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Draft A

Remedial Investigation/Feasibility Study for the 100-DR-1, 100-DR-2, 100-HR-1, 100-HR-2, and 100-HR-3 Operable Units

Prepared for the U.S. Department of Energy
Assistant Secretary for Environmental Management

U.S. DEPARTMENT OF ENERGY | Richland Operations Office
P.O. Box 550
Richland, Washington 99352

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Executive Summary

This document presents the results of a *Comprehensive Environmental Response, Compensation, and Liability Act of 1980* (CERCLA)[1] Remedial Investigation (RI)/Feasibility Study (FS) undertaken for a portion of the Hanford National Priorities List[2] (NPL) Site referred to as 100-D/H. 100-D/H represents areas impacted by nuclear operations at the D, DR, and H Reactor Areas. The area has been organized into four source operable units (OUs) (100-DR-1, 100-DR 2, 100-HR-1, and 100-HR 2) and a groundwater OU (100-HR-3). Site investigation and risk assessment work conducted for these OUs has resulted in a determination that contaminants in the vadose zone and groundwater pose a threat to the environment and that a CERCLA remedial action is warranted. Based on the 100-D/H RI/FS, the Proposed Plan issued by the U.S. Department of Energy, Richland Operations Office (DOE-RL) identifies a preferred alternative, as well as other alternatives considered for cleanup of the 100-D/H Operable Units, in order to receive comments from the Tribal Nations and the public. A Record of Decision (ROD) will be issued that identifies the final remedial alternative selected for 100-D/H and will document responses made to Tribal Nations and public comments. Remedial action decisions will address the integrated cleanup of contaminated soil, solid waste burial grounds, groundwater, and releases from and/or as a result of reactor operations. The objective for all these decisions is to protect human health and the environment, including restoring groundwater to drinking water standards and achieving ambient water quality criteria in the Columbia River that protect aquatic life.

There are 343 sites identified in the 100-D/H Areas and two operating groundwater pump-and-treat systems. All 343 sites were evaluated using the Tri-Parties site evaluation process for determining the status of each waste site. Forty-eight sites were closed out, rejected or not accepted as waste sites. The three reactors that are considered waste sites are not addressed in this document, but will be addressed in a future CERCLA decision. Also, one site, 100-D-58 (a septic tank and leach field), has been closed under

[1] *Comprehensive Environmental Response, Compensation, and Liability Act of 1980*, 42 USC 9601, et seq., Pub. L. 107-377, December 31, 2002. Available at: http://epw.senate.gov/cercla.pdf.

[2] 40 CFR 300, "National Oil and Hazardous Substances Pollution Contingency Plan," Appendix B, "National Priorities List," *Code of Federal Regulations*. Available at: http://www.gpo.gov/fdsys/pkg/CFR-2010-title40-vol27/xml/CFR-2010-title40-vol27-part300-appB.xml.

Washington State Department of Health regulations. The remaining 291 waste sites are evaluated in the RI/FS to determine the need for remedial action.

This RI/FS, which supports the Proposed Plan, has the following objectives:

- Provide information concerning the physical environmental setting for 100-D/H.
- Draw conclusions concerning the nature and extent of contamination present in 100-D/H and the potential for migration of contamination.
- Evaluate the potential for adverse effects on human health and the environment if no action is taken and exposure occurs.
- Develop and evaluate an appropriate range of remedial action alternatives for 100-D/H to address unacceptable risk to human health and the environment.

This RI/FS was prepared based on information gathered from historical studies, investigations, process knowledge, data collected during implementation of interim action RODs, and recent field investigations. Soil and groundwater cleanup actions and assessments have been performed since the early 1990s. The recently completed RI work was conducted to provide information to supplement the considerable body of information previously collected regarding site contamination. This supplemental work included five test pits, 17 groundwater monitoring wells, and 10 soil borings/temporary monitoring wells to resolve remaining data needs identified in the 100-D/H Work Plan[3]. In addition, a select network of wells was sampled to determine spatial and temporal variations in groundwater contamination.

100-D/H Background

The 100-D/H Areas encompass 20 km^2 (7.8 mi^2) adjacent to the Columbia River in the northwest portion of the Hanford Site. This section of the Columbia River is within the Hanford Reach, a non-tidal free-flowing section of the Columbia River. The Reach extends from Priest Rapids Dam downstream to the slack waters of Lake Wallula, created by McNary Dam. The Hanford Site contains some of the most important archaeological sites in the region. Hanford Site cultural resources are diverse, ranging from early

[3] DOE/RL-2008-46-ADD1, 2010, *Integrated 100 Area Remedial Investigation/Feasibility Study Work Plan, Addendum 1: 100-DR-1, 100-DR-2, 100-HR-1, 100-HR-2, and 100-HR-3 Operable Units*, Rev. 0, U.S. Department of Energy, Richland Operations Office, Richland, Washington. Available at: http://www5.hanford.gov/arpir/?content=findpage&AKey=0084374.

prehistoric times to the Atomic Age. Cultural resources surveys across the 100 Areas are routinely conducted as part of site evaluation to protect culturally sensitive areas. The results of these surveys are used in planning an appropriate remedial action.

The 100-D/H Areas include three deactivated nuclear reactors and support facilities that produced plutonium from 1945 to 1967. The reactors were built to irradiate uranium fuel rods to produce plutonium and other special nuclear materials. The reactors and processes associated with operations generated large quantities of liquid and solid wastes. Solid wastes included sludge, reactor components, and various other contaminated items associated with reactor operations. Waste generated from reactor operations was contaminated with radionuclides, hazardous chemicals, or both. Liquid wastes were released to the environment by discharging effluent to temporary surface impoundments, cribs, ditches, and the Columbia River. Solid waste was generally placed in burial grounds. The waste sites in 100-D/H included storage tanks, ponds, trenches, cribs, French drains, solid waste burial grounds, retention basins, pipelines, and spills/leaks.

Physical/Environmental Setting

The conceptual site model includes consideration of the physical and chemical characteristics of vadose materials, geologic features of the area, local groundwater characteristics, and the interaction of these elements with the Columbia River. The physical characteristics of the study area influence the movement of contaminants within the environment.

The topography is relatively flat inland from the Columbia River; elevation changes are greatest near the Columbia River, where the riverbank slopes steeply. The semiarid climate at 100-D/H has occasional high winds, and the majority of the land surface is an undisturbed shrub-steppe community. Riparian areas immediately adjacent to the river shoreline represent unique ecological communities.

The Hanford formation is the dominant material in the vadose zone (unsaturated zone) and consists of a sand and gravel unit that increases in thickness away from the river. The unconfined aquifer is predominantly within the Ringold Formation unit E in the 100-D Area. The unconfined aquifer is predominantly within the Hanford formation in the 100-H Area with the horn area being a transition from Ringold unit E to Hanford unit. The changing river levels directly influence groundwater elevations close to the river with decreasing effects inland. Groundwater flow is normally toward the river except

when the river is high, which causes groundwater to flow away from or parallel to the river.

Nature and Extent of Contamination

This document describes the current distribution of contaminants in environmental media, predicts the migration rate of contaminants through the physical setting (fate and transport), and evaluates the potential for contaminants to enter the Columbia River. Disposal of large volumes of liquid effluent to the vadose zone during reactor operations contributed to significant alterations in local hydrologic conditions and resulted in the accelerated transport of contaminants to deeper portions of the vadose zone and unconfined aquifer groundwater in 100-D/H. Today, contaminant migration rates are currently much slower than during operating periods because those discharges have stopped.

Contaminants in the vadose zone include radionuclides, anions, and metals. The analytical results from the RI characterization indicated the localized presence of hexavalent chromium [Cr(VI)] in the deep vadose zone to the water table. Cr(VI) is the most widespread contaminant in the groundwater beneath 100-D/H. Other groundwater contaminants are total chromium, strontium-90, and nitrate. Chromium is collocated with the Cr(VI) plume. Strontium-90 is present in the groundwater in localized areas within the 100-H Area and at one well in 100-D Area. Nitrate is present over larger areas but within boundaries of the Cr(VI) plume.

Waste site cleanup in 100-D/H began in 1997 under an interim action ROD[4] and is ongoing. These cleanups will continue to meet the requirements of the interim action ROD until a new ROD is issued. Interim action waste site cleanup consists primarily of removing and disposing of contaminated material followed by backfill and revegetation to protect human health and the environment.

Groundwater cleanup was initiated in 1997 under the same interim action ROD (EPA/ROD/R10-95/126) with the startup of the first pump-and-treat system at 100-H. The use of pump-and-treat was expanded under a 2004 interim action ROD Explanation

[4] EPA/ROD/R10-95/126, 1995, *Interim Remedial Action Record of Decision for the 100-BC-1, 100-DR-1, and 100-HR-1 Operable Units, Hanford Site, Benton County, Washington*, U.S. Environmental Protection Agency, Region 10, Seattle, Washington. Available at: http://www.epa.gov/superfund/sites/rods/fulltext/r1095126.pdf.

of Significant Differences[5] to include a stand alone pump-and-treat system to capture a newly discovered portion of the plume. Routine monitoring of the 100-HR-3 OU revealed an expansion of the plume across the horn area towards 100-H Area and an increase in Cr(VI) concentrations in the 100-D Area. The 2009 interim action ROD Explanation of Significant Differences[6] increased the groundwater extraction rate and the capacity of the pump-and-treat systems to capture the Cr(VI) plume. Two pump-and-treat systems are currently operating and are designed to remediate the Cr(VI) plume and provide a measure of protection to the Columbia River. Concentrations and plume footprint areas in groundwater are declining.

Exposure Assessment

Scenarios of how human and environmental receptors might come into contact with contaminants, with resultant health impacts, were evaluated. The principal contaminants identified in the soil beneath one or more waste sites include radionuclides, metals, polychlorinated biphenyls, and polynuclear aromatic hydrocarbons. The risk assessment identified chromium, Cr(VI), nitrate, and strontium-90 as the principal groundwater contaminants. Potential remedial technologies in the FS mitigate these soil and groundwater contaminants.

In addition, 143 waste sites in 100-D/H had closeout verification data collected following the implementation of interim action removal and disposal that was quantitatively evaluated. New soil screening levels (SSLs) and preliminary remediation goals (PRGs) were established for each environmental media of interest (soil and groundwater), each type of contaminant (hazardous substances and radionuclides), human and ecological receptors, and each potentially complete exposure pathway. The SSLs and PRGs are based on updated U. S. Environmental Protection Agency (EPA) guidance and a conservative scenario that includes assumptions of uniform vadose zone contamination (100:0 initial source distribution model for low distribution coefficient [K_d] contaminants and 70:30 initial source distribution model for high K_d contaminants) and an

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[6] EPA, Ecology, and DOE, 2009, *Explanation of Significant Differences for the 100-HR-3 and 100-KR-4 Operable Units Interim Action Record of Decision: Hanford Site Benton County, Washington*, U.S. Environmental Protection Agency, Washington State Department of Ecology, and U.S. Department of Energy, Olympia, Washington. Available at: http://www5.hanford.gov/arpir/?content=findpage&AKey=0096029.

infiltration/recharge rate based on irrigated agriculture for SSLs and conservation land use for PRGs.

Alternatives Development

The FS portion of the RI/FS consists of four phases: development of remedial action objectives (RAOs), screening of remedial technologies, development of remedial alternatives, and detailed analysis of alternatives. Remedial technologies were assembled into alternatives that address contamination on a media- or source-specific basis.

RAOs are identified for groundwater, surface water, and soil. RAOs are general descriptions of what a cleanup under CERCLA is expected to accomplish. These are narrative statements that define the extent to which waste sites require cleanup to protect human health and the environment. To meet RAOs, PRGs are established for each contaminant, receptor, exposure pathway, and environmental media of interest. Soil PRGs are provided for the protection of groundwater and surface water based on site-specific data for the 100-D/H Area and specific parameters including a natural recharge rate scenario (i.e., no irrigation) and an irrigation scenario. Both PRGs are used for remedial alternative development and evaluation.

A range of general response actions to meet RAOs is identified for the vadose zone and groundwater contaminants of concern (COCs). Response actions include a range of technologies and process options for vadose zone and groundwater. Process options and technologies for the range of response actions are evaluated for relative effectiveness, implementability, and cost.

The remedial technologies, retained from the screening process, were combined into remedial alternatives that provide a range of technologies for integrated waste site and groundwater remediation. The remedial alternatives were developed to achieve the RAOs and be responsive to National Contingency Plan[7] (NCP) and CERCLA programmatic goals. Alternatives evaluated include:

- Alternative 1 (No Action [as required by the NCP])

[7] 40 CFR 300, "National Oil and Hazardous Substances Pollution Contingency Plan," *Code of Federal Regulations*. Available at: http://www.gpo.gov/fdsys/pkg/CFR-2010-title40-vol27/xml/CFR-2010-title40-vol27-part300.xml.

- Alternative 2 RTD and Grouting for Waste Site and Pump-and-Treat with Biological Treatment for Groundwater
- Alternative 3 RTD and Grouting of Waste Site and Increased Capacity Groundwater Pump-and-Treat for Groundwater
- Alternative 4 RTD for Waste Sites and Pump-and-Treat for Groundwater

Alternatives Evaluation

Alternatives were evaluated individually and comparatively against the CERCLA criteria except public acceptance. Two criteria, overall protection of human health and the environment and compliance with applicable or relevant and appropriate requirements, are "threshold criteria." The next five are "balancing criteria" and include long-term effectiveness; reduction of toxicity, mobility, and volume (TMV) through treatment; short-term effectiveness; implementability; and cost. The Washington State acceptance modifying criteria has been addressed by state support for issuance of this RI/FS report and the 100-D/H Proposed Plan. The remaining modifying criterion, community acceptance, will be evaluated after the Proposed Plan goes through the Tribal Nations and public comment process as reflected in the responsiveness summary that will be included in the 100-D/H CERCLA ROD. The purpose of the detailed and comparative analysis is to develop the information necessary to recommend a preferred alternative in a Proposed Plan. The analysis showed:

- Alternative 1 (No Action) does not meet threshold criteria for all sites.
- Alternative 2 RTD and Grouting for Waste Site and Pump-and-Treat with Biological Treatment for Groundwater meets threshold criteria, performs well for long-term effectiveness and short-term effectiveness, and moderately well for reduction of TMV and implementability.
- Alternative 3 RTD and Grouting of Waste Site and Increased Capacity Groundwater Pump-and-Treat for Groundwater meets threshold criteria, performs well for long-term effectiveness, short-term effectiveness, and implementability, and performs moderately well for reduction of TMV.

- Alternative 4 RTD for Waste Sites and Pump-and-Treat for Groundwater meets threshold criteria, performs well for long-term effectiveness and implementability, and performs moderately well for short-term effectiveness and reduction of TMV.

Alternative 3 performs better than Alternatives 2 and 4 for short-term effectiveness. Costs for Alternative 4 are lower than for Alternative 2, and Alternative 3 has the highest cost. The analysis presented in this RI/FS provides enough information to be able to recommend a preferred alternative in the Proposed Plan that describes the proposed final remedies and PRGs.

There will be a period of time between when the ROD is approved and when the required Remedial Design/Remedial Action Work Plan (RD/RAWP) is prepared and issued. During this time period, DOE-RL plans to continue remedial activities such as waste site RTD remediation and groundwater pump-and-treat. In order for the these actions to be consistent with the final action remedy selection, the current interim action RD/RAWPs will be modified using the *Hanford Federal Facility Agreement and Consent Order* (Tri-Party Agreement) (Ecology, EPA, and DOE, 1989)[8] change notice process to include the final cleanup levels specified in the ROD.

[8] Ecology, EPA, and DOE, 1989, *Hanford Federal Facility Agreement and Consent Order*, 2 vols., as amended, Washington State Department of Ecology, U.S. Environmental Protection Agency, and U.S. Department of Energy, Olympia, Washington. Available at: http://www.hanford.gov/?page=81.

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Terms

1D	one-dimensional
3D	three-dimensional
AEA	*Atomic Energy Act of 1954*
AEC	Atomic Energy Commission
ARARs	applicable or relevant and appropriate requirements
ARCL	allowable residual contamination level
AUF	area use factor
AWQC	Ambient Water Quality Criteria
B.P.	before present
BAF	bioaccumulation factor
BCG	Biota Concentration Guide
BDL	below detection limit
bgs	below ground surface
BHC	benzene hexachloride
CAS	Chemical Abstract Service
CCC	criteria continuous concentration
CERCLA	*Comprehensive Environmental Response, Compensation, and Liability Act of 1980*
CFR	*Code of Federal Regulations*
CLARC	"Cleanup Levels and Risk Calculations"
CMC	criteria maximum concentration
COC	contaminant of concern
COEC	contaminant of ecological concern
COPC	contaminant of potential concern
Cr(VI)	hexavalent chromium
CRBG	Columbia River Basalt Group
CRC	Columbia River Corridor
CSM	conceptual site model
CTUIR	Confederated Tribes of the Umatilla Indian Reservation
CV	coefficient of variation

CVP	cleanup verification package
CWA	*Clean Water Act of 1972*
CY	calendar year
DOC	dissolved organic carbon
DOE	U.S. Department of Energy
DOE-RL	DOE Richland Operations Office
DQA	data quality assessment
DQO	data quality objective
DWS	drinking water standard
Ecology	Washington State Department of Ecology
EcoSSL	Ecological Soil Screening Level
EIS	Environmental Impact Statement
ELCR	excess lifetime cancer risk
EPA	U.S. Environmental Protection Agency
EPC	exposure point concentration
EQL	estimated quantitation limit
ERA	ecological risk assessment
ERAGS	*Ecological Risk Assessment Guidance for Superfund: Process for Designing and Conducting Ecological Risk Assessments: Interim Final* (EPA 540-R-97-006)
ERDF	Environmental Restoration Disposal Facility
FSB	Fuel Storage Basin
FY	fiscal year
GHG	greenhouse gas
GRA	general response actions
HDPE	high-density polypropylene
HEAST	Health Effects Assessment Summary Table
HEIS	Hanford Environmental Information System
HHRA	human health risk assessment
HHT	human health toxicity
HI	hazard index
HMS	Hanford Meteorological Station

HQ	hazard quotient
IAMP	*Interim Action Monitoring Plan for the 100-HR-3 and 100-KR-4 Operable Units*
ICP	inductively coupled plasma
ID	identification
IEUBK	Integrated Exposure Uptake Biokinetic
IRIS	Integrated Risk Information System
ISRM	in situ redox manipulation
ISS	interim safe storage
K_d	distribution coefficient
LFI	limited field investigation
LIDAR	Light Detection and Ranging
LOAEC	lowest observed adverse-effect concentration
LOAEL	lowest observed adverse-effect level
LOEC	lowest observed effect concentration
LSFF	Large Sodium Fire Facility
MATC	maximum allowable toxicant concentration
MCL	maximum contaminant level
MCLG	maximum contaminant level goal
MDL	method detection limit
MNA	monitored natural attenuation
MTCA	"Model Toxics Control Act" (WAC 173-340)
NAVD88	*North American Vertical Datum of 1988*
NBD	no background
NCP	"National Oil and Hazardous Substances Pollution Contingency Plan" (40 CFR 300)
NEPA	*National Environmental Policy Act of 1969*
NEPA	National Environmental Protection Act
NESHAP	National Emission Standards Hazardous Air Pollutants
NJDEP	New Jersey Department of Environmental Protection
NOAEC	no observed adverse-effect concentration
NOAEL	no observed adverse-effect level

NPL	"National Priorities List" (40 CFR 300, Appendix B)
OEHHA	Office of Health Hazard Assessment
ORNL	Oak Ridge National Laboratory
ORP	oxygen reduction potential
OSE	orphan site evaluation
OU	operable unit
PA/SI	preliminary assessment/site investigation
PAH	polycyclic aromatic hydrocarbons
PCB	polychlorinated biphenyl
PEF	particulate emission factor
PNNL	Pacific Northwest National Laboratory
PQL	practical quantitation limit
PRG	preliminary remediation goal
PVC	polyvinyl chloride
QA	quality assurance
QAPjP	Quality Assurance Project Plan
QC	quality control
QRA	qualitative risk assessment
RAG	remedial action goal
RAO	remedial action objective
RBSLs	risk-based screening levels
RCBRA	River Corridor Baseline Risk Assessment
RCRA	*Resource Conservation and Recovery Act of 1976*
RCW	*Revised Code of Washington*
RDR/RAWP	remedial design report/remedial action work plan
RESRAD	RESidual RADioactivity
RfD	reference dose
RI/FS	remedial investigation/feasibility study
RLM	Ringold Formation lower mud
RME	reasonable maximum exposure

ROD	record of decision
RPO	remedial process optimization
RSVP	Remaining Site Verification Package
RTD	removal, treatment, and disposal
RUM	Ringold Formation upper mud
SAP	sampling and analysis plan
SDWA	*Safe Drinking Water Act of 1974*
SESP	Surface Environmental Surveillance Program
SMDP	scientific management decision point
SOF	sum of fractions
SSL	soil screening level
STOMP	*Subsurface Transport Over Multiple Phases*
SVOCs	semivolatile organic compounds
TBC	to be considered
TMV	toxicity, mobility, and volume
TOC/DOC	total organic carbon/dissolved organic carbon
TPA	Tri-Party Agreement
TPH	total petroleum hydrocarbon
Tri-Cities	Kennewick, Pasco, and Richland
TRV	toxicity reference value
TSD	treatment, storage, and disposal
UCL	upper confidence level
USFWS	U.S. Fish and Wildlife Service
USGS	U.S. Geological Survey
VOC	volatile organic compound
WAC	*Washington Administrative Code*
WIDS	Waste Information Data System

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1 Introduction

In 1989, the U.S. Department of Energy (DOE), the U.S. Environmental Protection Agency (EPA), and the Washington State Department of Ecology (Ecology) signed the *Hanford Federal Facility Agreement and Consent Order* (Ecology et al., 1989a), hereinafter called the Tri-Party Agreement (TPA) to provide a framework for the cleanup of the Hanford Site (Figure 1-1). The scope of the agreement addressed the *Comprehensive Environmental Response, Compensation, and Liability Act of 1980* (CERCLA) remediation of inactive hazardous waste sites and active waste management, *Resource Conservation and Recovery Act of 1976* (RCRA) corrective action for solid waste management units, and closure of RCRA treatment, storage, and disposal (TSD) units across the Hanford Site.

This document presents the results of a CERCLA Remedial Investigation (RI)/Feasibility Study (FS) undertaken for 100-D/H (Figure 1-1). The information contained in this RI/FS supports a Proposed Plan, which will go through a public review and provide the basis for a Record of Decision (ROD). The ROD for 100-D/H will apply to the source operable units (OUs) 100-DR-1, 100-DR-2, 100-HR-1, and 100-HR-2 and to the 100-HR-3 Groundwater OU (Figure 1-2).

Much of Chapter 1 is devoted to summarizing the assessment and remediation work, treatability tests, and other relevant studies. This historical information is presented to provide a comprehensive picture of current 100-D/H site conditions and establishes a foundation for the remainder of the RI/FS document.

For the purpose of CERCLA cleanup, four sections of the Hanford Site were placed on the "National Oil and Hazardous Substances Pollution Contingency Plan" (40 CFR 300), Appendix B, "National Priorities List," hereinafter called NPL, as separate areas: 100 Area (Reactor Operations), 200 Area (Irradiated Fuel Reprocessing and Waste Management), 300 Area (Nuclear Fuel Production and Research and Development), and 1100 Area (Equipment and Maintenance). Because of the large number of waste sites, unplanned releases, and extensive groundwater contamination, the 100 Area was further divided into source and groundwater OUs for management of the investigation and remediation.

The list of waste sites for 100-D/H has been refined over time. During operations, waste disposal locations were constructed and operated as needed. Eventually, these locations were assigned an identification number. As technology evolved, computer databases were developed to store and track waste site information. Waste Information Data System (WIDS) is the database of waste site information for the Hanford Site. It assigns standardized identification numbers (site codes) and tracks the status of each waste site. As a result of the potential listing on the NPL (40 CFR 300, Appendix B), a preliminary assessment/site investigation (PA/SI) was conducted. This PA/SI identified potential waste sites by geographic area across the Hanford Site. A hazard ranking score resulted in four areas (100, 200, 300, and 1100) to be added to the NPL (40 CFR 300, Appendix B). Waste sites identified within the geographic areas included 100-KE and 100-KW areas and the nearby environs. These waste sites were included in WIDS and formed the basis for the preliminary list of waste sites in the 100-D/H geographic area. Since the PA/SI, additional efforts have been conducted to ensure that all waste sites posing a threat to human health and the environment (HHE) are identified through the Nonoperational Area Evaluation process, including the Orphan Site Evaluation and Discovery Site processes.

In 1991, the Tri-Parties determined there was a need to prioritize the CERCLA investigations and identify early actions to address waste sites and groundwater contamination. *Hanford Past-Practice Strategy* (DOE/RL-91-40), hereinafter called Past-Practice Strategy, provided the basis for prioritizing investigations and cleanup actions across the Hanford Site. This strategy emphasized the need to address waste sites and groundwater contamination that may pose a near-term impact to public health and the environment. In addition, the strategy proposed a bias for action to clean up waste sites and existing contamination where the need for a remedy was evident.

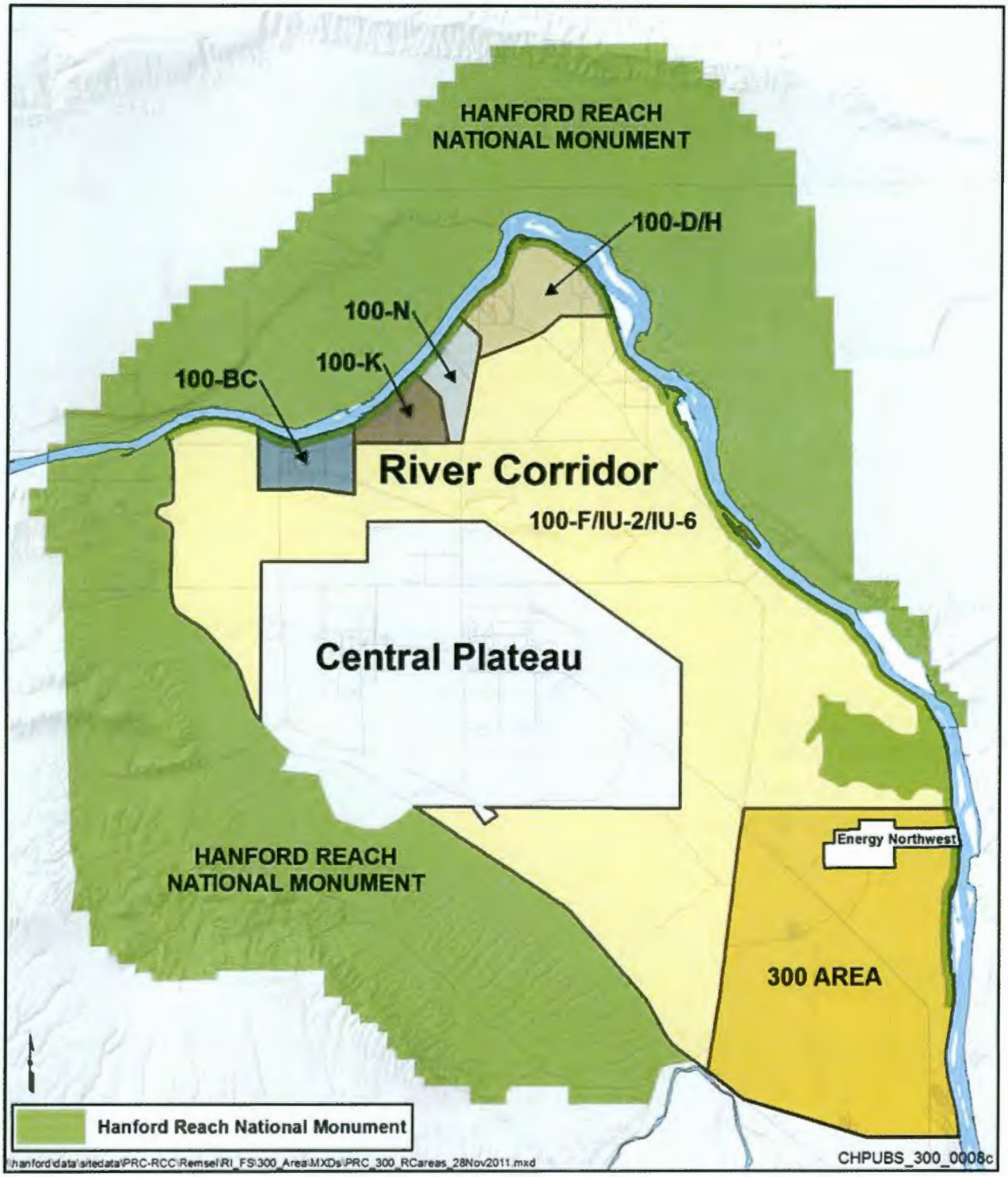

Figure 1-1. Hanford Site Map

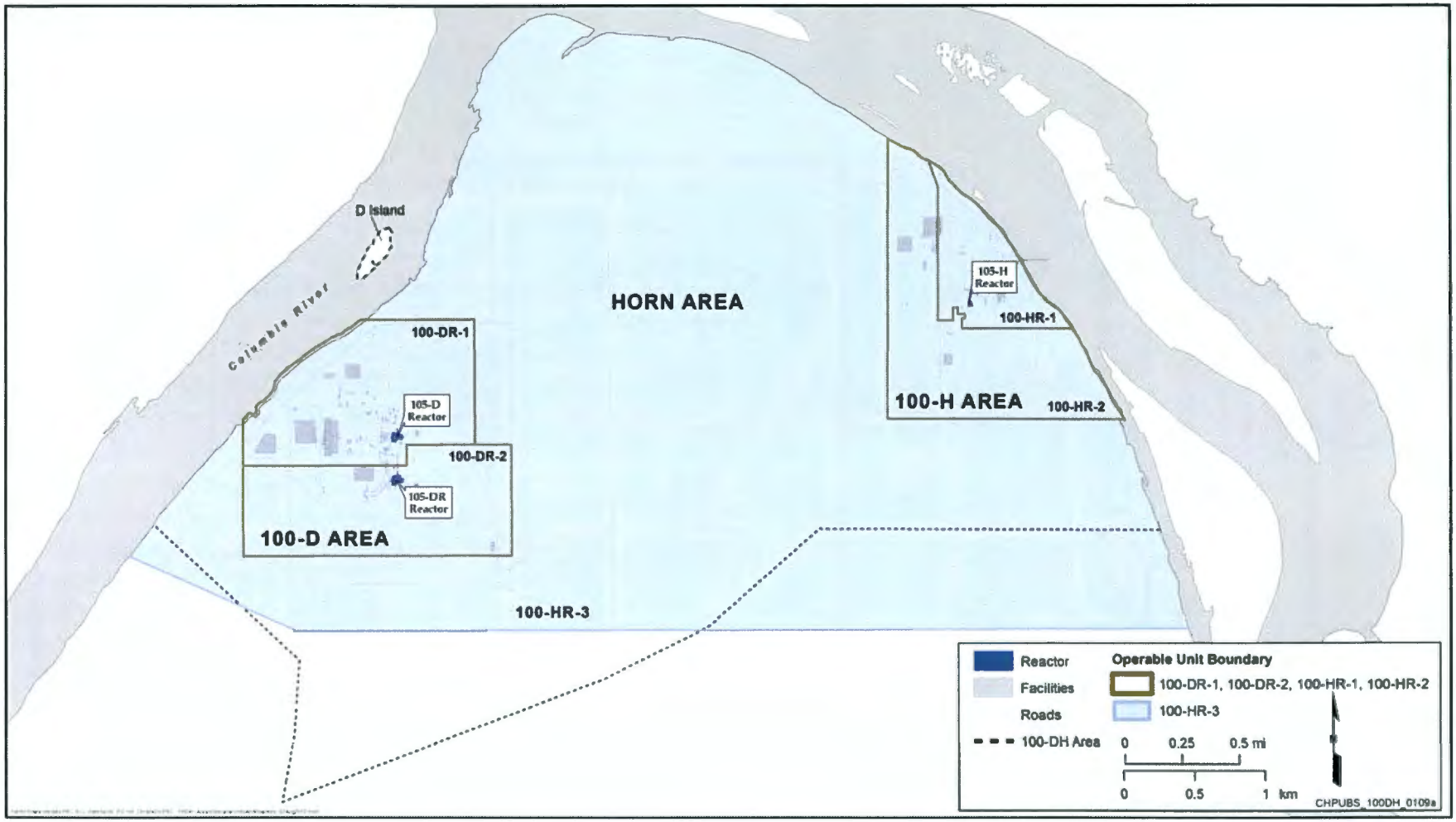

Figure 1-2. Location of 100-D/H Area OUs and Reactors

For 100-D/H, the Past-Practice Strategy (DOE/RL-91-40) resulted in specific actions and priority investigations. Limited field investigations (LFIs) were initiated where liquid waste disposal sites were considered responsible for local groundwater contamination. These LFIs were an initial step in characterizing the nature and extent of contamination in the vadose zone, structures, and debris that received radioactive liquid effluent discharges. Radionuclides, metals, and organics were analyzed in the LFI samples. The following reports document these investigations:

- *Limited Field Investigation Report for the 100-DR-1 Operable Unit* (DOE/RL-93-29)
- *RCRA Facility Investigation/Corrective Measures Study Work Plan for the 100-DR-2 Operable Unit, Hanford Site, Richland, Washington* (DOE/RL-93-46)
- *Limited Field Investigation Report for the 100-HR-1 Operable Unit* (DOE/RL-93-51)
- *Limited Field Investigation Report for the 100-HR-2 Operable Unit* (DOE/RL-94-53)
- *Limited Field Investigation Report for the 100-HR-3 Operable Unit* (DOE/RL-93-43)

The LFIs indicated that liquid disposal sites in 100-DR-1, 100-DR-2, 100-HR-1, and 100-HR-2 OUs were primarily responsible for the continuing release of hexavalent chromium [Cr(VI)] above established limits to the groundwater. For the 100-HR-3 OU (100-D/H groundwater), it was established that Cr(VI) in groundwater was entering the Columbia River at concentrations considered toxic to aquatic organisms. This led to the selection of interim actions to remediate source and groundwater contamination within the 100-DR-1, 100-DR-2, 100-HR-1, 100-HR-2, and 100-HR-3 OUs under the following interim action RODs:

- *Interim Action Record of Decision for the 100-BC-1, 100-BC-2, 100-DR-1, 100-DR-2, 100-FR-1, 100-FR-2, 100-HR-1, 100-HR-2, 100-KR-1, 100-KR-2, 100-IU-2, 100-IU-6, and 200-CW-3 Operable Units, Hanford Site, Benton County, Washington (100 Area Remaining Sites)* (hereinafter called 100 Area Remaining Sites ROD [EPA/ROD/R10-99/039]), July 1999
- *Interim Remedial Action Record of Decision for the 100-BC-1, 100-DR-1, and 100-HR-1 Operable Units, Hanford Site, Benton County, Washington* (EPA/ROD/R10-95/126), September 1995
- *Interim Remedial Action Record of Decision for the 100-BC-1, 100-BC-2, 100-DR-1, 100-DR-2, 100-FR-2, 100-HR-2, and 100-KR-2 Operable Units, Hanford Site (100 Area Burial Grounds), Benton County, Washington* (hereinafter called 100 Area Burial Grounds ROD [EPA/ROD/R10-00/121]), September 2000
- *Record of Decision for the 100-HR-3 and 100-KR-4 Operable Units Interim Remedial Actions, Hanford Site, Benton County, Washington* (hereinafter called 100-H/K ROD [EPA/ROD/R10-96/134]), March 1996

Current River Corridor cleanup work is progressing based on Interim Action RODs. An objective of waste site cleanup is to remove sources of contamination and contaminated environmental media that are close to the Columbia River, and place them in the Environmental Restoration Disposal Facility (ERDF) for final disposal on the Central Plateau. Reducing the concentrations of contaminants entering the Columbia River and restoring the groundwater to beneficial use remain the key objective of groundwater remediation within 100-D/H. Interim Remedial Action Objectives (RAOs) for the cleanup of waste sites within the 100-DR-1, 100-DR-2, 100-HR-1, and 100-HR-2 OUs focused on protecting human health from contaminants in the soil, controlling the sources of groundwater contamination to minimize the effects to groundwater resources, and protecting the Columbia River from further adverse effects. For the

100-HR-3 Groundwater OU, interim action RAOs focused on Cr(VI) as the key effect posed by the site to groundwater and surface water.

DOE is the lead federal agency at Hanford, per CERCLA, *Superfund Implementation* (Executive Order 12580), and the TPA (Ecology et al., 1989a). DOE develops implementation strategies and conducts response actions in this lead federal agency role. With implementation of the Past-Practice Strategy (DOE/RL-91-40) and progress with the interim remedial actions, DOE prepared *Hanford Site Cleanup Completion Framework* (DOE/RL-2009-10), hereinafter called Cleanup Completion Framework, to describe the cleanup strategy (Table 1-1). One of the principal components of the framework is the River Corridor, which consists of approximately 570 km^2 (220 mi^2) of the Hanford Site along the Columbia River. It includes a contiguous area that extends from the 100 and the 300 Areas to the Central Plateau boundaries (Figure 1-1).

Table 1-1. Overarching Goals for Hanford Site Cleanup

Goal	Description
1	Protect the Columbia River.
2	Restore groundwater to its beneficial use to protect human health, the environment, and the Columbia River.
3	Clean up River Corridor waste sites and facilities to protect groundwater and the Columbia River, shrink the active cleanup footprint to the Central Plateau, and support reasonably anticipated future land uses.
4	Clean up Central Plateau waste sites, tank farms, and facilities to protect groundwater and the Columbia River; minimize the footprint of areas requiring long-term waste management activities; and support reasonably anticipated future land uses.
5	Safely manage and transfer legacy materials scheduled for offsite disposition, including special nuclear material (including plutonium), spent nuclear fuel, transuranic waste, and immobilized high-level waste.
6	Consolidate waste treatment, storage, and disposal operations on the Central Plateau.
7	Develop and implement institutional controls and long-term stewardship activities that ensure protection of human health and the environment after cleanup activities are completed.
Source: *Hanford Site Cleanup Completion Framework* (DOE/RL-2009-10).	

Ecology is the lead regulatory agency for 100-D/H. The lead regulatory agency has the primary responsibility for overseeing all remedial action activities to ensure they meet applicable requirements.

For sites in the River Corridor, remedial actions are expected to restore groundwater to drinking water standards (DWSs) and protect the aquatic life in the Columbia River, by achieving ambient water quality criteria (AWQC) at groundwater discharge points to the river. Unless technically impracticable, the objectives will be achieved within a reasonable time. If cleanup levels are not achievable in a reasonable time or are determined to be technically impracticable, programs will be implemented to prevent further migration of the plume, prevent exposure to contaminated groundwater, and evaluate further risk reduction opportunities or seek an applicable or relevant and appropriate requirements (ARAR) waiver.

Current River Corridor cleanup work is progressing pursuant to interim action RODs. A primary objective of this work has been to remove potential sources of contamination and contaminated environmental media that are close to the Columbia River. In addition, interim actions have been occurring to address groundwater contamination.

To complete cleanup, the River Corridor has been divided into six geographic decision areas, including 100-D/H, to achieve source and groundwater remedy decisions (Figure 1-1). These decisions will provide comprehensive coverage for all areas within the River Corridor and will incorporate interim action cleanup activities. Cleanup levels will be established that will protect human health and the environment. These levels will comply with ARARs and consider the remedial action goals (RAGs) previously used in the implementation of interim action RODs for River Corridor OUs. The proposed cleanup levels (preliminary remediation goals [PRGs]) are numeric values that meet ARARs and are protective of human health and the environment. These PRGs will be used to assess the effectiveness of the selected remedial alternatives.

Chapter 1 summarizes the assessment and remediation work that was completed before preparation of this RI/FS Report. In addition to the 100-D/H specific work, Chapter 1 describes other relevant work that supports remedy selection for 100-D/H and explains the nature and extent of contamination across all six River Corridor areas. This RI/FS report builds on this body of previous work to provide a comprehensive understanding of current site conditions and evaluate a set of alternatives for addressing the remaining human health and environmental risks at 100-D/H.

For the purpose of this RI/FS, the following definitions are used:

- **Shallow vadose zone:** from ground surface to a depth of 4.6 m (15 ft). This depth interval is evaluated for protection of human health and ecological receptors as well as protection of groundwater and surface water.
- **Deep vadose zone:** from a depth of 4.6 m (15 ft) to the water table. This depth interval is evaluated for protection of groundwater and surface water. Residual contaminant concentrations in this zone are evaluated for human health protection to provide risk management information.

This RI/FS for 100-D/H was undertaken in accordance with *Integrated 100 Area Remedial Investigation/ Feasibility Study Work Plan* (DOE/RL-2008-46), hereinafter called the Integrated Work Plan, for the 100 Area, which contains the planning elements common to all the Hanford Site 100 Area source and groundwater OUs, and *Integrated 100 Area Remedial Investigation/Feasibility Study Work Plan, Addendum 1: 100-DR-1, 100-DR-2, 100-HR-1, 100-HR-2, and 100-HR-3 Operable Units* (DOE/RL-2008-46-ADD1), hereinafter called 100-D/H Work Plan, which is specific to 100-D/H. These work plans were developed and approved by Ecology to outline the requirements for an RI/FS supporting cleanup decisions for the OUs within the 100 Area NPL (40 CFR 300, Appendix B).

This introductory chapter is followed by the RI portion of the report (Chapters 2 through 7), the FS portion of the report (Chapters 8 through 10), and a list of the references used in preparing this report (Chapter 11).

- Chapter 2—Study Area Investigation
- Chapter 3—Physical Characteristics of the Study Area
- Chapter 4—Nature and Extent of Contamination
- Chapter 5—Contaminant Fate and Transport
- Chapter 6—Human Health Risk Assessment
- Chapter 7—Ecological Risk Assessment
- Chapter 8—Identification and Screening of Technologies
- Chapter 9—Development and Screening of Alternatives

- Chapter 10—Detailed Analysis of Alternatives
- Chapter 11—References

This RI/FS report includes extensive data used to perform calculations and assessments. Summaries of data are provided in this document and appendices, and clickable links may be used to take the reader to more detailed information contained in particular studies, databases, or reports found in the Administrative Record. Appendices are as follows:

- Appendix A—Site Map
- Appendix B— Summary of Previous Investigations/Remediation and Annotated Bibliography
- Appendix C—Aquifer Tube Water Quality Field Data
- Appendix D—Analytical Data and Text on Data Protocols/Quality Assurance/Quality Control
- Appendix E—Nature and Extent Summaries and Waste Site Table
- Appendix F—Fate and Transport Modeling Documentation
- Appendix G—Summary of Risk Characterization Results with Inclusion of Background Concentrations
- Appendix H—Ecological Risk Assessment Calculation Brief
- Appendix I—Technology Screening—Not Retained Technologies
- Appendix J—Alternative Development Supporting Documentation
- Appendix K—Nonoperational Area Evaluation
- Appendix L—100-DH Riparian/Nearshore Evaluation
- Appendix M—Data from New Characterization Boreholes and Wells and Development of Geologic Cross Sections
- Appendix N—Potential Applicable or Revelvant and Appropriate Requirements
- Appendix O—Summary Statistics

1.1 Purpose and Scope of Report

The RI/FS process is outlined in EPA and DOE RI/FS guidance (*Guidance for Conducting Remedial Investigations and Feasibility Studies Under CERCLA, Interim Final* [EPA/540/G-89/004], hereinafter called CERCLA RI/FS Guidance; *Remedial Investigation/Feasibility Study (RI/FS) Process, Elements and Techniques* [DOE/EH-94007658]). The RI/FS process is the methodology that the *Superfund Amendments and Reauthorization Act of 1986* program has established for characterizing the nature and extent of risks posed by uncontrolled hazardous waste sites and for evaluating potential remedial options.

This RI/FS was prepared in accordance with the previously referenced guidance as well as *CERCLA Compliance with Other Laws Manual: Interim Final* (EPA/540/G-89/006) and *CERCLA Compliance with Other Laws Manual: Part I* (EPA/540/G-89/009). These documents provide information on the regulations and standards that govern the RI/FS process, as well as an overview of requirements for each chapter of the RI/FS.

This RI/FS has the following objectives:

- Provide information concerning the physical environmental setting and site characterization.
- Draw conclusions concerning the nature and extent of contamination present at the site, the potential for migration of contamination, and the potential for adverse human health and environmental effects if no action is taken at the site and exposure occurs. This goal is achieved by evaluating historical and operational information about the site, identifying contaminants of potential concern (COPCs), evaluating potential migration pathways, and understanding potential effects to receptors, by estimating exposure (dose) effects in consideration of contaminant toxicity.
- Develop and evaluate an appropriate range of remedial action alternatives for the site that address unacceptable risk to human health or the environment.

This report also fulfills DOE's responsibility under the *National Environmental Policy Act of 1969* (NEPA) to assess NEPA values when evaluating CERCLA remedial actions. EPA and DOE-RL will issue a ROD for the 100-D/H OUs that will include responses to the comments received. After the ROD is issued, a remedial design/remedial action will be developed, approved, and then implemented.

The conceptual site model (CSM) is used in this RI/FS report to present what is known about 100-D/H. The American Society for Testing and Materials *Standard Guide for Developing Conceptual Site Models for Contaminated Sites* (ASTM E1689-95) defines the CSM as "a written or pictorial representation of an environmental system and the biological, physical, and chemical processes that determine the transport of contaminants from sources through environmental media to environmental receptors within the system." For the 100-D/H Work Plan (DOE/RL-2008-46-ADD1), the CSM was used to integrate relevant site information, determine whether information or data were missing (data gaps), and identify additional information to be collected. In Chapters 2 through 7 of this report, the model is refined by the additional information and then used to identify and evaluate potential risk to human health and the environment.

Figure 1-3 presents the basic elements associated with a CSM:

- **Source**—the location where a contaminant enters the physical setting. The primary sources of contaminants were releases related to reactor operations and are described in Chapter 1. Secondary sources are created when contaminants are mixed in the vadose zone and then the groundwater. Reactor operations at 100-D/H have ceased, so remaining primary sources are minimal and are expected to be removed through interim remedial actions; therefore, this document focuses on secondary contaminant sources in the vadose zone and groundwater along with potential risk to human health and the environment. These secondary sources are described in Chapter 4.
- **Release Mechanisms**—the actions necessary to release contaminants to the environment through resuspension of contaminated particulate matter, corrosion, and liquid waste discharges to the vadose zone, plant intrusion, animal burrowing, and erosion. Release mechanisms and relevant physical features are introduced in Chapter 3 and discussed in Chapter 5 in the context of fate and transport modeling.
- **Transport**—movement of a radiological, chemical, or physical agent in the environment from a secondary source, where human or ecological exposure could occur. Contaminants introduced into the environment can be transported between environmental media such as air, vadose zone, groundwater, and surface water because of interconnecting release mechanisms. Transport is discussed in Chapter 5.

- **Exposure**—the process by which a contaminant or physical agent in the environment comes into direct contact with the body, tissues, or exchange boundaries of humans, plants, or animals (for example, ingestion, inhalation, dermal absorption, or root uptake). Contaminants in the environment move from sources to potential receptors via pathways. An exposure pathway is complete when a receptor encounters contaminated environmental media. Potential exposure scenarios are discussed in Chapters 6 and 7.

- **Receptors**—humans and other organisms (for example, plants, animals, and other species) that may come into contact with the contaminants. Chapters 6 and 7 evaluate exposure to receptors.

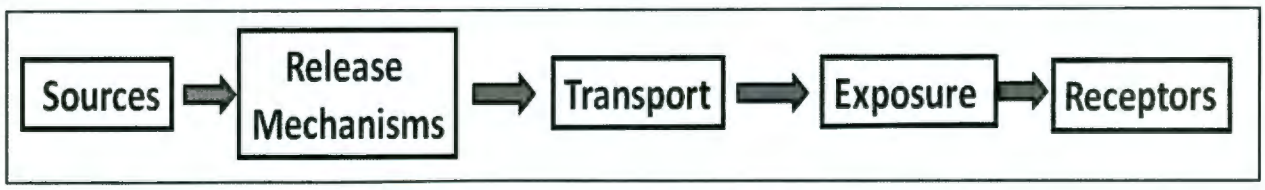

Figure 1-3. Conceptual Site Model

In Chapters 8 through 10, the refined model is used to identify technologies, develop remedial alternatives, and evaluate the effectiveness of potential remedial actions.

The identification of data needs in the 100-D/H Work Plan (DOE/RL-2008-46-ADD1) led to development of a sampling and analysis plan (SAP) that established characterization activities specific to 100-D/H (*Sampling and Analysis Plan for the 100-DR-1, 100-DR-2, 100-HR-1, 100-HR-2, and 100-HR-3 Operable Units Remedial Investigation/Feasibility Study* [DOE/RL-2009-40], hereinafter called the 100-D/H SAP). The approved 100-D/H SAP (DOE/RL-2009-40) includes a field sampling plan that provides the sampling strategy and techniques that were used to obtain the RI/FS data presented in this report. The 100-D/H SAP (DOE/RL-2009-40) also provides a Quality Assurance Project Plan (QAPjP) to ensure that data collected meet the appropriate quality assurance (QA) and quality control (QC) requirements.

1.2 Site Background

The Hanford Site encompasses approximately 1,517 km^2 (586 mi^2) in Benton, Franklin, and Grant counties in south-central Washington State within the semiarid Pasco Basin of the Columbia Plateau. The Site stretches approximately 50 km (30 mi) north to south and about 40 km (24 mi) east to west, immediately north-northwest of the confluence of the Yakima and Columbia Rivers; the cities of Kennewick, Pasco, and Richland (the Tri-Cities); and the city of West Richland. The Columbia River flows 80 km (50 mi) through the northern part of the Hanford Site and, turning south, forms part of the Site's eastern boundary, while the Yakima River runs near the southern boundary of the Hanford Site, joining the Columbia River at the city of Richland. The central portion of the Hanford Site is punctuated by two small east-west trending ridges, Gable Butte and Gable Mountain. Lands adjoining the site to the west, north, and east are principally range and agricultural. State Routes 240 and 24 skirt the southwestern and northern portions of the Site, respectively.

The Hanford Site area is culturally rich, experiencing a history of multiple occupations by both Native and non-Native Americans. For thousands of years, Native American peoples have inhabited the lands within and around the Hanford Site (*Tribal Distribution in Washington* [Spier, 1936]; and *Handbook of North American Indians: Volume 12, Plateau* [Walker and Sturtevant, 1998]). Non-Native American presence in the mid-Columbia began in 1805 with the arrival of the Lewis and Clark Expedition along the Columbia and Snake Rivers. In the late 19th and early 20th centuries, non-Native people began intensive settlement on the Hanford Site, establishing an early settler and farming landscape. Farmstead communities existed from 1880 to 1943, located primarily in the upland environment adjacent to the Columbia River. The area became one of

the premier orchard regions in the state of Washington following formation of the Hanford Irrigation and Development Company in 1905.

The River Corridor includes approximately 8,300 acres of historical farmsteads, of which approximately 5,000 acres were orchard lands. Figure 1-4 shows the historical farmsteads and orchard lands within the 100-D/H area. Within the farmstead areas and specifically on the orchard lands, lead arsenate was applied as a pesticide. The farming life at Hanford came to an abrupt halt in 1943 when the U.S. government took possession of the land to produce weapons-grade plutonium as part of the Manhattan Project. Lead arsenate use in Washington State effectively terminated in 1948, when DDT dichlorodiphenyltrichloroethane became widely available to the public (*Re-establishing Apples Orchards in the Chelan-Manson Area* [Benson et al., 1969]).

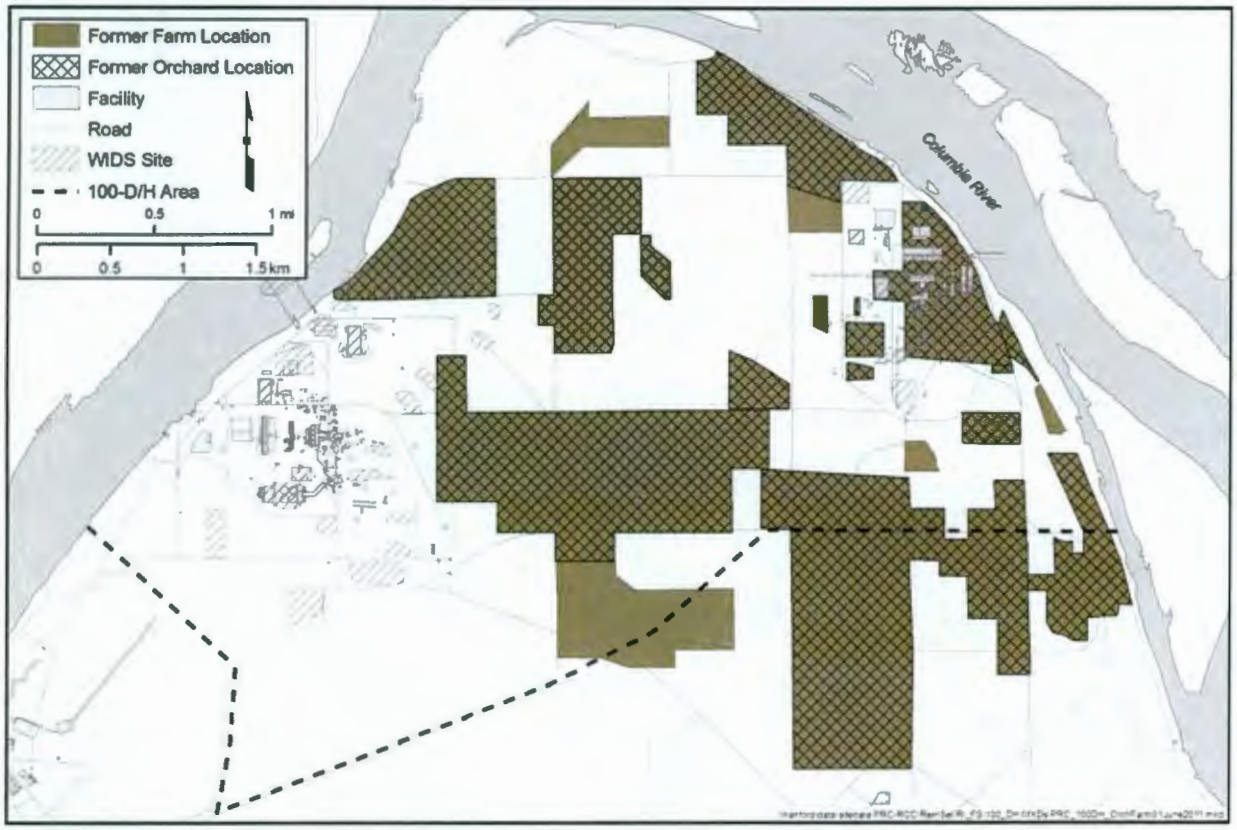

Figure 1-4. Historical Farm and Orchard Land Areas in 100-D/H

The persistence of residuals from lead arsenate that was applied as a pesticide before Hanford operations began is a concern that merits an assessment of potential effects to human health and the environment. To address this concern, the Tri-Parties have established the 100-OL-1 orchard lands OU (TPA Change Notice C-12-01). An RI of the 100-OL-1 OU will be conducted to determine if actions are needed to mitigate potential environmental or human health effects. If results from the RI indicate a need for action, an FS will be conducted to identify and evaluate a range of remedial alternatives.

1.2.1 Site Description

The Hanford Site is divided into numerically designated areas. The areas served as the location for reactor, chemical separation, and related activities for the production and purification of special nuclear materials and other nuclear facilities. The reactors and their ancillary/support facilities were located along the shore of the Columbia River in the 100 Area, because of the need for large quantities of water to cool

the reactors. Between 1943 and 1964, nine plutonium production reactors were built along the Columbia River in six areas: 100-BC, 100-K, 100-N, 100-D, 100-H, and 100-F.

The areas associated with this investigation are 100-D and 100-H.

The 100 Areas contained all of the reactors used to produce plutonium in fuel slugs irradiated in the reactor. The 100-D/H Area encompasses 20 km^2 (7.8 mi^2) in the northern portion of the Hanford Site in the 100 Area. 100-D/H includes two reactor areas (100-D and 100-H) and adjacent areas, as shown on Figure 1-2. Most of the waste sites associated with the investigation are in 100-D and 100-H. As shown on Figure 1-1, the Columbia River bounds about half of the site. The area between the reactor areas is commonly referred to as the "horn." The horn was used for agricultural purposes until 1943. A few isolated waste sites are located in the horn, but the area is relatively undisturbed. Appendix A includes detailed site plans of 100-D/H.

1.2.2 Hanford Site and Operational History

This section provides an overview of the history of the Hanford Site and summarizes the history of 100-D/H, including operational and process history. It describes the reactors and support facilities, cooling water systems, and radioactive and nonradioactive waste streams. It also describes the waste disposal facilities that were used during site operations and locations where contaminants were accidentally released. Finally, this section indicates the types of contaminants that are likely to be in various locations at 100-D/H, based on historical information and previous investigations.

1.2.2.1 Hanford Site History

The Hanford Site was selected for plutonium production for military nuclear weapons in 1942 as part of the Manhattan Project because of the availability of water from the Columbia River, access to power from the Bonneville and Grand Coulee Dams, its remote location, and its relatively small population. Land acquisition for the Hanford Site took place in February 1943 and was one of the largest land procurements (approximately 160,000 ha [400,000 ac]) carried out during World War II. Site construction, which began the following month, brought the first three reactors (B, D, and F) online by April 1945.

Between 1947 and 1955, the Atomic Energy Commission (AEC) added five new reactors (C, H, DR, KE, and KW) at the Hanford Site and boosted the output of the three Manhattan Project reactors (B, D, and F). The five new reactors were built with the intent of replacing some of the older Manhattan Project reactors, whose graphite blocks were showing signs of deformation, and increasing the plutonium output. Incremental improvements in the basic components of the World War II Manhattan Project reactors and a construction program that incorporated these improvements into the new reactors accounted for doubling the plutonium output at the Hanford Site in 1952 and 1953.

The period from 1956 through 1964 saw the most intense defense production at the Hanford Site, including the construction of a new dual-purpose reactor (N Reactor) capable of generating electricity and producing plutonium. Construction of the N Reactor, which featured a new closed-loop, primary cooling system, was completed in 1963, with plutonium production beginning in 1964. The N Reactor's 800-megawatt steam plant began producing electricity in 1966 and was the world's largest nuclear power plant for many years.

By the 1960s, however, the nation's plutonium stockpile was much larger than deemed necessary, and plutonium production at the Hanford Site gradually decreased. In 1964, the AEC shut down the H, DR, and F Reactors, followed by the D Reactor in 1967 and B Reactor in 1968. The C, KE, and KW Reactors were shut down in 1971. The N Reactor was shut down in 1986 and transitioned to cold standby in 1989, signaling the close of the Hanford Site's production mission and the start of its cleanup mission. During the Manhattan Project and Cold War, more than 67,000 kg (147,000 lb) of plutonium were produced at

the Hanford Site, 13,000 kg (29,000 lb) of which were fuel-grade plutonium. The Hanford Site produced the entire nation's nuclear arsenal plutonium between 1945 and 1963, and accounted for more than 65 percent of all plutonium in the history of U.S. plutonium production.

The environmental impacts associated with the ultimate disposition of the reactors were evaluated in the *Addendum (Final Environmental Impact Statement): Decommissioning of Eight Surplus Production Reactors at the Hanford Site, Richland, Washington* (DOE/EIS-0119F). The Environmental Impact Statement (EIS) ROD ("Record of Decision: Decommissioning of Eight Surplus Production Reactors at the Hanford Site, Richland, WA" [58 FR 48509], hereinafter called Reactor Decommissioning ROD) documented the selection of interim safe storage (ISS) for the reactors. (ISS is the provision of an upgraded, weather resistant shell to isolate the reactor core until remedial activities are conducted.) Following a period of up to 75 years for radioactive decay of short and intermediate half-life radionuclides, the reactors are planned to be disposed of in 200 Area Burial Grounds.

1.2.2.2 100-D/H Operations

Before beginning reactor construction at 100-D/H, the area supported orchard development, livestock grazing, and irrigated farming. Figure 1-5 is a 1941 aerial photo of the area before reactor construction. By the mid-1940s, the D Reactor was thought to be near the end of its life because of the growth and distortion of its graphite core.

The DR Reactor was built from late 1947 to early 1949 as a replacement for D Reactor. However, the D Reactor graphite distortion was controlled, and both reactors operated until 1964 when the DR Reactor was shut down. The D Reactor was shut down in 1967. Construction of H Reactor began in 1948, and the reactor operated from 1949 until 1965.

After reactor shutdown at 100-D/H, the 181-D Pump Station and the 182-D Reservoir were connected to the sitewide Export Water System to provide a backup water supply for the site. This system remains active today. The 182-D Reservoir is discussed in Section 3.8.1.

Fuel rods were placed in the nuclear fission reactors in the 100 Area and irradiated to transmute uranium to plutonium. The fuel was then taken to the 200 Area, where liquid chemical processes were used to separate plutonium and uranium from the fission products. Materials that had passed through the reactors for manufacture, or materials contacting items that had passed through the reactor, were radiologically contaminated with generally short-lived radioisotopes. These materials represented the majority of the waste produced. Active physical barriers and strong administrative measures were in place to minimize radiological hazards throughout the Hanford Site production areas. These measures affected the placement of disposal locations for various waste streams.

These materials became contaminants when they entered the environment, either through planned release, planned disposal, or through unplanned releases such as spills and leaks. Waste resulting from supporting production operations was disposed of in each area according to phase, quantity, radioactivity, and composition (for example, liquids, solids, high/low mass, or volume, high-level, low-level, strictly chemical, and septic). Thus, liquid and solid disposal locations were constructed and waste management practices were developed to manage these materials consistently among similar facilities at the Hanford Site (although practices changed over time). Liquid wastes from reactor operations and associated facilities were released to the vadose zone and the Columbia River. Solid wastes were disposed of in burial grounds associated with the facilities. More detailed discussions on the nature and extent of the contaminants associated with these processes are provided in Chapter 4. Sites for wastes intentionally or unintentionally released to or buried within 100-D/H included ponds, trenches, cribs, French drains, solid waste burial grounds, and unplanned releases, each of which is described in the following text.

Figure 1-5. 100-D/H in 1941 Before Reactor Construction

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- **Ponds.** Unlined, liquid discharge sites that were designed to receive low concentration liquid waste. Two typical 100-D/H Ponds are shown on Figure 1-6.

- **Trenches.** Shallow, narrow, unlined surface liquid waste sites of variable length that received sludge or liquid waste (cooling water, contaminated water and sludge, fuel rupture effluent, and decontamination solutions). Trenches typically were 15 to 40 m (50 to 130 ft) long, 3 to 5 m (10 to 17 ft) wide, and 2 to 6 m (6 to 20 ft) deep.

- **Cribs.** Subsurface liquid waste disposal sites for percolating wastewater into the ground without exposure to the atmosphere. The cribs typically were 3 × 3 × 3 m (10 × 10 × 10 ft) boxes, shored with wooden railroad ties and filled with gravel. Early waste management practices used cribs to receive low-level radioactive waste for disposal and to provide a physical barrier against surface exposure. Cribs received contaminated water and sludge, contaminated process tube effluent, fuel storage basin effluent, spent laboratory solutions, and potassium borate solutions.

Materials Used or Produced in 100-D/H

Many materials were used or produced in the reactor operations and related manufacturing processes in 100-D/H. These materials included the following:

- Process inputs:
  - Raw materials processed through the reactor, such as uranium fuel and cooling water
  - Process chemicals for water conditioning and inhibiting corrosion (for example, sodium dichromate, chlorine, and sulfuric acid)
  - Materials used for reactor maintenance, such as acids and solvents
- Waste products:
  - Radioactively and chemically contaminated materials (solid and liquid waste)
  - Radioactively and chemically contaminated cooling water

Figure 1-6. 100-D Area Ponds in 1992

- **French drains.** Subsurface liquid waste disposal sites designed to transport wastewater below the ground. These drains are usually built with a 1 m (3 ft) diameter, open (or gravel filled) pipe that is vertically placed less than 5 m (16 ft) below ground surface (bgs) (Figure 1-7).

- **Solid waste burial grounds.** Landfills (Figure 1-8) used to dispose of radioactive and nonradioactive hazardous substances, construction debris, contaminated equipment and soil, reactor parts, and burnable low-level radioactive materials. The 100 Area burial grounds may have also received pieces of spent nuclear fuel (*Estimates of Solid Waste Buried in 100 Area Burial Grounds* [WHC-EP-0087]; *Historical Events—Reactors and Fuels Fabrication* [RL-REA-2247]).

- **Unplanned release sites.** Waste sites caused by unplanned spills or releases from retention basins, ponds, pipelines (Figure 1-9), or other facilities and equipment (for example, tanks, tanker trucks, and transfer lines) used to handle liquid waste.

Figure 1-7. French Drain at Sodium Dichromate Railcar Unloading Station in 1997(Waste Site 100-D-12)

Figure 1-8. 118-H-1 Burial Ground Excavation in 2007

Figure 1-9. Extensive Excavation to Remove Pipelines near the D and DR Reactors (approximately 1996)

1.2.2.3 Reactor Mechanics and Layout

This section describes the mechanics and layout of the reactors and associated facilities. All reactor areas used the same nomenclature for numbering the reactors and associated facilities.

Reactors. The D and H Reactors were graphite-moderated, water-cooled units used to produce weapons-grade plutonium. Each reactor structure (Figure 1-10) includes a concrete foundation, steel base plate, cast iron bottom shield, cubical stack of graphite blocks, cast iron thermal shield walls/top, steel/Masonite® biological shield walls/top, and aluminum process tubes to hold the uranium fuel and carry the cooling water. Each reactor facility (designated as 105-D, 105-DR, and 105-H) includes a reactor block, control rod and safety rod facilities, reactor control room, fuel storage basin and associated fuel handling equipment, fans and ducts for the ventilation and recirculating gas systems, and supporting offices, shops, and laboratories.

1.2.2.4 Cooling Water

This section describes how cooling water was obtained and prepared for use in the reactors. It also describes the fate of the cooling water as it passed through the reactors and was subsequently discharged to the river or to the vadose zone.

A continuous supply of high-quality cooling water was essential to reactor operations to prevent reactor core damage from heat generated by the fission reactions. The D, DR, and H Reactors each used on average (over reactor operating lifetime) about 95,000 L/min (25,000 gal/min) of water obtained from the Columbia River. Over the lifetime of D and DR Reactor operations, approximately 4.5 trillion L

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(1.2 trillion gal) of cooling water were produced and used. At the H Reactor, approximately 2 trillion L (500 billion gal) of cooling water were used.

Figure 1-10. D Reactor during Construction (1943 to 1944)

Water Treatment. Water for reactor cooling was treated extensively before passing through the reactors. Figures 1-11 and 1-12 illustrate the general process in the D and H Reactors, respectively. The water treatment process shown is generally applicable to the two water treatment plants in 100-D and the water treatment plant in 100-H. The raw water was pumped from the Columbia River at the 181 River Pump House to the 182 Reservoir and Pump House, and then to the 183 Head House and Water Treatment Plant, where alum, sulfuric acid, sodium hydroxide, and chlorine were added (*100-D Area Technical Baseline Report* [WHC-SD-EN-TI-181]; *100-H Area Technical Baseline Report* [BHI-00127]). Sodium dichromate was added as a corrosion inhibitor later in the process at the 190 Process Pump House (100-D and 100-H) and at the 183-DR Water Treatment Plant (100-DR) as discussed below. In the 183 Water Treatment Plant (Figure 1-13), the water was subjected to chemical mixing, flocculation, settling, and filtration (through granular anthracite coal, sand, and gravel), and the treated water was stored in clear wells. Water for filter backwash was supplied from the clear wells by pumps in the 183 Water Treatment Plant. Backwash flowed through the filter media and was subsequently discharged through a waste valve into an "upstream" process sewer that discharged to the river at a 1904 or a 1907 Outfall (*Hazards Summary Report Volume 3 – Description of the 100-B, 100-C, 100-D, 100-DR, 100-F and 100-H Production Reactor Plants* [HW-74094 VOL3]).

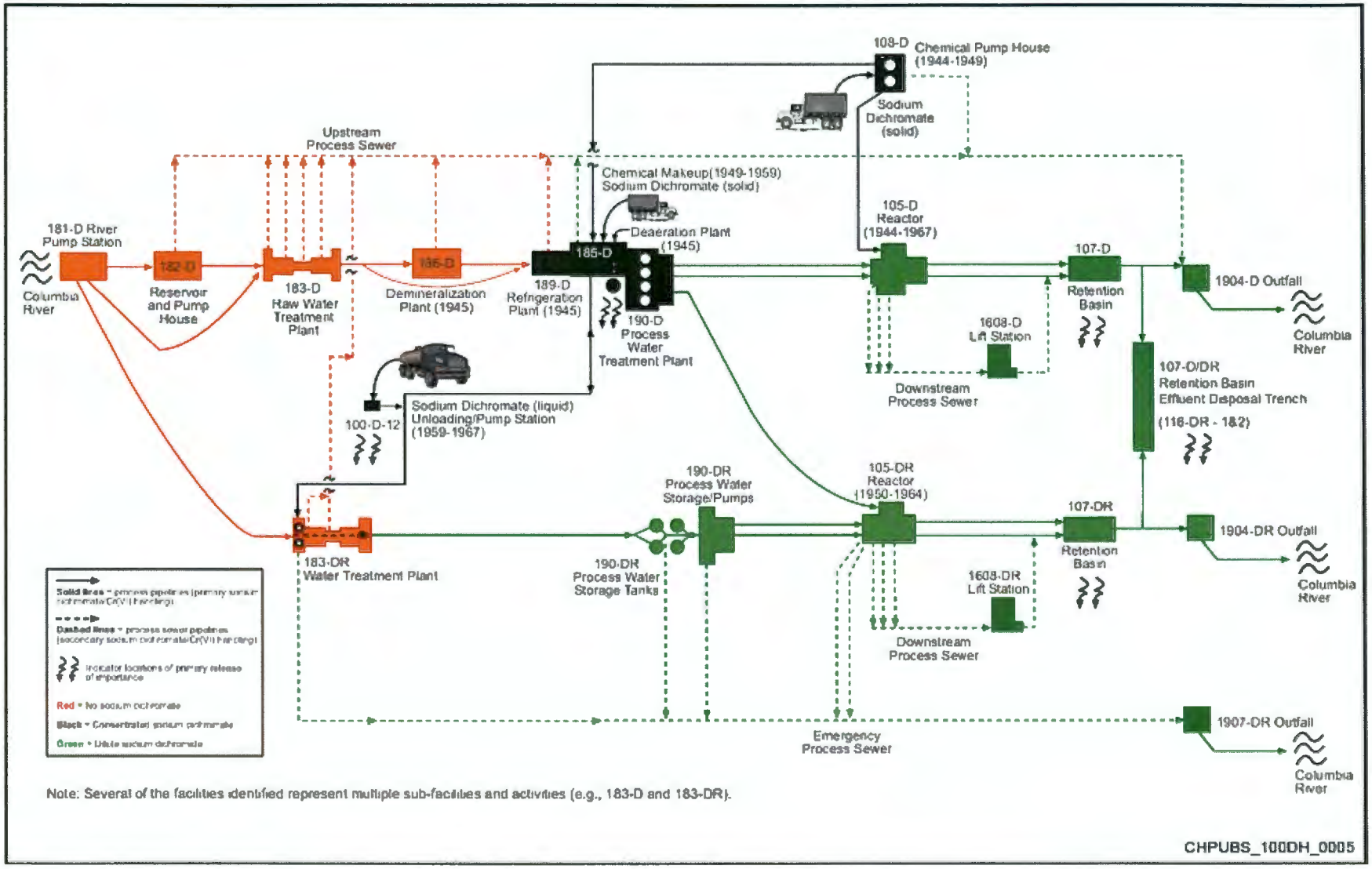

Figure 1-11. 100-D Water Treatment and Sodium Dichromate Use Flow Diagram

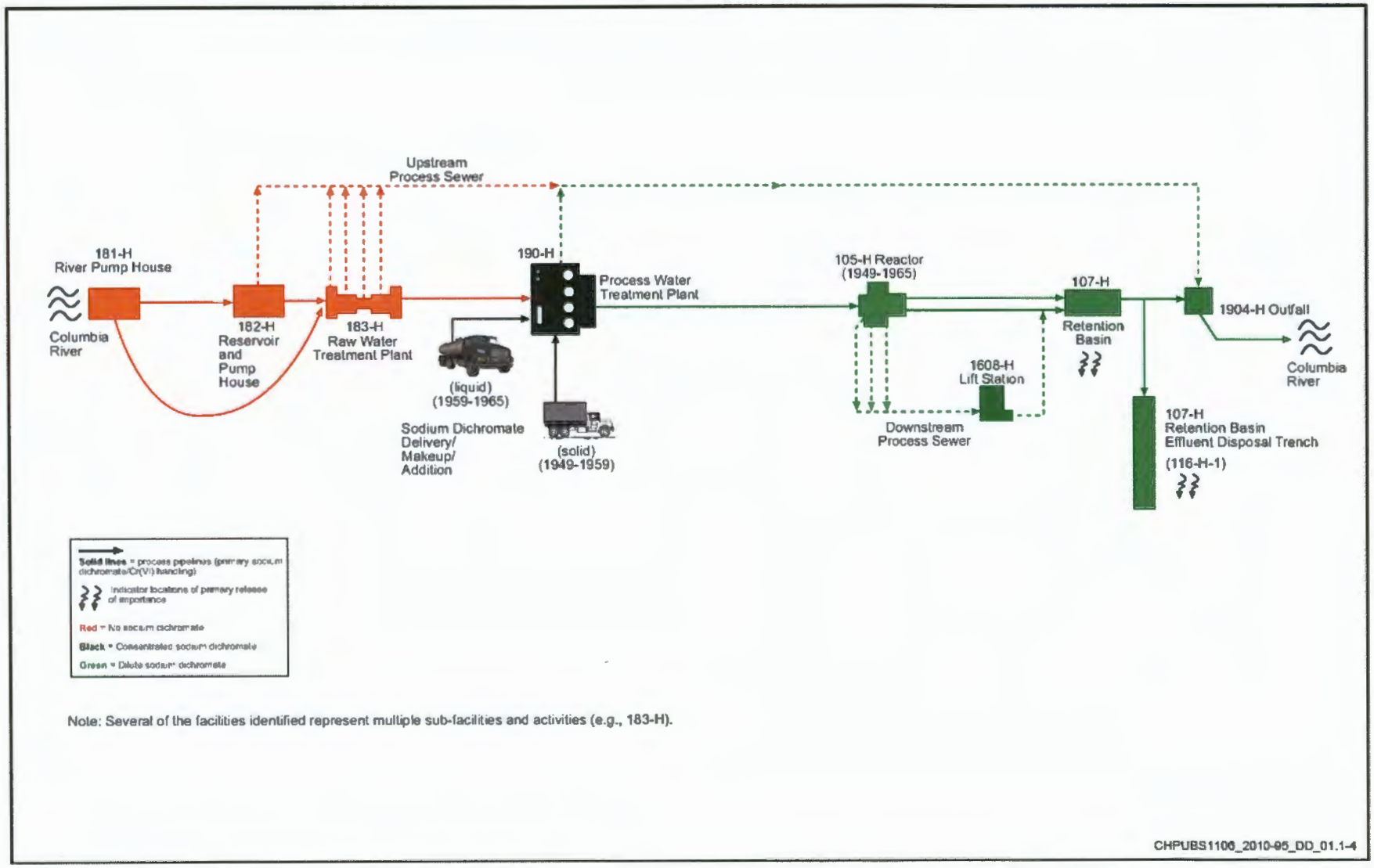

Figure 1-12. 100-H Water Treatment and Sodium Dichromate Use Flow Diagram

Figure 1-13. Aerial View of 183-D Water Treatment Plant in 1945

The 100-D upstream process sewers were nonradioactive and discharged at the 1904-D Outfall. The 100-DR and 100-H upstream process sewers received nonradioactive process sewer drains from the 183 and 190 facilities and some of the potentially low contamination work areas at the 105 Buildings and discharged to the 1904 Outfall. The process sewer to the 1907 Outfall also provided emergency DR Reactor cooling water sewers (that is, an alternative downstream [discharged from the reactor process] process sewer in the event of a failure of the primary downstream sewer to the 1904 Outfall).

The 186 Demineralization Plant was constructed as part of the water treatment system for D Reactor to remove dissolved calcium and magnesium salts, but it was never used for this purpose beyond startup tests at the D Reactor because the demineralization step was found to be unnecessary (*Manhattan Project Buildings and Facilities of the Hanford Site: A Construction History* [WHC-MR-0425]). The water from 186-D was sent to the 185-D De-aeration Plant to remove dissolved gases and entrained air (another step that was later found to be unnecessary) and was then pumped at the 190 Process Pump House to the 105 Reactor Building.

1.2.2.5 Sodium Dichromate Use

Sodium dichromate ($Na_2Cr_2O_7$-$2H_2O$), a corrosion inhibitor, was a key additive to cooling water at the 190 Process Pump House (100-D and 100-H) and at the 183-DR Water Treatment Plant (100-DR). Chromium was present in the dichromate anion with a +6 valence state (that is, Cr[VI]). More than

3,000 metric tons (3,300 tons) of Cr(VI) were used in conditioning the cooling water between 1945 and 1967 for the D and DR Reactors and about 1,200 metric tons (1,322 tons) for the H Reactor (100-D/H Work Plan [DOE/RL-2008-46-ADD1]). The reactor coolant had a near neutral pH, with Cr(VI) concentrations averaging 700 µg/L (2,000 µg/L as sodium dichromate).

From the D and H Reactors startup to final shutdown, the cooling water for each reactor contained 2,000 µg/L of sodium dichromate. The cooling water flow rate for each reactor was about 95,000 L/min (25,000 gal/min) from startup until 1955, when production began to increase, and cooling water flow for each reactor increased to about 190,000 L/min (50,000 gal/min) by the time the reactors were shut down. The following text summarizes the process of sodium dichromate preparation and use for reactor cooling water. The 100-D and 100-H facilities are shown in Appendix A.

100-D Area Sodium Dichromate Operations. The facilities and waste sites where sodium dichromate was handled at 100-D, based on process history information, are presented on Figure 1-14.

A 10 wt% sodium dichromate solution was used initially in 1944 (*Chemical Feed System for Process Water Treatment Buildings 105, 108 and 185* [SPEC-HW-2036]), which was increased to a 15 wt% sodium dichromate solution as the standard intermediate concentration by 1952 (*A Proposal for Liquid Sodium Dichromate Facilities for the 100-C and 100-D Areas* [HW-27270]). Batches of 10 to 1 wt% sodium dichromate solution were pumped from the 108-D Building via an underground pipeline to storage tanks in the 185-D Building. Batches of 10 or 15 wt% sodium dichromate water solution could also be transferred from the 108-D Building via an overhead pipeline to a storage tank in the 105-D Reactor Building valve pit.

The 10 to 15 wt% sodium dichromate solution was metered into the reactor cooling water stored in the 190-D and 190-DR storage/pump tanks to provide a sodium dichromate concentration of 1,800 to 2,000 µg/L. From there, it was pumped through the reactor facilities from 1944 through 1967. The 2,000 µg/L sodium dichromate reactor cooling water solution had a near neutral pH, with a concentration of Cr(VI) at about 700 µg/L. The sodium dichromate concentration in the reactor cooling water was reduced to 1,000 µg/L during reactor tests from 1964 to 1967 with a corresponding Cr(VI) concentration of 350 µg/L.

Concentrated sodium dichromate materials included solid sodium dichromate dihydrate ($Na_2Cr_2O_7$-$2H_2O$) and 70 wt% sodium dichromate-water solutions delivered to 100-D. Solid sodium dichromate dihydrate was received (in bags and/or drums) and processed in 100-D from 1944 until 1959. Shipments of 45 kg (100 lb) bags of solid sodium dichromate dihydrate were received at the 108-D Building from 1944 to 1950, then at the 185-D Building until 1955. Shipments of 226.8 kg (500 lb) drums of solid sodium dichromate dihydrate were received and stored at the 185-D Building from 1955 until 1959. Based on historical information for the 1713-DA Essential Materials Warehouse, supplies of 45 kg (100 lb) bags of solid sodium dichromate dihydrate also may have been stored at the 1713-DA Essential Materials Warehouse from 1944 until about 1955. It is not known when the 1713-DA Building was removed, but it was not seen in aerial photos after 1955. The shipments of bags and drums of solid sodium dichromate dihydrate were replaced with shipments of 70 wt% sodium dichromate water solutions beginning in 1959 and continued until D Reactor was shut down in 1967.

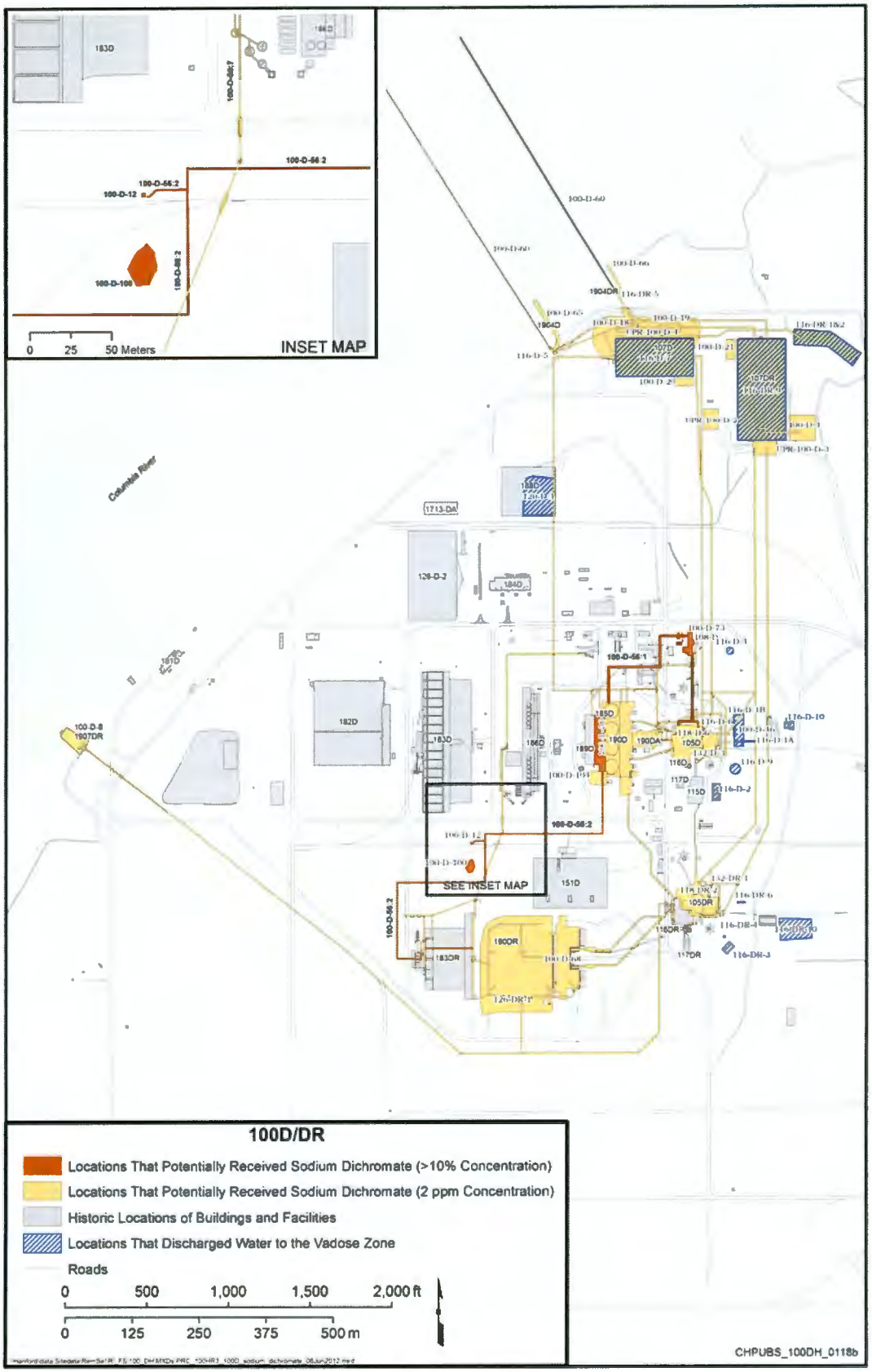

Figure 1-14. Facilities and Waste Sites Where Dichromate Was Handled at 100-D

In 1959, a tank truck/railroad car Unloading/Transfer Station (100-D-12) was installed adjacent to the railroad spur between the 183-D and the 183-DR Water Treatment Plants. The concentrated sodium dichromate solutions were transferred by hose from railroad cars or tanker trucks to the pumping facility (*100-D Area Technical Baseline Report* [WHC-SD-EN-TI-181]). The 100-D-12 Unloading/Transfer Station included a water dilution/mixing valve and a transfer pump (on a concrete pad) and an underground transfer pipeline (with isolation valves) that tied into the sodium dichromate underground transfer line from 185-D to 183-DR. The water dilution/mixing valve was used to dilute the delivered solutions to 70 wt% sodium dichromate, as necessary.

A 133,000 L (35,000 gal) storage tank was installed outside of the south side of the 185-D Building (Figure 1-14) to store the 70 wt% sodium dichromate solutions received at 100-D-12. A recirculation/transfer pump, valves, and piping connected the outside storage tank to sodium dichromate tanks inside the 185-D Building. Beginning in 1959, 70 wt% sodium dichromate solutions were transferred from the 185-D Building to the 183-DR inside storage tanks via underground piping. The isolation valves in the underground line allowed the alternate use of the line for transfers of 70 wt% solutions from the 100-D-12 Unloading/Transfer Station to the 185-D large storage tank, and from 185-D to 183-DR. From 1959 until 1964, the 183-DR Head House received 70 wt% sodium dichromate solutions, which were likely diluted to an intermediate 10 to 15 wt% concentration. From 1959 to 1967, the 70 wt% sodium dichromate solution was diluted to an intermediate concentration (that is, 10 to 15 wt% sodium dichromate solution) in the 185-D Building before feeding to the metering pumps in the 190-D Building.

100-H Area Sodium Dichromate Operations. The facilities and waste sites where sodium dichromate was handled at 100-H, based on process history information, are presented on Figure 1-15. The 100-H facilities are shown in Appendix A. Between late 1949 and early 1965, approximately 2 trillion L (500 billion gal) of reactor coolant containing 2,000 μg/L of sodium dichromate (except for the last year at 1,000 μg/L sodium dichromate) passed through H Reactor. The total amount of sodium dichromate in the reactor coolant volume was approximately 4 million kg (2 million lb), assuming a concentration of 2,000 μg/L, except for the last operating year when the sodium dichromate concentration was reduced to 1,000 μg/L. The residual footprint of the 190-H sodium dichromate handling area is addressed as the 100-H-46 waste site.

Multiple mixing steps to progress from highly concentrated sodium dichromate solutions to diluted reactor coolant solutions were not used at 100-H. River water was treated for impurities and pumped to the 190-H Building where sodium dichromate was added to make a cooling water solution of approximately 2,000 μg/L sodium dichromate. The bag-mixing process in the 190-H Building used solid sodium dichromate from 1949 to 1959 and 70 wt% sodium dichromate solutions from 1959 to 1965 ("Historical Information for 100-D/DR Area Uses of Chromic Acid and Sodium Dichromate: Supplement to IOM 129547" [Schwab, 2008]). In 1959, a 56,781 L (15,000 gal) horizontal storage tank was installed in the 190-H Building to receive, store, and supply a 70 wt% sodium dichromate solution to the batch mixing tanks also located in the 190-H Building.

Downstream from Reactors. Cooling water picked up other contaminants during passage through the reactors. These contaminants included activated elements in the water caused by the high neutron flux in the reactor cores (for example, calcium-41, chromium-51, and zinc-65), activation products from reactor components including the graphite reactor cores, steel process tube end pieces, process tubes, fuel cladding (for example, tritium, carbon-14, cobalt-60, nickel-63, europium-152, europium-154, and europium-155), fuel element fission products (for example, cesium-137 and strontium-90), and transmutation products (for example, plutonium-239 and -240). The total radioactivity of the reactor cooling water during normal operation was about 0.2 pCi/L (*100-D Area Technical Baseline Report* [WHC-SD-EN-TI-181]; *100-H Area Technical Baseline Report* [BHI-00127]).

Figure 1-15. Facilities and Waste Sites Where Dichromate Was Handled at 100-H

Cooling water effluent was near boiling after passing through the reactors (downstream) (*100-D Area Technical Baseline Report* [WHC-SD-EN-TI-181]; *100-H Area Technical Baseline Report* [BHI-00127]). During initial reactor operations, the effluent then traveled by pipeline to the 107-D (116-D-7), 107-DR (116-DR-9), and 107-H (116-H-7) Retention Basins where it was held for a short time (hours) to allow thermal cooling and very short-lived radionuclide decay. During the later years of reactor operation, the contaminated effluent was held on one side of a retention basin, then switched to the other side of the basin to provide longer holding times before river discharge. However, this "alternating side" process failed because of thermal stress between the hot and cold sides of the retention basins. Instead, the retention basins were operated as single units (that is, effluent flowed through both basin compartments) before river discharge. In addition, during the later years of operations, the effluent was redirected to nearby Pluto Cribs (replaced in the early 1950s by the 116-DR-1, 116-DR-2, and 116-H-1 liquid disposal trenches), following fuel cladding failures.

Sludge accumulated in the retention basins from the diatomaceous earth (solid slurry) purges used to clear process tube film buildup and from natural wind-blown sand accumulations. Sludge was removed and placed in 107 Basin Sludge Burial Trenches. The principal radionuclides detected in the Retention Basin Systems included europium-152, europium-154, europium-155, cobalt-60, cesium-137, and strontium-90 (short-lived radionuclides such as chromium-51 and zinc-65 have since decayed away) (*Radiological Characterization of the Retired 100 Areas* [UNI-946]).

The retention basins and effluent lines developed numerous documented leaks likely resulting from the thermal stress produced by the hot water exiting the reactors. The leakage rates at each retention basin were reportedly about 10,000 L/min (2,641 gal/min) (*100-D Area Technical Baseline Report* [WHC-SD-EN-TI-181]; *100-H Area Technical Baseline Report* [BHI-00127]; and *Radiological Characterization of the Retired 100 Areas* [UNI-946]). These leaks were not adequately repaired, and coolant loss to the subsurface was sufficient to create long-standing groundwater mounds in that area (*Unconfined Underground Radioactive Waste and Contamination* [HW-27337]; *Tabulation of Radioactive Liquid Waste Disposal Facilities* [HW-33305]; *Unconfined Underground Radioactive Waste and Contamination – 100 Areas* [HW-46715]; *Status of the Ground Water Beneath Hanford Reactor Areas: January, 1962 to January, 1963* [HW-77170]; *Radiological Characterization of the Retired 100 Areas* [UNI-946]).

From the retention basins, the effluent was transferred through large pipes to the 116-D-5 (1904-D), 116-DR-5 (1904-DR), and 116-H-5 (1904-H) Outfall structures and then into pipes that discharged at the bottom center of the Columbia River. The 100-D and 100-DR effluent lines pass through D Island (Figure 1-16) while the 100-H effluent line is further downstream. Overflow from all three-outfall structures could also discharge directly to the shore of the river through nearby spillways.

Significant Releases from Retention Basins and Effluent Lines

Several significant releases from the retention basins and effluent lines have been documented (*100-D Area Technical Baseline Report* [WHC-SD-EN-TI-181]):

- In 1950, a major leak from the northern side of the 107-D (116-D-7) Retention Basin became evident by the presence of water between the road and the fence line.
- In 1951, the 107-DR (116-DR-9) Retention Basin experienced excessive leakage at the inlet. In 1953, some of the contaminated soil from this leak was used as fill material at the southern end of the basin.
- In 1951, leaks were reported along the effluent lines approximately 46 m (150 ft) southeast of the 107-D (116-D-7) Retention Basin, contaminating the immediate vicinity of the basin.
- In 1967, a field test was conducted where the entire reactor effluent volume was discharged to the 116-DR-1/2 Trench for several months.

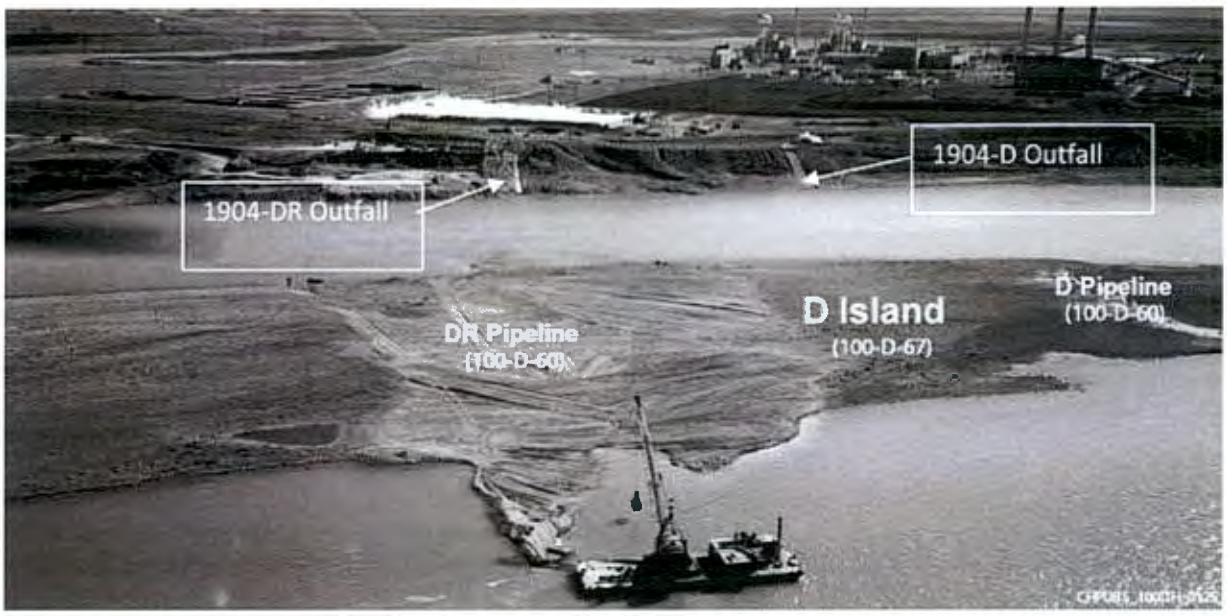

Figure 1-16. Aerial View of D and DR River Effluent Pipelines and Outfalls in 1956

During production, fuel element and infrastructure failures (for example, pipe leaks) led to releases of radiologically and chemically contaminated materials to the environment (*Unconfined Underground Radioactive Waste and Contamination* [HW-27337], *Tabulation of Radioactive Liquid Waste Disposal Facilities* [HW-33305], *Unconfined Underground Radioactive Waste and Contamination – 100 Areas* [HW-46715]; *Radiological Characterization of the Retired 100 Areas* [UNI-946]). Most fuel-element failures involved natural uranium or enriched uranium fuels. Fuel cladding failures occurred when corrosion or swelling of the aluminum cladding covering a uranium fuel slug caused the cladding to break open, releasing uranium oxide particles that contained plutonium isotopes and fission products (cesium-137 and strontium-90) into the cooling water.

During their years of operation, several hundred fuel cladding failures occurred at each of the D and H Reactors (*Fuel-Element Failures in Hanford Single-Pass Reactors, 1944-1971* [PNWD-2161 HEDR]; *100-D Area Technical Baseline Report* [WHC-SD-EN-TI-181]; *100-H Area Technical Baseline Report* [BHI-00127]). Fuel cladding failures resulted in highly radioactive cooling water that was released to the soil column or to the Columbia River. Fuel cladding failures also occurred within the fuel storage basins themselves. Some leakage was reported for the 100-D and 100-H fuel storage basins (*100-D Area Technical Baseline Report* [WHC-SD-EN-TI-181]; and *100-H Area Technical Baseline Report* [BHI-00127]); however, the leak rate was small, and the leak location was never identified. At D Reactor, however, at least one fuel storage basin leak was documented as being present at the rear of the building in May 1957.

From the early 1950s until the reactors were shut down in the mid-1960s, the highly radioactive water was segregated and drained to the 116-DR-1 Basin Trench (107 Liquid Disposal Trench), 116-DR-2 Basin Trench (107-DR Liquid Waste Disposal Trench), and 116-H-1 Basin Trench (107-H Liquid Waste Disposal Trench). Some other facilities received smaller quantities of radioactively contaminated water from the 105 Fuel Storage Basins and from special maintenance or process tests via the downstream process sewer and the 1608 Pump House (the downstream process sewer lift station), including two trenches east of D Reactor (116-D-1A and 116-D-1B).

From the late 1940s through the late 1960s, the 105 Storage Basin Trenches received water and sludge from the fuel storage basins. The trenches were typically about 30 m (100 ft) long, 3 m (10 ft) wide, and covered with 2 to 5 m (6 to 15 ft) of soil.

1.2.2.6 Other Radioactive Waste Streams

These waste streams included decontamination solutions, sludge, solid waste, and air emissions. Although cooling water was the dominant waste stream at 100-D/H because of the quantities used, other radioactive and chemical waste streams contributed to the contamination observed in 100-D/H soil and groundwater. Figure 1-17 presents the facilities and waste sites where strontium-90 was known to be present at 100-H, based on process history information. One well at 100-D (199-D-5-12) had historical readings of strontium-90 above the DWS until it went dry. Well 199-D-5-132 was drilled in a similar location as 199-D-5-12 and confirmed that the strontium-90 is still above the DWS. This is discussed in Chapter 4.

The miscellaneous cribs and trenches designation includes facilities (except 107 Liquid Waste Disposal Trenches) within 60 m (200 ft) of the reactor buildings that were used for disposal of liquid waste resulting from fuel failures, decontamination activities, and liquid and sludge from the irradiated fuel storage basins (*Radiological Characterization of the Retired 100 Areas* [UNI-946]). The principal radionuclides associated with these facilities include cobalt-60, cesium-137, strontium-90, europium-152, europium-154, europium-155, carbon-14, and tritium. The 117 Cribs at the D, DR, and H Reactors received low-activity radioactive condensate and water seal water drainage from 117 Building seal pits. The 108 Cribs (116-D-3 and 116-D-4) were underground French drains covered with approximately 2.5 m (8 ft) of soil that received contaminated liquid effluents from the 108-D Building, which housed the 100-D main maintenance shop. The liquid waste included contaminated water, decontamination solutions, solvents, and low-level fission products (*100-D Area Technical Baseline Report* [WHC-SD-EN-TI-181]). The 116-D-3 Crib also received effluent from a cask decontamination pad at the 108-D Building (*Radiological Characterization of the Retired 100 Areas* [UNI-946]).

Decontamination Solutions. Decontamination solutions were used routinely to clean facility equipment and surfaces at 108-D Building, at the reactor fuel slug decontamination facilities decontamination stations in 100-D, and at the H Reactor fuel slug decontamination facility wash pad next to the 105-H Fuel Storage Basin. Known decontamination solutions included chromic, citric, oxalic, nitric, sulfamic, and sulfuric acids (neutralized with sodium carbonate before disposal), and sodium fluoride. Other chemicals, including organic solvents, were used in some decontamination processes at the D and H Reactors at various times and locations (*Hazard Ranking System Evaluation of CERCLA Inactive Waste Sites at Hanford* [PNL-6456]; *100-D Area Technical Baseline Report* [WHC-SD-EN-TI-181]; *100-H Area Technical Baseline Report* [BHI-00127]).

Decontamination solutions contained both radionuclide and chemical contaminants and were generally disposed of in cribs, trenches, or French drains near the buildings in which they were used. Occasionally, decontamination solutions were routed to the downstream process sewer that drained to the 1608 Waste Water Pump House (lift station); the decontamination solutions were combined with the cooling water before being discharged to the river via the retention basins. The 105-DR and 105-H facilities also had process drains to an emergency process sewer that drained to the 1907 Outfall. The process sewer to the 1907 Outfall also received waste from the 183 Water Treatment Plant. The Cr(VI) concentrations in these solutions and volumes discharged to cribs and drains are not known. Near the 108-D Building, decontamination solutions were discharged into two small cribs, 116-D-3 (1951 to 1967) and 116-D-4 (1956 to 1967) (*Technical Activities Report Heat, Water, and Mechanical Studies* [HW-22346]). Laboratory solutions derived from corrosion tests also included Cr(VI) and were disposed of in the 116-D-4 Crib.

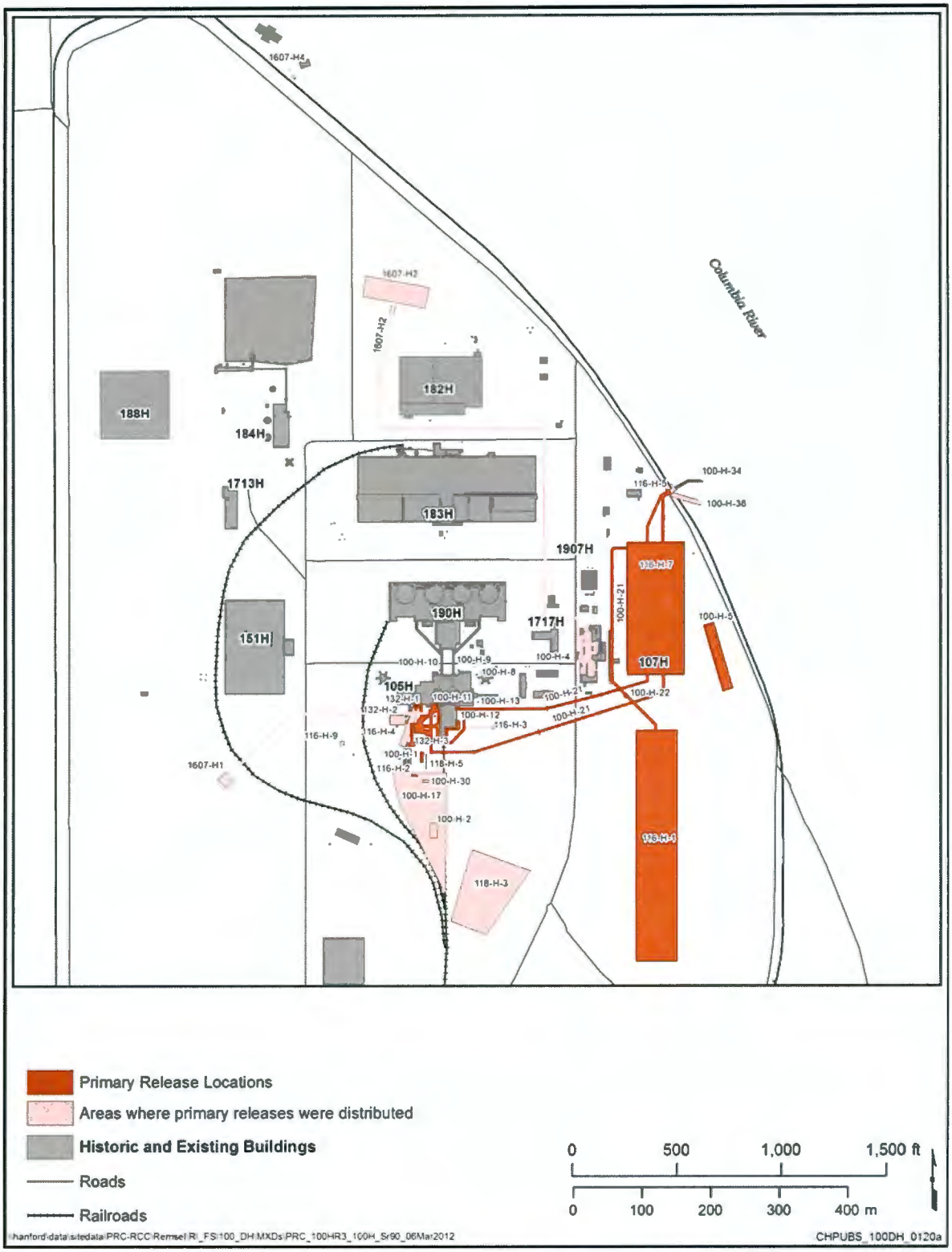

Figure 1-17. 100-H Strontium-90 Potential Sources

Burial Grounds. Burial grounds were used for the disposal of solid waste. The primary radionuclides at these locations were cobalt-60 and europium-152, although europium-154, europium-155, cesium-134, cesium-137, strontium-90, and nickel-63 are present (*Radiological Characterization of the Retired 100 Areas* [UNI-946]). Radioactive solid waste consisted of reactor components, contaminated equipment, tools, air filters, and miscellaneous contaminated items. This waste was primarily disposed of in the 118-D-1, 118-D-2, 118-D-3, 118-D-5, 118-H-1, and 118-H-3 Burial Grounds.

Other radioactive solid waste buried "in place" (that is, not at burial grounds) at 100-D/H included building foundations, belowgrade concrete structures, and other materials from demolished buildings. Starting in the 1970s, most 100 Area solid waste was transferred to the 200 Area burial grounds (*100-D Area Technical Baseline Report* [WHC-SD-EN-TI-181]; *100-H Area Technical Baseline Report* [BHI-00127]).

Air Emissions. A carbon dioxide and helium gas atmosphere was maintained around the reactor cores. Facilities supporting air treatment and air handling processes included the 115 and 117 Buildings, the 116 Stacks, and belowgrade tunnels. The reactor buildings were connected to the 115 and 117 Buildings via belowgrade concrete tunnels and ventilation ductwork.

Reactor and support facility ventilation systems first provided fresh air to staff areas, then to zones of increasing contamination, and finally to exhaust stacks. As the air passed through the reactor areas, it became contaminated with carbon-14, iodine-129, and tritium from radioactive gases, contaminated water vapor, and airborne particles. Originally, contaminated ventilation air was released directly to the atmosphere via 61 m (200 ft) tall concrete stacks (the 116 Stacks). However, air filtration systems were installed in 1960 to minimize the release of radionuclides. Two types of filter banks were used: a high efficiency particulate absolute bank and a halogen (activated charcoal) bank (*100-D Area Technical Baseline Report* [WHC-SD-EN-TI-181]; *100-H Area Technical Baseline Report* [BHI-00127]). Sections of the 115 and 117 Buildings and their associated tunnels and ductwork were contaminated with cobalt-60, cesium-134, cesium-137, carbon-14, tritium, europium-152, europium-154, europium-155, iodine-129, and strontium-90.

1.2.2.7 Nonradioactive Waste

This section summarizes the types of nonradioactive waste generated at 100-D/H. This waste had the potential to contribute to soil and groundwater contamination and included coal-fired powerhouse waste, septic system waste, a variety of other liquid wastes, and solid waste.

Hydroelectric power and coal were used as sources of energy for the Hanford Site. The D, DR, and H Reactors themselves did not generate electricity. Anthracite coal was stored in coal pits. Ash slurry (water based) from the coal fired 184-D and 184-H Power Houses was transported by pipeline to the 126-D-1 and 126-H-1 Ash Pits (Figure 1-18). Leakage in the pipeline and seepage in the ash pits were potential liquid contamination sources (*100-D Area Technical Baseline Report* [WHC-SD-EN-TI-181]; *100-H Area Technical Baseline Report* [BHI-00127]).

Sanitary liquid waste was routed by sewer lines to septic systems. Five septic systems were in 100-D and four in 100-H. There are no records of radiological waste being disposed of to these systems; however, detergents, cleaning compounds, and solvents may have been disposed of that contributed to local nitrate contamination in the vadose zone and groundwater. In addition, fertilizer use on pre-Hanford Site agricultural lands likely contributed to local 100-D/H nitrate contamination. The nitrate plumes are presented on Figures 1-19 and 1-20. The facilities and pre-Hanford Site agricultural lands where nitrate was known to be present at 100-D and 100-H, based on previous historical information, are also presented on Figures 1-19 and 1-20.

Figure 1-18. 184-D Power House with Coal Pit (above) and Ash Pit (to the right) in 1944

Additional nonradioactive liquid waste, including hazardous waste and hazardous substances, were used at various areas and discharged to liquid waste sites (*100-D Area Technical Baseline Report* [WHC-SD-EN-TI-181]; *100-H Area Technical Baseline Report* [BHI-00127]). The areas where liquid waste was handled and disposed of included the following:

- Oil containing polychlorinated biphenyls (PCBs) in electrical transformers and hydraulic machinery
- Boiler water treatment chemicals for the 184-D and 184-H Power Houses (for example, sodium sulfate, tri-sodium phosphate, and chromates) that ended up in boiler sludge (for which disposal methods are not known)
- Zeolite water softener regeneration solutions containing salt were disposed of to the upstream process sewer from the 184-D and 184-H Power Houses
- Fuel oil and diesel stored in underground and aboveground tanks just west of the 184-DR Steam Generating Building, at the confluence of the railroad tracks north of the 184-D Power House, and at 184-H located between 105-H and 190-H (waste sites 100-H-48 and 100-H-52)
- Gasoline and batteries for emergency electrical power (gasoline was stored in an aboveground tank at the rear of the 1621-D Facility)
- Oils, paints, and solvents used and stored in the 1714-D, 1715-D, 1716-D, 1717-D, 1722-D, 1715-H, 1716-H, 1717-H, and 1722-H Buildings
- Fluids from automotive repair and service performed at the 1716-D and 1716-H Buildings
- Additional wastewater generated from various cleaning processes (for which disposal locations are unknown)

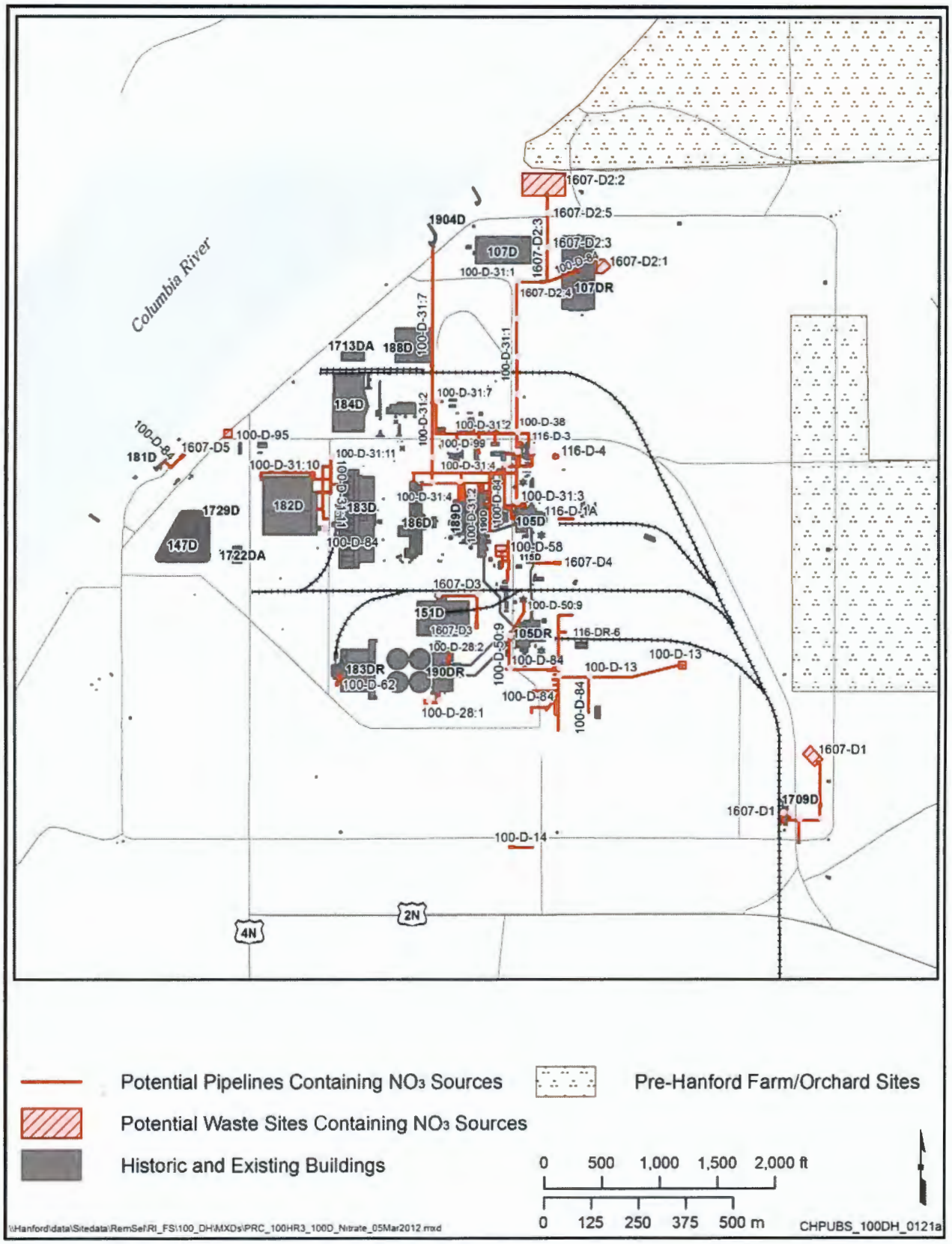

Figure 1-19. 100-D/DR Nitrate Potential Sources

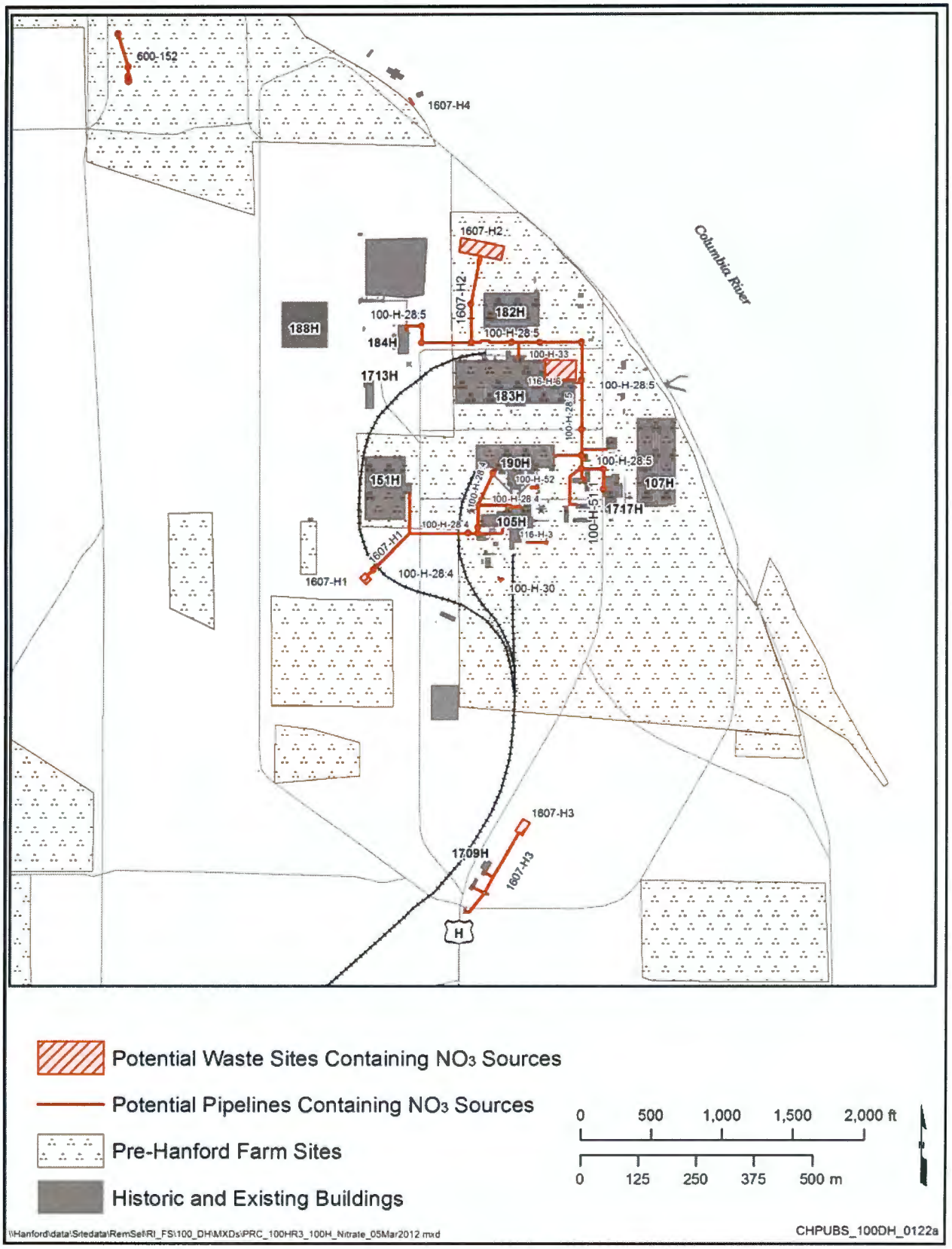

Figure 1-20. 100-H Nitrate Potential Sources

Nonradioactive solid waste included paper, trash, pieces of metal, and plastic parts. Some combustible waste was disposed of at the 128-D-2, 628-3, and 128-H-1 Burn Pits. Other solid waste consisted of relatively uncontaminated concrete, metal parts, and other materials from decontamination and demolition activities.

1.2.3 Previous and Ongoing Investigations and Remediation

This subsection summarizes the significant investigation and remediation activities for facilities, waste sites, and groundwater at 100-DH. Since the beginning of reactor operations, investigations were conducted to determine impacts to the environment, including the Columbia River. With the issuance of the TPA in 1989 (Ecology et al., 1989a), investigation activities transitioned to CERCLA cleanup activities, which have been ongoing to protect HHE within the River Corridor including 100-DH. Investigations and remediation activities were carried out pursuant to various remedial and removal action decision documents for facilities, waste sites, and groundwater.

The relevant data and conclusions from investigations and remediation activities (see Appendix B) provide supporting information that is analyzed and evaluated in this RI/FS. The following are examples of the various data sets used to develop this RI/FS:

- Vadose zone contaminants
- Groundwater contaminants
- Geologic contact information, fate and transport parameters (e.g., distribution coefficient [K_d] dispersivity, hydraulic conductivity, and soil bulk density)
- Well and borehole information (e.g., drill depth, screen length, and screen depth)
- Groundwater elevations and river stage
- Geographic information system shape files (e.g., aerial photography, Columbia River, and locations of wells and boreholes, salmon redds, facilities, roads, and waste sites)

Characterization of the vadose zone and associated waste sites has been an important consideration in Hanford plant operations since the 1940s. Some early reports (e.g., *Underground Waste Disposal at Hanford Works: An Interim Report Covering the 200 West Area* [HW-9671]; *The Underground Disposal of Liquid Wastes at the Hanford Works* [HW-17088]) examined the issues related to waste disposal at injection wells, shallow burial cribs, and surface ponds. Early characterization efforts combined well drilling and geophysical logging to evaluate rates of contaminant migration in the vadose zone and in the aquifer. Little attention was focused on nonradionuclides.

Radiological Characterization of the Retired 100 Areas (UNI-946) presents the results of vadose zone investigations in 1975 at solid and liquid waste sites. Soil samples were collected and analyzed mainly to determine the inventory of radionuclides in retention basins and in the vadose zone. In general, up to 70 percent of the radionuclide inventory was determined to be within the retention basins.

Analytical data used in this RI/FS (provided in Appendix D) include the data reduction protocols and QA reports. Summaries of facility demolition activities, vadose zone investigation and remedial activities, groundwater investigation and remedial activities, and previous risk assessments are provided below.

The various 100-D/H decision documents are summarized in Table 1-2. Appendix B presents an annotated bibliography of the related CERCLA documentation for the River Corridor.

Table 1-2. Summary of Decision Documents for 100-D/H

Decision Document	Summary
Reactors	
Action Memorandum: USDOE Hanford 100 Area National Priorities List, 105-F and 105-DR Reactor Buildings and Ancillary Facilities, Hanford Site, Benton County, Washington (Wagoner et al., 1998), July	Calls for decontamination and demolition of the contaminated reactor buildings (except for the reactor blocks) and ancillary facilities, and disposal of the waste. Calls for ISS enclosure over reactor blocks.
Action Memorandum: USDOE Hanford 100 Area National Priorities List, 105-D and 105-H Reactor Facilities and Ancillary Facilities, Hanford Site, Benton County, Washington (Wilson and Klein, 2000), December	Calls for decontamination and demolition of the contaminated reactor buildings (except for the reactor blocks) and ancillary facilities, and disposal of the waste. Calls for ISS enclosure over reactor blocks.
Source Operable Units	
Interim Remedial Action Record of Decision for the 100-BC-1, 100-DR-1, and 100-HR-1 Operable Units, Hanford Site, Benton County, Washington (EPA/ROD/R10-95/126), September 1995	Sets forth two approaches to remediation: • **Observational approach**—relies on historical information and LFIs. • **Plug-in approach**—allows for selection and application of remedial actions at similar sites. Selected remedial actions include the following: removal of contaminated soil, structures, and debris using the observational approach; treatment by thermal desorption or soil washing; disposal at the ERDF; and backfill followed by revegetation.
Record of Decision for the 100-IU-1, 100-IU-3, 100-IU-4, and 100-IU-5 Operable Units Remedial Action, Hanford Site, Benton County, Washington (EPA/ROD/R10-96/151), February 1996	This ROD addressed a waste site in the horn area (identified as 100-IU-4), which was a sodium dichromate barrel landfill. The ROD said no further action was needed following RTD to residential cleanup standards.
Amendment to the Interim Remedial Action Record of Decision for the 100-BC-1, 100-DR-1, and 100-HR-1 Operable Units, Hanford Site, Benton County, Washington (EPA/AMD/R10-97/044), April 1997	Addition of 34 waste sites throughout 100-BC, 100-D, 100-F, 100-H, and 100-K to previous ROD; termination of the soil-washing step for volume reduction; and emphasis on revegetation of remediated waste sites.
Interim Action Record of Decision for the 100-BC-1, 100-BC-2, 100-DR-1, 100-DR-2, 100-FR-1, 100-FR-2, 100-HR-1, 100-HR-2, 100-KR-1, 100-KR-2, 100-IU-2, 100-IU-6, and 200-CW-3 Operable Units, Hanford Site, Benton County, Washington (100 Area Remaining Sites) (EPA/ROD/R10-99/039), July 1999	Removal, treatment, and disposal of contaminated soil, structures, and debris for sites where sufficient information exists; plug-in approach for sites with limited information that meet the waste site profile; disposal of equipment and debris from 105-B, 105-D, 105-H, 105-KE, and 105-KW Reactor Buildings consistent with previous CERCLA disposal for areas associated with the 105-C, 105-F, and 105-DR Reactor Buildings.

Table 1-2. Summary of Decision Documents for 100-D/H

Decision Document	Summary
Interim Remedial Action Record of Decision for the 100-BC-1, 100-BC-2, 100-DR-1, 100-DR-2, 100-FR-2, 100-HR-2, and 100-KR-2 Operable Units, Hanford Site (100 Area Burial Grounds), Benton County, Washington (EPA/ROD/R10-00/121), September 2000	Selected remedies include the following: remove contaminated soil, structures, and associated debris; treat this waste as required to meet disposal facility requirements; dispose of contaminated materials at the ERDF; and backfill excavated areas with clean material, followed by revegetation.
Explanation of Significant Differences for the 100 Area Remaining Sites Interim Remedial Action Record of Decision (EPA et al., 2004), April	Addition of 28 waste sites. Addition of "Compliance with Floodplain and Wetland Environmental Review Requirements" (10 CFR 1022) and "Procedures for Implementing the National Environmental Policy Act and Assessing the Environmental Effects Abroad of EPA Actions" (40 CFR 6). Revise institutional controls in accordance with *Sitewide Institutional Controls Plan for Hanford CERCLA Response Actions* (DOE/RL-2001-41).
Groundwater OUs	
Record of Decision for the 100-HR-3 and 100-KR-4 Operable Units Interim Remedial Actions, Hanford Site, Benton County, Washington (EPA/ROD/R10-96/134), March 1996	Initiates the use of ion exchange technology to remove Cr(VI) from groundwater using a system of extraction and injection wells.
Interim Remedial Action Record of Decision Amendment for the 100-HR-3 Operable Unit, Hanford Site, Benton County, Washington (EPA/AMD/R10-00/122), October 1999	Alters the selected remedial action by deploying a new technology (in situ oxidation reduction [redox] manipulation [ISRM]) for remediation of the Cr(VI) plume in 100-D.
Explanation of Significant Difference for the 100-HR-3 Operable Unit, Hanford Site, Benton County, Washington (EPA/ESD/R10-03/606), April 2003	Revises the project schedule and cost estimate associated with the ISRM barrier. Explains that addition of an evaporation pond also invokes an additional ARAR.
Explanation of Significant Differences for the 100-HR-3 and 100-KR-4 Operable Units Interim Action Record of Decision: Hanford Site Benton County, Washington (EPA et al., 2009)	The cost and schedule for the remedy are revised to reflect the expanded DX and HX groundwater pump-and-treat systems. A total of 15 known waste sites and 47 candidate waste sites are added to the scope. The explanation of significant difference also provides for the reinjection of treated water to downgradient locations to contain plumes.

Note: Complete reference citations are provided in Chapter 11.

1.2.3.1 Previous Facility Demolition Activities

Since its original construction, 100-D/H has included 128 facilities, including 3 reactors, storage buildings, offices, retention basins, maintenance shops, process plants, an electric substation, storage tanks, pump stations, and outfall structures. Until the structures over a source site have been removed, no soil remediation can be completed. Therefore, the facilities (including contaminated pipelines associated

with them) are, and have been, undergoing removal to clear the way for the remedial work that focuses on contamination in the vadose zone. In addition, facility decontamination, demolition and disposal removes potential sources of continuing release to the environment as many facilities contain contaminants of concern.

The facilities at 100-D and 100-H were the first to be declared excess after their reactors were shut down starting in 1965 (Figure 1-21). Follow-on housekeeping and decommissioning activities began in 100-D and 100-H as part of a Sitewide initiative in 1973, after deactivation of the reactors. This activity progressed as resources allowed, from 1974 through 1990, with buildings demolished, surplus equipment salvaged or redeployed, and active operations maintained at a minimal level.

D, DR, and H Reactors. As stated in the 1993 Reactor Decommissioning ROD (58 FR 48509), DOE regards the safe storage of the reactors followed by deferred dismantlement, safe storage followed by one-piece removal, and immediate one-piece removal alternatives as equally favorable based solely on the evaluation of environmental impacts. DOE uses the CERCLA process to decommission and dismantle reactors based on the joint EPA/DOE policy on decommissioning signed in 1995 and incorporated into the TPA (Ecology et al., 1989a). Since the Reactor Decommissioning ROD (58 FR 48509) in 1993, documentation has been prepared and implemented under CERCLA resulting in placement of six of the eight surplus reactors (C, D, DR, F, H, and K) into ISS designed to prevent deterioration and release of contamination from the reactors. ISS for the three reactors was complete in 2005 to ensure the safety of the reactors for up to 75 years as part of the remediation activities in 100-D/H. Figure 1-22 shows the D and DR Reactors in their ISS configuration. Figure 1-23 shows H Reactor in its ISS configuration.

Figure 1-21. D and DR Reactors following Shutdown, before ISS (2000)

Figure 1-22. D and DR Reactors following ISS (circa 2005)

Figure 1-23. H Reactor after ISS (2005)

DOE decided to broaden the decommissioning approach for these eight surplus reactors, including the D, DR, and H Reactors, retaining the deferred one-piece removal option, as selected in the Reactor Decommissioning ROD (58 FR 48509), and considered an immediate dismantlement option in a Supplemental Analysis. "Amended Record of Decision for the Decommissioning of Eight Surplus Reactors at the Hanford Site, Richland, WA" (75 FR 43158), issued on July 23, 2010, confirms the one piece removal alternative and includes the additional alternative of an immediate dismantlement. The D, DR, and H Reactors removal will be conducted as CERCLA non-time-critical removal actions.

DOE evaluated the coordination of decommissioning actions with the completion of interim field remediation for 100-D/H. Based on October 2012 field remediation information, all waste sites in the immediate vicinity of the reactors are interim closed-out according to the Interim Action ROD. Until reactor removal is complete, DOE will continue to conduct routine reactor maintenance, surveillance, and radiological monitoring to ensure continued HHE protection during the ISS period. All other waste sites in the immediate vicinity of the reactors have been remediated according to the interim action ROD. It is not anticipated that remedial action will be needed for contaminant releases from the ISS reactor structures before their removal.

Fuel Storage Basins and Facilities. Upon closure of the reactors, the 105-D and 105-DR Fuel Storage Basin shielding water elevations were lowered; and the basins were cleaned out in 1984. Miscellaneous basin equipment and sludge were packaged and disposed in the 200 Area burial grounds (*Fuel Storage Basins Cleanup and Stabilization Project Report* [UNI-3958]). The remaining shielding water was processed and released to two ponds (116-D-10 and 116-D-11) that were excavated specifically to receive the treated shielding water, and an asphalt emulsion was applied to the basin walls and floors in an effort to fix any remaining contamination. The 105-H Fuel Storage Basin was drained to within 1.2 m (4 ft) of the basin bottom and was backfilled with soil, burying the sludge and miscellaneous equipment that remained in the basin (*Radiological Characterization of the Retired 100 Areas* [UNI-946]).

The contaminants in the discharged water included water treatment chemicals and radioactive isotopes dissolved in the cooling water from breached fuel cladding. Primary contaminants in fuel storage basin sludge samples included cesium-137, nickel-63, cobalt-60, and europium-152, -154, and -155, although plutonium-238, plutonium-239/240, tritium, strontium-90, and uranium were also present. Primary contaminants in fuel storage basin water samples were cesium-137 and strontium-90, although plutonium-238, plutonium-239/240, europium-155, and uranium were also present (*Radiological Characterization of the Retired 100 Areas* [UNI-946]).

Remaining building and facility waste exists as contamination in demolished ductwork, concrete, paint, equipment, insulation, and remaining process piping and tanks. Of the 128 facilities, 115 facilities have been demolished or removed from 100-D/H. Table 1-3 summarizes the status of the 100-D/H facilities.

1.2.3.2 Previous Vadose and Waste Site Investigations and Remediation

The behavior of contaminants in the vadose zone has been an important consideration in Hanford plant operations since the 1940s. Some reports (e.g., *Underground Waste Disposal at Hanford Works: An Interim Report Covering the 200 West Area* [HW-9671]; *The Underground Disposal of Liquid Wastes at the Hanford Works, Washington* [HW-17088]) examined the issues related to waste disposal at injection wells, shallow burial cribs, and surface ponds. Groundwater monitoring via wells began in the late 1940s to evaluate the rate of migration through the vadose zone and in the aquifer. Although most attention was focused on radionuclides, primarily within the 200 Area, groundwater monitoring around the 107-F Waste Disposal Trench and the 108-B Crib was reported for some chemicals. Waste site designs (116-KE-2 and possibly 116-KE-3 and 116-KW-2) sometimes included wells where geophysical logging could assess radionuclide movement through the aquifer. Continued waste site use depended on the

vertical migration of contaminants, and sites were shut down when contamination reached certain predetermined concentrations in groundwater at these wells. As such, hydrologic and geochemical processes in the vadose zone were of interest, but were not well understood.

Table 1-3. Summary of Facility Status

Area	Total Number of Facilities	Demolished[a]	Removed[a]	Active[a]	Inactive[a]
100-D	82	63	10	5[b]	4[c]
100-H	46	39	3	3[d]	1[e]
100-D/H	128	102	13	8	5

a. Status:

Active: Facility is occupied and in use (supports Hanford Site missions).

Inactive: Facility is no longer in use and is waiting decommissioning and demolition.

Demolished: Facility has been removed to grade (slab or foundation remains).

Removed: Facility foundation has been removed and any substructure is 0.3 to 0.9 m (1 to 3 ft) below grade.

b. Active facilities in 100-D include the 151-D Primary Substation Switch House, the 181-D River Pump House, the 182-D Reservoir (which supply water to the 200 Area), the 1601-D Transfer Building, and the 6508-S3 Emergency Siren.

c. Inactive facilities in 100-D include the 105-D/DR Safe Storage Enclosures, the 183-D Water Treatment Plant Power Operation Facility, and the 152-D1 Electrical Substation.

d. 100-H active facilities include the 1601-H Transfer Building, the 1713-H Warehouse, and the 6508-S2 Emergency Siren.

e. 100-H inactive facility is the 105-H Safe Storage Enclosure.

Vadose Zone Investigations. The vadose zone at the Hanford Site has been extensively studied since the 1980s. *Unsaturated Water Flow at the Hanford Site: A Review of Literature and Annotated Bibliography* (PNL-5428) provided an overview of the status of vadose zone studies in 1985. By 1992, a significant amount of data had been collected from lysimeters at a wide range of sites at Hanford ("Variations in Recharge at the Hanford Site" [Gee et al., 1992]). Recharge (sometimes called deep percolation) measurements using lysimetry and other techniques at the Hanford Site have been extensive over the past two decades (*Compendium of Data for the Hanford Site (Fiscal Years 2004 to 2008) Applicable to Estimation of Recharge Rates* [PNNL-17841]). Recharge rates applicable to different soil and surface cover conditions at the Hanford Site are listed in *Regulatory Basis and Implementation of a Graded Approach to Evaluation of Groundwater Protection* (DOE/RL-2011-50).

During the construction, operations, and remediation years, the native vegetation and topsoil was scraped off a large portion of 100-D/H. Based on results from "Variations in Recharge at the Hanford Site" (Gee et al., 1992), this condition affected a significant change in vadose zone dynamics with a substantial increase in vadose zone water flux since construction. In addition, water was added to historical waste site locations for dust control during remediation activities. Once remediation is complete and native vegetation is reestablished, recharge will decline to pre-construction rates. Recharge rates are discussed in Chapter 5. Vadose zone contaminant (radiological and nonradiological) characterization studies started at 100-DH in 1975 to evaluate contaminant inventories, concentrations, and distribution at inactive solid and liquid waste sites, reactors, and associated facilities. In the early 1990s, LFIs assessed the nature and extent of effluent discharges to the vadose zone at high-priority waste sites. Several column leaching studies assessed Cr(VI) transport from contaminated vadose zone material to groundwater. Because of the

presence of Cr(VI) in the groundwater, Cr(VI) source identification investigations were performed at 100-D/H.

Key documents applicable to 100-D/H waste site sources (including facilities and groundwater for completeness) are summarized in Table 1-2. Table B-1 (located in Appendix B) provides a brief scope of work, conclusions and implications, and links to the references for key investigations and remediation activities.

Waste Site Remediation. Remediation and characterization of the waste sites in 100 D and 100 H began in 1996 under the authority provided by the interim action RODs and RCRA closure plans, and continues to the present. Remediation consists mainly of 1) RTD of contaminated soil, debris, and waste material; and 2) verification sampling and computer modeling (as needed) to determine whether direct exposure and groundwater protection cleanup requirements have been achieved. After remediation, the excavations are backfilled with approved material, and native shrub steppe flora are planted. Remediation follows the observational approach, including use of radiological field screening data, in-process samples, and direct visual observation. Sample collection and analysis are used to verify achievement of remedial action goals for direct exposure, protection of groundwater, and protection of surface water. Interim cleanup requirements, as described in the 100-D/H Work Plan (DOE/RL-2008-46-ADD1) were achieved at all interim closed out and no action waste sites. As indicated in Table 1-4, closed out sites meet applicable cleanup standards.

Table 1-4. Summary of Waste Site Status (as of November 2012)

OU	Number of Sites[a]	Closed Out[b]	Interim Closed Out[c]	No Action[d]	Not Accepted[e]	Rejected[f]	Accepted[g]
100-DR-1	154	1	72	19	6	10	46
100-DR-2	61	9	27	3	5	3	14
Total 100-D	**215**	**10**	**99**	**22**	**11**	**13**	**60**
100-HR-1	81	1	32	14	5	3	26
100-HR-2	47	0	13	2	3	4	25
Total 100-H	**128**	**1**	**45**	**16**	**8**	**7**	**51**
Total in 100-D/H	**343**	**11**	**144**	**38**	**19**	**20**	**111**

Notes: Additional information is provided in Appendix E.

a. Total number of sites.

b. **Closed Out:** A reclassification status indicating that because of actions taken, a waste management unit meets applicable cleanup standards or closure requirements.

c. **Interim Closed Out:** A historical reclassification status indicating that because of actions taken, a waste management unit meets cleanup standards specified in an interim action ROD or Action Memorandum, but for which a final ROD has not been issued This reclassification status is no longer used. An "Interim" reclassification category and a "Closed Out" reclassification status are used instead.

d. **No Action:** A reclassification status indicating that a waste site does not require any further remedial action under RCRA Corrective Action, CERCLA, or other cleanup standards based on an assessment of quantitative data collected for the waste site.

e. **Not Accepted:** A classification status indicating an assessment was made that a WIDS site is not a waste management unit and is not within the scope of the *Hanford Federal Facility Agreement and Consent Order Action Plan* (Ecology et al., 1989b), Section 3.1. This classification requires lead regulatory agency approval.

Table 1-4. Summary of Waste Site Status (as of November 2012)

OU	Number of Sites[a]	Closed Out[b]	Interim Closed Out[c]	No Action[d]	Not Accepted[e]	Rejected[f]	Accepted[g]

f. **Rejected:** A reclassification status indicating a waste site does not require remediation under RCRA Corrective Action, CERCLA, or other cleanup standards based on qualitative information such as a review of historical records, photographs, drawings, walkdowns, ground penetrating radar scans, and shallow test pits. Such investigations do not include quantitative measurements.

g. **Accepted:** A classification status indicating an assessment has been made that a WIDS site is a waste management unit as defined in the *Hanford Federal Facility Agreement and Consent Order Action Plan* (Ecology et al., 1989b), Section 3.1.

There are 215 waste sites in the 100-DR-1 and 100-DR-2 OUs, and 128 waste sites in the 100-HR-1 and 100-HR-2 OUs. Of the 343 sites identified in 100-D/H, 232 have been closed out or interim closed, or given no action, not accepted, or rejected status as shown on Table 1-4. These status categories generally indicate that a site meets the remedial action goals of the interim action RODs. A total of 111 accepted waste sites remain in 100-D/H according to the interim action in 100-D/H as of November 2012.

Waste Site Cleanup Documentation. Following completion of the interim remedial actions at a waste site in accordance with the applicable interim action ROD, a cleanup document is prepared. This document contains verification information confirming the attainment of interim remedial action goals. This documentation usually includes a description of the interim remedial action conducted, sampling results, disposal information, and a chronology of events.

Tables 1-5 through 1-8 summarize the reclassification of the source OU waste sites and identify Cr(VI) waste sites. Waste site locations are presented in Appendix A. There are two active (operational) sites in 100-D/H. The 100-D-58 Septic System is an accepted waste site, consisting of the modern septic system that supports interim remedial action field facilities, while 100-D-55 (Gravel Pit 21) is classified as a not accepted site.

Table 1-5. 100-DR-1 Operable Unit Waste Sites (as of November 2012)

Reclassification Status	Waste Sites	Total
Closed Out	120-D-1[a]	1
Interim Closed Out	100-D-1, 100-D-2, 100-D-4,[b] 100-D-5, 100-D-6, 100-D-7,[b] 100-D-9, 100-D-18,[b] 100-D-19,[b] 100-D-20,[b] 100-D-21,[b] 100-D-22,[b] 100-D-25, 100-D-29, 100-D-31:1, 100-D-31:2, 100-D-31:3, 100-D-31:4, 100-D-31:5, 100-D-31:6, 100-D-31:7, 100-D-31:8, 100-D-31:9, 100-D-31:10, 100-D-32,[b] 100-D-42, 100-D-45, 100-D-48:1,[b] 100-D-48:2,[b] 100-D-48:3,[b] 100-D-48:4, 100-D-49:1,[b] 100-D-49:2,[b] 100-D-49:3,[b] 100-D-49:4, 100-D-52, 100-D-56:1,[b] 100-D-56:2,[b] 100-D-61, 116-D-1A,[b] 116-D-1B,[b] 116-D-2,[b] 116-D-4,[b] 116-D-5,[b] 116-D-6,[b] 116-D-7,[b] 116-D-9,[b] 116-D-10,[b] 116-DR-1&2,[b] 116-DR-5,[b] 116-DR-9,[b] 118-D-6:2, 118-D-6:3, 118-D-6:4, 120-D-2,[b] 126-D-2, 128-D-2, 130-D-1, 132-D-1, 132-D-2, 132-D-3,[b] 132-D-4, 628-3, 1607-D2:1, 1607-D2:2, 1607-D2:3, 1607-D2:4,1607-D4, 1607-D5, UPR-100-D-2,[b] UPR-100-D-3,[b] UPR-100-D-4[b]	72

Table 1-5. 100-DR-1 Operable Unit Waste Sites (as of November 2012)

Reclassification Status	Waste Sites	Total
No Action	100-D-3, 100-D-24, 100-D-50:5, 100-D-50:10, 100-D-70, 100-D-74, 100-D-75:3, 100-D-80:1, 100-D-82, 100-D-83:4, 100-D-84:1, 100-D-85:1, 100-D-86:2, 100-D-87, 100-D-88, 100-D-90, 116-D-3,[b] UPR-100-D-1, UPR-100-D-5	19
Not Accepted	100-D-10, 100-D-26, 100-D-34, 100-D-38, 100-D-57, 100-D-91	6
Rejected	100-D-33, 100-D-35, 100-D-41, 100-D-59, 100-D-79, 100-D-92, 100-D-93, 100-D-95, 126-D-1,[b] 126-D-3	10
Accepted	100-D-8, 100-D-30,[b] 100-D-31:11, 100-D-31:12, 100-D-50:1, 100-D-50:2, 100-D-50:3, 100-D-50:4, 100-D-50:6, 100-D-50:7, 100-D-50:8, 100-D-50:9, 100-D-60, 100-D-63, 100-D-65,[b] 100-D-66,[b] 100-D-67, 100-D-69, 100-D-71, 100-D-72, 100-D-73,[b] 100-D-75:1, 100-D-75:2, 100-D-76, 100-D-80:2, 100-D-81, 100-D-83:1, 100-D-83:2, 100-D-83:3, 100-D-83:5, 100-D-84:2, 100-D-85:2, 100-D-86:1, 100-D-86:3, 100-D-96, 100-D-97, 100-D-98:1, 100-D-98:2, 100-D-99, 100-D-101, 100-D-102, 100-D-104,[b] 100-D-105, 100-D-107, 118-D-6:1,[b] 1607-D2:5	46

a. The 120-D-1 (100-D Ponds) is a RCRA TSD unit that has been closed under RCRA.

b. Site associated with sodium dichromate usage.

1

Table 1-6. 100-DR-2 Operable Unit Waste Sites (as of November 2012)

Reclassification Status	Waste Sites	Total
Closed Out	100-D-27, 122-DR-1:1, 122-DR-1:2, 122-DR-1:3, 122-DR-1:4, 122-DR-1:5, 122-DR-1:6, 122-DR-1:7,[a] 1607-D3	9
Interim Closed Out	100-D-12,[b] 100-D-13, 100-D-15, 100-D-23, 100-D-28:1, 100-D-43, 100-D-46,[b] 100-D-47, 100-D-53, 100-D-54, 100-D-64, 116-D-8, 116-DR-4,[b] 116-DR-6,[b] 116-DR-7, 118-D-1, 118-D-4, 118-D-5, 118-DR-1, 118-DR-2:2,[b] 116-DR-8,[b] 116-DR-10,[b] 126-DR-1, 132-DR-1,[b] 132-DR-2, 1607-D1, 600-30	27
No Action	100-D-68,[b] 100-D-94, 128-D-1	3
Not Accepted	100-D-11, 100-D-36, 100-D-37, 100-D-55,100-D-89	5
Rejected	100-D-17, 100-D-28.2, 100-D-40	3
Accepted	100-D-14, 100-D-58, 100-D-62, 100-D-77,[b] 100-D-78, 100-D-103, 100-D-106, 118-D-2:1, 118-D-2:2,[b] 118-D-3:1,[b] 118-D-3:2 116-DR-3, 118-DR-2:1,[b] 100-D-100[b]	14

a. The 122-DR-1 (105-DR Sodium Fire Facility) is a RCRA TSD unit that has been closed under RCRA.

b. Site associated with sodium dichromate usage.

1

Table 1-7. 100-HR-1 Operable Unit Waste Sites (as of November 2012)

Reclassification Status	Waste Sites	Total
Closed Out	116-H-6[a,b]	1
Interim Closed Out	100-H-1,[b] 100-H-3, 100-H-4, 100-H-5,[b] 100-H-9, 100-H-10, 100-H-11, 100-H-12, 100-H-13, 100-H-14,[b] 100-H-17,[b] 100-H-21,[b] 100-H-22,[b] 100-H-24, 100-H-30,[b] 100-H-31, 100-H-41, 116-H-1,[b] 116-H-2,[b] 116-H-3,[b] 116-H-5,[b] 116-H-7,[b] 116-H-9,[b] 118-H-6:2, 118-H-6:3, 118-H-6:4, 118-H-6:5,[b] 118-H-6:6, 132-H-1, 1607-H2, 1607-H3, 1607-H4	32
No Action	100-H-7, 100-H-8, 100-H-28:1, 100-H-28:6, 100-H-28:8, 100-H-33, 100-H-35, 100-H-40, 100-H-45, 100-H-49:2, 100-H-50, 100-H-51:5, 100-H-53, 100-H-54:4	14
Not Accepted	100-H-6, 100-H-18, 100-H-19, 100-H-20, 100-H-26	5
Rejected	100-H-39, 100-H-47, 100-H-49:3	3
Accepted	100-H-28:2, 100-H-28:3, 100-H-28:4, 100-H-28:5, 100-H-28:7, 100-H-34,[b] 100-H-36, 100-H-42, 100-H-43, 100-H-44, 100-H-46, 100-H-48, 100-H-49:1, 100-H-51:1, 100-H-51:2, 100-H-51:3, 100-H-52, 100-H-51:6, 100-H-54, 100-H-56, 100-H-57, 100-H-59, 116-H-4,[b] 118-H-6:1,[b] 126-H-2, 132-H-3	26

a. 116-H-6 (183-H Solar Evaporation Basins) is a RCRA TSD unit that has been closed under RCRA but is in modified post-closure care.

b. Site associated with sodium dichromate usage.

2

Table 1-8. 100-HR-2 Operable Unit Waste Sites (as of November 2012)

Reclassification Status	Waste Sites	Total
Closed Out	None	0
Interim Closed Out	132-H-2, 100-H-2,* 100-H-37, 118-H-1:1, 118-H-1:2, 118-H-2, 118-H-3, 118-H-4, 118-H-5, 128-H-1, 1607-H1, 600-151, 600-152	13
No Action	128-H-2, 128-H-3	2
Not Accepted	100-H-15, 100-H-27, 600-258	3
Rejected	100-H-16, 100-H-32, 126-H-1, 100-H-55	4
Accepted	100-H-38, 100-H-58, 600-380, 600-381, 600-382:1, 600-382:2, 600-382:3, 600-382:4, 600-382:5, 600-383:1, 600-383:2, 600-383:3, 600-383:4, 600-383:5, 600-383:6, 600-383:7, 600-383:8, 600-383:9, 600-383:10, 600-384:1, 600-384:2, 600-384:3, 600-384:4, 600-384:5, 600-385	25

* Site associated with sodium dichromate usage.

Waste Site Consideration in the RI/FS. All 100-D/H waste sites were evaluated in this RI/FS to determine if they are protective of HHE. While the unique factors of each site were considered individually, the overall consideration of waste sites can be described generally based on classification/ reclassification status.

The Hanford WIDS summarizes information about known and suspected areas of contamination. These areas are defined as locations that may require action to mitigate a potential environmental impact (*Tri-Party Agreement Handbook Management Procedures*, Guideline Number TPA-MP-14, "Maintenance of the Waste Information Data System (WIDS)" [RL-TPA-90-0001]). All 100-D/H waste sites were considered part of this RI/FS process to determine whether the sites are protective of human health and the environment.

- Sites with a "closed out" status were reviewed to confirm that this determination had been made under appropriate regulatory authority. Where a closed out status was appropriate, no further review of site information was performed, and the site was not considered further within the RI/FS.

- Sites with a "rejected" or "not accepted" status were reviewed to determine whether new information, if available, is consistent with the existing documented basis for rejection or nonacceptance. Where the existing classification/reclassification was appropriate, the site was not considered further within the RI/FS process. Two rejected or not accepted sites at 100-D/H (100-D-10 and 100-D-59) were found to have information that was inconsistent with the existing determinations. The existing determinations are documented for each site in accordance with *Tri-Party Agreement Handbook Management Procedures,* Guideline Number TPA-MP-14, "Maintenance of the Waste Information Data System (WIDS)" (RL-TPA-90-0001).

- Sites with a "no action" or "interim closed" reclassification status based on confirmatory or verification data are all considered within the overall RI and are quantitatively evaluated against preliminary remediation goals (PRGs) as described in Chapters 5 through 7 to determine if further action(s) or institutional controls are needed. Sites with a no action or interim closed reclassification with a basis other than direct data (for example, historic decommissioning data) were considered on a site-by-site basis as described in Chapter 8.

- Sites with an "accepted" classification status fit within two broad general subcategories:

  - Sites where an interim remedial action requirement has been identified in interim decision documents, but for which interim remedial action had not been completed (via an approved waste site reclassification). These sites were considered within the RI from the standpoint that a remedial action determination has already been made. Because site-specific data were not yet available, these sites were carried into the FS.

  - Candidate sites under the 100 Area Remaining Sites ROD (EPA/ROD/R10-99/039) are sites for which an interim remedial action determination has not yet been made. The 100 Area Remaining Sites ROD (EPA/ROD/R10-99/039) established a process whereby new and existing sites that did not have sufficient information to warrant a remedial action determination or exclusion from consideration as a formal waste site could be evaluated in order to make this determination. These sites are referred to as "candidate sites" or "confirmatory sites" under the interim action framework. Until such time as the ROD is issued, the candidate process to add these waste sites will be retained under the interim action ROD; and these sites will continue to be dispositioned according to that process, including site-specific evaluation for protection of human health and the environment.

RCRA Treatment, Storage, and Disposal Unit Closure. Three RCRA treatment, storage, and disposal (TSD) units (183-H Solar Evaporation Basins, 100-D Ponds, and 105-DR Large Sodium Fire Facility [LSFF]) have undergone closure within 100-D/H as discussed below:

- **183-H Solar Evaporation Basin:** The 183-H Facility was constructed in 1949 and used for 100-H water treatment until mid-1960. This facility consisted of 16 concrete basins that were used in treating Columbia River water for subsequent use as reactor coolant. In 1973, four of these basins were converted for use in evaporation of other mixed waste. Approximately 1.6 million kg (3.6 million lb) of waste per year were treated by solar evaporation. The facility received routine and nonroutine waste, which consisted of spent acid etch solutions, metal constituents in the form of precipitates, and unused chemicals and spent solutions from miscellaneous processes. Most of the water treatment structures, including 12 additional adjoining basins, were demolished in 1974. The four remaining basins were inactive until July 1973, when radioactive and dangerous (mixed) waste from the 300 Area fuel fabrication facilities was shipped to the basins for storage and treatment. The last shipment of waste to the basins took place on November 8, 1985. By the fall of 1996, the basins had been completely demolished, and demolition waste was disposed of in the adjacent 183-H clear wells and at the ERDF. The site was not clean closed under RCRA because fluoride and nitrate concentrations are above the Method B cleanup levels of "Model Toxics Control Act—Cleanup" (WAC 173-340[3]), hereinafter called MTCA. Therefore, the unit was closed in place under the modified closure provisions of the RCRA Hanford permit with post-closure care. The modified closure certification was accepted by Ecology on May 13, 1997 (*183-H Solar Evaporation Basins Postclosure Plan* [DOE/RL-97-48]). Because clean closure was not achieved, dangerous waste constituents remain in the vadose zone beneath the waste site. In addition, RCRA closures do not have authority to address the cleanup of radiological contamination. As such, accepted WIDS site 100-H-33 was created to address the radiological contamination that is within the same footprint as 116-H-6.

- **100-D Ponds (120-D-1):** The 100-D Ponds were approximately 500 m (1,640 ft) northeast of D Reactor and were used from 1977 to 1994 for percolation of effluent. The ponds, which were constructed from the former 188-D Coal Ash Disposal Basin, consisted of two interconnected surface ponds. The original pond was constructed by excavating an area 9 m (30 ft) deep in the eastern half of the 188-D Basin. The site was modified in 1979 to form a two-compartment pond, one overflowing to the other. The modification resulted in a combined surface dimension of 50 by 67 m (160 by 220 ft).

  After the construction of the ponds, all process sewer liquid effluents were diverted to the ponds instead of the Columbia River. The northern pond was a percolation pond and the southern pond was a settling pond. The ponds received corrosive waste from the regeneration of ion exchange columns located in the 185-D/189-D complex. The ponds also received nonhazardous waste from the 183-D sand-filter backwash, small quantities of filtered chlorinated water from hydraulic test loops, and fuel discharge trampoline tests.

  The estimated flow rate was 170,000 L/day (45,000 gal/day). In August 1996, contaminated sediment was removed from the 100-D Ponds as part of a DOE-RL voluntary cleanup action (*100-D Ponds Closure Plan* [DOE/RL-92-71]). On August 27, 1999, Ecology accepted this TSD as clean closed.

- **105-DR Large Sodium Fire Facility (122-DR-1):** The LSFF was in the former supply and exhaust fan wing of the 105-DR Reactor Building (*105-DR Large Sodium Fire Facility Closure Plan* [DOE/RL-90-25]). The LSFF operated from 1972 to 1986 and was used to study the fire and safety aspects associated with sodium and other alkali metal fires for application to liquid metal reactors. It is estimated that 20,000 kg (44,100 lb) of sodium and other alkali metals were used at the facility for the burn tests.

The facility closure plan divided the facility into seven areas for sampling and closure purposes. Four of the seven RCRA subunits of the LSFF (122-DR-1) were closed out under the *105-DR Large Sodium Fire Facility Closure Plan* (DOE/RL-90-25). These subunits included the exhaust fan room, small fire room, large fire room, sodium handling room, and an office area (122-DR-1:1); the gravel scrubber (122-DR-1:3); the 117-DR Seal Pit Crib (122-DR-1:6); and the outdoor storage area (122-DR-1:7). Certification of closure of these LSFF TSD subunits is documented in *105-DR Large Sodium Fire Facility Soil Sampling Data Evaluation Report* (WHC-SD-EN-TI-307), and in an Ecology letter regarding closure certification for the 122-DR-1 Subsites 1, 3, 6, and 7 ("Ecology Acceptance of Closure Certification for the 105-DR Large Sodium Fire Facility (T-1-1)" [Wilson, 1996]). The 122-DR-1:6 Crib closure was based on process knowledge without further sampling. The closure only applies to use of this crib by LSFF operations. This crib and its associated influent piping are also separately identified as the 116-DR-8 and 100-D-50:8 waste sites, respectively, to address residual contamination associated with discharges to the crib.

The LSFF Closure Plan deferred the remaining three LSFF TSD subunits (122-DR-1:2, 122-DR-1:4, and 122-DR-1:5) to the CERCLA process (TPA [Ecology et al., 1989a]). These three subunits were excavated, and remedial activities are presented in the cleanup verification package (*Cleanup Verification Package for the 105-DR Large Sodium Fire Facility (122-DR-1:2, 100-D-53/122-DR-1:4, 132-DR-2/122-DR-1:5), the 119-DR Exhaust Stack Sampling Building (100-D-64), and the 100-D-23 and 100-D-54 Dry Wells* [CVP-2003-00018]). An Ecology letter ("Ecology Acceptance of Closure Certification for the 105-DR Large Sodium Fire Facility (T-1-1)" [Wilson, 1996]) regarding clean closure certification for the 122-DR-1 Subsites 2, 4, and 5 (*Cleanup Verification Package for the 105-DR Large Sodium Fire Facility (122-DR-1:2, 100-D-53/122-DR-1:4, 132-DR-2/122-DR-1:5), the 119-DR Exhaust Stack Sampling Building (100-D-64), and the 100-D-23 and 100-D-54 Dry Wells* [CVP-2003-00018]) acknowledged receipt and accepted DOE's closure certification and professional engineer's certification of closure for the *105-DR Large Sodium Fire Facility Closure Plan* (DOE/RL-90-25), dated June 3, 2004.

In total, 50 waste sites with closed out, rejected, or not accepted classification/reclassification statuses were reconsidered to ensure that each had a sufficient existing basis for these determinations. Those sites with sufficient existing bases are identified in Table 1-9 and will not be addressed further in this RI/FS. Three sites are special cases: Reactor Core Sites 118-D-6:1, 118-DR-2:1, and 118-H-6:1. These sites are discussed in this section. However, all 100-D and 100-H waste sites identified in Appendix C of *Hanford Federal Facility Agreement and Consent Order Action Plan* (Ecology et al., 1989b), hereinafter called TPA Action Plan, will be included in the ROD in order for the remedy decision to be documented, even if no further remedial activities are recommended.

Table 1-9. Sites Not Addressed Further in the RI/FS

Classification/ Reclassification Status	Waste Sites
Closed Out	100-D-27, 116-H-6, 120-D-1, 1607-D3, 122-DR-1:1, 122-DR-1:2, 122-DR-1:3, 122-DR-1:4, 122-DR-1:5, 122-DR-1:6, 122-DR-1:7
Not Accepted	100-D-11, 100-D-26, 100-D-34, 100-D-36, 100-D-37, 100-D-38, 100-D-55, 100-D-57, 100-D-89, 100-D-91, 100-H-6, 100-H-15, 100-H-18, 100-H-19, 100-H-20, 100-H-26, 100-H-27, 600-258

Table 1-9. Sites Not Addressed Further in the RI/FS

Classification/ Reclassification Status	Waste Sites
Rejected	100-D-17, 100-D-28:2, 100-D-33, 100-D-35, 100-D-40, 100-D-41, 100-D-79, 100-D-92, 100-D-93, 100-D-95, 126-D-1, 126-D-3, 100-H-16, 100-H-32, 100-H-39, 100-H-47, 100-H-55, 116-H-6, 126-H-1
Accepted*	100-D-58

* Indicates sites with basis for exclusion discussed in text.

Rejected and Not Accepted Waste Sites with Inadequate Existing Bases. Two sites were identified for which the existing basis warrants reconsideration.

100-D-10. The 100-D-10 site was identified to address a reported former outfall in the river embankment upstream from the 100-D-8, 1907-DR Outfall. The site was not accepted on the basis that it received only storm water drainage from the 190-DR Tank Pit. However, storm water drainage from this area was actually routed to the 100-D-50:1 Process Sewer, which discharged to the 100-D-8, 1907-DR Outfall. The 100-D-10 Outfall is visible in pre-Manhattan era aerial photographs, and there is no apparent association with later Hanford Site operations. While this may not warrant consideration of the feature as a waste site, the site should be reconsidered for accepted or not accepted status based on the correct information. For the purposes of the FS, this site will be considered as a candidate site for future evaluation under remedial actions.

100-D-59. The 100-D-59 site consists of a French drain at the former acid transfer station south of the 183-D Head House. This French drain was designed to receive any overflow from sulfuric acid transfer operations. The site was previously reclassified as rejected on the basis that any acids discharged to the drain would rapidly neutralize in alkaline soil. However, this determination was made before it was known that some sulfuric acids used at the Hanford Site were contaminated with mercury that may have accumulated significantly in soil. Because it is not known if the 100-D-59 French drain received mercury-contaminated acids, this site will be considered as a candidate site for future evaluation under remedial actions. However, it should also be noted that the 100-D-59 site is immediately adjacent to the 100-D-72 site and may be incidentally addressed by investigations or remedial activities for the latter site.

Waste Sites Requiring No Further Consideration. Waste sites requiring no further consideration are those that are or will be closed under another regulation, or need to be reclassified using the process in *Tri-Party Agreement Handbook Management Procedures*, Guideline Number TPA-MP-14, "Maintenance of the Waste Information Data System (WIDS)" (RL-TPA-90-0001). The waste sites removed from further consideration in this report are the following:

100-D-58. The 100-D-58 accepted waste site was constructed in 1998 and consists of an active modern septic system servicing 100-D area field remediation facilities. The site receives sanitary sewage from the M0-980 building and restroom trailer. The site was designed for conventional pressure distribution and conforms to the 1993 Department of Health design standard ("Wastewater and Reclaimed Water Use Fees" [WAC 246-272]). There are no radiation zones or known contamination areas in or near this site. For these reasons, this site will not be addressed further under the CERCLA RI/FS process. Following cessation of use, this system will be abandoned in accordance with "Wastewater and Reclaimed Water Use Fees" (WAC 246-272) requirements.

100-D-50:10. The 100-D-50:10 construction camp potable water supply pipelines subsite encompasses residual cast iron pipelines formerly used to supply potable water to the temporary construction camp southeast of the DR Reactor. This subsite was reclassified as no action based on a determination that the potable water supply was not associated with any constituents that would present an adverse risk to human health and the environment. This determination remains appropriate.

116-D-3. The 116-D-3 site was identified as a crib associated with the former 108-D facility. Based on review of historical drawings, geophysical investigation, and excavation, this site was determined to be a duplicate of the 116-D-4 site. The 116-D-3 site was reclassified as rejected in 2000; this reclassification was amended to no action in 2003. Another potential location for this crib has been identified separately as the 100-D-76 waste site. The 116-D-4 and 100-D-76 sites are addressed further in this RI/FS, but the redundant 116-D-3 site will not be considered further.

128-D-1. The 128-D-1 site was identified as a potential burn pit but was determined to be a duplicate of the 128-D-2 or 628-3 burn pit waste sites. The site was reclassified as no action based on this determination. This decision remains appropriate.

1.2.3.3 Nonoperational Area Evaluation Summary

In 2011, an evaluation of the River Corridor nonoperational areas was completed. The nonoperational evaluation considered the five contaminant transport mechanisms, physical features, and climate conditions that could influence transport and used surface and near-surface information from a number of available sources:

- Orphan site evaluations (OSEs)
- Air emission reports
- Environmental monitoring programs
- Statistical modeling

Appendix K describes the nonoperational evaluation process for the River Corridor, data and information used, and conclusions and recommendations. It also includes specific results and conclusions for 100-D/H.

Orphan Site Evaluation Summary. The OSEs are an important element of the nonoperational area evaluation. The purpose of the nonoperational area evaluation is to increase confidence that waste disposal or releases requiring characterization and cleanup within a given land parcel of the Hanford Site River Corridor are identified. Key elements of the OSEs include a comprehensive review of historical information, aerial photographs, and a field investigation. Results from these activities are reviewed with DOE-RL and the lead regulatory agency. Potential orphan sites are evaluated under the *Tri-Party Agreement Handbook Management Procedures*, Guideline Number TPA-MP-14, "Maintenance of the Waste Information Data System (WIDS)" (RL-TPA-90-0001) process. The OSEs were recently completed on the highest potential impact areas of 100-D/H to identify additional waste sites that may require characterization and possibly remediation (*100-D Area Orphan Sites Evaluation Report* [OSR-2006-0001]; *100-H Area Orphan Sites Evaluation Report* [OSR-2008-0002]). The OSE for the remainder of 100-D/H, primarily the horn area, was completed in 2011 (*100-F/IU-2/IU-6 Area – Segment 4 Orphan Sites Evaluation Report* [OSR-2011-0001]).

The 100-D OSE (OSR-2006-0001) identified 30 new waste sites, and the 100-H OSE (OSR-2008-0002) identified 15 new waste sites. The OSE for the remainder of 100-D/H (OSR-2011-0001) identified six waste sites.

Air Emissions. Two groups of sources of Hanford Site stack air emissions had the potential to affect the River Corridor by air deposition. The first source group, where most of the Hanford Site stack air emissions occurred between 1944 and 1972, were the facilities in the 200 Area that separated plutonium and uranium from irradiated reactor fuel. The second source group, the nine production nuclear reactors in the 100 Area, had stacks to exhaust ventilation air from the working areas of the reactor facilities. These were minor sources of emissions compared to emissions coming from the 200 Area facilities that separated plutonium and uranium from irradiated reactor fuel. Any hot spots existing in the River Corridor area as a result of historic stack emission deposition would have decayed or attenuated to negligible levels over the past 40 or more years since the majority of the air emissions occurred. Also, potential fugitive dust from surface waste disposal sites would have similarly decayed and dispersed to less than the allowable annual public exposure levels. Aerial radiation surveys of the Hanford Site and widespread soil sampling over many years support this conclusion (*An Aerial Radiological Survey of the Hanford Site and Surrounding Area, Richland, Washington* [EGG-10617-1062]).

Environmental Monitoring Programs. Data from ongoing monitoring investigations were also used to supplement the RI. Contaminant source, meteorological, air, surface water and sediment, and ecological investigations are described in Chapter 2.

Statistical Modeling. Statistical modeling was used to support the data analyses and development of technical recommendations, such as additional sampling for the nonoperational areas in the River Corridor. The process used established approaches and datasets from the Hanford Site Central Plateau and adapted them to the River Corridor. In addition to the CSM developed for the Central Plateau, the CSM for the River Corridor also addressed the potential for overland flow and potential effects on riparian and near-shore areas. Statistical analysis was used to represent the conceptual models and incorporate the available data to support a quantitative base for the probability that a (undiscovered) waste site might exist in the nonoperational areas. As a result of these efforts, no additional waste sites or areas affected by waste site releases were found in the nonoperational areas of 100-D/H that pose a threat to human health and the environment.

1.2.3.4 Previous Groundwater Investigation Activities

Groundwater monitoring projects are established under *General Environmental Protection Program* (DOE Order 5400.1) to meet the requirements of *Radiation Protection of the Public and the Environment* (DOE Order 5400.5), which deals with radiation protection of the public and the environment, and federal and state regulations. The TPA (Ecology et al., 1989a) is a legally binding document that requires the investigation of groundwater contamination and establishes a process for evaluating and implementing appropriate response actions.

Historical groundwater monitoring results for 100-D/H are presented in Chapter 4, and the locations of all 100-D/H groundwater monitoring wells are illustrated in Appendix A. Groundwater from wells in 100-D/H are sampled for contaminants of potential concern (COPCs) based on the results of the data quality objectives process. Sampling of groundwater at specific wells is scheduled for collection every quarter to 2 years, depending on the data need. Groundwater data for 100-D/H are used to create maps and plots that illustrate groundwater flow, water table elevations, hydrogeochemistry, and contaminant concentration trends and distribution. The results have been published annually in Hanford Site Groundwater Monitoring Reports since 1980 (for example, *Hanford Site Groundwater Monitoring and Performance Report for 2009: Volumes 1 & 2* [DOE/RL-2010-11], hereinafter called Hanford Site 2009 Groundwater Report) and are discussed in Chapter 4.

Aquifer Characteristics and Ground-Water Movement at Hanford [HW-60601]) depicted the general aquifer characteristics of the Hanford Site, including one of the first Sitewide groundwater flow maps

showing general directions and average rates of groundwater flow. Aquifer testing and aquifer properties were evaluated and summarized in this report. Hanford Site operations actively discharged a variety of water and liquid waste to the surface at locations such as B Pond and Gable Mountain Pond in the 200 Area, and retention ponds and trenches in the 100 Area. Groundwater mounds developed at these locations and affected groundwater flow across much of the Site, including 100-D/H. The Hanford Site water table changes over the period from 1950 to 1980 are documented by Pacific Northwest Laboratory (now Pacific Northwest National Laboratory [PNNL]) in a 1986 report (*Hanford Site Water Table Changes 1950 Through 1980: Data Observations and Evaluation* [PNL-5506]). This report described detailed water level changes at 5-year intervals at a network of wells across the site.

In 1967, the disposal of reactor coolant effluent to trenches was discussed for 100-F and 100-D, with proposed tests at 100-BC and 100-K (*Program Review—Ground Disposal of Reactor Effluent* [DUN-3259]). This report describes the 100-D test conducted at the 116-DR-1&2 Trench. Effluent was initiated from the 107-DR basin on March 7, 1967, and from the 107-D basin on March 9, 1967. Initial flows were 17,034 L/min (4,500 gal/min). On March 17, flows were increased to 104,099 L/min (27,500 gal/min), with liquid levels stabilizing about 3 m (10 ft) above the base of the trench. The test continued at this rate through June 26, 1967, when the D Reactor was shut down. Over this period, approximately 1.3×10^{10} L (3.4×10^{9} gal) of effluent were infiltrated through the trench. The estimated infiltration rate was 5,678 L/day (1,500 gal/day) per square foot of trench bottom. Effluent Cr(VI) concentrations of approximately 350 μg/L were estimated. The report provides several figures showing effects to groundwater, including descriptions of the large groundwater mound that developed. The effluent caused an additional 2.7 to 3 m (9 to 10 ft) of groundwater mounding beyond that caused by ongoing operations. The temperature of effluent exiting the reactor was approximately 95°C (203°F), and groundwater temperatures in excess of 50°C (122°F) were observed near the mound. A portion of this water appears to have migrated into and across the horn and likely is the source for the large, dilute Cr(VI) plume observed recently in horn area groundwater.

In the 1970s, concerns increased about radiological contamination of groundwater at Hanford; and researchers began to investigate various groundwater issues, from the vertical distribution of radioactive contamination (*Vertical Contamination in the Unconfined Groundwater at the Hanford Site, Washington* [PNL-2724]) to general radiological groundwater contamination (*Radiological Status of the Ground Water Beneath the Hanford Site: January – December 1980* [PNL-3768]).

By the mid-1980s, routine sampling began to include nonradiological constituents such as nitrate and Cr(VI). By 1984, quarterly groundwater monitoring had identified Cr(VI) at four wells in 100-H. As sampling continued, Cr(VI) continued to be found in various wells in 100-H. Cr(VI) was then reported at 100-D in 1987 at three wells near the D and DR Reactors, which have since been decommissioned. By 1988, it had become clear that action would be needed under CERCLA or RCRA in the 100-HR-3 OU. The initial study work plan was developed under the RCRA corrective action that was compatible with CERCLA (*RCRA Facility Investigation/Corrective Measures Study Work Plan for the 100-HR-3 Operable Unit, Hanford Site, Richland, Washington* [DOE/RL-88-36]). This work plan established the OU setting, objectives, procedures, tasks, and schedule for conducting the remedial field investigation and corrective measures study for the 100-HR-3 Groundwater OU. In 1993 and 1994, an LFI report (*Limited Field Investigation Report for the 100-HR-3 Operable Unit* [DOE/RL-93-43]) was completed along with the qualitative risk assessment (QRA) (*Qualitative Risk Assessment for the 100-HR-3 Groundwater Operable Unit* [WHC-SD-EN-RA-007]).

The Tri-Parties decided that all groundwater would be cleaned up under the CERCLA remedial action process that included the 100-HR-3 OU. In 1995, the focused feasibility study (*100-HR-3 Operable Unit Focused Feasibility Study* [DOE/RL-94-67]) and the proposed plan (*Proposed Plan for Interim Remedial*

Measure at the 100-HR-3 Operable Unit [DOE/RL-94-102]) were finalized. At the end of 1995, the pilot-scale treatability test summary report (*The Pilot-Scale Treatability Test Summary for the 100-HR-3 Operable Unit* [DOE/RL-95-83]) was issued, and the interim action ROD followed in April 1996. Table 1-10 summarizes the chronology of reports describing the interim action groundwater monitoring at the 100-HR-3 OU, including the annual monitoring of the pump-and-treat systems and changes that were made to the injection well/extraction well network. Also included in these documents is the ISRM monitoring and general groundwater monitoring within 100-D/H. Over time, the understanding of the contaminant distributions has evolved as more monitoring wells have been added to the network.

Table 1-10. Chronology of Groundwater Reports for 100-D/H

Year	Pump-and-Treat System	ISRM	Annual Groundwater Monitoring Reports	Aquifer Sampling Tube Data Reports	Annual Environmental Monitoring
1980	—	—	PNL-3768	—	—
1981	—	—	PNL-4237	—	—
1982	—	—	PNL-4659	—	—
1983	—	—	PNL-5041	—	PNL-5038, PNL-5039
1984	—	—	PNL-5408	—	PNL-5407
1985	—	—	—	—	PNL-5817
1986	—	—	—	—	PNL-6120
1987	—	—	PNL-6315-1, -2	—	PNL-6464
1988	—	—	PNL-6315-3, -4	—	PNL-6825
1989	—	—	PNL-7120	—	PNL-7346
1990	—	—	PNL-8073	—	PNL-7930
1991	—	—	PNL-8284	—	PNL-8148
1992	—	—	—	—	PNL-8682
1993	—	—	PNL-10082	—	PNL-9823
1994	—	—	PNL-10698	—	PNL-10574
1995	—	—	PNNL-11141	—	PNNL-11139
1996	EPA/ROD/R10-96/134	—	PNNL-11470	BHI-00778	PNNL-11472
1997	DOE/RL-97-96	—	PNNL-11793	—	PNNL-11795
1998	DOE/RL-99-13	—	PNNL-12086	BHI-01153	PNNL-12088
1999	DOE/RL-2000-01	EPA/AMD/R10-00/122	PNNL-13116	—	PNNL-13230
2000	DOE/RL-2001-04	DOE/RL-2000-74	PNNL-13404	Interoffice Memo No. 078404 (Lee and Raidl, 2000)	PNNL-13487
2001	DOE/RL-2002-05	DOE/RL-2001-01	PNNL-13788	BHI-01494	PNNL-13910
2002	DOE/RL-2003-09	DOE/RL-2003-05	PNNL-14187	BHI-01624	PNNL-14295
2003	DOE/RL-2004-21	DOE/RL-2004-06	PNNL-14548	PNNL-14444	PNNL-14687
2004	DOE/RL-2005-18	DOE/RL-2005-39	PNNL-15070	—	PNNL-15222

Table 1-10. Chronology of Groundwater Reports for 100-D/H

Year	Pump-and-Treat System	ISRM	Annual Groundwater Monitoring Reports	Aquifer Sampling Tube Data Reports	Annual Environmental Monitoring
2005	DOE/RL-2006-08	DOE/RL-2005-97	PNNL-15670	—	PNNL-15892
2006	DOE/RL-2006-76	DOE/RL-2007-19	PNNL-16346	—	PNNL-16623
2007	DOE/RL-2008-05	DOE/RL-2008-10	DOE/RL-2008-01	SGW-35028	PNNL-17603
2008	DOE/RL-2009-15	DOE/RL-2009-01	DOE/RL-2008-66	—	PNNL-18427
2009	DOE/RL-2010-11	DOE/RL-2010-11	DOE/RL-2010-11	—	PNNL-19445

Note: Complete reference citations are provided in Chapter 11.

As a result of the implementation of these interim remedies and associated monitoring, the extent of the plumes has been mapped. Figures 1-24 and 1-25 depict the Cr(VI) plume in the spring and fall for 2011 in the 100-HR-3 OU, respectively. Figure 1-26 shows that the 100-D Cr(VI) trend splits (southern plume). Figure 1-27 shows the 100-D northern plume. Figure 1-28 shows the 100-H Cr(VI) plume. The contaminated footprint is approximately 0.8 km^2 (0.31 mi^2). Major features to note are the high concentrations in 100-D, with low values across the horn into 100-H. The plumes in 100-D are separated by an area of little or no contamination. The most concentrated plume is upgradient from the ISRM Barrier. These high concentrations appear to be breaking through the barrier. Even though the former HR-3 and DR-5 groundwater pump-and-treat systems were deemed undersized, it is clear from the trend plots accompanying Figures 1-26 through 1-28 that they reduced Cr(VI) concentrations in the unconfined aquifer. Many wells show declining concentrations. However, the plume area is sufficiently large that the overall effect of the relatively small systems can be obscured by the sheer size of the groundwater volume being addressed. Consequently, the remedial process optimization (RPO) process was initiated and has significantly increased the number of wells involved in the remedy together with increased treatment capacity to accelerate the cleanup of the groundwater plumes.

Columbia River Studies. River Corridor studies involving groundwater (often referred-to in this context as groundwater seeps, pore water, or groundwater upwelling) that are pertinent to Columbia River water quality and ecological risk include the following:

- *Sampling and Analysis of 100 Area Springs* (DOE/RL-92-12)
- *Chromium in River Substrate Pore Water and Adjacent Groundwater: 100-D/DR Area, Hanford Site, Washington* (BHI-00778)
- *Field Summary Report for Remedial Investigation of Hanford Site Releases to the Columbia River, Hanford Site, Washington: Collection of Surface Water, Pore Water, and Sediment Samples for Characterization of Groundwater Upwelling*, hereinafter called Columbia River RI Report (WCH-380)
- *Columbia River Component Risk Assessment, Volume I:Screening Level Ecological Risk Assessment* and *Volume II: Baseline Human Health Risk Assessment* (DOE/RL-2010-117)

Pore water and Aquifer Tube Studies. In addition to the groundwater investigations, several pore water and aquifer tube studies have been performed that are relevant to 100-D/H and are summarized in Appendix B, Table B-1.

The first (1994) pore water study in the Hanford Reach of the Columbia River was designed to collect substrate water quality and contaminant data for determining the potential exposure and risk to ecological receptors from groundwater discharge to the river, particularly from Cr(VI) (*Preliminary Determination of Chromium Concentration Within Pore Water and Embryonic Chinook Salmon at Hanford Reach Spawning Area in Proximity to 100-HR-3 Operable Unit* [BHI-00156]). Cr(VI) concentrations below the former 11 µg/L AWQC (since lowered to 10 µg/L), as measured in the river substrate, was the PRG. Embryonic Chinook salmon were selected as the target receptor for the study because they have limited mobility during their early life stages (egg and sac-fry), spend most of their time within or near the river substrate, and thus could be chronically exposed to Cr(VI) from subsurface groundwater discharge. The appropriate season for pore water sampling is in the fall (during low river stage and relatively high groundwater discharge to the river). Salmon redds were identified by aerial surveys to establish when salmon were spawning in the Hanford Reach and to determine locations where pore water samples should be collected for Cr(VI) analysis.

The pore water sampling study adjacent to 100-D encompassed 3,100 linear m (10,200 linear ft) of the riverbed offshore of 100-D, corresponding to the mapped extent of Cr(VI) contamination in 100-D (*Chromium in River Substrate Pore Water and Adjacent Groundwater: 100-D/DR Area, Hanford Site, Washington* [BHI-00778]). Cr(VI) concentrations ranged from not being detected at many sample locations to a maximum of 632 µg/L. Results are presented in Table B-1.

Recent pore water studies were conducted from 2008 to 2010 for the entire River Corridor, including 100-D/H (*Field Summary Report for Remedial Investigation of Hanford Site Releases to the Columbia River, Hanford Site, Washington: Collection of Surface Water, Pore Water, and Sediment Samples for Characterization of Groundwater Upwelling* [WCH-380, Rev. 1], hereinafter called Columbia River RI Report). These studies showed Cr(VI) values above the AWQC at several river sites opposite the reactor areas during very low river stage. Although the majority of the sites did not show Cr(VI) above detection limits, values up to 331 µg/L and 46 µg/L were observed for the D and H areas, respectively. The sample locations and detailed results for the pore water study are provided in Chapters 2 and 4 of this RI/FS report.

Aquifer tubes have been installed along the Columbia River throughout the River Corridor. Aquifer tubes consist of small, stainless steel screens that are typically driven into the aquifer/hyporheic zone along the riverbank (Figure 1-29) using a percussion method. Data are used to evaluate effectiveness of upgradient treatment systems, such as groundwater pump-and-treat systems and permeable reactive barriers, and to quantify contaminant entry into the river.

The tubes monitor shallow groundwater of the uppermost, unconfined aquifer and typically terminate 1 to 2 m (3 to 6.5 ft) below the water table in the unconsolidated, permeable sediments. Ringold Formation upper mud (RUM) sediments typically cannot be penetrated using the percussion method for tube installation. Sampling of these tubes is governed by a SAP (*Sampling and Analysis Plan for Aquifer Sampling Tubes*, hereinafter called SAP for Aquifer Sampling Tubes [DOE/RL-2000-59]), revised in 2009. A polyethylene tube is attached to the aquifer tube, and a peristaltic pump is used to collect a water sample from the screened interval. Specific conductivity of the sample water varies with river stage, reflecting a mixture with either more groundwater (higher specific conductivity) or more surface water (lower specific conductivity) in the aquifer tube sample. Typically, aquifer tube samples collected during low river stage are more representative of groundwater conditions. Aquifer sampling tube data reports are listed in Table 1-10.

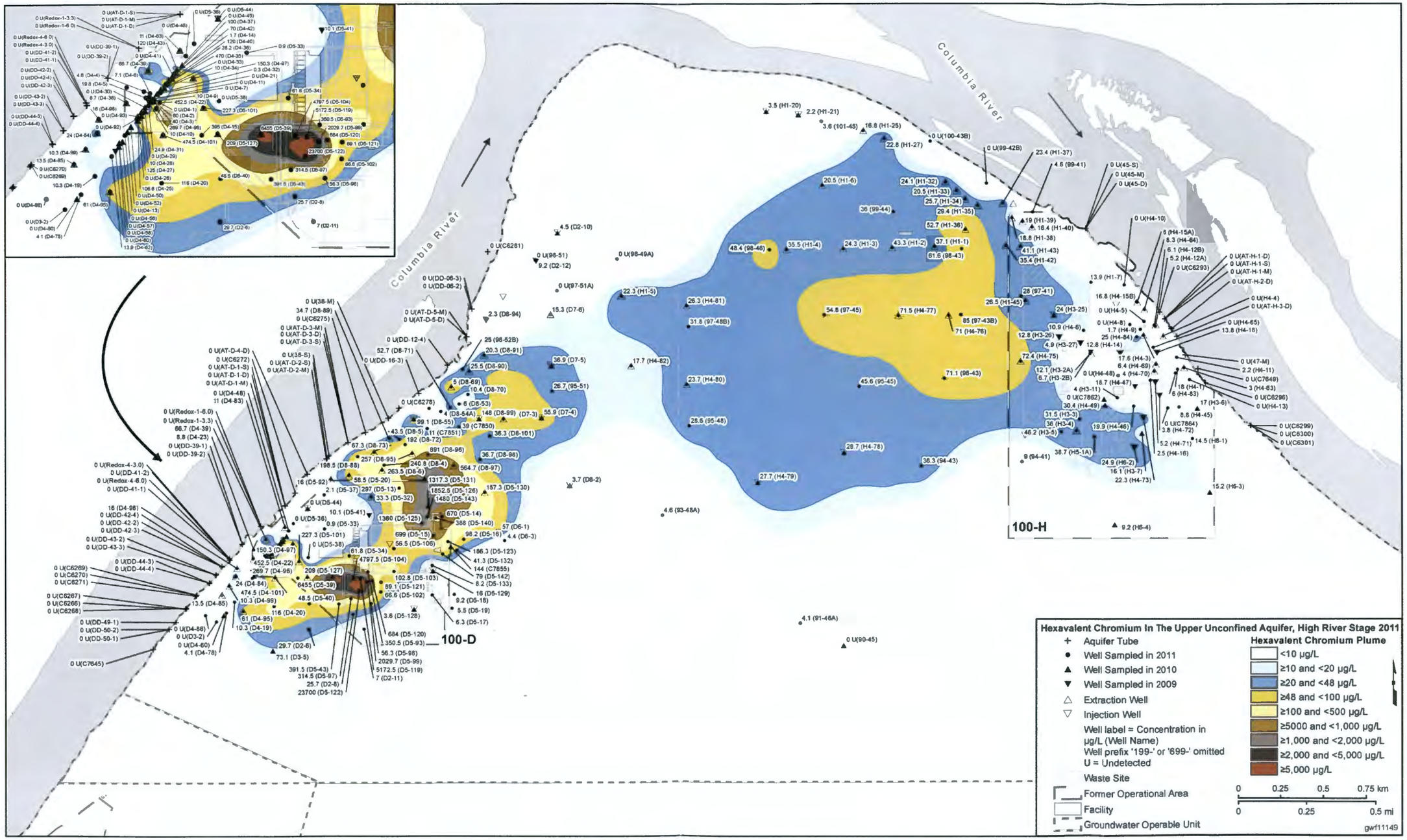

Figure 1-24. 100-HR-3 OU Cr(VI) (Spring 2011)

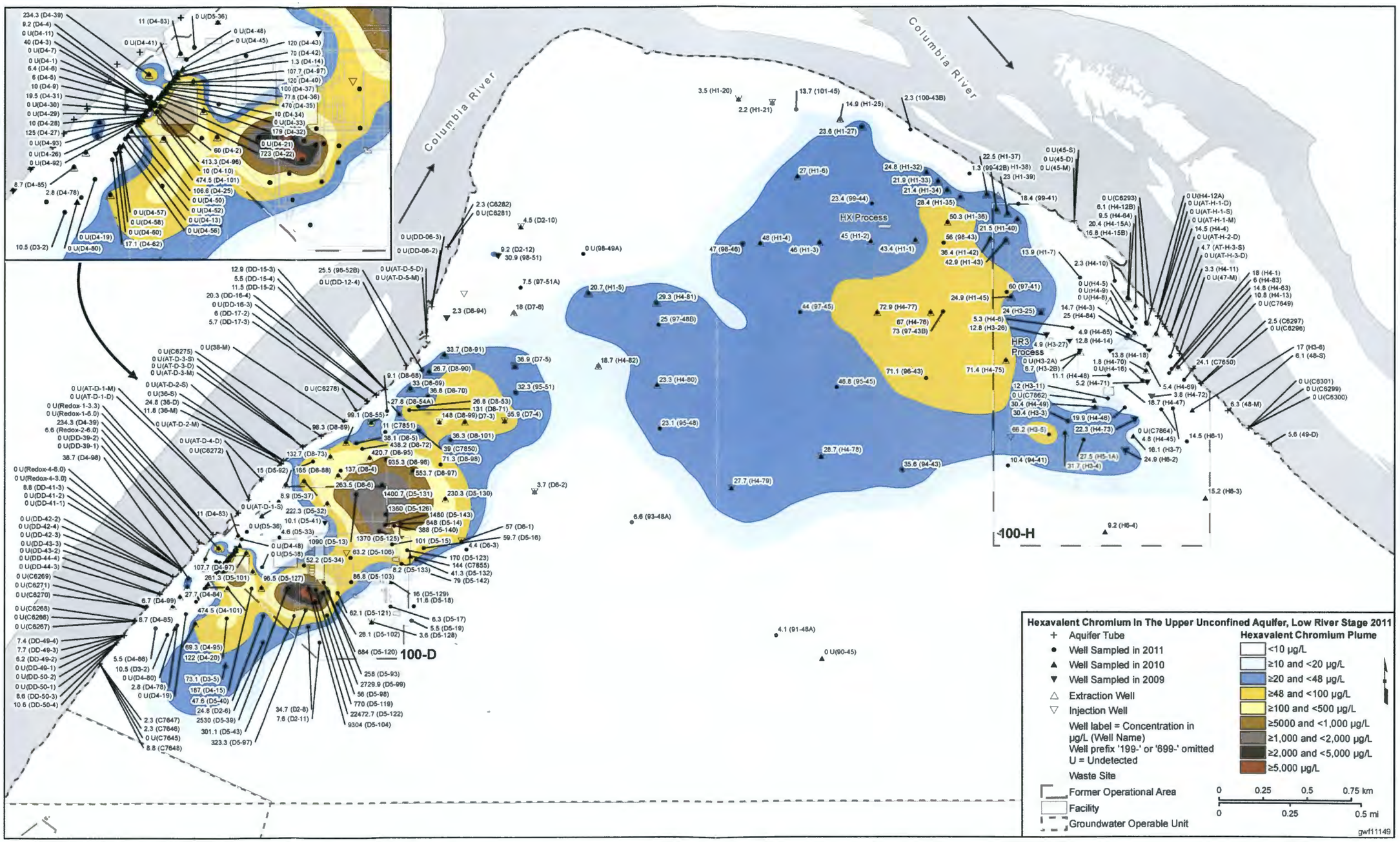

Figure 1-25. 100-HR-3 OU Cr(VI) Map (Fall 2011)

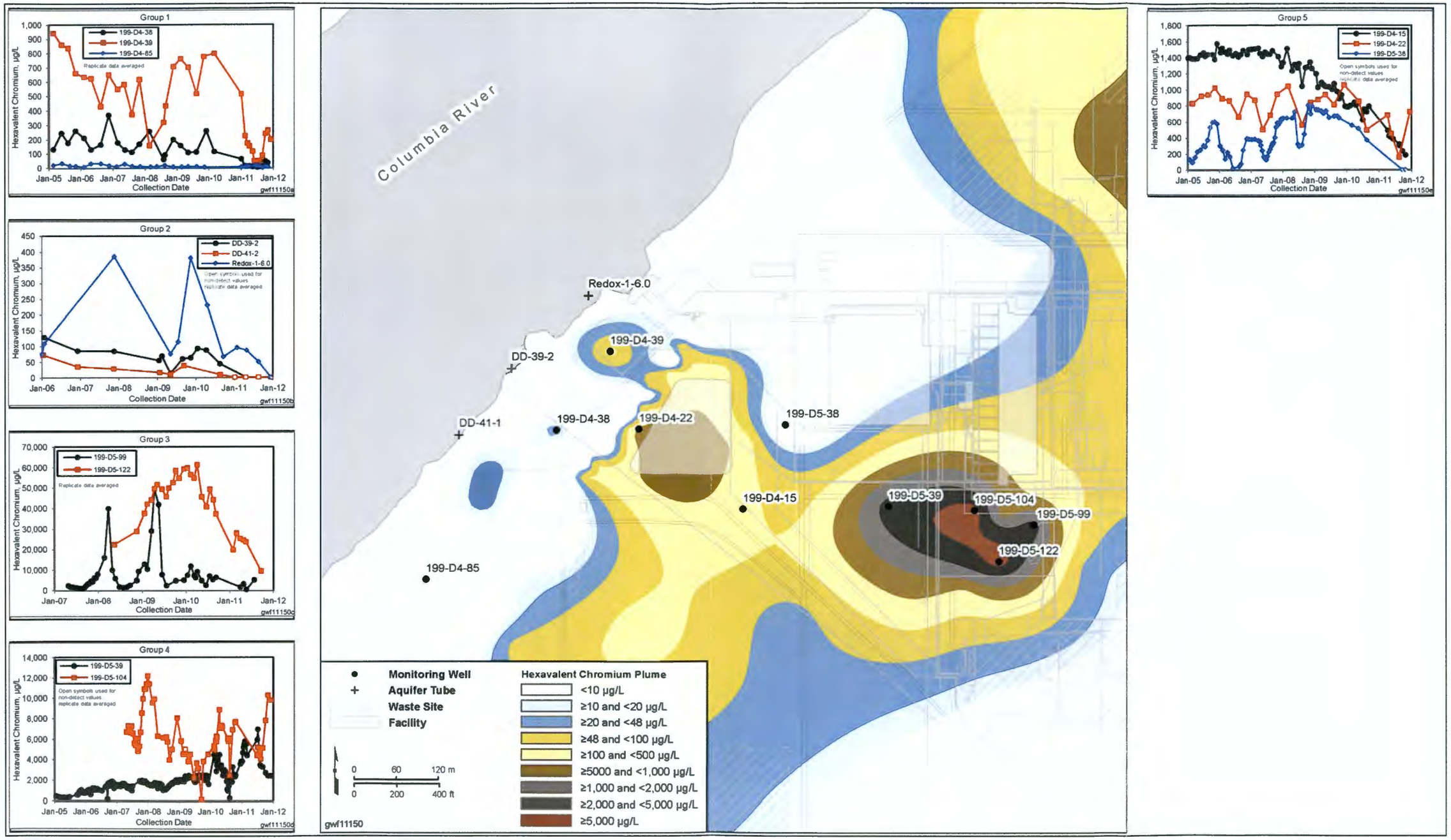

Figure 1-26. 100-D Area Cr(VI) Trend Plots (Southern Plume)

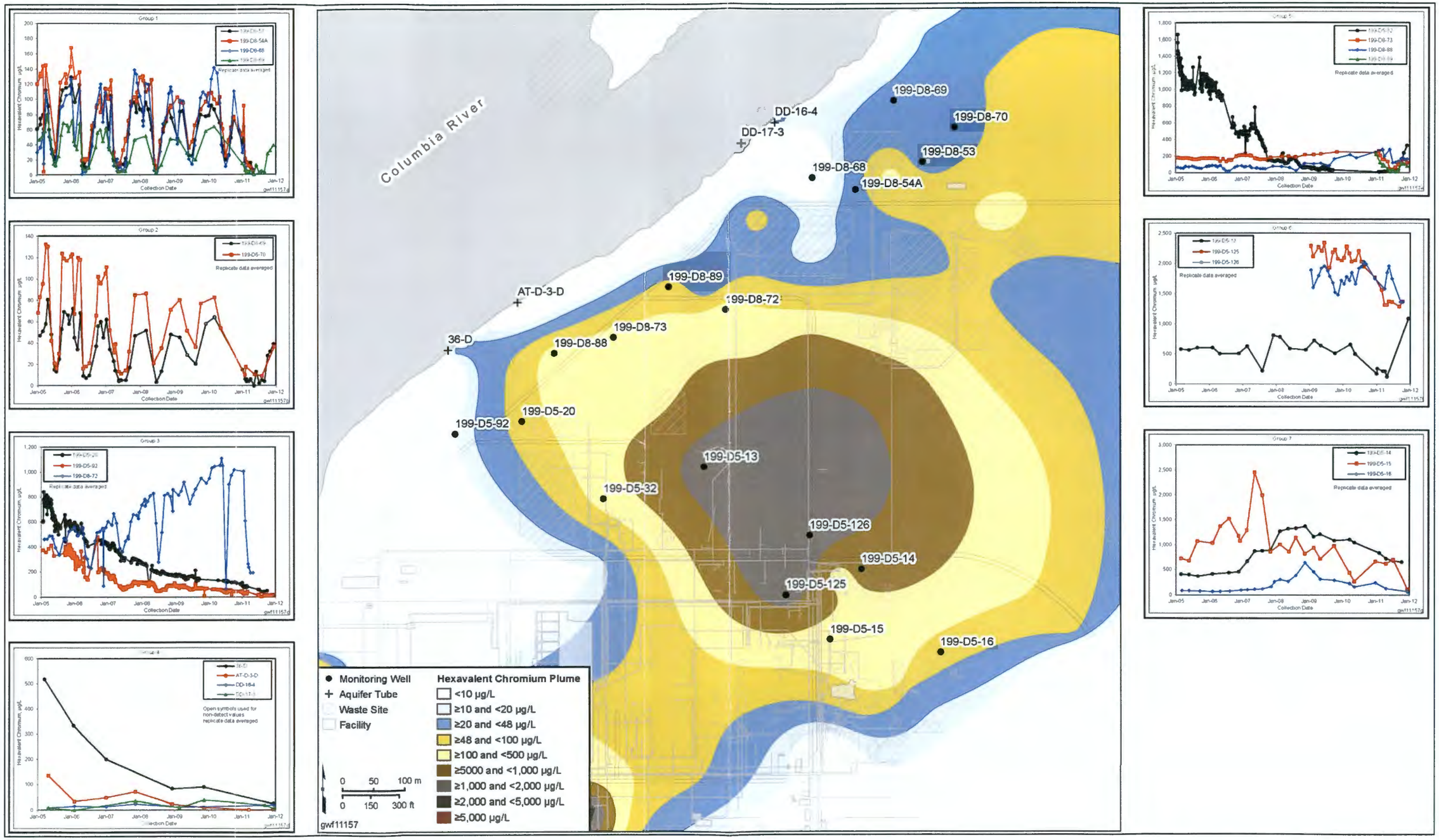

Figure 1-27. 100-D Area Cr(VI) Trend Plots (Northern Plume)

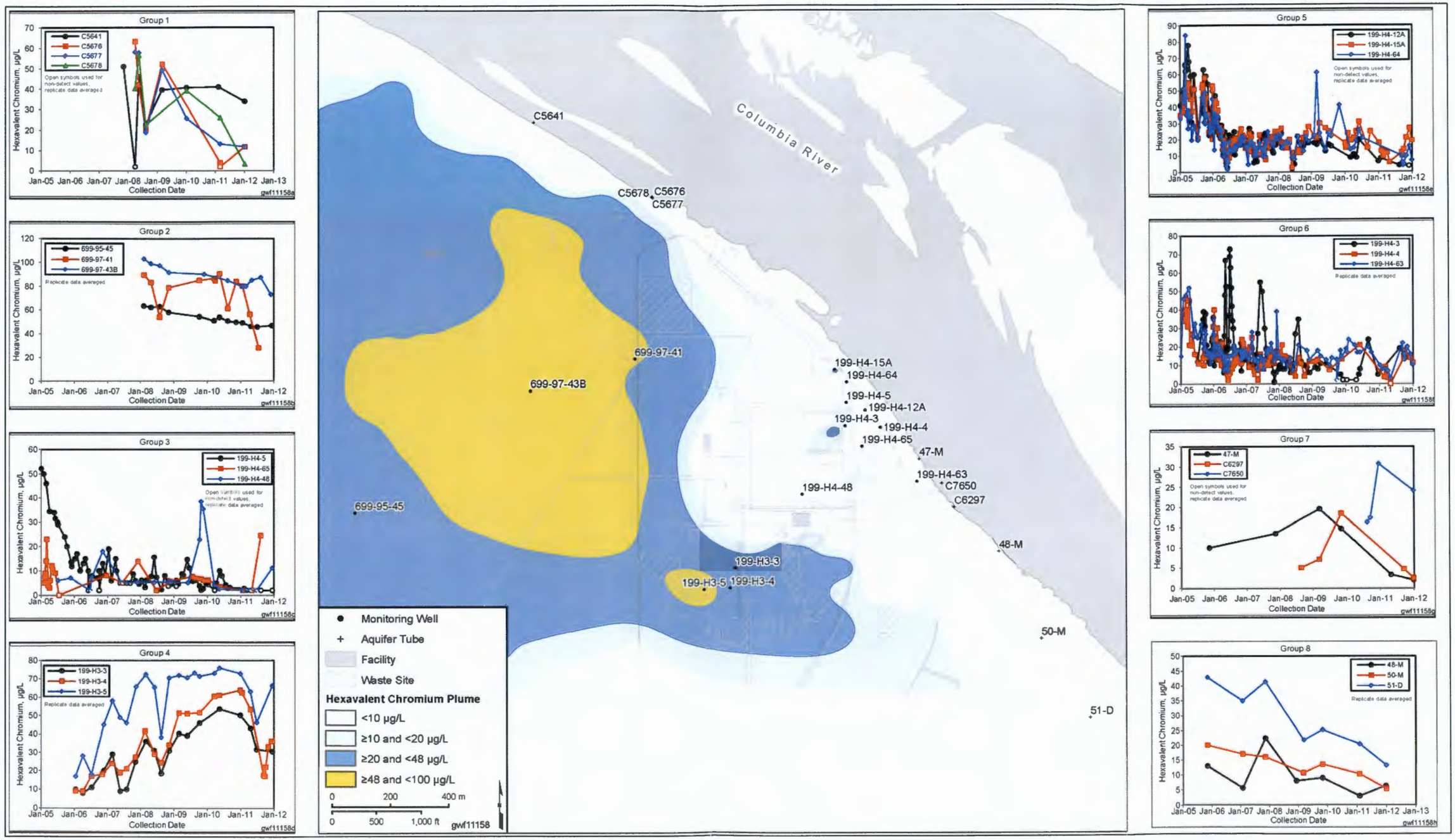

Figure 1-28. 100-H Area Cr(VI) Trend Plots

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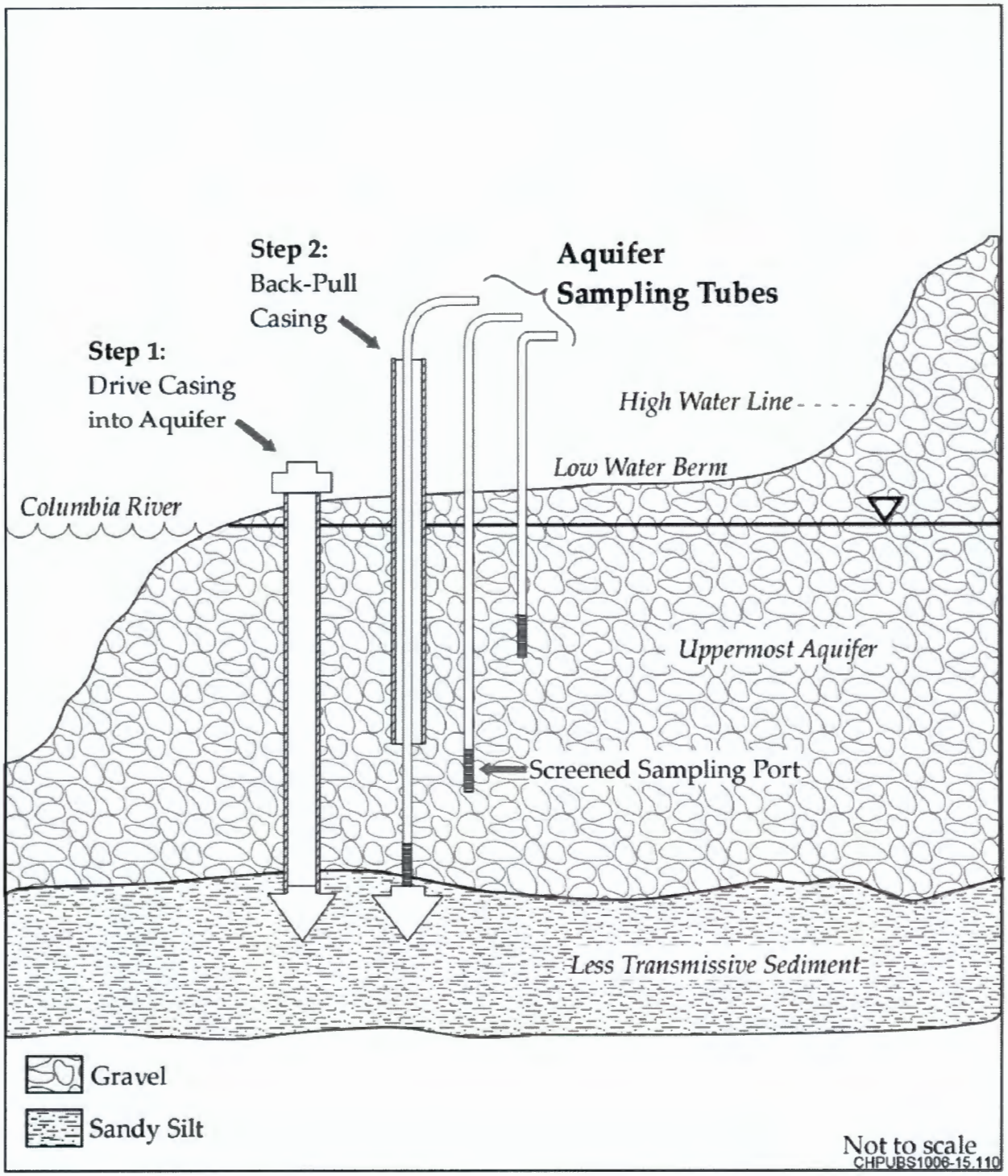

Figure 1-29. Main Components of Aquifer Sampling Tube Installation

Chromium Source Investigation. Three new wells were installed in the northern plume in an attempt to refine the location of the source area. These wells served to better define the high-concentration portion of the northern plume but did not identify a hot spot similar to that associated with the southern plume where Cr(VI) reached concentrations of 69,700 µg/L in 2010. The shallow and intermediate vadose zone was also sampled in 2009 using innovative drilling technology in an attempt to identify vadose zone

chromium sources. This work did not find significant levels of Cr(VI) at depth, but total chromium was elevated beneath waste site 100-D-104, where soil staining is evident.

Chromium in the Ringold Formation Upper Mud Unit. Aquifer tests were performed to gather additional data on the deep chromium contamination. The aquifer tests were performed using existing monitoring wells in 100-H and grouped into three sets of wells, with each set containing three wells. Each of the three-well sets had wells completed at increasing depths in the unconfined aquifer. Some rebound in Cr(VI) concentrations occurred as a result of the shutdown of the 100-HR-3 pump-and-treat system for a rebound study, but a zone to the north and northeast of H Reactor building was identified as a potential pathway for downward migration of Cr(VI)-contaminated groundwater during reactor operations. Overall, Cr(VI) within the tested zone appears to be of finite extent, which should be amenable to remediation via pump-and-treat.

The results suggest that the most likely explanation for the origin of the Cr(VI) in the RUM unit underlying 100-H is from contaminated cooling water that passed through the H Reactor. The cooling water contained up to 1,000 µg/L of Cr(VI), and was subsequently discharged to the ground. This water formed a mound that provided sufficient hydraulic force to push into the upper RUM unit and mix with existing groundwater in the unit, resulting in concentrations of one-tenth to one-thirtieth of the original cooling water. Concentrations decline inland, consistent with a reactor mound. The areal extent and relatively high continuous concentrations rule out localized contamination during well drilling.

1.2.3.5 Treatability Studies

During the RI/FS process, a range of technologies is identified to mitigate unacceptable risks. However, many of these technologies have little data supporting their performance at a scale comparable to the size of the contaminated site. As such, treatability studies are conducted to fill data gaps on performance characteristics of a promising technology and to reduce uncertainty in the implementation of a technology. One such technology is the passive in-situ reduction of Cr(IV).

The ISRM system is a passive reactive barrier that started as a single well treatability test at 100-D in 1997. The barrier was created by injecting a solution of sodium dithionite into the aquifer. Cr(VI) in the groundwater moving through the treatment zone was changed to relatively insoluble trivalent chromium. The treatability test ended in calendar year (CY) 1999 after six wells were treated. The ISRM technology was initially judged a success.

The ISRM barrier depends on the presence of naturally occurring iron in the aquifer to create treatment zones that reduce and immobilize Cr(VI). When data indicated that Cr(VI) was breaking through the ISRM treatment zones in several locations, scientists proposed that fortifying the barrier with additional iron could offer a sustainable long-term repair. As such, the *Nano-size Iron Injection Test* was initiated. Laboratory and field tests were performed in 2007 and 2008 to evaluate the effectiveness of injecting tiny particles of iron into the aquifer to repair portions of the ISRM barrier, as shown on Figure 1-30. The goals of the test were to disperse the iron at least 7 m (23 ft) from the injection well and to determine whether the iron could effectively decrease Cr(VI) concentrations in the groundwater. Existing monitoring wells indicated the iron reached more than 3 m (9.8 ft) from the injection well but not as far as 12 m (39 ft). A well drilled in 2009 verified that iron was dispersed more than 7 m (23 ft) from the injection well in significant amounts, which accomplished a primary goal of the test.

The injection modestly increased hydraulic conductivity and rapidly appeared in wells 3 m (10 ft) downgradient and upgradient. Injection affected the oxidation-reduction potential and dissolved oxygen in wells 3 m (10 ft) away, but no effects were observed in a well 12.8 m (42 ft) away.

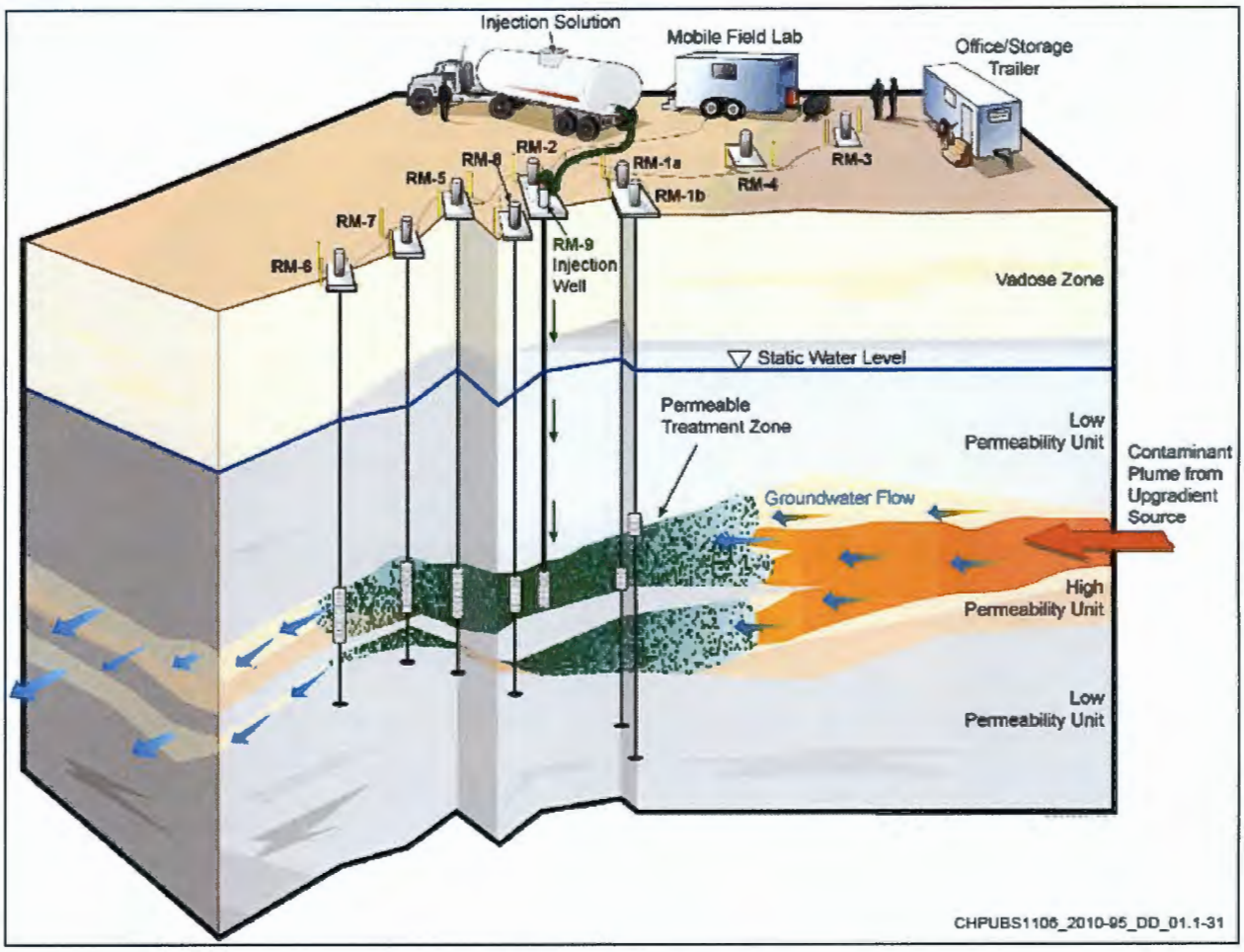

Figure 1-30. Nano-size Iron Injection

Electrical conductivity, electrical resistivity, and induced polarization showed that the injected iron intruded more than 7 m (23 ft) laterally into the formation. Groundwater from the injection and nearby monitoring wells showed that Cr(VI) and total chromium concentrations decreased to near detection limits. Concentrations of dissolved oxygen, nitrate, and uranium also decreased. Sulfate concentrations decreased immediately after treatment but later rebounded to pre-injection levels. Dissolved and total manganese and iron concentrations increased (*Treatability Test Report on Mending the In Situ Redox Manipulation Barrier Using Nano-Size Zero Valent Iron* [DOE/RL-2009-35]). One factor in evaluating a treatability test is the cost to implement the technology. The cost of the nano-sized iron was approximately a third of the implementation cost. When the material cost was estimated to implement this technology across 100-D/H, it was considered prohibitive. As such, the technology was judged to be an ineffective option.

Horizontal Directional Drilling. A technology demonstration of horizontal directional drilling was conducted in 100-D to evaluate the capability of this technology in difficult geological conditions and to determine the feasibility of emplacing a horizontal well to intercept a groundwater Cr(VI) plume. The primary goals of this demonstration were to drill through the 25 m (82 ft) thick vadose zone and emplace a 90 m (295 ft) long screen in the unconfined aquifer. Secondary objectives were to minimize the loss of drilling fluid to the vadose zone and aquifer and to place the screen within 1.5 m (5 ft) of the middle of the aquifer. The field demonstration was performed from November 2009 through January 2010.

This test did not result in successful installation of a horizontal groundwater well. The principal impediments to casing installation were difficulty in removing cuttings from the nearly horizontal casing and inability of the downhole hammer to advance through the unconsolidated Hanford formation. The results and analysis of this technology demonstration may be useful to project planners, scientists, and contractors who are considering similar types of projects at the Hanford Site.

Other Research. PNNL completed an in situ biostimulation treatability test at 100-D in CY 2009. The purpose of biostimulation is to induce the reduction of chromium, nitrate, and oxygen and to remove these compounds from the groundwater. Test data indicated that injected materials were successfully distributed to the target radius from the injection wells. Microbial activity and the ability to reduce the targeted species were observed throughout the monitored zone, and low oxygen, nitrate, and chromium concentrations were maintained for the duration of monitoring. Aquifer permeability reduction within the test zone was moderate while the injected substances and associated organic degradation products persisted for a period of at least 1 year. Further evaluation of this technology is discussed in Chapter 8.

Other Studies. Other studies at 100-D/H include radiological and other surveys, and treatability studies to test various technologies with application to cleanup of Cr(VI). Table B-1 (in Appendix B) presents a brief description of these, and Chapter 8 presents the pilot tests and treatability tests reviewed and considered in the FS technology screening.

1.2.3.6 Groundwater Remediation

Three CERCLA interim action remedies were initiated at the 100-HR-3 OU. These include the original 100-HR-3 groundwater pump-and-treat system in 100-H (which treated groundwater from both 100-D and 100-H), the DR-5 pump-and-treat system in 100-D, and the ISRM barrier in 100-D. The Interim ROD for 100-D/H groundwater (100-HR-3 and 100-KR-4 Interim ROD [EPA/ROD/R10-96/134]) required pump-and-treat to address Cr(VI), and an amendment to the interim ROD (100-HR-3 Interim ROD Amendment [EPA/AMD/R10-00/122]) required the ISRM barrier. Following are the RAOs of these systems:

- Protect aquatic receptors in the river bottom substrate from contaminants in groundwater entering the Columbia River.
- Protect human health by preventing exposure to contaminants in the groundwater.
- Provide information that will lead to the remedy.

For the interim ROD, groundwater in monitoring wells adjacent to the river should contain no more than 20 µg/L of Cr(VI) (100-HR-3 and 100-KR-4 Interim ROD [EPA/ROD/R10-96/134]). Protecting human health means groundwater should meet drinking water standards (for example, no more than 100 µg/L of total chromium [EPA] and no more than 48 µg/L of Cr(VI) [Ecology]), or institutional controls should prevent human consumption of groundwater.

The remedial design report and remedial action work plan (*Remedial Design and Remedial Action Work Plan for the 100-HR-3 and 100-KR-4 Groundwater Operable Units' Interim Action* [DOE/RL-96-84]) were issued in September of 1996. Construction was completed by June 30, 1997, and full-time operation of the HR-3 pump-and-treat system began July 1, 1997. In 1996, an amendment to the interim ROD was issued to modify the selected remedial action by deploying a new technology (ISRM) for remediation of the Cr(VI) plume in 100-D.

Concurrently with the startup of the 100-HR-3 pump-and-treat system, *Interim Action Monitoring Plan for the 100-HR-3 and 100-KR-4 Operable Units* (DOE/RL-96-90), hereinafter called IAMP, established

the general monitoring plan for the pump-and-treat systems in the 100-HR-3 and the 100-KR-4 OUs. The major constituents being monitored semiannually are Cr(VI), co-contaminants such as nitrates, and strontium-90. The IAMP (DOE/RL-96-90) was modified by a letter from Ecology to DOE ("Sampling Changes to the 100-HR-3 and 100-KR-4 Operable Units (OU)" [Wanek, 1998]) and was modified through the TPA (Ecology et al., 1989a) Change Notice process to add wells. With these modifications, the general sampling program was developed and carried forward into the interim ROD.

The 100-HR-3 pump-and-treat system began operation in 1997 with five extraction wells at 100-H and two extraction wells at 100-D using ion exchange resin columns to remove Cr(VI). The treated effluent was discharged into injection wells at 100-H. A series of modifications starting in CY 2000, and ending in CY 2010 added five extraction wells to the system. In 2004, the DR-5 pump-and-treat system was installed to treat high Cr(VI) concentrations in groundwater underlying the central part of 100-D. Again through a series of modifications the DR-5 system used five extraction wells and two injection wells to treat the groundwater.

The 100-HR-3 pump-and-treat system was very effective in reducing the size of the Cr(VI) plume beneath 100-H and reducing concentrations to less than the aquatic criteria near the river. However, concentration reductions in the northern plume and central part in 100-D have not been as effective because of the continuing sources of Cr(VI) in the vadose zone and the relatively small size of the DR-5 system.

Figure 1-31 shows the locations of the extraction and injection wells of the former DR-5 and 100-HR-3 system. The 100-HR-3 pump-and-treat system was shut down in May 2011 and removed 405.65 kg (894.3 lb) of Cr(VI) during its period of operation, treating more than 4.2 billion L (1.1 billion gal) of water. The 100-DR-5 pump-and-treat system was shut down in March 2011 and removed 337.58 kg (774.8 lb) of Cr(VI) over its lifetime, treating 384.2 million L (101.5 million gal) of water.

Within the context of the TPA (Ecology et al., 1989a), two target milestones are driving remedial activities at the 100-HR-3 OU: 2012 Columbia River Protection Milestone (M-016-110-T01) and 2020 Groundwater Plume Remediation Milestone (M-016-110-T02) (Ecology et al., 1989a):

> ***M-016-110-T01:*** *DOE shall take actions necessary to contain or remediate Cr(VI) groundwater plumes in each of the 100 Area NPL Operable Units such that Ambient Water Quality Criteria for Cr(VI) are achieved in the hyporheic zone and river water column (due date: December 31, 2012)*

> ***M-016-110-T02:*** *DOE shall take actions necessary to remediate Cr(VI) groundwater plumes such that Cr(VI) will meet drinking water standards in each of the 100 Area NPL Operable Units (due date: December 31, 2020).*

Remedial Process Optimization for 100-D/H. In response to the two target TPA (Ecology et al., 1989a) milestones, DOE initiated the RPO framework. The RPO framework provides a systematic approach for evaluating and improving site remediation systems. These activities include refinement of the following five tasks:

1. Review the conceptual site model and implications for site remediation.
2. Review the design and performance of the existing 100-HR 3 OU ex situ remedial systems and treatability actions and identify system or process modifications to improve performance.
3. Identify and screen in situ and ex situ remedial technologies with the potential to improve remedial performance at the site.

4. Develop potential remedial action alternatives for the site based on the screened technologies.

5. Develop pre-conceptual designs and costs for three pump-and-treat technologies that were identified in the screening process for inclusion in one or more of the proposed remedial action alternatives.

Additional information on the RPO efforts is discussed in Chapters 8 and 9 of the FS.

RPO Expansion—DX and HX Pump-and-Treat Systems. Historically, groundwater extraction has focused primarily on protecting the river and extracting groundwater from nearby waste sites. In addition, treated water was injected at quantities and locations that were optimized to minimize discharge of contaminated water to the river and maximize flow of contaminated water toward the extraction wells. However, as a result of actions identified in CERCLA 5-year reviews (Section 1.2.5), the pump-and-treat systems were expanded and improved to increase overall system up-time, reduce individual well down-time, and capture and treat more contaminated groundwater.

In the 100-HR-3 OU, pump-and-treat operations were expanded along the river, including for the first time, the horn area. Beneath the horn area is a lower concentration, dispersed plume migrating across the horn area from 100-D toward 100-H. While Cr(VI) concentrations in this region are generally less than 100 µg/L, they still exceed the aquatic criteria, and the contaminated groundwater had previously discharged to the river near known salmon redds.

Additional extraction and injection wells were installed in the 100-HR-3 OU, and two new water treatment plants were installed: 100-DX in 100-D and 100-HX in 100-H (Figure 1-32). DOE expanded the pump-and-treat systems in an effort to protect the river better and to comply with the 100-HR-3 and 100-KR-4 Interim ROD (EPA/ROD/R10-96/134), as amended by *Explanation of Significant Differences for the 100-HR-3 and 100-KR-4 Operable Units Interim Action Record of Decision: Hanford Site Benton County, Washington* (EPA et al., 2009). The expanded systems are anticipated to be a component of the remedy in the 100-HR-3 OU. The location, drilling, installation, and development of 47 new extraction wells and 23 new injection wells are described in *Sampling and Analysis Plan for Installation of 100-HR-3 Groundwater Operable Unit Remedial Process Optimization Wells* (DOE/RL-2009-09). Groundwater data needs were met by collecting one groundwater sample from each new well and analyzing for Cr(VI), anions, metals, tritium, technetium-99, and strontium-90.

The criteria for operation of the expanded DX and HX pump-and-treat systems will be established in an upcoming revision of *Remedial Design/Remedial Action Work Plan for the 100-HR-3 Groundwater Operable Unit Interim Actions* (DOE/RL-2009-56), which will require regulatory approval. It is anticipated that the DX pump-and-treat system will maintain separation of the 100-D northern and southern plumes, and operation will remain flexible downstream from the ISRM barrier to enable extraction (initially) or injection (eventually). The HX system will continue to protect the river with extraction and injection wells in the unconfined aquifer and will also extract groundwater from two wells within the first water-bearing unit of the RUM in support of the Interim Action ROD.

Groundwater is currently extracted, treated by ion exchange, and reinjected into the unconfined aquifer. Total groundwater extraction in 100-D is approximately 2,044 L/min (540 gal/min); however, to afford contingency for system optimization, the remedial design can accommodate a total flow of 2,300 L/min (600 gal/min). Total groundwater extraction in 100-H is approximately 2,763 L/min (730 gal/min); however, to afford contingency for system optimization, the remedial design can accommodate a total flow of 3,000 L/min (800 gal/min). Water from each system is transferred to separate facilities and treated in six ion exchange treatment trains, each of which has four columns (lead, lag 1, lag 2, and polish).

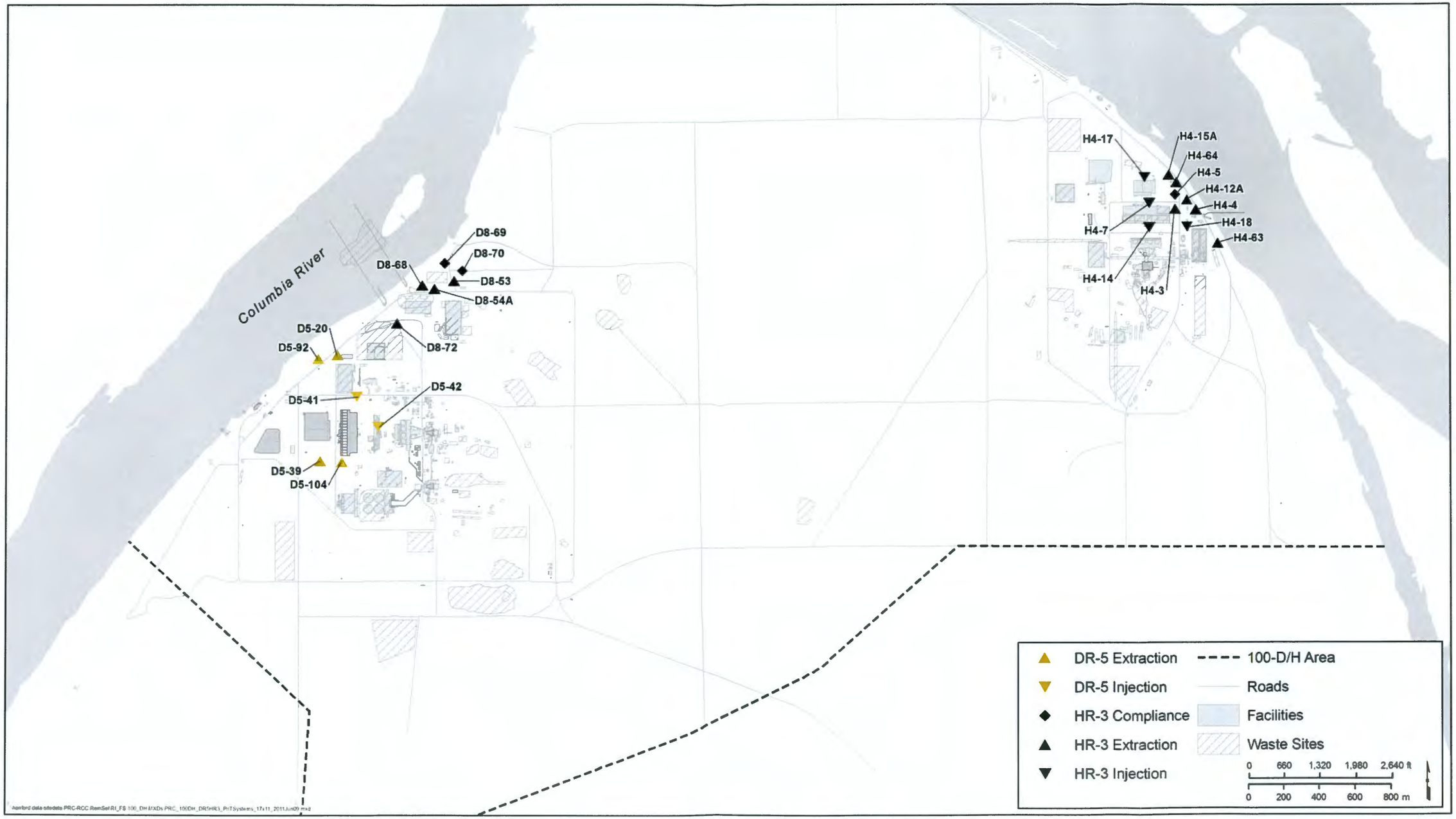

Figure 1-31. Original DR-5 and HR-3 Pump-and-Treat Systems

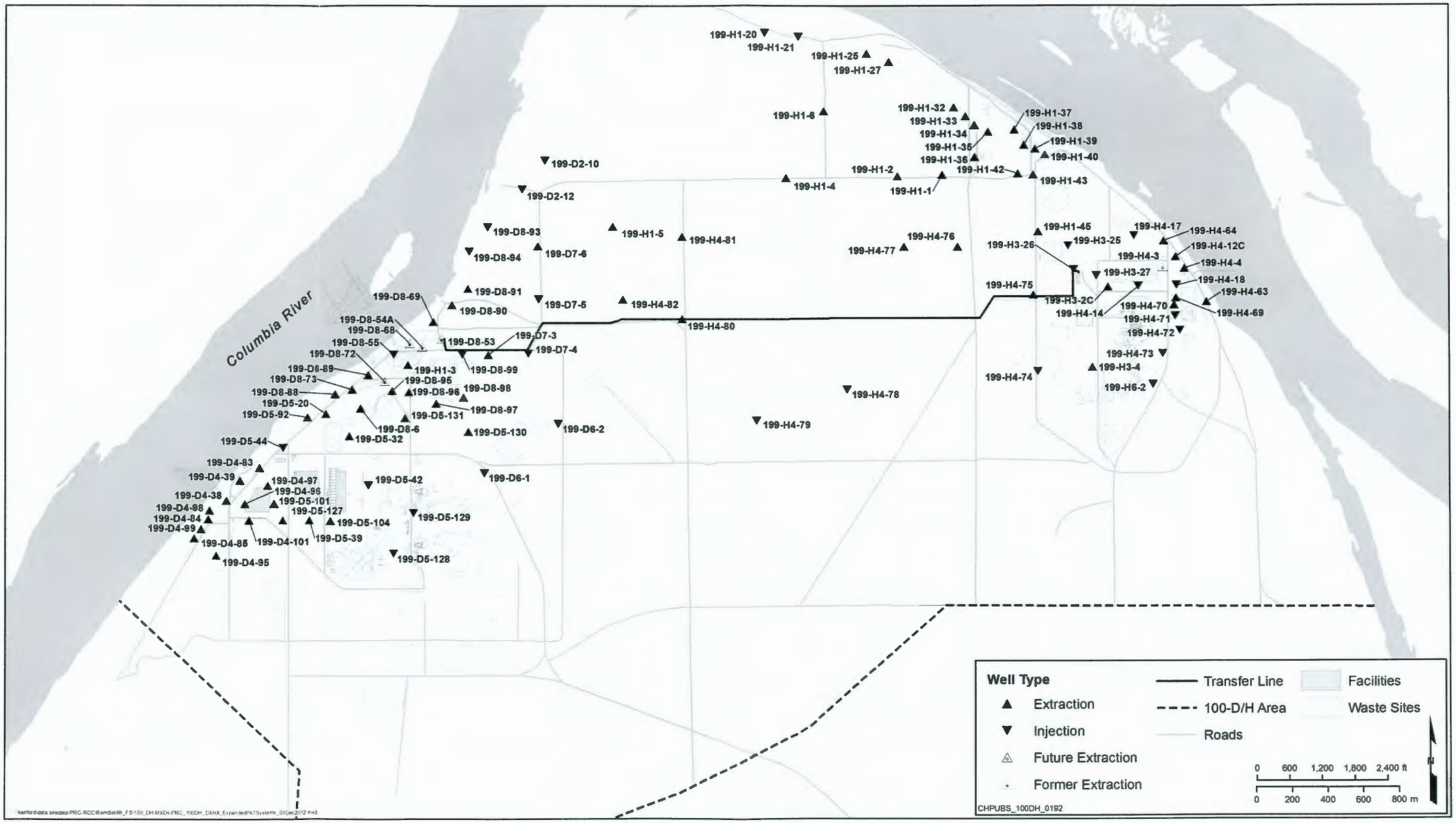

Figure 1-32. Expanded DX and HX Pump-and-Treat Systems

The combined total capacity of the expanded DX and HX pump-and-treat systems is approximately four times the combined total capacity of the former HR-3 and DR-5 pump-and-treat systems. The HR-3 (1,100 L/min [300 gal/min]) and DR-5 (190 L/min [50 gal/min]) ion exchange treatment plants have been placed on "cold standby" status for possible future use to treat hot spots or for other special applications. Table 1-11 summarizes information on schedule, design capacity, and numbers of extraction and injection wells in the pump-and-treat systems.

Table 1-11. Original and Expanded Pump-and-Treat Systems

System	Actual/Scheduled Operation	Design Capacity (L/min [gal/min])	Number of Extraction Wells	Number of Injection Wells
HR-3[a]	June 1997 to May 2011	1,100 (300)	10	4
HX[b]	October 2011 to present	3,000 (800)	33	15
DR-5[a]	July 2004 to March 2011	190 (50)	5	2
DX[b]	December 2010 to present	2,300 (600)	39	16
Total by December 2011 (DX + HX)		**5,300 (1,400)**	**72**	**31**

Notes:
Values shown are approximate based on current information and may change as further system improvements and designs occur.
Original and added wells are included in these numbers.
a. HR-3 and DR-5 are not included in the total, as they are no longer operating.
b. DX and HX refer to expanded pump-and-treat systems that focus on remediation of 100-D and 100-H, respectively.

In Situ Redox Manipulation Barrier and Fortification Test. An ISRM permeable reactive barrier is being used to treat Cr(VI) in the 100-D southern plume, in conjunction with the DX pump-and-treat system (*Fiscal Year 2008 Annual Summary Report for the In Situ Redox Manipulation Operations* [DOE/RL-2009-01]). The ISRM barrier consists of 65 wells spaced across the width of the 100-D southern plume parallel to the shoreline. The ISRM barrier was established by injecting sodium dithionite, which reacts with iron in the soil, into the aquifer through these wells to create a permeable treatment zone where contaminated groundwater can flow. The treatment zone reduces Cr(VI) to Cr(III). The majority of the remaining chemical reaction byproducts (predominantly sulfate) was then pumped out of the treated portion of the aquifer and transferred to the ISRM Evaporation Pond. The ISRM Pond is no longer used and is scheduled to be decommissioned.

The ISRM barrier continued to convert Cr(VI) to a less-toxic, less-mobile form (Cr[III]) within a portion of the aquifer. Concentrations in some downgradient wells remained above the remedial action goal of 20 µg/L (for interim action ROD) because the northeastern segment of the barrier was not working effectively. Therefore, new DX extraction wells were installed downgradient from the barrier to treat this area, as agreed to by the Tri-Parties. The ISRM barrier will continue to provide a measure of conversion to Cr(III) until the amendments are exhausted.

1.2.3.7 Riparian and Near-shore Areas

The River Corridor has been divided into three environmental zones for purposes of investigation (RCBRA [DOE/RL-2007-21, Volume I]; Integrated Work Plan [DOE/RL-2008-46]): the upland, riparian, and near-shore aquatic zones. Summary definitions of these environmental zones are presented in Section 3.9. These zones are identified here for describing the investigations in the riparian and near-shore areas.

Riparian and near-shore environments are of specific interest in the 100 and 300 Areas. The riparian zone contains plant communities requiring more water than the shrub-steppe vegetation of the upland zone, and because of the shallow water table, the riparian zone is generally green throughout the year (*Literature Review of Environmental Documents in Support of the 100 and 300 Area River Corridor Baseline Risk Assessment* [PNNL-SA-41467], hereinafter called RCBRA Literature Review). While the wildlife and food webs of the upland and riparian zones overlap, some wildlife species occur specifically within the riparian zone (*DQO Summary Report for the 100 Area and 300 Area Component of the RCBRA* [BHI-01757]). The near-shore zone is more frequently under water and is capable of sustaining aquatic biota.

There are few waste sites located within the riparian zone. However, releases and contaminant transport from waste sites could have resulted in hazardous or radioactive constituents being released to riparian and near-shore media. Groundwater from the Hanford Site discharges into the Columbia River through seeps, springs, and other upwelling locations. Discharge of groundwater could also have resulted in hazardous or radioactive constituents being released to riparian or near-shore zones.

Investigations historically conducted in the riparian and near-shore areas of 100-D/H are summarized in the RCBRA Literature Review (PNNL-SA-41467). In addition to these historical investigations, other sampling and analytical data have been collected from riparian and near-shore areas as part of the Surface Environmental Surveillance Program (SESP). The data from the SESP are summarized in the Annual Environmental Reports for the Hanford Site. Finally, investigations of riparian and near-shore areas were conducted as part of the RCBRA ecological risk assessment (ERA) (RCBRA [DOE/RL-2007-21, Volume 1]; *100 Area and 300 Area Component of the RCBRA Sampling and Analysis Plan* [DOE/RL-2005-42], hereinafter called the RCBRA SAP).

Investigation of Ground-Water Seepage from the Hanford Shoreline of the Columbia River (PNL-5289) identified riverbank springs and groundwater seeps along the length of the Hanford Site shoreline in 1983. Contaminant data specific to the 100-D Area riverbank springs indicated slightly elevated concentrations of nitrates, with a maximum concentration of 2,000 μg/L. Tritium and nitrate were detected in riverbank spring water samples in the 100-H Area. The maximum concentration of tritium in spring water samples was 4,000 pCi/L, compared with a maximum of 64,900 pCi/L in a groundwater sample collected from a well. The maximum concentration of tritium in an adjacent surface water sample was 65 pCi/L. The maximum nitrate concentration in a 100-H spring water sample was 5,750 μg/L.

Sampling of riverbank springs and adjacent surface water was performed along the Hanford Reach from the 100-B Area to the Hanford Town Site in 1991. Two springs were sampled at 100-D (*Sampling and Analysis of 100 Area Springs* [DOE/RL-92-12]). Riverbank spring water from these two locations had the highest concentrations of chromium (72 and 124 μg/L) measured during this entire study. In addition, tritium (1,200 and 3,100 pCi/L), strontium-90 (1.8 and 4.5 pCi/L), technetium-99 (0.3 to 4.9 pCi/L), and total uranium (0.9 and 1 pCi/L) were detected in riverbank spring water. Columbia River water samples collected near these riverbank spring sampling locations were below detectable limits, except for total uranium; total uranium concentrations in surface water were judged to be similar to concentrations in background locations. Samples of riverbank sediments had the highest concentrations of chromium (maximum of 122 mg/kg) for all sediment sampling locations. Gross beta concentrations in sediments ranged from 18 to 19 pCi/g. Concentrations of all of radionuclides were below detection or similar to concentrations in background locations. Five springs were sampled at 100-H. Riverbank spring water from these locations contained detectable levels of chromium (16 to 52 μg/L), tritium (400 to 3,800 pCi/L), strontium-90 (0.4 to 13 pCi/L), and total uranium (0.7 to 1.2 pCi/L). River water samples collected in the vicinity of these riverbank springs had concentrations below the detectable levels for chromium, strontium-90, and technetium-99. Total uranium levels in river water samples at 100-H were detected but were similar to levels in background locations. In addition to elevated chromium

concentrations (maximum of 122 mg/kg), riverbank spring sediment from 100-H had detectable levels of strontium-90, cesium-137, potassium-40, radium-226, thorium-228, and thorium-232 (*Sampling and Analysis of 100 Area Springs* [DOE/RL-92-12]).

Also in 1991, six sediment samples were collected from four locations in the vicinity of 100-D (*100 Area Columbia River Sediment Sampling* [WHC-SN-EN-TI-198]). Maximum concentrations of potassium-40, radium-226, uranium-233/234, uranium-238, thorium-228, and thorium-232 were no different than concentrations in upstream background samples. Two other radionuclides, cesium-137 and europium-152, had concentrations higher than those of the background samples (cesium-137, maximum 1.3 pCi/g; europium-152, maximum 0.9 pCi/g). Nine radionuclides were detected in the 100-D Area, but they were not detected in the background samples or in samples from the 100-B/C or the 100-K Areas. These radionuclides were cobalt-60 (maximum 0.41 pCi/g), europium-154 (maximum 0.04 pCi/g), europium-155 (maximum 0.04 pCi/g), radium-228 (maximum 0.54 pCi/g), thorium-231 (maximum 0.29 pCi/g), thorium-234 (maximum 0.69 pCi/g), uranium-235 (maximum 0.02 pCi/g), neptunium-237 (maximum 0.48 pCi/g), and americium-241 (maximum 0.24 pCi/g). Sediment samples were collected from four locations near the 100-H Area. Concentrations of most radionuclides detected (potassium-40, radium-226, thorium-228 and thorium-232) were similar to sediment concentrations from a reference site at Vernita, upstream from the Hanford Site. Sediment concentrations of europium-152 and uranium isotopes that were detected were above the reference level at Vernita. The maximum concentration for europium-152 was 1.8 pCi/g; for uranium-233/234, the maximum concentration was 2.3 pCi/g; and for uranium-238, the maximum concentration was 2.3 pCi/g. Plutonium-239/240 was detected in a single sediment sample, with a concentration of 0.07 pCi/g.

The SESP project does not routinely monitor Columbia River water near 100-D and 100-H. The nearest routinely monitored locations are the annual cross-river transects at 100-N and 100-F. Riverbank spring locations near 100-D and 100-H have been monitored by the SESP. The trends in metals concentrations in spring samples are reported to have been consistent over the past several years. With the exception of chromium, concentrations of metals in spring samples in 100-D and 100-H were below Washington State chronic ambient surface water quality criteria in "Water Quality Standards for Surface Waters of the State of Washington" (WAC 173-201A). Concentrations of radionuclides detected in springs in 2009 were reported to be similar to those in previous years. Potassium-40, cesium-137, and uranium isotopes were the only radionuclides reported above minimum detectable concentrations. Concentrations of radionuclides and metals in 100-D and 100-H sediments were similar to levels detected in previous years (*Hanford Site Environmental Report for Calendar Year* 2009 [PNNL-19455], hereinafter called 2009 Sitewide Environmental Report).

Investigations of riparian and near-shore areas were conducted in support of the RCBRA. Riparian and near-shore areas were selected where affected media (seeps, springs, or runoff) may have created exposure pathways to biota (RCBRA SAP [DOE/RL-2005-42]). Riparian sampling locations also were identified based on radiation field survey results (RCBRA SAP [DOE/RL-2005-42], Appendix C; *DQO Summary Report for the 100 Area and 300 Area Component of the RCBRA* [BHI-01757], Appendix H). Radiation survey results and detection of chromium in groundwater, aquifer tube, and biota (bivalve) samples provided the primary basis for selection of riparian and near-shore study sites in 100-D/H (RCBRA SAP [DOE/RL-2005-42], Table C-1). Eight near-shore (aquatic) study sites were near 100-D/H. Eight riparian study sites were also upstream and downstream from 100-D/H, and between the 100-D and 100-H operational areas.

Sample collection rationale and techniques varied by area and medium. Investigation areas characterized by data collected under the SAP included the upland, riparian, and near-shore river zones. Sites selected for sampling were identified based on existing data demonstrating a range of contaminant concentrations.

Reference sites were identified using evidence and knowledge of areas not affected by contaminant release and selected based on physical and ecological similarity to onsite investigation areas.

Media collected in the upland and riparian zones included soil, vegetation, invertebrates, small mammals, and kingbirds (kingbirds in the riparian zone only). Near-shore media included sediment, interstitial pore water, surface water, benthic macroinvertebrates, clams, and sculpin. Toxicity testing was performed on soil, sediment, and water to provide Hanford Site-specific information on the ecological effects of contaminant mixtures and contaminant bioavailability. The results of these tests are used to make informed inferences on the toxicity of contaminants to Hanford Site biota. A more detailed discussion of the results from the RCBRA in riparian and near-shore areas is presented in Chapter 4 of this RI/FS report.

1.2.4 Risk Assessments

Risk assessments have been conducted for the 100 Area to provide the foundation for establishing the need for remedial action to protect human health and the environment. Three key risk assessments, i.e., the qualitative risk assessments (QRAs) performed in the early 1990s, RCBRA (DOE/RL-2007-21), and CRC (DOE/RL-2010-117), are summarized below. The results of RCBRA and the CRC are described in more detail (and used) in Sections 4, 6, and 7 of this RI/FS.

Qualitative Risk Assessments. QRAs were conducted to define the basis for remedial actions under Interim Action RODs (Past-Practice Strategy [DOE/RL-91-40]). Assessment of human health risks in the QRAs was based on frequent use and occasional use scenarios, which reflected current guidance for that time. COPCs were identified from the historical site data and data collected during the LFIs, taking into consideration Hanford Site background activity of radionuclides and inorganic concentrations in the vadose zone and risk-based screening using residential exposure parameters (*Hanford Site Risk Assessment Methodology* [DOE/RL-91-45]). Human health risks presented in the QRAs were based on the maximum concentrations detected in waste site vadose zone material and in groundwater. Human health risks were quantified for a limited set of exposure pathways (soil ingestion, fugitive dust or volatile inhalation, and external exposure). Ecological risks were estimated using a streamlined approach, focusing on a single organism, the Great Basin pocket mouse, using the assumption that the waste site was the home range.

River Corridor Baseline Risk Assessment. The RCBRA (DOE/RL-2007-21, Volumes I and II) has been conducted to characterize current and potential future risks to human health and the environment that may be posed by releases of contaminants in the River Corridor. The RCBRA supports the current remediation decisions and consists of a human health risk assessment (HHRA) and an ecological risk assessment (ERA).

The HHRA (DOE/RL-2007-21, Volume II) provides as assessment of residual risks for remediated waste sites using the residential land use exposure scenario that was the basis for the cleanup values for the interim action ROD cleanup. In addition, the HHRA provides an assessment of residual risks for remediated waste sites and broad areas using a broad range of hypothetical receptors, including adults and children living in the River Corridor, Tribal members, recreational users, and adults working on the site. A screening level groundwater risk assessment is also completed to evaluate potential risks associated with potential exposure to contaminated groundwater.

One of the objectives of the RCBRA is to determine if the interim actions were protective of ecological receptors, which is achieved through the evaluations conducted in the ERA (DOE/RL-2007-21, Volume I). The scope of this ERA addresses upland areas, including remediated CERCLA waste sites, the White Bluffs and Hanford Townsites, and the 300 Area. In addition, the ERA evaluates the riparian and near-shore aquatic zones as well as areas of groundwater emergence on the south and west shoreline of the Columbia River. The ERA approach is based on an overall conceptual site model that summarizes

what is known about site conditions (including the location of contamination sources) and describes transport and exposure pathways through various environmental media that may be important in evaluating potential exposure to ecological receptors. Where possible, multiple lines of evidence were employed to comprehensively evaluate the potential for adverse effects on plants, invertebrates, and wildlife.

Columbia River Component Risk Assessment. The CRC (DOE/RL-2010-117) provides a comprehensive HHRA (Volume II) and a screening-level ERA (Volume I). The intent of the CRC HHRA was to complete the assessment of the "bank-to-bank" Hanford Reach and downstream areas (i.e., Lake Wallula) of the Columbia River, characterizing risk in areas not previously addressed under the RCBRA (DOE/RL-2007-21). Human exposure scenarios include an avid angler, casual user, subsistence farmer, and a Native American (Yakama Nation) subsistence fisher. The CRC HHRA identifies fish consumption as the largest potential contribution to overall human health risks.

The CRC ERA(DOE/RL-2010-117, Volume I) also uses analytical chemistry collected from surface water, sediment, pore water, island soils, and fish to evaluate the potential for risk to ecological receptors including aquatic life living within the Columbia River and wildlife frequenting or inhabiting the islands within the river. Based on a screening-level ecological risk assessment, the CRC ERA(DOE/RL-2010-117, Volume I) identifies some contaminants as contaminants of potential ecological concern (COPECs); mostly metals. The CRC further considered whether COPECs are attributable to Hanford Site-related sources.

Conclusions from the CRC HHRA (DOE/RL-2010-117, Volume II) are discussed in Section 6.4.2, and the CRC ERA (DOE/RL-2010-117, Volume I) are reviewed in Section 7.6.2.

1.2.5 CERCLA Five-Year Review

Effectiveness of the interim actions discussed previously is evaluated through the CERCLA 5-year review process. This review determines whether the selected remedy remains protective of human health and the environment. Since the issuance of the first interim ROD, there have been two 5-year reviews for the 100 Area NPL (40 CFR 300, Appendix B) Site. *The Second CERCLA Five-Year Review Report for the Hanford Site* (DOE/RL-2006-20) listed five issues and recommended six actions for 100-D/H:

- **Issue 8.** Groundwater monitoring data indicate there is an unidentified chromium vadose source in the 100-D Area near the demolished 190-DR clear wells.
  - **Action 8-1.** Complete a field investigation to investigate additional sources of chromium groundwater contamination within the 100-D Area. Perform additional geologic and geochemical investigations of the vadose zone in the 100-D Area. Investigations were conducted for both the southern and northern plume (*Investigation of Hexavalent Chromium Source in the Southwest 100-D Area* [SGW-38757]; *Report on Investigation of Hexavalent Chromium Source in the Northern 100-D Area* [DOE/RL-2010-40]). In addition, several boreholes and wells have been installed as part of the RI/FS (100-D/H Work Plan [DOE/RL-2008-46-ADD1]). This completed the required action.
- **Issue 9.** There is less than adequate data to characterize potential chromium groundwater contamination between the 100-D and 100-H Area, in the area known as the "horn."
  - **Action 9-1.** Perform additional characterization of the aquifer for chromium contamination between the 100-D and 100-H Area, in the area known as the "horn," and evaluate the need to perform remedial action to meet the RAOs of the 100-D ROD for interim action. This issue will also be addressed in the record of decision. This action was previously completed and is

summarized in *Hydrogeological Summary Report for 600 Area Between 100-D and 100-H for the 100-HR-3 Groundwater Operable Unit* (DOE/RL-2008-42).

- **Action 9-2.** Incorporate the "horn" area into the 100-HR-3 interim ROD treatment zone if Action 9-1 indicates the "horn" contains a groundwater chromium plume that needs immediate remediation. DOE has completed the RPO evaluation of the pump-and-treat system and is currently implementing the results (*100-HR-3 Remedial Process Optimization Modeling Technical Memorandum* [SGW-40044]). DOE installed additional extraction and injection wells throughout the horn area in FY 2009 and FY 2010 as part of RPO. This completed the required action.

- **Issue 10.** Some of the groundwater wells near the 182-D reservoir show conductivity values similar to values expected for raw water, indicating some leakage from the reservoir.
  - **Action 10-1.** Issue direction to the operating contractor to change operations to minimize leakage from the 182-D reservoir further. Direction was given to the contractor, and the action was completed. The leaks and their effect on groundwater flow have significantly diminished since the reduction of storage volume in the reservoir in 2004, to the point that influences on groundwater flow from reservoir leakage are indistinguishable from those created by nearby pump-and-treat activities (*Project Work Plan: Hanford 100-D Area Treatability Demonstration: Accelerated Bioremediation through Polylactate Injection* [PNNL-SA-50369]).

- **Issue 11.** A few wells within the in situ oxidation reduction (redox) manipulation (ISRM) barrier have shown break through much sooner than expected.
  - **Action 11-1.** Perform initial limited iron amendments to the ISRM barrier to evaluate whether this enhances performance. Results of the iron amendment tests are documented in *Treatability Test Report on Mending the In Situ Redox Manipulation Barrier Using Nano-Size Zero Valent Iron* (DOE/RL-2009-35). This completed the required action.

- **Issue 12.** Groundwater samples from deeper wells extending below the aquitard exceed the drinking water standard (100 µg/L) for chromium. The extent of chromium contamination in this zone is not well understood.
  - **Action 12-1.** Perform additional characterization of the aquifer below the initial aquitard. DOE installed three wells in the horn area, screened in the RUM unit (*Hydrogeological Summary Report for 600 Area Between 100-D and 100-H for the 100-HR-3 Groundwater Operable Unit* [DOE/RL-2008-42]), and continued to monitor three wells in the 100-H Area. Five wells (three in 100-H and two in 100-D) were installed as part of the 100-D/H Work Plan (DOE/RL-2008-46-ADD1). The wells have been drilled through the RUM and screened within the first water-bearing unit encountered. This completed the required action.

The third 5-year review was published in March 2012 (*Hanford Site Third CERCLA Five-Year Review Report* [DOE/RL-2011-56]). There were no issues identified for the 100-D/H Area.

1.2.6 Summary

Chapter 1 summarized historical information, prior assessments and remediation work, treatability tests, and other relevant studies. This information provides a picture of current 100-D/H site conditions and establishes a foundation for the remainder of the RI/FS document.

Hanford-related contamination of the 100-D/H Area began with reactor construction in 1943 and continued until related operations ceased. Radiological and chemical contamination of soil and groundwater resulted that remains to date. Characterization efforts have delineated the nature and extent of groundwater and vadose zone contamination. Risks to human health and the environment were recognized early, resulting in operational actions to limit transport of contaminants to potential receptors. Despite those actions, contamination levels exceeding standards have resulted. Interim remedial actions, including groundwater pump-and-treat and in situ treatments have been deployed to address groundwater contamination. Similarly, demolition of surface facilities and excavation of contaminated soil have been performed to begin the process of restoring the land and groundwater to beneficial use.

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2 Study Area Investigation

The study area investigation included the vadose zone and groundwater in 100-D/H, as guided by an approved work plan. Development of the 100-D/H Work Plan (DOE/RL-2008-46-ADD1) was based on review and evaluation of relevant documented information and data. The work plan identified additional information to support a remedial alternative evaluation and decision. This chapter describes the data needs (Table 2-1), the data collected to fill them, and the corresponding scope of work (including field activities, tests, analyses, and data sources) that was designed and carried out in the RI/FS. Chapters 3, 4, and 5 present results of the RI/FS activities. These chapters include data collected pursuant to the work plan referenced above as well as from previous studies and historical information to identify the nature and extent of contamination. The scope of work is outlined in the Integrated Work Plan (DOE/RL-2008-46), 100-D/H Work Plan (DOE/RL-2008-46-ADD1), and the 100-D/H SAP (DOE/RL-2009-40).

Highlights

- The investigation addressed the data gaps identified in the 100-D/H Work Plan (DOE/RL-2008-46-ADD1).
- Ten soil boreholes (five of which were converted to temporary wells), five test pits, twelve unconfined aquifer monitoring wells, and five RUM wells were installed.
- Seventy wells were installed as part of the RPO. Data from that effort were incorporated into the analysis.
- Spatial and temporal groundwater sampling was conducted at 52 wells.

The following sections of this chapter describe the field activities, other investigations and ongoing activities that contributed to this RI/FS. These sections summarize the scope of work, document any deviations from the work plan, and explain the rationale for the deviations. They also present details of investigation activities conducted under other scopes of work that may affect the development of remedial action alternatives, including *Remedial Investigation Work Plan for Hanford Site Releases to the Columbia River* (DOE/RL-2008-11), hereinafter called Columbia River RI Work Plan, the RCBRA (DOE/RL-2007-21), and ongoing groundwater and aquifer tube monitoring.

This chapter summarizes recent field activity, and subsequent chapters describe the results of this work and integrate it with the historical information (summarized in Chapter 1) to update the CSM and to identify and evaluate options for achieving RAOs.

2.1 Remedial Investigation Activities

The RI field activities included boreholes, test pits, groundwater monitoring well installation, spatial and temporal groundwater monitoring, and associated sampling and analysis for each activity. Table 2-1 presents the relationship of the field efforts and the data needs that were identified in the 100-D/H Work Plan (DOE/RL-2008-46-ADD1). Table 2-2 includes the supplemental investigations identified in the work plan and other investigations that may affect cleanup decisions for 100-D/H. Table 2-3 summarizes the field program and Table 2-4 shows the field samples collected. Figures 2-1, 2-2, and 2-3 present the locations where field sampling was conducted for 100-D, 100-H, and the horn, respectively.

Table 2-1. Data Gaps and Work Conducted per the RI/FS Work Plan for 100-D/H

Data Gap	Data Need	Scope of Work	Work Conducted/Section with Discussion	Data Gap Filled?
1. Vadose zone contaminant nature and extent needed to assess protection of groundwater beneath unremediated waste sites.	Characterize below unremediated waste sites to assess nature and extent of contamination in the vadose zone.	Continue interim remedial actions as they have demonstrated to be efficient in obtaining the necessary data during remediation. Obtain data documenting the remaining residual contamination following completion of the interim remedial action.	Interim remedial action and sampling have continued at 100-D/H waste sites. Data for sites that were reclassified before June 2012 have been evaluated through the risk evaluation process presented in this report. The verification data collected during the interim remedial actions are presented in Appendices D and E and evaluated in Chapters 5, 6, and 7. Knowledge and experience from the implementation of interim remedial actions is also used in the evaluation of technologies in Chapter 8.	Yes
2. Vadose zone contaminant nature and extent needed to assess protection of groundwater beneath remediated waste sites.	Characterize beneath and adjacent to remediated waste sites to assess the nature and extent of contamination in the vadose zone.	Drill 10 boreholes and install 2 groundwater monitoring wells; also excavate 5 test pits. Collect and analyze samples to assess vertical extent of contamination in the vadose zone. Excavate test pits to assess potential Cr(VI) sources and contaminant concentrations to maximum depths of 6.1 to 7.6 m (20 to 25 ft).	Ten boreholes drilled for waste site characterization were located as follows (borehole ID): • 116-D-1B Trench (C7855) • 116-D-7 Retention Basin (C7851) • 116-DR-1&2 Trench (C7852) • 116-DR-9 Retention Basin (C7850) • 116-H-1 Trench (C7864) • 116-H-4 Crib (C7862) • 116-H-6 Solar Evaporation Basin (C7860) • 116-H-7 Retention Basin (C7861) • 118-D-6:3 Reactor Fuel Storage Basin (FSB) (C7857) • 118-H-6:3 Reactor FSB (C7863) Groundwater monitoring wells were installed at the following waste sites (Well ID; borehole ID; SAP Reference number): • 100-D-56:1 Pipeline (Well 199-D5-143; C8375; Well 9 redrill) • 100-D-12 French Drain (Well 199-D5-144; C8668; Well R5 redrill) • 116-D-1A Trench (Well 199-D5-132; C7622; Well 4) A test pit was excavated at each of the following waste sites: • 100-D-12 French Drain (site has a well also) • 100-D- 4 Trench • 116-D-4 Crib • 116-H-2 Trench	Yes

Table 2-1. Data Gaps and Work Conducted per the RI/FS Work Plan for 100-D/H

Data Gap	Data Need	Scope of Work	Work Conducted/Section with Discussion	Data Gap Filled?
			• 1607-H4 Septic System Samples were collected in boreholes through the vadose zone, including at the Hanford formation/Ringold Formation contact (where present). Soil samples from boreholes and test pits were analyzed for location specific target analytes, field screening parameters, and batch leach testing. Borehole information and analytical data are presented in Chapters 3 and 4, and Appendices C, D, and M.	
3. Vadose zone contaminant nature and extent needed to assess protection of groundwater beneath and around reactor structures.	Characterize beneath and around the reactor structures to assess nature and extent of contamination in the vadose zone.	Drill two boreholes near the reactor structures in areas most likely to contain soil contamination. Collect and analyze samples to assess vertical extent of contamination in the vadose zone.	Soil boreholes were drilled at two FSBs. (These boreholes also meet the data needs in Data Gap 2.) • 118-D-6:3 Reactor FSB (C7857) • 118-H-6:3 Reactor FSB (C7863) Samples were collected through the vadose zone, including at the Hanford formation/ Ringold Formation contact (where present). Samples were analyzed for location specific target analytes, field screening parameters, and batch leach testing. Analytical results from these two boreholes are presented in Chapter 3 (Section 3.4.3),Chapter 4 (Sections 4.3.1) and Appendix D.	Yes
4. Unidentified waste sites (orphan/discovery sites) may exist in 100-D/H. Unidentified sites may include chromium contamination in surface soil because of undocumented spills.	Identify new waste sites and potential sources of contamination.	Complete the OSE process in the horn and conduct a supplemental survey of 100-D Operational Area.	The OSE process was completed in the horn area (*100-F/IU-2/IU-6 Area – Segment 4 Orphan Sites Evaluation Report* [OSR-2011-0001]). The evaluation identified six additional waste sites (600-380, 600-381, 600-382, 600-383, 600-384, and 600-385). A supplemental survey will be conducted at 100-D when remediation has been completed.	Yes

Table 2-1. Data Gaps and Work Conducted per the RI/FS Work Plan for 100-D/H

Data Gap	Data Need	Scope of Work	Work Conducted/Section with Discussion	Data Gap Filled?
5. The nature and extent of contamination in the unconfined aquifer above cleanup standards has not been defined in selected areas.	Define the extent of groundwater contamination above cleanup standards in select areas of the unconfined aquifer.	Install three new aquifer tubes and five new wells at approved locations in 100-D. Install three new aquifer tubes and five new wells at approved locations in 100-H. Sample the wells for groundwater COPCs presented in the 100-D/H SAP (DOE/RL-2009-40).	**100-D:** A cluster of new aquifer tubes and seven new wells were installed. One additional aquifer tube was added to the cluster of three planned and represents a second deep zone. The aquifer tubes and wells were placed and sampled according to the 100-D/H Work Plan (DOE/RL-2008-46-ADD1) to define the strontium-90 and Cr(VI) plumes, with one additional aquifer tube installed. Samples were collected and analyzed from the Hanford/Ringold lithologic change and from groundwater. The analytical data from the aquifer tubes are presented in Chapter 4 (Section 4.4) and in Appendix D. The groundwater data are presented in Chapter 4 and Appendix D.	Yes[a]
			Aquifer tubes (Sampling point 1) identifiers are: • C7645 • C7646 • C7647 • C7648 The well identifiers are: • 199-D3-5 (C7620; Well 2) • 199-D5-133 (C7621; Well 3) • 199-D5-132 (C7622; Well 4–also for Data Gap 2) • 199-D6-3 (C7623; Well 5) • 199-D5-143 (C8375, Well 9 redrill) • 199-D5-140 (C7866, mislocated Well 9) • 199-D5-144 (C8668, Well R5 redrill- also for Data Gap 2	
			100-H: A cluster of two new aquifer tubes and five new wells were installed. The deep locations for the aquifer tube cluster were not installed because of encountering the RUM at a shallow depth. There was insufficient water at the deep locations for aquifer tube installation. The installed aquifer tubes and wells were placed and sampled according to the 100-D/H Work Plan (DOE/RL-2008-46-ADD1) to define the strontium-90, Cr(VI), and nitrate plumes as outlined in the work plan. Samples were collected and analyzed at the Hanford/Ringold lithologic change and from groundwater. The analytical data from the aquifer tubes are presented in Chapter 4 (Section 4.4) and in Appendix D. The groundwater data are presented in Chapter 4 and Appendix D.	

Table 2-1. Data Gaps and Work Conducted per the RI/FS Work Plan for 100-D/H

Data Gap	Data Need	Scope of Work	Work Conducted/Section with Discussion	Data Gap Filled?
			Aquifer tube (Sampling point 8) identifiers are: • C7649 • C7650 The well identifiers are: • 199-H3-6 (C7626; Well 6) • 199-H3-7 (C7627; Well 7) • 199-H6-3 (C7628; Well 10) • 199-H6-4 (C7629; Well 11) • 199-H1-7 (C7630; Well 12)	
6. The level of groundwater contamination entering the Columbia River (in particular, the hyporheic zone) is not well known.	Evaluate the utility and adequacy of aquifer tubes in supporting the understanding of groundwater contamination entering the Columbia River. Collect groundwater upwelling data.	Continue collecting aquifer tube sampling data and information per the existing program. Collect groundwater upwelling samples in the Columbia River (Columbia River RI Work Plan [DOE/RL-2008-11]). A task was included in the Integrated Work Plan (DOE/RL-2008-46) for evaluating and developing an approach to obtain data that will demonstrate compliance with ambient water quality criteria in the river, for proposed new ROD.	The existing aquifer tube-sampling program is being continued. Upwelling samples were collected as per the Columbia River RI Work Plan (DOE/RL-2008-11). Results are presented in *Field Summary Report for Remedial Investigation of Hanford Site Releases to the Columbia River, Hanford Site, Washington: Collection of Surface Water, Pore Water, and Sediment Samples for Characterization of Groundwater Upwelling* (WCH-380, Rev. 0) and discussed in Chapter 4 of this report.	Yes

Table 2-1. Data Gaps and Work Conducted per the RI/FS Work Plan for 100-D/H

Data Gap	Data Need	Scope of Work	Work Conducted/Section with Discussion	Data Gap Filled?
7. The nature and extent of contaminants beneath the unconfined aquifer has not been evaluated.	Collect physical and hydrogeologic information to further support the evaluation of contaminant fate and transport beneath the unconfined aquifer.	100-D: Drill and sample soil and groundwater from two new boreholes (R4 and R5) drilled through the RUM and into the Ringold unit B. 100-H: Drill and sample soil and groundwater from three new boreholes (R1, R2, and R3) drilled through the RUM and into the Ringold unit B. Collect soil samples at 1.5 m (5 ft) into the RUM at the eight wells installed during the pump-and-treat system expansion.	**100-D:** Two new boreholes (R4 and R5) were planned to extend into the RUM, down to the Ringold Formation unit B (presumed) aquifer. These were completed as wells screened in the first water-bearing unit of the RUM. The well identifiers are as follows: • 199-D5-134 (C7624; Well R4) • 199-D5-141 (C7625; Well R5 – mislocated) **100-H:** Three new boreholes (R1, R2, and R3) were drilled through the RUM and into the Ringold Formation unit B (presumed) aquifer. These were completed as wells screened in the first water-bearing unit of the RUM. The well identifiers are as follows: • 199-H3-9 (C7639; Well R1) • 199-H3-10 (C7640; Well R2) • 199-H2-1 (C7631; Well R3) Results from the boreholes are presented in Chapter 3. **Soil Samples:** Split-spoon soil samples at 1.5 m (5 ft) total depth into the RUM were collected from eight wells that were installed as part of the expansion of the 100-D/H pump-and-treat system. Samples were collected in 100-D wells at the Hanford/ Ringold lithologic change. For new wells near waste sites, additional split-spoon samples were collected above and below the Hanford formation/Ringold Formation unit E contact. For deep wells, split-spoon soil samples were collected from above the water table; within the unconfined aquifer; within the deep unconfined aquifer at the top of the RUM; at two depths within the RUM (outside of any water producing zone); and within the Ringold Formation unit B (presumed), per the 100-D/H SAP (DOE/RL-2009-40). Analytical results from the soil boreholes are summarized in Chapter 4 (Section 4.3) and presented in Appendix D. Groundwater samples were collected during drilling of R1 through R5 from the unconfined aquifer, water-bearing units of the RUM, and the Ringold Formation unit B aquifer (presumed) for field screening parameters and COPC analysis. Groundwater results for the RUM wells are presented in Chapter 4 (Section 4.4) and in Appendix D.	Yes
		Collect samples in the 100-D wells at the Hanford/Ringold geologic contact.		

Table 2-1. Data Gaps and Work Conducted per the RI/FS Work Plan for 100-D/H

Data Gap	Data Need	Scope of Work	Work Conducted/Section with Discussion	Data Gap Filled?
8. It is unknown if contamination within the RUM will adversely affect aquatic receptors in the Columbia River.	Update bathymetric data for the river within 100-D/H to support calculations of contaminant transport to the river and ecological receptors.	Evaluate digital bathymetric data recently compiled by PNNL.	The data were evaluated to provide a better understanding of the relationship between the riverbed and the groundwater flow in the adjacent aquifer. A summary of the bathymetry is included in Section 2.1.7, and the data are incorporated into geologic cross sections in Appendix M. The groundwater upwelling sample results were used to determine if 100-D/H ecological receptors are adversely affected by unconfined aquifer or RUM contamination.	Yes
9. The rate of chemical and hydraulic exchange between the aquifer and the river in the nearshore is unknown.	Collect geochemical and hydrogeologic data to evaluate nearshore area groundwater contaminant fate and transport.	The nearshore area is directly affected by river stage. Available data to provide adequate understanding of groundwater flow paths, contaminant migration, and mixing in the nearshore area have been limited. TPA (Ecology et al., 1989a) milestones state that compliance with cleanup standards in this area is a target.	No specific data collection activities were proposed in this RI. Data from other efforts were used in the RI/FS as defined in the *Remedial Design and Remedial Action Work Plan for the 100-HR-3 and 100-KR-4 Groundwater Operable Units' Interim Action* (DOE/RL-96-84) and associated *Interim Action Monitoring Plan for the 100-HR-3 and 100-KR-4 Operable Units* (DOE/RL-96-90), the SAP for Aquifer Sampling Tubes (DOE/RL-2000-59), and the Columbia River RI Work Plan (DOE/RL-2008-11). Relevant results from these activities are discussed in Chapters 3, 4 and 5.	Yes
10. The mechanism to explain the persistence of the Cr(VI) plume is unknown.	Collect soil and water samples from the following units: (1) vadose zone, (2) deep vadose zone, (3) rewetted zone, (4) unconfined aquifer, (5) above the RUM, and (6) within the RUM.	Soil and water analyses were needed to determine the potential for each unit to contain sufficient contamination to be a continuing source of groundwater contamination.	Soil and water samples were collected and analyzed per the 100-D/H SAP (DOE/RL-2009-40). Sample locations are identified under Data Gaps 5 and 7, and summarized in Appendix C. Analytical results are presented in Chapter 4 (Section 4.3) and in Appendix D.	Yes

Table 2-1. Data Gaps and Work Conducted per the RI/FS Work Plan for 100-D/H

Data Gap	Data Need	Scope of Work	Work Conducted/Section with Discussion	Data Gap Filled?
11. Potential alternative remedial technologies have not been sufficiently investigated.	Evaluate alternative potential remedial technologies.	Groundwater contamination above aquatic standards and drinking water maximum contaminant levels had been detected in 100-D, 100-H, and the horn area. Interim remedial actions are currently operating to address contaminated areas in 100-D, 100-H, and the horn area. The RI collected data necessary for comparison of potential final remedies in the FS.	Data were collected during the RI and incorporated into the evaluation of technologies presented in Chapter 8 and Appendix I.	Yes
12. Insufficient data are available to support fate and transport modeling.	Collect additional data to support future fate and transport modeling. Assess the physical and hydraulic properties of soil and confirm contaminant distribution coefficients (K_d) to support modeling.	On selected soil samples, physical properties, hydraulic properties, contaminant concentrations, and leaching behavior were evaluated.	Soil samples from each of the deep boreholes, the eight monitoring wells installed during expansion of the pump-and-treat system, seven boreholes drilled through remediated waste sites, and one test pit were collected and analyzed per the 100-D/H SAP (DOE/RL-2009-40). Fate and transport modeling results are presented in Chapter 5 and Appendix F.	Yes
13. Data are needed to better define the spatial and temporal distribution of groundwater contamination.	Collect and analyze groundwater samples from select groundwater monitoring wells.	Additional groundwater data were needed that are spatially representative of 100-D/H, reflect river stage influence, and include groundwater COPCs.	A total of 53 existing wells were scheduled for sampling and analysis for the temporal and spatial analysis. Of these, 52 were included in the analysis. Well 199-D5-41 was sampled once, and then converted to an injection well per *Tri-Party Agreement Change Notice Form: DOE/RL-2009-40, Sampling and Analysis Plan for the 100-DR-1, 100-DR-2, 100-HR-1, 100-HR-2, and 100-HR-3 Operable Units Remedial Investigation/ Feasibility Study, Rev. 0* (TPA-CN-368). The data from this well were not used in the spatial and temporal analysis because the well was sampled only once and did not provide the statistical basis needed. Groundwater results from this sampling effort are presented in Chapters 4 and 6 and Appendix D.	Yes

Table 2-1. Data Gaps and Work Conducted per the RI/FS Work Plan for 100-D/H

Data Gap	Data Need	Scope of Work	Work Conducted/Section with Discussion	Data Gap Filled?
14. Leakage (current and future) from the 182-D Reservoir and export water lines may affect groundwater flow, contaminant transport, and effectiveness of remedies.	Evaluate future needs for the 182-D Reservoir. Collect water level and contaminant concentration data near the 182-D Reservoir.	Future operation needs for the 182-D Reservoir may require maintenance of higher water levels. Data are needed to monitor leakage, effects on groundwater flow and contaminant transport, and potential effects to remedies.	The water system master plan (*Hanford Site Water System Master Plan* [HNF-5828]) evaluates the future needs and infrastructure solutions for the 182-D Reservoir and export water lines. Automated water level monitoring and quarterly sampling at Wells 199-D5-38, 199-D5-33, and 199-D5-34 will continue as part of standard remedy performance evaluation. The Export Water System, including the 182-D Reservoir, is discussed in Chapter 3 (Section 3.7.1).	Yes

a. Cr(VI) was identified in Well 199-D3-5 resulting in greater confidence regarding the location of contamination in 100-D. The edge of the plume was defined using quantile kriging methods described in the *Hanford Site Groundwater Monitoring for 2011* (DOE/RL-2011-118). The Cr(VI) results will guide the remedial action design but should not affect the selection of a remedy.

RUM – Ringold Formation upper mud

Table 2-2. Supplemental Investigations and Other Primary Investigations

Scope of Work Identified	Section with Discussion
Evaluating and developing approaches to obtain data that will demonstrate compliance with AWQC in the river for the ROD. In April 2008, a technical review panel was convened to evaluate groundwater interactions with the Columbia River (*Technical Evaluation of the Interaction of Groundwater with the Columbia River at the Department of Energy Hanford Site, 100-D Area* [SGW-39305]). The panel suggested that the current mixing/dilution conceptual model should be re-evaluated. In addition, data may be needed to show representativeness of contaminant concentrations for compliance. Therefore, evaluation will include determination of whether 1:1 dilution assumption for groundwater entering the river is valid, and may include evaluation of whether data from aquifer tube samples are representative. Data collected as part of the RI for site releases to the Columbia River may be useful in this evaluation.	Section 2.1.13, River Corridor Supplemental Investigation
Collecting data and developing River Corridor background values in soil for antimony, boron, molybdenum, and selenium. Site-specific background values for these constituents may be needed to determine final soil cleanup values where calculated risk-based concentrations and/or ecological protection concentrations are less than background. Interim remedial actions have used Washington State background values for antimony and selenium; interim soil RAGs for boron and molybdenum are above expected site-specific background values.	Section 2.1.13, River Corridor Supplemental Investigation
Re-evaluating soil cleanup level for Cr(VI) to support the ROD. The lowest soil RAG for Cr(VI) under the interim RODs is 2.0 mg/kg. However, the calculated MTCA ("Deriving Soil Concentrations for Groundwater Protection" [WAC 173-340-747(3)(a)]) soil RAG value may be below the current limits of analytical quantitation in environmental samples, depending on the soil-partitioning value and groundwater-to-river dilution attenuation factor used, and soil cleanup values may default to the limits of quantitation. Because there is uncertainty in analytical detection and quantitation of Cr(VI) near the limits of detection, it may be necessary to consider the realistic capabilities of analytical performance in determination of a soil cleanup value.	Section 2.1.13, River Corridor Supplemental Investigation
Determining a site-specific soil-partitioning value for antimony. This value is necessary for calculation of the MTCA ("Deriving Soil Concentrations for Groundwater Protection" [WAC 173-340-747(3)(a)] soil RAG values for antimony. Antimony is not a significant contaminant in the River Corridor, and determination will include review of scientific literature, which suggests antimony soil partitioning values in the range of 1.4 to 45 mL/g.	Section 2.1.13, River Corridor Supplemental Investigation
Re-evaluate soil cleanup levels for arsenic to support the ROD. The soil RAG for arsenic under the interim RODs is 20 mg/kg, based on the TPA (Ecology et al., 1989a) to use the MTCA ("Unrestricted Land Use Soil Cleanup Standards" [WAC 173-340-740(2)] Method A value (*Remedial Design Report/Remedial Action Work Plan for the 100 Area* [DOE/RL-96-17]). The MTCA ("Unrestricted Land Use Soil Cleanup Standards" [WAC-173-340-740(2)]) Method A value is also 20 mg/kg. The MTCA ("Unrestricted Land Use Soil Cleanup Standards" [WAC 173-340-740(3)] Method B and "Deriving Soil Concentrations for Groundwater Protection" [WAC 173-340-747(3)(a)]) soil values for arsenic are below the site arsenic background of 6.5 mg/kg. Selection of a soil cleanup level for arsenic in the River Corridor will be accomplished through development of RODs.	Section 2.1.13, River Corridor Supplemental Investigation

Table 2-2. Supplemental Investigations and Other Primary Investigations

Scope of Work Identified	Section with Discussion
Other Primary Investigations that Potentially Affect Feasibility Study Decisions for Waste Sites and Groundwater Contamination	
Columbia River RI Work Plan (DOE/RL-2008-11).	Section 2.1.7, Surface Water and Sediment Investigation
River Corridor Baseline Risk Assessment (DOE/RL-2007-21).	Chapter 6, Human Health Risk Assessment Chapter 7, Ecological Risk Assessment Section 3.9, Demography and Land Use
Annual Groundwater Monitoring.	Section 2.1.10, Groundwater Investigations Section 4.4, Groundwater Contamination Section 4.5, Distribution of Contaminants
Ongoing Aquifer Tube Sampling.	Section 2.1.10, Groundwater Investigations Section 4.4, Groundwater Contamination Section 4.5, Distribution of Contaminants
Columbia River Component Risk Assessment (DOE/RL-2010-117, Vol. 1[Eco Risk] and Vol. II [HHRA]).	Section 7.5.2, Results and Conclusions of the CRC Section 3.9, Demography and Land Use
Data Summary Report for Hanford Site Coal Ash Characterization (WCH-506).	N/A

Sources: DOE/RL-96-17, *Remedial Design Report/Remedial Action Work Plan for the 100 Area.*

DOE/RL-2007-21, *River Corridor Baseline Risk Assessment, Volume I: Ecological Risk Assessment.*

DOE/RL-2008-11, *Remedial Investigation Work Plan for Hanford Site Releases to the Columbia River.*

DOE/RL-2010-117, *Columbia River Component Risk Assessment, Volume I: Screening-Level Ecological Risk Assessment.*

Ecology et al., 1989a, *Hanford Federal Facility Agreement and Consent Order.*

SGW-39305, *Technical Evaluation of the Interaction of Groundwater with the Columbia River at the Department of Energy Hanford Site, 100-D Area.*

WAC 173-340-747, "Model Toxics Control Act—Cleanup," "Deriving Soil Concentrations for Groundwater Protection."

WAC 173-340-740, "Model Toxics Control Act—Cleanup," "Unrestricted Land Use Soil Cleanup Standards."

WCH-506, *Data Summary Report for Hanford Site Coal Ash Characterization.*

Table 2-3. Summary of 100-D/H RI Field Program

Type	100-D	100-H	Total
New boreholes that have been decommissioned	3	2	5
New boreholes converted to temporary wells	2	3	5
New test pits	3	2	5
New permanent wells (screened in the unconfined aquifer)	7	5	12
New permanent wells (screened in the first water-bearing unit in the RUM)	2	3	5
New aquifer tubes per location	4	2	6
Spatial and Temporal Uncertainty Monitoring Wells			52

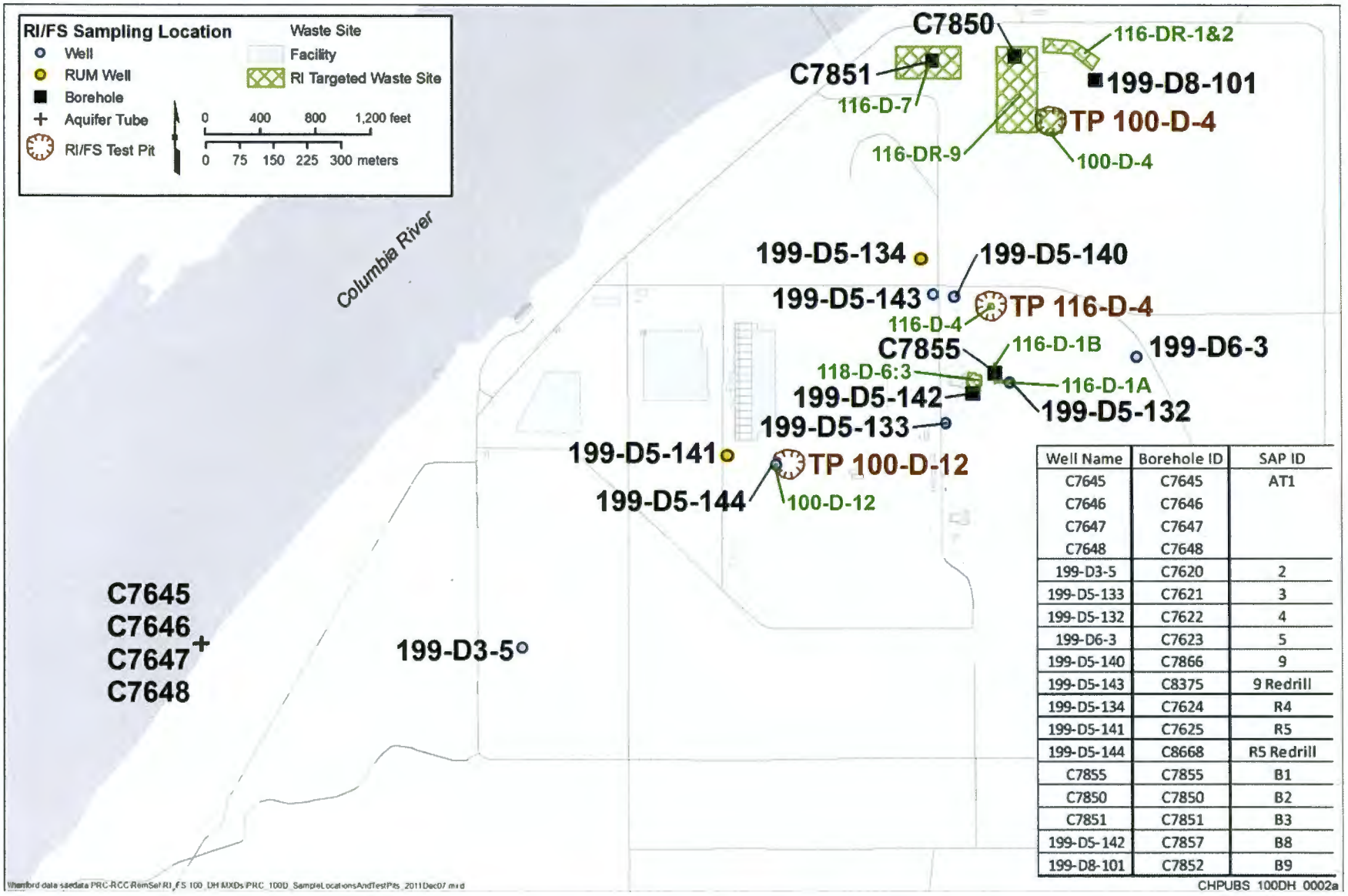
1

Table 2-4. Number of Field Samples Collected for 100-D/H RI

Source	Soil Samples*	Groundwater Samples*
New boreholes (decommissioned)	70	5
New boreholes (converted to temporary monitoring wells in unconfined aquifer)	57	5
New test pits	27	N/A
New permanent wells (screened in the unconfined aquifer)	154	41
New permanent wells (screened in the first water-bearing unit in the RUM)	63	30
New aquifer tubes	N/A	18
Spatial and Temporal Uncertainty Monitoring Wells	N/A	156

* The number of samples taken reflects the number of intervals sampled (*Sampling and Analysis Plan for the 100-DR-1, 100-DR-2, 100-HR-1, 100-HR-2, and 100-HR-3 Operable Units Remedial Investigation/Feasibility Study* [DOE/RL-2009-40]). The samples from each interval were then split amongst several laboratories for different analyses.

2



Well Name	Borehole ID	SAP ID
C7645 C7646 C7647 C7648	C7645 C7646 C7647 C7648	AT1
199-D3-5	C7620	2
199-D5-133	C7621	3
199-D5-132	C7622	4
199-D6-3	C7623	5
199-D5-140	C7866	9
199-D5-143	C8375	9 Redrill
199-D5-134	C7624	R4
199-D5-141	C7625	R5
199-D5-144	C8668	R5 Redrill
C7855	C7855	B1
C7850	C7850	B2
C7851	C7851	B3
199-D5-142	C7857	B8
199-D8-101	C7852	B9

Figure 2-1. Map Showing 100-D RI Sampling Locations

Well Name	Borehole ID	SAP ID
199-H3-6	C7626	6
199-H3-7	C7627	7
C7649 C7650	C7649 C7650	AT8
199-H6-3	C7628	10
199-H6-4	C7629	11
199-H1-7	C7630	12
199-H3-9	C7639	R1
199-H3-10	C7640	R2
199-H2-1	C7631	R3
C7862	C7862	B4
C7864	C7864	B5
199-H4-83	C7861	B6
199-H3-11	C7863	B7
199-H4-84	C7860	B10

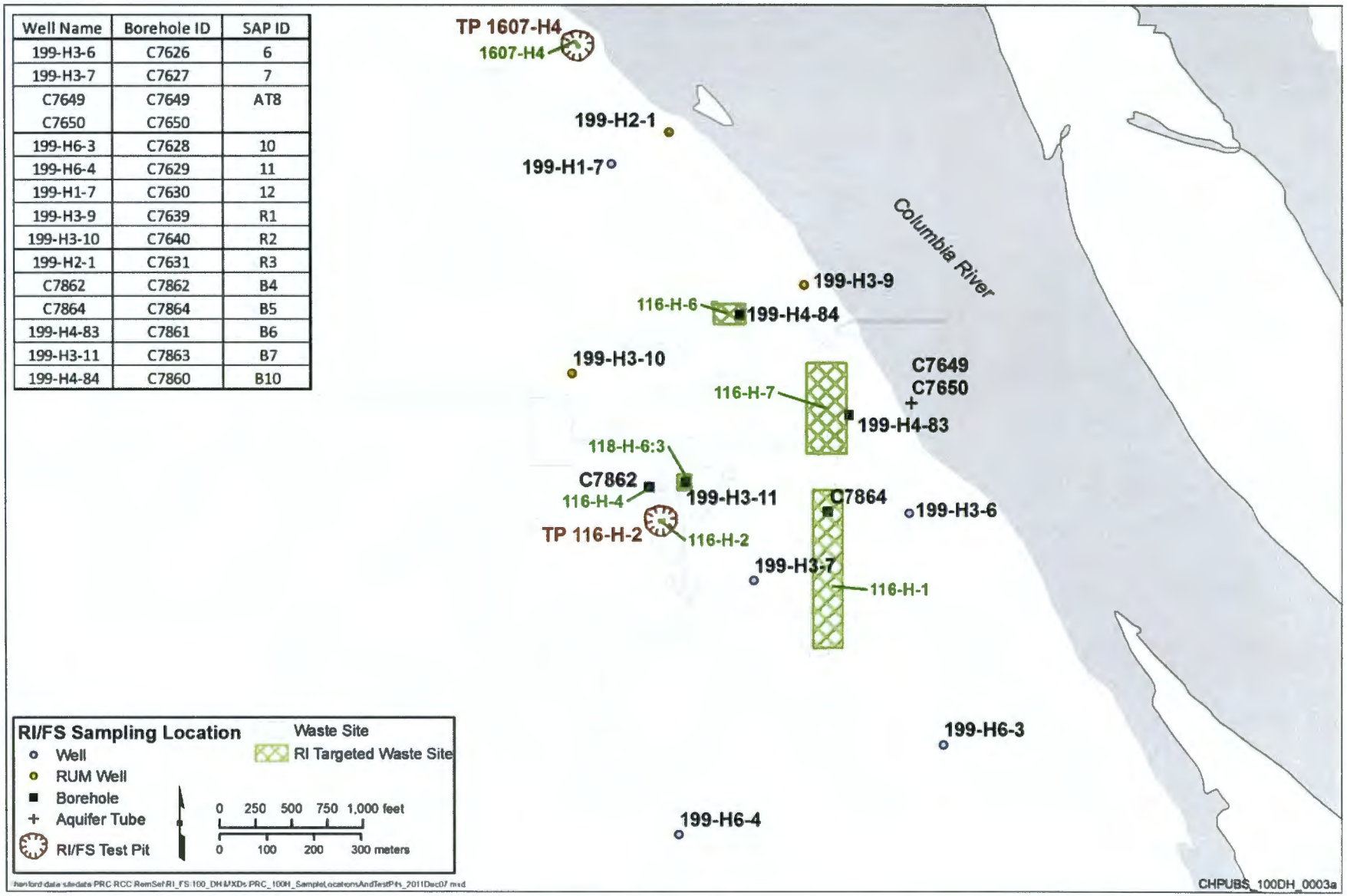


Figure 2-2. Map Showing 100-H RI Sampling Locations

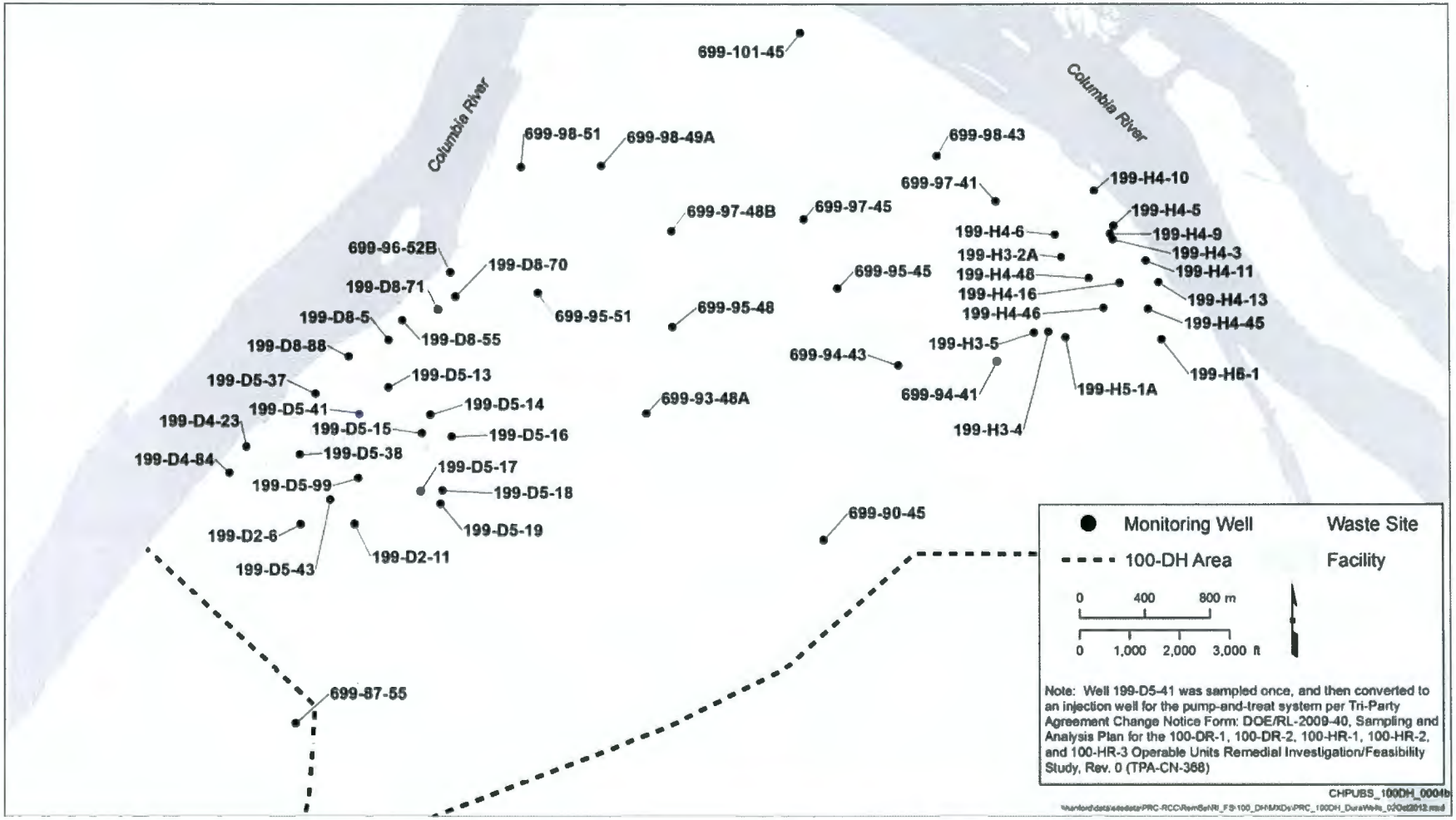


Figure 2-3. 100-D/H RI Spatial/Temporal Groundwater Monitoring Well Locations

The 100-D/H SAP (DOE/RL-2009-40) provides additional details, such as specific sample intervals, sampling and analytical methodology, and technical memorandums that summarize each field activity. Appendix C includes specific information for each borehole and sampling interval, including details of the field effort for soil and groundwater sampling, respectively. Soil samples were typically collected at 1.5 m (5 ft) depth intervals during drilling. Actual soil and groundwater sample depths may have some minor variability from the depths planned in the 100-D/H SAP (DOE/RL-2009-40) because of the depth where the water table was encountered and the formation conditions encountered. Some variability in sample location is expected and allowed under the 100-D/H SAP (DOE/RL-2009-40).

The following sections present details of investigations conducted under the 100-D/H Work Plan (DOE/RL-2008-46-ADD1), as well as investigation activities conducted under other scopes of work that may affect the FS decisions, including the Columbia River RI Work Plan (DOE/RL-2008-11) and the RCBRA (DOE/RL-2007-21).

Variations of 100-D/H Work Plan (DOE/RL-2008-46-ADD1) typically resulted in additional data collection and were as follows:

- Well Drilling
  - Well 199-D5-141 (C7625; Well R5) was incorrectly located. A replacement well, Well 199-D5-144 (C8668, Well R5 redrill), was drilled in the originally planned location. Samples were collected and analyzed during drilling from both of these wells. Drilling depth and sampling for Well 199-D5-144 (C8668, Well R5 redrill) was conducted under *Tri-Party Agreement Change Notice Form: DOE/RL-2009-40 Sampling and Analysis Plan for the 100-DR-1, 100-DR-2, 100-HR-1, 100-HR-2, and 100-HR-3 Operable Units Remedial Investigation Feasibility Study, Rev. 0* (TPA-CN-460).
  - Well 199-D5-140 (C7866, Well 9) was incorrectly located and not placed beneath the 100-D-56 sodium dichromate pipeline. A replacement well, Well 199-D5-143 (C8375; Well 9 redrill), was drilled in the originally planned location. Samples were collected and analyzed during drilling from both of these wells.
- Aquifer Tubes
  - One additional aquifer tube was added to the cluster of three planned at 100-D and represents a second deep zone.
  - The deep zone aquifer tubes at the 100-H cluster location were not installed as a result of encountering the Ringold Formation upper mud (RUM) at a shallow depth. There was insufficient water at the deep locations for aquifer tube installation.

Other approved deviations include the following:

- **Spatial and Temporal Sampling.** Well 199-D5-41 was sampled once and then converted to an injection well for the pump-and-treat system per *Tri-Party Agreement Change Notice Form: DOE/RL-2009-40, Sampling and Analysis Plan for the 100-DR-1, 100-DR-2, 100-HR-1, 100-HR-2, and 100-HR-3 Operable Units Remedial Investigation/Feasibility Study, Rev. 0* (TPA-CN-368).
- **Temporary Well Installation in 5 of 10 Boreholes.** Five boreholes were drilled into waste sites and completed as 10 cm (4 in.) temporary wells: 116-DR-1&2 (Trench), 118-D-6:3 (FSB), 116-H-6 (Solar Evaporation Basin), 116-H-7 (Retention Basin), and 118-H-6:3 (FSB). The well names and

associated borehole IDs are as follows: 199-D5-142 (C7857), 199-D8-101 (C7852), 199-H4-83 (C7861), 199-H4-84 (C7860), and 199-H3-11 (C7863).

2.1.1 RI Datasets Used in RI/FS

Historical and RI data are evaluated in this report. Appendix D provides additional details on the dataset along with data. The following is a list of the available data that were compiled for the RI/FS dataset:

- Data collected as part of ongoing site sampling programs or before initiation of the current RI/FS field investigation activities. Data sources include the 70 RPO wells, existing monitoring wells, decommissioned well geologic data, aquifer test and rebound study, the horn study, and the northern and southern plume investigations, among others:
  - Waste site remedial action soil analytical data (Cleanup Verification Package [CVP] and Remaining Site Verification Packages [RSVP] data) for the 17 waste sites investigated in the RI. This dataset was used in the CSM evaluation of the nature and extent of soil contamination (Chapter 4, Section 4.9), groundwater protection (Chapter 5), HHRA (Chapter 6), and ERA (Chapter 7). Other CVP and RSVP data are presented in Appendices D and E.
  - Field investigation soil analytical data (LFI data). This dataset was used in the evaluation of the nature and extent of soil contamination (Chapter 4), groundwater protection (Chapter 5), HHRA (Chapter 6), and ERA (Chapter 7).
  - Groundwater analytical data (January 1, 2006 to December 31, 2011). This dataset was used in evaluating the nature and extent of groundwater contamination (Chapter 4) and provides the basis for the initial plumes for groundwater modeling (Appendix F). Historical groundwater data was included to add to the understanding of the nature and extent of contamination, especially as it relates to the sources of contamination.
  - Well and borehole drilling and well construction information. This dataset was used in the development of hydrogeologic cross sections, aquifer isopach map, and RUM surface contour map (Chapter 3 and Appendix M) and groundwater model development (Chapter 5 and Appendix F).
  - Fate-and-transport parameters (for example, geochemical parameters, hydrogeologic parameters, and soil physical properties). This dataset was used in the development of the groundwater model and fate-and-transport evaluations (Chapter 5 and Appendix F).
  - Geologic information. This dataset was used in the development of the hydrogeologic cross sections, aquifer isopach map, and RUM surface contour map (Chapter 3 and Appendix M) and groundwater model development (Chapter 5 and Appendix F).
  - Groundwater levels and river stage. This dataset was used in the development of groundwater flow maps and groundwater model developments (Chapter 5 and Appendix F).
- CVP data collected for the 144 interim closed-out waste sites per the interim action ROD as part of the ongoing interim waste site remediation are used to develop and refine the CSM, are qualitatively discussed in Chapter 4 (Section 4.3.2), and are used in surface and groundwater protection, the human health risk assessment, and the ERA in Chapters 5, 6, and 7.
- Data collected during the RI/FS field investigation activities.

- Soil analytical data. Depth specific soil samples collected during RI borehole and well installation are used to evaluate contaminant distribution in the vadose zone and to develop/refine the CSM (Chapter 4), groundwater protection (Chapter 5), and HHRA (Chapter 6).
- Groundwater analytical data:
  - Spatial and temporal groundwater monitoring data. This dataset was used in the HHRA (Chapter 6) and understanding of spatial and temporal distribution of groundwater contaminants (Chapter 4).
  - Groundwater samples collected from RI boreholes and monitoring wells. Depth discrete groundwater samples were used to establish the vertical distribution of contaminants in groundwater (Chapter 4) and to develop/refine the CSM.
- Soil physical properties (grain size, moisture content, and porosity). These data were used in the groundwater model development (Appendix F), fate and transport modeling, and preliminary remediation goals (PRGs) development.
- Hydraulic conductivity. These data were used in the groundwater model development (Appendix F).
- Geophysical logging. The geophysical logs from RI boreholes are presented in Appendix M. These data help with the understanding of the CSM and transport of contaminants through the vadose zone.
- Distribution coefficient data for metals. This dataset is used in the evaluation of fate-and-transport of metals (Chapter 5).

Analytical data used in the RI/FS (Appendix D) were collected and analyzed in a fixed laboratory using approved methods with specific QA/QC requirements. Detection limits, precisions, accuracy, and completeness were assessed to determine whether the chemical and radiochemical data obtained were the right type, quality, and quantity to support regulatory decision making.

2.1.2 Historical Information Review

Historical information for 100-D/H was researched and considered during the 100-D/H Work Plan (DOE/RL-2008-46-ADD1) development and in the preparation of this report. Section 1.2.3 and Appendix B summarize those reports containing relevant or significant information. In addition, a summary of site history and 100-D/H operational and process history is presented in Section 1.2.2.

2.1.3 Surface Features

Surface feature mapping, such as high-resolution topography, was conducted using Light Detection and Ranging (LIDAR) technology for 100-D/H in 2008. LIDAR is an optical remote sensing technology that measures properties of scattered light to find range and/or other information of a distant target. The current accuracy of the LIDAR mapping is estimated at 0.11 m (4.3 in.). LIDAR data were used to create a topographic map of 100-D/H for defining surface relief/elevation differences. Surface topography (Section 3.1) establishes part of the framework needed to evaluate contaminant fate and transport. LIDAR was also used in conducting the non-operational area evaluation, discussed in Appendix K.

2.1.4 Contaminant Source Investigations

The OSE is a systematic approach to review land parcels and identify potential waste sites within the River Corridor that are not currently listed in existing CERCLA decision documents (RODs). The OSE

is discussed as part of the nonoperational area evaluation in Chapter 1 and is included in Data Gap 4 (Table 2-1).

The OSE process in 100-D was completed in February 2009 and identified 30 new waste sites (*100-D Area Orphan Sites Evaluation Report* [OSR-2006-0001]). In 100-H, the OSE was also completed in February 2009 (*100-H Area Orphan Sites Evaluation Report* [OSR-2008-0002]) and identified 15 new waste sites. The OSE for the remainder of 100-D/H, primarily the horn area, was completed in 2011 (*100-F/IU-2/IU-6 Area – Segment 4 Orphan Sites Evaluation Report* [OSR-2011-0001]) and identified six new waste sites. The waste site numbers for these orphan sites are 600-380, 600-381, 600-382, 600-383, 600-384, and 600-385. Evaluation of these new waste sites will be conducted to determine their status (that is, "no action," "not accepted," "rejected," or "accepted") and remediation will be carried out as appropriate. In addition, discovery site identification continues during ongoing remedial actions.

2.1.5 Meteorological Investigations

The Hanford Meteorological Station (HMS) is operated by Mission Support Alliance for DOE (http://www.hanford.gov/hms/). HMS provides a range of Hanford Site weather forecast products and real-time meteorological data, and currently maintains an extensive historical database of meteorological and climatological data. Meteorological measurements have been made at HMS since late 1944. Information specific to precipitation and wind speed have the potential to affect remedial actions, as discussed in Section 3.2. No additional meteorological data were collected as part of this RI/FS.

2.1.6 Air Investigations

Hanford Site contractors monitor radionuclide airborne emissions from site facilities through several programs. The Near-Facility Environmental Monitoring Program measures concentrations of radionuclides in the ambient air on the Hanford Site in or near facilities and operations. The Hanford Site Environmental Surveillance Program measures the ambient air at Sitewide locations away from facilities, offsite around the Site perimeter, and in nearby and distant communities. In addition, emissions from stacks, vents, or other types of point sources are monitored individually by analyzing samples extracted from the outflow at each point of release. The data collected by each program are used to assess the effectiveness of emission treatment and control systems and pollution management practices, and to determine compliance with state and federal regulatory requirements. These regulations include a radiological standard, which requires that Hanford Site emissions will be controlled such that no member of the public in any area of unrestricted access receives greater than 10 mrem/yr total effective dose equivalent. In some cases, remedial activities are provided with project-specific point source and/or ambient air sampling to assemble project-specific data. DOE provides information to the Washington State and EPA clean air offices describing the emissions and resultant maximum public dose from ongoing CERCLA activities. This information addresses contributions both from point sources and from all fugitive or diffuse sources of emissions of radionuclides.

Nonradioactive air pollutants are emitted from a variety of sources at the Hanford Site. These emissions are monitored at the source when activities are known to generate actual or potential pollutants of concern. DOE provides information to Washington State and EPA clean air offices describing the emissions. The following text summarizes the most recent information regarding Hanford Site air monitoring activities (2009 Sitewide Environmental Report [PNNL-19455]).

2.1.6.1 Air Monitoring Near Facilities and Operations

Ambient air is monitored at locations on the Hanford Site near facilities and operations. Samplers are located primarily within approximately 500 m (1,640 ft) of projects or facilities having a known potential for, or history of, environmental radiation releases. This ambient monitoring is termed near-facility

environmental monitoring. Monitoring locations are associated largely with major nuclear facilities and waste storage, disposal, or cleanup activities. Occasional adjustments are made in the number or location of the monitoring stations as changes in the sources of emissions may occur.

2.1.6.2 Air Monitoring at Hanford Sitewide and Offsite Locations

As part of the Hanford Site Environmental Surveillance Program, near facility ambient air samples are collected at four continuously operating locations associated with 100-D and four locations associated with 100-H (Figure 2-4). In addition, 11 additional ambient air monitors are operated at locations representing the Hanford Site perimeter, along with seven monitoring stations in nearby communities of Basin City, Benton City, Kennewick, Mattawa, Othello, Pasco, and Richland, Washington, and one in a distant community (Yakima, Washington).

Samples are collected from known or expected air transport pathways, which are generally downwind of potential or actual airborne releases and downgradient of liquid discharges. Airborne particle samples are collected at each station biweekly and monitored for gross alpha and gross beta concentrations. Biweekly samples are combined into quarterly composite samples and analyzed for gamma emitting radionuclides. Samples of atmospheric water vapor are collected every 4 weeks and analyzed for tritium at approximately 20 locations. All air sample results showed very low radiological concentrations in 2010, with resultant exposure to any public individual remaining well below the dose standard of 10 mrem/yr total effective dose equivalent. A detailed discussion of the air sampling and results are presented in the 2009 Sitewide Environmental Report (PNNL-19455) in Section 8.2. Table 8.2.3 of the same report provides sample locations and a list of analyses collected at each location.

Ambient air sampling is the primary method used in monitoring fugitive emissions. Hanford Site contractors also monitor for other effects from airborne emissions or other releases from site facilities. This is done through sampling of various environmental media besides the air, as part of the Surface Environmental Surveillance Program. Routine monitoring includes sampling of surface contamination, external radiation doses, soil, vegetation, and animals. All estimated and measured environmental doses from Hanford Site activities remain much lower than EPA and DOE standards. While not a required action for the CERCLA remedial action, the Washington State Department of Health also conducts independent sampling and analysis of various media, including ambient air, soil, and biota, both on and off the Hanford Site. This independent sampling and analysis routinely confirms little or no environmental impacts outside of the Hanford Site's most closely controlled work areas. A discussion of the nature and extent of air contaminants is presented in Section 4.8. Historic fugitive dust emissions or stack emissions have been evaluated as a potential fate-and-transport pathway for contaminants in non-operational areas. The nonoperational area evaluation discussed in Chapter 1 and presented in Appendix K, summarizes different surveillance programs, including the OSE, and provides different statistical analyses to identify the potential for effects in nonoperational areas.

No additional air monitoring, with the exception of in-process monitoring at the immediate worksite during select borehole, well, and test pit activities, has been conducted as part of this RI/FS.

2.1.7 Surface Water and Sediment Investigations

An investigation of pore water, surface water, and sediment was conducted to identify the nature and extent of contaminants entering the Columbia River, specifically by groundwater upwelling. The effort was performed according to the Columbia River RI Work Plan (DOE/RL-2008-11), and activities planned before the RI/FS as outlined in the 100-D/H Work Plan [DOE/RL-2008-46-ADD1]). As such, Data Gaps 6 and 9 (Table 2-1) denote the importance of addressing groundwater discharge and surface water/groundwater mixing to support decision making in a ROD. The following sections provide details on these investigations.

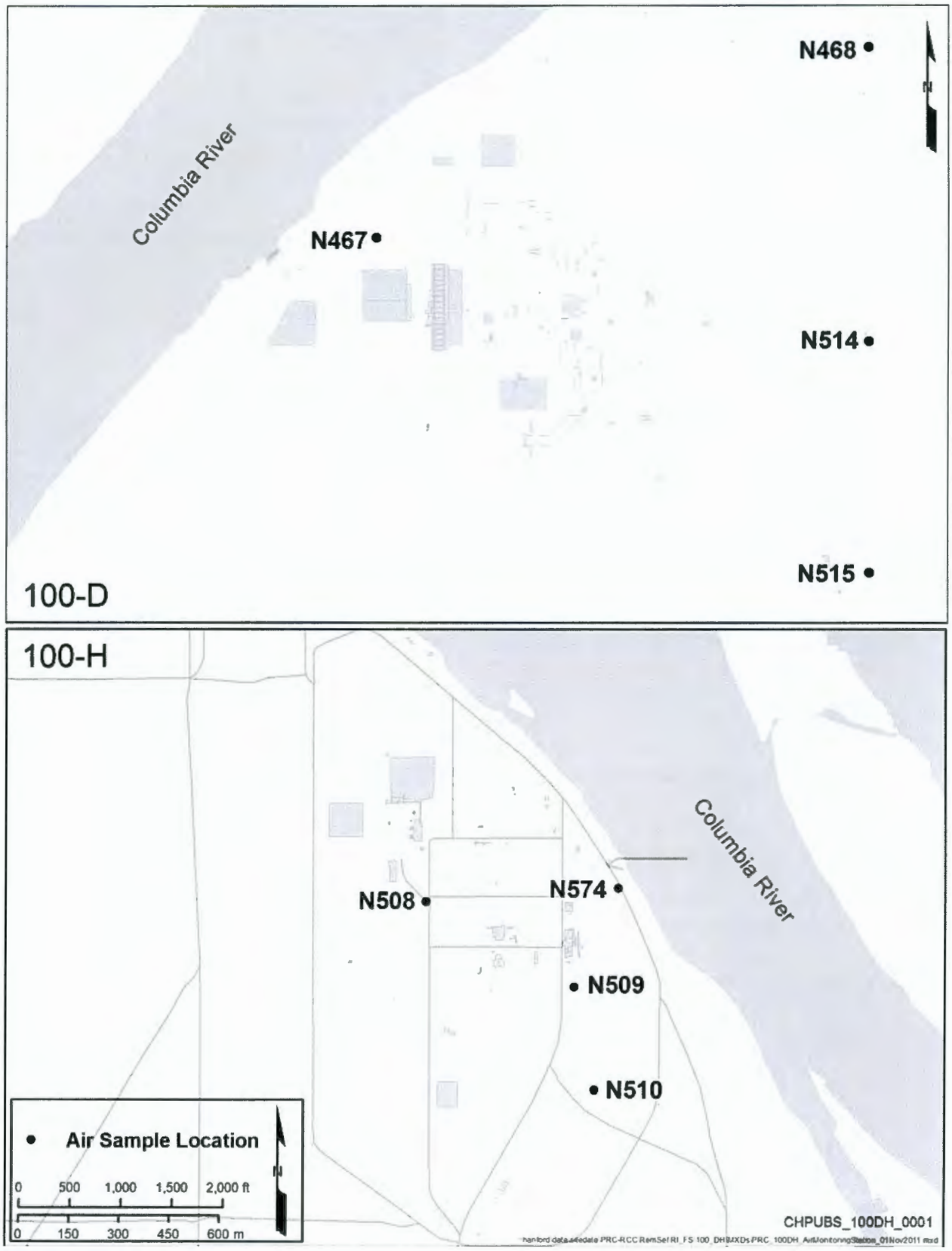


Figure 2-4. Collection of Ambient Air Samples Associated with 100-D and 100-H

Investigation data collected during the RI to evaluate groundwater discharge to surface water has been integrated with data sets collected as part of the Columbia River RI Work Plan (DOE/RL-2008-11) and presented in the CRC (DOE/RL-2010-117). The data were collected to provide an evaluation, presented in Appendix L, of the conceptual model of potential fate, transport, and exposure pathways in riparian and nearshore areas in 100-D/H.

2.1.7.1 Groundwater Upwelling and Discharge into the Columbia River (Pore Water, Surface Water, and Sediment Sampling)

Groundwater beneath the Hanford Site discharges to the Columbia River through seeps and upwelling to the riverbed. This flow path provides a means to transport Hanford Site contaminants that may have leached into groundwater to reach the Columbia River.

The availability of historical data to understand preferential groundwater flow paths, contaminant migration restrictions, and groundwater and river water mixing in the nearshore area is limited. Groundwater discharges into the Columbia River along the Hanford Reach. Mostly laminar flow results in a complex flow regime (*Technical Evaluation of the Interaction of Groundwater with the Columbia River at the Department of Energy Hanford Site, 100-D Area* [SGW-39305]) between river water and groundwater upwelling at the bottom of the river, and between river water and groundwater seeps at shoreline locations. Data were collected near 100-D/H in 2009 and 2010 to address the uncertainty related to the level of contaminants entering the Columbia River, including the contaminant transport mechanisms. Pore water sampling in the Columbia River was conducted during three phases, as outlined in the Columbia River RI Work Plan (DOE/RL-2008-11).

The first phase of the Columbia River RI Work Plan (DOE/RL-2008-11) pore water sampling was a technology demonstration to verify that proposed equipment was usable in the variable conditions found in the Hanford Reach section of the Columbia River. The second phase consisted of two subphases. Phase IIa focused on identifying riverbed areas where groundwater was entering the Columbia River, as evidenced by variations in conductivity and temperature measurements. In Phase IIb, a subset of the Phase IIa locations that showed evidence for groundwater entering the river were revisited in order to collect and analyze pore water samples for Cr(VI) as an indicator contaminant. Phase III sampling identified a subset of the previous sample locations for sampling and analysis of pore water, surface water (defined as water 0.3 m [1 ft] above the riverbed), and collocated sediment for a wide range of potential contaminants.

Pore water data (conductivity and temperature) were collected in Phase IIa using a multi-sensor water sampling probe capable of being inserted approximately 30 cm (12 in.) into the riverbed at five cross-river transects in 100-D and six in 100-H. Each transect had five sample locations. Additionally, 10 locations surrounding the established transects were sampled.

Pore water sampling for Phase IIb was conducted at a subset of the Phase IIa locations that indicated groundwater upwelling based on conductivity and temperature variances between the river and pore water, and were deemed most likely to show contamination. These sample locations were approved by the Tri-Parties and are shown on Figures 2-5 and 2-6 (Columbia River RI Report [WCH-380, Rev. 1]).

Figure 2-5. Remedial Investigation for Hanford Site Releases to the Columbia River—Phase IIb Indicator Contaminant Sample Locations at 100-D

Figure 2-6. Remedial Investigation for Hanford Site Releases to the Columbia River—Phase IIb Indicator Contaminant Sample Locations at 100-H

Pore water samples for Phase III (Figures 2-7 and 2-8) were collected from established upwelling locations, with the focus on sites where the indicator contaminant (Cr[VI]) was detected in the Phase IIb pore water samples. For Phase III sampling, the Tri-Parties selected six sample locations each near 100-D and 100-H for collection of pore water, surface water, and sediment. The Tri-Parties also chose one additional pore water sample location in 100-D, and two additional pore water sample locations for Cr(VI) only in 100-H. Phase III characterization samples were analyzed for a range of radiological and nonradiological analytes as shown in Table 2-5.

Pore water and sediment samples were successfully obtained from these locations and analyzed for a range of radiological and nonradiological analytes (listed in Table 2-5). Because of volume limitations, not all media and/or analyses could be collected or conducted. Chapter 4 presents the results from the sampling efforts. Table 2-6 provides information on the number of pore water samples collected during each sampling phase and the collection period.

2.1.7.2 Surface Water Sampling

During Phase III, the influence of contaminants on the water immediately above groundwater upwelling locations was determined by taking surface water samples. River water and pore water samples were collected concurrently approximately 0.3 m (12 in.) above the riverbed. Table 2-6 provides information on the number of surface water samples and the collection period at 100-D and 100-H.

2.1.7.3 Sediment Sampling

Sediment samples collected during Phase III of the study were from the locations shown on Figures 2-7 and 2-8. Samples were analyzed for a range of radiological and nonradiological analytes, as listed in Table 2-5. Sediment samples were obtained as close to the pore water sample locations as reasonably possible, with a preference given to locations with sediment deposits. Sample volume was limited in some locations because of the dominance of cobbles on the riverbed. Sediment samples could be collected at only five of the six specified sample locations at each of 100-D and 100-H. In locations where sediment sample volume was limited, not all analyses could be performed. Table 2-6 presents information on sediment samples and the collection period.

2.1.7.4 Additional Surface Water, Sediment, and Island Soil Sampling

In addition to the sampling described in the preceding sections, supplemental samples of surface water, sediment, and island soil samples were collected for identifying the nature and extent of potential releases of contaminants associated with operations at the Hanford Site during the RI at locations described in *Field Summary Report for Remedial Investigation of Hanford Site Releases to the Columbia River, Hanford Site, Washington: Collection of Surface Water, River Sediments, and Island Soil* (WCH-352) and *Data Summary Report for the Remedial Investigation of Hanford Site Releases to the Columbia River, Hanford Site, Washington: January 2011* (WCH-398), hereinafter called Hanford Site Releases Data Summary. Because of high Columbia River conditions that restricted access, no soil samples were collected from D-Island. Figures 5-10a through 5-11c in Hanford Site Releases Data Summary (WCH-398) show these sample locations near 100-D and 100-H. Table 2-7 provides a summary of the number of additional samples collected.

Figure 2-7. Remedial Investigation for Hanford Site Releases to the Columbia River—Phase III Characterization Sample Locations for 100-D

Figure 2-8. Remedial Investigation for Hanford Site Releases to the Columbia River—Phase III Characterization Sample Locations for 100-H

Table 2-5. Summary of Analyses Requested for Surface Water, Sediment, and Pore Water Sampling during Columbia River RI

Analytical Parameter	EPA Method	Surface Water	Sediment	Pore Water
Metals (filtered)	6010/7470	X	--	X
Metals (unfiltered)	6010/7470	X	--	X
Metals (solid)	6010/7471	--	X	--
Cr(VI)	7196A	X	X	X
VOCs	8260B	X	X	--
SVOCs	8270C	X	X	--
Pesticides	8081	X	X	--
PCBs	8082	X	X	--
Petroleum hydrocarbons	8115	--	X	--
Thorium isotopes	Isotopic Th	X	X	--
Plutonium isotopes	Isotopic Pu	X	X	--
Tritium	LSC	X	--	X
Total beta radiostrontium	GFPC	X	X	X
Uranium isotopes	Isotopic U	X	X	--
Carbon-14	LSC	X	X	--
Gamma-emitting radionuclides	GEA	X	X	--
Technetium-99	LSC/GFPC	X	X	--
TOC / DOC	415.1 & 415.1 M	X	X	X
Grain Size	ASTM D422-63	--	X	--
AVS/SEM[a]	N/A	--	X	--
Nitrate and anions	300.0 & 353.2	X	--	X
Hardness	130.1	X	--	--
Alkalinity	310.1	X	--	--
Field parameters[b]	Field instruments	X	--	X

Source: ASTM D422-63(2007), *Standard Test Method for Particle-Size Analysis of Soils.*

Note: Analyses will be sample specific; not all samples were analyzed by all methods in this table.

a. Sediment samples were extracted and analyzed for acid volatile sulfides (AVS) (EPA-821-R-91-100, *Draft Analytical Method for Determination of Acid Volatile Sulfide in Sediment, Determination of Acid Volatile Sulfide and Selected Simultaneously Extractable Metals in Sediment*). The simultaneously extracted metal (SEM) extracts were analyzed for all other metals by ICP-MS in accordance with PNNL standard operating procedures.

b. Field parameters for surface water samples were measured in the field and consisted of temperature, specific conductivity, dissolved oxygen, and pH. Field parameters for pore water consisted of temperature and conductivity.

Table 2-6. Summary of 100-D and 100-H Sample Collection during the Columbia River Remedial Investigation

Sample Phase	Sample Dates	Parameters of Interest	Number of Samples/Stations
100-D: Phase IIa	January/February/March 2009	Conductivity Mapping	77
100-D:Phase IIb	September/October 2009	Indicator Contaminant Screening	30
100-D:Phase III	February 2010	Groundwater Upwelling Characterization	6 (1)*
100-H:Phase IIa	January/February/March 2009	Conductivity Mapping	91
100-H:Phase IIb	October 2009	Indicator Contaminant Screening	30
100-H:Phase III	January/February 2010	Groundwater Upwelling Characterization	6 (2)*

* The number in parentheses represents the count of sample sites sampled only for Cr(VI) in pore water during Phase III sampling.

2.1.8 Geological Investigations

Geological investigations were conducted to address Data Gaps 2, 3, 5, 7, 10, and 12 listed in Table 2-1. Geological characterization and physical and hydraulic property data needs were identified to support development/refinement of the CSM and performance of analytical and numerical modeling within 100-D/H. In addition, geologic data were needed to gain a better understanding of the hydrogeologic conditions, aquifer interactions, and contaminant mobility through the vadose zone and within the unconfined and semi-confined aquifers. To address these data needs, the following wells were installed (Appendix C provides a crosswalk):

Table 2-7. Summary of Additional Samples Collected in the Vicinity of 100-D/H

Media Collected	Number of Samples	
	100-D	100-H
Island Soil	9	10
Surface Water	2	1
Sediment*	13	22

* Includes shoreline, shallow, and core samples.

- Twelve permanent wells were installed in the unconfined aquifer: Wells 199-D3-5 (C7620, Well 2), 199-D5-133 (C7621, Well 3), 199-D5-132 (C7622, Well 4), 199-D6-3 (C7623, Well 5), 199-D5-140 (C7866, Well 9), 199-D5-143 (C8375, Well 9 redrill), 199-D5-144 (C8668, Well R5 redrill), 199-H3-6 (C7626, Well 6), 199-H3-7 (C7627, Well 7), 199-H6-3 (C7628, Well 10), 199-H6-4 (C7629, Well 11), and 199-H1-7 (C7630, Well 12).
- Five wells were installed in the first water-bearing unit in the RUM: Wells 199-D5-134 (C7624, Well R4), 199-D5-141 (C7625, R5), 199-H2-1 (C7631, Well R3), 199-H3-9 (C7639, Well R1), and 199-H3-10 (C7640, Well R2).
- Five boreholes were drilled into waste sites and subsequently decommissioned: 116-DR-9 (Retention Basin), 116-D-7 (Retention Basin), 116-D-1B (Trench), 116-H-4 (Pluto Crib), and 116-H-1 (Trench).

- Five boreholes were drilled into waste sites and completed as 10 cm (4 in.) temporary polyvinyl chloride (PVC) wells: 116-DR-1&2 (Trench), 118-D-6:3 (FSB), 116-H-6 (Solar Evaporation Basin), 116-H-7 (Retention Basin), and 118-H-6:3 (FSB).

- Seventy RPO wells were installed per *Sampling and Analysis Plan for Installation of 100-HR-3 Groundwater Operable Unit Remedial Process Optimization Wells* (DOE/RL-2009-09). Geologic and characterization data from those wells were incorporated into the evaluation of site conditions. Well construction details, geologic information, and other data for the RPO wells are included in *Borehole Summary Report for the Installation of 70 Remedial Process Optimization, Pump-and-Treat Expansion Wells, For the 100-HR-3 Operable Unit* (SGW-48612), hereinafter called the Borehole Summary Report for RPO Wells.

Five boreholes were converted to temporary monitoring wells during installation activities to obtain a representative water sample from the unconfined aquifer. These wells had 3 m (10 ft) PVC screens installed to straddle the water table. The use of temporary monitoring wells is consistent with the 100-D/H SAP (DOE/RL-2009-40). Table C-4 (Appendix C) includes pertinent well location information while Table C-4 identifies samples collected in accordance with the 100-D/H SAP (DOE/RL-2009-40) requirements. Deviations in the number and depth of a particular sample are generally due to insufficient sample recovery. Other conditions that may cause a minor deviation include differences in planned depth for the water table, and differences in the expected geologic material changes. These deviations are identified in Table C-4 (Appendix C).

Well 199-D5-140 (C7866, Well 9) was intended to characterize the location of the 100-D-56 sodium dichromate pipeline at a 90° bend where a hole in the pipe was observed. However, the well was drilled at the wrong location because of coordinate issues. No change notice was needed for drilling, sampling, and installing the "replacement" well in the correct location because it was done per the SAP requirements. The new well number 199-D5-143 (C8375) was located as specified in the 100-D/H SAP (DOE/RL-2009-40).

Well 199-D5-141 (C7625, R5) was drilled at a location west of the planned location to avoid overhead power lines and the 100-D-100 remediation activities scheduled for summer 2011. Sampling was conducted in Well 199-D5-141 (C7625, Well R5) as planned in the 100-D/H SAP (DOE/RL-2009-40), which filled Data Gap 7. A shallower borehole was drilled at the original planned location. This 10 cm (4 in.) temporary PVC well was drilled approximately 3 m (10 ft) into the RUM. The new well number is 199-D5-144 (borehole C8668). Sampling from Well 199-D5-144 (C8668, Well R5 redrill) was conducted per *Tri-Party Agreement Change Notice Form: DOE/RL-2009-40 Sampling and Analysis Plan for the 100DR-1, 100-DR-2, 100-HR-1, 100-HR-2, and 100-HR-3 Operable Units Remedial Investigation Feasibility Study, Rev. 0* (TPA-CN-460).

2.1.8.1 Bathymetric Data

To evaluate flow paths of contaminants to aquatic receptors, updated and accurate bathymetric data for the river were needed (Data Gap 8). Recently collected bathymetric data were combined with groundwater fate-and-transport analysis to evaluate contaminant risks to potential ecological receptors and related portions of the river. Preliminary evaluation of the RUM surface using near river wells was sufficient to indicate that the RUM intersects the Columbia River. No additional data were proposed for the area as part of the RI/FS; however, the existing data were further evaluated to better define the river bathymetry. Bathymetry of the Columbia River near 100-D/H is shown on Figures 2-9 and 2-10.

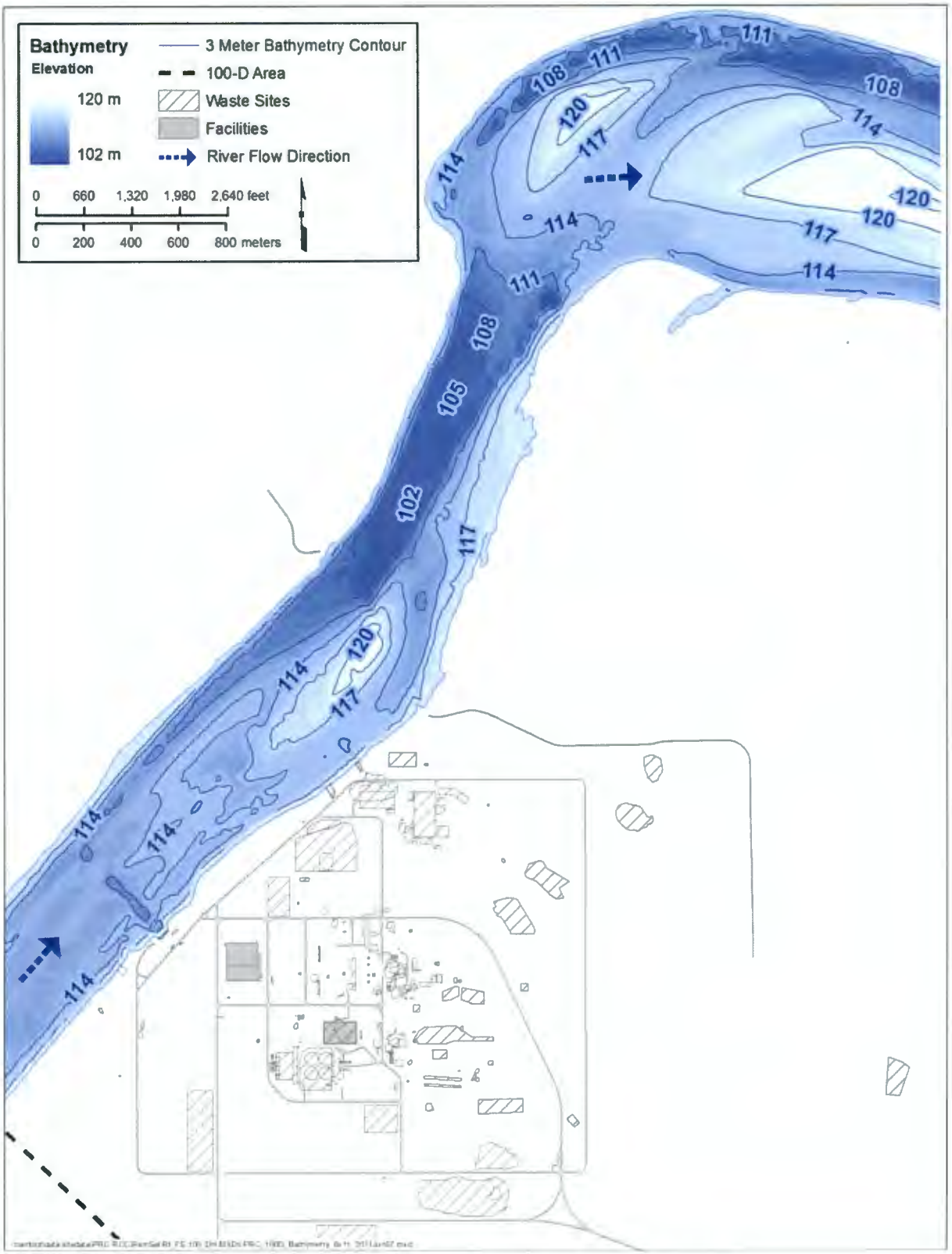


Figure 2-9. Bathymetry of the Columbia River near 100-D

Figure 2-10. Bathymetry of the Columbia River near 100-H

Cr(VI) and other contaminants have been detected above cleanup standards for drinking water and protection of aquatic receptors in the RUM unit and deeper Ringold units in 100-H groundwater. Additional information on the topography of the RUM unit surface relative to the topography of the river bottom was needed to evaluate the discharge locations of RUM unit groundwater

The development of a high-resolution bathymetry dataset for the Columbia River through the Hanford Reach was a continuation of FY 2009 work that focused on retrieving, assembling, and processing 66 km (41 mi) of existing bathymetry and terrestrial topographic data (*Development of a High-Resolution Bathymetry Dataset for the Columbia River through the Hanford Reach* [PNNL-19878]). At the conclusion of the FY 2009 work, it was determined that additional data needed to be collected over a 30 km (19 mi) section to supplement existing bathymetric and topographic data to fill significant data gaps in the central portion of the Hanford Reach. The hydrographic surveys were conducted in 2010 and the resulting data were merged to produce a single high-resolution (1 m [3.3 ft]) dataset for the Hanford Reach. Bathymetry data are incorporated in geologic cross sections presented in Appendix M while Chapter 4 presents a discussion of the updated bathymetric results.

2.1.8.2 Geophysical Logging

To gain a better understanding of the area geology, geophysical logging was conducted at each RI borehole (27 total). Logging was conducted using S.M. Stoller Corporation's Spectral Gamma Logging System and Neutron Moisture Logging System to identify natural and manufactured gamma emitting radionuclides and soil moisture, respectively, present near the boreholes. The starting point for logging, either the ground surface or top of the casing, was recorded for each well or borehole. Borehole logging was performed through temporary casing to produce a geophysical log of the entire length of the borehole. The log reports are located in the borehole summary reports (*Borehole Summary Report for the installation of 16 Resource Protection Wells in the 100-HR-3 Groundwater Operable Unit in Support of the Integrated 100 Areas RI/FS: 100-D/H Decisional Unit* [SGW-49912]; *Borehole Summary Report for 100-HR-3 Washington Closure Hanford RI/FS Boreholes* [SGW-50131]). The geophysical logging results are presented in Chapter 4, with the logs included in Appendix M.

2.1.8.3 Remedial Process Optimization Wells

Seventy RPO wells were installed in 100-D/H as part of an expansion for the pump-and-treat system. This expansion was conducted to meet the remedial objectives set forth in *Record of Decision for the 100-HR-3 and 100-KR-4 Operable Units Interim Remedial Actions, Hanford Site, Benton County, Washington* (EPA/ROD/R10-96/134), hereinafter called 100-H/K ROD. Figure 2-11 shows the locations of these wells. Drilling and sampling of these RPO wells were conducted as per the following control documents, and amended by the 100-D/H Work Plan (DOE/RL-2008-46-ADD1) and 100-D/H SAP (DOE/RL-2009-40):

- *ARRA Description of Work for the Installation of Fourteen Remedial Process Optimization Wells for the 100-D Area of the 100-HR-3 Operable Unit, FY2009* (SGW-41535)
- *Description of Work for the Installation of 35 Remedial Process Optimization Wells in the 100-H Area for the 100-HR-3 Operable Unit, Fiscal Year 2009* (SGW-41534)
- *ARRA FY2010 Description of Work for the Installation of 18 Scenario 5 Remedial Process Optimization Wells for the 100-D Area, 100-HR-3 Operable Unit* (SGW-44089)
- *ARRA FY2010 Description of Work for the Installation of 15 Scenario 5 Remedial Process Optimization Wells for the 100-H Area, 100-HR-3 Operable Unit* (SGW-44142)

- *Sampling and Analysis Plan for Installation of 100-HR-3 Groundwater Operable Unit Remedial Process Optimization Wells* (DOE/RL-2009-09, Rev. 0)
- *Sampling and Analysis Plan for Installation of 100-HR-3 Groundwater Operable Unit Remedial Process Optimization Wells* (DOE/RL-2009-09, Rev. 1)
- *Sampling and Analysis Plan for Installation of 100-HR-3 Groundwater Operable Unit Remedial Process Optimization Wells* (DOE/RL-2009-09, Rev. 2) for Scenario 5 wells project expansion
- Approximately 1.5 m (5 ft) into the RUM or at total depth (nine RPO wells only). Intended to test the vertical hydraulic conductivity of the aquitard and to test for the presence of contaminants.

Table 2-8 presents the vadose zone soil matrix, aquifer soil matrix, and RUM sampling analytes and physical property tests (nine wells only) that were conducted at the selected RPO wells specifically for the RI.

Results from these data are discussed in Chapter 4. The geologic interpretation from soil borehole logs is discussed in general in Chapter 3, and in detail in the Borehole Summary Report for RPO Wells (SGW-48612). The geology encountered in the new wells is generally consistent with previous reports.

2.1.9 Vadose Zone Investigations

The RI work is intended to develop and refine the CSM. One important aspect of the characterization is to provide information on the nature and extent of contaminant distribution in the vadose zone. As part of this effort, characterization wells, boreholes, and test pits were conducted at the locations described in the 100-D/H Work Plan (DOE/RL-2008-46-ADD1) and 100-D/H SAP (DOE/RL-2009-40).

Data needs specific to sources (soil) identified in the 100-D/H Work Plan (DOE/RL-2008-46-ADD1) and summarized in Table C-4 (Appendix C) are described in this section. Data needs were addressed in accordance with the 100-D/H Work Plan (DOE/RL-2008-46-ADD1) and 100-D/H SAP (DOE/RL-2009-40), except where noted, relative to unremediated and remediated waste sites and reactor areas (Data Gaps 1, 2, and 3, respectively). In addition, soil investigations were undertaken to evaluate the persistence of Cr(VI) (Data Gaps 7 and 10, respectively). The data collected were also used for development of fate and transport modeling (Data Gap 12).

2.1.9.1 Characterize Below Unremediated Waste Sites to Assess Nature and Extent of Contamination in the Vadose Zone

Characterization beneath unremediated waste sites was identified as Data Gap 1 in the 100-D/H Work Plan (DOE/RL-2008-46-ADD1). Waste sites with a high potential to affect groundwater are considered "high priority." At 100-D/H, those waste sites primarily handled high concentrations of sodium dichromate liquid. The waste sites identified in Chapter 1include 100-D-100, 100-D-12, 100-D-56, and 100-D-30, among others. Interim remedial actions have been effective in documenting the remaining residual contamination following the completion of RTD activities and are useful for assessing the nature and extent of contamination in the vadose zone. Data collected as part of the ongoing interim waste site remediation are used as a component of the CSM and in evaluating the nature and extent of contamination. CVP data collected during ongoing remediation are generally discussed in Chapter 4.

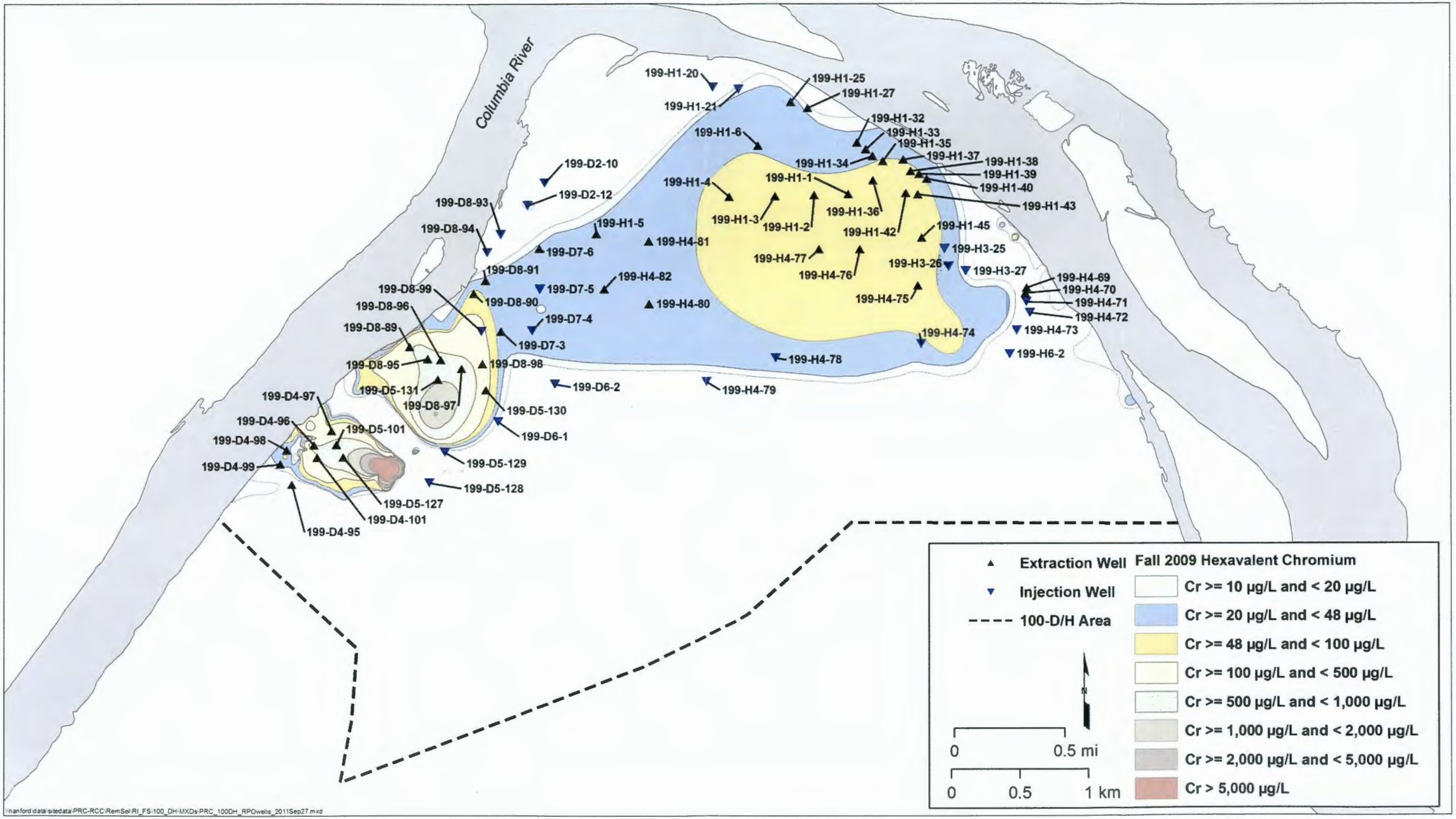


Figure 2-11. Remedial Process Optimization Well Locations

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Table 2-8. Soil Sample Chemical Analyses and Physical Property Tests for Nine RPO Boreholes

Radionuclides	Nonradionuclides*		Physical Properties
Barium-133	Antimony	Lithium	Grain size (sieve analysis)
Cesium-137	Arsenic	Manganese	
Cobalt-60	Barium	Molybdenum	
Europium-152	Beryllium	Nickel	
Europium-154	Boron	Selenium	
Europium-155	Cadmium	Silver	
Strontium-90	Cr(VI)	Thallium	
	Chromium (Total)	Vanadium	
	Copper	Zinc	
	Lead		

* Includes geochemical analyses for K_d and batch leach testing (see Chapter 5)

Remedial actions in 100-D/H began in 1996 under the authority of an interim ROD (*Interim Remedial Action Record of Decision for the 100-BC-1, 100-DR-1, and 100-HR-1 Operable Units, Hanford Site, Benton County, Washington* [EPA/ROD/R10-95/126]) and continue today. Cleanup has primarily consisted of RTD, which generates additional characterization data to address many of the vadose zone data gaps and helps refine overall site knowledge. Contaminated soil and debris are removed and disposed to the ERDF or another offsite facility (as appropriate) until the interim cleanup levels are met. Activities are guided during excavation using data obtained through field measurements or quick turnaround laboratory analyses. Sequencing of waste site cleanup is based on the TPA (Ecology et al., 1989a) milestone framework. Within this framework, knowledge of operational processes and past releases was used to target and prioritize specific waste sites or areas with contaminants that presently exist in groundwater or that could adversely affect it. Effective implementation of waste site cleanup prevents human and environmental exposure to soil contamination and further degradation of groundwater, thereby increasing the likelihood for success of groundwater cleanup actions (such as pump-and-treat).

As of November 2012, there were 343 waste sites, including subsites, identified in 100-D/H. Data needs associated with soil remedial actions in 100-D/H were met by planning and scheduling the remedial actions, collecting data to verify interim cleanup of waste sites, and obtaining concurrence from regulators on the achievement of interim RAGs for direct exposure, protection of groundwater, and protection of surface waters. Appendix E (Table E-1) documents data from "Interim Closed Out" and "No Action" sites that were incorporated into this RI/FS report.

2.1.9.2 Characterize Beneath and Beside Remediated Waste Sites

The need to provide additional characterization beneath and beside remediated waste sites (Data Gap 2) was addressed by installing 14 boreholes in 100-D, 13 boreholes in 100-H (Appendix C, Tables C-5 and C-6), and 5 test pits (Appendix C, Table C-7) to assess the vertical extent of contamination in the vadose zone. Characterization data were used to develop/refine the current CSM, including model input parameters and assumptions addressed in Chapter 5.

To fill the data need, boreholes and test pits were identified and sampled to refine the CSM, confirm modeling inputs, and provide data on the vertical distribution of contaminants. The RI activities required to address the data needs for Data Gap 2 involved drilling 10 boreholes, installing 2 permanent monitoring wells, excavating 5 test pits, and collecting and analyzing soil samples to assess the vertical extent of contamination in the vadose zone. Boreholes were drilled and sampled, and all but five boreholes were converted to 10 cm (4 in.) temporary PVC wells or permanent stainless steel wells (Appendix C, Table C-5). Characterization data collected beneath remediated waste sites were used to develop/refine the current CSM, including model input parameters and assumptions. Input parameters and assumptions used for remediated waste sites were compared against field data to identify the accuracy of model inputs and assumptions that affect contaminant migration predictions.

To determine which interim closed sites required additional characterization, all accepted waste sites having undergone an interim remedial action were evaluated with consideration of the following:

- Depth of remedial action relative to depth of the site's engineered structure
- Depth of contamination reported in historical documents relative to depth of remedial action
- Omission of historically reported contaminants during closure sampling analysis
- Closure sample concentrations relative to current "Model Toxics Control Act—Cleanup" (WAC 173-340, hereinafter called MTCA, Method B cleanup levels)
- Trends indicating contaminant concentration increases with depth
- Proximity to groundwater contaminant plumes
- Historically documented effects to groundwater
- Type of waste site (for example, high volume liquid effluent site, low volume liquid effluent site, or sludge trench)

Sites selected for characterization were identified during the work plan process in coordination between Ecology and DOE. Vadose zone soil samples were collected for characterization purposes during the field activities in locations as outlined in Table 2-1. Five boreholes were completed as 10 cm (4 in.) temporary PVC wells in order to obtain representative groundwater samples. Table C-8 (Appendix C) identifies the waste sites that were investigated and provides justification for selection, which was documented in Section 4.8.1 of the 100-D/H Work Plan (DOE/RL-2008-46-ADD1). Drilling of boreholes and wells, as outlined in Table 2-1, was conducted to address Data Gap 2.

Table C-5 (Appendix C) summarizes borehole and monitoring well sampling and location information, and Table C-7 (Appendix C) summarizes test pit information. Sampling and location information for the five boreholes completed as temporary monitoring wells (199-D8-101, 199-D5-142, 199-H4-84, 199-H4-83, and 199-H3-11) is presented in Section 2.1.

Samples were screened in the field for radiological contamination using field instruments and visually inspected for Cr(VI), as indicated by soil staining. Soil samples were generally collected from boreholes for analytical testing, field screening, and batch leach testing according to the 100-D/H SAP (DOE/RL-2009-40). Sampling typically was conducted at 4.5, 3, 1.5, and 0.6 m (15, 10, 5, and 2 ft) above the water table, at the water table, and 1.5 m (5 ft) into the aquifer. Location-specific target analytes specified in the 100-D/H SAP (DOE/RL-2009-40) include both a subset of the master list of target analytes (presented in Table 2-9) and additional analytes selected based on previous investigations and history of the waste site.

Table 2-9. Master List of Soil Target Analytes from 100-D/H Work Plan

Radionuclides	Nonradionuclides		
Americium-241	1,1-Dichloroethene	Beta-BHC	Manganese
Barium-133	4,4'-DDT	bis(2-ethylhexyl)phthalate	Mercury
Carbon-14	Acetone	Boron	Molybdenum Nickel
Cesium-137	Antimony	Cadmium	Nitrate (as N)
Cobalt-60	Aroclor-1016 (PCB)	Carbon tetrachloride	Nitrite (as N)
Europium-152	Aroclor-1221 (PCB)	Chloroform	Pyrene
Europium-154	Aroclor-1232 (PCB)	Chromium (total)	Selenium
Europium-155	Aroclor-1242 (PCB)	Chrysene	Silver
Neptunium-237	Aroclor-1248 (PCB)	Cobalt	Strontium
Nickel-63	Aroclor-1254 (PCB)	Copper	Sulfate
Plutonium-238	Aroclor-1260 (PCB)	Di-n-butyl phthalate	Thallium
Plutonium-239/240	Arsenic	Dibenz[a,h]anthracene	Tin
Strontium-90	Barium	Dieldrin	Trichloroethene
Technetium-99	Benzene	Fluoranthene	Uranium (total)
Tritium	Benzo(a)anthracene	Fluoride	Vanadium
Uranium-233/234	Benzo(a)pyrene	Cr(VI)	Vinyl Chloride
Uranium-235	Benzo(b)fluoranthene	Indeno(1,2,3-cd)pyrene	Zinc
Uranium-238	Benzo(k)fluoranthene	Lead	
	Beryllium	Lithium	

Source: *Integrated 100 Area Remedial Investigation/Feasibility Study Work Plan Addendum 1: 100-DR-1, 100-DR-2, 100-HR-1, 100-HR-2, and 100-HR-3 Operable Units* (DOE/RL-2008-46-ADD1)

Additional samples were collected, including at major formation and lithology changes, at the discretion of the geologist or sampler based on soil characteristics and field screening results; these samples were analyzed for physical properties (such as, grain size, porosity, moisture content, and bulk density, and for saturated samples, saturated hydraulic conductivity).

The split-spoon soil samples were collected in 0.76 m (2.5 ft) long (including shoe), 10 cm (4 in.) diameter split-spoon samplers lined with four 15 cm (6 in.) long Lexan® or stainless steel liners. The sampler was driven the full 0.76 m (2.5 ft) or until refusal (as determined by the onsite field geologist), whichever came first.

Groundwater samples were collected from open boreholes using a submersible pump with rates ranging from 3.29 L/min (0.87 gal/min) in C7851 to 38 L/min (10 gal/min) in C7864. Boreholes C7850 and C7855 were pumped at 22.7 and 7.6 L/min (6.0 and 2.0 gal/min), respectively. Before sampling, each borehole was purged long enough to provide stabilized field readings, but not necessarily three casing volumes. If significant drawdown occurred to where pumping could not be sustained, and a sample could not be collected, an alternative means of sampling was followed. A submersible pump was used for each of the boreholes, with the exception of C7862, which had inadequate recharge. Borehole C7862 was sampled using a "Kabis" sampler.

Boreholes C7852, C7857, C7860, C7861, and C7863 did not produce adequate water for sampling. Because of the difficulty of obtaining water samples, these five boreholes were completed as 10 cm (4 in.)

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temporary PVC wells. The temporary wells were developed before sampling. The field filtered water samples were analyzed for Cr(VI) and other metals to support K_d determination.

Boreholes C7850, C7851, C7855, C7862, and C7864 were not completed as monitoring wells and were decommissioned with DOE, Ecology, and EPA approval, in accordance with "Minimum Standards for Construction and Maintenance of Wells" (WAC 173-160), after geophysical logging and sampling were completed. The boreholes were backfilled to 0.6 m (2 ft) above static water level (to account for variability of the water table) with 10 to 20 mesh Colorado Silica Sand. The remaining borehole was filled with granular bentonite to within 0.9 m (3 ft) of ground surface. A cement seal was then placed from 0.9 m (3 ft) bgs to ground surface and marked with the name and date of the decommissioned borehole.

One test pit each was excavated and sampled at the100-D-12 Pump Station/French Drain, 100-D-4 Sludge Trench, 116-D-4 Liquid Waste Trench/Crib, 1607-H4 Septic Tank and Drain Field, and 116-H-2 Liquid Waste Trench/Crib waste sites (Appendix C, Table C-7). Table C-8 (Appendix C) provides justification for selection of these waste sites for sampling, as developed in the 100-D/H Work Plan (DOE/RL-2008-46-ADD1).

The 100-D-12 French Drain trench was excavated and sampled to a depth of 7.6 m (25 ft). The test pits at 100-D-4, 116-D-4, 116-H-2, and 1607-H-4 were excavated and sampled to a depth 6.2.m (19 ft). In addition to collection and submittal for laboratory analysis, samples were screened in the field for radiological contamination (using field instruments) and Cr(VI) (by visual observation).

Sampling commenced at the maximum depth of remedial action. The maximum depths of remedial action at waste sites are presented in Table 2-10.

Table 2-10. Maximum Depth of Remedial Action at Select Waste Sites

Waste Site	Depth m (ft) bgs
100-D-12	2.4 (8.0)
100-D-4	2.9 (9.5)
116-D-4	2.9 (9.5)

Waste Site	Depth m (ft) bgs
116-H-2	2.6 (8.3)
1607-H4	3.6 (11.0)

Samples were generally collected at 0.6 m (2 ft) intervals at the discretion of the geologist/sampler based on field screening results. One sample was also collected at the bottom of each excavation. Samples were collected for location-specific target analyte analysis, field screening, and batch leach testing. Excavations were backfilled immediately on completion of sampling.

2.1.9.3 Characterize beneath and around Reactor Structures

Additional characterization was needed for interim closed areas adjacent to the 105-D and 105-H Reactor facilities and soil underlying these reactors (Data Gap 3). Justification for characterization at these two reactors includes reports of leakage from FSBs during reactor operations, reports of contamination beyond the depth of remedial actions, the quantity of liquids managed, and the lack of sampling performed beneath the FSBs and around/beneath the reactors (Appendix C, Table C-8). To address Data Gap 3, boreholes were drilled adjacent to the 118-D-6:3 FSB (105-D Reactor) and through the118-H-6:3, FSB (105-H Reactor). The 118-H-6:3 FSB is also collocated with waste subsites 118-H-6:2; 105-H Reactor Ancillary Support Areas, Below-Grade Structures, and Underlying Soils, and 118-H-6:6 FSB Deep Zone Side-Slope Soils. One borehole was drilled within the boundary of each of the two reactor

FSBs, and samples were collected and analyzed to assess the vertical extent of contamination and to refine the 100-D/H CSM.

Remediation of the 118-H-6:3 FSB included removing the below grade structure and disposing of contaminated materials, including the soil underlying the former FSB floor and the side slopes. The 118-D-6:3 below grade structure remains in place; thus, the borehole was drilled as close as possible to the FSB. Copies of the borehole logs, detailed sampling summaries, well construction summaries, well summary sheets, geophysical logs, and final surveys are located in the borehole summary reports.

Additional characterization was not required for the 105-DR ISS reactor facilities, per the 100-D/H Work Plan (DOE/RL-2008-46-ADD1), because there was no historical evidence that the FSB leaked, and soil samples collected beneath the FSB floor indicated no contamination was present.

2.1.9.4 Evaluate Reasons for the Persistence of Cr(VI)

During the RI, data were collected to evaluate the potential for contaminants to be entrained within the soil matrix and be a continuing source of groundwater contamination (Data Gap 10). Samples were targeted within the upper and lower vadose zone, periodically rewetted zone, unconfined aquifer, RUM surface, and the first water-bearing unit of the RUM (Chapter 4). The soil data obtained during RI sampling and analysis activities, along with data from the RPO wells, were evaluated at the specified locations to decrease uncertainty about contaminant sources.

2.1.9.5 Develop Additional Data Needed for Modeling

Insufficient data to support fate and transport modeling were identified as Data Gap 12. The fate and transport of site contaminants in the environment is highly dependent on the effluent volume discharge and contaminant specific K_d, which quantifies the partitioning of a contaminant between a solid phase and an aqueous phase. Data needed to develop a K_d and to conduct accurate fate and transport modeling include: physical properties, hydraulic conductivity, batch leach test results, contaminant concentrations, and field screening parameters. Data to support contaminant fate and transport modeling were collected during the RI. Results of batch leach testing are discussed in Section 5.5, with details on sampling provided in the 100-D/H SAP (DOE/RL-2009-40).

Data to support contaminant fate and transport modeling were developed for selected RI soil samples as described in Chapter 5, including information on physical properties, hydraulic conductivity, batch leach test results, and field screening parameters, presented in Chapter 3.

2.1.10 Groundwater Investigations

Although considerable groundwater data have been gathered for 100-D/H, some additional data from monitoring wells and aquifer tubes were needed to support remedy decision making, as described below. This included additional investigation of selected waste sites. The rationale for selection of waste sites for additional groundwater characterization is summarized in Table C-8 (Appendix C).

Data needs specific to groundwater, as identified in the 100-D/H Work Plan (DOE/RL-2008-46-ADD1) and summarized in Table 2-1, are described in this section. Data collected to fill data gaps included sampling during drilling of the RPO wells within 100-D/H. Table 2-2 includes the supplemental investigations identified in the Integrated Work Plan (DOE/RL-2008-46), and other investigations that may potentially affect the remedy decision for 100-D/H.

2.1.10.1 Characterize Beneath the Unconfined Aquifer

The unconfined aquifer in 100-D is primarily within the Ringold unit E. At 100-H, the Ringold unit E is not present and the unconfined aquifer is within the Hanford formation. The horn represents a transitional

area where the unconfined aquifer is present in pockets of Ringold unit E or in the Hanford formation where the Ringold unit E is not present. The surface of the RUM is considered the base of the unconfined aquifer in 100-D/H, with the presence of silt and clay layers acting as a hydraulic barrier.

Several confined water-bearing sandy gravel units are present within and below the RUM. These water-bearing units may provide pathways for Cr(VI) to migrate between stratigraphic units under certain hydrogeologic conditions. Cr(VI)-contaminated groundwater from the RUM may discharge to the river, adversely affecting aquatic resources, and/or affecting portions of the RUM that have the potential for future use as a drinking water resource. As presented in *Aquifer Testing and Rebound Study in Support of the 100-H Deep Chromium Investigation* (SGW-47776), the first water-bearing unit in the RUM appears to be connected to the Columbia River in some locations and connected to the unconfined aquifer in other portions of 100-H. Previous investigations identified Cr(VI) within the first water-bearing unit of the RUM at 100-H, with concentrations ranging from below detection on the western side of 100-H to levels approaching 200 µg/L in one location, where extraction is currently occurring. Previous investigations of the RUM in other locations of 100-D/H were limited to one well location in 100-D and three wells across the horn, with low Cr(VI) levels identified in one of the horn area wells. Additional data were collected to further define the extent of contamination in the RUM and to support contaminant fate-and-transport evaluation.

Before the RI, only Well 199-D8-54B was completed in the first water-bearing unit of the RUM underlying 100-D. This well is located in an area of relatively low Cr(VI) concentrations in the northern plume. Data had not been collected beneath the 100-D southern plume, where the highest Cr(VI) concentrations are present in the unconfined aquifer. Cr(VI) concentrations in the southern plume are an order of magnitude greater than encountered in the northern plume. In 100-H, two wells (199-H3-2C and 199-H4-12C) and one piezometer (199-H4-15CS) are screened in the first water-bearing unit of the RUM. Cr(VI) concentrations in these wells/piezometer range from 55 to 153 µg/L during 2011. Piezometer host 199-H4-15 has three other screens (R, Q, and P) below the RUM at 59.1 to 59.7 m (194 to 196 ft), 89.9 to 90.5 m (295 to 297 ft), and 99.1 to 99.7 m (325 to 327 ft) bgs respectively. Cr(VI) has not been detected at concentrations above the AWQC in groundwater samples collected from these deeper sampling points.

Five deep boreholes were drilled as part of the RI to address the need for additional deeper characterization data. Two wells were placed at 100-D and three at 100-H. In 100-D, monitoring well 199-D5-134 (C7624, Well R4) was drilled slightly downgradient of the northern plume hot spot. Well 199-D5-141 (C7625, Well R5) was drilled near the 100-D-12 Pump Station/French Drain. In 100-H, groundwater monitoring well 199-H3-9 (C7639, Well R1) was drilled near the 116-H-7 Retention Basin. Well 199-H3-10 (C7640, Well R2) was drilled near the 183-H Clearwells/Disposal Pit. Well 199-H2-1 (C7631, Well R3) was drilled adjacent to the river, not bordering any known waste sites.

Well locations were selected to augment the existing well coverage in 100-H and to perform general characterization in 100-D in areas where deep investigations had not yet been carried out. Methods for collecting soil and groundwater samples during RUM drilling are described in Section 2.1.8. Table C-5 (Appendix C) summarizes borehole/well locations, depth to the Hanford formation/Ringold Formation unit E contact (if present), RUM surface contact, and well construction information. Table C-6 (Appendix C) summarizes the samples planned in the 100-D/H SAP (DOE/RL-2009-40) versus those actually collected during drilling.

Groundwater samples were collected at discrete depth intervals during drilling. Samples were collected from the unconfined aquifer, water-bearing units in the RUM, and deeper water-bearing units (presumed to be the Ringold Formation unit B). Laboratory analysis was conducted for analytes listed in Table 2-11. In addition, one field filtered groundwater sample was collected for analysis of Cr(VI) and other metals to

support K_d determination and to refine the nature and extent of Cr(VI) contamination. The sampling methodology used is described in Section 2.1.8. The deep boreholes were completed as monitoring wells in the first water-bearing unit of the RUM.

Table 2-11. Groundwater COPCs and Additional Analytes for 100-D/H

Contaminants of Potential Concern				
Radionuclides	**Nonradionuclides**			
Strontium-90 Technetium-99 Tritium	1,1-Dichloroethene Antimony Arsenic Benzene Beryllium Cadmium Carbon Tetrachloride	Chloroform Chromium (total) Cobalt Copper Fluoride Cr(VI)	Lead Manganese Mercury Nickel Nitrate (as N) Nitrite (as N) Selenium	Silver Sulfate Thallium Trichloroethene Uranium Vanadium Vinyl Chloride Zinc
Additional Analytes				
Radionuclides	**Nonradionuclides***			
Gross alpha Gross beta Cesium-137 Cobalt-60 Europium-152 Europium-154	Cyanide Pesticides Polychlorinated biphenyls Polycyclic aromatic hydrocarbons Semivolatile organic compounds			

Source: *Sampling and Analysis Plan for the 100-DR-1, 100-DR-2, 100-HR-1, 100-HR-2, and 100-HR-3 Operable Units Remedial Investigation/Feasibility Study* (DOE/RL-2009-40)

* Semivolatile organic compounds, polycyclic aromatic hydrocarbons, polychlorinated biphenyls, pesticides, and cyanide were analyzed for select wells and sampling events, as identified in Table 2-19 of DOE/RL-2009-40.

Split-spoon soil samples were collected at various depth intervals in the vadose zone and unconfined aquifer, at the aquifer/RUM contact, at additional depth intervals within the RUM, and within the Ringold Formation unit B. Soil samples were screened in the field for radiological and Cr(VI) contamination and laboratory analyzed for target analytes. Additional split-spoon samples were collected at major formation and lithology changes for analysis of physical properties (grain size, porosity, moisture content, bulk density, and vertical hydraulic conductivity). Samples were also collected at approximately 1.5 m (5 ft) intervals throughout the entire borehole intervals for field screening and geologic logging.

Figure 2-12 shows the general well construction design and well construction details provided in the Well Construction Summary Reports, which are included in the borehole summary reports. Table C-5 (Appendix C) summarizes the well construction, locations, and depth to the upper contact of the Ringold Formation unit E and the RUM encountered during drilling of each borehole.

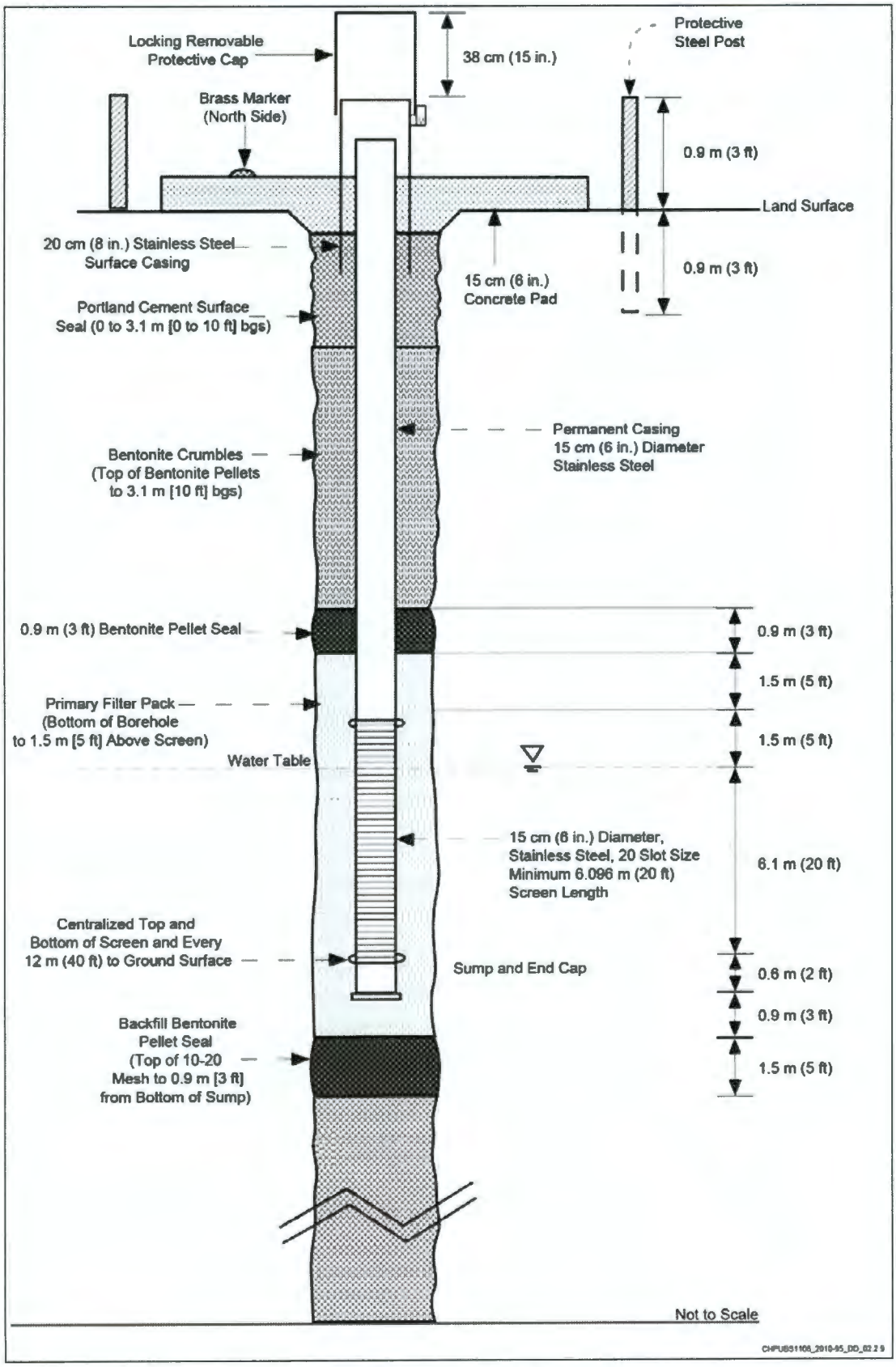

Figure 2-12. General Well Design

Following construction, wells were developed by pumping at sustainable flow rates up to 4.2 L/sec (55 gal/min). Development was continued until the well produced clear water consisting of low turbidity (less than or equal to 5 nephelometric turbidity units) and stabilized (at least three consecutive measurements within 10 percent of each other) temperature, pH, and specific conductance measurements were obtained. Water level drawdown and recovery was monitored with pressure transducer and datalogger equipment. These deep wells were not sampled further for the RI, but will be incorporated into the monitoring well network that is sampled periodically and included in annual reports (such as *Hanford Site Groundwater Monitoring Report for 2010* [DOE/RL-2011-01]).

Remedial Process Optimization Wells. As discussed in Section 2.1.8, 70 RPO wells were installed in 100-D/H as part of an expansion for the pump-and-treat system. Data were collected from these wells to support the RI effort in addition to the primary purpose of increasing efficiency of the pump-and-treat system. Groundwater samples collected specifically for the RI were from the unconfined aquifer. Results from these data are discussed in Chapter 3 and Chapter 4.

2.1.10.2 Refine Delineation of Nature and Extent of Groundwater Contamination in the Unconfined Aquifer

As part of the RI, it was necessary to determine the extent of select contaminants (Cr[VI] and strontium-90) at concentrations above cleanup standards (e.g., PRGs) in select locations in the unconfined aquifer in 100-D and 100-H (Data Gap 5). To address Data Gap 5, groundwater monitoring wells and aquifer tubes were installed at 100-D/H. Table 2-1 lists the wells and aquifer tubes installed to address this data gap. Information about the construction of these wells and aquifer tubes is summarized in Appendix C.

Each well was drilled to the top of the RUM and screened across the entire saturated thickness to characterize the unconfined aquifer. Soil samples were typically collected at 1.5 m (5 ft) intervals for geologic archive samples and field screening. Additionally, soil samples for each well were generally collected at 4.6, 3, 1.5, and 0.6 m (15, 10, 5, and 2 ft) above the water table, at the water table, 1.5 m (5 ft) below the water table, and at the bottom of the unconfined aquifer for analysis for location-specific target analytes, field screening parameters, and batch leach testing. Split-spoon samples were also collected from each borehole at major formations and lithology changes to provide site-specific physical property data (grain size, porosity, moisture content, bulk density, and vertical hydraulic conductivity) to support modeling efforts.

During the drilling of each well, groundwater samples were collected at 1.5 m (5 ft) intervals through the unconfined aquifer, beginning at approximately 1.5 m (5 ft) into the unconfined aquifer, for information on the vertical distribution of contaminants. Depth discrete groundwater samples were collected in each of the permanent wells during drilling starting at a depth of approximately 27 m (88 ft) in 100-D and starting at approximately 9 to 13 m (29 to 43 ft) in 100-H. Groundwater sampling continued to either the RUM contact for shallow wells or to the bottom of the borehole for deep (RUM) wells. All water samples from the wells were analyzed for groundwater COPCs (Table 2-11).

Wells were constructed and developed as described in Section 2.1.10.1. The wells were equipped with 3.1 to 4.6 m (10 to 15 ft) long screens, with construction details included in the borehole summary reports. In addition, copies of the borehole logs, detailed sampling summaries, geophysical logs, and final surveys are located in the borehole summary reports (Section 2.2). Table C-5 (Appendix C) summarizes borehole/well locations, depth to the Hanford formation/Ringold Formation unit E contact (if present), RUM surface contact, and well construction information.

At 100-D, Monitoring Well 199-D3-5 (C7620, Well 2) and Wells 199-D6-3(C7623, Well 5), 199-D5-140 (C7866, Well 9), and 199-D5-144 (C8668, Well R5 redrill) were installed to define the extent of the Cr(VI) plume to the south, southeast, and east, along with aquifer tubes (C7645, C7646, C7647, and C7648). The aquifer tubes were used in this area in lieu of installing monitoring wells to protect cultural resources. Monitoring Well 199-D3-5 (C7620, Well 2) was intended to define the extent of the Cr(VI) plume to the south; however, Cr(VI) was detected at that location, and therefore the edge of the plume has not been identified. If needed, additional wells may be installed during remedial design to address this data gap. In addition to defining the Cr(VI) plume, Monitoring Well 199-D6-3 (C7623, Well 5) was installed to define the extent of the strontium-90 plume, and to determine if potential sources of contamination are present in groundwater southeast of the 105-D Reactor. Monitoring Well 199-D5-133 (C7621, Well 3) was also installed to define the extent of the strontium-90 plume.

During the northern plume investigation (*Report on Investigation of Hexavalent Chromium Source in the Northern 100-D Area* [DOE/RL-2010-40]), a borehole had been drilled near the 100-D-56 Pipeline and had not identified high concentrations of Cr(VI) in soil. Therefore, during the RI, Monitoring Well 199-D5-140 (C7866, Well 9) was installed to investigate Cr(VI) sources near the 100-D-56 Pipeline where a hole in the pipeline, as a result of corrosion, was noted during remediation activities. However, because of an initial incorrect location of Well 199-D5-140 (C7866), a replacement well, Well 199-D5-143 (C8375, Well 9-redrill) was drilled in the planned location and sampled in accordance with the 100-D/H SAP (DOE/RL-2009-40). Since the northern plume investigation did not identify a Cr(VI) source, this well location was also selected to evaluate whether Cr(VI) concentrations in groundwater are higher near the pipeline than in nearby Wells 199-D5-125 and 199-D5-126, which have Cr(VI) concentrations between 1,500 and 2,000 µg/L.

From 1987 to 1999, groundwater samples from Monitoring Well 199-D5-12 (located east of the 105-D Reactor) had the highest strontium-90 concentrations identified in the unconfined aquifer underlying 100-D. This well was decommissioned in late 1999, with the last groundwater sample collected having strontium-90 concentrations exceeding the DWS of 8 pCi/L by approximately five times. Since that time, no wells have been available for sampling near this former well location. During the RI, Monitoring Well 199-D5-132 (C7622, Well 4) was installed as a replacement through the 116-D1-1A Trench to monitor both strontium-90 and Cr(VI). Well 199-D5-132 is located approximately 46 m (150 ft) and hydraulically downgradient from former Well 199-D5-12 and, therefore, monitors contaminants that may have sources located near the former well.

A cluster of two new aquifer tubes (C7649 and C7650), Monitoring Well 199-H3-6 (C7626, Well 6), and Wells 199-H3-7 (C7627, Well 7) and 199-H6-3 (C7628, Well 10) were installed in 100-H to define the extent of strontium-90 and to monitor nitrate concentrations. Specifically, the strontium-90 plume near waste sites 116-H-1 and 116-H-7, and along the river, had not been well defined.

Monitoring Wells 199-H6-3 (C7628, Well 10) and Well 199-H6-4 (C7629, Well 11) were installed to evaluate the southern extent of the strontium-90 plume, south of Well 199-H6-1 (and the former 107-H Liquid Disposal Trench [Waste Site 116-H-1]) in 100-H. In addition, the wells were placed to allow further evaluation of potential effects related to the 1607-H3 Septic System near a former guardhouse location (Facility 1720-H).

Monitoring Well 199-H1-7 (C7630, Well 12) was installed to assess groundwater effects north of the 1607-H3 Septic System where sufficient well coverage had not been available. Monitoring Well 199-H1-7 was also placed to monitor for polychlorinated biphenyls (PCBs), specifically Aroclor-1254), which was detected in an unfiltered groundwater sample from Well 199-H4-10 at

a concentration of 8.3 µg/L in November 2005. Aroclor-1254 was also detected in CVP samples collected during remediation of the 1607-H2 waste site.

Aquifer tubes are 0.64 cm (0.25 in.) outer diameter (0.43 cm [0.17 in.] inner diameter) polyethylene tubes that have a 15 cm (6 in.) long screen at the lower end. The tubes were implanted into the aquifer by driving a temporary steel casing into the ground and inserting a tube into the casing. The end of each tube was fitted with a screened section, which acts as the sampling port. The temporary steel casing was driven either by a hydraulic ram attached to a vehicle or by a hand-carried pneumatic air hammer. The steel casing was then backpulled, leaving the tube (and the stainless steel drive point) in place. Water is withdrawn from the tube using a peristaltic pump. The tubing exposed at ground surface is of minimal length (several feet) and is protected from wildlife and the elements by PVC conduit. Figure 2-13 shows the main components of aquifer tube installation. Each individual tube was driven to a different depth. These aquifer tubes were added to the SAP for Aquifer Sampling Tubes (DOE/RL-2000-59) to ensure that new aquifer tubes were installed and sampled consistent with existing aquifer tubes. Table C-10 summarizes information on the two new aquifer tube clusters that were installed as part of the RI.

2.1.10.3 Evaluate Reasons for the Persistence of Cr(VI)

Groundwater samples were analyzed for COPCs as specified in the 100-D/H Work Plan. In addition, for the low and transition river stage sampling rounds, groundwater samples from select wells were analyzed for cyanide, pesticides, PCBs, polycyclic aromatic hydrocarbons (PAHs), and/or semivolatile organic compounds (SVOCs), as in the 100-D/H Work Plan (DOE/RL-2008-46-ADD1).

2.1.10.4 Evaluate the 182-D Reservoir

The Export Water System, originally installed in the mid-1940s, provides redundant water supply capability using pumps located in 181-B/182-B Buildings as the primary source, with pumps in 181-D/182-D Buildings and 182-B Building diesel pumps as backup sources. Concerns over leakage from the 182-D Reservoir and the potential for any leakage to affect remedial actions led to a need to evaluate the reservoir uses (Data Gap 14). The current water system master plan (*Hanford Site Water System Master Plan* [HNF-5828], hereinafter called the Water System Master Plan) evaluated the future needs and infrastructure for the reservoir and export water lines. The Master Plan considers future operation needs for the 182-D Reservoir (Data Gap 14).

The 182-D Reservoir has a capacity of 94.6 million L (25 million gal); however, this has been administratively reduced to about 18.9 million L (5 million gal). Both operational and out-of-service pumps are present at 100-D/H. Use of the 182-D Reservoir is scheduled to continue for several years. Improvements to the 100-BC Export Water System, proposed for mid-2014, may facilitate some additional use of the system at 100-D/H during upgrades at 100-BC, depending on the status of use and water system expansion of operations at the 200 East and 200 West Areas. The current schedule has the 182-D Reservoir and associated pump stations closed by mid-2016, dependent on funding and other site priorities (Water System Master Plan [HNF-5828]). Automated groundwater elevation monitoring has been conducted historically near the reservoir. Quarterly sampling at selected wells near the reservoir will continue as part of standard remedy performance evaluation.

2.1.10.5 Ongoing Groundwater Monitoring

The 100-D/H RI groundwater data were evaluated with groundwater data from ongoing sampling of monitoring wells and aquifer tubes (Chapter 4). Monitoring wells and aquifer tubes in 100-D/H are sampled according to an established schedule and analyzed for specified constituents. The RI/FS unconfined aquifer wells, RUM wells, and aquifer tubes will be included in future groundwater sampling events to aid in monitoring the presence of selected COPCs in groundwater.

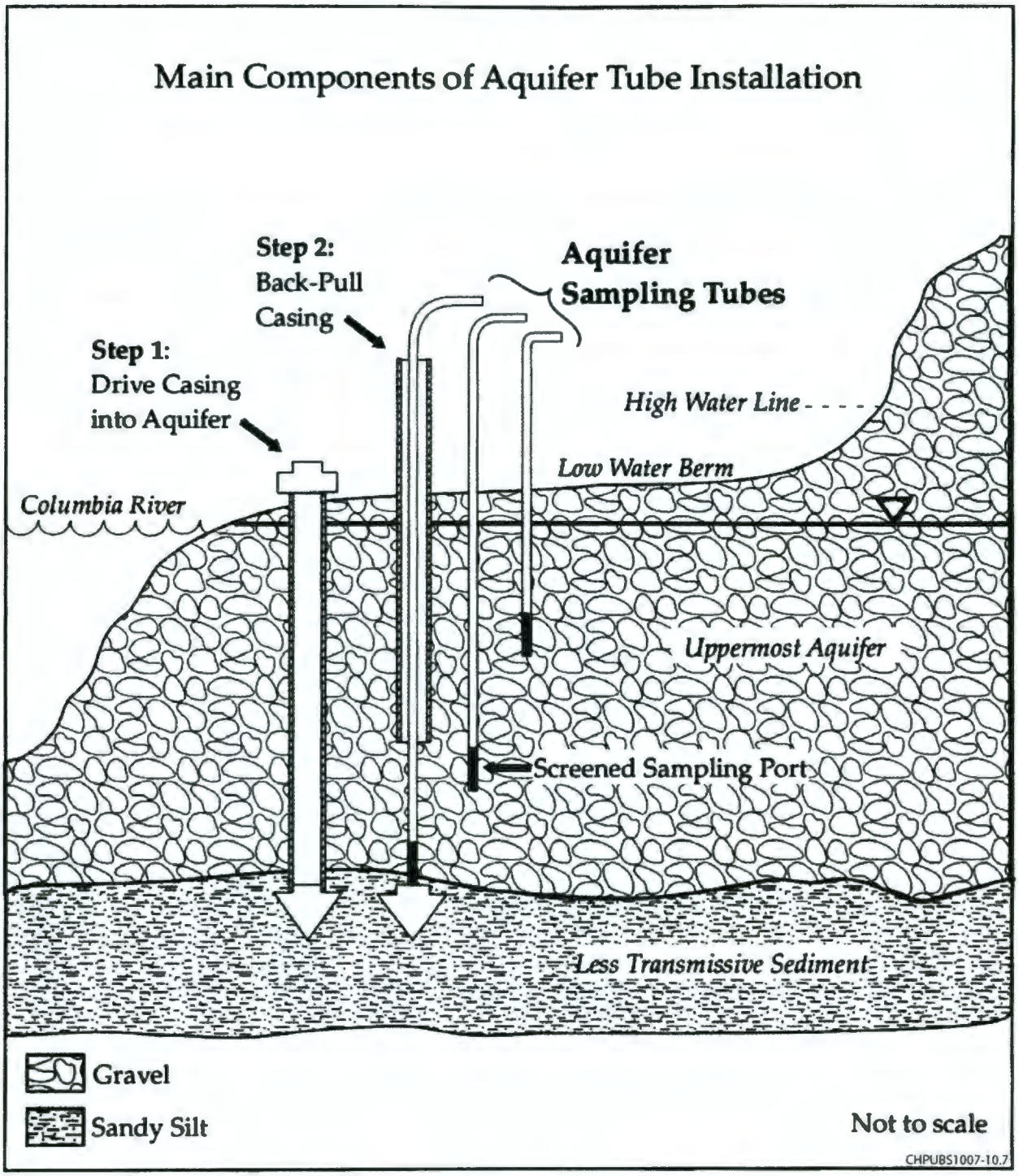

Figure 2-13. Main Components of Aquifer Tube Installation

The following list presents the main guiding documents governing monitoring well and aquifer tube sampling and analysis in 100-D/H. Numerous TPA Change Notices have been issued to update these documents as wells have been added or decommissioned and are available in the Administrative Record:

- *Sampling and Analysis Plan for Soil and Groundwater at the In Situ Bioremediation Design Test Wells at the Hanford 100-D Area* (DOE/RL-2010-56)
- 100-D/H SAP (DOE/RL-2009-40), as amended
- SAP for Aquifer Sampling Tubes (DOE/RL-2000-59)
- RCBRA SAP (DOE/RL-2005-42)
- *Supplement to the 100-HR-3 and 100-KR-4 Remedial Design Report and Remedial Action Workplan for the Expansion of the 100-KR-4 Pump and Treat System* (DOE/RL-2006-75)
- IAMP (DOE/RL-96-90), as amended
- *Remedial Design and Remedial Action Work Plan for the 100-HR-3 and 100-KR-4 Groundwater Operable Units' Interim Action* (DOE/RL-96-84)
- *Data Summary Report for the Remedial Investigation of Hanford Site Releases to the Columbia River, Hanford Site, Washington* (WCH-398)

2.1.11 Dense Chrome Theory Evaluation

Historical handling activities of sodium dichromate at 100-D (100-D-12 and former Railcar Unloading Station) have led to a theory regarding the potential presence of a dense Cr(VI) plume at the base of the unconfined aquifer, at the Ringold Formation unit E/RUM contact. This theory relates to Data Gaps 5 and 10, nature and extent of contamination in the unconfined aquifer and reasons for the persistence of the Cr(VI) plume, respectively. Concentrated solutions at 3,340 g/L being transferred off the rail car that leaked into the ground would have migrated through the vadose zone and entered the groundwater with relatively little dilution. Once in the groundwater, the density of the concentrated solution may have been sufficient to cause that solution to drop through the unconfined aquifer and collect on the RUM surface, because of the presence of silts/clays. Subsequently, this mass would diffuse and advect over the years within the groundwater flow regime to the current distribution. The maximum distribution observed at 199-D5-122 is about 69,700 μg/L, approximately 10,000 times less than the concentrated solution.

The vertical distribution of Cr(VI) in the unconfined aquifer was investigated previously (*Report on Investigation of Hexavalent Chromium in the Southwest 100-D Area* [DOE/RL-2009-92]). The investigation plan is described in *Field Investigation Plan for the Source of the Southwestern Chromium Plume in the 100-D Area* (DOE/RL-2006-74). Six wells near the 100-D South Plume "hot spot" were investigated using both Kabis and Solinst sampling devices. The only stratification in concentration was observed in Well 199-D5-99 at the base of the aquifer. Since that time, no additional work had been conducted to investigate Cr(VI) concentration stratification and determine if there are remnants in the current concentration distribution that support this theory. Therefore, passive samplers were used within four existing monitoring wells in 100-D/H to provide additional vertical stratification data. This additional work was performed outside the scope of the 100-D/H RI under a sample instruction. Results of the analyses are presented and discussed in Section 4.5.2.

2.1.12 Ecological Investigations

Ecological investigations have been conducted within or as part of the RI/FS for the 100-DR-1, 100-DR-2, 100-HR-1, 100-HR-2, and HR-3 OUs. These investigations have included work completed in support of the RCBRA (DOE/RL-2007-21), specifically Volume 1; the CRC (DOE/RL-2010-117), specifically Volume 1; and the Surface Environmental Surveillance Program (SESP) operated by Pacific Northwest National Laboratory (PNNL).

Ecological investigations for the RCBRA were identified through the data quality objective (DQO) process (*DQO Summary Report for the 100 Area and 300 Area Component of the RCBRA* [BHI-01757]) and as specified in the RCBRA SAP (DOE-RL-2005-42). Data collection consisted of sampling biotic media (e.g., plant, invertebrate, small mammal, bird egg, and fish tissue) and co-located abiotic media (e.g., soils, pore water, and sediment) for various contaminants of potential concern. Various field measures (e.g., species density and abundance) and biological assays (laboratory toxicity tests conducted on biota introduced to Hanford abiotic media) were also conducted. These data were included as appropriate as part of the ecological risk assessment presented in Chapter 7 and the riparian and nearshore evaluation in Appendix L of this report.

Ecological investigations for the CRC were identified through the DQO process (*DQO Summary Report for the Remedial Investigation of Hanford Site Releases to the Columbia River* [WCH-265]) and as specified in the Columbia River RI Work Plan and its corresponding SAP (DOE/RL-2008-11, Appendix A). Biota data collection focused on collecting fish tissue samples in 2009 and 2010. Associated abiotic media (surface water, sediment, and a groundwater upwelling study) were also collected to use in conjunction with the fish tissue samples. These data were included as appropriate as part of the ecological risk assessment presented in Chapter 7 and the riparian and nearshore evaluation in Appendix L of this report.

Through the SESP, PNNL monitors and surveys the Hanford Site plant and animal resources to establish potential radiological exposures as a result of site activities; assess the condition of endangered, threatened, or sensitive species; and evaluate breeding locations, habitat use, and distribution of key wildlife species. The following text describes the most recently published information regarding Hanford Site ecological monitoring activities (*Summary of the Hanford Site Environmental Report for Calendar Year 2008* [PNNL-18427-SUM]). Section 3.9 summarizes the ecology of the Hanford Site and Section 4.2.5 summarizes the results of the biota monitoring.

2.1.12.1 Vegetation Monitoring

In 2008, vegetation samples were collected on or adjacent to former waste disposal sites and from locations downwind and near or within the boundaries of operating facilities and remedial action sites to monitor for radioactive contaminants.

2.1.12.2 Fish and Wildlife Monitoring

Fish and wildlife on the Hanford Site are monitored for Site-produced contaminants. In 2008, sucker, common carp, smallmouth bass, and deer were collected at locations on and around the Hanford Site. Tissue samples were analyzed for strontium-90 and gamma emitters, including cesium-137. Since the 1990s, strontium-90 and cesium-137 have been the most frequently measured radionuclides in fish and wildlife samples. In addition, liver tissues from fish and deer were monitored for 17 trace metals.

2.1.12.3 Plant Communities and Population Surveys

Plant populations monitored on the Hanford Site include species listed by Washington State as endangered, threatened, or sensitive, and species listed as review group 1. Monitoring data are used to

develop baseline information and to monitor for changes resulting from Hanford Site operations. Surveys for rare animal species were conducted in 2008 as part of annual compliance review activities. More than 100 plants listed as endangered, threatened, sensitive, or on the view or watch list are found on the Hanford Site.

2.1.12.4 Wildlife Populations Surveys

Four fish and wildlife species on the Hanford Site are surveyed annually: fall Chinook salmon, steelhead trout, bald eagle, and mule deer. The number of fall Chinook salmon spawning nests (redds) in the Hanford Reach is estimated by aerial surveys. In addition, two aerial surveys were conducted to identify possible steelhead trout spawning areas. Roadside surveys were conducted for mule deer on the Hanford Site to assess age and sex ratios, and the frequency of testicular atrophy in males.

2.1.12.5 Habitat and Species Characterizations

Ecological monitoring on the Hanford Site includes characterizing breeding locations, habitat use, and distribution of key wildlife species. In 2008, characterization studies focused on the Woodhouse's toad and the burrowing owl, a Washington State candidate species and federal species of concern in this region. Toads were monitored using radio telemetry. Burrowing owl distributions and nesting habitats were evaluated.

2.1.12.6 Contaminated Biota

Radiological surveys are conducted around active and inactive waste sites at the Hanford Site to detect surface radiological contamination, including biointrusion, from plants and animals (including insects). The results from these surveys are used to determine trends, assess environmental impacts, and identify corrective actions, as appropriate. None of the 100 Area sites falls within the priority ranking for contamination incidents at the Hanford Site (most incidents are reported in the 200 Area). A total of 18 contamination episodes, mostly animal-related, were reported across the entire 100 Area in 2010 (*Quarterly Environmental Radiological Survey Summary, Fourth Quarter Calendar Year 2010* [HNF-SP-0665]).

2.1.13 River Corridor Supplemental Investigations

To support information needs for the entire River Corridor, the following supplemental activities from the Integrated Work Plan (DOE/RL-2008-46) were carried out separately from the RI field investigation:

- **Evaluated groundwater and surface water interactions for the River Corridor.** Flow paths in the groundwater/river zone of interaction vary with daily and seasonal fluctuations in river stage. River water infiltrates the banks during high river stages, moves inland, and then reverses flow as the river stage subsides and moves back through the hyporheic zone and discharges to the riverbed. Monitoring and modeling studies suggest that this back-and-forth motion of groundwater and the river is cyclical in response to the diurnal river stage cycles, which typically include two high and low stages in response to upstream hydroelectric dam power peaking demands. Review of modeling suggests there is a significant back-and-forth motion in the groundwater near the river that results in a substantial reduction in the groundwater flow velocity in the aquifer. It will experience numerous changes in flow direction before it eventually reaches the water column in the river. This concept is further discussed in Chapter 4 (Section 4.9).

- **Analyzed samples to determine River Corridor background concentration values for antimony, boron, cadmium, lithium, mercury, molybdenum, selenium, silver, and thallium.** Site-specific background values for these constituents were needed to determine soil cleanup values because calculated risk based concentrations and/or ecological protection concentrations were less than

Washington State or expected site-specific background values. Provisional data have been calculated and are presented in *Soil Background Data for Interim Use at the Hanford Site* (ECF-HANFORD-11-0038). Background values are discussed further in Section 4.1.

- **Reevaluated the soil cleanup level for Cr(VI) to support the ROD.** The lowest soil RAG for Cr(VI) under the interim action RODs is 2.0 mg/kg.

Based on the evaluation of soil cleanup levels and analytical methods, the accepted modeling approach was used to establish PRGs for this RI/FS. The development of PRGs for groundwater and surface water protection are presented in Chapter 5.

- **Determined a site-specific contaminant K_d for antimony.** Different K_d values have been identified at the Hanford Site for antimony. A site-specific value is needed to calculate soil RAG values (MTCA, "Deriving Soil Concentrations for Groundwater Protection" [WAC 173-340-747(3)(a)]). The summary of a scientific literature review conducted for this task is presented below:

  - The 1.4 mL/g K_d value is based on testing of Rainier Mesa tuff and does not appear to be comparable to Hanford Site soil types.
  - The 0 to 40 mL/g K_d range appears to be based largely on experience and general knowledge rather than on specific test results. A 1977 paper considered in establishing this range presents a K_d of approximately 65 mL/g antimony desorption from soil. This appears to be one of the few references available that presents actual K_d desorption data; the value supports the conclusion that desorption values are "much greater" than sorption values.

The 45 mL/g K_d value is a calculated value based on a theoretical correlation between K_d and the soil-to-plant concentration factor; it does not represent a value from experimental determination. This value is used by EPA and identified in "Cleanup Levels and Risk Calculations" (CLARC) database (Ecology, 2009), hereinafter called CLARC database.

  - The 3.76 mL/g K_d value comes from actual static batch equilibrium testing on sand/clay soil at a pH of 7.6, and appears to be a reasonable approximation of Hanford Site soil types. This value is based on sorption, not desorption.
  - Based on this review, a K_d value of 3.76 mL/g was used in the groundwater modeling presented in Chapter 5. This is a conservative value since it assumes a higher level of mobility than suggested by the technical review of the literature. The K_d value used, while conservative, results in a maximum concentration of antimony reaching groundwater at a peak year greater than 1,000 years, and the elimination of antimony as a groundwater COPC. A higher K_d value would have no effect on this result.

- **Reevaluated soil cleanup levels for arsenic to support the ROD.** The soil RAG for arsenic under the interim RODs is 20 mg/kg, based upon the TPA (Ecology et al., 1989) stipulation to use the MTCA ("Unrestricted Land Use Soil Cleanup Standards" [WAC 173-340-740(2)]) Method A value (*Remedial Design Report/Remedial Action Work Plan for the 100 Area* [DOE/RL-96-17]). The MTCA ("Unrestricted Land Use Soil Cleanup Standards" [WAC 173-340-740(2)]) Method A value is also 20 mg/kg. However, this 20 mg/kg value for arsenic exceeds the 1×10^{-6} individual cancer risk based on the MTCA ("Unrestricted Land Use Soil Cleanup Standards" [WAC 173-340-740(3)]) Method B value (0.67 mg/kg), and the MTCA ("Deriving Soil Concentrations for Groundwater Protection" [WAC 173-340-747(3)(a)]), groundwater protection value (0.00737 mg/kg). Both of these values are below the Hanford Site arsenic background concentration of 6.5 mg/kg.

Arsenic is a statewide concern because of historical smelter operations and pesticide use in agricultural areas (for example, orchards). The state of Washington has established programs to evaluate arsenic contamination continue to consider the Method A soil cleanup level of 20 mg/kg as a trigger for action. The Method A level (20 mg/kg) is proposed for continued use, which is consistent with other cleanup actions throughout the state.

2.2 Field Activity Documentation

As discussed in previous sections, field investigations have been conducted in 100-D/H to address the concerns discussed in the Data Needs Table 2-1, to supplement information received from the LFIs, and in response to results from ongoing remedial actions (for example, CERCLA 5-year reviews). The results of these field investigations are summarized in a variety of documents and tables. The following two borehole summary reports contain the borehole logs, detailed sampling summary, well summary sheets, and the final survey data:

- *Borehole Summary Report for the Installation of 16 Resource Protection Wells in the 100-HR-3 Groundwater Operable Unit in Support of the Integrated 100 Areas RI/FS: 100-D/H Decisional Unit* (SGW-49912)
- *Borehole Summary of Ten Characterization Boreholes in the 100-DR-1, 100-DR-2 and 100-HR-1, 100-HR-2 Source Operable Units in Support of the Integrated 100 Areas RI/FS: 100-D/H Decisional Unit* (SGW-50131)
- *Borehole Summary Report for the Installation of 70 Remedial Process Optimization, Pump-and-Treat Expansion Wells, for the 100-HR-3 Operable Unit* (SGW-48612)

The additional field data not contained within these reports are located in Appendix C.

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3 Physical Characteristics of the Study Area

This chapter describes the physical and environmental characteristics of 100-D/H, including the information obtained during the RI and ongoing monitoring activities. Physical characteristics are important components of the CSM because they help in presenting and understanding the nature and extent of contamination (as described in Chapter 4) and contaminant fate and transport (described in Chapter 5). The topics of this chapter include important surface features, meteorology, hydrology, geology, soil, hydrogeology, artificial water systems, demography and land use, ecology, and cultural resources. Some topics, such as regional geology and meteorology, concern the Hanford Site as a whole, while others are more specific to 100-D/H.

Highlights

- The topography in 100-D/H is relatively flat inland from the Columbia River, with a break in slope on the east side of 100-D and changes are greatest near the Columbia River where the riverbank slopes steeply.
- 100-D/H is characterized by a semiarid climate, shrub-steppe community with occasional high winds.
- The vadose zone consists of the highly transmissive and heterogeneous Hanford formation, underlain by the slightly less transmissive Ringold Formation unit E in 100-D. The less transmissive RUM unit underlies the Ringold Formation unit E and serves as an aquitard forming the base of the unconfined aquifer. The RUM contains poorly defined water-bearing layers.
- At 100-H, the Ringold Formation unit E is largely absent with the Hanford formation constituting both the vadose zone and unconfined aquifer.
- The unconfined aquifer is predominantly within Ringold Formation unit E at 100-D and predominantly within the Hanford formation in the horn and 100-H.
- Groundwater flows more readily across the Horn and 100-H because of the difference between the geologic units.
- The vadose zone is thicker at 100-D than at 100-H.
- River stage affects groundwater near the river, influencing groundwater elevations over 700 m (2,300 ft) inland at 100-D and over 640 m (2,100 ft) inland at 100-H. When river stage is low (October), groundwater flow is from 100-D/H toward the river. When river stage is high (June), water can enter the near-river aquifer and mix with 100-D/H groundwater.

3.1 Surface Features

Natural and manmade forces have modified the surface topography of the Hanford Site (*200 Areas Remedial Investigation/Feasibility Study Implementation Plan-Environmental Restoration Program* [DOE/RL-98-28]). The Columbia River, Pleistocene catastrophic flooding, Holocene eolian forces, interim remedial actions, and the construction of roads and buildings to support site missions have modified the Hanford Site topography. Basalt ridges and low-relief plains dominate the land surface of the Hanford Site. East-west trending anticlinal ridges are present south of 100-D/H. The surface topography of 100-D/H (Figure 3-1) was updated in FY 2010 using LIDAR, as described in Section 2.1.3.

The topography in 100-D/H consists of relatively steep banks rising up from the Columbia River and then generally flat to slightly undulating inland. Surface outburst channel features from ice age flooding events (discussed in Section 3.4) can be seen on Figure 3-1. The east side of 100-D slopes significantly and is several meters lower in elevation extending across the horn area into 100-H. This break in slope appears to coincide with the eastern edge of the Ringold Formation unit E, which is not present across the horn or at 100-H. Surface elevations range from approximately 154 m (505 ft) along the western boundary of 100-D/H to 115 m (377 ft) south of H Reactor.

3.2 Meteorology

The Hanford Site is located in a structural and topographic depression of the Columbia Plateau called the Pasco Basin. The area has a semiarid climate with dry and warm conditions. The Columbia Basin's large

size and complex topography contribute to substantial spatial variations in wind, temperature, precipitation, and other meteorological parameters, which are further affected by mountain barriers (*Hanford Site National Environmental Policy Act (NEPA) Characterization* [PNNL-6415], hereinafter called the NEPA Characterization Report). The Cascade Range to the west creates a rain shadow effect over eastern Washington State, minimizing precipitation at 100-D and 100-H, while the Rocky Mountains and ranges in southern British Columbia protect the sites from the more severe polar air masses from Canada (*Hanford Site Climatological Summary 2004 with Historical Data* [PNNL-15160]).

Climatologic data are monitored at the HMS near the 200 Area southwest of the 100-D/H boundary and other locations throughout the Hanford Site. Data gathered at the station are representative of conditions in 100-D/H. The station is approximately 12.9 km (8 mi) southwest of the 100-D/H boundary. From 1945 through 2009, the recorded maximum temperature was 45°C (113°F) during July 2002 and August 1961, and the recorded minimum temperature was -30.6°C (-23°F) measured twice in February 1950 (NEPA Characterization Report [PNNL-6415]). The monthly average temperature ranges from a low of -0.24°C (31.7°F) in January to a high of 24.6°C (76.3°F) in July. Annual average relative humidity at the HMS is 54 percent (NEPA Characterization Report [PNNL-6415]). It is highest during the winter months, averaging about 76 percent, and lowest during the summer, averaging approximately 36 percent (NEPA Characterization Report [PNNL-6415]). Tables 3-1 and 3-2 present the average monthly minimum and maximum temperatures, respectively, at the Hanford Site from 1945 through 2009.

Since 1947, annual precipitation at the Hanford Site has varied from approximately 7.6 to 31.3 cm (3.0 to 12.3 in.), with an annual average of 17.2 cm (6.8 in.). As shown in Table 3-3, most precipitation occurs during late fall and winter, with more than half occurring from November through February. Snowfall accounts for approximately 38 percent of precipitation at the Hanford Site from December through February (NEPA Characterization Report [PNNL-6415]) and for the majority of the moisture that infiltrates the ground. Average snowfall ranges from 0.25 cm (0.1 in.) during October to a maximum of 13.2 cm (5.2 in.) during December, and decreases to 1.3 cm (0.5 in.) during March. The highest monthly snowfall recorded at the HMS was 59.4 cm (23.4 in.) in January 1950.

Surface winds blow predominantly from the northwest during winter and summer months, and from the southwest during spring and fall. Local winds in the 100 Area and along the Columbia River are strongly influenced by near-river topography (NEPA Characterization Report [PNNL-6415]). Average monthly wind speeds at the Hanford Site are lowest during winter, averaging 10 to 11 km/h (6 to 7 mi/h) (Table 3-4). The highest average wind speeds, ranging from 14 to 16 km/h (8 to 10 mi/h), have been reported during summer. The fastest wind speeds recorded at HMS are usually associated with flow from the southwest. However, the summertime drainage winds from the northwest frequently exceed speeds of 47 km/h (30 mi/h).

Strong winds occasionally create blowing dust, and dust suppression measures are necessary during construction, demolition, and remedial actions to prevent the spread of contamination during periods of high winds. Methods used to minimize wind-related concerns in 100-D/H include applying dust suppression water and soluble adhesives. Wind and dust can limit the progress of work, and at times, it is necessary to stop work.

The wind speed class with the highest frequency of occurrence at HMS is 6.5 to 11 km/h (4 to 7 mi/h). Winds in that category occur 37 percent of the time. The speed class with the second highest frequency of occurrence is 13 to 19 km/h (8 to 12 mi/h), at 25 percent. Winds averaging over 40 km/h (25 mi/h) only occur 1 percent of the time on an annual basis, with the highest frequency (1.6 percent) in March (*Hanford Site Climatological Summary 2004 with Historical Data* [PNNL-15160]).

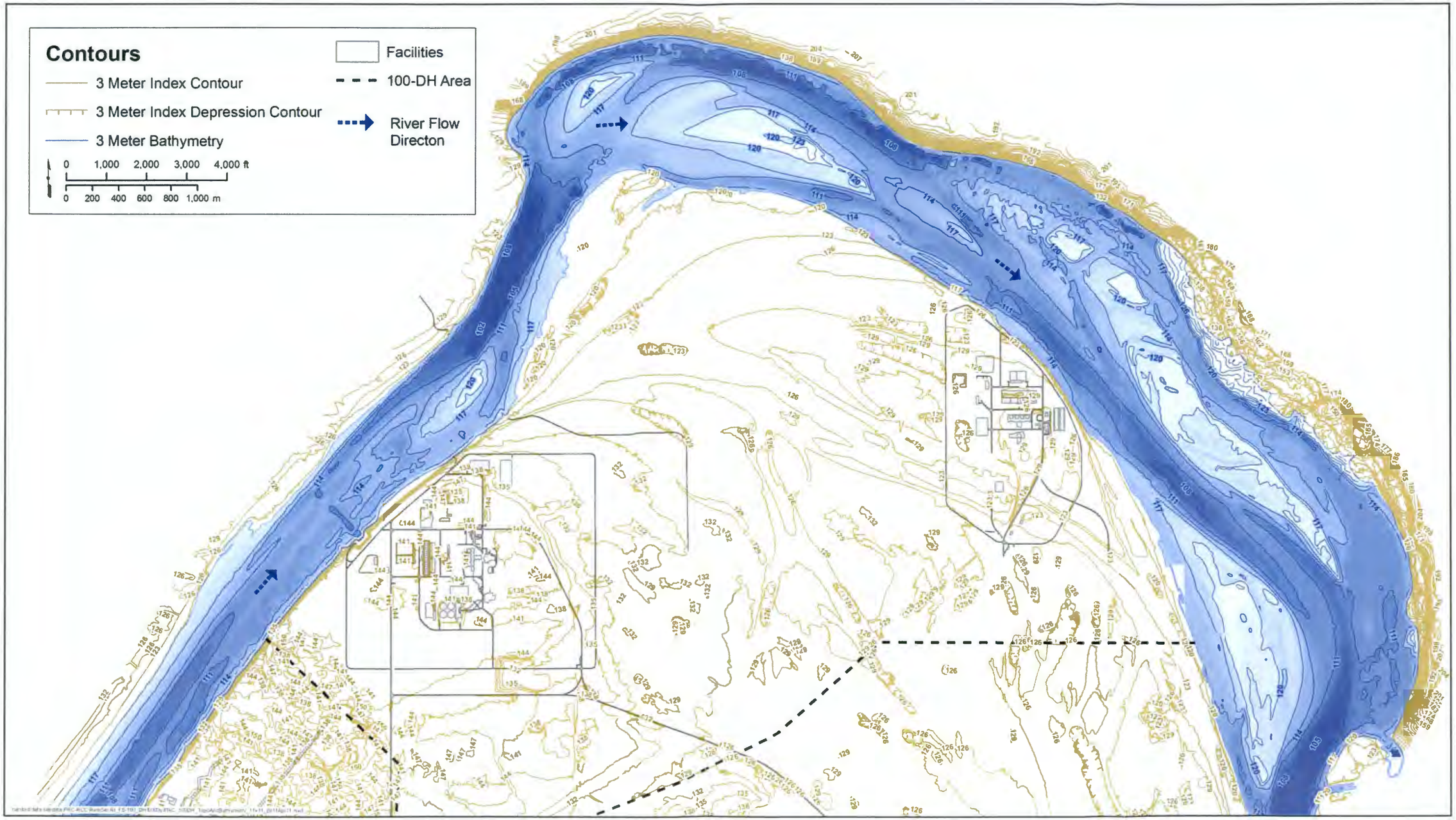

Figure 3-1. 100-D/H Topography and Columbia River Bathymetry (2010)

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Table 3-1. Monthly Minimum Temperatures from 1945 through 2009

1945 to 2009	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
Average (°C)	-14	-11	-6	-2	2	7	9	9	4	-3	-8	-13
Average (°F)	6	12	21	29	35	44	49	49	39	27	18	9
Lowest (°C)	-30	-31	-14	-6	-2	3	4	5	-1	-14	-25	-26
Lowest (°F)	-22	-23	6	21	28	37	39	41	30	7	-13	-14
Highest (°C)	-4	-2	0	3	9	11	14	13	9	1	-2	-5
Highest (°F)	24	29	32	37	48	52	58	56	48	34	28	23

Source: http://www.hanford.gov/page.cfm/hms/products/minmonth.

Table 3-2. Monthly Maximum Temperatures from 1945 through 2009

1945–2009	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
Average (°C)	14	17	21	27	34	37	41	40	35	27	18	14
Average (°F)	57	62	70	81	93	99	105	104	95	80	65	57
Lowest (°C)	2	8	17	22	27	30	36	36	30	22	12	4
Lowest (°F)	36	46	63	71	81	86	96	96	86	72	54	39
Highest (°C)	22	22	28	34	40	44	45	45	41	32	24	21
Highest (°F)	72	72	83	94	104	111	113	113	106	89	76	69

Source: http://www.hanford.gov/page.cfm/hms/products/maxmonth.

Table 3-3. Average, Minimum, and Maximum Monthly Precipitation from 1947 through 2009

1947–2009	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
Average (cm)	2.40	1.66	1.26	1.30	1.32	1.40	0.63	0.73	0.81	1.40	2.26	2.64
Average (in.)	0.95	0.65	0.50	0.51	0.52	0.55	0.25	0.29	0.32	0.55	0.89	1.04
Minimum (cm)	0.20	0.03	0.05	0.03	0.05	0.03	0.00	0.00	0.00	0.03	0.05	0.18
Minimum (in.)	0.08	0.01	0.02	0.01	0.02	0.01	0.00	0.00	0.00	0.01	0.02	0.07
Maximum (cm)	6.27	5.33	4.72	5.66	5.16	7.42	4.47	3.45	3.40	6.91	6.78	9.37
Maximum (in.)	2.47	2.10	1.86	2.23	2.03	2.92	1.76	1.36	1.34	2.72	2.67	3.69

Source: http://www.hanford.gov/page.cfm/hms/products/totprcp.

Table 3-4. Monthly Extremes for Prevailing Wind Directions, Average Speeds, and Peak Gusts at 15.2 m (50 ft) Level 1945 through 2004

Month	Prevailing Direction	Average Speed (km/h [mi/h])	Highest Average		Lowest Average		Peak Gusts		
			km/h (mi/h)	Year	km/h (mi/h)	Year	Speed (km/h [mi/h])	Direction	Year
January	NW	10 (6.3)	16.6 (10.3)	1972	4.7 (2.9)	1985	129 (80)	SW	1972
February	NW	11 (7.0)	17.9 (11.1)	1999	7.4 (4.6)	1963	105 (65)	SSW	1999*
March	WNW	13 (8.2)	17.2 (10.7)	1977	9.5 (5.9)	1958	113 (70)	SW	1956
April	WNW	14 (8.8)	17.9 (11.1)	1972	12 (7.2)	2004	117 (73)	SSW	1972
May	WNW	14 (8.9)	17.2 (10.7)	1983	9.3 (5.8)	1957	114 (71)	SSW	1948
June	NW	15 (9.1)	17.2 (10.7)	1983	12 (7.3)	1982	116 (72)	SW	1957
July	NW	14 (8.6)	17.2 (10.7)	1983	11 (6.8)	1955	111 (69)	WSW	1979
August	WNW	13 (8.0)	15 (9.5)	1996	10 (6.0)	1956	106 (66)	SW	1961
September	WNW	12 (7.4)	15 (9.2)	1961	8.7 (5.4)	1957	105 (65)	SSW	1953
October	NW	11 (6.6)	15 (9.1)	1946	7.1 (4.4)	1952	116 (72)	SW	1997
November	NW	10 (6.4)	16 (10.0)	1990	4.7 (2.9)	1956	108 (67)	WSW	1993
December	NW	9.7 (6.0)	13 (8.3)	1968	5.3 (3.3)	1985	114 (71)	SW	1955
Annual	NW	12 (7.6)	14 (8.8)	1999	10 (6.2)	1989	129 (80)	SW	January 1972

Source: *Hanford Site Climatological Summary 2004 with Historical Data* (PNNL-15160), Table 5.1.

3.3 Surface Water Hydrology

The Columbia River is the only perennial surface water feature associated with 100-D/H. The Columbia River influences site hydrogeology and contaminant migration, and is used as an onsite and regional water supply.

3.3.1 Columbia River

The Columbia River is the only natural surface water feature near 100-D/H and the study area to the west, north, and east (Figure 3-1). The Columbia River has played a major role in the depositional and erosional processes that helped produce the sedimentary and geologic features across the Hanford Site.

The stretch of Columbia River along the 100 Areas is referred to as the Hanford Reach. The Hanford Reach extends from river mile 396 approximately 1.6 km (1 mi) below Priest Rapids Dam downstream 81.6 km (51 mi) to river mile 346.5 north of the 300 Area north of the city of Richland, Washington (Figure 1-1). In May 2000, the Hanford Reach was incorporated into the 70,820 ha (175,000 ac) Hanford Reach National Monument ("Establishment of the Hanford Reach National Monument" [65 FR 37253]). River flows here are managed by controlling discharges from Priest Rapids Dam for generating power, controlling floods, and promoting salmon egg and embryo survival.

The Columbia River is noted for its very high quality water, exhibiting low suspended and dissolved solids load, low nutrient content, and absence of microbial contaminants (*Site Characterization Plan: Reference Repository Location, Hanford Site, Washington* [DOE/RW-0164]). While the river has produced large, episodic floods in the past, the construction of multiple dams on the Columbia River has considerably reduced the likelihood of future large scale flooding (*Final Environmental Impact Statement Disposal of Hanford Defense High-Level, Transuranic and Tank Wastes: Hanford Site, Richland, Washington* [DOE/EIS-0113]).

Columbia River flows typically peak from April through June, during spring run-off from regional and high elevation snowmelt. Flows are lowest from September through October. Flow rates range from approximately 1,020 to 10,300 m^3/s (36,000 to 362,000 ft^3/s) (*Hydrodynamic Simulation of the Columbia River, Hanford Reach, 1940-2004* [PNNL-15226]), depending on the releases from Priest Rapids Dam. It has an average flow rate of approximately 3,250 m^3/s (115,000 ft^3/s).

Construction of the Priest Rapids Dam began in 1956, with power generation starting in 1959. Priest Rapids operates as a run-of-the-river dam rather than a storage dam. Hourly to daily release rates of the Priest Rapids Dam further manage river stage to control the potential for flooding from the Columbia River at the Hanford Site. The nearest dam downstream from 100-D/H is McNary Dam. Construction of the McNary Dam began in 1947 with power generation starting in 1954. The dams result in a diurnal cycle of river stage in response to power generation at the dams, in addition to the annual cycle of river stage in response to snowmelt and seasonal runoff.

The depth and width of the Columbia River varies with changes in discharges and flow rates. River width within the Hanford Reach can vary from approximately 300 to 1,000 m (1,000 to 3,300 ft). Varying with flow rate, river width fluctuations cause repeated wetting and drying of the shoreline area (NEPA Characterization Report [PNNL-6415]).

The Columbia River stage is measured and recorded hourly at both 100-D and 100-H. As previously discussed, the river stage fluctuates throughout the year, depending on season. The Priest Rapids Dam, which is upstream from the 100 Area, ultimately controls the volume of river water flowing through the Hanford Reach. Figure 3-2 shows the daily average and annual river discharge rates at Priest Rapids Dam from 2005 through 2009. The river responses at the 100-D and 100-H gauge stations are similar. The highest discharge rates generally occur in the spring, when snowmelt is contributing to surface runoff into the river. Fluctuations in river stage directly influence groundwater levels, as described in Section 3.7.2.

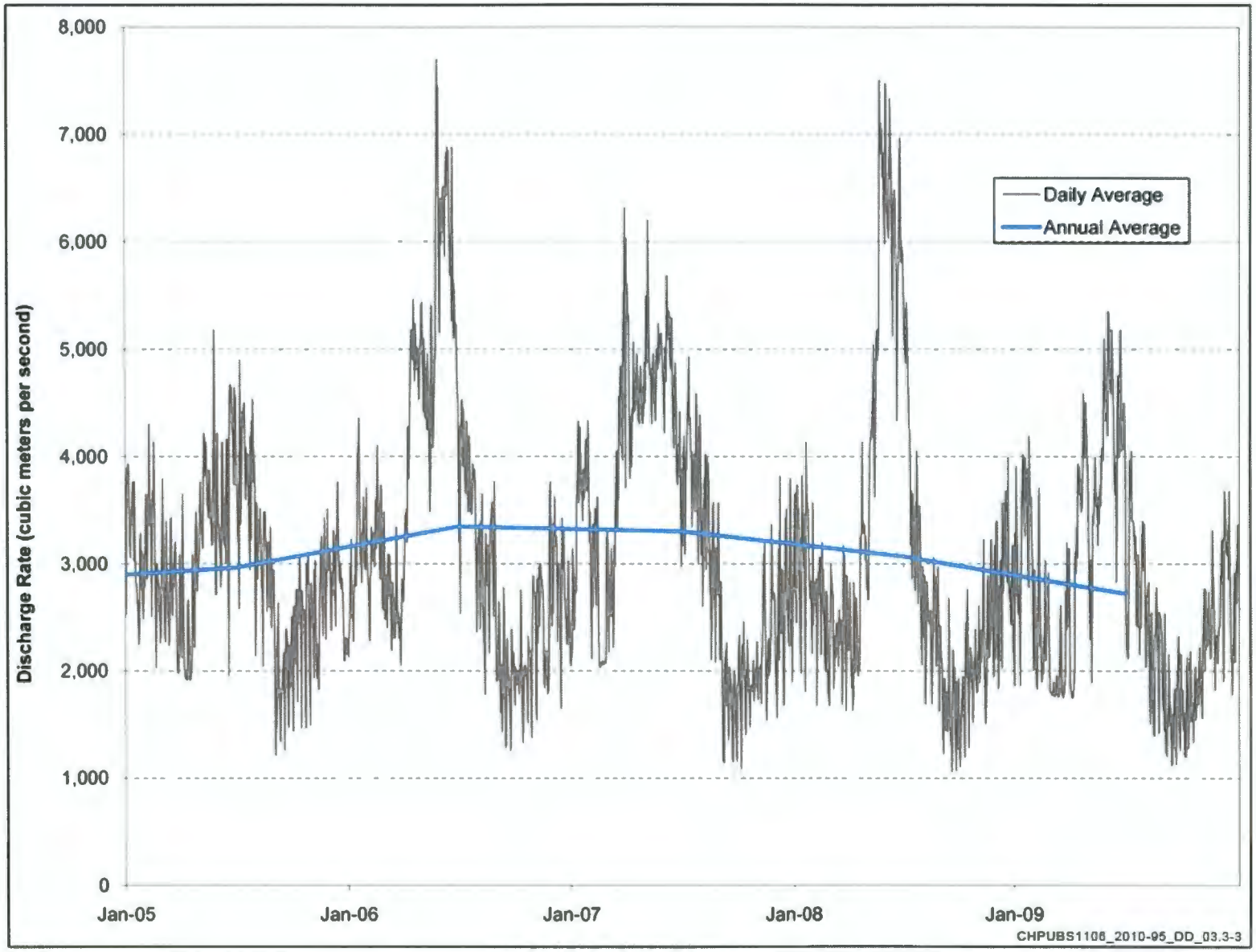

Figure 3-2. Daily Average and Annual Average Discharge Rates at Priest Rapids Dam (2005–2009)

3.3.2 Surface Water Use

Surface water from the Columbia River is withdrawn at 100-D for Hanford Site water use (*Hanford Site Environmental Report for Calendar Year 2006* [PNNL-16623]). The 182-D Reservoir was constructed and began use in 1945. The reservoir provided cooling water to D Reactor and a secondary supply of raw water to the rest of the Hanford Site. Section 3.8 provides additional details regarding the water supply system on the site. The Columbia River also provides the primary source of drinking water for the downstream municipalities of Richland, Kennewick, and Pasco, Washington. Numerous agricultural production operations downstream of Hanford also withdraw water from the Columbia River for irrigation.

3.4 Geology

The Hanford Site and Pasco Basin lie within the Columbia Plateau of southeastern Washington State. This broad plain, situated between the Cascade Mountains to the west and the Rocky Mountains to the east, is underlain by a thick sequence of volcanic Columbia River basalt, which forms the basement rock for the region (*Hydrogeologic Model for the Gable Gap Area, Hanford Site* [PNNL-19702]). Tectonic folding and faulting, which began with extrusion of the Columbia River Basalt Group (CRBG) basalts, continues to the present. The last basalt flows to reach the Pasco Basin occurred between 8.5 and 10.5 million years ago ("The Saddle Mountains: The Evolution of an Anticline in the Yakima Fold Belt" [Reidel, 1984]; *Site Characterization Plan: Reference Repository Location, Hanford Site, Washington* [DOE/RW-0164]). Unconsolidated sediments of the late Miocene, Pliocene, and Pleistocene ages (suprabasalt sediments) have accumulated up to 520 m (1,700 ft) thick in the Pasco Basin, the result of ancestral Columbia and possibly Snake/Clearwater River deposition (*Geologic Setting of the 100-HR-3 Operable Unit, Hanford Site, South-Central Washington* [WHC-SD-EN-TI-132]).

During the Ice Age (Pleistocene epoch), massive cataclysmic floods repeatedly occurred, interrupted by interglacial periods of several tens of thousands of years. Three episodes of cataclysmic flooding are recognized in the Pasco Basin. The oldest ice age floods were at least 770,000 years ago (±20,000 years); however, the first floods may have occurred closer to the beginning of the Ice Age, 2.6 million years ago. Gravels associated with the last (most recent) episode of flooding are found throughout the Pasco Basin. Fine grained deposits associated with that flood event contain volcanic material dating between 13,000 and 15,000 years ago (*Geology and Hydrology of the Hanford Site: A Standardized Text for Use in Westinghouse Hanford Company Documents and Reports* [WHC-SD-ER-TI-003]).

3.4.1 Geologic Setting

100-D/H lies on the northern flank of the Wahluke Syncline where the Columbia River winds around the north end of the Hanford Site and flows to the southeast towards Wallula Gap. The suprabasalt sediments at 100-D/H are as much as 108 m (355 ft) thick (*Geologic Setting of the 100-HR-3 Operable Unit, Hanford Site, South-Central Washington* [WHC-SD-EN-TI-132]).

Numerous investigations have been conducted in 100-D/H that contributed to the understanding of the geology of the area. As part of the RI, data from recently drilled wells were evaluated and combined with historical information and data from wells drilled since 1996 to support the interim actions. The information has been integrated to form an updated interpretation of 100-D/H geology. The general stratigraphic relationships of the units have remained unchanged, but the local geometry and thickness relationships are more detailed.

Geologic units from shallowest to deepest are Holocene sediments, Hanford formation, Ringold Formation, and the Columbia River Basalt Group. *100 Area Stratigraphic Database Development*

(ECF-100NPL-11-0070) describes the process used to create geologic maps and cross sections presented in Appendix M.

A partial listing of previous reports used to supplement the RI data include (but are not limited to) the following documents.

- *Limited Field Investigation Report for the 100-HR-3 Operable Unit* (DOE/RL-93-43)
- *Geologic Setting of the 100-HR-3 Operable Unit, Hanford Site, South Central Washington* (WHC-SD-EN-TI-132)
- *Geology of the Northern Part of the Hanford Site: An Outline of Data Sources and the Geologic Setting of the 100 Areas* (WHC-SD-EN-TI-011)
- *Remedial Design Report and Remedial Action Work Plan for the 100-HR-3 Groundwater Operable Unit In Situ Redox Manipulation* (DOE/RL-99-51)
- *Description of Work for the Installation of Two NABIR Wells at the 100-H Area, FY2006* (WMP-29720)
- *Sampling and Analysis Plan for Installation of 100-HR-3 Groundwater Operable Unit Remedial Process Optimization Wells* (DOE/RL-2009-09)
- *Borehole Summary Report for the Installation of 70 Remedial Process Optimization, Pump-and-Treat Expansion Wells, for the 100-HR-3 Operable Unit* (SGW-48612), which describes 70 new wells drilled to support the DX and HX interim action pump-and-treat expansions and included specific data collected in support of the RI/FS
- *Aquifer Testing and Rebound Study in Support of the 100-H Deep Chromium Investigation* (SGW-47776)
- *Report on Investigation of Hexavalent Chromium Source in the Northern 100-D Area* (DOE/RL-2010-40)
- *Report on Investigation of Hexavalent Chromium Source in the Southwest 100-D Area* (DOE/RL-2009-92)
- *Hydrogeological Summary Report for the 600 Area Between 100-D and 100-H for the 100-HR-3 Groundwater Operable Unit* (DOE/RL-2008-42)
- *Integrated 100 Area Remedial Investigation/Feasibility Study Work Plan, Addendum 1: 100-DR-1, 100-DR-2, 100-HR-1, 100-HR-2 and 100-HR-3 Operable Units* (DOE/RL-2008-46-ADD1)
- *Sampling and Analysis Plan for the 100-DR-1, 100-DR-2, 100-HR-1, 100-HR-2, and 100-HR-3 Operable Units Remedial Investigation/Feasibility Study* (DOE/RL-2009-40)

Additional documents are listed in Appendix B and Chapter 11, References.

Geologic data obtained from the recent drilling of RI and RPO boreholes have considerably improved the knowledge of 100-D/H stratigraphic relationships. Before drilling these boreholes, the geographic location of where the unconfined aquifer matrix transitions from the Ringold Formation unit E to the Hanford formation was not as well defined. The location of this lithologic transition, particularly to the east of 100-D, is important because the Hanford formation is more transmissive than the Ringold Formation unit E, which influences groundwater flow (Section 3.7). Data from the new boreholes also provides better delineation of the RUM surface.

3.4.2 Stratigraphy

Stratigraphic units at 100-D/H are listed below. Three of these units are most important to understanding groundwater contamination effects, the Hanford formation, Ringold Formation unit E, and the RUM. The cataclysmic flood deposits that form a major portion of the sediment present in the Pasco Basin are not formally named in the stratigraphic nomenclature. Therefore, this unit is informally referred to as the Hanford formation. Figure 3-3 presents a geologic/hydrostratigraphic column for 100-D/H.

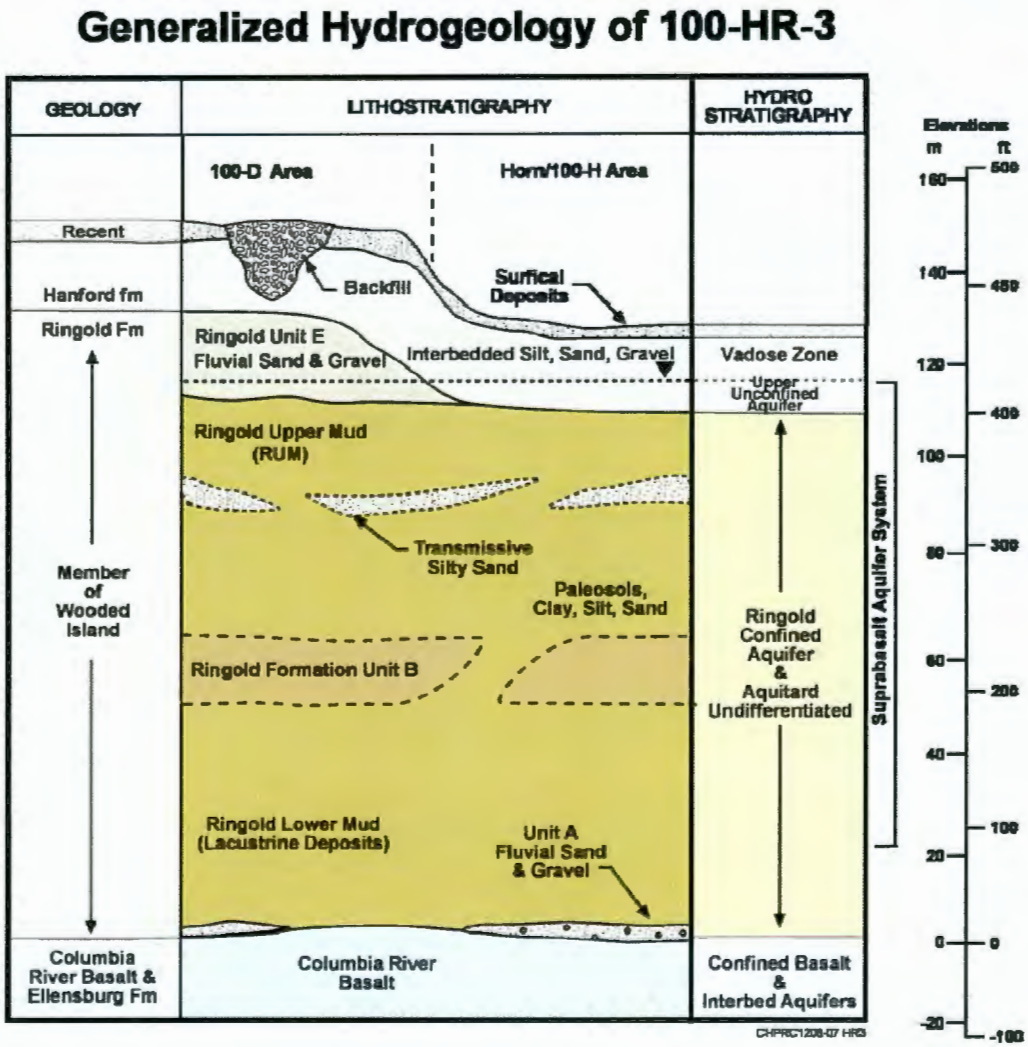

Figure 3-3. Generalized Geology and Hydrostratigraphy of 100-D/H

Specific stratigraphic units are listed below (from youngest to oldest) and described in more detail in the sections that follow:

- Recent eolian or anthropogenic deposits (sand; sand and gravel)
- Hanford formation (sand and gravel)
- Ringold Formation unit E (sand and gravel)

- RUM (silt, fine sand, and clay—includes water-bearing gravely to sandy layers)
- Ringold Formation unit C (sand and gravel)
- Ringold Formation lower mud (RLM) unit (silt and clay)
- Ringold Formation unit B (sand within RLM)
- CRBG (Columbia River Basalt Group flows interlayered with Ellensburg Formation sediments)
- Ellensburg Formation (sedimentary interbeds [tuff, paleosols, and sand] between CRBG basalt flows, including the Rattlesnake Ridge Interbed)

The sediments that overlie the basalts are divided into two primary units. The Ringold Formation is of late Miocene to middle Pliocene age (approximately 10.5 to 3 million years before present [B.P.]) (*Sedimentology and Stratigraphy of the Miocene-Pliocene Ringold Formation, Hanford Site, South-Central Washington* [WHC-SA-0740-FP]). The Ringold Formation is overlain by the informally named Hanford formation of Pleistocene age (approximately 1 million to 12,000 B.P.) (*Geology and Ground-Water Characteristics of the Hanford Reservation of the U.S. Atomic Energy Commission, Washington* [Newcomb et al., 1972]). Holocene surficial deposits of silt, sand, and gravel form a relatively thin veneer at the surface (*Geology and Hydrology of the Hanford Site: A Standardized Text for Use in Westinghouse Hanford Company Documents and Reports* [WHC-SD-ER-TI-003]; "Long History of Pre-Wisconsin, Ice Age Cataclysmic Floods: Evidence from Southeastern Washington State" [Bjornstad et al., 2001]).

The unconsolidated deposits at 100-D/H are underlain by Miocene-aged (approximately 17 to 8.5 million years B.P.) basalt of the CRBG that is interbedded with the Ellensburg Formation (*Geologic Setting of the 100-HR-3 Operable Unit, Hanford Site, South-Central Washington* [WHC-SD-EN-TI-132]). The CRBG may exceed 3,050 m (10,000 ft) in thickness locally including the interbedded sediments of the Ellensburg Formation. The Ellensburg Formation consists of a series of sedimentary units (epiclastic and volcaniclastic) that are interbedded with many of the basalt flows of the CRBG (*Revisions in Stratigraphic Nomenclature of the Columbia River Basalt Group* [USGS Bulletin 1457-G]).

The physical properties of these formations influence the distribution of contaminants in the subsurface. The Ringold Formation at 100-D/H includes several formational units including the Ringold Formation unit E, the RUM, the Ringold unit B, and the RLM. The Hanford formation comprises most of the vadose zone throughout 100-D/H. In addition, the Ringold Formation unit E is present in 100-D, where it comprises the unconfined aquifer and part of the vadose zone. Elsewhere in 100-D/H, the unconfined aquifer is composed almost entirely of Hanford formation sediments, including the area across the horn. Aquifer testing and slug testing data indicate that the horizontal hydraulic conductivity of the Hanford formation is generally three times greater than the Ringold Formation unit E; although variable degree of cementation can influence the transmissivity in both units (Tables 3-9 through 3-14; Appendix M). The RUM is present throughout 100-D/H, forming the base of the unconfined aquifer. The RUM appears to be thinnest near 100-H where the overlying Ringold Formation unit E has been eroded. The RUM includes interbedded sandy zones that form semiconfined (100-H only) to confined water-bearing units. These relationships are shown on Figure 3-3.

3.4.2.1 Surface Deposits

Recent deposits include eolian sands and river alluvium, which were placed over the past 10,000 years, and backfill materials deposited by humans. Construction backfill varies in depth, depending on the excavated depth of waste sites and building foundations, and backfill material may cover larger graded

areas to depths of 0.3 m (1 ft) or more. Backfill deposits may be up to 8 m (26 ft) thick near reactors and clearwells, but are generally less than 5 m (16 ft) thick in other areas. Because of anthropogenic activities associated with construction of the reactors and supporting facilities, the Holocene deposits were likely removed or altered because of extensive grading in the 1950s. Outside of those areas, the Holocene deposits are relatively thin (0.3 m [1 ft]) (*Geologic Setting of the 100-HR-3 Operable Unit, Hanford Site, South-Central Washington* [WHC-SD-EN-TI-132]).

3.4.2.2 Hanford Formation

The Hanford formation consists predominantly of unconsolidated sediments that cover a wide range of grain sizes, from boulder-sized gravel to sand, silty sand, and silt. The Hanford formation is an informal name used to describe these Pleistocene-age cataclysmic flood deposits (*Draft Environmental Assessment: Reference Repository Location Hanford Site, Washington* [DOE/RW-0017]). The Hanford formation facies consists of moderately to very poorly sorted, large to very large, cobble- to boulder-sized clasts in open framework gravels that include discrete sand lenses, with little, or no, silt and clay-sized material. The gravel-dominated Hanford formation is highly basaltic, ranging from approximately 50 to 80 percent basalt (*Geology of the Northern Part of the Hanford Site: An Outline of Data Sources and the Geologic Setting of the 100 Areas* [WHC-SD-EN-TI-011]). The sand fractions are also high in basalt content, with the remaining portion composed of feldspar, quartz, and traces of mica. The grains typically are subround to round gravel and subangular to subround in the sand grain fraction. The gravel-dominated facies typically are well stratified and contain little to no cementation (*Geologic Setting of the 100-HR-3 Operable Unit, Hanford Site, South-Central Washington* [WHC-SD-EN-TI-132]). Discrete sand lenses are present in 100-D/H, which may serve as preferential flow paths or collection zones for vadose zone contaminants. Caliche (calcium carbonate crust) is occasionally observed on Hanford formation gravels.

The Hanford formation has traditionally been classified into three separate lithofacies: gravel-dominated, sand-dominated, and interbedded sand and silt-dominated (*Standardized Stratigraphic Nomenclature for Post-Ringold-Formation Sediments Within the Central Pasco Basin* [DOE/RL-2002-39]). Beneath 100-D/H, only the gravel and sand dominated facies are present to depths of approximately 17 m (55 ft) bgs. Thicknesses range from 5 to 22 m (17 to 73 ft), with greatest thickness underlying the southwest-central part of 100-D/H. The unit generally thins to the north and east (Integrated Work Plan [DOE/RL-2008-46]).

3.4.2.3 Hanford/Ringold Contact

The erosional unconformity surface between the Hanford formation sediments and the underlying Ringold Formation sediment is referred to as the "Hanford formation/Ringold Formation contact". Hydrologic property differences exist across the Hanford/Ringold contact because of differences in the physical nature of the two units and actions of scouring by the paleofloods. Pleistocene-age cataclysmic glacial outburst floods have eroded into the older Ringold Formation sediment and removed and/or reworked some of the Ringold sediments.

Ringold Formation unit E is a denser, compact and well-graded formation versus the looser, coarser-grained Hanford gravel-dominated facies. The contact may be well defined in some locations but gradational in others, suggesting a mixing of materials near the end of a flood cycle. The pattern and flow path of these paleoflood channels are preserved in the topographic expression of the contact, and to some degree, in the surface topography.

To the east of 100-D, the Ringold Formation unit E is eroded away over much of the area, resulting in the Hanford formation laying unconformably on the RUM. There are locations where Ringold Formation unit E is present across the horn into 100-H, which indicates uneven erosion of that unit. Hanford

formation material that was deposited over the Ringold Formation erosional surface formed a contact surface (disconformity) between the two stratigraphic units. Since these two stratigraphic units have different lithologies, they also have unique hydraulic properties, which influence groundwater flow pathways and contaminant distribution as the contact is encountered. Because the overlying Hanford formation exhibits substantially greater hydraulic conductivity that the Ringold unit E or RUM Formations, the contact may retard the vertical migration of wastewater and precipitation down toward the aquifer and may contribute to lateral migration. Vertical hydraulic conductivity data, presented in detail in Section 3.6.1, indicates that the saturated hydraulic conductivity of the Hanford formation is greater than that of Ringold Formation unit E, with substantial local variability. In 100-D, the water table is generally below this contact. However, where the Hanford formation/Ringold Formation contact occurs below the water table, groundwater flows will tend to have a greater velocity within the Hanford formation portion with water migrating more slowly through Ringold Formation unit E. This condition may occur in areas of the horn. At 100-H, the Ringold Formation unit E is essentially absent.

The Hanford/Ringold contact may also affect groundwater discharge to the Columbia River. During operation of the reactors at 100-D and 100-H, groundwater elevations were substantially raised because of leakage and intentional discharges of reactor cooling water to the ground near the reactors. Seepage of groundwater along the Columbia River shoreline was been documented at 100-D and 100-H as early as 1963 (*Status of the Ground Water Beneath Hanford Reactor Areas January, 1962 to January, 1963* [HW-77170]). At 100-D, thermal springs were present along 609.6 m (2,000 ft) of riverbank, indicating a preference for horizontal flow instead of vertical flow where the hydraulic conductivity of the stratigraphic units changed. Thermal springs were also present at 100-H, extending 914.4 m (3,000 ft) along the riverbank. The groundwater at 100-D/H (as well as at the other 100 Area reactors) exhibited substantially elevated temperature because of the releases of near-boiling spent cooling water from the reactors. Groundwater seepage in these locations was still observed in 1991 (*Riverbank Seepage of Groundwater Along the 100 Areas Shoreline, Hanford Site* [WHC-EP-0609]) near both 100-D and 100-H, although less common. The recent groundwater seepage conditions are associated with re-equilibration of groundwater elevations following periods of seasonal high river state and temperatures have returned to typical background conditions.

Where the Hanford formation/Ringold Formation unit E contact occurs below the river level (approximately 115 m [364 ft] average elevation in 100-D/H) or the water table, it may form a preferential hydrogeologic flow path. This flow path can transport groundwater to other portions of the saturated Hanford formation/Ringold Formation contact. In 100-D, where substantial deposits of the Ringold Formation unit E are present, the Hanford formation/Ringold Formation unit E contact is predominantly above the water table.

3.4.2.4 Ringold Formation

The Miocene- to Pliocene-age (8.5 to 3.4 m.y. B.P.) Ringold Formation is a combination of alluvial and lacustrine deposits produced by the ancestral Columbia River and other regional river systems. Across the Hanford Site, the Ringold Formation is as much as 185 m (606 ft) thick. The Ringold Formation consists of nonindurated and semi-indurated clay, silt, fine- to coarse-grained sand, and variably cemented, multilithic, granule to cobble gravel. Ringold Formation sediments have been classified into five sediment facies associations, including fluvial gravel, fluvial sand, overbank deposits, lacustrine deposits, and alluvial fan deposits (*Geologic Setting of the 100-HR-3 Operable Unit, Hanford Site, South-Central Washington* [WHC-SD-EN-TI-132]; *Geology of the Northern Part of the Hanford Site: An Outline of Data Sources and the Geologic Setting of the 100 Areas* [WHC-SD-EN-TI-011]).

The typical stratigraphic units within the Ringold Formation are generally identified, from shallowest to deepest, as: unit E, RUM (confining layer), units B and C, Ringold lower mud, Ringold Formation gravel

unit A. Beneath 100-D/H, the Ringold Formation does not contain all of the commonly encountered stratigraphic units found elsewhere across the Hanford Site.

Ringold Formation unit E. The Ringold Formation unit E is the youngest Ringold age sediment present in the area; it occurs consistently in the 100-D Area and directly overlies the RUM. In the horn and 100-H Area, the presence of Ringold Formation unit E sediment is limited. The Ringold Formation unit E is composed of fluvial matrix-supported gravels and sands with intercalated fine- to coarse-grained sand and silt layers. Ringold Formation unit E lithology is between 35 and 90 percent felsic consisting mainly of metamorphic, intermediate volcanic, and felsic volcanics (*Geology of the Northern Part of the Hanford Site: An Outline of Data Sources and the Geologic Setting of the 100 Areas* [WHC-SD-EN-TI-011]). Micaceous sand is occasionally encountered. Grain-size distributions tend to be bimodal, with granule and coarse sand fractions generally absent. Cementation of the Ringold Formation unit E increases to the south and west in 100-D. The Ringold Formation unit E generally pinches out between the northeast portion of 100-D and the horn, with occasional sporadic occurrences identified in the horn, but is generally not present beneath 100-H.

Ringold Formation upper mud unit. The RUM underlying Ringold Formation unit E gravels are dominated by a fine grained overbank paleosol facies association that is up to 61 m (200 ft) thick (*Geologic Setting of the 100-HR-3 Operable Unit, Hanford Site, South-Central Washington* [WHC-SD-EN-TI-132]). Regionally, the RUM is essentially continuous with an undulating surface with depressions and topographic highs that locally affect aquifer thickness. It appears to dip to the north or be partly eroded in the most northwestern portion of the horn area. The top of the RUM ranges between elevation 104.5 and 115 m (342.8 and 377.3 ft) (*North American Vertical Datum of 1988* [NAVD88]). Boreholes drilled to basalt in 100-N and 100-H suggest that about 61 m (200 ft) of overbank-paleosol strata and 23 to 30.5 m (76 to 100 ft) of lacustrine deposits lie beneath the horn area (*Geologic Setting of the 100-HR-3 Operable Unit, Hanford Site, South-Central Washington* [WHC-SD-EN-TI-132]).

The upper part of the RUM sometimes contains gravel in a silt/clay matrix that represents a transition zone (reworked interval) above the more massive silt or clay. The silt and clay rich RUM has low hydraulic conductivity values relative to the Hanford formation and Ringold Formation unit E. The RUM is considered an aquitard and forms the base of the unconfined aquifer. Within the RUM, thin sand-to-gravel layers form zones with variable hydraulic conductivities that range from low to high and form confined or semiconfined aquifers (Section 3.6.1). Beneath a localized area of 100-H inland from the reactor, a shallow water-bearing unit within the RUM has been shown to be potentially hydraulically connected to the unconfined aquifer (Section 3.6.1), which could provide a pathway for contaminants to migrate. The top surface of the RUM is found between 28 and 33 m (91 and 109 ft) bgs near 100-D and between 11 and 40 m (37 and 66 ft) bgs near 100-H.

The surface topography of the RUM is presented on (Figure 3-4). This map was constructed using historical borehole data and includes RUM surface data obtained from the RPO and RI boreholes/wells. The map displays the RUM surface elevation in 1 m (3.3 ft) contour intervals. Individual RI borehole RUM elevation data points are listed in Table C-5 of Appendix C. This data was also used to develop hydrogeologic cross sections and surface contour maps (see Sections 3.6 and 4.5.2, and Appendix M). The maps and cross sections identify scour features in the RUM surface that are the result of river channel migration and/or glacial floods that ultimately laid down the Hanford formation. These scour features are also expressed east of the abrupt drop in surface elevation at the eastern edge of 100-D, coincident with the Ringold Formation unit E pinchout/erosional truncation zone (*Geologic Setting of the 100-HR-3 Operable Unit, Hanford Site, South-Central Washington* [WHC-SD-EN-TI-132]). The subsurface structure indicates north to northeast trending channel scouring in the RUM unit that is roughly parallel to the modern Columbia River.

The RUM surface exhibits varying complexity which may be related to the amount of available information. The higher density of boreholes near 100-D and 100-H provide for a more detailed delineation of channels and surface undulations in comparison to the horn. There are 3 to 4 m (10 to 13 ft) deep scour channels eroded into the surface at 100-D and 100-H that indicate the migration of the main channel of the Columbia River to the east over time. The northern tip of the horn area exhibits significant erosion of the RUM in the near-shore area where the Columbia River bends to the southeast.

Other characteristics of the RUM surface (Figure 3-4) include the undulating surface and localized depressions, which exhibit a horseshoe-like feature at 100-D. Higher elevations in the RUM surface are present close to the Columbia River, trending parallel and adjacent to the river. This ridge feature can be seen in cross section B to B' (Appendix M, Figure M-6) where the rise in the RUM surface is apparent at Well 199-D8-89, with the corresponding depression further inland at Wells 199-D5-134 and 199-D5-140.

A high RUM surface also appears to be northeast of 100-D. Depressions are at several locations. One depression is north of the 100-D Area, and another appears to circle the inland side of the D/DR Reactors. A third, less noticeable but possibly more important depression is between the river and the Cr(VI) plume hot spots. A portion of the third depression extends from near the 100-D south plume hot spot toward the north end of the ISRM barrier. This depression may have some effect on the shape of the south plume, contributing to the extension of the Cr(VI) plume to the west at Well 199-D3-5 (see Chapter 4).

In the central horn area, a broad high area of RUM is observed, corresponding to an area of relatively thin aquifer. Two low areas can be seen at 100-H, both trending from the northwest to the southeast. One shallow channel in the RUM extends northwest just west of H Reactor, and can be seen near 100-H (cross section D to D'; Appendix M, Figure M-8). The second depression, which may be related to the Cr(VI) distribution in the first water-bearing unit of the RUM, parallels the river along the shoreline.

Localized topographic highs and lows are identified on the RUM surface based on borehole data. Of particular interest in 100-D is the RUM depression just west of the D and DR Reactors that is approximately 2 m (6 ft) deep, which may provide a preferential groundwater flow path in a direction nearly parallel to the river.

Ringold Formation unit B and Ringold Formation lower mud unit. The Ringold unit B separates and differentiates the fine-grained sediment of the RUM from the underlying fine-grained sediment of the RLM. Fine sand to silty sand deposits of the Ringold unit B overlie the RLM unit and are approximately 15 to 24.5 m (50 to 80 ft) thick beneath 100-D/H. These Ringold unit B sands are inferred to be equivalent to fluvial gravel deposits of unit B (and possibly unit D) to the south in the Cold Creek Syncline. Ringold units A and C, which are present in other parts of the Cold Creek Syncline to the south of Gable Mountain, have not been found beneath 100-D/H. The RLM consists of fine-grained (silt- and clay-dominated) deposits (*Geologic Setting of the 100-HR-3 Operable Unit, Hanford Site, South-Central Washington* [WHC-SD-EN-TI-132]) that are approximately 23 to 30.5 m (75 to 100 ft) thick beneath 100-D/H.

3.4.2.5 Columbia River Basalt Group and Ellensburg Formation

The identified basalt flows of the CRBG number approximately 300 and maximum total thickness is approximately 4,600 m (15,000 ft) in the Pasco Basin. The CRBG erupted in the Miocene Epoch (17 to 8.5 million years ago) and has been divided into four formations from youngest to oldest: Saddle Mountain Basalt, Wanapum Basalt, Grand Ronde Basalt, and Imnaha Basalt (*Geology and Hydrology of the Hanford Site: A Standardized Text for Use in Westinghouse Hanford Company Documents and Reports* [WHC-SD-ER-TI-003]).

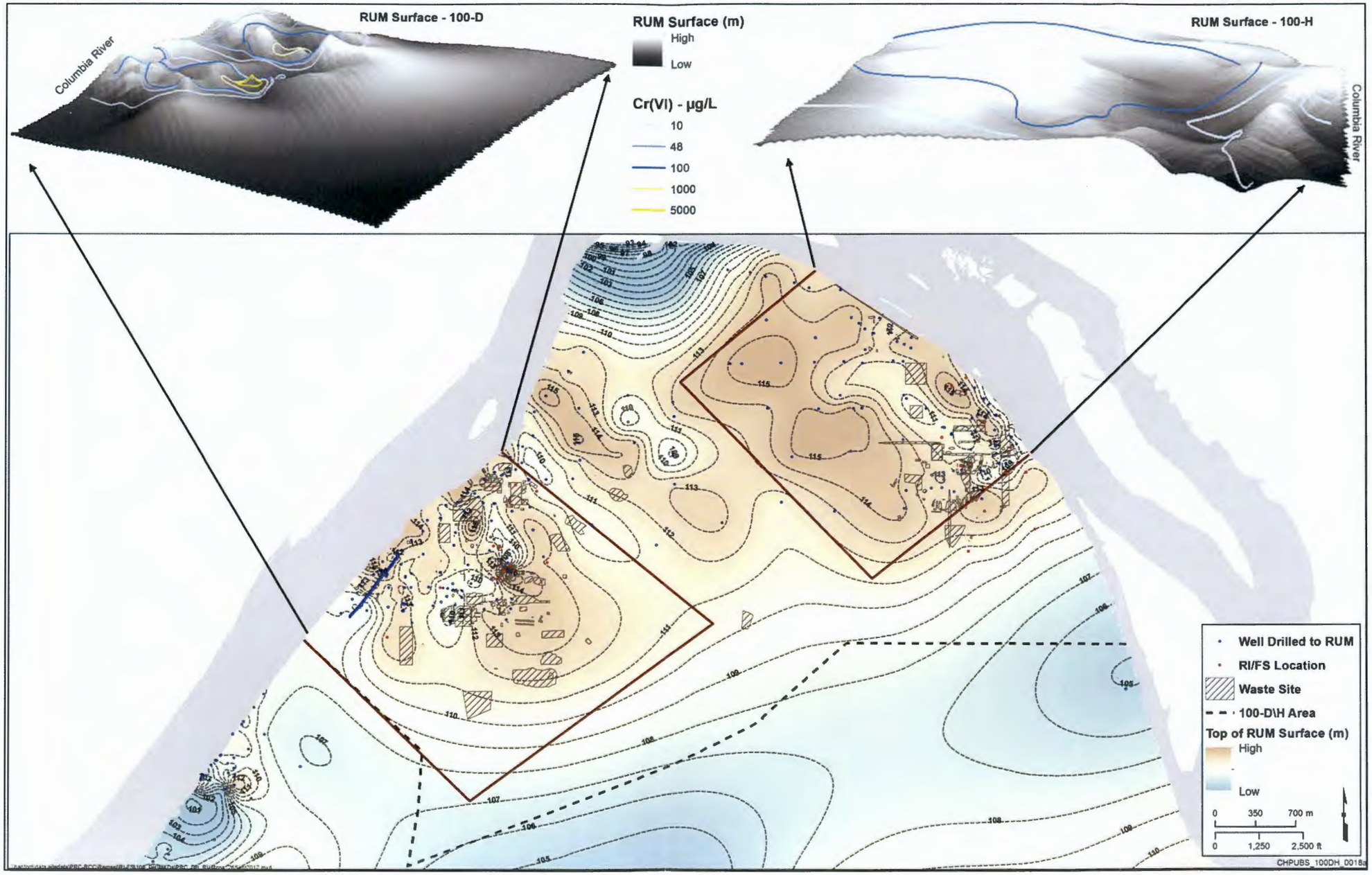

Figure 3-4. 100-D/H RUM Surface Elevation

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Sedimentary interbeds of the Ellensburg Formation occur between basalt flows. These interbed sediments (tuffaceous sands, silts, and clays) and the porous/fractured basalt flow tops and flow bottoms, form confined "interflow" aquifer zones that may extend across the Pasco Basin (*Site Characterization Plan: Reference Repository Location, Hanford Site, Washington* [DOE/RW-0164]).

The Elephant Mountain Member of the Saddle Mountain Basalt Formation is the upper basalt unit beneath 100-D/H (*Geologic Setting of the 100-HR-3 Operable Unit, Hanford Site, South-Central Washington* [WHC-SD-EN-TI-132]).

3.4.3 Hydrogeologic Cross Sections

Geological characterization and physical and hydraulic property data needs were identified to support development and refinement of the CSM and modeling for 100-D/H. Soil and groundwater data obtained from 27 RI boreholes were incorporated with existing data to further the understanding of the stratigraphy, hydrogeology, and contaminant mobility through the vadose zone and within the unconfined and semiconfined aquifers. Characterization data collected for the RI are described in Chapter 2.

Hydrogeologic cross sections, surface contour maps, and isopach maps of 100-D/H present detailed representations of the 100-D/H geology. Sufficient information exists to define the unconfined aquifer system in 100-D/H, as the majority of boreholes were drilled to confirm the depth to the RUM that forms the base of the unconfined aquifer. Figure 3-5 presents the trend lines used to construct five hydrogeologic cross sections, and includes the locations of the 17 RI wells. Hydrogeologic cross sections are presented in Appendix M, with analytical data presented on cross sections in Section 4.5.2.

Hydrogeologic information about the Ringold units below the RUM is far more limited than for the Hanford formation and Ringold unit E sediments. Several wells at 100-H drilled in the 1990s were deep enough to provide information on the RUM and a deeper water bearing zone, presumed to be Ringold Formation unit B. The horn study (*Hydrogeological Summary Report for 600 Area Between 100-D and 100-H for the 100-HR-3 Groundwater Operable Unit* [DOE/RL-2008-42]), RPO effort (*Sampling and Analysis Plan for Installation of 100-HR-3, Groundwater Operable Unit Remedial Process Optimization Wells* [DOE/RL-2009-09]), and the RI characterization improved understanding of the relationships and properties of these units.

Cross section A to A' (Figure M-5) is located in the southern portion of 100-D and trends from the Columbia River inland towards the east. The cross section shows the unconfined aquifer being present within the Ringold Formation unit E, and relatively thin. The undulating surface of the RUM is also evident, with a scour depression in the RUM surface present at Wells 199-D5-141 and 199-D5-104. The RUM surface rises slightly toward the river in this area, which may act as an impediment to contaminant migration. Not evident in the cross section, but shown in plan view in Figure 3-4, is the dip in the RUM surface in the southern portion of 100-D (coinciding with Well 199-D3-5), where the surface has a downward slope towards the south.

Cross section B to B' (Figure M-6) begins at the Columbia River and extends to the east, through northern 100-D. Several depressions occur in the RUM surface along this cross section. The dip in the RUM extends across a wider area in cross section B to B' and is more pronounced than in cross section A to A'. As in cross section A to A', the RUM surface rises as it gets closer to the river.

Hydrogeologic cross section C to C' (Figure M-7) runs parallel to the Columbia River from the ISRM well locations towards the horn area in the north. Cross section C to C' shows the unconfined aquifer completely within the Ringold Formation unit E near 100-D, with the RUM forming the bottom of the unconfined aquifer. As the cross section extends to the north, the Ringold Formation unit E is no longer identified in boring logs, with the aquifer being present entirely in Hanford formation material, which is

more conductive than the Ringold Formation unit E. The abrupt change in geology near Well 199-D8-55 and Well 199-D8-68 aligns roughly with the southern edge of surface outburst channels during historic flood events (Figure 3-1) that crossed the horn, eroding the Ringold Formation unit E. In addition, the outburst channels may have developed preferential pathways across the horn, resulting in the wide distribution of Cr(VI) during reactor operations through this zone. This cross section also shows the undulation of the RUM surface through the area, and a significant drop in elevation at the horn.

Cross section D to D' (Figure M-8) spans 100-D/H, beginning just north of 100-D and extending across the horn to 100-H. The RUM surface across the horn has a slight rise near 100-H but otherwise unremarkable. The unconfined aquifer occurs within the Ringold Formation unit E at 100-D, but transitions to the Hanford formation across the horn and at 100-H, with some few exceptions. The transition of the aquifer matrix from Ringold Formation unit E to the Hanford formation likely facilitated the lateral spread of Cr(VI) across the horn area and 100-H, because of greater hydraulic conductivity values of the Hanford formation. The spreading would have dramatically increased with the significant groundwater mounding that was caused by reactor operations and the 1967 cooling water injection test.

Running parallel to the Columbia River at 100-H is cross section E to E' (Figure M-9). Consistent with the 100-H portion of cross section D to D', the unconfined aquifer is present in the Hanford formation. This cross section shows several wells with screens completed in a confined transmissive lens (fine sand) that occurs within the RUM unit. The cross section also indicates that the unconfined aquifer is very thin at 100-H.

3.4.4 Remedial Process Optimization Wells

Seventy wells were drilled in 2009 and 2010 to enhance efficacy of the remedial action following the 2008 remedial process optimization (RPO) analysis of the HR-3 and DR-5 interim action pump-and-treat systems. The RPO identified the systems as undersized relative to the extent of the Cr(VI) plumes. The 70 wells were drilled within 100-D, across the horn, and in 100-H. The sampling and analysis plan is described in *Sampling and Analysis Plan for Installation of 100-HR-3 Groundwater Operable Unit Remedial Process Optimization Wells* (DOE/RL-2009-09). The RPO sampling is briefly summarized in Section 2.1.8. Physical and chemical data collected from RPO boreholes and wells are incorporated into isopach maps, surface contour maps, hydrogeologic cross sections, and modeling.

3.5 Vadose Zone

This section describes the general characteristics of the vadose zone underlying 100-D/H. The vadose zone (unsaturated zone) extends from ground surface to the water table of the uppermost aquifer. The hydraulic and chemical properties of this region control the downward movement of liquids and contaminants released near ground surface. Also called the zone of aeration, it includes the soil at the surface, the capillary fringe zone above the principal water-bearing zone, the periodically rewetted zone, and the combined rock, soil, air, and moisture interface linking the two. As the water table fluctuates in response to river stage and changes in recharge rates, the periodically rewetted zone experiences either wet or drying conditions. The capillary fringe is the edge of that wetted surface, where water seeps into the vadose zone material because of tension saturation. The thickness of the capillary fringe is typically small in sand and gravel formations (e.g., a centimeter or two), whereas the periodically rewetted zone in areas near the river at 100-D/H may be as much as 2 m (6 ft) thick.

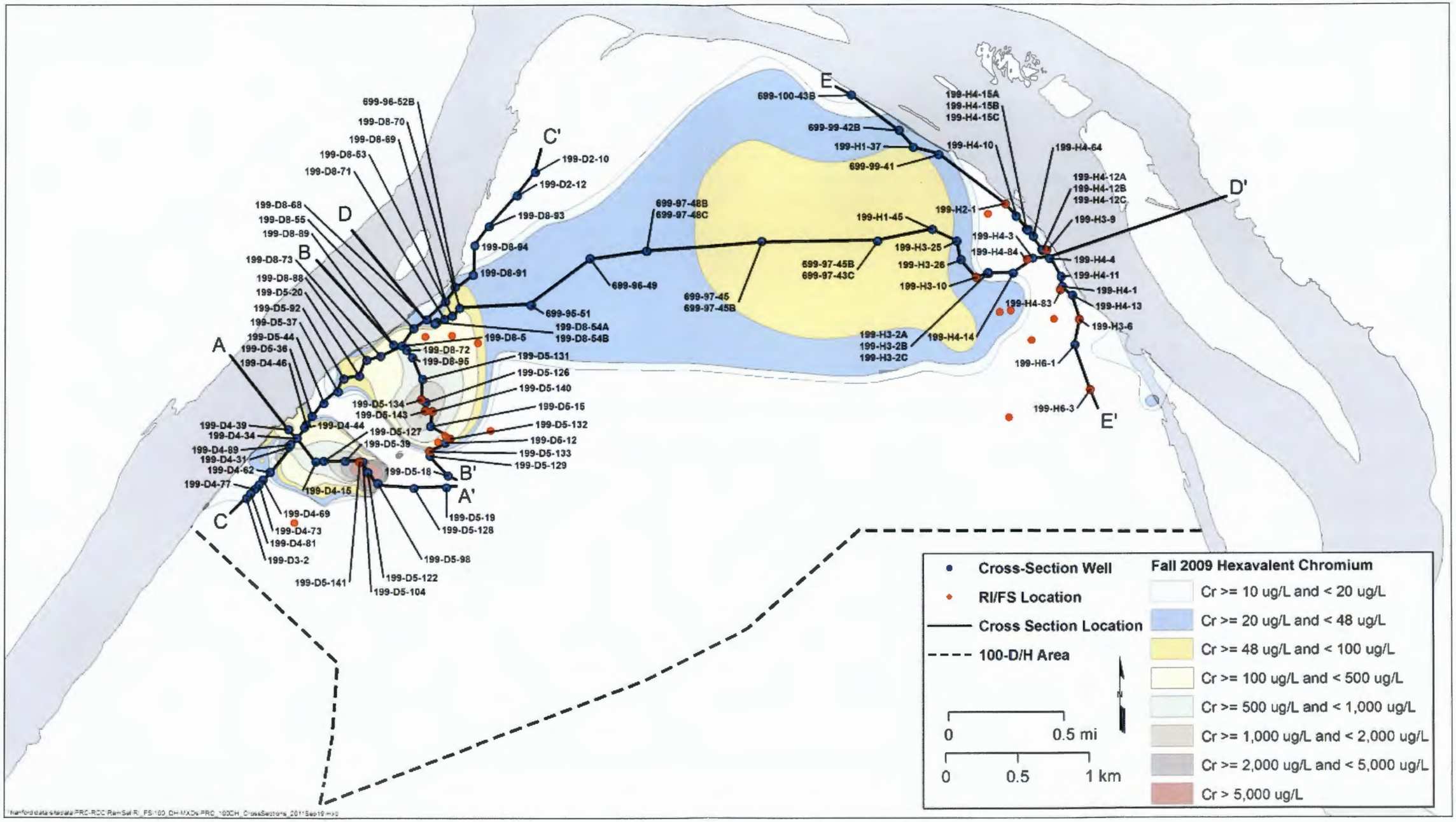

Figure 3-5. Trendline Location Map for 100-D/H Hydrogeologic Cross Sections and RI Wells

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The dominant stratigraphic unit in the vadose zone underlying 100-D/H is the Hanford formation. In 100-D/H, the upper part of the vadose zone to depths from 0.3 to 13.7 m (1 to 45 ft), has been disturbed in a nonuniform fashion by site grading and construction activities in the mid to late 1940s, by site operations between 1945 and 1967, and by waste site remediation and facilities decommissioning activities since reactor shutdown. The D, DR, and H Reactor areas were cleared of vegetation and regraded. Select areas away from the reactors were stripped and graded to support specific facilities, but outside the construction areas the existing plant community and soil profiles were not disrupted, especially within the horn area.

The vadose zone at 100-D includes surficial soil, Hanford formation sediments, and Ringold Formation unit E sediments. In 100-H, the Ringold Formation unit E is absent. There are pockets of Ringold Formation unit E across the horn. An investigation of the physical and geochemical properties of the vadose zone in the 100 Area is presented in *Geochemical Characterization of Chromate Contamination in the 100 Area Vadose Zone at the Hanford Site* (PNNL-17674).

Vadose zone thickness, which also represents the depth to groundwater, ranges from 0 to 27 m (0 to 89 ft), with an average thickness of 20 m (65.4 ft) in 100-D and an average thickness of 11.3 m (37.1 ft) in 100-H. Across 100-D/H, the vadose zone is typically thinner near the Columbia River and has extensive topographic variability. The surface topography has a significant drop in the north, closer to the horn, consistent with the locations of the outwash channels.

Water that infiltrates the ground surface is either retained by capillary forces or passed downward toward the water table, as gravitational flow. Movement of moisture in the vadose zone is influenced by overall soil moisture content, the hydraulic properties of soil, vegetation cover, and timing of precipitation events.

3.5.1 Soil Types

Holocene deposits of eolian loess, silt, sand, and gravel form surficial deposits across 100-D/H. These deposits overlie the Hanford formation in a relatively thin (less than 1 m [3 ft] thick) veneer in most locations. During the past 10,000 years, a mix of eolian and alluvial processes deposited this soil. In some portions of 100-D/H, the surface is reworked construction backfill. This backfill material typically consisted of Hanford formation gravel, sometimes mixed with construction debris. Debris pits and piles created during construction have generally been addressed as waste sites. Recent (1995 to present) backfill practices rely almost exclusively on excavated Hanford formation gravel or fill imported from local or offsite borrow pits. This backfill is generally located near existing or former manmade structures and varies in depth, depending on the excavation depth of waste sites and building foundations. Additionally, backfill may cover larger graded areas to a depth up to 0.3 m (1 ft). Because of human activities associated with construction of the reactors and supporting facilities, the Holocene deposits may have been removed or altered. The key waste sites in the operational areas are shown on Figures 2-1 and 2-2 and the extent of disturbed soil is visible in the areal views shown on Figures 2-5, 2-6, 2-7, and 2-8.

Soil Survey Hanford Project in Benton County Washington (BNWL-243) describes 15 soil series on the Hanford Site, which consist of sand, sandy loams, and silty loams. The following five soil series are present within 100-D/H (Figure 3-6):

- **Burbank Loamy Sand.** Burbank loamy sand is a dark-colored, coarse-textured soil underlain by gravel. Its surface soil is usually about 40 cm (16 in.) thick but may be as much as 75 cm (30 in.) thick. The gravel content of its subsoil ranges from 20 to 80 percent.

- **Ephrata Sandy Loam.** Ephrata sandy loam is found on level topography on the Hanford Site. Its surface is darkly colored and its subsoil is dark grayish-brown, medium-textured soil underlain by gravelly material that may continue for many = feet.

- **Ephrata Stony Loam.** Ephrata stony loam is similar to Ephrata sandy loam. It differs by the presence of many large hummocky ridges that consist of debris from melting glaciers. Areas between hummocks may contain many boulders several feet in diameter.

- **Riverwash.** Riverwash consists of the wet and periodically flooded areas of sand, gravel, and boulders adjacent to the Columbia River.

- **Rupert Sand.** Rupert sand is brown to grayish brown coarse sand. The color grades to dark grayish brown at a depth of about 90 cm (35 in.). Rupert soil typically develops under grass, sagebrush, and hopsage in coarse sandy alluvial deposits that are mantled by windblown sand. The relief characteristically consists of hummocky terraces and dune-like ridges. The soil is correlated as Quincy sand from an earlier survey.

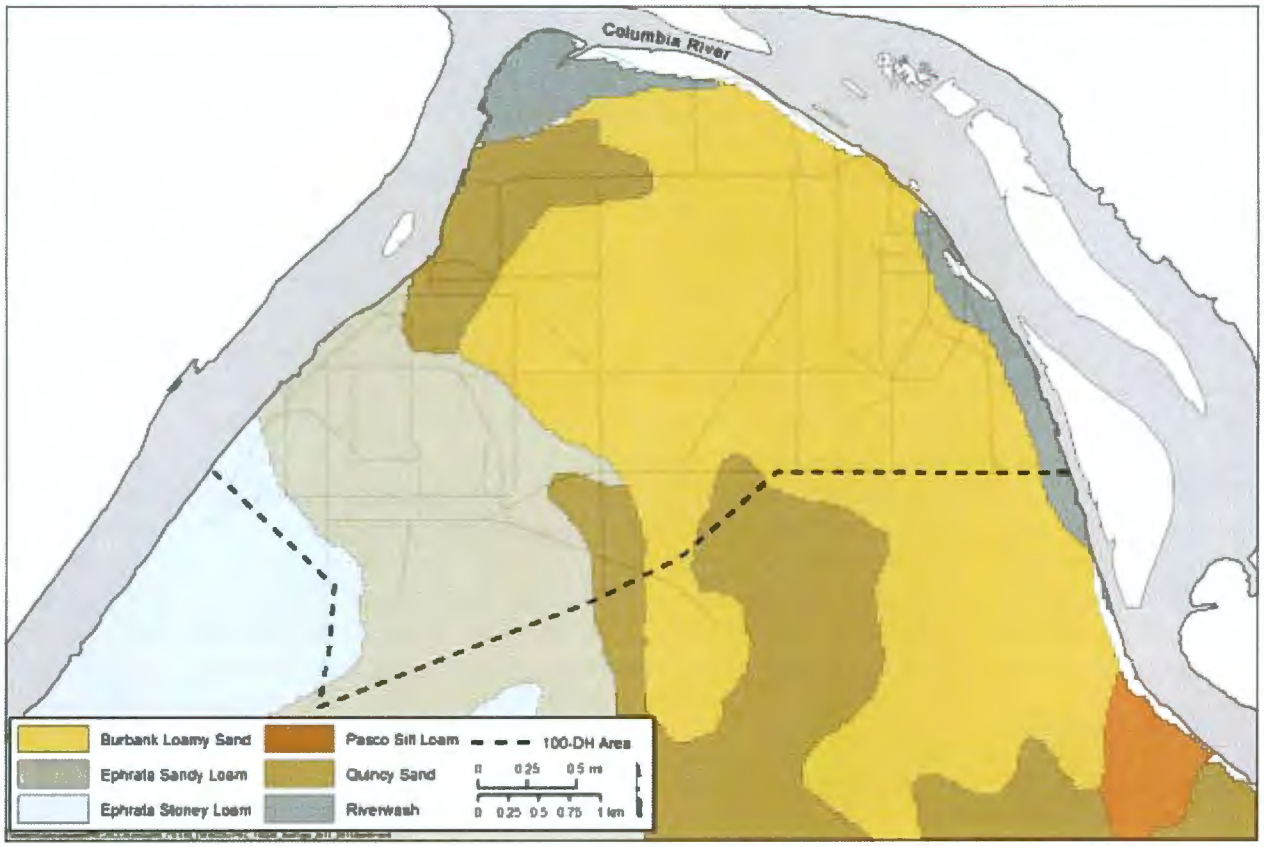


Figure 3-6. Soil Series in the Vicinity of 100-D/H Areas

The Burbank Loamy Sand and the Ephrata Sandy Loam cover around 70 percent of 100-D/H. Ephrata Sandy Loam covers the majority of 100-D, with an area of Ephrata Stony Loam along the west boundary and areas of Rupert (Quincy) Sand near the northeast and southeast boundary. 100-H is primarily Burbank Loamy Sand, with an area of Riverwash along the east boundary. The horn is primarily Burbank Loamy Sand, with an area of Rupert (Quincy) Sand along the northwest boundary and an area of Riverwash along the north boundary. Many small areas of backfill exist where construction activities and interim remedial actions have been completed. Each of these soil types has different characteristics.

The characteristics of the soil, such as permeability, are critical to modeling and understanding the effects of infiltration and subsequent recharge to the aquifer. Tables 3-5 and 3-6 present recharge estimates applicable to different conditions for 100-D/H surface soil. For comparison, "Hanford Site Vadose Zone Studies: An Overview" (Gee et. al., 2007) notes a range from nearly zero to 100 mm/yr depending on the soil type and vegetation cover.

Table 3-5. Estimated Recharge Rate

Major Soil Type	Estimated Recharge Rate (mm/yr)			
	No Vegetation	Cheatgrass	Young Shrub-Steppe	Mature Shrub-Steppe
Rupert Sand	44	22	8	4.0
Burbank Loamy Sand	52	26	6	3.0
Ephrata Sandy Loam	17	8.5	3.0	1.5
Ephrata Stony Loam	17	8.5	3.0	1.5
Riverwash	92	46	N/A	N/A

Source: PNNl-14702, *Vadose Zone Hydrogeology Data Package for Hanford Assessments*, Table 4.15.

Table 3-6. Estimated Recharge Rates and Variation—Disturbed Conditions

Condition	Best Estimate (mm/yr)	Estimated Standard Deviation (mm/yr)	Minimum (mm/yr)	Maximum (mm/yr)
Rupert Sand	44	22	22	88
Burbank Loamy Sand	52	26	26	101
Ephrata Sandy Loam	17	8.5	8.5	34
Ephrata Stony Loam	17	8.5	8.5	34
Riverwash	92	46	46	101

Source: PNNL-14702, *Vadose Zone Hydrogeology Data Package for Hanford Assessments*.

3.5.2 Soil Moisture Variations

Unsaturated flow of moisture/liquid in the vadose zone is highly complex and influenced by the hydraulic properties of soil and vegetation cover. Movement of moisture in the vadose zone is mainly vertically downward under gravity drainage, controlled by the unsaturated hydraulic conductivity and the difference in hydraulic head between two points (that is, hydraulic gradient). In 100-D, the moisture content in the Hanford formation or Ringold Formation unit E vadose zone ranges from 1.0 to 10.0 percent. The soil moisture content in the vadose zone of the Hanford formation in 100-H ranges from 1.8 to 6.0 percent. The moisture content in the vadose zone is largely dependent upon the grain size distribution, with finer-grained zones able to retain more moisture because of their smaller pore size and greater number of pores available.

The Hanford formation comprises the majority of the vadose zone beneath 100-D/H, although a lower water table in 100-D results in the Ringold Formation unit E also becoming unsaturated. Unsaturated flow through the Hanford formation may be influenced by the depositional environment. The flood deposits that constitute the Hanford formation tend to fine upward within each depositional sequence, resulting in alternating coarser and finer grains vertically. Cross beds found in the Hanford formation may also locally influence vertical migration of soil water, though the magnitude and extent of influence is not known.

Much of the operational area within 100-D/H was denuded of the native plant and soil cover, optimizing conditions for deep percolation of precipitation through the vadose zone. During historical operations, water was intentionally discharged to the ground surface, and under both historical and ongoing remedial actions, water may be applied to the ground surface for dust control. Under current conditions, low-moisture conditions dominate vadose zone with the majority of moisture resulting from infiltration of natural precipitation. However, periodic elevation of the water table because of fluctuations in river stage, leaks from site infrastructure (for example, 181-D River Pump House, 182-D Reservoir, and pipelines), and local application of dust suppression water during remedial actions can also contribute to increased soil moisture.

Within 100-D/H, vadose zone thickness varies because of natural and anthropogenic influences, such as changes in Columbia River stage as well as changes in topography. If groundwater is contaminated, mobile contaminants may be introduced into the basal vadose zone over large areas with the rising water table. With reduction in artificial recharge, precipitation is the main source of recharge; however, fluctuations in river stage and the flux from artificial recharge may affect the fate of contaminants. In the vadose zone, the pressure head is negative under unsaturated conditions (*200 Areas Remedial Investigation/Feasibility Study Implementation Plan – Environmental Restoration Program* [DOE/RL-98-28]). This reflects the fact that water in the unsaturated zone is held in soil pores under negative pressure by surface tension forces (*200 Areas Remedial Investigation/Feasibility Study Implementation Plan – Environmental Restoration Program* [DOE/RL-98-28]). If the volume of water in the vadose zone is less than the volume that can be retained by surface tension forces (field capacity), no water is available to migrate. Typically, this is the condition in the vadose zone under the native shrub-steppe vegetation on a fully developed soil profile. When this vegetation/soil cover is disturbed, transpiration is essentially zero because of lack of vegetation; consequently, a larger percentage of precipitation and any anthropogenic water are able to infiltrate into the vadose zone readily and, if a sufficient volume is present, migrate to the water table. Physically, as additional water is added to the vadose zone, it will migrate vertically under the force of gravity when the moisture content exceeds that which can be retained by the soil capillary forces.

3.5.3 Physical Soil Properties

During the RPO and RI/FS field investigations, numerous soil samples were collected within the Hanford formation, Ringold Formation unit E, RUM, first water-bearing unit in the RUM, and Ringold Formation unit B for evaluation of physical properties such as particle size, percent moisture, bulk density, and calculated porosity. Sample details are included in Appendix M. Samples included were collected from the vadose zone, aquifer matrices, and aquitard(s), with each stratigraphic formation clearly identified. Samples are listed in order, from shallow to deep, in order to show the variance in physical properties over the soil profile.

Particle size analysis was performed in accordance with *Standard Test Method for Particle-Size Analysis of Soils* (ASTM D422-63). Sediment moisture content was determined in accordance with *Standard Test Methods for Laboratory Determination of Water (Moisture) Content of Soil and Rock by Mass* (ASTM D2216-05). Density was determined in accordance with *Standard Test Method for Density of Soil*

in Place by the Drive-Cylinder Method (ASTM D2937-04) and is reported both as wet density and as dry density. Porosity is a calculated value determined by the following equation and reported as a percent.

$$Porosity = 1 - \frac{Bulk\ Density}{Particle\ Density}$$

For purposes of calculating porosity, normal particle density is assumed to be 2.65 g/cm^3 (165.434 lb/ft^3), the approximate particle density of quartz.

The following bullet summarizes the results of physical testing of the vadose zone samples:

- **Grain size:** The majority of the vadose zone consists of sandy gravels. However, grab samples collected from test pits at depths less than 6.1 m (20 ft) bgs often have more sand content than deeper soil. These samples may represent reworked soils with backfill, considering samples were collected in waste sites that had limited excavations conducted. Occasionally, sand layers are present in the vadose zone at greater depths (for example, Wells 199-D5-134 and 199-D5-132 had 67 to 70 percent sand at depths from 16.8 to 17.7 m [55 to 58 ft] bgs).

PNNL conducted a historical study (*Vadose Zone Hydrogeology Data Package for Hanford Assessments* [PNNL-14702]) on bulk densities in the Hanford formation versus the Ringold Formation unit E, which are used for comparison to recent RI and RPO physical property data (Table 3-7). Bulk density is used in vadose zone fate and transport calculations. The site wide statistical mean values were essentially identical for the Hanford formation and the Ringold Formation unit E.

Table 3-7. Statistical Mean Values for Site Wide Sample

Soil Class	Sample Size	K_s (cm/sec)	% gravel	Bulk Density	
				g/cm^3	kg/m^3
Hanford silty sand	38	8.58E-05	0.2	1.61	1610
Hanford fine sand	36	3.74E-04	0.6	1.60	1600
Hanford coarse sand	81	2.27E-03	2.6	1.67	1670
Hanford gravelly sand	16	6.65E-04	25.8	1.94	1940
Hanford sandy gravel	28	3.30E-04	51.4	1.93	1930
Hanford gravel	40	1.46E-03	67.6	1.97	1970
Ringold Formation gravel	18	4.13E-04	46.1	1.90	1900

Source: PNNL-14702, *Vadose Zone Hydrogeology Data Package for Hanford Assessments*.

3.6 Hydrogeology

The understanding of the hydrogeologic framework of 100-D/H is based on subsurface investigations conducted during the operational phase of the reactors up through the interim remedial actions and the current RI/FS. The three main hydrogeologic units include the following:

- Vadose zone (discussed in Section 3.5)
- Suprabasalt aquifer system
- Confined basalt aquifer system

This section describes the saturated hydrogeology of 100-D/H, beginning with descriptions of the main aquifer and aquitard units of the suprabasalt aquifer system. This system includes all sediments between the water table and the top surface of the basalt. The structure of the aquifer system is one of the controlling factors for groundwater flow between the various aquifers, aquitards, and the Columbia River, which forms a regional discharge boundary for shallow groundwater beneath the Hanford Site.

At 100-D/H, the unconfined aquifer is the zone between the water table and the surface of the RUM. At 100-D, the unconfined aquifer is present in the Ringold Formation unit E, and at 100-H, it is present in the Hanford formation, since the Ringold Formation unit E is absent. Within the RUM, there are several zones of sand and gravel which are water-bearing units. These water bearing units may be connected to each other, to the unconfined aquifer, or to the Columbia River. The extent of this aquifer interconnection varies spatially across 100-D/H and is may have been temporally dependent on the overlying hydrologic conditions, such as elevated head pressures which existed during operations because of high-volume cooling water discharges.

As presented in Section 3.4.1, the stratigraphic units identified within the Ringold Formation include the Ringold Formation unit E, the RUM, the Ringold unit B, and the RLM. Aquifers found below the upper surface of the RUM are typically confined or semiconfined, but leakage between the units may also occur. In addition, these various units may not continuous in all locations making them difficult to differentiate during drilling activities. The hydrostratigraphy of the suprabasalt aquifer system underlying 100-D/H is summarized in Table 3-8.

The hydraulic properties of an unconsolidated water-bearing unit are greatly influenced by the lithology of the aquifer matrix, with sand and gravel deposits generally being more transmissive than silt and clay units. However, poorly sorted sand and gravel deposits that contain intercalated silts and clays will see a significant reduction in hydraulic conductivity over well-sorted sand and gravel deposits because the pores are filled with fines, thereby reducing the interconnectedness of pores. Hydraulic conductivity is generally defined as the flow volume over time through a cross-sectional area (presented in cm/sec, m/day, or ft/day). Variations in the hydraulic conductivity of the aquifer matrix directly affect the ease with which groundwater flows through the unit, which affects the ability of contaminants to migrate (see Chapter 5). Because of the lower energy depositional environment, the RLM has the lower vertical hydraulic conductivity of the two aquitards (*Geologic Setting of the 100-HR-3 Operable Unit, Hanford Site, South-Central Washington* [WHC-SD-EN-TI-132]).

3.6.1 Unconfined Aquifer

Deposits comprising the unconfined aquifer at 100-D/H include the Hanford formation and Ringold Formation unit E. The thickness of the unconfined aquifer is determined by the difference between the water table elevation and the surface of the RUM, which forms the base of the unconfined aquifer. Aquifer thickness is greatest where deep scour channels occur in the RUM. An isopach map (showing aquifer thickness) is presented on Figure 3-7, using 1 m (3.3 ft) contour intervals. This map was created by subtracting RUM surface elevations from water table elevations measured from August through September 2010, and includes water table elevations from recent RI and RPO wells. Areas of darker blue indicate places where aquifer thickness is greatest and generally correspond to channels in the RUM surface.

Table 3-8. Hydrostratigraphy of the Suprabasalt Aquifer System at 100-D/H

Geologic Unit	Hydrostratigraphic Description	Principal Facies/ Principal Sediment Type	Approximate Range of Saturated Thickness (m [ft])
Hanford formation	Unconfined aquifer in 100-H and the eastern portions of the horn	Cataclysmic flood deposits/sandy gravel, loose	0 to 9 (0 to 30)
Ringold Formation unit E	Unconfined aquifer in 100-D and the western portion of the horn	Medium to high energy fluvial deposits/sandy gravel to sand, weakly to semiconsolidated	0 to 18 (0 to 60)
RUM	Aquitard defining the base of the unconfined aquifer	Paleosol and overbank deposits/sandy silt and silty clay, well-compacted	40 to 49 (130 to 160)
RUM, "first water-bearing unit"	First water-bearing unit (a confined to semiconfined water-bearing zone)	Low energy alluvial and paleosols/silty sand to fine sand, single-grained structure forming an interbed within the otherwise compact, fine-textured RUM	1 to 9 (3 to 30)
Ringold Formation unit B	Confined aquifer	Low energy alluvial deposits and paleosols/sand, loose	15 to 24 (50 to 80)
RLM	Aquitard	Lacustrine deposits/silt and clay, massive and compact	23 to 38 (75 to 125)

The unconfined aquifer thickness at 100-D/H generally thins from west to east from 100-D toward 100-H. Thickness of the unconfined aquifer ranges from near 0 to 12 m (39 ft) across the area. The thickness of the unconfined aquifer mimics the topography of the RUM (*Hydrogeological Summary Report for 600 Area Between 100-D and 100-H for the 100-HR-3 Groundwater Operable Unit* [DOE/RL-2008-42]). Aquifer thickness is greater beneath 100-D, where the unconfined aquifer matrix consists solely of Ringold Formation unit E sediments. The unconfined aquifer matrix in the horn and 100-H Areas consists of Hanford formation sediments where Ringold Formation unit E sediments are typically absent because of erosion. The aquifer is also influenced by the river stage, which causes fluctuations in the water table. Areas closest to the river are most affected by these fluctuations, with the effect muted farther inland.

The location for the transition of the aquifer matrix from Ringold Formation unit E to the Hanford formation has been updated based on information collected during drilling of the 70 RPO wells and RI boreholes and wells. As shown on Figure 3-8, pockets of Ringold Formation unit E have been identified farther east and south than previously identified in *100-HR-3 Remedial Process Optimization Modeling Data Package* (SGW-40781). In addition, the location of isolated pockets of Ringold Formation unit E across the horn provides further evidence that this transition is an erosional feature. Recent RI drilling generally supports earlier observations that the more transmissive Hanford formation is the dominant aquifer matrix in most of the horn area and 100-H.

The change in aquifer matrix from 100-D toward the horn and 100-H is important hydrogeologically because Hanford formation sediments typically have higher horizontal hydraulic conductivity values than Ringold Formation unit E sediments. During reactor operations, the groundwater mound beneath the retention basins and cooling water trench at 100-D pushed water to the north and east into the Hanford

formation. Once groundwater entered the Hanford formation, it would be less likely to move back south into the less transmissive Ringold Formation unit E.

3.6.1.1 Horizontal Hydraulic Conductivity – Unconfined Aquifer

Saturated horizontal hydraulic conductivity data from wells drilled before 2009 as part of the interim remedial actions are provided in *100-HR-3 Remedial Process Optimization Modeling Data Package* (SGW-40781, Rev. 0) and are listed in Table M-5 (Appendix M). The data provided in the model data package are grouped by geologic unit and include information on the data source and method of analysis. Well locations with estimates for hydraulic conductivity cover a broad area at 100-D, but are limited at 100-H to the area around H Reactor. Among the listed well locations are three wells in the horn area for which hydraulic conductivity values were available at the time.

Hydraulic conductivity values corresponding to the Hanford formation and the Ringold Formation unit E provided the basis for the development of the hydraulic conductivity distribution in the 100-D/H and across the horn that is described in *Conceptual Framework and Numerical Implementation of 100 Areas Groundwater Flow and Transport Model* (SGW-46279). Slug test data and analysis from RI wells is included (*Analysis of Slug Test Data at the 100-HR-3 Operable Unit* [ECF-100HR3-12-0011]) in Appendix D. Variability of the horizontal hydraulic conductivity in the unconfined aquifer reflects the variable cementation and sediment heterogeneity of the Ringold Formation unit E and the Hanford formation.

Based on this dataset, approximate ranges of hydraulic conductivity for each area are summarized in Table 3-9, assuming an average value where multiple entries are provided for the same well location. It should be noted that the values for Wells 199-D8-3 and 199-H4-10 are well outside the range of available hydraulic conductivity estimates from all other wells and for that reason two sets of ranges were calculated for 100-D and 100-H, including and excluding those data points, respectively. Also, one of the hydraulic conductivity estimates available in the horn was an order of magnitude higher than the other two.

The distribution of hydraulic conductivity values in each area is illustrated in the cumulative frequency plot shown in Figure 3-9. The hydraulic conductivity frequency plot suggests that the distributions in 100-D and 100-H are not significantly different. Both datasets are characterized by some extreme (and potentially suspect) values, resulting in higher mean values which differ by a factor of three between the two areas. When those extreme values are excluded, the mean hydraulic conductivity in the Hanford formation of 100-H is about two times higher than that in the Ringold Formation unit E of 100-D. Similarly, the median hydraulic conductivity value is 0.026 cm/s for the Ringold Formation unit E and 0.039 cm/s for the Hanford formation. It is important to note that in localized areas where horizontal conductivity is higher, preferential flow pathways may exist.

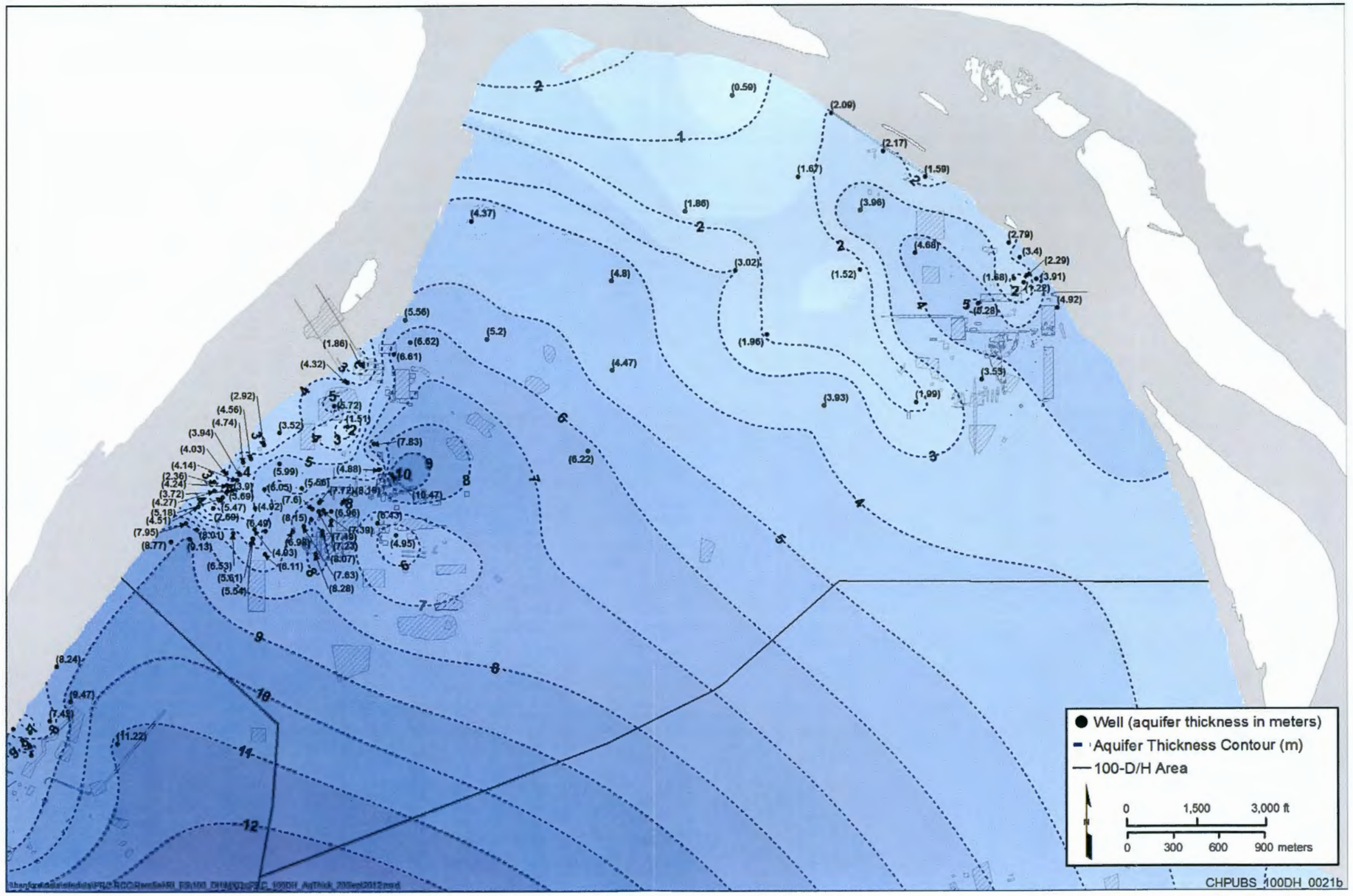


Figure 3-7. Unconfined Aquifer Thickness at 100-D/H for August to September 2010

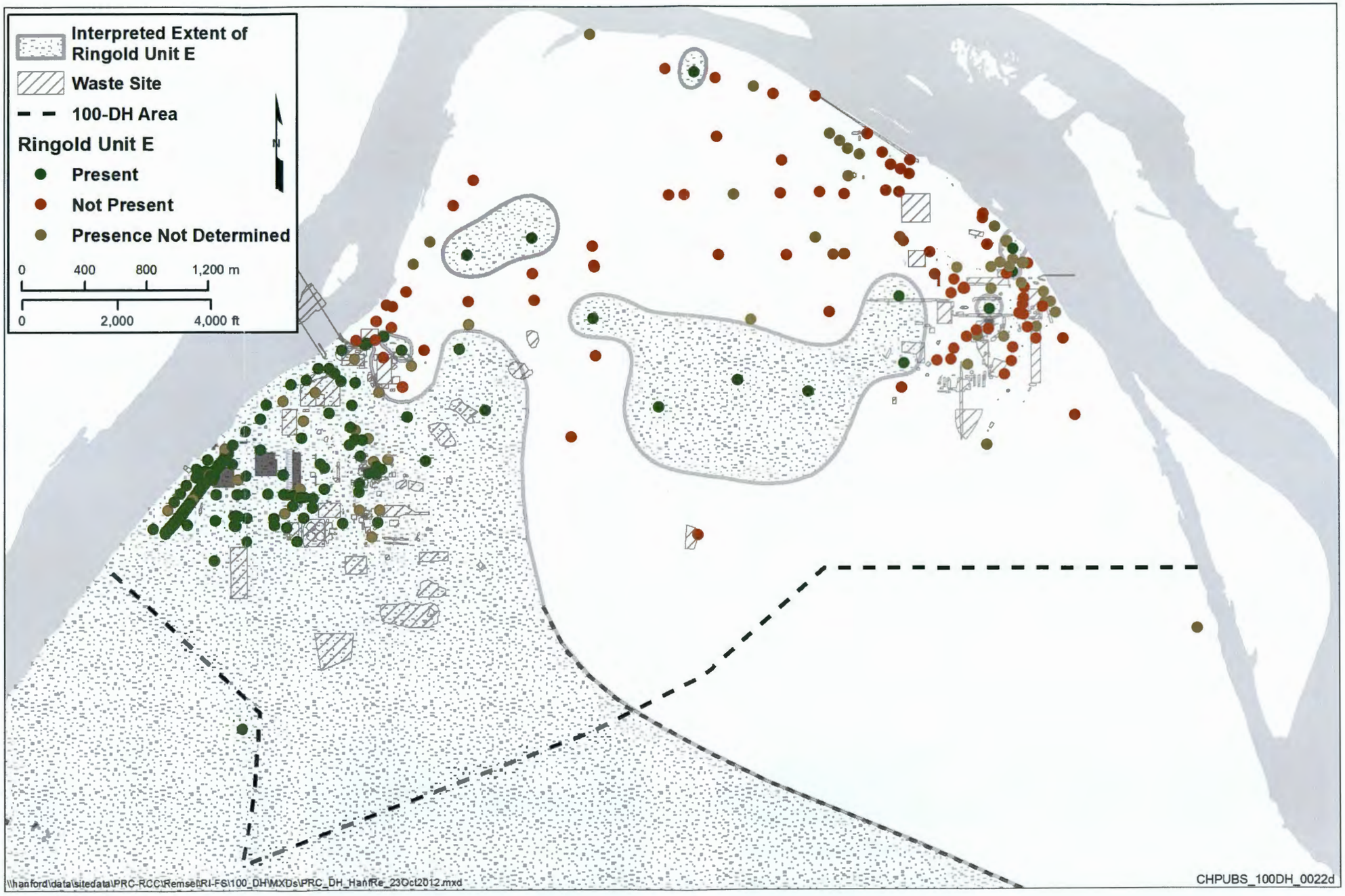


Figure 3-8. Hanford/Ringold Formation Unit E Presence at 100-D/H

Table 3-9. 100-D/H Estimated Ranges of Horizontal Hydraulic Conductivity Values Using Model Data Package Data

Area	Formation	Horizontal Hydraulic Conductivity K_H	Average Horizontal Hydraulic Conductivity K_H
100-D	Ringold unit E	0.004 to 0.648 cm/s (10.0 to 1837.0 ft/d)	0.059 cm/s (167.0 ft/d)
100-D (excluding outliers)	Ringold unit E	0.004 to 0.187 cm/s (10.0 to 530.0 ft/d)	0.040 cm/s (114.8 ft/d)
100-H	Hanford	0.018 to 1.863 cm/s (50.0 to 5,290.8 ft/f)	0.170 cm/s (483.0 ft/d)
100-H (excluding outliers)	Hanford	0.018 to 0.670 cm/s (50.0 – 1,900.0 ft/f)	0.086 cm/s (242.6 ft/d)
Horn	Hanford	0.018 to 0.279 cm/s (50.0 to 790.0.0 ft/f)	0.106 cm/s (300.0 ft/d)

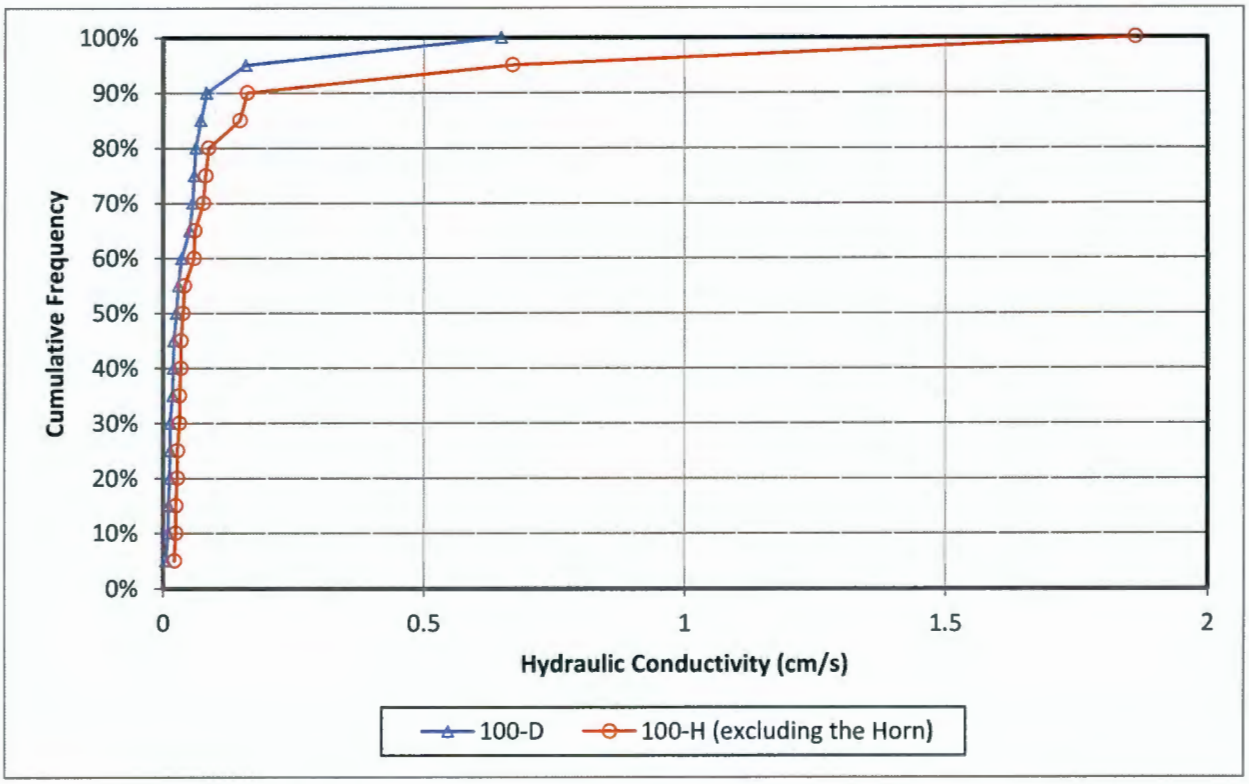


Figure 3-9. Cumulative Frequency of Hydraulic Conductivity Estimates in 100-D/H

A field investigation of the horn area between 100-D and 100-H was conducted in 2007 and 2008 to characterize the extent, concentration, and movement of Cr(VI) in groundwater (*Hydrogeological Summary Report for 600 Area Between 100-D and 100-H for the 100-HR-3 Groundwater Operable Unit* [DOE/RL-2008-42]). As part of this study, new wells were drilled and development data were analyzed to calculate hydraulic conductivity estimates in the unconfined aquifer. As no aquifer tests were performed

for this characterization effort, the well development data could provide the basis only for rough estimates of hydraulic conductivity, especially considering that well development data reflect short-term aquifer response and can largely overestimate the specific capacity of the well. The estimated hydraulic conductivities varied between 0.013 and 2.242 cm/s (36 and 6,354 ft/d).

Slug test data from 16 RI wells in 100-D and 100-H were analyzed to estimate hydraulic conductivity and specific storage of the water-bearing unit at each well as described in *Analysis of Slug Test Data at the 100-HR-3 Operable Unit* (ECF-100HR3-12-0011), included in Appendix D. Calculated hydraulic conductivity values for wells in 100-D and 100-H screened in the Hanford formations and the Ringold Formation unit E are tabulated on Table 3-10.

Table 3-10. Estimated Hydraulic Conductivity Values from Slug Tests in 100-D and 100-H Area Wells

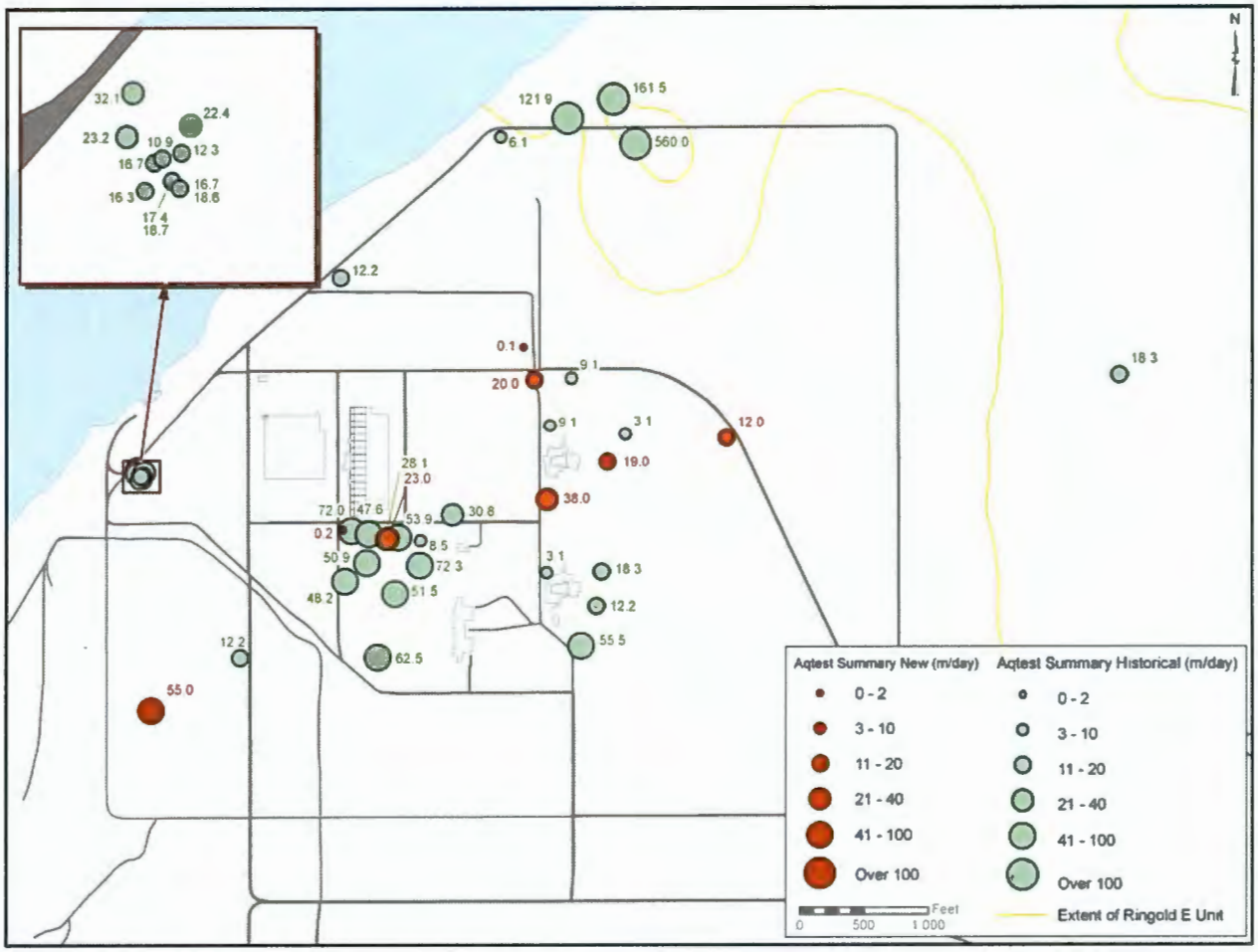
Well Name	Area	Geologic Unit	Horizontal Hydraulic Conductivity KH (cm/s)	Horizontal Hydraulic Conductivity KH (ft/d)	Vertical Anisotropy Ratio (KV/KH)
199-D3-5	100-D	Hanford and Ringold E	0.064	181.4	0.1
199-D5-132	100-D	Ringold E	0.022	62.4	0.1
199-D5-133	100-D	Ringold E	0.044	124.7	0.1
199-D5-143	100-D	Ringold E	0.023	65.2	0.1
199-D5-144	100-D	Ringold E	0.027	76.5	0.1
199-D6-3	100-D	Ringold E	0.014	39.7	0.1
199-H3-6	100-H	Hanford	0.044	124.7	0.01
199-H3-7	100-H	Hanford	0.031	87.9	0.1
199-H6-3	100-H	Hanford	0.031	87.9	0.1
199-H6-4	100-H	Hanford	0.140	396.9	0.1

Source: ECF-100HR3-12-0011, *Analysis of Slug Test Data at the 100-HR-3 Operable Unit.*

Hydraulic conductivity estimates for the Ringold Formation unit E based on the analysis of slug test data in 100-D varied between 0.014 and 0.044 cm/s (40.0 to 124.7 ft/d) with an average value of 0.026 cm/s (73.7 ft/d). The range of hydraulic conductivity estimates for the Hanford formation in 100-H was 0.031 to 0.140 cm/s (87.9 to 396.9 ft/d) with an average value of 0.062 cm/s (174.3 ft/d). These estimates are within the range of values historically reported in these areas and geologic units, and within the ranges reported in the literature for the particular soil types (e.g., *Groundwater* [Freeze and Cherry, 1979], Table 2.2)

Figures 3-10 and 3-11 illustrate the relative magnitude of the hydraulic conductivity estimates in 100-D and 100-H, respectively, distinguishing between older and recent aquifer tests (RI slug tests). These illustrations suggest that hydraulic conductivities in the Hanford formation are higher than those in the Ringold Formation unit E in this part of the River Corridor; however, the difference is not as significant as seen in other areas across the Hanford Site (e.g., the Central Plateau). In addition, hydraulic

conductivities in 100-H are relatively uniform, unlike those in 100-D where a higher degree of variation is identified, although a spatial pattern of those variations is not evident in those plots.

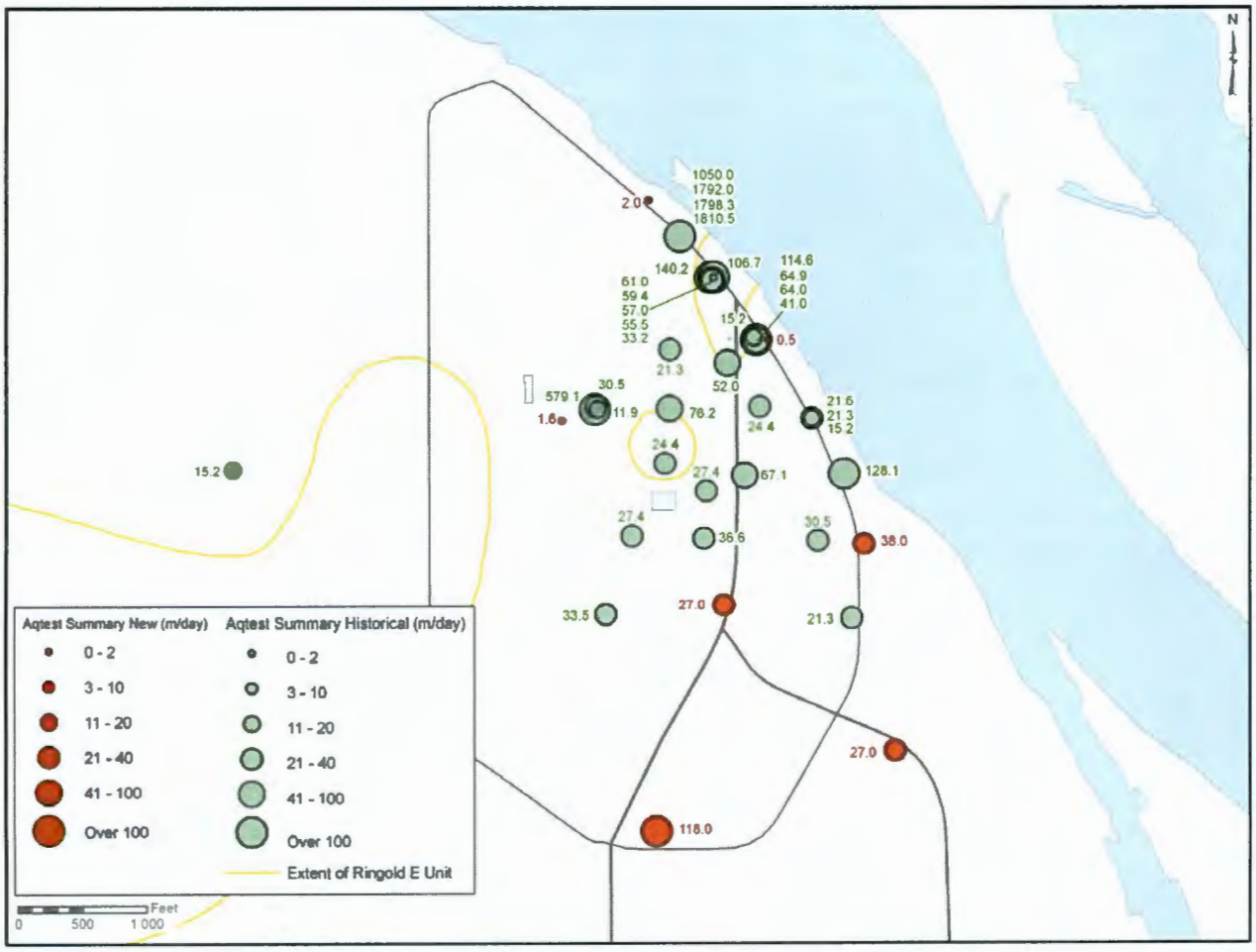


Source: ECF-100HR3-12-0011, *Analysis of Slug Test Data at the 100-HR-3 Operable Unit.*

Figure 3-10. Hydraulic Conductivity Estimates by Magnitude in 100-D

3.6.1.2 Vertical Hydraulic Conductivity – Unconfined Aquifer

Aquifer characterization activities were performed before the treatability test and subsequent installation of the In Situ REDOX Manipulation barrier (ISRM) in 100-D (*100-D Area In Situ Redox Treatability Test for Chromate-Contaminated Groundwater* [PNNL-13349]). As part of the characterization effort, two constant-rate aquifer tests were conducted to provide information that could be used to evaluate possible changes in the subsurface hydrologic conditions. The pumping tests were conducted before and following the treatability test with aquifer responses at seven observation wells were analyzed individually. The results of this analysis included estimates of the horizontal hydraulic conductivity in the unconfined aquifer, vertical anisotropy, storativity and specific yield. Horizontal hydraulic conductivity estimates were included in *100-HR-3 Remedial Process Optimization Modeling Data Package* (SGW-40781, Rev. 0) as presented earlier. Evaluation of the vertical anisotropy in the vicinity of the tested wells resulted in a range of vertical-to-horizontal ratios of 0.006 to 0.031 with a mean value of 0.015 (±0.010), with the second term corresponding to one standard deviation.



Source: ECF-100HR3-12-0011, *Analysis of Slug Test Data at the 100-HR-3 Operable Unit.*

Figure 3-11. Hydraulic Conductivity Estimates by Magnitude in 100-H

Soil samples were collected during the RI from the vadose zone, unconfined aquifer, RUM, and Ringold Formation unit B for vertical hydraulic conductivity analysis. The results from the sieve analyses and permeameter tests are tabulated on Table M-4 (Appendix M). Vertical hydraulic conductivity estimates are summarized in Table 3-11.

Based on these values, the vertical hydraulic conductivity in the Hanford formation ranges between 2.1×10^{-6} cm/s and 3.94×10^{-2} cm/s with a mean value of 1.16×10^{-2} cm/s. Vertical hydraulic conductivities in the Ringold Formation unit E vary between 2.45×10^{-4} cm/s to 8.01×10^{-2} cm/s with a mean value of 3.24×10^{-2} cm/s. It should be noted that there is considerable variation within both units.

Assumed vertical anisotropy ratios that were used for the analysis of the RI well slug test data are listed in Table 3-10. In all cases the vertical anisotropy ratio was 0.1, except for Well 199-H3-6 in 100-H where the ratio was 0.01. However, the horizontal hydraulic conductivity estimates were not particularly sensitive to the anisotropy ratio and therefore a ratio of 0.1 is considered reasonable.

Table 3-11. Estimated Vertical Hydraulic Conductivity Values from Soil Samples in 100-D and 100-H Area Wells

Well ID	Vertical Hydraulic Conductivity, K-sat (cm/sec)	Depth (m)	Depth (ft)	Stratigraphic Unit (based on geologic log)[a]	Lithologic Description[b]
199-H3-9	2.89×10^{-2}	14.40	47.24	Hanford	Gravelly sand
	3.8×10^{-5}	15.93	52.26	RUM	Silty sand
	4.24×10^{-3}	23.23	76.21	RUM	Sand
	5.7×10^{-6}	30.79	101.0	RUM	Silty sand
199-H3-10	3.64×10^{-3}	16.54	54.27	Hanford	Sandy gravel
	1.80×10^{-6}	17.36	56.96	RUM	Silty sand
	2.42×10^{-3}	68.15	223.6	Lower water bearing unit (presumably unit B)	Sand with silt
199-H2-1	2.20×10^{-6}	11.61	38.09	RUM contact	Silty sand
	1.30×10^{-4}	11.90	39.04	RUM contact	Sand with silt
	5.0×10^{-3}	48.63	159.5	RUM	Sand with silt
199-D5-133	9.33×10^{-3}	15.8	51.84	Hanford	Gravelly sand
	5.21×10^{-2}	31.62	103.7	Unit E	Sand
199-D5-141	8.99×10^{-4}	23.30	76.44	Unit E	Sandy gravel
	2.45×10^{-4}	24.77	81.27	Unit E	Sandy gravel
	8.01×10^{-2}	32.54	106.8	Unit E	Sand
	1.4×10^{-8}	34.98	114.8	RUM	Silty sand
	9.54×10^{-3}	96.15	315.5	Lower water-bearing unit (presumably unit B)	Sand
199-D5-143	3.64×10^{-3}	11.62	38.12	Hanford	Gravel with sand
	3.94×10^{-2}	13.11	43.01	Hanford	Sandy gravel
	1.23×10^{-2}	14.66	48.10	Hanford	Sandy gravel
	2.1×10^{-6}	16.35	53.64	Hanford	Sandy gravel
	5.73×10^{-2}	17.70	58.07	Unit E	Gravel with sand
	3.64×10^{-3}	30.86	101.2	Ringold Unit E	Gravel with sand

Table 3-11. Estimated Vertical Hydraulic Conductivity Values from Soil Samples in 100-D and 100-H Area Wells

Well ID	Vertical Hydraulic Conductivity, K-sat (cm/sec)	Depth (m)	Depth (ft)	Stratigraphic Unit (based on geologic log)[a]	Lithologic Description[b]
	2.7×10^{-8}	33.19	108.9	RUM	Silt with sand
C7850	5.06×10^{-4}	12.42	40.75	Hanford	Gravel with sand
	6.93×10^{-3}	19.37	63.55	Ringold Unit E	Gravelly sand

Notes: Values are based on data tabulated in Table M-4, Appendix M; hydraulic conductivity was calculated in the laboratory by Method D2434 Permeability.

a. Samples indicated as RUM are from the first water bearing unit in the RUM.

b. Lithology based on sieve analysis.

3.6.2 Confined Aquifer Zones within the Ringold Formation

The RUM contains near-horizontal sandy water-bearing units between the RUM surface and Ringold Formation unit B, a deeper water-bearing unit. These units represent confined or semiconfined units with variable conductivity and interconnectivity. The recognized aquifer units within the RUM are identified as the Ringold Formation unit C or D, generally the first water-bearing unit, and the lower water-bearing unit, which is presumed to be the Ringold Formation unit B (Figure 3-3). Other water-bearing units may also be present; however, these discontinuous units are not formally recognized in the nomenclature.

At 100-D, the first water bearing unit in the RUM is typically identified as Ringold Formation unit C. This unit may be absent at 100-D, with the first water bearing unit at 100-D being the Ringold Formation unit B (*Geology of the Northern Part of the Hanford Site: An Outline of Data Sources and the Geologic Setting of the 100 Areas* [WHC-SD-EN-TI-011]).

Three wells are currently completed within the RUM in the vicinity of the horn. The first water-bearing unit within the RUM is semiconfined to confined, based on its observed piezometric head being higher than the overlying shallow unconfined unit in some areas, but otherwise not well defined in this area.

At 100-H, the RUM first water-bearing unit is confined to semiconfined and described as a very fine-to-fine sand unit in RI borehole logs. The unit may occur as a discontinuous sand layer. Observations of the piezometric head in this unit during local pumping and during injection into the shallow unit indicate that it is not fully isolated from the overlying unconfined unit in the area inland of the H Reactor; near the river, however, it appears to be semiconfined to confined and isolated from the overlying unit. In wells completed in this unit near the river, the groundwater exhibits rapid, nearly simultaneous head changes that coincide directly with river stage fluctuations. The inland wells do not exhibit simultaneous responses to river stage changes, but are responsive to head changes in the overlying unconfined unit; this indicates that the unit identified as the first water-bearing unit of the RUM in the inland area of 100-H may not be in direct hydraulic communication with the similar unit near the river. The first water-bearing unit is approximately 0.5 to 7 m (1.6 to 23 ft) thick and occurs at elevations ranging from approximately 95 to 105 m (312 to 345 ft) (*North American Vertical Datum of 1988* [NAVD88]). Groundwater levels in this unit respond to changes in river stage, and when the river stage is moderate to low, this unit shows slightly higher pressure head than the unconfined aquifer near the river. In addition, the lack of lag time

between Columbia River stage changes and the water table fluctuations in the nearby wells completed in the RUM indicates a hydraulic connection between these units.

The lower sand unit is approximately 6 to 10 m (20 to 33 ft) thick and occurs at elevations ranging from approximately 20 to 30 m (66 to 98 ft) (*North American Vertical Datum of 1988* [NAVD88]). When first encountered during drilling Well 199-H3-9, the lower water-bearing unit sands caused a sand heave of more than 15 m (50 ft) into the borehole, indicating good water production in that unit.

Five of the RI well boreholes were drilled to the lower water-bearing unit in the RUM, presumed to be Ringold Formation unit B. These are Wells: 199-D5-134 (C7624, Well R4), 199-D5-141 (C7625, Well R5), 199-H2-1 (C7631, Well R3), 199-H3-9 (C7639, Well R1), and 199-H3-10 (C7640, Well R2). Laboratory results (presented in Chapter 4) did not indicate contamination in the lower water bearing unit encountered during drilling. As a result, the lower half of each borehole was sealed and the five deeper RI boreholes were completed with each well screened across the first water-bearing unit within the RUM. Piezometer 199-H4-15CR indicates the pressure head for this confined water-bearing zone is 1 to 2 m (3 to 7 ft) higher than for the unconfined aquifer near the Columbia River in 100-H.

3.6.2.1 Horizontal Hydraulic Conductivity – Water-Bearing Units within the RUM

Slug tests were completed on five RI wells installed in the first water-bearing unit of the RUM (*Analysis of Slug Test Data at the 100-HR-3 Operable Unit* [ECF-100HR3-12-0011]). In 100-D, hydraulic conductivity values estimated for the sandy unit at wells 199-D5-134 and 199-D5-141 were 1.2×10^{-4} cm/sec (0.66 ft/day) and 2.3×10^{-4} cm/sec (0.34 ft/day), respectively. In 100-H, three wells were screened in the first water-bearing unit of the RUM with hydraulic conductivities ranging from 6.9×10^{-4} cm/sec (1.96 ft/day) to 2.3×10^{-3} cm/sec (6.52 ft/day). The higher value at 199-H3-2C may be related to leaky aquifer conditions identified at this location. The 100-D RI results had hydraulic conductivity values that were two orders of magnitude less than those at Well 199-H3-2C. Table 3-12 summarizes the estimated hydraulic conductivities from the slug test data.

Table 3-12. Estimated Hydraulic Conductivity Values from Slug Tests

Well Name	Area	Geologic Unit	Horizontal Hydraulic Conductivity, K_H (cm/s)	Horizontal Hydraulic Conductivity, K_H (ft/d)	Vertical Anisotropy Ratio (K_V/K_H)
199-D5-134	100-D	First water-bearing unit in RUM	1.2×10^{-4}	0.34	0.1
199-D5-141	100-D	First water-bearing unit in RUM	2.3×10^{-4}	0.65	0.1
199-H2-1	100-H	First water-bearing unit in RUM	2.3×10^{-3}	6.52	0.1
199-H3-9	100-H	First water-bearing unit in RUM	6.9×10^{-4}	1.96	1.0
199-H3-10	100-H	First water-bearing unit in RUM	1.9×10^{-3}	5.39	0.1

Source: ECF-100HR3-12-0011, *Analysis of Slug Test Data at the 100-HR-3 Operable Unit.*

3.6.2.2 Vertical Hydraulic Conductivity –Water-Bearing Units within the RUM

During installation of the 70 DX/HX RPO wells across the 100-HR-3 OU, efforts were made to collect RUM surface samples at each borehole location for permeameter tests. Tests were not conducted when poor sample recovery occurred (e.g., Well 199-D7-5). In addition, soil samples were collected from 14 boreholes during the RI to evaluate the physical properties of the RUM. Each borehole log was conducted to verify the lithology of the depth interval that is representative of the first water-bearing unit of the RUM. The vertical hydraulic conductivity estimates are tabulated in Table 3-11.

The estimated vertical hydraulic conductivity values in the first water-bearing unit of the RUM varied between 1.40×10^{-8} and 5.0×10^{-3} cm/sec. However, samples from two of the locations (Well 199-H2-1 and 199-H3-9) had hydraulic conductivity values of 4.24×10^{-3} and 5.0×10^{-3} cm/s, respectively. These two locations skew the results for the remainder of the sample set. The hydraulic conductivity values without including those results range from 1.4×10^{-8} to 3.8×10^{-5} cm/s, which is consistent with expected results.

Samples from two different depths were collected in 199-H1-35 and 199-H1-36. The second sample from Well 199-H1-35 was collected 1.5 m [5 ft] into RUM and it had a vertical hydraulic conductivity value of about 1.0×10^{-6} cm/sec, approximately one order of magnitude lower than the value estimated from the sample collected from the RUM surface. The 199-H1-36 sample (1.5 m [5 ft] into the RUM) appears to have greater sand content than the sample collected at the RUM surface. The vertical hydraulic conductivity value for the 199-H1-36 RUM surface sample was at 1.0×10^{-6} cm/sec. The 199-H4-71 and 199-H4-73 samples have a greater percentage of sand/gravel than RUM samples with low vertical hydraulic conductivity values.

Analysis of an aquifer test performed at Well 199-H3-2C, as part of the deep chromium investigation in 100-H (*Aquifer Testing and Rebound Study in Support of the 100-H Deep Chromium Investigation* [SGW-47776]), suggests that there is hydraulic connection between the unconfined aquifer and a shallow water-bearing unit in the RUM at that location. The borehole log of this well shows higher permeability sediments above its screened interval. Vertical hydraulic conductivity estimates for neighboring Wells 199-H4-71 and 199-H4-73 are more than one order of magnitude higher than those of other RUM wells nearby. On the other hand, the aquifer test data at Well 199-H4-12C did not allow for similar inferences, because of its proximity to the Columbia River and the river's effect on water levels in the aquifer. Other nearby wells such as 199-H4-70 and 199-H3-27, have estimated vertical hydraulic conductivities consistent with typical RUM values (1.0×10^{-6} cm/sec, or 0.0028 ft/day), which suggests that the extent of a higher vertical conductivity zone could be limited in that area.

3.6.3 Columbia River Basalt Group Hydrogeology

The basalt confined aquifer system extends throughout the Pasco Basin. The upper basalt confined aquifer is an interflow zone consisting of fractured Elephant Mountain Member basalt flow-bottom, Rattlesnake Ridge interbed sediments, and underlying fractured Pomona Member basalt flow-top (see Figure 3-3). Piezometer 199-H4-15CP monitors a fracture zone in the Elephant Mountain Member basalt and consistently exhibits an artesian head, with water flowing from the well when the well cap is opened. Well 199-H4-2 monitors the upper basalt confined aquifer and also exhibits an artesian conditions. It should be noted that the pressure differential exhibited between the basalt aquifer unit and the aquifers within the overlying unconsolidated units is not a demonstration of an actual upward gradient (i.e., flow of groundwater from the deeper units to the shallower units) in the absence of defined flow paths.

Early groundwater maps of the upper confined basalt aquifer system show groundwater flow to the southwest under 100-D/H, based on very limited hydraulic head data between the Columbia River and Gable Mountain – Gable Butte (*Hydrochemistry and Hydrogeologic Conditions Within the Hanford Site*

Upper Basalt Confined Aquifer System [PNL-10817]). Recent Hanford Site groundwater maps for the upper basalt confined aquifer (*Hanford Site Groundwater Monitoring and Performance Report for 2009* [DOE/RL-2010-11]) have not portrayed the piezometric surface for the upper basalt confined aquifer in the 100 Area because of the limited dataset.

3.7 Groundwater Flow Regime

The understanding of groundwater movement at 100-D/H Area and its effects on migration of associated contaminants is based on knowledge of historical conditions as well as current operating conditions that affect groundwater elevation, flow direction, and velocity. Hydrogeologic characterization of aquifer material at 100-D/H presented in preceding subsections provides part of the picture, with understanding of the dramatic effects of water management related to reactor operations and recent operation of groundwater remedial actions providing the rest of the story. This section focuses on groundwater flow patterns and rates under historical (predevelopment and operational) and recent conditions. Natural an artificial hydrologic processes influenced groundwater flow patterns and contaminant distribution at 100-D/H.

The groundwater regime at 100-D/H can be separated into the following phases:

- Historical conditions, including the following:
  - Predevelopment conditions, during which time there was little to no anthropogenic activity
  - Pre-Hanford Site operations, when irrigated agriculture was implemented at numerous locations near the river, including operation of a substantial irrigation canal across the site to transport irrigation water
  - Operational conditions, during which time reactors were constructed and operated at the 100-D/H Areas, and substantial artificial recharge occurred because of disposal of wastewater into the vadose zone
  - Post-operational conditions, during which time effects from reactor and related operations ceased, and groundwater conditions commenced recovery to conditions showing many of the features of the predevelopment time frame
- Current (recent) conditions, during which time interim remedial actions have been undertaken. These remedial actions include waste site remediation in the vadose zone (source control), and groundwater pump-and-treat systems for the groundwater.

The following section describes the groundwater flow regime in terms of these historical and current (recent) conditions. It focuses on conditions in the unconfined aquifer caused by groundwater contamination and related remedial activities within the Ringold Formation unit E and Hanford formation, and concludes with discussion of the underlying RUM and interactions between the RUM and unconfined aquifer.

3.7.1 Historical Groundwater Flow Conditions

General patterns of groundwater flow before the commencement of operations at 100-D/H can be inferred from early maps of groundwater levels and from the distribution of natural recharge and discharge boundaries at the site. Together, these indicate that groundwater flow directions and rates in the area of the D, DR, and H Reactors were dictated by the natural locations of recharge and discharge, leading to general patterns of flow from the south-southeast toward the Columbia River near the D and DR Reactors, and from the south-southwest toward the Columbia River near the H Reactor. Groundwater

ultimately discharged to the Columbia River and fluctuations in groundwater levels within the unconfined aquifer resulted from natural changes in the stage of the Columbia River.

During the period of irrigated agriculture operations on the Hanford Site area, some undefined amount of artificial recharge likely occurred. This recharge would have been mostly related to local conditions beneath fields irrigated using flood or rill distribution techniques, and from leakage from the Hanford Ditch (an irrigation canal used to convey water from the vicinity of 100-K Area on the upstream side to the vicinity of the former Hanford Townsite on the downstream side). The hydrogeologic effects of the period of irrigated agricultural operations at Hanford are not defined quantitatively.

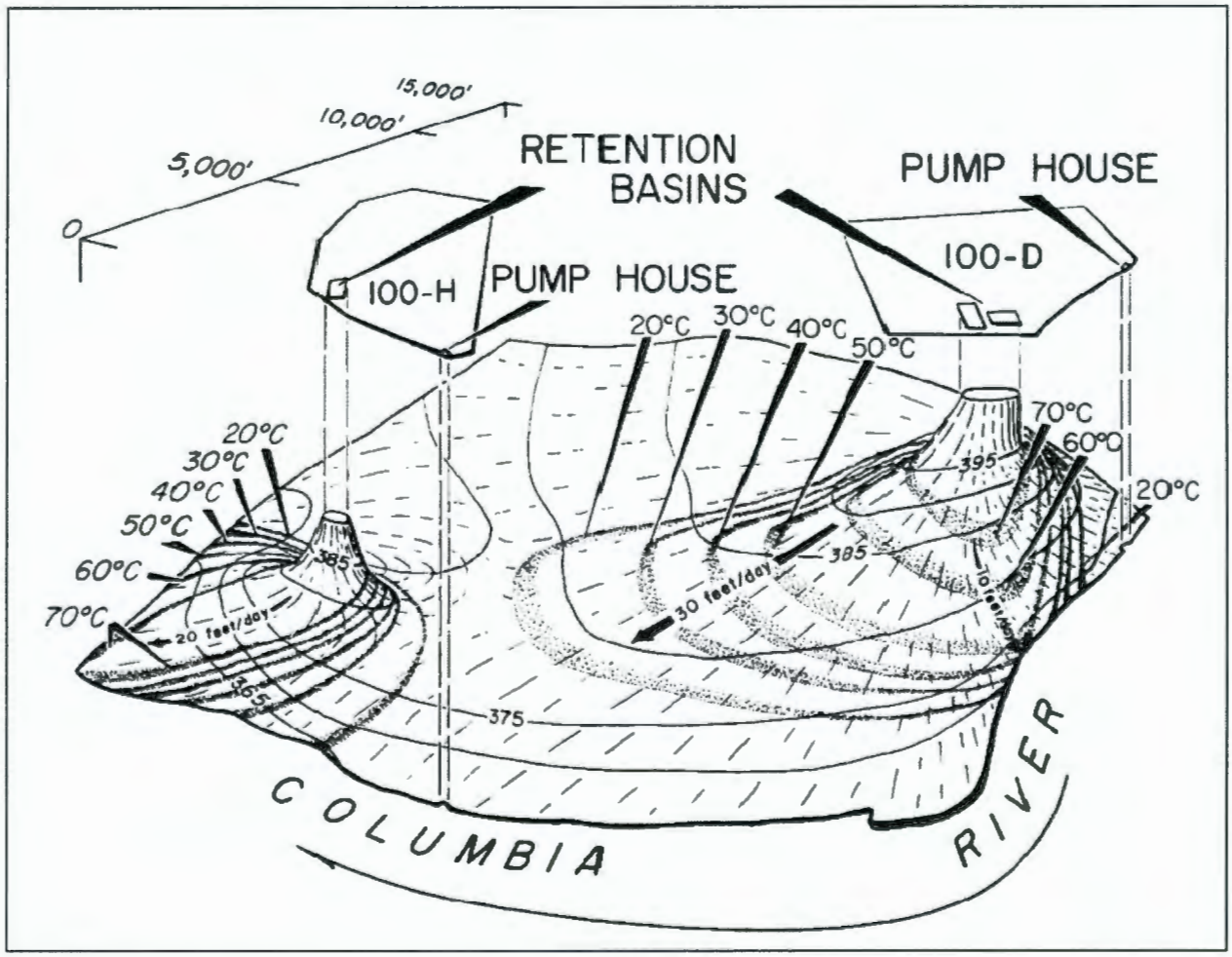
With the industrial development of the Hanford Site, various anthropogenic influences have dominated the directions and rates of groundwater flow. During operation of the D, DR, and H Reactors, large volumes of spent cooling water (high-temperature effluent from the reactors, containing Cr(VI) and other chemicals used to maintain water quality) were discharged to the retention basins where the water was held up, allowing the water to cool somewhat and short-lived radionuclides to decay. After the holding period, the contaminated cooling water was discharged directly into the Columbia River. Reactor cooling water entered the vadose zone in the vicinity of the reactor operations under two common conditions. First, leaks developed in the retention basins as a result of thermal expansion and contraction, allowing cooling water to leak from the basins into the underlying vadose zone. This contributed substantial quantities of localized artificial recharge to the underlying groundwater. Secondly, episodic fuel element failures also contaminated the cooling water with radioactive fission products and fuel residues; under these upset conditions, the cooling water stream was discharged directly to the vadose zone via engineered infiltration trenches instead of to the river. The discharge of contaminated cooling water to the vadose zone reduced the amount of radioactive contaminants that ultimately reached the river. The local artificial recharge conditions caused by discharges of contaminated cooling water from these trenches, and leaks from the retention basins, resulted in the buildup of extensive groundwater mounds in the unconfined aquifer beneath the reactor operating areas at 100-D and 100-H. Monitoring well hydrographs from 100-D and 100-H indicate that wastewater infiltration elevated groundwater levels as much as 10 m (33 ft) at 100-D and 7 m (21 ft) at 100-H. These extensive mounds altered groundwater flow patterns and groundwater velocity for years and account for the observed current distribution of groundwater contaminants across the entire width of the horn area from 100-D to 100-H. Operation of the three reactors ceased in 1964 (DR Reactor), 1967 (D Reactor), and 1965 (H Reactor). The artificial recharge mounds dissipated fairly quickly and groundwater flow began to exhibit pre-Hanford conditions.

A contemporary report of observations of the groundwater mounding effects of discharges of reactor cooling water to the vadose zone in the Hanford 100 Areas during reactor operations is presented in *Status of the Ground Water Beneath Hanford Reactor Areas January, 1962 to January, 1963* (HW-77170), which presents detailed descriptions of the groundwater mounds observed at all of the Hanford reactor areas, including detailed description of the groundwater temperature effects caused by discharge of high volumes of near-boiling cooling water.

3.7.1.1 Groundwater Mounding at 100-D

Groundwater mounding beneath the 107-D and 107-DR Retention Basins began shortly after reactor operations started in 100-D. By 1963, both basins had developed contraction/expansion cracks that allowed a large fraction of the conveyed cooling water to leak from the basins into the underlying vadose zone. The study of thermal and hydraulic effects, published in *Status of the Ground Water Beneath Hanford Reactor Areas January, 1962 to January, 1963* (HW-77170), clearly indicate the evolution of a groundwater mound, consisting largely of reactor cooling water, that extended all the way from 100-D to 100-H, with a peak elevation of greater than 122 m (400 ft) amsl beneath 107-D and 107-DR Retention Basins. The thermal effects of the 100-D cooling water recharge were measured in the intake water for

100-H. The calculated groundwater velocity between 100-D and 100-H was 9.1 m/day (30 ft/day). Groundwater temperature had been raised between 10°C and 50°C (50°F and 122°F) by 1963. The inferred extent of the groundwater mound associated with operation of the D and DR Reactors, along with the associated measured thermal effects, are shown in Figure 3-12. Thermal springs, caused by discharge of reactor cooling water from the exposed aquifer face near the river, were observed to extend over 600 m (2,000 ft) along the river shore downstream from the 100-D cooling water retention basins.



Source: HW-77170, *Status of the Ground Water Beneath Hanford Reactor Areas January, 1962 to January, 1963.*

Figure 3-12. Perspective Drawing of the Water Table at the 100-D and 100-H Areas in 1962

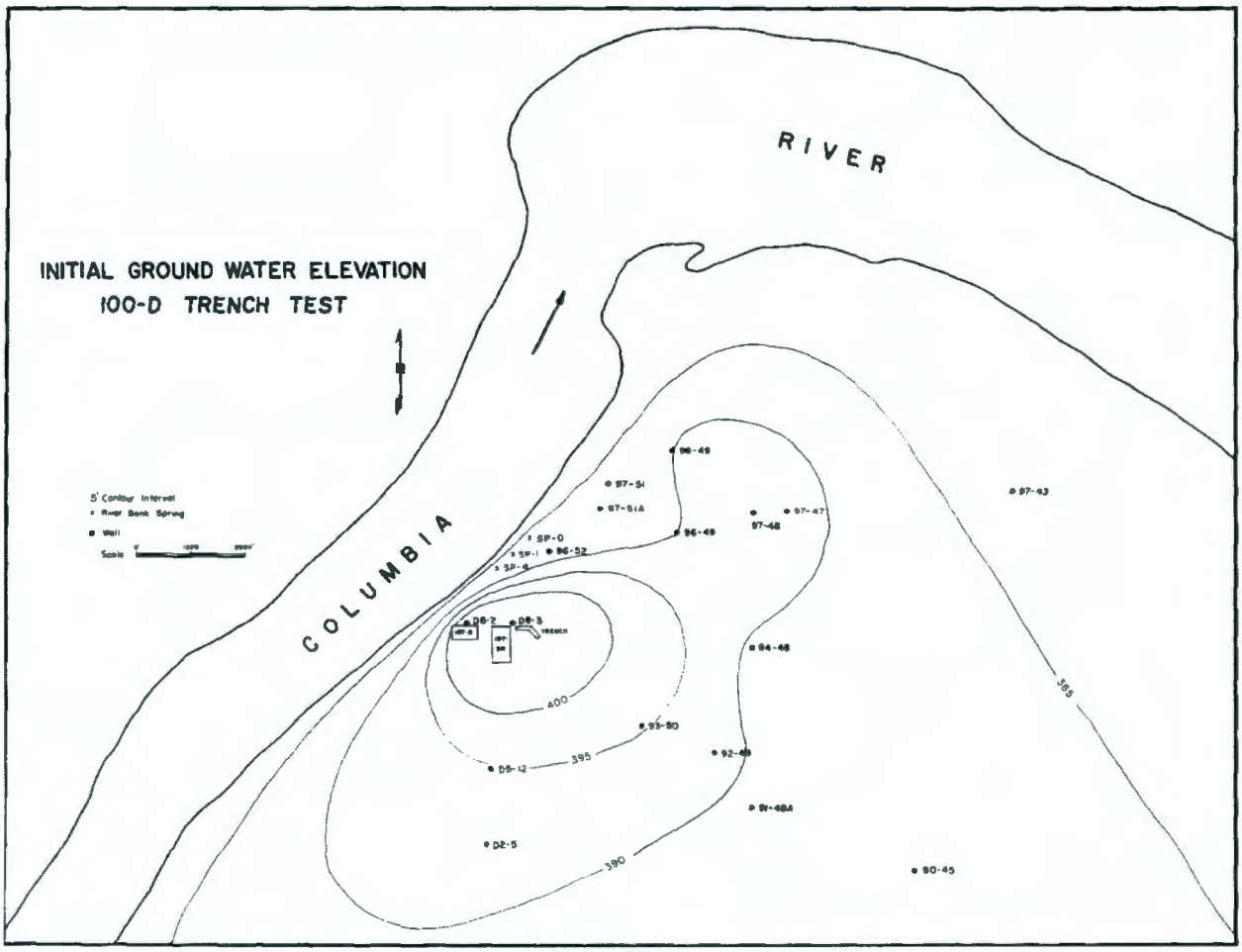
Continuing releases of fission and activation products in cooling water into the adjacent Columbia River was a concern for reactor operations by the 1960s. In order to assess some alternatives to continued release of cooling water directly to the river, a series of tests involving continuous discharge of reactor cooling water streams directly to the vadose zone was developed. One such test was conducted at 100-D during the last 4 months of operation of the D Reactor and is described in *Ground Disposal of Reactor Coolant Effluent* (BNWL-CC-1352). This test was performed from March to June 1967 and involved directing the entire cooling water discharge from operation of D Reactor (DR Reactor having been shut down in December 1964) into the 116-DR-1 and 116-DR-2 Waste Water Trenches. The objective of the test was to observe the reduction in fission and activation product activity concentrations produced by the increased time of travel for cooling water to enter the river, compared to the historical practice of direct discharge of cooling water. The test involved monitoring the nuclide activity concentrations in the

effluent cooling water and comparing those measurements to the activity concentrations observed in samples collected from the thermal springs that emerged at the river shore. Measurement of changes in groundwater temperature and elevation were also conducted to document the effects of the discharge on the physical groundwater system. The test was found to be effective at reducing activity concentrations of target nuclides (e.g., iodine-131, chromium-51, and zinc-65). During the test, a large volume of reactor coolant effluent was discharged to the 116-DR-1 and 116-DR-2 Trenches at a constant rate of 104,100 L/min (27,500 gal/min). This resulted in a total subsurface discharge of more than 13 billion L (3.4 billion gal) of cooling water effluent over the course of the test.

Groundwater elevations and temperatures increased beneath 100-D, with no significant decrease in the infiltration rate over time. Approximately 25 percent of the discharged volume was accounted for in the groundwater mound that formed. Detectable increases in groundwater elevation were measured as far as 1.6 km (1 mi) from the trenches.

Figures 3-13 (a, b, and c) illustrate water table conditions before, during, and after the 4-month field test in 1967. The effect of the groundwater mound on groundwater lateral flow shows the radial flow from the groundwater mound created with the discharge. This large effluent discharge increased the groundwater gradient of the already-established mound and accelerated groundwater flow to the northeast and east away from the trenches. In addition, the resulting increased head of the enlarged groundwater mound would have applied additional vertical pressure on the underlying aquitard (RUM). This potentially resulted in some water migrating vertically into the underlying RUM, resulting in contamination in that unit, such as at Well 699-97-48C. However, the anisotropic nature of the contact between the RUM and the overlying unconsolidated formations (i.e., either Ringold Formation unit E or Hanford formation), as well as the anisotropic contact between the Hanford formation and the Ringold unit E would have made lateral flow away from the mound the preferential pathway; rapidly in the Hanford, and slightly slower in the Ringold unit E. In addition, the relatively low hydraulic conductivity of the RUM and the relatively short duration of the injection test (i.e., 4 months) would tend to minimize the vertical distribution effects of this test condition. This scenario is discussed further in Chapter 4.

Hydrographs from wells near the infiltration trench indicate that extensive groundwater mounding occurred in response to the infiltration, persisted for the duration of the test; the mound did not fully dissipate until about 1968 or 1969, although it was largely gone by September 1967 (Figure 3-14). During the test, groundwater elevation rose to nearly meet the ground surface in the immediate vicinity of the disposal trench, with a water table elevation of 129 m (415 ft) and a ground surface elevation of approximately 133 m (436 ft). The effects of the artificial recharge were compounded by the fact the Columbia River exhibited substantially higher-than-average annual peak river stages during the period of 1961 through 1972; this condition would have prolonged the decay of the groundwater mound established by artificial recharge at 100-D/H. By June 1967, the researchers were no longer able to clearly discern changes in groundwater elevation related to the cooling water discharge from the effects of the annual peak river stage, which occurred that month and reached an elevation of 134 m (440 ft) amsl at Priest Rapids Dam.

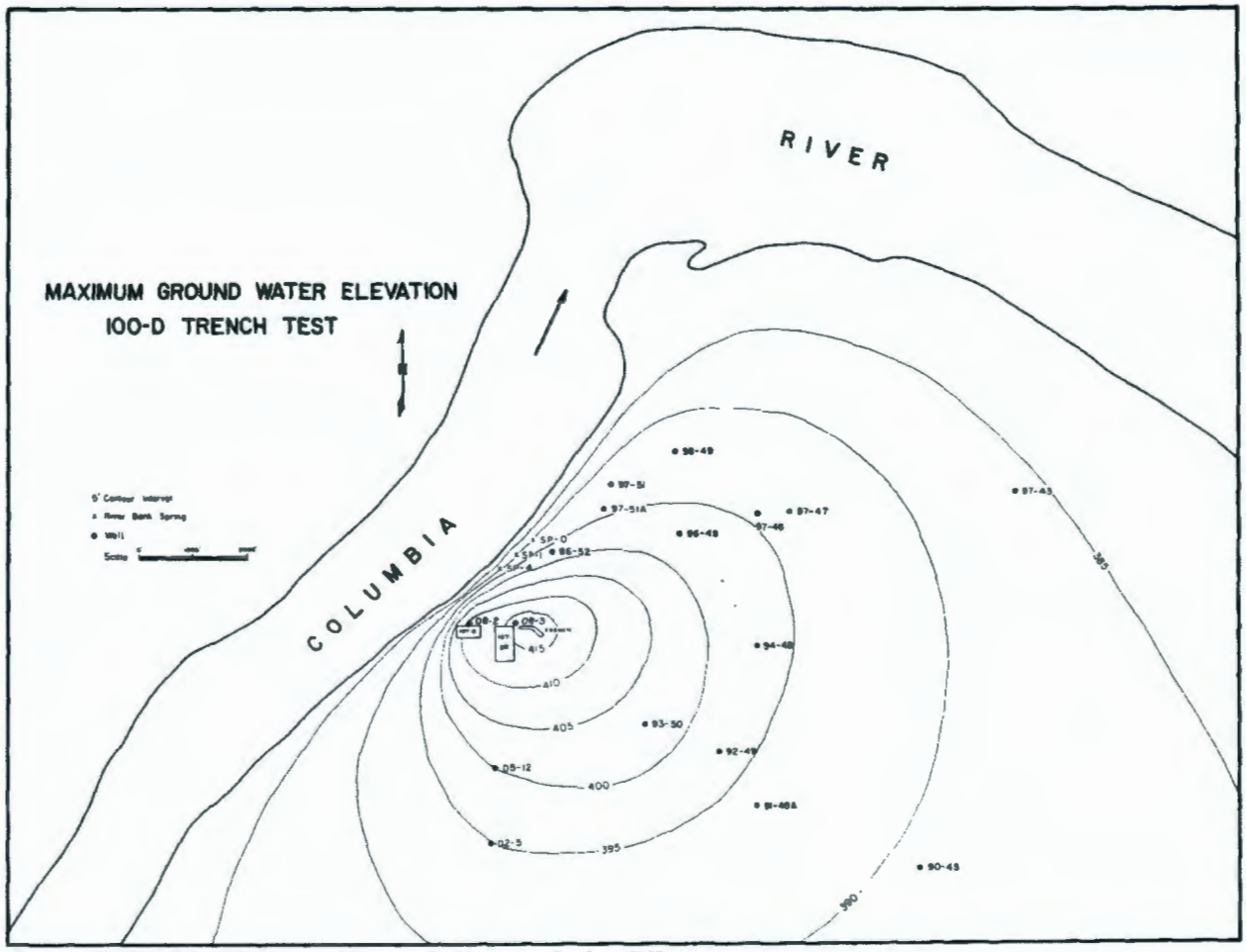


Source: BNWL-CC-1352, *Ground Disposal of Reactor Coolant Effluent.*

Figure 3-13a. Hydrologic Effects of the 1967 Infiltration Test

In addition to the reactor cooling water discharges at 100-D/H Areas, other anthropogenic activities have produced local artificial recharge to the shallow unconfined aquifer. In particular, chronic leakage from the 182-D Reservoir has produced a local groundwater mound beneath the footprint of the reservoir. This presence of this mound is apparent in two sets of observations:

- The Cr(VI) plume in the general 100-D Area has been divided by a body of uncontaminated water that is generally centered on the 182-D Reservoir. This condition does not appear to be related to natural groundwater flow.
- The basic geochemistry of groundwater samples collected from monitoring wells immediately adjacent to the reservoir is substantially different from the surrounding groundwater and exhibits characteristics consistent with the geochemistry of the Columbia River.

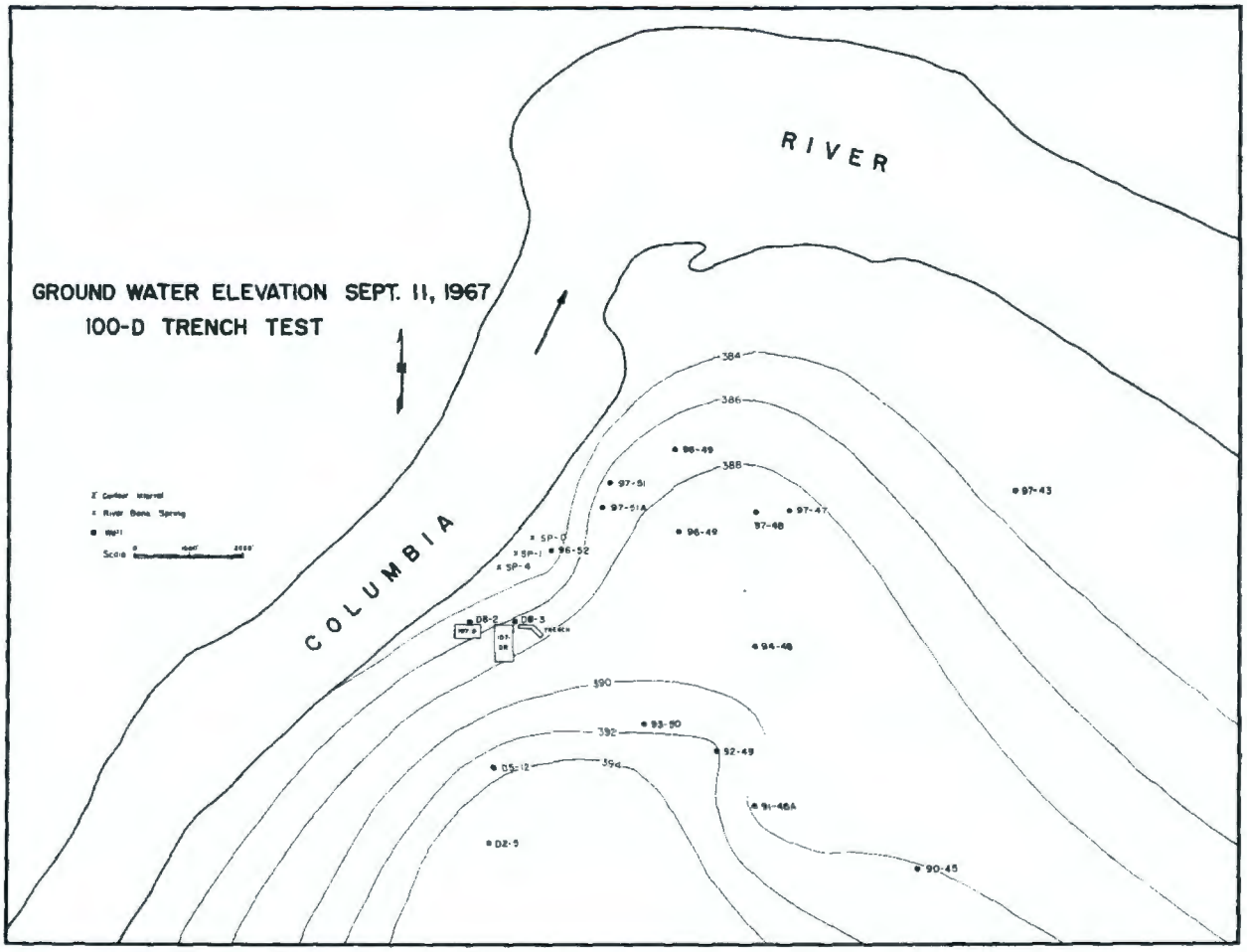


Source: BNWL-CC-1352, *Ground Disposal of Reactor Coolant Effluent.*

Figure 3-13b. Hydrologic Effects of the 1967 Infiltration Test

The 182-D Reservoir was constructed as part of the D Reactor cooling water treatment system and has also been used to store raw river water for site use, including export to the 200 Areas of the Central Plateau. The reservoir is still used as one of two sources of untreated, nonpotable water to supply the Hanford Site. Up until the last few years, the 182-D Reservoir chronically leaked enough to sustain a local groundwater mound. In recent years, the reservoir has been operated under administrative controls that limit the operating head within the reservoir to a pre-determined water level. This has reduced the apparent leakage from the reservoir substantially. Although the reservoir is expected to continue to leak, the effects on the underlying shallow unconfined aquifer are becoming apparent as the inferred distribution of the uncontaminated water that divides the two Cr(VI) plume segments appears to the shrinking. An area of very low Cr(VI) concentration is still observed associated with Wells 199-D5-33 and 199-D5-44, located near the reservoir.

Recent efforts to address the leakage have included reducing the operating water level in the reservoir and attempting to seal concrete cracks and construction joints. As presented in Chapter 4, the result has been a reduction in leakage and diminished effects on the local groundwater flow, which is seen by the merging of the northern and southern Cr(VI) plumes at 100-D.



Source: BNWL-CC-1352, *Ground Disposal of Reactor Coolant Effluent.*

Figure 3-13c. Hydrologic Effects of the 1967 Infiltration Test

3.7.1.2 Groundwater Mounding at 100-H

In 100-H, a substantial groundwater mound formed under conditions similar to those at 100-D, however, the overall magnitude of the mound was smaller, both in its height (which still approached local ground surface) and in areal extent. Again, similar to 100-D and the other Hanford reactors, cooling water was stored before treatment in the 182-H Reservoir, which may have leaked unspecified quantities of water during operation. Spent cooling water left the reactor and was held up in the 116-H-7 (107-H) Retention Basin before discharge to the Columbia River. Retention basin leaks at 100-H developed and substantial quantities of water were inadvertently released to the vadose zone beneath the basin. In addition, contaminated cooling water generated during upset conditions (e.g., fuel ruptures) at H Reactor was diverted from the retention basin to the 116-H-1 Trench and allowed to infiltrate into the vadose zone soil. Cooling water leaking from the retention basin and discharged to the trench was the source of the observed groundwater mound at 100-H.

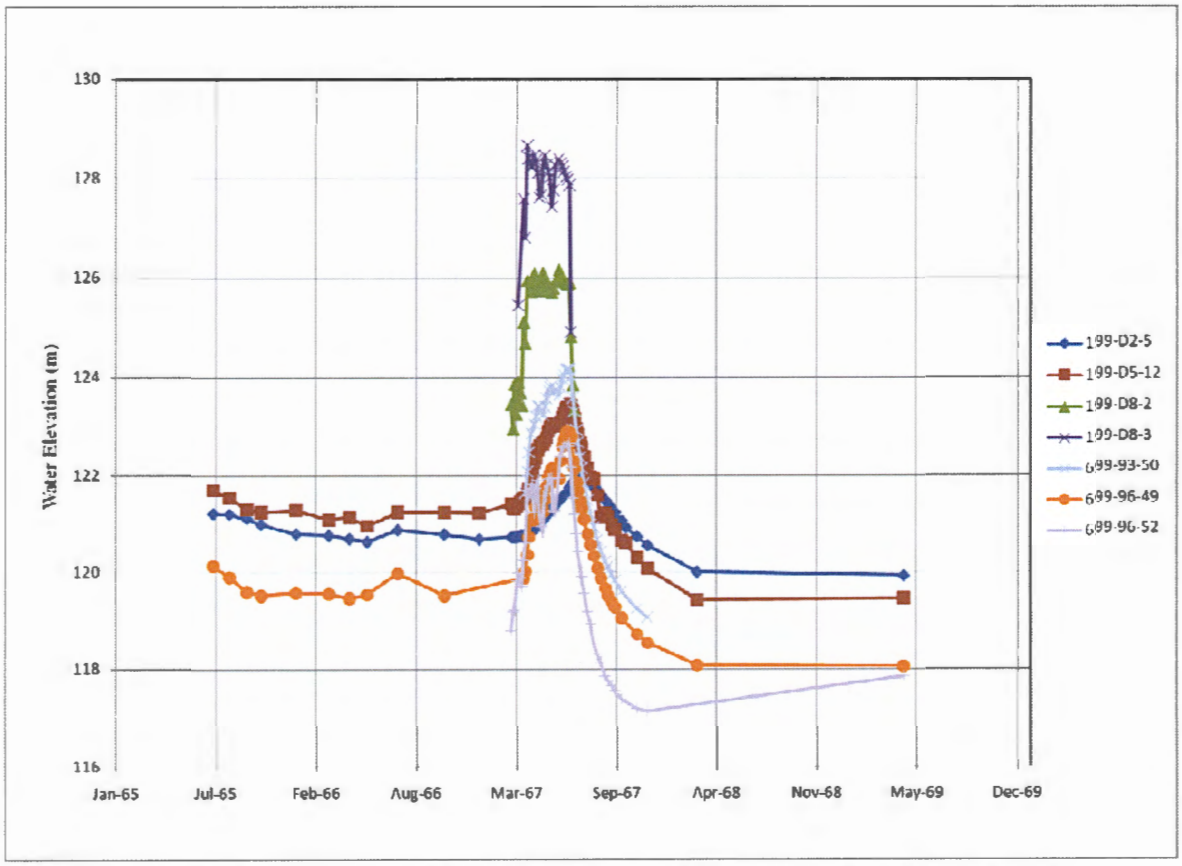


Figure 3-14. Hydrographs of Selected Wells near the 116-DR-1&2 Infiltration Trench

The groundwater mound at 100-H reached nearly to the ground surface and, like the mound observed at 100-D, exhibited elevated temperature greater than 70°C (157°F). The groundwater mound at 100-H would also have consistent largely of spent reactor cooling water, which would have displaced most of the naturally flowing groundwater in the aquifer during the operations period of the reactor. The 100-H mound formed within the coarse-textured Hanford formation that underlies the reactor area and sits directly atop the underlying RUM; the shallow unconfined aquifer at 100-H is found entirely within the Hanford formation. Groundwater beneath 100-H flowed rapidly toward the adjacent river where it discharged. Thermal springs were observed for about 1,000 m (3,000 ft) downstream from 100-H during reactor operations. Because of the 100-H facilities' proximity to the river and the high hydraulic conductivity of the underlying Hanford formation, the groundwater of the 100-H mound flowed rapidly toward the river at velocity of about 6.1 m/day (20 ft/day) and did not develop the broad extent observed at 100-D. The mounding quickly dissipated after H Reactor operations ceased in 1965.

During the early to mid-1960s, the reactor cooling water mound from 100-D operations extended all the way across the horn to 100-H, and temperature increases in cooling water withdrawn for 100-H operations were partially attributed to the elevated groundwater temperature because of the 100-D contribution.

3.7.1.3 Pump and Treat System Influences

In 1997 and 2004, two interim groundwater pump-and-treat systems (HR-3 and DR-5, respectively) were installed in 100-D/H. Although these systems worked effectively at reducing contaminant concentrations in the unconfined aquifer and the first water-bearing unit in the RUM, the systems had only localized influence on the groundwater flow regime. The DR-5 and HR-3 systems were shut down in April and

May 2011, respectively. The new 100-DX and 100-HX systems came online December 17, 2010, and October 1, 2011, respectively.

3.7.1.4 Horizontal Hydraulic Gradients and Flow Velocities

While reactor-related activities took place (including disposal of wastewater to the subsurface), groundwater flow velocities between the wastewater disposal areas and the Columbia River increased considerably from predevelopment rates. Several years after operations started in the 100-D and 100-H Areas, it was discovered that leaks from the retention basins and associated pipelines caused significant groundwater mounding under the retention basins, greatly increasing the gradients and groundwater velocities between the basins and the Columbia River. The gradients formed were sufficient to cause riverbank thermal springs near the retention basins in both areas.

In 1962, a study was undertaken to determine the effects that the thermally hot groundwater might have on reactor operations (*Status of the Ground Water Beneath Hanford Reactor Areas January, 1962 to January, 1963* [HW-77170]). At that time, groundwater velocities near the 100-D and 100-H Retention Basins were estimated to range from about 3.5×10^{-3} cm/sec (10 ft/day) to about 1.06×10^{-2} cm/sec (30 ft/day). Figure 3-14 depicts approximate groundwater elevations in 1962. Groundwater velocity at 100-D exhibited the greatest variation; velocity directly toward the river from the retention basins was about 3.5×10^{-3} cm/sec (10 ft/day), somewhat moderated by the presence of the Ringold unit E material in the aquifer, the velocity of groundwater flowing across the horn toward 100-H was substantially greater at about 1.06×10^{-2} cm/sec (30 ft/day). The velocity at 100-H, flowing from the vicinity of the retention basin toward the river, was about 7.0×10^{-3} cm/sec (20 ft/day). Given the period of reactor operations in both areas, the groundwater elevation contours presented on Figure 3-14 likely represent the approximate size and configuration of the groundwater mounds at their peaks, although the mound beneath 100-D would have grown substantially during the cooling water injection test conducted in 1967. The calculated groundwater velocity during the 100-D injection test was 1.75×10^{-2} cm/sec (50 ft/day) based on reduction of measured iodine-131 activity concentrations in the cooling water and in the groundwater subsequently discharged at thermal springs along the river (*Ground Disposal of Reactor Coolant Effluent* [BNWL-CC-1352]).

Discharges of wastewater to the various trenches and basins declined with the sequential cessation of reactor operations 1964 (DR Reactor), 1965 (H Reactor) and 1967 (D Reactor). Water level data obtained since 1967 suggest that conditions approaching predevelopment horizontal hydraulic gradients were largely restored by about 1968 or 1969 (Figure 3-13).

The effects of wastewater infiltration on patterns of groundwater flow and contaminant migration near the 100-D and 100-H Area reactors and associated trenches and basins are detailed further in Chapter 5. Water level maps are used to depict patterns of flow inland from the reactors and associated wastewater disposal areas and the likely effect of these groundwater flow patterns on contaminant migration.

3.7.1.5 Vertical Gradients

During operation of the reactors, infiltration and overland flow of contaminated cooling water from surface features and from leaks at the 100-D and 100-H Area retention basins created significant vertical (downward) fluxes within the vadose zone that would have increased the potential for vertical migration of contaminants released to the aquifer. Although historical water level data from River Corridor reactor areas during this period are from wells with similar screened intervals (making direct assessments of vertical gradients difficult), qualitative evaluation of the mounding conditions suggests that vertical hydraulic gradients exerted by the intense artificial recharge must have been significant.

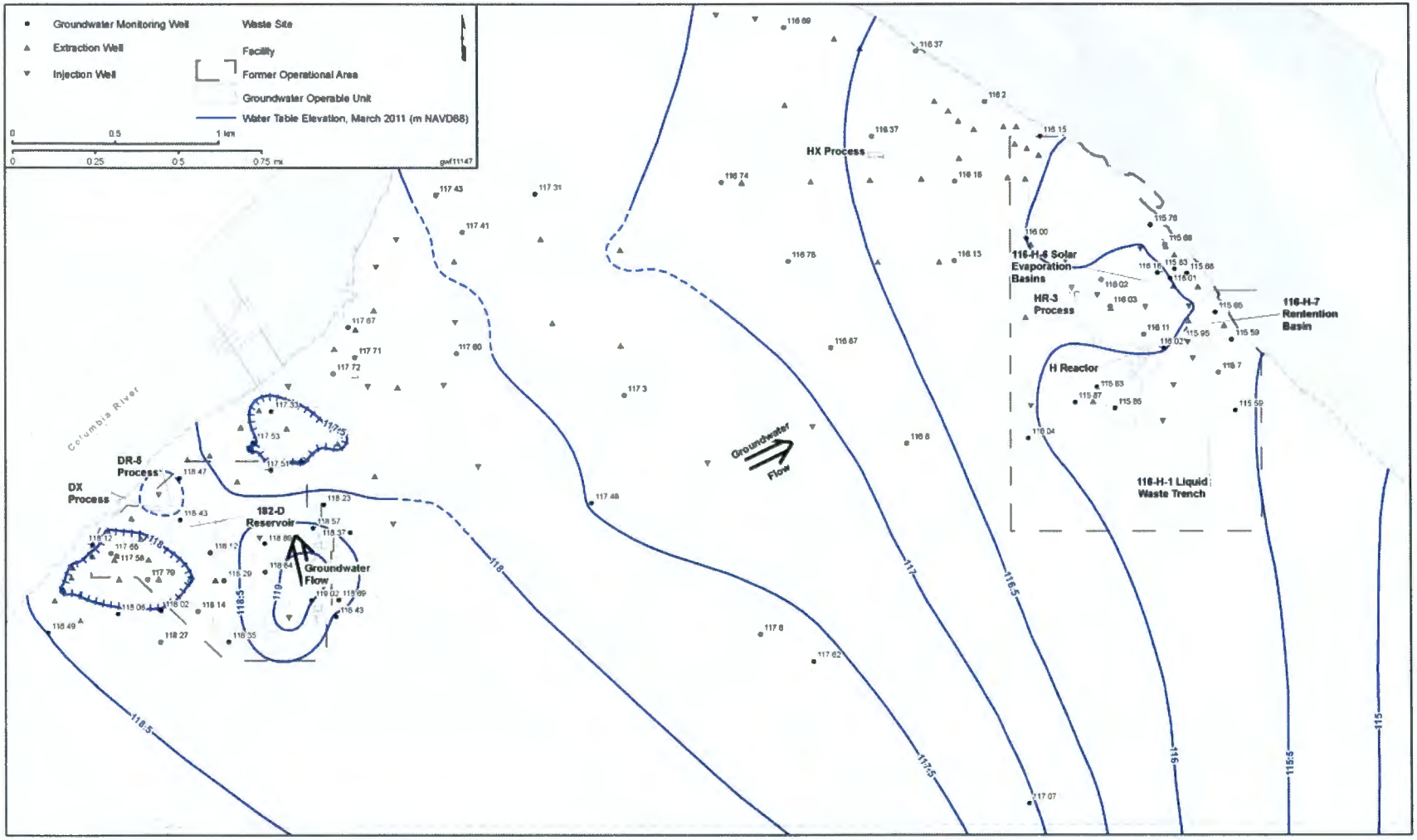
3.7.2 Current Groundwater Flow Conditions

Since the cessation of reactor operations and associated wastewater disposal, hydraulic gradients and groundwater flow have largely returned to their predevelopment direction toward the Columbia River, with variations in response to changes in the stage of the now actively managed Columbia River, which are dictated by the spring snowmelt, summer season, and controlled releases at the Priest Rapids Dam. Throughout the year, hydraulic gradients steepen toward the river during low river stage (fall and winter), and flatten or may reverse near the shoreline during high river stage. Superimposed on these longer-term fluctuations are daily and weekly fluctuations arising from controlled releases at the Priest Rapids Dam. Historically, the water table elevation ranges from approximately 117 m (384 ft) in the 100-D and central horn areas to approximately 115 m (377 ft) in 100-H (*North American Vertical Datum of 1988* [NAVD88]). The seasonal high river stage on the Hanford Reach of the Columbia River coincides with the spring snowmelt and typically extends from May through July and seasonal low river stage is generally from September through the early winter.

Data obtained from river gauges along the Hanford Reach indicate that high river stage can be more than 3 m (10 ft) higher than low river stage. River stage can also fluctuate several meters over short periods (hours to days), based on operations at Priest Rapids Dam (*Remedial Design and Remedial Action Work Plan for the 100-HR-3 and 100-KR-4 Groundwater Operable Units' Interim Action* [DOE/RL-96-84]). Depending on local geology, changing river stage can influence groundwater elevations up to several hundred meters inland. The groundwater level response to changes in river stage is slower and of less magnitude farther inland than near the river. However, effects have been observed as far inland as Gable Gap, approximately 3,600 m (2.2 mi) to the southeast (*Hanford Site Groundwater Monitoring for Fiscal Year 2008* [DOE/RL-2008-66]). Groundwater elevations have varied by up to 1.0 m/day (3.3 ft/day) in some wells nearest the river and up to approximately 1.8 m (6 ft) over the season in a few wells (*Monitoring Groundwater and River Interaction Along the Hanford Reach of the Columbia River* [PNL-9437]).

Water table maps are presented to illustrate groundwater elevations and groundwater flow under three different seasonal conditions: when the average river stage is at the annual low (September), when the river stage is intermediate (March), and when the river stage peaks (June). Groundwater elevation data displayed on the June 2010 and September 2010 contour maps represent pre-DX/HX pump-and-treat system groundwater elevations. However, the DR-5 and HR-3 pump-and-treat systems were still operating. The March 2011 contour map represents conditions where only the DX system was online.

The March 2011 groundwater contour map represents flow conditions during intermediate river stage (i.e., average conditions; Figure 3-15) over the entire 100-HR-3 OU. Figure 3-15 illustrates that under current conditions, groundwater enters the 100-HR-3 from the south and generally flows toward the Columbia River. Much of the regional flow is toward the northeast and 100-H. A lesser portion flows north/northwest toward 100-D, which is now influenced by pumping and injection. From the area northeast of 100-D, groundwater flows across the horn to the east-northeast and toward 100-H. Evidence indicates that the DX pump-and-treat system is influencing the groundwater flow regime beneath 100-D. Two groundwater depressions near the river are caused by DX extraction wells. In addition, a groundwater mound is nearly centered beneath the reactors, which is caused by DX injection wells. Flow away from the injection wells is designed to push contaminants toward the extraction wells.



Source: NAVD88, *North American Vertical Datum of 1988.*

Figure 3-15. 100-D/H Water Table Map – March 2011

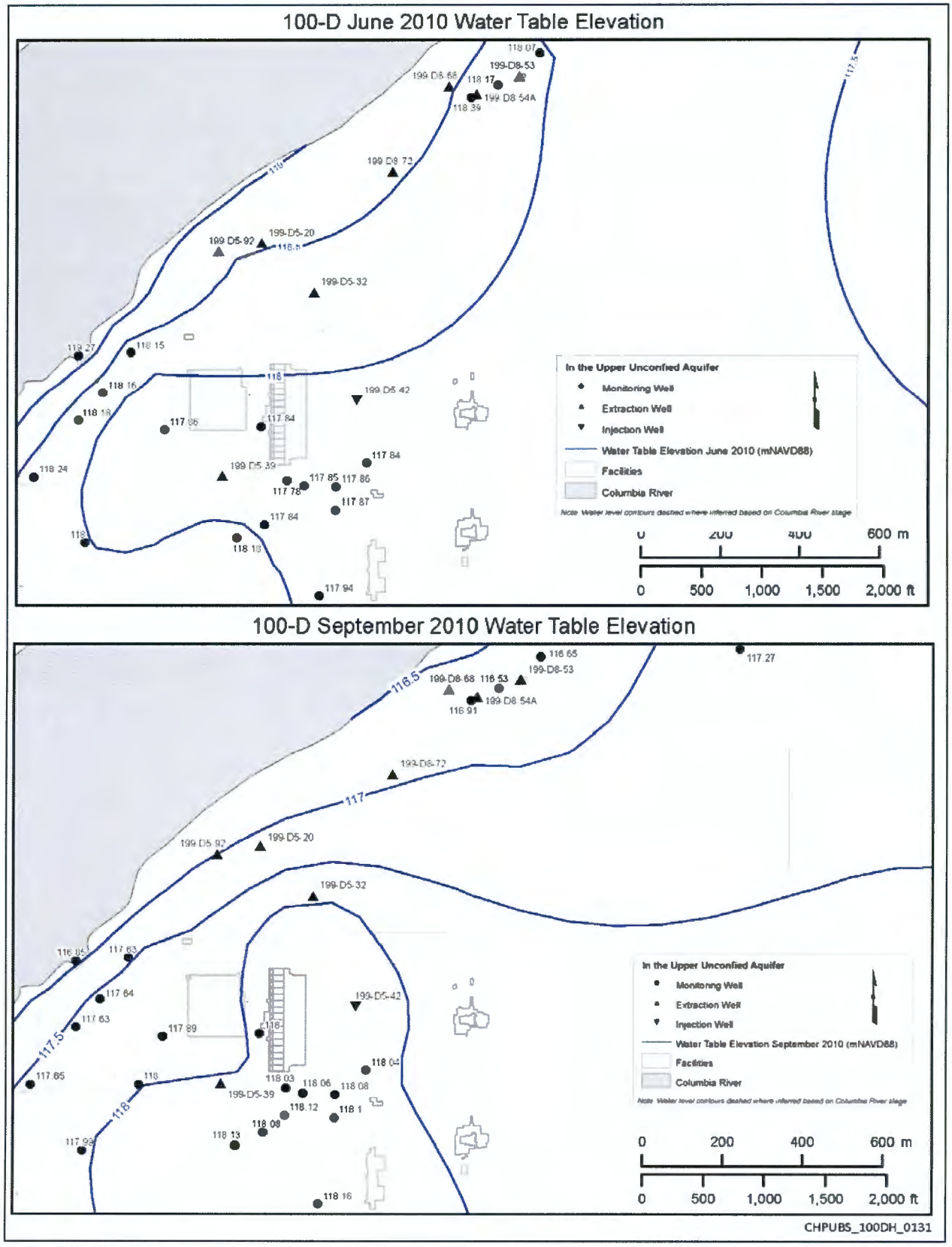
Two additional groundwater contour maps were constructed to show close-up profiles of 100-D (Figure 3-16) and 100-H (Figure 3-17) during high and low river stages. These maps show the effects of river stage on the groundwater flow regime near the river, where the effect is more pronounced than it is inland or in the horn area. The map shows high river stage conditions at 100-D in June 2010, where the 119 m (390 ft) equipotential line is near the river, and the next two consecutive equipotential lines decrease inland to 118.5 and 118 m (388.8 and 387 ft). This decrease indicates the river is flowing into the aquifer, so the river is referred to as a "losing stream." The flow is generally southeast, away from the river. The 118 m (387 ft) equipotential line is distorted around extraction Well 199-D5-39, which was extracting groundwater for the former DR-5 pump-and-treat system. In 100-D, the September 2010 map on Figure 3-16 shows low river stage conditions, where the 118 m (387 ft) equipotential line is inland; then, the next three consecutive equipotential lines decrease from 117.5 to 116.5 m (385 to 382 ft) at the river. This decrease indicates groundwater is discharging to the river. The general flow direction is northwest, north, and northeast depending on the location in 100-D. The influence of the former DR-5 pump-and-treat system is observed with the 118 m (387 ft) equipotential line.

The June 2010 map on Figure 3-17 represents high river stage conditions in 100-H, where the 117 m (383.8 ft) equipotential line is near the river, with the next equipotential line decreasing inland to 116 m (380.5 ft). This potentiometric head difference indicates that during period of high river stage, water enters the aquifer from the river and may migrate some distance inland at a velocity determined by the head difference. The oval-shaped 116 m (380.5 ft) equipotential line is likely a combination of effects from high river stage and the former HR-3 pump-and-treat extraction wells that were operating. The September 2010 map on Figure 3-17 represents low river stage conditions in 100-H, where equipotential lines converge toward the river from high to low elevation. This indicates groundwater is discharging to the river. The general flow direction is northeast and east depending on the location in 100-H. The influence of the former HR-3 pump-and-treat system is evident near Well 199-H4-3, which causes a steeper hydraulic gradient.

Groundwater flow direction reversals have been documented in 100-D and 100-H (*Conceptual Site Models for Groundwater Contamination at 100-BC-5, 100-KR-4, 100-HR-3, and 100-FR-3 Operable Units* [BHI-00917]; *Geohydrologic Characterization of the Area Surrounding the 183-H Solar Evaporation Basins* [PNL-6728]). Over the course of each year, however, groundwater exhibits a net discharge to the Columbia River from 100-D/H.

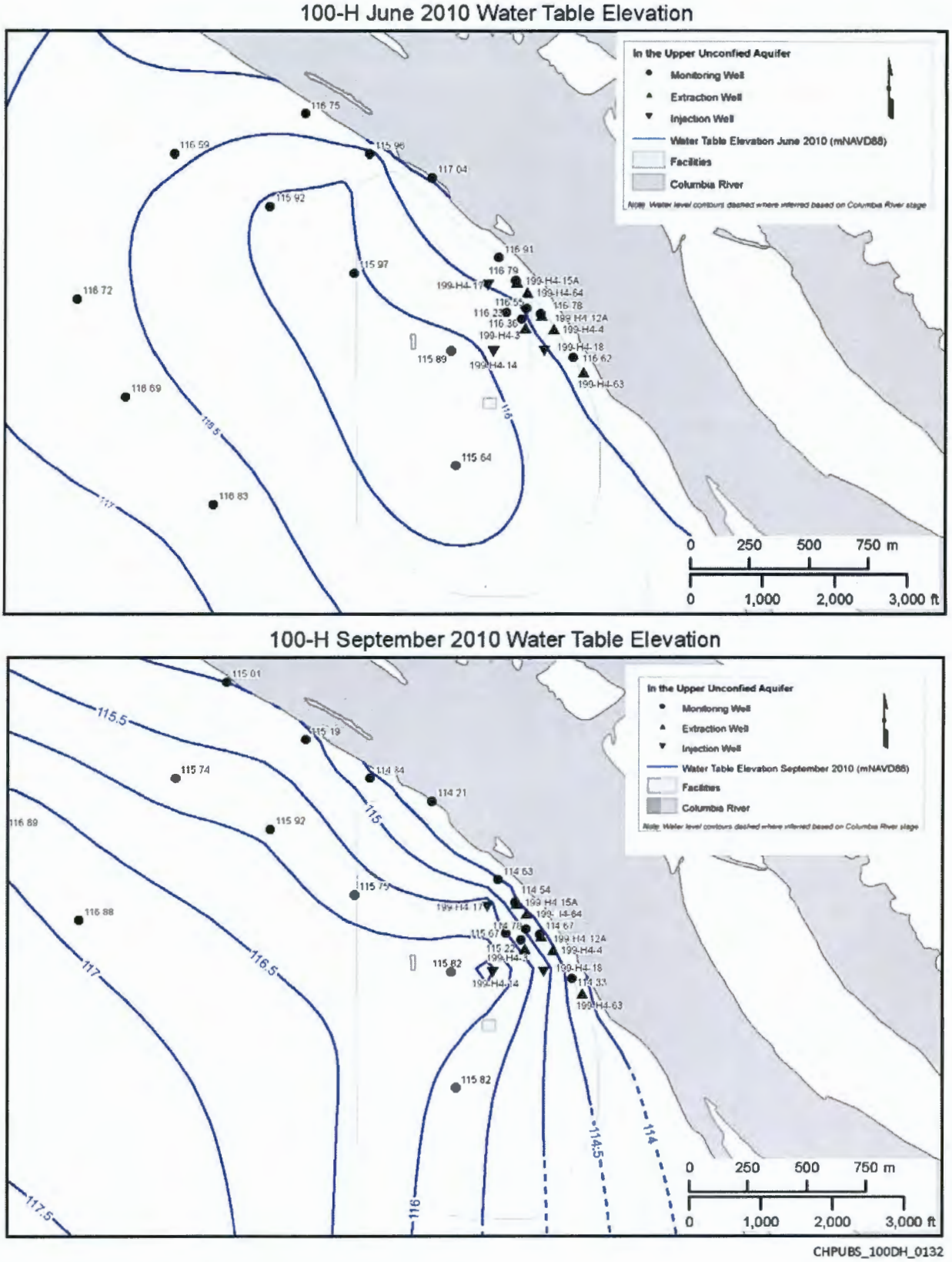
Figures 3-18 and 3-19 show river stage elevation at 100-D and 100-H river gauges versus groundwater elevations in selected wells in each area, respectively. Figures 3-18 and 3-19 show the annual and diurnal cycles in river stage fluctuations and the translation of those stage changes into the adjacent aquifer. The river stage fluctuates as much as 4.6 m (15 ft) during the year and some days by as much as 2.7 m (9 ft) (*Monitoring Groundwater and River Interaction Along the Hanford Reach of the Columbia River* [PNL-9437]), depending on how water is released from Priest Rapids Dam upstream from the Hanford Site. The fluctuations in river stage create a cyclic rise and fall of the water table in the aquifer adjacent to the river, the effects of which can be observed hundreds of meters inland. This zone between the high and low water table is termed the periodically rewetted zone.

Figure 3-15 shows that the 100-DX pump-and-treat system has a significant influence on groundwater flow at 100-D. With the startup of the 100-HX pump-and-treat system, which includes pumping of the RUM first water-bearing unit from Wells 199-H4-12C and 199-H3-2C in fall 2010, hydraulic gradients are now altered. Operation of the DX and HX pump-and-treat systems will result in gradient effects caused by extraction and injection wells.



Source: NAVD88, *North American Vertical Datum of 1988.*

Figure 3-16. 100-D Water Table Maps – June and September 2010



Source: NAVD88, *North American Vertical Datum of 1988.*

Figure 3-17. 100-H Water Table Maps – June and September 2010

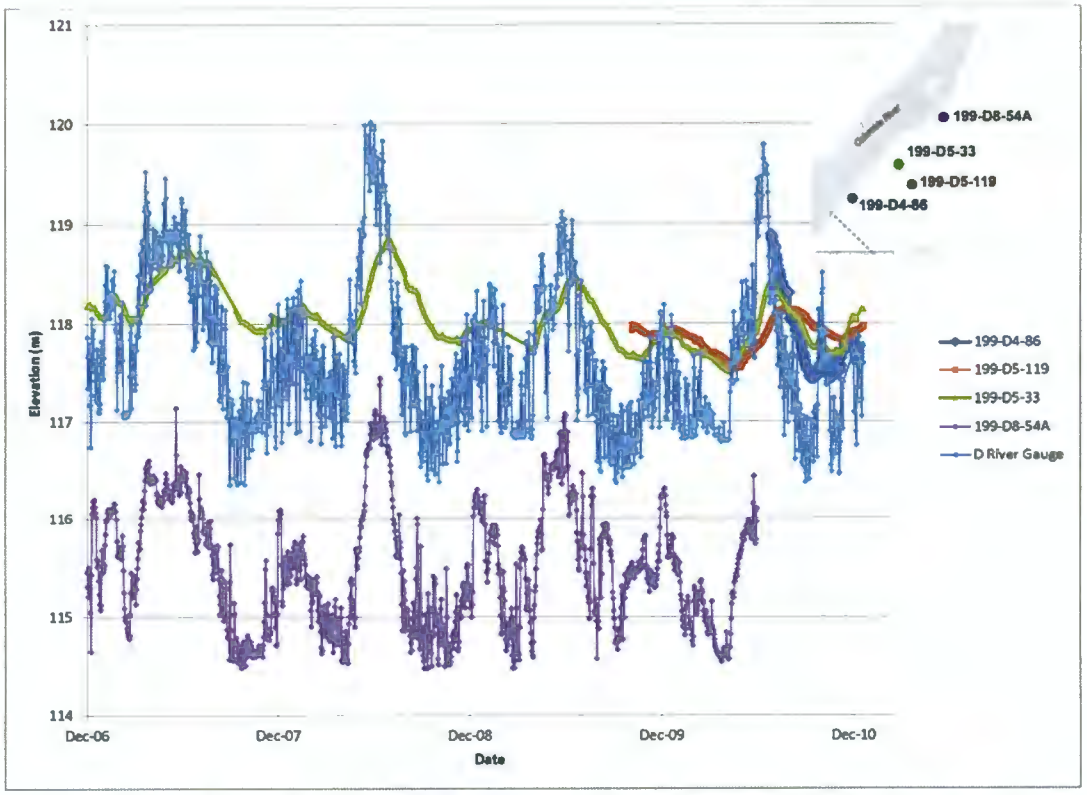


Figure 3-18. River Stage at 100-D Compared to Water Elevations in Selected Wells

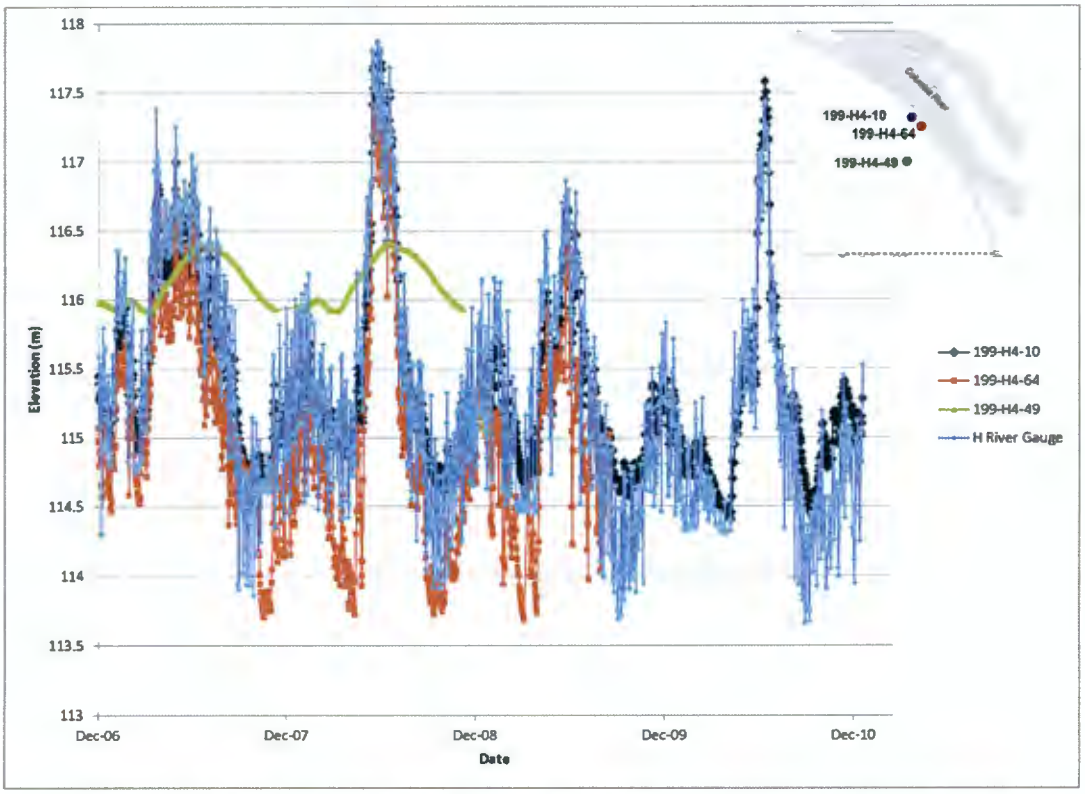


Figure 3-19. River Stage at 100-H Compared to Water Elevations in Selected Wells

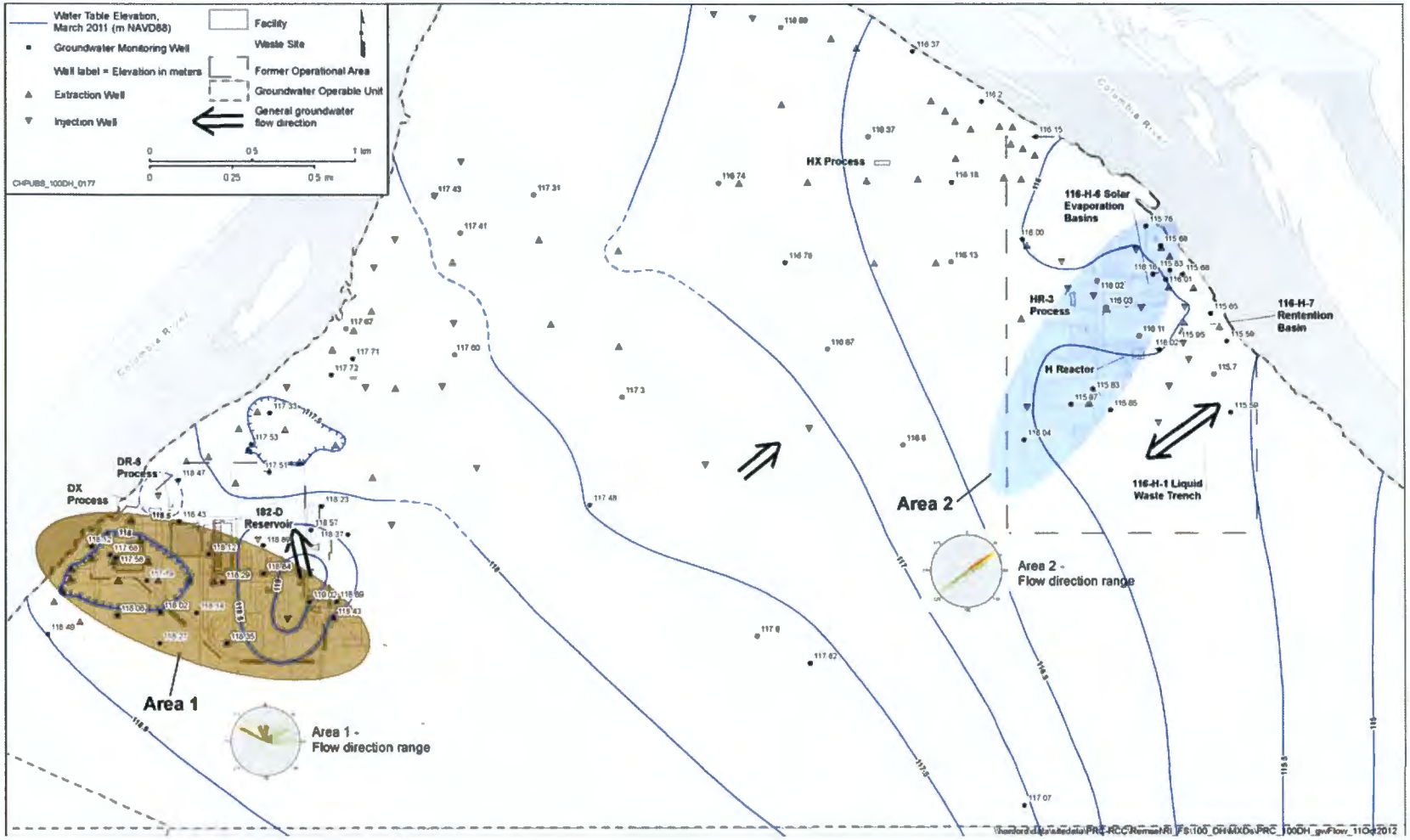
3.7.2.1 Horizontal Gradients and Groundwater Velocities

Current hydraulic gradients within the unconfined aquifer are generally toward the Columbia River, but the gradients show some seasonal variation in response to changes in river stage. Gradients steepen toward the river during low river stage (fall and winter), and flatten or may reverse near the river shoreline during high river stage (spring). Local gradients are also influenced by the operation of the pump-and-treat systems in 100-D/H.

The 100-HR-3 groundwater pump-and-treat systems were reconfigured and expanded in 2011 following RPO activities. Four groundwater pump-and-treat systems operated for all or part of 2011:

- In the 100-D Area, the DX system operated for the entire calendar year, while the DR-5 system operated from January to April.
- In the 100-H Area, the HR-3 system operated from January to May, while the HX system operated from late September to December.
- To evaluate the variation in the groundwater gradient direction and magnitude at 100-D/H, as influenced by current pump-and-treat operations, a three-point gradient analysis was performed using groundwater levels measured during 2011. This activity is described in *Estimation of Gradient and Flow Velocity in the 100-D/H Area* (ECF-100DH4-12-0013) in Appendix F. A "mesh" of triangles was created between monitoring wells that are outfitted with dataloggers and transducers that record groundwater levels continuously. With some exceptions (detailed below), each triangle in the mesh, referred to as an element, is defined by three monitoring wells. A gradient vector consisting of a magnitude and azimuth (direction) is calculated for each element, using groundwater levels measured in the three wells (Figure 3-20). For this analysis, weekly gradients were calculated for each element, using weekly average groundwater elevation measurements in monitoring wells. The presence of extraction or injection wells within any one three-point element introduces some degree of uncertainty in the net calculated gradient. Injection and extraction wells may exert effects on the direction or magnitude of gradient within the element.

The three-point gradient method is most effective if water levels vary linearly between the three wells used to define the triangular element. If an injection or extraction well lies inside an element, however, water level mounding or depression generated by the injection or extraction well will result in a different gradient than would be calculated assuming a planar water table passing through the three monitoring wells. Element triangles were therefore drawn such that injection wells lie outside of the triangles. If it was not possible to draw appropriate triangles using existing monitoring wells, water levels at the triangle vertices were inferred from weekly average water level maps prepared for *Calendar Year 2010 Annual Summary Report for the 100-HR-3 and 100-KR-4 Pump-and-Treat Operations and 100-NR-2 Groundwater Remediation* (DOE/RL-2011-25). These water level maps were calculated using a universal kriging technique that explicitly accounts for the effects of injection or extraction on groundwater levels (*Collection and Mapping of Water Levels to Assist in the Evaluation of Groundwater Pump-and-Treat Remedy Performance* [SGW-42305]).



Source: NAVD88, *North American Vertical Datum of 1988.*

Figure 3-20. Water Table Elevation Map for 100-D/H (March 2011) Indicating Inferred Groundwater Flow Directions

Results of the three-point gradient analysis suggest geographic variations in average hydraulic gradients that can be broadly grouped in three general areas (shown in Figure 3-20) as follows:

- **Area 1:** The area near the 100-D southern plume that is now the target of the 100-DX remedy (example Element #7).
- **Area 2:** The area near 100-H that is now the target of the 100-HX remedy (example Element #76).
- **Area 3:** The area between 100-D and 100-H, referred to as the horn. Because of relatively sparse data during 2011, a hydraulic gradient rose diagram is not presented for the horn area.

Radial diagrams illustrating gradient magnitude and direction in Areas 1 and 2 are presented on Figures 3-21 and 3-22. At 100-D, an azimuth direction of approximately 310 degrees would indicate a flow direction perpendicular to the Columbia River. At 100-H, an azimuth direction of approximately 45 degrees would indicate a flow direction perpendicular to the Columbia River. The radial diagrams illustrate the variations in weekly average gradient for representative elements in each of the two general areas identified above. The direction that the lines point indicates the calculated azimuth direction (that is, the flow direction). The length of the line indicates the relative magnitude of the groundwater gradient. The line colors reflect the general seasonality of the observations: blue indicating spring, green indicating summer, yellow indicating fall, and red indicating winter.

In Area 1, hydraulic gradients during 2011 varied in magnitude from approximately 0.0014 to 0.0023 for the period May through August, and from approximately 0.0017 to 0.0031, for the period September through April. The gradient direction was to the north/northwest toward the Columbia River for most of the year; however, gradients shifted to the north/northeast for a brief period from May through August 2011, coinciding with high stage in the Columbia River. The flow direction in this area exhibited a range of approximately 200 degrees azimuth over the high river-stage period of May to August, but only 70 degrees azimuth during the period September through April. Time-series and radial graphics showing the weekly gradients for Area 1 are presented on Figure 3-21.

In Area 2, hydraulic gradients varied in magnitude from about 0.0014 to 0.0036 for the period May through August, and from approximately 0.0015 to 0.0046 for the period September through April. The gradient direction was generally north/northeast toward the Columbia River; however, gradients shifted to the south/southeast for a brief period from May through August 2011, coinciding with high stage in the Columbia River. The flow direction in this area exhibited a range of approximately 128 degrees azimuth over the high river-stage period of May to August, but only 70 degrees azimuth during the period September through April. Time-series and radial graphics showing the weekly gradients for Area 2 are presented on Figure 3-22.

Hydraulic gradients in Area 3 during 2011, consisting of the horn, are difficult to enumerate because of widely varying monitoring frequencies at the wells that comprise the gradient elements. Available data indicate that areas relatively close to the shore of the Columbia River varied in azimuth from north/northwest during times of very low river stage (that is, toward the Columbia River at the northern side of 100-D), to south/southeast (that is, away from the Columbia River) during times of very high river stage, with periods of relatively flatter gradients to the west and west-southwest at times of intermediate river stage. At locations more distant from the Columbia River, gradients appear to be more systematically to the west across the horn.

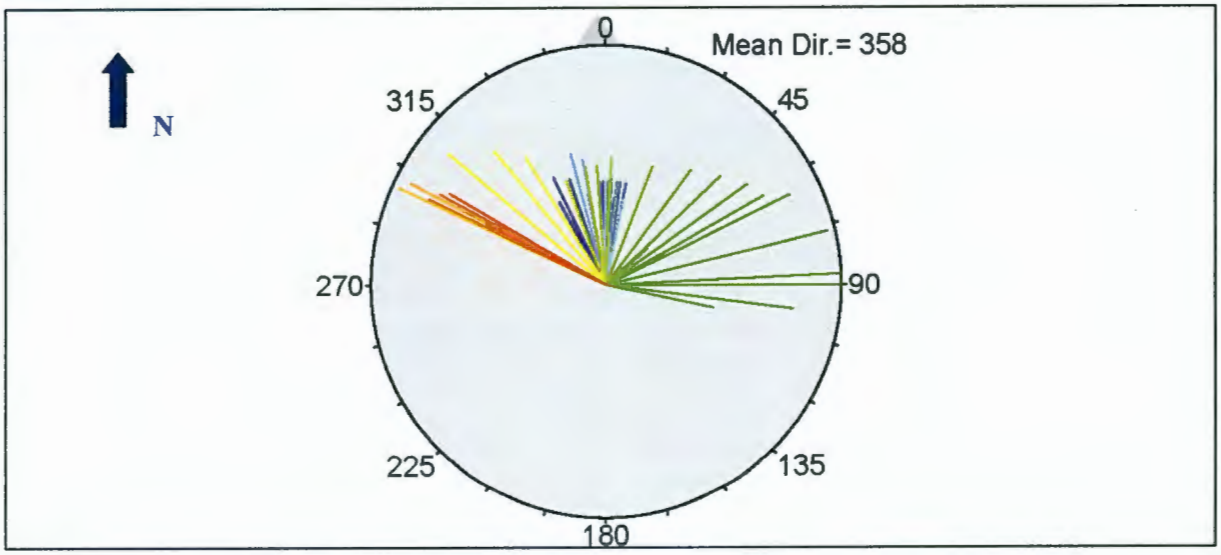


Figure 3-21. Weekly Average Gradient in 2011 for a Representative Element in Area 1

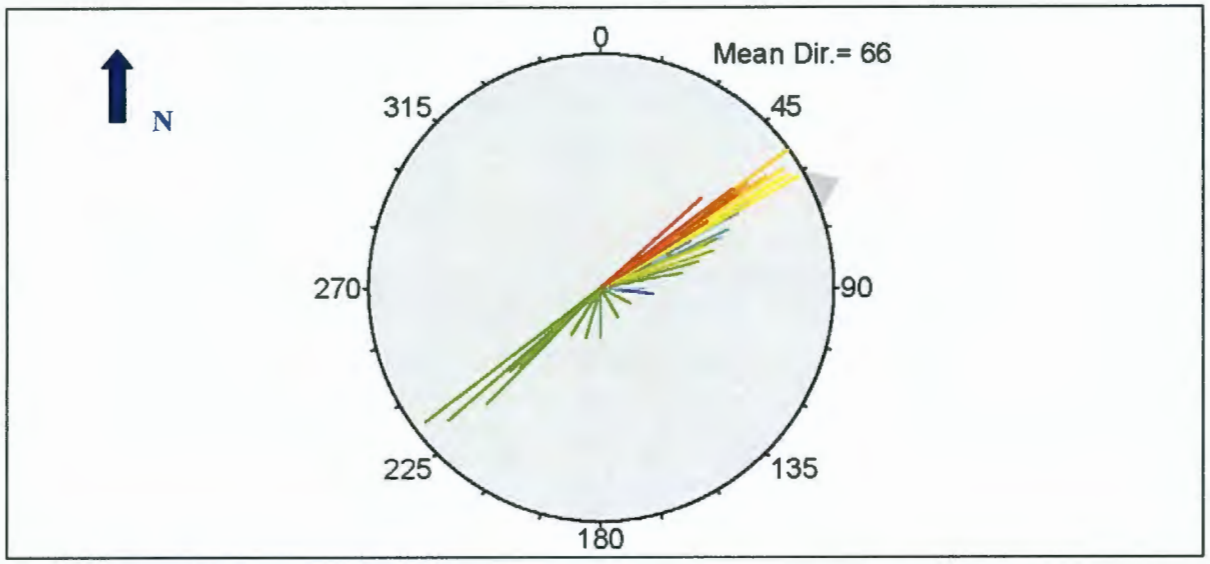


Figure 3-22. Weekly Average Gradient in 2011 for a Representative Element in Area 2

Representative ranges of average gradient magnitude and direction, hydraulic conductivity, and groundwater velocity for Areas 1 and 2 are summarized in Table 3-13. The tabulated range of values for gradient direction in Table 3-13 has the following limitations:

- Excludes eccentric elements
- Excludes elements with limited temporal coverage
- Does not reflect complete reversal that occurred at some wells primarily in response to groundwater pump-and-treat operations at nearby wells and in response to the changes in the Columbia River stage

Table 3-13. Typical Hydraulic Gradients and Groundwater Velocities for 2011

Area	Typical Hydraulic Gradient Magnitude	Typical Flow Direction (Azimuth)	Hydraulic Conductivity in Corresponding Formation (cm/sec)	Groundwater Flow Velocity in Corresponding Formation (cm/sec)
1	0.001 - 0.005	223 - 358	2.31×10^{-4}	1.29×10^{-6} - 6.43×10^{-5}
2	0.0008 - 0.003	23 - 106	6.94×10^{-4}	3.09×10^{-6} - 1.16×10^{-5}

3.7.2.2 Vertical Gradients

Four factors influence the evaluation of vertical hydraulic gradients across 100-D/H, making a detailed evaluation of vertical hydraulic gradients difficult:

- Throughout much of 100-D/H, the thickness of the unconfined aquifer is quite small, ranging from 2 to 3 m (6.6 to 9.9 ft) in areas of 100-H and the horn, up to about 8 to 10 m (26.2 to 32.8 ft) in areas of 100-D.
- The current monitoring well network consists mainly of wells screened within the Ringold Formation unit E within 100-D and in the Hanford formation in 100-H and most of the horn. Although screened intervals vary between wells, the screened intervals of neighboring wells often overlap because of the desire to monitor certain intervals within the aquifer.
- Natural stresses, such as recharge, that would result in significant vertical gradients are limited, except close to the Columbia River where three-dimensional flow occurs in response to stage-driven recharge-discharge cycles.
- Operation of the extensive pump-and-treat extraction and injection wells, which by design, generates vertically and horizontally convergent/divergent flow, overwhelming ambient vertical gradient patterns.

During the RI, groundwater levels were measured in boreholes drilled to the lower water bearing units (199-D5-134, 199-D5-141, 199-H2-1, 199-H3-9, and 199-H3-10) to supplement the existing dataset. In addition, static potentiometric groundwater surface levels were measured in completed wells, which were all screened in the first water-bearing unit of the RUM. Water levels were collected during drilling and subsequently collected from the completed well.

At 100-D, there are three wells completed in the first water bearing unit of the RUM. These are: 199-D5-134, 199-D5-141, and 199-D8-54B. Of these wells, only Well 199-D8-54B was installed as a nested well pair, with its sister Well 199-D8-54A, which is completed in the unconfined aquifer. Initial evaluation of the water levels during drilling for the RI wells indicates the potential for a slight upward gradient in both 199-D5-134 and 199-D5-141. However, water level information collected during drilling can be misleading because of the raising and lowering of drill casing during drilling activities which may open up or seal off various water bearing units. A more reliable evaluation can only be conducted following well completion and development.

Both RI wells at 100-D have had a minimal number (one to two) of water level measurements taken since the well was completed and developed. RI Well 199-D5-134 is located near both extraction and injection wells, which influence the water table. Cross gradient Well 199-D5-131 is currently an extraction well,

and cross gradient Well 199-D5-42 is an injection well. As shown in Figure 3-23; top, the water levels at Well 199-D5-134 track closely to the nearest downgradient Well 199-D5-13.

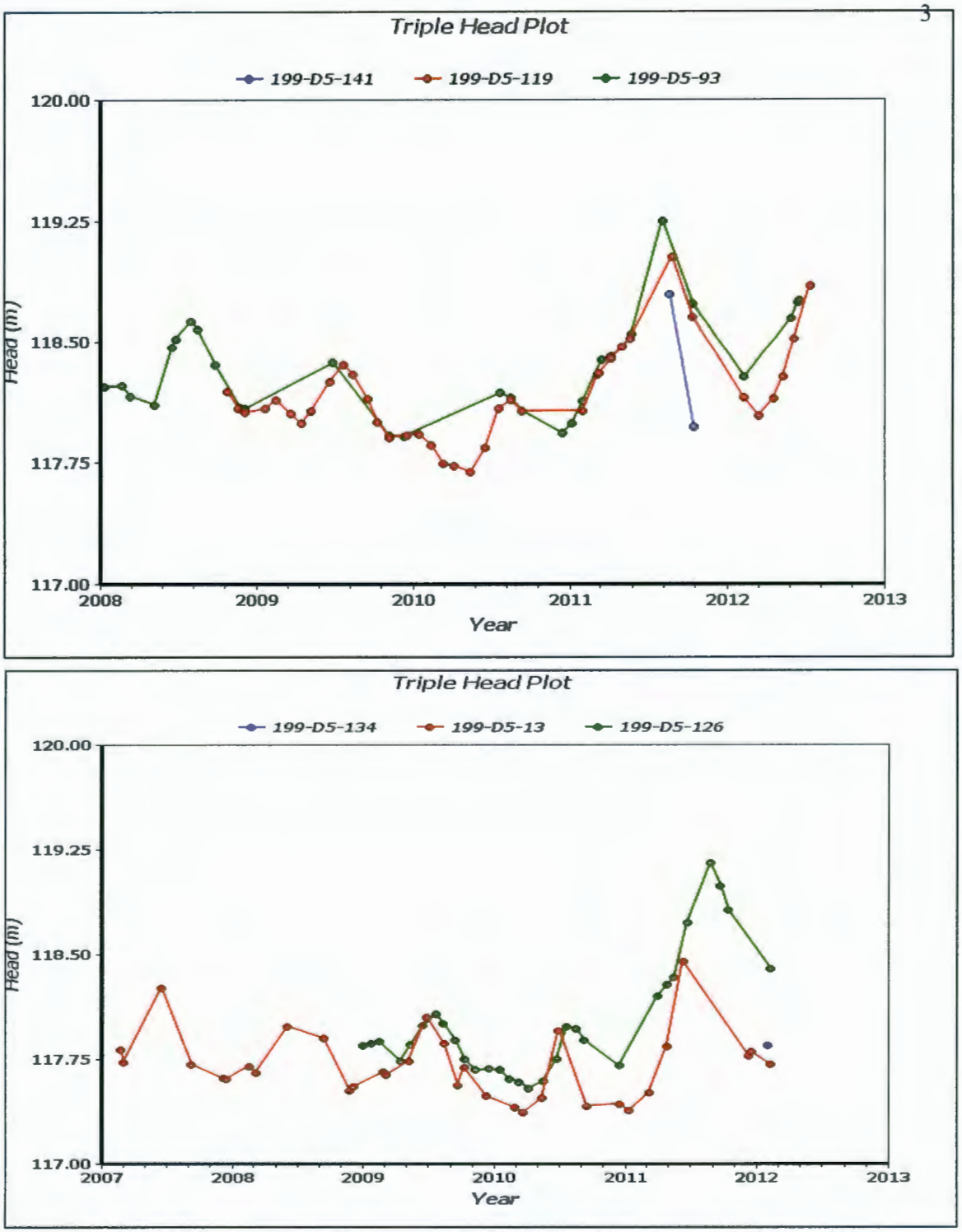


Figure 3-23. Comparison of Piezometric Head Observed in Selected Wells Completed Within the Shallow Unconfined Aquifer and the Uppermost Water-Bearing Unit of the RUM at 100-D

Nearby Well 199-D5-126 also tracked closely with that well until mid-2010, which correlates with the startup of the DX pump-and-treat system, indicating that this well is being influenced by that system. At Well 199-D5-141, water levels in the nearby Wells 199-D5-119 and 199-D5-93 both track closely with the RI well (Figure 3-23; bottom). While this interpretation is based on limited data, it suggests that the vertical gradient between the first water bearing unit of the RUM (as exhibited in Well 199-D5-93) and the unconfined aquifer at Well 199-D5-134 is near zero, with a slightly downward vertical gradient at Well 199-D5-141.

Only one well pair is present at 100-D. These are Wells 199-D8-54A and 199-D8-54B. Typically, well pairs are constructed close together and provide excellent information when evaluating the vertical gradient between aquifers. As shown in Figure 3-24, water levels in these two wells are essentially identical, with the head difference being consistently lower by 0.01 to 0.35 m (0.03 to 1.2 ft) in the unconfined aquifer as compared to the RUM. This indicates that there is small, but consistently upward, gradient between the unconfined aquifer and the first water bearing unit of the RUM. Since none of the wells at 100-D have been completed in the lower water-bearing units, the vertical hydraulic gradient associated with those lower units cannot be determined.

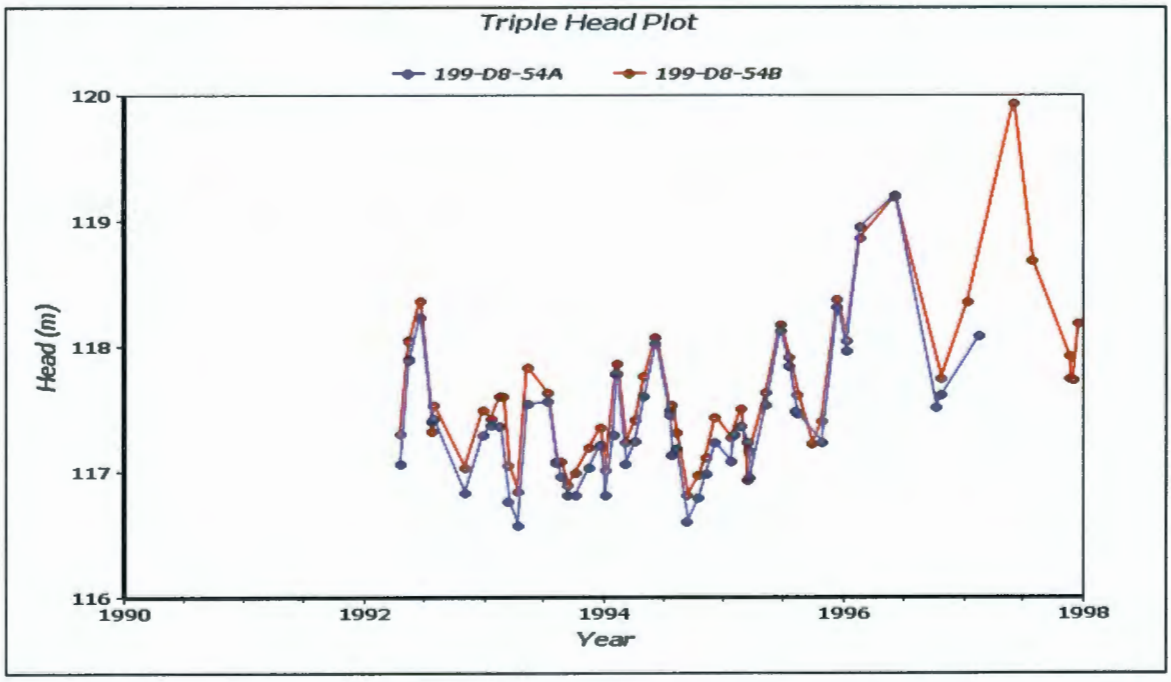


Figure 3-24. Comparison of Piezometric Head Observed in Selected Wells Completed Within the Shallow Unconfined Aquifer and the Uppermost Water-Bearing Unit of the RUM at 100-D

At 100-H, there are six wells that are screened within the first water bearing unit of the RUM. Three of the older RUM wells are paired with wells screened within the unconfined aquifer, as shown in Table 3-14. Since Wells 199-H3-2C and 199-H4-12C are currently being used as extraction wells, current data from those well sets are not suitable for evaluation of the vertical hydraulic gradient and older information must be used.

Table 3-14. Paired Wells at 100-H Area Completed Within the Shallow Unconfined Aquifer and the First Water-Bearing Unit of the RUM

Stratigraphic Unit Monitored	Well Pairs		
Screened in unconfined aquifer	199-H3-2A 199-H3-2B	199-H4-12A 199-H4-12B	199-H4-15A 199-H4-15B
Screened in first water-bearing unit of the RUM	199-H3-2C (extraction well)	199-H4-12C (extraction well)	199-H4-15CS

Figure 3-25 shows the water levels for the H-River Gauge along with Wells 199-H4-12A and 199-H4-12C, which are located adjacent to the Columbia River. Before connection of the pump-and-treat system, water levels in both wells track closely with the river stage changes. Well 199-H4-12A, however, has considerably lower hydraulic head than either the river or Well 199-H4-12C, which is completed in the RUM. This indicates an upward vertical gradient in this area, opposite of that found inland. It should also be noted that the RUM well hydraulic head falls in the mid-range of the river gauge measurements, with apparently instantaneous response to river stage change. This indicates a hydraulic connection between the aquifer in the RUM and the river (discussed further in Section 3.7.4, Section 3.7.6, and Section 4.5.1).

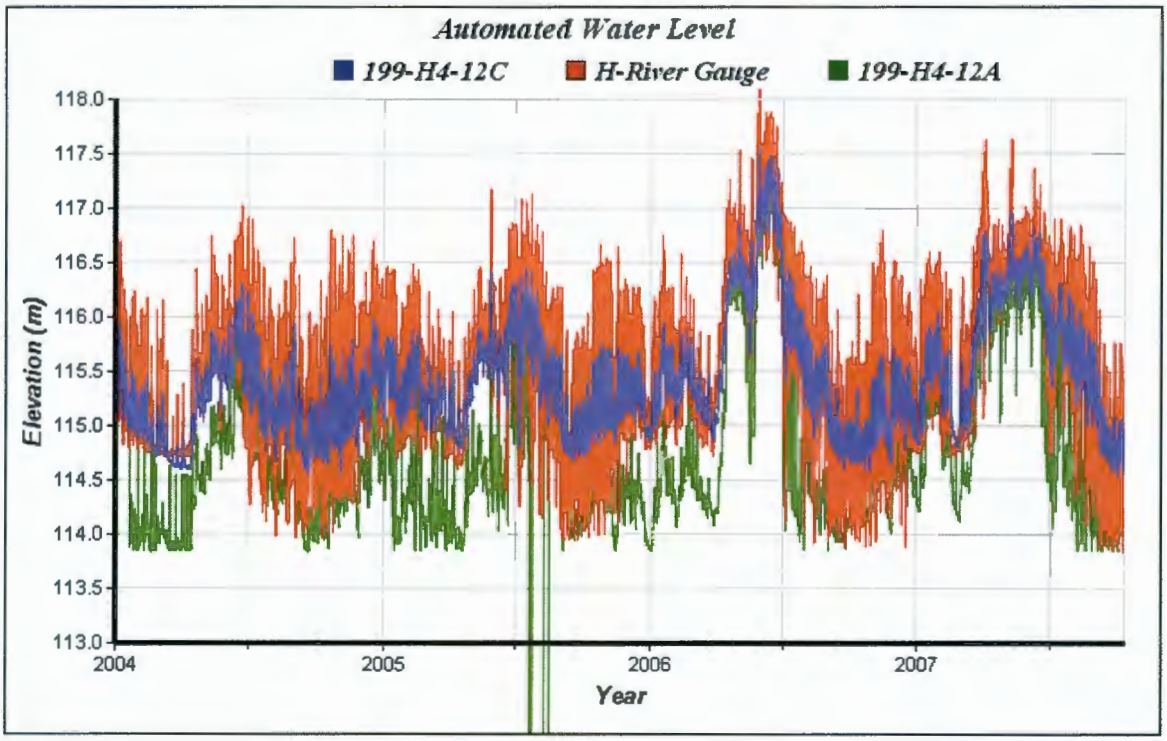


Figure 3-25 Comparison of Piezometric Head Observed in Selected Wells Completed Within the Shallow Unconfined Aquifer and the Uppermost Water-Bearing Unit of the RUM at 100-H

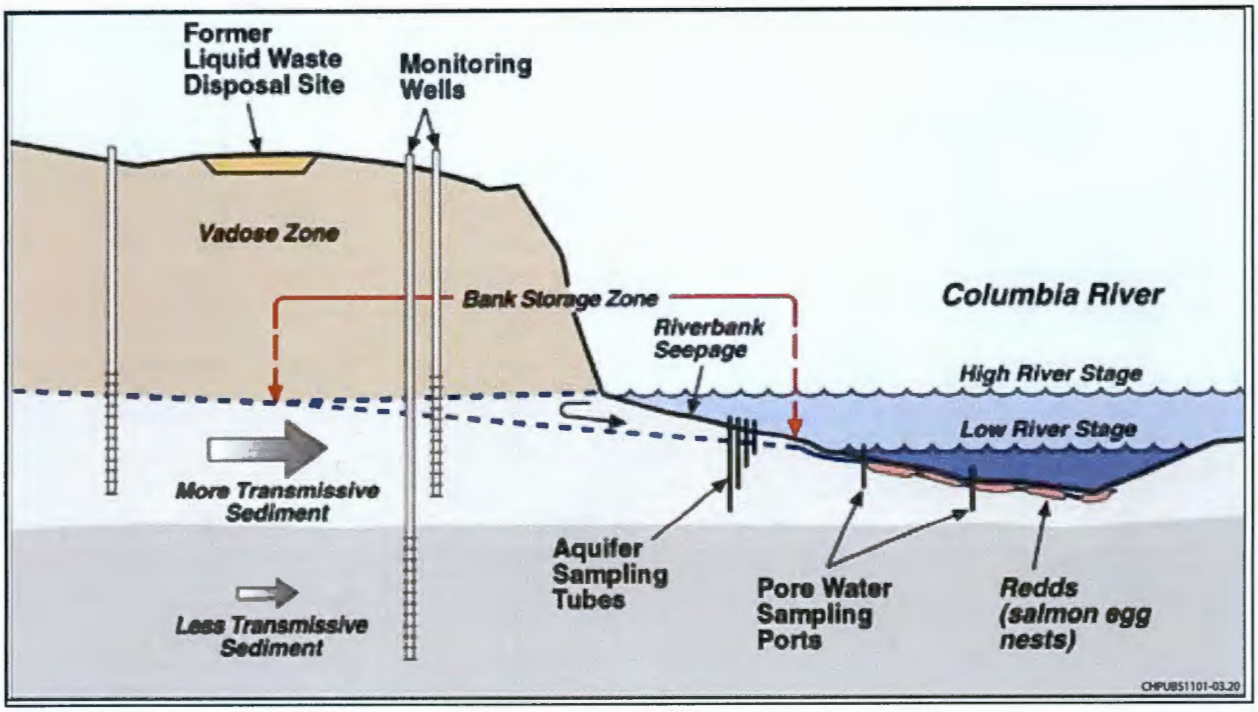
RI Wells 199-H3-9, 199-H3-10, and 199-H2-1 were not nested with adjacent wells, making evaluation of the vertical gradient difficult. Water level data for these wells is also limited to single point, which does not allow for trend analysis. In addition, under current conditions, groundwater levels in Wells 199-H3-9 and 199-H3-10 are likely influenced by the extraction pumping at Wells 199-H3-2C and 199-H4-12C, which are screened in the first water-bearing unit of the RUM. The hydraulic head differences of Wells 199-H4-15A (unconfined), 199-H4-15B (unconfined), 199-H4-15CS (first water-bearing unit in the RUM), 199-H4-15CR (Ringold unit B), 199-H4-15CQ (water bearing unit in the RLM), and 199-H4-15CP (basalt unit) are considerable. Hydraulic head of the first water bearing unit in the RUM is occasionally lower than the head in the unconfined aquifer, but has been as much as 0.5 m (1.6 ft) higher. The well completed in the Ringold unit B (199-H4-15CR) has similar head values to that of the RUM well (199-H4-15CS). Well 199-H4-15CQ, completed in the RLM, has a higher head and therefore upward gradient, when compared to the shallower aquifers, but there is considerable variation in the amount of head present. The hydraulic head of the basalt is consistently about 4 m (13 ft) higher than the other wells, indicating a strong upward gradient in the basalt aquifer. The presence of a demonstrable piezometric head difference between aquifer units is not evidence of the movement of groundwater upward or downward between the units. The movement of water in response to the observed head differences is dependent upon the existence of hydraulic conduit that would allow the movement of water. In most instances where substantial piezometric head differences are identified, the sustained pressure differential is indication of the absence of a direct hydraulic communication between the units.

3.7.3 Groundwater and Surface Water Interactions

Groundwater and surface water interactions are important for understanding the flux of contaminants entering the Columbia River. The zone of interaction is represented by the boundary between groundwater and river water below the river and near the shoreline. Groundwater discharge into the river occurs as seeps or springs release groundwater that flows across the riparian zone to the river, and via direct subsurface discharge of groundwater into the river channel substrate. Section 4.5 discusses recent pore water, surface water, and sediment sampling results, and Figure 3-26 illustrates the zone of interaction and riverbank seepage.

Groundwater flow, especially near the river, is strongly influenced by river stage, which varies seasonally and is directly controlled by the upstream Priest Rapids Dam. The rise and fall of river stage creates a dynamic zone of interaction between groundwater and river water; river stage influences flow patterns, transport rates, contaminant concentrations, and attenuation rates within the system (*Zone of Interaction Between Hanford Site Groundwater and Adjacent Columbia River: Progress Report for the Groundwater/River Interface Task Science and Technology Groundwater/Vadose Zone Integration Project* [PNNL-13674]).

Physical, chemical, and biological processes that potentially alter the characteristics of approaching groundwater occur within the zone of interaction. Data suggest that physical processes (for example, changes in gradient and physical mixing of river water with groundwater) are the primary influences on contaminant concentrations and fluxes where groundwater discharges into the river. Chemical processes (for example, precipitation reactions involving varying concentrations of calcium carbonate, pH, or reduction-oxidation conditions) may render contaminants less mobile as they adsorb to sediments or precipitate.



Note: Modified from PNNL-13674, *Zone of Interaction Between Hanford Site Groundwater and Adjacent Columbia River: Progress Report for the Groundwater/River Interface Task Science and Technology Groundwater/Vadose Zone Integration Project.*

Figure 3-26. Zone of Interaction and River Bank Seepage

Riverbank seepage to the river, as shown on Figure 3-26, is visible as the river stage declines following seasonal periods of high water. Conversely, during high river stage, these seep areas are submerged as river water enters the riverbanks and forms a layered system or a mixture during interaction with approaching groundwater. Data concerning the seeps and the riverbank indicate that the riverbank storage water composition varies dramatically from almost entirely river water during high river stage to primarily groundwater during low river stage (*Zone of Interaction Between Hanford Site Groundwater and Adjacent Columbia River: Progress Report for the Groundwater/River Interface Task Science and Technology Groundwater/Vadose Zone Integration Project* [PNNL-13674]). A cross-sectional depiction of groundwater flow towards the river is presented in Figure 3-27.

Along the 100 Area, riverbank seepage composed of contaminated groundwater creates potential pathways for contaminants to enter the Columbia River (*Investigation of Ground-Water Seepage from the Hanford Shoreline of the Columbia River* [PNL-5289]). Potential mixing of river water with groundwater may produce lower contaminant concentrations in the seep discharges than can be found in upgradient groundwater. These lower contaminant concentrations may be attributed to the bank storage phenomenon, where infiltrated river water stored in the riverbank during high river stage returns to the river via seeps during lower river stage (*Hanford Site Environmental Report for Calendar Year 2007* [PNNL-17603]).

3.7.4 Aquifer Intercommunication

Aquifer intercommunication occurs when groundwater moves vertically between aquifers, such as the unconfined aquifer and first water-bearing unit of the RUM, or between the first water-bearing unit of the RUM and lower water-bearing units.

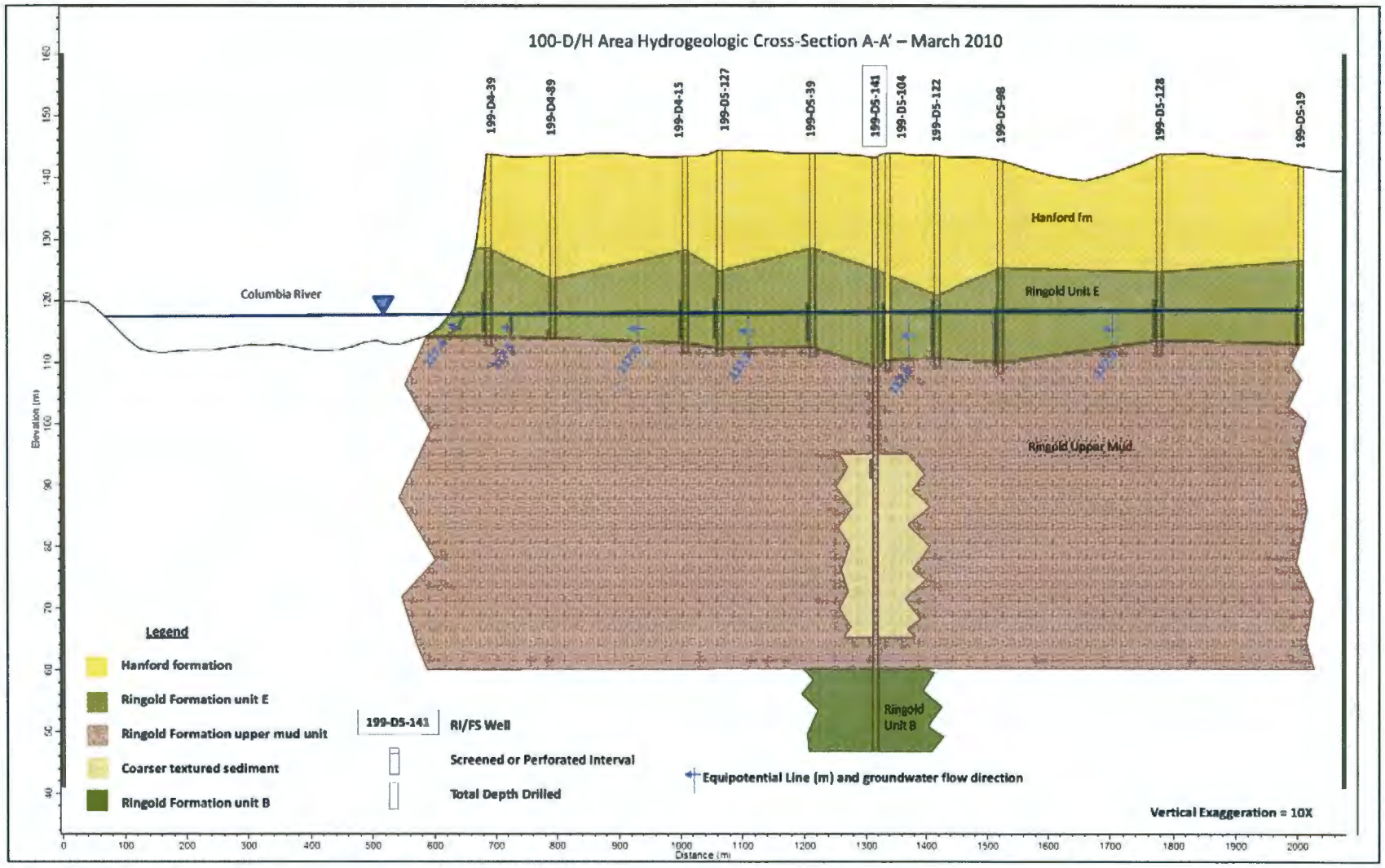


Figure 3-27. Groundwater Equipotential Lines and Streamflow Lines at 100-D

For groundwater movement to occur between aquifers, a difference in each aquifer's potentiometric head must exist, and a permeable flow path must exist between the two through which water can flow. Intercommunication between aquifers can occur by one mechanism or a combination of several mechanisms:

- Natural vertical head differences between aquifers push water through the intervening aquitard.
- Differences in head resulting from anthropogenic activities provide a driving force that pushes water through an intervening aquitard.
- Erosional unconformities provide a pathway for groundwater to move between aquifers (for instance, where the RUM unit surface may have been eroded by Pleistocene floods, possibly exposing shallow water-bearing sands to the unconfined aquifer).
- Erosional unconformities are considered the most likely significant mechanism for direct physical interconnection between the unconfined aquifer and water-bearing units within the upper RUM.
- The potential for pathways along poorly constructed wells or boreholes also exists; however, older wells suspected of having poor construction have been decommissioned.

Although vertical head differences offer one line of evidence of intercommunication between aquifer (and aquitard) units, another line of evidence is groundwater quality data. The presence of groundwater contaminants in underlying units provides evidence of communication with overlying units. The aquifer interconnections at 100-D have not been evaluated primarily because contamination has not been identified in the lower aquifers.

At 100-H, wells completed in the first water bearing unit of the RUM currently exhibit concentrations of Cr(VI) at levels above 100 µg/L (see Section 4.5.1). These data indicate that in the past, most likely under operational conditions when vertical head gradients were elevated, aquifer intercommunication occurred. This condition is indicated by the close similarity in measured piezometric heads in Wells 199-H3-2A, 199-H3-2B, and 199-H3-2C in years before initiating pump-and-treat activities at 100-H (Figure 3-28). The measured heads in these three wells from 1986 to 1997 were nearly identical and exhibited seasonal transients of the same timing and magnitude; this condition indicates that these aquifer units are not hydraulically isolated.

As part of *Aquifer Testing and Rebound Study in Support of the 100-H Deep Chromium Investigation* (SGW-47776), aquifer intercommunication was tested at groundwater monitoring Wells 199-H3-2C, 199-H4-12C, and Piezometer 199-H4-15CS. Both step tests and constant rate pumping tests were conducted, and nearby wells were monitored for response. During the constant rate pump test of Well 199-H3-2C, the unconfined aquifer was found to exhibit characteristics of a leaky aquifer, with groundwater levels in adjacent wells showing drawdown in response to pumping of the first water-bearing unit in the RUM (Figure 3-29). This is further supported by the almost immediate rising of groundwater levels in these wells after the pump was turned off. The lithologic description in the borehole log suggests that the shallow portion of the RUM unit at Well 199-H3-2C contains a greater percentage of sand, which is the likely cause of the hydraulic connection. This indicates intercommunication between the unconfined aquifer and the first water-bearing unit of the RUM at the nested well set 199-H3-2A, 199-H3-2B, and 199-H3-2C.

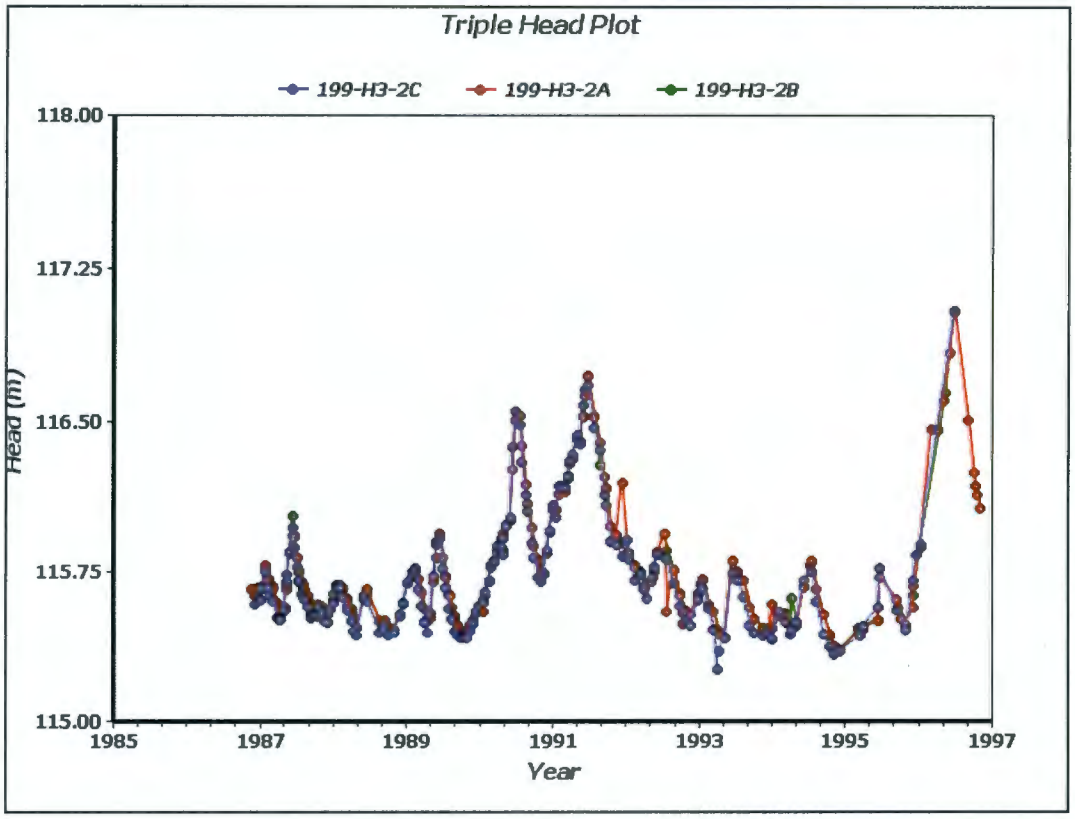


Figure 3-28. Piezometric Head Trends in Three Paired Wells at 100-H

The pump test also shows a response to river stage in RUM Well 199-H4-12C and Piezometer 199-H4-15CS, indicating communication with the Columbia River. The communication to the river is also discussed in Section 3.7.2, Section 3.7.6, and Section 4.5.1. The associated nested wells, however, did not indicate intercommunication between the unconfined aquifer and the RUM at those locations during testing. To confirm that the response was not related to leakage, the nested well completion of Well 199-H4-15 was evaluated in *Aquifer Testing and Rebound Study in Support of the 100-H Deep Chromium Investigation* (SGW-47776). These wells were determined to have had seal integrity.

An evaluation of the differences in hydraulic head (Section 3.7.2) indicates that the lower aquifer does not consistently have an upward vertical gradient, but can exhibit different gradients at different locations. This is due to the heterogeneity of the aquifer material and the overlying mud unit. As a result, wells screened in the first water bearing unit of the RUM have nearly the same water table elevation as the wells in the unconfined aquifer in completed wells, with only slight differences. A head difference is present in Well 199-H4-12C as compared to 199-H4-12A; but not evident at Well 199-H3-2C as compared to 199-H3-2A, as discussed previously.

Upward gradients will respond to increased downward pressure such as the increased hydraulic pressure created during reactor operations which resulted in a groundwater mound in both 100-D and 100-H, and may be reversed under such operational conditions. Evidence for these overwhelming conditions includes the river response in RUM wells and the thermal response noted during reactor operation. During reactor operations, thermal springs at the edge of the Columbia River at 100-H were measured with temperatures up to 74°C (165°F) (*Status of the Ground Water Beneath Hanford Reactor Areas January, 1962 to January, 1963* [HW-77170]).

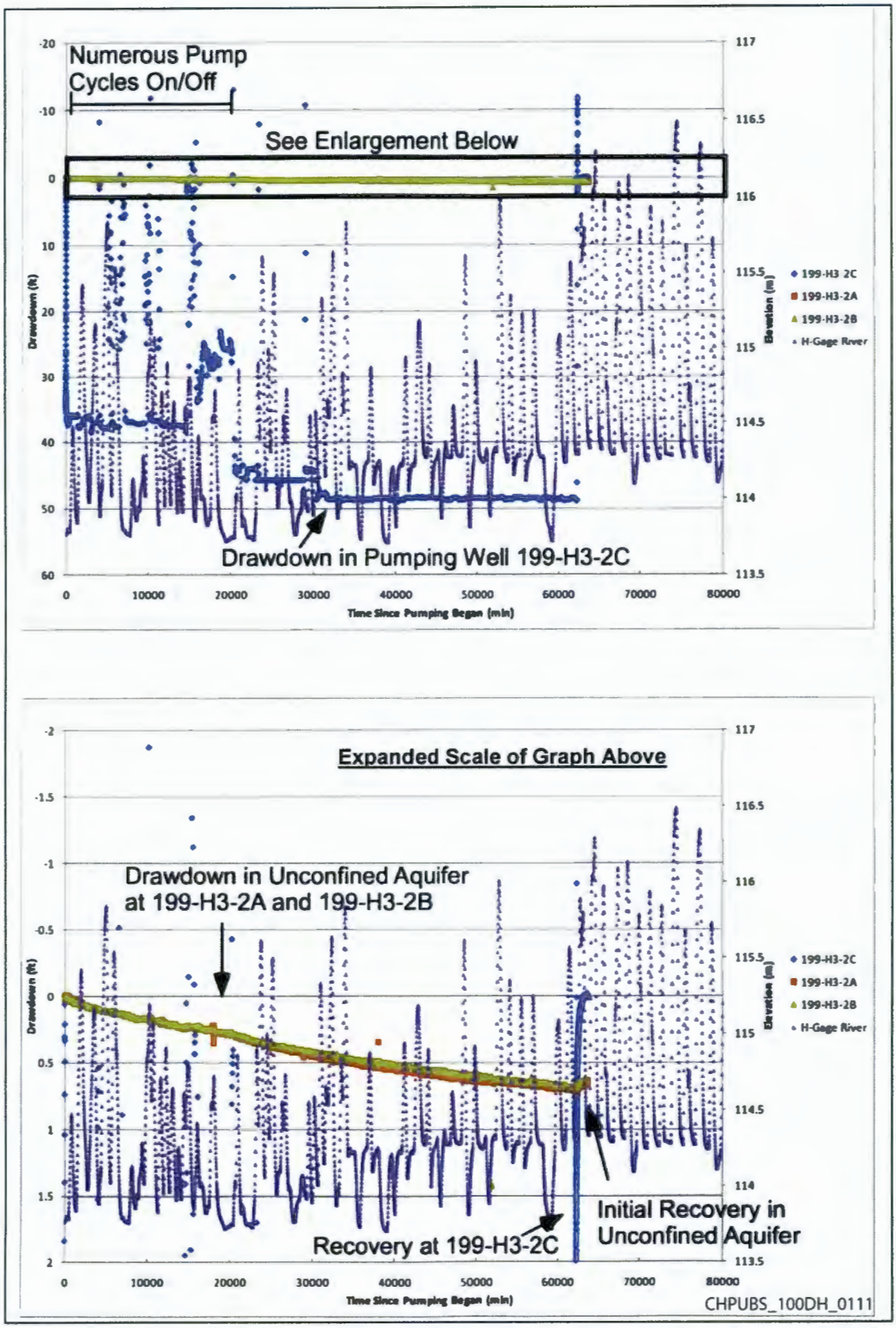


Figure 3-29. Hydrographs for Constant Rate Pump Test at Well 199-H3-2C

Further evidence of a hydraulic connection between the unconfined aquifer and first water-bearing unit in the RUM at a localized area of 100-H includes the results of a comparison of the geochemistry of groundwater from the unconfined aquifer, first water-bearing unit in the RUM, and river water. First, analysis of groundwater geochemistry from the first water-bearing unit in the RUM (Section 3.7.6) indicates that groundwater from RUM wells near the river have a chemical signature similar to that of river water. Second, the chemical signature of groundwater from RUM Well 199-H3-2C is similar to groundwater from nested Wells 199-H3-2A and 199-H3-2B, which are completed in the unconfined aquifer. Third, in addition to having similar chemical signatures between groundwater and river water, the ability of Well 199-H4-12C to sustain about 38 L/min (10 gal/min) pumping provides further supporting evidence that the RUM aquifer is drawing water from another source, such as the river or other aquifers.

3.7.5 Additional Effects to the Groundwater Flow Regime

The current interim remedies for 100-HR-3 OU groundwater contaminated with Cr(VI) consist of in situ chemical treatment and pump-and-treat. These interim remedies are intended to prevent Cr(VI) from reaching the Columbia River at concentrations exceeding the ambient water quality criterion. These remedial systems (discussed in Chapter 1) have a significant effect on the groundwater flow regime in the 100-HR-3 OU.

The updated remediation systems will drastically alter hydraulic gradients and flow in the unconfined aquifer beneath 100-D/H. The 100-DX/HX system is designed to effectively capture and treat Cr(VI)-contaminated groundwater before it enters the Columbia River at concentrations exceeding aquatic water quality criteria. This system will continue to evolve as extraction and injection wells are turned on and off to account for seasonal river stage variations and plume configuration changes. The RD/RAWP will address current and future issues with the 100-DX/HX system.

Another aspect of soil remedial activities potentially affecting groundwater is dust control. During remedial action, it is important to control fugitive dust (and the contamination it may contain). Water is applied to control airborne dust on haul roads, at excavation sites, and at soil stockpiles. If the water volume applied exceeds the holding capacity of vadose zone soil, it could move deeper into the vadose zone and eventually serve as a source of groundwater recharge. As a result, water is applied only to the extent needed to control dust to meet worker protection needs, and mitigate airborne contamination concerns for that day's planned excavation activities.

3.7.6 Groundwater Geochemistry

Groundwater data were evaluated for the distribution of the major ions in various wells within 100-D/H. The major ions evaluated include the common positively charged cations [calcium (Ca^{+2}), sodium (Na^{+}), potassium (K^{+}), and magnesium (Mg^{+2})] and the common negatively charged anions [chloride (Cl^{-}), sulfate (SO_4^{-2}), carbonate (CO_3^{-2}) and bicarbonate (HCO_3^{-})]. The relative equivalent concentrations of these ions were compared in wells with different geology, various levels of contamination, and the water of the Columbia River. To compare the concentrations of the ions, laboratory analytical results are collected. The concentrations are then converted from µg/L or mg/L into the milliequivalents per liter of the ion, based on its atomic weight.

The equivalent ionic concentrations vary greatly across 100-D/H (Appendix M), but when the distribution is plotted as a graphic diagram, patterns develop. Radial plot diagrams showing relative ion concentrations for various wells, based on the data in Table M-6 (Appendix M), and for the Columbia River, are presented on Figures 3-30 and 3-31. Groundwater monitoring wells were evaluated from various geologic units and locations at 100-D (Figure 3-30), across the horn (Figure 3-31), and from 100-H (Figure 3-31).

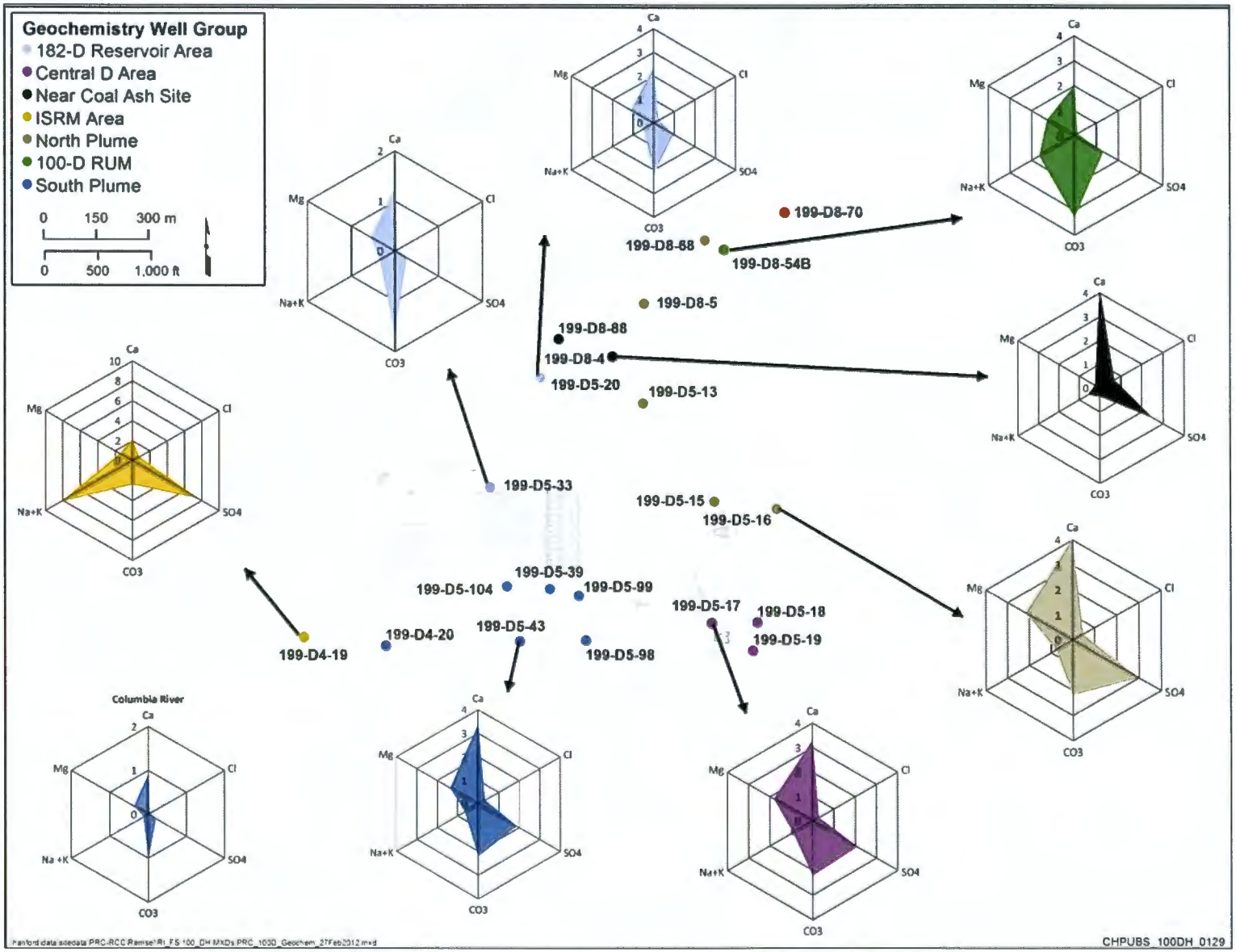


Figure 3-30. Major Ion Chemistry of 100-D Groundwater

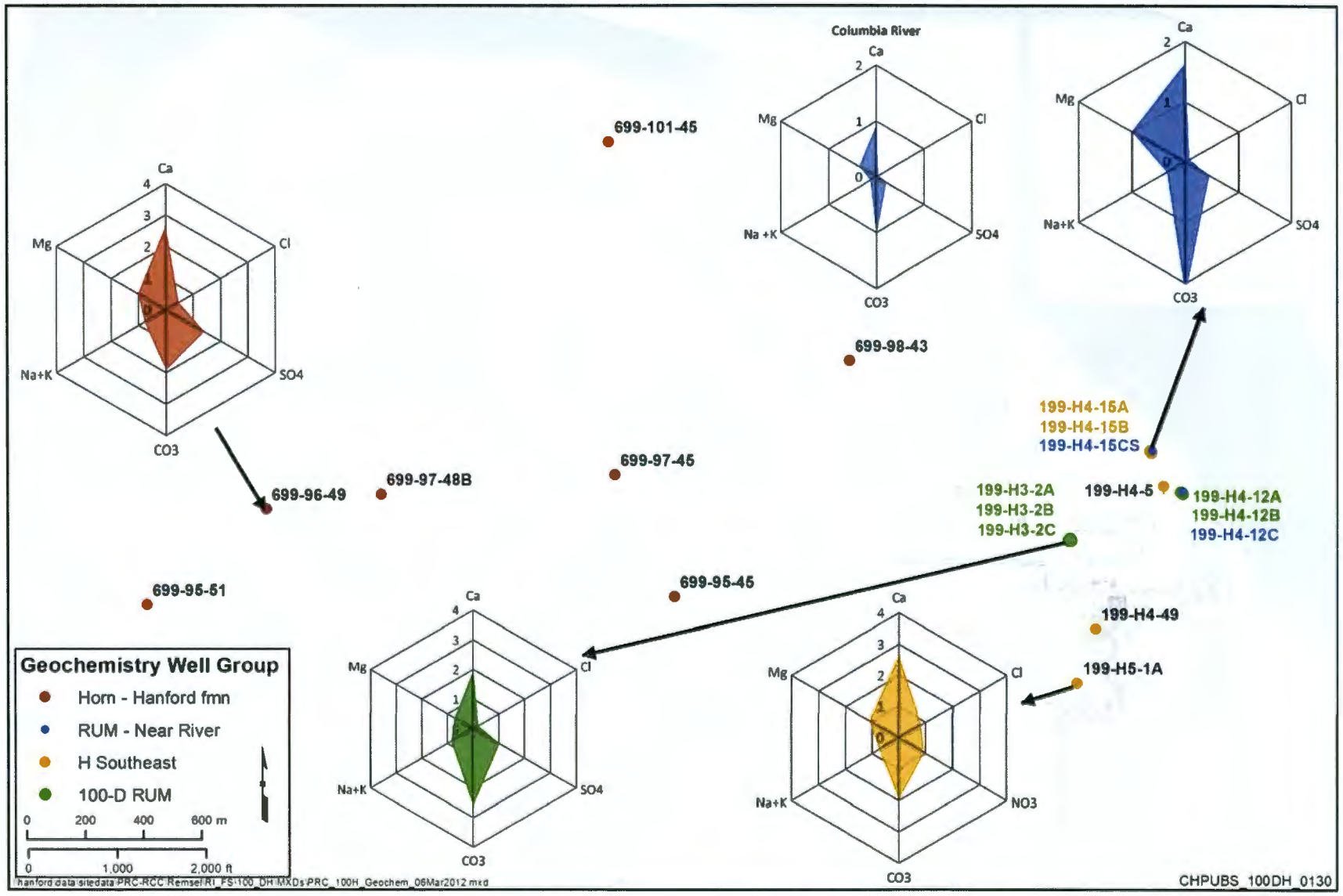


Figure 3-31. Major Ion Chemistry of 100-H and horn Groundwater

Not all wells had adequate data for conducting a geochemical analysis; however, based on the available data, several groups with a similar chemistry, ion pattern, or distribution emerged during the evaluation. At 100-D/H, the primary patterns correlate with the geologic unit in which the well is completed. There are differences between wells completed in the Hanford formation, those completed in the Ringold Formation unit E, and those completed in the RUM or lower Ringold Formation units that are apparent in the chemical pattern moving west to east across the horn. The other controlling factors are the contaminant levels and various sources.

Groundwater monitoring wells with similar geochemical signatures are shown in Figures 3-30 and 3-31 with similar colored dots. A detailed description of the geochemistry similarities and the data used to develop the radial plot diagrams are included in Appendix M.

Wells in 100-D are generally completed and screened within the Ringold Formation unit E, and the geochemical signature is similar in most wells within that formation. Slight variations in the pattern are evident in areas with higher levels of Cr(VI), with slightly higher levels of magnesium, sodium, and potassium changing the pattern in the Stiff diagram. At Well 199-D5-33, the geochemical pattern is most similar to that found in river water. This is likely due to some continued leakage of the 182-D Reservoir, which holds raw river water.

Other areas with variation include the ISRM barrier, as see in Well 199-D4-19. The reducing agent used in the barrier consists of sodium dithionite ($Na_2S_2O_4$). As expected, the sodium and sulfate levels in the groundwater at the ISRM well are much higher than levels in other wells within 100-D/H. This pattern indicates the continued presence of reaction products from the placement of the ISRM barrier.

Also in 100-D is waste site 126-D-1 Coal Ash Pit. The relatively high calcium and sulfate concentrations exhibited in Well 199-D8-4 are consistent with effects from deposition of fly and bottom ash, as well as flue-gas desulfurization residues. Flue gas desulfurization residue consists primarily of gypsum (calcium sulfate) and is frequently combined with fly ash. The chemical distribution found in flue gas desulfurization is consistent with the geochemical pattern identified in the wells downgradient from waste site 126-D-1, including Wells 199-D8-88 and 199-D8-4. No metals were identified in association with these wells, with the exception of a reported value of 60 µg/L of zinc in Well 199-D8-4, which is lower than the ambient water quality criterion of 91 µg/L but higher than typical concentrations in that well.

North of 100-D toward the horn are Wells 199-D8-70 and 699-95-51. The chemical signature in these wells is less distinct. The wells in the northern 100-D/horn transitional area have lower levels of carbonate than in other areas. Well 699-101-45, located to the far north of the horn, is also considered transitional and has the same pattern. The higher levels of sodium plus potassium in these wells are consistent throughout the horn and are coupled with lower magnesium levels than those found in 100-D. The wells in the horn also tend to have lower chloride levels than wells in 100-H, which are associated with the Hanford formation.

Wells completed in the Hanford formation at 100-H have a similar chemical pattern. The concentrations of magnesium, chloride, and sodium plus potassium give the Stiff diagram a distinct shape, as represented by Well 199-H4-49 on Figure 3-31.

Well 199-D8-54B shows a significantly different geochemical pattern than other wells at 100-D. This well is completed in the first water bearing unit of the RUM, and therefore the groundwater chemistry reflects the chemistry within that aquifer. As shown in the other RUM wells within 100-H and the horn, most of the wells completed in this aquifer have a similar pattern, with much higher carbonate, sodium, and potassium levels. However, Well 199-D8-54B is unexpectedly similar in chemical pattern to a group of nested wells (199-H3-2A, 199-H3-B, and 199-H3-C). It is also similar to nested Wells 199-H4-12A

and 199-H4-12B, which are completed in the Hanford formation. However, the RUM Well 199-H4-12C has a different signature. It is undetermined why wells with such different geology would present such similar patterns in groundwater chemistry.

Three sets of nested wells are in 100-H. The nested wells consist of at least three wells or piezometers completed in multiple aquifers and are within a few feet of each other. The nested groups are 199-H3-2, 199-H4-12, and 199-H4-15. Wells with the suffix of "A" and "B" are completed in the unconfined aquifer at 100-H. Wells 199-H3-2C, 199-H4-12C, and 199-H4-15CS are completed in the RUM. The remaining piezometers associated with 199-H4-15 are completed in the Ringold Formation unit B (-15CQ), the RLM (-15CR), and the basalt (-15CP) units, which all have distinct geochemistry.

Monitoring Wells 199-H3-2A, 199-H3-2B, 199- H3-2C, 199-H4-12A, and 199-H4-12B have a similar chemical pattern. The consistency of the chemical pattern is expected for the "A" and "B" wells, since they are completed in the same geologic unit. The deep well, 199-H3-2C, was completed in the RUM. Monitoring Well 199-H3-2C was shown to be connected to the overlying unconfined aquifer wells (199-H3-2A and 199-H3-2B) during the aquifer rebound test (*Aquifer Testing and Rebound Study in Support of the 100-H Deep Chromium Investigation* [SGW-47776]).

The other two RUM well/piezometers at 100-H (199-H4-12C and 199-H4-15CS) are near the Columbia River. Both of these wells have a geochemistry that is similar to river water, and dis-similar to the chemistry found in the associated nested wells. The water levels in these wells also respond to changes in river stage (*Aquifer Testing and Rebound Study in Support of the 100-H Deep Chromium Investigation* [SGW-47776]). The observation of similar geochemistry supports the theory that the RUM is hydrologically connected to the river in that location. It also supports the theory that the RUM wells near the river are not connected to the inland RUM wells.

3.7.7 Time Series Evaluation

The geochemistry of four wells was also evaluated over a time series: 199-D4-15 (southern 100-D), 199-D5-14 (northern 100-D), 199-H4-48 (central 100-H), and 199-H4-6 (northern 100-H). The wells in 100-D showed no significant change over time, with geochemical patterns consistent with other wells in the geographic area. In 100-H, Well 199-H4-48 had little change from 1999 through 2010. The pattern for 1992 had very little chloride in the geochemical signature, which is quite different from the pattern in all of the later years. This change corresponds with a drop in Cr(VI) concentrations from 150 µg/L in 1996 to 20 µg/L in 1999 in Well 199-H4-48. The other well in 100-H that was evaluated over time is 199-H4-6.

This well showed little change over time, with small fluctuations of sulfate from 1988 through 1996. In 2010, the chloride levels in 199-H4-6 increased, thus changing the pattern, but there was no corresponding change in Cr(VI) levels, and the cause of this fluctuation has not been determined.

Review of geochemical data suggests the following:

- Groundwater chemistry at the ISRM barrier (Well 199-D4-19) has been altered, showing high sulfate levels.
- Groundwater downgradient of the coal ash pit is consistent with the chemistry of fly ash; however, no other contamination has been linked to it.
- It is likely that the 182-D Reservoir continues to leak, affecting the groundwater. This idea is supported by the chemistry of Well 199-D5-33 and other nearby wells.

- RUM Well 199-H3-2C is hydrologically connected to the overlying unconfined aquifer.
- RUM wells near the Columbia River at 100-H are hydrologically connected to the Columbia River.

3.8 Artificial Water Systems

This section describes the water systems at 100-D/H that affect the groundwater flow regime.

3.8.1 Export Water System

The Export Water System provides water service within the 100 and 200 Areas of the Hanford Site, as well as certain facilities in the 600 Area (Figure 3-32) for process water, fire protection, dust suppression, and other nonpotable uses. At 100-BC and 100-D, river pumping stations draw raw water from the Columbia River and feed it into two large-capacity reservoirs (182-B and 182-D). In turn, pump stations at the reservoirs move water into a network of pipelines traversing the 100 Area and connecting to moderately sized distribution reservoirs on the Central Plateau. The Export Water System was originally installed in the mid-1940s and has been in constant use for more than 60 years. By 2004, the majority of the water lines in 100-D were removed from service. The only remaining active lines are the 1.1 m (3.5 ft) diameter export water line and one 0.3 m (1 ft) diameter fire suppression water line.

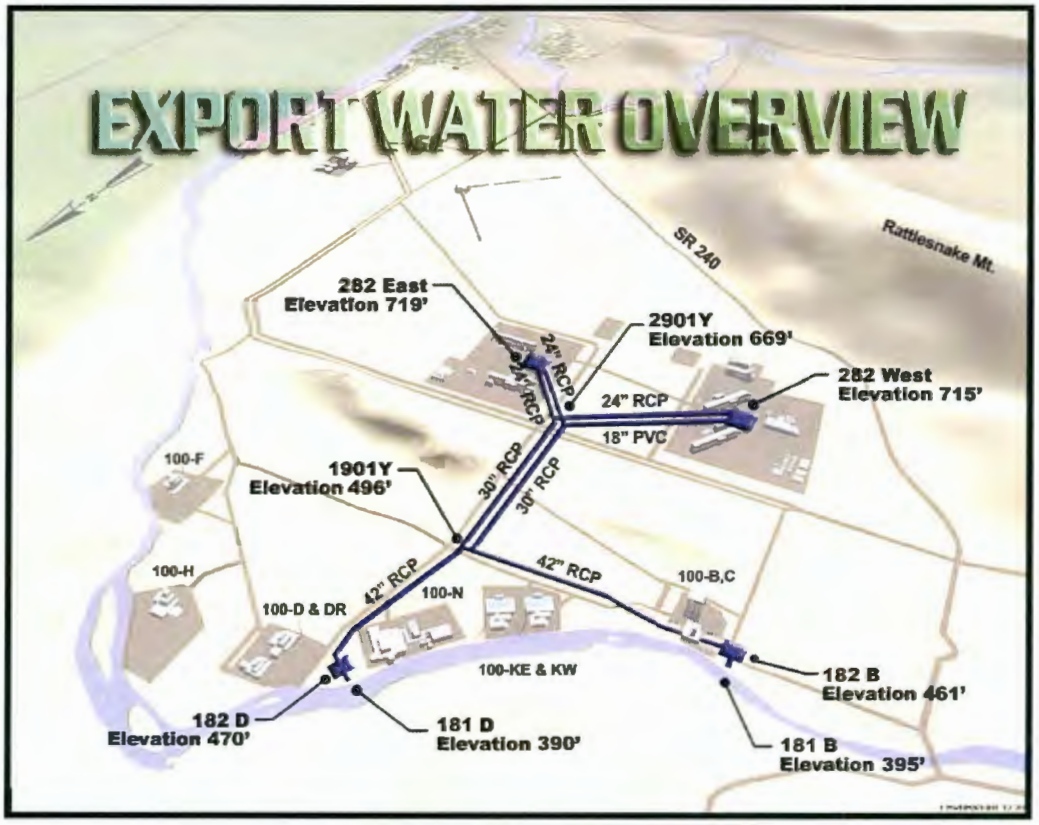


Figure 3-32. Export Water Distribution Piping

The 182-D Reservoir is one of two remaining structures at the Hanford Site that stores large quantities of untreated raw water. Chapter 2 discusses the current plans for the 182-D Reservoir and export water lines. Currently, the 182-D Reservoir serves two roles:

- Rarely needed emergency backup facility to the primary reservoir in 100-BC (*November 6 & 7 Facilitated Session and the 182-D Reservoir Repair and Modification Report and Long Term Export Water Supply System Alternatives Study* [FH, 2008])

- Source of water operated during 242-A Evaporator campaigns to reduce waste volumes in the single- and double-shell tanks in the 200 Area

Leaks from the 182-D Reservoir occurred chronically from the beginning of reactor operations, potentially influencing the local groundwater flow regime. It has been demonstrated that historically the reservoir may have leaked several hundred gallons per minute. Leakage from the reservoir and export water lines could affect groundwater flow, contaminant mobility, and the effectiveness of groundwater cleanup at 100-D. Low specific conductance readings in groundwater and historically elevated groundwater levels in monitoring wells near the reservoir confirm that significant leakage had occurred and was an ongoing, chronic condition.

Until 2004, the 182-D Reservoir chronically leaked enough to sustain a local water table mound that created a "clean zone" between the northern and southern plumes in 100-D (see Chapter 4). This mound appears to have diverted groundwater flow north and south of the 182-D Reservoir, with corresponding diversion and local dilution of contaminants in groundwater. However, these leaks and their effect on groundwater flow have diminished since the reduction of storage volume in the reservoir in 2004 (from 95 million L [25 million gal] to 19 million L [5 million gal]) (Water System Master Plan [HNF-5828]). This is to the point that influences on groundwater flow from reservoir leakage are indistinguishable from those created by nearby pump-and-treat activities (*Calendar Year 2010 Annual Summary Report for the 100-HR-3and 100-KR-4 Pump and Treat Operations, and 100-NR-2 Groundwater Remediation* [DOE/RL-2011-25]).

Monitoring of water levels in the reservoir and groundwater levels in nearby monitoring wells (199-D5-33, 199-D5-38, and 199-D5-34) is conducted automatically via pressure transducers to coordinate reduced operation consistent with the Water System Master Plan (HNF-5828).

In response to the reservoir leakage information, a specific issue (Issue 10) was included in *The Second CERCLA Five-Year Review Report for the Hanford Site* (DOE/RL-2006-20) for DOE to provide direction to its operating contractor to conduct changes to the operation of the reservoir to minimize leakage. Those actions were completed and documented in the closeout of the Five-Year Review issue. Specific actions implemented include the following:

- The 182-B Reservoir is designated as the primary export water source for the Hanford Site and is used for emergency backup (maintained in a standby configuration).

- The original design water level for the 182-D Reservoir was a depth of 4.6 m (15 ft) with 0.3 m (1 ft) of freeboard; however, while in standby operations, the water levels within 182-D Reservoir have been reduced via operating controls over the past 7 years using a graded approach. Water levels are now maintained at depths of 0.3 to 0.9 m (1 to 3 ft).

- During emergency operations, the water levels within 182-D Reservoir are allowed to rise from 0.6 to 1.8 m (2 and 6 ft).

- Flow from the 182-D Reservoir is required infrequently for emergency operation, most recently to support 242-A Evaporation operations during a pump maintenance shutdown at 182-B Reservoir.

- The floor of the reservoir was cleaned and caulked.

DOE recently evaluated various options for the Export Water System (*Export Water System Options Study* [North Wind, 2010]), including whether the 182-D Reservoir was necessary to support the continued Hanford Site cleanup mission. The Water System Master Plan (HNF-5828) identified the preferred infrastructure solution for the Export Water System and included monitoring requirements for the 182-D Reservoir and export water lines. The Water System Master Plan (HNF-5828) calls for maintaining the present Export Water System for the next 10 years while a new export water system is designed, permitted, and constructed in 100-K. Ultimately all Export Water System-related facilities in 100-D would be demolished. The 182-D Reservoir and pump station will be removed and the area brought to grade with clean fill. In the meantime, monitoring of the 182-D Reservoir will continue, as specified in the Water System Master Plan (HNF-5828).

3.8.2 Reactor Cooling Water Systems

Other facilities that released large quantities of fluid over long periods included the 107-D and 107-DR Retention Basins, the 116-DR-1 and 116-DR-2 Trenches, and the 120-D-1 Ponds. These long-term discharges created groundwater mounds under the discharge facilities that overwhelmed the natural hydraulic gradient for some distance away from the discharge site and had substantial effects on contaminant migration patterns in the unconfined aquifer. A substantial groundwater mound beneath the entire 100-D Area, centered beneath the retention basins, had a maximum observed height of about 5 m (15 ft) above the natural static water table (*100-HR-3 Remedial Process Optimization Modeling Data Package* [SGW-40781]).

These long-term releases created a radial flow regime that was established early in the operations period and sustained until operations ceased. Groundwater mounding dissipated quickly (that is, within weeks or a few months) after cessation of releases from these sources.

In 100-H, the opportunities for groundwater mounding that formed from liquid discharges were much less complex. The 116-H-7 (107-H) Retention Basin, held held-up spent reactor cooling water before discharge to the river; it leaked chronically throughout the operations period. In addition, contaminated cooling water was discharged from the retention basin to the 116-H-1 Trench and was allowed to infiltrate into the vadose zone. As previously described, a slightly smaller groundwater mound persisted beneath the 100-H Area until cessation of operations in 1965, after which the mound quickly dissipated.

3.8.2.1 Demography and Land Use

Demographics. A detailed discussion of the population surrounding the Hanford Site, including adjacent counties and cities, is presented in the NEPA Characterization Report (PNNL-6415). The 2009 population estimate from the U.S. Census Bureau was that 47,530 people lived in the city of Richland, the closest population center to the Hanford Site. An estimated 58,650 people lived in Pasco and 67,810 people lived in Kennewick. Population groups near the Hanford Site include Native Americans and various ethnic minorities. Native American descendants living near the Hanford Site include members of the following federally recognized groups: the Confederated Tribes and Bands of the Yakama Nations, the Confederated Tribes of the Umatilla Indian Reservation (CTUIR), the Nez Perce Tribe, and the Confederated Tribes of the Colville Reservation. Members of other unrecognized Tribes members, such as the Wanapum, also live in the area. There is no continuous human inhabitation immediately adjacent to 100-D/H.

The economy in the region near the Hanford Site is driven by three major sectors: DOE and its contractors operating the Hanford Site; Energy Northwest, which operates the nuclear-powered Columbia Generating Station on land leased from DOE; and the agricultural community, including a substantial food-processing component. Additional employment sectors driving the local economy include "other major employers," such as non-DOE contractor employers in the region, tourism, and healthcare.

Land Use. The Columbia River is a critical resource for the people and ecology of the Pacific Northwest. The 80.5 km (50 mi) stretch of the Columbia River flowing through the Hanford Site is referred to as the Hanford Reach. It is a non-tidal, free-flowing stretch of the Columbia in the United States. The river, islands, gravel bars, sloughs, riparian areas, and dune field of the Hanford Reach provide a variety of habitats that are now rare along the Columbia River. As one of the largest rivers in North America, its waters support a multitude of uses that are vital to the economic and environmental well-being of the region. The river is particularly important in sustaining the culture of Native Americans. The Columbia River downstream of the Hanford Site is the primary source of municipal drinking water for cities of Richland, Pasco, and Kennewick; river water is withdrawn for irrigation at numerous locations below the Hanford Reach.

Land use in the River Corridor is currently controlled by the DOE and the U.S. Fish and Wildlife Service (USFWS), which jointly manage this federally owned land to protect natural and cultural resources while conducting cleanup activities. Such management is consistent with *Final Hanford Comprehensive Land Use Plan Environmental Impact Statement* (DOE/EIS-0222-F), hereinafter called Hanford CLUP, and *Supplement Analysis: Hanford Comprehensive Land-Use Plan Environmental Impact Statement* (DOE/EIS-0222-SA-01) for the site, and reflects the requirements of *Hanford Reach National Monument: Final Comprehensive Conservation Plan and Environmental Impact Statement Adams, Benton, Grant and Franklin Counties, Washington* (USFWS, 2008) for the Hanford Reach National Monument. It is both the DOE and the USFWS expectation that this joint management of the Hanford Site will continue for many years into the future and that the property will remain under federal ownership.

Interim RODs for CERCLA cleanup activities in the River Corridor recognized that the reasonably anticipated future land use in the River Corridor had not been well defined. Since that time, DOE has issued the Hanford CLUP (DOE/EIS-0222-F), the Hanford Reach National Monument has been established, and those documents define conservation and preservation as the future use of the lands along the river. In a memorandum (*Hanford Reach National Monument* [Clinton, 2000]), the President directed the Secretary of Energy to consult with the Secretary of the Interior on how best to protect the lands around the Hanford Reach National Monument permanently. Much of the area contains shrub-steppe habitat and other areas of scientific and historic interests. The President specifically included the possibility of adding lands to the Hanford Reach National Monument as they are remediated. EPA and the state of Washington believe that the cleanup actions in the River Corridor should also support the potential for future residential use.

When soil cleanup goals were initially established for the River Corridor, the TPA (Ecology et al., 1989a) signatories agreed that it was appropriate to protect for a range of potential exposures in the future so that cleanup actions did not limit future use of the site. Such a goal addressed the interests of a number of Hanford Site stakeholders, including the Future Site Uses Working Group. Interim action cleanup requirements were based upon consideration of state MTCA (WAC 173-340) cleanup requirements for unrestricted surface use for chemical contaminants and a dose based standard of 15 mrem per year for radiological constituents based on DOE guidance for a residential exposure. For the purpose of establishing final cleanup requirements for the River Corridor cleanup, the Tri-Party agencies believe it is appropriate to continue to use the interim action ROD cleanup requirements, updated to reflect revised MTCA (WAC 173-340) values and excess cancer risk for radiological constituents. Final cleanup values will also be established to protect groundwater and surface water resources and address ecological risk considerations.

Because the interim action cleanup values in River Corridor RODs were developed to accommodate a variety of future land use options the resultant cleanup actions will be protective of the reasonably foreseeable land uses that DOE and the USFWS anticipate for the River Corridor.

Tribal Interests. Tribal fishing rights are recognized on rivers within the lands ceded by treaty, including the Columbia River, which flows through the Hanford Site. In addition to fishing rights, the Tribes retain the privilege to hunt, gather roots and berries, and pasture horses and cattle on "open and unclaimed lands." It is the position of DOE that the Hanford Site, which was assembled from lands acquired from private owners and lands withdrawn from the public domain into a federal enclave with no public entry, is not open and unclaimed land. While reserving all rights to assert their respective positions, the Tribes are participants in DOE's land use planning process, and DOE considers Tribal Nation concerns in that process.

3.9 Ecology

The Hanford Site is located in a mid-latitude area with a semiarid climate. This portion of the Columbia Basin provides a unique habitat, having the last free-flowing section of the Columbia River passing through it, supporting a rich diversity of plant and animal species (2009 Sitewide Environmental Report [PNNL-19455]). Species diversity is maintained through the long-standing management practices of DOE, which leaves most of the land relatively undisturbed. Only about 6 percent of Hanford Site land has been disturbed or is actively used by DOE for waste disposal and storage. The native terrestrial and aquatic ecological resources found on the Hanford Site are becoming increasingly rare throughout the Columbia Basin region.

Three key ecological study zones have been identified for purposes of investigation in the River Corridor: the upland, riparian, and near-shore river zones (RCBRA Literature Review [PNNL-SA-41467]; RCBRA [DOE/RL-2007-21, Volume I]).

- **Upland zone**—Consists of land adjacent to the main channel of the Columbia River above the high-water mark of the Columbia River. Terrestrial and generally dry, the upland zone is not influenced by river flow and depends on precipitation for its water supply.

  Within the operational areas, most of the upland zone is highly disturbed, consisting of barren or gravel areas or non-native annual species. The upland environment outside the operational areas is relatively undisturbed and consists of relatively native shrub-steppe vegetation habitat.

- **Riparian zone**—extends from the point on the riverbank where upland vegetation is no longer dominant to the shoreline of the Columbia River. Typically narrow, the riparian zone varies in width, depending on the slope of the riverbank. The transition from the upland zone vegetation to riparian vegetation is generally abrupt. The vegetation that grows in the riparian zone along the river shoreline is thicker and taller than that in the upland area, attracting a broader range of wildlife species. The small mammals, birds, and reptiles common to the upland environment are also likely to inhabit the riparian environment (RCBRA Report [DOE/RL-2007-21, Volume I]).

- **Near-shore aquatic zone**—Consists of a narrow band of the Columbia River adjacent to the shoreline. The near-shore aquatic zone evaluated in this report extends from the low water mark on the shoreline into the river channel to a water depth of roughly 1.8 m (6 ft). The CRC (DOE/RL-2010-117, Volume I) evaluates environmental conditions for depths greater than 1.8 m (6 ft). The aquatic vegetation found in the near-shore zone supports aquatic insect populations, benthic taxa (species and organisms that live in or on the bottom of the river), birds, and fish. At least 45 species of fish live in the Columbia River adjacent to the Hanford Site, and some use the river as a migration route to and from upstream spawning areas. The shoreline areas provide rearing habitat for many fish species, including spawning habitat for threatened and endangered fish species (RCBRA Report [DOE/RL-2007-21, Volume I]).

Large-scale distribution of vegetation types within the upland zone and surrounding the riparian and near-shore zones before the 2000 wildfire is presented in Figure 3-33. Table H-1 (Appendix H) presents a description and extensive and list of species known or potentially occurring on the Hanford Site classified by habitat type.

3.9.1 Threatened and Endangered Species

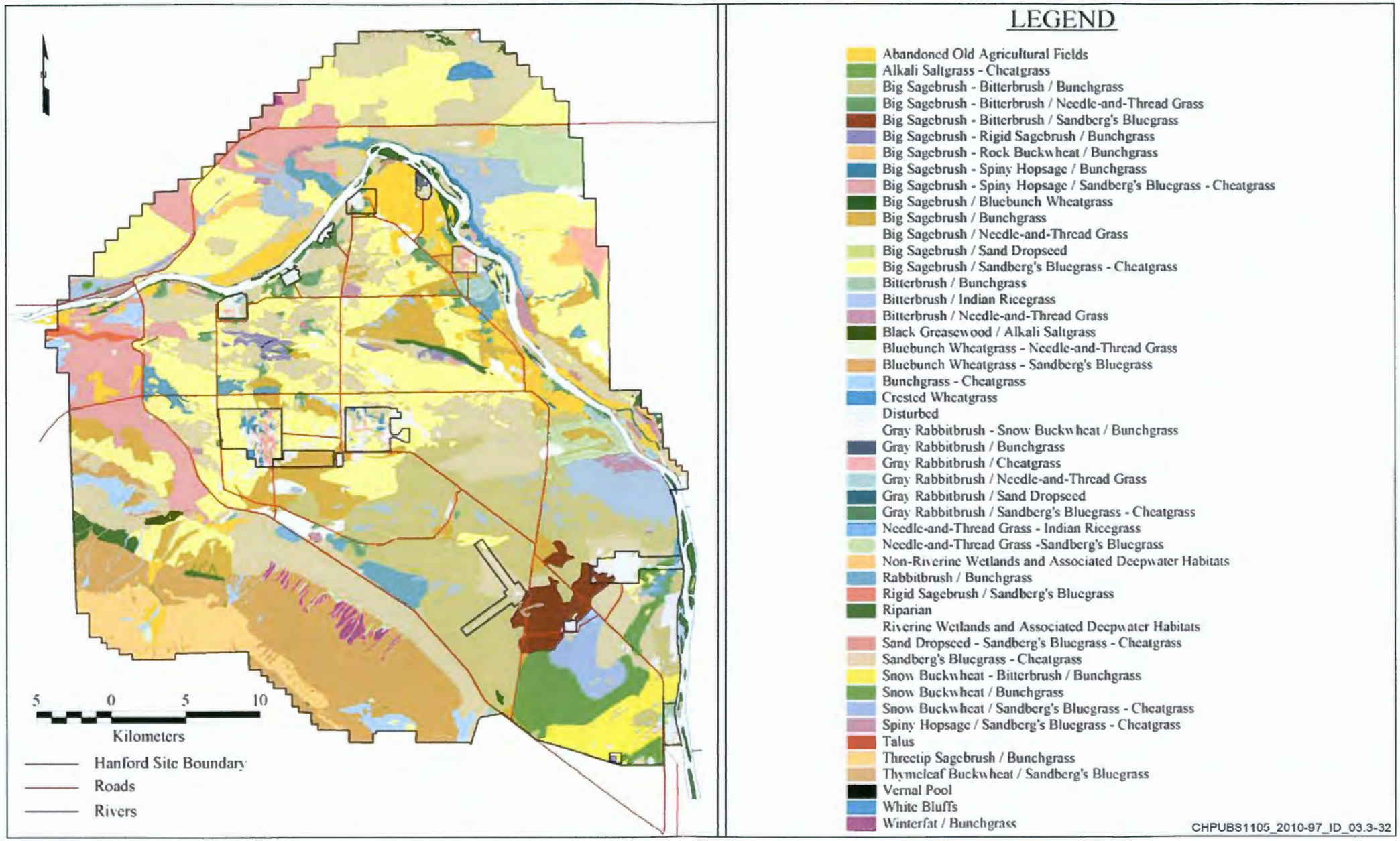
Several species are recognized by state or federal agencies as having special status based on the species' risk of extinction. Threatened and endangered species are considered at risk, and as such, these species were not identified for sacrificial sampling or subsequent analyses for the risk assessment effort. Data for selected surrogate species were required for contaminant or biological characterization based on the guild in which the special-status species were identified (Table 5-1 of *Risk Assessment Work Plan for the 100 Area and 300 Area Component of the RCBRA* [DOE/RL-2004-37]; Chapter 7 of this report). The list of state and federally listed species of concern, including candidate, sensitive, and monitored species thought or known to occur on the Hanford Site is updated regularly in the NEPA Characterization Report (PNNL-6415). No plants, invertebrates, reptiles, amphibians, or mammals on the federal list of threatened and endangered wildlife and plants are known to occur on the Hanford Site (RCBRA Literature Review [PNNL-SA-41467]).

Two species of federally listed endangered fish, the Upper Columbia River spring-run Chinook salmon and the steelhead, occur in the Hanford Reach. The spring-run Chinook salmon do not spawn in the Hanford Reach, but they use it as a migration corridor. Steelhead spawning has been observed in the Hanford Reach. The bull trout is listed as threatened by the National Marine Fisheries Service but is not considered a resident species and is rarely observed in the Hanford Reach (*100-B/C Pilot Project Risk Assessment Report* [DOE/RL-2005-40]).

Table H-2 provide flora and fauna species listed by the State of Washington as threatened or endangered, including candidate, sensitive, and monitored species thought or known to occur on the Hanford Site.

3.9.2 River Corridor Food Web and Receptors

Consideration of ecological receptors in the risk assessment requires an understanding of relationships among biotic community members. One such relationship, trophic transfer of contaminants, is an important element in ecological risk assessments. To develop a conceptual model based on trophic guilds, ERAGS (EPA-540-R-97-006) recommends defining the functional ecosystem components with regard to their role in the food web. Given the complexity of trophic interactions, food webs are a simplification of the ecosystem showing broad relationships limited to trophic transfer. At a base level, some organisms prey on plants (herbivores), plants and animals (omnivores), or just animals (carnivores). More specific feeding classes exist with a particular trophic category. Considering the terrestrial environment, for example, pollen-feeding animals may be relatively unimportant in terms of nutrient and energy transfer through the food web, but they are important as plant pollinators. The same generalities are applicable to considerations of trophic linkages in the aquatic environment (for example, many aquatic invertebrates consume periphyton and use this autotrophic component of the aquatic food web as a refuge from predation). Ultimately, depiction of trophic-level relationships from a functional perspective allows for ready identification of the feeding guilds most at risk from ingestion of contaminated plant and animal materials (RCBRA Report [DOE/RL-2007-21, Volume I]).



Source: PNNL-6415, *Hanford Site National Environmental Policy Act (NEPA) Characterization.*

Figure 3-33. Distribution of Vegetation Types and Area before the 2000 Fire

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This trophic framework is used to describe a simplified structure for the ecological community of the RCBRA (DOE/RL-2007-21) (Figure 3-33). For the most part, trophic linkages among aquatic and terrestrial biota are stronger within habitats than between habitats. In recognition of this, receptors are delineated into aquatic, near-shore, and terrestrial food webs. Some organisms can use both aquatic and terrestrial habitats. For example, bats and kingbirds are aerial insectivores that live on land and meet their dietary demands primarily through the consumption of emergent aquatic insects. The highest trophic level consists of avian predators that can traverse all environments.

Hanford Site-specific receptors are recommended as surrogates for the MTCA ("Terrestrial Ecological Evaluation Procedures" [WAC 173-340-7490]) feeding guilds because they represent relevant ecological endpoints that also address management goals (*DQO Summary Report for the 100 Area and 300 Area Component of the RCBRA* [BHI-01757]). Receptor trophic-based guilds are representative of the upland, riparian, and near-shore environments and include decomposers, producers, and consumers (herbivores, omnivores, insectivores, and carnivores). While categories such as omnivory and herbivory are useful constructs to simplify a complex ecosystem, animals do not typically restrict themselves to narrowly defined food sources. Considerable dietary overlap exists among the middle trophic levels because all species are, to some degree, opportunists. Other species are primarily insectivorous only at times when insects are abundant (*Washington Department of Fish and Wildlife's Priority Habitat and Species Management Recommendations, Vol. IV: Birds – Sage Sparrow*, Amphispiza belli [WDFW, 2003]). Given the dietary overlap, it would be an artificial distinction to focus on a specific category; modeling specific diets (for example, strict herbivory) is done to set the exposure bounds in trophic-transfer analyses.

3.10 Cultural Resources

The Hanford Site contains some of the most important archaeological sites in the region. Many of these sites are eligible for listing on "National Register of Historic Places" (36 CFR 60). In addition, other natural resources and sacred sites important to the cultures of the regional Tribal Nations are preserved at the Hanford Site (*Data Compendium for the Columbia River Comprehensive Impact Assessment* [PNL-9785]). Long-term (that is, more than 50 years) restricted access has minimized looting and vandalism of historic, cultural, and archaeological sites. Furthermore, hydroelectric and agricultural development have not destroyed these culturally significance sites, as has been experienced elsewhere in the Columbia River Basin.

While rapid Hanford Site development did not accommodate protection of important Native American locations, Hanford Site planners, directors of onsite construction activity, and Tribal Nations leaders work together for the protection of important Native American locations. The cultural resources of the Hanford Site are important to many people interested in their historic preservation. The National Register of Historic Places criteria (*National Register of Historic Places Multiple Property Documentation Form-Historic, Archaeological and Traditional Cultural Properties of the Hanford Site, Washington* [DOE/RL-97-02]) offer three suitable categories for chronicling historic, archaeological, and traditional cultural properties of the Hanford Site:

- Prehistoric era (10,000 years B.P. to 1805)
- Homestead and Townsite era (1805 to 1945)
- Manhattan Project and Cold War era (post-1945 to 1990)

DOE has undertaken an ongoing, comprehensive preservation planning effort for the Hanford Site. The results of these efforts have implemented protective programs for conserving cultural resources (*National Register of Historic Places Multiple Property Documentation Form – Historic, Archaeological*

and Traditional Cultural Properties of the Hanford Site, Washington [DOE/RL-97-02]; *Programmatic Agreement Among the U.S. Department of Energy, Richland Operations Office, The Advisory Council on Historic Preservation, and the Washington State Historic Preservation Office for the Maintenance, Deactivation, Alteration, and Demolition of the Built Environment on the Hanford Site, Washington* [DOE/RL-96-77]; and *Hanford Cultural Resources Management Plan* [DOE/RL-98-10]). Cultural resource surveys are routinely conducted as part of site evaluation and preparation before excavation to protect culturally sensitive areas. The results of these surveys are used in the site selection process and applied in the various sampling and analysis plans. Additionally, the creation of the Hanford Reach National Monument (*Hanford Reach National Monument: Final Comprehensive Conservation Plan and Environmental Impact Statement Adams, Benton, Grant and Franklin Counties, Washington* [USFWS, 2008] and "Hanford Reach National Monument; Adams, Benton, Franklin and Grant Counties, WA" [73 FR 72519]) provides an additional means for the preservation and maintenance of the wide range of cultural resources present along the river.

Artifacts discovered across the Hanford Site provide evidence of the Site's occupational characteristics, use durations and periods, and multiple land uses (for example, ceremonial, and religious sites, and burial grounds). Hanford Site cultural resources are diverse, ranging from early prehistoric times to the Atomic Age. Native American archaeological sites are associated with prehistoric and ethnographic villages and activities, as well as sacred and ceremonial areas, such as mountains and rivers where food and medicinal plants were gathered and dispersed across the landscape (*U.S. Department of Energy's Hanford Cultural Resources Laboratory Oral History and Ethnography Task Annual Report* [PNNL-14237]). Many sites and natural features along the Columbia River are regarded as sacred or important to the cultural heritage of members of the CTUIR, Yakama Indian Nation, the Nez Perce Tribe, and the Wanapum. A cultural resources review process was followed for any data collection activities. As with other areas across the Hanford Site, disturbance maps and reports have been prepared for many areas. Tribal Nation leaders review the locations and potential effects to these resources before site activities begin (*Hanford Cultural Resources Management Plan* [DOE/RL-98-10]).

3.10.1 Prehistoric Era

Approximately a dozen prehistoric sites are in 100-D/H, most of which are within 0.4 km (0.25 mi) of the Columbia River. In addition, numerous historic sites are associated with the pre-Hanford Site agricultural period. In general, archaeological sites on the Hanford Reach, including 100-D/H, tend to be on the alluvial flats and lower terraces near the shorelines and islands of the Columbia River. Shoreline sites are generally long and narrow, parallel to the river. Inland prehistoric sites have been discovered on Gable Butte, Rattlesnake Mountain, and near the few isolated seeps. Prehistoric settlement patterns and seasonal rounds in this section of the Columbia Basin were associated with nonagricultural practices that included fishing, upland root gathering, and hunting. Archaeological evidence suggests that pre-contact settlement patterns consisted of consolidated winter villages and dispersed summer camps. Winter villages consisted of long tule mat lodges placed in shallow, bermed pits. Summer camps were associated with seasonal procurement strategies. Long-term prehistoric settlement sites (winter) tend to have pit houses and tool assemblages used for stone tool manufacture and plant and animal preparation. In contrast, short-term seasonal use sites have no pit houses; however, they contain artifacts similar to long-term use sites. Seasonal use of the area centered on the fall fish migrations and winter villages. Seasonal rounds began in the spring with the maturing of plants in the lowland areas and gradually moved to the higher elevations as plant maturation continued into the early fall. Fishing continued from April through September, and hunting occurred in the winter months. Collected food reserves were stored for later winter consumption when plant and fish supplies were the lowest of the year. Archaeological investigations conducted in the Columbia Plateau have enabled the creation of a cultural chronology dating back to the end of the Pleistocene, which is summarized in the following paragraphs

(*Cultural Resources Report for the 100-HR-3 Resource Process Optimization Wells Project, Benton County, Washington* [SGW-44410]).

The Windust Phase (11,000 to 8,000 years B.P.) represents the oldest known Paleo-Indian culture in the Columbia Plateau region. Although archaeological evidence is limited, the people of this period are believed to have been highly mobile hunters and foragers. The food source was primarily large mammals, supplemented with small mammals and fish. Population numbers were low. Living areas are believed to have been in rock shelters and caves. No evidence of constructed dwellings or storage features exists, which further supports the theory of a highly mobile culture. Artifacts from this phase include projectile points, cobble tools, scrapers, gravers and burins, hammer stones, grooved stones, used flakes, bone awls, ocher beads, and antler wedges. Supporting evidence of a Paleo-Indian culture on the Hanford Reach includes a Windust-style projectile point, which was discovered near 100-K in 2001. This projectile point is the oldest known Paleo-Indian point discovered to date at the Hanford Site.

The Cascade/Vantage Phase (8,000 to 4,500 years B.P.) sites include leaf-shaped Cascade projectile points, stemmed projectile points, ovate knives, edge-ground cobble tools, microblades, hammer stones, core tools, and scrapers. The people of this period are believed to have been mobile foragers who relied in part on fish, mussel shells, seeds, and animals. Generally, Vantage Phase sites are at the confluence of major rivers and their tributaries, near intersections of larger side canyons, and along rapids.

People of the Frenchman Springs Phase (4,500 to 2,500 years B.P.) are believed to have been more dependent than their predecessors on the use of natural resources from upland areas. The people from this period also shifted from tools manufactured from fine-grained basalt to cryptocrystalline silica and petrified wood, probably the result of increased upland exploitation. During this period, a shift from chipped stone to ground stone and cobbled implements occurred. Mortars and pestles were first used during this period, suggesting increased reliance on seeds and roots. Semi-subterranean house pits were in use during this period, although not at every location. Research suggests there were both mobile and sedentary foragers with an increased reliance on upland resources.

The Cayuse I Phase (2,500 to 1,200 years B.P.) is characterized by the use of pit houses. The pit houses had level floors, vertical walls with step-like benches, and basal-notched and corner-notched projectile points. The Cayuse II Phase (1,200 to 900 years B.P.) differs only slightly from the earlier phase in that it contains a different pit house design. These pit houses lack the wall benches that characterize the previous phase. Projectile points remain very similar. In the Cayuse III Phase (900 to 250 years B.P.), the number of corner-notched projectile points decreases, and the use of stemmed and side-notched points increases. The number of trade goods also increases during this period. In general, the Cayuse Phase contained well-developed ground stone technologies, small corner-notched and side-notched projectile points, scrapers, lanceolate and pentagonal knives, net weights, pestles, grinding stones, hopper mortars, and cobble implements. During the Cayuse period, populations increased their reliance on fish and root collecting and reduced their reliance on hunting. Horses were introduced in about 1730, increasing the hunting and transportation capabilities. The Cayuse III Phase was also the period with the largest pre-contact populations.

Sahaptin-speaking Wanapum occupied the region of the Columbia River between the Wenatchee and Snake Rivers. Pre-contact population numbers were estimated to be as high as 10,000 before the beginning of the 1800s. By the early to middle 1800s, several epidemics reduced the population to a fraction of its original size. In the mid-1800s, a large group of indigenous people lived at Priest Rapids, referred to by early traders as Priest's Rapids People. Below Priest Rapids, the Wanapum resided at 15 different village locations. Randomly scattered between these village sites along this portion of the Columbia River were areas where small family groups also resided and places where food was cached.

Generally, the Wanapum wintered along the shoreline of the Columbia River, relying on stored foods collected during the yearly seasonal rounds. Plant collecting began in the low elevations in the spring and culminated each year in the upland areas near the end of the summer and early fall months. Midsummer was a time of hunting large and small game, with seasonal camps near the foothills. By fall, the Wanapum would return to the river to pursue the fall fish migrations and prepare for the upcoming winter. Figure 3-34 shows a temporary camp, and Figure 3-35 shows a dugout canoe.

Note: Building in background and car to the left.

Figure 3-34. Native American Temporary Camp in 1945

Figure 3-35. Dugout Canoe in 1945

3.10.2 Homestead and Townsite Era

The Lewis and Clark expedition of 1805 was the initial group of explorers and traders into the lower portion of the Hanford Reach. Their travels began the exploration and subsequent settlement of the region. The explorers sought trade items from the Native Americans and trade routes for traded goods. They were later followed by gold miners, livestock producers, and homesteaders.

By the 1860s, the discovery of gold in the region resulted in a large influx of miners traveling on their way to the gold fields. Several locations along the Hanford Reach such as Ringold, White Bluffs, and Wahluke, were part of the transportation routes used by miners and support industries. Numerous locations with gold mining features believed to be created by Euro-American and Chinese people remain along the shoreline of the Hanford Reach. The mining industry created a demand for beef, and the Columbia Basin was quickly discovered to be the ideal location for livestock production.

A noticeable increase in Euro-American settlement began in eastern Washington in the late 1800s. The initial permanent settlement of non-Indians into the area began slowly with livestock producers that supported gold miners in Alaska and Idaho. Pasture was free for the taking and very abundant. Ranchers relied on the bountiful supply of bunch grass and open rangeland to graze thousands of cattle, and later, sheep and horses. The open range was also an ideal winter pasture. It lasted from the 1880s to about 1910, as homesteaders settled into the area and began to plow up the rangeland to plant crops. Even though the open rangeland was no longer available, livestock remained an important economic commodity to agricultural producers. Agriculture gradually replaced the open-range livestock operations that had dominated the area during the latter part of the 1800s and early 1900s.

Homesteaders developed the agricultural landscape in the Columbia Basin by removing unwanted sagebrush and bunchgrass and plowing the land. The opportunity was brought about by the passage of the *Homestead Act of 1862*, which declared that anyone 21 years of age or older who was willing to live on and develop 65 ha (160 ac) of public land for 5 years was the legal owner. Near the turn of the century, many would-be homesteaders moved west to begin a new life. Many of the homesteaders traveled by one of the three transcontinental railroads (Northern Pacific; Great Northern; or Chicago, Milwaukee, St. Paul & Pacific Railroad) to the Columbia Basin area. Local transportation systems in the Columbia Valley were very limited at that time, so many of the new settlers arrived by river transportation.

Steamboat and ferry service were the primary transportation systems on the Columbia River in the early non-Indian settlement of the area. New agricultural towns of Hanford and White Bluffs, as well as small communities of Allard-Vernita, Wahluke, and Fruitvale, in addition to local rural residents, relied almost exclusively on river transportation during the early development of the area. Initially, when population numbers were low, canoes and ferry operations met the demand. However, as the population increased, steamboat owners took advantage of the opportunity to earn large profits. Many steamboats operated on the Hanford Reach carrying the larger cargoes, while canoes and ferries carried small cargoes of people, animals, and equipment primarily from one shore to the other. At least 10 ferry services operated on the Hanford Reach during their peak. The earliest known ferry service began at White Bluffs in 1859. A ferry service began operation in 100-D in 1880 to transport Chinese gold miners across the Columbia River; Native Americans had previously used this location for a canoe crossing. A store and hotel were established near the crossing. A ferry operated at this location until the Manhattan Project took control of the area in 1943 and closed the ferry after 63 years of operation. Figure 3-36 shows a ferry crossing in the horn area. As increasing numbers of farmers moved into the region, it became apparent that more water, other than small amounts of rain, was needed to produce higher crop yields. Irrigation projects were under construction throughout eastern Washington shortly after the turn of the 20th century.

Note: Ferry boat in approximate center of photograph.

Figure 3-36. Ferry Crossing in Horn Area in 1941

Many irrigation projects began as small-scale, privately funded projects, usually with insufficient funding, and the Hanford Site area was no exception. The Hanford Site area was sought after by developers and producers for its unique geographical ability to produce agricultural crops, especially fruit, 2 to 3 weeks ahead of harvests in surrounding areas. In the early 1900s, wheat and livestock were the primary agricultural commodities produced in Benton County.

By the early 1900s, land speculators began constructing large-scale, privately funded irrigation canals to supply water to thousands of acres in the White Bluffs, Hanford, Fruitvale, Vernita, and Richland areas. Various irrigation techniques were initiated to produce the most affordable irrigation system, which included pumping from wells and canals, and directly from the Columbia River. Poor economic conditions brought about by weak commodity prices and the Depression of the 1930s created economic hardships on most local residents that continued until the area was acquired by the government under the *War Powers Act of 1941* for the Manhattan Project.

3.10.3 Manhattan Project and Cold War Era

The federal government selected the Hanford Site for the location of the Manhattan Project in 1942, and in 1943, approximately 1,500 local residents were removed from their lands for the war effort.

The following year, the Hanford Site was created to support the nation's production of plutonium during World War II. Plutonium production at the Hanford Site continued until 1965, when President Lyndon Johnson declared that the nation's plutonium stockpile had exceeded its needs, and the production of plutonium was gradually decreased. The shutdown of N Reactor in 1986 and its transition to cold standby in 1989 with the end of the Cold War signaled the close of the production mission at the Hanford Site and the start of its environmental cleanup mission, which continues in earnest today. Section 1.2.2 presents additional information on the Manhattan Project and the Cold War Era.

3.11 Summary of Physical Setting

Within this chapter, the key elements of potential contaminant pathways within the environment are discussed. These include a number of important elements for the CSM such as the interrelationships between the geology, the ecology, and the hydrologic cycle. The relationship between these elements and the vadose zone, groundwater, riparian zone, and Columbia River are discussed. The discussion also includes a description of the plants and animals that need to be considered as part of the remedy selection. Historical use of the land by various Native American tribes has resulted in the designation of culturally sensitive sites within 100-D/H. Mitigative or evasive measures may be required to protect these sites during remedial actions, which are proposed at the culmination of this report.

The study area is in the Pasco Basin of Washington. The monthly average temperature ranges from a low of -0.24°C (31.7°F) in January to a high of 24.6°C (76.3°F) in July. Surface winds are predominantly from the northwest and are frequently the result of cooler air draining from the mountains to the northwest. On average, the highest wind speeds occur in March. Average annual precipitation is 17.2 cm (6.8 in.) with most of this occurring during the late fall and winter months when evapotranspiration is lowest. Natural recharge rates to groundwater from precipitation vary from approximately 0 to 100 mm/yr (0 to 3.94 in./yr), depending on plant cover and soil type. In operational areas where the vegetation and topsoil have been removed, a large fraction of this water travels down through the vadose zone, leaching any available contaminants as it drains. The Columbia River is the dominant surface water feature at the site. Columbia River flows typically peak from April through June during spring runoff because of regional and high elevation snowmelt. Flows are lowest from September through October. Flow rates range from approximately 1,020 to 10,300 m^3/s (36,000 to 362,000 ft^3/s) (*Hydrodynamic Simulation of the Columbia River, Hanford Reach, 1940-2004* [PNNL-15226]), depending on the releases from Priest Rapids Dam. At high river stage, the river water can flow impede on the groundwater aquifer along the shoreline and some distance inland, which is very site specific and depends heavily on the stratigraphy of the location.

At 100-D/H, the stratigraphy varies considerably from 100-D, across the horn to 100-H. Basalts are overlain by Ringold Formation material across the entire site. The Ringold Formation consists of Miocene-Pliocene age sediments with several identified units, with both semiconfined and confined aquifers; however not all units exist in all areas across 100-D/H. Ringold Formation unit E is primarily present at 100-D, but is only identified in small pockets across the horn, and has not been identified at 100-H. As a result, the unconfined aquifer is present within the Ringold Formation unit E at 100-D, and in the overlying Hanford formation at 100-H. This difference has a significant effect on groundwater movement in the area, and subsequently on the contaminant distribution, because of a higher hydraulic conductivity within the Hanford formation materials allowing faster groundwater movement in that formation.

Another feature of the geology that is important to groundwater flow is the surface of the Ringold Formation upper mud unit, the RUM. The RUM is a low transmissive confining aquitard consisting predominantly of silt, with some interbedded sandy zones. Results of recent geologic investigations

indicate that the RUM surface has an undulating topography with relief up to several meters. The undulating surface potentially slows groundwater flow with downward dips in the surface potentially controlling flow direction.

The Ringold Formation unit E has been eroded over much of the area east of 100-D, although occasional pockets remain beneath the horn and 100-H. The upper part of the RUM sometimes contains gravel in a silt/clay matrix that may be a transition zone (reworked interval) above the more massive silt or clay. Within the RUM, thin sand-to-gravel lenses form zones with variable hydraulic conductivities that range from low to high.

Across 100-D/H, the vadose zone is comprised of Hanford formation material, although a lower water table in 100-D results in the Ringold Formation unit E also becoming unsaturated. Cross-beds found in the Hanford formation also may locally influence vertical migration of contaminated liquids, although the extent of influence is not known. However, the vertical hydraulic conductivity of even the finer-grained layers is much greater than the annual flux from precipitation and other sources, such that the vadose zone can transmit as much water as is left over after evapotranspiration at the surface.

Vadose zone thickness (and depth to groundwater) ranges from 0 to 27 m (0 to 89 ft), with an average thickness of 11.3 m (37.1 ft) in 100-H and an average thickness of 20 m (65.4 ft) in 100-D. Average vadose zone thickness within 100-D/H is estimated to be 15.6 m (51.3 ft). The vadose zone in 100-D consists of a thin veneer of permeable surface soil, underlain by Hanford formation sediments and Ringold Formation unit E sediments. The vadose zone in 100-H consists of a thin veneer of permeable surface soil underlain by Hanford formation sediments, with the horn being a transitional area geologically.

The Hanford Site's arid climate keeps the vadose zone soil moisture relatively low. Historically, effluent discharge to the soil column increased soil moisture beneath waste sites. Soil moisture ranged from less than 2 percent to 10 percent, with one sample slightly exceeding 10 percent. Samples collected from near the water table generally had higher moisture content than those samples collected higher in the vadose zone, the exception being those samples near the surface that occasionally exhibit higher moisture content because of precipitation. Vadose zone soil data show that the Ringold Formation unit E has similar grain size, porosity, and bulk densities as those of the Hanford formation, yet the Ringold Formation unit E has greater moisture content (at 100-D) than that of the Hanford formation.

During reactor operations at 100-D/H, the rates and direction of groundwater flow in the unconfined aquifer were controlled by the underlying geology, while being strongly influenced by surface and subsurface discharge of cooling water effluent and operation of the water/wastewater infrastructure. Infiltration and overland flow of contaminated cooling water from surface features and from leaks at the 100-D and 100-H Area retention basins created significant vertical (downward) fluxes within the vadose zone that would have increased any potential for vertical migration of contaminants released to the aquifer. Since reactor operations ceased, 100-D/H hydrogeology has been influenced by the following:

- Groundwater remediation activities
- Operation of the Export Water System and other water/wastewater infrastructure
- Application of dust suppression water during waste site remediation
- Annual and diurnal fluctuations in Columbia River stage

Groundwater flows into 100-D/H from the south and then regionally bends toward the lower hydraulic heads at 100-H. At 100-D some groundwater discharges to the Columbia River, however most of the groundwater flows from 100-D across the horn to 100-H. River stage affects groundwater near the river,

influencing groundwater elevations more than 700 m (2,300 ft) inland at 100-D, and more than 640 m (2,100 ft) inland at 100-H. When river stage is low, natural groundwater flow is from 100-D/H toward the river. When river stage is high, water can flow from the river inland and mix with 100-D/H groundwater.

Current hydraulic gradient magnitudes and directions within the unconfined aquifer are generally toward the Columbia River, but they show some seasonal variation in response to changes in river stage. Gradients steepen toward the river during low river stage (fall and winter), and flatten or may reverse near the river shoreline during high river stage (spring). Local gradients are also heavily influenced by the operation of the pump-and-treat systems in the 100-D and 100-H Areas. During 2011, groundwater pump-and-treat remedies were reconfigured and expanded following RPO activities. Four groundwater pump-and-treat systems operated for all or part of 2011:

- In the 100-D Area, the DX system operated for the entire year, while the DR-5 system operated from January to April.
- In the 100-H Area, the HR-3 system operated from January to May, while the HX system operated from late September to December.

Because of the operation of these remedies, and the influence of the Columbia River stage, groundwater levels and hydraulic gradients within the unconfined aquifer varied widely during 2011.

Near the 100-D southern plume, the hydraulic gradients during 2011 varied in magnitude from about 0.0002 to about 0.008. The gradient direction was generally north/northwest toward the Columbia River; however, gradients shifted to the north/northeast for a brief period from May through August 2011, coinciding with high stage in the Columbia River. The flow velocity ranged from 0.0000013 to 0.000064 cm/sec (0.004 to 0.181 ft/day). Near 100-H, the gradient direction was generally north/northeast toward the Columbia River; however, gradients shifted to the south/southeast for a brief period from May through August 2011, coinciding with high stage in the Columbia River. The flow velocity ranged from 0.0000031 to 0.000012 cm/sec (0.009 to 0.034 ft/day).

Intercommunication between different aquifers is indicated at nested Wells 199-H3-2A, 199-H3-2B, and 199-H3-2C within 100-H. During a step test and constant rate pump test of Well 199-H3-2C, the unconfined aquifer exhibited characteristics of a leaky aquifer, with groundwater levels in nearby water table wells showing drawdown in response to pumping in the first water-bearing unit in the RUM. Geochemical data evaluated during this RI also indicate a connection between the aquifers at Well 199-H3-2C. Aquifer connectivity was not identified in the nested well sets at Wells 199-H4-15CS and 199-H4-12C.

An examination of the vertical hydraulic gradient between the various aquifers indicated that a weak upward gradient is present in some locations in 100-D/H, with other areas showing either a downward or equipotential gradient. These conditions, combined with a less cemented and thinner zone between the surface of the RUM and the first water bearing unit of the RUM are conducive to allowing transmission of contaminated groundwater from the unconfined aquifer to the first water bearing unit in the RUM. Under operating conditions, the groundwater mound formed at both the D and H Reactors would have allowed migration to the lower aquifer.

Groundwater monitoring wells in the southern Cr(VI) plume in 100-D, with a few exceptions, have similar geochemical pattern. Wells in the southern plume have higher relative levels of calcium, with high levels of carbonate and sulfate. The geochemical signature in the northern plume is similar to that found in the southern plume, with the exception of higher sodium plus potassium, sulfate, and magnesium levels. Wells with a different geochemical pattern are those affected by local contamination or other

groundwater additives. The reducing agent used in the barrier consists of sodium dithionite ($Na_2S_2O_4$). As expected, the sodium and sulfate levels in the groundwater at the ISRM well are much higher than levels in other wells within 100-D/H. The wells in the horn have a similar geochemical signature to each other.

Monitoring Wells 199-H3-2A, 199-H3-2B, 199- H3-2C, 199-H4-12A, and 199-H4-12B have a similar geochemical pattern. Of these wells, only one (Well 199-H3-2C), was completed in the RUM. The other two RUM wells with adequate geochemical data are Well 199-H4-12C and 199-H4-15CS. These wells are located near the Columbia River, and have a geochemistry similar to river water but different from the chemistry found in the associated nested wells. The water levels in these wells also respond to changes in river stage (*Aquifer Testing and Rebound Study in Support of the 100-H Deep Chromium Investigation* [SGW-47776]). The observation of similar geochemistry supports the theory that the RUM is hydrologically connected to the river in that location.

The Hanford Site area was a seasonal home to human inhabitants dating back at least 11,000 years. For hundreds of years before the end of the Frontier Period in 1890, the Wanapum and other tribes used the area as a seasonal homeland. These peoples would take advantage of the fish runs in the fall and winter along the river before returning to higher ground in the spring and summer. European settlement followed before the taking of the land by the federal government in 1943.

4 Nature and Extent of Contamination

Evaluation of the nature and extent of contamination describes contaminant concentrations found in the environmental media in the study area. Contamination is determined from recently collected RI and RPO data, data from the Columbia River RI Work Plan (DOE/RL-2008-11), data from the RCBRA (DOE/RL-2007-21, Volume I), data available from previous limited field investigations (LFIs), ongoing air and water monitoring, completed interim remediation (that is, CVP data), and historical operational process information. Reported concentrations of the various analytes are compared to vadose zone background concentrations as a initial screening tool to identify contaminants of potential concern (COPCs) associated with 17 RI waste sites within 100-D/H.

Following the comparison to background levels, the contaminants are described in relation to their nature and extent. As such, this chapter focuses principally on vadose zone and groundwater COPCs. Uncertainties associated with the data, as they relate to the nature and extent of contamination, also are described.

Chapter 4 of the 100-D/H Work Plan (DOE/RL-2008-46-ADD1) described the major features of the CSM. These concepts provide the basic framework for interpreting the data collected under the RI to fulfill the data gaps and data needs developed in the 100-D/H Work Plan (DOE/RL-2008-46-ADD1). Section 4.7 of this RI report presents refinement and discussion of the CSM in the context of the results. This chapter continues to develop the CSM with nature and extent information regarding 100-D/H media (soil, groundwater, air, biota, and surface water/sediment).

Highlights

- Analytes detected in the vadose zone were compared to background concentrations. Initial screening identified more than 70 analytes above background concentrations (see Table 4-6).
- COPC concentrations in the vadose zone vary with depth. However, most contaminant concentrations generally decrease with depth. Higher concentrations are typically in the upper half of the vadose zone.
- Cr(VI) plumes in groundwater are associated with past reactor operations at 100-D and 100-H. Migration from 100-D across the Horn toward 100-H has resulted in a large connected Cr(VI) plume within D/H with an area greater than 10 km^2 (3.86 mi^2).
- Cr(VI) contamination has been identified in the first water-bearing unit within the RUM at 100-H near the river, and in one well in the horn.
- Nitrate plumes are present primarily in 100-D coincident with the Cr(VI) plume with elevated nitrate at selected wells in 100-H.
- Strontium-90 is present in a small plume in 100-H east of the reactor and in one well in 100-D. These localized areas are associated with the fuel storage basins.
- Several likely continuing sources of Cr(VI) to groundwater contamination are identified at waste sites (for example,100-D-100, 100-D-104, and 100-D-30) undergoing active remediation.

Contaminants in the vadose zone, periodically rewetted zone (PRZ), and groundwater resulted from various activities during reactor operations. Under current conditions, the primary contributor to groundwater contamination at 100-D/H is vadose zone contamination from unremediated waste sites (for example, Cr(VI) from 100-D-100, 100-D-30, and 100-D-104). Contaminants from waste sites and facilities were transported through the vadose zone, into the PRZ, and then into the groundwater. Less mobile contaminants tend to stay bound to soil particles in the vadose zone and PRZ, while more mobile contaminants tend to move through the vadose zone and PRZ into the groundwater, due to infiltration and changing groundwater elevations caused by Columbia River stage changes.

Much of the data collected during implementation of interim remedial actions has been documented in CVPs and LFIs, which are incorporated into the discussion of the nature and extent of contamination. Information is also presented to describe the current understanding of contamination attributed to 100-D/H in the Columbia River, biota, and air, and is summarized from the Hanford Site Releases Data

Summary (WCH-398), RCBRA (DOE/RL-2007-21, Volume I), and *Hanford Site Environmental Report for Calendar Year 2011* (DOE/RL-2011-119). Section 4.3 describes vadose zone contamination associated with locations selected for new boreholes/wells under the RI. Section 4.4 describes groundwater contamination. Sections 4.4.5, 4.5, and 4.6 discuss Columbia River surface water/sediments, biota, and air, respectively.

4.1 Background Concentrations

Background substances are usually naturally occurring (present in the environment in forms not influenced by human activity) or anthropogenic (natural and/or artificial forms present in the environment due to human activities not related to the CERCLA site(s) under consideration). Some chemicals may be present in background because of both natural and artificial conditions, such as naturally occurring arsenic and arsenic from historical agricultural pesticide applications (*Guidance for Comparing Background and Chemical Concentrations in Soil for CERCLA Sites* [EPA 540-R-01-003]).

The identification of background concentrations of substances in soil is one step in determining if potential waste sites require remedial action. These concentrations are also important because in some instances, calculated risk-based benchmarks (substance concentrations that may have the potential to present risk to human or ecological receptors) are less than background levels. Where benchmarks are less than background levels, cleanup goals generally default to background (rather than the calculated values) because CERCLA typically does not require cleanup to concentrations below background levels.

The background concentrations used in this section represent 90th percentile values that are determined from a range of Hanford Site background sample concentrations. For example, the 104 total chromium background sample concentrations used to calculate the 90th percentile soil value (18.5 mg/kg) ranged from 2.9 to 30.6 mg/kg. Similarly, the 104 lead background sample concentrations used to calculate the 90th percentile soil value (10.2 mg/kg) ranged from 1.1 to 26.6 mg/kg. As such, contaminant concentrations may exceed the 90th percentile background value and still remain within the range of natural Hanford Site background. As part of the RI, supplemental investigations developed River Corridor background soil values for antimony, boron, cadmium, lithium, mercury, molybdenum, selenium, silver, and thallium. The investigation results are in *Soil Background Data for Interim Use at the Hanford Site* (ECF-HANFORD-11-0038).

In addition to background concentrations of metals, orchard lands are potential contributors of arsenic and lead to the soil. Collocated within the historical orchard land areas are waste sites related to releases from Hanford Site operations (Figure 4-1). The 100-OL-1 OU has been established in the 100 Area and sections of 100-DH to address residual lead and arsenic contamination in the soil from pre-Hanford agricultural pesticide use. The contaminants associated with these waste sites will continue to be evaluated and addressed through the RI/FS process for the various areas (100-BC, 100-K, 100-N, 100-D/H, or 100-F) where the individual waste sites are geographically located. During implementation of the selected remedy at these waste sites, contaminants present will be remediated as needed to meet the cleanup levels prescribed in the applicable ROD. Should contaminants associated with historical orchard lands (for example, lead, and arsenic) be present at any particular waste site, that contamination will not be remediated beyond the waste site footprint as part of the ROD. Any contaminants remaining outside the waste site footprint will be addressed as part of the remedial investigation for the 100-OL-1 OU. This approach will allow reclassification of individual waste sites that meet the cleanup standards (for non-orchard lands related contaminants) of the applicable decision area ROD while supporting the broad area investigation of historical orchard lands as part of the 100-OL-1 OU.

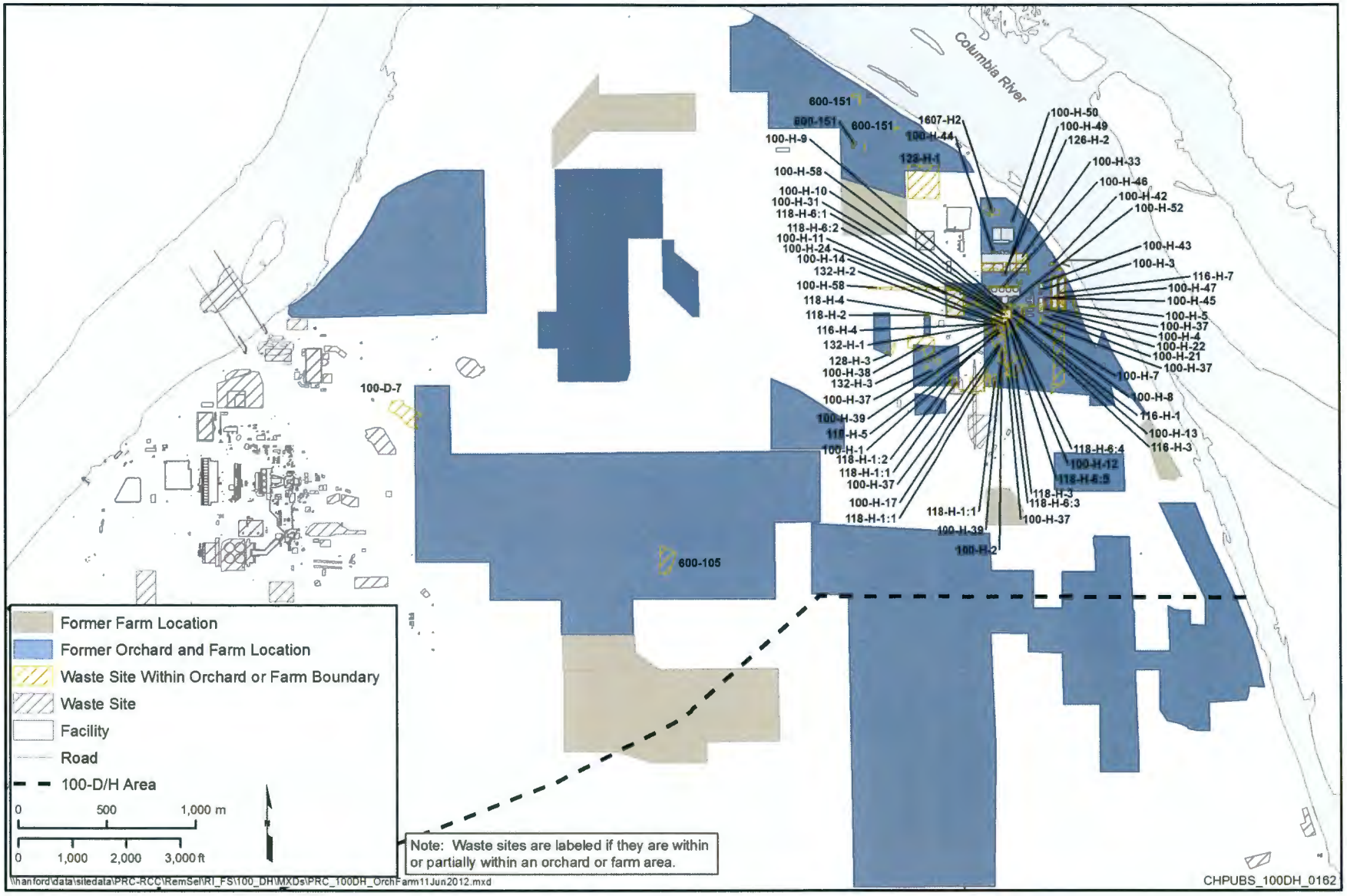

Figure 4-1. Locations of Former Orchards, Farms, and Waste Sites in 100-D/H

Details for the handoff of actions between the decision area RODs and the 100-OL-1 OU investigations will be established in the remedial design report/remedial action work plan (RDR/RAWP) documents associated with each decision area ROD. An example of this approach as implemented for the interim action RODs is provided by *Federal Facility Agreement and Consent Order, Modify Remedial Design Report/Remedial Action Work Plan for the 100 Area (DOE/RL-96-17, Rev. 6) to Add Section 3.6.10 Residual Pesticides from Agriculture Use* (TPA-CN-401).

Soil background values are benchmarks to define contamination, as well as identify preliminary COPCs. Soil analytes that do not have established background concentrations, but are detected at concentrations greater than method detection limits are also considered preliminary COPCs and are further evaluated in Chapters 5, 6, and 7. The groundwater background values are primarily reference points and are not applied to the discussion of groundwater contamination. This is because filtered samples were used to develop groundwater background values due to the variability of geochemical conditions across 100-D/H. Tables 4-1 and 4-2 present background soil and groundwater concentrations, respectively, derived for the Hanford Site.

Table 4-1. Background Concentrations in Hanford Site Soil

CAS Number	Analyte	Abbreviation	90th Percentile	Reference
Radionuclides (pCi/g)				
14596-10-2	Americium-241	Am-241	--	--
14762-75-5	Carbon-14	C-14	--	--
10045-97-3	Cesium-137[a]	Cs-137	1.05	DOE/RL-96-12
10198-40-0	Cobalt-60	Co-60	0.00842	DOE/RL-96-12
14683-23-9	Europium-152	Eu-152	--	
15585-10-1	Europium-154	Eu-154	0.0334	DOE/RL-96-12
14391-16-3	Europium-155	Eu-155	0.0539	DOE/RL-96-12
13981-37-8	Nickel-63	Ni-63	--	--
13981-16-3	Plutonium-238	Pu-238	0.00378	DOE/RL-96-12
15117-48-3	Plutonium-239/240[a]	Pu-239/240	0.0248	DOE/RL-96-12
10098-97-2	Strontium-90[a]	Sr-90	0.178	DOE/RL-96-12
14133-76-7	Technetium-99	Tc-99	--	--
10028-17-8	Tritium	H-3	--	--
13966-29-5	Uranium-233/234	U-233/234	1.10	DOE/RL-96-12
15117-96-1	Uranium-235	U-235	0.109	DOE/RL-96-12
7440-61-1	Uranium-238	U-238	1.06	DOE/RL-96-12

Table 4-1. Background Concentrations in Hanford Site Soil

CAS Number	Analyte	Abbreviation	90th Percentile	Reference
Nonradionuclides (mg/kg)				
7429-90-5	Aluminum	Al	11,800	DOE/RL-92-24
7440-36-0	Antimony	Sb	0.13	ECF-HANFORD-11-0038
7440-38-2	Arsenic	As	6.47	DOE/RL-92-24
7440-39-3	Barium	Ba	132	DOE/RL-92-24
7440-41-7	Beryllium	Be	1.51	DOE/RL-92-24
7440-42-8	Boron	B	3.89	ECF-HANFORD-11-0038
7440-43-9	Cadmium	Cd	0.56	ECF-HANFORD-11-0038
7440-47-3	Chromium (Total)	Cr	18.5	DOE/RL-92-24
18540-29-9	Chromium (Hexavalent)	Cr(VI)	--	--
7440-48-4	Cobalt	Co	15.7	DOE/RL-92-24
7440-50-8	Copper	Cu	22.0	DOE/RL-92-24
7439-92-1	Lead	Pb	10.2	DOE/RL-92-24
7439-93-2	Lithium	Li	13.3	ECF-HANFORD-11-0038
7439-96-5	Manganese	Mn	512	DOE/RL-92-24
7439-97-6	Mercury	Hg	0.01	ECF-HANFORD-11-0038
7439-98-7	Molybdenum	Mo	0.47	ECF-HANFORD-11-0038
7440-02-0	Nickel	Ni	19.1	DOE/RL-92-24
7782-49-2	Selenium	Se	0.78	ECF-HANFORD-11-0038
7440-22-4	Silver	Ag	0.167	ECF-HANFORD-11-0038
7440-24-6	Strontium metal (strontium)	Sr	--	--
7440-28-0	Thallium	Tl	0.18	ECF-HANFORD-11-0038
7440-62-2	Vanadium	V	85.1	DOE/RL-92-24
7440-66-6	Zinc	Zn	67.8	DOE/RL-92-24
16984-48-8	Fluoride	F^-	2.81	DOE/RL-92-24
14797-55-8	Nitrate	NO_3^-	52	DOE/RL-92-24
14797-65-0	Nitrite	NO_2^-	[b]	DOE/RL-92-24
75-35-4	1,1-Dichloroethene	--	--	--

Table 4-1. Background Concentrations in Hanford Site Soil

CAS Number	Analyte	Abbreviation	90th Percentile	Reference
106-46-7	1,4-Dichlorobenzene	--	--	--
83-32-9	Acenaphthene	--	--	--
120-12-7	Anthracene	--	--	--
12674-11-2	Aroclor-1016	--	--	--
11104-28-2	Aroclor-1221	--	--	--
11141-16-5	Aroclor-1232	--	--	--
53469-21-9	Aroclor-1242	--	--	--
12672-29-6	Aroclor-1248	--	--	--
11097-69-1	Aroclor-1254	--	--	--
11096-82-5	Aroclor-1260	--	--	--
71-43-2	Benzene	--	--	--
56-55-3	Benzo(a)anthracene	--	--	--
50-32-8	Benzo(a)pyrene	--	--	--
205-99-2	Benzo(b)fluoranthene	--	--	--
207-08-9	Benzo(k)fluoranthene	--	--	--
191-24-2	Benzo(ghi)perylene	--	--	--
117-81-7	Bis(2-ethylhexyl)phthalate	--	--	--
86-74-8	Carbazole	--	--	--
56-23-5	Carbon tetrachloride	CCl_4	--	--
67-66-3	Chloroform	$CHCl_3$	--	--
218-01-9	Chrysene	--	--	--
53-70-3	Dibenz[a,h]anthracene	--	--	--
84-74-2	Di-n-butylphthalate	--	--	--
107-21-1	Ethylene Glycol	--	--	--
206-44-0	Fluoranthene	--	--	--
193-39-5	Indeno(1,2,3-cd)pyrene	--	--	--
75-09-2	Methylene Chloride	CH_2Cl_2	--	--
87-86-5	Pentachlorophenol	--	--	--

Table 4-1. Background Concentrations in Hanford Site Soil

CAS Number	Analyte	Abbreviation	90th Percentile	Reference
85-01-8	Phenanthrene	--	--	--
129-00-0	Pyrene	--	--	--
127-18-4	Tetrachloroethene	PCE	--	--
79-01-6	Trichloroethylene	TCE	--	--
108-88-3	Toluene	--	--	--
68334-30-5	Total Petroleum Hydrocarbons	TPH	--	--

Sources: *Hanford Site Background: Part 1, Soil Background for Nonradioactive Analytes* (DOE/RL-92-24).
Hanford Site Background: Part 2, Soil Background for Radionuclides (DOE/RL-96-12).
Soil Background Data for Interim Use at the Hanford Site (ECF-HANFORD-11-0038).

a. Cesium-137, strontium-90, and plutonium-239/240 are anthropogenic radionuclides whose background values only apply to surface soil samples.

b. Insufficient data above the reporting limit to provide for a distribution fit.

-- = either a background study has not been performed for this analyte (i.e., strontium) or the constituent does not occur naturally in the environment (i.e., the organic constituents).

Table 4-2. Hanford Site Groundwater Background Concentrations for COPCs in 100-D/H Groundwater

Constituent	Units	90th Percentile
Nonradionuclides		
Antimony (filtered)	µg/L	55.1
Arsenic (filtered)	µg/L	7.85
Barium (filtered)	µg/L	105
Beryllium (filtered)	µg/L	2.29
Cadmium (filtered)	µg/L	0.916
Chloride (unfiltered)	µg/L	15,630
Chromium (total, filtered)	µg/L	2.4
Cobalt (filtered)	µg/L	0.916
Copper (filtered)	µg/L	0.81
Cyanide	µg/L	8.41
Fluoride	µg/L	1,047
Lead (filtered)	µg/L	0.917

Table 4-2. Hanford Site Groundwater Background Concentrations for COPCs in 100-D/H Groundwater

Constituent	Units	90th Percentile
Manganese (filtered)	µg/L	38.5
Mercury (filtered)	µg/L	0.003
Nickel (filtered)	µg/L	1.56
Nitrate (unfiltered)	µg/L	26,871
Nitrite (unfiltered)	µg/L	93.7
Selenium (filtered)	µg/L	10.5
Sulfate (unfiltered)	µg/L	47,014
Thallium (filtered)	µg/L	1.67
Uranium	µg/L	9.85
Vanadium (filtered)	µg/L	11.5
Zinc (filtered)	µg/L	21.8
Radionuclides		
Strontium-90	pCi/L	0.0146
Tritium	pCi/L	119

Source: *Hanford Site Background: Part 3, Groundwater Background* (DOE/RL-96-61).

Note: The organic COPCs 1,1-dichloroethene, 1,1,2-trichloroethane, benzene, carbon tetrachloride, chloroform, tetrachloroethene, TCE, and vinyl chloride are assumed to have natural background concentrations of zero.

4.2 Sources

Section 1.2.2 discusses the site history of 100-D/H. The primary sources of contamination in 100-D/H are liquid and solid wastes generated and released during the operation of the reactors and support facilities, and from unplanned releases. The reactor operations responsible for generating and releasing contaminants to the environment have all been discontinued. Secondary sources are contaminants remaining in the vadose zone and within the aquifer matrix.

This section discusses what is considered a primary source and what is considered a secondary source, and highlights certain COPCs because of their observed distribution or persistence in the environment at 100-D/H. The same individual contaminants may be found in both the original primary source material that was released (for example, liquid and solid waste streams discharged to the environment), and in the secondary sources that remain (for example, contaminated vadose zone soil). Contaminants that are currently present in secondary sources were typically released as primary source material. Limited primary source material may be encountered during the implementation of remedial activities in structures, pipelines, and other process components. Residual material remaining in piping is typically found as pipe scaling and has a limited potential to be released as a secondary source to the vadose zone. Assessment of the potential for continuing releases from remaining secondary sources is an element of the remedial investigation.

Liquid waste sources can be classified into two types: high volume/low concentration liquid wastes and low volume/high concentration liquid wastes. The volumes of liquid effluent waste streams varied over orders of magnitude. The largest volume streams were generated as steam condensate, cooling water, and unplanned releases. To generate the cooling water solutions for the 105-D, 105-DR, and 105-H Reactors, concentrated sodium dichromate solid and liquid feed solutions were mixed in the cooling water system to achieve the required coolant concentration. As discussed in Chapter 1, the primary contaminants related to the cooling water include Cr(VI), tritium, strontium-90, and various radionuclides, with the radionuclides being a result of the cooling water passing through the reactors.

Solid wastes in 100-D/H were generated in facilities and managed mainly in burial grounds. According to WIDS, the burial ground waste consists of numerous trenches and vertical steel pipes of various sizes that contain radioactive solid waste from the 105-D, 105-DR, and 105-H Reactors. Solid wastes were also disposed into burn pits and dumping areas, and as unplanned releases.

4.2.1 Primary Sources

The primary sources of contamination in 100-D/H are three water-cooled nuclear reactors (105-D, 105-DR, and 105-H [Figure1-2]), and the structures (for example, fuel storage basins) and processes (for example, sodium dichromate process) associated with reactor operations. The three reactor buildings remain intact today in a safe storage enclosure. Most of the associated structures and facilities near the reactor have been demolished or removed. The reactors were built to irradiate uranium-enriched fuel rods from which plutonium and other special nuclear materials could be extracted. The reactors and processes associated with operations generated large quantities of liquid and solid wastes. Effluent generated during operations consisted primarily of contaminated reactor cooling water, fuel storage basin water, and decontamination solutions. Cooling water consisted of river water treated to remove dissolved solids and enhanced with chemicals to reduce corrosion. Cooling water contaminants consisted of fuel materials, fission and irradiation byproducts, and Cr(VI) (used as a corrosion inhibitor). Solid wastes consisted of sludge, reactor components, and various other contaminated items. Waste generated from reactor operations was contaminated with radionuclides, hazardous chemicals, or both.

The target analyte list for contaminants in soil was based on process knowledge, as described in *100-D/100-H Target Analyte List Development for Soil* (WCH-322). COPCs in groundwater were developed for the 100-D/H SAP (DOE/RL-2009-40) as described in *Identification of Contaminants of Potential Concern for Groundwater Risk Assessment at the 100-HR-3 Groundwater Operable Unit* (ECF-100HR3-10-0469). Tables 2-3 through 2-18 of the 100-D/H SAP (DOE/RL-2009-40) present the waste site-specific target analytes and analytical methods for determination of the analytes.

Liquid Effluent Waste Sources. The volumes of liquid effluent waste streams discharged to specific waste sites varied over several orders of magnitude. The largest volume streams were generated as steam condensate, cooling water, and unplanned releases of cooling water. The primary contaminants related to the cooling water include Cr(VI), carbon-14, tritium, strontium-90, and various other radionuclides.

Concentrated Water Treatment Chemical Waste Sources. Substantial volumes of chemicals were used to condition the cooling water used by the reactors. These include chlorine, sulfuric acid, alum, sodium chloride, sodium hydroxide, and sodium dichromate dihydrate. These chemicals were stored in bulk at the water treatment head houses for each reactor (183-D and 183-H) and were metered into the cooling water stream at various points ultimately to provide a continuous

stream of cooling water with low solids and conditioned for corrosion resistance. Over the course of operations, varying volumes of these chemicals were released to the environment, either routinely or episodically, in the vicinity of the chemical storage and handling areas. The sources consisted of low-volume, high-concentration sodium dichromate and variable volumes of low-concentration sodium dichromate in liquid effluent. To generate the cooling water solutions for the 105-D, 105-DR, and 105-H Reactors, concentrated sodium dichromate feed solutions were processed through an infrastructure system that diluted the higher-strength source materials to achieve the required coolant composition. Reactor operations at 100-D/H used both concentrated sodium dichromate solution and granular sodium dichromate (see Figures 1-14 and 1-15).

Solid Waste Primary Sources. The primary solid waste source area types are buildings, burial sites, and solid waste sites. The 118-D-3 and 118-D-4 Burial Grounds were the primary disposal sites for radioactive solid wastes at 100-D. The primary disposal site for radioactive solid waste from the 105-H Reactor was the 118-H-1 Burial Ground. However, numerous other burial grounds received radioactive waste at 100-D/H. Solid wastes disposed to these waste sites include a variety of radiologically contaminated and irradiated materials consisting of reactor hardware including irradiated dummy fuel elements, splines, rods, thimbles, and various other solid, and potentially liquid, waste in containers. These waste sites consist of numerous trenches and vertical steel pipes of various sizes that contain radioactive solid waste from 105-D, 105-DR, and 105-H Reactors. Waste from the 105-N Reactor was also disposed at 100-D. Occasional fires at burial grounds were the source of unplanned releases.

Coal Ash Sites. Coal-fired power plants were associated with the D and H reactors. Coal ash is considered a solid waste issue at 100-D/H. There are two coal ash waste sites in 100-D/H, including two sites that are classified as "rejected" waste sites—126-D-1 and 126-H-1. Coal ash sites are not considered to constitute hazardous wastes; therefore, these sites are not considered further under CERCLA.

Nonoperational Areas and Orphan Sites. The nonoperational areas at 100-D/H have been evaluated through the OSE process described in Appendix K. This evaluation includes not only the potential for anthropogenic disposal activities but also considers windblown dust emissions, stack emissions, overland flow, and possible contaminant placement because of biointrusion by potential carriers such as wasps. An historical evaluation was performed inside the exclusion area and walk-downs conducted outside the exclusion area. New discoveries of waste sites not associated with existing waste sites is unlikely.

Secondary Sources. Contaminants released to the environment during reactor operations contaminated the vadose zone beneath facilities and waste sites. These secondary sources of contamination pose potential human health and the environment exposures through numerous pathways (for example, direct contact, inhalation, and/or ingestion of contaminated soil, groundwater, and/or surface water). Contaminants from waste sites and facilities were transported through the vadose zone, into the PRZ, and then into the groundwater. Less mobile contaminants tend to stay bound to soil particles in the vadose zone and PRZ, while more mobile contaminants tend to move through the vadose zone and PRZ into the groundwater due to driving forces (during reactor operations and under natural rainfall conditions). As groundwater elevations rise and fall across the PRZ due to Columbia River stage changes, contaminants that are more mobile have the potential to leach into the groundwater. This includes contaminated soil in the PRZ, which is the lower portion of the vadose zone that is contacted by groundwater during periods of high groundwater elevation.

Chapter 1 presents the operational periods of the facilities and reactors. The reactor processes responsible for generating and releasing primary sources to the environment have all been discontinued. Contaminants remaining as secondary sources may continue to migrate through the environment, depending on environmental conditions, and the individual constituent properties. Section 4.2.3 lists the constituents detected in 100-D/H groundwater samples collected since 2005.

Chapters 6 and 7, respectively, discuss the evaluation of risks posed by the identified secondary sources to human health and the environment through direct exposure. Interim actions continue to address the risks posed by contaminants. It is anticipated that most sources of contamination will have been addressed through interim actions before the ROD is released. The potential for secondary sources to provide a significant ongoing source of contamination to groundwater is evaluated through the comparison of post remedial action contaminant concentrations to the screening levels for groundwater and surface water protection in Chapter 5.

The following sections briefly discuss contaminants seen in the vadose zone and in groundwater. Contaminants present in the vadose zone have the potential to affect human health and the environment through direct exposure, and are identified in Chapter 8, Tables 8-2 and 8-3. Four of these contaminants are shown to have affected groundwater at 100-D/H—Cr (VI), total chromium, nitrate, and Sr-90. The RI results and pertinent historical data for the vadose zone and groundwater are presented in more detail in Sections 4.3 and 4.4, respectively.

4.2.2 Sources of Specific Contaminants at 100-D/H

The major contaminants of interest at 100-D/H originated from chemical materials used during reactor operations. The following paragraphs discuss the processes that contributed these contaminants to the environment.

4.2.2.1 Hexavalent Chromium

In the hexavalent state (Cr(VI)), chromium is present as a soluble oxyanion and because of its mobility and widespread presence, has a potential effect on human health and the environment (100-HR-3 and 100-KR-4 Interim ROD [EPA/ROD/R10-96/134]; 100-D/H Work Plan [DOE/RL-2008-46-ADD1]). Cr(VI) is present in the groundwater at 100-D/H at concentrations exceeding aquatic ("Toxics Criteria for Those States Not Complying with Clean Water Act Section 303(c)(2)(B)" [40 CFR 131.36]) and MTCA (WAC 173-340) B levels.

During operation of the D, DR, and H Reactors and associated facilities, numerous locations received highly concentrated sodium dichromate solutions. This stock solution was fed into the cooling water treatment system for mixing and dilution before entering the reactors. After passing through each reactor, the low-concentration sodium dichromate solutions were discharged to retention basins and selected trenches and cribs. After operations, reactor decontamination wastes were discharged to the 116-DR-1&2 Trench. Figures 1-15 and 1-16 show facilities where sodium dichromate was handled.

Sodium dichromate dihydrate ($Na_2Cr_2O_7 \cdot 2H_2O$), the chemical form of the treatment product containing Cr(VI), was delivered as a solid and concentrated 70 wt % liquid by rail tanker cars and was transferred to aboveground bulk storage tanks. It was added to the reactor cooling water to inhibit corrosion (*100-D Area Technical Baseline Report* [WHC-SD-EN-TI-181]). Figure 1-12 shows the general flow path of the sodium dichromate. Solid sodium dichromate was stored and mixed with water at the 108-D Building Chemical Pump House and the 185-D Deaerating Plant from 1955 until 1959. The concentrated solution, containing about 700 g/L sodium dichromate

dihydrate, was metered into the cooling water feed stream to achieve a working concentration of to 2,000 μg/L. By 1964, that amount was reduced to 1,000 μg/L (see Section 1.2.2.5.).

The quantities of sodium dichromate received, handled, and processed each month in 100-D/H were essentially the amount needed to provide the 2,000 μg/L (ppb) concentration of sodium dichromate in the reactor cooling water. At an average cooling water usage rate of approximately 30,000 gal/min at each at the three reactors (105-D, 105-DR, and 100-H), consumption was approximately 0.23 kg of sodium dichromate from 0.32 L (0.085 gal) of stock solution per minute per reactor. This led to approximately 467 L (123.4 gal) per day of stock solution, which required one 19,000 L (5,000 gal) railcar every 41 days per reactor. With the presence of two operating reactors at 100-D, more than one railcar per month was required.

Because of the volume of solution transferred, spills and leaks of concentrated liquid solutions of sodium dichromate materials during receiving, handling, and processing activities near the 100-D-12 waste site, 108-D Building, 185-D Building, and the 100-D-56 pipeline likely occurred on a regular basis. Spills and leaks in these areas upstream from the 190 Building are the most likely source of observed Cr(VI) groundwater contamination. Spills of sodium dichromate at cooling water support facilities had the greatest potential for environmental contamination. Decontamination wastes produced in 100-D/H from the reactor were commingled with other liquids and were routed for disposal in various trenches.

Much of the cooling water was discharged directly to the Columbia River through the outfall pipe system. Discharges of cooling water to the ground downstream from the reactor through leaks in retention basins and trenches and cribs typically infiltrated through the vadose zone into the aquifer and eventually discharged to the Columbia River through the groundwater flow system.

4.2.2.2 Radionuclides

The primary radionuclides associated with reactor operations that resulted in vadose zone and/or groundwater concerns at 100-D/H are cesium-137, cobalt-60, europium-152 and -154, strontium-90, tritium, and technetium-99.

Strontium-90. Strontium-90 is a fission product associated with reactor operations. It most commonly entered the environment in reactor cooling water contaminated by episodic fuel cladding failures. The post-reactor cooling water system was monitored for fuel cladding failures and the contaminated cooling water was redirected to one of the 107-D, 107-DR, or 107-H Retention Basins reserved for this purpose. During routine reactor operations, no single basin was designated to receive the contaminated cooling water, so all three basins received this waste stream. Strontium-90 was also discharged to the vadose zone at the 105-D, 105-DR, and 105-H Fuel Storage Basins and related cribs during reactor operations.

Tritium. Tritium was formed primarily by neutron activation of lithium during reactor operations. Tritium in the southern portion of 100-D is believed to be related to historical releases of tritium at 100-N. Isolated detections of elevated tritium near the 105-D and 105-DR retention basins is consistent with the expected release of tritium in the contaminated cooling water following fuel cladding failure events.

Technetium-99. Technetium-99 is also a fission product associated with reactor operations, as is strontium-90. This radionuclide followed essentially the same pathway as did strontium-90, and is found primarily in association with fuel storage basins, cribs, and trenches.

Uranium. The source of uranium isotopes is primarily from the reactor fuel. During fuel failures, uranium entered the cooling water stream. It is also associated with the spent fuel in the fuel storage basins.

4.2.2.3 Other Contaminants

Other contaminants identified in 100-D/H include nitrate, trichloroethane, total chromium, chloroform, boron, molybdenum, antimony, and strontium. While these contaminants were all likely included in some aspect of reactor operations and maintenance, the sources of nitrate, total chromium, and chloroform are best understood and are discussed in the following subsections.

Nitrate, Lead, and Arsenic. Lead and arsenic are present in the soil as a result of pre-Hanford Site agricultural activities (discussed in Section 4.1). Nitrate may be associated with former agricultural activities, discharge of nitric acid washes/rinses during reactor and support facility decontamination, and human waste discharged to septic systems.

Total Chromium. Chromium occurs naturally in the environment and is typically precipitated as a low-solubility hydroxide molecule, $Cr(OH)_3$. As such, chromium is not mobile. Elevated levels of chromium in 100-DH is associated with the discharge of sodium dichromate dihydrate ($Na_2Cr_2O_7 \cdot 2H_2O$), which contains Cr(VI). Cr(VI) ions can also be subject to chemical reduction under moderately reducing conditions, or by reaction with reducing agents such as ferrous iron. Ferrous iron is very effective at reducing Cr(VI) to Cr(III).

Chloroform. Chloroform was detected during the spatial and temporal sampling events. The chloroform is generally coincident with the Cr(VI) plumes at low concentrations of several micrograms per liter. No specific source has been identified, but it is a known degradation product of organic compounds. Chloroform most likely originated as a residue from chlorination of cooling water to control microbial growth.

4.3 Vadose Zone Contamination

This section describes the nature (type and concentration) and extent (distribution) of contamination in the vadose zone due to industrial activities related to the operation of three 100-D/H nuclear reactors. The descriptions of soil contamination represent data collected during previous limited field investigations (*Limited Field Investigation Report for the 100-DR-1 Operable Unit* [DOE/RL-93-29], *Limited Field Investigation Report for the 100-DR-2 Operable Unit* [DOE/RL-94-73], *Limited Field Investigation Report for the 100-HR-1 Operable Unit* [DOE/RL-93-51], and *Limited Field Investigation Report for the 100-HR-2 Operable Unit* [DOE/RL-94-53]), site closeout sampling, ongoing interim waste site remediation, and the current RI for constituents with concentrations that exceed background soil concentrations.

Vertical profile figures for the RI boreholes and test pits, plus applicable LFI boreholes, show the distribution of contamination in the vadose zone. Only depth discrete soil analytical results are used to illustrate the nature and extent of the preliminary COPCs in these profiles. The profiles provide visual depictions of the analytes relative to background concentrations (if available), sample depths, waste site structures, depths of remedial action, lithology, stratigraphy, and water table depths (if encountered). Within each profile, data collected below the depth of the interim action excavation defines existing conditions at the 17 interim closed-out waste sites identified for additional characterization in the 100-D/H Work Plan (DOE/RL-2008-46-ADD1). Radiological data decayed through year 2012 are presented to provide a more direct comparison to data obtained at multiple sampling events. Figures 2-1 and 2-2 show the locations of the RI boreholes, test pits, and wells.

Appendix D (Tables 1–26) provide the analytical results for residual contamination at the 17 RI characterization sites. Appendices B and D summarize the analytical results for residual contamination at the other 100-D/H closed-out, interim closed-out, and no action waste sites. The closeout verification data reflect soil concentrations used to closeout waste sites according to the interim action RODs. The data presented are from the shallow zone (0 to 4.6 m [0 to 15 ft] bgs) and/or the deep zone >4.6 m (>15 ft) bgs soil concentrations from CVP or RSVP documents. The concentrations typically represent the 95 percent upper confidence limit (UCL) based on the arithmetic mean of the data obtained from statistical sampling. The lateral extent of contamination at waste sites is generally defined by the boundary of the excavated footprint associated with soil remediation (for example, RTD).

Batch leach testing results for the determination of vadose zone K_d values to support modeling are presented in Chapter 5 and Appendix C (section C.2)and are not included in the vertical profiles. The batch leach testing results obtained from RI boreholes are from the same sample depth intervals as the vertical profiles reported in this chapter.

The following subsections of Section 4.3 present the soil analyte exclusion process (4.3.1), the waste site vadose zone profiles (4.3.2 to 4.3.19), a discussion of key waste sites currently undergoing interim action (4.3.20), the RI well soil and sediment results (4.3.21), an RPO well soil sampling summary (4.3.22), a discussion of the periodically rewetted zone (4.3.23), a discussion of soil results uncertainty (4.3.24), and a summary of vadose zone nature and extent (4.3.25).

4.3.1 Soil Analytes Excluded

The soil analytical data sets applicable to RI waste site sampling include constituents characterized as having short half-lives (for example, <3 years), common laboratory contaminants, essential nutrients, and essentially nontoxic substances. These constituents are commonly not discussed as detections and are primarily an artifact of the sampling and analysis process, not observed above background concentrations, or not a human health concern (that is, nontoxic) per *Risk Assessment Guidance for Superfund Volume I Human Health Evaluation Manual (Part A): Interim Final* (EPA/540/1-89/002), hereinafter called the risk assessment guide. Table 4-3 lists the 100-D/H soil target analytes excluded from further consideration in this document.

Table 4-3. 100-D/H Soil Analytes Excluded from Further Consideration*

	Analyte	Exclusion Rationale	Daughters (Half Life)
Radionuclides	Cerium-144	Half-life less than 3 years (284.91 days)	Pr-144m (1.2 min), Pr-144 (17.28 min), and Nd-144 (stable)
	Cesium-134	Half-life less than 3 years (2.065 years)	Ba-134 (stable)
	Cobalt-58	Half-life less than 3 years (70.86 days)	Ni-58 (stable)
	Iron-59	Half-life less than 3 years (44.495 days)	Co-59 (stable)
	Manganese-54	Half-life less than 3 years (312.03 days)	Fe-54 (stable)
	Ruthenium-103	Half-life less than 3 years (39.26 days)	Rh-103m (56.12 min), and Rh-103 (stable)

Table 4-3. 100-D/H Soil Analytes Excluded from Further Consideration*

	Analyte	Exclusion Rationale	Daughters (Half Life)
	Ruthenium-106	Half-life less than 3 years (373.59 days)	Rh-106 (29.9 sec) and Pd-106 (stable)
	Sodium-22	Half-life less than 3 years (2.6019 years)	Ne-22 (stable)
	Tin-113	Half-life less than 3 years (115.09 days)	In-113m (1.658 hours) and In-113 (stable)
	Uranium-240	Half-life less than 3 years (14.1 hours)	Np-240 (7.22 min), and Np-240 (1.03 hours)
	Analyte	**Exclusion Rationale**	**Half Life**
	Radium-224	Decay daughter of Thorium-232/ Radium-228; in equilibrium with parent	3.66 days
	Thorium-234	Decay daughter of Uranium-238; in equilibrium with parent	24.1 days
	Actinium-228	Decay daughter of Thorium-232/ Radium-228; in equilibrium with parent	6.15 hours
	Lead-212	Decay daughter of Thorium-232/ Radium-228; in equilibrium with parent	10.64 hours
	Lead-214	Decay daughter of Radium-226; in equilibrium with parent	26.8 minutes
	Thorium-228	Decay daughter of Th-232/Radium-228; in equilibrium with parent	1.91 years
	Potassium-40	Naturally occurring background radiation	1.25 billion years
	Thorium-230	Only potential source from naturally occurring background radiation (insufficient in growth time for the Hanford Site introduced uranium as decay daughter of Uranium-234)	75.38 thousand years
	Radium-226	Only potential source from naturally occurring background radiation (insufficient in growth time for the Hanford Site introduced uranium as decay daughter of Uranium-234/Thorium-230)	1.6 thousand years
	Radium-228	Decay daughter of Thorium-232. Will be in equilibrium with parent	5.75 years
	Thorium-232	Naturally occurring background radiation	14 billion years

Table 4-3. 100-D/H Soil Analytes Excluded from Further Consideration*

	Analyte	Exclusion Rationale	Daughters (Half Life)
Nonradionuclides	**Analyte**	**Exclusion Rationale**	**Half Life**
	Calcium	Essential nutrient	NA
	Chloride	Essential nutrient	NA
	Iron	Essential nutrient	NA
	Magnesium	Essential nutrient	NA
	Sodium	Essential nutrient	NA
	Potassium	Essential nutrient	NA
	Phosphate	Essential nutrient	NA
	Ammonia	No soil toxicity information available	NA
	Zirconium	No soil toxicity information available	NA

Note: Half-life information was taken from the Radiochemistry Society website (RS, 2011).

* List is from *100-D/100-H Decision Unit Target Analyte List Development for Soil* (WCH-322).

4.3.2 100-D-4 Trench Characterization

The 100-D-4 Trench received sludge and effluent in 1953 from the 107-D/DR retention basins. The interim remedial action excavation was to 2.9 m (9.5 ft) bgs, possibly less than the depth of the original bottom of the trench. Because this site only received sludge and effluent over a short time, it is not considered a high-volume liquid waste site, and may not have affected groundwater during operations. The residual contaminants detected during CVP interim close-out sampling included cesium-137, europium-152, strontium-90, uranium-238, Cr(VI), and PCBs (Aroclor-1254 and -1260).

A test pit was excavated through it during the RI (Figure 4-2) to characterize the trench. The soil samples were analyzed to evaluate the vertical extent of contamination in the vadose zone to a depth of 5.8 m (19 ft) bgs. The results of these RI samples and the CVP sample results are presented in Appendix D (Table 71). The RI test pit results for contaminants detected above background levels and for contaminants detected that do not have background values are also presented in Figure 4-3.

Of the 18 radioactive and 56 non-radioactive contaminants analyzed-for in the RI test pit samples, four were detected or were present above background (see Figure 4-3). Between the CVP and RI results, 11 contaminants were detected above background concentrations in the vadose zone beneath the trench. The detected contaminant concentrations generally decreased with depth, with exception of strontium (metal) and tin. Only Cr(VI) and nitrate in nearby groundwater monitoring wells exceeded drinking water standards (see Figures 4-66 and 4-90 for well locations). Soil concentrations detected or present above background levels are compared to soil concentrations protective of groundwater and surface water (i.e., PRGs and SSLs) in Chapter 5. The CVP and RI data are also used in Chapter 6 for the human health risk evaluation.

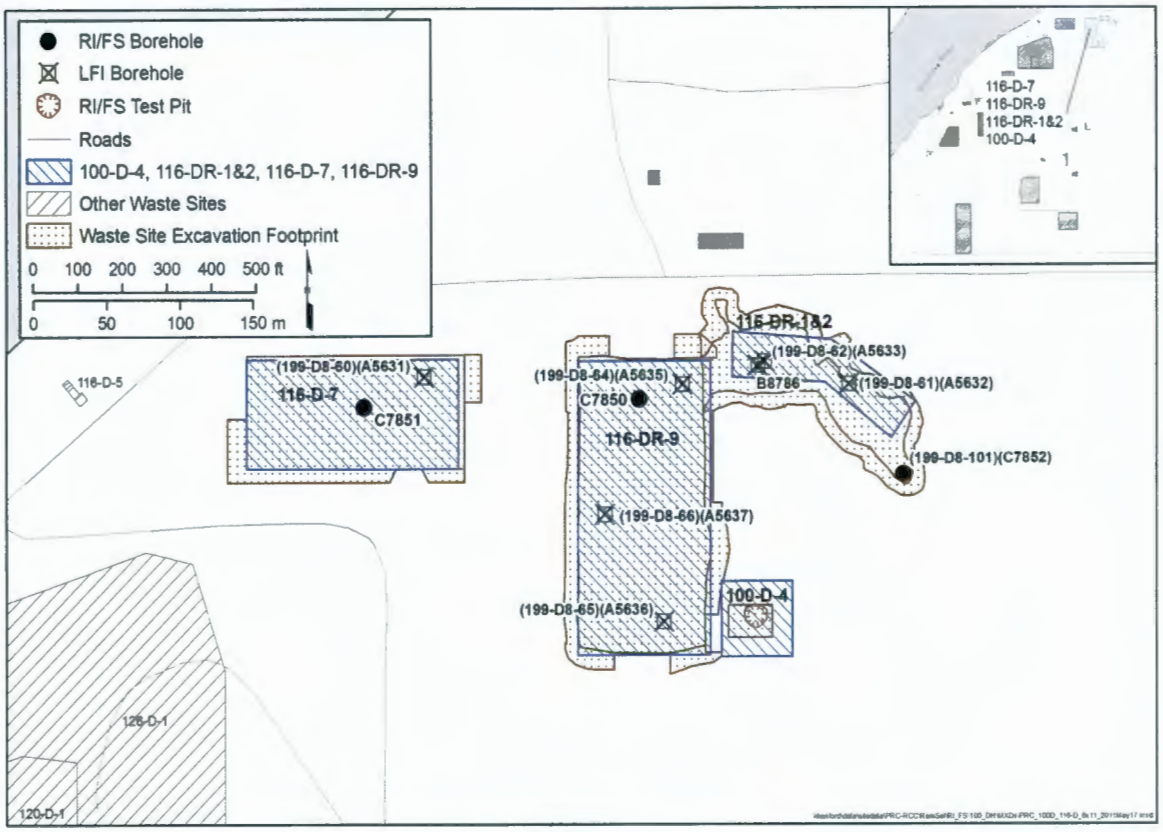

Figure 4-2. 100-D-4, 116-D-7, 116-DR-1 & 2, and 116-DR-9 Location Map

4.3.3 100-D-12 French Drain Characterization

The 100-D-12 Pumping Station/French Drain received concentrated sodium dichromate (70%) and sulfuric acid solutions during operations. The volume of liquid received and the dates of operation are not well documented for this site; however, it is suspected of being a major source of Cr(VI) for the 100-D south groundwater plume. The interim remedial action excavation was to 2.4 m (7.9 ft), potentially less than the depth of the French Drain structure. The CVP interim closeout sample analysis for 100-D-12 only included Cr(VI).

To ensure the proper placement of the borehole prescribed for this site, a test pit was first completed to about 7.6 m (25 ft) bgs to sample the soil and visually inspect the subsurface soils for sodium dichromate staining (Figure 4-4). After establishing the borehole location, the test pit was backfilled and a borehole was drilled to evaluate the vertical extent of contamination in the vadose zone to the water table at 25.9 m (85.1 ft) bgs. The results of the test pit, borehole, and CVP sample results are presented in Appendix D (Table 72). The RI test pit and borehole results for contaminants detected above background levels and for contaminants detected that do not have background values are also presented in Figure 4-5.

Of the 18 radioactive and 56 nonradioactive contaminants analyzed-for in the RI test pit and borehole samples, 10 were detected (see Figure 4-5). Between the CVP and RI results, 10 contaminants, including total chromium and Cr(VI), were detected or were present above background concentrations in the vadose zone beneath the site. Contaminant concentrations generally decreased with depth, except for strontium-90, with its highest concentration (2.2 pCi/g) near the vadose zone-groundwater interface. Only nitrate and Cr(VI) exceed drinking

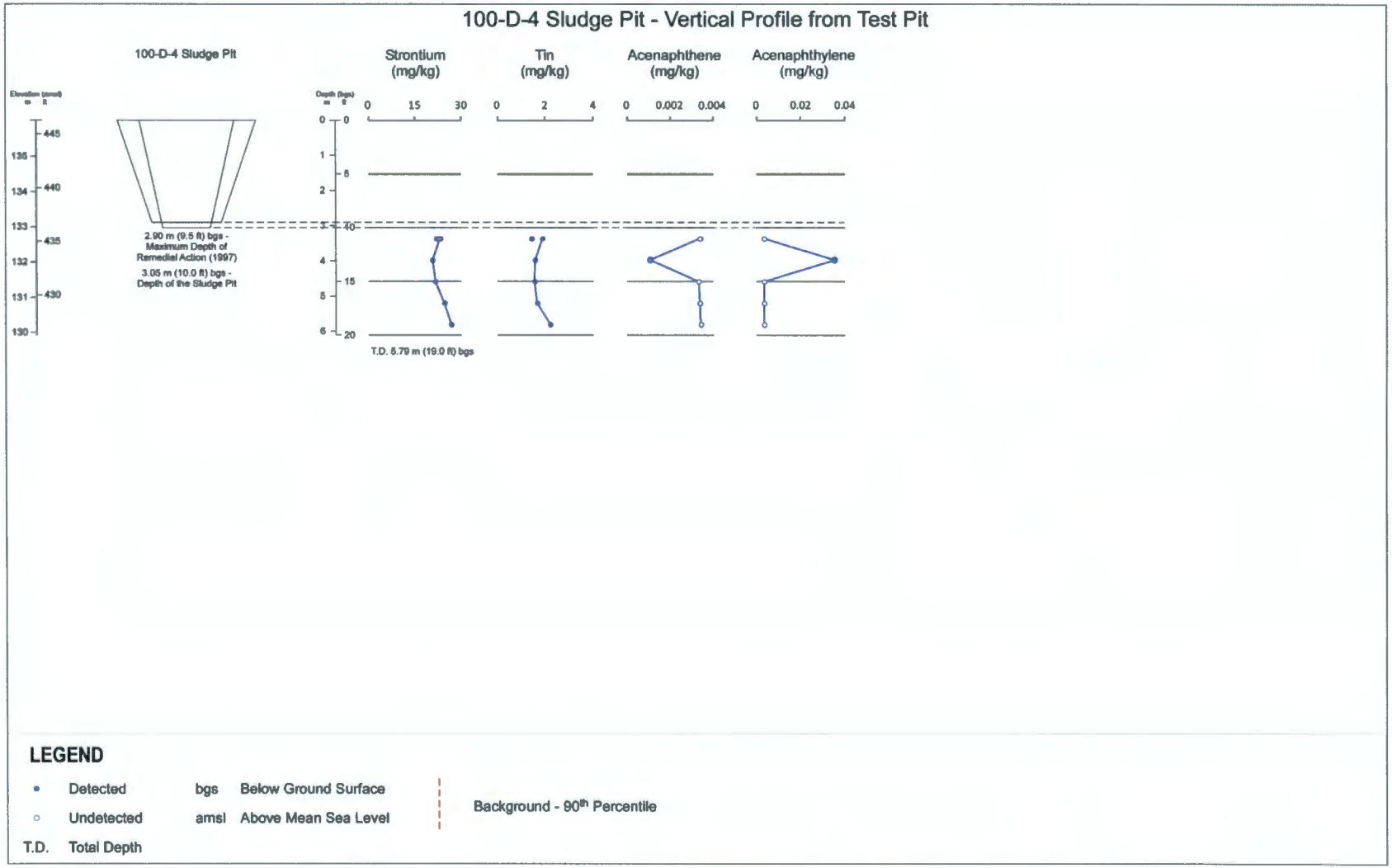

Figure 4-3. 100-D-4 Trench Vertical Profiles of Contamination in Test Pit

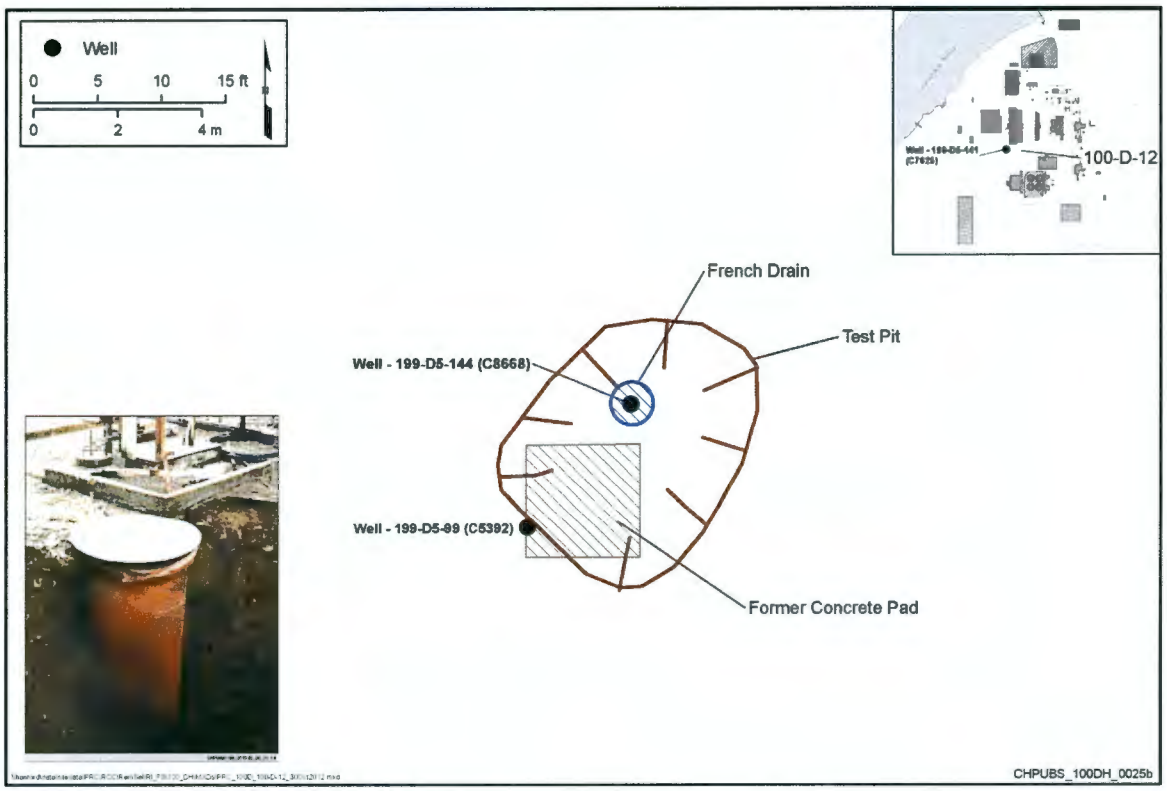

Figure 4-4. 100-D-12 Location Map

water standards in nearby groundwater monitoring wells (see Figures 4-66 and 4-90 for well locations). The total chromium and Cr(VI) in the vadose zone at 100-D-12 may be associated with the 100-D-100 waste site, which is being remediated. Soil concentrations detected or present above background are compared to soil concentrations protective of groundwater and surface water (i.e., PRG, SSL) in Chapter 5. The CVP and RI data are also used in Chapter 6 for the human health risk evaluation.

A second borehole (C7625) was drilled during the RI ~108 m (355 ft) northwest of 100-D-12 (Figure 4-4). Information from this borehole is included here because of its proximity to 100-D-12 and its location about 10 m (32 ft) west of the 100-D-72 waste site (which includes a concrete encasement that protected the service piping for air, steam, filtered water, lime slurry, and sulfuric acid, plus drained acid waste to a neutralization pit). These components were associated with the storage and flow of sulfuric acid to the 183-D Head House. This borehole was also drilled and samples were collected through the vadose zone to the water table (26 m [85.2 ft] bgs). Appendix D (Table 73) summarizes the results of the borehole samples and Figure 4-6 presents vertical profiles for contaminants detected and present above background levels. Of the 18 radioactive and 56 nonradioactive contaminants analyzed-for in the borehole samples, eight contaminants were identified in the vadose zone above background concentrations from a depth of 7.6 m (25 ft) bgs to the water table. The maximum tritium concentration was 18.6 pCi/g (at 70 ft bgs), while the maximum total chromium, molybdenum, nickel, and strontium (metal) concentrations are detected about 18.9 m (62 ft) bgs. Only nitrate and Cr (VI) exceed drinking water standards in nearby groundwater monitoring wells (see Figures 4-66 and 4-90 for well locations). Soil concentrations detected or present above background are compared to soil concentrations protective of groundwater and surface water (i.e., PRG, SSL) in Chapter 5.

4.3.4 100-D-56:1 Pipeline Characterization

The 100-D-56:1 Pipeline is an abandoned 7.6 cm (3 in.) diameter underground chemical supply line that was used from 1944 to 1950. The pipeline transported sodium silicate and sodium dichromate liquids between the 108-D, 185-D, and 190-D buildings. During remedial action 1,500 L (400 gal) of sodium dichromate was removed from the pipeline and a hole was noted at a 90-degree bend in the pipeline. The pipeline interim remedial action excavation was to 2 m (6.6 ft) bgs.

An RI borehole was needed to better define the vertical extent of contamination at the leak location (Figure 4-6). The borehole was drilled and samples were collected through the vadose zone to the water table (25.1 m [82.5 ft] bgs). The results of the borehole and interim closeout CVP samples are presented in Appendix D (Table 74). The RI borehole results for contaminants detected above background levels and for contaminants detected that do not have background values are also presented in Figure 4-7.

Of the 18 radioactive and 56 nonradioactive contaminants analyzed-for in the RI test pit and borehole samples, seven were detected (Figure 4-7). Between the CVP and RI results, 16 contaminants were detected or were present above background concentrations in the vadose zone beneath the site. While Cr(VI) was not detected in the borehole results, it was measured to a depth of 2 m (6.6 ft) in the CVP results. The greatest concentrations of strontium-90, total chromium, lithium, molybdenum, and strontium (metal) were detected 50 to 70 ft bgs in the vadose zone. Only nitrate and Cr(VI) exceeded drinking water standards in nearby groundwater monitoring wells (Figures 4-66 and 4-90 for well locations). Soil concentrations detected or present above background are compared to soil concentrations protective of groundwater and surface water (i.e., PRG, SSL) in Chapter 5. The CVP and RI data are also used in Chapter 6 for the human health risk evaluation. During the RI, a second borehole (C7866) was drilled near the origin of the 100-D-56:1 pipeline, within the 100-D-101 waste site boundary, which includes the soil beneath the 108-D Chemical Pump House (Figure 4-7). This borehole was also drilled and samples were collected through the vadose zone to the water table (25.1 m [82.5 ft] bgs). The results of the borehole samples are summarized in Appendix D (Table 75) and Figure 4-7 presents vertical profiles for contaminants detected and present above background levels. Of the 18 radioactive and 56 nonradioactive contaminants analyzed-for in the borehole samples, seven contaminants were detected or were present above background concentrations in the vadose zone from 5 m (16 ft) bgs to the water table 25.8 m (84.7 ft) bgs. The maximum total chromium, molybdenum, and nickel concentrations are detected about 24.6 m (81 ft) bgs. Other contaminant trends varied, but generally decreased with depth and are typically low concentration single detections (i.e., mercury) above background. Only nitrate and Cr (VI) exceed drinking water standards in nearby groundwater monitoring wells (see Figures 4-66 and 4-90 for well locations). Soil concentrations detected or present above background are compared to soil concentrations protective of groundwater and surface water (i.e., PRG, SSL) in Chapter 5.

4.3.5 116-D-1A Trench Characterization

The 116-D-1A Trench received 200,000 L (52,834 gal) of FSB effluent and sludge from 1947 to 1952. This material contained 1,000 kg (2,200 lb) sodium dichromate and a radiological inventory of 4.7 curies (Figure 4-8). The site is categorized as a low-volume, high-concentration liquid waste site that was not expected to affect groundwater during operations. The interim action excavation to 4.6 m (15 ft) bgs was potentially less than the depth of the original trench, so a residual source of contamination may remain in the vadose zone that could affect groundwater quality.

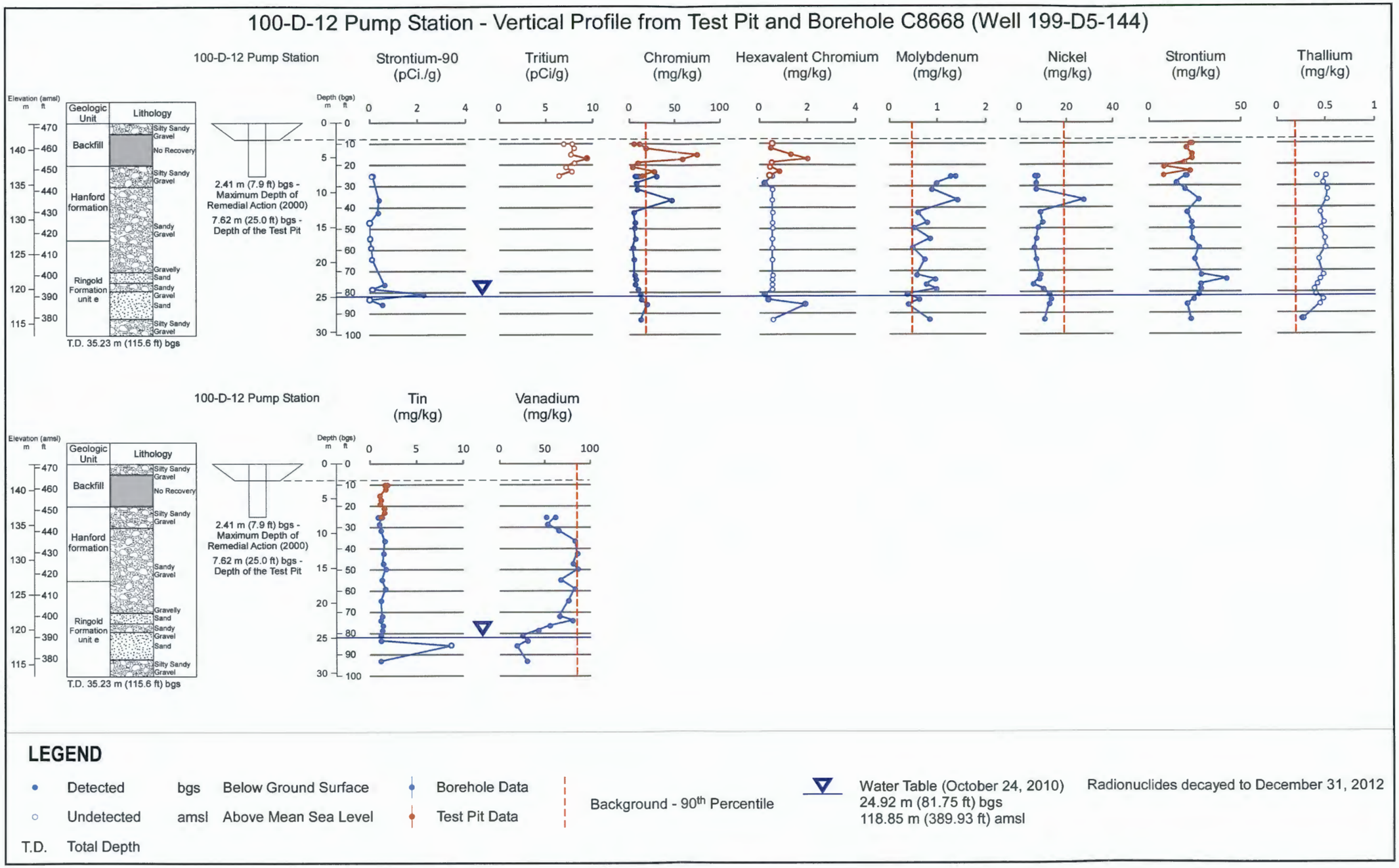

Figure 4-5. 100-D-12 Vertical Profile of Contamination in Borehole C8668 (Well 199-D5-144)

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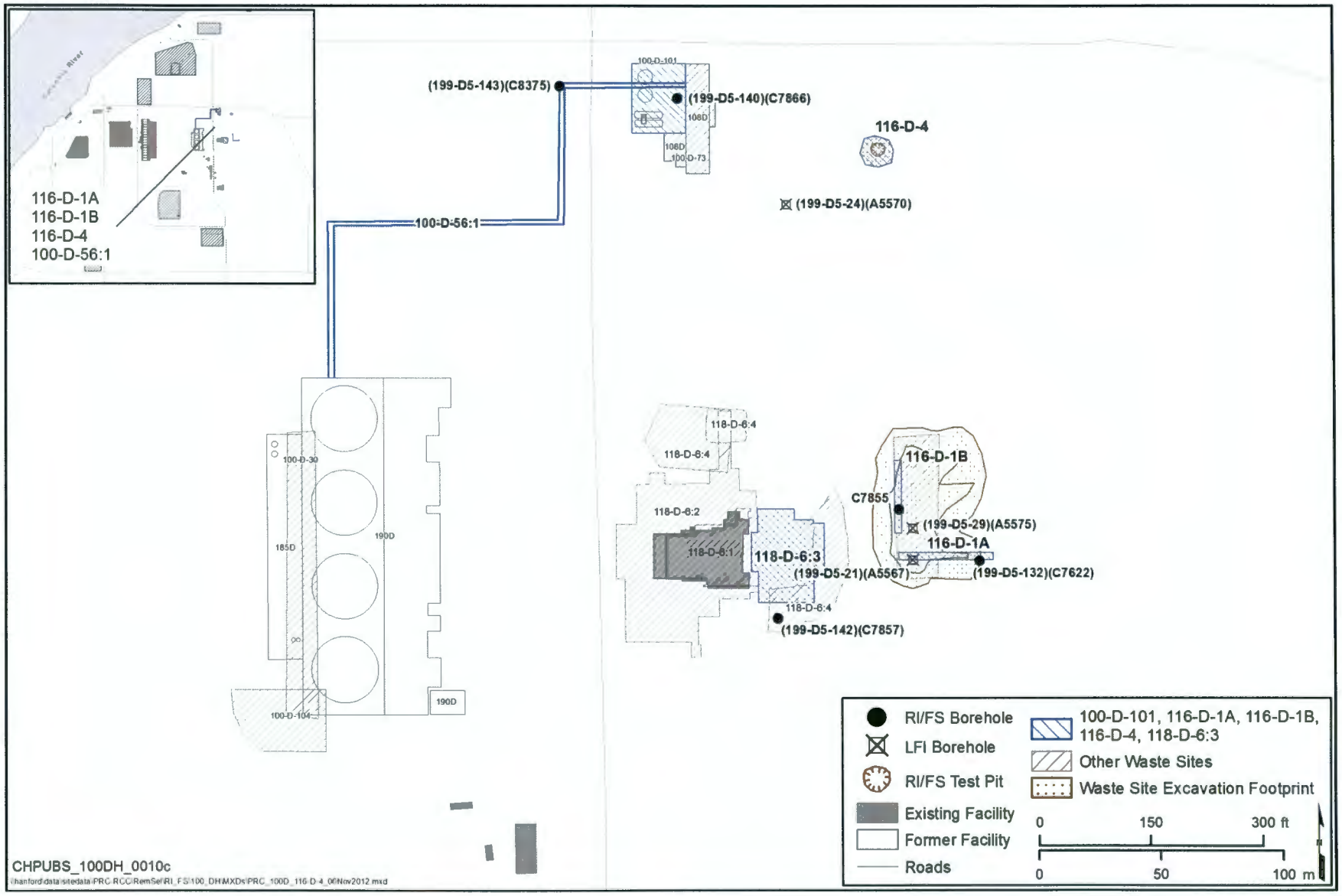

Figure 4-6. 100-D-56:1, 116-D-1A, 116-D-1B, 116-D-4, and 118-D-6:3 Location Map

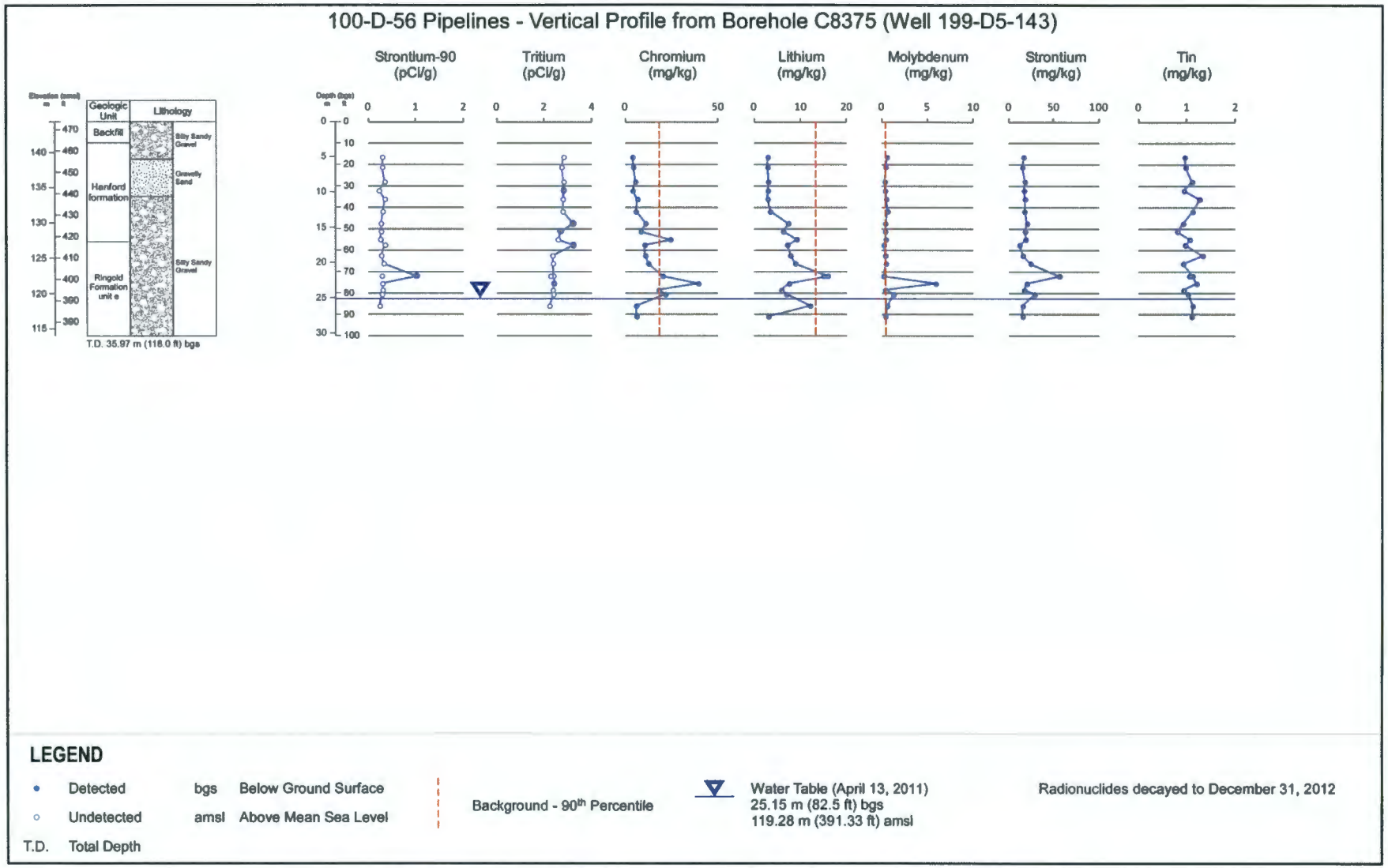

Figure 4-7. 100-D-56:1 Vertical Profile of Contamination in Borehole C8375 (Well 199-D5-143)

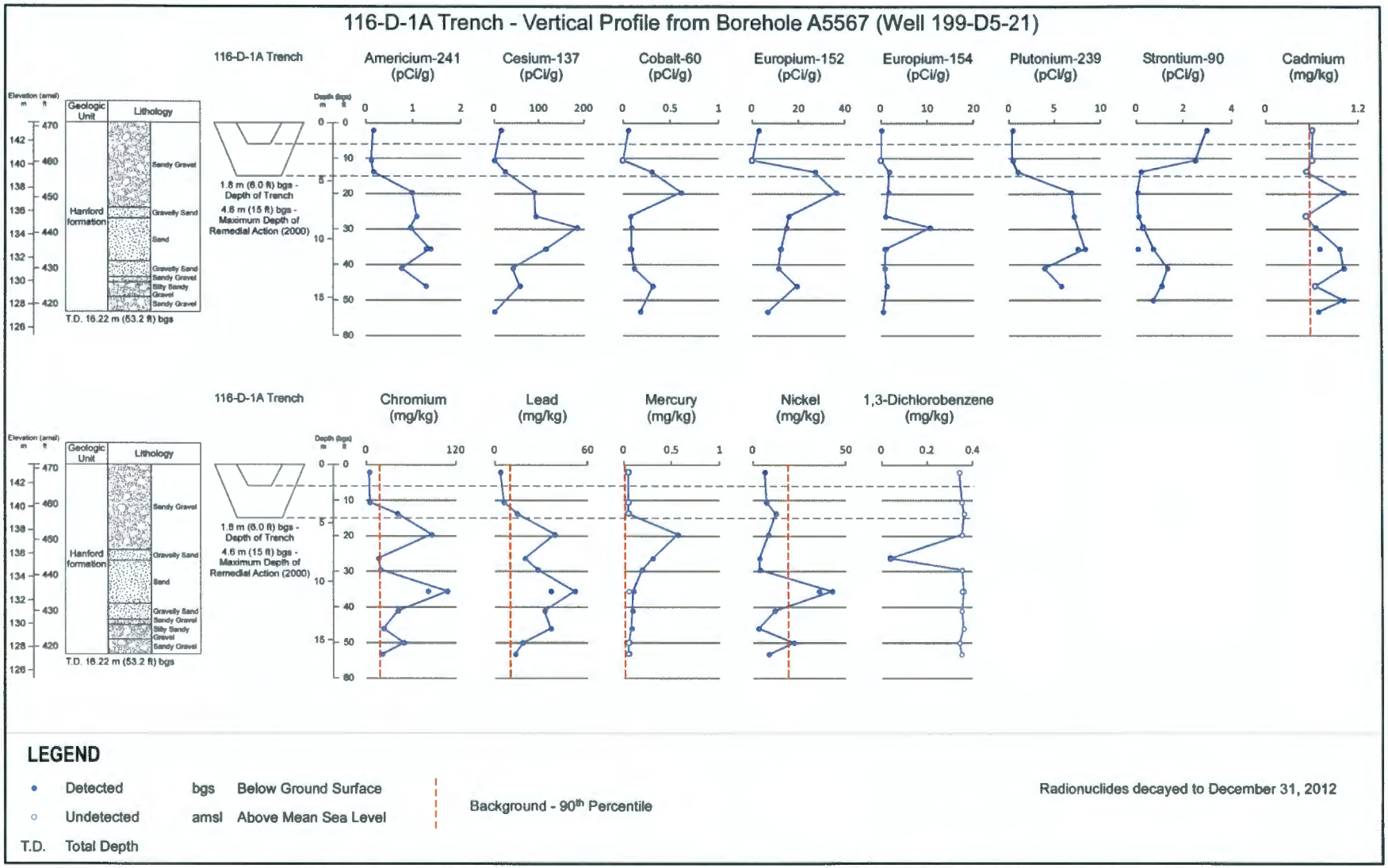

Figure 4-8. 116-D-1A Vertical Profiles of Contamination in LFI Borehole A5567 (Well 199-D5-21)

An RI borehole was drilled (Figure 4-6) and soil samples were collected and analyzed to evaluate the extent of contamination through the vadose zone to the water table (26 m [85.5 ft] bgs). The borehole data plus interim closeout CVP and LFI data for 116-D-1A are summarized in Appendix D (Table 76). Vertical profiles of RI and LFI borehole data for contaminants detected and present above background levels are also presented in Figures 4-8, 4-9, and 4-10.

Between the CVP, LFI, and RI sample results for 116-D-1A, 26 contaminants were detected or were present above background levels. The profiles show that contaminant trends vary at this site; however, higher concentrations are generally in the upper half of the vadose zone. An exception to the typical contaminant distribution at this site involves the arsenic concentration (167 mg/kg) at the groundwater-vadose zone interface while the other arsenic concentrations were below background values. This arsenic concentration is considered an outlier that is not representative of arsenic concentrations at 116-D-1A because four batch leach samples collected from the same interval had concentrations that were < 2 mg/kg (see Appendix C for the batch leach result summary). Only nitrate, Cr(VI), and strontium-90 exceeded drinking water standards in nearby groundwater monitoring wells (see Figures 4-66, 4-90, and 4-95 for well locations). Soil concentrations detected or present above background are compared to soil concentrations protective of groundwater and surface water (i.e., PRG, SSL) in Chapter 5. The CVP, LFI, and RI data are also used in Chapter 6 for the human health risk evaluation.

4.3.6 116-D-1B Trench Characterization

The 116-D-1B Trench received 8,000,000 L (2,113,376 gal) of sludge and effluent from the FSB from 1953-1967. The effluent and sludge had a 2.6-curie radiological inventory. The trench was a high-volume liquid waste site and the contamination affected the vadose zone groundwater during operations. The interim action excavation to 4.6 m (15 ft) bgs was potentially less than the depth of the original trench, so a residual source of contamination may remain in the vadose zone that could affect groundwater quality.

An RI borehole was drilled (Figure 4-6) and soil samples were collected and analyzed to evaluate the extent of contamination through the vadose zone to the water table (26.5 m [86.9 ft]) bgs. The RI borehole data plus interim closeout CVP and LFI data for 116-D-1B are summarized in Appendix D (Table 77). Vertical profiles of the RI and LFI borehole data for contaminants detected and present above background levels are also presented in Figures 4-11, 4-12, 4-13, and 4-14.

The combined RI, CVP, and LFI sample results indicate that 45 contaminants were detected or were present above background concentrations beneath 116-D-1B. The profiles show that contaminant trends vary, with higher concentrations generally present in the upper half of the vadose zone. However, total chromium, barium, molybdenum, and delta-BHC have elevated concentrations near the water table. Only nitrate and Cr(VI) exceed drinking water standards in nearby groundwater monitoring wells (see Figures 4-66 and 4-90 for well locations). Soil concentrations detected or present above background are compared to soil concentrations protective of groundwater and surface water (i.e., PRG, SSL) in Chapter 5. The CVP, LFI, and RI data are also used in Chapter 6 for the human health risk evaluation.

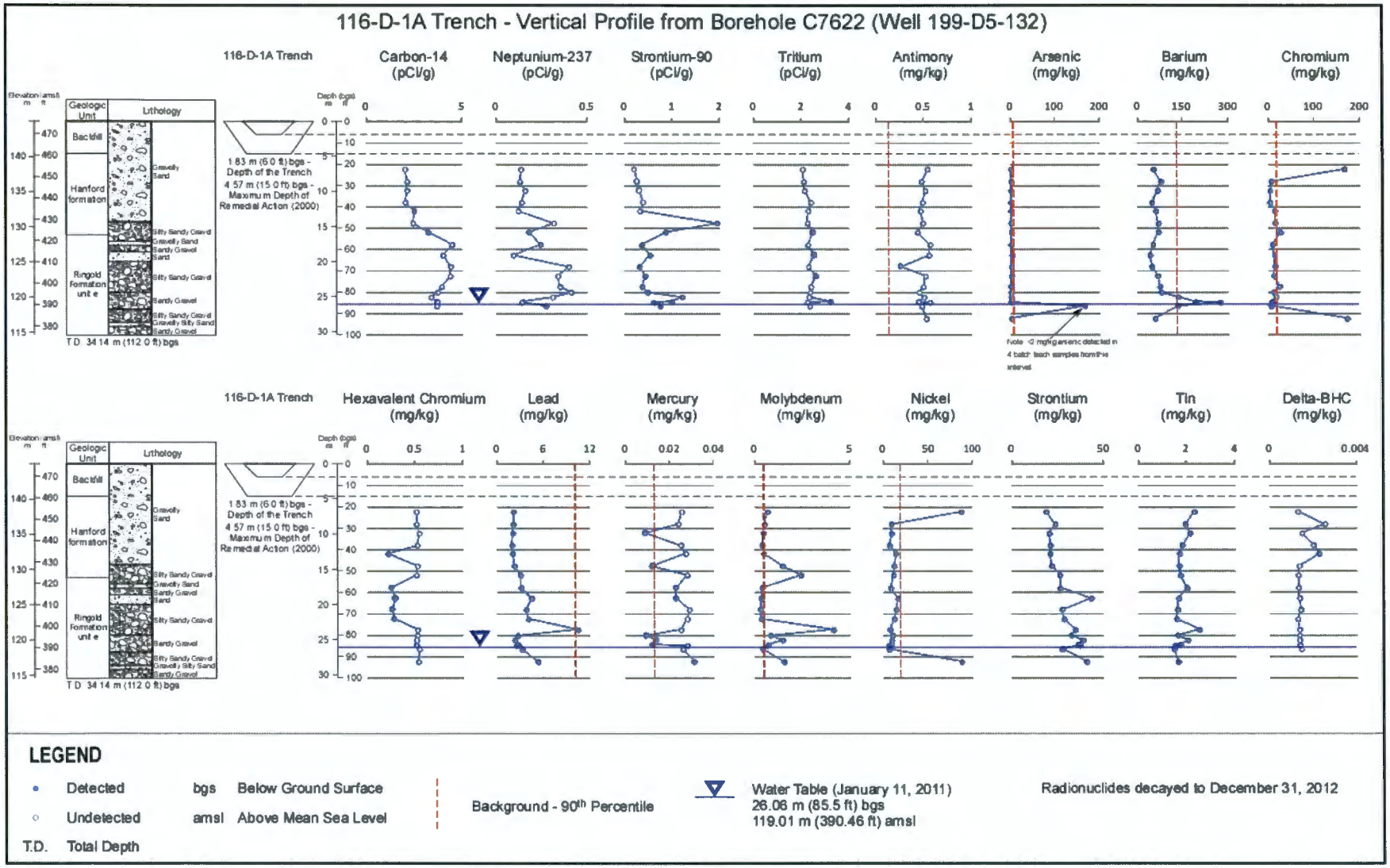

Figure 4-9. 116-D-1A Vertical Profiles of Contamination in RI Borehole C7622 (Well 199-D-5-132)

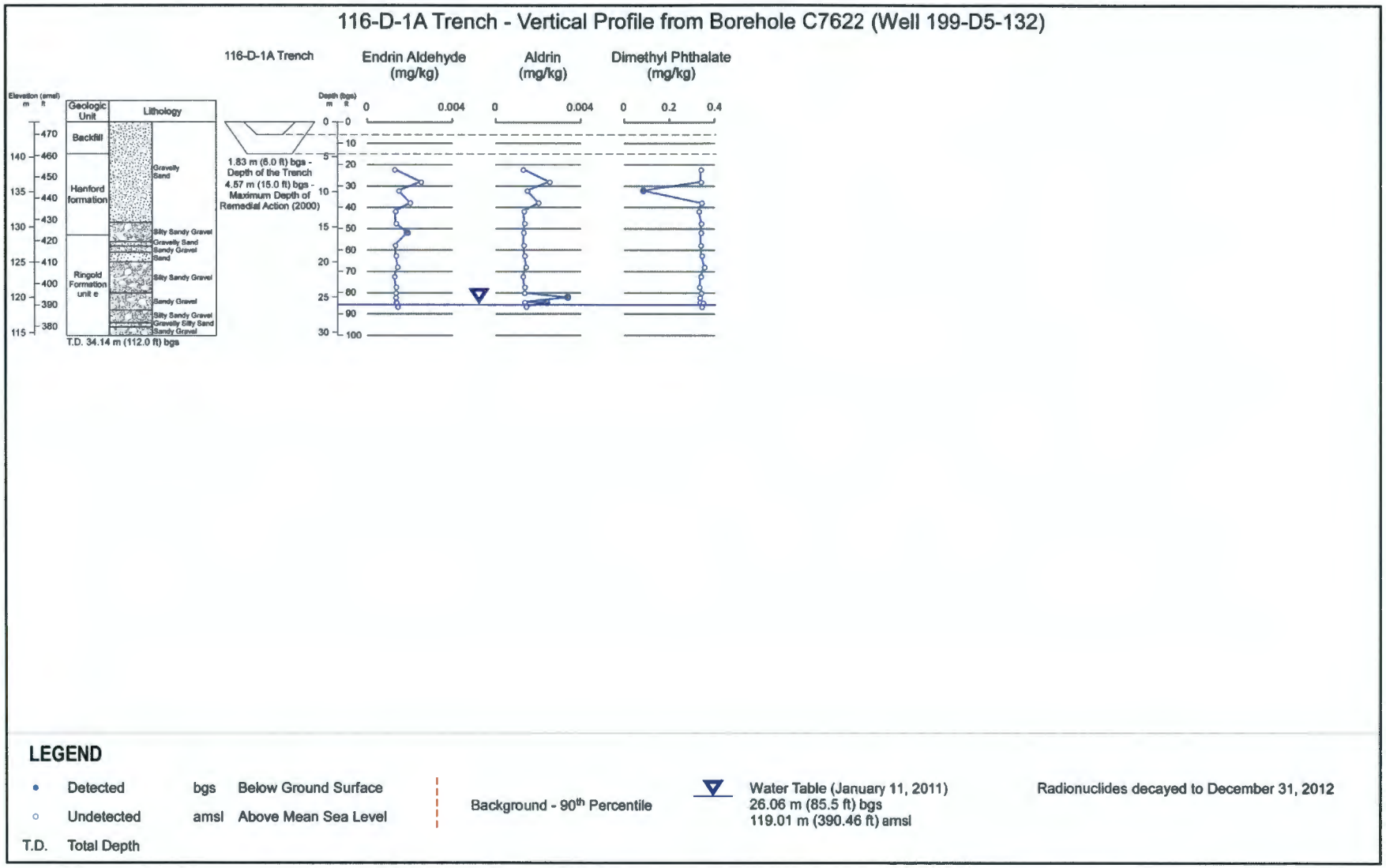

Figure 4-10. 116-D-1A Vertical Profiles of Contamination in Remedial Investigation Borehole C7622 (Well 199-D5-132)

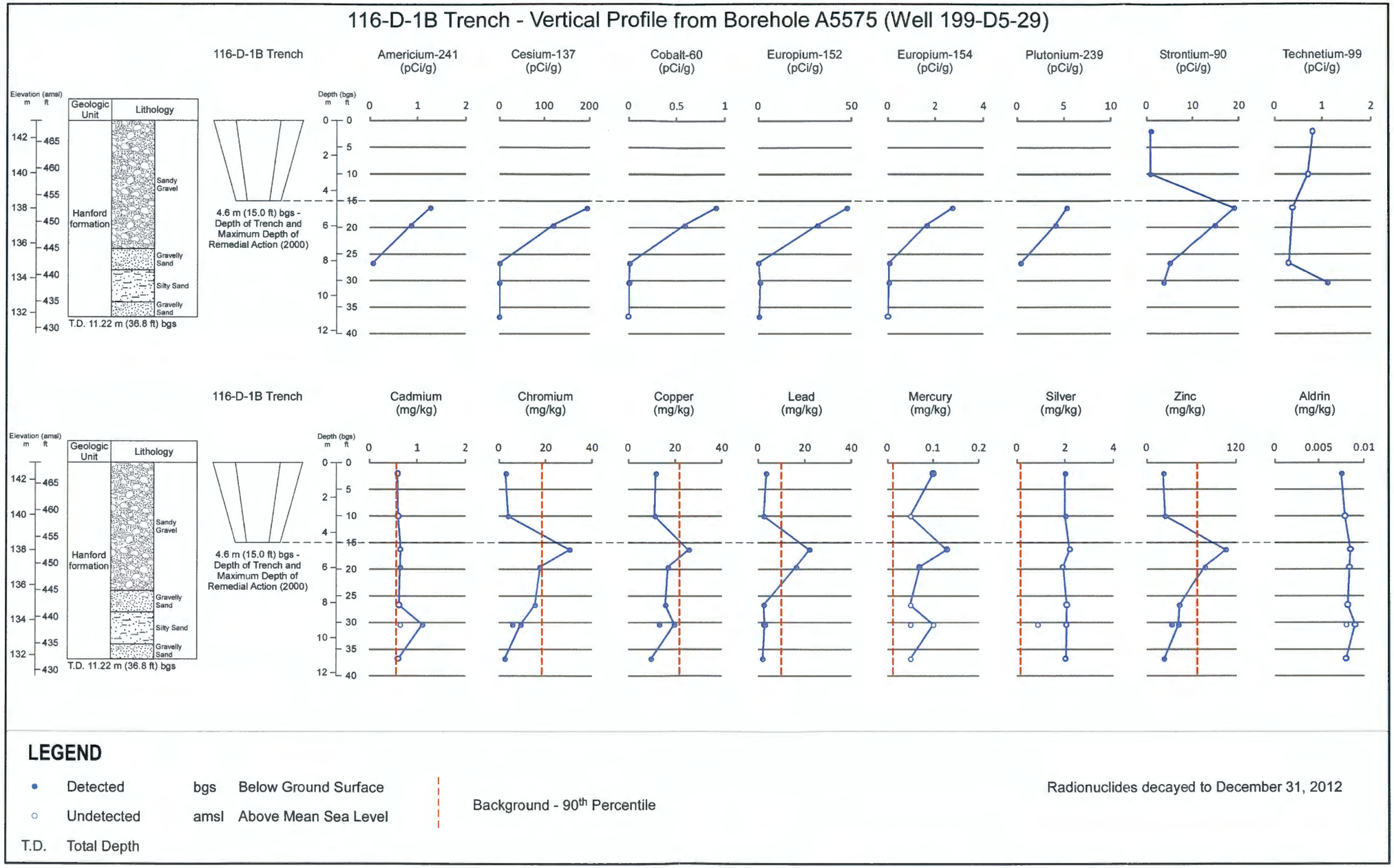

Figure 4-11. 116-D-1B Vertical Profiles of Contamination in LFI Borehole A5575 (Well 199-D5-29)

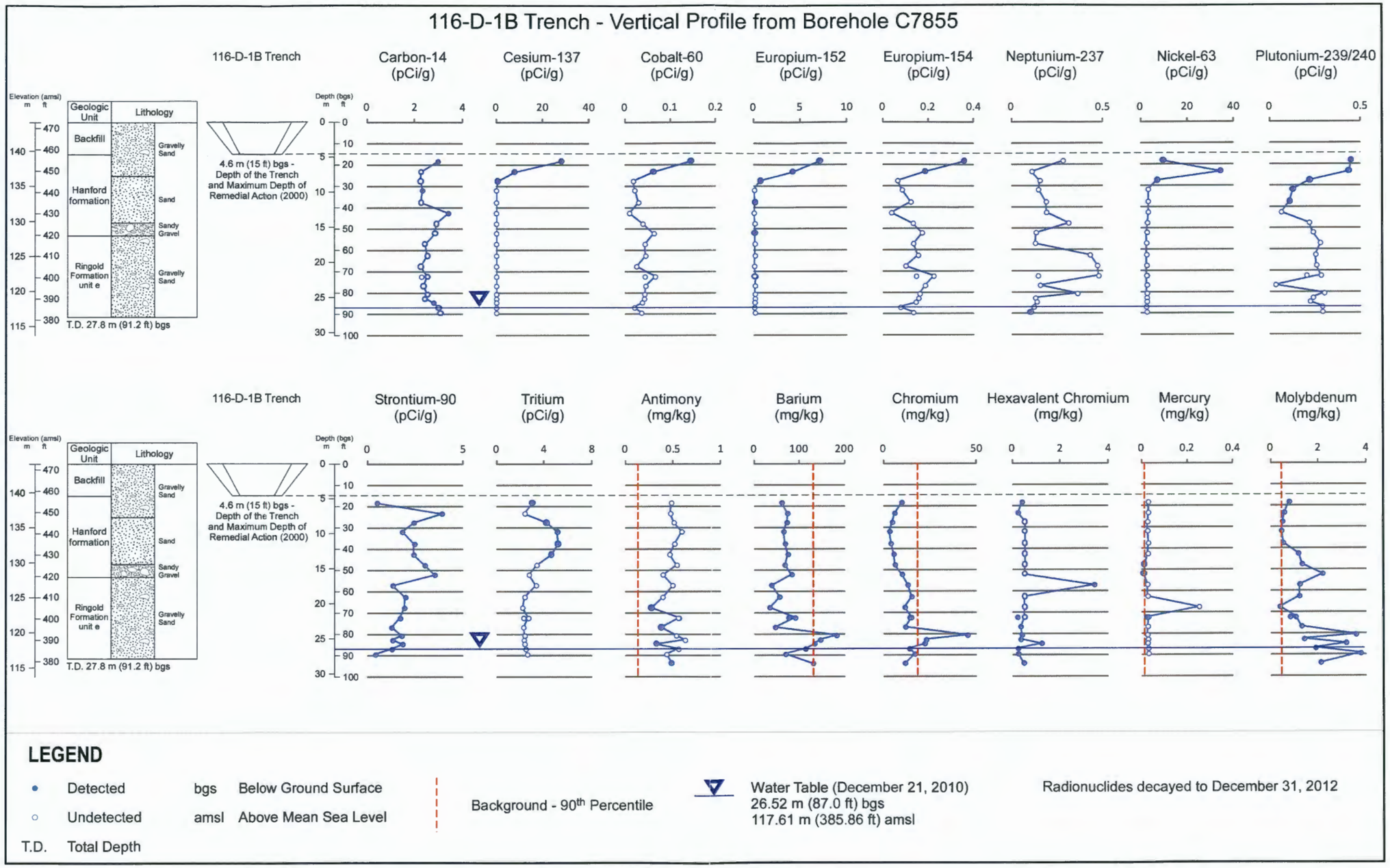

Figure 4-12. 116-D-1B Vertical Profiles of Contamination in Remedial Investigation Borehole (C7855)

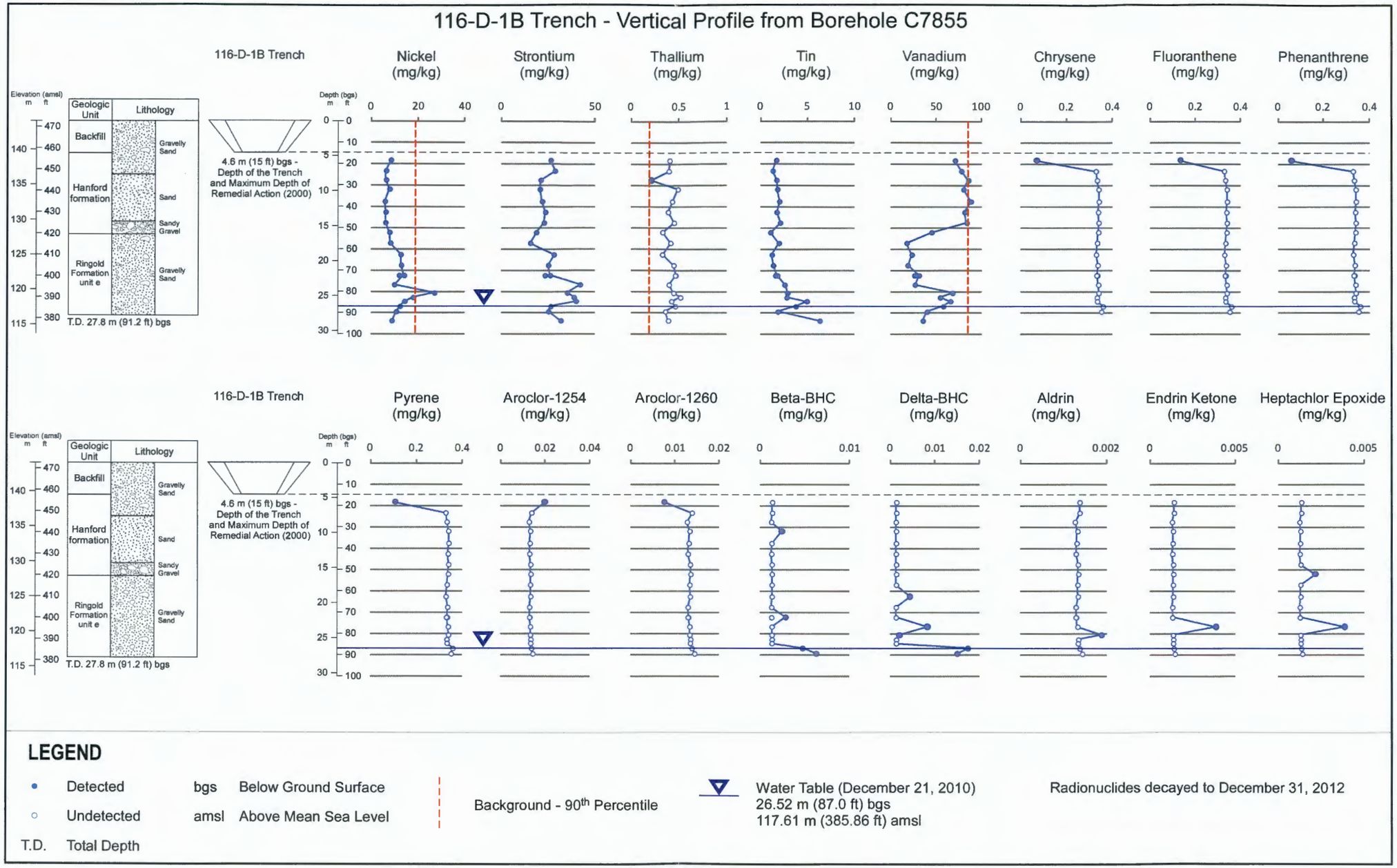

Figure 4-13. 116-D-1B Vertical Profiles of Contamination in Remedial Investigation Borehole (C7855)

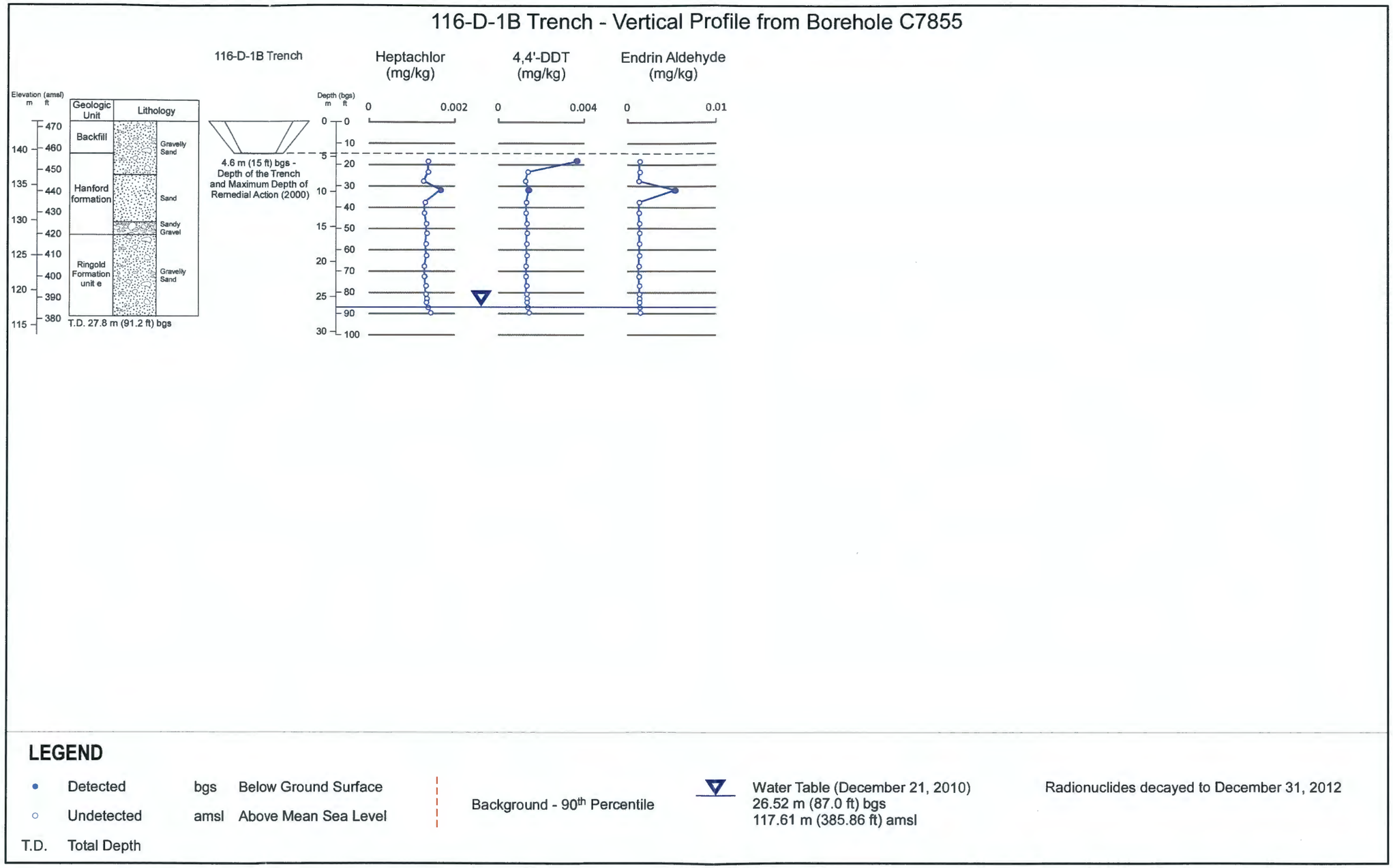

Figure 4-14. 116-D-1B Vertical Profiles of Contamination in Remedial Investigation Borehole (C7855)

4.3.7 116-D-4 Crib Characterization

The 116-D-4 Crib received 30,000 L (7,925 gal) of decontamination fluids, solvents and low-level fission products from the 108-D Building from 1956-1967. The interim remedial excavation to 2.8 m (9 ft) bgs for this low-volume waste site was potentially less than the depth of the crib structure. However, only Cr(VI) was detected at the excavation depth in the interim closeout CVP sampling effort.

An RI test pit was excavated (Figure 4-6) and soil samples were collected and analyzed to evaluate the extent of contamination to a depth of 5.8 m (19 ft) bgs. An earlier LFI borehole was drilled to 7 m (23 ft) to investigate this site. The test pit, CVP, and LFI data for 116-D-4 are summarized in Appendix D (Table 78). Vertical profiles of the test pit and LFI borehole data for contaminants detected and present above background levels are also presented in Figures 4-15 and 4-16.

Between the CVP, LFI, and RI sample results for 116-D-4, 11 contaminants were detected or were present in the vadose zone above background levels. Contaminant concentrations generally decreased with depth beneath the site and only nitrate and Cr(VI) exceeded drinking water standards in nearby groundwater monitoring wells (see Figures 4-66 and 4-90 for well locations). Soil concentrations detected or present above background are compared to soil concentrations protective of groundwater and surface water (i.e., PRG, SSL) in Chapter 5. The CVP, LFI, and RI data are also used in Chapter 6 for the human health risk evaluation.

4.3.8 116-D-7 Retention Basin Characterization

The 116-D-7 Retention Basin received 105-D Reactor cooling water from 1944-1967. After radioactive decay and thermal cooling, the effluent was discharged to the Columbia River. Due to cooling water leaks and spills, the radionuclide inventory near the basin ranged from 5 to 400 Ci during operations. The basin contamination extended beyond the depth of the interim remedial excavation [7.4 m (24.3 ft)] and reached the water table during operations.

An RI borehole was drilled (Figure 4-2) and soil samples were collected and analyzed to evaluate contamination in the vadose zone to the depth of the water table (19 m [62.3 ft]) bgs. The 116-D-7 RI borehole, 1992 LFI borehole (which extended to 11.2 m [36.6 ft] bgs), and the interim closeout CVP data are summarized in Appendix D (Tables 79, 80, and 81). The RI and LFI borehole data for contaminants detected or present above background levels are presented in Figures 4-17, 4-18, and 4-19.

Between the CVP, LFI, and RI sample results for 116-D-7, 21 contaminants were detected or were present in the vadose zone above background levels. The profiles show that higher contaminant concentrations are typically in the upper half of the vadose zone and contaminant concentrations generally decreased with depth, except for tritium and barium. Only Cr(VI) and nitrate exceeded drinking water standards in nearby groundwater monitoring wells (Figures 4-66 and 4-90 for well locations). Soil concentrations detected or present above background are compared to soil concentrations protective of groundwater and surface water (i.e., PRG, SSL) in Chapter 5. The CVP, LFI, and RI data are also used in Chapter 6 for the human health risk evaluation. Concentrations above background are compared to soil concentrations protective of groundwater and surface water (i.e., PRG, SSL) in Chapter 5. The data are also used in Chapter 6 to perform an evaluation of human health risk.

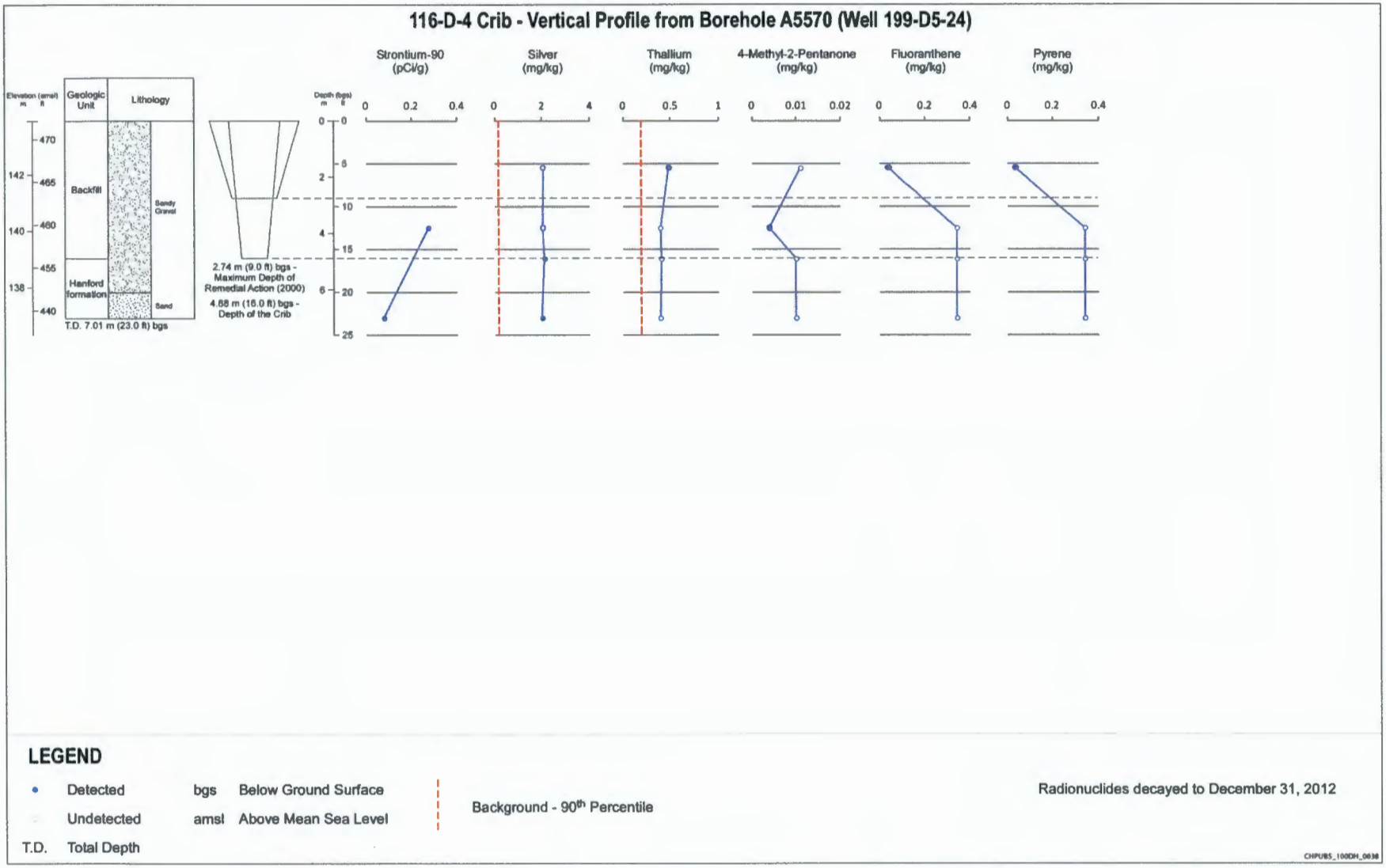

Figure 4-15. 116-D-4 Vertical Profiles of Contamination in LFI Borehole A5570 (Well 199-D5-24)

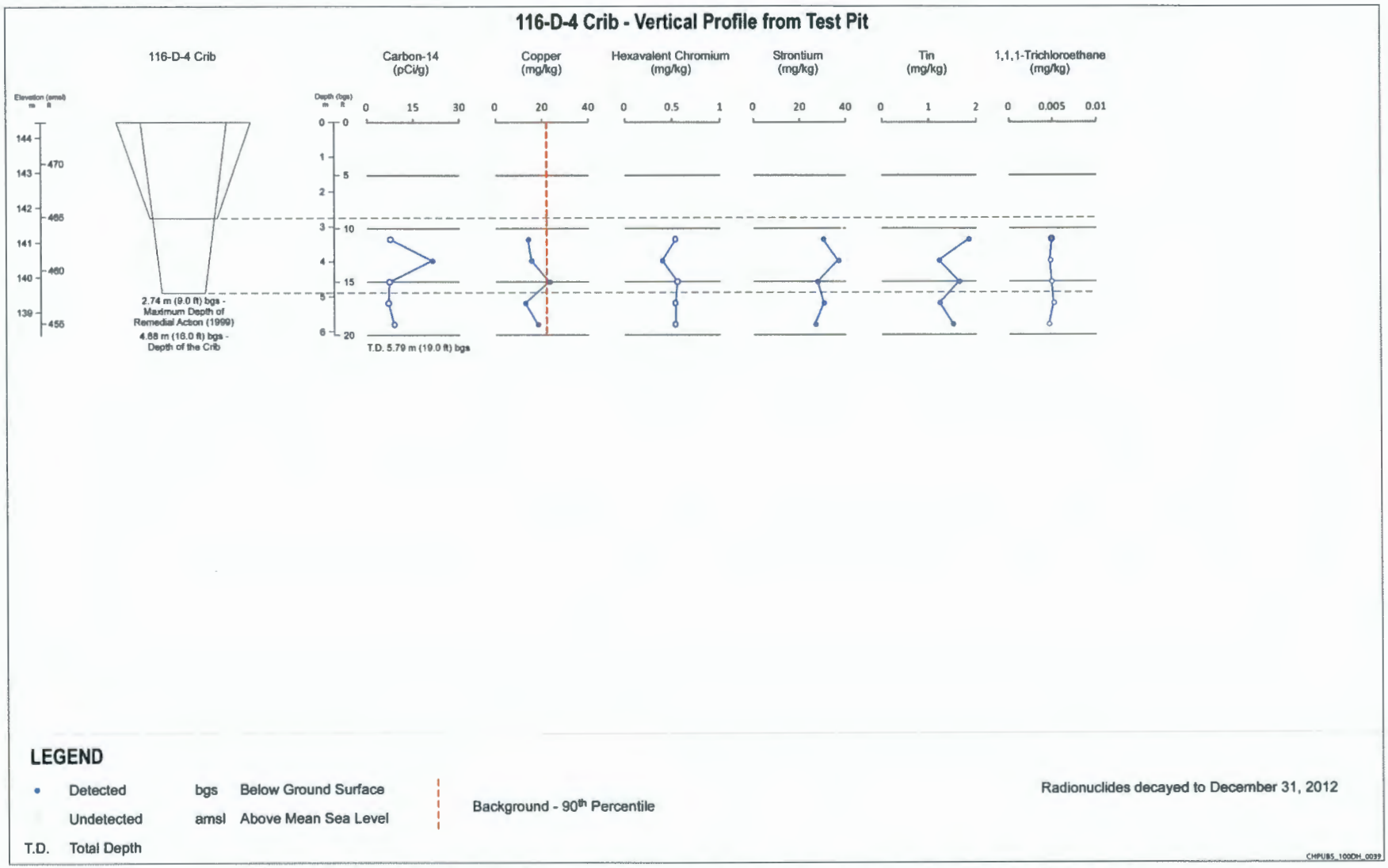

Figure 4-16. 116-D-4 Vertical Profiles of Contamination in Remedial Investigation Test Pit 116-D-4

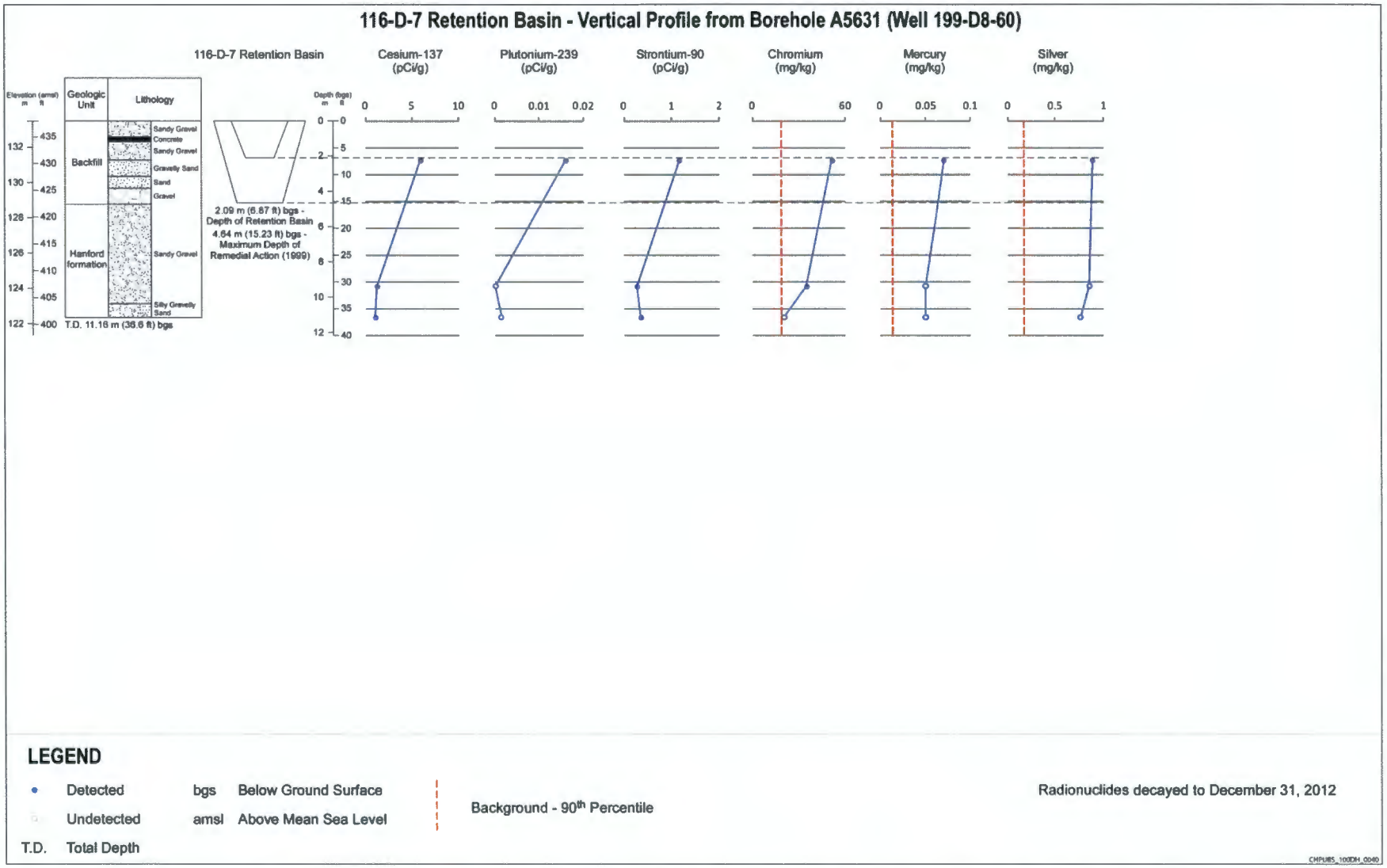

Figure 4-17. 116-D-7 Vertical Profiles of Contamination in LFI Borehole A5631 (Well 199-D8-60)

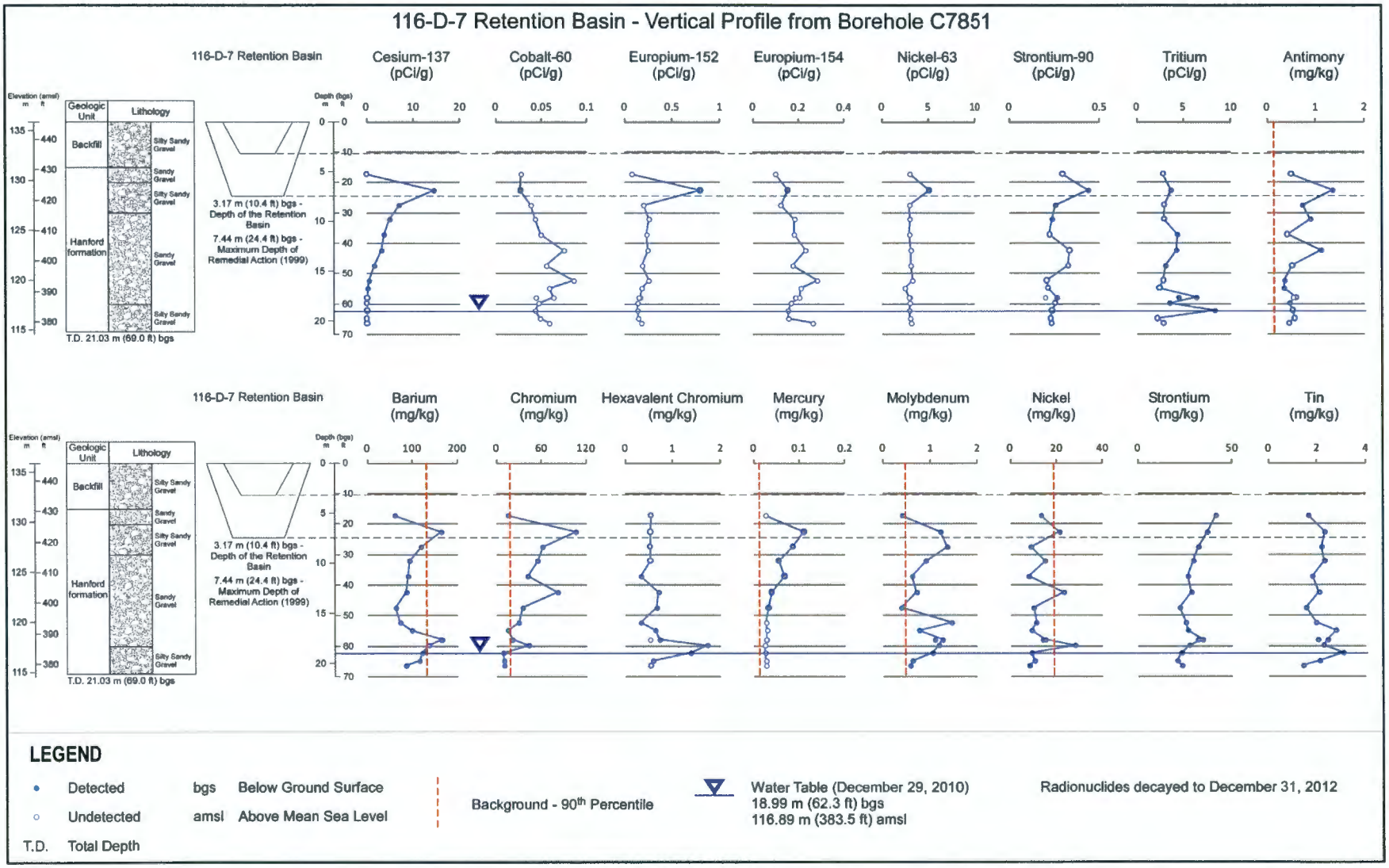

Figure 4-18. 116-D-7 Vertical Profiles of Contamination in Remedial Investigation Borehole C7851

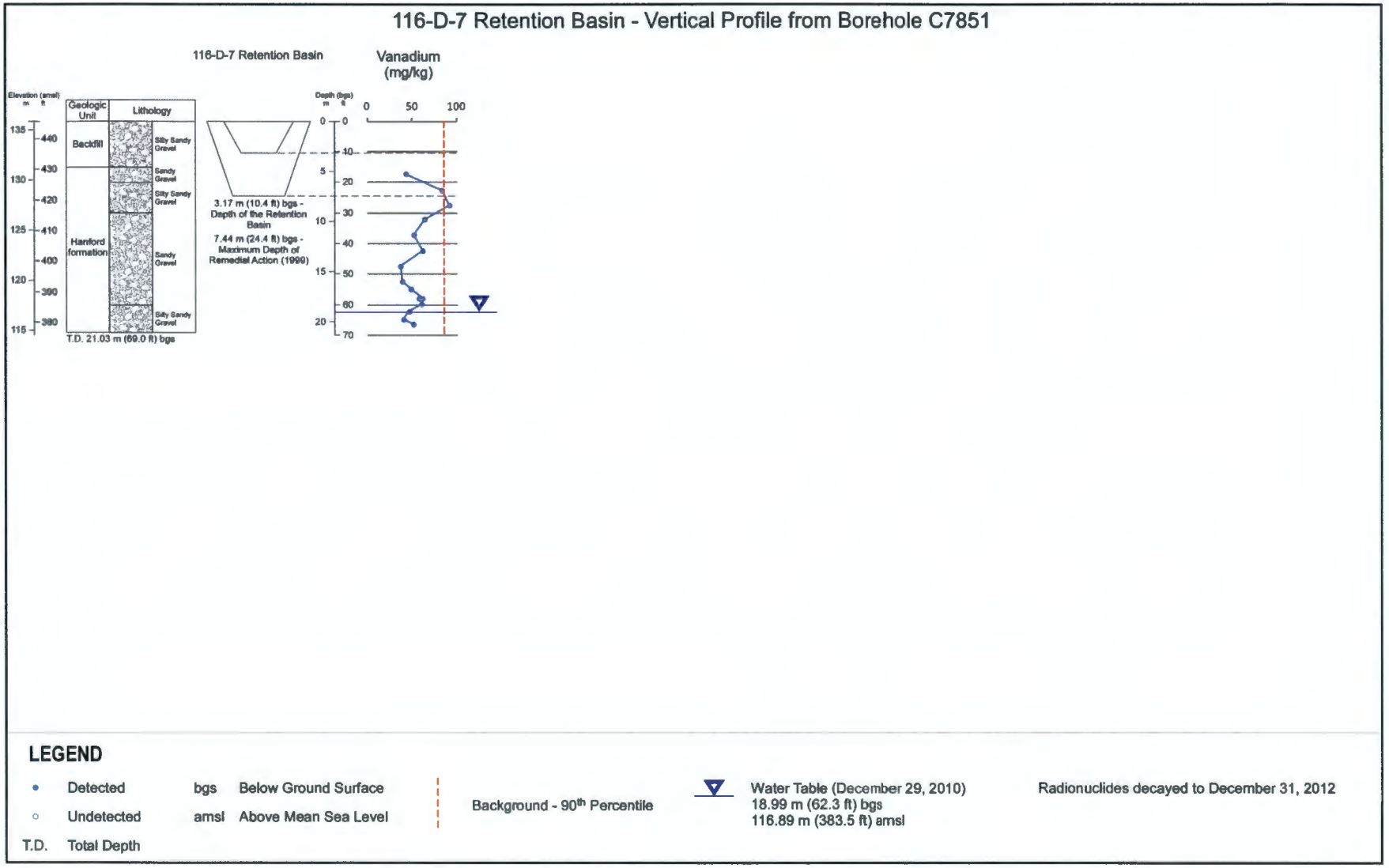

Figure 4-19. 116-D-7 Vertical Profiles of Contamination in Remedial Investigation Borehole C7851

4.3.9 116-DR-1&2 Trench Characterization

The 116-DR-1&2 Trench received 40 million L (10.5 million gal) of effluent, 40 kg (88 lb) of sodium dichromate, and a 3.1-curie radiological inventory from 1950-1967. The trench contamination extended beyond the depth of the interim remedial excavation [5.0 m (16.4 ft)] and reached the water table during operations.

An RI borehole was drilled (Figure 4-2) and soil samples were collected and analyzed to evaluate contamination through the vadose zone to the depth of the water table (19.6 m [64.2 ft]) bgs. Previous investigations for this site included three LFI boreholes and interim closeout CVP samples. The 116-D-7 RI borehole, previous LFI boreholes, and the CVP data are summarized in Appendix D (Tables 82 and 83). Vertical profiles of the RI and LFI borehole results for contaminants detected or present above background levels are presented in Figures 4-20, 4-21, 4-22, 4-23, and 4-24.

An evaluation of the CVP, LFI, and RI sample results for 116-DR-1&2 indicate that 26 contaminants were detected or were present in the vadose zone above background levels. Contaminant concentrations generally decreased with depth. Only Cr(VI) and nitrate are in nearby groundwater wells in excess of drinking water standards (see Figures 4-66 and 4-90 for well locations). Soil concentrations detected or present above background are compared to soil concentrations protective of groundwater and surface water (i.e., PRG, SSL) in Chapter 5. The CVP, LFI, and RI data are also used in Chapter 6 for the human health risk evaluation.

Twenty-six contaminants were identified in the vadose zone beneath the site. Contaminant concentrations generally decrease with depth. Only Cr(VI) and nitrate are in nearby groundwater wells in excess of drinking water standards (see Figures 4-66 and 4-90). Soil concentrations above background are compared to soil concentrations protective of groundwater and surface water (i.e., PRG, SSL) in Chapter 5. The data are also used in Chapter 6 to perform an evaluation of human health risk.

4.3.10 116-DR-9 Retention Basin Characterization

The 116-DR-9 Retention Basin received 105-DR Reactor cooling water from 1950 to 1967. After radioactive decay and thermal cooling the effluent was discharged to the Columbia River. Due to cooling water leaks and spills, the radionuclide inventory near the basin ranged from 5 to 400 Ci during operations. The basin contamination extended beyond the depth of the interim remedial excavation (4.75 m [15.6 ft]) and reached the water table during operations.

An RI borehole was drilled (Figure 4-2) and soil samples were collected and analyzed to evaluate contamination in the vadose zone to the depth of the water table (19.6 m [64.2 ft]) bgs. The 116-D-9 RI borehole, three previous LFI boreholes, and the interim closeout CVP data are summarized in Appendix D (Tables 84 and 85). The RI and LFI borehole data for contaminants detected or present above background levels are presented in Figures 4-25, 4-26, 4-27, 4-28, and 4-29.

Between the CVP, LFI, and RI sample results for this retention basin, 27 contaminants were detected or were present in the vadose zone above background levels. Contaminant concentrations generally decreased with depth, except for carbon-14 and tin (their concentrations generally increased with depth to the water table). Only nitrate and Cr(VI) are in nearby groundwater wells in excess of drinking water standards (see Figures 4-66 and 4-90 for well locations). Soil concentrations detected or present above background are compared to soil concentrations protective of groundwater and surface water (i.e., PRG, SSL) in Chapter 5. The CVP, LFI, and RI data are also used in Chapter 6 for the human health risk evaluation.

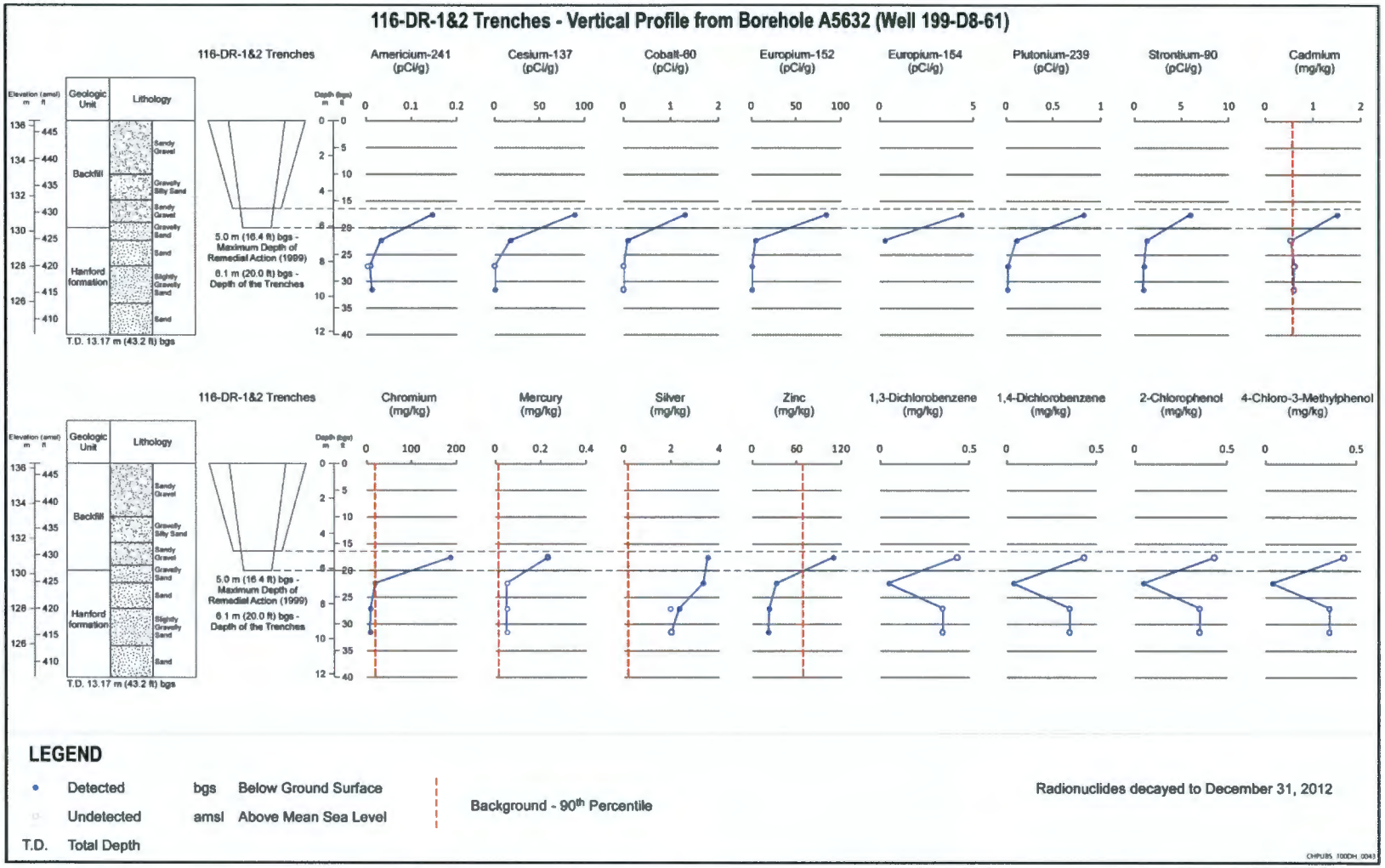

Figure 4-20. 116-DR-1&2 Vertical Profiles of Contamination in LFI Borehole A5632 (Well 199-D8-61)

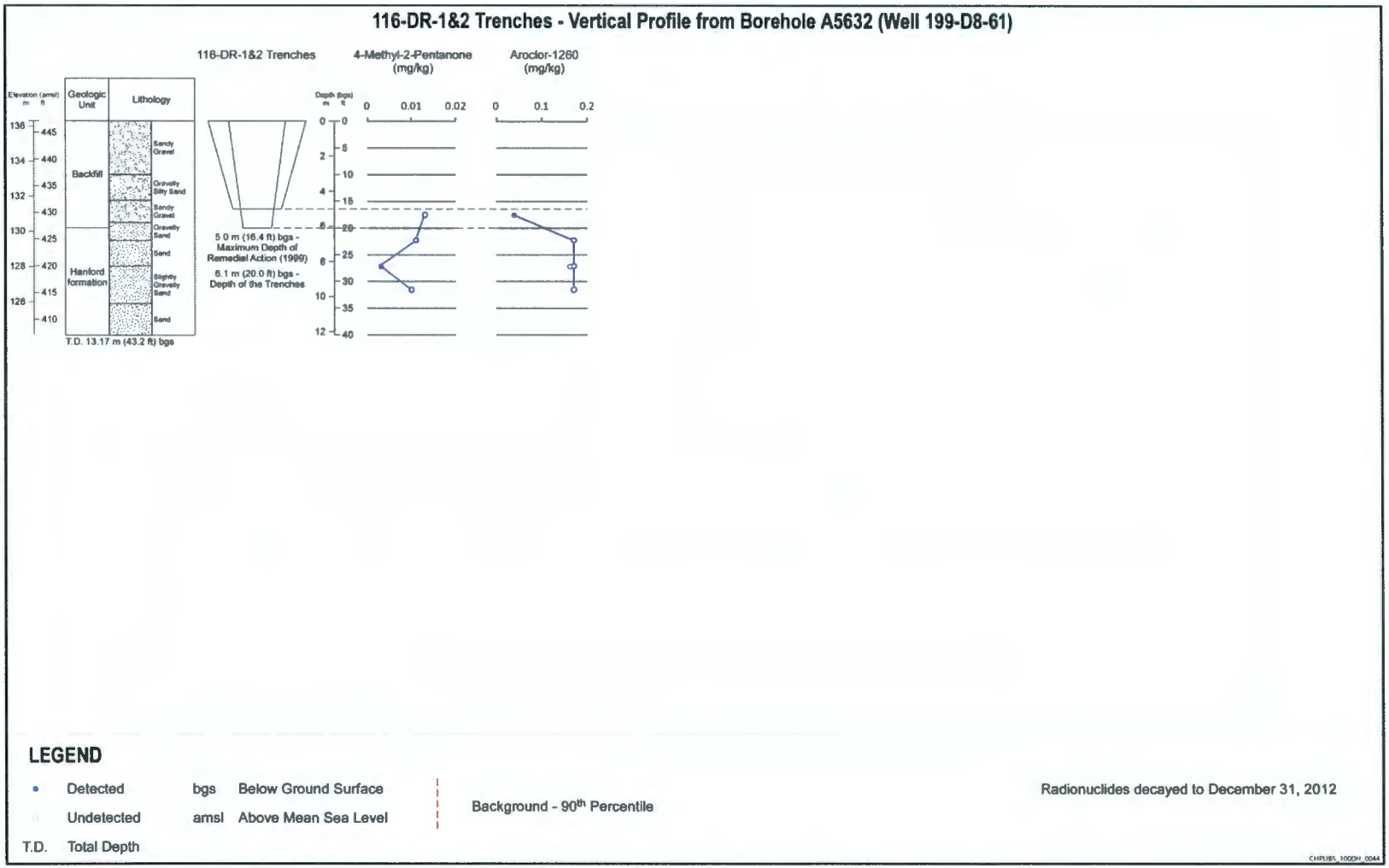

Figure 4-21. 116-DR-1&2 Vertical Profiles of Contamination in LFI Borehole A5632 (199-D8-61)

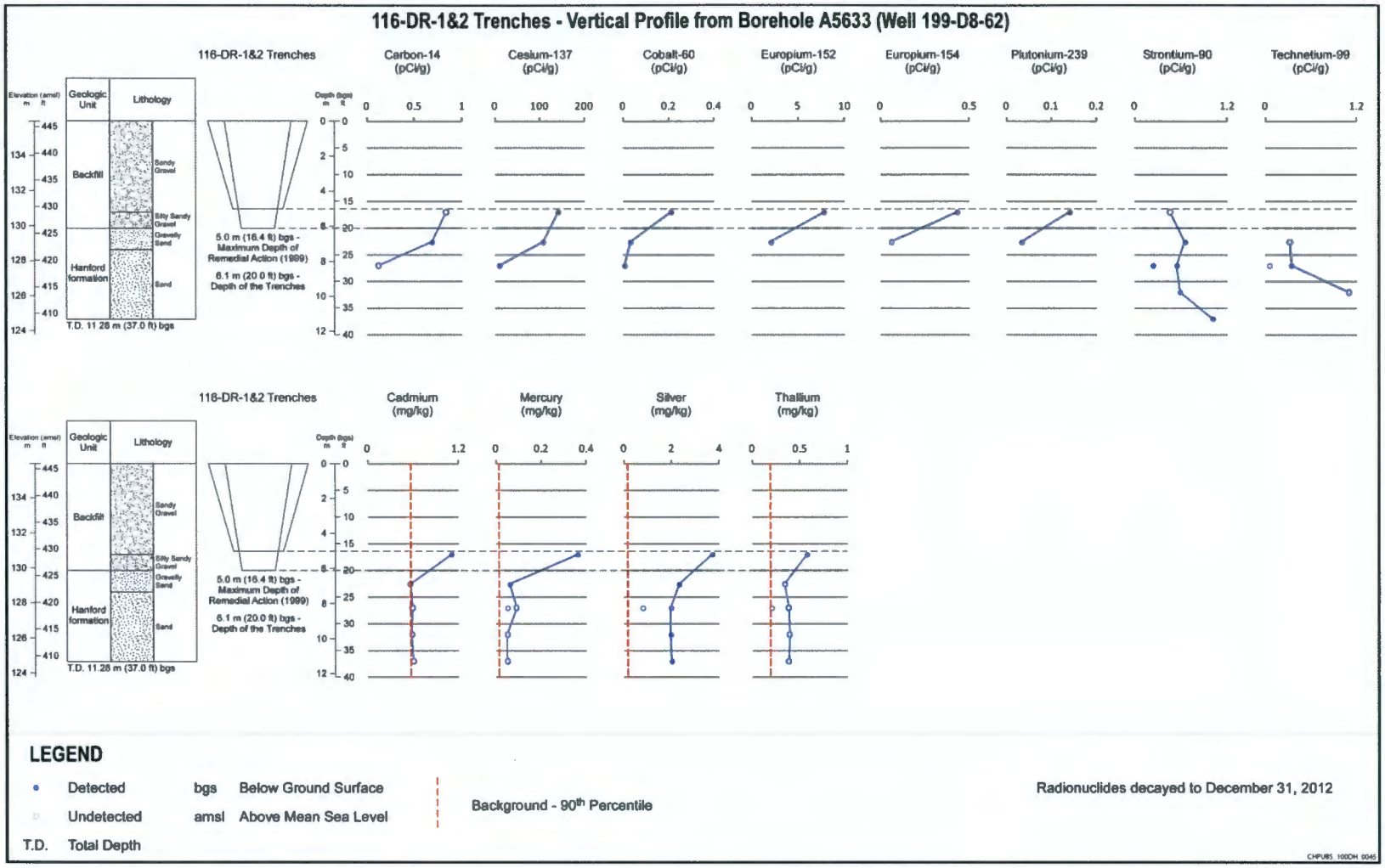

Figure 4-22. 116-DR-1&2 Vertical Profiles of Contamination in LFI Borehole A5633 (Well 199-D8-62)

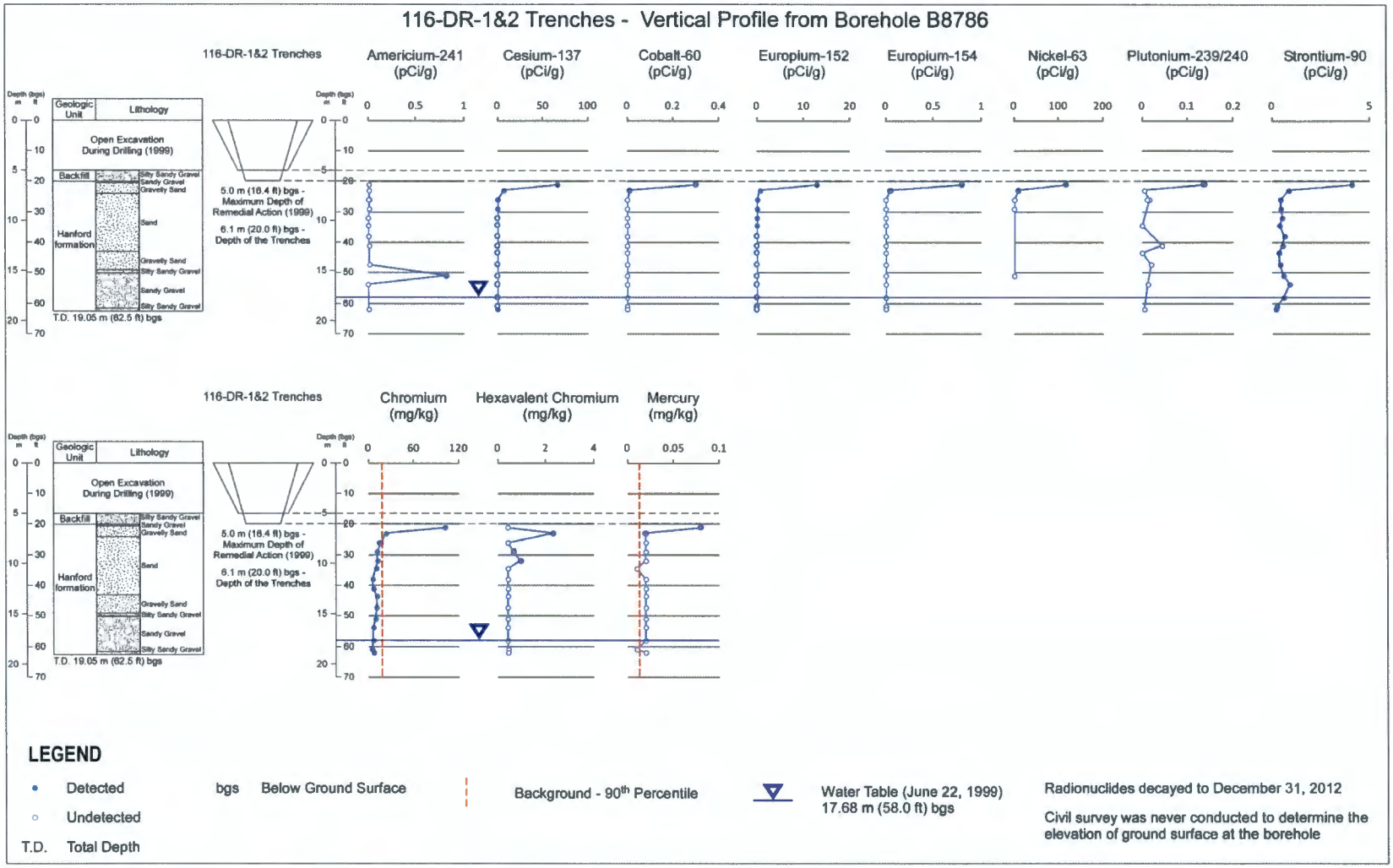

Figure 4-23. 116-DR-1&2 Vertical Profiles of Contamination in Field Remediation Borehole B8786

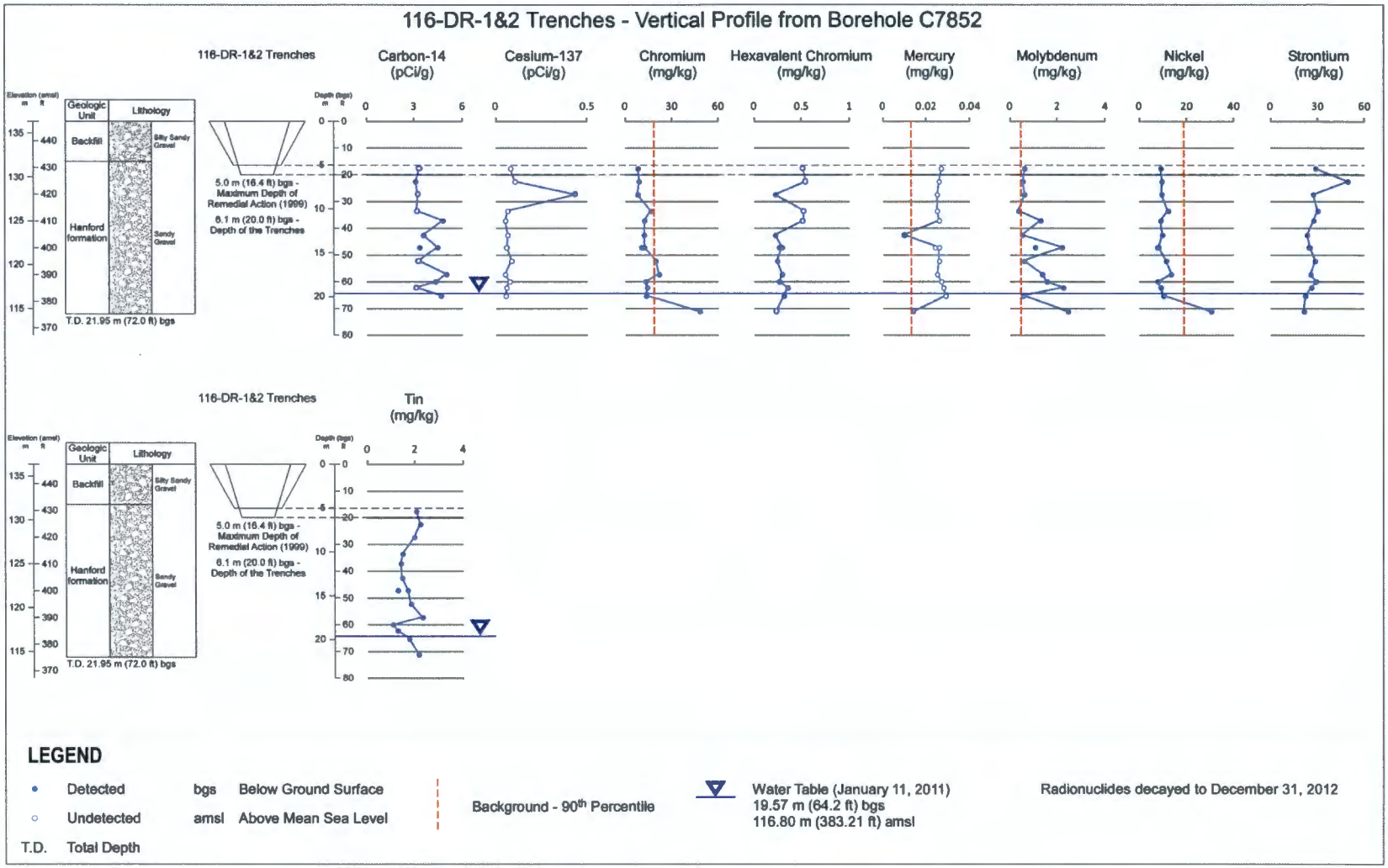

Figure 4-24. 116-DR-1&2 Vertical Profiles of Contamination in Remedial Investigation Borehole C7852

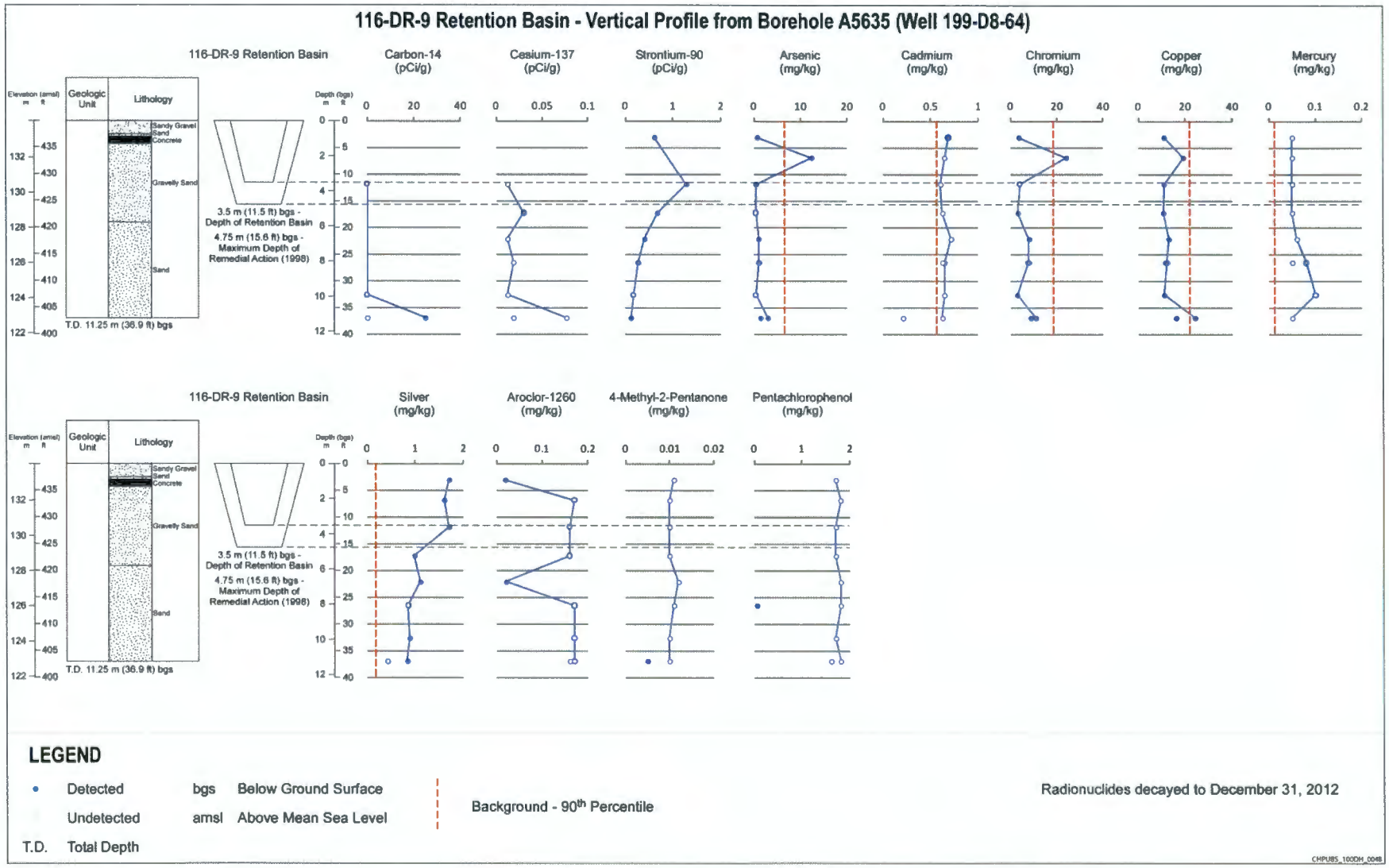

Figure 4-25. 116-DR-9 Vertical Profiles of Contamination in LFI Borehole A5635 (Well 199-D8-64)

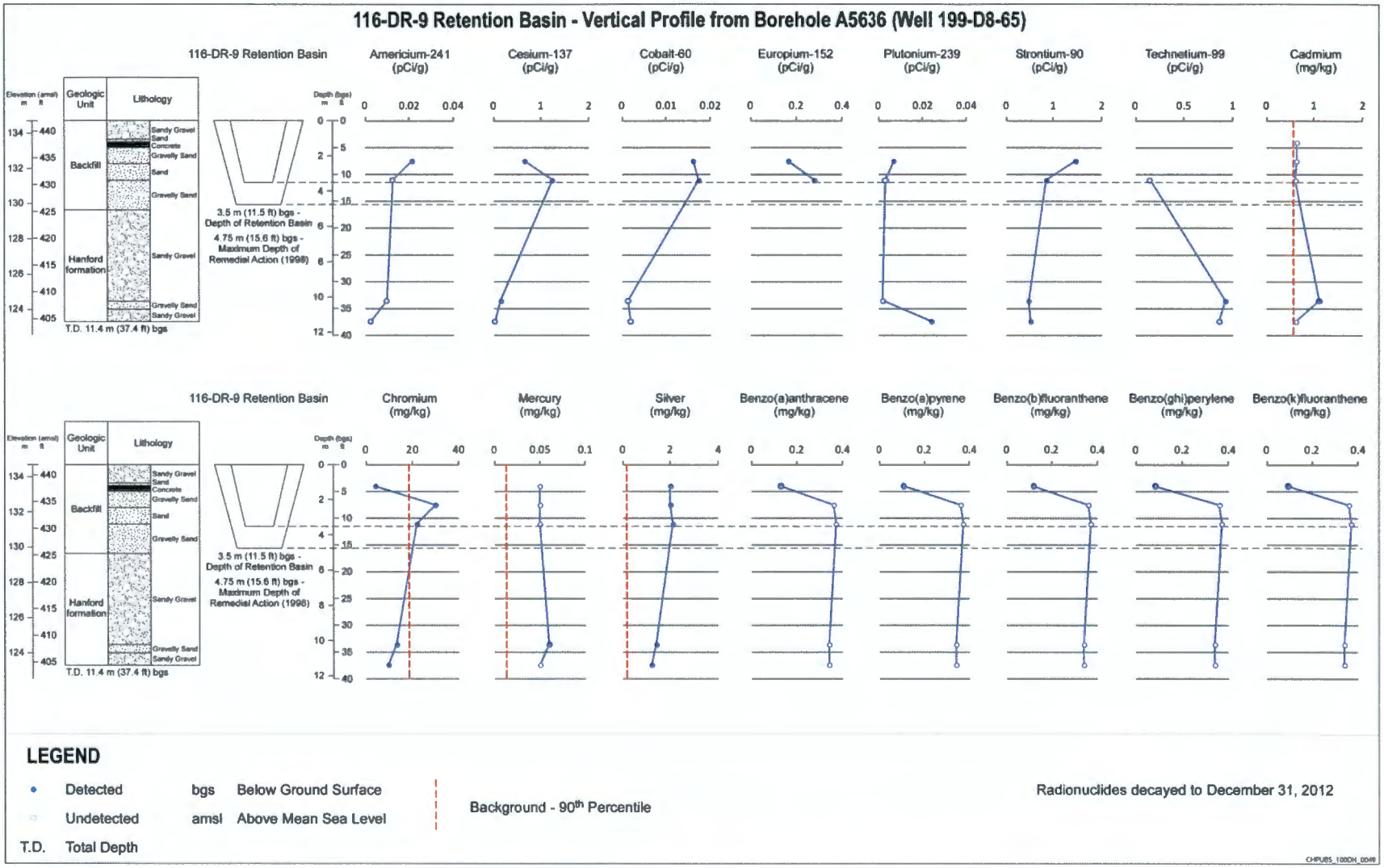

Figure 4-26. 116-DR-9 Vertical Profiles of Contamination in LFI Borehole A5636 (Well199-D8-65)

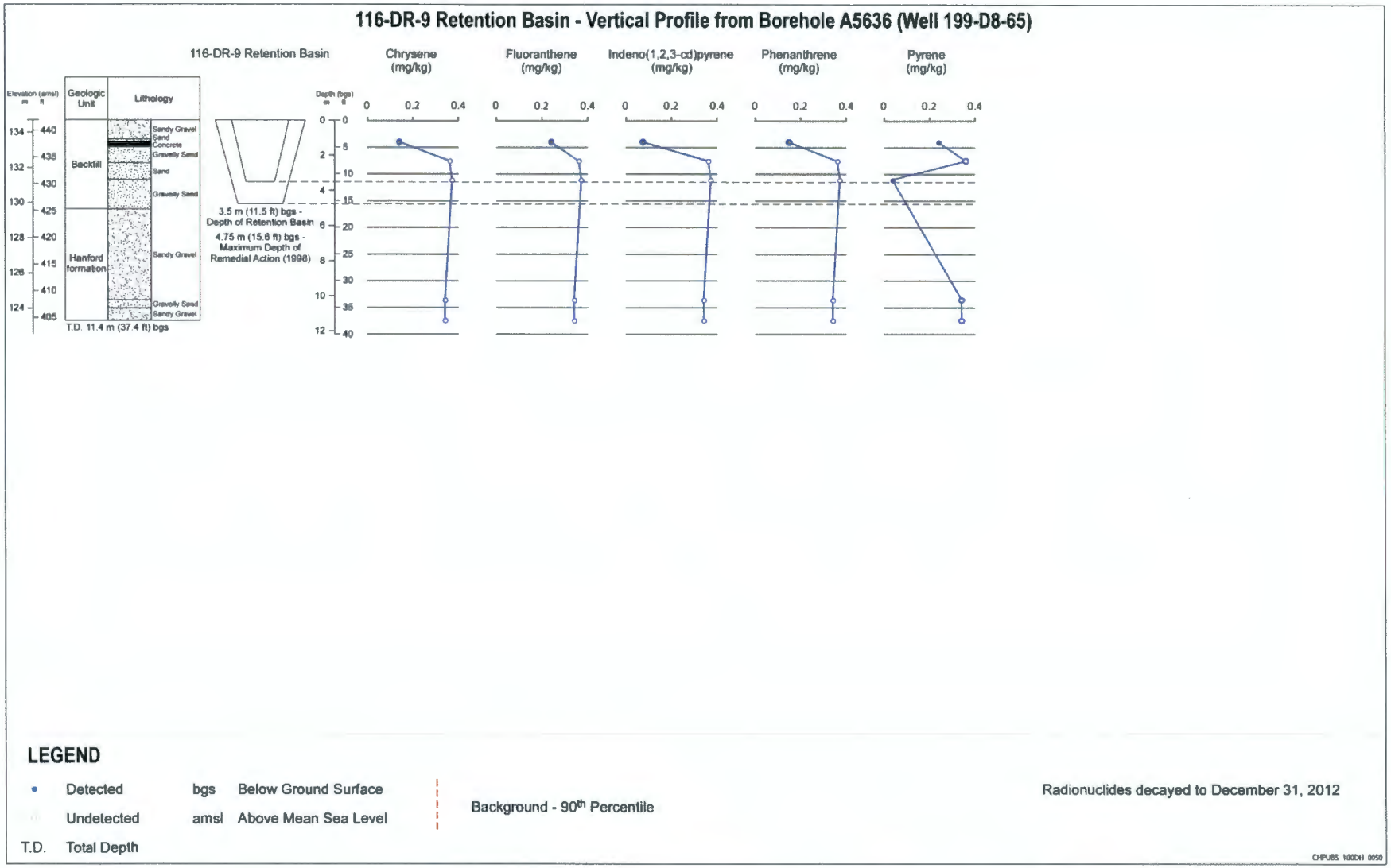

Figure 4-27. 116-DR-9 Vertical Profiles of Contamination in LFI Borehole A5636 (Well 199-D8-65)

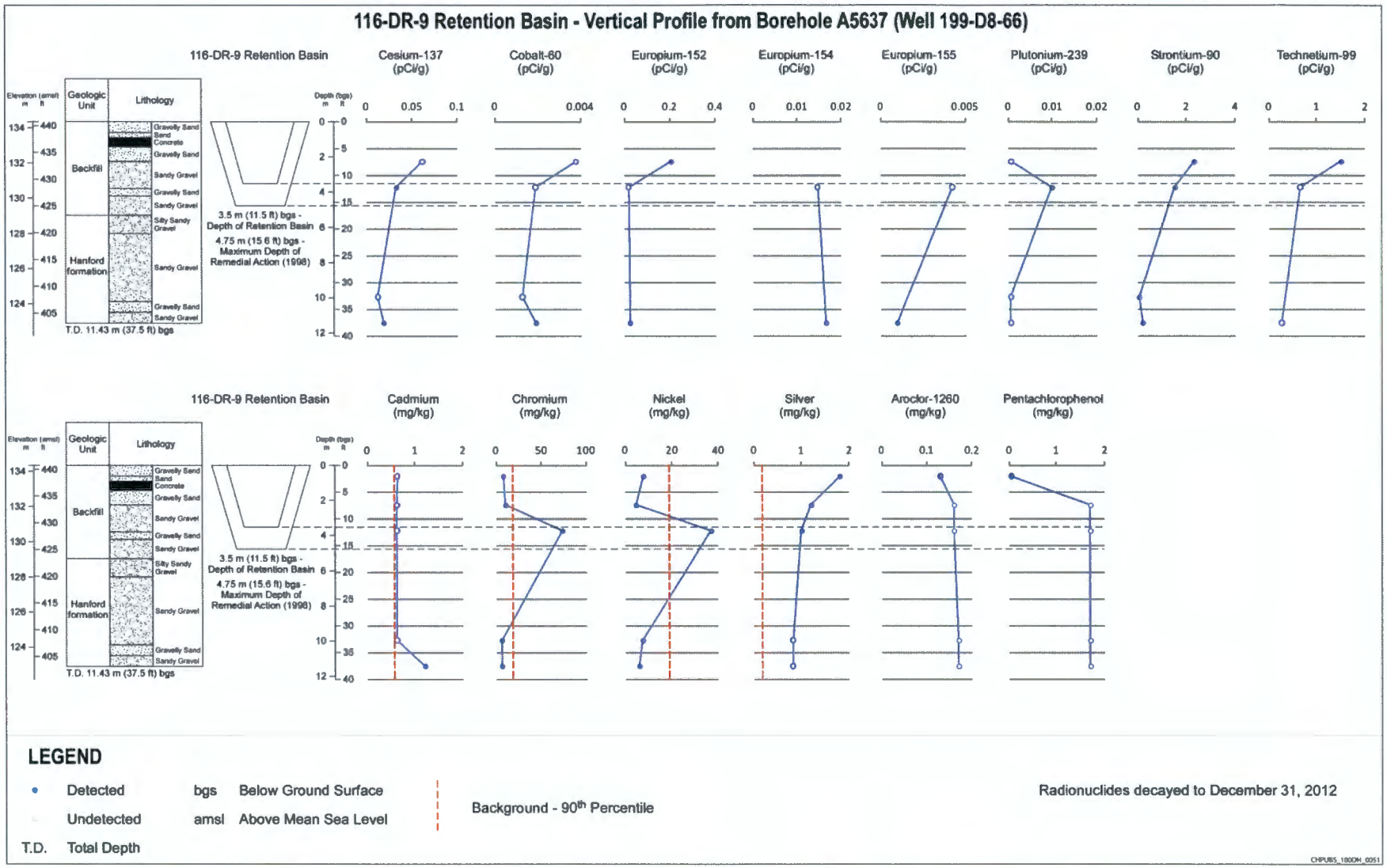

Figure 4-28. 116-DR-9 Vertical Profiles of Contamination in LFI Borehole A5637 (Well 199-D8-66)

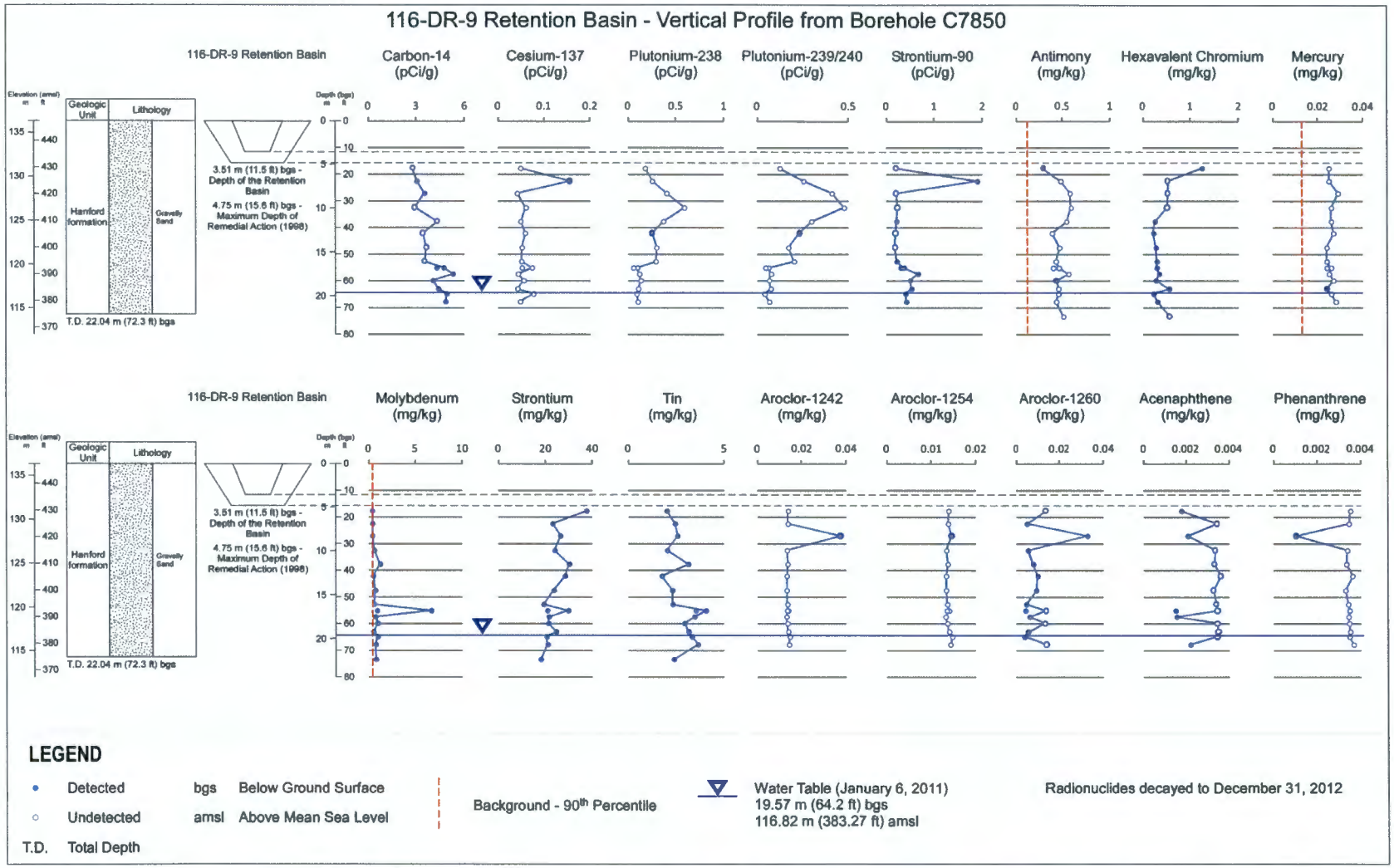

Figure 4-29. 116-DR-9 Vertical Profiles of Contamination in Remedial Investigation Borehole C7850

4.3.11 118-D-6 Reactor Fuel Storage Basin Characterization

The 118-D-6 Reactor FSB stored irradiated fuel elements from 1944 to 1967. The cooling water was not removed from the basin until 1985. During interim remedial action, the floor and walls of the basin were left in place, limiting the depth of excavation to less than the engineered structure. Only concrete samples from the FSB floor were collected during the interim closeout CVP effort. Soil samples were not collected from beneath the basin floor, which reportedly leaked during operations.

An RI borehole (C7857) was drilled adjacent to the FSB (Figure 4-6) and soil samples were collected and analyzed to evaluate vadose zone contamination to the depth of the water table (25 m [82.2 ft]) bgs. The borehole data are summarized in Appendix D (Table 86). The concrete CVP sample results are not included in Appendix D (Table 86). The RI borehole data for contaminants detected or present above background levels are presented in Figures 4-30 and 4-31.

The RI results for the FSB indicate that 19 contaminants were detected or were present in the vadose zone above background levels. Most contaminant concentrations generally decreased with depth. However, barium, total chromium, molybdenum, and nickel had their highest concentrations between 60 and 80 ft bgs. Only nitrate and Cr(VI) are in nearby groundwater wells in excess of drinking water standards (see Figures 4-66 and 4-90 for well locations). Soil concentrations detected or present above background are compared to soil concentrations protective of groundwater and surface water (i.e., PRG, SSL) in Chapter 5. The RI data are also used in Chapter 6 for the human health risk evaluation.

4.3.12 116-H-1 Trench Characterization

The 116-H-1 Trench received effluent from the 116-H-6 retention basin during reactor fuel element failure shut downs from 1952 to 1965 and. The trench received 90,000,000 L (24,000,000 gal) of effluent that included 90 kg (41 lb) of sodium dichromate and a radiological inventory of 33 Ci. The effluent reached the water table during operations, and contamination extended beyond the depth of the interim remedial excavation (4.6 m [15 ft] bgs). The trench is located near the 100-H strontium-90 plume.

An RI borehole (C7864) was drilled adjacent to the trench (Figure 4-32) and soil samples were collected and analyzed to evaluate the vertical extent of vadose zone contamination to the water table (13.3 m [43.5 ft] bgs). In addition, LFI and excavation boreholes (A5724 and C3048, respectively) were drilled historically (1992 and 2000, respectively). The CVP, LFI, and RI data are summarized in Appendix D (Tables 87 and 88). The RI and LFI borehole data for contaminants detected or present above background levels are presented in Figures 4-33, 4-34, 4-35, 4-36 and 4-37.

The CVP, LFI, and RI analytical results for the 116-H-1 Trench indicate that 43 contaminants were detected or were present in the vadose zone above background levels. Contaminant concentrations generally decreased with depth. However, higher concentrations of antimony, total chromium, copper, lead, and molybdenum were present near the water table. Only Cr(VI) and strontium-90 are in nearby groundwater wells in excess of drinking water standards (see Figures 4-66 and 4-95 for well locations). Soil concentrations detected or present above background are compared to soil concentrations protective of groundwater and surface water (i.e., PRG, SSL) in Chapter 5. The RI data are also used in Chapter 6 for the human health risk evaluation.

4.3.13 116-H-2 Trench Characterization

The 116-H-2 Trench received effluent from the 105-H Reactor and the 1608-H Pump House from 1950 to 1965 and. The trench received 600,000,000 L (160,000,000 gal) of effluent that included 600 kg (273 lb) of sodium dichromate and had a radiological inventory of 1.4 Ci. This trench is a high-volume liquid waste site that extends beyond the depth of the interim remedial excavation [2.6 m (8.5 ft) and likely affected groundwater quality during operations.

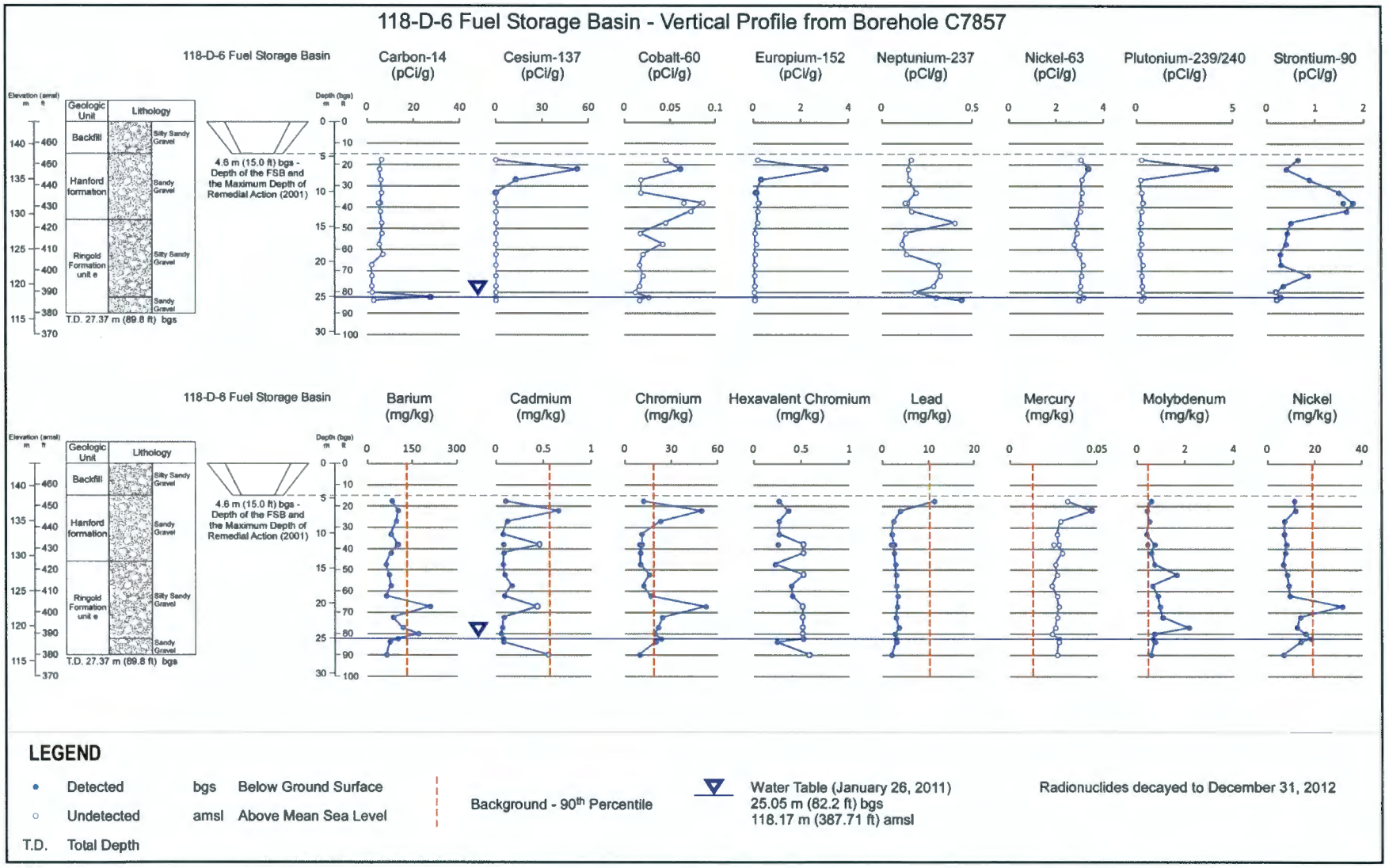

Figure 4-30. 118-D-6 Vertical Profiles of Contamination in Remedial Investigation Borehole C7857

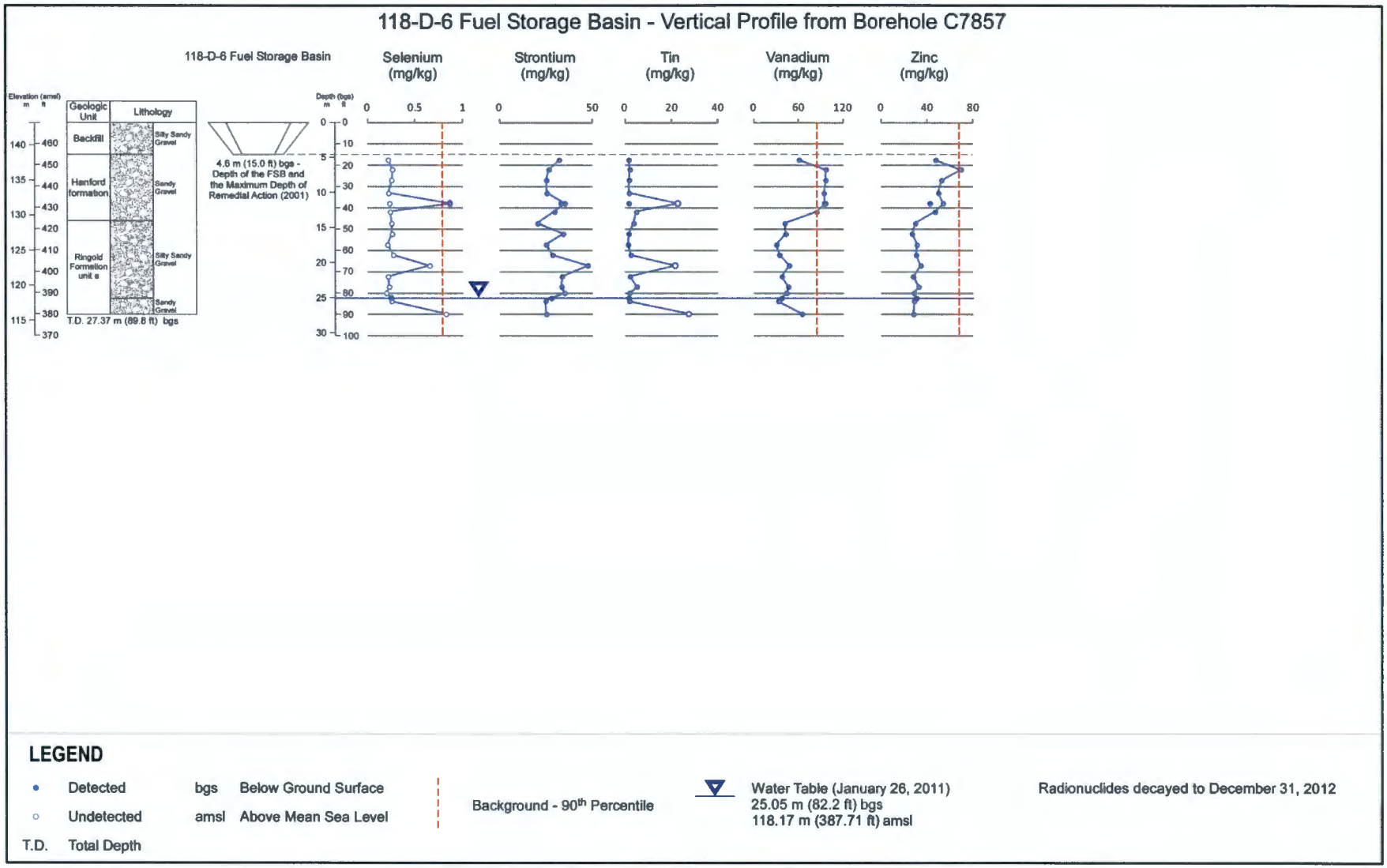

Figure 4-31. 118-D-6 Vertical Profiles of Contamination in Remedial Investigation Borehole C7857

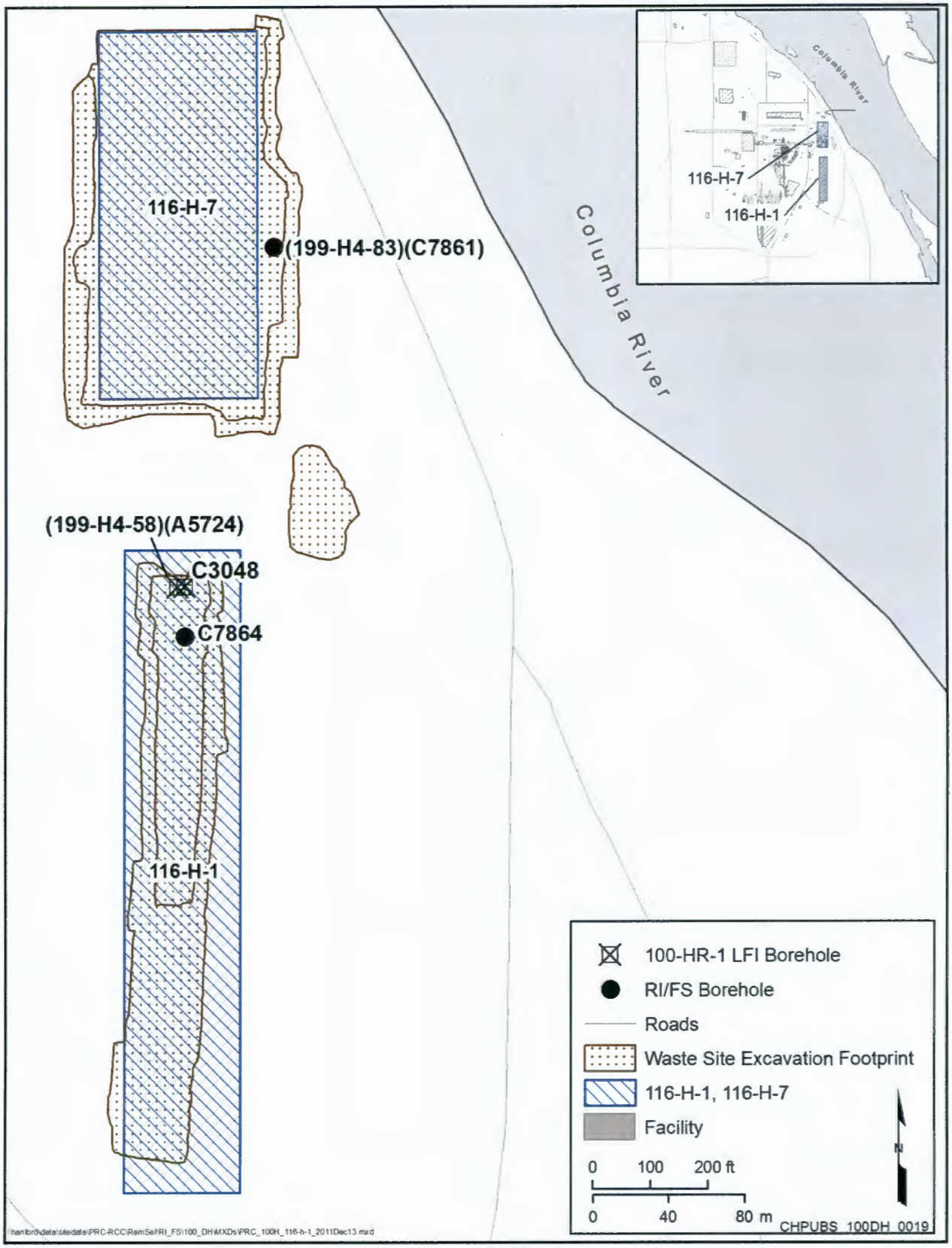

Figure 4-32. 116-H-1 and 116-H-7 Location Map

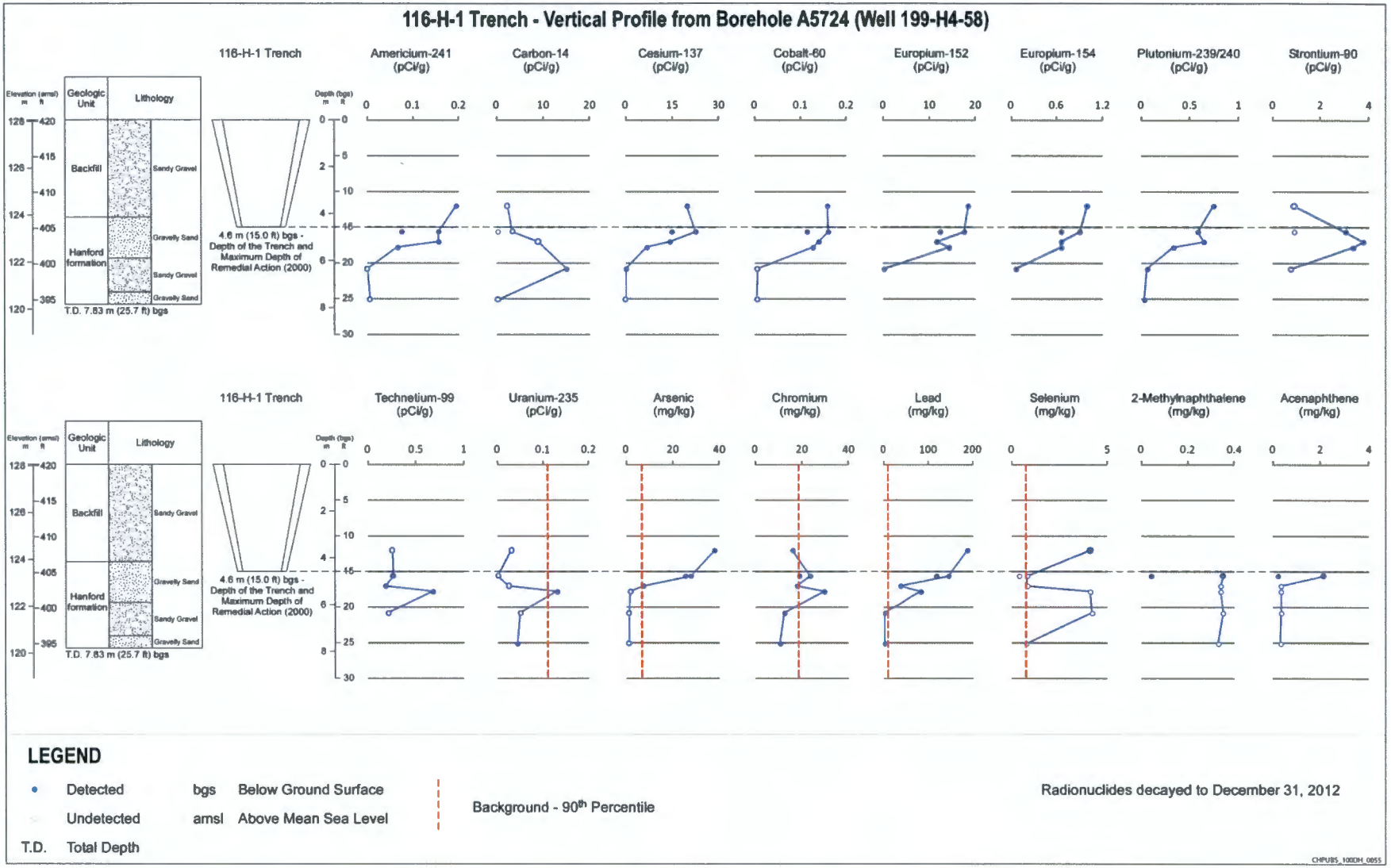

Figure 4-33. 116-H-1 Vertical Profiles of Contamination in LFI Borehole A5724 (Well 199-H-58)

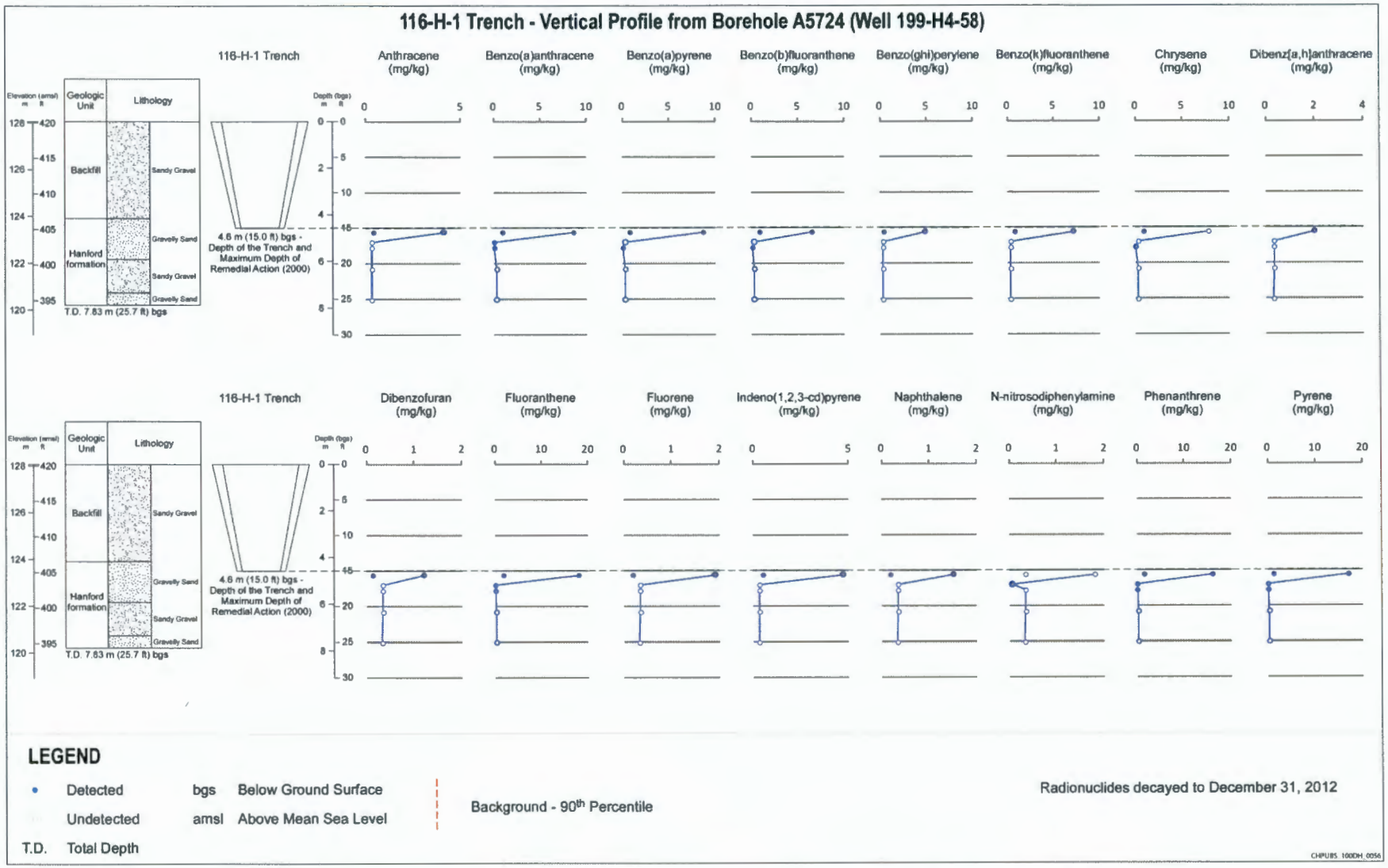

Figure 4-34. 116-H-1 Vertical Profiles of Contamination in LFI Borehole A5724 (Well 199-H4-58)

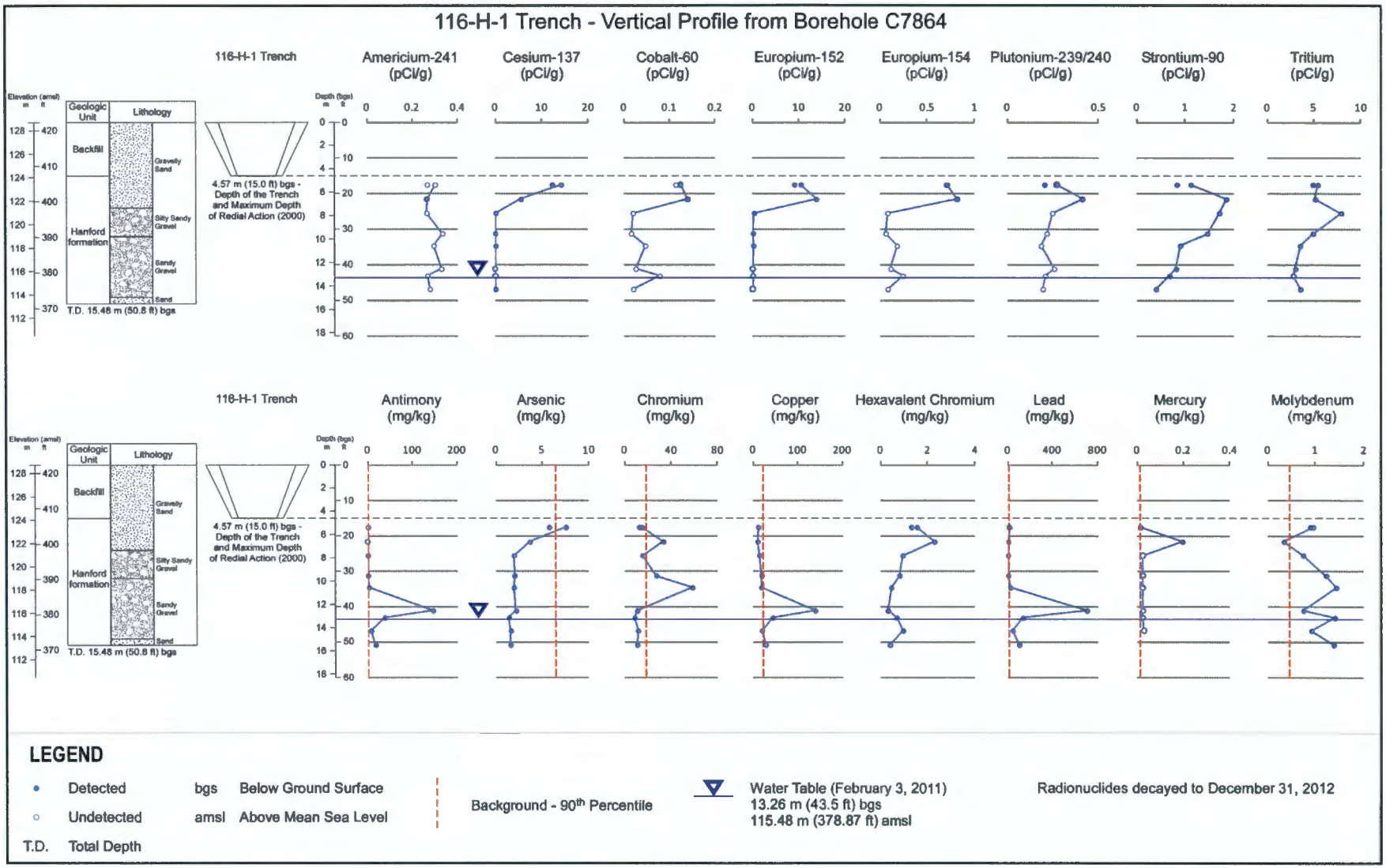

Figure 4-35. 116-H-1 Vertical Profiles of Contamination in Remedial Investigation Borehole C7864

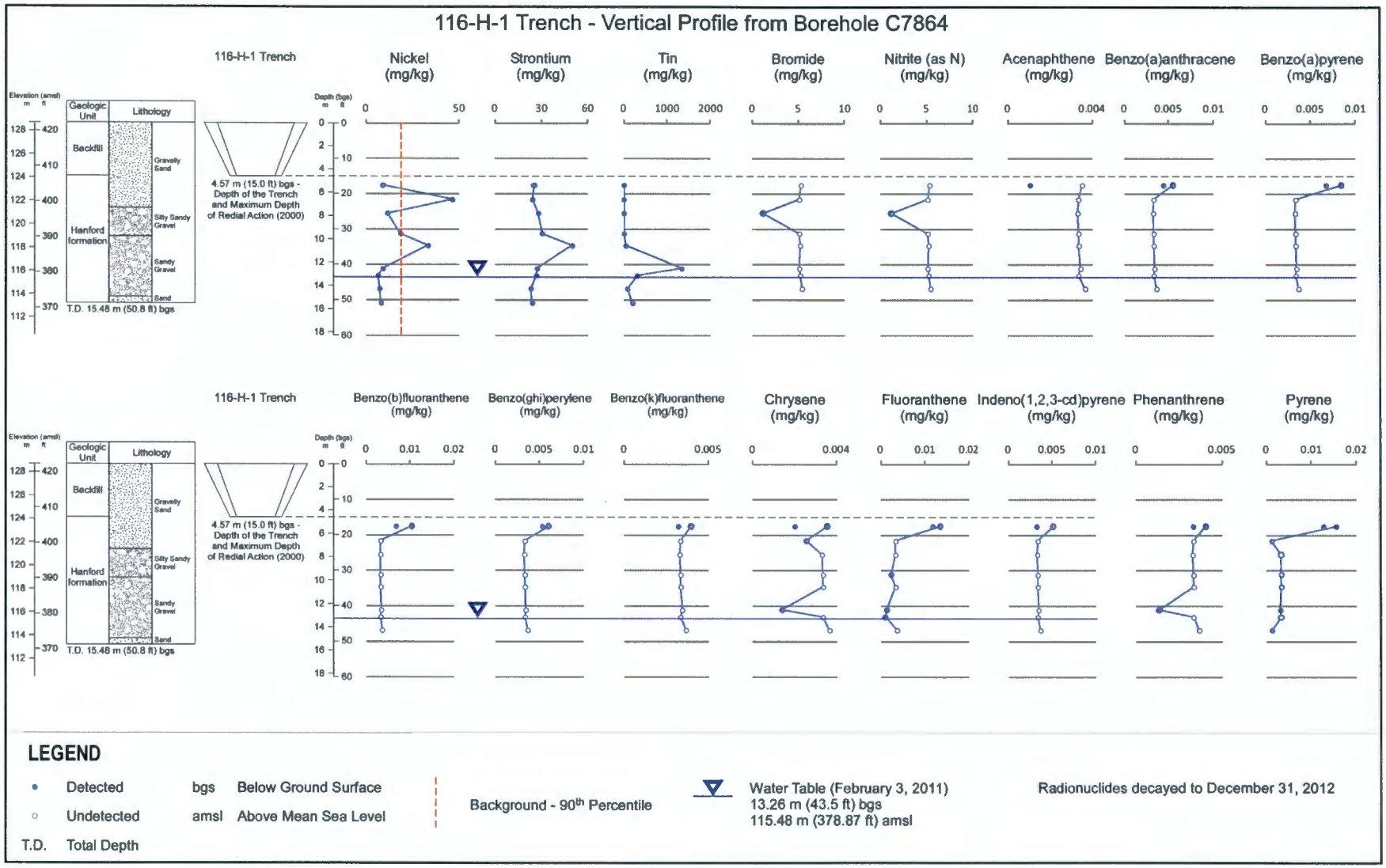

Figure 4-36. 116-H-1 Vertical Profiles of Contamination in Remedial Investigation Borehole C7864

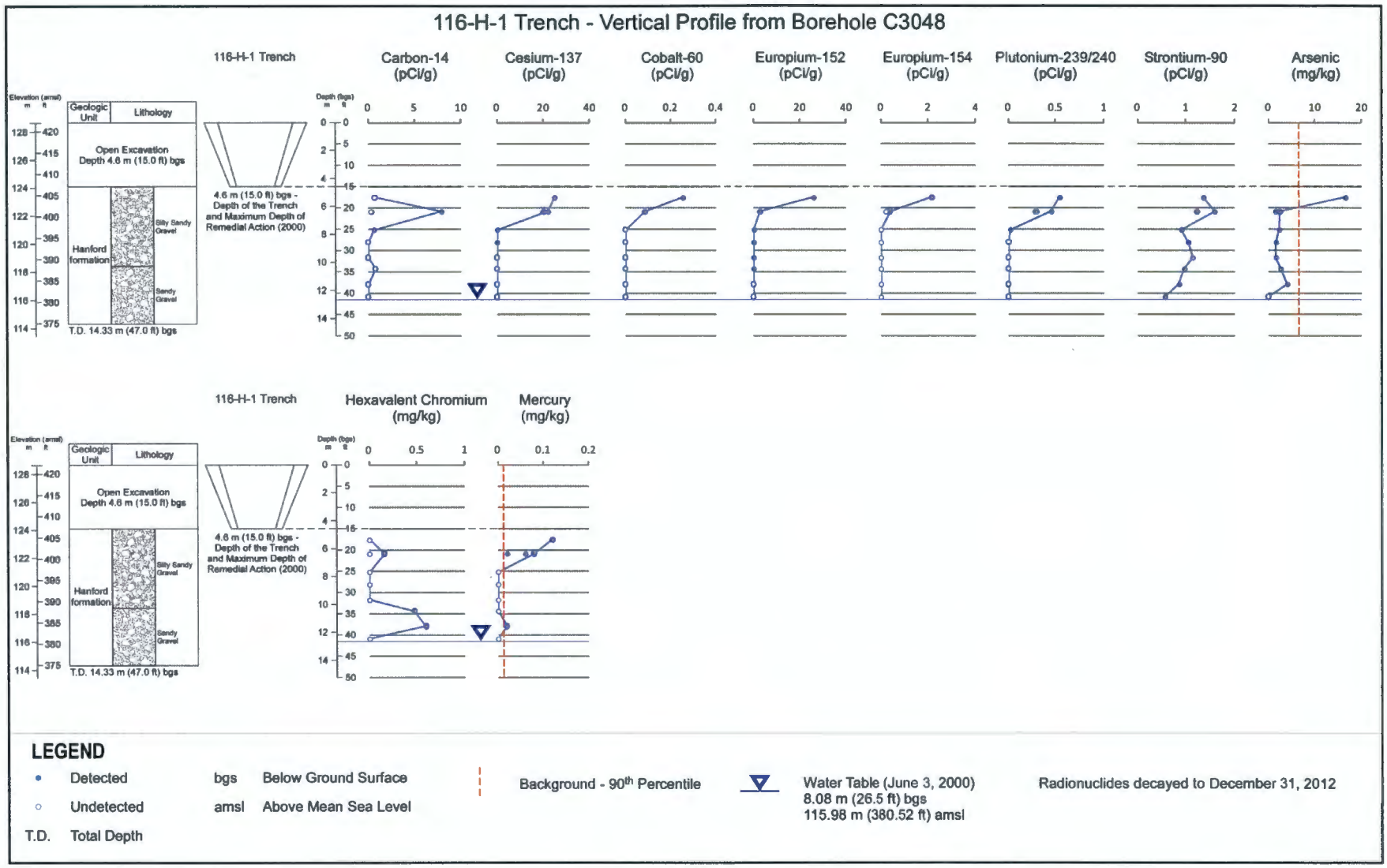

Figure 4-37. 116-H-1 Vertical Profiles of Contamination in Excavation Borehole C3048

An RI test pit was excavated through the trench location (Figure 4-38) and soil samples were collected and analyzed to evaluate the vertical extent of contamination to a depth of 5.8 m (19 ft). The CVP, LFI, and RI data are summarized in Appendix D (Table 89). The RI and LFI borehole data for contaminants detected or present above background levels are presented in Figures 4-39 and 4-40.

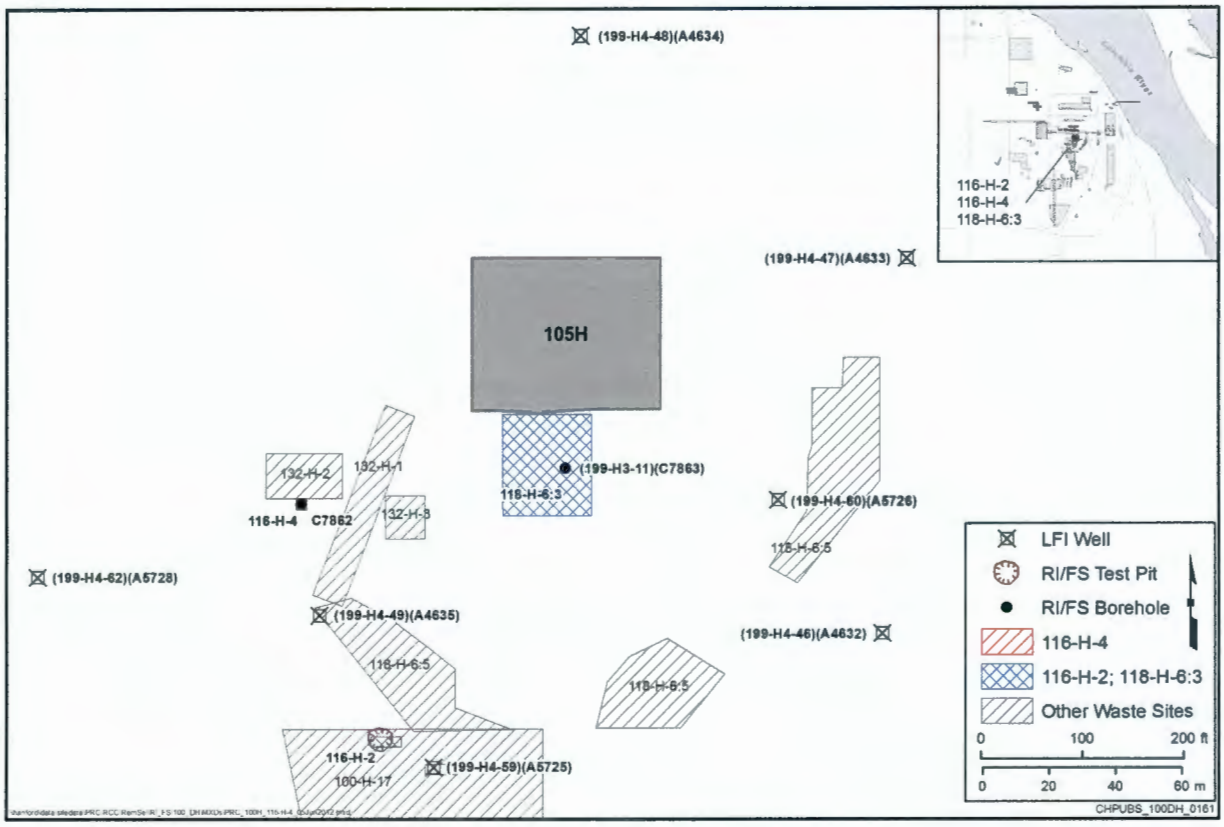

Figure 4-38. 116-H-2, 116-H-4, and 118-H-6:3 Location Map

Between the CVP, LFI, and RI sample results for this trench, 12 contaminants were detected or were present above background levels in the vadose zone. Only Cr(VI) was detected in groundwater wells in excess of drinking water standards (see Figure 4-66 for well locations). Soil concentrations detected or present above background are compared to soil concentrations protective of groundwater and surface water (i.e., PRG, SSL) in Chapter 5. The RI data are also used in Chapter 6 for the human health risk evaluation.

4.3.14 116-H-4 Crib Characterization

The 116-H-4 Crib received effluent from the 105-H Reactor from 1950 to 1952, and the 1,000 L (254 gal) of effluent received included 1,000 kg (454 lb) of sodium dichromate and had a radiological inventory of 270 Ci. Contaminated material was removed from this site in 1960 and placed in the 118-H-5 Burial Ground to facilitate construction of the 117-H Building. The depth of the soil removed was not well documented and it is not known if contamination in the soil column was adequately removed. The crib was considered a significant source of sodium dichromate.

An RI borehole (C7862) was drilled through the crib (Figure 4-38) and soil samples were collected and analyzed to evaluate the vertical extent of analytes in the vadose zone from a depth of 1.46 m (4.8 ft) bgs to the water table (13.7 m [44.8 ft]) bgs. No other soil data are available from this site. Summary data for the RI borehole are presented in Appendix D (Table 90). Figure 4-41 presents vertical profiles of borehole contamination detected for contaminants without background values or present above background.

The RI results for the crib indicate that nine contaminants were detected or were present in the vadose zone above background concentrations. Contaminant concentrations generally decrease with depth, although carbon-14 and tritium had higher concentrations at about 40 ft bgs. Only Cr(VI) was detected in nearby groundwater wells in excess of drinking water standards (see Figure 4-66 for well locations). Soil concentrations detected or present above background are compared to soil concentrations protective of groundwater and surface water (i.e., PRG, SSL) in Chapter 5. The RI data are also used in Chapter 6 for the human health risk evaluation.

4.3.15 116-H-6 and 100-H-33 Solar Evaporation Basins

The 116-H-6 and 100-H-33 waste site designations address the contaminated soil associated with the 183-H solar evaporation basins. Historically, 116-H-6 pertains to the chemical contamination beneath the site, which has been "closed-out" under RCRA ("Closure Certification for the 183-H Solar Evaporation Basins (T-1-4)" [96-EAP-246]), while 100-H-33 addresses radiological contamination. In this subsection, discussion of 116-H-6 is synonymous with 100-H-33, unless otherwise noted. The waste site and borehole/sample locations are shown in Figure 4-42.

The 116-H-6 Solar Evaporation Basin site is a RCRA TSD unit that consists of four basins. The facility was used from 1949 to 1985 to evaporate various liquid waste streams, including neutralized, spent acid etch solutions containing technetium-99 and uranium. The basins were demolished in 1995 and 0.6 m (2 ft) of soil was removed from beneath the site. This soil removal action was based on 1991 soil data from eight boreholes sampled within and adjacent to the site boundary. Analytical data from the boreholes showed high levels of contamination up to 0.6 m (2 ft) below the bottom of the basin (this equated to a remediation depth of about 2.7 m (9 ft) bgs).

However, below Basin 1, soil removal continued to 4.6 m (15 ft) below the former structure (*183-H Solar Evaporation Basins Postclosure Plan* [DOE/RL-97-48]), indicating excavation to about 6.1 m (20 ft) bgs. A test pit was then dug below the Basin 1 excavation to 7.6 m (25 ft) bgs (*183-H Solar Evaporation Basins Postclosure Plan* [DOE/RL-97-48]). These test pit samples indicated nitrate and fluoride soil contamination above industrial standards (MTCA Method B [WAC 173-340] and Method C *183-H Solar Evaporation Basins Postclosure Plan* [DOE/RL-97-48]). Due to these results, the test pit soil from Basin 1 was disposed at the ERDF, and the site was backfilled. Protection of groundwater was demonstrated through modeling and a modified RCRA closure for 116-H-6 that included groundwater monitoring was approved. During the RI, an additional borehole (199-H4-84) associated with Basin 1 was drilled and sampled within the site boundary to the water table 12.6 m (41.5 ft) bgs. Summary data for the boreholes are presented in Appendix D (Tables 91, 92, and 93). Vertical profiles of borehole contamination detected for contaminants without background values or present above background are presented in Figures 4-43 through 4-51.

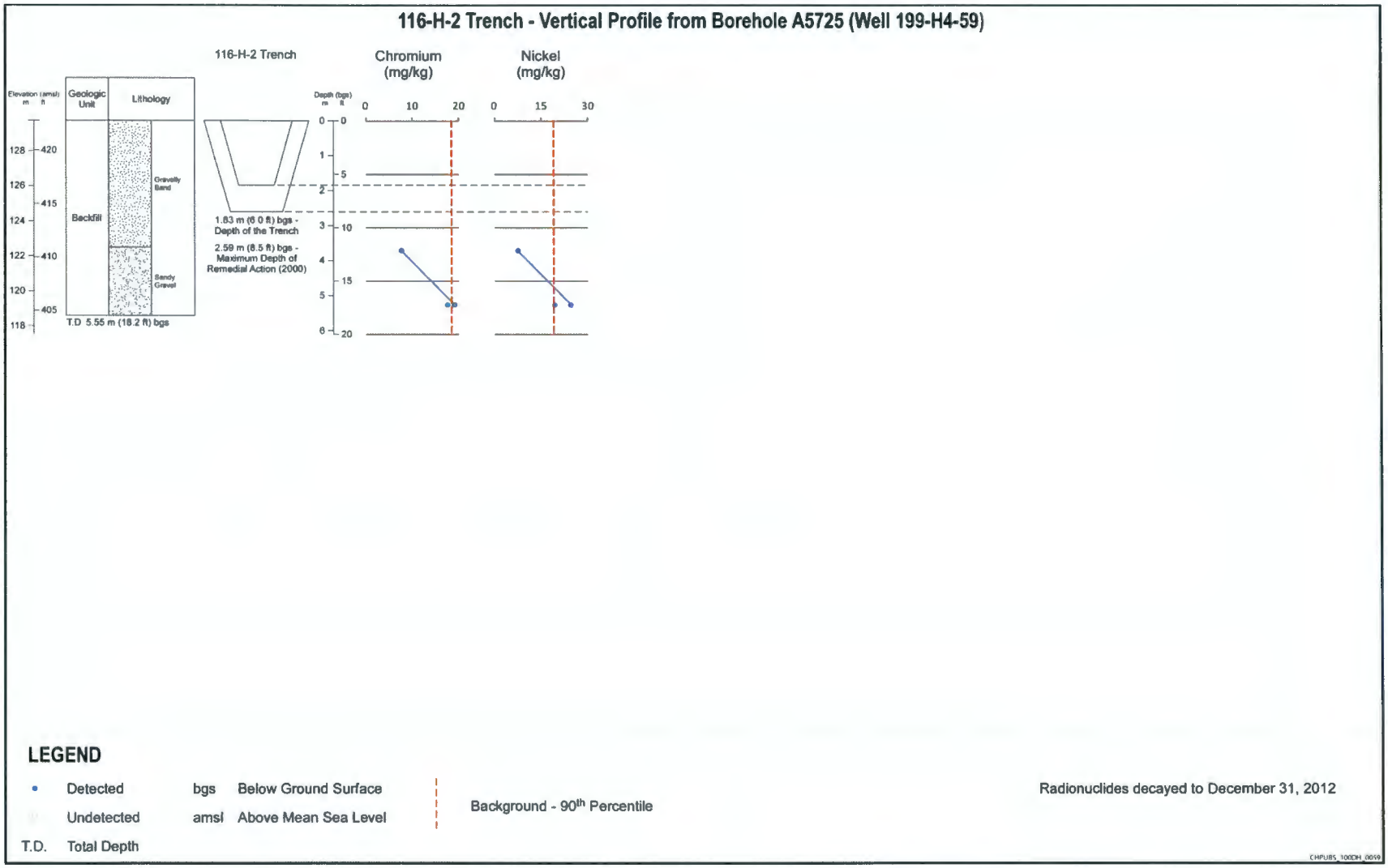

Figure 4-39. 116-H-2 Vertical Profiles of Contamination in LFI Borehole A5725 (Well 199-H4-59)

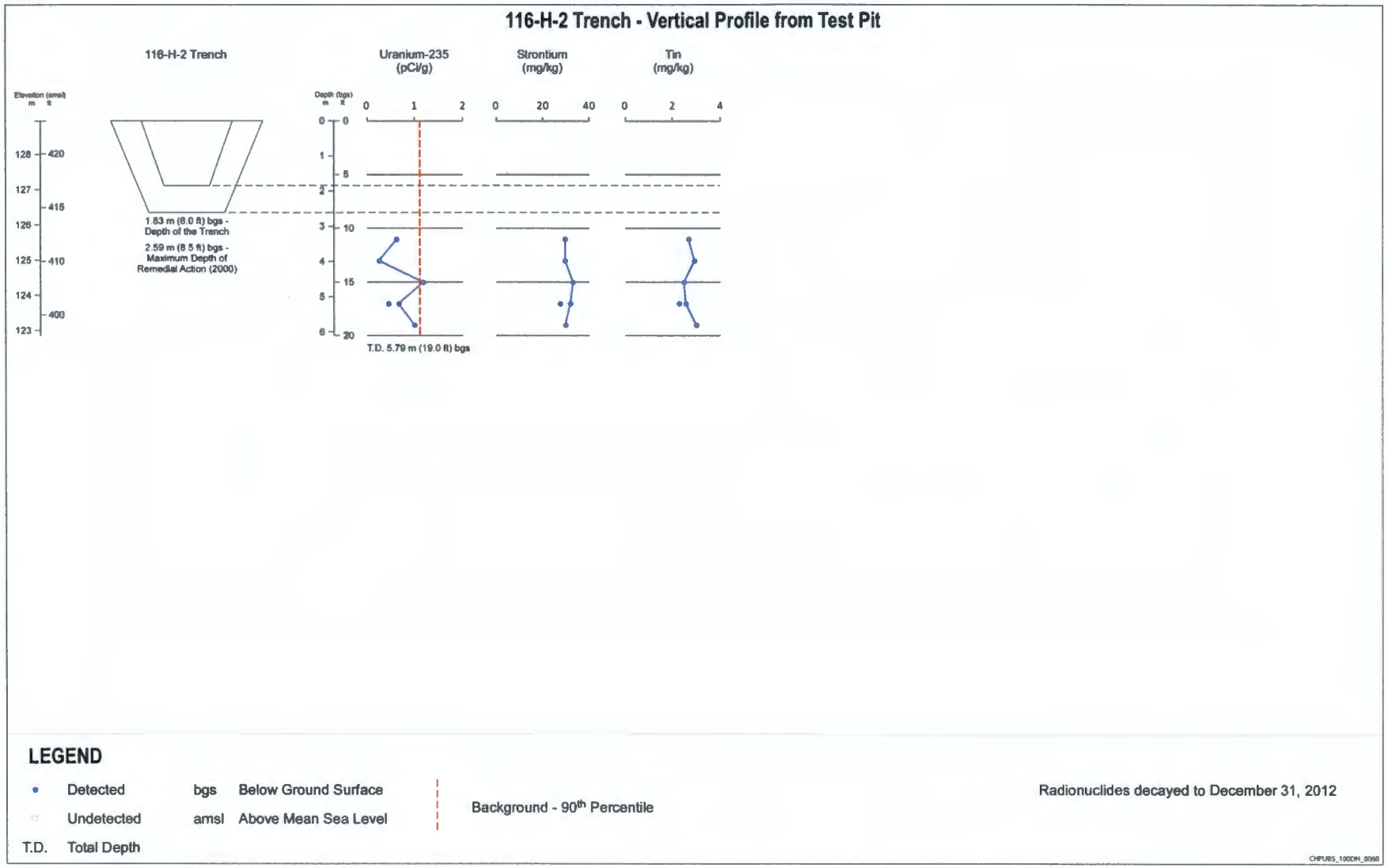

Figure 4-40. 116-H-2 Vertical Profiles of Contamination in Remedial Investigation Test Pit

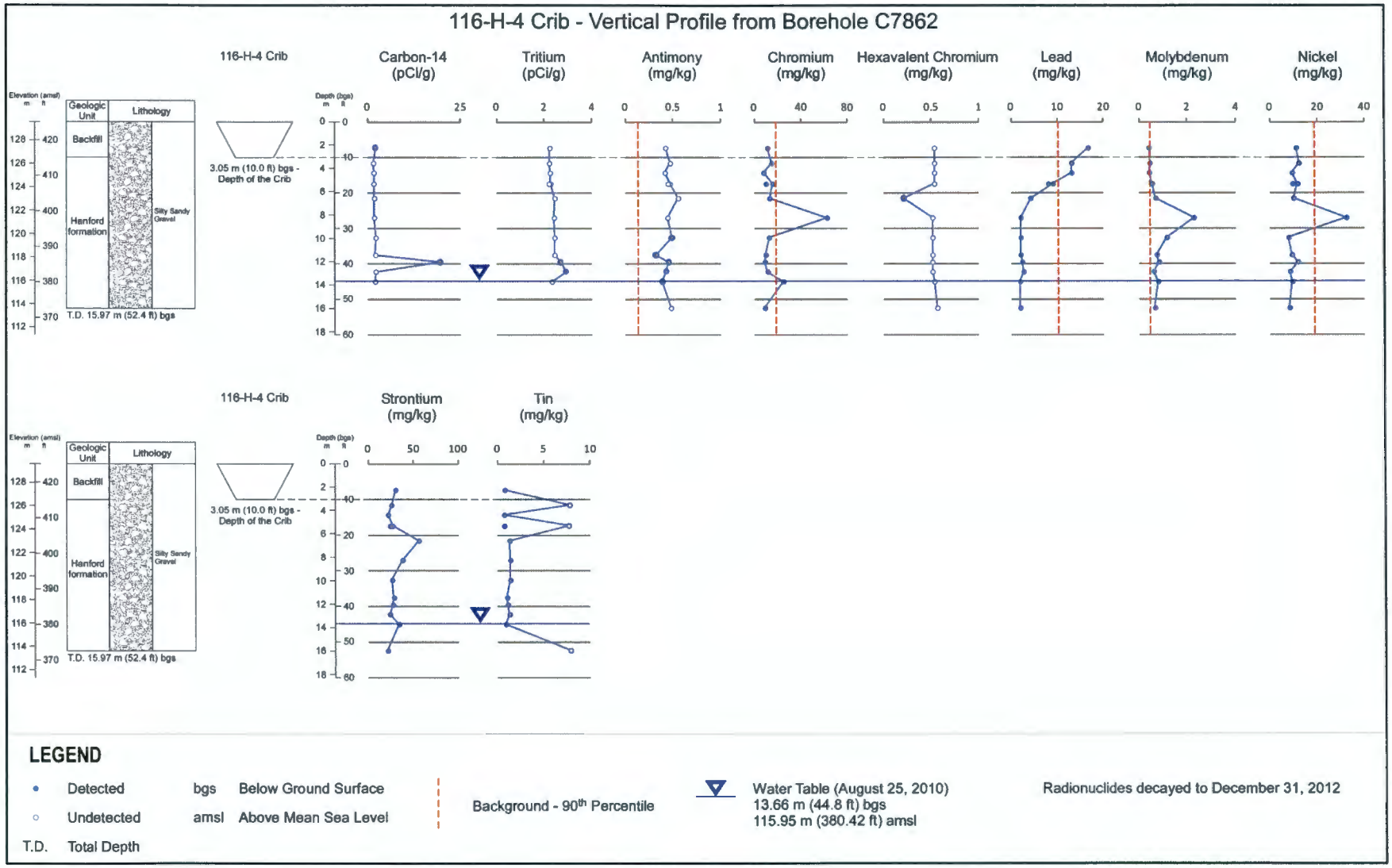

Figure 4-41. 116-H-4 Crib Vertical Profiles of Contamination in Remedial Investigation Borehole C7862

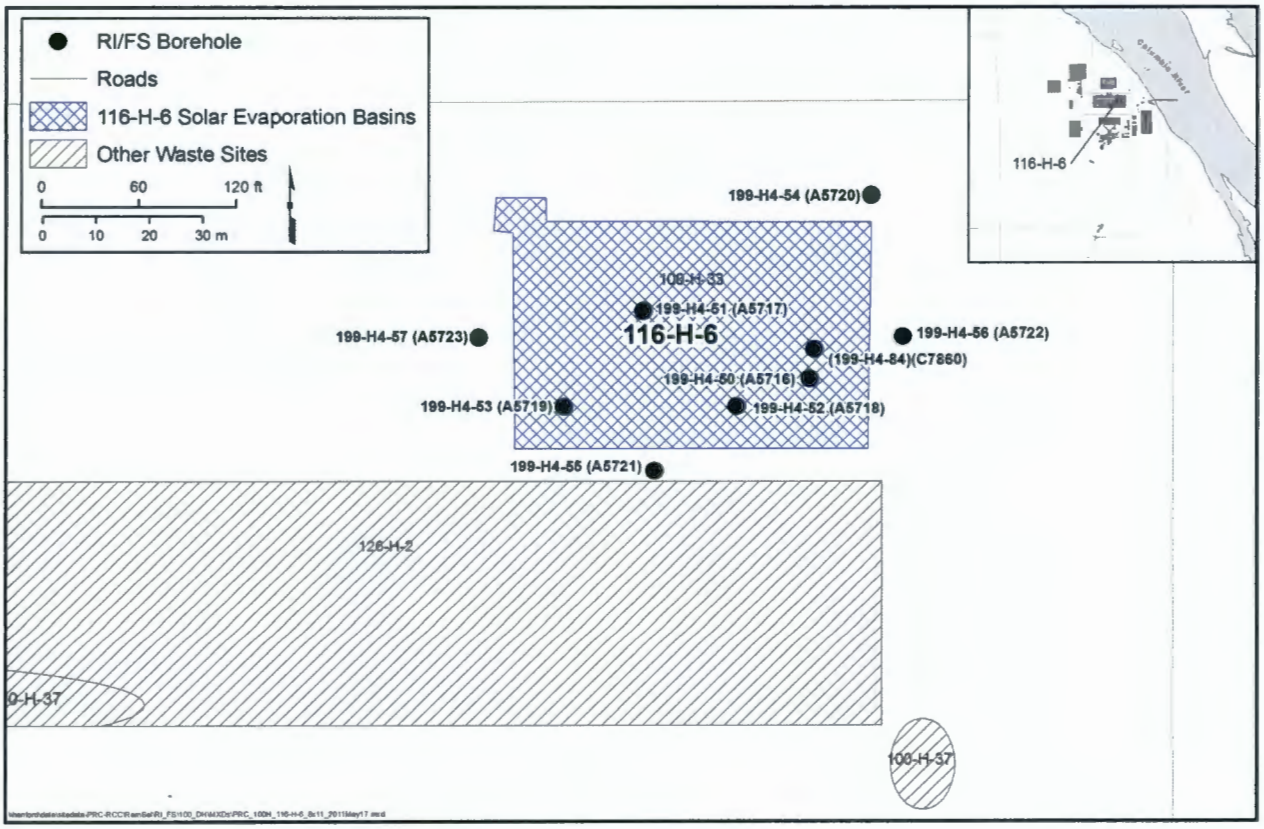

Figure 4-42. 116-H-6 Location Map

An evaluation of the borehole and test pit sample results for the Solar Evaporation Basin site indicates that 24 contaminants were detected or were present in the vadose zone above background concentrations within the site boundary. Contaminant trends in individual boreholes indicate that technetium-99, strontium-90, and tritium concentrations increase with depth, but their levels are typically <2 to 7 pCi/g. Nitrate reaches a maximum of 304 mg/kg at 10.2 m (33.4 ft) bgs, while Cr (VI) concentrations are <2 mg/kg beneath the site. Only eight contaminants (cobalt-60, technetium-99, antimony, cadmium, lead, selenium, nitrate, and fluoride) were detected or were present above background levels from boreholes adjacent to the site. Detecting fewer contaminants adjacent to the site suggests that transport is mainly vertical beneath the site with little lateral spreading. Cr (VI) is the only contaminant detected above the drinking water standards beneath this site. Soil concentrations detected or present above background are compared to soil concentrations protective of groundwater and surface water (i.e., PRG, SSL) in Chapter 5. The RI data are also used in Chapter 6 for the human health risk evaluation.

4.3.16 116-H-7 Retention Basin Characterization

The 116-H-7 Retention Basin received cooling water from the 105-H Reactor from 1949 to 1965. After radioactive decay and thermal cooling, the effluent was discharged from this concrete basin to the river. The basin, a high-volume liquid site that leaked, affected groundwater during operations and, thus, contamination extended beyond the CVP interim remedial excavation depth (4.75 m [15.6 ft) bgs.

An RI borehole (C7861, Figure 4-32) was drilled and sampled to evaluate the vertical extent of contamination through the vadose zone to the water table (10.7 m [35 ft]) bgs. Summary data for the CVP, LFI borehole, and RI borehole are presented in Appendix D (Tables 94, 95, and 96). Vertical profiles of borehole contamination detected or present above background are in Figures 4-52 and 4-53.

The CVP, LFI, and RI analytical results for 116-H-7 indicate that 21 contaminants were detected or present above background in the vadose zone beneath the site. Contaminant trends vary at this site with many concentrations generally decreasing with depth. However, strontium-90, antimony, strontium (metal), and molybdenum have increased concentrations toward the water table. Only nitrate and Cr(VI) are detected in nearby groundwater wells in excess of drinking water standards (see Figure 4-66 for well locations). Chapter 5 compares soil concentrations detected or present above background to soil concentrations protective of groundwater and surface water (i.e., PRG, SSL). The RI data are also used in Chapter 6 for the human health risk evaluation.

4.3.17 118-H-6 Reactor Fuel Storage Basin Characterization

The 118-H-6 Reactor FSB was used to store irradiated fuel elements from 1949 to 1965. The basin leaked during operations and contamination extended beyond the depth of remedial excavation (7.5 m [26.5 ft]). The FSB was also identified as a site that should be characterized to determine if leaked contamination from it might now be located under the 105-H ISS reactor structure.

An RI borehole (C7863, Figure 4-38) was drilled and sampled to evaluate the vertical extent of contamination through the vadose zone to the water table (14.6 m [48 ft] bgs). Summary data for the CVP and RI borehole are presented in Appendix D (Table 97). Vertical profiles of borehole contamination detected or present above background are in Figure 4-54.

The CVP and RI analytical results for the FSB indicate that 21 contaminants were detected or present above background in the vadose zone. Contaminant trends generally decrease with depth at this site. However, strontium-90, total chromium, mercury, molybdenum, and tin concentrations are greater toward the water table. Only Cr(VI) is detected in nearby groundwater wells in excess of DWSs (see Figures 4-66 for well locations). The absence or low concentration of Cr(VI), total chromium, and strontium-90 in the RI borehole results suggest that the vadose zone beneath the FSB is not contributing to local groundwater quality and historical FSB leaks are not likely under the ISS 105-H Reactor. Chapter 5 compares soil concentrations detected or present above background to soil concentrations protective of groundwater and surface water (i.e., PRG, SSL). The RI data are also used in Chapter 6 for the human health risk evaluation.

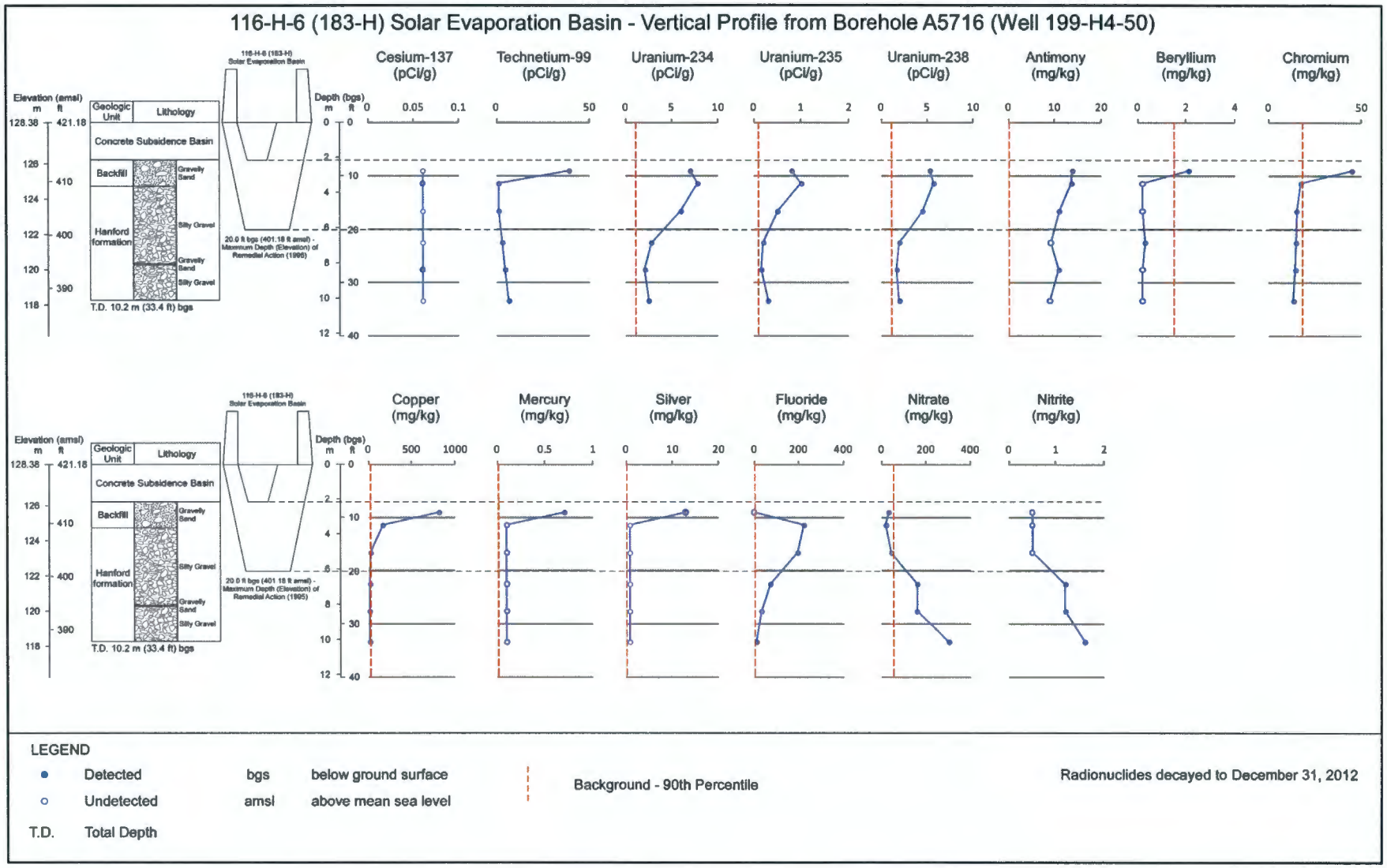

Figure 4-43. 116-H-6 Vertical Profiles of Contamination in RCRA Borehole A5716 (Well 199-H4-50)

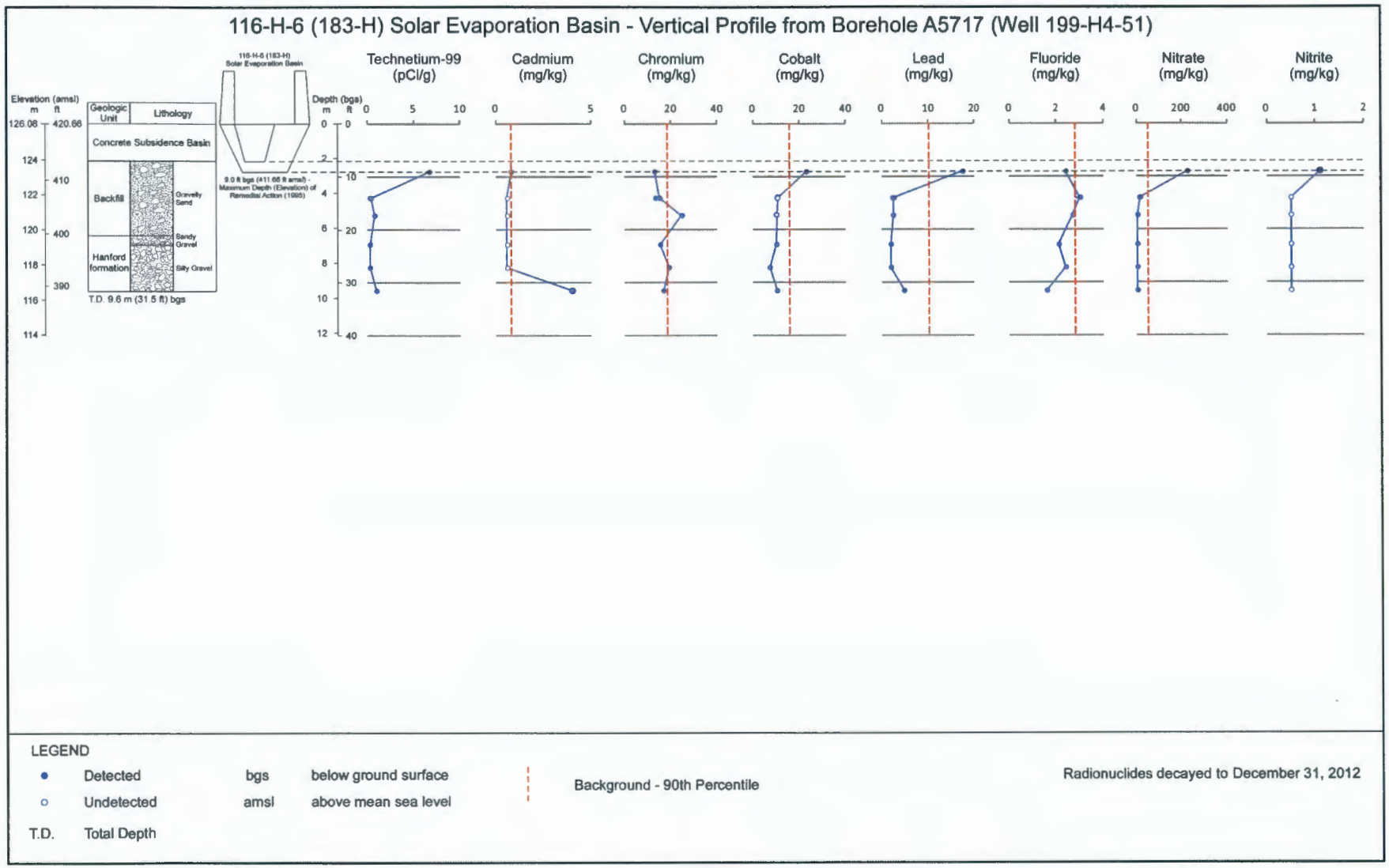

Figure 4-44. 116-H-6 Vertical Profiles of Contamination in RCRA Borehole A5717 (Well 199-H4-51)

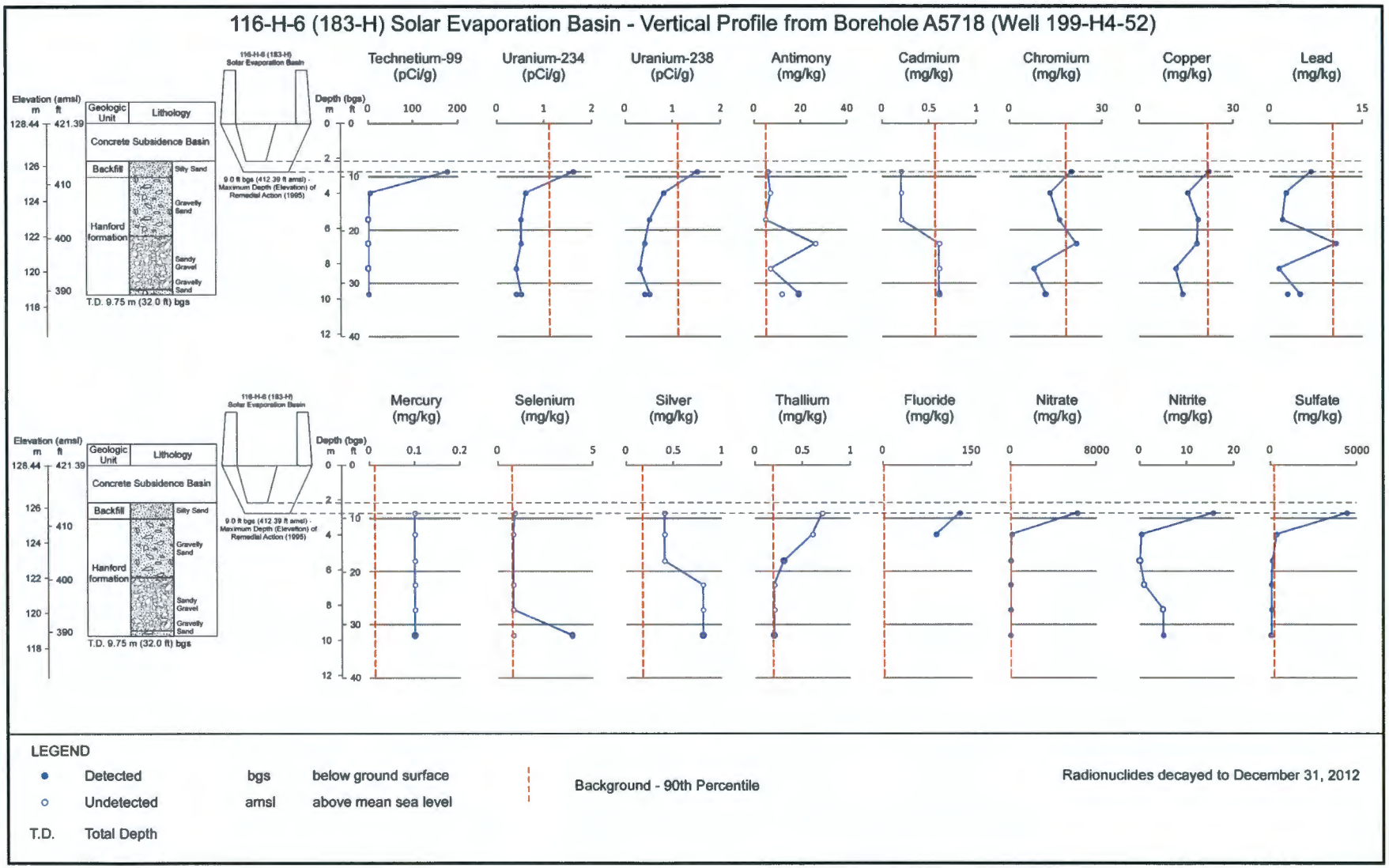

Figure 4-45. 116-H-6 Vertical Profiles of Contamination in RCRA Borehole A5718 (Well 199-H4-52)

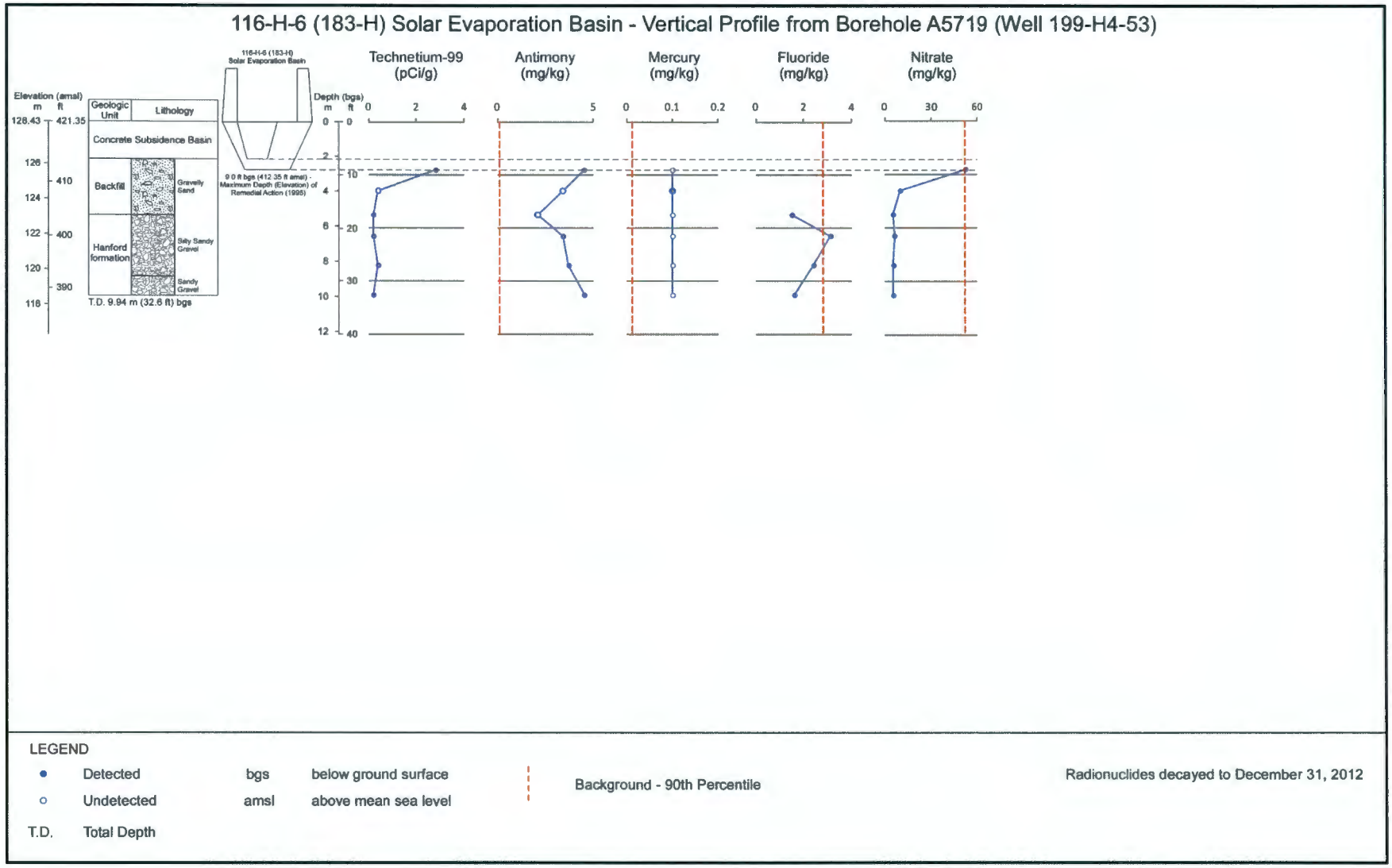

Figure 4-46. 116-H-6 Vertical Profiles of Contamination in RCRA Borehole A5719 (Well 199-H4-53)

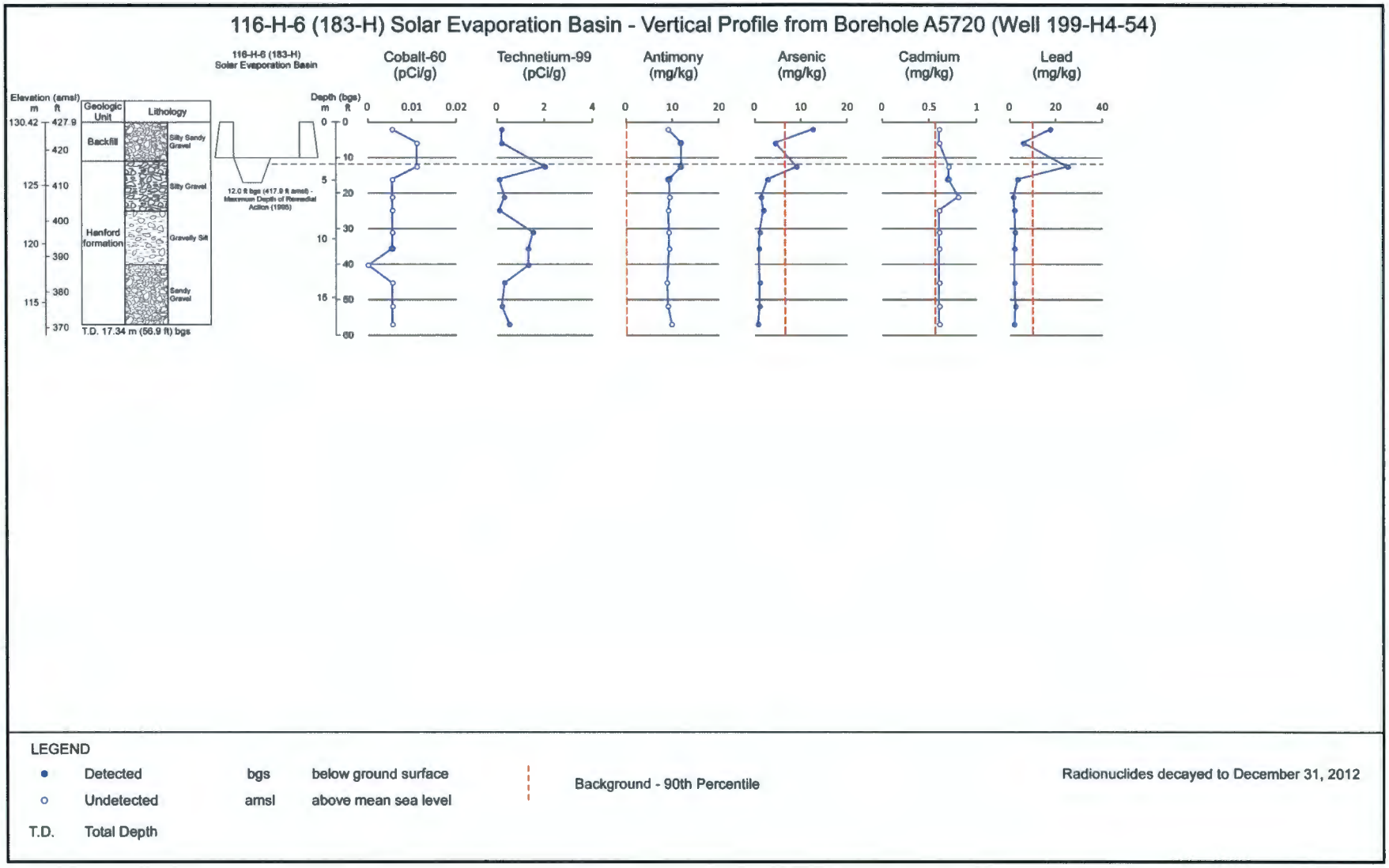

Figure 4-47. 116-H-6 Vertical Profiles of Contamination in RCRA Borehole A5720 (Well 199-H4-54)

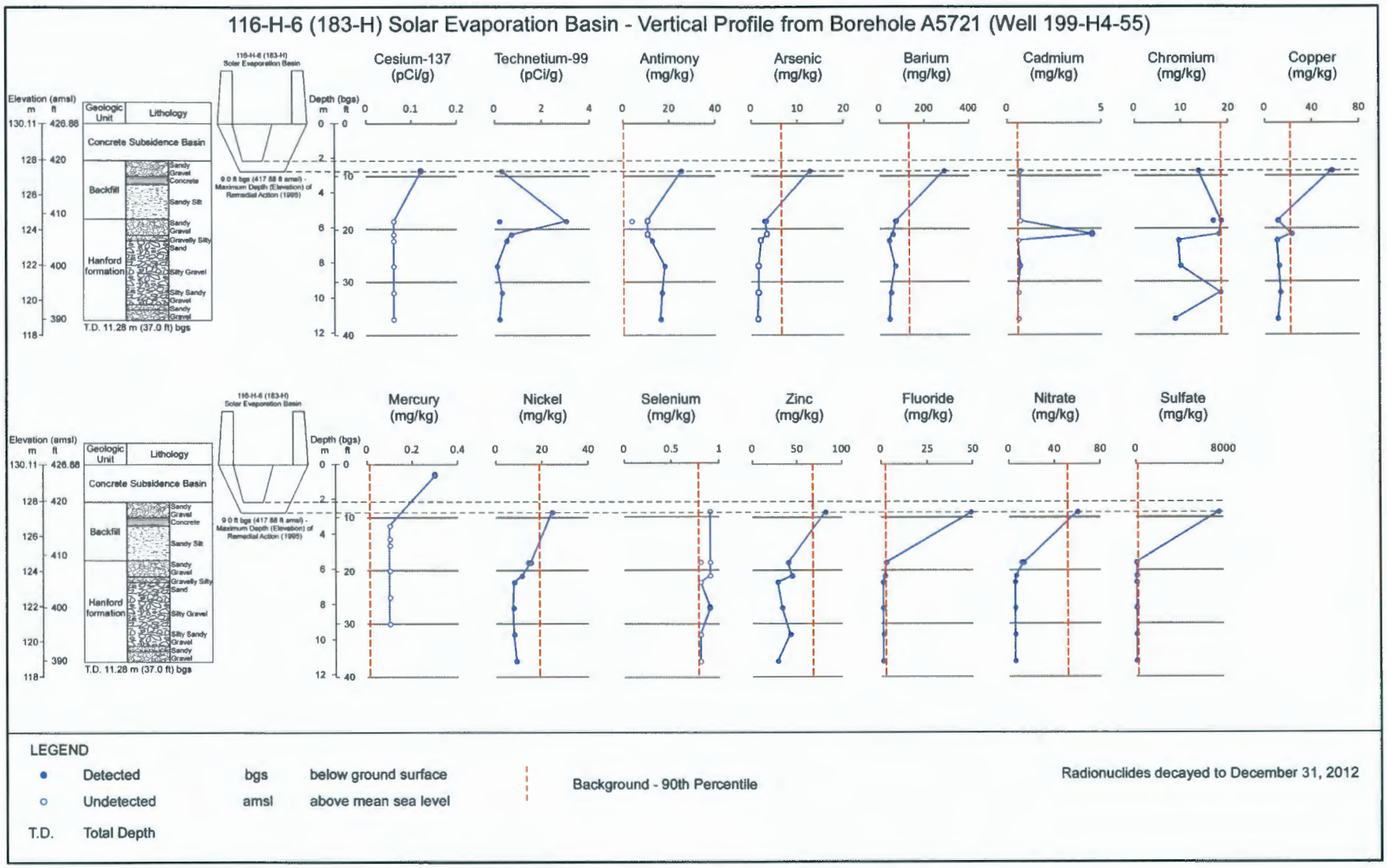

Figure 4-48. 116-H-6 Vertical Profiles of Contamination in RCRA Borehole A5721 (Well 199-H4-55)

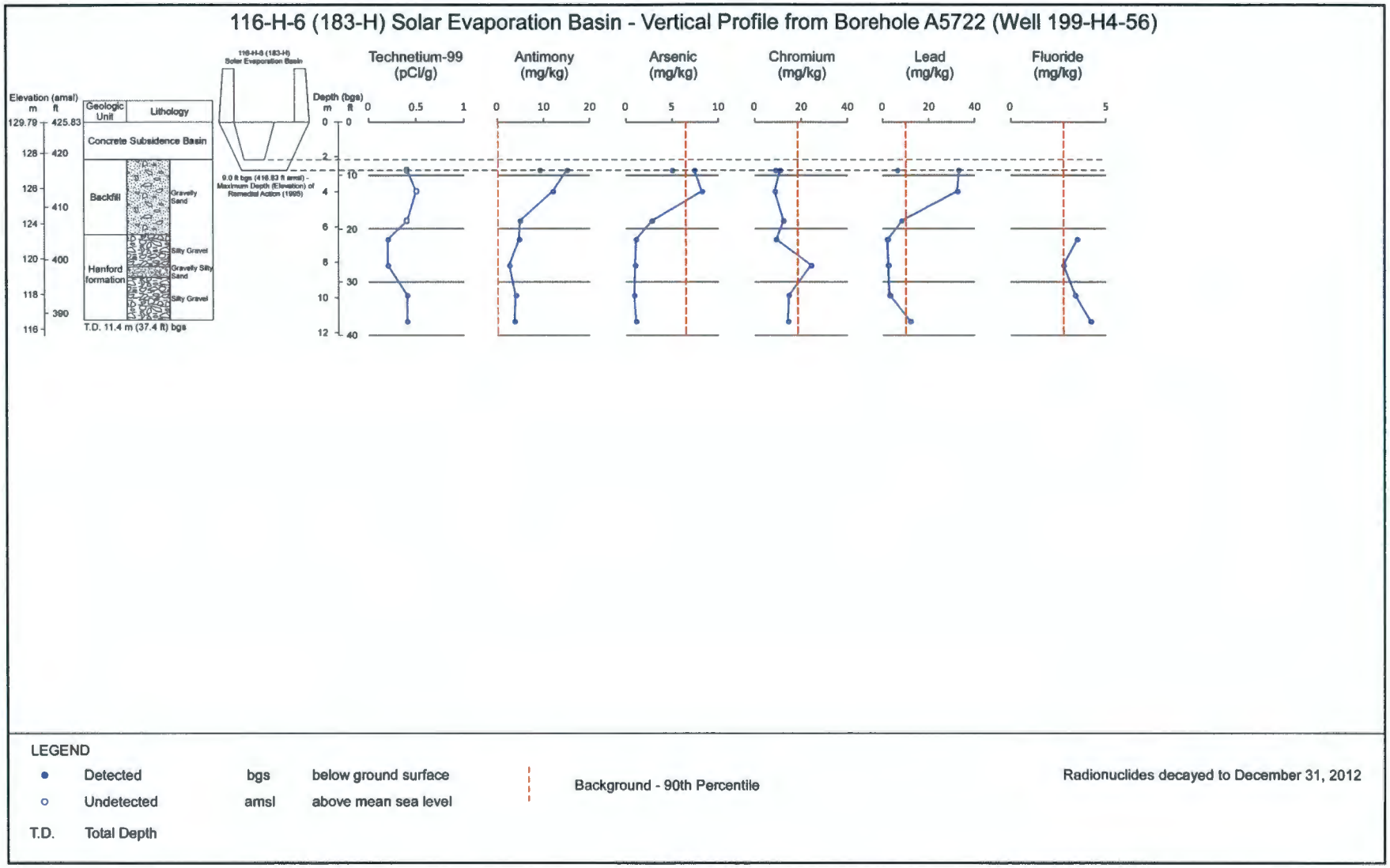

Figure 4-49. 116-H-6 Vertical Profiles of Contamination in RCRA Borehole A5722 (Well 199-H4-56)

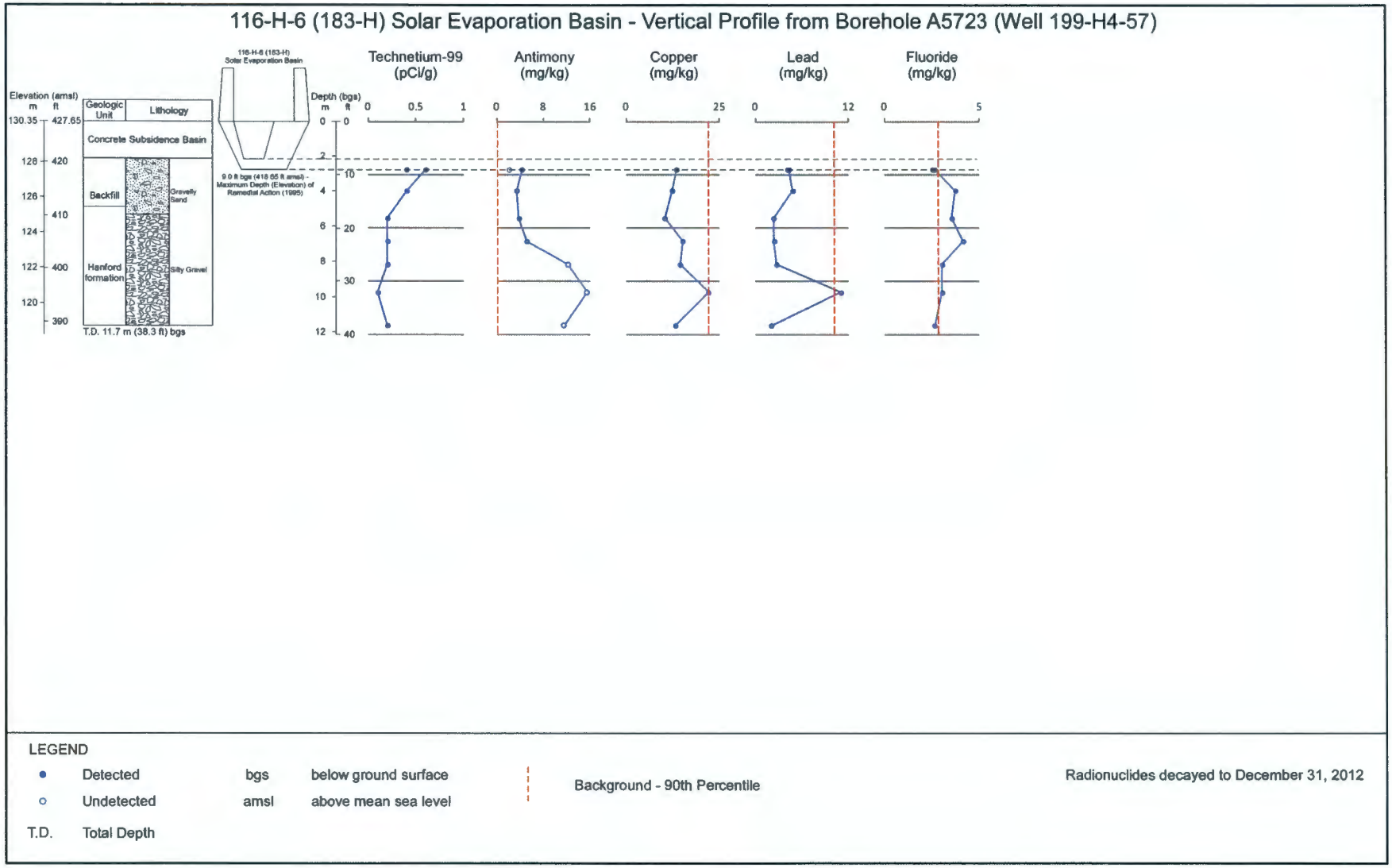

Figure 4-50. 116-H-6 Vertical Profiles of Contamination in RCRA Borehole A5723 (199-H4-57)

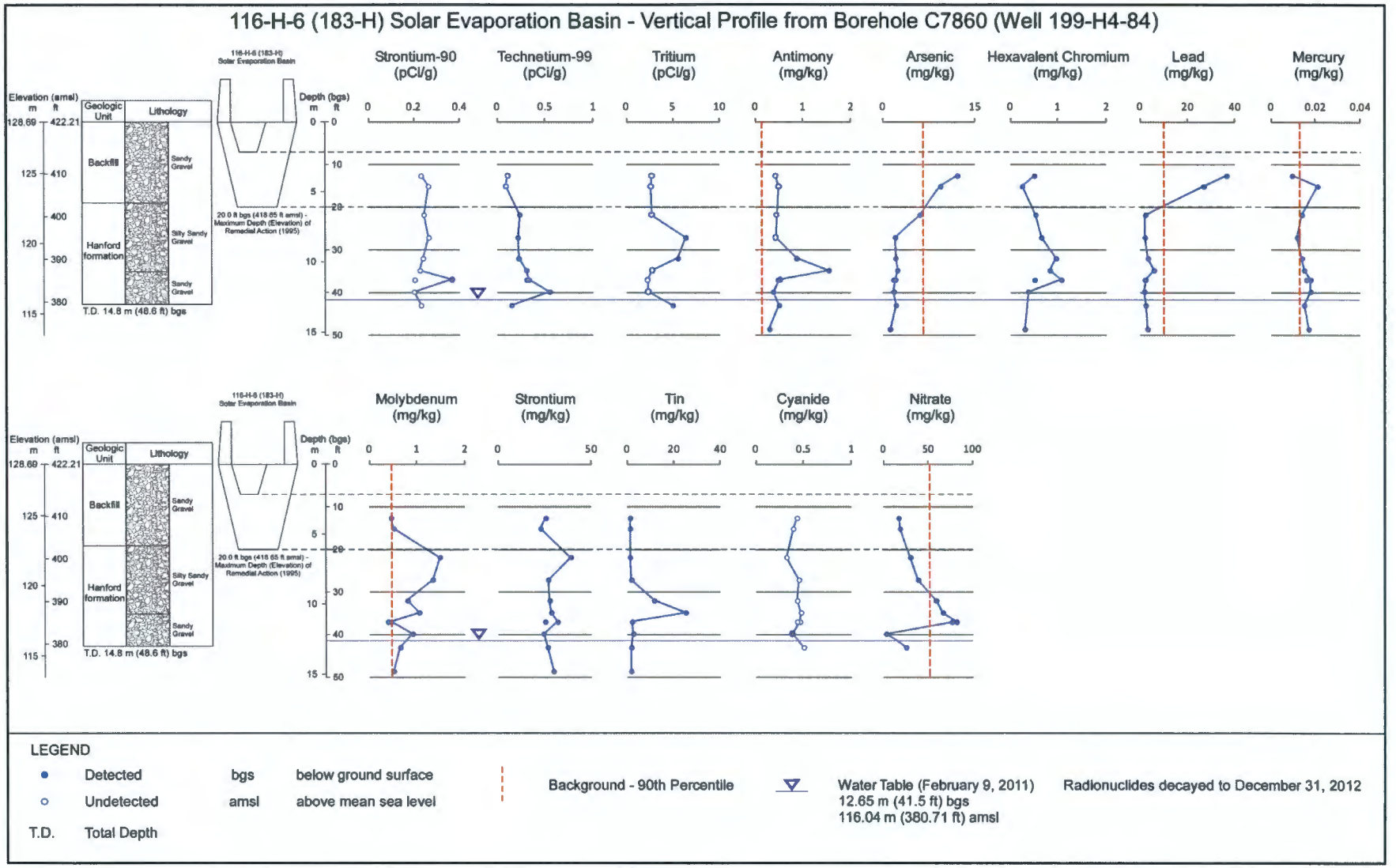

Figure 4-51. 116-H-6 Vertical Profiles of Contamination in Remedial Investigation Borehole C7860 (Well 199-H4-84)

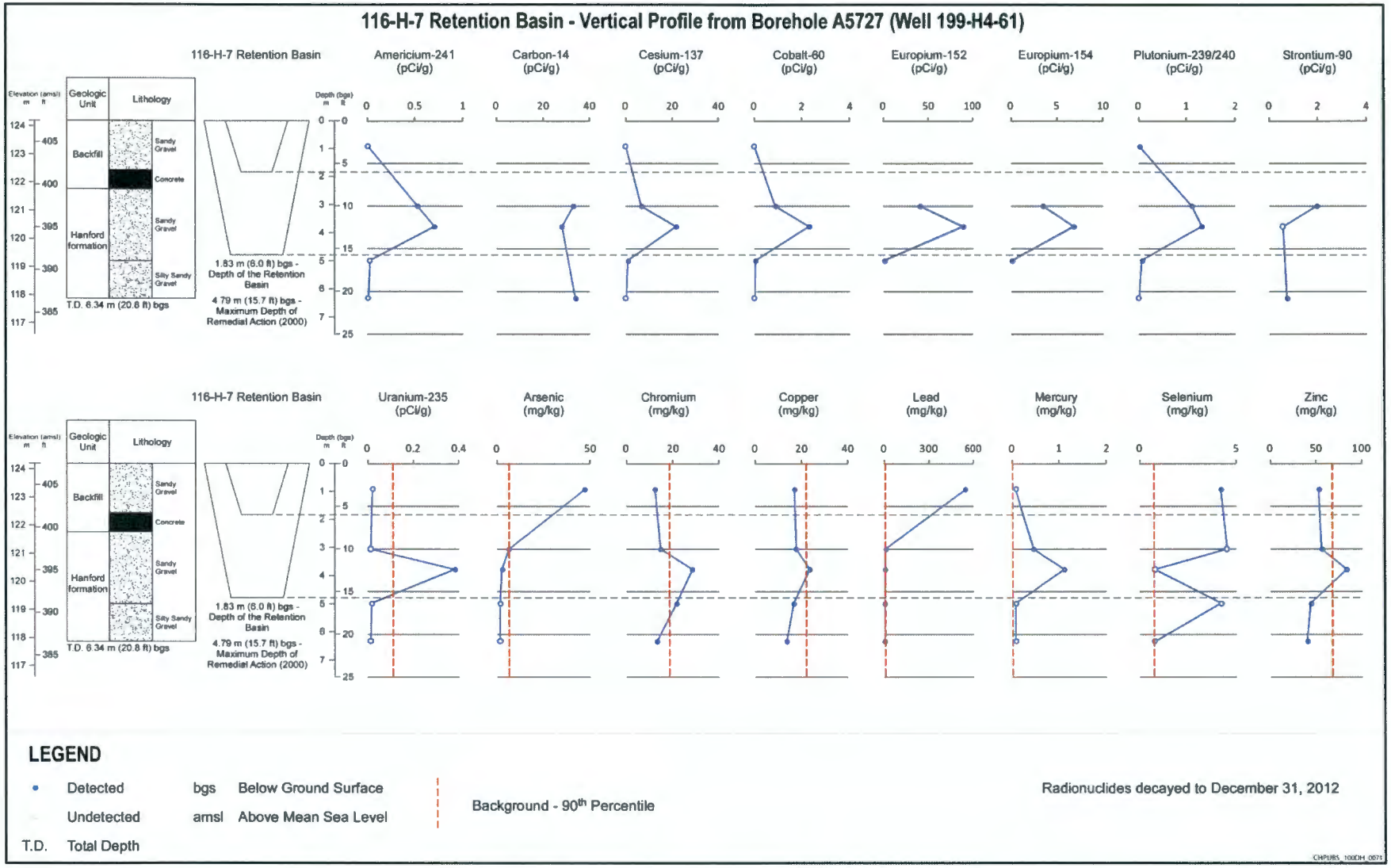

Figure 4-52. 116-H-7 Vertical Profiles of Contamination in LFI Borehole A5727 (Well 199-H4-61)

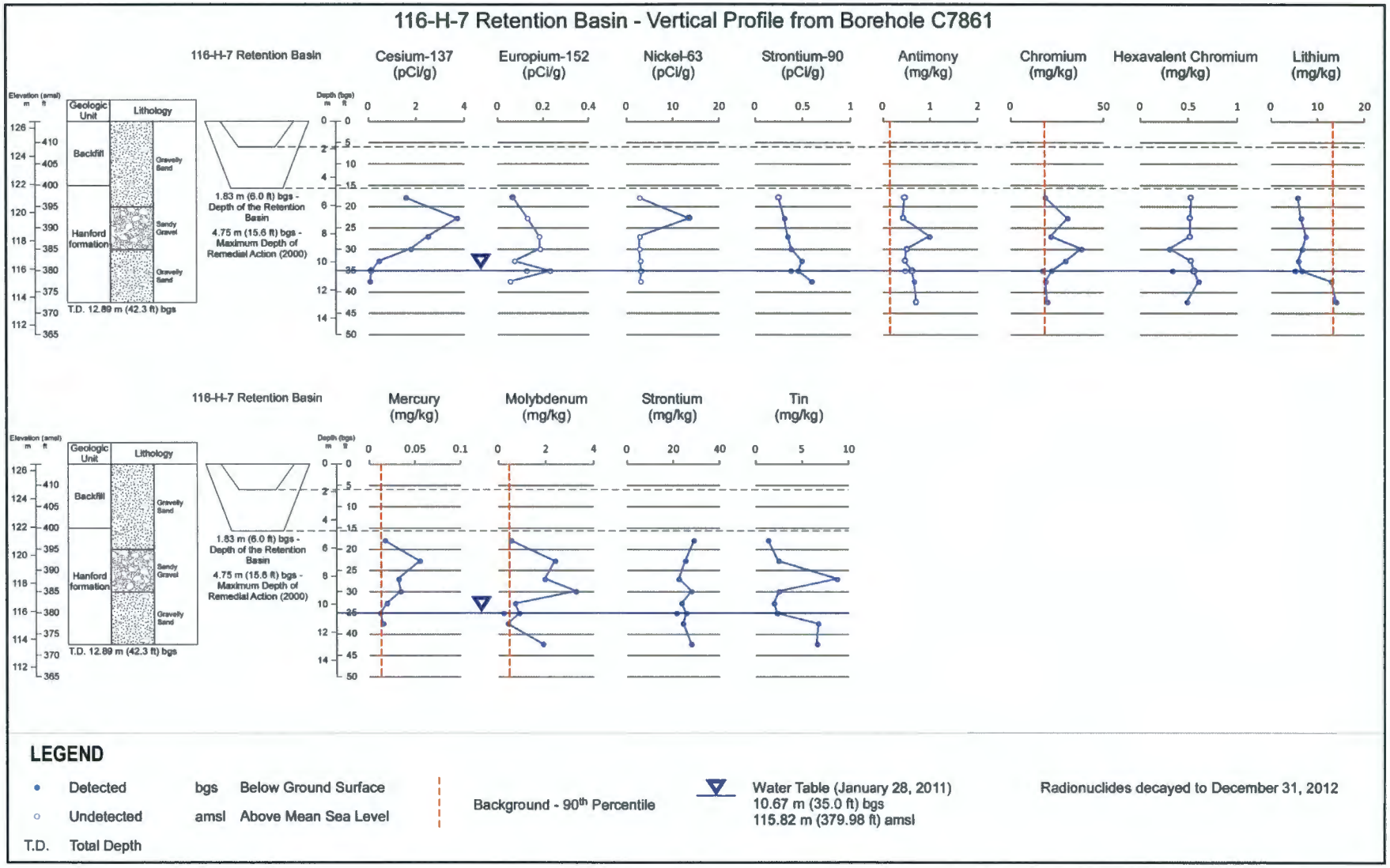

Figure 4-53. 116-H-7 Vertical Profiles of Contamination in Remedial Investigation Borehole C7861

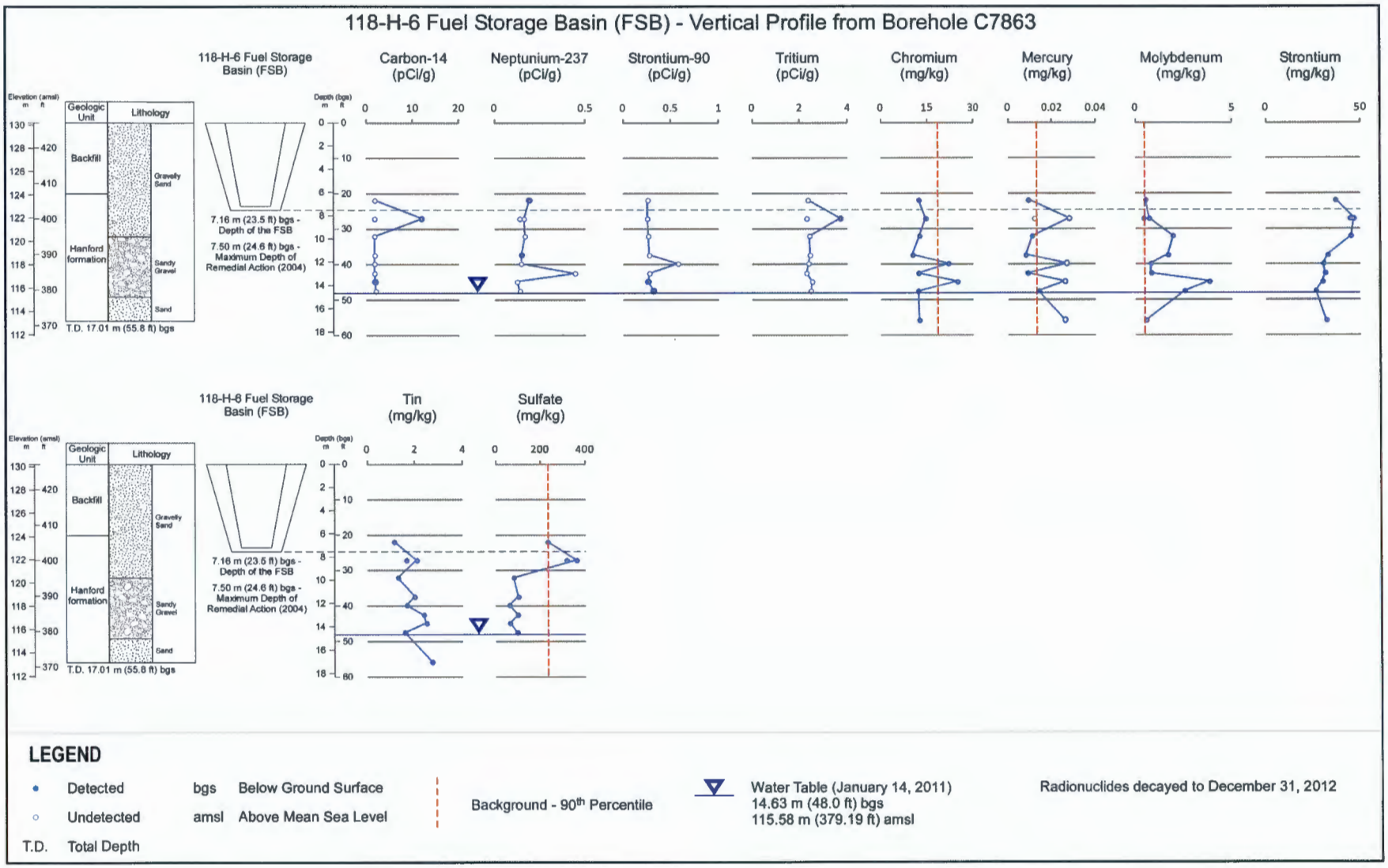

Figure 4-54. 118-H-6 Vertical Profiles of Contamination in Remedial Investigation Borehole C7863

4.3.18 *1607-H4 Septic System Characterization*

The 1607-H4 Septic System received sanitary sewage from the 181-H Pump House from 1948 to 1965. During interim remedial action, the site was excavated to 3.6 m (11.8 ft) bgs and elevated metal and PAH concentrations were detected in tank sludge and CVP samples collected during cleanup verification. In addition, the site is located in an area with a relatively shallow water table (8.8 m [28.9 ft] bgs).

An RI test pit was excavated through the trench location (Figure 4-55) and soil samples were collected and analyzed to evaluate the vertical extent of contamination to a depth of 5.6 m (19 ft). The CVP and RI data are summarized in Appendix D (Table 98). The RI borehole data for contaminants detected or present above background levels are presented in Figure 4-56 and 4-57.

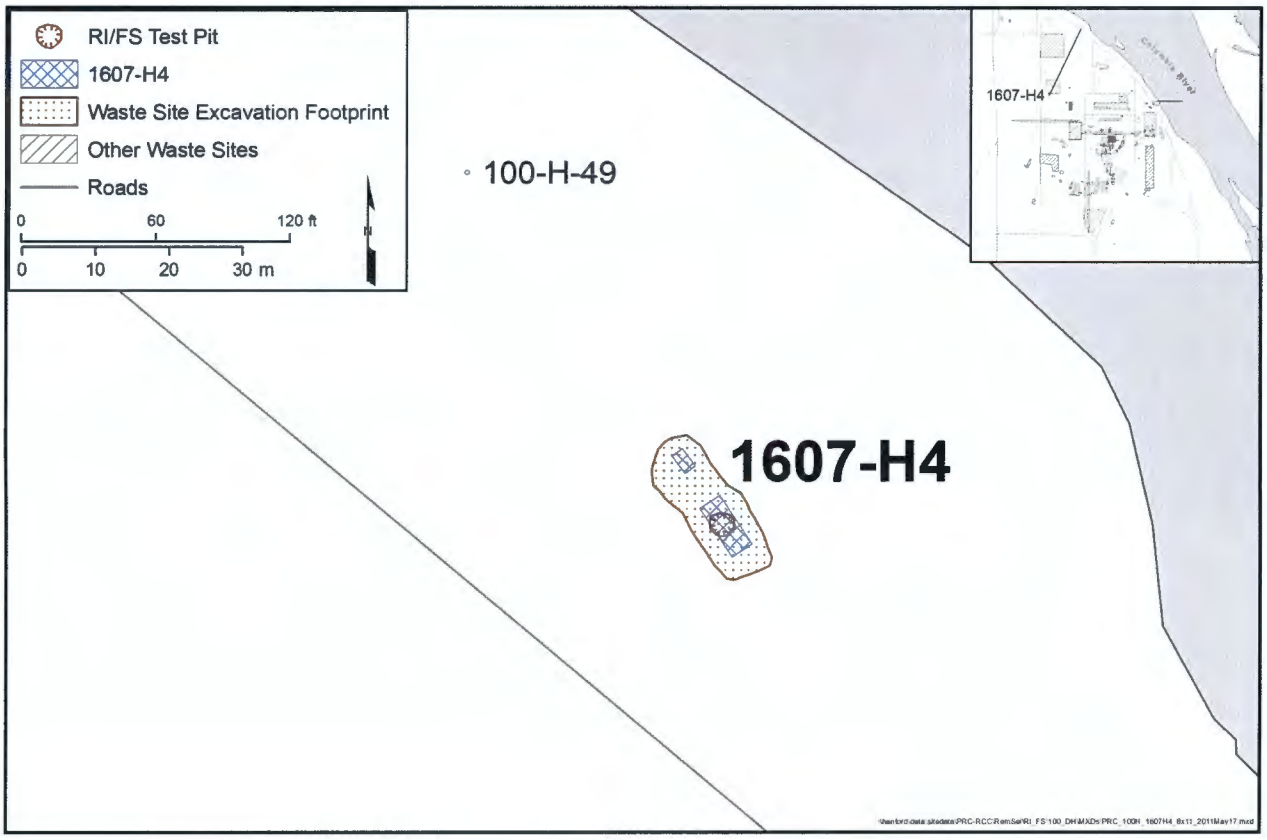

Figure 4-55. 1607-H4 Location Map

Between the CVP and RI sample results for this trench, 21 contaminants, including 15 PAHs, were detected or were present above background in the vadose zone. Contaminant concentrations decrease with depth, with the exception of lead. Lead concentrations increase with depth to 5.8 m (19 ft). Only Cr(VI) is detected in nearby groundwater wells in excess of drinking water standards (see Figures 4-66 for well locations). Chapter 5 compares soil concentrations detected or present above background to soil concentrations protective of groundwater and surface water (i.e., PRG, SSL). The RI data are also used in Chapter 6 for the human health risk evaluation.

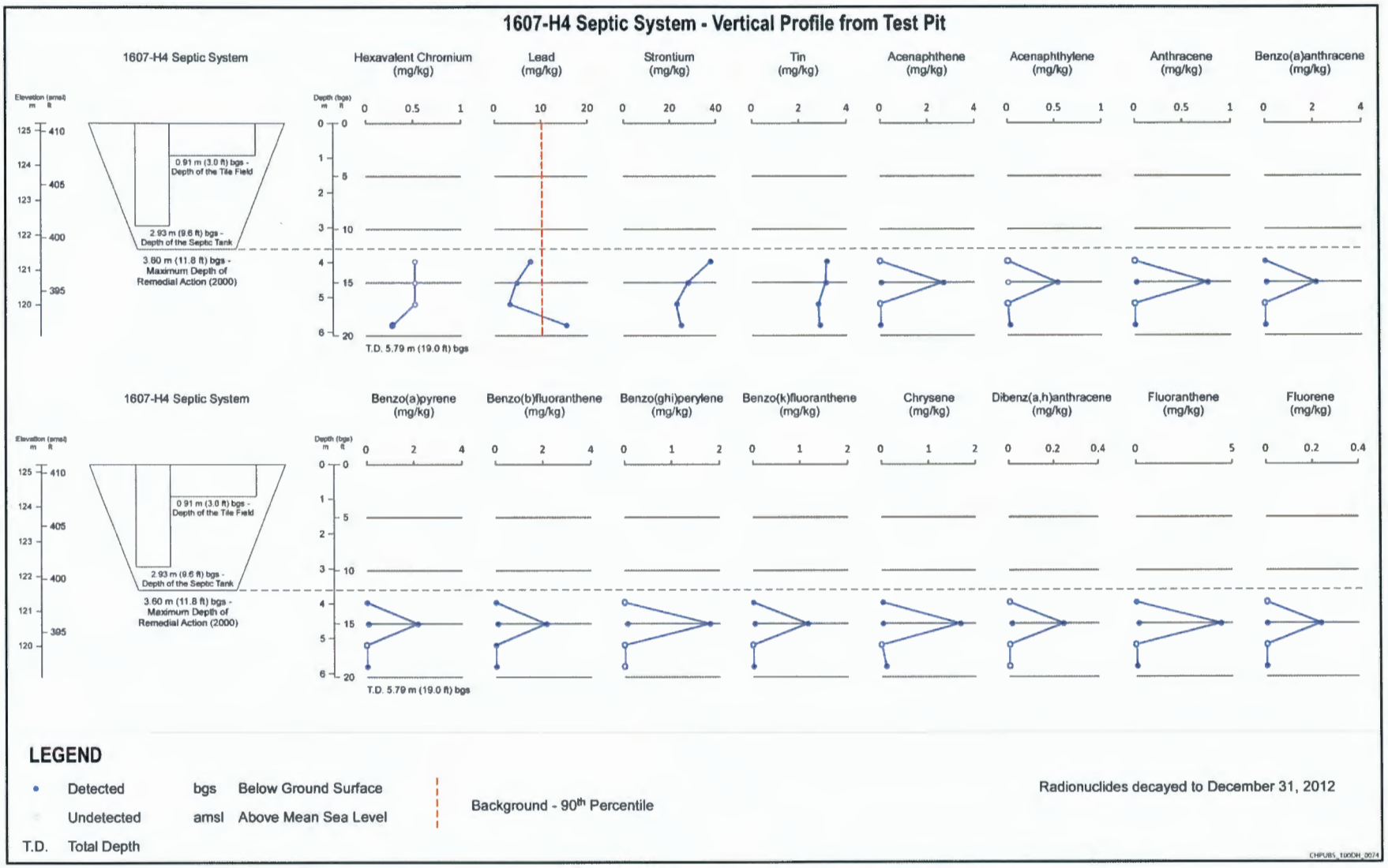

Figure 4-56. 1607-H4 Vertical Profiles of Contamination in Remedial Investigation Test Pit

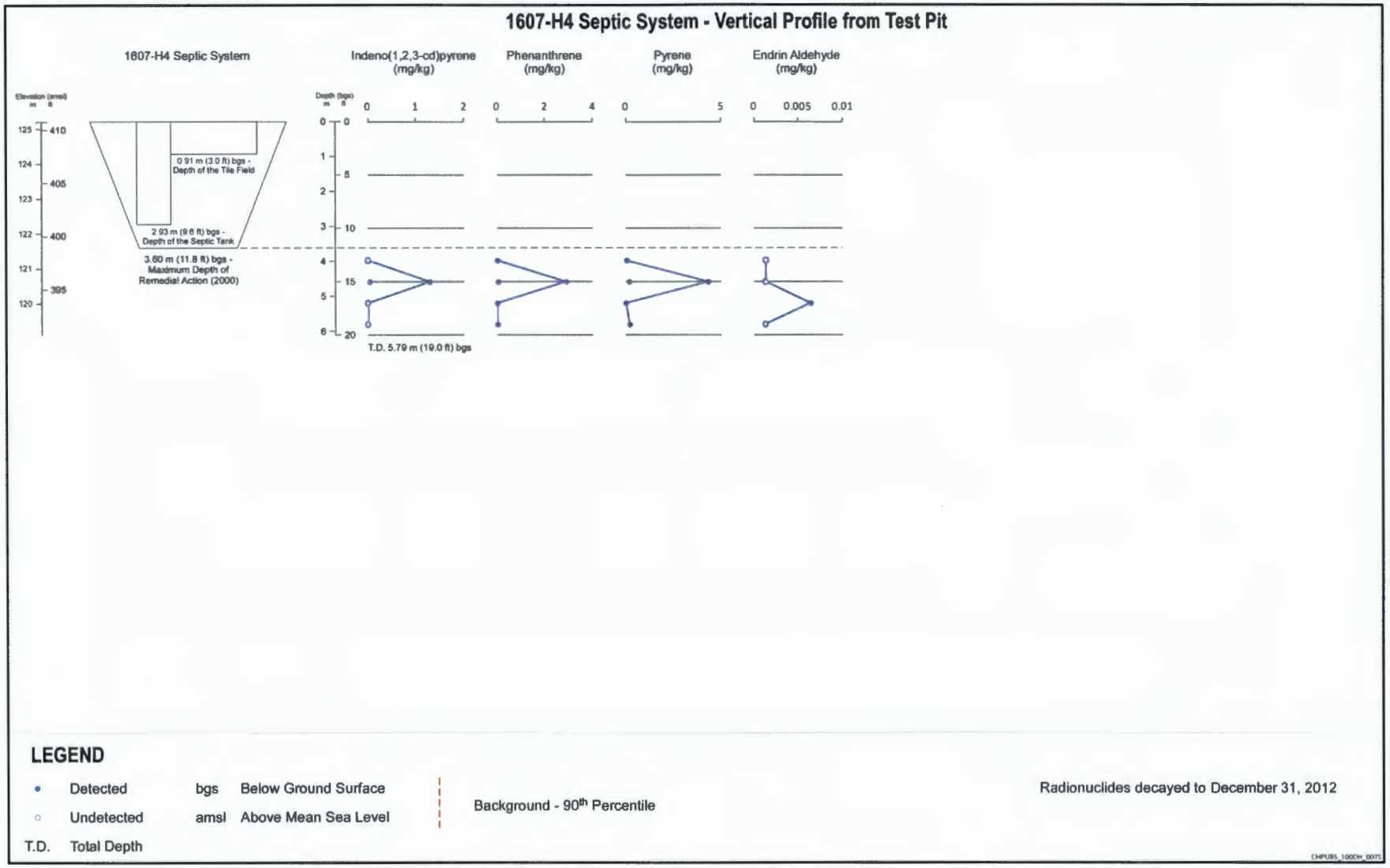

Figure 4-57. 1607-H4 Vertical Profiles of Contamination in Remedial Investigation Test Pit

4.3.19 New RI Well Soil and Sediment Sampling

This summary describes the nature and extent of vadose zone contamination detected or present above background and aquifer sediment and groundwater concentrations for the wells and boreholes identified in Table 4-4. The well locations are shown in Figures 2-2 and 2-3. Vadose zone soil and aquifer sediments were collected for each well within 1.5 m to 4.6 m (5 to 15 ft) of the water table to characterize contaminants associated with the groundwater-vadose zone interface. Groundwater grab samples were also collected from these locations. The Conceptual Site Model (Section 4.9) discusses contaminant interactions between the vadose zone and aquifer and Appendix D Tables 99 through 112 summarize the soil and aquifer sediment data. Vertical profiles of contaminants detected or present above background levels and not previously presented in Chapter 4 (e.g., see Figure 4-5, 100-D-12 profile) are presented in Appendix D (Figures 1 through 14).

Table 4-4. Identification of 100-D/H RI Wells with Vadose Zone and Aquifer Sediment Samples

Well Identification	Borehole Identification	RI Area, 100-D/H SAP (DOE/RL-2009-40) Well ID*
199-D3-5	C7620	100-D, Well 2
199-D5-133	C7621	100-D, Well 3
199-D5-132	C7622	100-D, Well 4
199-D6-3	C7623	100-D, Well 5
199-D5-140	C7866	100-D, Well 9
199-D5-143	C8375**	100-D, Replacement Well 9
199-D5-134	C7624	100-D, Well R4
199-D5-141	C7625	100-D, Well R5
199-D5-144	C8668**	100-D, Replacement Well R5
199-H3-6	C7626	100-H, Well 6
199-H3-7	C7627	100-H, Well 7
199-H6-3	C7628	100-H, Well 10
199-H6-4	C7629	100-H, Well 11
199-H1-7	C7630	100-H, Well 12
199-H3-9	C7639	100-H, Well R1
199-H3-10	C7640	100-H, Well R2
199-H2-1	C7631	100-H, Well R3

* Wells for 100-D and 100-H RI are identified and described in *Integrated 100 Area Remedial Investigation/Feasibility Study Work Plan Addendum 2: 100-KR-1, 100-KR-2, and 100-KR-4 Operable Units* (DOE/RL-2008-46-ADD2).

**Profiles for 199-D5-144 (100-D-12) and 199-D5-143 (100-D-56:1) are in Figures 4-5 and 4-7, respectively.

With few exceptions, low-levels of radioactive contamination were detected in soil and sediment samples adjacent to the water table. The maximum concentrations for cesium-137 (C7623), strontium-90 (C7624, C7626, and C7639), and tritium (C7625, C7626, C7627, and C7630) were 0.241 pCi/g, 0.906 pCi/g, and 18.6 pCi/g, respectively.

The results for nonradioactive contaminants detected or present above background concentrations in the new RI wells are summarized as follows:

- Boreholes C7624, C7625, C7626, and C7628 showed elevated total chromium, nickel, and molybdenum concentrations. Concentrations of chromium and nickel were highest in C7628 at 2,900 and 1,390 mg/kg, respectively (see Appendix D, Figure 8). Elevated concentrations of copper and cobalt were also present in the samples with the most elevated total chromium results.
- Hexavalent chromium was detected above and below the water table in boreholes C7620, C7621, C7623, C7624, C7626, C7628, C7629, C7640, and C7631. The maximum Cr(VI) concentration was 1.17 mg/kg in C7629.
- Barium was reported in one or more samples from C7623 and C7624 at a maximum concentration of 192 mg/kg.
- Thallium was detected in C7627 and C7630 at a maximum concentration of 0.278 mg/kg.
- A single detection of uranium (9.73 mg/kg) was reported just above the water table in C7626.
- Low-level detections of 2-hexanone and styrene were present in boreholes C7627 and C7629.
- Concentrations of strontium (metal) and tin were consistent with the results from other 100-D/H RI borehole samples.
- With a few exceptions, the metal detections in the groundwater sediment samples collected from 1.5 m (5 ft) into the unconfined aquifer were similar to those found in the vadose zone soils.

The maximum Cr(VI) and total chromium concentrations in the groundwater grab samples from these wells exceeded 1,400 ppb in 199-D5-134. This elevated concentration was located in the northern Cr(VI) groundwater plume.

4.3.20 Potentially Significant Cr (VI) Waste Sites Undergoing Active Interim Remediation

Interim remedial actions are presently being performed at several 100-D sites with known or potential Cr(VI) contamination. All these sites are associated with pre-reactor handling and use of concentrated sodium dichromate solutions and are identified in Chapter 1 on Figure 1-14. Interim remediation will continue at these sites, and is expected to be complete before issuance of a final action ROD. However, brief summaries of the current state and data for these sites, as of early September 2012, are provided to support an understanding of ongoing remediation at sites of particular potential relevance as sources of Cr(VI) contamination. Although a technical evaluation of future effect to groundwater has not been performed, one or more of these sites very strongly suggest a continuing source of aquifer contamination near the Cr(VI) groundwater plume. 100-D-100 appears to be the worst-case Cr(VI) site, based on concentrations observed in the deep vadose zone. Closeout verification data from these sites will be evaluated at the completion of interim remedial actions to verify protection of human health and the environment. Results from these and other accepted waste sites will be integrated into the final ROD as results are available.

100-D-73. The 100-D-73 waste site consists of the footprint of the former 108-D Building, where concentrated Cr(VI) solution was initially prepared during historical operations. Remediation of the site

has been performed up to 4.6 m (15 ft) bgs, including removal of stained concrete. Further excavation of this site is not anticipated, and no significant Cr(VI) inventory was identified during remediation. Elevated Cr(VI) concentrations in soil have been found only in the south-central portion of the site, with a maximum Cr(VI) concentration of 16.8 mg/kg identified at 3 m (10 ft) bgs. Total chromium concentrations in soil samples collected near this waste site are generally within typical Hanford Site background concentrations (18.5 mg/kg), with a maximum detected concentration of 19.8 mg/kg.

100-D-30. The 100-D-30 waste site addresses residual sodium dichromate contamination in soil and concrete rubble associated with the former 185-D facility sodium dichromate trench and mixing tanks. Stained soils and concrete were observed during initial remediation, with analytical sample results for Cr(VI) up to 108 mg/kg in soil samples collected from waste material. Initial remediation extended up to 3.5 m (12 ft) bgs at the location of a former sump in the pipe trench. Further subsurface characterization in the sump area detected slightly elevated Cr(VI) and total chromium concentrations at depth (*Report on Investigation of Hexavalent Chromium Source in the Northern 100-D Area* [DOE/RL-2010-40]). Based on these detections, additional remediation has been performed in the sump area, currently extending to a maximum depth of approximately 15 m (50 ft). Cr(VI) concentrations up to 113 mg/kg (at approximately 10.7 m [35 ft] bgs) have been observed during further remediation, and additional material removal is planned.

100-D-104. The 100-D-104 waste site addresses an area of vadose zone contamination, including Cr(VI), discovered immediately southeast of the former 185-D Building and 100-D-30 waste site. Upon discovery, contaminated soil was initially removed to a depth of approximately 6 m (20 ft). Multiple different colors of soil staining were still observed at this depth, and separate analytical samples showed a range of contaminant concentrations: Cr(VI) was quantified between 0.24 and 286 mg/kg; total chromium was quantified between 2.8 mg/kg and 303 mg/kg; sulfate results ranged from undetected to 4,590 mg/kg. The disparity in the nature of the staining within a small area is suggestive of multiple historical releases. The most likely source of the contamination is a former acid neutralization French drain located at nearly the exact location of the staining, which would account for the elevated sulfate levels observed in some samples. An external sodium dichromate storage tank was also located immediately nearby and may have had releases to the drain or immediate vicinity.

Additional subsurface characterization was performed at the locations of the acid neutralization French drain and the sodium dichromate storage tank (*Report on Investigation of Hexavalent Chromium Source in the Northern 100-D Area* [DOE/RL-2010-40]). No significant Cr(VI) was detected beneath the former storage tank, with a maximum result of 0.25 mg/kg at approximately 7.5 m (25 ft) bgs. Total chromium was detected above background levels at up to 112 mg/kg at a total depth of approximately 5 m (20 ft), decreasing to 28.1 mg/kg at a depth of approximately 7.5 m (25 ft). Higher contamination levels were observed in samples collected beneath the former French drain, with results of up to 14.2 mg/kg Cr(VI) (at a total depth of approximately 6.5 m [21 ft]) and 265 mg/kg total chromium (at a total depth of approximately 9 m [30 ft] bgs). Cr(VI) was detected up to the maximum depth characterized, with 0.38 mg/kg at the deepest interval (approximately 19 m [62 ft] bgs). Based on these data, the 100-D-104 site has thus far been remediated to a total depth of approximately 15 m (50 ft). Two separate areas of visual staining are apparent at this depth, and additional material removal is planned.

100-D-100. The 100-D-100 waste site addresses an area of stained soil discovered adjacent to the former railroad spur servicing the 183-DR Head House. The stained area is also near the former railcar unloading station (100-D-12 waste site), but on the opposite (southern) side of the former railroad junction. Initial surficial sampling at the stained area showed up to 2,110 mg/kg of Cr(VI) present. However, at 0.3 m (1 ft) bgs, the Cr(VI) concentration decreased significantly (87 mg/kg), with a corresponding total

chromium concentration of 150 mg/kg. A higher proportion of silt was observed in this sample relative to other shallow samples collected.

Initial characterization of soil at the 100-D-100 waste site extended to 4.6 m (15 ft) bgs, where the maximum Cr(VI) and total chromium detections were 17.6 mg/kg and 31 mg/kg, respectively. Sulfate concentrations above Hanford Site background (up to 920 mg/kg) were quantified in several of the samples, suggesting that sulfuric acid may also have been released at this location. Remediation of this site has extended to approximately 15 m (50 ft) bgs thus far, and has revealed significant visual staining and soil contamination. Hexavalent chromium has been detected at up 709 mg/kg, and remediation is planned to continue to additional depth to remove Cr(VI)-contaminated soil.

100-D-77. The 100-D-77 waste site consists of the footprint of the former 183-DR facility, used for water treatment for the 105-DR Reactor, including handling, storage, and injection of sodium dichromate. Remediation has included both the former head house and yard area, where sodium dichromate and sulfuric acid solutions were stored, and the sample room area, where sodium dichromate was injected into cooling water. Remediation has extended to a maximum depth of 8.5 m (28 ft) bgs in the head house area, and no further excavation is expected at either area. No significant residual Cr(VI) inventory was identified during remediation. Cr(VI) was detected in residual structural concrete components with up to 7.7 mg/kg in a former acid trap. Stained soils have been observed, but the highest soil Cr(VI) concentration detected thus far was 2.38 mg/kg, with a corresponding total chromium concentration of 59.7 mg/kg. Remediation was driven primarily by removal of subgrade structural components and mercury contamination in soil above interim action RAGs. Mercury was likely present as a result of spills of contaminated sulfuric acid.

4.3.21 RI Well Soil and Sediment Sampling Summary

This summary describes the nature and extent of vadose zone contamination above background and aquifer sediment and groundwater concentrations for the wells identified in Table 4-4. The wells locations are shown in Figures 2-2 and 2-3. Vadose zone soil and aquifer sediments were collected each within 1.5 m to 4.6 m (5 to 15 ft) of the water table to characterize contaminants associated with the groundwater vadose zone interface. Groundwater grab samples were also collected from these locations. These data are used to describe contamination associated with the PRZ. Contaminant interactions between the vadose zone and aquifer are discussed in Section 4.3.5 Periodically Rewetted Zone. Summary data on soil and aquifer sediment are presented in Appendix D Tables 27 through 40. Appendix D also provides vertical profiles of contaminants detected above background and not previously presented (e.g., see Figure 4-5, 100-D-12 profile).

With exception, low-levels on contamination above background were detected in soil and sediment samples adjacent to the water table. Cesium-137 was detected in C7623 at a maximum of 0.241 pCi/g. Strontium-90 was detected in three boreholes (C7624, C7626, and C7639) at a maximum concentration of 0.906 pCi/g. Tritium was detected in four boreholes (C7625, C7626, C7627, and C7630) at a maximum concentration of 18.6 pCi/g.

Samples in several boreholes (C7624, C7625, C7626, and C7628) showed elevated chromium nickel and molybdenum concentrations. Concentrations of chromium and nickel were highest in C7628 at 2,900 and 1,390 mg/kg, respectively (see Appendix D, Figure 8-C7628 profile). Elevated concentrations of copper and cobalt were also present in samples with the most elevated chromium results. Chapter 5 evaluates the impact to groundwater of these elevated concentrations detected above background. Cr(VI) was detected in several boreholes (C7620, C7621, C7623, C7624, C7626, C7628, C7629, C7640, and C7631) both above and below the water table. The maximum Cr(VI) concentration was 1.17 mg/kg in C7629. Barium above background was reported in one or more samples from C7623 and C7624 at a maximum

concentration of 192 mg/kg. Thallium was detected in C7627 and C7630 at a maximum concentration of 0.278 mg/kg. A single detection of uranium (metal) at 9.73 mg/kg was reported just above the water table in C7626. Very low-level detections of 2-hexanone and styrene were reported in samples from boreholes C7627 and C7629. Concentrations of nonradiological analytes for which a background value is not available (for example, strontium and tin) are consistent with concentrations measured in other 100-D/H RI samples.

With a few exceptions, the concentrations of metals seen in the sediment samples collected from 1.5 m (5 ft) into the unconfined aquifer were similar to those found in the vadose zone soils. Visually, the exceptions can be observed in the various profiles presented in Appendix D. The maximum Cr(VI) and chromium concentrations in groundwater from these wells exceeded 1,400 ppb in 199-D5-134. The elevated concentrations are located within the northern Cr(VI) groundwater plume. Additional discussions of current and future impacts to groundwater are presented later in Chapter 4 and in Chapter 5, respectively.

4.3.22 RPO Soil Sampling Summary to Support RI/FS

Soil samples were collected from 9 of 70 RPO boreholes (199-D4-89, 199-D5-128, 199-D7-5, 199-D7-6, 199-D8-89, 199-H1-2, 199-H1-35, 199-H1-36, and 199-H1-4) and analyzed for select radionuclides, metals, and physical properties. The RPO wells are shown on Figure 4-58. In addition, samples were collected from the RUM surface and submitted for Cr(VI) analysis. RPO locations were also selected to provide increased spatial distribution for data collection, particularly within the RUM. The data provide additional information for physical and hydrogeologic parameters to support possible future fate and transport evaluations, particularly beneath the unconfined aquifer.

Several metals and one radionuclide (strontium-90 as total beta radiostrontium) were detected above the 90th percentile of established background concentrations (see Table 4-1). Table 4-5 presents a summary of analytes that were detected. The detection limit for antimony is greater than the background concentration of 0.13 mg/kg. Antimony was detected in only two samples. Silver concentrations were not detected above background concentrations; however, the detection limit is slightly above background in eight samples. Boron was detected above background in all but one sample. The detections are not included in Table 4-5 because they are flagged as estimated values as a result of interference.

During RPO drilling activities, one soil sample was collected from each borehole within the RUM for laboratory permeameter testing and for Cr(VI) analysis. The samples are used to determine whether Cr(VI) is leaching out of the RUM as a long-term continuous source and whether the RUM is an effective aquitard for the unconfined aquifer beneath 100-D/H. The permeameter testing results are discussed in Chapter 3, Section 3.6. The analytical results for Cr(VI) are presented in Appendix D, Table 70. The laboratory detected Cr(VI) in only one soil sample from the RUM (199-H1-7). However, this sample had a laboratory "B" qualifier, which indicates it was below the laboratory quantitation limit.

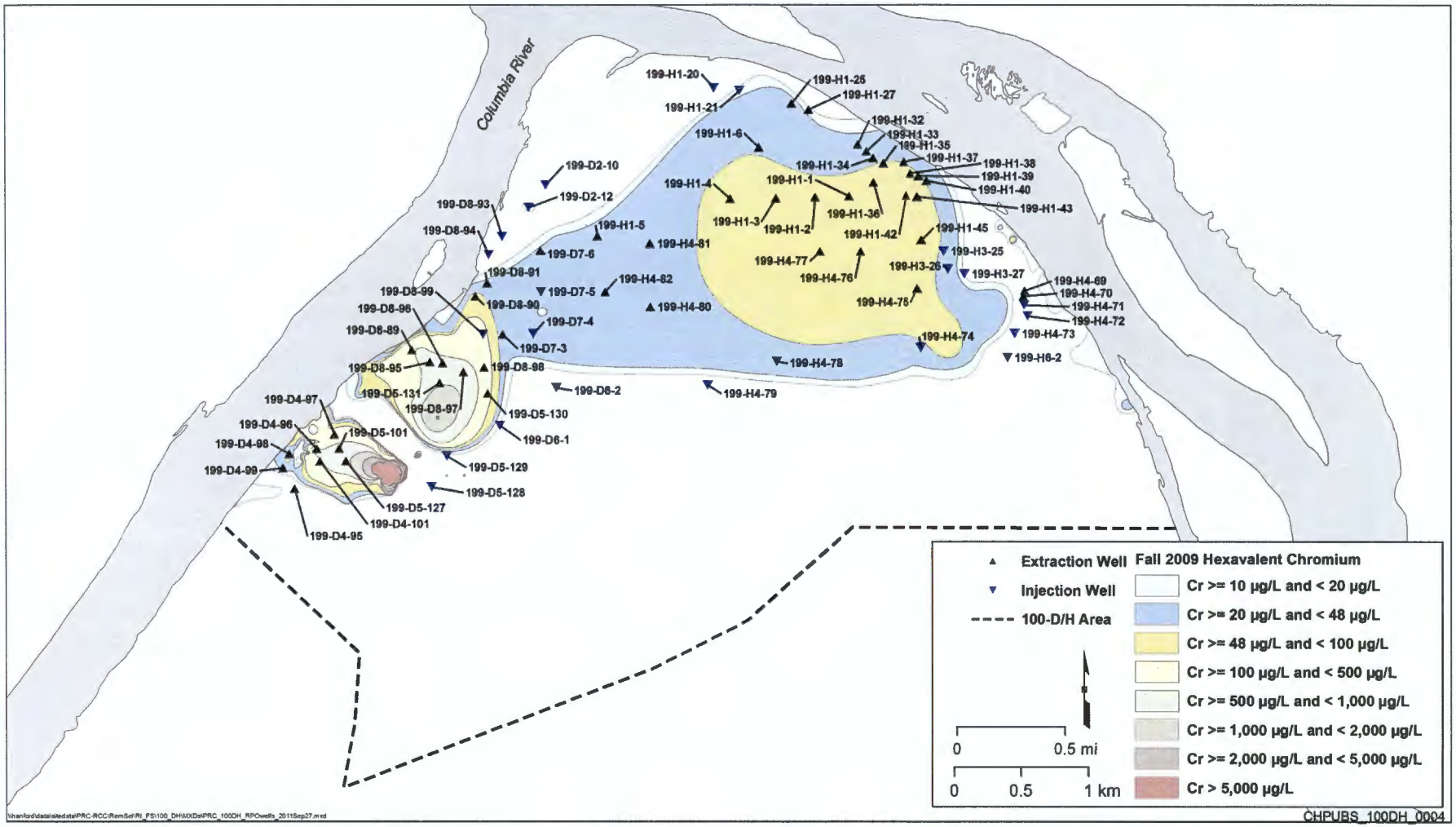

Figure 4-58. Location of RPO Wells in 100-D/H

Table 4-5. Analytical Samples from RPO Wells

Well ID	Sample ID	Sample Bottom Depth (m)	Sample Bottom Depth (ft)	Analyte	Result (in mg/kg unless noted)
199-D4-96	B22HT8	29.26	96	Chromium	19.3
	B22HT7	31.09	102	Lead	15.5
	B22HT5	25.91	85	Molybdenum	1.24
	B22HV0	32.49	106.6	Total beta radiostrontium	2.4 (pCi/g)
199-D7-5	B23RD8	19.35	63.5	Chromium	24.3
	B23RD8	19.35	63.5	Lead	10.6
	B23RD8	19.35	63.5	Lithium	15.5
	B23RD4	12.19	40	Molybdenum	0.48
	B23RD5	14.05	46.1	Molybdenum	0.61
	B23RD6	15.45	50.7	Molybdenum	0.74
	B23RD8	19.35	63.5	Nickel	20.6
	B23RD5	14.05	46.1	Selenium	1.08
	B23RD6	15.45	50.7	Selenium	1.16
	B23RD8	19.35	63.5	Selenium	1.89
199-D7-6	B244W2	7.50	24.6	Antimony	1.43
	B244W2	7.50	24.6	Chromium	655
	B244W2	7.50	24.6	Copper	95
	B244W5	13.26	43.5	Manganese	654
	B244W2	7.50	24.6	Manganese	970
	B244W3	10.79	35.4	Molybdenum	0.99
	B244W2	7.50	24.6	Molybdenum	147
	B244W2	7.50	24.6	Nickel	78.2
	B244W5	13.26	43.5	Selenium	0.96
	B244W3	10.79	35.4	Selenium	1.12
	B244W2	7.50	24.6	Selenium	1.39
	B244W5	13.26	43.5	Thallium	0.21

Table 4-5. Analytical Samples from RPO Wells

Well ID	Sample ID	Sample Bottom Depth (m)	Sample Bottom Depth (ft)	Analyte	Result (in mg/kg unless noted)
199-D8-89	B22HX1	19.54	64.1	Chromium	203
	B22HX1	19.54	64.1	Molybdenum	3.07
	B22HX1	19.54	64.1	Nickel	106
199-H1-2	B24DF0	16.61	54.5	Selenium	1.32
199-H1-35	B22HY0	14.63	48	Barium	138
199-H1-36	B23511	14.17	46.5	Barium	160
	B23511	14.17	46.5	Manganese	709
	B23508	11.83	38.8	Molybdenum	0.56
	B23509	14.17	46.5	Molybdenum	0.61
	B23511	14.17	46.5	Selenium	0.85
	B23506	9.60	31.5	Selenium	0.9
	B23509	14.17	46.5	Selenium	1.43
	B23508	11.83	38.8	Selenium	1.92
199-H1-4	B24DF1	14.97	49.1	Manganese	542
	B24DF1	14.97	49.1	Selenium	0.94

4.3.23 Evaluation of Water Addition to Wells and Boreholes during Sampling

The wells and boreholes installed during the RI field activities at 100-D/H were drilled using the cable tool method or using Foremost AP-1000 diesel-percussion-hammer drill rigs, commonly referred to as Becker Hammer rigs (*Borehole Summary Report for the Installation of 16 Resource Protection Wells in the 100-HR-3 Groundwater Operable Unit in Support of the Integrated 100 Areas RI/FS: 100-D/H Decisional Unit* [SGW-49912]), which is standard practice at the Hanford Site. Periodically, water was added to the hole to allow removal of drill cuttings from the dry, unconsolidated sediments of the Hanford formation and Ringold Formation unit E. The intent is to provide sufficient water for removing cuttings and advancing the borehole without disturbing the underlying material, which is being tested for a variety of mobile and immobile contaminants. An analysis was conducted of the effects to the representativeness of the RI characterization samples (*Data Quality Evaluation of Vadose Zone Soil Sampling Data Collection During RI Drilling for the 100 Area Operable Units* [ECF-100KR4-11-0166]). Typically, one gallon (0.13 ft^3) of water was sufficient to provide some cohesion to the cuttings, allowing the sample to be retrieved. However, occasionally 5 to 20 gal (0.67 to 2.67 ft^3) were used. Most of the additions were completed at least 0.6 m (2 ft) above the planned split spoon sample interval.

Water was added during drilling at 19 of the 27 wells and borings drilled during the RI. Seven of these locations have a subset of samples that potentially could have impacts to the characterization of mobile constituents ranging from one to six of the dozen or more samples collected at each location. Three

locations (199-D3-5, 199-D5-134, and 199-D5-143) were affected to the extent that four to six of the split spoon samples collected might have received some impact.

At these locations, the analytical results for mobile contaminants, such as Cr(VI), were evaluated further. Sample results from the entire thickness of the vadose zone were slightly above or below the detection limits in each of the boreholes with potentially impacted samples. The results were consistent regardless of the addition of water during drilling. This indicates that the sample results were not affected.

Evaluation of mobile constituent data from a large number of boreholes does not reveal any particular trends. There are occasional changes up to plus or minus 0.5 mg/kg for Cr(VI), which may be a result of some redistribution during drilling or more likely reflects the actual distribution with depth. Variations appear to be within the bounds of measurement error. The data from sample intervals with added water were consistent with data from internals above and/or below the interval. These observations suggest that the samples provide us with a reasonable and representative estimate of subsurface conditions.

In conclusion, the intent of the drilling was to provide representative samples for physical property and contaminant analysis. Occasionally, the addition of water was required to provide either additional density for the drilling air in the Becker Hammer or cohesion to remove cuttings using the cable tool method so the drilling could progress. Review of the vadose zone conditions indicates that the large matric potentials will tend to wick water preferentially in the lateral direction. This was confirmed in many instances by the neutron logs that measured the presence of higher water content at the depth where water was added. Consequently, it does not appear that the additional water would have significantly contacted the zone of the split-spoon in most of the split-spoon samples collected. While there are samples affected that do increase the uncertainty at some locations, there does not appear to be a bias introduced to these data that would change the conclusions of the nature and extent and fate and transport analyses and would not change the selection of remedies and combination of remedies that are described in the Proposed Plan.

4.3.24 Summary Vadose Zone Preliminary COPCs at RI Waste Sites

Analytes detected (if no background values are available) or present above background values within the vadose zone are generally indicators of anthropogenic waste management effects on the environment. Table 4-6 identifies the preliminary COPCs at waste sites identified in the 100-D/H SAP (DOE/RL-2009-40) associated with borehole, well, and test pit soil sample collection. The mobility and risk associated with contamination in the vadose zone are further evaluated in Chapters 5, 6, and 7 to assess the need for remedial action.

4.3.25 Summary of Vadose Zone Nature and Extent

Soil samples were collected during limited field investigations, interim remedial actions, and the RI to support an evaluation of the nature and extent of existing contamination in the vadose zone at 100-D/H. Soil data from these efforts are used to identify the type, concentration, and distribution of contamination detected (if no background values are available) or present above background concentrations in the vadose zone. The preliminary COPCs identified in Table 4-6 provide an indicator of anthropogenic impacts associated with discharging effluent to the soil and other waste management practices. Various radionuclides, metals, semivolatile organic compounds, volatile organic compounds, PCBs, pesticides, and anions are identified as preliminary COPCs in the vadose zone. Their concentrations and distributions vary by contaminant and location.

Table 4-6. Preliminary COPCs Present in the Vadose Zone Above Background

Radionuclides	Metals	SVOCs	SVOCs continued
Americium-241	antimony	1,3dichlorobenzene	pyrene
Carbon-14	arsenic	1,4, dichlorobenzene	2-methylnaphthalene
Cesium-137	barium	2chlorophenol	dibenzofuran
Cobalt-60	lithium	4chloro3methylphenol	Naphthalene
Europium-152	cadmium	acenaphthene	N-nitrosodiphenylamine
Europium-154	chromium	acenaphthylene	VOCs
Europium-155	copper	anthracene	4-methyl-2-pentonone
Neptunium-237	Cr(VI)	benzo(a)anthracene	1,1,1-trichloroethane
Nickel-63	lead	benzo(b)fluoranthene	**Pesticides**
Plutonium-238	mercury	benzo(g,h,i)perylene	Aldrin
Plutonium-239/240	molybdenum	benzo(a)pyrene	Beta-BHC
Strontium-90	nickel	benzo(k)fluoranthene	delta-BHC
Technetium-99	selenium	chrysene	endrin aldehyde
Tritium	silver	dibenzo(a,h)anthracene	endrin ketone
Uranium-233/234	strontium	dimethylphthalate	heptachlor
Uranium-235	thallium	fluoranthene	Heptachlor epoxide
Uranium-238	tin	fluorene	4,4-DDT
PCBs	vanadium	indeno(1,2,3cd)pyrene	**Anions**
Aroclor-1242	zinc	pentachlorophenol	Cyanide
Aroclor-1254	boron	phenanthrene	fluoride
Aroclor-1260			nitrate
			nitrite
			Sulfate

The concentrations of most radionuclides decrease with depth. Radionuclides like americium-241, cesium-137, and europium-152 were mainly detected in the upper half of the vadose zone. The distribution of other radionuclides, such as carbon-14, neptunium-237, and technetium-99, are characterized typically as sporadic or single detections. Strontium-90 appears to be the most widespread radionuclide associated with historical 100-D/H sources, extending throughout the vadose zone at some waste sites.

Metals are the second most common group of analytes detected or present above background levels in 100-D/H. Strontium (metal), and tin were consistently detected in RI samples, but their presence reflects the lack of an established background level and does not appear to be indicative of Hanford Site operations. Residual Cr(VI) and total chromium were frequently detected in the vadose associated with remediated waste sites during RI sampling, but the maximum Cr(VI) concentration (4.07 mg/kg) was detected at 116-H-7 at 4.8 m (15.7 ft) bgs and their concentrations both generally decrease with depth. Antimony, barium, boron, cadmium, copper, mercury, selenium, silver, thallium, vanadium, and zinc were only identified at a limited number of waste sites.

PCBs, semivolatile organics, volatile organics, pesticides, and anions are generally present infrequently, at low concentrations, or single detections in the vadose zone.

The mobility and risk associated with contamination in the vadose zone are further evaluated in Chapters 5, 6, and 7 to assess the need for remedial action.

4.4 Groundwater Contamination

This section presents a comprehensive interpretation of sampling results conducted to address additional data needs for spatial and temporal distribution of contaminants as identified in the 100-D/H Work Plan (DOE/RL-2008-46-ADD1). Concentration trends over time and summary statistics for groundwater COPCs are based on groundwater data from wells sampled over a 5-year period (from January 2006 through December 2010). The nature and extent evaluation used data through the end of 2011. Figure 2-3 presents the location of the groundwater monitoring wells and the aquifer tubes in 100-D/H area. Effects on contaminant concentrations and distributions from changes in Columbia River stage are discussed.

The 100-D/H Work Plan (DOE/RL-2008-46-ADD1), Section 4.8, identified the following data need associated with evaluation of the spatial and temporal distribution of contaminants in groundwater.

Data Need No. 13: Collect and analyze groundwater samples from select groundwater wells. As a result of the uncertainties identified in the RCBRA Report (DOE/RL-2007-21), the Integrated Work Plan (DOE/RL-2008-46) added activities that would help reduce uncertainties, verify conclusions of the HHRA presented in the RCBRA Report (DOE/RL-2007-21, Volume II), and ensure that contaminants were not inadvertently overlooked based on the use of the existing groundwater dataset. Section 3.6.5.1 of the Integrated Work Plan (DOE/RL-2008-46) identifies the following activities to reduce uncertainties:

- Identify existing and/or install new monitoring wells that are spatially representative of the groundwater. This set of wells will represent locations where a receptor potentially could contact groundwater.
- Conduct multiple rounds of sampling to obtain temporal representation of the unconfined aquifer from influence of river stage. Additional rounds of sampling at spatially representative monitoring wells will represent current groundwater conditions and capture the influence of river fluctuations on COPC concentrations.
- Analyze all spatially representative monitoring wells for a focused list of groundwater COPCs identified for each round of sampling. Analyzing each of the monitoring wells for COPCs will provide a dataset that is representative of potential releases to the groundwater.
- Evaluate sample results from characterization activities to support final remedial action decisions for groundwater.

To address data gap 13, a total of 52 existing wells were sampled and results were analyzed for spatial and temporal distribution. The sampling locations are presented on Figure 2-3.

The contaminant plume areas are discussed geographically as the 100-D southern plume, 100-D northern plume, 100-H plume, and horn area plume, and are mainly based on the distribution of Cr(VI) concentrations. The other contaminants are primarily collocated with the Cr(VI) plume. The highest concentrations of contaminants have been identified in the southern plume of 100-D. Slightly lower concentrations are present in the 100-D northern plume and at 100-H. The horn area plume, which is characterized by even lower contaminant concentrations, is the region between 100-D and 100-H.

For analytes that have shown consistent detections above action levels, plume maps were developed to show the spatial extent of contamination in the unconfined aquifer at 100-D, 100-H, and the horn. Plume maps were created for Cr(VI), nitrate, strontium-90, zinc, carbon tetrachloride, sulfate, and tritium.

4.4.1 Groundwater Data Collected for Spatial and Temporal Analysis

As a result of the uncertainties identified in the RCBRA (DOE/RL-2007-21), a rigorous analysis of groundwater data was performed for the purpose of identifying COPCs and reported in the 100-D/H Work Plan (DOE/RL-2008-46-ADD1). A total of 31 groundwater COPCs were identified through the activities of the 100-D/H Work Plan (DOE/RL-2008-46-ADD1) and are listed in the 100-D/H SAP (DOE/RL-2009-40), Table 1-2. A total of 52 monitoring wells were selected to represent the 100-HR-3 Groundwater OU spatially; three sampling rounds were collected from each location for those analytes identified as COPCs. The sampling rounds were collected at low, transitional, and high river stage to represent the temporal variability in aquifer constituents during the year.

Seasonal variations in river stage affect aquifer conditions by causing temporary changes in the water table. These elevation changes affect flow directions and rates, causing local changes in contaminant concentrations. For example, high river stage conditions may cause an influx of clean water from the river, thereby lowering contaminant concentrations. When the aquifer further inland experiences the high river elevation as a pressure pulse, the higher water table may affect a contaminated section of unsaturated sediments, causing contaminant concentrations to rise. Conversely, when the river stage is at the lowest levels, the groundwater flow direction near the river is generally toward the river, also causing contaminant plumes to migrate toward the river. Further inland, contaminant concentrations in the aquifer may decrease because contaminated soils are above the water table and, therefore, cannot interact with groundwater to release contaminants. To characterize the dynamic groundwater conditions and associated contaminant levels adequately, sampling was conducted during periods when the river stage and water table are high, when both are low, and at some interval between or transitional to extreme conditions.

The Columbia River stage can vary 3 to 4 m (9.8 to 13.1 ft) between low and high elevation, which is based on a 30-day moving average selected to show the influence that river dynamics have on groundwater levels. This can cause water table fluctuations of several meters, depending on the hydraulic properties of local sediments and the distance of the observation point from the river. Examples of seasonal river changes are shown on Figure 4-59. The daily averaged elevations depict a cyclic pattern of maximum to minimum river stage from year to year. These periodic or cyclic changes are engineered by upstream dams and reservoirs used for flood control, hydroelectric production, and salmon spawning programs. For any given year, the highest river stages occur from May through June while the lowest levels occur from September through October, possibly to mid-November. The intervals between the maximum and minimum river stage from approximately December through April and July through August are periods when the aquifer is in transition. The change from low to high elevations occurs gradually over about four months, when levels are increasing from the low in the fall of the year to the May/June maximum. The change from high to low levels is sharp, occurring over a two-month interval through July and August.

To illustrate that the maximum and minimum river stages are predictable and therefore useful for setting the 100-HR-3 OU risk assessment sampling schedule, trends of daily averaged elevation measurements covering the same 360-day period from September through August of the following year are superimposed on Figure 4-60. For example, data from September 1, 2005 is overlain on data from September 1, 2006 and September 1, 2007. Such a comparison illustrates the repeatable cycle of seasonal variations, allowing the timing of river fluctuations to set the schedule for the 100-HR-3 OU RI groundwater sampling. This schedule, as discussed in the Integrated Work Plan (DOE/RL-2008-46), began in October 2009 and was completed in June 2010.

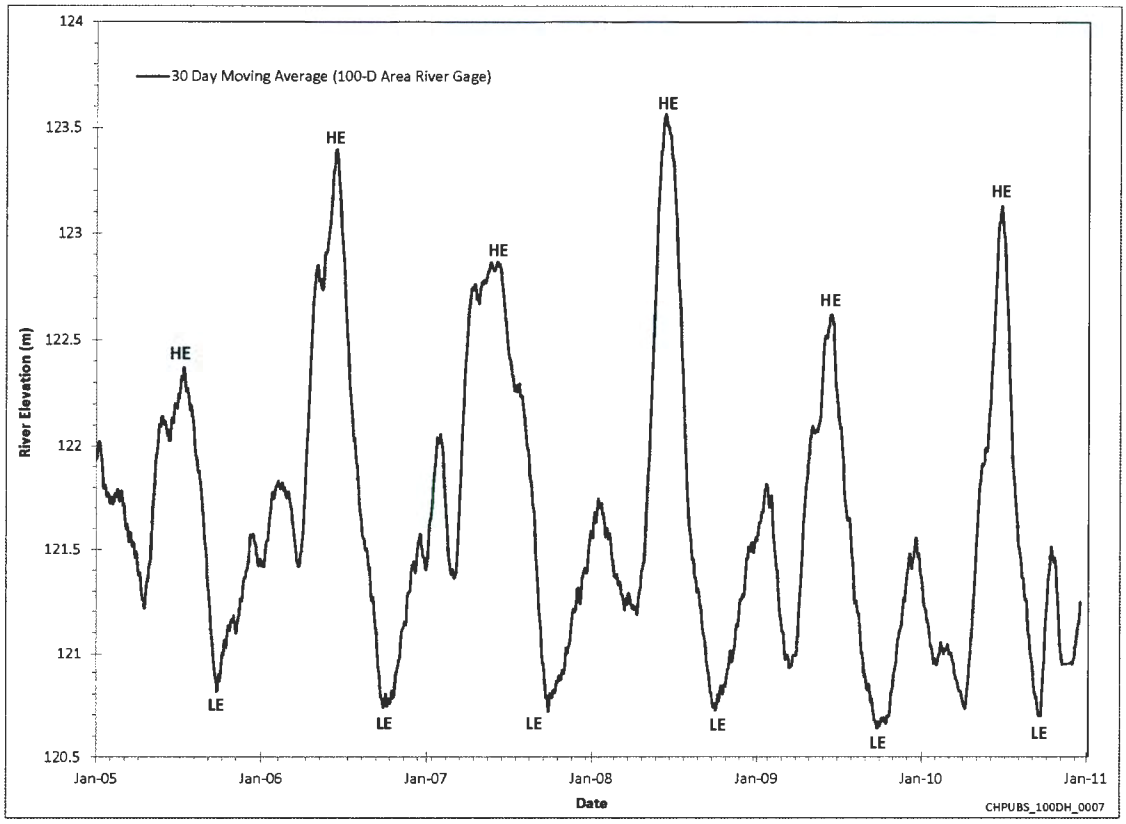

Figure 4-59. Seasonal Fluctuations in River Elevations Illustrating the Cyclic Nature of Maximum High and Low Elevations Over Multiple Years

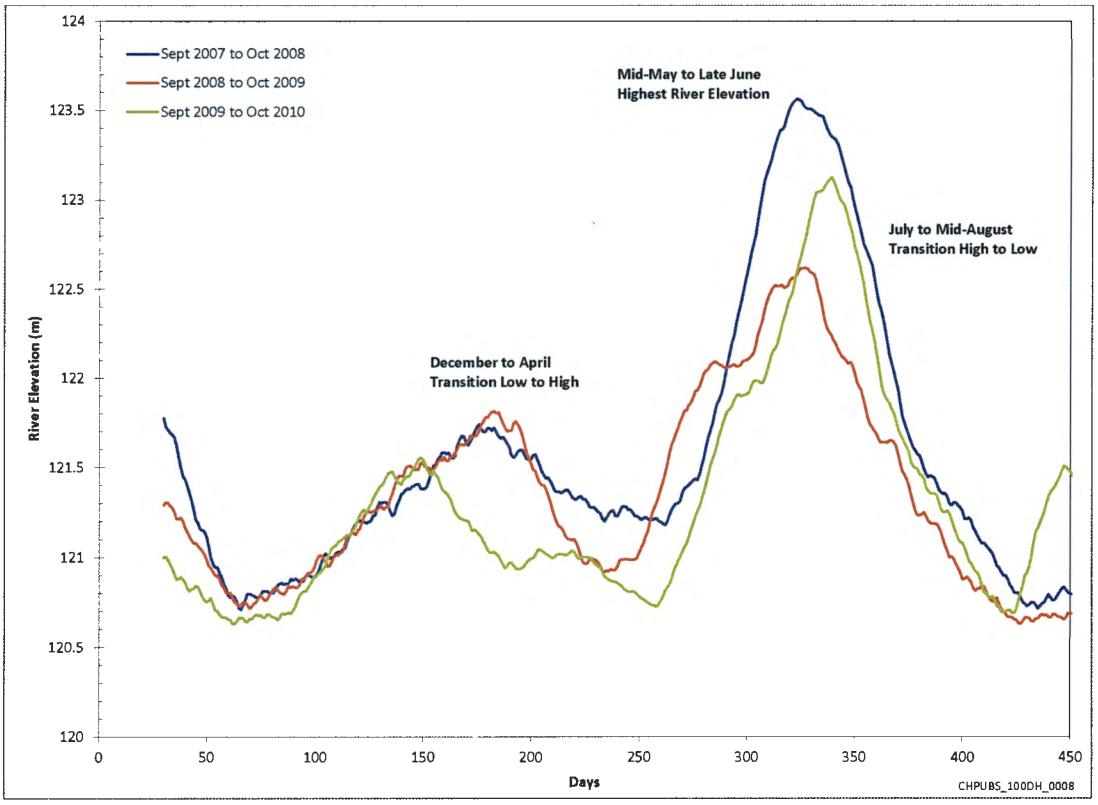

Figure 4-60. Annual Trends in River Elevations at a Single Location

With this prescribed period over which groundwater samples could be collected, the water table was low in October 2009, and at or near a maximum in June 2010. The transitional period occurred during the winter of 2009 to 2010 through the early spring of 2010. Thus, sampling of the groundwater network was scheduled in October 2009, March 2010, and June 2010. The final sampling intervals based on actual sampling dates are compared to a trend line of river elevation data on Figures 4-59 and 4-60. First, each sampling event was completed within the predetermined periods for low water table from mid-September to mid-November, transitional aquifer conditions occurring from December 2009 through April 2010, and maximum aquifer levels from May through June 2010. It should be noted, however, that an extremely high river stage occurred in July 2010. This anomaly was a result of unusual snowfall and temperatures, and could not have been predicted. Second, each sampling event was completed within 30 days, thus minimizing effects from dynamic river fluctuations. Based on the previous discussion, the chemistry data from groundwater samples collected during these three sampling events are fully representative of the dynamic groundwater conditions at the 100-HR-3 OU.

In monitoring wells, the water table response becomes more muted as distance from the river increases. Figures 4-61 and 4-62 show hydrographs for river gages at 100-D and 100-H, and adjacent wells screened in the unconfined aquifer. Groundwater levels in well 199-D8-70 are fairly close to that of the river elevation versus wells 199-D2-11 and 199-D5-99, where the groundwater level responses are much more seasonal. Similar responses are observed in 100-H, but the hydraulic conductivity of the aquifer matrix is higher, causing the wells to be in much higher hydraulic communication with the river. Groundwater level responses in these wells tend to follow the river more closely. Further inland at well 199-H5-1A, the response is more muted. Overall, the response of the river can be measured relatively far inland in the aquifer.

The analytical data are presented in Appendix D, incorporated into the historical summary statistics, and included in the contaminant distribution discussions. Further evaluations of this dataset, including the evaluations of COPCs, are presented in Chapter 6, Human Health Risk Assessment.

4.4.1.1 Historical Groundwater Evaluation

Uncertainties associated with the groundwater dataset were identified in the RCBRA. These uncertainties relate to the ability of the groundwater dataset collected from 1998 to 2008 to represent current baseline conditions and potential exposure within each groundwater OU. Analytical data used for the screening level assessment were collected to fulfill a variety of state and federal regulations, including the *Atomic Energy Act of 1954*, RCRA, CERCLA, and Section 173 of the *Washington Administrative Code*. Although the monitoring data can be used for risk assessment purposes, there are uncertainties associated with its use. Specifically, target analytes, sampling frequencies, and MDLs (or reporting limits) are different between programs because the information is used to meet different requirements.

As a result of the uncertainties identified in the RCBRA (DOE/RL-2007-21), a rigorous analysis of groundwater data for the purpose of identifying COPCs was performed in the 100-D/H Work Plan (DOE/RL-2008-46-ADD1). The groundwater dataset used for COPC identification consisted of sampling and analysis data collected from 98 monitoring wells from the 100-HR-3 Groundwater OU. The sampling and analysis data were collected between January 7, 1992, and November 20, 2008, and include four consecutive quarterly rounds collected during 1992 and 1993 and reported in 100-HR-3 LFI (DOE/RL-93-43), which were also used for the ecological component of the qualitative risk assessment (*Qualitative Risk Assessment for the 100-HR-3 Groundwater Operable Unit* [WHC-SD-EN-RA-007]).

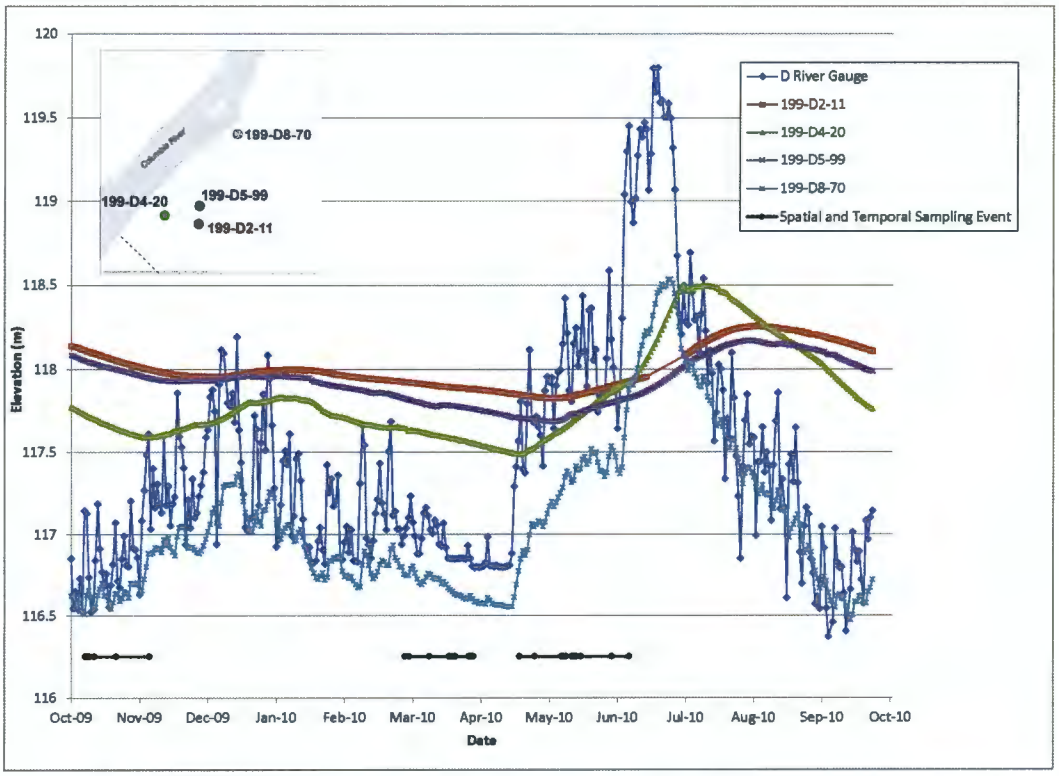

Figure 4-61. Hydrographs of the Columbia River at 100-D and Groundwater Levels in Adjacent Monitoring Wells

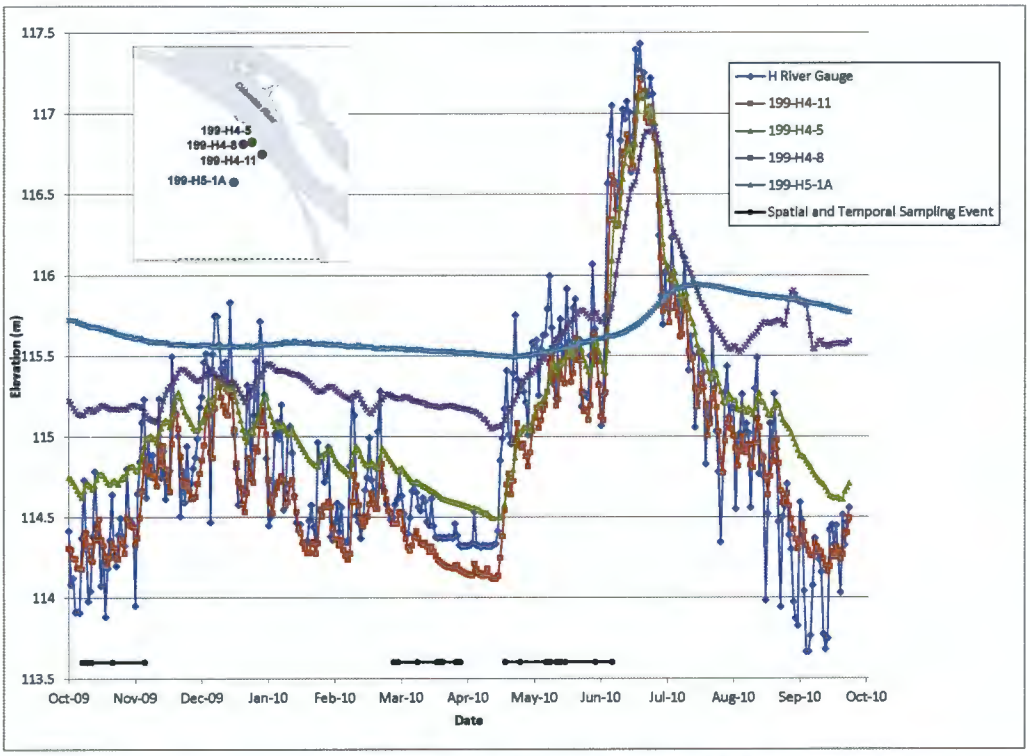

Figure 4-62. Hydrographs of the Columbia River at 100-H and Groundwater Levels in Adjacent Monitoring Wells

In total, 31 groundwater COPCs were identified through the activities of the 100-D/H Work Plan (DOE/RL-2008-46-ADD1) and are listed in the 100-D/H SAP (DOE/RL-2009-40), Table 1-2. The process used to develop the vadose zone soil target analyte lists and groundwater COPCs is described in Section 4.4 of the 100-D/H Work Plan (DOE/RL-2008-46-ADD1). Step 4 of the COPC identification process identifies the agency review of monitoring well locations and groundwater COPCs. This step of the process allows the agency to adjust the COPC identification process by adding additional analytes or sample locations on a site-specific basis. Table 4-7 lists the additional analytes and the monitoring well locations that were included as a result of Step 4 of the COPC identification process.

Table 4-7. Additional Groundwater Analytes and Locations for Analysis

Pesticides by Method 8081	PCBs by Method 1668A	PCBs by Method 8082	Polynuclear Aromatic hydrocarbons by Method 8310
199-D5-15[a] 199-D8-71[b]	199-D5-15[a] 199-D8-55[b] 199-D8-71[b] 199-H4-10[a] 199-H4-13[a] 199-H4-48[a]	199-D4-84[a] 199-D5-13[a] 199-D5-15[a] 199-D5-17[a] 199-D5-99[a] 199-D8-55[b] 199-D8-71[b] 199-D8-88[a] 199-H3-2A[a] 199-H4-3[b] 199-H4-10[a] 199-H4-11[b] 199-H4-13[a] 199-H4-16[a] 199-H4-45[b] 199-H4-48[a]	199-D4-84[a] 199-D5-13[a] 199-D5-15[a] 199-D5-17[a] 199-D5-99[a] 199-D8-55[b] 199-D8-71[b] 199-D8-88[a] 199-H3-2A[a] 199-H4-3[b] 199-H4-10[a] 199-H4-11[b] 199-H4-13[a] 199-H4-16[a] 199-H4-45[b] 199-H4-48[a]
Semivolatile Organic Compounds by Method 8270		**Radionuclides[c]**	**Cyanide by Method 9012**
199-D4-84[a] 199-D5-13[a] 199-D5-15[a] 199-D5-17[a] 199-D5-99[a] 199-D8-55[b] 199-D8-71[b] 199-D8-88[a]	199-H3-2A[a] 199-H4-3[b] 199-H4-10[a] 199-H4-11[b] 199-H4-13[a] 199-H4-16[a] 199-H4-45[b] 199-H4-48[a]	All wells in monitoring well network.	199-H4-3[b]

a. Collected at the low river stage

b. Collected at the low river stage and high river stage

c. Radionuclides include gross alpha, gross beta, cesium-137, cobalt-60, europium-152, and europium-154.

The analytical performance requirements (required analytical method) and the lowest chemical-specific ARAR are listed in the 100-D/H SAP (DOE/RL-2009-40). The action level is listed to ensure that the estimated quantitation limit (EQL) is adequate for confirming the presence or absence of the COPC at the

corresponding level. In total, 52 monitoring wells were selected to represent the 100-HR-3 Groundwater OU spatially; three sampling rounds were collected from each location for those analytes identified as COPCs and radionuclides listed in Table 4-1. The sampling rounds were collected at low, transitional, and high river stage to represent the temporal variability in aquifer constituents during the year. This dataset was used to perform the risk assessment presented in Chapter 6.

4.4.1.2 Groundwater Evaluation for the Unconfined Aquifer

The nature and extent of contamination in groundwater was based on the last six years of data, which were considered representative of current groundwater conditions (that is, samples collected between January 1, 2006 and December 31, 2011). The nature and extent evaluation uses a subset of data from the 100-D/H Work Plan (DOE/RL-2008-46-ADD1) for wells screened in the unconfined aquifer, as well as all spatial and temporal wells considered in the groundwater risk assessment. A total of 99 wells, including the 52 spatial and temporal wells, were considered. Figure 4-63 provides the locations of wells considered in the nature and extent evaluation. Groundwater data for 100-D/H were compiled and statistically analyzed and the results are presented in Appendix O, Table O-1 through Table O-3. These tables present the summary statistics for each analyte identified as a historical COPC in the 100-D/H Work Plan (DOE/RL-2008-46-ADD1) and the 100-D/H SAP (DOE/RL-2009-40) and list the background concentrations in Hanford Site groundwater (*Hanford Site Background: Part 3, Groundwater Background* [DOE/RL-96-61]) where available, and the action level for each analyte.

The additional analytes that were requested for each well listed in Table 4-7 are presented in Table O-4 through Table O-19. These tables list the additional analytes by well, provide summary statistics (where applicable), and list the background concentrations and action level for each analyte.

For the purpose of COPC identification, action levels are screening levels derived from chemical-specific ARARs and/or risk based concentrations using default exposure assumptions (it should be noted that some of the exposure pathways in these screening levels are incomplete).

Following are the sources of action levels from federal regulations:

- 40 CFR 141 National Primary Drinking Water Regulations, MCLs, secondary MCLs, and nonzero MCLGs established under the *Safe Drinking Water Act of 1974* (SDWA)
- National Recommended Water Quality Criteria, Ambient Water Quality Criteria (AWQC) established under Section 304 of the *Clean Water Act of 1977*
- "Water Quality Standards" (40 CFR 131) for states not complying with Section 303 of the *Clean Water Act of 1977*

Following are the sources of the action levels from Washington State regulations:

- "Water Quality Standards for Surface Waters of the State of Washington" (WAC 173-201A)
- "Groundwater Cleanup Standards" (WAC 173-340-720)
- "Surface Water Cleanup Standards" (WAC 173-340-730)
- "Group A Public Water Supplies," "Maximum Contaminant Levels (MCLs) and Maximum Residual Disinfectant Levels (MRDLs)" (WAC 246-290-310)

While surface water and AWQC standards are considered for the identification of action levels, it must be noted that these standards only apply for groundwater where it enters the Columbia River. For the upland parts of groundwater, only drinking water standards are applicable.

The following evaluation specifically identifies when the action level is a DWS or an AWQC. The evaluation presented in this section focuses on the following analytes:

- Analytes that are identified as COPCs in the groundwater risk assessment provided in Section 6.3 that warrant further evaluation in the FS.
- Analytes identified as historical COPCs in the 100-D/H Work Plan (DOE/RL-2008-46-ADD1) as a result of uncertainties resulting from limitations in the analytical data (inadequate MDLs or anomalous results). Analytical data used in the groundwater risk assessment provided in Section 6.3 and data from a larger population of wells sampled over a longer sampling period were evaluated to determine these analytes do not warrant further evaluation in the FS.
- Additional analytes that were identified through Step 4 of the COPC identification process in the 100-D/H Work Plan (DOE/RL-2008-46-ADD1) and do not warrant further evaluation in the FS.

The results summary is provided in the following subsections: 4.4.1.3 (COPCs by plume area), 4.4.1.4 (analytes of interest—nondetected), 4.4.1.5 (analytes of interest—100-D groundwater plume area), 4.4.1.6 (analytes of interest—100-H groundwater plume area), 4.4.1.7 (analytes of interest—horn groundwater plume area), 4.4.1.8 (additional analytes requested by agencies), and 4.4.1.9 (conclusions).

COPCs Warranting Further Evaluation in FS.
Section 6.3 identifies the COPCs that warrant further evaluation in the FS for each of the exposure areas evaluated in the groundwater risk assessment. The COPCs are discussed in the following paragraphs as applicable to each exposure area (100-D plume area, 100-H plume area, and the horn plume area).

100-D Plume Area. Cr(VI), chromium, and nitrate are identified in the 100-D plume areas as COPCs that warrant further evaluation in the FS. Concentrations of these COPCs are widely distributed and consistently present above the DWS (nitrate) or the AWQC (Cr(VI) and chromium). The following paragraphs provide a summary for each COPC. Additional information regarding trend plots and contours is provided in Section 4.5.

Cr(VI) was detected in 95 percent of the unfiltered and 92 percent of the filtered groundwater samples. Cr(VI) was reported above the AWQC of 10 µg/L in 89 percent of the detected unfiltered results and 92 percent of the detected filtered results. Although all monitoring wells within the plume area were compared to the AWQC value of 10 µg/L, only near-river wells would need to meet this criterion. Concentrations of filtered Cr(VI) ranged between 2.0 and 69,700 µg/L. With the exception of one unfiltered result, all MDLs were less than or equal to the AWQC of 10 µg/L. Note that an August 2010 groundwater sample from Well 199-D5-122, reports the site maximum Cr(VI) concentration of 69,700 µg/L. This well is located in the 100-D southern plume.

Chromium (total) was detected in 94 percent of the unfiltered and 93 percent of the filtered groundwater samples. Chromium (total) was reported above the AWQC of 65 µg/L in 63 percent of the detected unfiltered results and 62 percent of the detected filtered results. Concentrations of unfiltered chromium (total) range between 3.5 and 61,500 µg/L and filtered chromium (total) ranged between 3.2 and 10,500 µg/L. All MDLs were less than the AWQC of 65 µg/L. Although all monitoring wells within the groundwater plume area were compared to the AWQC, only near-river wells would need to meet this criterion.

Nitrate was detected in 100 percent of the unfiltered groundwater samples. Nitrate was reported above the DWS of 45,000 µg/L in 39 percent of the detected unfiltered results. Concentrations of unfiltered nitrate ranged between 1,160 and 116,000 µg/L.

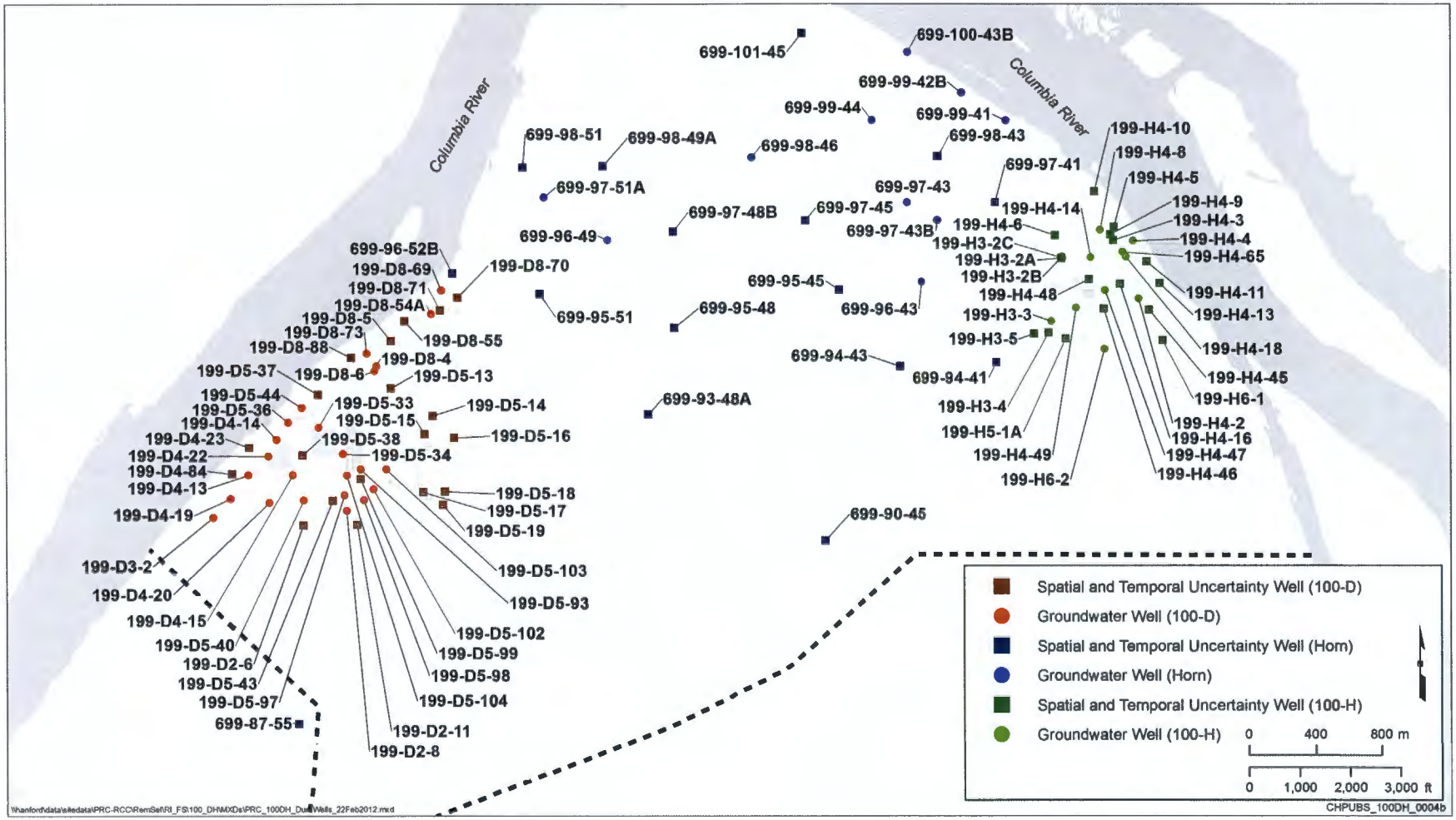

Figure 4-63. Well Locations Used in the Nature and Extent Evaluation.

100-H Plume Area. Cr(VI), strontium-90, and nitrate are identified in the 100-H plume area as COPCs that warrant further evaluation in the FS. Concentrations of Cr(VI) and strontium-90 are widely distributed and consistently present at concentrations above the AWQC (Cr(VI)) or the DWS (strontium-90). Nitrate in the 100-H plume area is not widely distributed but is present at concentrations above the DWS in localized areas. The following paragraphs provide a summary for each COPC. Additional information regarding trend plots and contours are provided in Section 4.5.

Cr(VI) was detected in 93 percent of the unfiltered and 89 percent of the filtered groundwater samples. Cr(VI) was reported above the AWQC of 10 μg/L in 61 percent of the detected unfiltered results and 51 percent of the detected filtered results. Concentrations of filtered Cr(VI) ranged between 2 and 75 μg/L. All MDLs were less than the AWQC of 10 μg/L. Although all monitoring wells within the plume area were compared to the AWQC value of 10 μg/L, only near-river wells would need to meet this criterion.

Strontium-90 was detected in 42 percent of the unfiltered groundwater samples. Strontium-90 was reported above the DWS of 8 pCi/L in 38 percent of the detected unfiltered results. Concentrations of unfiltered strontium-90 ranged between 1.5 pCi/L and 110 pCi/L. All MDLs were less than the DWS of 8 pCi/L.

Nitrate was detected in 100 percent of the unfiltered groundwater samples. Nitrate was reported above the DWS of 45,000 μg/L in 3.2 percent of the detected unfiltered results. Concentrations of unfiltered nitrate ranged between 416 and 253,000 μg/L. The maximum nitrate concentration of 253,000 μg/L was measured in well 199-H4-3 during May 2006. Nitrate concentrations measured at 199-H4-3 during 2010 and 2011 range between 27,400 and 49,100 μg/L.

Horn Plume Area. Cr(VI) and chromium are identified in the horn plume area as COPCs that warrant further evaluation in the FS. Concentrations of these COPCs are widely distributed and consistently present at concentrations above the AWQC (Cr(VI) and chromium). The following paragraphs provide a summary for each COPC. Additional information regarding trend plots and contours are provided in Section 4.5.

Cr(VI) was detected in 90 percent of the unfiltered and 87 percent of the filtered groundwater samples. Cr(VI) was reported above the AWQC of 10 μg/L in 89 percent of the detected unfiltered results and 89 percent of the detected filtered results. Concentrations of filtered Cr(VI) ranged between 2.9 and 117 μg/L. All MDLs were less than the AWQC of 10 μg/L. Although all monitoring wells within the plume area were compared to the AWQC value of 10 μg/L, only near-river wells would need to meet this criterion.

Chromium (total) was detected in 89 percent of the unfiltered and 91 percent of the filtered groundwater samples. Chromium (total) was reported above the AWQC of 65 μg/L in 16 percent of the detected unfiltered results and 13 percent of the detected filtered results. Concentrations of filtered chromium (total) ranged between 4.3 and 113 μg/L. All MDLs were less than the AWQC of 65 μg/L. Although all monitoring wells within the groundwater plume area were compared to the AWQC, only near-river wells would need to meet this criterion.

Historical COPCs—Nondetected

Historical COPCs are those analytes that were identified in the 100-D/H SAP (DOE/RL-2009-40) or those analytes for which a maximum concentration exceeding an action level was reported during the spatial and temporal sampling (Section 6.3). The following provides descriptions of those historical COPCs that were not detected in the spatial and temporal dataset or the dataset representing a larger population of wells and a longer sampling timeframe. Nondetected historical COPCs include radionuclides and VOCs.

Gross gamma analytes (cesium-137, cobalt-60, europium-152, and europium-154) were identified as additional analytes in the 100-D/H SAP (DOE/RL-2009-40). Gross gamma analytes were analyzed in all of the RI monitoring network wells during all sampling rounds. Gross gamma analytes were not detected in any of the groundwater samples analyzed from any plume area. All MDLs were less than their

respective DWSs. Based on the results of this evaluation and the groundwater risk assessment presented in Section 6.3, gross gamma analytes are not identified as COPCs to be further evaluated in the FS.

1,1-Dichloroethene was identified as a historical COPC in the 100-D/H SAP (DOE/RL-2009-40) not because it was detected but because the laboratory MDLs were not adequate for determining its presence at or below 0.057 µg/L. The action level for 1,1-dichloroethene of 0.057 µg/L is based on the national recommended water quality criteria for "Human Health for Consumption of Water plus Organism;" however, it defaults to the EQL of 2 µg/L reported in the 100-D/H SAP (DOE/RL-2009-40) when the analytical method cannot achieve a detection level of 0.057 µg/L. 1,1-Dichloroethene was not detected in any groundwater sample from any plume area and all MDLs are less than the EQL listed in the 100-D/H SAP (DOE/RL-2009-40). Based on the results of this evaluation and the groundwater risk assessment presented in Section 6.3, 1,1-dichloroethene is not identified as a COPC for further evaluation in the FS.

Benzene was identified as a historical COPC in the 100-D/H SAP (DOE/RL-2009-40) because it was detected above the groundwater cleanup standard and most MDLs were greater than the groundwater cleanup standard. Benzene was not detected in any unfiltered groundwater sample from any plume area. The action level for benzene of 0.8 µg/L is based on the MTCA ("Groundwater Cleanup Standards" [WAC 173-340-720]) level; however, it defaults to the EQL of 1.5 µg/L reported in the 100-D/H SAP (DOE/RL-2009-40) when the analytical method cannot achieve the groundwater cleanup standard. Benzene was not detected in any groundwater sample from any plume area and all MDLs are less than the EQL listed in the 100-D/H SAP (DOE/RL-2009-40). Based on the results of this evaluation and the groundwater risk assessment presented in Section 6.3, benzene is not identified as a COPC for further evaluation in the FS.

Trichloroethene was identified as a historical COPC in the 100-D/H SAP (DOE/RL-2009-40) not because it was detected but because the MDLs were not adequate for determining its presence at or below the groundwater cleanup standard. The action level for trichloroethene of 0.49 µg/L is based on the MTCA ("Groundwater Cleanup Standards" [WAC 173-340-720]) groundwater cleanup level; however, it defaults to the EQL of 1 µg/L reported in the 100-D/H SAP (DOE/RL-2009-40) when the analytical method cannot achieve the groundwater cleanup standard. Trichloroethene was not detected in any groundwater sample from the 100-D or 100-H plume area and all MDLs are less than or equal to the EQL listed in the 100-D/H SAP (DOE/RL-2009-40). Based on the results of this evaluation and the groundwater risk assessment presented in Section 6.3, trichloroethene is not identified as a COPC for further evaluation in the FS.

Vinyl chloride was identified as a historical COPC in the 100-D/H SAP (DOE/RL-2009-40) not because it was detected but because the MDLs were not adequate for determining its presence at or below the national recommended water quality criteria for "Human Health for Consumption of Water plus Organism." The action level for vinyl chloride of 0.025 µg/L is based on the national recommended water quality criteria for "Human Health for Consumption of Water plus Organism;" however, it defaults to the EQL of 5 µg/L reported in the 100-D/H SAP (DOE/RL-2009-40) when the analytical method cannot achieve a detection level of 0.025 µg/L. Vinyl chloride was not detected in any groundwater sample from any plume area and all MDLs are less than the EQL listed in the 100-D/H SAP (DOE/RL-2009-40). Based on the results of this evaluation and the groundwater risk assessment presented in Section 6.3, vinyl chloride is not identified as a COPC for further evaluation in the FS.

Historical COPCs—100-D Groundwater Plume Area

The following subsections describe historical COPCs that were detected at least once in the 100-D groundwater plume area and include radionuclides, VOCs, anions, and metals. As described earlier, historical COPCs are either those analytes that were identified as COPCs in the 100-D/H SAP

(DOE/RL-2009-40) or those analytes for which a maximum concentration exceeding an action limit was reported during the spatial and temporal sampling (Section 6.3).

Radionuclides. Gross alpha and gross beta were identified as additional analytes in the 100-D/H SAP (DOE/RL-2009-40). Gross alpha and gross beta were analyzed in all of the RI monitoring network wells during all sampling rounds as well as in the larger populations of wells over the longer timeframe. Gross alpha was detected in 14 percent of the unfiltered groundwater samples and gross beta was detected in 84 percent of the unfiltered samples. Gross alpha was detected at concentrations ranging between 1.2 and 6.8 pCi/L, which are less than the DWS of 15 pCi/L. Gross beta was detected at concentrations ranging between 2.3 and 220 pCi/L. Gross beta concentrations are consistent with the presence of tritium. Based on the results of this evaluation and the groundwater risk assessment presented in Section 6.3, gross alpha and gross beta are not identified as COPCs to be further evaluated in the FS.

Strontium-90 was identified as a historical COPC in the 100-D/H Work Plan (DOE/RL-2008-46-ADD1) because it was detected above the DWS of 8 pCi/L. Strontium-90 was detected in 20 of 109 (18 percent) of the unfiltered groundwater samples, with concentrations ranging between 0.48 and 45 pCi/L. With the exception of a single result reported at well 199-D5-132, all strontium-90 results (detected concentrations and MDLs) were less than the DWS of 8 pCi/L. Strontium-90 was reported at a concentration of 45 pCi/L at well 199-D5-32; this is the only result reported at this well during the specified period because it was installed during the RI to fill data gap 2 and data gap 5. Additionally, well 199-D5-12, located south of the 116-D-1A liquid waste stream, historically reported strontium-90 concentrations above the DWS (with concentrations up to 52.6 pCi/L) until it was decommissioned in 2002. Based on the results of this evaluation and the groundwater risk assessment presented in Section 6.3, strontium-90 is retained as a COPC for further evaluation in the FS.

Technetium-99 was identified as a historical COPC in the 100-D/H Work Plan (DOE/RL-2008-46-ADD1) because it was detected above the DWS of 900 pCi/L. Technetium-99 was detected in 2 of 72 (2.8 percent) of the unfiltered groundwater samples with concentrations ranging between 12 and 16 pCi/L. All results (detected concentrations and MDLs) were less than the DWS of 900 pCi/L. Based on the results of this evaluation and the groundwater risk assessment presented in Section 6.3, technetium-99 is not retained as a COPC for further evaluation in the FS.

Tritium was identified as a historical COPC in the 100-D/H Work Plan (DOE/RL-2008-46-ADD1) because it was detected above the DWS of 20,000 pCi/L. Tritium was detected in 175 of 223 (78 percent) of the unfiltered groundwater samples with concentrations ranging between 180 and 20,000 pCi/L. All results (detected concentrations and MDLs) are less than the DWS of 20,000 pCi/L. Tritium was reported at 199-D4-19 with concentrations that range between 9,050 and 20,000 pCi/L with concentrations reducing with time. Tritium was below the DWS of 20,000 pCi/L at all other monitoring wells. Tritium concentrations at or just below the DWS are associated with a trend at 199-D4-19. Based on the results of this evaluation and the groundwater risk assessment presented in Section 6.3, tritium is not retained as a COPC for further evaluation in the FS.

Volatile Organic Compounds. 1,2-Dichloroethane was detected above the action level of 0.38 μg/L in samples collected for the RI in the H Plume area and is included in the nature and extent evaluation. 1,2-Dichloroethane was not identified as a historical COPC in the 100-D/H Work Plan (DOE/RL-2008-46-ADD1) because it was never reported above the MDL in the 115 samples included in the dataset that were analyzed. The action level for 1,2-dichloroethane of 0.38 μg/L is based on the CWA criteria for "Human Health for Consumption of Water plus Organism;" however, it defaults to the EQL of 5 μg/L reported in the 100-D/H SAP (DOE/RL-2009-40) when the analytical method cannot achieve a detection level of 0.38 μg/L. 1,1-Dichloroethene was not detected in any groundwater sample from any plume area and all

MDLs are less than the EQL listed in the 100-D/H SAP (DOE/RL-2009-40). Based on the results of this evaluation and the groundwater risk assessment presented in Section 6.3, 1,2-dichloroethane is not retained as a COPC for further evaluation in the FS.

Bromodichloromethane was detected above the action level of 0.27 µg/L in samples collected for the RI and is included in the nature and extent evaluation. Bromodichloroethane was not identified as a historical COPC in the 100-D/H Work Plan (DOE/RL-2008-46-ADD1) and results for bromodichloromethane were not reported for samples collected for purposes other than the RI. The action level for bromodichloromethane is 0.27 µg/L based on the *National Recommended Water Quality Criteria* (EPA, 2009b), Human Health Water + Organism" value; however, it defaults to the EQL of 5 µg/L reported in the 100-D/H SAP (DOE/RL-2009-40) when the analytical method cannot achieve the action level. All results (detected concentrations and MDLs) are less than the EQL listed in the 100-D/H SAP (DOE/RL-2009-40). Based on the results of this evaluation and the groundwater risk assessment presented in Section 6.3, bromodichloromethane is not retained as a COPC for further evaluation in the FS.

Carbon tetrachloride was identified as a historical COPC in the 100-D/H Work Plan (DOE/RL-2008-46-ADD1) because it was detected above the action level, and most MDLs were greater than the action level. The action level for carbon tetrachloride defaults to the EQL of 1 µg/L reported in the 100-D/H SAP (DOE/RL-2009-40) when the analytical method cannot achieve 0.23 µg/L. Carbon tetrachloride was detected twice in 199-D2-6 with values of 1.7 µg/L on 8/2/2009 (transitional river stage) and 2.6 µg/L on 10/8/2010 (low river stage), both at concentrations greater than the EQL of 1 µg/L. Well 199-D2-6 (see Figure 4-63 for well location) was sampled and analyzed for carbon tetrachloride during a subsequent transitional river stage (3-30-2010) for the spatial and temporal sampling and the result (0.063U) was reported below the EQL. No other carbon tetrachloride results were reported for 199-D2-6 during a low river stage. Carbon tetrachloride was detected once in 199-D5-18 (2.7 µg/L) at a concentration greater than the EQL of 1 µg/L. Carbon tetrachloride was analyzed in two subsequent sampling rounds at this well and reported as nondetected concentrations less than the action level. All MDLs are less than or equal to the EQL listed in the 100-D/H SAP (DOE/RL-2009-40). Based on the results of this evaluation and the groundwater risk assessment presented in Section 6.3, carbon tetrachloride is identified as a COPC for further evaluation in the FS.

Chloroform was identified as a historical COPC in the 100-D/H Work Plan (DOE/RL-2008-46-ADD1) because it was detected above the action level, and most MDLs were greater than the action level. MDLs were not adequate for determining the presence of chloroform at or below the action level of 1.4 µg/L. The groundwater action level for chloroform defaults to the EQL of 5 µg/L reported in the 100-D/H SAP (DOE/RL-2009-40) when the analytical method cannot achieve the action level. Chloroform concentrations above the EQL of 5 µg/L were reported in four wells (199-D5-13, 199-D5-38, 199-D8-5, and 199-D8-88). Chloroform concentrations above the cleanup level were reported in three or more sampling rounds and appear to be associated with a trend at 199-D8-5 (three of four rounds; 4.4 to 8.3 µg/L). Infrequent detections of chloroform at concentrations above the EQL were reported at 199-D5-13 (one of four sampling rounds; 3.1 to 6.4 µg/L), 199-D5-38 (one of four sampling rounds; 1.9 to 5.8 µg/L), and 199-D8-88 (two of four sampling rounds; 3.2 to 8 µg/L). All MDLs are less than the EQL of 5 µg/L. Based on the results of this evaluation and the groundwater risk assessment presented in Section 6.3, chloroform is identified as a COPC.

Tetrachloroethene was detected above the action level of 0.081 µg/L in samples collected for the RI and is included in the nature and extent evaluation. Tetrachloroethene was not identified as a historical COPC in the 100-D/H Work Plan (DOE/RL-2008-46-ADD1). The action level for tetrachloroethene is 0.081 µg/L based on the MTCA ("Groundwater Cleanup Standards" [WAC 173-340-720]) groundwater cleanup

level; however, it defaults to the EQL of 5 µg/L reported in the 100-D/H SAP (DOE/RL-2009-40) when the analytical method cannot achieve the action level. All results (detected concentrations and MDLs) are less than the EQL listed in the 100-D/H SAP (DOE/RL-2009-40). Based on the results of this evaluation and the results of the groundwater risk assessment presented in Section 6.3, tetrachloroethene is not identified as a COPC for further evaluation in the FS.

Anions. Fluoride was identified as a historical COPC in the 100-D/H Work Plan (DOE/RL-2008-46-ADD1) because it was detected above the action level, and some MDLs were greater than the cleanup level of 960 µg/L. Fluoride was detected in 178 of 365 (49 percent) of unfiltered groundwater samples with concentrations ranging between 14 µg/L and 550 µg/L. All fluoride results are less than the action level. Fluoride concentrations in unfiltered samples are also less than the 90th percentile Hanford Site background level of 1,047 µg/L. Based on the results of this evaluation and the results of the groundwater risk assessment presented in Section 6.3, fluoride is not identified as a COPC for further evaluation in the FS.

Nitrite was identified as a historical COPC in the 100-D/H Work Plan (DOE/RL-2008-46-ADD1) because it was detected above the DWS, and some MDLs were greater than the DWS of 3,300 µg/L. Nitrite was detected in 88 of 458 (19 percent) of unfiltered samples with concentrations ranging between 27 and 3,190 µg/L. All results (detected concentrations and MDLs) were less than the DWS of 3,300 µg/L. Nitrite concentrations in unfiltered samples are also greater than the 90th percentile Hanford Site background level of 94 µg/L. Based on the results of this evaluation and the results of the groundwater risk assessment presented in Section 6.3, fluoride is not identified as a COPC.

Sulfate was identified as a historical COPC in the 100-D/H Work Plan (DOE/RL-2008-46-ADD1) because it was detected above the secondary DWS of 250,000 µg/L. Sulfate was detected in 100 percent of unfiltered groundwater samples with concentrations ranging between 11,100 and 549,000 µg/L. Sulfate concentrations above the secondary DWS were reported in 199-D4-23, 199-D4-84, and 199-D4-19. Sulfate concentrations above the DWS of 250,000 µg/L were reported in 2 of 19 sampling rounds at 199-D4-23; all 12 sampling rounds at 199-D4-84; and 5 of 15 sampling rounds at 199-D4-19. The results of this evaluation indicate that sulfate has historically been detected in groundwater samples. Sulfate concentrations above the secondary DWS are associated with a trend at 199-D4-84 and 199-D4-19 where concentrations have ranged up to 2.2 times greater than the secondary DWS of 250,000 µg/L. The presence of elevated concentrations of sulfate in these wells is associated with the presence of sodium dithionite, which is associated with the ISRM barrier at the OU and is not the result of a site release. With the exception of sulfate in the above wells, sulfate concentrations are less than the secondary DWS. Based on the results of this evaluation and the results of the groundwater risk assessment presented in Section 6.3, fluoride is not retained as a COPC.

Metals. Antimony was identified as a historical COPC in the 100-D/H Work Plan (DOE/RL-2008-46-ADD1) because it was detected above the action level, and most MDLs were greater than the action level. Antimony was detected in 16 of 249 unfiltered samples (6.4 percent) and 15 of 262 (5.7 percent) of the filtered groundwater samples. Antimony concentrations in unfiltered and filtered groundwater samples range between 0.65 and 68 µg/L. Unfiltered and filtered samples collected before the RI were analyzed by Method 6010. MDLs for these results range between 2.5 and 720 µg/L and detected concentrations range between 3.7 and 68 µg/L. All but one antimony result reported by Method 6010 were flagged with a "B" qualifier or with a "Y" review qualifier. The "B" qualifier indicates the analyte was detected at a value less than the required detection limit, but greater than or equal to the MDL. The "Y" review qualifier indicates the result is suspect, but there is insufficient evidence to show the result is valid or invalid. Samples collected for the RI were analyzed using trace methods identified in the 100-D/H SAP (DOE/RL-2009-40). MDLs for these samples range between 0.3 and 1.1 µg/L and the two detected concentrations range between 0.65 and 1.5 µg/L. The results of this evaluation indicate that antimony has

historically been detected in groundwater samples at a low frequency (6.2 percent in unfiltered samples and 6.9 percent in filtered samples) with concentrations up to 12 times greater than the action level. Some historical detections of antimony are flagged with a "B" qualifier. Antimony concentrations are not associated with a specific location or a trend. Antimony concentrations associated with samples collected for the RI are not above the action level of 5.6 μg/L. With the exception of five sample results flagged with a "Y" review qualifier, all antimony concentrations are below the 90th percentile Hanford Site background level of 55 μg/L. Based on the results of this evaluation and the results of the groundwater risk assessment presented in Section 6.3, antimony is retained as a COPC and warrants further evaluation in the FS.

Arsenic was identified as a historical COPC in the 100-D/H Work Plan (DOE/RL-2008-46-ADD1) because it was detected above the action level and all MDLs were greater than the action level. MDLs were not adequate for determining the presence of arsenic at or below the Action level of 0.018 μg/L. The action level for arsenic of 0.018 μg/L is based on the national recommended water quality criteria for "Human Health for Consumption of Water plus Organism;" however, the action level defaults to the EQL of 4 μg/L reported in the 100-D/H SAP (DOE/RL-2009-40) when the analytical method cannot achieve 0.018 μg/L. Arsenic was detected in 130 of 139 (93 percent) of the unfiltered and 128 of 146 (8 percent) of the filtered groundwater samples. Arsenic concentrations range between 0.42 and 5.7 in unfiltered groundwater samples and between 0.4 and 5.8 μg/L in filtered groundwater samples. Minimum, maximum, and 90th percentile concentrations for (filtered) background concentrations of arsenic are 0.5, 8.8, and 7.85 μg/L, respectively. Arsenic concentrations in unfiltered and filtered samples are less than the 90th percentile Hanford Site background concentration. Based on the results of this evaluation and the results of the groundwater risk assessment presented in Section 6.3, arsenic is not retained as a COPC for further evaluation in the FS.

Beryllium was identified as a historical COPC in the 100-D/H Work Plan (DOE/RL-2008-46-ADD1) because it was detected above the DWS, and most MDLs were greater than the DWS of 4 μg/L. Beryllium was detected in 7 of 249 (2.8 percent) of the unfiltered samples and in 5 of 262 (1.9 percent) of the filtered groundwater samples. Beryllium concentrations in unfiltered and filtered groundwater samples range between 0.092 and 0.31 μg/L. All beryllium results (detected concentrations and MDLs) are less than the DWS of 4 μg/L. In addition, all beryllium concentrations are less than the 90th percentile Hanford Site background level of 2.3 μg/L. Based on the results of this evaluation and the results of the groundwater risk assessment presented in Section 6.3, beryllium is not retained as a COPC for further evaluation in the FS.

Cadmium was identified as a historical COPC in the 100-D/H Work Plan (DOE/RL-2008-46-ADD1) because it was detected above the AWQC and most MDLs were greater than the AWQC of 0.25 μg/L. Cadmium was detected in 2 of 249 (1.2 percent) of the unfiltered samples and in 2 of 262 (0.76 percent) of the filtered groundwater samples. Cadmium concentrations in unfiltered and filtered groundwater samples range between 0.11 and1.7 μg/L. Unfiltered and filtered samples collected for purposes other than the RI were analyzed by Method 6010. MDLs for these results range between 0.3 and 30 μg/L. Samples collected for the RI used trace methods identified in the 100-D/H SAP (DOE/RL-2009-40). MDLs for these samples range between 0.055 and 0.2 μg/L. Cadmium was detected once above the AWQC in a filtered sample from 199-D4-84 (0.031 μg/L); however, cadmium was not detected in the corresponding unfiltered sample. Additionally, cadmium was analyzed for but not detected in five previous sample rounds. Cadmium was detected once above the AWQC in a filtered sample from 199-D5-93; however, cadmium was not detected in the corresponding unfiltered sample (less than 0.91 μg/L). Additionally, cadmium results (detected concentrations and MDLs) reported by Method 6010 are not accurate at concentrations at or near the AWQC. Cadmium concentrations above the AWQC are not associated with a specific location or with a trend. Detected concentrations of cadmium in unfiltered and filtered samples are below the 90th percentile Hanford Site background level of 0.92 μg/L. However, the MDLs reported using Method 6010 do not have sufficient accuracy to attain the AWQC. Based on the results of this evaluation and the results of the groundwater risk assessment presented in Section 6.3, cadmium is retained as a COPC for

further evaluation in the FS. Although all monitoring wells within the groundwater plume area were compared to the AWQC, only near-river wells would need to meet this criterion.

Cobalt was identified as a historical COPC in the 100-D/H Work Plan (DOE/RL-2008-46-ADD1) because it was detected above the action level, and some MDLs were greater than the action level of 2.6 µg/L. Cobalt was detected in 40 of 249 (16 percent) of unfiltered samples and 56 of 262 (21 percent) of filtered groundwater samples. Cobalt concentrations in unfiltered and filtered groundwater samples range between 0.099 and 32.5 µg/L. Unfiltered and filtered samples collected for purposes other than the RI were analyzed by Method 6010. MDLs for these results range between 2 and 70 µg/L, unfiltered concentrations ranged between 0.53 and 31 µg/L, and filtered concentrations ranged between 0.68 and 33 µg/L. Most cobalt results (42 of 47) reported by Method 6010 were flagged with a "B" qualifier or flagged with a "C" qualifier. The "C" qualifier indicates that the analyte was detected in both the sample and the associated QC blank, and the sample concentration was less than or equal to five times the blank concentration. Cobalt concentrations for unfiltered and filtered samples flagged with a "B" ranged between 2.9 and 19 µg/L (all results above action level). Cobalt concentrations for unfiltered and filtered samples flagged with a "C" ranged between 0.53 and 32.5 µg/L (28 of 30 results above action level). Samples collected for the RI used trace methods identified in the 100-D/H SAP (DOE/RL-2009-40). MDLs for these samples range between 0.05 and 0.22 µg/L, detected concentrations ranged between 0.099 and 3.0 µg/L for unfiltered samples, and between 0.1 and 2.9 µg/L for filtered samples. Most cobalt results (48 of 60) reported by trace methods were flagged with a "B" qualifier, flagged with a "C" qualifier, or flagged with both a "B" and a "C" qualifier. Cobalt concentrations for unfiltered and filtered samples flagged with a "B" qualifier ranged between 0.1 and 0.58 µg/L (all results below action level). Cobalt concentrations for unfiltered and filtered samples flagged with a "C" or a "BC" qualifier ranged between 0.099 and 2.8 µg/L (2 of 26 results above action level). Cobalt concentrations above the action level in samples collected for the RI were reported at 199-D4-84 and 199-D4-23. At 199-D4-84, cobalt was detected above the action level during two sampling rounds: one round where both the filtered (2.8 µg/L) and unfiltered (2.8 µg/L) samples were above the action level, and one round where the filtered (2.7 µg/L) sample was above the action level but the corresponding unfiltered sample (2.4 µg/L) was below the action level. Cobalt was analyzed for in three RI sampling rounds at 199-D4-23 with unfiltered and filtered concentrations ranging between 2.7 and 3.0 µg/L, all above the action level (two results were flagged with a "C" qualifier). The results of this evaluation indicate that cobalt has historically been detected in groundwater samples (25 percent of unfiltered samples and 32 percent of filtered samples) at concentrations up to 13 times greater than the action level; however, most cobalt results are flagged with either a "B" or C" qualifier. Cobalt concentrations above the action level are not associated with a specific location or with a trend. Cobalt results (detected concentrations and MDLs) reported by Method 6010 are not accurate at concentrations at or near the action level. Cobalt concentrations in filtered samples are above the 90th percentile Hanford Site background level of 0.92 µg/L. Based on the results of this evaluation and the results of the groundwater risk assessment presented in Section 6.3, cobalt is retained as a COPC for further evaluation in the FS.

Copper was identified as a historical COPC in the 100-D/H Work Plan (DOE/RL-2008-46-ADD1) because it was detected above the AWQC, and some MDLs were greater than the AWQC of 9.0 µg/L. Copper was detected in 56 of 249 (22 percent) of unfiltered samples and 44 of 262 (17 percent) of filtered groundwater samples. Copper concentrations in unfiltered samples range between 0.12 and 116 µg/L and filtered groundwater samples range between 0.17 and 18.7 µg/L. Unfiltered and filtered samples collected for purposes other than the RI were analyzed by Method 6010. MDLs for these results range between 0.7 and 70 µg/L, unfiltered concentrations ranged between 4.6 and 116, and filtered concentrations ranged between 4.7 and 18.7 µg/L. Many copper results (42 of 47 results) reported by Method 6010 were flagged with a "B" qualifier or flagged with "C" qualifier. Copper concentrations for unfiltered and filtered

samples flagged with a "B" ranged between 4.6 and 24 μg/L (5 of 14 results above AWQC). Copper concentrations for unfiltered and filtered samples flagged with a "C" ranged between 7.2 and 18.7 μg/L (25 of 28 results above AWQC). Copper was analyzed in eight sample rounds at 199-D5-15 where the highest copper concentration was reported. Copper concentrations above the AWQC were reported in two of eight sampling rounds at this well. One unfiltered result was reported at 116 μg/L and flagged with a "G" review qualifier, indicating the result has been reviewed and determined to be correct; the corresponding filtered result was reported below the MDL. Another unfiltered result at 199-D5-15 reported at 12 μg/L and flagged with a "C" qualifier; the corresponding filtered result (9.6 μg/L) was also above the action level. All remaining rounds at 199-D5-15 reported copper at concentrations less than the AWQC. Samples collected for the RI used trace methods identified in the 100-D/H SAP (DOE/RL-2009-40). MDLs for these samples range between 0.1 and 1.0 μg/L and detected concentrations in unfiltered and filtered samples range between 0.12 and 2.3 μg/L. The results of this evaluation indicate that copper has historically been detected in groundwater samples (31 percent of unfiltered samples and 25 percent of filtered samples) at concentrations up to 13 times greater than the AWQC, and many copper results are flagged with either a "B" or C" qualifier. Copper concentrations above the AWQC are not associated with a specific location or with a trend. Copper results (detected concentrations and MDLs) reported by Method 6010 are not accurate at concentrations at or near the AWQC. Copper concentrations associated with samples collected for the RI are less than the AWQC of 9 μg/L. Copper concentrations in filtered samples are above the 90th percentile Hanford Site background level of 0.81 μg/L. Based on the results of this evaluation and the results of the groundwater risk assessment presented in Section 6.3, copper is retained as a COPC for further evaluation in the FS. Although all monitoring wells within the groundwater plume area were compared to the AWQC, only near-river wells would need to meet this criterion.

Iron was detected above the secondary DWS of 300 μg/L in samples collected for the RI and is included in the nature and extent evaluation. Iron affects aesthetic qualities relating to public acceptance of drinking water. These regulations are not federally enforceable, but are intended as guidelines for states. Iron concentrations in groundwater are compared to the AWQC of 1,000 μg/L. Iron was detected in 154 of 249 (62 percent) of unfiltered and 66 of 262 (25 percent) of filtered groundwater samples. All samples were analyzed by Method 6010. MDLs for these samples ranged between 9 and 330 μg/L, detected iron concentrations in unfiltered samples ranged between 17 and 14,000 μg/L, and ranged between 9.7 and 348 μg/L in filtered samples. Iron concentrations greater than 1,000 μg/L were detected only in unfiltered samples from 199-D4-84, 199-D3-2, 199-D4-14, and 199-D5-93. All filtered samples were less than 1,000 μg/L. Iron present in wells 199-D4-84, 199-D3-2, 199-D4-14, and 199-D5-93 is likely associated with the ISRM barrier. Iron concentrations in filtered water samples are less than the 90th percentile Hanford Site background level of 570 μg/L. Based on the results of this evaluation and the results of the groundwater risk assessment presented in Section 6.3, iron is not retained as a COPC for further evaluation in the FS.

Lead was identified as a historical COPC in the 100-D/H Work Plan (DOE/RL-2008-46-ADD1) because it was detected above the water quality standards for surface waters of the state, and some MDLs were greater than the water quality standard of 2.1 μg/L. Lead was detected in 15 of 62 (24 percent) of unfiltered and 9 of 64 (14 percent) of filtered groundwater samples. Lead concentrations in unfiltered groundwater samples range between 0.2 and 0.52 μg/L and in filtered groundwater samples range between 0.29 and 3.7 μg/L. All MDLs were less than the action level of 2.1 μg/L. Lead was reported above the action level of 2.1 μg/L at two wells (199-D5-142 and 199-D8-101). A single detection of lead was reported at 199-D5-142 (2.24 μg/L) and at 199-D8-101 (3.66 μg/L) and both lead results were flagged with a "B" laboratory qualifier. The "B" qualifier indicates the analyte was detected at a value less than the required detection limit, but greater than or equal to the MDL, indicating that the result is an estimation. Both wells were installed for the RI and are the only results available for these wells. Additionally, lead from these wells was not analyzed by the trace methods identified in the 100-D/H SAP

but was analyzed by Method 6010, which is not accurate for measuring trace levels of lead. Lead concentrations in filtered samples are above the 90th percentile Hanford Site background level of 0.92 µg/L. Based on the results of this evaluation and the results of the groundwater risk assessment presented in Section 6.3, lead is retained as a COPC for further evaluation in the FS. Although all monitoring wells within the plume area were compared to the State water quality standard, only near-river wells would need to meet this standard.

Manganese was identified as a historical COPC in the100-D/H Work Plan (DOE/RL-2008-46-ADD1) because it was detected above the secondary DWS of 50 µg/L. Manganese affects aesthetic qualities relating to public acceptance of drinking water. These regulations are not federally enforceable, but are intended as guidelines for states. Manganese concentrations in groundwater are compared to the MTCA ("Surface Water Cleanup Standards" [WAC 173-340-730]) surface water cleanup level of 907 µg/L. Manganese was detected in 60 of 249 (24 percent) of unfiltered and 51 of 262 (19 percent) of filtered groundwater samples. Manganese concentrations in unfiltered samples range between 0.55 and 814 µg/L and filtered groundwater samples range between 0.43 and 163 µg/L. All manganese results (detected concentrations and MDLs) were less than the surface water cleanup level of 907 µg/L. Manganese concentrations in filtered samples are above the 90th percentile Hanford Site background level of 39 µg/L. Based on the results of this evaluation and the results of the groundwater risk assessment presented in Section 6.3, manganese is not identified as a COPC for further evaluation in the FS.

Mercury was identified as a historical COPC in the 100-D/H Work Plan (DOE/RL-2008-46-ADD1) because it was detected above the AWQC, and some MDLs were greater than the AWQC of 0.012 µg/L. The AWQC for mercury of 0.012 µg/L is based on the AWQC; however, it defaults to the EQL of 0.5 µg/L reported in the 100-D/H SAP (DOE/RL-2009-40) when the analytical method cannot achieve the AWQC. Mercury was not detected in any unfiltered groundwater samples (61 samples) and was detected in 3 of 61 (4.9 percent) filtered groundwater samples. All mercury results (detected concentrations and MDLs) were less than the EQL of 0.5 µg/L. Mercury concentrations in filtered samples are above the 90th percentile Hanford Site background level of 0.003 µg/L. Based on the results of this evaluation and the results of the groundwater risk assessment presented in Section 6.3, mercury is not retained as a COPC for further evaluation in the FS.

Nickel was identified as a historical COPC in the 100-D/H Work Plan (DOE/RL-2008-46-ADD1) because it was detected above the AWQC of 52 µg/L. Nickel was detected in 73 of 249 (29 percent) unfiltered and 47 of 262 (18 percent) filtered groundwater samples. Nickel concentrations in unfiltered samples range between 4 and 69 µg/L and filtered groundwater samples range between 0.59 and 68 µg/L. MDLs for these samples ranged between 2.2 and 66.5 µg/L. MDLs for samples collected prior to the RI and for the RI were less than the AWQC; five samples collected in 2011 were reported with MDLs greater than the AWQC. All samples were analyzed by Method 6010. Nickel concentrations in filtered samples above the AWQC of 52 µg/L were reported at 199-D4-84 and 199-D4-19. Nickel was detected once at 199-D4-84 in unfiltered (69 µg/L) and filtered (68 µg/L) samples from the same sample round; however, both of these results were flagged with a "C" qualifier. Nickel was analyzed in one previous and four subsequent sample rounds at 199-D4-84 at concentrations less than the AWQC. Nickel was detected once at 199-D4-19 in an unfiltered (54 µg/L) and filtered (56 µg/L) sample from the same sample round. Both of these results were flagged with a "C" qualifier. Nickel was analyzed in one previous and three subsequent sample rounds at 199-D4-19 at concentrations less than the AWQC. Nickel concentrations above the AWQC are not associated with a specific location or with a trend; additionally, these results were flagged with a "C" qualifier indicating they are associated with laboratory contamination. Nickel concentrations in filtered samples are above the 90th percentile Hanford Site background level of 1.6 µg/L. Based on the results of this evaluation and the results of the groundwater risk assessment presented in Section 6.3, nickel is retained as a COPC for further evaluation in the FS.

Selenium was identified as a historical COPC in the 100-D/H Work Plan (DOE/RL-2008-46-ADD1) because it was detected above the AWQC and some MDLs were greater than the AWQC of 5 µg/L. Selenium was detected in 69 of 73 (95 percent) of unfiltered and 71 of 75 (95 percent) of filtered groundwater samples. Selenium concentrations in unfiltered samples range between 0.38 and 10.7 µg/L and filtered samples range between 0.44 and 10.5 µg/L. All samples were analyzed by trace methods identified in the 100-D/H SAP (DOE/RL-2009-40). Except for one sample collected in 2011 after the RI, MDLs for these samples were reported at 0.6 µg/L. All selenium concentrations are less than or equal to the 90th percentile Hanford Site background level of 11 µg/L. Based on the results of this evaluation and the results of the groundwater risk assessment presented in Section 6.3, selenium is not retained as a COPC for further evaluation in the FS.

Silver was identified as a historical COPC in the 100-D/H Work Plan (DOE/RL-2008-46-ADD1) because it was detected above the action level, and some MDLs were greater than the action level of 2.6 µg/L. The action level for silver of 2.6 µg/L is based on "Water Quality Standards for Surface Waters of the State of Washington" (WAC 173-201A); however, it defaults to the EQL of 10 µg/L identified in the 100-D/H SAP (DOE/RL-2009-40) when the analytical method cannot achieve 2.6 µg/L. Silver was detected in 23 of 249 (9.2 percent) unfiltered and 20 of 262 (7.6 percent) filtered groundwater samples. Silver concentrations in unfiltered samples range between 0.13 and 34 µg/L and filtered groundwater samples range between 0.29 and 33 µg/L. All of the unfiltered and filtered samples collected for purposes other than the RI were analyzed by Method 6010. MDLs for this method ranged between 0.5 and 110 µg/L and detected concentrations ranged between 5.3 and 34 µg/L. All silver detected results reported by Method 6010 were flagged with a "B" qualifier, flagged with "C" qualifier, or flagged with both a "B" and a "C" qualifier. Silver concentrations for unfiltered and filtered samples flagged with a "B" ranged between 5.3 and 20 µg/L. Silver concentrations for unfiltered and filtered samples flagged with a "C" or "BC" ranged between 6.5 and 34 µg/L. Most of the samples collected for the RI were analyzed by trace methods and the remaining samples were analyzed by Method 6010 as identified in the 100-D/H SAP (DOE/RL-2009-40). MDLs for these samples range between 0.04 and 0.2 µg/L and detected concentrations ranged between 0.13 and 0.29 µg/L for unfiltered and filtered samples. All historical detections of silver are flagged with a combination of "B" and "C" qualifiers. Silver concentrations are not associated with a specific location or a trend and silver results (detected concentrations and MDLs) reported by Method 6010 are not accurate at concentrations at or near the action level. Silver concentrations associated with samples collected for the RI are not above 2.6 µg/L and are also below the 90th percentile Hanford Site background level of 5.3 µg/L. Based on the results of this evaluation and the results of the groundwater risk assessment presented in Section 6.3, silver is retained as a COPC for further evaluation in the FS.

Thallium was identified as a historical COPC in the 100-D/H Work Plan (DOE/RL-2008-46-ADD1) because it was detected above the action level and some MDLs were greater than the action level of 0.24 µg/L. The action level for thallium is 0.24 µg/L based on the CWA, "Human Health for Consumption of Water plus Organism;" however, it defaults to the EQL of 2 µg/L identified in the 100-D/H SAP (DOE/RL-2009-40) when the analytical method cannot achieve the action level. All samples were analyzed by trace methods identified in the 100-D/H SAP (DOE/RL-2009-40). Thallium was detected in 4 of 60 (6.7 percent) unfiltered samples and 6 of 62 (9.7 percent) of filtered groundwater samples. All thallium results (detected concentrations and MDLs) are less than the EQL of 2 µg/L, and thallium concentrations in unfiltered and filtered samples are below the 90th percentile Hanford Site background level of 1.7 µg/L. Based on the results of this evaluation and the results of the groundwater risk assessment presented in Section 6.3, thallium is not retained as a COPC for further evaluation in the FS.

Uranium was identified as a historical COPC in the 100-D/H Work Plan (DOE/RL-2008-46-ADD1) because it was detected above the DWS of 30 µg/L. Uranium was detected in all unfiltered groundwater samples (160 samples) and all filtered groundwater samples (20 samples). All uranium results (detected

concentrations and MDLs) were less than the DWS of 30 µg/L. Uranium concentrations in unfiltered and filtered samples are below the 90th percentile Hanford Site background level of 9.9 µg/L. Based on the results of this evaluation and the results of the groundwater risk assessment presented in Section 6.3, uranium is not retained as a COPC for further evaluation in the FS.

Vanadium was identified as a historical COPC in the 100-D/H Work Plan (DOE/RL-2008-46-ADD1) because it was detected above the action level of 80 µg/L. Vanadium was detected in 116 of 249 (47 percent) unfiltered samples and 106 of 262 (40 percent) of filtered groundwater samples. All samples were analyzed by Method 6010 as identified in the 100-D/H SAP (DOE/RL-2009-40). Samples collected for purposes other than the RI reported MDLs that range between 4.1 and 140 µg/L (17 samples collected in 2006 report MDLs equal to 140 µg/L). Samples collected for the RI reported MDLs that ranged between 4.1 and 12 µg/L. All detected vanadium concentrations are less than the action level of 80 µg/L. Vanadium concentrations in filtered samples are above the 90th percentile Hanford Site background level of 12 µg/L. Based on the results of this evaluation and the results of the groundwater risk assessment presented in Section 6.3, vanadium is not retained as a COPC for further evaluation in the FS.

Zinc was identified as a historical COPC in the 100-D/H Work Plan (DOE/RL-2008-46-ADD1) because it was detected above the action level and some MDLs were greater than the action level of 91 µg/L. Zinc was detected in 130 of 249 (52 percent) unfiltered and 120 of 262 (46 percent) filtered groundwater samples. All samples were analyzed by Method 6010 as identified in the 100-D/H SAP (DOE/RL-2009-40). MDLs for samples analyzed by Method 6010 ranged between 4 and 20 µg/L, detected concentrations for unfiltered samples ranged between 6.4 and 764 µg/L, and ranged between 5.2 and 329 µg/L for filtered samples. Some zinc results reported by Method 6010 were flagged with a "B" qualifier, with a "C" qualifier, or with both a "B" and a "C" qualifier. Zinc concentrations in unfiltered and filtered samples (62 of 250 detected results) flagged with a "B" ranged between 4.0 and 40 µg/L. Zinc concentrations in unfiltered and filtered samples (39 of 250 detected results) flagged with a "C" or with a "BC" ranged between 7 and 332 µg/L. Zinc concentrations above 91 µg/L in filtered samples were reported in 11 wells (199-D2-6, 199-D3-2, 199-D4-15, 199-D4-20, 199-D4-23, 199-D4-84, 199-D5-15, 199-D5-36, 199-D5-38, 199-D5-40, and 199-D5-44). A single occurrence of zinc at concentrations greater than the water quality standard were reported in filtered samples at seven of the 11 wells including 199-D2-6, 199-D4-23, 199-D5-15, 199-D5-36, 199-D5-38, 199-D5-40, and 199-D5-44. Zinc concentrations at these seven wells ranged between 96 and 215 µg/L and each sample was collected between November 14, 2006, and December 7, 2006. Between four and eight additional sample rounds at these locations reported zinc at concentrations less than the water quality standard. Similar results are observed in 199-D4-15 as zinc was reported at a concentration of 260 µg/L on November 20, 2006; zinc was reported above the water quality standard in a subsequent sample round at this well but this result was flagged with a "C" qualifier. These results suggest that zinc was potentially introduced in the laboratory after the sample was collected in the field between November and December 2006.

Zinc concentrations from filtered samples above the water quality standard were reported in four sampling rounds at 199-D3-2 with concentrations ranging between 116 and 230 µg/L; a sample collected in 2011 was less than the water quality standard (18 µg/L),. Zinc concentrations from filtered samples above the water quality standard were reported in five of six sample rounds at 199-D4-20, with concentrations ranging between 92 and 147 µg/L. Zinc concentrations from filtered samples above the water quality standard were reported in six sampling rounds at 199-D4-84, with concentrations ranging between 205 and 329 µg/L. The results of this evaluation indicate that zinc has historically been detected in groundwater samples. Zinc concentrations above the water quality standard in eight wells (199-D2-6, 199-D4-15, 199-D4-23, 199-D5-15, 199-D5-36, 199-D5-38, 199-D5-40, and 199-D5-44) are likely associated with a source of zinc that was introduced in the laboratory after the sample was collected in the field. Zinc concentrations are associated with a trend at 199-D3-2, 199-D4-20, and 199-D4-84 where

concentrations have ranged up to 3.6 times greater than the water quality standard of 91 µg/L. The presence of elevated concentrations of zinc in these wells is associated with the presence of sodium dithionite, which is associated with the ISRM barrier at the OU and is not the result of a site release. With the exception of the above wells, zinc concentrations are less than the water quality standard. Based on the results of this evaluation and the results of the groundwater risk assessment presented in Section 6.3, zinc is not retained as a COPC for further evaluation in the FS.

Summary of the 100-D Groundwater Plume Area Groundwater Evaluation. Table 4-8summarizes the outcome of the analysis. Contaminants that warrant further evaluation in the FS are chromium (total), Cr(VI), nitrate, and strontium-90.

Table 4-8. Summary of 100-D Groundwater Plume Area Contaminant Evaluation

Category	Constituent
Retained as a COPC	
Contaminant of potential concern[a] (contaminants that warrant further evaluation in FS)	Chromium, Cr(VI), nitrate, strontium-90
Detected at levels above action level and background (uncertain status)	Antimony, cadmium, carbon tetrachloride, chloroform, cobalt, copper, lead, nickel, silver,
Not retained as a COPC	
Detected above action level, but not associated with a Hanford Site release or isolated instances and/or suspect data	Sulfate, zinc
Detected in groundwater but below action level, EQL, or background concentrations	Arsenic, beryllium, bromodichloromethane, fluoride, gross alpha, iron, manganese, mercury, nitrite, selenium, technetium-99, tetrachloroethene, thallium, tritium, uranium, vanadium,
Not detected in groundwater	1,1-Dichloroethene, 1,2-dichloroethane, benzene, Cesium-137, cobalt-60, europium-152. europium-154, europium-155, trichloroethene, vinyl chloride

Based on evaluation of data collected January 2006 through December 2011.

Historical COPCs—100-H Groundwater Plume Area

The following subsections describe historical COPCs that were detected at least once in the 100-H groundwater plume area and include radionuclides, VOCs, anions, and metals. As described earlier, historical COPCs are either those analytes that were identified as COPCs in the 100-D/H SAP (DOE/RL-2009-40), or those analytes for which a maximum concentration exceeding an action level was reported during the spatial and temporal sampling (Section 6.3).

Radionuclides. Gross alpha and gross beta were identified as additional analytes in the 100-D/H SAP (DOE/RL-2009-40). Gross alpha and gross beta were analyzed in all of the RI monitoring network wells during all sampling rounds, as well as in the in the larger populations of wells over the longer period. Gross alpha was detected in 27 percent of the unfiltered groundwater samples and gross beta was detected in 94 percent of the unfiltered samples. Gross alpha was detected at concentrations ranging between 1.2 and 7.9 pCi/L, which are less than the DWS of 15 pCi/L. Gross beta was detected at concentrations ranging between 3.5 and 74 pCi/L; gross beta concentrations are consistent with the presence of tritium.

Based on the results of this evaluation and the results of the groundwater risk assessment presented in Section 6.3, gross alpha and gross beta are not retained as COPCs for further evaluation in the FS.

Technetium-99 was identified as a historical COPC in the 100-D/H Work Plan (DOE/RL-2008-46-ADD1) because it was detected above the DWS of 900 pCi/L. Technetium-99 was detected in 43 of 157 (27 percent) of the unfiltered groundwater samples with concentrations ranging between 6.5 and 870 pCi/L. The results of this evaluation indicate that technetium-99 has been historically detected at concentrations less than the DWS of 900 pCi/L. Based on the results of this evaluation and the results of the groundwater risk assessment presented in Section 6.3, technetium-99 is not retained as a COPC for further evaluation in the FS.

Tritium was identified as a historical COPC in the 100-D/H Work Plan (DOE/RL-2008-46-ADD1) because it was detected above the DWS of 20,000 pCi/L. Tritium was detected in 185 of 190 (97 percent) of the unfiltered groundwater samples with concentrations ranging between 7.3 and 11,000 pCi/L. The results of this evaluation indicate that tritium has been historically detected at concentrations less than the DWS of 20,000 pCi/L. Based on the results of this evaluation and the results of the groundwater risk assessment presented in Section 6.3, tritium is not retained as a COPC for further evaluation in the FS.

Volatile Organic Compounds. 1,2-Dichloroethane was detected above the action level of 0.38 µg/L in samples collected for the RI, and is included in the nature and extent evaluation. 1,2-Dichloroethane was not identified as a historical COPC in the 100-D/H Work Plan (DOE/RL-2008-46-ADD1) because it was never reported above the MDL in the 115 samples included in the 100-D/H Work Plan (DOE/RL-2008-46-ADD1) dataset that were analyzed. The action level for 1,2-dichloroethane of 0.38 µg/L is based on the CWA criteria for "Human Health for Consumption of Water plus Organism;" however, it defaults to the EQL of 5 µg/L reported in the 100-D/H SAP (DOE/RL-2009-40) when the analytical method cannot achieve a detection level of 0.38 µg/L. 1,1-Dichloroethene was not detected in any groundwater sample from any plume area and all MDLs are less than the EQL listed in the 100-D/H SAP (DOE/RL-2009-40). Based on the results of this evaluation and the groundwater risk assessment presented in Section 6.3, 1,2-dichloroethane is not identified as a COPC for further evaluation in the FS.

Bromodichloromethane was detected above the action level of 0.27 µg/L in samples collected for the RI and is included in the nature and extent evaluation. Bromodichloroethane was not identified as a historical COPC in the 100-D/H Work Plan (DOE/RL-2008-46-ADD1), and results for bromodichloromethane were not reported for samples collected for purposes other than the RI. Bromodichloromethane was not detected in any of the groundwater samples in the 100-H plume. All MDLs were less than the EQL of 5.0 µg/L. Bromodichloromethane is not retained as a COPC.

Carbon tetrachloride was identified as a historical COPC in the 100-D/H Work Plan (DOE/RL-2008-46-ADD1) because it was detected above the action level, and most MDLs were greater than the action level. The action level for carbon tetrachloride defaults to the EQL of 1 µg/L reported in the 100-D/H SAP (DOE/RL-2009-40) when the analytical method cannot achieve 0.23 µg/L. Carbon tetrachloride was detected in 2 of 55 unfiltered groundwater samples (3.6 percent). Carbon tetrachloride was detected once in 199-H4-10 (0.088 µg/L) at a concentration less than the EQL of 1 µg/L. Carbon tetrachloride was detected once in 199-H4-11 (2 µg/L) at a concentration greater than the EQL of 1 µg/L. Carbon tetrachloride was analyzed at 199-H4-10 in one previous and one subsequent sampling round and reported with nondetected concentrations less than or equal to the EQL. Carbon tetrachloride was analyzed in two subsequent sampling rounds at 199-H4-11 and reported at nondetected concentrations less than the EQL. All MDLs are less than or equal to the EQL listed in the 100-D/H SAP (DOE/RL-2009-40). The presence of carbon tetrachloride does not suggest it is associated with a specific

location or a trend. Based on the results of this evaluation and the groundwater risk assessment presented in Section 6.3, carbon tetrachloride is retained as a COPC.

Chloroform was identified as a historical COPC in the 100-D/H Work Plan (DOE/RL-2008-46-ADD1) because it was detected above the action level, and most MDLs were greater than the action level. The action level for chloroform defaults to the EQL of 5 μg/L reported in the 100-D/H SAP (DOE/RL-2009-40) when the analytical method cannot achieve the action level. Chloroform was detected in 39 of 55 (71 percent) of unfiltered groundwater samples with concentrations ranging between 0.32 μg/L and 2 μg/L. All results (detected concentrations and MDLs) are less than the EQL of 5 μg/L. Based on the results of this evaluation and the groundwater risk assessment presented in Section 6.3, chloroform is not retained as a COPC.

Tetrachloroethene was not identified as a historical COPC in the 100-D/H Work Plan (DOE/RL-2008-46-ADD1). Tetrachloroethene was not detected in any of the groundwater samples in the 100-H plume. All MDLs are less than the EQL of 5μg/L. Based on the results of this evaluation and the groundwater risk assessment presented in Section 6.3, tetrachloroethene is not retained as a COPC.

Anions. Fluoride was identified as a historical COPC in the 100-D/H Work Plan (DOE/RL-2008-46-ADD1) because it was detected above the action level, and some MDLs were greater than the action level of 960 μg/L. Fluoride was detected in 147 of 209 (70 percent) of unfiltered groundwater samples, with concentrations ranging between 49 μg/L and 240 μg/L. All fluoride results (detected concentrations and MDLs) are less than the action level. Fluoride concentrations in unfiltered samples are also less than the 90th percentile Hanford Site background level of 1,047 μg/L. Based on the results of this evaluation and the results of the groundwater risk assessment presented in Section 6.3, fluoride is not identified as a COPC for further evaluation in the FS.

Nitrite was identified as a historical COPC in the 100-D/H Work Plan (DOE/RL-2008-46-ADD1) because it was detected above the DWS, and some MDLs were greater than the DWS of 3,300 μg/L. Nitrite was detected in 31 of 201 (15 percent) of unfiltered samples with concentrations ranging between 28 and 2,270 μg/L. All nitrite results (detected concentrations and MDLs) were less than the DWS of 3,300 μg/L. Nitrite concentrations in unfiltered samples are greater than the 90th percentile Hanford Site background level of 94 μg/L. Based on the results of this evaluation and the results of the groundwater risk assessment presented in Section 6.3, nitrite is not retained as a COPC for further evaluation in the FS.

Sulfate was identified as a historical COPC in the 100-D/H Work Plan (DOE/RL-2008-46-ADD1) because it was detected above the secondary DWS of 250,000 μg/L. Sulfate was detected in 100 percent of unfiltered groundwater samples (173 samples) with concentrations ranging between 16,200 and 149,000 μg/L. All sulfate results are less than the secondary DWS of 250,000 μg/L. Sulfate concentrations in unfiltered samples are greater than the 90th percentile Hanford Site background level of 47,000 μg/L. Based on the results of this evaluation and the results of the groundwater risk assessment presented in Section 6.3, sulfate is not retained as a COPC for further evaluation in the FS.

Metals. Antimony was identified as a historical COPC in the 100-D/H Work Plan (DOE/RL-2008-46-ADD1) because it was detected above the action level, and most MDLs were greater than the action level. Antimony was detected in 16 of 170 unfiltered samples (9.4 percent) and 15 of 184 (8.1 percent) of the filtered groundwater samples. Antimony concentrations in unfiltered and filtered groundwater samples ranged between 0.34 and 42 μg/L. Most unfiltered and filtered samples collected for purposes other than the RI were analyzed by Method 6010. MDLs for these results range between 4 and 72 μg/L and detected concentrations range between 2.7 and 42 μg/L. Five of seven antimony detected results reported by Method 6010 were flagged with a "Y" review qualifier. For samples analyzed using trace methods identified in the 100-D/H SAP (DOE/RL-2009-40), the MDLs ranged between 0.3 and 0.6 μg/L and the

detected concentrations range between 0.34 and 1.0 μg/L. The results of this evaluation indicate that antimony has historically been detected in groundwater samples at a low frequency of detection (11 percent in unfiltered samples and 9.3 percent in filtered samples) with concentrations up to 7.5 times greater than the standard. Some historical detections of antimony are flagged with a "Y" qualifier. Antimony concentrations are not associated with a specific location or a trend and antimony results (detected concentrations and MDLs) reported by Method 6010 are not accurate at concentrations at or near the action level. Antimony concentrations reported by methods identified in the 100-D/H SAP (DOE/RL-2009-40) are not above the action of 5.6 μg/L. All antimony concentrations are below the 90th percentile Hanford Site background level of 55 μg/L. Based on the results of this evaluation and the results of the groundwater risk assessment presented in Section 6.3, antimony is retained as a COPC and warrants further evaluation in the FS.

Arsenic was identified as a historical COPC in the 100-D/H Work Plan (DOE/RL-2008-46-ADD1) because it was detected above the action level, and all MDLs were greater than the action level. The action level for arsenic of 0.018 μg/L is based on the national recommended water quality criteria for "Human Health for Consumption of Water plus Organism;" however, the action level defaults to the EQL of 4 μg/L reported in the 100-D/H SAP (DOE/RL-2009-40) when the analytical method cannot achieve 0.018 μg/L. Arsenic was detected in 52 of 53 (98 percent) of the unfiltered and 54 of 56 (96 percent) of the filtered groundwater samples. Arsenic concentrations range between 1.4 and 3.7 μg/L in unfiltered groundwater and between 1.3 and 3.6 μg/L in filtered groundwater samples. Minimum, maximum, and 90th percentile concentrations for (filtered) background concentrations of arsenic are 0.5, 8.8, and 7.85 μg/L, respectively. All arsenic concentrations in unfiltered and filtered samples are less than the 90th percentile Hanford Site background concentration. Based on the results of this evaluation and the results of the groundwater risk assessment presented in Section 6.3, arsenic is not retained as a COPC for further evaluation in the FS.

Beryllium was identified as a historical COPC in the 100-D/H Work Plan (DOE/RL-2008-46-ADD1) because it was detected above the DWS, and most MDLs were greater than the DWS of 4 μg/L. Beryllium was detected in 2 of 170 (1.2 percent) of the unfiltered samples and in 8 of 184 (4.4 percent) of the filtered groundwater samples. Beryllium concentrations in unfiltered and filtered groundwater samples range between 0.072 and 1.2 μg/L. All beryllium results (detected concentrations and MDLs) are less than the DWS of 4 μg/L. In addition, all beryllium concentrations are less than the 90th percentile Hanford Site background level of 2.3 μg/L. Based on the results of this evaluation and the results of the groundwater risk assessment presented in Section 6.3, beryllium is not retained as a COPC for further evaluation in the FS.

Cadmium was identified as a historical COPC in the 100-D/H Work Plan (DOE/RL-2008-46-ADD1) because it was detected above the AWQC, and most MDLs were greater than the AWQC of 0.25 μg/L. Cadmium was detected in 2 of 170 (1.2 percent) of the unfiltered samples and in 3 of 184 (1.6 percent) of the filtered groundwater samples. Cadmium concentrations in unfiltered and filtered groundwater samples range between 0.39 and 4.1 μg/L. Almost all unfiltered and filtered samples collected for purposes other than the RI were analyzed by Method 6010. MDLs for these results range between 0.3 and 4 μg/L. For samples analyzed using the trace methods identified in the 100-D/H SAP (DOE/RL-2009-40), the MDLs range between 0.1 and 0.2 μg/L. Cadmium was detected above the AWQC in four wells (199-H3-5, 199-H4-5, 199-H4-9, and 199-H4-13). A single cadmium concentration above the AWQC was reported in one of four sampling rounds conducted at 199-H3-5; the unfiltered sample (4.1 μg/L) was flagged with a "B" qualifier. A single cadmium concentration above the AWQC was reported in one of 11 sample rounds conducted at 199-H4-5; the filtered sample (1.2 μg/L) was flagged with a "B" qualifier and in the corresponding unfiltered sample, cadmium was reported as not detected. Cadmium concentrations above the AWQC were reported in one of 12 sample rounds conducted at 199-H4-9; both the unfiltered (2.4 μg/L) and filtered sample (1.8 μg/L) were flagged with a "B" qualifier. Cadmium concentrations above the AWQC were reported in one of 11 sample rounds conducted at 199-H4-13; the filtered sample

(0.39 μg/L) was flagged with a "B" qualifier and in the corresponding unfiltered sample, cadmium was reported as not detected. The results of this evaluation indicate that cadmium has historically been detected in groundwater samples at a low frequency (1.3 percent in unfiltered and 1.9 percent in filtered samples). Cadmium concentrations above the AWQC are not associated with a specific location or with a trend. Cadmium concentrations in unfiltered and filtered samples are above the 90th percentile Hanford Site background level of 0.92 μg/L. However, cadmium concentrations above background are considered estimated concentrations because all results are flagged with a "B" qualifier. Additionally, cadmium MDLs reported by Method 6010 are not accurate at concentrations at or near the AWQC. Based on the results of this evaluation and the results of the groundwater risk assessment presented in Section 6.3, cadmium is retained as a COPC for further evaluation in the FS.

Chromium was identified as a historical COPC in the 100-D/H Work Plan (DOE/RL-2008-46-ADD1) because it was detected above the AWQC of 65 μ/L. Chromium was detected in 131 of 170 (77 percent) of unfiltered samples and 126 of 184 (66 percent) of filtered groundwater samples. Chromium concentrations in unfiltered samples range between 4.8 and 215 μg/L and filtered samples range between 4.9 and 79 μg/L. Chromium concentrations in filtered samples above the AWQC of 65 μg/L were reported at 199-H3-4 and 199-H3-5. Filtered chromium concentrations above the AWQC were reported in one of three sample rounds at 199-H3-4; the filtered sample (66 μg/L) was reported at a slightly higher concentration than the corresponding unfiltered sample (62 μg/L). Filtered chromium concentrations above the AWQC were reported in all four samples at 199-H3-5, with concentrations ranging between 71 and 79 μg/L. Filtered chromium concentrations at 199-H3-5 appear to be associated with a trend. Chromium concentrations at all other wells do not appear to be associated with a location or a trend. Chromium concentrations in filtered and unfiltered samples are greater than the 90th percentile Hanford Site background level of 2.4 μg/L. Based on the results of this evaluation and the results of the groundwater risk assessment presented in Section 6.3, chromium is retained as a COPC for further evaluation in the FS.

Cobalt was identified as a historical COPC in the 100-D/H Work Plan (DOE/RL-2008-46-ADD1) because it was detected above the action level, and some MDLs were greater than the action level of 2.6 μg/L. Cobalt was detected in 21 of 170 (12 percent) unfiltered samples and 37 of 184 (20 percent) filtered groundwater samples. Cobalt concentrations in unfiltered and filtered groundwater samples range between 0.062 and 29 μg/L. Unfiltered and filtered samples collected for purposes other than the RI were analyzed by Method 6010. The MDLs for these results range between 0.5 and 7 μg/L, unfiltered concentrations ranged between 4.1 and 27 μg/L, and filtered concentrations ranged between 4.3 and 29 μg/L. Most cobalt results (9 of 11) reported by Method 6010 were flagged with a "B" qualifier or flagged with "C" qualifier. Cobalt concentrations for unfiltered and filtered samples flagged with a "B" ranged between 4.1 and 7.9 μg/L (all results above action level). Cobalt concentrations for unfiltered and filtered samples flagged with a "C" ranged between 25 and 29 μg/L (all results above action level). For samples analyzed using trace methods identified in the 100-D/H SAP (DOE/RL-2009-40), the MDLs ranged between 0.05 and 2 μg/L, detected concentrations for unfiltered samples ranged between 0.062 and 0.9 μg/L, and concentrations ranged between 0.083 and 2.8 μg/L for filtered samples. A cobalt concentration was reported above the action level at 199-H3-5 for a sample analyzed by trace methods identified in the 100-D/H SAP (DOE/RL-2009-40). Cobalt concentrations above the action level were reported in one of three sample rounds conducted at 199-H3-5; the filtered sample (2.8 μg/L) was flagged with a "B" qualifier and the corresponding unfiltered sample was not detected. Cobalt concentrations above the action level are not associated with a specific location or with a trend. Cobalt results (detected concentrations and MDLs) reported by Method 6010 are not accurate at concentrations at or near the action level. Cobalt concentrations in filtered samples are above the 90th percentile Hanford Site background level of 0.92 μg/L. Based on the results of this evaluation and the results of the groundwater risk assessment presented in Section 6.3, cobalt is retained as a COPC for further evaluation in the FS.

Copper was identified as a historical COPC in the 100-D/H Work Plan (DOE/RL-2008-46-ADD1) because it was detected above the AWQC, and some MDLs were greater than the AWQC of 9.0 µg/L. Copper was detected in 41 of 170 (24 percent) unfiltered samples and 21 of 162 (13 percent) of filtered groundwater samples. Copper concentrations in unfiltered samples range between 0.12 and 28 µg/L and filtered groundwater samples range between 0.17 and 13.3 µg/L. Unfiltered and filtered samples collected for purposes other than the RI were analyzed by Method 6010. MDLs for these results ranged between 0.7 and 7 µg/L, unfiltered concentrations ranged between 1.4 and 28 µg/L, and filtered concentrations ranged between 1.2 and 13 µg/L. Many copper results (16 of 22 results) reported by Method 6010 were flagged with a "B" qualifier, flagged with "C" qualifier, or flagged with a "BC" qualifier. Copper concentrations for unfiltered and filtered samples flagged with a "B" ranged between 2.9 and 6.4 µg/L (no results above AWQC). Copper concentrations for unfiltered and filtered samples flagged with a "C" or a "BC" ranged between 1.2 and 13 µg/L (3 of 11 results above AWQC). Copper was analyzed in two sample rounds at 199-H4-18 where the highest copper concentrations were reported. Copper concentrations above the AWQC were reported in both sample rounds at this well; the unfiltered copper results ranged between 25 and 28 µg/L and the corresponding filtered sample results were reported with MDLs less than the AWQC. Copper concentrations above the AWQC from 199-H4-5 and 199-H4-13 were flagged with a "C;" the remaining 10 sampling rounds at each well reported all nondetects with MDLs below the AWQC. For samples analyzed using trace methods identified in the 100-D/H SAP (DOE/RL-2009-40), the MDLs range between 0.1 and 0.2 µg/L and detected concentrations in unfiltered and filtered samples ranged between 0.12 and 2.8 µg/L. and. Copper concentrations above the AWQC are not associated with a specific location or with a trend where many copper results reported by Method 6010 are flagged with either a "B" or C" qualifier. Copper results (detected concentrations and MDLs) reported by Method 6010 are not accurate at concentrations at or near the AWQC. Copper concentrations associated with samples collected for the RI are less than the AWQC of 9 µg/L. Copper concentrations in filtered samples are above the 90th percentile Hanford Site background level of 0.81 µg/L. Based on the results of this evaluation and the results of the groundwater risk assessment presented in Section 6.3, copper is retained as a COPC for further evaluation in the FS.

Iron was detected above the secondary DWS of 300 µg/L in samples collected for the RI and is included in the nature and extent evaluation. The MCL for iron is based on the secondary DWS. Iron affects aesthetic qualities relating to public acceptance of drinking water. These regulations are not federally enforceable, but are intended as guidelines for states. Iron concentrations in groundwater are compared to the AWQC of 1,000 µg/L. Iron was detected in 126 of 170 (74 percent) of unfiltered and 88 of 184 (47 percent) of filtered groundwater samples. All samples were analyzed by Method 6010. MDLs for these samples ranged between 9 and 50 µg/L, detected iron concentrations in unfiltered samples ranged between 10 and 7,840 µg/L and ranged between 9.5 and 426 µg/L in filtered samples. Iron concentrations greater than 1,000 µg/L were detected only in unfiltered samples from 199-H4-3 (3 of 9 rounds), 199-H4-9 (2 of 12 rounds), and 199-H4-18 (1 of 2 rounds). Iron concentrations in filtered water samples are less than the background level of 570 µg/L. Based on the results of this evaluation and the results of the groundwater risk assessment presented in Section 6.3, iron is not retained as a COPC for further evaluation in the FS.

Lead was identified as a historical COPC in the 100-D/H Work Plan (DOE/RL-2008-46-ADD1) because it was detected above the action level, and some MDLs were greater than the action level of 2.1 µg/L. Lead was detected in 5 of 51 (9.8 percent) unfiltered and 6 of 53 (11 percent) filtered groundwater samples. Lead concentrations in unfiltered samples range between 0.22 and 0.71 µg/L and filtered groundwater samples range between 0.21 and 2.5 µg/L. Except for three sample collected for purposes other than the RI, all samples were analyzed by trace methods identified in the 100-D/H SAP (DOE/RL-2009-40). MDLs for the trace methods were less than the action level of 2.1 µg/L. Lead results (detected concentrations and MDLs) reported by Method 6010 are not accurate at concentrations at or near the AWQC. Lead

concentrations in unfiltered and filtered samples are less than the 90th percentile Hanford Site background level of 0.92 µg/L. Based on the results of this evaluation and the results of the groundwater risk assessment presented in Section 6.3, lead is retained as a COPC for further evaluation in the FS.

Manganese was identified as a historical COPC in the 100-D/H Work Plan (DOE/RL-2008-46-ADD1) because it was detected above the secondary DWS of 50 µg/L. The action level for manganese is based on the secondary DWS. Manganese affects aesthetic qualities relating to public acceptance of drinking water. These regulations are not federally enforceable, but are intended as guidelines for states. Manganese concentrations in groundwater are compared to the MTCA ("Surface Water Cleanup Standards" [WAC 173-340-730]) level of 907 µg/L. Manganese was detected in 38 of 170 (22 percent) of unfiltered and 39 of 184 samples (21 percent) of filtered groundwater samples. Manganese concentrations in unfiltered samples range between 0.53 and 156 and filtered groundwater samples range between 0.53 and 93 µg/L. All manganese results (detected concentrations and MDLs) were less than the surface water cleanup level of 907 µg/L Manganese concentrations in filtered samples are above the 90th percentile Hanford Site background level of 39 µg/L. Based on the results of this evaluation and the results of the groundwater risk assessment presented in Section 6.3, manganese is not retained as a COPC for further evaluation in the FS.

Mercury was identified as a historical COPC in the 100-D/H Work Plan (DOE/RL-2008-46-ADD1) because it was detected above the AWQC, and some MDLs were greater than the AWQC of 0.012 µg/L. The action level for mercury of 0.012 µg/L is based on the AWQC; however, it defaults to the EQL of 0.5 µg/L reported in the 100-D/H SAP (DOE/RL-2009-40) when the analytical method cannot achieve the AWQC. Mercury was detected in 1 of 50 (2 percent) unfiltered and 1 of 49 (2 percent) filtered groundwater samples. All mercury results (detected concentrations and MDLs) are less than the EQL of 0.5 µg/L. Mercury concentrations in filtered samples are above the 90th percentile Hanford Site background level of 0.003 µg/L. Based on the results of this evaluation and the results of the groundwater risk assessment presented in Section 6.3, mercury is not retained as a COPC for further evaluation in the FS.

Nickel was identified as a historical COPC in the 100-D/H Work Plan (DOE/RL-2008-46-ADD1) because it was detected above the AWQC of 52 µg/L. Nickel was detected in 61 of 170 (36 percent) unfiltered samples and 45 of 184 (24 percent) filtered groundwater samples. Nickel concentrations in unfiltered samples range between 4 and 37 µg/L and filtered groundwater samples range between 0.72 and 36 µg/L. MDLs for these samples ranged between 2.4 and 66.5 µg/L. MDLs for samples collected prior to the RI and for the RI were less than the AWQC; 12 samples collected in 2011 were reported with MDLs greater than the AWQC. All samples were analyzed by Method 6010. All detected concentrations and most MDLs were less than the AWQC. Nickel concentrations in filtered samples are above the 90th percentile Hanford Site background level of 1.6 µg/L. Based on the results of this evaluation and the results of the groundwater risk assessment presented in Section 6.3, nickel is retained as a COPC for further evaluation in the FS.

Selenium was identified as a historical COPC in the 100-D/H Work Plan (DOE/RL-2008-46-ADD1) because it was detected above the AWQC, and some MDLs were greater than the AWQC of 5 µg/L. Selenium was detected in 51 of 53 (96 percent) unfiltered and 51 of 55 (93 percent) filtered groundwater samples. All selenium results (both detected concentrations and MDLs) are less than the AWQC. All selenium concentrations and MDLs are less than the AWQC. All selenium concentrations are less than the 90th percentile Hanford Site background level of 11 µg/L. Based on the results of this evaluation and the results of the groundwater risk assessment presented in Section 6.3, selenium is not identified as a COPC for further evaluation in the FS.

Silver was identified as a historical COPC in the 100-D/H Work Plan (DOE/RL-2008-46-ADD1) because it was detected above the action level, and some MDLs were greater than the action level of 2.6 µg/L. The action level for silver of 2.6 µg/L is based on "Water Quality Standards for Surface Waters of the

State of Washington" (WAC 173-201A); however, it defaults to the EQL of 10 µg/L identified in the 100-D/H SAP (DOE/RL-2009-40) when the analytical method cannot achieve 2.6 µg/L. Silver was detected in 10 of 170 (5.9 percent) unfiltered and 7 of 184 (3.8 percent) filtered groundwater samples. Silver concentrations in unfiltered samples range between 0.1 and 30 µg/L and filtered groundwater samples ranged between 0.32 and 33 µg/L. Most of the unfiltered and filtered samples collected for purposes other than the RI were analyzed by Method 6010. MDLs for this method ranged between 0.5 and 11 µg/L and detected concentrations ranged between 6.1 and 33 µg/L. All but two silver results reported by Method 6010 were flagged with a "B" qualifier, flagged with "C" qualifier, or flagged with both a "B" and a "C" qualifier. Silver concentrations for unfiltered and filtered samples flagged with a "B" ranged between 6.2 and 17 µg/L. Silver concentrations for unfiltered and filtered samples flagged with a "C" or "BC" ranged between 6.1 and 33 µg/L. Silver concentrations for the two sample results without qualifiers were 7.8 and 8.2 µg/L. The samples collected for the RI were analyzed by trace methods, although Method 6010 is identified in the 100-D/H SAP (DOE/RL-2009-40). MDLs for these samples range between 0.1 and 0.2 µg/L and detected concentrations ranged between 0.32 and 1.0 µg/L for unfiltered and filtered samples. Most historical detections of silver are flagged with a combination of "B" and "C" qualifiers. Silver concentrations are not associated with a specific location or a trend and silver results (detected concentrations and MDLs) reported by Method 6010 are not accurate at concentrations at or near the action level. Silver concentrations associated with samples collected for the RI are not above the standard of 2.6 µg/L and are also below the 90th percentile Hanford Site background level of 5.3 µg/L. Based on the results of this evaluation and the results of the groundwater risk assessment presented in Section 6.3, silver is retained as a COPC for further evaluation in the FS.

Thallium was identified as a historical COPC in the 100-D/H Work Plan (DOE/RL-2008-46-ADD1) because it was detected above the action level and some MDLs were greater than the action level of 0.24 µg/L. The standard for thallium of 0.24 µg/L is based on the CWA, "Human Health for Consumption of Water plus Organism;" however, it defaults to the EQL of 2 µg/L identified in the 100-D/H SAP (DOE/RL-2009-40) when the analytical method cannot achieve the action level. Thallium was detected in 2 of 49 (4.1 percent) of unfiltered and 4 of 51 (7.8 percent) of filtered groundwater samples. All thallium results (detected concentrations and MDLs) are less than the EQL of 2 µg/L and thallium concentrations in unfiltered and filtered samples are below the 90th percentile Hanford Site background level of 1.7 µg/L. Based on the results of this evaluation and the results of the groundwater risk assessment presented in Section 6.3, thallium is not retained as a COPC for further evaluation in the FS.

Uranium was identified as a historical COPC in the 100-D/H Work Plan (DOE/RL-2008-46-ADD1) because it was detected above the DWS of 30 µg/L. Uranium was detected in 155 of 158 (98 percent) unfiltered samples and all 28 of the filtered groundwater samples. Uranium concentrations range between 0.38 and 86 µg/L in unfiltered groundwater samples and between 0.42 and 13 µg/L in filtered groundwater samples. All MDLs were less than the DWS of 30 µg/L. Uranium was reported above the DWS in an unfiltered sample collected at 199-H4-3 (86 µg/L). However, 16 subsequent sampling rounds reported unfiltered uranium concentrations ranging between 6.7 and 29 µg/L (all less than the DWS). The uranium concentration at 199-H4-3 that is greater than the DWS is not associated with a trend. Uranium concentrations in unfiltered and filtered samples are greater than the 90th percentile Hanford Site background level of 9.9 µg/L. Based on the results of this evaluation and the results of the groundwater risk assessment presented in Section 6.3, uranium is not retained as a COPC for further evaluation in the FS.

Vanadium was identified as a historical COPC in the 100-D/H Work Plan (DOE/RL-2008-46-ADD1) because it was detected above the action level of 80 µg/L. Vanadium was detected in 40 of 170 (24 percent) unfiltered and 43 of 184 (23 percent) filtered groundwater samples. All vanadium results (detected concentrations and MDLs) are less than the action level of 80 µg/L. Vanadium concentrations in filtered samples are above the 90th percentile Hanford Site background level of 12 µg/L. Based on the

results of this evaluation and the results of the groundwater risk assessment presented in Section 6.3, vanadium is not retained as a COPC for further evaluation in the FS.

Zinc was identified as a historical COPC in the 100-D/H Work Plan (DOE/RL-2008-46-ADD1) because it was detected above the action level, and some MDLs were greater than the action level of 91 µg/L. Zinc was detected in 51 of 170 (30 percent) unfiltered and 37 of 184 (20 percent) filtered groundwater samples. All samples were analyzed by Method 6010. The MDLs ranged between 4 and 20 µg/L, detected concentrations for unfiltered samples ranged between 1 and 236 µg/L, and ranged between 1.6 and 88 µg/L for filtered samples. Some zinc results (57 of 88 results) reported by Method 6010 were flagged with a "B" qualifier, a "C" qualifier, or with both a "B" and a "C" qualifier. Zinc concentrations in unfiltered and filtered samples (33 of 88 results) flagged with a "B" ranged between 4 and 24 µg/L. Zinc concentrations in unfiltered and filtered samples (24 of 88 results) flagged with a "C" or with a "BC" ranged between 1 and 120 µg/L. Only one zinc result was reported above 91 µg/L and it was from an unfiltered sample at 199-H4-13 (236 µg/L); however, the corresponding filtered sample (88 µg/L) was less than the action level (91 µg/L). All subsequent detected concentrations and MDLs were less than the action level (nine sampling rounds) at this well. The zinc concentration above the action level is not associated with a specific location or with a trend and all remaining zinc results were flagged with a "B" or a "C" laboratory qualifier. Zinc concentrations are also greater than the 90th percentile Hanford Site background level of 22 µg/L. Based on the results of this evaluation and the results of the groundwater risk assessment presented in Section 6.3, zinc is retained as a COPC for further evaluation in the FS.

Summary of the 100-H Groundwater Plume Area Groundwater Evaluation. Table 4-9 summarizes the outcome of the analysis. Contaminants that warrant further evaluation in the FS are chromium (total), Cr(VI), nitrate, and strontium-90.

Table 4-9. Summary of 100-H Groundwater Plume Area Contaminant Evaluation

Category	Constituent
Retained as a COPC	
Contaminant of potential concern[a] (contaminants that warrant further evaluation in FS)	Chromium, Cr(VI), nitrate, strontium-90
Detected at levels above action level and background (uncertain status)	Antimony, cadmium, carbon tetrachloride, cobalt, copper, lead, nickel, silver, zinc
Not retained as a COPC	
Detected above action level, but not associated with a Hanford Site release or isolated instances and/or suspect data	uranium
Detected in groundwater but below action level, EQL, or background concentrations	Arsenic, beryllium, chloroform, fluoride, gross alpha, gross beta, manganese, mercury, nitrite, selenium, sulfate, technetium-99, thallium, tritium, vanadium
Not detected in groundwater	1,1-Dichloroethene, 1,2-dichloroethane, benzene, bromodichloromethane, cesium-137, cobalt-60, europium-152. europoium-154, europium-155, tetrachloroethene, trichloroethene, vinyl chloride

Based on evaluation of data collected January 2006 through December 2011.

Historical COPCs—Horn Groundwater Plume Area

The following subsections describe the analytes of interest that were detected at least once in the Horn groundwater plume area and include radionuclides, VOCs, anions, and metals. As described earlier, analytes of interest are either those analytes that were identified as COPCs in the 100-D/H SAP (DOE/RL-2009-40) or those analytes for which a maximum concentration exceeding an action level was reported during the spatial and temporal sampling (Section 6.3).

Radionuclides. Gross alpha and gross beta were identified as additional analytes in the 100-D/H SAP (DOE/RL-2009-40). Gross alpha and gross beta were analyzed in all of the RI monitoring network wells during all sampling rounds, as well as in the larger populations of wells over the longer period. Gross alpha was detected in 26 percent of the unfiltered groundwater samples and gross beta was detected in 79 percent of the unfiltered samples. Gross alpha was detected at concentrations ranging between 1.4 and 7.8 pCi/L, which is less than the DWS of 15 pCi/L. Gross beta was detected at concentrations ranging between 2.4 and 21 pCi/L; gross beta concentrations are consistent with the presence of tritium. Based on the results of this evaluation and the results of the groundwater risk assessment presented in Section 6.3, gross alpha and gross beta are not retained as COPCs for further evaluation in the FS.

Strontium-90 was identified as a historical COPC in the 100-D/H Work Plan (DOE/RL-2008-46-ADD1) because it was detected above the DWS. Strontium-90 was detected in 2 of 56 (3.6 percent) of the unfiltered groundwater samples with concentrations of 2.4 and 4.2 pCi/L. All strontium-90 results (detected concentrations and MDLs) were all less than the DWS of 8 pCi/L. The results of this evaluation and the groundwater risk assessment presented in Section 6.3, strontium-90 is not retained as a COPC for further evaluation in the FS.

Technetium-99 was identified as a historical COPC in the 100-D/H Work Plan (DOE/RL-2008-46-ADD1) because it was detected above the DWS. Technetium-99 was detected in 1 of 54 (1.9 percent) of the unfiltered groundwater samples at a concentration of 12 pCi/L. All technetium-99 results (detected concentrations and MDLs) are less than the DWS of 900 pCi/L. Based on the results of this evaluation and the groundwater risk assessment presented in Section 6.3, technetium-99 is not retained as a COPC for further evaluation in the FS.

Tritium was identified as a historical COPC in the 100-D/H Work Plan (DOE/RL-2008-46-ADD1) because it was detected above the DWS. Tritium was detected in 177 of 187 (95 percent) of the unfiltered groundwater samples with concentrations ranging between 200 and 6,030 pCi/L. All tritium results (detected concentrations and MDLs) are less than then DWS of 20,000 pCi/L. Based on the results of this evaluation and the groundwater risk assessment presented in Section 6.3, tritium is not retained as a COPC for further evaluation in the FS.

Volatile Organic Compounds. 1,2-Dichloroethane was detected above the action level of 0.38 µg/L in samples collected for the RI and is included in the nature and extent evaluation. 1,2-Dichloroethane was not identified as a historical COPC in the 100-D/H Work Plan (DOE/RL-2008-46-ADD1). 1,2-Dicholoroethane was not detected in any groundwater sample in the horn area plume. All MDLs were less than the EQL of 5 µg/L. Based on the results of this evaluation and the groundwater risk assessment presented in Section 6.3, 1,2-dichloroethane is not retained as a COPC for further evaluation in the FS.

Bromodichloromethane was detected above the action level of 0.27 µg/L in samples collected for the RI and is included in the nature and extent evaluation. Bromodichloroethane was not identified as a historical COPC in the 100-D/H Work Plan (DOE/RL-2008-46-ADD1). Bromodichloromethane was not detected in any of the groundwater samples in the horn area plume. All MDLs were less than the EQL of 5 µg/L. Based on the results of this evaluation and the groundwater risk assessment presented in Section 6.3, bromodichloromethane is not retained as a COPC for further evaluation in the FS.

Carbon tetrachloride was identified as a historical COPC in the 100-D/H Work Plan (DOE/RL-2008-46-ADD1) because it was detected above the action level, and most MDLs were greater than the action level. The action level for carbon tetrachloride defaults to the EQL of 1 μg/L reported in the 100-D/H SAP (DOE/RL-2009-40) when the analytical method cannot achieve the action level. Carbon tetrachloride was detected in 10 of 64 unfiltered groundwater samples (16 percent). Carbon tetrachloride was detected once in 699-94-43 (1.4 J μg/L), 699-95-48 (1.5 J μg/L), 699-95-51 (1.3 J μg/L), 699-96-52B (1.3 J μg/L), 699-97-41 (1.1 J μg/L), 699-97-48B (1.1 J μg/L), 699-98-43 (1.0 J μg/L), and 699-98-49A (1.7 J μg/L) at concentrations slightly greater than the EQL of 1 μg/L. Three subsequent sampling rounds were conducted at 699-97-41, 699-94-43, 699-95-48, 699-95-51, 699-96-52B, 699-97-48B, 699-98-43, and 699-98-49A, each reporting nondetected concentrations with MDLs less than or equal to the EQL. Carbon tetrachloride was detected twice in 699-95-45 (1.4 μg/L and 0.16 μg/L) at concentrations greater than and less than the EQL of 1 μg/L. Two subsequent sampling rounds were reported with nondetected concentrations less than or equal to the EQL. All MDLs are less than or equal to the EQL listed in the 100-D/H SAP (DOE/RL-2009-40). The infrequent presence of carbon tetrachloride above the EQL does not suggest it is associated with a specific location or a trend. Based on the results of this evaluation and the groundwater risk assessment presented in Section 6.3, carbon tetrachloride is retained as a COPC for further evaluation in the FS.

Chloroform was identified as a historical COPC in the 100-D/H Work Plan (DOE/RL-2008-46-ADD1) because it was detected above the action level, and most MDLs were greater than the action level. The action level for chloroform defaults to the EQL of 5 μg/L reported in the 100-D/H SAP (DOE/RL-2009-40) when the analytical method cannot achieve the action level. Chloroform was detected in 33 of 64 (52 percent) unfiltered groundwater samples. All results (detected concentrations and MDLs) are less than the EQL of 5 μg/L. Based on the results of this evaluation and the groundwater risk assessment presented in Section 6.3, chloroform is not retained as a COPC for further evaluation in the FS.

Tetrachloroethane was detected above the action level of 0.081 μg/L in samples collected for the RI and is included in the nature and extent evaluation. Tetrachloroethene was not identified as a historical COPC in the 100-D/H Work Plan (DOE/RL-2008-46-ADD1). Tetrachloroethene was detected in 11 of 64 unfiltered groundwater samples (17 percent). The action level for tetrachloroethene is 0.081 μg/L based on the MTCA ("Groundwater Cleanup Standards" [WAC 173-340-720]) groundwater cleanup level; however, it defaults to the EQL of 5 μg/L reported in the 100-D/H SAP (DOE/RL-2009-40) when the analytical method cannot achieve the action level. All results (detected concentrations and MDLs) are less than the EQL listed in the 100-D/H SAP (DOE/RL-2009-40). Based on the results of this evaluation and the results of the groundwater risk assessment presented in Section 6.3, tetrachloroethene is not retained as a COPC for further evaluation in the FS.

Trichloroethene was identified as a historical COPC in the 100-D/H Work Plan (DOE/RL-2008-46-ADD1) because it was detected above the action level, and most MDLs were greater than the action level. The action level for trichloroethene of 0.49 μg/L is based on the MTCA ("Groundwater Cleanup Standards" [WAC 173-340-720]) groundwater cleanup level; however, it defaults to the EQL of 1 μg/L reported in the 100-D/H SAP (DOE/RL-2009-40) when the analytical method cannot achieve the action level. Trichloroethene was detected in 3 of 64 (4.7 percent) unfiltered groundwater samples. All results (detected concentrations and MDLs) are less than the EQL listed in the 100-D/H SAP (DOE/RL-2009-40). Based on the results of this evaluation and the results of the groundwater risk assessment presented in Section 6.3, trichloroethene is not retained as a COPC for further evaluation in the FS.

Anions. Fluoride was identified as a historical COPC in the 100-D/H Work Plan (DOE/RL-2008-46-ADD1) because it was detected above the action level, and some MDLs were greater than the action level of

960 µg/L. Fluoride was detected in 151 of 198 (76 percent) of unfiltered groundwater samples. All results (detected concentrations and MDLs) were less than the action level of 960 µg/L. Fluoride concentrations in unfiltered samples are also less than the 90th percentile Hanford Site background level of 1,047 µg/L. Based on the results of this evaluation and the results of the groundwater risk assessment presented in Section 6.3, fluoride is not retained as a COPC for further evaluation in the FS.

Nitrate was identified as a historical COPC in the 100-D/H Work Plan (DOE/RL-2008-46-ADD1) because it was detected above the DWS of 45,000 µg/L. Nitrate was detected in all unfiltered groundwater samples (198 samples). All results (detected concentrations and MDLs) were less than the DWS of 45,000 µg/L. Nitrate concentrations in unfiltered samples are greater than the 90th percentile Hanford Site background level of 26,900 µg/L. Based on the results of this evaluation and the results of the groundwater risk assessment presented in Section 6.3, nitrate is not retained as a COPC for further evaluation in the FS.

Nitrite was identified as a historical COPC in the 100-D/H Work Plan (DOE/RL-2008-46-ADD1) because it was detected above the DWS, and some MDLs were greater than the DWS of 3,300 µg/L. Nitrite was detected in 20 of 196 (10 percent) of unfiltered samples. All results (detected concentrations and MDLs) were less than the DWS of 3,300 µg/L. Nitrite concentrations in unfiltered samples are greater than the 90th percentile Hanford Site background level of 94 µg/L. Based on the results of this evaluation and the results of the groundwater risk assessment presented in Section 6.3, nitrite is not retained as a COPC for further evaluation in the FS.

Sulfate was identified as a historical COPC in the 100-D/H Work Plan (DOE/RL-2008-46-ADD1) because it was detected above the secondary DWS of 250,000 µg/L. Sulfate was detected in 100 percent of unfiltered samples (198 samples). All results (detected concentrations and MDLS) were less than the secondary DWS of 250,000 µg/L. Sulfate concentrations in unfiltered samples are greater than the 90th percentile Hanford Site background level of 47,000 µg/L. Based on the results of this evaluation and the results of the groundwater risk assessment presented in Section 6.3, sulfate is not retained as a COPC for further evaluation in the FS.

Metals. Antimony was identified as a historical COPC in the 100-D/H Work Plan (DOE/RL-2008-46-ADD1) because it was detected above the action level, and most MDLs were greater than the action level. MDLs were not adequate for determining the presence of antimony at or below the action level of 5.6 µg/L. Antimony was detected in 3 of 196 unfiltered samples (1.5 percent) and 9 of 197 (4.6 percent) of the filtered groundwater samples. Antimony concentrations in unfiltered samples range between 4.1 and 46.7 µg/L and filtered groundwater samples range between 0.32 and 75 µg/L. Most unfiltered and filtered samples collected for purposes other than the RI were analyzed by Method 6010 (287 of 3939 results). MDLs for these results range between 4 and 720 µg/L and detected concentrations range between 4.1 and 75 µg/L. Seven of 11 antimony results reported by Method 6010 were flagged with a "B" qualifier, a "C" qualifier, and/or a "Y" review qualifier. For samples analyzed using trace methods identified in the 100-D/H SAP (DOE/RL-2009-40), the MDLs ranged between 0.3 and 0.6 µg/L and the single detected concentration was 0.32 µg/L. Some historical detections of antimony are flagged with a "B," a "C," and/or a "Y" qualifier and the presence of antimony does not suggest it is associated with a specific location or a trend. Antimony concentrations reported by trace methods as identified in the 100-D/H SAP (DOE/RL-2009-40) are not above the action level of 5.6 µg/L. With the exception of two sample results flagged with a "Y" qualifier, all detected antimony concentrations are below the 90th percentile Hanford Site background level of 55 µg/L. Based on the results of this evaluation and the results of the groundwater risk assessment presented in Section 6.3, antimony is retained as a COPC and warrants further evaluation in the FS.

Arsenic was identified as a historical COPC in the 100-D/H Work Plan (DOE/RL-2008-46-ADD1) because it was detected above the action level and all MDLs were greater than the action level. The action level for arsenic of 0.018 μg/L is based on the national recommended water quality criteria for "Human Health for consumption of Water plus Organism;" however, the action level defaults to the EQL of 4 μg/L reported in the 100-D/H SAP (DOE/RL-2009-40) when the analytical method cannot achieve the action level. Arsenic was detected in 46 of 51 (90 percent) of the unfiltered and 46 of 51 (90 percent) of the filtered groundwater samples. Arsenic concentrations range between 0.61 and 7.5 μg/L in unfiltered groundwater samples and between 0.48 and 7.2 μg/L in filtered groundwater samples. Minimum, maximum, and 90th percentile concentrations for (filtered) background concentrations of arsenic are 0.5, 8.8, and 7.85 μg/L, respectively. All arsenic concentrations in unfiltered and filtered samples are less than the 90th percentile Hanford Site background concentration. Based on the results of this evaluation and the results of the groundwater risk assessment presented in Section 6.3, arsenic is not retained as a COPC for further evaluation in the FS.

Beryllium was identified as a historical COPC in the 100-D/H Work Plan (DOE/RL-2008-46-ADD1) because it was detected above the DWS, and most MDLs were greater than the DWS of 4 μg/L. Beryllium was detected in 1 of 196 samples (0.51 percent) of the unfiltered samples and in 2 of 197 (1.0 percent) of the filtered groundwater samples. All beryllium results (detected concentrations and MDLs) were less than the DWS of 4 μg/L. In addition, all beryllium detected concentrations are less than the 90th percentile Hanford Site background level of 2.3 μg/L. Based on the results of this evaluation and the results of the groundwater risk assessment presented in Section 6.3, beryllium is not retained as a COPC for further evaluation in the FS.

Cadmium was identified as a historical COPC in the 100-D/H Work Plan (DOE/RL-2008-46-ADD1) because it was detected above the AWQC and most MDLs were greater than the AWQC of 0.25 μg/L. Cadmium was detected in 1 of 196 (0.51 percent) of the unfiltered samples and in 3 of 197 (1.5 percent) of the filtered groundwater samples. Cadmium concentrations in unfiltered and filtered groundwater samples range between 4.3 and 6.1 μg/L. Most unfiltered and filtered samples collected for purposes other than the RI were analyzed by Method 6010 (291 of 393 results) and the MDLs for these results range between 0.45 and 30 μg/L. For samples analyzed using trace methods identified in the 100-D/H SAP (DOE/RL-2009-40), the MDLs range between 0.1 and 0.2 μg/L. Cadmium was detected above the AWQC in four wells (699-94-41, 699-97-43B, 699-99-41 and 699-99-42B). A cadmium concentration above the AWQC was reported in one of ten sampling rounds conducted at 699-94-41; the unfiltered sample (6 μg/L) was flagged with a "Y" review qualifier and cadmium was not detected in the corresponding filtered sample. A cadmium concentration above the AWQC was reported in one of seven sample rounds conducted at 699-97-43B; the filtered sample (6 μg/L) was flagged with a "B" qualifier and a "Y" review qualifier, and cadmium in the corresponding unfiltered sample was not detected. A cadmium concentration above the AWQC was reported in one of eight sampling rounds conducted at 699-99-41; the filtered sample (4.3 μg/L) was flagged with a "Y" review qualifier, and cadmium in the corresponding unfiltered sample was not detected. A cadmium concentration above the AWQC was reported in one of six sample rounds conducted at 699-99-42B; the filtered sample (6.1 μg/L) was flagged with a "Y" review qualifier. Historical cadmium results are either flagged with a "B" qualifier and/or a "Y" review qualifier and the presence of cadmium above the AWQC does not suggest it is associated with a specific location or with a trend. Cadmium concentrations in unfiltered and filtered samples are above the 90th percentile Hanford Site background level of 0.92 μg/L. Based on the results of this evaluation and the results of the groundwater risk assessment presented in Section 6.3, cadmium is retained as a COPC for further evaluation in the FS.

Cobalt was identified as a historical COPC in the 100-D/H Work Plan (DOE/RL-2008-46-ADD1) because it was detected above the action level, and some MDLs were greater than the action level of

2.6 µg/L. Cobalt was detected in 3 of 196 (1.5 percent) of unfiltered samples and 27 of 197 (14 percent) of filtered groundwater samples. Cobalt concentrations in unfiltered samples range between 0.074 and 0.21 µg/L and filtered groundwater samples range between 0.06 and 6.2 µg/L. Most unfiltered and filtered samples collected for purposes other than the RI were analyzed by Method 6010 (291 of 393 results); the MDLs for these results range between 1.7 and 70 µg/L. Cobalt was not detected in unfiltered samples and concentrations ranged between 4 and 6.2 µg/L in filtered samples. Some of the results (5 of 8 samples) were flagged with a "B" qualifier, with concentrations ranging between 4 and 6.2 µg/L. For samples analyzed using trace methods identified in the 100-D/H SAP (DOE/RL-2009-40), the MDLs range between 0.05 and 0.1 µg/L, detected concentrations for unfiltered samples ranged between 0.074 and 0.21 µg/L, and ranged between 0.06 and 0.5 µg/L for filtered samples. Most cobalt results are flagged with a "B" qualifier and the presence of cobalt concentrations above the action level does not suggest it is associated with a specific location or with a trend. Cobalt concentrations associated with samples analyzed using trace methods identified in the 100-D/H SAP (DOE/RL-2009-40) are not above the action level of 2.6 µg/L. Cobalt concentrations in filtered samples are above the 90th percentile Hanford Site background level of 0.92 µg/L. Based on the results of this evaluation and the results of the groundwater risk assessment presented in Section 6.3, cobalt is retained as a COPC for further evaluation in the FS.

Copper was identified as a historical COPC in the 100-D/H Work Plan (DOE/RL-2008-46-ADD1) because it was detected above the AWQC, and some MDLs were greater than the AWQC of 9.0 µg/L. Copper was detected in 40 of 196 (20 percent) unfiltered samples and 31 of 197 (16 percent) of filtered groundwater samples. Copper concentrations in unfiltered samples range between 0.1 and 6.3 µg/L and range between 0.1 and 8.8 µg/L in filtered groundwater samples. Most unfiltered and filtered samples for purposes other than the RI were analyzed by Method 6010 (291 of 393 results). The MDLs for these results range between 2 and 70 µg/L, unfiltered concentrations ranged between 4 and 6.3 µg/L, and filtered concentrations ranged between 4 and 7 µg/L. For samples analyzed using the trace methods identified in the 100-D/H SAP (DOE/RL-2009-40), the MDLs range between 0.1 and 0.2 µg/L, unfiltered concentrations range between 0.1 and 2.8 µg/L, and filtered concentrations range between 0.2 and 8.8 µg/L. MDLs associated with historical concentrations (reported by Method 6010) are not accurate at concentrations at or near the AWQC. MDLs for copper in samples analyzed using trace methods identified in the 100-D/H SAP (DOE/RL-2009-40) are below the AWQC. Copper concentrations in filtered samples are above the 90th percentile Hanford Site background level of 0.81 µg/L. Based on the results of this evaluation and the results of the groundwater risk assessment presented in Section 6.3, copper is retained as a COPC for further evaluation in the FS.

Iron was detected above the secondary DWS of 300 µg/L in samples collected for the RI and is included in the nature and extent evaluation. The action level for iron is based on the secondary DWS. Iron affects aesthetic qualities relating to public acceptance of drinking water. These regulations are not federally enforceable, but are intended as guidelines for states. Iron concentrations in groundwater are compared to the AWQC of 1,000 µg/L. Iron was detected in 131 of 196 (67 percent) of unfiltered and 57 of 197 (29 percent) of filtered groundwater samples. All samples were analyzed by Method 6010. The MDLs ranged between 9 and 330 µg/L, detected iron concentrations ranged between 12 and 2,840 µg/L in unfiltered groundwater samples, and between 11 and 2,050 µg/L in filtered groundwater samples. Iron concentrations greater than 1,000 µg/L were detected at 699-90-45 and 699-97-51A. Iron concentrations above 1,000 µg/L were reported in only two of eight unfiltered samples at 699-97-51A with concentrations above the AWQC ranging from 1,070 to 2840 µg/L. Iron concentrations above 1,000 µg/L were reported in both unfiltered and filtered samples at 699-90-45, with concentrations ranging between 1,780 and 2,490 µg/L. The results of this evaluation indicate that iron concentrations have historically been detected in groundwater. Iron concentrations above the AWQC of 1,000 µg/L at 699-90-45 are a result of the corrosion of the carbon steel well casing that was installed in 1961. The presence of iron at

well 699-97-51A does not appear to be associated with a trend. Except for iron reported at 699-90-45, iron concentrations in filtered water samples are less than the background level of 570 µg/L. Based on the results of this evaluation and the results of the groundwater risk assessment presented in Section 6.3, iron is not retained as a COPC for further evaluation in the FS.

Lead was identified as a historical COPC in the 100-D/H Work Plan (DOE/RL-2008-46-ADD1) because it was detected above the action level, and some MDLs were greater than the action level of 2.1 µg/L. Lead was detected in 6 of 51 (12 percent) of unfiltered and 2 of 51 (3.9 percent) of filtered groundwater samples. All lead results (detected concentrations and MDLs) are less than the action level. Lead concentrations in unfiltered and filtered samples are less than the 90th percentile Hanford Site background level of 0.92 µg/L. Based on the results of this evaluation and the results of the groundwater risk assessment presented in Section 6.3, lead is retained as a COPC for further evaluation in the FS.

Manganese was identified as a historical COPC in the 100-D/H Work Plan (DOE/RL-2008-46-ADD1) because it was detected above the secondary DWS of 50 µg/L. The action level for manganese is based on the secondary DWS. Manganese affects aesthetic qualities relating to public acceptance of drinking water. These regulations are not federally enforceable, but are intended as guidelines for states. Manganese concentrations in groundwater are compared to the MTCA ("Surface Water Cleanup Standards" [WAC 173-340-730]) level of 907 µg/L. Manganese was detected in 52 of 196 (27 percent) of unfiltered and 46 of 197 samples (29 percent) of filtered groundwater. All manganese results (detected concentrations and MDLs) were less than the MTCA ("Surface Water Cleanup Standards" [WAC 173-340-730]) level of 907 µg/L. Manganese concentrations in filtered samples are above the 90th percentile Hanford Site background level of 39 µg/L. Based on the results of this evaluation and the results of the groundwater risk assessment presented in Section 6.3, manganese is not retained as a COPC for further evaluation in the FS.

Mercury was identified as a historical COPC in the 100-D/H Work Plan (DOE/RL-2008-46-ADD1) because it was detected above the AWQC, and some MDLs were greater than the AWQC of 0.012 µg/L. Mercury was not detected in any unfiltered (52 samples) or filtered (52 samples) groundwater samples. The action level for mercury of 0.012 µg/L is based on the AWQC; however, it defaults to the EQL of 0.5 µg/L reported in the 100-D/H SAP (DOE/RL-2009-40) when the analytical method cannot achieve the AWQC. All MDLs are less than the EQL of 0.5 µg/L reported in the 100-D/H SAP (DOE/RL-2009-40). MDLs for mercury in filtered samples are above the 90th percentile Hanford Site background level of 0.003 µg/L. Based on the results of this evaluation and the results of the groundwater risk assessment presented in Section 6.3, mercury is not retained as a COPC for further evaluation in the FS.

Nickel was identified as a historical COPC in the 100-D/H Work Plan (DOE/RL-2008-46-ADD1) because it was detected above the AWQC of 52 µg/L. Nickel was detected in 11 of 196 (5.6 percent) of unfiltered and 12 of 197 (6.1 percent) of filtered groundwater samples. MDLs for samples collected prior to the RI and for the RI were less than the AWQC; 10 samples collected in 2011 were reported with MDLs greater than the AWQC. Except for these 10 results, all nickel results (detected concentrations and MDLs) were less than the AWQC. Nickel concentrations in filtered samples are above the 90th percentile Hanford Site background level of 1.6 µg/L. Based on the results of this evaluation and the results of the groundwater risk assessment presented in Section 6.3, nickel is retained as a COPC for further evaluation in the FS.

Selenium was identified as a historical COPC in the 100-D/H Work Plan (DOE/RL-2008-46-ADD1) because it was detected above the AWQC of 5 µg/L, and some MDLs were greater than the AWQC. Selenium was detected in 50 of 51 (98 percent) of unfiltered and 49 of 51 (96 percent) of filtered groundwater samples. Selenium concentrations in unfiltered samples ranged between 0.91 and 7.1 µg/L and filtered samples ranged between 0.78 and 7.3 µg/L. Selenium concentrations above the AWQC were reported at 699-95-51. Selenium was detected once at 699-95-51 in an unfiltered (7.1 µg/L) and filtered

(7.3 μg/L) sample from the same sample round. Selenium was analyzed in two subsequent sample rounds at 699-95-51 at concentrations less than the AWQC. All selenium concentrations are less than the 90th percentile Hanford Site background level of 11 μg/L. Based on the results of this evaluation and the results of the groundwater risk assessment presented in Section 6.3, selenium is not retained as a COPC for further evaluation in the FS.

Silver was identified as a historical COPC in the 100-D/H Work Plan (DOE/RL-2008-46-ADD1) because it was detected above the action level, and some MDLs were greater than the action level of 2.6 μg/L. The action level for silver is 2.6 μg/L based on the WAC 173-201A surface water standard; however, it defaults to the EQL of 10 μg/L identified in the 100-D/H SAP (DOE/RL-2009-40) when the analytical method cannot achieve the action level. Silver was detected in 5 of 196 (2.6 percent) of unfiltered and 5 of 197 (2.5 percent) of filtered groundwater samples. Silver concentrations in unfiltered samples ranged between 0.28 and 12 μg/L and filtered groundwater samples ranged between 5.6 and 13 μg/L. Most of the unfiltered and filtered samples collected for purposes other than the RI were analyzed by Method 6010 (291 of 393 results). The MDLs for this method ranged between 1.7 and 110 μg/L and detected concentrations ranged between 4 and 13 μg/L. Eight of nine detected silver results reported by Method 6010 were flagged with a "B" qualifier and one received both a "B" and a "C" qualifier. Silver concentrations for the seven unfiltered and filtered samples flagged with a "B" ranged between 4 and 13 μg/L. The single silver concentration flagged with a "BC" was 8.2 μg/L (filtered). The silver concentration for the single unflagged sample result was 12 μg/L (unfiltered). For samples analyzed by trace methods identified in the 100-D/H SAP (DOE/RL-2009-40), the MDLs range between 0.1 and 0.2 μg/L and the only detected result was 0.28 μg/L for an unfiltered sample. Most historical detections of silver are flagged with a combination of "B" and "C" qualifiers. Silver concentrations are not associated with a specific location or a trend. Silver results (detected concentrations and MDLs) reported by Method 6010 are not accurate at concentrations at or near the action level. Silver concentrations reported by trace methods identified in the 100-D/H SAP (DOE/RL-2009-40) are less than the action level of 2.6 μg/L and are also below the 90th percentile Hanford Site background level of 5.3 μg/L. Based on the results of this evaluation and the results of the groundwater risk assessment presented in Section 6.3, silver is retained as a COPC for further evaluation in the FS.

Thallium was identified as a historical COPC in the 100-D/H Work Plan (DOE/RL-2008-46-ADD1) because it was detected above the action level, and some MDLs were greater than the action level of 0.24 μg/L. The action level for thallium of 0.24 μg/L is based on the AWQC; however, it defaults to the EQL of 2 μg/L identified in the 100-D/H SAP (DOE/RL-2009-40) when the analytical method cannot achieve the action level. Thallium was not detected in any of the unfiltered (51 samples) or filtered samples (51 samples). All MDLs are less than the EQL of 2 μg/L identified in the 100-D/H SAP (DOE/RL-2009-40). Based on the results of this evaluation and the results of the groundwater risk assessment presented in Section 6.3, thallium is not retained as a COPC for further evaluation in the FS.

Uranium was identified as a historical COPC in the 100-D/H Work Plan (DOE/RL-2008-46-ADD1) because it was detected above the DWS of 30 μg/L. Uranium was detected in all 54 unfiltered and all 16 of the filtered groundwater samples. All uranium results (detected concentrations and MDLs) are less than the DWS of 30 μg/L. Uranium concentrations in unfiltered and filtered samples are less than the 90th percentile Hanford Site background level of 9.9 μg/L. Based on the results of this evaluation and the results of the groundwater risk assessment presented in Section 6.3, uranium is not retained as a COPC for further evaluation in the FS.

Vanadium was identified as a historical COPC in the 100-D/H Work Plan (DOE/RL-2008-46-ADD1) because it was detected above the action level of 80 μg/L. Vanadium was detected in 76 of 196 (39 percent) of unfiltered and 69 of 197 (35 percent) of filtered groundwater samples. Vanadium concentrations in

filtered and unfiltered samples range between 4.2 and 38 μg/L, which are below the action level. MDLs for unfiltered and filtered samples range between 4.1 and 140 μg/L. Samples collected for purposes other than the RI were analyzed by Method 6010 and reported MDLs that range between 4.1 and 140 μg/L (two samples collected in 2006 report MDLs equal to 140 μg/L). Samples collected for the RI were also analyzed by Method 6010 as identified in the 100-D/H SAP (DOE/RL-2009-40); however, these MDLs ranged between 12 and 17 μg/L. Vanadium concentrations are less than the action level of 80 μg/L. Vanadium concentrations in filtered samples are above the 90th percentile Hanford Site background level of 12 μg/L. Based on the results of this evaluation and the results of the groundwater risk assessment presented in Section 6.3, vanadium is not identified as a COPC for further evaluation in the FS.

Zinc was identified as a historical COPC in the 100-D/H Work Plan (DOE/RL-2008-46-ADD1) because it was detected above the action level, and some MDLs were greater than the action level of 91 μg/L. Zinc was detected in 31 of 156 (16 percent) unfiltered and 22 of 197 (11 percent) filtered groundwater samples. All samples were analyzed by Method 6010. MDLs for samples analyzed by Method 6010 ranged between 4 and 9.6 μg/L, detected concentrations ranged between 4.2 and 444 μg/L in unfiltered groundwater samples, and ranged between 4 and 364 μg/L for filtered groundwater samples. Some zinc results reported by Method 6010 were flagged with a "B" qualifier or with both a "B" and a "C" qualifier. Zinc concentrations in unfiltered and filtered samples (25 of 53 results) flagged with a "B" ranged between 4 and 29 μg/L. Zinc concentrations in unfiltered and filtered samples flagged with a "BC" qualifier ranged between 5 and 93 μg/L. Zinc concentrations above the action level were reported at 699-87-55,699-97-43, 699-98-46, 699-99-41, 699-99-42B). Zinc concentrations above the action level were reported in one of six sample rounds at 699-87-55; the unfiltered (384 μg/L) and the filtered sample (364 μg/L) from the same round were greater than the action level. The four previous and two subsequent sample rounds at 699-87-55 reported concentrations less than the action level. Zinc concentrations above the action level were reported in one of three sample rounds at 699-97-43; the filtered sample (93 μg/L) was greater than the action level but the corresponding unfiltered sample (70 μg/L) was less than the action level. The one previous and one subsequent sample round at 699-97-43 reported concentrations less than the action level. Zinc concentrations above the action level were reported in one of eight sample rounds at 699-98-46; the unfiltered sample (93 μg/L – flagged with a "C" laboratory qualifier) was greater than the action level but the corresponding filtered sample (less than 4 μg/L) was less than the action level. The five previous and two subsequent sample rounds at 699-98-46 reported concentrations less than the action level. Zinc concentrations above the action level were reported in one of eight sample rounds at 699-99-44; the unfiltered (444 μg/L) and the filtered sample (163 μg/L) from the same round were greater than the action level. The five previous and two subsequent sample rounds at 699-99-44 reported concentrations less than the action level. Zinc concentrations above the action level were reported in one of six sample rounds at 699-99-42B; the unfiltered (306 μg/L) and the filtered sample (295 μg/L) from the same round were greater than the action level. The five previous sample rounds at 699-99-42B reported concentrations less than the action level. Zinc concentrations above the action level are not associated with a specific location or with a trend. Zinc concentrations are also greater than the 90th percentile Hanford Site background level of 22 μg/L. Based on the results of this evaluation and the results of the groundwater risk assessment presented in Section 6.3, zinc is retained as a COPC for further evaluation in the FS.

Summary of the Horn Groundwater Plume Area Groundwater Evaluation. Table 4-10 summarizes the outcome of the analysis. Contaminants that warrant further evaluation in the FS are chromium (total), Cr(VI), and carbon tetrachloride.

Table 4-10. Summary of Horn Groundwater Plume Area Contaminant Evaluation

Category	Constituent
Is retained as a COPC	
Contaminant of potential concern[a] (contaminants that warrant further evaluation in FS)	Chromium, Cr(VI), carbon tetrachloride
Detected at levels above action level and background (uncertain status)	Antimony, cadmium, cobalt, copper, nickel, silver, zinc
Is not retained as a COPC	
Detected above action level, but not associated with a Hanford Site release or isolated instances and/or suspect data	Iron
Detected in groundwater but below action level, EQL, or background concentrations	Arsenic, beryllium, chloroform, fluoride, gross alpha, gross beta, lead, manganese, nitrate, nitrite, selenium, sulfate, strontium-90, technetium-99, tetrachloroethene, trichloroethene, tritium, uranium, vanadium,
Not detected in groundwater	1,2-dichloroethane, 1,1-dichloroethene, benzene, bromodichloromethane, cesium-137, cobalt-60, europium-152. europium-154, europium-155, mercury, thallium, vinyl chloride

Based on evaluation of data collected January 2006 through December 2011.

Additional Analytes Requested by Agencies

As described earlier, additional analytes and sample locations were added through Step 4 of the COPC identification process described in Section 4.4 of the 100-D/H Work Plan (DOE/RL-2008-46-ADD1). Table 4-7 lists the additional groundwater analytes and locations for analysis. In general, the following analytical methods were added:

- Pesticides by Method 8081 at three well locations
- PCBs by Method 1668A at six well locations
- PCBs by Method 8082 at 16 well locations
- PAHs by Method 8310 at 16 well locations
- SVOCs by Method 8270 at 16 well locations
- Radionuclides at all well locations within the monitoring well network
- Cyanide by Method 9012 at one well location

The following describes the analytes from each of the methods including pesticides, dioxin-like PCB congeners, PCB aroclors, PAHs, SVOCs, and radionuclides (discussed in previous section). The following subsection discusses the results of the additional analytes and well locations by analytical method. Summary statistics for each well representing the additional analytes are presented in Tables O-4 through O-19.

Pesticides by Method 8081. Pesticides were analyzed at three wells including 199-D5-15, 199-D8-71, and 199-H4-48. Pesticides were not detected in any of the samples analyzed.

Dioxin-like PCB Congeners by Method 1668A. Dioxin-like PCB congeners were analyzed at the following 6 wells: 199-D5-15, 199-D8-55, 199-D8-71, 199-H4-10, 199-H4-13, and 199-H4-48. There are twelve dioxin-like PCB congeners, which have associated toxicity information allowing the calculation of an action level. The following discusses only these twelve PCB congeners:

- Two PCB congeners were detected at 199-D5-15, both at concentrations less than their action level.
- One PCB congener was detected at 199-H4-10 and 199-H4-48 at concentrations less than its action level.
- Two PCB congeners were detected at 199-D8-55 at concentrations less than their respective action level.
- Five PCB congeners were detected at 199-D8-71, all at concentrations less than their action level.
- Seven PCB congeners were detected at 199-H4-13 with six of seven PCB congeners at concentrations less than their action level.
- One PCB congener (2,3',4,4',5-Pentachlorobiphenyl) was detected in 199-H4-13 at a concentration greater than the action level during the low river stage; however, two subsequent rounds reported the congener as nondetected or at a concentration less than the action level.

PCB Aroclors by Method 8082. PCB aroclors were analyzed at the following 16 wells: 199-D4-84, 199-D5-13, 199-D5-15, 199-D5-17, 199-D5-99, 199-D8-55, 199-D8-71, 199-D8-88, 199-H3-2A, 199-H4-10, 199-H4-11, 199-H4-13, 199-H4-16, 199-H4-3, 199-H4-45, and 199-H4-48. PCB aroclors were not detected in any of the samples analyzed.

Polynuclear Aromatic Hydrocarbons by Method 8310. PAHs were analyzed at the following 16 wells: 199-D4-84, 199-D5-13, 199-D5-15, 199-D5-17, 199-D5-99, 199-D8-5, 199-D8-71, 199-D8-88, 199-H3-2A, 199-H4-10, 199-H4-11, 199-H4-13, 199-H4-16, 199-H4-3, 199-H4-45, and 199-H4-48. PAHs were not detected in any of the samples analyzed.

Semivolatile Organic Compounds by Method 8270. SVOCs were analyzed at the following 16 wells: 199-D4-84, 199-D5-13, 199-D5-15, 199-D5-17, 199-D5-99, 199-D8-55, 199-D8-71, 199-D8-88, 199-H3-2A, 199-H4-10, 199-H4-11, 199-H4-13, 199-H4-16, 199-H4-3, 199-H4-45, and 199-H4-48. Except for one well, SVOCs were either not detected or detected at concentrations less than the action level.

Bis(2-ethylhexyl)phthalate was detected in one of three samples collected from 199-D8-88 at a concentration (2.1 µg/L) above the action level of 1.2 µg/L. Bis(2-ethylhexyl)phthalate is a common laboratory contaminant that is introduced into the sample after it is collected in the field.

Cyanide by Method 9012. Cyanide was analyzed at well 199-H4-3. Cyanide was not detected in any of the three sampling rounds performed at this well.

Conclusions for Additional Analytes. The following summarizes the conclusions that can be made from the evaluation presented above and identifies uncertainties associated with the results of the evaluation. Additional analytical methods were added for up to 16 well locations and include the following types of analyte classes: pesticides, dioxin-like PCB congeners, PCB aroclors, PAHs, SVOCs, cyanide, and radionuclides (summarized previously).

- Pesticides, PCB aroclors, and PAHs were not detected in any of the samples analyzed.

- Dioxin-like PCB congeners were detected at least once in five wells but detected concentrations were less than the action level. A dioxin-like PCB congener was detected once at concentrations above the action level at one well but is not associated with a trend.

- Except for one well, SVOCs were either not detected or detected at concentrations less than the action level. Bis(2-ethylhexyl)phthalate was detected in one of three samples collected from 199-D8-88 at a concentration (2.1 μg/L) above the action level of 1.2 μg/L. Bis(2-ethylhexyl)phthalate is a common laboratory contaminant that is introduced into the sample after it is collected in the field.

- Cyanide was not detected in any of the samples analyzed (three sampling rounds at one well).

4.4.1.3 Groundwater Evaluation for the Confined Aquifer

The nature and extent of contamination in groundwater in the confined aquifer was based on the last 6 years of data (samples collected between January 1, 2006 and December 31, 2011). The nature and extent evaluation described in this section uses data for wells screened in the first water bearing unit of the RUM. A total of 11 wells were included in the evaluation and are listed in Table 4-11. Figure 4-63 provides the locations of the wells considered in the nature and extent evaluation. Groundwater data for 100-D/H were compiled and statistically analyzed and the results are presented in Appendix O (Tables O-20 through Table O-22). These tables present the summary statistics for each analyte where data were available from the completed well and met the 6-year period criteria. The tables also list the background concentrations in Hanford Site groundwater (*Hanford Site Background: Part 3, Groundwater Background* [DOE/RL-96-61]) where available, and the corresponding action level.

Table 4-11. Monitoring Wells Constructed in the Confined Aquifer (First Water Bearing Unit of the Ringold Upper Mud)

100-D Area	100-H Area	Horn Area
199-D5-141 199-D8-54B	199-H2-1 199-H3-2C 199-H3-9 199-H3-10 199-H4-12C 199-H4-15CS	699-97-43C 699-97-45B 699-97-48C

The evaluation of the data for the confined aquifer is similar to that performed for the unconfined aquifer. The results summary is provided in the following subsections for the 100-D groundwater plume area, the 100-H groundwater plume area, and the horn groundwater plume area.

Confined Aquifer Analytes—100-D Area

The following subsections describe the analytical data that were available from completed wells and met the 6 year time frame criteria at 100-D, and includes radionuclides, anions, and metals. These analytes include all data collected during the specified period.

Radionuclides. Gross alpha and gross beta were analyzed in both of the confined aquifer wells. Gross alpha was detected in 29 percent of the groundwater samples and gross beta was detected in 100 percent of the samples. Gross alpha was detected at concentrations ranging between 2.2 and 3.3 pCi/L, which are less than the DWS of 15 pCi/L. Gross beta was detected at concentrations ranging between 9.2 and 13 pCi/L. Gross beta concentrations are generally consistent with the presence of tritium or strontium-90.

Based on the results of this evaluation, gross alpha is not retained for further evaluation in the FS, but gross beta is retained because the presence of strontium-90 was not determined.

Tritium was analyzed because it was detected in the unconfined aquifer above the DWS of 20,000 pCi/L. Tritium was not detected in any of the seven groundwater samples collected from the confined aquifer wells. All results (non-detected estimated concentrations and MDLs) are less than the DWS of 20,000 pCi/L. Based on the results of this evaluation, tritium is not retained for further evaluation in the FS.

Anions. Fluoride was detected in six of seven (86 percent) of groundwater samples with concentrations ranging between 92 µg/L and 265 µg/L. All fluoride results (detected concentrations and MDLs) are less than the action level. Fluoride concentrations in these unfiltered samples are also less than the 90th percentile Hanford Site background level of 1,047 µg/L. Based on the results of this evaluation, fluoride is not retained for further evaluation in the FS.

Nitrate was detected in five of seven (71 percent) of groundwater samples with concentrations ranging between 2,230 µg/L and 2,528 µg/L. All nitrate results (detected concentrations and MDLs) are less than the DWS and action level. Based on the results of this evaluation, nitrate is not retained for further evaluation in the FS for the confined aquifer.

Nitrite was detected in two of seven (29 percent) of groundwater samples with concentrations ranging between 175 µg/L and 210 µg/L. All nitrite results (detected concentrations and MDLs) are less than the DWS. Nitrite concentrations in the two unfiltered samples with detected quantities are greater than the 90th percentile Hanford Site background level of 94 µg/L. Based on the results of this evaluation, nitrite is not retained for further evaluation in the FS.

Sulfate was detected in all seven (100 percent) groundwater samples with concentrations ranging between 54,200 µg/L and 66,000 µg/L. All sulfate results are greater than the 90th percentile Hanford Site background level of 47,014 µg/L, but are less than the secondary DWS. Based on the results of this evaluation, sulfate is not retained for further evaluation in the FS.

Metals. Antimony was not detected in any of seven unfiltered or seven filtered groundwater samples. Most unfiltered and filtered samples collected for purposes other than the RI were analyzed by Method 6010. MDLs for samples analyzed by Method 6010 ranged between 4 and 72 µg/L (six of seven MDLs for unfiltered samples and eight of nine MDLs for filtered samples were greater than the action level). Samples collected for the RI were analyzed using trace methods identified in the 100-D/H SAP (DOE/RL-2009-40), the MDLs ranged between 0.3 and 0.6 µg/L. Antimony MDLs reported by Method 6010 are not accurate at concentrations at or near the action level (5.6 µg/L). Based on these uncertainties, antimony is retained for further evaluation in the FS.

Arsenic was detected in one of one (100 percent) of the unfiltered and one of one (100 percent) of the filtered groundwater samples. Arsenic concentrations measured 4.2 µg/L in both the unfiltered and filtered groundwater samples. Both arsenic concentrations in unfiltered and filtered samples are less than the 90th percentile Hanford Site background concentration. Based on the results of this evaluation, arsenic is not retained for further evaluation in the FS.

Beryllium was not detected in any of the unfiltered or filtered samples. All MDLs were less than or equal to the DWS. Beryllium is not retained for further evaluation in the FS.

Cadmium was not detected in any of the unfiltered or filtered samples. Most unfiltered and filtered samples collected for purposes other than the RI were analyzed by Method 6010. MDLs for samples analyzed by Method 6010 ranged between 0.91 and 4 µg/L (all greater than the AWQC). Cadmium

MDLs reported by Method 6010 are not accurate at concentrations at or near the action level. Based on these uncertainties, cadmium is retained for further evaluation in the FS.

Chromium was detected in three of seven (43 percent) of unfiltered samples and one of seven (14 percent) of filtered groundwater samples. Detected chromium concentrations range between 3.4 and 11 µg/L in unfiltered samples and measured 5.8 µg/L in the filtered sample. Total chromium concentrations are less than the AWQC and DWS. All MDLs were less than the AWQC and DWS. Chromium is not retained for further evaluation in the FS.

Cobalt was not detected in any of the unfiltered or filtered samples. Most unfiltered and filtered samples collected for purposes other than the RI were analyzed by Method 6010. MDLs for samples analyzed by Method 6010 ranged between 4 and 7 µg/L (all 7 MDLs for the unfiltered and filtered samples were greater than the action level). Samples collected for the RI were analyzed using trace methods identified in the 100-D/H SAP (DOE/RL-2009-40), the MDL was 0.1 µg/L. Cobalt MDLs reported by Method 6010 are not accurate at concentrations at or near the action level. Based on these uncertainties, cobalt is retained for further evaluation in the FS.

Copper was detected in one of seven unfiltered samples (14 percent) and zero of seven filtered groundwater samples. The detected concentration of copper measured 4 µg/L, which is above the background value of 0.81 µg/L, but less than the AWQC. Most unfiltered and filtered samples collected for purposes other than the RI were analyzed by Method 6010. MDLs for samples analyzed by Method 6010 ranged between 4 and 7 µg/L (all 7 MDLs for the unfiltered and filtered samples were less than the AWQC). Samples collected for the RI were analyzed using trace methods identified in the 100-D/H SAP (DOE/RL-2009-40), the MDL was 0.2 µg/L. Copper MDLs reported by Method 6010 may not be accurate at concentrations at or near the AWQC. Based on these uncertainties, copper is retained for further evaluation in the FS.

Cr(VI) was detected in five of nine (56 percent) of unfiltered samples and four of six (67 percent) of filtered groundwater samples. Cr(VI) concentrations detected in unfiltered samples range between 2.7 and 14.6 µg/L, and range between 2 and 9 µg/L in the filtered samples. The Cr(VI) concentration in one unfiltered sample measured above the AWQC of 10 µg/L. All MDLs were less than or equal to the DWS and the surface water quality standard for the state of Washington. Based on the uncertainty associated with one sample result above the AWQC, Cr(VI) is retained for further evaluation in the FS.

Iron was detected above the secondary DWS of 300 µg/L in samples collected for the RI and is included in the nature and extent evaluation. Iron affects aesthetic qualities relating to public acceptance of drinking water. Iron was detected in 6 of 7 (86 percent) of unfiltered and 5 of 7 (71 percent) of filtered groundwater samples. All iron concentrations and MDLs in unfiltered and filtered water samples are less than the 90th percentile Hanford Site background concentration of 570 µg/L. Iron is not retained for further evaluation in the FS.

Manganese affects aesthetic qualities relating to public acceptance of drinking water. These regulations are not federally enforceable, but are intended as guidelines for states. Manganese concentrations in groundwater are compared to the MTCA ("Surface Water Cleanup Standards" [WAC 173-340-730]) level of 907 µg/L. Manganese was detected in four of seven (57 percent) of unfiltered and three of seven samples (43 percent) of filtered groundwater samples. Manganese concentrations in unfiltered samples range between 4 and 4.8, and between 4.9 and 6.2 µg/L in filtered groundwater samples. All manganese results (detected concentrations and MDLs) were less than the MTCA ("Surface Water Cleanup Standards" [WAC 173-340-730]) level of 907 µg/L and the 90th percentile Hanford Site background level of 39 µg/L. Manganese is not retained for further evaluation in the FS.

Nickel was not detected in any of the seven unfiltered or seven filtered samples. All MDLs are all above the background groundwater concentration, but less than the AWQC except for one MDL reported at Well 199-D5-141. A total of four samples were collected at well 199-D5-141, three of the four MDLs were less than the AWQC. Nickel is retained for further evaluation in the FS.

Silver was not detected in any of the unfiltered or filtered samples. Most unfiltered and filtered samples collected for purposes other than the RI were analyzed by Method 6010. MDLs for samples analyzed by Method 6010 ranged between 4 and 11 µg/L (all greater than the AWQC). Samples collected for the RI were analyzed using trace methods identified in the 100-D/H SAP (DOE/RL-2009-40), the MDLs ranged between 0.1 and 0.2 µg/L. Silver MDLs reported by Method 6010 are not accurate at concentrations at or near the AWQC. Based on these uncertainties, silver is retained for further evaluation in the FS.

Uranium was detected in one of one unfiltered samples (100 percent). Filtered groundwater samples were not analyzed. The uranium result (3.2 µg/L) is less than the background concentration and DWS. Uranium is not retained for further evaluation in the FS.

Vanadium was detected in seven of seven (100 percent) unfiltered and seven of seven (100 percent) filtered groundwater samples. The concentrations in the unfiltered and filtered samples range between 16 and 41 µg/L and 16 and 46 µg/L, respectively. All vanadium results are greater than the background concentration but less than the action level of 80 µg/L. Vanadium is not retained for further evaluation in the FS.

Zinc was detected in three of six (50 percent) unfiltered and two of six (33 percent) filtered groundwater samples. Zinc concentrations in unfiltered samples range between 6.3 and 353 µg/L and range between 40 and 264 in filtered groundwater samples. Zinc was detected above the AWQC in both the filtered and filtered samples from 199-D5-141 (353 and 264 µg/L, respectively). Zinc is retained for further evaluation in the FS.

Data are also available for barium, calcium, magnesium, potassium sodium, and strontium, which are not retained for further analysis. Calcium, magnesium, potassium, and sodium are essential nutrients, and the barium and strontium concentrations are below the CWA "Human Health Water and Organism" and MTCA ("Groundwater Cleanup Standards" [WAC 173-340-720(4)(b)(iii)(A) and (B)]) levels, respectively.

Confined Aquifer Analytes—100-H Area

The following subsections describe the analyte data that were available from completed wells and met the 6-year period criteria and include radionuclides, VOCs, anions, and metals. These analytes include all data collected during the specified period.

Radionuclides. Gross alpha and gross beta were analyzed in five of the six confined aquifer wells. Gross alpha was detected in 20 percent of the groundwater samples and gross beta was detected in 90 percent of the samples. Gross alpha was detected at concentrations ranging between 3.3 and 7.8 pCi/L, which are less than the DWS of 15 pCi/L. The highest gross alpha concentration of 14 pCi/L was measured at Well 199-H3-10. Gross beta was detected at concentrations ranging between 3.8 and 10 pCi/L. Gross beta concentrations are generally consistent with the presence of tritium or strontium-90. Based on the results of this evaluation, gross alpha and gross beta are not retained for further evaluated in the FS.

Strontium-90 was detected in one of eight (13 percent) of the groundwater samples with a concentration measuring 2.2 pCi/L, with a minimum detectable activity of 1.8 pCi/L. The only detected strontium-90 concentration was measured in Well 199-H3-9. All results (detected concentrations and MDLs) are less than the DWS of 8 pCi/L. Strontium-90 is retained for further evaluation in the FS, however, since the single sample from Well 199-H3-9 had a detectable result.

Technetium-99 was detected in 2 of 15 (13 percent) of the groundwater samples with concentrations ranging between 7.9 and 12 pCi/L. All results (detected concentrations and MDLs) are less than the DWS of 900 pCi/L. The highest technetium-99 concentration was measured in Well 199-H4-12C. Technetium-99 is not retained for further evaluation in the FS.

Tritium was detected in 2 of 15 (13 percent) of the groundwater samples with concentrations ranging between 270 and 400 pCi/L. All results (detected concentrations and MDLs) are less than the DWS of 20,000 pCi/L. The highest tritium concentration was measured in well 199-H2-1. Tritium is not retained for further evaluation in the FS.

Cesium-137, cobalt-60, europium-152, europium-154, and europium-155, were not detected in any of the samples collected from the confined aquifer in the 100-H area and all MDLs were less than their DWS. Based on the results of this evaluation, cesium-137, cobalt-60, europium-152, europium-154, and europium-155 are not retained for further evaluation in the FS.

Data are also available for other radionuclides that are not retained for further analysis. Beryllium-7, cesium-134, and ruthenium-106 have half-lives less than three years. Potassium-40 levels are attributable to background radiation levels, and antimony-125 has no defined action level.

Volatile Organic Compounds. 1,2-Dichloroethane and bromodichloromethane were detected above their action levels in samples collected from the unconfined aquifer for the RI in the H Plume area and are included in the nature and extent evaluation. 1,2-Dichloroethane and bromodichloromethane were not detected in any of the samples collected from the confined aquifer in the 100-H area and the MDLs were less than their EQL. Based on the results of this evaluation, 1,2-dichloroethane and bromodichloromethane are not retained for further evaluation in the FS.

1,1-Dichloroethene, benzene, carbon tetrachloride, tetrachloroethene, trichloroethene, and vinyl chloride were not detected in any of the samples collected from the confined aquifer in the 100-H area and the MDLs were less than or equal to their action level or EQL (as applicable). Based on the results of this evaluation, 1,1-dichloroethene, benzene, carbon tetrachloride, chloroform, tetrachloroethene, trichloroethene, and vinyl chloride are not retained for further evaluation in the FS.

Chloroform was detected in one of two (50 percent) of the groundwater samples with a concentration of 3.7 µg/L. The groundwater action level for chloroform defaults to the EQL of 5 µg/L reported in the 100-D/H SAP (DOE/RL-2009-40) when the analytical method cannot achieve the action level. All chloroform results (detected concentrations and MDLs) are less than the EQL. Chloroform is not retained for further evaluation in the FS.

Data are also available for 1,1,1-trichloroethane, 1,1,2,2-tetrachloroethane, 1,1,2-trichloroethane, 1,1-dichloroethane, 1,2-dichloroethene (total), 1,2-dichloropropane, 1,4-dichlorobenzene, 1,4-dioxane, 1-butanol, 2-butanone, 2-hexanone, 4-methyl-2-pentanone, acetone, bromoform, bromomethane, carbon disulfide, chlorobenzene, chloroethane, chloromethane, cis-1,2-dichloroethylene, cis-1,3-dichloropropene, dibromochloromethane, ethyl cyanide, ethylbenzene, methylene chloride, styrene, tetrahydrofuran, toluene, trans-1,2-dichloroethylene, trans-1,3-dichloropropene, and xylenes (total), but none of these organics were detected or retained for further analysis.

Anions. Fluoride was identified as a historical COPC in the unconfined aquifer for the 100-D/H Work Plan (DOE/RL-2008-46-ADD1). Fluoride was detected in 10 of 15 (67 percent) of groundwater samples with concentrations ranging between 26 µg/L and 230 µg/L. All fluoride results (detected concentrations and MDLs) are less than the action level. Fluoride concentrations in unfiltered samples are also less than the 90th percentile Hanford Site background level of 1,047 µg/L. Fluoride is not retained for further evaluation in the FS.

Nitrate was detected in all 15 (100 percent) of the groundwater samples with concentrations ranging between 770 µg/L and 7,530 µg/L. All nitrate results of detected concentrations are less than the DWS and the 90th percentile Hanford Site background level. Nitrate is not retained in the confined aquifer for further evaluation in the FS.

Nitrite was detected in 1 of 15 (6.7 percent) of groundwater samples at a concentration of 348 µg/L. All nitrite results (detected concentrations and MDLs) are less than the DWS, but the single detected value in the unfiltered sample is greater than the 90th percentile Hanford Site background level of 94 µg/L. Nitrite is not retained for further evaluation in the FS.

Sulfate was detected in all 15 (100 percent) of the groundwater samples with concentrations ranging between 20,200 µg/L and 74,500 µg/L. All sulfate results are less than the secondary DWS, and all but one are less than the 90th percentile Hanford Site background level. Sulfate is not retained for further evaluation in the FS.

Data are also available for other anions that are not retained for further analysis. There are no action levels for bromide or phosphate, and chloride and cyanide concentrations are all below the Clean Water Act - Freshwater Criterion Continuous Concentration (CCC).

Metals. Antimony was detected in 1 of 12 unfiltered samples (8.3 percent) and none of the 10 filtered groundwater samples. Antimony was measured at a concentration of 4.3 µg/L in a single unfiltered sample, which is less than the action level of 5.6 µg/L. Most unfiltered and filtered samples collected for purposes other than the RI were analyzed by Method 6010. MDLs for samples analyzed by Method 6010 ranged between 4 and 72 µg/L, and most MDLs were greater than the action level). Samples collected for the RI were analyzed using trace methods identified in the 100-D/H SAP (DOE/RL-2009-40); the MDL was 0.6 µg/L. Antimony MDLs reported by Method 6010 are not accurate at concentrations at or near the action level. Based on these uncertainties, antimony is retained for further evaluation in the FS.

Arsenic was detected in two of two (100 percent) of the unfiltered groundwater samples. Arsenic concentrations range between 2.1 and 5.1 µg/L in the unfiltered samples. The arsenic concentrations are less than the 90th percentile Hanford Site background concentration. Based on the results of this evaluation, arsenic is not retained for further evaluation in the FS.

Beryllium was detected in 1 of 12 (8.3 percent) of the unfiltered and none of filtered groundwater samples. All beryllium results (detected concentrations and MDLs) were less than or equal to the DWS. Beryllium is not retained for further evaluation in the FS.

Cadmium was not detected in any of the 12 unfiltered or 10 filtered samples. Most unfiltered and filtered samples collected for purposes other than the RI were analyzed by Method 6010. MDLs for samples analyzed by Method 6010 ranged between 0.2 and 4 µg/L, with most greater than the AWQC. Samples collected for the RI were analyzed using trace methods identified in the 100-D/H SAP (DOE/RL-2009-40), the MDL was reported as 0.2 µg/L. Cadmium MDLs reported by Method 6010 are not accurate at concentrations at or near the action level. Based on these uncertainties, cadmium is retained for further evaluation in the FS.

Chromium was identified as a historical COPC in the unconfined aquifer for the 100-D/H Work Plan (DOE/RL-2008-46-ADD1). Total chromium was detected in 11 of 12 (92 percent) of unfiltered samples and 9 of 10 (90 percent) of filtered groundwater samples. Chromium concentrations in unfiltered samples range between 2.3 and 319 µg/L and filtered samples range between 85 and 142 µg/L. Chromium concentrations in filtered samples above the AWQC of 65 µg/L were reported at three wells (199-H3-9, 199-H4-12C, and 199-H4-15CS). Chromium concentrations above the AWQC (and DWS) were reported in the two sampling rounds at 199-H3-9. Chromium concentrations in unfiltered samples were above the

AWQC in all six sampling rounds and above the DWS in three of six rounds at 199-H4-12C. Chromium concentrations in unfiltered samples were above the AWQC in both sampling rounds and above the DWS in one of two rounds at 199-H4-15CS. Chromium is retained for further evaluation in the FS.

Cobalt was detected in 1 of 12 unfiltered samples (8.3 percent) and none of the filtered groundwater samples. Cobalt was measured at a detected concentration of 1.2 µg/L, which is below the action level. Most unfiltered and filtered samples collected for purposes other than the RI were analyzed by Method 6010. MDLs for samples analyzed by Method 6010 ranged between 0.1 and 7 µg/L (10 of 11 MDLs for unfiltered samples and all 10 MDLs for filtered samples were greater than the action level). Samples collected for the RI were analyzed using trace methods identified in the 100-D/H SAP (DOE/RL-2009-40); the MDL was reported at 0.1 µg/L. Cobalt MDLs reported by Method 6010 are not accurate at concentrations at or near the action level. Based on these uncertainties, cobalt is retained for further evaluation in the FS.

Copper was identified as a historical COPC in the unconfined aquifer for the 100-D/H Work Plan (DOE/RL-2008-46-ADD1). Copper was detected in 4 of 12 unfiltered samples (33 percent) and 1 of 10 filtered groundwater samples (10 percent). Copper was reported above the AWQC in two wells (199-H3-10 and 199-H4-12C). Copper was detected in one of six unfiltered samples at 199-H4-12C, but at a concentration below the AWQC. Most unfiltered and filtered samples collected for purposes other than the RI were analyzed by Method 6010. MDLs for samples analyzed by Method 6010 ranged between 4 and 7 µg/L. Samples collected for the RI were analyzed using trace methods identified in the 100-D/H SAP (DOE/RL-2009-40); the MDL was 0.2 µg/L. Copper MDLs reported by Method 6010 may not be accurate at concentrations at or near the AWQC. Based on these uncertainties, copper is retained for further evaluation in the FS.

Cr(VI) was detected in 60 of 62 (97 percent) of unfiltered samples and all 15 of filtered groundwater samples. Cr(VI) concentrations in unfiltered samples range between 30 and 287 µg/L and filtered samples range between 62 and 140 µg/L. Cr(VI) concentrations in unfiltered samples above the AWQC of 10 µg/L were reported at 199-H3-2C, 199-H3-9, 199-H4-12C, and 199-H4-15CS. Cr(VI) concentrations above the AWQC were reported in all 16 sampling rounds at 199-H3-2C. Cr(VI) concentrations above the AWQC were reported in both sampling rounds at 199-H3-9. Cr(VI) concentrations above the AWQC were reported in all 31 sampling rounds at 199-H4-12C. Cr(VI) concentrations above the AWQC were reported in all 11 sampling rounds at 199-H4-15CS. Cr(VI) is retained for further evaluation in the FS.

Iron was detected above the secondary DWS of 300 µg/L in samples collected for the RI and is included in the nature and extent evaluation. Iron affects aesthetic qualities relating to public acceptance of drinking water. Iron was detected in 6 of 10 (60 percent) of unfiltered and 2 of 10 (20 percent) of filtered groundwater samples. All iron concentrations in unfiltered and filtered water samples are less than the 90th percentile Hanford Site background concentration of 570 µg/L. Iron is not retained for further evaluation in the FS.

Lead was detected in one of two (50 percent) unfiltered groundwater samples. Lead concentrations in the unfiltered sample collected from Well 199-D3-10 measured 2.37 µg/L, which is above the AWQC. Based on these uncertainties, lead is retained for further evaluation in the FS.

Manganese affects aesthetic qualities relating to public acceptance of drinking water. These regulations are not federally enforceable, but are intended as guidelines for states. Manganese concentrations in groundwater are compared to the MTCA ("Surface Water Cleanup Standards" [WAC 173-340-730]) level of 907 µg/L. Manganese was detected in 3 of 12 (25 percent) of unfiltered and 2 of 10 samples (20 percent) of filtered groundwater samples. Manganese concentrations in unfiltered samples range between 4.8 and 125 and filtered groundwater samples range between 4.4 and 10 µg/L. All manganese

results (detected concentrations and MDLs) were less than the surface water cleanup level of 907 µg/L. Manganese concentrations in filtered samples are less than the 90th percentile Hanford Site background level of 39 µg/L. Manganese is not retained for further evaluation in the FS.

Mercury was analyzed in the confined aquifer samples. The action level for mercury of 0.012 µg/L is based on the AWQC; however, it defaults to the EQL of 0.5 µg/L reported in the 100-D/H SAP (DOE/RL-2009-40) when the analytical method cannot achieve the AWQC. Mercury was not detected in any of the unfiltered samples and was not analyzed in filtered samples. All mercury MDLs are less than the EQL of 0.5 µg/L. Mercury is not retained for further evaluation in the FS.

Nickel was detected in 3 of 12 (25 percent) unfiltered samples and 0 of 10 filtered groundwater samples. Nickel concentrations in unfiltered samples range between 5.0 and 22 µg/L. Except for two MDLs reported at 199-H4-12C, all results (detected concentrations and MDLs) are less than the AWQC. A total of six samples were collected at Well 199-H4-12C; both detected results were less than the AWQC (5 and 22 µg/L). Based on the results of this evaluation, nickel is not retained for further evaluation in the FS.

Selenium was not detected in two unfiltered groundwater samples. All selenium MDLs are less than the AWQC and the 90th percentile Hanford Site background level of 11 µg/L. Selenium is not retained for further evaluation in the FS.

Silver was not detected in any of the unfiltered samples but was detected in one of 10 (10 percent) filtered samples. Most unfiltered and filtered samples collected for purposes other than the RI were analyzed by Method 6010. MDLs for samples analyzed by Method 6010 ranged between 0.2 and 11 µg/L (all greater than the AWQC). The single silver detection was reported at 199-H4-15CS (5.2 µg/L). This result was flagged with a "C" laboratory qualifier. The "C" qualifier indicates that the analyte was detected in both the sample and the associated QC blank, and the sample concentration was less than or equal to five times the blank concentration. Samples collected for the RI were analyzed using trace methods identified in the 100-D/H SAP (DOE/RL-2009-40), the MDL was reported as 0.2 µg/L. Silver MDLs reported by Method 6010 are not accurate at concentrations at or near the AWQC. Based on these uncertainties, silver is retained for further evaluation in the FS.

Thallium was not detected in either of the unfiltered samples. Samples collected for the RI were analyzed using trace methods identified in the 100-D/H SAP (DOE/RL-2009-40), and the MDL was reported as 0.1 µg/L, which is below the 90th percentile Hanford Site background level and the action level. Thallium MDLs reported by Method 6010 are not accurate at concentrations at or near the action level. Thallium is not retained for further evaluation in the FS.

Uranium was detected in all 15 samples (100 percent) of the unfiltered samples. Filtered groundwater samples were not analyzed. All uranium results were less than the DWS. Uranium is not retained for further evaluation in the FS.

Vanadium was detected in 11 of 12 (92 percent) unfiltered and all 10 (100 percent) filtered groundwater samples. All vanadium results (detected concentrations and MDLs) are less than the action level of 80 µg/L. Vanadium is not retained for further evaluation in the FS.

Zinc was detected in 6 of 12 (50 percent) unfiltered and 5 of 10 (50 percent) filtered groundwater samples. Zinc concentrations in unfiltered samples range between 3.6 and 291 µg/L and range between 4 and 74 in filtered groundwater samples. Zinc was reported in unfiltered samples at concentrations above the AWQC in two wells (199-H3-10, and 199-H4-12C). Zinc concentrations above the AWQC were reported in the single sampling round at 199-H3-10, and in one of six rounds at 199-H4-12C. All zinc results are less than the DWS of 5,000 µg/L. Zinc is retained for further evaluation in the FS.

Data are also available for aluminum, barium, calcium, magnesium, potassium sodium, and strontium, which are not retained for further analysis. Aluminum concentrations are below the MCLs; calcium, magnesium, potassium, and sodium are essential nutrients; and the barium and strontium concentrations are below the CWA "Human Health Water and Organism" and MTCA ("Groundwater Cleanup Standards" [WAC 173-340-720(4)(b)(iii)(A) and (B)]) levels, respectively.

Confined Aquifer Analytes—Horn Area

The following subsections describe the analyte data that were available from completed wells and met the 6-year period criteria in the Horn, and include radionuclides, anions, and metals. These analytes include all data collected during the specified period.

Radionuclides. Gross alpha and gross beta were identified as additional analytes in the 100-D/H SAP (DOE/RL-2009-40). Gross alpha and gross beta were analyzed in all three of the confined aquifer wells. Gross alpha was detected in 6 of 31 (19 percent) of the groundwater samples and gross beta was detected in 30 of 31 (97 percent) of the samples. Gross alpha was detected at concentrations ranging between 2 and 9.1 pCi/L, which are less than the DWS of 15 pCi/L. Gross beta was detected at concentrations ranging between 2.9 and 12 pCi/L. Gross beta concentrations are generally consistent with the presence of tritium and strontium-90. While the concentrations of strontium-90 were not analyzed, gross beta had a maximum value of 5.8 pCi/L and, therefore, the presence of strontium-90 is not expected. Based on the results of this evaluation, gross alpha and gross beta are not retained to be further evaluated in the FS.

Tritium was detected in 7 of 22 (32 percent) of the groundwater samples with concentrations ranging between 390 and 780 pCi/L. All results (detected concentrations and MDLs) are less than the DWS of 20,000 pCi/L. Tritium is not retained for further evaluation in the FS.

Anions. Fluoride was detected in 18 of 21 (66 percent) of groundwater samples with concentrations ranging between 67 μg/L and 371 μg/L. All fluoride results (detected concentrations and MDLs) are less than the action level. Fluoride concentrations in unfiltered samples are also less than the 90th percentile Hanford Site background level of 1,047 μg/L. Based on the results of this evaluation, fluoride is not retained for further evaluation in the FS.

Nitrate was detected in all 21 (100 percent) groundwater samples with concentrations ranging between 436 μg/L and 18,300 μg/L. All nitrate results (detected concentrations and MDLs) were less than the DWS. Nitrate is not retained in the confined aquifer for further evaluation in the FS.

Nitrite was detected in 4 of 21 (19 percent) of groundwater samples with concentrations ranging between 153 μg/L and 267 μg/L. All nitrite results (detected concentrations and MDLs) are less than the DWS. Nitrite concentrations in unfiltered samples are greater than the 90th percentile Hanford Site background level of 94 μg/L. Based on the results of this evaluation, nitrite is not retained for further evaluation in the FS.

Sulfate was detected in all 21 (100 percent) groundwater samples with concentrations ranging between 11,400 μg/L and 56,000 μg/L. All sulfate results are less than the secondary DWS. Based on the results of this evaluation, sulfate is not retained for further evaluation in the FS.

Data are also available for other anions that are not retained for further analysis. There are no action levels for bromide or phosphate, and chloride concentrations are all below the Clean Water Act - Freshwater Criterion Continuous Concentration (CCC).

Metals. Antimony was not detected in any of the 22 unfiltered samples or 22 filtered groundwater samples. All unfiltered and filtered samples were analyzed by Method 6010. MDLs for samples analyzed by Method 6010 ranged between 4 and 60 μg/L (18 of 22 MDLs for unfiltered samples and 17 of 22 MDLs for filtered samples were greater than the action level). Antimony MDLs reported by Method 6010

are not accurate at concentrations at or near the action level. Based on these uncertainties, antimony is retained for further evaluation in the FS.

Beryllium was not detected in any of the unfiltered or filtered samples. All MDLs were less than or equal to the DWS. Beryllium is not retained for further evaluation in the FS.

Cadmium was not detected in any of the unfiltered or filtered samples. All unfiltered and filtered samples were analyzed by Method 6010. The MDLs range between 0.45 and 4 µg/L, with the latter value greater than 90th percentile Hanford Site background level and the action level. Cadmium MDLs reported by Method 6010 are not accurate at concentrations at or near the action level. Based on these uncertainties, cadmium is retained for further evaluation in the FS.

Total chromium was detected in 9 of 22 (41 percent) of unfiltered samples and 7 of 22 (32 percent) of filtered groundwater samples. Chromium concentrations in unfiltered samples range between 3.6 and 58 µg/L and filtered samples range between 3.1 and 55 µg/L. All chromium results (detected concentrations and MDLs) are less than the AWQC, but all detected concentrations exceed the 90th percentile Hanford Site background level. Chromium is retained for further evaluation in the FS.

Cobalt was not detected in any of the 7 unfiltered samples and was detected in 1 of 22 filtered groundwater samples. The single cobalt detection was reported at 699-97-48C (5.9 µg/L); all other cobalt results were reported as nondetects. All unfiltered and filtered samples were analyzed by Method 6010. MDLs for samples analyzed by Method 6010 ranged between 4 and 5.9 µg/L (all MDLs greater than the action level). Cobalt MDLs reported by Method 6010 are not accurate at concentrations at or near the action level. Based on these uncertainties, cobalt is retained for further evaluation in the FS.

Copper was detected in 2 of 22 unfiltered samples (9.1 percent) and 2 of 22 filtered groundwater samples (9.1 percent). All copper results (detected concentrations and MDLs) were less than the AWQC but all detected concentrations were greater than the 90th percentile Hanford Site background level. Based on these uncertainties, copper is retained for further evaluation in the FS.

Cr(VI) was detected in 12 of 34 (35 percent) of unfiltered samples and 5 of 15 (33 percent) of filtered groundwater samples. Cr(VI) concentrations in unfiltered samples range between 3.7 and 53 µg/L and concentrations in filtered samples range between 27 and 42 µg/L. Cr(VI) concentrations in filtered samples above the surface water quality criteria of 10 µg/L were reported at Well 699-97-48C, which is the only well with detected concentrations in filtered samples. Cr(VI) concentrations in unfiltered samples above the AWQC were reported in all five sampling rounds at 699-97-48C. Cr(VI) concentrations above the AWQC were reported in all eight sampling rounds at 699-97-48C. Cr(VI) is retained for further evaluation in the FS.

Iron was detected above the secondary DWS of 300 µg/L in samples collected for the RI and is included in the nature and extent evaluation. Iron affects aesthetic qualities relating to public acceptance of drinking water. Iron was detected in 17 of 22 (77 percent) of unfiltered and 14 of 22 (64 percent) of filtered groundwater samples. Iron concentrations in groundwater are compared to the AWQC of 1,000 µg/L. A single detection of iron above the AWQC was reported in the unfiltered sample from at 699-97-43C; the remaining 13 filtered and unfiltered results were less than the AWQC. Iron is not retained for further evaluation in the FS.

Manganese affects aesthetic qualities relating to public acceptance of drinking water. These regulations are not federally enforceable, but are intended as guidelines for states. Manganese concentrations in groundwater are compared to the MTCA ("Surface Water Cleanup Standards" [WAC 173-340-730]) level of 907 µg/L. Manganese was detected in 20 of 22 (91 percent) of unfiltered and 18 of 22 samples (82 percent) of filtered groundwater samples. Manganese concentrations in unfiltered samples range between 5.5 and 602 and filtered groundwater samples range between 7.1 and 567 µg/L. All manganese

results (detected concentrations and MDLs) were less than the surface water cleanup level of 907 µg/L. Manganese is not retained for further evaluation in the FS.

Nickel was detected in 1 of 22 (4.6 percent) unfiltered samples and 1 of 22 (4.6 percent) filtered groundwater samples. All nickel results (detected concentrations and MDLs) are less than the AWQC. Based on the results of this evaluation, nickel is not retained for further evaluation in the FS.

Silver was not detected in any of the unfiltered or filtered samples. All unfiltered and filtered samples were analyzed by Method 6010. MDLs for samples analyzed by Method 6010 ranged between 4 and 7 µg/L (all greater than the AWQC). Silver MDLs reported by Method 6010 are not accurate at concentrations at or near the AWQC. Based on these uncertainties, silver is retained for further evaluation in the FS.

Vanadium was detected in 18 of 22 (82 percent) unfiltered and 18 of 22 (82 percent) filtered groundwater samples. All vanadium results (detected concentrations and MDLs) are less than the action level of 80 µg/L. Vanadium is not retained for further evaluation in the FS.

Zinc was detected in 10 of 22 (45 percent) unfiltered and 8 of 22 (36 percent) filtered groundwater samples. Zinc concentrations in unfiltered samples range between 4 and 121 µg/L and range between 7.6 and 92 in filtered groundwater samples. Zinc was detected above the AWQC in both the filtered and filtered sample from 699-97-45B (92.4 and 121 µg/L, respectively); all seven subsequent rounds were less than the AWQC. Zinc is retained for further evaluation in the FS.

Data are also available for barium, calcium, magnesium, potassium sodium, and strontium, which are not retained for further analysis. Calcium, magnesium, potassium, and sodium are essential nutrients, and the barium and strontium concentrations are below the CWA "Human Health Water and Organism" and MTCA ("Groundwater Cleanup Standards" [WAC 173-340-720(4)(b)(iii)(A) and (B)]) levels, respectively.

Summary of the Confined Aquifer Groundwater Evaluation. Table 4-12 summarizes the outcome of the analysis. Contaminants that warrant further evaluation in the FS include gross beta, strontium-90, antimony, cadmium, chromium, cobalt, copper, Cr(VI), lead, silver, and zinc. In each of the evaluated areas, antimony, cadmium, cobalt, copper, lead, silver, and zinc are carried into the FS for evaluation based on uncertainty associated with the analytical method (MDLs above the action level). In 100-D, gross beta was carried forward to the FS based on uncertainty with the specific radionuclide representing gross beta and nickel was carried forward based on detections in one well. Also in 100-D, Cr(VI) was carried to the FS based on a single detection above the surface water quality standard in one well location. In 100-H, total chromium and Cr(VI) are found in several locations along the river, without comparable concentrations in the unconfined aquifer above. Strontium-90 is also found in the same locations as Cr(VI) at 100-H. Total chromium and Cr(VI) are found in one well within the Horn area.

4.5 Distribution of Contaminants

Data were collected to better describe the nature and extent of contamination in the various stratigraphic units and enhance the understanding of the plumes. Analytical data from groundwater monitoring wells, remediation wells, and RPO wells were included in the evaluation. The 100-D/H Work Plan (DOE/RL-2008-46-ADD1), Section 4.8, identified the following data needs associated with obtaining a better understanding of the nature and extent of contamination in groundwater.

Data Need No. 5: Define the extent of groundwater contamination above cleanup standards in select areas of the unconfined aquifer. These data are needed to verify that the area southwest of the ISRM barrier was clean of contamination while the two new aquifer tubes at 100-H (C7649 and C7650) were installed to determine the extent of contamination between the 116-H-7 Retention Basins and the river.

Table 4-12. Summary of Confined Aquifer Evaluation

Category	Area Evaluated		
	100-D	100-H	Horn
Analyte Retained for Further Evaluation in the FS			
Detected at levels above action level and background	Gross Beta Antimony* Cadmium* Cobalt* Copper* Hexavalent Chromium Nickel Silver* Zinc*	Strontium-90 Antimony* Cadmium* Chromium Cobalt* Copper* Hexavalent Chromium Lead* Silver* Zinc*	Antimony* Cadmium* Chromium Cobalt* Copper* Hexavalent Chromium Silver* Zinc*

*Analytes are carried forward to the FS for evaluation because the MDL for Method 6010 is above the action level for those analytes.

To address this data gap, four new aquifer tubes and seven new wells were installed at 100-D and two new aquifer tubes and five new wells were installed at 100-H (Table 2-1). Sample locations are presented on Figures 2-1 and 2-2.

Data Need No. 7: Collect physical and hydrogeologic parameters from soil samples to support the determination of contaminant fate and transport beneath the unconfined aquifer. 100-D: Only one well (199-D8-54B) had been installed in the RUM in 100-D, in an area of relatively low concentrations in the unconfined aquifer in the north chromium plume. Cr(VI) has been detected in the well above water quality standards. At 100-H, groundwater contaminant concentrations remain above the aquatic and drinking water standards in wells completed beneath the unconfined aquifer. Additional contaminant and hydrogeologic information is needed in the RUM to evaluate potential adverse impacts of groundwater discharging from the RUM through seeps and upwelling in the bottom of the river. Additional soil samples locations were selected to address spatial variability of hydraulic properties of the RUM. To address this data gap, additional wells were installed into the RUM, and soil and groundwater samples were collected at the locations shown on Figure 2-1 and Figure 2-2. Five wells were drilled into the RUM: 199-D5-134 (C7624, Well R4), 199-D5-141 (C7625, Well R5), 199-H2-1 (C7631, Well R3), 199-H3-9 (C7639, Well R1), and 199-H3-10 (C7640, Well R2). These wells were screened in the first water-bearing unit within the RUM. Water quality data include hydraulic conductivity testing (including slug tests and permeameter testing), temperature, pH, oxidation-reduction potential, dissolved oxygen, and turbidity, which were collected during sampling, well development, and slug tests. Analytical data from samples collected during drilling are presented in conjunction with the discussions of the specific contaminants. The vertical distribution of Cr(VI) is presented in Section 4.5.2.

Data Need No. 10: Collect soil and water samples from the following units: (1) vadose zone, (2) deep vadose zone, (3) rewetted zone, (4) unconfined aquifer, (5) above the RUM, and (6) within the RUM. These data are needed to evaluate alternative CSM components regarding whether groundwater contamination is from vadose zone sources (in areas of past handling and storage of high concentration sodium dichromate and

in the periodically wetted zone), within the unconfined aquifer, above the RUM Unit, or within the RUM Unit and diffusing to the unconfined aquifer.

To address this data gap, soil and groundwater samples were collected at the locations shown on Figure 2-1 and Figure 2-2.

Groundwater data at 100-D were collected from seven unconfined aquifer wells, two wells drilled into the RUM, two boreholes converted to wells, three boreholes during drilling, and four aquifer tubes. At 100-H, groundwater data were collected from five unconfined aquifer wells, three wells drilled into the RUM, three boreholes converted to wells, two boreholes during drilling, and two aquifer tubes (Table 2-3). These 17 new monitoring wells and 6 aquifer tubes were installed to address Data Needs 5, 10, and 13 (Chapter 2).

Boreholes C7852, C7857, C7860, C7861, and C7863 were originally intended as temporary borings to collect soil samples and grab groundwater samples. Because of lithologic conditions that prohibited the collection of grab groundwater samples, the borings were converted to temporary Monitoring Wells 199-D8-101, 199-D5-142, 199-H4-84, 199-H4-83, and 199-H3-11, respectively. Five wells were drilled into the RUM: 199-D5-134 (C7624, Well R4), 199-D5-141 (C7625, Well R5), 199-H2-1 (C7631, Well R3), 199-H3-9 (C7639, Well R1), and 199-H3-10 (C7640, Well R2). These wells were screened in the first water-bearing unit within the RUM. Water quality data including conductivity, temperature, pH, oxidation-reduction potential, dissolved oxygen, and turbidity were collected during sampling, well development, and slug tests. Analytical data from samples collected during drilling are presented in conjunction with the discussions of the specific contaminants.

The following sections describe the nature and extent of Cr(VI), nitrate, strontium-90, and other contaminants in groundwater. Contaminants are discussed in order of the size of the footprint of the groundwater plume exceeding the applicable standards. Table 4-13 summarizes information on these plume areas.

Table 4-13. Approximate Areal Extent of 100-D/H Plumes for 2009 and 2011

Contaminant	Cr(VI)		Cr(VI)		Nitrate		Strontium-90	
Standard	10 μg/L in km^2 (mi^2)		48 μg/L in km^2 (mi^2)		45,000 μg/L in km^2 (mi^2)		8 pCi/L in km^2 (mi^2)	
Year	2009	2011	2009	2011	2009	2011	2009	2011
100-D	3.9 (1.5)	2.12 (0.82)	2.7 (1.0)	1.0 (0.38)	1.5 (0.57)	1.12 (0.43)	0.0 (0.0)	0.03 (0.01)
100-H	2.7 (1.0)	0.8 (0.31)	1.0 (0.38)	0.05 (0.02)	0.03 (0.01)	0.12 (0.05)	0.2 (0.09)	0.09 (0.03)
Horn Area	13 (4.8)	4.34 (1.68)	4.7 (1.8)	0.74 (0.29)	0.0 (0.0)	0.21 (0.08)	0.0 (0.0)	0.0 (0.0)
Total	19 (7.4)	7.26 (2.80)	8.3 (3.2)	1.78 (0.69)	1.5 (0.58)	1.44 (0.56)	0.2 (0.09)	0.12 (0.04)

a. "Water Quality Standards for Surface Waters of the State of Washington" (WAC 173-201A). This standard only applies to groundwater that discharges to surface water at the interface.

b. "Model Toxics Control Act—Cleanup" (WAC 173-340(4)(b)(iii)).

c. "National Primary Drinking Water Regulations," "Maximum Contaminant Levels for Inorganic Contaminants" (40 CFR 141.62) (modified, 10,000 μg/L × 1/0.226).

d. "National Primary Drinking Water Regulations," "Maximum Contaminant Levels for Radionuclides" (40 CFR 141.66).

Additional analytes discussed include those evaluated in the FS, based on the spatial and temporal analysis, as well as other analytes with detections.

4.5.1 Hexavalent Chromium

Hexavalent chromium is present in groundwater at 100-D, 100-H, and across the horn between the two reactor areas. Currently, Cr(VI) is primarily found within the unconfined aquifer of 100-D/H, with concentrations above 10 µg/L present in the first water-bearing unit of the RUM at 100-H. The total footprint of the plume areas exceeding the 10 µg/L AWQC, which applies to areas that discharge to surface water, is approximately 19 km^2 (7.4 mi^2). The total footprint of the plume areas exceeding 48 µg/L MTCA (WAC 173-340) DWS, which applies to the whole plume, is approximately 8.3 km^2 (3.2 mi^2). The highest Cr(VI) concentrations in 100-D/H are located west of the 105-DR Reactor, commonly referred to as the southern 100-D plume "hot spot." Concentrations in the horn and 100-H are significantly lower, often below 100 µg/L. The Cr(VI) contamination in the horn area groundwater, and to some degree in 100-H, is believed to have originated in 100-D and migrated east with groundwater flow. The spread of Cr(VI) across the horn likely occurred during 105-D and 105-DR reactor operations, when the groundwater mound associated with the retention basins and cooling water effluent trenches was at its greatest extent.

A contributor to groundwater contamination at 100-D/H was the infiltration test at the 116-DR-1&2 Trench. The large volume of cooling water discharged to the 116-DR-1&2 Trench infiltrated the vadose zone, reaching the water table and expanding the groundwater mound already present from normal reactor operations. This created high hydraulic head conditions, forcing the water to migrate from the Ringold Formation unit E at 100-D into the Hanford formation of the horn. Across the horn, the geology transitions from Ringold Formation unit E to the Hanford formation dominating in the aquifer. Moving eastward toward 100-H, Hanford formation material dominates the unconfined aquifer, with smaller pockets of Ringold Formation unit E present (see Figure 3-8). In general, groundwater flow will follow the path of least resistance. This means that groundwater moving across the horn would tend to remain in the Hanford formation, where there is less resistance to water flow. In addition, the groundwater mound would not migrate easily to the far northern portion of the horn, where the aquifer is also present in the Hanford formation, because of the restricted flow caused by thin aquifer in that area (see Figure 3-7).

4.5.1.1 100-D Area

The unconfined aquifer of the 100-D southern plume has the highest Cr(VI) concentrations in 100-D/H, with a maximum value of 69,700 µg/L (Well 199-D5-122). In contrast, the concentrations across the horn are consistently below 100 µg/L and concentrations in 100-H are below 2,000 µg/L. The highest concentration in the northern plume at 100-D was 2,310 µg/L in well 199-D5-125, reported in June 2010. Monitoring Well 199-D5-122, which is located in the hot spot of the southern plume, has had levels over 60,000 µg/L reported in January, April, and August of 2010. Concentrations in this well have declined in response to the operation of the DX pump-and-treat system, which started in December 2010. As more wells have been installed at 100-D, the confidence in the plume location has improved. The area of highest concentrations in the southern plume (Well 199-D5-122) remains located in a central area near waste site 100-D-100 and 100-D-12. Figure 4-64 shows the waste sites associated with sodium dichromate use and disposal. These waste sites are potential source areas for the associated Cr(VI) groundwater plumes.

In the northern plume, the highest Cr(VI) concentrations are located at Well 199-D5-125. Hexavalent chromium concentrations in northern plume monitoring wells (199-D5-14, 199-D5-15, 199-D5-16, 199-D5-125, and 199-D5-126) have generally increased or remained relatively stable. Because waste site remediation is ongoing in 100-D, sources may remain in the vadose zone that are contributing to the groundwater plume. Potential source areas for the northern plume were investigated in 2009 (*Report on Investigation of Hexavalent Chromium Source in the Northern 100-D Area* [DOE/RL-2010-40]). Results

indicated that the closest waste site that could be a source for the northern plume (100-D-104) is located approximately 250 m (820 ft) from the highest groundwater concentrations, making it an unlikely candidate.

Interim remedial actions have or will address source areas associated with the northern plume. These waste sites include 116-D-1A and 116-D-1B cribs (300 m [985 ft] southeast of Well 199-D5-125) and the site of the former 185-D and 190-D buildings (350 m [1,150 ft] southwest of Well 199-D5-125). The 185-D building occupied what is now the 100-D-30 and 100-D-104 waste sites, where Cr(VI) has been detected in near-surface soil. Ongoing remediation will provide additional information on the location of any potential sources of persistent contamination at the 100-D northern plume.

The lower concentrations of Cr(VI) within the northern plume are located near the 116-DR-1&2 Trench, 116-D-7 Retention Basin, and 116-DR-9 Retention Basin. These waste sites received large volumes of cooling water effluent consisting of low concentrations of Cr(VI) and radioactive compounds. The northern Cr(VI) plume extends to the northeast to encompass the area of these three waste sites indicating that they contributed to the Cr(VI) plume (Figure 4-64).

An alternate theory to the northern plume origin is based on historical leakage from the 182-D Reservoir, associated piping, and its location relative to the two plumes. It has been hypothesized that the northern plume has split off from the southern plume and is part of the same source area. The natural flow of groundwater in the aquifer tends to be eastward from 100-D, with groundwater levels at approximately 118 m (387 ft), toward 100-H, where groundwater levels are approximately 116 m (380 ft). However, leaks from the 182-D Reservoir and associated piping, in addition to the artificially enhanced recharge through the disturbed surface, have caused slight groundwater mounding in some portions of 100-D. The groundwater mound causes a component of the groundwater to flow toward the river, disrupting the natural flow across the horn. Historical leaks of the 182-D Reservoir may have begun after the plume near the railcar unloading station had begun to migrate to the northeast. As the reservoir began to show signs of wear, leaks from the reservoir and associated piping could have split the plume into two portions. Consequently, both the 100-D southern and the 100-D northern plumes could have originated from the same source.

In addition, the groundwater geochemistry in Well 199-D5-33 shows a good correlation to Columbia River water (Section 3.8). This indicates that although leakage from the reservoir has decreased since water levels were drawn down, the reservoir continues to leak and contribute to the aquifer below. The reduction in leakage has allowed the space between the two plumes at 100-D to lessen. However, clean water introduced from reservoir leakage continues to affect contaminant distribution.

Seasonal Change. Changes in the groundwater plume shape and concentration can occur for several reasons. When river stage is high (in the spring), hydraulic head in the river is greater than groundwater hydraulic head. As a result, river water moves inland into bank storage. This causes dilution at the groundwater/surface water interface where clean river water is mixing with Cr(VI) contaminated groundwater, causing Cr(VI) concentrations to be lower in samples collected from monitoring wells and aquifer tubes near the river. As river stage drops, more groundwater discharges to the river from the aquifer(s) causing contaminant levels in nearshore areas to increase. The seasonal variation in Cr(VI) concentrations in groundwater is often greatest adjacent to the river, with less variation and a lag in response time observed farther inland. However, variations in the RUM surface can also affect the distance that seasonal variations may be expected.

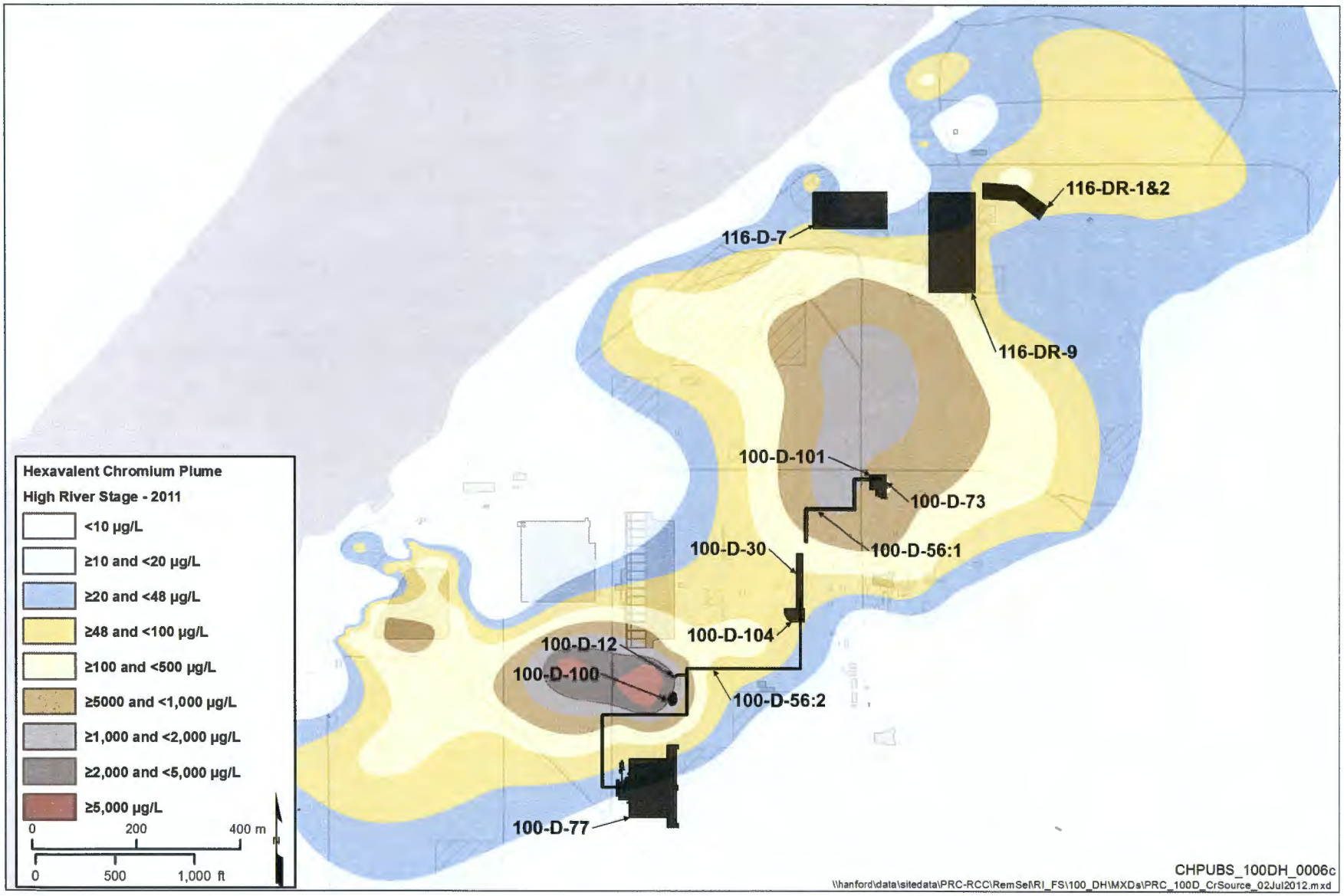

Figure 4-64. 100-D Spring 2011 Cr(VI) Plume and Waste Sites Associated with Sodium Dichromate Use

Seasonal Cr(VI) concentration trends are most often observed in aquifer tube data. In fall 2009, Cr(VI) concentrations in aquifer tube samples were lower than in spring. Only two (33 percent) of the fall aquifer tube samples had higher Cr(VI) concentrations than the spring samples. In 2011, this trend was even more evident as the fall sample results were below detection (Figure 4-65). This is atypical of most seasonal conditions where the spring freshet will effectively suppress contaminant concentrations in groundwater measured in shoreline aquifer tubes. To further demonstrate the variability in concentrations over time as a result of river stage fluctuations, Figures 4-66 and 4-67 show the plume shape and concentration changes across 100-D in low and high river of 2011.

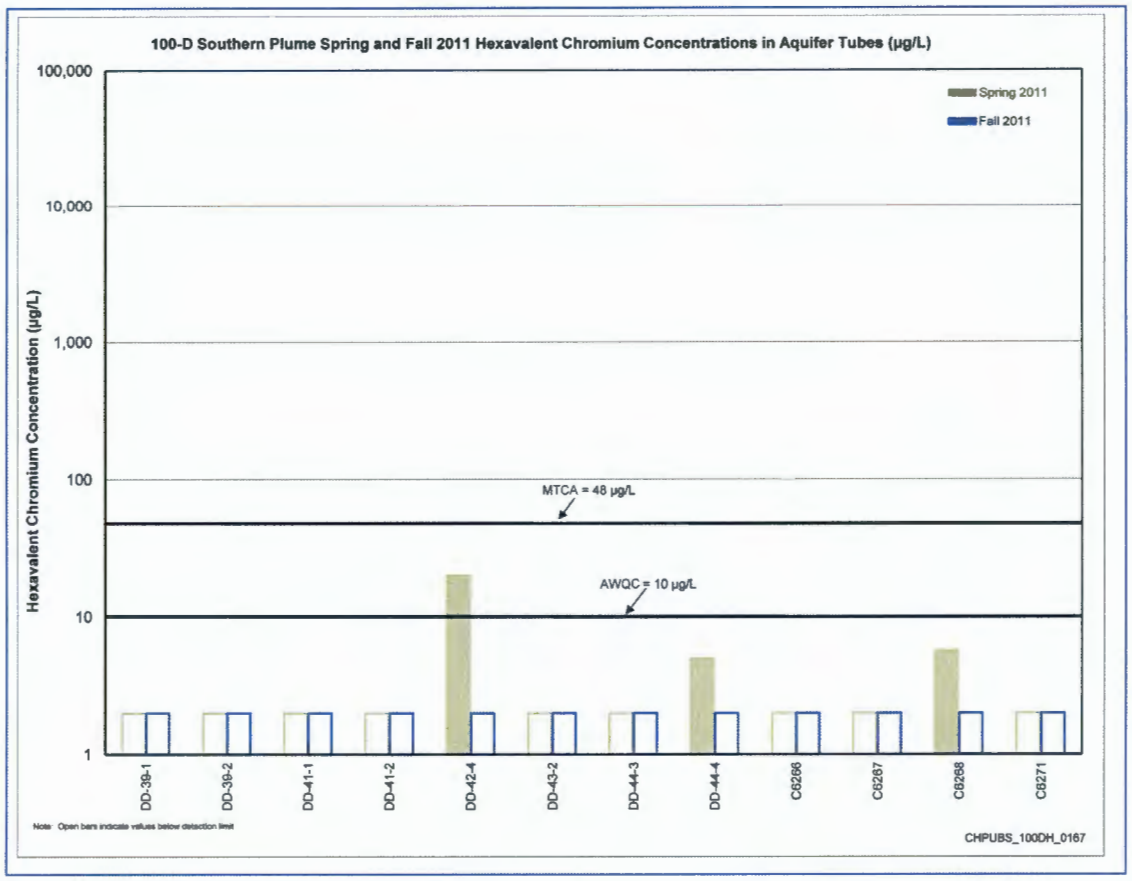

Figure 4-65. 100-D Southern Plume Cr(VI) Concentrations in Aquifer Tubes (Fall and Spring 2011)

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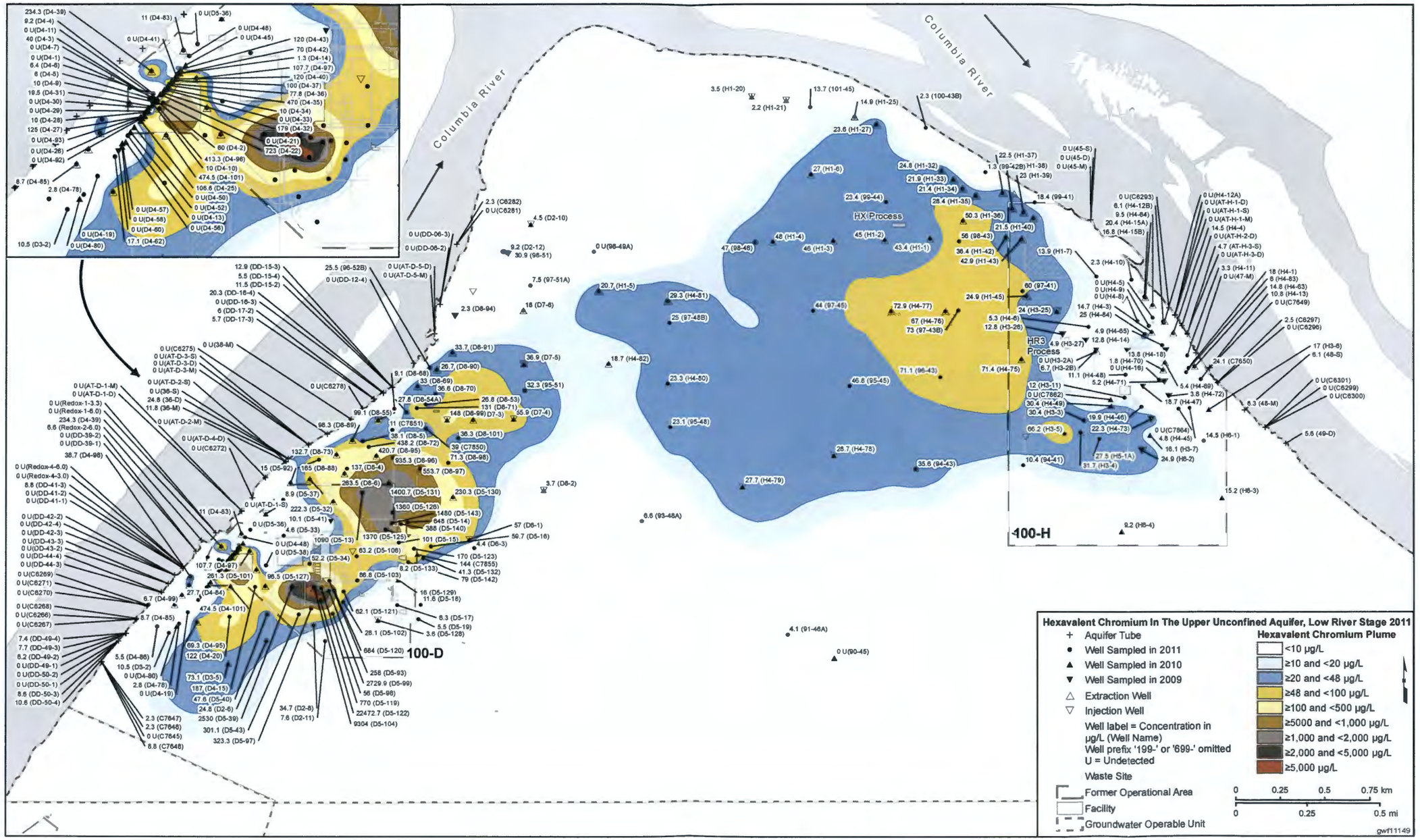

Figure 4-66. Cr(VI) Plume at 100-D/H – Low River Stage (Fall 2011)

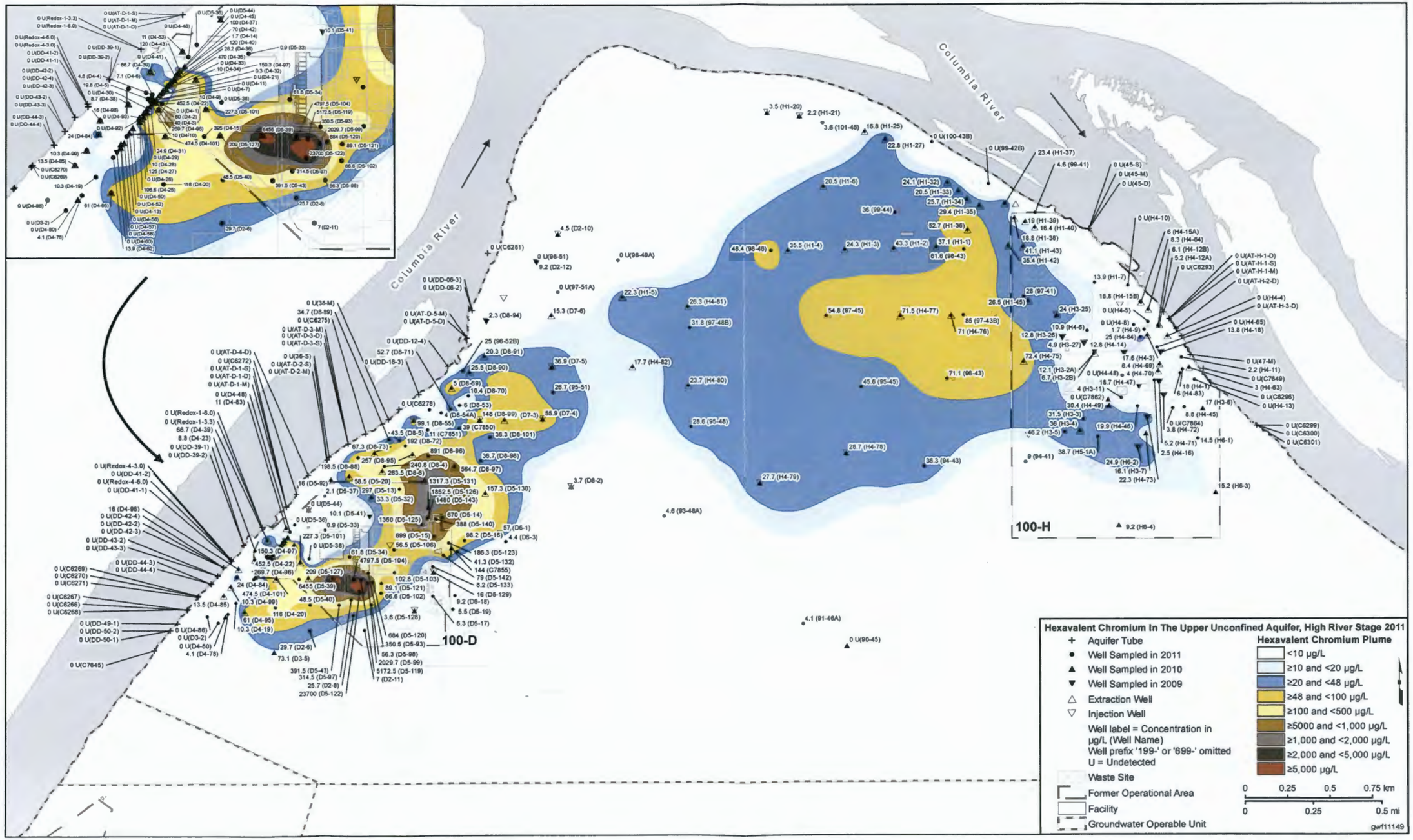

Figure 4-67. Cr(VI) Plume at 100-D/H – High River Stage (Spring 2011)

Cr(VI) concentrations in northern plume monitoring wells (199-D8-69 and 199-D8-70), which are located on the edge of the plume near the river, have decreased over time (Figure 4-68). A strong seasonal variation was exhibited in these near-river wells, with lowest concentrations often below the AWQC of 10 µg/L in Wells 199-D8-69 and 199-D8-70 during summer sampling rounds. As shown on Figure 4-68, the seasonal fluctuation has been greatly reduced as a result of the influence of the DX pump-and-treat system, which started operation in December 2010. Seasonal variations are also present in the southern plume wells. However, the seasonal trends are not as dramatic as in Wells 199-D8-69 and 199-D8-70, and so are not presented, and overall concentrations in these wells are decreasing with time.

The Cr(VI) concentrations in monitoring wells closer to the middle of the northern plume (199-D8-88, 199-D8-55, and 199-D8-73), near the river, were increasing slightly from 2005 through 2010. In 2011, concentrations dropped in response to the DX pump-and-treat system in Wells 199-D8-73 and 199-D8-88. Analytical results from after the startup of the remediation system still show some seasonal fluctuation, but the effects are muted. Well 199-D8-55 has not been monitored since May 2010 and has been converted for use as an injection well for the DX pump-and-treat system.

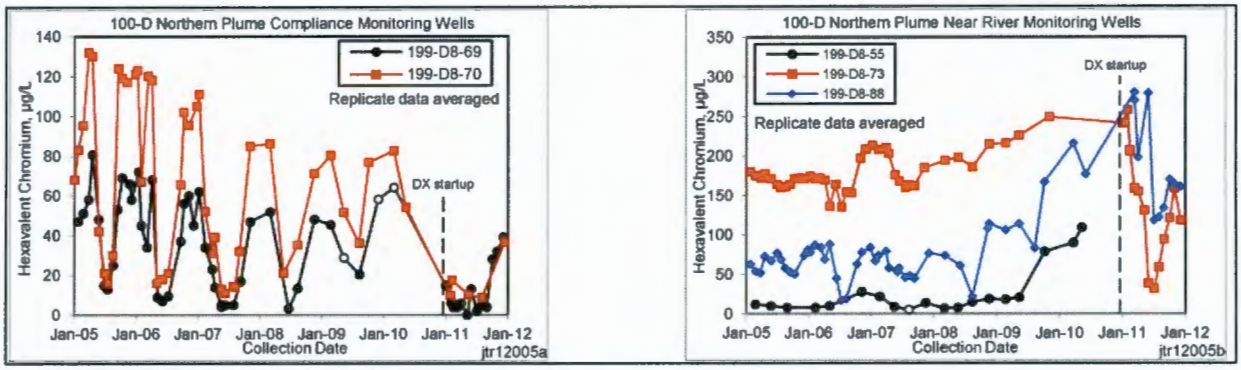

Figure 4-68. Trend Plots for Select Wells in the Northern Cr(VI) Plume at 100-D

Remediation Effects. In addition to the influence of the Columbia River, ongoing remediation activities of the vadose zone and unconfined aquifer affect contaminant distribution in the groundwater. Groundwater remediation has been ongoing since 1997 (HR-3 pump-and-treat system) in the northern plume, and since 2004 (DR-5 pump-and-treat system) in the southern plume. As discussed in Section 1.2.3, the older DR system removed substantial Cr(VI) mass but was under designed, pumping only 50 gallons per minute (gal/min). The DX pump-and-treat system, operating at 500 gal/min, began operating in December 2010 and has already affected Cr(VI) concentrations in the southern plume. The areal extent of the Cr(VI) plume in the unconfined aquifer has essentially remained the same to date. However, the DX pump-and-treat system has removed a significant amount of mass from the southern and northern plumes, reducing concentrations in many wells. Cr(VI) concentrations in Well 199-D5-122, located in the southern plume hot spot, decreased significantly from a high of 69,700 µg/L in August 2010 to 9,400 µg/L in September 2011 (Figure 4-69).

Groundwater remediation activities in 100-D also included the installation of the ISRM barrier, which was intended to reduce Cr(VI) to a more stable, trivalent form. The ISRM barrier, which intersects the southern end of the Cr(VI) plume, has been largely effective on the south end of the barrier. In response to the ISRM barrier, Cr(VI) concentration trends in groundwater samples from wells both upgradient (199-D5-38, 199-D4-15, 199-D4-20, and 199-D4-22) and downgradient (199-D4-38, 199-D4-23, 199-D4-84, and 199-D4-85) of the barrier are generally decreasing. The barrier was designed to treat levels of Cr(VI) up to 20 µg/L using ferric iron. As a result of the higher concentrations encountered at the northern end of the barrier, along with higher groundwater velocities (which resulted in reduced

treatment time), some breakthrough was occurring on the northern end of the barrier. As a result, the pump-and-treat system was expanded to capture Cr(VI) that passed through the barrier.

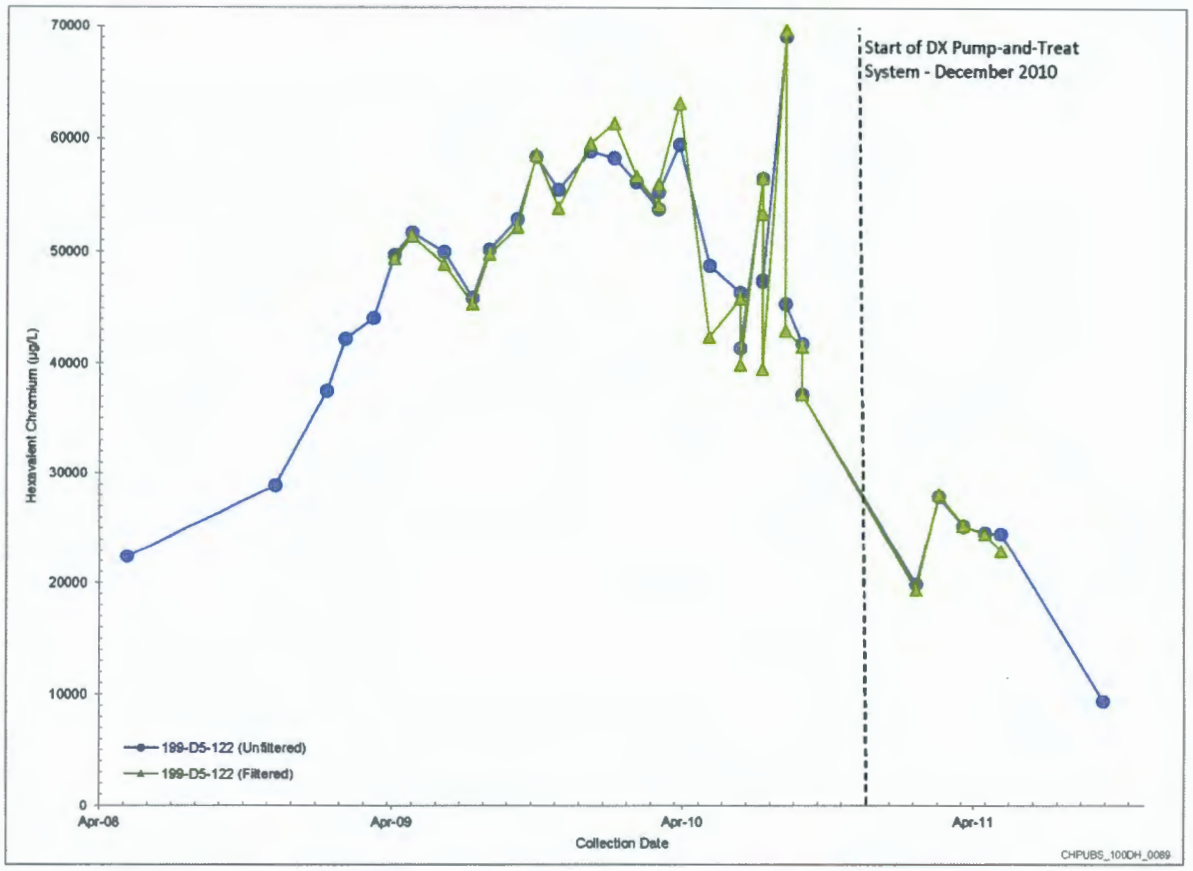

Figure 4-69. Cr(VI) Concentrations Over Time: Well 199-D5-122

Ongoing vadose zone remediation at 100-D-100 has removed soil with Cr(VI) present in concentrations up to 774 mg/kg at 10.7 to 13.7 m (35 to 45 ft) bgs and 334 mg/kg at 15.2 m (50 ft) bgs. Remediation of waste sites not only removes contaminated soil, but also reduces the potential for contamination to affect the groundwater (for example, Well 199-D5-122). Groundwater samples from Wells 199-D5-102, 199-D5-98, and 199-D5-99 have all shown a decrease in Cr(VI) concentrations, but the response may be associated with the pump-and-treat operations or with removal of source material. Other well locations have not shown a response to date (Well 199-D5-104), with concentrations remaining stable.

RI Wells in 100-D. Nine groundwater monitoring wells and five boreholes were installed within or adjacent to the 100-D northern and southern Cr(VI) plumes to provide additional data and delineation of the extent of contamination, as part of the RI (Figure 2-1). Of the nine wells, seven were completed in the unconfined aquifer and two were completed in the first water-bearing unit of the RUM. Groundwater samples were collected at discrete depth intervals from open boreholes during drilling. Cr(VI) was detected in groundwater samples during drilling in the unconfined aquifer from each of the RI wells and boreholes at concentrations above 10 µg/L. Detections were at various depths within the unconfined aquifer. The result from Well 199-D6-3 was 17.60 µg/L at 28.65 m (94 ft) bgs; however, the duplicate sample result was below detection, thus introducing some uncertainty in this result.

Well 199-D3-5 was installed to define the southern extent of the Cr(VI) plume (Figure 2-1) and support data gaps 5 and 13. This well was placed to the south of the 100-D southern hot spot. Contamination was

identified at this location, but the well also provides more information on the plume configuration. Cr(VI) concentrations increased with depth in this well, with a result of 73.10 µg/L from the sample at the RUM surface, bottom of the unconfined aquifer. This well location correlates with a dip in the RUM surface, which slopes to the south/southwest in that area, away from the 100-D southern plume hot spot (Figure 3-4). As discussed in Section 3.7.1 (Horizontal Hydraulic Gradients and Flow Velocities), groundwater flow direction can shift toward the west (azimuth of 270 degrees) depending on river stage. These two factors indicate that groundwater flow may follow the surface of the RUM in this area. The presence of Cr(VI) at the RUM surface is consistent with such a flow pattern.

During discrete depth sampling, the maximum Cr(VI) concentrations in groundwater ranged from a low of 11 µg/L, at a depth of 21 m (69 ft) in borehole C7851, to a high value of 6,520 µg/L at a depth of 29.1 m (95.5 ft) in 199-D5-141 (C7625, Well R5). Figure 4-70 shows the maximum Cr(VI) concentrations in groundwater samples collected during drilling, along with the associated sample depth. Vertical distribution data are presented in Section 4.5.3.

Historically, Cr(VI) concentrations in groundwater from the first water-bearing unit in the RUM at 100-D have been consistently below 48 µg/L, with sample results below 10 µg/L. The single exception is one sample result from Well 199-D8-54B, which had a concentration of 14.6 µg/L in May 2008, the maximum concentration for the well. However, the corresponding duplicate sample had a result of 8.3 µg/L. With that exception, all other groundwater samples from Well 199-D8-54B have been below 10 µg/L. Well 199-D5-134 (C7624, Well R4) and Well 199-D5-141 (C7625, Well R5) were installed as part of this RI/FS effort to confirm the results of the existing well (199-D8-54B).

At Well 199-D5-141, samples collected during drilling indicated Cr(VI) concentrations up to 2,590 µg/L in the unconfined aquifer. Cr(VI) was not detected in the first water-bearing unit in the RUM at this location. As there is no contamination present in the underlying aquifer within the RUM at a location where high concentrations are present in the unconfined aquifer, data indicate there is no hydraulic connection between the two water-bearing units at the well. Well 199-D5-141 was screened across the water-bearing unit in the RUM and samples from the completed well represent that aquifer. Monitoring Well 199-D5-144 (C8668, Well R5 redrill) was located closer to the 100-D-12 waste site (and upgradient from Well 199-D5-141); however, Cr(VI) values in the unconfined aquifer were lower than those detected in groundwater from Well 199-D5-141.

In Well 199-D5-134 (C7624, Well R4), samples collected during drilling indicated Cr(VI) concentrations up to 1,670 µg/L in the unconfined aquifer. Also from samples collected during drilling, both total chromium and Cr(VI) were detected in the first water-bearing unit of the RUM. Total chromium concentrations were at 12.6 µg/L and Cr(VI) concentrations were at 12.2 µg/L. An evaluation of the boring logs and daily reports indicates that the sample was collected following difficulty in sample collection, which resulted in a delay of more than 2 days. Based on an evaluation of the sample results at this well, presented with the vertical distribution discussion, it is likely that this sample was contaminated from groundwater originating from the unconfined aquifer, and is not representative of the first water-bearing unit in the RUM. A post-installation sample from January 2012 had a concentration below detection, confirming that contamination is not present in the confined aquifer.

During drilling activities, groundwater samples were collected from the second water-bearing unit in the RUM, presumed to be the Ringold Formation unit B, in Wells 199-D5-141 and 199-D5-134 to evaluate the presence of contaminants in the lower aquifer. Total chromium and Cr(VI) concentrations in these samples were below the laboratory detection limits. Vertical distribution data are presented in Section 4.5.2.

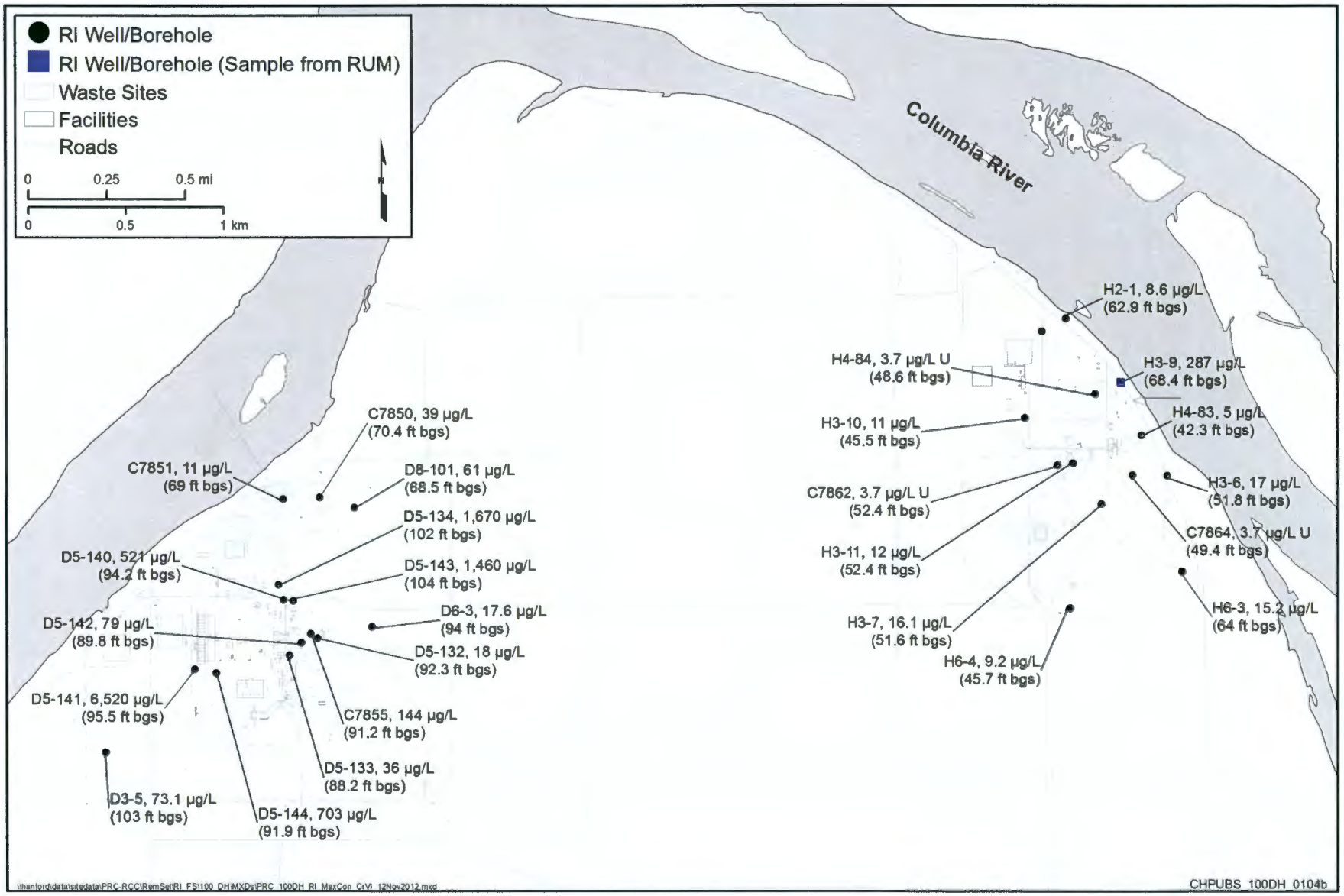

Figure 4-70. Maximum Cr(VI) Concentrations in Borehole Water Samples from RI Wells

4.5.1.2 Horn Area

Groundwater in the horn generally exhibits much lower Cr(VI) concentrations than those found in the 100-D plumes, although concentrations still exceed the AWQC of 10 µg/L and the MTCA ("Groundwater Cleanup Standards" [WAC 173-340-720]) level of 48 µg/L at many locations. Figure 4-71 shows the Cr(VI) concentrations for wells completed in the RUM, including those in the horn. The horn has very few waste sites; therefore, the Cr(VI) detected in the groundwater likely migrated across the horn from 100-D rather than having originated from local releases. Other waste sites in the area, such as 600-105, are located to the south with Cr(VI) levels below 5 µg/L, and are unlikely contributors to the main Cr(VI) plume.

Concentration trends in the horn groundwater monitoring wells are generally decreasing or stable. The maximum Cr(VI) detection was 117 µg/L in Well 699-97-43B (October 2007). Cr(VI) concentrations in aquifer tube samples located along the eastern side of the horn (44-D, C5634, C5637, C5641, and C5674) are also generally stable or decreasing.

Three wells in the horn are completed in the RUM: Wells 699-97-43C, 699-97-45B, and 699-97-48C. Cr(VI) concentrations have been consistently below the laboratory detection limits from Wells 699-97-43C and 699-97-45B, the area closest to 100-H. Groundwater samples from Well 699-97-48C (Figure 4-72), located downgradient from the 116-DR-1&2 Trench, have shown an overall increasing Cr(VI) concentration trend.

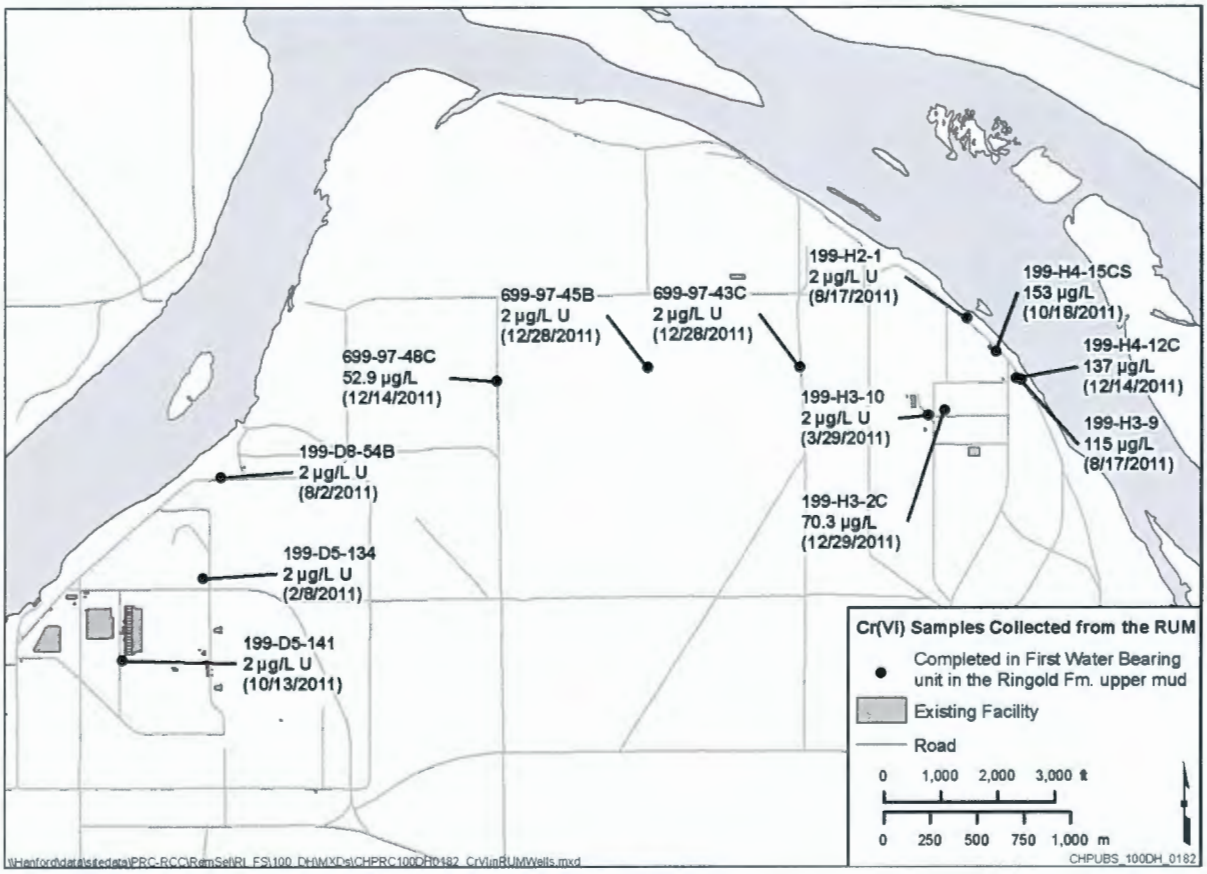

Figure 4-71. Monitoring Wells Completed in the RUM

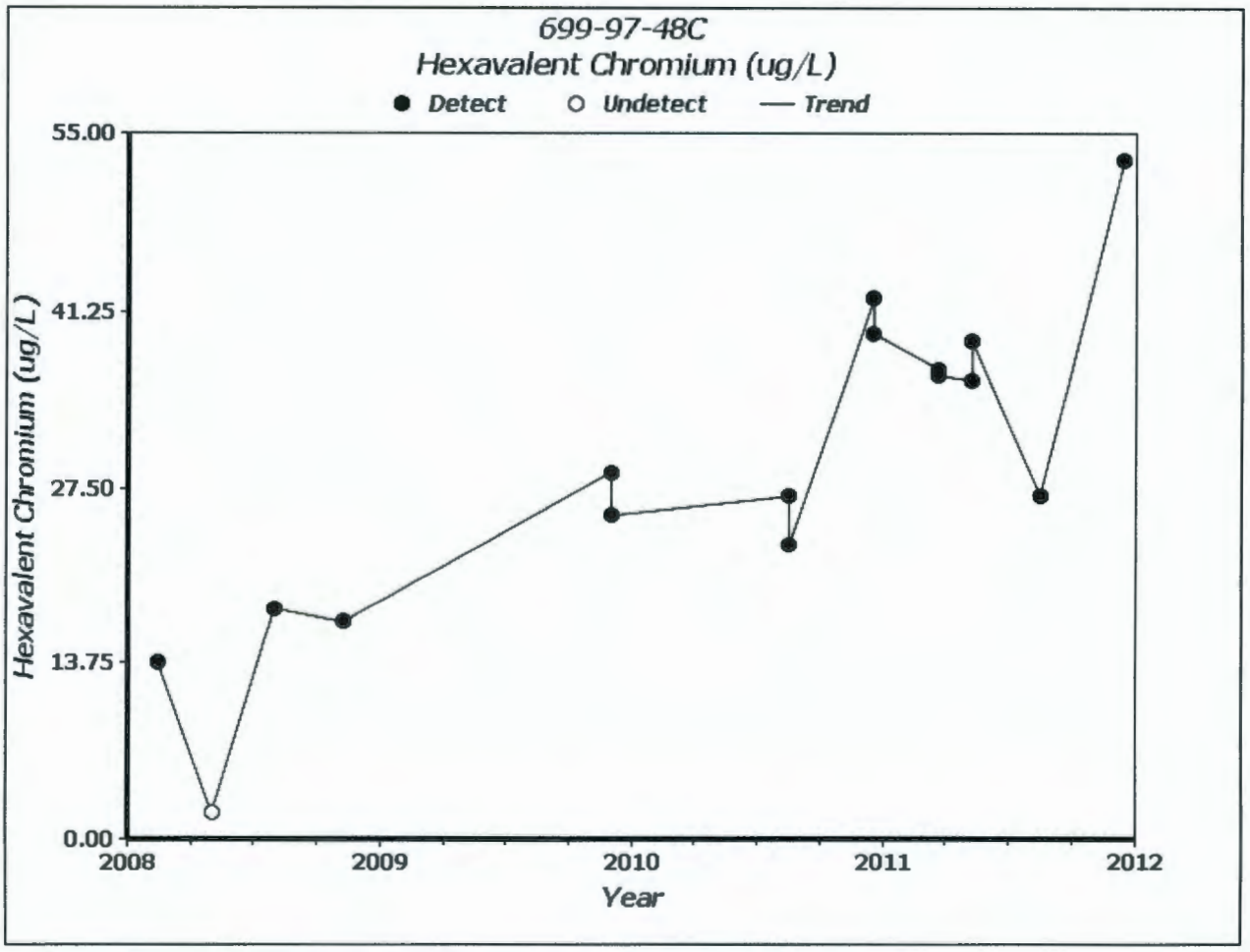

Figure 4-72. Cr(VI) Concentrations in RUM Well 699-97-48C

Seasonal Changes. Aquifer tubes in the horn were not sampled for Cr(VI) in fall 2009 or spring 2010. In groundwater monitoring wells, the Cr(VI) concentrations greater than 10 µg/L were measured in nine (82 percent) of the fall 2009 and spring 2010 groundwater samples (Figure 4-73). The seasonal variation in groundwater monitoring wells across the horn is not consistent in the area.

Overall, the Cr(VI) concentrations across the horn near 100-D are below 48 µg/L, with concentrations generally showing a decreasing trend in the unconfined aquifer. Because the mass of Cr(VI) continues to migrate to the east with the groundwater flow, Cr(VI) concentrations increase on the eastern side of the horn with concentrations as high as 85 µg/L at Well 699-97-43B. The HX pump-and-treat system extraction and injection wells have largely remediated the area at 100-H and have formed a barrier to further migration of Cr(VI) from the horn towards the river.

No monitoring wells were drilled in the horn as part of the RI. However, 25 RPO wells were installed in the horn from 2009 to 2010. The additional sampling, together with previous monitoring, indicates that the Cr(VI) plume underlying the horn has remained relatively stable, and is slowly migrating toward 100-H. There continues to be an area of groundwater with concentrations slightly greater than the MTCA ("Groundwater Cleanup Standards" [WAC 173-340-720]) level of 48 µg/L near 100-H.

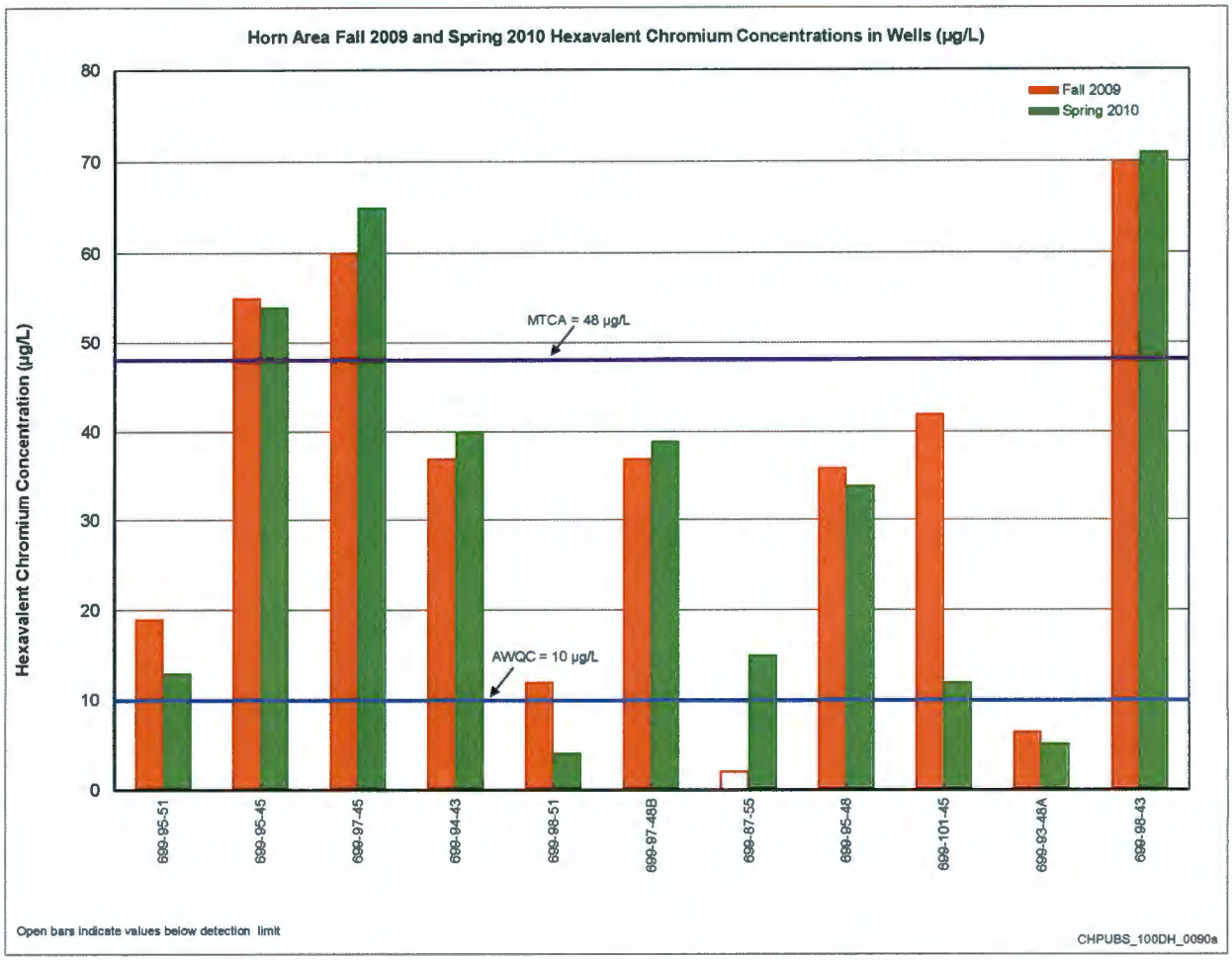

Figure 4-73. Horn Area Cr(VI) Concentrations in Groundwater from Wells – Fall 2009 and Spring 2010

4.5.1.3 100-H Area

The Cr(VI) plume at 100-H (Figure 4-74) is characterized by much lower concentrations than the 100-D plumes (Figures 4-66 and 4-67). Cr(VI) concentrations in the unconfined aquifer are not detected above 100 µg/L in 100-H; however, samples from most areas of 100-H exceed the AWQC of 10 µg/L. The maximum concentration in the unconfined aquifer was 91.8 µg/L in Well 199-H1-43 (March 2010). Monitoring wells completed in the first water bearing unit of the RUM had concentrations as high as 287 µg/L (Well 199-H3-9).

Facilities and waste sites associated with former sodium dichromate handling are potential sources of Cr(VI) contamination. In 100-H, these include the following facilities and waste sites: 116-H-1 Trench, 116-H-2 Trench and associated overflow (100-H-17 waste site), 116-H-4 Pluto Crib, 183-H Solar Evaporation Basins (116-H-6 waste site), 116-H-7 Retention Basin, and 105-H Reactor Fuel Storage Basin. Other potential or suspected sources of Cr(VI) include the 190-H sodium dichromate handling facilities (100-H-46 waste site), 100-H-21 effluent pipelines, and 100-H-5 sludge trench. The relationship of these waste sites to the current Cr(VI) plume is shown on Figure 4-74. The groundwater mound at 100-H was not as extensive as at 100-D, but originated primarily from the 116-H-7 Retention Basin and leaked at rates as high as 38,000 L/min (10,000 gpm).

100-D is an additional source of Cr(VI) in the unconfined aquifer at 100-H, primarily because of the 1967 infiltration test, when cooling water discharges to the 116-H-7 were redirected to the 116-DR-1 & 2 Trench. As discussed in Chapter 1 (Section 1.2.3), the infiltration test resulted in approximately 1.3×10^{10} L (3.4×10^{9} gal) of cooling water effluent being discharged to the 116-DR-1&2 Trench (*Program Review—Ground Disposal of Reactor Effluent* [DUN-3259]). The effluent caused an additional 2.7 to 3 m (9 to 10 ft) of groundwater mounding beyond that caused by ongoing operations, with Cr(VI) concentrations estimated at 350 μg/L. The subsequent groundwater mound migrated eastward, affecting the unconfined aquifer in the horn and 100-H.

Unlike 100-D, contamination has been detected in the first water-bearing unit of the RUM at 100-H. As discussed in Chapter 3, the geologic conditions at 100-H are different from those at 100-D. The key stratigraphy at 100-H, as compared to 100-D, includes the following features:

- Thinner vadose zone and unconfined aquifer
- Ringold Formation unit E is not present
- Variation in RUM material between the unconfined aquifer and the first water-bearing unit

At 100-D, the RUM is typically identified at about 27 to 35 m (90 to 115 ft) bgs. This same unit is identified at 100-H at about 10 to 18 m (35 to 60 ft) bgs, with some variability in both areas. The first portion of the RUM is therefore about 17 m (55 ft) thinner at 100-H, making it more vulnerable to hydraulic head changes in the overlying aquifer. More importantly, the first portion of the RUM appears to have a greater sand and gravel component at 100-H than the same zone at 100-D, contributing to the vulnerability of the first water bearing unit in the RUM.

These stratigraphic features, combined with high head conditions created from the groundwater mound at the 116-H-7 Retention Basin during reactor operations, are the likely reasons for contamination in the RUM at 100-H and not in 100-D. The pressure of the mound at 100-H could have pushed the contaminated groundwater into the first of the lower aquifers. Section 4.5.3 discusses the vertical distribution of Cr(VI), and includes the analytical sample results from the deeper units.

Seasonal Change. Figure 4-66 and 4-67 show the seasonal variation of Cr(VI) plume configuration at 100-D/H. As in 100-D, seasonal variability at 100-H is observed mainly adjacent to the river, with minimal seasonal variation inland. Figure 4-75 shows Cr(VI) fall 2009 and spring 2010 concentrations for 100-H wells. 100-H aquifer tubes were not sampled for Cr(VI) in fall 2009 or spring 2010. Cr(VI) concentrations greater than the AWQC of 10 μg/L were measured in 10 (48 percent) of the fall 2009 groundwater samples and 13 (62 percent) of the spring 2010 samples. Concentrations greater than the MTCA ("Groundwater Cleanup Standards" [WAC 173-340-720]) level of 48 μg/L were measured in three (14 percent) of the fall 2009 groundwater samples and four (19 percent) of the spring 2010 groundwater samples. In contrast to other 100-D/H areas described previously, fall 2009 concentrations were lower than spring 2010 concentrations. The fall 2009 concentrations were greater than spring 2010 concentrations in only eight (38 percent) of the monitoring wells. This pattern is more typical of the seasonal variations along the Columbia River, and was identified at 100-H during the 2009 to 2010 time frame, resulting in part from the different lithology of the aquifer matrix.

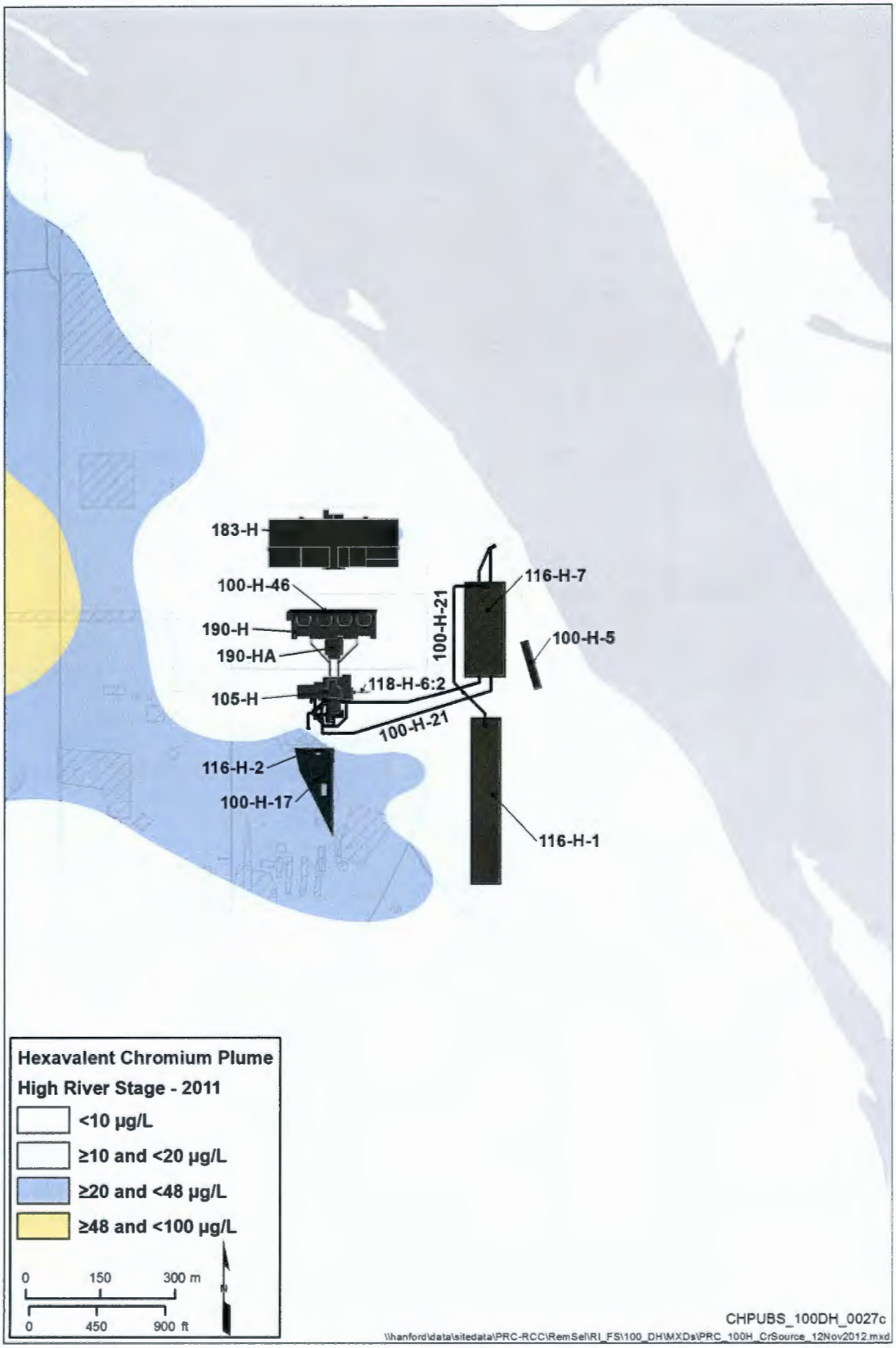

Figure 4-74. 100-H: Cr(VI) Spring 2010 Plume and Waste Sites Associated with Sodium Dichromate Use

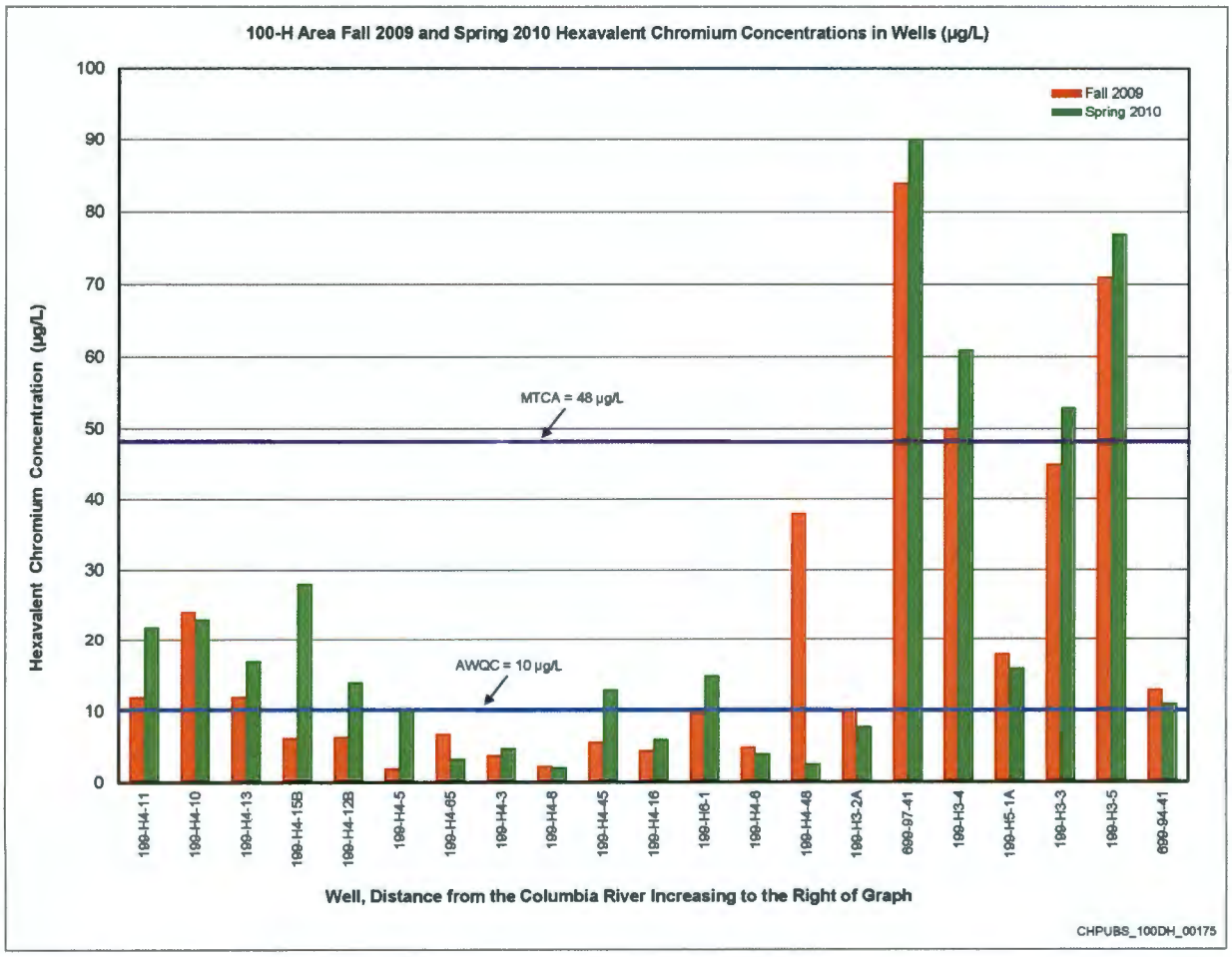

Figure 4-75. 100-H Area Cr(VI) Concentrations in Groundwater from Wells – Fall 2009 and Spring 2010

Remediation Effects. Unlike the Cr(VI) plumes in 100-D, the 100-H plume has diminished substantially in recent years, as evidenced by decreasing Cr(VI) concentrations in the unconfined aquifer extraction wells (199-H4-3, 199-H4-63, and 199-H4-15A). This result is directly caused by the operation of the interim remedy HR-3 pump-and-treat system. On October 1, 2011, the HX pump-and-treat system was started with a capacity of 3,000 L/min (800 gal/min). This new system is expected to facilitate remediation because of the expanded capture area. The remediation system is aided by the hydrogeology of the area, which includes an aquifer matrix (that is, Hanford formation) with more favorable hydraulic properties than at 100-D, and a relatively thin unconfined aquifer. The Cr(VI) concentrations in the majority of unconfined aquifer wells at 100-H show a decreasing trend. Figure 4-76 shows representative trends from wells in the northern, northwestern, and southeastern portions of 100-H (Wells 199-H4-15B, 199-H4-8, and 199-H4-45, respectively).

Monitoring wells in the southwestern portion of 100-H do not follow the same trend, exhibiting either stable or increasing Cr(VI) concentrations. This suggests continued migration of Cr(VI) with groundwater flow in the unconfined aquifer from the horn to the southeast.

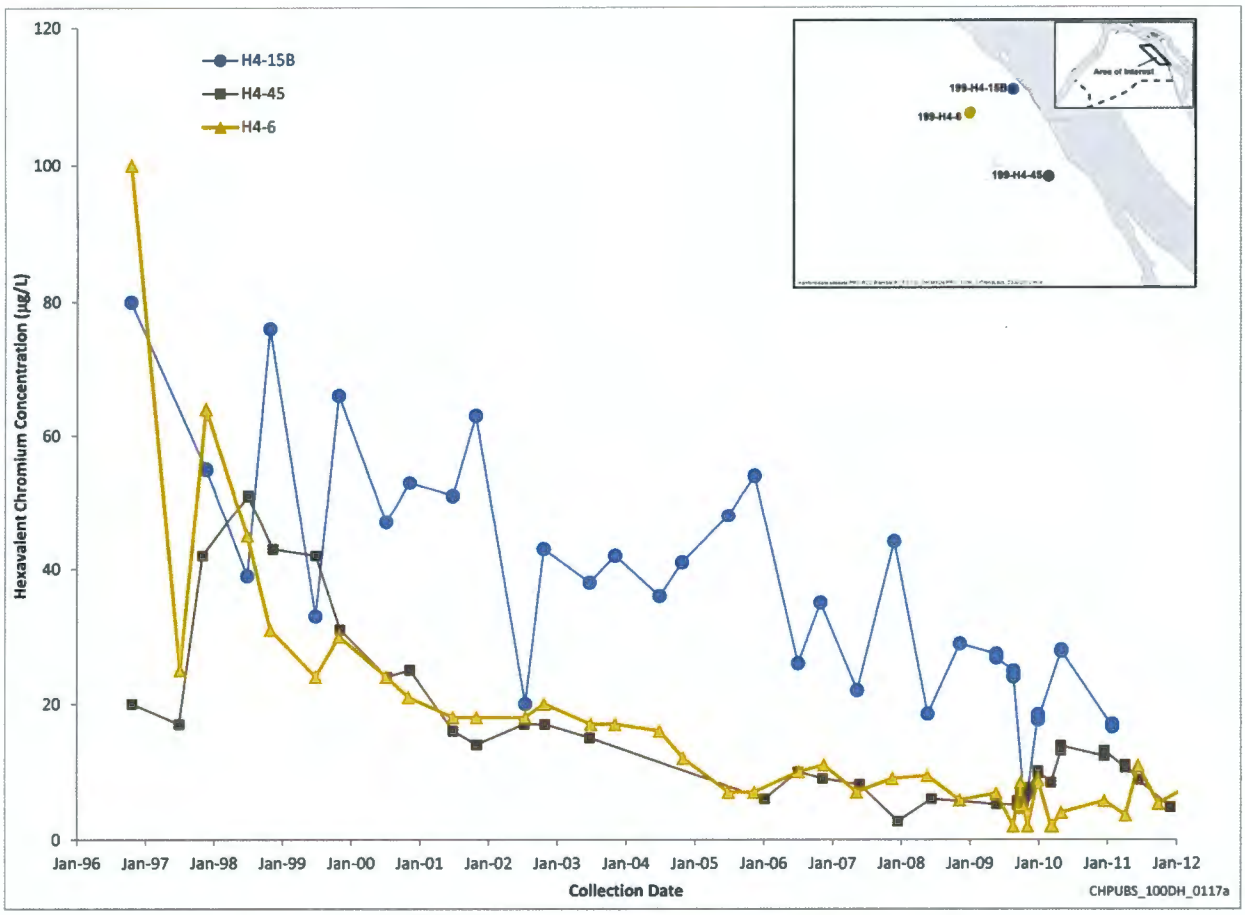

Figure 4-76. Cr(VI) Trends in Groundwater for Select Wells in 100-H

Cr(VI) in the RUM. The RUM is generally considered an aquitard, which means it cannot transmit significant amounts of water, but may store water. Aquitards, which are often considered leaky confining beds, can transmit small amounts of water between stratigraphic units. Along the river corridor, the RUM surface forms the base of the unconfined aquifer, and contains several water-bearing sandy gravel lenses. At 100-H, the material present between the RUM surface and the first water-bearing unit is typically described as sandy silt or gravelly silt. This can be compared to the same zone at 100-D, which is generally described as silt and silty clay material with lenses of "thick" or "hardened" clay. In addition, based on the existing RUM wells, the material in this zone is nearly 17 m (55 ft) thicker at 100-D than in most areas of 100-H, with some variability between wells. In areas such as at 100-H, where the first water-bearing unit in the RUM is closer to the RUM surface, and the RUM material itself is more permeable and may allow water to be transmitted between stratigraphic units, a hydraulic connection between the unconfined aquifer and the water-bearing units within the RUM may be present.

Under certain conditions, this connection may transmit contamination in addition to water. One such condition may be the discharge of large volumes of cooling water that occurred near the former 105-H Reactor, which caused a mound of groundwater to form 4.9 to 10.1 m (16 to 33 ft) above the natural water table. At 100-H, the groundwater mound was primarily associated with the 116-H-7 and 116-H-1 Retention Basins, which is located just south of the highest levels of Cr(VI) contamination within the first water bearing unit of the RUM. The high head conditions associated with the groundwater mound during operations may have driven groundwater contaminated with Cr(VI), consistent with cooling water, into the first water-bearing unit in the RUM via a hydraulic connection between this unit and the unconfined

aquifer (Section 3.7.3 and Section 3.7.4). This is supported by the observed higher concentrations of Cr(VI) in the RUM as compared to the overlying unconfined aquifer (Figure 4-77).

Another condition that supports a connection between the retention basins and the presence of Cr(VI) in the RUM along the Columbia River shoreline is the undulating surface of the RUM itself.

In 2009, an aquifer test and rebound study was conducted at 100-H (*Aquifer Testing and Rebound Study in Support of the 100-H Deep Chromium Investigation* [SGW-47776]). Testing was conducted at three 100-H wells/piezometers that were completed in the first water-bearing unit of the RUM (Wells 199-H3-2C, 199-H4-12C, and 199-H4-15CS). All three of these wells showed a marked response to the aquifer tests conducted from August 20 to November 11, 2009. During that time, step-drawdown pump tests and subsequent constant rate pump tests were conducted at all three of the wells. Details on the aquifer characteristics of these wells and their interactions are discussed in Section 3.7.2, Section 3.7.3, and Section 3.7.4. The wells closest to the Columbia River had Cr(VI) concentrations above 20 μg/L prior to the pumping and rebound tests, with the Cr(VI) concentrations at inland Well 199-H3-2C slightly below 20 μg/L. After pumping was suspended, Cr(VI) concentrations in these three wells continued to show a gradually increasing trend, reaching 148 μg/L in Well 199-H4-12C by March 2011 and 153 μg/L in piezometer 199-H4-15CS.

Three additional wells were completed in the RUM at 100-H as part of this RI: Wells 199-H2-1, 199-H3-9, and 199-H3-10. Figure 4-77 shows Cr(VI) concentration trends from the wells currently completed within the RUM, with results from the last sampling event from 2011 posted. As discussed in Section 4.4.2, Cr(VI) was detected at 8.6 μg/L in the first water-bearing unit of the RUM in Well 199-H2-1 during drilling. Post-installation samples from Well 199-H2-1 were below detection. Cr(VI) concentrations at Well 199-H3-9 were as high as 287 μg/L during drilling. Cr(VI) was not detected in the first water-bearing unit of the RUM at Well 199-H3-10.

A comparison of groundwater levels in 199-H4-15 nested piezometers suggests that an upward hydraulic gradient exists between the unconfined aquifer and lower water-bearing units below the RUM (Chapter 3, Section 3.7.2.2). This relationship would tend to retard migration of contaminants from the upper units except under unique circumstances, such as the presence of high hydraulic head. However, the steepness of the upward vertical gradient has decreased in recent years. This decrease in vertical gradient may help explain concentration trends in both the semiconfined and the confined water-bearing units in the Ringold Formation. Piezometer 199-H4-15CS is completed in the first water-bearing unit in the RUM, and 199-H4-15CQ is completed in the second water-bearing unit, presumed to be Ringold unit B. Piezometer 199-H4-15CR is completed in the RLM, and 199-H4-15CP is completed in the basalt unit. The Cr(VI) concentrations for the 100-H deep piezometer cluster 199-H4-15CP, 199-H4-15CQ, and 199-H4-15CR are all below the AWQC value of 10 μg/L.

RI Wells. Eight RI groundwater monitoring wells and five boreholes were installed in or adjacent to the 100-H plume to provide additional data and to further delineate the extent of contamination. Five of the monitoring wells were drilled into the top of the RUM to an average depth of 19.2 m (62.9 ft) bgs and screened in the unconfined aquifer. Groundwater samples were collected from boreholes during drilling at depth discrete intervals. The Cr(VI) concentrations in groundwater samples from the unconfined aquifer (boreholes and wells) ranged from nondetect in a few locations to 25 μg/L at a depth of 14.3 m (47.0 ft) in Well 199-H4-84. Sample results for the remaining locations were between 3.7 and 16.1 μg/L.

Three monitoring wells were drilled into the first water-bearing unit in the RUM to an average depth of 64.8 m (212.6 ft). Within the first water-bearing unit of the RUM, the Cr(VI) concentrations ranged from below detection to a maximum value of 287 μg/L at a depth of 20.8 m (68.4 ft) in Well 199-H3-9 (C7639, Well R1). Figure 4-77 shows Cr(VI) maximum concentrations and associated depths in all new RI wells.

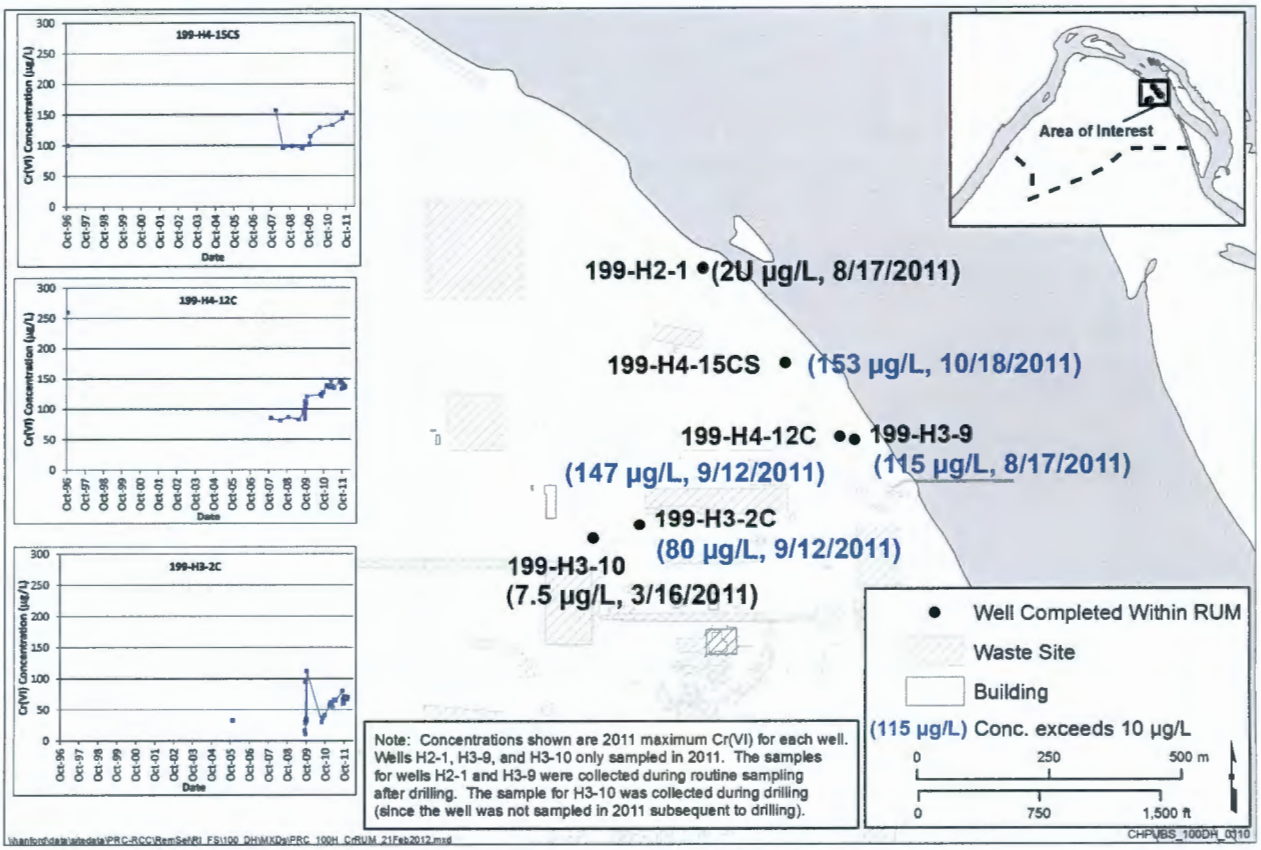

Figure 4-77. Cr(VI) Distribution in the First Water-Bearing Unit in the RUM

During drilling activities, groundwater samples were collected from Wells 199-H2-1, 199-H3-9, and 199-H3-10 to evaluate the presence of contamination, including total chromium and Cr(VI) in some of the deeper water-bearing units within the RUM, with the first of these presumed to be unit B. Total chromium and Cr(VI) concentrations in these groundwater samples were below the laboratory detection limits. Vertical distribution data are presented later in this section.

4.5.1.4 Hexavalent Chromium in Aquifer Tubes

Additional Cr(VI) aquifer tube data from the 2011 annual report (*Hanford Site Groundwater Monitoring for 2011* [DOE/RL-2011-118]) are presented on Figures 4-78 through 4-83 and discussed in the following paragraphs.

Figure 4-78 shows the historical range and the maximum Cr(VI) concentrations in 100-D aquifer tubes. At most sites, the 2011 concentrations were at the lower end of the historical range. Figure 4-79 shows fall 2011 Cr(VI) concentrations with depth in a cross-section near the ISRM barrier and southern Cr(VI) plume from upstream to downstream aquifer tubes at 100-D. The Cr(VI) concentrations in 2011 were the highest in aquifer tube DD-50 at 10.6 µg/L. Cr(VI) concentrations in aquifer tubes downgradient of the northern plume have generally declined since the late 1990s (Figure 4-80), although exhibiting some seasonal variation. Figure 4-80 shows results for aquifer tubes near the northern Cr(VI) plume. Throughout 100-D, Cr(VI) concentrations are significantly lower in 2011 than in previous years, with analytical results below 25 µg/L at all sampled locations except Redox-1-6.0. Concentrations in Redox-1-6.0 were at 96.70 µg/L in January 2011, but decreased to below detection by fall of 2011. The Cr(VI) concentrations in aquifer tube Redox-1-6, which had a value of 384 µg/L in 2009.

Figure 4-81 shows the historical range and the maximum Cr(VI) concentrations in 100−H aquifer tubes. Figures 4-82 and 4-83 show the fall 2011 Cr(VI) concentrations with depth in a cross-section through aquifer tubes from the east side of the horn area and through 100-H. Cr(VI) concentrations in aquifer tubes in the main 100-H area were below 10 µg/L with the exception of aquifer tube C7650, which had a concentration of 26.6 µg/L in December 2011. Concentrations along the horn area were all less than the AWQC value of 10 µg/L, excluding C6287.

Six new aquifer tubes were installed to meet the criteria of data need 5 (100-D/H Work Plan [DOE/RL-2008-46-ADD1]). The four new aquifer tubes at 100-D (C7645, C7646, C7647, C7648) were installed to define the lateral extent of contamination southwest of the ISRM barrier, while the two new aquifer tubes at 100-H (C7649 and C7650) were installed to define the extent of contamination between the 116-H-7 Retention Basins and the river. Data collected from the four 100-D aquifer tubes indicated that Cr(VI) concentrations are less than 10 µg/L. At 100-H, aquifer tube C7649 had Cr(VI) concentrations below detection limits. Aquifer tube C7650 had 26.6 µg/L of Cr(VI) detected in fall 2011, with 2010 results ranging from 6.6 µg/L in August to 30.8 µg/L in December. Aquifer tube data from 100-D/H generally indicate that Cr(VI) concentrations are less than the AWQC of 10 µg/L, with few exceptions.

4.5.2 Vertical Distribution of Hexavalent Chromium

The distribution of contaminants is important to understand not just horizontally, but also vertically, to ensure that the plume is well defined. For example, a well that is completed in the top of an aquifer may not indicate the presence of an analyte that is denser than water, and is therefore only present at the bottom of that unit or in a deeper water-bearing zone. To evaluate the vertical distribution of Cr(VI), existing wells were sampled at discrete depths, and wells installed during the RI were sampled at discrete depths during drilling.

Four existing wells were sampled in early 2011 using rigid porous polyethylene (RPP) samplers: Well 199-D5-99, 199-D5-122, 199-D5-126, 199-D, and 699-97-45. Each well was equipped with four RPP passive samplers placed at different depth intervals within each well to evaluate the vertical stratification of Cr(VI) within the unconfined aquifer. Groundwater monitoring Well 199-D5-99 is located near the former 100-D-12 French Drain waste site, in the southern 100-D plume, Well 199-D5-122 is located in the hot spot of the southern 100-D plume, and Well 199-D5-126 is located within the hot spot of the northern 100-D plume. Monitoring Well 699-97-45 is located in the horn, and unlike the other wells is screened in the Hanford formation.

Predetermined depth intervals of each RPP sampler were based on where the water table and RUM surface were encountered at each well location. RPP sampler placement was as follows: at the water table, straddling the screen at the RUM surface, at 0.6 to 0.9 m (2 to 3 ft) above the RUM surface, and between the upper and intermediate RPP. The results of the study are reported in *Cr(VI) Density Stratification Study, 100-D Area, Hanford Site, Washington* (SGW-49739).

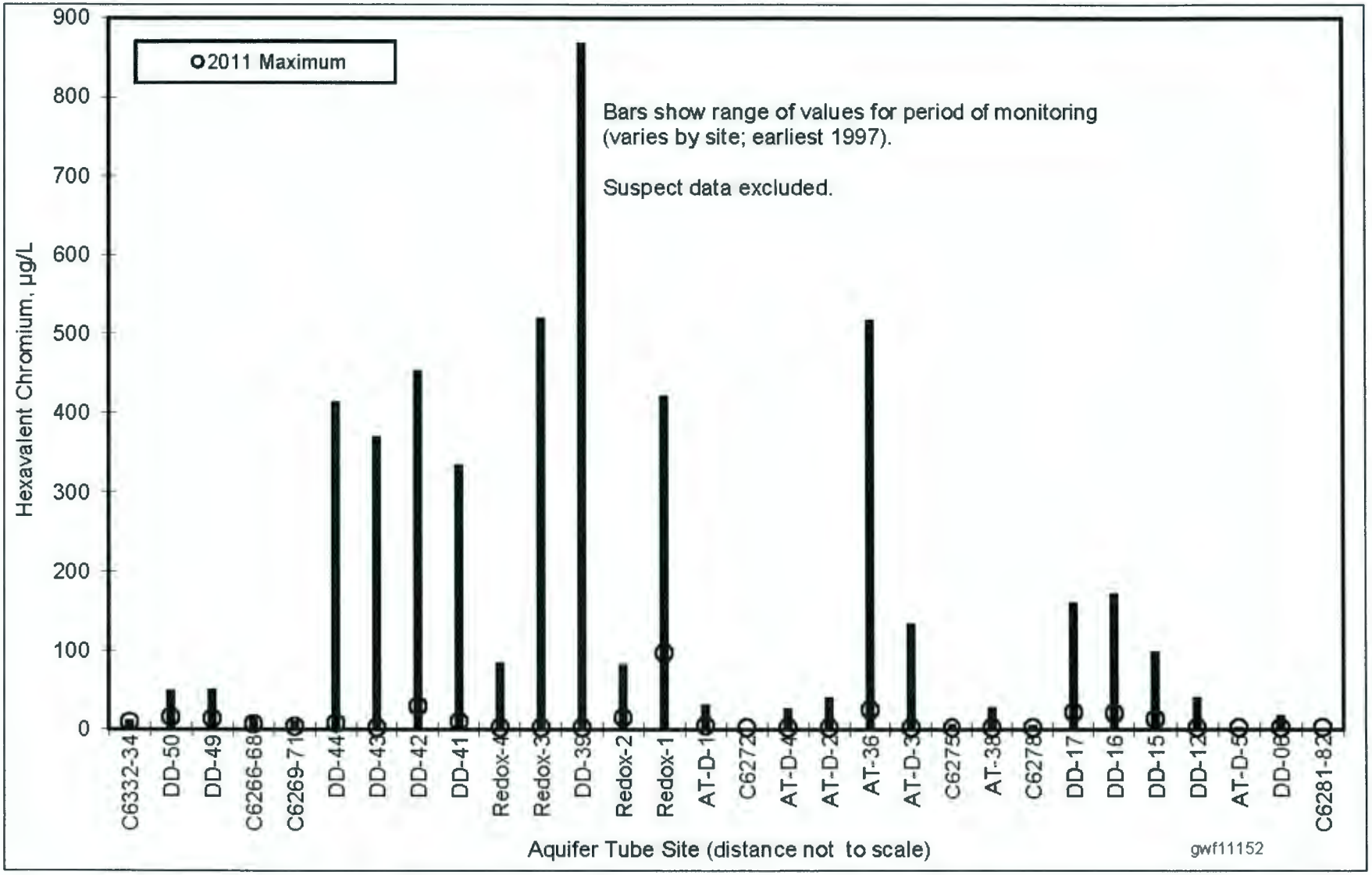

Figure 4-78. 100-D Area Aquifer Tubes – Historical Data Range

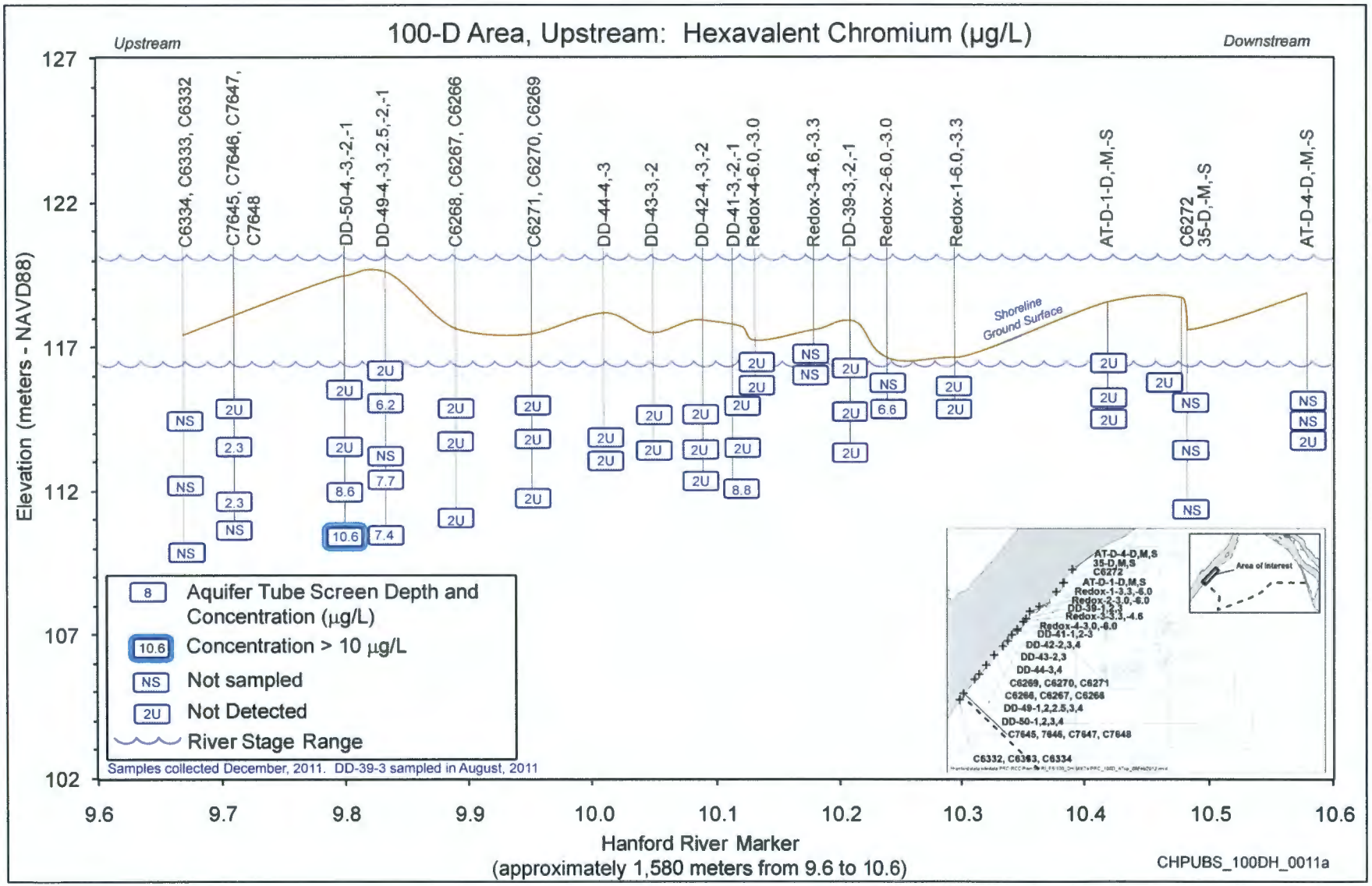

Figure 4-79. 100-D Area Upstream Aquifer Tubes

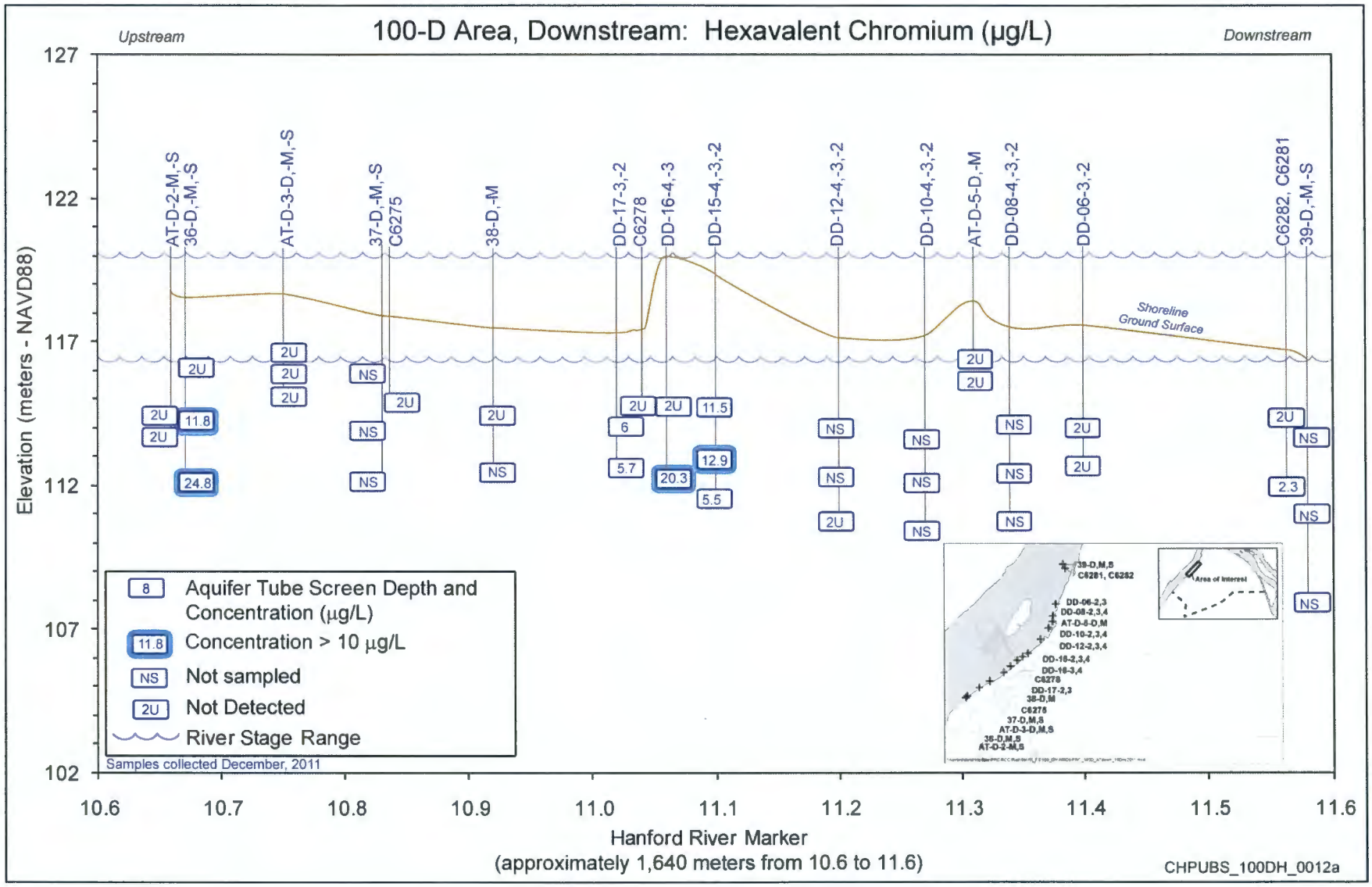

Figure 4-80. 100-D Area Downstream Aquifer Tubes

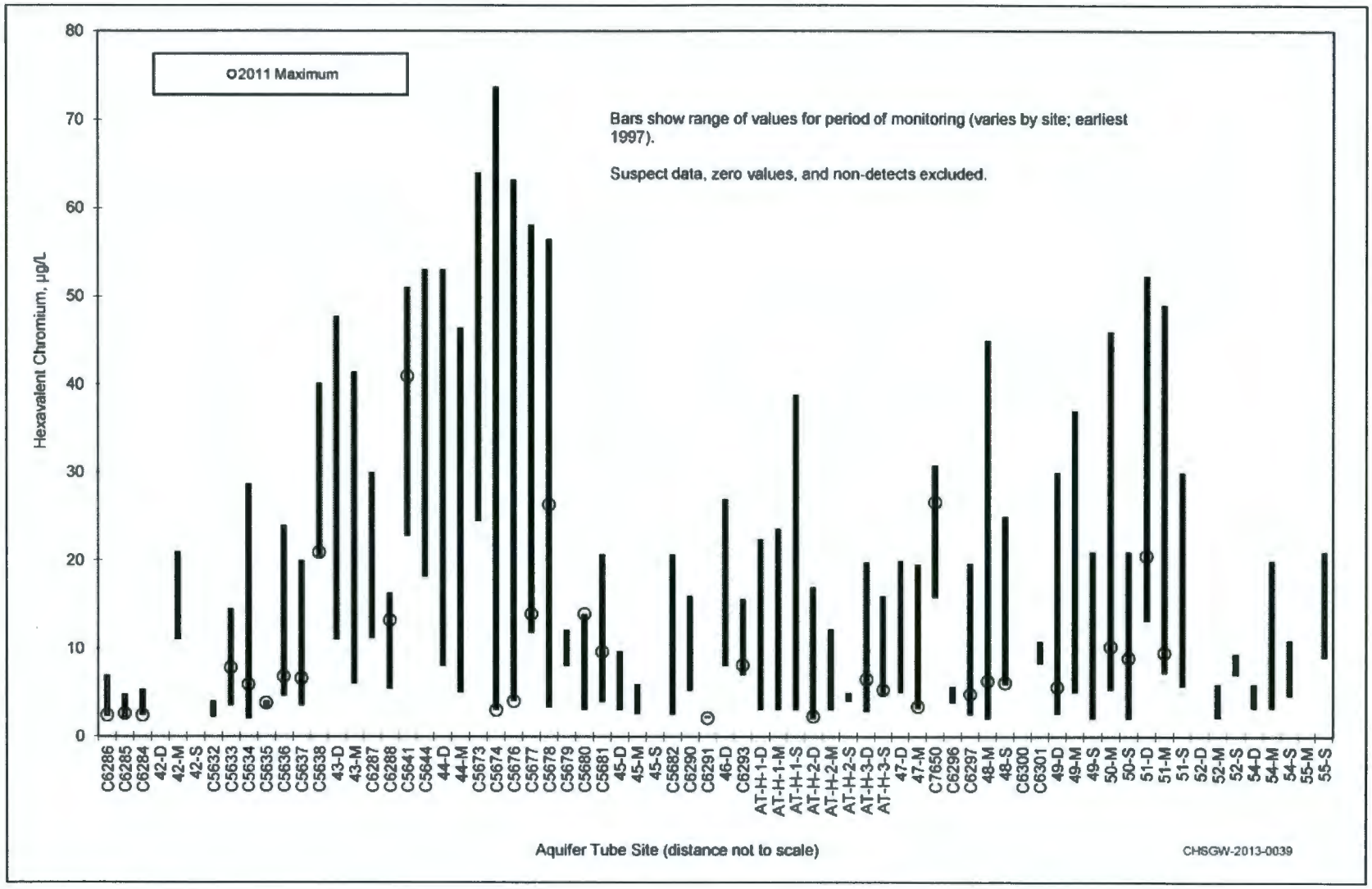

Figure 4-81. 100-H Area Aquifer Tubes – Historical Data Range

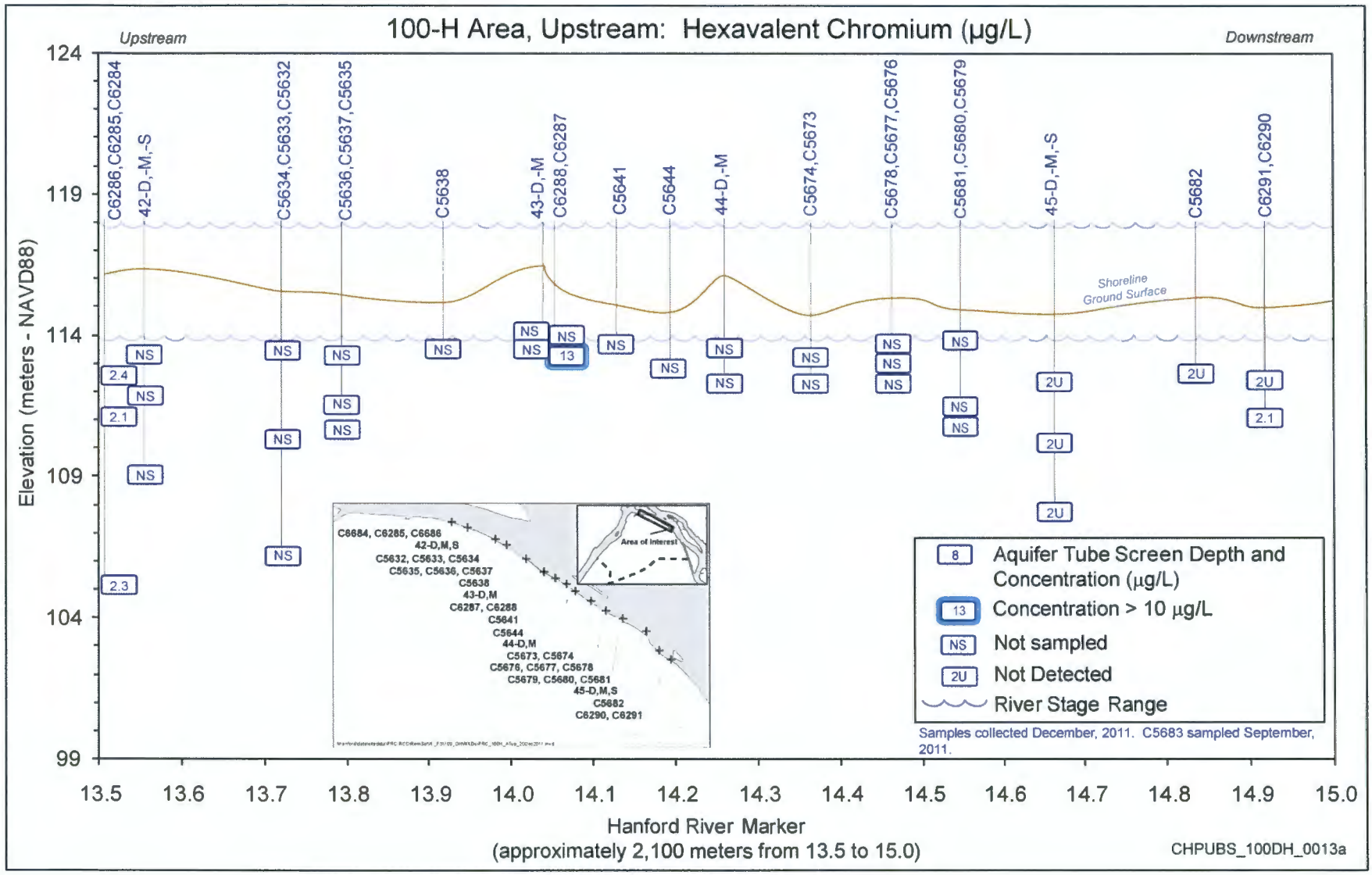

Figure 4-82. 100-H Area Upstream Aquifer Tubes

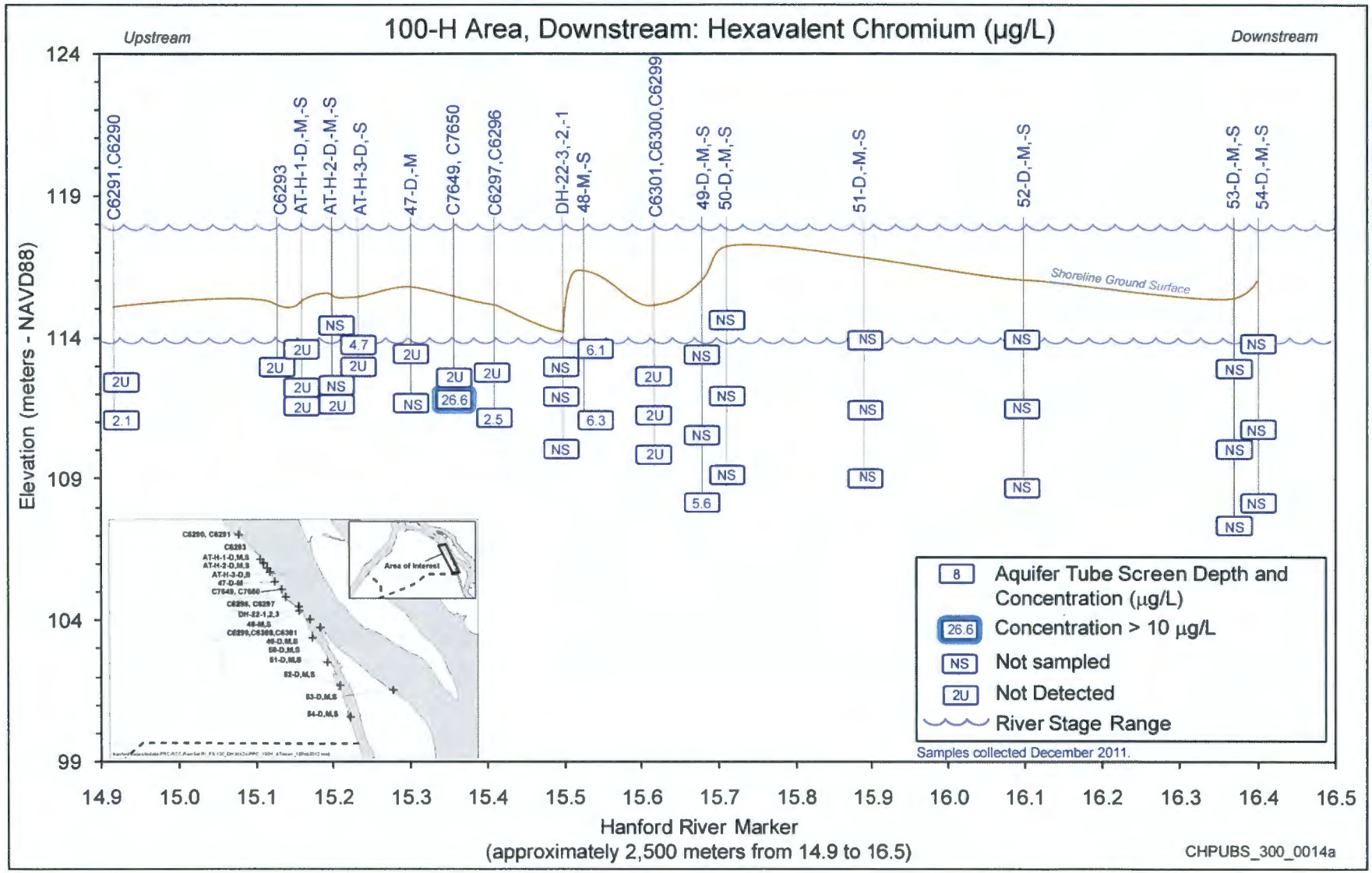

Figure 4-83. 100-H Area Downstream Aquifer Tubes

As shown in Table 4-14, the Cr(VI) results from Wells 199-D5-99 and 199-D5-122 indicate the presence of some vertical stratification in the unconfined aquifer with higher concentrations near the bottom of the aquifer. The RPP sampling conducted in Well 199-D5-99 showed the greatest vertical Cr(VI) stratification at 100-D, because concentrations in the upper 75 percent of the unconfined aquifer were at approximately 1,500 µg/L, then increased 9,960 µg/L in the RPP sampler placed at the RUM surface. This well is located near waste site 100-D-100 and the former railcar unloading facility.

Concentrations in Well 199-D5-122 were higher in the lower 75 percent of the unconfined aquifer, with concentrations of about 26,000 µg/L. The RPP sample from the top of the aquifer had Cr(VI) at 6,590 µg/L. However, as with Well 199-D5-99, the variation is found in a single well and lacks an apparent downward or increasing trend. This is more indicative of variation within the water column than actual stratification trends.

In Well 699-97-45, lower Cr(VI) concentrations are present at the bottom of the aquifer. However, the analytical results only vary by a relatively small amount between depths, resulting in inconclusive results in that well. Vertical stratification of Cr(VI) was not apparent in Well 199-D5-126.

Table 4-14. Vertical Distribution of Cr(VI) in Four 100-D/H Unconfined Aquifer Wells

Well Name (Borehole ID)	General Location	Sample Depth m (ft) bgs	Date Collected	Sample ID (HEIS #)	Cr(VI) Sample Results (µg/L)
199-D5-99 (C5392)	100-D, Southern Plume	26.46 (86.80)	1/17/2011	B2BDM5	1,440
		28.80 (94.50)	1/17/2011	B2BDM6	1,460
		32.46 (106.50)	1/17/2011	B2BDM7	1,490
		33.38 (109.50)	1/17/2011	B2BDM8	9,960
199-D5-122 (C5936)	100-D, Southern Plume	26.44 (86.75)	1/17/2011	B2BDN0	6,590
		28.96 (95.00)	1/17/2011	B2BDN1	25,700
		31.85 (104.50)	1/17/2011	B2BDN2	26,200
		32.77 (107.50)	1/17/2011	B2BDN3	26,900
199-D5-126 (C6390)	100-D, Northern Plume	26.42 (86.68)	1/31/2011	B2BDN5	1,510
		28.73 (94.25)	1/31/2011	B2BDN6	1,510
		32.54 (106.75)	1/31/2011	B2BDN7	1,520
		33.45 (109.75)	1/31/2011	B2BDN8	1,510
699-97-45 (C5659)	Horn	9.75 (32.00)	1/31/2011	B2BDP0	53.9
		9.91 (32.50)	1/31/2011	B2BDP1	53.2
		11.09 (36.40)	1/31/2011	B2BDP2	55.6
		12.01 (39.40)	1/31/2011	B2BDP3	24.1

During the RI, 17 monitoring wells and 10 soil borings (5 of which were completed as temporary wells) were installed. Groundwater samples were collected during drilling at discrete depth intervals to characterize the vertical extent of contaminants. Most of the wells were completed in the unconfined aquifer; however, five wells were completed in the first water-bearing unit of the RUM.

Table 4-15 and Table 4-16 present the groundwater analytical results for total chromium and Cr(VI) by depth for the wells and boreholes drilled as part of the RI/FS field effort. As shown in Table 4-15, vertical stratification of Cr(VI) is indicated to some extent in the 100-D southern plume unconfined aquifer. Stratification is not indicated in the 100-D northern plume or the 100-H plume.

Laboratory data qualifiers are included in Tables 4-15 and 4-16, with "D" indicating a dilution factor, "U" indicating the analyte was not detected above the limiting criteria shown, and "B" indicating that the analyte concentration was near the detection limit for that test method. It should also be noted that the laboratory methods used to determine total chromium and Cr(VI) are different. Method 6010 Metals by inductively coupled plasma (ICP) or Method 200.8 (which has a lower detection limit) is used to determine total chromium, and Method 7196, a color metric method, is used to determine Cr(VI). Method 7196 is susceptible to interference from colored matrices and chemical interference. Because of the potential for interference, as well as differences in sample preparation and analysis procedures, it is generally thought that the total chromium methodology provides a more accurate reading than the Cr(VI) method. However, using current technology, a method is not available that provides both total chromium and Cr(VI) results. Results for individual wells and boreholes are discussed in the following paragraphs.

100-D, Southern Plume: Unconfined Aquifer – These wells are located within the southern plume of 100-D, where the highest concentrations of Cr(VI) have been identified. The analytical results indicate present-day vertical stratification in areas where highly concentrated sodium dichromate was handled. However, it should be noted that this stratification is not well defined or consistent in the aquifer.

Table 4-15. Chromium and Cr(VI) Sample Results from RI Drilling – 100-D

Area	Well Name (Borehole ID, SAP ID)	Sample Depth bgs m (ft)	Chromium (µg/L)	Cr(VI) (µg/L)	Turbidity (NTU)	Specific Conductance (µS/cm)
100-D, Southern Plume: Unconfined Aquifer	199-D3-5 (C7620, Well 2)	27.5 to 28.1 (90.3 to 92.3)	14.4	8	--	--
		27.7 to 28.1 (90.9 to 92.3)	12.4	3.7 (U)	13.2	533
		29.3 to 29.7 (96 to 97.4)	15.8 (D)	2 (U)	9.35	575
		30.8 (101.2)	48.2 (D)	27	--	--
		30.8 (101.2)	49.2 (D)	27	5.96	596
		31.4 (103)	84.7 (D)	73.1	8.45	577
	199-D5-144 (C8668, Well R5 redrill)	28.01 (91.90)	684	703	--	--
		28.01 (91.90)	684	636	293	727
		29.11 (95.51)	403	--	--	--
		29.11 (95.51)	407	--	8.31	533

Table 4-15. Chromium and Cr(VI) Sample Results from RI Drilling – 100-D

Area	Well Name (Borehole ID, SAP ID)	Sample Depth bgs m (ft)	Chromium (µg/L)	Cr(VI) (µg/L)	Turbidity (NTU)	Specific Conductance (µS/cm)
		30.27 (99.31)	304	284	19.5	535
		31.49 to 31.64 (103.3 to 103.8)	257	238	--	--
		31.49 to 31.64 (103.3 to 103.8)	246	241	13.7	535
		32.61 (107)	103	98	302	600
100-D, Southern Plume: RUM Wells	199-D5-141 (C7625, Well R5)	27.5 (90.3)	2,070	2,010 (D)	--	--
		27.5 (90.3)	1,990 (D)	2,100	50.8	642
		29.1 (95.5)	6,080 (D)	6,520 (D)	--	--
		29.1 (95.5)	6,290 (D)	6,510 (D)	7.42	560
		30.6 (100.5)	5,300 (D)	5,440 (D)	4.14	536
		32.5 (106.5)	961 (D)	986	1.36	447
		34.1 (112)	2,470 (D)	2,590 (D)	24.1	432
		49.5 (162.5)	1 (UD)	2 (U)	742	449
		94.1 (308.8)	0.5 (U)	2 (U)	571	338
	199-D5-133 (C7621, Well 3)	26.9 (88.2)	36.9	31.1	--	--
		26.9 (88.2)	36.7 (D)	36	3.97	651
		28.3 (92.7)	16.5 (D)	2 (U)	336	596
		29.8 (97.8)	27.2 (D)	7.4	--	--
		29.8 (97.8)	20.4 (D)	9.9	25.4	626
		31.4 (103)	22.3 (D)	2 (U)	214	624
	199-D5-132 (C7622, Well 4)	27 (88.7)	28.9 (D)	15.5	--	--
		27 - 28.1 (88.7 - 92.3)	24.5	18	93.1	675
		29.4 (96.4)	17.5 (D)	7.1	--	--
		29.4 (96.4)	19.7 (D)	6.9	45.8	667
		31.1 (102)	34.5 (D)	16.1	8.47	654
		32. (105)	29 (D)	9	91.6	646
	199-D6-3	28.7 (94)	22	17.6	--	--

Table 4-15. Chromium and Cr(VI) Sample Results from RI Drilling – 100-D

Area	Well Name (Borehole ID, SAP ID)	Sample Depth bgs m (ft)	Chromium (µg/L)	Cr(VI) (µg/L)	Turbidity (NTU)	Specific Conductance (µS/cm)
	(C7623, Well 5)	28.7 (94)	35.7	2 (U)	572	846
100-D, Northern Plume: Unconfined Aquifer		30.2 (99)	39.9 (D)	2 (U)	--	--
		30.2 (99)	39.2 (D)	2 (U)	166	825
		30.9 (101.5)	37.3 (D)	8.2	121	840
	199-D5-140 (C7866, Well 9)	27.5 - 27.8 (90.2 - 91.3)	358 (D)	290	228	656
		27.5 - 27.8 (90.2 - 91.3)	336	327	--	--
		28.7 (94.2)	540 (D)	521	8.2	654
		28.7 (94.2)	580 (D)	513	--	--
		30.2 (99)	460	425	100	657
		31.5 (103.3)	450 (D)	376	62.4	631
	199-D5-143 (C8375, Well 9 redrill)	27.7 (91)	1,330	1,210 (D)	24.3	576
		27.7 (91)	1,260 (D)	--	--	--
		28.9 (95)	1,240 (D)	1,160 (D)	101	566
		28.9 (95)	1,250 (D)	1,140 (D)	--	--
		31.2 (102.5)	1,360 (D)	1,260 (D)	22.5	566
		31.7 (104)	1,460 (D)	1,460 (D)	13.2	557
100-D, Northern Plume: Unconfined Aquifer Boreholes	199-D8-101 (C7852, Waste Site 116-DR-1&2)	21.9 (72)	34.1	27	0.64	727
		21.9 (72)	35.6	27	--	--
	199-D5-142 (C7857, Waste Site 118-D-6)	26.5 - 27.4 (87.1 - 89.8)	80	79	12.2	161
		26.5 - 27.4 (87.1 - 89.8)	82.1	78	--	--
	Borehole C7850 (Waste Site 116-DR-9)	21.5 (70.4)	33.5	23	--	--
		21.5 (70.4)	31.4	39	936	614
	Borehole C7851 (Waste Site 116-D-7)	20.3 - 21 (66.7 - 69)	33	8	--	--
		20.3 - 21 (66.7 - 69)	18.9	11	>1,000	588

Table 4-15. Chromium and Cr(VI) Sample Results from RI Drilling – 100-D

Area	Well Name (Borehole ID, SAP ID)	Sample Depth bgs m (ft)	Chromium (µg/L)	Cr(VI) (µg/L)	Turbidity (NTU)	Specific Conductance (µS/cm)
	Borehole C7855 (Waste Site 116-D-1B)	27.8 (91.2)	151	143	--	--
		27.8 (91.2)	150	144	364	684
100-D, Northern Plume: RUM Well	199-D5-134 (C7624, Well R4)	28 (92)	1,480	1,560 (D)	--	--
		28 (92)	1,420	1,480	10.4	584
		29.6 (97)	1,290 (D)	1,430	--	--
		29.6 (97)	1,350 (D)	1,460	15.3	578
		31.1 (102)	1,250 (D)	1,670	35	567
		32.7 (107.3)	1,090 (D)	1,090 (D)	35.8	549
		41.3 (135.5)	12.6 (D)	12.2	32.1	462
		46.9 (154)	1.31 (BD)	2 (U)	259	330
		82 (268.9)	1.27 (BD)	2 (U)	82	389

Notes:

Data shown excludes soil samples collected through water extraction (WE) methods due to comparability of values. WE concentrations are consistently lower the acid extraction methods. Also excluded from dataset are results reported in water units (that is, µg/L) for soil samples and "R" and "Y" flagged data.

Turbidity and conductivity were analyzed once per depth interval. Turbidity and conductivity values, which are field measurements, were tied to analytical samples where possible. Some samples were filtered.

Shaded cells indicate the sample was collected from a water-bearing unit within the RUM.

B = Analyte was detected but the result is near the detection limit of the test method.

D = Analyte was reported at a secondary dilution factor.

U = Analyzed for but not detected above limiting criteria.

µS/cm = microSiemens per centimeter

NTU = nephelometric turbidity units

NS = Not specified

"—" indicates analyte was not sampled for at that location.

Table 4-16. Chromium and Cr(VI) Sample Results from RI Drilling – 100-H

Area	Well Name (Borehole ID, SAP ID)	Sample Depth bgs - m (ft)	Chromium (µg/L)	Cr(VI) (µg/L)	Turbidity (NTU)	Specific Conductance (µS/cm)
100-H: Unconfined Aquifer	199-H3-6 (C7626, Well 6)	15.2 (49.8)	14.8 (D)	2.5 (B)	3.37	473
		15.2 - 15.8 (49.8 - 51.8)	13.6	17	3.31	471
		16.4 (53.9)	11.5 (D)	2.7 (B)	--	--
	199-H3-7 (C7627, Well 7)	15.1 (49.5)	11.4	5.1	--	--
		15.1 (49.5)	22.2	13	65	524
		15.7 (51.6)	25.4 (D)	12.6	36.9	502
		15.7 (51.6)	25.1 (D)	16.1	--	--
	199-H6-3 (C7628, Well 10)	14.8 (48.5)	15.4	2 (U)	9.21	621
		14.8 (48.5)	23 (D)	6	--	--
		16.2 (53.1)	20.1 (D)	2 (U)	159	618
		16.2 (53.1)	22.8 (D)	2 (U)	--	--
		19.5 (64)	34.4 (D)	15.2	34.7	601
	199-H6-4 (C7629, Well 11)	13.9 (45.7)	12.9	8.7	--	--
		13.9 (45.7)	13.6 (D)	9.2	2.58	477
		13.9 (45.7)	13.3 (D)	8	--	--
		14.6 (48)	19.2 (D)	6.3	4.45	475
		16.2 (53)	14.4 (D)	6.8	5.53	478
		18.4 (60.5)	12.2 (D)	3.2 (B)	51.5	462
	199-H1-7 (C7630, Well 12)	NS		--	--	--
		NS		--	--	--
100-H: Unconfined Aquifer, Boreholes	199-H4-84 (C7860, Waste Site 116-H-6)	14.1 - 14.8 (46.2 - 48.6)	3.91	3.7 (U)	--	--
		14.1 - 14.8 (46.2 - 48.6)	4.2	3.7 (U)	73.8	344
	199-H4-83 (C7861, Waste Site 116-H7)	12.2 - 12.9 (40 - 42.3)	7.29	4	--	--
		12.2 - 12.9 (40 - 42.3)	7.58	5	43.3	466

Table 4-16. Chromium and Cr(VI) Sample Results from RI Drilling – 100-H

Area	Well Name (Borehole ID, SAP ID)	Sample Depth bgs - m (ft)	Chromium (µg/L)	Cr(VI) (µg/L)	Turbidity (NTU)	Specific Conductance (µS/cm)
	199-H3-11 (C7863, Waste Site 118-H-6)	16.5 (54.2)	25.3	11	--	--
		16.5 (54.2)	26.2	12	7.74	596
	Borehole C7862 (Waste Site 116-H-4)	15.2 - 16 (49.9 - 52.4)	2 (U)	3.7 (U)	--	--
		15.2 - 16 (49.9 - 52.4)	2 (U)	3.7 (U)	>1,000	526
	Borehole C7864 (Waste Site 116-H-1)	15.1 (49.4)	6.84	3.7 (U)	--	--
		15.1 (49.4)	7.54	3.7 (U)	18.3	490
100-H: RUM Wells	199-H2-1 (C7631, Well R3)	9.2 (30.1)	7.44	2 (U)	--	--
		9.2 (30.1)	10.1 (D)	2 (U)	--	--
		9.2 (30.1)	9.06 (BD)	3.7 (U)	46.8	228
		10.64 (34.9)*	8.8 (BD)	5.9	239	241
		19.2 (62.9)	11.4 (D)	8.6	298	265
		48.2 (158.3)	1 (UD)	2 (U)	13.7	360
		54.7 (179.6)	2.87 (BD)	2 (U)	79.2	356
	199-H3-9 (C7639, Well R1)	12.3 (40.4)	7.84	6	--	--
		12.3 (40.4)	7.84 (BD)	3.7 (U)	6.88	478
		13.8 (45.2)	3.81 (BD)	3.1 (B)	--	--
		13.8 (45.2)	3.58 (BD)	2.9 (B)	87.7	410
		14.2 (46.5)	8.85 (BD)	2 (U)	38.3	321
		20.8 (68.4)	319 (D)	287	599	259
		40.8 (134)	4.14 (BD)	2 (U)	132	332
		53.9 (177)	4.09 (BD)	2 (U)	9.74	369
	199-H3-10 (C7640, Well R2)	13.9 (45.5)	13.8	7.5	--	--
		13.9 (45.5)	13.3 (D)	11	6.54	470
		15.2 (49.9)	10.5 (D)	3.7 (B)	2.98	451
		15.2 (49.9)	10.2 (D)	3.3 (B)	6.85	450
		16.1 (52.8)	13.1 (D)	2 (B)	914	289

Table 4-16. Chromium and Cr(VI) Sample Results from RI Drilling – 100-H

Area	Well Name (Borehole ID, SAP ID)	Sample Depth bgs - m (ft)	Chromium (µg/L)	Cr(VI) (µg/L)	Turbidity (NTU)	Specific Conductance (µS/cm)
		60.4 (198)	2.05 (BD)	2 (U)	>1,000	347
		68.2 (223.6)	2.32 (BD)	2 (U)	54.1	377

Notes:

Data shown excludes soil samples collected through water extraction (WE) methods due to comparability of values. WE concentrations are consistently lower the acid extraction methods. Also excluded from dataset are results reported in water units (i.e., µg/L) for soil samples and "R" and "Y" flagged data.

Turbidity and conductivity were analyzed once per depth interval. Turbidity and conductivity values are field measurements. The sample may have been subsequently filtered.

Shaded cells indicate the sample was collected from a water-bearing unit within the RUM.

* = This depth was recorded incorrectly as 106.9 m, which is well beyond the total depth drilled of 57.6 m.

B = Analyte was detected but the result is near the detection limit of the test method.

D = Analyte was reported at a secondary dilution factor.

U = Analyzed for by not detected above limiting criteria.

µS/cm = microSiemens per centimeter

NTU = nephelometric turbidity units NS = Not specified

"—" indicates analyte was not sampled for at that location.

Well 199-D3-5 (C7620, Well 2) – Cr(VI) concentrations in groundwater are greater than the AWQC of 10 µg/L at this location. The Cr(VI) concentrations are less than total chromium as expected because Cr(VI) typically represents only a portion of the chromium oxidation states present. Cr(VI) concentrations in groundwater increase with depth up to 73.1 µg/L at the 31.4 m (103 ft) depth, with a corresponding total chromium result of 84.7 µg/L. This indicates that the majority of chromium at this location consists of Cr(VI). No post-installation groundwater samples were collected from this well in 2011.

Well 199-D5-144 (C8668, Well R5 redrill) – This well is located adjacent to waste site 100-D-100, which is undergoing excavation. Analytical results in this location were lower than those identified in a slightly downgradient well (199-D5-141). Groundwater sample results for both Cr(VI) and total chromium concentrations were nearly identical in this location, with results in all sample locations being well over the MTCA ("Groundwater Cleanup Standards" [WAC 173-340-720]) level of 48 µg/L. The concentrations at the top of the RUM were the lowest detected in groundwater, at 103 and 98 µg/L for total chromium and Cr(VI), respectively. No post-installation groundwater samples were collected from this well in 2011.

100-D, Southern Plume: RUM Well – This well extends into the RUM and groundwater samples were collected from both the unconfined aquifer and water-bearing units within the RUM.

Well 199-D5-141 (C7625, Well R5): In the unconfined aquifer, Cr(VI) concentrations are relatively high and distributed over the entire thickness. Cr(VI) concentrations ranging from 986 to 6,520 µg/L, as presented in Table 4-15. The total chromium and Cr(VI) mass in these samples are nearly equal, indicating that chromium is predominantly in the mobile hexavalent form and that little natural reduction is occurring in the unconfined aquifer at this location. Total chromium appears to be less than Cr(VI) in many of the samples throughout the depth intervals, indicating a potential error in laboratory analysis.

Overall, the results suggest that the chromium in the unconfined aquifer is in a very mobile state. Within the RUM, neither total chromium nor Cr(VI) was detected. The well was screened within the RUM, and neither total chromium nor Cr(VI) was detected in post-installation groundwater samples collected in August and October 2011.

100-D, Northern Plume: Unconfined Aquifer – These wells are located within the northern plume of 100-D and historically have had significantly lower Cr(VI) concentrations in groundwater than in the southern plume. However, there is also a hot spot with localized higher concentrations.

Well 199-D5-133 (C7621, Well 3) – No significant contaminant spikes are observed near the PRZ. The groundwater sample results from the upper 10 m of the 25 m (33 of the 82 ft) thick unconfined aquifer are slightly elevated, however, suggesting a shallow intrusion of contaminated groundwater within the unconfined aquifer. At a depth of approximately 27.7 m (91 ft), the soil boring log indicates that the felsic-rich material is no longer present and the silt content increases. The felsic material tends to be more alkaline and the Cr(VI) would remain in that valence state. This is consistent with the analytical results, which show lower Cr(VI) where the silt content of the geologic material increases. There appears to be some reduction to trivalent chromium in this lower portion of the aquifer. A post-installation groundwater sample collected from this well in August 2011 identified total chromium and Cr(VI) concentrations at 12 μg/L and 8.2 μg/L, respectively. The total chromium values are slightly lower than the 16.5 to 36.9 μg/L detected during drilling. The post-installation results are consistent with the concentrations of Cr(VI), which ranged from 36 μg/L to below detection during drilling activities.

Well 199-D5-132 (C7622, Well 4) – In the unconfined aquifer, Cr(VI) concentrations range from 6.9 to 18 μg/L and total chromium values range from 17.5 to 34.5 μg/L. Total chromium is higher than Cr(VI) in all samples, and the ratio indicates a lower mobility within the aquifer at this location, with the exception of the uppermost sample. Slightly higher Cr(VI) concentrations were detected near the top and near the bottom of the aquifer. The stratigraphic units noted in the borehole do not indicate any significant variability in lithology except for a slight increase in gravel content near the bottom of the well. A post-installation sample collected from this well in August 2011 had total chromium at 43 and 41 μg/L, with Cr(VI) detected at 41.3 μg/L. The total chromium and Cr(VI) concentrations are slightly higher than identified during drilling.

Well 199-D6-3 (C7623, Well 5) – In the unconfined aquifer, Cr(VI) concentrations ranged from below detection to 17.6 μg/L at a depth of 28.7 m (94 ft) bgs, the shallowest sample depth. Laboratory results from the duplicate sample at that depth were below detection, introducing some uncertainty to the analytical results from that sample interval. Total chromium results in the unconfined aquifer ranged from 22 to 39.9 μg/L, and are relatively consistent throughout the aquifer thickness. A post-installation groundwater sample collected from this well in August 2011 had total chromium and Cr(VI) concentrations of 10 μg/L and 4.4 μg/L, respectively. These values are consistent with the concentration range identified during borehole groundwater sampling activities.

Well 199-D5-140 (C7866, Well 9) – Total chromium concentrations in the unconfined aquifer during drilling ranged from 336 to 580 μg/L, while Cr(VI) concentrations ranged from 290 to 521 μg/L. The total chromium and Cr(VI) track closely in most samples, indicating that most of the Cr(VI) in groundwater is in the mobile hexavalent oxidation state. A post-installation groundwater sample collected from this well in June 2011 had total chromium concentrations of 372 and 375 μg/L. The Cr(VI) concentration from the same sample date was 388 μg/L, which is slightly higher than the total chromium level. This difference is likely within the range of laboratory error. Analytical values are consistent with the concentration range identified during borehole groundwater sampling activities. The well was decommissioned in mid-June 2011 for continued waste site remediation activities.

Well 199-D5-143 (C8375, Well 9 redrill) – In the unconfined aquifer, total chromium concentrations ranged from 1,240 to 1,460 µg/L, while Cr(VI) concentrations ranged from1,140 to 1, 460 µg/L. The total chromium and Cr(VI) results are nearly identical, indicating that most of the chromium is in the hexavalent oxidation state and mobile. These concentrations are consistent with the location of the well in the 100-D northern plume. A post-installation groundwater sample collected from this well in August 2011 had total chromium and Cr(VI) detected at 1,420 µg/L and 1,480 µg/L, respectively. These values are consistent with the concentration range identified during borehole groundwater sampling activities.

100-D, Northern Plume: Unconfined Aquifer Boreholes – These boreholes were installed near selected waste sites in the 100-D northern plume. Because the boreholes were drilled primarily to determine soil conditions, they were not extended to the top of the RUM, as were the wells. It should be noted that two of these boreholes were converted to temporary wells to obtain groundwater samples because of low water production within the borehole at the time of drilling.

Temporary Well 199-D8-101 (C7852, Waste Site 116-DR-1&2) – Groundwater samples were collected from one location during drilling, the upper 1.5 m (5 ft) of the unconfined aquifer. Samples from that location were analyzed as duplicates with results being nearly identical. Total chromium concentrations were detected at about 35 µg/L and Cr(VI) concentrations were at 27 µg/L, indicating that the chromium in the unconfined aquifer at this location is primarily in the mobile Cr(VI) state. Three post-installation groundwater samples were collected from this well in 2011, with samples collected in April, June, and July. Total chromium concentrations showed a decreasing trend from April through July (64 µg/L, 53 µg/L, and 13 µg/L, respectively). Cr(VI) concentrations also showed a similar decreasing trend from April through July (61 µg/L, 50 µg/L, and 9 µg/L, respectively). The decreasing concentrations can be attributed to pump-and-treat system operations. The previous borehole total chromium and Cr(VI) groundwater results fall in the middle of the post-installation results.

Temporary Well 199-D5-142 (C7857, Waste Site 118-D-6) – Groundwater samples were collected in the upper 1.5 m (5 ft) of the unconfined aquifer. The duplicate samples had total chromium at concentrations of 80 and 82.1 µg/L and Cr(VI) concentrations were at 79 and 78 µg/L, indicating that the chromium in the unconfined aquifer at this location is primarily in the mobile Cr(VI) state. A post-installation groundwater sample collected from this well in April 2011 had total chromium and Cr(VI) detected at 21.6 µg/L and 16 µg/L, respectively. These values are approximately four times lower than previously identified during borehole groundwater sampling activities and are attributed to the nearby extraction and injection wells.

Borehole C7850 (Waste Site 116-DR-9) – Groundwater samples were collected from the upper 1.5 m (5 ft) of the unconfined aquifer. Total chromium was detected at 31.4 and 33.5 µg/L and Cr(VI) was detected at 23 and 39 µg/L, indicating that the chromium in the aquifer at this location is primarily in the mobile Cr(VI) state.

Borehole C7851 (Waste Site 116-D-7) – Groundwater samples were collected from the upper 1.5 m (5 ft) of the unconfined aquifer. Total chromium was detected at 18.9 to 33 µg/L, while Cr(VI) was detected at 8 to 11 µg/L. The duplicate sample results for total chromium were not as similar as would be expected for water samples, introducing some uncertainty in the data quality. The lithology identified in the borehole log did not indicate any conditions affecting sample collection. The values of Cr(VI) at this location are considered more accurate.

Borehole C7855 (Waste Site 116-D-1B) – Groundwater samples were collected from the upper 1.5 m (5 ft) of the unconfined aquifer. Samples from that location were analyzed as duplicates with concentrations being nearly identical. Total chromium concentrations were reported at 150 and 151 µg/L, and Cr(VI) concentrations were reported at 143 and 144 µg/L. This indicates that the chromium in the aquifer at this location is primarily in the mobile Cr(VI) state.

100-D, Northern Plume: RUM Well – This well extends into the first water-bearing unit in the RUM and groundwater samples were collected from both the unconfined aquifer and water-bearing units within the RUM.

199-D5-134 (C7624, Well R4) – In the unconfined aquifer, Cr(VI) concentrations are relatively high and uniformly distributed over the entire thickness of the aquifer. Concentrations of total chromium in groundwater decrease from values of 1,480 μg/L near the top of the unconfined aquifer to 1,090 μg/L at the surface of the RUM. The Cr(VI) concentrations range from 1,090 to 1,670 μg/L, with concentrations decreasing below the base of the unconfined aquifer, which is approximately 40 m (131 ft) bgs. The total chromium and Cr(VI) mass are nearly equal, indicating that chromium is predominantly in the mobile hexavalent form and that little natural reduction is occurring in the aquifer at this location. Total chromium was reported as less than Cr(VI) from samples at a depth of about 31 m (100 ft), indicating a potential error in laboratory analysis, or matrix interference.

At this well location, the RUM was encountered at 33.1 m (108.5 ft) bgs. Three samples were collected for laboratory analysis from water-bearing units within the RUM. Total chromium and Cr(VI) were both detected from the first water-bearing unit, with a concentration of 12.6 and 12.2 μg/L, respectively. An evaluation of the boring logs and daily reports indicates that the sample was collected following difficulty during drilling, resulting in a delay in sample collection of more than two days. Because the well drilling was not yet completed, and high concentrations (1,090 μg/L) of total chromium and Cr(VI) were detected in the aquifer above the RUM surface, it is possible that this sample was contaminated from groundwater originating in the unconfined aquifer. Two additional lower groundwater samples were collected within the RUM, and neither total chromium nor Cr(VI) was detected. The first water-bearing unit in the RUM was screened to allow for future sampling from that unit.

100-H: Unconfined Aquifer – These wells are located within 100-H and screened in the unconfined aquifer. The Cr(VI) concentrations in the unconfined aquifer at 100-H have been significantly lower than in 100-D, but remain higher than those found in the horn area.

199-H3-6 (C7626, Well 6) – Total chromium concentrations in the unconfined aquifer were relatively consistent with depth. Concentrations of Cr(VI) show some variability and range from 2.5 to 17 μg/L at the same location. Two separate laboratories analyzed the sample by the same analytical method and reported considerably different results, with the higher results (17 μg/L) being reported for the filtered sample, which is typically considered more reliable and usually lower than results from unfiltered samples, where 2.5 μg/L was reported. No post-installation groundwater samples were collected from this well in 2011.

199-H3-7 (C7627, Well 7) – Total chromium concentrations in groundwater ranged from 11.4 to 25.4 μg/L at this well. Cr(VI) concentrations are less than total chromium as expected, ranging from 5.1 to 16.1 μg/L. The exception is the filtered sample pair where Cr(VI) was higher (13 μg/L) than the total chromium value (11.4) μg/L. Two separate laboratories analyzed one sample from the same location and reported considerably different results, with the higher results (13 μg/L) being reported for the filtered sample, which is typically considered more reliable and usually lower than results from unfiltered samples, where 5.1 μg/L was reported. No post-installation groundwater samples were collected from this well in 2011.

199-H6-3 (C7628, Well 10) – Total chromium concentrations in groundwater range from 15.4 to 34.4 μg/L, with values for Cr(VI) less than total chromium as expected, ranging from undetected up to 15.2 μg/L. The highest Cr(VI) reported at this well (15.2 μg/L) was collected from the bottom of the unconfined aquifer, approximately 2.3 m (7.5 ft) below the previous sample depth. No post-installation groundwater samples were collected from this well in 2011.

199-H6-4 (C7629, Well 11) – Total chromium concentrations in groundwater range from 12.2 to 19.2 μg/L, with values for Cr(VI) less than total chromium as expected, ranging from 3.2 to 9.2 μg/L. Cr(VI) appears to be uniformly distributed through the entire thickness of the aquifer. In contrast to Well 199-H6-3 (C7628, Well 10), the lowest Cr(VI) in groundwater reported at this well (3.2 μg/L) was collected from the bottom of the unconfined aquifer, approximately 3.3 m (10.8 ft) below the previous sample depth. No post-installation groundwater samples were collected from this well in 2011.

199-H1-7 (C7630, Well 12) – Total chromium and Cr(VI) concentrations in groundwater had similar concentrations. The two results presented for total chromium are 16 μg/L and 14 μg/L; however, both results are "B" flagged, indicating the result was close to the detection limit for the test method, accounting for the variability. A post-installation groundwater sample collected from this well in August 2011 had total chromium and Cr(VI) detected at 16 μg/L and 14 μg/L, respectively. These values are consistent with the concentration range identified during borehole groundwater sampling activities.

100-H: Vadose Zone Boreholes – These boreholes were installed near selected waste sites in 100-H. Because the boreholes were drilled primarily to evaluate contaminant concentrations in the vadose zone, they were only drilled into the top of the aquifer. Three of these boreholes were converted to temporary wells screened in approximately 1.5 m (5 ft) into the aquifer to obtain groundwater samples.

199-H4-84 (C7860, Waste Site 116-H-6) – Total chromium concentrations in groundwater are approximately 4 μg/L, with Cr(VI) not detected. The duplicate sample results are nearly identical. A post-installation groundwater sample collected from this well in June 2011 had total chromium and Cr(VI) detected at 28.6 μg/L and 25 μg/L, respectively. The total chromium and Cr(VI) concentrations are higher than previously identified in borehole groundwater samples. It is not known what caused this increase, but this well will continue to be monitored.

199-H4-83 (C7861, Waste Site 116-H7) – Total chromium was detected in groundwater at 7.3 and 7.6 μg/L, with Cr(VI) concentrations of 4 and 5 μg/L. The duplicate sample results are essentially the same. A post-installation groundwater sample collected from this well in June 2011 had total chromium and Cr(VI) detected at 8.3 μg/L and 6 μg/L, respectively. These values are consistent with the concentration range identified during borehole groundwater sampling activities.

199-H3-11 (C7863, Waste Site 118-H-6) – Total chromium was detected at 25.3 and 26.2 μg/L, with Cr(VI) concentrations of 11 and 12 μg/L. The Cr(VI) concentrations are less than total chromium. This is expected where Cr(VI) represents a single oxidation state within the total chromium concentration present. A post-installation groundwater sample collected from this well in June 2011 had total chromium and Cr(VI) concentrations of 6.6 μg/L and 4 μg/L, respectively. These values are lower than previously identified during borehole groundwater sampling activities.

Borehole C7862 (Waste Site 116-H-4) – Concentrations of both Cr(VI) and total chromium were below detection limits.

Borehole C7864 (Waste Site 116-H-1) – Total chromium concentrations were reported at 7.5 and 6.8 μg/L. The Cr(VI) concentrations were below detection limits.

100-H: RUM Wells – These wells extend into the RUM and groundwater samples were collected from both the unconfined aquifer and lower water-bearing units within the RUM.

199-H2-1 (C7631, Well R3) – In the unconfined aquifer, Cr(VI) concentrations are below the detection limit at all depth intervals except for at the RUM surface. The sample at the bottom of the unconfined aquifer had a Cr(VI) concentration of 5.9 μg/L. Total chromium concentrations in the unconfined aquifer range from 7.44 to 10.1 μg/L, with the analytical results flagged as near the detection limit in two of the

samples. Total chromium and Cr(VI) were detected in the first water-bearing unit in the RUM at 11.4 μg/L and 8.6 μg/L, respectively. Concentrations in samples collected from deeper water-bearing units were below the detection limits for both total chromium and Cr(VI). Neither total chromium nor Cr(VI) was detected in a post-installation groundwater sample collected from the first water-bearing unit in August 2011. Given the low levels detected during drilling, some variation in concentrations is not unexpected. Additional sampling will be needed to determine if low levels of Cr(VI) and total chromium are present in that location.

199-H3-9 (C7639, Well R1) – Total chromium concentrations in the unconfined aquifer range from 3.58 μg/L to 8.85 μg/L, with no discernible vertical pattern to the concentration distribution observed. The Cr(VI) concentrations in the unconfined aquifer range from below detection to 6 μg/L; however, the sample with a result of 6 μg/L was a duplicate to a sample that reported nondetectable concentrations. The higher result was reported for the unfiltered sample, which can have interference from the presence of color or chemicals in groundwater. The filtered sample result is considered more representative of conditions.

The groundwater sample collected from the first water-bearing unit of the RUM had total chromium and Cr(VI) detected at 319 μg/L and 287 μg/L, respectively. This sample represents borehole water during drilling and, therefore, the high contaminant concentrations may be caused by high turbidity values (599 NTU) present during sampling. A groundwater sample collected in August 2011 had Cr(VI) detected at 115 μg/L, which is likely to be more representative of aquifer conditions, because the sample was collected from a completed monitoring well. This result confirms the presence of high levels of Cr(VI) in the first water-bearing unit of the RUM.

Within the next two water-bearing units of the RUM, the total chromium concentrations in groundwater were significantly lower. Concentrations for the two lower water-bearing units were reported at approximately 4 μg/L. The results were flagged by the laboratory as being estimated values. The corresponding Cr(VI) concentrations in groundwater were below detection limits.

199-H3-10 (C7640, Well R2) – Total chromium concentrations in the unconfined aquifer range from 10.2 μg/L to 13.8 μg/L, with no discernible vertical concentration trends observed. The Cr(VI) concentrations in the unconfined aquifer range from 2 to 11 μg/L, with slightly lower concentrations distributed in deeper portions of the unconfined aquifer. Total chromium and Cr(VI) concentrations in groundwater samples from the first water-bearing unit of the RUM and two deeper water-bearing units were less than their respective detection limits. No post-installation groundwater samples have been collected from this well in 2011.

Key to understanding potential vertical stratification of Cr(VI) is an understanding of the underlying geology and evaluating the concentration patterns in relation to that geology. Localized variations in stratigraphy often result in different contaminant distribution trends. The geologic features and associated Cr(VI) concentrations are presented on Figures 4-84 through 4-89. The cross-section locations are presented on Figure 4-84.

Cross-section A to A' (Figure 4-85) transects the southern plume at 100-D. As presented in Tables 4-15 and 4-16, vertical Cr(VI) stratification in the unconfined aquifer appears to be present beneath 100-D where high concentrations of 70 percent sodium dichromate solution was handled during reactor operations. However, concentrations are not consistently increasing or decreasing with depth across 100-D/H. A trend toward some vertical stratification is most prominent in the unconfined aquifer at RI Well 199-D3-5 (Table 4-15), not shown on the cross-section. At Well 199-D3-5, Cr(VI) concentrations increase with depth to the surface of the RUM.

At Well 199-D5-141 (C7625, Well R5), high concentrations of Cr(VI) are present in the unconfined aquifer; however, a Cr(VI) stratification trend was not observed. The Cr(VI) concentrations are elevated at several mid-level depths within the aquifer. Concentrations decline to 986 µg/L at 32.46 m (106.5 ft), but rise to 2,590 µg/L at the RUM surface, where a depression exists. As shown on cross-section A to A', the Cr(VI) concentrations from the first water-bearing unit in the RUM and the lower water-bearing units were below detection limits. Therefore, Cr(VI) is limited to the unconfined aquifer in this location.

As shown in cross-section B to B' (Figure 4-86) and Table 4-15, vertical Cr(VI) stratification in the unconfined aquifer is not apparent in the 100-D northern plume. An area of higher Cr(VI) concentrations is observed in the unconfined aquifer at Wells 199-D5-134 (C7624, Well R4), 199-D5-126, 199-D5-143 (C8375, Well 9 redrill), and 199-D5-140 (C7866, Well 9). A slight depression in the RUM is also present in this area and appears to extend farther toward the river than at cross-section A to A', with a gentle topographical rise in the RUM surface that appears to impede contaminant transport in this location.

RI Well 199-D5-134 was drilled into the lower water-bearing units of the RUM to evaluate the vertical extent of Cr(VI) beneath the northern plume. A concentration of 12.2 µg/L Cr(VI) was detected during drilling from the first water-bearing unit in the RUM. The well was screened across this stratigraphic unit during completion and a laboratory result from a post-installation sample, collected on January 30, 2012, was below the detection limits. In addition, samples collected from the lower water-bearing units did not have Cr(VI) detected.

At cross-section C to C', which is located in 100-D parallel to the river, the variation in the RUM surface is apparent (Figure 4-87). However, few vertical profile samples have been collected. Existing sampling results along this cross-section do not indicate stratification in Cr(VI) with depth.

Cross-section D to D' (Figure 4-88) provides a transect from 100-D through the horn area to100-H. Based on both historical and recent vertical sampling in the unconfined aquifer, Cr(VI) stratification is limited across the horn. Concentrations of Cr(VI) along this cross-section are generally below 100 µg/L, except at 100-D, with many wells having concentrations below the DWS of 48 µg/L.

Cr(VI) has been detected in the first water-bearing unit of the RUM in wells located in the horn at concentrations below the DWS. The presence of Cr(VI) in the RUM in this area is likely the result of the high hydraulic head conditions during reactor operations at 100-D forcing contaminants and water into the RUM, considered an aquitard, which is by definition able to transmit limited amounts of water between geologic units. To the east of Well 699-97-48C, Cr(VI) concentrations within the RUM diminish to below detection. This indicates that the influence of 100-D operations did not extend to 100-H within the RUM.

Cross-section E to E' (Figure 4-89) runs parallel to the Columbia River along 100-H. As shown on the cross-section, the unconfined aquifer in this area is thinner than at 100-D. There is minimal Cr(VI) stratification at 100-H, and Cr(VI) concentrations in the unconfined aquifer are generally below the DWS. Within the RUM, Cr(VI) has been identified in the first water-bearing unit but not in the lower water-bearing units. Low levels of Cr(VI) were detected in Well 199-H2-1, with concentrations during drilling of 3.7 µg/L (flagged as below detection limit), 5.9 µg/L and 8.6 µg/L at depths of 9.17, 10.64, and 19.17 m (30.1, 34.9, and 62.9 ft) bgs, respectively. Following completion, an analytical sample collected on August 17, 2011, was below the detection limits. Well 199-H2-1 delineates the northern edge of the Cr(VI) plume within the RUM. Concentrations farther south were detected at levels up to 287 µg/L (during drilling) in Well 199-H3-9. The high concentrations near the river are likely related to reactor operations, with the high head conditions associated with the nearby 116-H-7 Retention Basin overcoming an upward hydraulic gradient of the confined aquifer and forcing contaminated water into the first water-bearing unit of the RUM. These higher Cr(VI) concentrations were not detected in the RUM to the west of an apparent ridgeline trending parallel to the river, which is located slightly west of the

retention basins. Concentrations in Well 199-H3-2C, west of the "ridge," are typically between 50 and 80 μg/L. The extent of contamination in the first water-bearing unit of the RUM is not delineated to the south, but appears to follow the area of downward RUM surface slope along the river.

Deeper water bearing units, such as those within the Ringold Formation unit B, the Ringold Formation lower mud, and the basalt units are also presented in cross sections A-A', B-B', D-D', and E-E', Concentrations in these lower units are consistently below 10 μg/L.

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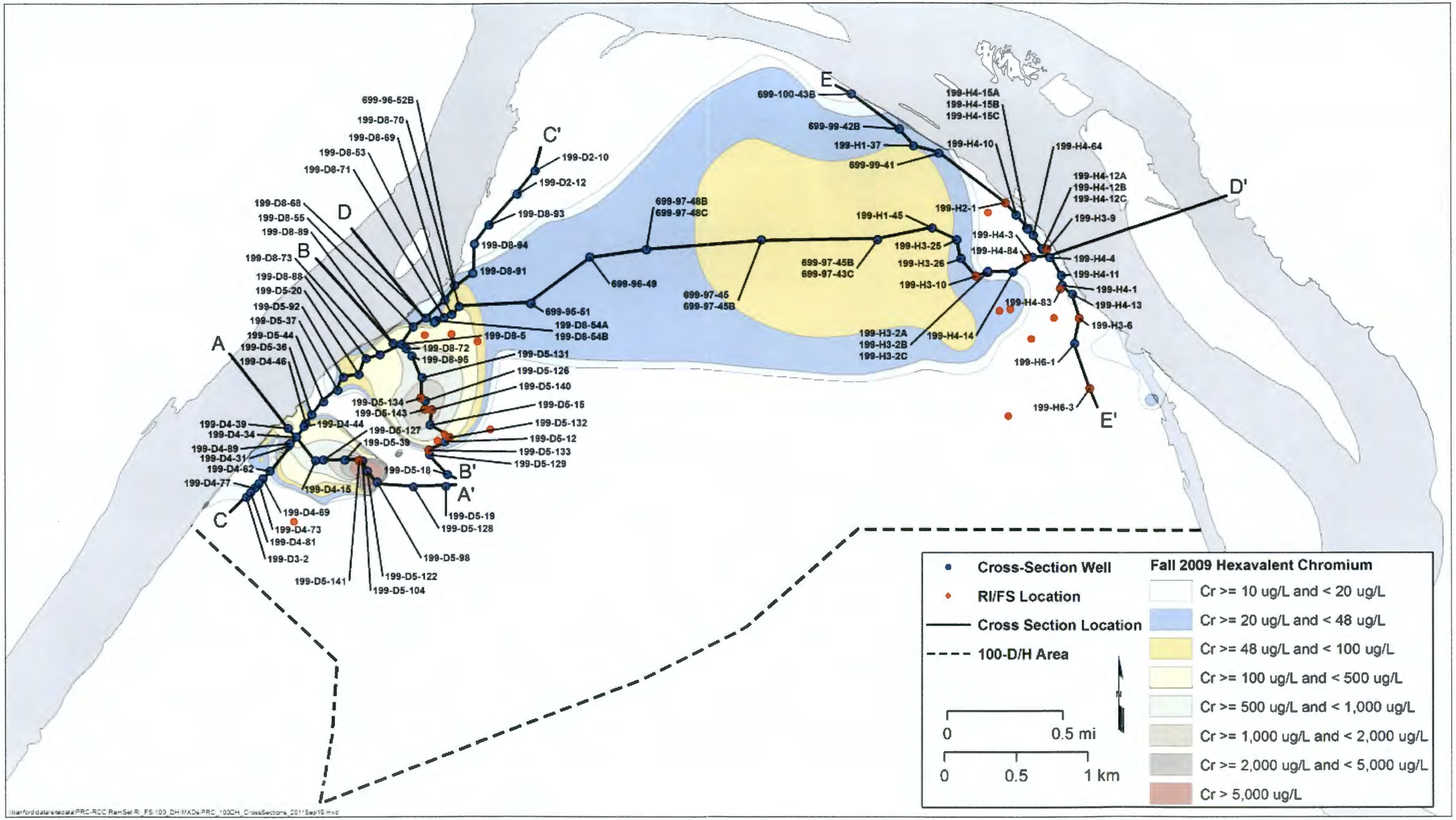

Figure 4-84 Trendline Location Map for 100-D/H Hydrogeologic Cross Sections and RI Wells

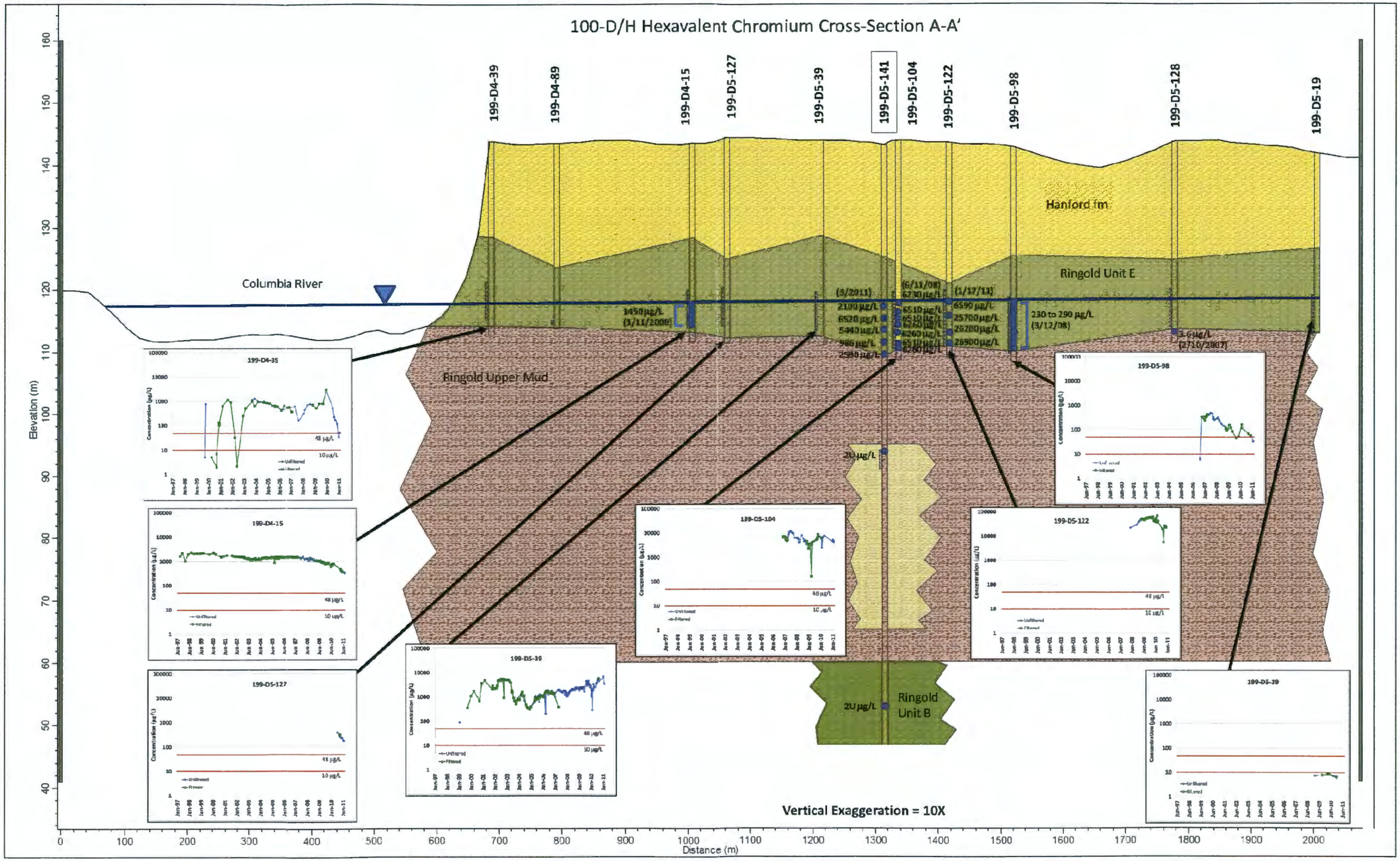

Figure 4-85. Vertical Distribution of Cr(VI) Concentrations in Groundwater shown on Cross-Section A-A'

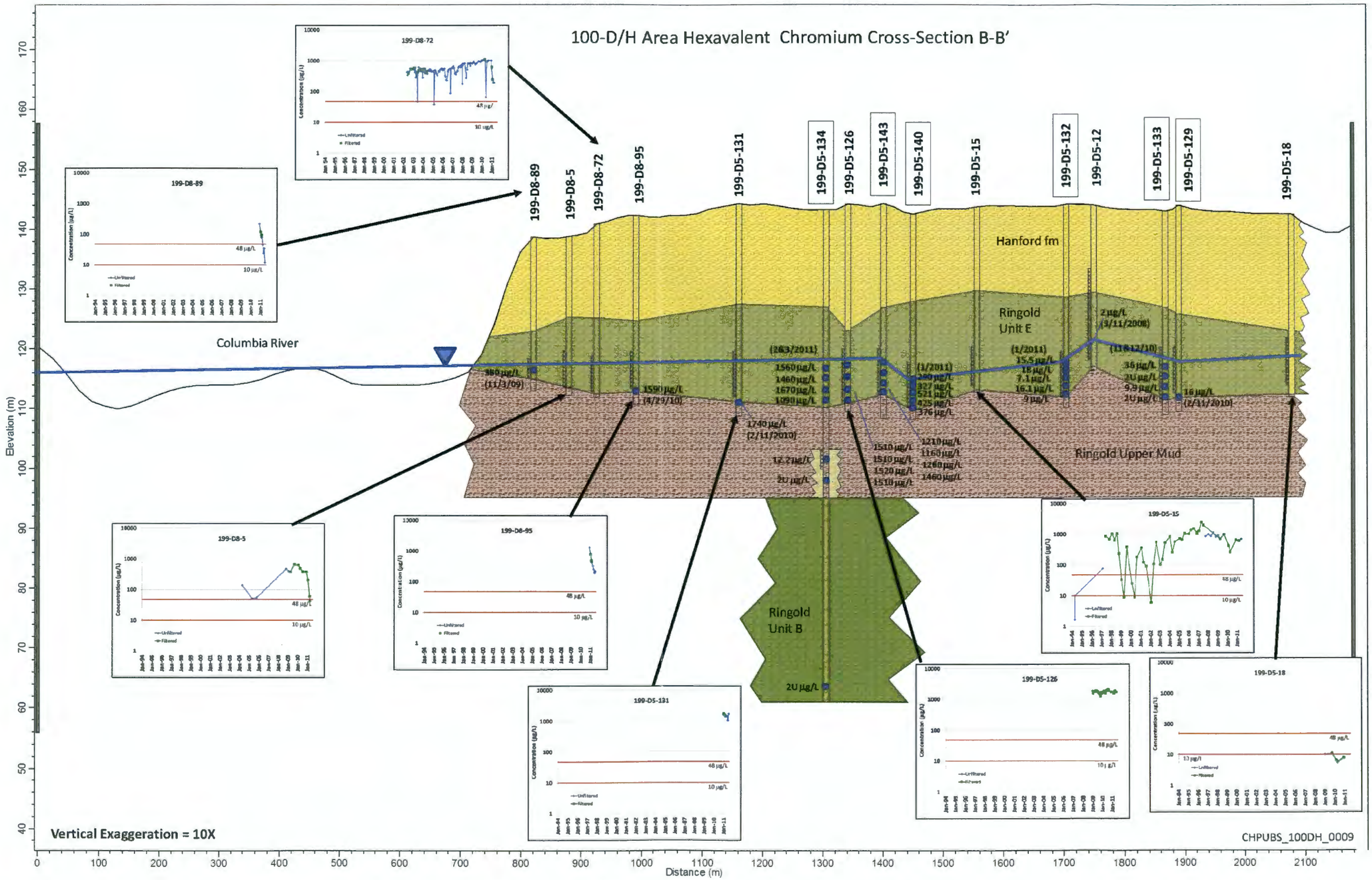

Figure 4-86. Vertical Distribution of Cr(VI) Concentrations in Groundwater shown on Cross-Section B-B'

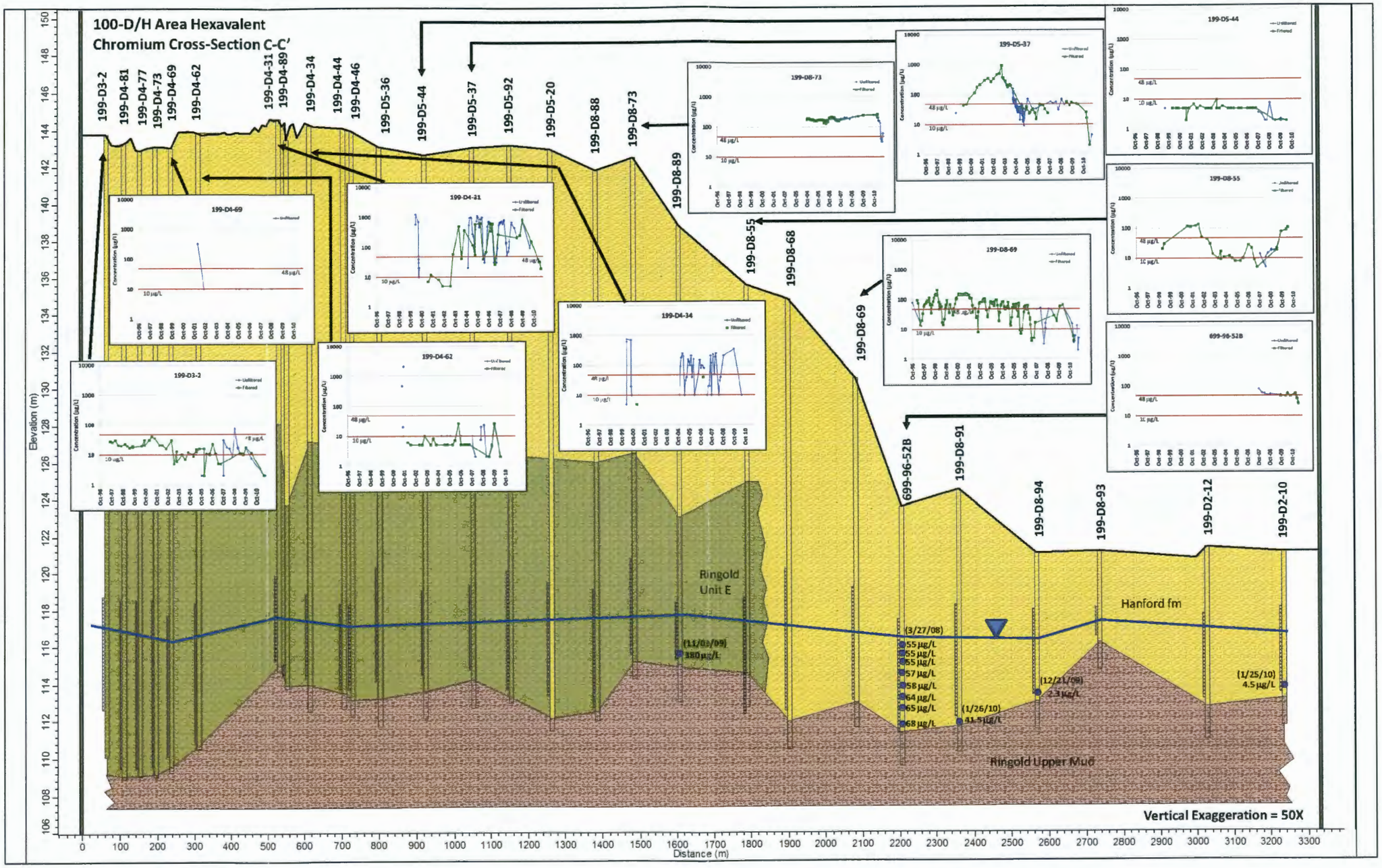

Figure 4-87. Vertical Distribution of Cr(VI) Concentrations in Groundwater shown on Cross-Section C-C'

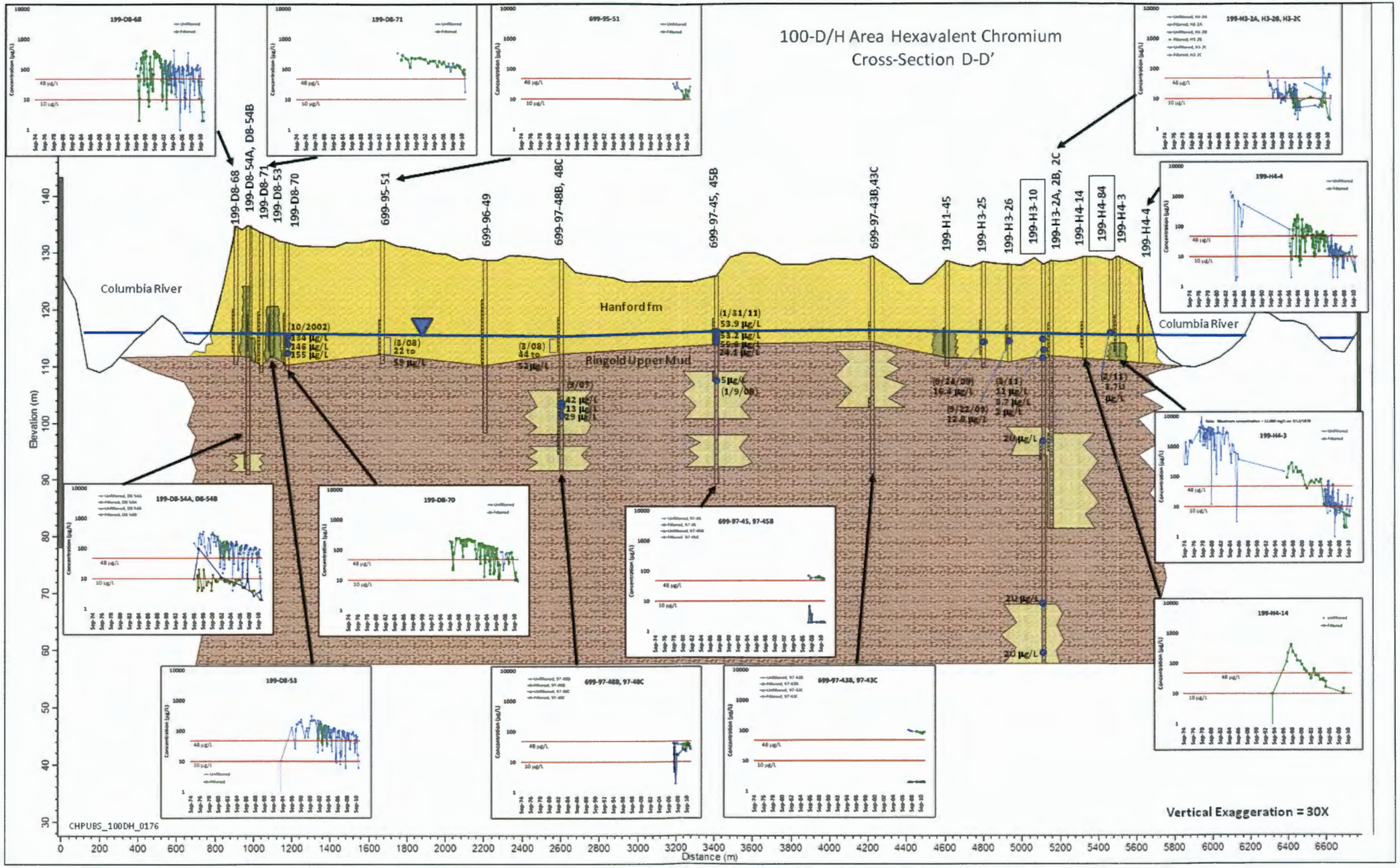

Figure 4-88. Vertical Distribution of Cr(VI) Concentrations in Groundwater shown on Cross-Section D-D'

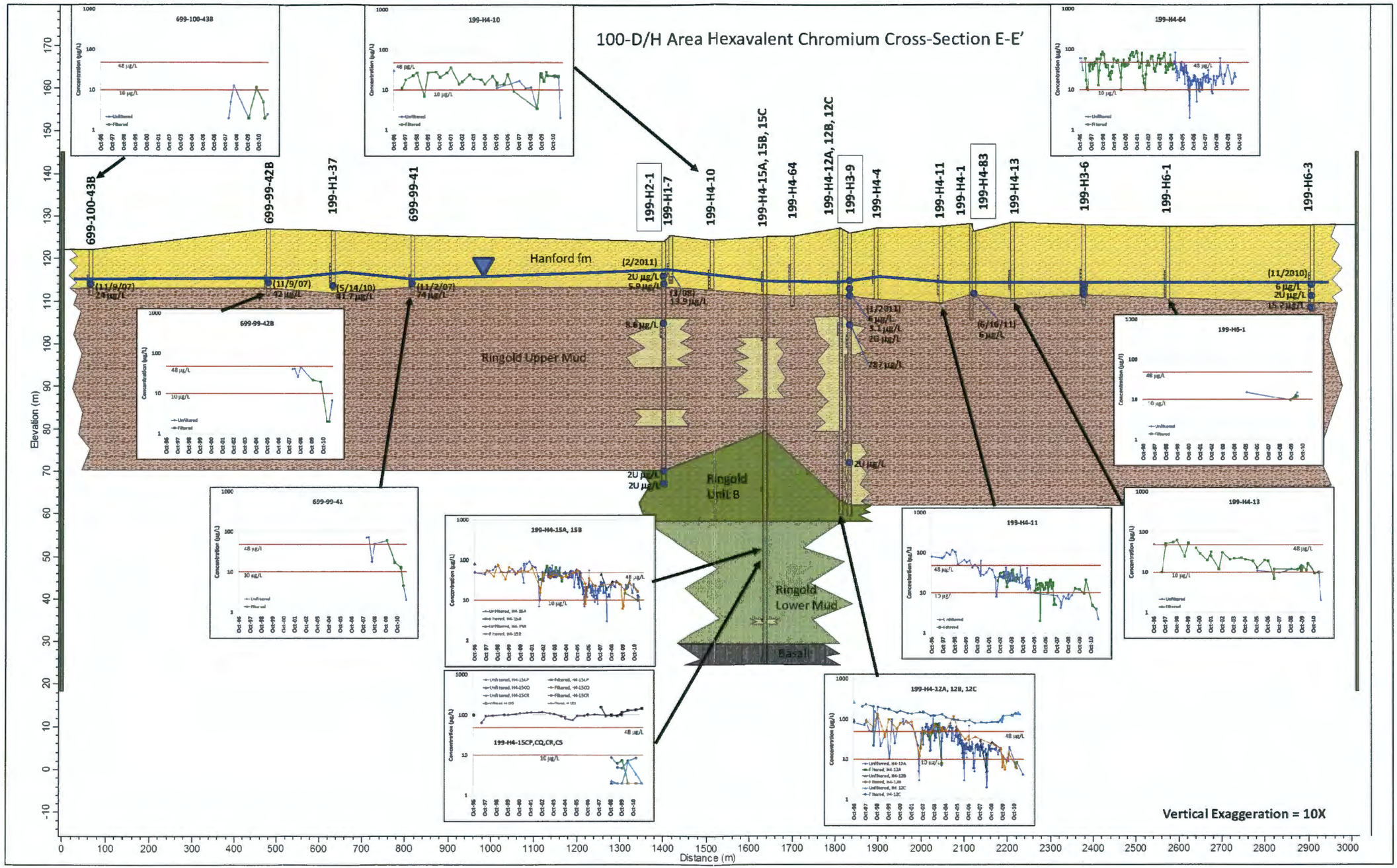

Figure 4-89. Vertical Distribution of Cr(VI) Concentrations in Groundwater shown on Cross-Section E-E'

4.5.2 Nitrate

Nitrate may be expressed as nitrate (NO_3) or as nitrate-nitrogen (NO_3-N). The DWSs for NO_3-N and NO_3 are 10,000 and 45,000 µg/L, respectively. Nitrate is present in the unconfined aquifer at concentrations exceeding the 45,000 µg/L DWS, primarily in 100-D (Figure 4-90). Nitrate concentrations in groundwater in the horn are below the DWS, and a small area in 100-H exceeds the DWS. The total footprint of the 100-D/H plume areas exceeding the DWS is approximately 1.5 km^2 (0.58 mi^2). Nitrate has not been detected in the water-bearing units of the RUM. Aquifer tube concentrations during 2011 were below the DWS in 100-D/H. The primary source of nitrate in 100-D/H is nitric acid used during reactor operations as a decontamination solution. Nitric acid and other decontamination solutions were disposed in cribs, trenches, and French drains near the building where they were used. These solutions were also occasionally combined with reactor cooling water and discharged to the river (*100-D Area Technical Baseline Report* [WHC-SD-EN-TI-181]). Another reactor operation related source is from oxidation of ammonia discharged in the condensate solution. Secondary contributors include septic systems, sewer lines, and former agricultural practices.

100-D. Within the unconfined aquifer underlying 100-D, nitrate is present in two general areas, with most of the plume area slightly above the DWS. Nitrate concentrations in groundwater from the southern plume extraction wells (199-D5-39 and 199-D5-104) are currently stable near 45,000 µg/L. However, because of the startup of the DX pump-and-treat system in December 2010, previously identified trends in the remaining portion of the plume are no longer apparent. For example, Well 199-D4-15, which is located between two extraction wells, had stable values around 60,000 µg/L. Concentrations subsequently decreased to 44,300 µg/L in April 2011, the first monitoring event following the DX system startup. At the north end of the ISRM barrier, nitrate concentrations in Well 199-D5-36 increased sharply from 5,000 µg/L in 2010 to 46,500 µg/L in August 2011. Nitrate concentrations in groundwater from other areas of the southern portion of the plume are also fluctuating, with concentrations actually increasing in some locations (199-D5-17) in response to changes in the groundwater flow regime.

Figure 4-91 shows seasonal variation in nitrate concentrations in the unconfined aquifer (100-D southern plume wells) during fall and spring 2011. Nitrate concentrations in groundwater were greater in fall 2009 than in spring 2010 in 20 wells (77 percent). Since the startup of the DX pump-and-treat system, the seasonal variation patterns are not as consistent.

Groundwater from the northern plume extraction wells (199-D5-20, 199-D8-53, 199-D8-68, and 199-D8-54A) have nitrate concentrations that are generally stable or increasing slightly and often exhibit seasonal variation (with lowest concentrations in the spring). Nitrate concentrations in the unconfined aquifer exceed the DWS in fall sampling rounds in Well 199-D8-72, but are often below the DWS in spring rounds. A sample collected from 199-D8-72 in March 2011 had a result below the DWS, at the lowest concentration observed in this well (21,800 µg/L).

Concentrations of nitrate in northern plume monitoring wells 199-D5-14, 199-D5-13, and 199-D8-4 are above the DWS, but are either stable or decreasing. With the exception of samples collected during spring, most wells in the northern plume have concentrations above 45,000 µg/L. The effect of the DX pump-and-treat system in the northern plume has not resulted in dramatic changes in nitrate concentrations in most locations, with a few exceptions. In October 2011, a dramatic decrease in concentration was observed in the groundwater sample from Well 199-D8-5. Overall, the plume appears to have migrated farther north than previously delineated. This is likely a result of the extraction wells to the north, which are focused on Cr(VI) removal.

Of the RI wells installed at 100-D, nine were sampled for nitrate as required in the SAP. Groundwater samples from all nine wells had nitrate concentrations above the 45,000 µg/L DWS at various depths.

Within the unconfined aquifer, maximum concentrations ranged from 51,400 µg/L at Well 199-D5-141 (C7625, Well R5) to 81,000 µg/L at Well 199-D5-133 (C7621, Well 3). Monitoring Well 199-D5-141 also had the lowest nitrate concentration detected in the RI wells, with a result of 28,400 µg/L detected at a depth of 34.1 m (112 ft). Figure 4-92 shows maximum nitrate concentrations in the RI wells at 100-D/H, and Table 4-17 presents the analytical results from samples collected at discrete depths during drilling. The four RI aquifer tubes at 100-D (C7645, C7646, C7647, and C7648) are located southwest of the ISRM Barrier and had nitrate concentrations less than 10,000 µg/L, well below the 45,000 µg/L DWS.

Monitoring wells screened in the first water-bearing unit of the RUM include one previously existing well and two RI wells: Wells 199-D8-54B, 199-D5-134 (C7624, Well R4), and 199-D5-141 (C7625, Well R5). Concentrations in groundwater samples from the first water-bearing unit in the RUM were below 10,000 µg/L, well below the DWS, in each location.

In summary, nitrate concentrations in the unconfined aquifer exceed the DWS of 45,000 µg/L, primarily within two areas of 100-D. These areas are located near the reactors and generally coincide with the 100-D north and south Cr(VI) plumes, indicating they are derived from nitric acid and septic sources associated with 105-D and 105-DR operations, as presented in Chapter 1 and shown on Figure 1-20. Nitrate was detected in groundwater at very low concentrations (much less than DWS) in the first water-bearing unit of the RUM.

Horn Area. Underlying the horn, nitrate concentrations in the unconfined aquifer are well below the DWS of 45,000 µg/L. Figure 4-93 shows seasonal variation in nitrate concentrations in the unconfined aquifer of the horn during fall and spring 2011. However, most wells in that area were only sampled once in 2011. Nitrate concentrations in groundwater samples from wells screened in the first water-bearing unit of the RUM (699-97-43C, 699-97-45B, and 699-97-48C) are well below the DWS.

100-H. Figure 4-90 shows the nitrate plume in the unconfined aquifer at 100-D/H. Nitrate concentrations in the unconfined aquifer above the DWS of 45,000 µg/L are found in isolated areas at 100-H.

- Groundwater samples from Well 199-H4-3 have a history of elevated nitrate concentrations, with a maximum concentration exceeding 3,000,000 µg/L in 1986. Concentrations were consistently below the DWS in 2008 through 2010, but increased to 72,200 µg/L in October 2011. A seasonal fluctuation is indicated in this location.

- Well 199-H1-27, an extraction well located north of 100-H, had nitrate detected in a groundwater sample at 69,500 µg/L in December 2011. This was much higher than the 10,500 µg/L detected in a borehole groundwater characterization sample collected in 2010, and higher than in nearby wells. Because there have only been two sampling events at this well, no trend has been established.

- Well 199-H4-75, an extraction well located on the western (upgradient) boundary of 100-H, had nitrate detected at 167,000 µg/L in 2011. This was the highest concentration in the region. Much lower nitrate concentrations (less than 30,000 µg/L) were detected in the surrounding unconfined aquifer in wells. Concentrations of other anions in the groundwater sample from 199-H4-75 also were high (for example, sulfate at 359,000 µg/L).

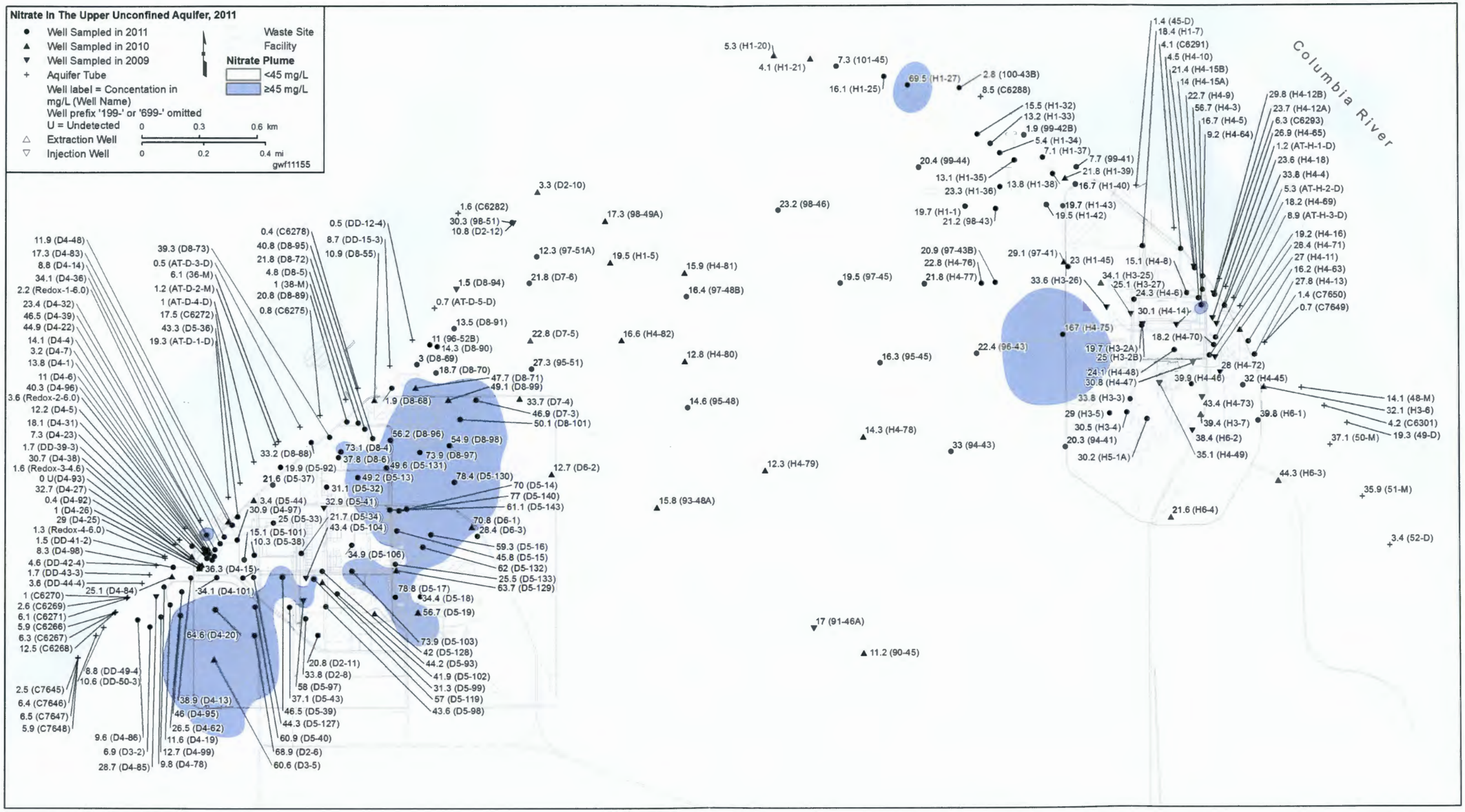

Figure 4-90. Nitrate Plume for 2011 at 100-D/H

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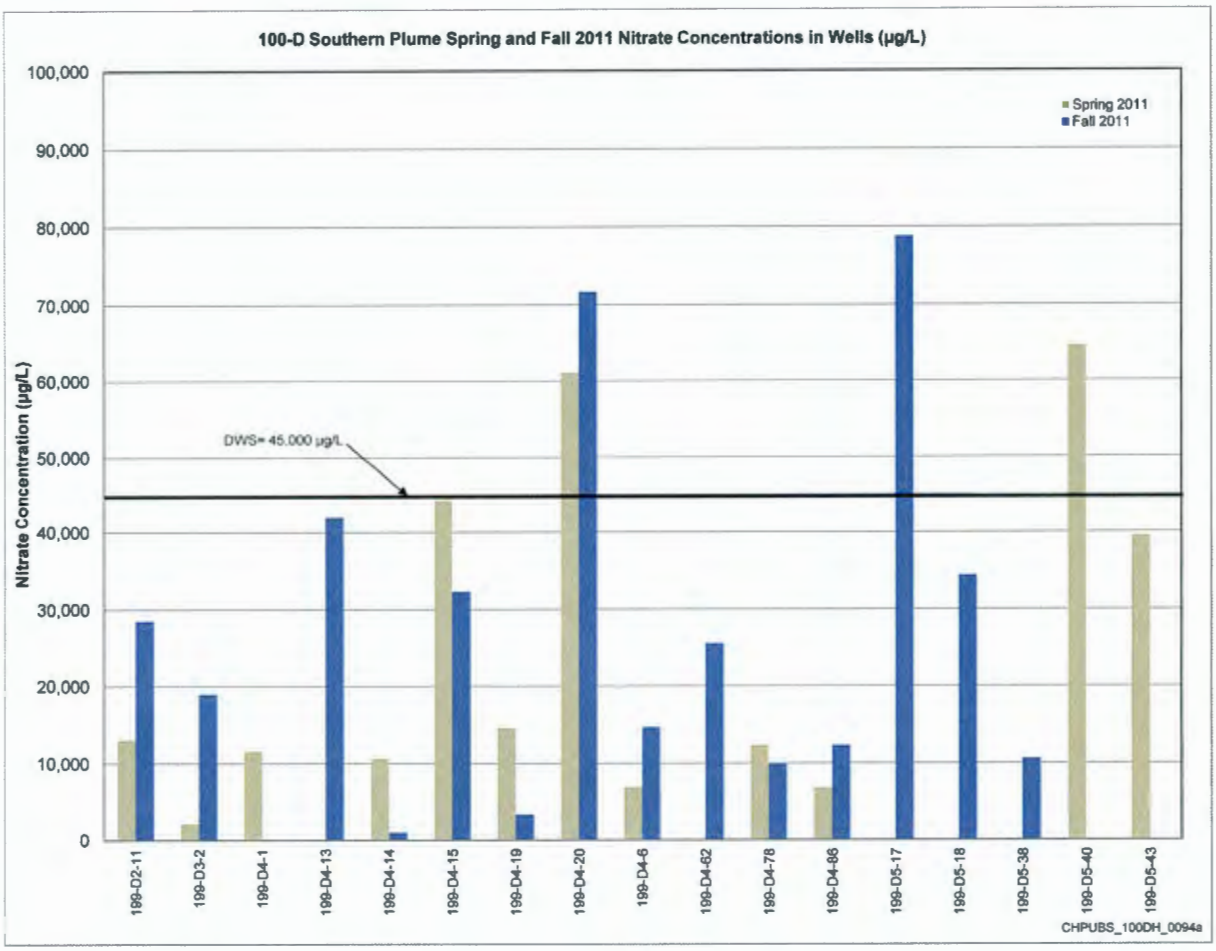

Figure 4-91. 100-D Area Southern Plume Nitrate Concentrations in Wells – Spring and Fall 2011

Groundwater samples collected during RI drilling activities show that nitrate concentrations did not vary significantly with depth in the unconfined aquifer. Nitrate concentrations in groundwater samples from the first water-bearing unit in the RUM are much lower (less than DWS) than in the unconfined aquifer. Figure 4-92 shows the maximum nitrate concentrations in groundwater samples collected from RI wells.

Nitrate concentrations in groundwater from 100-H Area aquifer tubes are generally low, excluding several aquifer tubes downstream from the operational area, where historical concentrations slightly exceeded the DWS. In February 2011, groundwater samples from aquifer tubes 50-M and 51-M had nitrate concentrations of 37,000 and 35,900 µg/L, respectively. A borehole groundwater sample from RI Well 199-H6-3 had 44,300 µg/L of nitrate detected during drilling. This well is located to the west of aquifer tube 51-M, indicating that the nitrate plume extends farther to the southwest than previously interpreted.

Figure 4-94 presents the fall and spring 2011 nitrate concentrations in the unconfined aquifer underlying 100-H. Some seasonal variation in concentrations is expected at 100-H; however, wells were not sampled for nitrate in timeframes adequate to show variation. In fall 2009, nitrate concentrations were greater than spring 2010 concentrations in 11 (58 percent) of the 100-H wells.

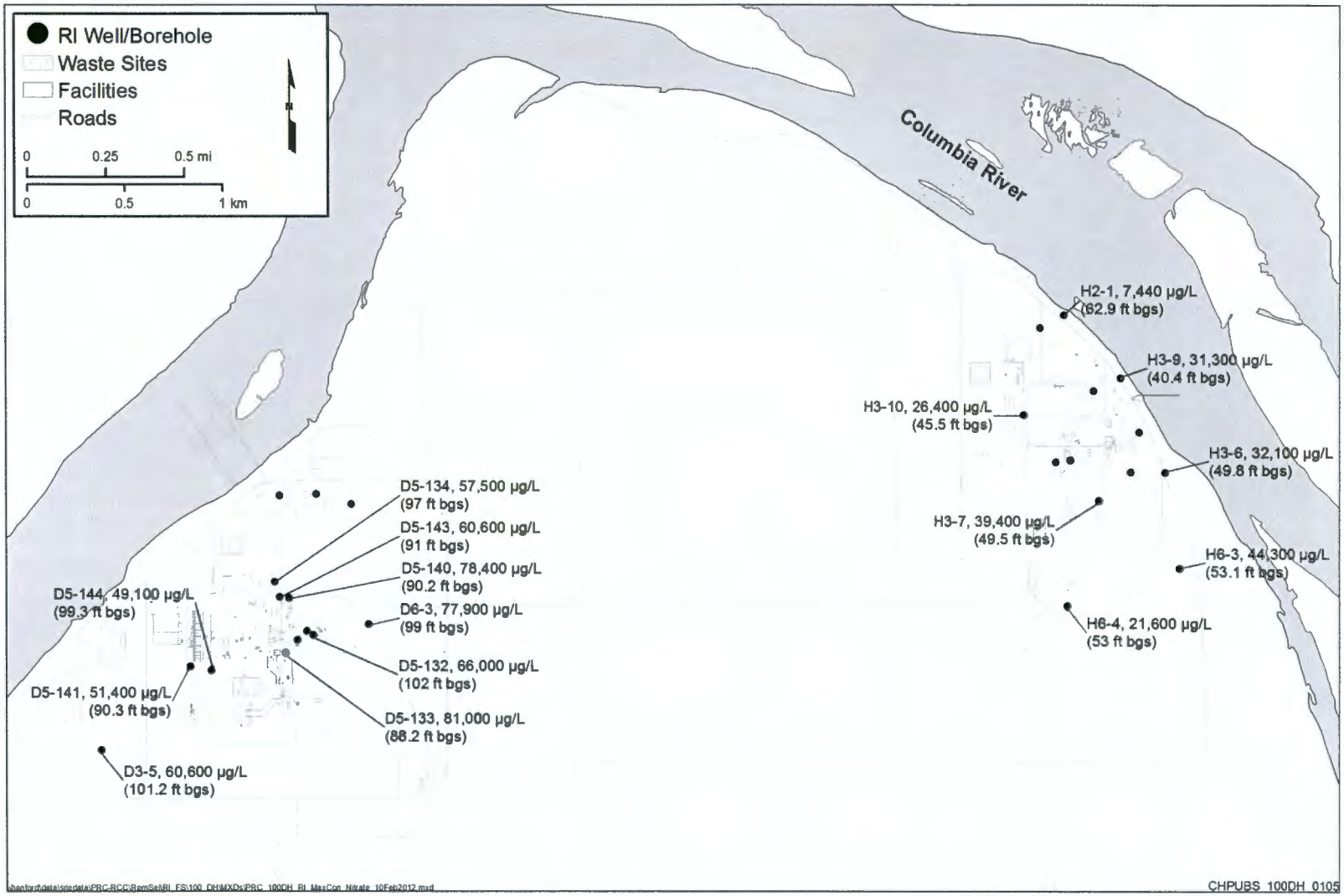

Figure 4-92. Maximum Nitrate Concentrations in Borehole Water Samples from RI Wells

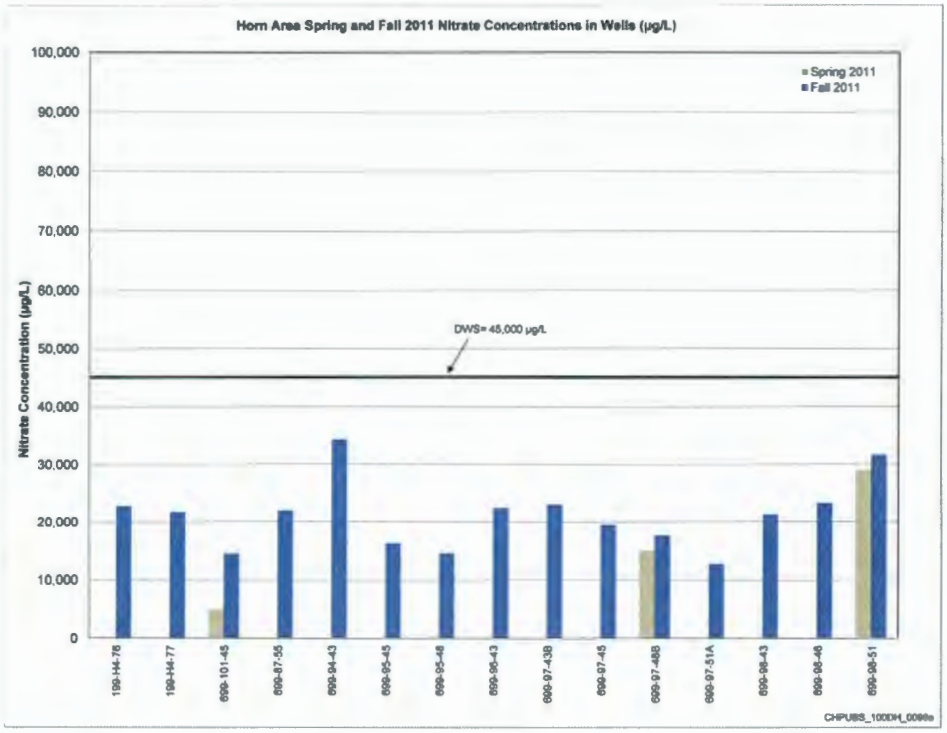

Figure 4-93. Horn Area Nitrate Concentrations in Monitoring Wells Fall and Spring 2011

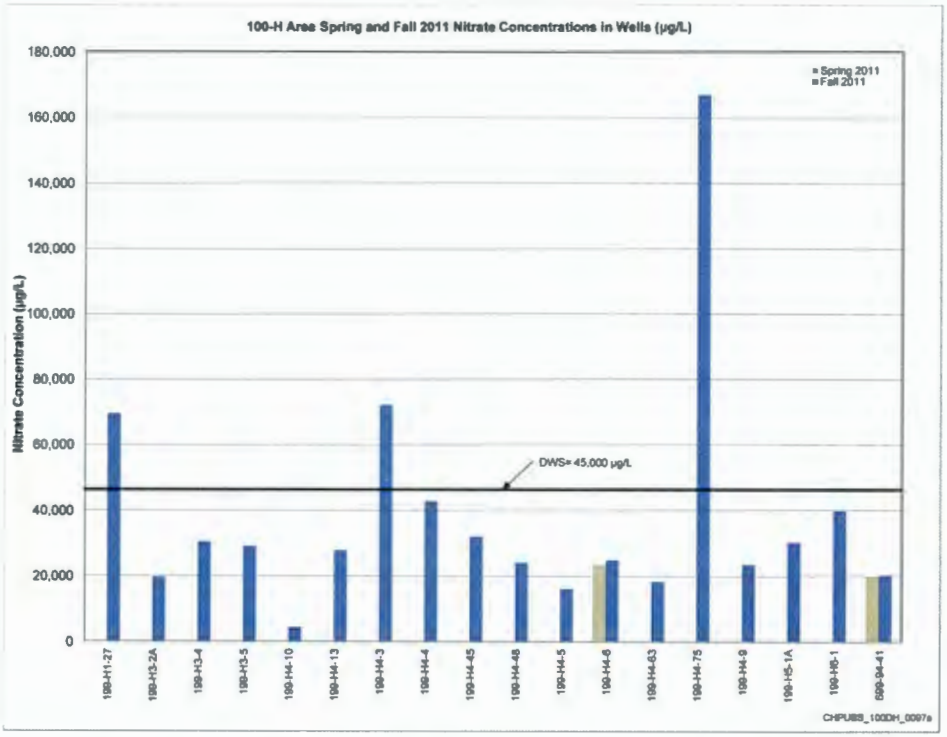

Figure 4-94. 100-H Area Nitrate Concentrations in Monitoring Wells Spring and Fall 2011

Table 4-17. Nitrate Sample Results from RI Drilling – 100-D

Area	Well Name (Borehole ID, SAP ID)	Sample Depth bgs m (ft)	Nitrate (µg/L)
100-D, Southern Plume: Unconfined Aquifer	199-D3-5 (C7620, Well 2)	27.5 to 28.1 (90.3 to 92.3)	39,200 (D)
		27.7 to 28.1 (90.9 to 92.3)	--
		29.3 to 29.7 (96 to 97.4)	47,800 (D)
		30.8 (101.2)	59,300 (D)
		30.8 (101.2)	60,600 (D)
		31.4 (103)	59,300 (D)
	199-D5-144 (C8668, Well R5 redrill)	28.01 (91.90)	--
		28.01 (91.90)	42,100 (D)
		29.11 (95.51)	--
		29.11 (95.51)	41,600 (D)
		30.27 (99.31)	41,200 (D)
		31.49 to 31.64 (103.3 to 103.8)	49,100 (D)
		31.49 to 31.64 (103.3 to 103.8)	47,800 (D)
		32.61 (107)	47,400 (D, N)
100-D, Southern Plume: RUM Wells	199-D5-141 (C7625, Well R5)	27.5 (90.3)	--
		27.5 (90.3)	51,400 (D)
		29.1 (95.5)	43,000 (D)
		29.1 (95.5)	42,500 (D)
		30.6 (100.5)	40,800 (D)
		32.5 (106.5)	29,400 (D)
		34.1 (112)	28,400 (D)
		49.5 (162.5)	2,090 (D)
		94.1 (308.8)	168 (U, D)
100-D, Northern Plume: Unconfined Aquifer	199-D5-133 (C7621, Well 3)	26.9 (88.2)	--
		26.9 (88.2)	81,000 (D)
		28.3 (92.7)	68,600 (D)
		29.8 (97.8)	72,600(D)
		29.8 (97.8)	73,000 (D)
		31.4 (103)	74,800 (D)
	199-D5-132 (C7622, Well 4)	27 (88.7)	--
		27 - 28.1 (88.7 - 92.3)	62,900 (D)
		29.4 (96.4)	64,600 (D)
		29.4 (96.4)	63,300 (D)

Table 4-17. Nitrate Sample Results from RI Drilling – 100-D

Area	Well Name (Borehole ID, SAP ID)	Sample Depth bgs m (ft)	Nitrate (µg/L)
100-D, Northern Plume: Unconfined Aquifer		31.1 (102)	66,000 (D)
		32. (105)	65,500 (D)
	199-D6-3 (C7623, Well 5)	28.7 (94)	--
		28.7 (94)	77,900 (D)
		30.2 (99)	77,900 (D)
		30.2 (99)	77,000 (D)
		30.9 (101.5)	77,000 (D)
	199-D5-140 (C7866, Well 9)	27.5 (90.2)	78,400 (D)
		27.5 - 27.8 (90.2 - 91.3)	--
		28.7 (94.2)	75,700 (D)
		28.7 (94.2)	76,600 (D)
		30.2 (99)	73,500 (D)
		31.5 (103.3)	74,400 (D)
	199-D5-143 (C8375, Well 9 redrill)	27.7 (91)	60,600 (D)
		27.7 (91)	--
		29. (95)	58,900 (D)
		29. (95)	57,500 (D)
		31.2 (102.5)	57,100 (D)
		31.7 (104)	54,900 (D)
	199-D8-101 (C7852, Waste Site 116-DR-1&2)	21.9 (72)	66,400 (D)
		21.9 (72)	--
	199-D5-142 (C7857, Waste Site 118-D-6)	26.5 - 27.4 (87.1 - 89.8)	--
100-D, Northern Plume: Unconfined Aquifer Boreholes		26.5 - 27.4 (87.1 - 89.8)	--
	Borehole C7850 (Waste Site 116-DR-9)	21.5 (70.4)	--
		21.5 (70.4)	--
	Borehole C7851 (Waste Site 116-D-7)	20.3 - 21 (66.7 - 69)	--
		20.3 - 21 (66.7 - 69)	--
	Borehole C7855 (Waste Site 116-D-1B)	27.8 (91.2)	--
		27.8 (91.2)	--
Northern Plume: PUM	199-D5-134 (C7624, Well R4)	28 (92)	--
		28 (92)	56,700 (D)

Table 4-17. Nitrate Sample Results from RI Drilling – 100-D

Area	Well Name (Borehole ID, SAP ID)	Sample Depth bgs m (ft)	Nitrate (µg/L)
		29.6 (97)	57,500 (D)
		29.6 (97)	56,700 (D)
		31.1 (102)	53.100 (D)
		32.7 (107.3)	54,000 (D)
		41.3 (135.5)	2,900 (D)
		46.9 (154)	2,310 (D)
		82 (268.9)	1,560 (D)

Notes:

Bold values exceed the DWS of 45,000 µg/L

Shaded cells indicate the sample was collected from a water-bearing unit below the RUM surface

"D" flag indicates that analysis was conducted at a secondary dilution factor.

"N" flag indicates that the spike sample recovery was outside of the control limits.

"—" indicates analyte was not sampled for at that location.

Nitrate concentrations in wells screened in the first water-bearing unit of the RUM (199-H3-2C, 199-H4-12C, and piezometer 199-H4-15CS) are below the DWS and exhibit stable trends. The piezometer nest consisting of 199-H4-15CP, 199-H4-15CQ, and 199-H4-15CR is screened in various lower Ringold Formation water-bearing units and the basalt aquifer. Groundwater samples from all three piezometers had nitrate concentrations less than 10,000 µg/L in 2011. Three RI wells were also screened in the first water-bearing unit of the RUM: Well 199-H2-1 (C7631, Well R3), Well 199-H3-9 (C7639, Well R1), and Well 199-H3-10 (C7640, Well R2). Nitrate concentrations from the RI wells were less than 10,000 µg/L in the RUM.

Of the 13 RI wells installed at 100-H, 9 were sampled for nitrate as required in the SAP. Groundwater samples from these 9 wells had nitrate concentration less than the 45,000 µg/L DWS. The concentrations in the unconfined aquifer range from 5,710 µg/L at Well 199-H2-1 to 44,300 µg/L at Well 199-H6-3. Analytical results collected at discrete depth intervals during drilling are presented in Table 4-18. The two RI aquifer tubes at 100-H (C7649 and C7650), located between the 116-H-7 Retention Basins and the river, had nitrate detected in groundwater at concentrations less than 10,000 µg/L. In summary, little nitrate remains in 100-H above the DWS, except in a few isolated wells.

Table 4-18. Nitrate Sample Results from RI Drilling – 100-H

Area	Well Name (Borehole ID, SAP ID)	Sample Depth bgs m (ft)	Nitrate (µg/L)
100-H: Unconfined Aquifer	199-H3-6 (C7626, Well 6)	15.2 (49.8)	32,100 (D)
		15.2 - 15.8 (49.8 - 51.8)	--
		16.4 (53.9)	31,300 (D)
	199-H3-7 (C7627, Well 7)	15.1 (49.5)	--
		15.1 (49.5)	39,400

Table 4-18. Nitrate Sample Results from RI Drilling – 100-H

Area	Well Name (Borehole ID, SAP ID)	Sample Depth bgs m (ft)	Nitrate (µg/L)
		15.7 (51.6)	38,200
		15.7 (51.6)	38,300
	199-H6-3 (C7628, Well 10)	14.8 (48.5)	44,100
		14.8 (48.5)	--
		16.2 (53.1)	44,300
		16.2 (53.1)	43,800
		19.5 (64)	41,800
	199-H6-4 (C7629, Well 11)	13.9 (45.7)	--
		13.9 (45.7)	20,100
		13.9 (45.7)	20,200
		14.6 (48)	20,400
		16.2 (53)	21,600
		18.4 (60.5)	21,300
	199-H1-7 (C7630, Well 12)	NS	18,400
		NS	--
100-H: RUM Wells	199-H2-1 (C7631, Well R3)	9.2 (30.1)	--
		9.2 (30.1)	5,930
		9.2 (30.1)	5,710
		10.64 (34.9)[a]	6,550
		19.2 (62.9)	7,440
		48.2 (158.3)	2,060
		54.7 (179.6)	2,230
	199-H3-9 (C7639, Well R1)	12.3 (40.4)	--
		12.3 (40.4)	31,300
		13.8 (45.2)	22,700
		13.8 (45.2)	22,500
		14.2 (46.5)	14,700
		20.8 (68.4)	5,580
		40.8 (134)	1,930
		53.9 (177)	3,000
	199-H3-10 (C7640, Well R2)	13.9 (45.5)	--
		13.9 (45.5)	26,400

Table 4-18. Nitrate Sample Results from RI Drilling – 100-H

Area	Well Name (Borehole ID, SAP ID)	Sample Depth bgs m (ft)	Nitrate (µg/L)
		15.2 (49.9)	25,000
		15.2 (49.9)	24,400
		16.1 (52.8)	23,900
		60.4 (198)	3,520
		68.2 (223.6)	1,850

Notes:

Bold values exceed the DWS of 45,000 µg/L

Shaded cells indicate the sample was collected from a water-bearing unit within the RUM.

"D" flag indicates that analysis was conducted at a secondary dilution factor.

"—" indicates analyte was not sampled for at that location.

NS = not specified

4.5.3 Strontium-90

The total footprint of the strontium-90 plume area exceeding the DWS of 8 pCi/L is approximately 0.2 km^2 (0.09 mi^2). Strontium-90 was produced as a fission product in reactor fuel during the plutonium production operations. Contamination of water by fission products occurred commonly in the reactor fuel storage basins; water was released from the fuel storage basins by unplanned releases (i.e., leaks and spills), as well as during planned releases (e.g., basin water overflow during reactor defueling operations). In addition, reactor cooling water became contaminated with fuel and fission product residues during reactor fuel failure incidents. During these incidents, the reactor cooling water was typically diverted from the normal discharge directly to the river to liquid disposal trenches for discharge of the contaminated cooling water directly to the vadose zone. Leaks from the cooling water retention basins, as well as the intentional discharges of contaminated cooling water to the disposal trenches, accounts for most of the observed strontium-90contamination in the vadose zone and groundwater. Figure 4-95 presents the strontium-90 plume at 100-D/H.

100-D. Contamination of soil and groundwater by strontium-90 at 100-D Area is apparently related to both planned and unplanned releases of fission product-contaminated wastewater to the environment. Overflow water from the fuel storage basins was routed to the 116-D-1A and 116-D-1B liquid waste trenches. Both of these trench areas exhibit full-thickness vadose zone contamination by strontium-90. Well 199-D-12, located just south of 116-D-1A liquid waste trench, historically exhibited strontium-90 in groundwater well above the DWS (i.e., up to 52.6 pCi/L) until it was decommissioned in 2002 (Figure 4-96). Well 199-D5-132, installed during the RI immediately adjacent to 116-D-1A Trench, exhibits strontium-90 in groundwater at about 44 pCi/L. Strontium-90 contamination observed in both of these wells is consistent with historical releases of contaminated wastewater to the trenches. Strontium-90 has been consistently below the minimum detectable activity (MDA) in all sampling events in nearby upgradient existing Well 199-D5-16. Well 199-D5-15, located 127 m (418 ft) downgradient of the 116-D-1A and 116-D-1B trenches, has exhibited variable Sr-90 concentrations in groundwater ranging from about 1 to 5 pCi/L since 1992, with the most recent samples exhibiting no detectable activity.

Other wells near 116-D-7 and 116-DR-9 Retention Basins and 116-DR-1&2 Trench have historically exhibited strontium-90 in groundwater. These retention basins and trench received the single-pass reactor

cooling water discharges from both 105-D and 105-DR reactors. Chronic leakage from the retention basins, as well as the intentional discharge of radiologically contaminated cooling water to the trench, resulted in the observed residual Sr-90 in groundwater in this vicinity. Monitoring Well 199-D8-68, located on the northern end of 100-D, had several DWS exceedances up until 2005, with no exceedances since that time.

Strontium-90 has not been detected in Well 199-D8-54B, which is completed in the first water-bearing unit of the RUM. Concentration trends in aquifer tubes (DD-15-2, DD-15-3, DD-15-4, DD-17-2, and DD-17-3) are variable, but all strontium-90 concentrations are below the DWS.

Of the 13 RI wells installed at 100-D, nine were sampled for strontium-90 as required in the SAP. Six of the nine wells had groundwater samples with concentrations less than the 8 pCi/L DWS. Wells 199-D3-5 (C7620, Well 2) and 199-D5-132 (C7622, Well 4) both had detections that exceeded the DWS. Well 199-D3-5 had a maximum activity of strontium-90 in the unconfined aquifer of 8.5 pCi/L, at a depth of 31.4 m (103 ft) bgs. An analytical result of 4.5 pCi/L was also reported in the same borehole at 30.8 m (101.2 ft) bgs, the sample interval but slightly shallower. However, the duplicate sample at that depth was below the MDA. In addition, the gross beta results do not correlate with either the 8.5 or 4.5 pCi/L results, and there were known laboratory issues with strontium-90 results during the analysis period, introducing uncertainty to those results.

Strontium-90 activity at Well 199-D5-132 (C7622, Well 4) had a maximum value of 65 pCi/L in a fine-grained interval at a depth of 29.6 m (96.4 ft) bgs. The aquifer matrix at that depth and the sample interval above it, which had a reported value of 45 pCi/L, had a higher percentage of silt than other sample intervals in that borehole. The higher silt percentage may have retarded movement of the strontium-90. Well 199-D5-132 was drilled just outside the footprint of the 116-D-1A Trench, indicating that 116-D-1A is the likely source of strontium-90 to groundwater. In addition, strontium-90 concentrations in the unconfined aquifer at Well 199-D5-132 (C7622, Well 4) are similar to the historical strontium-90 concentrations detected in Well 199-D5-12, which had concentrations of 30 pCi/L detected in 1999 before decommissioning. Results from soil borehole C7857 had also had low activities, with levels consistently below 2 pCi/g. However, the maximum value was encountered slightly deeper in borehole C7857. These factors indicate that the fuel storage basin is also a likely source of strontium-90 in groundwater in that area, but not conclusively.

Sr-90 was detected in only one of four samples of the hyporheic zone from the four RI aquifer tubes at 100-D (C7645, C7646, C7647, and C7648) located southwest of the ISRM barrier. Strontium-90 was detected in aquifer tube C7646 at 3.2 pCi/L. Table 4-19 presents analytical data for groundwater aquifer grab samples collected during drilling activities. Figure 4-97 shows maximum strontium-90 concentrations identified in the nine RI wells.

100-H. A plume of strontium-90 is present in the unconfined aquifer at 100-H, although the concentrations are at or less than the DWS over the majority of 100-H (Figure 4-95). Strontium-90 was likely released to the environment during historical reactor operations (i.e., to the vadose zone soil with subsequent migration to the underlying shallow unconfined aquifer) through planned and unintentional releases of contaminated water from the fuel storage basin and from releases of contaminated reactor cooling water to the 116-H-7 retention basin and the 116-H-1 Trench. Other waste sites that received radiologically contaminated liquids may also have contributed to the observed Sr-90 in soil and groundwater at 100-H. The Sr-90 plume has persisted because of the moderate to low mobility of strontium-90 in water, and its half-life of 28.79[1] years.

[1] Half-life from Radiochemistry Society website (RS, 2011) was accessed January 2012.

Strontium-90 concentrations observed in groundwater have generally declined over the past 20 years at 100-H; however, some locations continue to exhibit variable concentrations in excess of the 8 pCi/L. For example, extraction Well 199-H4-63, located midway between the 116-H-7 Retention Basin and the Columbia River, has exhibited a general downward concentration trend since 1996. Concentrations of Sr-90 in groundwater increased from 16 pCi/L to 31 pCi/L at this well during 2011 (inset Figure 4-98). The cause of this increase is not apparent and there may be several influences, including plume migration under the influence of the groundwater pump-and-treat system, seasonal transient effects of groundwater elevation, or the mobilization of strontium-90 under the effects of addition of water for dust control during remedial actions at nearby waste sites. Concentrations of strontium-90 in other nearby groundwater monitoring wells within the plume (e.g., 199-H4-11 and 199-H4-45) exhibit a similar trend, with general decreases in concentration since the early 1990s, and increases in the most recent year (inset Figure 4-98).

The possibility for strontium-90 concentrations to be affected by the pump-and-treat system operation became apparent in October 2009 when the pump-and-treat system was shut down for a Cr(VI) concentration rebound test at 100-H Area. Monitoring Wells 199-H3-2A, 199-H4-13, 199-H4-45, and extraction Well 199-H4-63 each exhibited a substantial increase in concentration in the period immediately following the system shutdown. Peak measured Sr-90 concentrations during these transients were 13 pCi/ L, 23 pCi/L, 35 pCi/L, and 110 pCi/L in Wells 199-H3-2A, 199-H4-63, 199-H4-45, and 199-H4-13, respectively (see inset graphs in Figure 4-98).

Strontium-90 exhibits variable mobility in the vadose zone and aquifer system within the Hanford 100 Area. At 100-K, Sr-90 exhibits a moderate degree of mobility, such that high-concentration plumes have migrated away from the apparent release points, and transient concentrations approaching the MCL-equivalent of 8 pCi/L have been observed in aquifer tubes monitoring the Columbia River hyporheic zone. Similar conditions are apparent at 100-H, where Sr-90 concentrations exceeding the 8 pCi/L have been observed in nearshore wells and in samples collected from aquifer tubes completed in the hyporheic zone (e.g., Aquifer Tubes 47-D and 47-M, located at the river shore near Well 199-H4-11). Concentrations in the unconfined aquifer at 100-H exceeded the Sr-90 DWS of 8 pCi/L in five groundwater samples in fall 2011, and in three of the spring 2011 groundwater samples (Figure 4-99). The areal extent of the inferred strontium-90 plume varies seasonally; the concentration variation likely results from seasonal contact with contaminated vadose zone soil during periods of high river stage, which generally corresponds with higher groundwater elevations in the near-river portions of 100-H.

Strontium-90 concentrations in aquifer tubes are variable with some locations exceeding the DWS. Most aquifer tubes have concentrations below the reporting limit. The exceptions are aquifer tubes 47-D, 47-M, and C7649, which have concentrations fluctuating around the 8 pCi/L DWS.

Of the RI wells installed at 100-H, seven were analyzed for strontium-90. Groundwater samples from six of seven wells had concentrations below the 8 pCi/L DWS. Strontium-90 was detected in Well 199-H3-6 at 8.2 pCi/L, slightly over the DWS. Figure 4-97 shows the maximum strontium-90 concentrations in the seven RI wells. Table 4-20 presents the strontium-90 results collected from discrete depth intervals during drilling. Results indicate a relatively localized area of strontium-90 that exceeds the DWS in the unconfined aquifer at 100-H, with no strontium-90 exceedances detected in lower water-bearing units in the Ringold Formation.

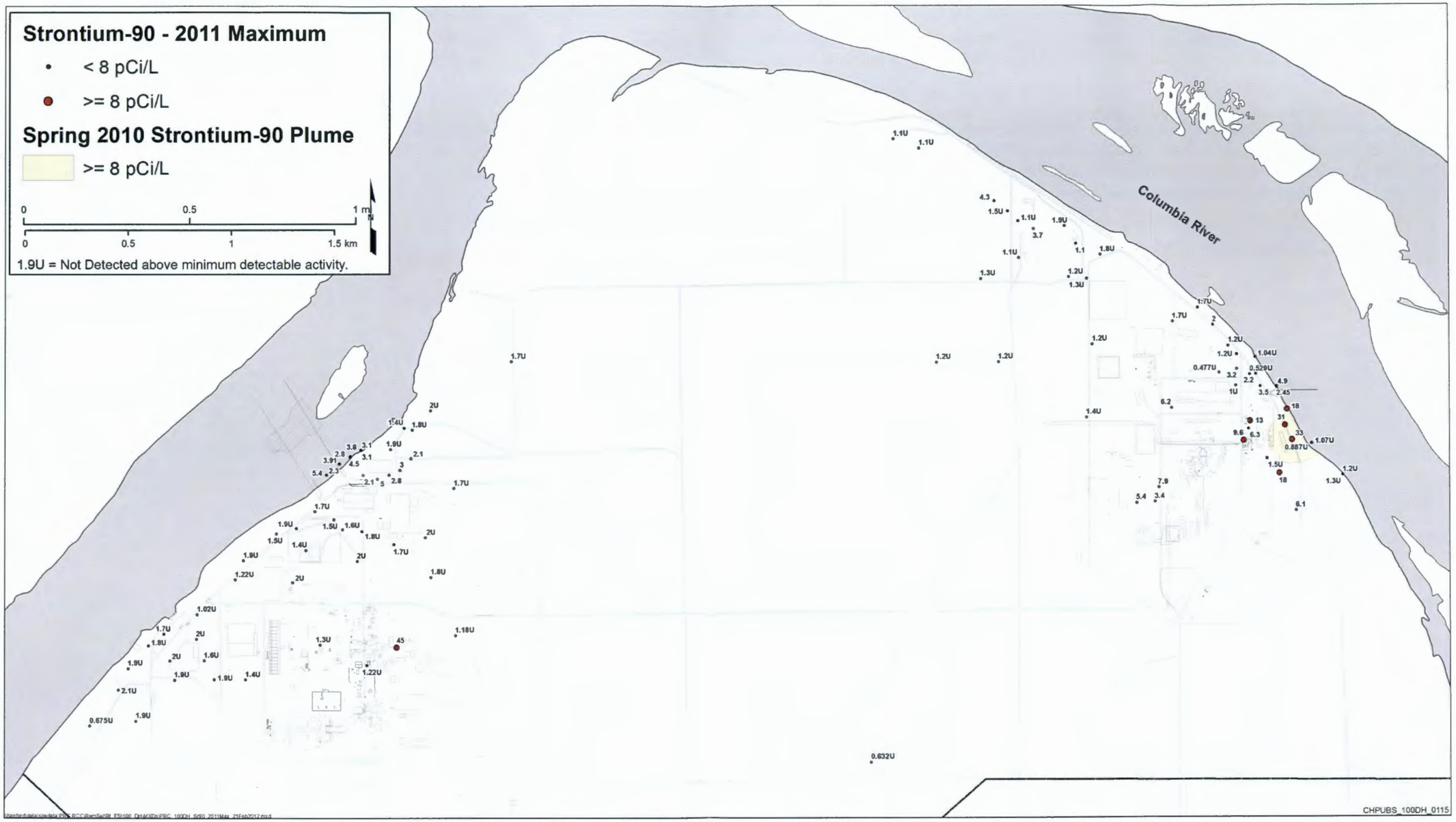

Figure 4-95. Spring 2010 Strontium-90 Plume and 2011 Maximum Strontium-90 Detections in the Unconfined Aquifer at 100-D/H

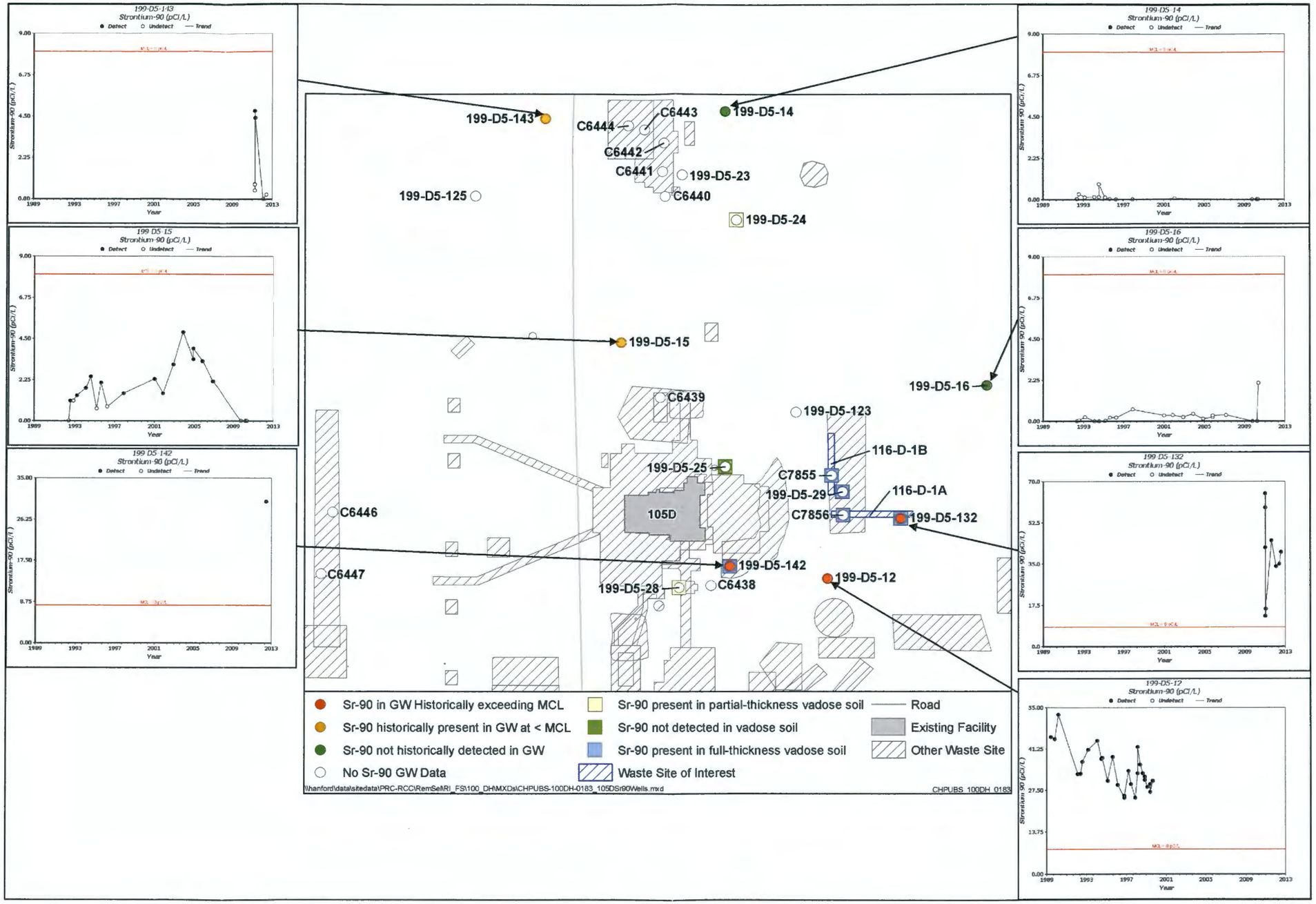

Figure 4-96. Strontium-90 at 100-D

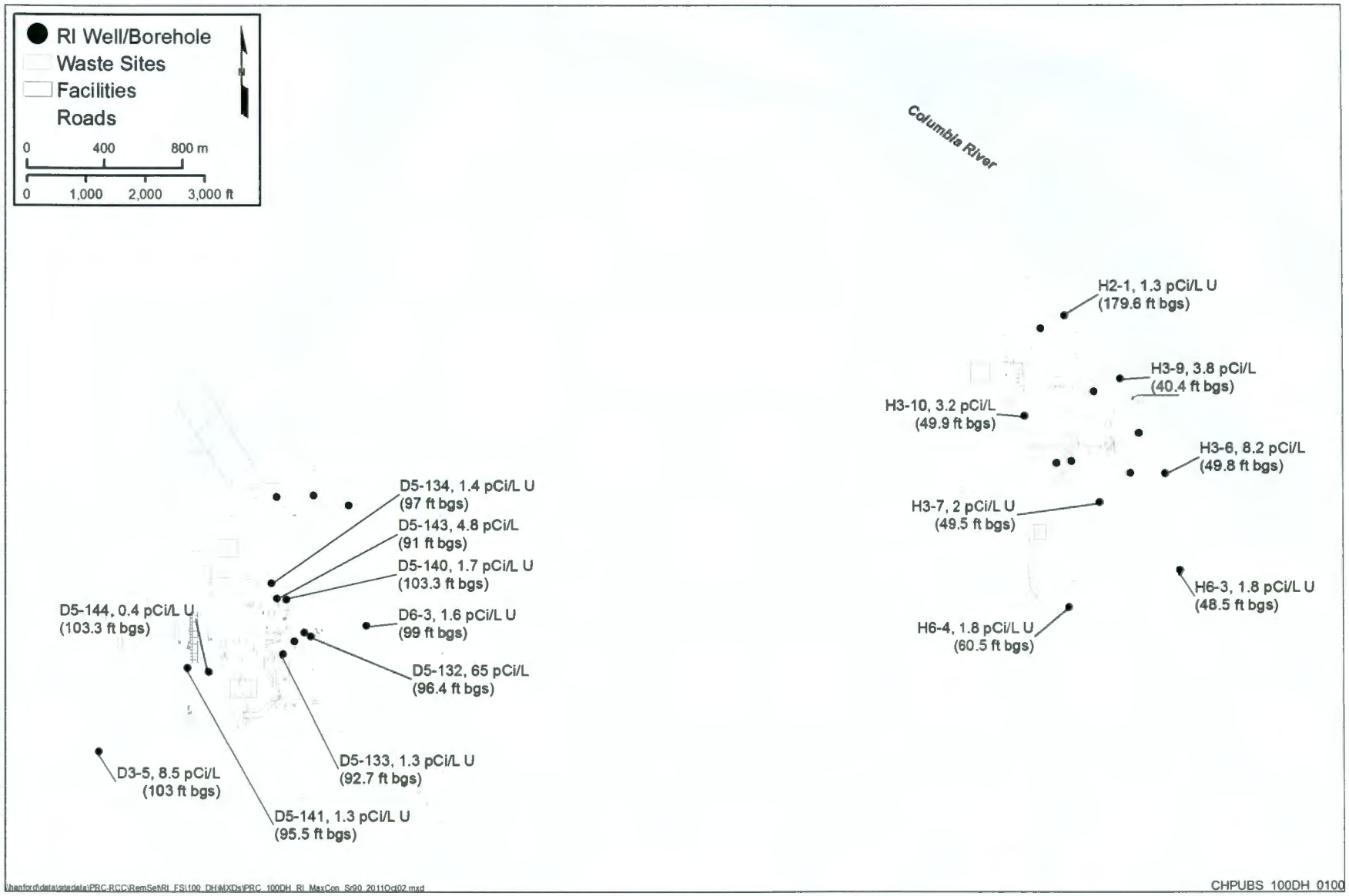

Figure 4-97. Maximum Strontium-90 Concentrations in Borehole Water Samples from RI Wells

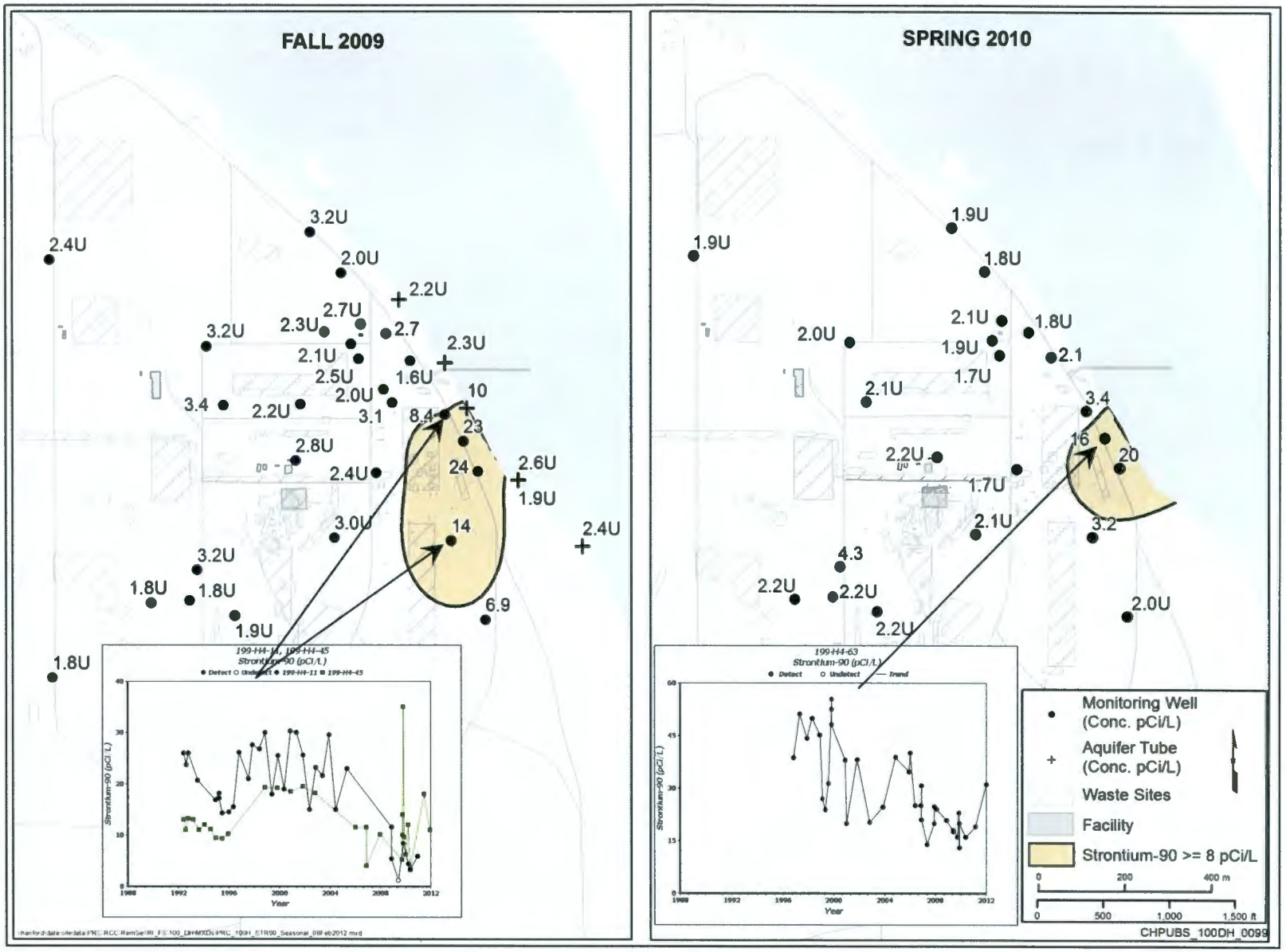

Figure 4-98. 100-H Fall and Spring Strontium-90 Plume in the Unconfined Aquifer

Table 4-19. Strontium-90 Sample Results from RI Drilling – 100-D

Area	Well Name (Borehole ID, SAP ID)	Sample Depth bgs m (ft)	Strontium-90 (pCi/L)
100-D, Southern Plume: Unconfined Aquifer	199-D3-5 (C7620, Well 2)	27.5 to 28.1 (90.3 to 92.3)	--
		27.7 to 28.1 (90.9 to 92.3)	2 (U)
		29.3 to 29.7 (96 to 97.4)	2.1 (U)
		30.8 (101.2)	4.5
		30.8 (101.2)	2.1 (U)
		31.4 (103)	8.5
	199-D5-144 (C8668, Well R5 redrill)	28.01 (91.90)	--
		28.01 (91.90)	0.466 (U)
		29.11 (95.51)	0.55 (U)
		29.11 (95.51)	0.575 (U)
		30.27 (99.31)	0.519 (U)
		31.49 to 31.64 (103.3 to 103.8)	--
		31.49 to 31.64 (103.3 to 103.8)	0.438 (U)
		32.61 (107)	0.456 (U)
100-D, Southern Plume: RUM Wells	199-D5-141 (C7625, Well R5)	27.5 (90.3)	
		27.5 (90.3)	1.6 (U)
		29.1 (95.5)	1.3 (U)
		29.1 (95.5)	1.5 (U)
		30.6 (100.5)	1.4 (U)
		32.5 (106.5)	1.7 (U)
		34.1 (112)	1.6 (U)
		49.5 (162.5)	1.7 (U)
		94.1 (308.8)	1.4 (U)
100-D, Northern Plume: Unconfined Aquifer	199-D5-133 (C7621, Well 3)	26.9 (88.2)	--
		26.9 (88.2)	1.3 (U)
		28.3 (92.7)	1.3 (U)
		29.8 (97.8)	1.6 (U)
		29.8 (97.8)	1.5 (U)
		31.4 (103)	1.4 (U)
	199-D5-132 (C7622, Well 4)	27 (88.7)	--
		27 - 28.1 (88.7 - 92.3)	42
		29.4 (96.4)	65
		29.4 (96.4)	59

Table 4-19. Strontium-90 Sample Results from RI Drilling – 100-D

Area	Well Name (Borehole ID, SAP ID)	Sample Depth bgs m (ft)	Strontium-90 (pCi/L)
		31.1 (102)	13
		32. (105)	16
	199-D6-3 (C7623, Well 5)	28.7 (94)	--
		28.7 (94)	1.8 (U)
		30.2 (99)	1.6 (U)
		30.2 (99)	1.7 (U)
		30.9 (101.5)	1.8 (U)
	199-D5-140 (C7866, Well 9)	27.5 (90.2)	--
		27.5 - 27.8 (90.2 - 91.3)	1.7 (U)
		28.7 (94.2)	1.9 (U)
		28.7 (94.2)	1.8 (U)
		30.2 (99)	1.8 (U)
		31.5 (103.3)	1.7 (U)
	199-D5-143 (C8375, Well 9 redrill)	27.7 (91)	--
		27.7 (91)	4.8
		29. (95)	1.6 (U)
		29. (95)	1.6 (U)
		31.2 (102.5)	1.6 (U)
		31.7 (104)	4.4
	199-D5-134 (C7624, Well R4)	28 (92)	--
		28 (92)	1.7 (U)
		29.6 (97)	1.7 (U)
		29.6 (97)	1.4 (U)
100-D, Northern Plume: RUM Well		31.1 (102)	1.5 (U)
		32.7 (107.3)	1.5 (U)
		41.3 (135.5)	1.6 (U)
		46.9 (154)	2 (U)
		82 (268.9)	1.7 (U)

Notes:

Bold values exceed the DWS of 8 pCi/L

Shaded cells indicate samples collected from a water-bearing unit below the RUM surface.

"U" flag indicates analyte was not detected above the minimum detectable activity (MDA) shown.

"—" indicates analyte was not analyzed for at that location.

4.5.4 Tritium

Historically, tritium was detected in the unconfined aquifer at concentrations greater than the DWS of 20,000 pCi/L in several wells at 100-D. Tritium is not present in the horn or 100-H above the DWS. By 1995, concentrations had decreased in most wells. Tritium concentrations in groundwater from Well 199-D5-17, located near 105-DR Reactor, displays typical trends for most wells (Figure 4-100) in 100-D. The primary sources of tritium are reactor operations at 105-D and 105-DR.

100-N has also contributed to the tritium now found in the unconfined aquifer underlying the southern portion of 100-D. As discussed in *Hanford Site Ground-Water Monitoring for 1993* (PNL-10082) a tritium plume was present at the 1325-N Crib (waste site 116-N-3). This plume later migrated to the northeast as shown in *Hanford Site Groundwater Monitoring for Fiscal Year 1997* (PNNL-11793, Plate 3). The remnants of this tritium plume are still identified by tritium concentrations in groundwater at or near 20,000 pCi/L in the southwestern portion of 100-D. Tritium has not been detected above the DWS of 20,000 pCi/L in wells/piezometers completed in the first water-bearing unit of the RUM. Figure 4-101 shows the tritium plume in the unconfined aquifer at 100-D/H in 2010, as well as the maximum values of tritium detected in 2011. Activities of tritium in the 2011 groundwater samples were below the DWS of 20,000 pCi/L.

In 2011, the maximum tritium concentration in the unconfined aquifer underlying 100-D was identified in Well 199-D8-89 at 24,000 pCi/L, from February 2011. This also represents the only location with concentrations over 20,000 pCi/L. The duplicate sample was reported at 180 pCi/L, which is more consistent with historical concentrations in that area. The groundwater sample result from January 2012 was 430 pCi/L, which indicates that the high value from early 2011 may be in error. The February 2011 result, which was flagged as suspect, is undergoing further review.

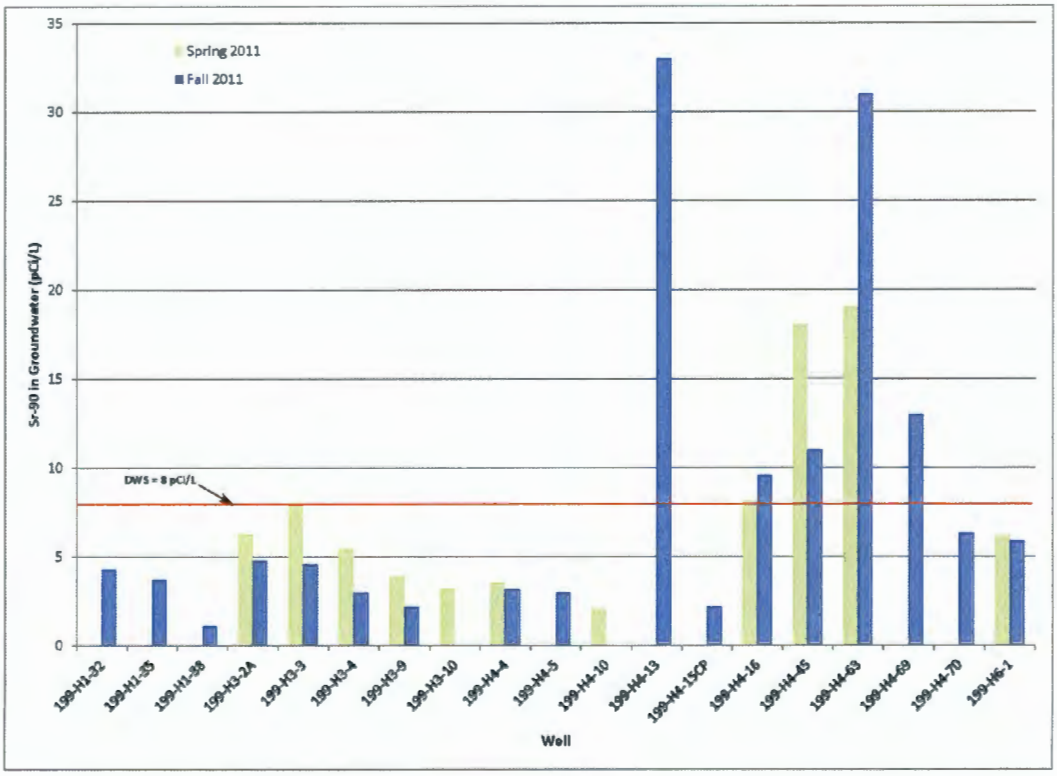

Figure 4-99. 100-H Area Strontium-90 Concentrations in Wells Spring and Fall 2011

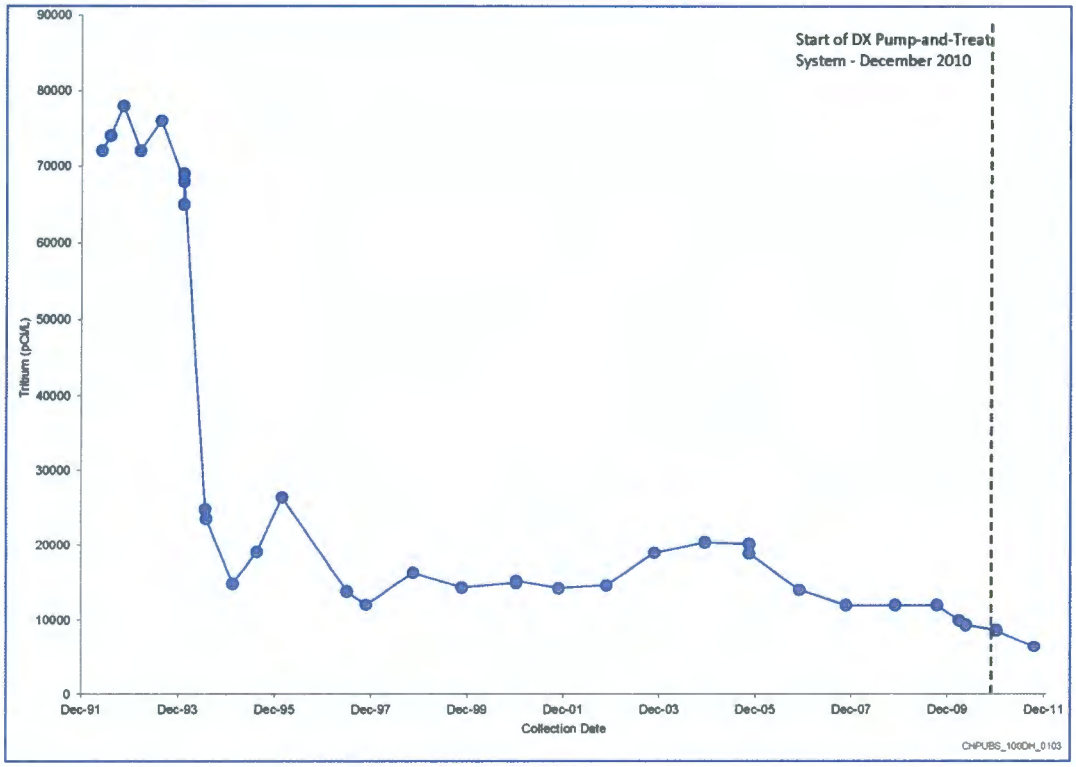

Figure 4-100. Tritium Concentrations in Groundwater over Time in Well 199-D5-17

Of the 25 RI wells installed throughout 100-D/H, 15 were sampled for tritium. Tritium was identified in groundwater samples during drilling in Wells 199-D5-132 and 199-D5-133 at 11,000 and 10,000 pCi/L, respectively. Elevated concentrations in this location are associated with the fuel storage basin and Fuel Storage Basin Trench (100-DR-1). Figure 4-102 shows that the maximum concentration in groundwater sample from Well 199-D6-3 was 20,000 pCi/L, which is equal to the DWS. This well also had the highest reliable tritium concentration detected at 100-D/H. Tritium contamination identified in the unconfined aquifer at Well 199-D6-3 may be associated with the upgradient 118-D-4 Burial Grounds, which had known reactor components and tritium. Soil samples collected from the borehole for Well 199-D6-3 did not have tritium detected, which provides further evidence of an upgradient source. Results from 2011 groundwater sampling were an order of magnitude lower, at 2,600 pCi/L. This introduces uncertainty regarding the actual tritium concentrations at that location.

Groundwater samples collected during drilling of RI Well 199-D3-5 had a maximum tritium concentration of 17,000 pCi/L. This well is located downgradient of the 118-D-2:1 Burial Ground, which has both tritium and strontium-90 identified among the potential COPCs.

4.5.5 Zinc

Historically, elevated concentrations of zinc in the unconfined aquifer have been found beneath 100-D. Zinc has been detected in the unconfined aquifer and the first water-bearing unit of the RUM. Detections in both aquifer units have been sporadic in most locations and do not have a consistent trend. The maximum concentration (from 2007 to 2011) identified in an aquifer tube was at Redox-4-6.0, which is located downgradient of the ISRM barrier. The concentration of 119 µg/L is at the high end of background concentrations at Hanford.

Figure 4-101. Spring 2010 Tritium Plume and Maximum 2011 Concentrations

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Table 4-20. Strontium-90 Sample Results from RI Drilling – 100-H

Area	Well Name (Borehole ID, SAP ID)	Sample Depth bgs m (ft)	Strontium-90 (pCi/L)
100-H: Unconfined Aquifer	199-H3-6 (C7626, Well 6)	15.2 (49.8)	--
		15.2 - 15.8 (49.8 - 51.8)	8.2
		16.4 (53.9)	5.9
	199-H3-7 (C7627, Well 7)	15.1 (49.5)	--
		15.1 (49.5)	2 (U)
		15.7 (51.6)	2.7 (U)
		15.7 (51.6)	2 (U)
	199-H6-3 (C7628, Well 10)	14.8 (48.5)	--
		14.8 (48.5)	1.8 (U)
		16.2 (53.1)	2.5 (U)
		16.2 (53.1)	2.6 (U)
		19.5 (64)	1.8 (U)
	199-H6-4 (C7629, Well 11)	13.9 (45.7)	--
		13.9 (45.7)	2 (U)
		13.9 (45.7)	2 (U)
		14.6 (48)	2.2 (U)
		16.2 (53)	1.9 (U)
		18.4 (60.5)	1.8 (U)
	199-H1-7 (C7630, Well 12)	NS	--
		NS	1.7 (U)
100-H: RUM Wells	199-H2-1 (C7631, Well R3)	9.2 (30.1)	--
		9.2 (30.1)	1.7 (U)
		9.2 (30.1)	2 (U)
		10.64 (34.9)[a]	1.9 (U)
		19.2 (62.9)	1.6 (U)
		48.2 (158.3)	1.4 (U)
		54.7 (179.6)	1.3 (U)
	199-H3-9 (C7639, Well R1)	12.3 (40.4)	--
		12.3 (40.4)	3.8
		13.8 (45.2)	1.6 (U)
		13.8 (45.2)	1.4 (U)
		14.2 (46.5)	1.6 (U)

Table 4-20. Strontium-90 Sample Results from RI Drilling – 100-H

Area	Well Name (Borehole ID, SAP ID)	Sample Depth bgs m (ft)	Strontium-90 (pCi/L)
		20.8 (68.4)	1.8 (U)
		40.8 (134)	1.6 (U)
		53.9 (177)	1.7 (U)
	199-H3-10 (C7640, Well R2)	13.9 (45.5)	1.7 (U)
		13.9 (45.5)	3.2
		15.2 (49.9)	1.6 (U)
		15.2 (49.9)	1.8 (U)
		16.1 (52.8)	1.6 (U)
		60.4 (198)	1.7 (U)
		68.2 (223.6)	1.3 (U)

Notes:

Bold values exceed the DWS of 8 pCi/L

Shaded cells indicate samples collected from a water-bearing unit below the RUM surface.

"U" flag indicates analyte was not detected above the MDA (minimum detectable activity) shown.

"—" indicates analyte was not analyzed for at that location.

NS = not specified.

Sources of zinc contribution to groundwater have not been isolated, but possibilities include trace amounts of zinc in iron oxide that was liberated during reduction associated with the ISRM, and/or trace zinc that may have been present in the sodium dithionite if it was manufactured using the zinc process. However, because zinc exceedances are present in the unconfined aquifer in other areas of 100-D/H, the ISRM barrier could not have been the only source.

Another possible source is from mobilization of zinc from the soil under acidic conditions, which did occur during reactor operations. As presented in *Solubility and Mobility of Copper, Zinc and Lead in Acidic Environments* (Reddy et al., 1995), the availability and mobility of zinc will increase in low pH environments.

Concentrations in groundwater samples from wells correlating with the 100-D southern Cr(VI) plume (199-D5-17, 199-D5-18, and 199-D5-40) have exceeded 91 µg/L, the standard for zinc where water is discharged to surface water. Groundwater samples in 100-H have also had zinc concentrations above the 91 µg/L level. The elevated concentrations are not consistent, and do not have a trend. In the first water-bearing unit of the RUM underlying 100-D (199-D8-54B), zinc was identified at concentrations just below 91 µg/L. Zinc has also been detected in wells completed in the RUM in the horn, and in 100-H, at concentrations ranging from at or near the detection limits to as high as 89 µg/L on one occasion. In 100-H, zinc concentrations in groundwater from Well 199-H4-2, which is screened in the basalt, have been well below the water quality standard consistently since late 1994. In these lower aquifers, as with the unconfined aquifer, there is no apparent trend or consistent detection.

During drilling activities for RI wells, water samples were collected at discrete depth intervals and analyzed for zinc. The resulting maximum zinc concentrations in the unconfined aquifer underlying

100-D ranged from 17 to 331 µg/L, and at 100-H unconfined aquifer concentrations ranged from 12 to 291 µg/L. No apparent distribution trends were observed.

4.5.6 Carbon Tetrachloride

Carbon tetrachloride is present in the unconfined aquifer in limited amounts underlying 100-D/H. A source has not been identified, although it could be associated with liquid waste site discharges.

The distribution of carbon tetrachloride in 100-D/H groundwater in fall 2009 is shown on. The detection limit (1 µg/L) exceeds the 0.23 µg/L (CWA – Human Health Water + Organism), so only exceedances of the detection limit are highlighted on the base map. Concentrations in the unconfined aquifer from monitoring wells at 100-D, 100-H, and the horn all show decreasing trends and most recent concentrations are below the detection limit. In 100-D/H, groundwater from the first water-bearing unit of the RUM has had a limited number of sampling events.

Based on the carbon tetrachloride groundwater sampling data collected from the RI wells, only one sample had carbon tetrachloride detected above the detection limit. Well 199-H3-9 (C7639, Well R1), at a depth of 40.8 m (134 ft) bgs, had a detected value of 2.7 µg/L. The sample is "J" flagged, meaning the sample was detected above the MDL but less than the practicable quantitation limit. All groundwater samples from aquifer tubes were below the detection limit of 1 µg/L.

4.5.7 Chloroform

Chloroform is a minor contaminant present in the unconfined aquifer in limited amounts at 100-D/H. A source has not been identified, though it could be associated with liquid waste site discharges or the biodegradation of carbon tetrachloride.

Chloroform has been sporadically detected in the unconfined aquifer underlying 100-D/H, with concentrations ranging from below detection to 53 µg/L (Well 199-H4-47 in 1992). Since 1998, chloroform has not been detected in groundwater underlying 100-D/H above 10 µg/L. Except for chloroform detected in Well 199-H4-5, chloroform concentrations are not associated with a specific location or with a trend.

The highest concentration detected was from Well 199-D5-143 (3.4 µg/L). Chloroform concentrations in groundwater samples from the RI aquifer tubes were below action levels. Groundwater samples from the first water-bearing unit of the RUM have not been analyzed for chloroform.

4.5.8 Sulfate

Sulfate is present in the unconfined aquifer underlying a large portion of 100-D but with only occasional detections at 100-H. Sulfate in the unconfined aquifer at 100-D, not associated with the ISRM barrier, is a result of sulfuric acid being used primarily as a decontamination solution with some used in water treatment. In addition, mercury-contaminated, commercial-grade sulfuric acid was used for cooling water pH adjustment at 100-K (1968 to 1977). Although this period was after the shutdown of the 100-D/H reactors, mercury contamination associated with sulfuric acid has been identified during the remediation of the 100-D-77 waste site at the 183-DR Head House.

Before 2005, sulfate concentrations in the unconfined aquifer underlying the southern area of 100-D exceeded 1,000,000 µg/L. These high concentrations were associated with injections of sodium dithionite solution at the ISRM barrier, which elevated sulfate concentrations in groundwater along the barrier and in some downgradient wells and aquifer tubes. However, since 2005, concentrations have dropped to less than 500,000 µg/L. The secondary DWS for sulfate is 250,000 µg/L.

A portion of the 183-H Solar Evaporation Sedimentation Basins was used as an evaporation treatment facility for neutralized acid wastes. Four of the basins were converted for use as solar evaporators after cessation of reactor operations at 100-H. The neutralized nitric and sulfuric acid wastes, generated by reactor fuel fabrication processes in the 300 Area were transferred to the open-topped basins and allowed to evaporate. The basins apparently leaked substantial amounts of waste to the vadose zone, creating a high-concentration groundwater plume of nitrate, sulfate, mercury, and other metals.

Sulfate has been analyzed for in groundwater samples from wells/piezometers in the first water-bearing unit of the RUM as well as in the RLM unit in the recent 5-year period, with no exceedances of 250,000 µg/L (DWS).

4.5.9 Other Contaminants Evaluated in the RI

As presented in the 100-D/H SAP (DOE/RL-2009-40), groundwater COPCs and several additional analytes were identified for evaluation during the RI. In addition to the statistical evaluation of historical data presented in Section 4.2.1, the analytical data from the RI groundwater monitoring wells were also evaluated. The following contaminants that were detected in borehole groundwater samples from RI wells, yet not discussed separately, are shown in Table 4-21.

Radionuclides

Groundwater analysis conducted during the RI resulted in the detection of two radionuclides, other than those discussed separately: europium-154 and technetium-99. The remaining radionuclides that were analyzed per the 100-D/H SAP (DOE/RL-2009-40) were not detected above the MDA.

Europium-154 was detected in a groundwater sample from Well 199-D3-5 (C7620, Well 2) at 57 pCi/L. However, the total analytical error reported by the laboratory was 71 pCi/L, making this result questionable.

Technetium-99 was detected in groundwater samples collected from several RI wells, as presented in Table 4-22.

Values for gross alpha ranged from below the MDA to 14 pCi/L in Well 199-H3-10 (C7640, Well R2), with all but two detections below 10 pCi/L. Gross beta values ranged from below the MDA to 130 pCi/L in Well 199-D5-132 (C7622, Well 4). With the exception of the detections in that one location (Well 199-D5-132), the remaining gross beta results were below 22 pCi/L, with most detections below 10 pCi/L.

Metals, Ions and Anions

Several metals were detected in groundwater as shown in Table 4-21. The maximum detections are presented in Table 4-23.

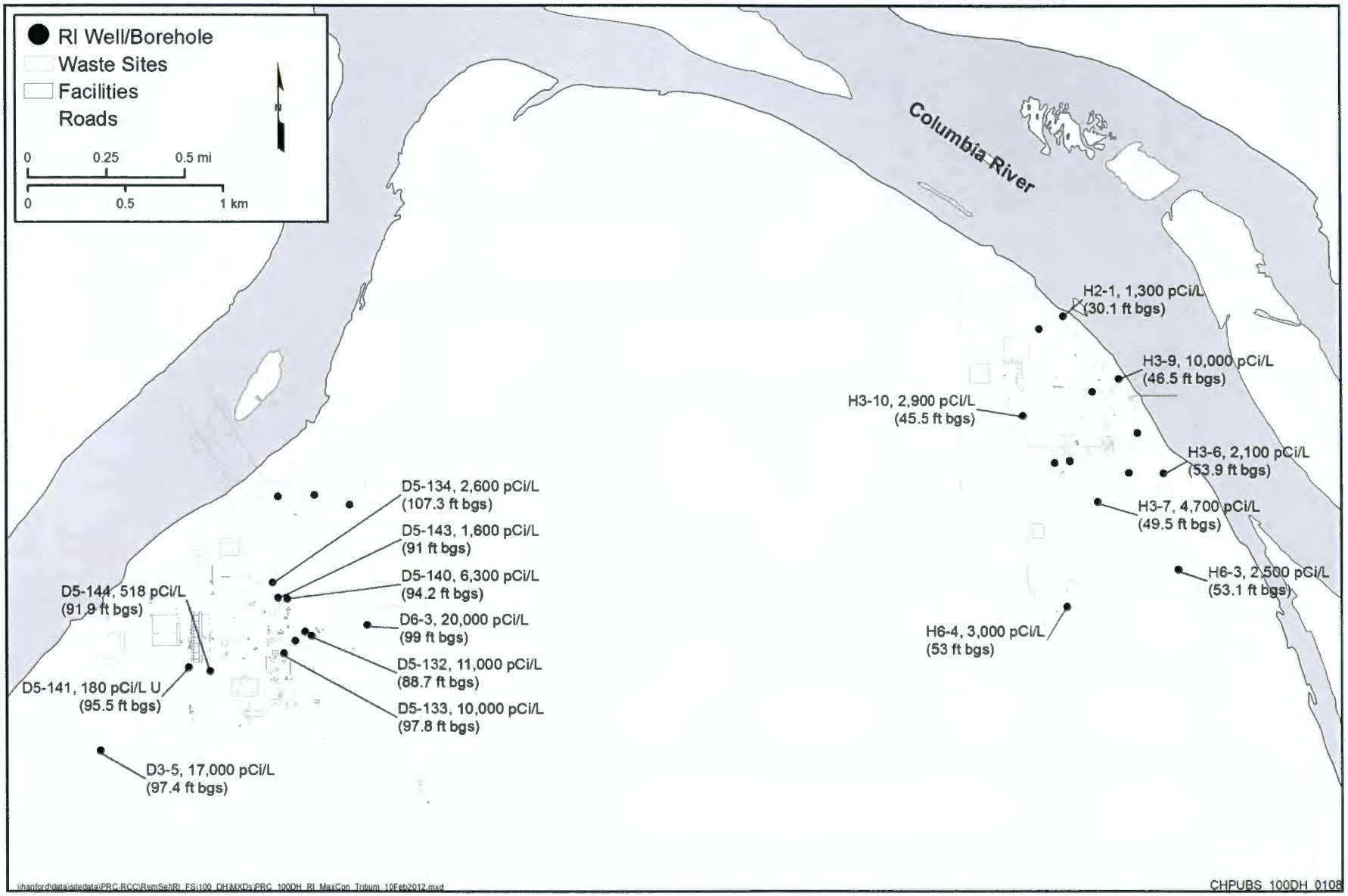

Figure 4-102. Maximum Tritium Concentrations in Borehole Water Samples from RI Wells

Table 4-21. Other Contaminants Evaluated in the RI and Detected

Radionuclides	Metals, Ions, and Anions		Volatile Organic Carbon
Technetium-99	Aluminum	Phosphorus	Acetone
Europium-154	Barium	Potassium	Tetrachloroethene
Gross alpha	Boron	Selenium	Toluene
Gross beta	Calcium	Silicon	
	Cobalt	Sodium	
	Copper	Uranium	
	Iron	Vanadium	
	Lead		
	Magnesium		
	Manganese		
	Molybdenum		
	Nickel		

Table 4-22. Detections of Technetium-99 in RI Wells – Borehole Groundwater Samples

Well Name	Boring ID	SAP ID	Result (pCi/L)
199-D3-5	C7620	Well 2	190
199-H6-4	C7629	Well 6	68
199-H2-1	C7631	Well R3	100
199-H3-9	C7639	Well R1	18
199-H3-10	C7640	Well R2	10

Table 4-23. Maximum Detections of Metals, Ions and Anions in RI Wells – Borehole Groundwater Samples

Analyte	Maximum Concentration (µg/L)	Analyte	Maximum Concentration (µg/L)
Aluminum	3,320	Molybdenum	36.8
Barium	471	Nickel	20.3
Boron	96.2	Phosphorus	55.7 (one detection)
Calcium	118,000	Potassium	7,420
Cobalt	8.12	Selenium	5.36 (one detection)
Copper	136	Silicon	20,800
Iron	1600	Uranium	6.75
Lead	8.36	Vanadium	37.3
Magnesium	32,900		
Manganese	777		

Volatile Organic Compounds

Groundwater samples collected from RI wells during drilling had the following VOCs detected: acetone had a maximum detection of 4.9 μg/L, tetrachloroethene had a maximum detection of 2.8 μg/L, and toluene had a single detection of 1.3 μg/L. No other VOCs were detected.

4.5.10 Summary of Contamination in Groundwater at 100-D/H

The nature and extent of contamination in groundwater has been monitored extensively since 1997 to evaluate the implementation of the interim remedies. In 2009, an RPO was initiated to optimize and expand the pump-and-treat systems in 100-D/H. This effort resulted in installation of 70 production wells that provided additional information in support of this RI Report. In addition, the 10 RI characterization boreholes and the 17 RI characterization wells have provided significant new information that expands the general understanding of the nature and extent of contamination in groundwater.

The sampling and analysis of groundwater was conducted under the 100-D/H SAP (DOE/RL-2009-40, as modified by Change Notice for Modifying Approved Documents/Workplans In Accordance with the Tri-Party Agreement Action Plan, Section 9.0, Documentation and Records: Interim Action Monitoring Plan for the 100-HR-3 and 100-KR-4 Operable Units, DOE/RL-96-90, Rev. 0 [TPA-CN-298]). The RPO work was conducted under a separate SAP (Sampling and Analysis Plan for Installation of 100-HR-3 Groundwater Operable Unit Remedial Process Optimization Wells [DOE/RL-2009-09]). The collective body of information, including the annual monitoring information, indicates that the nature and extent of contamination in groundwater at 100-D/H is well understood in terms of the magnitude of the contamination present and the geography of the contaminant plumes.

Groundwater contamination statistics were calculated for the 100-D, horn, and 100-H portions of 100-D/H. Cr(VI) is the largest single groundwater contaminant with plumes in 100-D, horn, and 100-H. The aerial extent (in square kilometers) of the various contaminant plumes within 100-D/H are shown in Table 4-13. In 100-D, the Cr(VI) plume is subdivided into the 100-D southern and northern plumes. The 100-D southern plume originates near the 105-DR Reactor near the 100-D-12 and 100-D-100 waste sites. Figure 4-64 shows the relationship of the Cr(VI) plume to the potential source areas. The 100-D southern plume contains the highest concentrations of Cr(VI) at the Hanford Site with concentrations in excess of 60,000 μg/L (69,700 μg/L in Well 199-D5-122 in August 2010). Some vertical stratification of Cr(VI) is observed in the unconfined aquifer in the 100-D southern plume, but the stratification is not consistent throughout. RI data indicated that in the 100-D southern plume, where there are moderately high concentrations of Cr(VI), high concentrations are present at the bottom of the unconfined aquifer.

The 100-D northern plume is located closer to the 105-D Reactor near portions of the piping system that transmitted concentrated sodium dichromate from the unloading station to the head house next to Building 183-D. Figure 4-64 shows the relationship of the plume to the nearby waste sites. A single waste site has not yet been identified that coincides with the area of higher concentrations at the 100-D northern plume. There is therefore some potential that leakage from the 182-D Reservoir has separated a single larger plume into the two distinct areas. Leakage from the reservoir is apparent in the geochemistry from nearby Well 199-D5-33, which has a geochemical signature similar to the Columbia River (Section 3.8).

A large diffuse plume of Cr(VI) is located in the horn between 100-D and 100-H (Figures 4-66 and 4-67). Groundwater from the unconfined aquifer underlying the horn generally exhibits much lower Cr(VI) concentrations than are present in the 100-D plumes, although concentrations at many locations still exceed the AWQC of 10 μg/L AWQC and the MTCA ("Groundwater Cleanup Standards" [WAC 173-340-720]) level of 48 μg/L. The horn has very few waste sites, and the Cr(VI) detected in shallow groundwater likely migrated across the horn with groundwater flow from 100-D, rather than having originated from local releases.

The Cr(VI) plume in the unconfined aquifer at 100-H (Figures 4-66 and 4-67) is characterized by much lower concentrations than the 100-D plumes, and has mostly been remediated in this area. Cr(VI) concentrations in the unconfined aquifer are less than 100 µg/L; however, most portions of the plume underlying 100-H exceed the AWQC of 10 µg/L. Figure 4-74 shows the relationship of primary sources to the Cr(VI) plume. In the unconfined aquifer, higher groundwater flow rates and the relative success of the interim remedy pump-and-treat system in this area have resulted in low remaining concentrations.

Cr(VI) has been identified underlying 100-H within the first water-bearing unit of the RUM, with the highest concentrations near the Columbia River, as well as in one location of the horn near 100-D. The contamination within the RUM at 100-H is likely a result of the high hydraulic head conditions created from the groundwater mound at the 116-H-7 Retention Basin during reactor operations. The pressure of the mound pushed the contaminated groundwater into the lower unit. As discussed in Chapter 3, the material of the RUM between the unconfined aquifer and the first water bearing unit within the RUM consists of more transmissive material, which also appears to be thinner at 100-H, as compared to 100-D.

The details of Cr(VI) in groundwater were described in Section 4.5. In addition, the rebound testing conducted in 2009 (*Aquifer Testing and Rebound Study in Support of the 100-H Deep Chromium Investigation* [SGW-47776]), resulted in an increase of Cr(VI) concentrations in the RUM Well 199-H4-12C, which does not appear to be hydraulically connected to the unconfined aquifer (based on current data). At Well 699-97-48C, high hydraulic head would have been present as a result of discharges to the 116-DR-1&2 Trench and retention basins at 100-D. As discussed in Chapter 3, the presence of the Ringold Formation unit E in locations just northeast of the trenches appears to have resulted in a preferential pathway across the horn. The geologic conditions also would have been conducive to high hydraulic head near Well 699-97-48C.

Several other constituents are important in groundwater at 100-D/H. The interim action RODs (100-HR-3 and 100-KR-4 Interim ROD [EPA/ROD/R10-96/134] and 100-HR-3 Interim ROD Amendment [EPA/AMD/R10-00/122]) listed the following potential co-contaminants at 100-D/H: nitrate, strontium-90, tritium, uranium, and technetium-99. These co-contaminants have been monitored extensively in the years since these RODs were issued. Technetium-99 is present at very low levels and does not warrant further classification as a co-contaminant, but continued monitoring is required through the current RCRA permit. Uranium levels are below the DWS, but increased from concentrations around 10 µg/L to a concentration of 28.9 µg/L in Well 199-H4-3, in response to unusually high water table elevations during 2011.

The nitrate plumes are defined as areas greater than the DWS (45,000 µg/L), and overlap a portion of the Cr(VI) plumes at 100-D. The south nitrate plume extends over to the 118-D-3 waste site east of the 105-DR Reactor. The northern nitrate plume extends from the 105-D Reactor up to the area of the retention basins and west of the 116-DR-1&2 Trench. Nitrate concentrations in groundwater above the DWS in 100-H are limited to two small areas (Figure 4-90). The RI results confirmed the extent of the plumes in both reactor areas.

Strontium-90 is found in the unconfined aquifer at both 100-D and 100-H. A small plume has historically been monitored at 100-H in the area east of the 105-H Reactor near the 116-H-7 Retention Basin and the 116-H-1 Trench (Figure 4-98). Both of these are likely historical release points for Sr-90-contaminated water. The strontium-90 plume at 100-H exhibits small seasonal variations apparently related to the water table elevations. Strontium-90 was detected in RI Well 199-H3-6 at 8.2 pCi/L. At 100-D, strontium-90 is found in groundwater near 105-D Reactor and the 116-D-1A and 116-D-1B wastewater trenches. These are both likely historical release points for Sr-90 contaminated water; the fuel storage basin at 105-D Reactor may have leaked contaminated water to the vadose zone and the wastewater trenches received

fuel storage basin overflow and other radiologically-contaminated wastewater. Sr-90 has been detected in groundwater near 105-D Reactor for about 20 years, with persistent detections in Well 199-D5-12 at concentrations as high as 52.6 pCi/L in March 1990. This well was decommissioned in 2002 when groundwater levels dropped below the pump intake level, but the well had not been sampled since late 1999. RI Well 199-D5-132 (C7622, Well 4) was installed to replace this well and provide an evaluation point for strontium-90 near 116-D-1A trench. Elevated levels of strontium-90 were detected in borehole groundwater samples at all depth intervals in Well 199-D5-132 (Table 4-19) at concentrations consistent with those historically detected in Well 199-D5-12. Persistent detections of Sr-90 in groundwater near the 107-D and 107-DR Retention Basins have declined to below the DWS in recent years. Discontinuous low-level detections of Sr-90 have been measured in groundwater near 105-DR Reactor.

As discussed in Section 4.4.3.2, tritium has been detected historically in the unconfined aquifer at concentrations exceeding the DWS value of 20,000 pCi/L near the ISRM barrier (Figure 4-101). The plume has been documented as being a remnant from the 100-N tritium plume associated with the 1325-N Crib. Concentrations in the unconfined aquifer near that area have since declined to less than the DWS. During the RI, tritium was detected in borehole groundwater samples from Wells 199-D5-132 and 199-D5-133 at 11,000 and 10,000 pCi/L, respectively, associated with the fuel storage basin and 100-DR-1. The groundwater sample from Well 199-D3-5 had a maximum detection of 17,000 pCi/L, likely associated with a nearby burial ground. At RI Well 199-D6-3, tritium was detected in a borehole groundwater sample at 20,000 pCi/L. However, a groundwater sample collected from the completed well in 2011 was 2,600 pCi/L, which introduces uncertainty regarding actual tritium concentrations in the unconfined aquifer at this location.

Small amounts of other constituents are present in the unconfined aquifer underlying 100-D/H. Those that were identified in the groundwater include zinc, carbon tetrachloride, chloroform, and sulfate. Zinc shows sporadic values greater than the aquatic limit (91 µg/L) at several wells in 100-D/H. Carbon tetrachloride is found in very small amounts. The carbon tetrachloride overlaps a portion of the Cr(VI) plumes. Elevated chloroform was detected in groundwater from several wells within the 100-D north Cr(VI) plume. Sulfate is present in the unconfined aquifer at concentrations exceeding the secondary drinking water standard of 250,000 µg/L only at or downgradient of the ISRM barrier, where it is a byproduct of barrier chemical placement. Sulfate is detected in the unconfined aquifer in other areas underlying 100-D associated with sulfuric acid use. Additional analytes, such as antimony, cadmium, cobalt, silver, and nickel, were also identified in groundwater; however, these analytes had infrequent detections, which result in uncertain status. All of these analytes were evaluated in the risk assessment and are discussed further in Section 4.4 and Chapter 6.

In conclusion, the nature and extent of contamination in groundwater at 100-D/H is complex. The major contaminant in groundwater is Cr(VI), which covers an area in excess of 7 km^2 of unconfined aquifer. Delineation of the plume boundaries is fairly well understood. To the southwest, Well 199-D3-5 was intended to delineate the plume along that boundary, but Cr(VI) was identified in borehole groundwater samples during drilling. Therefore, as part of the RD/RA work plan or remedy implementation, additional delineation may be needed to ensure capture and/or treatment of the entire southern plume. Underlying 100-H, Cr(VI) contamination is present in the first water-bearing unit of the RUM, but the plume boundaries have not been fully delineated in that area to the south and southwest. Contamination is also present in one well located in the horn (Well 699-97-48C), near 100-D. Other contaminants (primarily metals) have been detected and are evaluated in the risk assessment and discussed in Section 4.4 and Chapter 6.

4.6 Evaluation of Potential Effects on the Columbia River Adjacent to 100-D/H

Appendix L presents an evaluation of contaminants in riparian and nearshore media and the Columbia River. This evaluation addresses, on a reactor area basis, the potential for Hanford Site contaminants in soil or groundwater to migrate to riparian or nearshore areas or to the Columbia River at concentrations that could be of concern to ecological receptors.

The Appendix L evaluation supplements the analysis of the River Corridor-wide ecological risks presented in the ecological risk assessment (ERA) (RCBRA, Volume 1 [DOE/RL-2007-21]). The ERA identifies on a site-wide basis some contaminants of ecological concern (COECs) in riparian and nearshore media (soil, sediment, and water) that could warrant further evaluation.

Appendix L also addresses COECs identified in the Columbia River Component (CRC) ERA (DOE/RL 2010-117, Volume I), specifically those identified for 100-D/H. The following text describes the results of the two risk assessments, including the types of data collected to complete the assessments.

Table 4-24 lists the combined COECs from both the RCBRA (DOE/RL-2007-21) and the CRC (DOE/RL-2010-117). The evaluation of the HHE risk presented in the CRC (DOE/RL-2010-117, Volume II) addresses all the data collected throughout the Hanford Reach and downstream to McNary Dam, as directed in the Columbia River RI Work Plan (DOE/RL-2008-11). Chapters 6 and 7 summarize the Appendix L analysis, including results regarding which, if any, of the following COECs could be attributed to sources within the 100-D or 100-H.

Table 4-24. Riparian, Nearshore, and Riverine COECs from the RCBRA and CRC

COEC	Receptors	Media
Aluminum [1]	Fish Aquatic Invertebrates Aquatic Plants	Pore Water
Arsenic [2]	Terrestrial Plants	Riparian Soil
Cadmium 2	Aquatic Plants and Invertebrates	Sediment
Chromium [1]	Fish Aquatic Invertebrates Aquatic Plants	Pore Water
Chromium [1,2]	Aquatic Plants and Invertebrates and the Bufflehead	Sediment
Chromium [2]	Terrestrial Plants and Invertebrates	Riparian Soil
Cr(VI) [1,2]	Aquatic Plants and Invertebrates	Sediment
Cr(VI) [1,2]	Fish Aquatic Invertebrates Aquatic Plants	Pore Water
Lead [1]	Fish Aquatic Invertebrates Aquatic Plants	Pore Water

Table 4-24. Riparian, Nearshore, and Riverine COECs from the RCBRA and CRC

COEC	Receptors	Media
Lead [2]	Terrestrial Plants	Riparian Soil
Manganese [2]	Aquatic Plants and Invertebrates	Sediment
Manganese [2]	Aquatic Plants and Invertebrates	Pore Water
Mercury [2]	Terrestrial Invertebrates	Riparian Soil
Nickel [1]	Fish Aquatic Invertebrates Aquatic Plants	Pore Water
Nitrate [1]	Fish Aquatic Invertebrates Aquatic Plants	Pore Water
TPH–Diesel [2]	Terrestrial Invertebrates	Riparian Soil
Uranium [2]	Aquatic Plants and Invertebrates	Pore water
Zinc [2]	Terrestrial Plants and Invertebrates and Kingbirds	Riparian Soil

Notes:

Evaluation on whether 100-D/H represents a potential source is presented in Appendix L.

1. COECs presented in the executive summary of the CRC (DOE/RL-2010-117)

2. COECs presented in Sections 8.4 and 8.5 of the RCBRA (DOE/RL-2007-21)

COEC = contaminant of ecological concern

CRC = Columbia River Component Risk Assessment, Volume 1: Screening Level Ecological Risk Assessment (DOE/RL-2010-117)

RCBRA = River Corridor Baseline Risk Assessment, Volume 1: Ecological Risk Assessment (DOE/RL-2007-21)

4.6.3 Summary of Results and Conclusions of RCBRA and CRC

The RCBRA (DOE/RL-2007-21) evaluated ecological risks at 48 nearshore study sites potentially affected by contamination from Hanford Site sources in comparison to reference sites. Study sites were selected in areas where known contaminated groundwater plumes enter the Columbia River and in areas between the plumes. Twenty-two COPECs were identified for the nearshore environment and sixteen of these (all inorganics) were identified for further consideration. The RCBRA (DOE/RL-2007-21) concluded that across the Hanford Reach of the Columbia River (that is, River Corridor-wide), five COPECs are COECs (cadmium, chromium, Cr(VI), manganese, and uranium) in the nearshore environment that may present an unacceptable level of risk for one or more of the assessment endpoint entities (aquatic plants, aquatic invertebrates, amphibians, fish, and wildlife). These results are based primarily on the comparisons of COPEC concentrations to toxicity benchmarks, measures of exposure and effects in biota, or the results of wildlife exposure analyses (RCBRA Report [DOE/RL-2007-21], Volume 1).

The RCBRA (DOE/RL-2007-21) also evaluated ecological risks at 18 representative riparian study sites located adjacent to, or where they may be directly affected by, known contaminated media (that is, groundwater seeps, soil, and sediment). In addition, data from the 100-B/C area pilot study and the 100-NR-2 ecological study were evaluated. As with the nearshore environment, 22 COPECs were identified for the riparian environment. The RCBRA (DOE/RL-2007-21) identified 9 of the identified

22 COPECs (arsenic, chromium, copper, lead, mercury, selenium, TPH-diesel, vanadium, and zinc) as possibly presenting some level of risk for one or more of the assessment endpoint entities (terrestrial plants, invertebrates, and wildlife). This is based on soil bioassays, comparison of COPEC concentrations to plant or terrestrial invertebrate benchmarks, or the results of wildlife exposure analyses. However, conclusions in the RCBRA (DOE/RL-2007-21) were that on a River Corridor-wide basis (combined 100 and 300 Areas), only six of these COPECs should be considered COECs (arsenic, chromium, lead, mercury, TPH-diesel, and zinc). Appendix L discusses these RCBRA-specified COECs with respect to ecological risk within 100-D/H.

The CRC (DOE/RL-2010-117, Volume 1) included an ecological risk assessment that combines both screening and baseline elements. Abiotic media were compared to screening benchmarks for surface water, sediment, and pore water to identify COPECs. Soil concentrations were compared to plant and invertebrate benchmarks, while desktop food web models were used to evaluate risks to wildlife. A baseline assessment was conducted to assess risk to fish using tissue residue data. The CRC (DOE/RL-2010-117, Volume 1) concluded there were eight COECs (aluminum, chromium, Cr(VI), lead, manganese, mercury, selenium, and uranium) within pore water, surface water, island soils, and sediment. The evaluation included distinct conclusions for the reach adjacent to the100 Area versus those for the reach adjacent to the 100-D/H Source OUs. Six COECs were identified for the 100-D/H Source OUs, as presented in Table 4-24. Appendix L discusses these CRC-specified COECs with respect to ecological risk within the 100-D/H Source OUs.

4.6.4 Columbia River Surface Water, Pore Water, and Sediment Investigation

In 2004, a process was established to compile, classify, and manage environmental data (for example, surface water and sediment) associated with the Columbia River in *Columbia River Component of the River Corridor Baseline Risk Assessment: Basis and Assumptions on Project Scope* (DOE/RL-2004-49). The CRC database was created because of these efforts and was documented in *Existing Source Information Summary Report Compilation/Evaluation Effort: December 2004 to September 2005* (WCH-64). The subsequent *Columbia River Component Data Evaluation Summary Report* (WCH-91) described the activities that were undertaken to evaluate the data collected in the compilation effort and to assist in defining the extent of Hanford Site-related contamination. The compiled data were used to identify potential data gaps in the spatial, temporal, and chemical composition of the existing dataset. The *Columbia River Component Data Gap Analysis* (WCH-201) presented the results of that analysis and provided the foundation for the sampling plan that was documented in the Columbia River RI Work Plan (DOE/RL-2008-11).

The scope of the Columbia River RI Work Plan (DOE/RL-2008-11) and *DQO Summary Report for the Remedial Investigation of Hanford Site Releases to the Columbia River* (WCH-265) included the following fieldwork component, which has generated data necessary to fill data gaps in the understanding of current conditions in the Columbia River. The data from the field activities were evaluated in both ecological and human health risk assessments reported in the CRC (DOE/RL-2010-117).

Sampling to fulfill the needs defined in the Columbia River RI Work Plan (DOE/RL-2008-11) was initiated in October 2008 and completed in June 2010. Media sampled included surface water, pore water, sediment (shoreline, shallow, cores), island soil, and six species of fish. The results of the biota sampling are discussed in Appendix L and Section 4.6.1. The RI field activities associated with the collection of sediment, river water, and island soil in the Columbia River adjacent to and downstream from the Hanford Site and in nearby tributaries are documented in Field Summary Report for Columbia River RI (WCH-352). Field Summary Report for Columbia River RI (WCH-352) describes the sampling locations, identifies samples collected, and describes modifications and additions made to the SAP that was provided as

Appendix A to the Columbia River RI Work Plan (DOE/RL-2008-11). Groundwater upwelling field activities and data collection are documented in the Columbia River RI Report (WCH-380).

Groundwater Upwelling Investigation at 100-D/H

Groundwater beneath the Hanford Site discharges to the Columbia River via seeps and upwelling to the riverbed. This flow path for groundwater provides a means for transporting Hanford Site-associated contaminants that entered the groundwater from past waste disposal practices to the Columbia River. As discussed in Section 2.1.7 of this report, the nearshore groundwater conditions are directly affected by river stage. The greatest contaminant flux and highest concentrations at exposure locations are postulated to occur during periods of low river stage. During this period, the hydraulic gradient toward the river is greatest and mixing between river water and groundwater is minimal.

Sediment samples collected from the locations shown on Figures 2-5, 2-6, 2-7, and 2-8 were analyzed for a range of radiological and nonradiological analytes as described in Table 2-5. Sediment samples were obtained as close to the pore water sample locations as reasonably possible, with a preference given to locations with fine sediment deposits. Sample volume was limited in some locations because of the dominance of cobbles on the riverbed. In locations where sediment sample volume was limited, not all analyses could be performed at each location. Information on the number of sediment samples collected and the period in which they were obtained is presented in Table 2-7. Additional sediment, island soil, and surface water samples were collected in areas identified in *Columbia River Component Data Gap Analysis* (WCH-201) and the Columbia River RI Work Plan (DOE/RL-2008-11).

To address the uncertainty related to the level of contamination entering the Columbia River via upwelling, including contaminant transport mechanisms, data were collected near 100-D/H. Pore water, surface water, and sediment sampling in the Columbia River was conducted in 2009 and 2010, as outlined in the Columbia River RI Work Plan (DOE/RL-2008-11). The following paragraphs discuss the sediment, surface water, and pore water samples presented in the CRC (DOE/RL-2010-117, Volume I) as described in *DQO Summary Report for the Remedial Investigation of Hanford Site Releases to the Columbia River* (WCH-265) and *Sampling and Analysis Instructions for the Remedial Investigation of Hanford Site Releases to the Columbia River* (WCH-286) as these data relate to 100-D/H. The aquifer tube results are presented and discussed in Section 4.5.

The influence of contaminants on the water quality immediately above groundwater upwelling locations was determined by taking surface water samples. River water was collected concurrently during pore water sample collection at approximately 0.3 m (12 in.) above the riverbed. At 100-D/H, surface water sample analysis at all sample locations included the analytes listed in Table 2-5. Information on the number of surface water samples collected and the period in which they were obtained is presented in Table 2-7.

To determine if surface water or groundwater was being collected, specific conductance and temperature were used as indicators. The specific conductance or conductivity in the Columbia River is typically lower (130 to 145 μS/cm) than groundwater (400 to 600 μS/cm). In addition, in this region of the Columbia River, surface water temperatures typically range from approximately 0.5 °C (33 °F) in the winter months to more than 27 °C (80 °F) during the late summer months, whereas groundwater typically stays between 7 °C (45 °F) and 15 °C (60 °F) (Columbia River RI Report [WCH-380]).

Phase II(a) and Phase II(b) Sampling

Pore water samples collected as part of the Phase II(a) groundwater upwelling investigation defined in the Columbia River RI Work Plan (DOE/RL-2008-11) helped to delineate areas of groundwater upwelling into the river bottom. Measurements of conductivity and temperature in pore water were used to guide the selection of Phase II(b) stations that were sampled for indicator contaminants. As described in Section 2.1.7, Cr(VI) was the indicator contaminant in both the 100-D and 100-H areas. Further discussion

of this investigation is found in the Columbia River RI Report (WCH-380). Additional discussion of the analytical results from these areas can be found in Hanford Site Releases Data Summary (WCH-398).

Conductivity measurements made at all of the Phase II(b) sample locations near 100-D indicated the presence of groundwater. Of the 30 sites where pore water was measured for Cr(VI), 11 sample results exceeded the AWQC (10 µg/L). The two highest Cr(VI) pore water concentrations (112 and 331 µg/L) were collected from two separate regions with water depths less than 0.9 m (3 ft) below the low water mark. The highest Cr(VI) pore water concentration (331 µg/L) was located just upstream from the 181-D River Pump Station where an aquifer tube showed a concentration of 380 µg/L and a well inland had a concentration of 700 µg/L (see Figure 4-63). The second highest concentration (112 µg/L) was located about 9.7 m (32 ft) off shore from an area where Cr(VI) groundwater plume estimates of 100 µg/L are found in the wells.

Thirty Phase II(b) sample locations were selected near 100-H. Pore water conductivity measured during Phase II(b) showed the presence of groundwater at all locations. Fifteen Cr(VI) sample results exceeded the AWQC (10 µg/L; results ranged from 12 to 46 µg/L). Three of these fifteen locations were downriver near the White Bluffs Townsite boat launch. One station had a strontium-90 value of 6.78 pCi/L, which is comparable to the predicted groundwater plume concentration (8 pCi/L) in that area.

Phase III Sampling

Phase III sample locations were a subset of the previous sample locations for characterization sampling and analysis of pore water, surface water (0.3 m [1 ft] above the riverbed), and collocated sediment for a broad range of analyses defined in the Columbia River RI Work Plan (DOE/RL-2008-11).

Sediment samples were collected as close to the pore water sample location as reasonably possible, with a preference given to locations with sediment deposits. Bulk sediment Cr(VI) concentrations ranged from 0.2 to 4.7 mg/kg with the highest concentration found in 100-D, which corresponds to the concentrations as measured in pore water. Additional Phase III analytical result discussions can be found in Hanford Site Releases Data Summary (WCH-398).

Six sample locations were selected for Phase III sampling in 100-D. Samples from pore water and proximal river water were collected at all six sites. Pore water conductivity values ranged from 213 µS/cm to 560 µS/cm. Typical groundwater conductivity values range from 400 µS/cm to 600 µS/cm. Laboratory results for surface water Cr(VI), total uranium, and strontium-90 were below detection limits. Pore water sample results for Cr(VI) ranged from 9 to 640 µg/L, with the maximum value found at station T100D3A where a high value of 331 µg/L was detected during Phase II(b) (Columbia River RI Report [WCH-380, Rev. 1]). A single detection of strontium-90 (1.5 pCi/L) was reported at a station along the Hanford Site shoreline adjacent to the 100-D island. Tritium was also detected from 353 to 14,100 pCi/L with the maximum level (14,100 pCi/L) found in an area adjacent to the 100-D strontium-90 groundwater plume.

Six sample locations were selected for Phase III sampling in 100-H. Samples from pore water and proximal river water were collected at all six sites. Pore water conductivity values ranged from 184 to 343 µS/cm, perhaps indicating some degree of suppression or mixing as a result of the river stage at the time of the Phase III measurements. Typical groundwater conductivity values range from 400 to 600 µS/cm. A site upriver of the 100-H Reactor area yielded the maximum pore water conductivity value during Phase III. Laboratory results for Cr(VI), total uranium, and strontium-90 in surface water were not detected during Phase III. Pore water results for Cr(VI) ranged from 7 to 50 µg/L, with the maximum value found at the station upriver of 100-H. Tritium was also detected over a range of 454 to 1,250 pCi/L, with the maximum found at the same upriver station as the Cr(VI) maximum. The amounts of strontium-90 detected in pore water results were all below detection with the exception of a 6 pCi/L result from a location adjacent to the 100-H strontium-90 groundwater plume (Columbia River RI Report [WCH-380, Rev. 1]).

Sediment and Surface Water Samples near 100-D/H

In addition to the sampling performed during the groundwater upwelling investigation, samples of sediment, surface water, and island soil were obtained from select locations to develop a better understanding of the nature and extent of potential contaminants released from the Hanford Site and to support subsequent human health and ecological risk assessments.

For 100-D/H, the following sampling efforts were conducted:

- A shallow sediment core was collected near the 181-D river pump station.
- Several shallow sediment samples were collected from the downstream end of the 100-D island, and further downstream from the island both shallow sediment and shoreline sediments were taken at locations conducive to sediment deposition in the river.
- Shallow sediments were collected, as well as three surface water samples, along the Grant County shoreline of the river.
- Ten soil samples and several shoreline sediment samples were collected from Island 3 (upriver from 100-H).
- Soil and shoreline samples were collected from Locke Island (adjacent to and downstream from 100-H).

The analytical results for these samples are presented in Hanford Site Releases Data Summary (WCH-398). The CRC (DOE/RL-2010-117) presents an evaluation of the HHE risk represented by all the data collected, as directed in the Columbia River RI Work Plan (DOE/RL-2008-11). This evaluation addresses all the data collected throughout the Hanford Reach and downstream to McNary Dam.

Conclusions:

The fieldwork associated with the Columbia River RI was completed in accordance with the requirements defined in the Columbia River RI Work Plan (DOE/RL-2008-11). Based on the data collected, it was confirmed that groundwater upwelling does occur in the Columbia River. The potential impacts to ecological and human receptors were evaluated in the CRC risk assessments (DOE/RL-2010-117) and are discussed in Chapters 6 and 7.

4.6.5 Nature and Extent of Contamination in Riparian and Nearshore Areas

Evaluation of contaminants in riparian and nearshore media involved developing a CSM of the riparian and nearshore environment along 100-D/H (Appendix L). This CSM addressed, on a reactor area basis, the potential for Hanford Site contaminants in soil or groundwater to migrate to riparian or nearshore areas at concentrations that could be of concern to ecological receptors. The point of departure for this CSM was the analysis of the River Corridor-wide ecological risks presented in the RCBRA ERA (DOE/RL-2007-21, Volume I). The RCBRA identified on a sitewide basis some COECs in riparian and nearshore media (soil, sediment, and water) that could warrant further evaluation. Sources, fate and transport pathways, and exposure points were identified to provide a framework for evaluating sampling and analytical data in the riparian and nearshore area. Data characterizing riparian area soils, near-river groundwater, groundwater from aquifer tube samples, pore water samples, seep samples, sediments, and surface water integrated across the 100-D/H RI/FS and the RCBRA were evaluated to determine if Hanford Site contaminants could migrate to riparian and nearshore areas at concentrations posing an ecological risk or could have been responsible for the measured concentrations observed. The spatial distributions of contaminants across the different media were compared to determine if there might be transport from onsite soils and groundwater to riparian/nearshore areas.

Concentrations at exposure points in sediment and all aqueous media (groundwater, aquifer tubes, seeps, pore water, and surface water) were compared with ecological screening levels to identify contaminants posing an ecological risk. Concentrations of some metals in pore water and sediments were higher than screening levels for aquatic plants or invertebrates; however, these appeared to be unrelated to Hanford Site sources, based on the relative distribution of concentrations between near-river groundwater and nearshore media, or comparison with reference areas. Concentrations of total chromium and Cr(VI) in pore water were higher than AWQC, and could be associated with Cr(VI) contamination in groundwater. Based on the results of the evaluation in Appendix L, with the exception of total chromium and Cr(VI), detected concentrations of contaminants in riparian or nearshore media are not reliably detectable at levels of ecological concern, or are not associated with contamination in soil or groundwater resulting from Hanford Site operations.

4.7 Biota

This section summarizes ecological sampling or biological monitoring data that have been collected for 100-D/H. Biota data are useful to understand biological receptors, which are evaluated in Chapter 7.

Biota data from two main environmental sampling projects conducted at the Hanford Site were reviewed and summarized for this section. The SESP is a multimedia environmental surveillance project conducted by PNNL. The primary goal of the SESP is to measure concentrations of radionuclides and chemicals in environmental media to demonstrate compliance with applicable environmental quality standards and public exposure limits, and to assess environmental effects. Project personnel annually collect samples of ambient air, surface water, agricultural products, fish, wildlife, and sediments. Soil and vegetation samples are collected about every five years. SESP analytical capabilities include the measurement of radionuclides at environmental concentrations. In selected media, SESP can also measure environmental concentrations of nonradiological constituents including metals, anions, VOCs, and total organic carbon (TOC). The SESP sampling design is described in *Environmental Monitoring Plan United States Department of Energy Richland Operations Office* (DOE/RL-91-50).

Fish tissue has been a part of monitoring at the Hanford Site for many years, resulting in a variety of species and fish tissue in the database of historical samples. Within the historical fish tissue dataset, there is considerable inconsistency in species evaluated, tissue type (whole body, fillet, skin on, skin off), and analytes. Additionally, multiple collection and analysis approaches, as well as variability in species life spans, are believed to have introduced significant variability in analytical results. Fish tissue sampling was part of the CRC HHRA (DOE/RL-2010-117, Volume II). The Columbia River RI Work Plan (DOE/RL-2008-11), DQO process (*Data Quality Assessment Report for the Remedial Investigation of Hanford Site Releases to the Columbia River, Hanford Site, Washington* (WCH-381) and SAP (*Sampling and Analysis Instruction for the Remedial Investigations of Hanford Site Releases to the Columbia River* (WCH-286) for the CRC (DOE/RL-2010-117, Volume II) defined a consistent sampling and analysis approach among species, tissue types, and analytes. Therefore, only fish tissue data from 2009 to 2010 were used in the CRC HHRA (DOE/RL-2010-117, Volume II); the 2009 to 2010 program focused on target fish species intended to be most representative of the exposure scenarios identified for the CRC HHRA (DOE/RL-2010-117, Volume II):

- Common carp (*Cyprinus carpio*)
- Mountain whitefish (*Prosopium williamsoni*)
- Walleye (*Stizostedion vitreum*)
- Smallmouth bass (*Micropterus dolomieui*)
- Bridgelip sucker (*Catostomus columbianus*)

- White sturgeon (*Acipenser transmonatnus*)

These six fish species are year-round resident fish that reflect a range of trophic levels and have a higher rate of harvest and consumption among the local population. As described in the Columbia River RI Work Plan (DOE/RL-2008-11), salmon were not sampled as part of this study because they spend most of their life cycle in the ocean as opposed to the Hanford Site Study Area.

For all species except sturgeon, fish tissue samples were composite samples composed of tissue from approximately five fish. Generally, five samples of each fish species were collected from each area, and each sample included separate fillet, carcass (which included the head and skeleton of the fish), and combined liver and kidney tissue for analysis. For carp, sufficient tissue mass was available to obtain separate liver and kidney samples. Fillet samples for all of these species except sturgeon were prepared with the skin on, because skin for these types of fish is often left on during preparation, and consumed. Sturgeon samples were not composited, and thus samples represent tissue from individual fish. Sturgeon fillet samples were collected with the skin off, and separate liver and kidney samples were prepared.

Biota data are also summarized from ecological samples collected to support the RCBRA ERA (DOE/RL-2007-21, Volume I). The primary goal of RCBRA is to evaluate current and potential future risks to the environment posed by releases of hazardous substances. RCBRA appraises relevant sources of contamination, exposure pathways, and contaminants for several environmental media and receptors including surface soil, vegetation, soil invertebrates, small mammals, and birds. RCBRA analytical capabilities include the measurement of radionuclides, metals, anions, SVOCs, herbicides, and pesticides at environmental concentrations, as well as physical properties (pH, moisture, particle size) in selected media. Most of the RCBRA environmental samples were collected in 2006 and 2007. The RCBRA sampling and analytical specifications are documented in the RCBRA SAP (DOE/RL-2005-42).

Figure 4-103 shows the SESP and RCBRA biota sample locations. The terrestrial plant and animal species collected and the tissues analyzed are as follows:

- Perennial vegetation: stems and leaves (combined)
  - Dominant shrub: current year's growth
  - Dominant grass: current year's growth
  - Balsamroot: leaves, roots
- Terrestrial invertebrate: whole body composites
- Mouse: whole body composites; kidney and liver (combined)
- Mule Deer: antler
- Bird: Western Kingbird organs, crop

Table H-23 in Appendix H summarizes plant tissue samples collected within 100-D/H for the SESP and RCBRA projects. Samples collected for RCBRA were analyzed for radioactive and nonradioactive constituents. The samples collected for SESP were analyzed for radionuclides and total uranium only. The table also shows a summary of plant tissue samples collected from several reference areas (unaffected areas) as a part of the RCBRA project. The reference samples were analyzed for the same suite of analytes as the RCBRA study site samples. The plant tissue sample results from the 100-D/H study sites are within the range of the results for the reference area samples.

Appendix H, Table H-24 summarizes the invertebrate tissue samples collected within 100-D/H for the RCBRA project. The samples were analyzed for metals and radionuclides only. Because of insufficient sample volumes, organic constituents were not analyzed. The table also shows a summary of invertebrate tissue samples collected from several reference areas (unaffected areas) as a part of the RCBRA project. The

reference samples were analyzed for the same suite of analytes as the RCBRA study site samples. For the inorganic analytes, the invertebrate tissue sample results from the 100-D/H study sites are within the range of the results for the reference area samples with the exception of aluminum, arsenic, total uranium, lead, nickel, silicon, and zinc, which show slightly higher concentrations. For the radionuclides, the invertebrate tissue sample results from the 100-D/H study sites are within the range of the results for the reference area samples, except technetium-99 and uranium-233/234, which show slightly higher concentrations

Appendix H, Table H-25 summarizes the mouse tissue samples collected within 100-D/H for the RCBRA project. The samples were analyzed for radioactive and nonradioactive constituents. The table also shows a summary of mouse tissue samples collected from several reference areas (unaffected areas) as a part of the RCBRA project. The reference samples were analyzed for the same suite of analytes as the RCBRA study site samples. For the inorganic analytes, the mouse tissue sample results from the 100-D/H study sites are within the range of the results for the reference area samples with the exception of boron, sodium, and tin, which show slightly higher concentrations. For the radionuclides, the invertebrate tissue sample results from the 100-D/H study sites are within the range of the results for the reference area samples with the exception of potassium-40 and radium-228, which show slightly higher concentrations.

Appendix H, Table H-26 summarizes two mule deer antler samples collected for SESP. The samples were analyzed for strontium-90 only. No anomalies were observed for strontium-90. There are no comparable reference samples for the mule deer antler samples.

Table H-27 in Appendix H, summarizes the western kingbird organ and crop tissue samples collected within 100-D/H for the RCBRA project. The samples were analyzed for metals and radionuclides only. Because of insufficient sample volumes, organic constituents were not analyzed. The table also shows a summary of bird tissue samples collected from several reference areas (unaffected areas) as a part of the RCBRA project. The reference samples were analyzed for the same suite of analytes as the RCBRA study site samples. For the inorganic analytes, the bird tissue sample results from the 100-D/H study sites are within the range of the results for the reference area samples with the exception of boron, phosphorus, and zinc, which show slightly higher concentrations. For the radionuclides, the invertebrate tissue sample results from the 100-D/H study sites are within the range of the results for the reference area samples with the exception of potassium-40, which shows slightly higher concentrations.

4.8 Air

Atmospheric releases of radioactive materials from Hanford Site facilities and operations to the surrounding region are potential sources of human exposure. On the Hanford Site, radioactive constituents in air are monitored onsite near facilities and operations, at Sitewide locations away from facilities, and offsite around the Site perimeter, as well as in nearby and distant communities. As discussed in Section 2.1.6, Hanford Site contractors monitor radionuclide airborne emissions from Site facilities through several programs. The Near-Facility Environmental Monitoring Program measures concentrations of radionuclides in the ambient air on the Hanford Site near facilities and operations. The Hanford Site Environmental Surveillance Program measures the ambient air at Sitewide locations away from facilities, around the perimeter of the Site, and offsite in nearby and distant communities. In addition, emissions from stacks, vents, or other types of point sources are monitored individually by analyzing samples extracted from the outflow at each point of release. Currently, no point source releases are associated with 100-D/H.

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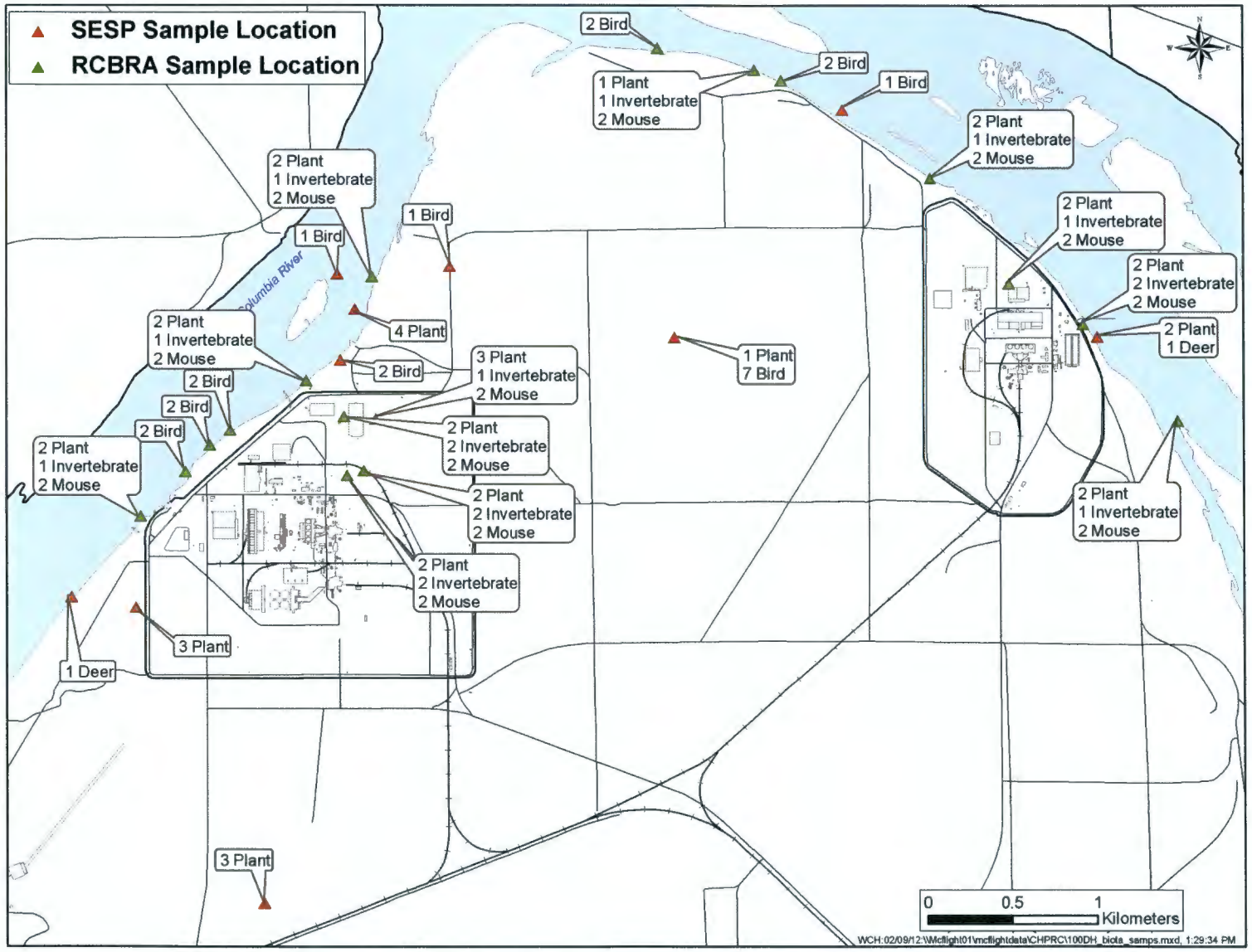

Figure 4-103. Location of SESP and RCBRA Biota Samples

The data collected by each program are used to assess the effectiveness of emission treatment and control systems and pollution management practices, and to determine compliance with state and federal regulatory requirements. Pollution management practices include controlling fugitive emissions during remediation and monitoring to ensure mitigation measures are adequate for emission control and worker exposure. Additional description of the ambient air sampling activities is available in the 2009 Sitewide Environmental Report (PNNL-19455). There were no notifications of air sample exceedances to WDOH for samples collected at 100-D/H in 2009. Air sample locations are shown on Figure 2-4.

4.9 Conceptual Site Model

The purpose of this CSM is to describe the features, events, and processes that resulted in the observed environmental contamination at 100-D/H and affect the future migration of existing contamination in soil and groundwater at the Hanford Site. The CSM relies on a comprehensive review of all available data, including field data if available, radiological surveys, process history, analogous site information, personal interviews, engineering drawings and as-builts, and any other available information. The CSM is based on the following:

- Primary sources of contamination are the liquids and solids that were used during reactor operations (Chapters 1 and 3).
- Physical features of the site, both natural and artificial, including soil, groundwater, surface water, climatic, and biologic features that affect the potential migration of contaminants and exposure to potential receptors (Chapter 3).
- Secondary sources (primarily vadose zone material or groundwater) that became contaminated as a result of releases of primary sources of contamination into the environment; these secondary sources resulted from operation and activities that have contributed to contamination of other environmental media (primarily groundwater and secondarily surface water or riparian soil) (Chapter 4).
- Description of the environmental pathways, driving forces, and transport mechanisms through which contaminants migrate from the reactor areas and associated waste sites through the ground to the river (Chapter 5).
- Potential exposure pathways and receptors for site contaminants (Chapters 6 and 7).

The resulting CSM integrates all of these elements to provide a basis for understanding contaminant fate and transport in the environment. This understanding is an important part of the RI/FS process and provides a technical basis for the description and understanding of Site conditions, assessment of the actual and potential risks posed by Site conditions, and evaluation of the need for remedial action(s). A summary diagram illustrating the multiple aspects of the contamination condition at 100-D/H is shown on Figure 4-104. As indicated in this illustration, primary contaminant sources were released by numerous mechanisms, producing varying potential secondary sources. Contaminants are subject to numerous transport mechanisms and may reach potentially exposed receptors at multiple exposure points associated with the vadose zone, groundwater, and surface water.

The approach to presentation of the CSM in this section is to synthesize our knowledge of Site conditions and operating history to provide a description for understanding the interrelations of the various contaminant migration pathways. Chapters 1 through 4 provide a framework for the conceptualization and characterization process. These results are synthesized in Chapter 5 to describe the fate of contaminants in the system as they move through the vadose zone and aquifer to the river. Chapters 6 and 7 determine the ecological and human health risks posed by these contaminants from the distribution and amounts of contaminants

present at potential exposure points. For those contaminants with an actionable risk, the CSM is used in Chapters 8 through 10 to identify appropriate remedial technologies and to evaluate remedial alternatives.

The FS identifies specific remedial technologies that are applicable to the individual contaminants in their specific conditions or locations at the site. The applicable technologies are then assembled into definable remedial alternatives that will address the combinations of contaminants and their locations to interrupt the linkages between the conceptual model elements shown on Figure 4-104. The selected remedial alternatives that have been identified are then evaluated in detail to compare their relative effectiveness in reducing or eliminating the risks posed by the site contaminants.

Target analytes in soil include Cr(VI), nitrate, arsenic, barium, total chromium, mercury, lead, carbon-14, cesium-137, cobalt-60, europium-152, nickel-63, plutonium-239, plutonium-240, strontium-90, uranium-238, uranium-233, uranium-234, tritium, other radionuclides, TCE, and total petroleum hydrocarbons. A complete list of target analytes is provided in Table 2-9. These analytes are evaluated in Chapters 5, 6, and 7 to determine if they pose a risk to HHE, based on concentrations and distribution.

Soil contaminants are found distributed over various portions of the thickness of the vadose zone, depending on the location of their initial release, the quantity of water or other liquid discharged with them, and their relative mobility in soil. Most of the metallic contaminants (for example, lead, arsenic, barium, mercury, cesium-137, and radioisotopes of cobalt, europium, nickel, plutonium, and uranium) are found near the points of historical release.

Contaminants that migrated to groundwater have developed into identified groundwater plumes. Cr(VI) is recognized as a principal COPC in groundwater at 100-D/H because of its mobility, widespread presence, and potential effect to HHE. Other COPCs include aluminum, nitrate, arsenic, barium, total chromium, mercury, nickel, lead, cesium-137, cobalt-60, europium-152, nickel-63, plutonium-239, plutonium-240, strontium-90, technetium-99, uranium-238, uranium-233, uranium-234, tritium, and other radionuclides (Table 2-11). The COPCs for groundwater are evaluated to determine if they pose a risk to HHE and should be evaluated in the FS. Nitrate concentrations in the unconfined aquifer underlying 100-D show stable trends and a declining trend at 100-H. Strontium-90 concentrations in the unconfined aquifer underlying 100-D are consistent with levels found previously in the same location, with little change. Underlying 100-H, strontium-90 concentrations in the unconfined aquifer fluctuate seasonally, but exhibit a stable plume size. Tritium concentrations in the unconfined aquifer are generally declining.

4.9.3 Physical Environment of 100-D/H

The physical environment of 100-D/H is an arid to semiarid low-elevation habitat environment located in southeast Washington State adjacent to the Columbia River. The site was the location of established farming communities in the early part of the 20th Century and was selected for establishment of the Hanford Site for production of plutonium in 1943. The site receives 17.2 cm (6.8 in.) of mean annual precipitation and supports only a low-growing shrub-steppe plant community in upland areas. The native plants are adapted to use the available moisture by rooting sufficiently deep to take advantage of moisture stored over the winter in the upper few meters of soil. The infiltration rates used in the vadose zone models are selected from the upper end of available rates based on about 30 years of field measurements (lysimeter studies) and long-term isotopic recharge studies that necessarily incorporate the effects of the history of all land surface changes at the measurement sites, including past wild fires.

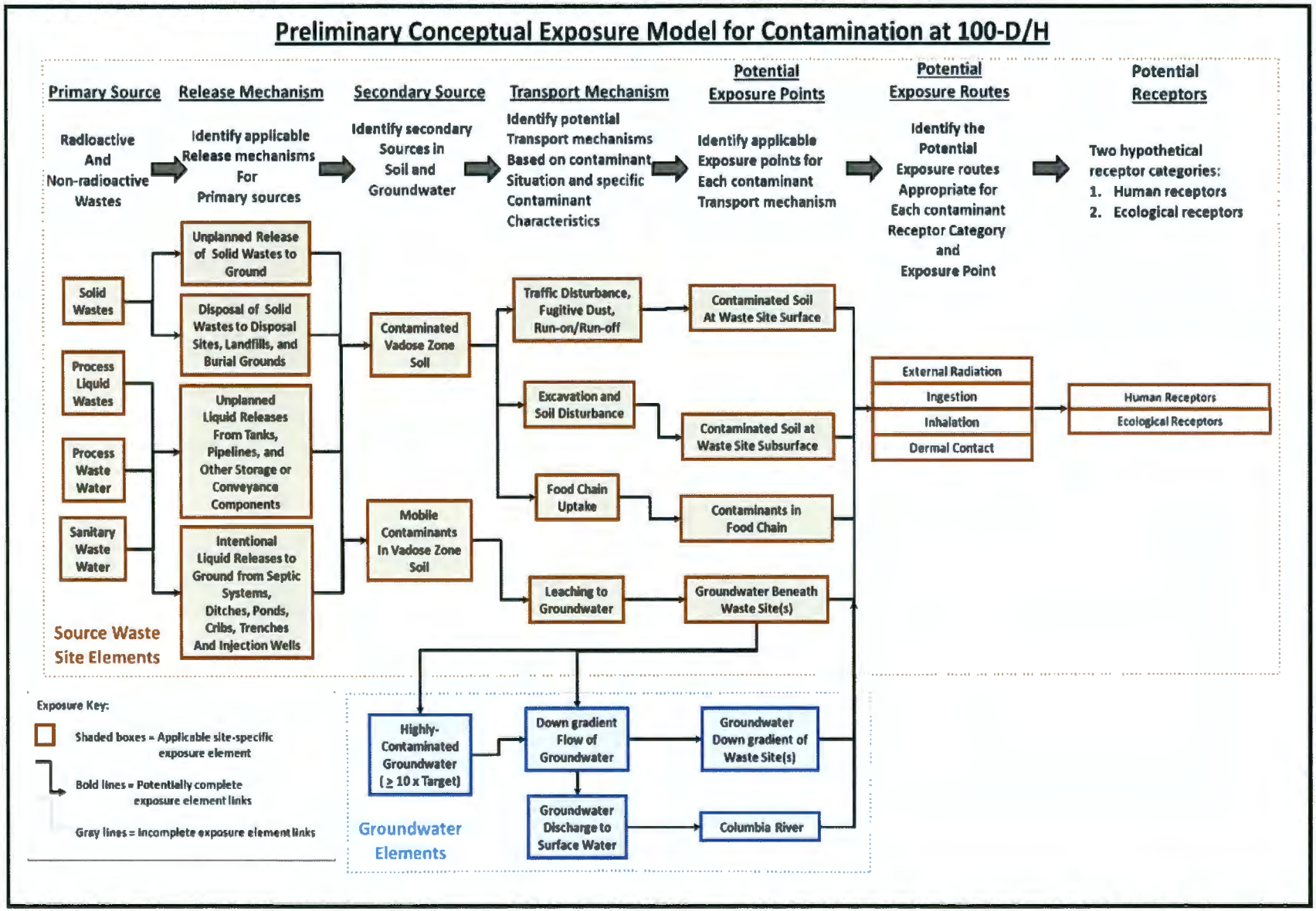

Figure 4-104. Preliminary Conceptual Exposure Model for Contamination at 100-D/H

The Columbia River, which is flow-controlled by dam operations, is adjacent to 100-D/H and has long been used as a source of drinking and irrigation water. At the Hanford Site, the river was used during reactor operations as a source of cooling water. During development of the Hanford Site operations, the previously established farming operations were razed and the heavy industrial operations to support the reactor operations were constructed.

Construction activity in the reactor areas (which is less than one percent of the Hanford Site) resulted in removal of much of the vegetation and topsoil. Much of the surface was maintained as bare gravel, and weeds were strictly controlled. Without any native vegetation or topsoil, significant portions of the rainwater and snowmelt water could infiltrate the soil unabated.

The vadose zone in 100-D consists primarily of Hanford formation gravels with portions of Ringold Formation unit E sands and gravels near the water table. The gravel-dominated facies of the Hanford formation are typically well-stratified and contain little cementation, allowing for water to pass through the material more easily than through the more cemented Ringold Formation unit E. Lenses of black sand have been identified beneath 100-D that are finer-grained than typical Hanford formation gravel. These fine-grained sand lenses tend to reduce the vertical hydraulic conductivity of the vadose zone matrix. Underlying 100-H and the northern portion of the horn, the vadose zone is dominated by the highly conductive, coarse-textured sand/gravel of the Hanford formation. These sediments are capable of draining significant amounts of water vertically, also allowing for faster horizontal water movement. Ground surface elevation in 100-D is about 143 m (470 ft) above mean sea level. The surface elevation drops to about 122 m (400 ft) above mean sea level in the central portion of the horn, and is about 128 m (420 ft) near 100-H. The current depth to the water table beneath 100-D/H ranges from about 27 m (88 ft) bgs in the central portion of 100-D to 14 m (46 ft) bgs in central 100-H, and less than 5 m (16 ft) bgs near the Columbia River and in the northern portion of the horn between the D and H Reactor areas.

The shallow unconfined aquifer is found within the Ringold unit E Formation beneath most of the 100-D operating area and within the Hanford formation at 100-H (Figure 3-8). In the intervening area of the horn, the shallow unconfined aquifer is variably within the Ringold Formation unit E and Hanford formation. This difference in aquifer matrix contributes to a higher groundwater flow velocity beneath 100-H and in the northern portions of the horn. During reactor operations, the water table at 100-D and the area of the horn near 100-D, rose into the overlying Hanford formation in some locations.

The base of the unconfined aquifer is delineated by the surface of the RUM, and undulations in the RUM surface may also affect localized groundwater flow, especially where depressions exist (Figure 3-4). The upper part of the RUM sometimes contains gravel in a silt/clay matrix that may represent a transition zone (reworked interval) above the more massive silt or clay. Within the RUM, thin sand-to-gravel lenses form zones with variable hydraulic conductivities that range from low to high. Beneath a localized area of 100-H, the first water-bearing unit within the RUM has been shown to be hydraulically connected to the unconfined aquifer (see Section 3.7.4), and could provide a pathway for contaminants to migrate.

4.9.4 Contamination Sources

Historical releases of various liquid and solid wastes were the primary sources that resulted in contamination of the vadose zone and underlying groundwater. Contamination migrated through the vadose zone to groundwater. In turn, the contaminated groundwater migrated downgradient into the Columbia River. The control of discharge of contaminants from groundwater into surface water is important to recognize as a key objective of remedial actions. Ongoing remedial actions at 100-D/H, such as the groundwater pump-and-treat systems, are actively reducing the potential for impacts to the river.

4.9.4.1 Primary Sources

The primary sources identified at 100-D/H consisted of low-volume, highly concentrated water treatment chemicals, widely variable volumes of liquid effluent discharged or released from various points in the reactor process, and solid waste. Releases took place during operations at the three 100-D/H reactors and during waste management operations that followed cessation of reactor operations in the early 1970s. The addition of contaminants to primary sources at 100-D/H ceased with the end of reactor operations. Figure 4-105 is an aerial view showing 100-D and 100-H during operations.

Figure 4-105. Aerial Photograph of D and H Reactors During Operations

Primary contaminant sources consisted of low-volume, highly concentrated sodium dichromate, widely variable volumes of liquid effluent discharged or released from various points in the reactor process, and solid waste. Specific primary sources of contaminants include the following:

- Episodic planned disposal of solid waste materials including chemical wastes, construction materials and debris, repair and maintenance wastes, and radiologically contaminated tools, materials, and reactor components (some highly radioactive and irradiated fuel fragments) placed in burial grounds.

- Episodic planned disposal and unplanned releases of liquid waste materials, including radiologically contaminated decontamination solutions associated with reactor repair and maintenance activities, off-specification or surplus water treatment chemicals, reactor cooling gas condensate, and FSB leakage. The conditioning processes added specified concentrations of chemicals, including alum, chlorine, sodium dichromate, and sulfuric acid. This category of primary source material includes spills, leaks, and wash-down of high-concentration sodium dichromate dihydrate stock solution and moderate-concentration sodium dichromate dihydrate working solution. The historical release of concentrated sodium dichromate dihydrate solution appears to account for persistent groundwater plumes near the 105-D and 105-DR Reactors water treatment facilities. Reactor cooling gas condensate releases and spent nitric acid disposal at cribs adjacent to the reactors appear to account for persistent plumes of nitrate near the reactors. Only minor amounts of nitrate are observed at 100-H.

- Groundwater contaminant plumes beneath 100-D/H are mainly attributed to the primary source materials and a limited number of confirmed, or potential, release points. Before entering the 105-D, 105-DR, and 105-H reactors, cooling water was treated for corrosion control by adding sodium dichromate working solution to the water to achieve an operating sodium dichromate dihydrate concentration of 2,000 µg/L. The nearly continuous planned disposal, as well as the unplanned releases, of large volumes of this cooling water appears to have caused the extensive plume of Cr(VI) in the unconfined aquifer underlying the horn area between 100-D and 100-H.

- Cr(VI) contamination has been observed in groundwater within the first water-bearing unit of the RUM in one well in the horn and a localized area near the river at 100-H. Groundwater mounding resulting from reactor operations at 100-D is thought to be responsible for the Cr(VI) contamination in Well 699-97-48C, which is completed in the RUM and located in the horn, but downgradient from the 116-DR-1&2 Trench. Groundwater mounding due to reactor operations at 100-H is thought to have caused the low Cr(VI) contamination in the first water bearing unit of the RUM in the 100-H area. The increased hydraulic head resulting from these groundwater mounds likely pushed contaminated cooling water into the uppermost water-bearing unit of the RUM at areas where the RUM material is more porous, and therefore more transmissive, or through erosion exposures. Contamination in this lower aquifer zone is a potential secondary source to the unconfined aquifer above where an upward gradient exists.

- Reactor cooling water was contaminated with short-lived activation products and, following reactor fuel cladding failures, with the entire suite of uranium and mixed fission and activation products present in the irradiated fuel. Releases of cooling water are responsible for most of the soil and groundwater contamination observed near the cooling water retention basins, trenches, and cribs.

- Historical septic systems and the disposal of nitric acid from reactor operations are believed to have caused or significantly contributed to the nitrate plume in the unconfined aquifer underlying 100-D and 100-H.

- Other chemical wastes generated and released at 100-D/H included the water treatment chemicals that were received, stored, and used in large volumes in the head house areas. These include strong mineral acid and caustic as well as toxic materials (for example, sodium dichromate dihydrate solution). Following the fuel cladding failure, highly radioactive liquid had to be removed from the reactor pile to recover operations. This liquid was discharged to the subsurface disposal structures.

Contaminants introduced into the environment included metals, radionuclides, and solvents (Tables 2-12 and 2-13). Soil contaminants are found distributed over various portions of the thickness of the vadose zone, depending on the location of their initial release, the quantity of water or other liquid discharged with them, and their relative mobility in soil. In some instances, contaminants that migrated to groundwater have developed into identified groundwater plumes. Cr(VI) is recognized as a COPC in soil and groundwater at 100-D/H because of its mobility, widespread presence, and potential effect to HHE. Analytes in soil and groundwater are evaluated in Chapters 5, 6, and 7 to determine if they pose a risk to HHE, based on concentrations and distribution, and should be further evaluated in the FS.

Previous actions undertaken under interim action RODs and CERCLA removal actions have addressed the environmental threats posed by majority of primary sources. Remedial actions will address any remaining primary sources but the focus is on control of contamination associated with secondary sources that may result in either direct contact exposure to identified receptors, or be released and transported to groundwater or surface water, where potential exposures may occur.

4.9.4.2 Low-Concentration/ High-Volume Waste Sites

Total chromium and Cr(VI), as well as mixed fission products, are key contaminants for this type of waste site, which includes retention basins and selected trenches such as the 116-DR-1&2 Trench and 116-H-1 Trench. These contaminants are primarily associated with cooling water, which made up the greatest percentage of the low-concentration/high-volume wastes, with process sewers being a secondary contributor.

Chronic leaks in the conveyance system piping, retention basins, and infiltration from trenches were sufficient to create substantial groundwater mounds beneath the reactor areas. The most dramatic effects of planned liquid releases occurred in 1967, when the entire cooling water waste stream from the 105-D Reactor was discharged directly to the ground via the 116-DR-1&2 Trench, resulting in a substantial

temporary increase in the magnitude of the 100-D groundwater mound present under normal operating conditions. As a result of normal operations combined with the effect of the 1967 infiltration test, the contaminated cooling water from 100-D spread over an area of about 16.6 km^2 (6.4 mi^2). The affected area includes the developed industrial portions of 100-D and 100-H areas (about 2.8 and 1.8 km^2 [1.0 and 0.7 mi^2], respectively), and about 12 km^2 (4.6 mi^2) of the horn, located between the two reactor sites.

As these large fluxes of coolant water with a low concentration of Cr(VI) passed through the saturated vadose zone and aquifer matrix, a small fraction would tend to reduce and become immobile. In the presence of acidic conditions and where the ferrous ion is available, the +6 oxidation state of chromium will reduce to its +3, or trivalent, state, which is both less mobile and less toxic. Where the ferrous ion is present with sulfide, this process will take place in both neutral and alkaline conditions. The reduction process would be expected to continue over time, lacquering a small fraction of the total flux onto the sediment, resulting in relatively large concentrations of Cr(III) in sediments associated with these low-concentration/high-volume sites.

After operations ceased, the large groundwater mound beneath 100-D that extended out across the horn, and the comparatively smaller mound below the retention basin at 100-H both collapsed as water drained down to the water table. As the groundwater mound receded, it left relatively normal groundwater levels formed by the newly emplaced cooling water. In areas of the horn near 100-D, where the groundwater mound was the highest, contaminants present in the groundwater with low to moderate K_d values could have been "stranded" in the deeper areas of vadose zone as this mound collapsed. These less mobile contaminants would remain available for downward migration to the water table, if a driving force such as water were present in sufficient amounts. Remnants of cooling water would potentially be present in the vadose zone in a volume roughly equal to the specific retention (porosity minus specific yield) or field capacity. This is approximately 2-5 percent of the total volume, depending on the sediment composition (sand versus silt versus gravel).

Key aspects of contaminant migration from low-concentration/high-volume waste sites include the following:

- Very large volumes of contaminated cooling water containing relatively low concentrations of Cr(VI) and radionuclides were discharged. Although the Cr(VI) concentration of the cooling water was low relative to the highly concentrated feed stock, the residual Cr(VI) plume across the horn still exceeds the cleanup target concentration over a large area (Table 4-13).
- A small fraction of the daily Cr(VI) flux would be reduced to Cr(III) under reducing conditions as it passed through the sediment, dropping out of solution and building up in the vadose zone and aquifer matrix. This is evident where there are large total chromium concentrations remaining relative to the remaining Cr(VI) concentrations.
- Cooling water effluent moved vertically through the vadose zone, and flowed laterally in the unconfined aquifer. These high-volume discharges created a transient water table mound beneath the reactor operating areas. Distribution of contaminants inland also occurred because of the mounding.
- Mounded cooling water effluent entered the first water-bearing unit in the RUM in a localized area underlying the 105-H Reactor, where the RUM thins and there is a hydraulic connection between this unit and the overlying unconfined aquifer.

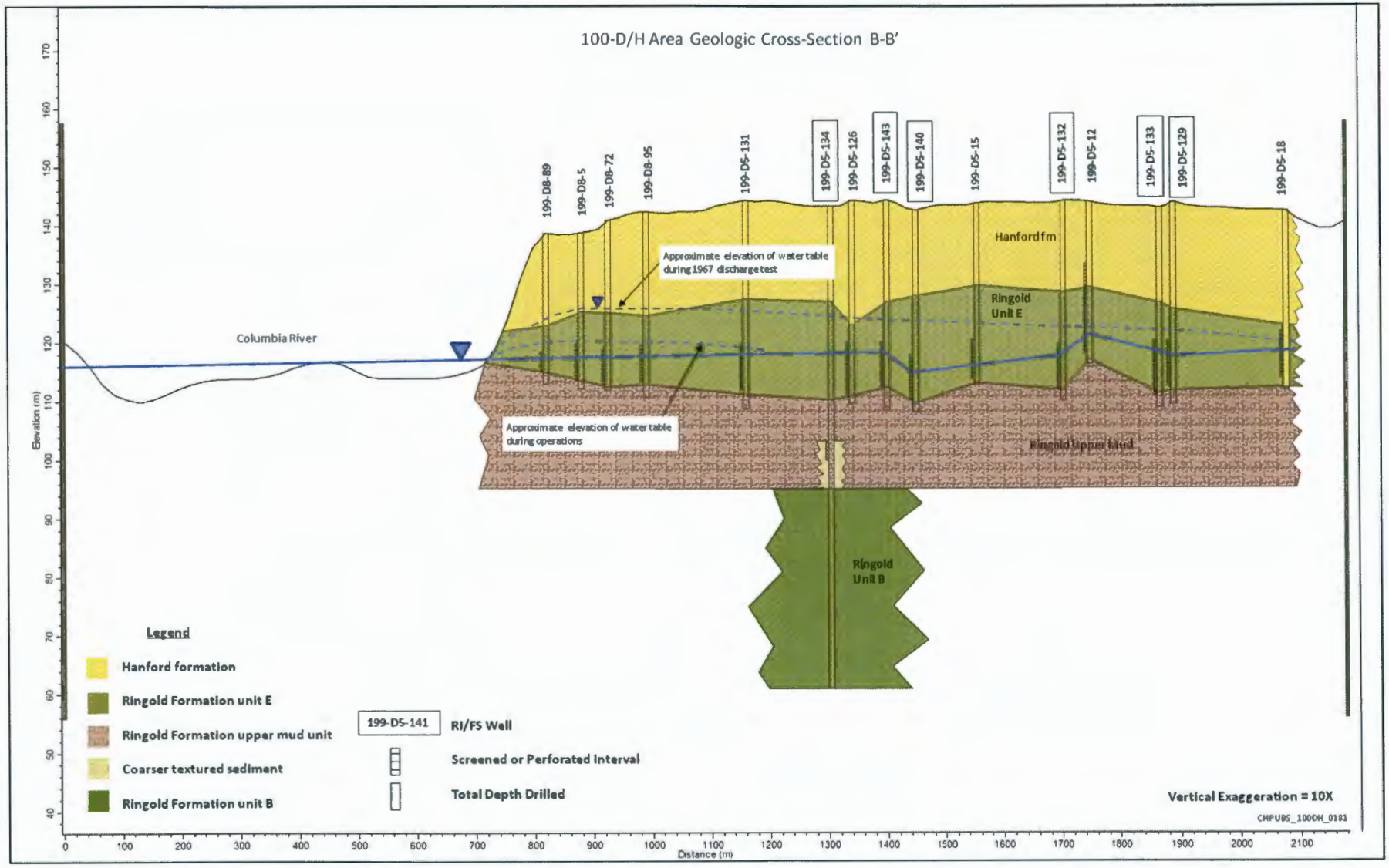

Figure 4-106. Estimated Groundwater Elevations During Operations and 1967 Infiltration Test at Cross Section B to B'

- Groundwater movement follows the path of least resistance, taking with it the contamination from the cooling water. During operations, the elevated water table at 100-D rose to a level of approximately 125 m (410 ft) amsl (*Status of the Ground Water Beneath Hanford Reactor Areas January, 1962 to January, 1963* [HW-77170]). The plan view map of the cross section is presented in Figure 4-84. At this elevation, the water table extended into the Hanford formation, above the Ringold Formation unit E at 100-D in some locations. Figure 4-106 shows a cross section to the west of the highest groundwater elevations and even in that location, the water table shows a significant elevation change. In 1967, during the infiltration test, the water table rose to over 126.5 m (415 ft) amsl (*Ground Disposal of Reactor Coolant Effluent* [BNWL-CC-1352]), well into the Hanford formation at some locations.

- The large groundwater mound at 100-D migrated across the horn, tending to move within the Hanford formation material instead of back into tighter sediments of the Ringold Formation unit E. However, where Ringold Formation unit E was present, the elevated water table would have likely remained above the Ringold to Hanford contact, allowing for contamination in those areas. Figure 4-107 presents a conceptual graphical depiction of flow from the waste sites through the aquifer matrix at variable flow velocities.

- Contaminants were transported through the vadose zone or unconfined aquifer according to their relative mobility. Highly mobile constituents, such as Cr(VI) and tritium, migrated with groundwater, while less mobile constituents such as strontium-90 migrated more slowly through the vadose zone and aquifer system.

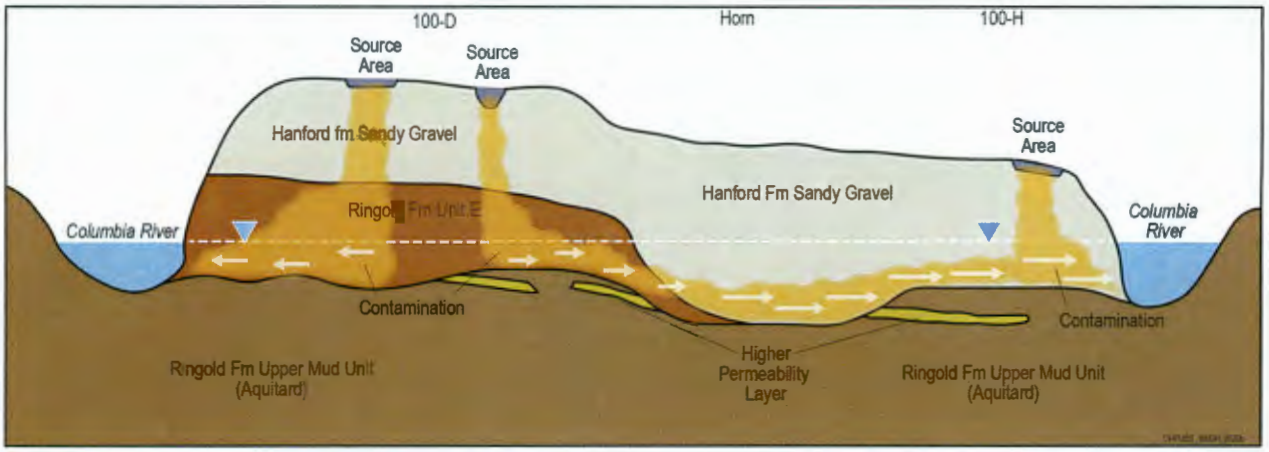

Figure 4-107. Conceptual Representation of Contaminant Flow Through Aquifer Material

4.9.4.3 High-Concentration/Low-Volume Waste Sites

The high-concentration/low-volume waste sites were liquid and solid waste disposal sites and surface spills. The routine discharges tended to be episodic and related to specific operation or maintenance functions (for example, reactor refueling, decontamination, repair activities, and deionization system regeneration). Additional release mechanisms include leaks of concentrated solutions from storage locations and conveyance systems, as well as discharges to cribs and French drains. These waste sites are significant because of high concentrations of contaminants such as Cr(VI) in the sodium dichromate feed stock.

At the 105-D, 105-DR, and 105-H Reactors, the sodium dichromate stock solution was generated and managed in several ways over the life of the reactors. Initially, sodium dichromate was procured as a crystalline solid at a purity grade of greater than 99 percent. The solid sodium dichromate was dissolved in water to make a 70 wt% solution. The 70 wt% solution was subsequently diluted to a 15 wt% working

solution, which was then pumped through pipelines to the cooling water head houses and metered directly into the reactor cooling water to achieve the final cooling water concentration of about 2,000 µg/L Cr(VI).

The highly concentrated stock solution was released to the environment through spills and washout of vehicles and containers and also through leaks in the conveyance system. These releases, most of which occurred at or near the ground surface, resulted in gross contamination of the vadose zone at many locations within 100-D/H. This includes the 100-D-100 waste site, which coincides with the 105-DR reactor sodium dichromate pipeline (100-D-56 waste site) as well as the railcar solution transfer station. The historical releases may account for potential areas of residual contamination in the vadose zone in this area. Similar conditions have been observed at the 105-H Reactor but at a much lower magnitude of contamination.

In addition to the typical high concentration liquids used at the reactors, a special case of these liquid wastes is the post operation use of the 183-H Solar Evaporation Sedimentation Basins. These basins were originally used as part of the water treatment facility. Subsequently, four of the basins were converted for use as solar evaporators, after cessation of operations at the 105-H Reactor. The neutralized nitric and sulfuric acid wastes, generated by reactor fuel fabrication processes in the 300 Area, were transferred to the open-topped basins and allowed to evaporate. The basins apparently leaked substantial amounts of waste to the vadose zone. The main contaminants in groundwater associated with the 183-H Solar Evaporation Basin are Cr(VI) and nitrate.

Key aspects of high-concentration/low-volume waste sites (many of which have been remediated under interim actions) and migration of contaminants associated with them include the following:

- Initially, dry sodium dichromate was used to prepare feedstock at the 108-D Building Chemical Pump House. The concentrated solution was pumped to tanks in the 105-D and 185-D Buildings. This preparation process was relocated in 1950 to the 185-D De-aeration Plant. Two piping lines (100-D-56) were installed between the 185-D De-aeration Plant and the 183-DR Head House for the transfer of the feedstock. Spills during product handling and leaks from piping have contributed to Cr(VI) contamination near the 183-DR Head House. Dry sodium dichromate was also used at 100-H (1949 to 1959), with a similar process as that conducted at 100-D.

- High-concentration sodium dichromate (70 wt% solution) was received at the railcar unloading site and transferred into storage tanks near the 190-D and 190-H Building (Section 1.2.2). Complete transfer of these railcar or tanker fluids into the pumping facility did not occur. Residuals were drained from the transfer hoses between the pumping station and railcars and tankers. These residuals and rinse water were discharged directly into a nearby French drain about 0.9 m (3 ft) in diameter. This was an important primary source for the present-day high-concentration Cr(VI) plumes near the 105-D and 105-DR Reactors. There is presently not a high-concentration Cr(VI) plume near the 105-H Reactor.

- The other important source for sodium dichromate concentrated solution is leakage from the pipelines that transferred the concentrated solution to the head houses upstream from the reactors, where it was input to the cooling water stream.

- Some of the high-concentration solutions may have been the result of cleaning operations at the 108-D Building Chemical Pump House where rinse water was discharged to the process sewer. The process sewer discharged directly into the Columbia River and later to the 116-DR-1&2 Trench. One likely scenario is that at the end of operations, concentrated solution was cleaned out via the process sewer, ending up in the 116-DR-1&2 Trench. Therefore, discharges to the 116-DR-1&2 Trench may have been both low-concentration (cooling water) and high-concentration solutions.

- Sulfuric acid was used at 100-D/H primarily as a decontamination solution, with some used in water treatment. Mercury-contaminated commercial-grade sulfuric acid was used for cooling water pH adjustment at 100-K (1968 to 1977) and it is possible that mercury contaminated acid was also used 100-D and 100-H. Although this period was after the shutdown of the 100-D/H reactors, mercury contamination in soil associated with sulfuric acid has been identified during the remediation of the 100-D-77 waste site at the 183-DR Head House and at 100-H-44, near the 183-H Head House. Since other locations with acid staining that did not have mercury have also been identified, however, it is not a certainty that the acid was cross-contaminated.

- 100-H historically exhibited some mercury contamination in groundwater, apparently related to the use of the 183-H Settling Basins as a solar evaporation pond for chemical waste following cessation of reactor operations. In addition, mercury contamination of sulfuric acid has been identified at the 183-DR Head House and may be associated with the 183-H Head House.

- The 100-D/H nitrate groundwater plumes originate near each reactor and the sources are likely related to reactor operations. The likely source of nitrate may include historical use of nitric acid-based solvents in the reactor buildings, including laboratory areas, as a decontamination solution. Another reactor operation-related source is from oxidation of ammonia discharged in the condensate solution. Some additional contribution from septic sources is possible. Detailed analysis of the vadose zone profile at 1607-H-4 in the RI did not find nitrate above background (52 mg/kg).

- Elevated sulfate in groundwater at 100-D/H has originated from multiple sources. Sulfuric acid was primarily used at 100-D/H as a decontamination solution, with some possibly used to adjust cooling water pH. Sulfuric acid was also used to produce alum from bauxite ore to be used in cooling water treatment. The highest sulfate concentrations currently are found near the ISRM barrier at 100-D near 100-DR Reactor and result from the oxidation of the sodium dithionite reductant used to establish that reactive treatment zone.

4.9.5 Secondary Sources

Secondary sources are the environmental media (for example, soil, surface water, and groundwater) that were affected by the initial releases of primary sources and, subsequently, retained sufficient levels of mobile contamination to function as continuing sources of contamination to adjacent soil, surface water, groundwater, and/or air.

The historical releases of primary contaminant source material to the environment resulted in contaminated vadose zone material beneath facilities and waste sites and contaminated groundwater. The resulting contamination, with varying mobility, is subject to leaching to groundwater, to transport by surface run-on or run-off, and/or to transport by wind as particulates. Surface run-off and wind are not considered important pathways contributing to current contamination levels (see Appendix K). If not remediated, this contaminated material acts as a secondary source with potential for the further spread of contaminants through the environment and potential exposure to human and ecological receptors. The main secondary source of concern at 100-D/H is vadose zone soil, including the PRZ, and possibly in low-conductivity zones of the unconfined aquifer. While Cr(VI) is the main secondary source of concern, other COPCs also may be present in these zones.

Chapters 6 and 7 discuss the evaluation of risks posed by the identified secondary sources to human health and the environment, respectively, through direct exposure. The potential for secondary sources to provide a significant ongoing source of contamination to groundwater is evaluated through the comparison of contaminant EPCs in vadose zone materials to the SSLs and PRGs (Section 5.6) protective of groundwater and surface water.

Waste remediation has been oriented toward removal of secondary sources of contamination at waste sites (Section 1.2.2). Confirmation sampling and RI characterization data (Section 4.3) indicates that cleanup goals have been achieved within the vadose zone; however, groundwater monitoring indicates that the potential for residual contamination in soil exists. During high river stage, groundwater rises into contaminated vadose zone materials, increasing the rate at which contaminants are leached to the groundwater and causing a temporary peak in concentrations. Contaminants with higher K_d values would also tend to leach, but at significantly lower rates, as discussed in Chapter 5.

Based on the historical and current presence of specific groundwater plumes in 100-D/H, the following general locations are potential areas of residual contamination in the vadose zone that may contribute to groundwater contamination:

- Groundwater monitoring around the 105-D and 105-DR reactors indicates potential for residual contamination in these areas.
  - The FSB leaks, disposal cribs, and trenches were historical sources of Cr(VI) and mixed fission product (i.e., strontium-90, tritium, and cesium-137) contamination. Strontium-90 and Cr(VI) groundwater contamination remains in these areas and indicates potential for residual contamination in the vadose zone.
- Groundwater monitoring around the high concentration sodium dichromate off-loading, mixing, and delivery system.
  - Chromium contaminated vadose zone soil underlying the sodium dichromate dihydrate solution transfer facilities are potential areas of residual Cr(VI) contamination. These areas include the vicinity of the 100-D-100 waste site and other related conveyance systems.
- High volume/low concentration cooling water disposal areas are potential areas of residual contamination contributing to groundwater.
  - Various cribs and trenches (such as 116-D-1A, 116-D-1B, and 116-D-4) received low concentration, high volumes of contaminated cooling water. Other associated waste site areas include the 116-DR-1&2 Trench and the 116-D-7 and 116-DR-9 retention basins.
  - Groundwater beneath the footprint of the 116-DR-1&2 Trenches, associated with the former groundwater mound, likely accounts for most, if not all, of the residual groundwater Cr(VI) across the horn. The collapse of the groundwater mound likely resulted in contaminants remaining in the vadose zone, which may serve as a residual source to groundwater.
  - Areas at 100-H that were historical sources of Cr(VI) include the 116-H-7 retention basin and various cribs and trenches. Monitoring in these areas has not indicated the presence of residual contamination.
- The 183-H solar evaporation basin (116-H-6) area may contain residual contamination in the vadose zone.

No areas of high concentration Cr(VI) contamination are observed in the unconfined aquifer beneath 100-H. This indicates that the significant secondary sources, areas that received high concentration stock solutions or where spills may have occurred, have been remediated and residual contamination is not present.

In addition, the interim action pump-and-treat remedy made significant progress at 100-H in cleaning up Cr(VI) derived from the 105-H Reactor in the unconfined aquifer. Additional small areas of this contamination remain and additional Cr(VI) is migrating in groundwater from the horn area into 100-H.

4.9.6 Release Mechanisms

Primary release mechanisms are the processes during operations that resulted in the initial distribution of contaminants to the environment. Secondary release mechanisms are the processes that result in the redistribution of secondary source contaminants to other environmental media.

4.9.6.1 Primary Liquid Waste Release Mechanisms

The primary release mechanisms of liquid wastes at 100-D/H fall into two general categories: intentional or planned releases and unplanned releases. As discussed previously, the two groups of liquid wastes are high-volume/low-concentration liquids and low-volume/high-concentration liquids. These types of discharges were directly related to reactor operations, with discharges generally being released to various cribs, trenches, retention basins, and other engineered structures. Occasionally, planned releases allowed for discharge directly to the ground surface. For example, contaminated reactor cooling water during upset conditions, reactor cooling gas condensate, and captured FSB leakage were released directly to the land surface and allowed to infiltrate. Other planned releases at 100-D/H that appear to have made substantial effects on vadose soil and groundwater include releases to numerous French drains, cribs, and trenches.

Unplanned releases were primarily from leaks of the retention basins, but also included releases from tanks, spills, and leaks from conveyance systems. Liquid wastes were also released from pipelines through leaks at joints or material failure between joints as a result of corrosion or other damage.

The timing aspects of the liquid source releases range from episodic, short-term releases (for example, spills or transfer leaks at the head house storage areas) to regular periodic releases (for example, gas dryer condensate releases, fuel storage basin overflows, and sedimentation basin flushes) and near-continuous discharges of spent reactor cooling water.

4.9.6.2 Primary Dry Waste Release Mechanisms

Contaminants associated with dry solid waste were released to the environment through intentional disposal at waste sites or through unplanned releases of particulate material. The contaminants may transfer to the environment through leaching or dissolution. Dry granular or crystalline chemical products or contaminated soil particulates may also become windborne, suspended in surface run-off, or transferred to the surface through physical contact with a contaminated surface. These releases were evaluated during that waste site discovery process and data indicate that airborne contaminants are not significant (Appendix K). Intentional/planned releases of solid waste are believed to account for the large majority of historical dry waste releases to the environment. Dry contaminants also include powdered Cr(VI) that may have been spilled during operations and was subsequently either swept up or washed down.

Solid wastes were typically disposed through burial in landfills and burial grounds as a planned release. Unplanned releases of solid wastes included spills of dry sodium dichromate or other solid chemicals. Waste site remediation is ongoing and is removing these types of wastes.

4.9.6.3 Secondary Source Release Mechanisms

Contaminated material that remains in the environment is considered a potential ongoing secondary source of some contaminants released to air, groundwater, surface water, or to the riparian zone. Secondary sources remaining in pipelines and control structures in the form of pipe scale, corrosion products, sludge, and sediment may be released through structural failure of the pipeline and exposed to net infiltration. The following secondary release mechanisms for contaminants are grouped by importance, with some being present as both historical and current mechanisms:

Historical release mechanisms with a minor contribution to the environment:

- Volatilization of contaminants in near-surface soil to the atmosphere or soil gas (applicable to VOCs and tritium)
- Transport of contaminants from surface soil in surface water run-off, both as dissolved constituents and suspended particles (applicable to all contaminants)

Historical release mechanisms with more significant potential to affect the environment:

- Desorption and/or dissolution from contaminated vadose zone soil and transport with infiltrating precipitation or other water sources (applicable to soluble and mobile contaminants)
- Direct release of reactor process-related chemicals

Current release mechanisms with minor potential to effect the environment:

- Resuspension of particulates in air (applicable to all contaminants) from contaminants at the soil surface
- Transport of contaminants from surface soil in surface water run-off, both as dissolved constituents and suspended particles (applicable to all contaminants)
- Biotic uptake (applicable to soluble and mobile contaminants located in the shallow vadose and riparian zone) and translocation in plants and animals

Current release mechanisms with more significant potential to effect the environment:

- Desorption and/or dissolution from contaminated vadose zone soil and transport with infiltrating precipitation or other water sources (applicable to soluble and mobile contaminants)
- Groundwater discharge to surface water and to the riparian ground surface when contaminated groundwater discharges in seeps and springs associated with seasonal high river stages (applicable to soluble contaminants contained in groundwater)
- Groundwater discharge to surface water through upwelling into the river
- Contaminants with a high K_d held within the PRZ, which may be released by dissolution, ion exchange, or advective flow when groundwater elevations periodically re-enter this zone
- Desorption and/or dissolution from contaminated soil within the saturated material below the water table as a result of groundwater fluctuations and flow

4.9.7 Transport Mechanisms

The driving forces of contamination are either artificial or natural. The artificial forces during operations were related to the reactor operations and waste disposal practices, including the large groundwater mound at 100-D and the smaller mound at 100-H. The practice of disposing high volumes of liquid waste has contaminated the vadose zone and groundwater. Maintaining safe work conditions during remediation by applying water to control dust is postulated to have been a transient driving force. However, the long-term driving force is the natural system, as described by the hydrologic cycle.

The hydrologic cycle plays an important role in the CSM. Most of the precipitation occurs during the fall and winter months, when evaporation and plant use are the lowest. This water is stored in the upper few meters of the soil column and is available for plants during the dry summer months. A small fraction of water may percolate below the root zone, where it will continue to drain essentially undisturbed vertically through the vadose zone to the water table.

At 100-D/H, the groundwater currently flows toward the Columbia River, which forms the discharge boundary for the shallow unconfined aquifer. The transition area between the aquifer and the river is called the hyporheic zone. The Columbia River is free-flowing through 100-D/H and river stage, which can vary as much as 3 m (10 ft), and is controlled by the Priest Rapids Dam. When the river rises, the river water pushes into the riverbank, pushing back on the aquifer and causing the water table to rise in the nearby aquifer. When the river stage drops, groundwater in the aquifer flows again into the river.

In addition to discharge of groundwater to the river through the hyporheic zone, groundwater seasonally discharges in springs or seeps at elevations above the river stage. This occurs generally during the period following seasonal high river stage in the early summer. As the river stage recedes after the spring thaw, groundwater that has become elevated as it equilibrated with the high river stage may drain directly to the ground surface in the riparian zone.

4.9.8 Contaminant Migration

Contaminants migrate along flow paths. Understanding of the flow path for contaminant migration is important in the development of the remedial alternatives in Chapters 8 and 9. The goal of a remedial alternative or combination of alternatives is to break the flow path and isolate or remove the contaminant. The major components of the flow path are illustrated on Figure 4-108. The upland zone to the right of the figure is the location of the reactors and facilities and associated waste sites. Once contaminants entered the ground through leaks at basins or pipes, planned releases at cribs and trenches, and other unplanned releases, the contaminant fluids combine with the ambient water already in the vadose zone soil plus precipitation and begin to leach down toward the water table.

Excavation of waste sites is intended to remove leachable material before it affects the groundwater. This type of remediation removes contaminants from the surface and near-surface soil, thus reducing, or eliminating, the potential for direct-contact exposures, or migration of these near-surface contaminants in surface water run-on/run-off or as wind-blown suspended particles. In some instances, the more mobile constituents, particularly Cr(VI), tritium, and nitrate, have already migrated deep into the vadose zone and/or into the underlying groundwater.

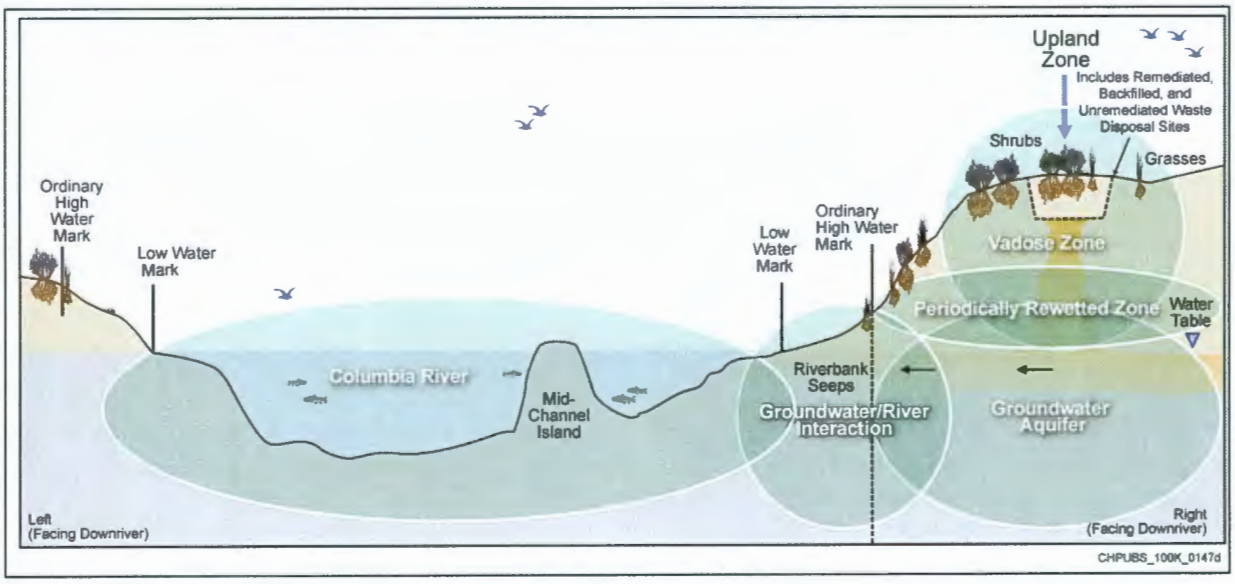

Figure 4-108. Schematic of Areas of Contaminant Interaction with Vadose Zone Soil, Groundwater, and Surface Water

Contaminants in 100-D/H include highly mobile constituents that do not adsorb readily to the geologic materials in either the vadose zone or the aquifer. These contaminants include Cr(VI), nitrate, sulfate, and tritium, which migrate readily with water. Strontium-90 has been found in the deep vadose zone and in groundwater, but tends to adsorb readily to the geologic materials in the vadose zone and aquifer and does not move much further under natural conditions. In the presence of acids, strontium-90 and other metals would be more mobile, but the buffering capacity of the sediment would neutralize the leachate and reverse the process.

Where mobile contaminants remain in the vadose zone, precipitation or other water sources have the potential to mobilize those contaminants. Although there is no evidence that residual wastewater from operations is still present to act as a driving force, both precipitation and dust suppression water may contribute to water infiltrating through the vadose zone. As seen during interim remedial action activities, such as at waste sites 100-D-100, 100-D-30, and 100-D-104, highly mobile contaminants such as Cr(VI) can be left in place throughout the vadose zone from near the surface to the water table, unless mobilized. As contaminants are driven down through the vadose zone, which is about 26 m (85 ft) thick in 100-D, they would do so in pulses associated with the presence of water, the driving force. The concentrations of residual contaminants at remediated sites, potential secondary sources, were evaluated in their respective CVP reports and reported to be below the applicable cleanup criteria.

Residual contaminants that remain in the vadose zone after the cessation of waste discharges can migrate downward by any of four mechanisms:

- They may continue to move by gravity drainage of residual wastewater within the vadose zone (this process is not believed to be continuing at this time).
- They may be mobilized in the fraction of annual precipitation that actually percolates deep into the vadose zone to recharge into the aquifer.
- They may be mobilized into groundwater from the vadose zone during seasonal increases in groundwater table elevation resulting from high river stages.
- They may be mobilized in water added for dust control during remedial actions (for example, excavation) and migrate deeper into the vadose zone.

Other factors that affect contaminant migration include persistence and chemical changes, including radioactivity. These factors relate directly to the way the individual constituent reacts in the environment.

The persistence of various contaminants determines how long they are available to the environment and for transport to the different receptors. If a contaminant remains in the environment for a long time, and is highly mobile, it is more likely to be transported from the vadose zone to the groundwater, and eventually to the surface water. Persistence is defined by how long it takes a particular contaminant to be transformed into a less toxic or less available form, or how long it takes the contaminant physically to leave the affected area. Radionuclides undergo radioactive decay at varying rates specific to the individual nuclide. Chemicals may also degrade, decay, or undergo chemical transformation that reduces the residual mass of the contaminant available for transport or direct exposure.

Chemical Changes

Several constituents also may be altered into a different valence state as a result of the chemistry of the water or lithologic composition of sediments in which they are present, resulting in a change to the mobility. For example, Cr(III) adsorbs and precipitates out of solution and becomes immobile. The chemistry of sodium dichromate is important for this reason.

The geochemistry of sodium dichromate is important. Chromium is typically present in the environment in one of two oxidation states (chromium(III) or Cr(VI)).When chromium is in the hexavalent state (with a +6 valence), the chemical form is present as a soluble oxyanion, either as the dichromate anion ($Cr_20_7^{2-}$) or chromate anion (CrO_4^{2-}), depending primarily on pH. The dichromate anion is dominant in acidic conditions and the chromate anion is dominant in alkaline conditions. Chromium(III) is typically precipitated as a low-solubility hydroxide molecule, $Cr(OH)_3$, and has low mobility. Most soil types, including those found at the Hanford Site, tend to be negatively charged as well, so there is no significant force of attraction between the chromium anions and the sediment, such that typically the adsorption is assumed to be very low for dichromate passing through the sediment. However, at locations where iron and bacteria are available to react chemically and biologically with the dichromate anion, reaction occurs and immobile chromium(III) forms can precipitate out of solution. Chapter 5 discusses in detail the fate and transport for contaminants in 100-D/H.

The ionic forms of Cr(VI) are relatively stable at the oxidation state typically found in soil and groundwater at 100-D/H and the constituent tends to remain mobile. The source of the Cr(VI) in the environment was the sodium dichromate used for corrosion control in reactor cooling water. This compound is acidic in its concentrated form. However, the dichromate, or chromate, ion can react with other metals in the environment to form compounds of lesser solubility. These compounds can include potassium dichromate (which is about one tenth as soluble as sodium dichromate) and lead chromate (which is essentially insoluble in water). The Cr(VI) ions can also be subject to chemical reduction under moderately reducing conditions, or upon reaction with reducing agents such as ferrous iron. This reduction appears to be the case in both soil and groundwater. Ferrous iron is very effective at reducing Cr(VI) to Cr(III), producing a very low-solubility hydroxide molecule.

In groundwater, where iron, hydrogen sulfide, and bacteria are available to react chemically and biologically with the dichromate anion, reaction occurs and immobile Cr(III) forms can precipitate out of solution. Ongoing research by Lawrence Berkley National Laboratory, PNNL, and the University of Oklahoma as a joint project on reduction of Cr(VI) by microbial communities indicates that three different bacteria are present at 100-H that are capable of reducing Cr(VI) anoxically (*Microbial Community Changes During Sustained Cr(VI) Reduction at the 100H Site in Hanford, WA* [Chakraborty et al., 2010]). Other research ("Enhanced Microbial Reduction of Cr(VI) and U(VI) by Different Natural Organic Matter Fractions" [Baohua and Chen, 2003]) indicates that the presence of microbial activity for reduction of metals is also dependent on the pH and humic acid present. Section 5.6, Contaminant Persistence, discusses chemical reduction of chromium in more detail.

Radionuclide Decay

The primary radionuclides associated with reactor operations that resulted in vadose zone and/or groundwater concerns at 100-D/H are strontium-90 and tritium. The half-lives of these radionuclides are presented in Table 4-25.

Table 4-25. Half-Lives of Select Radionuclides

Analyte	Half-life*
Strontium-90	28.79 years
Tritium	12.32 years

* Half-lives were obtained from the Radiochemistry Society (RS, 2011) website in Feb. 2012.

4.9.8.1 Vadose Zone

Historical contaminant migration at 100-D/H was driven by the release of large volumes of reactor cooling water to the ground surface, along with natural forces. With the cessation of operations, most discharges and releases ceased. However, contamination may continue to migrate under the influence of the hydrologic cycle and continue to interact chemically with the sediment matrix. The following are key features of the fate and transport:

- Waste site remediation has focused on reducing risk posed by direct contact exposures and achieving groundwater and surface water protection. These actions frequently included excavation of contaminated soil to remove contaminants that may pose a threat to groundwater. Although direct contact exposures are generally assessed within the upper 4.6 m (15 ft) of the vadose zone, the interim remedial actions were implemented to achieve defined RAGs and were not constrained to a specified depth of remediation (may have been shallower or deeper than 4.6 m [15 ft]). Sources contributing the majority of water to the surface during operations were related to reactor operations and waste disposal practices, including the release of varying quantities of wastewater to the vadose zone. Large quantities of contaminated cooling water were discharged to the ground at both 100-D and 100-H areas during reactor operations. Precipitation, dust suppression, and leakage from the FSBs and retention basins became the driving forces for contaminant movement through the vadose zone. Net infiltration from precipitation (recharge) on the nonvegetated surface soil would have been about 17 mm/yr (0.67 in./yr). Dust-suppression water was used during demolition and remediation of waste sites; this may have produced local recharge transient events. Once waste sites are revegetated, the plants transpire most of the natural precipitation, limiting infiltration deep into the vadose zone. Revegetation of waste sites at 100-D/H is variable and is expected to accelerate after completion of remedial activities.

- Some of the contaminants may remain dissolved in the pore water of the vadose zone material, but the mass in this phase is likely to be low (given the relative small volume of water) and slowly leachable (because it is located in smaller pores).

- Known chemical reactions within the vadose zone can reduce Cr(VI) to its less toxic and less mobile trivalent state in conjunction with sorption and precipitation ("Factors Affecting Chemical and Biological Reduction of Hexavalent Chromium in Soil (Losi et al., 1994). Chromium reduction by ferrous iron and chemical precipitation with barium sulfate are two reactions that occur; iron constitutes approximately five percent of the composition of both the Ringold Formation unit E and the Hanford formation.

- Chemical analysis of the vadose zone material at RI wells indicates that much of the chromium in the vadose zone is in reduced form. This is consistent with known attenuation mechanisms for Cr(VI) described by *Monitored Natural Attenuation of Inorganic Contaminants in Ground Water Volume 2 – Assessment for Non-Radionuclides Including Arsenic, Cadmium, Chromium, Copper, Lead, Nickel, Nitrate, Perchlorate, and Selenium* (EPA/600/R-07/140).

- Cr(VI) was discharged into the surrounding environment as a dissolved ionic species in various liquids. The historical records information described in a previous section shows Cr(VI) was released into the environment primarily as dissolved sodium dichromate dihydrate in two types of solutions: the low concentration reactor coolant and the high concentration 70 weight percent stock solution used to make reactor coolant. The differences in solution chemistry, associated production facilities, and discharge locations have had a substantial effect on current Cr(VI) distribution in the subsurface.

- With regard to ultimate Cr(VI) distribution in the environment, the significant solution properties are Cr(VI) concentration, pH, and specific density. The approximate Cr(VI) concentration was 466 g/L in the 70 percent by weight solution. This solution was acidic (pH about 1.5) and significantly more dense than water (specific gravity of 1.7 g/cm^3). The main aquifer contamination from this stock solution appears to have originated near 100-D-100, 100-D-102, and 100-D-104.

- As described in Chapter 1, sodium dichromate dihydrate solid ($Na_2Cr_2O_7$-$2H_2O$) and 70 wt% sodium dichromate-water solutions were delivered to 100-D/H. The dry material was received in bags and/or drums at 100-D from 1944 until 1959, using the 108-D Building until 1950, and then the 185-D Building. Shipments of 226.8 kg (500 lb) drums of solid sodium dichromate dihydrate were received and stored at the 185-D Building from 1955 until 1959. Based on historical information for the 1713-DA Essential Materials Warehouse, supplies of 45 kg (100 lb) bags of solid sodium dichromate dihydrate also may have been stored at the 1713-DA Essential Materials Warehouse from 1944 until about 1955. It is not known when the 1713-DA Building was removed, but it was not seen in aerial photos after 1955. The shipments of bags and drums of solid sodium dichromate dihydrate were replaced with shipments of 70 wt% sodium dichromate water solutions beginning in 1959 and continued until the 105-D Reactor was shut down in 1967. In 1959, a tank truck/railroad car Unloading/Transfer Station (100-D-12) was installed adjacent to the railroad spur between the 183-D and the 183-DR Water Treatment Plants. The concentrated sodium dichromate solutions were transferred by hose from railroad cars or tanker trucks to the pumping facility (*100-D Area Technical Baseline Report* [WHC-SD-EN-TI-181]).

The bag-mixing process in the 190-H Building used solid sodium dichromate from 1949 to 1959 and 70 wt% sodium dichromate solutions from 1959 to 1965. In 1959, a 56,781 L (15,000 gal) horizontal storage tank was installed in the 190-H Building to receive, store, and supply a 70 wt% sodium dichromate solution to the batch mixing tanks also located in the 190-H Building.

The delivery of the 70 percent solution into the storage tanks was not completely efficient, and yellowish-stained soil around the storage tank location indicated losses to the subsurface. In addition, some leakage in the transfer pipes or connection between the transfer pipes and the mixing tanks is plausible. The fraction of delivered 70 percent solution lost to the subsurface is not known at either 100-D or 100-H.

- Following discharge of these concentrated Cr(VI) fluids into the subsurface, vertical migration occurred. The density of the fluid would have facilitated vertical migration into the subsurface with little lateral movement. However, very little information is available that describes the initial distribution of Cr(VI) from this fluid in the subsurface; and several factors suggest a broad range of possibilities. The vadose zone and the unconfined aquifer are about 25 m (82 ft) and generally 8 to 12 m (26 to 39 ft) thick, respectively, near 100-D-100. Column studies conducted by PNNL show an initial large fraction of Cr(VI) leaching in the first pore volume followed by diminishing returns, as the additional leachate contains increasingly smaller fractions tending asymptotically to zero.

- The current RTD remediation strategy in the vadose zone appears to be protective once contaminated soils are removed from the affected waste sites. However, the potential remains for residual contamination within unremediated portions of the vadose zone, particularly near historical release points that produced groundwater contamination. Undefined secondary sources could gradually leach into the groundwater for a number of years and will be monitored in the groundwater. There remains uncertainty in how this residual contamination might behave within 100-D/H.

The practice of disposing high volumes of liquid waste to waste facilities not only created a groundwater mound but also, because the water was contaminated with Cr(VI) and fission products, caused widespread distribution of contaminants underlying 100-D/H.

Residual contaminants with low K_d values (less mobile) may be retained within the vadose zone at, or near, historical release points. These contaminants may be mobilized and reach the water table under the driving forces of seasonal precipitation recharge or transient anthropogenic recharge events. The 100-D/H Work Plan identified several locations where the source areas in the shallow vadose zone had been remediated but where the potential remained for residual contamination below the depth of remediation. Several characterization boreholes and wells were drilled at these locations to evaluate the presence of residual contamination; these results indicated no significant residual contamination at the locations where data gaps and uncertainty were identified. Following are potential locations for this type of source:

- Cr(VI)-contaminated vadose zone soil underlying the sodium dichromate solution transfer facilities associated with the water treatment chemical handling at 105-D, 105-DR, and 105-H Reactors. Specifically, these areas of interest are near the 108-D Building and its associated liquid waste cribs, the overhead and underground pipelines that transferred concentrated sodium dichromate solutions between 108-D Building and the 183-D and 183-DR Head Houses, and the former railcar unloading station identified as 100-D-12. These areas are likely to have contributed to other constituents present in soil and groundwater that are not identified as COPCs (for example, sodium, aluminum, and sulfate). Remediation activities near these facilities have identified contamination present throughout the entire thickness of the vadose zone.

- Vadose zone soil underlying the various liquid process waste disposal facilities at the 105-D, 105-DR, and 105-H Reactors (that is, cribs and trenches).

- Vadose zone soil beneath the reactor cooling water retention basins and discharge facilities at 100-D/H. The soil may contain residual Cr(VI), fission products, and other cooling water-related constituents.

- Vadose zone soil underlying the fuel storage basins and the basin leak disposal cribs/injection wells, contaminated with Cr(VI) and mixed fission products (for example, strontium-90, tritium, Cs-137) (presented in Section 1.2.2). This is supported by the continued presence of the strontium-90 plume near the fuel storage basin.

Based on observations and measurements of site-related contaminants, contaminated groundwater beneath the 100-D/H Reactor areas may be a pathway of contamination discharging to the river and to the riparian/river shore area during high river stage. Prevention of the pathway from groundwater to surface water is a key consideration in the FS.

4.9.8.2 Periodically Rewetted Zone

The PRZ is the portion of the vadose zone and aquifer system that lies between the seasonal high and low groundwater elevation levels. This zone has the potential to function as a secondary source for some contaminants, particularly those that exhibit K_d values greater than 1 ml/g. Contaminants may enter the PRZ under two common conditions: by downward migration from an overlying vadose zone source, or by emplacement from contaminated groundwater during high water conditions, where the contaminant(s) may be retained by the soil matrix in that zone, and then re-enter the groundwater at the next high water period. Contaminants within the PRZ may be retained by various mechanisms, including ion exchange processes, or simply by retention of contaminated water within small pore structures by capillary action when water levels decline.

As groundwater rises into the PRZ during high water, groundwater may contact contaminants that are present in the overlying soil. As the water table falls, the contaminant may leach out of the soil and migrate into the aquifer below. This rewetting action thereby allows for potential ongoing release of contaminants to the groundwater. The rate of downward migration through the vadose zone depends on the K_d value of that COPC and the presence of driving forces, such as water. This action essentially replenishes the contaminants in the PRZ until remediation activities remove the overlying source. Assuming that contamination is present in the overlying soil, this action can result in two different situations: immediate or nearly immediate response to the change in water level, or a delayed response.

Examination of the time series of Cr(VI) concentrations, a highly mobile contaminant, and the associated groundwater elevation hydrograph for a well near the highest-concentrated portion of the plume underlying 100-D (i.e., Well 199-D5-99) shows the effect of this rewetted zone. In Well 199-D5-99, the Cr(VI) concentration reaches its maximum transient concentration in the period following seasonal maximum transient groundwater elevation (Figure 4-109). Groundwater rising into the overlying zone of soil contamination increases the rate at which contaminants are leached to the groundwater, causing a temporary peak in concentrations. Contaminants with higher K_d values would also tend to leach, but at significantly lower rates, as discussed in Section 5.

The delayed response shown by Cr(VI) at Well 199-D5-99 is a result of both the distance of the well from the river, and the fact that the water table is present within the less permeable or transmissive Ringold Formation unit E, which slows the response. In addition, at average water table elevations, the lower zone would be flushed of contaminants on a regular basis meaning that only higher water periods would introduce the overlying contaminant into the groundwater. At 100-H, this response is also present (as in Well 199-H4-4), even though there appears to be less of a delay since the well is close to the river and the aquifer is within Hanford formation material.

Ongoing remedial actions at the 100-D-100 unplanned release site suggest that Cr(VI) is present in the vadose zone throughout the soil column thickness. In addition, because the Cr(VI) at the groundwater interface is localized, the fluctuations in Cr(VI) concentrations in groundwater from Well 199-D5-99 are likely more directly related to the vadose source and not specifically a source in the PRZ. Mobile contaminants located within the PRZ are most likely a continuation of the residual, overlying contamination in the vadose zone.

Strontium-90, which has a higher K_d, exhibits some seasonal variations at 100-H, apparently related to changes in water table elevation. The response to groundwater fluctuations and associated change in the shape of the plume indicates the presence of strontium-90 in the soil above the aquifer. In addition, discontinuous low-level detections of strontium-90 have been measured in groundwater in the immediate vicinity of 105-DR Reactor. As not all waste sites have been remediated near the likely sources, it is not yet clear if strontium-90 is present throughout the vadose zone or in the limited area near the water table.

4.9.8.3 Groundwater

In the unconfined aquifer, groundwater impacts at 100-D/H are primarily due to contaminated cooling water. However, elevated Cr(VI) concentrations at 100-D also indicate that some fraction of the 70 percent solution has reached the unconfined aquifer. The maximum Cr(VI) concentrations in a number of wells exceed reactor coolant chromium concentrations (up to 700 µg/L) at the 100-D southern hot spot. These wells have shown Cr(VI) levels up to 69,700 µg/L, concentrations that could not be achieved if the source were the reactor coolant only.

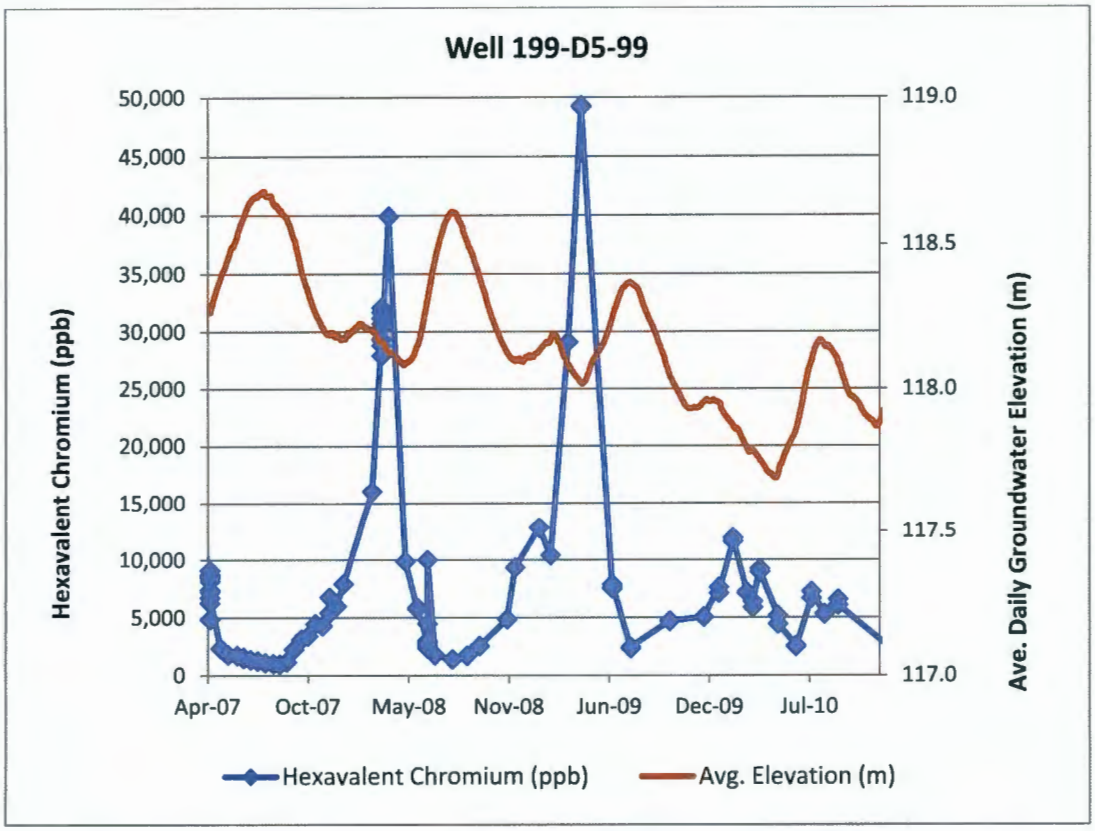

Figure 4-109. Cr(VI) Response to Groundwater Level Changes in Well 199-D5-99

The lower concentration, but very high volumes of cooling water that were discharged during operations resulted in spreading Cr(VI) and other analytes over a broad area since the cooling water mound flowed with groundwater. Groundwater flow during operations was generally toward the east (Figure 3-54), with some component of flow at 100-D being more westerly, toward the Columbia River. During the 1967 infiltration test, this flow direction was accentuated, with the high hydraulic head of the mound forcing water through the aquifer at a higher than normal rate, and hydraulic head.

Groundwater monitoring data indicate that Cr(VI) from concentrated solution is present at the 100-D south plume. The flow path of the high concentration plume is consistent with groundwater flow direction. In addition, while the concentrations of Cr(VI) have been as high as 69,700 µg/L at the south plume, the density does not appear to control the movement of the contamination. Depth discrete data collected during the RI and during previous investigations indicate that, while there is uneven distribution of Cr(VI) in the unconfined aquifer, it is not consistent throughout the aquifer. Current pump-and-treat operations are greatly reducing the Cr(VI) concentrations in this area, as exhibited in Well 199-D5-122.

Contaminants near the bottom of the unconfined aquifer do appear to have been influenced by the depressions identified in the RUM surface. The undulating surface of the RUM (Section 3.4.2) has a marked depression that coincides with the 100-D southern Cr(VI) plume hot spot (Figure 3-4). The depression extends to the south and then curves to the west and toward the Columbia River. Analytical results from Well 199-D3-5 (C7620, Well 2), where the highest Cr(VI) in the well is at the top of the RUM surface, can be explained by contaminants following the RUM surface (Section 4.5.1 – Vertical Distribution of Cr(VI)). Contaminants within a low spot would also experience lower flow rates and would not disperse as readily as in the upper portions of the aquifer.

Groundwater flow across the horn is primarily controlled by the hydrogeology of the area. The Ringold Formation unit E is the primary material of the unconfined aquifer at 100-D. Within the horn, the unconfined aquifer occurs primarily within the Hanford formation, as it is at 100-H. However, in some areas of the horn, erosional remnants of Ringold Formation unit E are present. Since groundwater flows more easily through the Hanford formation due to hydrogeologic properties, a preferred pathway would have been created for contamination movement from 100-D across the horn. As shown conceptually on Figure 4-110, the presence of Ringold Formation unit E may have directed flow (as shown by the arrow) across the horn to 100-H.

Cr(VI) contamination is also present in areas where there is Ringold Formation unit E present. This is theorized to be a result groundwater rising above the Ringold Formation unit E during extremely high groundwater periods. As the amount of cooling water discharge increased, such as during the 1967 test, contaminated cooling water potentially overtopped the Ringold/ Hanford contact elevation due to the exaggerated groundwater mound.

The groundwater mound also had the effect of displacing the original groundwater volume in the unconfined aquifer with cooling water effluent. Contaminants related to cooling water, such as Cr(VI) and fission products, were distributed throughout the unconfined aquifer and up into the deeper vadose zone. As the mound collapsed, contaminants with higher K_d values would have been stranded in the lower portion of the vadose zone. Contaminants with high K_d values can be retained by various sorption processes onto the silt, sand, and gravel matrices within the aquifer (saturated conditions). These contaminants may continue to enter groundwater by dissolution, diffusion, or ionic exchange processes but likely at a very low rate. In areas where groundwater moves very slowly (for example, most of the 100-D area), high concentrations of contaminants adsorbed on the aquifer matrix may slowly diffuse and disperse into groundwater as it interacts with the affected matrix, resulting in persistent downgradient distribution of elevated contaminant concentrations in groundwater. This is most likely to have occurred in the area between the 116-DR-1&2 Trench and the middle of the horn, where the drop in water levels during collapse of the mound would have been greatest.

The first water bearing unit within the RUM has been identified as contaminated in several locations. The RUM consists of gravel in a silt/clay matrix and is considered an aquitard, a leaky confining unit with limited groundwater movement. Within the RUM, thin sand/gravel lenses form discontinuous water-bearing units with variable transmissive properties. Along with variations in RUM surface topography, there is also variation in the thickness of the silt/clay units between the various water-bearing units of the RUM.

Samples collected from water-bearing units within the underlying RUM unit did not exhibit site-related contamination at 100-D. Farther to the east, within the horn, contamination has been identified in the RUM at Well 699-97-48C. This well is located downgradient from the discharge point for 105-D and 105-DR cooling water during the 1967 infiltration test and would have experienced high hydraulic head conditions, forcing contaminated cooling water into this unit. As shown in Figure 4-111, this theory is supported by the water table elevation from operations, which shows a preference for water movement near Well 699-97-48C. The well is located between two remnants of Ringold Formation unit E, which would have acted as a "pinch point" for groundwater moving eastward through that area.

Aquifer tests (see Chapter 3, Section 3.7.4) and geochemistry analysis (Section 3.7.6) indicate that a hydraulic connection exists between the first water-bearing unit in the RUM and the Columbia River near Well 199-H4-12C, and Well 199-H4-15CS. Based on boreholes placed near the 105-H Reactor, the upper confining portion of the RUM appears to thin at this location, likely caused by shallow sediments being reworked and eroded by flood events. These wells coincide with a downward slope of the RUM surface toward the river, located just east of the 183-H Solar Evaporation Basins.

During the RI, several wells were drilled into the first water-bearing unit of the RUM. Contamination was identified in this unit at 100-H, with Cr(VI) concentrations as high as 287 µg/L in a groundwater sample collected during drilling of Well 199-H3-9. However, a comparison of the RUM thickness to the Cr(VI) concentrations did not show any obvious pattern. An analytical model of the RUM contamination was developed (*Evaluation of Potential Hydraulic Capture and Plume Recovery from the Ringold Upper Mud (RUM) in the 100-HR-3 Operable Unit (OU)* [ECF-100HR3-12-0025] in Appendix F) to assess the current state of capture from the two pumping wells in this unit (discussed in the FS).

Groundwater generally moves through the Hanford formation with the direction of flow determined by seasonal variations in the water table inland from the river and in response to Columbia River stage variations. During the high-river stage period in early to mid-summer, groundwater flows southeasterly across the northern portion of the horn area; during periods of general low river stage, groundwater generally discharges into the Columbia River from inland areas. Erosion of the Ringold Formation unit E and parts of the RUM in the horn area have created an undulated RUM surface topography that may expose some of the water-bearing units within the RUM, particularly in the northern portion of the horn area. This condition may account for the historical entry of contaminants (for example, Cr(VI)) into shallow water-bearing units of the RUM, as observed at some localized portions beneath the horn area.

The groundwater flow system beneath the Hanford Site remains a primary pathway for contaminants to migrate away from source areas and, for some contaminants, to discharge into the river. Characterization of hydrogeology at the 100 Area requires understanding of the properties and behavior of the vadose zone, groundwater, and surface water sources, interfaces, and interactions. Both natural and anthropogenic hydrologic processes have influenced groundwater flow patterns and contaminant distribution in the subsurface underlying 100-D/H. The effects of natural processes on contaminant migration are ongoing, while the effects of anthropogenic operations (for example, the high-volume liquid discharges into the 116-H-7 Retention Basin, 116-H-1 Trench, 116-D-7 and 116-DR-9 Retention Basins, and 116-DR-1 and -2 Trenches) have diminished over time with the cessation of reactor operations. However, some residual effects have not completely dissipated, and other processes continue to influence contaminant migration, particularly ongoing pump-and-treat operations.

Groundwater flow directions close to the Columbia River are influenced by river stage. Generally, natural groundwater flow patterns transport COPCs toward the Columbia River. Groundwater flow toward the river dominates at low river stage and surface water dominates the near-shore aquifer flow during periods of peak high river stage. In spring, when the river stage is high, the water table near the river flattens and river water may flow a limited distance into the unconfined aquifer. High river stages can be more than 4 to 5 m (13 to 16 ft) greater than low river stage. The river stage can also fluctuate several meters over short periods (that is, hours to days), based on Priest Rapids Dam operations. River stage fluctuations influence groundwater elevations and flow directions several hundred meters inland from the river. The magnitude of the influence is tempered with increasing distance from the river. Groundwater flow through the first water-bearing unit in the RUM is tempered by recharge water having to move more slowly through sediments with lower hydraulic conductivities of 1×10^{-7} cm/s. As a result, the groundwater flow within the RUM is much slower and Cr(VI) will have a longer residence time.

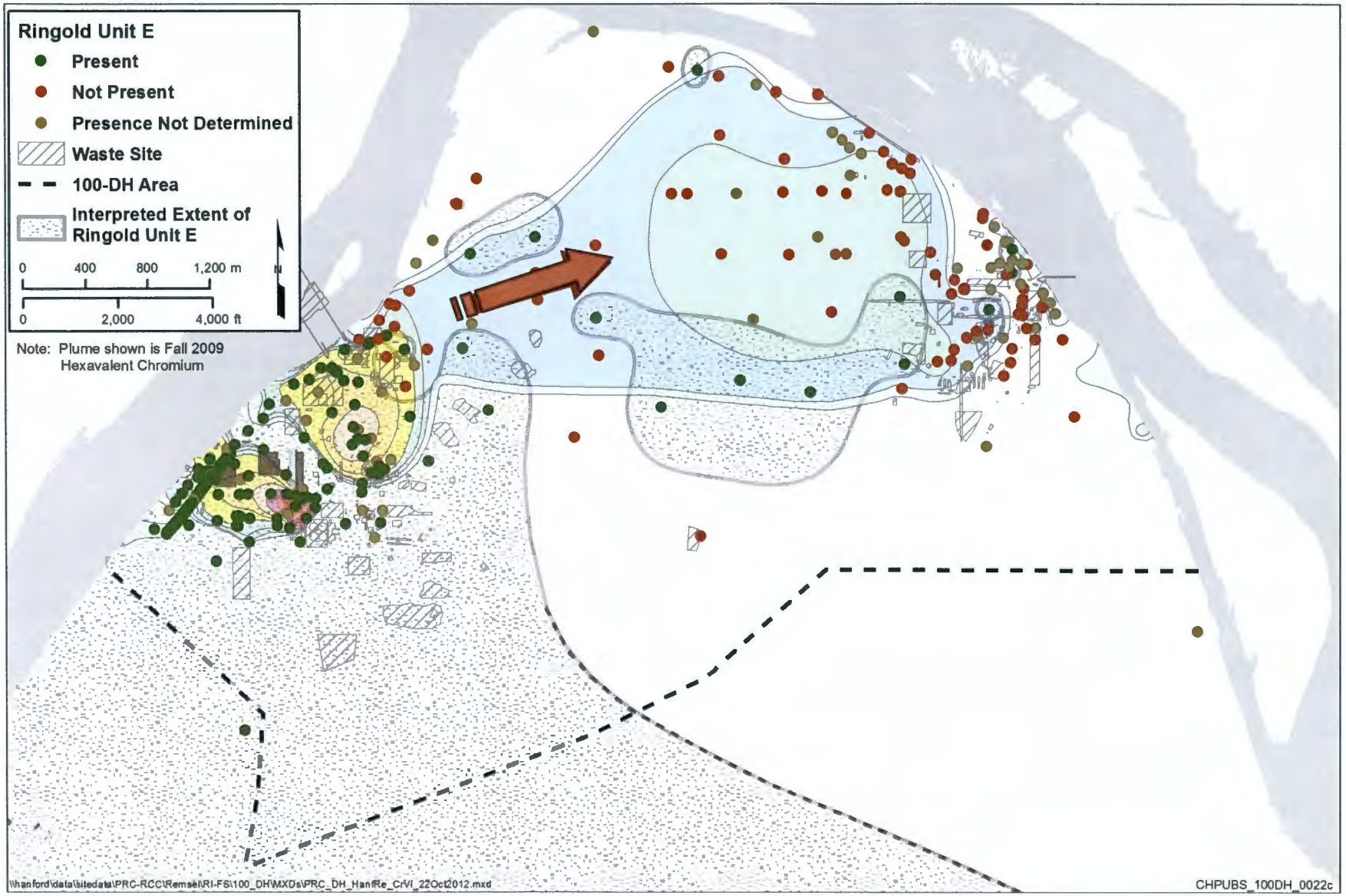

Figure 4-110. Cr(VI) Plume with Geologic Formations of the Unconfined Aquifer

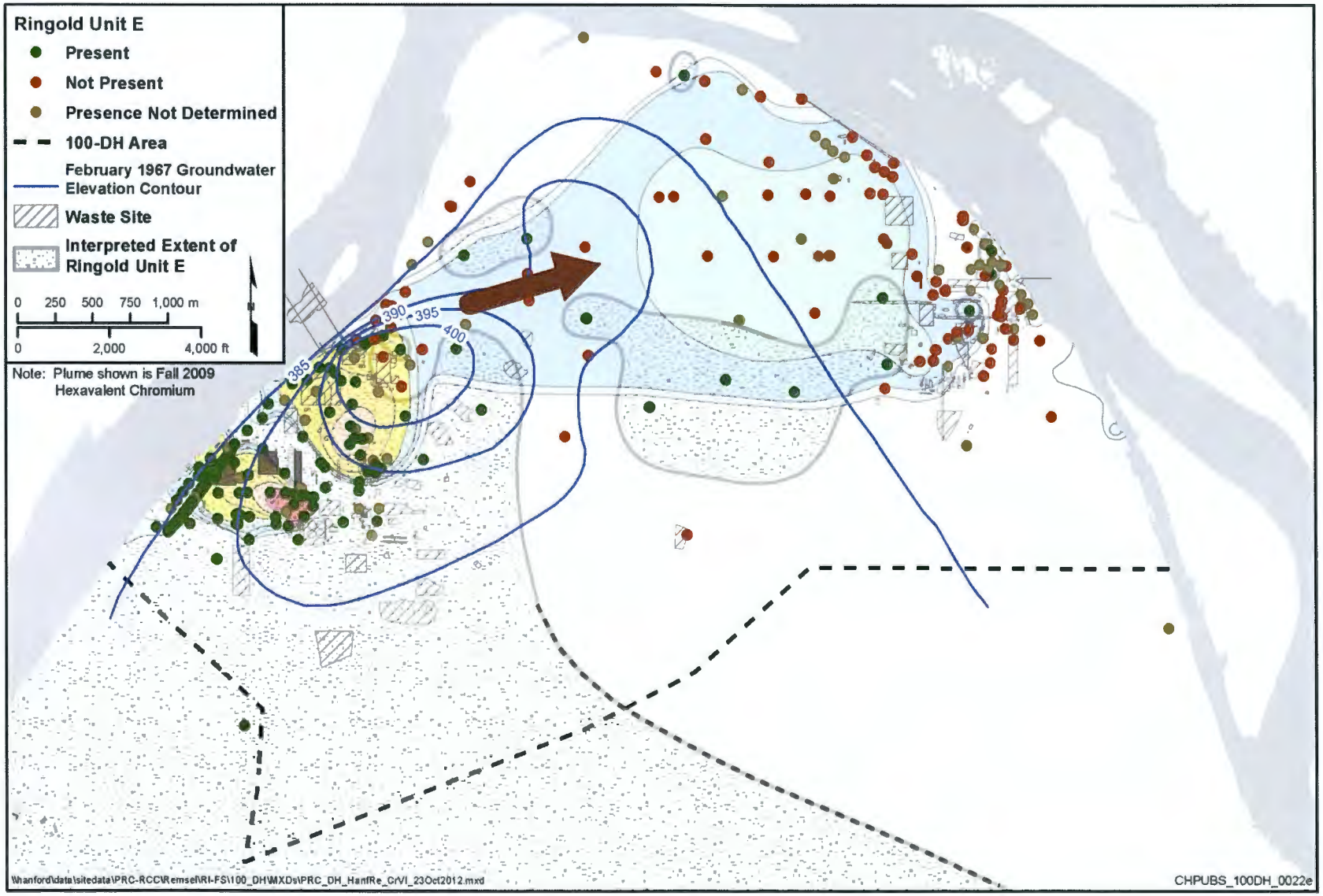

Figure 4-111. Cr(VI) Plume with Geologic Formations of the Unconfined Aquifer

Important Groundwater Concepts

Groundwater flow and contaminant fate and transport processes beneath 100-D/H are highly complex. The main concepts regarding contaminant effects in groundwater include the following:

- Remaining contamination at 100-D/H is primarily associated with Cr(VI) in the following general locations:
  - The high concentration 100-D south plume, which is apparently associated with stock solution releases at 100-D-100, no high concentration source area has been identified at the 100-D north plume.
  - The Cr(VI) plume within the unconfined aquifer underlying 100-D and the horn originated from 105-D and 105-DR Reactor operations. Most of the effect to the horn likely occurred during the 1967 infiltration test at 116-DR-1&2, associated with the 105-D Reactor.
  - Relatively low concentrations across the horn associated with the 116-DR-1&2 Trench 1967 infiltration test.
  - The first water-bearing unit in the RUM in the horn (slightly east of 100-D) at Well 699-97-48C was likely affected from the 116-DR-1&2 Trench and 1967 infiltration test. The effect appears to be limited in areal extent to a small area of the horn, and also limited to low concentrations of Cr(VI).

Contamination in the first water bearing unit of the RUM at 100-H, which is likely associated with the 100-H groundwater mound during operations.

- The Cr(VI) plume within the unconfined aquifer underlying the western portion of 100-H originated from 105-D and 105-DR Reactor operations. Cr(VI) contamination in the unconfined aquifer along the eastern portion of 100-H was a result of 105-H Reactor operations and has been largely remediated.
- Natural attenuation of Cr(VI) is largely attributed to the reduction to Cr(III), with some adsorption and precipitation. The formation of low-solubility Cr(VI) salts may also occur in the environment. Adsorption may facilitate the reduction process. Reductants associated with the aquifer matrix are most important; iron is an important component that is abundant within Hanford sediments. Much smaller plumes of nitrate and strontium-90 are also present in 100-D/H.
- The first water-bearing unit of the RUM at 100-H was affected from 105-H Reactor operations and the resultant groundwater mound. This may be associated with the thinner aquifer and the absence of the Ringold Formation unit E at 100-H, compared to 100-D.
- The interim action DX/HX pump-and-treat system continues to decrease the areal extent of Cr(VI) and reduce Cr(VI) concentrations in the unconfined aquifer.
- Contaminants that are collocated with Cr(VI), such as nitrate, are also being removed from the aquifer by the pump-and-treat system. This aspect is important for remedy selection and design.

4.9.8.4 Hyporheic Zone

The hyporheic zone is a significant interface where groundwater transitions from the aquifer into the surface water. The CRC (DOE/RL-2010-117) evaluated contamination within this zone, and associated ecologic risks are discussed in Section 7.6.4.5. It can be characterized in physiochemical terms by the presence of at least 10 percent advected stream water in the subsurface ("Retention and Transport of Nutrients in a Third-Order Stream in Northwest California: Hyporheic Processes" [Triska et al., 1989])

and can be considered a temporally dynamic area of subsurface mixing between the surface water and groundwater beneath and laterally to a stream channel.

The hyporheic zone in the immediate vicinity of the river is monitored at 100-D/H by analysis of samples collected from a series of shallow sampling structures colloquially called aquifer tubes. The aquifer tube samples have confirmed the presence of 100-D/H contaminants in subsurface water in this zone. Concentrations are variable and somewhat dependent on river stage at the time of sampling. When the river stage is high and river water is actively entering a bank storage condition, concentrations tend to be at their minimum. When river stage declines, contaminant concentrations typically increase as the river once again becomes a gaining stream receiving discharge of groundwater from beneath 100-D/H.

The water particle flow direction moves according to river stage. At high river stage, flow paths are toward the river. When flow reverses, flow should approximately follow the same streamline in the opposite direction with potentially a small component of deflection downstream. "Influence of Oscillating Flow on Hyporheic Zone Development" (Maier and Howard, 2011) indicates that a daily stream stage fluctuation of 0.8 m (2.6 ft) generates between 1.75 and 2.5 m (5.7 and 8.2 ft) of particle movement into and out of the stream bank, and 1.25 to 1.5 m (4 to 5 ft) within the riffle. For streams with a daily stream stage change of 4 m (13 ft), which can occur on the Columbia River, the particle oscillation into the bank extends as much as 7 m (23 ft) while the vertical particle movement within the riffle is only between 3 and 3.5 m (10 and 11.5 ft). During this process, surface water infuses into the hyporheic zone and shoreline portion of the aquifer. River water refreshes some mineral surfaces and provides additional nourishment to bacteria in the form of organic carbon, phosphates, and other nutrients.

Under steady-state flow conditions, flow paths are generally smooth and residence times of surface water in the hyporheic zone are relatively long. Stream-stage fluctuations create fluctuating head gradients and flow reversals, depending on the magnitude of the stream-stage fluctuation ("Influence of Oscillating Flow on Hyporheic Zone Development" [Maier and Howard, 2011]). Once Cr(VI) enters the river or encounters a significant portion of river water in the hyporheic zone, it will be reduced relatively rapidly because of the presence of iron and a reducing environment. The oscillating particle effect demonstrated by "Influence of Oscillating Flow on Hyporheic Zone Development" (Maier and Howard, 2011) for rivers with more extreme fluctuations of stream stage results in a more dynamic exchange of biological and chemical parameters, including iron.

4.9.8.5 River/Riparian Zone

The riparian zone along the river is subject to periodic flooding and deposition of sediment and other detritus along with the floodwater that contains contaminants from upstream. This includes both the immediately upstream portions of the Hanford Site as well as the main upstream flow from Canada and northern and central Washington. Therefore, non-Hanford Site contaminants may be introduced to the surface, vadose zone, and groundwater portion of the riparian zone through flooding. Most recently, significant riparian flooding was observed in 2011 as a result of the high run-off associated with the 2010 to 2011 snow pack.

Over the years of the Hanford Site operations, direct discharges also have affected the river. Data from the Columbia River RI Work Plan (DOE/RL-2008-11) and *DQO Summary Report for the Remedial Investigation of Hanford Site Releases to the Columbia River* (WCH-265) suggest historical discharge from 100-D/H operations affected the Columbia River. Large quantities of cooling water were discharged directly to the river via outfall pipes. Upstream data indicate that industrial and mining sources also contributed contaminants to the Columbia River.

Upwelling studies showed several locations where measurable Cr(VI) was upwelling in the riverbed adjacent to the known groundwater plumes in 100-D/H. Evaluation of the flow regime on both sides of the river indicates that groundwater from 100-D/H does not upwell beyond the thalweg on the far side of

the river. The upwelling studies (Columbia River RI Report [WCH-380]) indicate that during low flow stage in the river, water that is largely Hanford Site groundwater can upwell into the river. Given the large diurnal changes in river stage, this upwelling condition is likely to be ephemeral as groundwater flow reverses direction into and out of the river several times per day in response to operations upstream at Priest River Dam.

4.9.8.6 Pathways

A final aspect of the CSM is the different exposure pathways in which humans, animals, and plants could potentially be affected by the presence of contaminants in the environment. The exposure pathways and risk assessments for humans are described in Chapter 6 and the exposure pathways and risk assessments for biota are described in Chapter 7. The remedies are developed and evaluated in Chapters 8, 9, and 10; the remedies are chosen to break the exposure pathways, thereby providing effective mitigation of the potential risks that the contaminants pose. The general CSM elements that describe the full evolution of contaminants in the environment are described in Chapter 1 and are represented in the flow diagram on Figure 4-112.

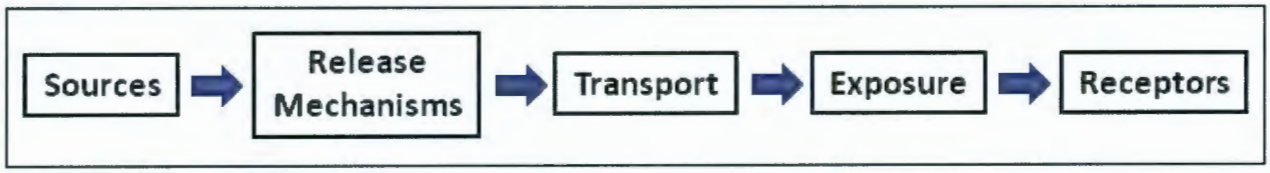

Figure 4-112. Simplified CSM Element Diagram

The potential exposure points for contaminants in soil and water are determined by the location of the contaminants and the potentially applicable transport mechanisms. The following potential exposure points are identified for contaminants at 100-D/H:

- The **ground surface within 100-D/H** at areas where contaminated soil is located in the surface, or near-surface, such that exposure could occur by transient contact with surface soil, or through minimally intrusive activities.
- The **subsurface portion of the vadose zone within 100-D/H** where contaminated soil may be encountered during intrusive activities (for example, excavation for construction). Excavation activities can also relocate contaminants to the ground surface where they may become subject to surface exposure.
- **Food chain** effects may occur where contaminants from the surface or subsurface soil are taken up by plants or animals, thereby entering the food chain. This incorporates two types of potential exposures: direct contact and ingestion by the initially exposed organism(s), and subsequent exposure to second order consumers by ingestion of the contaminated organisms.
- **Contaminated groundwater beneath 100-D/H** could potentially be extracted and consumed, thereby creating an exposure point for contaminated groundwater potentially anywhere within the aquifer where it would be feasible to produce a useful volume of water.
- **Contaminated groundwater at locations outside 100-D/H** where contaminants may migrate from the source area. This groundwater could potentially be extracted and consumed, thereby creating an exposure point for contaminated groundwater potentially anywhere within the aquifer where it would be feasible to produce a useful volume of water.

- **Surface and subsurface soil within the riparian zone** near the river where contaminants may migrate via overland flow of contaminated groundwater discharging seasonally from seeps above the river stage elevation.

- **Surface water of the Columbia River** adjacent to, and downstream from, 100-D/H, where contaminants migrate from the site via overland flow processes or by the interaction of contaminated groundwater with surface water of the river.

The CSM summarizes the primary and secondary sources of contaminants at 100-D/H, the mechanisms for contaminant release into the environment, and contaminant distribution through the environment. The migration of contaminants includes a discussion on the driving forces and transport mechanisms whereby contaminants have a pathway to cause exposure to a receptor. The evaluation of risk from contaminants is provided in Chapters 6 and 7. This evaluation provides a basis for a remedial action to break the pathway for exposure for a contaminant that poses a risk to human health or the environment. The evaluation of potential remedies is presented in Chapters 8 through 10.

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5 Contaminant Fate and Transport

This chapter presents an evaluation of the anticipated behavior of selected contaminants in the vadose zone and groundwater at 100-D/H. An approach is presented for the assessment of anticipated future behavior of vadose zone contaminants that may function as secondary sources of contamination. The approach describes how these contaminants are released into the environment to affect underlying groundwater. A simulation approach is also presented to describe the future behavior of contaminants already present in groundwater at 100-D/H. Factors affecting the fate and transport of contaminants and results, and uncertainties in the information and methods are discussed, concluding with a summary of the chapter as a whole.

Highlights

- The disposal of large volumes of liquid effluent to the vadose zone during reactor operations resulted in accelerated transport of contaminants to deeper portions of the vadose zone and the unconfined aquifer in 100-D/H.
- Contaminant migration rates are currently much slower, because liquid effluent discharges have stopped.
- For previously remediated waste sites, there were no exceedances of soil screening levels protective of groundwater and surface water.
- Groundwater contaminant flow and transport modeling indicates that the groundwater pump-and-treat systems provide protection to the Columbia River along the shoreline in almost all areas.
- The existing groundwater pump-and-treat systems are actively remediating the Cr(VI) plumes.
- Strontium-90 and nitrate concentrations in 100-D/H groundwater above the MCLs are within the capture zone of the recovery wells. Concentrations and plume footprint areas in groundwater will decline over time, although the rate of decline is not uniform across the area.

The purposes of the fate and transport information provided in this chapter are as follows:

- Describe the development of soil screening levels (SSLs) and PRGs for contaminated vadose zone soil in 100-D/H. In addition, the chapter will describe the application of the SSLs and PRGs to observed soil conditions to support assessment of potential threats to groundwater and surface water.
- Use the SSLs and PRGs to evaluate whether contaminants present in the vadose zone at 16 waste sites characterized during this RI and during the preceding LFI, as well as interim closed waste sites, may act as a secondary source of groundwater contamination. Waste sites that are not yet remediated were carried forward into the FS for evaluation without SSL/PRG evaluation with the COPCs identified for those waste sites based on process knowledge.
- Establish a process for evaluating ongoing groundwater and vadose zone remediation activities and comparing remedial alternatives being considered for completing cleanup actions at 100-D/H.

Understanding contaminant fate and transport in the environment is an important part of the RI/FS process. Projections of future contaminant behavior and concentrations at points of exposure are needed to assess potential threats to human health and the environment. These simulations are especially important for sites where contaminants are long-lived or where groundwater contaminant plumes may migrate beyond the area covered by a monitoring well network. Contaminant fate and transport was simulated using a one-dimensional (1D) computer model for the vadose zone and a three-dimensional (3D) computer model for groundwater contaminants.

This chapter describes key processes affecting the fate and transport of 100-D/H COPCs in environmental media, and the effect these processes have on the distribution of COPCs in the future. The information presented in this chapter was used to calculate SSLs and PRGs that are protective of groundwater and surface water under the modeling scenarios presented. Remediated waste site constituent concentrations are compared to the screening levels to identify waste sites requiring consideration in the FS for

groundwater or surface water protection. The results from groundwater flow and transport models developed to simulate existing COPC fate and transport in groundwater for the 2012 through 2087 period are also presented. In addition, the assumptions and model input parameters detailed in this chapter are important for future waste site remediation efforts and meeting the cleanup standards. The cleanup verification process, including demonstration of how cleanup standards are achieved, can involve the evaluation of the conceptual site model at the individual waste sites against the assumptions used to develop the SSLs and PRGs. To the extent a significant deviation from the groundwater/surface water protection SSLs and PRGs assumptions is observed, site-specific conditions can be used to revise the fate and transport models to evaluate the potential for the waste site to act as a source of groundwater contamination.

One hundred twenty-seven interim remediated and three un-remediated waste sites were evaluated to determine if further action may be needed for the protection of groundwater and surface water. No waste sites exhibited exposure point concentrations (EPCs) greater than the SSLs. None of the EPCs for metals fell outside the reported background concentration range for Hanford Site soils.

The remaining waste sites that are not yet remediated are carried into the FS for evaluation. The COPCs identified for those waste sites are based on process knowledge.

5.1 Evaluation Process for Assessment of Protectiveness of Groundwater and Surface Water

The evaluation of the potential for vadose zone contamination to affect groundwater and/or surface water followed a specific set of logical steps shown on Figure 5-1. This process evaluated the potential for secondary source contaminants to migrate to groundwater and subsequently discharge to surface water at concentrations that would pose a threat to human or ecological receptors. This evaluation did not include assessment of potential for effects of direct contact exposures to shallow or surface contamination (direct contact exposure assessment is provided in Chapter 6). The activities associated with these steps are as follows:

- Available data that describe the nature and extent of residual vadose zone soil contamination at a particular waste site were assembled. This included laboratory analysis of soil samples collected from the vadose, field measurements of specific contaminant concentrations, qualitative and quantitative measurements of radionuclides present in the vadose zone, measurements of soil physical properties (for example, moisture, particle size distribution), and field observations during drilling and/or excavation. These data were generated from process knowledge and operating history, from specific waste site characterization activities (for example, LFIs and RI activities), or from completion and verification measurements (CVP) associated with completion of vadose zone remedial activities.

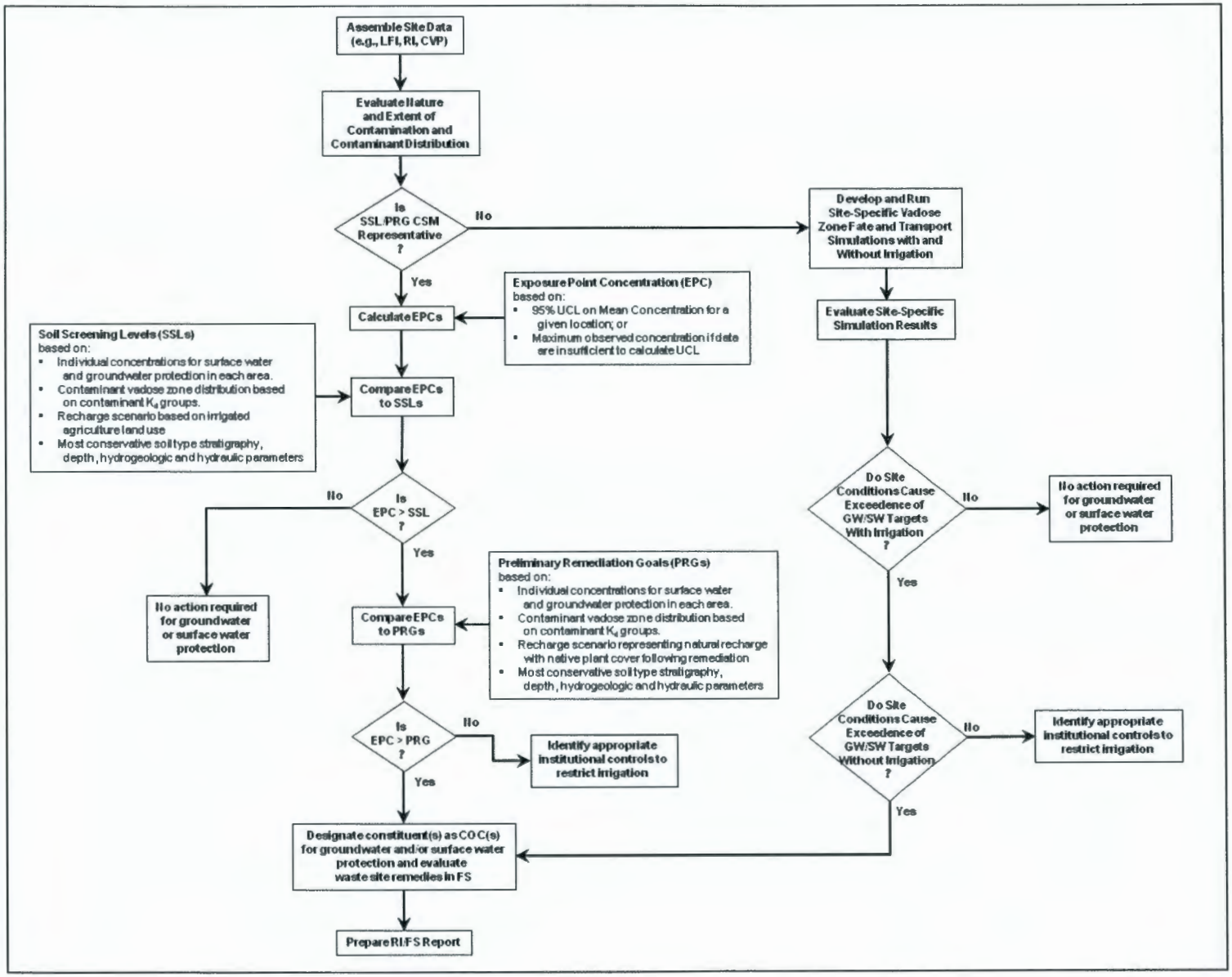


Figure 5-1. Screening Process for Groundwater/Surface Water Protection at Sites with Measurement Data Describing Site Conditions

- The data were assembled to provide a description of residual contamination conditions that included concentrations of contaminants and their locations with the vertical and horizontal extent of the waste site.

- The individual waste site conditions were then compared to the generic conceptual site models developed for the SSL and PRG development. If the known site conditions were similar to those used to describe the SSL and PRG simulations (that is, 100:0 or 70:30 profile, depending on individual contaminant's Kd value), then the evaluation followed the SSL and PRG comparison pathway. If, however, the known site conditions differed from the default simulation such that these were non-conservative initial conditions, then the waste site was evaluated using a site-specific contaminant transport simulation. Conditions indicating that default scenarios are not representative included the presence of past or persistent groundwater plumes associated with a specific waste site or operating area or an observed vertical distribution of a contaminant, or contaminants, within the vadose zone that were inconsistent with the default initial distribution (that is, 100:0 or 70:30 profile) such that the default initial condition would be non-conservative with respect to peak groundwater concentration for that contaminant.

- Waste sites for which the default conditions were not representative were subsequently evaluated individually in a site-specific analysis.

- EPCs were derived for each COPC based on the site-specific data at hand, and were assigned either the 95 percent UCL on the mean concentration or the maximum observed concentration if available data were insufficient to derive the 95 percent UCL.

- The EPCs for each contaminant at a waste site were then compared to the SSLs. The SSLs represent conservative protection values (protective of groundwater and surface water) based on the maximum reasonably foreseeable recharge scenario in the 100 Area (that is, irrigated agriculture). If the EPC is less than the SSL, then that contaminant was identified as requiring no further action and the assessment moved on to the next contaminant.

- If the site-specific contaminant EPC exceeded the SSL, then the EPC was subsequently compared to the PRG for that contaminant. The PRGs represent protection values (protective of groundwater and surface water) based on the expected land use in the 100 Area (that is, conservation activities with native vegetation). If the EPC exceeded the SSL, but was less than the PRG, then the affected waste site was identified for application of institutional controls that will prevent irrigation in the future at the site. If the EPC exceeded the PRG, then the contaminant was identified as a COC and the site was included in the FS for identification of appropriate remedial alternatives to mitigate risks to groundwater and surface water posed by the vadose zone contamination.

In cases where the waste site conditions were not adequately represented by the default SSL and/or PRG simulations, then the waste site and its affected contaminants were evaluated using a site-specific vadose zone transport simulation. This simulation used the same general fate and transport modeling approach used for the SSL and PRG development, except that site-specific conditions were substituted where appropriate. Site-specific simulations were evaluated as follows:

- Site-specific results under the irrigation recharge scenario were evaluated to determine if the site conditions resulted in exceedance of the contaminant-specific groundwater or surface water protection criteria (for example, MCLs or AWQC). If the site conditions did not cause an exceedance of any of the criteria, then the site was identified as requiring no further action. Regarding groundwater or surface water protection.

- The site-specific results under the native vegetation recharge scenario were evaluated next to determine whether the site conditions resulted in exceedance of the groundwater or surface water protection criteria. If the site conditions did not cause an exceedance of any of the protection criteria under the native vegetation (no irrigation) recharge scenario, then the affected waste site was identified for application of institutional controls that will prevent irrigation in the future at the site.

- If the site-specific results under the native vegetation (no irrigation) recharge scenario indicated that the site conditions would result in an exceedance of any of the groundwater or surface water protection criteria, then the exceeding contaminants were identified as COCs for that site. In addition, the waste site was included in the FS for identification of appropriate remedial alternatives to mitigate risks to groundwater and surface water posed by the vadose zone contamination.

The assessment of vadose zone contaminant migration focused on evaluation of waste sites that have been characterized during the 100-D/H Area LFI, the current RI, or are sites at which planned soil remediation is complete and characterization describing the post-remediation conditions are available (that is, the "previously remediated sites"). In contrast, this assessment does not evaluate waste sites that are not yet remediated because these were carried directly forward into the FS for evaluation, with the COPCs identified based on process knowledge. The process followed for evaluation of previously remediated waste sites provided a basis for confirming the completion of the soil remediation at these sites. The same process was applied to the following 16 previously remediated sites located in 100-D/H using LFI, RI, and CVP/RSVP data:

- 116-D-1B Trench
- 116-D-7 Retention Basin
- 116-DR-1&2 Trench
- 116-DR-9 Retention Basin
- 116-H-1 Trench
- 116-H-4 Pluto Crib
- 116-H-6 Solar Evaporation Basin
- 116-H-7 Retention Basin
- 118-D-6:3 Reactor Fuel Storage Basin
- 118-H-6:2, 118-H-6:3, and 118-H-6:6 Reactor Fuel Storage Basin
- 100-D-12 French Drain
- 116-D-1A Trench
- 100-D- 4 Trench
- 116-D-4 Crib
- 116-H-2 Trench/Crib
- 1607-H4 Septic System

Interim remedial action has continued, and vadose zone sampling was conducted at 142 of these interim-remediated waste sites, which were remediated per *Interim Remedial Action Record of Decision for the 100-BC-1, 100-DR-1, and 100-HR-1 Operable Units, Hanford Site, Benton County, Washington* (EPA/ROD/R10-95/126) through December 2012 (Table 5-1). CVP and/or RSVP data are available for the completed waste sites and evaluated through the risk assessment activities.

Table 5-1. Previously-Remediated Waste Sites in 100-D/H

100-D Area				100-H Area	
100-DR-1 OU	100-D-48:1	116-D-10	*100-DR-2 OU*	*100-HR-1 OU*	100-H-12
100-D-1	100-D-48:2	116-D-1A	100-D-12	100-H-17	100-H-14
100-D-18	100-D-48:3	116-D-2	100-D-13	100-H-21	118-H-6:3
100-D-19	100-D-48:4	116-D-4	100-D-15	100-H-24	118-H-6:6
100-D-2	100-D-49:2	116-D-5	100-D-28:1	100-H-28:1	118-H-6:5
100-D-20	100-D-49:3	116-D-6	100-D-43	100-H-28:6	118-H-6:4
100-D-21	100-D-49:4	116-D-7	100-D-47	100-H-3	1607-H2
100-D-22	100-D-50:5	116-D-9	100-D-94	100-H-35	1607-H3
100-D-24	100-D-52	116-DR-1 & 2	116-D-8	100-H-4	1607-H4
100-D-29	100-D-56:1	116-DR-5	116-DR-10	100-H-41	
100-D-3	100-D-56:2	116-DR-9	116-DR-4	100-H-45	*100-HR-2 OU*
100-D-31:1	100-D-61	118-D-6:4	116-DR-6	100-H-49:2	100-H-37
100-D-31:10	100-D-7	120-D-2	116-DR-7	100-H-5	100-H-40
100-D-31:2	100-D-70	126-D-2	116-DR-8	100-H-50	118-H-1:1
100-D-31:3	100-D-74	128-D-2	118-D-1	100-H-51:4	118-H-1:2
100-D-31:4	100-D-75:3	130-D-1	118-D-4	100-H-51:5	118-H-2
100-D-31:5	100-D-80:1	132-D-1	118-D-5	100-H-53	118-H-3
100-D-31:6	100-D-82	1607-D2:1	118-DR-1	100-H-7	118-H-4
100-D-31:7	100-D-83:4	1607-D2:2	118-DR-2:2	100-H-8	118-H-5
100-D-31:8	100-D-84:1	1607-D2:3	122-DR-1:2	116-H-1	128-H-1
100-D-31:9	100-D-85:1	1607-D2:4	1607-D1	116-H-3	128-H-2
100-D-32	100-D-87	1607-D4	600-30	116-H-5	128-H-3
100-D-4	100-D-88	1607-D5		116-H-7	1607-H1
100-D-42	100-D-9	628-3		116-H-9	600-151
100-D-45	100-D-90	UPR-100-D-5		100-H-11	600-152

5.2 Overview of the 100-D/H Conceptual Site Model

The CSM presented in Section 4.9 described how the operating history at 100-D/H contributed to sources of environmental contamination and presented how the primary sources related to secondary contamination sources and the integration of contaminant migration to known and potential receptor

exposure points. Releases occurred during reactor operations. The retention basins are located between the river and the coal plant. The water treatment operations are shown from the 182 Reservoir and the 183-D and 183-DR clearwells and head houses.

5.3 Contaminant Persistence

The persistence of various contaminants in the environment determines how long they are available for transport to different receptors. If a contaminant remains in the environment for a long time and is highly mobile, it is more likely to be transported from the vadose zone to groundwater and, eventually, to surface water. Persistence is defined by how long it takes a particular contaminant to be transformed into a less toxic or less available form, or how long it takes the contaminant to physically leave the affected area. Radionuclides undergo radioactive decay at varying rates specific to the individual nuclides. Nonradioactive chemicals may also degrade, decay, or undergo chemical transformation that reduces the residual mass of the contaminant available for transport or direct exposure. The following paragraphs discuss the persistence of the selected COPCs.

5.3.1 Persistence of Nonradioactive Chemical Constituents

The persistence of chemical constituents, and alternatively, the degradation of these compounds at 100-D/H is primarily driven by biological and geochemical reduction-oxidation processes, potential biological uptake, and physical processes (for example, volatilization and water solubility). The nonradioactive chemical constituents identified for this assessment include metals (chromium measured as Cr(VI), which is generally present as a dissolved oxyanion or as a metallic salt, and total chromium, which includes Cr(VI) and chromium in other valence states, and zinc), other metals, nonmetallic oxyanions (nitrate and sulfate), and VOCs (carbon tetrachloride and chloroform). These constituents are subject to a variety of transformational processes.

Both Cr(VI) and zinc are relatively stable and persistent in the vadose and groundwater environment at 100-D/H. Zinc is generally present as a divalent cation and does not undergo transformation under ambient conditions. Chromium is typically present in the environment in one of two oxidation states (trivalent or hexavalent). Trivalent chromium is typically precipitated as a low-solubility hydroxide molecule, $Cr(OH)_3$ and, as such, is not mobile and exhibits low mammalian toxicity. Cr(VI), however, is highly toxic and is typically present under ambient conditions at 100-D/H as a soluble oxyanion, $Cr_2O_7^{-2}$ or CrO_4^{-2}, depending primarily on pH (the dichromate oxyanion is dominant in acidic conditions; the chromate oxyanion is dominant in alkaline conditions). The ionic forms of Cr(VI) are relatively stable at the oxidation state typically found in soil and groundwater at 100-D/H, and the constituent tends to remain mobile.

While the largest primary source of Cr(VI) contamination in 100-D/H was sodium dichromate dihydrate used for corrosion control in reactor cooling water, another source is important to consider in evaluating groundwater data. Sodium dichromate is an acidic compound in its concentrated form. The dichromate, or chromate, ion can react with other metals in the environment to form compounds of lesser solubility. These compounds can include potassium dichromate (which is about one-tenth as soluble as sodium dichromate dihydrate) and lead chromate (which is essentially water insoluble). Cr(VI) ions can also be subject to chemical reduction under moderately reducing conditions, or by reaction with reducing agents such as ferrous iron. Ferrous iron is very effective at reducing Cr(VI) to Cr(III), producing a very low-solubility hydroxide molecule. Many of the metals of interest identified in soil and groundwater at 100-D/H are not considered very mobile with the exception of some species of chromium. Metals, such as zinc, arsenic, cadmium, and lead, are persistent in the environment but they are less mobile. However, chromium may be present in various oxidation and ionic states that affect the mobility in the environment. Cr(VI) is moderately mobile and toxic, whereas the reduced trivalent form exhibits low mobility and is

nontoxic. Mercury can be very mobile when it is in the presence of acid as may have been the case during reactor operations.

The mobility, and therefore persistence, of metals in vadose soil and within an aquifer system is influenced by several factors including: soil type, the cation exchange capacity of the soil (the ability to hold cations [that is, positively-charged ions] under a given condition), pH, and the presence of other metals. Cationic metals, such as $^{90}Sr^{+2}$ tend to bind more easily to silt and clay particles in soil. This is a function of greater surface area being available for binding. In addition, the silt and clay fractions commonly exhibit a higher capacity for cation exchange and therefore for sorbing metal ions. Cation exchange is a substantial contributor to reduction in mobility of many metals in the vadose and aquifer system at 100-D/H.

Some metals (for example, arsenic and chromium) commonly exist in the environment as complex anions (for example, arsenate, arsenite, and chromate). These metal oxyanions are generally water-soluble and are not retained by soil particles to a large degree. Some soil particles (for example, organic matter and some clay minerals) do exhibit measureable anion exchange capacity under certain conditions of pH and reduction-oxidation potential. Anion exchange does not play a large role in limiting mobility of most oxyanions at 100-D/H.

Nitrate is a common plant nutrient and is a relatively stable oxyanion of nitrogen and oxygen. Its presence in groundwater beneath 100-D/H may be related to historical planned and unplanned releases of cooling water treatment chemicals, as well as from the use of nitric acid for various decontamination and cleaning activities in the reactors. Nitrate occurs in groundwater in proximity to the reactors and septic systems. It is highly water-soluble and remains stable in vadose zone soil and groundwater, and surface water under oxidizing conditions typically encountered at 100-D/H. Nitrate is subject to chemical or biological reduction to nitrite or ultimately to diatomic nitrogen by soil and water microorganisms under low-oxygen conditions. Reduction of nitrate to diatomic nitrogen generally results in removal of the nitrogen from the soil/water system. Nitrate is mobile, relatively stable, and persistent in groundwater.

Sulfate is a common plant nutrient, however, its presence in groundwater beneath 100-D/H may be related to historical planned and unplanned releases of cooling water treatment chemicals, as well as from the use of sulfuric acid for various decontamination and cleaning activities in the reactors. Sulfate is widespread with the highest concentrations observed downgradient of the in situ reduction-oxidation manipulation (ISRM) reactive barrier. The sulfate at this location resulted from oxidation of the residual sulfhydryl (-SH) groups from the sodium dithionite reagent used to establish the ISRM barrier. Sulfate is mobile, relatively stable, and persistent in groundwater.

Carbon tetrachloride and chloroform will degrade very slowly, if at all, under typical dissolved oxygen concentrations in groundwater beneath 100-D/H. However, carbon tetrachloride and chloroform can be reductively dechlorinated by facultative[1] and obligate[2] anaerobic microorganisms under anoxic conditions. Additionally, carbon tetrachloride and chloroform may volatilize from the land surface or surface water directly to the atmosphere. Carbon tetrachloride and chloroform dissolved in soil moisture or groundwater can partition to soil gas and then migrate to the atmosphere; however, gas exchange from the deep vadose (for example, below a few meters below ground surface) or from groundwater accounts for only a tiny potential loss. Once in the atmosphere, these compounds can be destroyed through photolytic oxidation. The potential for volatilization or biologically mediated degradation is dependent

[1] Can survive in both aerobic and anaerobic conditions.

[2] Can survive only in anaerobic conditions.

upon the specific physical and chemical characteristics of a constituent and the size and nature of the microbial populations. The chlorinated solvents presented at 100-D/H are expected to be persistent in soil and groundwater.

5.3.2 Persistence of Radiological Constituents

Radiological constituent persistence is primarily controlled by radioactive decay processes that can transform the parent isotope into another isotope of the same element or into another element. The daughter product of decay may be a radionuclide or a stable isotope. Radionuclides with relatively high mobility and longer half-lives ($T_{1/2}$) are of more environmental concern than radionuclides with lower mobility and shorter $T_{1/2}$. This is primarily due to the potential for constituents with higher mobility and longer half-lives to reach the saturated zone at higher activities and greater potential to migrate though groundwater. Chapter 6 identifies one radionuclide as a groundwater COPC (strontium-90). Strontium-90 is a beta particle-emitting fission product with a $T_{1/2}$ of 28.8 years. The beta decay daughter product of strontium-90 is yttrium-90 ($T_{1/2}$ = 64.1 hours), which then beta decays to the stable zirconium-90 isotope. The radionuclides, their half-lives, and daughter products are further explained in *Composite Analysis for Low-Level Waste Disposal in the 200 Area Plateau of the Hanford Site* (PNNL-11800).

The persistence in groundwater of these nuclides is also affected by their individual unique chemical and physical behaviors. Strontium-90 commonly remains as an exchangeable divalent cation in the environment. As such, it is not highly mobile and tends to be retained on soil particles near its point of release. Retention of strontium on soil particles by cation exchange processes, however, is subject to competition by other common cations (for example, calcium). This competition can increase strontium mobility under some environmental conditions. This is a consideration with regard to the chemistry of dust suppression water and fixatives used in application/operation procedures.

5.4 Vadose Contaminant Migration Assessment

Concepts affecting contaminant transport in the vadose zone are presented in this section, followed by factors affecting contaminant transport in the saturated zone. Quantitative applications of these parameters and boundary conditions to develop analytical and numerical models of transport through the vadose and saturated zones are presented, with a discussion of each factor affecting contaminant migration. The results of the application of these models to develop groundwater and surface water protection comparison criteria and predict future conditions are also presented.

Contaminants released from 100-D/H sources were transported through the vadose zone and, in some cases, reached the water table. This discussion focuses on factors affecting contaminant transport through the unsaturated and saturated zones of the unconsolidated matrix above the basalt. The most significant factors affecting ongoing subsurface contaminant migration are the type of surface cover and its effect on net infiltration or recharge rates; the physical, chemical, and hydraulic characteristics of the matrix; and the physical and chemical properties of the contaminant (Section 5.6.1).

Once contaminants reached groundwater, mobile contaminants traveled with groundwater in the general direction of flow. Contaminated groundwater can migrate downgradient to discharge directly into the adjacent Columbia River. Contaminated groundwater may also be seasonally discharged in springs or seeps to flow overland across the riparian zone to discharge into the river. Seasonal seep discharges may be a limited and localized source of recontamination of the ground surface in the riparian zone.

The assessment of vadose zone contaminant migration is focused on evaluation of waste sites that have been characterized during the 100-D/H LFI, the current RI, or are sites at which planned soil remediation is complete and characterization describing the post-remediation conditions are available (that is, the

"previously remediated sites"). Unremediated sites are elevated directly to the FS without undergoing the evaluation described here. This provides a basis for confirming the completion of the soil remediation at these sites. The same process was applied to 16 previously remediated sites located in 100-D/H using LFI, RI, and CVP/RSVP data.

Contaminant migration from 100-D/H waste sites through the vadose zone to the underlying aquifer is controlled by the driving forces, interactions between water and sediments, and interactions between the contaminants and sediments specific to the OUs. Driving forces include gravity; matric potential gradients; recharge, which is the result of competition between precipitation, evaporation, transpiration, infiltration, run-off, and run-on; and artificial discharges, such as those from septic tank leach fields, lagoons, pipe and tank leaks, and irrigation. The types, thicknesses, and properties of the sediments can all affect the rate and direction of solute and water movement to the aquifer. A contaminant's concentration in the groundwater and its concentration in the downgradient Columbia River, including the peak concentration, are dependent on the solute flux from the vadose zone; aquifer thickness, properties, and flux rates; travel distance; groundwater and river water mixing; and the location sampled. Each contaminant's decay rate (if applicable) and propensity to sorb to vadose zone or aquifer materials can also be important controlling factors on the peak concentration, from which the PRG or the screening level is calculated.

5.4.1 Surface Cover, Infiltration, and Recharge

The net infiltration into the vadose zone is driven by the competition between processes of precipitation, evaporation, transpiration, run-off, and run-on. In a semiarid or arid climate, downward fluxes resulting from this competition are episodic and usually infrequent. A number of studies have been carried out at the Hanford Site to ascertain representative long-term averages of the episodic fluxes (that is, recharge rates), such as those compiled in *Vadose Zone Hydrogeology Package for Hanford Site Assessments* (PNNL-14702), hereinafter called Vadose Zone Hydrogeology Package) for the 100 Area.
The 100 Area-specific recharge rates in the Vadose Zone Hydrogeology Package (PNNL-14702) varied with surface soil type and provided an estimate of the range of possible recharge rates for various land uses. The four surface soil types identified in the 100 Area were the Ephrata Sandy Loam, Ephrata Stony Loam, Burbank Loamy Sand, and Rupert Sand. However, recharge rates for the Ephrata Sandy Loam and the Ephrata Stony Loam were described as being identical (Vadose Zone Hydrogeology Package [PNNL-14702]). Therefore, the three different surface soil types were assumed to represent recharge rate variability for modeling purposes.

The long-term natural driving force for flow and transport through the vadose zone is the downward movement of water. This movement is expressed as follows (*Compendium of Data for the Hanford Site [Fiscal Years 2004 to 2008] Applicable to Estimation of Recharge Rates* [PNNL-17841]):

- Infiltration refers to water usually resulting from precipitation that enters the ground. Enhanced infiltration may result where surface depressions act as terminuses for overland flow.
- Deep percolation or deep drainage refers to water that has percolated or drained below the zone of evaporation and the influence of plant roots.
- Recharge is water that flows to the water table, and is the primary mechanism for transporting contaminants from the vadose zone to groundwater.

Direct measurement of naturally occurring recharge resulting from surface infiltration at the Hanford Site is not practical. The measurement is made indirectly because of the thickness of the vadose zone and the time necessary for water to travel from the land surface to the water table. In place of direct measurements of recharge at the water table, measurements and analyses of deep drainage in the

unsaturated zone are used to approximate the recharge. The terms can be equated, as long as the climate, land use, and land cover remain the same. Consequently, the terms "deep percolation" or "deep drainage" are often used synonymously with recharge.

There is ample evidence that revegetation of the disturbed land at the Hanford Site occurs both with and without human intervention. Data collected from the Prototype Hanford Barrier in 200 East Area indicate the tall sagebrush (*Artemisia tridentata*) community begins to reduce net infiltration very soon after planting. The species richness of the plant community on the Prototype Hanford Barrier dropped from 35 in 1997 to 12 in 2007. The dominance of tall sagebrush on the surface may continue to reduce the species richness on the surface (Figure 5-2).

Grass cover has decreased from initial levels on the barrier surface, and continued decreasing from 2004 to 2007. Cheatgrass (*Bromus tectorum*) and Russian thistle (*Salsola kali*) are nearly nonexistent on the barrier surface. The western and northern side slopes of the barrier, which were not planted with sagebrush, show less plant cover but higher species diversity than the barrier surface. This may be due to the influence of windblown material and seeds from adjacent land, or the lack of shrubs competing for resources. Insects and small mammals are present in the barrier surface, which indicates the restored barrier surface is beginning to function like a recovering ecosystem.

Numerous studies have estimated recharge rates for the vadose zone system at the Hanford Site under various surface cover conditions. One such study (*Estimated Recharge Rates at the Hanford Site*, PNL-10285) cites the results of radioisotopic tracer studies that were used to estimate recharge rates under various covers. This included an evaluation of the Ephrata Sandy Loam and Ephrata Stony Loam soil types present at 100-D/H, where the chlorine-36 tracer study indicated a recharge rate of 2.6 mm/yr under shrub and bunchgrass cover. The same report describes estimated recharge rates of 4.9 mm/yr and 17.3 mm/yr for cheatgrass and bare ground, respectively.

Source: Figure 4.1 from PNNL-17176, *200-BP-1 Prototype Hanford Barrier Annual Monitoring Report for Fiscal Years 2005 Through 2007.*

Figure 5-2. Prototype Hanford Barrier Cover in 2007 Dominated by Sagebrush (*Artemisia tridentata*), 13 Years after Plant Community Establishment

The recharge rate affects the velocity of porewater through the vadose zone. The flow velocity in the vadose zone is expected to have been greatest beneath the ponds, French drains, trenches, and cribs during the operational periods when percolation was at its greatest. A similar increase would have occurred in the vadose zone beneath unlined ditches. The velocity of downward movement is expected to have decreased after the waste disposal ceased, as the subsurface water content profile began to equilibrate to new surface conditions. After the waste disposal operations ended, alterations to the surface cover (including excavation of contaminated soil, backfilling the excavation with clean fill, revegetation, and stabilization) began to alter the net infiltration rate into the vadose zone.

The recharge input values to the STOMP models for the SSL calculation (based on irrigated agriculture land use) and the PRG calculation (based on conservation land use) were obtained from the vadose zone data package compiled in the Vadose Zone Hydrogeology Package (PNNL-14702). These data provided the basis for stipulating recharge rates in the two sequential models used to derive SSLs and PRGs. The first simulation, called the pre-2010 model, was used to establish the initial matric potential distribution in the vadose zone for subsequent modeling in the post-2010 model, which simulated the migration of water and contaminants to the underlying aquifer. Summarized in the following paragraphs, the recharge rates are discussed in detail in *STOMP 1-D Modeling for Determination of Preliminary Remediation Goals for 100 Area D, H, and K Source Areas* (ECF-HANFORD-11-0063) in Appendix F.

Two different scenarios based on future land use were evaluated in the post-2010 simulations. The first scenario, based on conservation land use and termed as the native vegetation recharge scenario, included the maturation of shrub-steppe in three periods from bare soil (Period 1) to mixed grass and shrub cover (Period 2) to mature shrub-steppe cover (Period 3). The second scenario, termed the irrigation recharge scenario, included the same bare soil period as for the native vegetation scenario (Period 1) followed by application of irrigation rates that were represented by infiltration increments over the native vegetation rate (Periods 2 and 3) (derivation of these increments is described below). Recharge for native vegetation is a function of both the surface soil type and the kind and extent of vegetation cover. The major soil types found at 100-D/H are Ephrata Sandy Loam and Stony Loam, Burbank Loamy Sand, and Rupert Sand. The initial recharge rates assigned to these soil types (that is, 1.5 mm/yr and 17.0 mm/yr, for established shrub-steppe and bare soil, respectively) were taken from PNNL-14702, Section 4.5. Recharge rates for each scenario were determined using the rates for the area-specific soil type(s). Rates were assumed to change over time in step function fashion for the two scenarios.

Recharge rates for each type of land cover, and for each surface soil type, were applied to the top boundary of the model from the year 0 to 1944 for the pre-operational period. The pre-operational period is an arbitrarily long period used simply to establish a steady state moisture profile for the specified recharge rate. The second period was from 1944 to 2010 for the operational period (Table 5-2) to reflect the effect of surface cover changes during the operational era (with vegetation removed) on soil moisture condition in the soil profile. Note that future land use has no impact on the pre-2010 recharge rates (that is, the recharge rates are identical for both recharge scenarios for the pre-2010 model).

Three recharge periods were specified in the post-2010 simulations to represent changes in recharge rates following the assumed future land use of each recharge scenario. For both scenarios, bare soil was assumed to continue to be the land cover above the waste site during the first recharge period, from 2010 to 2015.

Table 5-2. Recharge Rates for Pre-2010 Simulations at 100-D/H

Soil Type	Recharge Scenario	Recharge Rate for Different Period (mm/yr)	
		0–1944*	1944–2010
Ephrata Sandy Loam and Stony Loam	Native Vegetation	1.5	17.0
	Irrigation	1.5	17.0
Burbank Loamy Sand	Native Vegetation	3.0	52.0
	Irrigation	3.0	52.0
Rupert Sand	Native Vegetation	4.0	44.0
	Irrigation	4.0	44.0

* The period from CY 0 through 1944 is an arbitrarily long simulation period only for ensuring a steady state in soil column moisture contents and fluxes is reached for the specified recharge rate; there is no particular significance to starting from CY 0.

For the native vegetation recharge scenario, the second recharge period of 30 years in duration based on transition period duration information in *Regulatory Basis and Implementation of a Graded Approach to Evaluation of Groundwater Protection* (DOE/RL-2011-50), and represents grasses and shrubs covering bare soil. The third recharge period represents the establishment of a mature shrub-steppe that continues for the remainder of the simulation period. Thus, recharge rates decreases with time in this native vegetation recharge scenario as the vegetation cover transitions from bare soil towards a mature shrub-steppe state that is maintained thereafter.

Recharge rates for the irrigation scenario were estimated using the same approach employed to assess interim remediation at 100 Area waste sites (100 Area RDR/RAWP [DOE/RL-96-17]). These site assessments used levels calculated from RESidual RADioactivity (RESRAD) simulations for radioactive contaminants of interest that assumed total recharge was a combination of irrigation and nonirrigation (base case, or native vegetation) recharge rates. As the base case rates used in the RESRAD simulations differed from those adopted from the Vadose Zone Hydrogeology Package (PNNL-14702), the RESRAD equation for total recharge was solved to determine the rate attributable to irrigation alone. Based on that approach, the calculated nonirrigation total recharge rate was calculated to be 11.6 mm/yr (0.5 in./yr), and the recharge attributable to irrigation alone was 68.4 mm/yr (2.7 in./yr). This resultant irrigation rate attributable to irrigation alone was added to the native vegetation recharge rates to determine a recharge rate (total) for the irrigation scenario for each soil type in the SSL and PRG estimate simulations. The resulting recharge rates for native vegetation and irrigation recharge scenarios for each surface soil type are shown in Table 5-3.

For the SSL calculation, the maximum foreseeable recharge scenario (irrigation) was applied for each surface soil type (Table 5-3) at 100-D/H. For the PRG value calculation, the reasonably anticipated land use scenario (conservation with native vegetation) was applied for each surface soil type (Table 5-3).

Table 5-3. Recharge Rates for Post-2010 Simulations at 100-D/H

Soil Type	Recharge Scenario	Recharge Rate (mm/yr)		
		2010–2015 (Period 1)	2015–2045 (Period 2)	After 2045 (Period 3)
Ephrata Sandy Loam and Stony Loam	Native Vegetation	17.0	3.0	1.5
	Irrigation	17.0	71.4	69.9
Burbank Loamy Sand	Native Vegetation	52.0	6.0	3.0
	Irrigation	52.0	74.4	71.4
Rupert Sand	Native Vegetation	44.0	8.0	4.0
	Irrigation	44.0	76.4	72.4

5.4.2 Stratigraphy

The characteristics of material in the vadose zone affecting contaminant mobility are the particle size, permeability, and organic content of the lithologies present beneath the waste site. The primary mechanism for transport in the vadose zone is the flow of infiltrating water in response to gravitational and capillary forces. The pore networks (represented by grain-size distributions in each vertical lithologic sequence, the hydraulic and transport properties of each lithologic unit in the sequence, and the thickness of each lithologic unit) affect water flow and contaminant transport through the vadose zone. The unsaturated hydraulic conductivity of each lithologic unit varies with moisture content and, therefore, is a function of matric potential. The effects of the different lithologic units and variations in their individual thicknesses in 100-D/H on screening level and PRG values were determined by running STOMP simulations for a number of stratigraphic columns that represented the range of variations in 100-D/H.

The Hanford formation, Ringold Formation unit E, and the RUM were described in Chapter 3 (Section 3.4). Borehole data were used to identify representative stratigraphic columns for 100-D and 100-H. Two lithologic units are present in the 100-D vadose zone: the gravel-dominated Hanford formation and the Ringold Formation unit E. Only the Hanford formation is present in 100-H. Because of its coarse texture and higher hydraulic conductivity, the Hanford formation transmits water and dissolved or suspended contaminants more rapidly than the underlying Ringold Formation unit E does.

The contact between the Hanford formation and the Ringold Formation unit E forms a textural discontinuity that can result in temporary perching of water atop the interface during saturated vertical flow conditions in the vadose zone. During historical high-volume water discharges, a substantial quantity of water carrying dissolved Cr(VI) and other contaminants was transmitted vertically and laterally through the Hanford formation to locations considerably distant from the points of release. This was particularly notable at the 116-DR-1&2 Trench, where a substantial groundwater mound formed in the Hanford formation beneath the trench, and extended for thousands of meters inland both upstream and downstream, and across the horn area. The variability in stratigraphy observed at 100-D/H was recognized during design of the vadose transport simulation model used to evaluate the potential for migration of contaminants from the vadose zone to groundwater. The representative stratigraphic columns shown in Figures 5-3 and 5-4 illustrate how the variability was integrated into the transport simulations.

Using the June 2008 water table elevations to represent the highest water table, a conservative (smaller) thickness of the vadose zone was computed for each well and borehole. Borehole data were also used to estimate the thickness of each lithologic unit within the vadose zone and within the unconfined aquifer. The boreholes were divided into groups based on the proportion of each lithologic unit and total vadose zone thickness. A representative stratigraphic column was selected for each borehole group, resulting in six stratigraphic columns for 100-D (Figure 5-3) and two stratigraphic columns for 100-H (Figure 5-4) to be used in the STOMP simulations. Each column was assumed to contain also clean backfill to represent conditions following interim remediation. Backfill was assumed to replace the uppermost 4.5 m (15 ft) of each column. Additional details regarding the development of these stratigraphic columns are presented in *STOMP 1-D Modeling for Determination of Soil Screening Levels and Preliminary Remediation Goals for 100 Area D, H, and K Source Areas* (ECF-HANFORD-11-0063) in Appendix F.

5.4.3 Sorption

Mobility of constituents can be characterized using the soil/water K_d. This parameter is dependent upon the geochemical characteristics of the constituent, the properties of the transporting water, and the nature of the soil matrix. In general, organic constituents with lower molecular weights have lower K_d values than those with higher molecular weights. The K_d values of metallic radionuclides and nonradiological metals are primarily influenced by the charge sign (positive or negative) and magnitude of charge of the dominant species in a given geochemical environment (that is, positively charged ions tend to become attached to the negatively charged soil particles, while negatively charged ions tend to be repelled from soil particles and remain in solution).

Tritium is often used as a tracer for water molecules in column breakthrough testing and is assumed to define the zero K_d condition. It is conceivable that tritium substituted for hydrogen in a water molecule or hydroxyl species can exchange with water molecules adsorbed to solids or with hydroxyl groups on the surfaces of solid hydrous oxides (*K_d Values for Agricultural and Surface Soils for Use in Hanford Site Farm, Residential, and River Shoreline Scenarios: Technical Report for Groundwater Protection Project—Characterization of Systems Task* [PNNL-16531]).

Several metals of environmental concern exist in vadose zone material in more than one oxidation state: trivalent and pentavalent arsenic and trivalent and hexavalent chromium. The oxidation state and mineral speciation of these metals determines their relative mobility. Cr(VI), originally released as high-solubility sodium dichromate dihydrate, is relatively mobile in solution, being only weakly sorbed. Cr(VI); however, for other mineral compounds and a fraction of soil residues, it may be present in soil as relatively low-solubility mineral species such as potassium dichromate or lead chromate. Alternatively, Cr(III) is relatively immobile, being generally present as relatively insoluble precipitates, such as chromic hydroxide, $Cr(OH)_3$ (*Behavior of Metals in Soils* [EPA/540/S-92/018]).

Arsenic, Cr(VI) as chromate, sulfate, and nitrate are predominantly anionic species in the oxygenated, neutral to slightly basic pH porewater and groundwater observed in the subsurface at 100-D/H. Anionic species typically have relatively low K_d values and are considered to exhibit high to slight mobility in the 100-D/H subsurface environment. Cr(VI) may exist as the chromate ions $HCrO_4^-$ (predominant at pH <6.5) or CrO_4^{2-} (predominant at pH 6.5) and as the dichromate ion $Cr_2O_7^{2-}$ (predominant at concentrations >10 mM and at pH 2-6). In low ionic strength solutions, only the hexavalent chromate anion, CrO_4^{2-}, is found in oxidizing and near-neutral pH conditions.

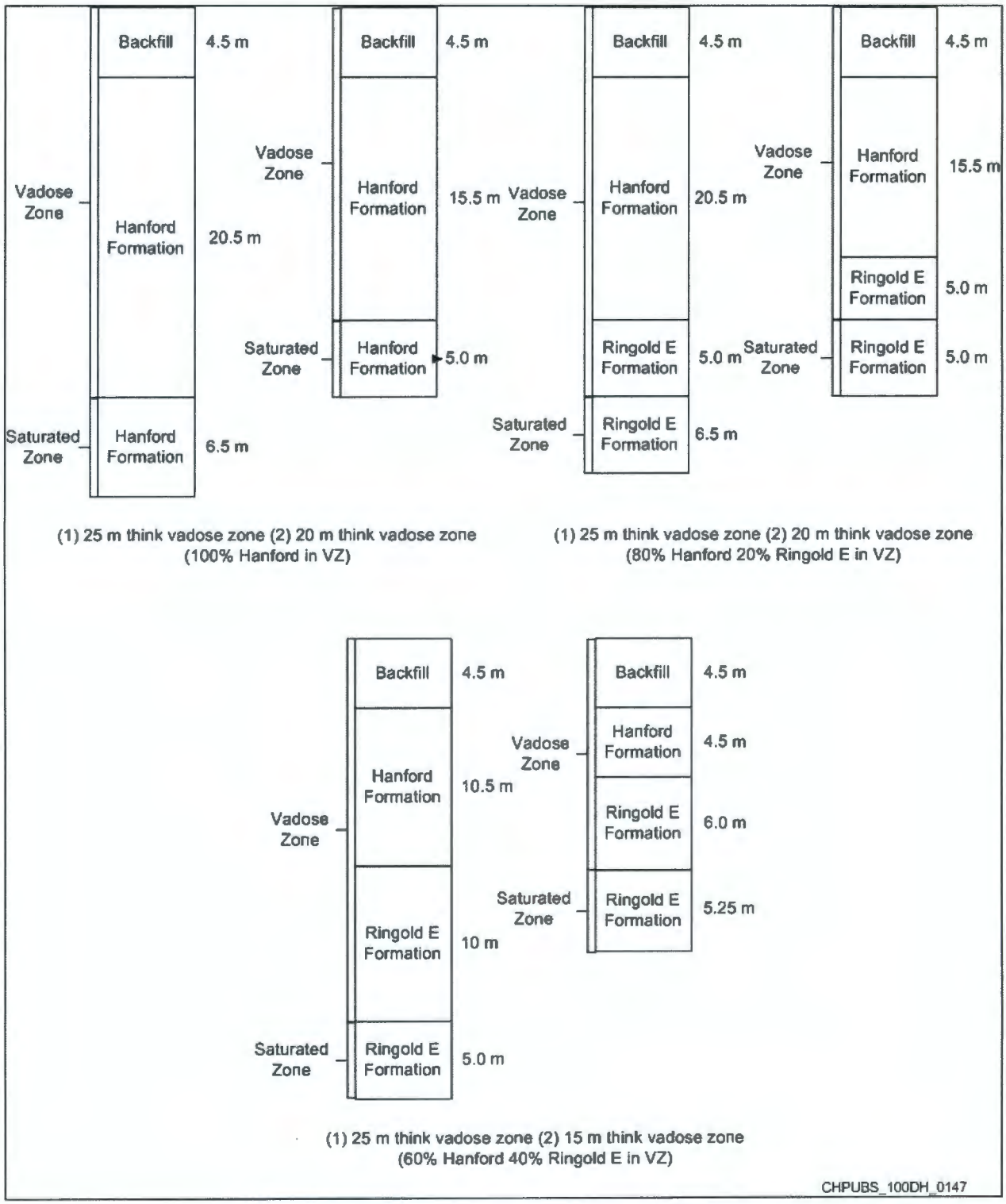


Figure 5-3. Six Representative Stratigraphic Columns for 100-D

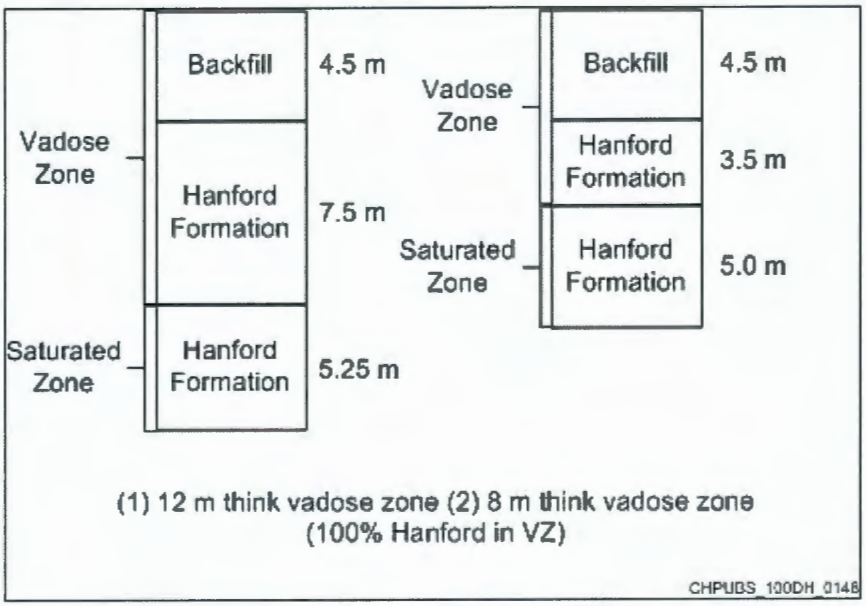


Figure 5-4. Two Representative Stratigraphic Columns for 100-H

Strontium-90 and uranium are most likely to exist as cationic species in the 100-D/H subsurface environment. Cationic species, which have higher K_d values than the anions, are typically considered moderately mobile to essentially immobile. Cations are adsorbed by clay minerals, oxides, and organic matter. Adsorption is pH dependent, increasing with increasing pH. Strontium-90 and uranium may become immobilized by forming precipitates with phosphate, carbonate, and hydroxide (*Behavior of Metals in Soils* [EPA/540/S-92/018]).

Table 5-4 summarizes the mobility of these contaminants. These contaminants are grouped by their relative mobility and the K_d values. The K_d values for the contaminants are provided in *STOMP 1-D Modeling for Determination of Soil Screening Levels and Preliminary Remediation Goals for 100 Areas D, H, and K Source Areas, Table A-1* (ECF-HANFORD-11-0063), in Appendix F, for the set of 192 nonradionuclide COPCs and 28 radionuclide COPCs. Analysis of batch leaching test results for soil samples collected from 100-D/H have led to derivation of a nonzero K_d for residual Cr(VI) as described in subsequent text.

Contaminants in the slight mobility group exhibit a high degree of interaction with vadose zone and aquifer solids and, as a result, migrate slowly through the vadose zone and aquifer. Their concentration in the vadose zone decreases rapidly with increasing depth, and their dissolved concentration in groundwater decreases dramatically with distance from a source or release point. The decrease in concentration is due to the relatively large fraction of the contaminant that interacts with, and become sorbed to, the solid materials in the vadose zone and aquifer. For this discussion and comparison, the slight mobility group includes contaminants that exhibit K_d values greater than 1 but less than 30 mL/g.

Table 5-4. Mobility of 100-D/H Contaminants of Potential Concern

Assumed K_d Value for Mobility Group	Inorganics	Radionuclides	Organics
High 0	Nitrate Sulfate	Tritium	None
Moderate 1	Hexavalent Chromium Uranium	None	Carbon Tetrachloride Chloroform
Slight 30	Arsenic	Sr-90	None
Immobile	Zinc Chromium (Cr+3)	Am-241 Cs-137 Pu-238 Pu-239/240 Co-60 Ni-63 Eu-152 Eu-154	None

Increasing Kd

Increasing interaction with soil

Increasing Rate of Migration

Contaminants considered essentially immobile sorb so strongly to vadose zone material that no migration is observed with infiltrating water under near-neutral pH in the vadose zone. Liquid waste sources with highly acidic or basic pH values or those that contained complexing agents may have transported these contaminants into the vadose zone at the time of disposal, but migration decreased as the liquid waste equilibrated with the vadose zone material. These constituents are not expected to reach the unconfined aquifer except at waste sites with subsurface release mechanisms and a very thin vadose zone. Those that may have reached the unconfined aquifer are not expected to migrate further through the aquifer. The low mobility contaminants are identified as those that exhibit K_d values greater than 30 mL/g, for comparison purposes.

The K_d values used for the STOMP simulations were taken from *Calculation of Nonradiological Soil Concentrations Protective of Groundwater Using the Fixed Parameter 3-Phase Equilibrium Partitioning Equation for the 100 Areas and 300 Area* (ECF-HANFORD-10-0442) in Appendix F. Simulations were run for 1,000 years to produce peak groundwater concentrations for a subset of the entire range of K_d values required for all COPCs. The subset was comprised of 26 K_d values ranging from 0 to 16 mL/g, because these values yield peak groundwater concentrations within the simulation period for the range of vadose zone thicknesses and lithologies simulated in this evaluation. Peak concentrations were estimated for the remaining K_d values by scaling with retardation factor (*STOMP 1-D Modeling for Determination of Soil Screening Levels and Preliminary Remediation Goals for 100 Area D, H, and K Source Areas* [ECF-HANFORD-11-0063] in Appendix F).

5.4.4 Matric Potential

The matric potential is a measure of the attractive forces between water and porous or fractured materials that are important during variably saturated flow conditions (*Vadose Zone Processes* [Selker et al., 1999]). Moisture content and hydraulic conductivity are functions of matric potential. These functions are typically nonlinear and must be determined for each medium. The combination of

matric potential gradients and gravity constitute the most important driving forces for vadose zone flow. The soil covers discussed in the preceding section will cause variations in the moisture and matric potential, in accordance with the net infiltration allowed by each cover.

Like pressure head, matric potential can be measured in the field and in the laboratory. In situ measurements of matric potential in the shallow Hanford Site vadose zone have been made using tensiometers and heat-dissipation sensors in lysimeters, pits, and boreholes (*Compendium of Data for the Hanford Site (Fiscal Years 2004 to 2008) Applicable to Estimation of Recharge Rates* [PNNL-17841]; *Hydrologic Characterizations Using Vadose Zone Monitoring Tools: Status Report* [PNNL-14115]; and *Soil Water Balance and Recharge Monitoring at the Hanford Site – FY09 Status Report* [PNNL-18807]).

The nonlinear relationship between water content and matric potential, frequently called the moisture retention or characteristic curve, can usually be measured in the laboratory. The much greater nonlinearity of the hydraulic conductivity and matric potential constitutive relation, termed the relative permeability, can typically be measured only over a small range of matric potential values. The remainder of the matric potential range must be inferred, because the hydraulic conductivity can decrease several orders of magnitude for a much smaller decrease in matric potential.

The "A Closed-form Equation for Predicting the Hydraulic Conductivity of Unsaturated Soils" (van Genuchten, 1980) alpha and *n* parameters used in the STOMP 1D simulations were selected to represent materials from 100-D/H and help define the relationship between moisture content in variably saturated media, the matric potential, and relative permeability. The inputs used in the simulations are described in detail in *STOMP 1-D Modeling for Determination of Soil Screening Levels and Preliminary Remediation Goals for 100 Areas D, H, and K Source Areas* (ECF-HANFORD-11-0063) in Appendix F.

5.5 Batch Leach Tests

Batch leach tests were conducted on soil samples from selected boreholes, wells, and test pits during the 100-D/H RI to establish estimated K_d values to support modeling needs, as described in the 100-D/H SAP (DOE/RL-2009-40). Contamination present in porewater within the bulk soil matrix was not analyzed, or accounted for separately, because the associated contaminant mass is included within the bulk leachate concentrations. The K_d calculations for each contaminant and each dilution ratio were performed using the analytical results from bulk soil analysis and leach testing of material collected from the same location.

5.5.1 Batch Leach Test Methodology

Batch leach tests were performed on soil and aquifer sediment samples using a leach procedure based on *Standard Test Method for Shake Extraction of Solid Waste with Water* (ASTM D3987-06). The procedure was performed using a 2 mm sieve to include the entire sand fraction based on the U.S. Department of Agriculture scheme for soil grain-size classification. Where insufficient sample mass with less than 2 mm particle diameter was available based on actual field conditions, a 3/8 in. mesh screen was used instead. Demineralized water, pH-adjusted according to EPA's West Coast recommendation, was used as the leaching liquid. Selected soil samples were leached at soil to water weight ratios of 1 to 1, 1 to 2.5, and 1 to 5, with one test in each series duplicated. Soil/water mixtures were placed in clean, water-tight sample containers (extraction vessels) and rotated end over end through the vessel centerline at a rate of about 30 rotations per minute for 18 hours. Following 18 hours of mixing, the soil/water slurry was filtered using a 0.45 μm filter. The leachate was analyzed for pH and conductivity. The leachate, after the 18-hour extraction period, and untreated soil were analyzed for arsenic, barium, cadmium, total chromium and Cr(VI), lead, selenium, and silver. Metals analysis for leachate and soil digests was performed using EPA Methods 6010, 6020, or 200.8 for ICP metals, as applicable (bulk soil was digested using EPA

Method 3050B or 3051 for metals and EPA Method 3060A for Cr(VI) to prepare for analysis). Separate aliquots of material were used for bulk soil analysis and leaching.

The K_d is calculated as the ratio of the contaminant sorbed to material to the contaminant in solution by the following equation:

$$K_d = \frac{(C_S \times M_S)(C_L \times V_L)}{M_S} \times \frac{1000}{C_L}$$

where:

K_d = soil-water distribution coefficient (mL/g)
C_S = contaminant concentration in bulk soil matrix before leaching (μg/g)
M_S = dry mass of soil used for leaching (g)
C_L = contaminant concentration in leachate (μg/L)
V_L = liquid volume used for leaching (L)

For each vadose zone soil sample, four replicate samples were analyzed for total soil metal concentrations. The average of the four measurements was used in the calculation of K_d. If one or more of the four replicates was found to be below reporting limit, the sample concentration was not considered reliable enough to report a K_d value. This was done because the reporting limit varied among replicates, with the reporting limit for one replicate often being several times that of another. This variation precluded the use of surrogate values such as half-reporting limits because of the significant uncertainty introduced by the variable reporting limits. In most cases, more than one or all four replicates were below reporting limit. For duplicate samples, the larger K_d of the two was reported. Often, an average soil concentration was calculated but the leachate water concentration was below reporting limit: in this event, the reporting limit was substituted in the calculation of a minimum K_d value, and a greater than (>) sign was placed before the calculated K_d value in the table.

The batch leaching of soil samples collected from 100-D/H was conducted on many uncontaminated soil samples collected during the RI characterization process. As a result, most of the 251 samples selected for batch leach testing were found to contain either no detectable residues of the analytes of interest in the bulk sample or no detectable analyte of interest in the extract; in those situations, the estimate of K_d is not quantifiable. The only analyte consistently detected in bulk samples and in the batch leaching extract was barium. Chromium was detected in a few batch leaching extract samples. Cr(VI) was detected in 77 sample intervals of 251 samples collected from 29 locations. Cr(VI) was detected in only 9 of 251 batch leach extracts from the same 29 locations. From this, it can be inferred for Cr(VI) that a significant fraction of chromium is non-leachable and that much of the chromium may be in Cr(III) form. There are, however, anomalies that are a key component of the CSM (Section 4.9) that may provide an explanation for the north plume that currently does not have an identified source. The details of the batch leach testing results are presented separately in *100-D and 100-H Remedial Investigation Distribution Coefficient Calculations* (0100X-CA-V0059).

5.5.2 Development of a Hexavalent Chromium Distribution Coefficient for Vadose Simulations from Batch Leach Testing Results

The results of the batch leach testing for Cr(VI) were further evaluated to identify a single derived K_d value to represent Cr(VI) behavior in the vadose zone model, which were then used to calculate peak groundwater concentrations used to derive SSLs and PRGs.

The results of leach tests described in *100-D and 100-H Remedial Investigation Distribution Coefficient Calculations* (0100X-CA-V0059) were analyzed (along with data from other OUs) to estimate an area-wide K_d value for residual Cr(VI) in the vadose zone. The assessment of K_d relies on collected field data and the corresponding laboratory analysis outlined in the 100-D/H SAP (DOE/RL-2009-40) to recommend a K_d value for use in the 100 Area. All methods used to calculate a value for K_d were outlined in the 100-D/H SAP (DOE/RL-2009-40) for each respective OU along the River Corridor. The objective of this evaluation is to recommend a K_d for use in the River Corridor, including 100-D/H. Details of the analysis are described in *Evaluation of Hexavalent Chromium Leach Test Data Conducted on Vadose Zone Sediment Samples from the 100 Area* (ECF-HANFORD-11-0165) in Appendix F. The batch leach testing evaluation of data collected in 100-D/H is presented in Appendix C. A total of 31 boreholes and wells were included in the batch leach testing, and most had very low leachate levels (< 100 μg/L) with two exceptions, at boreholes C7862 and C7866 that had higher leachate concentrations. These results, along with results from other River Corridor OUs, were used in *Evaluation of Hexavalent Chromium Leach Test Data Conducted on Vadose Zone Sediment Samples from the 100 Area* (ECF-HANFORD-11-0165) in Appendix F.

To establish a representative K_d value for use in the River Corridor, the calculated K_d values were adjusted for the amount of water used during the tests normalizing the values to the smallest soil:water extract ratio. This resulted in a 90th percentile exceedance K_d value of 0.8 mL/g (here, a 90th percentile exceedance means there is a 0.9 probability that the K_d value will be greater than or equal to 0.8 mL/g). Based on the batch leach results for soil samples collected from all of the 100 Area, a K_d value of 0.8 mL/g was designated as a conservative estimate for the lower limit on residual Cr(VI) K_d value for the River Corridor.

The results of batch leach testing using the method specified in the SAP are subject to some degree of uncertainty because of the test method and the computational approach to calculating resulting K_d. Specific areas of uncertainty identified to apply to the derivation of contaminant-specific K_ds are described in *Evaluation of Hexavalent Chromium Leach Test Data Conducted on Vadose Zone Sediment Samples from the 100 Area* (ECF-HANFORD-11-0165) (Appendix F). The following general topics may produce some uncertainty in derivation of K_d values from batch leach testing measurements:

- Differences in the pH of the extract solutions used to prepare the solid phase and liquid phase for analysis
- Dilution effects of batch leaching at differing solid to liquid ratios
- Variations in the linearity of the measured K_d values
- Effects of the coarse material (that is, gravel fraction) on the effective K_d of the formation
- Potential dilution effects of adding potable water to boreholes during drilling

5.6 Vadose Zone Modeling Methods and Results

Vadose zone transport simulations for this activity were performed using the STOMP code with a series of input values for sensitive variables based to the extent possible on conditions observed or measured at representative locations within the individual OUs. The model development for vadose zone models used to support this RI is comprehensively documented in *Model Package Report: Vadose Zone Model for the River Corridor* (SGW-50776). The numerical approach for calculations made using this model is described in detail in *STOMP 1-D Modeling for Determination of Soil Screening Levels and Preliminary Remediation Goals for 100 Areas D, H, and K Source Areas* (ECF-HANFORD-11-0063) in Appendix F,

but a brief summary is presented here. One-dimensional numerical models were constructed to represent the key facets of the conceptual model and were solved using the STOMP code (*STOMP: Subsurface Transport Over Multiple Phases Version 2.0: Theory Guide* [PNNL-12030]). The STOMP-W (water) mode was used to solve the Richards equation and the advection-dispersion equation that govern unsaturated water flow and dilute solute transport, respectively, under variably saturated conditions in porous media. The STOMP numerical simulations were performed to obtain the magnitude and time of peak groundwater concentrations for COPCs for the various recharge rates, sediment types and thicknesses, and hydraulic properties applicable to 100-D/H. Numerical transport simulations were run to simulate 1,000 years from the present (based on regulator comment). Thus, only a subset of the 100 Area COPCs that were likely to have peak groundwater concentrations occur within that period were simulated. Peak concentrations and times for other COPCs were estimated through scaling by a retardation coefficient. Simulated peak groundwater concentrations were then used to compute SSLs and PRGs.

Conceptually, the model represents a column of sediments that comprise a vadose zone underlain by an aquifer. Recharge-driven flow moves downward through the vadose zone, where it encounters contamination that is eventually transported to an underlying aquifer, across which a pressure gradient drives horizontal flow. At the start of each vadose transport simulation, the vadose zone is composed of a cover of clean fill with constant thickness as well as contaminated and uncontaminated sediments of varying thickness. The aquifer constitutes the base of the column with a minimum thickness of 5 m (16 ft), so that a 5 m (16 ft) long monitoring well screen could be simulated. Within the 100-D/H source OU, the vadose zone is composed of either the Hanford formation alone or a combination of the Hanford formation and Ringold Formation unit E. In contrast, the saturated zone can comprised of only the Hanford formation or only the Ringold Formation unit E. Underlying 100-D, the Ringold Formation unit E/RUM contact forms the base of the unconfined aquifer. Underlying most of the horn area and 100-H, the Hanford formation/RUM contact forms the base of the unconfined aquifer. The derived K_d for Cr(VI) of 0.8 mL/g was applied to Cr(VI) in all vadose zone strata in the model.

The STOMP code input parameters are summarized in Table 5-5 for the screening level and PRG calculations for 100-D/H.

Table 5-5. Summary of Selected Primary Fate and Transport Simulation Input Parameters Used with 1-D Model Implemented in the STOMP Code for Screening Level and Preliminary Remediation Goal Calculations in 100-D and 100-H Operable Units

Input Parameter (units)	Input Parameter Value		
Simulation Duration			
Simulation to establish initial hydraulic conditions (yr)	Calendar years 0 to 2010 (arbitrary long period to reach a steady state)		
Simulation to predict contaminant transport (yr)	Calendar years 2010 to 3010		
Upper Boundary Condition: Recharge (Deep Percolation) for Different Surface Soils [stepwise constant]			
Native Vegetation Recharge Scenario	**Ephrata sandy loam and stony loam**	**Burbank loamy sand**	**Rupert sand**
Recharge before 1944 (mm/yr)	1.5	3.0	4.0
Recharge 1944 to 2010 (mm/yr)	17.0	52.0	44.0

Table 5-5. Summary of Selected Primary Fate and Transport Simulation Input Parameters Used with 1-D Model Implemented in the STOMP Code for Screening Level and Preliminary Remediation Goal Calculations in 100-D and 100-H Operable Units

Input Parameter (units)	Input Parameter Value		
Recharge 2010 to 2015 (mm/yr)	17.0	52.0	44.0
Recharge 2015 to 2045 (mm/yr)	3.0	6.0	8.0
Recharge after 2045 (mm/yr)	1.5	3.0	4.0
Irrigation Recharge Scenario	**Ephrata sandy loam and stony loam**	**Burbank loamy sand**	**Rupert sand**
Recharge before 1944 (mm/yr)	1.5	3.0	4.0
Recharge 1944 to 2010 (mm/yr)	17.0	52.0	44.0
Recharge 2010 to 2015 (mm/yr)	17.0	52.0	44.0
Recharge 2015 to 2045 (mm/yr)	71.4	74.4	76.4
Recharge after 2045 (mm/yr)	69.9	71.4	72.4
Lateral Boundary Condition: Hydraulic Gradient (Saturated Portion)			
100-D hydraulic gradient (m/m)	0.0011		
100-H hydraulic gradient (m/m)	0.0021		

Hydraulic Parameters					
		Vadose Zone		Saturated Zone	
100-D Operable Units	**Backfill**	**Hanford formation**	**Ringold Formation**	**Hanford formation**	**Ringold Formation**
n_T total porosity (m^3/m^3)	0.276	0.28	0.28	0.28	0.28
n_D diffusive porosity (m^3/m^3)	0.262	0.25	0.28	0.25	0.28
α van Genuchten water retention function inverse air entry matric potential (1/cm)	0.019	0.029	0.013	0.021	0.008
n van Genuchten water retention function exponential fitting parameter (dimensionless)	1.4	1.378	1.538	1.458	1.66
s_r residual saturation (dimensionless)	0.162	0.02	0.054	0.04	0.093
$K_{s,h}$ saturated horizontal hydraulic conductivity (m/d)	0.517	4.03	0.819	55.5	22.4
$K_{s,v}$ saturated vertical hydraulic conductivity (m/d)	0.517	0.403	0.0819	5.55	2.24

Table 5-5. Summary of Selected Primary Fate and Transport Simulation Input Parameters Used with 1-D Model Implemented in the STOMP Code for Screening Level and Preliminary Remediation Goal Calculations in 100-D and 100-H Operable Units

Input Parameter (units)	Input Parameter Value				
		Vadose Zone		Saturated Zone	
100-H Operable Units	**Backfill**	**Hanford formation**	**Ringold Formation**	**Hanford formation**	**Ringold Formation**
n_T total porosity (m^3/m^3)	0.276	0.28	0.28	0.28	0.28
n_D diffusive porosity (m^3/m^3)	0.262	0.25	0.28	0.25	0.28
α van Genuchten water retention function inverse air entry matric potential (1/cm)	0.019	0.029	0.013	0.021	0.008
n van Genuchten water retention function exponential fitting parameter (dimensionless)	1.4	1.378	1.538	1.458	1.66
s_r residual saturation (dimensionless)	0.162	0.02	0.054	0.04	0.093
$K_{s,h}$ saturated horizontal hydraulic conductivity (m/d)	0.517	4.03	0.819	97.6	3.70
$K_{s,v}$ saturated vertical hydraulic conductivity (m/d)	0.517	0.403	0.0819	9.76	0.370

Both 100-D and 100-H Operable Units	**Backfill**	**Hanford formation**	**Ringold Formation**
ρ_p particle density (g/cm^3)	*Calculated from bulk density and porosity;* $\rho_p = \rho_b / (1 - n_T)$		
ρ_b bulk density (g/cm^3)	1.94	1.93	1.93
m Mualem relative permeability function fitting parameter (dimensionless)	$m = (n-1)/n$		
β Mualem relative permeability function exponential term	0.5		

Transport Parameters	
D_m molecular diffusion (m^2/s)	Conventional model with $D_m = 0$
α_L longitudinal dispersivity (m)	0 (dispersivity neglected; conservative assumption with regard to peak concentration)

Table 5-5. Summary of Selected Primary Fate and Transport Simulation Input Parameters Used with 1-D Model Implemented in the STOMP Code for Screening Level and Preliminary Remediation Goal Calculations in 100-D and 100-H Operable Units

Input Parameter (units)	Input Parameter Value
α_T/α_L dispersivity anisotropy ratio (dimensionless)	Not applicable (one-dimensional model)
K_d distribution coefficient (mL/g)	26 K_d values over a range were simulated directly, and the results interpolated to provide values for specific contaminants based on their respective K_d. The 26 simulated K_d values were: 0, 0.00001, 0.0001, 0.001, 0.01, 0.02, 0.04, 0.06, 0.08, 0.1, 0.2, 0.3, 0.4, 0.5, 0.6, 0.7, 0.8, 0.9, 1.0, 2.0, 4.0, 8.0, 10.0, 12.0, 14.0, and 16.0. The specific contaminant K_d values for all contaminants evaluated (191 nonradionuclides and 28 radionuclides) are listed in ECF-HANFORD-10-0063 (found in Appendix F) in Attachment A, Tables A-1, A-2, and A-3; and Attachment B, Tables B-1, B-2, and B-3.

Sources: Mualem, 1976, "A New Model for Predicting the Hydraulic Conductivity of Unsaturated Porous Media."
van Genuchten, M.Th, 1980, "A Closed-form Equation for Predicting the Hydraulic Conductivity of Unsaturated Soils."
Note: Details on the basis for all parameters in this table are found in Appendix F (*STOMP 1-D Modeling for Determination of Soil Screening Levels and Preliminary Remediation Goals for 100 Area D, H, and K Source Areas* [ECF-HANFORD-11-0063]).

Based on observations of contaminant distribution made from analysis of soil samples from RI borings and wells, spatial distributions of contamination were identified for use in initial flow and transport simulations. Numerous contaminants were found to be distributed throughout the thickness of the vadose zone; others exhibited limited vertical distribution. For the source distribution, all the contaminants were grouped into two categories, one with low K_d (<2 mL/g) and another with high K_d (≥2 mL/g). The process of identification of low K_d range and high K_d range is presented in *Conceptual Basis for Distribution of Highly Sorbed Contaminants in 100 Areas Vadose Zone* (SGW-51818) in Appendix F. A uniform concentration of 1 mg/kg was applied in the entire vadose zone below the clean backfill up to 0.5 m (1.6 ft; two grid blocks) above the water table for the low K_d contaminants referred to here as 100:0 profile. Initial concentration in the 0.5 m (1.6 ft) zone above the water table was not applied due to physical presence of capillary fringe and water table movement in the periodically rewetted zone that would result from river stage fluctuations. Placing the initial mass at the water table can also result in boundary effects and extreme-concentration gradients. For the high K_d contaminants, 70 percent of the vadose zone below the clean backfill was assumed to be contaminated and referred to as the 70:30 profile.

The use of a unit-concentration source, typically 1 mg/kg, is a common practice in simulation of transport of multiple contaminants in a simulated system where transport is controlled by K_d. The transport results for individual constituents can then be estimated by simply scaling the results for the unit-concentration source against the actual constituent concentration. Although the same contaminant distribution was applied to calculate both the SSLs and the PRGs, a different infiltration/recharge scenario was applied to each simulation. The SSL simulation uses a conservative recharge scenario based on irrigated agriculture land use. The PRG simulation, alternatively, uses a recharge scenario based on the expected future land use: conservation, with a native plant population re-established at the land surface.

5.6.1 Initial Contaminant Distribution

The initial simulation configuration was developed assuming that interim remedial actions have been undertaken and that 4.6 m (15 ft) of clean backfill is present starting at the ground surface. Note that this assumption is approximately representative of conditions at any particular waste site, where interim action excavations may have proceeded to greater or lesser depth.

For the initial source distribution of contaminants in the vadose zone used to simulate transport for the purposes of calculating SSLs and PRGs, all the contaminants were grouped into two categories: one with lower sorption ($K_d < 2$ mL/g) and another one with higher sorption ($K_d \geq 2$ mL/g). The basis for segregating contaminants by their K_d values for purposes of assigning an initial vertical distribution of contamination in the vadose zone is discussed in *Conceptual Basis for Distribution of Highly Sorbed Contaminants in 100 Areas Vadose Zone* (SGW-51818). For lower K_d contaminants and strontium-90, a uniform unit-source concentration of 1 mg/kg was applied in the entire vadose zone below the clean backfill to a depth 0.5 m (1.6 ft) above the water table (that is, down to the capillary fringe) and is referred to as the 100:0 profile. For the higher K_d category, a uniform concentration unit-source of 1 mg/kg was applied in the upper 70 percent of the vadose zone below the clean backfill and is referred to as the 70:30 profile.

An exception to the assignment of initial source distributions based on K_d is made in the case of strontium-90, owing to the observed distribution of this COPC at depths greater than the upper 70 percent of the profile in several locations. This distribution is a legacy of the greater mobility of this contaminant in the operational era under different geochemical and hydraulic conditions than are prevalent in the present or anticipated in the future. Accordingly, despite a K_d value higher than the 2 mL/g threshold, the 100:0 profile is applied for evaluation of strontium-90.

5.6.2 Simulation Duration

The simulation time for calculating screening values and PRGs was limited to 1,000 years based on regulator comment. The peak groundwater concentration within the 1,000-year simulation was used to determine the SSL and PRG values.

5.7 Groundwater/Surface Water Protection Screening Level and Preliminary Remediation Goal Development

The SSLs and PRGs were developed to provide an basis to identify waste site contaminants that may pose a threat for continuing contribution to groundwater and surface water contamination (*STOMP 1-D Modeling for Determination of Soil Screening Levels and Preliminary Remediation Goals for 100 Area D, H, and K Source Areas* [ECF-HANFORD-11-0063] in Appendix F). The SSLs and PRGs are based on an assumption of uniform vadose zone contamination over a bounding portion of the soil column (100:0 initial source profile for lower K_d contaminants plus strontium-90, 70:30 profile for higher K_d contaminants except strontium-90). SSLs were calculated assuming a recharge scenario based the maximum foreseeable recharge scenario, irrigated agriculture. In contrast, PRGs were calculated assuming a recharge scenario based on the expected land use, conservation. Comparison of site EPCs to SSLs provided a conservative screening level to identify those constituents that likely pose a continuing threat under the bounding condition of irrigated agriculture land use. Those sites that failed this screening were then compared to PRGs based on expected land use to identify those constituents that likely pose a continuing threat under conservation land use.

SSLs and PRGs represent the maximum concentration, whether mass concentration (for nonradionuclides) or activity concentration (for radionuclides) in soil, of specific contaminants that can remain in the vadose zone without causing an exceedance of a potential ARAR (Chapter 8) or risk-based

value for protection of groundwater or surface water. The value of a SSL or PRG for a particular COPC depends on a small number of key factors:

- COPC initial vertical distribution (the portion of the vadose zone that is contaminated, and the distance of that contamination relative to the water table below)
- Recharge rate, which is a function of land cover (surface soil type and vegetation cover); for future conditions, this in turn is a function of the expected land use (irrigated agriculture or conservation with native vegetation)
- Interactions between vadose zone geology and water movement (hydraulic parameters)
- COPC characteristics (sorption and decay, where applicable)

Some of these key factors are represented in a conservative approach to ensure the PRGs are not overestimated. For example, the COPC initial distributions are intentionally bounding relative to observed COPC vertical distributions in the vadose zone (Chapter 4).

Peak concentrations were calculated by running multiple simulations using STOMP for a set of K_d values that represent the range of K_d values of contaminants for 100-D/H. Values for specific constituents were selected from within the results based on their particular representative K_d. These results were used to calculate PRGs and SSLs for each constituent.

The peak concentrations beyond the simulated K_d values were estimated based on interpolation among the peak concentrations for the two closest K_d values (which encompass the K_d value of the contaminant) for which STOMP calculations have been performed. For the contaminants with higher K_d values, the peak concentration within 1,000 years was less than 0.0001 μg/L for most cases. In such cases, the PRG is calculated using 0.0001 μg/L peak concentration, and the ">" sign is placed before the PRG value. These cases occurred at different K_d value for different soil columns; this maximum condition for 100-D/H occurred at $K_d > 1$.

The practical quantitation limit (PQL) for soil concentration for aroclors is 0.017 mg/kg. If the screening value or the PRG value for aroclors was calculated less than 0.017 mg/kg, then it was set to 0.017 mg/kg.

The modeled PRG value selected for 100-D/H for Cr(VI) was calculated to be 21 mg/kg. The PRG value was subsequently constrained to a maximum value of 6 mg/kg because the K_d value used in the model was derived from experiments with soil concentration less than 6 mg/kg. A PRG level of 2 mg/kg has been selected as the level used from the interim action ROD.

In the RI/FS process, waste sites are evaluated using PRGs. Known measured concentrations of COPCs are compared to the appropriate set of PRG values. If the concentration for one or more COPC exceeds either the groundwater-specific or surface water-specific PRG, then the site is carried into the FS, where options for addressing risks posed by the site will be determined and evaluated using the CERCLA criteria.

PRGs were determined for each COPC by simulating peak groundwater concentrations for all representative stratigraphic columns and surface soil types, assuming a particular recharge scenario and contaminant source distribution, and then selecting the smallest PRG value calculated from the resulting peak concentrations. Screening levels, which identified analytes to be designated as COPCs at each 100-D/H waste site, were determined in a similar manner to PRGs, but the most conservative source distribution and recharge scenarios were used in the SSL simulations. This section summarizes the modeling approach for calculation of SSL and PRG values that are protective of both surface water and

groundwater, describes their application, and identifies the underlying assumptions, conservatism, and uncertainties in the calculations. Calculation of a PRG or SSL is straightforward:

- Simulate variably saturated transport of a unit concentration or activity of waste-derived COPCs from a specified interval in the vadose zone in which flow and solute transport are driven by a particular recharge scenario.
- Identify the peak groundwater concentration or activity.
- Divide the peak into the potential ARAR (Chapter 8) or risk-based value for groundwater or surface water and correct for units to give the groundwater-specific or surface water-specific PRG or screening level.
- Repeat the calculation for different COPCs, contaminant mass distribution, net infiltration rate, vadose zone geology, or vadose zone thickness.

Calculation of PRG values for the 100-D/H source OU included the use of 1D numerical fate and transport simulations for some COPCs and scaling computations for the remaining COPCs. The STOMP code was selected to perform the simulations based on its ability to simulate the vadose zone features, events, and processes relevant to calculating PRGs in the 100 Area. In addition, STOMP was selected to satisfy the other code criteria and attributes identified in *Regulatory Basis and Implementation of a Graded Approach to Evaluation of Groundwater Protection* (DOE/RL-2011-50), which describes the basis for using STOMP in this type of evaluation.

Several methodologies, model inputs, and assumptions for computing remedial goals were used to determine interim remedial action goals as part of the 100 Area RD/RAWP (DOE/RL-96-17, Appendix B). The remedial action goals for nonradioactive constituents were developed using the "100 times rule" (WAC 173-340, Method B) and remedial action goals for radioactive constituents were developed using the RESRAD model to estimate resultant doses. Although development of the interim action remedial action goals included transport simulations using RESRAD only for radionuclides, the PRGs in this calculation for both radioactive and nonradioactive constituents were calculated with STOMP.

5.7.1 Identification of Peak Groundwater Concentrations

Peak groundwater concentrations were calculated along a portion of the domain's downgradient boundary corresponding to the top 5 m (16 ft) of the aquifer. The average concentration for the topmost 5 m (16 ft) was assumed a reasonable estimate of the groundwater concentration that would be measured within a 6 m (20 ft) long monitoring well screen that straddled the water table. The concentration was conservatively estimated by calculating it at the aquifer beneath the downgradient edge of the footprint of a representative waste site, from which the peak groundwater concentration of contaminant and breakthrough time were determined. For COPCs not simulated in STOMP, peak groundwater concentration and year of occurrence were calculated by scaling with the retardation factor (*STOMP 1-D Modeling for Determination of Soil Screening Levels and Preliminary Remediation Goals for 100 Area D, H, and K Source Areas* [ECF-HANFORD-11-0063] in Appendix F).

5.7.2 Site-Specific Modeling

The graded approach for vadose zone modeling permitted application of site-specific modeling in cases where the individual waste site conditions were not adequately (conservatively) represented by the default SSL and/or PRG simulations) as described in Section 5.1.

A review of the RI borehole profile data reported in Chapter 4 was conducted to identify specific waste sites and specific higher sorption COPCs ($K_d > 2$) that may not be conservatively represented by the 70:30 profile that was used to develop SSL and PRG values. The RI borehole profile data that exhibited concentrations of such higher K_d constituents were identified and then the list reduced to eliminate those cases that were as follows:

- From boreholes that did not sample the lower 30 percent of the vadose zone
- For COPCs for which there are no background values
- Reporting concentrations in the lower 30 percent of the vadose zone were within the range of background
- For COPCs with $K_d > 25$
- For strontium-90

The reason for the exclusion of COPCs with $K_d > 25$ was that results from vadose zone modeling to develop SSLs show that COPCs with K_d values higher than 25 result in nonrepresentative (NR) values based on peak groundwater concentrations simulated within 1,000 years for the 100:0 profile; thus there is no need to evaluate these cases further. Strontium-90 was excluded because it was decided to assign the 100:0 profile to this constituent through the 100-D/H area based on its prevalence throughout the vadose zone in many locations, presence in groundwater, and recognition that this contaminant is a recognized risk driver in the 100 Area.

Based on the evaluation above, the following waste sites and COPCs were identified as potential cases for which the 70:30 profile may be non-conservative:

- 116-D-1A (trench), neptunium-237
- 116-D-7 (retention basin), antimony
- 116-DR-9 (retention basin), acenaphthene
- 116-H-1 (trench), phenanthrene
- 116-H-1 (trench), antimony
- 116-H-4 (pluto crib), antimony
- 116-H-6 (solar evaporation basin), antimony
- 116-H-7 (retention basin), antimony
- 116-H-7 (retention basin), molybdenum
- 118-H-6 (reactor fuel storage basin), neptunium-237

For each case on the above list, the conservatism of the 70:30 profile was tested by simulating these cases again using the most appropriate representative stratigraphic column (Figures 5-3 and 5-4) for each of these sites. Simulations were performed in pairs: once with the actual vertical contamination profile reported in Chapter 4, and again using the 70:30 profile. The peak groundwater concentration predicted by the model were obtained from in each case in these pairs of simulations and compared. The conservatism of the 70:30 profile was considered validated if:

1. The simulated peak groundwater concentration obtained from observed contaminant distribution was less than the peak groundwater concentration obtained from the 70:30 profile.
2. The simulated peak groundwater concentration was less than the MCL for the constituent simulated.

The results of the above analysis for the identified sites and COPCs that were identified as potentially non-conservative revealed that the 70:30 profile was conservative in all instances; therefore no site-specific evaluations were required for the purpose of establishing SSL and PRG values. The evaluation of the conservatism of the 70:30 profile is included in *STOMP 1-D Modeling for Determination of Soil Screening Levels and Preliminary Remediation Goals for 100 Area D, H, and K Source Areas* (ECF-HANFORD-11-0063) in Appendix F.

5.7.3 Comparison of Vadose Zone EPCs to SSLs and PRGs

The definition of, and process for identification of, COPCs is presented in Section 6.2.1.3. The EPCs of the residual contamination of COPCs for each waste site and soil group (for example, overburden, shallow, shallow focused, and deep) as indicated in the CVP data for the previously remediated waste sites are calculated through the process presented in Chapter 6 and summarized here. The EPC of each waste site and soil group is the UCL-95 of the mean concentration or the maximum detected if too few detections were available to estimate a UCL-95 value. The waste site-specific EPC of each constituent was compared to the model-derived SSLs protective of groundwater and protective of surface water. No waste sites or constituents had EPCs that exceeded the groundwater protection or surface water protection SSLs in 100-D/H.

If a waste site soil group COPC EPC exceeded the model-derived SSL protective of groundwater or the model-derived SSL protective of surface water, then it was carried through to the second step for waste site assessment. The second step compares the EPCs of the COPCs that emerged from the screening level (SSL comparison) to model-derived PRGs protective of groundwater and surface water.

Recharge rates in 100-D/H would be greatest in the future under an irrigated agriculture land use. This land use is the basis for the screening analysis based on the irrigation recharge scenario defined for SSL derivation. The SSL represents the maximum constituent mass or activity soil concentration that will not result, under the irrigation recharge scenario, in a downgradient groundwater or surface water concentration exceeding the federal and state criteria listed in *STOMP 1-D Modeling for Determination of Soil Screening Levels and Preliminary Remediation Goals for 100 Area D, H, and K Source Areas* (ECF-HANFORD-11-0063) in Appendix F.

Recharge rates in 100-D/H, as well as in the rest of the 100 Area, are expected to decrease after demolition and remediation activities are complete and the native xerophyte plant cover is reestablished under the reasonably anticipated land use of conservation. This is the basis for the native vegetation (that is, non-irrigated) recharge scenario defined for the PRG derivation. The PRG values represent the maximum constituent mass or activity soil concentration, under the native recharge scenario, that will not result in downgradient groundwater or surface water concentration exceeding the federal and state criteria listed in *STOMP 1-D Modeling for Determination of Soil Screening Levels and Preliminary Remediation Goals for 100 Areas D, H, and K Source Areas* (ECF-HANFORD-11-0063) in Appendix F.

There were no waste sites with SSL exceedances; therefore, no sites were screened against the PRGs. Human health and ecological screening is conducted as part of the risk assessment. The results of the SSL comparison are shown in Tables 5-12 and 5-13 for 100-D and 100-H waste sites, respectively. Uncertainties that may affect the interpretation of the comparison of site-specific EPCs to the SSLs and PRGs are discussed in Section 5.7.

5.7.4 COPCs Retained for Assessment in Feasibility Study

As noted in the preceding subsection, no previously remediated waste sites exceeded screening levels, and none of those sites are carried into the FS. The waste sites that are not yet remediated are carried into the FS for evaluation, with the COPCs identified for those waste sites based on process knowledge.

5.8 Groundwater Contaminant Migration Assessment

The behavior of contaminants currently known to exist in groundwater at 100-D/H was evaluated using computer simulations to describe estimated future conditions. The approach to simulating groundwater contaminant migration and the technical basis for selecting specific input values are described in the following subsections.

5.8.1 Factors Affecting Contaminant Mobility in the Saturated Zone

The transport velocity of contaminants in the saturated zone is dependent on the groundwater flow velocity and the specific retardation factor of each individual contaminant. The groundwater flow velocity, and hence the rate of contaminant transport, increases with increasing hydraulic conductivity and hydraulic gradients, yet decreases with increasing porosity. The retardation factor of contaminants in a given stratigraphic unit increases with increasing K_d. Each of these variables is dependent, in turn, upon other factors: historical discharges, natural recharge, artificial recharge (in this case, effluent disposal to trenches), sorption, groundwater treatment systems, and Columbia River stage variations. Discussion of each of these factors follows.

5.8.1.1 Historical Discharges

Site-related discharges of liquids and contaminants at 100-D/H entered the groundwater system at various times and locations during the historical operations at the OU. Low-concentration Cr(VI) and, periodically during upset conditions, low-concentration fission products entered the ground at locations of reactor cooling water discharge. These locations included leakage from the 116-D-7, 116-DR-9, and 116-H-7 Retention Basins, and direct discharges to the 116-DR-1&2 and 116-H-1 Trenches. These discharges resulted in the development of a large groundwater mound that extended for thousands of meters in all directions from the release points. The mound consisted primarily of discharged reactor cooling water that displaced the original groundwater. This groundwater mound would have exhibited a fairly uniform Cr(VI) content of about 700 μg/L, the concentration of that constituent in the working cooling water stream. This water stream likely also included various nonradiological contaminants associated with the cooling water treatment process, in addition to nuclear fuel and activation and fission products associated with periodic reactor fuel failures. The groundwater flow velocities associated with the cooling water mound during reactor operations at 100-D were measured at 3 m/day (10 ft/day) between the retention basins and the river. Across the horn area (between 100-D and 100-H) the groundwater flow velocities were 10 m/day (30 ft/day). Groundwater flow velocity between the retention basin at 100-H and the river was measured at 6.1 m/day (20 ft/day). After cessation of cooling water discharges, the groundwater mounds rapidly subsided to near-natural groundwater elevations; however, the body of contaminated water was still in place and is assumed to have moved slowly (at a groundwater flow velocity of about 0.3 m/day [1 ft/day]) toward the Columbia River. Underlying 100-H, Cr(VI) in the footprint of the former groundwater plume remains mobile; concentrations have decreased over that area since the implementation of the interim action pump-and-treat system in 1997.

Other historical releases to ground that apparently affected groundwater include spills and leaks of water treatment chemicals (including sodium dichromate dihydrate solution at high concentration, sodium hydroxide solution, sulfuric acid solution, sodium chloride solution, and alum solution [hydrated potassium aluminum sulfate]). These releases appear to have occurred over the operating period in or near the water treatment facilities at the D, DR, and H Reactors. French drains located near the bulk chemical receiving area at 100-D received spills and leaks. Numerous areas of chromium-stained vadose soil have been identified at both 100-D and 100-H as described in Section 4.2. The current groundwater plumes are a result of some of these releases reaching groundwater. Residual contamination in the vadose zone

(particularly Cr[VI]) may potentially contribute to continuing groundwater contamination in these areas; however, the actual timing and concentration/mass of released contaminants is uncertain.

The 183-H sedimentation basins (originally part of the H Reactor cooling water treatment facilities) were removed from service at the end of reactor operations in 1965. In 1973, the eastern portion of the sedimentation basins were converted for use as an evaporation basin for radioactive waste generated in the 300 Area. Waste evaporation activities continued until 1985, when the remaining basin was closed. Groundwater monitoring at nearby wells indicates that contaminants from the evaporation operation entered groundwater soon after the operation commenced. Substantial concentrations of uranium, nitrate, sulfate, and sodium were observed in groundwater.

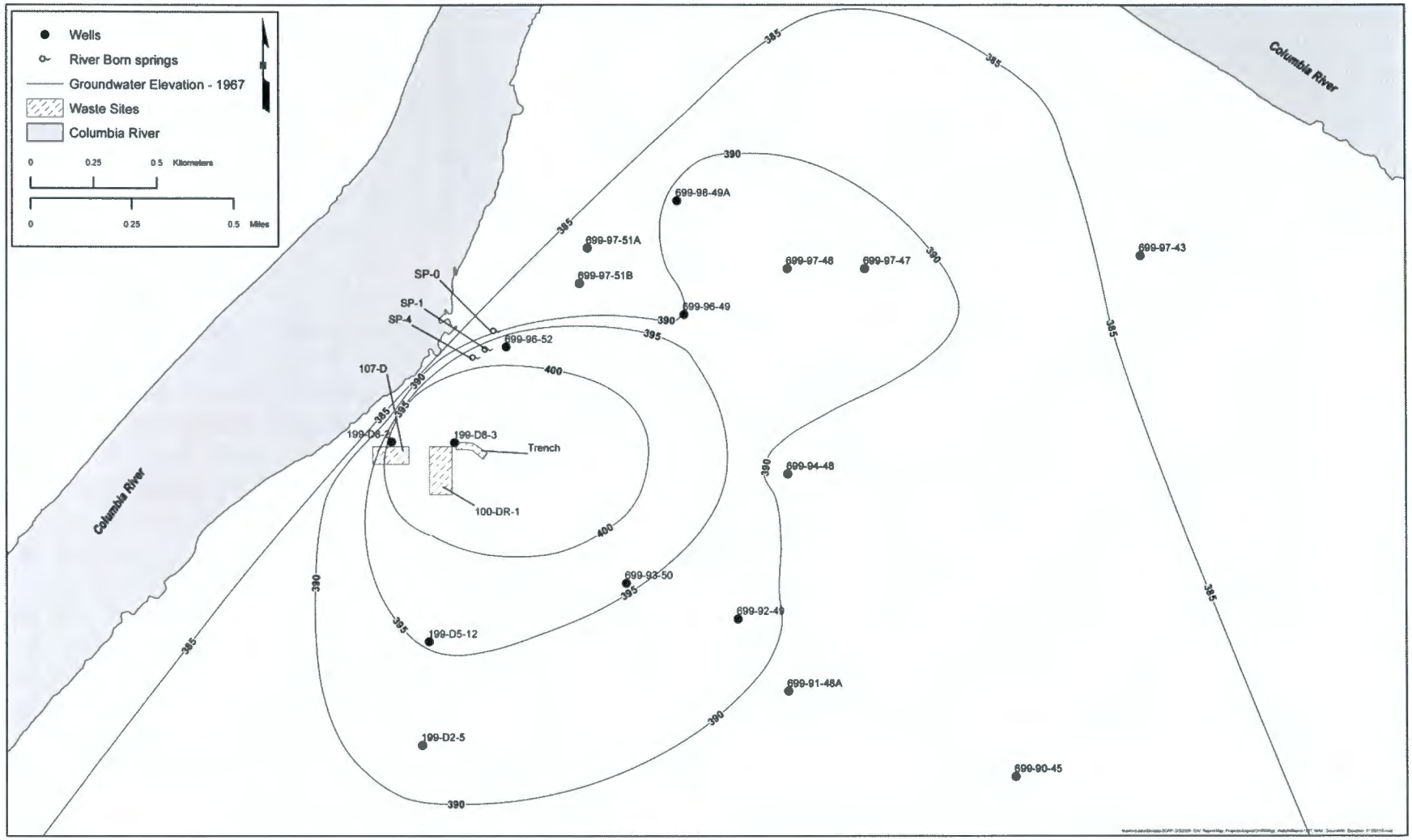
5.8.1.2 Recharge

The recharge rate at 100-D/H has changed significantly from the period of reactor operations to current conditions. Large volumes of liquid wastes percolating through the vadose zone caused a mound in the water table in 100-D/H during operations. After the production ceased, the large-volume effluent disposal ended, and the hydraulic gradients began to return to natural conditions.

Groundwater recharge results from the net infiltration of precipitation, leaks, and liquid waste disposal. The high recharge rates present during operations affected groundwater elevations and resulted in changes in groundwater flow velocity and direction. During the operation of the D/DR Reactors, cooling water containing Cr(VI), other water treatment chemicals, and radionuclides was briefly held at leaking retention basins (*Status of the Ground Water Beneath Hanford Reactor Areas* [HW-77170]). An estimated combined leakage rate of 5,663,369 L/day (1,496,104 gal/day) resulted in mounding of the unconfined aquifer water table until February 1967 (Figure 5-5). An additional cause of recharge water unique to the source area remediation is the addition of water for dust suppression. This water is applied during excavation activities to prevent fugitive dust from contaminated and potentially contaminated soil. Water is applied in 18,927 L (5,000 gal) trucks which is equivalent to 13.2 L/min (3.5 gal/min) for each truck added assuming 100 percent infiltration. However, it is assumed that a large fraction of dust-suppression water either evaporates, or is removed along with excavated soil.

5.8.1.3 Effluent Disposal to Trenches

To evaluate the fate of effluent discharge to infiltration trenches and the associated effect on groundwater flow directions, an infiltration test was conducted between March and June 1967 (*Ground Disposal of Reactor Cooling Effluent* [BNWL-CC-1352]). During this test, 12,745,480,000 L (3,367,000,000 gal) of reactor coolant effluent were disposed to the joined 107-DR disposal trenches (116-DR-1&2 Trench). This is equivalent to approximately 141,616,500 L/day (37,400,000 gal/day), approximately 25 times the daily infiltration rate from the leaking storage basin. Hydrographs from wells near the infiltration area indicate that significant groundwater mounding occurred in response to the infiltration, and did not fully dissipate until 1968 to 1969 (Figure 5-6). A detailed discussion on this infiltration test is presented in Chapter 3, Section 3.7.1.



Source: Modified from BNWL-CC-1352, *Ground Disposal of Reactor Cooling Effluent.*

Figure 5-5. 107-DR Disposal Trenches 1 and 2 Infiltration Test Base Map and Initial Water-Level Distribution

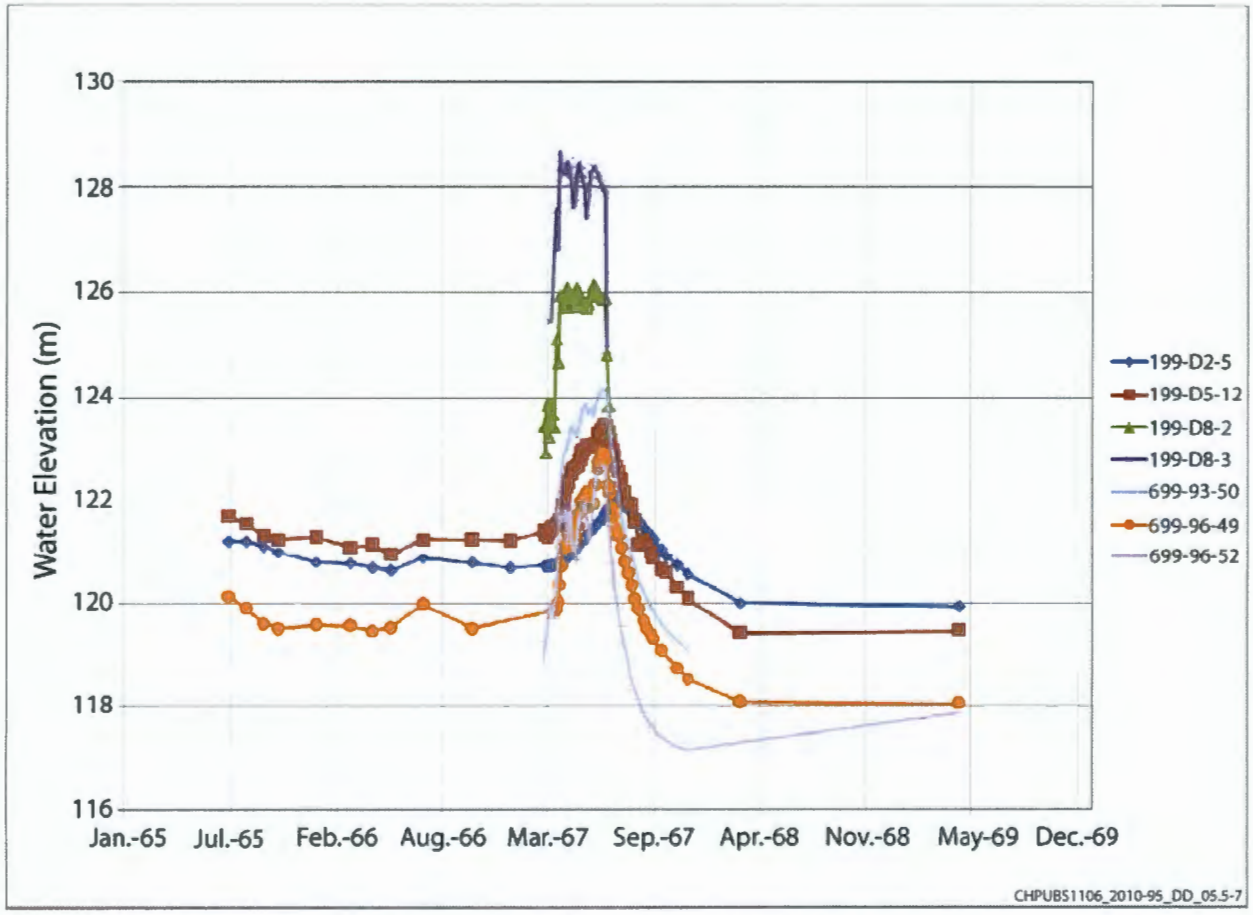


Figure 5-6. Water Level Hydrographs at Selected Wells near 107-DR Disposal Trenches 1 and 2

To characterize the extent of mounding created by the infiltration test and to evaluate its effect on flow direction and potential contaminant migration pathways, an analysis was undertaken as part of this RI/FS to accomplish the following:

- Map groundwater levels measured during and after the infiltration test.
- Evaluate the resulting pattern of hydraulic gradients and resulting groundwater flow directions during each of these periods.

To accomplish this, groundwater levels that were measured in monitoring wells every few days during and after the infiltration test were compiled. Groundwater levels were converted to elevations and placed on a map using a water-level mapping technique that incorporates mounding in response to injection of water (*Collection and Mapping of Water Levels to Assist in the Evaluation of Groundwater Pump-and-Treat Remedy Performance* [SGW-42305]). This technique combines universal kriging of the groundwater levels with a linear trend and an additional term to account for aquifer response to infiltration at a point source of water.

Because the Columbia River influences groundwater elevations, an estimate of the river stage throughout the infiltration test period was required to construct the groundwater level maps. Although groundwater level data are available throughout the period of interest, only recent river stage data are available. To obtain estimated river stage elevations throughout the period of interest, an empirical function was developed for each river gauge location by developing a correlation between daily river gauge values for

the period 2006 to 2009 and daily average dam discharge rates from the Columbia River gauge below Priest Rapids Dam for the same period. This empirical function was then used to calculate an approximate river stage at each river gauge for each event on which groundwater levels are available during the period of the infiltration test. Linear interpolation between the river gauge water level estimates provided a continuous estimate of the river stage along the shoreline for inclusion in the groundwater level mapping.

Semi-annual groundwater level maps were prepared using combined groundwater levels and river stage data. Two examples of groundwater level maps are provided on Figure 5-7, illustrating approximate groundwater flow patterns during and at some time following the infiltration test. The analysis suggests that substantial mounding had developed during reactor operations before the infiltration test. The infiltration test caused further mounding, which resulted in an increase in the hydraulic gradient and groundwater flow velocity across the horn area. Maps reflecting conditions during the infiltration test (such as Figure 5-7, inset 1), and for the time following operations until mounding dissipated, indicate that the infiltration of effluent water provided a mechanism to transport contaminants at relatively low concentrations inland of source areas associated with the D Reactor, and across the horn area toward the H Reactor, under conditions of both high and low river stage. Similar but lesser effects would be expected to have occurred because of leakage from the retention basins. Groundwater level maps that were prepared using data obtained sometime after the cessation of operations and the infiltration test (such as Figure 5-7, inset 2) suggest that migration from the D Reactor toward the H Reactor may occur under current conditions during times of high river stage. However, at times of low river stage, migration would be primarily toward the Columbia River as discussed in Section 3.7.2.

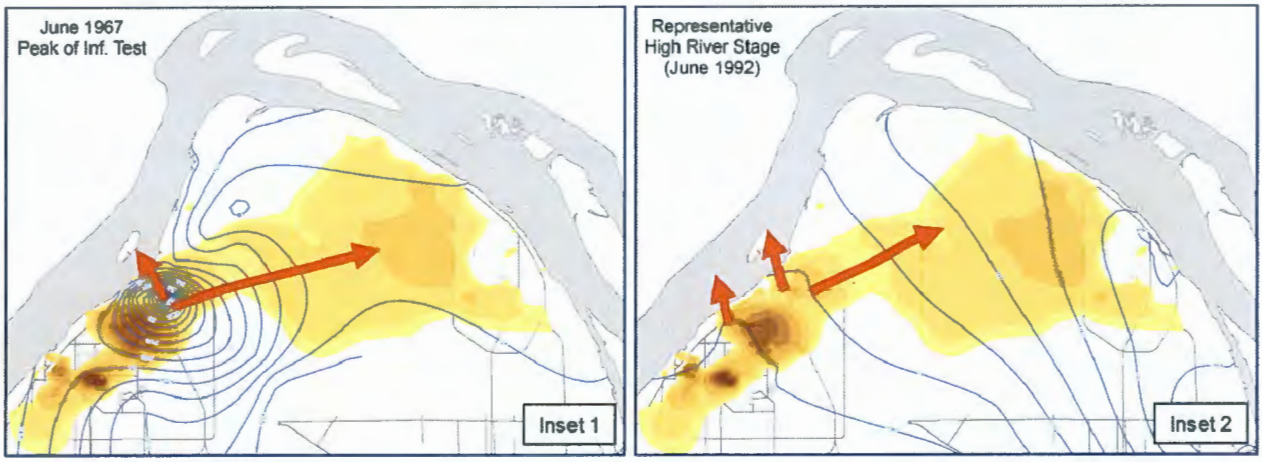


Figure 5-7. Groundwater Level Distributions and Potential Plume Migration Pathways due to the 100-D Trench Test

5.8.1.4 Sorption

The groundwater constituents identified in Chapter 4 and discussed in detail in the annual groundwater reports for 100-D/H are total chromium, Cr(VI), sulfate, nitrate, zinc, strontium-90, carbon tetrachloride, tritium, and chloroform. Groundwater COPCs are identified in Chapter 6. This list is limited to contaminants in groundwater at present. Continuing contributions of mass from vadose zone source sites are not included because these sites were determined to meet a protectiveness criterion (see vadose zone evaluation in Section 5.4) or will be subject to a remedy through the FS process to preclude functioning as a continuing source. Cr(VI), tritium, nitrate, and sulfate are highly mobile (Table 5-4) and migrate at the same velocity as groundwater under ambient geochemical conditions. Strontium-90 in 100-D/H groundwater is a divalent cation and is considered slightly mobile in near-neutral or slightly basic

groundwater. Chloroform is a moderately mobile chlorinated solvent Cr(VI) is assumed to exhibit a K_d of zero in the groundwater transport simulations.

5.8.1.5 Groundwater Treatment Systems

The 100-D/H pump-and-treat systems are described in detail in Chapter 1. The 100-HR-3 pump-and-treat system began operating in 1997. The 100-DR-5 system was added in 2007. The 100-DX system began operating in 2010. The 100-HX system began operating in 2011. Note that the 100-HR-3 and DR-5 systems are no longer operational. The 100-DX/HX pump-and-treat system was installed to expand the area of influence and to increase the capacity of the treatment system. The systems are operated to intercept Cr(VI) before it reaches the Columbia River, with the objective of limiting concentrations of Cr(VI) at concentrations that do not exceed the 10 µg/L AWQC. The estimated future effects of pump-and-treat operations under selected scenarios are discussed in the FS portion of this report.

The capture efficiency of these systems was evaluated using a numerical groundwater flow model. The modeling approach and the results of the evaluation are presented later in this report.

The ISRM barrier discussed in Chapter 1 was installed to chemically reduce dissolved Cr(VI) in groundwater to the essentially immobile and less toxic Cr(III) species. The barrier was constructed by injecting sodium dithionite with potassium carbonate and potassium bicarbonate pH buffers with the purpose of creating a residual reduced zone within the aquifer that would provide ongoing removal of Cr(VI) from groundwater. The ISRM barrier has exhibited variable performance. The barrier has been supplemented by extraction wells and an ex situ treatment system in the highest concentration portion of the plume to prevent it from extending to the Columbia River.

5.8.1.6 Columbia River Stage Variations

As discussed in Chapter 3 (Section 3.3.1), the Columbia River is a discharge boundary for the aquifer system, and the unconfined aquifer is in direct communication with the river along the shoreline of 100-D/H. Changing river stage influences groundwater elevations several hundred meters inland, but with diminishing influence at increasing distance from the river. At the river, high river stage can be more than 3 m (10 ft) higher than low river stage. Groundwater elevations have varied by up to 1.0 m/day (3.3 ft/day) in some wells nearest the river and up to approximately 1.8 m (6 ft) over the season in a few wells. This results in the PRZ being largest for sites nearest the river and smaller with increasing distance from the river. During high river stages, the inland flow direction in areas near the river and the reduced hydraulic gradients in more inland areas reduce the annual net groundwater flow velocity toward the Columbia River and the migration rate of contaminants dissolved in the groundwater. Groundwater in the horn area appears to be more dramatically influenced by high river stage conditions; the groundwater flow direction appears to move directly from the river on the upstream side of the horn area into the aquifer and move toward the downstream side of the horn area near the 100-H Area.

5.8.2 Saturated Zone Modeling Methods and Results

A groundwater flow and contaminant transport model has been developed and calibrated for purposes of remedy design evaluation in the 100 Area. The model development and calibration was comprehensively documented in *Conceptual Framework and Numerical Implementation of 100 Areas Groundwater Flow and Transport Model* (SGW-46279) in Appendix F. The groundwater flow model was constructed using the U.S. Geological Survey (USGS) modular groundwater flow model, MODFLOW ("A Modular Three-Dimensional Finite-Difference Ground-Water Flow Model" [McDonald and Harbaugh, 1988]). Particle tracking was performed using the USGS program, MODPATH (*User's Guide for MODPATH/ MODPATH-PLOT, Version 3: A Particle Tracking Post-Processing Package for MODFLOW, the U.S. Geological Survey Finite-Difference Ground-Water Flow Model* [Pollock, 1994]). The modular

3D multispecies transport model for simulation of advection, dispersion, and chemical reactions of contaminants in groundwater systems (MT3DMS) was used to simulate the contaminant plume migration (*MT3DMS: A Modular Three-Dimensional Multi-Species Transport Model for Simulation of Advection, Dispersion, and Chemical Reactions of Contaminants in Groundwater Systems; Documentation and User's Guide* [Zheng and Wang, 1999]).

A summary description of the model development and deployment is presented below. Additional details are presented in *Modeling of RI/FS Design Alternatives for 100-HR-3* (ECF-100HR3-11-0114) in Appendix F.

5.8.2.1 Model Structure

The finite difference grid for the groundwater flow model encompasses all 100 Area OUs. The grid is constructed so that the northwest and northeast boundaries of the flow model parallel and abut the Columbia River. The model extends southward, toward Gable Butte and Gable Mountain. The grid spacing is relatively coarse (about 100 m [328 ft]) throughout much of the domain, but it is refined (15 m [49 ft]) near each 100 Area OU to support remedy evaluations.

Groundwater flow is simulated as 3D using four layers. These layers represent the Hanford formation (always present in Layer 1, across the model domain) and the Ringold Formation unit E (typically represented by Layers 2 through 4, except east of 100-D where it is absent and therefore all model layers represent the Hanford formation). Throughout much of the western half of the modeled area (including 100-K and 100-D), the water table lies within the Ringold Formation unit E sands, whereas toward the east and north of the modeled area (including 100-H and 100-F), the water table lies within the Hanford formation sands and gravels. Near 100-BC, the water table fluctuates between the two formations.

The base of the model was set as the top of the RUM where present and the top of the basalt where the RUM is absent, which typically occurs in the southern portions of the model approaching Gable Butte. The geologic characterization was compiled as part of the model data packages (*100-HR-3 Remedial Process Optimization Modeling Data Package* [SGW-40781]; *100-KR-4 Remedial Process Optimization Modeling Data Package* [SGW-41213]; *Geohydrologic Data Package in Support of 100-BC-5 Modeling* [SGW-44022]; *Geohydrologic Data Package in Support of 100-FR-3 Modeling* [SGW-47040]). This characterization depicts the lateral facies transition from Ringold Formation unit E in the west and south of the model domain, to the Hanford formation sands and gravels in the east and north of the model domain, between 100-D and 100-H.

The establishment of the initial plume condition for the simulation was intended to describe an approximation of the current contaminant distribution and applied a conservative approach. Contaminant measurements at individual monitoring wells were distributed uniformly across the model layers at measurement locations. Subsequent transport simulations allow for movement of contaminants between the layers in the saturated zone. The model domain of the saturated zone is subject to contaminant distribution uncertainty because of variability in actual well construction and screen placement. The placement of contaminants across the full thickness of the aquifer in the initial condition is expected to be conservative in light of vertical profile measurements that indicate actual substantial variation in vertical contaminant distribution.

The principal aquifer property specified in the flow model is the spatially varying hydraulic conductivity of the saturated aquifer materials. The hydraulic conductivity distribution in the model was developed based on the information included in the model data package and a pilot-point approach implemented in the model calibration process. Estimates of hydraulic conductivity compiled as part of the model data package were tabulated and assigned to their corresponding aquifer unit. The values for the aquifer

hydraulic conductivity that resulted from the model evaluation process are 19 m/day (62 ft/day) for Ringold Formation unit E in 100-D and 63 m/day (206 ft/day) for the Hanford formation.

Natural recharge resulting from precipitation was specified based on information included in *Groundwater Data Package for Hanford Assessments* (PNNL-14753). An electronic version of the recharge package developed in that report was obtained, and the data were spatially distributed to the model grid cells and then adjusted during the model calibration process. Based on the results of the model calibration process, the recharge value was set equal to 12 mm/yr throughout much of the model domain. This natural recharge rate (12 mm/yr) appears to differ from natural recharge rates applied for vadose zone simulations to develop SSLs and PRGs but is in fact consistent when considered in context. The mean recharge rate applied for the groundwater model is applicable to the large area represented by that model (encompassing all of the 100 Area), which includes substantial variability in land surface soil types and vegetation cover. Moreover, this rate was calibrated to present day hydraulic conditions. In contrast, the recharge rates applied for vadose zone modeling apply to the smaller, local-scale waste site representations. For present day conditions (bare soil), the recharge rates for waste sites range from 17 to 52 mm/yr (see Table 5-5 for recharge rates under bare soil, 2010–2015). The groundwater model surface area averages recharge from these bare soil areas (such as waste sites) with larger areas of native vegetation cover with recharge rates ranging from 1.5 to 4.0 mm/yr (see Table 5-5 for recharge rates under mature shrub-steppe native vegetation cover, after 2045). Thus, the 12 mm/yr rate is consistent when considered in context as a spatially averaged value over diverse land cover conditions during the model simulation period.

Artificial recharge (for example, infiltration from surface reservoirs) was specified based on reported values (*In Situ Redox Manipulation (ISRM) Annual Report Fiscal Year 2007* [DOE/RL-2008-10]).

Effective porosity and specific yield values for the entire aquifer were identified from published sources and revised during the model calibration and are equal to 18 and 10 percent, respectively. Both values are within the range of values documented in previous investigations for the Hanford Site (*Development of a Three-Dimensional Ground-Water Model of the Hanford Site Unconfined Aquifer System: FY 1995 Status Report* [PNNL-10886]). Riverbed conductance values were also determined during calibration, separately for the stretches of the Columbia River within each area, to reflect the variability in geologic conditions in each area.

The groundwater flow model was calibrated to data included in the model data packages for each OU, through a combined manual and automated process. The model calibration was facilitated by using the PEST parameter estimation software program (*User's Manual for PEST Version 11* [Doherty, 2010]) and post-processing programs that calculate water level responses to stresses. The model was calibrated to data from January 2006 to December 2010. Calibration focused on the transient response of water levels to transient pumping and river stage stresses, and how these compare to values measured at wells at each OU. In addition, maps of water level contours calculated by the model were compared to contours included in published reports to ensure that the simulated hydraulic gradient magnitude and direction agree with prior independent interpretations.

Further details about all parameter values used in the model are included in the comprehensive modeling report (*Conceptual Framework and Numerical Implementation of 100 Areas Groundwater Flow and Transport Model* [SGW-46279]).

5.8.2.2 Contaminant Transport Processes

The migration of Cr(VI) in response to current and projected extraction and injection well operations in 100-D/H was simulated to support remedy design evaluation; total chromium is anticipated to follow

similar patterns. In addition to modeling of Cr(VI), transport simulations were performed for strontium-90 and nitrate to evaluate corresponding migration patterns because of the current and projected extraction and injection well operations. Transport simulations were based on the following:

- Transient flow fields calculated by the groundwater flow model
- An initial distribution for each COC in groundwater
- A dual-domain formulation representing plume migration in a dual-porosity continuum with mass transfer between the mobile and immobile domains

The dual-domain concept applies multiple K_d values to the migration of selected contaminants to better describe their overall observed migration. For example, a highly mobile contaminant that typically exhibits a K_d of 0 may be assigned a small K_d value for a portion of the aquifer system. This small K_d reflects a migration retardation effect that may actually result from any of a number of physical processes, ranging from ionic exchange to retention of dissolved contaminants within low-permeability portions of the aquifer. The dual-domain approach is used to address some of the uncertainty in contaminant transport estimates when exact parameter values are not well known and observed contaminant behavior suggests that actual migration is variable.

Nitrate and strontium-90 passing through the ion exchange treatment system are not removed under the current treatment process. They are therefore recirculated in the aquifer via injection at the injection wells connected to each treatment plant. Nitrate and strontium-90 concentrations injected back into the aquifer are equal to the blended influent concentration at the treatment plant.

Recent studies by PNNL (such as *Geochemical Characterization of Chromate Contamination in the 100 Area Vadose Zone at the Hanford Site* [PNNL-17674]) suggest (based on a series of column and batch experiments to investigate Cr(VI) mobility during advective transport under saturated conditions) that Cr(VI) within the vadose zone of the 100 Area exhibits migration characteristics that may be more complex than can be represented using simple advection. According to these tests, although the majority of the mass is highly mobile and migrates by advection, Cr(VI) mass can be held in heterogeneous parts of the aquifer of low hydraulic conductivity. This immobile Cr(VI) constitutes a longer-term continuing source to the mobile domain, facilitated by mass transfer between the domains. Based on these observations, the migration of Cr(VI) can be described by a dual-domain (or dual-porosity) approach that divides the aquifer into two domains: the mobile and immobile, using the bimodal grain-size distribution noted in Chapter 3. Advective transport occurs predominantly in the mobile domain, while mass transfer occurs by diffusion between the mobile and immobile domains.

MT3DMS, which is discussed in detail in the modeling analysis (*Modeling of RI/FS Design Alternatives for 100-HR-3* [ECF-100HR3-11-0114]), supports the use of a dual-domain formulation to simulate the transport of a contaminant in groundwater. The following parameters must be specified for the dual-domain formulation: the fraction of mobile and immobile domains; the mass transfer coefficient between the mobile and immobile domains; and K_d describing sorption within the mobile and immobile domains. For the 100 Area transport model, it was assumed that for Cr(VI) sorption occurs within the immobile domain, and that no sorption occurs within the mobile domain. Sorption occurs both in the mobile and immobile domain for strontium-90. No sorption was assumed for nitrate, which was simulated using a single-domain, single-porosity formulation. The parameter values for the dual-domain formulation for transport simulation are listed in *Modeling of RI/FS Design Alternatives for 100-HR-3* (ECF-100HR3-11-0114), Table 3-1 (Appendix F). Further details on the development of the dual-domain parameters can be found in the comprehensive modeling report (*Conceptual Framework and Numerical Implementation of 100 Areas Groundwater Flow and Transport Model* [SGW-46279]).

Radioactive decay was considered for strontium-90 ($T_{1/2}$ = 28.8 years).

5.8.2.3 Contaminant Initial Conditions

Initial conditions for the COCs in groundwater within the 100-D/H were obtained based on maximum sampled COC concentrations at each monitoring location during CY 2011 as reflected on the plume depictions presented in *Hanford Site Groundwater Monitoring for 2011* (DOE/RL-2011-118, Section 2.5). For Cr(VI) the plume depiction for Fall 2011 was used. The initial Cr(VI) distribution in the 100-HR-3 OU is shown on Figure 5-8. Figures 5-9 and 5-10 show the resulting distributions for strontium-90 and nitrate, respectively.

The contaminant plumes depicted on Figures 5-8 through 5-10 are the starting point for the contaminant transport simulations as of January 2011. Note that solid triangles depict pump-and-treat wells, pointing up for extraction wells and pointing down for injection wells. Open circles are used on the figures to depict inactive wells at a given time. Therefore, these figures show most of the wells near 100-H as inactive because they did not enter service until the 100-HX system became operational in October 2011. The wells used for the old 100-HR-3 system were still active in January 2011. In later figures that represent 2012, these wells will be depicted as triangles pointing in the appropriate direction as the systems are turned on later in 2011 and shut down at the end of 2012.

5.8.3 Model Deployment

The groundwater flow and transport model was used to simulate flow conditions and Cr(VI), strontium-90, and nitrate plume migration patterns, assuming the reported operation of the interim pump-and-treat system in the 100-HR-3 OU during 2011–2012, which includes the 100-DX and 100-HX treatment plants and the associated extraction and injection wells. *Modeling of RI/FS Design Alternatives for 100-HR-3* (ECF-100HR3-11-114), Table 3-2 (Appendix F) includes a list of all pump-and-treat wells with their corresponding pumping rates. For the base case simulation, pump-and-treat extraction and injection wells w assumed to be turned off at the end of CY 2012 and ambient flow conditions are simulated thereafter. The model input parameters are summarized in Table 5-6.

Predictive simulations were based on transient-state (that is, time-varying) conditions in the aquifer that reflect water level changes because of river stage variation. The modeling period corresponds to a 77-year corresponding to the period 2011 to 2012 with current pump-and-treat systems operating, and a 75-year simulation period thereafter without pump-and-treat operations, representing a "no further action" condition. For the first 27 years, the modeling period consists of a series of 12 monthly stress periods that are repeated in the same sequence. The stress periods correspond to monthly average river stages, each representing the average river stage for the particular calendar month over the period 2006 to 2010 (excluding 2007 values, when the river stage variation pattern was inconsistent to the other years). It is assumed that these conditions are representative of the typical conditions in the field and that future conditions will not vary significantly from these conditions. For the remaining 50-year period, a single stress period is used, with the river stage elevation remaining constant, reflecting annual average conditions corresponding to 2006 to 2010 average elevations for the month of January. This approach allows for generation of a substantial body of simulation information to evaluate the apparent effects of seasonal river stage transients on transport, but also provides for efficient long-term transport estimate and plume behavior calculations.

Modeling results will support system performance evaluation considering attainment of river protection and aquifer cleanup levels. For this purpose, an estimate of hydraulic containment in 2012 and plume depictions at selected intervals are developed and discussed in the following text.

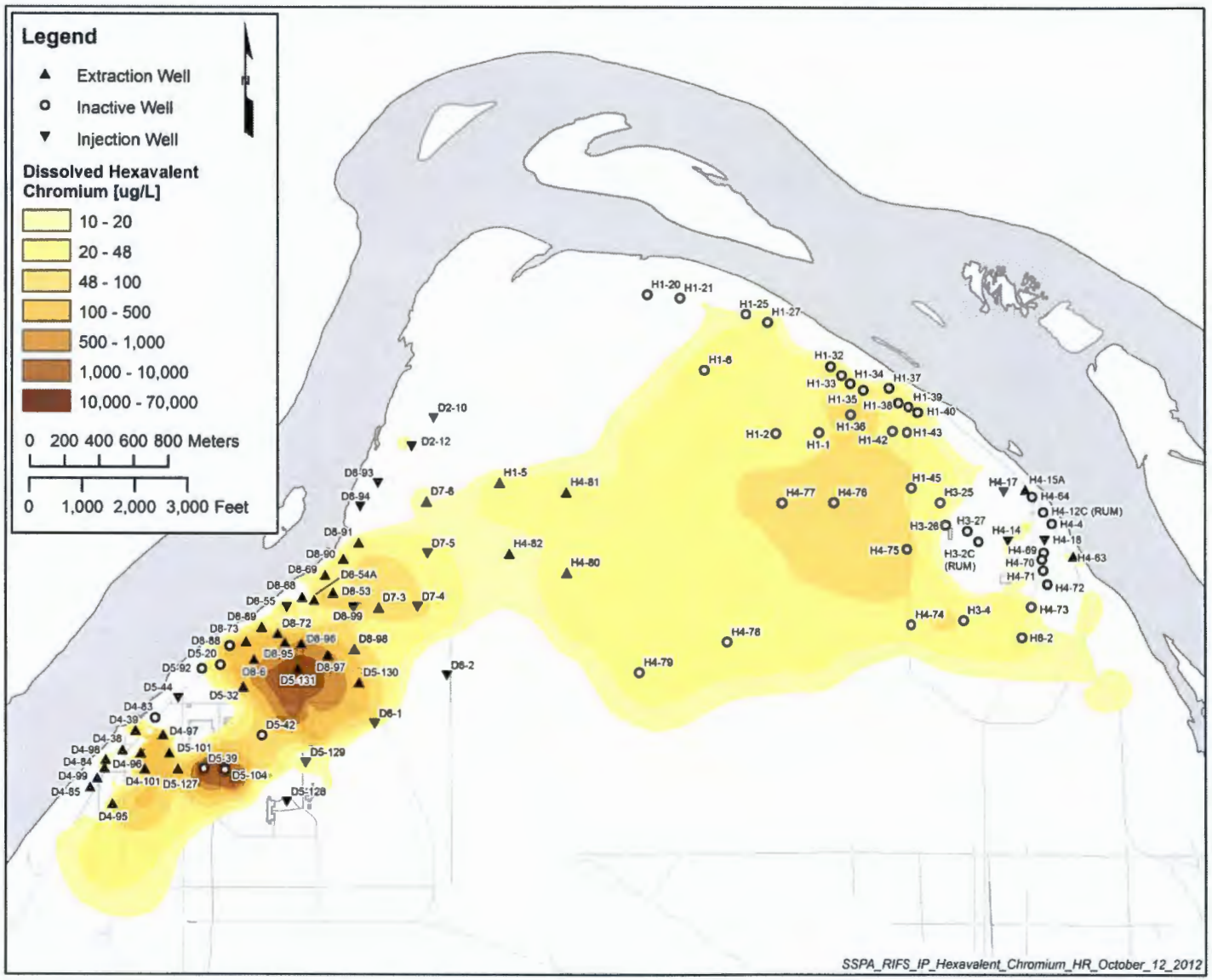


Figure 5-8. 100-DX/HX Pump-and-Treat Well Configuration and Approximate Extent of Dissolved Cr(VI) in 100-D/H

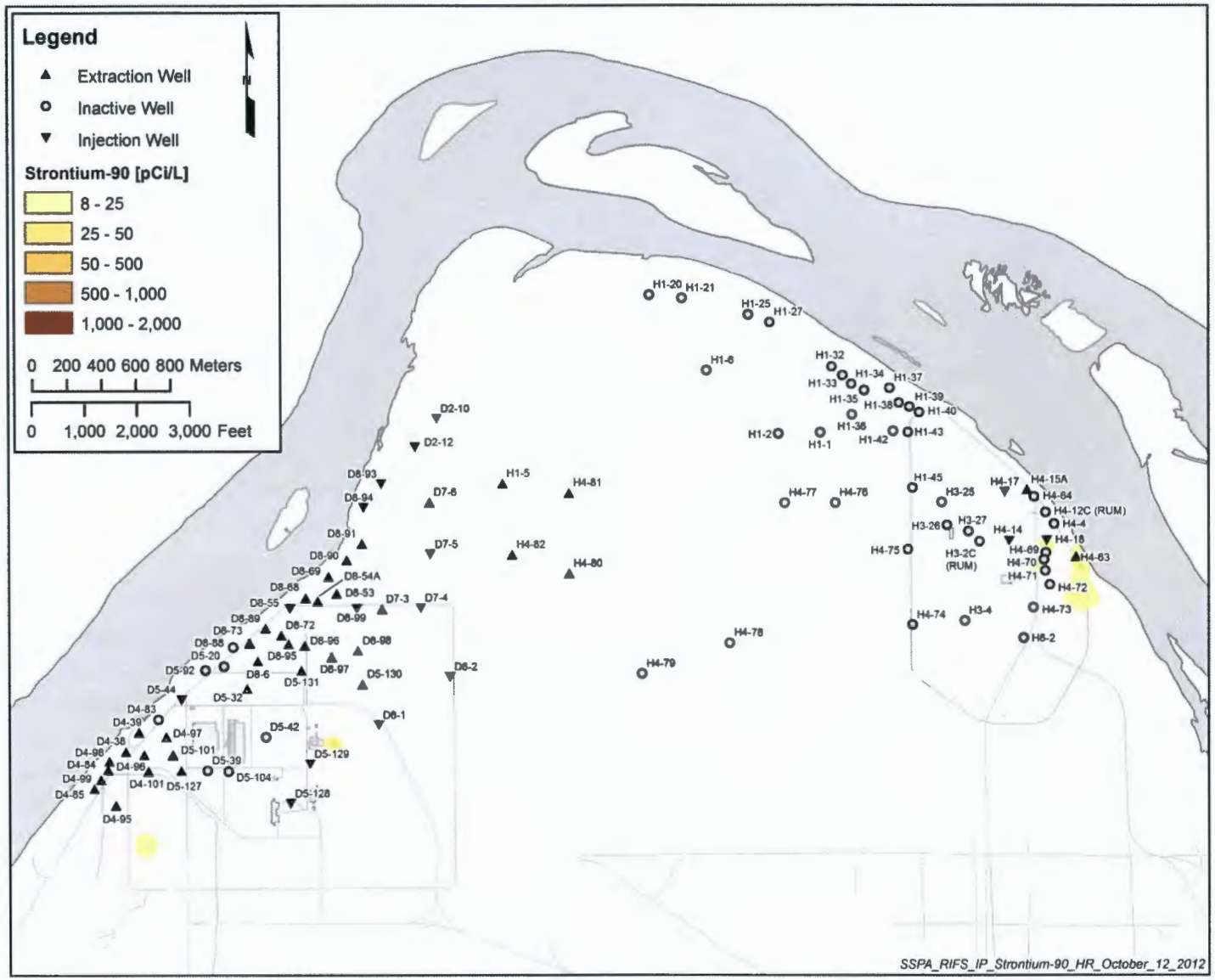


Figure 5-9. 100-DX/HX Pump-and-Treat Well Configuration and Approximate Extent of Strontium-90 in 100-D/H

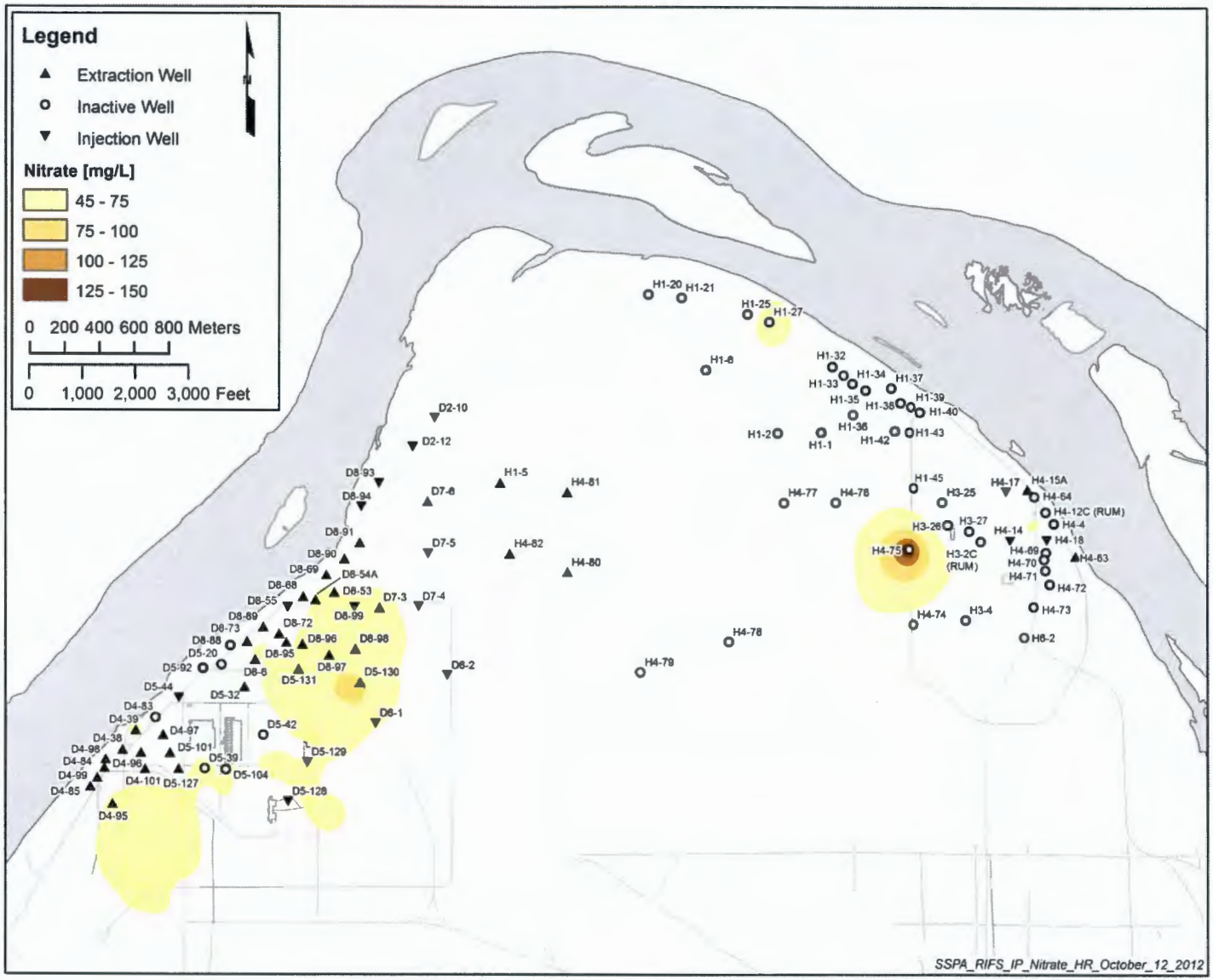


Figure 5-10. 100-DX/HX Pump-and-Treat Well Configuration and Approximate Extent of Nitrate in 100-D/H

Table 5-6. Summary of Selected Primary Fate and Transport Simulation Input Parameters Used with Groundwater Flow and Transport Model Implemented in the MODFLOW and MT3DMS Codes for in the 100-HR-3 Groundwater Operable Unit

Input Parameter (units)	Input Parameter Value	
Simulation Duration		
Simulation of historical conditions (used for model calibration)	5 years – January 2006 through December 2010 Monthly stress periods	
Simulation of future conditions	77 years Monthly stress periods for first 27 years followed by a single stress period of 50 years	
Upper Boundary Condition: Recharge		
Recharge Boundary	Recharge values reported in PNNL-14753, *Groundwater Data Package for Hanford Assessments*, were uniformly scaled during the model calibration process to provide improved fit to measured groundwater elevations. Resulted in a "typical value" for groundwater recharge of 12 mm/yr[a,b] throughout the model domain.	
Lateral Boundary Conditions		
Constant Head Boundaries	Used to represent time-variant hydraulic head distribution in model cells representing a) the Western Gap and b) the Gable Gap.	
General Head Boundaries	Used to represent flow into and out of the model domain along a) the southeast model boundary between Gable Mountain and the Columbia River, and b) the western boundary of the model. Stress-period specific, spatially variable values specified on the basis of a map of site-wide groundwater elevations representing typical groundwater level conditions in 2006–2010.	
River Boundary	River stage data from six gauges located in the vicinity of each Operable Unit plus USGS Gauge 12472800 (located below Priest Rapids Dam) were processed and summarized to monthly average stage values for application in each stress period.	
Lower Boundary Condition		
No Flow Boundary	The lower boundary of the model is a no-flow boundary, in keeping with the stratigraphy selected to choose relatively impermeable units (aquitard, basalt, or mud) to serve as the lower boundary.	
Sources and Sinks		
Pumping Stresses	Extraction and injection rates for 100 Area pump-and-treat systems for January 2006 through December 2012 for following systems included: DR-5, HR-3, DX, HX, KX, KR4, and KW. No further pumping after December 2012 ("no further action").	
Hydraulic Parameters		
Specific yield (unitless)	0.10	
Specific storage (1/day)	0.000005	
	Hanford formation	Ringold Formation

Table 5-6. Summary of Selected Primary Fate and Transport Simulation Input Parameters Used with Groundwater Flow and Transport Model Implemented in the MODFLOW and MT3DMS Codes for in the 100-HR-3 Groundwater Operable Unit

Input Parameter (units)	Input Parameter Value	
K_h saturated horizontal hydraulic conductivity (m/d)	Spatially variable; mean zonal value in: 63[a]	Spatially variable; mean zonal value in: 19[a]
Vertical anisotropy ratio (K_v/K_h)	0.1	
Transport Parameters		
Total porosity (unitless)	0.225	
Mobile porosity (unitless)	0.18	
Immobile porosity (unitless)	0.045	
First-order dual-domain mass transfer rate (1/day)	0.01	
ρ_b bulk density (g/mL)	1.72	
K_d distribution coefficient (mL/g)	Contaminant-dependent and modeled using dual-domain approach; see SGW-46279 and ECF-100HR4-11-0114 for details.	

a. Denotes calibrated value.

b. Recharge rate represents an average over a large area represented by the groundwater model with spatially distributed surface soil types and vegetative cover under present day conditions. This value is not directly comparable to recharge rates applied for vadose zone modeling of waste sites that represent a subset of the surface soil type and vegetation cover range applicable to the larger area groundwater model. The values used for the vadose zone models used to develop SSL and PRG values are within the range of recharge rates represented in the groundwater model for present day conditions.

Note: Details on the basis for all parameters in this table are found in *Conceptual Framework and Numerical Implementation of 100 Areas Groundwater Flow and Transport Model* [SGW-46279].

5.8.3.1 Hydraulic Containment in 2012

Based on the groundwater modeling results, a systematic approach was developed and applied to the estimated hydraulic containment in 2012. Although a single depiction of capture can be calculated using particle tracking when a model simulates quasi-steady-state conditions, an estimate of the approximate extent of hydraulic capture was calculated with the transient model. The approach was similar to that described in "The Capture Efficiency Map: The Capture Zone Under Time-Varying Flow" (Festger and Walter, 2002) and "Sources of Water to Wells for Transient Cyclic Systems" (Reilly and Pollock, 1996), focusing on the evaluation of the temporal variation in capture because of changing flow patterns and hydraulic gradients:

- Releasing particles near the end of each of the 12 monthly stress periods and simulating their migration using a very low effective porosity, ensuring that particle travel times are essentially instantaneous
- Recording the instantaneous fate of each particle during each stress period
- Calculating a capture zone for each stress period based on the "snapshot" of aquifer conditions at the time of the particle release; in this case, producing 12 instantaneous snapshots of the extent of capture

- Constructing a capture efficiency map by counting the number of times a particle originating from a location was captured by a well, and dividing this count by the total number of releases (that is, 12)

Figure 5-11 shows the current capture efficiency of the 100-DX/HX pump-and-treat system for the unconfined aquifer underlying 100-D/H. The calculated capture efficiency suggests there are areas where the capture is very efficient and areas where the capture is less efficient, although always above 50 percent. Areas of efficient capture encompass almost the entire Cr(VI) plume footprint providing river protection along much of the shoreline, particularly close to the pump-and-treat wells. Areas where capture is less efficient include only parts of the plume footprint near the shoreline, where pump-and-treat wells are absent or relatively sparsely placed. The ability to place wells close to the shoreline is constrained by cultural resource issues, ecological resource issues, and topographical limitations because of the steep riverbank in many locations.

This evaluation considered a "no further action" condition as the baseline. The continued operation of the current pump-and-treat system, as well as selected optimization schemes for extraction and injection well placement and operation, are evaluated in the FS presented in Chapter 8 and 9.

5.8.3.2 Contaminant Plume Migration

To present a simulated baseline of groundwater contaminant migration at 100-HR-3 OU, the results of simulations described as Alternative 1, which does not include continued active pump-and-treat groundwater remediation in future years (no further action after December 2012), are shown in the following figures:

- Figures 5-12 to 5-17 show the simulated dissolved Cr(VI) plume distribution in December in the years 2012, 2015, 2020, 2040, 2060, 2070, and 2087, respectively, based on the current well configuration and treatment system. (Note: Figure 5-8 showed the initial condition for dissolved Cr(VI) plume in December 2010.)

- Figures 5-18 to 5-27 show the simulated plume distributions for strontium-90 in December in the years 2012, 2015, 2020, 2030, 2040, 2050, 2060, 2070, and 2087, respectively. (Note: Figure 5-9 showed the initial condition for dissolved strontium-90 plume in December 2010.)

- Figures 5-28 to 5-35 show the simulated plume distributions for nitrate in December in the years 2012, 2015, 2020, 2030, 2040, 2060, 2070, and 2075, respectively. (Note: Figure 5-10 showed the initial condition for dissolved nitrate plume in December 2010.)

The depicted plumes were simulated assuming the current pump-and-treat system is turned off on December 31, 2012. Each model simulation is run either until the modeling period expires or the COC is below the respective AWQC or MCL.

Plume migration patterns under Alternative 1 conditions (with no active pump-and-treat operations after December 2012) indicate that all of the mobile contaminants migrate toward, and ultimately discharge into the Columbia River. The predicted concentrations of the contaminants decrease according to their natural decay rate (for example, for radionuclides), or because of dispersion as they move through the aquifer. Those contaminants with low attenuation (that is, low K_d) move more rapidly toward the river than those with higher attenuation (that is, higher K_d).

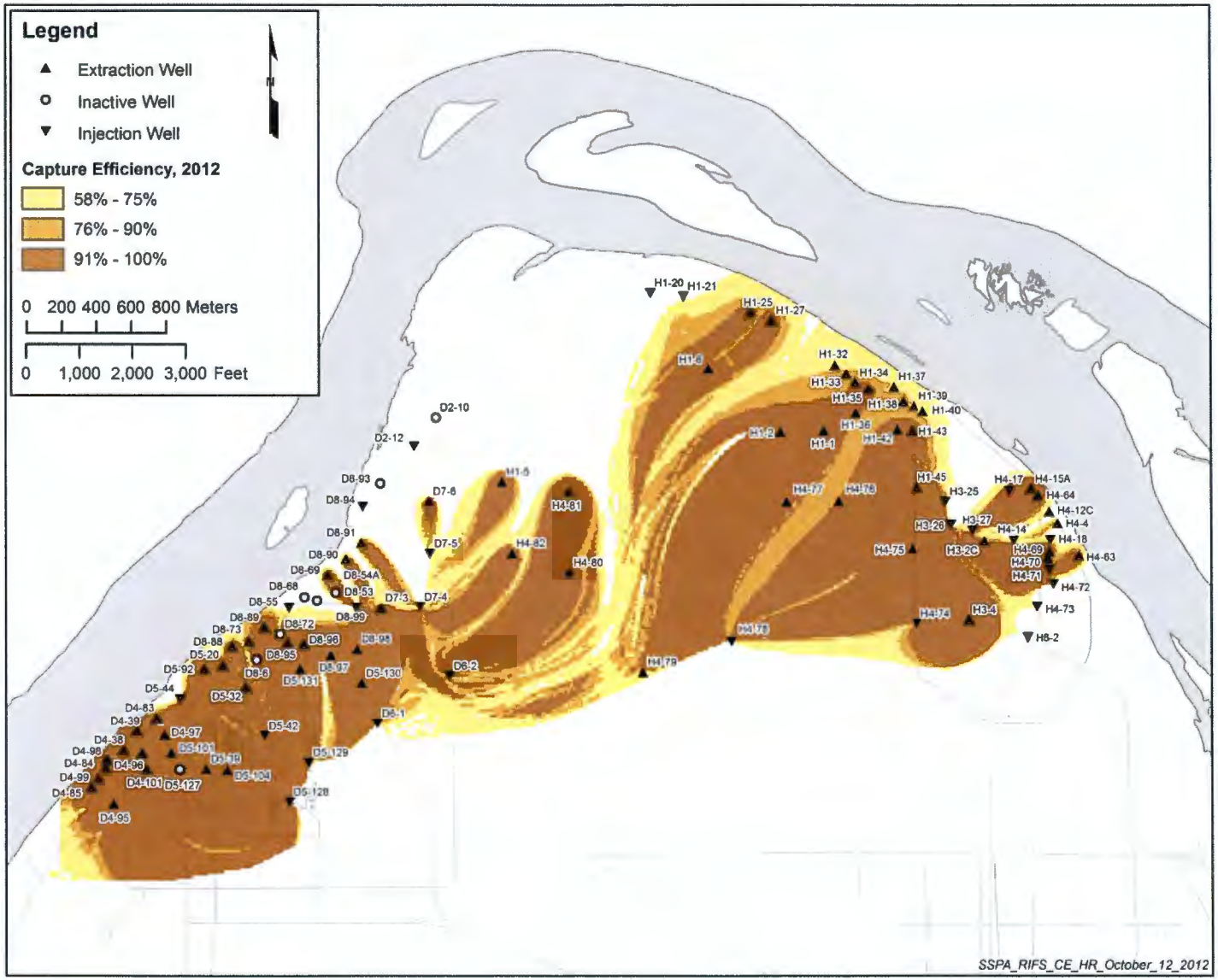


Figure 5-11. Approximate Extent of Capture throughout 100-D/H for January to December 2012

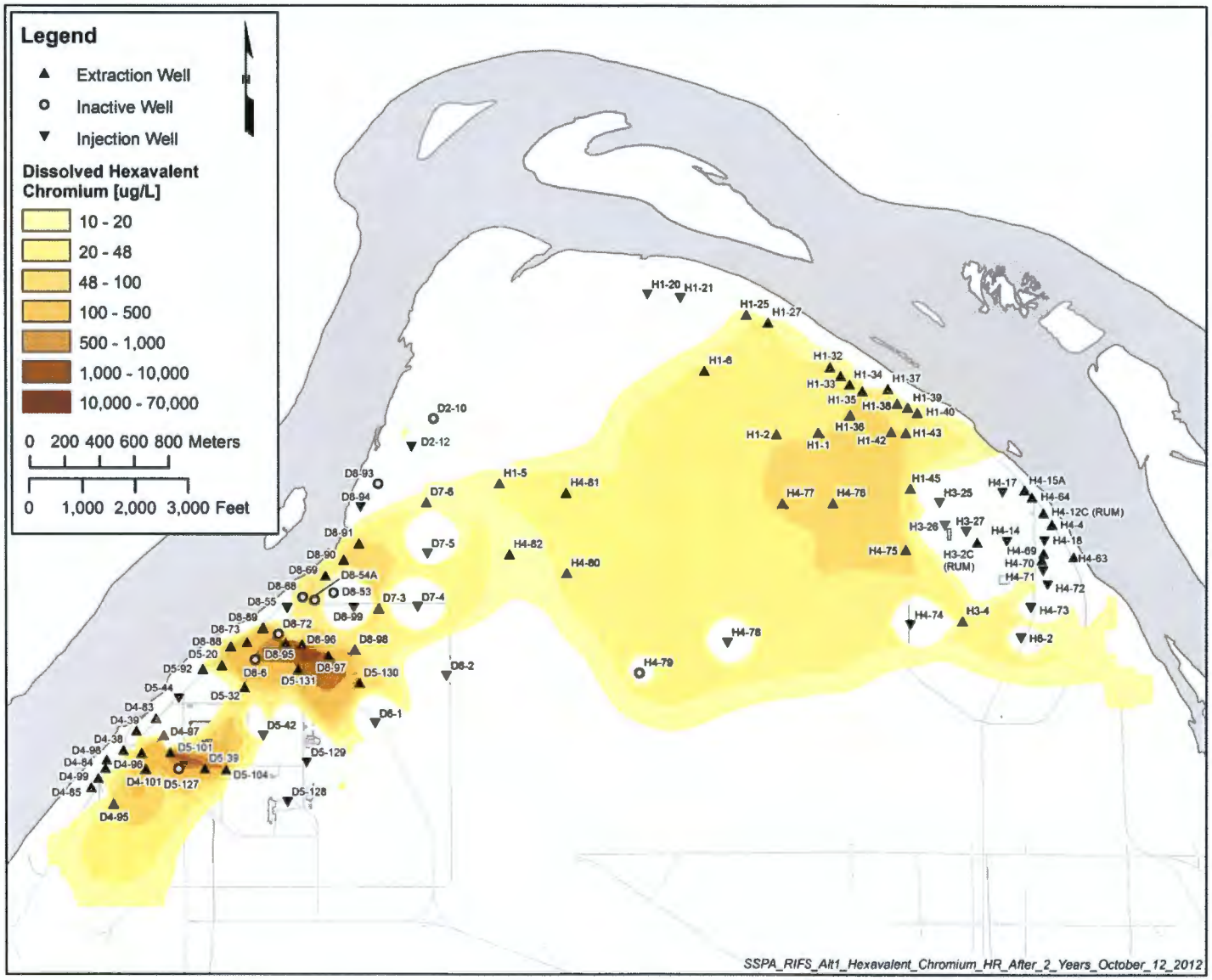


Figure 5-12. Model-Simulated Dissolved Cr(VI) Distribution After 2 Years (Based on No Further Action)

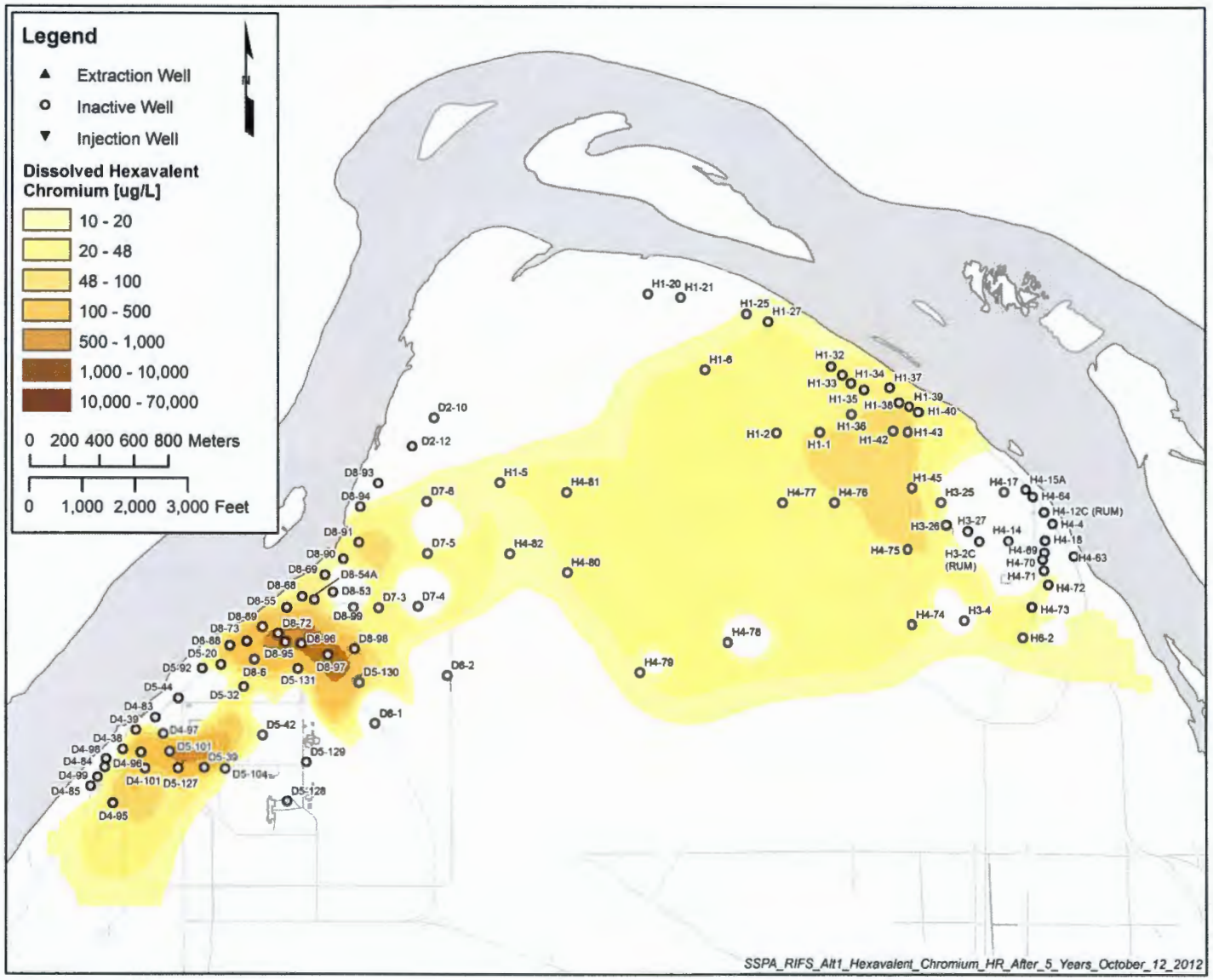


Figure 5-13. Model-Simulated Dissolved Cr(VI) Distribution After 5 Years (Based on No Further Action)

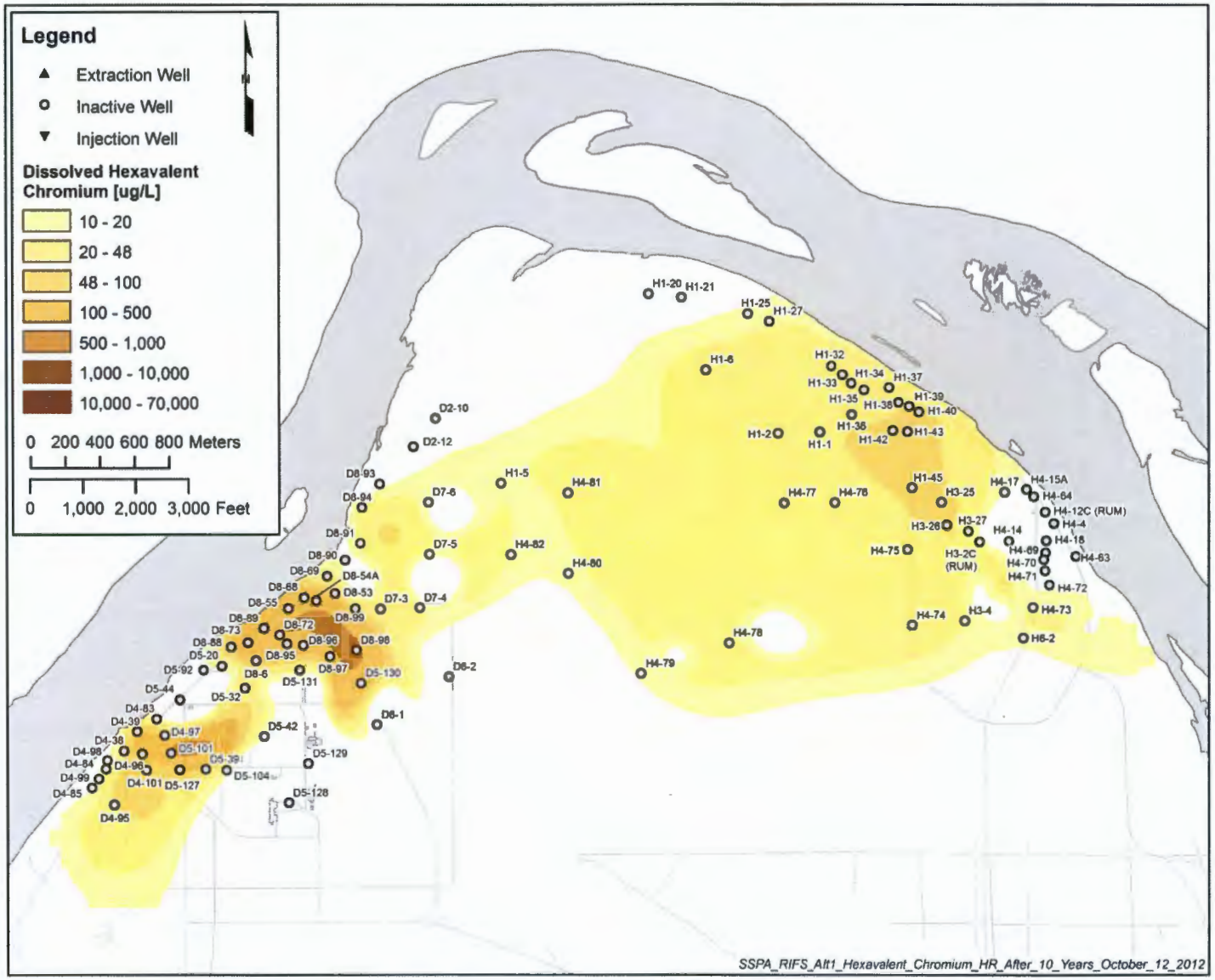


Figure 5-14. Model-Simulated Dissolved Cr(VI) Distribution After 10 Years (Based on No Further Action)

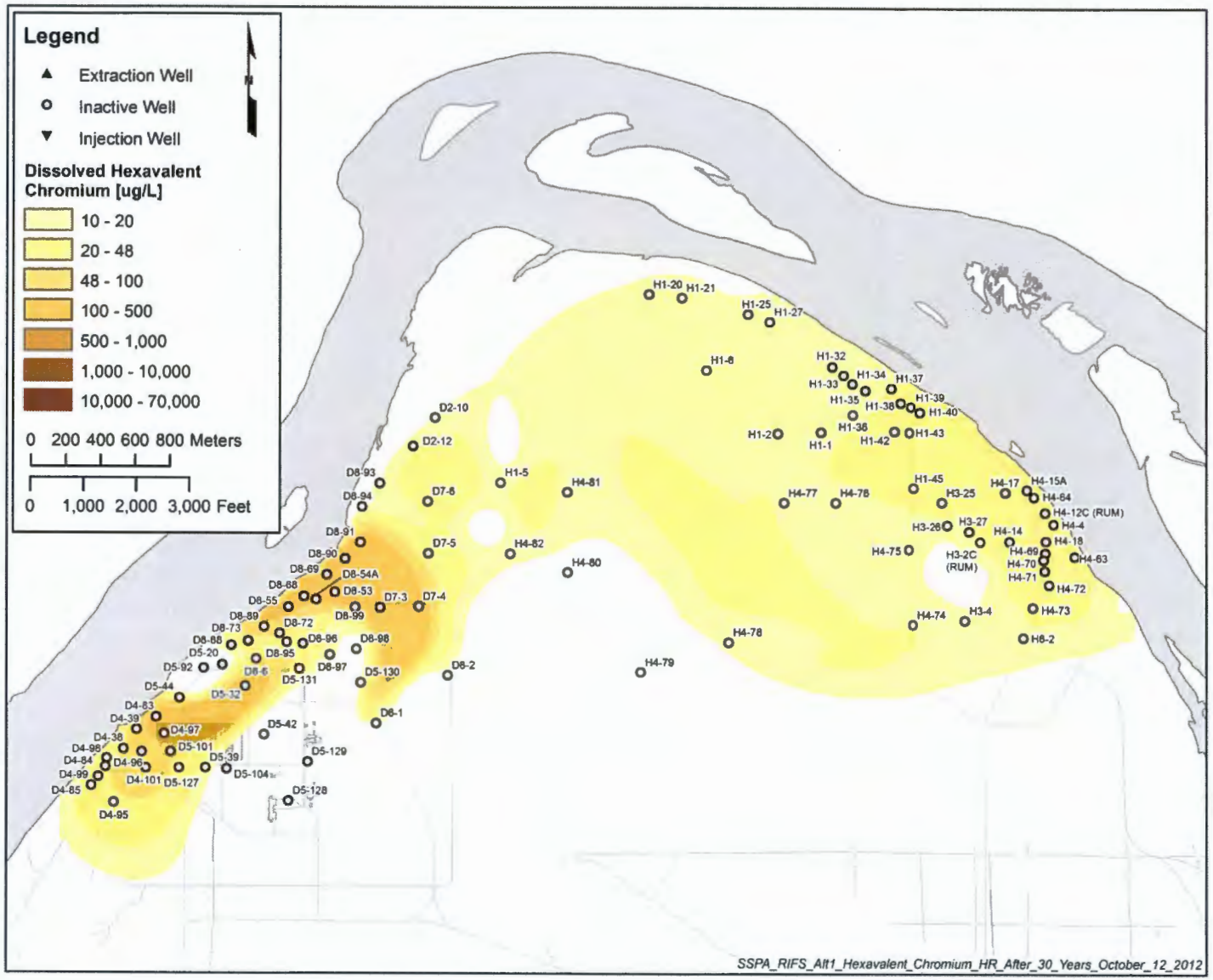


Figure 5-15. Model-Simulated Dissolved Cr(VI) Distribution After 30 Years (Based on No Further Action)

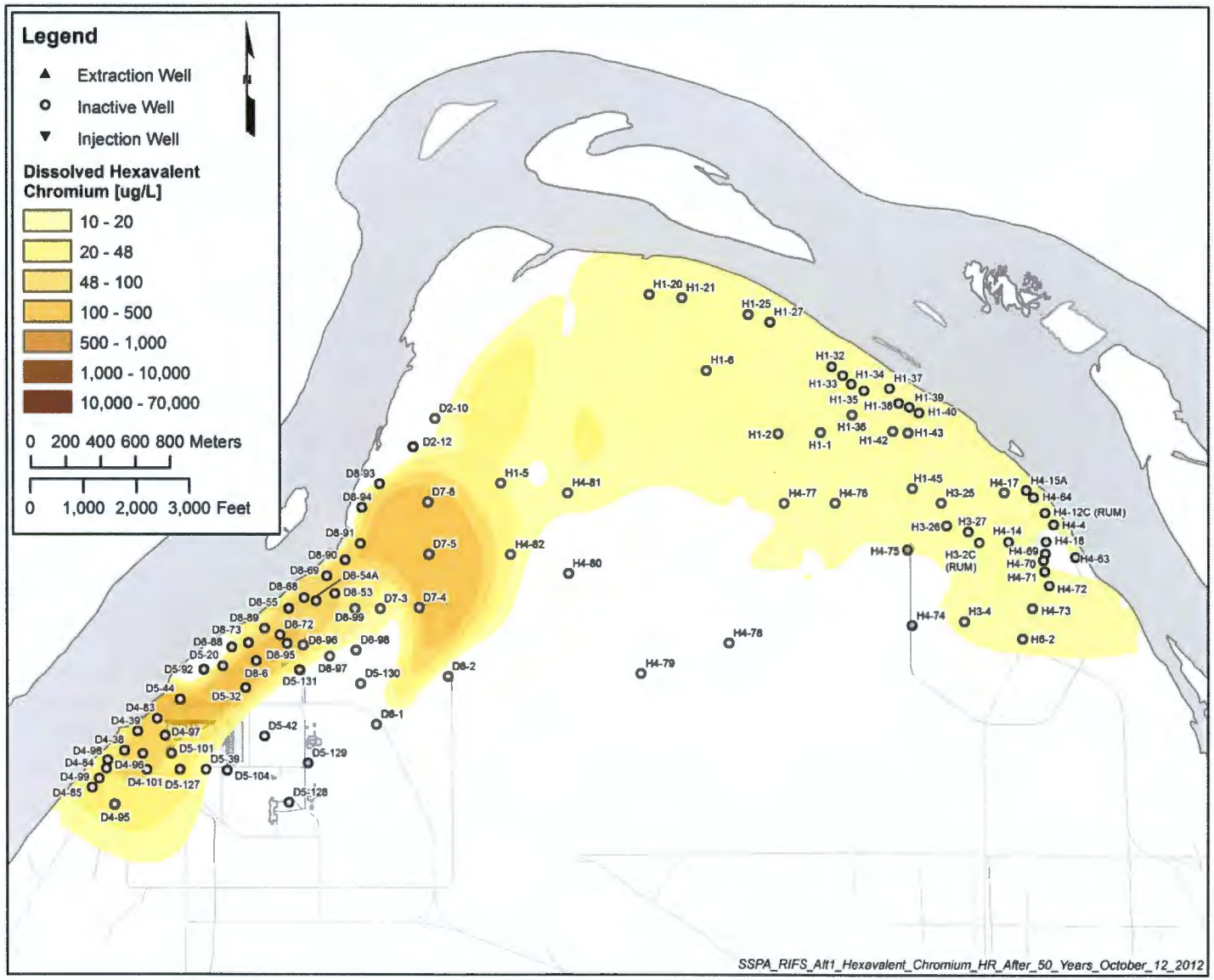


Figure 5-16. Model-Simulated Dissolved Cr(VI) Distribution After 50 Years (Based on No Further Action)

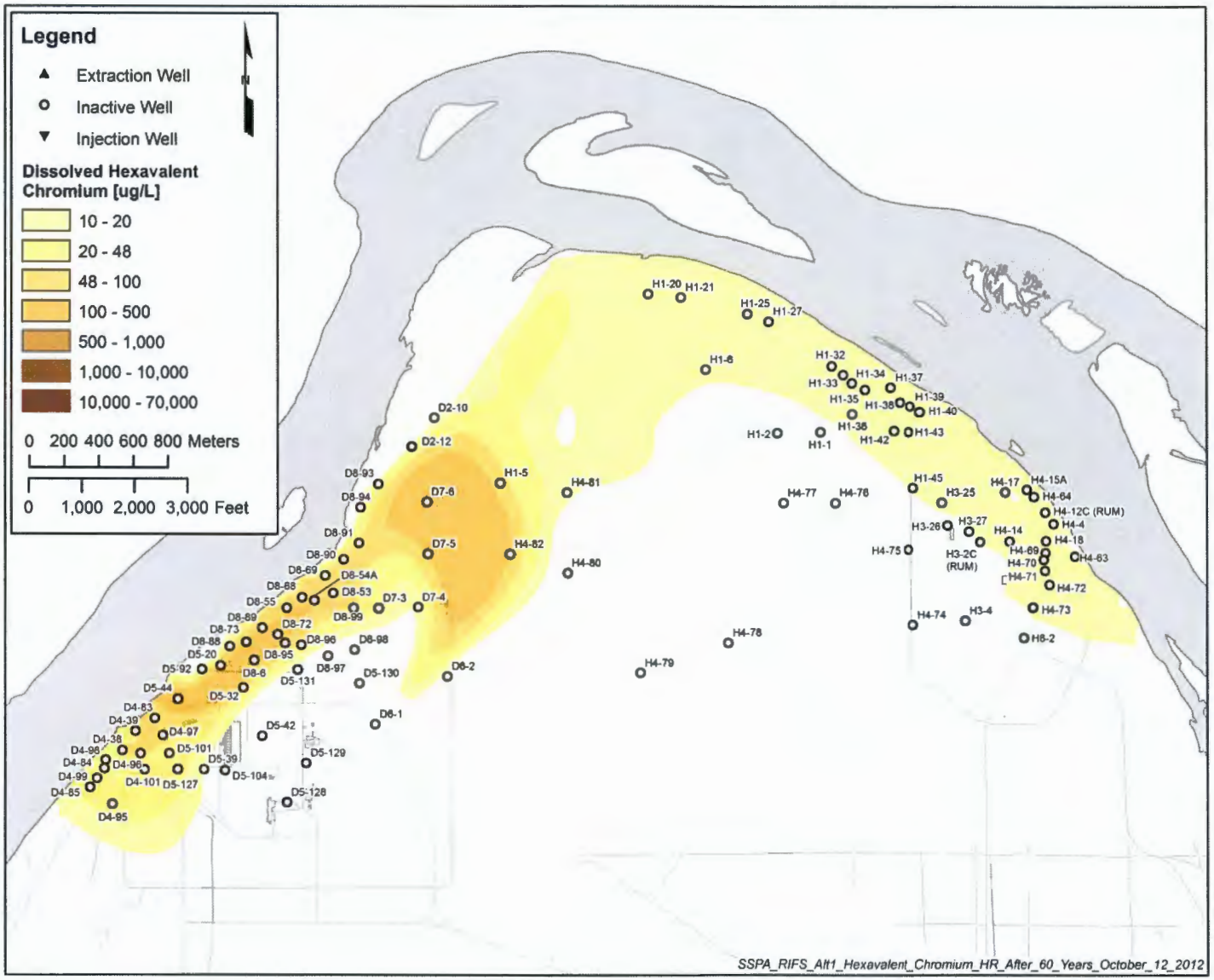


Figure 5-17. Model-Simulated Dissolved Cr(VI) Distribution After 60 Years (Based on No Further Action)

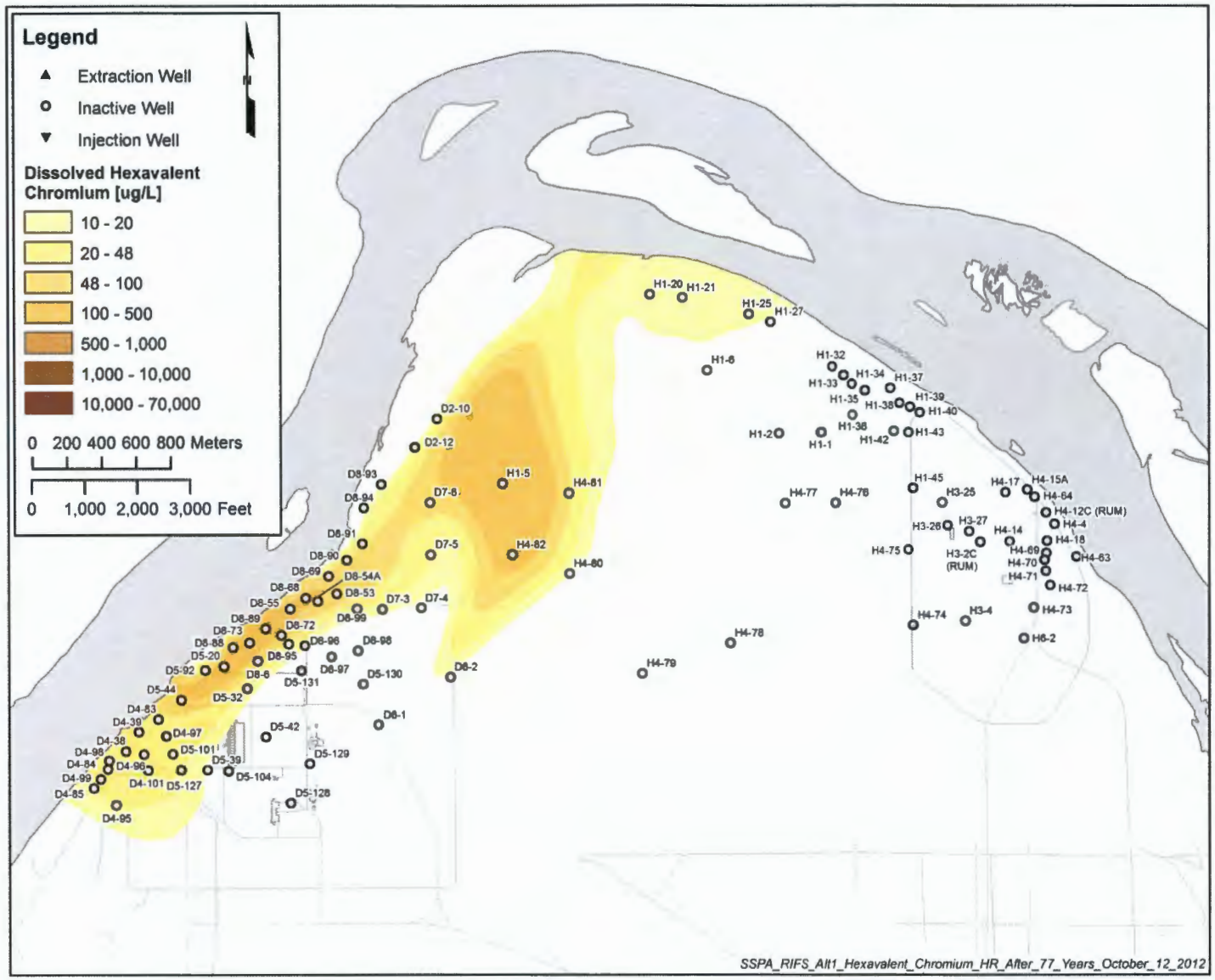


Figure 5-18. Model-Simulated Dissolved Cr(VI) Distribution After 77 Years (Based on No Further Action)

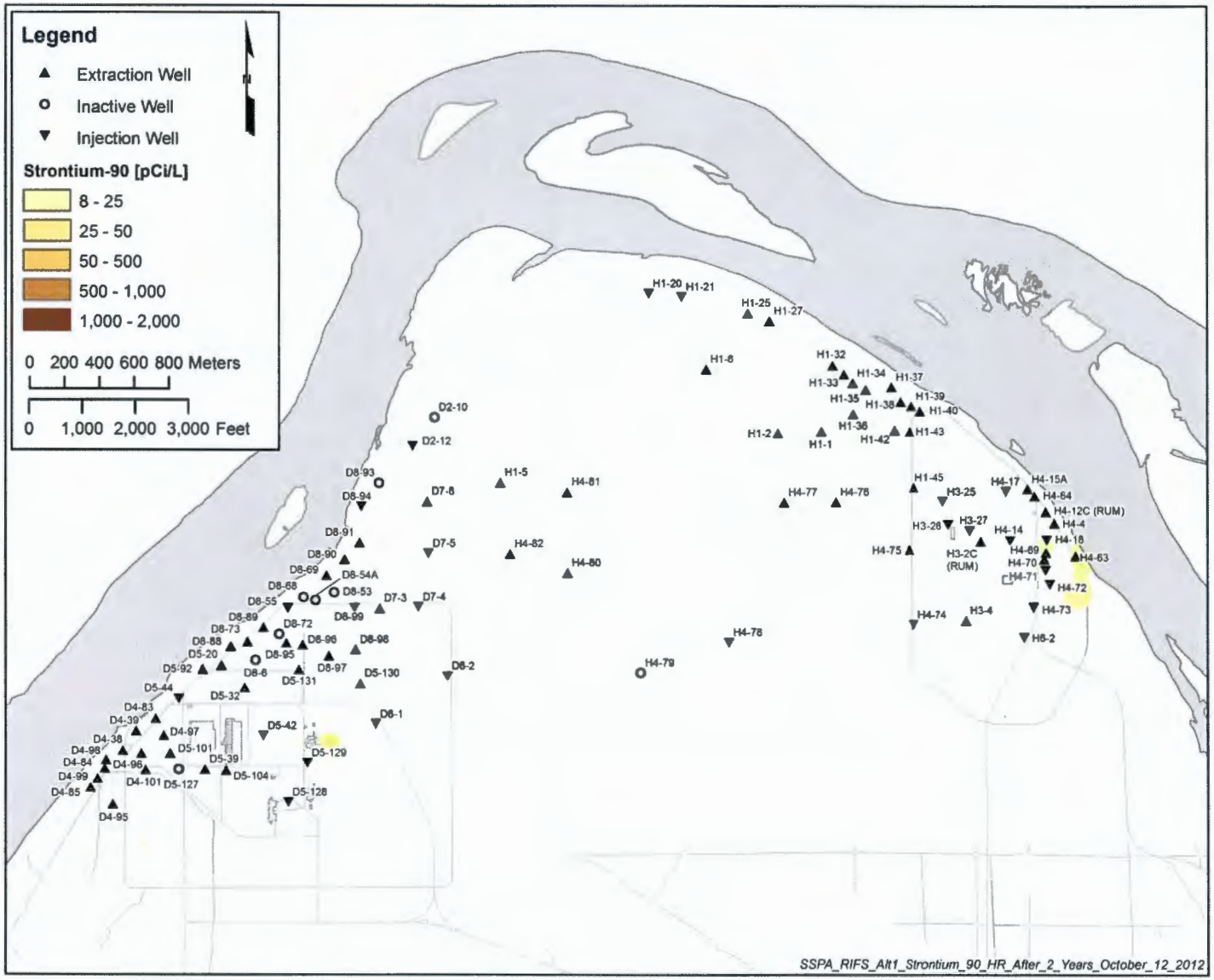


Figure 5-19. Model-Simulated Strontium-90 Distribution After 2 Years (Based on No Further Action)

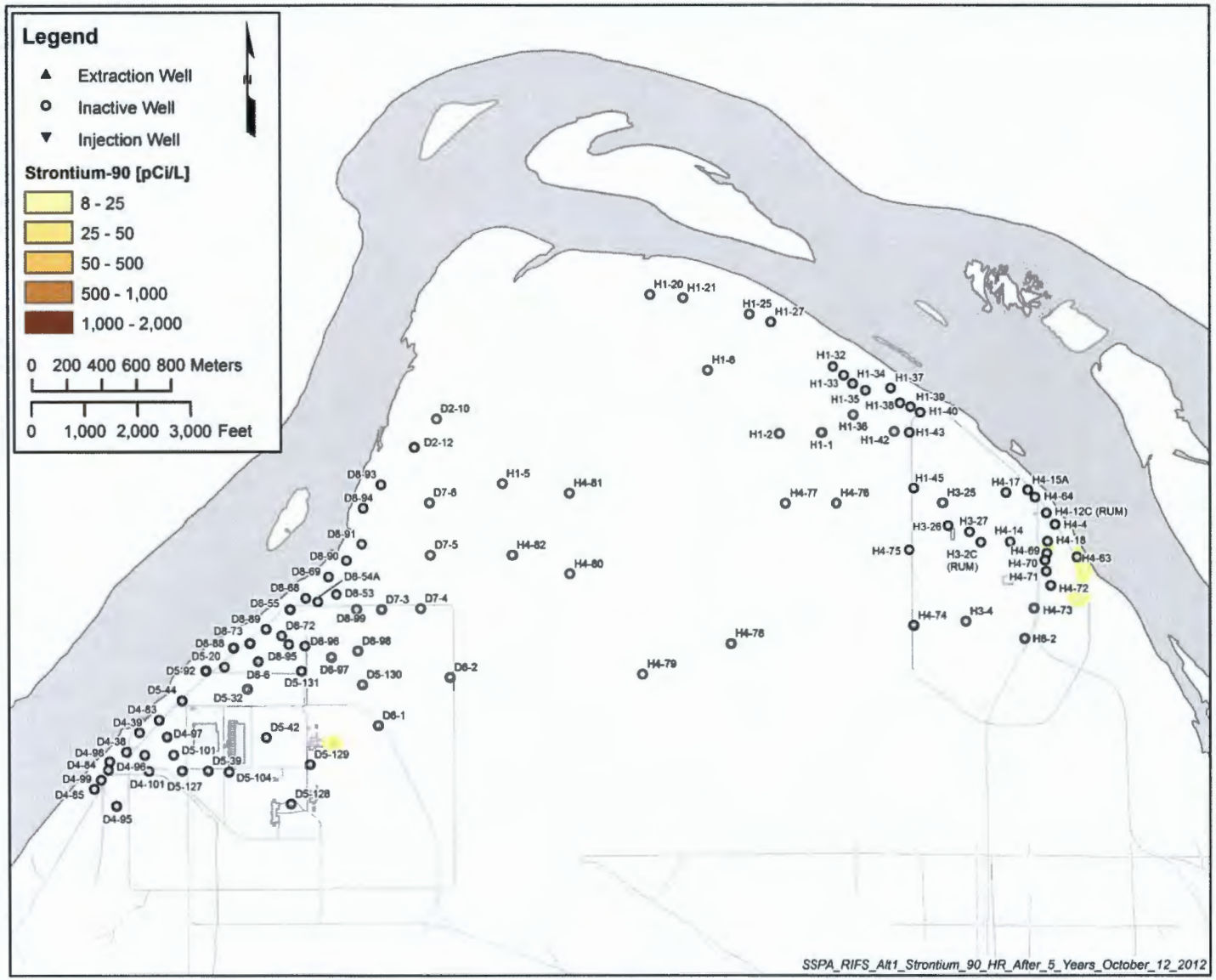


Figure 5-20. Model-Simulated Strontium-90 Distribution After 5 Years (Based on No Further Action)

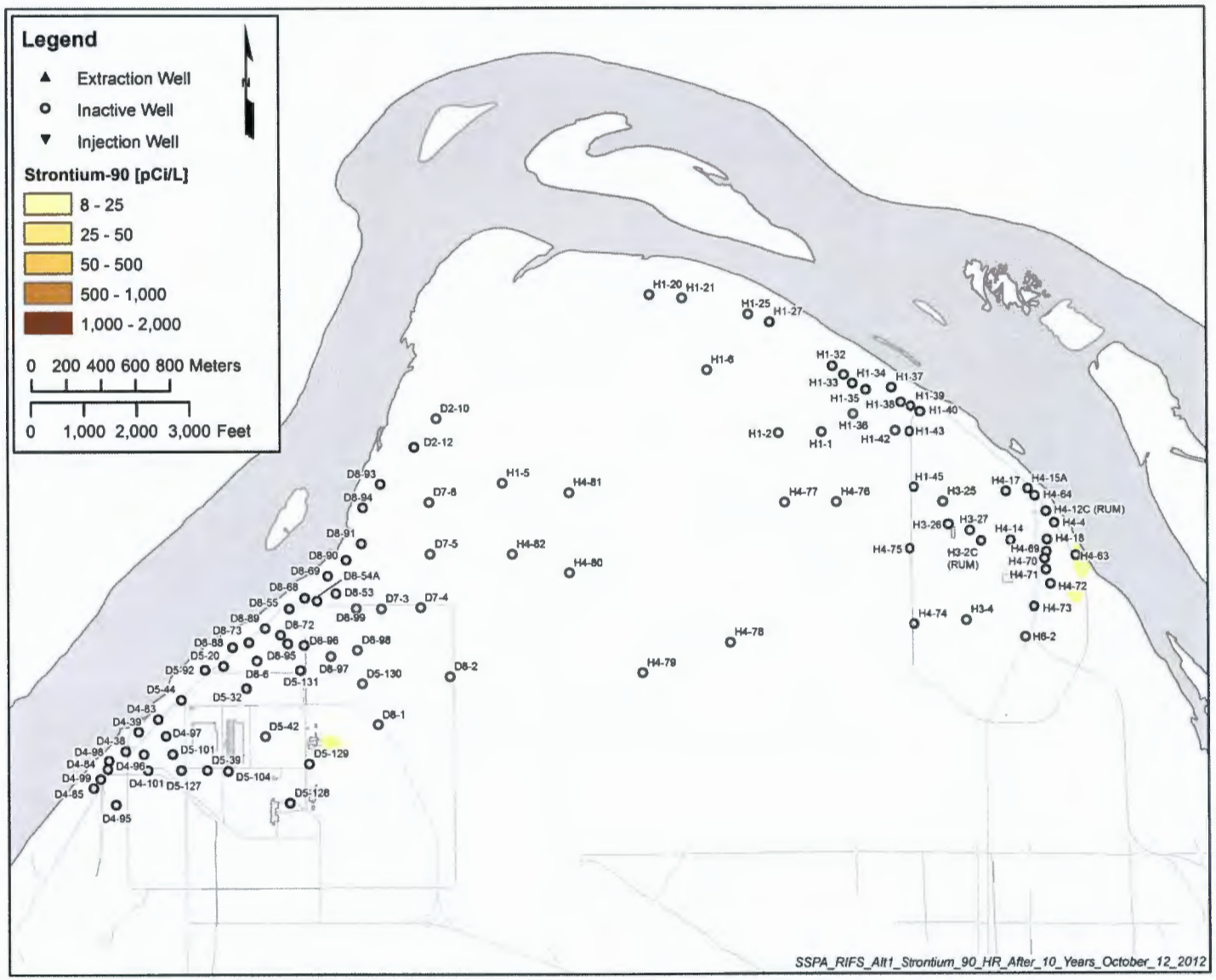


Figure 5-21. Model-Simulated Strontium-90 Distribution After 10 Years (Based on No Further Action)

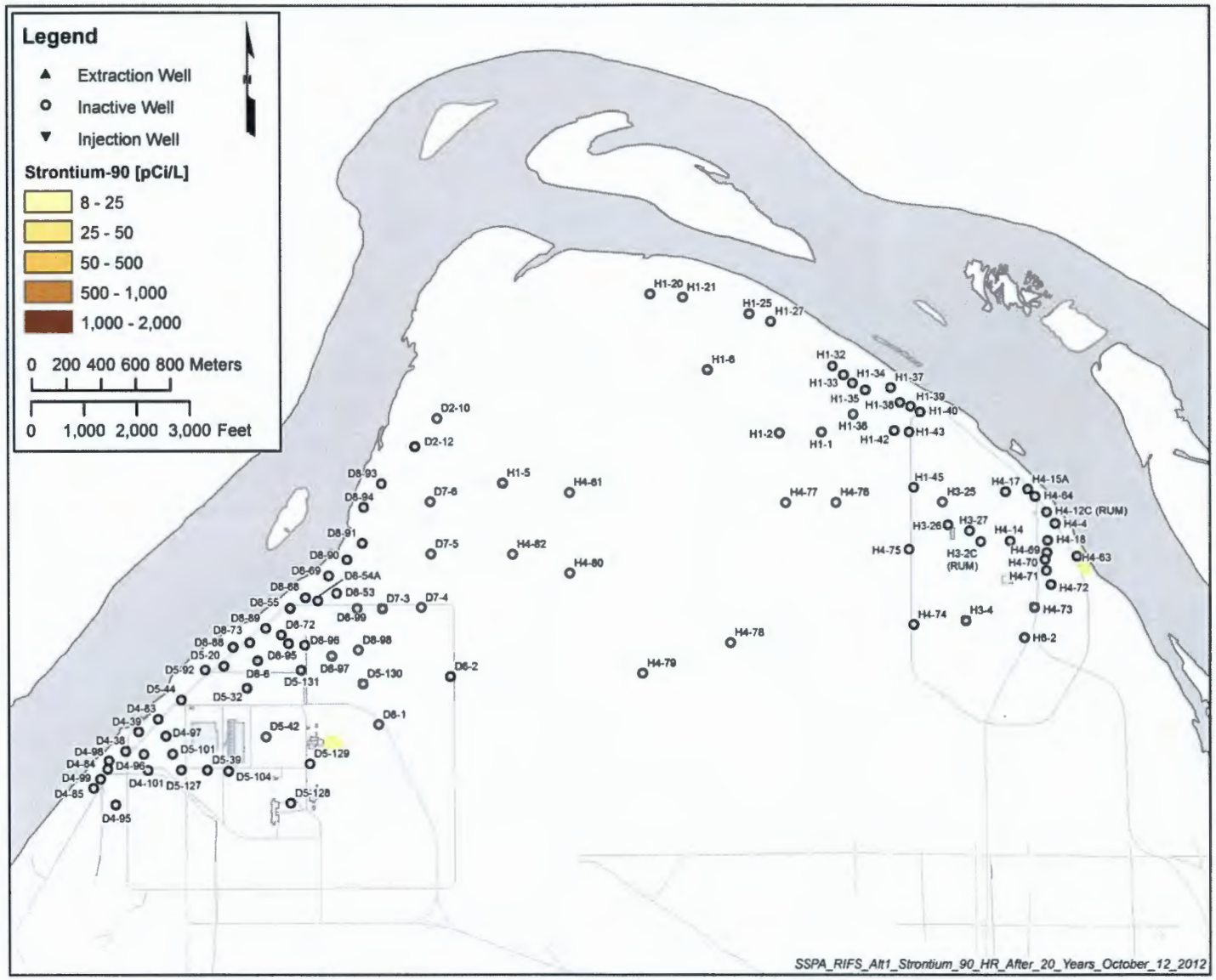


Figure 5-22. Model-Simulated Strontium-90 Distribution After 20 Years (Based on No Further Action)

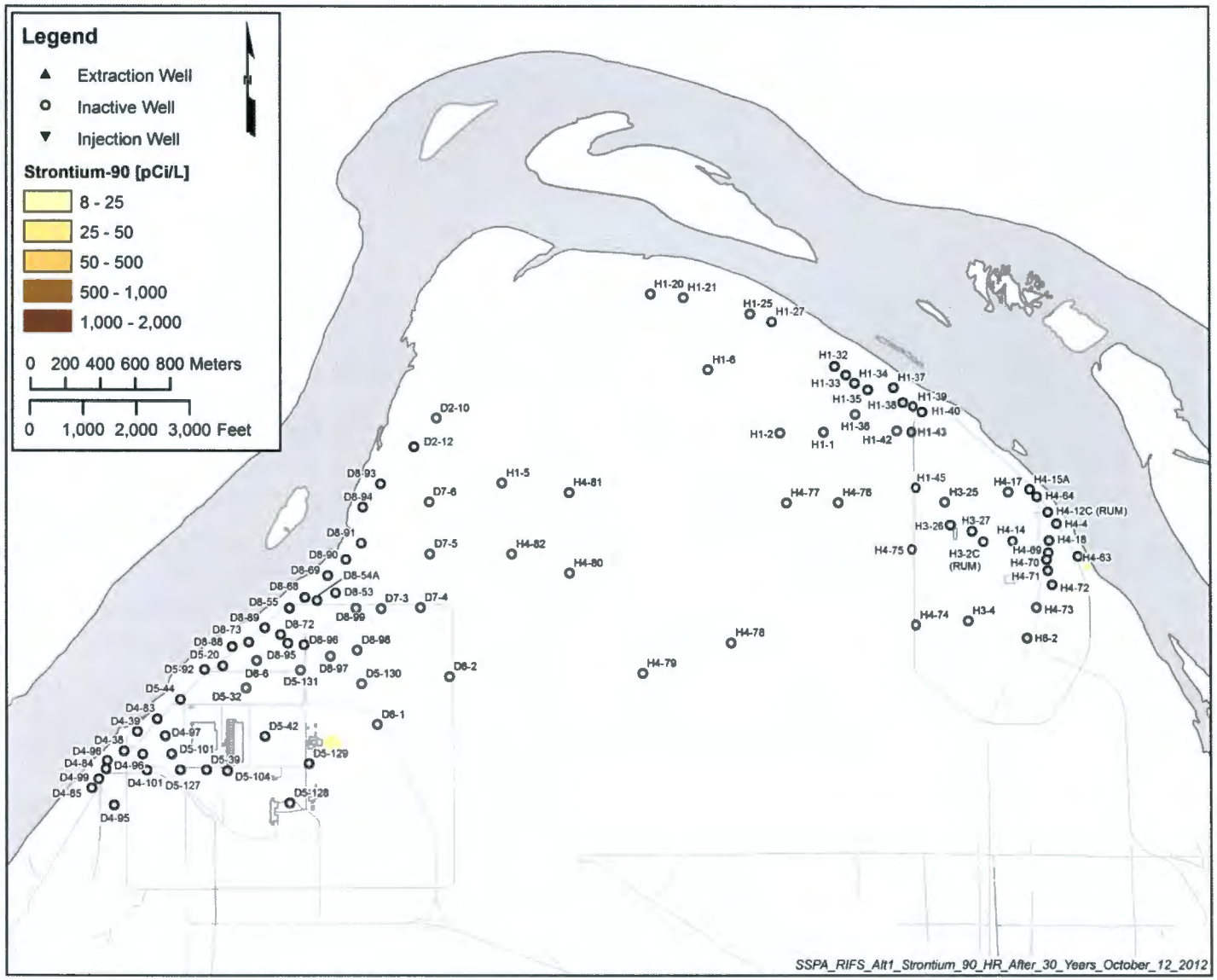


Figure 5-23. Model-Simulated Strontium-90 Distribution After 30 Years (Based on No Further Action)

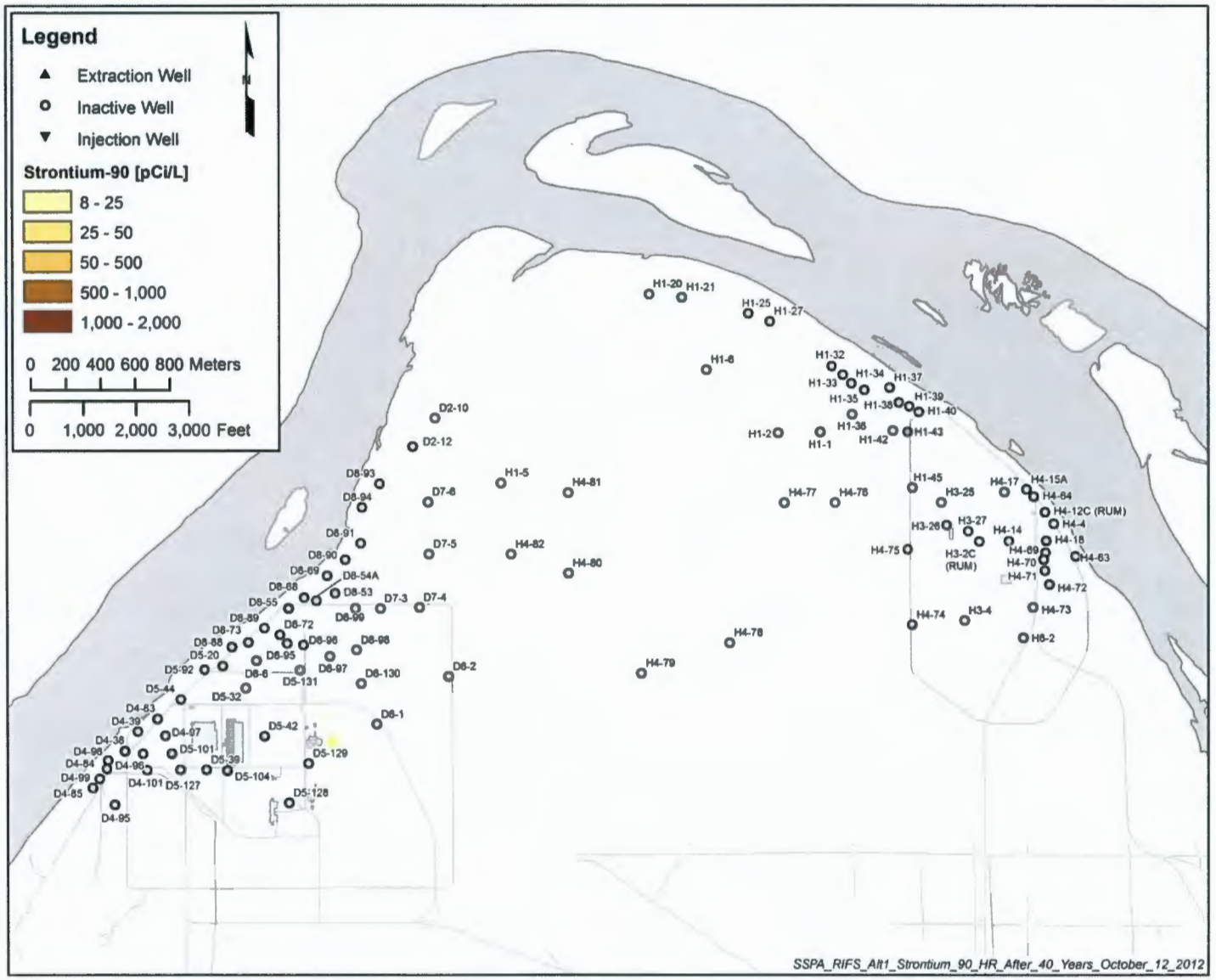


Figure 5-24. Model-Simulated Strontium-90 Distribution After 40 Years (Based on No Further Action)

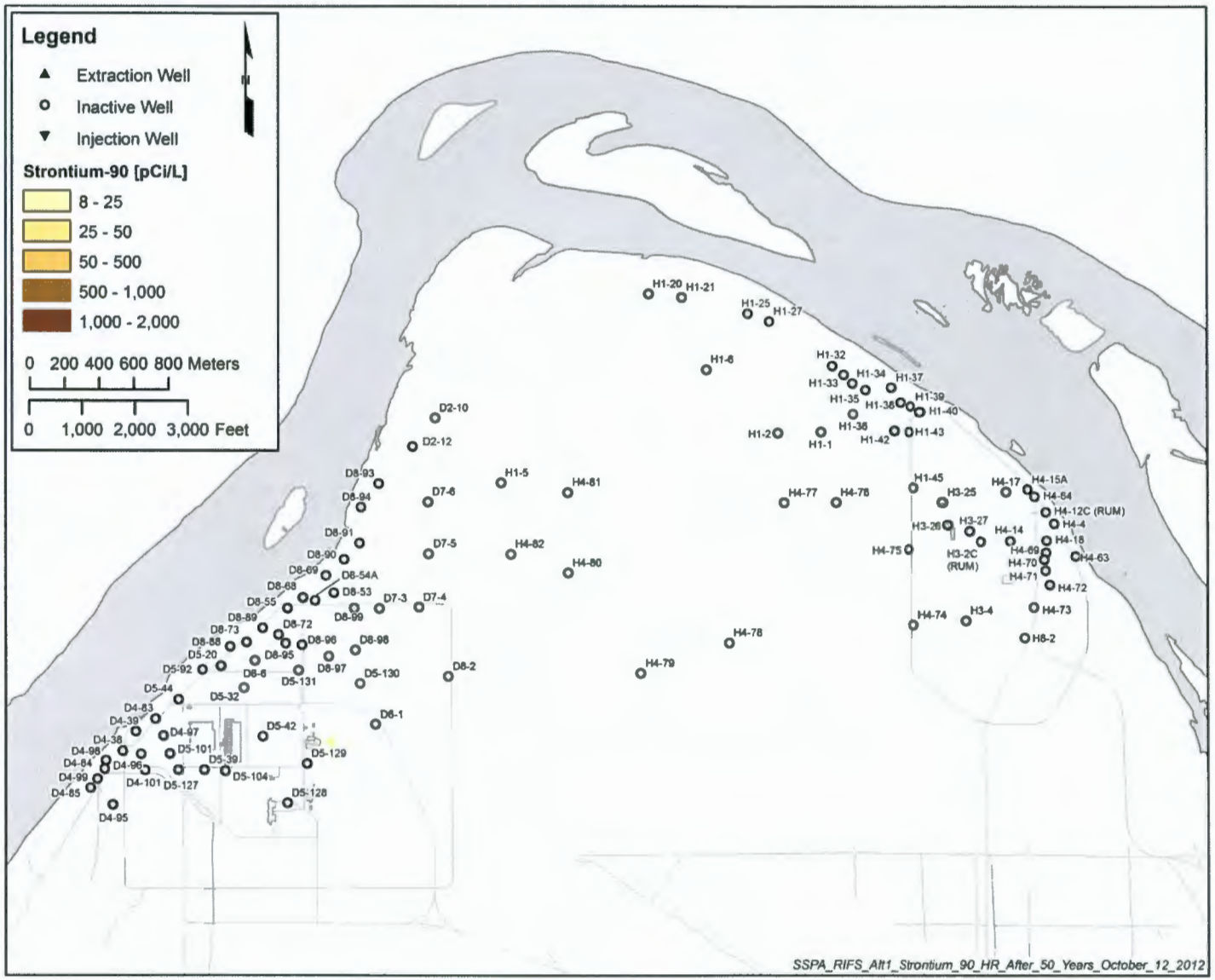


Figure 5-25. Model-Simulated Strontium-90 Distribution After 50 Years (Based on No Further Action)

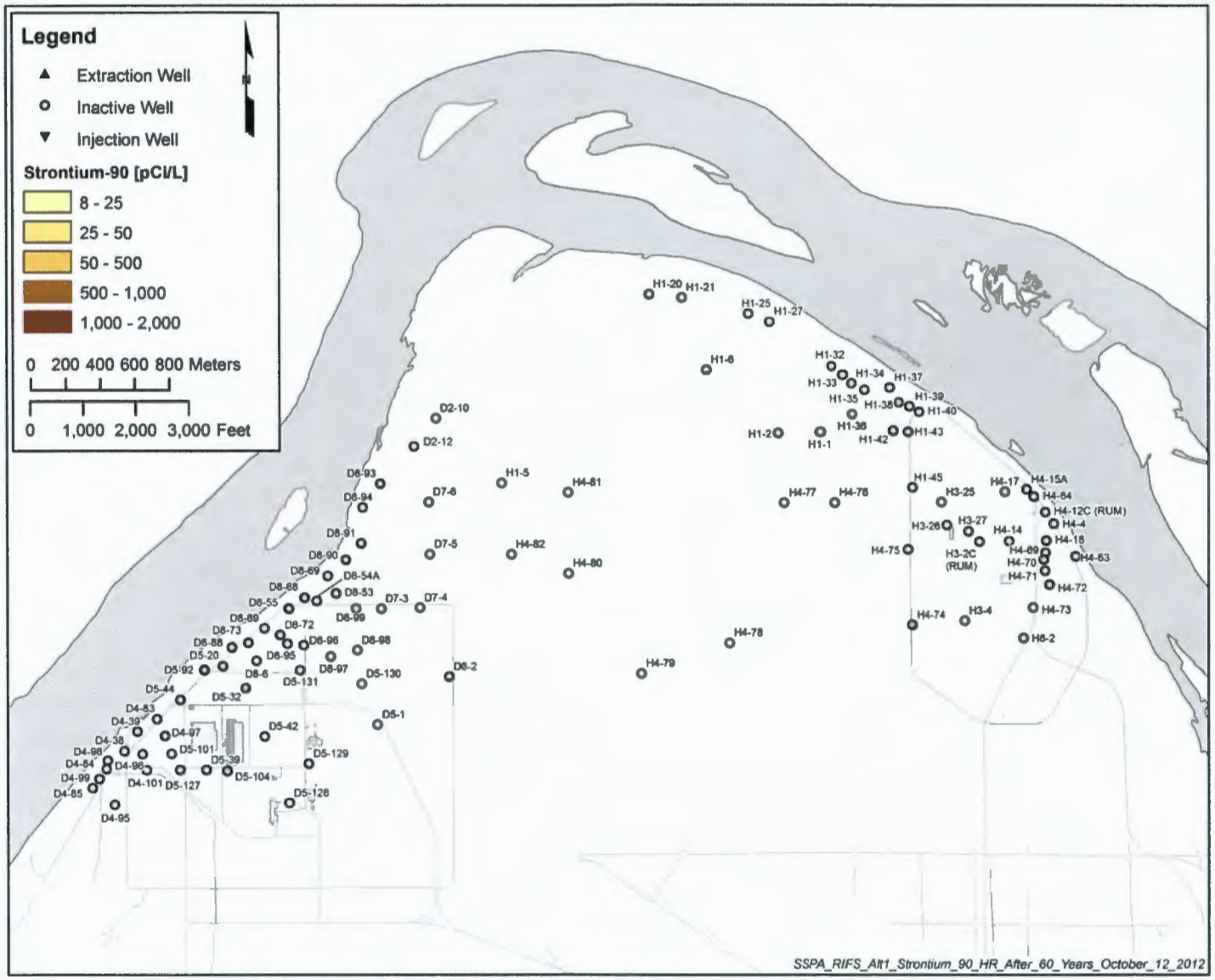


Figure 5-26. Model-Simulated Strontium-90 Distribution After 60 Years (Based on No Further Action)

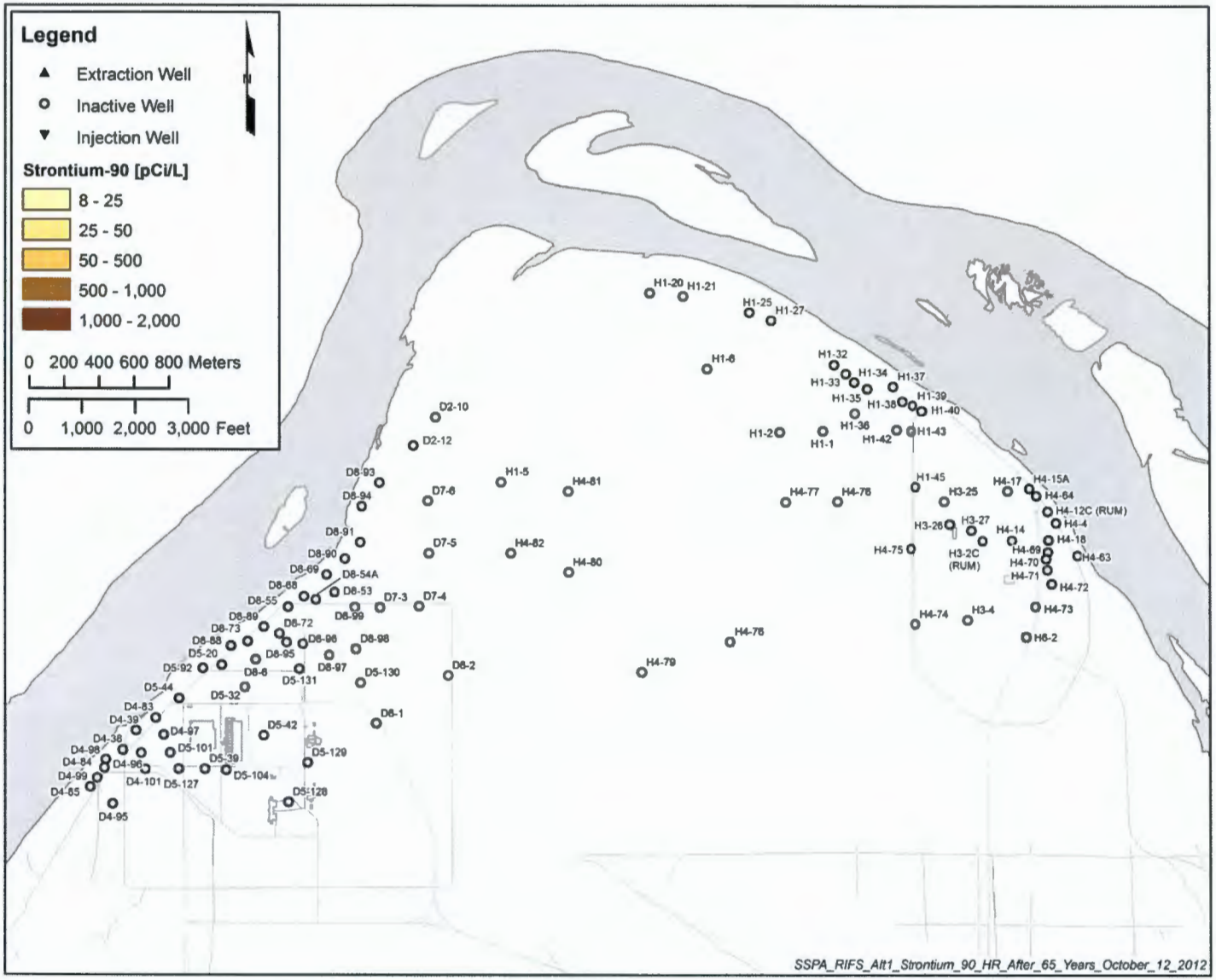


Figure 5-27. Model-Simulated Strontium-90 Distribution After 65 Years (Based on No Further Action)

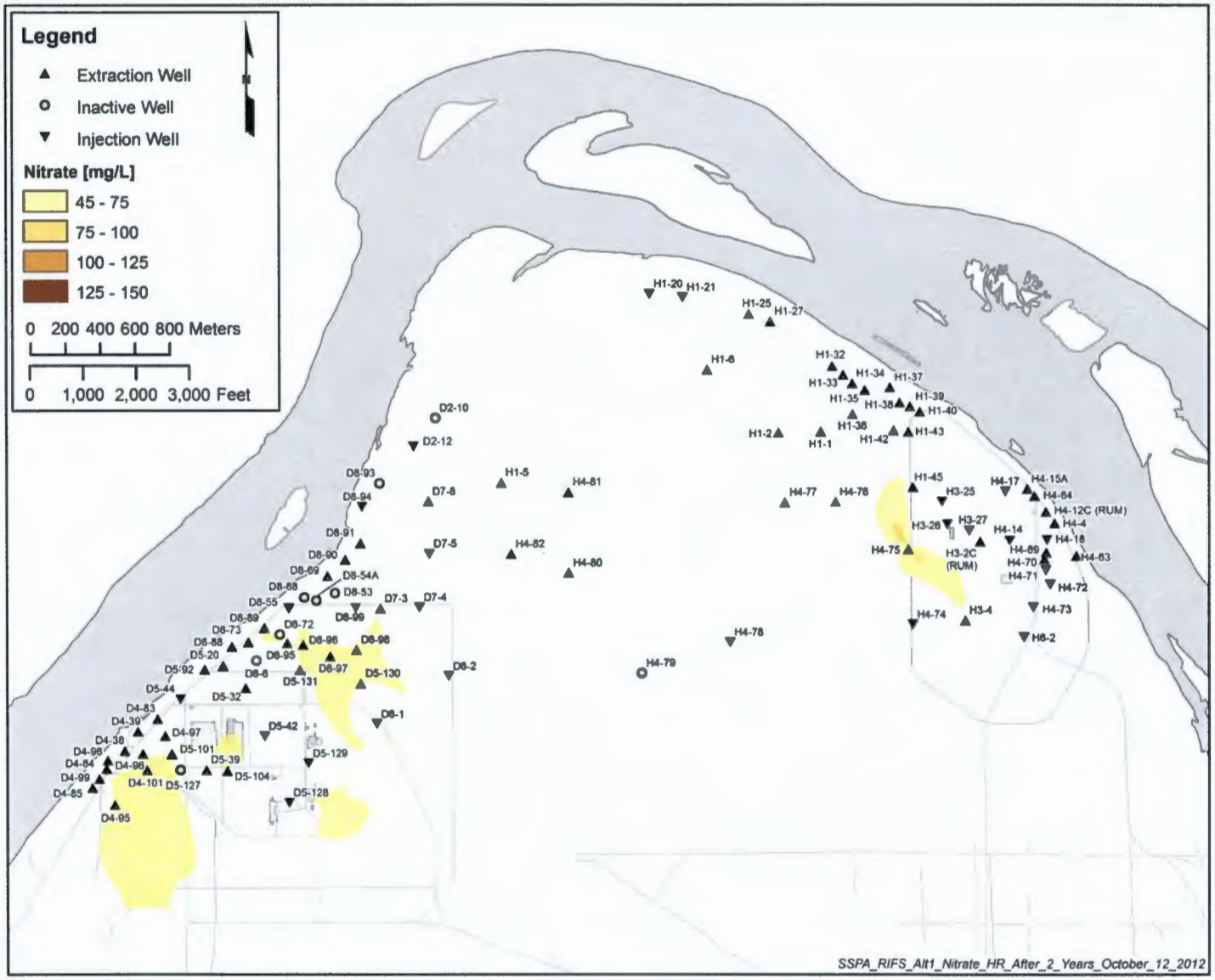


Figure 5-28. Model-Simulated Nitrate Distribution After 2 Years (Based on No Further Action)

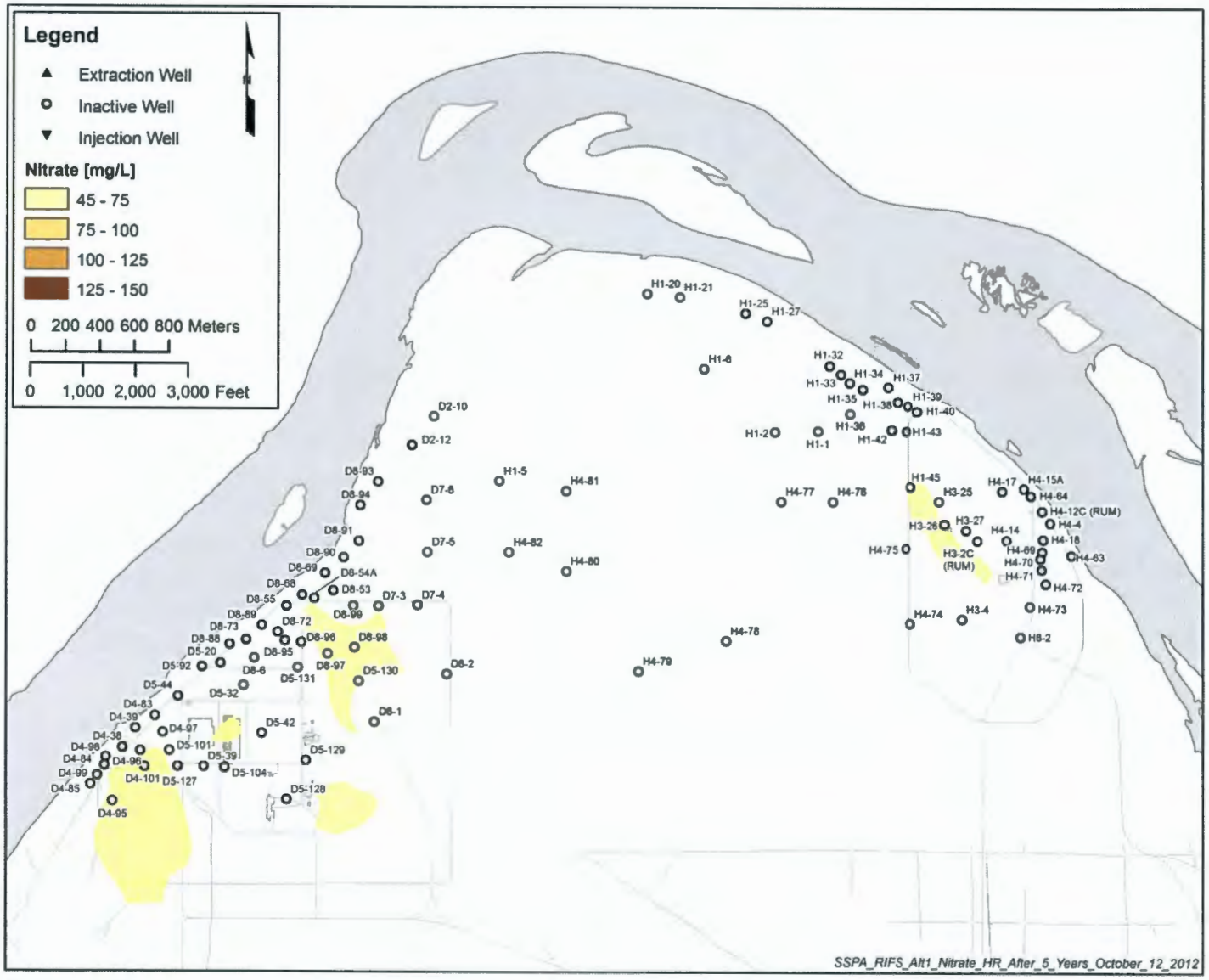


Figure 5-29. Model-Simulated Nitrate Distribution After 5 Years (Based on No Further Action)

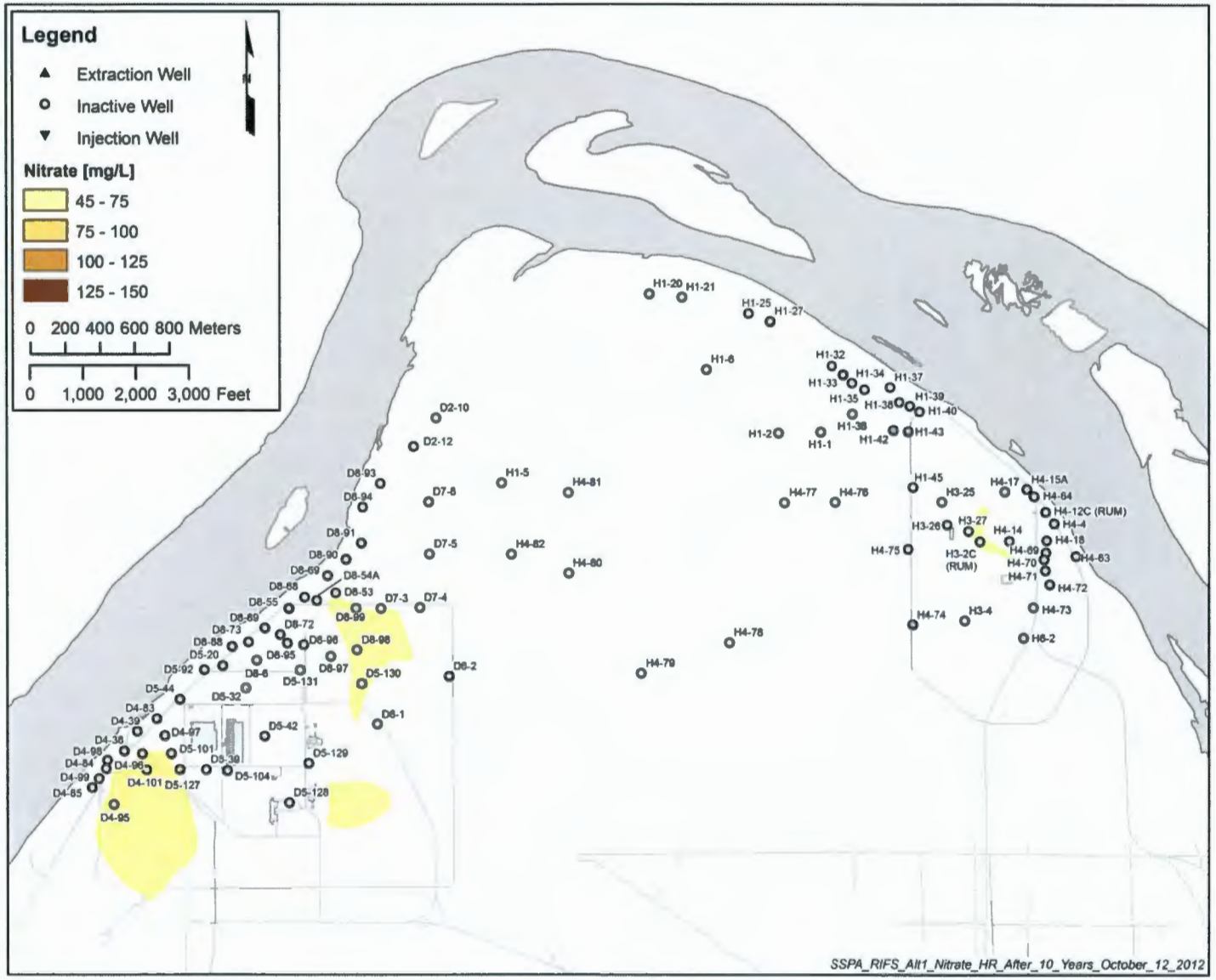


Figure 5-30. Model-Simulated Nitrate Distribution After 10 Years (Based on No Further Action)

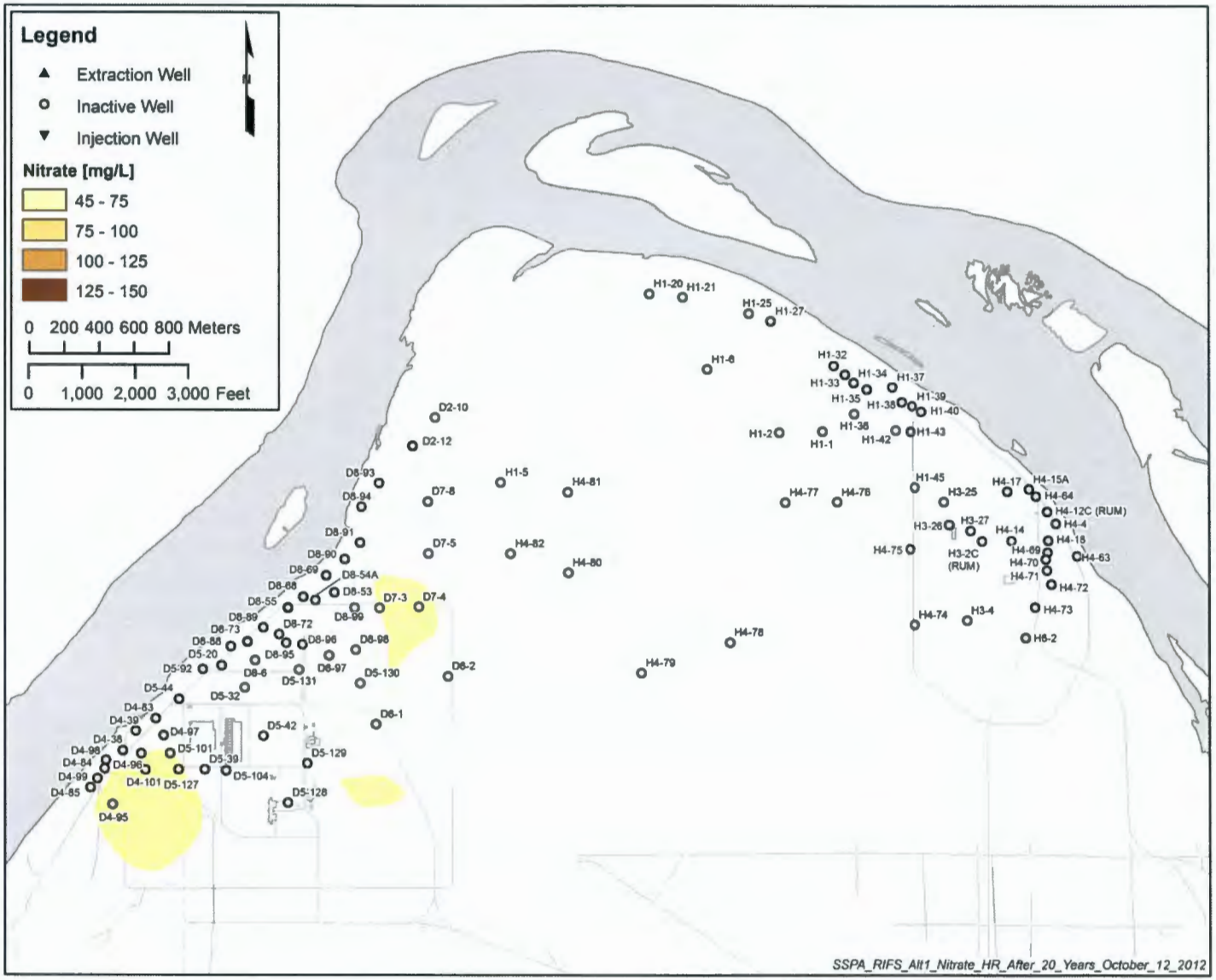


Figure 5-31. Model-Simulated Nitrate Distribution After 20 Years (Based on No Further Action)

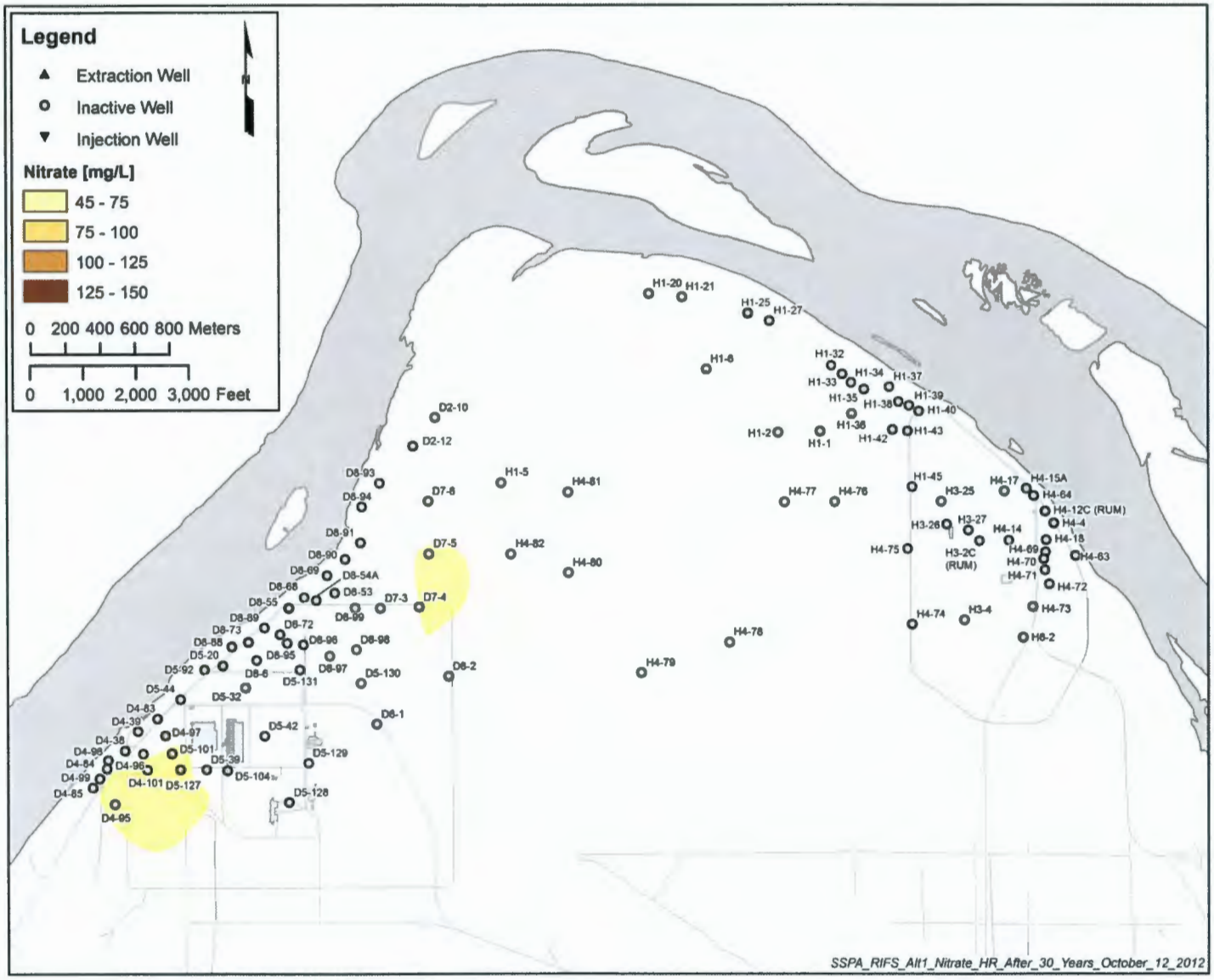


Figure 5-32. Model-Simulated Nitrate Distribution After 30 Years (Based on No Further Action)

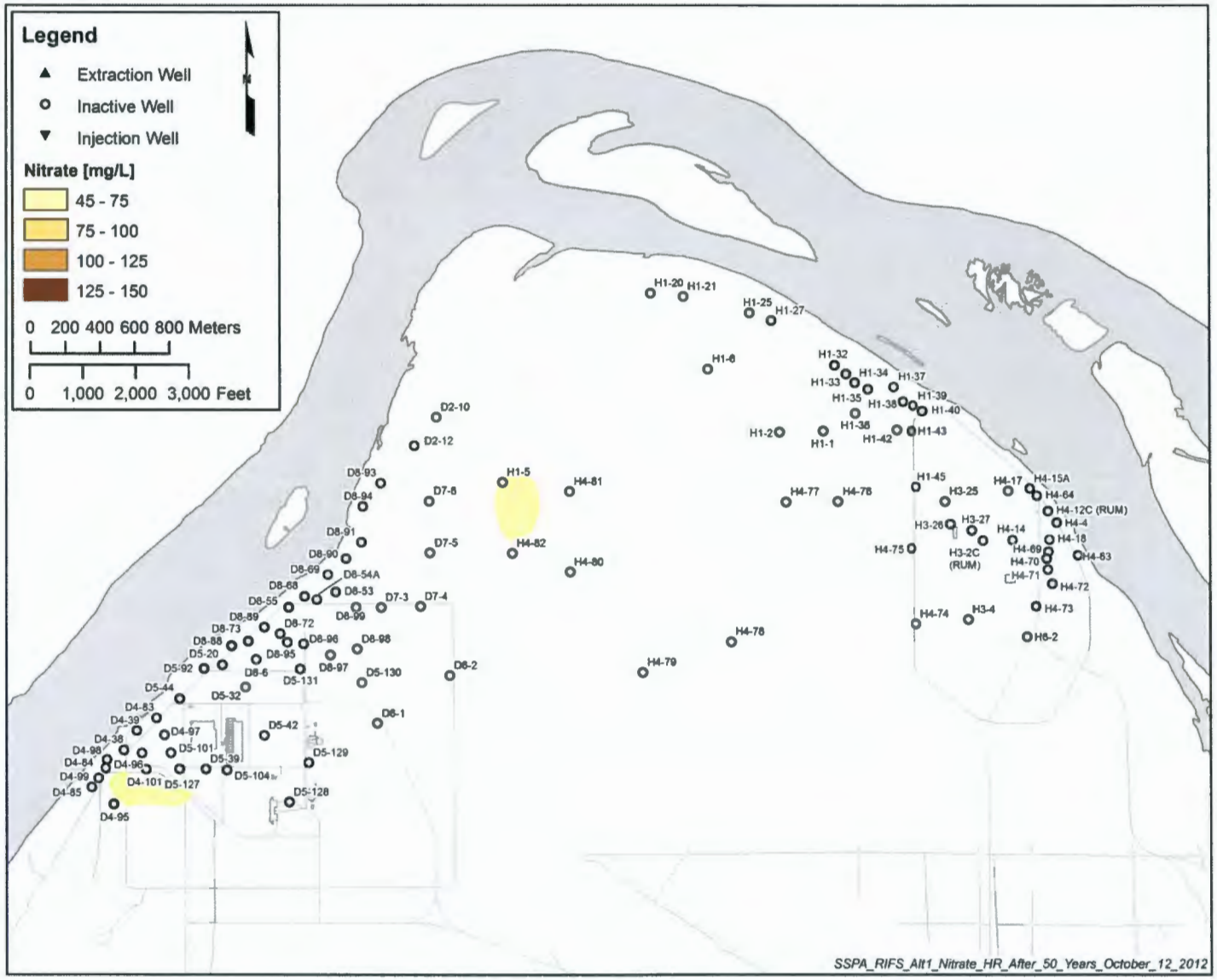


Figure 5-33. Model-Simulated Nitrate Distribution After 50 Years (Based on No Further Action)

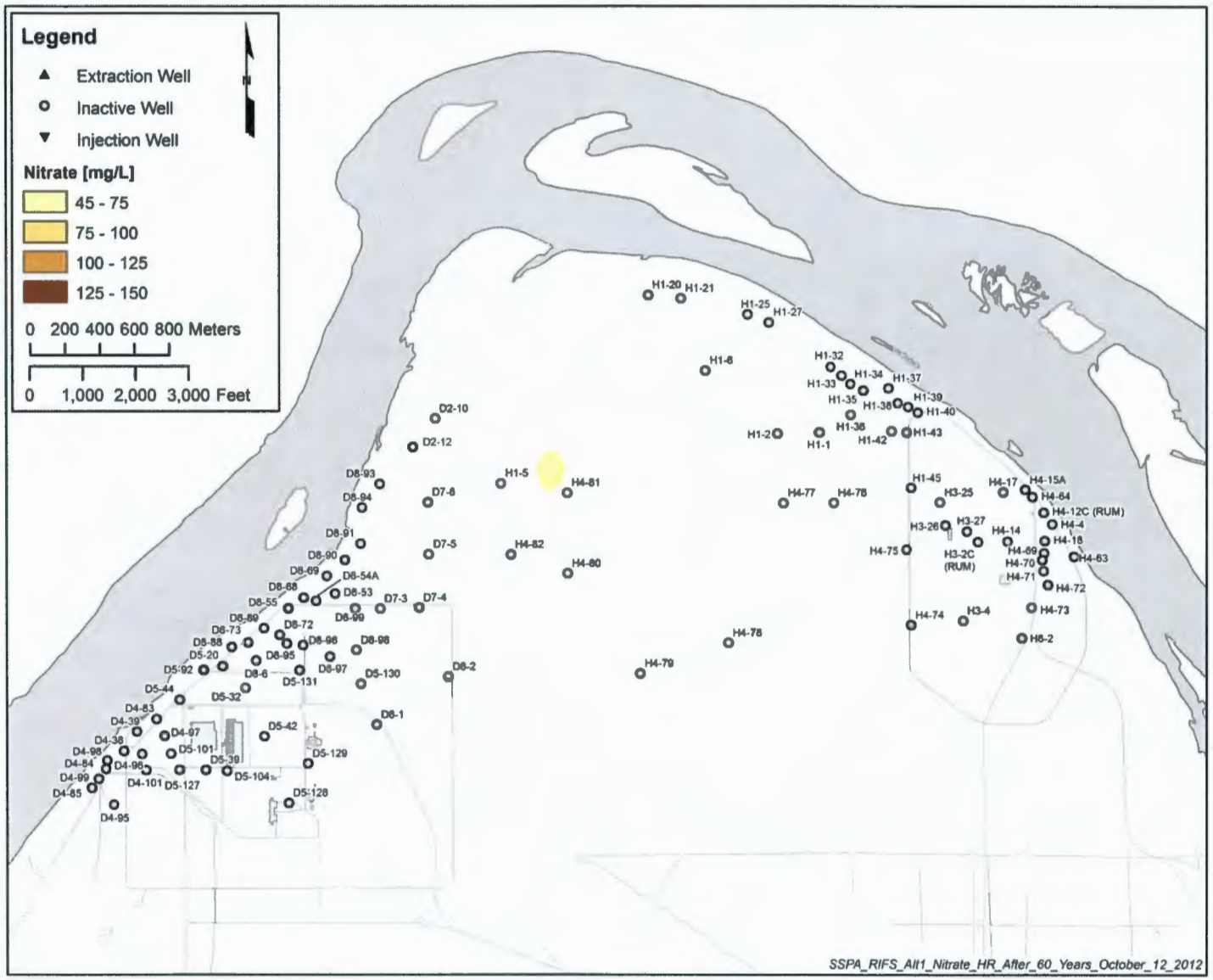


Figure 5-34. Model-Simulated Nitrate Distribution After 60 Years (Based on No Further Action)

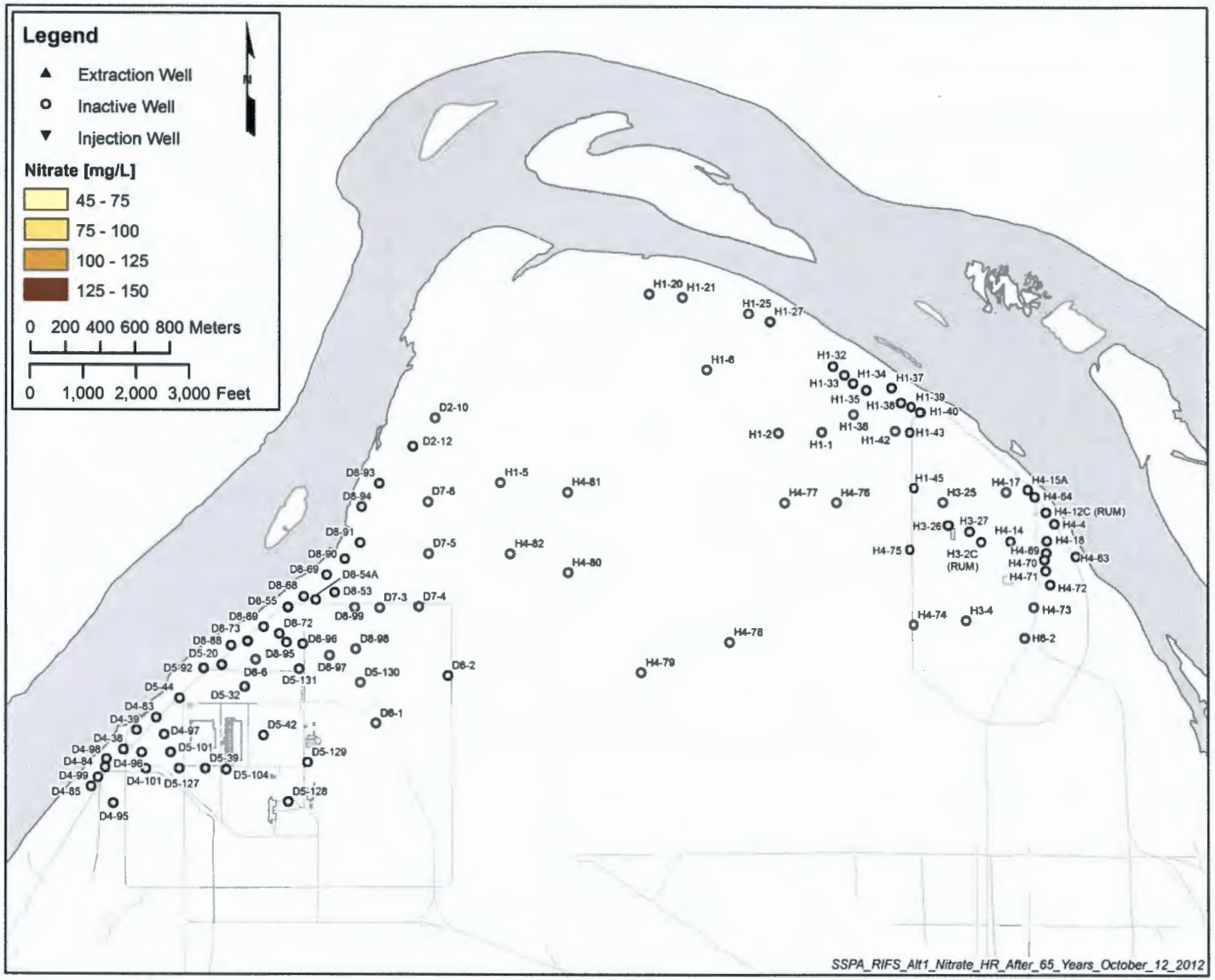


Figure 5-35. Model-Simulated Nitrate Distribution After 65 Years (Based on No Further Action)

Prediction results for Cr(VI) in groundwater underlying 100-D/H show that the highest concentrations of Cr(VI) persist along the shoreline of 100-D (Figure 5-18). The shoreline is where initial concentrations above 40,000 µg/L attenuate slowly and stay above the 10 µg/L level for more than 75 years under natural fate and transport conditions (if the pump-and-treat system is turned off at the end of CY 2012). The highest concentrations remain between the 100 and 500 µg/L concentration contour interval. Therefore, the model simulation predicts that after 75 years of natural attenuation Cr(VI) concentrations in groundwater would exceed the AWQC value of 10 µg/L at points where groundwater discharges to surface water.

When groundwater pump-and-treat remedial systems are initially developed and evaluated in an FS, a two-tiered approach to defining the system components (number of wells, well locations, and flow rates) is used. This two-tiered approach includes simulating the performance of a remedial systems (that is, mass removal and time to cleanup) using the COCs maximum concentration over times, as well as using the COCs EPC value over time. The maximum concentration simulations provide the most conservative analysis of the systems performance and the EPC simulation provides reasonably expected results (with a statistical confidence level of 95 percent). From these simulations, the systems components are identified, including a range from short times to cleanup using aggressive mass removal approaches to longer periods that remove less mass and take advantage of various degrees of natural attenuation. The remedial alternatives resulting from these simulations (using the maximum and the EPC approach) cover a broad range of performance, times to cleanup, certainty in achieving the predicted performance and overall remedy life-cycle cost. Remedy system performance data and groundwater compliance demonstration data that show cleanup levels have been met will be generated using guidance provided by Washington State ("Groundwater Cleanup Standards" [WAC 173-340-720(9)(d)(i)]) and EPA (*Methods for Evaluating The Attainment Of Cleanup Standards Volume 2: Ground Water* [EPA 230-R-92-014]). These guidance documents support the use of a broad and robust monitoring well network and the use of a statistical data presentation such as an EPC. Therefore, the remedial alternatives performance criteria are generally defined using the statistical EPC approach, while understanding the implications of the COC maximum concentrations and where in the OU those maximum concentrations occur for the longest time (e.g., hot spots and isolated areas above cleanup levels). The monitoring well network and performance monitoring program will be defined in the remedial design phase of the project. The groundwater transport model is used to present the predicted efficacy of alternative pump-and-treat strategies in the FS.

5.9 Uncertainties that Apply to Groundwater and Vadose Zone Modeling

This uncertainty discussion is based primarily on the current vadose zone and groundwater modeling objectives, and the use of these models to evaluate future conditions under no action and active remediation scenarios. Although these uncertainties exist and must be considered in decision making, conservative assumptions incorporated into the vadose zone and groundwater transport simulations can reduce the effects of uncertainty on successfully remediating 100-D/H waste sites and groundwater.

5.9.1 Uncertainty in the Conceptual Site Model

Conceptual model uncertainty is often the main uncertainty when using models to predict future contaminant fate and transport. Assumed values for vadose zone and aquifer physical properties, together with assumed values for contaminant transport properties, contribute to overall predictive uncertainty. Assumptions of spatially invariant material properties are often necessary to develop initial flow and transport models to perform and obtain acceptable calibration, despite the recognition that the processes that deposited the soil materials produce stratified and heterogeneous sequences. Local variation in vadose and/or aquifer material properties can result in contaminant transport variations. Conceptual

model uncertainty is discussed below for both vadose zone transport modeling and for groundwater transport simulations.

The assumption of constant effective porosity and saturated hydraulic conductivity values for a given stratigraphic unit is made in calculating screening levels and PRGs protective of groundwater and surface, as well as in converting the Darcy flux (as calculated by MODFLOW) to average linear groundwater velocity (as used in MODPATH and MT3DMS) for fate and transport calculations. In addition, the heterogeneity in the form of discontinuous structures (lenses), bar structures, and overbank deposits that is common at a scale below the grid size of the 100-D/H groundwater models is not accounted for in the Tier 1 screening level calculations or the Tier 2 STOMP 1D PRG model calculations. Some of these features can lead to locally faster contaminant movement than predicted by models that assume spatially invariant properties, although over broad areas, the average values for predictions will be similar whether small-scale heterogeneity is—or is not—represented. The effects of these local-scale uncertainties on predictions of groundwater and surface water protection metrics are minimized to the practical extent possible by building in conservatism using the lowest screening level and calculated PRG resulting from the STOMP simulations.

Perhaps one of the largest uncertainties in the CSM for groundwater simulations at 100-D/H is the potential for continued contribution of contaminants to groundwater from residual vadose zone sources. The groundwater contamination transport simulations discussed previously do not include the effects of any continuing releases to groundwater (that is, they assess the behavior of existing groundwater contaminant plumes only). The potential for continued release of contaminants from contaminated vadose zone soil remains uncertain. After cessation of reactor operations and cooling water treatment and disposal activities at 100-D/H, the driving force of artificial water discharge to the soil for downward movement of mobile contaminant has been largely eliminated. For example, the south groundwater Cr(VI) plume underlying 100-D has exhibited persistent elevated concentrations. This may be the result of some degree of continuing contribution from the vadose zone. The nature of this potential contribution is also uncertain. There are localized regions within the vadose zone that contain measureable quantities of mobile contaminants. Potentially, contributions may occur from natural or artificial recharge water (for example, dust-control water used during demolition) moving downward through the vadose zone and carrying mobile contaminants to groundwater. Historical groundwater monitoring data indicate that in some locations (for example, Wells 199-D5-99 and 199-D5-122), groundwater Cr(VI) concentrations exhibit increasing concentration transient trends apparently associated with periods of anthropogenic increases in groundwater elevation. This suggests the possibility that groundwater entering portions of the deep vadose zone at those locations at elevations above the normal natural seasonal fluctuation range may mobilize residual soil contaminants. Similar uncertainty may also exist at other areas of historical releases to the ground.

5.9.2 Uncertainty in the Initial Contaminant Distribution

Uncertainties with estimating contaminant distribution are primarily associated with the interpolation of individual sample contaminant concentration and the representativeness of individual samples with respect to the region surrounding the sample. The sample contaminant concentration is a minor contributor to overall uncertainty because of stringent quality controls applied by analytical laboratories. However, the representativeness in time and space of samples, together with the uncertainty associated with the interpolation of those point sample values to make a continuous distribution, is likely the greatest contributor to overall uncertainty in the initial contaminant distribution for both vadose zone and groundwater simulations.

The distribution of groundwater contaminants across the entire thickness of the saturated model domain for groundwater simulations is presented at a conservative representation of the contaminant conditions.

Investigative approaches such as collection of groundwater grab samples during well drilling help reduce this uncertainty by providing information on vertical contaminant distribution in groundwater.

Evaluation of the vertical distribution of contaminants across the full thickness of the vadose zone for the SSL and PRG simulations for low K_d contaminants and upper 70 percent of vadose zone for high K_d contaminants is believed to be representative (with conservative bias) based on observations of actual contaminant distribution made during the RI. However, data from the RI boreholes revealed cases where it was suspected this 70:30 profile might be non-conservative. A process was followed to identify cases that needed further evaluation. The conservatism in the identified cases was tested by comparing simulations of actual contamination profiles with the 70:30 profile to determine which yielded higher peak groundwater concentration predictions. The results of this testing demonstrated the 70:30 profile was conservative (that is, resulted in more restrictive SSL and PRG values than would be obtained using available profile data) for all cases evaluated. This was the case because simulating actual borehole concentration profiles yielded lower peak groundwater concentrations than for the 70:30 profile. This evaluation is described in Section 5.7.2. Had this evaluation revealed non-conservative cases, then under the graded approach to vadose zone modeling in which sites with observed conditions not well represented by the default SSL and PRG simulation conditions would have been reconsidered with site-specific modeling as described in Section 5.1, and presented on Figure 5-1. However, in 100-D/H no such cases occurred and site-specific modeling was not necessary.

A major exception to this process was made for strontium-90, which was simulated using the 100:0 profile despite a higher K_d value because of the prevalence of this constituent through the vadose zone and in groundwater in many locations, and in recognition of the risk associated with this constituent in the River Corridor.

The default SSL and PRG simulation calculations assumed that the contaminants are distributed uniformly over the vadose zone thickness (full thickness for low K_d contaminants and strontium-90, upper 70 percent of the thickness for all other high K_d contaminants) beneath the remediated waste site. If contaminants are actually limited to the near-surface portion of the vadose zone (a non-conservative condition relative to the assumptions), then the SSL and PRG simulations will be inaccurate and may lead to an overly conservative assessment of the potential threat to groundwater or surface water.

Additional site-specific vadose zone characterization and site-specific transport simulations would contribute to reducing this uncertainty. The need for additional vadose zone characterization is somewhat moderated by the consideration of robust remedial alternatives such as the remove-treat-dispose (RTD) approach to contaminated soil remediation. In cases where site-specific data indicate that the contaminant is present through the entire vadose zone thickness, particularly for contaminants with higher K_ds, the SSLs and PRGs may not be representative and the site-specific data will be used in site-specific transport simulations.

The majority of the residual contamination is expected to occur in the fine-grained (<2 mm size) portion of the sediments in the vadose zone. However, considerable uncertainty exists in the spatial variation in fraction of fine-grained material within the vadose zone. For modeling, the residual contaminant concentration determined in the laboratory on the fine-grained sediments is applied to the bulk volume, thereby increasing the initial mass estimate. This overestimation of initial mass, in turn, leads to overestimation of peak groundwater contamination in the vadose zone transport modeling, resulting in low (conservative) SSL and PRG values compared to what would be calculated with a more realistic treatment of the bulk volume.

Additional uncertainty with respect to initial contaminant concentrations is introduced by measurement of contaminants at concentrations that exceed the 90th percentile background concentration but are less than

the maximum of the background concentration range. This condition is observed at several sites in 100-D/H where arsenic EPCs exceed the 90th percentile concentration, but are less than the maximum reported background concentration (*Soil Background for Interim Use at the Hanford Site* [ECF-HANFORD-11-0038] in Appendix D). The initial contaminant concentration rules applied in general vadose zone modeling (100:0 and 70:30 profiles) function as a bounding condition with respect to the predicted peak groundwater concentration, so the impact of not using these measurements would have only a small likelihood of results in a less bounding initial condition.

5.9.3 Uncertainty in Contaminant Transport Parameters

Parameters that affect contaminant transport include the K_d value of the contaminant and soil porosity.

The K_d value of a contaminant for a soil type represents the degree of partitioning of the contaminant to the surface of the soil particles compared to the dissolved concentration. A high K_d value is typically found in contaminants such as aroclors (PCBs), which are generally considered hydrophobic. The lower the K_d, the more likely it is that the contaminant will move with water through the vadose zone. A K_d value can vary based on the water quality and chemistry (such as pH), the concentration of the contaminant, the type of sorbent, and the availability of sorption sites within the soil matrix. The K_d values used in evaluating the transport were primarily based on the assumption of dilute concentrations of contaminants in moisture within the vadose zone. These values tend to represent the more mobile K_d conditions for a particular contaminant, and therefore are considered bounding (conservative relative to prediction of contaminant arrival at a downgradient location).

Soil porosity is variable throughout the soil column based on compaction and soil type heterogeneity. As soil porosity increases, so does the contaminant mass flux to the water table; accordingly, the uncertainty in actual formation porosity will be reflected in uncertainty in the contaminant mass flux. A best estimate of the porosity is used in constructing the model. However, it is acknowledged that there is uncertainty in these estimates. *Vadose Zone Hydrogeology Data Package for Hanford Assessments* (PNNL-14702, Appendix B) provides estimates of the uncertainty in soil hydraulic parameters, including porosity, based on variability in data used to derive these model parameters. Based on the information presented in that reference, the porosity of Hanford formation sediments can be represented as normally distributed with a standard deviation of no more than 0.10. Ringold Formation sediments are slightly more variable, where porosity can be represented as normally distributed with a standard deviation no larger than 0.14.

5.9.4 Uncertainties, Assumptions, and Limitations Specific to Vadose Modeling

Uncertainties based on the numerical equations used in modeling are expected to be small. *Regulatory Basis and Implementation of a Graded Approach to Evaluation of Groundwater Protection* (DOE/RL-2011-50) provides a summary evaluation of the comparisons of field data and results to the model simulations of similar conditions using STOMP. The evaluations indicate that the STOMP code adequately simulates the natural processes. *Model Package Report: Vadose Zone Model for the River Corridor* (SGW-50776) includes a detailed evaluation of uncertainties, assumptions, and limitations to the vadose zone model used for this study.

The representativeness of soil samples collected during drilling and the resultant chemical analyses of those samples are subject to some degree of uncertainty. A limited number of soil samples in some boreholes may have been affected by water added during drilling in the vadose zone. The most likely effect of this condition, if it can be confirmed to have occurred, is that the added water may have wetted the underlying soil above its natural condition, and if wetting was sufficiently great, some movement of mobile contaminants may have occurred in soil immediately beneath the drill string at the point of water addition. The magnitude of the effect of this condition is difficult to quantify; however, it is believed that

in most instances of water intrusion into the formation, the volume of water was modest and should not have caused dramatic removal of mobile contaminants. Additional uncertainties related to specific measurements (for example, batch leaching tests) are discussed earlier in this chapter.

Assumptions within the model input parameters have an effect on the simulation outcomes. The key assumptions used for 100-D/H are as follows (with other assumptions presented in *STOMP 1-D Modeling for Determination of Soil Screening Levels and Preliminary Remediation Goals for 100 Areas D, H, and K Source Areas* [ECF-HANFORD-11-0063] in Appendix F):

- The vadose zone is considered homogeneous in nature, within the stratigraphic cross sections developed for the simulations, without consideration to the presence of thin finer grained material, which can retard the downward migration of contaminants. This constitutes a balanced representation of the vadose zone with respect to contaminant arrival time and peak groundwater concentration. If preferential pathways exist, these would function to decrease arrival time and, potentially, increase peak groundwater concentration. Such pathways are difficult to identify in most cases, but if present would be surmised to have had the largest impact prior to the present.

- Based on current revegetation activities, revegetation of a waste site after remediation is typically occurring within one to two growing seasons. In the modeling, revegetation of the area is assumed to start after 5 years, with bare soil present for the first 5 years. This assumption results in more water infiltrating to the vadose zone than may actually occur.

- The estimated recharge scenario used in development of SSLs and PRGs includes a progression from bare ground through developing shrub-steppe plant community to a long-term mature shrub-steppe community. This recharge scenario may be subject to specific uncertainty because of the potential for wildfire effects. Wildfires occur periodically (and can be characterized by a recurrence frequency), and the effects of these events would likely result in a net increase of the long-term recharge rate to groundwater underlying affected areas. This effect is due to the removal of the mature plant communities at the ground surface (effectively to bare ground) in a fire event, followed by a plant recovery succession, and culminating in the mature shrub-steppe community if the time until the next fire event allows. The magnitude of this effect on average recharge rates is not quantified, but would depend on factors such as the fire event recurrence frequency, the intensity of individual fire events, and the recovery periods for specific plant communities. A typical fire cycle would include the fire year, during which the surface is assumed to be bare ground and recharge is maximized, followed by rapid establishment of cheatgrass (*Bromus tectorum*) over 1 to 2 years. Subsequent re-establishment of the young shrub-steppe community follows, with eventual development of the mature shrub-steppe. A realistic treatment of the fire cycle should address uncertainty in the fire recurrence frequency. To account for this uncertainty, the recharge rates used in the vadose zone models are selected from the upper end of available rates based on about 30 years of field measurements (lysimeter studies) and long-term isotopic recharge studies that necessarily incorporate the effects of the history of all land surface changes at the measurement sites, including past wildfires.

- Groundwater is assumed to have negligible mixing with the Columbia River. In calculating the values for surface water protection, the point of calculation is the upper 5 m (16 ft) of groundwater at the downgradient edge of the waste site. No attenuation or decay of contaminants is assumed between the source area and groundwater further downgradient of the waste, or at the river. This results in conservative SSL and PRG estimates because most waste sites are located some distance from the river, and some mixing will occur between the waste site and locations downgradient as well as in the river.

- The 100:0 and 70:30 profiles (for low and high K_d contaminants, respectively, except that strontium-90 was evaluated using the 100:0 profile despite its higher K_d value) for SSL use an irrigation recharge scenario and assume the entire vadose zone below backfill is contaminated below clean fill for low K_d contaminants, while the upper 70 percent of the vadose zone below backfill is contaminated for high Kd contaminants. No waste sites were identified in the analysis for site-specific analysis, based on evaluation of the conservatism of the 70:30 profile for sites identified as potentially being unrepresentative in the RI borehole data reported in Chapter 4. The PRG development used these same initial source distributions, but with a recharge scenario based on reestablishment of natural net infiltration. Both the SSL and PRG simulations applied a derived Kd for Cr(VI) of 0.8 mL/g based on the results of the batch leach testing at the 100 Area.

- The initial conditions for matric potential at the start of the flow and transport simulations represent a wetter vadose zone than is expected for such gravel-dominated sediments in an arid climate, thus allowing significantly higher water and solute flux values. This is conservative in that it results in more rapid movement of water through the vadose zone.

- The median hydraulic gradient value for each source area may be too low by several-fold for waste sites near the Columbia River and may be several times too high for waste sites that are far inland from the river.

- The assumption of a 5 m (16 ft) thick aquifer may be non-conservative for those waste sites at locations where the aquifer thickness is less than 5 m (16 ft). As noted in Section 3.6.1, the saturated thickness of the unconfined aquifer thins towards the 100-H Area. The mapping of aquifer thickness presented in Section 3.6.1 shows, for times of year with lower river stages (for example, September) that the 5 m (16 ft) assumption is, at least seasonally, not representative. The process for developing the representative stratigraphic columns for the 100-H waste sites (Section 5.4.2) that incorporated a 5 m (16 m) aquifer included consideration of the seasonal fluctuation in aquifer thickness; however, the conservatism was placed on minimizing the vadose zone thickness by using the highest annual water table. This was, at least for some portion of the year, non-representative with respect to dilution in the saturated zone.

5.9.5 Uncertainties, Assumptions, and Limitations Specific to Groundwater Modeling

Uncertainties based on the numerical equations used in modeling are expected to be small. A groundwater flow and contaminant transport model has been developed and calibrated for remedy design evaluation purposes in the 100 Area. The model development and calibration is documented in a comprehensive modeling report (*Conceptual Framework and Numerical Implementation of 100 Areas Groundwater Flow and Transport Model* [SGW-46279]).

The groundwater flow model grid encompasses all 100 Area OUs. The model finite-difference grid is constructed so that the north and northeast boundaries of the flow model parallel and abut the Columbia River. The model extends southward, toward Gable Butte and Gable Mountain. The grid spacing is relatively coarse (about 100 m [328 ft]) throughout much of the domain, but it is refined (15 m [49 ft]) in the area of each 100 Area OU to support remedy evaluations.

Assumptions within the model input parameters have an effect on the simulation outcomes. The key assumptions used for 100-D/H are as follows, with other assumptions and the specific input parameters presented in *Conceptual Framework and Numerical Implementation of 100 Areas Groundwater Flow and Transport Model* (SGW-46279) in Appendix F.

Predictive simulations were based on transient-state (that is, time-varying) conditions in the aquifer that reflect water level changes because of river stage variation. The modeling period corresponds to a 77-year period (CY 2011 to 2087). For the period 2011 to 2037, the modeling period consists of a series of 12 monthly stress periods that are repeated in the same sequence. The stress periods correspond to monthly average river stages, each representing the average river stage for the particular calendar month over the period 2006 to 2010 (excluding 2007 values, when the river stage variation pattern was inconsistent with the other years). It is assumed that these conditions are representative of the typical conditions in the field and that future conditions will not vary significantly.

Groundwater flow is simulated as 3D using four layers to represent the Hanford formation (always present in Layer 1) and the Ringold Formation unit E (typically represented by Layers 2 through 4). The base of the model is assumed to be the top of the RUM where present and the top of the basalt where the RUM is absent. Underlying 100-D, the water table mainly occurs within the Ringold Formation unit E. Across the horn area, east and northeast of 100-D, the water table occurs primarily within the Hanford formation. The water-bearing units within the RUM are not included in this representation; as noted earlier, Cr(VI) contamination occurs in the first water-bearing unit of the RUM but is limited in areal extent beneath 100-H and at one location in the Horn, near 100-D. Further east into the horn area, no Cr(VI) contamination is observed in this unit. Thus, the uncertainty introduced to the model from omitting this feature is considered low. In *Evaluation of Potential Hydraulic Capture and Plume Recovery from the Ringold Upper Mud (RUM) in the 100-HR-3 Operable Unit (OU)* (ECF-100HR3-12-0025), screening-level calculations are performed in support of the 100-HR-3 RI/FS to evaluate time-dependent zones-of-contribution and contaminant recovery under various pumping scenarios from wells screened in the upper portion on the RUM.

The principal aquifer property specified in the flow model is the spatially varying hydraulic conductivity of the saturated aquifer materials. Estimates of hydraulic conductivity compiled as part of the model data package were tabulated and assigned to their corresponding aquifer unit. Following are the mean values for the aquifer hydraulic conductivity that resulted from the model calibration process:

- 19 m/day (62 ft/day) for the Ringold Formation unit E
- 63 m/day(206 ft/day) for the Hanford formation

Areal recharge from precipitation was specified based on information included in *Groundwater Data Package for Hanford Assessments* (PNNL-14753). An electronic version of the recharge package developed in this report was obtained, and the data were spatially distributed to the model grid cells and were subsequently adjusted during model calibration. Based on the results of the model calibration, the recharge value was set equal to 12 mm/yr throughout much of the model domain. This value is a spatially based average of distributed recharge rates applied over a large area with variable surface soil types and vegetation cover types for present day conditions represented by the groundwater model. As such, this rate is not directly comparable to, but is consistent with, recharge rates applied for vadose zone models of individual waste sites.

Effective porosity and specific yield values for the entire aquifer were identified from published sources and revised during the model calibration and are equal to 18 and 10 percent, respectively. Both values are within the range of values documented in previous investigations for the Hanford Site (*Development of a Three-Dimensional Ground-Water Model of the Hanford Site Unconfined Aquifer System: FY 1995 Status Report* [PNNL-10886]).

The initial distribution of each COC in groundwater within the 100-HR-3 OU was obtained using maximum sampled COC concentrations at each monitoring location during the period 2009 to 2010.

It is assumed that no continuous source is present in the aquifer or vadose zone that would affect the contaminant distribution.

5.10 Summary of Contaminant Fate and Transport

Intentional and unintentional releases of primary waste source materials occurred during nuclear material production at the Hanford Site. The EPCs of each remediated waste site, soil group, and COPC, as well as the results of vadose zone soil analysis for soil samples collected during the previous CVP, LFI and this RI, were compared to the SSLs calculated using the irrigation recharge scenario with 100:0 and 70:30 profiles as well as the PRGs calculated using the nominal scenario (that is, non-irrigated, natural recharge conditions) with 100:0 and 70:30 profiles. After excluding COPCs with peak concentration times greater than 1,000 years, no waste sites were found to exhibit EPCs greater than the SSLs for residual contamination in the vadose zone. None of the EPC concentrations for metals fell outside the reported background concentration range for Hanford Site soils. Metals are believed to be representative of naturally occurring background concentrations, with potential contribution from historical agricultural application of lead arsenate pesticides to orchards that pre-dated Hanford Site operations near 100-H. Waste sites that have not yet been remediated were carried into the FS for evaluation with COPCs based on process knowledge.

Groundwater contaminant flow and transport modeling over an extended future period and historical monitoring indicate that the groundwater pump-and-treat systems have provided, and will continue to provide, protection to the Columbia River along the shoreline in almost all areas.

The source area waste sites that have been remediated under interim action did not exceed the SSLs or PRGs protective of groundwater and surface water for Cr(VI). While the RI data indicate that contamination in the vadose zone has been remediated, groundwater monitoring indicates that there is potential for low level residual contamination. Unremediated waste sites have significant inventory in the shallow vadose zone. At some locations (for example, the 100-D-100 and the 100-D-104 waste sites), RTD down to the water table may be required to completely remediate the contaminated soil.

In 100-H, pumping is currently being conducted within the first water-bearing unit of the RUM. Characterization of the RUM confirmed that Cr(VI) contamination consistent with cooling water is present in this horizon below the unconfined aquifer only in localized portions of 100-H. The cross sections presented in Chapter 3 indicate that a significant thickness of silt occurs between the contaminated RUM water-bearing unit beneath 100-H and the channel of the river, blocking the pathway further out from the shoreline. Therefore, continued pumping of this water-bearing unit will capture the Cr(VI) and protect the river farther out into the channel from the shoreline.

Plume migration patterns, as estimated by the model, indicate a diminishing footprint of the Cr(VI) plume because of pump-and-treat operations. Concentrations in groundwater above 20 μg/L appear sufficiently controlled by the current combined extraction/injection activity across the area of interest as shown by the hydraulic containment in 2012 evaluation (Section 5.8.3.1). Model results indicate that concentrations between 10 and 20 μg/L are fairly well contained, except in areas west of 100-D and east of 100-H where the plume slowly discharges to the river. Based on the modeling results, Cr(VI) concentrations in groundwater underlying 100-D/H will decline over time, although the rate of decline is not uniform across the area of interest.

Strontium-90 concentrations in groundwater above the MCL are within the capture zone of the recovery wells at 100-D, but small areas are outside the capture zone at 100-H. Recirculated strontium-90 concentrations reinjected into the aquifer are always below the MCL and modeling results suggest that concentrations will decline over time.

Nitrate concentrations in groundwater above the MCL are within the capture zone of the recovery wells although a small area is outside the capture zone in 100-D south. Recirculated nitrate concentrations injected back into the aquifer are always below the MCL and modeling results suggest that concentrations will decline slowly over time.

Simulation of the base case groundwater contaminant plume migration indicates that there is a clear basis for remedial action to address the existing plumes underlying 100-D/H. Turning off the pump-and-treat systems at the end of 2012 (as represented in the base case groundwater modeling) will result in the existing plumes persisting and slowly discharging into the Columbia River. Without the implementation of remedial action, such as the current pump-and-treat systems, unacceptably large concentrations of groundwater contaminants (for example, Cr[VI]) will continue to discharge to the Columbia River.

The evaluation of EPCs indicated that remediated waste sites should not contribute to continuing groundwater contamination based on lack of any exceedances of SSLs protective of groundwater or surface water (Section 5.7.3). Uncertainties remain regarding the potential for continued contribution of residual vadose zone contaminants to underlying groundwater. Strategies for addressing potential residual contamination will be discussed in the FS. Remedial alternatives evaluated in the FS portion of this report will consider monitoring requirements that will verify the assumptions for vadose zone contaminant behavior. Existing groundwater plumes of Cr(VI), strontium-90, and nitrate near the reactor condensate cribs, the FSBs, the retention basins, the cribs, and the cooling water head houses should be considered for specific monitoring of potential future vadose zone contributions.

Chapter 5 describes and predicts how quickly or slowly contaminants migrate and their potential to enter the Columbia River. The potential to be harmful depends on specific human and environmental receptors, as well as exposure times and patterns that might bring receptors and contaminates into contact. The ways that the contaminants could come into contact with, and affect, human health and the environment are called pathways. Chapter 6 addresses the human health pathway; scenarios of how humans might come into contact with contaminants in the setting with resultant health effects are evaluated. Chapter 7 addresses the biological receptor pathway. Scenarios of how plant, animal, bird, or invertebrate species might come into contact with contaminants in the setting and be affected are evaluated in Chapter 7.

6 Human Health Risk Assessment

The integration of past and current HHRAs supports the development of remedial alternatives for waste sites and contaminated groundwater in the 100-D/H decision area. These risk assessments have been integrated with the cleanups performed under the interim action RODs to identify the need for further remedial action and, if needed, to develop PRGs.

Highlights

- Principal soil contaminants identified at one or more waste sites through the risk assessment included radionuclides, metals, PCBs, and PAHs.
- The baseline risk assessment identified Cr(VI), chromium, strontium-90, and nitrate as final groundwater COPCs for evaluation of potential remedial technologies in the FS.
- Data and process knowledge indicate that human health PRGs would be exceeded at unremediated waste sites and provide the basis for action.

As described in the previous sections, the remedial actions completed to date in the River Corridor were implemented primarily under interim action RODs. There is a requirement under CERCLA to perform a baseline risk assessment (BRA) to characterize current and potential threats to human health and the environment before final action RODs for final remedies can be issued. The RCBRA was prepared to address the regulatory requirement that a baseline risk assessment be performed. The RCBRA Report (DOE/RL-2007-21, Volume II) is a comprehensive HHRA for the River Corridor considering relevant sources of contamination, exposure pathways, and contaminants to evaluate current and potential future risks posed by hazardous substance releases. The following is the purpose of the RCBRA, as described in Section 1.1:

> *The purpose of the RCBRA is to characterize current and potential future risks to human health and the environment that may be posed by releases of hazardous substances in the River Corridor of the Hanford Site. DOE is required to assess human and ecological risk under CERCLA, Resource Conservation and Recovery Act of 1976 (RCRA), National Environmental Policy Act of 1969, and DOE orders. The "National Oil and Hazardous Substances Contingency Plan" (40 Code of Federal Regulations [CFR] 300), which implements CERCLA, specifically requires a site-specific baseline risk assessment to determine the need for action at sites, determine levels of contaminants that can remain onsite and still be protective, and provide a basis for comparing health impacts of various cleanup alternatives (40 CFR 300.430[d][4]).*

Per the risk assessment guide (EPA/540/1-89/002), a baseline risk assessment is an "analysis of the potential adverse health effects (current or future) caused by hazardous substance releases from a site in the absence of any actions to control or mitigate these releases (that is, under an assumption of no action)."

The baseline risk assessment is part of the CERCLA RI/FS process. The RI/FS is the methodology that the CERCLA program has established for characterizing the nature and extent of contamination associated with releases of hazardous substances to the environment, for assessing the potential risks posed by the environmental contamination to human and ecological receptors, and for developing and evaluating remedial options. Because the RI/FS is a process designed to support risk management decision making for CERCLA sites, the assessment of human health and environmental risk serves an essential role in the RI/FS process. The baseline risk assessment provides information to assist in the development, evaluation, and selection of appropriate response alternatives. The results of the baseline risk assessment are used to determine whether additional response action is necessary at the Site; support development of PRGs; support selection of the "no action" remedial alternative where it is appropriate; and document the magnitude of risk and primary contributors (for example, chemicals and exposure pathways) to risk at a site.

Interim action RODs were written for River Corridor sites to allow cleanup activities to move forward as potential risks were identified. However, final remedy selection must be completed in order for the NPL (40 CFR 300, Appendix B) CERCLA sites in the River Corridor to reach final closeout. One of the key evaluations needed to establish final action RODs for sites in the River Corridor was a baseline risk assessment (*Risk Assessment Work Plan for the 100 Area and 300 Area Component of the RCBRA* [DOE/RL-2004-37]). The RCBRA (DOE/RL-2007-21, Volume II) HHRA and the companion ecological risk assessment (DOE/RL-2007-21, Volume I) provided an evaluation of ecological and human health risk from residual contamination at waste sites remediated under the interim action RODs and from potentially affected environmental media under various exposure scenarios. Unacceptable risks are present in the River Corridor at waste sites that are identified in the IARODs but have yet to be remediated. The determination of the presence of unacceptable risk and basis for action at yet-to-be remediated waste sites is supported by field investigation data as well as information gathered through implementation of the observational-approach soil cleanup actions in the River Corridor over the past 15 years. The Site-specific risk information provided by the RCBRA (DOE/RL-2007-21) would be used to support final action RODs for the River Corridor.

6.1 Role of the RCBRA and the RI/FS Risk Assessment

The RCBRA (DOE/RL-2007-21) provided the following range of analyses:

- Assessment of residual risks for remediated waste sites using the Unrestricted Land Use exposure scenario that was the basis for the remedial action goals for the interim action ROD cleanup in the 100 Area
- An assessment of risks for several yet-to-be remediated waste sites using a broad range of exposure scenarios
- Assessment of residual risks for remediated waste sites and broad areas[1] using a broad range of exposure scenarios

Portions of these analyses were considered in the HHRA approach used to develop soil PRGs that are presented in the RCBRA (DOE/RL-2007-21). The following issues are addressed in this chapter as part of the integration of RCBRA and the RI/FS, which will support the development of final action RODs for the 100 Area decision areas:

- Incorporation of PRG values from the RCBRA (DOE/RL-2007-21) for radioisotopes and chemicals based on updated regulatory guidance
- Inclusion of all decision units[2] associated with a remediated waste site
- Inclusion of analytical data from focused sampling designs
- Analysis time frame (that is, waste sites cleaned up after the analysis conducted in the RCBRA [DOE/RL-2007-21])
- Use of EPCs consistent with the waste site decision units (for example, shallow zone, deep zone) and based on current EPA guidance

[1] The term "broad area" is used in the RCBRA to refer to an exposure area that could potentially be as large as an individual interim action ROD decision area or as large as the entire River Corridor.

[2] The floor and sidewalls of an excavated waste site are divided into one or more decision units. A sample design is developed for each decision unit. See Section 6.2.2.2 for additional information.

The following sections discuss the integration of the RCBRA (DOE/RL-2007-21) and the RI/FS risk assessment:

- Section 6.1.1 summarizes the evaluation of residual risks performed in the RCBRA (DOE/RL-2007-21) for waste sites cleaned up under the interim action ROD. The results from this evaluation have been compared with the PRGs developed in the RCBRA (DOE/RL-2007-21) for use in the RI/FS.
- Section 6.1.2 and 6.1.3 describe Unrestricted Land Use and other scenarios used in the River Corridor, their associated uncertainties, and the way they have been incorporated into the RI/FS.

The HHRA supporting the RI/FS is presented in two sections. Section 6.2 presents the methods and the results for the soil risk assessment and Section 6.3 presents the methods and results for the analysis of groundwater risks.

The soil risk assessment supporting the RI/FS (Section 6.2) provides the data analysis (Section 6.2.1), estimated EPCs (Section 6.2.2), exposure assessment (Section 6.2.3), toxicity assessment (Section 6.2.4), risk characterization (Section 6.2.5), and the uncertainties assessment (Section 6.2.6).

The groundwater risk assessment supporting the RI/FS (Section 6.3) discusses findings and uncertainties of the RCBRA (DOE/RL-2007-21) (Section 6.3.1). The Integrated Work Plan (DOE/RL-2008-46) adds activities that would help reduce uncertainties, verify conclusions, and ensure that no contaminants were inadvertently overlooked based on the use of the existing dataset. The risk assessment involves the following steps: identification of COPCs (Section 6.3.2), exposure assessment (Section 6.3.3), toxicity assessment (Section 6.3.4), risk characterization (Section 6.3.5), risk characterization using action levels (Section 6.3.6), the tap water risk characterization (Section 6.3.7), and the uncertainties assessment (Section 6.3.9). The results of Section 6.3 will be used to identify COPCs, which represent contaminants that will be evaluated in the FS to define the COCs and guide the selection of remedial alternatives.

Section 6.4 presents conclusions of the riparian and near shore environment from the RCBRA (DOE/RL-2007-21, Volume II) and conclusions from the CRC (DOE/RL-2010-117, Volume II) are in Section 6.4.1. Section 6.5 presents a summary and conclusions for the soil risk assessment (Section 6.5.1) and the groundwater risk assessment (Section 6.5.2).

6.1.1 Evaluation of Residual Risks for Interim Action ROD Cleanups from the RCBRA

This section discusses the results of the screening-level evaluation presented in Chapter 2 of the RCBRA (DOE/RL-2007-21, Volume II). It also compares the results from the screening-level evaluation to the methodology used to develop the interim action remedial action goals and describes how analytical data from CVP/RSVP were used in the screening evaluation. Finally, the screening-level risk results from the RCBRA (DOE/RL-2007-21, Volume II) are compared to the results of the soil risk assessment. The risk results from the soil risk assessment are based on guidance and exposure assumptions that have been updated since the interim action remedial action goals were published. The methods used in the risk assessment are described in Sections 6.1.2 and 6.1.3.

Chapter 2 of the RCBRA (DOE/RL-2007-21, Volume II) presents a screening-level assessment of residual cancer risks and noncancer hazards for the remediated waste sites using the exposure scenarios that were the basis of the residential remedial action goals for the interim action ROD cleanups in the 100 Area. This assessment was done to provide information about the residual risks and noncancer hazards associated with post-interim action conditions at the remediated waste sites and help assess whether residual conditions are protective of human health.

Interim action ROD cleanup activities for the 100 Areas were based on an unrestricted scenario that was the basis for the remedial action goals. The interim action ROD residential scenario for radionuclides is a Rural Residential scenario that, in addition to direct contact, includes food chain exposure pathways (for example, ingestion of homegrown produce, beef, and milk). The interim action ROD residential scenario for chemicals is based on the MTCA Method B Soil Cleanup Levels ("Unrestricted Land Use Soil Cleanup Standards" [WAC 173-340-740]). The MTCA (WAC 173-340) Method B levels are based solely on incidental soil ingestion and do not address the food exposure pathways that were included for the radionuclide Rural Residential scenario[3]. The interim action remedial action goal for arsenic was based on the MTCA Method A soil cleanup level ("Unrestricted Land Use Soil Cleanup Standards" [WAC 173-340-740]). The interim action remedial action goal for lead was calculated using *Guidance Manual for the Integrated Exposure Uptake Biokinetic Model for Lead in Children* (EPA/540/R-93/081). It should be noted that the radionuclide PRGs for the residential scenario used in the soil risk assessment incorporate exposure assumptions that were updated to reflect current EPA guidance as described in Section 6.1.2.

CVPs or RSVPs were prepared to document completion of interim action ROD cleanup actions in accordance with the applicable decision document and support waste site reclassification. The screening-level calculations presented in Chapter 2 of the RCBRA (DOE/RL-2007-21) use the interim action ROD risk assessment models, but differ from the calculations used in the CVPs and RSVPs to document the interim action ROD cleanups.

Twenty-eight waste sites from the 100-D Source OU and eight wastes sites from the 100-H Source OU were evaluated in the RCBRA (DOE/RL-2007-21). Sixty-seven additional waste sites at the 100-D Source OU and 39 additional waste sites at the 100-H Source OU have been remediated since 2005, and are not addressed in the RCBRA. Residual cumulative cancer risks from chemicals evaluated in the RCBRA are less than 1×10^{-5} using the interim action ROD residential scenario (that is, MTCA Method B Unrestricted Land Use scenario). This is with the exception of 100-H-21, where the risk driver is arsenic with a reasonable maximum exposure (RME) concentration of 13.8 mg/kg, which is less than the direct exposure remedial action goal of 20 mg/kg published in the 100 Area RDR/RAWP (DOE/RL-96-17).

Residual cumulative cancer risks from radionuclides for all remediated waste sites are less than 1×10^{-4} based on the interim action ROD Rural Residential scenario with the exception of the following waste sites:

- 100-D-48:3
- 116-DR-9

The noncancer hazard indices (HIs) for chemicals do not exceed a threshold of 1 at the 28 100-D remediated waste sites and the 8 100-H waste sites. A summary of the risk assessment results for a residential scenario using approaches from both the RCBRA and the RI/FS is provided in Tables 6-1 to 6-3.

[3] Note that for beryllium, cadmium, and Cr(VI), the interim action remedial action goal for direct contact is based on the inhalation pathway.

Table 6-1. Summary of Residential Scenario Risk Assessment Results for Chemical Carcinogens

Waste Site Name	RI/FS Decision Unit	Chemical Carcinogens and Cancer Risk Drivers			
		RCBRA Chemical Risk	RCBRA Chemical Risk Driver	RI/FS Chemical Risk	RI/FS Chemical Risk Driver
100-D Source OU Waste Sites					
100-D-4	Shallow	2×10^{-7}	None	3.8×10^{-7}	None
100-D-12	Shallow	--	--	--	--
100-D-20	Shallow	--	--	--	--
100-D-21	Shallow	--	--	--	--
100-D-22	Shallow	--	--	--	--
100-D-48:1	Shallow	--	--	--	--
100-D-48:2	Shallow	--	--	--	--
100-D-48:3	Shallow	--	--	--	--
100-D-48:4	Shallow	--	--	--	--
100-D-49:2	Shallow	--	--	--	--
100-D-49:4	Shallow	--	--	3.4×10^{-7}	None
100-D-52	Shallow	--	--	--	--
116-D-1A	Shallow	--	--	--	--
116-D-2	Shallow	--	--	--	--
116-D-4	Shallow	--	--	--	--
116-D-7	Shallow	--	--	--	--
116-D-9	Shallow	--	--	--	--
116-DR-1&2	Shallow	--	--	--	--
116-DR-4	Shallow	--	--	--	--
116-DR-6	Shallow	--	--	--	--
116-DR-7	Shallow	--	--	--	--
116-DR-9	Shallow	--	--	--	--
118-DR-2:2	Shallow	5×10^{-6}	Arsenic (5×10^{-6})	4.9×10^{-6}	Arsenic (4.8×10^{-6})
122-DR-1:2	Shallow	5×10^{-6}	Arsenic (5×10^{-6})	5.0×10^{-6}	Arsenic (4.7×10^{-6})
1607-D2:1	Shallow	2×10^{-6}	Arsenic (2×10^{-6})	1.7×10^{-6}	Arsenic (1.7×10^{-6})
1607-D2:3	Shallow	--	--	3.5×10^{-10}	None
1607-D2:4	Shallow	--	--	--	--
1607-D4	Shallow_Focused	2×10^{-6}	Arsenic (2×10^{-6})	2.3×10^{-6}	Arsenic (2.3×10^{-6})

Table 6-1. Summary of Residential Scenario Risk Assessment Results for Chemical Carcinogens

Waste Site Name	RI/FS Decision Unit	Chemical Carcinogens and Cancer Risk Drivers			
		RCBRA Chemical Risk	RCBRA Chemical Risk Driver	RI/FS Chemical Risk	RI/FS Chemical Risk Driver
100-H Source OU Waste Sites					
100-H-5	Shallow	6×10^{-6}	Arsenic (6×10^{-6})	6.9×10^{-6}	Arsenic (6.9×10^{-6})
100-H-17	Shallow	5×10^{-6}	Arsenic (5×10^{-6})	5.1×10^{-6}	Arsenic (5.1×10^{-6})
100-H-21	Shallow	2×10^{-5}	Arsenic (2×10^{-5})	2×10^{-5}	Arsenic (2.0×10^{-5})
100-H-24	Shallow	6×10^{-6}	Arsenic (6×10^{-6})	6.4×10^{-6}	Arsenic (6.3×10^{-6})
116-H-1	Shallow	1×10^{-5}	Arsenic (1×10^{-5})	9.9×10^{-6}	Arsenic (9.9×10^{-6})
116-H-7	Shallow	8×10^{-6}	Arsenic (8×10^{-6})	9.6×10^{-6}	Aroclor-1260 (1.3×10^{-6}) Arsenic (8.3×10^{-6})
1607-H2	Shallow	1×10^{-5}	Arsenic (1×10^{-5})	1.1×10^{-5}	Arsenic (1.1×10^{-5})
	Shallow_Focused	--	--	2.3×10^{-6}	Arsenic (2.3×10^{-6})
1607-H4	Shallow	Not Evaluated	Not Evaluated	1.5×10^{-5}	Arsenic (1.1×10^{-5}) Benzo(a)pyrene (2.8×10^{-6})

Notes:

Chemical drivers shown have an associated risk greater than 1×10^{-6}.

The risk value for the individual drivers is shown in parentheses after the name of the risk driver chemical.

Risks are based on reasonable maximum EPCs.

Source:

RCBRA data: *River Corridor Baseline Risk Assessment, Volume II: Human Health Risk Assessment* (DOE/RL-2007-21), Volume II, Part 2, Table 2-10.

RI/FS data: (DOE/RL-2010-95), Appendix G, Table G-17 (100-D) and G-36 (100-H).

-- = Carcinogenic COPCs were not identified.

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Table 6-2. Summary of Residential Scenario Risk Assessment Results for Noncarcinogens

Waste Site Name	RI/FS Decision Unit	Noncancer Hazard Index and Noncancer Hazard Drivers			
		RCBRA Hazard Index	RCBRA Chemical Hazard Driver	RI/FS Hazard Index	RI/FS Chemical Hazard Driver
100-D Source OU Waste Sites					
100-D-4	Shallow	0.001	None	0. 053	None
100-D-12	Shallow	--	--	--	--
100-D-20	Shallow	0.01	None	0.011	--
100-D-21	Shallow	--	--	--	--

Table 6-2. Summary of Residential Scenario Risk Assessment Results for Noncarcinogens

Waste Site Name	RI/FS Decision Unit	Noncancer Hazard Index and Noncancer Hazard Drivers			
		RCBRA Hazard Index	RCBRA Chemical Hazard Driver	RI/FS Hazard Index	RI/FS Chemical Hazard Driver
100-D-22	Shallow	--	--	0.01	--
100-D-48:1	Shallow	0.01	None	0.008	None
100-D-48:2	Shallow	0.01	None	0.006	--
100-D-48:3	Shallow	0.01	None	0.01	None
100-D-48:4	Shallow	0.01	None	0.015	None
100-D-49:2	Shallow	0.005	None	0.004	--
100-D-49:4	Shallow	0.01	None	0.11	None
100-D-52	Shallow	0.01	None	0.011	None
116-D-1A	Shallow	0.01	None	0.009	None
116-D-2	Shallow	0.01	None	0.006	--
116-D-4	Shallow	0.01	None	0.005	--
116-D-7	Shallow	0.01	None	0.011	None
116-D-9	Shallow	0.01	None	0.007	--
116-DR-1&2	Shallow	0.01	None	0.006	--
116-DR-4	Shallow	0.01	None	0.006	--
116-DR-6	Shallow	0.01	None	0.006	--
116-DR-7	Shallow	0.01	None	0.007	--
116-DR-9	Shallow	0.01	None	0.008	None
118-DR-2:2	Shallow	0.14	None	0.15	None
122-DR-1:2	Shallow	0.13	None	0.23	None
1607-D2:1	Shallow	0.06	None	0.06	None
1607-D2:3	Shallow	0.002	None	0.007	None
1607-D2:4	Shallow	0.01	None	0.009	None
1607-D4	Shallow_Focused	0.07	None	0.56	None
100-H Source OU Waste Sites					
100-H-5	Shallow	0.19	None	0.20	None
100-H-17	Shallow	0.15	None	0.15	None
100-H-21	Shallow	0.58	None	0.56	None
100-H-24	Shallow	0.17	None	0.17	None
116-H-1	Shallow	0.29	None	0.28	None

Table 6-2. Summary of Residential Scenario Risk Assessment Results for Noncarcinogens

Waste Site Name	RI/FS Decision Unit	Noncancer Hazard Index and Noncancer Hazard Drivers			
		RCBRA Hazard Index	RCBRA Chemical Hazard Driver	RI/FS Hazard Index	RI/FS Chemical Hazard Driver
116-H-7	Shallow	0.23	None	0.24	None
1607-H2	Shallow	0.54	None	0.52	None
	Shallow_Focused	Not Evaluated	Not Evaluated	0.07	None
1607-H4	Shallow	--	--	0.32	None

Notes:

Chemical drivers shown have an associated hazard quotient (HQ) greater than 1. The HQ for the individual drivers is shown in parentheses after the name of the risk driver chemical.

HIs are based on reasonable maximum EPCs.

Sources:

RCBRA data: *River Corridor Baseline Risk Assessment, Volume II: Human Health Risk Assessment* (DOE/RL-2007-21), Part 2, Table 2-10.

RI/FS data: Appendix G, Table G-17 (100-D) and G-36 (100-H).

--- = Noncarcinogenic COPCs were not identified.

Table 6-3. Summary of Residential Scenario Risk Assessment Results for Radionuclides

Waste Site Name	RI/FS Decision Unit	Radionuclides and Radiological Risk Drivers			
		RCBRA Radiological Risk	RCBRA Radiological Risk Driver	RI/FS Radiological Risk	RI/FS Radiological Risk Driver
100-D-4	Shallow	3×10^{-5}	None	1.7×10^{-5}	None
100-D-12	Shallow	--	--	--	--
100-D-20	Shallow	9×10^{-5}	None	5.3×10^{-5}	None
100-D-21	Shallow	8×10^{-6}	None	6.4×10^{-7}	None
100-D-22	Shallow	--	--	3.1×10^{-5}	None
100-D-48:1	Shallow	4×10^{-5}	None	2.4×10^{-5}	None
100-D-48:2	Shallow	1×10^{-4}	None	5.1×10^{-5}	None
100-D-48:3	Shallow	2×10^{-4}	Cesium-137 (2×10^{-4})	1.4×10^{-4}	Strontium-90 (1.2×10^{-4})
100-D-48:4	Shallow	7×10^{-5}	None	4.0×10^{-5}	None
100-D-49:2	Shallow	3×10^{-5}	None	1.3×10^{-5}	None
100-D-49:4	Shallow	1×10^{-4}	None	5.9×10^{-5}	None
100-D-52	Shallow	4×10^{-6}	None	1.3×10^{-5}	None

Table 6-3. Summary of Residential Scenario Risk Assessment Results for Radionuclides

Waste Site Name	RI/FS Decision Unit	Radionuclides and Radiological Risk Drivers			
		RCBRA Radiological Risk	RCBRA Radiological Risk Driver	RI/FS Radiological Risk	RI/FS Radiological Risk Driver
116-D-1A	Shallow	7×10^{-5}	None	4.2×10^{-5}	None
116-D-2	Shallow	7×10^{-6}	None	3.0×10^{-6}	None
116-D-4	Shallow	4×10^{-6}	None	2.6×10^{-6}	None
116-D-7	Shallow	5×10^{-5}	None	2.0×10^{-5}	None
116-D-9	Shallow	2×10^{-5}	None	2.8×10^{-5}	None
116-DR-1&2	Shallow	5×10^{-5}	None	2.3×10^{-5}	None
116-DR-4	Shallow	1×10^{-5}	None	2.8×10^{-6}	None
116-DR-6	Shallow	4×10^{-5}	None	3.5×10^{-5}	None
116-DR-7	Shallow	2×10^{-5}	None	9.8×10^{-6}	None
116-DR-9	Shallow	4×10^{-4}	Cesium-137 (4×10^{-4})	2.6×10^{-4}	Cesium-137 (2.3×10^{-4})
118-DR-2:2	Shallow	1×10^{-4}	None	2.3×10^{-4}	Technetium-99 (1.6×10^{-4})
122-DR-1:2	Shallow	6×10^{-6}	None	3.1×10^{-6}	None
1607-D2:1	Shallow	6×10^{-6}	None	1.9×10^{-6}	None
1607-D2:3	Shallow	1×10^{-5}	None	7.1×10^{-6}	None
1607-D2:4	Shallow	4×10^{-6}	None	2.2×10^{-6}	None
1607-D4	Shallow_Focused	--	--	--	--
100-H Source OU Waste Sites					
100-H-5	Shallow	1×10^{-5}	None	3.8×10^{-6}	None
100-H-17	Shallow	5×10^{-5}	None	5.5×10^{-5}	None
100-H-21	Shallow	6×10^{-5}	None	5.6×10^{-5}	None
100-H-24	Shallow	--	--	--	--
116-H-1	Shallow	1×10^{-4}	None	6.1×10^{-5}	None
116-H-7	Shallow	5×10^{-5}	None	2.3×10^{-5}	None
1607-H2	Shallow	9×10^{-6}	None	3.7×10^{-6}	None
	Shallow_Focused	Not Evaluated	Not Evaluated	--	--
1607-H4	Shallow	--	--	4.6×10^{-6}	None

Table 6-3. Summary of Residential Scenario Risk Assessment Results for Radionuclides

		Radionuclides and Radiological Risk Drivers			
Waste Site Name	**RI/FS Decision Unit**	**RCBRA Radiological Risk**	**RCBRA Radiological Risk Driver**	**RI/FS Radiological Risk**	**RI/FS Radiological Risk Driver**

Notes:

Radionuclide drivers shown have an associated risk greater than 1×10^{-4}.

The risk value for the individual drivers is shown in parentheses after the name of the risk driver chemical.

Risks are based on reasonable maximum EPCs.

Source s:

RCBRA data: *River Corridor Baseline Risk Assessment, Volume II: Human Health Risk Assessment* (DOE/RL-2007-21), Part 2, Table 2-10.

RI/FS data: Appendix G, Table G-17 (100-D) and G-36 (100-H).

-- = Radionuclide COPCs were not identified.

6.1.2 RI/FS Risk Assessment (Unrestricted Land Use)

As shown in Tables 6-1 to 6-3, the risk assessment results are similar between the RCBRA (DOE/RL-2007-21, Volume II) and the RI/FS for the residential scenario. Differences in results are generally attributed to the COPC identification process, the method used to calculate EPCs, and the PRG value used for comparison. The soil risk assessment provided in this chapter supplements the RCBRA (DOE/RL-2007-21, Volume II) because there are several key differences between the scope and purpose of the RCBRA (DOE/RL-2007-21, Volume II) and the scope and purpose of the RI/FS. Differences between the RCBRA (DOE/RL-2007-21, Volume II) and the RI/FS in the methodologies used for assessing residual risks are described in Table 6-4; these include methods for COPC identification, selection of exposure factors used for the remedial action goals and PRGs, inclusion of all decision units associated with a waste site, and inclusion of analytical data from focused sampling designs. As a result of these differences, the soil risk assessment provided in the RI/FS more directly supports the evaluation of remedial alternatives in the FS. Table 6-4 also provides the methods used for preparing the closeout documentation.

RAOs are narrative statements that define the extent to which waste sites require cleanup to protect human health and the environment. Further, PRGs (also used as risk-based screening levels [RBSLs]) are the numeric values that would be expected to achieve the RAOs presented in Chapter 8. The 100-D/H OU PRGs are developed in the RCBRA (DOE/RL-2007-21) and presented in this chapter.

For the 100-D/H Source OU, the results of the soil risk assessment presented in this chapter will be used to determine whether additional remedial action may be necessary for waste sites where remediation has been completed, and whether the goals and objectives of the interim action RODs have been met, as demonstrated by verification sampling and analysis. It is important to note that another objective of the soil risk assessment is to determine and affirm a basis for action. Although the RI/FS risk assessment and the RCBRA (DOE/RL-2007-21) focus on the protection of human health and the environment at waste sites that have been remediated, there are significant potential risks at unremediated sites that require continuation of cleanup actions. The risk-based screening evaluation for the residential scenario in this chapter provides information necessary to resolve the following questions and provides information needed to support final remedial decisions that will ensure protection of human health and the environment:

- Are residual conditions for cleanup actions completed under the interim action RODs protective of human health and the environment based on comparison to RBSLs calculated in accordance with current EPA guidance?
- Are there waste sites with a no action or interim closed out reclassification status that should be carried into the FS?
- What uncertainties are associated with the risk results that require a risk management decision?

Waste sites evaluated in the River Corridor were Interim Closed using remedial action goals related to direct contact soil exposure by human receptors. These remedial action goals are reported in the 100 Area RDR/RAWP (DOE/RL-96-17). The remedial action goals for radionuclides have not been revised since originally published in 1996. Remedial action goals in the 100 Area of the River Corridor (for direct contact) were based on a Rural Residential exposure scenario. The interim action ROD residential scenario for radionuclides is a Rural Residential scenario that, in addition to direct contact, includes food chain exposure pathways (for example, ingestion of homegrown produce, beef, and milk). Since the 100 Area RDR/RAWP (DOE/RL-96-17) was originally published, EPA has published a change in policy associated with health protectiveness thresholds as well as updates in guidance associated with several exposure assumptions. PRGs presented in this chapter incorporate exposure assumptions that were updated to reflect current EPA guidance (see Table 6-4).

The interim action ROD residential scenario for chemicals is based on the MTCA Method B Soil Cleanup Levels ("Unrestricted Land Use Soil Cleanup Standards" [WAC 173-340-740]). The MTCA Method B levels are based solely on incidental soil ingestion and do not address the food exposure pathways that were included for the radionuclide Rural Residential scenario. The MTCA (WAC 173-340) Method B cleanup levels developed in this chapter are similar to those published in the most recent version of the 100 Area RDR/RAWP (DOE/RL-96-17) with the exception of those chemicals with remedial action goals based on the inhalation exposure route.

In addition to performing the risk-based screening evaluation, another purpose for updating the PRGs is to determine whether the remedial action goals developed and reported in the 100 Area RDR/RAWP (DOE/RL-96-17) are protective when compared to current guidance. Chapter 8 provides a summary of the remedial action goals reported in the 100 Area RDR/RAWP in addition to the PRGs presented in this chapter.

6.1.3 RI/FS Soil Risk Assessment (Reasonably Anticipated Future Land Use Scenarios)

The RCBRA (DOE/RL-2007-21) evaluated risks for a range of exposure scenarios that represent a range of upper bound and reasonably anticipated receptors and activities. When soil cleanup levels were initially established for the River Corridor, the TPA signatories agreed that it was appropriate to protect for a range of potential exposures in the future so that interim cleanup actions did not limit future use of the Site. The Resident Monument Worker and the Casual Recreational User scenario represent reasonably anticipated future land use.

PRGs are presented in this section for both scenarios (resident Monument worker and the casual recreational user), as well as residential PRGs, for use in the risk-based screening evaluation. CVP and RSVP data are compared to these PRGs. When the total risk for a waste site is less than 1×10^{-4} for radionuclides based on the residential scenario or 1×10^{-5} for chemicals based on MTCA Method B Soil Cleanup Levels ("Unrestricted Land Use Soil Cleanup Standards" [WAC 173-340-740]), then protection of the resident Monument worker and casual recreational user is also achieved. The results of these comparisons can be used in risk management decisions (presented in Section 6.2.5.5) and show that the total risk calculated for the Residential and Resident Monument Worker scenarios are essentially

identical. The Residential PRGs are slightly lower than the Resident Monument Worker PRGs because the Residential exposure scenario includes the food chain pathways.

The Resident Monument Worker scenario was evaluated in the RCBRA (DOE/RL-2007-21) as an occupational scenario and was applied on a local and broad area scale. In the RCBRA (DOE/RL-2007-21), the resident Monument worker spent a fraction of the day on the waste site at his residence (local area) and spent a fraction of the same day in a region as large as an individual ROD decision area (comparable to an OU) and potentially as large as the entire River Corridor conducting work activities (broad area). To incorporate the use of this exposure scenario in the RI/FS process, the scenario was modified to assume that the broad area concentration was equal to the RME broad area upland surface soil concentration reported in the RCBRA (DOE/RL-2007-21). The PRG value represents the concentration of soil the resident Monument worker is exposed to on the waste site (local area).

With the exception of the soil ingestion rate and exposure time, the exposure assumptions used to calculate the resident Monument worker local area PRGs are the same as those that would be used to provide an RME for the residential exposure scenario. With the exception of the soil ingestion rate, the exposure assumptions used to calculate the resident Monument worker broad area risks are the same as those that would be used to provide an RME for the Industrial Worker exposure scenario defined in *Risk Assessment Guidance for Superfund Volume I: Human Health Evaluation Manual* *Supplemental Guidance* *"Standard Default Exposure Factors" Interim Final* (OSWER Directive 9285.6-03). Some exposure assumptions were updated based on recent EPA guidance or modified to conform to recommended EPA methodology for calculation of PRGs. Exposure assumptions that were updated based on recent guidance include inhalation rates, PEFs, and the external gamma shielding factor. The exposure assumptions that were modified to correlate to standard PRGs equations include soil ingestion rates, indoor time fraction, onsite exposure time, and use of decay factors. These updates and modifications allow a PRG to be developed to confirm that cleanup actions at the waste site are protective of reasonably anticipated future land uses. Table 6-5 summarizes the modifications made to the Resident Monument Worker exposure scenario for use as a PRG.

The Casual User scenario was evaluated in the RCBRA (DOE/RL-2007-21, Volume II) as a recreational scenario and was applied on a broad area scale; the casual user spent time enjoying recreational activities (broad area) only in a region as large as an individual ROD OU, and potentially as large as the entire River Corridor. Similar to the Resident Monument Worker, this exposure scenario was used to calculate forward risk estimates. To incorporate the use of this exposure scenario in the RI/FS process, the scenario was modified to develop a PRG assuming that all of the casual user time was spent on the waste site (local area). This assumption is the only modification made to this exposure scenario; no changes were made to the exposure assumptions used to calculate PRG values. This modification allows a conservative PRG to be developed to confirm that cleanup actions at the waste site are protective.

Some exposure assumptions for the Casual Recreational User scenario were updated based on recent EPA guidance or modified to conform to recommended EPA methodology for calculation of PRGs. Exposure assumptions that were updated based on recent guidance include the incidental soil ingestion rate, the inhalation rate, PEF, time spent on the local area and broad area scale, external gamma shielding factor, and radiological decay. The exposure assumptions that were modified to correlate to standard PRGs equations include soil ingestion rates and use of decay factors. These updates and modifications allow a PRG to be developed to confirm that cleanup actions at the waste site are protective of human health and the environment. Table 6-6 summarizes the modifications made to the Casual Recreational User exposure scenario for use as a PRG.

Table 6-4. Comparison of Methods and Assumptions Used for the Residential Scenario

Parameter	Method Used in Closeout Documentation	Method Used in RCBRA	Method Used in RI/FS	Overall Effect on RI/FS
Basis of PRG Values for Radioisotopes and Chemicals				
Residential PRG value for radioisotopes	Radionuclide cancer risk is evaluated using the interim action ROD Rural Residential exposure scenario reported in *Remedial Design Report/Remedial Action Work Plan for the 100 Area* DOE/RL-96-17. Radionuclide remedial action goals were calculated based on a dose threshold of 15 millirems per year (mrem/yr).	Radionuclide cancer risk is evaluated using the interim action ROD Rural Residential exposure scenario reported in *Remedial Design Report/Remedial Action Work Plan for the 100 Area* DOE/RL-96-17. Radionuclide remedial action goals were calculated based on a dose threshold of 15 mrem/yr. In the RCBRA, these remedial action goals were converted to RBSLs based on a risk threshold of 1×10^{-4} (pg 2-41 of the RCBRA). The interim action ROD Rural Residential exposure scenario is considered a Local Area exposure scenario (located on a waste site).	Radionuclide cancer risk is evaluated using the residential exposure scenario. This exposure scenario is similar to the interim action ROD Rural Residential scenario but incorporates updates to reflect recent EPA guidance as identified in the following text.	The residential scenario used in the RI/FS reflects updates in methodology (risk-based versus dose-based threshold) and recent recommendations in exposure assumptions. RBSL/PRG values differ slightly between the remedial action goals reported in the closeout documentation, RCBRA and the RI/FS for key COPCs (gamma emitters and strontium-90). Risk-based PRG values reported in the RI/FS for gamma emitters and strontium-90 are slightly lower than the remedial action goals reported in the closeout document and in the RCBRA. Risk-based PRG values reported in the RI/FS for some alpha emitters are greater than the remedial action goals reported in the closeout document and in the RCBRA.
Updates to EPA guidance for residential PRG	External gamma shielding factor is 0.8 based on *Risk Assessment Guidance for Superfund: Volume I—Human Health Evaluation Manual (Part B, Development of Risk-Based Preliminary Remediation Goals)* (EPA, 1991). Outdoor time fraction is 0.2 (5 hours/day over 350 days/year), which was obtained from *Hanford Guidance for Radiological Cleanup* (WDOH/320-015). Annual dose rate is 15 mrem/year based on "Radiation Site Cleanup Standards" (40 CFR 196).	External gamma shielding factor is 0.7, which is based on the default value recommended in the RESRAD code. Outdoor time fraction is 0.2 (5 hours/day over 350 days/year) from WDOH/320-015. Target cancer risk value is 1×10^{-4} based on the recommendations published in *Radiation Risk Assessment at CERCLA Sites: Q & A* (EPA/540/R/99/006).	External gamma shielding factor is 0.4 from *Soil Screening Guidance for Radionuclides: User's Guide* (EPA/540-R-00-007). Outdoor time fraction is 0.12 (3 hours/day over 350 days/year) published in the Exposure Factors Handbook (EPA/600/P-95/002Fb). Target cancer risk value is 1×10^{-4} based on the recommendations published in *Radiation Risk Assessment at CERCLA Sites: Q & A* (EPA/540/R/99/006).	The gamma-shielding factor was revised from 0.7 to 0.4. The current assumption accounts for a 60 percent reduction in external exposure due to shielding from structures rather than a 30 percent reduction. The use of the updated assumption results in slightly less exposure and a less conservative PRG value (higher). The outdoor time fraction was revised from 0.2 to 0.12. The current assumption assumes the resident spends 3 hours/day outside rather than 5 hours/day. Use of the updated assumption results in less exposure and a less conservative PRG value (higher). The protective threshold value was updated from a dose-based value to a risk-based value. The overall outcome is that updated PRG values used in the RI/FS are slightly lower for beta- and gamma-emitting radioisotopes and higher for alpha-emitting radioisotopes.
MTCA Method B soil cleanup levels for unrestricted land use	Separate MTCA Method B levels were calculated for incidental soil ingestion and inhalation.	MTCA Method B levels are based solely on incidental soil ingestion.	Separate MTCA Method B levels were calculated for incidental soil ingestion and inhalation.	Chemicals that only report toxicity values for the inhalation exposure route are not included in the RCBRA evaluation (beryllium, cadmium, cobalt, Cr(VI), and nickel). Remedial action goals are reported for chemicals that only report toxicity values for the inhalation exposure route (beryllium, cadmium, cobalt, Cr(VI), and nickel). The RI/FS separately reports cancer risks and noncancer HIs for both incidental soil ingestion and inhalation exposure routes.
MTCA Method B inhalation cleanup levels for unrestricted land use	*Remedial Design Report/Remedial Action Work Plan for the 100 Area,* DOE/RL-96-17 reports remedial action goals for beryllium, cadmium, Cr(VI) based on the inhalation exposure pathway, based on WAC 173-340-750 (3), 1996. A PEF value of 1.0×10^{7} m^3/kg was used to convert air concentrations to soil concentrations. The PEF value of 1.0×10^{7} m^3/kg is based on the default mass loading factor in RESRAD. This is roughly two orders of magnitude smaller than EPA's default PEF of 1.4×10^{9} m^3/kg.	MTCA Method B inhalation cleanup levels were not evaluated in the RCBRA.	MTCA Method B inhalation cleanup levels were calculated for the inhalation exposure route. A PEF value of 7.3×10^{10} m^3/kg is used to convert air concentrations to soil concentrations. This PEF uses meteorological data from Boise, Idaho, and Hanford Site-specific annual wind speed. The PEF of 7.3×10^{10} m^3/kg is within a factor of two of EPA's default PEF of 1.4×10^{9} m^3/kg published in the Supplemental Guidance for Developing Soil Screening Levels for Superfund Sites (OSWER 9355.4-24).	Inhalation pathway cleanup levels that use a PEF value based on the default mass loading factor in RESRAD are lower values (more conservative) than those cleanup levels that are based on EPA methodology.

Table 6-4. Comparison of Methods and Assumptions Used for the Residential Scenario

Parameter	Method Used in Closeout Documentation	Method Used in RCBRA	Method Used in RI/FS	Overall Effect on RI/FS
Data Analysis				
Waste site decision units and analysis time frame	The floor and sidewalls of an excavated waste site are divided into one or more decision units. A sample design is developed for the decision unit. Sampling requirements for each decision unit are described in *100 Area Remedial Action Sampling and Analysis Plan* (DOE/RL-96-22).	For Local Area exposure scenarios (including the interim action ROD Rural Residential scenario), the RCBRA used only the CVP/RSVP datasets from shallow zone decision units. These datasets are from waste sites that were excavated/remediated through calendar year 2005. The shallow zone decision unit is typically represented by soils from the excavation floor if at or above 4.6 m (15 ft) and any sidewalls from grade level (0 m [0 ft]) to a depth of 4.6 m (15 ft).	The RI/FS used CVP/RSVP datasets from all decision units associated with an excavated/remediated waste site through July 2011. In addition to the shallow zone decision unit, the RI/FS evaluates the risk contribution from soils associated with the overburden, staging pile footprint area, and the deep zone decision units.	The RI/FS soil risk assessment is intended to supplement the analysis in Chapter 2 of the RCBRA. The RI/FS soil risk assessment results will be used to identify waste sites that warrant evaluation of remedial alternatives in the FS. The RI/FS soil risk assessment can also be used to disposition the waste site from an interim status to final closure status when risk thresholds are not exceeded.
Statistical and focused sample designs	The layout and orientation of sampling designs are based on the size, shape, and depth of the Site. The datasets from the sample design are used to confirm attainment of RAOs.	When both focused and statistical samples exist for an analyte at a waste site, only the statistical samples were used to calculate the representative concentrations. An uncertainty analysis was performed to evaluate the effect the selection of focused and/or statistical samples has on the risk assessment results. Representative concentrations for these waste sites are also calculated using the combined focused and statistical samples. The statistical representative concentrations were compared to the combined focused and statistical samples and shown in Table C3-11 in Appendix C, Section C-3, "Representative Concentrations."	The approach used to evaluate the dataset for each sample design is the same as that used for the closeout documentation.	Evaluation of only the data from statistical sample designs when focused sample data are also collected has the potential to understate risk. Frequently focused sample results are collected in areas with the highest potential for contamination to be present. The RI/FS soil risk assessment results will be used to identify waste sites that warrant evaluation of remedial alternatives in the FS. The RI/FS soil risk assessment can also be used to disposition the waste site from an interim status to final closure status when risk thresholds are not exceeded.
COPC Identification	Closeout documentation did not incorporate a COPC identification step. All detected analytes with remedial action goals reported in *Remedial Design Report/Remedial Action Work Plan for the 100 Area* (DOE/RL-96-17) were evaluated in the closeout documentation. It should be noted that the remedial action goals listed in DOE/RL-96-17 do not include analytes that meet exclusion criteria.	The COPC refinement process includes a number of complementary steps and criteria, including a preselected list of contaminants that were excluded and a list that were included, as determined and agreed upon among the Tri-Parties. Additional selection steps include evaluation of all data according to detection status, statistical comparisons of Hanford Site data to background and reference site data, and an analyte-specific evaluation. Each interim action ROD area has a separate list of COPCs.	COPC identification uses the exclusion criteria defined in Section 6.2.1.3 of this Chapter. The inclusion list and other refinement steps used in the RCBRA were not incorporated into the RI/FS. When a COPC was detected at least once in a waste site decision unit (and it did not meet the exclusion criteria) it was carried into all risk calculations.	COPC refinement in RCBRA often included analytes that were not detected at the waste site. The inclusion of analytes that were not detected at a waste site decision unit results in an overstatement of risk. The method used to identify COPCs in the RI/FS is similar to the method used in the closeout documentation. The RI/FS and closeout documentation did not evaluate analytes that met exclusion criteria. Although two different COPC identification processes were used in the RCBRA and the RI/FS, similar risk drivers were identified in the risk characterization step of the analysis as shown in Tables 6-1 to 6-3.
Exposure point concentrations	The primary statistical calculation to support closeout documentation was the 95 percent UCL on the arithmetic mean of the data for waste sites closed using a statistical/random sampling design. Statistical calculations were performed in compliance with *Statistical Guidance for Ecology Site Manager* (Ecology Publication 92-54). This guidance addresses two kinds of data distributions: normal, and lognormal. This guidance also implements the substitution method where a proxy value of one-half the detection limit is assigned to nondetected results. For small datasets (n<10) a nonparametric distribution was assumed. When a nonradionuclide was detected in fewer than 50 percent of the samples collected and for focused sampling designs, the maximum detected value was used for comparison purposes. For radionuclides, a 95 UCL was always calculated using a nonparametric method based on the "z" statistic.	Representative concentrations pertain to sampled medium, whereas EPCs also include modeled concentrations in other exposure media. In general, the process used in the RCBRA follows EPA guidance as provided in the *ProUCL Version 4.0 User Guide* (EPA/600/R-07/038). The ProUCL software was not used to calculate representative concentrations.	*Calculating Upper Confidence Limits for Exposure Point Concentrations at Hazardous Waste Sites* (OSWER 9285.6-10) is the EPA guidance for UCL calculation and ProUCL 4.00.05 serves as the companion software package for this guidance. ProUCL 4.00.05 contains rigorous parametric and nonparametric (including bootstrap methods) statistical methods that can be used on full datasets without nondetects and on datasets with below detection or nondetect observations. Both ProUCL and *Calculating Upper Confidence Limits for Exposure Point Concentrations at Hazardous Waste Sites* (OSWER 9285.6-10) were used to recalculate the UCLs for the 100-D/H Source OU.	*ProUCL Version 4.0 User Guide* (EPA/600/R-07/038) draws from guidance documented in *Calculating Upper Confidence Limits for Exposure Point Concentrations at Hazardous Waste Sites* (OSWER 9285.6-10). Methodologies for calculating 95 UCLs are similar between the RCBRA and the RI/FS. The methodology used in the closeout documentation addresses only two data distributions for the 95 UCL calculation and implemented the substitution of one-half the detection limit value for nondetected results.

Table 6-4. Comparison of Methods and Assumptions Used for the Residential Scenario

Parameter	Method Used in Closeout Documentation	Method Used in RCBRA	Method Used in RI/FS	Overall Effect on RI/FS
Waste Site Specific Information				
Exclusion of focused sample design data from waste site 1607-H2	Both focused and statistical sample design datasets were evaluated in the closeout documentation.	Focused sample design datasets were not evaluated. Only statistical sample design datasets were evaluated.	Both focused and statistical sample design datasets were evaluated in the RI/FS.	Exclusion of some datasets has the potential to understate risks in the RCBRA.
Exclusion of shallow zone waste site 1607-H4	Both focused and statistical sample design datasets were evaluated in the closeout documentation.	COPCs on OU-specific list were not detected.	All analytes detected at 1607-H4 were identified as COPCs and carried forward into the risk characterization step of the analysis	Exclusion of some datasets has the potential to understate risks in the RCBRA.
Chemical Risk for 100-D-49:4	Aroclor-1254 was included in the closeout documentation for this waste site.	Analyte-specific evaluation for 100-D/H COPCs excluded Aroclor-1254.	COPC selection process for RI/FS included all detected analytes, which includes Aroclor-1254 for 100-D-49:4.	May have the potential to overstate risks.
Chemical Risk for 1607-D2:3	bis(2-ethylhexyl)phthalate was included in the closeout documentation for this waste site.	Analyte-specific evaluation for 100-D/H COPCs did not include bis(2-ethylhexyl)phthalate.	COPC selection process for RI/FS included all detected analytes, which includes bis(2-ethylhexyl)phthalate for 1607-D2:3	May have the potential to overstate risks.

PEF = particulate emission factor

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6.1.4 Other Residential Land Use Scenarios in RCBRA

The RCBRA (DOE/RL-2007-21, Volume II) also evaluated three residential scenarios that describe exposures related to a rural land-use pattern that involves home-produced foods. The Subsistence Farmer scenario envisions a substantial quantity of home-produced foods, but not a diet composed solely of such foods. The two Native American Resident scenarios, however, envision a complete subsistence lifestyle where all foods are grown at the home or (in the case of fish) caught in the Columbia River. Residential receptors are assumed to spend effectively all of their time in the area around a residence located on a remediated waste site to assign all soil-related exposures protectively to that site.

Table 6-5. Summary of Differences in Exposure Assumptions for the Resident Monument Worker between the RCBRA and RI/FS Risk Assessment

Parameter	RCBRA Resident Monument Worker	RI/FS Resident Monument Worker
Soil ingestion rate	A soil ingestion rate of 100 mg/day is assumed for this receptor. The soil ingestion rate is apportioned to the local area and the broad area based on the amount of time the receptor spends at each area. The RCBRA (DOE/RL-2007-21) allocated 52.2 mg/day to the residential portion (local area) of this scenario and 25 mg/day to the occupational portion (broad area) of this scenario.	A soil ingestion rate of 100 mg/day is assumed for this receptor. The RI/FS allocated 76.2 mg/day to residential portion (local area) of this scenario and 23.8 mg/day to the occupational portion (broad area) of this scenario for a total of 100 mg/day.
Inhalation rate	The RCBRA (DOE/RL-2007-21) assumed an inhalation rate of 0.63 m^3/hour based on an inhalation rate of 15 m^3/day.	The RI/FS assumed an inhalation rate of 0.83 m^3/hour based on an inhalation rate of 20 m^3/day.
Particulate Emission Factor	The RCBRA (DOE/RL-2007-21) used a PEF of 1.08×10^8 m^3/kg for the local area and a PEF of 4.3×10^8 m^3/kg for the broad area.	The RI/FS used the EPA default PEF of 7.3×10^{10} m^3/kg for the local area and a PEF of 2.6×10^{10} m^3/kg for the broad area.
Time spent on the local area and broad area scale	The RCBRA (DOE/RL-2007-21) assumed an exposure time of 13 hours/day spent at the residence (local area), 8 hours spent onsite at work (broad area), and 3 hours offsite (neither local nor broad area) for a total of 24 hours/day.	The RI/FS assumed that an exposure time of 16 hours/day was spent at the residence (local area) and 8 hours/day onsite at work (broad area) for a total of 24 hours/day.
Indoor and outdoor exposure time	The RCBRA (DOE/RL-2007-21) assumed that the resident spent 13 hours/day indoors, 8 hours/day outdoors, and 3 hours/day offsite.	The RI/FS assumed that the resident spent 13 hours/day indoors and 3 hours/day outdoors (local area) and the worker spent 8 hours/day outdoors (broad area).
Gamma shielding factor	The RCBRA (DOE/RL-2007-21) used an external gamma shielding factor of 0.7.	The RI/FS used an external gamma shielding factor of 0.4 based on current guidance.
Radiological decay factors	Decay of radioisotopes over the exposure duration was not accounted for.	Decay of radioisotopes over the exposure duration was incorporated.

Table 6-6. Summary of Differences in Exposure Assumptions for the Casual Recreational User between the RCBRA and RI/FS Risk Assessment

Parameter	RCBRA Casual User	RI/FS Casual Recreational User
Soil ingestion rate	A soil ingestion rate of 100 mg/day for an adult and 200 mg/day for a child were assumed for this receptor. Soil ingestion at the waste site was assumed proportional to the fraction of waking hours spent at the Site.	A soil ingestion rate of 100 mg/day for an adult and 200 mg/day for a child were assumed for this receptor. All soil ingestion was assumed to occur at the waste site.
Inhalation rate	The RCBRA (DOE/RL-2007-21) assumed an inhalation rate of 1 m^3/hour for an adult and 1 m^3/hour for a child based on EPA recommended short-term exposure values for light activity.	The RI/FS assumed an inhalation rate of 0.83 m^3/hour for an adult, based on an inhalation rate of 20 m^3/day, and 0.417 m^3/hour for a child, based on an inhalation rate of 10 m^3/day (*Risk Assessment Guidance for Superfund: Volume I—Human Health Evaluation Manual (Part B, Development of Risk-Based Preliminary Remediation Goals): Interim* [EPA/540/R-92/003]).
Particulate Emission Factor	The RCBRA (DOE/RL-2007-21) used a PEF of 4.3×10^8 m^3/kg for the broad area.	The RI/FS used the EPA default PEF of 7.3×10^{10} m^3/kg (*Supplemental Guidance for Developing Soil Screening Levels for Superfund Sites* [OSWER 9355.4-24]).
Time spent on the local area and the broad area scale	The RCBRA (DOE/RL-2007-21) assumed an exposure time of 6 hours/day is spent onsite, all in the broad area.	The RI/FS assumed an exposure time of 6 hours/day is spent onsite, all in the local area.
Gamma shielding factor	The RCBRA (DOE/RL-2007-21) did not apply a gamma-shielding factor (all exposure is assumed to occur outdoors).	The RI/FS did not apply a gamma-shielding factor (all exposure is assumed to be occurring outdoors).
Radiological decay factors	Decay of radioisotopes over the exposure duration was not accounted for.	Decay of radioisotopes over the exposure duration was incorporated.

PRGs were not calculated in the RCBRA (DOE/RL-2007-21, Volume II) for these additional residential scenarios. Direct contact and food chain exposure associated with radiological contaminants for unrestricted land use are represented by the Rural Residential scenario described in Section 6.1.2.

DOE, through discussions with the Tribes ("Contract No. DE-AC06-96RL13200 – Native American Scenarios in Remedial Investigation and Feasibility Study Risk Assessments and Assuming Responsibility and Configuration Control of the Soil Inventory Model" [0702827]), has agreed to include quantitative analysis of Native American scenarios in risk assessments supporting RI/FS documents. The two scenarios considered are provided by the CTUIR and the Yakama Nation. The RCBRA (DOE/RL-2007-21, Volume II) presents the risks and hazards calculated for both Native American exposure scenarios from direct contact, external gamma exposure, inhalation, and food chain pathways from remediated waste sites. The groundwater risk assessment presented in Section 6.3 presents the results of both Native American scenarios for potentially complete exposure pathways associated with groundwater. The groundwater risk assessment presents the risks and hazards calculated for groundwater used as a source of drinking water and as a source of steam for sweat lodge (see Section 6.3.8.5.1). The results from the RCBRA for remediated waste sites and the results from the groundwater risk assessment can be summed to obtain a cumulative estimate of risk for all exposure pathways included in the CTUIR and Yakama Nation exposure scenarios. These tribal scenarios have been evaluated and presented in Hanford Site risk assessments to assist interested parties in providing input on remedial alternatives (*Feasibility Study Report for the 200-ZP-1 Groundwater Operable Unit* [DOE/RL-2007-28]), and have not been used for development of PRGs as part of alternatives analyses in the FS.

The results of the local area risk assessment for the residential scenarios indicate that present-day RME cancer risk is frequently greater (11 of 28 remediated sites at the 100-D Source OU and seven of seven remediated sites at the 100-H Source OU for the Subsistence Farmer scenario) than 1×10^{-4} and that RME chemical HI frequently (4 of 28 remediated sites at the 100-D Source OU and seven of seven remediated sites at the 100-H Source OU for the Subsistence Farmer scenario) exceeds the threshold HI of 1. A summary of risks and noncancer hazards associated with the Subsistence Farmer scenario is provided in Table 6-7. Present-day RME cancer risks greater than 1×10^{-4} for the Subsistence Farmer exposure scenario are almost entirely related to one of three factors:

- External irradiation from short-lived radionuclides including europium-152, cesium-137, and cobalt-60
- Exposure to arsenic from ingestion of garden produce
- Exposure to the short-lived radionuclide strontium-90 from ingestion of produce and livestock products

By the year 2075, the RCBRA (DOE/RL-2007-21) calculated the Subsistence Farmer RME cancer risks above 1×10^{-4} are related overwhelmingly to arsenic exposure from produce ingestion. Because the CTUIR Resident and Yakama Resident scenarios use very high (subsistence level) Hanford Site-raised food ingestion rates, strontium-90 still plays a significant role in food-related exposures at year 2075 for these scenarios. By the year 2150, however, CTUIR Resident and Yakama Resident cancer risks above 1×10^{-4} are dominated by arsenic exposure from ingestion of garden produce.

- The RCBRA (DOE/RL-2007-21, Volume II) Subsistence Farmer cancer risk and chemical HI results were frequently above threshold criteria. The Subsistence Farmer reported cancer risk and chemical HI results above threshold criteria whereas the closeout documentation reported that residual chemical concentrations met or were below threshold criteria. The two major differences were identified between the risk assessment methods used in the RCBRA (DOE/RL-2007-21, Volume II) and the basis of the interim remedial action goals. These differences were as follows: Residential interim action remedial action goals for chemicals are MTCA Method B ("Unrestricted Land Use Soil Cleanup Standards" [WAC 173-340-740]), which is an RME scenario based on incidental soil ingestion and does not address the food exposure pathways historically evaluated for radionuclides.
- The interim action remedial action goal for arsenic is 20 mg/kg, which is an "adjusted" value established by the State of Washington to address a range of natural background levels MTCA ("Tables" [WAC 173-340-900]).

These differences largely explain why some waste sites that were remediated to meet the interim action RAGs still appear to present high levels of residual risk under the Subsistence Farmer scenario:

One of the primary uncertainties for site-specific results relates to modeled exposure concentrations in foods, particularly garden produce. Further discussion of the potential biases in modeled food chain exposures is provided in the RCBRA (DOE/RL-2007-21). As discussed in Section 5.9.4.2 of the RCBRA (DOE/RL-2007-21), in the case of the noncancer HI results for produce ingestion of mercury, uranium, and copper, a large conservative bias is anticipated because a linear plant uptake model was applied to soil concentrations that are far above naturally occurring levels. In the case of arsenic, produce ingestion provides the largest contribution to total cancer risk, even though the range of site soil concentrations is relatively small. Uncertainty in produce concentrations is attributable to intrinsic variability related to soil conditions, plant species and tissue type, harvest time, and other variables. A review of recommended plant-soil ratios from a number of sources, as described in Section 5.9.2.4 of the RCBRA

(DOE/RL-2007-21, Volume II), shows that the range of soil to plant transfer ratios for arsenic (from 0.006 to 1.125) is approximately a factor of 200. The value of 0.53 used in the HHRA, from the RESRAD computer code that has been used to perform dose assessment at the Hanford Site and other DOE facilities, is near the upper end of this range. The high-end values for plant-soil concentrations, many of which were used in the RCBRA (DOE/RL-2007-21, Volume II) to assess exposure through food pathways, may result in a scenario that provides exposures to nonradionuclide contaminants higher than an RME. PRGs identified in this document for nonradiological analytes are based on MTCA procedures, which do not include food chain pathways.

Table 6-7. Summary of Excess Lifetime Cancer Risks and Noncancer Hazards for the Subsistence Farmer Scenario Reported in the RCBRA

Waste Site Name	Present Day Total Cancer Risk	COPC	Pathway	Present Day Hazard Index	COPC	Pathway
100-D Source OU Waste Sites						
100-D-4	6×10^{-5}	None	--	0.12	None	--
100-D-12	--[a]	None	--	--[a]	None	--
100-D-20	2×10^{-4}	Cesium-137	External Irradiation	0.25	None	--
		Europium-152	External Irradiation			
100-D-21	2×10^{-5}	None	--	--[a]	None	--
100-D-22	--[a]	None	--	--[a]	None	--
100-D-48:1	9×10^{-5}	None	--	0.14	None	--
100-D-48:2	2×10^{-4}	Cesium-137	External Irradiation	0.15	None	--
		Europium-152	External Irradiation			
		Cobalt-60	External Irradiation			
100-D-48:3	4×10^{-4}	Cesium-137	External Irradiation	0.17	None	--
		Strontium-90	Milk Ingestion			
		Cesium-137	Milk Ingestion			
100-D-48:4	2×10^{-4}	Cesium-137	External Irradiation	0.21	None	--
		Europium-152	External Irradiation			
		Strontium-90	Milk Ingestion			
100-D-49:2	6×10^{-5}	None	--	0.11	None	--

Table 6-7. Summary of Excess Lifetime Cancer Risks and Noncancer Hazards for the Subsistence Farmer Scenario Reported in the RCBRA

Waste Site Name	Present Day Total Cancer Risk	COPC	Pathway	Present Day Hazard Index	COPC	Pathway
100-D-49:4	2×10^{-4}	Europium-152	External Irradiation	0.56	None	--
100-D-52	7×10^{-6}	None	--	0.13	None	--
116-D-1A	2×10^{-4}	Cesium-137	External Irradiation	0.16	None	--
		Europium-152	External Irradiation			
		Strontium-90	Milk Ingestion			
		Strontium-90	Produce Ingestion			
116-D-2	2×10^{-5}	None	--	0.14	None	--
116-D-4	8×10^{-6}	None	--	0.13	None	--
116-D-7	1×10^{-4}	None	--	0.18	None	--
116-D-9	7×10^{-5}	None	--	0.16	None	--
116-DR-1&2	1×10^{-4}	None	--	0.14	None	--
116-DR-4	3×10^{-5}	None	--	0.15	None	--
116-DR-6	1×10^{-4}	None	--	0.13	None	--
116-DR-7	4×10^{-5}	None	--	0.15	None	--
116-DR-9	7×10^{-4}	Cesium-137	External Irradiation	0.054	None	--
		Cesium-137	Milk Ingestion			
118-DR-2:2	8×10^{-4}	Arsenic	Produce Ingestion	2.9	Arsenic	Produce Ingestion
122-DR-1:2	5×10^{-4}	Arsenic	Produce Ingestion	2.6	Arsenic	Produce Ingestion
1607-D2:1	2×10^{-4}	Arsenic	Produce Ingestion	1.2	Arsenic	Produce Ingestion
1607-D2:3	3×10^{-5}	None	--	0.63	None	--
1607-D2:4	9×10^{-6}	None	--	0.13	None	--
1607-D4	2×10^{-4}	Arsenic	Produce Ingestion	1.4	Arsenic	Produce Ingestion

Table 6-7. Summary of Excess Lifetime Cancer Risks and Noncancer Hazards for the Subsistence Farmer Scenario Reported in the RCBRA

Waste Site Name	Present Day Total Cancer Risk	COPC	Pathway	Present Day Hazard Index	COPC	Pathway
100-H Source OU Waste Sites						
100-H-5	7×10^{-4}	Arsenic	Produce Ingestion	4.7	Arsenic	Produce Ingestion
					Mercury	Beef Ingestion
100-H-17	7×10^{-4}	Arsenic	Produce Ingestion	3	Arsenic	Produce Ingestion
100-H-21	2×10^{-3}	Arsenic	Produce Ingestion	12	Arsenic	Produce Ingestion
100-H-24	7×10^{-4}	Arsenic	Produce Ingestion	3.6	Arsenic	Produce Ingestion
116-H-1	1×10^{-3}	Arsenic	Produce Ingestion	5.9	Arsenic	Produce Ingestion
116-H-7	1×10^{-3}	Arsenic	Produce Ingestion	4.7	Arsenic	Produce Ingestion
1607-H2	1×10^{-3}	Arsenic	Produce Ingestion	69	Mercury	Beef Ingestion
					Mercury	Produce Ingestion

Notes:

Risk drivers shown have an associated risk greater than 1×10^{-4}.

No COCs were identified.Source:

RCBRA data: *River Corridor Baseline Risk Assessment, Volume II: Human Health Risk Assessment* (DOE/RL-2007-21), Part 2 (Tables 5-102 and 5-104).

6.2 Soil Risk Assessment

Section 6.1.1 summarized the evaluation of residual risks performed in the RCBRA (DOE/RL-2007-21, Volume II) for waste sites cleaned up under the interim action ROD. Section 6.1.2 described how elements of the RCBRA (DOE/RL-2007-21, Volume II) were updated to reflect current guidance, risk assessment methodologies, and toxicity information to support the FS. Section 6.2 provides the updated soil risk assessment, which implements the updates described in Section 6.1.2.

The following paragraphs describe the 100-D/H Source OU soil risk assessment followed:

- Identify all waste sites with a "no action" or "interim closed out" reclassification status.
- Obtain verification sampling and analysis data for all "no action" and "interim closed out" waste sites that have been remediated through July 2011.4

[4] These are waste sites for which interim action cleanups had been completed under interim action RODS and for which the CVPs were completed through July 2011.

- Compute EPCs for each detected analyte measured at a waste site, using the EPA's ProUCL version 4.00.05 software (ProUCL Version 4.00.05 User Guide (Draft) [EPA/600/R-07/038]).
- Compare EPCs to direct contact RBSLs selected to represent baseline conditions and reasonably anticipated future Hanford Site use.
- Calculate cancer risk and noncancer hazards for each detected analyte.
- Compare cancer risks and noncancer hazards to acceptable state and federal target risk and noncancer thresholds.
- Determine whether the "no action" or "interim closed out" waste site should be carried forward into the FS to select remedial alternatives.

This soil risk assessment follows the risk assessment guide (EPA/540/1-89/002). The following sections describe the four-step process. Because this soil risk assessment is intended to complement the analysis performed in the RCBRA (DOE/RL-2007-21, Volume II), where applicable, a brief description is provided to describe the similarities in approach.

6.2.1 Data Analysis

This section describes the sources of data used in the risk assessment (Section 6.2.1.1), describes the data quality assessment (DQA) and data validation process (Section 6.2.1.2), and identifies COPCs in vadose zone material that are accessible for human exposures (Section 6.2.1.3). During the course of this risk assessment, analytes were evaluated to identify COPCs and prioritize those estimated to pose an unacceptable risk and warrant evaluation in the FS.

6.2.1.1 Sources of Analytical Data Used in Risk Assessment

This soil risk assessment includes vadose zone material samples for remediated waste sites with a "no action" or "interim closed out" reclassification status collected within the 100-DR-1, 100-DR-2, 100-HR-1, and 100-HR-2 Source OUs. Waste sites where remediation and verification sampling and analysis were assessed by the end of July 2011 are included in the soil risk assessment.

All samples were collected and analyzed in accordance with the requirements stated in *100 Area Remedial Action Sampling and Analysis Plan* (hereinafter called 100 Area SAP [DOE/RL-96-22]). Data collected under the 100 Area SAP (DOE/RL-96-22) are used to meet the purpose and objectives of the 100 Area RDR/RAWP (DOE/RL-96-17), which describes the design and the implementation of the remedial action processes required by the following:

- *Interim Action Record of Decision for the 100-BC-1, 100-BC-2, 100-DR-1, 100-DR-2, 100-FR-1, 100-FR-2, 100-HR-1, 100-HR-2, 100-KR-1, 100-KR-2, 100-IU-2, 100-IU-6, and 200-CW-3 Operable Units, Hanford Site, Benton County, Washington (100 Area Remaining Sites)* (EPA/ROD/R10-99/039)
- *Amendment to the Interim Remedial Action Record of Decision for the 100-BC-1, 100-DR-1, and 100-HR-1 Operable Units, Hanford Site, Benton County, Washington* (EPA/AMD/R10-97/044)
- *Declaration of the Record of Decision for the 100-BC-1, 100-BC-2, 100-DR-1, 100-DR-2, 100-FR-2, 100-HR-2, and 100-KR-2 Operable Units, Hanford Site (100 Area Burial Grounds), Benton County, Washington* (EPA/541/R-00/121)

Remediation of waste sites in the 100-D/H Source OUs began in 1996. The constituents are identified for each waste site based on process knowledge, site history, and site-specific discussions with the lead regulatory agency. Constituents analyzed include the COPCs for the waste site; as a result different constituents are analyzed at each waste site. Therefore, only constituents reported at each waste site are

included in risk calculations. Analytical results for each waste site are included in the associated closeout documentation, which is listed in Appendix C, Table C-1, of the 100-D/H Work Plan (DOE/RL-2008-46-ADD1). Both the 100-D/H Work Plan (DOE/RL-2008-46-ADD1) and the 100 Area RDR/RAWP (DOE/RL-96-17) were reviewed and approved by the Tri-Parties.

Ninety-five 100-D Source OU waste sites have verification sampling data and are included in this soil risk assessment. Twenty-eight of these 100-D Source OU waste sites were evaluated in the RCBRA (DOE/RL-2007-21, Volume II). An additional thirteen 100-D Source OU sites, referred to as associated waste sites, have been remediated, but are included in another waste site's sampling and closeout documentation. Forty-seven 100-H Source OU waste sites have verification sampling and analysis data and are included in this soil risk assessment. Eight of these 36 100-H Source OU waste sites were evaluated in the RCBRA (DOE/RL-2007-21, Volume II). An additional 10 100-H Source OU sites, referred to as consolidated sites, have been remediated but are included in another waste site's sampling and closeout documentation. A summary of the waste sites, associated decision unit(s), and reclassification status for 100-D Source OU and 100-H Source OU is provided in Tables G-1 and G-2, respectively. Waste site decision units are defined in Section 6.2.2.2. The waste sites listed in Tables G-1 and G-2 are a subset of the waste sites that were listed in Appendix C, Table C-1, of the 100-D/H Work Plan (DOE/RL-2008-46-ADD1). Summaries of the remediated waste sites and consolidated waste sites for the 100-D and 100-H Source OUs is provided in Tables 6-8 and 6-9, respectively.

Table 6-8. Summary of Remediated Waste Sites and Associated Waste Sites in the 100-D Source OUs

Waste Site Totals	Remediated Waste Site	Associated Waste Sites[a]
100-D Source OU		
100-DR-1		
	100-D-1	--
	100-D-18	
	100-D-19	
	100-D-2	
	100-D-20	
	100-D-21	
	100-D-22	
	100-D-24	
	100-D-29	
	100-D-3	
	100-D-31:1[b]	
	100-D-31:10	
	100-D-31:2[b]	
	100-D-31:3	
	100-D-31:4	
	100-D-31:5	

Table 6-8. Summary of Remediated Waste Sites and Associated Waste Sites in the 100-D Source OUs

Waste Site Totals	Remediated Waste Site	Associated Waste Sites[a]	
	100-D-31:6		
	100-D-31:7		
	100-D-31:8		
	100-D-31:9		
	100-D-32		
	100-D-4		
	100-D-42[c]		
	100-D-45[c]		
	100-D-48:1	100-D-49:1	UPR-100-D-4
	100-D-48:2	UPR-100-D-2	UPR-100-D-3
	100-D-48:3	100-D-5	100-D-6
	100-D-48:4	--	
	100-D-49:2		
	100-D-49:3		
	100-D-49:4		
	100-D-50:5		
	100-D-52		
	100-D-56:1		
	100-D-56:2		
	100-D-61		
	100-D-7		
	100-D-70		
	100-D-74		
	100-D-75:3		
	100-D-80:1		
	100-D-82		
	100-D-83:4		
	100-D-84:1		
	100-D-85:1		
	100-D-87		
	100-D-88		

Table 6-8. Summary of Remediated Waste Sites and Associated Waste Sites in the 100-D Source OUs

Waste Site Totals	Remediated Waste Site	Associated Waste Sites[a]
	100-D-9	
	100-D-90	
	116-D-10	
	116-D-1A	116-D-1B
	116-D-2	--
	116-D-4	
	116-D-5	
	116-D-6	
	116-D-7	
	116-D-9	
	116-DR-1 & 2	
	116-DR-5	
	116-DR-9	100-D-25
	118-D-6:4	--
	120-D-2	
	126-D-2	
	128-D-2	
	130-D-1	
	132-D-1	
	1607-D2:1	
	1607-D2:2	
	1607-D2:3	
	1607-D2:4	
	1607-D4	
	1607-D5	
	628-3	
	UPR-100-D-5	
100-DR-1 Source Operable Unit Totals	**74**	**8**
100-DR-2		
	100-D-12	--

Table 6-8. Summary of Remediated Waste Sites and Associated Waste Sites in the 100-D Source OUs

Waste Site Totals	Remediated Waste Site	Associated Waste Sites[a]
	100-D-13	
	100-D-15	
	100-D-28:1	
	100-D-43[c]	
	100-D-47	
	100-D-94	
	116-D-8	
	116-DR-10	
	116-DR-4	
	116-DR-6	
	116-DR-7	
	116-DR-8	
	118-D-1	
	118-D-4	
	118-D-5	
	118-DR-1	
	--[d]	100-D-46
	118-DR-2:2	--
	122-DR-1:2	100-D-23 100-D-53 100-D-54 100-D-64
	1607-D1	--
	600-30	
100-DR-2 Source Operable Unit Totals	**21**	**5**
100-DR-1 and 100-DR-2	**95**	**13**
100-D Area Total	**108**	

a. Associated waste sites are those sites for which remediation and closeout documentation were consolidated with another remediated waste site.

b. Sample results are consolidated for the 100-D-31:1 and 100-D-31:2 waste sites.

c. Sample results are consolidated for the 100-D-42, 100-D-43, and 100-D-45 waste sites.

d. Consolidated with 116-D-1A (100-DR-1) remediated waste site.

Table 6-9. Summary of Remediated Waste Sites and Consolidated Waste Sites in the 100-H Source OUs

Waste Site Totals	Remediated Waste Site	Associated Waste Sitesa	
100-HR-1 Source OU			
	100-H-17	100-H-30	116-H-2
	100-H-21	100-H-1	100-H-22
	100-H-24	--	
	100-H-28:1		
	100-H-28:6		
	100-H-3		
	100-H-35		
	100-H-4		
	100-H-41		
	100-H-45		
	100-H-49:2		
	100-H-5		
	100-H-50		
	100-H-51:4		
	100-H-51:5		
	100-H-53		
	100-H-7		
	100-H-8		
	116-H-1		
	116-H-3		
	116-H-5		
	116-H-7		
	116-H-9		
	100-H-11[b]	100-H-10[b]	
	100-H-12[b]	100-H-9[b]	
	100-H-14[b]	100-H-13[b]	
	118-H-6:3[b]	118-H-6:2[b]	
	118-H-6:6[b]	100-H-31[b]	
	118-H-6:5	--	
	118-H-6:4		

Table 6-9. Summary of Remediated Waste Sites and Consolidated Waste Sites in the 100-H Source OUs

Waste Site Totals	Remediated Waste Site	Associated Waste Sitesa
	1607-H2	
	1607-H3	
	1607-H4	
100-HR-1 Source Operable Unit Totals	**33**	**9**
100-HR-2 Source OU		
	-- [c]	100-H-2
	100-H-37	--
	100-H-40	
	118-H-1:1	
	118-H-1:2	
	118-H-2	
	118-H-3	
	118-H-4	
	118-H-5	
	128-H-1	
	128-H-2	
	128-H-3	
	1607-H1	
	600-151	
	600-152	
100-HR-2 Source Operable Unit Totals	**14**	**1**
100-HR-1 and 100-HR-2 Totals	**47**	**10**
100-H Area Total	**57**	

a. Associated waste sites are those sites for which remediation and closeout documentation were consolidated with another remediated waste site.

b. Sample results are consolidated for the 118-H-6:2,118-H-6:3,118-H-6:6,100-H-9, 100-H-10,100-H-11,100-H-12,100-H-13,100-H-14, and 100-H-31 waste sites

c. Consolidated with 100-H-17 (100-DR-1)

The following sources of analytical data were used in the soil risk assessment:

- All verification sampling and analysis data reside in the Hanford Environmental Information System (HEIS) database.

- All closeout verification data used in this soil risk assessment are included in Appendix D of this report.

6.2.1.2 Data Quality Assessment and Data Validation

A DQA is performed and reported in each closeout documentation report. The DQA compares the verification sampling approach and resulting analytical data with the sampling and data quality requirements specified by the project objectives and performance specifications. The DQA determines if the data are of the right type, quality, and quantity to support Hanford Site cleanup verification decisions within specified error tolerances. The DQA also determines if the analytical data are found acceptable for decision-making purposes and if the sample design was sufficient for the purpose of cleanup Hanford Site verification. The cleanup verification sample analytical data and detailed DQA are summarized in the appendices associated with the CVPs. The results of each DQA are incorporated by reference and no further DQA was performed as part of this risk assessment.

All the analytical data are evaluated, and a portion validated for compliance with QA project plan requirements as documented in the 100 Area RDR/RAWP (DOE/RL-96-17). Data evaluation is performed to determine whether the laboratory carried out all steps required by the SAP and the laboratory contract governing the conduct of analysis and reporting of the data. This evaluation also examines the available laboratory data to determine whether an analyte is present or absent in a sample and the degree of overall uncertainty associated with that determination. Data validation was done in accordance with validation procedures as part of data evaluation.

6.2.1.3 Identification of COPCs

For the purposes of this soil risk assessment, a COPC is defined as an analyte suspected of being associated with site-related activities that represent a potential threat to human health and the environment, and whose data are of sufficient quality for use in a quantitative BRA.

All analytes detected at least once in a waste site decision unit for the waste sites included in the soil risk assessment are identified as COPCs. As described in Section 6.2.2.2, the floor and sidewalls of an excavated waste site are divided into one or more decision units (for example, shallow zone, deep zone, overburden, or staging pile area). Verification sampling and analysis data are collected according to sample design requirements for the type of decision unit. For the purpose of this soil risk assessment, an "exposure area" and a "decision unit" are operationally defined as being the same. Verification sampling and analysis data are subsequently grouped to calculate EPCs.

The contributions from naturally occurring metals and anthropogenic radioisotopes are discussed in the risk characterization section in accordance with CERCLA Soil Background Comparisons Guidance (EPA 540-R-01-003). The risk characterization will discuss elevated background concentrations and their contribution to Hanford Site risks as well as naturally occurring elements that are not CERCLA hazardous substances, pollutants, and contaminants but exceed the RBSLs.

The RCBRA (DOE/RL-2007-21) identifies a subset of analytes that is excluded from consideration as COPCs by agreement among the Tri-Parties based on relevant Hanford Site data. The following exclusion lists employed in the RCBRA (DOE/RL-2007-21) were also applied to the waste site verification data during the data reduction steps described in Section 6.2.2.2 and listed in Appendix G, Tables G-5 and G-6:

Radionuclides with a half-life of less than 3 years: Radionuclides with half-lives less than 3 years would not be present as a result of historical Hanford Site operations because of radioactive decay that would have occurred since operations ceased.

Essential nutrients: Essential nutrients that are present at relatively low concentrations and are toxic only at high concentrations need not be considered in a quantitative risk assessment.

Water quality or soil physical property measurements: These analytes were measured to obtain information on water quality or soil properties to understand potential confounding factors for bioassays conducted for soil, sediment, or water or to interpret their influence on the toxicity of COPCs (for example, grain size for soils, water hardness for metal effects).

Background radionuclides (potassium-40, radium-226, radium-228, thorium-228, thorium-230, and thorium-232): These background radionuclides were identified by consensus of Tri-Party managers as not directly related to Hanford Site operations or processes.

The RCBRA (DOE/RL-2007-21) includes two additional steps to identify COPCs that the soil risk assessment did not apply:

Evaluate analytes that are commonly reported in waste site cleanup verification reports based on frequency of detection. Inclusion list analytes were not consistently reported in the CVP and RSVP data; therefore, this step was not implemented.

Evaluate remaining analytes as candidate COPCs, based on comparisons to Hanford Site background, reference areas, and an "analyte-specific" evaluation.

As a result of not applying these last two steps used in the RCBRA (DOE/RL-2007-21) to identify COPCs, more analytes are identified as COPCs in this soil risk assessment than were identified in the RCBRA (DOE/RL-2007-21, Volume II). Identifying all detected analytes (except those on the exclusion list) as COPCs is a more streamlined approach that is consistent with CERCLA Soil Background Comparisons Guidance (EPA 540-R-01-003).

6.2.2 Exposure Point Concentrations

Calculating Upper Confidence Limits for Exposure Point Concentrations at Hazardous Waste Sites (hereinafter called Calculating UCL for EPCs [OSWER 9285.6-10]) states that, "an exposure point concentration (EPC) is a conservative estimate of the average chemical concentration in an exposure medium." *Supplemental Guidance to RAGS: Calculating the Concentration Term* (hereinafter called RAGS Supplemental Guidance [OSWER Publication 9285.7-081]) states that, "because of the uncertainty associated with estimating the true average concentration at a site, the 95 percent UCL of the arithmetic mean should be used for this variable." Use of the 95 percent UCL of the arithmetic mean yields risk estimates that correspond to an RME. Instances where a value different from a UCL is used as the EPC are clearly stated in this risk assessment. Reasons and/or justifications are also provided.

Calculating UCL for EPCs (OSWER 9285.6-10) further states that, "The EPC is determined for each individual exposure unit within a site. An exposure unit is the area throughout which a receptor moves and encounters an environmental medium for the duration of the exposure. Unless there is site-specific evidence to the contrary, an individual receptor is assumed to be equally exposed to media within all portions of the exposure unit over the time frame of the risk assessment." For this soil risk assessment, the "exposure unit" and the "decision unit" are operationally defined as being the same. As previously described, one or more decision units are included within a waste site, including shallow vadose zone material (0 to 4.6 m [0 to 15 ft] bgs), deep vadose zone material (greater than 4.6 m [15 ft] bgs), overburden material, and staging pile area footprint material.

Statistical Guidance for Ecology Site Managers (Ecology Publication 92-54) has been used to calculate EPCs for all closeout documentation to date. *Statistical Guidance for Ecology Site Managers* (Ecology Publication 92-54) was published in 1992, and this guidance has been superseded by Calculating

UCL for EPCs (OSWER 9285.6-10), which was published in 2002. For this soil risk assessment, UCLs were recalculated for all waste sites and decision units to incorporate the updated guidance in Calculating UCL for EPCs (OSWER 9285.6-10). UCLs that incorporate updated guidance use more rigorous statistical methods to estimate exposure concentrations and eliminate the use of the simple substitution method for nondetects (where a proxy value of one-half the detection limit is assigned to all nondetected results). Calculating UCL for EPCs (OSWER 9285.6-10) notes that because of the complicated formulas used to compute UCLs, there is no general rule about which substitution rule will yield an appropriate UCL. The uncertainty associated with the substitution method increases and its appropriateness decreases as the detection limit becomes larger and as the number of nondetects in the dataset increases.

The following sections describe the statistical methodology used for closeout documentation (Section 6.2.2.1) and the statistical methodology used for this soil risk assessment (Section 6.2.2.2). Although both evaluations used the same dataset, the differences in statistical methodologies may result in differences in the EPC values between the closeout documentation and this risk assessment for the same COPCs in a waste site decision unit.

6.2.2.1 Statistical Evaluation Methodology Used for Closeout Documentation

For waste sites closed using a statistical/random sampling design, the primary statistical calculation to support cleanup verification was the 95 percent UCL on the arithmetic mean of the data. Statistical calculations were performed in compliance with *Statistical Guidance for Ecology Site Managers* (Ecology Publication 92-54). This guidance addresses two kinds of data distributions: normal, and lognormal. For normal data, the guidance recommends a UCL on the mean based on the Student's t-statistic. For lognormal data, the guidance recommends the Land method using the H-statistic. This guidance also implements the substitution method where a proxy value of one-half the detection limit is assigned to nondetected results.

Small datasets (n<10) were evaluated in accordance with Section 5.2.1.4 of *Statistical Guidance for Ecology Site Managers* (Ecology Publication 92-54) and a nonparametric distribution was assumed. When a nonradionuclide was detected in fewer than 50 percent of the samples collected and for focused sampling designs, the maximum detected value was used for comparison purposes.

6.2.2.2 Statistical Evaluation Methodology Used for the Soil Risk Assessment

Calculating UCL for EPCs (OSWER 9285.6-10) is the EPA guidance for UCL calculation and ProUCL 4.00.05 serves as the companion software package for this guidance. ProUCL 4.00.05 contains rigorous parametric and nonparametric (including bootstrap methods) statistical methods that can be used on full datasets without nondetects and on datasets with nondetect observations. Both ProUCL and Calculating UCL for EPCs (OSWER 9285.6-10) were used to recalculate the UCLs for the 100-D/H Source OU.

To ensure that waste sites and decision units are grouped correctly and UCLs are accurately recalculated, all waste sites, decision unit groupings, and sample numbers were individually verified against the original closeout documentation. *Waste Site Evaluation Process for the 100-DR-1, 100-DR-2, 100-HR-1, and 100-HR-2 Source Operable Units* (ECF-100DR1-11-0003), which is provided in Appendix G, documents the process used to confirm a complete list of waste sites with a reclassification status of "interim closed out" or "no action" through July 2011. Verification of sample numbers associated with each waste site was confirmed, along with the decision unit grouping with which the sample is associated. This list of samples is used to verify that the sampling results are complete. The analytical data that have undergone this review process become the final dataset used to calculate the UCLs and associated summary statistics used in this risk assessment. Tables G-3 and G-4 (Appendix G) list the sample numbers associated with each waste site decision unit, along with the date the sample was collected, the type of sample design used, and the Washington State plane coordinates of the sample location.

6.2.2.2.1 *Waste Site Decision Units*

Verification sampling and analysis data that are associated with the samples listed in Tables G-3 and G-4 (Appendix G) are from several different decision units within a waste site, including shallow vadose zone material, deep vadose zone material, overburden material, and staging pile area footprint material. The following describes the basis of each decision unit and briefly describes the sample designs used.

The floor and sidewalls of an excavated waste site are divided into one or more decision units. A sample design is developed for the decision unit. Sample design requirements for each decision unit are described in the 100 Area SAP (DOE/RL-96-22). In practice, the shallow zone decision unit is typically represented by material from the excavation floor if at or above 4.6 m (15 ft) and any sidewalls from grade level (0 m) to a depth of 4.6 m (0 to 15 ft). The deep zone decision unit is represented by material from the excavation floor (if below 4.6 m [15 ft]) and by any sidewall materials below 4.6 m (15 ft). As needed, decision subunits and an associated sampling design are also established for suspect clean overburden stockpiles (that is, to verify suitability for backfill material) and the footprint of the staging pile area. The layout and orientation of the sampling designs are based on the size, shape, and depth of the site. Sampling of a waste site decision unit to confirm attainment of RAOs was performed according to one of three types of sampling designs: focused sampling design, random or statistical sampling, or a combination of both.

The decision unit naming convention is summarized in Table 6-10.

Table 6-10. Summary and Definition of Decision Unit Types

Decision Unit Name	Depth	Sampling Design Description
Shallow	0 to 4.6 m (0 to 15 ft) bgs	Samples collected using a statistical sampling design
Deep	Greater than 4.6 m (15 ft) bgs	
Overburden	Not applicable	
Staging pile area	Not applicable	
Shallow_Focused	0 to 4.6 m (15 ft) bgs	Samples collected using a focused sampling design
Deep_Focused	Greater than 4.6 m (15 ft) bgs	
Staging Pile Area_Focused	Not applicable	

The process used to calculate EPCs for each waste site and decision unit is documented in *Computation of Exposure Point Concentrations for the 100-DR-1, 100-DR-2, 100-HR-1, and 100-HR2 Source Operable Units* (ECF-100DR1-11-0004), which is provided in Appendix G, and the purpose is to document the data processing and reduction steps, methodology, decision logic, assumptions, input files, and output files used to determine the EPCs.

6.2.2.2.2 *Data Processing and Reduction*

This section describes the data processing and reduction steps that are taken prior to the calculation of UCLs. Figures 6-1 and 6-2 show each of the data processing and data reduction steps, and the number of records associated with each step for the 100-D and 100-H Source OUs, respectively.

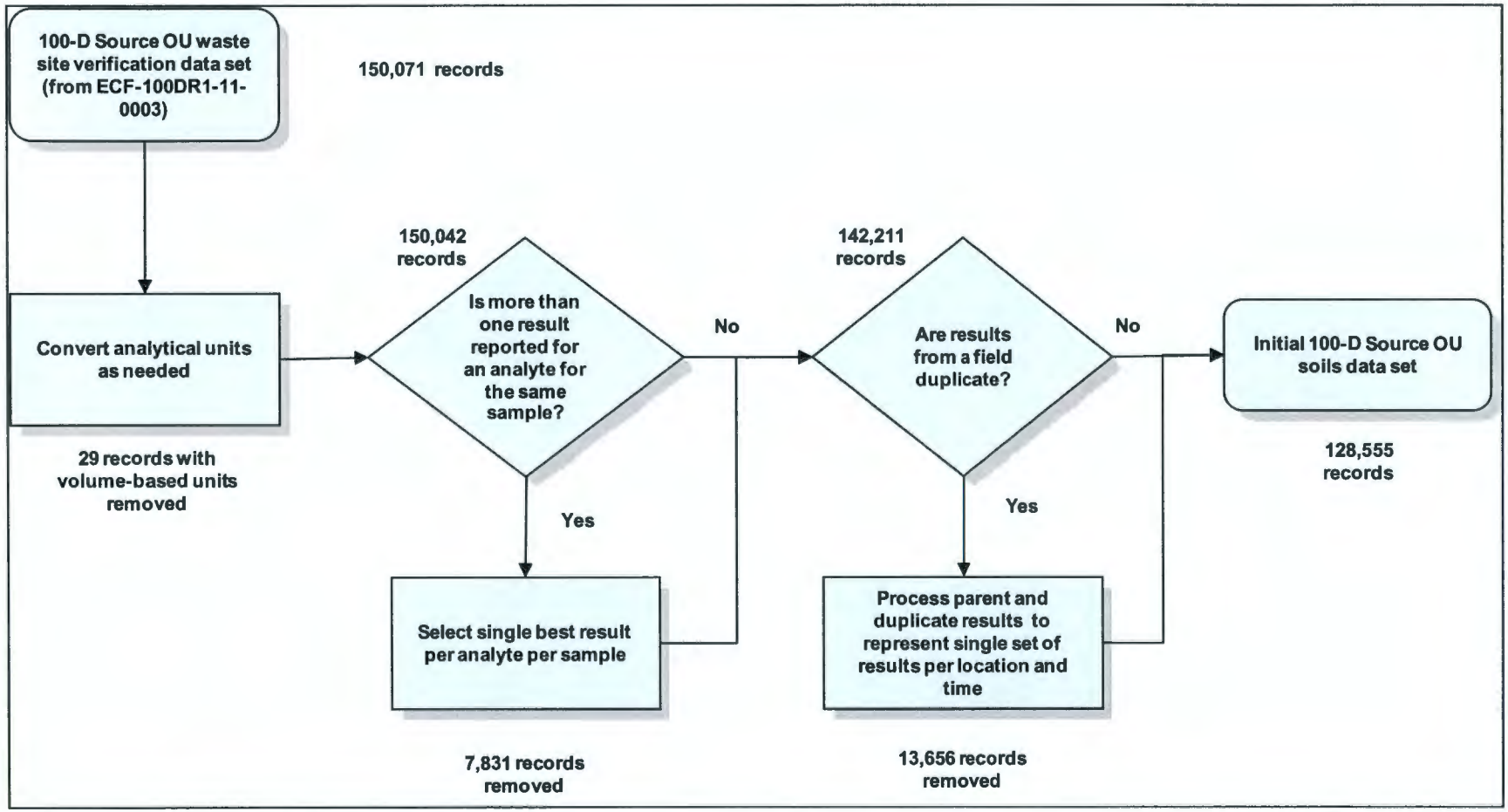

Figure 6-1. Data Processing and Reduction Steps for 100-D Source OU

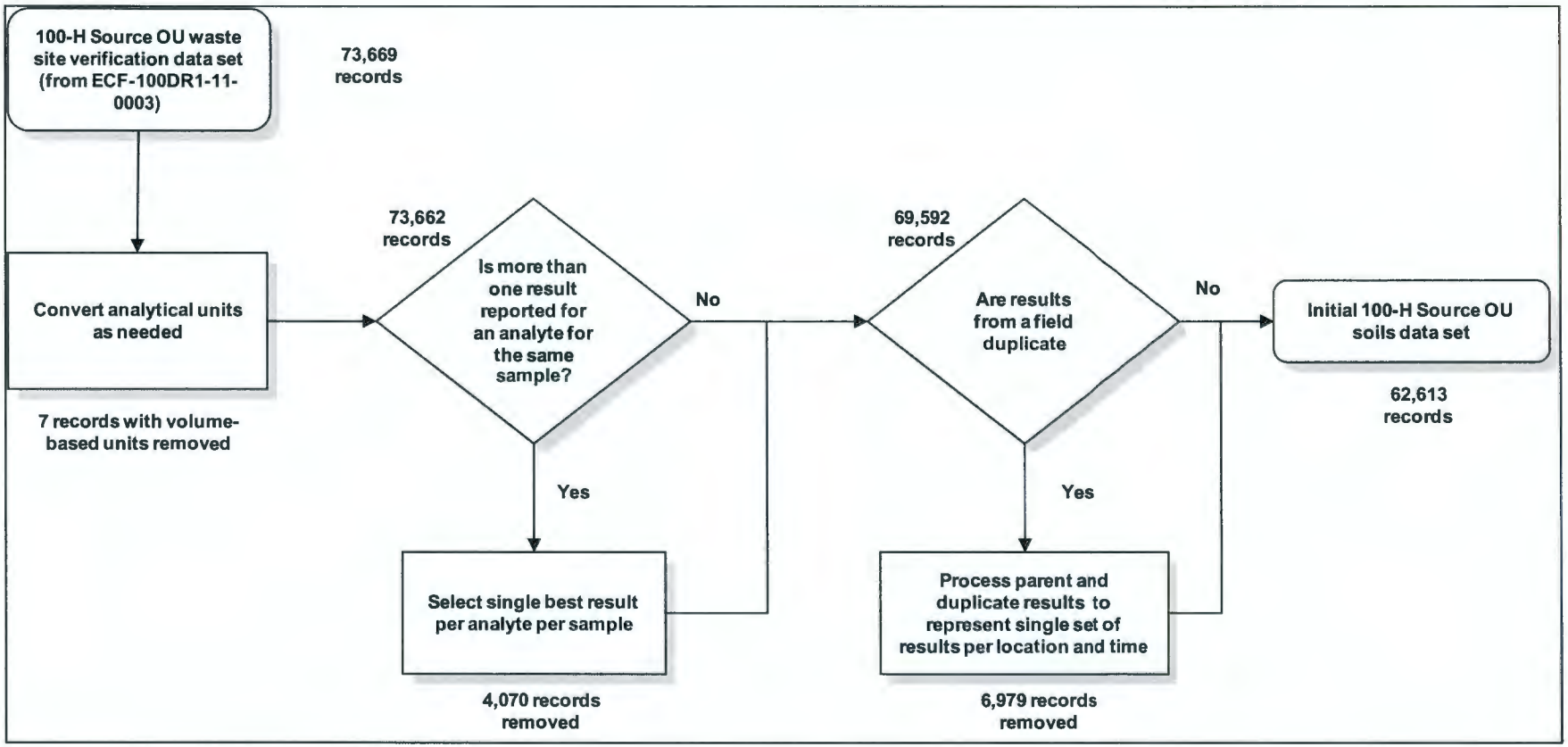

Figure 6-2. Data Processing and Reduction Steps for 100-H Source OU

6.2.2.2.3 *Laboratory and Data Validation Flags*

Analytical data are received from the laboratory with data qualification flags; validation qualifiers are assigned during the data validation process. The following rules are applied to determine how the sample results can be used for calculating UCLs.

- All sample results flagged with a "U" data qualifier or combination of qualifiers that include a "U," such as a "UJ," are considered nondetected concentrations.
- All sample results without a "U" data qualifier are considered detected concentrations, including results without a qualifier or with an "E" or a "J" qualifier.
- Sample results that are rejected and flagged with an "R" validation qualifier are not used for calculating UCLs.

where:

U = Analyzed for but not detected above limiting criteria.

J = Estimated value.

E = Reported value is estimated because of interference (inorganics).

R = Do not use. Further review indicates the result is not valid.

6.2.2.2.4 *Analytes Reported by Multiple Analytical Methods*

Often, a sample is analyzed for an analyte using more than one analytical method, resulting in multiple results for the analyte from the same location and sample date. When analytes are reported by more than one analytical method for a sample, the results are processed to select the method that provides the most reliable results. Considerations for determining data to be retained include method-associated sample size, detection frequency, and detection limits. The most conservative (that is, health-protective) use of these types of data is the goal. Larger sample size, higher detection frequencies, and lower detection limits are given higher priority for method selection.

For example, lead may be analyzed using EPA Method 200.8 (*Methods for the Determination of Metals in Environmental Samples, Supplement I* [EPA-600/R-94/111]) with an EQL of 0.5 mg/kg, or EPA Method 6010 (*Test Methods for Evaluating Solid Waste: Physical/Chemical Methods, Third Edition; Final Update IV-B* [SW-846], hereinafter called SW 846) with an EQL of 5.0 mg/kg. For a sample with lead concentrations reported by both methods, the results reported by EPA Method 200.8 (*Methods for the Determination of Metals in Environmental Samples, Supplement I* [EPA-600/R-94/111]) are chosen over EPA Method 6010 (SW 846 [SW-846]) because of the more sensitive detection limit.

6.2.2.2.5 *Field Duplicate Results*

Field QC samples (field duplicates) are collected in the field and analyzed by the laboratory as unique samples. The parent sample and field QC samples are collected from the same location (that is, sample node) and same date, resulting in more than one sample per location and date. Because multiple sets of analytical results cannot be used to quantify risk (that is, this would result in multiple-counting of a chemical), the results for the same location and date are reduced to a single result for each reported analyte. The most conservative (that is, health-protective) result is the goal. The following criteria are used to reduce multiple sample results for one location and date to a single result:

- If two or more detections are reported, the maximum concentration is used.
- If one detection and one or more nondetections are reported, the detected concentration is used. If two or more nondetections are reported, the lowest detection limit is used.

6.2.2.2.6 *Identify Analytes for 95 Percent UCL Calculation*

After extracting and processing the dataset, it is further reduced to identify a subset of analytes that require computation of a UCL. Analytes that meet any of the exclusion criteria or that were not detected in any of the samples analyzed with the 100-D/H Source OU are not carried forward into the statistical calculations and EPC selection. The analyte identification steps and the number of records associated with each of the steps are presented on Figure 6-3 for the 100-D Source OU and Figure 6-4 for the 100-H Source OU.

6.2.2.2.7 *Apply Exclusion Criteria*

The first step used to identify analytes that require a 95 percent UCL calculation is to apply exclusion criteria. Analytes that do not meet the exclusion criteria are carried forward into the next step of the process. Analytes that meet exclusion criteria are eliminated from further consideration. The following were excluded:

- Radionuclides that have half-lives of less than 3 years and that are not significant daughter products
- Background radionuclides that are not directly related to Hanford Site operations or processes (potassium-40, radium-226, radium-228, thorium-228, thorium-230, and thorium-232)
- Essential nutrients (minerals) (calcium, magnesium, potassium, and sodium)
- Analytes without known toxicity information (for example, delta-BHC, endrin ketone, and sulfate)[5]

A total of 49 analytes for the 100-D Source OU and 37 analytes for the 100-H Source OU meet the exclusion criteria and are listed in Tables G-5 and G-6 (Appendix G), respectively. Sampling dates, minimum and maximum detected concentrations, minimum and maximum MDLs, and the basis for their exclusion are provided in these tables.

6.2.2.2.8 *Identify Nondetected Analytes*

The next step used to identify analytes that require a 95 percent UCL calculation is to identify nondetected analytes. Analytes that are measured at appropriate sampling locations, have adequate detection limits, and that have not been detected in any of the samples from the 100-D Source OUs or (separately) from the 100-H Source OUs are eliminated from further consideration. Any analyte that is detected at least once in the 100-D Source OU or (separately) at least once in the 100-H Source OU is carried forward to the next step of the process.

A total of 75 analytes were not detected in the 100-D Source OU and 83 analytes were not detected in the 100-H Source OU and are listed in Tables G-7 and G-8 (Appendix G), respectively. The tables also provide sampling dates, total number of samples, and minimum and maximum MDLs.

6.2.2.2.9 *95 Percent UCL Calculation Methodology*

A discussion of waste site decision units was provided earlier in this section. It should be noted that calculated UCLs and EPCs selected for shallow zone and deep zone decision units represent verification data collected from the floor and the sidewall of the excavated waste site. As a result, risks are overstated because the UCL and the EPC do not take credit for the existing clean backfill that covers the remediated waste site.

Analytical data for all analytes that have been detected at least once in each waste site decision unit are extracted from the dataset and subsequently formatted so they can be directly imported into ProUCL where 95 percent UCL calculations and summary statistics are performed.

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[5] Note that this exclusion criterion includes the water quality or soil physical property measurements described in Section 6.2.1.3 of this chapter. The sources of analyte-specific toxicity values and the recommended reference hierarchy is provided in Section 6.2.4.2.

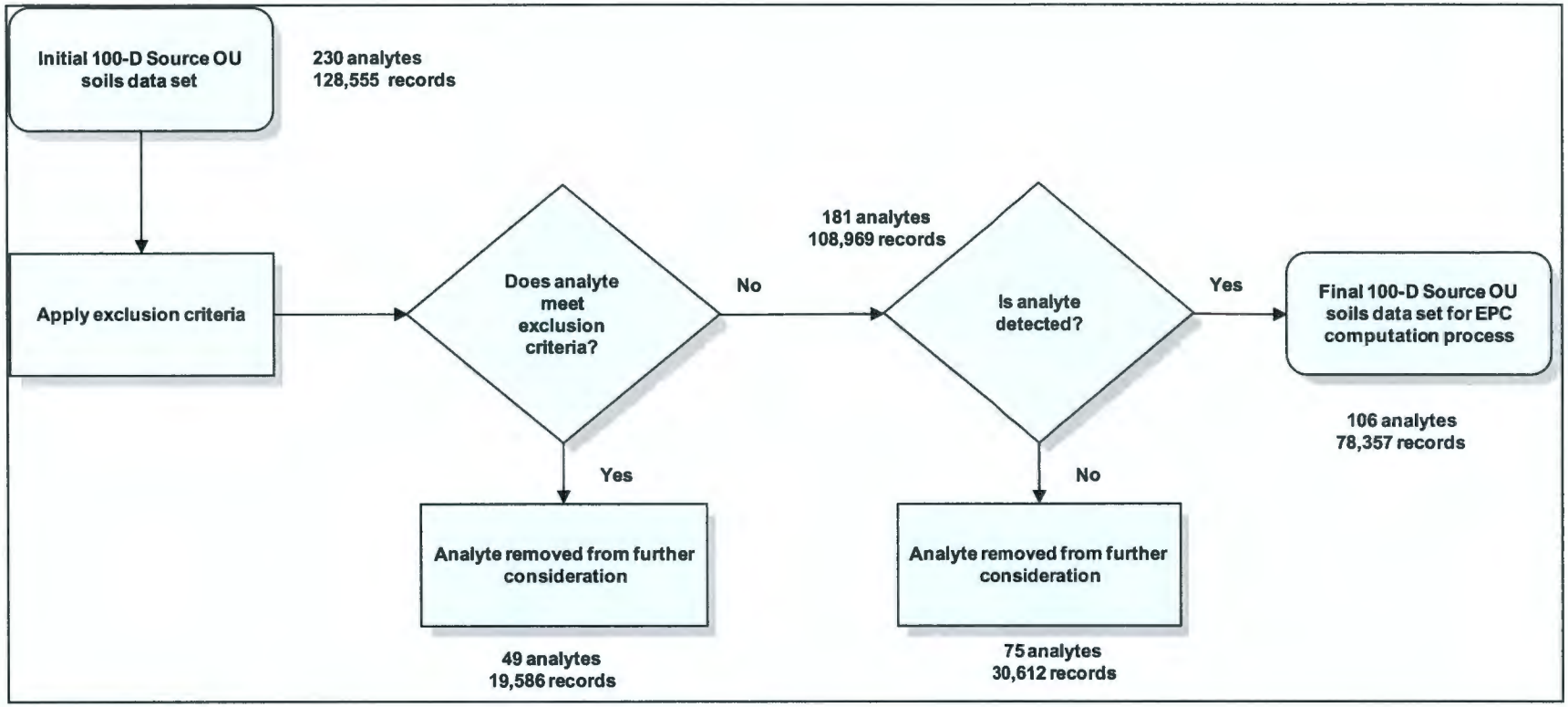

Figure 6-3. Analyte Identification Steps for 100-D Source OU

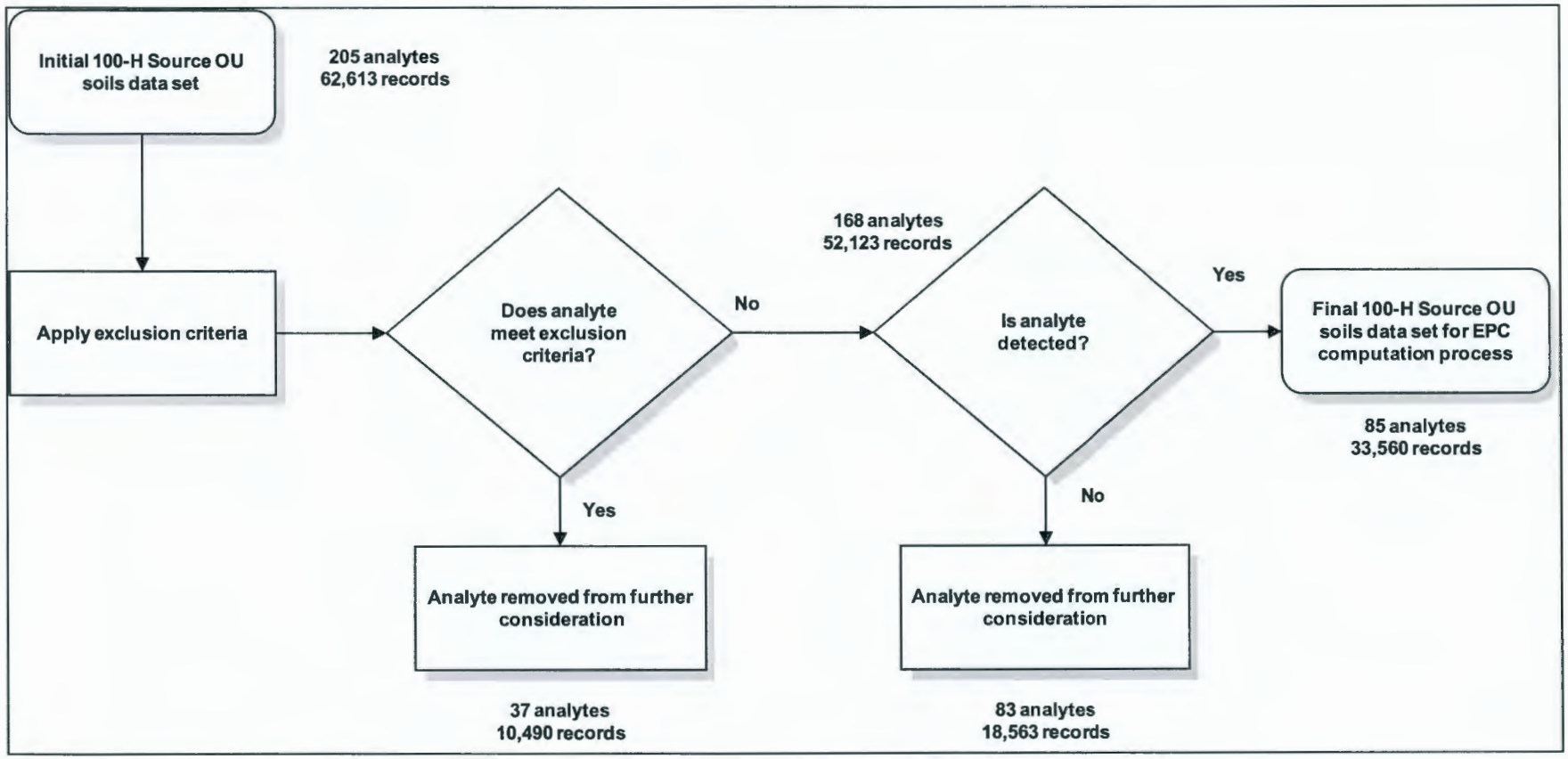

Figure 6-4. Analyte Identification Steps for 100-H Source OU

The following information is obtained from the UCL calculations and summary statistics generated for each waste site decision unit:

- Waste site decision unit name
- Analyte name and Chemical Abstract Service (CAS) Registry number
- Total number of sample results, total number of detects, and total number of nondetects
- Minimum and maximum detection limits for each detected analyte (when available)[6]
- Minimum and maximum detected concentrations for each analyte
- Coefficient of variation (CV) for each analyte
- The UCL value, the UCL basis, and comments and/or warning statements for each analyte

For most datasets, ProUCL recommends a single UCL as the decision statistic. When a single decision statistic is recommended, this UCL is selected. However, ProUCL will recommend more than one decision statistic for some datasets. The most conservative (that is, health-protective) result, that is not greater than the maximum observed concentration, is the goal when selecting the UCL to represent the EPC. When more than one decision statistic is given, the following logic is used to select the UCL:

If more than one UCL is recommended as a decision statistic and the UCLs are less than or equal to the maximum observed concentration, then the highest recommended UCL is selected as the decision statistic.

If more than one UCL is recommended as a decision statistic and the UCLs are greater than the maximum observed concentration, then the maximum observed concentration is selected as the decision statistic.

If more than one UCL is recommended as a decision statistic, at least one is less than the maximum observed concentration, and at least one is greater than the maximum observed concentration, then the maximum observed concentration is selected as the decision statistic. There were 12 analytes in 100-D and 8 analytes in 100-H where more than one UCL was recommended and at least one of the UCLs was greater than the maximum observed concentration.

6.2.2.2.10 *Selection of EPCs*

The following logic was used to select the EPC for each detected analyte in a waste site decision unit:

- For samples collected in accordance with a focused sampling design, the maximum detected concentration is selected as the EPC for every detected analyte.
- For samples collected in accordance with a statistical sampling design, the following logic is applied:
  - If a valid 95 percent UCL can be calculated, then the highest potential 95 percent UCL value (if more than one potential UCL value is recommended) is selected.
  - If the recommended 95 percent UCL is greater than the maximum detected concentration, then the maximum detected concentration is selected.
  - If a valid 95 percent UCL cannot be calculated, then the maximum detected concentration is selected.

[6] Minimum and maximum detection limits are summarized in the ProUCL output only when a valid UCL can be calculated.

Selection of the EPC value using the above decision logic is presented on Figure 6-5. A summary of the EPCs for each detected analyte in a given waste site decision unit is provided in Table G-9 for the 100-D Source OU and Table G-10 for the 100-H Source OU (Appendix G).

6.2.2.2.11 *Use of Maximum Detected Concentrations to Estimate the EPC*

The EPC defaults to the maximum detected concentration when the following conditions are met:

- When samples are collected using a focused sampling design
- When a valid 95 percent UCL cannot be calculated because of a limited number of detections (less than 5)
- When a valid 95 percent UCL is greater than the maximum detected concentration

The sampling plan for a focused decision unit was designed to sample the areas of suspected contamination. The results from this type of sampling design can introduce bias into statistical analyses to estimate means, such as calculations of UCLs. RAGS Supplemental Guidance (OSWER Publication 9285.7-081) states "a value other than the 95 percent UCL can be used, provided the risk assessor can document that high coverage of the true population mean occurs (that is, the value equals or exceeds the true population mean with high probability)." Because the sampling design for these decision units focused on areas of suspected contamination, the conclusion that maximum detected concentration exceeds the true population mean in a focused decision unit can be made with certainty. Additionally, the closeout documentation for the focused decision units used the maximum detected concentration to determine whether the remedial action remedial action goal has been attained (Section 3.6.3 of the 100 Area RDR/RAWP [DOE/RL-96-17]). Because of the potential for statistical bias and to maintain consistency with the 100 Area RDR/RAWP (DOE/RL-96-17) the maximum detected concentration is selected as a conservative estimate of the EPC for the focused decision units.

ProUCL has minimum size requirements to compute UCLs. For datasets of at least five results, a UCL is not calculated when there is only one detected result in the dataset. ProUCL notes that in cases where the number of available detected samples is small (fewer than five), the estimation of the EPC term is decided upon on a site-specific basis. ProUCL generates warning messages regarding the potential deficiencies associated with a small dataset. For small datasets with very few detected values (fewer than five) a valid UCL cannot be calculated. For risk assessment purposes, the maximum concentration is used as a conservative representation of the EPC.

Some of the distributional methods employed by ProUCL can produce very high estimates of the UCL (particularly the Land method). Calculating UCL for EPCs (OSWER 9285.6-10) acknowledges that the Land method can produce extremely high values for the UCL when data exhibit high variance and the sample size is small. RAGS Supplemental Guidance (OSWER Publication 9285.7-081) recognizes the problem of extremely high UCLs, and recommends the maximum detected concentration become the default when the calculated UCL exceeds this value. However, when the recommended UCL exceeds the maximum detected concentration, ProUCL advises that an alternative UCL (that is, Chebyshev inequality) be selected instead of the maximum detected concentration for an EPC. When the recommended UCL is greater than the maximum detected result, the maximum detected value is selected as the EPC for the 100-D/H Source OU. ProUCL displays a warning message when the recommended 95 percent UCL of the mean exceeds the observed maximum concentration.

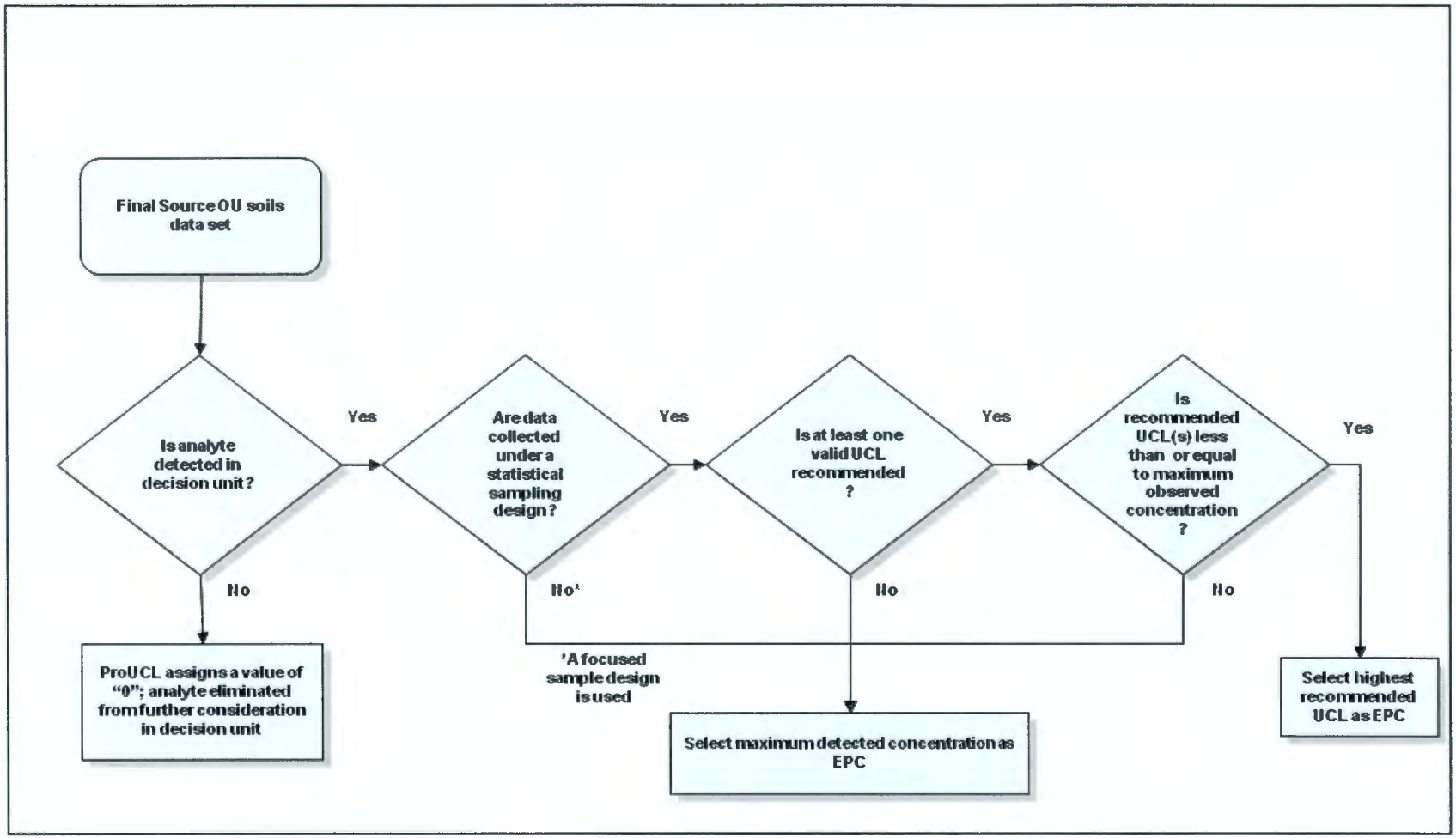

Figure 6-5. Decision Logic for Selection of the EPC Value

6.2.2.3 Methodology Used to Calculate Total Uranium Concentrations from Isotopic Uranium Concentrations

Uranium analytical data are reported for all the 100-DH Source OU waste site decision units as isotopic uranium (reported in units of pCi/g) and not as total uranium (reported in units of µg/kg). Because total uranium (µg/kg) is needed to support the 100-D/H Area Source OU FS, an additional step is performed to calculate a mass-based total uranium concentration (µg/kg) from the activity-based isotopic uranium concentrations (pCi/g) reported for each waste site decision unit. This step entails obtaining the uranium isotope analytical data for each sample, converting the data from activity- to mass-based concentrations, and then summing the converted values for detected concentrations to produce a mass-based total uranium value. For sample results where all uranium isotope results are reported as nondetects, the individual values are not summed, but the maximum nondetect value is retained and put in the ProUCL file, flagged as a nondetect. The pCi/g to µg/kg conversions and subsequent summations are performed using specific activities for the uranium isotopes and appropriate conversion factors, as shown in the calculation example provided in Table 6-11. As mentioned previously, only detected concentrations are included in the summations. In the Table 6-11 example, U-235 is a nondetect and thus not included in the summation. The calculated total uranium values are assigned an analyte name of Total_U_Isotopes in the datasets and then a ProUCL input file (as described in Section 6.2.2.2) containing the Total_U_Isotopes data is produced for each waste site decision unit.

Table 6-11. Example Conversion from Activity- to Mass-Based Concentration (pCi/g to µg/kg) for Uranium Isotopes and Summation to Produce a Mass-Based Total Uranium Concentration (µg/kg)

Uranium Isotope	Measured Activity (pCi isotope/ g soil)[a] (ND or D)	Specific Activity (Bq isotope/ g isotope)[b]	Specific Activity (pCi isotope/ g isotope)[c]	Conversion Factor (µg isotope/ g isotope)	Conversion Factor (g soil/ kg soil)	Calculated Concentration (µg isotope/ kg soil)[d]
U-233/234[e]	0.649 (D)	2.302E+08	6.222E+09	1,000,000	1,000	0.10
U-235	0.031 (ND)	7.995E+04	2.161E+06	1,000,000	1,000	14 (not summed)
U-238	0.338 (D)	1.243E+04	3.359E+05	1,000,000	1,000	1,006
Total Uranium Concentration (Total_U_Isotopes) (µg total uranium/kg soil) =						**1,006**

a. Example analytical data shown for illustration purposes only.

b. *Table of Isotopes* (Firestone and Shirley, 1998).

c. Formula = specific activity (Bq/g) / 3.7E+10 Bq/Ci × 1.0E+12 pCi/Ci.

d. Formula = measured activity (pCi/g) / specific activity (pCi/g) × conversion factor (µg/g) × conversion factor (g/kg).

e. Values presented are for uranium-234; uranium-234 is assumed to be the dominant isotope in undifferentiated uranium-233/234

6.2.3 Exposure Assessment

This section defines the exposure scenarios used for various land use and receptor activities, describes the potential exposure pathways resulting from Hanford Site contaminants, and provides the methodology for calculating the RBSLs for direct contact, based on currently available Hanford Site information.

The conceptual exposure model is formulated according to EPA guidance, taking into consideration information on contaminant sources, release mechanisms, routes of migration, potential exposure points, potential routes of exposure, and potential receptor groups associated with the 100-D/H Source OUs. This results in a set of exposure pathways that reflect an RME.

An exposure pathway can be described as the physical course that a COPC takes from the point of release to a receptor. The route of exposure is the means by which a COPC enters a receptor. For an exposure pathway to be complete, all of the following components must be present:

- A source
- A mechanism of chemical release and transport
- An environmental transport medium
- An exposure point
- An exposure route
- A receptor or exposed population

In the absence of any one of these components, an exposure pathway is considered incomplete; therefore, it creates no risk or hazard[11].

6.2.3.1 Contaminant Sources

The primary sources of contamination in 100-D/H Source OU are three water-cooled nuclear reactors (105-D, 105-DR, and 105-H) and the structures (for example, fuel storage basins) and processes (for example, sodium dichromate process) associated with reactor operations. The reactors were built to irradiate uranium-enriched fuel rods from which plutonium and other special nuclear materials could be extracted. Effluent generated during operations consisted primarily of contaminated reactor cooling water, fuel storage basin water, and decontamination solutions.

Liquid and solid wastes from reactor operations and associated facilities were released to the vadose zone column and the Columbia River. Wastes released to the environment created secondary sources of contamination such as surface impoundments, cribs, ditches, burial grounds, and unplanned release sites. Contaminant sources (that is, facilities and waste sites) are described in Sections 4.2 and 5.3 of this report.

6.2.3.2 Release Mechanisms and Environmental Transport Media

The primary COPC release mechanisms and transport pathways at 100-D and 100-H are discussed in Sections 5.4 and 5.5, and include the following:

- Migration of contaminated liquids through the vadose zone column through infiltration, percolation, or leaching
- Direct contact and external radiation from vadose zone material containing COPCs (receptor contact with shallow vadose zone material replaces release and transport)
- Emission of dusts and vapors during former plant operations
- Generation of dust emanating from shallow vadose zone material to ambient air from wind, or during maintenance or excavation activities occurring at the 100-D/H Source OU
- Volatilization of COPCs emanating from shallow vadose zone material to ambient air at the 100-D/H Source OU

6.2.3.3 Potentially Complete Human Exposure Pathways and Receptors

Based on the current understanding of land use conditions near the 100-D/H Source OU, the most plausible exposure pathways for calculating PRGs and characterizing the human health risks have been

[11] With the exception of external irradiation from radionuclides, environmental contaminants must cross a cellular barrier and enter the body of a receptor for exposure to occur.

identified (represented on Figures G-1 and G-2 in Appendix G). The groundwater risk assessment is provided in Section 6.3.

For the purpose of this soil risk assessment, shallow vadose zone material is represented by samples collected from 0 to 4.6 m (0 to 15 ft) bgs, and deep vadose zone material is represented by samples collected from depths greater than 4.6 m (15 ft) bgs (Section 6.2.2.2, Table 6-10). Groundwater is represented by samples collected from the unconfined aquifer and discussed in Section 6.3.

6.2.3.3.1 *Residential Scenario*

PRGs (also used as RBSLs) developed for the Residential scenario are the numeric values that represent the RAOs presented in Chapter 8. The results of comparing EPCs to the RBSLs in this soil risk assessment will be used to help determine whether additional remedial action is necessary for waste sites where remediation has been completed, and whether the goals and objectives of the interim action RODs have been met, as demonstrated by verification sampling and analysis.

The Residential scenario for radiological and nonradiological analytes is based on two different conceptual exposure models. The exposure pathways for radionuclides include direct contact in addition to dust inhalation, consumption of homegrown foodstuffs (for example, produce, beef, and milk), and the leaching pathway (includes drinking water ingestion and fish ingestion). The exposure pathways for nonradiological analytes in vadose zone material include direct contact from incidental ingestion and inhalation of vapors and dust in ambient air.

The Residential scenarios described in the following paragraphs are consistent with the exposure scenario and ARARs used to develop the interim action remedial action goals for soil presented in the 100 Area RDR/RAWP (DOE/RL-96-17). This exposure scenario is also evaluated in the RCBRA (DOE/RL-2007-21, Volume II) to determine whether cleanup actions completed under the interim action RODs are protective of human health relative to the range of exposure scenarios evaluated in this risk assessment.

Radiological. Consistent with the 100 Area RDR/RAWP (DOE/RL-96-17), the RESRAD code is used to evaluate exposure to radiological contaminants in vadose zone material. Revisions to this exposure scenario reflect updates in guidance since the 100 Area RDR/RAWP (DOE/RL-96-17) was originally published in 1996. With the exception of changes resulting from updates in guidance, the Residential scenario is the same as that published in the 100 Area RDR/RAWP (DOE/RL-96-17) (see Table 6-4). Exposure assumptions that were updated to reflect current EPA guidance include a decrease in the external gamma shielding factor (increased shielding) and a decrease in the outdoor time fraction. Health protective levels were also updated from a target annual dose rate of 15 mrem/yr to a target risk of 1×10^{-4} to be consistent with guidance recommended in *Radiation Risk Assessment At CERCLA Sites: Q & A (*EPA/540/R/99/006). A detailed description of this exposure scenario is provided in *Documentation of Preliminary Remediation Goals (PRGs) for Radionuclides Using the IAROD Exposure Scenario for the 100 and 300 Area Remedial Investigation/Feasibility Study (RI/FS) Report* (ECF-HANFORD-10-0429). A summary of the exposure assumptions that were modified as a result of updates to EPA guidance was provided in Table 6-4.

For radiological PRG development, a subsistence farming setting is used. This assumes that each interim remediated waste site decision unit has 1) the potential to be developed into a residence with a basement, 2) vegetable and fruit crops grown in a backyard garden, and 3) a pasture that is used to raise livestock sufficient for meat and milk production. A downgradient well is installed where exposure could potentially occur from contaminants leaching from the vadose zone material to groundwater beneath the residence (that is, the leaching pathway). The resident could potentially come into direct contact with soil

from the remediated waste site and potentially inhale dust in ambient air. The resident could potentially consume crops raised in a backyard garden and consume meat (beef and poultry) and milk from livestock raised on the pasture. Based on established land uses and the proclamation of "Establishment of the Hanford Reach National Monument" (65 FR 37253), it is unlikely that land within the l00-D/H OU will be used for residential purposes.

The Residential scenario evaluates residential pathways that include exposure to shallow vadose zone material from residential yards or groundwater from domestic wells. Potential routes of exposure to shallow vadose zone material evaluated in the RESRAD code include direct external exposure, incidental material ingestion, and inhalation of dust generated from wind or from yard maintenance activities. This scenario also evaluates residential exposure to radiological contaminants through food chain pathways (uptake of contamination from vadose zone material to plants and animals). Food chain pathways include the consumption of fruits and vegetables grown in a backyard garden and consumption of meat and milk from livestock raised on the pasture. From the leaching pathway, this scenario evaluates residential consumption of drinking water from a downgradient well, use of the well for irrigating crops and watering livestock, and residential consumption of fish raised in a pond supplemented with water from the downgradient well.

Nonradiological. The Residential scenario for nonradiological analytes measured in soil is also consistent with the exposure scenario used for the interim action remedial action goals for soil presented in the 100 Area RDR/RAWP (DOE/RL-96-17). The exposure scenario for protection of human health is based on MTCA ("Unrestricted Land Use Soil Cleanup Standards" "Method B Soil Cleanup Levels for Unrestricted Land Use" [WAC 173-340-740(3)] and "Cleanup Standards to Protect Air Quality" "Method B Air Cleanup Levels" [WAC 173-340-750(3)]). The MTCA (WAC 173-340) Standard Method B soil cleanup levels are based on exposure to a child receptor that includes incidental ingestion, and use residential exposure frequency and duration assumptions. The MTCA (WAC 173-340) Standard Method B air cleanup levels are based on exposure to child and adult receptors, includes inhalation of vapors and dust in ambient air, and assumes residential exposure frequency and duration assumptions. For arsenic and lead, MTCA ("Tables" [WAC 173-340-900]), Table 740-1 Method A, soil cleanup level for unrestricted land use of 20 mg/kg and 250 mg/kg were used.

Groundwater. Groundwater within the 100-HR-3 OU is currently contaminated, and withdrawal is prohibited as a result of institutional controls placed on it by DOE through the interim action ROD; however, institutional controls will be evaluated as part of the final remedy. Under current Hanford Site use conditions, no complete human exposure pathways to groundwater are assumed to exist. In addition, groundwater currently discharges to the Columbia River through upwelling and seeps. Groundwater within this OU is not anticipated to become a future source of drinking water until cleanup criteria are met and groundwater is restored to its highest beneficial use. However, groundwater in this risk analysis is evaluated for drinking water use and undiluted groundwater concentrations are compared to DWSs and aquatic criteria to support the determination of the basis for action and to support the development of PRGs for evaluating remedial alternatives in the FS. It is noted that aquatic water quality criteria are only directly applicable where groundwater discharges to surface water.

The Residential scenario for radiological and nonradiological analytes measured in groundwater is also consistent with the remedial action goals documented in the interim action RODs and in the 100 Area RDR/RAWP (DOE/RL-96-17). Groundwater concentrations are compared to current maximum contaminant levels (MCLs) for radionuclides, which are set at 4 mrem/yr for the sum of the doses from beta particle and photon emitters, 15 pCi/L for gross alpha emitter activity (including Ra-226, but excluding uranium and radon), and 5 pCi/L combined for Ra-226 and Ra-228. A mass-based concentration MCL has been established for uranium as 30 μg/L. The exposure scenario for protection of human health is based on the MTCA Standard Method B ("Groundwater Cleanup Standards" "Standard

Method B Potable Groundwater Cleanup Levels" [WAC 173-340-720 (4)(b)]). The MTCA (WAC 173-340) Standard Method B groundwater cleanup levels are based on exposure to child and adult receptors, include drinking water ingestion and inhalation of vapors, and makes residential exposure frequency and duration assumptions.

6.2.3.3.2 *Resident Monument Worker Scenario*

Land use within the River Corridor's 100 and 600 Areas is predominantly conservation/preservation. In 2000, Presidential Proclamation 7319 (*Establishment of the Hanford Reach National Monument* was signed creating the Hanford Reach National Monument, to be managed by USFWS and DOE ("Establishment of the Hanford Reach National Monument" [65 FR 37253]). The Monument was established to protect the biological, historic, and scientific objects contained within. To support continued protection of natural and cultural resources, the proclamation stated that the Monument would not be developed for residential or commercial use in the future ("Establishment of the Hanford Reach National Monument" [65 FR 37253]).

This exposure scenario was included in the subset of occupational scenarios presented in the RCBRA (DOE/RL-2007-21, Volume II). The Resident Monument Worker scenario is a site-specific scenario that envisions a resident employee of the Hanford Reach National Monument. These receptors are assumed to be exposed primarily in an outdoor environment as they lead tours, conduct ecological education, or perform similar activities. When not working, these receptors are envisioned to live in an onsite residence associated with the Monument. By use of a domestic well at their residence, these receptors may also be exposed to groundwater contaminants through domestic water use. Exposure to groundwater as a domestic source of water by the resident Monument worker is not included in the soil PRG value that is calculated for this exposure scenario. The risks from exposure to 100-HR-3 groundwater from use as a domestic source of water can be separately added to provide a total risk from exposure to soil and groundwater.

The Resident Monument Worker scenario for radiological and nonradiological analytes in vadose zone material is based on the same conceptual exposure model. The exposure pathways include direct contact and inhalation of vapors and dust in ambient air. Adults could potentially be exposed to Hanford Site contaminants in shallow vadose zone material at their residence through direct external exposure, incidental ingestion, dermal absorption, and inhalation. During working activities, these adults may also be potentially exposed to contaminants in shallow vadose zone material by direct external exposure, incidental soil ingestion, dermal absorption, and inhalation. No food chain pathways are included in this exposure scenario.

When the total risk for a waste site is less than 1×10^{-4} for radionuclides based on the Residential scenario or 1×10^{-5} for chemicals based on the MTCA ("Human Health Risk Assessment Procedures" [WAC 173-340-708(5)]) cumulative risk threshold, then protection of the resident Monument worker is achieved. The results of these comparisons can be used in risk management decisions (presented in Section 6.2.5.5) and show that the total risk calculated for the Resident and the Resident Monument Worker scenarios are essentially identical. The Residential PRGs are slightlylower than the Resident Monument Worker PRGs because the Residential exposure scenario includes the food chain pathways.

6.2.3.3.3 *Casual Recreational User Scenario*

As discussed previously, the reasonably anticipated future land use within the River Corridor's 100 and 600 Areas is predominantly conservation/preservation. The casual recreational user is selected as a receptor to represent potential exposures from recreational use along the River Corridor. This exposure scenario was included in the subset of recreational use scenarios presented in the RCBRA (DOE/RL-2007-21, Volume II). The Casual Recreational User scenario is a site-specific scenario representing occasional recreational use that focuses on activities such as walking and picnicking in areas along the Columbia River

where paths and benches are likely to exist. These receptors are assumed to be exposed entirely in an outdoor environment. This scenario also assumes that drinking water is obtained from an offsite source.

PRGs are presented in this section for the casual recreational user that represents a reasonably anticipated future land use. Casual recreational user PRG values are developed for radiological and nonradiological contaminants. When the total risk for a waste site is less than 1×10^{-4} based on the Residential scenario or 1×10^{-5} for chemicals based on the MTCA ("Human Health Risk Assessment Procedures" [WAC 173-340-708(5)]) cumulative risk threshold, then protection of the casual recreational user is achieved. The results of these comparisons (presented in Section 6.2.5.5) can be used in risk management decisions.

The Casual Recreational User scenario for radiological and nonradiological analytes in vadose zone material is based on the same conceptual exposure model. The exposure pathways include direct contact and inhalation of vapors and dust in ambient air. Adults and children could potentially be exposed to Hanford Site contaminants in shallow vadose zone material along the river through direct external exposure, incidental ingestion, dermal absorption, and inhalation of vapors and dust in ambient air.

6.2.3.4 Quantification of Potential Exposures

Quantification of potential exposures in this risk assessment is evaluated through the comparison of EPCs to PRGs (which are also used as RBSLs). *Risk Assessment Guidance for Superfund Volume I – Human Health Evaluation Manual (Part B, Development of Risk-Based Preliminary Remediation Goals): Interim* (EPA/540/R-92/003), hereinafter called Risk Assessment Guidance Volume I, Part B, provides guidance on using EPA toxicity values and exposure information to calculate PRGs. Once the BRA has been performed, PRGs can be derived using site-specific risks. PRGs developed in the FS will usually be based on site-specific risks and ARARs and not on screening levels. PRGs are obtained from two general sources: concentrations based on ARARs (for example DWS), and concentrations based on risk assessment. It should be recognized that the PRGs that are ARAR-based are also considered risk-based. Exposure assumptions published by the state and EPA and toxicity values published by EPA are used to derive risk-based PRGs.

PRGs based on risk assessment equations include the Resident Monument Worker and the Casual Recreational User scenarios. PRGs for these scenarios are calculated using methodologies published in Risk Assessment Guidance Volume I, Part B (EPA/540/R-92/003) and the Superfund Radionuclide PRG download and calculation web site (EPA, 2010b). Toxicity values and exposure values published by EPA are used to derive risk-based PRGs.

The Residential scenario for chemicals is based on the MTCA Method B Soil Cleanup Levels ("Unrestricted Land Use Soil Cleanup Standards" [WAC 173-340-740]) and MTCA Method B Air Cleanup Levels ("Method B Air Cleanup Levels" [WAC 173-340-750]). PRGs for soil ingestion are calculated using the equations provided in MTCA ("Unrestricted Land Use Soil Cleanup Standards" [WAC 173-340-740(3)]). PRGs for the inhalation pathway are calculated using the equations provided in MTCA ("Cleanup Standards to Protect Air Quality" "Method B Air Cleanup Levels" [WAC 173-340-750(3)]). Air cleanup levels are converted to soil concentrations using EPA published volatilization factors for analytes that meet the operational definition of a volatile and a PEF for analytes that are not volatile. MTCA Method A soil cleanup levels for unrestricted land use, obtained from MTCA ("Tables" [WAC 173-340-900]), Table 740-1 are used as PRGs for arsenic and lead.

In addition to the guidance listed previously, radionuclide PRGs for the resident are calculated using the RESRAD code. The RESRAD code was used to calculate PRGs for the Residential scenario because of unique exposure pathways. The RESRAD code was used for the Residential scenario because this scenario includes the food chain pathway and the leaching to groundwater pathway. According to *User's*

Manual for RESRAD Version 6 (ANL/EAD-4), the RESRAD model and computer code were developed as a multifunctional tool to assist in developing cleanup criteria and assessing the dose or risk associated with residual radioactive material.

Table 6-12 summarizes the PRG values for each exposure scenario.

6.2.3.4.1 *Calculation of Residential PRGs using RESRAD*

The radionuclide PRGs for the Residential scenario are calculated using RESRAD, Version 6.5 (ANL, 2009b) model and code according to the guidance specified in *User's Manual for RESRAD Version 6* (ANL/EAD-4). The RESRAD model was used to calculate single radionuclide concentrations that correspond to a target cancer risk level of 1×10^{-4} for the Residential scenario. For the purpose of this soil risk assessment, the single radionuclide concentrations described in this section are used as PRGs for the Residential scenario.

The RESRAD model allows for the use of site-specific chemical and physical parameters to estimate single radionuclide concentrations. The potentially complete exposure pathways considered are direct contact, inhalation pathway, the food chain pathway, and leaching of contaminants in the vadose zone through the vadose zone column to the groundwater table. Exposure routes associated with the direct contact and inhalation pathways include external gamma exposure, incidental ingestion, and inhalation of dust. Exposure routes associated with the food chain exposure pathway include consumption of homegrown produce, meat, and milk. Exposure routes associated with the leaching pathway include crop irrigation, aquatic food consumption, and drinking water ingestion. A detailed description of methodology, inputs, assumptions, and results of the calculations is presented in *Documentation of Preliminary Remediation Goals (PRGs) for Radionuclides Using the IAROD Exposure Scenario for the 100 and 300 Area Remedial Investigation/Feasibility Study (RI/FS) Report* (ECF-HANFORD-10-0429) in Appendix G.

6.2.3.4.2 *Calculation of Unrestricted Land Use PRGs using MTCA Equations*

The direct contact nonradiological PRGs for unrestricted land use (that is, the resident) are calculated using equations and input parameters described in MTCA ("Unrestricted Land Use Soil Cleanup Standards" [WAC 173-340-740(3)]). The Standard Method B soil cleanup levels for unrestricted land use are based on ingestion and were calculated for noncarcinogens and carcinogens using equation 740-1 and equation 740-2, respectively. Standard Method B soil cleanup levels for unrestricted land use are based on an acceptable cancer risk level of 1×10^{-6} for nonradiological carcinogens or a hazard quotient (HQ) of 1 for noncarcinogens.

Reference dose (RfD) and carcinogenic potency factors are determined using the recommended reference hierarchy as described in "Human Health Toxicity Values in Superfund Risk Assessments" (Cook, 2003), hereinafter called Superfund HHT Risk Assessment Values. A detailed description of methodology, inputs, assumptions, and the results of the calculations is presented in *Calculation of Standard Method B Direct Contact Soil Cleanup Levels for Unrestricted Land Use* (ECF-HANFORD-10-0044) (Appendix G).

The inhalation nonradiological PRGs for unrestricted land use (that is, the resident) are calculated using equations and input parameters described in MTCA ("Cleanup Standards to Protect Air Quality," "Method B Air Cleanup Levels" [WAC 173-340-750(3)]). The Method B air PRGs are calculated for noncarcinogens and carcinogens using equation 750-1 and equation 750-2, respectively.

Air PRGs are converted to soil concentrations using EPA-published volatilization factors for analytes that meet the operational definition of a volatile and a PEF for analytes that are not volatile. Method B soil PRGs for the inhalation pathway are based on an acceptable cancer risk level of 1×10^{-6} for carcinogens

or an HQ of 1 for noncarcinogens. Inhalation RfD and inhalation carcinogenic potency factors are determined using the recommended reference hierarchy as described in Superfund HHT Risk Assessment Values (Cook, 2003). A detailed description of methodology, inputs, and assumptions and the results of the calculations are presented in *Calculation of Inhalation Pathway Preliminary Remediation Goals Using Standard Method B Air Cleanup Levels for the 100 Areas and 300 Area Remedial Investigation/Feasibility Study Reports* (ECF-HANFORD-11-0033) in Appendix G.

6.2.3.4.3 Calculation of Resident Monument Worker PRGs for Radiological Analytes using EPA Equations

The radiological PRGs for the resident Monument worker are calculated using equations consistent with those published on the EPA Preliminary Remediation Goals for Radionuclides Web site. Resident Monument worker PRGs are based on an acceptable cancer risk level of 1×10^{-4} for carcinogens. A detailed description of methodology, inputs, and assumptions and the results of the calculations is presented in *Documentation of Radiological Preliminary Remediation Goals in Soil for a Resident Monument Worker Exposure Scenario for the 100 Areas and 300 Area Remedial Investigation/Feasibility Study (RI/FS) Reports* (ECF-HANFORD-11-0142).

6.2.3.4.4 Calculation of Casual Recreational User PRGs for Radiological Analytes using EPA Equations

The radiological PRGs for the casual recreational user are calculated using equations consistent with those published on the EPA Preliminary Remediation Goals for Radionuclides Web site. Casual recreational user radiological PRGs are based on an acceptable cancer risk level of 1×10^{-4} for carcinogens. A detailed description of methodology, inputs, assumptions, and the results of the calculations is presented in *Calculation of Radiological Preliminary Remediation Goals in Soil for a Casual Recreational User Scenario for the 100 Areas and 300 Area Remedial Investigation/Feasibility Study (RI/FS) Reports* (ECF-HANFORD-10-0446).

6.2.3.4.5 Calculation of Casual Recreational User PRGs for Nonradiological Analytes using EPA Equations

The nonradiological PRGs for the casual recreational user are calculated using equations consistent with those published on "Regional Screening Levels for Chemical Contaminants at Superfund Sites" (hereinafter called Regional Screening Levels [EPA, 2009a]). Casual recreational user nonradiological PRGs are based on an acceptable cancer risk level of 1×10^{-6} for carcinogens or an HQ of 1 for noncarcinogens. RfD and carcinogenic potency factors are determined using the recommended reference hierarchy as described in Superfund HHT Risk Assessment Values (Cook, 2003). A detailed description of methodology, inputs and assumptions and the results of the calculations are presented in *Calculation of Nonradiological Preliminary Remediation Goals in Soil for a Casual Recreational User Scenario for the 100 Areas and 300 Area Remedial Investigation/Feasibility Study (RI/FS) Reports* (ECF-HANFORD-10-0445).

6.2.4 Toxicity Assessment

This toxicity assessment evaluates the relationship between the magnitude of exposure to a contaminant at the 100-D/H Source OU and the likelihood of adverse health effects to potentially exposed populations. This assessment provides, where possible, a numerical estimate of the increased likelihood of adverse effects associated with contaminant exposure. The toxicity assessment contains two steps—hazard characterization and dose-response evaluation—as discussed in the following sections.

Table 6-12. Summary of Risk-based Screening Levels for the 100-D/H Source OU

Analyte	90th Percentile Background	Method A Soil PRG	Residential PRG	Method B Direct Contact Soil PRG (Carcinogen)	Method B Direct Contact PRG (Noncarcinogen)	Method B Inhalation Soil PRG (Carcinogen)	Method B Inhalation Soil PRG (Noncarcinogen)	Casual Recreational User PRG (Carcinogen)	Casual Recreational User PRG (Noncarcinogen)	Resident Monument Worker PRG (Carcinogen)
Radionuclides (pCi/g)										
Americium-241	--	--	155	--	--	--	--	2,570	--	275
Carbon-14	--	--	81	--	--	--	--	328,000	--	52,000
Cesium-137	1.1	--	4.4	--	--	--	--	100	--	6.2
Cobalt-60	0.0084	--	3.1	--	--	--	--	63	--	3.3
Europium-152	--	--	3.7	--	--	--	--	66	--	3.8
Europium-154	0.033	--	4.4	--	--	--	--	78	--	4.8
Europium-155	0.054	--	327	--	--	--	--	5,870	--	354
Neptunium-237	--	--	8.9	--	--	--	--	202	--	15
Nickel-63	--	--	608	--	--	--	--	575,000	--	91,600
Plutonium-238	0.0038	--	236	--	--	--	--	3,820	--	605
Plutonium-239/240	0.025	--	203	--	--	--	--	3,340	--	539
Technetium-99	--	--	1.5	--	--	--	--	114,000	--	17,300
Total beta radiostrontium	0.18	--	2.3	--	--	--	--	5,060	--	518
Tritium	--	--	623	--	--	--	--	15,400	--	1,270,000
Uranium-233/234	1.1	--	133	--	--	--	--	5,810	--	931
Uranium-234	1.1	--	133	--	--	--	--	5,810	--	931
Uranium-235	0.11	--	16	--	--	--	--	295	--	22
Uranium-238	1.1	--	54	--	--	--	--	1,090	--	93
Metals (mg/kg)										
Aluminum	11,800	--	--	--	80,000	--	>1,000,000	--	912,000	--
Antimony	0.13	--	--	--	32	--	--	--	365	--
Arsenic	6.5	20	--	0.67	24	42,400	500,000	4.5	253	--
Barium	132	--	--	--	16,000	--	>1,000,000	--	182,000	--
Beryllium	1.5	--	--	--	160	76,000	667,000	>1,000,000	1,820	--
Boron	3.9	--	--	--	16,000	--	>1,000,000	--	182,000	--
Cadmium	0.56	--	--	--	40	101,000	333,000	>1,000,000	821	--
Chromium	19	--	--	--	120,000	--	--	--	>1,000,000	--
Cobalt	16	--	--	--	24	20,300	200,000	920,000	274	--
Copper	22	--	--	--	3,200	--	--	--	36,500	--
Cr(VI)	--	--	--	--	240	2,170	>1,000,000	98,600	2,740	--
Iron	32,600	--	--	--	56,000	--	--	--	639,000	--
Lead	10	250	--	--	--	--	--	--	--	--
Lithium	13	--	--	--	160	--	--	--	1,830	--
Manganese	512	--	--	--	11,200	--	>1,000,000	--	128,000	--
Mercury	0.013	--	--	--	24	--	>1,000,000	--	274	--

Table 6-12. Summary of Risk-based Screening Levels for the 100-D/H Source OU

Analyte	90th Percentile Background	Method A Soil PRG	Residential PRG	Method B Direct Contact Soil PRG (Carcinogen)	Method B Direct Contact PRG (Noncarcinogen)	Method B Inhalation Soil PRG (Carcinogen)	Method B Inhalation Soil PRG (Noncarcinogen)	Casual Recreational User PRG (Carcinogen)	Casual Recreational User PRG (Noncarcinogen)	Resident Monument Worker PRG (Carcinogen)
Molybdenum	0.47	--	--	--	400	--	--	--	4,560	--
Nickel	19	--	--	--	1,600	701,000	>1,000,000	>1,000,000	18,200	--
Selenium	0.78	--	--	--	400	--	>1,000,000	--	4,560	--
Silver	0.17	--	--	--	400	--	--	--	4,560	--
Strontium	--	--	--	--	48,000	--	--	--	548,000	--
Tin	--	--	--	--	48,000	--	--	--	548,000	--
Total_U_Isotopes	3.2	--	--	--	240	--	>1,000,000	--	2,740	--
Uranium	3.2	--	--	--	240	--	>1,000,000	--	2,740	--
Vanadium	85	--	--	--	400	--	--	--	4,560	--
Zinc	68	--	--	--	24,000	--	--	--	274,000	--
Polyaromatic Hydrocarbons (mg/kg)										
Acenaphthene	--	--	--	--	4,800	--	--	--	40,100	--
Anthracene	--	--	--	--	24,000	--	--	--	201,000	--
Benzo(a)anthracene	--	--	--	1.4	--	>1,000,000	--	1.7	--	--
Benzo(a)pyrene	--	--	--	0.14	--	166,000	--	0.17	--	--
Benzo(b)fluoranthene	--	--	--	1.4	--	>1,000,000	--	1.7	--	--
Benzo(k)fluoranthene	--	--	--	1.4	--	>1,000,000	--	1.7	--	--
Chrysene	--	--	--	14	--	>1,000,000	--	17	--	--
Dibenz[a,h]anthracene	--	--	--	1.4	--	>1,000,000	--	1.7	--	--
Fluoranthene	--	--	--	--	3,200	--	--	--	26,800	--
Fluorene	--	--	--	--	3,200	--	--	--	26,800	--
Indeno(1,2,3-cd)pyrene	--	--	--	1.4	--	>1,000,000	--	1.7	--	--
Pyrene	--	--	--	--	2,400	--	--	--	20,100	--
Polychlorinated Biphenyls (mg/kg)										
Aroclor-1242	--	--	--	0.50	--	320,000	--	2.6	--	--
Aroclor-1248	--	--	--	0.50	--	319,963	--	2.6	--	--
Aroclor-1254	--	--	--	0.50	1.6	320,000	--	2.6	13	--
Aroclor-1260	--	--	--	0.50	--	320,000	--	2.6	--	--
Anions (mg/kg)										
Fluoride	2.8	--	--	--	4,800	--	>1,000,000	--	54,700	--
Nitrate	52	--	--	--	568,000	--	--	--	>1,000,000	--
Nitrite	--	--	--	--	24,000	--	--	--	274,000	--
Nitrogen in Nitrate	--	--	--	--	128,000	--	--	--	>1,000,000	--
Nitrogen in Nitrite	--	--	--	--	8,000	--	--	--	91,300	--
Nitrogen in Nitrite and Nitrate	--	--	--	--	128,000	--	--	--	>1,000,000	--

Table 6-12. Summary of Risk-based Screening Levels for the 100-D/H Source OU

Analyte	90th Percentile Background	Method A Soil PRG	Residential PRG	Method B Direct Contact Soil PRG (Carcinogen)	Method B Direct Contact PRG (Noncarcinogen)	Method B Inhalation Soil PRG (Carcinogen)	Method B Inhalation Soil PRG (Noncarcinogen)	Casual Recreational User PRG (Carcinogen)	Casual Recreational User PRG (Noncarcinogen)	Resident Monument Worker PRG (Carcinogen)
Other Organics (mg/kg)										
1,1-Dichloroethene	--	--	--	--	4,000	--	102	--	8,773	--
1,2-Dichlorobenzene	--	--	--	--	7,200	--	546	--	34,000	--
2-(2-methyl-4-chlorophenoxy) propionic acid	--	--	--	--	80	--	--	--	713	--
2,4,5-Trichlorophenol	--	--	--	--	8,000	--	--	--	71,300	--
2,4-DB(4-(2,4-Dichlorophenoxy)butanoic acid)	--	--	--	--	640	--	--	--	5,703	--
2,4-Dichlorophenol	--	--	--	--	240	--	--	--	2,140	--
2,4-Dinitrophenol	--	--	--	--	160	--	--	--	1,426	--
2-Butanone	--	--	--	--	48,000	--	28,700	--	464,000	--
2-Chloronaphthalene	--	--	--	--	6,400	--	--	--	73,000	--
2-Hexanone	--	--	--	--	400	--	160	--	3,599	--
2-Methylnaphthalene	--	--	--	--	320	--	--	--	2,680	--
4,4'-DDD (Dichlorodiphenyldichloroethane)	--	--	--	4.2	--	>1,000,000	--	24	--	--
4,4'-DDE (Dichlorodiphenyldichloroethylene)	--	--	--	2.9	--	>1,000,000	--	17	--	--
4,4'-DDT (Dichlorodiphenyltrichloroethane)	--	--	--	2.9	40	>1,000,000	--	20	421	--
Acetone	--	--	--	--	72,000	--	190,000	--	789,000	--
Aldrin	--	--	--	0.059	2.4	37,200	--	0.33	21	--
Alpha-BHC	--	--	--	0.16	640	101,322	--	0.90	5,703	--
Alpha-Chlordane	--	--	--	2.9	40	>1,000,000	>1,000,000	19	410	--
beta-1,2,3,4,5,6-Hexachlorocyclohexane (beta-BHC)	--	--	--	0.56	--	344,000	--	3.2	--	--
Bis(2-ethylhexyl) phthalate	--	--	--	71	1,600	>1,000,000	--	405	14,300	--
Butylbenzylphthalate	--	--	--	526	16,000	--	--	2,980	143,000	--
Carbazole	--	--	--	50	--	--	--	283	--	--
Chlordane	--	--	--	2.9	40	>1,000,000	>1,000,000	19	410	--
Chloroform	--	--	--	32	800	0.24	100	11	4,908	--
Dibenzofuran	--	--	--	--	80	--	--	--	913	--
Dieldrin	--	--	--	0.063	4.0	39,600	--	0.35	36	--
Diethylphthalate	--	--	--	--	64,000	--	--	--	570,313	--
Di-n-butylphthalate	--	--	--	--	8,000	--	--	--	71,300	--
Dinoseb(2-secButyl-4,6-dinitrophenol)	--	--	--	--	80	--	--	--	713	--
Endosulfan I	--	--	--	--	480	--	--	--	4,280	--
Endosulfan II	--	--	--	--	480	--	--	--	4,280	--
Ethylbenzene	--	--	--	91	8,000	2.3	1,045	90	50,140	--

Table 6-12. Summary of Risk-based Screening Levels for the 100-D/H Source OU

Analyte	90th Percentile Background	Method A Soil PRG	Residential PRG	Method B Direct Contact Soil PRG (Carcinogen)	Method B Direct Contact PRG (Noncarcinogen)	Method B Inhalation Soil PRG (Carcinogen)	Method B Inhalation Soil PRG (Noncarcinogen)	Casual Recreational User PRG (Carcinogen)	Casual Recreational User PRG (Noncarcinogen)	Resident Monument Worker PRG (Carcinogen)
Gamma-BHC (Lindane)	--	--	--	0.91	24	588,319	--	6.0	246	--
Heptachlor epoxide	--	--	--	0.11	1.0	70,100	--	0.62	9.3	--
Isophorone	--	--	--	1,053	16,000	--	50,482	5,962	142,578	--
Methoxychlor	--	--	--	--	400	--	--	--	3,560	--
Methylene chloride	--	--	--	133	4,800	11	965	337	35,700	--
Naphthalene	--	--	--	--	1,600	1.4	25	62	2,240	--
Phenol	--	--	--	--	24,000	--	11,614	--	213,867	--
Toluene	--	--	--	--	6,400	--	4,770	--	63,800	--
Total petroleum hydrocarbons - diesel range	--	2,000	--	--	--	--	--	--	--	--
Total petroleum hydrocarbons - diesel range extended to C36	--	--	--	--	--	--	--	--	--	--
Total petroleum hydrocarbons - gasoline range	--	30	--	--	--	--	--	--	--	--
Total petroleum hydrocarbons - motor oil (high boiling)	--	2,000	--	--	--	--	--	--	--	--
Xylenes (total)	--	--	--	--	16,000	--	103	--	10,346	--

6.2.4.1 Hazard Characterization

Hazard characterization identifies the types of toxic effects that a chemical can exert. For the toxicity assessment, chemicals can be divided into two broad groups—noncarcinogens and carcinogens—based on their effects on human health.

Carcinogens are those contaminants that are known or suspected causes of cancer following exposure. Noncarcinogenic compounds are associated with a wide variety of systemic effects, such as liver toxicity or developmental effects. Some contaminants (for example, arsenic) are capable of eliciting both carcinogenic and noncarcinogenic responses; therefore, these contaminants are evaluated for both effects.

For cancer effects, EPA has developed a carcinogen classification system (*Guidelines for Carcinogen Risk Assessment* [EPA/630/P-03/001F]) that uses a weight of evidence approach for classifying the likelihood that a chemical is a human carcinogen. Information considered in developing the classification includes human studies of the association between cancer incidence and exposure, as well as long-term animal studies under controlled laboratory conditions. Other supporting evidence considered includes short-term tests for genotoxicity, metabolic and pharmacokinetic properties, toxicological effects other than cancer, structure-activity relationships, and physical and chemical properties of the chemical.

For noncancer effects, toxicity values are derived based on the critical toxic endpoint (that is, the most sensitive adverse effect following exposure). Table G-11 (Appendix G) lists the COPCs detected at the 100-D/H Source OU area that have been identified as having documented systemic effects.

6.2.4.1.1 Dose-response Evaluation

The magnitude of toxicity of a contaminant depends on the dose to a receptor. Dose refers to exposure to a contaminant concentration over a specified period. Human exposures are generally classified as acute (typically less than 2 weeks), subchronic (about 2 weeks to 7 years), or chronic (7 years to a lifetime). This HHRA specifically addresses chronic exposure. Acute exposures and risks are evaluated only when chronic exposure estimates pose a high risk. A dose-response curve describes the relationship between the degree of exposure (i.e., dose) and the incidence of the adverse effects (that is, response) in the exposed population. EPA uses this dose-response information to establish toxicity values for particular chemicals, as described in the following sections.

Reference Doses for Noncancer Effects. The toxicity value describing the dose-response relationship for noncancer effects is the RfD value. For noncarcinogenic effects, the body's protective mechanisms must be overcome before an adverse effect is manifested. If exposure is high enough and these protective mechanisms (or thresholds) are exceeded, adverse health effects can occur. EPA attempts to identify the upper bound of this tolerance range in the development of noncancer toxicity values. EPA uses the apparent toxic threshold value, in conjunction with uncertainty factors based on the strength of the toxicological evidence, to derive an RfD value. EPA defines an RfD value as follows:

In general, the RfD is an estimate (with uncertainty spanning perhaps an order of magnitude) of a daily exposure to the human population (including sensitive subgroups) that is likely to be without an appreciable risk of deleterious effects during a lifetime. The RfD is generally expressed in units of mg/kg-day.

Available chronic RfD values for the oral and inhalation exposure routes are used to calculate PRGs. Because EPA has not derived toxicity values specific to skin contact, dermal slope factors and RfD values were derived from oral toxicity factors in accordance with EPA guidance. The RfD values for the contaminants evaluated in the 100-D/H Source OU are summarized in Table G-11 (Appendix G).

Slope Factors for Cancer Effects. The dose-response relationship for cancer effects is expressed as a cancer slope factor that converts estimated intake directly to excess lifetime cancer risk (ELCR). Slope

factors are expressed in units of risk per level of exposure (or intake). The data used for estimating the dose-response relationship are taken from lifetime animal studies or human occupational or epidemiological studies where excess cancer risk has been associated with exposure to the chemical. However, because risk at low intake levels cannot be directly measured in animal or human epidemiological studies, a number of mathematical models and procedures have been developed to extrapolate from the high doses used in the studies to the low doses typically associated with environmental exposures. The model choice leads to uncertainty associated with the carcinogenic response at very low levels of exposure. EPA assumes linearity at low doses when uncertainty exists about the mechanism of action of a carcinogen and when information suggesting nonlinearity is absent.

It is assumed, therefore, that if a cancer response occurs at the dose levels used in the study, then there is some probability that a response will occur at all lower exposure levels (that is, a dose-response relationship with no threshold is assumed). Moreover, the dose-response slope chosen is usually the 95 percent UCL on the mean on the actual dose-response curve observed in the laboratory studies. As a result, uncertainty and conservatism are built into the EPA risk extrapolation approach. EPA has stated that cancer risks estimated by this method produce estimates that "provide a rough but plausible upper limit of risk." The cancer slope factors used in this assessment are summarized in Table G-11 (Appendix G).

6.2.4.2 Toxicity Values

The analyte-specific toxicity values presented in Table G-11 (Appendix G) are determined using the following recommended reference hierarchy as described in Superfund HHT Risk Assessment Values (Cook, 2003):

- Tier 1—The EPA Integrated Risk Information System (IRIS) database
- Tier 2—The EPA Provisional Peer Reviewed Toxicity Values
- Tier 3—Other Toxicity Values

6.2.4.2.1 *Tier 1—IRIS*

The preferred source of toxicity data is EPA's IRIS database. Expert toxicologists at EPA have derived the values in this database and the values have undergone a thorough review and validation both within and outside EPA. If a toxicity value is available in IRIS, that value is preferred to any other value.

6.2.4.2.2 *Tier 2—Provisional Peer Reviewed Toxicity Values*

If a toxicity value is not available in IRIS, the next source is EPA's Provisional Peer Reviewed Toxicity Values. This source includes toxicity values that have been developed by the Office of Research and Development/National Center for Environmental Assessment/Superfund Health Risk Technical Support Center. This database is not available to the public, but is accessible to EPA risk assessors via EPA's intranet. These values are also published at Regional Screening Levels (EPA, 2009a).

6.2.4.2.3 *Tier 3—Other Toxicity Values*

Tier 3 includes additional EPA and non-EPA sources of toxicity information, including the following:

- The California Environmental Protection Agency (CalEPA)'s Toxicity Criteria Database contains toxicity values that are peer-reviewed and address both cancer and noncancer effects.
- Agency for Toxic Substances and Disease Registry database Minimal Risk Levels for Hazardous Substances are peer-reviewed estimates of the daily human exposure to hazardous substances that is likely to be without appreciable risk of adverse noncancer health effects over a specified duration of exposure.

- Toxicity values in *Health Effects Assessment Summary Tables: FY 1997 Update* (EPA 540-R-97-036), hereinafter called HEAST.

When Tier 1, Tier 2, or Tier 3 toxicity values are not available for a COPC, the toxicity values from the National Center for Environmental Assessment are used. These values can be found in the Risk Assessment Information System (ORNL, 2010).

A derived RfD for nitrate was calculated from the RfD reported in IRIS (1.6 mg/kg-day) for nitrate as nitrogen (NO3-N) using the mass fraction of nitrogen in nitrate. The mass fraction of nitrogen in nitrate = mol wt N/mol wt NO_3^- = (14 g/mol)/(62 g/mol) = 0.226. The derived RfD for nitrate = (1.6 mg NO_3 - N/kg-day) × (1 mg NO_3^-/0.226 mg NO_3-N) = 7.1 mg NO_3-/kg-day.

A derived RfD for nitrite was calculated from the RfD reported in IRIS (0.1 mg/kg-day) for nitrite as nitrogen (NO_2-N) using the mass fraction of nitrogen in nitrite. The mass fraction of nitrogen in nitrite = mol wt N/mol wt NO_2^- = (14 g/mol)/(46 g/mol) = 0.304. The derived RfD for nitrite = (0.1 mg NO_2-N/kg-day) × (1 mg NO_2^-/0.304 mg NO_2-N) = 0.3 mg NO_2^-/kg-day.

Toxic equivalence factors were used to calculate toxicity values for dioxins, furans, and carcinogenic polycyclic aromatic hydrocarbons (PAHs) as described in MTCA ("Human Health Risk Assessment Procedures," hereinafter called HHRA Procedures [WAC 173-340-708(8)(D)(iii)(A)]).

For several nonradionuclide analytes, the toxicity value used was obtained from a different source than recommended by the EPA Superfund hierarchy (Superfund HHT Risk Assessment Values [Cook, 2003]). The following paragraphs summarize the differences in toxicity values.

For consistency with previous Hanford Site analyses of trichloroethene, the oral cancer slope factor of 0.089 $(mg/kg\text{-}day)^{-1}$ and inhalation unit risk of 2.5E-05 $(\mu g/m^3)^{-1}$ published in HEAST (EPA 540-R-97-036) are used for this assessment. HEAST (EPA 540-R-97-036) has not been updated since 1997 and does not reflect the most current source of information. The oral cancer slope factor and inhalation unit risk currently implemented in Regional Screening Levels (EPA, 2009a) are established by the CalEPA Office of Environmental Health Hazard Assessment (OEHHA). The oral slope factor of 0.0059 $(mg/kg\text{-}day)^{-1}$ and the Inhalation Unit Risk of 2.0E-06 $(\mu g/m^3)^{-1}$ derived by OEHHA are presented in *Public Health Goal for Trichloroethylene in Drinking Water* (OEHHA, 2009). If the revised EPA value were used to calculate the MTCA ("Unrestricted Land Use Soil Cleanup Standards" [WAC 173-340-740]) soil cleanup level, the soil concentration would increase from 11 to 169 mg/kg. If the revised EPA value were used to calculate the MTCA ("Cleanup Standards to Protect Air Quality" [WAC 173-340-750]) soil cleanup level, the soil concentration would increase from 0.17 to 2.2 mg/kg. Use of the HEAST (EPA 540-R-97-036) values in this assessment has the potential to overestimate cancer risk.

For fluoride, the oral RfD of 0.06 mg/kg-day published on IRIS is used for this assessment. The value reported on IRIS has not been updated since 1989 and does not reflect the most current source of information. The oral RfD currently implemented by in the Regional Screening Levels (EPA, 2009a) is established by the CalEPA OEHHA. The oral RfD derived by OEHHA is 0.04 mg/kg-day as documented in *Chronic Toxicity Summary: Fluorides Including Hydrogen Fluoride* (OEHHA, 2003). If the revised CalEPA value were used to calculate the MTCA ("Unrestricted Land Use Soil Cleanup Standards" [WAC 173-340-740]) soil cleanup level, the soil concentration would decrease from 4,800 to 3,200 mg/kg. Use of the IRIS value in this assessment has the potential to under-estimate noncancer hazards.

For Cr(VI), the current assessment considers cancer effects only for inhalation exposures. Note that an oral RfD and a reference concentration are available for assessment of noncancer effects. An oral cancer slope factor has recently been published by the New Jersey Department of Environmental Protection

(NJDEP). The oral cancer slope factor derived by NJDEP is 0.5 $(mg/kg\text{-}day)^{-1}$, as presented in *Derivation of an Ingestion-Based Soil Remediation Criterion for Cr^{+6} Based on the NTP Chronic Bioassay Data for Sodium Dichromate Dihydrate* (NJDEP, 2009). If the NJDEP value were used to calculate the MTCA ("Unrestricted Land Use Soil Cleanup Standards" [WAC 173-340-740]) level, the soil concentration would decrease from 240 to 2.0 mg/kg. Assessing only inhalation cancer effects from Cr(VI) has the potential to under-estimate cancer risk.

When evaluating toxicity, 1,1-dichloroethane is not considered a carcinogen by Ecology. Therefore, the oral slope factor of 5.7E-03 $(mg/kg\text{-}day)^{-1}$ and inhalation unit risk of 1.5E-06 $(\mu g/m^3)^{-1}$ presented in the Regional Screening Levels (EPA, 2009a) are not used to evaluate toxicity. This is consistent with the CLARC database (Ecology, 2009). In addition, the reference concentration of 0.7 (mg/m^3) published by HEAST (EPA 540-R-97-036) is used to evaluate noncarcinogenic inhalation risk. If the revised EPA value were used to calculate the MTCA ("Unrestricted Land Use Soil Cleanup Standards" [WAC 173-340-740]) level, the soil concentration would decrease from 16,000 to 175 mg/kg. If the revised EPA value were used to calculate the MTCA ("Cleanup Standards to Protect Air Quality" [WAC 173-340-750]) soil cleanup level, the soil concentration would decrease from 573 to 2.8 mg/kg. This is also consistent with the CLARC database (Ecology, 2009).

The analyte-specific toxicity values, decay constants, and half-life presented in Table G-11 (Appendix G) are determined using the recommended values from the HEAST Radionuclides Table.

6.2.5 Risk Characterization

The risk characterization step is completed through the comparison of the EPC to the PRG. This comparison step is used to determine whether the post-remediation soil concentrations are protective of human health. It is also used to determine whether current material concentrations have the potential to exceed an HI greater than 1 or the upper end of the NCP (40 CFR 300) risk range for cumulative carcinogenic site risk to an individual based on RME for both current and future land use.

Although this risk assessment produces numerical estimates of risk, it should be recognized that these numbers might not predict actual health outcomes because they are based largely on hypothetical assumptions. Their purpose is to provide a frame of reference for risk management decision making. Interpretation of the risk estimates provided should consider the nature and weight of evidence supporting these estimates, as well as the magnitude of uncertainty surrounding them.

For the purpose of this risk characterization step, the potential for unacceptable human health risk is identified using the following risk thresholds:

- ELCR values are compared to the "target range" of 10^{-6} to 10^{-4} that is generally used by regulatory agencies. MTCA (WAC 173-340) states that cancer risks resulting from multiple hazardous substances should not exceed 1×10^{-5} for unrestricted land use. ELCR values within or exceeding this target range require a risk management decision that includes evaluating site-specific characteristics and exposure scenario factors to assess whether remedial action is warranted.

- An HI (the sum of the ratios of the chemical intake to the RfDs for all COPCs) greater than 1 indicates that some potential exists for adverse noncancer health effects associated with exposure to the COPCs.

6.2.5.1 Cancer Risk Estimation Method

To estimate the cancer risks from exposure to an individual nonradiological carcinogen from all exposure routes considered, the following equation is used:

$$Risk_I = \frac{EPC_{soil}}{PRG_{carcinogen}} \times TR$$

where:

$Risk_I$ = ELCR for individual chemical or radioisotope (unitless)

EPC_{soil} = EPC in soil (µg/kg or pCi/g)

$PRG_{carcinogen}$ = Soil PRG based on 10^{-6} carcinogenic effect for chemical (µg/kg) or 10^{-4} carcinogenic effect for radioisotope (pCi/g)

TR = Target ELCR of 10^{-6} for individual hazardous substance or 10^{-4} for individual radioisotope

To estimate the cancer risks from exposure to multiple carcinogens from all exposure routes considered, the following equation is used. The following equation is consistent with that published in "Regional Screening Values for Chemical Contaminants at Superfund Sites" (2009a).

$$Risk_T = \sum_i \frac{EPC_{soil}}{PRG_{carcinogen}} \times TR$$

where:

$Risk_T$ = Total ELCR for all chemicals and radioisotopes

EPC_{soil} = EPC in soil (µg/kg or pCi/g)

$PRG_{carcinogen}$ = Soil PRG based on 10^{-6} carcinogenic effect for chemical (µg/kg) or 10^{-4} carcinogenic effect for radioisotope (pCi/g)

TR = Target ELCR of 10^{-6} for individual hazardous substance or 10^{-4} for individual radioisotope

i = The sum of the ratios for the ith chemical

6.2.5.2 Noncancer Risk Estimation Method

For noncancer effects, the likelihood that a receptor will develop an adverse effect is estimated by comparing the predicted level of exposure for a particular chemical with the highest level of exposure that is considered protective (that is, its RfD). The ratio of the chronic daily intake divided by RfD is the HQ. To estimate the HQ from all exposure routes considered for an individual hazardous substance, the following equation is used:

$$HQ = \frac{EPC_{soil}}{PRG_{noncarcinogen}}$$

where:

HQ = HQ for individual chemical

EPC_{soil} = EPC in soil (µg/kg)

$PRG_{noncarcinogen}$ = Preliminary remediation goal based on HQ=1 noncarcinogenic effects (µg/kg)

To estimate the HI from all exposure routes considered for multiple hazardous substances, the following equation is used. The following equation is consistent with that published in Regional Screening Levels (2009a).

$$HI_T = \sum_i \frac{EPC_{soil}}{PRG_{noncarcinogen}}$$

where:

HI_T = Total HI for all chemicals

EPC_{soil} = Exposure point concentration in soil (µg/kg)

$PRG_{noncarcinogen}$ = Preliminary remediation goal based on HQ=1 noncarcinogenic effects (µg/kg)

i = The sum of the ratios for the ith chemical

6.2.5.3 Comparisons of Lead and Arsenic to MTCA A Soil Cleanup Levels

Potential risks from lead concentrations were evaluated using a different method than what is conventionally used for other carcinogens and noncarcinogens (as described in previous sections). For direct contact pathways, the EPCs for lead were compared to the MTCA ("Tables" [WAC 173-340-900], Table 740-1), Method A soil cleanup level for Unrestricted Land Use of 250 mg/kg.

The Method A cleanup level is based on EPA's Integrated Exposure Uptake Biokinetic (IEUBK) model, which is available on the EPA website. The IEUBK model is designed to calculate the probability of blood-lead concentrations for children between 6 months and 84 months (that is, up to 7 years) of age who have been exposed to lead through various sources (for example, air, water, soil, dust, and in utero contributions from the mother) to exceed a specific blood lead concentration.

Additionally, arsenic EPCs were compared to the MTCA ("Tables" [WAC 173-340-900], Table 740-1), Method A soil cleanup level for Unrestricted Land Use of 20 mg/kg.

6.2.5.4 Consideration of Background in Risk Assessment

CERCLA Soil Background Comparisons Guidance (EPA 540-R-01-003) provides national policy considerations for application of background data in risk assessment and remedy selection. This policy recommends an approach that addresses site-specific background issues in the risk characterization. CERCLA Soil Background Comparisons Guidance (EPA 540-R-01-003) indicates the following:

- COPCs that have both release-related and background-related sources should be included in the risk assessment. When concentrations of naturally occurring elements at a site exceed risk-based screening levels, that information should be discussed qualitatively in the risk characterization.
- CERCLA Soil Background Comparisons Guidance (EPA 540-R-01-003) defines background constituents as the following: anthropogenic—natural and artificial substances present in the environment as a result of human activities (not specifically related to the CERCLA release in question), and naturally occurring-substances present in the environment in forms that have not been influenced by human activity.

6.2.5.4.1 Sources of Background Concentrations

The 90th percentile and maximum background concentrations for the Hanford Site have been developed for both inorganic chemicals and radionuclides and are considered representative of both naturally occurring and anthropogenic substances. The maximum inorganic background concentrations used in this

evaluation are identified as the "overall maximum concentrations" in the Non-Rad Soil Background document (DOE/RL-92-24), Summary Table 1, and the 90th percentile inorganic background concentrations are identified as the "lognormal distribution 90th percentiles" in the Non-Rad Soil Background document (DOE/RL-92-24), Summary Table 2. The exceptions to this are described in the following paragraph. Two types of sampling were conducted to determine the inorganic background values: systematic random sampling, and judgment sampling. The overall maximum concentrations were determined by considering the analytical results from both systematic random samples and judgmental samples. The 90th percentile values were calculated using the analytical results from the systematic random samples only.

The Hanford Site background values for antimony, boron, cadmium, lithium, mercury, molybdenum, selenium, silver, and thallium are documented in *Soil Background Data for Interim Use at the Hanford Site* (ECF-HANFORD-11-0038). Boron was not analyzed for in the Non-Rad Soil Background document (DOE/RL-92-24) and the analytical data associated with the remaining analytes in the Non-Rad Soil Background document (DOE/RL-92-24) are considered unusable for statistical analyses because of elevated MDLs. The background concentration values documented in *Soil Background Data for Interim Use at the Hanford Site* (ECF-HANFORD-11-0038) reference *A Review of Metal Concentrations Measured in Surface Soil Samples Collected On and Around the Hanford Site* (PNNL-18577), hereinafter called Review of Metal Concentrations. The ECF documents a review of the datasets from the Non-Rad Soil Background document (DOE/RL-92-24) and Review of Metal Concentrations (PNNL-18577), which indicates the data are comparable and issues associated with elevated detection limits were eliminated as a result of improvements in analytical methods used for Review of Metal Concentrations (PNNL-18577). It is noted that *Soil Background Data for Interim Use at the Hanford Site* (ECF-HANFORD-11-0038) recalculates the percentile values based on using a nonparametric (Kaplan-Meier) method, consistent with the methodology used in the Non-Rad Soil Background document (DOE/RL-92-24). Review of Metal Concentrations (PNNL-18577) calculated the 90th percentile values based on an assumption of normally distributed data.

The background concentration values documented in *Soil Background Data for Interim Use at the Hanford Site* (ECF-HANFORD-11-0038) for selenium reference *Natural Background Soil Metals Concentrations in Washington State* (Ecology Publication 94-115) because neither the Non-Rad Soil Background document (DOE/RL-92-24) nor Review of Metal Concentrations (PNNL-18577) had adequate analytical results.

Radionuclide background values (lognormal 90th percentile and maximum) are identified in the Rad Soil Background document (DOE/RL-96-12), Table 5-1. The background values for naturally occurring radionuclides were determined primarily by analyzing a subset of the inorganic systematic random samples from the vadose zone (upper 30 cm [76 in.] of the soil column). The background values for the anthropogenic radionuclides were determined from analytical results from surface sampling (upper 2.5 cm [1 in.] of the soil column).

The composition of background samples described in the Non-Rad Soil Background document (DOE/RL-92-24), Rad Soil Background document (DOE/RL-96-12), and Review of Metal Concentrations (PNNL-18577) is representative of the sedimentary facies in the vadose zone at the 100-D/H Source OU. These background data are recommended for use in environmental restoration activities on the Hanford Site to maintain consistency between projects, and they have been peer reviewed for technical credibility. Table G-12 (Appendix G) lists the maximum and 90th percentile background concentration values for inorganic chemicals and radionuclides.

6.2.5.4.2 *Comparison of Site and Background Risk Contributions*

Understanding the contribution to risk from naturally occurring elements is important because remedial action goals are not set at concentrations below natural background levels under CERCLA. Similarly, MTCA ("Overview of Cleanup Standards" [WAC 173-340-700(6)(d)]) states that:

In some cases, cleanup levels calculated using the methods specified in this chapter are less than natural background levels or levels that can be reliably measured. In those situations, the cleanup level shall be established at a concentration equal to the practical quantitation limit or natural background concentration, whichever is higher.

CERCLA Soil Background Comparisons Guidance (EPA 540-R-01-003) states:

When background concentrations are high relative to the concentrations of released hazardous substances, pollutants, and contaminants, a comparison of site and background concentrations may help risk managers make decisions concerning appropriate remedial actions. The contribution of background concentrations to risks associated with CERCLA releases may be important for refining specific RAGs [remedial action goals] for contaminants of concern that warrant remedial action.

The 90th percentile value is used as a fixed benchmark concentration for determining which contaminants should be evaluated for purposes of background risk. To assist in risk management decisions concerning appropriate remedial actions, a comparison of background risks to risks from CERCLA releases is provided using the approach described in the following text:

EPCs from each decision unit are compared to the background value for metals and radionuclides listed in Table G-12 (Appendix G). A comparison of EPCs to the lognormal 90th percentile value for each decision unit is provided in Table G-13 (Appendix G) for the 100-D Source OU and Table G-14 (Appendix G) for the 100-H Source OU. Risk estimates are calculated as follows:

- If the EPC is less than or equal to the background value, then a risk estimate or an HQ is not calculated.
- If the EPC is greater than the background value, then a risk estimate or an HQ is calculated.
- If a background value is not available for an analyte, then a risk estimate or an HQ is calculated.
- The total ELCR is summed for all analytes with EPCs greater than their background value.
- The HI is summed for all analytes with EPCs greater than their respective background value.

6.2.5.5 *Summary of Risk Estimates by Exposure Scenario*

This section summarizes the risk estimates for each of the exposure scenarios considered for the 100-D/H Source OU.

6.2.5.5.1 *Residential Scenario*

PRGs developed for the Residential scenario are the numeric values that represent the RAOs presented in Chapter 8. PRGs are established to help determine the need for remedial action at unremediated waste sites. The PRGs are also used to compare EPCs to the RBSLs in this soil risk assessment that will be used to help determine whether additional remedial action is necessary for waste sites where remediation has been completed, and whether the goals and objectives of the interim action RODs have been met, as demonstrated by verification sampling and analysis. A complete description of the Residential exposure scenario is provided in Section 6.2.3.3.1.

For completeness in analysis, all risk estimates for each remediated waste site decision unit are provided in Appendix G. The risk estimates, which include all COPCs regardless of their EPCs relative to background concentrations, are presented in Tables G-15 through G-23 (100-D Residential scenario) and Tables G-34 to G-42 (100-H Residential scenario).

Appendix G also includes risk estimates for each remediated waste site decision unit, which include only those COPCs with EPCs greater than background values or that do not have a background value. These risk estimates are presented in Tables G-24 to G-33 (100-D Residential scenario) and Tables G-43 to G-52 (100-H Residential scenario). Only these risk estimates without background contributions are summarized and discussed in the risk characterization because this information is used for decisions concerning appropriate remedial actions.

100-D Source OU. Risk estimates were calculated for each decision unit within a remediated waste site including shallow vadose zone material, deep vadose zone material, overburden material, and staging pile area footprint material. The results without background contribution for the Residential scenario are presented in Tables G-24 to G-26 (Appendix G).

An overall summary of the total risk estimates and noncancer hazards (if applicable) for the residential scenario from each of the remediated waste sites is provided in Tables 6-13 and Table 6-14 for shallow zone material, Table 6-15 and Table 6-16 for overburden material, Table 6-17 and Table 6-18 for staging piles, and Table 6-19 for the deep zone. These tables list the OU that each remediated waste site resides in, the reclassification status, the remediated waste site, the associated waste site (if applicable), the decision unit reported with an exceedance (if applicable), the total ELCR and the risk driver and percent contribution (if applicable), and the hazard index and the noncancer hazard driver and percent contribution (if applicable).

Shallow Zone. A total of 92 remediated waste sites are reported with CVP/RSVP data associated with the shallow zone in the 100-D Source OU. The following lists the sample designs that were applied to the remediated waste sites evaluated:

- Twenty remediated waste sites were sampled using a focused sampling design.
- Forty-seven remediated waste sites were sampled using a statistical sampling design (with three sites having two statistically distinct decision units).
- Twenty-five remediated waste sites were sampled using both a statistical and a focused sampling design (with two sites having one focused and three statistically distinct decision units; three sites having one focused and two statistically distinct decision units and two sites with two focused decision units and one statistical decision unit).

Radionuclide Results. As presented in Table 6-13, the potential total ELCR is greater than or equal to the upper risk threshold of 1×10^{-4} at nine remediated waste sites, is within the target risk range of 10^{-4} to 10^{-6} at 27 remediated waste sites, and is less than the lower risk threshold of 1×10^{-6} at three remediated waste sites. Risks were not reported at 21 remediated waste sites because radiological COPC concentrations were less than background. Radiological COPCs were not reported at 32 remediated waste sites.

Table 6-13. Summary of Total Risks from Radionuclides for the 100-DR-1 and 100-DR-2 Shallow Zone Waste Sites for the Residential Scenario

Classification Status	Remediated Waste Site	Associated Waste Sites[a]	Decision Unit with Exceedance	Total ELCR[b]	Risk Driver and % Contribution
100-DR-1 OU					
Interim Closed Out	116-DR-9	100-D-25	Shallow	2.6×10^{-4} $< 1 \times 10^{-4}$ (2038)	Cesium-137 (2.3×10^{-4} – 89%)
	100-D-42, 100-D-43, 100-D-45	--	Shallow Focused	1.2×10^{-4} $< 1 \times 10^{-4}$ (2012)	Cobalt-60 (3.9×10^{-5} – 34%) Nickel-63 (7.6×10^{-5} – 66%)
	100-D-48:3	100-D-5 100-D-6	Shallow	1.2×10^{-4} $< 1 \times 10^{-4}$ (2009)	Strontium-90 (1.2×10^{-4} – 97%)
	118-D-6:4	--	Shallow 2	1.2×10^{-4} $< 1 \times 10^{-4}$ (2022)	Cesium-137 (6.5×10^{-5} – 53%) Europium-152 (3.9×10^{-5} – 31%) Strontium-90 (1.6×10^{-5} – 13%)
	100-D-48:1	100-D-49:1 UPR-100-D-4	None	1×10^{-4} to 1×10^{-6}	None
	100-D-48:2	UPR-100-D-2 UPR-100-D-3			
	116-D-1A	100-D-46 116-D-1B			
	100-D-20 100-D-22 100-D-4 100-D-48:4 100-D-49:2 100-D-49:4 100-D-52 116-D-5 116-D-7 116-D-9 116-DR-1&2 116-DR-5 128-D-2 132-D-1 1607-D2:3 1607-D2:4	--			
	100-D-29 100-D-32 1607-D2:1	--	None	$< 1 \times 10^{-6}$	None

Table 6-13. Summary of Total Risks from Radionuclides for the 100-DR-1 and 100-DR-2 Shallow Zone Waste Sites for the Residential Scenario

Classification Status	Remediated Waste Site	Associated Waste Sites[a]	Decision Unit with Exceedance	Total ELCR[b]	Risk Driver and % Contribution
	100-D-1 100-D-21 100-D-31:1, 100-D-31:2 100-D-31:10 100-D-31:3 100-D-31:4 100-D-31:7 100-D-31:8 100-D-31:9 100-D-49:3 116-D-10 116-D-2 116-D-4 1607-D2:2	--	None	No COPCs reported above background	None
	100-D-2 100-D-31:5 100-D-31:6 100-D-56:1 100-D-56:2 100-D-61 100-D-7 100-D-9 120-D-2 126-D-2 130-D-1 1607-D4 1607-D5 628-3	--	None	No COPCs detected	None
No Action	100-D-24 100-D-3 UPR-100-D-5	--	None	No COPCs reported above background	None

Table 6-13. Summary of Total Risks from Radionuclides for the 100-DR-1 and 100-DR-2 Shallow Zone Waste Sites for the Residential Scenario

Classification Status	Remediated Waste Site	Associated Waste Sites[a]	Decision Unit with Exceedance	Total ELCR[b]	Risk Driver and % Contribution
	100-D-50:5 100-D-70 100-D-74 100-D-75:3 100-D-80:1 100-D-82 100-D-83:4 100-D-84:1 100-D-85:1 100-D-87 100-D-88 100-D-90	--	None	No COPCs detected	None
100-DR-2					
Interim Closed Out	--[c]	100-D-43	Shallow Focused	1.2×10^{-4} $< 1 \times 10^{-4}$ (2012)	Cobalt-60 (3.9×10^{-5} – 34%) Nickel-63 (7.6×10^{-5} – 66%)
	--[d]	100-D-46	None	1×10^{-4} to 1×10^{-6}	None
	118-DR-2:2	--	Shallow	2.2×10^{-4} $< 1 \times 10^{-4}$ (>100,000 years)	Technetium-99 (1.6×10^{-4} – 74%) Strontium-90 (3.9×10^{-5} – 18%)
	116-D-8		Shallow Focused 2	1.7×10^{-4} $< 1 \times 10^{-4}$ (2035)	Cesium-137 (1.7×10^{-4} –100%)
	100-D-47		Shallow Focused	1.0×10^{-4} $< 1 \times 10^{-4}$ (2009)	Europium-152 (4.2×10^{-5} – 40%) Strontium-90 (5.2×10^{-5} – 50%)
	116-DR-10 116-DR-6 116-DR-7 116-DR-8 118-D-1 118-D-4 118-D-5 118-DR-1	--	None	1×10^{-4} to 1×10^{-6}	None
	100-D-13	--	None	No COPCs reported above background	None
	116-DR-4				

Table 6-13. Summary of Total Risks from Radionuclides for the 100-DR-1 and 100-DR-2 Shallow Zone Waste Sites for the Residential Scenario

Classification Status	Remediated Waste Site	Associated Waste Sites[a]	Decision Unit with Exceedance	Total ELCR[b]	Risk Driver and % Contribution
	122-DR-1:2	100-D-23 100-D-53 100-D-54 100-D-64			
	100-D-12 100-D-15 100-D-28:1 1607-D1 600-30		None	No COPCs detected	None
No Action	100-D-94	--	None	No COPCs detected	None

Note:

Results summarized from Table G-24, Residential Scenario Risk Estimates and Noncancer Hazards for the 100-D Source OU Waste Site Decision Units Without Background Contribution (Appendix G).

The following three waste sites do not report shallow zone sample data: 100-D-18, 100-D-19, and 116-D-6.

a. Associated waste sites are those sites for which remediation and closeout documentation were consolidated with another remediated waste site.

b. Total ELCR represents risk contributions from radiological COPCs.

c. Remediated waste site 100-D-43 (100-DR-2 OU) is associated with remediated waste sites 100-D-42 and 100-D-45 (100-DR-1 OU).

d. Remediated waste site 100-D-46 (100-DR-2 OU) is associated with remediated waste site 116-D-1A (100-DR-1).

Table 6-14. Summary of Total Carcinogenic Risks and Noncancer Hazards from Direct Contact for the 100-DR-1 and 100-DR-2 Shallow Zone Waste Sites for the Residential Scenario

Classification Status	Remediated Waste Site	Associated Waste Sites[a]	Decision Unit with Exceedance	Total ELCR[b]	Risk Driver and % Contribution	Hazard Index	Noncancer Hazard Driver and % Contribution
100-DR-1 OU							
Interim Closed Out	100-D-31:4	--	Shallow	1.7×10^{-5}	Benzo(a)pyrene (1.4×10^{-5} – 81%) Benzo(b)fluoranthene (1.2×10^{-6} – 7%)	<1	None
	1607-D5		Shallow	5.2×10^{-6}	Benzo(a)anthracene (1.4×10^{-6} – 28%) Benzo(a)pyrene (3.0×10^{-6} – 57%)		

Table 6-14. Summary of Total Carcinogenic Risks and Noncancer Hazards from Direct Contact for the 100-DR-1 and 100-DR-2 Shallow Zone Waste Sites for the Residential Scenario

Classification Status	Remediated Waste Site	Associated Waste Sites[a]	Decision Unit with Exceedance	Total ELCR[b]	Risk Driver and % Contribution	Hazard Index	Noncancer Hazard Driver and % Contribution
	1607-D2:2		Shallow	1.1×10^{-6}	Aroclor-1254 (8.9×10^{-7} – 74%) Aroclor-1260 (3.1×10^{-7} – 26%)		
	100-D-1 100-D-31:1, 100-D-31:2 100-D-31:10 100-D-31:3 100-D-31:6 100-D-31:7 100-D-31:8 100-D-31:9 100-D-4 100-D-42, 100-D-43, 100-D-45 100-D-49:4 100-D-61 100-D-7 100-D-9 116-D-5 116-DR-5 118-D-6:4 126-D-2 128-D-2 130-D-1 132-D-1 1607-D2:3 1607-D4 628-3	--	None	$< 1 \times 10^{-6}$	None	<1	None

Table 6-14. Summary of Total Carcinogenic Risks and Noncancer Hazards from Direct Contact for the 100-DR-1 and 100-DR-2 Shallow Zone Waste Sites for the Residential Scenario

Classification Status	Remediated Waste Site	Associated Waste Sites[a]	Decision Unit with Exceedance	Total ELCR[b]	Risk Driver and % Contribution	Hazard Index	Noncancer Hazard Driver and % Contribution
	100-D-2 100-D-29 100-D-31:5 100-D-32 100-D-56:1 100-D-56:2 116-D-10 120-D-2	--	None	No COPCs reported above background	None	<1	None
	1607-D2:1					No COPCs reported above background	None
	100-D-20	--	None	No COPCs detected	None	No COPCs reported above background	None
	100-D-21					No COPCs Reported	None
	100-D-22					No COPCs reported above background	None
	100-D-48:1	100-D-49:1 UPR-100-D-4				<1	None
	100-D-48:2	UPR-100-D-2 UPR-100-D-3				No COPCs reported above background	None
	100-D-48:3	100-D-5 100-D-6				<1	None
	100-D-48:4	--					

Table 6-14. Summary of Total Carcinogenic Risks and Noncancer Hazards from Direct Contact for the 100-DR-1 and 100-DR-2 Shallow Zone Waste Sites for the Residential Scenario

Classification Status	Remediated Waste Site	Associated Waste Sites[a]	Decision Unit with Exceedance	Total ELCR[b]	Risk Driver and % Contribution	Hazard Index	Noncancer Hazard Driver and % Contribution
	100-D-49:2					No COPCs reported above background	None
	100-D-49:3						
	100-D-52						
	116-D-1A	100-D-46 116-D-1B				<1	None
	116-D-2	--				No COPCs reported above background	None
	116-D-4						
	116-D-7					<1	None
	116-D-9					No COPCs reported above background	None
	116-DR-1&2						
	116-DR-9	100-D-25				<1	None
	1607-D2:4	--				No COPCs reported above background	None
No Action	100-D-87	--	Shallow Focused	1.1×10^{-5}	Arsenic (1.1×10^{-5} – 95%)	<1	None
	100-D-24 100-D-74	--	None	$< 1 \times 10^{-6}$	None	<1	None
	100-D-82					No COPCs reported above background	None
	100-D-84:1 100-D-88 100-D-70					<1	None

Table 6-14. Summary of Total Carcinogenic Risks and Noncancer Hazards from Direct Contact for the 100-DR-1 and 100-DR-2 Shallow Zone Waste Sites for the Residential Scenario

Classification Status	Remediated Waste Site	Associated Waste Sites[a]	Decision Unit with Exceedance	Total ELCR[b]	Risk Driver and % Contribution	Hazard Index	Noncancer Hazard Driver and % Contribution
	100-D-50:5	--	None	No COPCs reported above background	None	No COPCs reported above background	None
	100-D-75:3 100-D-83:4					<1	None
	100-D-85:1					No COPCs reported above background	None
	UPR-100-D-5					<1	None
	100-D-3	--	None	No COPCs detected	None	No COPCs reported above background	None
	100-D-80:1 100-D-90					No COPCs detected	None
100-DR-2							
Interim Closed Out	--[c]	100-D-43	None	$< 1 \times 10^{-6}$	None	<1	None
	--[d]	100-D-46	None	No COPCs detected	None	<1	None
	100-D-13 100-D-15 100-D-28:1 116-D-8 116-DR-10 116-DR-8 118-DR-1 118-DR-2:2	--	None	$< 1 \times 10^{-6}$	None	<1	None
	122-DR-1:2	100-D-23 100-D-53 100-D-54 100-D-64					

Table 6-14. Summary of Total Carcinogenic Risks and Noncancer Hazards from Direct Contact for the 100-DR-1 and 100-DR-2 Shallow Zone Waste Sites for the Residential Scenario

Classification Status	Remediated Waste Site	Associated Waste Sites[a]	Decision Unit with Exceedance	Total ELCR[b]	Risk Driver and % Contribution	Hazard Index	Noncancer Hazard Driver and % Contribution
	1607-D1 600-30	--					
	100-D-47 118-D-1 118-D-4 118-D-5	--	None	No COPCs reported above background	None	<1	None
	100-D-12		None	No COPCs detected	None	No COPCs detected	None
	116-DR-4 116-DR-6 116-DR-7					No COPCs reported above background	None
No Action	100-D-94	--	None	No COPCs reported above background	None	<1	None

Note:

Results summarized from Table G-24, Residential Scenario Risk Estimates and Noncancer Hazards for the 100-D Source OU Waste Site Decision Units Without Background Contribution (Appendix G).

The following three waste sites do not report shallow zone sample data: 100-D-18, 100-D-19, and 116-D-6.

a. Associated waste sites are those sites for which remediation and closeout documentation were consolidated with another remediated waste site.

b. Total ELCR represents risk contributions from nonradiological carcinogenic COPCs.

c. Remediated waste site 100-D-43 (100-DR-2 OU) is associated with remediated waste sites 100-D-42 and 100-D-45 (100-DR-1 OU).

d. Remediated waste site 100-D-46 (100-DR-2 OU) is associated with remediated waste site 116-D-1A (100-DR-1).

Table 6-15. Summary of Total Risks from Radionuclides for the 100-DR-1 and 100-DR-2 Overburden Material for the Residential Scenario

Classification Status	Remediated Waste Site	Associated Waste Sites[a]	Decision Unit with Exceedance	Total ELCR[b]	Risk Driver and % Contribution
100-DR-1 OU					
Interim Closed Out	100-D-48:3	100-D-5 100-D-6	None	1×10^{-4} to 1×10^{-6}	None
	100-D-48:2	UPR-100-D-2 UPR-100-D-3	None		
	100-D-31:10 100-D-31:3, 100-D-31:4 100-D-31:8 100-D-4 116-D-5 116-DR-5 1607-D2:3	--	None		
	100-D-29	--	None	$< 1 \times 10^{-6}$	
	100-D-31:1, 100-D-31:2 100-D-31:7 100-D-31:9 100-D-32 126-D-2	--	None	No COPCs reported above background	
	100-D-31:5 100-D-31:6 100-D-42, 100-D-43, 100-D-45 100-D-56:1 100-D-56:2	--	None	No COPCs reported	
No Action	UPR-100-D-5	--	None	1×10^{-4} to 1×10^{-6}	
100-DR-2					
Interim Closed Out	100-D-47 116-DR-8 118-D-5	--	None	1×10^{-4} to 1×10^{-6}	None
	118-D-I	--	None	$< 1 \times 10^{-6}$	
	116-DR-I0 118-D-4	--	None	No COPCs reported above background	
	100-D-28:1		None	No COPCs reported	

Table 6-15. Summary of Total Risks from Radionuclides for the 100-DR-1 and 100-DR-2 Overburden Material for the Residential Scenario

Classification Status	Remediated Waste Site	Associated Waste Sites[a]	Decision Unit with Exceedance	Total ELCR[b]	Risk Driver and % Contribution

Note:

Results summarized from Table G-24, Residential Scenario Risk Estimates and Noncancer Hazards for the 100-D Source OU Waste Site Decision Units Without Background Contribution (Appendix G).

Remediated waste site 100-D-43 (100-DR-2 OU) is associated with remediated waste sites 100-D-42 and 100-D-45 (100-DR-1 OU).

Remediated waste site 100-D-46 (100-DR-2 OU) is associated with remediated waste site 116-D-1A (100-DR-1).

a. Associated waste sites are those sites for which remediation and closeout documentation were consolidated with another remediated waste site.

b. Total ELCR represents risk contributions from radiological COPCs.

Table 6-16. Summary of Total Carcinogenic Risks and Noncancer Hazards from Direct Contact for the 100-DR-1 and 100-DR-2 Overburden for the Residential Scenario

Classification Status	Remediated Waste Site	Associated Waste Sites[a]	Decision Unit with Exceedance	Total ELCR[b]	Risk Driver and % Contribution	Hazard Index	Noncancer Hazard Driver and % Contribution
100-DR-1 OU							
Interim Closed Out	100-D-31:3, 100-D-31:4	--	None	1.2×10^{-6}	None	<1	None
Interim Closed Out	100-D-31:1, 100-D-31:2 100-D-31:10 100-D-31:6 100-D-31:7 100-D-31:8 100-D-31:9 116-D-5 116-DR-5 126-D-2	--	None	$< 1 \times 10^{-6}$	None	<1	None

Table 6-16. Summary of Total Carcinogenic Risks and Noncancer Hazards from Direct Contact for the 100-DR-1 and 100-DR-2 Overburden for the Residential Scenario

Classification Status	Remediated Waste Site	Associated Waste Sites[a]	Decision Unit with Exceedance	Total ELCR[b]	Risk Driver and % Contribution	Hazard Index	Noncancer Hazard Driver and % Contribution
	100-D-48:2	UPR-100-D-2 UPR-100-D-3	None	No COPCs reported	None	<1	None
	100-D-48:3	100-D-5 100-D-6				No COPCs reported above background	
	1607-D2:3	--					
Interim Closed Out	100-D-29 100-D-31:5 100-D-32 100-D-42, 100-D-43, 100-D-45 100-D-56:1 100-D-56:2	--	None	No COPCs reported above background	None	<1	None
	100-D-4	--	None	No COPCs reported	None	<1	None
No Action	UPR-100-D-5	--	None	$< 1 \times 10^{-6}$	None	<1	None
100-DR-2							
Interim Closed Out	100-D-28:1	--	None	1.9×10^{-6}	None	<1	None

Table 6-16. Summary of Total Carcinogenic Risks and Noncancer Hazards from Direct Contact for the 100-DR-1 and 100-DR-2 Overburden for the Residential Scenario

Classification Status	Remediated Waste Site	Associated Waste Sites[a]	Decision Unit with Exceedance	Total ELCR[b]	Risk Driver and % Contribution	Hazard Index	Noncancer Hazard Driver and % Contribution
	116-DR-10 116-DR-8 118-D-1	--		$< 1 \times 10^{-6}$	$< 1 \times 10^{-6}$		
	100-D-47 118-D-4 118-D-5	--		No COPCs reported above background	None		

Note:

Results summarized from Table G-24, Residential Scenario Risk Estimates and Noncancer Hazards for the 100-D Source OU Waste Site Decision Units Without Background Contribution (Appendix G).

a. Associated waste sites are those sites for which remediation and closeout documentation were consolidated with another remediated waste site.

b. Total ELCR represents risk contributions from nonradiological carcinogenic COPCs.

Table 6-17. Summary of Total Risks from Radionuclides for the 100-DR-1 and 100-DR-2 Staging Piles for the Residential Scenario

Classification Status	Remediated Waste Site	Associated Waste Sites[a]	Decision Unit with Exceedance	Total ELCR[b]	Risk Driver and % Contribution
100-DR-1 OU					
Interim Closed Out	116-D-10	--	None	1×10^{-4} to 1×10^{-6}	None
	116-DR-5 132-D-1			$< 1 \times 10^{-6}$	
	100-D-7 116-D-5			No COPCs reported above background	
	100-D-56:2 130-D-1 628-3			No COPCs reported	

Table 6-17. Summary of Total Risks from Radionuclides for the 100-DR-1 and 100-DR-2 Staging Piles for the Residential Scenario

Classification Status	Remediated Waste Site	Associated Waste Sites[a]	Decision Unit with Exceedance	Total ELCR[b]	Risk Driver and % Contribution
100-DR-2					
Interim Closed Out	118-D-1	--	None	$< 1 \times 10^{-6}$	None
	100-D-28:1 1607-D1			No COPCs reported	

Note:

Results summarized from Table G-24, Residential Scenario Risk Estimates and Noncancer Hazards for the 100-D Source OU Waste Site Decision Units Without Background Contribution (Appendix G).

a. Associated waste sites are those sites for which remediation and closeout documentation were consolidated with another remediated waste site.

b. Total ELCR represents risk contributions from radiological COPCs.

Table 6-18. Summary of Total Carcinogenic Risks and Noncancer Hazards from Direct Contact for the 100-DR-1 and 100-DR-2 Staging Piles for the Residential Scenario

Classification Status	Remediated Waste Site	Associated Waste Sites[a]	Decision Unit with Exceedance	Total ELCR[b]	Risk Driver and % Contribution	Hazard Index	Noncancer Hazard Driver and % Contribution
100-DR-1 OU							
Interim Closed Out	132-D-1	--	None	1.8×10^{-6}	Benzo(a)pyrene (1.3×10^{-6} – 69%)	<1	None
	100-D-56:2 100-D-7 116-D-5 116-DR-5 130-D-1 628-3	--	None	$< 1 \times 10^{-6}$	None	<1	None
	116-D-10	--	None	No COPCs reported above background	None	<1	None

Table 6-18. Summary of Total Carcinogenic Risks and Noncancer Hazards from Direct Contact for the 100-DR-1 and 100-DR-2 Staging Piles for the Residential Scenario

Classification Status	Remediated Waste Site	Associated Waste Sites[a]	Decision Unit with Exceedance	Total ELCR[b]	Risk Driver and % Contribution	Hazard Index	Noncancer Hazard Driver and % Contribution
100-DR-2							
Interim Closed Out	100-D-28:1 118-D-1 1607-D1	--	None	$< 1 \times 10^{-6}$	None	<1	None

Note:

Results summarized from Table G-24, Residential Scenario Risk Estimates and Noncancer Hazards for the 100-D Source OU Waste Site Decision Units Without Background Contribution (Appendix G).

a. Associated waste sites are those sites for which remediation and closeout documentation were consolidated with another remediated waste site.

b. Total ELCR represents risk contributions from nonradiological carcinogenic COPCs.

Table 6-19. Summary of Total Risks from Radionuclides for the 100-DR-1 and 100-DR-2 Deep Zone Waste Sites for the Residential Scenario

Classification Status	Remediated Waste Site	Associated Waste Sites[a]	Decision Unit with Exceedance	Total ELCR[b]	Risk Driver and % Contribution
100-DR-1					
Interim Closed Out	116-D-1A	100-D-46 116-D-1B	Deep	1.6×10^{-2} $< 1.0 \times 10^{-4}$ (2203)	Cesium-137 (9.3×10^{-3} – 58%) Cobalt-60 (2.5×10^{-4} – 1.6%) Europium-152 (5.3×10^{-3} – 33%) Europium-154 (3.6×10^{-4} – 2.2%) Strontium-90 (9.3×10^{-4} – 5.8%)
	116-D-7	--	Deep	9.7×10^{-3} $< 1.0 \times 10^{-4}$ (2125)	Cesium-137 (7.1×10^{-4} – 7.3%) Cobalt-60 (7.4×10^{-4} – 7.6%) Europium-152 (7.3×10^{-3} – 75%) Europium-154 (8.3×10^{-4} – 8.6%) Nickel-63 (1.2×10^{-4} – 1.2%)

Table 6-19. Summary of Total Risks from Radionuclides for the 100-DR-1 and 100-DR-2 Deep Zone Waste Sites for the Residential Scenario

Classification Status	Remediated Waste Site	Associated Waste Sites[a]	Decision Unit with Exceedance	Total ELCR[b]	Risk Driver and % Contribution
	116-DR-1&2		Deep	6.7×10^{-3} $< 1.0 \times 10^{-4}$ (2148)	Cesium-137 (1.7×10^{-3} – 25%) Cobalt-60 (1.1×10^{-4} – 1.7%) Europium-152 (3.4×10^{-3} – 51%) Europium-154 (1.8×10^{-4} – 2.7%) Strontium-90 (1.3×10^{-3} – 19%)
	118-D-6:4		Deep Focused	2.5×10^{-3} $< 1.0 \times 10^{-4}$ (2143)	Cesium-137 (1.9×10^{-3} – 77%) Europium-152 (4.7×10^{-4} – 19%)
	100-D-48:1	100-D-49:1 UPR-100-D-4	Deep	2.8×10^{-3} $< 1.0 \times 10^{-4}$ (2093)	Cesium-137 (6.8×10^{-4} – 24%) Cobalt-60 (2.6×10^{-4} – 9.2%) Europium-152 (1.8×10^{-3} – 62%) Europium-154 (1.1×10^{-4} – 3.9%)
	100-D-49:2	--	Deep	2.5×10^{-3} $< 1.0 \times 10^{-4}$ (2117)	Cesium-137 (1.4×10^{-3} – 57%) Cobalt-60 (2.5×10^{-4} – 10%) Europium-152 (6.7×10^{-4} – 27%)
	116-DR-9	100-D-25	Deep	1.2×10^{-3} $< 1.0 \times 10^{-4}$ (2064)	Cesium-137 (2.4×10^{-4} – 21%) Europium-152 (7.0×10^{-4} – 61%)
	100-D-18	--	Deep	7.5×10^{-4} $< 1.0 \times 10^{-4}$ (2066)	Cesium-137 (4.1×10^{-4} – 54%) Europium-152 (2.7×10^{-4} – 36%)
	100-D-49:4		Deep	3.3×10^{-4} $< 1.0 \times 10^{-4}$ (2027)	Cesium-137 (4.8×10^{-5} – 14%) Europium-152 (2.3×10^{-4} – 69%)
	100-D-48:2	UPR-100-D-2 UPR-100-D-3	Deep	3.0×10^{-4} $< 1.0 \times 10^{-4}$ (2034)	Cesium-137 (1.1×10^{-4} – 36%) Europium-152 (8.8×10^{-5} – 29%) Strontium-90 (7.8×10^{-5} – 26%)
	100-D-48:3	100-D-5 100-D-6	Deep	2.0×10^{-4} $< 1.0 \times 10^{-4}$ (2028)	Cesium-137 (6.1×10^{-5} – 30%) Strontium-90 (1.2×10^{-4} – 60%)
	100-D-19	--	Deep Focused	1.4×10^{-4} $< 1.0 \times 10^{-4}$ (2042)	Nickel-63 (1.3×10^{-4} – >99%)
	116-D-5 116-D-6	--	None	1×10^{-4} to 1×10^{-6}	None
100-DR-2					
Interim Closed Out	--[c]	100-D-46	Deep	1.6×10^{-2} $< 1.0 \times 10^{-4}$ (2203)	Cesium-137 (9.3×10^{-3} – 58%) Cobalt-60 (2.5×10^{-4} – 1.6%) Europium-152 (5.3×10^{-3} – 33%) Europium-154 (3.6×10^{-4} – 2.2%) Strontium-90 (9.3×10^{-4} – 5.8%)

Table 6-19. Summary of Total Risks from Radionuclides for the 100-DR-1 and 100-DR-2 Deep Zone Waste Sites for the Residential Scenario

Classification Status	Remediated Waste Site	Associated Waste Sites[a]	Decision Unit with Exceedance	Total ELCR[b]	Risk Driver and % Contribution
	116-DR-6	--	Deep	6.3×10^{-4} $< 1.0 \times 10^{-4}$ (2048)	Cesium-137 (1.5×10^{-4} – 23%) Europium-152 (3.9×10^{-4} – 62%)
	118-DR-2:2		Deep	6.8×10^{-4} $< 1.0 \times 10^{-4}$ (2140)	Cesium-137 (2.7×10^{-4} – 37%) Cobalt-60 (1.4×10^{-4} – 19%) Europium-152 (1.6×10^{-4} – 22%)
	118-D-1	--	Deep	$< 1 \times 10^{-6}$	None
	122-DR-1:2	100-D-23 100-D-53 100-D-54 100-D-64	None	No COPCs reported above background	None

Note:

Results summarized from Table G-24, Residential Scenario Risk Estimates and Noncancer Hazards for the 100-D Source OU Waste Site Decision Units Without Background Contribution (Appendix G).

a. Associated waste sites are those sites for which remediation and closeout documentation were consolidated with another remediated waste site.

b. Total ELCR represents risk contributions from radiological COPCs.

c. Remediated waste site 100-D-46 (100-DR-2 OU) is associated with remediated waste site 116-D-1A (100-DR-1).

Nine remediated waste sites report concentrations of Hanford Site-related COPCs that are equal to or exceed the upper range of the target threshold for the Residential scenario. The cancer risk levels for the Residential scenario are as follows:

- The 116-DR-9 waste site (shallow decision unit) reports a total ELCR of 2.6×10^{-4}. The primary contributor to risk is cesium-137 (2.3×10^{-4}; 89 percent contribution). The EPC of cesium-137 is 10 pCi/g, which is greater than the residential RBSL of 4.4 pCi/g and is also greater than the direct exposure remedial action goal of 6.2 pCi/g, published in the 100 Area RDR/RAWP (DOE/RL-96-17). Activities of all radionuclides will decay to a total ELCR of less than 1.0×10^{-4} by year 2038.

- The 118-DR-2:2 waste site (shallow decision unit) reports a total ELCR of 2.2×10^{-4}. The primary contributors to risk include technetium-99 (1.6×10^{-4}; 74 percent contribution) and strontium-90 (3.9×10^{-5}; 18 percent contribution). The EPC of technetium-99 is 2.4 pCi/g, which is greater than the residential RBSL of 1.5 pCi/g. However, the EPC is less than the current direct exposure remedial action goal of 5.8 pCi/g published in the 100 Area RDR/RAWP (DOE/RL-96-17). Activities of all radionuclides will not decay to a total ELCR of less than 1.0×10^{-4} within a reasonable timeframe as a result of the presence of technetium-99.

- The 116-D-8 (shallow focused 2 decision unit) reports a total ELCR of 1.7×10^{-4}. The primary contributor to risk is cesium-137 (1.7×10^{-4}; 100 percent contribution). The EPC of cesium-137 is 7.6 pCi/g, which is greater than the residential RBSL of 4.4 pCi/g and is also greater the current direct exposure remedial action goal of 6.2 pCi/g published in the 100 Area RDR/RAWP (DOE/RL-96-17). Activities of all radionuclides will decay to a total ELCR of less than 1.0×10^{-4} by year 2035.

- The 100-D-42, 100-D-43, 100-D-45 waste sites (shallow focused decision unit) report a total ELCR of 1.2×10^{-4}. The primary contributors to risk include cobalt-60 (3.9×10^{-5}; 34 percent contribution) and nickel-63 (Ni-63) (7.6×10^{-5}; 66 percent contribution). Activities of all radionuclides will decay to a total ELCR of less than 1.0×10^{-4} by year 2012.

- The 100-D-48:3 waste site (shallow decision unit) reports a total ELCR of 1.2×10^{-4}. The primary contributor to risk is strontium-90 (1.2×10^{-4}; 97 percent contribution). The EPC of strontium-90 is 2.7 pCi/g, which is slightly greater than the residential RBSL of 2.3 pCi/g. However, the EPC is less than the current direct exposure remedial action goal of 4.5 pCi/g published in the 100 Area RDR/RAWP (DOE/RL-96-17). Activities of all radionuclides have decayed to a total ELCR of less than 1.0×10^{-4} in year 2009.

- The 118-D-6:4 waste site (shallow 2 decision unit) reports a total ELCR of 1.2×10^{-4}. The primary contributors to risk are cesium-137 (6.5×10^{-5}; 53 percent contribution), europium-152 (3.9×10^{-5}; 31 percent contribution), and strontium-90 (1.6×10^{-5}; 13 percent contribution). Activities of all radionuclides will decay to a total ELCR of less than 1.0×10^{-4} by year 2022.

- The 100-D-47 waste site (shallow focused decision unit) reports a total ELCR of 1.0×10^{-4}. The primary contributors to risk include europium-152 (4.2×10^{-5}; 40 percent contribution) and strontium-90 (5.2×10^{-5}; 50 percent contribution). Activities of all radionuclides have decayed to a total ELCR of less than 1.0×10^{-4} in year 2009.

Nonradiological Results (Direct Contact). As presented in Table 6-14, the potential cumulative ELCR is greater than the 1×10^{-6} for four remediated waste sites and is less than the 1×10^{-6} for 44 remediated waste sites. Risks were not reported at 19 remediated waste sites because nonradiological carcinogenic COPCs were less than background. Nonradiological carcinogenic COPCs were not reported at 25 remediated waste sites.

As presented in Table 6-14, three remediated waste sites report individual carcinogens greater than the WAC 173-340-740 acceptable cancer risk level of 1×10^{-6}, two of these three remediated waste sites are greater than the MTCA ("Human Health Risk Assessment Procedures" [WAC 173-340-708(5)]) cumulative risk threshold of 1×10^{-5}. The cancer risk levels for the residential scenario are as follows:

- The 100-D-31:4 waste site (shallow decision unit) reports a cumulative nonradiological ELCR of 1.7×10^{-5}. The primary contributors to risk include benzo[a]pyrene (1.4×10^{-5}; 81 percent contribution) and benzo[b]fluoranthene (1.2×10^{-6}; 6.8 percent contribution).
- The 1607-D5 waste site (shallow decision unit) reports a cumulative nonradiological ELCR of 5.2×10^{-6}. The primary contributors to risk include benzo[a]anthracene (1.4×10^{-6}; 28 percent contribution) and benzo[a]pyrene (3.0×10^{-6}; 57 percent contribution).
- The 100-D-87 (shallow focused decision unit) reports a total ELCR of 1.1×10^{-5}. The primary contributor to risk is arsenic (1.1×10^{-5}; 95 percent contribution).

The EPC for arsenic at 100-D-87 is 7 mg/kg and is the only contributor to risk. Although the EPC is greater than the lognormal 90th percentile value of 6.5 mg/kg, it is consistent with naturally occurring levels at the Hanford Site (see Table G-13 [Appendix G]) and considered naturally occurring and are not the result of a Hanford Site release. Additionally, the EPC for arsenic is less than the direct exposure remedial action goal of 20 mg/kg published in the 100 Area RDR/RAWP (DOE/RL-96-17).

For the 100-D-31:4 remediated waste site (shallow decision unit), the EPCs for benzo(a)pyrene (1.9 mg/kg) and benzo(b)fluoranthene (1.6 mg/kg) are greater than their PRGs. A summary of the benzo(a)pyrene and benzo(b)fluoranthene results follows:

- The EPC for benzo(a)pyrene is based on maximum detected concentration. Twelve soil samples were collected from the shallow decision unit and analyzed for benzo(a)pyrene. Benzo(a)pyrene was detected in three of 12 samples, measured concentrations range between 0.24 and 1.9 mg/kg (one result is greater than the risk based screening level of 0.14 mg/kg).
- The EPC for benzo(b)fluoranthene is based on maximum detected concentration. Twelve soil samples were collected from the shallow decision unit and analyzed for benzo(b)fluoranthene. Benzo(a)fluoranthene was detected in three of 12 samples, measured concentrations range between 0.01 and 1.6 mg/kg (one result greater than the risk based screening level of 1.4 mg/kg).

As presented in Table 6-14, the potential HI from noncancer effects from direct contact without background contribution is less than the EPA target HI of 1 and the MTCA ("Unrestricted Land Use Soil Cleanup Standards" [WAC 173-340-740]) target HI of 1 for 69 of 92 remediated waste sites. An HI was not reported for 19 remediated waste sites because COPC concentrations were less than background. Nonradiological COPCs were not reported at four remediated waste sites.

As presented in Table G-26 (Appendix G), all lead and arsenic EPCs are less than their respective Method A soil cleanup levels of 250 mg/kg and 20 mg/kg for unrestricted land use.

Nonradiological Results (Inhalation). As presented in Table G-28 (Appendix G), the potential cumulative ELCR for the inhalation pathway from all nonradiological carcinogenic COPCs without background contribution ranges from 3.3×10^{-16} to 7.7×10^{-8}. The potential cumulative ELCR is less than the "Cleanup Standards to Protect Air Quality" (WAC 173-340-750) Method B risk value of 1×10^{-6} for individual carcinogens for 65 remediated waste sites. Risks were not reported at eight remediated waste sites because COPC concentrations were less than background. Nonradiological carcinogenic COPCs were not reported at 19 remediated waste sites.

As presented in Table G-28 (Appendix G), the potential HI is less than the EPA target HI of 1 and the "Cleanup Standards to Protect Air Quality" (WAC 173-340-750) Method B target HI of 1 for 65 remediated waste sites. An HI was not reported for 23 remediated waste sites because COPC concentrations were less than background. Nonradiological COPCs were not reported at four remediated waste sites.

Overburden. A total of 32 remediated waste sites are reported with CVP/RSVP data associated with overburden material in the 100-D Source OU. The following sample designs were applied to the remediated waste sites evaluated:

- Thirty-one remediated waste sites were sampled using a statistical sampling design (include one site with two statistically distinct decision units.
- One remediated waste site was sampled using both a statistical and a focused sampling design and was subdivided into two focused decision units and two statistical decision units.

Radionuclide Results. As presented in Table 6-15, the potential total ELCR is within the target risk range of 10^{-4} to 10^{-6} for overburden material associated with 14 remediated waste sites and less than the lower target risk threshold of 1×10^{-6} for overburden material associated with two remediated waste sites. Risks were not reported at eight remediated waste sites because radiological COPCs concentrations were less than background. Radiological COPCs were not reported at eight remediated waste sites.

Nonradiological Results (Direct Contact). As presented in Table 6-16, the potential cumulative ELCR is greater than 1×10^{-6} for overburden material associated with three remediated waste sites and is less than 1×10^{-6} for overburden material associated with 14 remediated waste sites. Risks were not reported in overburden material associated with 11 remediated waste sites because nonradiological carcinogenic COPCs were less than background. Risks were not reported in overburden material associated with four remediated waste sites because no COPCs were reported. Although overburden material associated with three remediated waste sites report a total ELCR greater than 1×10^{-6}; there were no individual carcinogens reported with risks greater than the target risk level of 1×10^{-6}.

As presented in Table 6-16, the potential HI for direct contact from noncancer effects without background contribution is less than the EPA target HI of 1 and the MTCA ("Unrestricted Land Use Soil Cleanup Standards" [WAC 173-340-740]) target HI of 1 for overburden material associated with 30 remediated waste sites. Hazards were not reported in overburden material associated with two remediated waste sites because nonradiological carcinogenic COPCs were less than background.

As presented in Table G-26 (Appendix G), all lead and arsenic EPCs are less than their respective Method A soil cleanup levels of 250 mg/kg and 20 mg/kg for unrestricted land use.

Nonradiological Results (Inhalation). As presented in Table G-30 (Appendix G), the potential cumulative ELCR for the inhalation pathway from all nonradiological carcinogenic COPCs without background contribution ranges from 1.9×10^{-14} to 3.5×10^{-8}. The potential cumulative ELCR is less than the MTCA ("Cleanup Standards to Protect Air Quality" [WAC 173-340-750]) Method B risk threshold of 1×10^{-6} for individual carcinogens for overburden material associated with 21 remediated waste sites. Risks were not reported at eight remediated waste sites because nonradiological carcinogenic COPCs were less than background. Nonradiological carcinogenic COPCs were not reported in three remediated waste sites.

As presented in Table G-30 (Appendix G), the potential HI is less than the EPA target HI of 1 and the MTCA ("Cleanup Standards to Protect Air Quality" [WAC 173-340-750]) Method B target HI of 1 for overburden material associated with 26 remediated waste sites. Risks were not reported at six remediated waste sites because nonradiological carcinogenic COPCs were less than background.

Staging Pile Area. A total of 11 remediated waste sites are reported with CVP/RSVP data associated with staging pile areas in the 100-D Source OU. The following sample designs were applied to the remediated waste sites evaluated:

- Nine remediated waste sites were sampled using a statistical sampling design.
- One remediated waste site was sampled using a focused sampling design.
- One remediated waste site was sampled using both a statistical and a focused sampling design and was subdivided into two distinct statistical decision units.

Radionuclide Results. As presented in Table 6-17, the potential total ELCR is within the target risk range of 10^{-4} to 10^{-6} for staging pile material associated with one remediated waste site and less than the lower risk threshold of 1×10^{-6} for staging pile material associated with three remediated waste sites. Risks were not reported in staging pile material associated with two remediated waste sites because radiological COPC concentrations were less than background. Radiological COPCs were not reported at five remediated waste sites.

Nonradiological Results (Direct Contact). As presented in Table 6-18, the potential cumulative ELCR from direct contact for all nonradiological carcinogenic COPCs without background contribution is greater than 1×10^{-6} for staging pile material associated with one remediated waste site and is less than 1×10^{-6} for staging pile material associated with nine remediated waste sites. Risks were not reported at one remediated waste site because nonradiological carcinogenic COPC concentrations were less than background.

As presented in Table 6-18, staging pile material associated with one remediated waste site reports individual carcinogens greater than the WAC 173-340-740 acceptable cancer risk level of 1×10^{-6}; however, it is less than the MTCA ("Human Health Risk Assessment Procedures" [WAC 173-340-708(5)]) cumulative risk threshold of 1×10^{-5}. The cancer risk levels for the residential scenario are as follows:

- The 132-D-1 (staging pile area decision unit) reports a cumulative ELCR of 1.8×10^{-6}. The primary contributor to risk is benzo[a]pyrene (1.3×10^{-6}; 69 percent contribution).

As presented in Table 6-18, the potential HI for direct contact from noncancer effects without background contribution is less than the EPA target HI of 1 and the MTCA ("Unrestricted Land Use Soil Cleanup Standards" [WAC 173-340-740]) target HI of 1 for staging pile material associated with the 11 remediated waste sites.

As presented in Table G-26 (Appendix G), all lead and arsenic EPCs are less than their respective Method A soil cleanup levels of 250 and 20 mg/kg for unrestricted land use.

Nonradiological Results (Inhalation). As presented in Table G-32 (Appendix G), the potential cumulative ELCR for the inhalation pathway from all nonradiological carcinogenic COPCs without background contribution ranges from 9.0×10^{-16} to 2.8×10^{-10}. The potential cumulative ELCR is less than the MTCA ("Cleanup Standards to Protect Air Quality" [WAC 173-340-750]) Method B risk risk value of 1×10^{-6} for individual carcinogens for staging pile material associated with the 11 remediated waste sites.

As presented in Table G-32 (Appendix G), the potential HI is less than the EPA target HI of 1 and the MTCA ("Cleanup Standards to Protect Air Quality" [WAC 173-340-750]) Method B target HI of 1 for staging pile material associated with nine remediated waste sites. HIs were not reported for two remediated waste sites because COPC concentrations were less than background.

Deep Zone. Deep vadose zone samples were evaluated to identify remediated waste sites where exposure to residual contamination could present a potential risk from an inadvertent exposure through deep excavation activities. While this exposure would be industrial in nature, the RBSLs (developed for the Residential exposure scenario) were used for convenience as screening values to identify such sites in order to allow institutional controls to be established to control access to deep contamination.

A total of 18 remediated waste sites are reported with CVP/RSVP data associated with the deep zone in the 100-D Source OU. The following lists the sample designs that were applied to the remediated waste sites evaluated:

- One remediated waste site was sampled using a focused sampling design.
- Sixteen remediated waste sites were sampled using a statistical sampling design.
- One remediated waste site was sampled using both a statistical and a focused sampling design.

The remaining 77 remediated waste sites were not excavated deeper than 4.6 m (15 ft) bgs and are not discussed in this section.

Radionuclide Results. As presented in Table 6-19, the total ELCR is greater than the upper risk threshold of 1×10^{-4} at 14 remediated waste sites; is within the target risk range of 10^{-4} to 10^{-6} at two remediated waste site; and is less than the lower risk threshold of 1×10^{-6} at one remediated waste site. Risks were not reported at one remediated waste site because COPC concentrations were less than background.

100-H Source OU. Risk estimates were calculated for each decision unit within a remediated waste site including shallow vadose zone material, deep vadose zone material, overburden material, and staging pile area footprint material. The results for the Residential scenario are presented in Tables G-43 to G-52 (Appendix G).

An overall summary of the total risk estimates and noncancer hazards (if applicable) for the residential scenario from each of the remediated waste sites is provided in Table 6-20 and Table 6-21 for shallow zone material, Table 6-22 and Table 6-23 for overburden material, Table 6-24 and Table 6-25 for staging piles, and Table 6-26 for the deep zone. These tables list the OU that each remediated waste site resides in, the reclassification status, the remediated waste site, the associated waste site (if applicable), the decision unit reported with an exceedance (if applicable), the total ELCR and the risk driver and percent contribution (if applicable), and the hazard index and the noncancer hazard driver and percent contribution (if applicable).

Shallow Zone. A total of 42 remediated waste sites are reported with CVP/RSVP data associated with the shallow zone in the 100-H Source OU. The following lists the sample designs that were applied to the sites evaluated:

- Fifteen remediated waste sites were sampled using a focused sampling design.
- Seventeen remediated waste sites were sampled using a statistical sampling design (with one site having two statistically distinct decision units and three sites having three statistically distinct decision units).
- Ten remediated waste sites were sampled using both a statistical and a focused sampling design (with two sites having one focused and two statistically distinct decision units and one site having one focused and three statistically distinct decision units).

Radionuclide Results. As presented in Table 6-20, the potential total ELCR from all radiological COPCs without background contribution is greater than the upper risk threshold of 1×10^{-4} for two remediated

waste sites, is within the target risk range of 10^{-4} to 10^{-6} for 16 remediated waste sites, and less than the lower risk threshold of 1×10^{-6} for two remediated waste sites. Risks were not reported at five remediated waste sites because radiological COPC concentrations were less than background. Radiological COPCs were not reported at 17 remediated waste sites.

Two remediated waste sites report concentration of Hanford Site-related COPCs that exceed the upper range of the threshold for the Residential scenario. The cancer risk levels for the Residential scenario are as follows:

- The 116-H-5 waste site (shallow decision unit) reports a total ELCR of 1.1×10^{-4}. The primary contributor to risk is strontium-90 (1.1×10^{-4}; 96 percent contribution). The EPC of strontium-90 is 2.4 pCi/g, which is greater than the residential RBSL of 2.3 pCi/g. However, the EPC is less than the current direct exposure remedial action goal of 4.5 pCi/g published in the 100 Area RDR/RAWP (DOE/RL-96-17). Activities of all radionuclides will decay to a total ELCR of less than 1.0×10^{-4} by year 2016.

- The 118-H-1:1waste site (shallow 2 decision unit) reports a total ELCR of 1.2×10^{-4}. The primary contributor to risk is strontium-90 (1.0×10^{-4}; 87 percent contribution). The EPC of strontium-90 is 2.3 pCi/g, which is equal to the residential RBSL of 2.3 pCi/g. However, the EPC is less than the current direct exposure remedial action goal of 4.5 pCi/g published in the 100 Area RDR/RAWP (DOE/RL-96-17). Activities of all radionuclides will decay to a total ELCR of less than 1.0×10^{-4} by year 2016.

Nonradiological Results (Direct Contact). As presented in Table 6-21, the potential cumulative ELCR is greater than 1×10^{-6} for 21 remediated waste sites and is less than 1×10^{-6} for 12 remediated waste sites. Risks were not reported at nine remediated waste sites because nonradiological carcinogenic COPC concentrations were less than background.

As presented in Table 6-21, 21 remediated waste sites report individual carcinogens greater than the MTCA ("Unrestricted Land Use Soil Cleanup Standards" [WAC 173-340-740]) acceptable cancer risk level of 1×10^{-6}, 19 of the remediated waste sites are greater than the MTCA ("Human Health Risk Assessment Procedures" [WAC 173-340-708(5)]) cumulative risk threshold of 1×10^{-5}. The cancer risk levels for the residential scenario are as follows:

- 100-H-21 (shallow decision unit) reports a cumulative nonradiological ELCR of 2.0×10^{-5}. The primary contributor to risk is arsenic (2.0×10^{-5}; 100 percent contribution).

- 100-H-28:6 (shallow focused decision unit) reports a cumulative nonradiological ELCR of 1.1×10^{-5}. The primary contributor to risk is arsenic (1.1×10^{-5}; 100 percent contribution).

- 100-H-3 (shallow decision unit) reports a cumulative nonradiological ELCR of 2.3×10^{-5}. The primary contributor to risk is arsenic (2.2×10^{-5}; 97 percent contribution).

- 100-H-35 (shallow focused 3 decision unit) reports a cumulative nonradiological ELCR of 2.8×10^{-5}. The primary contributor to risk is arsenic (2.8×10^{-5}; 100 percent contribution).

- 100-H-37 (shallow focused decision unit) reports a cumulative nonradiological ELCR of 2.0×10^{-5}. The primary contributor to risk is arsenic (2.0×10^{-5}; 100 percent contribution).

- 100-H-4 (shallow focused decision unit) reports a cumulative nonradiological ELCR of 1.4×10^{-5}. The primary contributor to risk is arsenic (1.3×10^{-5}; 95 percent contribution).

Table 6-20. Summary of Total Risks from Radionuclides for the 100-HR-1 and 100-HR-2 Shallow Zone Waste Sites for the Residential Scenario

Classification Status	Remediated Waste Site	Associated Waste Sites[a]	Decision Unit with Exceedance	Total ELCR[b]	Risk Driver and % Contribution
100-HR-1					
Interim Closed Out	116-H-5	--	Shallow	1.1×10^{-4} $< 1.0 \times 10^{-4}$ (2016)	Strontium-90 (1.1×10^{-4} – 96%)
	100-H-17	100-H-30 116-H-2	None	1×10^{-4} to 1×10^{-6}	None
	100-H-21	100-H-1 100-H-22			
	100-H-4 100-H-5 116-H-1 116-H-3 116-H-7 118-H-6:5	--			
	116-H-9	--	None	$< 1 \times 10^{-6}$	None
	118-H-6:4 1607-H2 1607-H4	--	None	No COPCs reported above background	None
	100-H-24 100-H-3 100-H-41 1607-H3	--	None	No COPCs detected	None
No Action	100-H-49:2	--	None	1×10^{-4} to 1×10^{-6}	None
	100-H-35 100-H-53	--	None	No COPCs reported above background	None
	100-H-28:1 100-H-28:6 100-H-45 100-H-50 100-H-51:4 100-H-51:5 100-H-7 100-H-8	--	None	No COPCs detected	None

Table 6-20. Summary of Total Risks from Radionuclides for the 100-HR-1 and 100-HR-2 Shallow Zone Waste Sites for the Residential Scenario

Classification Status	Remediated Waste Site	Associated Waste Sites[a]	Decision Unit with Exceedance	Total ELCR[b]	Risk Driver and % Contribution
100-HR-2					
Interim Closed Out	--[c]	100-H-2	None	1×10^{-4} to 1×10^{-6}	None
	118-H-1:1	--	Shallow 2	1.2×10^{-4} $< 1.0 \times 10^{-4}$ (2016)	Strontium-90 (1.0×10^{-4} – 87%)
	100-H-37 118-H-2 118-H-3 118-H-4 118-H-5 1607-H1 600-152	--	None	1×10^{-4} to 1×10^{-6}	None
	118-H-1:2	--	None	$< 1 \times 10^{-6}$	None
	128-H-1 600-151	--	None	No COPCs detected	None
No Action	100-H-40 128-H-2 128-H-3	--	None	No COPCs detected	None

Note:

Results summarized from Table G-43, Residential Scenario Risk Estimates and Noncancer Hazards for the 100-H Source OU Waste Site Decision Units Without Background Contribution (Appendix G).

The following five waste sites do not report shallow zone sample data: 100-H-11, 100-H-12, 100-H-14, 118-H-6:3, and 118-H-6:6.

a. Associated waste sites are those sites for which remediation and closeout documentation were consolidated with another remediated waste site.

b. Total ELCR represents risk contributions from radiological COPCs.

c. Remediated waste site 100-H-2 (100-HR-2 OU) is associated with remediated waste site 100-H-17 (100-HR-1).

Table 6-21. Summary of Total Carcinogenic Risks and Noncancer Hazards from Direct Contact for the 100-HR-1 and 100-HR-2 Shallow Zone Waste Sites for the Residential Scenario

Classification Status	Remediated Waste Site	Associated Waste Sites[a]	Decision Unit with Exceedance	Total ELCR[b]	Risk Driver and % Contribution	Hazard Index	Noncancer Hazard Driver and % Contribution
100-HR-1							
Interim Closed Out	118-H-6:5	--	Shallow 1	6.0×10^{-5}	Arsenic (6.0×10^{-5} – >99%)	1.7	Arsenic (HQ = 1.7 – >99%)
	100-H-3		Shallow	2.3×10^{-5}	Arsenic (2.2×10^{-5} – 97%)	<1	None
	100-H-41		Shallow Focused	2.2×10^{-5}	Arsenic (1.2×10^{-5} – 55%) Benzo(a)pyrene (7.1×10^{-6} – 33%)		
	100-H-21	100-H-1 100-H-22	Shallow	2.0×10^{-5}	Arsenic (2.0×10^{-5} – 100%)		
	1607-H4	--	Shallow	1.5×10^{-5}	Arsenic (1.1×10^{-5} – 75%) Benzo(a)pyrene (2.8×10^{-6} – 19%)		
	100-H-4		Shallow Focused	1.4×10^{-5}	Arsenic (1.3×10^{-5} – 95%)		
	116-H-3		Shallow	1.2×10^{-5}	Arsenic (1.2×10^{-5} – 100%)		
	1607-H2		Shallow	1.1×10^{-5}	Arsenic (1.1×10^{-5} – 97%)		
	116-H-1		Shallow	9.9×10^{-6}	Arsenic (9.9×10^{-6} – 100%)		
	116-H-7		Shallow	1.3×10^{-6}	Aroclor-1260 (1.3×10^{-6} – 100%)		
	100-H-24 116-H-5 116-H-9 1607-H3	--	None	$< 1 \times 10^{-6}$	None	<1	None
	100-H-17	100-H-30 116-H-2--	None	No COPCs reported above background	None	<1	None
	100-H-5	--				No COPCs reported above background	None
	118-H-6:4						
No Action	100-H-35	--	Shallow Focused 3	2.8×10^{-5}	Arsenic (2.8×10^{-5} – 100%)	<1	None
	100-H-49:2		Shallow Focused	1.4×10^{-5}	Arsenic (1.3×10^{-5} – 96%)		
	100-H-8		Shallow Focused	1.1×10^{-5}	Arsenic (1.1×10^{-5} – 100%)		
	100-H-28:6		Shallow Focused	1.1×10^{-5}	Arsenic (1.1×10^{-5} – 100%)		

Table 6-21. Summary of Total Carcinogenic Risks and Noncancer Hazards from Direct Contact for the 100-HR-1 and 100-HR-2 Shallow Zone Waste Sites for the Residential Scenario

Classification Status	Remediated Waste Site	Associated Waste Sites[a]	Decision Unit with Exceedance	Total ELCR[b]	Risk Driver and % Contribution	Hazard Index	Noncancer Hazard Driver and % Contribution
	100-H-28:1 100-H-51:4 100-H-51:5 100-H-53	--	None	$< 1 \times 10^{-6}$	None	<1	None
	100-H-45 100-H-50 100-H-7	--	None	No COPCs reported above background	None	<1	None
100-HR-2							
Interim Closed Out	--[c]	100-H-2	None	No COPCs reported above background	None	<1	None
	600-151	--	Shallow 2	9.0×10^{-5}	Arsenic (8.9×10^{-5} – >99%)	2.5	Arsenic (HQ = 2.5 – >99%)
	118-H-3		Shallow 1	2.4×10^{-5}	Arsenic (2.4×10^{-5} – >99%)	No COPCs reported above background	None
	100-H-37		Shallow Focused	2.0×10^{-5}	Arsenic (2.0×10^{-5} – 100%)	<1	None
	128-H-1		Shallow 3	1.7×10^{-5}	Arsenic (1.6×10^{-5} – 97%)		
	1607-H1		Shallow	1.5×10^{-5}	Arsenic (1.4×10^{-5} – 95%)		
	118-H-1:1 118-H-1:2 600-152	--	None	$< 1 \times 10^{-6}$	None	<1	None
	118-H-2	--	None	No COPCs reported above background	None	<1	None
	118-H-4						
	118-H-5						
No Action	128-H-3	--	Shallow Focused	1.7×10^{-5}	Arsenic (1.7×10^{-5} – 98%)	<1	None
	100-H-40		Shallow Focused	1.2×10^{-5}	Arsenic (1.2×10^{-5} – 97%)		
	128-H-2	--	None	$< 1 \times 10^{-6}$	None	<1	None

Table 6-21. Summary of Total Carcinogenic Risks and Noncancer Hazards from Direct Contact for the 100-HR-1 and 100-HR-2 Shallow Zone Waste Sites for the Residential Scenario

Classification Status	Remediated Waste Site	Associated Waste Sites[a]	Decision Unit with Exceedance	Total ELCR[b]	Risk Driver and % Contribution	Hazard Index	Noncancer Hazard Driver and % Contribution

Note:

Results summarized from Table G-43, Residential Scenario Risk Estimates and Noncancer Hazards for the 100-H Source OU Waste Site Decision Units Without Background Contribution (Appendix G).

The following five waste sites do not report shallow zone sample data: 100-H-11, 100-H-12, 100-H-14, 118-H-6:3, and 118-H-6:6.

a. Associated waste sites are those sites for which remediation and closeout documentation were consolidated with another remediated waste site.

b. Total ELCR represents risk contributions from nonradiological carcinogenic COPCs.

c. Remediated waste site 100-H-2 (100-HR-2 OU) is associated with remediated waste site 100-H-17 (100-HR-1).

Table 6-22. Summary of Total Risks from Radionuclides for the 100-HR-1 and 100-HR-2 Overburden Material for the Residential Scenario

Classification Status	Remediated Waste Site	Associated Waste Sites[a]	Decision Unit with Exceedance	Total ELCR[b]	Risk Driver and % Contribution
100-HR-1					
Interim Closed Out	100-H-21	100-H-1 100-H-22	None	1×10^{-4} to 1×10^{-6}	None
	116-H-5 118-H-6:4	--		1×10^{-4} to 1×10^{-6}	
	116-H-7 1607-H2			No COPCs reported above background	
	1607-H3			No COPCs reported	
100-HR-2					
Interim Closed Out	118-H-1:1 1607-H1	--	None	1×10^{-4} to 1×10^{-6}	None
	128-H-1			No COPCs reported	

Table 6-22. Summary of Total Risks from Radionuclides for the 100-HR-1 and 100-HR-2 Overburden Material for the Residential Scenario

Classification Status	Remediated Waste Site	Associated Waste Sites[a]	Decision Unit with Exceedance	Total ELCR[b]	Risk Driver and % Contribution

Note:

Results summarized from Table G-43, Residential Scenario Risk Estimates and Noncancer Hazards for the 100-H Source OU Waste Site Decision Units Without Background Contribution (Appendix G).

a. Associated waste sites are those sites for which remediation and closeout documentation were consolidated with another remediated waste site.

b. Total ELCR represents risk contributions from radiological COPCs.

Table 6-23. Summary of Total Carcinogenic Risks and Noncancer Hazards from Direct Contact for the 100-HR-1 and 100-HR-2 Overburden Material for the Residential Scenario

Classification Status	Remediated Waste Site	Associated Waste Sites[a]	Decision Unit with Exceedance	Total ELCR[b]	Risk Driver and % Contribution	Hazard Index	Noncancer Hazard Driver and % Contribution
100-HR-1							
Interim Closed Out	1607-H2		Overburden	2.5×10^{-5}	Arsenic (2.4×10^{-5} – 95%)	<1	None
	100-H-21	100-H-1 100-H-22	Overburden	1.7×10^{-5}	Arsenic (1.7×10^{-5} – 100%)		
	116-H-7 116-H-5 1607-H3	--	None	$< 1 \times 10^{-6}$	None	<1	None
	118-H-6:4	--	None	No COPCs reported above background	None	<1	None
100-HR-2							
Interim Closed Out	128-H-1		Overburden	6.1×10^{-5}	Arsenic (6.1×10^{-5} – >99%)	1.7	Arsenic (HQ = 1.7 – >99%)
	1607-H1		Overburden	2.2×10^{-5}	Arsenic (2.2×10^{-5} – >99%)	<1	None
	118-H-1:1	--	None	$< 1 \times 10^{-6}$	None	<1	None

Table 6-23. Summary of Total Carcinogenic Risks and Noncancer Hazards from Direct Contact for the 100-HR-1 and 100-HR-2 Overburden Material for the Residential Scenario

Classification Status	Remediated Waste Site	Associated Waste Sites[a]	Decision Unit with Exceedance	Total ELCR[b]	Risk Driver and % Contribution	Hazard Index	Noncancer Hazard Driver and % Contribution

Note:

Results summarized from Table G-43, Residential Scenario Risk Estimates and Noncancer Hazards for the 100-H Source OU Waste Site Decision Units Without Background Contribution (Appendix G).

a. Associated waste sites are those sites for which remediation and closeout documentation were consolidated with another remediated waste site.

b. Total ELCR represents risk contributions from nonradiological carcinogenic COPCs.

Table 6-24. Summary of Total Risks from Radionuclides for the 100-HR-1 and 100-HR-2 Staging Piles for the Residential Scenario

Classification Status	Remediated Waste Site	Associated Waste Sites[a]	Decision Unit with Exceedance	Total ELCR[b]	Risk Driver and % Contribution
100-HR-1					
Interim Closed Out	116-H-5	--	None	1×10^{-4} to 1×10^{-6}	None
100-HR-2					
Interim Closed Out	118-H-4	--	None	No COPCs reported above background	None
	118-H-1:1			1×10^{-4} to 1×10^{-6}	
	128-H-1			No COPCs reported	

Note:

Results summarized from Table G-43, Residential Scenario Risk Estimates and Noncancer Hazards for the 100-H Source OU Waste Site Decision Units Without Background Contribution (Appendix G).

a. Associated waste sites are those sites for which remediation and closeout documentation were consolidated with another remediated waste site.

b. Total ELCR represents risk contributions from radiological COPCs.

Table 6-25. Summary of Total Carcinogenic Risks and Noncancer Hazards from Direct Contact for the 100-HR-1 and 100-HR-2 Staging Piles for the Residential Scenario

Classification Status	Remediated Waste Site	Associated Waste Sites[a]	Decision Unit with Exceedance	Total ELCR[b]	Risk Driver and % Contribution	Hazard Index	Noncancer Hazard Driver and % Contribution
100-HR-1							
Interim Closed Out	116-H-5	--	Staging Pile Area	1.6×10^{-5}	Arsenic (1.5×10^{-5}– 96%)	<1	None
100-HR-2							
Interim Closed Out	128-H-1		Staging Pile Area 2	8.1×10^{-5}	Arsenic (8.1×10^{-5} – >99%)	2.3	Arsenic (HQ = 2.3 – 100%)
	118-H-1:1	--	None	$< 1 \times 10^{-6}$	None	<1	None
	118-H-4	--	None	No COPCs reported above background	None	<1	None

Note:

Results summarized from Table G-43, Residential Scenario Risk Estimates and Noncancer Hazards for the 100-H Source OU Waste Site Decision Units Without Background Contribution (Appendix G).

a. Associated waste sites are those sites for which remediation and closeout documentation were consolidated with another remediated waste site.

b. Total ELCR represents risk contributions from nonradiological carcinogenic COPCs.

Table 6-26. Summary of Total Risks from Radionuclides for the 100-HR-1 and 100-HR-2 Deep Zone Waste Sites for the Residential Scenario

Classification Status	Remediated Waste Site	Associated Waste Sites[a]	Decision Unit with Exceedance	Total ELCR[b]	Risk Driver and % Contribution
100-HR-1					
Interim Closed Out	116-H-1	--	Deep	3.0×10^{-3} $< 1.0 \times 10^{-4}$ (2110)	Cesium-137 (1.1×10^{-3} - 36%) Europium-152 (1.6×10^{-3} - 53%) Europium-154 (1.5×10^{-4} - 5%) Strontium-90 (1.2×10^{-4} - 4%)
	116-H-7		Deep	2.8×10^{-3} $< 1.0 \times 10^{-4}$ (2098)	Cesium-137 (5.0×10^{-4} - 18%) Cobalt-60 (2.1×10^{-4} - 7%) Europium-152 (1.8×10^{-3} - 62%) Europium-154 (2.0×10^{-4} - 7%) Strontium-90 (1.1×10^{-4} - 4%)
	116-H-3		Deep	9.4×10^{-4} $< 1.0 \times 10^{-4}$ (2056)	Cesium-137 (2.3×10^{-4} - 25%) Europium-152 (6.3×10^{-4} - 67%)
	118-H-6:2,:3,:6, 100-H-9,10,11,12,13,14,31	100-H-10 100-H-13 100-H-31 100-H-9 118-H-6:2	Deep 3	8.7×10^{-4} $< 1.0 \times 10^{-4}$ (2108)	Cesium-137 (4.4×10^{-4} - 51%) Strontium-90 (2.8×10^{-4} - 32%)
	100-H-21	100-H-1 100-H-22	Deep	1.9×10^{-4} $< 1.0 \times 10^{-4}$ (2019)	Cesium-137 (1.0×10^{-4} - 52%) Europium-152 (5.9×10^{-5} - 31%)
	116-H-5	--	None	1×10^{-4} to 1×10^{-6}	None
	100-H-5		None	No COPCs reported above background	None
	1607-H2	--	None	No COPCs detected	None
No deep zone decision units reported in 100-HR-2 Operable Unit					

Note:

Results summarized from Table G-43, Residential Scenario Risk Estimates and Noncancer Hazards for the 100-H Source OU Waste Site Decision Units Without Background Contribution (Appendix G).

a. Associated waste sites are those sites for which remediation and closeout documentation were consolidated with another remediated waste site.

b. Total ELCR represents risk contributions from radiological COPCs.

- 100-H-40 (shallow focused decision unit) reports a cumulative nonradiological ELCR of 1.2×10^{-5}. The primary contributor to risk is arsenic (1.2×10^{-5}; 97 percent contribution).
- 100-H-41 (shallow focused decision unit) reports a cumulative nonradiological ELCR of 2.2×10^{-5}. The primary contributors to risk are arsenic (1.2×10^{-5}; 55 percent contribution) and benzo(a)pyrene (7.1×10^{-6}; 33 percent contribution).
- 100-H-49:2 (shallow focused decision unit) reports a cumulative nonradiological ELCR of 1.4×10^{-5}. The primary contributor to risk is arsenic (1.3×10^{-5}; 96 percent contribution).
- 100-H-8 (shallow focused decision unit) reports a cumulative nonradiological ELCR of 1.1×10^{-5}. The primary contributor to risk is arsenic (1.1×10^{-5}; 100 percent contribution).
- 116-H-1 (shallow decision unit) reports a cumulative nonradiological ELCR of 9.9×10^{-6}. The primary contributor to risk is arsenic (9.9×10^{-6}; 100 percent contribution).
- 116-H-3 (shallow decision unit) reports a cumulative nonradiological ELCR of 1.2×10^{-5}. The primary contributor to risk is arsenic (1.2×10^{-5}; 100 percent contribution).
- 116-H-7 (shallow decision unit) reports a cumulative nonradiological ELCR of 1.3×10^{-6}. The primary contributor to risk is aroclor-1260 (1.3×10^{-6}; 100 percent contribution).
- 118-H-3 (shallow 1 decision unit) reports a cumulative nonradiological ELCR of 2.4×10^{-5}. The primary contributor to risk is arsenic0 (2.4×10^{-5}; >99 percent contribution).
- 118-H-6:5 (shallow 1 decision unit) reports a cumulative nonradiological ELCR of 6.0×10^{-5}. The primary contributor to risk is arsenic (6.0×10^{-5}; > 99 percent contribution).
- 128-H-1 (shallow 3 decision unit) reports a cumulative nonradiological ELCR of 1.7×10^{-5}. The primary contributor to risk is arsenic0 (1.6×10^{-5}; 97 percent contribution).
- 128-H-3 (shallow focused decision unit) reports a cumulative nonradiological ELCR of 1.7×10^{-5}. The primary contributor to risk is arsenic0 (1.7×10^{-5}; 98 percent contribution).
- 1607-H1 (shallow decision unit) reports a cumulative nonradiological ELCR of 1.5×10^{-5}. The primary contributor to risk is arsenic (1.4×10^{-5}; 95 percent contribution)
- 1607-H2 (shallow decision unit) reports a cumulative nonradiological ELCR of 1.1×10^{-5}. The primary contributor to risk is arsenic (1.1×10^{-5}; 97 percent contribution).
- 1607-H4 (shallow decision unit) reports a cumulative nonradiological ELCR of 1.5×10^{-5}. The primary contributors to risk are arsenic (1.1×10^{-5}; 75 percent contribution) and benzo(a)pyrene (2.8×10^{-6}; 19 percent contribution).
- 600-151 (shallow 2 decision unit) reports a cumulative nonradiological ELCR of 9.0×10^{-5}. The primary contributor to risk is arsenic (8.9×10^{-5}; >99 percent contribution).

Arsenic is a primary contributor to risk at 20 of the 21 remediated waste sites. With the exception of arsenic concentrations at 118-H-6:5 and 600-151, the EPCs for arsenic at the remaining 19 waste sites range from 6.6 to 18.5 mg/kg. Although the EPCs are greater than the lognormal 90th percentile value of 6.5 mg/kg, they are consistent with naturally occurring levels at the Hanford Site (Table G-14 [Appendix G]) and are not the result of a site release. Additionally, the EPCs for arsenic at these 19 remediated waste sites are less than the direct exposure RAG of 20 mg/kg published in the 100 Area

RDR/RAWP (DOE/RL-96-17). The following paragraphs discuss arsenic concentrations measured at the 118-H-6:5 and 600-151 remediated waste sites.

Although aroclor-1260 at 116-H-7 (shallow decision unit) and benzo(a)pyrene at 100-H-4 (shallow decision unit) and 1607-H4 (shallow decision unit) are greater than the acceptable risk threshold value of 1×10^{-6} for individual carcinogens, they are not greater than the MTCA ("Human Health Risk Assessment Procedures" [WAC 173-340-708(5)]) cumulative risk threshold of 1×10^{-5}.

As presented in Table 6-21, the potential HI for direct contact from noncancer effects without background contributions is greater than the target HI of 1 at two remediated waste sites and is less than the target HI at 38 remediated waste sites. An HI was not reported for two remediated waste sites because nonradiological COPC concentrations were less than background.

Two remediated waste sites report a HI greater than the MTCA ("Human Health Risk Assessment Procedures" [WAC 173-340-708(5)]) target HI of 1. The HI for the residential scenario is as follows:

- 118-H-6:5 (shallow 1 decision unit) reports an HI of 1.7. The primary contributor to the HI is arsenic (HQ = 1.7; > 99 percent contribution).
- 600-151 (shallow 2 decision unit) reports an HI of 2.5. The primary contributor to the HI is arsenic (HQ = 2.5; >99 percent contribution).

The following paragraphs provide a discussion of arsenic concentrations measured at the 118-H-6:5 and 600-151 remediated waste sites.

A comparison of arsenic and lead EPCs to their respective Method A soil cleanup level of 20 mg/kg and 250 mg/kg, respectively, is provided in Table G-45. Except for arsenic EPCs reported at remediated waste sites 118-H-6:5 and 600-151, all arsenic EPCs are less than the Method A soil cleanup level of 20 mg/kg for unrestricted land use. Except for lead EPCs reported at remediated waste site 600-151, all lead EPCs are less than the Method A soil cleanup level of 250 mg/kg.

For 118-H-6:5 remediated waste site, the arsenic EPCs for the shallow 1 decision unit (39.6 mg/kg) and the shallow focused decision unit (27 mg/kg) are greater than the remedial action goal of 20 mg/kg published in the 100 Area RDR/RAWP (DOE/RL-96-17). A summary of the arsenic results for the 118-H-6:5 remediated waste site follows:

- Twelve soil samples were collected from the shallow 1 decision unit and analyzed for arsenic. Arsenic concentrations range between 6.52 and 66.2 mg/kg (six results greater than the Method A soil cleanup level of 20 mg/kg). Arsenic concentrations greater than the Method A cleanup level range between 23.5 and 66.2 mg/kg.
- Two soil samples were collected and analyzed from this decision unit. Arsenic concentrations from the focused decision unit range between 17 and 27 mg/kg (one result greater than the Method A soil cleanup level of 20 mg/kg).

For 600-151 remediated waste site, the arsenic EPCs for the shallow 1 (31.8 mg/kg), shallow 2 (59.6 mg/kg), and shallow 3 (54 mg/kg) decision units are greater than the remedial action goal of 20 mg/kg published in the 100 Area RDR/RAWP (DOE/RL-96-17). A summary of the arsenic results for the 600-151 remediated waste site follows:

- Eighteen soil samples were collected from the shallow 1 decision unit and analyzed for arsenic. Arsenic concentrations range between 3.2 and 74.4 mg/kg (four results greater than the Method A soil

cleanup level of 20 mg/kg). Arsenic concentrations greater than the Method A cleanup level range between 21.6 and 74.4 mg/kg.

- Twelve soil samples were collected from the shallow 2 decision unit and analyzed for arsenic. Arsenic concentrations range between 7 and 104 mg/kg (nine results greater than the Method A soil cleanup level of 20 mg/kg). Arsenic concentrations greater than the Method A cleanup level range between 22.4 and 104 mg/kg.

- Thirteen soil samples were collected from the shallow 3 decision unit and analyzed for arsenic. Arsenic concentrations range between 8.7 and 68.3 mg/kg (eight results greater than the Method A soil cleanup level of 20 mg/kg). Arsenic concentrations greater than the Method A cleanup level range between 26 and 68 mg/kg.

For 600-151 remediated waste site, the lead EPCs for the shallow 2 (267 mg/kg) and shallow 3 (276 mg/kg) decision units are greater than the remedial action goal of 250 mg/kg published in the 100 Area RDR/RAWP (DOE/RL-96-17). A summary of the lead results for the 600-151 remediated waste site follows:

- Twelve soil samples were collected from the shallow 2 decision unit and analyzed for lead. Lead concentrations range between 12 and 518 mg/kg (three results greater than the Method A soil cleanup level of 250 mg/kg). Lead concentrations greater than the Method A cleanup level range between 286 and 518 mg/kg.

- Thirteen soil samples were collected from the shallow 3 decision unit and analyzed for lead. Lead concentrations range between 6.7 and 641 mg/kg (two results greater than the Method A soil cleanup level of 250 mg/kg). Lead concentrations greater than the Method A cleanup level are 408 and 641 mg/kg.

Nonradiological Results (Inhalation). As presented in Table G-47 (Appendix G), the potential cumulative ELCR for the inhalation pathway from all nonradiological carcinogenic COPCs without background contribution ranges from 6.3×10^{-14} to 4.6×10^{-7}. The potential cumulative ELCR is less than the MTCA ("Cleanup Standards to Protect Air Quality" [WAC 173-340-750]) Method B risk value of 1×10^{-6} for individual carcinogens for 37 remediated waste sites. Risks were not reported at five remediated waste sites because nonradiological carcinogenic COPC concentrations were less than background.

As presented in Table G-47 (Appendix G), the potential HI from the inhalation pathway from noncancer effects without background contributions is less than the EPA target HI of 1 and the MTCA ("Cleanup Standards to Protect Air Quality" [WAC 173-340-750]) Method B target HI of 1 for 38 remediated waste sites. An HI was not reported for four remediated waste sites because nonradiological COPC concentration were less than background.

Overburden. Nine remediated waste sites are reported with CVP/RSVP data associated with overburden in the 100-H Source OU. All nine remediated waste sites were sampled using a statistical sampling design.

Radiological Results. As presented in Table 6-22 (Appendix G), the potential total ELCR from all radiological COPCs without background contribution is within the target risk range of 10^{-4} to 10^{-6} for overburden material associated with five remediated waste sites. Risks were not reported in overburden material associated with two remediated waste sites because radiological COPC concentrations were less than background. Radiological COPCs were not reported in overburden material associated with two remediated waste sites.

Nonradiological Results (Direct Contact). As presented in Table 6-23, the potential cumulative ELCR from direct contact for all nonradiological carcinogenic COPCs without background contribution is greater than 1×10^{-6} for overburden material associated with four remediated waste sites and is less than 1×10^{-6} for overburden material associated with four remediated waste sites. Risks were not reported in overburden material associated with one remediated waste site because nonradiological carcinogenic COPC concentrations were less than background.

As presented in Table 6-23, overburden material associated with four remediated waste sites report individual carcinogens greater than the WAC 173-340-740 acceptable cancer risk level of 1×10^{-6}, all four are also greater than the MTCA ("Human Health Risk Assessment Procedures" [WAC 173-340-708(5)]) cumulative risk threshold of 1×10^{-5}. The cancer risk levels for the residential scenario are as follows:

- 100-H-21 (overburden) reports a cumulative nonradiological ELCR of 1.7×10^{-5}. The primary contributor to risk is arsenic (1.7×10^{-5}; 100 percent contribution).
- 128-H-1 (overburden) reports a cumulative nonradiological ELCR of 6.1×10^{-5}. The primary contributor to risk is arsenic (6.1×10^{-5}; > 99 percent contribution).
- 1607-H1 (overburden) reports a cumulative nonradiological ELCR of 2.2×10^{-5}. The primary contributor to risk is arsenic (2.2×10^{-5}; > 99 percent contribution).
- 1607-H2 (overburden) reports a cumulative nonradiological ELCR of 2.5×10^{-5}. The primary contributor to risk is arsenic (2.4×10^{-5}; 95 percent contribution).

Arsenic is the primary contributor to risk in overburden from four remediated waste sites. With the exception of the arsenic EPC in overburden material from 128-H-1, the EPCs for arsenic in overburden material from the three remaining remediated waste sites range from 11 to 16 mg/kg. Although these EPCs are greater than the lognormal 90th percentile value of 6.5 mg/kg, they are consistent with naturally occurring levels at the Hanford Site (see Table G-14 [Appendix G]) and not the result of a site release. Additionally, the EPCs for arsenic in overburden material from these three remediated waste sites are less than the direct exposure remedial action goal of 20 mg/kg published in the 100 Area RDR/RAWP (DOE/RL-96-17). Following is a discussion of arsenic concentrations measured in overburden material from the 128-H-1 remediated waste site.

As presented in Table 6-23, the potential HI for direct contact from noncancer effects without background contribution is greater than the target HI of 1 in overburden material from one remediated waste site and is less than the target HI in overburden material from eight remediated waste sites.

Overburden material associated with the 128-H-1 waste site reports a HI greater than the MTCA ("Human Health Risk Assessment Procedures" [WAC 173-340-708(5)]) target HI of 1. The HI for the residential scenario is as follows:

- 128-H-1 (overburden) reports an HI of 1.7. The primary contributor to the HI is arsenic (HQ = 1.7; > 99 percent contribution).

Table G-45 provides a comparison of arsenic and lead EPCs to their Method A soil cleanup levels of 20 mg/kg and 250 mg/kg, respectively. Except for arsenic (40.5 mg/kg) and lead (254 mg/kg), EPCs reported in overburden material associated with remediated waste site 128-H-1, all arsenic and lead EPCs are less than the Method A soil cleanup level of 20 mg/kg and 250 mg/kg, respectively, for unrestricted land use. A summary of the arsenic and lead results in overburden material associated with the 128-H-1 remediated waste site follows:

- Twelve soil samples were collected from the overburden material and analyzed for arsenic. Arsenic concentrations range between 15.1 and 56.8 mg/kg (nine results greater than the Method A soil cleanup level of 20 mg/kg). Arsenic concentrations greater than the Method A cleanup level range between 23.5 and 56.8 mg/kg.

- Twelve soil samples were collected from the overburden material and analyzed for lead. Lead concentrations range between 73.6 and 406 mg/kg (four results greater than the Method A soil cleanup level of 250 mg/kg). Lead concentrations greater than the Method A cleanup level range between 278 and 406 mg/kg.

Nonradiological Results (Inhalation). As presented in Table G-49 (Appendix G), the potential cumulative ELCR for the inhalation pathway from all nonradiological carcinogenic COPCs without background contribution ranges from 5.9×10^{-11} to 7.7×10^{-8}. The potential cumulative ELCR is less than the MTCA ("Cleanup Standards to Protect Air Quality" [WAC 173-340-750]) Method B risk value of 1×10^{-6} for individual carcinogens for overburden material associated with the nine remediated waste sites.

As presented in Table G-49 (Appendix G), the potential HI for the inhalation pathway from noncancer effects without background contributions is less than the EPA target HI of 1 and the MTCA ("Cleanup Standards to Protect Air Quality" [WAC 173-340-750]) Method B target HI of 1 for overburden material associated with the nine remediated waste sites.***Staging Pile Area.*** Four remediated waste sites are reported with CVP/RSVP data associated with a staging pile area in the 100-H Source OU. The four remediated waste sites were sampled using a statistical sampling design, with one site having two statistically distinct decision units.

Radiological Results. As presented in Table 6-24, the potential total ELCR from all radiological COPCs without background contribution are within the target risk range of 10^{-4} to 10^{-6} for staging piles associated with two remediated waste sites. Risks were not reported for one staging pile associated with one remediated waste site because radiological COPC concentrations were less than background. Radiological COPCs were not reported at one staging pile area associated with one remediated waste site.

Nonradiological Results (Direct Contact). As presented in Table 6-25, the potential cumulative ELCR from direct contact for all nonradiological carcinogenic COPCs without background contributions is greater than 1×10^{-6} for staging pile material associated with two remediated waste sites and is less than 1×10^{-6} for staging pile material associated with one remediated waste site. Risks were not reported in staging pile material associated with one remediated waste site because nonradiological carcinogenic COPC concentrations were less than background.

As reported in Table 6-25, staging pile material associated with two remediated waste sites report individual carcinogens greater than the WAC 173-340-740 acceptable cancer risk level of 1×10^{-6}, and are also greater than the MTCA ("Human Health Risk Assessment Procedures" [WAC 173-340-708(5)]) cumulative risk threshold of 1×10^{-5}. The cancer risk levels for the residential scenario are as follows:

- 116-H-5 (staging pile area) reports a cumulative nonradiological ELCR of 1.6×10^{-5}. The primary contributor to risk is arsenic (1.5×10^{-5}; 96 percent contribution).

- 128-H-1 (staging pile area footprint 2) reports a cumulative nonradiological ELCR of 8.1×10^{-5}. The primary contributor to risk is arsenic (8.1×10^{-5}; > 99 percent contribution).

Arsenic is the primary contributor to risk in staging pile material from two remediated waste sites. With the exception of the arsenic EPC in staging pile material from 128-H-1, the EPC for arsenic at 116-H-5 is 10.3 mg/kg. Although this EPC is greater than the lognormal 90th percentile value of 6.5 mg/kg, it is

consistent with naturally occurring levels at the Hanford Site (see Table G-14 [Appendix G]) and not the result of a site release. Additionally, the EPC for arsenic at the 116-H-5 waste site is less than the direct exposure remedial action goal of 20 mg/kg published in the 100 Area RDR/RAWP (DOE/RL-96-17). A discussion of arsenic concentrations measured in staging pile material from the 128-H-1 remediated waste site follows.

As presented in Table 6-25, the potential HI for direct contact from noncancer effects without background contribution is greater than the target HI of 1 in staging pile material from one remediated waste site and is less than the target HI in staging pile material from three remediated waste sites.

Staging pile area material associated with the 128-H-1 waste site reports a HI greater than the MTCA ("Human Health Risk Assessment Procedures" [WAC 173-340-708(5)]) target HI of 1. The HI for the residential scenario is as follows:

- 128-H-1 (staging pile area 2) reports an HI of 2.3. The primary contributor to the HI is arsenic (HQ = 2.3; > 99 percent contribution).

A comparison of arsenic and lead EPCs to their respective Method A soil cleanup level of 20 mg/kg and 250 mg/kg, respectively is provided in Table G-45. Except for the arsenic (40.5 mg/kg) EPC reported in staging pile material associated with remediated waste site 128-H-1, all arsenic and lead EPCs are less than the Method A soil cleanup level of 20 mg/kg and 250 mg/kg respectively, for unrestricted land use. A summary of the arsenic results in staging pile material associated with the 128-H-1 remediated waste site follows:

- Twelve soil samples were collected from the staging pile area 2 decision unit and analyzed for arsenic. Arsenic concentrations range between 12.9 and 97.7 mg/kg (nine results greater than the Method A soil cleanup level of 20 mg/kg). Arsenic concentrations greater than the Method A cleanup level range between 24.8 and 97.7 mg/kg.

Nonradiological Results (Inhalation). As presented in Table G-51 (Appendix G), the potential cumulative ELCR for the inhalation pathway from all nonradiological carcinogenic COPCs without background contribution ranges from 1.6×10^{-14} to 1.3×10^{-9}. The total cumulative ELCR is less than the MTCA ("Cleanup Standards to Protect Air Quality" [WAC 173-340-750]) Method B risk value of 1×10^{-6} for individual carcinogens for staging piles associated with the four remediated waste sites.

As presented in Table G-51 (Appendix G), the potential HI for the inhalation pathway from noncancer effects without background contributions is less than the EPA target HI of 1 and the MTCA ("Cleanup Standards to Protect Air Quality" [WAC 173-340-750]) Method B target HI of 1 for staging piles associated with four remediated waste sites.

Deep Zone. Deep vadose zone samples were evaluated to identify remediated waste sites where exposure to residual contamination could present a potential risk from an inadvertent exposure through deep excavation activities. While industrial in nature, the RBSLs (developed for the Residential exposure scenario) were used for convenience as screening values to identify such sites in order to allow institutional controls to be established to control access to deep contamination.

Twelve remediated waste sites are reported with CVP/RSVP data associated with the deep zone in the 100-H Source OU:

- Five remediated waste sites were sampled using a statistical sampling design.
- One remediated waste site was sampled using a focused sampling design.

- One remediated waste site was sampled using both a statistical and a focused sampling design (consisting of three statistical decision units and two focused decision units).

The remaining 35 remediated waste sites were not excavated deeper than 4.6 m (15 ft) bgs and are not discussed in this section. The Residential scenario results for the deep vadose zone are summarized by decision unit in Table G-52 (Appendix G).

Radiological Results. As presented in Table 6-26, the total ELCR is greater than the upper risk threshold of 1×10^{-4} for nine remediated waste sites and is within the target risk range of 10^{-4} to 10^{-6} for one remediated waste site. Risks were not reported at one remediated waste site because radiological COPC concentrations were less than background. Radiological COPCs were not detected at one remediated waste site.

6.2.5.5.2 *Resident Monument Worker Scenario*

PRGs developed for the Resident Monument Worker scenario represent reasonably anticipated future land use. The results of this comparison are used to confirm that cleanup actions are protective of the reasonably anticipated future land uses that DOE and the USFWS anticipate for the River Corridor. The Resident Monument Worker scenario is described in Section 6.2.3.3.

For completeness in analysis, all risk estimates for each remediated waste site decision unit are provided in Appendix G. The risk estimates, which includes all radiological COPCs regardless of their EPCs relative to the background concentration are presented in Tables G-53 through G-56 (100-D Resident Monument Worker scenario) and Tables G-64 through G-67 (100-H Resident Monument Worker scenario).

Appendix G also includes risk estimates for each remediated waste site decision unit, which include only those radiological COPCs with EPCs greater than background values or that do not have a background value in Tables G-57 through G-60 (100-D Resident Monument Worker scenario in Appendix G) and Tables G-68 through G-71 (100-H Resident Monument Worker scenario in Appendix G). Only these results are discussed in the risk characterization because it is this information that is used for decisions concerning appropriate remedial actions.

100-D Source OU. Risk estimates were calculated for the shallow vadose zone material, overburden material, and staging pile area material decision units within a remediated waste site. Risk estimates were not calculated for the deep zone decision units because the direct contact exposure pathway is incomplete. The results for the Resident Monument Worker scenario are presented in Table G-57 (Appendix G).

An overall summary of the cumulative risk estimates for the resident Monument worker scenario for each of the remediated waste sites evaluated is provided in Table G-61 for the shallow zone, Table G-62 for overburden material, and Table G-63 for staging piles. These tables list the OU that each remediated waste site resides in, the reclassification status, the remediated waste site, the associated waste site (if applicable), the decision unit reported with an exceedance (if applicable), the total ELCR, and the risk driver and percent contribution (if applicable).

Shallow Zone. As presented in Table G-61, the total ELCR for radionuclides is greater than the upper risk threshold of 1×10^{-4} at two remediated waste sites, is within the target risk range of 10^{-4} to 10^{-6} at 29 remediated waste sites, and is less than the lower risk threshold of 1×10^{-6} at eight remediated waste sites. Risks were not reported at 21 remediated waste sites because radiological COPC concentrations were less than background. Radiological COPCs were not reported at 32 remediated waste sites. Following are the results of the Resident Monument Worker scenario compared to the Residential scenario.

Nine remediated waste sites report concentrations of Hanford Site-related COPCs that are equal to or exceed the upper range of the target threshold for the Residential scenario (see Table 6-13). Whereas only two remediated waste sites report concentrations of Hanford Site-related COPCs that exceed the upper range of the target threshold for the resident Monument worker scenario. Following are the cancer risk levels for the resident Monument worker scenario:

- The 116-D-8 waste site (shallow focused 2 decision unit) reports a total ELCR of 1.2×10^{-4} for the resident Monument worker. The primary contributor to risk is cesium-137 (1.2×10^{-4}; 100 percent contribution).
- The 116-DR-9 waste site (shallow decision unit) reports a total ELCR of 1.8×10^{-4} for the resident Monument worker. The primary contributors to risk include cesium-137 (1.6×10^{-4}; 92 percent contribution) and europium-152 (1.1×10^{-5}; 6.5 percent contribution).

Nonradiological Results (Direct Contact and Inhalation). As described in Section 6.2.3.3, the MTCA Method B Soil Cleanup Levels ("Unrestricted Land Use Soil Cleanup Standards" [WAC 173-340-740]) are the PRG values used to achieve protection of the resident Monument worker. Table 6-14 provides the results for the residential scenario.

Overburden. As presented in Table G-62, the total ELCR for radionuclides is within the target risk range of 10^{-4} to 10^{-6} for overburden material associated with six remediated waste sites and less than the lower risk threshold of 1×10^{-6} for overburden material associated with ten remediated waste sites. Risks were not reported for seven remediated waste sites because radiological COPC concentrations were less than background. Radiological COPCs were not reported at nine remediated waste sites.

Results for the Residential scenario are similar to the Resident Monument Worker scenario, as overburden material associated with remediated waste sites did not exceed the upper risk threshold of 1×10^{-4}.

Nonradiological Results (Direct Contact and Inhalation). As described in Section 6.2.3.3, the MTCA Method B Soil Cleanup Levels ("Unrestricted Land Use Soil Cleanup Standards" [WAC 173-340-740]) are the PRG values used to achieve protection of the resident Monument worker. Table 6-16 provides the results for the residential scenario.

Staging Pile Area. As presented in Table G-63, the total ELCR for radionuclides is less than the lower target threshold of 1×10^{-6} for staging pile area material associated with four remediated waste sites. Risks were not reported at two remediated waste sites because radiological COPC concentrations were less than background. Radiological COPCs were not reported at five remediated waste sites.

Results for the Residential scenario are similar to the Resident Monument Worker scenario, as staging piles associated with remediated waste sites did not exceed the upper risk threshold of 1×10^{-4}.

Nonradiological Results (Direct Contact and Inhalation). As described in Section 6.2.3.3, the MTCA Method B Soil Cleanup Levels ("Unrestricted Land Use Soil Cleanup Standards" [WAC 173-340-740]) are the PRG values used to achieve protection of the resident Monument worker. Table 6-18 provides the results for the residential scenario.

100-H Source OU. Risk estimates were calculated for the shallow vadose zone material, overburden material, and staging pile area material decision units within a remediated waste site. Risk estimates were not calculated for the deep zone decision units because the direct contact exposure pathway is incomplete (that is, samples are collected from depths greater than 4.6 [15 ft] bgs). The results without background contribution for the Resident Monument Worker scenario results are presented in Table G-68 (Appendix G).

An overall summary of the cumulative risk estimates for the resident Monument worker scenario from each of the remediated waste sites evaluated is provided in Table G-72 for the shallow zone material, Table G-73 for overburden materials, and Table G-74 for staging piles. These tables list the OU that each remediated waste site resides in, the reclassification status, the remediated waste site, the associated waste site (if applicable), the decision unit reported with an exceedance (if applicable), the total ELCR, and the risk driver and percent contribution (if applicable).

Shallow Zone. As presented in Table G-72, the potential total ELCR for radionuclides is within the target risk range of 10^{-4} to 10^{-6} for ten remediated waste sites and less than the lower risk threshold value of 1×10^{-6} for ten remediated waste sites. Risks were not reported at five remediated waste sites because COPC concentrations were less than background. Radiological COPCs were not reported at 17 remediated waste sites. Following are the results of the Resident Monument Worker scenario compared to the Residential scenario.

Two remediated waste sites report concentrations of Hanford Site-related COPCs that exceed the upper risk threshold of 1×10^{-4} for the Residential scenario (see Table 6-20). Whereas, shallow zone remediated waste sites do not exceed the upper risk threshold of 1×10^{-4} for the Resident Monument Worker scenario.

Nonradiological Results (Direct Contact and Inhalation). As described in Section 6.2.3.3, the MTCA Method B Soil Cleanup Levels ("Unrestricted Land Use Soil Cleanup Standards" [WAC 173-340-740]) are the PRG values used to achieve protection of the resident Monument worker. Table 6-21 provides the results for the residential scenario.

Overburden. As presented in Table G-73, the total ELCR for radionuclides is within the target risk range of 10^{-4} to 10^{-6} for overburden material associated with two remediated waste sites and is less than the lower risk threshold of 1×10^{-6} for overburden material associated with three remediated waste sites. Risks were not reported at two remediated waste sites because radiological COPC concentrations were less than background. Radiological COPCs were not reported at two remediated waste sites.

Results for the Residential scenario are similar to the Resident Monument Worker scenario, as overburden material associated with remediated waste sites did not exceed the upper risk threshold of 1×10^{-4}.

Nonradiological Results (Direct Contact and Inhalation). As described in Section 6.2.3.3, the MTCA Method B Soil Cleanup Levels ("Unrestricted Land Use Soil Cleanup Standards" [WAC 173-340-740]) are the PRG values used to achieve protection of the resident Monument worker. Table 6-23 provides the results for the residential scenario

Staging Pile Area. As presented in Table G-74, the total ELCR is within the target risk range of 10^{-4} to 10^{-6} for staging pile area material associated with one remediated waste site and is less than the lower risk threshold of 1×10^{-6} for staging pile area material associated with one remediated waste site. Risks were not reported in staging pile material associated with one remediated waste site because radiological COPC concentrations were less than background. Radiological COPCs were not reported in staging pile material associated with one remediated waste site.

Results for the Residential scenario are similar to the Resident Monument Worker scenario, as staging piles associated with remediated waste sites did not exceed the upper risk threshold of 1×10^{-4}.

Nonradiological Results (Direct Contact and Inhalation). As described in Section 6.2.3.3, the MTCA Method B Soil Cleanup Levels ("Unrestricted Land Use Soil Cleanup Standards" [WAC 173-340-740]) are the PRG values used to achieve protection of the resident Monument worker. Table 6-25 provides the results for the residential scenario.

6.2.5.5.3 *Casual Recreational User Scenario*

PRGs developed for the Casual Recreational User scenario represent reasonably anticipated future land use. The results of this comparison are used to confirm that cleanup actions are protective of the reasonably anticipated future land uses that DOE and USFWS anticipate for the River Corridor. The Casual Recreational User scenario is described in Section 6.2.3.3.

For completeness in analysis, risk estimates for each remediated waste site decision unit are provided in Appendix G, which includes all COPCs regardless of their EPCs relative to the background values. The risk estimates are provided in Tables G-75 through G-78 (100-D Casual Recreational User scenario) and Tables G-89 through G-92 (100-H Casual Recreational User scenario).

Appendix G also includes risk estimates for each remediated waste site decision unit, which include only those COPCs with EPCs greater than background values or that do not have a background value, in Tables G-79 through G-82 (100-D Casual Recreational User scenario) and Tables G-93 through G-96 (100-H Casual Recreational User scenario). Only these results are discussed in the risk characterization because it is this information that is used for decisions concerning appropriate remedial actions.

100-D Source OU. Risk estimates were calculated for the shallow vadose zone material, overburden material, and staging pile area material decision units within a remediated waste site. Risk estimates were not calculated for the deep zone decision unit because the direct contact exposure pathway is incomplete (that is, samples are collected from depths greater than 4.6 m [15 ft] bgs). The results for the Casual Recreational User scenario are presented in Table G-79 (Appendix G).

An overall summary of the cumulative risk estimates and noncancer hazards for the casual recreational user scenario for each of the remediated waste sites evaluated is provided in Table G-83 and Table G-84 for shallow zone material, Table G-85 and Table G-86 for overburden material, and Table G-87 and Table G-88 for staging piles. These tables list the OU that each remediated waste site resides in, the reclassification status, the remediated waste site, the associated waste site (if applicable), the decision unit reported with an exceedance (if applicable), the total ELCR, and the risk driver and percent contribution (if applicable).

Shallow Zone. As presented in Table G-83, the total ELCR for radionuclides is within the target risk range of 10^{-4} to 10^{-6} at 14 remediated waste sites and is less than 1×10^{-6} at 25 remediated waste sites. Risks were not reported at 21 remediated waste sites because radiological COPC concentrations were less than background. Radiological COPCs were not detected at 32 remediated waste sites. Following are the results of the Casual Recreational User scenario compared to the Residential scenario.

Nine remediated waste sites report concentrations of Hanford Site-related COPCs (radionuclides) that are equal to or exceed the upper range of the target threshold for the Residential scenario (see Table 6-13). Whereas shallow zone remediated waste sites did not exceed the upper range of the target threshold for the Casual Recreational User scenario.

As presented in Table G-84, the potential cumulative ELCR for nonradionuclides is within the target risk range of 10^{-4} to 10^{-6} for three remediated waste sites and less than 1×10^{-6} for 62 remediated waste sites. Risks were not reported at eight remediated waste sites because nonradiological carcinogenic COPC concentrations were less than background. Nonradiological carcinogenic COPCs were not detected at 19 decision units. Following are the results of the Casual Recreational User scenario compared to the Residential scenario.

For the Residential scenario, three remediated waste sites report individual carcinogens greater than the WAC 173-340-740 acceptable cancer risk level of 1×10^{-6}, two of these three remediated waste sites are greater than the MTCA ("Human Health Risk Assessment Procedures" [WAC 173-340-708(5)])

cumulative risk threshold of 1×10^{-5} (see Table 6-14). For the Casual Recreational User scenario, one remediated waste site (100-D-31:4) is greater than the MTCA ("Human Health Risk Assessment Procedures" [WAC 173-340-708(5)]) cumulative risk threshold of 1×10^{-5}.

As presented in Table G-84, the potential HI from direct contact for noncancer effects without background contribution is less than the EPA target HI of 1 for 69 remediated waste sites. An HI was not reported at 19 remediated waste sites because nonradiological COPC concentrations were less than background. Nonradiological COPCs were not detected at four remediated waste sites. The results of the Casual Recreational User scenario compared to the Residential scenario follow.

Noncancer hazards for the Residential scenario are similar to the Casual Recreational User scenario as shallow remediated waste sites were less than the EPA target HI of 1 and the MTCA ("Unrestricted Land Use Soil Cleanup Standards" [WAC 173-340-740]) target HI of 1.

Overburden. As presented in Table G-85,the total ELCR from all radiological COPCs without background contribution is within the target risk range of 10^{-4} to 10^{-6} for overburden material associated with two remediated waste sites and is less than the lower risk threshold of 1×10^{-6} for overburden material associated with 14 remediated waste sites. Risks were not reported at 11 remediated waste sites because radiological COPC concentrations were less than background. Radiological COPCs were not reported at five remediated waste sites. Following are the results of the Casual Recreational User scenario compared to the Residential scenario.

Results for the Residential scenario are similar to the Casual Recreational User scenario, as overburden material associated with remediated waste sites did not exceed the upper risk threshold of 1×10^{-4}.

As presented in Table G-86, the total ELCR for nonradionuclides from direct contact for all nonradiological carcinogenic COPCs without background contribution is within the target risk range of 10^{-4} to 10^{-6} for overburden material associated with one remediated waste site and is less than 1×10^{-6} for overburden material associated with 20 remediated waste sites. Risks were not reported for overburden material associated with nine remediated waste sites because nonradiological carcinogenic COPC concentrations were less than background. Risks were not reported for overburden material associated with two remediated waste sites because nonradiological carcinogenic COPCs were not reported. Following are results of the Casual Recreational User scenario compared to the Residential scenario.

For the Residential scenario, overburden material associated with three remediated waste sites report a total ELCR greater than 1×10^{-6}; however, there were no individual carcinogens reported with risks greater than the target risk level of 1×10^{-6}. For the Casual Recreational User scenario, overburden material associated with one remediated waste site reports a total ELCR greater than 1×10^{-6}; similarly, there were no individual carcinogens reported with risks greater than the target risk level of 1×10^{-6}.

As presented in Table G-86, the potential HI from direct contact for noncancer effects without background contribution is less than the EPA target HI of 1 for overburden material associated with 30 remediated waste sites. An HI was not reported for overburden material associated with two remediated waste sites because COPC concentrations were less than background. Following are results of the Casual Recreational User scenario compared to the Residential scenario.

Noncancer hazards for the Residential scenario are similar to the Casual Recreational User scenario as shallow remediated waste sites were less than the EPA target HI of 1 and the MTCA ("Unrestricted Land Use Soil Cleanup Standards" [WAC 173-340-740]) target HI of 1.

Staging Pile Area. As presented in Table G-87, the potential total ELCR from direct contact for all radiological COPCs without background contribution is less than the lower target risk threshold value of

1×10^{-6} for staging pile area material associated with four remediated waste sites. Risks were not reported in staging pile material associated with two remediated waste sites because radiological COPC concentrations were less than background. Radiological COPCs were not detected in staging pile area material associated with five remediated waste sites. Following are results of the Casual Recreational User scenario compared to the Residential scenario.

Results for the Residential scenario are similar to the Casual Recreational User scenario for radionuclides, as staging piles associated with remediated waste sites did not exceed the upper risk threshold of 1×10^{-4}.

As presented in Table G-88, the potential cumulative ELCR from direct contact for all nonradiological carcinogenic COPCs without background contribution is within the target risk range of 10^{-4} to 10^{-6} for staging pile area material associated with one remediated waste site and is less than 1×10^{-6} for staging pile material associated with 10 remediated waste sites. Following are results of the Casual Recreational User scenario compared to the Residential scenario.

For the Residential scenario, staging pile material associated with one remediated waste site reports individual carcinogens greater than the WAC 173-340-740 acceptable cancer risk level of 1×10^{-6}; however, it is less than the MTCA ("Human Health Risk Assessment Procedures" [WAC 173-340-708(5)]) cumulative risk threshold of 1×10^{-5} (see Table 6-18). For the Casual Recreational User scenario, staging pile material associated with one remediate waste site reports a total ELCR greater than 1×10^{-6}; similarly there were no individual carcinogens reported with risks greater than the target risk level of 1×10^{-6}.

As presented in Table G-88, the potential HI from direct contact for noncancer effects without background contribution is less than the EPA target HI of 1 for the staging pile area material associated with 11 remediated waste sites. Following are results of the Casual Recreational User scenario compared to the Residential scenario.

Noncancer hazards for the Residential scenario are similar to the Casual Recreational User scenario as shallow remediated waste sites were less than the EPA target HI of 1 and the MTCA ("Unrestricted Land Use Soil Cleanup Standards" [WAC 173-340-740]) target HI of 1.

100-H Source OU. Risk estimates were calculated for the shallow, overburden, and staging pile area decision units within a remediated waste site. Risk estimates were not calculated for the deep zone decision unit because the direct contact exposure pathway is incomplete (that is, samples are collected from depths greater than 4.6 m [15 ft] bgs). The results for the Casual Recreational User scenario are summarized by decision unit in Tables G-93 (Appendix G).

An overall summary of the total risk estimates and noncancer hazards (if applicable) for the casual recreational user scenario from each of the remediated waste sites evaluated are provided in Tables G-97 and G-98 for shallow zone material, Tables G-99 and G-100 for overburden material, and Tables G-101 and G-102 for staging piles. These tables list the OU that each remediated waste site resides in, the reclassification status, the remediated waste site, the associated waste site (if applicable), the decision unit reported with an exceedance (if applicable), the total ELCR and the risk driver and percent contribution (if applicable), and the hazard index and the hazard driver and percent contribution (if applicable).

Shallow Zone. As presented in Table G-97, the total ELCR from all radiological COPCs without background contribution is within the target risk range of 10^{-4} to 10^{-6} at four remediated waste sites and is less than 1×10^{-6} at 16 remediated waste sites. Risks were not reported at five remediated waste sites because radiological COPC concentrations were less than background. Radiological COPCs were not

reported at 17 remediated waste sites. Following are results of the Casual Recreational User scenario compared to the Residential scenario.

Two remediated waste sites report concentrations of Hanford Site-related COPCs that exceed the upper risk threshold of 1×10^{-4} for the Residential scenario (see Table 6-20). Whereas, shallow zone remediated waste sites do not exceed the upper risk threshold of 1×10^{-4} for the Casual Recreational User scenario.

As presented in Table G-98, the total ELCR from all nonradiological COPCs without background contribution is within the target risk range of 10^{-4} to 10^{-6} for 20 remediated waste sites and less than 1×10^{-6} for 17 remediated waste sites. Risks were not reported at five remediated waste sites because nonradiological carcinogenic COPC concentrations were less than background. Following are results of the Casual Recreational User scenario compared to the Residential scenario.

For the Residential scenario, 21 remediated waste sites report individual carcinogens greater than the WAC 173-340-740 acceptable cancer risk level of 1×10^{-6}, 19 of the remediated waste sites are greater than the MTCA ("Human Health Risk Assessment Procedures" [WAC 173-340-708(5)]) cumulative risk threshold of 1×10^{-5} (see Table 6-21). For the Casual Recreational User scenario, 20 remediated waste sites report individual carcinogens greater than the WAC 173-340-740 acceptable cancer risk level of 1×10^{-6}; however, all of the remediated waste sites are less than the MTCA ("Human Health Risk Assessment Procedures" [WAC 173-340-708(5)]) cumulative risk threshold of 1×10^{-5}.

As presented in Table G-98, the potential HI from direct contact for noncancer effects without background contribution is less than the EPA target HI of 1 for 40 remediated waste sites. An HI was not reported at two remediated waste sites because nonradiological COPC concentrations were less than background. Following are results of the Casual Recreational User scenario compared to the Residential scenario.

For the Residential scenario, two remediated waste sites report a HI greater than the EPA target HI of 1 and the MTCA ("Human Health Risk Assessment Procedures" [WAC 173-340-708(5)]) target HI of 1. For the Casual Recreational User scenario, noncancer hazards were less than the EPA target HI of 1 and the MTCA ("Unrestricted Land Use Soil Cleanup Standards" [WAC 173-340-740]) target HI of 1.

Overburden. As presented in Table G-99, the potential total ELCR from all radiological COPCs without background contribution is within the target risk range of 10^{-4} to 10^{-6} for overburden material associated with one remediated waste site and less than the lower risk threshold value of 1×10^{-6} for overburden material associated with four remediated waste sites. Risks were not reported in overburden material associated with two remediated waste sites because radiological COPC concentrations were less than background. Radiological COPCs were not detected in overburden material associated with two remediated waste sites. Following are results of the Casual Recreational User scenario compared to the Residential scenario.

Results for the Residential scenario are similar to the Casual Recreational User scenario, as overburden material associated with remediated waste sites did not exceed the upper risk threshold of 1×10^{-4}.

As presented in Table G-100, the potential cumulative ELCR from direct contact for all nonradiological carcinogenic COPCs without background contribution ranges is within the target risk range of 10^{-4} to 10^{-6} for overburden material associated with four remediated waste sites and is less than 1×10^{-6} for overburden material associated with five remediated waste sites. Following are results of the Casual Recreational User scenario compared to the Residential scenario.

For the Residential scenario, overburden material associated with four remediated waste sites report individual carcinogens greater than the WAC 173-340-740 acceptable cancer risk level of 1×10^{-6}; all four are also greater than the MTCA ("Human Health Risk Assessment Procedures"

[WAC 173-340-708(5)]) cumulative risk threshold of 1×10^{-5}. For the Casual Recreational User scenario, overburden material associated with four remediated waste sites report individual carcinogens greater than the MTCA ("Unrestricted Land Use Soil Cleanup Standards" [WAC 173-340-740]) acceptable cancer risk level of 1×10^{-6}; however, all remediated waste sites are less than the MTCA ("Human Health Risk Assessment Procedures" [WAC 173-340-708(5)]) cumulative risk threshold of 1×10^{-5}.

As presented in Table G-100, the potential HI from direct contact for noncancer effects without background contribution is less than the EPA target HI of 1 for overburden material associated with the nine remediated waste sites. Following are results of the Casual Recreational User scenario compared to the Residential scenario.

For the Residential scenario, one remediated waste site reports a HI greater than the EPA target HI of 1 and the MTCA ("Human Health Risk Assessment Procedures" [WAC 173-340-708(5)]) target HI of 1. For the Casual Recreational User scenario, noncancer hazards were less than the EPA target HI of 1 and the MTCA ("Unrestricted Land Use Soil Cleanup Standards" [WAC 173-340-740]) target HI of 1.

Staging Pile Area. As presented in Table G-101, the total ELCR from all radiological COPCs without background contributions is less than the lower risk threshold of 1×10^{-6} for staging pile area material associated with two remediated waste sites. Risks were not reported in staging pile material associated with one remediated waste site because radiological COPC concentrations were less than background. Radiological COPCs were not detected in staging pile area material associated with one remediated waste site. The results of the Casual Recreational User scenario compared to the Residential scenario follow.

Results for the Residential scenario are similar to the Casual Recreational User scenario for radionuclides, as staging piles associated with remediated waste sites did not exceed the upper risk threshold of 1×10^{-4}.

As presented in Table G-102, the total ELCR from direct contact for all nonradiological carcinogenic COPCs without background contribution is within the target risk range of 10^{-4} to 10^{-6} for staging pile area material associated with two remediated waste site and is less than 1×10^{-6} for staging pile area material associated with two remediated waste sites. The results of the Casual Recreational User compared to the Residential scenario follow.

For the Residential scenario, staging pile material associated with two remediated waste sites report individual carcinogens greater than the MTCA ("Unrestricted Land Use Soil Cleanup Standards" [WAC 173-340-740]) acceptable cancer risk level of 1×10^{-6}; both are also greater than the MTCA ("Human Health Risk Assessment Procedures" [WAC 173-340-708(5)]) cumulative risk threshold of 1×10^{-5} (see Table 6-25). For the Casual Recreational User scenario, staging pile material associated with two remediated waste sites report individual carcinogens greater than the WAC 173-340-740 acceptable cancer risk level of 1×10^{-6}; one staging pile associated with a remediated waste site is also greater than the MTCA ("Human Health Risk Assssment Procedures" [WAC 173-340-708(5)]) cumulative risk threshold of 1×10^{-5}.

As presented in Table G-102, the potential HI from direct contact for noncancer effects without background contribution is less than the EPA target HI of 1 for staging pile area material associated with the four remediated waste sites. The results of the Casual Recreational User scenario compared to the Residential scenario follow.

For the Residential scenario, staging pile material associated with one remediated waste site reports an HI greater than the EPA target HI of 1 and the MTCA ("Human Health Risk Assessment Procedures" [WAC 173-340-708(5)]) target HI of 1. For the Casual Recreational User scenario, noncancer hazards

were less than the EPA target HI of 1 and the MTCA ("Unrestricted Land Use Soil Cleanup Standards" [WAC 173-340-740]) target HI of 1.

6.2.6 Uncertainties in the Soil Risk Assessment

The purpose of this soil risk assessment is to determine whether a further remedial action is warranted under CERCLA. Estimating and evaluating health risk from exposure to environmental contaminants is a complex process with inherent uncertainties. Uncertainty reflects limitations in knowledge, and simplifying assumptions must be made to quantify health risks.

In this assessment, uncertainties are associated with sampling and analysis data, sampling design, the EPCs, radiological decay, exposure, toxicity assumptions, and risk characterization.

6.2.6.1 Uncertainties Associated with Sampling and Analysis Data

Sampling and analysis data used in this soil risk assessment represent post-remediation conditions of waste sites with a "no action" or an "interim closed out" remediation status. All soil samples were collected in accordance with the requirements stated in the 100 Area SAP (DOE/RL-96-22). These data were collected specifically to determine whether the remedial action processes implemented under the work plan met the RAOs and remedial action goals stated in the interim action RODs listed in Section 6.2.1.1.

Some uncertainties may be associated with the changing requirements associated with the analysis of COCs identified in each ROD. When remediation initially began in 1996 in the 100 Area, only those analytes identified as COCs were analyzed and reported by the laboratory. However, as remediation continued, analytical methods improved, guidance was superseded, and reporting requirements changed. Currently, analytes identified as COCs are analyzed using a methods-based approach, which requires each laboratory to report the concentration of the COC and all associated target analytes included in the analytical method.

Waste sites associated with the earliest interim action RODs are generally the radioactive high volume liquid effluent sites. In general, verification samples collected to determine whether RAOs had been met report fewer analytes than those that have been remediated more recently. The majority of waste sites typicallyinclude verification samples analyzed using a methods-based approach. These generally include burial grounds and waste sites identified during the discovery process. If a method-based approach were used, risks may be slightly higher but would remain protective of human health. This conclusion is supported by results of the method-based approach used for RI samples collected for this report.

6.2.6.2 Uncertainties Associated with Sampling Design and Exposure Point Concentrations

Calculating UCL for EPCs (OSWER 9285.6-10) recommends using a 95 percent UCL on the mean for estimating EPCs. Section 6.2.2.2 describes the methodology for calculating the EPCs for detected analytes.

When the following conditions were met, the maximum concentration rather than the 95 percent UCL was selected as the EPC:

- Samples are collected using a focused sampling design.
- A valid 95 percent UCL cannot be calculated because of a limited number of detections (fewer than five).
- A valid 95 percent UCL is greater than the maximum detected concentration.

When these conditions are met, statistical bias is introduced, resulting in the potential to overstate risk.

In addition, EPCs selected for shallow zone and deep zone decision units represent verification data collected from the floor and the sidewall of the excavated waste site. As a result, risks are likely overstated because the EPC does not take credit for the existing clean backfill that covers the remediated waste site.

6.2.6.3 Adjustments in EPCs Associated with Decay of Radioisotopes

Section 6.2.5.2 provides a summary of the risk estimates by exposure scenario evaluated. The results of the soil risk assessment for the Residential scenario identified a group of waste sites with concentrations of Hanford Site-related COPCs that result in individual risks greater than the upper risk threshold value of 1×10^{-4}. Table G-103 and Table G-104 (Appendix G) list the 100-D and 100-H waste sites and the applicable decision unit, each radioisotope reported for the waste site decision unit, the year the samples were collected, the EPCs, the half-life for each radioisotope, and the year that each radioisotope decays to an activity level equal to the residential RBSL. The tables also present the number of years required for radioisotope decay to reach a total risk estimate (based on all radionuclides reported) less than the upper risk threshold value of 1×10^{-4}. Deep vadose zone samples were evaluated to identify remediated waste sites where exposure to residual contamination could present a potential risk from an inadvertent exposure through deep excavation activities. While this exposure would be industrial in nature, the RBSLs (developed for the Residential exposure scenario) were used for convenience as screening values to identify such sites in order to allow institutional controls to be established to control access to deep contamination.

The elapsed time at which the activity level would decay below the residential RBSL is based on the radioactive decay law using the following equation:

$$\tau = \frac{\log \frac{A_E}{A_O}}{\log 0.5} \times \tau_{\frac{1}{2}}$$

where:

A_E = remaining amount of substance (the PRG) (pCi/g)

A_O = original amount of substance (the EPC) (pCi/g)

$\tau_{½}$ = half-life of the substance (years)

τ = elapsed amount of time (years)

The number of years required for total risk to be less than 1×10^{-4} (represented by "t") was back-calculated using the following inequality for a waste site with "n" radionuclides reported:

$$\left(\frac{EPC_1 \times 10^{\left[\frac{t}{t_{\frac{1}{2}}} \times log\left(\frac{1}{2}\right)\right]}}{PRG_1}\right) + \cdots + \left(\frac{EPC_n \times 10^{\left[\frac{t}{t_{\frac{1}{2}}} \times log\left(\frac{1}{2}\right)\right]}}{PRG_n}\right) < 1 \times 10^{-4}$$

The following lists the year that concentrations of radioisotopes currently measured in shallow decision units decay to activity levels less than residential RBSLs and the year that the total ELCR is less than 1×10^{-4}:

- Strontium-90 concentrations at 100-D-48:3 decayed to levels less than residential RBSLs in year 2007. Activities of all radionuclides decayed to a total ELCR of less than 1.0×10^{-4} in year 2009.

- Activities of all radionuclides decayed to a total ELCR of less than 1.0×10^{-4} by year 2009 at 100-D-47.
- Europium-152 and nickel-63 concentrations at 100-D-42, 100-D-43, and 100-D-45 decayed to a total ELCR of less than 1.0×10^{-4} in 2012.
- Strontium-90 concentrations at 118-H-1:1 decayed to levels less than the residential RBSL in year 2011. Activities of all radionuclides will decay to a total ELCR of less than 1.0×10^{-4} by year 2016.
- Strontium-90 concentrations at 116-H-5 decay to levels less than the residential RBSL in year 2013. Activities of all radionuclides will decay to a total ELCR of less than 1.0×10^{-4} by year 2016.
- Activities of all radionuclides decayed to a total ELCR of less than 1.0×10^{-4} by year 2022 at 118-D-6:4.
- Cesium-137 concentrations at 116-DR-9 decay to levels less than residential RBSLs in year 2035. Activities of all radionuclides will decay to a total ELCR of less than 1.0×10^{-4} by year 2038.
- Cesium-137 concentrations at 116-D-8 decay to levels less than residential RBSLs in year 2035. Activities of all radionuclides will decay to a total ELCR of less than 1.0×10^{-4} by year 2035.
- Technetium-99 is detected at the 118-DR-2:2 shallow decision unit at concentrations that result in risks above 1 × 10-4. Decay does not occur within a reasonable period for technetium-99 because the half-life is 213,000 years and is not included in the above calculations.

The following lists the year that concentrations of radioisotopes currently measured in deep decision units decay to activity levels less than residential RBSLs:

- Activities of all radionuclides decayed to a total ELCR of less than 1.0×10^{-4} by year 2019 at 100-H-21.
- Cesium-137 concentrations at 100-D-48:2 decayed to levels less than residential RBSLs in year 2003. Activities of all radionuclides will decay to a total ELCR of less than 1.0×10^{-4} by year 2034.
- Strontium-90 concentrations at 100-D-48:3 decayed to levels less than residential RBSLs in year 2008. Activities of all radionuclides will decay to a total ELCR of less than 1.0×10^{-4} by year 2028.
- Europium-152 concentrations at 100-D-49:4 decay to levels less than residential RBSLs in year 2016. Activities of all radionuclides will decay to a total ELCR of less than 1.0×10^{-4} by year 2027.
- Cesium-137 and strontium-90 concentrations at 118-H-6:2, 118-H-6:3,118-H-6:6, 100-H-9, 100-H-10, 100-H-11, 100-H-12, 100-H-13,100-H-14, and 100-H-31 decay to levels less than residential RBSLs in year 2069. Activities of all radionuclides will decay to a total ELCR of less than 1.0×10^{-4} by year 2108.
- Cesium-137 and europium-152 concentrations at 116-DR-6 decay to levels less than residential RBSLs in year 2026. Activities of all radionuclides will decay to a total ELCR of less than 1.0×10^{-4} by year 2048.
- Cesium-137 and europium-152 concentrations at 116-DR-9 decay to levels less than residential RBSLs in year 2037. Activities of all radionuclides will decay to a total ELCR of less than 1.0×10^{-4} by year 2064.
- Cesium-137, cobalt-60, and europium-152 concentrations at 116-H-3 decay to levels less than residential RBSLs in year 2036. Activities of all radionuclides will decay to a total ELCR of less than 1.0×10^{-4} by year 2056.

- Cesium-137 concentrations at 118-DR-2:2 decay to levels less than residential RBSLs in year 2041. Activities of all radionuclides will decay to a total ELCR of less than 1.0×10^{-4} by year 2140.
- Nickel-63 concentrations at 100-D-19 (focused) decay to levels less than residential RBSL in year 2041. Activities of all radionuclides will decay to a total ELCR of less than 1.0×10^{-4} by year 2042.
- Cesium-137 and europium-152 concentrations at 100-D-18 decay to levels less than residential RBSLs in year 2060. Activities of all radionuclides will decay to a total ELCR of less than 1.0×10^{-4} by year 2066.
- Cesium-137, cobalt-60, europium-152, europium-154, and strontium-90 concentrations at 116-H-7 decay to levels less than residential RBSLs in year 2098.
- Cesium-137, cobalt-60, europium-152, and europium-154 concentrations at 100-D-48:1 decay to levels less than residential RBSLs in year 2083. Activities of all radionuclides will decay to a total ELCR of less than 1.0×10^{-4} by year 2093.
- Cesium-137, cobalt-60, europium-152, europium-154, and nickel-63 concentrations at 116-D-7 decay to levels less than residential RBSLs in year 2083. Activities of all radionuclides will decay to a total ELCR of less than 1.0×10^{-4} by year 2125.
- Cesium-137, europium-152, europium-154, and strontium-90 concentrations at 116-H-1 decay to levels less than residential RBSLs in year 2102. Activities of all radionuclides will decay to a total ELCR of less than 1.0×10^{-4} by year 2110.
- Cesium-137, cobalt-60, and europium-152 concentrations at 100-D-49:2 decay to levels less than residential RBSLs in year 2113. Activities of all radionuclides will decay to a total ELCR of less than 1.0×10^{-4} by year 2117.
- Cesium-137, cobalt-60, europium-152, europium-154, and strontium-90 concentrations at 116-DR-1&2 decay to levels less than residential RBSLs in year 2122. Activities of all radionuclides will decay to a total ELCR of less than 1.0×10^{-4} by year 2148.
- Cesium-137 and europium-152 concentrations at 118-D-6:4 decay to levels less than residential RBSLs in year 2138. Activities of all radionuclides will decay to a total ELCR of less than 1.0×10^{-4} by year 2143.
- Cesium-137, cobalt-60, europium-152, europium-154, and strontium-90 concentrations at 116-D-1A decay to levels less than residential RBSLs in year 2196. Activities of all radionuclides will decay to a total ELCR of less than 1.0×10^{-4} by year 2203.

6.2.6.4 Uncertainties Associated with Exposure Assumptions

The exposure assumptions used to develop the RBSLs for each exposure scenario represent an RME. For estimating the RME, 95 percentile values (or upper-bound estimates of national averages) are generally used for exposure assumptions, and exposed populations and exposure scenarios are also selected to represent upper-bound exposures. The intent of the RME, as discussed by the EPA Deputy Administrator and the Risk Assessment Council "Guidance on Risk Characterization for Risk Managers and Risk Assessors" (Habicht, 1992) is to present risks as a range from central tendency to high-end risk (above the 90th percentile of the population distribution). This descriptor is intended to estimate the risks that are expected to occur in small but definable "high-end" segments of the subject population ("Guidance on Risk Characterization for Risk Managers and Risk Assessors" [Habicht, 1992]). EPA distinguishes between those scenarios that are possible but highly improbable and those that are

conservative but more likely to occur within a population, with the latter being favored in risk assessment. In general, these assumptions are intended to be conservative and yield an upper bound of the true risk or hazard.

6.2.6.5 Uncertainties Associated with Toxicity Assessment

The toxicological database was also a source of uncertainty. EPA has outlined some of the sources of uncertainty as defined in the risk assessment guide (EPA/540/1-89/002) and in Superfund HHT Risk Assessment Values (Cook, 2003). These sources may include or result from the extrapolation from high to low doses and from animals to humans. This is contingent on the species, gender, age, and strain differences in the uptake, metabolism, organ distribution, and target site susceptibility of a toxin. The human population's variability with respect to diet, environment, activity patterns, and cultural factors are also sources of uncertainty.

Traditionally, EPA has developed toxicity criteria for carcinogens by assuming that all carcinogens are nonthreshold contaminants. However, EPA recently has published revised cancer guidelines (*Guidelines for Carcinogen Risk Assessment I* [EPA/630/P-03/001F]) in which they have modified their former position of assuming nonthreshold action for all carcinogens. This new guidance emphasizes establishing the specific toxicokinetic mode of action that leads to development of cancer. In the future, toxicity criteria for carcinogens in the United States will be developed assuming no threshold for contaminants that exhibit genotoxic modes of action, or where the mode of action is not known. However, currently available EPA toxicity criteria for carcinogens were all derived assuming a nonthreshold model.

In most of the world, nonthreshold toxicity criteria are developed only for those carcinogens that appear to cause cancer through a genotoxic mechanism (International Toxicity Estimates for Risk database [TERA, 2011]). Specifically, for genotoxic contaminants, the cancer dose-response model is based on high-dose to low-dose extrapolation and assumes there is no lower threshold for the initiation of toxic effects. Cancer effects observed at high doses are found in laboratory animals or are extrapolated from occupational or epidemiological studies. Cancer effects observed at low doses are commonly found in environmental exposures. These models are essentially linear at low doses, so no dose is without some risk of cancer.

Slope Factors for Cr(VI). The oral RfD of 0.003 mg/kg-day published by IRIS is used to develop the MTCA ("Unrestricted Land Use Soil Cleanup Standards" [WAC 173-340-740]) direct contact soil cleanup level for Cr(VI). NJDEP has recently published an oral carcinogenic potency factor of 0.5 $(mg/kg\text{-}day)^{-1}$ (*Derivation of an Ingestion-Based Soil Remediation Criterion for Cr^{+6} Based on the NTP Chronic Bioassay Data for Sodium Dichromate Dihydrate* [NJDEP, 2009]). If the NJDEP value were used to calculate the MTCA ("Unrestricted Land Use Soil Cleanup Standards" [WAC 173-340-740]) direct contact soil cleanup level, the concentration would decrease from 240 mg/kg to 2.0 mg/kg. The use of the oral RfD published by IRIS may result in underestimating risk.

6.2.6.6 Uncertainties Associated with Risk Characterization

In the risk characterization, the assumption was made that the total risk of developing cancer from exposure to Hanford Site contaminants is the sum of the risk attributed to each individual contaminant. Likewise, the potential for the development of noncancer adverse effects is the sum of the HQs estimated for exposure to each individual contaminant. This approach, in accordance with EPA guidance, did not account for the possibility that constituents act synergistically or antagonistically, resulting in an overestimation or underestimation of risk.

6.2.6.6.1 *Uncertainties in Risk Estimates Associated with Remedial Investigation and Limited Field Investigation Soil Data*

In addition to the waste site closeout remediation data (CVP/RSVP), two additional sources of data were considered for use in the RI/FS and the soil risk assessment. These sources of data include: 1) vadose zone data collected for the RI to fill data gaps associated with the nature and extent of contamination or associated with understanding the fate and transport of contaminants, and 2) LFI data collected in 1992 from the 100-D/H OU. These data were collected for purposes other than fulfilling needs of the risk assessment; as such, they were not used to evaluate quantitative risks. However, these data were evaluated qualitatively by comparing concentrations of analytes to RBSLs to determine whether the results could be useful for risk management decisions. The results of this comparison are provided in Appendix G, Attachment G-1.

Soil data identified as useful for informing risk management decisions include those collected to fill Data Gaps 2, 3, and 7. Chapter 2, Table 2-1 lists the data gaps and the work conducted per the 100-D/H Work Plan (DOE/RL-2008-46-ADD1). Twelve boreholes (7 from 100-D and 5 from 100-H), 5 test pits (3 from 100-D and 2 from 100-H), and 14 monitoring wells (7 from 100-D and 7 from 100-H) were drilled for the RI. In general, the comparison of soil concentrations from RI data to RBSLs are consistent with those risk results reported for closeout documentation data (CVP/RSVP), because most boreholes and test pits were collected through waste sites that were previously remediated.

In the early 1990s, an LFI was performed in the 100-DR-1 and 100-DR-2 OUs, the 100-HR-1 and 100-HR-2 OUs, and the 100-HR-3 OU. Results of the qualitative risk evaluation show elevated risk results at some waste sites. However, use of the LFI data over state risks because these waste sites have been subsequently remediated under the interim action ROD.

6.3 Groundwater Risk Assessment

EPA guidance provided in "Summary of Key Existing EPA CERCLA Policies for Groundwater Restoration" (Woolford and Reeder, 2009, page 4), clarifies EPA's policies for determining whether a groundwater remedial action is warranted under CERCLA. In discussing the role of the baseline risk assessment, "Summary of Key Existing EPA CERCLA Policies for Groundwater Restoration" (Woolford and Reeder, 2009) quotes the preamble to the NCP (40 CFR 300):

"The results of the baseline risk assessment are used to determine whether remediation is necessary, to help provide justification for performing remedial action, and to assist in determining what exposure pathways need to be remediated."

"Summary of Key Existing EPA CERCLA Policies for Groundwater Restoration" (Woolford and Reeder, 2009) then continues to clarify when a CERCLA remedial action is appropriate (page 5):

"A CERCLA remedial action generally is appropriate[12] in various circumstances, including: a standard that helps define protectiveness (e.g., a federal or state MCL or nonzero MCLG for current or potential drinking water aquifers) is exceeded; when the estimated risk calculated in a risk assessment exceeds a noncarcinogenic level for an adverse health effect or the upper end of the NCP risk range for 'cumulative carcinogenic site risk to an individual based on reasonable maximum exposure for both current and future land use;[13] the noncarcinogenic hazard index is greater than one (using reasonable maximum exposure assumptions for either the current or reasonably anticipated future land use); or the site contaminants

[12] See EPA 540-R-97-013, *Rules of Thumb for Superfund Remedy Selection.*

[13] See Clay, 1991, "Role of the Baseline Risk Assessment in Superfund Remedy Selection Decisions."

cause adverse environmental impacts.[14] It is important to note that all conditions do not need to be present for action and the conditions may be independent of each other."

EPA guidance provided in "Role of the Baseline Risk Assessment in Superfund Remedy Selection Decisions" (Clay, 1991) describes how to use the baseline risk assessment to make risk management decisions such as determining whether remedial action under CERCLA Section 104 or Section 106 is necessary. The "Role of the Baseline Risk Assessment in Superfund Remedy Selection Decisions" (Clay, 1991) describes the following conditions when a CERCLA action is generally warranted:

- The baseline risk assessment indicates that a cumulative site risk to an individual using RME assumptions for either current or future land use exceeds the 10^{-4} ELCR end of the risk range.
- For groundwater actions, MCLs and nonzero maximum contaminant limit goals (MCLGs) will generally be used to gauge whether remedial action is warranted.
- Chemical-specific standards that define acceptable risk levels also may be used to determine whether an exposure is associated with an unacceptable risk to human health and the environment and whether remedial action is warranted.

Protectiveness of human health is evaluated by comparing groundwater concentrations within the groundwater OU to existing federal or state MCLs or nonzero MCLGs. Protectiveness of aquatic receptors is determined by the comparison of groundwater concentrations at the point of discharge to surface water to water quality criteria established under Section 304 or Section 303 of the *Clean Water Act of 1977* as well as Washington State water quality standards. The point of compliance for surface water cleanup levels is defined in the MTCA ("Surface Water Cleanup Standards" [WAC 173-340-730(7)(a)]) as the point or points at which hazardous substances are released to surface waters of the state. MTCA ("Surface Water Cleanup Standards" [WAC 173-340-730(7)(b)]) indicates that no mixing zone shall be allowed to demonstrate compliance with surface water cleanup levels.

Groundwater concentrations are compared to MTCA ("Ground Water Cleanup Standards" [WAC 173-340-720]) and MTCA ("Surface Water Cleanup Standards" [WAC 173-340-730]) to determine whether EPCs result in a HI greater than one. The EPCs also are used to calculate ELCRs that are compared to the upper end of the NCP (40 CFR 300) risk range for cumulative carcinogenic site risk to an individual based on RME for both current and future land use.

EPA guidance provided in "Clarification of the Role of Applicable, or Relevant and Appropriate Requirements in Establishing Preliminary Remediation Goals under CERCLA" (Fields, 1997) clarifies the relationship between two statutory mandates of CERCLA: (1) protect human health and the environment, and (2) attain or waive, if justified, based on site-specific circumstances, ARARs. It remains EPA's policy that ARARs will generally be considered protective, absent multiple contaminants or pathways of exposure. However, the guidance clarifies that, in rare situations, even absent multiple pathways or contaminants, PRGs should be set at levels more protective than required by a given ARAR, where application of the ARAR would not be protective of human health and the environment.

The RCBRA (DOE/RL-2007-21) evaluated groundwater data collected from 1998 to 2008. During the development of the Integrated Work Plan (DOE/RL-2008-46) approximately one year of additional groundwater data were collected and evaluated. The Integrated Work Plan (DOE/RL-2008-46) identified the need to collect representative spatial and temporal samples from a subset of wells. These data were collected over an 8-month period between October 7, 2009 and June 11, 2010. In this RI/FS, three

[14] See EPA 540-R-97-013, *Rules of Thumb for Superfund Remedy Selection.*

analyses of groundwater data are presented. Section 4.4.1.2 uses all available information to describe the nature and extent of contamination in groundwater. Section 4.4.1.2 provides summary statistics for groundwater data collected over the last 6 years that were used for COPC identification. Finally, Section 6.3 provides the groundwater risk assessment based on the representative spatial and temporal dataset.

A groundwater risk assessment was performed for the 100-HR-3 Groundwater OU. The 100-HR-3 Groundwater OU includes all groundwater in the 100-D, 100-H and horn area impacted by waste sites. There are four primary groundwater plumes within the 100-HR-3 Ground OU. Contaminant plume areas are identified geographically as the 100-D southern plume, 100-D northern plume, 100-H plume, and horn area plume, and are mainly based on the distribution of Cr(VI) concentrations. Other contaminants are primarily collocated with the Cr(VI) plume. The 100-HR-3 Groundwater OU risk assessment followed the strategy outlined as follows:

- Evaluate current groundwater data to identify contaminants present in groundwater in the OU. Analytical measurement data collected to resolve spatial, chemical, and temporal uncertainties described in the Integrated Work Plan (DOE/RL-2008-46) were used.
- Identify action levels for detected contaminants, using ARARs or risk based concentrations to establish a basis for screening analytes.
- Compare the detected contaminant concentrations to action levels in order to identify COPCs within the entire 100-HR-3 Groundwater OU. These analytes are also used to provide a comprehensive evaluation of contribution to cumulative risk and total hazard.
- Conduct the risk characterization step on the set of COPCs, including a determination that action levels have been exceeded.

Results of this groundwater risk assessment indicate that concentrations of contaminants in the 100-HR-3 Groundwater OU exceed action levels, and warrant investigation in an FS to address groundwater contamination within the OU. The COPCs represent contaminants that will be evaluated in the FS to define the COCs and to develop and select remedial alternatives. The groundwater risk assessment identifies multiple contaminants within the 100-D Source, horn, and 100-H Source exposure areas that exceed chemical-specific ARARs or risk-based concentrations. HHRA Procedures (WAC 173-340-708(5)(a), and WAC 173-340-708(6)(b)) require that cleanup levels be adjusted downward to take into account exposure to multiple hazardous substances or multiple pathways of exposure. This adjustment needs to be made only if, without this adjustment, the HI would exceed 1, or the total ELCR would exceed 1 in 100,000 (1×10^{-5}).

Additionally, several local and regional Tribes have ancestral ties to the Hanford Reach of the Columbia River and surrounding lands. DOE has requested that each Tribe provide an exposure scenario that reflects their traditional activities. At this time, the CTUIR (*Exposure Scenario for CTUIR Traditional Subsistence Lifeways* [Harris and Harper, 2004]) and the Yakama Nation (*Yakama Nation Exposure Scenario for Hanford Site Risk Assessment* [Ridolfi, Inc., 2007]) have provided scenarios. A quantitative groundwater risk assessment is included for both Tribal use scenarios to evaluate each of the potentially complete groundwater exposure pathways. The results for the Native American risk assessment are provided in *Native American Risk Assessment for the 100-HR-3 Groundwater Operable Unit* (ECF-100HR3-10-0477) (Appendix G). Section 6.3.7 provides a summary of this evaluation.
A quantitative evaluation of human health risk to a resident from exposure to tap water is included for comparison to the Native American Risk Assessment. This comparison is provided because the Native American scenarios and the EPA tap water scenario include the same exposure pathways and exposure routes but have different exposure assumptions. The EPA tap water scenario includes RME assumptions

whereas the Native American scenarios include high-end exposure assumptions. The Native American scenarios are discussed in more detail in the uncertainty section (Section 6.3.8.5.1). The results of the comparison show how the similarities and differences that result in use of RME and high-end assumptions. The results of the tap water risk assessment are provided in *Tap Water Risk Assessment for the 100-HR-3 Groundwater Operable Unit* (ECF-100HR3-10-0478) (Appendix G).

6.3.1 Findings of the River Corridor Baseline Risk Assessment

The RCBRA (DOE/RL-2007-21) provides a screening level groundwater risk assessment for the 100-HR-3 Groundwater OU to evaluate potential risks associated with groundwater exposure. The results of the groundwater screening level risk assessment indicate potential risk above EPA thresholds within the 100-HR-3 Groundwater OU. Noncancer chemical hazard results were also above the EPA's threshold value of 1.

Uncertainties associated with the groundwater dataset were identified in the RCBRA (DOE/RL-2007-21). These uncertainties relate to the ability of the groundwater dataset collected from 1998 to 2008 to represent current baseline conditions and potential exposure within each groundwater OU. Analytical data used for the screening level assessment were collected to fulfill a variety of state and federal regulations, including RCRA, CERCLA, the *Atomic Energy Act of 1954;* and Section 173 of the *Washington Administrative Code*. Although the monitoring data can be used for risk assessment purposes, there are uncertainties associated with its use. Specifically, target analytes, sampling frequencies, and MDLs (or reporting limits) are different between programs because the information is used to meet different requirements.

As a result of the uncertainties identified in the RCBRA, the Integrated Work Plan (DOE/RL-2008-46) added activities that would help reduce uncertainties, verify conclusions of the HHRA presented in the RCBRA, and ensure that no contaminants were inadvertently overlooked based on the use of the existing dataset. Section 3.6.5.1 of the Integrated Work Plan (DOE/RL-2008-46) identifies the following activities to reduce uncertainties:

Identify existing and/or install new monitoring wells that are spatially representative of the groundwater. This set of wells will represent locations where a receptor potentially could contact groundwater.

Conduct multiple rounds of sampling to obtain temporal representation of the unconfined aquifer from influence of river stage. Additional rounds of sampling at spatially representative monitoring wells will represent current groundwater conditions and capture the influence of river fluctuations on COPC concentrations.

Analyze all spatially representative monitoring wells for a focused list of groundwater COPCs identified for each round of sampling. Analyzing each of the monitoring wells for COPCs will provide a dataset that is representative of potential releases to the groundwater.

Evaluate sample results from characterization activities to support final remedial action decisions for groundwater.

The RCBRA evaluated exposure to groundwater for three residential scenarios (Subsistence Farmer, CTUIR Resident, and Yakama Resident scenarios) and the residential component of the resident Monument worker exposure scenario. Direct exposure to contaminants in groundwater was evaluated for household uses of groundwater in each of these scenarios, such as drinking and cooking (ingestion) and bathing (dermal absorption). If VOCs were measured in groundwater, indirect exposure by inhalation of VOCs in air may occur while bathing or when using groundwater in the home for other purposes. The inhalation pathway for VOCs associated with household use of groundwater is evaluated for VOCs that are identified

as COPCs in groundwater. Additionally, ingestion, inhalation, and dermal exposures to COPCs in groundwater used in a sweat lodge were evaluated in the CTUIR Resident and Yakama Resident scenarios.

The results of the screening level groundwater risk assessment provided in the RCBRA (DOE/RL-2007-21) identified Cr(VI) in the 100-HR-3 Groundwater OU as the primary contributor to risk through ingestion and dermal contact with groundwater.

6.3.2 Identification of Contaminants of Potential Concern

The first step of this groundwater risk assessment is data evaluation to select the COPCs for protection of human health and the environment. A preliminary COPC evaluation was conducted to support the 100-D/H Work Plan (DOE/RL-2008-46-ADD1) and the 100-D/H SAP (DOE/RL-2009-40). The work plan effort evaluated groundwater analytical data from the 100-HR-3 Groundwater OU collected over a 16-year period (1992 to 2008). Table 6-27 presents the 31 COPCs based on the evaluation of historical data in the work plan for the entire 100-HR-3 Groundwater OU.

Table 6-27. List of Historical Contaminants of Potential Concern in the 100-HR-3 Groundwater OU

Metals		
Antimony	Arsenic	Beryllium
Cadmium	Chromium	Cobalt
Copper	Cr(VI)	Lead
Manganese	Mercury	Nickel
Selenium	Silver	Thallium
Uranium	Vanadium	Zinc
Volatile Organic Compounds		
1,1-Dichloroethene	Benzene	Carbon tetrachloride
Chloroform	Trichloroethene	Vinyl chloride
Radiological		
Strontium-90	Technetium-99	Tritium
Anions		
Fluoride	Nitrate (as N)	Nitrite (as N)
Sulfate		

Source: *Sampling and Analysis Plan for the 100-DR-1, 100-DR-2, 100-HR-1, 100-HR-2, and 100-HR-3 Operable Units Remedial Investigation/Feasibility Study* (DOE/RL-2009-40), Table 1-2.

The COPCs identified during the work plan phase were validated by using groundwater samples analyzed using the methods documented in the 100-D/H SAP (DOE/RL-2009-40), Table 2-19. The groundwater dataset used for COPC identification consists of sampling and analysis data collected from 52 monitoring wells within the 100-HR-3 OU. The monitoring well network represents locations where human or ecological receptors could potentially encounter groundwater within the OU. The primary exposure pathway for humans is through groundwater obtained from a residential or community water well, assuming development of the land for future human habitation.

Identification of groundwater COPCs for the 100-HR-3 Groundwater OU is a two-step process. The first step of the process establishes a set of analytes that will be carried forward to identify COPCs for each of the exposure areas. These analytes are then evaluated in the risk characterization step of the groundwater risk assessment where a set of COPCs are identified for evaluation in the FS. Figure 6-6 presents a summary of the COPC identification process. The left side of Figure 6-6 illustrates the two steps used to identify COPCs. The right side of Figure 6-6 illustrates how the results of the Tap Water scenario analysis are used to confirm that COPCs are the major risk and hazard drivers.

The process used to identify data for COPC selection and the selection of action levels for this groundwater risk assessment are described in Sections 6.3.2.1 and 6.3.2.2. The methodology used to calculate EPCs is described in Section 6.3.2.3. The exposure assessment and toxicity assessment are presented in Sections 6.3.3 and 6.3.4, respectively. Finally, the risk characterization step for each of the exposure areas is described in Section 6.3.5 and 6.3.6, and the EPA Tap Water scenario is described in Section 6.3.7. The primary objective of this groundwater risk assessment is to provide information necessary to identify what remedial actions will be necessary in the remedy selected for the 100-HR-3 Groundwater OU.

6.3.2.1 Data Used to Identify Contaminants of Potential Concern

The sampling and analysis data used to identify COPCs were collected over an 8-month period between October 7, 2009 and June 11, 2010. Three sampling events were used to capture the effects that temporal fluctuations of river stage have on groundwater conditions. Samples collected from mid-May to mid-June 2010 represent the aquifer when the river stage is at its highest elevation. Samples collected from early October 2009 to early November 2009 represent the aquifer when the river is at its lowest elevation. Samples collected from mid-March to mid-April 2010 represent the aquifer when the river is transitioning from high to low river stage.

All monitoring wells used in this monitoring network were screened in the unconfined aquifer. All of the wells in the network were existing monitoring or compliance wells and are listed in Table 6-28, which lists each well in the 100-HR-3 Groundwater OU; Figure 6-7 shows their locations.

The analytical dataset for the 100-HR-3 Groundwater OU risk assessment is extracted from the HEIS database. After extraction, the analytical data are processed to obtain a single set of results per sampling location and time of collection. A total of 27,354 records were obtained from HEIS, and a total of 113 analytes were included in the dataset prior to analytical data processing. After analytical data processing (as described in the next section), the final dataset used for the COPC identification process contained a total of 16,202 records, with 113 analytes included in the dataset.

6.3.2.1.1 *Analytical Data Processing*

The dataset obtained from HEIS included the following types of information:

- Analytical results from both unfiltered and filtered samples
- Data qualification and data validation flags, including rejected results
- Results for a given analyte reported by more than one analytical method
- Parent, field duplicate, and field split sample results

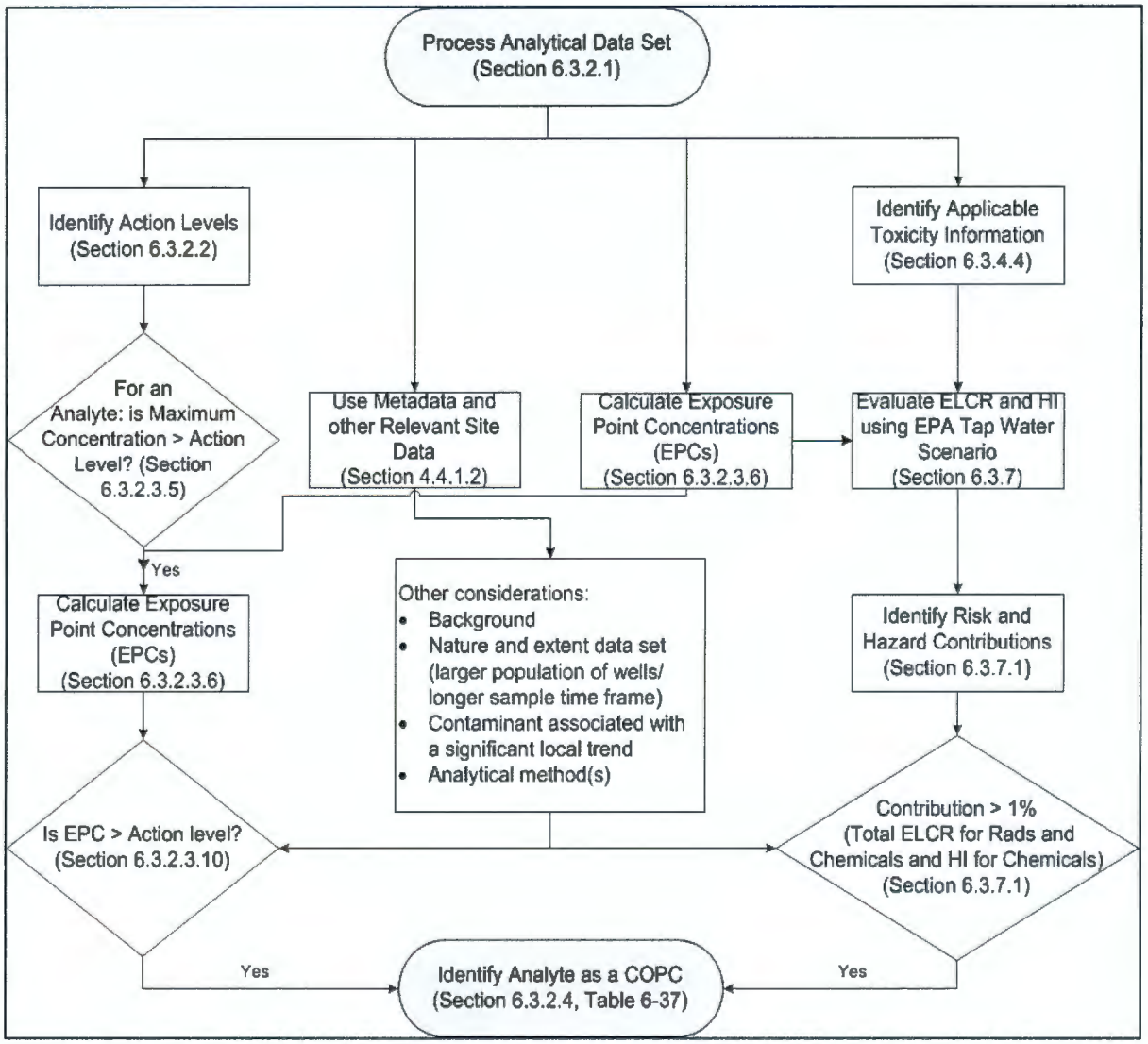

Figure 6-6. Overview of COPC Identification Process

Table 6-28. Monitoring Wells Used in the Groundwater Risk Assessment from the 100-HR-3 Groundwater OU

Well Name			
100-D Source Exposure Area			
199-D2-11	199-D2-6	199-D4-23	199-D4-84
199-D5-13	199-D5-14	199-D5-15	199-D5-16
199-D5-17	199-D5-18	199-D5-19	199-D5-37
199-D5-38	199-D5-43	199-D5-99	199-D8-5
199-D8-55	199-D8-70	199-D8-71	199-D8-88

Table 6-28. Monitoring Wells Used in the Groundwater Risk Assessment from the 100-HR-3 Groundwater OU

Well Name			
100-H Source Exposure Area			
199-H3-2A	199-H4-10	199-H4-11	199-H4-13
199-H4-16	199-H4-3	199-H4-45	199-H4-46
199-H4-48	199-H4-5	199-H4-6	199-H4-9
199-H6-1			
Horn Exposure Area			
199-H3-4	199-H3-5	199-H5-1A	699-101-45
699-87-55	699-90-45	699-93-48A	699-94-41
699-94-43	699-95-45	699-95-48	699-95-51
699-96-52B	699-97-41	699-97-45	699-97-48B
699-98-43	699-98-49A	699-98-51	

The analytical data were processed using the steps described in the following paragraphs and thus identify one set of results per sampling location and date of sample collection. The data processing steps and the numbers of records associated with each step are presented on Figure 6-8.

Descriptions of the data processing steps follow.

6.3.2.1.2 *Sample Results*

Only analytical results from unfiltered samples are used in identifying COPCs; results from filtered samples that may have been collected in support of other monitoring or compliance programs are excluded. Unfiltered sample results represent total concentrations of the analytes, while filtered sample results represent only dissolved concentrations. Use of filtered sampling results might lead to underestimation of chemical and radiological concentrations (for example, in water from an unfiltered tap).

The risk assessment guide (EPA/540/1-89/002) addresses this issue in providing guidance on estimating exposure concentrations in groundwater:

While filtration of groundwater samples provides useful information for understanding chemical transport within an aquifer, the use of filtered samples for estimating exposure is very controversial, because these data may underestimate chemical concentrations in water from an unfiltered tap. Therefore, data from unfiltered samples should be used to estimate exposure concentrations.

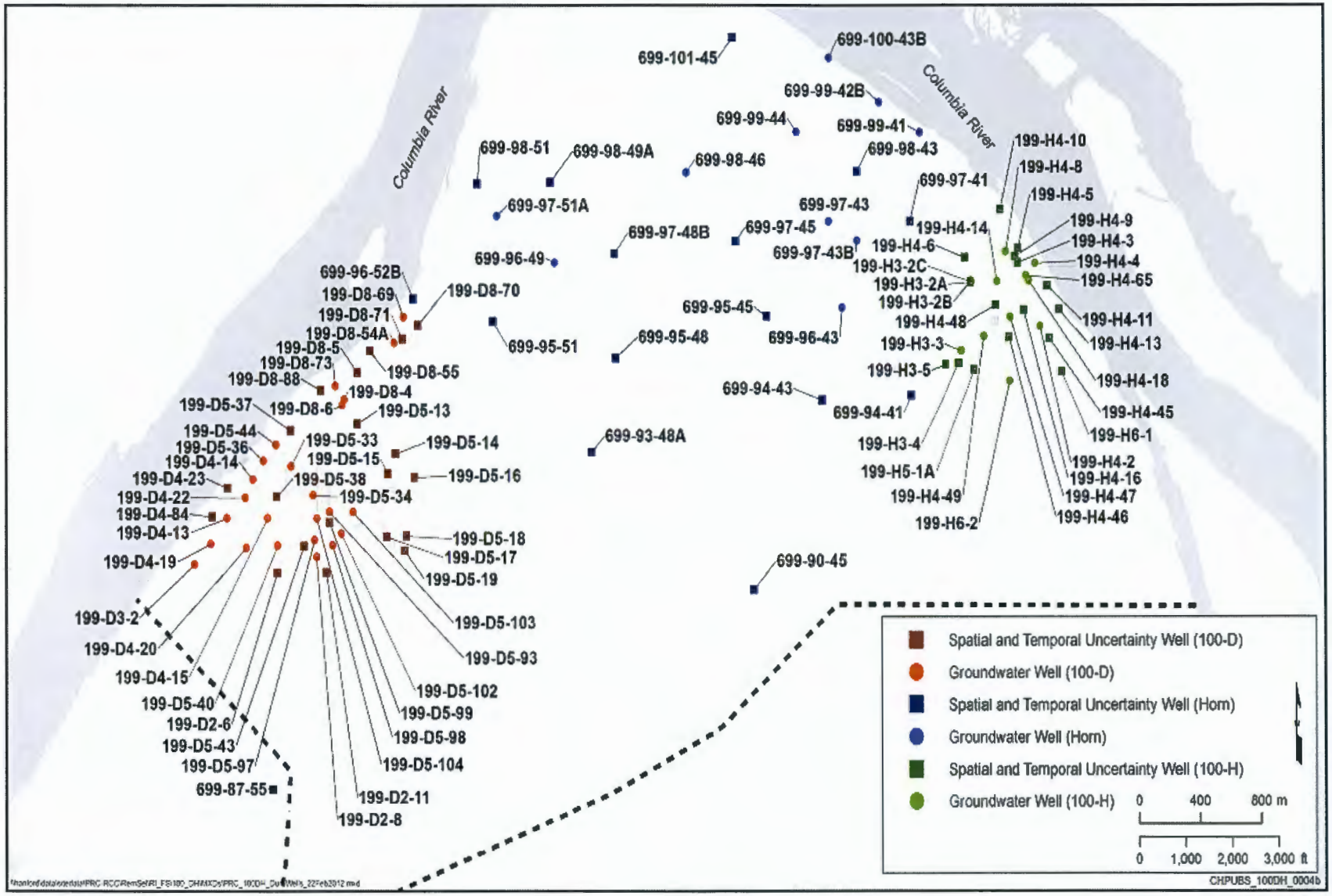

Figure 6-7. Monitoring Well Locations within the 100-HR-3 Groundwater OU

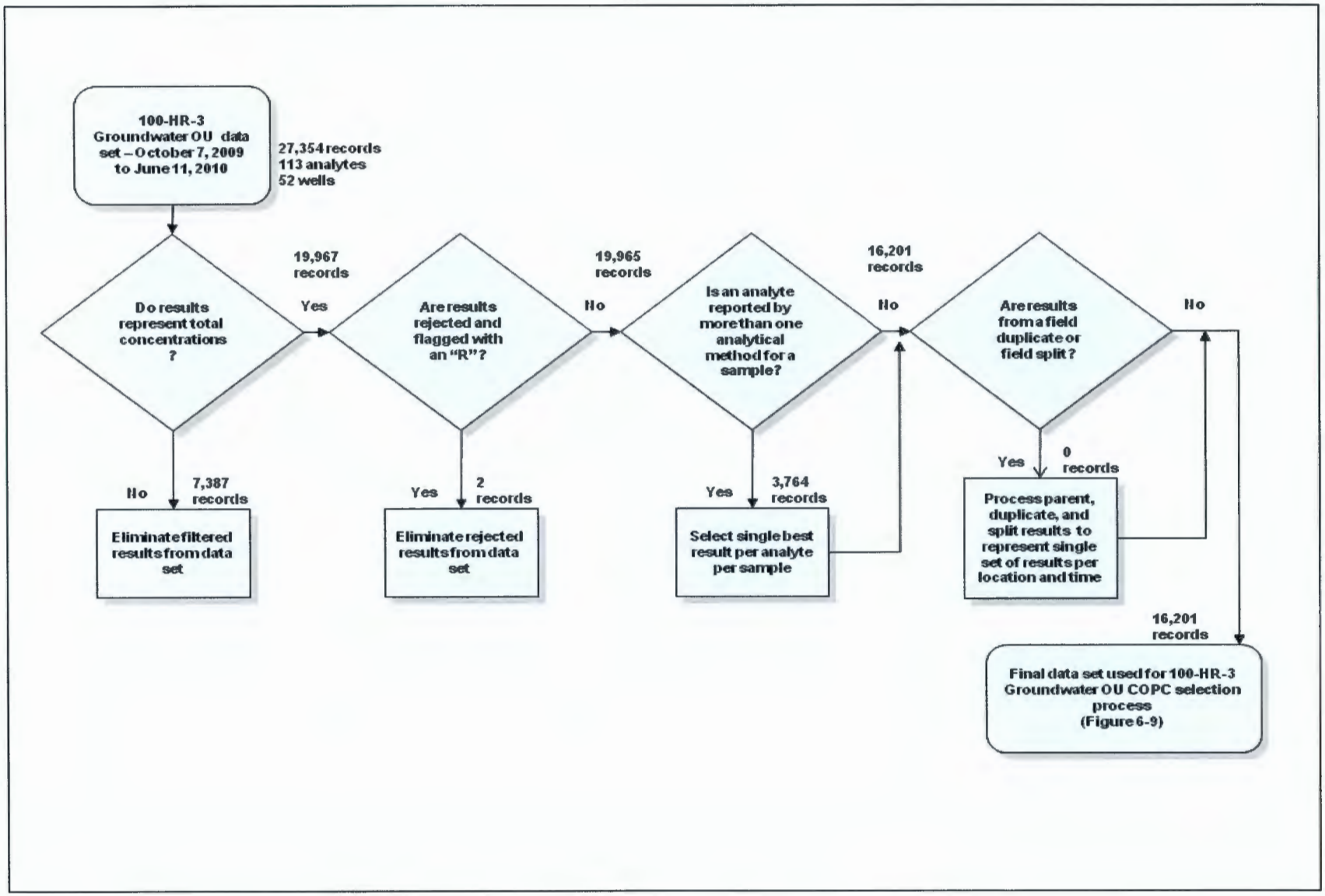

Figure 6-8. Data Processing and Reduction Steps for the 100-HR-3 Source OU

6.3.2.1.3 *Laboratory and Data Validation Flags*

Analytical data are received from the laboratory with data qualification flags. Validation qualifiers are assigned during the data validation process. The following rules determine how flagged and/or qualified sample results are used in identifying COPCs.

- Sample results flagged with a "U" data qualifier or combinations of qualifiers that include a "U," such as a "UJ," are considered nondetected results.
- Sample results without a "U" data qualifier are considered detected concentrations, including results with no qualifier or with a "J" data qualifier.
- Sample results that are rejected and flagged with an "R" validation qualifier are not used in identifying COPCs.

where:

U	=	Analyzed for but not detected above limiting criteria
J	=	Estimated value
R	=	Do not use. Further review indicates the result is not valid

6.3.2.1.4 *Analytes Reported by Numerous Analytical Methods*

Often analytes are reported by more than one analytical method. Therefore, multiple results for an analyte at the same location and sample date are possible. Because multiple sets of analytical results cannot be used to quantify risk (that is, this would result in multiple counting of a chemical), the set of data that best represents the actual concentration will be retained. The results are processed to select the method that provides the most reliable results. Considerations for determining data to be retained include method-associated sample size, detection frequency, and detection limits. The most conservative (that is, health-protective) use of these types of data will be the goal. Larger sample size, higher detection frequencies, and lower detection limits are given higher priority for method selection.

For example, lead may be analyzed using EPA Method 200.8 (*Methods for the Determination of Metals in Environmental Samples, Supplement I* [EPA-600/R-94/111]) with an EQL of 2 µg/L or EPA Method 6010 in SW 846 [SW-846] with an EQL of 50 µg/L. For a sample with lead concentrations reported using both methods, the results reported by EPA Method 200.8 (*Methods for the Determination of Metals in Environmental Samples, Supplement I* [EPA-600/R-94/111]) is selected over EPA Method 6010 (SW 846 [SW-846]) because of the more sensitive detection limit.

6.3.2.1.5 *Field Duplicate and Field Split Results*

Field QC samples (field duplicates and field splits) are collected in the field and analyzed by the laboratory as unique samples. The parent sample and QC samples are collected from the same location (that is, monitoring well) on the same date, resulting in more than one sample per location and date. The following criteria are used to reduce multiple sample results for an individual location and date to a single result:

- If two or more detections exist, the maximum concentration is used.
- If at least one detection and one or more nondetected results exist, the detected concentration is used.
- If only (two or more) nondetected results exist, the lowest detection limit is used.

6.3.2.2 Identify Action Levels

For the purpose of risk assessment and identification of COPCs, action levels are screening levels derived from chemical-specific ARARs and/or risk based concentrations using default exposure assumptions. All sources of action levels for each of the 113 analytes reported in the HEIS database for the 100-HR-3 OU are identified in Table 6-29.

Although the term "action level" is used for screening purposes, the term "action level" is not used to determine remediation levels nor does it imply that a groundwater action should be taken. Cleanup levels for groundwater contaminants are developed in the ROD.

The sources of action levels from federal regulations are:

- "National Primary Drinking Water Regulations" (40 CFR 141), MCLs, secondary MCLs, and nonzero MCLGs established under the *Safe Drinking Water Act of 1974* (SDWA)
- *National Recommended Water Quality Criteria* (EPA, 2009b), Ambient Water Quality Criteria (AWQC) established under Section 304 of the *Clean Water Act of 1977*
- "Water Quality Standards" (40 CFR 131) for states not complying with Section 303 of the *Clean Water Act of 1977*

The sources of the action levels from Washington State regulations are:

- "Water Quality Standards for Surface Waters of the State of Washington" (WAC 173-201A)
- "Groundwater Cleanup Standards" (WAC 173-340-720)
- "Surface Water Cleanup Standards" (WAC 173-340-730)
- "Group A Public Water Supplies," "Maximum Contaminant Levels (MCLs) and Maximum Residual Disinfectant Levels (MRDLs)" (WAC 246-290-310)

While surface water and AWQC standards are considered for the identification of action levels, it must be noted that these standards only apply for groundwater where it enters the Columbia River. For the upland parts of groundwater, only DWSs are applicable.

Derivation of State of Washington groundwater cleanup levels is provided in a separate calculation brief (*Calculation of Standard Method B Groundwater Cleanup Levels for Potable Groundwater for the 100 Areas and 300 Area Remedial Investigation/Feasibility Study Reports* [ECF-100NPL-10-0462]). Derivation of State of Washington surface water cleanup levels is provided in a separate calculation brief (*Calculation of Standard Method B Surface Water Cleanup Levels for the 100 Areas and 300 Area Remedial Investigation/Feasibility Study Reports* [ECF-100NPL-10-0463]).

6.3.2.3 COPC Identification Process

Section 6.3.2.1 defined the analytical dataset and described the analytical data processing steps used in this section for identifying groundwater COPCs. Section 6.3.2.2 identified the action levels used in this section for identifying groundwater COPCs. The COPC identification process described in this section is used to identify a subset of analytes to be carried forward into the risk characterization step provided in Section 6.3.5. This step of the process uses sampling and analysis data collected from the 52 monitoring wells in the 100-HR-3 Groundwater OU. The purpose of grouping all sampling and analysis data together from each exposure area (that is, the 100-D Source exposure area, the 100-H Source exposure area, and the horn exposure area) is to identify those analytes with detected concentrations above the lowest available action level before an EPC is calculated. A detailed description of the screening process is provided in *Identification of Contaminants of Potential Concern for Groundwater Risk Assessment at the 100-HR-3 Groundwater Operable Unit* (ECF-100HR3-10-0469) (Appendix G). The COPC identification steps, number of records, and number of analytes associated with each step are depicted on Figures 6-9 and 6-10 for the 100-HR-3 Groundwater OU and listed as follows:

- Apply exclusion criteria
- Identify nondetected analytes
- Identify analytes with maximum detected concentrations less than action levels
- Identify analytes with maximum detected concentrations greater than action levels
- Calculate EPCs for analytes with maximum detected concentrations greater than action levels
- Identify analytes with EPCs less than action level
- Identify analytes with EPCs greater than action level

6.3.2.3.1 *Summary of Nature and Extent Evaluation*

Section 4.4.1.2 presents the nature and extent of contamination in groundwater based on the last 6 years of data (that is, samples collected between January 2006 and December 2010). All monitoring wells within the 100-HR-3 Groundwater OU that are screened in the unconfined aquifer were included in the nature and extent evaluation (see Figure 6-7). The nature and extent evaluation reviewed all analytes identified as historical COPCs. As described previously, historical COPCs were identified in the work plan using data collected over a 16-year period (1992 to 2008) (see Table 6-27 for a list of historical COPCs). The nature and extent evaluation also includes the review of analytes that were not identified in Table 6-27 but report concentrations greater than an action level using the current RI data. The dataset used for the nature and extent evaluation is considered to be representative of current groundwater conditions based on the overall spatial coverage of monitoring wells across the OU and based on the inclusion of RI data that were collected to resolve uncertainties identified in the RCBRA (DOE/RL-2007-21) and the 100-D/H Work Plan (DOE/RL-2008-46-ADD1). This analysis is included to confirm that analytes that are identified as COPCs using RI data are consistent with the observations and characteristics of the data from a larger population of wells and analytical results collected over a longer period.

6.3.2.3.2 *Apply Exclusion Criteria*

The first step in the groundwater COPC identification process is to apply certain exclusion criteria. Analytes that met one or more of the exclusion criteria were eliminated as COPCs. The eliminated analytes are listed in Table 6-30. Analytes that did not meet any of the exclusion criteria were carried forward into the next step. Following are the exclusion criteria:

- Naturally occurring radionuclides associated with background radiation

Radionuclides that have half-lives of less than 3 years and are not significant daughter products

- Essential nutrients (minerals)
- Analytes without known toxicity information

One naturally occurring radionuclide associated with background radiation (potassium-40) was measured in groundwater from the 100-HR-3 Groundwater OU and was eliminated as a COPC.

Radioisotopes with half-lives less than or equal to 3 years are eliminated from further consideration, because only a small fraction of their original activity remains after 30 years of decay since the reactors ceased operation. Four radioisotopes met this exclusion criterion (antimony-125, beryllium-7, cesium-134, and ruthenium-106) and were eliminated from further consideration as COPCs. These radioisotopes were reported with nondetectable concentrations. Additionally, these isotopes are not significant daughter products associated with a decay chain.

Essential nutrients are those analytes considered essential for human nutrition. The essential nutrients calcium, magnesium, potassium and sodium were detected in the groundwater in the 100-HR-3 OU, but are excluded from further consideration as COPCs.

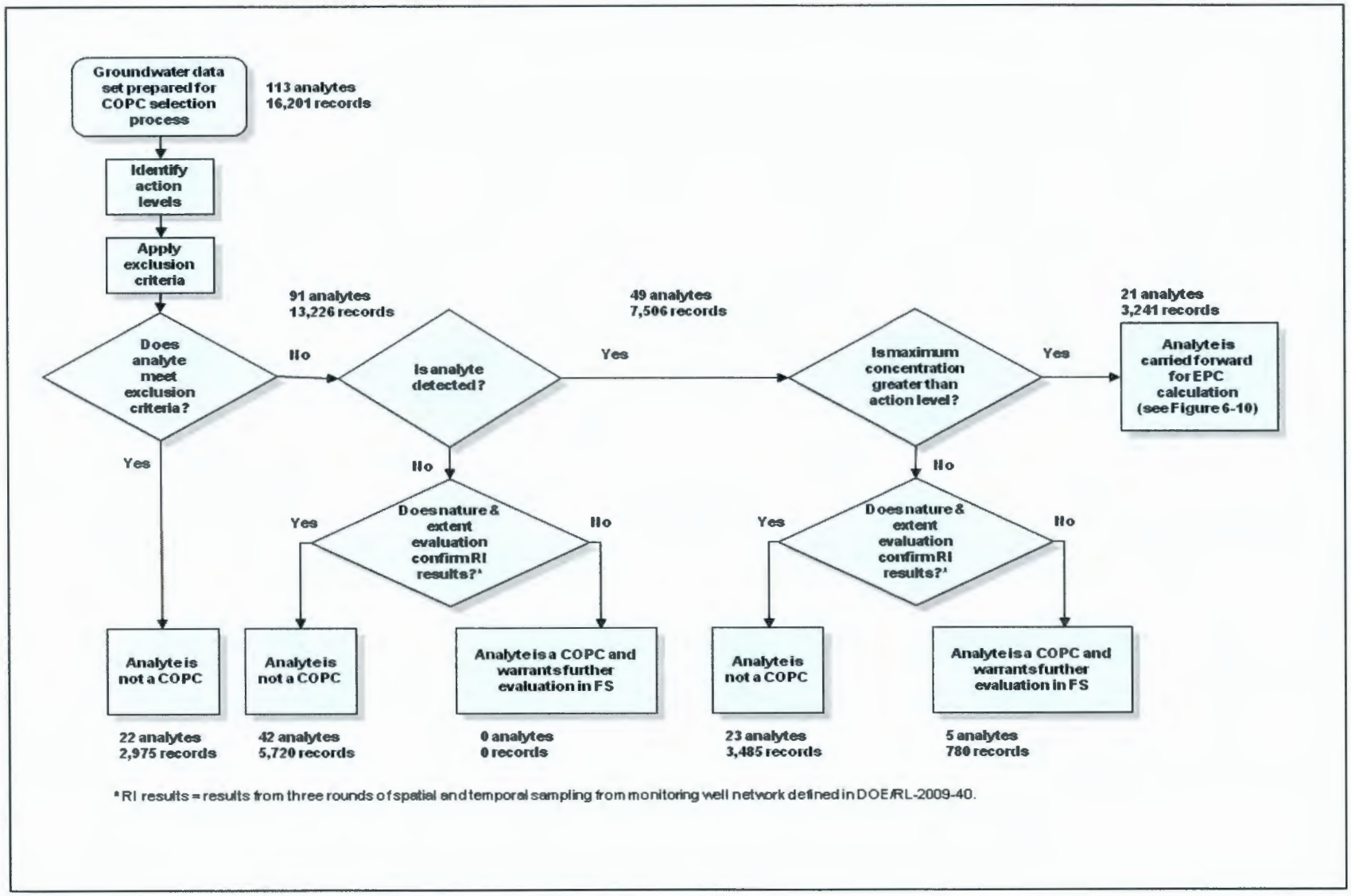

Figure 6-9. COPC Identification Process for the 100-HR-3 Source OU: Part 1

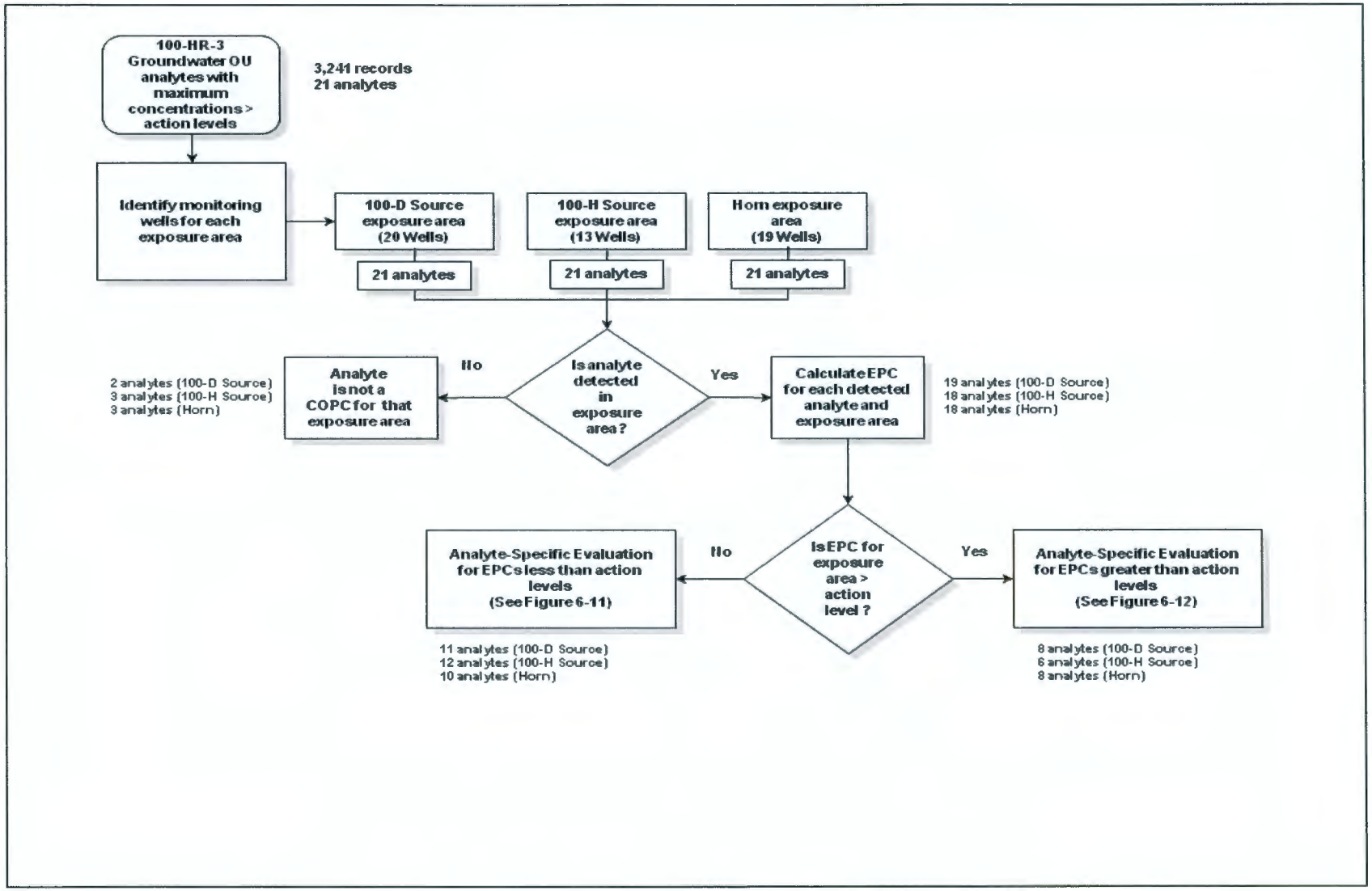

Figure 6-10. COPC Identification Process for the 100-HR-3 Groundwater OU: Part 2

Analytes without an action level were identified in Table 6-29. Because of the lack of promulgated standards (see Table 6-30), these analytes were not evaluated herein because this section focuses on comparing detected concentrations against action levels. However, the overall contribution of these analytes (and all other detections) were evaluated in the EPA Tap Water scenario (Section 6.3.7), using all available toxicity information. For example, chloromethane does not have a promulgated standard, but toxicity information is published and was used to evaluate the risk for this contaminant. For some analytes without an action level, toxicological information that could be considered in assessing any risks they may present is not available. Twelve analytes were eliminated from further consideration as COPCs because they do not have an action level nor do they have available toxicological information.

6.3.2.3.3 *Identify Nondetected Analytes*

The next step in the groundwater COPC identification process was to identify nondetected analytes. Chemicals and radionuclides that have been analyzed for, but not detected in any sample (collected from appropriate locations with adequate detection limits), were eliminated as COPCs. All analytes detected at least once were carried forward to the next step.

A total of 42 analytes were not detected in the 100-HR-3 OU groundwater samples collected for the RI. These analytes are listed in Table 6-31, each with sampling dates, minimum and maximum MDLs, the action level, the basis of the action level, and the level of exceedance.

Benzene, 1,1-dichloroethene, and vinyl chloride were not detected in the RI samples and were identified as historical COPCs in the work plan. These three analytes were not detected in samples collected specifically for the RI nor were they detected in the larger population of monitoring wells described previously in Section 4.4.1.2 or in Section 6.3.2.3.1. All MDLs associated with these analytes were less than the action level or the EQL (as applicable) listed in the 100-D/H SAP (DOE/RL-2009-40). Therefore, these three analytes are not COPCs and will not be carried forward into the FS.

6.3.2.3.4 *Identify Analytes with Maximum Detected Concentrations Less than Action Levels*[15]

This step identifies analytes with maximum concentrations less than action levels. In this screening, the maximum concentration of each analyte detected in groundwater was compared to its action level, to identify analytes not likely to contribute significantly to overall risk. If the maximum detected concentration of an analyte was less than its action level, the analyte was eliminated as a COPC, unless the nature and extent evaluation indicates otherwise.

Twenty-eight analytes were detected at least once and had maximum detected concentrations less than their respective action levels. A list of these analytes is presented in Table 6-32, each with sampling dates, minimum and maximum MDLs, minimum and maximum detected concentrations, the action levels, and the basis for each action level.

[15] See Section 6.3.2.2 for the definition of an action level.

Table 6-29. Summary of Federal and State Water Quality Criteria used as Action Levels for the 100-HR-3 Groundwater OU

				Groundwater					Surface Water									
				40 CFR 141[a]		WAC 246-290-310[b]	WAC 173-340-720[c]		Clean Water Act National Recommended Water Quality Criteria[d]			WAC 173-201A[e]	40 CFR 131[f]			WAC 173-340-730[g]	Action Level Value	
CAS Number	Analyte Name	Alternate Analyte Name	Units	Federal MCL	Federal MCLG	State MCL	Groundwater Method A Cleanup Levels	Groundwater Method B Unrestricted Land Use	Acute Freshwater CMC	Freshwater CCC	Human Health Water + Organism	Freshwater CCC	Freshwater CMC	Freshwater CCC	Human Health Water + Organism	Surface Water Method B Unrestricted Land Use	Action Level	Action Level Basis
630-20-6	1,1,1,2-Tetrachloroethane	--	µg/L	--	--	--	--	1.7	--	--	--	--	--	--	--	6.2	1.7	WAC 173-340-720(4)(b)(iii)(A) and (B)
71-55-6	1,1,1-Trichloroethane	--	µg/L	200	200	--	--	16,000	--	--	--	--	--	--	--	925,926	200	40 CFR 141 – Primary Federal MCL
79-34-5	1,1,2,2-Tetrachloroethane	--	µg/L	--	--	--	--	0.22	--	--	0.17	--	--	--	0.17	6.5	0.17	Clean Water Act -- Human Health Water + Organism
79-00-5	1,1,2-Trichloroethane	--	µg/L	5.0	3.0	--	--	0.77	--	--	0.59	--	--	--	0.60	25	0.59	Clean Water Act -- Human Health Water + Organism
75-34-3	1,1-Dichloroethane	--	µg/L	--	--	--	--	1,600	--	--	--	--	--	--	--	73,549	1,600	WAC 173-340-720(4)(b)(iii)(A) and (B)
75-35-4	1,1-Dichloroethene	1,1-Dichloroethylene	µg/L	7.0	7.0	--	--	400	--	--	330	--	--	--	0.057	23,148	0.057	40 CFR 131 -- Human Health Water + Organism
96-18-4	1,2,3-Trichloropropane	--	µg/L	--	--	--	--	0.0015	--	--	--	--	--	--	--	0.017	0.0015	WAC 173-340-720(4)(b)(iii)(A) and (B)
96-12-8	1,2-Dibromo-3-chloropropane	--	µg/L	0.20	--	--	--	0.055	--	--	--	--	--	--	--	0.70	0.055	WAC 173-340-720(4)(b)(iii)(A) and (B)
106-93-4	1,2-Dibromoethane	--	µg/L	0.050	--	--	--	0.022	--	--	--	--	--	--	--	0.22	0.022	WAC 173-340-720(4)(b)(iii)(A) and (B)
107-06-2	1,2-Dichloroethane	--	µg/L	5.0	--	--	--	0.48	--	--	0.38	--	--	--	0.38	59	0.38	Clean Water Act -- Human Health Water + Organism
540-59-0	1,2-Dichloroethene (Total)	1,2-Dichloroethylene Mixed Isomers	µg/L	--	--	--	--	72	--	--	--	--	--	--	--	2,102	72	WAC 173-340-720(4)(b)(iii)(A) and (B)
78-87-5	1,2-Dichloropropane	--	µg/L	5.0	--	--	--	1.2	--	--	0.50	--	--	--	--	44	0.50	Clean Water Act -- Human Health Water + Organism

Table 6-29. Summary of Federal and State Water Quality Criteria used as Action Levels for the 100-HR-3 Groundwater OU

				Groundwater					Surface Water								Action Level Value	
				40 CFR 141[a]		WAC 246-290-310[b]	WAC 173-340-720[c]		Clean Water Act National Recommended Water Quality Criteria[d]			WAC 173-201A[e]	40 CFR 131[f]			WAC 173-340-730[g]		
CAS Number	Analyte Name	Alternate Analyte Name	Units	Federal MCL	Federal MCLG	State MCL	Groundwater Method A Cleanup Levels	Groundwater Method B Unrestricted Land Use	Acute Freshwater CMC	Freshwater CCC	Human Health Water + Organism	Freshwater CCC	Freshwater CMC	Freshwater CCC	Human Health Water + Organism	Surface Water Method B Unrestricted Land Use	Action Level	Action Level Basis
106-46-7	1,4-Dichlorobenzene	--	µg/L	75	75	--	--	8.1	--	--	63	--	--	--	400	22	8.1	WAC 173-340-720(4)(b)(iii)(A) and (B)
123-91-1	1,4-Dioxane	--	µg/L	--	--	--	--	4.0	--	--	--	--	--	--	--	--	4.0	WAC 173-340-720(4)(b)(iii)(A) and (B)
71-36-3	1-Butanol	N-Butanol	µg/L	--	--	--	--	800	--	--	--	--	--	--	--	82,044	800	WAC 173-340-720(4)(b)(iii)(A) and (B)
78-93-3	2-Butanone	Methyl ethyl ketone	µg/L	--	--	--	--	4,800	--	--	--	--	--	--	--	492,264	4,800	WAC 173-340-720(4)(b)(iii)(A) and (B)
591-78-6	2-Hexanone	--	µg/L	--	--	--	--	80	--	--	--	--	--	--	--	3,429	80	WAC 173-340-720(4)(b)(iii)(A) and (B)
108-10-1	4-Methyl-2-pentanone	4-Methyl-2-penatone	µg/L	--	--	--	--	640	--	--	--	--	--	--	--	61,002	640	WAC 173-340-720(4)(b)(iii)(A) and (B)
67-64-1	Acetone	--	µg/L	--	--	--	--	7,200	--	--	--	--	--	--	--	738,397	7,200	WAC 173-340-720(4)(b)(iii)(A) and (B)
107-02-8	Acrolein	--	µg/L	--	--	--	--	4.0	--	3.0	6.0	--	--	--	320	--	3.0	Clean Water Act -- Freshwater CCC
107-05-1	Allyl chloride	--	µg/L	--	--	--	--	2.1	--	--	--	--	--	--	--	62	2.1	WAC 173-340-720(4)(b)(iii)(A) and (B)
7429-90-5	Aluminum	--	µg/L	50	--	--	--	16,000	750	87	--	--	--	--	--	5,185	50	40 CFR 141 - Federal MCL
7440-36-0	Antimony	Antimony (metallic)	µg/L	6.0	6.0	6.0	--	6.4	--	--	5.6	--	--	--	14	1,037	5.6	Clean Water Act -- Human Health Water + Organism
7440-38-2	Arsenic	Arsenic, inorganic	µg/L	10	--	10	--	0.058	340	150	0.018	190	360	190	0.018	0.098	0.018	Clean Water Act -- Human Health Water + Organism
7440-39-3	Barium	--	µg/L	2,000	2,000	2,000	--	3,200	--	--	1,000	--	--	--	--	129,630	1,000	Clean Water Act -- Human Health Water + Organism

Table 6-29. Summary of Federal and State Water Quality Criteria used as Action Levels for the 100-HR-3 Groundwater OU

				Groundwater					Surface Water								Action Level Value	
				40 CFR 141[a]		WAC 246-290-310[b]	WAC 173-340-720[c]		Clean Water Act National Recommended Water Quality Criteria[d]			WAC 173-201A[e]	40 CFR 131[f]			WAC 173-340-730[g]		
CAS Number	Analyte Name	Alternate Analyte Name	Units	Federal MCL	Federal MCLG	State MCL	Groundwater Method A Cleanup Levels	Groundwater Method B Unrestricted Land Use	Acute Freshwater CMC	Freshwater CCC	Human Health Water + Organism	Freshwater CCC	Freshwater CMC	Freshwater CCC	Human Health Water + Organism	Surface Water Method B Unrestricted Land Use	Action Level	Action Level Basis
71-43-2	Benzene	--	µg/L	5.0	--	--	--	0.80	--	--	2.2	--	--	--	1.2	23	0.80	WAC 173-340-720(4)(b)(iii)(A) and (B)
7440-41-7	Beryllium	Beryllium and compounds	µg/L	4.0	4.0	4.0	--	32	--	--	--	--	--	--	--	273	4.0	40 CFR 141 - Primary Federal MCL
7440-42-8	Boron	Boron and borates only	µg/L	--	--	--	--	3,200	--	--	--	--	--	--	--	--	3,200	WAC 173-340-720(4)(b)(iii)(A) and (B)
75-27-4	Bromodichloromethane	--	µg/L	--	--	--	--	0.71	--	--	0.55	--	--	--	0.27	28	0.27	40 CFR 131 -- Human Health Water + Organism
75-25-2	Bromoform	--	µg/L	--	80	--	--	5.5	--	--	4.3	--	--	--	4.3	219	4.3	Clean Water Act -- Human Health Water + Organism
74-83-9	Bromomethane	--	µg/L	--	--	--	--	11	--	--	47	--	--	--	48	968	11	WAC 173-340-720(4)(b)(iii)(A) and (B)
7440-43-9	Cadmium	Cadmium (water)	µg/L	5.0	5.0	5.0	--	8.0	2.0	0.25	--	0.91	3.9	1.0	--	20	0.25	Clean Water Act -- Freshwater CCC
75-15-0	Carbon disulfide	--	µg/L	--	--	--	--	800	--	--	--	--	--	--	--	13,295	800	WAC 173-340-720(4)(b)(iii)(A) and (B)
56-23-5	Carbon tetrachloride	--	µg/L	5.0	--	--	--	0.34	--	--	0.23	--	--	--	0.25	2.7	0.23	Clean Water Act -- Human Health Water + Organism
10045-97-3	Cesium-137	--	pCi/L	200	--	--	--	--	--	--	--	--	--	--	--	--	200	40 CFR 141 - Primary Federal MCL
16887-00-6	Chloride	--	µg/L	250,000	--	250,000	--	--	860,000	230,000	--	230,000	--	--	--	--	230,000	Clean Water Act -- Freshwater CCC
108-90-7	Chlorobenzene	--	µg/L	100	100	--	--	160	--	--	130	--	--	--	680	5,034	100	40 CFR 141 - Primary Federal MCL
75-00-3	Chloroethane	Ethylchloride	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--
67-66-3	Chloroform	--	µg/L	80	--	--	--	1.4	--	--	5.7	--	--	--	5.7	56	1.4	WAC 173-340-720(4)(b)(iii)(A) and (B)

Table 6-29. Summary of Federal and State Water Quality Criteria used as Action Levels for the 100-HR-3 Groundwater OU

				Groundwater					Surface Water								Action Level Value	
				40 CFR 141[a]		WAC 246-290-310[b]	WAC 173-340-720[c]		Clean Water Act National Recommended Water Quality Criteria[d]			WAC 173-201A[e]	40 CFR 131[f]			WAC 173-340-730[g]		
CAS Number	Analyte Name	Alternate Analyte Name	Units	Federal MCL	Federal MCLG	State MCL	Groundwater Method A Cleanup Levels	Groundwater Method B Unrestricted Land Use	Acute Freshwater CMC	Freshwater CCC	Human Health Water + Organism	Freshwater CCC	Freshwater CMC	Freshwater CCC	Human Health Water + Organism	Surface Water Method B Unrestricted Land Use	Action Level	Action Level Basis
74-87-3	Chloromethane	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--
126-99-8	Chloroprene	2-Chloro-1,3-butadiene	µg/L	--	--	--	--	160	--	--	--	--	--	--	--	2,412	160	WAC 173-340-720(4)(b)(iii)(A) and (B)
7440-47-3	Chromium	--	µg/L	100	100	100	--	24,000	570	65	--	156	550	180	--	19,444	65	Clean Water Act -- Freshwater CCC
156-59-2	cis-1,2-Dichloroethylene	--	µg/L	70	70	--	--	80	--	--	--	--	--	--	--	2,336	70	40 CFR 141 - Primary Federal MCL
10061-01-5	cis-1,3-Dichloropropene	--	µg/L	--	--	--	--	0.44	--	--	0.34	--	--	--	--	34	0.34	Clean Water Act -- Human Health Water + Organism
7440-48-4	Cobalt	--	µg/L	--	--	--	--	4.8	--	--	--	--	--	--	--	2.6	2.6	WAC 173-340-730(3)(b)(iii)(A) and (B)
10198-40-0	Cobalt-60	--	pCi/L	100	--	--	--	--	--	--	--	--	--	--	--	--	100	40 CFR 141 - Primary Federal MCL
7440-50-8	Copper	--	µg/L	1,300	1,300	--	--	640	13	9.0	1,300	--	17	11	--	2,881	9.0	Clean Water Act -- Freshwater CCC
124-48-1	Dibromochloromethane	--	µg/L	60	60	--	--	0.52	--	--	0.40	--	--	--	0.41	21	0.40	Clean Water Act -- Human Health Water + Organism
74-95-3	Dibromomethane	Methylene bromide	µg/L	--	--	--	--	80	--	--	--	--	--	--	--	4,216	80	WAC 173-340-720(4)(b)(iii)(A) and (B)
75-71-8	Dichlorodifluoromethane	--	µg/L	--	--	--	--	1,600	--	--	--	--	--	--	--	84,312	1,600	WAC 173-340-720(4)(b)(iii)(A) and (B)
97-63-2	Ethyl methacrylate	--	µg/L	--	--	--	--	720	--	--	--	--	--	--	--	26,365	720	WAC 173-340-720(4)(b)(iii)(A) and (B)
100-41-4	Ethylbenzene	--	µg/L	700	700	--	--	4.0	--	--	530	--	--	--	3,100	16	4.0	WAC 173-340-720(4)(b)(iii)(A) and (B)
14683-23-9	Europium-152	--	pCi/L	200	--	--	--	--	--	--	--	--	--	--	--	--	200	40 CFR 141 - Primary Federal MCL

Table 6-29. Summary of Federal and State Water Quality Criteria used as Action Levels for the 100-HR-3 Groundwater OU

				Groundwater					Surface Water									
				40 CFR 141[a]		WAC 246-290-310[b]	WAC 173-340-720[c]		Clean Water Act National Recommended Water Quality Criteria[d]			WAC 173-201A[e]	40 CFR 131[f]			WAC 173-340-730[g]	Action Level Value	
CAS Number	Analyte Name	Alternate Analyte Name	Units	Federal MCL	Federal MCLG	State MCL	Groundwater Method A Cleanup Levels	Groundwater Method B Unrestricted Land Use	Acute Freshwater CMC	Freshwater CCC	Human Health Water + Organism	Freshwater CCC	Freshwater CMC	Freshwater CCC	Human Health Water + Organism	Surface Water Method B Unrestricted Land Use	Action Level	Action Level Basis
15585-10-1	Europium-154	--	pCi/L	60	--	--	--	--	--	--	--	--	--	--	--	--	60	40 CFR 141 - Primary Federal MCL
14391-16-3	Europium-155	--	pCi/L	600	--	--	--	--	--	--	--	--	--	--	--	--	600	40 CFR 141 - Primary Federal MCL
16984-48-8	Fluoride	--	µg/L	4,000	4,000	4,000	--	960	--	--	--	--	--	--	--	15,600	960	WAC 173-340-720(4)(b)(iii)(A) and (B)
12587-46-1	Gross alpha	--	pCi/L	15	--	--	--	--	--	--	--	--	--	--	--	--	15	40 CFR 141 - Primary Federal MCL
12587-47-2	Gross beta	--	mrem/year	4.0	--	--	--	--	--	--	--	--	--	--	--	--	4.0	40 CFR 141 - Primary Federal MCL
18540-29-9	Hexavalent chromium	Cr(VI)	µg/L	--	--	--	--	48	16	11	--	10	15	10	--	486	10	40 CFR 131 -- Freshwater CCC
7439-89-6	Iron	--	µg/L	300	--	300	--	11,200	--	1,000	300	--	--	--	--	9,074	300	40 CFR 141 - Federal Secondary MCL
78-83-1	Isobutyl alcohol	--	µg/L	--	--	--	--	2,400	--	--	--	--	--	--	--	246,132	2,400	WAC 173-340-720(4)(b)(iii)(A) and (B)
7439-92-1	Lead	Lead and compounds	µg/L	15	--	--	15	--	65	2.5	--	2.1	65	2.5	--	--	2.1	WAC 173-201A
7439-93-2	Lithium	--	µg/L	--	--	--	--	32	--	--	--	--	--	--	--	--	32	WAC 173-340-720(4)(b)(iii)(A) and (B)
7439-96-5	Manganese	Manganese (water)	µg/L	50	--	50	--	2,240	--	--	50	--	--	--	--	907	50	40 CFR 141 - Federal Secondary MCL
7487-94-7	Mercury	Mercuric chloride	µg/L	2.0	2.0	2.0	--	4.8	1.4	0.77	--	0.012	2.1	0.012	0.14	0.78	0.012	40 CFR 131 -- Freshwater CCC
126-98-7	Methacrylonitrile	--	µg/L	--	--	--	--	0.80	--	--	--	--	--	--	--	82	0.80	WAC 173-340-720(4)(b)(iii)(A) and (B)
80-62-6	Methyl methacrylate	--	µg/L	--	--	--	--	11,200	--	--	--	--	--	--	--	960,219	11,200	WAC 173-340-720(4)(b)(iii)(A) and (B)

Table 6-29. Summary of Federal and State Water Quality Criteria used as Action Levels for the 100-HR-3 Groundwater OU

				Groundwater					Surface Water									
				40 CFR 141[a]		WAC 246-290-310[b]	WAC 173-340-720[c]		Clean Water Act National Recommended Water Quality Criteria[d]			WAC 173-201A[e]	40 CFR 131[f]			WAC 173-340-730[g]	Action Level Value	
CAS Number	Analyte Name	Alternate Analyte Name	Units	Federal MCL	Federal MCLG	State MCL	Groundwater Method A Cleanup Levels	Groundwater Method B Unrestricted Land Use	Acute Freshwater CMC	Freshwater CCC	Human Health Water + Organism	Freshwater CCC	Freshwater CMC	Freshwater CCC	Human Health Water + Organism	Surface Water Method B Unrestricted Land Use	Action Level	Action Level Basis
75-09-2	Methylene chloride	--	µg/L	5.0	--	--	--	5.8	--	--	4.6	--	--	--	4.7	960	4.6	Clean Water Act -- Human Health Water + Organism
7439-98-7	Molybdenum	--	µg/L	--	--	--	--	80	--	--	--	--	--	--	--	1,296	80	WAC 173-340-720(4)(b)(iii)(A) and (B)
7440-02-0	Nickel	Nickel soluble salts	µg/L	--	100	100	--	320	470	52	610	137	1,400	160	610	1,103	52	Clean Water Act -- Freshwater CCC
14797-55-8	Nitrate	--	µg/L	45,000	45,000	45,000	--	113,600	--	--	45,000	--	--	--	--	--	45,000	40 CFR 141 - Primary Federal MCL
14797-65-0	Nitrite	--	µg/L	3,300	3,300	3,300	--	5,280	--	--	--	--	--	--	--	--	3,300	40 CFR 141 - Primary Federal MCL
14265-44-2	Phosphate	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--
7782-49-2	Selenium	--	µg/L	50	50	50	--	80	--	5.0	170	5.0	20	5.0	--	2,701	5.0	Clean Water Act -- Freshwater CCC
7440-22-4	Silver	--	µg/L	100	--	100	--	80	3.2	--	--	2.6	3.4	--	--	25,926	2.6	WAC 173-201A
7440-24-6	Strontium	Strontium, Stable	µg/L	--	--	--	--	9,600	--	--	--	--	--	--	--	25,926	9,600	WAC 173-340-720(4)(b)(iii)(A) and (B)
10098-97-2	Strontium-90	--	pCi/L	8.0	--	--	--	--	--	--	--	--	--	--	--	--	8.0	40 CFR 141 - Primary Federal MCL
100-42-5	Styrene	--	µg/L	100	100	--	--	1,600	--	--	--	--	--	--	--	38,409	100	40 CFR 141 - Primary Federal MCL
14808-79-8	Sulfate	--	µg/L	250,000	--	250,000	--	--	--	--	--	--	--	--	--	--	250,000	40 CFR 141 – Secondary Federal MCL
14133-76-7	Technetium-99	--	pCi/L	900	--	--	--	--	--	--	--	--	--	--	--	--	900	40 CFR 141 - Primary Federal MCL
127-18-4	Tetrachloroethene	Perchloroethylene (PCE)	µg/L	5.0	--	--	--	0.081	--	--	0.69	--	--	--	0.80	0.39	0.081	WAC 173-340-720(4)(b)(iii)(A) and (B)
7440-28-0	Thallium	Thallium (soluble salts)	µg/L	2.0	0.50	2.0	--	--	--	--	0.24	--	--	--	1.7	--	0.24	Clean Water Act -- Human Health Water + Organism

Table 6-29. Summary of Federal and State Water Quality Criteria used as Action Levels for the 100-HR-3 Groundwater OU

				Groundwater					Surface Water									
				40 CFR 141[a]		WAC 246-290-310[b]	WAC 173-340-720[c]		Clean Water Act National Recommended Water Quality Criteria[d]			WAC 173-201A[e]	40 CFR 131[f]			WAC 173-340-730[g]	Action Level Value	
CAS Number	Analyte Name	Alternate Analyte Name	Units	Federal MCL	Federal MCLG	State MCL	Groundwater Method A Cleanup Levels	Groundwater Method B Unrestricted Land Use	Acute Freshwater CMC	Freshwater CCC	Human Health Water + Organism	Freshwater CCC	Freshwater CMC	Freshwater CCC	Human Health Water + Organism	Surface Water Method B Unrestricted Land Use	Action Level	Action Level Basis
7440-31-5	Tin	--	µg/L	--	--	--	--	9,600	--	--	--	--	--	--	--	519	519	WAC 173-340-730(3)(b)(iii)(A) and (B)
108-88-3	Toluene	--	µg/L	1,000	1,000	--	--	640	--	--	1,300	--	--	--	6,800	19,384	640	WAC 173-340-720(4)(b)(iii)(A) and (B)
156-60-5	trans-1,2-dichloroethylene	--	µg/L	100	100	--	--	160	--	--	140	--	--	--	--	32,818	100	40 CFR 141 - Primary Federal MCL
10061-02-6	trans-1,3-dichloropropene	--	µg/L	--	--	--	--	0.44	--	--	0.34	--	--	--	--	34	0.34	Clean Water Act - - Human Health Water + Organism
110-57-6	trans-1,4-dichloro-2-butene	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--
79-01-6	Trichloroethene	Trichloroethylene	µg/L	5.0	--	--	--	0.49	--	--	2.5	--	--	--	2.7	6.6	0.49	WAC 173-340-720(4)(b)(iii)(A) and (B)
75-69-4	Trichloromonofluoro-methane	Trichlorofluoro-methane	µg/L	--	--	--	--	2,400	--	--	--	--	--	--	--	--	2,400	WAC 173-340-720(4)(b)(iii)(A) and (B)
10028-17-8	Tritium	--	pCi/L	20,000	--	--	--	--	--	--	--	--	--	--	--	--	20,000	40 CFR 141 - Primary Federal MCL
7440-61-1	Uranium	Uranium (soluble salts)	µg/L	30	--	--	--	48	--	--	--	--	--	--	--	778	30	40 CFR 141 - Primary Federal MCL
7440-62-2	Vanadium	Vanadium and compounds	µg/L	--	--	--	--	80	--	--	--	--	--	--	--	--	80	WAC 173-340-720(4)(b)(iii)(A) and (B)
108-05-4	Vinyl acetate	--	µg/L	--	--	--	--	8,000	--	--	--	--	--	--	--	820,441	8,000	WAC 173-340-720(4)(b)(iii)(A) and (B)
75-01-4	Vinyl chloride	--	µg/L	2.0	--	--	--	0.061	--	--	0.025	--	--	--	2.0	7.7	0.025	Clean Water Act - - Human Health Water + Organism
1330-20-7	Xylenes (total)	Xylenes (mixture)	µg/L	10,000	10,000	--	--	1,600	--	--	--	--	--	--	--	--	1,600	WAC 173-340-720(4)(b)(iii)(A) and (B)
7440-66-6	Zinc	Zinc (metallic)	µg/L	5,000	--	5,000	--	4,800	120	120	7,400	91	110	100	--	16,548	91	WAC 173-201A

Table 6-29. Summary of Federal and State Water Quality Criteria used as Action Levels for the 100-HR-3 Groundwater OU

				Groundwater					Surface Water									
				40 CFR 141[a]		WAC 246-290-310[b]	WAC 173-340-720[c]		Clean Water Act National Recommended Water Quality Criteria[d]			WAC 173-201A[e]	40 CFR 131[f]			WAC 173-340-730[g]	Action Level Value	
CAS Number	Analyte Name	Alternate Analyte Name	Units	Federal MCL	Federal MCLG	State MCL	Groundwater Method A Cleanup Levels	Groundwater Method B Unrestricted Land Use	Acute Freshwater CMC	Freshwater CCC	Human Health Water + Organism	Freshwater CCC	Freshwater CMC	Freshwater CCC	Human Health Water + Organism	Surface Water Method B Unrestricted Land Use	Action Level	Action Level Basis

Note:

That 40 CFR 131, "Water Quality Standards," *National Recommended Water Quality Criteria* (EPA, 2009b), and WAC 173-201A, "Water Quality Standards for Surface Waters of the State of Washington" only apply in locations where groundwater has the potential to discharge to the Columbia River. Sources:

40 CFR 131, "Water Quality Standards."

40 CFR 141, "National Primary Drinking Water Regulations."

Ecology Publication 94-06, *Model Toxics Control Act Cleanup Regulation Chapter 173-340 WAC.*

EPA, 2009b, *National Recommended Water Quality Criteria.*

WAC 173-201A, "Water Quality Standards for Surface Waters of the State of Washington."

WAC 173-340-720(4)(b)(ii)(A) and (B), "Groundwater Cleanup Standards," "Noncarcinogens and Carcinogens."

WAC 173-340-730(3)(b)(iii)(A) and (B), Surface Water Cleanup Standards," "Noncarcinogens and Carcinogens."

WAC 246-290-310, "Group A Public Water Supplies," "Maximum Contaminant Levels (MCLs) and Maximum Residual Disinfectant Levels (MRDLs)."

Table 6-30. Summary of Groundwater Analytes that Meet Exclusion Criteria in the 100-HR-3 Groundwater OU

Analyte Name	Analyte Class	Begin Sampling Date	End Sampling Date	Total Samples	Total Detects	Frequency of Detection	Units	Minimum Detection Limit	Maximum Detection Limit	Minimum Detected Result	Maximum Detected Result	Basis for Exclusion
Bromide	ANION	10/7/2009	6/11/2010	117	86	73.50%	µg/L	90	450	98	320	No Action Level/No Toxicity Values
Phosphate	ANION	10/7/2009	6/11/2010	117	3	2.56%	µg/L	429	2,150	460	1,260	No Action Level/No Toxicity Values
Bismuth	METAL	10/7/2009	6/11/2010	122	7	5.74%	µg/L	0	23	23	38	No Action Level/No Toxicity Values
Calcium	METAL	10/7/2009	6/11/2010	156	156	100.00%	µg/L	--	--	34,200	157,000	Essential Nutrient
Magnesium	METAL	10/7/2009	6/11/2010	156	156	100.00%	µg/L	--	--	774	39,600	Essential Nutrient
Potassium	METAL	10/7/2009	6/11/2010	156	156	100.00%	µg/L	--	--	1,870	7,190	Essential Nutrient
Silicon	METAL	10/7/2009	6/11/2010	122	122	100.00%	µg/L	--	--	7,510	22,800	No Action Level/No Toxicity Values
Sodium	METAL	10/7/2009	6/11/2010	156	156	100.00%	µg/L	--	--	4,200	38,100	Essential Nutrient
Antimony-125	RAD	10/7/2009	6/11/2010	156	0	0.00%	pCi/L	-4.30E+00	6.5	--	--	Half-Life less than 3 years
Beryllium-7	RAD	10/7/2009	6/11/2010	156	0	0.00%	pCi/L	-3.28E+01	32	--	--	Half-Life less than 3 years
Cesium-134	RAD	10/7/2009	6/11/2010	156	0	0.00%	pCi/L	-2.81E+00	2.7	--	--	Half-Life less than 3 years
Gross beta	RAD	10/7/2009	6/11/2010	156	116	74.36%	pCi/L	0.055	6.3	3.4	58	No Action Level/No Toxicity Values
Potassium-40	RAD	10/7/2009	6/11/2010	156	1	0.64%	pCi/L	-8.60E+01	37	58	58	Background Radiation
Ruthenium-106	RAD	10/7/2009	6/11/2010	156	0	0.00%	pCi/L	-2.79E+01	26	--	--	Half-Life less than 3 years
1-Chloro-1,1-difluoroethane	VOC	10/9/2009	10/9/2009	1	1	100.00%	µg/L	--	--	56	56	No Action Level/No Toxicity Values
Acetonitrile	VOC	3/18/2010	6/11/2010	104	0	0.00%	µg/L	2.0	2.0	--	--	No Action Level/Toxicity Values Available
Chloroethane	VOC	10/7/2009	6/11/2010	156	0	0.00%	µg/L	0.085	1.0	--	--	No Action Level/No Toxicity Values
Chloromethane	VOC	10/7/2009	6/11/2010	156	1	0.64%	µg/L	0.077	1.0	0.10	0.10	No Action Level/Toxicity Values Available
Ethyl cyanide	VOC	10/7/2009	6/11/2010	156	0	0.00%	µg/L	1.2	2.0	--	--	No Action Level/No Toxicity Values
Iodomethane	VOC	3/18/2010	6/11/2010	104	0	0.00%	µg/L	0.092	0.092	--	--	No Action Level/No Toxicity Values
Tetrahydrofuran	VOC	10/7/2009	6/11/2010	156	0	0.00%	µg/L	1.1	2.0	--	--	No Action Level/No Toxicity Values
trans-1,4-Dichloro-2-butene	VOC	3/18/2010	6/11/2010	104	0	0.00%	µg/L	0.29	0.29	--	--	No Action Level/No Toxicity Values
Trichloroacetyl chloride	VOC	10/9/2009	10/9/2009	1	1	100.00%	µg/L	--	--	1.5	1.5	No Action Level/No Toxicity Values

Table 6-31. Summary of Analytes that Were Not Detected in the 100-HR-3 Groundwater OU

Analyte Name	Analyte Class	Begin Sample Date	End Sample Date	Total Samples	Total Detects	Frequency of Detection	Units	Minimum Detection Limit	Maximum Detection Limit	Action Level	Action Level Basis	Level of Exceedance
Cesium-137	RAD	10/7/2009	6/11/2010	156	0	0.00%	pCi/L	-2.15	2.96	200	40 CFR 141 - Primary Federal MCL	-0.011
Cobalt-60	RAD	10/7/2009	6/11/2010	156	0	0.00%	pCi/L	-2.98	2.09	100	40 CFR 141 - Primary Federal MCL	-0.030
Europium-152	RAD	10/7/2009	6/11/2010	156	0	0.00%	pCi/L	-6.76	6.52	200	40 CFR 141 - Primary Federal MCL	-0.034
Europium-154	RAD	10/7/2009	6/11/2010	156	0	0.00%	pCi/L	-6.14	7.94	60	40 CFR 141 - Primary Federal MCL	-0.10
Europium-155	RAD	10/7/2009	6/11/2010	156	0	0.00%	pCi/L	-5.03	4.24	600	40 CFR 141 - Primary Federal MCL	-0.008
1,4-Dichlorobenzene	SVOC	10/7/2009	6/11/2010	156	0	0.00%	µg/L	0.12	1	8.1	WAC 173-340-720(4)(b)(iii)(A) and (B)	0.015
1,1,1,2-Tetrachloroethane	VOC	3/18/2010	6/11/2010	104	0	0.00%	µg/L	0.09	0.09	1.7	WAC 173-340-720(4)(b)(iii)(A) and (B)	0.054
1,1,1-Trichloroethane	VOC	10/7/2009	6/11/2010	156	0	0.00%	µg/L	0.067	1	200	40 CFR 141 - Federal MCL	0.00034
1,1,2,2-Tetrachloroethane	VOC	10/7/2009	6/11/2010	156	0	0.00%	µg/L	0.098	1	0.17	Clean Water Act -- Human Health Water + Organism	0.58
1,1,2-Trichloroethane	VOC	10/7/2009	6/11/2010	156	0	0.00%	µg/L	0.063	1	0.59	Clean Water Act -- Human Health Water + Organism	0.11
1,1-Dichloroethane	VOC	10/7/2009	6/11/2010	156	0	0.00%	µg/L	0.068	1	1,600	WAC 173-340-720(4)(b)(iii)(A) and (B)	0.000043
1,1-Dichloroethene	VOC	10/7/2009	6/11/2010	156	0	0.00%	µg/L	0.051	1	0.057	40 CFR 131 -- Human Health Water + Organism	0.89
1,2,3-Trichloropropane	VOC	3/18/2010	6/11/2010	104	0	0.00%	µg/L	0.15	0.15	0.0015	WAC 173-340-720(4)(b)(iii)(A) and (B)	103
1,2-Dibromo-3-chloropropane	VOC	3/18/2010	6/11/2010	104	0	0.00%	µg/L	0.41	0.41	0.055	WAC 173-340-720(4)(b)(iii)(A) and (B)	7.5
1,2-Dibromoethane	VOC	3/18/2010	6/11/2010	104	0	0.00%	µg/L	0.13	0.13	0.022	WAC 173-340-720(4)(b)(iii)(A) and (B)	5.9
1,2-Dichloroethene (Total)	VOC	10/7/2009	6/11/2010	156	0	0.00%	µg/L	0.13	1	72	WAC 173-340-720(4)(b)(iii)(A) and (B)	0.0018
1,2-Dichloropropane	VOC	10/7/2009	6/11/2010	156	0	0.00%	µg/L	0.097	1	0.50	Clean Water Act -- Human Health Water + Organism	0.19
1,4-Dioxane	VOC	3/18/2010	6/11/2010	104	0	0.00%	µg/L	7.6	7.6	4.0	WAC 173-340-720(4)(b)(iii)(A) and (B)	1.9
1-Butanol	VOC	10/7/2009	6/11/2010	156	0	0.00%	µg/L	12	100	800	WAC 173-340-720(4)(b)(iii)(A) and (B)	0.015
2-Hexanone	VOC	10/7/2009	6/11/2010	156	0	0.00%	µg/L	0.22	1.0	80	WAC 173-340-720(4)(b)(iii)(A) and (B)	0.0028
4-Methyl-2-pentanone	VOC	10/7/2009	6/11/2010	156	0	0.00%	µg/L	0.12	1.0	640	WAC 173-340-720(4)(b)(iii)(A) and (B)	0.00019
Acrolein	VOC	3/18/2010	6/11/2010	104	0	0.00%	µg/L	2.8	2.8	3.0	Clean Water Act -- Freshwater CCC	0.93
Allyl chloride	VOC	3/18/2010	6/11/2010	104	0	0.00%	µg/L	0.091	0.11	2.1	WAC 173-340-720(4)(b)(iii)(A) and (B)	0.044
Benzene	VOC	10/7/2009	6/11/2010	156	0	0.00%	µg/L	0.045	1.0	0.80	WAC 173-340-720(4)(b)(iii)(A) and (B)	0.057
Chlorobenzene	VOC	10/7/2009	6/11/2010	156	0	0.00%	µg/L	0.15	1.0	100	40 CFR 141 - Primary Federal MCL	0.0015
Chloroprene	VOC	3/18/2010	6/11/2010	104	0	0.00%	µg/L	0.086	0.097	160	WAC 173-340-720(4)(b)(iii)(A) and (B)	0.00054
cis-1,2-Dichloroethylene	VOC	10/7/2009	6/11/2010	156	0	0.00%	µg/L	0.083	1.0	70	40 CFR 141 - Primary Federal MCL	0.0012
cis-1,3-Dichloropropene	VOC	10/7/2009	6/11/2010	156	0	0.00%	µg/L	0.073	1.0	0.34	Clean Water Act -- Human Health Water + Organism	0.21

Table 6-31. Summary of Analytes that Were Not Detected in the 100-HR-3 Groundwater OU

Analyte Name	Analyte Class	Begin Sample Date	End Sample Date	Total Samples	Total Detects	Frequency of Detection	Units	Minimum Detection Limit	Maximum Detection Limit	Action Level	Action Level Basis	Level of Exceedance
Dibromochloromethane	VOC	10/7/2009	6/11/2010	156	0	0.00%	µg/L	0.057	1.0	0.40	Clean Water Act -- Human Health Water + Organism	0.14
Dibromomethane	VOC	3/18/2010	6/11/2010	104	0	0.00%	µg/L	0.21	0.21	80	WAC 173-340-720(4)(b)(iii)(A) and (B)	0.0026
Dichlorodifluoromethane	VOC	3/18/2010	6/11/2010	104	0	0.00%	µg/L	0.070	0.084	1,600	WAC 173-340-720(4)(b)(iii)(A) and (B)	0.00004
Ethyl methacrylate	VOC	3/18/2010	6/11/2010	104	0	0.00%	µg/L	0.11	0.11	720	WAC 173-340-720(4)(b)(iii)(A) and (B)	0.00015
Ethylbenzene	VOC	10/7/2009	6/11/2010	156	0	0.00%	µg/L	0.086	1.0	4.0	WAC 173-340-720(4)(b)(iii)(A) and (B)	0.022
Isobutyl alcohol	VOC	3/18/2010	6/11/2010	104	0	0.00%	µg/L	8.7	8.7	2,400	WAC 173-340-720(4)(b)(iii)(A) and (B)	0.0036
Methacrylonitrile	VOC	3/18/2010	6/11/2010	104	0	0.00%	µg/L	0.050	0.50	0.80	WAC 173-340-720(4)(b)(iii)(A) and (B)	0.063
Methyl methacrylate	VOC	3/18/2010	6/11/2010	104	0	0.00%	µg/L	0.26	0.26	11,200	WAC 173-340-720(4)(b)(iii)(A) and (B)	0.00002
Styrene	VOC	10/7/2009	6/11/2010	156	0	0.00%	µg/L	0.036	1.0	100	40 CFR 141 - Primary Federal MCL	0.00036
trans-1,2-Dichloroethylene	VOC	10/7/2009	6/11/2010	156	0	0.00%	µg/L	0.083	1.0	100	40 CFR 141 - Primary Federal MCL	0.00083
trans-1,3-Dichloropropene	VOC	10/7/2009	6/11/2010	156	0	0.00%	µg/L	0.083	1.0	0.34	Clean Water Act -- Human Health Water + Organism	0.24
Trichloromonofluoromethane	VOC	3/18/2010	6/11/2010	104	0	0.00%	µg/L	0.041	0.11	2,400	WAC 173-340-720(4)(b)(iii)(A) and (B)	0.00002
Vinyl acetate	VOC	3/18/2010	6/11/2010	104	0	0.00%	µg/L	0.17	0.18	8,000	WAC 173-340-720(4)(b)(iii)(A) and (B)	0.00002
Vinyl chloride	VOC	10/7/2009	6/11/2010	156	0	0.00%	µg/L	0.032	1.0	0.025	Clean Water Act -- Human Health Water + Organism	1.3

Note:

Shading indicates that an analyte was identified in the list of COPCs in DOE/RL-2009-40, *Sampling and Analysis Plan for the 100-DR-1, 100-DR-2, 100-HR-1, 100-HR-2, and 100-HR-3 Operable Units Remedial Investigation/Feasibility Study*.

Table 6-32. Summary of Groundwater Analytes That Do Not Exceed an Action Level in the 100-HR-3 Groundwater OU

Analyte Name	Analyte Class	Begin Sample Date	End Sample Date	Total Samples	Total Detects	Frequency of Detection	Units	Minimum Detection Limit	Maximum Detection Limit	Minimum Detected Result	Maximum Detected Result	Action Level	Action Level Basis
Chloride	ANION	10/7/2009	6/11/2010	156	156	100.00%	µg/L	--	--	3,960	44,900	230,000	Clean Water Act -- Freshwater CCC
Fluoride	ANION	10/7/2009	6/11/2010	156	63	40.38%	µg/L	60	300	60	343	960	WAC 173-340-720(4)(b)(iii)(A) and (B)
Nitrite	ANION	10/7/2009	6/11/2010	156	10	6.41%	µg/L	9.9	591	1,140	2,270	3,300	40 CFR 141 - Primary Federal MCL
Antimony	METAL	10/7/2009	6/11/2010	156	13	8.33%	µg/L	0.30	1.1	0.61	1.0	5.6	Clean Water Act -- Human Health Water + Organism
Barium	METAL	10/7/2009	6/11/2010	156	156	100.00%	µg/L	--	--	25	133	1,000	Clean Water Act -- Human Health Water + Organism
Beryllium	METAL	10/7/2009	6/11/2010	156	5	3.21%	µg/L	0.050	0.11	0.10	0.31	4.0	40 CFR 141 - Federal MCL
Boron	METAL	10/7/2009	6/11/2010	122	44	36.07%	µg/L	19	19	9.7	102	3,200	WAC 173-340-720(4)(b)(iii)(A) and (B)
Cadmium	METAL	10/7/2009	6/11/2010	156	2	1.28%	µg/L	0.055	0.20	0.11	0.22	0.25	Clean Water Act -- Freshwater CCC
Copper	METAL	10/7/2009	6/11/2010	156	104	66.67%	µg/L	0.10	0.20	0.10	2.8	9.0	Clean Water Act -- Freshwater CCC
Lead	METAL	10/7/2009	6/11/2010	156	23	14.74%	µg/L	0.10	0.20	0.20	0.71	2.1	WAC 173-201A
Molybdenum	METAL	10/7/2009	6/11/2010	122	115	94.26%	µg/L	4.0	4.0	0.56	12	80	WAC 173-340-720(4)(b)(iii)(A) and (B)
Nickel	METAL	10/7/2009	6/11/2010	156	33	21.15%	µg/L	4.0	4.0	2.4	39	52	Clean Water Act -- Freshwater CCC
Silver	METAL	10/7/2009	6/11/2010	156	6	3.85%	µg/L	0.040	0.20	0.13	1.00	2.6	WAC 173-201A
Strontium	METAL	10/7/2009	6/11/2010	156	156	100.00%	µg/L	--	--	138	938	9,600	WAC 173-340-720(4)(b)(iii)(A) and (B)
Tin	METAL	10/7/2009	6/11/2010	122	11	9.02%	µg/L	0.050	39	0.055	43	519	WAC 173-340-730(3)(b)(iii)(A) and (B)
Uranium	METAL	10/7/2009	6/11/2010	156	156	100.00%	µg/L	--	--	0.29	13	30	40 CFR 141 - Federal MCL
Vanadium	METAL	10/7/2009	6/11/2010	156	66	42.31%	µg/L	4.1	12	5.4	33	80	WAC 173-340-720(4)(b)(iii)(A) and (B)
Gross alpha	RAD	10/7/2009	6/11/2010	156	34	21.79%	pCi/L	-2.90	11	2.0	7.9	15	40 CFR 141 - Primary Federal MCL
Technetium-99	RAD	10/7/2009	6/11/2010	155	8	5.16%	pCi/L	-17	3.6	7.9	35	900	40 CFR 141 - Primary Federal MCL
Tritium	RAD	10/7/2009	6/11/2010	156	142	91.03%	pCi/L	-13	170	180	12,000	20,000	40 CFR 141 - Primary Federal MCL
2-Butanone	VOC	10/7/2009	6/11/2010	156	1	0.64%	µg/L	0.52	1.0	10	10	4,800	WAC 173-340-720(4)(b)(iii)(A) and (B)
Acetone	VOC	10/7/2009	6/11/2010	156	2	1.28%	µg/L	0.34	1.0	0.82	6.9	7,200	WAC 173-340-720(4)(b)(iii)(A) and (B)
Bromoform	VOC	10/7/2009	6/11/2010	156	1	0.64%	µg/L	0.094	1.0	0.58	0.58	4.3	Clean Water Act -- Human Health Water + Organism
Bromomethane	VOC	10/7/2009	6/11/2010	156	1	0.64%	µg/L	0.084	1.0	0.97	0.97	11	WAC 173-340-720(4)(b)(iii)(A) and (B)
Carbon disulfide	VOC	10/7/2009	6/11/2010	156	1	0.64%	µg/L	0.050	1.0	0.076	0.076	800	WAC 173-340-720(4)(b)(iii)(A) and (B)
Toluene	VOC	10/7/2009	6/11/2010	156	3	1.92%	µg/L	0.062	1.0	0.062	0.18	640	WAC 173-340-720(4)(b)(iii)(A) and (B)
Trichloroethene	VOC	10/7/2009	6/11/2010	156	3	1.92%	µg/L	0.21	1.0	0.26	0.33	0.49	WAC 173-340-720(4)(b)(iii)(A) and (B)
Xylenes (total)	VOC	10/7/2009	6/11/2010	156	2	1.28%	µg/L	0.11	1.0	0.44	0.46	1,600	WAC 173-340-720(4)(b)(iii)(A) and (B)

Note:
Shading indicates that an analyte was identified in the list of COPCs in DOE/RL-2009-40, *Sampling and Analysis Plan for the 100-DR-1, 100-DR-2, 100-HR-1, 100-HR-2, and 100-HR-3 Operable Units Remedial Investigation/Feasibility Study.*

Antimony, beryllium, cadmium, copper, fluoride, lead, nickel, nitrite, silver, technetium-99, tritium, trichloroethene, uranium, and vanadium were identified as historical COPCs in the 100/D-H Work Plan (DOE/RL-2008-46-ADD1). A discussion of these 14 analytes is provided in the following paragraphs.

Beryllium, fluoride, nitrite, technetium-99, tritium, uranium, and vanadium were detected in groundwater at concentrations below their respective action level in samples collected for the RI and in the larger population of wells described previously in Section 4.4.1.2 and Section 6.3.2.3.1. Beryllium, fluoride, lead, nitrite, tritium, uranium, and vanadium are not COPCs and will not be carried forward into the risk characterization section or into the FS. It should also be noted that concentrations of beryllium and fluoride in filtered groundwater samples are less than their 90th percentile Hanford Site background value.

Antimony, cadmium, copper, lead, nickel, silver, and trichloroethene were detected in groundwater samples collected for the RI at concentrations below their respective action level. However, these analytes were detected at concentrations above their respective action level or EQL (as applicable) in the larger population of wells described previously in Section 4.4.1.2 and Section 6.3.2.3.1. The following text discusses the results for these six analytes.

Detections of antimony, cadmium, copper, lead, and silver above the action level or EQL (as applicable) were from the larger population of wells sampled in the past 6 years as described previously in Section 4.4.1.2 and Section 6.3.2.3.1. All antimony, cadmium, copper, lead, and silver results (detected concentrations and MDLs) reported by Method 6010 (SW 846 [SW-846]) were greater than the action level or EQL (as applicable). Antimony, cadmium, copper, lead, and silver concentrations reported by Method 6010 are considered uncertain because this method cannot accurately report trace concentrations. Groundwater samples analyzed by Method 6010 generally report MDLs greater than the action level, resulting in nondetected concentrations greater than the action level. Similarly, detected concentrations are reported as estimates (flagged with a "B" qualifier) at concentrations greater than the action level and are below the contract-required calibration range of the instrument. Some results are also flagged with a "C" qualifier indicating that the analyte was detected in both the sample and the associated QC blank, and the sample concentration is less than or equal to five times the blank concentration. Additionally, antimony, cadmium, copper, lead, and silver concentrations above the action level are not associated with a specific location or with a trend. Although antimony, cadmium, copper, lead, and silver were detected at concentrations less than the action level in samples analyzed for the RI by Method 200.8 (*Methods for the Determination of Metals in Environmental Samples, Supplement 1* [EPA-600/R-94/111]), their historical presence with infrequent detections above the action level result in an uncertain status. Therefore, antimony, cadmium, copper lead, and silver are retained as COPCs and they warrant further evaluation in the FS.

Detections of nickel above the AWQC were from the larger population of wells sampled over the past 6 years as described previously in Section 4.4.1.2 and Section 6.3.2.3.1. Nickel was detected above the AWQC in the 100-D plume, but nickel concentrations were less than the action level in the 100-H and horn plume areas. All nickel results reported at concentrations greater than the AWQC were flagged with a "C" qualifier indicating that the analyte was detected in both the sample and the associated QC blank, and the sample concentration is less than or equal to five times the blank concentration. Additionally, nickel concentrations in the 100-D plume area above the AWQC were not associated with a specific location or with a trend. Although nickel was detected at concentrations less than the AWQC in samples collected for the RI, its presence over the action level in the 100-D plume over the past 6 years results in an uncertain status. Therefore, nickel in the 100-D plume area is retained as a COPC and warrants further evaluation in the FS. Although all monitoring wells within the groundwater plume areas were compared to the AWQC, only near-river wells would need to meet this criterion.

Detections of trichloroethene above the action level were from the larger population of wells sampled over the past 6 years as described previously in Section 4.4.1.2 and Section 6.3.2.3.1. The action level for trichloroethene is 0.49 µg/L based on the MTCA ("Groundwater Cleanup Standards" [WAC 173-340-720]) level; however, it defaults to the EQL of 1 µg/L reported in the 100-D/H SAP (DOE/RL-2009-40) when the analytical method cannot achieve the action level. Trichloroethene was detected infrequently in the horn area plume (6 percent frequency) at concentrations less than the EQL. Therefore, trichloroethene is not retained as a COPC and will not be carried forward into the risk characterization section or into the FS.

6.3.2.3.5 *Identify Analytes with Maximum Detected Concentrations Greater than Action Levels*[16]

This step identifies analytes with maximum concentrations greater than their respective action levels. Such analytes have the potential to contribute to overall risk. If the maximum detected concentration of an analyte is greater than its action level, the analyte is carried forward into the next step of the analysis for calculation of EPCs.

Twenty-one analytes were detected in the RI data at least once, with maximum detected concentrations greater than their respective action levels. A list of these analytes is presented in Table 6-33, each with sampling dates, minimum and maximum MDLs, minimum and maximum detected concentrations, the action level, and the basis of the action level.

6.3.2.3.6 *Calculate EPCs for Each Analyte with Maximum Detected Concentrations Greater than Action Levels*[17]

COPCs are identified by comparing statistical EPC estimates to action levels for each analyte and exposure area. EPCs are calculated as the 90th percentile value for each analyte with a maximum detected concentration greater than the action level from the groundwater dataset collected specifically for the RI. The MDL is used as the concentration for nondetect results in the percentile calculations. The 90th percentile exposure is identified in Guidelines for Exposure Assessment (EPA/600/Z-92/001) for describing and characterizing health risks and produces risk estimates corresponding to an RME. A description of the methodology used to calculate the 90th percentile values is provided in *Calculation of Exposure Point Concentrations for the 100-HR-3 Groundwater Operable Unit* (ECF-100HR3-10-0473) (Appendix G).

In general, Calculating UCL for EPCs (OSWER 9285.6-10) recommends using a 95 percent UCL on the average for estimating EPCs. However, experience at the Hanford Site indicates that averages and UCLs cannot be reliably calculated for groundwater datasets. The 100-HR-3 Groundwater OU exhibits an aquifer setting where multiple groundwater contaminants are present in overlapping plumes, and the highest concentrations of the various COPCs have different locations within the plumes.

Use of the 90th percentile value from a distribution of groundwater concentration data as an estimate of the EPC is a different approach for estimating EPCs than that provided in Calculating UCL for EPCs (OSWER 9285.6-10). However, as described in the following text, the 90th percentile exposure concentration is identified in other EPA risk assessment guidance as appropriate for describing and characterizing health risks; its use yields risk estimates that correspond to an RME.

According to *An Examination of EPA Risk Assessment Principles and Practices* (EPA/100/B-04/001), the RME is an appropriate exposure scenario for risk calculations, within the realistic range of exposure, since the goal of the Superfund program is to protect against high-end, not worst-case, exposures. The "high end" is defined as that part of the exposure distribution that is above the 90th percentile, but

[16] See Section 6.3.2.2 for the definition of an action level.

[17] See Section 6.3.2.2 for the definition of an action level.

Table 6-33. Summary of Analytes that Exceed an Action Level in the 100-HR-3 Groundwater OU

Analyte Name	Analyte Class	Begin Sample Date	End Sample Date	Total Samples	Total Detects	Frequency of Detection	Units	Minimum Detection Limit	Maximum Detection Limit	Minimum Detected Result	Maximum Detected Result	Action Level	Action Level Basis
Nitrate	ANION	10/7/2009	6/11/2010	155	155	100.00%	µg/L	--	--	7,880	99,200	45,000	40 CFR 141 - Primary Federal MCL
Sulfate	ANION	10/7/2009	6/11/2010	156	156	100.00%	µg/L	--	--	24,900	438,000	250,000	40 CFR 141 – Secondary Federal MCL
Aluminum	METAL	10/7/2009	6/11/2010	156	65	41.67%	µg/L	5.0	10	5.4	188	50	40 CFR 141 – Secondary Federal MCL
Arsenic	METAL	10/7/2009	6/11/2010	156	147	94.23%	µg/L	0.40	0.80	0.61	7.5	0.018	Clean Water Act -- Human Health Water + Organism
Chromium	METAL	10/7/2009	6/11/2010	156	156	100.00%	µg/L	--	--	5.6	4,460	65	Clean Water Act -- Freshwater CCC
Cobalt	METAL	10/7/2009	6/11/2010	156	44	28.21%	µg/L	0.050	0.22	0.062	3.0	2.6	WAC 173-340-730(3)(b)(iii)(A) and (B)
Cr(VI)	METAL	10/7/2009	6/11/2010	156	144	92.31%	µg/L	2.0	2.0	2.6	6,390	10	40 CFR 131 -- Freshwater CCC
Iron	METAL	10/7/2009	6/11/2010	156	110	70.51%	µg/L	18	18	17	7,840	300	40 CFR 141 – Secondary Federal MCL
Lithium	METAL	10/7/2009	6/11/2010	122	102	83.61%	µg/L	4.0	4.0	2.6	133	32	WAC 173-340-720(4)(b)(iii)(A) and (B)
Manganese	METAL	10/7/2009	6/11/2010	156	24	15.38%	µg/L	3.3	4.0	0.60	122	50	40 CFR 141 – Secondary Federal MCL
Mercury	METAL	10/7/2009	6/11/2010	156	1	0.64%	µg/L	0.050	0.10	0.11	0.11	0.012	40 CFR 131 -- Freshwater CCC
Selenium	METAL	10/7/2009	6/11/2010	156	150	96.15%	µg/L	0.60	0.60	0.38	7.1	5.0	Clean Water Act -- Freshwater CCC
Thallium	METAL	10/7/2009	6/11/2010	156	6	3.85%	µg/L	0.050	0.10	0.10	1.0	0.24	Clean Water Act -- Human Health Water + Organism
Zinc	METAL	10/7/2009	6/11/2010	156	36	23.08%	µg/L	5.2	6.0	0.90	260	91	WAC 173-201A
Strontium-90	RAD	10/7/2009	6/11/2010	156	19	12.18%	pCi/L	-14	2.6	2.2	27	8.0	40 CFR 141 - Primary Federal MCL
1,2-Dichloroethane	VOC	10/7/2009	6/11/2010	156	1	0.64%	µg/L	0.10	1.0	0.67	0.67	0.38	Clean Water Act -- Human Health Water + Organism
Bromodichloromethane	VOC	10/7/2009	6/11/2010	156	2	1.28%	µg/L	0.082	1.0	0.67	0.68	0.27	40 CFR 131 -- Human Health Water + Organism
Carbon tetrachloride	VOC	10/7/2009	6/11/2010	156	14	8.97%	µg/L	0.063	1.0	0.088	2.7	0.23	Clean Water Act -- Human Health Water + Organism
Chloroform	VOC	10/7/2009	6/11/2010	156	113	72.44%	µg/L	0.10	1.0	0.12	8.3	1.4	WAC 173-340-720(4)(b)(iii)(A) and (B)
Methylene chloride	VOC	10/7/2009	6/11/2010	156	18	11.54%	µg/L	0.11	1.0	0.12	11	4.6	Clean Water Act -- Human Health Water + Organism
Tetrachloroethene	VOC	10/7/2009	6/11/2010	156	8	5.13%	µg/L	0.088	1.0	0.093	0.43	0.081	WAC 173-340-720(4)(b)(iii)(A) and (B)

Note:

Shading indicates that the analyte is identified in the list of COPCs in DOE/RL-2009-40, *Sampling and Analysis Plan for the 100-DR-1, 100-DR-2, 100-HR-1, 100-HR-2, and 100-HR-3 Operable Units Remedial Investigation/Feasibility Study.*

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below the 99.9th percentile. The approach is consistent with the peer-reviewed *Guidelines for Exposure Assessment* (EPA/600/Z-92/001). Groundwater concentrations directly reflect potential exposures and risks, so a 90th percentile concentration reflects an RME scenario.

Groundwater datasets at the Hanford Site are highly skewed, with a large proportion of below detection limit (BDL) values. *Data Quality Assessment: Statistical Methods for Practitioners* (EPA/240/B-06/003) provides guidance for estimating statistical parameters (whether means or upper percentiles) depending on the variability in the dataset. The variability of the dataset is assed in terms of the CV and the proportion of observations that are BDL. For datasets with CVs greater than 0.5 and 50 percent or more observations that are BDL, EPA recommends using upper percentiles as opposed to means to develop summary statistics

Therefore, the rationale for using a 90th percentile value as an estimate of the EPC is consistent with the definition of an RME scenario, and is an appropriate statistic for groundwater datasets in this groundwater OU. Additional statistical evaluation of the 100-HR-3 Groundwater OU datasets that support the selection of the 90th percentile value as the EPC is provided in *Calculation of Exposure Point Concentrations for the 100-HR-3 Groundwater Operable Unit* (ECF-100HR3-10-0473) (Appendix G). This evaluation includes an estimation of the 95 percent UCL value for each detected analyte, along with the analysis of variability, to assess the reliability of the 95 percent UCL estimates. Results of the evaluation indicate that, for the majority of analytes, a reliable and meaningful 95 percent UCL estimate cannot be calculated, because of (1) an insufficient number of samples, (2) an insufficient number of detections, or (3) a high variance of the data. Therefore, the 90th percentile is adopted as the estimated EPC for all analytes. A comparison of the 90th percentile and 95 percent UCL values is provided in the uncertainty analysis (Section 6.3.8.2).

A flowchart depicting the COPC identification process and the number of analytes associated with each process step is provided on Figure 6-10. The steps in the sequence are described in the following sections.

6.3.2.3.7 *Identify Monitoring Wells in Each Exposure Area*

Three exposure areas are identified for the 100-HR-3 Groundwater OU including: (1) the 100-D Source exposure area, (2) the 100-H Source exposure area, and (3) the horn exposure area. Table 6-28 lists the monitoring wells associated with each exposure area.

6.3.2.3.8 *Identify Nondetected Analytes in Each Exposure Area*

Analytes that have not been detected in any of the groundwater samples from an exposure area are eliminated as COPCs for that exposure area. The analytes 1,2-dichloroethane and mercury were eliminated as COPCs in the 100-D Source exposure area. The analytes 1,2-dichloroethane, bromodichloromethane, and tetrachloroethene were eliminated as COPCs in the 100-H Source exposure area. The analytes bromodichloromethane, mercury, and thallium were eliminated as COPCs in the horn exposure area. All analytes detected at least once in an exposure area are carried forward to the next step of the process for that exposure area.

6.3.2.3.9 *Identify Analytes with 90th Percentile Values Less than Action Levels in Each Exposure Area.*

The 90th percentile values are compared to the lowest available action level for protection of human health and aquatic receptors. Comparisons of EPCs to action levels for the 100-D Source, 100-H Source, and horn exposure areas are provided in Tables 6-34, 6-35, and 6-36, respectively. A flow chart depicting this comparison is provided on Figure 6-11.

100-D Source Exposure Area. Eleven of the 19 analytes have been detected at least once in groundwater and have 90th percentile values less than their respective action levels (Table 6-34).

Seven of the eleven analytes (cobalt, manganese, selenium, strontium-90, sulfate, thallium, and zinc) were identified as historical COPCs in the work plan. A discussion of all analytes with EPCs less than the action level is provided in the following paragraphs.

Aluminum was detected in groundwater samples collected for the RI and the EPC is less than the secondary MCL. Detections of aluminum above the secondary MCL were reported in groundwater samples collected for the RI. Aluminum was not analyzed in the larger population of wells sampled over the past 6 years as described previously in Section 4.4.1.2 and Section 6.3.2.3.1. The secondary MCL is not federally enforceable, but intended as a guideline for states. As such, aluminum concentrations in groundwater are compared to the AWQC of 87 μg/L. Aluminum concentrations are less than the AWQC of 87 μg/L. Based on these results, aluminum is not retained as a COPC and will not be carried forward into the risk characterization section.

Cobalt was detected in groundwater samples collected for the RI and the EPC is less than the action level. Detections of cobalt above the action level were reported in groundwater samples collected for the RI and the larger population of wells as described previously in Section 4.4.1.2 and Section 6.3.2.3.1. Samples collected prior to the RI were analyzed by Method 6010 (SW 846 [SW-846]) where cobalt results (detected concentrations and MDLs) are generally greater than the action level. Cobalt results reported by Method 6010 (SW 846 [SW-846]) are considered uncertain because this method cannot accurately report trace concentrations. Additionally MDLs reported by Method 6010 are greater than the action level, resulting in nondetected concentrations greater than the action level. Most detected concentrations from Method 6010 were flagged with a "B" or a "C" laboratory qualifier. Sample results flagged with a "B" qualifier are considered estimates because concentrations are below the contract-required calibration range of the instrument. Sample results flagged with a "C" qualifier indicates that the analyte was detected in both the sample and the associated QC blank, and the sample concentration was less than or equal to five times the blank concentrations. Three RI samples were reported with cobalt concentrations greater than the action level; two of these three results were also flagged with a "C" laboratory qualifier. Although most cobalt concentrations were less than the action level in RI samples analyzed by trace methods, its historical presence with infrequent detections above the action level results in an uncertain status. Therefore, cobalt is retained as a COPC and warrants further evaluation in the FS.

Iron was detected in groundwater samples collected for the RI and the EPC is less than the secondary MCL. Detections of iron above the secondary MCL were reported in groundwater samples collected for the RI and the larger population of wells sampled over the past 6 years as described previously in Section 4.4.1.2 and Section 6.3.2.3.1. The secondary MCL is not federally enforceable, but intended as a guideline for states. As such, iron concentrations in groundwater are compared to the AWQC of 300 μg/L. Iron concentrations are less than the AWQC of 300 μg/L. In addition, iron concentrations in unfiltered and filtered water samples are less than the background level of 760 μg/L. Based on these results, iron is not retained as a COPC and will not be carried forward into the risk characterization section or into the FS.

Lithium was detected in groundwater samples collected for the RI and the EPC is less than the action level. Detections of lithium above the action level were reported in groundwater samples collected for the RI. Lithium was not analyzed in the larger population of wells sampled over the past 6 years as described previously in Section 4.4.1.2 and Section 6.3.2.3.1. A single detection of lithium was measured at well 199-D8-71 at a concentration greater than the action level of 32 μg/L. However, lithium concentrations in unfiltered and filtered water samples are less than the background level of 11,321 μg/L. Based on these results, lithium is not retained as a COPC and will not be carried forward into the risk characterization section or into the FS.

Table 6-34. Comparison of EPCs to Action Levels for the 100-D Source Exposure Area

Analyte Name	Analyte Class	Total Number of Samples	Number of Detects	Frequency of Detection	Units	Minimum Detection Limit	Maximum Detection Limit	Minimum Detected Result	Maximum Detected Result	90th Percentile of RI Data	Action Level	Action Level Basis	90th Percentile > Action Level?
Nitrate	ANION	60	60	100.00%	µg/L	--	--	10,800	99,200	69,500	45,000	40 CFR 141 – Primary Federal MCL	Yes
Sulfate	ANION	60	60	100.00%	µg/L	--	--	24,900	438,000	161,500	250,000	40 CFR 141 – Secondary Federal MCL	No
Aluminum	METAL	60	19	31.67%	µg/L	5	10	5.9	42	24	50	40 CFR 141 – Secondary Federal MCL	No
Arsenic	METAL	60	56	93.33%	µg/L	0.8	0.8	0.61	2.9	2.6	0.018	Clean Water Act -- Human Health Water + Organism	Yes
Chromium	METAL	60	60	100.00%	µg/L	--	--	7.7	4,460	925	65	Clean Water Act -- Freshwater CCC	Yes
Cobalt	METAL	60	23	38.33%	µg/L	0.05	0.22	0.099	3.0	1.3	2.6	WAC 173-340-730(3)(b)(iii)(A) and (B)	No
Cr(VI)	METAL	60	60	100.00%	µg/L	--	--	7.9	6,390	992	10	40 CFR 131 -- Freshwater CCC	Yes
Iron	METAL	60	39	65.00%	µg/L	18	18	22	265	106	300	40 CFR 141 - Secondary Federal MCL	No
Lithium	METAL	47	42	89.36%	µg/L	4	4	4.3	133	21	32	WAC 173-340-720(4)(b)(iii)(A) and (B)	No
Manganese	METAL	60	3	5.00%	µg/L	3.3	4	5.5	47.0	4.0	50	40 CFR 141 - Secondary Federal MCL	No
Selenium	METAL	60	56	93.33%	µg/L	0.6	0.6	0.38	6.5	4.4	5.0	Clean Water Act -- Freshwater CCC	No
Thallium	METAL	60	4	6.67%	µg/L	0.05	0.1	0.12	1.0	0.10	0.24	Clean Water Act -- Human Health Water + Organism	No
Zinc	METAL	60	18	30.00%	µg/L	5.2	6	6.4	260	34	91	WAC 173-201A	No
Strontium-90	RAD	60	3	5.00%	pCi/L	-14	2.4	2.3	3.7	0.67	8.0	40 CFR 141 - Primary Federal MCL	No
Bromodichloromethane	VOC	60	2	3.33%	µg/L	0.082	1	0.67	0.68	1.0	0.27	40 CFR 131 -- Human Health Water + Organism	Yes
Carbon tetrachloride	VOC	60	2	3.33%	µg/L	0.063	1	2.6	2.7	1.0	0.23	Clean Water Act -- Human Health Water + Organism	Yes
Chloroform	VOC	60	50	83.33%	µg/L	1	1	0.12	8.3	5.1	1.4	WAC 173-340-720(4)(b)(iii)(A) and (B)	Yes
Methylene chloride	VOC	60	6	10.00%	µg/L	0.11	1	0.16	0.27	1.0	4.6	Clean Water Act -- Human Health Water + Organism	No
Tetrachloroethene	VOC	60	1	1.67%	µg/L	0.088	1	0.14	0.14	1.0	0.081	WAC 173-340-720(4)(b)(iii)(A) and (B)	Yes

Table 6-35. Comparison of EPCs to Action Levels for the 100-H Source Exposure Area

Analyte Name	Analyte Class	Total Number of Samples	Number of Detects	Frequency of Detection	Units	Minimum Detection Limit	Maximum Detection Limit	Minimum Detected Result	Maximum Detected Result	90th Percentile of RI Data	Action Level	Action Level Basis	90th Percentile > Action Level?
Nitrate	ANION	38	38	100.00%	µg/L	--	--	16,700	46,900	39,800	45,000	40 CFR 141 – Primary Federal MCL	No
Sulfate	ANION	39	39	100.00%	µg/L	--	--	38,000	88,700	79,700	250,000	40 CFR 141 - Secondary Federal MCL	No
Aluminum	METAL	39	13	33.33%	µg/L	10	10	6.1	188	41	50	40 CFR 141 - Secondary Federal MCL	No
Arsenic	METAL	39	39	100.00%	µg/L	--	--	1.4	3.7	3.3	0.018	Clean Water Act -- Human Health Water + Organism	Yes
Chromium	METAL	39	39	100.00%	µg/L	--	--	7.3	39	31	65	Clean Water Act -- Freshwater CCC	No
Cobalt	METAL	39	16	41.03%	µg/L	0.050	0.10	0.062	0.90	0.43	2.6	WAC 173-340-730(3)(b)(iii)(A) and (B)	No
Cr(VI)	METAL	39	34	87.18%	µg/L	2.0	2.0	2.6	29	26	10	40 CFR 131 -- Freshwater CCC	Yes
Iron	METAL	39	29	74.36%	µg/L	18	18	17	7,840	444	300	40 CFR 141 - Secondary Federal MCL	Yes
Lithium	METAL	32	27	84.38%	µg/L	4.0	4.0	4.4	23	14	32	WAC 173-340-720(4)(b)(iii)(A) and (B)	No
Manganese	METAL	39	8	20.51%	µg/L	4.0	4.0	12	120	35	50	40 CFR 141 - Secondary Federal MCL	No
Mercury	METAL	39	1	2.56%	µg/L	0.050	0.10	0.11	0.11	0.10	0.012	40 CFR 131 -- Freshwater CCC	Yes
Selenium	METAL	39	38	97.44%	µg/L	0.60	0.60	0.83	3.2	2.7	5.0	Clean Water Act -- Freshwater CCC	No
Thallium	METAL	39	2	5.13%	µg/L	0.050	0.10	0.10	0.28	0.10	0.24	Clean Water Act -- Human Health Water + Organism	No
Zinc	METAL	39	9	23.08%	µg/L	6.0	6.0	2.8	30	16	91	WAC 173-201A	No
Strontium-90	RAD	39	12	30.77%	pCi/L	-7.8	2.6	3.2	27	14	8.0	40 CFR 141 - Primary Federal MCL	Yes
Carbon tetrachloride	VOC	39	2	5.13%	µg/L	0.063	1.0	0.088	2.0	1.0	0.23	Clean Water Act -- Human Health Water + Organism	Yes
Chloroform	VOC	39	31	79.49%	µg/L	1.0	1.0	0.55	1.7	1.4	1.4	WAC 173-340-720(4)(b)(iii)(A) and (B)	No
Methylene chloride	VOC	39	5	12.82%	µg/L	0.11	1.0	0.13	11	1.0	4.6	Clean Water Act -- Human Health Water + Organism	No

Table 6-36. Comparison of EPCs to Action Levels for the Horn Exposure Area

Analyte Name	Analyte Class	Total Number of Samples	Number of Detects	Frequency of Detection	Units	Minimum Detection Limit	Maximum Detection Limit	Minimum Detected Result	Maximum Detected Result	90th Percentile of RI Data	Action Level	Action Level Basis	90th Percentile > Action Level?
Nitrate	ANION	57	57	100.00%	µg/L	--	--	7,880	33,900	29,550	45,000	40 CFR 141 - Primary Federal MCL	No
Sulfate	ANION	57	57	100.00%	µg/L	--	--	30,000	97,300	78,350	250,000	40 CFR 141 – Secondary Federal MCL	No
Aluminum	METAL	57	33	57.89%	µg/L	5	10	5.4	150	54	50	40 CFR 141 - Secondary Federal MCL	Yes
Arsenic	METAL	57	52	91.23%	µg/L	0.4	0.8	0.6	7.5	5.5	0.018	Clean Water Act -- Human Health Water + Organism	Yes
Chromium	METAL	57	57	100.00%	µg/L	--	--	6	88	76	65	Clean Water Act -- Freshwater CCC	Yes
Cobalt	METAL	57	5	8.77%	µg/L	0.05	0.10	0.07	0.21	0.10	2.6	WAC 173-340-730(3)(b)(iii)(A) and (B)	No
Cr(VI)	METAL	57	50	87.72%	µg/L	2	2	4	90	71	10	40 CFR 131 -- Freshwater CCC	Yes
Iron	METAL	57	42	73.68%	µg/L	18	18	18	2490	422	300	40 CFR 141 - Secondary Federal MCL	Yes
Lithium	METAL	43	33	76.74%	µg/L	4	4	3	16	12	32	WAC 173-340-720(4)(b)(iii)(A) and (B)	No
Manganese	METAL	57	13	22.81%	µg/L	4	4	1	122	11	50	40 CFR 141 - Secondary Federal MCL	No
Selenium	METAL	57	56	98.25%	µg/L	0.6	0.6	0.9	7.1	3.2	5.0	Clean Water Act -- Freshwater CCC	No
Zinc	METAL	57	9	15.79%	µg/L	1	6	6	46	12	91	WAC 173-201A	No
Strontium-90	RAD	57	4	7.02%	pCi/L	-9.70	1.00	2.20	4.20	0.90	8.0	40 CFR 141 - Primary Federal MCL	No
1,2-Dichloroethane	VOC	57	1	1.75%	µg/L	0.1	1.0	0.7	0.7	1.0	0.38	Clean Water Act -- Human Health Water + Organism	Yes
Carbon tetrachloride	VOC	57	10	17.54%	µg/L	0.1	1.0	0.2	1.7	1.3	0.23	Clean Water Act -- Human Health Water + Organism	Yes
Chloroform	VOC	57	32	56.14%	µg/L	0.1	1.0	0.2	1.0	1.0	1.4	WAC 173-340-720(4)(b)(iii)(A) and (B)	No
Methylene chloride	VOC	57	7	12.28%	µg/L	0.1	1.0	0.1	0.6	1.0	4.6	Clean Water Act -- Human Health Water + Organism	No
Tetrachloroethene	VOC	57	7	12.28%	µg/L	0.1	1.0	0.1	0.4	1.0	0.081	WAC 173-340-720(4)(b)(iii)(A) and (B)	Yes

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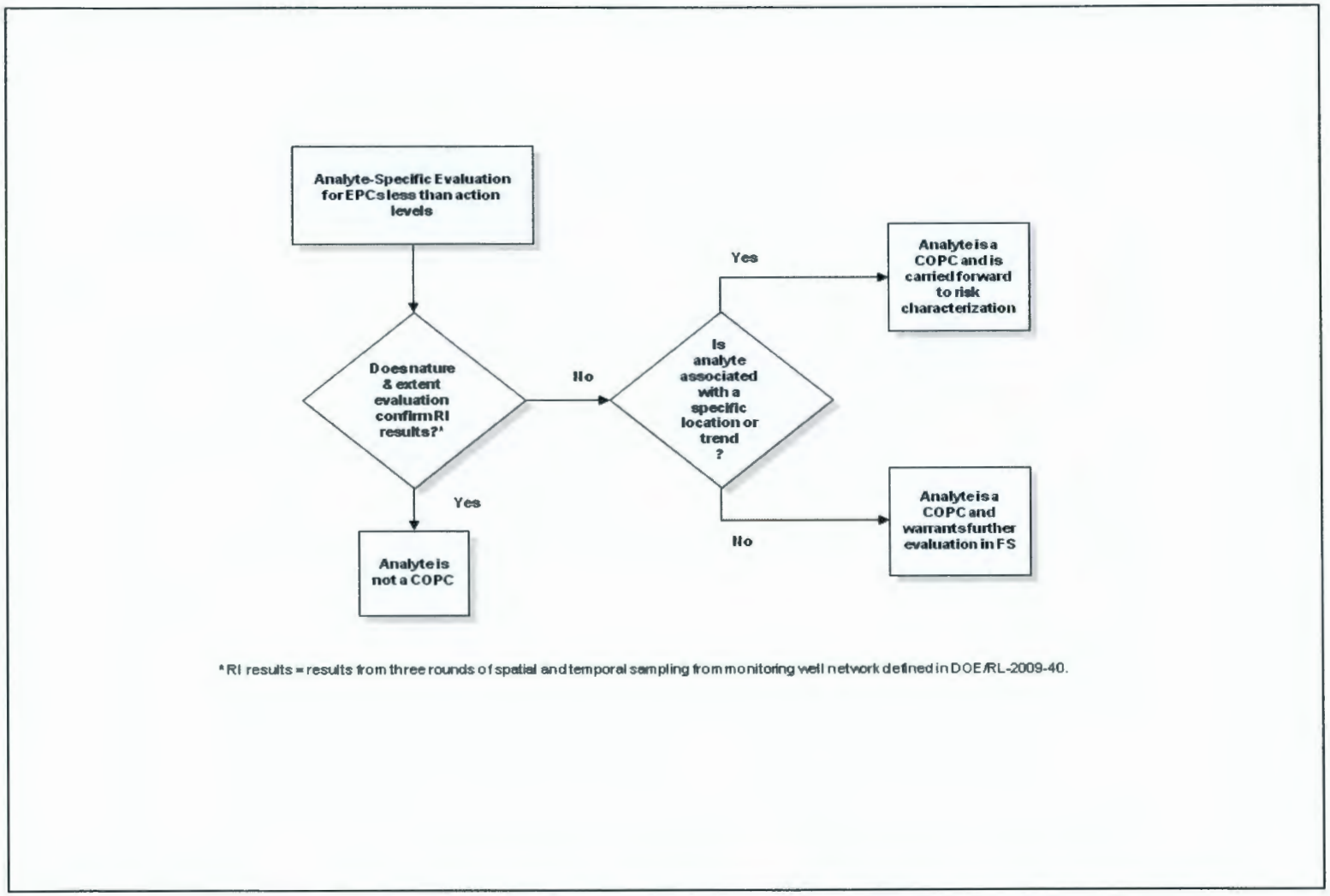

Figure 6-11. Analyte-Specific Evaluation Process for EPCs Less than Action Levels

Manganese was detected in groundwater samples collected for the RI and the EPC is less than the secondary MCL. Detections of manganese above the secondary MCL were reported in groundwater samples collected for the RI and the larger population of wells sampled over the past 6 years as described previously in Section 4.4.1.2 and Section 6.3.2.3.1. The secondary MCL is not federally enforceable, but intended as a guideline for states. As such, manganese concentrations are compared to the MTCA ("Surface Water Cleanup Standards" [WAC 173-340-730]) of 907 μg/L. Manganese concentrations reported in RI samples and the larger population of wells are less than 907 μg/L. Based on these results, manganese is not retained as a COPC and will not be carried forward into the risk characterization section or into the FS.

Methylene chloride was detected in groundwater samples collected for the RI and the EPC is less than the action level. Methylene chloride was not analyzed in the larger population of wells sampled over the past 6 years as described previously in Section 4.4.1.2 and Section 6.3.2.3.1. All detections of methylene chloride were less than the action level of 4.6 μg/L. Based on these results, methylene chloride is not retained as a COPC and will not be carried forward into the risk characterization section.

Selenium was detected in groundwater samples collected for the RI and the EPC is less than the action level. Selenium concentrations above the action level of 5 μg/L were measured in four RI wells and in the larger population of wells sampled over the past 6 years as described previously in Section 4.4.1.2 and Section 6.3.2.3.1. However, all selenium concentrations in filtered and unfiltered groundwater samples are less than or equal to the 90th percentile Hanford Site background level of 11 μg/L. Therefore, selenium is not retained a COPC and will not be carried forward into the risk characterization section or the FS.

Strontium-90 was detected in groundwater samples collected for the RI and the EPC is less than the action level. With the exception of a single result reported at well 199-D5-132, all strontium-90 concentrations in RI samples and the larger population of wells were less than the DWS of 8 pCi/L. Strontium-90 was reported at a concentration of 45 pCi/L at well 199-D5-32, this is the only result reported at this well during the specified time period because it was installed during the RI to fill data gap 2 and data gap 5. Additionally, well 199-D5-12, located south of the 116-D-1A liquid waste stream, historically reported strontium-90 concentrations above the DWS (with concentrations up to 52.6 pCi/L) until it was decommissioned in 2002. Based on these results, strontium-90 is retained as a COPC and will be carried forward into the FS for further evaluation.

Sulfate was detected in groundwater samples collected for the RI and the EPC is less than the secondary MCL. Sulfate concentrations above the secondary MCL were measured in two RI wells and in the larger population of wells sampled over the past 6 years as described previously in Section 4.4.1.2 and Section 6.3.2.3.1. Sulfate concentrations are associated with a trend at wells 199-D4-23, 199-D4-84, 199-D4-13, and 199-D4-19 where concentrations are above the secondary MCL. The presence of sulfate in these wells is associated with sodium dithionite, which is used for the ISRM barrier at the OU and is not the result of a Hanford Site release. Therefore, sulfate is not a COPC and will not be carried forward into the risk characterization or into the FS.

Thallium was detected in groundwater samples collected for the RI and the EPC is less than the action level. Thallium was detected in four groundwater samples collected for the RI at concentrations above the action level but below the EQL of 2 μg/L identified in the 100-D/H SAP (DOE/RL-2009-40). In addition, the EPC is less than the EQL of 2 μg/L. Thallium concentrations detected in the larger population of wells were also less than the EQL. It should also be noted that concentrations of thallium in filtered groundwater samples are less than the 90th percentile Hanford Site background level of 1.7 μg/L. Based on these results, thallium is not retained as a COPC and will not be carried forward into the risk characterization section or into the FS.

Zinc was detected in groundwater samples collected for the RI and the EPC is less than the AWQC. Zinc concentrations above the AWQC were measured in five RI wells and in the larger population of wells sampled over the past 6 years as described previously in Section 4.4.1.2 and Section 6.3.2.3.1. Detections of zinc above the AWQC were reported in the larger population of wells during 2006. It is likely that the presence of zinc in these samples is associated with a source of zinc that was introduced in the laboratory. Zinc is also associated with a trend at 199-D3-2, 199-D4-20, and 199-D4-84 where concentrations are above the AWQC. The presence of zinc in these wells is associated with sodium dithionite, which is used for the ISRM barrier at the OU and is not the result of a Hanford Site release. Therefore, zinc is not retained as a COPC and will not be carried forward into the risk characterization or into the FS.

100-H Source Exposure Area. Twelve of 18 analytes have been detected at least once in groundwater and have 90th percentile values less than their respective action level (Table 6-35).

Nine analytes (chloroform, chromium, cobalt, manganese, nitrate, selenium, sulfate, thallium, and zinc) were identified as historical COPCs in the work plan. A discussion of all analytes with EPCs less than the action level is provided in the following paragraphs.

Aluminum was detected in groundwater samples collected for the RI and the EPC is less than the secondary MCL. Detections of aluminum above the secondary were reported in groundwater samples collected for the RI. Aluminum was not analyzed in the larger population of wells sampled over the past 6 years as described previously in Section 4.4.1.2 and Section 6.3.2.3.1. The secondary MCL is not federally enforceable, but intended as a guideline for states. As such, aluminum concentrations in groundwater are compared to the AWQC of 87 µg/L. Aluminum concentrations are greater than the AWQC of 87 µg/L in the unfiltered samples at 199-H4-9 where concentrations range between 118 and 188 µg/L. However, filtered aluminum results from 199-H4-9 were less than the AWQC. Based on these results, aluminum is not retained as a COPC and will not be carried forward into the risk characterization section.

Chloroform was detected in groundwater samples collected for the RI and the EPC is less than the action level. The action level for chloroform is 1.4 µg/L based on the MTCA ("Groundwater Cleanup Standards" [WAC 173-340-720]) level; however, it defaults to the EQL of 5 µg/L reported in DOE/RL-2009-40 when the analytical method cannot achieve the action level. Detections of chloroform in RI samples and the larger population of wells sampled over the past 6 years are less than the EQL. Therefore, chloroform is not retained as a COPC and will not be carried forward into the risk characterization section or into the FS.

Chromium was detected in groundwater samples collected for the RI and the EPC is less than the AWQC of 65 µg/L. Detections of chromium above the AWQC were measured in the larger population of wells sampled over the past 6 years as described previously in Section 4.4.1.2 and Section 6.3.2.3.1. Except for chromium detected at 199-H3-5, chromium concentrations above the action level are not associated with a specific location or with a trend. Chromium concentrations above the action level are associated with a trend at 199-H3-5; however, Cr(VI) is collocated at this well with concentrations greater than the State water quality standard. The results of this evaluation indicate that chromium is present in groundwater; however, infrequent detections above the action level result in an uncertain status. Therefore, chromium is retained as a COPC and warrants further evaluation in the FS. Although all monitoring wells within the groundwater plume area were compared to the AWQC, only near-river wells would need to meet this criterion.

Cobalt and zinc were detected in groundwater samples collected for the RI and the EPCs are less than the action level. Detections of cobalt and zinc above the action levels were measured in RI samples and the larger population of wells sampled over the past 6 years as described previously in Section 4.4.1.2 and Section 6.3.2.3.1. Detections of cobalt and zinc above the action level are not associated with a specific

location or with a trend and many results were flagged with a combination of "B" [18] and "C" [19] qualifiers. The results of this evaluation indicate that although cobalt and zinc were detected at concentrations less than the action level in samples collected for the RI and the larger population of wells, their historical presence with infrequent detections above the action level result in an uncertain status. Therefore, cobalt and zinc are retained as COPCs and warrant further evaluation in the FS.

Selenium and sulfate were detected in groundwater samples collected for the RI and their EPCs are less than the action level or secondary MCL, respectively. Detections of these analytes in RI samples and the larger population of wells sampled over the past 6 years as described previously in Section 4.4.1.2 and Section 6.3.2.3.1 were measured at concentrations less than their action level or secondary MCL. In addition, selenium concentrations in filtered and unfiltered samples are less than the 90th percentile Hanford Site background level. Based on these results, selenium and sulfate are not retained as COPCs and will not be carried forward into the risk characterization section or into the FS.

Lithium was detected in groundwater samples collected for the RI and the EPC is less than the action level. Lithium was not analyzed in the larger population of wells sampled over the past 6 years as described previously in Section 4.4.1.2 and Section 6.3.2.3.1. All detections of lithium are less than the action level of 32 µg/L. Additionally, lithium concentrations in unfiltered and filtered water samples are less than the background level of 11,321 µg/L. Based on these results, lithium is not retained as a COPC and will not be carried forward into the risk characterization section or into the FS.

Manganese was detected in groundwater samples collected for the RI and the EPC is less than the action level. Detections of manganese above the action level were reported in groundwater samples collected for the RI and the larger population of wells sampled over the past 6 years as described previously in Section 4.4.1.2 and Section 6.3.2.3.1. The secondary MCL is not federally enforceable, but intended as a guideline for states. As such, manganese concentrations are compared to the MTCA ("Surface Water Cleanup Standards" [WAC 173-340-730]) of 907 µg/L. Manganese concentrations reported in RI samples and the larger population of wells are less than 907 µg/L. Based on these results, manganese is not a COPC and will not be carried forward into the risk characterization section or into the FS.

Methylene chloride was detected in groundwater samples collected for the RI and the EPC is less than the action level. Methylene chloride was not analyzed in the larger population of wells sampled over the past 6 years as described previously in Section 4.4.1.2 and Section 6.3.2.3.1. All detections of methylene chloride were less than the action level of 4.6 µg/L. Based on these results, methylene chloride is not a COPC and will not be carried forward into the risk characterization section.

Thallium was detected in groundwater samples collected for the RI and the EPC is less than the action level. In addition, the EPC is less than the EQL of 2 µg/L identified in the 100-D/H SAP (DOE/RL-2009-40). Thallium concentrations detected in the larger population of wells sampled over the past 6 years as described previously in Section 4.4.1.2 and Section 6.3.2.3.1 were also less than the EQL. Based on these results, thallium is not retained as a COPC and will not be carried forward into the risk characterization section or into the FS. It should also be noted that concentrations of thallium in filtered groundwater samples are less than the 90th percentile Hanford Site background level of 1.7 µg/L.

Nitrate was detected in groundwater samples collected for the RI and the EPC is less than the MCL. Detections of nitrate above the MCL were reported in the RI samples and the larger population of wells

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[18] "B" laboratory qualifier indicates that the analyte (inorganics) was detected at a value less than the contract-required detection limit, but greater than or equal to the instrument detection limit or the MDL, as appropriate.

[19] A "C" laboratory qualifier indicates that the analyte (inorganics) was detected in both the sample and the associated QC blank, and the sample concentration was less than or equal to five times the blank concentration

sampled over the past 6 years as described previously in Section 4.4.1.2 and Section 6.3.2.3.1. Nitrate concentrations at or above the MCL were measured at 199-H4-3, 199-H4-46, and 199-H6-1. Concentrations range between 27,400 and 253,000 µg/L in these wells. Nitrate is retained as a COPC because it is associated with a trend and will be carried forward into the risk characterization section.

Horn Exposure Area. Ten of 19 analytes have been detected at least once in groundwater and have 90th percentile values less than their respective action level (Table 6-36).

Eight analytes (chloroform, cobalt, manganese, nitrate, selenium, strontium-90, sulfate, and zinc) were identified as historical COPCs in the work plan. A discussion of all analytes with EPCs less than action levels is provided in the following paragraphs.

Chloroform was detected in groundwater samples collected for the RI and the EPC is less than the action level. The action level for chloroform is 1.4 µg/L based on the MTCA ("Groundwater Cleanup Standards" [WAC 173-340-720]) level; however, it defaults to the EQL of 5 µg/L reported in DOE/RL-2009-40 when the analytical method cannot achieve the action level. Detections of chloroform are less than the EQL in the RI samples and in the larger population of wells. Therefore, chloroform is not retained a COPC and will not be carried forward into the risk characterization or into the FS.

Nitrate, sulfate, and strontium-90 were detected in groundwater samples collected for the RI and their EPCs are less than the MCL or secondary MCL. Detections of these analytes were reported in RI samples and the larger population of wells sampled over the past 6 years as described previously in Section 4.4.1.2 and Section 6.3.2.3.1. All measured concentrations were less than the MCL or the secondary MCL. Based on these results, nitrate, strontium-90, and sulfate are not retained as COPCs and will not be carried forward into the risk characterization section or into the FS.

Lithium was detected in groundwater samples collected for the RI and the EPC is less than the action level. Lithium was not analyzed in the larger population of wells sampled over the past 6 years as described previously in Section 4.4.1.2 and Section 6.3.2.3.1. All detections of lithium are less than the action level of 32 µg/L. Based on these results, lithium is not retained as a COPC and will not be carried forward into the risk characterization section or into the FS.

Manganese was detected in groundwater samples collected for the RI and the EPC is less than the secondary MCL. Detections of manganese above the secondary MCL were reported in groundwater samples collected for the RI and the larger population of wells sampled over the past 6 years as described previously in Section 4.4.1.2 and Section 6.3.2.3.1. The secondary MCL is not federally enforceable, but intended as a guideline for states. As such, manganese concentrations are compared to the MTCA ("Surface Water Cleanup Standards" [WAC 173-340-730]) of 907 µg/L. Manganese concentrations reported in RI samples and the larger population of wells are less than 907 µg/L. Based on these results, manganese is not retained as a COPC and will not be carried forward into the risk characterization section or into the FS.

Methylene chloride was detected in groundwater samples collected for the RI and the EPC is less than the action level. Methylene chloride was not analyzed in the larger population of wells sampled over the past 6 years as described previously in Section 4.4.1.2 and Section 6.3.2.3.1. All detections of methylene chloride were less than the action level of 4.6 µg/L. Based on these results, methylene chloride is not retained as a COPC and will not be carried forward into the risk characterization section.

Cobalt and zinc were detected in groundwater samples collected for the RI and their EPCs are less than the respective action levels. These analytes have been detected at concentrations above the action level in the larger population of wells sampled over the past 6 years as described previously in Section 4.4.1.2 and Section 6.3.2.3.1. Samples collected prior to the RI were analyzed by Method 6010 (SW 846 [SW-846])

where results (detected concentrations and MDLs) are generally greater than the action level. Cobalt and zinc results reported by Method 6010 (SW 846 [SW-846]) are considered uncertain because this method cannot accurately report trace concentrations. Additionally MDLs reported by Method 6010 are greater than the action level resulting in nondetected concentrations greater than the action level. Most detected concentrations from Method 6010 (SW 846 [SW-846]) were flagged with a "B" or a "C" laboratory qualifier. Sample results flagged with a "B" qualifier are considered estimates because concentrations are below the contract-required calibration range of the instrument. Sample results flagged with a "C" qualifier indicate that the analyte was detected in both the sample and the associated QC blank, and the sample concentration was less than or equal to five times the blank concentrations. Although cobalt and zinc concentrations are less than action levels in RI samples, their historical presence with infrequent detections above the action level result in an uncertain status. Therefore, cobalt and zinc are retained as COPCs and warrant further evaluation in the FS.

Selenium was detected in groundwater samples collected for the RI and the EPC is less than the action level. Detections of selenium above the action level were measured in RI samples and in the larger population of wells sampled over the past 6 years as described previously in Section 4.4.1.2 and Section 6.3.2.3.1. Selenium concentrations in filtered and unfiltered samples are less than the 90th percentile Hanford Site background level. Based on these results, selenium is not retained as a COPC and will not be carried forward into the risk characterization section or into the FS.

6.3.2.3.10 *Identify COPCs with 90th Percentile Values Greater than Action Levels in Each Exposure Area.*

The 90th percentile values are compared to the lowest available action level for protection of human health and aquatic receptors. Comparisons of EPCs to action levels for the 100-D Source, 100-H Source, and horn exposure areas are provided in Tables 6-30, 6-31, and 6-32, respectively. A flow chart depicting this comparison is provided on Figure 6-12.

100-D Source Exposure Area. Eight of the 19 analytes have been detected at least once in groundwater and have 90th percentile values greater than their respective action levels (Table 6-34). A discussion of all analytes reported with an EPC greater than the action level is provided in the following paragraphs.

Arsenic, carbon tetrachloride, chloroform, chromium, Cr(VI), and nitrate were identified as historical COPCs in the work plan and are also listed on Table 6-34 because EPCs are greater than their respective action levels.

Arsenic is detected in groundwater samples collected for the RI and the EPC is above the action level. Detections of arsenic above the action level have also been measured in the larger population of wells sampled over the past 6 years as described previously in Section 4.4.1.2 and Section 6.3.2.3.1. Arsenic concentrations in all filtered and unfiltered samples are less than the 90th percentile Hanford Site background value of 7.85 µg/L. Based on these results, arsenic is a not retained as a COPC and will not be carried forward into the risk characterization section or into the F

Bromodichloromethane was detected in groundwater samples collected for the RI and the EPC is greater than the action level. Bromodichloromethane was not analyzed in the larger population of wells sampled over the past 6 years as described previously in Section 4.4.1.2 and Section 6.3.2.3.1. The action level for bromodichloromethane is 0.27 µg/L based on the MTCA ("Groundwater Cleanup Standards" [WAC 173-340-720]) level; however, it defaults to the EQL of 5 µg/L reported in DOE/RL-2009-40 when the analytical method cannot achieve the action level. Detections of bromodichloromethane in RI samples are less than the EQL. Therefore, bromodichloromethane is not retained as a COPC and will not be carried forward into the risk characterization section or into the FS.

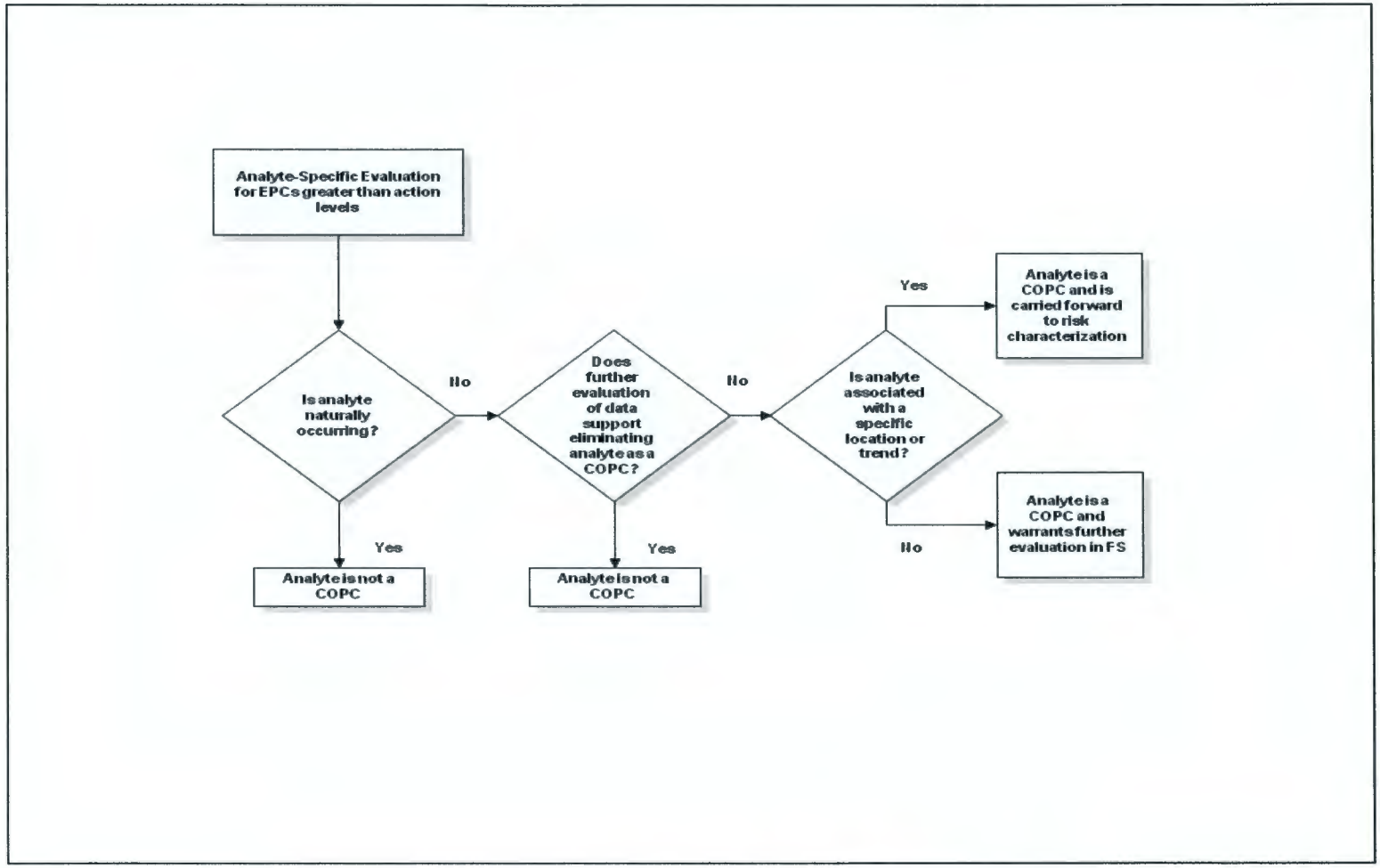

Figure 6-12. Analyte-Specific Evaluation Step for EPCs Greater Than Action Levels

Tetrachloroethane was detected in groundwater samples collected for the RI and the EPC is greater than the action level. Tetrachloroethene was not analyzed in the larger population of wells sampled over the past 6 years as described previously in Section 4.4.1.2 and Section 6.3.2.3.1. The action level for tetrachloroethene is 0.081 µg/L based on the MTCA ("Groundwater Cleanup Standards" [WAC 173-340-720]) level; however, it defaults to the EQL of 5 µg/L reported in DOE/RL-2009-40 when the analytical method cannot achieve the action level. Detections of tetrachloroethene in RI samples are less than the EQL. Therefore, tetrachloroethene is not retained as a COPC and will not be carried forward into the risk characterization section or into the FS.

Carbon tetrachloride was detected in groundwater samples collected for the RI and the EPC is greater than the action level. The action level for carbon tetrachloride is 0.23 µg/L based on the *National Recommended Water Quality Criteria* (EPA, 2009b), Human Health Water + Organism" value; however, it defaults to the EQL of 1 µg/L reported in the 100-D/H SAP (DOE/RL-2009-40) when the analytical method cannot achieve the action level. Detections of carbon tetrachloride were measured in the larger population of wells sampled over the past 6 years as described previously in Section 4.4.1.2 and Section 6.3.2.3.1 at concentrations greater than the EQL. Carbon tetrachloride was detected twice in 199-D2-6 and detected once in 199-D5-18 at concentrations greater than the EQL; however, carbon tetrachloride concentrations from two subsequent sample rounds were less than the detection limit. The results of the evaluation indicate that carbon tetrachloride has been historically detected in groundwater. Detections of carbon tetrachloride above the EQL were infrequent and were not associated with a specific location or with a trend resulting in an uncertain status. Therefore, carbon tetrachloride is a COPC and warrants further evaluation in the FS.S.

Chloroform, chromium, Cr(VI), and nitrate were detected in groundwater samples collected for the RI. Their EPCs are greater than the action level. Concentrations of chloroform, chromium, Cr(VI), and nitrate are widely distributed and are consistently present at concentrations above the DWS (nitrate), AWQC (chromium and Cr(VI)), or the EQL (chloroform). The distribution of these analytes within the groundwater OU are discussed in detail in Section 4.5.1. Based on the results of this evaluation, chloroform, chromium, Cr(VI), and nitrate are COPCs and are carried forward into the risk characterization section.

100-H Source Exposure Area. Six of the 18 analytes have been detected at least once in groundwater and have 90th percentile values greater than their respective action levels (Table 6-35). A discussion of all analytes reported with an EPC greater than the action level is provided in the following paragraphs.

Arsenic, carbon tetrachloride, Cr(VI), mercury, and strontium-90 were identified as historical COPCs in the work plan and are also listed on Table 6-35 because EPCs are greater than their respective action levels.

Arsenic is detected in groundwater samples collected for the RI and the EPC is above the action level. Detections of arsenic above the action level have also been measured in the larger population of wells sampled over the past 6 years as described previously in Section 4.4.1.2 and Section 6.3.2.3.1. Arsenic concentrations in all filtered and unfiltered samples are less than the 90th percentile Hanford Site background value of 7.85 µg/L. Based on these results, arsenic is a not a COPC and will not be carried forward into the risk characterization section or into the FS.

Carbon tetrachloride was detected in groundwater samples collected for the RI and the EPC is greater than the action level. The action level for carbon tetrachloride is 0.23 µg/L based on the *National Recommended Water Quality Criteria* (EPA, 2009b), Human Health Water + Organism" value; however, it defaults to the EQL of 1 µg/L reported in DOE/RL-2009-40 when the analytical method cannot achieve the action level. Detections of carbon tetrachloride above the EQL were measured in the larger population of wells sampled over the past 6 years as described previously in Section 4.4.1.2 and Section 6.3.2.3.1. Carbon tetrachloride was detected once in 199-H4-11 at a concentration greater than the EQL; however, two subsequent sample rounds were less than the detection limit. The results of this evaluation indicate that carbon

tetrachloride is detected infrequently at concentrations greater than the EQL and results in an uncertain status. Therefore, carbon tetrachloride is retained as a COPC and warrants further evaluation in the FS.

Iron was detected in groundwater samples collected for the RI and the EPC is greater than the secondary MCL. Detections of iron above the secondary MCL were reported in groundwater samples collected for the RI and the larger population of wells sampled over the past 6 years as described previously in Section 4.4.1.2 and Section 6.3.2.3.1. The secondary MCL is not federally enforceable, but intended as a guideline for states. As such, iron concentrations in groundwater are compared to the AWQC of 300 μg/L. Iron concentrations in unfiltered samples are greater than the AWQC of 300 μg/L; however all but one iron concentration in filtered samples were less than the AWQC. Additionally, iron concentrations in filtered water samples are less than the background level of 760 μg/L. Based on these results, iron is not retained a COPC and will not be carried forward into the risk characterization section or into the FS.

Mercury was detected in groundwater samples collected for the RI and the EPC is above the action level. The action level for mercury is 0.012 μg/L based on the AWQC; however, it defaults to the EQL of 0.05 μg/L reported in the 100-D/H SAP (DOE/RL-2009-40) when the analytical method cannot achieve the AWQC concentration. Detections of mercury were measured at concentrations less than the EQL of 0.05 μg/L in RI samples and in the larger population of wells sampled over the past 6 years as described previously in Section 4.4.1.2 and Section 6.3.2.3.1. Based on these results, mercury is not retained as a COPC and will not be carried forward into the risk characterization section or into the FS.

Cr(VI) and strontium-90 were detected in groundwater samples collected for the RI and their EPCs are greater than the State water quality standard or MCL, respectively. Concentrations of Cr(VI) and strontium-90 are widely distributed and are consistently present at concentrations above the State water quality standard (Cr(VI)) or the MCL (strontium-90). The distribution of these analytes within the groundwater OU is discussed in detail in Section 4.5.1. Based on the results of this evaluation, Cr(VI) and strontium-90 are both retained as COPCs and are carried forward into the risk characterization section.

Horn Exposure Area. Eight of the 18 analytes have been detected at least once in groundwater and have 90th percentile values greater than their respective action levels (Table 6-36). A discussion of all analytes reported with an EPC greater than the action level is provided in the paragraphs in the following text.

Arsenic, carbon tetrachloride, total chromium, and Cr(VI) were identified as historical COPCs in the work plan and are also listed on Table 6-36 because EPCs are greater than their respective action levels.

Aluminum was detected in groundwater samples collected for the RI and the EPC is greater than the secondary MCL. Detections of aluminum above the secondary were reported in groundwater samples collected for the RI. Aluminum was not analyzed in the larger population of wells sampled over the past 6 years as described previously in Section 4.4.1.2 and Section 6.3.2.3.1. The secondary MCL is not federally enforceable, but intended as a guideline for states. As such, aluminum concentrations in groundwater are compared to the AWQC of 87 μg/L. Aluminum concentrations are less than the AWQC of 87 μg/L. Based on these results, aluminum is not retained as a COPC and will not be carried forward into the risk characterization section.

Arsenic is detected in groundwater samples collected for the RI and the EPC is above the action level. Detections of arsenic above the action level have also been measured in the larger population of wells sampled over the past 6 years as described previously in Section 4.4.1.2 and Section 6.3.2.3.1. Arsenic concentrations in all filtered and unfiltered samples are less than the 90th percentile Hanford Site background value of 7.85 μg/L. Based on these results, arsenic is not retained as a COPC and will not be carried forward into the risk characterization section or into the FS.

Iron was detected in groundwater samples collected for the RI and the EPC is greater than the secondary MCL. Detections of iron above the secondary MCL were reported in groundwater samples collected for the RI and the larger population of wells sampled over the past 6 years as described previously in Section 4.4.1.2 and Section 6.3.2.3.1. The secondary MCL is not federally enforceable, but intended as a guideline for states. As such, iron concentrations in groundwater are compared to the AWQC of 300 µg/L. Iron concentrations are greater than the AWQC of 300 µg/L. However, iron concentrations in filtered water samples are less than the background level of 760 µg/L. Based on these results, iron is not a COPC and will not be carried forward into the risk characterization section or into the FS.

Tetrachloroethane was detected in groundwater samples collected for the RI and the EPC is greater than the action level. Tetrachloroethene was not analyzed in the larger population of wells sampled over the past 6 years as described previously in Section 4.4.1.2 and Section 6.3.2.3.1. The action level for tetrachloroethene is 0.081 µg/L based on the MTCA ("Groundwater Cleanup Standards" [WAC 173-340-720]) level; however, it defaults to the EQL of 5 µg/L reported in the 100-D/H SAP (DOE/RL-2009-40) when the analytical method cannot achieve the action level. Detections of tetrachloroethene in RI samples are less than the EQL. Therefore, tetrachloroethene is not retained as a COPC and will not be carried forward into the risk characterization section or into the FS.

1,2-Dichloroethane was detected in groundwater samples collected for the RI and the EPC is greater than the action level. 1,2-Dichloroethane was not analyzed in the larger population of wells sampled over the past 6 years as described previously in Section 4.4.1.2 and Section 6.3.2.3.1. The action level for 1,2-dichloroethane is 0.38 µg/L based on the MTCA ("Groundwater Cleanup Standards" [WAC 173-340-720]) groundwater cleanup level; however, it defaults to the EQL of 5 µg/L reported in DOE/RL-2009-40 when the analytical method cannot achieve the action level. Detections of 1,2-dichloroethane in RI samples are less than the EQL. Therefore, 1,2-dichloroethane is not retained as a COPC and will not be carried forward into the risk characterization section or into the FS.

Carbon tetrachloride was detected in groundwater samples collected for the RI and the EPC is greater than the action level. The action level for carbon tetrachloride is 0.23 µg/L based on the *National Recommended Water Quality Criteria* (EPA, 2009b), Human Health Water + Organism" value; however, it defaults to the EQL of 1 µg/L reported in the 100-D/H SAP (DOE/RL-2009-40) when the analytical method cannot achieve the action level. Detections of carbon tetrachloride greater than the EQL were measured in RI samples and the larger population of wells sampled over the past 6 years as described previously in Section 4.4.1.2 and Section 6.3.2.3.1. Detections of carbon tetrachloride above the EQL were infrequent and were not associated with a specific location or with a trend, resulting in an uncertain status. Therefore, carbon tetrachloride is retained as a COPC and warrants further evaluation in the FS.

Chromium and Cr(VI) were detected in groundwater samples collected for the RI and their EPCs are greater than the AWQC or the State water quality standard. Concentrations of chromium and Cr(VI) are widely distributed and are consistently present at concentrations above the AWQC. The distribution of these analytes within the groundwater OU is discussed in detail in Section 4.5.1. Based on the results of this evaluation, chromium and Cr(VI) are retained as COPCs and are carried forward into the risk characterization section.

6.3.2.4 Summary of COPCs

Table 6-37 presents a summary of the COPCs identified for the 100-HR-3 Groundwater OU. This list of COPCs represents the analytes most likely to contribute to overall risk within each 100-HR-3 Groundwater OU exposure area.

Table 6-37. Summary of Groundwater COPCs Identified for the 100-HR-3 Groundwater OU

100-D Source Exposure Area		
Metals	**VOCs**	**Nonradioactive Anions**
Antimony[a]	Carbon tetrachloride[a]	Nitrate
Cadmium[a]	Chloroform	
Chromium		
Cobalt[a]		
Copper[a]		
Cr(VI)	**Radionuclides**	
Lead[a]	Strontium-90	
Nickel[a]		
Silver[a]		
100-H Source Exposure Area		
Metals	**Radionuclides**	**Nonradioactive Anions**
Antimony[a]	Strontium-90	Nitrate[b]
Cadmium[a]		
Cobalt[a]	**VOCs**	
Copper[a]	Carbon tetrachloride[a]	
Cr(VI)		
Lead[a]		
Nickel[a]		
Silver[a]		
Zinc[a]		
Horn Exposure Area		
Metals	**VOCs**	
Antimony[a]	Carbon tetrachloride	
Cadmium[a]		
Chromium		
Cobalt[a]		
Copper[a]		
Cr(VI)		
Lead[a]		
Nickel[a]		
Silver[a]		
Zinc[a]		

a. EPC did not exceed action level, but infrequent detections above action level result in uncertain status and warrant further evaluation in the FS.

b. EPC did not exceed an action level but retained as a COPC due to localized contamination.

Chromium, Cr(VI), chloroform, and strontium-90 were identified as COPCs because the 90th percentile concentration exceeded the action level. As described in Section 6.3.2.3, nitrate and strontium-90 in the 100-H Source exposure area are identified as COPCs because it is associated with a localized source of contamination.

The COPC identification process identified ten analytes for the 100-HR-3 Groundwater OU that are considered COPCs and warrant further evaluation in the FS. The occurrence of antimony, cadmium, carbon tetrachloride, cobalt, copper, lead, nickel, silver, and zinc in groundwater is uncertain because these analytes historically have been detected in groundwater at concentrations above their respective action level; however, their presence was not associated with a specific location or a trend and the analytical methods used were not of sufficient accuracy for risk characterization purposes. In addition, the EPCs for these analytes (except carbon tetrachloride in the horn area) are less than their respective action level. Therefore, antimony, cadmium, carbon tetrachloride, cobalt, copper, lead, nickel, silver, and zinc are retained as COPCs where they warrant further evaluation in the FS; however, they will not move forward into the risk characterization section.

6.3.3 Exposure Assessment

The exposure assessment component of the risk assessment typically identifies the populations that may be exposed, the routes by which these receptors may become exposed, and the magnitude, frequency, and duration of potential exposures.

6.3.3.1 Contaminant Sources

Contaminant sources (that is, facilities and waste sites) were previously discussed in Section 6.2.3.1 and are listed in Sections 4.2.1 and 5.2 of this report.

6.3.3.2 Release Mechanisms and Environmental Transport Media

The primary COPC release mechanisms and transport pathways evaluated at the 100-D/H Source OU are discussed in Sections 5.3 and 5.4, and include the following:

- Direct contact with groundwater containing COPCs
- Volatilization of COPCs in groundwater from showering or household activities
- Discharge of groundwater to the Columbia River through upwelling and seeps

6.3.3.3 100-HR-3 Groundwater Operable Unit Exposure Areas

The 100-HR-3 Groundwater OU is generally distinguished by the presence of Cr(VI) plumes within the 100-D, 100-H, and the horn area as discussed in detail in Chapter 4. The 100-D Source exposure area represents the northern and southern plume sources in the 100-D area. The 100-H Source exposure area represents the plume sources within the 100-H Area. The horn exposure area represents the portion of the plume that is located in the horn area. The horn exposure area is located downgradient from the 100-D sources where contaminant concentrations have migrated over time. The primary objectives for evaluating each exposure area are to provide information necessary to determine the need for remedial action and to use this information to select the best remedy. These objectives are achieved by performing the following steps for each exposure area:

1. EPCs for each COPC are compared to action levels for understanding the potential for exposure to groundwater contaminants and the associated health risks.
2. Specific locations are identified within the exposure area for evaluating remedial alternatives in the FS.

The basis for each exposure area and the known or suspected sources are described in the following text. Exposure areas and the location of associated monitoring wells are shown on Figures 2-1 and 2-2. Table 6-28 lists the monitoring wells included in each exposure area.

6.3.3.4 Potentially Complete Human and Aquatic Exposure Pathways and Receptors

This section describes the potentially complete exposure pathways and receptors that are specifically addressed in the action levels (see Section 6.3.2.2) evaluated in this groundwater risk assessment.

6.3.3.4.1 *Action Levels Used to Evaluate Protection of Human Health*

All of the action levels for use as a drinking water source consider ingestion as a complete and significant pathway for exposure. Washington State regulations assume that inhalation of vapors for VOCs is also a complete and significant exposure pathway. Washington State regulations do not include the dermal contact exposure route in the equations for calculation of groundwater cleanup levels, whereas federal regulations consider dermal contact exposure a complete but insignificant groundwater contaminant exposure pathway. Elimination of the dermal contact exposure route from action levels may result in an overestimation of the cleanup level; uncertainties associated with exclusion of this exposure route are addressed in Section 6.3.6.4.

For groundwater with the potential to impact surface water, federal water quality standards assume that exposure to humans occurs through ingestion of water and consumption of fish tissue, and Washington State regulations assume that exposure occurs through consumption of fish tissue. These federal standards are developed for protection of human health where groundwater discharges to surface water that is used as a drinking water source and used for fishing. Washington State regulations as defined in MTCA ("Surface Water Cleanup Standards" [WAC 173-340-730(3)(b)]) developed surface water standards that assume that exposure occurs through consumption of fish tissue.

6.3.3.4.2 *Action Levels Used to Evaluate Protection of Aquatic Receptors*

The objectives and methodology for deriving the numerical AWQC are described in *Guidelines for Deriving Numerical National Water Quality Criteria for the Protection of Aquatic Organisms and Their Uses* (PB85-227049). The AWQC are intended to provide a reasonable level of protection of all except a small fraction (0.05) of the taxa, unless a commercially or recreationally important species is very sensitive. Protection of the following aquatic organisms and their uses are defined in *Guidelines for Deriving Numerical National Water Quality Criteria for the Protection of Aquatic Organisms and Their Uses* (PB85-227049) as prevention of unacceptable long-term and short-term effects:

- Commercially, recreationally, and other important species
- Fish and benthic invertebrate assemblages in rivers and streams
- Fish, benthic invertebrate, and zooplankton assemblages in lakes, reservoirs, estuaries, and oceans

Numeric values are expressed as two numbers, the criteria maximum concentration (CMC) and criteria continuous concentration (CCC), which provide an appropriate degree of protection of aquatic organisms and their uses from acute and chronic toxicity to animals, toxicity to plants, and bioaccumulation by aquatic organisms. The CMC is an estimate of the highest concentration of a material in surface water to which an aquatic community can be exposed briefly without resulting in an unacceptable effect. EPA derives acute criteria from 48- to 96-hour tests of lethality or immobilization. The CCC is an estimate of the highest concentration of a material in surface water to which an aquatic community can be exposed indefinitely without resulting in an unacceptable effect. EPA derives chronic criteria from longer-term (often greater than 28 days) tests that measure survival, growth, reproduction or, in some cases, bioconcentration. The CMC and the CCC are two of the six parts of the aquatic life criterion.

The other four parts are the acute averaging period, chronic averaging period, acute frequency of allowed exceedance, and chronic frequency of allowed exceedance. The lower of the CMC or the CCC is the numeric water quality criteria used as the action level for protection of freshwater species.

6.3.4 Toxicity Assessment

The toxicity assessment component evaluates the relationship between the magnitude of exposure to an analyte and the likelihood of adverse health effects to potentially exposed populations. Similar to the exposure assessment, the comparison to action levels takes into consideration the likelihood of an adverse health effect to occur to the potentially exposed population. The risk-based concentrations, such as the MTCA ("Groundwater Cleanup Standards" [WAC 173-340-720]), are developed using toxicological information published at EPA's IRIS database and EPA's hierarchy of toxicity values described in Section 6.2.2. The assignment of action levels to COPCs is described in Section 6.3.2.2.

6.3.4.1 State and Federal Maximum Contaminant Levels for Nonradionuclides

The MCLG is the maximum level of a contaminant in drinking water at which no known or anticipated adverse health effects occur, allowing for an adequate margin of safety. MCLGs are nonenforceable health goals. EPA establishes the MCL, an enforceable standard, based on the MCLG. The MCL is the maximum permissible level of a contaminant in water that is delivered to any user of a public water system. Prior to the *Safe Drinking Water Act of 1974* amendments in 1996, the MCL was set as close to the MCLG as was feasible. The 1996 Amendments to the SDWA permit consideration of costs and benefits in establishing an MCL. Primary MCLs are legally enforceable standards and protect public health by limiting the levels of contaminants in drinking water. Secondary MCLs are nonenforceable guidelines regulating those contaminants that may cause cosmetic effects (such as skin or tooth discoloration) or aesthetic effects (such as taste, odor, or color) in drinking water. The secondary MCLs are recommended standards but are not federally enforceable.

Six-Year Review Chemical Contaminants Health Effects Technical Support Document (EPA 822-R-03-008) describes how MCLGs are derived. MCLGs are developed using an oral RfD for contaminants that exhibit a threshold toxic effect. The RfD is an estimate (with uncertainty spanning perhaps an order of magnitude) of a daily oral exposure to the human population (including sensitive subgroups) that is likely to be without an appreciable risk of deleterious noncancer effects during a lifetime. EPA generally assumes that the relative source contribution from drinking water is 20 percent of the RfD, unless other exposure data for the chemical are available. This allows 80 percent of the total exposure to come from sources other than drinking water, such as exposure from food, inhalation, or dermal contact.

6.3.4.2 Maximum Contaminant Levels for Radionuclides in Drinking Water

Current MCLs for radionuclides are set at 4 mrem/yr for the sum of the doses from beta particle and photon emitters, 15 pCi/L for gross alpha emitter activity (including Ra-226, but excluding uranium and radon), and 5 pCi/L combined for Ra-226 and Ra-228. A mass-based concentration MCL of 30 μg/L has been established for uranium. The current MCLs for beta emitters specify that MCLs are to be calculated based on an annual dose equivalent of 4 mrem to the total body or any internal organ. It is further specified that the calculation be performed based on a 2 L (0.5 gal)/day drinking water intake using the 168-hour data listed in *Maximum Permissible Body Burdens and Maximum Permissible Concentrations of Radionuclides in Air and in Water for Occupational Exposure* (NBS Handbook 69).

6.3.4.3 Washington State Regulations

Toxicological parameter values are obtained from the CLARC database (Ecology, 2010) compendium of technical information related to the calculation of cleanup levels under the MTCA (WAC 173-340).

The sources for the oral cancer potency values and RfDs are provided in the CLARC database. The sources for identifying RfD s and carcinogenic potency factors are defined in MTCA ("Human Health Risk Assessment Procedures" [WAC 173-340-708(7) and WAC 173-340-708(8)]).

6.3.4.4 Toxicity Values

The sources of toxicity values for human health are the same as those described in Section 6.2.4.2 of the report.

As discussed in Section 6.3.3.4.2, the lower of the CMC or the CCC is the numeric water quality criteria used as the action level for protection of freshwater species. *Technical Support Document for Water Quality-based Toxics Control* (EPA/505/2-90-001) explains that development of national numerical water quality criteria for the protection of aquatic organisms is a complex process that uses information from many areas of aquatic toxicology. After it is decided that a national criterion is needed for a particular material, all available information concerning toxicity to and bioaccumulation by aquatic organisms is collected and reviewed for acceptability. If enough acceptable data for 48- to 96-hour toxicity tests on aquatic animals are available, they are used to derive the acute criterion. If sufficient data on the ratio of acute to chronic toxicity concentrations are available, they are used to derive the chronic or long-term exposure criteria. The chronic criteria can also be calculated directly if sufficient data are available. If justified, one or both of the criteria may be related to another water quality characteristic (for example, pH, temperature, or hardness). Separate criteria are developed for fresh water and salt water.

6.3.5 Risk Characterization

Risk characterization is the final step of the HHRA process. In this step, the toxicity values are combined with the estimated chemical intakes for the receptor populations in order to estimate both carcinogenic and noncarcinogens risks quantitatively. The risk characterization step is completed through the comparison of the EPC to the action level using the equations presented in Section 6.3.5.1. As described earlier in this section, the comparison to action levels determines whether existing groundwater concentrations are protective of human health and the environment. It is also used to determine whether current groundwater concentrations have the potential to exceed an HI greater than 1 or the upper end of the NCP risk range for cumulative carcinogenic site risk to an individual based on RME for both current and future land use.

6.3.5.1 Protectiveness Evaluation

Protectiveness of human health is determined by the comparison of 90th percentile groundwater concentrations to existing federal or state MCLs. Similarly, protectiveness of human and aquatic receptors is determined by the comparison of 90th percentile groundwater concentrations to water quality criteria established under Section 304 or 303 of the *Clean Water Act of 1977* and Washington State water quality standards.

This risk characterization step is included to address the presence of multiple exposure pathways or the potential for exposure to multiple contaminants. This step is also included to address the requirements of MTCA (HHRA Procedures [WAC 173-340-708(5)(a) and WAC 173-340-708(6)(b)]). These regulations require that cleanup levels be adjusted downward to take into account exposure to multiple hazardous substances or multiple pathways of exposure. This adjustment needs to be made only if without this adjustment, the HI would exceed 1 or the total ELCR would exceed 1 in 100,000 (1×10^{-5}).

To determine the potential to exceed an HI greater than 1 or the upper end of the NCP risk range for cumulative carcinogenic site risk to an individual based on RME for both current and future land use, the following standards are used:

- WAC 173-340-720, "Groundwater Cleanup Standards"
- WAC 173-340-730, "Surface Water Cleanup Standards"

- *National Recommended Water Quality Criteria* (EPA, 2009b)

For the purposes of this evaluation, the potential for unacceptable human health risk is identified using the following risk thresholds:

- ELCR values are compared to the "target range" of 10^{-6} to 10^{-4} that is generally used by EPA. MTCA (WAC 173-340) states that cancer risks resulting from multiple hazardous substances should not exceed 1×10^{-5} for unrestricted land use. ELCR values within or exceeding the target range require a risk management decision that includes evaluating site-specific characteristics and exposure scenario factors to assess whether remedial action is warranted.

- An HI (the sum of the ratios of the chemical intake to the RfDs for all COPCs) greater than 1 indicates that some potential exists for adverse noncancer health effects associated with exposure to the COPCs.

Although this groundwater risk assessment produces numerical estimates of risk, it should be recognized that these numbers might not predict actual health outcomes because they are based largely on hypothetical assumptions. Their purpose is to provide a frame of reference for risk management decision making. Interpretation of the risk estimates provided should consider the nature and weight of evidence supporting these estimates, as well as the magnitude of uncertainty surrounding them.

Protectiveness of humans from exposure to beta/photon emitters is determined by an annual dose equivalent to the body or any internal organ and determined by comparison to an activity concentration in drinking water for alpha emitters; therefore, the sum of fractions is used determine the annual dose from exposure to beta/photon emitters.

6.3.5.1.1 *Cancer Risk Estimation Method*

The potential for cancer effects is evaluated by estimating the ELCRs. This risk is the incremental increase in the probability of developing cancer during one's lifetime in addition to the background probability of developing cancer (that is, if no exposure to Hanford Site chemicals occurs). To estimate the cancer risks from exposure to an individual carcinogen from all exposure routes considered, the following equation is used.

$$Risk_I = \frac{EPC_{water}}{CUL_{carcinogen}} \times TR$$

where:

$Risk_I$ = ELCR for individual chemical

EPC_{water} = 90th percentile concentration in groundwater (µg/L)

$CUL_{carcinogen}$ = groundwater cleanup level based on 10^{-6} carcinogenic effect (µg/L)

TR = target ELCR for individual hazardous substance for unrestricted land use (10^{-6})

To estimate the cancer risks from exposure to multiple carcinogens from all exposure routes considered, the following equation is used.

$$Risk_T = \sum_i \frac{EPC_{water}}{CUL_{carcinogen}} \times TR$$

where:

$Risk_T$ = total ELCR for all chemicals

EPC_{water} = 90th percentile concentration in groundwater (µg/L)

$CUL_{carcinogen}$ = groundwater cleanup level based on 10^{-6} carcinogenic effect (µg/L)

TR = target ELCR for individual hazardous substance for unrestricted land use (10^{-6})

i = the sum of the ratios for the ith chemical

6.3.5.1.2 *Noncancer Risk Estimation Method*

For noncancer effects, the likelihood that a receptor will develop an adverse effect is estimated by comparing the predicted level of exposure for a particular chemical with the highest level of exposure that is considered protective (that is, its RfD). The ratio of the chronic daily intake divided by RfD is the HQ.

When the HQ for a chemical exceeds 1 (that is, exposure exceeds RfD), a concern exists for potential noncancer health effects. To estimate the HQ from all exposure routes considered for an individual hazardous substance, the following equation is used.

$$HQ = \frac{EPC_{water}}{CUL_{noncarcinogen}}$$

where:

HQ = HQ for individual chemical

EPC_{water} = 90th percentile concentration in groundwater (µg/L)

$CUL_{noncarcinogen}$ = groundwater cleanup level based on HQ = 1 noncarcinogenic effects (µg/L)

To estimate the HI from all exposure routes considered for multiple hazardous substances, the following equation is used.

$$HI_T = \sum_i \frac{EPC_{water}}{CUL_{noncarcinogen}}$$

where:

HI_T = total HI for all chemicals

EPC_{water} = 90th percentile concentration in groundwater (µg/L)

$CUL_{noncarcinogen}$ = groundwater cleanup level based on HQ=1 noncarcinogenic effects (µg/L)

i = sum of the ratios for the ith chemical

6.3.5.1.3 *Estimating the Sum of Fractions and 4 mrem/yr Dose Equivalent*

An annual cumulative dose equivalent of 4 mrem to the total body or any internal organ from beta and photon emitters is considered protective of human health. The sum of fractions is used to determine whether the contribution of each radioisotope is greater than the cumulative annual dose equivalent of 4 mrem. The following equation is used to determine whether the 4 mrem standard is exceeded when a mixture of radioisotopes is present:

$$Sum\ of\ Fractions = \left(\frac{A\left(\frac{pCi}{L}\right)}{MCL_A\left(\frac{pCi}{L}\right)} + \frac{B\left(\frac{pCi}{L}\right)}{MCL_B\left(\frac{pCi}{L}\right)} + \bullet\bullet\bullet \right)$$

where:

A = EPC activity concentration of specific beta/photon emitting nuclide A

B = EPC activity concentration of specific beta/photon emitting nuclide B

MCL_A = derived single-nuclide beta/photon emitting MCL-equivalent activity concentration for nuclide A

MCL_B = derived single-nuclide beta/photon emitting MCL-equivalent activity concentration for nuclide B

The 4 mrem standard is not exceeded if the sum of fractions is less than 1. Each fraction is converted to a dose equivalent of 4 mrem/year by multiplying the fraction by 4.

6.3.6 Risk Characterization Results Using Action Levels by Exposure Area

Action levels that are considered protective of human health and the environment were used to identify COPCs that warrant further evaluation in the FS. The lowest of the available action levels was selected for comparison if more than one action level exists for a certain analyte. The analytes listed in Tables 6-38, 6-39, and 6-40 are considered COPCs because the 90th percentile groundwater concentration is greater than the lowest available action level, or the analyte is measured at concentrations above the lowest action level in a localized area.

Table 6-38. Summary of Current Conditions 90th Percentile Groundwater Concentrations, Federal and State MCLs, and WAC 173-340-720 Groundwater Cleanup Levels for the 100-D Source Exposure Area (Human Health Action Levels)

COPCs	Units	90th Percentile Value	Federal MCL	State MCL	MTCA, "Groundwater Cleanup Standards" (WAC 173-340-720) Cleanup Levels		
					Noncarcinogens	Carcinogens at 10^{-6} Risk Level	Carcinogens at 10^{-5} Risk Level
Chloroform	µg/L	5.1	80	--	80	1.4	14
Chromium	µg/L	925	100	100	24,000	--	--
Cr(VI)	µg/L	992	--	--	48	--	--
Nitrate	µg/L	69,500	45,000	45,000	113,600	--	--

Source: WAC 173-340-720, "Model Toxics Control Act—Cleanup," "Groundwater Cleanup Standards."

Table 6-39. Summary of Current Conditions 90th Percentile Groundwater Concentrations, and Federal and State Water Quality Standards for the 100-D Source Exposure Area (Aquatic Action Levels)

COPCs	Units	90th Percentile Value	AWQC		WAC 173-201A	40 CFR 131 Water Quality Standards	
			Freshwater CMC (acute)	Freshwater CCC (chronic)	Freshwater CCC (chronic)	Freshwater CMC (acute)	Freshwater CCC (chronic)
Chloroform	µg/L	5.1	--	--	--	--	--
Chromium	µg/L	925	570	65	156	550	180
Cr(VI)	µg/L	992	16	11	10	15	10
Nitrate	µg/L	69,500	--	--	--	--	--

Sources:

40 CFR 131, "Water Quality Standards."

WAC 173-201A, "Water Quality Standards for Surface Waters of the State of Washington."

Table 6-40. Summary of 90th Percentile Current Groundwater Concentrations and Associated Cancer Risk and Noncancer Hazard Index for the 100-D Source Exposure Area

COPC	Units	90th Percentile Value	MTCA, "Groundwater Cleanup Standards (WAC 173-340-720) Cleanup Levels			
			Noncarcinogens	HQ	Carcinogens at 10^{-6} Risk Level	ELCR
Chloroform	µg/L	5.1	80	0.06	1.4	3.6×10^{-6}
Total ELCR					–	**3.6×10^{-6}**
Chromium	µg/L	925	24,000	0.04	--	--
Cr(VI)	µg/L	992	48	21	--	--
Nitrate	µg/L	69,500	113,600	0.61	--	--
Hazard Index				**21**		

Source: WAC 173-340-720, "Model Toxics Control Act—Cleanup," "Groundwater Cleanup Standards."

6.3.6.1 100-D Source Exposure Area

Groundwater in the 100-D Source exposure area is evaluated as a potential drinking water source and near-shore groundwater has the potential to discharge to the Columbia River. Table 6-38 provides a summary of the COPCs, the 90th percentile groundwater concentration, federal and state MCLs, and the MTCA ("Groundwater Cleanup Standards" [WAC 173-340-720]) for carcinogenic and noncarcinogenic effects. Table 6-39 provides a summary of the COPCs, the 90th percentile groundwater concentration, and

federal and state surface water quality standards. These standards (listed in Tables 6-38 and 6-39) represent the action levels that were exceeded by at least one COPC.

6.3.6.1.1 *Protectiveness Evaluation for Human Health*

The protectiveness evaluation for human health is performed to help determine whether a CERCLA remedial action is appropriate. *Rules of Thumb for Superfund Remedy Selection* (EPA 540-R-97-013) states that a remedial action is generally appropriate when a regulatory standard that helps define protectiveness (a federal or state MCL or nonzero MCLG for current or potential drinking water aquifers) is exceeded.

The 90th percentile groundwater concentration for chromium is greater than the federal and state MCL developed for the protection of human health. Chromium is identified as a COPC indicating the need to evaluate potential remedial technologies for chromium in the FS. Of the 20 monitoring wells in the 100-D Source exposure area, 11 monitoring wells were reported with concentrations of chromium above 100 μg/L. A detailed discussion of the chromium plume is also provided in Section 4.5.

A federal and state MCL is not available for Cr(VI); therefore, the protectiveness evaluation was not performed. Cr(VI) is discussed in the protectiveness evaluation for aquatic organisms and the risk evaluation.

The 90th percentile groundwater concentration for nitrate is greater than the federal and state MCL developed for the protection of human health. Nitrate is identified as a final COPC indicating the need to evaluate potential remedial technologies for nitrate in the FS. Of the 20 monitoring wells in the 100-D Source exposure area, nine monitoring wells were reported with concentrations of nitrate above 45,000 μg/L. A detailed discussion of the nitrate plume is provided in Section 4.5.

The 90th percentile groundwater concentration for chloroform is less than the federal and state MCL developed for the protection of human health. Chloroform is not an identified as a COPC indicating a need for further review in the FS is not established based on the results of this evaluation.

6.3.6.1.2 *Protectiveness Evaluation for Aquatic Receptors*

As described in the exposure assessment, groundwater discharges to the Columbia River through upwelling and seeps. The point of compliance for surface water cleanup levels is defined in the MTCA ("Surface Water Cleanup Standards" [WAC 173-340-730(7)(a)]) as the point or points at which hazardous substances are released to surface waters of the state. MTCA ("Surface Water Cleanup Standards" [WAC 173-340-730(7)(b)]) indicates that no mixing zone shall be allowed to demonstrate compliance with surface water cleanup levels. Groundwater EPCs from each exposure area within the 100-HR-3 Groundwater OU are compared to determine whether groundwater concentrations discharging to the Columbia River are in compliance with federal and state standards.

The 90th percentile groundwater concentration for chromium is greater than the federal freshwater AWQC value of 65 μg/L. Chromium is identified as a COPC, indicating the need to evaluate potential remedial technologies for chromium in the FS. Of the 20 monitoring wells, 13 monitoring wells were reported with concentrations of chromium above the freshwater AWQC value of 65 μg/L. It is assumed that a portion of the dissolved concentrations of total chromium are present in the form of Cr(VI) and total chromium is not presented separately from Cr(VI) in the nature and extent evaluation and the FS.

The 90th percentile groundwater concentration for Cr(VI) is greater than the "Water Quality Standards for Surface Waters of the State of Washington" (WAC 173-201A) freshwater AWQC value of 10 μg/L. Cr(VI) is identified as a COPC, indicating the need to evaluate potential remedial technologies for Cr(VI) in the FS. Of the 20 monitoring wells, 19 monitoring wells were reported with concentrations of chromium above the "Water Quality Standards for Surface Waters of the State of Washington" (WAC 173-201A) freshwater AWQC value of 10 μg/L. Cr(VI) concentrations above the AWQC were

also measured in 25 additional wells, based on the results from the larger population of wells and longer sampling timeframe. The following lists the approximate distance from the Columbia River for those wells reporting concentrations greater than 10 µg/L.

199-D2-11 (1,134 m [3,720 ft])	199-D5-13 (602 m [1,975 ft])	199-D5-93 (624 m [2,047 ft])
199-D2-6 (832 m [2,730 ft])	199-D5-132 (1,269 m [4,163 ft])	199-D5-97 (647 m [2,123 ft])
199-D2-8 (1,008 m [3,307 ft])	199-D5-14 (983 m [3,070 ft])	199-D5-98 (769 m [2,523 ft])
199-D3-2 (241 m [790 ft])	199-D5-140 (950 m [3,117 ft])	199-D5-99 (659 m [2,162 ft])
199-D4-14 (212 m [696 ft])	199-D5-142 (1,219 m [3,999 ft])	199-D8-101 (470 m [1,542 ft])
199-D4-15 (488 m [1,601 ft])	199-D5-143 (889 m [2,917 ft])	199-D8-4 (224 m [735 ft])
199-D4-19 (250 m [820 ft])	199-D5-15 (1,035 m [3,396 ft])	199-D8-5 (143 m [469 ft])
199-D4-20 (500 m [1,640 ft])	199-D5-16 (1,235 m [4,052 ft])	199-D8-54A (174 m [571 ft])
199-D4-22 (247 m [810 ft])	199-D5-17 (1,368 m [4,488 ft])	199-D8-55 (106 m [348 ft])
199-D4-23 (81 m [266 ft])	199-D5-18 (1,510 m [4,954 ft])	199-D8-6 (251 m [823 ft])
199-D4-84 (120 m [394 ft])	199-D5-34 (670 m [2,198 ft])	199-D8-69 (93 m [305 ft])
199-D5-102 (1,045 m [3,428 ft])	199-D5-37 (161 m [528 ft])	199-D8-70 (188 m [617 ft])
199-D5-103 (1,028 m [3,373 ft])	199-D5-38 (294 m [964 ft])	199-D8-71 (185 m [607 ft])
199-D5-104 (811 m [2,661 ft])	199-D5-40 (537 m [1,762 ft])	199-D8-73 (136 m [446 ft])
199-D5-122 (921 m [3,022 ft])	199-D5-43 (649 m [2,129 ft])	199-D8-88 (106 m [348 ft])

Although all monitoring wells within the plume area were compared to the AWQC concentration, only near-river wells would need to meet this criterion. Section 4.5.1 provides a detailed discussion of the Cr(VI) plume.

Federal and state water quality standards for the protection of freshwater organisms are not published for chloroform or nitrate; therefore, a protectiveness evaluation is not included. Chloroform and nitrate are evaluated in the protectiveness evaluation for human health in Section 6.3.6.1.1 and the risk evaluation is presented in Section 6.3.6.1.3.

6.3.6.1.3 *Risk Evaluation*

The potential cumulative ELCR for the 100-D Source exposure area from all nonradiological carcinogenic COPCs is 3.6×10^{-6}, which is less than the MTCA (HHRA Procedures [WAC 173-340-708]) risk threshold of 1×10^{-5} for multiple hazardous substances and less than the upper NCP threshold of 1×10^{-4}. Table 6-40 shows the only contributor to risk is chloroform (3.6×10^{-6}, 100 percent contribution). Chloroform is not identified as a COPC based on the results of this evaluation. As discussed previously, the nature and extent evaluation of groundwater presented in Section 4.5 also supports the conclusion of this analysis. Over the past 6 years, chloroform has been associated with a trend in 12 wells (199-D8-88, 199-D2-6, 199-D2-11, 199-D4-84, 199-D5-13, 199-D5-14, 199-D5-15, 199-D5-16, 199-D5-37, 199-D5-38, 199-D5-99, and 199-D8-5) where concentrations have ranged between 1.1 to 5.9 times greater than the action level of 1.4 µg/L. However, there have been no measured concentrations above the 10^{-5} level of 14 µg/L.

The HI for the 100-D Source exposure area is 21, which is greater than the EPA and MTCA (WAC 173-340) target HI of 1. The primary contributor to the noncancer HI is Cr(VI) (HQ=21,

97 percent contribution). The individual HQs for chloroform, chromium, and nitrate are each less than 1. The primary noncancer health effects associated with exposure to Cr(VI) is nasal septum atrophy. Cr(VI) is identified as a COPC based on the results of this evaluation. Chromium, nitrate, and zinc are not identified as COPCs based on the results of this evaluation.

6.3.6.2 100-H Source Exposure Area

Groundwater in the 100-H Source exposure area is evaluated as a potential drinking water source and near-shore groundwater has the potential to discharge to the Columbia River. Table 6-41 provides a summary of the COPCs, the 90th percentile groundwater concentration, federal and state MCLs, and the MTCA ("Groundwater Cleanup Standards" [WAC 173-340-720]) for carcinogenic and noncarcinogenic effects. Table 6-42 provides a summary of the COPCs, the 90th percentile groundwater concentration, and federal and state water quality standards. These standards (listed in Tables 6-41 and 6-42) represent the action levels that were exceeded by at least one COPC.

Table 6-41. Summary of Current Conditions 90th Percentile Groundwater Concentrations, Federal and State MCLs, and WAC 173-340-720 Groundwater Cleanup Levels for the 100-H Source Exposure Area (Human Health Action Levels)

COPCs	Units	90th Percentile Value	Federal MCL	State MCL	MTCA, "Groundwater Cleanup Standards" (WAC 173-340-720) Cleanup Levels		
					Noncarcinogens	Carcinogens at 10^{-6} Risk Level	Carcinogens at 10^{-5} Risk Level
Strontium-90	pCi/L	14	8	--	--	--	--
Cr(VI)	µg/L	26	--	--	48	--	--
Nitrate	µg/L	39,800	45,000	45,000	113,600	--	--

Source: WAC 173-340-720, "Model Toxics Control Act—Cleanup," "Groundwater Cleanup Standards."

Table 6-42. Summary of Current Conditions 90th Percentile Groundwater Concentrations, and Federal and State Water Quality Standards for the 100-H Source Area (Aquatic Action Levels)

COPCs	Units	90th Percentile Value	AWQC		WAC 173-201A	40 CFR 131 Water Quality Standards	
			Freshwater CMC (acute)	Freshwater CCC (chronic)	Freshwater CCC (chronic)	Freshwater CMC (acute)	Freshwater CCC (chronic)
Strontium-90	pCi/L	14	--	--	--	--	--
Cr(VI)	µg/L	26	16	11	10	15	10
Nitrate	µg/L	39,800	--	--	--	--	--

Sources:

40 CFR 131, "Water Quality Standards,"

WAC 173-201A, "Water Quality Standards for Surface Waters of the State of Washington."

6.3.6.2.1 *Protectiveness Evaluation for Human Health*

The 90th percentile groundwater concentration for strontium-90 is greater than the federal MCL developed for the protection of human health. As Table 6-43 shows, potential exposure to groundwater as a drinking water source would result in a dose greater than 4 mrem per year from strontium-90. Of the 13 wells, three monitoring wells (199-H4-11, 199-H4-13, and 199-H4-45) were reported with strontium-90 concentrations greater than the MCL of 8 pCi/L. Strontium-90 is identified as a COPC, indicating the need to evaluate potential remedial technologies for strontium-90 in the FS. A detailed discussion of the strontium-90 plume is provided in Section 4.5.

Table 6-43. Summary of 90th Percentile Current Groundwater Concentrations and Associated Sum of Fractions for the 100-H Source Exposure Area

Final COPC	Units	90th Percentile Value	Federal MCL	Individual Fraction
Strontium-90	pCi/L	14	8	1.8
Sum of Fractions				**1.8**
Cumulative Annual Dose (mrem)				**7.0**

Note:
MCL; derived single-nuclide MCL-equivalent activity concentration.

A federal and state MCL is not available for Cr(VI); therefore, the protectiveness evaluation was not performed. Cr(VI) is discussed in the protectiveness evaluation for aquatic organisms and the risk evaluation.

The 90th percentile groundwater concentration for nitrate is less than the federal and state MCL developed for the protection of human health. Nitrate has only been detected in Wells 199-H4-3, 199-H4-46, and 199-H6-1 at concentrations above the MCL, indicating its presence is localized downgradient of the following sources: 105-H reactor or the 1607-H1 septic system (199-H4-46), the solar evaporation basin (199-H4-3) and the 116-H-1 trench (199-H6-1). Although the 90th percentile groundwater concentration is less than the federal MCL, nitrate is identified as a COPC and it warrants further evaluation in the FS. A detailed discussion of the nitrate plume is provided in Section 4.5.

6.3.6.2.2 *Protectiveness Evaluation for Aquatic Receptors*

Federal and state water quality standards for the protection of freshwater organisms are not published for strontium-90 or nitrate; therefore, a protectiveness evaluation is not included Strontium-90 is evaluated in the protectiveness evaluation for human health in Section 6.3.6.2.1. Nitrate is evaluated in the protectiveness evaluation for human health in Section 6.3.6.2.1 and the risk evaluation is presented in Section 6.3.6.2.3.

The 90th percentile groundwater concentration for Cr(VI) is greater than the "Water Quality Standards for Surface Waters of the State of Washington" (WAC 173-201A) freshwater AWQC value of 10 µg/L. Of the 13 monitoring wells, 10 monitoring wells were reported with concentrations of Cr(VI) above the "Water Quality Standards for Surface Waters of the State of Washington" (WAC 173-201A) freshwater AWQC value of 10 µg/L. Cr(VI) is identified as a COPC, indicating the need to evaluate potential remedial technologies for Cr(VI) in the FS. Cr(VI) concentrations above the AWQC were also measured in 15 additional wells, based on the results from the larger population of wells and longer sampling timeframe. The following lists the approximate distance from the Columbia River for those wells reporting concentrations greater than 10 µg/L.

199-H1-7 (256 m [840 ft])
199-H3-2A (482 m [1,581 ft])
199-H3-2C (472 m [1,549 ft])
199-H3-3 (746 m [2,447 ft])
199-H3-4 (792 m [2,598 ft])
199-H3-5 (868 m [2,848 ft])
199-H4-10 (54 m [177 ft])
199-H4-11 (55 m [180 ft])
199-H4-13 (55 m [180 ft])
199-H4-14 (332 m [1,089 ft])
199-H4-18 (150 m [492 ft])
199-H4-3 (162 m [531 ft])
199-H4-4 (62 m [203 ft])
199-H4-45 (205 m [673 ft])
199-H4-46 (422 m [1,384 ft])
199-H4-48 (413 m [1,355 ft])
199-H4-49 (566 m [1,857 ft])
199-H4-5 (107 m [351 ft])
199-H4-6 (415 m [1,361 ft])
199-H4-65 (152 m [499 ft])
199-H4-84 (200 m [656 ft])
199-H4-9 (152 m [499 ft])
199-H5-1A (717 m [2,352 ft])
199-H6-1 (295 m [968 ft])
199-H6-2 (575 m [1,886 ft])

Although all monitoring wells within the plume area were compared to the AWQC concentration, only near-river wells would need to meet this criterion. A discussion of the Cr(VI) plume is presented in Section 4.5.

6.3.6.2.3 *Risk Evaluation*

No carcinogenic COPCs were identified in the 100-H Source exposure area.

Table 6-44 shows the HI for the 100-H Source exposure area is 0.89, which is less than the EPA and MTCA (WAC 173-340) target HI of 1. The individual HQs for Cr(VI) and nitrate are less than one. Cr(VI) and nitrate are not identified as COPCs based on the results of this evaluation.

Table 6-44. Summary of 90th Percentile Current Groundwater Concentrations and Associated Cancer Risk and Noncancer Hazard Index for the 100-H Source Exposure Area

COPC	Units	90th Percentile Value	MTCA, "Groundwater Cleanup Standards" (WAC 173-340-720) Cleanup Levels			
			Noncarcinogens	HQ	Carcinogens at 10^{-6} Risk Level	ELCR
Cr(VI)	µg/L	26	48	0.54	--	--
Nitrate	µg/L	39,800	113,600	0.35	--	--
Hazard Index				**0.89**		

Source: WAC 173-340-720, "Model Toxics Control Act—Cleanup," "Groundwater Cleanup Standards."

6.3.6.3 *Horn Exposure Area*

Groundwater in the horn exposure area is evaluated as a potential drinking water source and near-shore groundwater has the potential to discharge to the Columbia River. Table 6-45 provides a summary of the COPCs, the 90th percentile groundwater concentration, federal and state MCLs, national recommended water quality criteria (human health water + organism), and MTCA ("Groundwater Cleanup Standards" [WAC 173-340-720]) for carcinogenic and noncarcinogenic effects. Table 6-46 provides a summary of the COPCs, the 90th percentile groundwater concentration, and federal and state water quality standards. These standards (listed in Tables 6-45 and 6-46) represent the action levels that were exceeded by at least one COPC.

Table 6-45. Summary of Current Conditions 90th Percentile Groundwater Concentrations, and Federal and State MCLs, and WAC 173-340-720 Groundwater Cleanup Levels for the Horn Exposure Area

COPCs	Units	90th Percentile Value	Federal and State MCL	Human Health Water + Organism	MTCA, "Groundwater Cleanup Standards" (WAC 173-340-720) Cleanup Levels		
					Noncarcinogens	Carcinogens at 10^{-6} Risk Level	Carcinogens at 10^{-5} Risk Level
Chromium	µg/L	76	100	--	24,000	--	--
Cr(VI)	µg/L	71	--	--	48	--	--

Source: WAC 173-340-720, "Model Toxics Control Act—Cleanup," "Groundwater Cleanup Standards."

Table 6-46. Summary of Current Conditions 90th Percentile Groundwater Concentrations, and Federal and State Water Quality Standards for the Horn Exposure Area (Aquatic Action Levels)

COPCs	Units	90th Percentile Value	AWQC		WAC 173-201A	40 CFR 131 Water Quality Standards	
			Freshwater CMC (acute)	Freshwater CCC (chronic)	Freshwater CCC (chronic)	Freshwater CMC (acute)	Freshwater CCC (chronic)
Chromium	µg/L	76	570	65	156	550	180
Cr(VI)	µg/L	71	16	11	10	15	10

Sources:

40 CFR 131, "Water Quality Standards."

WAC 173-201A, "Water Quality Standards for Surface Waters of the State of Washington."

6.3.6.3.1 *Protectiveness Evaluation for Human Health*

The 90th percentile groundwater concentration for chromium is less than the federal and state MCL developed for the protection of human health. Chromium is not identified as a COPC and a need for further review in the FS is not established based on the results of this evaluation. A detailed discussion of the chromium plume is provided in Section 4.5.

A federal MCL is not available for Cr(VI); therefore, the protectiveness evaluation was not performed. Cr(IV) is discussed in the protectiveness evaluation for aquatic organisms presented in Section 6.3.6.3.1 and the risk evaluation presented in Section 6.3.6.3.3.

6.3.6.3.2 *Protectiveness Evaluation for Aquatic Receptors*

The 90th percentile groundwater concentration for Cr(VI) is greater than the "Water Quality Standards for Surface Waters of the State of Washington" (WAC 173-201A) freshwater AWQC value of 10 µg/L. Of the 19 monitoring wells, 16 monitoring wells were reported with concentrations of Cr(VI) above the "Water Quality Standards for Surface Waters of the State of Washington" (WAC 173-201A) freshwater AWQC value of 10 µg/L. Cr(IV) is identified as a COPC, indicating the need to evaluate potential remedial technologies for Cr(VI) in the FS. Cr(VI) concentrations above the AWQC were also measured in eight additional wells, based on the results from the larger population of wells and longer sampling timeframe. The following lists the approximate distance from the Columbia River for those wells reporting concentrations greater than 10 µg/L.

699-100-43B (22 m [72 ft])	699-95-51 (595 m [1,952 ft])	699-97-48B (1,217 m [3,993 ft])
699-101-45 (232 m [761 ft])	699-96-43 (1,171 m [3,842 ft])	699-97-51A (437 m [1,430 ft])
699-87-55 (1,384 m [4,541 ft])	699-96-49 (891 m [2,923 ft])	699-98-43 (461 m [1,512 ft])
699-93-48A (1,553 m [5,095 ft])	699-96-52B (54 m [177 ft])	699-98-46 (1,046 m [3,432 ft])
699-94-41 (1,170 m [3,839 ft])	699-97-41 (522 m [1,713 ft])	699-98-51 (230 m [754 ft])
699-94-43 (1,645 m [5,397 ft])	699-97-43 (796 m [2,611 ft])	699-99-41 (51 m [167 ft])
699-95-45 (1,469 m [4,819 ft])	699-97-43B (792 m [2,598 ft])	699-99-42B (59 m [194 ft])
699-95-48 (1,430 m [4,692 ft])	699-97-45 (1,228 m [4,029 ft])	699-99-44 (488 m [1,601 ft])

Although all monitoring wells within the plume area were compared to the AWQC concentration, only near-river wells would need to meet this criterion. Section 4.5.1 provides a detailed discussion of the distribution of Cr(VI) in the 100-HR-3 Groundwater OU.

6.3.6.3.3 *Risk Evaluation*

No carcinogenic COPCs were identified in the horn exposure area.

Table 6-47 shows the HI for the horn exposure area is 1.7, which is greater than the EPA and MTCA (WAC 173-340) target HI of 1. The primary contributor to the noncancer HI is Cr(VI) (HQ=1.5; 88 percent contribution). The individual HQ for chromium is less than 1. The primary noncancer health effects associated with exposure to Cr(VI) is nasal septum atrophy. Cr(VI) is identified as a COPC based on the results of this evaluation. Chromium is not identified as COPC based on the results of this evaluation.

Table 6-47. Summary of 90th Percentile Current Groundwater Concentrations and Associated Cancer Risk and Noncancer Hazard Index for the Horn Exposure Area

COPC	Units	90th Percentile Value	MTCA, "Groundwater Cleanup Standards" (WAC 173-340-720) Cleanup Levels			
			Non Carcinogens	HQ	Carcinogens at 10^{-6} Risk Level	ELCR
Chromium	µg/L	76	24,000	<0.01	--	--
Cr(VI)	µg/L	71	48	1.5	--	--
Hazard Index				**1.7**		

Source: WAC 173-340-720, "Model Toxics Control Act—Cleanup," "Groundwater Cleanup Standards."

6.3.7 Risk Characterization Results of the EPA Tap Water Scenario

This section summarizes the results for each of the exposure pathways associated with use of groundwater as a drinking water (tap water source). As described in Regional Screening Levels (EPA, 2009a), the EPA Tap Water scenario reflects a RME scenario. The EPA Tap Water scenario is consistent with a residential exposure scenario because it incorporates default residential exposure assumptions. The results of the *Tap Water Risk Assessment for the 100-HR-3 Groundwater Operable Unit* (ECF-100HR3-10-0478) (Appendix G). Potentially complete exposure routes for the EPA Tap Water scenario include exposure of adult and children residents to groundwater used as a drinking water source and include the following:

- Ingestion of drinking water
- Inhalation of volatiles when showering and other domestic purposes
- Dermal contact with skin while showering and using groundwater for other domestic purposes (such as washing dishes)

It should be noted that EPA considers external radiation to be a significant exposure route only for radionuclides in soil (risk assessment guide [EPA/540/1-89/002]). External radiation from radionuclides in water is considered insignificant because of its shielding effects. EPA does not publish radionuclide cancer slope factors to quantify cancer risk from external or dermal exposure to radioactive analytes in groundwater. Radionuclide cancer risk is therefore calculated in this evaluation only for ingestion and inhalation exposure routes.

6.3.7.1 Use of Groundwater as a Potential Tap Water Source

In order to provide a comprehensive evaluation of current risks associated with the 100-HR-3 Groundwater OU, potential exposure to groundwater as a tap water source is evaluated under this scenario. Potential routes of exposure to groundwater include ingestion, dermal contact, and inhalation of volatiles during household activities. Results from this analysis are used to provide baseline conditions for all analytes with available toxicity information. Table 6-48 provides a summary of the risk estimates by exposure route for each exposure area evaluated. Table 6-49 and Table 6-50 show the details of contribution to risk and hazard, respectively, by contaminant for the 100-D Source exposure area, Table 6-51 and Table 6-52 for the 100-H Source exposure area, and Table 6-53 and Table 6-54 for the horn exposure area. The results in Tables 6-49 through 6-54 provide overall summaries of the EPA Tap Water scenario analysis for all detected analytes identified in Section 6.3.2.4.

Table 6-48. Summary of Risk Estimates from Use of Groundwater as a Potential Drinking Water Source Using EPA Tap Water Equations

	100-D Source Exposure Area		100-H Source Exposure Area		Horn Exposure Area	
Exposure Route	**ELCR**	**HI**	**ELCR**	**HI**	**ELCR**	**HI**
			Nonradionuclide Analytes			
Ingestion	7.1×10^{-5}	10	7.7×10^{-5}	1.5	1.4×10^{-4}	2.0
Dermal	5.7×10^{-6}	3.8	9.1×10^{-7}	0.12	6.3×10^{-6}	0.30
Inhalation	8.7×10^{-7}	<0.01	2.4×10^{-7}	<0.01	5.0×10^{-7}	<0.01
Total	**7.7×10^{-5}**	**14**	**7.8×10^{-5}**	**1.6**	**1.4×10^{-4}**	**2.3**
			Radionuclide Analytes			
Ingestion	1.0×10^{-5}	--	1.9×10^{-5}	--	5.2×10^{-6}	--
Inhalation	1.3×10^{-6}	--	5.7×10^{-7}	--	6.4×10^{-7}	--
Total	**1.1×10^{-5}**	**--**	**2.0×10^{-5}**	**--**	**5.9×10^{-6}**	**--**
Total ELCR*	**8.8×10^{-5}**	**--**	**9.8×10^{-5}**	**--**	**1.5×10^{-4}**	**--**

* Total cumulative ELCR represents the sum of the total nonradionuclide ELCR and the total radionuclide ELCR.

-- = Indicates HI not applicable

Table 6-49. 100-D Source Exposure Area - Summary of Tap Water Scenario Cancer Risk Results for Nonradiological and Radiological Analytes in Groundwater

Analyte Name	90th Percentile Concentration in Groundwater (mg/L or pCi/L)	Volatile[a]	Risk (Ingestion)	Risk (Dermal)	Risk (Inhalation)	Total Risk	% Contribution
Acetone	0.0010	Yes	--	--	--	--	--
Aluminum	0.024	--	--	--	--(b)	--	--
Arsenic	0.0026	--	5.72E-05	3.08E-07	--(b)	5.75E-05	65
Barium	0.10	--	--	--	--(b)	--	--
Beryllium	0.00010	--	--	--	--(b)	--	--
Boron	0.067	--	--	--	--(b)	--	--
Bromodichloromethane	0.0010	Yes	9.22E-07	7.27E-08	1.84E-07	1.18E-06	1.33
Cadmium	0.00020	--	--	--	--(b)	--	--
Carbon tetrachloride	0.0010	Yes	1.93E-06	4.99E-07	7.45E-08	2.51E-06	2.8
Chloroform	0.0051	Yes	2.35E-06	2.07E-07	5.82E-07	3.14E-06	3.6
Chromium	0.92	--	--	--	--(b)	--	--
Cobalt	0.0013	--	--	--	--(b)	--	--
Copper	0.00073	--	--	--	--(b)	--	--
Fluoride	0.12	--	--	--	--(b)	--	--
Cr(VI)	0.99	--	--	--	--(b)	--	--
Iron	0.11	--	--	--	--(b)	--	--
Lithium	0.021	--	--	--	--(b)	--	--
Manganese	0.0040	--	--	--	--(b)	--	--
Methylene chloride	0.0010	Yes	1.12E-07	4.04E-09	2.33E-09	1.18E-07	0.13
Molybdenum	0.0044	--	--	--	--(b)	--	--

Table 6-49. 100-D Source Exposure Area - Summary of Tap Water Scenario Cancer Risk Results for Nonradiological and Radiological Analytes in Groundwater

Analyte Name	90th Percentile Concentration in Groundwater (mg/L or pCi/L)	Volatile[a]	Risk (Ingestion)	Risk (Dermal)	Risk (Inhalation)	Total Risk	% Contribution
Nickel	0.0095	--	--	--	--(b)	--	--
Nitrate	70	--	--	--	--(b)	--	--
Selenium	0.0044	--	--	--	--(b)	--	--
Silver	0.00020	--	--	--	--(b)	--	--
Strontium	0.63	--	--	--	--(b)	--	--
Tetrachloroethene	0.0010	Yes	8.03E-06	4.62E-06	2.93E-08	1.27E-05	14.3
Tin	0.039	--	--	--	--(b)	--	--
Uranium	0.0041	--	--	--	--(b)	--	--
Vanadium	0.026	--	--	--	--(b)	--	--
Zinc	0.034	--	--	--	--(b)	--	--
Strontium-90	0.67	--	7.03E-07	--	--(b)	7.03E-07	0.79
Technetium-99	16	--	8.32E-07	--	--(b)	8.32E-07	0.94
Tritium	8,800	Yes	8.43E-06	--	1.29E-06	9.73E-06	11
Total Cumulative ELCR			8.05E-05	5.71E-06	2.17E-06	8.84E-05	100

a. Volatile contaminants as defined by EPA, 2009a, "Regional Screening Levels for Chemical Contaminants at Superfund Sites," or as defined by EPA 540-R-97-036, *Health Effects Assessment Summary Tables: FY 1997 Update*, "April 16, 2001 Update: Radionuclide Toxicity," "Radionuclide Table: Radionuclide Carcinogenicity – Slope Factors."

b. Nonvolatile constituents are not considered in the inhalation exposure route.

-- = Indicates toxicity criteria not available to quantify contaminant's cancer risk via this exposure route.

Shading identifies analytes with a contribution greater than 1 percent to total cumulative risk.

Table 6-50. 100-D Source Exposure Area - Summary of Tap Water Scenario Noncancer Hazard Results for Nonradiological Analytes in Groundwater

Analyte Name	90th Percentile Concentration in Groundwater C_w (mg/L)	Volatile[a]	HQ (Ingestion)	HQ (Dermal)	HQ (Inhalation)	Total HQ	% Contribution
Acetone	0.0010	Yes	<0.01	--	<0.01	<0.01	0.00022
Aluminum	0.024	--	<0.01	<0.01	--(b)	<0.01	0.0047
Arsenic	0.0026	--	0.23	<0.01	--(b)	0.24	1.7
Barium	0.10	--	0.014	<0.01	--(b)	0.015	0.10
Beryllium	0.00010	--	<0.01	<0.01	--(b)	<0.01	0.017
Boron	0.067	--	<0.01	<0.01	--(b)	<0.01	0.065
Bromodichloromethane	0.0010	Yes	<0.01	<0.01	--	<0.01	0.010
Cadmium	0.00020	--	0.011	<0.01	--(b)	0.012	0.085
Carbon tetrachloride	0.0010	Yes	0.039	0.011	<0.01	0.050	0.35
Chloroform	0.0051	Yes	0.014	<0.01	<0.01	0.016	0.11
Chromium	0.92	--	0.017	<0.01	--(b)	0.024	0.17
Cobalt	0.0013	--	0.12	<0.01	--(b)	0.12	0.85
Copper	0.00073	--	<0.01	<0.01	--(b)	<0.01	0.0035
Fluoride	0.12	--	0.053	<0.01	--(b)	0.053	0.37
Cr(VI)	0.99	--	9.1	3.8	--(b)	13	90
Iron	0.11	--	<0.01	<0.01	--(b)	<0.01	0.029
Lithium	0.021	--	0.28	<0.01	--(b)	0.29	2.0
Manganese	0.0040	--	<0.01	<0.01	--(b)	<0.01	0.036
Methylene chloride	0.0010	Yes	<0.01	<0.01	<0.01	<0.01	0.003
Molybdenum	0.0044	--	0.024	<0.01	--(b)	0.024	0.17

Table 6-50. 100-D Source Exposure Area - Summary of Tap Water Scenario Noncancer Hazard Results for Nonradiological Analytes in Groundwater

Analyte Name	90th Percentile Concentration in Groundwater C_w (mg/L)	Volatile[a]	HQ (Ingestion)	HQ (Dermal)	HQ (Inhalation)	Total HQ	% Contribution
Nickel	0.0095	--	0.013	<0.01	--(b)	0.013	0.094
Nitrate	70	--	0.27	<0.01	--(b)	0.27	1.9
Selenium	0.0044	--	0.024	<0.01	--(b)	0.024	0.17
Silver	0.00020	--	<0.01	<0.01	--(b)	<0.01	0.0083
Strontium	0.63	--	0.029	<0.01	--(b)	0.029	0.20
Tetrachloroethene	0.0010	Yes	<0.01	<0.01	<0.01	<0.01	0.031
Tin	0.039	--	<0.01	<0.01	--(b)	<0.01	0.013
Uranium	0.0041	--	0.038	<0.01	--(b)	0.038	0.27
Vanadium	0.026	--	0.14	<0.01	--(b)	0.15	1.0
Zinc	0.034	--	<0.01	<0.01	--(b)	<0.01	0.022
Total HI			10.4	3.81	<0.01	14.2	100

a. Volatile contaminants as defined by EPA, 2009a, "Regional Screening Levels for Chemical Contaminants at Superfund Sites."

b. Nonvolatile constituents are not considered in the inhalation exposure route.

-- = Indicates toxicity criteria not available to quantify contaminant's hazard via this exposure route.

Shading identifies analytes with a contribution of greater than 1 percent to HI.

Table 6-51. 100-H Source Exposure Area - Summary of Tap Water Scenario Cancer Risk Results for Nonradiological and Radiological Analytes in Groundwater

Analyte Name	90th Percentile Concentration in Groundwater (mg/L or pCi/L)	Volatile[a]	Risk (Ingestion)	Risk (Dermal)	Risk (Inhalation)	Total Risk	% Contribution
Aluminum	0.041	--	--	--	--(b)	--	--
Antimony	0.00061	--	--	--	--(b)	--	--
Arsenic	0.0033	--	7.41E-05	3.99E-07	--(b)	7.45E-05	76
Barium	0.067	--	--	--	--(b)	--	--
Boron	0.037	--	--	--	--(b)	--	--
Bromoform	0.0010	--	1.17E-07	7.89E-09	--(b)	1.25E-07	0.13
Bromomethane	0.0010	Yes	--	--	--	--	--
Carbon disulfide	0.0010	Yes	--	--	--	--	--
Carbon tetrachloride	0.0010	Yes	1.93E-06	4.99E-07	7.45E-08	2.51E-06	2.6
Chloroform	0.0014	Yes	6.45E-07	--	1.60E-07	8.05E-07	0.8
Chloromethane	0.0010	Yes	--	--	--	--	--
Chromium	0.031	--	--	--	--(b)	--	--
Cobalt	0.00043	--	--	--	--(b)	--	--
Copper	0.0013	--	--	--	--(b)	--	--
Fluoride	0.11	--	--	--	--(b)	--	--
Cr(VI)	0.026	--	--	--	--(b)	--	--
Iron	0.44	--	--	--	--(b)	--	--
Lithium	0.014	--	--	--	--(b)	--	--
Manganese	0.035	--	--	--	--(b)	--	--
Mercury	0.00010	--	--	--	--(b)	--	--

Table 6-51. 100-H Source Exposure Area - Summary of Tap Water Scenario Cancer Risk Results for Nonradiological and Radiological Analytes in Groundwater

Analyte Name	90th Percentile Concentration in Groundwater (mg/L or pCi/L)	Volatile[a]	Risk (Ingestion)	Risk (Dermal)	Risk (Inhalation)	Total Risk	% Contribution
Methylene chloride	0.0010	Yes	1.12E-07	4.04E-09	2.33E-09	1.18E-07	0.12
Molybdenum	0.0054	--	--	--	--(b)	--	--
Nickel	0.0089	--	--	--	--(b)	--	--
Nitrate	40	--	--	--	--(b)	--	--
Nitrite	1.6	--	--	--	--(b)	--	--
Selenium	0.0027	--	--	--	--(b)	--	--
Strontium	0.39	--	--	--	--(b)	--	--
Tin	0.039	--	--	--	--(b)	--	--
Toluene	0.0010	Yes	--	--	--	--	--
Uranium	0.0061	--	--	--	--(b)	--	--
Vanadium	0.012	--	--	--	--(b)	--	--
Zinc	0.016	--	--	--	--(b)	--	--
Strontium-90	14	--	1.48E-05	--	--(b)	1.48E-05	15
Technetium-99	8.8	--	4.57E-07	--	--(b)	4.57E-07	0.47
Tritium	3,900	Yes	3.74E-06	--	5.73E-07	4.31E-06	4.4
Total Cumulative ELCR			9.59E-05	9.10E-07	8.10E-07	9.76E-05	100

a. Volatile contaminants as defined by EPA, 2009a, "Regional Screening Levels for Chemical Contaminants at Superfund Sites," or as defined by EPA 540-R-97-036, *Health Effects Assessment Summary Tables: FY 1997 Update*, "April 16, 2001 Update: Radionuclide Toxicity," "Radionuclide Table: Radionuclide Carcinogenicity – Slope Factors."

b. Nonvolatile constituents are not considered in the inhalation exposure route.

-- = Indicates toxicity criteria not available to quantify contaminant's cancer risk via this exposure route.

Shading identifies analytes with a contribution greater than 1 percent to total cumulative risk.

Table 6-52. 100-H Source Exposure Area - Summary of Tap Water Scenario Noncancer Hazard Results for Nonradiological Analytes in Groundwater

Analyte Name	90th Percentile Concentration in Groundwater C_w (mg/L)	Volatile[a]	HQ (Ingestion)	HQ (Dermal)	HQ (Inhalation)	Total HQ	% Contribution
Aluminum	0.041	--	<0.01	<0.01	--(b)	<0.01	0.070
Antimony	0.00061	--	0.042	<0.01	--(b)	0.043	2.6
Arsenic	0.0033	--	0.30	<0.01	--(b)	0.31	19
Barium	0.067	--	<0.01	<0.01	--(b)	<0.01	0.60
Boron	0.037	--	<0.01	<0.01	--(b)	<0.01	0.31
Bromoform	0.0010	--	<0.01	<0.01	--(b)	<0.01	0.090
Bromomethane	0.0010	Yes	0.020	<0.01	<0.01	0.022	1.4
Carbon disulfide	0.0010	Yes	<0.01	<0.01	<0.01	<0.01	0.021
Carbon tetrachloride	0.0010	Yes	0.039	0.011	<0.01	0.050	3.0
Chloroform	0.0014	Yes	<0.01	<0.01	<0.01	<0.01	0.27
Chloromethane	0.0010	Yes	--	--	<0.01	<0.01	0.008
Chromium	0.031	--	<0.01	<0.01	--(b)	<0.01	0.048
Cobalt	0.00043	--	0.039	<0.01	--(b)	0.039	2.4
Copper	0.0013	--	<0.01	<0.01	--(b)	<0.01	0.053
Fluoride	0.11	--	0.048	<0.01	--(b)	0.049	3.0
Cr(VI)	0.026	--	0.23	0.097	--(b)	0.33	20
Iron	0.44	--	0.017	<0.01	--(b)	0.018	1.1
Lithium	0.014	--	0.20	<0.01	--(b)	0.20	12
Manganese	0.035	--	0.040	<0.01	--(b)	0.046	2.8
Mercury	1.00E-04	--	<0.01	<0.01	--(b)	<0.01	0.60
Methylene chloride	0.0010	Yes	<0.01	<0.01	<0.01	<0.01	0.030
Molybdenum	0.0054	--	0.029	<0.01	--(b)	0.030	1.8
Nickel	0.0089	--	0.012	<0.01	--(b)	0.013	0.77
Nitrate	40	--	0.15	<0.01	--(b)	0.16	9.5
Nitrite	1.6	--	0.14	<0.01	--(b)	0.14	8.8

Table 6-52. 100-H Source Exposure Area - Summary of Tap Water Scenario Noncancer Hazard Results for Nonradiological Analytes in Groundwater

Analyte Name	90th Percentile Concentration in Groundwater C_w (mg/L)	Volatile[a]	HQ (Ingestion)	HQ (Dermal)	HQ (Inhalation)	Total HQ	% Contribution
Selenium	0.0027	--	0.015	<0.01	--(b)	0.015	0.91
Strontium	0.39	--	0.018	<0.01	--(b)	0.018	1.1
Tin	0.039	--	<0.01	<0.01	--(b)	<0.01	0.11
Toluene	0.0010	Yes	<0.01	<0.01	<0.01	<0.01	0.028
Uranium	0.0061	--	0.055	<0.01	--(b)	0.056	3.4
Vanadium	0.012	--	0.067	<0.01	--(b)	0.067	4.1
Zinc	0.016	--	<0.01	<0.01	--(b)	<0.01	0.090
Total HI			1.51	0.12	<0.01	1.63	100

a. Volatile contaminants as defined by EPA, 2009a, "Regional Screening Levels for Chemical Contaminants at Superfund Sites."
b. Nonvolatile constituents are not considered in the inhalation exposure route.
-- = Indicates toxicity criteria not available to quantify contaminant's hazard via this exposure route.
Shading identifies analytes with a contribution of greater than 1 percent to HI.

Table 6-53. Horn Exposure Area - Summary of Tap Water Scenario Cancer Risk Results for Nonradiological and Radiological Analytes in Groundwater

Analyte Name	90th Percentile Concentration in Groundwater (mg/L or pCi/L)	Volatile[a]	Risk (Ingestion)	Risk (Dermal)	Risk (Inhalation)	Total Risk	% Contribution
1,2-Dichloroethane	0.0010	Yes	1.35E-06	6.41E-08	1.29E-07	1.55E-06	1.0
2-Butanone	0.0010	Yes	--	--	--	--	--
Acetone	0.0010	Yes	--	--	--	--	--
Aluminum	0.054	--	--	--	--(b)	--	--
Antimony	0.00074	--	--	--	--(b)	--	--
Arsenic	0.0055	--	1.23E-04	6.61E-07	--(b)	1.23E-04	83

Table 6-53. Horn Exposure Area - Summary of Tap Water Scenario Cancer Risk Results for Nonradiological and Radiological Analytes in Groundwater

Analyte Name	90th Percentile Concentration in Groundwater (mg/L or pCi/L)	Volatile[a]	Risk (Ingestion)	Risk (Dermal)	Risk (Inhalation)	Total Risk	% Contribution
Barium	0.067	--	--	--	--(b)	--	--
Boron	0.024	--	--	--	--(b)	--	--
Carbon tetrachloride	0.0013	Yes	2.51E-06	6.49E-07	9.68E-08	3.26E-06	2.2
Chloroform	0.0010	Yes	4.61E-07	4.05E-08	1.14E-07	6.16E-07	0.41
Chromium	0.076	--	--	--	--(b)	--	--
Cobalt	0.00010	--	--	--	--(b)	--	--
Copper	0.0014	--	--	--	--(b)	--	--
Fluoride	0.26	--	--	--	--(b)	--	--
Cr(VI)	0.071	--	--	--	--(b)	--	--
Iron	0.42	--	--	--	--(b)	--	--
Lithium	0.012	--	--	--	--(b)	--	--
Manganese	0.011	--	--	--	--(b)	--	--
Methylene chloride	0.0010	Yes	1.12E-07	4.04E-09	2.33E-09	1.18E-07	0.079
Molybdenum	0.0085	--	--	--	--(b)	--	--
Nickel	0.0049	--	--	--	--(b)	--	--
Nitrate	30	--	--	--	--(b)	--	--
Nitrite	0.12	--	--	--	--(b)	--	--
Selenium	0.0032	--	--	--	--(b)	--	--
Silver	0.00020	--	--	--	--(b)	--	--
Strontium	0.36	--	--	--	--(b)	--	--

Table 6-53. Horn Exposure Area - Summary of Tap Water Scenario Cancer Risk Results for Nonradiological and Radiological Analytes in Groundwater

Analyte Name	90th Percentile Concentration in Groundwater (mg/L or pCi/L)	Volatile[a]	Risk (Ingestion)	Risk (Dermal)	Risk (Inhalation)	Total Risk	% Contribution
Tetrachloroethene	0.0010	Yes	8.03E-06	4.62E-06	2.93E-08	1.27E-05	8.5
Tin	0.039	--	--	--	--(b)	--	--
Toluene	0.0010	Yes	--	--	--	--	--
Trichloroethene	0.0010	Yes	1.32E-06	2.21E-07	1.24E-07	1.67E-06	1.1
Uranium	0.0042	--	--	--	--(b)	--	--
Vanadium	0.022	--	--	--	--(b)	--	--
Xylenes (total)	0.0010	Yes	--	--	--	--	--
Zinc	0.012	--	--	--	--(b)	--	--
Strontium-90	0.90	--	9.51E-07	--	--(b)	9.51E-07	0.64
Technetium-99	1.9	--	9.88E-08	--	--(b)	9.88E-08	0.066
Tritium	4,350	Yes	4.17E-06	--	6.39E-07	4.81E-06	3.2
Total Cumulative ELCR			1.42E-04	6.26E-06	1.14E-06	1.49E-04	100

Note:

Shading identifies analytes with a contribution greater than 1 percent to total cumulative risk.

a. Volatile contaminants as defined by EPA, 2009a, "Regional Screening Levels for Chemical Contaminants at Superfund Sites," or as defined by EPA 540-R-97-036, *Health Effects Assessment Summary Tables: FY 1997 Update*, "April 16, 2001 Update: Radionuclide Toxicity," "Radionuclide Table: Radionuclide Carcinogenicity – Slope Factors."

b. Nonvolatile constituents are not considered in the inhalation exposure route.

-- = Indicates toxicity criteria not available to quantify contaminant's cancer risk via this exposure route.

Table 6-54. Horn Exposure Area - Summary of Tap Water Scenario Noncancer Hazard Results for Nonradiological Analytes in Groundwater

Analyte Name	90th Percentile Concentration in Groundwater C_w (mg/L)	Volatile[a]	HQ (Ingestion)	HQ (Dermal)	HQ (Inhalation)	Total HQ	% Contribution
1,2-Dichloroethane	0.0010	Yes	<0.01	<0.01	<0.01	<0.01	0.063
2-Butanone	0.0010	Yes	<0.01	<0.01	<0.01	<0.01	0.0021
Acetone	0.0010	Yes	<0.01	--	<0.01	<0.01	0.0014
Aluminum	0.054	--	<0.01	<0.01	--(b)	<0.01	0.066
Antimony	0.00074	--	0.050	<0.01	--(b)	0.052	2.3
Arsenic	0.0055	--	0.50	<0.01	--(b)	0.50	22
Barium	0.067	--	<0.01	<0.01	--(b)	<0.01	0.43
Boron	0.024	--	<0.01	<0.01	--(b)	<0.01	0.14
Carbon tetrachloride	0.0013	Yes	0.051	0.014	<0.01	0.065	2.8
Chloroform	0.0010	Yes	<0.01	<0.01	<0.01	<0.01	0.14
Chromium	0.076	--	<0.01	<0.01	--(b)	<0.01	0.086
Cobalt	0.00010	--	<0.01	<0.01	--(b)	<0.01	0.40
Copper	0.0014	--	<0.01	<0.01	--(b)	<0.01	0.042
Fluoride	0.26	--	0.12	<0.01	--(b)	0.12	5.3
Cr(VI)	0.071	--	0.65	0.27	--(b)	0.92	40
Iron	0.42	--	0.017	<0.01	--(b)	0.017	0.73
Lithium	0.012	--	0.17	<0.01	--(b)	0.17	7.4
Manganese	0.011	--	0.013	<0.01	--(b)	0.015	0.64
Methylene chloride	0.0010	Yes	<0.01	<0.01	<0.01	<0.01	0.02
Molybdenum	0.0085	--	0.046	<0.01	--(b)	0.047	2.0

Table 6-54. Horn Exposure Area - Summary of Tap Water Scenario Noncancer Hazard Results for Nonradiological Analytes in Groundwater

Analyte Name	90th Percentile Concentration in Groundwater C_w (mg/L)	Volatile[a]	HQ (Ingestion)	HQ (Dermal)	HQ (Inhalation)	Total HQ	% Contribution
Nickel	0.0049	--	<0.01	<0.01	--(b)	<0.01	0.30
Nitrate	30	--	0.11	<0.01	--(b)	0.12	5.0
Nitrite	0.12	--	0.011	<0.01	--(b)	0.011	0.48
Selenium	0.0032	--	0.018	<0.01	--(b)	0.018	0.78
Silver	0.00020	--	<0.01	<0.01	--(b)	<0.01	0.052
Strontium	0.36	--	0.016	<0.01	--(b)	0.017	0.72
Tetrachloroethene	0.0010	Yes	<0.01	<0.01	<0.01	<0.01	0.19
Tin	0.039	--	<0.01	<0.01	--(b)	<0.01	0.078
Toluene	0.0010	Yes	<0.01	<0.01	<0.01	<0.01	0.020
Trichloroethene	0.0010	Yes	--	--	<0.01	<0.01	0.015
Uranium	0.0042	--	0.039	<0.01	--(b)	0.039	1.7
Vanadium	0.022	--	0.12	<0.01	--(b)	0.12	5.4
Xylenes (total)	0.0010	Yes	<0.01	<0.01	<0.01	<0.01	0.015
Zinc	0.012	--	<0.01	<0.01	--(b)	<0.01	0.048
Total HI			1.98	0.30	<0.01	2.28	100

Note:

Shading identifies analytes with a contribution of greater than 1 percent to HI.

a. Volatile contaminants as defined by EPA, 2009a, "Regional Screening Levels for Chemical Contaminants at Superfund Sites."

b. Nonvolatile constituents are not considered in the inhalation exposure route.

-- = Indicates toxicity criteria not available to quantify contaminant's hazard via this exposure route.

100-D Source Exposure Area. The total ELCR is 1.1×10^{-5} for nonradiological analytes and 1.2×10^{-4} for radiological analytes. The radiological ELCR and nonradiological ELCRs are within the EPA range of 1×10^{-4} to 1×10^{-6}.

As shown in Table 6-49, the major contributors to the total cumulative ELCR (those analytes that contribute greater than 1 percent of total cumulative ELCR) are tetrachloroethene (1.3×10^{-5}; 14 percent contribution), tritium (9.7×10^{-6}; 11 percent contribution), chloroform (3.1×10^{-6}; 3.6 percent contribution), carbon tetrachloride (2.5×10^{-6}; 2.8 percent contribution), and bromodichloromethane (1.2×10^{-6}; 1.3 percent contribution). Contribution to ELCR is elevated for arsenic (5.8×10^{-5}; 65 percent contribution) where measured concentrations are within natural background values.

As shown in Table 6-50, the HI is 14, which is greater than the EPA target HI of 1. The primary contributor to the noncancer HI (those analytes that contribute greater than 1 percent of total HI) is Cr(VI) (HQ of 13; 90 percent contribution). All remaining individual analytes (arsenic, lithium, nitrate, and vanadium) that contribute greater than one percent of the HI also report a HQ less than 1.

100-H Source Exposure Area. The total ELCR is 7.8×10^{-5} for nonradiological analytes and 2.0×10^{-5} for radiological analytes. Both total ELCRs are within the EPA range of 1×10^{-4} to 1×10^{-6}.

As shown in Table 6-51, the major contributors to the total cumulative ELCR (those analytes that contribute greater than 1 percent of total cumulative ELCR) are strontium-90 (1.5×10^{-5}; 15 percent contribution), tritium (4.3×10^{-6}; 4.4 percent contribution), and carbon tetrachloride (2.5×10^{-6}; 2.6 percent contribution). Contribution to ELCR is elevated for arsenic (7.4×10^{-5}; 76 percent contribution) where measured concentrations are within natural background values.

As shown in Table 6-52, the HI is 1.6, which is greater than the EPA target HI of 1.0. All individual analytes (antimony, arsenic, bromomethane, carbon tetrachloride, cobalt, fluoride, Cr(VI), iron, lithium, manganese, molybdenum, nitrate, nitrite, strontium, uranium, and vanadium) that contribute greater than one percent of the HI also report a HQ less than 1.

Horn Exposure Area. The total ELCR is 1.4×10^{-4} for nonradiological analytes and 5.9×10^{-6} for radiological analytes. The nonradiological ELCR is greater than the EPA upper target risk threshold of 1×10^{-4} and the radiological ELCR is within the EPA range of 1×10^{-4} to 1×10^{-6}.

As shown in Table 6-53, the major contributors to the total cumulative ELCR (those analytes that contribute greater than 1 percent of total cumulative ELCR) are tetrachloroethene (1.3×10^{-5}; 8.5 percent contribution), tritium (4.8×10^{-6}; 3.2 percent contribution), carbon tetrachloride (3.3×10^{-6}; 2.2 percent contribution), trichloroethene (1.7×10^{-6}; 1.1 percent contribution), and 1,2-dichloroethane (1.6×10^{-6}; 1.0 percent contribution). Contribution to ELCR is elevated for arsenic (1.2×10^{-4}; 83 percent contribution) where measured concentrations are within natural background values.

As shown in Table 6-54, the HI is 2.3, which is greater than the EPA target HI of 1.0. All individual analytes (antimony, arsenic, carbon tetrachloride, fluoride, Cr(VI), iron, lithium, molybdenum, nitrate, uranium, and vanadium) that contribute greater than one percent of the HI also report a HQ less than 1.

6.3.8 Uncertainties in Groundwater Risk Assessment

The purpose of this groundwater risk assessment is to determine whether a groundwater remedial action is warranted under CERCLA. Estimating and evaluating health risk from exposure to environmental contaminants is a complex process with inherent uncertainties. Uncertainty reflects limitations in knowledge, and simplifying assumptions must be made to quantify health risks.

In this assessment, uncertainties relate to the selection of COPCs and the development of media concentrations to which receptors may be exposed, the assumptions about exposure and toxicity, and the characterization of health risks. Uncertainties exist regarding the quantification of health risks in terms of several assumptions about exposure and toxicity, including Hanford Site-specific and general uncertainties. Based on the anticipation of uncertainty when quantifying exposure and toxicity, the health risks and hazards presented in this groundwater risk assessment are more likely to provide an upper boundary on risk.

6.3.8.1 Uncertainties Associated with Sampling and Analysis Data

Sampling and analysis data used in this groundwater risk assessment were collected specifically to address the uncertainties identified in the RCBRA (DOE/RL-2007-21) and the 100-D/H Work Plan (DOE/RL-2008-46-ADD1). These uncertainties were generally associated with the chemical, spatial, and temporal representativeness of the dataset used to evaluate current baseline conditions in the RCBRA. Uncertainties with chemical representativeness were related to the analysis of varying analytical methods between monitoring wells within the OU. Uncertainties with spatial and temporal representativeness were associated with varying sampling frequencies between monitoring wells as a result of differing monitoring programs.

Current baseline conditions are presented by groundwater data collected over an 8-month period between October 7, 2009 and June 11, 2010. Three sampling events were used to capture the effects that temporal fluctuations of river stage have on groundwater conditions. The COPCs identified during the work plan phase were validated by using groundwater samples analyzed for the analytical methods documented in the 100-D/H SAP (DOE/RL-2009-40). The groundwater dataset used for COPC identification consists of sampling and analysis data collected from 52 monitoring wells within the 100-HR-3 Groundwater OU. The monitoring well network represents locations where human or ecological receptors could potentially encounter groundwater within the OU. The primary exposure pathway for humans is through groundwater obtained from a residential or community water well, assuming development of the land for future human habitation. The primary exposure pathways for aquatic organisms are direct discharge of groundwater to the Columbia River or through seeps.

All samples were analyzed using methods that could accurately measure analytes to concentrations equal to or less than the lowest action level. When analytical methods could not achieve the lowest action level, the action level defaulted to the MDL that could reasonably be achieved. These detection limits are documented in Table 2-19 of the 100-D/H SAP (DOE/RL-2009-40).

Technetium-99 was reported at a concentration of 2,100 pCi/L in sample number B257L3 at well 199-D5-18 (collected on May 12, 2010). This result is flagged with a "Y" review qualifier and a subsequent review of the results indicates that samples were misidentified. The review indicates that an aliquot (either B24949 or B24952) from well 299-E27-24 was inadvertently substituted for B257L3 during sample preparation and analysis at the laboratory. Additionally, two previous sampling rounds (March 30, 2010 and October 21, 2009) reported nondetected concentrations and gross beta measurements at this well from all three results consistently report concentrations between 7.5 and 9.5 pCi/L. Technetium-99 results from 199-D5-18 reported in 1992 were also reported as nondetected concentrations. The technetium-99 result for sample B257L3 was not included in the data set used to calculate exposure point concentrations.

6.3.8.2 Uncertainties Associated with Exposure Point Concentrations

The protectiveness and groundwater risk assessment methodology uses an RME concentration for each COPC for the entire OU rather than performing the evaluation on a specific well or location. In general, EPA Superfund guidance recommends using a 95 percent UCL on the arithmetic mean for estimating

EPCs that reflect a RME. However, experience indicates that averages and UCLs cannot be reliably calculated for Hanford Site groundwater datasets.

Groundwater datasets at the Hanford Site are highly skewed, with a large proportion of BDL values. *Data Quality Assessment: Statistical Methods for Practitioners* (EPA/240/B-06/003) provides guidance for estimating parameters (whether means or upper percentiles) depending on the variability in the dataset, as expressed as the CV and the proportion of observation that are BDL. For datasets with CVs greater than 1 and 50 percent or more observations that are BDL, *Data Quality Assessment: Statistical Methods for Practitioners* (EPA/240/B-06/003) recommends using upper percentiles as opposed to means to develop summary statistics.

EPA's ProUCL software is used to estimate EPCs and statistics for comparison with standards and background levels, in accordance with EPA Superfund risk assessment guidance. ProUCL contains computational methods for parametric and nonparametric UCL, upper prediction limits (UPLs) and upper tolerance limits for use with datasets without non-detects as well as datasets with BDL observations. These computational methods can address skewed datasets with and without BDL observations. However, in practice, ProUCL will provide warning flags for 95 percent UCLs from datasets that are both highly skewed and that contain a large proportion (50 percent or greater) BDL observations.

Use of the 90th percentile value from a distribution of groundwater concentration data as an estimate of the EPC is an alternative approach for estimating EPCs in cases where ProUCL does not provide reliable UCL values. However, use of the 90th percentile exposure concentration to develop an EPC is consistent with other EPA risk assessment guidance for describing and characterizing health risks. *Guidance for Risk Characterization* (EPA, 1995) states that risk assessments should provide an evaluation of risks at the high end of the distribution of exposure. Conceptually, the high end of the distribution means above the 90th percentile of the population distribution, but not higher than the individual in the population with the highest exposure (*Guidelines for Exposure Assessment* [EPA/600/Z-92/001]), which is comparable to the definition of RME as defined in the risk assessment guide (EPA/540/1-89/002). Therefore, use of the 90th percentile as the basis for a groundwater EPC yields risk estimates that correspond to an RME.

To illustrate the problem with using the 95% UCL for the groundwater data sets described in this report, Table 6-55 presents a few statistics for each contaminant, including the frequency of detection, 90th percentile, mean, and 95% UCL. The mean in Table 6-55 is calculated (by the ProUCL software) using only the detected concentration values; nondetect results are not used. For the 95 percent UCL recommended by ProUCL for censored datasets (i.e., some concentrations were below the detection limit), the nondetect results were used (by the same software, ProUCL) in the calculation of the 95 percent UCL using a Kaplan-Meier statistical method (a nonparametric method). For highly skewed and/or highly censored datasets (i.e., those when the frequency of detection is low), these differing statistical approaches with respect to the dataset can lead to large differences between the two calculated values. This is especially true when the frequency of nondetects exceeds 40 percent. For example, the calculated mean concentration value for aluminum in the 100-D Source Exposure Area is 19 μg/L. This mean value is based on only the 19 detected values. When ProUCL used its algorithms to calculate the 95 percent UCL, the recommended calculated value was 13 μg/L, which is smaller than the mean value calculated by the software. This is due to the consideration of the 41 nondetect values in calculating the 95 percent UCL, for which the method detection limit is used as the observed concentration for these measurements. This situation occurs for many of the contaminants in the 100-HR-3 Groundwater OU. A similar situation exists for the other exposure areas in Table 6-55.

Table 6-55. Percentile Concentrations and Summary Statistics for the 100-HR-3 Groundwater OU Dataset

COPC	Units	Number of Detections	Number of Nondetects	Frequency of Nondetects	Coefficient of Variation	90th Percentile of RI Data	Max	Mean	95% UCL	Is 95% UCL > 90th Percentile?	Action Level*
100-D Source Exposure Area											
Acetone	µg/L	1	59	98%	N/A	1	0.82	0.82	Not Calculated	N/A	7,200
Aluminum	µg/L	19	41	68%	0.55	24	42	19	13	No	50
Arsenic	µg/L	56	4	7%	0.29	2.6	2.9	1.8	1.9	No	0.018 (4.0)
Barium	µg/L	60	0	0%	0.28	100	133	73	78	No	1,000
Beryllium	µg/L	5	55	92%	0.46	0.1	0.31	0.17	0.17	Yes	4.0
Boron	µg/L	19	28	60%	0.50	67	102	49	38	No	3,200
Bromodichloromethane	µg/L	2	58	97%	0.011	1	0.68	0.68	0.68	No	0.27 (5.0)
Cadmium	µg/L	2	58	97%	0.46	0.2	0.22	0.17	0.12	No	0.25
Carbon tetrachloride	µg/L	2	58	97%	0.027	1	2.7	2.7	2.7	Yes	0.23 (1.0)
Chloride	µg/L	60	0	0%	0.41	30,000	37,300	18,800	20,473	No	230,000
Chloroform	µg/L	50	10	17%	0.99	5.1	8.3	2.2	3.1	No	1.4 (5.0)
Chromium	µg/L	60	0	0%	2.0	925	4,460	412	905	No	65
Cobalt	µg/L	23	37	62%	1.4	1.3	3	0.82	0.55	No	2.6 (4.0)
Copper	µg/L	38	22	37%	0.78	0.73	2.3	0.48	0.47	No	9.0
Fluoride	µg/L	13	47	78%	0.35	115	148	86	70	No	960
Gross alpha	pCi/L	7	53	88%	0.38	3.5	6.8	4.3	4.1	Yes	15
Gross beta	pCi/L	44	16	27%	0.65	14	35	8.9	8.8	No	--
Cr(VI)	µg/L	60	0	0%	2.3	992	6,390	494	1,534	Yes	10
Iron	µg/L	39	21	35%	0.84	106	265	70	66	No	300
Lead	µg/L	14	46	77%	0.30	0.35	0.52	0.34	0.26	No	2.1
Lithium	µg/L	42	5	11%	1.4	21	133	15	26	Yes	32
Manganese	µg/L	3	57	95%	1.2	4	47	19	47	Yes	50

Table 6-55. Percentile Concentrations and Summary Statistics for the 100-HR-3 Groundwater OU Dataset

COPC	Units	Number of Detections	Number of Nondetects	Frequency of Nondetects	Coefficient of Variation	90th Percentile of RI Data	Max	Mean	95% UCL	Is 95% UCL > 90th Percentile?	Action Level*
Methylene chloride	µg/L	6	54	90%	0.18	1	0.27	0.22	0.22	No	4.6
Molybdenum	µg/L	43	4	9%	0.78	4.4	8.7	2.1	3.1	No	80
Nickel	µg/L	13	47	78%	0.91	9.5	39	14	7.8	No	52
Nitrate	µg/L	60	0	0%	0.44	69,500	99,200	45,011	49,284	No	45,000
Selenium	µg/L	56	4	7%	0.49	4.4	6.5	2.8	3	No	5.0
Silver	µg/L	4	56	93%	0.32	0.2	0.26	0.19	0.23	Yes	2.6
Strontium	µg/L	60	0	0%	0.36	629	938	452	488	No	9,600
Strontium-90	pCi/L	3	57	95%	0.27	0.67	3.7	2.8	3.7	Yes	8.0
Sulfate	µg/L	60	0	0%	0.66	161,500	438,000	118,847	162,675	Yes	250,000
Technetium-99	pCi/L	2	57	97%	0.20	16	16	14	13	No	900
Tetrachloroethene	µg/L	1	59	98%	N/A	1	0.14	0.14	Not Calculated	N/A	0.08 (5.0)
Thallium	µg/L	4	56	93%	1.1	0.1	1	0.38	0.17	Yes	0.24 (2.0)
Tin	µg/L	9	38	81%	2.9	39	43	5	11	No	519
Tritium	pCi/L	48	12	20%	0.91	8,800	12,000	3,808	5,854	No	20,000
Uranium	µg/L	60	0	0%	0.38	4.1	4.8	2.8	3	No	30
Vanadium	µg/L	30	30	50%	0.30	26	33	20	17	No	80
Zinc	µg/L	18	42	70%	1.5	34	260	57	37	Yes	91
100-H Source Exposure Area											
Aluminum	µg/L	13	26	67%	1.2	41	188	49	35	No	50
Antimony	µg/L	4	35	90%	0.28	0.61	1	0.72	0.65	Yes	5.6
Arsenic	µg/L	39	0	0%	0.26	3.3	3.7	2.3	2.5	No	0.018 (4.0)
Barium	µg/L	39	0	0%	0.29	67	72	45	48	No	1,000
Boron	µg/L	17	15	47%	0.33	37	56	29	27	No	3,200
Bromoform	µg/L	1	38	97%	N/A	1	0.58	0.58	Not Calculated	N/A	4.3

Table 6-55. Percentile Concentrations and Summary Statistics for the 100-HR-3 Groundwater OU Dataset

COPC	Units	Number of Detections	Number of Nondetects	Frequency of Nondetects	Coefficient of Variation	90th Percentile of RI Data	Max	Mean	95% UCL	Is 95% UCL > 90th Percentile?	Action Level*
Bromomethane	µg/L	1	38	97%	N/A	1	0.97	0.97	Not Calculated	N/A	11
Carbon disulfide	µg/L	1	38	97%	N/A	1	0.076	0.076	Not Calculated	N/A	800
Carbon tetrachloride	µg/L	2	37	95%	1.3	1	2	1	0.82	No	0.23 (1.0)
Chloride	µg/L	39	0	0%	0.24	37,600	44,900	27,013	28,774	No	230,000
Chloroform	µg/L	31	8	21%	0.27	1.4	1.7	1	1.1	No	1.4 (5.0)
Chloromethane	µg/L	1	38	97%	N/A	1	0.1	0.1	Not Calculated	N/A	--
Chromium	µg/L	39	0	0%	0.44	31	34	17	19	No	65
Cobalt	µg/L	16	23	59%	0.81	0.43	0.9	0.29	0.2	No	2.6 (4.0)
Copper	µg/L	21	18	46%	0.79	1.3	2.8	0.8	0.65	No	9.0
Fluoride	µg/L	8	31	79%	0.23	106	114	91	89	No	960
Gross alpha	pCi/L	10	29	74%	0.41	4	7.9	4	3.3	No	15
Gross beta	pCi/L	34	5	13%	0.76	30	58	19	27	No	4.0
Cr(VI)	µg/L	34	5	13%	0.62	26	29	13	14	No	10
Iron	µg/L	29	10	26%	3.5	444	7,840	414	1,575	Yes	300
Lead	µg/L	5	34	87%	0.58	0.23	0.71	0.37	0.29	Yes	2.1
Lithium	µg/L	27	5	16%	0.54	14	23	8.6	11	No	32
Manganese	µg/L	8	31	79%	0.81	35	120	47	25	No	50
Mercury	µg/L	1	38	97%	N/A	0.1	0.11	0.11	Not Calculated	N/A	0.012 (0.5)
Methylene chloride	µg/L	5	34	87%	2.0	1	11	2.4	2.3	Yes	4.6
Molybdenum	µg/L	29	3	9%	1.1	5.4	7.8	1.9	3.3	No	80
Nickel	µg/L	12	27	69%	0.58	8.9	18	8.3	6.3	No	52
Nitrate	µg/L	38	0	0%	0.20	39,800	46,900	30,037	31,686	No	45,000
Nitrite	µg/L	8	31	79%	0.22	1,560	2,270	1,609	1,485	No	3,300

Table 6-55. Percentile Concentrations and Summary Statistics for the 100-HR-3 Groundwater OU Dataset

COPC	Units	Number of Detections	Number of Nondetects	Frequency of Nondetects	Coefficient of Variation	90th Percentile of RI Data	Max	Mean	95% UCL	Is 95% UCL > 90th Percentile?	Action Level*
Selenium	µg/L	38	1	3%	0.37	2.7	3.2	1.7	1.8	No	5.0
Strontium	µg/L	39	0	0%	0.18	391	477	321	337	No	9,600
Strontium-90	pCi/L	12	27	69%	0.77	14	27	11	7.2	No	8.0
Sulfate	µg/L	39	0	0%	0.19	79,700	88,700	62,613	65,752	No	250,000
Technetium-99	pCi/L	5	34	87%	0.82	8.8	35	14	16	Yes	900
Thallium	µg/L	2	37	95%	0.64	0.1	0.28	0.19	0.28	Yes	0.24 (2.0)
Tin	µg/L	1	31	97%	N/A	39	0.11	0.11	Not Calculated	N/A	519
Toluene	µg/L	1	38	97%	N/A	1	0.062	0.062	Not Calculated	N/A	640
Tritium	pCi/L	39	0	0%	0.29	3,900	4,400	2,636	2,843	No	20,000
Uranium	µg/L	39	0	0%	1.0	6.1	13	2.6	3.4	No	30
Vanadium	µg/L	6	33	85%	0.52	12	23	12	13	Yes	80
Zinc	µg/L	9	30	77%	0.67	16	30	14	9.3	No	91
Horn Exposure Area											
1,2-Dichloroethane	µg/L	1	56	98%	N/A	1	0.67	0.67	Not Calculated	N/A	0.38
2-Butanone	µg/L	1	56	98%	N/A	1	10	10	Not Calculated	N/A	4,800
Acetone	µg/L	1	56	98%	N/A	1	6.9	6.9	Not Calculated	N/A	7,200
Aluminum	µg/L	33	24	42%	1.0	54	150	34	30	No	50
Antimony	µg/L	9	48	84%	0.13	0.74	0.95	0.77	0.73	No	5.6
Arsenic	µg/L	52	5	9%	0.51	5.5	7.5	3.2	3.4	No	0.018 (4.0)
Barium	µg/L	57	0	0%	0.27	67	80	48	51	No	1,000
Boron	µg/L	8	35	81%	0.43	24	35	23	21	No	3,200
Carbon tetrachloride	µg/L	10	47	82%	0.35	1.3	1.7	1.2	1.1	No	0.23 (1.0)

Table 6-55. Percentile Concentrations and Summary Statistics for the 100-HR-3 Groundwater OU Dataset

COPC	Units	Number of Detections	Number of Nondetects	Frequency of Nondetects	Coefficient of Variation	90th Percentile of RI Data	Max	Mean	95% UCL	Is 95% UCL > 90th Percentile?	Action Level*
Chloride	µg/L	57	0	0%	0.30	23,600	26,600	15,768	16,798	No	230,000
Chloroform	µg/L	32	25	44%	0.53	1	1	0.44	0.45	No	1.4 (5.0)
Chromium	µg/L	57	0	0%	0.65	76	88	37	43	No	65
Cobalt	µg/L	5	52	91%	0.41	0.1	0.21	0.15	0.14	Yes	2.6 (4.0)
Copper	µg/L	45	12	21%	0.84	1.4	2.8	0.68	0.7	No	9.0
Fluoride	µg/L	42	15	26%	0.45	263	343	167	159	No	960
Gross alpha	pCi/L	17	40	70%	0.31	3.5	5.6	3.4	3.5	No	15
Gross beta	pCi/L	38	19	33%	0.46	12	21	8	7.3	No	4.0
Cr(VI)	µg/L	50	7	12%	0.64	71	90	40	51	No	10
Iron	µg/L	42	15	26%	2.1	422	2,490	271	498	Yes	300
Lead	µg/L	4	53	93%	0.56	0.2	0.66	0.38	0.43	Yes	2.1
Lithium	µg/L	33	10	23%	0.38	12	16	8.1	8	No	32
Manganese	µg/L	13	44	77%	1.4	11	122	33	24	Yes	50
Methylene chloride	µg/L	7	50	88%	0.65	1	0.62	0.33	0.25	No	4.6
Molybdenum	µg/L	43	0	0%	0.57	8.5	12	4.7	5.4	No	80
Nickel	µg/L	8	49	86%	0.31	4.9	7.9	5.5	5.1	Yes	52
Nitrate	µg/L	57	0	0%	0.33	29,550	33,900	20,073	21,558	No	45,000
Nitrite	µg/L	2	55	96%	0.13	118	1,380	1,265	1,380	Yes	3,300
Selenium	µg/L	56	1	2%	0.50	3.2	7.1	2	2.3	No	5.0
Silver	µg/L	2	55	96%	0.79	0.2	1	0.64	1	Yes	2.6
Strontium	µg/L	57	0	0%	0.21	360	409	280	293	No	9,600
Strontium-90	pCi/L	4	53	93%	0.32	0.9	4.2	2.9	2.7	Yes	8.0
Sulfate	µg/L	57	0	0%	0.20	78,350	97,300	62,219	65,061	No	250,000
Technetium-99	pCi/L	1	56	98%	N/A	1.9	12	12	Not Calculated	N/A	900
Tetrachloroethene	µg/L	7	50	88%	0.71	1	0.43	0.2	0.14	No	0.081 (5.0)

Table 6-55. Percentile Concentrations and Summary Statistics for the 100-HR-3 Groundwater OU Dataset

COPC	Units	Number of Detections	Number of Nondetects	Frequency of Nondetects	Coefficient of Variation	90th Percentile of RI Data	Max	Mean	95% UCL	Is 95% UCL > 90th Percentile?	Action Level*
Tin	µg/L	1	42	98%	N/A	39	1.3	1.3	Not Calculated	N/A	519
Toluene	µg/L	2	55	96%	0.34	1	0.18	0.15	0.18	No	640
Trichloroethene	µg/L	3	54	95%	0.13	1	0.33	0.31	0.27	No	0.49 (1.0)
Tritium	pCi/L	55	2	4%	0.52	4,350	4,700	2,569	3,286	No	20,000
Uranium	µg/L	57	0	0%	0.43	4.2	5	2.6	2.8	No	30
Vanadium	µg/L	30	27	47%	0.28	22	29	17	16	No	80
Xylenes (total)	µg/L	2	55	96%	0.031	1	0.46	0.45	0.44	No	1,600
Zinc	µg/L	9	48	84%	0.82	12	46	17	10	No	91

* Value in parentheses () represents the estimated quantitation limit reported in *Sampling and Analysis Plan for the 100-DR-1, 100-DR-2, 100-HR-1, 100-HR-2, and 100-HR-3 Operable Units Remedial Investigation/Feasibility Study* (DOE/RL-2009-40).

Other situations exist for which the recommended 95 percent UCL value is either not calculated (because the frequency of detection is too small) or where the UCL value is higher than the mean but only nominally higher and within less than a single significant digit. For example, the chloroform EPC calculation in the Horn Exposure Area has a 95 percent UCL of 0.45 against a mean value of 0.44.

This discussion only highlights the limits of the 95 percent UCL calculations and the need to use judgment in the establishment of the final EPCs. One advantage of using the 90th percentile is that fewer assumptions are used in the calculation and it can be consistently used for data sets regardless of skewness, frequency of detection issues, multiple detection limits, and other similar factors that can influence the 95 percent UCL calculation. Finally, future monitoring efforts should facilitate the calculation of the 95 percent UCL value to allow future compliance monitoring to provide the requisite data.

A comparison of the 90th percentile values used for the protectiveness and groundwater risk assessments and the 95 percent UCLs is presented in Table 6-55. Table 6-55 also shows the frequency of detection and CVs for the COPCs in groundwater. As shown in Table 6-55, the datasets for most of the COPCs are characterized by a high proportion of BDL values, high CVs, or both; for those COPCs, the 90th percentile is the most appropriate statistic for an EPC. In addition, the 90th percentile concentrations are greater than the 95 percent UCL values for COPCs that are risk drivers in groundwater, such as Cr(VI), nitrate, and strontium-90 in the 100-D and 100-H Areas.

6.3.8.2.1 *100-D Source Exposure Area*

The 90th percentile concentrations of RI data are greater than the 95 percent UCL for 26 of 35 analytes reported on Table 6-55. The 95 percent UCL is greater than the 90th percentile concentration for beryllium, carbon tetrachloride, gross alpha, Cr(VI), lithium, manganese, silver, strontium-90, sulfate, thallium, and zinc. Although the 90th percentile concentrations are less than the 95 percent UCL concentration for beryllium, gross alpha, lithium, manganese, silver, strontium-90, sulfate, thallium, and zinc, both concentrations were less than the action level or EQL (as applicable) and use of the 95 percent UCL would not result in a different conclusion. Although the 90th percentile concentrations are less than the 95 percent UCL concentration for carbon tetrachloride and Cr(VI), both concentrations were greater than the action level or EQL (as applicable) and use of the 95 percent UCL would not result in a different conclusion.

A 95 percent UCL was not calculated for acetone and tetrachloroethene because only one detection was reported for each of these analytes. Therefore, a comparison could not be made.

6.3.8.2.2 100-H- Source Exposure Area

The 90th percentile concentrations of RI data are greater than the 95 percent UCL for seven of 41 analytes reported on Table 6-55. The 95 percent UCL is greater than the 90th percentile concentration for antimony, iron, lead, methylene chloride, technetium-99, thallium and vanadium. Although the 90th percentile concentration is less than the 95 percent UCL concentration for iron, both concentrations were greater than the action level and use of the 95 percent UCL would not result in a different conclusion. The 95 percent UCL is greater than the 90th percentile concentration for iron. Although the 90th percentile concentrations are less than the 95 percent UCL concentration for antimony, lead, methylene chloride, technetium-99, thallium, and vanadium, both concentrations were less than the action level or EQL (as applicable) and use of the 95 percent UCL would not result in a different conclusion.

A 95 percent UCL was not calculated for bromoform, bromomethane, carbon disulfide, chloromethane, mercury, tin, and toluene, because only one detection was reported for each of these analytes. Therefore, a comparison could not be made.

6.3.8.2.3 Horn Exposure Area

The 90th percentile concentrations of RI data are greater than the 95 percent UCL for eight of 42 analytes reported on Table 6-55. The 95 percent UCL is greater than the 90th percentile concentration for cobalt, iron, lead, manganese, nickel, nitrite, silver, and strontium-90. Although the 90th percentile concentration is less than the 95 percent UCL concentration for iron, both concentrations were greater than the action level and use of the 95 percent UCL would not result in a different conclusion. The 95 percent UCL is greater than the 90th percentile concentration for iron. Although the 90th percentile concentrations are less than the 95 percent UCL concentration for cobalt, iron, lead, manganese, nickel, nitrite, silver, and strontium-90, both concentrations were less than the action level or EQL (as applicable) and use of the 95 percent UCL would not result in a different conclusion.

A 95 percent UCL was not calculated for 1,2-dichloroetane, 2-butanone, acetone, technetium-99, and tin, because only one detection was reported for each of these analytes. Therefore, a comparison could not be made.

For the 100-D Source exposure area, the 90th percentile concentrations for chloroform, chromium, and nitrate are greater than the 95 percent UCL values. For Cr(VI), the 95 percent UCL of 1,534 µg/L is greater than the 90th percentile value of 992 µg/L. Both Cr(VI) concentrations are similar, indicating that Cr(VI) is distributed throughout the 100-D Source exposure area and both are greater than the freshwater CCC value of 10 µg/L.

For the 100-H Source exposure area, the 90th percentile value for Cr(VI), nitrate, and strontium-90 are greater than the 95 percent UCL. For Cr(VI), both the 95 percent UCL value of 14 µg/L and the 90th percentile value of 34 µg/L are greater than the freshwater CCC value of 10 µg/L. For strontium-90, the 90th percentile value of 14 pCi/L is greater than the MCL value of 8 pCi/L, whereas the 95 percent UCL value of 7.2 is not greater than the MCL. For nitrate, both the 90th percentile value of 39,800 µg/L and the 95 percent UCL value of 31,868 µg/L are less than the MCL value of 45,000 µg/L.

For the horn exposure area, the 90th percentile values for carbon tetrachloride, chromium, and Cr(VI) are greater than the 95 percent UCL. For carbon tetrachloride, the 95 percent UCL value of 1.1 µg/L and the 90th percentile value of 1.3 µg/L are greater than the MTCA ("Groundwater Cleanup Standards" [WAC 173-340-720]) groundwater cleanup level of 0.23 µg/L. For chromium, the 90th percentile value of 76 µg/L is greater than the freshwater CCC value of 65 µg/L, whereas the 95 percent UCL value of 54 µ/L is not greater than the criterion. For Cr(VI), the 95 percent UCL value of 51 µg/L and the 90th percentile value of 71 µg/L are greater than the freshwater CCC value of 10 µg/L.

6.3.8.3 Uncertainties Associated with Exposure Assessment

The exposure assumptions used to develop the action levels represent an RME. For estimating the RME, 95 percent UCL values (or upper-bound estimates of national averages) are generally used for exposure assumptions, and exposed populations and exposure scenarios are also selected to represent upper-bound exposures. The intent of the RME, as discussed by the EPA Deputy Administrator and the Risk Assessment Council ("Guidance on Risk Characterization for Risk Managers and Risk Assessors" [Habicht, 1992]), is to present risks as a range from central tendency to high-end risk (above the 90th percentile of the population distribution). This descriptor is intended to estimate the risks that are expected to occur in small but definable "high-end" segments of the subject population ("Guidance on Risk Characterization for Risk Managers and Risk Assessors" [Habicht, 1992]). EPA distinguishes between those scenarios that are possible but highly improbable and those that are conservative but more likely to occur within a population, with the latter being favored in risk assessment. In general, these assumptions are intended to be conservative and yield an overestimate of the true risk or hazard.

6.3.8.3.1 *Uncertainties Associated with Dermal Contact Exposure*

The action levels for use as a drinking water source consider ingestion and inhalation of vapors as complete and significant pathways for exposure. For the action levels, the dermal contact pathway is considered a complete but insignificant pathway of exposure for the contaminants detected in groundwater. The exclusion of the dermal contact exposure route from the action levels may have the potential to underestimate the actual cleanup level.

EPA considers the dermal contact route to be significant if it contributes at least 10 percent of the exposure derived from the oral pathway. These results are based on comparing two main household daily uses of water: as a source for drinking, and for showering or bathing (*Risk Assessment Guidance for Superfund Volume I: Human Health Evaluation Manual [Part E, Supplemental Guidance for Dermal Risk Assessment]: Final* [EPA/540/R/99/005]). Exhibits B-3 and B-4 of *Risk Assessment Guidance for Superfund Volume I: Human Health Evaluation Manual [Part E, Supplemental Guidance for Dermal Risk Assessment]: Final* [EPA/540/R/99/005]) provide a screening tool to focus the dermal risk assessment on those chemicals that are more likely to make a contribution to the overall risk. Exhibit B-3 indicates that dermal exposure exceeds 10 percent of drinking water for carbon tetrachloride, chromium, and Cr(VI). The ratio of the dermal absorbed dose from dermal to oral is 40 percent for chromium, 42 percent for Cr(VI), and 17 percent for carbon tetrachloride. Based on this comparison, the action level concentrations may have the potential to underestimate exposure to these COPCs.

6.3.8.3.2 *Uncertainties Associated with Action Levels that include the Fish Consumption Exposure Pathway*

Water quality standards used as action levels to identify COPCs have been developed to include exposure to groundwater contaminants through direct contact (groundwater ingestion and fish consumption). These specific action levels are:

- "Water Quality Standards" (40 CFR 131) for states not complying with Section 303 of the *Clean Water Act of 1977*, Human Health Water + organism
- National Recommended Water Quality Criteria, Ambient Water Quality Criteria (AWQC) established under Section 304 of the *Clean Water Act of 1977*, Human Health Water + organism

These water quality standards were used to identify COPCs in groundwater based on the potential for impacts to surface water. While groundwater adjacent to the Columbia River can discharge to the river through the hyporheic zone, contaminants potentially in groundwater undergo dilution in the river flows to concentrations indistinguishable from levels upstream. Correspondingly, this limits potential accumulation of groundwater contaminants into fish to levels indistinguishable from levels upstream. Based on these factors, potential exposure pathways from groundwater through fish consumption along the Hanford Site are considered incomplete.

6.3.8.3.3 *Uncertainties Associated with Toxicity Assessment*

The toxicological database was also a source of uncertainty. EPA has outlined some of the sources of uncertainty as defined in the risk assessment guide (EPA/540/1-89/002) and in Superfund HHT Risk Assessment Values (Cook, 2003). These sources may include or result from the extrapolation from high to low doses and from animals to humans. This is contingent on the species, gender, age, and strain differences in the uptake, metabolism, organ distribution, and target site susceptibility of a toxin. The human population's variability with respect to diet, environment, activity patterns, and cultural factors are also sources of uncertainty.

Traditionally, EPA has developed toxicity criteria for carcinogens by assuming that all carcinogens are nonthreshold contaminants. However, EPA has recently published revised cancer guidelines (*Guidelines for Carcinogen Risk Assessment I* [EPA/630/P-03/001F]) in which they have modified their former position of assuming nonthreshold action for all carcinogens. This new guidance emphasizes establishing the specific toxicokinetic mode of action that leads to development of cancer. In the future, toxicity criteria for carcinogens in the United States will be developed assuming no threshold for contaminants that exhibit genotoxic modes of action, or where the mode of action is not known. However, currently available EPA toxicity criteria for carcinogens were all derived assuming a nonthreshold model.

In most of the world, nonthreshold toxicity criteria are developed only for those carcinogens that appear to cause cancer through a genotoxic mechanism (International Toxicity Estimates for Risk database [TERA, 2011). Specifically, for genotoxic contaminants, the cancer dose response model is based on high to low dose extrapolation and assumes there is no lower threshold for the initiation of toxic effects. Cancer effects observed at high doses are found in laboratory animals or are extrapolated from occupational or epidemiological studies. Cancer effects observed at low doses are commonly found in environmental exposures. These models are essentially linear at low doses, so no dose is without some risk of cancer.

6.3.8.3.4 *Slope Factors for Carbon Tetrachloride*

For consistency with previous Hanford Site analyses of carbon tetrachloride, an oral slope factor of 0.13 $(mg/kg\text{-}day)^{-1}$ and oral RfD of 0.0007 (mg/kg-day) are used to develop the MTCA ("Groundwater Cleanup Standards" [WAC 173-340-720]) groundwater cleanup level and were previously published in IRIS (December 7, 2009). These values do not reflect the most current oral toxicity values published for carbon tetrachloride.

The oral slope factor and oral RfD currently recommended by EPA for carbon tetrachloride are established by IRIS. The source of these toxicity values is consistent with the hierarchy of toxicity values recommended in Superfund HHT Risk Assessment Values (Cook, 2003). The current oral slope factor is 0.07 $(mg/kg\text{-}day)^{-1}$ and the oral RfD is 0.004 (mg/kg-day). The current IRIS slope factor is lower than the value of 0.13 $(mg/kg\text{-}day)^{-1}$ previously published in IRIS, and the current IRIS oral RfD is higher than the value of 0.0007 (mg/kg-day) previously published in IRIS.

If the current oral slope factor was used to calculate the MTCA ("Groundwater Cleanup Standards" [WAC 173-340-720]) groundwater cleanup level, the groundwater concentration would increase from 0.34 to 0.63 μg/L. The groundwater risk for carbon tetrachloride in the horn exposure area would decrease from 3.8×10^{-6} to 2.1×10^{-6} and the cumulative risk would remain less than 1×10^{-5}.

If the current oral RfD were used to calculate the MTCA ("Groundwater Cleanup Standards" [WAC 173-340-720]) groundwater cleanup level, the groundwater concentration would increase from 5.6 to 32 μg/L. The HQ at the 90th percentile would decrease from 0.23 to 0.04. Use of the oral cancer potency factor and oral RfD previously published in IRIS provides an upper bound on risks when compared to the oral slope factor and oral RfD currently published in IRIS.

6.3.8.3.5 *Slope Factors for Trichloroethene*

The oral cancer potency factor of 0.089 $(mg/kg\text{-}day)^{-1}$ is used to develop the MTCA ("Groundwater Cleanup Standards" [WAC 173-340-720]) groundwater cleanup level was obtained from the CLARC database. The factor is obtained from a draft EPA risk assessment prepared in 2001, which has since been updated; therefore, in this case, the oral cancer potency factor does not reflect the most current source of information.

The oral slope factor currently recommended by EPA for trichloroethene was published in IRIS in September 2011. The source of this toxicity value is consistent with the hierarchy of toxicity values recommended in Superfund HHT Risk Assessment Values (Cook, 2003). The oral slope factor is 0.046 $(mg/kg\text{-}day)^{-1}$ (IRIS). Trichloroethene is considered an "early-life exposure" carcinogen for kidney cancer, because it acts through a mutagenic mode-of-action. Accounting for kidney cancer risks from early-life exposure would result in a slightly more conservative value (by a factor of 2) for the oral slope factor.

The newly revised EPA value is lower than the value of 0.089 $(mg/kg\text{-}day)^{-1}$ for oral exposures published in the CLARC database (Ecology, 2009). However, the EPA value accounting for early life exposure is slightly greater than the value published by Ecology in the CLARC database (Ecology, 2009).

If the revised EPA value was used to calculate the MTCA ("Groundwater Cleanup Standards" [WAC 173-340-720]) level, the groundwater concentration would increase from 0.49 to 0.95 µg/L.

6.3.8.3.6 *Slope Factors for Cr(VI)*

The oral RfD of 0.003 mg/kg-day published by IRIS is used to develop the MTCA ("Groundwater Cleanup Standards" [WAC 173-340-720]) level for Cr(VI). An oral carcinogenic potency factor has recently been published (*Derivation of an Ingestion-Based Soil Remediation Criterion for Cr^{+6} Based on the NTP Chronic Bioassay Data for Sodium Dichromate Dihydrate* [NJDEP, 2009]). The oral carcinogenic potency factor derived is 0.5 $(mg/kg\text{-}day)^{-1}$, as presented in *Derivation of an Ingestion-Based Soil Remediation Criterion for Cr^{+6} Based on the NTP Chronic Bioassay Data for Sodium Dichromate Dihydrate* (NJDEP, 2009). If the NJDEP value were used to calculate the MTCA ("Groundwater Cleanup Standards" [WAC 173-340-720]) level, the groundwater concentration would decrease from 48 to 0.18 µg/L.

6.3.8.4 *Uncertainties Associated with Risk Characterization*

As discussed in Section 6.3.4, MCLs for radionuclides are set at 4 mrem/yr for the sum of the doses from beta particle and photon emitters, 15 pCi/L for gross alpha emitter activity (including Ra-226, but excluding uranium and radon), and 5 pCi/L combined for Ra-226 and Ra-228. A mass concentration MCL has been established for uranium as 30 µg/L. At this time, no additional federal or state standards are associated with evaluating the effects of exposure to radionuclides. Risks were estimated for radioisotopes identified as COPCs using inputs and equation 720-2 from MTCA ("Groundwater Cleanup Standards" [WAC 173-340-720(4)(iii) (B)]) and radionuclide slope factors from HEAST (EPA 540-R-97-036). Exposure inputs and equation 720-2 were selected to calculate risks to be consistent with the risk evaluation section for 100-H Source exposure area that is presented in Section 6.3.6.2.3. Cancer risks for strontium-90 in the 100-H Source exposure area were also calculated using the tap water scenario and presented in Table 6-56. Table 6-56 shows the MCL concentration for strontium-90, and the EPC reported in the 100-H Source exposure area does not individually exceed the 10^{-4} ELCR end of the NCP (40 CFR 300) risk range. Although the 90th percentile value for strontium-90 does not exceed the upper end of the risk range, strontium-90 was retained as a contaminant of potential concern for further evaluation in the FS because it was present in localized areas at concentrations greater than the DWS.

Table 6-56. Summary of 90th Percentile Current Groundwater Concentrations, Associated Cancer Risk, and Associated Sum of Fractions for Radioactive COPCs

COPC	90th Percentile Value (pCi/L)	Federal or State MCL (pCi/L)	ELCR at Federal MCL	Individual Fraction	Individual ELCR
Strontium-90	14	8	8.5×10^{-6}	1.8	1.5×10^{-5}
Sum of Fractions				**1.8**	-
Cumulative ELCR for Radioactive COPCs				-	**1.5×10^{-5}**

6.3.8.4.1 *Uncertainties Associated with the Native American Risk Assessments*

As discussed in Section 6.1.3, the RCBRA (DOE/RL-2007-21, Volume II) evaluated three residential scenarios that describe exposure related to rural land-use patterns that involved exposure assumptions that represented subsistence use. Of the three residential scenarios, two Native American scenarios were evaluated including CTUIR and the Yakama Nation. Although groundwater within the 100-DH OU is not anticipated to become a source of drinking water, contaminants in groundwater were assessed using the two Native American scenarios to provide estimates of human health risks under the assumption of full-time occupancy in the future. In addition, the risks calculated using the Native American scenarios were compared with risks estimated using EPA's standard default assumptions for residential tap water use (the Tap Water scenario). As described in Regional Screening Levels (EPA, 2009a), the residential Tap Water scenario reflects an RME scenario.

The groundwater risk assessment provided in this RI/FS provides an update to address the uncertainties associated with the assessment of groundwater risks presented in the RCBRA (DOE/RL-2007-21, Volume II) (see Section 6.3.2). The uncertainties in the RCBRA (DOE/RL-2007-21, Volume II) were associated with the ability of the groundwater dataset collected from 1998 to 2008 to represent current baseline conditions and potential exposure within each groundwater OU. The following paragraphs discuss the uncertainties with risks associated with groundwater contaminants based on current baseline conditions.

The Native American and Tap Water scenarios addressed direct exposure to contaminants in groundwater associated with household uses of groundwater, such as drinking and cooking (ingestion) and bathing (dermal absorption). If VOCs were measured in groundwater and identified as COPCs, indirect exposure by inhalation of VOCs in air while bathing or when using groundwater in the home for other purposes was also addressed. The CTUIR and Yakama Nation scenarios incorporated ingestion, inhalation, and dermal exposures to COPCs in groundwater used in a sweat lodge.

Exposure parameters for drinking water ingestion, VOC inhalation, and dermal absorption differ between the Native American exposure scenarios and the EPA tap water scenario. Examples of these differences include the following: exposure frequency (Native American 365 day/yr; EPA tap water 350 day/yr); exposure duration (Native American 70 years; EPA tap water 30 years); drinking water ingestion rate (Native American 4 L/day [1 gal/day]; EPA tap water 2 L/day [0.5 gal/day]); and inhalation rate (CTUIR 25 m^3/day [883 ft^3/day], Yakama Nation 26 m^3/day [918 ft^3/day]; EPA tap water 20 m^3/day [706 ft^3/day]). As a result, the Native American exposure scenarios both produce higher total ELCR and HI than the EPA Tap Water scenario. Depending on the contaminants and pathways involved, as described in the following paragraphs, ELCR and HI for the Native American scenarios may be 4- to 5-fold greater than for the Tap Water scenario, drinking water ingestion, VOC inhalation, and dermal absorption exposure pathways. COPCs are the same between each of the exposure scenarios; the percent contribution for each COPC is higher for the Native American scenarios than the EPA Tap Water scenario.

The largest uncertainties associated with the Native American scenarios are with the use of groundwater in a sweat lodge. EPCs for air in a sweat lodge were calculated for the CTUIR Resident and Yakama Resident scenarios. Appendix 4 of *Exposure Scenario for CTUIR Traditional Subsistence Lifeways* (Harris and Harper, 2004) provides equations for estimating air-phase contaminant concentrations for volatile and semivolatile COPCs in the water used to create steam in the lodge, as well as separate equations for nonvolatile COPCs. Inhalation exposure to nonvolatile COPCs in the sweat lodge was evaluated in the CTUIR and Yakama Nation Resident scenarios in spite of concerns with the model for calculating these air-phase EPCs. The *Exposure Scenario for CTUIR Traditional Subsistence Lifeways* (Harris and Harper, 2004) equation for calculating air-phase EPCs for nonvolatile analytes (Equation 3-2) calculates the concentration of a nonvolatile COPC in air as a function of the concentration of water vapor produced by the volatilization of water poured over hot rocks in a sweat lodge. Because nonvolatile contaminants have

no vapor pressure, Equation 3-2 does not have a common physical basis with volatile chemicals. It is possible that inhalation of nonvolatile COPCs might occur by an alternative physical model, such as respiration of respirable-size aerosols, if such aerosols were formed when water is poured over the hot rocks in a lodge. However, a model of resuspension of nonvolatile impurities in aerosol form is inconsistent with other mechanical processes involving steam. For example, EPA does not address this pathway in shower volatilization models (*Volatilization Rates from Water to Indoor Air Phase II* [EPA 600/R-00/096]). It is also inconsistent with the widespread use of steam distillation for commercial water purification.

As described in Section 6.1.4, the RCBRA (DOE/RL-2007-21) presents the risks and hazards calculated for both Native American exposure scenarios from direct contact, external gamma exposure, inhalation, and food chain pathways at remediated waste sites. The groundwater risk assessment presents the risks and hazards calculated for groundwater used as a source of drinking water and as a source of steam for sweat lodge use. The results from the RCBRA (DOE/RL-2007-21, Volume II) for remediated waste sites and the results from the groundwater risk assessment are presented in Table 6-57. The risks and hazards can be summed to obtain a cumulative estimate of risk and hazard for all exposure pathways included in the CTUIR and Yakama Nation exposure scenarios.

Groundwater within the 100-HR-3 OU is currently contaminated, and withdrawal is prohibited as a result of institutional controls placed on it by DOE through the interim action ROD. Under current site use conditions, no complete human exposure pathways to groundwater are assumed to exist. Groundwater within this OU is not anticipated to become a future source of drinking water until cleanup criteria are met, and groundwater is restored to its highest beneficial use.

6.4 Risk Assessment Conclusions of the Riparian and Nearshore Environment from RCBRA

Human health risks were assessed in areas outside the footprints of waste sites as part of the RCBRA (DOE/RL-2007-21, Volume II) and the CRC risk assessment (DOE/RL-2010-117, Volume II). The following sections summarize the conclusions obtained from these two risk assessments. Table 6-58 presents a summary of the total risks and noncancer hazards associated with the riparian and near-shore area and the Columbia River. Several investigations conducted on effluent pipelines that discharged to the Columbia River are also summarized in the following sections.

6.4.1 Risk Assessment Conclusions from the River Corridor Baseline Risk Assessment

The assessment of human health risks was based on "broad area" environmental data that characterized concentrations of COPCs in upland and riparian surface soils, river water and sediment, and fish tissue. The exposure scenarios considered for riparian and near-shore areas were avid angler, casual user, and Tribal scenarios, including nonresident Tribal scenario, and ingestion of fish in the CTUIR and Yakama Nation Resident scenarios. The Casual Recreational User scenario addresses occasional recreational use and is focused on activities such as walking and picnicking in riparian areas near the river. The avid angler is focused on individuals who are not engaged in a subsistence lifestyle. The avid angler application is associated with exposure in the near-shore region of the River Corridor, and takes into consideration potential exposures to sediments and fish. The nonresident Tribal scenario is focused on individuals engaged in a subsistence lifestyle who reside offsite but use the River Corridor for various activities such as hunting, gathering plants, and fishing.

EPCs in soil in the riparian environment were calculated using MULTI INCREMENT® sampling from riparian locations in 100-D/H OU (RCBRA SAP [DOE/RL-2005-42]). Discrete sediment samples used to

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calculate EPCs were obtained from sites in the River Corridor selected from locations of known groundwater plumes, areas of groundwater discharge to the river, results of past biota sampling locations, or areas of fine-grained sediment deposits. Data from sculpin, clams, and benthic macroinvertebrates (primarily crayfish) were used to estimate fish ingestion risks to avid angler and nonresident Tribal receptors.

The results of the broad area risk assessment in the 100-D/H OU area for the Casual Recreational User and Avid Angler scenarios showed that lifetime cancer risks generally were near 1×10^{-6} and were below a noncancer HI of 1 for direct exposures to soil, sediment, and surface water.

Risks for riparian soils were higher than a 1×10^{-4} cancer risk and above a noncancer HI of 1 for the nonresident Tribal scenario. Modeled concentrations of arsenic from riparian soil into native vegetation provided the largest contribution to cancer risks and noncancer HIs. However, as discussed in the RCBRA (DOE/RL-2007-21, Volume II), uncertainties in the food chain modeling methods considerably overstate risks from plant ingestion exposure pathways for arsenic. There were no cancer risks estimated from fish ingestion for any of the scenarios evaluated, because no carcinogenic COPCs were detected in fish tissue samples in 100-D/H. The noncancer HI for fish ingestion with the nonresident Tribal scenario exceeded 1. In the 100-D/H OU, nickel detected in sculpin was the driver for noncancer risks from fish ingestion. The RCBRA (DOE/RL-2007-21, Volume II) concluded that sculpin caught close to the OUs would not be expected to reflect risks potentially associated with food fish with large home ranges. Noncancer HIs calculated with the combination of localized concentrations in sculpin with subsistence ingestion rates are probably overstated.

Based on the results from this analysis, there are no additional COPCs identified in riparian soils, near-shore sediments, and surface water that warrant further evaluation in the FS. The COPCs identified for waste sites are inclusive of the riparian area. Uncertainties in the estimation of human health risks, as described in the RCBRA, suggest that these risks have been considerably overstated.

6.4.2 Risk Assessment Conclusions from the Columbia River Component

The CRC HHRA (DOE/RL-2010-117, Volume II) provides a comprehensive assessment of human health risks for the Hanford Reach. The intent of the CRC HHRA (DOE/RL-2010-117, Volume II) was to complete the assessment of the "bank-to-bank" Hanford Reach and downstream areas (that is, Lake Wallula) of the Columbia River, characterizing risk in areas not previously addressed under the RCBRA. Human exposure scenarios addressed in the CRC HHRA (DOE/RL-2010-117, Volume II) were an avid angler, casual user, hypothetical future resident, and a Native American (Yakama Nation) subsistence fisher. As discussed in the CRC HHRA (DOE/RL-2010-117, Volume II), fish ingestion exposure provided the largest contribution to overall human health risks. A fish sampling program was specifically created to support the CRA HHRA (DOE/RL-2010-117, Volume II) and provided a consistent sampling and analysis approach among species, tissue types, and analytes (Columbia River RI Work Plan [DOE/RL-2008-11]). The fish species targeted in the sampling program were intended to be the most representative of the exposure scenarios identified in the HHRA, and included the following:

- Common carp (Cyprinus carpio)
- Mountain whitefish (Prosopium williamsoni)
- Walleye (*Sander vitreus*)
- Smallmouth bass (Micropterus dolomieui)
- Bridgelip sucker (Catostomus columbianus)
- White sturgeon (Acipenser transmontanus)

Table 6-57. Summary of Cancer Risks and Noncancer Hazards for Soil and Groundwater Exposure Pathways Associated with the CTUIR and the Yakama Nation Exposure Scenarios

Environmental Medium/ Exposure Pathway	CTUIR Total ELCR	CTUIR Risk Drivers (Contributes > 1 × 10^{-6})	CTUIR Hazard Index	CTUIR Hazard Drivers	Yakama Nation Total ELCR	Yakama Nation Risk Drivers (Contributes > 1 × 10^{-6})	Yakama Nation Hazard Index	Yakama Nation Hazard Drivers
100-DSource								
Remediated Waste Sites (Direct Contact and Food Chain Pathways)	4.0×10^{-2} to 2×10^{-5}	Arsenic, cesium-137, europium-152, strontium-90, technetium-99	0.048 to 380	Arsenic, mercury	4.0×10^{-2} to 3×10^{-5}	Arsenic, cesium-137, europium-152, strontium-90, technetium-99	0.16 to 430	Arsenic, mercury
Groundwater as a Potential Drinking Water Source	3.4×10^{-4}	Bromodichloro-methane, carbon tetrachloride, chloroform, tetrachloroethene, tritium,	23	Cr(VI)	3.7×10^{-4}	Bromodichloro-methane, carbon tetrachloride, chloroform, tetrachloroethene, tritium,	23	Cr(VI)
Groundwater as a Potential Source of Steam from Sweat Lodge Use (Includes Vaporized Nonvolatiles)	5.0×10^{-1}	Cr(VI)	99	Cr(VI)	1.0×10^{-0}	Cr(VI)	716	Arsenic, barium, cadmium, cobalt, Cr(VI), manganese, nickel
Groundwater as a Potential Source of Steam from Sweat Lodge Use (Excludes Vaporized Nonvolatiles)	--	--	--	--	1.6×10^{-4}	Arsenic, bromodichloro-methane, carbon tetrachloride, tetrachloroethene, tritium	48	Cr(VI)
100-HSource								
Remediated Waste Sites (Direct Contact and Food Chain Pathways)	4.0×10^{-2} to 2×10^{-5}	Arsenic, cesium-137, europium-152, strontium-90, technetium-99	0.048 to 380	Arsenic, mercury	4.0×10^{-2} to 3×10^{-5}	Arsenic, cesium-137, europium-152, strontium-90, technetium-99	0.16 to 430	Arsenic, mercury
Groundwater as a Potential Drinking Water Source	4.0×10^{-4}	Carbon tetrachloride strontium-90, tritium.	3.3	Cr(VI)	4.3×10^{-4}	Strontium-90, tritium. carbon tetrachloride	3.3	Cr(VI)
Groundwater as a Potential Source of Steam from Sweat Lodge Use (Includes Vaporized Nonvolatiles)	1.8×10^{-2}	Cr(VI)	13	Cr(VI), arsenic, barium	1.3×10^{-1}	Cr(VI)	96	Arsenic, barium, cobalt, Cr(VI), manganese, nickel, uranium
Groundwater as a Potential Source of Steam from Sweat Lodge Use (Excludes Vaporized Nonvolatiles)	--	--	--	--	6.7×10^{-5}	Arsenic, carbon tetrachloride, chloroform, tritium	1.5	--
Horn Area								
Soil (Direct Contact and Food Chain Pathways)	4.0×10^{-2} to 2×10^{-5}	Arsenic, cesium-137, europium-152, strontium-90, technetium-99	0.048 to 380	Arsenic, mercury	4.0×10^{-2} to 3×10^{-5}	Arsenic, cesium-137, europium-152, strontium-90, technetium-99	0.16 to 430	Arsenic, mercury
Groundwater as a Potential Drinking Water Source	5.7×10^{-4}	1,2-dichloroethane, tetrachloroethene, trichloroethene, tritium	4.4	Cr(VI)	6.2×10^{-4}	1,2-dichloroethane, tetrachloroethene, trichloroethene, tritium	4.4	Cr(VI)

Table 6-57. Summary of Cancer Risks and Noncancer Hazards for Soil and Groundwater Exposure Pathways Associated with the CTUIR and the Yakama Nation Exposure Scenarios

Environmental Medium/ Exposure Pathway	CTUIR Total ELCR	CTUIR Risk Drivers (Contributes > 1 × 10^{-6})	CTUIR Hazard Index	CTUIR Hazard Drivers	Yakama Nation Total ELCR	Yakama Nation Risk Drivers (Contributes > 1 × 10^{-6})	Yakama Nation Hazard Index	Yakama Nation Hazard Drivers
Groundwater as a Potential Source of Steam from Sweat Lodge Use (Includes Vaporized Nonvolatiles)	4.9×10^{-2}	Cr(VI)	14	Cr(VI), arsenic, barium, manganese	3.1×10^{-1}	Cr(VI)	100	Arsenic, barium, cobalt, fluoride, Cr(VI), manganese, nickel
Groundwater as a Potential Source of Steam from Sweat Lodge Use (Excludes Vaporized Nonvolatiles)	--	--	--	--	9.6×10^{-5}	1,2-dichloroethane, arsenic, carbon tetrachloride, chloroform, tetrachloroethene, trichloroethene, tritium	3.6	Cr(VI)

Table 6-58. Summary of Total Risks and Noncancer Hazards for the Riparian and Near-Shore Areas in the 100-D/H OUs

Environment/Exposure Media	Excess Lifetime Cancer Risk	Primary Risk Drivers	Noncancer Hazard Index	Primary Noncancer Hazards	Comment	Source
Casual User Scenario						
Riparian Soil	3.0×10^{-6}	None	0.02	None	--	RCBRA - DOE/RL-2007-21, Tables 4-14 and 4-16
Avid Angler Scenario						
Near-Shore – sediment, river water, dust	2.0×10^{-6}	None	0.006	None	--	RCBRA - DOE/RL-2007-21, Tables 4-17 and 4-19
Fish ingestion - Sculpin	0	None	4.3	Metals	Screening-level result employing near-shore COPC concentrations in sculpin, a small fish with a limited home range.	
Nonresident Tribal Scenario						
Soil, sediment, water	6.0×10^{-5}	Arsenic	0.78	None	--	RCBRA - DOE/RL-2007-21, Tables 4-24 and 4-26
Plants and game	1.0×10^{-2}	Arsenic	80	Arsenic	Ingestion of contaminants in plants and game were modeled using high end biotransfer factors, which overstated concentrations accumulated from soil. Uncertainties associated with the large range of published bio-transfer factors.	
Fish Ingestion - Sculpin	NA	None	25	Metals	Screening-level result employing near-shore COPC concentrations in sculpin, a small fish with a limited home range.	
Casual User Scenario (Child - Columbia River)						
100-A Study Area COPCs	0	None	0.005	None	Risks in each	CRC -

Table 6-58. Summary of Total Risks and Noncancer Hazards for the Riparian and Near-Shore Areas in the 100-D/H OUs

Environment/Exposure Media	Excess Lifetime Cancer Risk	Primary Risk Drivers	Noncancer Hazard Index	Primary Noncancer Hazards	Comment	Source
in Surface Water					media summed across chemical carcinogens and radionuclides.	DOE/RL-2010-117, Tables 6-1 and 6-2.
100-A Study Area COPCs in Sediment	$4.0\ 10^{-7}$	None	0.02	None		
100-B Study Area COPCs in Surface Water	0	None	0.003	None		
100-B Study Area COPCs in Sediment	$4.0\ 10^{-7}$	None	0.02	None		
100-B Study Area COPCs in Island Soil	8×10^{-7}	None	0.02	None		
Casual User Scenario (Adult - Columbia River)						
100-A Study Area COPCs in Surface Water	0	None	0.001	None	Risks in each media summed across chemical carcinogens and radionuclides.	CRC - DOE/RL-2010-117, Tables 6-13 and 6-14.
100-A Study Area COPCs in Sediment	3.0×10^{-6}	None	0.003	None		
100-B Study Area COPCs in Surface Water	0	None	0.0009	None		
100-B Study Area COPCs in Sediment	2×10^{-6}	None	0.002	None		
100-B Study Area COPCs in Island Soil	5×10^{-7}	None	0.003	None		
Avid Angler Scenario (Child - Columbia River)						
100 Area, Fish Ingestion	1×10^{-6}	Carbon-14	7	PCBs (dioxin and non-dioxin like)	--	CRC - DOE/RL-2010-117, Table 6-25
Avid Angler Scenario (Youth - Columbia River)						
100-A Study Area COPCs in Surface Water	0	None	0.001	None	Risks in each media summed across chemical carcinogens and radionuclides.	CRC - DOE/RL-2010-117, Tables 6-28 and 6-29.
100-A Study Area COPCs in Sediment	7.0×10^{-7}	None	0. 005	None		
100-A Study Area COPCs in Fish	4×10^{-6a}	Carbon-14	7	PCBs (dioxin and non-dioxin like)		
100-B Study Area COPCs in Surface Water	0	None	0.0008	None		
100-B Study Area COPCs in Sediment	5.0×10^{-7}	None	0.003	None		

Table 6-58. Summary of Total Risks and Noncancer Hazards for the Riparian and Near-Shore Areas in the 100-D/H OUs

Environment/Exposure Media	Excess Lifetime Cancer Risk	Primary Risk Drivers	Noncancer Hazard Index	Primary Noncancer Hazards	Comment	Source
100-B Study Area COPCs in Island Soil	3.0×10^{-7}	None	0.006	None		
100-B Study Area COPCs in Fish	4×10^{-6}[a]	Carbon-14	7	PCBs (dioxin and non-dioxin like)		
Avid Angler Scenario (Adult - Columbia River)						
100-A Study Area COPCs in Surface Water	1.0×10^{-7}	None	0.0007	None	Risks in each media summed across chemical carcinogens and radionuclides.	CRC - DOE/RL-2010-117, Tables 6-34 and 6-35.
100-A Study Area COPCs in Sediment	3.0×10^{-6}	None	0. 003	None		
100-A Study Area COPCs in Fish	3.0×10^{-5}[a]	Carbon-14	7	PCBs (dioxin and non-dioxin like)		
100-B Study Area COPCs in Surface Water	0	None	0.0005	None		
100-B Study Area COPCs in Sediment	2.0×10^{-6}	None	0.002	None		
100-B Study Area COPCs in Island Soil	5.0×10^{-7}	None	0.003	None		
100-B Study Area COPCs in Fish	3.0×10^{-5}[a]	Carbon-14	7	PCBs (dioxin and non-dioxin like)		

Notes:

Carbon-14 was detected in one sucker fillet at a concentration slightly higher than the minimum detectable activity of the instrument and is likely a false positive result. Risk contributions of carbon-14 were low relative to the contribution of risk from PCBs and chlorinated pesticides. Carbon-14 was not detected in near-shore groundwater, seeps, or sediment, but was detected in one soil sample collected from the riparian area.

Zero values indicate that there were no COPCs for that medium; therefore, no risks or noncancer hazards were calculated.

Risks presented in this table are for COPCs identified in the Study Area (that is, along the River Corridor sites. COPCs for Reference Areas are presented in the CRC report. Note that risks associated with Reference Area COPCs typically are greater than risks associated with Study Area COPCs.

Risk estimates for the CTUIR and Yakama Nation scenarios are provided in *River Corridor Baseline Risk Assessment, Volume II: Human Health Risk Assessment* (DOE/RL-2007-21) and *Columbia River Component Risk Assessment, Volume II: Baseline Human Health Risk Assessment* (DOE/RL-2010-117) risk assessment reports.

Separate fillets, carcass (including the head and skeleton), and combined livers and kidneys were analyzed. Fillet samples for all species except sturgeon were prepared with the skin on, because the skin for these types of fish is often left on during preparation and consumption.

As described previously, the fish consumption pathway provided the largest contribution (99 percent contribution) to human health risks (evaluated for the Avid Angler and Native American scenarios). The fish consumption pathway was evaluated using two separate approaches. In the first approach, risk was quantified assuming a receptor consumed a varied diet consisting of all six species evaluated. In a second approach, risk was quantified for each individual fish species. Although the concentrations of COPCs, and hence, estimated hazard/risk, varied among the different species, the relative magnitude of risk remained similar among all six fish species. Relative magnitude of risk for the Avid Angler scenario was generally in the range of 2×10^{-3} to 8×10^{-3}, with bass and carp having the overall lowest and highest associated cancer risk, respectively. PCBs, chlorinated pesticides (notably dieldrin), cobalt, lithium, and mercury were the primary risk drivers through fish ingestion. Throughout the 100 Area sub-area (where the 100-D/H OU is located), all of the risk drivers in fish also were identified as COPCs in upstream reference areas. Carbon 14 was the only radionuclide consistently detected among fish tissue samples although at a very low (1 percent) frequency of detection. Carbon-14 was also only sporadically detected in abiotic media.

PCBs, mercury, and chlorinated pesticides in fish tissue, which are primary risk drivers, are prevalent in fish tissue in many waterbodies, because of their widespread historical use, atmospheric deposition and, consequently, high prevalence in abiotic media. The results from Chapter 4 and Riparian and Near-Shore CSM in Appendix L show that there are unlikely to be sources or transport pathways from Hanford Site soils or groundwater that would have resulted in transport of PCBs, mercury, or chlorinated pesticides to Columbia River media (sediment or surface water) where they could have been accumulated into fish tissue. Based on the absence of transport pathways for these contaminants from the 100-D/H OU sites or groundwater, coupled with comparable risks associated with fish caught in reference areas, it is unlikely that Hanford Site activities in the 100-D/H OU are associated with the fish ingestion risks projected in the CRC HHRA (DOE/RL-2010-117, Volume II).

Results from the risk characterization indicate that the risks related to exposure to surface water and sediment are very small relative to that from the fish ingestion pathway. Cumulative risks for the Casual Recreational User scenario (which included direct contact exposure pathways to sediment and surface water) were 7×10^{-6} in the 100 Area sub-area. Arsenic in sediment within most of the exposure points accounted for over half of the cumulative risk. Of the radionuclides, cobalt-60, europium-152, and cesium-137 constitute the majority of radiation cancer risk. Cesium-137 is a known constituent of worldwide atmospheric fallout and was found largely in reference areas.

Risks from island soil exposures were relatively minor compared to risks from other abiotic media, cobalt-60 in soils collected from island soils was a contributor to risk; however, it was reported at a low frequency of detection (1 of 69 island soils) and at low concentration (0.016 pCi/g) (the residential PRG for cobalt-60 is 3.3 pCi/g). Cobalt-60 was not detected in the soils collected from the 100 Area sub area.

In the early 1990s, the upstream half (12.5 acres) of 100-D Island (100-D-67) was surveyed using the Ultrasonic Ranging and Data System (USRADS) (*100-D Island USRADS Radiological Surveys Preliminary Report – Phase II* [BHI-00134]). Areas of elevated radiation readings were found to be discrete radioactive particles (specks) that were in the silt 10.1 to 25.4 cm (4 to 10 in.) beneath the surface and between the 4-6 inch diameter cobbles that make up the bulk of the soil on 100-D Island. During the USRADS surveys in April 1992, the specks that were found were removed and a portion of them were counted in the laboratory. The only radionuclide found in the majority of the specks was cobalt-60. In 1992, the highest activity speck contained 22 micro-Curies of cobalt-60 with the average specks containing 2.5 micro-Curies. Calculations based on the maximum number of specks found in a volume of soil show that the soil activity due to cobalt-60 in 1992 was 0.45 pCi/g.

The Washington State Department of Health (WDOH) conducted a risk assessment on cobalt-60 present in particulates on 100-D Island (*100-D Island Radiological Survey* [WDOH/ERS-96-1101]). The carcinogenic risk associated with the cobalt-60 particles was stated to be the result of two pathways: external exposure and ingestion. The maximum potential dose rate from external exposure was estimated to be 0.04 mrem/year based on a recreational scenario. The WDOH study (*100-D Island Radiological Survey* [WDOH/ERS-96-1101]) also reported the carcinogenic risk from external exposure and ingestion of soil to be 2.7×10^{-8} and 2.3×10^{-11}, respectively, and concluded that the risks from radioactive specks were not sufficient to justify further surveys to locate and remove them. Since 1993, cobalt-60 has decayed through almost four halflives resulting in present day risks that are considerably less than these values. In 2004, the 100-D Island was surveyed using Laser-Assisted Ranging and Data System (LARADS). The results of the survey showed that levels of gamma-emitting radionuclides were present at or slightly above background levels, with maximum readings between background and 5,000.

Based on conclusions from previous studies and because of radioactive decay, it is concluded that no further remedial action is warranted for 100-D Island.

6.4.3 Risk Assessment Conclusions for River Pipelines

During operations, water used in fuel production to cool the reactors was discharged to the Columbia River via effluent pipelines. The release of this cooling water ended when the associated reactors and facilities were shut down. Today, the three inactive 100-D/H effluent pipelines remain in their original locations in the Columbia River channel. Past characterization efforts obtained samples of the river effluent pipelines from the 100-BC, 100-D, and 100-F areas. Characterization data collected during the river pipeline evaluations were used to evaluate potential risks from contaminants within the pipelines. The RCBRA (DOE/RL-2007-21) provided a summary of the previous characterization efforts and risk assessment for these pipelines in Section 8.2.2.

In 1984, *River Discharge Lines Characterization Report* (UNI-3262) discussed samples of scale (flakes of mostly rust) from the interior surfaces and enclosed sediment of the effluent pipelines from the 105-C, 105-DR, and 105-F Reactors. The pipelines were also visually inspected underwater by a diver, and their positions and physical conditions were assessed. Samples of scale and sediment were analyzed for radionuclides. The major radionuclides detected included cobalt-60, cesium-137, europium-152, europium-154, and europium-155. Radionuclide concentrations were greater in the scale than in the sediment. Direct beta-gamma radiation measurements were also obtained for interior and exterior pipe surfaces. The human health risk assessment determined that elevated human radiological exposure could occur if portions of the river pipelines became dislodged and washed ashore (RCBRA [DOE/RL-2007-21], Section 8.2.2).

In 1994, a comprehensive geophysical survey (*Columbia River Effluent Pipeline Survey* [WHC-SD-EN-TI-278]) located and mapped the reactor effluent pipelines. The study relied mainly on remote sensing geophysical techniques, including navigation and echo sounding, side-scanning radar, sub-bottom profiling, seismic reflection profiling, and ground-penetrating radar. The results indicated that the pipelines have neither broken loose nor moved from their original locations. However, portions of some pipelines are no longer buried.

In 1995, pipe scale and sediment from the interior of the effluent pipelines from the 100-BC and 100-D areas were sampled and physically characterized using a robotic transporter (*100 Area River Effluent Pipelines Characterization Report* [BHI-00538]). Analytical data from these two pipelines were intended to complement the 1984 radionuclide data (*River Discharge Lines Characterization Report* [UNI-3262]) and were expected to represent "worst case" conditions with respect to radiological

contamination. This assumption was based on the long years of pipeline service and the volume of effluent known to have been discharged from the 105-B and 105-D/DR Reactors.

Evaluations of human health and ecological risk have been performed for the river effluent pipelines, as they are today, located on or beneath the river channel bottom, and for a scenario in which a pipeline section breaks away from the main pipeline and is washed onto the shore of the river. Both the 1996 risk assessment effort (*100 Area River Effluent Pipelines Characterization Report* [BHI-00538]) and the 1998 risk assessment effort (*100 Area River Effluent Pipelines Risk Assessment* [BHI-01141]) relied on data collected from the 1984 and 1995 characterization work. The evaluation of human health and ecological risk performed in 1998 (*100 Area River Effluent Pipelines Risk Assessment* [BHI-01141]) concluded that the concentrations of chromium and mercury in the scale and sediment within the pipelines pose minimal ecological risk, because they have been in contact with river water without dissolving since the reactors were shut down. The 1998 risk evaluation results indicated pipelines present no unacceptable risks and, therefore, there are no remediation requirements under CERCLA. This is supported by the following:

- Minimal deteriorated condition of the pipelines
- Continued decrease of radionuclide concentrations due to decay (radioactivity would be less than 15 mrem/yr above background by Year 2022)
- Inaccessible location
- Unavailability of significant contaminants to affect human health and the environment

Based on available information, no elevated risk levels are expected to be associated with these pipelines.

6.5 Summary and Conclusions

The soil and groundwater risk assessments for the 100-D/H source and groundwater OUs accomplish the following objectives:

- Proposes direct contact PRGs in soil for use in the FS consistent with values presented in the RCBRA (DOE/RL-2007-21).
- Evaluates the effectiveness of source interim actions for the 100-D/H Source OU.
- Qualitatively evaluates soil data from RI and LFI soil borings and wells to determine whether results could be useful for risk management decisions.
- Confirms that previously remediated waste sites meet RAOs and remedial action goals published in the 100 Area RDR/RAWP (DOE/RL-96-17). In addition, confirms that waste sites achieve the direct contact PRGs proposed for the FS. In other words, sites cleaned up under interim action do not need to be revisited in the FS to demonstrate protection of human health.
- Identifies the waste sites and COCs in the vadose zone that require further evaluation in the FS.
- Confirms that waste sites that have not been remediated are carried forward into the FS based on process history and/or sampling results.
- Identifies the COCs in groundwater that require further evaluation in the FS.

The methodology used to assess risks for the RI/FS uses PRGs developed in the RCBRA and incorporates the most current agency guidance. COPCs in the vadose zone and groundwater were identified in a conservative manner, using exclusions identified in the RCBRA to identify COPCs. The methods for

developing EPCs are based on EPA's ProUCL guidance manual. The Residential scenario used to develop PRGs and characterize risks to human health from contaminants in the vadose zone is drawn from the scenario that was used to develop cleanup levels for the 100 Area RDR/RAWP (DOE/RL-96-17), and was brought up to date to be consistent with the most recent regulatory guidance. PRGs for the vadose zone were developed to reflect a range of exposure scenarios and include those that represent the RAOs (Residential scenario) and reasonably anticipated future land use (Resident Monument Worker and Casual Recreational User).

Contaminant concentrations in groundwater were compared with a range of groundwater and surface water standards for protection of human health and aquatic organisms. In addition, risks from contaminants in groundwater were assessed using Tribal scenarios based on assumptions provided by the CTUIR and Yakama Nation. The EPA Tap Water scenario is also evaluated to provide a similar scenario using exposure assumptions that represent a RME.

Cumulative risks were calculated for multiple contaminants and multiple exposure pathways by exposure media (that is, soil or groundwater). Cumulative risks summed across soil and groundwater were not calculated for the Residential scenario because the RME for this scenario does not include combined exposures to both media; therefore, they are presented separately.

RI and LFI data were compared to PRGs developed in the RCBRA. Soil samples collected from depth intervals ranging from 0 to 4.6 m (15 ft) bgs were combined and compared to PRGs, including those that represent the RAOs (Residential scenario) and reasonably anticipated future land use (Resident Monument Worker and Casual Recreational User). Soil samples collected from depth intervals greater than 4.6 m (15 ft) bgs were combined and compared to residential PRGs.

The protection of groundwater and surface water from contaminants currently in the vadose zone was discussed in Chapter 5. The ecological risk assessment that evaluates the protection of terrestrial receptors is discussed in Chapter 7.

6.5.1 Conclusions for the Soil Risk Assessment

The principal contaminants in the vadose zone are radionuclides and metals. The radionuclides can be categorized as being related to waste disposal, including cesium-137, cobalt-60, europium-152, europium-154, and strontium-90.

6.5.1.1 Shallow Zone Results for Closeout Verification Data

Cancer risks associated with all radionuclides at remediated waste sites within the top 4.6 m (15 ft) of soil are in the 10^{-4} range for both the 100-D and 100-H Source OUs, based on the Residential exposure scenario. Two waste sites in the 100-D Source OU and one waste site in the 100-H Source OU were reported with individual COPCs greater than 1×10^{-4}. Cancer risks associated with the Resident Monument Worker scenario are similar to the Residential scenario. Cancer risks for the resident Monument worker have a cumulative ELCR approximately 0.75 times lower than the unrestricted (resident)R. Cancer risks for a Casual Recreational User scenario are approximately two orders of magnitude lower than the Residential scenarios. This slight exceedance of target risk thresholds is a result of health protective levels being updated from a target annual dose rate of 15 mrem/yr to a target risk of 1×10^{-4} to be consistent with *Radiation Risk Assessment At CERCLA Sites: Q & A* (EPA/540/R/99/006). In addition, the radionuclides related to waste disposal have relatively short half-lives. It is anticipated that concentrations would decay to levels corresponding to EPA's target risk range within 50 years.

Concentrations of strontium-90 in the 100-D-48:3 shallow decision unit, concentrations of cobalt-60 and nickel-63 in the 100-D-42, 100-D-43, and 100-D-45 shallow focused decision unit, and concentrations of

europium-152 and strontium-90 in the 100-D-47 shallow focused decision unit have decayed to residential RBSLs and do not warrant further evaluation in the FS. The following waste sites contain Hanford Site-related COPCs in the top 4.6 m (15 ft) and warrant further evaluation as COCs in the FS:

- 116-D-8 shallow focused 2 decision unit contained cesium-137 at a concentration of 7.63 pCi/g, resulting in a risk of 1.7×10^{-4}, when sampled in 2011. Activities of all radionuclides will decay to a total ELCR of less than 1.0×10^{-4} by year 2035.
- 116-DR-9 shallow decision unit contained cesium-137 at a concentration of 10 pCi/g, resulting in a risk of 2.0×10^{-4}, when sampled in 1999. Cesium-137 concentrations will decay to levels less than the residential RBSL of 4.4 pCi/g by year 2035. Activities of all radionuclides will decay to a total ELCR of less than 1.0×10^{-4} by year 2038.
- 118-D-6:4 shallow 2 decision unit contained cesium-137, europium-152, and strontium-90 at concentrations of 2.9 pCi/g, 1.4 pCi/g, and 0.36 pCi/g resulting in a risk of 1.2×10^{-4}, when sampled in 2010. Activities of all radionuclides will decay to a total ELCR of less than 1.0×10^{-4} by year 2022.
- 118-DR-2:2 shallow decision unit contained technetium-99 at a concentration of 2 pCi/g when sampled in 2000, resulting in a risk of 2.2×10^{-4}. The technetium-99 concentration is greater than the residential RBSL of 1.5 pCi/g and does not decay to the residential RBSL within a reasonable period.
- 116-H-5 shallow decision unit contained strontium-90 at a concentration of 2.4 pCi/g, resulting in a risk of 1.1×10^{-4}, when sampled in 2011. Activities of all radionuclides will decay to a total ELCR of less than 1.0×10^{-4} by year 2016.
- 118-H-1:1 shallow 2 decision unit contained strontium-90 at a concentration of 2.3 pCi/g, resulting in a risk of 1.2×10^{-4}, when sampled in 2010. Activities of all radionuclides will decay to a total ELCR of less than 1.0×10^{-4} by year 2016.

For nonradiological contaminants, the COPCs that are the largest contributors to calculated risks and HIs are metals at the 100-D Source OU and metals, PCBs, and PAHs at the 100-H Source OU. For all exposure scenarios, for waste sites that have been remediated under interim action RODs, the cancer risks and noncancer HIs for nonradioactive contaminants fell within EPA's target risk ranges. Concentrations of arsenic in vadose zone material are associated with cancer risks higher than 10^{-6} under unrestricted (residential) exposure assumptions. Two waste sites were reported with PAH concentrations, and one waste site was reported with Aroclor-1260 greater than the acceptable risk value of 1×10^{-6} for individual carcinogens but less than the MTCA (HHRA Procedures [WAC 173-340-708(5)]) cumulative risk threshold of 1×10^{-5} for multiple contaminants and multiple pathways. The concentrations of arsenic in vadose zone material posing risks greater than 10^{-6} are consistent with Sitewide naturally occurring background in vadose zone material. No waste sites require further evaluation in the FS based on the presence of nonradiological COPCs.

6.5.1.2 Shallow Zone Results for RI and LFI Data

Soil samples were collected from depths ranging between 0 to 4.6 m (15 ft) bgs for two RI soil borings (both from 100-H Source OU), five RI test pits (three from 100-D Source OU and two from 100-H Source OU), and seven LFI test pits (all from 100-D Source OU). The conclusions from the evaluation of the shallow zone RI and LFI data are consistent with the conclusions of the soil risk assessment.

For the 100-D Source OU, four LFI sample locations (116-D-1A Trench, 116-D-7 Retention Basin, 116-DR-9 Retention Basin, and 116-D-2 Crib) report soil concentrations greater than residential RBSLs. These four waste sites have been subsequently remediated under the interim action ROD. The soil risk

assessment did not identify risks associated with these sites based on current conditions. At the 108-D/Sodium Dichromate Tanks test pit, benzo(a)pyrene was reported as an estimated concentration of 180 µg/kg in the 1.5 m (5 ft) bgs depth interval. The benzo(a)pyrene concentration of 180 µg/kg is slightly greater than the residential RBSL of 137 µg/kg. Benzo(a)pyrene is the only analyte reported at a concentration greater than a RBSL at this location. In addition, the 108-D/Sodium Dichromate Tanks test pit is located within the footprint of the 100-D-101 waste site that is identified as an accepted WIDS waste site.

For the 100-H Source OU, three LFI sample locations (116-H-1 Trench, 116-H-7 Retention Basin, and 116-H-9 Crib) report soil concentrations greater than residential RBSLs. These three waste sites have been subsequently remediated under the interim action ROD. The soil risk assessment did not identify risks associated with these sites based on current conditions.

6.5.1.3 Deep Zone Results for Closeout Verification Data

Deep vadose zone samples were evaluated to identify remediated waste sites where exposure to residual contamination could present a potential risk from an inadvertent exposure through deep excavation activities. While this exposure would be industrial in nature, the RBSLs (developed for the Residential exposure scenario) were used for convenience as screening values to identify such sites in order to allow institutional controls to be established to control access to deep contamination.

Eighteen waste sites represented by the following 20 decision units are reported with concentrations of one or more radioisotopes (cesium-137, cobalt-60, europium-152, europium-154, nickel-63, or strontium-90) in the deep zone. These waste sites will decay to residential RBSLs within 2 and 185 years:

- 100-D-18 deep decision unit
- 100-D-19 deep focused decision unit
- 100-D-48:1 deep decision unit
- 100-D-48:2 deep decision unit
- 100-D-48:3 deep decision unit
- 100-D-49:2 deep decision unit
- 100-D-49:4 deep decision unit
- 116-D-1A deep decision unit
- 116-D-7 deep decision unit
- 116-DR-1&2 deep decision unit
- 116-DR-6 deep decision unit
- 116-DR-9 deep decision unit
- 118-D-6:4 deep decision unit
- 118-D-6:4 deep focused decision unit
- 118-DR-2:2 deep decision unit
- 116-H-1 deep decision unit
- 116-H-3 deep decision unit
- 116-H-7 deep decision unit
- 118-H-6:2, 118-H-6:3, 118-H-6:6, 100-H-9, 100-H-10, 100-H-11, 100-H-12, 100-H-13, 100-H-14, and 100-H-31 deep 2 decision unit
- 118-H-6:2, 118-H-6:3, 118-H-6:6, 100-H-9, 100-H-10, 100-H-11, 100-H-12, 100-H-13, 100-H-14, and 100-H-31 deep 3 decision unit

6.5.1.4 Deep Zone Results for RI and LFI Data

Soil samples were collected from depths greater than 4.6 m (15 ft) bgs from the following locations:

- Twelve RI soil borings (seven from 100-D Source OU and five from 100-H Source OU)
- Fifteen of the RI wells (seven from 100- D Source OU and eight from 100-H Source OU)
- Five RI test pits (three from 100-D Source OU and two from 100-H Source OU)
- Twenty-three LFI soil borings (18 from 100-D Source OU and five from 100-H Source OU)
- Six LFI wells (all from 100-H Source OU)

The conclusions from the evaluation of the deep zone RI and LFI data are consistent with the conclusions of the soil risk assessment.

For the 100-D Source OU, RI soil boring/well samples from 116-D-1B Trench (C7855), 116-D-7 Retention Basin (C7851), and the 118-D-6 Reactor Fuel Storage Basin (C7857) and 100-D RUM Well R5 redrill (C8668) report radionuclide concentrations greater than residential RBSLs. LFI soil boring samples from 116-D-1A Trench (199-D5-21), 116-D-1B Trench (199-D5-29), 116-DR-1&2 Trench (199-D8-62), 116-D-2 Crib (199-D5-22), 116-D-9 Crib (199-D5-26), and the 132-D-3 Pumping Station (199-D5-28) also report radionuclide concentrations greater than residential RBSLs. Radionuclide concentrations from each of the above soil borings were decayed to determine the year that activities would be reduced to levels less than the residential RBSL. The following summarizes the results of the comparisons for the previously listed waste sites:

- LFI data, CVP/RSVP closeout data, and RI soil boring data are available for the 116-D-1A Trench. This site is a potential source for groundwater contamination in the D northern Cr(VI) groundwater plume. The RI data indicate that individual risks from all detected analytes are less than the risk threshold of 1×10^{-4}. The results of the LFI data analysis and the risk assessment for the deep decision unit identify similar radioisotopes as contributors to risk (cesium-137, cobalt-60, europium-152, europium-154, and strontium-90). These radioisotopes are present at depths ranging between 5.2 and 16.2 m (17 and 53.2 ft) bgs. Concentrations of all isotopes decay to levels less than residential RBSLs between years 2174 and 2196.

- LFI and RI soil boring data are available for the 116-D-1B Trench. The results of the RI data analysis and the LFI data analysis identify similar radioisotopes as contributors to risk (cesium-137, cobalt-60, europium-152, europium-154, and strontium-90) and concentrations of all isotopes decay to levels less than residential RBSLs between years 2092 and 2177. Cesium-137 and europium-152 radioisotopes are present at depths ranging between 4.8 and 7.1 m (15.7 and 23.2 ft) bgs and strontium-90 is present at depths ranging between 6.3 and 15.8 m (20.7 and 52 ft) bgs in the RI soil boring. Cesium-137, cobalt-60, europium-152, europium-154, and strontium-90 are present at depths ranging between 4.3 and 8.1 m (14 and 26.7 ft) bgs. The 116-D-1B Trench is a consolidated waste site associated with the 116-D-1A Trench; therefore, the risk assessment results reported for the 116-D-1A Trench apply to the 116-D-1B Trench.

- LFI data, CVP/RSVP closeout data, and RI soil boring data are available for the 116-D-7 Retention Basin. The RI data analysis identifies cesium-137 as a contributor to risk and the risk assessment for the 116-D-7 waste site (deep decision unit) identifies cesium-137, cobalt-60, europium-152, europium-154, and nickel-63 as contributors to risk. Cesium-137 is present at depths ranging between 6.1 and 9.8 m (19.9 and 32 ft) bgs. The LFI data indicate that individual risks from all detected analytes are less than the risk threshold of 1×10^{-4}. Based on the results of the RI data analysis and

the risk assessment, radioisotope concentrations decay to levels less than residential RBSLs by year 2063 and year 2083, respectively.

- LFI data, CVP/RSVP closeout data, and RI soil boring data are available for the 116-DR-1&2 Trench. This site is historically a source for groundwater contamination in the D northern and horn Cr(VI) groundwater plumes. However, the RI data indicate that individual risks from all detected analytes remaining in the soil at the present day are less than the risk threshold of 1×10^{-4}. The results of the LFI data analysis and the risk assessment for the deep decision unit identify similar radioisotopes as contributors to risk (cesium-137, cobalt-60, europium-152, europium-154, and strontium-90). Concentrations of all isotopes decay to levels less than residential RBSLs between years 2163 and 2122. These radioisotopes are present at depths ranging between 4.5 and 6.8 m (14.8 and 22.3 ft) bgs.

- LFI data, CVP/RSVP closeout data, and RI soil boring data are available for the 116-DR-9 Retention Basin. The RI data and LFI data indicate that individual risks from all detected analytes are less than the risk threshold of 1×10^{-4}. The risk assessment for the 116-DR-9 waste site (deep decision unit) identifies cesium-137 and europium-152 as contributors to risk. Concentrations of these isotopes decay to levels less than residential RBSLs by year 2037.

- RI soil boring data are available for the 118-D-6 Reactor Fuel Storage Basin. The RI data analysis identifies cesium-137 as a contributor to risk. Concentrations of cesium-137 decay to activity levels less than the residential RBSL by year 2120. Cesium-137 is present at depths ranging between 5.9 and 8.2 m (19.5 and 27 ft) bgs.

- RI soil boring data are available for the 100-D RUM Well R5 Redrill. The RI data analysis identifies strontium-90 as a contributor to risk. Concentrations of strontium-90 decay to activity levels less than the residential RBSL by year 2012. Strontium-90 is present at depths ranging between 24 and 24.7 m (78.6 and 81.1 ft) bgs.

- LFI data and CVP/RSVP closeout data are available for the 116-D-2 Crib. The LFI data analysis identifies cesium-137 and strontium-90 as contributors to risk. Cesium-137 and strontium-90 were present at depths ranging between 5.2 and 6.1 m (17 and 20 ft) bgs. This waste site was subsequently remediated under the interim action ROD. The soil risk assessment did not identify risks associated with this site based on current conditions.

- LFI data and CVP/RSVP closeout data are available for the 116-D-9 Crib. The LFI data analysis identifies strontium-90 as a contributor to risk. Strontium-90 was present at depths ranging between 5.3 and 6.4 m (17.3 and 20.9 ft) bgs. The soil risk assessment did not identify risks associated with this site based on current conditions.

- LFI data are available for the 132-D-3 Pumping Station; soil samples were not collected from this site as part of the closeout documentation because this is a facility. The LFI data analysis identifies strontium-90 as a contributor to risk. Strontium-90 is present at depths ranging between 7.6 and 8.2 m (25 and 27 ft) bgs. Concentrations of strontium-90 decayed to activity levels less than the residential RBSL by year 1999.

- LFI data and CVP/RSVP closeout data are available for the 116-DR-7 Crib. The LFI data analysis identifies europium-152 as a contributor to risk. Europium-152 is present at depths ranging between 7.6 and 9.0 m (25 and 29.5 ft) bgs. Concentrations of strontium-90 decayed to activity levels less than the residential RBSL by year 2006. The soil risk assessment did not identify risks associated with this site based on current conditions.

For the 100-H Source OU, RI soil boring samples from the 116-H-1 Trench (C7864) report radionuclide concentrations greater than residential RBSLs. LFI soil boring samples from the 116-H-1 Trench (199-H4-58), the 116-H-7 Retention Basin (199-H4-61), and Well 199-H4-49 also report radionuclide concentrations greater than residential RBSLs. Radionuclide concentrations from each of the above soil borings were decayed to determine the year that activities would be reduced to levels less than the residential RBSL.

- LFI data, CVP/RSVP closeout data, and RI soil boring data are available for the 116-H-1 Trench. The RI and LFI data analysis identifies cesium-137 and europium-152 as contributors to risk. These radioisotopes are present at depths ranging between 4.6 and 6.6 m (15.1 and 21.6 ft) bgs. The risk assessment for 116-H-1 Trench waste site (deep decision unit) identifies cesium-137, europium-152, europium-154, and strontium-90 as contributors to risk. Concentrations of radioisotopes decay to activity levels less than the residential RBSL between year 2065 (RI and LFI data) and year 2101 (CVP/RSVP closeout data).

- LFI data, CVP/RSVP closeout data, and RI soil boring data are available for the 116-H-7 Retention Basin. The RI data indicate that individual risks from all detected analytes are less than the risk threshold of 1×10^{-4}. Europium-152 concentrations in LFI data are reported above the residential RBSL and are present at depths ranging between 4.5 and 5.0 m (14.8 and 16.4 ft) bgs. Concentrations of europium-152 in LFI data decayed to activity levels less than the residential RBSL in year 1994. The risk assessment for 116-H-7 Retention Basin waste site (deep decision unit) identifies cesium-137, cobalt-60, europium-152, europium-154, and strontium-90 as contributors to risk. Cesium-137, cobalt-60, europium-152, europium-154, and strontium-90 concentrations at 116-H-7 decay to levels less than residential RBSLs by year 2070.

- LFI data are available for 199-H4-49 monitoring well. The LFI data analysis identifies technetium-99 as a contributor to risk and is present at depths ranging between 10.7 and 11.4 m (35 and 37.5 ft) bgs. Concentrations of technetium-99 do not decay within a reasonable timeframe because the half-life for this isotope is 213,000 years.

The results from several of the waste sites are based on small datasets, which creates uncertainties in obtaining reliable EPCs in vadose zone material. The uncertainties relating to small datasets could result in risks either being over- or understated. EPCs selected for shallow zone and deep zone decision units represent verification data collected from the floor and sidewalls of the excavated waste site. EPCs developed from the floor and sidewalls of the excavated waste site overstate risk because the contaminant is assumed to be uniformly distributed across the entire decision unit, and exposure is assumed to occur at the surface. However, these sample locations are actually at depth and take no credit for the existing clean backfill that covers the remediated waste site. If the contaminants are disturbed in the future, their distribution within the decision unit would be blended with the clean backfill, resulting in an overall reduction of the EPC for the decision unit. The approach for identifying COPCs is conservative because it excludes few contaminants, and therefore probably overstates risks. The exposure factors and toxicity values used to develop the PRGs generally are conservative and tend to provide upper-bound estimates of risks in vadose zone material.

Based on the results of the soil risk assessment for the 100-D/H Source OU, cleanups in vadose zone material conducted as part of the interim actions appear to have been effective in reducing human health risks to within EPA's target risk range. In some cases, residual risks are higher than the State of Washington's cancer risk threshold; however, in all cases, the contaminant exceeding the State of Washington's cancer risk threshold is arsenic and is present at concentrations consistent with naturally occurring background. Cleanup of shallow vadose zone material (4.6 m [15 ft]) to achieve residential or

unrestricted uses is also protective of a range of exposure scenarios, including those for a casual recreational user and a resident Monument worker. Deep vadose zone samples (from remediated waste sites) were used to identify locations where institutional controls should be implemented to prevent in advertent exposure through deep excavation activities. While this exposure would be industrial in nature, the RBSLs (developed for the Residential exposure scenario) were used for convenience to identify sites where institutional controls should be established to control access to deep contamination. These sites do not pose significant risks because there is no current exposure pathway for deep contamination. Concentrations in deep vadose material will decay to the Residential RBSLs within185 years. In addition, data and process knowledge indicate that human health PRGs would be exceeded at unremediated waste sites and provides the basis for action. Table 8-6 provides the contaminants that are anticipated to exceed human health PRGs for unremediated waste sites.

6.5.2 Conclusions for the Groundwater Risk Assessment

The 100-HR-3 Groundwater OU was evaluated as three separate exposure areas including the 100-D Source, 100-H Source, and horn exposure areas. The 100-D Source exposure area represents the plume sources in 100-D, including the northern and southern Cr(VI) plumes. The 100-H Source exposure area represents the plume sources in 100-H. The horn exposure area represents the portion of the Cr(VI) plume that is located in the horn area where 100-D Sources have dispersed over time.

100-D Source Exposure Area. The contaminants in groundwater that are the largest contributors to calculated risks, dose, and HIs are Cr(VI) and total chromium, chloroform, and nitrate. The EPCs in groundwater were compared with AWQC and state water quality standards for protection of human health and aquatic organisms, federal and state primary and secondary MCLs, and state groundwater cleanup levels.

The EPC for nitrate is greater than the federal and state MCLs developed for the protection of human health. Nitrate is identified as a COPC, indicating the need to evaluate potential remedial technologies for these analytes in the FS.

Metals concentrations in groundwater higher than ambient water quality standards are chromium and Cr(VI). The EPCs for chromium and Cr(VI) are both higher than the AWQC for protection of aquatic receptors. In addition, the EPC for chromium is greater than the federal MCL. Therefore, the EPCs for both chromium species are greater than the MCL or AWQC developed for the protection of human health or aquatic organisms, indicating the need to evaluate potential remedial technologies for these analytes in the FS.

Strontium-90 was reported at a concentration above the DWS at well 199-D5-32, this is the only result reported at this well. Additionally, well 199-D5-12, located south of the 116-D-1A liquid waste stream, historically reported strontium-90 concentrations above the DWS until it was decommissioned in 2002. Strontium-90 is identified as a COPC, indicating the need to evaluate potential remedial technologies in the FS.

The EPC for chloroform is greater than the MTCA Method B groundwater cleanup level, which is based on a 1×10^{-6} target cancer risk level. However, the cumulative risk for chloroform is less than the MTCA (HHRA Procedures [WAC 173-340-708]) cumulative risk level of 1×10^{-5} for multiple contaminants. The EPC for chloroform is also less than federal and state MCLs and federal and state water quality criteria developed for the protection of human health or aquatic organisms. The results of this evaluation for chloroform do not indicate the need to evaluate potential remedial technologies in the FS.

Based on the results of the groundwater risk assessment, chromium, Cr(VI), and nitrate are identified as COCs in the 100-D Source exposure area and indicate the need to evaluate potential remedial technologies in the FS.

The COPC identification process identified eight analytes for the 100-D source exposure area that are considered COPCs and warrant further evaluation in the FS. The occurrence of antimony, cadmium, carbon tetrachloride, cobalt, copper, lead, nickel, and silver, and their nature and extent evaluation, indicates these analytes historically have been detected in groundwater at concentrations above their respective action level, but their presence was not associated with a specific location or a trend. Therefore, these analytes warrant further evaluation in the FS.

In addition to comparison to action levels, risks were evaluated using the Native American scenarios and the EPA Tap Water scenario. The total cumulative ELCRs for the 100-D Source exposure area for the CTUIR and Yakama Nation exposure scenarios are 8.9×10^{-4} and 9.3×10^{-4}, respectively, when groundwater is used as a drinking water source. The total cumulative ELCRs for both Native American scenarios are greater than the EPA upper target risk threshold of 1×10^{-4}. The primary contributors to risk for the CTUIR scenario are arsenic, carbon tetrachloride, chloroform, tetrachloroethene, strontium-90, and tritium. The primary contributors to risk for the Yakama Nation scenario are arsenic, carbon tetrachloride, chloroform, tetrachloroethene, strontium-90, and tritium. The total ELCR for the EPA Tap Water scenario is 2.0×10^{-4}, which is greater than the EPA upper target risk threshold of 1×10^{-4}. The primary contributors to risk for the Tap Water scenario are arsenic, carbon tetrachloride, chloroform, tetrachloroethene, and tritium. Arsenic is a primary contributor to risk for each of the scenarios (approximately 25 percent); however, levels of arsenic in groundwater are considered naturally occurring. The remaining analytes that are reported contribute less than 5 percent of the total cumulative risk. The total HI for the 100-D Source exposure area is 26 for both the CTUIR and Yakama Nation exposure scenarios. The HI for the EPA tap water equations is 14. Cr(VI) is the primary contributor to the noncancer HI for the Native American scenarios, as well as the primary contributor to the EPA Tap Water exposure scenario.

The cumulative ELCR is 5.0×10^{-1} for the CTUIR scenario and 7.6×10^{-1} for the Yakama Nation scenario when groundwater is used as a source of steam for a sweat lodge. The cumulative risk for the Native American scenarios is greater than the EPA upper target risk threshold of 1×10^{-4}. The individual ELCR values for cobalt and Cr(VI) are greater than the EPA upper target risk threshold of 1×10^{-4}. The HI for the 100-D Source exposure area is 99 for the CTUIR scenario and 205 for the Yakama Nation scenario when groundwater is used as a source of steam for a sweat lodge, which is greater than the EPA target HI of 1. The primary contributors to the noncancer HI are Cr(VI), cobalt, nickel, and barium.

100-H Source Exposure Area. The principal contaminants in groundwater are strontium-90 and Cr(VI). The EPCs in groundwater were compared with AWQCs and state water quality standards for protection of human health and aquatic organisms, federal and state primary and secondary MCLs, and state groundwater cleanup levels.

The EPC for strontium-90 is greater than the federal MCL developed for the protection of human health. Strontium-90 is identified as a COPC, indicating the need to evaluate potential remedial technologies for strontium-90 in the FS.

The EPC for Cr(VI) is greater than the AWQC developed for the protection of aquatic receptors. Cr(VI) is identified as a COPC, indicating the need to evaluate potential remedial technologies for Cr(VI) in the FS.

Although the EPC for nitrate is less than the MCL developed for the protection of human health, it is present at concentrations above the MCL in localized areas. Nitrate is identified as a COPC; its presence warrants design considerations for any engineered controls or remedial actions performed in this OU.

Based on the results of the groundwater risk assessment, the following COPCs are identified as COCs in the 100-H Source exposure area and indicate the need to evaluate potential remedial technologies in the FS: Cr(VI), nitrate, and strontium-90.

The COPC identification process identified nine analytes for the 100-H source exposure area that are not COPCs but that were moved forward into the FS because they have uncertainty associated with the dataset and their occurrence in groundwater. The analytes in the 100-H source exposure area include antimony, cadmium, carbon tetrachloride, chromium, cobalt, copper, nickel, silver, and zinc. The nature and extent evaluation indicates these analytes historically have been detected in groundwater at concentrations above their respective action level, but their presence was not associated with a specific location or a trend. Therefore, these analytes warrant further evaluation in the FS.

In addition to comparison to action levels, risks were evaluated using the Native American scenarios and the EPA Tap Water scenario. The total cumulative ELCRs for the 100-H Source exposure area for the CTUIR and Yakama Nation exposure scenarios are 4.0×10^{-4} and 4.3×10^{-4}, respectively, when groundwater is used as a drinking water source. The total cumulative ELCRs for both Native American scenarios are greater than the EPA upper target risk threshold of 1×10^{-4}. The primary contributors to risk for the CTUIR scenario are arsenic, carbon tetrachloride, chloroform, strontium-90, technetium-99, and tritium. The primary contributors to risk for the Yakama Nation scenario are arsenic, carbon tetrachloride, chloroform, strontium-90, technetium-99, and tritium. The total ELCR for the EPA Tap Water scenario is 9.8×10^{-5}, which is within the EPA range of 1×10^{-4} to 1×10^{-6}. The primary contributors to risk for the Tap Water scenario are arsenic, carbon tetrachloride, chloroform, strontium-90, and tritium. Arsenic is a primary contributor to risk for each of the scenarios (approximately 75 percent); however, levels of arsenic in groundwater are considered naturally occurring. The remaining analytes that are reported contribute less than 25 percent of the total cumulative risk. The total HI for the 100-H Source exposure area is 3.3 for both the CTUIR and Yakama Nation exposure scenarios. The HI for the EPA tap water equations is 1.6. No individual COPC in the 100-H Source exposure area had a HQ greater than the EPA target HI of 1 for the Native American scenarios or the EPA Tap Water exposure scenario.

The cumulative ELCR is 1.8×10^{-2} for the CTUIR scenario and 3.7×10^{-2} for the Yakama Nation scenario when groundwater is used as a source of steam for a sweat lodge. The cumulative risk for the Native American scenarios is greater than the EPA upper target risk threshold of 1×10^{-4}. The individual ELCR value for Cr(VI) is greater than the EPA upper target risk threshold of 1×10^{-4}. The HI for the 100-H Source exposure area is 13 for the CTUIR scenario and 28 for the Yakama Nation scenario when groundwater is used as a source of steam for a sweat lodge, which is greater than the EPA target HI of 1. The primary contributors to the noncancer HI are Cr(VI), cobalt, nickel and barium.

Horn Exposure Area. The principal contaminants in groundwater are chromium and Cr(VI). The EPCs in groundwater were compared with AWQC and state water quality standards for protection of human health and aquatic organisms, federal and state primary and secondary MCLs, and state groundwater cleanup levels.

Metals concentrations in groundwater higher than ambient water quality standards are chromium and Cr(VI). The EPCs for chromium and Cr(VI) both are higher than the AWQC for protection of aquatic receptors. Chromium and Cr(VI) are COPCs, indicating the need to evaluate potential remedial technologies for chromium and Cr(VI) in the FS.

Based on the results of the groundwater risk assessment the following COPCs are identified as COCs and indicate the need evaluate potential remedial technologies in the FS: chromium and Cr(VI).

The COPC identification process identified seven analytes for the horn exposure area that are not COPCs but were moved forward into the FS because they have uncertainty associated with the dataset and their occurrence in groundwater. The analytes in the horn exposure area include antimony, cadmium, cobalt, copper, nickel, silver, and zinc. The nature and extent evaluation indicates these analytes historically have been detected in groundwater at concentrations above their respective action level, but their presence was not associated with a specific location or a trend. Therefore, these analytes warrant further evaluation in the FS.

In addition to comparison to action levels, risks were evaluated using the Native American scenarios and the EPA Tap Water scenario. The total cumulative ELCRs for the horn exposure area for the CTUIR and Yakama Nation exposure scenarios are 5.7×10^{-4} and 6.2×10^{-4}, respectively, when groundwater is used as a drinking water source. The total cumulative ELCRs for both Native American scenarios are greater than the EPA upper target risk threshold of 1×10^{-4}. The primary contributors to risk for the CTUIR scenario are 1,2-dichloroethane, arsenic, carbon tetrachloride, chloroform, tetrachloroethene, trichloroethene, strontium-90, and tritium. The primary contributors to risk for the Yakama Nation scenario are 1,2-dichloroethane, arsenic, carbon tetrachloride, chloroform, tetrachloroethene, trichloroethene, strontium-90, and tritium. The total ELCR for the EPA Tap Water scenario is 1.5×10^{-4}, which is greater than the EPA upper target risk threshold of 1×10^{-4}. The primary contributors to risk for the Tap Water scenario are 1,2-dichloroethane, arsenic, carbon tetrachloride, tetrachloroethene, trichloroethene, and tritium. Arsenic is a primary contributor to risk for each of the scenarios (approximately 85 percent); however, levels of arsenic in groundwater are considered naturally occurring. The remaining analytes that are reported contribute less than 15 percent of the total cumulative risk. The total HI or the horn exposure area is 4.5 for both the CTUIR and Yakama Nation exposure scenarios. The HI for the EPA tap water equations is 2.3. Cr(VI) is the primary contributor to the noncancer HI for the Native American scenario. No individual COPCs had a HQ greater than the EPA target HI of 1 for the EPA Tap Water exposure scenario.

The cumulative ELCR is 4.9×10^{-2} for the CTUIR scenario and 9.9×10^{-2} for the Yakama Nation scenario when groundwater is used as a source of steam for a sweat lodge. The cumulative risk for the Native American scenarios is greater than the EPA upper target risk threshold of 1×10^{-4}. The individual ELCR value for Cr(VI) is greater than the EPA upper target risk threshold of 1×10^{-4}. The HI for the horn exposure area is 14 for the CTUIR scenario and 29 for the Yakama Nation scenario when groundwater is used as a source of steam for a sweat lodge, which is greater than the EPA target HI of 1. The primary contributors to the noncancer HI are Cr(VI) and barium.

The key uncertainties in the assessment of groundwater risks are with the assessment of dermal contact exposure pathways, selection of the toxicity value for carbon tetrachloride, and recent developments with the toxicity value for Cr(VI). The evaluation of potential risks from VOCs is based on ingestion and inhalation exposure pathways and does not consider exposure through dermal contact with water. Not including the dermal contact exposure pathway potentially results in risks from these contaminants being understated. The cancer slope factor used by the State of Washington to develop the Method B groundwater cleanup level for carbon tetrachloride has not yet been updated to reflect current agency guidelines. Use of this toxicity value overstates risks from carbon tetrachloride by approximately a factor of two fold. Use of a current cancer slope factor would decrease the groundwater risk from carbon tetrachloride from 3.8×10^{-6} to 2.1×10^{-6}. Ingestion exposure to Cr(VI) is currently assessed as a noncarcinogen for purposes of developing groundwater cleanup levels for protection of human health, and Cr(VI) currently does not have a federal MCL. However, some state agencies, particularly the NJDEP, have developed a cancer slope factor for Cr(VI). Assessing ingestion of Cr(VI) in groundwater as a carcinogen is not yet incorporated into regulatory requirements or guidance at this time; however, groundwater standards for protection of human health for Cr(VI) would be considerably lower if these were based on carcinogenic effects.

The results from the groundwater risk assessment were based on three additional rounds of groundwater sampling across the 100-HR-3 Groundwater OU, which were intended to provide a more definitive identification of COPCs. The results of this groundwater risk assessment did not identify any COPCs in addition to those identified in the work plan. The results of the groundwater risk assessment identified total chromium, Cr(VI), strontium-90, and nitrate as contaminants warranting further evaluation in the FS.

7 Ecological Risk Assessment

The integration of past and ongoing ERAs supports the development of remedial alternatives for waste sites and contaminated groundwater in the 100-D/H OUs. These risk assessments have been integrated with the cleanups performed under the interim action RODs to identify the need for further remedial action and development of ecological PRGs.

Highlights

- The ERA evaluated soil contaminant concentrations at 142 interim closed and no action waste sites.
- The ERA relied on ecological PRGs presented in the RCBRA that are protective of populations and communities. The exposure area and the relative size of the waste sites were used in conjunction with the ecological PRGs to determine where ecological protection is required.
- Concentrations of radionuclides in upland soil verification samples did not exceed screening levels.
- Interim remedial actions at 100-D/H under interim action ROD remedial action goals were protective of ecological receptors at all waste sites.
- An examination of the interrelationships between potential contaminant sources, transport mechanisms, exposure pathways, and receptors in the Columbia River concluded that chromium and Cr(VI) in the 100-HR-3 Groundwater OU contribute to potential ecological risks.
- Data and process knowledge indicate ecological PRGs will be exceeded at unremediated waste sites. Those exceedances will be evaluated through the ERA process, including consideration of waste site size and wildlife home ranges within a scientific management decision point to determine a basis for action.

As described in the previous chapters, the remedial actions completed to date in the River Corridor were implemented under interim action RODs. The RAOs in the 100 Area interim action RODs were developed to protect human health from direct contact with vadose zone material or to protect groundwater and surface water from contaminants leaching from vadose zone material. Protection of ecological receptors from direct contact with contaminated vadose zone material was not addressed directly in the interim action RODs, but indirectly with the assumptions that attainment of standards for protection of human health or that reduced contaminant leaching would also be protective of ecological receptors. Protection of ecological receptors from discharges into the river was considered in the interim action RODs through consideration of state water quality standards and federal ambient water quality criteria.

CERCLA requires a baseline risk assessment to characterize current and potential threats to human health and the environment before issuance of the ROD. The source and groundwater component of the RCBRA (DOE/RL-2007-21)[1] was prepared to address the regulatory requirement to perform a baseline risk assessment. The RCBRA (DOE/RL-2007-21) was a comprehensive examination of current and potential risks in areas potentially affected by Hanford Site processes within the 100 Area and 300 Area OUs. One of the objectives of the RCBRA (DOE/RL-2007-21) was to determine whether the interim actions were protective of ecological receptors (*Risk Assessment Work Plan for the 100 Area and 300 Area Component of the RCBRA* [DOE/RL 2004-37]). The scope of the RCBRA (DOE/RL-2007-21) addressed the following portions of the River Corridor:

- Upland areas, including remediated CERCLA waste sites within 100-K, 100-D, 100-F, 100-H, 100-BC, and 100-N Areas; the White Bluffs and Hanford Townsites; and the 300 Area.
- Riparian and near-shore aquatic zones on the southern and western shorelines of the Columbia River on the Hanford Site.

[1] All citations to the RCBRA (DOE/RL-2007-21) in this chapter are referring to *Volume I: Ecological Risk Assessment*.

- Groundwater and areas of groundwater emergence on the southern and western shorelines of the Columbia River on the Hanford Site

The RCBRA (DOE/RL-2007-21) used multiple measures of exposure, ecological effect, and ecosystem/receptor characteristics to evaluate risks at 20 study sites across the River Corridor associated with remediated waste sites (10 excavated/backfilled sites and 10 surface removal/native soil sites) and 10 reference areas, as described in the RCBRA SAP (DOE/RL-2005-42). The sites studied were selected from high-priority waste sites that had been remediated when the study was developed and represent the types of waste sites and remedial actions addressed by interim action RODs. Based on this set of study sites, the results from the RCBRA (DOE/RL-2007-21) identified contaminants in soil as contaminants of ecological concern (COECs). The principal COECs were metals and pesticides.

The study design of the ERA in the RCBRA (DOE/RL-2007-21) provided risk conclusions that applied across the entire River Corridor. The study design, coupled with results that identified COECs across the River Corridor, required development of an ERA approach for the RI/FS that allowed evaluation of risks on a site-by-site basis as well as supported development of PRGs. That approach incorporates the use of ecological SSLs and ecological PRGs, which have been developed using the tiered process outlined in *Tier 1 Risk-Based Soil Concentrations Protective of Ecological Receptors at the Hanford Site* (CHPRC-00784) and *Tier 2 Risk-Based Soil Concentrations Protective of Ecological Receptors at the Hanford Site* (CHPRC-01311), respectively, found in Appendix H of the 100-K RI/FS. This tiered process allows the incorporation of more sophisticated ERA methods and increasing levels of ecological site-specific and site relevant information to provide SSLs and PRGs that are more representative of Hanford Site conditions. Development of the risk-based concentration values (SSLs) and PRGs incorporates the problem formulation, the conceptual ecological exposure models, and selected bioaccumulation datasets developed in the RCBRA (DOE/RL-2007-21). These values were used to screen the 75 waste sites in the 100-D (100-DR-1 and 100-DR-2) OUs and 36 waste sites in the 100-H (100-HR-1 and 100-HR-2) OUs, with verification sampling and analytical information, to provide site-specific ecological risk information for each site.

The CRC (DOE/RL-2010-117)[2] used analytical chemistry collected from surface water, sediment, pore water, and island soil to evaluate the potential for risk to ecological receptors including aquatic life living within the Columbia River and wildlife frequenting or inhabiting the islands within the river. Based on a screening-level ERA using refined toxicity and distributional data, the CRC (DOE/RL-2010-117) identified contaminants in soil as contaminants of potential ecological concern (COPECs). COPECs principally were metals. The potential for these contaminants to have originated from 100-D or 100-H is discussed later in this chapter. Three of the 75 waste sites in the 100-D Source OU and 5 of the 36 waste sites in the 100-H Source OU report only deep-zone data and therefore are not included in the evaluation.

The following approach has been used for addressing ecological risks potentially associated with waste sites in the 100-D and 100-H OUs:

- **Updating the identification of COPCs (Section 7.1).** The RCBRA (DOE/RL-2007-21) went through a process to identify COPCs for ecological receptors based on a sitewide review of River Corridor data. This identification process has been updated to account for verification sampling data specifically in individual 100-D/H waste sites.

[2] All citations to the CRC (DOE/RL-2010-117) in this chapter are referring to *Volume I: Screening-Level Ecological Risk Assessment*.

- **Presenting the problem formulation (Section 7.2).** This section summarizes the problem formulation used in developing the risk-based concentration values used in this ERA as ecological SSLs. This problem formulation reflects conditions in upland environments across the Hanford Site and incorporates information developed from the RCBRA (DOE/RL-2007-21).

- **Presenting effects and exposure assessments (Section 7.3).** This section summarizes the quantitative assessments used in developing the Tier 1 and Tier 2 risk-based concentration values, including the wildlife exposure factors, biotransfer factors, and wildlife toxicity reference values (TRVs) (Appendix H, Tables H-1 and H-2). The data and methods used to develop risk-based concentrations protective of plants and soil invertebrates are discussed in this section. More detailed descriptions of the data and methods used to calculate all of the ecological risk-based concentrations in soil are presented in *Tier 1 Risk-Based Soil Concentrations Protective of Ecological Receptors at the Hanford Site* (CHPRC-00784) and *Tier 2 Risk-Based Soil Concentrations Protective of Ecological Receptors at the Hanford Site* (CHPRC-01311). In addition, these values are incorporated into the RCBRA (DOE/RL-2007-21).

- **Updating the ecological risk characterization for 100-D/H waste sites (Section 7.4).** Verification sampling and analysis data for the 95 waste sites in the 100-D and 47 waste sites in the 100-H were used to calculate EPCs, which were then compared with the ecological SSLs and, as appropriate, the PRGs. The results from these comparisons were used to identify receptors of interest and COECs to determine the need for further action at 100-D/H sites. In addition, the results of this risk characterization were used to determine which of the risk-based concentration values should be recommended for use as PRGs.

- **Analyzing risks in the riparian and near-shore areas, and the Columbia River (Section 7.5).** Final recommended COECs in riparian and island soil and the surface water and sediments of the Columbia River as identified in the RCBRA (DOE/RL-2007-21) and CRC (DOE/RL-2010-117) were evaluated as to the potential for attribution to the 100-D/H near-shore area.

- **Presenting preliminary scientific management decision point (SMDP) (Section 7.6).** Potential risks identified through the direct comparison of verification sampling soil data to SSLs and PRGs were considered in the context of additional factors. Uncertainties in the risk characterization, spatial information, data quality, magnitude and aerial extent of risk, and confidence in risk-based values were included with other factors to make recommendations of which, if any, risks should be addressed further in the FS. The process for developing final remediation goals was also discussed along with recommendations for the SMDP for evaluating waste sites as follows:
  - Size of the waste site relative to the home range of wildlife receptors (for example, developing and applying an area use factor [AUF] in the comparison of an EPC to the PRGs)
  - Estimation of exposure using a central tendency estimate such as the 95 percent UCL
  - Size of the waste site relative to the area of adjacent uncontaminated habitat
  - Nature and extent of residual contamination following remediation
  - Potential presence of exposure pathways following remediation
  - Number and frequency of exceedances of the risk thresholds (PRGs)

- Location of the samples exceeding thresholds, sample frequency, and proximity of other exceedances
- Depth at which exceedances of the risk thresholds (PRGs) occur

Section 6.4 evaluates the protection of aquatic receptors from groundwater that has the potential to discharge to the Columbia River. The approach used to identify COPCs that warrant further evaluation in the FS presented in Section 6.4 is based on comparison of groundwater concentrations to the lowest available chemical-specific ARARs published for the protection of human health and aquatic receptors. Thus, risks to aquatic receptors have been considered in the context of evaluating the risks groundwater may contribute to surface water at the groundwater/surface-water interface. Combining the evaluation of human health provides a streamlined approach that addresses the restoration of groundwater and the protection of aquatic receptors.

In addition to the analysis of waste sites, Chapter 7 summarizes an evaluation of ecological risks in riparian and near-shore areas based on the analysis developed in the RCBRA (DOE/RL-2007-21) and risk in the Columbia River developed for the CRC (DOE/RL-2010-117). Appendix L evaluates ecological risks identified within the Columbia River and the relationship among potential sources to the Columbia River in the 100-D/H OUs, transport pathways, and ecological receptors. The RCBRA (DOE/RL-2007-21) evaluated risks to an array of assessment endpoints using multiple measures of exposure, effect, and ecosystem/receptor characteristics at representative near-shore study sites. The study sites were selected to represent locations that may be adjacent to or directly affected by known contaminated media (groundwater seeps and springs, soil, sediment). The RCBRA (DOE/RL-2007-21) has been supplemented through the development of a conceptual model depicting the relationships among sources in the 100-D/H OUs and riparian and near-shore media (soil, sediment, pore water, and surface water). This conceptual model is presented as Appendix L.

7.1 Identification of Contaminants of Potential Concern

This section describes the sources of data used in the ERA, the DQA and data validation process, and the process for identifying COPCs in soil. CVP and RSVP data collected within 95 waste sites in the 100-D Source OU and 47 waste sites in the 100-H Source OU were used to identify COPCs. This chapter presents the risk assessment for individual waste sites using CVP/RSVP data. During this ERA, COPCs were examined to identify a refined list of COPECs estimated to pose site-related ecological risks to receptor populations.

7.1.1 Data Summary

Remediation of waste sites in the 100-D/H Source OUs began in 1996. Ninety-five 100-D Source OU waste sites have verification sampling data and are included in this soil risk assessment. Twenty-eight of these 100-D Source OU waste sites were evaluated in the RCBRA (DOE/RL-2007-21). An additional thirteen 100-D Source OU sites, referred to as associated waste sites, have been remediated, but are included in another waste site's sampling and closeout documentation.

Forty-seven 100-H Source OU waste sites have verification sampling and analysis data and are included in this soil risk assessment. Eight of these thirty-six 100-H Source OU waste sites were evaluated in the RCBRA (DOE/RL-2007-21). An additional ten 100-H Source OU sites, referred to as consolidated sites, have been remediated but are included in another waste site's sampling and closeout documentation. A summary of the waste sites, associated decision unit(s), and reclassification status for the 100-D and 100-H Source OUs is provided in Tables G-1 and G-2, respectively (Appendix G). Waste site decision units are defined in Section 6.2.2.2. The waste sites listed in Tables G-1 and G-2 (Appendix G) are

a subset of the waste sites that were listed in Appendix C, Table C-1, of the 100-D/H Work Plan (DOE/RL-2008-46-ADD1).

7.1.2 Data Quality Assessment

A DQA is performed and reported in each closeout documentation report. The DQA compares the verification sampling approach and resulting analytical data with the sampling and data quality requirements specified by the project objectives and performance specifications. The DQA determines whether the data are of the right type, quality, and quantity to support site cleanup verification decisions within specified error tolerances. The DQA also determines whether the analytical data are acceptable for decision-making purposes and whether the sample design was sufficient for clean site verification. The cleanup verification sample analytical data and detailed DQA are summarized in the appendices associated with the CVPs. The results of each DQA are incorporated by reference, and no further DQA was performed as part of this risk assessment.

All of the analytical data are evaluated and a portion is validated for compliance with quality assurance project plan requirements as documented in the 100 Area RDR/RAWP (DOE/RL-96-17). Data evaluation is performed to determine whether the laboratory carried out all steps required by the 100-D/H SAP (DOE/RL-2009-40) and the laboratory contract governing the conduct of analysis and reporting of the data. This evaluation also examines the available laboratory data to determine whether an analyte is present or absent in a sample and the degree of overall uncertainty associated with that determination.

7.1.3 Identification of COPCs

All analytes detected at least once in a waste site decision unit for the waste sites in the 100-D/H OUs, included in the risk assessment, are identified as COPCs except those exclusions described below. Verification sampling and analysis data are collected according to sample design requirements for the type of decision unit. For this ERA, an "exposure area" and a "decision unit" are operationally defined as being the same. Verification sampling and analysis data are subsequently grouped to calculate EPCs.

The contribution from naturally occurring metals and anthropogenic radioisotopes is discussed in the risk characterization section in accordance with *Guidance for Comparing Background and Chemical Concentrations in Soil for CERCLA Sites* (EPA 540-R-01-003). The risk characterization discusses elevated background concentrations and their contribution to site risks and naturally occurring elements that are not CERCLA hazardous substances, pollutants, and contaminants, but exceed the risk-based screening levels.

The RCBRA (DOE/RL-2007-21) identifies a subset of analytes excluded from consideration as COPCs by agreement among the Tri-Parties based on relevant Hanford Site data. The following exclusion list used in the RCBRA (DOE/RL-2007-21) was also applied to the waste site verification data during the data reduction steps described in Section 6.2.2.2:

- **Radionuclides with a half-life of less than 3 years:** Radionuclides with half-lives less than 3 years would not result from historical Hanford Site operation because radioactive decay would have occurred since operations ceased.
- **Essential nutrients:** Essential nutrients present at relatively low concentrations and toxic only at high concentrations were not considered in the quantitative risk assessment.
- **Water quality or soil physical property measurements:** These analytes were measured only to obtain information on water quality or soil properties to understand potential confounding factors for bioassays conducted for soil, sediment, or water or to interpret their influence on the toxicity of COPCs (for example, grain size for soil, water hardness for metal effects).

- **Background radionuclides (potassium-40, radium-226, radium-228, thorium-228, thorium-230, and thorium-232):** As identified and implemented in the RCBRA (DOE/RL-2007-21), these background radionuclides were identified by consensus of Tri-Party managers as not directly related to Hanford Site operations or processes.

A list of the analytes that meet the exclusion criteria for the soil risk assessment are listed in Section 6.2.2.2 and presented in Appendix G (Table G-3). The RCBRA (DOE/RL-2007-21) includes the following two additional steps to identify COPCs that the soil risk assessment did not apply:

- Analytes commonly reported in waste site cleanup verification reports based on frequency of detection. Inclusion list analytes were not consistently reported in the CVP and RSVP data; therefore, this step was not implemented.

- Remaining analytes evaluated as candidate COPCs, based on comparisons to Hanford Site background, reference areas, and an analyte-specific evaluation.

As a result of not applying the last two steps used in the RCBRA (DOE/RL-2007-21) to identify COPCs, more analytes are identified as COPCs in this risk assessment than were identified in the RCBRA (DOE/RL-2007-21). Identifying all detected analytes (except those on the exclusion list) as COPCs is a more streamlined approach consistent with *Guidance for Comparing Background and Chemical Concentrations in Soil for CERCLA Sites* (EPA 540-R-01-003).

In addition to the steps described above, aluminum and iron were excluded as COPCs for all decision units within the 100-D/H Area OUs. The EcoSSLs for aluminum and iron are based on soil pH (*Ecological Soil Screening Level for Aluminum: Interim Final* [OSWER Directive 9285.7 60] and *Ecological Soil Screening Level for Iron: Interim Final* [OSWER Directive 9285.7-69). The potential for aluminum toxicity is only identified in soils when the pH is 5.5 or less. While iron is essential for plant growth and is generally considered to be a micronutrient (*Soils and Soil Fertility* [Thompson and Troeh, 1973]), the potential for iron bioavailability is only identified when the pH is less than 5 or greater than 8 (*Ecological Soil Screening Level for Iron: Interim Final* [OSWER Directive 9285.7-69]). Oxidized environments (upland or well-aerated soils, such as those at the Hanford Site) promote the precipitation of ferric-oxide compounds, which are not available to plants for uptake. The main concern from an ecological risk perspective for iron is not direct chemical toxicity per se, but the effect of iron as a mediator in the geochemistry of other (potentially toxic) metals and the potential physical hazard of depositing flocculent (*Ecological Soil Screening Level for Iron: Interim Final* [OSWER Directive 9285.7-69]). These other COPECs are being evaluated with the screening levels identified in Section 7.3. While some individual measurements of pH indicate potential bioavailability of aluminum and iron, most measurements of soil pH in the River Corridor are within the ranges where these chemicals are not bioavailable. Data collected during the 2011 Hanford-wide field study indicated that pH in soils range between 5.8 and 8.7 (*Tier 2 Terrestrial Plant and Invertebrate Preliminary Remediation Goals (PRGs) for Nonradionuclides for Use at the Hanford Site* [ECF-HANFORD-11-0158]). Thus, aluminum and iron concentrations are not bioavailable and do not pose a risk to terrestrial ecological receptors.

The COPC list for these OUs was evaluated to develop a COPEC list in this risk assessment. A COPEC is defined as a COPC with concentrations exceeding both the background concentration and ecological screening level. The process to identify COPECs is discussed in Section 7.4.

7.2 Problem Formulation

The problem formulation includes the physical layout of the site, its history and ecology, and the development of an ecological conceptual site exposure model that evaluates potential exposure pathways

and identifies the representative species used to assess ecological risk to those and other similar species. The problem formulation includes identification of the important aspects of the 100-D/H Source OUs waste site decision units to be protected (referred to as assessment endpoints) and the means by which the assessment endpoints are evaluated (measures of exposure and effects).

7.2.1 Site Setting

The 100-D/H Area is in the northern portion of the Hanford Site adjacent to the Columbia River. The 100 Area reactors and associated facilities are on steep bluffs overlooking the river. The topography of the 100-D/H OU area is relatively flat inland from the Columbia River. The area has been disturbed and graded extensively by human activity since reactor construction began in the 1940s and through present-day waste site remedial activities. The surface elevation ranges from approximately 116 m (380 ft) above mean sea level at the Columbia River to 135 m (443 ft) above mean sea level on the eastern edge of 100-D. The upland environment is described in this section. The riparian and near-shore habitats are described in Appendix L, which evaluates the potential for exposures in the riparian and near-shore environments to be of concern and to have originated from 100-D or 100-H OU waste sites.

The predominant plant community in the 100 Area is sagebrush/Sandberg's bluegrass/cheatgrass. Currently, no plant species on the Hanford Site are federally listed as threatened or endangered under the *Endangered Species Act of 1973*. Plant species listed as threatened or endangered by Washington State include the awned halfchaff sedge (*Lipocarpha aristulata*), grand redstem (*Ammannia robusta*), lowland toothcup (*Rotala ramosior*), and persistentsepal yellowcress (*Rorippa columbiae*). These plant species are restricted to wetlands in the riparian zone of the Columbia River (NEPA Characterization Report [PNNL-6415]). Table 3-22 presents the complete list of state-listed flora.

Shrub and grassland habitats dominate the Hanford Site and support a diverse and abundant variety of wildlife species, including in the uplands of the River Corridor. The 100 Areas are mostly undisturbed or fully recovered and thus support these diverse and abundant wildlife communities. Wildlife use of the remaining disturbed and developed areas is expected to be reduced because these areas are less attractive and provide fewer of the needs of wildlife than do natural habitats. However, even these areas can be frequented by wildlife. Common species include large animals like Rocky Mountain elk (*Cervus elaphus*) and mule deer (*Odocoileus hemionus*); predators such as coyote (*Canis latrans*), bobcat (*Lynx rufus*), and badger (*Taxidea taxus*); and herbivores including deer mice (*Peromyscus maniculatus*), harvest mice (*Riethrodontonomys megalotis*), ground squirrels (*Spermophilus* spp.), voles (*Lemmiscus curtatus, Microtus* spp.), and black-tailed jackrabbits (*Lepus californicus*). The most abundant mammal on the Hanford Site is the Great Basin pocket mouse (*Perognathus parvus*). Other nonburrowing animals including cottontails (*Sylvilagus nutalli*), jackrabbits, snakes, and burrowing owls (*Athene cunicularia*) may use abandoned burrows of other animals.

No species that regularly frequent the Hanford Site are listed as threatened or endangered under the *Endangered Species Act of 1973*. Species listed as threatened or endangered by Washington State include the burrowing owl (*Athene cunicularia*), Merriam's shrew (*Sorex merriami*), and Washington ground squirrel (*Urocitellus washingtoni*). However, no species are known or expected to occur onsite because of the highly developed nature of this area. Table 3-23 presents the complete list of state listed fauna. Fauna previously identified at the site are listed in Appendix H, Attachment H-1.

Although the bald eagle has been removed from the list of federally endangered species, it is still protected under the *Bald and Golden Eagle Protection Act of 1940*. In addition, DOE continues to protect nest and roost sites on the Hanford Site under the *Bald Eagle Site Management Plan for the Hanford Site, South-Central Washington* (DOE/RL-94-150). Changes have been made to reduce the buffer zones surrounding winter night roosts and nest sites from 800 to 400 m (875 to 437.5 ft).

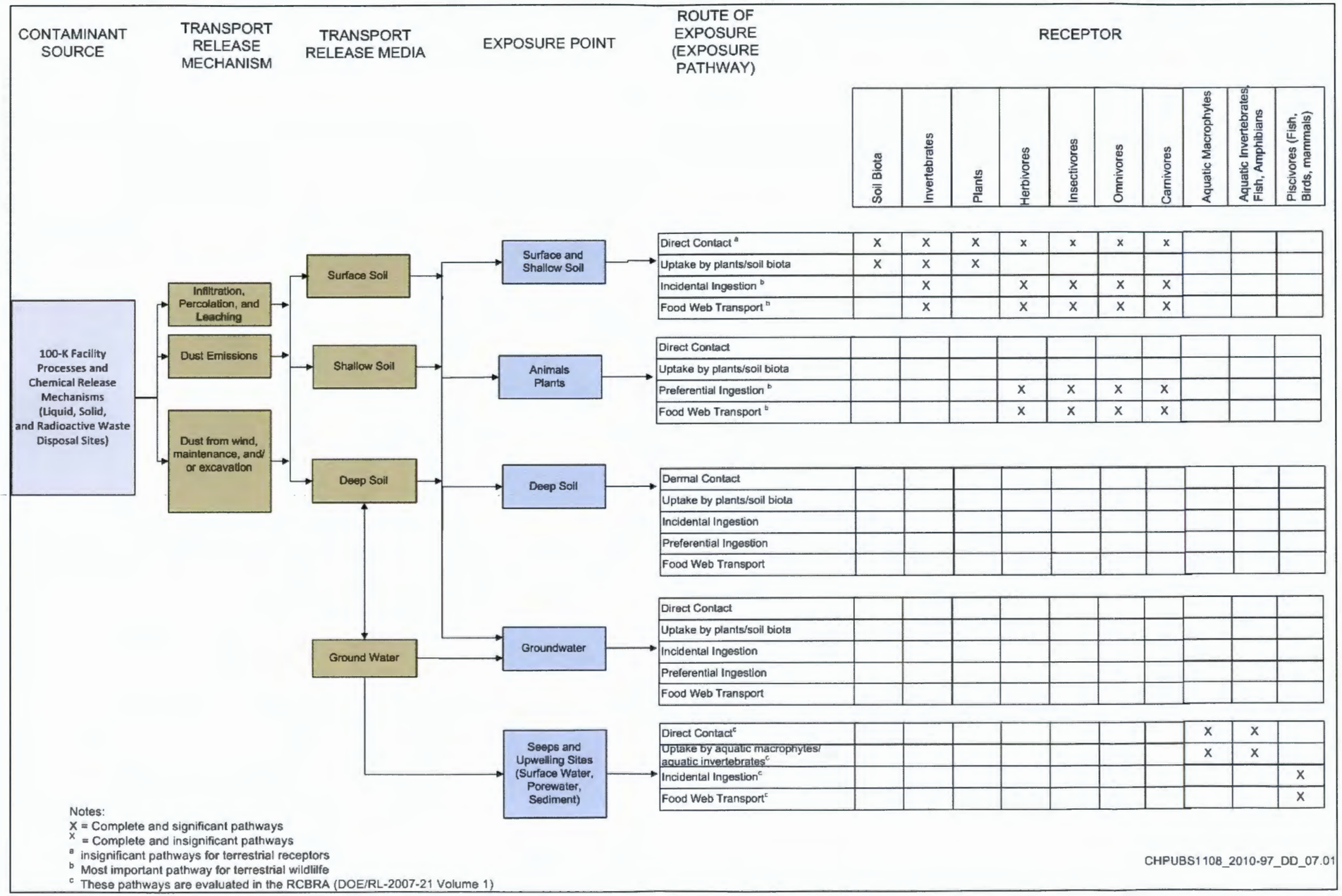
Bald eagles have generally been observed at the Hanford Site from November to March ("A Congregation of Wintering Bald Eagles" [Fitzner and Hanson, 1979]). During daylight hours, bald eagles perch along the Hanford Reach of the Columbia River and a few kilometers inland (*Bald Eagle Site Management Plan for the Hanford Site, South-Central Washington* [DOE/RL-94-150]). The primary perching areas occur in trees from the Hanford Townsite to the Vernita Bridge. Bald eagles predominantly forage on the banks of the river and the island where waterfowl roost and salmon carcasses are found. Two roosting sites are in this same area (*Bald Eagle Site Management Plan for the Hanford Site, South-Central Washington* [DOE/RL-94-150]). Although these areas along the Columbia River are primarily between 100-D and 100-H, additional consideration of these species is not required for this risk assessment. Additional discussion on site setting and site history is included in Sections 3.10 and 1.2, respectively.

7.2.2 Simplified Ecological Exposure Model for Upland Sites

Development of the ecological exposure model for this ERA involved characterizing the exposure pathways and ecological receptors associated with the habitat types in the upland environment of the waste sites within the 100-D/H OUs. Appropriate exposure pathways and representative endpoint species for the upland environment of the 100-D/H OUs were developed based on information from the RCBRA (DOE/RL-2007-21) and are discussed below. A full risk assessment of the riparian area or the islands within the Columbia River are not presented because they were already completed for the RCBRA (DOE/RL-2007-21) and CRC (DOE/RL-2010-117); however, the same models and receptors were used here as in those documents. The aquatic exposure models are described in Section L.2.4 (Appendix L) with the evaluation of the aquatic exposure pathways. Results of those exposure and effects evaluations (that is, the risk characterization) are discussed in Chapter 4, Appendix L, and Section 7.6 of this chapter with respect to the potential for the 100-D/H Source OUs to contribute to the final identified risks. With consideration of the ecological setting, land use, and COPC release mechanisms and transport pathways known at the 100-D/H Source OUs upland environments, the ecological exposure pathways considered most plausible are shown on Figure 7-1 and include the following:

- Direct contact of vegetation with analytes in surface soil.
- Direct contact with, or ingestion of, surface soil by terrestrial invertebrates (for example, beetles and ants).
- Direct contact with, or ingestion of, surface soil by terrestrial avian and mammalian wildlife.
- Dietary exposure of terrestrial and mammalian wildlife to COPCs bioaccumulated in food items (for example, plants or prey).
- Dietary exposure to emissions from radionuclides bioaccumulated and retained within the tissues of plants, terrestrial invertebrates, and terrestrial wildlife.
- External exposure of plants, terrestrial invertebrates, and terrestrial wildlife to emissions from radionuclides in soil.
- Ecological receptors are not likely to have complete exposure pathways to soil below the biologically active zone. Therefore, deep soil was not evaluated in this ERA.

A food web model for the upland environment of the Hanford Site (Figure 7-2) was developed based upon an understanding of the ecology of the area and documented in the previous ERAs.



Source: DOE/RL-2007-21, River Corridor Baseline Risk Assessment, Volume I: Ecological Risk Assessment.

Figure 7-1. Ecological Conceptual Site Model for Terrestrial Habitats in Hanford Site Upland Environment

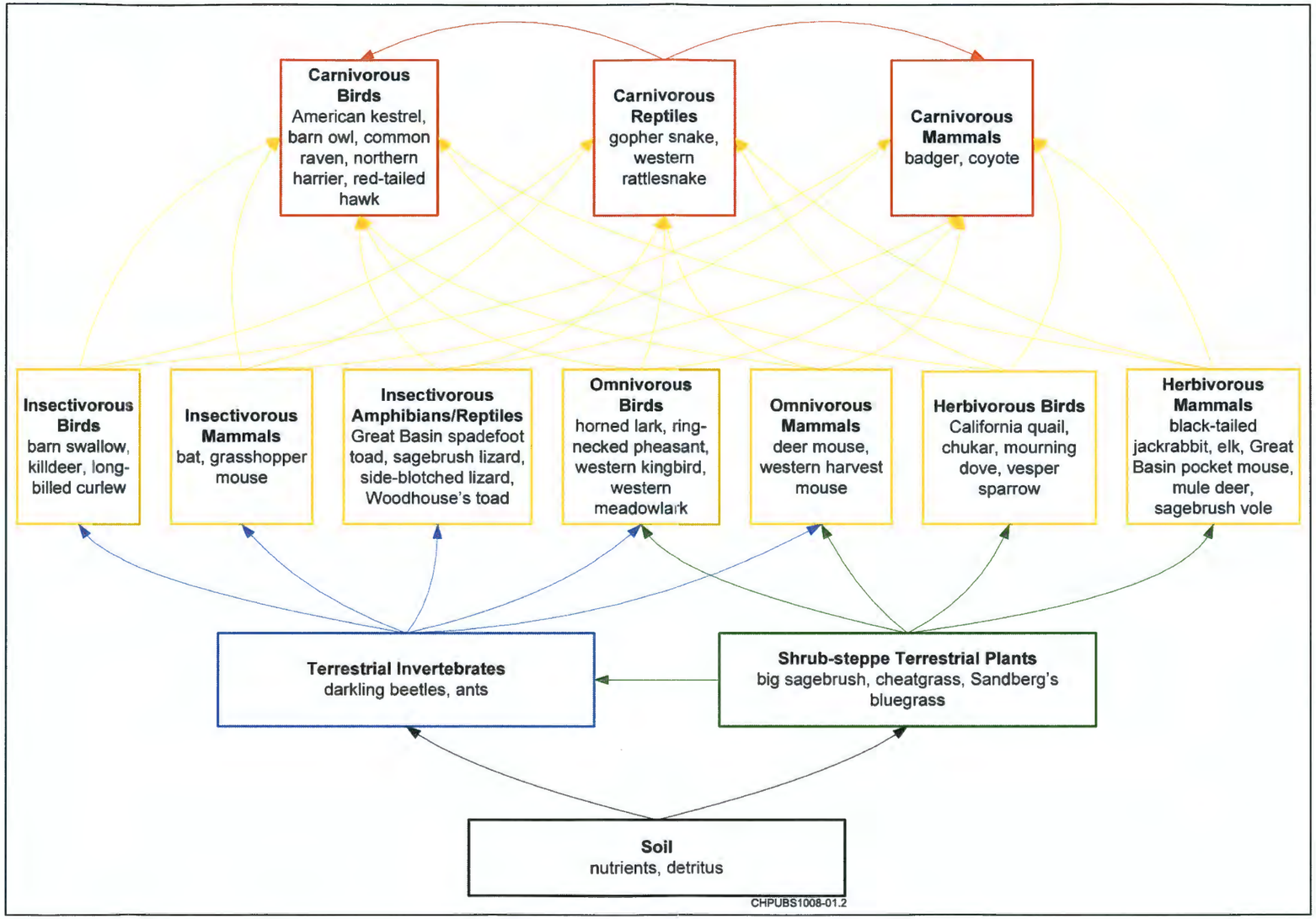


Figure 7-2. Hanford Site Upland Environment Terrestrial Food Web

The following entities (represented by trophic guilds) and their associated organizational level have been identified for evaluation:

- Terrestrial plants—community level
- Terrestrial invertebrates—community level
- Soil micro-organisms and microbial processes—community level
- Herbivorous birds—population level
- Herbivorous mammals—population level
- Insectivorous birds—population level
- Insectivorous mammals—population level
- Omnivorous birds—population level
- Omnivorous mammals—population level
- Carnivorous birds—population level
- Carnivorous mammals—population level
- Reptiles and amphibians[3]

Some endpoints entities are evaluated at the population level and others at the community level. As reported in *Summary Report: Risk Assessment Forum Technical Workshop on Population-level Ecological Risk Assessment* (EPA/100/R-09/006), "Define ecological risk assessment as estimating the likelihood or probability of adverse effects (e.g., mortality to single species of organisms, reduction in populations of nontarget organisms because of acute, chronic, and reproductive effects, or disruption in community and ecosystem level functions)." The EPA has developed guidance that can aid in distinguishing the assessment level including *Framework for Ecological Risk Assessment* (EPA/630/R-92/001), *Ecological Significance and Selection of Candidate Assessment Endpoints* (EPA/540/F-95/037), and *Generic Ecological Assessment Endpoints (GEAEs) for Ecological Risk Assessment* (EPA/630/P-02/004F). These guidelines intentionally do not specify a target level of organization to protect for an entity allowing flexibility in setting the target organizational level that works for the individual project. The organizational levels described above align with the management goals originally defined in *DQO Summary Report for the 100 Area and 300 Area Component of the RCBRA* (BHI-01757), which focuses on protecting individuals for special-status species, preventing adverse effects on Hanford biota from contaminants, protecting rare habitats, and minimizing contaminant loading into biota. With the ecosystem at the Hanford Site, maintaining the health of wildlife populations and the function of a plant community are appropriate as opposed to focusing on populations of particular plant species within that community.

As noted in Appendix A to *Generic Ecological Assessment Endpoints (GEAEs) for Ecological Risk Assessment* (EPA/630/P-02/004F), EPA's principles for ecological risk assessment and risk management at Superfund sites state that "Superfund's goal is to reduce ecological risks to levels that will result in the recovery and maintenance of healthy local populations and communities of biota." Should a special-status species of plant (such as an endangered species of native grass or forb) be present at a given waste site at the Hanford Site, protecting that population would be acceptable. However, the measurement endpoints described in the next section that align with these entities described above were selected appropriately to protect populations and communities. Although the endpoints identified may be expressed as single species toxicity tests, as these guidance documents express, interpretation of the

[3] Although part of the food web for the upland environment, effects data for reptiles and amphibians are limited. Therefore, SSLs were not developed for this trophic guild.

results relative to lowest observed effect concentration (LOEC) or lowest observed adverse effect level (LOAEL) endpoints for the protection of populations and communities is appropriate. Section III in *Issuance of Final Guidance: Ecological Risk Assessment and Risk Management Principles for Superfund Sites* (OSWER Directive 9285.7-28 P) states, "Levels that are expected to protect local populations and communities can be estimated by extrapolating from effects on individuals and groups of individuals using a lines-of-evidence approach. "The performance of multi-year field studies at Superfund sites to try to quantify or predict long-term changes in local populations is not necessary for appropriate risk management decisions to be made. Data from discrete field and laboratory studies, if properly planned and appropriately interpreted can be used to estimate local population or community-level effects." *Issuance of Final Guidance: Ecological Risk Assessment and Risk Management Principles for Superfund Sites* (OSWER Directive 9285.7-28 P) further states that "Superfund ERAs gather effects data on individuals in order to predict or postulate potential effects on local wildlife, fish, invertebrates, and plant populations and communities that occur in specific habitats at sites." Finally, as noted in the *Overview of the Ecological Risk Assessment Process in the Office of Pesticide Programs, U.S. Environmental Protection Agency—Endangered and Threatened Species Effects Determinations* (EPA, 2004), "If effects on the survival and reproduction of individuals are limited, it is assumed that risks at the population level from such effects will be of minor consequence."

To calculate ecological SSLs, endpoint representative species were selected for each entity identified above (trophic guilds/functional groups) that could use the site. For example, a red-tailed hawk may be considered representative of raptors visiting the site. Consistent with ERAGS (EPA 540-R-97-006); *Guidelines for Ecological Risk Assessment* (EPA/630/R-95/002F); and MTCA ("Site-Specific Terrestrial Ecological Evaluation Procedures" [WAC 173-340-7493]), endpoint species should preferably be ones that have ecological relevance, are of societal value, are susceptible to chemical stressors at the site, or allow risk managers to meet policy goals. These factors were used to select representative receptor species common to the Hanford Site upland environment that are within the trophic guilds identified above. Selected receptors are conservative indicators of the potential for risk to the trophic guilds identified for evaluation. The representative receptor species selected for each of the trophic guilds are as follows:

- Herbivorous birds—California quail (*Callipepla californica*)
- Herbivorous mammals—Great Basin pocket mouse (*Perognathus parvus*)
- Insectivorous birds—killdeer (*Charadrius vociferus*)
- Insectivorous mammals—northern grasshopper mouse (*Onychomys leucogaster*)
- Omnivorous birds—western meadowlark (*Sturnella neglecta*)
- Omnivorous mammals—deer mouse (*Peromyscus maniculatus*)
- Carnivorous birds (raptors)—red-tailed hawk (*Buteo jamaicensis*)
- Carnivorous mammals—badger (*Taxidea taxus*)

Unlike birds and mammals, methods to differentiate exposure and/or effects among different plant species or among invertebrate species are unavailable. Therefore, individual species for terrestrial vegetation and invertebrates were not selected to represent the plant or invertebrate populations and communities for evaluation.

7.2.3 Assessment Endpoints

Assessment endpoints are an expression of the important ecological values that are to be protected at a site (*Ecological Risk Assessment* [Suter, 1993]; *Guidelines for Ecological Risk Assessment* [EPA/630/R-95/002F]; *Ecological Risk Assessment for Contaminated Sites* [Suter et al., 2000]).

Assessment endpoints are based on known information concerning the analytes present, the study area, the ecological CSM, and risk hypotheses. The three components to each assessment endpoint are as follows: an *entity* (e.g., migratory birds), an *attribute* of that entity (e.g., individual survival), and a *measure* (e.g., a measurable value, such as an effect level). Measures are described following the general description of assessment endpoints (*Guidelines for Ecological Risk Assessment* [EPA/630/R-95/002F]; *Ecological Risk Assessment for Contaminated Sites* [Suter et al., 2000]).

The assessment endpoint entities for the 100-D/H Source OUs waste sites were selected based on the following principal criteria:

- Ecological relevance
- Societal relevance
- Susceptibility (or high exposure) to known or potential stressors at the Hanford Site

The attribute selected for each entity was based on the organizational level of the entity and the primary criteria used to select it. Entities and attributes were selected for community and population levels of assessment.

7.2.4 Measures of Exposure and Effects

Measures (formerly referred to as measurement endpoints) are measurable attributes used to evaluate the risk hypotheses and are predictive of effects on the assessment endpoints (*Guidelines for Ecological Risk Assessment* [EPA/630/R-95/002F]). The three categories of measures are as follows:

- Measures of exposure are used to evaluate intake of a contaminant from contact with environmental media (for example, soil). Measures of exposure can be an EPC of a COPC in an environmental medium or food item. A measure of exposure also can be a dose occurring through ingestion, inhalation, or dermal contact with a contaminant in an environmental medium. The SSLs were estimated by back-calculating from a target dose associated with the selected assessment endpoint to a corresponding concentration in soil (see Section 7.3.1 for further discussion).

  The measure of exposure represents the exposure appropriate for the assessment endpoint (for example, a wildlife population) throughout its exposure area (for example, the entire home range of the target species). Thus, the average exposure to multiple individuals (for example, the population of wildlife or the plant community) in a species is the basis for population- or community-level effects.

- Measures of effect are used to evaluate the response of an organism that is exposed to a stressor. Measures of effects used in this evaluation include TRVs for wildlife (Appendix H, Tables H-1 and H-2) and LOECs in soil for plants and soil invertebrates (Section 7.3.1). The maximum acceptable adverse effect levels generally selected for population- and community-level assessment endpoints are the lowest LOECs or LOAELs, when available.

- Measures of ecosystem and receptor characteristics are used to evaluate the ecosystem characteristics that influence the assessment endpoints, the distribution of stressors, and the characteristics of the assessment endpoints that may affect exposure or response to the stressor. Measures of ecosystem and receptor characteristics are used to characterize ecological risks as part of a baseline ecological risk assessment or evaluation. This ecological information was not used directly in calculating SSLs. However, measures of ecosystem and receptor characteristics may represent additional lines of evidence that can be used along with SSLs in evaluating remedial alternatives in the RI/FS.

7.3 Effects and Exposure Assessment

The effects and exposure assessments were conducted and integrated to develop two levels of thresholds for evaluating the 100-D/H data. This follows the tiered process referred to earlier and as described in ERAGS (EPA 540-R-97-006). The initial evaluation versus conservative thresholds (SSLs) helps to focus the evaluation on those COPEC-receptor-waste sites combinations that might require further evaluation. The additional evaluation completed with a comparison to PRGs helps identify which COPEC-receptor-waste sites combinations should be brought forward to the SMDP in Section 7.6. Comparisons to SSLs were used to identify COPEC receptor waste sites combinations for the SMDP in cases where the second tier of effect level (PRG) was not available or recommended (e.g., organics, radionuclides, and a few inorganics).

For wildlife, the effects assessment presents TRVs derived from literature-based toxicity information on COPCs that can be used in determining the potential for adverse effects to ecological receptors. The following two types of effects-based values are presented in this ERA: initial conservative values from published literature (for example, Ecology, EPA, and DOE guidance or compendiums), and more Hanford Site-specific values (values established using data collected at the Hanford Site). These values are used within food-chain exposure dose models from the exposure assessment to establish media benchmarks (thresholds). For plants and invertebrates, the effects data are incorporated more simply because the effects are measured relative to direct exposure. Thus, the concentration associated with an observed effect in the exposure medium (soil, water, sediment) becomes the benchmark (threshold).

The exposure assessment identifies exposure pathways associated with the representative receptor species listed in Section 7.2.2. The exposure assessment uses the following two types of exposure evaluations: the avian and mammalian SSLs, and the more site-specific avian and mammalian PRGs. It also describes the models used to calculate SSLs and PRGs.

The TRVs were combined with the exposure information to calculate SSLs and PRGs. This section presents the salient features of the effects and exposure assessments as they were used to calculate the SSLs and PRGs. The development of the nonradionuclide and radionuclide SSLs and PRGs is summarized in the exposure assessment for each receptor group (that is, plants, soil invertebrates, and wildlife). The methodology used to develop the SSLs is detailed in *Tier 1 Risk-Based Soil Concentrations Protective of Ecological Receptors at the Hanford Site* (CHPRC-00784). The methodology used to develop the PRGs for wildlife is detailed in *Tier 2 Risk-Based Soil Concentrations Protective of Ecological Receptors at the Hanford Site* (CHPRC-01311). The methodology used to develop the Hanford Site-specific risk thresholds and to select PRGs for plants and invertebrates is detailed in *Tier 2 Terrestrial Plant and Invertebrate Preliminary Remediation Goals (PRGs) for Nonradionuclides for Use at the Hanford Site* (ECF-HANFORD-11-0158). These documents are presented in Appendix H of the 100-K RI/FS Report (DOE/RL-2010-97).

The effects and exposure assessment is organized as follows:

- Section 7.3.1 presents the effects assessment with separate sections for radionuclides (Section 7.3.1.1) and nonradionuclides (Section 7.3.1.2) because of the method of their derivation. Within each of these sections, effects for plants and invertebrates are discussed separately from wildlife. For radionuclides, the effects assessment includes values that correspond to effects from a dose of radiation. For nonradionuclides, plant and invertebrate effects are described relative to direct exposure, whereas for wildlife, the effects are described relative to the ingested dose.

- Section 7.3.2 presents the exposure assessment with separate sections for plants and invertebrates (Section 7.3.2.1) and wildlife (Section 7.3.2.2). Exposure to wildlife is further broken out to describe

the food-chain models that estimate the concentration in ingested prey and how the assumptions of the model differ in the development of SSLs versus PRGs. Section 7.3.2.3 further describes specific differences in the modeling of wildlife exposure to radionuclides. The SSLs that result from the effects and exposure assessments are presented in these sections.

- Section 7.3.3 describes wildlife exposure through drinking from seeps along the Columbia River.
- Section 7.3.4 describes the PRGs that result from the effects and exposure assessment.
- Section 7.3.5 describes how soil and seep data were used to estimate EPCs for comparisons with the SSLs and PRGs.

7.3.1 Effects Assessment

The ecological effects assessment consists of an evaluation of available toxicity or other effects information to interpret the significance of the exposures to COPCs relative to potential adverse effects to ecological receptors. Data that can be used include literature-derived or site-specific single-chemical toxicity data (wildlife), site-specific ambient-media toxicity tests (plants and invertebrates), and site-specific field surveys (*Ecological Risk Assessment for Contaminated Sites* [Suter et al., 2000]). The effects data used in this ERA are represented by single-chemical toxicity data from literature sources and are summarized below for radionuclides and nonradionuclides. The effects levels presented are used either directly (for plants and invertebrates) or within exposure dose models (for wildlife) to establish concentrations in exposure media (for example, soil) that protect plant and invertebrate communities and wildlife populations.

7.3.1.1 Effects Assessment of Radionuclides

Radionuclide toxicity data for plants and wildlife are represented by DOE's Biota Concentration Guides (BCGs) for radionuclides, presented in *A Graded Approach for Evaluating Radiation Doses to Aquatic and Terrestrial Biota* (DOE-STD-1153-2002), hereinafter called Graded Approach for Radiation Doses to Biota. Two radionuclide effect thresholds, as determined by consensus of international radiation regulatory agencies, form the basis for effect thresholds used to develop screening levels of radionuclides in soil for the protection of plants and animals. General guidance from the International Council for Radiological Protection (*Recommendations of the International Commission on Radiological Protection* [ICRP-60]), *Proliferation Resistance Fundamentals for Future Nuclear Energy Systems* (IAEA STR-332), and *Sources and Effects of Ionizing Radiation* (UNSCEAR, 2000) with scientific annexes (Sales Publication No. E.00.IX.4) concluded that radiological doses to terrestrial plants and terrestrial vertebrates should not exceed 1.0 and 0.1 rad/day, respectively. If radiation exposure does not exceed these biota dose levels, the consensus opinion of the international radiological organizations is that ecological populations will be protected. DOE has adopted these effect thresholds and integrated them into Graded Approach for Radiation Doses to Biota (DOE-STD-1153-2002), which includes the following screening method and three detailed levels of analysis for demonstrating compliance with applicable dose limits for protection of biota:

- A general screening that involves comparing maximum radionuclide concentrations in environmental media (that is, soil) with a set of BCGs to evaluate compliance with the biota dose limits.
- Site-specific screening using more realistic site- representative lumped parameters (for example, bioaccumulation factors [BAFs]) in place of conservative default parameters, using mean radionuclide concentrations in place of maximum values, and considering time dependence and spatial extent of contamination.

- Site-specific analysis using a kinetic-allometric modeling methodology. Multiple parameters, which represent contribution to an organism's internal dose, can be modified to represent site- and organism-specific characteristics. These parameters include body mass, consumption rates of food or soil, inhalation rate, lifespan, and biological elimination rates. Development of the organism-specific characteristics involves using allometric equations that relate these parameters to body mass.
- Site-specific biota dose assessment involving the collection and analysis of biota samples.

BCGs can be calculated using dose models, equations, and default parameters presented in Graded Approach for Radiation Doses to Biota (DOE-STD-1153-2002). The values in soil, calculated using these default methods, are included in Table 6-4 of Graded Approach for Radiation Doses to Biota (DOE-STD-1153-2002). These dose models, equations, and default parameters are also incorporated into the RESRAD-BIOTA for Windows, Version 1.5 (ANL, 2009a) model (*RESRAD-BIOTA: A Tool for Implementing a Graded Approach to Biota Dose Evaluation, User's Guide, Version 1* [DOE/EH-0676]) to establish values protective of wildlife populations and plant communities. *Effects of Ionizing Radiation on Terrestrial Plants and Animals: A Workshop Report* (ORNL/TM-13141) also discusses populations of wildlife and communities of plants as the basis for the BCGs. RESRAD-BIOTA presents the following three levels of analysis, which correspond to the following levels in the graded approach:

- Level 1—general screening approach
- Level 2—site-specific screening with representative parameters
- Level 3—site-specific analysis using the kinetic/allometric modeling methodology

The BCGs for plants for this ERA were calculated using the Level 1 analysis in RESRAD-BIOTA and are shown in Table 7-1.

For wildlife (animals), more receptor-specific SSLs were developed using RESRAD-BIOTA for Windows, Version 1.5 (ANL, 2009a) with Level 3 assumptions. Values were established for eight species representing feeding guilds at the site. However, Hanford Site-specific tissue residue of radionuclides was insufficient for developing models so values from relevant published literature were used ("Derivation of Transfer Parameters for Use Within the ERICA Tool and the Default Concentration Ratios for Terrestrial Biota" [Beresford et al., 2008]). Final radionuclide SSLs for wildlife are listed in Table 7-2.

Because the dose from radionuclides is additive ("Principles and Issues in Radiological Ecological Risk Assessment" [Jones et al., 2003]), the total contribution of radionuclides known to be associated with Hanford Site processes was also calculated. A total radionuclide exposure estimate was calculated using the sum of fractions (SOF) method. With the SOF method, the contributions of radionuclides were reviewed to determine their contribution to dose. Contributions were considered significant if the radionuclide EPC was greater than the SSL and detected frequently.

7.3.1.2 Effects Assessment for Nonradionuclides

Effects data for the nonradionuclide COPCs are presented below for plants and invertebrates and for wildlife. Included is a description of the sources of the information used and an explanation of the selection of effects data. The overarching theme was to use the most recent of relevant toxicological information available as described within ERAGS (EPA 540-R-97-006) and MTCA ("Site-Specific Terrestrial Ecological Evaluation Procedures" [WAC 173-340-7493]).

Table 7-1. SSLs in Upland Soil for Plants and Soil Invertebrates

			EPA EcoSSLs			ORNL - ES/ER/TM-85/R3, ES/ER/TM-126/R2		DOE BCGs		Washington State Dept. of Ecology - MTCA (WAC 173-340, Table 749-3)		Lowest Screening Benchmark by Receptor Type					
Group	Soil Constituent	Units	Plants	Invertebrate	Reference	Plants	Invertebrate	Terrestrial Plant	Terrestrial Animal	Plant	Soil Biota	Plant	Inverts	Overall Lowest Screening Benchmark	Background Soil Concentrations[a]	SSL for Plants and Soil Invertebrates[b]	Basis
Radionuclides	Americium-241	pCi/g	---	---	---	---	---	21,500	3,890	---	---	21,500	---	21,500	---	21,500	Benchmark
	Antimony-125	pCi/g	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---
	Carbon-14	pCi/g	---	---	---	---	---	60,700	4,760	---	---	60,700	---	60,700	---	60,700	Benchmark
	Cesium-134	pCi/g	---	---	---	---	---	1,090	11.3	---	---	1,090	---	1,090	---	1,090	Benchmark
	Cesium 137	pCi/g	---	---	---	---	---	2,210	20.8	---	---	2,210	---	2,210	1.05	2,210	Benchmark
	Cobalt-60	pCi/g	---	---	---	---	---	6,130	692	---	---	6,130	---	6,130	0.00842	6,130	Benchmark
	Curium-244	pCi/g	---	---	---	---	---	153,000	4,060	---	---	153,000	---	153,000	---	153,000	Benchmark
	Europium-152	pCi/g	---	---	---	---	---	14,700	1,520	---	---	14,700	---	14,700	---	14,700	Benchmark
	Europium-154	pCi/g	---	---	---	---	---	12,500	1,290	---	---	12,500	---	12,500	0.0334	12,500	Benchmark
	Europium-155	pCi/g	---	---	---	---	---	153,000	15,800	---	---	153,000	---	153,000	0.0539	153,000	Benchmark
	Hydrogen-3 (tritium)	pCi/g	---	---	---	---	---	1,680,000	174,000	---	---	1,680,000	---	1,680,000	---	1,680,000	Benchmark
	Neptunium-237	pCi/g	---	---	---	---	---	8,150	3,860	---	---	8,150	---	8,150	---	8,150	Benchmark
	Nickel-63	pCi/g	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---
	Plutonium-238	pCi/g	---	---	---	---	---	17,500	5,270	---	---	17,500	---	17,500	0.00378	17,500	Benchmark
	Plutonium-239/240	pCi/g	---	---	---	---	---	12,700	6,110	---	---	12,700	---	12,700	0.0248	12,700	Benchmark
	Radium-226	pCi/g	---	---	---	---	---	288	50.6	---	---	288	---	288	0.815	288	Benchmark
	Radium-228	pCi/g	---	---	---	---	---	245	43.9	---	---	245	---	245	---	245	Benchmark
	Strontium-90	pCi/g	---	---	---	---	---	3,580	22.5	---	---	3,580	---	3,580	0.178	3,580	Benchmark
	Technetium-99	pCi/g	---	---	---	---	---	21,900	4,490	---	---	21,900	---	21,900	---	21,900	Benchmark
	Thorium-232	pCi/g	---	---	---	---	---	23,500	1,510	---	---	23,500	---	23,500	1.32	23,500	Benchmark
	Uranium-234	pCi/g	---	---	---	---	---	51,600	5,130	---	---	51,600	---	51,600	1.1	51,600	Benchmark
	Uranium-235	pCi/g	---	---	---	---	---	27,400	2,770	---	---	27,400	---	27,400	0.109	27,400	Benchmark
	Uranium-238	pCi/g	---	---	---	---	---	15,700	1,580	---	---	15,700	---	15,700	1.06	15,700	Benchmark

Table 7-1. SSLs in Upland Soil for Plants and Soil Invertebrates

			EPA EcoSSLs			ORNL - ES/ER/TM-85/R3, ES/ER/TM-126/R2		DOE BCGs		Washington State Dept. of Ecology - MTCA (WAC 173-340, Table 749-3)		Lowest Screening Benchmark by Receptor Type					
Group	Soil Constituent	Units	Plants	Invertebrate	Reference	Plants	Invertebrate	Terrestrial Plant	Terrestrial Animal	Plant	Soil Biota	Plant	Inverts	Overall Lowest Screening Benchmark	Background Soil Concentrations[a]	SSL for Plants and Soil Invertebrates[b]	Basis
Metals	Aluminum	mg/kg	Narrative Statement		OSWER Dir. 9285.7-60	50	---	---	---	50	---	50	---	50	11,800	11,800	Background
	Antimony	mg/kg	---	78	OSWER Dir. 9285.7-61	5	---	---	---	5	---	5	78	5	5.2	5.2	Background
	Arsenic, total all valence states	mg/kg	18	---	OSWER Dir. 9285.7-62	10	60	---	---	---	---	10	60	10	6.47	10	Benchmark
	Arsenic (III)	mg/kg	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---
	Arsenic (V)	mg/kg	---	---	---	---	---	---	---	10	60	10	60	10	---	10	Benchmark
	Barium	mg/kg	---	330	OSWER Dir. 9285.7-63	500	---	---	---	500	---	500	330	330	132	330	Benchmark
	Beryllium	mg/kg	---	40	OSWER Dir. 9285.7-64	10	---	---	---	10	---	10	40	10	1.51	10	Benchmark
	Bismuth	mg/kg	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---
	Boron	mg/kg	---	---	---	0.5	---	---	---	0.5	---	0.5	---	0.5	---	0.5	Benchmark
	Cadmium	mg/kg	32	140	OSWER Dir. 9285.7-65	4	20	---	---	4	20	4	20	4	0.78	4	Benchmark
	Chromium (total)[c,d]	mg/kg	---	---	OSWER Dir. 9285.7-66	1	0.4	---	---	42	42	1	0.4	0.4	18.5	18.5	Background
	Chromium(III)	mg/kg	---	---	OSWER Dir. 9285.7-66	1	0.4	---	---	42	42	1	0.4	0.4	---	0.4	Benchmark
	Chromium(VI)	mg/kg	---	---	OSWER Dir. 9285.7-66	---	---	---	---	---	---	---	---	---	---	---	---
	Cobalt	mg/kg	13	---	OSWER Dir. 9285.7-67	20	---	---	---	20	---	13	---	13	15.7	15.7	Background
	Copper	mg/kg	70	80	OSWER Dir. 9285.7-68	100	50	---	---	100	50	70	50	50	22	50	Benchmark
	Lead	mg/kg	120	1700	OSWER Dir. 9285.7-70	50	500	---	---	50	500	50	500	50	10.2	50	Benchmark
	Lithium[d]	mg/kg	---	---	---	2	---	---	---	35	---	2	---	2	33.5	33.5	Background
	Manganese[d]	mg/kg	220	450	OSWER Dir. 9285.7-71	500	---	---	---	1,100	---	220	450	220	512	512	Background
	Mercury	mg/kg	---	---	---	0.3	0.1	---	---	0.3	0.1	0.3	0.1	0.1	0.33	0.33	Background
	Molybdenum	mg/kg	---	---	---	2	---	---	---	2	---	2	---	2	6	6	Background
	Nickel	mg/kg	38	280	OSWER Dir. 9285.7-76	30	200	---	---	30	200	30	200	30	19.1	30	Benchmark
	Selenium	mg/kg	0.52	4.1	OSWER Dir. 9285.7-72	1	70	---	---	1	70	0.52	4.1	0.52	0.78	0.78	Background
	Silver	mg/kg	560	---	OSWER Dir. 9285.7-77	2	---	---	---	2	---	2	---	2	0.73	2	Benchmark
	Strontium	mg/kg	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---
	Thallium	mg/kg	---	---	---	1	---	---	---	1	---	1	---	1		1	Benchmark
	Tin	mg/kg	---	---	---	50	---	---	---	50	---	50	---	50	---	50	Benchmark
	Uranium	mg/kg	---	---	---	5	---	---	---	5	---	5	---	5	3.21	5	Benchmark
	Vanadium	mg/kg	---	---	OSWER Dir. 9285.7-75	2	---	---	---	2	---	2	---	2	85.1	85.1	Background
	Zinc[d]	mg/kg	160	120	OSWER Dir. 9285.7-73	50	200	---	---	86	200	50	120	50	67.8	67.8	Background

Table 7-1. SSLs in Upland Soil for Plants and Soil Invertebrates

			EPA EcoSSLs			ORNL - ES/ER/TM-85/R3, ES/ER/TM-126/R2		DOE BCGs		Washington State Dept. of Ecology - MTCA (WAC 173-340, Table 749-3)		Lowest Screening Benchmark by Receptor Type					
Group	Soil Constituent	Units	Plants	Invertebrate	Reference	Plants	Invertebrate	Terrestrial Plant	Terrestrial Animal	Plant	Soil Biota	Plant	Inverts	Overall Lowest Screening Benchmark	Background Soil Concentrations[a]	SSL for Plants and Soil Invertebrates[b]	Basis
General Inorganics	Ammonia/ammonium	mg/kg	---	---	---	---	---	---	---	---	---	---	---	---	9.23	9.23	Background
	Chloride	mg/kg	---	---	---	---	---	---	---	---	---	---	---	---	100	100	Background
	Cyanide	mg/kg	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---
	Fluoride	mg/kg	---	---	---	---	---	---	---	---	---	---	---	---	2.81	2.81	Background
	Iodine	mg/kg	---	---	---	4	---	---	---	4	---	4	---	4	---	4	Benchmark
	Nitrate/nitrite	mg/kg	---	---	---	---	---	---	---	---	---	---	---	---	52	52	Background
	Phosphate	mg/kg	---	---	---	---	---	---	---	---	---	---	---	---	0.785	0.785	Background
	Sulfate/sulfite	mg/kg	---	---	---	---	---	---	---	---	---	---	---	---	237	237	Background
	Total organic carbon	mg/kg	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---
Volatile Organics	1,1-dichloroethane	mg/kg	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---
	1,1-dichloroethene	mg/kg	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---
	1,1,1-trichloroethane	mg/kg	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---
	1,1,2-trichloroethane	mg/kg	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---
	1,1,2,2-tetrachloroethane	mg/kg	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---
	1,2-dichlorobenzene	mg/kg	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---
	1,2-dichloroethane (DCA)	mg/kg	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---
	1,3-dichlorobenzene	mg/kg	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---
	2-butanone (methyl ethyl ketone/MEK)	mg/kg	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---
	2-hexanone	mg/kg	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---
	Benzene	mg/kg	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---
	Butanol	mg/kg	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---
	Carbon tetrachloride	mg/kg	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---
	Chlorobenzene	mg/kg	---	---	---	---	40	---	---	---	40	---	40	40	---	40	Benchmark
	Chloroform	mg/kg	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---
	cis-1,2-dichloroethylene	mg/kg	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---
	Dichloromethane (methylene chloride)	mg/kg	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---
	Ethyl benzene	mg/kg	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---
	Methyl isobutyl ketone	mg/kg	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---
	n-butyl benzene	mg/kg	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---

Table 7-1. SSLs in Upland Soil for Plants and Soil Invertebrates

			EPA EcoSSLs			ORNL - ES/ER/TM-85/R3, ES/ER/TM-126/R2		DOE BCGs		Washington State Dept. of Ecology - MTCA (WAC 173-340, Table 749-3)		Lowest Screening Benchmark by Receptor Type					
Group	Soil Constituent	Units	Plants	Invertebrate	Reference	Plants	Invertebrate	Terrestrial Plant	Terrestrial Animal	Plant	Soil Biota	Plant	Inverts	Overall Lowest Screening Benchmark	Background Soil Concentrations[a]	SSL for Plants and Soil Invertebrates[b]	Basis
	Tetrachloroethylene	mg/kg	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---
	Toluene	mg/kg	---	---	---	200	---	---	---	200	---	200	---	200	---	200	Benchmark
	Trans-1,2-dichloroethylene	mg/kg	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---
	Trichloroethylene (TCE)	mg/kg	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---
	Xylene	mg/kg	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---
Polycyclic Aromatic Hydrocarbons	Acenaphthene	mg/kg	---	29	OSWER Dir. 9285.7-75	20	---	---	---	20	---	20	29	20	---	20	Benchmark
	Acenaphthylene	mg/kg	---	29	OSWER Dir. 9285.7-75	---	---	---	---	---	---	---	29	29	---	29	Benchmark
	Anthracene	mg/kg	---	29	OSWER Dir. 9285.7-75	---	---	---	---	---	---	---	29	29	---	29	Benchmark
	Benzo(a)pyrene	mg/kg	---	18	OSWER Dir. 9285.7-75	---	---	---	---	---	---	---	18	18	---	18	Benchmark
	Benzo(a)anthracene	mg/kg	---	18	OSWER Dir. 9285.7-75	---	---	---	---	---	---	---	18	18	---	18	Benchmark
	Benzo(b)fluoranthene	mg/kg	---	18	OSWER Dir. 9285.7-75	---	---	---	---	---	---	---	18	18	---	18	Benchmark
	Benzo(ghi)perylene	mg/kg	---	18	OSWER Dir. 9285.7-75	---	---	---	---	---	---	---	18	18	---	18	Benchmark
	Benzo[k]fluoranthene	mg/kg	---	18	OSWER Dir. 9285.7-75	---	---	---	---	---	---	---	18	18	---	18	Benchmark
	Chrysene	mg/kg	---	18	OSWER Dir. 9285.7-75	---	---	---	---	---	---	---	18	18	---	18	Benchmark
	Dibenz(ah)anthracene	mg/kg	---	18	OSWER Dir. 9285.7-75	---	---	---	---	---	---	---	18	18	---	18	Benchmark
	Fluoranthene	mg/kg	---	18	OSWER Dir. 9285.7-75	---	---	---	---	---	---	---	18	18	---	18	Benchmark
	Fluorene	mg/kg	---	29	OSWER Dir. 9285.7-75	---	30	---	---	---	30	---	29	29	---	29	Benchmark
	Indeno[1,2,3-cd]pyrene	mg/kg	---	18	OSWER Dir. 9285.7-75	---	---	---	---	---	---	---	18	18	---	18	Benchmark
	2-methylnaphthalene	mg/kg	---	29	OSWER Dir. 9285.7-75	---	---	---	---	---	---	---	29	29	---	29	Benchmark
	Naphthalene	mg/kg	---	29	OSWER Dir. 9285.7-75	---	---	---	---	---	---	---	29	29	---	29	Benchmark
	Phenanthrene	mg/kg	---	29	OSWER Dir. 9285.7-75	---	---	---	---	---	---	---	29	29	---	29	Benchmark
	Pyrene	mg/kg	---	18	OSWER Dir. 9285.7-75	---	---	---	---	---	---	---	18	18	---	18	Benchmark
	Total PAHs	mg/kg	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---
	Low molecular weight PAHs[e]	mg/kg	---	29	OSWER Dir. 9285.7-75	---	---	---	---	---	---	---	29	29	---	29	Benchmark
	High molecular weight PAHs[f]	mg/kg	---	18	OSWER Dir. 9285.7-75	---	---	---	---	---	---	---	18	18	---	18	Benchmark

Table 7-1. SSLs in Upland Soil for Plants and Soil Invertebrates

			EPA EcoSSLs			ORNL - ES/ER/TM-85/R3, ES/ER/TM-126/R2		DOE BCGs		Washington State Dept. of Ecology - MTCA (WAC 173-340, Table 749-3)		Lowest Screening Benchmark by Receptor Type					
Group	Soil Constituent	Units	Plants	Invertebrate	Reference	Plants	Invertebrate	Terrestrial Plant	Terrestrial Animal	Plant	Soil Biota	Plant	Inverts	Overall Lowest Screening Benchmark	Background Soil Concentrations[a]	SSL for Plants and Soil Invertebrates[b]	Basis
Petroleum	Gasoline range organics	mg/kg	---	---	---	---	---	---	---	---	100	---	100	100	---	100	Benchmark
	TPH-diesel	mg/kg	---	---	---	---	---	---	---	---	200	---	200	200	---	200	Benchmark
	TPH-kerosene	mg/kg	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---
Semivolatile Organics	Normal paraffin hydrocarbons	mg/kg	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---
	Phenol	mg/kg	---	---	---	70	30	---	---	70	30	70	30	30	---	30	Benchmark
	2-methylphenol (o-cresol)	mg/kg	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---
	4-methylphenol (p-cresol)	mg/kg	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---
	2,4-dinitrotoluene	mg/kg	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---
	Bis(2-ethylhexyl)phthalate[g]	mg/kg	---	---	---	100	---	---	---	100	---	100	---	100	---	100	Benchmark
	Polychlorinated biphenyls (PCB)[h,i]	mg/kg	---	---	---	40	---	---	---	40	---	40	---	40	---	40	Benchmark
	Aroclor-1016[h,i]	mg/kg	---	---	---	40	---	---	---	40	---	40	---	40	---	40	Benchmark
	Aroclor-1221[h,i]	mg/kg	---	---	---	40	---	---	---	40	---	40	---	40	---	40	Benchmark
	Aroclor-1232[h,i]	mg/kg	---	---	---	40	---	---	---	40	---	40	---	40	---	40	Benchmark
	Aroclor-1242[h,i]	mg/kg	---	---	---	40	---	---	---	40	---	40	---	40	---	40	Benchmark
	Aroclor-1248[h,i]	mg/kg	---	---	---	40	---	---	---	40	---	40	---	40	---	40	Benchmark
	Aroclor-1254[h,i]	mg/kg	---	---	---	40	---	---	---	40	---	40	---	40	---	40	Benchmark
	Aroclor-1260[h,i]	mg/kg	---	---	---	40	---	---	---	40	---	40	---	40	---	40	Benchmark
	Aroclor-1262[h,i,j]	mg/kg	---	---	---	40	---	---	---	40	---	40	---	40	---	40	Benchmark
Herbicide	Dichloroprop	mg/kg	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---
Pesticide	Aldrin	mg/kg	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---
	beta-1,2,3,4,5,6-hexachlorocyclohexane[k,l]	mg/kg	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---
	alpha-chlordane[m]	mg/kg	---	---	---	---	---	---	---	---	1	---	1	1	---	1	Benchmark
	gamma-chlordane[m]	mg/kg	---	---	---	---	---	---	---	---	1	---	1	1	---	1	Benchmark
	Dichlorodiphenyldichloroethylene	mg/kg	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---
	Dichlorodiphenyltrichloroethane	mg/kg	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---
	Dieldrin	mg/kg	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---
	Endosulfan I	mg/kg	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---
	Endosulfan II	mg/kg	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---
	Endosulfan sulfate	mg/kg	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---

Table 7-1. SSLs in Upland Soil for Plants and Soil Invertebrates

			EPA EcoSSLs			ORNL - ES/ER/TM-85/R3, ES/ER/TM-126/R2		DOE BCGs		Washington State Dept. of Ecology - MTCA (WAC 173-340, Table 749-3)		Lowest Screening Benchmark by Receptor Type					
Group	Soil Constituent	Units	Plants	Invertebrate	Reference	Plants	Invertebrate	Terrestrial Plant	Terrestrial Animal	Plant	Soil Biota	Plant	Inverts	Overall Lowest Screening Benchmark	Background Soil Concentrations[a]	SSL for Plants and Soil Invertebrates[b]	Basis
	Endrin aldehyde	mg/kg	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---
	Methoxychlor	mg/kg	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---

Note: Complete citations of OSWER Directives are provided in Chapter 11.

Sources: ES/ER/TM-85/R3, *Toxicological Benchmarks for Screening Contaminants of Potential Concern for Effects on Terrestrial Plants: 1997 Revision.*

ES/ER/TM-126/R2, *Toxicological Benchmarks for Contaminants of Potential Concern for Effects on Soil and Litter Invertebrates and Heterotrophic Process: 1997 Revision.*

MTCA (WAC 173-340, "Model Toxics Control Act—Cleanup").

a. Background soil concentrations are selected according to the following hierarchy: the 90th percentile of Hanford Site background; Washington State-wide background. See the text for further discussion of sources.

b. The selected PRG is the higher of either the background in soil or the overall lowest screening value between plants and soil invertebrates.

c. When chromium (total) not available, the lower of either Cr(III) or Cr(VI) as available was used as a surrogate.

d. MTCA plant and soil biota benchmarks were replaced by Washington State natural background concentration.

e. The low molecular weight PAHs screening values from EPA (OSWER Directive 9285.7-78 [*Ecological Soil Screening Levels for Polycyclic Aromatic Hydrocarbons (PAHs): Interim Final*]) represents the sum of the low molecular weight PAHs. For the purposes of this assessment, the benchmark was also applied to the individual low molecular weight PAHs.

f. The high molecular weight PAHs screening values from EPA (OSWER Directive 9285.7-78 [*Ecological Soil Screening Levels for Polycyclic Aromatic Hydrocarbons (PAHs): Interim Final*]) represents the sum of the high molecular weight PAHs. For the purposes of this assessment, the benchmark was also applied to the individual high molecular weight PAHs.

g. Values for diethyl phthalate were used as a surrogate for bis(2-ethylhexyl)phthalate.

h. Aroclor-1254 value was used as surrogate.

i. MTCA values represent screening value for PCB mixtures.

j. MTCA Aroclor-1260 values used as surrogate for Aroclor-1262.

k. Form of HCB not identified in ES/ER/TM-126/R2.

l. MTCA value based on benzene hexachloride, including lindane.

m. MTCA values based on chlordane.

--- = value not available

ORNL = Oak Ridge National Laboratory

TPH = total petroleum hydrocarbons

Table 7-2. Wildlife (Birds and Mammals) SSLs for Radionuclides

			NOAEL-Based Site-Specific SSLs									LOAEL-Based Site-Specific SSLs								
Group	Soil Constituent	Units	California Quail	Meadow-lark	Killdeer	Red-Tailed Hawk	Great Basin Pocket Mouse	Deer Mouse	Grasshopper Mouse	Badger	NOAEL Lowest	California Quail	Meadow-lark	Killdeer	Red-Tailed Hawk	Great Basin Pocket Mouse	Deer Mouse	Grass-hopper Mouse	Badger	LOAEL Lowest
Radionuclides	Americium-241	pCi/g	--	--	--	--	--	--	--	--	--	28,900	25,000	11,900	17,800	72,100	48,700	41,400	4,840	4,840
	Carbon-14	pCi/g	--	--	--	--	--	--	--	--	--	54	60	56	50	61	60	135	32	32
	Curium-244	pCi/g	--	--	--	--	--	--	--	--	--	389,000	252,000	105,000	207,000	2,300,000	722,000	499,000	50,800	50,800
	Cobalt-60	pCi/g	--	--	--	--	--	--	--	--	--	805	805	805	863	805	805	806	1,000	805
	Cesium-134	pCi/g	--	--	--	--	--	--	--	--	--	1,140	1,190	1,200	854	1,160	1,180	1,270	562	562
	Cesium 137	pCi/g	--	--	--	--	--	--	--	--	--	2,390	2,700	2,800	1,430	2,510	2,630	3,280	924	924
	Europium-152	pCi/g	--	--	--	--	--	--	--	--	--	1,740	1,740	1,740	1,880	1,740	1,740	1,740	2,220	1,740
	Europium-154	pCi/g	--	--	--	--	--	--	--	--	--	1,610	1,610	1,610	1,740	1,610	1,610	1,610	2,060	1,610
	Europium-155	pCi/g	--	--	--	--	--	--	--	--	--	33,400	33,400	33,400	37,300	33,400	33,400	33,400	48,600	33,400
	Hydrogen-3 (tritium)	pCi/g	--	--	--	--	--	--	--	--	--	1,430	1,280	936	1,130	3,270	2,290	2,830	420	420
	Neptunium-237	pCi/g	--	--	--	--	--	--	--	--	--	8,190	8,140	7,880	9,150	8,250	8,170	8,180	11,200	7,880
	Nickel-63	pCi/g	--	--	--	--	--	--	--	--	--									
	Plutonium-238	pCi/g	--	--	--	--	--	--	--	--	--	36,300	56,200	20,900	26,800	291,000	161,000	161,000	5,980	5,980
	Plutonium-239/240	pCi/g	--	--	--	--	--	--	--	--	--	38,800	60,300	22,300	28,400	324,000	175,000	176,000	6,270	6,270
	Radium-226	pCi/g	--	--	--	--	--	--	--	--	--	168	142	58	377	285	165	199	193	58
	Radium-228	pCi/g	--	--	--	--	--	--	--	--	--	169	140	55	418	306	165	203	193	55
	Antimony-125	pCi/g	--	--	--	--	--	--	--	--	--	4,580	4,580	4,580	5,040	4,580	4,580	4,580	6,130	4,580
	Strontium-90	pCi/g	--	--	--	--	--	--	--	--	--	521	302	151	112	706	519	413	91	91
	Technetium-99	pCi/g	--	--	--	--	--	--	--	--	--	5,360	11,500	137,000	280,000	8,670	12,100	412,000	128,000	5,360
	Thorium-232	pCi/g	--	--	--	--	--	--	--	--	--	5,070	12,900	5,340	12,400	34,400	32,500	86,200	4,560	4,560
	Uranium-234	pCi/g	--	--	--	--	--	--	--	--	--	12,700	21,800	6,370	40,900	30,300	24,800	51,600	14,200	6,370
	Uranium-235	pCi/g	--	--	--	--	--	--	--	--	--	6,340	7,810	4,360	10,200	8,600	8,130	9,630	8,060	4,360
	Uranium-238	pCi/g	--	--	--	--	--	--	--	--	--	8,020	10,400	5,150	22,100	11,900	11,000	13,900	13,400	5,150

NOAEL = no observed adverse-effect level

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Plants and Invertebrates. Single-chemical screening-level toxicity values for terrestrial plants and soil invertebrates were available from the following sources:

- EPA's EcoSSLs (http://www.epa.gov/ecotox/ecossl/)
- Screening benchmark concentrations in soil developed by ORNL; many of the ecological indicator soil concentrations published by Ecology were drawn from ORNL screening benchmark concentrations
- Washington State Department of Ecology's ecological indicator soil concentrations, found in "Site-Specific Terrestrial Ecological Evaluation Procedures" (WAC 173-340-7493(2)(a)(i)), Table 749-3

The lowest available plant or invertebrate value from these sources was selected as the SSL for each analyte because they represent direct exposure of the receptors to the media. These SSLs are presented in Table 7-1. Each source is summarized below.

EPA's EcoSSLs for plants and soil invertebrates were derived using data from tests performed within soil conditions favoring relatively high bioavailability for upland soil. The soil chemistry conditions of relatively high bioavailability were defined by organic matter content and by low soil pH. From the studies reviewed, the measure of toxic effects to either plants or soil invertebrates were grouped into one of the following four ecologically relevant endpoints: reproduction, population characteristics, growth, or physiological changes. Toxicity parameters used in deriving the EcoSSLs were the EC20 (effective concentration affecting 20 percent of a test population), the maximum acceptable toxicant concentration (MATC), and the EC10 (effect concentration affecting 10 percent of a test population). The MATC was calculated by EPA from studies that reported a no-observed-adverse-effects concentration (NOAEC) and a lowest observed adverse effects concentration (LOAEC). The MATC was calculated as the geometric mean of the NOAEC and LOAEC. Studies that reported only a LOAEC or only a NOAEC (for example, unbound studies) were not considered to provide a reliable assessment of the dose response and were not used for EcoSSL development. The EcoSSL for plants and soil invertebrates was calculated as the geometric mean of all the toxicity parameters from studies conducted under conditions of high bioavailability. Note that use of the EC20, MATC, and EC10 as toxicity parameters means that EcoSSLs for plants and soil invertebrates are not based on NOAECs; thus, the recommended value is at a level where effects have been observed but to a percent of individuals considered acceptable within the ERA practice as demonstrated by its use within the EcoSSL approach documents (*Guidance for Developing Ecological Soil Screening Levels* [OSWER Directive 9285.7-55]).

The ORNL benchmarks for the toxicity to plants from chemical analytes in soil were based on thresholds for effects on growth and reproduction derived from published toxicity studies conducted in soil or solution. The benchmarks are concentrations of chemicals that correspond to the LOEC for the 10th percentile of plant species tested. The ORNL benchmarks for toxicity to soil invertebrates and heterotrophic processes from analytes in soil represent thresholds (LOECs) for statistically significant effects on growth, reproduction, or activity. The toxicity benchmarks were derived by rank- ordering the LOEC values and selecting a value that approximated the 10th percentile.

If 10 or fewer values were available for a chemical, the lowest LOEC was used. If the 10th percentile fell between LOEC values, a value was chosen by interpolation. If a chemical concentration in soil represented a 50 percent or higher reduction in survivorship of plants, the concentration was divided by five to approximate the more sensitive endpoints of growth or production. Plant toxicity benchmarks for metals are usually lower than those for soil invertebrates or microbial processes, and they are lower than most PRGs calculated for wildlife.

Ecology's ecological indicator soil concentrations, presented in Table 749-3 of MTCA (WAC 173-340), represent soil concentrations expected to be protective at any MTCA (WAC 173-340) site and are provided for use in eliminating hazardous substances from further consideration under MTCA ("Site-Specific Terrestrial Ecological Evaluation Procedures" [WAC 173-340-7493(2)(a)(i)]). The ecological indicator soil concentrations for plants are based on benchmarks published in *Toxicological Benchmarks for Screening Potential Contaminants of Potential Concern for Effects on Terrestrial Plants: 1997 Revision* (ES/ER/TM-85/R3). The ecological indicator soil concentrations for soil biota are based on benchmarks published in *Toxicological Benchmarks for Contaminants of Concern for Effects on Soil and Litter Invertebrates and Heterotrophic Process: 1997 Revision* (ES/ER/TM-126/R2).

Wildlife (Birds and Mammals). Bird and mammal TRVs for both the no observed adverse-effect levels (NOAELs) and LOAELs were used in the SSL and PRG development. The TRVs were used within models relating the ingested dose of the chemicals (Section 7.3.2, Exposure Assessment) with the TRVs to establish SSLs or PRGs that represent adverse effects thresholds. The TRVs were obtained from various sources, with a focus on the most recent sources and those derived or endorsed by EPA and Ecology (as evidenced by their use in either EcoSSLs or the MTCA [WAC 173-340]). The primary literature sources used were EcoSSLs. The toxicity studies used were selected initially from the following sources, which have been listed in order of preference:

- OSWER Directives
  - 9285.7-56, *Ecological Soil Screening Levels for Dieldrin: Interim Final*
  - 9285.7-57, *Ecological Soil Screening Levels for DDT and Metabolites: Interim Final*
  - 9285.7-60, *Ecological Soil Screening Level for Aluminum: Interim Final*
  - 9285.7-61, *Ecological Soil Screening Levels for Antimony: Interim Final*
  - 9285.7-62, *Ecological Soil Screening Levels for Arsenic: Interim Final*
  - 9285.7-63, *Ecological Soil Screening Levels for Barium: Interim Final*
  - 9285.7-64, *Ecological Soil Screening Levels for Beryllium: Interim Final*
  - 9285.7-65, *Ecological Soil Screening Levels for Cadmium: Interim Final*
  - 9285.7-66, *Ecological Soil Screening Levels for Chromium: Interim Final*
  - 9285.7-67, *Ecological Soil Screening Levels for Cobalt: Interim Final*
  - 9285.7-68, *Ecological Soil Screening Levels for Copper: Interim Final*
  - 9285.7-69, *Ecological Soil Screening Level for Iron: Interim Final*
  - 9285.7-70, *Ecological Soil Screening Levels for Lead: Interim Final*
  - 9285.7-71, *Ecological Soil Screening Levels for Manganese: Interim Final*
  - 9285.7-72, *Ecological Soil Screening Levels for Selenium: Interim Final*
  - 9285.7-73, *Ecological Soil Screening Levels for Zinc: Interim Final*
  - 9285.7-75, *Ecological Soil Screening Levels for Vanadium: Interim Final*

- 9285.7-76, *Ecological Soil Screening Levels for Nickel: Interim Final*
- 9285.7-77, *Ecological Soil Screening Levels for Silver: Interim Final*
- 9285.7-78, *Ecological Soil Screening Levels for Polycyclic Aromatic Hydrocarbons (PAHs): Interim Final*

- MTCA (WAC 173-340), Table 749-5
- Other available literature—primarily *Toxicological Benchmarks for Wildlife: 1996 Revision* (ES/ER/TM-86/R3)
- NOAEL and LOAEL values selected for chemicals and reported in Integrated Risk Information System
- NOAEL and LOAEL values presented in wildlife toxicity assessments developed by the United States Army Center for Health Promotion and Preventive Medicine

An EPA panel of experts developed a process for reviewing and selecting TRVs for EcoSSL development for wildlife. The process was to select NOAELs to develop EcoSSLs for wildlife. Selected TRVs were either the highest NOAEL for population-level effects (for example, survival, growth, and reproduction endpoints) below the lowest LOAEL for population-level effects or the geometric mean of NOAELs, depending on the number and quality of data available. Selection of the TRVs for development of Hanford SSLs and PRGs attempted to use the work of this expert panel. Thus, for analytes that EPA has developed EcoSSLs for birds and mammals, those same NOAELs were used for wildlife SSL and PRG development for Hanford (see *Tier 1 Risk-Based Soil Concentrations Protective of Ecological Receptors at the Hanford Site* [CHPRC-00784], in Appendix H of the 100-K RI/FS Report [DOE/RL-2010-97], for a full detailed description). In some cases, the NOAEL-based TRV for the EcoSSL was the highest NOAEL below the lowest LOAEL identified for studies evaluating survival, growth, and reproduction endpoints. In these cases, the paired LOAEL from the study was selected as the LOAEL for Hanford SSL and PRG development. In other cases, the geometric mean of the NOAELs for growth and reproduction endpoints was selected to derive the EcoSSL. In these cases, the LOAEL for Hanford SSL and PRG development was selected as the lowest LOAEL from the EcoSSL dataset above the geometric mean NOAEL.

The only exception to this TRV selection process was for the arsenic TRV for avian receptors, in which case the selected study was not identified and reviewed by the EPA panel. The study "Main and Interactive Effects of Arsenic and Selenium on Mallard Reproduction and Duckling Growth and Survival" (Stanley et al., 1994), conducted by USFWS at Patuxent wildlife research center over a 92- to 173-day period, resulted in both a NOAEL and a LOAEL for reproductive effects. The EcoSSL document considered nine studies on the effects of arsenic to have sufficient quality to consider in developing the avian SSL. All of these studies were conducted over 70 days or less. "Arsenic Residues in Eggs from Laying Hens Fed with a Diet Containing Arsenic(III) Oxide" (Holcman and Stibilj, 1997) presented an unbound NOAEL that was selected because it was the lowest value. "Main and Interactive Effects of Arsenic and Selenium on Mallard Reproduction and Duckling Growth and Survival" (Stanley et al., 1994) was conducted by a reliable research group over a much longer time frame and produced bound results (that is, the NOAEL was bound by a LOAEL). The intent of the EcoSSLs is to provide a value that can provide a reliable conservative screen, whereas TRV selection for this ERA is for use in PRG development for remedial decisions. Given all of this information, the NOAEL and LOAEL from "Main and Interactive Effects of Arsenic and Selenium on Mallard Reproduction and Duckling Growth and Survival" (Stanley et al., 1994) were selected over the EcoSSL recommendation.

For analytes lacking EcoSSLs, other primary and secondary sources of studies were used. Whenever possible, the primary literature sources were obtained and evaluated. Appropriate toxicity studies were selected from these sources based on the following criteria:

- Studies were of chronic exposures or exposures during a critical stage of life (for example, reproduction).
- Exposure was oral through food ingestion to ensure data were representative of oral exposures expected for wildlife in the field.
- Emphasis was placed on studies of reproductive effects to ensure relevancy to population-level effects.
- Studies presented adequate information to evaluate and determine the magnitude of exposure and effects (or no-effects concentrations).

Specifically, toxicity studies were selected to serve as the TRV if exposure was chronic or was measured during a critical life stage, the dosing regime was sufficient to identify both a NOAEL and a LOAEL, and the study considered ecologically relevant effects (for example, growth, reproduction, or survival). If multiple studies for a given COPC met these criteria, the study generating the lowest reliable toxicity value was selected to be the TRV.

The full explanations of the TRVs selected, the method of calculating the SSLs and PRGs, and the resulting SSLs and PRGs are included in Appendix H of the 100-K RI/FS Report (DOE/RL-2010-97) (*Tier 1 Risk-Based Soil Concentrations Protective of Ecological Receptors at the Hanford Site* [CHPRC-00784] for SSLs and *Tier 2 Risk-Based Soil Concentrations Protective of Ecological Receptors at the Hanford Site* [CHPRC-01311] for PRGs).

7.3.2 Exposure Assessment

The exposure assessment for plants and invertebrates, wildlife, and radionuclides is summarized below. Additionally, a brief description of SSL and PRG development as a relationship between the effects assessment described in Section 7.3.1 and the exposure assessment is provided. For wildlife, this description is provided with sections for nonradionuclide SSLs, radionuclide SSLs, and nonradionuclide PRGs, which include details in the estimation of exposure.

7.3.2.1 Exposure Assessment for Terrestrial Plants and Soil Invertebrates

Terrestrial plants and soil invertebrates experience exposure primarily through the soil in which they live. This exposure occurs as a consequence of living in a contaminated medium (that is, receptors are directly exposed to COPCs). Although other exposure pathways (for example, dietary exposure for invertebrates or foliar uptake for plants) may contribute to total exposure for these receptors, exposure through the soil predominates. Consequently, estimates of exposure for terrestrial plants and soil invertebrates are represented by the concentration of COPCs in the soil (mg/kg). As such, the concentrations of chemicals in soil that correspond to adverse effects described in the effects assessment (Section 7.3.1) were also assigned as the SSLs. The assumption is the same for PRG selection for plants and invertebrates but is described separately in Section 7.3.4.

7.3.2.2 Exposure Assessment for Wildlife (Birds and Mammals)

In contrast to plants and soil invertebrates, birds and mammals experience chemical exposure through multiple pathways, including the ingestion of surface water, sediment/soil, biotic media (food), inhalation, and dermal contact. Modeling is often used to assess exposure via these multiple exposure pathways.

The end product, or exposure estimate, for birds and mammals is a dose estimate that quantifies the amount of chemical in milligrams per kilogram receptor body weight per day [mg/kg/day].

Following is the general form of the model used to estimate exposure of birds and mammals to chemicals in environmental media (*Ecological Risk Assessment for Contaminated Sites* [Suter et al., 2000]):

$$E_t = E_o + E_d + E_i$$

where:

E_t = total chemical exposure experienced by wildlife

E_o = oral exposure

E_d = dermal exposure

E_i = inhalation exposure

Oral exposure occurs through the consumption of contaminated food, water, or sediment/soil; dermal exposure occurs when contaminants are absorbed directly through the skin; and inhalation exposure occurs when volatile compounds or fine particulates are inhaled into the lungs. Although methods are available for assessing dermal exposure to humans (*Dermal Exposure Assessment: Principles and Applications* [EPA/600/8-91/011B]), data necessary to estimate dermal exposure generally are not available for wildlife (*Wildlife Exposure Factors Handbook*, Vol. 1 and II [EPA/600/R-93/187]). Similarly, methods and data necessary to estimate wildlife inhalation exposures are poorly developed (*Wildlife Exposure Factors Handbook*, Vol. I and II [EPA/600/R-93/187]) or limited.[4] Additionally, a wildlife receptor's exposure to contaminants by inhalation and dermal contact usually contributes little to its overall exposure. Dermal exposure also is likely to be low, even in burrow dwelling animals, because of the presence of protective dermal layers (for example, feathers, fur, or scales). Therefore, for the purposes of developing the SSL values, both dermal and inhalation exposure were assumed to be negligible[4]. Therefore, only oral exposures via ingestion of soil and food were included in the development of risk-based concentrations for birds and mammals.

Large mammalian wildlife using the upland 100-D/H Areas move down to the Columbia River riparian area and drink from the freshwater seeps and from the Columbia River. Bats and birds frequenting or residing in these areas also can use the seeps along the Columbia River to meet their daily needs. A semi-quantitative evaluation of the ingestion of seep water was performed and is discussed with the risk characterization in Section 7.4.4.

Total chemical exposure experienced by wildlife (E_t) is assumed to be equal to oral exposure (E_o). By replacing E_o with a generalized exposure model modified from *Ecological Risk Assessment for Contaminated Sites* (Suter et al., 2000) to include only soil and food ingestion, the previous equation was rewritten as follows:

[4] If the CSM had indicated that VOCs are a significant COPEC, focused analyses of the inhalation pathway may have been warranted, but VOCs were not significant at 100-D or 100-H. Risk-based concentrations or PRGs for this pathway, however, are beyond the scope of this report.

$$E_t = \left[\left[\sum_{i=1}^{n} B_{ij} \times P_i \times FIR\right] + \left[Soil_j \times P_s \times FIR\right]\right] x\, AUF$$

where:

Et =

total exposure (mg/kg/day)

$Soil_j$ = chemical concentration in soil (mg/kg dry weight)

P_s = soil ingestion rate as proportion of diet (unitless)

FIR = food intake rate (kg food/kg body weight/day, dry weight)

B_{ij} = chemical concentration in biota type (i) (mg/kg, dry weight)

P_i = proportion of biota type (i) in diet (unitless)

AUF = area use factor (area of site/home range [Appendix H, Table H-6] of receptor) (unitless)

The bird and mammal effects data (Section 7.3.1.1) were combined with the wildlife exposure model to calculate avian/mammal SSLs and PRGs for nonradionuclides. These SSLs and PRGs consist of soil concentrations associated with estimated dietary exposures equivalent to a selected effect level and were calculated using the following basic equation:

$$1 = \frac{TRV}{\sum (SSLorPRG \times DFI) \times \left[(Frac_v \times C_v) + (Frac_i \times C_i) + (Frac_m \times C_m) + (Frac_s)\right]}$$

where:

TRV = toxicity reference value (mg/kg body weight/day)

SSL = wildlife soil screening level (mg/kg)

PRG = wildlife preliminary remediation goal (mg/kg)

$Frac_v$ = fraction of diet represented by vegetation (unitless)

DFI = daily ingestion rate of all food items (kg/kg body weight/day dry weight)

C_v = concentration in vegetation tissue (mg/kg dry weight)

$Frac_i$ = fraction of diet represented by terrestrial invertebrates (unitless)

C_i = concentration in soil invertebrate tissue (mg/kg dry weight)

$Frac_m$ = fraction of diet represented by small mammals/birds (unitless)

C_m = concentration in small mammal tissue (mg/kg dry weight)

$Frac_s$ = fraction of diet represented by incidentally ingested soil (unitless)

The TRV denotes the level of toxicity of the chemical, as reported from literature sources. The wildlife SSLs and PRGs use the LOAELs, which is consistent with protecting ecological receptors at the population and community levels. The daily ingestion rate and dietary fractions are specific to bird and mammal receptors identified for the upland environment of the Hanford Site. The chemical concentration in the food item (vegetation, soil invertebrate, and small mammal) is estimated by BAFs or bioaccumulation regression models to extrapolate to the food source. This equation is solved for wildlife

SSLs or PRGs using the Excel goal-seek tool, such that exposure (the denominator) equals the TRV (the numerator).

For the purposes of this risk assessment, the LOAEL-based wildlife SSLs and wildlife PRGs were used to evaluate residual risks at the 100-D/H Source OUs remediated waste sites. The SSLs and PRGs were compared to EPCs developed for the 100-D/H OUs as described in Section 7.4.1.

Wildlife Exposure Factors. Within the exposure models described above, species-specific exposure parameters are required to estimate exposure. These include body weight, food ingestion rate, diet composition represented by dietary fractions, and percent or fraction of diet as incidental soil ingestion. The following assumptions were part of the calculation of wildlife exposures used to develop the wildlife SSLs and wildlife PRGs:

- For SSL and PRG development, wildlife was assumed to forage exclusively within the waste site being evaluated, resulting in an AUF of 1. In other words, the resulting SSLs and PRGs did not account for wildlife home range instead of assuming that prey tissue concentrations from food obtained outside the waste site boundaries might contain lower concentrations of contaminants. This assumption is discussed in more detailed in the risk conclusions and the SMDP discussed in Section 7.6, including accounting for home range and development of site-specific AUFs as warranted.
- Incidental soil ingestion was not included as part of the total dietary composition, but instead was added to the total; for calculation purposes, it was treated as a percentage of total dietary intake.
- All animals were assumed to be year-round residents, and migration away from areas contaminated with COPCs was not assumed.
- Bioavailability of analytes was assumed equivalent to the chemical form used for developing TRVs in the toxicity studies.
- 100 percent of the estimated soil concentrations (EPCs) were assumed bioavailable for uptake into tissues within the exposure models.

The exposure parameters and source references used for each representative receptor species are summarized in Appendix H (Table H-3). All weight-based exposure parameters are listed on a dry-weight basis. Species-specific biological information was unavailable for some parameters. When this occurred, allometric equations that express general biological relationships for broader classes of animals were used to estimate the exposure parameters ("Food Requirements of Wild Animals: Predictive Equations for Free-Living Mammals, Reptiles, and Birds" [Nagy, 2001]). These allometric conversions are detailed in *Tier 1 Risk-Based Soil Concentrations Protective of Ecological Receptors at the Hanford Site* (CHPRC-00784) included in Appendix H of the 100-K RI/FS (DOE/RL-2010-97).

Estimation of Bioaccumulation into Food Items. A major component of the desktop food-chain model described above is modeling the concentration of contaminates within the prey consumed by wildlife within the waste sites being evaluated. This modeled dose received through ingesting food was considered in the final estimate of the soil concentration that represents a toxic threshold (that is, the SSL or PRG). Bioaccumulation models and assumptions used within the calculation of wildlife SSLs and PRGs are described below. While some of them are the same as those within MTCA ("Site-Specific Terrestrial Ecological Evaluation Procedures" [WAC 173-340-7493]) promulgated in 2001, advancements in estimating bioaccumulation into food items were published as part of the initial (2003) and subsequent updates (2005 and 2007) to *Guidance for Developing Ecological Soil Screening Levels*

(OSWER Directive 92857-55). These models and assumptions represent the most recent equations used in ERA and are now the standard of practice; thus they were employed for developing SSLs and PRGs for Hanford.

- **Estimating Prey Tissue Concentration for SSLs**—The concentrations of COPCs in each food item were estimated rather than measured. For the purposes of exposure estimation, partitioning of analytes from environmental media to prey was estimated from literature values and models. The models presented in the EPA EcoSSLs methodology (*Guidance for Developing Ecological Soil Screening Levels* [OSWER Directive 9285.7-55]) were used preferentially for estimation of bioaccumulation into biota from soil. Consistent with the approach used for the EcoSSLs, regression-based models (if available) and median BAFs from the source selected by EPA were used. In the absence of applicable bioaccumulation models, a default value of 1 was assumed. In all cases, it was assumed that tissue uptake occurs under steady-state conditions. Bioaccumulation models used to derive wildlife SSLs are presented in Table H-4 (Appendix H). The wildlife SSLs are presented in Table 7-2 for radionuclides and Table 7-3 for nonradionuclides.

- **Estimating Prey Tissue Concentration for PRGs**—Development of the PRGs for birds and mammals focused on the integration of available site-specific bioaccumulation data for plants, terrestrial arthropods, and small mammals with data from the existing bioaccumulation models (that is, those from *Guidance for Developing Ecological Soil Screening Levels* [OSWER Directive 9285.7-55]) that were used to develop the EcoSSLs in order to develop a set of more site-specific and site-relevant bioaccumulation models.[5] The following Hanford Site-specific and literature-based datasets were used to develop these bioaccumulation models presented in Appendix H (Table H-5):

Hanford Site-specific bioaccumulation data have been collected in support of the RCBRA (DOE/RL-2007-21) and other projects at the site. Data representing tissue from terrestrial plants (foliage shoots, and other aboveground parts of grasses, shrubs, and trees), small mammals (whole individual mice or composites of multiple whole mice), and terrestrial arthropods (whole individual invertebrates or composites of multiple whole invertebrates) and collocated soil data were extracted from HEIS. Only paired samples in which the target analytes were detected in both tissue and soil were retained for the bioaccumulation database; observations that were nondetects in either the soil or tissue of a sample pair were excluded from consideration.

Literature Derived Bioaccumulation Data for Plants and Small Mammals. Data from previously developed and published bioaccumulation models for plants and small mammals were used to augment the Hanford Site-specific data. Specifically, the plant bioaccumulation databases from *Empirical Models for the Uptake of Inorganic Chemicals from Soil by Plants* (BJC/OR-133) and "Uptake of Inorganic Chemicals from Soil by Plant Leaves: Regressions of Field Data" (Efroymson et al., 2001) were used. In addition, the small mammal bioaccumulation database from *Development and Validation of Bioaccumulation Models for Small Mammals* (ES/ER/TM-219) was used. These data also represent the primary bioaccumulation data for inorganics integrated into *Guidance for Developing Ecological Soil Screening Levels* (OSWER Directive 9285.7-55). Electronic copies of the original databases were obtained from the authors to facilitate integration with Hanford Site-specific data.

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[5] These bioaccumulation models are defined as more site-specific and site-relevant because they are based on both site-specific data and data from published literature sources. This combining of Hanford Site-specific and literature data was performed to maximize utility of the Hanford Site-specific data collected over comparatively narrow concentration ranges by expanding the dataset to include literature data collected across a wider concentration range.

Table 7-3. Wildlife (Birds and Mammals) SSLs for Nonradionuclides

Group	Soil Constituent	Units	NOAEL-Based Site-Specific SSLs									LOAEL-Based Site-Specific SSLs								
			California Quail	Meadowlark	Killdeer	Red-Tailed Hawk	Great Basin Pocket Mouse	Deer Mouse	Grasshopper Mouse	Badger	NOAEL Lowest	California Quail	Meadowlark	Killdeer	Red-Tailed Hawk	Great Basin Pocket Mouse	Deer Mouse	Grasshopper Mouse	Badger	LOAEL Lowest
Metals	Aluminum	mg/kg	22,019.89	18,601.60	4,920.58	61,782.28	687.18	270.79	380.00	710.10	270.79	--	--	--	--	6,871.78	2,707.93	3,798.53	7,100.99	2,707.93
	Antimony	mg/kg	--	--	--	--	8.82	0.66	0.60	16.66	**0.60**	--	--	--	--	96.64	6.61	5.98	166.64	**5.98**
	Arsenic, Total all valence states	mg/kg	1,799.96	1,980.65	425.03	10,343.53	264.76	104.92	170.75	549.25	**104.92**	8,103.60	10,558.64	2,131.65	45,439.07	459.41	189.51	318.36	880.60	**189.51**
	Arsenic (III)	mg/kg	1,799.96	1,980.65	425.03	10,343.53	264.76	104.92	170.75	549.25	**104.92**	8,103.60	10,558.64	2,131.65	45,439.07	459.41	189.51	318.36	880.60	**189.51**
	Arsenic (V)	mg/kg	1,799.96	1,980.65	425.03	10,343.53	264.76	104.92	170.75	549.25	**104.92**	8,103.60	10,558.64	2,131.65	45,439.07	459.41	189.51	318.36	880.60	**189.51**
	Barium	mg/kg	1,228.88	1,270.92	659.91	14,442.04	2,081.99	1,889.09	4,605.48	18,842.90	**659.91**	2,463.67	2,547.94	1,323.00	28,953.51	3,469.98	3,148.48	7,675.79	31,404.83	**1,323.00**
	Beryllium	mg/kg	--	--	--	--	13.95	17.96	100.83	282.51	**13.95**	--	--	--	--	--	--	--	--	--
	Bismuth	mg/kg	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--
	Boron	mg/kg	63.94	86.46	139.66	796.74	39.76	49.95	284.01	766.77	**39.76**	222.00	300.21	484.92	2,766.47	132.92	166.97	949.39	2,563.21	**132.92**
	Cadmium	mg/kg	151.07	2.77	0.89	1,374.94	76.19	1.47	1.30	455.47	**0.89**	277.98	5.08	1.63	2,335.22	2,065.15	27.54	23.57	5,228.02	**1.63**
	Chromium (total)	mg/kg	334.34	96.52	36.52	1,286.46	320.38	74.72	77.98	752.34	**36.52**	349.42	100.88	38.17	1,355.27	1,284.17	299.49	312.56	3,535.56	**38.17**
	Chromium (III)	mg/kg	334.34	96.52	36.52	1,286.46	320.38	74.72	77.98	752.34	**36.52**	349.42	100.88	38.17	1,355.27	1,284.17	299.49	312.56	3,535.56	**38.17**
	Chromium (VI)	mg/kg	--	--	--	--	1,233.45	287.66	300.21	3,379.86	**287.66**	--	--	--	--	5,339.59	1,245.29	1,299.60	16,583.35	**1,245.29**
	Cobalt	mg/kg	1,425.33	305.33	108.78	1,601.40	2,174.40	260.93	250.06	1,346.06	**108.78**	1,460.92	312.95	111.50	1,632.76	3,233.42	388.02	371.85	1,868.88	**111.50**
	Copper	mg/kg	485.15	85.30	35.84	3,727.67	872.95	99.95	109.38	2,640.09	**35.84**	1,914.48	271.87	107.07	13,020.77	1,893.59	175.77	182.42	4,672.45	**107.07**
	Lead	mg/kg	247.02	48.68	15.51	978.92	1,204.17	151.05	153.49	2,005.03	**15.51**	537.35	114.72	35.58	2,433.19	2,544.20	331.98	336.43	4,108.35	**35.58**
	Lithium	mg/kg	--	--	--	--	3,189.29	1,258.37	1,749.15	257.42	**257.42**	--	--	--	--	6,378.59	2,516.73	3,498.30	514.83	**514.83**
	Manganese	mg/kg	16,368.56	24,183.51	9,588.41	113,951.05	4,227.40	4,115.32	18,430.08	20,464.11	**4,115.32**	31,822.67	48,820.09	19,635.66	221,536.11	5,828.07	5,798.20	27,720.38	28,212.66	**5,798.20**
	Mercury	mg/kg	3.09	0.35	0.04	24.60	0.49	0.03	0.03	8.67	**0.03**	35.51	21.26	3.59	133.86	7.98	1.87	3.25	43.36	**1.87**
	Molybdenum	mg/kg	34.51	27.03	17.90	97.66	1.67	1.40	2.77	7.12	**1.40**	345.10	270.35	179.02	976.56	16.67	13.96	27.66	71.20	**13.96**
	Nickel	mg/kg	1,080.61	79.37	30.84	6,037.33	303.26	17.77	16.29	637.16	**16.29**	1,911.91	136.40	52.86	11,078.24	675.83	35.81	32.58	1,438.31	**32.58**
	Selenium	mg/kg	5.57	3.72	1.72	157.63	2.05	1.19	1.83	32.28	**1.19**	10.47	8.17	4.29	417.07	2.97	1.90	3.19	59.89	**1.90**
	Silver	mg/kg	345.30	12.76	4.96	2,043.71	1,441.77	34.55	30.00	3,096.93	**4.96**	3,452.99	127.57	49.62	20,437.07	14,417.68	345.55	299.96	30,969.35	**49.62**
	Strontium	mg/kg	--	--	--	--	9,442.09	4,849.22	6,476.09	4,227.51	**4,227.51**	--	--	--	--	--	--	--	--	--
	Thallium	mg/kg	--	--	--	--	5.09	1.84	2.43	2.63	**1.84**	--	--	--	--	25.45	9.21	12.16	13.14	**9.21**
	Tin	mg/kg	82.17	127.93	231.25	1,852.26	186.81	251.72	2,690.84	5,107.00	**82.17**	204.21	317.95	574.73	4,603.40	279.41	376.51	4,024.76	7,638.67	**204.21**
	Uranium	mg/kg	2,501.56	2,690.66	785.38	18,729.66	609.72	393.49	748.09	1,694.30	**393.49**	--	--	--	--	1,217.46	785.70	1,493.73	3,383.09	**785.70**
	Vanadium	mg/kg	66.97	58.21	15.56	268.46	1,363.15	577.09	834.81	1,863.83	**15.56**	133.95	116.42	31.13	536.92	2,723.03	1,152.80	1,667.60	3,723.18	**31.13**
	Zinc	mg/kg	4,973.24	714.12	66.60	70,825.06	4,611.56	633.13	794.24	38,590.44	**66.60**	5,015.41	725.66	67.80	71,293.85	4,660.81	643.92	810.09	38,865.84	**67.80**
General Inorganics	Ammonia/ Ammonium	mg/kg	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--
	Chloride	mg/kg	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--

Table 7-3. Wildlife (Birds and Mammals) SSLs for Nonradionuclides

Group	Soil Constituent	Units	NOAEL-Based Site-Specific SSLs									LOAEL-Based Site-Specific SSLs								
			California Quail	Meadowlark	Killdeer	Red-Tailed Hawk	Great Basin Pocket Mouse	Deer Mouse	Grasshopper Mouse	Badger	NOAEL Lowest	California Quail	Meadowlark	Killdeer	Red-Tailed Hawk	Great Basin Pocket Mouse	Deer Mouse	Grasshopper Mouse	Badger	LOAEL Lowest
	Cyanide	mg/kg	--	--	--	--	27,970.79	20,692.77	78,122.51	38,060.72	**20,692.77**	--	--	--	--	--	--	--	--	**--**
	Fluoride	mg/kg	1,492.00	2,812.00	556.00	9,206.00	9,824.70	8,216.34	35,672.53	17,379.40	**556.00**	6,123.00	11,539.00	2,281.00	37,771.00	16,520.65	13,816.13	59,984.89	29,224.21	**2,281.00**
	Iodine	mg/kg	--	--	--	--	159.37	183.40	1,557.90	759.00	**159.37**	--	--	--	--	1,593.68	1,834.00	15,579.01	7,589.98	**1,593.68**
	Nitrate/Nitrite	mg/kg	--	--	--	--	206,421.95	152,710.84	576,537.26	280,884.80	**152,710.84**	--	--	--	--	460,072.60	340,361.45	1,284,984.44	626,035.16	**340,361.45**
	Phosphate	mg/kg	--	--	--	--	--	--	--	--	**--**	--	--	--	--	--	--	--	--	**--**
	Sulfate/Sulfite	mg/kg	--	--	--	--	--	--	--	--	**--**	--	--	--	--	--	--	--	--	**--**
	Total Organic Carbon	%	--	--	--	--	--	--	--	--	**--**	--	--	--	--	--	--	--	--	**--**
Volatile Organics	1,1-dichloroethane	mg/kg	3,614.96	217.36	82.67	13,954.67	20,357.19	573.95	502.38	22,894.29	**82.67**	7,229.93	434.72	165.33	27,909.35	--	--	--	--	**165.33**
	1,1-dichloroethene	mg/kg	3,614.96	217.02	82.54	11,432.85	12,214.32	343.83	300.94	12,238.43	**82.54**	7,229.93	434.04	165.08	22,865.71	--	--	--	--	**165.08**
	1,1,1-trichloroethane	mg/kg	3,614.96	216.69	82.42	8,935.68	407,143.89	11,443.76	10,015.86	349,074.28	**82.42**	7,229.93	433.38	164.84	17,871.35	--	--	--	--	**164.84**
	1,1,2-trichloroethane	mg/kg	3,614.96	217.23	82.62	12,031.27	407,143.89	11,472.08	10,041.41	420,572.26	**82.62**	7,229.93	434.46	165.24	24,062.54	--	--	--	--	**165.24**
	1,1,2,2-tetrachloroethane	mg/kg	3,614.96	216.67	82.41	9,548.86	3,635.79	102.18	89.43	3,255.35	**82.41**	7,229.93	433.33	164.82	19,097.72	36,357.95	1,021.81	894.31	32,553.53	**164.82**
	1,2-dichlorobenzene	mg/kg	87.76	90.99	82.05	4,343.45	282.35	294.23	854.29	17,612.27	**82.05**	175.53	181.99	164.11	8,686.89	--	--	--	--	**164.11**
	1,2-dichloroethane (DCA)	mg/kg	3,614.96	221.88	84.32	16,083.77	20,357.19	585.79	513.07	24,709.56	**84.32**	7,229.93	443.75	168.64	32,167.55	--	--	--	--	**168.64**
	1,3-dichlorobenzene	mg/kg	96.13	95.93	82.03	4,051.47	309.75	313.74	853.99	16,651.92	**82.03**	192.26	191.85	164.05	8,102.94	--	--	--	--	**164.05**
	2-butanone (Methyl Ethyl Ketone/MEK)	mg/kg	2,101.72	1,040.62	312.32	11,538.19	721,051.83	159,713.07	176,661.35	970,850.97	**312.32**	21,017.23	10,406.18	3,123.19	115,381.89	1,861,054.73	412,223.86	455,967.83	2,505,793.22	**3,123.19**
	2-hexanone	mg/kg	2,101.72	548.27	185.63	9,653.17	2,035.72	243.60	236.58	2,511.81	**185.63**	21,017.23	5,482.66	1,856.29	96,531.69	14,697.89	1,758.76	1,708.10	18,135.29	**1,708.10**
	Benzene	mg/kg	8,554.00	513.00	195.00	27,053.00	285.00	8.02	7.02	285.56	**7.02**	--	--	--	--	2,850.01	80.23	70.22	2,855.63	**70.22**
	Butanol	mg/kg	--	--	--	--	50,892.99	2,906.16	2,625.73	67,048.62	**2,625.73**	--	--	--	--	203,571.95	11,624.66	10,502.90	268,194.47	**10,502.90**
	Carbon Tetrachloride	mg/kg	3,614.96	216.30	82.28	7,382.46	6,514.30	182.77	159.96	4,903.54	**82.28**	7,229.93	432.60	164.56	14,764.92	--	--	--	--	**164.56**
	Chlorobenzene	mg/kg	3,614.96	216.31	82.28	6,672.38	7,939.31	222.77	194.96	5,560.69	**82.28**	7,229.93	432.63	164.57	13,344.76	15,756.47	442.11	386.93	11,035.82	**164.57**
	Chloroform	mg/kg	3,614.96	217.17	82.60	13,002.94	6,107.16	172.03	150.58	6,600.19	**82.60**	7,229.93	434.33	165.19	26,005.89	16,692.90	470.22	411.58	18,040.51	**165.19**
	Cis-1,2-dichloroethylene	mg/kg	3,614.96	217.05	82.55	13,446.00	18,402.90	518.11	453.49	20,270.86	**82.55**	7,229.93	434.10	165.11	26,892.00	--	--	--	--	**165.11**
	Dichloromethane (Methylene Chloride)	mg/kg	3,614.96	217.95	82.88	17,281.03	2,381.79	67.33	58.94	2,999.00	**58.94**	7,229.93	435.91	165.77	34,562.06	20,357.19	575.50	503.78	25,632.44	**165.77**
	Ethyl Benzene	mg/kg	159.00	182.00	194.00	12,721.00	342.00	383.68	1,357.00	33,025.00	**159.00**	--	--	--	--	1,027.00	1,151.00	4,073.00	99,076.00	**1,027.00**
	Methyl Isobutyl Ketone	mg/kg	2,101.72	572.87	192.73	10,211.45	721,051.83	90,039.69	87,995.70	915,291.62	**192.73**	21,017.23	5,728.72	1,927.26	102,114.45	1,861,054.73	232,394.92	227,119.32	2,362,392.98	**1,927.26**
	n-butyl Benzene	mg/kg	301.00	263.25	193.00	7,857.00	529.53	484.77	1,091.54	18,135.28	**193.00**	--	--	--	--	1,588.60	1,454.30	3,274.62	54,405.85	**1,454.30**
	Tetrachloroethylene	mg/kg	3,614.96	215.72	82.07	7,733.34	570.00	15.95	13.96	443.18	**13.96**	7,229.93	431.44	164.13	15,466.68	2,850.01	79.75	69.79	2,215.89	**69.79**
	Toluene	mg/kg	8,554.00	512.13	195.00	17,200.00	21,171.48	594.37	520.19	15,763.32	**195.00**	--	--	--	--	211,714.82	5,943.66	5,201.85	157,633.17	**5,201.85**

Table 7-3. Wildlife (Birds and Mammals) SSLs for Nonradionuclides

Group	Soil Constituent	Units	NOAEL-Based Site-Specific SSLs									LOAEL-Based Site-Specific SSLs								
			California Quail	Meadowlark	Killdeer	Red-Tailed Hawk	Great Basin Pocket Mouse	Deer Mouse	Grasshopper Mouse	Badger	NOAEL Lowest	California Quail	Meadowlark	Killdeer	Red-Tailed Hawk	Great Basin Pocket Mouse	Deer Mouse	Grasshopper Mouse	Badger	LOAEL Lowest
	Trans-1,2-dichloroethylene	mg/kg	3,614.96	217.05	82.55	11,881.41	18,402.90	518.11	453.49	18,869.15	**82.55**	7,229.93	434.10	165.11	23,762.81	--	--	--	--	**165.11**
	Trichloroethylene (TCE)	mg/kg	3,614.96	216.77	82.45	7,497.82	285.00	8.01	7.01	216.87	**7.01**	7,229.93	433.53	164.90	14,995.63	2,850.01	80.13	70.14	2,168.73	**70.14**
	Xylene	mg/kg	149.00	174.99	194.00	13,419.00	422.29	480.57	1,787.19	45,266.25	**149.00**	--	--	--	--	825.70	939.65	3,494.50	88,509.43	**825.70**
Polycyclic Aromatic Hydrocarbons	Acenaphthene	mg/kg	6,830.60	284.90	109.59	38,361.66	71,250.18	1,396.31	1,210.97	96,952.35	**109.59**	68,306.01	2,849.00	1,095.94	383,616.62	142,500.36	2,792.63	2,421.93	193,904.70	**1,095.94**
	Acenaphthylene	mg/kg	3,505.79	18.59	7.36	38,361.66	24,320.69	91.39	77.90	96,952.35	**7.36**	43,765.65	186.19	73.61	383,616.62	54,131.54	182.92	155.81	193,904.70	**73.61**
	Anthracene	mg/kg	3,405.23	169.72	67.83	38,361.66	178,810.67	4,783.60	4,213.42	554,013.42	**67.83**	43,404.77	1,716.22	678.31	383,616.62	--	--	--	--	**678.31**
	Benzo(a)pyrene	mg/kg	47.19	5.97	2.41	767.23	60.38	8.08	7.64	554.01	**2.41**	--	--	--	--	634.65	81.16	76.43	5,540.13	**76.43**
	Benzo(a)anthracene	mg/kg	117.98	5.21	2.03	767.23	306.81	7.26	6.40	554.01	**2.03**	--	--	--	--	3,635.53	73.37	64.00	5,540.13	**64.00**
	Benzo(b)fluoranthene	mg/kg	22.46	3.04	1.27	767.23	24.68	4.08	3.92	554.01	**1.27**	--	--	--	--	246.75	40.84	39.23	5,540.13	**39.23**
	Benzo(ghi)perylene	mg/kg	12.19	2.64	1.12	767.23	12.60	3.47	3.47	554.01	**1.12**	--	--	--	--	88.95	32.35	34.70	5,540.13	**32.35**
	Benzo[k]fluoranthene	mg/kg	136.25	3.25	1.27	767.23	405.73	4.56	3.92	554.01	**1.27**	--	--	--	--	4,069.49	45.63	39.23	5,540.13	**39.23**
	Chrysene	mg/kg	117.98	3.65	1.43	767.23	306.81	5.09	4.45	554.01	**1.43**	--	--	--	--	3,635.53	51.38	44.52	5,540.13	**44.52**
	Dibenz(ah)anthracene	mg/kg	43.63	3.54	1.42	767.23	54.29	4.86	4.41	554.01	**1.42**	--	--	--	--	542.86	48.58	44.13	5,540.13	**44.13**
	Fluoranthene	mg/kg	14.85	2.54	1.09	767.23	1,957.42	420.68	419.58	69,251.68	**1.09**	--	--	--	--	3,914.85	841.35	839.16	138,503.35	**839.16**
	Fluorene	mg/kg	6,830.60	44.59	17.54	38,361.66	50,892.99	156.71	133.55	69,251.68	**17.54**	68,306.01	445.91	175.36	383,616.62	101,785.97	313.43	267.10	138,503.35	**175.36**
	Indeno[1,2,3-cd]pyrene	mg/kg	48.73	2.90	1.15	767.23	62.64	4.00	3.57	554.01	**1.15**	--	--	--	--	626.38	40.03	35.67	5,540.13	**35.67**
	2-Methylnaphthalene	mg/kg	5.04	5.69	154.74	38,361.66	5.02	5.47	500.36	27,866.87	**5.02**	8.37	9.46	1,547.37	383,616.62	6.01	6.55	1,132.02	63,046.73	**6.01**
	Naphthalene	mg/kg	33.98	36.92	415.86	38,361.66	33.32	36.20	116.06	27,700.67	**33.32**	339.83	369.21	378.05	383,616.62	99.95	108.61	348.19	83,102.01	**99.95**
	Phenanthrene	mg/kg	4,329.17	235.76	94.31	38,361.66	301,134.27	6,731.36	5,919.25	554,013.42	**94.31**	56,061.03	2,405.58	943.13	383,616.62	--	--	--	--	**943.13**
	Pyrene	mg/kg	10.67	3.88	1.86	767.23	825.29	360.01	436.37	41,551.01	**1.86**	--	--	--	--	1,375.49	600.01	727.29	69,251.68	**600.01**
	Total PAHs	mg/kg	--	--	--	--	--	--	--	--	**--**	--	--	--	--	--	--	--	--	**--**
	Low MW PAHs	mg/kg	6,592.49	12,622.80	2,316.48	38,361.66	25,368.94	19,169.51	74,597.33	36,343.28	**2,316.48**	67,599.94	128,678.69	23,164.80	383,616.62	130,652.20	97,560.30	372,986.63	181,716.40	**23,164.80**
	High MW PAHs	mg/kg	39.51	72.42	46.33	767.23	29.05	39.00	699.35	340.72	**29.05**	--	--	--	--	156.91	208.68	3,491.06	1,700.82	**156.91**
Petroleum	Gasoline Range Organics	mg/kg	--	--	--	--	--	--	--	--	**--**	--	--	--	--	--	--	--	--	**--**
	TPH - Diesel	mg/kg	105,086.17	199,535.36	35,638.15	590,179.41	407,143.89	301,204.82	1,137,154.37	554,013.42	**35,638.15**	1,050,861.71	1,995,353.63	356,381.52	5,901,794.15	610,715.84	451,807.23	1,705,731.55	831,020.12	**356,381.52**
	TPH - Kerosene	mg/kg	105,086.17	199,535.36	35,638.15	590,179.41	407,143.89	301,204.82	1,137,154.37	554,013.42	**35,638.15**	1,050,861.71	1,995,353.63	356,381.52	5,901,794.15	610,715.84	451,807.23	1,705,731.55	831,020.12	**356,381.52**
Semivolatile Organics	Normal paraffin hydrocarbons	mg/kg	170,870.11	324,444.50	57,947.64	959,631.73	407,143.89	301,204.82	1,137,154.37	554,013.42	**57,947.64**	--	--	--	--	610,715.84	451,807.23	1,705,731.55	831,020.12	**451,807.23**
	Phenol	mg/kg	--	--	--	--	4,885.73	526.11	503.73	5,918.74	**503.73**	--	--	--	--	14,657.18	1,578.34	1,511.20	17,756.23	**1,511.20**

Table 7-3. Wildlife (Birds and Mammals) SSLs for Nonradionuclides

Group	Soil Constituent	Units	NOAEL-Based Site-Specific SSLs									LOAEL-Based Site-Specific SSLs								
			California Quail	Meadowlark	Killdeer	Red-Tailed Hawk	Great Basin Pocket Mouse	Deer Mouse	Grasshopper Mouse	Badger	NOAEL Lowest	California Quail	Meadowlark	Killdeer	Red-Tailed Hawk	Great Basin Pocket Mouse	Deer Mouse	Grasshopper Mouse	Badger	LOAEL Lowest
	2-methylphenol (ocresol)	mg/kg	--	--	--	--	127,436.04	10,037.62	9,293.33	134,503.46	**9,293.33**	--	--	--	--	--	--	--	--	--
	4-methylphenol (pcresol)	mg/kg	--	--	--	--	127,436.04	10,101.66	9,357.99	136,360.50	**9,357.99**	--	--	--	--	--	--	--	--	--
	2,4-dinitrotoluene	mg/kg	0.29	0.30	0.20	7.17	13.78	13.46	35.58	285.89	**0.20**	38.14	39.15	26.35	932.00	28.79	28.13	74.35	597.39	**26.35**
	Bis[2-ethylhexyl] phthalate[a]	mg/kg	111.06	0.35	0.14	263.03	1,733.20	5.35	4.55	3,599.39	**0.14**	--	--	--	--	17,332.00	53.52	45.42	35,993.87	**45.42**
	Total PCBs[b]	mg/kg	10.01	0.65	0.33	25.09	2.92	0.30	0.27	8.47	**0.27**	100.12	3.58	1.82	250.88	29.22	1.61	1.47	84.71	**1.47**
	Aroclor 1016[b]	mg/kg	6.45	0.64	0.33	21.75	35.21	2.75	2.47	150.41	**0.33**	64.48	3.55	1.82	217.53	88.14	5.30	4.85	376.56	**1.82**
	Aroclor 1221[b]	mg/kg	2.73	0.61	0.33	24.02	0.69	0.25	0.27	8.15	**0.25**	27.30	3.44	1.82	240.18	6.88	1.48	1.47	81.52	**1.47**
	Aroclor 1232[b]	mg/kg	2.19	0.59	0.33	26.24	0.55	0.24	0.27	8.81	**0.24**	21.94	3.40	1.82	262.36	5.48	1.44	1.47	88.10	**1.44**
	Aroclor 1242[b]	mg/kg	10.36	0.65	0.33	25.55	3.09	0.30	0.27	8.74	**0.27**	103.60	3.59	1.82	255.51	30.91	1.63	1.49	87.35	**1.49**
	Aroclor 1248[b]	mg/kg	9.41	0.65	0.33	24.33	0.35	0.06	0.06	1.06	**0.06**	94.05	3.58	1.82	243.35	3.47	0.35	0.32	10.55	**0.32**
	Aroclor 1254	mg/kg	11.52	0.65	0.33	27.26	3.48	0.30	0.27	9.11	**0.27**	115.21	3.59	1.82	272.65	34.76	1.62	1.47	91.11	**1.47**
	Aroclor 1260[b]	mg/kg	20.38	0.66	0.33	51.49	7.67	0.30	0.27	15.42	**0.27**	203.80	3.62	1.82	514.89	76.65	1.64	1.47	154.21	**1.47**
	Aroclor-1262[b]	mg/kg	37.83	71.83	12.83	212.46	27.69	20.48	77.33	37.67	**12.83**	378.31	718.33	128.30	2,124.65	276.86	204.82	773.26	376.73	**128.30**
Pesticide	Dichloroprop	mg/kg	--	--	--	--	--	--	--	--	**--**	--	--	--	--	--	--	--	--	**--**
	Aldrin	mg/kg	0.45	0.08	0.03	1.06	10.22	1.99	1.96	26.80	**0.03**	2.24	0.40	0.16	5.30	51.12	9.94	9.82	133.98	**0.16**
	beta-1,2,3,4,5,6-Hexachlorocyclohexane	mg/kg	4.11	3.65	2.72	112.24	1.87	1.73	3.97	66.95	**1.73**	6.17	5.47	4.08	168.46	9.36	8.67	19.87	334.76	**4.08**
	alpha-Chlordane	mg/kg	121.50	24.28	10.08	301.53	92.53	20.47	20.66	264.12	**10.08**	607.51	121.40	50.41	1,507.65	925.29	204.66	206.56	2,641.24	**50.41**
	gamma-Chlordane	mg/kg	121.50	24.19	10.04	301.53	92.53	20.40	20.57	264.12	**10.04**	607.51	120.97	50.22	1,507.65	925.29	203.98	205.75	2,641.24	**50.22**
	Dichlorodiphenyldichloroethylene	mg/kg	30.37	0.21	0.07	0.06	20.48	0.11	0.09	0.03	**0.03**	300.36	2.30	0.80	1.70	135.88	0.71	0.59	0.40	**0.40**
	Dichlorodiphenyltrichloroethane	mg/kg	30.37	0.30	0.10	2.53	20.48	0.16	0.14	1.41	**0.10**	300.36	3.47	1.19	46.28	135.88	1.05	0.88	12.68	**0.88**
	Dieldrin	mg/kg	1.93	0.06	0.02	1.64	0.28	0.01	0.01	0.35	**0.01**	6.07	0.20	0.08	5.16	0.57	0.02	0.02	0.69	**0.02**
	Endosulfan I	mg/kg	93.44	66.32	41.40	1,671.48	0.92	0.71	1.29	21.88	**0.71**	--	--	--	--	--	--	--	--	**--**
	Endosulfan II	mg/kg	93.44	66.32	41.40	1,671.48	0.92	0.71	1.29	21.88	**0.71**	--	--	--	--	--	--	--	--	**--**
	Endosulfan sulfate	mg/kg	62.89	55.40	41.40	2,159.84	0.61	0.56	1.29	27.15	**0.56**	--	--	--	--	--	--	--	--	**--**
	Endrin aldehyde	mg/kg	2.56	0.52	0.23	52.86	0.51	0.14	0.14	14.04	**0.14**	--	--	--	--	5.13	1.36	1.41	140.40	**1.36**
	Methoxychlor	mg/kg	--	--	--	--	59.78	11.20	10.92	441.01	**10.92**	--	--	--	--	119.56	22.39	21.84	882.02	**21.84**

Shaded cells represent the lowest chemical specific NOAEL-based and LOAEL-based SSLs

a. Values for diethyl phthalate and di-n-butyl phthalate were used as a surrogate for bis(2)ethylhexyl phthalate.
b. Aroclor-1254 value was used as surrogate.

The development of the plant bioaccumulation database is described in "Uptake of Inorganic Chemicals from Soil by Plant Leaves: Regressions of Field Data" (Efroymson et al., 2001) as follows:

> "Field and greenhouse studies in which concentrations of arsenic, cadmium, copper, lead, mercury, nickel, selenium, or zinc in both surface soil and collocated, aboveground plant tissue were analyzed were identified. Information regarding soil and plant concentrations, soil parameters, exposure time, chemical form, dry or wet weight, extraction method, plant species, and plant part was compiled in a spreadsheet. The database included the following number of observations per growth form: 525, graminoid; 544, forb/herb; 4, forb/herb or vine; 69, forb/herb or shrub; 16, shrub; 18, tree or shrub; 49, tree; and 107 unknown or composited samples. Approximately thirty percent of the data represented chemical concentrations in plant leaves, excluding stems, fruits and seeds; and the remaining aboveground samples included clippings, unspecified aboveground parts or shoots. Samples of fruits or seeds alone were excluded from the database. Tests in which salts (e.g., cadmium chloride, copper sulfate, sodium selenate) were added in solution to soil were excluded because of preliminary results that suggested that regressions of concentrations in plants on concentrations in soil were different for field and salt chemical forms.
>
> Only studies in which concentrations were expressed on an air- or oven-dry weight basis were used. Although most studies reported that plant material was washed, studies were not excluded if the extent of washing was not stated in the paper. Studies were used even if the individual investigators observed no correlation between concentrations of contaminants in soils and plants. Concentrations of chemicals in soil or plants were sometimes estimated from a figure, but only if estimates could be made within about 10 %. Data for species that are known to hyperaccumulate metals were excluded. Data for which measured concentrations were below detection thresholds were excluded.
>
> Each plant species or variety, soil type, location, and concentration of the test element in soil represented an independent observation in the dataset. Differences in exposure duration or above-ground plant part did not constitute separate observations; concentrations in soils or plants that differed on the basis of one of these two variables were averaged. The number of observations in these means, which ranged between 1 and 6, was not retained in the subsequent statistical analysis.
>
> Concentrations of contaminants in soil at the time of plant sampling were used if known. If these concentrations were not measured (as was often the case in pot studies), the initial concentration of the element measured in or added to soil was assumed to be equivalent to the final concentration. In field experiments, the change in soil concentration of an element over time was assumed to be minimal.
>
> Observations were included in the database if the total chemical concentration in soil was measured, either by extraction with strong acid or by extraction with moderately strong acid (e.g., 4N sulfuric acid) sometimes accompanied by heat. Studies in which concentrations of contaminants in soil were determined by a partial extraction with DTPA (diethylene triamine pentaacetic acid), weak acids, or water were excluded from analysis.
>
> For studies in which contaminant concentrations at multiple depths were measured, the concentration at the 0-10, 0-15, or 0-20 cm depth interval was recorded. Where only a single soil depth was measured, it ranged from 5 to 70 cm.
>
> Studies included contamination from the following sources: mine wastes (ores, tailings), smelter deposits, other industrial sources, vehicle and other urban emissions, wastewater effluents, composts, fertilizers, dredged materials, sewage sludges, fly ashes, flue dusts, nuclear waste, and arsenical pesticide residues. Where materials such as fertilizers were added to soil, data were excluded if mixing with soil did not occur. In addition, some measurements were taken from background

> locations. For example, chemical data for arsenic included the following sources: mine waste (24 observations), smelter operations (23 observations), fly ash disposal (18 observations), pesticide use (19 observations), nuclear waste (4 observations), unidentified urban sources (3 observations), background or no apparent anthropogenic source (13 observations), and unknown source (18 observations). Field studies in which a current, local atmospheric source of contaminants was present were excluded from the database."

Similarly, the development of the small mammal bioaccumulation database was described in *Development and Validation of Bioaccumulation Models for Small Mammals* (ES/ER/TM-219) as follows:

> "A literature search was performed for studies that reported chemical concentrations in co-located small mammal and soil samples. Data were restricted to only studies that reported whole body or carcass (whole body minus selected organs or other tissues) concentrations. To ensure relevancy of UFs and models to field situations, only field studies in which resident small mammals were collected were considered. All small mammal tissue burdens were therefore assumed to be at equilibrium with soil concentrations. To ensure comparability of data, only 'total' chemical analyses of both soil and small mammals (i.e., resulting from extractions of metals using concentrated acids) were included. Data resulting from DTPA, acetic acid, and other mild extraction methods were excluded. The mean (or composite) chemical concentration in soil and small mammal reported for each sampling location evaluated in each study was considered an observation. If data for multiple small mammal species were reported at a site, each was considered a separate observation. Soil and small mammal data in the database were reported as mg/kg dry weight. If studies reported small mammal concentrations in terms of wet weight, dry weight concentrations were estimated assuming a 68% water content (EPA, 1993). Data concerning soil characteristics [e.g., soil pH, % organic matter, cation exchange capacity, soil texture, etc.] were rarely reported and therefore do not appear in the database. Because chemical uptake was expected to vary according to small mammal diet preferences, each species was assigned to one of the three trophic groups: insectivore (diet consisting primarily of insects and other invertebrates), herbivore (diet consisting primarily of plant material), and omnivore (diet consisting of both animal and plant material). A summary of the small mammal species included in the database and the trophic groups to which they were assigned is presented in Table 1. To validate the models developed from the literature-derived data, soil and small mammal data collected as part of Comprehensive Environmental Response, Compensation, and Liability Act (CERCLA) remedial investigations at sites in Oklahoma (PTI 1995) and Montana (LaTier et al., 1995) were acquired as a validation dataset. Small mammal species in this validation dataset, however, represented only the herbivore and omnivore trophic groups. Validation data for insectivores were unavailable."[6]

Literature Derived Bioaccumulation Data for Terrestrial Arthropods. Estimating exposures to insectivorous or omnivorous wildlife involved estimating bioaccumulation into soil invertebrates. Soil invertebrate bioaccumulation models used for SSLs consisted of the earthworm models from *Development and Validation of Bioaccumulation Models for Earthworms* (ES/ER/TM-220) and "Literature-Derived Bioaccumulation Models for Earthworms: Development and Validation" (Sample et al., 1999). Hanford Site-specific observations (as detailed in the RCBRA [DOE/RL-2007-21] and *Central Plateau Ecological Risk Assessment Data Package Report* [DOE/RL-2007-50]) indicate that earthworms are nonexistent in upland soil and have little or no contribution to the invertebrate portion of bird and mammal diets at the Hanford Site. Rather, insects and other arthropods (for example, beetles, ants, and spiders) are the primary prey of invertebrate-feeding birds and mammals at the site. Consequently, the data collected to address site-specific bioaccumulation into invertebrate prey of birds and mammals focused on arthropods (RCBRA

[6] References in this passage can be found in the original source (*Development and Validation of Bioaccumulation Models for Small Mammals* [ES/ER/TM-219]); complete citation is provided in Chapter 11.

[DOE/RL-2007-21]). Additional bioaccumulation data for terrestrial arthropods were identified and extracted from published literature to supplement the Hanford Site-specific data. This database was largely developed to support bioaccumulation modeling for the U.S. Army Adaptive Risk Assessment Modeling Systems (ARAMS[7]) and was first presented in *Development of Terrestrial Exposure and Bioaccumulation Information for the Army Risk Assessment Modeling System (ARAMS)* (USACHPPM, 2004). A literature search was performed for studies that reported chemical concentrations in collocated biota and media samples[8]. Literature databases searched included those hosted by the Defense Technical Information Center (*Online Information for the Defense Community*, Public Technical Reports [DTIC, 2012]), EPA (ECOTOX database) and the U.S. National Library of Medicine (TOXLINE: Toxicology Data Network).

From the range of studies reviewed, 22 were identified as containing relevant data (i.e., reported collocated soil and biota concentrations). Terrestrial invertebrate data focused on studies of accumulation in insects or spiders and reported whole body concentrations. To ensure relevancy of the soil to biota factors and models to field situations, only field studies that collected resident terrestrial invertebrates were considered. Therefore, all terrestrial invertebrate residues were assumed to be at equilibrium with soil concentrations.

To ensure comparability of data, only "total" chemical analyses of both soil and biota (e.g., resulting from extractions of metals using concentrated acids) were included. Data resulting from acetic acid, diethylenetriaminepentaacetic acid, and other mild extraction methods were excluded. The mean (or composite) chemical concentration in media and biota reported for each sampling location evaluated in each study was considered an observation. If data for multiple species were reported at a site, each species was considered a separate observation. Soil and biota data in the terrestrial arthropod database were reported as mg/kg DW. If a study identified in the literature search reported biota concentrations in wet weight, then DW concentrations were either calculated using the water content presented in the study or estimated assuming water content percentages as presented in *Wildlife Exposure Factors Handbook* (EPA/600/R-93/187) when water content was not presented in the study.

Data concerning species, soil pH, percent organic matter (OM), cation exchange capacity (CEC), soil texture, and soil Ca concentration (mg/kg dry wt) were included in the database whenever reported. Additionally, class, order, and family taxonomic data were included for each species in the database. These data was used to develop uptake factors by taxon for terrestrial invertebrates. Because chemical uptake was expected to vary according to terrestrial invertebrate diet preferences, each species was assigned to one of three trophic groups: predator (diet consisting primarily of other insects), herbivore (diet consisting primarily of plant material), and detritivore (diet consisting primarily of organic matter in the leaf litter).

To ensure the accuracy of the terrestrial arthropod database, all data were verified by at least one reviewer. The reviewer would first exam the study for data presented and analytical methods used. The reviewer would then check all calculations and conversions necessary to obtain required units (e.g., mg/kg dry weight). Finally, a minimum of 25 percent of all data was checked. If an error was found during this check, then 100 percent of the data was verified. Unit conversion and transposition errors were the most common types of errors found; however these were infrequent. All errors were corrected.

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[7] ARAMS was previously known as the Army Risk Assessment Modeling System.

[8] Data usability requirements included: only paired/collocated samples with detects in both tissue and soil at levels above detection limits; terrestrial invertebrate data focuses on whole body tissue samples; only field studies, not laboratory studies, were included except where noted; only total chemical analyses of both soil and biota – data resulting from mild acid extraction methods were excluded; the mean or composite chemical concentration in media and biota reported per location in each study was considered an observation; data on distinct species were considered separate observations; all wet weight measurements were converted to dry weight using study specific water content or estimations from *Wildlife Exposure Factors Handbook* (EPA/600/R-93/187). Additional detail on data usability is found in *Tier 2 Risk-Based Soil Concentrations Protective of Ecological Receptors at the Hanford Site* (CHPRC-01311) within Appendix H of the 100-K RI/FS Report (DOE/RL-2010-97).

Development of Integrated Bioaccumulation Models. The Hanford Site-specific plant, soil invertebrate, and small mammal data were integrated with the literature-derived bioaccumulation data. Bioaccumulation analyses were performed once biota data were converted to standard units (mg/kg-dry weight). Analyses were restricted to observations where the chemical of interest was detected in both soil and the matched tissue sample; all observations in which either soil or tissue concentrations were nondetects were excluded from the analyses. Analyses consisted of development of BAFs and nonlinear regression analyses. BAFs are simply the ratio between concentrations measured in tissue and that in soil. BAFs for all paired soil-tissue observations and summary statistics (arithmetic mean, standard deviation, minimum, maximum, median, and 90th percentile) were calculated.

To evaluate if a log-linear relationship exists between the chemical concentration in soil and that in terrestrial biota, simple log-linear regressions were performed using SAS PROC REG (*SAS/STAT User's Guide* [SAS Institute, 1999]). Chemical concentrations in both soil and biota tissues were transformed to natural-log (ln) before regression analyses. Regression analyses were considered significant and suitable for estimation purposes if all three of the following criteria were met: $p \geq 0.05$, $r^2 > 0.2$, and a positive slope. If regression analyses did not meet one of these criteria, the median BAFs were used to estimate tissue concentrations in exposure models.

The wildlife SSLs for nonradionuclides are presented in Table 7-3, and the wildlife PRGs (metals only) are presented in Table 7-4. For the purposes of this ERA, the LOAEL-based SSLs (SSLs that used lowest effect levels from the effects assessment) were used to evaluate residual risks at the remediated 100-D/H waste sites. To focus the assessment on COPEC-receptor-waste site combinations that might require further evaluation, the SSLs were compared to EPCs developed for 100-D/H as described in Section 7.4.1. To identify which COPEC-receptor-waste sites combinations should be brought forward to the SMDP to identify community- or population-level effects to be addressed in the FS, EPCs were compared to PRGs for COPCs that exceeded SSLs and background, as described in Section 7.4.3.

7.3.2.3 Radionuclide Exposures

Exposure to radionuclides differs from chemical exposure. Terrestrial biota receives exposure to radionuclides through a combination of both internal and external pathways. Internal exposure is a function of radiation emitted from radionuclides retained in tissues. At a terrestrial site such as the 100-D/H OUs, external exposure is due to radiation from radionuclides in soil with which biota come into contact (or come near). For the purposes of developing SSLs, radionuclide exposure was estimated based on the internal and external radiation exposure models used to develop BCGs as described in Graded Approach for Radiation Doses to Biota (DOE-STD-1153-2002).

The BCGs for terrestrial plants and animals represent SSLs for radionuclides in soil for assessing ecological risks at the 100-D/H Source OUs waste sites (Table 7-1). The BCGs for radionuclides use conservative assumptions for internal and external exposure. While existing effects data support the application of these dose limits to representative individuals within populations of plants and animals, the assumptions and parameters applied in the derivation of the BCGs are based on a maximally exposed individual, representing a conservative approach for screening purposes. The following assumptions are used for estimating doses from external exposure for developing BCGs:

- The source medium is infinite in extent and contains uniform concentrations of radionuclides (that is, there are no hot spots).
- One hundred percent of the radionuclide energies are absorbed (despite the small size of some of the receptors).
- Organisms exposed to soil are uniformly surrounded by the source medium.

Table 7-4. Preliminary Remediation Goals for Wildlife (Birds and Mammals)

Analyte Group	Analyte	Units	California Quail	Western Meadowlark	Killdeer	Red-Tailed Hawk	Lowest Avian PRG	Great Basin Pocket Mouse	Deer Mouse	Grasshopper Mouse	Badger	Lowest Mammal PRG	Lowest Wildlife PRG
Metal	Silver	mg/kg	4,238	3,973	**983**	20,186	983	24,465	9,806	14,362	30,778	9,806	983
Metal	Aluminum	mg/kg	19,217	31,220	7,214	74,599	7,214	4,883	**3,988**	13,059	7,811	3,988	3,988
Metal	Arsenic	mg/kg	4,776	7,403	2,284	40,102	2,284	201	**127**	302	847	127	127
Metal	Boron	mg/kg	**54**	68	91	2,714	54	**32**	39	170	2,516	32	32
Metal	Barium	mg/kg	1,721	2,335	**1,687**	8,101	1,687	2,265	2,617	11,873	12,430	2,265	1,687
Metal	Beryllium	mg/kg	NTD	NTD	NTD	NTD	NTD	**14**	20	181	289	14	14
Metal	Cadmium	mg/kg	294	103	**29**	1,711	29	2,203	624	858	4,704	624	29
Metal	Cobalt	mg/kg	1,397	2,050	**484**	4,798	484	2,901	2,136	5,610	4,234	2,136	484
Metal	Chromium	mg/kg	193	221	**109**	610	109	544	**517**	1,424	1,765	517	109
Metal	Copper	mg/kg	423	461	**213**	12,881	213	233	193	1,217	4,631	193	193
Metal	Mercury	mg/kg	36	4.7	2	92	2	7.9	**1.6**	1.8	33	1.6	1.6
Metal	Lithium	mg/kg	--	--	--	--	--	**1,664**	1,797	8,347	6,522	1,664	1,664
Metal	Manganese	mg/kg	20,746	26,026	14,407	150,899	14,407	**3,322**	3,467	11,780	21,916	3,322	3,322
Metal	Molybdenum	mg/kg	125	117	95	515	95	5.9	**5.7**	14	38	5.7	5.7
Metal	Nickel	mg/kg	2,051	1,127	361	11,625	361	711	**247**	342	1,520	247	247
Metal	Lead	mg/kg	559	664	**156**	2,300	156	2,672	1,578	3,807	3,966	1,578	156
Metal	Antimony	mg/kg	NTD	NTD	NTD	NTD	NTD	97	**92**	366	325	92	92
Metal	Selenium	mg/kg	10	4.9	2.4	24	2	2.7	**1.4**	1.9	8.8	1.4	1.4
Metal	Strontium (Elemental)	mg/kg	--	--	--	--	--	**1,214**	1,449	6,540	8,256	1,214	1,214
Metal	Thallium	mg/kg	--	--	--	--	--	8.7	**6.2**	12	25	6.2	6.2
Metal	Uranium (Calculated Total)	mg/kg	2,002	339	139	82	82	812	123	119	**40**	40	40
Metal	Vanadium	mg/kg	81	107	**43**	505	43	260	297	4,531	3,596	260	43
Metal	Zinc	mg/kg	6,289	4,662	**856**	906	856	6,711	3,331	12,666	1,037	1,037	856

Notes: Bold values represent lowest PRG for that analyte.

Shaded values are based on NOAELs because of the lack of LOAELs.

NBD = no (or incomplete) bioaccumulation data (for estimation of dietary exposure)

NTD = no toxicity data (for selected analyte)

PRG = preliminary remediation goal

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The following assumptions are used in estimating doses from internal exposure for developing BCGs:

- All radionuclide decay energies are retained in tissue (100 percent of energies absorbed).
- Exposure for a given radionuclide includes all decay chain progeny.

All radionuclides are uniformly distributed such that all target tissues may be affected.

7.3.3 Drinking Water Exposure

The estimates of exposure from drinking water ingestion by wildlife include the use of a simplified model whereby the rate of ingestion is standardized to the body weight of the receptor on a per-kilogram basis. The simplified allometric scaling equations presented in "Scaling of Osmotic Regulation in Mammals and Birds" (Calder and Braun, 1983) were used to estimate water ingestion as the number of liters consumed per kilogram body weight per day. These rates of ingestion were then multiplied by the concentration of COPECs to calculate the total dose from the drinking water pathway:

$$Dose = \lfloor [Water \times DWIR] \rfloor x\, AUF$$

where:

Dose = drinking water exposure (mg/kg body weight/day)

Water = chemical concentration in seep water (mg/L)

DWIR = drinking water ingestion rate (L/kg body weight/day)

AUF = area use factor (area of site/home range of receptor) (unitless)

Drinking water ingestion was estimated for several species of birds and mammals expected in the 100-D/H riparian area along the Columbia River, with the initial assumption that they reside at the site and fulfill their drinking water requirements exclusively from the seeps, but only for 9 months of the year because the river stage is elevated from mid-April to mid-July, making the seeps inaccessible. Therefore, an AUF of 0.75 was employed for all species except bats. For bats, an AUF of 0.5 was used since bats use a combination of hibernating and seeking alternative sources of emergent insects during the winter months (*Living with Wildlife: Bats* [WDFW, 2004]).

Estimates are not included for small mammals as they maintain water balance through excreting concentrated urine, obtaining water from food, and generating water during metabolism ("Perognathus parvus" [Verts and Kirkland, 1988]). Estimating drinking water exposure can be complicated because the presence of seeps and observed concentrations depend on river stage and, for several species of birds, migration patterns are a factor. Assuming that wildlife meet their daily drinking water requirements from the seeps instead of a more available source, such as the river, is a conservative approach meant to evaluate a worst-case scenario. Therefore, though it represents an overestimate, the 95 percent UCL of the arithmetic mean concentration of the analyzed constituent was used as the EPC for simplicity. While filtered water data are used in evaluations of the effects on aquatic receptors because those concentrations are bioavailable, unfiltered concentrations are more appropriate for drinking water, as bioavailability may change within the digestive tract. Both were included to be comprehensive, as in rare cases, filtered measurements can be higher than unfiltered. Results were not pooled so as to not bias any one sampling event at which both measurements occurred.

7.3.4 PRGs

The PRGs presented in this chapter represent Hanford Site-specific values as presented in *Tier 2 Risk-Based Soil Concentrations Protective of Ecological Receptors at the Hanford Site*

(CHPRC-01311). Much of the modeling used to develop PRGs for wildlife is presented in this chapter as the PRGs build on the SSLs (*Tier 1 Risk-Based Soil Concentrations Protective of Ecological Receptors at the Hanford Site* [CHPRC-00784]), using the same receptors, exposure models, life history parameters, and TRVs. The only deviations from the SSL development were the use of bioaccumulation models that included exclusively arthropods as the invertebrate portion of receptors' diets[9] and integration of Hanford Site-specific data. The SSLs included prey tissue estimation models that were generic and included a wide variety of species, only some of which are likely to occur within the arid environment at Hanford. Most invertebrate data included in the food web models for SSL development for invertivores and omnivores relied on bioaccumulation data from earthworms and other soil invertebrates. Soil invertebrates such as earthworms are rarely encountered in the arid upland soil at the Hanford Site. Thus, modeling for PRG development (*Tier 2 Risk-Based Soil Concentrations Protective of Ecological Receptors at the Hanford Site* [CHPRC-01311]) incorporated additional Hanford Site-specific tissue data and data from other closely related ecosystems and more recent data specific to insects found at Hanford that had not been available when either the MTCA guidance ("Site-specific Terrestrial Ecological Evaluation Procedures" [WAC 173-340-7493]) or EPA EcoSSLs were developed.

The development of PRGs corresponds to an exposure and effects assessment, conducted as part of a baseline ecological risk assessment within ERAGS (EPA 540-R-97-006) and reflects *Ecological Risk Assessment and Management Principles for Superfund Sites* (OSWER Directive 9385.7-28 P), which encourages the use of site-specific ecological risk data to support cleanup decisions, whenever practicable. The process for developing PRGs is also consistent with Ecology's "Site-specific Terrestrial Ecological Evaluation Procedures" (WAC 173-340-7493). None of the differences were recalculations of the original datasets and models used to derive the WAC values. Rather, all of the changes from the WAC 173-340 Table 749-3 are based on updated exposure models (*Guidance for Developing Ecological Soil Screening Levels* [OSWER Directive 9285.7-55]) and toxicological literature reviews not available at the time WAC 173-340 Table 749-3 was developed. These PRGs are intended to be applied to all upland environments across the Hanford Site. Though additional receptors may also be present in riparian areas, the wildlife PRGs and the supporting bioaccumulation and exposure models and TRVs are applicable for riparian areas and can be used in conjunction with values for those additional receptors.

Hanford Site-specific wildlife PRGs are presented in Table 7-4. PRGs were researched for inorganic and organic constituents, but not radionuclides. Ultimately, PRGs were only recommended for inorganics, as data were limited for organics.[10] Confidence in the PRGs as a whole is greater than for the SSLs as they were developed specifically for use at the Hanford Site using site-specific data. Relative to each other, confidence in some PRGs is greater than in others. The additional confidence is due to a combination of the total number of Hanford Site-specific paired soil and tissue samples and the strength of the relationship between tissue and soil concentration (correlation). Details regarding the confidence in specific PRGs are included in the SMDP in Section 7.6 as needed.

PRGs for inorganic chemicals protective of plants and invertebrates are presented in Table 7-5. When Hanford Site-specific toxicological data on the effects of plants and soil invertebrates were available, these data were considered for PRG selection. These data are summarized in the following three documents:

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[9] Further detail on the estimation of invertebrate tissue concentrations is found in Section 7.3.2.2.

[10] Here in Chapter 7, if a second tier effect threshold (e.g., PRG) was not available or recommended, chemical-waste site combinations were retained for further evaluation in the SMDP (section 7.6) if the exposure point concentration exceeded the SSL).

- *Tier 2 Terrestrial Plant and Invertebrate Preliminary Remediation Goals (PRGs) for Nonradionuclides for Use at the Hanford Site* (ECF-HANFORD-11-0158), included in Appendix H of the 100-K RI/FS (DOE/RL-2010-97)
- RCBRA (DOE/RL-2007-21)
- *Ecological Soil Screening Levels for Arsenic and Lead in the Tacoma Smelter Plume Footprint and Hanford Site Old Orchards Ecology* (Ecology Publication 11-03-006).

Table 7-5. Final Recommended Soil PRGs for Plants and Invertebrates

Chemical	Plant NOEC (mg/kg)	Invertebrate NOEC (mg/kg)
Antimony	842	842
Arsenic	128	128
Barium	500	358
Beryllium	10	40
Boron	29.6	28.6
Cadmium	9.84	20
Chromium	259	149
Cobalt	15.7	15.7
Copper	70	58
Lead	9,090	1,700
Manganese	1,260	1,260
Mercury	0.3	12.5
Molybdenum	2	28
Nickel	38	280
Selenium	2.02	4.1
Silver	560	2.99
Thallium	1	0.459
Tin	838	838
Uranium	250	100
Vanadium	89.4	116
Zinc	621	8,980

All of the site specific toxicological thresholds presented in these documents are no observed-effect concentrations (NOECs). Thus, for each chemical studied in one or more of these documents, the greatest NOEC among these documents was selected as the PRG for that chemical. When Hanford Site-specific thresholds for plants and invertebrates were not presented in these three documents, the EcoSSL was selected as the PRG because it included more recent information than what was available when the MTCA (WAC 173-340) Table 749-3 was developed. When an EcoSSL was not available, the value from WAC was selected. The two exceptions were as follows:

- The Hanford Site-specific background value for cobalt was selected as the PRG for both plants and invertebrate. There is no WAC or EcoSSL value for invertebrates. The background value of 15.7 mg/kg is greater than the EcoSSL of 13 mg/kg. While the WAC plant value of 20 mg/kg is greater than the background value, it is based upon the value from ORNL and the original authors gave the value low confidence. Site-specific plant and invertebrate NOEC values of 11.2 mg/kg and 12.2 mg/kg were also available from the RCBRA (DOE/RL-2007-21), but this value was the highest concentration tested and was lower than background.

- The cadmium value for invertebrates of 20 mg/kg from WAC was selected as the PRG over the EcoSSL of 140 mg/kg. The WAC value was based upon an ORNL recommendation where the authors gave a moderate to high confidence in the recommendation, and this was considered of equal weight with the EcoSSLs so the lower of values of equal confidence was selected.

The final recommended PRG represented the most appropriate value, leaning toward the most recent data available that met the criteria set forth in ERAGS (EPA 540-R-97-006) and MTCA (WAC 173-340-7493) guidelines for selecting site-specific criteria. In selection of values that differ from MTCA (WAC 173-340) Table 749-3, when multiple recent toxicological data sources were available, the value of the highest confidence or the lower of two values with equally high confidence was chosen. The site-specific values are preferred over those from published literature in that they are more recent data not available at the time MTCA guidance or EcoSSLs were developed and they reflect the potential for toxicity under conditions found specifically at the site. However, with some COPECs, more recent site-specific sampling efforts were unable to obtain concentration ranges above those from published literature. With all of the site-specific studies conducted for the RCBRA (DOE/RL-2007-21), by Ecology and recently by CH2M HILL Plateau Remediation Company in the Central Plateau, no clear significant toxicity to plants and invertebrates attributable to site soil contaminants was observed; thus, recommended toxicological values are unbound NOECs. Hence, in some cases, published literature values above these unbound NOECs were selected as PRGs over site-specific values. Final selection of the PRGs for plants and invertebrates is discussed in detail in *Tier 2 Terrestrial Plant and Invertebrate Preliminary Remediation Goals (PRGs) for Nonradionuclides for Use at the Hanford Site* (ECF-HANFORD-11-0158). As with the wildlife PRGs, details regarding the confidence in specific PRGs are included in the SMDP in Section 7.6 as needed.

7.3.5 Estimation of Exposure Point Concentrations in Waste Sites

As mentioned earlier, assuming that wildlife forage exclusively within the boundaries of a waste site or that the data collected from within a waste site represent the central tendency of exposure to wildlife is a conservative assumption. In reality, the concentration of contaminants to which a wildlife population is exposed includes points both in and out of the waste site being investigated unless physical barriers prevent exposure. Thus, a true exposure estimate would include data points both in and out of the site boundary, and in some investigations for other sites, the points outside have been generated by either measured data or have been assumed to be at background. However, methods for this type of estimate of exposure are not defined in guidance and are not commonplace. What is common in ERA practice, and

what was done for this ERA, is to initially characterize risks assuming an AUF of 1 (all exposure is within the site) and then refine that assumption should the highly conservative exposure estimate and risk characterization suggest an unacceptable risk warranting further evaluation. Hence, this section describes the method that EPCs were derived within the waste sites that assumed an AUF of 1. The SMDP in Section 7.6 describes how AUFs should be used for evaluating waste sites.

In total, 95 waste sites in the 100-D Source OUs and 47 waste sites in the 100-H Source OUs were verification sampled and included in this ERA. Chapter 6 details the computation of the EPCs for the waste sites at the 100-D/H Source OUs. Briefly, the 95 percent UCL of the arithmetic mean was calculated as the EPC for each decision unit (shallow, overburden, staging pile area, and footprint staging pile and focused) within each waste site. Two separate statistical evaluations were performed, one used for the closeout documentation and one used for human health and ecological risk assessments, as follows:

- **Statistical Evaluation Used for Closeout Documentation:** For the closeout documentation, the primary statistical calculation to support cleanup verification was the 95 percent UCL on the arithmetic mean of the data. As discussed in *Statistical Guidance for Ecology Site Managers* (Ecology Publication 92-54), a 95 percent UCL on the mean based on the Student's t-test statistic was used for normally distributed data, and the Land method using the H-statistic was used for lognormal data. This guidance also uses proxy values of one-half the detection limits for nondetect values. For small datasets (n less than 10), typically the maximum detected concentration was used as the EPC.

- **Statistical Evaluation Used for Soil Risk Assessment:** Both *Calculating Upper Confidence Limits for Exposure Point Concentrations at Hazardous Waste Sites* (OSWER 9285.6-10) (the most recent EPA guidance for UCL calculation) and ProUCL 4.00.05 were used to recalculate EPCs for the human health and ecological risk assessments of the 100-D/H OU waste site decision units. Although *Statistical Guidance for Ecology Site Managers* (Ecology Publication 92-54) has been used to calculate EPCs for all closeout documentation to date, EPCs were recalculated according to *Calculating Upper Confidence Limits for Exposure Point Concentrations at Hazardous Waste Sites* (OSWER Directive 9285.6-10) to allow for the use of more rigorous statistical methods to estimate exposure concentration and to eliminate the use of the one-half the detection limit used in *Statistical Guidance for Ecology Site Managers* (Ecology Publication 92-54), which has the potential to underestimate exposure concentrations.

The process used to calculate EPCs for each waste site and decision unit is documented in *Computation of Exposure Point Concentrations for the 100-DR-1, 100-DR-2, 100-HR-1 and 100-HR-2 Source Operable Units* (ECF-100DR1-11-0004) (Appendix G). The purpose of *Computation of Exposure Point Concentrations for the 100-DR-1, 100-DR-2, 100-HR-1, and 100-HR-2 Source Operable Units* (ECF-100DR1-11-0004) is to document the data processing and reduction steps, methodology, decision logic, assumptions, input files, and output files used to determine the EPCs. EPCs generated for use in this evaluation for each waste site, decision unit, and detected analyte at the 100-D/H OUs are provided in Tables H-7 and H-8 (Appendix H).

For the drinking water evaluation included in Section 7.4.4, limited data are available and estimating exposure can be complicated as the presence of seeps and observed concentrations depend on river stage. For several species of birds, migration patterns are also a factor. EPCs of seep concentrations was used for simplicity, calculated using Pro UCL software in the same way as were soil EPCs.

7.4 Risk Characterization

The outcome of this step is a list of COPECs for each medium-pathway-receptor combination evaluated. Risks at the 100-D/H Source OUs waste sites were estimated using the HQ method as follows:

$$HQ = EPC/SSL \text{ or } PRG$$

where:

HQ = ecological hazard quotient (unitless)

EPC = soil concentration (µg/kg for nonradionuclides and pCi/g for radionuclides)

SSL = plant/invertebrate or wildlife soil screening level (µg/kg for nonradionuclides and pCi/g for radionuclides)

PRG = plant/invertebrate or wildlife preliminary remediation goal (µg/kg for nonradionuclides)

The HQ values less than 1.0 indicate that adverse effects associated with exposure to a given analyte are unlikely (ERAGS [EPA 540-R-97-006]). These analytes were not considered to present a significant risk and were excluded from further evaluation. An HQ greater than or equal to 1.0 indicates data are insufficient to exclude the potential for risk, but does not indicate that risks are actually present; therefore, these COPCs were carried forward for further evaluation.

In the screening evaluation, the soil EPC for each waste site and decision unit (as applicable) was compared to the plant/invertebrate SSL and the wildlife SSL for all COPCs including metals, pesticides, PCBs, and PAHs (as aroclors). The HQs for these comparisons are provided in Appendix H, Tables H-7 and H-8. COPCs with HQs equal to or greater than 1.0 were carried forward for further evaluation. Only metals failed the screen.[11] COPCs for which appropriate toxicity data were unavailable were not evaluated further, but were retained as uncertainties.

Because the plant/invertebrate and/or wildlife SSL values for 10 COPCs (arsenic, boron, lithium, mercury, manganese, molybdenum, selenium, strontium, thallium, and uranium) were higher than the corresponding PRG values, comparison of the EPCs for these chemicals with both SSLs and PRGs were reviewed to confirm they were below both the SSL and the PRG. For these 10 chemicals, if an EPC was greater than either the SSL or the PRG, the chemical was carried forward to the background evaluation for that specific waste site decision unit.

7.4.1 Risk Characterization for Radionuclides and Aroclors

Because the dose from radionuclides is additive, the total contributions of radionuclides were calculated using the SOF method. With the SOF method, contributions were considered significant if the EPC was greater than the SSL. The SOF equation is as follows:***

$$SOF = \sum_{j=1}^{n} Exposure_j / SSL_j$$

where:

SOF = sum-of-fractions

[11] Metals failing the SSL screen for at least one receptor are identified by waste site in the results section in Tables 7-6 and 7-7 and include: arsenic, barium, boron, chromium, copper, lead, lithium, manganese, mercury, molybdenum, selenium, silver, uranium/total uranium isotopes, vanadium, and zinc.

$Exposure_j$ = exposure concentration for radionuclides

SSL_j = soil screening level for radionuclides

For the purposes of this evaluation, the HQs for each radionuclide were summed within each decision unit to equal an SOF. If the SOF was greater than 1, individual detected radionuclide isotope COPCs were carried forward to the background evaluation.

For those COPCs that exceeded one or more SSLs, the EPC was compared to the background value and summarized in the subsequent tables (Appendix H, Tables H-9 and H-10) in Section 7.4.2.

Similarly, for Aroclors, HIs were calculated to evaluate additive effects. If the HI for Aroclors was greater than 1, the detected Aroclors were identified for further evaluation. This approach is conservative because the measurement of Aroclors as mixtures of PCB congeners does produce some overlap of congeners in multiple Aroclor mixtures. However, a total Aroclor HI is not an uncommon practice. While potential duplication could occur depending on which mixtures are detected, at most sites only one or two Aroclor mixtures are detected and tend to dominate. Also, by carrying the HI >1 forward, when a conclusion of no risk or no unacceptable risk is reached there is less uncertainty with the conclusion because of the additional conservatism in the approach.

7.4.2 Characterization Relative to Background

Background concentrations for inorganic analytes in soil at the Hanford Site are described in the Non-Rad Soil Background document (DOE/RL-92-24). That document provides the 90th percentile background concentrations for several inorganic analytes. For selected inorganic analytes not included in the Non-Rad Soil Background document (DOE/RL-92-24), the 90th percentile concentration has been obtained from PNNL as summarized in *Soil Background Data for Interim Use at the Hanford Site* (ECF-HANFORD-11-0038) and from the 100 Area RDR/RAWP (DOE/RL-96-17) for uranium. Background concentrations for radiological analytes in soil at the Hanford Site are described in the Rad Soil Background document (DOE/RL-96-12), which provides the 90th percentile concentration of background concentrations for several radiological analytes. Background concentrations were not identified for organic chemicals; therefore, all organic chemicals, with HQs greater than or equal to 1.0 were carried forward. COPC EPCs that were less than the 90th percentile background concentration were excluded from further evaluation. COPCs with EPCs not within the range of site background were carried forward for comparison to the PRGs.

7.4.3 Further (Refined) Characterization Relative to PRGs

In the PRG evaluation, the soil EPC for each waste site and decision unit (as applicable) was compared to the plant/invertebrate PRG and the wildlife PRG for all remaining COPCs. COPCs with HQs equal to or greater than 1.0 were retained as COPECs. COPECs were given further consideration under the SMDP. The methodology used in this step of the risk characterization is provided in Appendix H (*Ecological Risk Evaluation for the 100-DR-1, 100-DR-2, 100-HR-1, and 100-HR-2 Source Operable Units* [ECF-100DR1-11-0006]). For any chemical-waste site EPC that exceeded both the SSL and background, if no PRG is presented in Table 7-4 or 7-5, then the chemical-waste site combination was automatically retained for additional evaluation in the SMDP presented in Section 7.6.

7.4.4 Characterization of Risk through Ingestion of Drinking Water

Freshwater seep drinking ingestion HQs for inorganic chemicals were estimated as the ratio of estimated ingestion doses to TRVs. The TRVs used were the same as those used to develop the wildlife PRGs to evaluate soil as presented in *Tier 1 Risk-Based Soil Concentrations Protective of Ecological Receptors at*

the Hanford Site (CHPRC-00784) and *Tier 2 Risk-Based Soil Concentrations Protective of Ecological Receptors at the Hanford Site* (CHPRC-01311). The equation is as follows:

$$HQ = Dose/TRV$$

where:

HQ = ecological hazard quotient (unitless)

Dose = drinking water exposure (mg/kg body weight/day)

TRV = toxicological reference value (mg/kg body weight/day)

For radionuclides, the HQs for evaluating freshwater seep drinking water ingestion were simply a ratio of the measured concentrations in water to the BCGs for wildlife. The lowest water BCG of terrestrial or riparian animal receptors was taken from Graded Approach for Radiation Doses to Biota (DOE-STD-1153-2002) or was calculated using *RESRAD-BIOTA: A Tool for Implementing a Graded Approach to Biota Dose Evaluation, User's Guide, Version 1* (DOE/EH-0676) when not available. SOFs were calculated as described above. Also, as with the soil evaluation, the EPC represents the 95 percent UCL of the arithmetic mean concentration of the analyzed constituent. The equation is as follows:

$$HQ = (EPC*AUF)/BCG$$

where:

HQ = ecological hazard quotient (unitless)

EPC = radionuclide concentration in seep (pCi/L)

AUF = area use factor

BCG = biota concentration guide (pCi/L)

7.4.5 Screening Evaluation Results

The comparisons to plant/invertebrate and wildlife SSLs are provided in Appendix H (Table H-7 and Table H-8 for the 100-D and 100-H OUs, respectively). The results of the screening evaluation (that is, comparison of EPCs with SSLs) in soil is described below, and exceedances for 100-D and 100-H OUs are listed in Table 7-6 and Table 7-7, respectively.

Table 7-6. Summary of Chemicals in 100-D OU Surface Soils (0 to 15 ft [0 to 4.6 m]) Exceeding SSLs and Background

Waste Site/Decision Unit	Exceedances Based on Comparisons to SSLs and Background*	
	Plant/Invertebrate SSL-Based HQ	Wildlife SSL-Based HQ
100-D-13_Shallow_Focused	Boron (9.8) Selenium (1.7) Zinc (2.4)	Zinc (1.8)
100-D-15_Shallow_2	Zinc (1.9)	Zinc (1.4)
100-D-28:1_Shallow	Mercury (5.7) Selenium (3.3)	--

Table 7-6. Summary of Chemicals in 100-D OU Surface Soils (0 to 15 ft [0 to 4.6 m]) Exceeding SSLs and Background

Waste Site/Decision Unit	Exceedances Based on Comparisons to SSLs and Background*	
	Plant/Invertebrate SSL-Based HQ	Wildlife SSL-Based HQ
100-D-29_Shallow	Boron (10.7)	--
100-D-31:3, 100-D-31:4_Overburden	Selenium (2.3)	--
100-D-31:5_Overburden	Boron (40.2) Mercury (1.4) Selenium (3.3)	--
100-D-31:5_Shallow	Boron (21.0) Mercury (2.4) Selenium (2.9) Zinc (1.8)	Zinc (1.3)
100-D-31:6_Overburden	Boron (12.0) Mercury (8.7)	--
100-D-31:6_Shallow	Boron (11.7) Mercury (5.8)	--
100-D-31:8_Shallow_Focused_1	Vanadium (47.5)	Vanadium (3.0)
100-D-31:8_Shallow_Focused_2	Barium (4.8) Boron (338.0) Mercury (1.2) Molybdenum (1.2) Zinc (1.7)	Barium (1.2) Boron (1.3) Zinc (1.2)
100-D-42, 100-D-43, 100-D-45_Shallow	Copper (1.8)	--
100-D-47_Shallow_Focused	Vanadium (42.9)	Vanadium (2.8)
100-D-56:1_Overburden	Boron (11.3) Selenium (4.0)	Selenium (1.1)
100-D-56:1_Shallow	Selenium (3.3)	--
100-D-56:2_Shallow_Focused	Chromium (73.3)	--
100-D-61_Shallow	Boron (22.3) Selenium (2.0)	--
100-D-7_Shallow_1	Selenium (1.6)	--
100-D-7_Staging pile area footprint	Mercury (3.0)	--
100-D-70_Shallow_Focused	Copper (1.5) Zinc (1.6)	Zinc (1.2)
100-D-82_Shallow_Focused	Lead (2.7)	Lead (3.8)
100-D-83:4_Shallow_Focused	Mercury (9.5)	--
100-D-84:1_Shallow_Focused	Mercury (1.2) Vanadium (47.2)	Vanadium (3.0)
100-D-87_Shallow_Focused	Zinc (1.7)	Zinc (1.2)
100-D-88_Shallow_Focused	Vanadium (52.0)	Vanadium (3.3)
100-D-94_Shallow_Focused	Mercury (5.8)	--

Table 7-6. Summary of Chemicals in 100-D OU Surface Soils (0 to 15 ft [0 to 4.6 m]) Exceeding SSLs and Background

Waste Site/Decision Unit	Exceedances Based on Comparisons to SSLs and Background*	
	Plant/Invertebrate SSL-Based HQ	Wildlife SSL-Based HQ
116-D-8_Shallow	--	Lead (1.2)
116-D-8_Shallow_Focused_2	Selenium (2.3)	--
116-DR-5_Overburden	Zinc (1.5)	Zinc (1.1)
116-DR-5_Shallow	Chromium (63.6)	--
116-DR-8_Overburden_3	Lithium (13.9) Silver (1.1) Zinc (1.5)	Zinc (1.1)
116-DR-8_Shallow	Lithium (35.2)	--
118-D-1_Shallow_Focused	Total_U_Isotopes (1.0)	--
118-D-4_Shallow_Focused	Cadmium (1.2) Chromium (120.3) Vanadium (46.2)	Cadmium (2.9) Chromium (1.3) Vanadium (3.0)
118-D-6:4_Shallow_2	Mercury (12.0)	--
118-DR-2:2_Shallow	Mercury (1.6)	--
120-D-2_Shallow	Mercury (1.1)	--
126-D-2_Shallow_Focused	Boron (15.8) Lead (1.4)	Lead (2.0)
128-D-2_Shallow_1	Selenium (2.1)	--
132-D-1_Shallow	Mercury (10.0) Selenium (1.7)	--
132-D-1_Staging Pile Area Footprint	Mercury (6.8)	--
1607-D2-2_Shallow	Chromium (49.4) Mercury (4.8) Silver (5.9) Zinc (1.8)	Zinc (1.4)
1607-D5_Shallow	Barium (2.3) Boron (139.0) Lead (2.1)	Lead (3.0)
628-3_Shallow	Mercury (1.6)	--
628-3_Staging Pile Area_2	Mercury (1.1)	--
628-3_Staging Pile Area_3	Chromium (102.9)	Chromium (1.1)

* Analytes with exposure point concentrations consistent with background are excluded in these results.

HQ = hazard quotient

HQ = hazard quotient

NB = no background

SSL = soil screening level

Table 7-7. Summary of Chemicals in 100-H OU Surface Soils (0 to 15 ft [0 to 4.6 m]) Exceeding SSLs and Background

Waste Site/Decision Unit	Exceedances Based on Comparisons to SSLs and Background*	
	Plant/Invertebrate SSL-Based HQ	Wildlife SSL-Based HQ
100-H-21_Overburden	Arsenic (1.1)	Lead (1.0)
100-H-21_Shallow	Arsenic (1.3)	Lead (1.1)
100-H-28:1_Shallow_Focused	Barium (1.6) Boron (132.8)	--
100-H-3_Shallow	Arsenic (1.5) Boron (9.4) Mercury (2.6)	Lead (1.3)
100-H-35_Shallow_Focused_1	Chromium (51.8)	--
100-H-35_Shallow_Focused_2	Chromium (52.3) Mercury (1.0)	--
100-H-35_Shallow_Focused_3	Arsenic (1.9) Zinc (1.9)	Zinc (1.4)
100-H-37_Shallow_Focused	Arsenic (1.3) Lead (1.1)	Lead (1.5)
100-H-4_Shallow	Mercury (4.1)	--
100-H-4_Shallow_Focused	Boron (38.8) Uranium (2.0)	--
100-H-40_Shallow_Focused	Zinc (1.7)	Zinc (1.2)
100-H-49:2_Shallow_Focused	Boron (11.3) Chromium (47.0) Zinc (1.9)	Lead (1.3) Zinc (1.4)
100-H-51:4_Shallow_Focused	Zinc (1.6)	Zinc (1.2)
100-H-51:5_Shallow_Focused	Boron (52.6)	--
100-H-53_Shallow_Focused	Molybdenum (1.9) Zinc (1.6)	Lead (1.0) Zinc (1.2)
100-H-8_Shallow_Focused	Mercury (3.1)	Lead (1.3)
116-H-5_Staging Pile Area Footprint	Arsenic (1.0)	Lead (1.3)
116-H-7_Shallow	Chromium (49.1)	--
118-H-1:1_Overburden	Boron (21.1)	--
118-H-1:1_Shallow_1	Selenium (1.9)	--

Table 7-7. Summary of Chemicals in 100-H OU Surface Soils (0 to 15 ft [0 to 4.6 m]) Exceeding SSLs and Background

Waste Site/Decision Unit	Exceedances Based on Comparisons to SSLs and Background*	
	Plant/Invertebrate SSL-Based HQ	Wildlife SSL-Based HQ
118-H-1:1_Shallow_Focused	Boron (8.3) Chromium (55.0) Mercury (2.0)	--
118-H-3_Shallow_1	Arsenic (1.6) Lead (1.2) Zinc (2.7)	Lead (1.6) Zinc (2.0)
118-H-4_Staging Pile Area	Zinc (1.5)	Zinc (1.1)
118-H-5_Shallow	Boron (12.5)	--
118-H-6:5_Shallow_1	Arsenic (4.0) Lead (3.4)	Lead (4.8)
118-H-6:5_Shallow_2	Boron (24.4) Mercury (1.7)	--
118-H-6:5_Shallow_Focused	Arsenic (2.7) Lead (2.3)	Lead (3.2)
128-H-1_Overburden	Arsenic (4.1) Lead (5.1)	Lead (7.1)
128-H-1_Shallow_3	Arsenic (1.1) Boron (9.8) Lead (1.3)	Lead (1.8)
128-H-1_Shallow_4	Boron (8.9) Mercury (10.2)	Lead (1.3)
128-H-1_Staging pile area footprint_2	Arsenic (5.4) Lead (2.5)	Lead (3.5)
128-H-2_Shallow_Focused	Selenium (2.3)	--
128-H-3_Shallow_Focused	Arsenic (1.1) Lead (1.9)	Lead (2.6)
1607-H1_Overburden	Arsenic (1.4) Lead (1.2) Selenium (1.9)	Lead (1.7)
1607-H2_Overburden	Arsenic (1.6) Lead (1.1)	Lead (1.5)
1607-H2_Shallow	Chromium (510.0) Mercury (25.9)	Chromium (5.3) Lead (1.1) Mercury (1.4)

Table 7-7. Summary of Chemicals in 100-H OU Surface Soils (0 to 15 ft [0 to 4.6 m]) Exceeding SSLs and Background

Waste Site/Decision Unit	Exceedances Based on Comparisons to SSLs and Background*	
	Plant/Invertebrate SSL-Based HQ	Wildlife SSL-Based HQ
1607-H3_Overburden	Boron (9.5) Chromium (124.5)	Chromium (1.3)
1607-H4_Shallow	--	Lead (1.2)
600-151_Shallow_1	Arsenic (3.2) Boron (9.7) Lead (2.5) Selenium (1.8)	Lead (3.6)
600-151_Shallow_2	Arsenic (6.0) Lead (5.3)	Lead (7.5)
600-151_Shallow_3	Arsenic (5.4) Lead (5.5)	Lead (7.8)

* Analytes with exposure point concentrations consistent with background are excluded in these results.

HQ = hazard quotient

HQ = hazard quotient

NB = no background

SSL = soil screening level

7.4.5.1 100-D OU

The 100-D OU has 95 waste sites with CVP/RSVP data. Samples collected greater than 4.6 m (15 ft) bgs (deep and deep focused) were not included in the ERA; therefore, three (100-D-18, 100-D-19, and 116-D-6) of the 95 sites were not included in the ERA. No detections were observed at two waste sites (100-D-12 and 100-D-90). Therefore, plant/invertebrate and wildlife SSL HQs for 92 waste sites are provided in Appendix H, Table H-7. The SSL-based HQs were less than 1.0 for all COPCs in all of the decision units evaluated at 21 of the 92 waste sites. The following waste sites did not require further evaluation of ecological risk:

- 100-D Sites: 100-D-20, 100-D-21, 100-D-22, 100-D-3, 100-D-4, 100-D-48:1, 100-D-48:2, 100-D-48:3, 100-D-48:4, 100-D-49:2, 100-D-49:3, 100-D-49:4, and 100-D-80:1
- 116-D Sites: 116-D-1A, 116-D-2, 116-D-4, and 116-D-9
- 116-DR Sites: 116-DR-1,2, 116-DR-4, 116-DR-6, and 116-DR-7

The SSLs, background, and PRGs were not available for 22 COPCs. These COPCs were retained as an uncertainty and are discussed in Section 7.4.9. The EPCs for inorganic analytes barium, boron, cadmium, chromium, copper, lead, lithium, manganese, mercury, molybdenum, selenium, silver, total uranium isotopes, vanadium, and zinc exceeded 1 or both of the SSLs at the remaining waste sites, as presented in Appendix H, Table H-7. Within these waste sites, EPCs of analytes exceeded the plant/invertebrate SSLs,

while fewer analytes exceeded the wildlife SSLs. These waste site decision units were carried forward to the background evaluation.

7.4.5.2 100-H OU

The 100-H Source OU has 47 waste sites with CVP/RSVP data. Samples collected greater than 4.6 m (15 ft) bgs (deep and deep focused) were not included in the ERA. Five of the 47 waste sites (118-H-6:2, :3, and :6; 100-H-9, -10, -11, -12, -13, -14, and -31, which represents 5 remediated waste sites and 5 consolidated waste sites) were not included in the ERA. Therefore, plant/invertebrate and wildlife SSL HQs for 42 waste sites are provided in Table H-8 (Appendix H). The SSL-based HQs were less than 1.0 for all COPCs in all of the decision units evaluated at the following two waste sites: 100-H-24 and 116-H-3. These waste sites were eliminated from further evaluation of ecological risk.

The SSLs, background, and PRGs were not available for 13 COPCs. These COPCs were retained as an uncertainty and are discussed in Section 7.4.9. The EPCs for the inorganic analytes arsenic, barium, boron, chromium, lead, manganese, mercury, molybdenum, selenium, uranium, vanadium, and zinc exceeded one or both of the SSLs at the remaining waste sites. Within these waste sites, EPCs of analytes exceeded the plant/invertebrate SSLs, while fewer analytes exceeded the wildlife SSLs. These waste site decision units were carried forward to the background evaluation.

7.4.6 Background Evaluation

Although in exceedance of an SSL, EPCs for many of the COPCs within the remaining waste sites were below the 90th percentile background concentrations, so were eliminated from further evaluation. The comparisons of COPC EPCs to the 90th percentile background for the remaining waste sites are provided in Appendix H, Table H-9 and Table H-10 for 100-D OU and 100-H OU, respectively.

7.4.6.1 100-D OU

COPCs did not exceed the 90th percentile background concentrations in all of the decision units evaluated at 31 of the remaining waste sites. The background evaluation for the remaining waste sites is provided in Appendix H, Table H-9. The following 31 waste sites did not require further evaluation of ecological risks:

- 100-D Sites: 100-D-1, 100-D-2, 100-D-24, [100-D-31:1, 100-D-31:2], 100-D-31:10, 100-D-31:3, 100-D-31:4, 100-D-31:7, 100-D-31:9, 100-D-32, 100-D-50:5, 100-D-52, 100-D-74, 100-D75:3, 100-D-85:1, and 100-D-9
- 116-D Sites: 116-D-10 and 116-D-7
- 116-DR Sites: 116-DR-10 and 116-DR-9
- 118-D Site: 118-D-5
- 118-DR Site: 118-DR-1
- 120-D Site: 120-D-2
- 122-DR Site: 122-DR-1:2
- 130-D Site: 130-D-1
- 1607-D Sites: 1607-D2-1, 1607-D2-3, 1607-D2-4, and 1607-D4

- 600 Site: 600-30
- UPR-100 Site: UPR-100-D-5

Within the remaining waste sites, 46 decision units had COPC EPCs in exceedance of both an SSL and background. Barium, boron, cadmium, chromium, copper, lead, lithium, mercury, molybdenum, selenium, silver, vanadium, zinc, and total uranium isotopes were detected outside the range of background. The COPC EPCs detected in exceedance of background were carried forward to the risk assessment. Exceedances from the SSLs and background evaluations in soil are summarized in Table 7-6.

7.4.6.2 100-H OU

The COPCs did not exceed the 90th percentile background concentrations in all of the decision units evaluated at 13 of the remaining waste sites. The background evaluation for the remaining waste sites is provided in Appendix H, Table H-10. The following 13 waste sites did not require further evaluation of ecological risks:

- 100-H Sites: 100-H-17, 100-H-28:6, 100-H-41, 100-H-45, 100-H-5, 100-H-50, and 100-H-7
- 116-H Sites: 116-H-1 and 116-H-9
- 118-H Sites: 118-H-1:2, 118-H-2, and 118-H-6:4
- 600 Site: 600-152

Within the remaining waste sites, 41 waste site decision units had COPC EPCs in exceedance of both an SSL and background. Arsenic, barium, boron, chromium, lead, mercury, molybdenum, selenium, uranium, and zinc were detected outside the range of background. The COPC EPCs detected in exceedance of background were carried forward to the risk assessment. Exceedances from the SSLs and background evaluations in soil are summarized in Table 7-7.

7.4.7 PRG Evaluation Results

Further evaluation was conducted on those waste sites that were not eliminated in the SSL and background evaluations. Risks were evaluated based on the resulting HQs and are provided in Tables H-11 and H-12 (Appendix H) and summarized in Table 7-8 and Table 7-9 for the 100-D and 100-H OUs, respectively.

Table 7-8. Summary of 100-D OU Waste Sites Ecological Evaluation Based on PRGs for Surface Soils (0 to 15 ft [0 to 4.6 m])

Waste Site/Decision Unit	Exceedances Based on Comparisons to PRGs	
	Plant/Invertebrate HQ	Wildlife HQ
100-D-28:1_Shallow	Mercury (1.9)	Selenium (1.2)
100-D-31:5_Overburden	--	Selenium (1.2)
100-D-31:5_Shallow	--	Selenium (1.0)
100-D-31:6_Overburden	Mercury (2.9)	--
100-D-31:6_Shallow	Mercury (1.9)	--
100-D-31:8_Shallow_Focused_1	Vanadium (1.1)	Vanadium (2.2)

Table 7-8. Summary of 100-D OU Waste Sites Ecological Evaluation Based on PRGs for Surface Soils (0 to 15 ft [0 to 4.6 m])

Waste Site/Decision Unit	Exceedances Based on Comparisons to PRGs	
	Plant/Invertebrate HQ	Wildlife HQ
100-D-31:8_Shallow_Focused_2	Barium (4.4) Boron (5.9) Molybdenum (1.2)	Boron (5.3)
100-D-42, 100-D-43, 100-D-45_Shallow	Copper (1.6)	--
100-D-47_Shallow_Focused	--	Vanadium (2.0)
100-D-56:1_Overburden	Selenium (1.0)	Selenium (1.5)
100-D-56:1_Shallow	--	Selenium (1.2)
100-D-7_Staging pile area footprint	Mercury (1.0)	--
100-D-70_Shallow_Focused	Copper (1.3)	--
100-D-83:4_Shallow_Focused	Mercury (3.2)	--
100-D-84:1_Shallow_Focused	Vanadium (1.1)	Vanadium (2.2)
100-D-88_Shallow_Focused	Vanadium (1.2)	Vanadium (2.4)
100-D-94_Shallow_Focused	Mercury (1.9)	--
116-DR-8_Shallow	Lithium (2.0)	--
118-D-4_Shallow_Focused	Vanadium (1.0)	Vanadium (2.1)
118-D-6:4_Shallow_2	Mercury (4.0)	--
132-D-1_Shallow	Mercury (3.3)	--
132-D-1_Staging Pile Area Footprint	Mercury (2.3)	--
1607-D2-2_Shallow	Mercury (1.6) Silver (3.9)	--
1607-D5_Shallow	Barium (2.2) Boron (2.4)	Boron (2.2)

HQ = hazard quotient
PRG = preliminary remediation goal

Table 7-9. Summary of 100-H OU Waste Sites Ecological Evaluation Based on PRGs for Surface Soils (0 to 15 ft [0 to 4.6 m])

Waste Site/Decision Unit	Exceedances Based on Comparisons to PRGs	
	Plant/Invertebrate HQ	Wildlife HQ
100-H-28:1_Shallow_Focused	Barium (1.5) Boron (2.3)	Boron (2.1)
100-H-4_Shallow	Mercury (1.4)	--
100-H-53_Shallow_Focused	Molybdenum (1.9)	--
100-H-8_Shallow_Focused	Mercury (1.0)	--
118-H-6:5_Shallow_1	--	Lead (1.1)
128-H-1_Overburden	--	Lead (1.6)
128-H-1_Shallow_4	Mercury (3.4)	--
1607-H2_Shallow	Chromium (1.4) Mercury (8.6)	Chromium (1.9) Mercury (1.7)
600-151_Shallow_2	--	Lead (1.7)
600-151_Shallow_3	--	Lead (1.8)

HQ = hazard quotient
PRG = preliminary remediation goal

7.4.7.1 100-D OU

The following 15 waste sites did not exceed the plant/invertebrate or the wildlife PRGs (HQs were less than 1.0) and were eliminated from further evaluation (Appendix H, Table H-11):

- 100-D Sites: 100-D-13, 100-D-15, 100-D-29, [100-D-31:3, 100-D-31:4], 100-D-56:2, 100-D-61, 100-D-82, and 100-D-87
- 116-D Site: 116-D-8
- 116-DR Site: 116-DR-5
- 118-D Site: 118-D-1
- 118-DR Site: 118-DR-2:2
- 120-D Site: 120-D-2
- 126-D Site: 126-D-2
- 628-D Site: 628-D-3

The EPCs for the inorganic analytes barium, boron, copper, lithium, mercury, molybdenum, selenium, silver, and vanadium exceeded one or both groups of PRGs (plants/invertebrates, wildlife). These COPCs

will be retained as COPECs in one or more of the remaining 24 waste site decision units (Appendix H, Table H-11) and will be further addressed in the SMDP.

The risk assessment identified COPECs for the following waste site decision units because of potential ecological risks to plants or invertebrates that may be from Hanford Site releases (Table 7-8):

- 100-D-28:1_Shallow: Mercury, Selenium
- 100-D-31:5_Overburden: Selenium
- 100-D-31:5_ Shallow: Selenium
- 100-D-31:6_Overburden: Mercury
- 100-D-31:6_Shallow: Mercury
- 100-D-31:8_Shallow_Focused_1: Vanadium
- 100-D-31:8_Shallow_Focused_2: Barium, Boron, Molybdenum
- 100-D-42, 100-D-43, 100-D-45_Shallow: Copper
- 100-D-47_Shallow_Focused: Vanadium
- 100-D-56:1_Overburden: Selenium
- 100-D-56:1_Shallow: Selenium
- 100-D-7_Staging pile area footprint: Mercury
- 100-D-70_Shallow_Focused: Copper
- 100-D-83:4_Shallow_Focused: Mercury
- 100-D-84:1_Shallow_Focused: Vanadium
- 100-D-88_Shallow_Focused: Vanadium
- 100-D-94_Shallow_Focused: Mercury
- 116-DR-8_Shallow: Lithium
- 118-D-4_Shallow_Focused: Vanadium
- 118-D-6:4_Shallow_2: Mercury
- 132-D-1_Shallow: Mercury
- 1607-D2-2_Shallow: Mercury, Silver
- 1607-D5_Shallow: Barium, Boron

7.4.7.2 100-H OU

The following 19 waste sites did not exceed the plant/invertebrate PRGs or the wildlife PRGs (HQs were less than 1.0 and were eliminated from further evaluation) (Appendix H, Table H-12):

- 100-H Sites: 100-H-21, 100-H-3, 100-H-35, 100-H-37, 100-H-40, 100-H-49:2, 100-H-51:4, 100-H-51:5
- 116 Sites: 116-H-5, 116-H-7
- 118-H Sites: 118-H-3, 118-H-4, 118-H-5, 118-H-1:1
- 128-H Sites: 128-H-2, 128-H-3
- 1607 Sites: 1607-H1, 1607-H3, 1607-H4

The EPCs for the inorganic analytes barium, boron, chromium, lead, mercury, molybdenum exceeded one or both groups of PRGs (plants/invertebrates, birds/mammals). These COPCs will be retained as COPECs in one or more of the remaining four waste site decision units (Appendix H, Table H-12) and will be further addressed in the SMDP.

The risk assessment identified COPECs for the following waste site decision units because of potential ecological risks to plants or invertebrates that may be attributable to past site practices (Table 7-9):

- 100-H-28:1_Shallow_Focused: Barium, Boron
- 100-H-4_Shallow: Mercury
- 100-H-53_Shallow_Focused: Molybdenum
- 100-H-8_Shallow_Focused: Mercury
- 118-H-6:5_Shallow_1: Lead
- 128-H-1_Overburden: Lead
- 128-H-1_Shallow_4: Mercury
- 1607-H2_Shallow: Chromium, Mercury
- 600-151_Shallow_2: Lead
- 600-151_Shallow_3: Lead

7.4.8 Characterization of Drinking Water Ingestion

The EPCs from seep water along the 100-D and 100-H riparian areas of the Columbia River were evaluated for drinking water intake by birds and mammals representing feeding guilds in the upland and riparian areas of the Columbia River Corridor. The results of these comparisons for inorganics are provided in Appendix H (Table H-13 and Table H-14 for 100-D for 100-H, respectively). Under this scenario, doses of aluminum at 100-H and aluminum and nitrate at 100-D were greater than 1 percent (that is, HQ greater than 0.01) for one or more of the evaluated receptors, while exposure from all other chemicals to all other receptors produced HQs less than 0.01. Thus, other than for the chemical-source OU combinations listed above, exposure from chemicals to all receptors produced HQs less than 1, indicating no additional risk for wildlife exposure to nonradionuclides from drinking seeps at the 100-D/H Area. Further, the results of the evaluation presented in Appendix H should be considered acceptable for all of the chemical-source OU combinations (except the three listed), as inclusion of drinking ingestion to the exposure models presented in this chapter (Sections 7.4.1 through 7.4.3) would not have altered the outcomes. Inclusion of drinking ingestion in the development of SSLs and PRGs is not warranted. For those chemical-Source OU-receptor combinations listed, further discussion is provided below:

- **Aluminum**—For aluminum, the dose from ingestion of prey and soil is not significant in terrestrial environments with soil pH greater than 5.5 as the aluminum is bound and unavailable for biological uptake (OSWER Directive 9285.7-60). Hence, for wildlife residing in the circumneutral soil of the 100-D and 100-H Source OUs, drinking ingestion by wildlife represents the primary contribution to the total dose of aluminum and, even for mammals, yielded an HQ less than 0.1 (Appendix H, Tables H-13 and H-14) under the worst-case scenario, even for the more susceptible bats.

- **Nitrate**—While the drinking ingestion dose at 100-D yielded an HQ of 0.013 for bats, dietary ingestion shown in Table H-7 for 100-D is below 0.001 for all waste sites. Thus, dietary ingestion of nitrate would be insignificant relative to drinking ingestion from seeps, and total exposure from the combined prey ingestion and drinking water ingestion would not be at a level of concern. Similarly, the drinking ingestion dose at 100-H yielded an HQ of 0.26 for elk, 0.36 for badgers, and 0.47 for

bats, and the dietary ingestion shown in Table H-8 for 100-H is below 0.0001 for all waste sites. Dietary ingestion of nitrate would be insignificant relative to drinking ingestion from seeps, and total exposure from the combined prey ingestion and drinking water ingestion would not be at a level of concern. Thus, with the HQs being less than 1 under a worst-case scenario and with more available and uncontaminated sources of drinking water available (for example, the Columbia River), there is no unacceptable risk. Exposure to nitrate through drinking seep water does not warrant further evaluation, and inclusion in SSL or PRG development is not required.

Evaluation of radionuclide doses from wildlife drinking seep water is included in Appendix H, Table H-15. EPCs for seeps were compared to the lower of BCGs for terrestrial and riparian animals. The total SOF for wildlife drinking seep water from 100-D was 0.037. With the maximum SOF from terrestrial soil pathways from any waste site within the 100-D Source OUs being 0.04 (Appendix H, Table H-7), there is no additional risk for wildlife exposure to radionuclides from drinking seeps at 100-D. Similarly, the total drinking ingestion SOF for 100-H seep water was 0.12. Combined with a worst-case SOF from terrestrial soil pathways of 0.23 within 100-H waste sites (Appendix H, Table H-8), there is no risk for wildlife exposure to radionuclides from drinking seep water at 100-H.

Given the results provided in Appendix H, Tables H-13 through H-15, there is no significant risk to wildlife in the 100-D/H Source OUs from drinking freshwater seeps along the Columbia River in the 100-D/H riparian area. Further, the results of the evaluation presented in Appendix H should be considered acceptable for all of the chemical-source OU combinations because the inclusion of drinking ingestion in the exposure models presented in this chapter (Sections 7.4.1 through 7.4.3) would not have altered the risk outcomes or conclusions. Inclusion of drinking ingestion in the development of SSLs and PRGs is therefore not warranted.

7.4.9 Uncertainties Assessment

Uncertainties are present in all risk assessments because of the limitations of the available data and the need to make certain assumptions and extrapolations based on incomplete information. In addition, the use of various models (for example, uptake and food web exposures) carries associated uncertainty as to how well the model reflects actual conditions. Because conservative assumptions were generally used in the exposure and effects assessments, these uncertainties are more likely to overestimate rather than underestimate the likelihood and magnitude of risks to ecological receptors. The following uncertainties and limitations are associated with the proposed methodology and available data for the ERA:

- **Data Use**—The quantitative evaluation of chemical concentrations in soil included surface soil from the 0 to 4.6 m (15 ft) depth range. Ecology uses a standard point of compliance in soil of 4.6 m (15 ft) for demonstrating protection of ecological receptors (MTCA, "Terrestrial Ecological Evaluation Procedures" [WAC 173-340-7490(4)(b)]). This depth range may overestimate the depth to which many terrestrial receptors would be exposed. MTCA (WAC 173-340) identifies the biologically active zone as 1.8 to 4.6 m (6 to 15 ft) (MTCA [WAC 173-340]). Evaluation of data that extends beyond the biologically active zone could either overestimate or underestimate risk. For this ERA, the depth from 1.8 to 4.6 m (6 to 15 ft) is also included because human activities could bring materials from that depth to the surface, creating a complete exposure pathway.

  No toxicological data or background values were available for some COPCs (2,4,5-trichlorophenol, 2,4-DB(4-(2,4-dichlorophenoxy)butanoic acid), 2,4-dichlorophenol, 2,4-dinitrophenol, 2-chloronaphthalene, 4,4'-DDD (dichlorodiphenyldichloroethane), acetone, alpha-BHC, butylbenzylphthalate, carbazole, dibenzofuran, di-n-butylphthalate, dinoseb(2-sec butyl-4,6-dinitrophenol), gamma-BHC (Lindane), heptachlor epoxide, isophorone, nickel-63, neptunium-237, nitrogen in nitrate, nitrogen in nitrite, nitrogen in nitrite and nitrate, total petroleum hydrocarbons -

diesel range extended to C36, total petroleum hydrocarbons - motor oil (high boiling), and total petroleum hydrocarbons – gasoline range) or were limited for some COPC/receptor combinations. Therefore, SSLs could not be calculated for all receptors or COPCs. Exclusion of COPCs from SSL development may not adequately address aggregate risk at a site, although remedial alternatives protective of receptors with SSLs may also be protective of receptors lacking sufficient toxicity data. In addition, the absence of SSLs for plants and soil invertebrates can be addressed through site-specific bioassays, which are a component of Tier 2.

Bioavailability and toxicity of metals are functions of many factors including soil pH, with metals (e.g., aluminum, iron, lead, mercury) generally being more bioavailable and toxic at low pHs (*Guidance for Developing Ecological Soil Screening Levels* [OSWER Directive 9285.7-55]). The pH levels for soil used to develop plant toxicity values range from 3 to 8. (mean=6.3) (*Tier 1 Risk-Based Soil Concentrations Protective of Ecological Receptors at the Hanford Site* [CHPRC-00784]). The pH levels for soil used to develop invertebrate toxicity values were between 3.8 and 8.1 (mean = 5.6) (*Tier 1 Risk-Based Soil Concentrations Protective of Ecological Receptors at the Hanford Site* [CHPRC-00784]). The minimum soil pH reported in the RCBRA (DOE/RL-2007-21) in riparian and upland soil was 6.6. Because the range of pH values in soil associated with plant and soil invertebrate toxicity values within the published literature include values substantially lower than those present throughout most of the Hanford Site, the resulting SSLs for plants and soil invertebrates may not accurately represent toxicity. Because metals are more bioavailable at lower pH, the SSLs may overestimate concentrations in Hanford Site soil that would be toxic to plants and soil invertebrates; therefore, risk estimates may be overly conservative. Evaluating this potential overestimation of bioavailability was one of the goals of a 2011 Hanford Site field effort to collect soil with a pH range more reflective of Hanford Site soils (*Tier 2 Terrestrial Plant and Invertebrate Preliminary Remediation Goals (PRGs) for Nonradionuclides for Use at the Hanford Site* [ECF-HANFORD-11-0158]). With the exception of four samples collected from within the River Corridor, the range of pH values from samples collected for the 2011 study was between 5.8 and 8.7 with all but 5 of 67 samples above the minimum pH of 6.6 identified in previous RCBRA (DOE/RL-2007-21) soil samples. Further, oxidized environments (upland or well-aerated soils like those at the Hanford Site) promote the precipitation of ferric-oxide compounds, which are not available to plants for uptake. Thus, the PRGs more accurately reflect the actual bioavailability of potential contaminants within the Hanford Site soil than they do the SSLs developed using published data from laboratory studies and other sites.

With respect to TPH (both high boiling point motor oil and diesel extended to the C36 range), though no SSL or PRG was previously developed for soil at the Hanford Site, published literature is available to provide prospective. In "Ecotoxicity Test Data for Total Petroleum Hydrocarbons in Soil: Plants and Soil-Dwelling Invertebrates" (Efroymson et al., 2004), the authors compiled a literature review on toxicological effects to plant and invertebrates with the results suggesting invertebrates are more sensitive to petroleum hydrocarbons than plants. Using lube oil to represent TPH-motor oil, no-effect thresholds ranged from 15 to 1,490 mg/kg in soil and EC20 was found as low as 15 to 149 mg/kg. Conversely, lube oil NOAECs for plants ranged from 969 mg/kg to 12,000 mg/kg. MTCA lists ecological indicator soil concentrations (WAC 173-340, Table 749-3) for soil biota for diesel and gasoline range organics at 200 mg/kg and 100 mg/kg, respectively, based on original work published at ORNL (*Toxicological Benchmarks for Contaminants of Potential Concern for Effects on Soil and Litter Invertebrates and Heterotrophic Process: 1997 Revision* [ES/ER/TM-126/R2]). The highest concentration of TPH-diesel was at 160 mg/kg measured 126-D-2_Shallow_Focused, and the highest concentration of TPH-motor oil was at 188 mg/kg measured at 100-H-4_Shallow_Focused. Given these maximum concentrations are below the MTCA diesel range ecological indicator soil

concentration and on the low end of the range of NOAECs, no further evaluation of TPH is warranted.

- **Wildlife TRVs**—Data on the toxicity of many chemicals to the receptor species were sparse or lacking, requiring the extrapolation of data from other wildlife species or from laboratory studies with non-wildlife species. This is a typical limitation and extrapolation for ERAs because so few wildlife species have been tested directly for most chemicals. The uncertainties associated with toxicity extrapolation were minimized through the selection of the most appropriate test species for which suitable toxicity data were available. The factors considered in selecting a test species to represent a receptor species included taxonomic relatedness, trophic level, foraging method, and similarity of diet.

  A second uncertainty related to the derivation of TRVs applies to metals. Most of the toxicological studies on which the TRVs for metals were based used forms of the metal (such as salts) that have high water solubility and high bioavailability to receptors. Because the analytical samples on which site-specific exposure estimates were based measured total metal, regardless of form, and these highly bioavailable forms are expected to compose only a fraction of the total metal concentration, this is likely to overestimate potential risks for these chemicals. A recent study was conducted comparing the toxicity of laboratory-spiked soil versus aged field-collected soil and the predictive ability of the European Union's predicted no-effect concentrations for five metals. The study concluded that total metals concentrations in field-collected soil are poor indicators of toxicity ("Toxicity of Trace Metals in Soil as Affected by Soil Type and Aging After Contamination: Using Calibrated Bioavailability Models to Set Ecological Soil Standards" [Smolders et al., 2009]).

- **Chemical Mixtures**—The SSLs used in this assessment are based on exposure to individual analytes. Information on the ecotoxicological effects of chemical interactions is generally lacking, which required (as is standard for evaluations of ecological risk) that the chemicals be evaluated on a compound-by-compound basis during the comparison to SSLs. This could underestimate risks (if there are additive or synergistic effects among chemicals) or overestimate risks (if there are antagonistic effects among chemicals). Assessment of data in this report resulted in a description of potential exposure risks because of metals, which are typically known to be additive. In this case, effects may be underestimated.

- **Receptor Species Selection**—Reptiles were identified as being part of the food web present at the Hanford Site, but were not evaluated quantitatively even when exposure pathways were complete. A qualitative assessment of potential risk to these taxa can be made by using the results of quantitative evaluation for other fauna with similar diets and assumed similarity in metabolizing COPECs to make inferences. Considering the results of quantitative evaluation of avian receptors can indicate the potential for risks to these taxa. The uncertainty associated with the lack of toxicological data for reptiles and inferring risk from other fauna could either overestimate or underestimate risks.

  It was also assumed that reptiles were neither exposed to significantly higher concentrations of chemicals nor more sensitive to chemicals than the other receptor species evaluated in the food web model. This assumption was a source of uncertainty in the ERA. In addition, there is uncertainty associated with the use of specific receptor species to represent larger groups of organisms (for example, guilds).

- **Food Web Exposure Modeling**—While life history data are available for many of the wildlife species at the Hanford Site, Hanford Site-specific data were unavailable for several specific parameters included in the desktop food web models used to estimate exposure to wildlife. These factors included food ingestion rate, incidental soil ingestion as a percent or as a rate, home

range, and dietary composition established as the percent of stomach contents. As a result of this lack of Hanford Site-specific data, exposure parameters were modeled based on allometric relationships or on data from the same species in other portions of its range. Because diet composition as well as food and soil ingestion rates can differ among individuals and locations, published parameter values may not accurately reflect individuals at the Hanford Site. Consequently, SSLs may be either overconservative or underconservative. For example, the wildlife EcoSSLs were derived with a model that incorporates prey tissue items that compose 100 percent of the receptor's diet coming from the site, not accounting for food obtained in adjacent uncontaminated areas, whereas MTCA (WAC 173-340) values account for offsite prey consumption. Therefore, the assumed contributions of ingestion of analytes in prey tissues for the wildlife EcoSSLs are greater than those used to develop the MTCA (WAC 173-340) values and likely overestimate risk.

Ultimately, there is uncertainty with both the MTCA (WAC 173-340) and EPA values used as SSLs with respect to site-specificity. The wildlife PRGs employed in this ERA are more site-specific than the SSLs because prey concentrations were estimated with Hanford Site data. However, there is also uncertainty in those values associated with the percentage of diet obtained from the site. In applying the PRGs, the assumption was that 100 percent of the food ingestion was from the site, which, in many cases, is an overestimate. This assumption was evaluated on a case-by-case basis to aid the SMDP presented in Section 7.6.

- **Central Tendency versus Maximum Exposure Concentration Estimates**—As is typical in an ERA, a finite number of samples of environmental media is used to develop the exposure estimates. The maximum measured concentration provides a conservative estimate for sessile biota or those with a limited home range. The most realistic exposure estimates for mobile species with relatively large home ranges and for species populations (even those that are sessile or have limited home ranges) are those based upon an estimate of central tendency of chemical concentrations in each medium to which these receptors are exposed. This is reflected in the wildlife dietary exposure models contained in *Wildlife Exposure Factors Handbook* (EPA/600/R-93/187). It is possible, however, that receptors could spend additional time foraging at a nearby waste site and thus be exposed to analytes from more than one site. Thus, EPC estimates of contaminants in individual waste site media and food sources may not accurately represent contaminant exposure to a receptor ranging into other sites. However, assuming an AUF of 1 will likely result in a conservative estimate of exposure because offsite foraging would likely be conducted in uncontaminated areas. Given the mobility of the upper trophic-level receptor species used in the ERA, the use of maximum chemical concentrations as EPCs when UCLs were not calculated by ProUCL to estimate the exposure via food webs is very conservative. This conservatism was reduced to levels that are more realistic when the number of samples collected in a site was adequate in sample size to develop a UCL on the mean.

- **Comparisons to Background Concentrations**—Background concentrations were used to judge whether measured concentrations within waste sites reflect site-related activities, background, or a combination. If site chemical concentrations were consistent with these background levels, it was assumed that the concentrations were not site- related. Comparisons to background in this evaluation include the use of the 90th percentile of the background dataset as compared to the EPC. Thus, 10 percent of the background dataset is higher than the 90th percentile. Concentrations measured above background may be within the distribution of background variability and could represent a false positive risk. The possibility also exists that concentrations below background were indeed site-related, rendering the assumption false. However, the effect of this possibility is minimal because metals and radioisotopes at concentrations consistent with background conditions should exhibit

no different ecological effects than those common in areas not affected by releases, regardless of their source.

- **Risk Estimates Associated with Remedial Investigation and Limited Field Investigation Soil Data**—In addition to the waste site remediation data (CVP/RSVP), the following two sources of data were considered for use in the ERA. These sources of data include the following:
  - Vadose zone data collected for the RI to fill data gaps associated with the nature and extent of contamination or associated with understanding the fate and transport of contaminants
  - Limited field investigation data collected in 1992 from the 100-D/H OUs

These data were collected for purposes other than fulfilling needs of the risk assessment; as such, they were not used to evaluate risks quantitatively. However, these data were evaluated qualitatively by comparing concentrations of analytes to risk-based screening levels to determine whether the results could be useful for risk management decisions.

- RI and LFI data are described in Chapter 6, Section 6.2.6.6. All RI and LFI soil data from the soil borings and wells described in Chapter 6 were compared to the PRGs and SSLs used in the ERA. Detailed datasets and vertical profiles are provided in Section 4.2.2, and the soil borings/wells and associated depth intervals for data in the ERA are summarized in Appendix H (Tables H-16 and H-17).

- Similar to the CVP/RSVP data, soil data from each soil boring, well, or test pit were grouped by depth. Soil data were processed and reduced using the same methods as those described in Section 7.1. Soil samples collected from depth intervals ranging from 0 to 4.6 m (15 ft) bgs were combined, and the maximum detected concentration was compared to the Hanford Site background concentration and the lowest available ecological PRG value or the SSL when no PRG was available. Soil samples collected from depth intervals greater than 4.6 m (15 ft) bgs were not evaluated because they extend beyond the 4.6 m (15 ft) bgs standard point of compliance for ecological receptors defined by MTCA ("Terrestrial Ecological Evaluation Procedures" [WAC 173-340-7490(4)(b)]).

- A comparison of the range of detected concentrations to ecological PRGs or SSLs from each of these sample locations is provided in Appendix H, Tables H-18 and H-19. The wells and test pits that report detected concentrations greater than the ecological PRGs and SSLs for the 100-D and 100-H Source OUs are summarized in Table 7-10 and Table 7-11, respectively.

- For the 100-D Source OU (shown in Table 7-10), four LFI sample locations (100-D-12 Sodium Dichromate site, 116-D-4 Crib, 116-DR-9 Retention Basin, and 130-D-1 Underground Tank) report soil concentrations greater than ecological SSLs. Three waste sites (100-D-12 Sodium Dichromate site, 116-D-4 Crib, and 116-DR-9 Retention Basin) have been remediated under the interim action ROD. At the 130-D-1 Underground Tank (199-D5-27), bis(2-ethylhexyl)phthalate was detected at a concentration of 6.3 mg/kg in the 3 to 3.6 m (10 to 12 ft) bgs depth interval. The bis(2-ethylhexyl)phthalate concentration of 6.3 mg/kg is greater than the ecological SSL of 0.14 mg/kg. The 130-D-1 Underground Tank is an accepted waste site that will be remediated.

- For the 100-H Source OU (shown in Table 7-11), three LFI sample locations (116-H-1 Trench, 116-H-7 Retention Basin, and the 116-H-9 Crib) report soil concentrations greater than ecological SSLs. These three waste sites have been remediated under the interim action ROD.

Table 7-10. Summary of Ecological Risk Comparisons at 100-D Source OU for RI Data, CVP/RSVP Data, and LFI Data

Waste Site	RI Data	Shallow Zone Ecological Risks?	CVP/RSVP Data*	Shallow Zone Ecological Risks?	LFI Data	Shallow Zone Ecological Risks?
Soil Borings Installed to Characterize Residual Contamination Beneath the Remediated Waste Site						
100-D-4 Trench	100-D-4 Trench (Test Pit)	No individual risks > thresholds	CVP-98-00004	No individual risks > thresholds	--	--
100-D-12 Sodium Dichromate Site	100-D-12 French Drain (Test Pit)	No individual risks > thresholds	CVP-2000-00016	No COPCs detected	100-D-12 TP1	Chromium (1.5 m [5 ft] bgs)
					100-D-12 TP2	No individual risks > thresholds
					100-D-12 TP3	No individual risks > thresholds
100-D-56 Sodium Dichromate Pipeline (Well 9)	C8375	No samples collected from this depth range	Accepted	--	--	--
116-D-1A Trench (Well 4)	C7622	No samples collected from this depth range	CVP-2000-00010	No individual risks > thresholds	199-D5-21	No individual risks > thresholds
116-D-1B Trench	C7855	No samples collected from this depth range	--	--	199-D5-29	No individual risks > thresholds
116-D-4 Crib	116-D-4 Crib (Test Pit)	No individual risks > thresholds	CVP-2000-00008	No COPCS reported above background	199-D5-24	Thallium (0.9 to 1.7 m [3 to 5.5 ft] bgs)
116-D-7 Retention Basin	C7851	No samples collected from this depth range	CVP-99-00007	No individual risks > thresholds	199-D8-60	No individual risks > thresholds
116-DR-1&2 Trench	C7852	No samples collected from this depth range	CVP-2000-00002	No individual risks > thresholds	199-D8-61	No samples collected from this depth range
					199-D8-62	No samples collected from this depth range

Table 7-10. Summary of Ecological Risk Comparisons at 100-D Source OU for RI Data, CVP/RSVP Data, and LFI Data

Waste Site	RI Data	Shallow Zone Ecological Risks?	CVP/RSVP Data*	Shallow Zone Ecological Risks?	LFI Data	Shallow Zone Ecological Risks?
116-DR-9 Retention Basin	C7850	No samples collected from this depth range	CVP-99-00006	No individual risks > thresholds	199-D8-64	Bis(2-ethylhexyl)phthalate (2.7 to 3.6 m [9 to 11.8 ft] bgs)
					199-D8-65	No individual risks > thresholds
					199-D8-66	No individual risks > thresholds
118-D-6 Reactor Fuel Storage Basin	C7857	No samples collected from this depth range	No (concrete)	--	--	--
Wells Installed to Characterize Contamination in the Unconfined Aquifer						
100-D Well No. 2	C7620	No samples collected from this depth range	--	--	--	--
100-D Well No. 3	C7621	No samples collected from this depth range	--	--	--	--
100-D Well No. 5	C7623	No samples collected from this depth range	--	--	--	--
Well 9 (redrilled for data gap 2)	C7866	No samples collected from this depth range	--	--	--	--
Wells Installed to Characterize Contamination Beneath the Unconfined Aquifer in the RUM						
100-D RUM Well R4	C7624	No samples collected from this depth range	--	--	--	--
100-D RUM Well R5	C7625	No samples collected from this depth range	--	--	--	--
100-D RUM Well R5 Redrill	C8668	No samples collected from this depth range				

Table 7-10. Summary of Ecological Risk Comparisons at 100-D Source OU for RI Data, CVP/RSVP Data, and LFI Data

Waste Site	RI Data	Shallow Zone Ecological Risks?	CVP/RSVP Data*	Shallow Zone Ecological Risks?	LFI Data	Shallow Zone Ecological Risks?
Soil Borings Installed during LFI to Characterize Waste Site in 100-DR-1 and 100-DR-2 OUs						
116-D-6 French Drain	--	--	CVP-2000-00009	No individual risks > thresholds	199-D5-25	No samples collected from this depth range
116-D-2 Crib	--	--	CVP-2000-00013	No COPCS reported above background	199-D5-22	No individual risks > thresholds
116-D-9 Crib	--	--	CVP-2000-00012	No individual risks > thresholds	199-D5-26	No samples collected from this depth range
132-D-3 Pumping Station	--	--	RSVP-2005-033	Facility	199-D5-28	No samples collected from this depth range
116-D-5 Outfall Structure	--	--	Accepted Waste Site	--	199-D8-59	No samples collected from this depth range
116-DR-5 Outfall Structure	--	--	RSVP-2010-051	No individual risks > thresholds	199-D8-63	No samples collected from this depth range
116-D-3 French Drain	--	--	No Action Waste Site	--	199-D5-23	No samples collected from this depth range
130-D-1 Underground Tank	--	--	Accepted Waste Site	--	199-D5-27	Bis(2-ethylhexyl) phthalate (3 to 3.7 m [10 to 12 ft] bgs)
108-D/ Sodium Dichromate Tanks	--	--	Not listed as a WIDS waste site	--	108-D-TNKS-TP-1	No individual risks > thresholds
					108-D-TP-1	No individual risks > thresholds
116-DR-3 Trench	--	--	Accepted Waste Site	--	118-D-5 TP	No individual risks > thresholds
					116-DR-3 TP	No individual risks > thresholds
116-DR-7 Crib	--	--	CVP-2000-00019	No individual risks > thresholds	199-D5-30	No individual risks > thresholds

* Complete reference citations are provided in Chapter 11.

Table 7-11. Summary of Ecological Risk Comparisons at 100-H Source OU for RI Data, CVP/RSVP Data, and LFI Data

Waste Site	RI Data	Shallow Zone Ecological Risks?	CVP/RSVP Data*	Shallow Zone Ecological Risks?	LFI Data	Shallow Zone Ecological Risks?
Soil Borings Installed to Characterize Residual Contamination Beneath the Remediated Waste Site						
116-H-1 Trench	C7864	No samples collected from this depth range	CVP-2000-00026	No individual risks > thresholds	199-H4-58	Lead (3 to 3.7 m [10 to 12 ft] bgs)
116-H-2 Trench/Crib	116-H-2 Trench/Crib (Test Pit)	Bis(2-ethylhexyl)phthalate (2.7 to 3.4 m [9 to 11 ft] bgs; 3.4 to 4 m [11 to 13 ft] bgs)	CVP-2000-00031	No individual risks > thresholds	199-H4-59	No individual risks > thresholds
116-H-4 Pluto Crib	C7862	No individual risks > thresholds	Accepted Waste Site	--	--	--
116-H-6 Solar Evaporation Basin	C7860	No individual risks > thresholds	--	--	--	--
116-H-7 Retention Basin	C7861	No samples collected from this depth range	CVP-2000-00027	No individual risks > thresholds	199-H4-61	Lead (0.3 to 0.9 m [1 to 3 ft] bgs); carbon-14 (2.4 to 3 m [8 to 10 ft] bgs); mercury (2.4 to 3 m [8 to 10 ft] bgs; 3 to 3.8 m [9.8 to 12.4 ft] bgs)
118-H-6 Reactor Fuel Storage Basin	C7863	No samples collected from this depth range	CVP-2006-00003	No individual risks > thresholds	--	--
1607-H4 Septic System	1607-H4 Septic System (Test Pit)	Benzo(a)anthracene, benzo(b)fluoranthene, benzo(a)pyrene, benzo(k)fluoranthene, benzo(ghi)perylene, bis(2-ethylhexyl)phthalate, chrysene, fluoranthene, indeno(1,2,3-cd)pyrene (4 to 4.6 m [13 to 15 ft] bgs)	CVP-2000-0025	No individual risks > thresholds	--	--

Table 7-11. Summary of Ecological Risk Comparisons at 100-H Source OU for RI Data, CVP/RSVP Data, and LFI Data

Waste Site	RI Data	Shallow Zone Ecological Risks?	CVP/RSVP Data*	Shallow Zone Ecological Risks?	LFI Data	Shallow Zone Ecological Risks?
Wells Installed to Characterize Contamination in the Unconfined Aquifer						
100-H Well No. 6	C7626	No samples collected from this depth range	--	--	--	--
100-H Well No. 7	C7627	No samples collected from this depth range	--	--	--	--
100-H Well No. 10	C7628	No samples collected from this depth range	--	--	--	--
100-H Well No. 11	C7629	No samples collected from this depth range	--	--	--	--
100-H Well No. 12	C7630	No individual risks > thresholds	--	--	--	--
Wells Installed to Characterize Contamination Beneath the Unconfined Aquifer in the RUM						
100-H RUM Well R1	C7639	No samples collected from this depth range	--	--	--	--
100-H RUM Well R2	C7640	No samples collected from this depth range	--	--	--	--
100-H RUM Well R3	C7631	No samples collected from this depth range	--	--	--	--
Soil Borings Installed during LFI to Characterize Priority Waste Site in 100-HR-1 and 100-HR-2 OUs						
116-H-3 French Drain	--	--	CVP-2000-00032	No individual risks > thresholds	199-H4-60	No samples collected from this depth range
116-H-9 Crib	--	--	RSVP-2009-047	No individual risks > thresholds	199-H4-62	Aluminum, barium, cadmium, chromium, cobalt, copper, manganese, nickel, vanadium (0.9 to 1.6 m [3.1 to 5.3 ft] bgs)

Table 7-11. Summary of Ecological Risk Comparisons at 100-H Source OU for RI Data, CVP/RSVP Data, and LFI Data

Waste Site	RI Data	Shallow Zone Ecological Risks?	CVP/RSVP Data*	Shallow Zone Ecological Risks?	LFI Data	Shallow Zone Ecological Risks?
Limited Field Investigation - Monitoring Well Installation (Not associated with a Waste Site)						
--	--	--	--	--	199-H4-45	No samples collected from this depth range
--	--	--	--	--	199-H4-46	No samples collected from this depth range
--	--	--	--	--	199-H4-47	No samples collected from this depth range
--	--	--	--	--	199-H4-48	No samples collected from this depth range
--	--	--	--	--	199-H4-49	No samples collected from this depth range
--	--	--	--	--	199-H6-1	No samples collected from this depth range

* Complete reference citations are provided in Chapter 11.

- Two RI sample locations (116-H-2 Trench/Crib and 1607-H4 septic system) report soil concentrations greater than ecological SSLs. At the 116-H-2 Trench/Crib Test Pit, bis(2-ethylhexyl)phthalate was detected at a concentration of 0.76 mg/kg in the 3.4 to 4 m (11 to 13 ft) bgs depth interval at a concentration greater than the ecological SSL of 0.14 mg/kg. All bis(2-ethylhexyl)phthalate results were flagged with a "B" laboratory qualifier, indicating that the analyte was detected in both the sample and the associated QC blank, and the sample concentration is less than or equal to five times the blank concentration. At the 1607-H4 septic system test pit, bis(2-ethylhexyl)phthalate and eight polynuclear aromatic hydrocarbons were detected at concentrations greater than ecological SSLs in the 4 to 4.5 m (13 to 15 ft) bgs depth interval. Concentrations of PAHs and bis(2-ethylhexyl)phthalate ranged between slightly greater than the SSL to four times greater than the SSL. PAH and bis(2-ethylhexyl)phthalate concentrations were less than ecological SSLs in the 3.4 to 4 m (11 to 13 ft) bgs interval.

7.5 Assessment of Risks in Riparian, Near-Shore Media, and Columbia River

The RCBRA (DOE/RL-2007-21) evaluated soil, sediment, and water in riparian and near-shore areas. The remedial action goals used in the interim actions addressed risks to human health from direct contact with soil and threats to groundwater and surface water as a result of leaching from soil, but did not directly address risks to ecological receptors, except those protected through compliance with AWQC. The ERA conducted as part of the RCBRA (DOE/RL-2007-21) addresses residual contaminant concentrations at remediated waste sites in the upland zones and the transport of contaminants from waste sites to the Columbia River riparian and near-shore zones (Integrated Work Plan [DOE/RL-2008-46]). The CRC (DOE/RL-2010-117) evaluated island soil, sediment, water, and fish tissue in the Columbia River beyond the near-shore environment. Several investigations conducted on effluent pipelines that discharged to the Columbia River are also summarized in the following subsections.

7.5.1 Summary of Results and Conclusions of RCBRA

The RCBRA (DOE/RL-2007-21) evaluated ecological risks at 48 near-shore study sites potentially affected by contamination from Hanford Site sources in comparison to reference sites. Study sites were selected in areas where known contaminated groundwater plumes enter the Columbia River and in areas between the plumes. For the near-shore environment, 22 COPECs were identified and 16 of these (all inorganics) were further identified as COECs. The RCBRA (DOE/RL-2007-21) concluded that across the Hanford Reach of the Columbia River (that is, corridor-wide) five COECs (cadmium, chromium, Cr[VI], manganese, and uranium) in the near-shore environment may present an unacceptable level of risk for one or more of the assessment endpoint entities (aquatic plants, aquatic invertebrates, amphibians, fish, and wildlife). These results are based primarily upon the comparisons of COPEC concentrations to toxicity benchmarks, measures of exposure and effects in biota, or the results of wildlife exposure analyses (RCBRA [DOE/RL-2007-21]). The evaluation of these sediment COECs is summarized as follows:

- **Cadmium** was detected in 9 of 22 near-shore sediment samples (Appendix L, Tables L-68 through L-70). However, none of the samples exceeded the lower effects threshold (ecological screening level [ESL]); thus cadmium was not carried forward to the FS.

- **Total Chromium** was detected in 23 of 24 near-shore sediment samples. However, none of the samples exceeded the lower effects threshold (screening value from Ecology Publication 11-09-54); thus chromium was not carried forward to the FS.

- **Manganese** was detected in 22 of 22 near-shore sediment samples. However, none of the samples exceeded the lower effects threshold (screening level); thus manganese was not carried forward to the FS.

The evaluation of these pore water COECs is summarized as follows:

- **Cr(VI)** was detected in five of eight 100-D pore water samples (Appendix L, Table L-41) and one of two 100-H pore water samples (Appendix L, Table L-46). Within 100-D, near-shore filtered samples exceeded the ESL in pore water (2 of 8 samples), aquifer tubes (17 of 62 samples), seep (1 of 1 sample), and groundwater wells (84 of 103 samples). Within 100-H, all pore water concentrations were below the ESL in the 100-H Area, and seep data were not collected. However, near-shore filtered samples exceeded the ESL in aquifer tubes (41 of 105 samples) and groundwater wells (76 of 111 samples). Given the clear pathway from groundwater to the aquifer tubes and ultimately pore water, there is a clear pathway of Cr(VI) originating from the 100-HR-3 Groundwater OU wells within the vicinity of 100-D and 100-H that warrants further evaluation in the FS.

- **Manganese** was detected in seven of seven 100-D pore water samples and two of two 100-H pore water samples, but concentrations were less than the ESLs for all sediment and pore water samples collected in the 100-D and 100-H Areas. Filtered concentrations were also below the ESL, except for one aquifer tube sample. Thus, manganese in the 100-D and 100-H near-shore areas was not recommended for evaluation in the FS.

- **Uranium** was not detected in the nine samples collected from pore water near 100-D and 100-H Areas. Thus, uranium in the 100-D/H near-shore areas was not recommended for evaluation in the FS.

The RCBRA (DOE/RL-2007-21) identified 9 of the identified 22 COPECs (arsenic, chromium, copper, lead, mercury, selenium, TPH-diesel, vanadium, and zinc) as possibly presenting risk for 1 or more of the assessment endpoint entities (terrestrial plants, invertebrates, and wildlife). This is based on soil bioassays, comparison of COPEC concentrations to plant or terrestrial invertebrate benchmarks, or the results of wildlife exposure analyses. However, conclusions were that on a River Corridor-wide basis, only six of these COPECs should be considered further (arsenic, chromium, lead, mercury, TPH-diesel, and zinc). The evaluation of these COECs is summarized below.

As shown in Appendix L, Tables L-51 through L-61, concentrations of arsenic, chromium, lead, mercury, TPH-diesel, and zinc in the 100-D and 100-H riparian soil were all below the PRGs presented in Tables 7-4 and 7-5. Thus, none of these soil COECs was carried forward to the FS.

Final COECs identified within the RCBRA (DOE/RL-2007-21) are included in Table 7-12. These COECs were determined for the River Corridor as a whole. The potential or likelihood for the 100-D/H Source OUs to have contributed to the potential ecological risks identified for these COECs is discussed in Appendix L and summarized in the remainder of this section.

Table 7-12. Riparian, Near-shore and Riverine COECs from the RCBRA and CRC

COEC	Receptors	Media	Are 100-DR-1, 100-DR-2, 100-HR-1 and 100-HR-2 Potential Sources?	Is 100-HR-3 a Potential Source?
Aluminum[a]	Fish Aquatic Invertebrates Aquatic Plants	Pore Water	No	No
Arsenic[b]	Terrestrial Plants	Riparian Soil	No	No

Table 7-12. Riparian, Near-shore and Riverine COECs from the RCBRA and CRC

COEC	Receptors	Media	Are 100-DR-1, 100-DR-2, 100-HR-1 and 100-HR-2 Potential Sources?	Is 100-HR-3 a Potential Source?
Cadmium[b]	Aquatic plants and invertebrates	Sediment	No	No
Chromium[a]	Fish Aquatic invertebrates Aquatic plants	Pore water	No	Yes
Chromium[a,b]	Aquatic plants and invertebrates and the bufflehead	Sediment	No	No
Chromium[b]	Terrestrial plants and invertebrates	Riparian soil	No	No
Cr(VI)[a,b]	Aquatic plants and invertebrates	Sediment	No	No
Cr(VI)[a,b]	Fish Aquatic invertebrates Aquatic plants	Pore water	No	Yes
Lead[a]	Fish Aquatic invertebrates Aquatic plants	Pore water	No	No
Lead[b]	Terrestrial plants	Riparian soil	No	No
Manganese[b]	Aquatic plants and invertebrates	Sediment	No	No
Manganese[b]	Aquatic plants and invertebrates	Pore water	No	No
Mercury[b]	Terrestrial invertebrates	Riparian soil	No	No
Nickel[a]	Fish Aquatic invertebrates Aquatic plants	Pore water	No	No
Nitrate[a]	Fish Aquatic invertebrates Aquatic plants	Pore water	No	No
TPH–Diesel[b]	Terrestrial invertebrates	Riparian soil	No	No
Uranium[b]	Aquatic plants and invertebrates	Pore water	No	No

Table 7-12. Riparian, Near-shore and Riverine COECs from the RCBRA and CRC

COEC	Receptors	Media	Are 100-DR-1, 100-DR-2, 100-HR-1 and 100-HR-2 Potential Sources?	Is 100-HR-3 a Potential Source?
Zinc[b]	Terrestrial plants and invertebrates and kingbirds	Riparian soil	No	No

a. COECs presented in the executive summary of the CRC (*Columbia River Component Risk Assessment, Volume I: Screening-Level Ecological Risk Assessment* [DOE/RL-2010-117]).

b. COECs presented in Sections 8.4 and 8.5 of the RCBRA (*River Corridor Baseline Risk Assessment, Volume I: Ecological Risk Assessment* [DOE/RL-2007-21]).

7.5.2 Summary of Results and Conclusions of CRC

The CRC (DOE/RL-2010-117) included an ERA that combines both screening and baseline elements. Abiotic media were compared to screening benchmarks for surface water, sediment, and pore water to identify COPECs. Soil concentrations were compared to plant and invertebrate benchmarks, while desktop food web models were used to evaluate risks to wildlife. A baseline assessment was conducted to assess risk to fish using tissue residue data. The CRC (DOE/RL-2010-117) concluded that eight COECs were within sediment, pore water, island soil, and shoreline sediment (aluminum, chromium, Cr[VI], lead, manganese, mercury, selenium, and uranium). The evaluation included distinct conclusions for the reach adjacent to the 100 Area versus those for the reach adjacent to the 100-D/H Source OUs. Six COECs were identified for the 100-D/H Source OUs, as presented in Table 7-12. The potential or likelihood for the 100-D/H Source OUs to have contributed to the potential ecological risks identified for these COECs is discussed in Appendix L and summarized in the remainder of this section. The evaluation of these COECs is summarized as follows:

- **Aluminum** was detected in three of nine pore water samples in the 100-D/H near-shore areas. However, detections in all aqueous media were below ESLs. Therefore, aluminum is not considered a COEC and will not be carried forward to the risk characterization section or to the FS.

- **Cr(VI)** was detected in 5 of 10 pore water samples in the 100-D/H Area reach of the Columbia River, but was not collected from sediment. Within 100-D, near-shore filtered samples exceeded the ESL in pore water (2 of 8 samples), aquifer tubes (17 of 62 samples), seep (1 of 1 sample), and groundwater wells (84 of 103 samples). Within 100-H, all pore water concentrations were below the ESL and seep data were not collected. However, near-shore filtered samples exceeded the ESL in aquifer tubes (41 of 105 samples) and groundwater wells (76 of 111 samples). Given the clear pathway from groundwater to the aquifer tubes and ultimately pore water, there is a clear pathway of Cr(VI) originating from the 100-HR-3 Groundwater OU wells within the vicinity of 100-D and 100-H that warrants further evaluation in the FS.

- **Total Chromium** was detected in 23 of 24 near-shore sediment samples and 3 of 9 pore water samples in the 100-D/H near-shore areas. However, samples were less than the ESL for all sediment and pore water samples collected in the 100-D/H Areas and all aquifer tubes and seep samples collected in the 100-H Area. However, filtered total chromium samples were detected above the ESL in aquifer tubes (17 of 62 samples), seeps (1 of 8 samples), and groundwater wells (20 of 37 samples). While Cr(VI) concentrations are elevated in the same media, mean and maximum concentrations of

Cr(VI) are well below those of total chromium. Therefore, given the clear pathway from groundwater to the aquifer tubes and ultimately pore water, there is a potential pathway of total chromium originating from the 100-HR-3 Groundwater OU wells within the vicinity of 100-D/H that warrants further evaluation in the FS.

- **Lead** was not detected in pore water samples collected from the 100-D/H near-shore areas. Filtered concentrations within the 100-H aquifer tube, seep, and groundwater samples were below the ESL. Thus, the 100-HR-3 Groundwater OU wells within the vicinity of 100-D/H do not contribute to concentrations of lead observed in pore water at locations within the reach of the Columbia River.

- **Nickel** was detected in 2 of 9 pore water samples collected from the 100-D/H near-shore areas. The ESL was exceeded within 100-D in a limited number of aquifer tubes (8 of 62 filtered samples and 10 of 64 unfiltered samples) and groundwater wells (2 of 37 filtered samples and 3 of 36 unfiltered samples). Samples from all aqueous media were below the ESLs within the 100-H Area. Thus, the 100-HR-3 Groundwater OU wells within the vicinity of 100-D/H do not contribute to concentrations of nickel observed in pore water indicative of risk to aquatic plants in the Columbia River Reach adjacent to or downstream from the 100-D/H Areas.

- **Nitrate** was not collected from pore water in the 100-D/H near-shore areas. For the purposes of the ERA, nitrate was identified as a potential risk in the CRC (DOE/RL-2010-117) because of one anomalously high detection that exceeded the LOEC screening value identified at 37.64 mg/L. However, this value is actually an LC10 (effect noted in just 10 percent of organisms), which is an effect level more sensitive than commonly used. More importantly, as suggested by reference concentrations as high as 20.7 mg/L, nitrate is a common constituent in the Columbia River and its tributaries, a reflection of the agricultural land use prevalent in the area. Thus, nitrate is not of further concern with respect to ecological risks in the 100-D/H Areas.

7.5.3 100-D/H River Effluent Pipeline Investigations

During operations, water used in fuel production to cool the reactors was discharged to the Columbia River via effluent pipelines. The release of this cooling water ended when the reactors and facilities were shut down. Today, the three inactive 100-D/H effluent pipelines remain in their original locations in the Columbia River channel. Past characterization efforts obtained samples of the river effluent pipelines from the 100-BC, 100-D, and 100-F Areas. Characterization data collected during the river pipeline evaluations were used to evaluate potential risks from contaminants within the pipelines. The RCBRA (DOE/RL-2007-21) summarized the previous characterization efforts and risk assessment for these pipelines in Section 8.2.2.

In 1984, *River Discharge Lines Characterization Report* (UNI-3262) discussed samples of scale (flakes of mostly rust) from the interior surfaces and enclosed sediment of the effluent pipelines from the C, DR, and F Reactors. The pipelines were also visually inspected underwater by a diver, and their positions and physical conditions were assessed. Samples of scale and sediment were analyzed for radionuclides. The major radionuclides detected included cobalt-60, cesium-137, europium-152, europium-154, and europium-155. Radionuclide concentrations were greater in the scale than in the sediment. Direct beta-gamma radiation measurements were also obtained for interior and exterior pipe surfaces. The dose rates measured for direct contact with the interior of the pipe surfaces were less than 1 mrem/hour, and readings on the exterior were below the instrument's detection capability. Because the half-lives of all of these radionuclides is less than 30 years, the activity levels have declined by a factor of two to five and are no longer expected to be ecological risk drivers.

In 1994, a comprehensive geophysical survey (*Columbia River Effluent Pipeline Survey* [WHC-SD-EN-TI-278]) located and mapped the reactor effluent pipelines. The study relied mainly on remote sensing geophysical techniques, including navigation and echo sounding, side-scanning radar, sub-bottom profiling, seismic reflection profiling, and ground-penetrating radar. The results indicated that the pipelines have neither broken loose nor moved from their original locations. However, portions of some pipelines are no longer buried.

In 1995, pipe scale and sediment from the interior of the effluent pipelines from the 100-BC and 100-D Areas were sampled and physically characterized using a robotic transporter (*100 Area River Effluent Pipelines Characterization Report* [BHI-00538]). Analytical data from these two pipelines were intended to complement the 1984 radionuclide data (*River Discharge Lines Characterization Report* [UNI-3262]) and were expected to represent worst-case conditions with respect to radiological contamination. This assumption was based on the long years of pipeline service and the volume of effluent discharged from the B and D/DR Reactors.

The analytical results from the 1984 and 1995 effluent pipeline characterization studies at the B, C, D/DR, and F Reactors may reasonably be applied to effluent pipelines in 100-D/H, because operations among these reactors were similar. Evaluations of human health and ecological risk have been performed for the river effluent pipelines, as they are today located on or beneath the river channel bottom, and for a scenario in which a pipeline section breaks away from the main pipeline and is washed onto the shore of the river. Both the 1996 risk assessment effort (*100 Area River Effluent Pipelines Characterization Report* [BHI-00538]) and the 1998 risk assessment effort (*100 Area River Effluent Pipelines Risk Assessment* [BHI-01141]) relied on data collected from the 1984 and 1995 characterization work. The evaluation of human health and ecological risk performed in 1998 (*100 Area River Effluent Pipelines Risk Assessment* [BHI-01141]) concluded that the concentrations of chromium and mercury in the scale and sediment within the pipelines pose minimal ecological risk because they have been in contact with river water without dissolving since the reactors were shut down. The 1998 risk evaluation results indicated pipelines present no unacceptable risks and, therefore, no remediation requirements under CERCLA. This is supported by the following:

- Minimal deteriorated condition of the pipelines
- Continued decrease of radionuclide concentrations because of decay
- Inaccessible location
- Unavailability of significant contaminants to affect human health and the environment

Based on available information, no elevated risk levels are expected to be associated with these pipelines.

7.5.4 Summary of the Evaluation of Riparian Soil

The RCBRA (DOE/RL-2007-21) evaluated ecological risks at representative riparian study sites adjacent to, or where they may be directly affected by, known contaminated media (groundwater seeps, soil, or sediment). The RCBRA (DOE/RL-2007-21) concluded that six COECs identified for the riparian environment (arsenic, chromium, lead, mercury, zinc, and TPH-diesel) may present an unacceptable level of risk to one or more of the assessment endpoint entities based on soil bioassays, comparison of COPEC concentrations to plant or terrestrial invertebrate toxicity benchmarks, or the results of wildlife exposure analyses. The CRC (DOE/RL-2010-117) did not identify risks to terrestrial plants or invertebrates from exposure to island and riparian soil.

Most concentrations detected in riparian soil within the 100-D/H OU were below ESLs (in this case specifically the SSLs) described previously. Except for aluminum, bis(2-ethylhexyl)phthalate, thallium, and vanadium, all other detections were below PRGs (Appendix L, Tables L-51 through L-61). These

four chemicals are discussed below. Those chemicals below PRGs do not warrant further evaluation in the FS.

Unremediated waste sites in the riparian area were not evaluated in this analysis. Because those sites, listed in Table 8-4, have similar site histories to the sites currently evaluated, the predicted outcomes are anticipated to be similar as well. Some unremediated waste sites may have exceedances of PRGs, which would provide the basis for remedial action or further evaluation. Additional discussion is provided in Section 7.6.2.

7.5.4.1 Risks to Terrestrial Plants in the Riparian Area

Measurements of all chemicals within the riparian soil of the 100-D/H OUs were below plant ESLs (Tables L-51, L-52, L-55, and L-56) except thallium. Thallium was identified in the RCBRA (DOE/RL-2007-21) as being below background. Likewise, the CRC (DOE/RL-2010-117) did not identify risks to terrestrial plants from exposure to island and riparian soil. Therefore, no COPECs in 100-D/H riparian soil warrant further evaluation in the FS based on risks to terrestrial plants. This finding is also supported by the results of biological measures collected as part of the RCBRA (DOE/RL-2007-21) including plant bioassays on Sandberg's bluegrass (*Poa secunda*) and plant tissue testing. Though these lines of evidence carry less weight given their limited datasets and temporal variability (that is, they were conducted just once), the results support the same conclusion. There were no significant correlations with chemicals and bioassay measures, and there were no significant correlations between soil chemistry and plant tissue measurements.

7.5.4.2 Risks to Terrestrial Invertebrates in the Riparian Area

Concentrations of chromium, mercury, and zinc exceeded SSLs protective of terrestrial invertebrates in the 100-D riparian soil study area (2f, Rip 1, Rip 2, Rip 3, Rip 8, Rip 9, Rip 10); concentrations were higher than the terrestrial invertebrate LOEC (RCBRA [DOE/RL-2007-21], Table 5-70). However, no chemicals, including chromium, mercury, and zinc, had concentrations that exceeded the Hanford Site-specific PRGs protective of terrestrial invertebrates (Tables L-51, L-52, L-55, and L-56) except thallium. Thallium was identified in the RCBRA (DOE/RL-2007-21) as being below background. The CRC (DOE/RL-2010-117) did not identify risks to terrestrial invertebrates from exposure to island and riparian soil. Based on this analysis, no COPECs in riparian soil for terrestrial invertebrates warrant further evaluation in the FS based on risks to terrestrial invertebrates.

Terrestrial invertebrate tissue concentrations, which indicate contaminant uptake and bioavailability, were measured at riparian study sites and reference locations and some, but not all, chemicals were detected in terrestrial invertebrates. Statistical differences were found between terrestrial invertebrate tissue concentrations for certain chemicals between riparian study sites and reference sites. However, this line of evidence was ranked low because of the lack of detections in invertebrate tissue for certain chemicals and the possibility of bias because of sample collection methods. Statistical differences in tissue concentrations of mercury and zinc in terrestrial invertebrates were noted between River Corridor and reference study sites; this relationship is based on data across the entire River Corridor and should not be inferred as a relationship specific for the 100-D/H Areas. However, there is insufficient evidence of a correlation for chemicals between tissue concentrations in terrestrial invertebrates and concentrations in soil (RCBRA [DOE/RL-2007-21]).

7.5.4.3 *Risk to Wildlife* in the Riparian Area

Risk to wildlife in the riparian area was evaluated in the RCBRA (DOE/RL-2007-21) using both field measures and desktop food web modeling using models similar to those described in this ERA for SSLs. A separate desktop food web evaluation was included in this ERA using the SSLs and PRGs presented in

Tables 7-2 through 7-4. Results of these three analyses are described below. The results all suggest that there is no risk to wildlife in the riparian soil of the 100-D/H OUs.

For riparian soil, field ecological measures of the small mammal community were developed as qualitative information on the status of these populations. Estimated dietary contaminant exposures and chemical concentrations in bird or small mammal tissues were compared to ecological effects levels established for dietary ingestion or related to tissue residues. For selected chemicals (cadmium, chromium, lead, selenium, and PCBs), measured tissue concentrations in small mammals trapped in study sites were not greater than reference areas (RCBRA [DOE/RL-2007-21], Table 5-48), and were less than available tissue effect levels (RCBRA, page 5-91).

Dietary exposure to terrestrial birds and mammals estimated using wildlife exposure models and riparian soil concentrations across the River Corridor indicated potential exposure higher than LOAEL-based SSL values for copper, selenium, vanadium, and zinc (DOE/RL-2007-21, Section 8.4.1.3). Only zinc was identified as a final COEC for riparian soil exposure to birds and mammals. However, selenium and vanadium concentrations within the 100-D Area, 100-H Area, and horn area were within Hanford Site-wide background, and copper and zinc concentrations were below Hanford-specific ESLs (Appendix L, Tables L-57 and L-58) for wildlife and therefore do not warrant further evaluation in the FS.

Most concentrations detected in riparian soil within the 100-D/H Areas were below SSLs and PRGs. ESL results showed the following three chemicals within riparian soil had concentrations above wildlife ESLs within the 100-D/H OUs: aluminum, bis(2-ethylhexyl)phthalate, and vanadium (Appendix L, Section L4.5 and Tables L-53, L-54, L-57, and L-58). However, these analytes were not identified as COECs in the RCBRA (DOE/RL-2007-21) and do not warrant further evaluation in the FS. Aluminum was detected below background and is not bioavailable or considered toxic to wildlife at pH levels above 5.5 like those found in the 100-D/H riparian areas. Bis(2-ethylhexyl)phthalate concentrations exceeded SSLs in 2 of 21 samples. The SSLs were based on unbound no-effect levels in literature-based food chain models (that is, insufficient site-specific data were available to develop a PRG). Bis(2-ethylhexyl)phthalate, a common lab contaminant, was not identified as a final COEC in the RCBRA (DOE/RL-2007-21) or as a COEC in the CRC (DOE/RL-2010-117); thus, further evaluation is not warranted. The maximum detected concentrations of vanadium (60.1 and 55 mg/kg) for the 100-D/H OUs were less than the site background of 85 mg/kg. Additional discussion is provided in Appendix L, Section L4.5. No additional evaluation is warranted in the FS.

Within the RCBRA (DOE/RL-2007-21), information on dietary contaminant exposures was also compared to ecological effects levels for diet to assess risks to birds or mammals potentially exposed to contaminants in near-shore sediments, biota, and water. Only chromium was considered a final COEC. The single study site with which this risk was associated is not within the 100-D/H near-shore environment.

7.5.5 Summary of Evaluation of Near Shore and Columbia River

The results from the evaluation in Appendix L showed that a range of inorganic, organic, and radiological contaminants was detected in near-river groundwater samples collected from the 100-D/HR-3 OUs are not affecting the aquatic life exposed to pore water, surface water, or sediment in the Columbia River near the 100-DH OUs[12]. Numerous lines of evidence were considered as part of the evaluation. The evidence

[12] Both filtered and unfiltered water sample results were evaluated in the RCBRA Report (DOE/RL-2007-21). In some cases, the toxicity information or standards/criteria are based on dissolved metals concentrations (filtered samples). Therefore, exposure and the potential for risk from metals may be overestimated by using the unfiltered (or total metals) concentrations.

included, but was not limited to, the comparison of aquatic media (aquifer tube, pore water, spring/seep, and surface water) in the riparian and near-shore areas to ESLs, data quality, temporal significance, and correlations or the lack thereof with chemistry and observed responses in the bioassays and reference data. In general, data quality issues such as presence of contamination in blank samples, or elevated detection limits relative to the criteria in wells not nearest to the river, and the use of unfiltered data (potentially overestimating exposure) indicate data may overestimate risks initially identified through aquatic criteria comparisons.

Although the biological measures collected do not represent all seasonal conditions and river stage fluctuations, the results of pore water bioassays on aquatic invertebrates and amphibians also suggest little or no correlation between COPEC concentrations and observed responses in the bioassays, and the responses were not different from those of upstream references. Benthic invertebrate community structure data also suggest no differences between reference sites and locations adjacent to the Hanford Site. The results from this analysis confirm the results from the evaluation presented in Appendix L, that with the exception of total chromium and Cr(VI) in groundwater, no COECs affect aquatic life exposed to pore water or surface water in the Columbia River near 100-D/H.

7.5.5.1 Risk to Fish

No COECs in the RCBRA (DOE/RL-2007-21) or in the CRC (DOE/RL-2010-117) were identified for surface water exposures to fish.

Pore water concentrations at study sites were greater than the water standards or criteria for Cr(VI) (RCBRA [DOE/RL-2007-21], Section 8.5.1.4). The CRC (DOE/RL-2010-117) also indicated exceedances of water quality criteria (aluminum, chromium, Cr[VI], lead, nickel, and nitrate) in 100-D/H pore water samples. However, most other lines of evidence suggest that there is no unacceptable risk to fish in the Columbia River. And as described above in Section 7.5.1 and in Appendix L, Section L4, with the exception of total chromium and Cr(VI), these chemicals are not found in near-shore groundwater; therefore, there is no source for these COECs from the 100-HR-3 OU. In addition, these values are not necessarily indicative of risks to fish, because these screening values are based on water quality or plant or invertebrate risk.

In general, across the River Corridor, fish were smaller (in length and mass) at study sites relative to reference sites. However, many factors either confound or contribute to the size of fish captured, such as fishing pressure or ease of capture of the target size range. Correlation with capture size and chemical concentration or other factor (for example, habitat, nutrient availability) was not possible because it was not considered part of the original study design. There were no strong trends in fish histopathological observations between those collected at study sites and those from reference site locations. No tissue COPECs were correlated with histopathological endpoints associated with adverse effects at study sites. No exceedances of tissue effects levels for near-shore aquatic COPECs were measured in fish tissue. In addition, evidence of greater contaminant uptake in fish from study sites was not apparent for most COPECs and tissues.

For 100-D/H, total chromium and Cr(VI) in 100-HR-3 OU groundwater, which represents a potential source for pore water concentrations that exceed the fish surface water ESL, warrant further evaluation in the FS. Total chromium and Cr(VI) concentrations in multiple near-shore wells and aquifer tubes exceed ambient water quality criteria.

Other COPECs detected in pore water above ambient water criteria do not appear to be issues in groundwater or aquifer tubes, suggesting that the 100-HR-3 Groundwater OU is not the source of observed elevated concentrations. The exceedances for additional chemicals are discussed in more detail

in Appendix L. As explained in Appendix L, exceedances of ambient water quality criteria for other chemicals within aquatic media (pore water, seeps, aquifer tubes, groundwater, surface water) were either anomalous (that is, very low frequency) or because of laboratory reporting issues.

7.5.5.2 Risks to Aquatic Plants

Potential effects on aquatic plants were evaluated through results of a bioassay in sediment and comparison of sediment and pore water concentrations to SSLs (RCBRA [DOE/RL-2007-21], Tables 6-88 through 6-91). Based on the combined pore water and sediment concentrations, the RCBRA (DOE/RL-2007-21) identified cadmium, chromium, Cr(VI), manganese, and uranium as COECs warranting further evaluation for potential effects on aquatic plants, as noted in Section 8.5.1.1 (DOE/RL-2007-21). The CRC (DOE/RL-201-117) identified the final COECs for pore water and sediment within the 100-HR-3 OU as aluminum, chromium, Cr(VI), lead, nickel, and nitrate. For the 100-D and 100-H near-shore sampling sites, antimony and phosphorus were detected in sediment at concentrations greater than the upper threshold sediment biota ESL (Appendix L, Tables L-72 and L-74). Sediment COPECs/COECs are discussed in more detail below with risks to aquatic invertebrates and in more detail in Appendix L, with a conclusion that observed sediment concentrations do not warrant further evaluation. Pore water COPECs from the 100-D/H near-shore sampling sites are discussed in more detail in Appendix L, Section L4.2, which concluded that concentrations in the pore water, with the exception of Cr(VI), were not at levels warranting additional evaluation. Of the key plume contaminants in the reach of the Columbia River adjacent to 100-D/H OUs, Cr(VI) had concentrations of ecological relevance in the near-shore environment. Total chromium was above the ESL in near-shore groundwater wells, aquifer tubes, and seeps. Only total chromium and Cr(VI) represent a potential source for concentrations that exceeded water quality criteria at the point of exposure (pore water), warranting further evaluation in the FS.

Laboratory bioassays (that is, toxicity tests) were conducted with field-collected sediments. Significant relationships were determined with observed response within aquatic plant toxicity tests in association with confounding factors and some chemicals. Additionally, there were clear measures of exposure (that is, accumulation into plants), primarily for inorganic chemicals detected in pore water and sediment. However, of the significant relationships determined, none was with chemicals for which pore water concentrations were greater than aquatic plant benchmarks. Further, no risks to aquatic plants were noted based on toxicity testing.

7.5.5.3 Risks to Aquatic Invertebrates

The primary lines of evidence used to evaluate risks to aquatic invertebrates are field surveys, the results of bioassays, and comparison of sediment and water concentrations to ESLs

Abiotic Media Concentrations Compared to Literature Values. Pore water concentrations at study sites across the Hanford Reach were greater than chronic water standards or criteria for five COPECs (aluminum, cadmium, chromium, Cr(VI), and lead; RCBRA [DOE/RL-2007-21], Table 6-90). However, there are significant uncertainties relative to many of the conclusions based on pore water sampling. Further, all of these abiotic measurements represent a single point measurement within a dynamic river system with daily and seasonal fluctuations and flow volumes that can shift the composition of the substrates sampled. Exceedances should not be ignored as they can indicate exposure at levels presenting a risk. But because of the uncertainty in the representativeness of the measurements resulting from the dynamic environment, the exceedances should be considered along with other data that identify whether there is an ongoing source of the measurements. This analysis is presented in Appendix L. The interpretation of pore water results as an indication of adverse effects to aquatic invertebrates is the same as that for aquatic plants, given that the ESLs are protective of both plants and aquatic invertebrates:

total chromium and Cr(VI) in the 100-HR-3 Groundwater OU, which represents a potential source for pore water concentrations that exceed water quality criteria, warrant further evaluation in the FS.

For the River Corridor as a whole, sediment COECs (cadmium, chromium, and manganese) suggest a potential for adverse effects (RCBRA Report [DOE/RL-2007-21], Section 8.5.1.2). Likewise, total chromium and Cr(VI) in sediment were identified as COECs for the 100 Area in the CRC (DOE/RL-2010-117). For sediment samples collected within the 100-D and 100-H near-shore areas (Appendix L, Tables L-72 and L-74), concentrations were greater than upper threshold ESLs for antimony and phosphorus within the 100-D Area only.

Given the uncertainty with representativeness mentioned above, each of the COECs from the RCBRA (DOE/RL-2007-21), CRC (DOE/RL-2010-117), and 100-D/H near-shore sediment is discussed in detail in Appendix L. Concentrations of most Hanford-Reach sediment COECs are either below ESLs (cleanup standard from Ecology Publication 11-09-54) or below reference in the 100-D/H near-shore environment (explanations for the exceptions are described in Appendix L). This suggests that sediments upstream from the Hanford Site potentially contribute to concentrations observed in the 100-D/H near-shore sediments. Further, riparian soil for most of the COECs is lower than upstream sediment and Hanford Site reference soil concentrations, suggesting that the riparian soil in the 100-D/H Area is not a source of the observed sediment concentrations for the RCBRA (DOE/RL-2007-21) COECs identified. Biological measures such as amphipod bioassays, clam tubes, and community surveys from rock baskets show no clear indication of toxicity or correlation of response with COEC concentrations. Although they represent only a snapshot in time and do not represent all seasonal conditions and river stage fluctuations, these measures support the analysis that Hanford Site operations in 100-D/H do not adversely affect aquatic receptors exposed to sediment in the 100-D/H near-shore environment. Based on these findings, only total chromium and Cr(VI) in groundwater warrant further evaluation in the FS.

Direct Toxicity Measures. Risks to aquatic macroinvertebrates based on toxicity testing showed relationships with confounding factors and some chemicals. Histopathological measures of Asiatic clams (*Corbicula fluminea*) differed in study sites compared to reference sites; these measures also showed some negative relationships with chemicals. However, sediment bioassays at site Cr7/CR8 and 2f selected to represent 100-D/H showed no difference in amphipod (*Hyalella azteca*) growth or survival relative to reference sites. Likewise, survival and reproduction tests on water fleas in pore water showed no difference at sites representing 100-D/H relative to reference sites. Correlation between abiotic media chemistry and observed differences in measured effects from both bioassays was conducted across the Hanford Reach. Mercury was the only COPEC with a significant correlation that showed a potential negative effect with a significant regression; however, mercury was below sediment ESLs at the 100-D/H study sites. Clams were also monitored for survival. There was a statistical decrease in survival at study sites compared to reference sites, but there was no correlation of clam survival with COPECs. Together, these measures do not indicate substrate concentrations were toxic. However, they do not represent all seasonal conditions and river stage fluctuations.

Community Structure Measures. Key community metrics do not suggest that contaminant-related effects to benthic macroinvertebrates are evident in aquatic study sites as a group, as indicated by the comparison of Ephemeroptera, Plecotera, and Trichoptera data from study sites relative to reference sites. Most of the aquatic community measures did not differ between the study sites and reference sites. There were exceptions among the large number of aquatic community measures evaluated, but the agreement among measures was weak, and the biological significance to populations is not evident.

Measures of Exposure. Within the RCBRA (DOE/RL-2007-21), clear measures of exposure (accumulation), primarily for inorganic COPECs, were detected in water, sediment, and tissues. There

were no statistically significant correlations between COPEC concentrations in pore water or sediment with tissues of aquatic organisms, indicating a lack of significant COPEC bioaccumulation. Further, no tissue effect levels for COPECs in invertebrate tissue were exceeded.

Most histopathological measures of clams and mussels showed no significant differences between study and reference. While, there were exceptions, COPEC concentrations generally did not correlate with differences in histopathological measures.

Weight of Evidence. As stated previously, abiotic and biotic measures collected for the RCBRA (DOE/RL-2007-21) do not represent all seasonal conditions and river stage fluctuations. Abiotic measurements exceed literature-based screening values for some COPECs, and this line of evidence is generally given the lowest weight given the lack of site-specificity in the literature-based values. Although biological measures give a different perspective than the chemistry, given the limited dataset and the uncertainty with full representation of seasonal measurements, the results of the chemistry cannot be ignored.

Of the key groundwater plume contaminants investigated, total chromium and Cr(VI) had concentrations of ecological relevance in the near-shore environment for the 100-D Area, 100-H Area, and horn area. Total chromium and Cr(VI) in groundwater in the 100-HR-3 Groundwater OU, which represents a potential ongoing source for pore water concentrations that exceed water quality criteria, warrant further evaluation in the FS. This conclusion is applicable to both aquatic invertebrates and amphibians.

7.5.5.3 Risk to Near-Shore Wildlife

The RCBRA (DOE/RL-2007-21) evaluated risk to middle trophic-level wildlife including the kingbird, mink, and bufflehead. Risks to wildlife in the near-shore environment are primarily from ingestion of prey consisting of aquatic invertebrates, clams, and fish and from incidental ingestion of sediment. Only chromium risk to the bufflehead represented a risk warranting further evaluation, and the chromium was elevated at just one study site not within the 100-D/H near-shore environment. However, because of the limited time at the site (winter only) and the unlikeliness of a population of bufflehead ducks feeding over this single location long enough to cause chronic exposure, total chromium does not warrant additional consideration in the FS for exposures to near-shore middle-trophic level wildlife.

7.5.5.4 Transport Pathways for Cr(VI) from Groundwater to Surface water

At 100-D/H, groundwater flows toward the Columbia River. During major spring discharge events, river water may enter the banks and the adjacent groundwater system upstream from the Site and move laterally parallel to the river for some distance before discharging back into the river (*Technical Evaluation of the Interaction of Groundwater with the Columbia River at the Department of Energy Hanford Site, 100-D Area* [SGW-39305]). A daily 3 m change in river levels superimposed with seasonal changes or alterations of site groundwater flows by remediation efforts likely causes seasonal shifts in the regional groundwater flow system that will affect groundwater/surface water exchange through the hyporheic zone. In addition to the discharge of groundwater to the river through the hyporheic zone, groundwater seasonally discharges in seeps or springs above river stage, principally following seasonal high river stage in early summer. During operations, large volumes of reactor cooling water were discharged to the Columbia River. Under current conditions, the high-volume liquid effluent releases ended when reactor operations ceased in 1971.

Receptors in the riverbed and benthic and hyporheic zones can be exposed to contaminated (1) groundwater, (2) groundwater/surface-water mixtures, or (3) surface water. The unconfined aquifer beneath the 100-HR-3 OU discharges to the Columbia River via upwelling through the riverbed and, to a lesser extent, via riverbank springs that appear during low river stage. Sampling locations (for example,

near-river wells, riverbank springs, aquifer tubes, and near-shore river water) used for water quality monitoring near the Columbia River are discussed in the Riparian and Near-Shore CSM presented in Appendix L. As is discussed in Section 4, springs along the 100-D and 100-H Source OU shoreline have been monitored for many years as part of the Surface Environmental Surveillance Program (SESP) (2009 Sitewide Environmental Report [PNNL-19455]). Samples of spring water and associated fine-grained sediment collected during late summer/early fall have been analyzed for Cr(VI) and other waste effluent indicators. Annual sampling is conducted when Columbia River flow is at its seasonal low, resulting in the maximum flow of groundwater from the unconfined aquifer to the river. In addition, data were collected near 100-D and 100-H Source OUs during the CRC (DOE/RL-2010-117) to address the uncertainty related to the level of contamination entering the Columbia River via upwelling, including the contaminant transport mechanisms. Pore water, surface water, and sediment sampling in the Columbia River was conducted in 2009 and 2010, as outlined in the Columbia River RI Work Plan (DOE/RL-2008-11).

Based on available information, there is a pathway for migration of Cr(VI) in 100-HR-3 OU near-river groundwater to shoreline pore water. In addition, there is evidence (based on conductivity measurements) of pore water entry into Columbia River surface water. However, surface water samples collected at mid-channel depth within the Columbia River in the vicinity of 100-D and 100-H have not measured detectable levels of Cr(VI). The flux of Cr(VI) in groundwater is too small to produce significant Cr(VI) effects related to Hanford Site operation in Columbia River surface water. This is supported by a lack of detections of Cr(VI) in surface water and a conclusion that accumulation of Cr(VI) in fish tissue such as sculpin does not pose a significant risk (see Chapter 6).[13]

7.5.6 Conclusions

Table 7-12 presents the 13 COECs identified in the riparian and near-shore media from the RCBRA (DOE/RL-2007-21) and the CRC (DOE/RL-2010-117). For each COEC, RCBRA (DOE/RL-2007-21) and CRC (DOE/RL-2010-117) abiotic media data (soil, sediment, groundwater, pore water, aquifer tubes, seeps, and surface water) from reference areas, upstream sources, and onsite riparian and near-shore areas are discussed in Appendix L to determine the likelihood that the 100-D/H OUs were sources. The conclusion of Appendix L is that of the COECs in Table 7-12, only total chromium and Cr(VI) are related to the 100-D/H OUs in groundwater.

7.5.7 Risk Conclusions and Scientific Management Decision Point

COPCs were identified in ninety-five 100-D OU waste sites, which were reclassified as "interim closed," "no action," or to be determined through the TPA (Ecology et al., 1989a) process. The COPCs were identified in forty-seven 100-H OU waste sites reclassified as "interim closed" or "no action" through the TPA (Ecology et al., 1989a) process. EPCs of COPCs for each decision unit (for example, overburden, shallow-focused, shallow, staging pile footprint) at each waste site were compared to the plant/invertebrate SSL, the wildlife SSL, background, and plant/invertebrate PRG and wildlife PRG values. Within the 100-D OU, 12 waste sites were retained for additional consideration based on EPC exceedances of six COPECs (copper, lithium, mercury, selenium, silver, and vanadium). Within the 100-H OU, four waste sites were retained for additional consideration based on EPC exceedances of four COPECs (barium, boron, chromium, and mercury).

At the SMDP, the results of the ERA were considered in the context of other factors (for example, spatial coverage, data, chemical specifics, receptors at risk, and confidence in PRGs) to support recommendations on the COECs to be brought forward to the risk managers and considered for the FS.

[13] The noncancer HI above 1.0 for the Tribal scenario was driven by nickel, not Cr(VI).

This included agreement on the assessment endpoints, representative receptors, and complete exposure pathways that correspond to those COECs. The final recommendation for the SMDP is a conclusion that there were no potential risks to ecological receptors in the upland remediated waste sites and source OUs warranting further evaluation in the FS. As part of the assessment of contributions to ecological risks identified in the riparian and near-shore environments of the Columbia River (RCBRA [DOE/RL-2007-21]) and the main channel, far-shore, and island environment of the Columbia River in the CRC (DOE/RL-2010-117), total chromium and Cr(VI) in the 100-HR-3 Groundwater OU are recommended for further evaluation in the FS.

7.6 SMDP Considerations

Within the process for conducting ecological risk evaluations or assessments at CERCLA sites, several decision points occur at which risk managers, risk assessors, and other stakeholders agree on a path forward with respect to ecological risk associated with a site. Typical variations include the following risk assessment outcomes:

- No unacceptable potential risks to ecological receptors (for example, risks are sufficiently low and below risk-based thresholds such as SSLs or PRGs).
- Potential for risks to ecological receptors, but the risks do not warrant the evaluation of remedial alternatives in the FS because of a number of considerations.[14]
- Potential for risks to ecological receptors, but there is uncertainty in one or more components of the ERA that warrant the evaluation of remedial alternatives in the FS.
- Need to evaluate remedial alternatives in the FS based on the protection of another receptor or exposure pathway (for example, human health) that would address potential ecological risks.
- Potential for risk to ecological receptors warranting evaluation of remedial alternatives in the FS.

With the risk assessment outcomes listed above, agreement is needed on the following elements to assist in the evaluation of remedial alternatives in the FS: the COCs, the assessment endpoints, the exposure pathways, and the risk questions. To confidently achieve one of the risk assessment outcomes, a number of factors and supporting information were considered in the conclusion of the risk assessment to assist risk management decisions. These outcomes were considered within the context of other exposure pathways and receptors evaluated at the same site. Factors that were considered to interpret the results of the risk characterization and determine if the site requires evaluation of remedial alternatives in the FS include the following:

- Spatial characteristics of the remediated waste site (area and excavation depth of the remediated waste site)
- Proximity and size of nearby unremediated waste sites and unaffected habitat
- Number and location of samples collected at the site
- Data quality (presence of qualifiers, adequacy of detection limits)
- Frequency that risk-based thresholds are exceeded and the location(s) of those exceedances

[14] For example, a wildlife risk for a specific contaminant was driven by an estimated exposure to a badger, but the size of the site is 20 m^2 representing a minimal portion of the total required foraging area for a badger, and the site does not represent a preferential feeding area.

- Chemical-specific properties of each COC (for example, does it have the potential to biomagnify in the food web, or is it persistent in the environment?)
- Identification of specific receptors that have the potential for adverse health effects (feeding guild [plants, insects, or omnivorous, herbivorous, insectivorous, or carnivorous wildlife], proportion of receptors affected, likelihood of population- or community-level effects, home range of the receptors at risk relative to the area exceeding risk-based thresholds)
- Recalculation of the EPC based on the home range of the receptor or to estimate the residual risk after the removal action has been implemented
- Evaluation of PRG (that is, level of confidence, basis, relation to other PRGs such as those for human health or groundwater protection)

As shown in Appendix H (Table H-20), 19 waste sites within 100-D OU and 8 waste sites within the 100-H OU were reported with concentrations of COPECs greater than their respective PRGs. Figures showing the location and concentration of COPECs reported with an HQ greater than 1.0 are provided in Appendix H. During development of the evaluation, the factors above were evaluated and resulted in a recommendation, as part of the SMDP, that no waste sites be carried forward into the FS for evaluation of remedial alternatives. The decisions for 100-D/H OUs were based on a subset of the factors described above, including the following:

- Depth of samples[15] exceeding thresholds relative to the 4.6 m (15 ft) bgs standard point of compliance for ecological receptors defined by MTCA (WAC 173-340)
- Number and frequency of exceedances of the risk thresholds (PRGs)
- Magnitude of exceedance relative to the risk thresholds (the HQ)
- Confidence in the ecological risk thresholds defining the exceedances
- Quality of the sample data defining the exceedances
- Location of the samples exceeding thresholds, sample frequency, and proximity of other exceedances
- Area of exceedance relative to home range of receptor exceeding and relative to area of unaffected nearby habitat

Within these 27 waste sites, eleven inorganic metals were measured at concentrations above the PRGs identified in this chapter. After considering the factors listed above, the recommendation was not to require further evaluation in the FS or any remedial action. A summary of the rationale by chemical and receptor is provided below with the details for each specific waste site-decision unit-chemical combination being found in Appendix H, Table H-20.

Plants: Mercury (14 waste site decision units), vanadium (5 waste site decision units), molybdenum (2 waste site decision units), and copper (2 waste site decision units) were all measured at concentrations above plant PRGs. Molybdenum is not expected to adversely affect the plant communities as it is not documented as phytotoxic in the published literature. Samples for copper above the copper PRG

[15] For the purposes of the ecological risk assessment, it was assumed that soil up to 4.6 m (15 ft) bgs is accessible to ecological receptors because this soil can be brought to the surface by human activities, thereby becoming biologically accessible. In some cases, the database indicated soil was collected from a shallow depth, but further review conducted for the SMDP showed that soil was collected below 4.6 m (15 ft).

(58 mg/kg) were collected at 4.8 to 7 m (16 to 23 ft) which is below the standard point of compliance of 4.6 m (15 ft) and the maximum depth at which plant roots have been observed at the Hanford Site (3 m [9.8 ft]; *Rooting Depth and Distributions of Deep-Rooted Plants in the 200 Area Control Zone of the Hanford Site* [PNL-5247]). Most vanadium samples were just above background and also collected below where plant roots have been observed at the Hanford Site. Risk to plants from mercury are unlikely because of low confidence in the PRG and no exceedance of wildlife PRGs for a bioaccumulative compound. These were infrequent and in most cases spatially distinct exceedances that would not cause a community level effect. If localized adverse effects did occur, habitat fragmentation in the 100-D OU would not be likely given the level of ecological services the habitat is providing in the current condition and the available habitat refugia nearby (see Section 7.6.3)

Invertebrates: Barium and silver were measured at concentrations above terrestrial invertebrate PRGs at three and one waste site-decision units respectively. These were infrequent and in most cases spatially distinct exceedances that would not cause a community level effect. Considering these infrequent exceedances, if deep excavation were to occur, the elevated concentrations would be mixed with much lower concentration material resulting in a lower exposure concentration. At three of the waste site decision units, samples were from a depth below the maximum at which invertebrates have previously been observed at the Hanford Site (2.7 m [8.9 ft]; *Characterization of the Hanford 300 Area Burial Grounds: Task IV – Biological Transport* [PNL-2774]). Risk to the terrestrial invertebrate community are not expected at these waste site decision units and there is ample unimpacted habitat for available in adjacent areas and along the River Corridor.

Wildlife: Selenium and lead were measured at concentrations above wildlife PRGs at five and four waste site-decision units respectively. However, selenium measurements were sometimes deep (i.e., below the maximum depth at which Hanford Site wildlife have been observed to burrow [1 m {3.3 ft} pocket mouse] "Loose Rock As Biobarriers in Shallow Land Burial" [Cline et al., 1980]) and the size of the waste sites is small. When the size of the sites was considered relative to the home range of wildlife receptors (i.e., application of an AUF), HQs were below 1.0. The population density of small mammals and the number of individuals expected to reside within these small sites was also considered. The final conclusion was that there are no population level effects to avian and mammalian receptors at any of the remediated waste sites that were evaluated including those with some measured samples of selenium and lead above PRGs.

SMDP Conclusion: As indicated in Appendix H, Table H-20, consideration of factors listed above resulted in the conclusion of no unacceptable risks to terrestrial wildlife or plants and invertebrates exposed to vadose zone soil and a recommendation of no further action for the waste sites within the 100-DR-1, 100-DR-2, 100-HR-1, or 100-HR-2 Source OUs. For unremediated waste sites, remedial actions will consider the PRGs through the SMDP process. More detail in applying that process to unremediated sites is described in Sections 7.6.2 and 7.6.3.

7.6.1 Recommendations for Evaluating Wildlife in Future Assessments at Unremediated Waste Sites

Data and process knowledge indicate ecological PRGs will be exceeded at unremediated waste sites. Those exceedances will be evaluated through the ERA process, including consideration of such factors as waste site size and wildlife home ranges within a scientific management decision point, to determine a basis for action. PRGs will be presented in the proposed plans for protection of wildlife receptors. The PRGs will achieve protection of the populations of wildlife species constituting the food web at the Hanford Site (Figure 7-1), including a range of feeding guilds. The receptor species selected for quantitative development of PRGs are intended to represent the species within those feeding guilds.

As discussed in the technical support documents for ecological values in soil for wildlife (*Tier 1 Risk-Based Soil Concentrations Protective of Ecological Receptors at the Hanford Site* [CHPRC-0784]; *Tier 2 Risk-Based Soil Concentrations Protective of Ecological Receptors at the Hanford Site* [CHPRC-01311]), the values used to calculate PRGs are based on the assumption that the size of the waste site inhabited by a receptor is the same size as the area used by the animal, for example, its home range, breeding range, or feeding/foraging range. In other words, the PRGs assume that a wildlife receptor is exposed 100 percent of the time to the contaminants in a waste site. This ratio of the area of contamination to the home range is known as an AUF. An AUF = 1 is another way of stating the assumption that the contaminated area and home range are identical. An AUF of 1.0 means that an animal is exposed to site contaminants 100 percent of the time; depending on the home range of the animal in relation to the size of the waste site, assuming that the AUF is 1 in development of SSLs or PRGs may considerably overstate ecological risks. However, several wildlife receptors, particularly the carnivorous mammals and most birds, have home ranges much larger than most of the waste sites; applying PRGs for those receptors to most waste sites would overstate ecological risks.

The home ranges for the wildlife receptors used for PRG development are shown in Appendix H, Table H-6. In considering the home range data available for each species, it must be recognized that these ranges are reduced during breeding season. On the other hand, food sources in a semiarid environment such as the Hanford Site may be scarcer than what is reflected in the studies available, some of which were not conducted in similar habitats. While many biological studies have been conducted at the Hanford Site, studies specifically on home range or population density are not available for all species or guilds being evaluated.

Completion of remedial actions as part of the cleanup verification process based on ecological PRGs will incorporate a SMDP on a case-by-case basis to determine that the action is protective of ecological receptors. The SMDP approach and its use in remediation decision making will be presented in detail in the RDR/RAWP. Further, in cases where verification samples exceed the PRGs and these PRGs represent the limiting value (that is, the wildlife PRGs are lower than all other applicable PRGs), a risk management decision should be made similar to the SMDP described in Section 7.6.1. Particular attention should be given to the number of samples exceeding the PRGs, the spatial area represented by the samples, and the depth at which samples exceed the PRGs. Other key factors considered in the SMDP process include the following:

- Size of the waste site relative to home range of wildlife receptors (for example, developing and applying an AUF in the comparison of an EPC to the PRGs)
- Estimation of exposure using a central tendency estimate such as the 95 percent UCL
- Size of the waste site relative to area of adjacent uncontaminated habitat
- Nature and extent of residual contamination following remediation
- Potential presence of exposure pathways following remediation
- The number and frequency of exceedances of the risk thresholds (PRGs)
- The location of the samples exceeding thresholds, sample frequency, and proximity of other exceedances

PRGs are typically based on a concentration that may elicit adverse effects (that is, reduce survival, growth, or reproduction), as observed in low number of individuals exposed to chemicals in laboratory toxicity tests. For some chemicals, this is based on toxicity tests reporting a 20 percent effect level

(for example, mortality observed in 20 percent or more tested organisms or growth reduced by 20 percent). For other chemicals, this is the lowest concentration tested with undefined adverse effects. In considering the results of verification data for future remedial actions relative to the PRGs, consideration must be given to the origins of the toxicity data upon which the exceeded PRGs are based. This should be considered in the context of the risk management goal (protection of populations of wildlife), the selected assessment endpoint (reproduction, survival, and growth), and specific life history data for the selected wildlife receptors selected to represent the end points (for example, home range, population density).

7.6.2 Recommendations for Evaluating Plants and Invertebrates in Future Assessments at Unremediated Waste Sites

PRGs for terrestrial plants and invertebrates have been established for the Hanford Site (*Tier 2 Terrestrial Plant and Invertebrate Preliminary Remediation Goals (PRGs) for Nonradionuclides for Use at the Hanford Site* [ECF-HANFORD-11-0158]) and have been useful in screening waste sites for potential adverse effects to these communities. However, the use of these PRGs in selecting final remediation goals in the FS or the proposed plan should be considered on a site-specific basis except for waste sites where listed protected species have been identified (that is, federal or state listed and protected threatened or endangered species). This recommendation is based upon the following lines of evidence: no significant adverse toxicological effects observed at the highest available concentrations tested in site-specific bioassays; historical and ongoing biological surveys demonstrating no significant differences from control areas; and the limited likelihood of habitat fragmentation because of areas with elevated contaminants in soil. The plant and invertebrate PRGs can help identify where remedial actions have been effective. However, in cases where verification samples exceed these PRGs and these PRGs represent the limiting value (that is, the plant or invertebrate PRG is lower than all other applicable PRGs), a risk management decision should be made like the SMDP described in Section 7.6.1. Particular attention should be given to the number of samples exceeding the PRGs, the spatial area represented by the samples, and the depth at which samples exceed the PRGs.

Plant and invertebrate bioassays have been conducted at the Hanford Site on both plant and invertebrate species by DOE (RCBRA [DOE/RL-2007-21]; *Central Plateau Ecological Risk Assessment Data Package Report* [DOE/RL-2007-50]; *Tier 2 Terrestrial Plant and Invertebrate Preliminary Remediation Goals (PRGs) for Nonradionuclides for Use at the Hanford Site* [ECF-HANFORD-11-0158]) and by Ecology (*Ecological Soil Screening Levels for Arsenic and Lead in the Tacoma Smelter Plume Footprint and Hanford Site Old Orchards Ecology* [Ecology Publication 11-03-006]). Results of these studies have not shown significant adverse effects that can be clearly attributed to soil chemistry that have resulted as part of past operations or practices at Hanford. Scatter plots of the effects versus chemical concentrations show no clear patterns, and statistical tests have shown no correlation between effects and soil chemistry. As a result, the highest concentrations established have served as NOECs with no upper bounds, which have been established as PRGs. Sensitive species may demonstrate adverse effects at concentrations exceeding these NOECs. However, the risk management goal from *DQO Summary Report for the 100 Area and 300 Area Component of the RCBRA* (BHI-01757) was the maintenance of diversity and abundance of flora and fauna at the community or population level. As noted in Appendix A to *Generic Ecological Assessment Endpoints (GEAEs) for Ecological Risk Assessment* (EPA/630/P-02/004F), EPA's principles for ecological risk assessment and risk management at Superfund sites state, "Superfund's goal is to reduce ecological risks to levels that will result in the recovery and maintenance of healthy local populations and communities of biota." Comparing waste site chemical concentrations to LOECs could help identify potential community-level risks to plants and invertebrates and would adequately achieve the risk management goal. However, establishing a concentration gradient with site-specific weathered soil (as opposed to spiked laboratory tests with more highly bioavailable forms of chemicals) capable of

producing a LOEC has proven to be problematic. The concentrations have not been at levels high enough to demonstrate significant toxicity to native species (most of the plant tests have all been on native blue grass [*Poa secunda*], nematodes [*Caenorhabditis elegans*], and springtails [*Folsomia candida*]). Moreover, the chemicals present in the soil (mostly inorganic constituents and metals) are not known to be significant bioaccumulators. This points to the fact that existing concentrations at the Hanford Site may not be toxic to plants and invertebrates.

Numerous studies measuring the diversity and abundance and many other parameters have been part of biological surveys conducted at the Hanford Site. Among these are the SESP that has been conducted by PNNL for more than 20 years. The RCBRA (DOE/RL-2007-21) also included biological surveys for cryptogam, plants, invertebrates, and small mammals. These studies have included observations at both contaminated and uncontaminated sites across the Hanford Site. Overall, these studies document a complex and thriving ecosystem and show no clear distinction in measures at waste sites versus those at control sites. However only a portion of the areas studied include previously contaminated or remediated areas. Thus, there is no certainty that the same conclusion could be drawn from the remaining waste sites that have not yet been addressed.

At some sites, if significant effects to the plant community occur, a negative effect could be habitat fragmentation from reduced function of the plants or complete loss of the community. Habitat fragmentation is the discontinuity in spatial distribution of resources and conditions that affect occupancy, reproduction, or survival in a particular species ("What is Habitat Fragmentation?" [Franklin et al., 2002]). However, this is not likely at the Hanford Site if waste sites are left unremediated. In their current conditions, waste sites have a range of no to partial plant cover that supports a community of invertebrates such as ants and beetles, small burrowing mammals, birds, and carnivorous wildlife. The soil contains a seed bank from plants at the site and the surrounding plants outside the waste site. The surrounding shrub-steppe and grassland habitats would act as habitat refugia that ultimately would buffer the waste sites from extreme variation in the overall environmental condition and continue to support the ecosystem.

7.6.3 Evaluations of Sediment in Future Assessments and at Unremediated Waste Sites Below the Ordinary High Water Mark

Waste sites extending below the ordinary high water mark of the Columbia River should be assessed as an aquatic environment and, as such, should be evaluated for the protection of aquatic organisms described in the conceptual model in Appendix L. The evaluation of surface sediment data for future assessments will be against the freshwater sediment ESLs presented in Appendix L, Table L-4. These values are from a number of sources and are intended for screening measured concentrations for potential adverse effects to aquatic organisms exposed to sediments. However, not all of the ESLs presented are designed to be used as cleanup levels for evaluating remedial actions. The primary source of freshwater sediment PRGs are the cleanup screening levels published in *Development of Benthic SQVs for Freshwater Sediments in Washington, Oregon, and Idaho* (Ecology Publication 11-09-054). These values were specifically selected as thresholds for freshwater sediments in Washington, Oregon, and Idaho through the evaluation of field-collected toxicological data. The CRC (DOE/RL-2010-117) presented sediment LOECs for nine chemicals (acetone, alpha-BHC, chromium, dichlorodiphenyldichloroethane, heptachlor epoxide, phosphorous, silver, toluene, and TPH-diesel), but values from *Development of Benthic SQVs for Freshwater Sediments in Washington, Oregon, and Idaho* (Ecology Publication 11-09-054) were only available for four of these chemicals. Values for other chemicals rely on other sources and methods. These LOECs could be used as PRGs, such as the heptachlor epoxide value from "Development and Evaluation of Consensus-Based Sediment Quality Guidelines for Freshwater Ecosystems" (MacDonald et al., 2000), but others such as those derived through equilibrium

partitioning might require additional consideration. Recommended freshwater sediment PRGs are presented in Table 7-13. As with soil investigations described above, future assessments should include SMDP considerations (Section 7.6.2).

Table 7-13. Freshwater Sediment PRGs

Chemical	PRG
Antimony	12
Arsenic	120
Cadmium	5.4
Chromium	88
Copper	1,200
Lead	>1,300
Mercury	0.8
Nickel	110
Selenium	>20
Silver	1.7
Zinc	>4,200
TPH-Diesel	510

Note: > "Greater than" value indicates that the toxic level is unknown, but above the concentration shown.

8 Identification and Screening of Technologies

This chapter begins the feasibility study section of the RI/FS. The RI defines the nature and extent of contamination at the site and the potential risks to human health and the environment posed by site contaminants.

The FS identifies and evaluates alternative strategies to address the risks. The FS consists of three phases: screening of remedial technologies, development of remedial alternatives, and detailed analysis of selected alternatives. Remedial technologies are assembled into alternatives that address contamination on a media- or source-specific basis. Technologies are evaluated in this chapter to determine their effectiveness in removing the contaminants (described in Chapter 4) or interrupting the exposure pathway (described in Chapters 6 and 7).

Chapter 8 presents the following discussions:

- RAOs, ARARs, PRGs (Section 8.1)
- General response actions (GRAs) (Section 8.2)
- Identification and screening of remedial technologies and associated process options to clean up the contamination (Section 8.3)

Chapter 9 assembles the alternatives and Chapter 10 provides a detailed analysis of the alternatives to address contaminated media at 100-D/H.

Highlights

- Media specific RAOs are identified for groundwater, surface water, and soil.
- To meet RAOs, PRGs are established for each environmental medium of interest, contaminants, receptors, and exposure pathways.
- Of 343 sites in the 100-D and 100-H Areas, 291 are recommended for evaluation in the FS.
- A range of GRAs to meet RAOs is identified for waste sites and contaminated groundwater.
- Process options retained for vadose zone actions include no action, standard and deep excavation, disposal, in situ treatments: biological reduction, solidification, soil flushing, stabilization/ sequestration, and void-fill grouting; surface barriers, and institutional controls.
- Process options retained for groundwater include no action, MNA, pump-and-treat, in situ treatments: chemical stabilization, biological (anaerobic), combined biological and chemical, reactive chemical barrier, and flushing; ion exchange, and institutional controls.
- Process options and technologies for the range of GRAs are evaluated for relative effectiveness, implementability, and cost.

8.1 Remedial Action Objectives

RAOs are general descriptions of what the remedial action is expected to accomplish (that is, medium-specific or site-specific goals for protecting human health and the environment). They are defined as specifically as possible to address the following concerns:

- Media of interest (soil or groundwater)
- Types of contaminants (radionuclides and chemical constituents)
- Potential receptors (human and ecological)
- Exposure pathways (external radiation, direct contact, ingestion, or inhalation)

The RAOs provide a basis for evaluating the capability of a specific remedial alternative to achieve compliance with potential ARARs and/or an intended level of risk protection for human health and the environment in accordance with the NCP ("Remedial Investigation/Feasibility Study and Selection of Remedy" [40 CFR 300.430(e)(2)(i)]), and CERCLA RI/FS Guidance (EPA/540/G-89/004). RAOs are presented in Section 8.1.4. Background information used in developing the RAOs is presented in Sections 8.1.1 through 8.1.3.

8.1.1 Contaminants of Concern

In the RI/FS process, the results of the risk assessment and fate and transport evaluation are used to identify COPCs, which represent contaminants that will be evaluated in the FS to define the COCs and guide the selection of remedial alternatives.

8.1.1.1 Waste Site Soil

The evaluation of remedial actions for specific waste sites relies upon a comprehensive review of all available data for each site, including field data if available, radiological surveys, process history, analogous site information, personal interviews, engineering drawings and as-builts, and any other available information. The following analytes were identified as COCs in soil based on this risk evaluations for previously remediated waste sites with verification data: cesium-137, cobalt-60, europium-152, europium-154, nickel-63, strontium-90, and technetium-99. For yet-to-be remediated waste sites, additional COPCs are identified based on review of waste site history/processes and characterization of analogous waste sites and listed in Table 8-1. As a result of this comprehensive review, the characteristics of each site are sufficiently defined for the purpose of alternative development and comparison in the FS. During implementation of remedial actions, should field conditions vary from those presented in the FS and indicate a need to re-evaluate the efficacy of the selected remedial action, the appropriate remedy modification will be used, consistent with CERCLA guidance. Section 8.2.1 presents additional information on the waste sites. As discussed in Section 4.1, waste sites collocated within historic orchard lands will be remediated as needed to meet the cleanup levels for contaminants attributable to Hanford Site operations.

Table 8-1. Summary of Soil COPCs Based on Process Knowledge

Radionuclides	Metals	Polychlorinated Biphenyls	Polynuclear Aromatic Hydrocarbons
Carbon-14	Antimony	Aroclor-1016	Benzo(a)pyrene
Cesium-137	Arsenic	Aroclor-1221	Benzo(b)fluoranthene
Cobalt-60	Barium	Aroclor-1232	Benzo(k)fluoranthene
Europium-152	Cadmium	Aroclor-1242	Chrysene
Europium-154	Chromium, total	Aroclor-1248	Dibenz(a,h)anthracene
Nickel-63	Cr(VI)	Aroclor-1254	Indeno(1,2,3-cd)pyrene
Strontium-90	Copper	Aroclor-1260	Pyrene
Technetium-99	Lead		
	Mercury		
	Nickel		
	Silver		
	Zinc		

8.1.1.2 Groundwater

The following contaminants were identified in the groundwater risk assessment in Chapter 6 as COCs for the 100-HR-3 OU groundwater: chromium (total), Cr(VI), nitrate, and strontium-90.

The groundwater risk assessment in Chapter 6 identified seven analytes for the 100-D Area, nine analytes for the 100-H Area, and seven analytes for the horn area, that are COPCs that move forward into the FS because they have an uncertainty associated with the data set and occurrence in groundwater. The COPCs in the 100-D Area include antimony, cadmium, carbon tetrachloride, cobalt, copper, nickel, and silver. The COPCs in the 100-H Area include antimony, cadmium, carbon tetrachloride, chromium, cobalt, copper, nickel, silver, and zinc. The COPCs in the horn area include antimony, cadmium, cobalt, copper, nickel, silver, and zinc. The nature and extent evaluation indicates that historically, these analytes have been detected in groundwater at concentrations above their respective action level (Section 6.3.2.3), but their presence was not associated with a specific location or with a trend. Because of the uncertain status of the COPCs, treatment for the COPCs are not evaluated in the alternatives developed in Chapter 9. To assure protectiveness and confirm current understanding of the nature and extent of contamination and potential risks, these analytes will be analyzed as part of the performance monitoring. If monitoring identifies that remedial action is necessary for the COPCs, these changes will be addressed through a ROD change.

8.1.2 Potential Applicable or Relevant and Appropriate Requirements

Substantive standards of promulgated regulations pertaining to CERCLA response actions are identified through the ARAR identification process. The ARARs identification process is based on CERCLA Section 121(d) and EPA guidance (CERCLA RI/FS Guidance [EPA/540/G-89/004], *CERCLA Compliance with Other Laws Manual: Interim Final* [EPA/540/G-89/006], and *CERCLA Compliance with Other Laws Manual: Part II* [EPA/540/G-89/009). Section 121(d) requires, with exceptions, that any promulgated substantive ARAR standard, requirement, criterion, or limitation under any federal environmental law, or any more stringent state requirement pursuant to a state environmental statute, be met (or a waiver justified) for any hazardous substance, pollutant, or contaminant that will remain onsite after completion of remedial action. Additionally, NCP, "Remedial Design/Remedial Action, Operation and Maintenance" (40 CFR 300.435[b][2]) requires that ARARs be attained (unless waived) during the remedial action.

8.1.2.1 ARARs Evaluation Process

The ARARs evaluation prepared for this RI/FS was conducted in accordance with the NCP, "Remedial Investigation/Feasibility Study and Selection of Remedy" (40 CFR 300.430[f][1][ii][B][2]). The identification of ARARs is a two-step process. First, it must be determined if the law or regulation is applicable. If not applicable, it must be determined if the law or regulation is both relevant and appropriate. The terms "applicable" and "relevant and appropriate" are defined in NCP, "Definitions" (40 CFR 300.5) as follows.

"Applicable requirements" are those substantive standards that specifically address the situation at a CERCLA site. These requirements would legally apply to remedial actions in the absence of CERCLA authority. All jurisdictional prerequisites of the requirement must be met in order for the requirement to be applicable, including specific application to federal agencies (for example, through a waiver of federal sovereign immunity).

"Relevant and appropriate" requirements mean those environmental requirements such as cleanup standards that address problems or situations sufficiently similar to those encountered at the CERCLA site that their use is well-suited to the particular site (NCP, "General" [(40 CFR 300.400(g)(2)]).

A requirement that is relevant and appropriate may not meet one or more jurisdictional prerequisites for applicability but still make sense at the site, given the circumstances of the site and the release.

In evaluating the relevance and appropriateness of a requirement, the eight comparison factors in NCP, "General" (40 CFR 300.400[g][2]) are considered:

- The purpose of the requirement and the purpose of the CERCLA action
- The medium regulated or affected by the requirement and the medium contaminated or affected at the CERCLA site
- The substances regulated by the requirement and the substances found at the CERCLA site
- The actions or activities regulated by the requirement and the remedial action contemplated at the CERCLA site
- Any variances, waivers, or exemptions of the requirement and their availability for the circumstances at the CERCLA site
- The type of place regulated and the type of place affected by the release or CERCLA action
- The type and size of structure or facility regulated and the type and size of structure or facility affected by the release or contemplated by the CERCLA action
- Any consideration of use or potential use of affected resources in the requirement and the use or potential use of the affected resource at the CERCLA site

To be considered (TBC) information represents another category of nonpromulgated advisories or guidance issued by federal or state governments that are not legally binding and do not have the status of ARARs. In some circumstances, TBC information will be evaluated, along with ARARs, in determining the remedial action necessary to protect human health and the environment. TBC information complements ARARs in determining protectiveness at a CERCLA site or in assessing implementation of certain actions. For example, because cleanup standards do not exist for all contaminants, health advisories, which would be TBC information, may be helpful in defining cleanup levels.

Section 161 of the AEA, as amended, provides DOE the authority to establish DOE orders containing instructions and operational requirements considered important to protect human health and the environment from nuclear material, source material, and byproduct materials. While the requirements of DOE Orders must be met, they are not ARARs and are independent of the TBC and ARARs identification process at the Hanford Site.

Potential ARARs for 100-D/H are examined to determine if they fall into one of three categories: chemical-specific, location-specific, or action-specific requirements. These categories are defined as follows:

- Chemical-specific requirements are usually health- or risk-based numerical values or methodologies that, when applied to site-specific conditions, result in the establishment of public and worker safety levels and site cleanup levels.
- Location-specific requirements are restrictions placed on the concentration of dangerous substances or the conduct of activities solely because they occur in special geographic areas.
- Action-specific requirements are usually technology- or activity-based requirements or limitations triggered by remedial actions performed at the site.

8.1.2.2 Waivers from ARARs

The CERCLA lead agency delegated authority under Section 121 may waive ARARs, with EPA's concurrence, and select a remedial action that does not attain the same level of cleanup as that identified by the ARARs. In *Superfund Implementation* (Executive Order 12580), the president delegated Section 121 authority to DOE for cleanup of DOE facilities. Section 121 of the *Superfund Amendments and Reauthorization Act of 1986* identifies the following circumstances in which DOE may waive ARARs for onsite remedial actions:

- The remedial action selected is only a part of a total remedial action (such as an interim action), and the final remedy will attain the ARAR upon its completion.
- Compliance with the ARAR will result in a greater risk to human health and the environment than alternative options.
- Compliance with the ARAR is technically impracticable from an engineering perspective.
- An alternative remedial action will attain an equivalent standard of performance using another method or approach.
- The ARAR is a state requirement that the state has not consistently applied (or demonstrated the intent to apply consistently) in similar circumstances.

ARAR waivers can be established in the ROD or through a ROD modification.

8.1.2.3 Potential ARARs Identified

Potential federal and Washington State ARARs are presented in Table N-1 of Appendix N. When the final remedy selection is documented in the ROD, all federal and state ARARs with which the final remedy must comply are also finalized. Key potential ARARs are identified in following text.

Potential Chemical-Specific ARARs. The chemical-specific ARARs and TBCs that may affect remediation of 100-D/H OU are the substantive elements of the *Washington Administrative Code* regulations that implement the MTCA (WAC 173-340). Within this branch of the *Washington Administrative Code*, there are detailed regulations for developing standards for remedial actions involving soil cleanup (MTCA, "Unrestricted Land Use Soil Cleanup Standards" [WAC 173-340-740]) and groundwater cleanup standards (MTCA, "Groundwater Cleanup Standards"[WAC 173-340-720]). These standards are in the form of risk-based concentrations, or established by modeling, that help establish soil and groundwater cleanup standards for nonradioactive contaminants. Following is a list of additional Washington State and federal regulations:

- Substantive portions of MTCA (WAC 173-340) ("Selection of Cleanup Actions" [WAC 173-340-360] and MTCA "Overview of Cleanup Standards" [WAC 173-340-700] through MTCA "Site-Specific Terrestrial Ecological Evaluation Procedures" [WAC 173-340-7493]) (2007)
- Nonzero MCL goals and MCLs promulgated under the *Safe Drinking Water Act of 1974* (SDWA) ("National Primary Drinking Water Regulations" [40 CFR 141]) and/or by the state of Washington ("Group A Public Water Supplies" [WAC 246-290])
- The AWQC developed under the *Clean Water Act* (Section 304) and/or promulgated by the state of Washington ("Water Quality Standards for Groundwaters of the State of Washington" [WAC 173-200] and Surface Water Quality Standards [WAC 173-201A])

- The *Toxic Substances Control Act of 1976* (implemented via "Polychlorinated Biphenyls (PCBs) Manufacturing, Processing, Distribution in Commerce, and Use Prohibitions" [40 CFR 761])
- "National Primary and Secondary Ambient Air Quality Standards" (40 CFR 50)
- "National Emissions Standards for Hazardous Air Pollutants" (40 CFR 61)

Potential Location-Specific ARARs. Potential location-specific ARARs that have been identified for the 100-D/H OU include those that protect cultural, historic, and Native American sites and artifacts under the *Native American Graves Protection and Repatriation Act of 1990*, *Archeological and Historic Preservation Act of 1974*, and *National Historic Preservation Act of 1966*. In addition, those ARARs protect listed endangered and threatened species or their critical habitat under the *Endangered Species Act*. The *Migratory Bird Treaty Act of 1918* has been identified as substantive standards for DOE compliance in executive orders and a Memorandum of Understanding between DOE and USFWS, and is pertinent to CERCLA response actions when there is a potential to adversely affect protected bird species.

Potential Action-Specific ARARs. Action-specific ARARs that could be pertinent to possible remediation activities at 100-D/H relate to waste management activities, solid and dangerous waste regulations (for management of characterization and remediation wastes and performance standards for waste left in place), and radioactive waste management under AEA regulations. The other major category of action-specific ARARs concern standards for controlling emissions to the environment.

8.1.2.4 Waste Management Standards

Remedial action alternatives proposed in Chapter 9 of this FS have the potential to produce a variety of waste that contains both radioactive and chemical constituents. It is anticipated that most of the waste will be designated as low-level. However, quantities of PCB-contaminated waste, and asbestos and asbestos-containing material could be included in remediation waste. The majority of the waste will be in a solid form.

The storage, treatment, and disposal of hazardous waste and the hazardous component of mixed waste resulting from the remedial action would be subject to the substantive provisions of RCRA. In the State of Washington, RCRA is implemented through "Dangerous Waste Regulations" (WAC 173-303), which is an EPA-authorized State RCRA program. Treatment standards for dangerous or mixed waste that are subject to RCRA land disposal restrictions are specified in "Dangerous Waste Regulations," "Land Disposal Restrictions" (WAC 173-303-140), which incorporates "Land Disposal Restrictions" (40 CFR 268) by reference. Radioactive waste is managed by DOE under the authority of the AEA. EPA has regulatory authority over release of radioactive waste in context of a CERCLA action.

The *Toxic Substances Control Act of 1976* and regulations in "Polychlorinated Biphenyls (PCBs) Manufacturing, Processing, Distribution in Commerce, and Use Prohibitions" (40 CFR 761) generally govern the management and disposal of PCB wastes. The *Toxic Substances Control Act of 1976* regulations contain specific provisions for PCB waste, including PCB waste that contains a radioactive component. PCBs also are considered underlying hazardous constituents under RCRA and, thus, could be subject to "Dangerous Waste Regulations" (WAC 173-303) and "Land Disposal Restrictions" (40 CFR 268) requirements.

Removal and disposal of asbestos and asbestos-containing material are regulated under the *Clean Air Act of 1990* and "National Emission Standards for Hazardous Air Pollutants" (NESHAPs), "National Emission Standard for Asbestos" (40 CFR 61, Subpart M). These regulations provide for special precautions to prevent environmental releases or exposure to personnel of airborne emissions of asbestos fibers during remedial actions.

Waste generated through CERCLA remedial actions and designated as low-level radioactive waste that meets ERDF acceptance criteria (*Environmental Restoration Disposal Facility Waste Acceptance Criteria* [WCH-191]) is planned to be disposed at ERDF, which is engineered to meet appropriate performance standards for mixed radioactive and hazardous waste. ERDF is considered onsite with Hanford remedial actions for the purpose of management and/or disposal of waste.[1]

Waste designated as dangerous or mixed waste would be treated as appropriate to meet land disposal restrictions and ERDF acceptance criteria, and disposed of at ERDF. ERDF is an engineered facility that provides a high degree of protection to human health and the environment and meets RCRA minimum technical requirements for landfills, including standards for a double liner, a leachate collection system, leak detection, monitoring, and final cover. Construction and operation of ERDF was authorized using a separate CERCLA ROD (*Declaration of the Interim Record of Decision for the Environmental Restoration Disposal Facility* [hereinafter called ERDF ROD (EPA/ROD/R10-95/100)]; *Record of Decision Amendment: U.S. Department of Energy Environmental Restoration Disposal Facility Hanford Site – 200 Area Benton County, Washington* [EPA/AMD/R10-02/030]). *Explanation of Significant Differences: USDOE Environmental Restoration Disposal Facility, Hanford Site, Benton County, Washington* (hereinafter called ERDF ESD [EPA/ESD/R10-96/145]) modified the ERDF ROD (EPA/ROD/R10-95/100) to clarify the eligibility of waste generated during cleanup of the Hanford Site. Per ERDF ESD (EPA/ESD/R10-96/145), ERDF is eligible for disposal of any low-level waste, mixed waste, and hazardous/dangerous waste generated as a result of cleanup actions (for example, remedial/removal action waste and investigation-derived waste), provided the waste meets ERDF waste acceptance criteria requirements and appropriate CERCLA decision documents are in place.

8.1.2.5 Standards Controlling Emissions to the Environment

Remedial action alternatives proposed in Chapter 9 of this FS have the potential to generate airborne emissions of both radioactive and toxic/criteria airborne emissions. Implementation of these activities and associated air monitoring will be discussed in the RD/RAWP for 100-D/H.

8.1.2.6 Radiological Air Emissions

The federal *Clean Air Act of 1990* and amendments, and the "Washington Clean Air Act" (RCW 70.94) each require regulation of radioactive air emissions. The state implementing regulation "Ambient Air Quality Standards and Emission Limits for Radionuclides" (WAC 173-480) sets standards that are as stringent or more so than the standards under the federal *Clean Air Act of 1990* and amendments, including the Federal implementing regulation, NESHAPs "National Emission Standards for Emissions of Radionuclides Other than Radon from Department of Energy Facilities" (40 CFR 61, Subpart H). The EPA's partial delegation of the Subparts A and H authority to the State of Washington includes all substantive emissions monitoring, abatement, and reporting aspects of the federal regulation. These state standards protect the public by conservatively establishing exposure standards applicable to the

[1] CERCLA Section 104(d)(4), "where two or more noncontiguous facilities are reasonably related on the basis of geography, or on the basis of the threat or potential threat to the public health or welfare or the environment, the President may, at his discretion, treat these facilities as one." The preamble to the NCP (40 CFR 300) clarifies the stated EPA interpretation that when noncontiguous facilities are reasonably close to one another, and wastes at these sites are compatible for a selected treatment or disposal approach, CERCLA Section 104(d)(4) allows the lead agency to treat these related facilities as one for response purposes. This allows the lead agency to manage waste transferred between such noncontiguous facilities without having to obtain a permit. ERDF is considered to be onsite for response purposes under this remedial/removal/removal action. It should be noted that the scope of work covered in this remedial/removal/removal action is for a facility and waste contaminated with hazardous substances. Materials encountered during implementation of the selected remedial/removal/removal action that are not contaminated with hazardous substances are outside the authority of CERCLA and will be dispositioned by DOE.

maximally exposed public individual. Members of the public can travel on the Columbia River through the Hanford Reach, but they cannot "abide or reside" there (Supplemental Analysis for the CLUP).

"Ambient Air Quality Standards and Emission Limits for Radionuclides" (WAC 173-480) limits emissions of radionuclides to the ambient air by requirement that emissions of radionuclides in the air shall not cause a maximum effective dose equivalent of more than 10 mrem/yr to the whole body to any member of the public. Under the state implementing regulations, "Radiation Protection—Air Emissions" (WAC 246-247) in "Radiation Protection—Air Emissions," "Definitions" (WAC 246-247-030[15]), defines the member of the public (real or hypothetical) who abides or resides in an unrestricted area. This member of the public may receive the highest total effective dose equivalent from the emission unit(s) under consideration, taking into account all exposure pathways affected by the radioactive air emissions. In addition, by its adoption of the federal standard at NESHAPs, "Standard" (40 CFR 61.92), the state limits radionuclide airborne emissions from the Hanford Site (that is, facility) to not exceed amounts that would cause an exposure to any member of the public of greater than 10 mrem/yr effective dose equivalent. The state implementing regulation "Radiation Protection—Air Emissions" (WAC 246-247), which adopts the "Ambient Air Quality Standards and Emission Limits for Radionuclides" (WAC 173-480) standards, and the NESHAPs, "National Emission Standards for Emissions of Radionuclides Other than Radon from Department of Energy Facilities" (40 CFR 61, Subpart H) standard, requires verification of compliance with the 10 mrem/yr standard, and would be applicable to the remedial action.

"Radiation Protection—Air Emissions" (WAC 246-247) further addresses sources emitting radioactive airborne emissions by requiring monitoring of such sources (emission units). Such monitoring may involve various methods depending upon the configuration of the source. Most stacks or vents are monitored by extracting a sample of the effluent stream from the stack or vent, with subsequent analysis of the sample. Emissions that do not pass through a stack, vent, or other orifice, are termed diffuse emissions, and these are normally monitored by extraction of a sample of the ambient air, with subsequent laboratory analysis. The substantive provisions of "Radiation Protection—Air Emissions" (WAC 246-247) that require monitoring of radioactive airborne emissions potentially would be applicable to remedial action and would generally be an "applicable" ARAR.

The above state implementing regulations further require control of radioactive airborne emissions to the extent economically and technologically feasible ("General Standards" and associated definitions [WAC 246-247-040(3) and -040(4)]). To cover the substantive aspect of these requirements, best or reasonably achieved control technology could be addressed by ensuring that applicable emission control technologies (those successfully operated in similar applications) would be used when economically and technologically feasible (that is, based on cost/benefit). Controls will be administered as appropriate using the best methods from among those that are reasonable and effective.

8.1.2.7 Criteria/Toxic Air Emissions

Under "General Regulations for Air Pollution Sources" (WAC 173-400) and "Controls for New Sources of Toxic Air Pollutants" (WAC 173-460), requirements are established for the regulation of emissions of criteria/toxic air pollutants. The primary nonradioactive emissions resulting from remedial actions will be fugitive particulate matter. In accordance with "General Standards for Maximum Emissions" (WAC 173-400-040), reasonable precautions must be taken to (1) prevent the release of air contaminants associated with fugitive emissions resulting from excavation, materials handling, or other operations; and (2) prevent fugitive dust from becoming airborne from fugitive sources of emissions. The use of treatment technologies that would result in emissions of toxic air pollutants that would be subject to the substantive applicable requirements of "Controls for New Sources of Toxic Air Pollutants" (WAC 173-460) are not anticipated to be a part of remedial actions selected for 100-D/H.

If treatment of some waste encountered during the remedial action is required to meet ERDF waste acceptance criteria, the type of treatment anticipated would consist of solidification/stabilization techniques such as microencapsulation or grouting, and "Controls for New Sources of Toxic Air Pollutants" (WAC 173-460) would not be considered an ARAR. If more aggressive treatment is required that would result in the emission of regulated air pollutants, the substantive requirements of "Requirements for New Sources in Attainment or Unclassifiable Areas" (WAC 173-400-113[2]) and "Control Technology Requirements" (WAC 173-460-060) would be evaluated to determine potential applicability.

Emissions to the air will be minimized during implementation of remedial actions through use of standard industry practices such as the application of water sprays and fixatives. These techniques are considered reasonable precautions to control fugitive emissions as required by the regulatory standards.

8.1.2.8 Groundwater Beneficial Use

CERCLA and NCP establish separate requirements for a groundwater remedy: to be protective of human health and the environment and to meet ARARs. This is a concept of central importance to the development of the groundwater remedy for the 100-HR-3 OU. These separate requirements are further clarified in a memorandum ("Clarification of the Role of Applicable, or Relevant and Appropriate Requirements in Establishing Preliminary Remediation Goals Under CERCLA" [Fields, 1997]). Specifically, this memorandum clarifies that, in rare instances, even absent multiple pathways or contaminants, PRGs should be set at levels more protective than required by a given ARAR, where application of the ARAR would not be protective of human health and the environment.

The requirement to achieve threshold protectiveness and ARAR-based requirements is established by the NCP (40 CFR 300), which also establishes the requirement to return useable groundwater to beneficial use within a reasonable period. EPA generally defers to state agency definitions of useable groundwater provided under the various comprehensive state groundwater protection programs, administered by the states across the United States and a state's determination of groundwater usability at CERCLA sites (*Guidance on Remedial Actions for Contaminated Ground Water at Superfund Sites* [EPA/540/G-88/003]). The State of Washington defines groundwater as potable in "Groundwater Cleanup Standards" (WAC 173-340-720[2]), unless the exclusion criteria in "Groundwater Cleanup Standards" (WAC 173-340-720[2][a] through [c]) can be demonstrated (that is, insufficient yield, natural constituents that make it unsuitable as a drinking water source). The groundwater within the100-HR-3 OU does not meet the exclusion criteria; therefore, it is classified as potable and must be restored to beneficial use wherever practicable, and within a period that is reasonably consistent with NCP requirements. The state of Washington has further determined that the highest beneficial use for potable groundwater at most of the cleanup sites within the state, including the site, is as a potential source of domestic drinking water ("Groundwater Cleanup Standards" [WAC 173-340-720(1)(a)]).

Groundwater within the 100-HR-3 OU is currently contaminated, and withdrawal is prohibited because of institutional controls placed on it by DOE. Under current site use conditions, no groundwater wells are available for public consumption specific to 100-D/H. Further, regardless of land use designations for soil, groundwater within this OU is not anticipated to become a future source of drinking water until cleanup criteria are met and groundwater is restored to its highest beneficial use. However, groundwater in this risk analysis is evaluated for drinking water use to support the determination of the basis for action and to support the development of PRGs for evaluating remedial alternatives in the FS.

8.1.2.9 Surface Water Beneficial Use

Surface water beneficial use is considered because groundwater within the 100-HR-3 OU currently discharges to the Columbia River through upwelling and seeps. Surface Water Quality Standards, "Use Designations—Fresh Waters" (WAC 173-201A-600) and Surface Water Quality Standards,

"Table 602-Use Designations for Fresh Waters by Water Resource Inventory Area" (WAC 173-201A-602), identify the beneficial use (or designated uses) for rivers and streams of Washington. Designated uses for waters of Washington can include public water supply; protection for fish, shellfish, and wildlife; and recreational, agricultural, industrial, navigational, and aesthetic purposes. Water quality criteria are designed to protect the designated uses and are used to assess the general health of Washington surface waters and set permit limits.

The point of compliance for surface water quality standards is defined in the MTCA, "Surface Water Cleanup Standards" (WAC 173-340-730[7][a]) as the point or points at which hazardous substances are released to surface waters of the state. MTCA, "Surface Water Cleanup Standards" (WAC 173-340-730[7][b]) indicates that no mixing zone shall be allowed to demonstrate compliance with surface water cleanup levels.

Designated uses of the Columbia River, identified in Surface Water Quality Standards, "Table 602—Use Designations for Fresh Waters by Water Resource Inventory Area" (WAC 173-201A-602), include the following:

- Aquatic life uses—spawning and rearing
- Recreational uses—primary contact
- Water supply uses—drinking water, industrial water, agricultural water, and stock water
- Miscellaneous uses—wildlife habitat, harvesting, commercial/navigation, boating, and aesthetics

The groundwater risk assessment presented in Section 6.3 evaluates potential exposure of aquatic organisms to contaminants in the 100-HR-3 OU. This assessment uses the most stringent federal and state water quality criteria to support the basis for action and to support PRG development.

8.1.3 Remedial Action Objectives

Under CERCLA and the NCP (40 CFR 300), soil and groundwater remedies must (1) be protective of human health and the environment, and (2) meet ARARs (or satisfy criteria for an ARAR to be waived). RAOs must be developed to address COPCs, media of concern, potential receptors, and exposure pathways. RAOs are general descriptions of what a cleanup under CERCLA is expected to accomplish. They are narrative statements that define the extent to which waste sites require cleanup to protect human health and the environment.

The RAOs were based on existing River Corridor regulatory documents (for example, interim action RODs) and were expanded to cover gaps when integrating all media and resources for an area. Media specific RAOs were developed for groundwater (RAO 1), surface water (RAO 2), and soil (RAOs 3 through 5). The combined RAO list is as follows:

- **RAO 1.** Prevent unacceptable risk to human health from ingestion of and exposure to groundwater containing contaminant concentrations above federal and state standards and risk-based thresholds.
- **RAO 2.** Prevent unacceptable risk to human health and ecological exposure to surface water containing contaminant concentrations above federal and state standards and risk-based thresholds.
- **RAO 3.** Prevent unacceptable risk from contaminants migrating and/or leaching through soil that will result in groundwater concentrations that exceed federal and state standards and risk based thresholds for protection of surface water and groundwater.
- **RAO 4.** Prevent unacceptable risk to human health and ecological receptors from exposure to the upper 4.6 m (15 ft) of soil contaminated with nonradiological constituents at concentrations above the

unrestricted land use criteria for human health (provided in MTCA B) or soil contaminant levels for ecological receptors.

- **RAO 5.** Prevent unacceptable risk to human health and ecological receptors from exposure to the upper 4.6 m (15 ft) of soil and to structures and debris contaminated with radiological constituents. For human health and ecological receptors:
  - Prevent exposure to radiological constituents at concentrations at or above a dose rate limit that causes an excess cancer lifetime risk threshold of 10^{-6} to 10^{-4} above background for the residential exposure scenario.
  - Protect ecological receptors based on a dose rate limit of 0.1 rad/day for terrestrial wildlife populations, which is a TBC criterion.

8.1.4 Preliminary Remediation Goals

To meet the RAOs, PRGs are established. These goals generally are quantitative cleanup levels that would meet ARARs and risk-based levels, and would be protective of human health and the environment. The PRGs will be used to assess the effectiveness of the selected remedial alternatives in meeting the RAOs. Table 8-2 provides a summary of the 100-D/H Direct Contact Human Health PRGs, and Groundwater Protection and Surface Water Protection SSLs and PRGs. The interim action ROD remedial action goal identified in the 100 Area RDR/RAWP (DOE/RL-96-17) are also listed in Table 8-2 for direct comparison to the PRGs from this RI/FS. In this table, direct contact human health PRGs for radionuclides (highlighted in yellow) are the lowest of the PRGs calculated for the residential exposure scenario (based on a target cancer risk level) and the residential interim action remedial action goal (based on radiological dose) as defined in the 100 Area RDR/RAWP (DOE/RL-96-17). For non-radionuclides, direct contact human health PRGs (also highlighted in yellow) are the RI/FS 2007 MTCA Method B values. Green highlighting denotes the PRG described in Chapter 5 of this RI/FS above background for each analyte for groundwater/surface water protection except for Cr(VI), which is compared to the interim action remedial action goal of 2.0 mg/kg.

The PRGs represent a core component of the overall technology screening and remedial alternative development process in the FS. PRGs are numerical values expressed as concentrations for a chemical or radionuclide in an environmental media. A remedial action achievement of PRGs results in residual contamination that is protective of human health and the environment (NCP, "Remedial Investigation/Feasibility Study and Selection of Remedy" [40 CFR 300.430(e)(2)(i)]). PRGs are also used to identify the area and volume of environmental media that must be addressed; therefore, PRGs are determined before the development of the remedial alternatives.

Meeting PRGs and the potential ARARs and, by extension, achieving RAOs, can be accomplished by reducing concentrations (or activities) of contaminants to PRG levels or by eliminating potential exposure pathways/routes. Contaminant-specific and numeric soil PRGs for direct contact exposure, protection of groundwater, and protection of surface water typically are presented as concentrations, which for nonradionuclides are in milligrams per kilogram for soil and in picocuries per gram for radionuclides. Contaminant-specific and numerical cleanup levels for groundwater typically are expressed in micrograms per liter for nonradiological COCs and picocuries per liter for radiological COCs.

Residual risks following completion of remediation of the waste sites must meet the 10^{-4} to 10^{-6} ELCR for radiological and carcinogenic COCs and must be less than or equal to an HI value of 1.0 for hazardous substances (as described in Chapter 6 of this RI/FS) for direct contact exposure with soil by humans. These cumulative risk and hazard thresholds must be met for each waste site from the ground surface to 4.6 m (15 ft) bgs.

A summary of the 100-D/H Ecological PRGs is provided in Table 8-3 for the four receptor groups (plants, invertebrates, birds, and mammals) that will be used within the SMDP for ecological protection. If ecological PRGs are exceeded, the site managers will evaluate this exceedance using the SMDP process described in Chapter 7 of this RI/FS.

Finally, to demonstrate that cleanups have achieved the groundwater protection PRGs, the cleanup verification process can involve the evaluation of the conceptual site model at the waste sites against the assumptions used to develop these PRGs (described in Chapter 5 of this RI/FS). To the extent a significant deviation from the groundwater/surface water protection PRG assumptions is observed, site-specific conditions can be used to revise the fate and transport models to evaluate the potential for the waste site to act as a source of groundwater contamination.

8.1.4.1 Development Approach

PRGs are presented for each environmental media of interest (soil and groundwater), each type of contaminant (hazardous substances and radionuclides), human and ecological receptors, and each potentially complete exposure pathway. The following sections describe the approach that was taken to develop PRGs for each media, receptor, and exposure pathway.

8.1.4.2 Direct Contact Exposure PRGs for Nonradiological Contaminants

Development of the PRGs for direct contact exposure to nonradiological contamination for both human and ecological receptors is described in the following sections.

Human Exposure. For human receptors, soil PRGs developed for direct contact and inhalation exposure pathways are risk-based standards for hazardous substances. Risk-based standards for individual hazardous substances are established using applicable federal and state laws and risk equations. Risk-based standards for individual carcinogens in an unrestricted exposure scenario are based on an ELCR of 1×10^{-6} and an HQ of 1.0 for individual noncarcinogenic substances as described in MTCA, "Unrestricted Land Use Soil Cleanup Standards" (WAC 173-340-740[3][b][iii][B]). Additional information about exposure assumptions and the risk bases is provided in Section 6.2.3.3.1 of this RI/FS.

Consistent with this approach, the methodology described for unrestricted land use under MTCA, "Unrestricted Land Use Soil Cleanup Levels" (WAC 173-340-740[3]), is used to calculate the risk-based standards for soil ingestion. Risk-based standards for inhalation pathways use equations and input parameters described in "Cleanup Standards to Protect Air Quality" (WAC 173-340-750[3]) and EPA-published volatilization factors and particulate emission factors.

For arsenic, lead, and total petroleum hydrocarbon, Table 740-1 in the Method A "Soil Cleanup Levels for Unrestricted Land Use" (MTCA "Tables" [WAC 173-340-900]) is used as the PRG for direct contact exposure.

Soil PRG values are also developed for the direct-contact and inhalation pathways combined, using the casual recreational user exposure scenario. The casual recreational user scenario is used to represent the reasonably anticipated future land use for the operable unit(s). A complete description of the activities, exposure assumptions, and risk bases associated with casual recreational user scenario is provided in Section 6.2.3.3.3 of this RI/FS. The PRG values listed in Table 8-2 for this exposure scenario are provided to aid in determining whether the cleanup actions achieve the CERCLA threshold criteria.

Table 8-2. Summary of 100-D and 100-H Operable Units Human Health, Groundwater Protection, and Surface Water Protection Soil PRGs

Analyte Name	CAS No.	Units	Hanford Site Background Concentration[a]	Human Health PRGs						PRGs Protective of Groundwater and Surface Water									
				RI/FS Soil MTCA B—Direct Contact PRG[b]	RI/FS Residential PRG[c]	RI/FS Soil Method B Inhalation PRG[d]	RI/FS Casual Recreational User—Direct Contact PRG[e,f]	RI/FS Resident Monument Worker—Direct Contact PRG[g]	DOE/RL 96-17 Residential RAG[h]	100-D 70:30/100:0 Contaminant Source Model GWP SSL[i]	100-H 70:30/100:0 Contaminant Source Model GGWP SSL[i]	100-D 70:30/100:0 Contaminant Source Model GWP PRG[i]	100-H 70:30/100:0 Contaminant Source Model GWP PRG[i]	DOE/RL 96-17 GWP RAG[h]	100-D 70:30/100:0 Contaminant Source Model SWP SSL[i]	100-H 70:30/100:0 Contaminant Source Model SWP SSL[i]	100-D 70:30/100:0 Contaminant Source Model SWP PRG[i]	100-H 70:30/100:0 Contaminant Source Model SWP PRG[i]	DOE/RL 96-17 SWP RAG[h]
Radionuclides																			
Americium-241	14596-10-2	pCi/g	--	--	155	--	2,570	275	32	--[j]	--[j]	--[j]	--[j]	--	--[j]	--[j]	--[j]	--[j]	--
Carbon-14	14762-75-5	pCi/g	--	--	81	--	328,000	52,000	8.7	--[j]	--[j]	--[j]	--[j]	--	--[j]	--[j]	--[j]	--[j]	--
Cesium-137	10045-97-3	pCi/g	1.1	--	4.4	--	100	6.2	6.2	--[j]	--[j]	--[j]	--[j]	1,470	--[j]	--[j]	--[j]	--[j]	2,930
Cobalt-60	10198-40-0	pCi/g	0.0084	--	3.1	--	63	3.3	1.4	--[j]	--[j]	--[j]	--[j]	13,900	--[j]	--[j]	--[j]	--[j]	27,800
Curium-243	15757-87-6	pCi/g	--	--	30	--	527	37	22	--[j]	--[j]	--[j]	--[j]	--	--[j]	--[j]	--[j]	--[j]	--
Europium-152	14683-23-9	pCi/g	--	--	3.7	--	66	3.8	3.3	--[j]	--[j]	--[j]	--[j]	--	--[j]	--[j]	--[j]	--[j]	--
Europium-154	15585-10-1	pCi/g	0.033	--	4.4	--	78	4.8	3.0	--[j]	--[j]	--[j]	--[j]	--	--[j]	--[j]	--[j]	--[j]	--
Europium-155	14391-16-3	pCi/g	0.054	--	327	--	5,870	354	125	--[j]	--[j]	--[j]	--[j]	--	--[j]	--[j]	--[j]	--[j]	--
Iodine-129	15046-84-1	pCi/g	--	--	0.076	--	3,035	434	0.25	0.67	5.6	4.4	67	0.25	0.67	5.6	4.4	67	0.25
Neptunium-237	13994-20-2	pCi/g	--	--	8.9	--	202	15	2.4	142	1,292	--[j]	--[j]	0.90	142	1,292	--[j]	--[j]	1.8
Nickel-63	13981-37-8	pCi/g	--	--	608	--	575,000	91,600	4,013	--[j]	--[j]	--[j]	--[j]	83	--[j]	--[j]	--[j]	--[j]	166
Niobium-94	14681-63-1	pCi/g	--	--	1.4	--	26	1.7	2.4	--[n]	--[n]	--[n]	--[n]	--	--[n]	--[n]	--[n]	--[n]	--
Plutonium-238	13981-16-3	pCi/g	0.0038	--	236	--	3,820	605	39	--[j]	--[j]	--[j]	--[j]	--	--[j]	--[j]	--[j]	--[j]	--
Plutonium-239/240	PU-239/240	pCi/g	0.025	--	203	--	3,340	539	35	--[j]	--[j]	--[j]	--[j]	--	--[j]	--[j]	--[j]	--[j]	--
Technetium-99	14133-76-7	pCi/g	--	--	1.5	--	114,449	17,322	5.8	33	295	70	504	0.46	33	295	70	504	0.92
Total beta radiostrontium (strontium-90)	SR-RAD	pCi/g	0.18	--	2.3	--	5,060	518	4.5	25,884	99,314	--[j]	--[j]	28	--[j]	--[j]	--[j]	--[j]	55
Tritium	10028-17-8	pCi/g	--	--	623	--	15,400	1,265,000	459	1,824	11,539	2,501	15,547	13	1,824	11,539	2,501	15,547	25
Uranium-233/234	U-233/234	pCi/g	1.1	--	133	--	5,810	931	1.1	--[k]	--[k]	--[k]	--[k]	1.1	--[k]	--[k]	--[k]	--[k]	1.1
Uranium-235	15117-96-1	pCi/g	0.11	--	16	--	295	22	0.61	--[k]	--[k]	--[k]	--[k]	0.50	--[k]	--[k]	--[k]	--[k]	0.50
Uranium-238	U-238	pCi/g	1.1	--	54	--	1,093	93	1.1	--[k]	--[k]	--[k]	--[k]	1.1	--[k]	--[k]	--[k]	--[k]	1.1
Metals																			
Aluminum	7429-90-5	mg/kg	11,800	80,000	--	>1,000,000	912,453	--	--	--[j]	--[j]	--[j]	--[j]	--	--[j]	--[j]	--[j]	--[j]	--
Antimony	7440-36-0	mg/kg	0.13	32	--	--	365	--	32	14	122	--[j]	--[j]	5.0	13	114	--[j]	--[j]	5.0
Arsenic	7440-38-2	mg/kg	6.5	20	--	42,414	4.5	--	20	--[j]	--[j]	--[j]	--[j]	20	--[j]	--[j]	--[j]	--[j]	20
Barium	7440-39-3	mg/kg	132	16,000	--	>1,000,000	182,000	--	5,600	--[j]	--[j]	--[j]	--[j]	200	--[j]	--[j]	--[j]	--[j]	400
Beryllium	7440-41-7	mg/kg	1.5	160	--	75,991	1,825	--	10	--[j]	--[j]	--[j]	--[j]	1.5	--[j]	--[j]	--[j]	--[j]	1.5
Boron	7440-42-8	mg/kg	3.9	16,000	--	>1,000,000	183,000	--	7,200	6,138	52,134	--[j]	--[j]	320	--[n]	--[n]	--[n]	--[n]	--
Cadmium	7440-43-9	mg/kg	0.56	40	--	101,000	821	--	14	--[j]	--[j]	--[j]	--[j]	0.81	--[j]	--[j]	--[j]	--[j]	0.81

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Analyte Name	CAS No.	Units	Hanford Site Background Concentration[a]	Human Health PRGs						PRGs Protective of Groundwater and Surface Water									
				RI/FS Soil MTCA B—Direct Contact PRG[b]	RI/FS Residential PRG[c]	RI/FS Soil Method B Inhalation PRG[d]	RI/FS Casual Recreational User—Direct Contact PRG[e,f]	RI/FS Resident Monument Worker—Direct Contact PRG[g]	DOE/RL 96-17 Residential RAG[h]	100-D 70:30/100:0 Contaminant Source Model GWP SSL[i]	100-H 70:30/100:0 Contaminant Source Model GGWP SSL[i]	100-D 70:30/100:0 Contaminant Source Model GWP PRG[j]	100-H 70:30/100:0 Contaminant Source Model GWP PRG[j]	DOE/RL 96-17 GWP RAG[h]	100-D 70:30/100:0 Contaminant Source Model SWP SSL[i]	100-H 70:30/100:0 Contaminant Source Model SWP SSL[i]	100-D 70:30/100:0 Contaminant Source Model SWP PRG[j]	100-H 70:30/100:0 Contaminant Source Model SWP PRG[j]	DOE/RL 96-17 SWP RAG[h]
Chromium	7440-47-3	mg/kg	19	120,000	--	--	>1,000,000	--	80,000	--[j]	--[j]	--[j]	--[j]	19	--[j]	--[j]	--[j]	--[j]	19
Cobalt	7440-48-4	mg/kg	16	24	--	20,264	274	--	24	--[j]	--[j]	--[j]	--[j]	16	--[j]	--[j]	--[j]	--[j]	--
Copper	7440-50-8	mg/kg	22	3,200	--	--	36,500	--	2,960	74,214	80,889	--[j]	--[j]	59	1,044	1,138	--[j]	--[j]	22
Cr(VI)	18540-29-9	mg/kg	--	240	--	2,171	2,740	--	2.1	6.0[l]	6.0[l]	6.0[l]	6.0[l]	4.8	5.4	6.0[m]	2.1	2.1	2.0
Iron	7439-89-6	mg/kg	32,600	56,000	--	--	638,750	--	--	--[j]	--[j]	--[j]	--[j]	--	--[j]	--[j]	--[j]	--[j]	--
Lead	7439-92-1	mg/kg	10	250	--	--	--	--	353	--[j]	--[j]	--[j]	--[j]	10	--[j]	--[j]	--[j]	--[j]	10
Lithium	7439-93-2	mg/kg	13	160	--	--	1,825	--	160	--[j]	--[j]	--[j]	--[j]	34	--[n]	--[n]	--[n]	--[n]	--
Manganese	7439-96-5	mg/kg	512	11,200	--	>1,000,000	128,000	--	3,760	--[j]	--[j]	--[j]	--[j]	512	--[j]	--[j]	--[j]	--[j]	512
Mercury	7439-97-6	mg/kg	0.013	24	--	>1,000,000	274	--	24	--[j]	--[j]	--[j]	--[j]	0.33	--[j]	--[j]	--[j]	--[j]	0.33
Molybdenum	7439-98-7	mg/kg	0.47	400	--	--	4,563	--	400	5,979	9,193	--[j]	--[j]	8.0	96,887	148,954	--[j]	--[j]	--
Nickel	7440-02-0	mg/kg	19	1,600	--	701,000	18,250	--	1,600	--[j]	--[j]	--[j]	--[j]	19	--[j]	--[j]	--[j]	--[j]	27
Selenium	7782-49-2	mg/kg	0.78	400	--	>1,000,000	4,560	--	400	159	1,313	--[j]	--[j]	5.0	16	131	--[j]	--[j]	1.0
Silver	7440-22-4	mg/kg	0.17	400	--	--	4,563	--	400	--[j]	--[j]	--[j]	--[j]	8.0	--[j]	--[j]	--[j]	--[j]	0.73
Strontium	7440-24-6	mg/kg	--	48,000	--	--	547,500	--	48,000	--[j]	--[j]	--[j]	--[j]	960	--[j]	--[j]	--[j]	--[j]	--
Thallium	7440-28-0	mg/kg	0.19	--	--	--	--	--	--	--[j]	--[j]	--[j]	--[j]	--	--[j]	--[j]	--[j]	--[j]	--
Tin	7440-31-5	mg/kg	--	48,000	--	--	547,500	--	48,000	--[j]	--[j]	--[j]	--[j]	960	--[j]	--[j]	--[j]	--[j]	--
Total_U_Isotopes	Total_U_Isotopes	mg/kg	3.2	240	--	>1,000,000	2,737	--	240	39	331	--[j]	--[j]	3.2	1,005	8,590	--[j]	--[j]	3.2
Uranium (soluble salts)	7440-61-1	mg/kg	3.2	240	--	>1,000,000	2,737	--	240	39	331	--[j]	--[j]	3.2	1,005	8,590	--[j]	--[j]	3.2
Vanadium	7440-62-2	mg/kg	85	400	--	--	4,560	--	560	--[j]	--[j]	--[j]	--[j]	85	--[n]	--[n]	--[n]	--[n]	--
Zinc	7440-66-6	mg/kg	68	24,000	--	--	274,000	--	24,000	--[j]	--[j]	--[j]	--[j]	480	--[j]	--[j]	--[j]	--[j]	68
Other Inorganics																			
Chloride	16887-00-6	mg/kg	100	--	--	--	--	--	--	9,151	81,804	19,306	139,893	25,000	8,419	75,259	17,762	128,701	--
Cyanide	57-12-5	mg/kg	--	1,600	--	--	18,250	--	1,600	1,627	10,678	--[j]	--[j]	20	42	278	--[j]	--[j]	1.0
Fluoride	16984-48-8	mg/kg	2.8	4,800	--	>1,000,000	54,750	--	4,800	--[j]	--[j]	--[j]	--[j]	96	--[n]	--[n]	--[n]	--[n]	400
Nitrate	14797-55-8	mg/kg	52	568,000	--	--	>1,000,000	--	567,000[m]	1,647	14,725	3,475	25,181	4,430[m]	1,647	14,725	3,475	25,181	8,860[n]
Nitrite	14797-65-0	mg/kg	--	24,000	--	--	274,000	--	26,300[m]	121	1,080	255	1,847	329[m]	--[n]	--[n]	--[n]	--[n]	658[n]
Sulfate	14808-79-8	mg/kg	237	--	--	--	--	--	--	9,151	81,804	19,306	139,893	25,000	--[n]	--[n]	--[n]	--[n]	--

Table 8-2. Summary of 100-D and 100-H Operable Units Human Health, Groundwater Protection, and Surface Water Protection Soil PRGs

Analyte Name	CAS No.	Units	Hanford Site Background Concentration[a]	Human Health PRGs						PRGs Protective of Groundwater and Surface Water									
				RI/FS Soil MTCA B—Direct Contact PRG[b]	RI/FS Residential PRG[c]	RI/FS Soil Method B Inhalation PRG[d]	RI/FS Casual Recreational User—Direct Contact PRG[e,f]	RI/FS Resident Monument Worker—Direct Contact PRG[g]	DOE/RL 96-17 Residential RAG[h]	100-D 70:30/100:0 Contaminant Source Model GWP SSL[i]	100-H 70:30/100:0 Contaminant Source Model GGWP SSL[i]	100-D 70:30/100:0 Contaminant Source Model GWP PRG[i]	100-H 70:30/100:0 Contaminant Source Model GWP PRG[i]	DOE/RL 96-17 GWP RAG[h]	100-D 70:30/100:0 Contaminant Source Model SWP SSL[i]	100-H 70:30/100:0 Contaminant Source Model SWP SSL[i]	100-D 70:30/100:0 Contaminant Source Model SWP PRG[i]	100-H 70:30/100:0 Contaminant Source Model SWP PRG[i]	DOE/RL 96-17 SWP RAG[h]
Polychlorinated Biphenyls																			
Polychlorinated Biphenyls	1336-36-3	mg/kg	--	2.5	--	>1,000,000	13	--	0.50	--	--	--	--	--	--	--	--	--	
Aroclor-1016	12674-11-2	mg/kg	--	5.6	--	>1,000,000	46	--	0.50	--[j]	--[j]	--[j]	--[j]	0.017	--[j]	--[j]	--[j]	--[j]	0.017
Aroclor-1221	11104-28-2	mg/kg	--	0.50	--	320,000	2.0	--	0.50	0.39	2.5	--[j]	--[j]	0.017	0.017	0.017	--[j]	--[j]	0.017
Aroclor-1232	11141-16-5	mg/kg	--	0.50	--	320,000	2.0	--	0.50	0.39	2.5	--[j]	--[j]	0.017	0.017	0.017	--[j]	--[j]	0.017
Aroclor-1242	53469-21-9	mg/kg	--	0.50	--	320,000	2.6	--	0.50	--[j]	--[j]	--[j]	--[j]	0.017	--[j]	--[j]	--[j]	--[j]	0.017
Aroclor-1248	12672-29-6	mg/kg	--	0.50	--	320,000	2.6	--	0.50	--[j]	--[j]	--[j]	--[j]	0.017	--[j]	--[j]	--[j]	--[j]	0.017
Aroclor-1254	11097-69-1	mg/kg	--	0.50	--	320,000	2.6	--	0.50	--[j]	--[j]	--[j]	--[j]	0.017	--[j]	--[j]	--[j]	--[j]	0.017
Aroclor-1260	11096-82-5	mg/kg	--	0.50	--	320,000	2.6	--	0.50	--[j]	--[j]	--[j]	--[j]	0.017	--[j]	--[j]	--[j]	--[j]	0.017
Pesticides																			
2,4,5-TP(2-(2,4,5-Trichlorophenoxy)propionic acid)Silvex	93-72-1	mg/kg	--	640	--	--	5,703	--	640	4.3	37	8.5	71	5.0	0.86	7.3	1.7	14	--
2,4,5-T(2,4,5-Trichlorophenoxyacetic acid)	93-76-5	mg/kg	--	800	--	--	7,129	--	800	11	91	21	168	16	--[n]	--[n]	--[n]	--[n]	--
2,4-DB(4-(2,4-Dichlorophenoxy)butanoic acid)	94-82-6	mg/kg	--	640	--	--	5,703	--	640	13	106	25	217	13	642	5,451	1,286	11,115	--
2,4-Dichlorophenoxyacetic acid	94-75-7	mg/kg	--	800	--	--	8,004	--	640	3.9	33	7.6	58	7.0	5.5	47	11	83	--
4,4'-DDD (Dichlorodiphenyldichloroethane)	72-54-8	mg/kg	--	4.2	--	>1,000,000	24	--	4.2	--[j]	--[j]	--[j]	--[j]	0.037	--[j]	--[j]	--[j]	--[j]	0.0033
4,4'-DDE (Dichlorodiphenyldichloroethylene)	72-55-9	mg/kg	--	2.9	--	>1,000,000	17	--	2.9	--[j]	--[j]	--[j]	--[j]	0.026	--[j]	--[j]	--[j]	--[j]	0.0033
4,4'-DDT (Dichlorodiphenyltrichloroethane)	50-29-3	mg/kg	--	2.9	--	>1,000,000	20	--	2.9	--[j]	--[j]	--[j]	--[j]	0.026	--[j]	--[j]	--[j]	--[j]	0.0033
Aldrin	309-00-2	mg/kg	--	0.059	--	37,220	0.33	--	0.059	--[j]	--[j]	--[j]	--[j]	0.0017	--[j]	--[j]	--[j]	--[j]	0.0017
Alpha-BHC	319-84-6	mg/kg	--	0.16	--	101,322	0.90	--	0.16	0.016	0.13	--[j]	--[j]	0.0017	0.0030	0.025	--[j]	--[j]	0.0017
Alpha-Chlordane	5103-71-9	mg/kg	--	2.9	--	>1,000,000	19	--	2.9	--[j]	--[j]	--[j]	--[j]	0.025	--[j]	--[j]	--[j]	--[j]	0.017
Beta-BHC	319-85-7	mg/kg	--	0.56	--	344,111	3.1	--	0.56	0.067	0.57	--[j]	--[j]	0.0049	0.013	0.11	--[j]	--[j]	0.0055
Dalapon	75-99-0	mg/kg	--	2,400	--	--	21,387	--	2,400	7.7	68	16	117	20	417	3,669	862	6,285	--
Delta-BHC	319-86-8	mg/kg	--	--	--	--	--	--	--	--[n]	--[n]	--[n]	--[n]	--	--[n]	--[n]	--[n]	--[n]	--
Dicamba	1918-00-9	mg/kg	--	2,400	--	--	21,387	--	2,400	27	226	52	400	48	1,358	11,583	2,675	20,508	--
Dieldrin	60-57-1	mg/kg	--	0.063	--	39,648	0.35	--	0.063	--[j]	--[j]	--[j]	--[j]	0.0033	--[j]	--[j]	--[j]	--[j]	0.0033
Dinoseb(2-secButyl-4,6-dinitrophenol)	88-85-7	mg/kg	--	80	--	--	713	--	80	16	133	--[j]	--[j]	0.70	--[n]	--[n]	--[n]	--[n]	--
Endosulfan I	959-98-8	mg/kg	--	480	--	--	4,277	--	480	126	1,071	--[j]	--[j]	9.6	0.074	0.63	--[j]	--[j]	0.011
Endosulfan II	33213-65-9	mg/kg	--	480	--	--	4,277	--	480	126	1,071	--[j]	--[j]	9.6	0.074	0.63	--[j]	--[j]	0.011

Table 8-2. Summary of 100-D and 100-H Operable Units Human Health, Groundwater Protection, and Surface Water Protection Soil PRGs

Analyte Name	CAS No.	Units	Hanford Site Background Concentration[a]	Human Health PRGs						PRGs Protective of Groundwater and Surface Water									
				RI/FS Soil MTCA B—Direct Contact PRG[b]	RI/FS Residential PRG[c]	RI/FS Soil Method B Inhalation PRG[d]	RI/FS Casual Recreational User—Direct Contact PRG[e,f]	RI/FS Resident Monument Worker—Direct Contact PRG[g]	DOE/RL 96-17 Residential RAG[h]	100-D 70:30/100:0 Contaminant Source Model GWP SSL[i]	100-H 70:30/100:0 Contaminant Source Model GGWP SSL[i]	100-D 70:30/100:0 Contaminant Source Model GWP PRG[i]	100-H 70:30/100:0 Contaminant Source Model GWP PRG[i]	DOE/RL 96-17 GWP RAG[h]	100-D 70:30/100:0 Contaminant Source Model SWP SSL[i]	100-H 70:30/100:0 Contaminant Source Model SWP SSL[i]	100-D 70:30/100:0 Contaminant Source Model SWP PRG[i]	100-H 70:30/100:0 Contaminant Source Model SWP PRG[i]	DOE/RL 96-17 SWP RAG[h]
Endosulfan sulfate	1031-07-8	mg/kg	--	--	--	--	--	--	480	--[n]	--[n]	--[n]	--[n]	9.6	1.2	10	--[j]	--[j]	0.011
Endrin	72-20-8	mg/kg	--	24	--	--	214	--	24	20	123	--[j]	--[j]	0.20	0.023	0.14	--[j]	--[j]	0.039
Endrin Aldehyde	7421-93-4	mg/kg	--	--	--	--	--	--	24	--[n]	--[n]	--[n]	--[n]	0.20	2.9	18	--[j]	--[j]	0.039
Endrin ketone	53494-70-5	mg/kg	--	--	--	--	--	--	24	--[n]	--[n]	--[n]	--[n]	0.20	--[n]	--[n]	--[n]	--[n]	0.039
Gamma-BHC (Lindane)	58-89-9	mg/kg	--	0.91	--	588,319	6.0	--	0.77	0.070	0.59	--[j]	--[j]	0.0067	0.017	0.14	--[j]	--[j]	0.0038
Gamma-Chlordane	5566-34-7	mg/kg	--	2.9	--	>1,000,000	19	--	2.9	--[j]	--[j]	--[j]	--[j]	0.025	--[j]	--[j]	--[j]	--[j]	0.017
Heptachlor	76-44-8	mg/kg	--	0.22	--	140,292	1.3	--	0.22	0.15	0.98	--[j]	--[j]	0.0020	0.0017	0.0040	--[j]	--[j]	0.0020
Heptachlor epoxide	1024-57-3	mg/kg	--	0.11	--	70,146	0.62	--	0.11	--[j]	--[j]	--[j]	--[j]	0.0020	--[j]	--[j]	--[j]	--[j]	0.0020
Methoxychlor	72-43-5	mg/kg	--	400	--	--	3,564	--	400	--[j]	--[j]	--[j]	--[j]	4.0	--[j]	--[j]	--[j]	--[j]	1.7
Toxaphene	8001-35-2	mg/kg	--	0.91	--	569,934	5.1	--	0.91	--[j]	--[j]	--[j]	--[j]	0.20	--[j]	--[j]	--[j]	--[j]	0.20
Polynuclear Aromatic Hydrocarbons																			
Acenaphthene	83-32-9	mg/kg	--	4,800	--	--	40,139	--	4,800	3,727	30,259	--[j]	--[j]	96	2,496	20,261	--[j]	--[j]	129
Acenaphthylene	208-96-8	mg/kg	--	--	--	--	--	--	4,800	--[n]	--[n]	--[n]	--[n]	96	--[n]	--[n]	--[n]	--[n]	129
Anthracene	120-12-7	mg/kg	--	24,000	--	--	200,696	--	24,000	--[j]	--[j]	--[j]	--[j]	240	--[j]	--[j]	--[j]	--[j]	1,920
Benzo(a)anthracene	56-55-3	mg/kg	--	1.4	--	>1,000,000	1.7	--	1.4	--[j]	--[j]	--[j]	--[j]	0.015	--[j]	--[j]	--[j]	--[j]	0.015
Benzo(a)pyrene	50-32-8	mg/kg	--	0.14	--	165,799	0.17	--	0.14	--[j]	--[j]	--[j]	--[j]	0.015	--[j]	--[j]	--[j]	--[j]	0.015
Benzo(b)fluoranthene	205-99-2	mg/kg	--	1.4	--	>1,000,000	1.7	--	1.4	--[j]	--[j]	--[j]	--[j]	0.015	--[j]	--[j]	--[j]	--[j]	0.015
Benzo(ghi)perylene	191-24-2	mg/kg	--	--	--	--	--	--	2,400	--[n]	--[n]	--[n]	--[n]	48	--[n]	--[n]	--[n]	--[n]	192
Benzo(k)fluoranthene	207-08-9	mg/kg	--	1.4	--	>1,000,000	1.7	--	1.4	--[j]	--[j]	--[j]	--[j]	0.015	--[j]	--[j]	--[j]	--[j]	0.015
Chrysene	218-01-9	mg/kg	--	14	--	>1,000,000	17	--	14	--[j]	--[j]	--[j]	--[j]	0.12	--[j]	--[j]	--[j]	--[j]	0.10
Dibenz[a,h]anthracene	53-70-3	mg/kg	--	1.4	--	>1,000,000	1.7	--	1.4	--[j]	--[j]	--[j]	--[j]	0.030	--[j]	--[j]	--[j]	--[j]	0.030
Fluoranthene	206-44-0	mg/kg	--	3,200	--	--	26,760	--	3,200	--[j]	--[j]	--[j]	--[j]	64	--[j]	--[j]	--[j]	--[j]	18
Fluorene	86-73-7	mg/kg	--	3,200	--	--	26,760	--	3,200	3,218	26,141	--[j]	--[j]	64	5,532	44,930	--[j]	--[j]	260
Indeno(1,2,3-cd)pyrene	193-39-5	mg/kg	--	1.4	--	>1,000,000	1.7	--	1.4	--[j]	--[j]	--[j]	--[j]	0.33	--[j]	--[j]	--[j]	--[j]	0.33
Naphthalene	91-20-3	mg/kg	--	1,600	--	1.4	62	--	1,600	125	1,056	--[j]	--[j]	16	3,867	32,599	--[j]	--[j]	988
Phenanthrene	85-01-8	mg/kg	--	--	--	--	--	--	24,000	--[n]	--[n]	--[n]	--[n]	240	--[n]	--[n]	--[n]	--[n]	1,920
Pyrene	129-00-0	mg/kg	--	2,400	--	--	20,070	--	2,400	--[j]	--[j]	--[j]	--[j]	48	--[j]	--[j]	--[j]	--[j]	192

Table 8-2. Summary of 100-D and 100-H Operable Units Human Health, Groundwater Protection, and Surface Water Protection Soil PRGs

Analyte Name	CAS No.	Units	Hanford Site Background Concentration[a]	Human Health PRGs						PRGs Protective of Groundwater and Surface Water									
				RI/FS Soil MTCA B—Direct Contact PRG[b]	RI/FS Residential PRG[c]	RI/FS Soil Method B Inhalation PRG[d]	RI/FS Casual Recreational User—Direct Contact PRG[e,f]	RI/FS Resident Monument Worker—Direct Contact PRG[g]	DOE/RL 96-17 Residential RAG[h]	100-D 70:30/100:0 Contaminant Source Model GWP SSL[i]	100-H 70:30/100:0 Contaminant Source Model GGWP SSL[i]	100-D 70:30/100:0 Contaminant Source Model GWP PRG[i]	100-H 70:30/100:0 Contaminant Source Model GWP PRG[i]	DOE/RL 96-17 GWP RAG[h]	100-D 70:30/100:0 Contaminant Source Model SWP SSL[i]	100-H 70:30/100:0 Contaminant Source Model SWP SSL[i]	100-D 70:30/100:0 Contaminant Source Model SWP PRG[i]	100-H 70:30/100:0 Contaminant Source Model SWP PRG[i]	DOE/RL 96-17 SWP RAG[h]
Volatile Organic Compounds and Semivolatile Organic Compounds																			
1,1-Dichloroethene	75-35-4	mg/kg	--	4,000	--	102	8,773	--	--	0.54	4.6	1.1	8.6	--	0.010	0.037	0.010	0.070	--
1,1,1-Trichloroethane	71-55-6	mg/kg	--	160,000	--	3,658	320,909	--	--	25	196	47	426	--	115,537	384,000	219,841	384,000	--
1,2,4-Trichlorobenzene	120-82-1	mg/kg	--	34	--	14	257	--	800	1.6	14	--[j]	--[j]	7.0	2.1	18	--[j]	--[j]	45
1,2-Dichlorobenzene	95-50-1	mg/kg	--	7,200	--	546	34,038	--	7,200	165	1,355	448	5,249	60	115	949	313	3,675	540
1,3-Dichlorobenzene	541-73-1	mg/kg	--	--	--	--	--	--	2,400	--[o]	--[o]	--[o]	--[o]	24	99	817	292	3,571	80
1,4-Dichlorobenzene	106-46-7	mg/kg	--	185	--	1.5	64	--	42	3.4	29	14	182	0.33	9.1	76	36	485	0.97
2,4,5-Trichlorophenol	95-95-4	mg/kg	--	8,000	--	--	71,289	--	8,000	832	7,037	--[j]	--[j]	80	506	4,279	--[j]	--[j]	--
2,4,6-Trichlorophenol	88-06-2	mg/kg	--	80	--	95	515	--	91	1.1	9.0	3.0	35	0.80	0.39	3.2	1.1	12	0.42
2,4-Dichlorophenol	120-83-2	mg/kg	--	240	--	--	2,139	--	240	3.1	24	5.9	54	4.8	10	78	19	172	19
2,4-Dimethylphenol	105-67-9	mg/kg	--	1,600	--	--	14,258	--	1,600	27	216	58	577	32	64	513	137	1,369	111
2,4-Dinitrophenol	51-28-5	mg/kg	--	160	--	--	1,426	--	160	1.2	11	2.5	18	3.2	2.5	23	5.3	39	14
2,4-Dinitrotoluene	121-14-2	mg/kg	--	3.2	--	>1,000,000	18	--	160	0.33	0.33	0.33	0.46	3.2	0.33	0.33	0.33	0.33	0.33
2,6-Dinitrotoluene	606-20-2	mg/kg	--	80	--	--	714	--	80	1.3	11	2.5	21	1.6	9.5	79	18	151	136
2-Butanone	78-93-3	mg/kg	--	48,000	--	28,673	464,234	--	--	191	1,678	393	2,878	--	19,606	172,102	40,349	295,164	--
2-Chlorophenol	95-57-8	mg/kg	--	400	--	--	4,563	--	400	11	92	31	368	4.0	23	187	63	745	19
2-Chloronaphthalene	91-58-7	mg/kg	--	6,400	--	--	73,000	--	6,400	1,219	10,358	--[j]	--[j]	64	1,905	16,185	--[j]	--[j]	206
2-Hexanone	591-78-6	mg/kg	--	400	--	160	3,599	--	--	1.9	16	3.7	28	--	159	1,386	321	2,401	--
2-Methylnaphthalene	91-57-6	mg/kg	--	320	--	--	2,676	--	320	61	518	--[j]	--[j]	3.2	265	2,247	--[j]	--[j]	--
2-Methylphenol (cresol, o-)	95-48-7	mg/kg	--	4,000	--	52,024	35,645	--	4,000	124	1,021	366	4,464	80	3,745	30,937	11,074	135,213	--
2-Nitroaniline	88-74-4	mg/kg	--	800	--	>1,000,000	7,129	--	240	11	94	22	174	2.4	182	1,530	353	2,824	--
2-Nitrophenol	88-75-5	mg/kg	--	--	--	--	--	--	--	--[n]	--[n]	--[n]	--[n]	--	--[n]	--[n]	--[n]	--[n]	--
3,3'-Dichlorobenzidine	91-94-1	mg/kg	--	2.2	--	536,409	13	--	2.2	0.33	0.80	0.44	6.2	0.33	0.33	0.33	0.33	0.67	0.33
3-Nitroaniline	99-09-2	mg/kg	--	24	--	>1,000,000	214	--	24	0.33	2.4	0.56	4.5	0.33	7.1	60	14	111	--
4-Bromophenylphenyl ether	101-55-3	mg/kg	--	--	--	--	--	--	--	--[n]	--[n]	--[n]	--[n]	--	--[n]	--[n]	--[n]	--[n]	--
4-Chloro-3-methylphenol	59-50-7	mg/kg	--	8,000	--	--	71,289	--	4,000	553	4,581	1,805	22,927	80	10,534	87,328	34,409	384,000	--
4-Chloroaniline	106-47-8	mg/kg	--	5.0	--	--	28	--	320	0.33	0.33	0.33	0.33	6.4	0.38	3.2	0.74	6.1	--
4-Chlorophenylphenyl ether	7005-72-3	mg/kg	--	--	--	--	--	--	--	--[n]	--[n]	--[n]	--[n]	--	--[n]	--[n]	--[n]	--[n]	--
4-Methyl-2-pentanone	108-10-1	mg/kg	--	6,400	--	13,103	69,370	--	--	29	251	58	434	--	2,739	23,957	5,553	41,338	--

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Analyte Name	CAS No.	Units	Hanford Site Background Concentration[a]	Human Health PRGs						PRGs Protective of Groundwater and Surface Water									
				RI/FS Soil MTCA B—Direct Contact PRG[b]	RI/FS Residential PRG[c]	RI/FS Soil Method B Inhalation PRG[d]	RI/FS Casual Recreational User—Direct Contact PRG[e,f]	RI/FS Resident Monument Worker—Direct Contact PRG[g]	DOE/RL 96-17 Residential RAG[h]	100-D 70:30/100:0 Contaminant Source Model GWP SSL[i]	100-H 70:30/100:0 Contaminant Source Model GGWP SSL[i]	100-D 70:30/100:0 Contaminant Source Model GWP PRG[i]	100-H 70:30/100:0 Contaminant Source Model GWP PRG[i]	DOE/RL 96-17 GWP RAG[h]	100-D 70:30/100:0 Contaminant Source Model SWP SSL[i]	100-H 70:30/100:0 Contaminant Source Model SWP SSL[i]	100-D 70:30/100:0 Contaminant Source Model SWP PRG[i]	100-H 70:30/100:0 Contaminant Source Model SWP PRG[i]	DOE/RL 96-17 SWP RAG[h]
4-Methylphenol (cresol, p-)	106-44-5	mg/kg	--	400	--	55,967	3,564	--	400	12	102	37	446	8.0	453	3,740	1,339	16,348	--
4-Nitroaniline	100-01-6	mg/kg	--	50	--	>1,000,000	283	--	48	0.33	2.6	0.59	4.7	0.33	5.6	47	11	87	--
4-Nitrophenol	100-02-7	mg/kg	--	640	--	--	5,703	--	640	30	238	73	811	13	931	7,509	2,300	25,573	1,254
4,6-Dinitro-2-methylphenol	534-52-1	mg/kg	--	8.0	--	--	71	--	8.0	0.53	4.4	2.1	28	0.33	5.4	45	21	280	--
Acetone	67-64-1	mg/kg	--	72,000	--	189,926	789,195	--	72,000	269	2,379	561	4,069	720	27,598	243,946	57,513	384,000	--
Benzene	71-43-2	mg/kg	--	18	--	0.57	22	--	--	0.060	0.51	0.12	0.96	--	0.091	0.77	0.18	1.4	--
Bis(2-chloro-1-methylethyl)ether	108-60-1	mg/kg	--	14	--	4.8	71	--	14	0.33	0.33	0.33	0.59	0.33	2.3	20	4.5	36	7.5
Bis(2-Chloroethoxy)methane	111-91-1	mg/kg	--	240	--	--	2,139	--	0.91	1.9	16	3.8	28	0.33	91	799	188	1,368	0.33
Bis(2-chloroethyl) ether	111-44-4	mg/kg	--	0.91	--	0.27	4.4	--	0.91	0.33	0.33	0.33	0.33	0.33	0.33	0.33	0.33	0.33	0.33
Bis(2-ethylhexyl) phthalate	117-81-7	mg/kg	--	71	--	>1,000,000	405	--	71	--[j]	--[j]	--[j]	--[j]	0.60	--[j]	--[j]	--[j]	--[j]	0.36
Butylbenzylphthalate	85-68-7	mg/kg	--	526	--	--	2,981	--	16,000	882	3,653	--[j]	--[j]	320	158	654	--[j]	--[j]	250
Carbazole	86-74-8	mg/kg	--	50	--	--	283	--	50	9.5	80	--[j]	--[j]	0.44	--[n]	--[n]	--[n]	--[n]	--
Carbon tetrachloride	56-23-5	mg/kg	--	14	--	0.24	9.2	--	7.7	0.045	0.35	0.085	0.78	0.034	0.030	0.24	0.058	0.53	0.050
Chloroform	67-66-3	mg/kg	--	32	--	0.24	11	--	--	0.099	0.83	0.19	1.5	--	0.40	3.4	0.78	6.2	--
Dibenzofuran	132-64-9	mg/kg	--	80	--	--	913	--	160	177	1,039	--[j]	--[j]	3.2	19	111	--[j]	--[j]	--
Diethylphthalate	84-66-2	mg/kg	--	64,000	--	--	570,313	--	64,000	1,132	9,464	2,208	18,544	1,280	1,503	12,569	2,932	24,628	4,600
Dimethyl phthalate	131-11-3	mg/kg	--	--	--	--	--	--	80,000	--[n]	--[n]	--[n]	--[n]	1,600	15,340	130,554	30,140	232,149	14,400
Di-n-butylphthalate	84-74-2	mg/kg	--	8,000	--	--	71,289	--	8,000	1,634	13,817	--[j]	--[j]	160	2,043	17,272	--[j]	--[j]	540
Di-n-octylphthalate	117-84-0	mg/kg	--	--	--	--	--	--	1,600	--[n]	--[n]	--[n]	--[n]	32	--[n]	--[n]	--[n]	--[n]	--
Ethylbenzene	100-41-4	mg/kg	--	91	--	2.3	90	--	--	0.64	5.3	1.4	14	--	2.5	21	5.5	54	--
Ethylene glycol	107-21-1	mg/kg	--	160,000	--	>1,000,000	>1,000,000	--	160,000	596	5,320	1,253	9,104	320	61,066	384,000	128,473	384,000	--
Hexachlorobenzene	118-74-1	mg/kg	--	0.63	--	396,476	3.5	--	0.63	--[j]	--[j]	--[j]	--[j]	0.33	--[j]	--[j]	--[j]	--[j]	0.33
Hexachlorobutadiene	87-68-3	mg/kg	--	13	--	>1,000,000	73	--	13	--[j]	--[j]	--[j]	--[j]	0.33	--[j]	--[j]	--[j]	--[j]	0.33
Hexachlorocyclopentadiene	77-47-4	mg/kg	--	480	--	>1,000,000	4,277	--	480	--[j]	--[j]	--[j]	--[j]	5.0	--[j]	--[j]	--[j]	--[j]	48
Hexachloroethane	67-72-1	mg/kg	--	71	--	>1,000,000	405	--	71	3.6	31	--[j]	--[j]	0.31	1.6	14	--[j]	--[j]	0.38
Isophorone	78-59-1	mg/kg	--	1,053	--	50,482	5,962	--	1,050	3.1	26	6.0	47	9.2	0.56	4.7	1.1	8.6	1.7
Nitrobenzene	98-95-3	mg/kg	--	160	--	2.0	91	--	160	1.8	14	3.4	30	1.6	1.9	15	3.6	32	3.4
n-Nitrosodi-n-dipropylamine	621-64-7	mg/kg	--	0.14	--	91,190	0.81	--	0.33	0.33	0.33	0.33	0.33	0.33	0.33	0.33	0.33	0.33	0.33
n-Nitrosodiphenylamine	86-30-6	mg/kg	--	204	--	>1,000,000	1,156	--	204	15	127	--[j]	--[j]	1.8	2.8	24	--[j]	--[j]	1.9

Table 8-2. Summary of 100-D and 100-H Operable Units Human Health, Groundwater Protection, and Surface Water Protection Soil PRGs

Analyte Name	CAS No.	Units	Hanford Site Background Concentration[a]	Human Health PRGs						PRGs Protective of Groundwater and Surface Water									
				RI/FS Soil MTCA B—Direct Contact PRG[b]	RI/FS Residential PRG[c]	RI/FS Soil Method B Inhalation PRG[d]	RI/FS Casual Recreational User—Direct Contact PRG[e,f]	RI/FS Resident Monument Worker—Direct Contact PRG[g]	DOE/RL 96-17 Residential RAG[h]	100-D 70:30/100:0 Contaminant Source Model GWP SSL[i]	100-H 70:30/100:0 Contaminant Source Model GGWP SSL[i]	100-D 70:30/100:0 Contaminant Source Model GWP PRG[i]	100-H 70:30/100:0 Contaminant Source Model GWP PRG[i]	DOE/RL 96-17 GWP RAG[h]	100-D 70:30/100:0 Contaminant Source Model SWP SSL[i]	100-H 70:30/100:0 Contaminant Source Model SWP SSL[i]	100-D 70:30/100:0 Contaminant Source Model SWP PRG[i]	100-H 70:30/100:0 Contaminant Source Model SWP PRG[i]	DOE/RL 96-17 SWP RAG[h]
Methylene chloride	75-09-2	mg/kg	--	133	--	11	337	--	133	0.21	1.9	0.44	3.3	0.50	0.20	1.7	0.41	3.0	0.94
Pentachlorophenol	87-86-5	mg/kg	--	8.3	--	>1,000,000	35	--	8.3	0.33	2.5	1.1	15	0.33	0.33	0.92	0.42	5.6	0.33
Phenol	108-95-2	mg/kg	--	24,000	--	11,614	213,867	--	24,000	132	1,127	260	1,995	480	551	4,696	1,085	8,312	4,200
Tetrachloroethene	127-18-4	mg/kg	--	1.9	--	0.88	10	--	--	0.017	0.14	0.038	0.40	--	0.080	0.65	0.18	1.9	--
trans-1,2-Dichloroethylene	156-60-5	mg/kg	--	1,600	--	38	3,332	--	--	6.1	52	12	93	--	8.5	73	17	131	--
Tributyl Phosphate	126-73-8	mg/kg	--	109	--	--	616	--	185	12	98	--[j]	--[j]	3.3	30	251	--[j]	--[j]	--
Toluene	108-88-3	mg/kg	--	6,400	--	4,774	63,832	--	6,400	80	627	152	1,362	64	162	1,273	309	2,767	1,360
Trichloroethene	79-01-6	mg/kg	--	11	--	0.17	7.2	--	--	0.047	0.39	0.092	0.79	--	0.24	2.0	0.47	4.0	--
Vinyl Chloride	75-01-4	mg/kg	--	1.4	--	0.52	0.07	--	--	0.005	0.026	0.006	0.045	--	0.005	0.011	0.005	0.019	--
Xylenes (total)	1330-20-7	mg/kg	--	16,000	--	103	10,346	--	16,000	290	2,338	627	6,370	160	--[n]	--[n]	--[n]	--[n]	--
Other Organics																			
Total Petroleum Hydrocarbons–Diesel Range	TPHDIESEL	mg/kg	--	2,000	--	--	--	--	200	2,000[o]	2,000[o]	2,000[o]	2,000[o]	200	--[n]	--[n]	--[n]	--[n]	200
Total Petroleum Hydrocarbons–Motor Oil (high boiling)	TPH/OILH	mg/kg	--	2,000	--	--	--	--	200	2,000[o]	2,000[o]	2,000[o]	2,000[o]	200	--[n]	--[n]	--[n]	--[n]	200

Notes:

Yellow highlighting denotes the PRG selection for protection of human health. For radionuclide human health PRGs, the lowest of the PRGs calculated for the residential exposure scenario and the residential Interim Action ROD RAG defined in the 100 Area RDR/RAWP (DOE/RL-96-17) is highlighted in yellow. For nonradionuclide PRGs, the PRG is the RI/FS MTCA Method B value, except for arsenic and lead which set to the "Soil Cleanup Levels for Unrestricted Land Uses" [WAC 173-340-900], Table 740-1, "Method A Soil Cleanup Levels for Unrestricted Land Uses."

Green highlighting denotes the most conservative PRG above background for each analyte for GW/SW protection except for Cr(VI), which is compared to the interim action RAG of 2.0 mg/kg.

a. Hanford Site background values for nonradionuclides: DOE/RL-92-24, Vol. 1, *Hanford Site Background: Part 1, Soil Background for Nonradioactive Analytes*, ECF-HANFORD-11-0038, "Soil Background Data for Interim Use at the Hanford Site; Hanford Site Background Values for Radionuclides"; DOE/RL-96-12, *Hanford Site Background: Part 2, Soil Background for Radionuclides*.

b. ECF-HANFORD-10-0444, "Documentation of Standard Method B Contact Cleanup Levels for Unrestricted Land Use"; PRGs for arsenic, lead, and total petroleum hydrocarbons are based on WAC 173-340-900, Table 740-1, "Method A Soil Cleanup Levels for Unrestricted Land Uses."

c. ECF-HANFORD-10-0429, "Documentation of Preliminary Remediation Goals (PRGs) for Radionuclides Using the IAROD Exposure Scenario for the 100 and 300 Area Remedial Investigation/Feasibility Study (RI/FS) Report."

d. ECF-HANFORD-11-0033, "Calculation of Inhalation Pathway Preliminary Remediation Goals Using Standard Method B Air Cleanup Levels for the 100 Areas and 300 Area Remedial Investigation/Feasibility Study Reports."

e. ECF-HANFORD-10-0445, "Calculation of Nonradionuclides Preliminary Remediation Goals in Soil for a Casual Recreational User Scenario for the 100 Areas and 300 Area Remedial Investigation/Feasibility Study Reports."

f. ECF-HANFORD-10-0446, "Calculation of Radiological Preliminary Remediation Goals in Soil for a Casual Recreational User Scenario for the 100 Areas and 300 Area Remedial Investigation/Feasibility Study Reports."

g. ECF-HANFORD-11-0142, "Calculation of Radiological Preliminary Remediation Goals in Soil for a Resident Monument Worker Scenario for the 100 Areas and 300 Area Remedial Investigation/Feasibility Study Reports."

h. DOE/RL 96-17, *Remedial Design Report/Remedial Action Work Plan for the 100 Area*.

i. ECF-HANFORD-11-0063, "STOMP 1-D Modeling for Determination of Preliminary Remediation Goals for 100 Area Source Operable Units D, H, and K." A 70:30 initial source distribution is used for analytes with $K_d \geq 2$ mL/g; a 100:0 initial source distribution is used for analytes with $K_d < 2$ mL/g. The SSL and PRG value for all analytes defaults to the EQL when the calculated value is less than the EQL. EQL values are obtained from the Sampling and Analysis Plans for 100-K (DOE/RL-2009-41) and 100-D/H (DOE/RL-2009-40).

Table 8-2. Summary of 100-D and 100-H Operable Units Human Health, Groundwater Protection, and Surface Water Protection Soil PRGs

Analyte Name	CAS No.	Units	Hanford Site Background Concentration[a]	Human Health PRGs						PRGs Protective of Groundwater and Surface Water										
				RI/FS Soil MTCA B—Direct Contact PRG[b]	RI/FS Residential PRG[c]	RI/FS Soil Method B Inhalation PRG[d]	RI/FS Casual Recreational User—Direct Contact PRG[e,f]	RI/FS Resident Monument Worker—Direct Contact PRG[g]	DOE/RL 96-17 Residential RAG[h]	100-D 70:30/100:0 Contaminant Source Model GWP SSL[i]	100-H 70:30/100:0 Contaminant Source Model GGWP SSL[i]	100-D 70:30/100:0 Contaminant Source Model GWP PRG[i]	100-H 70:30/100:0 Contaminant Source Model GWP PRG[i]	DOE/RL 96-17 GWP RAG[h]	100-D 70:30/100:0 Contaminant Source Model SWP SSL[i]	100-H 70:30/100:0 Contaminant Source Model SWP SSL[i]	100-D 70:30/100:0 Contaminant Source Model SWP PRG[i]	100-H 70:30/100:0 Contaminant Source Model SWP PRG[i]	DOE/RL 96-17 SWP RAG[h]

j. The SSL or PRG value for groundwater protection or surface water protection is considered non-representative because there is no breakthrough of the analyte simulated within 1,000 years for the majority of soil columns (breakthrough is defined as concentrations above 1E-04 µg/L or 1E-04 pCi/L).

k. A SSL is calculated for total uranium (CAS # 7440-61-1) but not isotopic uranium because an MCL is not available for isotopic uranium. When total uranium analytical results (µg/kg) are available, EPCs are compared to the total uranium SSL. When only isotopic uranium results (pCi/g) are available, uranium is addressed by converting the isotopic uranium from activity-based (pCi/g) to mass-based (µg/kg) concentrations and summing to provide a mass-based total uranium EPC (identified as Total_U_Isotopes), as described in ECF-100DR1-11-0004, "Computation of Exposure Point Concentrations for the 100-DR-1, 100-DR-2, 100-HR-1 and 100-HR-2 Source Operable Units." The Total_U_Isotopes EPC is then compared to the total uranium SSL.

l. The SSL and PRG values for Cr(VI)) are set to a maximum value of 6.0 mg/kg because the K_d value used in the model was derived from experiments with soil concentrations less than 6 mg/kg.

m. Value converted from "as nitrogen" values in DOE/RL-96-17 using the following conversion factors as applicable: 4.43 g NO_3^-/g N and 3.29 g NO_2^-/ g N.

n. A groundwater protection or surface water protection SSL or PRG is not calculated because a groundwater or surface water cleanup level or MCL is not available for the analyte.

o. The SSL for total petroleum hydrocarbons is a default screening level obtained from WAC 173-340-900, Table 747-5, "Residual Saturation Screening Levels for TPH."

Table 8-3. Summary of 100-D and 100-H Operable Units Ecological Soil PRGs

Contaminant	Hanford Site Background Concentration[a]	Preliminary Remediation Goals for Protection of Ecological Receptors			
		Plant PRG[b]	Invertebrate PRG[b]	Avian PRG[b]	Mammal PRG[b]
Radionuclides (pCi/g)					
Americium-241	--	21,500	--[c]	11,900	4,840
Carbon-14	--	60,700	--[c]	50	32
Cesium-137	1.1	2,210	--[c]	1,430	924
Cobalt-60	0.0084	6,130	--[c]	805	805
Curium-243	--	--[c]	--[c]	--[c]	--[c]
Europium-152	--	14,700	--[c]	1,740	1,740
Europium-154	0.033	12,500	--[c]	1,610	1,610
Europium-155	0.054	153,000	--[c]	33,400	33,400
Iodine-129	--	--[c]	--[c]	--[c]	--[c]
Neptunium-237	--	8,150	--[c]	7,880	7,880
Nickel-63	--	--[c]	--[c]	--[c]	--[c]
Niobium-94	--	--[c]	--[c]	--[c]	--[c]
Plutonium-238	0.0038	17,500	--[c]	20,900	5,980
Plutonium-239/240	0.025	12,700	--[c]	22,300	6,270
Strontium-90	0.18	3,580	--[c]	112	91
Technetium-99	--	21,900	--[c]	5,360	8,670
Tritium	--	1,680,000	--[c]	936	420
Uranium-233/234	1.1	51,600	--[c]	6,370	14,200
Uranium-235	0.11	27,400	--[c]	4,360	8,060
Uranium-238	1.1	15,700	--[c]	5,150	11,000
Total Uranium (summed)	--	--[c]	--[c]	--[c]	--[c]
Chemicals (mg/kg)					
Aluminum	11,800	--[d]	--[d]	--[d]	--[d]
Antimony	0.13	842	842	--[c]	92
Arsenic	6.5	128	128	2,284	127
Barium	132	500	358	1,690	2,270
Beryllium	1.5	10	40	--[c]	14
Bismuth	--	--[c]	--[c]	--[c]	--[c]

Table 8-3. Summary of 100-D and 100-H Operable Units Ecological Soil PRGs

Contaminant	Hanford Site Background Concentration[a]	Preliminary Remediation Goals for Protection of Ecological Receptors			
		Plant PRG[b]	Invertebrate PRG[b]	Avian PRG[b]	Mammal PRG[b]
Boron	3.9	30	29	54	32
Cadmium	0.56	9.8	20	29	624
Chromium	19	259	149	109	517
Cobalt	16	16	16	484	2,140
Copper	22	70	58	213	193
Cr(VI)	--	--[c]	--[c]	--[c]	1,250
Lead	10	9,090	1,700	156	1,580
Lithium	13	2.0	--[c]	--[c]	1,664
Manganese	512	1,260	1,260	14,400	3,320
Mercury	0.013	0.30	13	2.0	1.6
Molybdenum	0.47	2.0	28	95	5.7
Nickel	19	38	280	361	247
Selenium	0.78	2.0	4.1	2.4	1.4
Silver	0.17	560	3.0	983	9,810
Strontium	--	--[c]	--[c]	--[c]	1,210
Thallium	0.19	1.0	0.46	--[c]	6.2
Tin	--	838	838	204	279
Total_U_Isotopes	3.2	250	100	82	40
Uranium (soluble salts)	3.2	250	100	82	40
Vanadium	85	89	116	43	260
Zinc	68	621	8,980	856	1,040
Chloride	100	--[c]	--[c]	--[c]	--[c]
Cyanide	--	--[c]	--[c]	--[c]	20,700
Fluoride	2.8	--	--	2,280	13,800
Nitrate	52	--	--[c]	--[c]	340,000
Nitrite	--	--	--[c]	--[c]	340,000
Sulfate	237	--	--[c]	--[c]	--[c]
Aroclor 1016	--	40	--[c]	1.8	4.9
Aroclor 1221	--	40	--[c]	1.8	1.5

Table 8-3. Summary of 100-D and 100-H Operable Units Ecological Soil PRGs

Contaminant	Hanford Site Background Concentration[a]	Preliminary Remediation Goals for Protection of Ecological Receptors			
		Plant PRG[b]	Invertebrate PRG[b]	Avian PRG[b]	Mammal PRG[b]
Aroclor 1232	--	40	--[c]	1.8	1.4
Aroclor 1242	--	40	--[c]	1.8	1.5
Aroclor 1248	--	40	--[c]	1.8	0.33
Aroclor 1254	--	40	--[c]	1.8	1.5
Aroclor 1260	--	40	--[c]	1.8	1.5
4,4'-DDE (Dichlorodiphenyldichloroethylene)	--	--[c]	--[c]	0.80	0.40
4,4'-DDT (Dichlorodiphenyltrichloroethane)	--	--[c]	--[c]	1.2	0.88
Aldrin	--	--[c]	--[c]	0.16	9.8
Alpha-Chlordane	--	--[c]	1.0	50	204
Beta-BHC	--	--[c]	--[c]	4.1	8.7
Dieldrin	--	--[c]	--[c]	0.079	0.021
Endosulfan I	--	--[c]	--[c]	41	0.71
Endosulfan II	--	--[c]	--[c]	41	0.71
Endosulfan sulfate	--	--[c]	--[c]	41	0.56
Endrin Aldehyde	--	--[c]	--[c]	0.23	1.4
Gamma-Chlordane	--	--[c]	1.0	50	204
Methoxychlor	--	--[c]	--[c]	--[c]	22
Acenaphthene	--	20	29	1,100	2,420
Acenaphthylene	--	--[c]	29	74	156
Anthracene	--	--[c]	29	678	4,210
Benzo(a)anthracene	--	--[c]	18	2.0	64
Benzo(a)pyrene	--	--[c]	18	2.4	76
Benzo(b)fluoranthene	--	--[c]	18	1.3	39
Benzo(ghi)perylene	--	--[c]	18	1.1	32
Benzo(k)fluoranthene	--	--[c]	18	1.3	39
Chrysene	--	--[c]	18	1.4	45
Dibenz[a,h]anthracene	--	--[c]	18	1.4	44
Fluoranthene	--	--[c]	18	1.1	839

Table 8-3. Summary of 100-D and 100-H Operable Units Ecological Soil PRGs

Contaminant	Hanford Site Background Concentration[a]	Preliminary Remediation Goals for Protection of Ecological Receptors			
		Plant PRG[b]	Invertebrate PRG[b]	Avian PRG[b]	Mammal PRG[b]
Fluorene	--	--[c]	29	175	267
Indeno(1,2,3-cd)pyrene	--	--[c]	18	1.2	36
Naphthalene	--	--[c]	29	340	100
Phenanthrene	--	--[c]	29	943	5,920
Pyrene	--	--[c]	18	1.9	600
1,1-Dichloroethene	--	--[c]	--[c]	165	301
1,1,1-Trichloroethane	--	--[c]	--[c]	165	10,016
1,2,4-Trichlorobenzene	--	--[c]	--[c]	--	--
1,2-Dichlorobenzene	--	--[c]	--[c]	164	282
1,3-Dichlorobenzene	--	--[c]	--[c]	164	310
2,4-Dinitrotoluene	--	--[c]	--[c]	26	28
2-Butanone	--	--[c]	--[c]	3,123	412,224
2-Hexanone	--	--[c]	--[c]	1,856	1,708
2-Methylnaphthalene	--	--[c]	29	8.4	6.0
2-Methylphenol (cresol, o-)	--	--[c]	--[c]	--[c]	9,290
4-Chlorophenylphenyl ether	--	--[c]	--[c]	--[c]	--[c]
4-Methyl-2-pentanone	--	--[c]	--[c]	1,927	227,119
4-Methylphenol (cresol, p-)	--	--[c]	--[c]	--[c]	9,360
Benzene	--	--[c]	--[c]	195	70
Bis(2-ethylhexyl) phthalate	--	100	--[c]	0.14	45
Carbon tetrachloride	--	--[c]	--[c]	165	160
Chloroform	--	--[c]	--[c]	165	412
Diethylphthalate	--	100	--[c]	--[c]	--[c]
Ethylbenzene	--	--[c]	--[c]	159	1,027
Methylene chloride	--	--[c]	--[c]	166	504
Phenol	--	70	30	--[c]	1,510
Tetrachloroethene	--	--[c]	--[c]	164	70
trans-1,2-Dichloroethylene	--	--[c]	--[c]	165	453
Toluene	--	200	--[c]	195	5,200

Table 8-3. Summary of 100-D and 100-H Operable Units Ecological Soil PRGs

Contaminant	Hanford Site Background Concentration[a]	Preliminary Remediation Goals for Protection of Ecological Receptors			
		Plant PRG[b]	Invertebrate PRG[b]	Avian PRG[b]	Mammal PRG[b]
Trichloroethene	--	--[c]	--[c]	165	70
Xylenes (total)	--	--[c]	--[c]	149	826
Total Petroleum Hydrocarbons–Diesel Range	--	--[c]	200	356,000	452,000

Note: The need for remedial action to protect ecological receptors will be based on population and community level effects. Exceedance of ecological PRGs initiates a scientific management decision point to determine a basis for action.

a. Hanford Site background values for nonradionuclides: DOE/RL-92-24, *Hanford Site Background: Part 1, Soil Background for Nonradioactive Analytes*; ECF-HANFORD-11-0038, "Soil Background Data for Interim Use at the Hanford Site"; DOE/RL-2010-95, *Remedial Investigation/Feasibility Study for the 100-DR-1, 100-DR-2, 100-HR-1, 100-HR-2,and 100-HR-3 Operable Units*; Hanford Site background values for radionuclides: DOE/RL-96-12, *Hanford Site Background: Part 2, Soil Background for Radionuclides*.

b. CHPRC-00784, *Tier 1 Risk-Based Soil Concentrations Protective of Ecological Receptors at the Hanford Site*; CHPRC-01311, *Tier 2 Risk-Based Soil Concentrations Protective of Ecological Receptors at the Hanford Site*; ECF-HANFORD-11-0158, "Tier 2 Terrestrial Plant and Invertebrate Preliminary Remediation Goals (PRGs) for Nonradionuclides for Use at the Hanford Site." These PRGs will be used within the SMDP process described in Chapter 7 of this RI/FS.

c. A PRG is not calculated because a toxicity value is not available for this receptor or analyte.

d. Aluminum ecotoxicity is only identified in soils with pH less than 5.5 (OSWER Directive 9285.7-60). Most soil pH measures at the Hanford Site are greater than 5.5 (DOE/Rl-2007-21; DOE/RL-2007-50; ECF-HANFORD-11-0158). Thus, aluminum at the Hanford Site does not present an ecological risk.

PRG = preliminary remediation goal

Risk-based standards for some contaminants are calculated to be less than area background values or PQLs. Where risk-based standards are less than area background concentrations, PRGs may be set at concentrations that are equal to the agreed upon site or area background concentrations. Area background values for selected nonradioactive contaminants in soil have been characterized for the Hanford Site (*Hanford Site Background: Part 1, Soil Background for Nonradioactive Analytes* [DOE/RL-92-24]). Similarly, where risk-based standards are less than PQLs, PRGs will default to the PQLs. Therefore, the PRGs for individual nonradioactive contaminants in solid waste and particulate reflect the value that is greatest among risk-based standards, area background values, or PQLs.

Ecological Exposure. Ecological PRGs for the protection of plants, soil invertebrates, and wildlife (birds and mammals) are developed using a tiered approach (*Tier 1 Risk-Based Soil Concentrations Protective of Ecological Receptors at the Hanford Site* [CHPRC-00784]). The objective of a tiered approach is to refine available generic screening levels (EcoSSLs in MTCA [WAC 173-340], Table 749-3, or BCGs), as needed, with additional literature-derived or site-specific information to more realistically represent Hanford Site-specific ecological risks. The ecological PRGs are developed in Section 7.3.4 of this RI/FS.

8.1.4.3 Direct Contact Exposure PRGs for Radiological Contaminants

The PRGs for direct-contact exposure to radiological contaminants for both human and ecological receptors are described in the following subsections.

Human Exposure. PRGs for radioactive wastes and radioactively contaminated soils for human receptor direct contact exposures are based on EPA radionuclide soil cleanup guidance. As established by the NCP (40 CFR 300), CERCLA cleanup actions generally should achieve a level of risk within the 10^{-4} to 10^{-6} ELCR based on the RME for an individual.). Demonstration that the 10^{-4} to 10^{-6} residual risk-range goal has been achieved will be accomplished through final verification sampling during closeout of individual sites using the residential exposure scenario. A complete description of the activities, exposure assumptions, and risk bases associated with the residential scenario is provided in Section 6.2.3.3.1 of this RI/FS. Table 8-2 presents a summary of the remedial action goals reported in the 100 Area RDR/RAWP (DOE/RL-96-17) and the PRGs developed for the residential scenario.

Soil PRG values are developed for the direct contact and inhalation pathways, combined, using the resident Monument worker and the casual recreational user exposure scenarios. The resident Monument worker scenario and the casual recreational use are both used to represent the reasonably anticipated future land use. A complete description of the activities, exposure assumptions, and risk bases associated with resident Monument worker and casual recreational user scenarios is provided in Section 6.2.3.3.2 and Section 6.2.3.3.3, respectively, of this RI/FS. PRG values for individual radioisotopes are based on an ELCR of 1×10^{-4}. The PRG values listed in Table 8-2 for this exposure scenario are provided to aid in determining whether the cleanup actions achieve CERCLA threshold criteria.

Ecological Exposure. BCGs are proposed for use as ecological PRGs for radionuclides for terrestrial plants and animals (including soil invertebrates).A discussion of the application of BCGs to radionuclide toxicity data is presented in Section 7.3. BCGs are also evaluated at the SMDP and considers potential population impacts for decisions. Additional evaluation may be conducted where biological exposures exceed BCGs.

8.1.4.4 Soil Concentrations Protective of Groundwater and Surface Water (SSLs and PRGs)

Fate and transport modeling of contaminants in the vadose zone was conducted to assess their potential impact on groundwater or surface water. Numerical models were constructed to represent the key factors of the conceptual model for 100-D/H and simulated using the STOMP code (STOMP Subsurface Transport Over Multiple Phases Version 2.0 Theory Guide [PNNL-12030]). Modeling with STOMP was performed with a bounding representation of waste distribution and for multiple stratigraphic columns representing the range of conditions within 100-D/H. The bounding representation of waste distribution consisted of a uniform distribution through the entire vadose zone thickness beneath the backfill for contaminants with $K_d < 2$ (referred to as the 100:0 initial distribution) and a uniform distribution through the upper 70 percent of the vadose zone thickness beneath the backfill for contaminants with $K_d \geq 2$ (referred to as the 70:30 initial distribution). Constituents that were persistent (that is, do not degrade or decay in a reasonable period) and that had a peak concentration in groundwater occurring within 1,000 years in the future were evaluated in this way. This modeling process, including assumptions and inputs, is described in Section 5.4 and modeling results in Section 5.6

SSL values were developed from STOMP simulated peak groundwater concentrations obtained for an irrigation recharge scenario representing a bounding future recharge case (based on irrigated agriculture) and for criterion protective of groundwater and surface water. Those sites/contaminants that failed the screening level (based on comparison of EPCs to SSLs) were next evaluated against PRG values developed from STOMP simulated peak groundwater concentrations obtained for a native vegetation recharge scenario representative of re-establishment of the native xerophytic plant communities on the land surface (Table 8-3) and the same criterion protective of groundwater and surface water. The derivation of these protection levels is described in Section 5.7.

8.1.4.5 COC Identification Based on Groundwater and Surface Water PRGs

A groundwater risk assessment was presented in Section 6.3 of this report. The list of COCs presented in Table 8-4 was determined in Section 6.3. The process used to identify COCs is described in Section 6.3.2.3 (COPC Identification Process) and in Section 6.3.5 (Risk Characterization). Based on the results of the groundwater risk assessment, the list of COCs include chromium (total), Cr(VI), and nitrate in the 100-D Area; strontium-90, Cr(VI), and nitrate in the100-H Area; and chromium (total) and Cr(VI) in the horn area. Nitrate has a 90th percentile concentration less than the DWS in the 100-H Area but has areas of localized contamination above the DWS and therefore retained as a COC.

8.2 General Response Actions

GRAs consistent with RAOs were identified for 100-D/H. GRAs are basic actions that might be undertaken to remediate a site, and are assembled based on nature and extent of contamination, as presented in the RI. For each GRA, several possible remedial technologies may exist, which can be further divided into a number of process options. This section discusses the remedial technology selection process.

Potential remedial technologies are selected for evaluation based on their potential ability to mitigate the identified risks or achieve compliance with ARARs for the remedial action. Technologies and process options selected for evaluation are assessed with respect to their implementability, effectiveness, and relative cost in accordance with CERCLA RI/FS Guidance (EPA/540/G-89/004) and the NCP ("Remedial Investigation/Feasibility Study and Selection of Remedy" [40 CFR 300.430(e)]). The selected final remedy must comply with ARARs and protect human health and the environment.

CERCLA RI/FS Guidance (EPA/540/G-89/004) suggests development and evaluation of a range of responses, including a no action alternative, to ensure identification and selection of an appropriate remedy. The technology screening process consists of the following steps:

- Identify GRAs that may meet RAOs, either individually or in combination with other GRAs.
- Identify, screen, and evaluate remedial technology types for each GRA.
- Select representative process option(s).

Following the technology screening, representative process options are assembled into remedial alternatives (presented in Chapter 9) that are evaluated further in the detailed and comparative analyses of alternatives (presented in Chapter 10).

GRAs identified for vadose zone soils, including waste sites, in 100-D/H include the following:

- No action
- Institutional controls
- Removal, ex situ treatment, processing, and disposal
- In situ treatment
- Containment

GRAs identified for contaminated groundwater in 100-D/H include the following:

- No action
- Institutional controls
- Monitored natural attenuation (MNA)
- Pump-and-treat (collection, ex situ treatment, and discharge)

- In situ treatment
- Containment

8.2.1 Target Remediation Areas

In accordance with CERCLA RI/FS Guidance (EPA/540/G-89/004), the FS is required to determine the areas or volumes of media to which GRAs might be applied. This section summarizes the waste sites and groundwater areas that will be evaluated in the FS, based on the PRGs and findings of the HHRA, ERA, and RI presented in the preceding chapters.

8.2.1.1 Waste Sites

Table 8-5 summarizes the 343 waste sites (including subsites) evaluated through the RI/FS and groups which sites are, and are not, carried through forward into the FS as described below.

Waste Sites Not Carried Forward into the FS (Chapter 1). As discussed in Chapter 1, the determination of which areas of 100-D/H are waste sites has been performed following specific procedures defined in TPA documents (*Hanford Federal Facility Agreement and Consent Order* [Ecology et al., 1989a] and Tri-Party Agreement Action Plan [Ecology et al., 1989b]). The areas that have been suspected waste sites are summarized and tracked in the WIDS database. As information is learned about the sites, or as they are remediated and confirmation data collected, the sites are classified or reclassified, depending on their status. Of the 343 sites (including subsites) as of June 2011 in 100-D/H, 49 sites were not carried forward per discussion in Chapter 1. Forty-eight sites were classified or reclassified as "Rejected," "Not Accepted," or "Closed Out," and one site will be closed out under Washington Department of Health Regulations (Table 8-5). These sites were not considered further in this RI/FS.

In addition, the waste sites for the three reactor core safe storage enclosures, 105-D, 105-DR, and 105-H, are not discussed in the FS. The waste sites are 118-D-6:1, 118-DR-2:1, and 118-H-6:1, respectively. In September 1993, DOE issued a NEPA ROD (58 FR 48509) that established a path forward for the Hanford Site reactors. An "Amended Record of Decision for the Decommissioning of Eight Surplus Production Reactors at the Hanford Site, Richland, WA" (75 FR 43158) was issued in July 2010. The NEPA ROD provided options for immediate dismantlement for reactor decommissioning, and one-piece disposal of the reactor cores or dismantlement after an ISS period of approximately 75 years, which allowed for decay of the radionuclide(s) that presented the major risk for site workers. The three reactor buildings are currently in ISS. As detailed in Cleanup Completion Framework (DOE/RL-2009-10), the NEPA ROD (58 FR 48509) indicated DOE's intent to complete these decommissioning actions consistent with the proposed cleanup schedule for remedial actions, which includes the D, DR, and H Reactors. Until reactor removal is complete, DOE will continue to conduct routine maintenance, surveillance, and radiological monitoring activities to ensure continued protection of human health and the environment during the ISS period.

Additional information developed through the risk assessments in Chapters 5, 6, and 7 do not change the determinations from Chapter 1. Figures 8-1 and 8-2 present the location of the 100-D and 100-H waste sites, respectively, that are not evaluated in the FS. Chapter 1 presents additional information on these waste sites not carried forward.

Waste sites Identified for No Further Action. There are 146 waste sites that are identified for no further action based on the risk assessments in Chapters 5, 6, and 7, and specific alternative evaluations are not developed for these sites. Locations of the 100-D and 100-H waste sites are shown on Figures 8-3 and 8-4, respectively.

Table 8-4. 100-HR-3 Groundwater OU PRGs

Contaminant	Units	90th Percentile Concentration	Dose (mrem/yr) Based on 90th Percentile Concentration	ELCR Based on 90th Percentile Concentration	HQ Based on 90th Percentile Concentration	Drinking Water Standard	WAC 173-340 Cleanup Levels		AWQC			WAC 173-201A	40 CFR 131 Water Quality Standard		100-HR-3 Groundwater OU PRG[a]
							Noncarcinogens at HQ = 1	Carcinogens at 1×10^{-5} Risk Level	Freshwater CMC (acute)	Freshwater CCC (chronic)	Human Health Water + Organism at 1×10^{-5} Risk Level	Freshwater CCC (chronic)	Freshwater CMC (acute)	Freshwater CCC (chronic)	
100-D Source Exposure Area															
COCs—for Remedial Technology Screening and Alternative Development															
Chromium	µg/L	925	--	--	0.04	100	24,000	--	570	65	--	156	550	180	65
Cr(VI)[c]	µg/L	992	--	--	21	--	48	--	16	11	--	10	15	10	10
Nitrate[b]	µg/L	69,500	--	--	0.61	45,000	113,600	--	--	--	45,000	--	--	--	45,000
COPCs															
Antimony	µg/L	--	--	--	--	6.0	6.4	--	--	--	56	--	--	14	6.0
Cadmium	µg/L	0.2	--	--	0.025	5.0	8.0	--	2.0	0.25	--	0.91	3.9	1.0	0.25
Carbon tetrachloride	µg/L	1.0	--	2.9×10^{-6}	0.18	5.0	5.6	3.4	--	--	2.3	--	--	--	2.3
Cobalt	µg/L	1.3	--	--	0.27	--	4.8	--	--	--	--	--	--	--	4.8
Copper	µg/L	0.73	--	--	<0.01	1,300	640	--	13	9.0	1,300	--	17	11	9.0
Nickel	µg/L	9.5	--	--	0.030	100	320	--	470	52	610	137	1,400	160	52
Silver	µg/L	0.2	--	--	<0.01	100	80	--	3.2	--	--	2.6	3.4	--	2.6
100-H Source Exposure Area															
COCs—for Remedial Technology Screening and Alternative Development															
Strontium-90	pCi/L	14	7.0	1.5×10^{-5}	--	8	--	--	--	--	--	--	--	--	8
Cr(VI)[c]	µg/L	26	--	--	0.54	--	48	--	16	11	--	10	15	10	10
Nitrate[b]	µg/L	39,800	--	--	0.35	45,000	113,600	--	--	--	45,000	--	--	--	45,000
COPCs															
Antimony	µg/L	0.61	--	--	0.095	6.0	6.4	--	--	--	56	--	--	14	6.0
Cadmium	µg/L	--	--	--	--	5.0	8.0	--	2.0	0.25	--	0.91	3.9	1.0	0.25
Carbon tetrachloride	µg/L	1.0	--	2.9×10^{-6}	0.18	5.0	5.6	3.4	--	--	2.3	--	--	--	2.3
Chromium	µg/L	31	--	--	<0.01	100	24,000	--	570	65	--	156	550	180	65
Cobalt	µg/L	0.43	--	--	0.090	--	4.8	--	--	--	--	--	--	--	4.8
Copper	µg/L	1.3	--	--	<0.01	1,300	640	--	13	9.0	1,300	--	17	11	9.0
Nickel	µg/L	8.9	--	--	0.028	100	320	--	470	52	610	137	1,400	160	52
Silver	µg/L	--	--	--	--	100	80	--	3.2	--	--	2.6	3.4	--	2.6
Zinc	µg/L	16	--	--	<0.01	5,000	4,800	--	120	120	7,400	91	110	100	91

Table 8-4. 100-HR-3 Groundwater OU PRGs

Contaminant	Units	90th Percentile Concentration	Dose (mrem/yr) Based on 90th Percentile Concentration	ELCR Based on 90th Percentile Concentration	HQ Based on 90th Percentile Concentration	Drinking Water Standard	WAC 173-340 Cleanup Levels		AWQC			WAC 173-201A	40 CFR 131 Water Quality Standard		100-HR-3 Groundwater OU PRG[a]
							Noncarcinogens at HQ = 1	Carcinogens at 1×10^{-5} Risk Level	Freshwater CMC (acute)	Freshwater CCC (chronic)	Human Health Water + Organism at 1×10^{-5} Risk Level	Freshwater CCC (chronic)	Freshwater CMC (acute)	Freshwater CCC (chronic)	
Horn Area Exposure Area															
COCs—for Remedial Technology Screening and Alternative Development															
Chromium	µg/L	76	--	--	<0.01	100	24,000	--	570	65	--	156	550	180	65
Cr(VI)[c]	µg/L	71	--	--	1.5	--	48	--	16	11	--	10	15	10	10
COPCs															
Antimony	µg/L	0.74	--	--	0.12	6.0	6.4	--	--	--	56	--	--	14	6.0
Cadmium	µg/L	--	--	--	--	5.0	8.0	--	2.0	0.25	--	0.91	3.9	1.0	0.25
Cobalt	µg/L	0.1	--	--	0.021	--	4.8	--	--	--	--	--	--	--	4.8
Copper	µg/L	1.4	--	--	<0.01	1,300	640	--	13	9.0	1,300	--	17	11	9.0
Nickel	µg/L	4.9	--	--	0.015	100	320	--	470	52	610	137	1,400	160	52
Silver	µg/L	0.2	--	--	<0.01	100	80	--	3.2	--	--	2.6	3.4	--	2.6
Zinc	µg/L	12	--	--	<0.01	5,000	4,800	--	120	120	7,400	91	110	100	91

Sources:
"Water Quality Standards" (40 CFR 131).
"Water Quality Standards for Surface Waters of the State of Washington" (WAC 173-201A).
"Model Toxics Control Act—Cleanup," "Groundwater Cleanup Standards" (WAC 173-340-720).
Notes: DWSs from "National Primary Drinking Water Regulations" (40 CFR 141).
a. The final cleanup levels achieved at the conclusion of the remedial action will correspond to a cumulative ELCR less than 1×10^{-5} and HI of less than 1.
b. Nitrate may be expressed as nitrate (NO_3) or as nitrate-nitrogen (NO_3-N). The DWS for NO_3-N is 10,000 µg/L and 45,000 µg/L for NO_3^-.
c. There is no DWS specific to Cr(VI).

Table 8-5. Waste Site Evaluation for 100-D/H RI/FS

100-D/H Waste Sites – 343 Total

NUMBER OF WASTE SITES FOR FURTHER EVALUATION

Sites not carried into FS (52)

Sites Closed, Not Accepted, or Rejected not carried into FS. (48)
100-D-27, 120-D-1, 1607-D3, 122-DR-1:1, 122-DR-1:2, 122-DR-1:3, 122-DR-1:4, 122-DR-1:5, 122-DR-1:6, 122-DR-1:7, 100-D-11, 100-D-26, 100-D-34, 100-D-36, 100-D-37, 100-D-38, 100-D-55, 100-D-57, 100-D-89, 100-D-91, 100-H-6, 100-H-15, 100-H-18, 100-H-19, 100-H-20, 100-H-26, 100-H-27, 600-258, 100-D-17, 100-D-28:2, 100-D-33, 100-D-35, 100-D-40, 100-D-41, 100-D-79, 100-D-92, 100-D-93, 100-D-95, 126-D-1, 126-D-3, 100-H-16, 100-H-32, 100-H-39, 100-H-47, 100-H-55, 116-H-6, 126-H-1, 100-H-49:3

Sites to be Closed under Washington Department of Health Regulations. (1)
100-D-58

Reactor Cores/Safe Storage Enclosures (3)
118-D-6:1, 118-DR-2:1, 118-H-6:1

Evaluate Based on Waste Site Status / Tank Removal / Reactor Site

Chapter 1

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Sites Identified for No Further Action (146)

Sites Pass Screening Levels for Human Health Risk Assessment, Groundwater/Surface Water Protection, Ecological Risk Assessment, Modeling Predictions. (125)
100-D-1, 100-D-2, 100-D-3, 100-D-4, 100-D-7, 100-D-9, 100-D-12, 100-D-13, 100-D-15, 100-D-20, 100-D-21, 100-D-22, 100-D-23, 100-D-24, 100-D-28:1, 100-D-29, 100-D-31:1, 100-D-31:10, 100-D-31:2, 100-D-31:3, 100-D-31:4, 100-D-31:5, 100-D-31:6, 100-D-31:7, 100-D-31:8, 100-D-31:9, 100-D-32, 100-D-42, 100-D-43, 100-D-45, 100-D-47, 100-D-48:4, 100-D-49:3, 100-D-50:5, 100-D-53, 100-D-54, 100-D-56:1, 100-D-56:2, 100-D-61, 100-D-64, 100-D-70, 100-D-74, 100-D-75:3, 100-D-80:1, 100-D-82, 100-D-83:4, 100-D-84:1, 100-D-85:1, 100-D-87, 100-D-88, 100-D-90, 100-D-94, 100-H-2, 100-H-3, 100-H-4, 100-H-7, 100-H-8, 100-H-17, 100-H-24, 100-H-28:1, 100-H-28:6, 100-H-30, 100-H-35, 100-H-37, 100-H-40, 100-H-41, 100-H-45, 100-H-49:2, 100-H-50, 100-H-51:4, 100-H-51:5, 100-H-53, 116-D-2, 116-D-4, 116-D-5, 116-D-6, 116-D-9, 116-D-10, 116-DR-4, 116-DR-5, 116-DR-7, 116-DR-8, 116-DR-10, 116-H-2, 116-H-5, 116-H-9, 118-D-1, 118-D-4, 118-D-5, 118-DR-1, 118-H-1:1, 118-H-1:2, 118-H-2, 118-H-3, 118-H-4, 118-H-5, 118-H-6:4, 118-H-6:5, 120-D-2, 126-D-2, 128-D-2, 128-H-1, 128-H-2, 128-H-3, 130-D-1, 132-D-1, 132-D-2, 132-D-3, 132-DR-1, 1607-D1, 1607-D2:1, 1607-D2:2, 1607-D2:3, 1607-D2:4, 1607-D4, 1607-D5, 1607-H1, 1607-H2, 1607-H3, 1607-H4, 600-151, 600-152, 600-30, 628-3, UPR-100-D-5

No Action Sites based on site specific evaluation. (21)
100-D-50:3, 100-D-50:10, 100-D-67, 100-D-68, 100-D-86:2, 100-H-9, 100-H-10, 100-H-13, 100-H-28:8, 100-H-31, 100-H-33, 116-D-3, 116-H-4, 118-D-6:2, 118-H-6:2, 128-D-1, 132-D-4, 132-DR-2, 132-H-1, 132-H-2, UPR-100-D-1

Evaluate in Chapters 5, 6, 7, and 8

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Waste Sites to be Remediated under the Interim Action RODs. (59)
100-D-8, 100-D-14, 100-D-30, 100-D-31:11, 100-D-31:12, 100-D-50:1, 100-D-50:4, 100-D-50:6, 100-D-50:7, 100-D-50:8, 100-D-50:9, 100-D-62, 100-D-65, 100-D-66, 100-D-69, 100-D-71, 100-D-72, 100-D-73, 100-D-75:2, 100-D-76, 100-D-77, 100-D-78, 100-D-80:2, 100-D-81, 100-D-83:1, 100-D-83:2, 100-D-83:3, 100-D-83:5, 100-D-84:2, 100-D-85:2, 100-D-86:1, 100-D-86:3, 100-D-97, 100-D-99, 100-D-100, 100-D-104, 100-H-28:2, 100-H-28:3, 100-H-28:4, 100-H-28:5, 100-H-42, 100-H-43, 100-H-44, 100-H-46, 100-H-48, 100-H-49:1, 100-H-51:1, 100-H-51:2, 100-H-51:3, 100-H-52, 116-DR-3, 118-D-2:1, 118-D-2:2, 118-D-3:1, 118-D-3:2, 126-DR-1, 126-H-2, 132-H-3, 1607-D2:5

Assume then Verify Interim Actions Achieve Required Standards

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Sites Identified for Further Action (86)

Sites with Deep Contamination Exceeding Human Health Criteria, but Incomplete Pathway. (32)
100-D-5, 100-D-6, 100-D-18, 100-D-19, 100-D-46, 100-D-48:1, 100-D-48:2, 100-D-48:3, 100-D-49:1, 100-D-49:2, 100-D-49:4, 100-H-1, 100-H-11, 100-H-12, 100-H-14, 100-H-21, 100-H-22, 116-D-1A, 116-D-1B, 116-D-7, 116-DR-1&2, 116-DR-6, 116-H-1, 116-H-3, 116-H-7, 118-D-6:3, 118-D-6:4, 118-H-6:3, 118-H-6:6, UPR-100-D-2, UPR-100-D-3, UPR-100-D-4

River Pipelines. (2)
100-D-60, 100-H-34

Waste Sites Remaining for Remedial Action. (52)
100-D-10, 100-D-50:2, 100-D-52, 100-D-59, 100-D-63, 100-D-75:1, 100-D-96, 100-D-98:1, 100-D-98:2, 100-D-101, 100-D-102, 100-D-103, 100-D-105, 100-D-106, 100-D-107, 100-H-5, 100-H-28:7, 100-H-36, 100-H-38, 100-H-51:6, 100-H-54, 100-H-56, 100-H-57, 100-H-58, 100-H-59, 116-D-8, 116-DR-9 [100-D-25], 118-DR-2:2, 600-380, 600-381, 600-382:1, 600-382:2, 600-382:3, 600-382:4, 600-382:5, 600-383:1, 600-383:2, 600-383:3, 600-383:4, 600-383:5, 600-383:6, 600-383:7, 600-383:8, 600-383:9, 600-383:10, 600-384:1, 600-384:2, 600-384:3, 600-384:4, 600-384:5, 600-385

Institutional Controls

52

Develop Remedial Action Alternatives and Cost Estimates to Achieve Final Cleanup

REMEDIAL INVESTIGATION

FEASIBILITY STUDY

CHPJBS_100DH_0124g

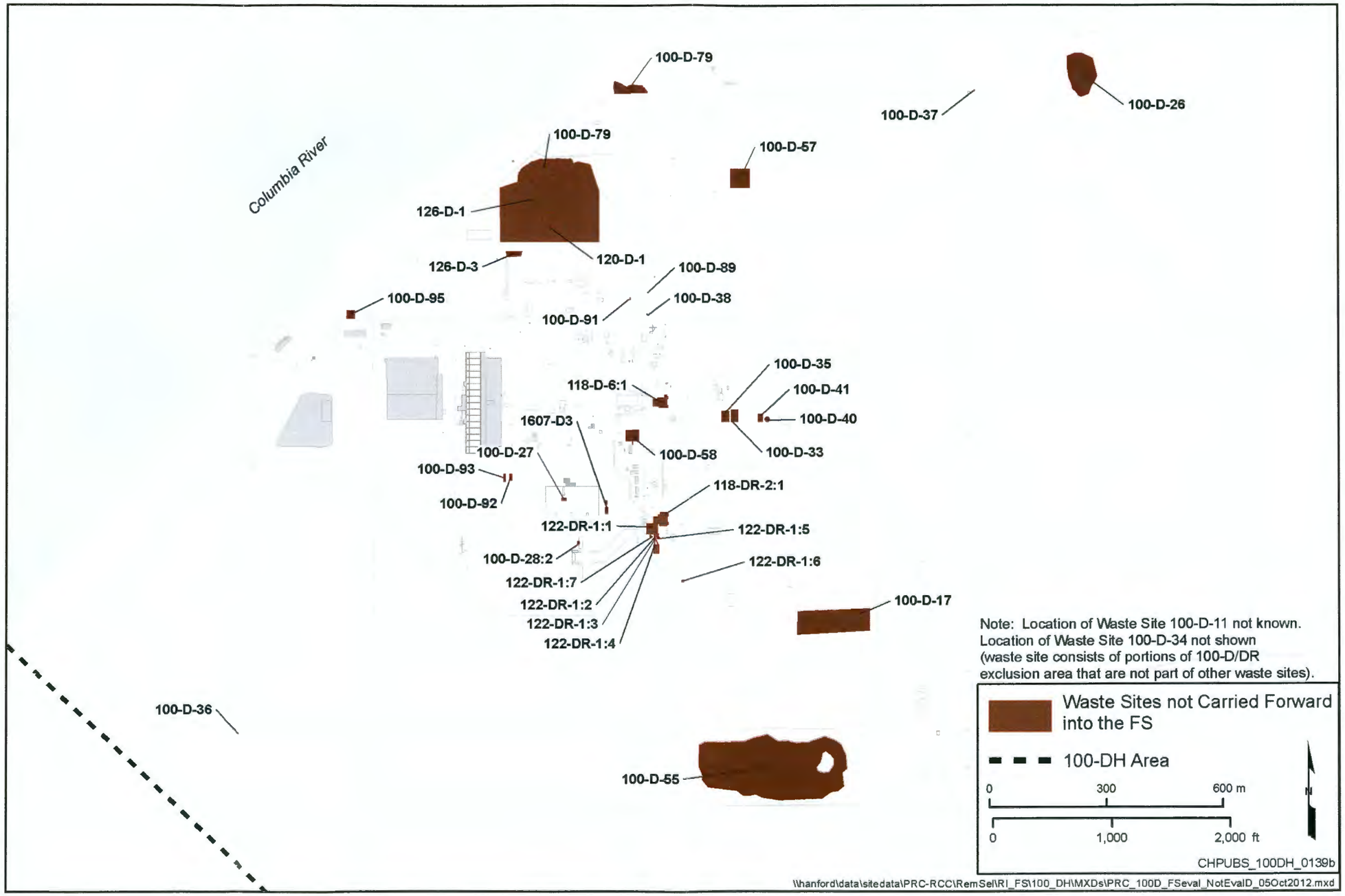

Figure 8-1. 100-D Waste Sites not Carried Forward into the FS

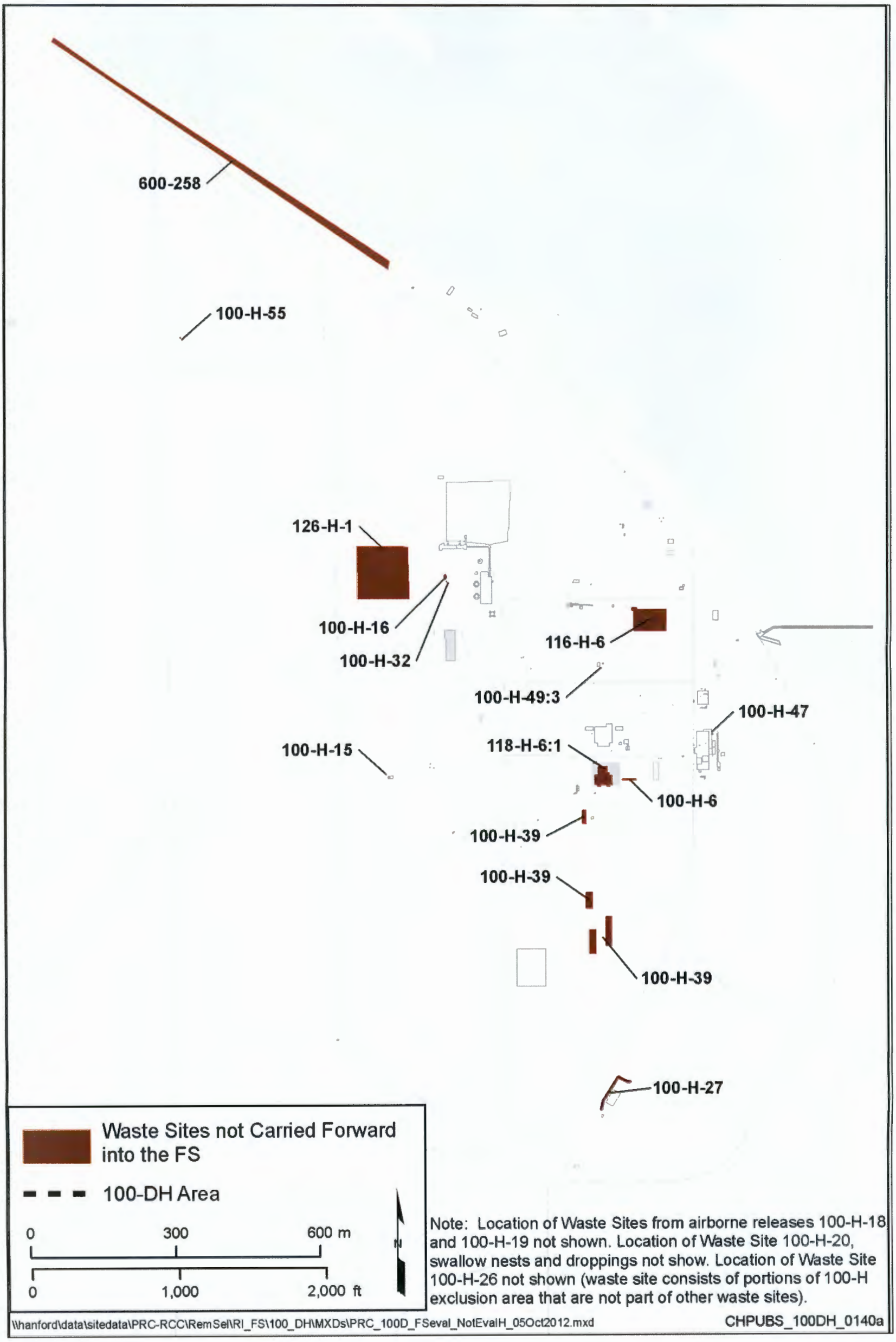

Figure 8-2. 100-H Waste Sites not Carried Forward into the FS

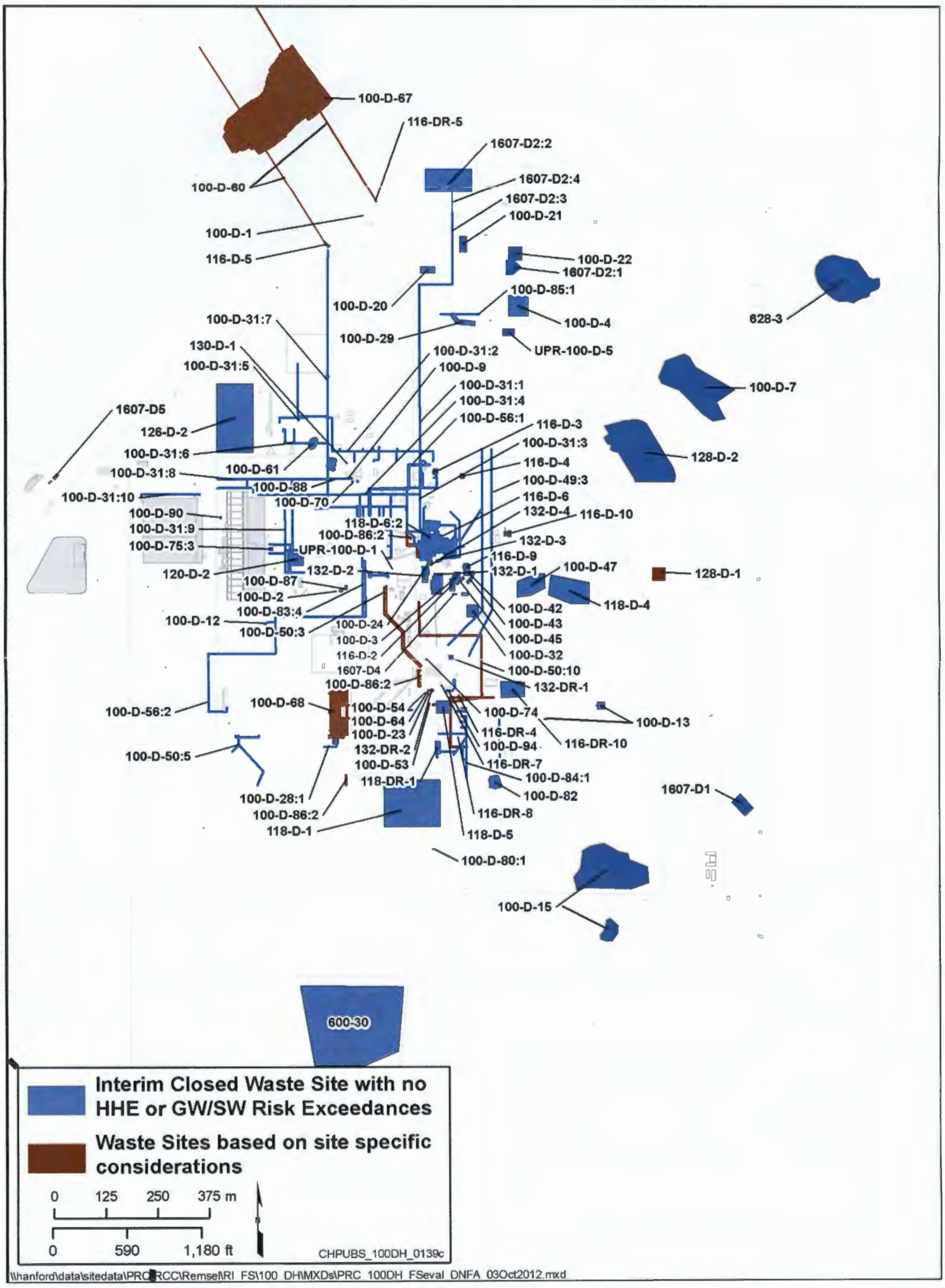

Figure 8-3. 100-D Waste Sites Identified for No Further Action

Figure 8-4. 100-H Waste Sites Identified for No Further Action

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The waste sites identified for no further action include the following:

- 125 interim closed or interim no action sites with verification data that have been quantitatively evaluated to the PRGs in Table 8-2 and indicate no unacceptable risk to human health and the environment
- 21 waste sites with site-specific evaluations indicating no further action

The site-specific evaluations for the 21 waste sites identified for no further action are described as follows:

- ***100-D-50:10.*** The 100-D-50:10 construction camp potable water supply pipelines subsite encompasses residual cast iron pipelines formerly used to supply potable water to the temporary construction camp southeast of the DR Reactor. This subsite was reclassified as "no action" based on a determination that potable water supply was not associated with any constituents that would present an adverse risk to human health and the environment. This determination remains appropriate for final action purposes.
- ***100-D-50:3.*** Based on a review of historical documentation, it is concluded that the 100-D-50:3, Reactor Cooling Water Pipelines from 190-D High Bay, were not put into operational use. The original operational plans to use the pipelines to supply process water to the DR Reactor were discontinued after determining that D Reactor could continue to be used. The 100-D-50:3 subsite does not pose a risk to human or ecological receptors, groundwater, or the Columbia River and is rejected from consideration as a waste site.
- ***100-D-67.*** The 100-D-67, D Island waste site was evaluated to investigate further its classification as a waste site in accordance with the RAOs and goals established in the Remaining Sites ROD (EPA, 1999). Evaluation of the 100-D-67 waste site, including risk assessment by the Washington State Department of Health, radiological surveys, and field investigations and sampling, indicate that residual contamination concentrations are protective of human health and the environment. The results of this evaluation indicate that the residual concentrations of radionuclide COCs/COPCs at this waste site do not preclude any future land uses (as bounded by a casual recreational user scenario) and are less than the PRG values in Table 8-2. Both the external exposure risk and ingestion risk in 1993 were below the radioactivity risk threshold of 10^{-6}. Since 1993, cobalt-60 has decayed through almost four half-lives so the present day risks will be considerably less than these values. Based on this evaluation, no further action is identified for the 100-D-67 waste site.
- ***100-D-68.*** The 100-D-68 site consists of the below grade remaining concrete structures for the former 190-DR Pump House. Remaining concrete was evaluated in accordance with an approved facility-specific closure plan and determined to meet the criteria for "no action." This determination remains appropriate for final action purposes.
- ***100-D-86:2.*** The 100-D-86:2 subsite consists of potential process sewer segments that were not captured by other waste sites. All of the potential segments in this subsite were found not to be present by direct test pitting and investigation at the suspected locations and the subsite was reclassified as no action. This determination remains appropriate for final action purposes.
- ***116-D-3.*** The 116-D-3 site was identified as a crib associated with the former 108-D facility. Based on review of historic drawings, geophysical investigation, and excavation, this site was determined to be a duplicate of the 116-D-4 site. The 116-D-3 site was reclassified as "rejected" in 2000; this reclassification was amended to "no action" in 2003. Another potential location for this crib has been

identified separately as the 100-D-76 waste site. The 116-D-4 and 100-D-76 sites are addressed further in this RI/FS, but no further action is identified for the redundant 116-D-3 site.

- ***118-D-6:2.*** The 118-D-6:2 subsite consists of the ancillary support areas and below grade structures for the D Reactor. This subsite is not inclusive of the safe storage enclosure and reactor core or the former fuel storage basin, which are addressed as separate subsites. This subsite was reclassified as "interim closed out" based on complete removal of all above- and below grade structures. Sampling was not required under the facility-specific SAP because there were no drivers for potential contamination associated with these facilities to enter into soil. This basis remains appropriate for no further final action considerations for this subsite. Other portions of the former 105-D facility are considered separately.

- ***128-D-1.*** The 128-D-1 site was identified as a potential burn pit, but was determined to be a duplicate of either the 128-D-2 or the 628-3 burn pit waste sites. The site was reclassified as "no action" based on this determination; this remains appropriate for final action purposes.

- ***132-D-4.*** The 132-D-4 site consists of the former 116-D Exhaust Stack for the D Reactor. The stack and foundation have been demolished and removed to 0.9 m (3 ft) below grade. The 132-D-4 site was reclassified to "interim closed out" based on sampling at the 116-C, 116-F, and 116-H stacks, which determined that contamination penetrated the stack concrete minimally and would be removed by demolition to below grade. A drain line contained within the stack concrete was determined to be analogous to drain lines in the 116-C and 116-F, which were determined to require no action based on dose modeling. These determinations remain appropriate for the residual concrete foundation.

- ***UPR-100-D-1.*** The UPR-100-D-1 site was initially identified as a small area of suspect oil-stained soil. A 2005 walkdown could not locate any evidence of the site, and noted that the area had been highly disturbed by demolition and remediation activities. The site was reclassified as "no action." This determination remains appropriate for final action purposes.

- ***132-DR-2.*** The 132-DR-2 site consists of the former 116-DR Exhaust Stack for the DR Reactor. Use of this stack for the 117-DR LSFF was addressed separately as the 122-DR-1:5 subsite. The stack and foundation have been demolished and removed to 0.9 m (3 ft) below grade. This site was reclassified to "interim closed out" based on removal of potentially contaminated concrete and radiological survey of surrounding soil. These determinations remain appropriate for the residual concrete foundation.

- ***100-H-28:8.*** The 100-H-28:8 subsite encompasses the former process water supply pipelines in subgrade concrete tunnels between the 190-H and 105-H facilities. These pipelines were completely removed during demolition of the 190-H facility and ISS activities for the H Reactor. The subsite was reclassified as "no action" based on removal of the pipelines and the absence of any evidence of leakage during operations. This determination remains appropriate for final action purposes.

- **116-H-4.** The 116-H-4 crib was removed in 1960 during construction of the 117-H Building and no requirements for interim action were previously determined. Remedial investigation samples from borehole C7862, as discussed in Chapter 4, were collected to characterize the waste site. The evaluation on the soil data from RI borehole was performed (Chapters 5, 6, and 7) and identified that concentrations did not exceed human health, groundwater protection, or surface water protection PRG values for analyzed constituents. The maximum detected concentrations for aluminum and lithium exceeded ecological PRG values; however, the maximum detectable concentrations were below Hanford background concentrations. Additionally, Cr(VI) and total chromium concentrations from groundwater samples collected at the borehole were both below detection limits. Based on having been previously removed and RI data evaluation, no further action is identified at this site.

- **100-H-33.** The 183-H Solar Evaporation Basins RCRA waste site (116-H-6) was closed out through a modified RCRA closure in 1997. The 100-H-33 waste site was created to address the radionuclide component of the 183-H Solar Evaporation Basins. Maximum concentrations of dangerous waste constituents of concern in 183-H Soil from the test pit following remediation in 1997, *183-H Solar Evaporation Basins Postclosure Plan* [DOE/RL-97-48]), are less than the PRG values presented in Table 8-2. In addition, sample results presented in Chapter 4 from the RI borehole identified in the 100-D/H Work Plan (DOE/RL-2008-46-ADD1) at the 183-H Solar Basin (C7860) did not exceed PRG values for analyzed constituents. Fate and transport modeling and risk assessment evaluation in Chapters 5, 6 and 7 did not identify unacceptable human health and the environment, groundwater, or surface water risk. For these reasons, no further action is identified in the remedial action alternatives for the 100-H-33 or 116-H-6 waste site.

- **132-H-1.** The 132-H-1 site encompasses the residual components of the demolished 116-H Reactor Stack, including the demolition burial trench. This stack was decommissioned, demolished, and buried in-place in 1983. Under interim actions, samples collected during decommissioning were used to model the residual condition, and no further actions were determined to be necessary for the site. However, the site was ultimately completely exhumed and removed during remediation of the adjacent 132-H-3 waste site. As such, no further action is warranted for the 132-H-1 site; the 132-H-3 site is addressed further in this RI/FS.

- **132-H-2.** The 132-H-2 site encompasses residual structural components of the demolished 117-H Filter Building. This building was decontaminated, decommissioned, and demolished in 1984, leaving portions of the below-grade structure and demolition rubble in place. Under interim actions, samples collected during decommissioning were used to model the residual condition conservatively, and no further actions were determined to be necessary for the site. However, the residual concrete was ultimately exhumed and removed during remediation of the adjacent 132-H-3 waste site. As such, no further action is warranted for the 132-H-2 site; the 132-H-3 site is addressed further in this RI/FS.

- **100-H-9, 100-H-10, and 100-H-13.** The 100-H-9, 100-H-10, and 100-H-13 sites were French drains adjacent to the H Reactor, likely associated with steam condensate discharges. These French drains and underlying soil were removed and disposed incidentally during ISS activities for the H Reactor. These sites were reclassified as "interim closed out" and the determination remains appropriate.

- **100-H-31.** The 100-H-31 site consisted of an area of PCB soil contamination associated with an unplanned release from a former electrical substation on the north side of the 105-H Reactor Building. PCBs have low mobility in soil based on soil distribution coefficients; therefore, these unplanned releases would not have resulted in contamination migration to significant depth. Soil at the location of the release was removed and disposed to a depth of approximately 4.6 m (15 ft) during ISS activities for H Reactor, and the site was reclassified as "interim closed out" based on removal of potentially contaminated soil. This basis is also appropriate for no further final action considerations.

- **118-H-6:2.** The 118-H-6:2 subsite consists of the ancillary support areas and below grade structures for the H Reactor. This subsite is not inclusive of the safe storage enclosure and reactor core or the former fuel storage basin, which are addressed as separate subsites. This subsite was reclassified as "interim closed out" based on complete removal of all above- and below-grade structures. Soil sampling was not required because the structures that were removed were not subjected to standing contaminated water and there was no mechanism for residual surface

concrete contamination to penetrate into the concrete and underlying soil. This basis remains appropriate for no further final action considerations for this subsite. Other portions of the former 105-H facility are considered separately.

The remaining 145 sites will be discussed further in this RI/FS and are evaluated further in Chapter 9 for remedial alternatives. Tables J-1 and J-2 in Appendix J identify the contaminants and basis for action (where there are risks) associated with waste sites remediated under interim actions. Table J-2 is a subset of Table J-1 that shows those sites that have groundwater/surface water protection as a basis for action.

Waste Sites to be Remediated under Interim Action RODs. A total of 59 waste sites are currently or anticipated to be remediated under the 100 Area Remaining Sites ROD (EPA/ROD/R10-99/039), or are anticipated to be in progress by the time the ROD is signed. The locations of the sites are shown on Figure 8-5.

Waste Sites Remaining for Further Action. The remaining 86 waste sites are expected to require further action after the ROD is issued. Thirty-two of the 86 waste sites have been interim or interim no action closed and had verification data with exceedances of deep human health protection criteria (residential scenario) for select radionuclide compounds (Chapter 6). Contamination was detected in deep zone verification soil samples collected from depths greater than 4.6 m (15 ft) bgs and, as a result, there is no direct exposure pathway. DOE has proposed to place deep excavation institutional controls to limit exposure (Table 9-4). Rough order of magnitude cost for excavating and removing the contaminants at all 32 sites is $320 million. However, these radionuclides will continue to decay to below human health protection criteria within 2 to 185 years and during this time (depending on the current concentration of individual constituent(s) of interest at each site), DOE or the federal government will maintain controls on the land to prevent exposure to these materials. For this reason, institutional controls will be maintained for these sites until unrestricted use is allowable. No further technology application is required and remedial action alternatives are not developed for these sites.

Two other waste sites associated with river effluent pipelines (100-D-60 and 100-H-34) were evaluated for risks, as discussed in Chapters 6 and 7. The risk assessments associated with river effluent pipeline investigation indicate no unacceptable human health risk, based on RME scenarios. No technology application is required and remedial action alternatives are not developed for the below river effluent pipeline waste sites in this FS. While no IC is required, an annual inspection under the RCRA permit (Permit Number WA7 89000 8967) is conducted along the shoreline to identify Hanford debris.

The remaining 52 waste sites are expected to be remediated after the ROD is issued. These sites are evaluated for remedial alternatives in Chapter 9. Risk drivers have been determined based on knowledge of the process that was performed at the sites and remediation results at similar sites in the River Corridor (Table J-1, Appendix J). The remedial approaches for the major risk drivers are developed for each alternative and presented in Chapter 9. These sites include three waste sites completed under interim actions that had exceedances of human health direct exposure risk PRGs for radionuclides.

The following 52 waste sites remain for remedial action; locations are shown on Figure 8-6:

- Twenty-nine sites for which interim remedial actions will not be completed until after the ROD is signed (100-D-50:2, 100-D-75:1, 100-H-36, 100-H-58, 100-D-106, 100-H-51:6, 100-H-59, 600-380, 600-382:1, 600-382:2, 600-382:3, 600-382:4, 600-382:5, 600-383:1, 600-383:2, 600-383:3, 600-383:4, 600-383:5, 600-383:6, 600-383:7, 600-383:8, 600-383:9, 600-383:10, 600-384:1, 600-384:2, 600-384:3, 600-384:4, 600-384:5, and 600-385)

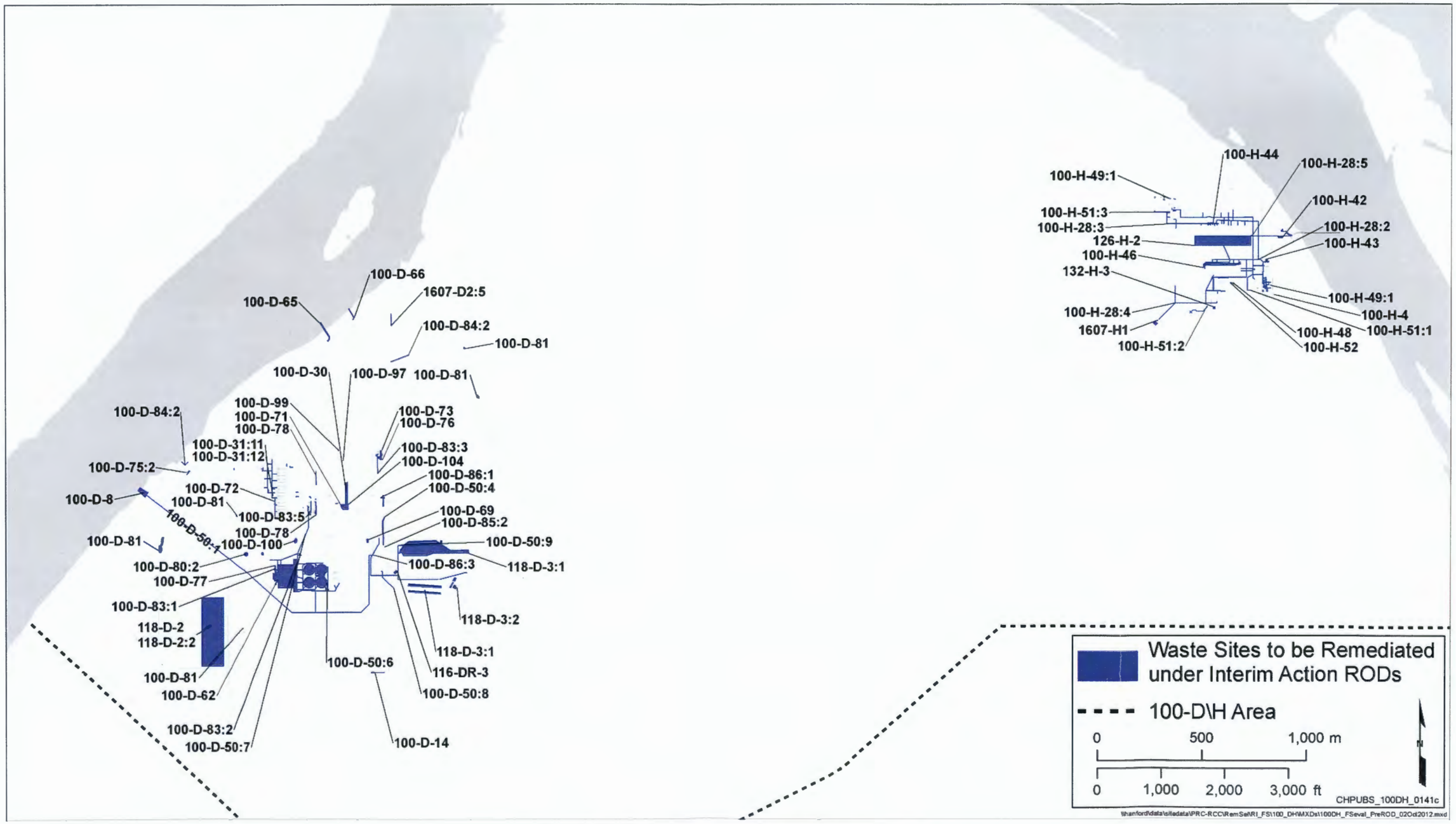

Figure 8-5. Waste Sites to be Remediated under Interim Action RODs

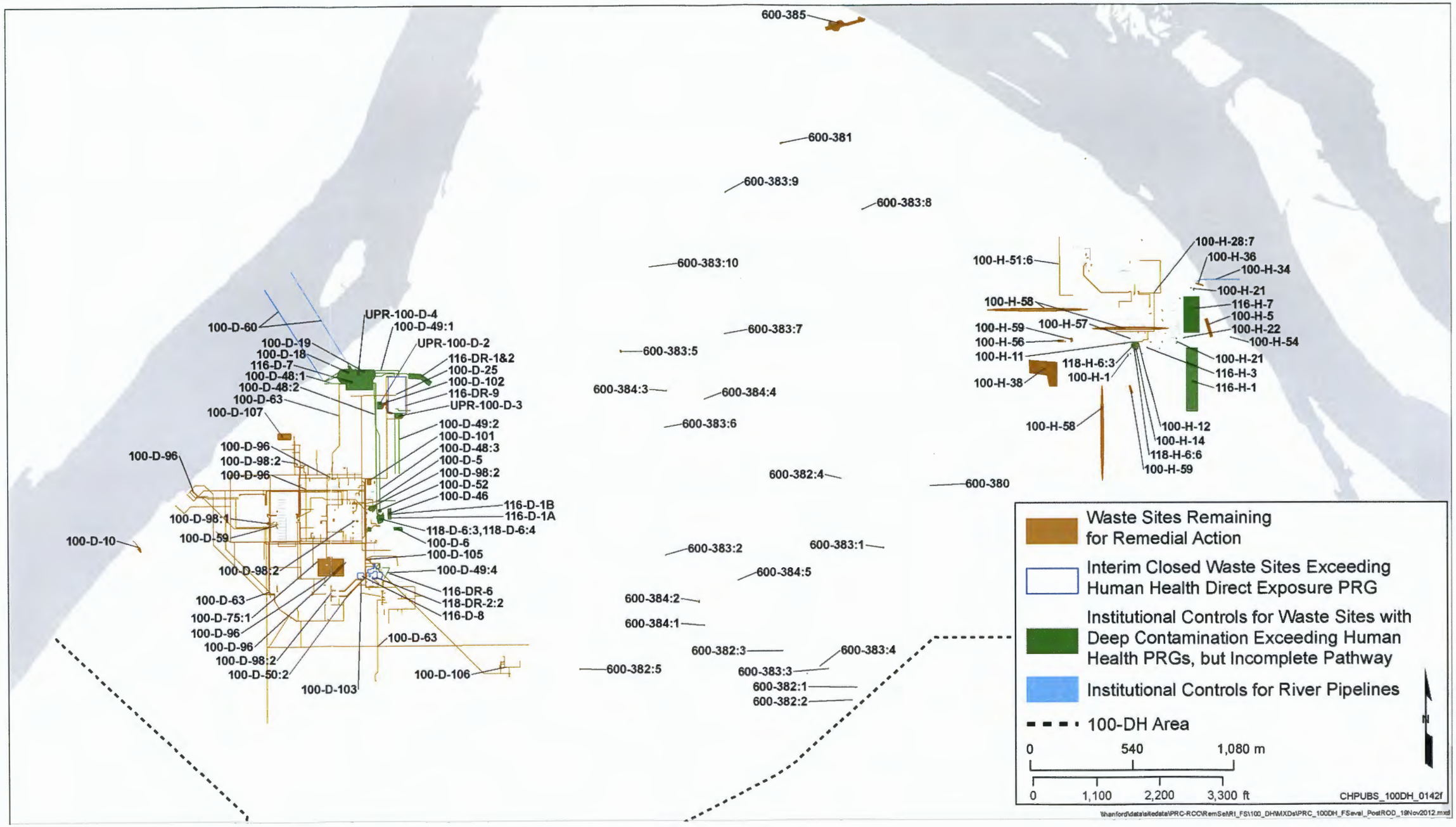

Figure 8-6. Waste Sites Remaining for Further Action

- Four sites that have undergone interim action but risk assessment indicates sites exceed shallow human health direct exposure criteria for radionuclides in a portion of the site (100-D-25, 116-DR-9, 116-D-8, and 118-DR-2:2)

- Four candidate sites that are being carried forward because of site-specific considerations (100-D-10, 100-D-52, 100-D-59, and 100-H-5)

- Fifteen candidate sites that have not yet undergone confirmatory evaluation under interim actions (100-D-63, 100-D-96, 100-D-98:1, 100-D-101, 100-D-102, 100-D-103, 100-D-105, 100-D-107, 100-H-54, 100-H-28:7, 100-H-38, 100-H-57, 100-D-98:2, 100-D-105, and 100-H-56)

The 100 Area Remaining Sites ROD (EPA/ROD/R10-99/039) established a process whereby new and existing sites that did not have sufficient information to make a remedial action determination could be evaluated to make this decision. These sites are referred to as "candidate sites" or "confirmatory sites" under the interim action framework. There are 15 waste sites identified as candidate sites that have not yet been evaluated as of June 2012. An additional four sites are being considered as candidate sites for final action based on site-specific considerations. Confirmatory evaluation to determine the need for remediation and confirmation of COCs will be performed for the 19 candidate sites. Remedial alternatives have been developed for all 19 under the assumption that remediation is determined to be warranted.

The 4 sites that have been remediated and interim closed under the interim actions ROD include 116-D-8, 116-DR-9, 100-D-25, and 118-DR-2:2. The risk assessment evaluation of verification sample data from the four sites indicates the sites exceed shallow human health direct exposure criteria for radionuclides (cesium-137, strontium-90, and technetium-99) in a portion of the sites (for example, side slopes of excavation in the shallow zone). The four sites are evaluated for remedial action alternatives in the FS for final remediation of residual risks.

The following bullets identify the four sites carried forward for inclusion in the FS, based on site-specific considerations:

- The 100-D-10 and 100-D-59 waste sites are not accepted and rejected waste sites, respectively, for which the existing basis warrants reconsideration as discussed in Section 1.2.3. Therefore, the sites are evaluated in the FS for final actions as candidate sites.

- The 100-D-52, 105-D Downcomer Insulation Space Dry Well waste site has been remediated and interim closed out. The data from verification sampling have been quantitatively evaluated and indicate no unacceptable risk to human health and the environment. However, this site history suggests cooling water effluent from D Reactor as a possible contributor of contamination. Contamination potentially associated with cooling water effluent includes Cr(VI) was not analyzed in interim closure verification samples and this site warrants consideration as a candidate site for future evaluation under final actions.

- The 100-H-5, 107-H Retention Basin Sludge Burial Site waste site has been remediated and interim closed out. The data from verification sampling have been quantitatively evaluated to the PRGs in Table 8-2 and indicate no unacceptable risk to human health and the environment. However, Cr(VI) analysis not included in interim closure verification samples, but the site was used for retention basin sludge disposal, which would contain incidental Cr(VI). This site warrants consideration as a candidate site for future evaluation under final actions.

Tables J-1 and J-2 in Appendix J identify the contaminants and basis for action (where there are risks) associated with waste sites remaining for remedial action. Table J-2 is a subset of Table J-1 that shows those sites that have groundwater/surface water protection as a basis for action.

8.2.1.2 Groundwater

Figures 4-66 and 4-67 illustrates areas exceeding the groundwater PRG for Cr(VI), and by inference, the areas exceeding the groundwater PRG for total chromium, which follows the same contaminant plume as Cr(VI). Figures 4-90 and 4-95 illustrate areas exceeding the groundwater PRG for nitrate and strontium-90, respectively.

As described in Chapter 4, Cr(VI) contamination has been identified in several locations (primarily in 100-H) in the uppermost water-bearing unit below the unconfined aquifer. There was no contamination deeper than this based on the RI characterization and there is a strong upward gradient within the lower portions of the Ringold Formation. Remediation of Cr(VI) contamination in the confined RUM aquifer is evaluated in the remedial action alternatives.

The CSM described in Section 4.9 identifies locations where confirmation sampling and RI characterization indicate cleanup goals have been achieved at waste sites, but groundwater monitoring indicates that a potential for residual contamination in soil exists. The four general locations where groundwater monitoring indicates potential residual contamination in the vadose zone that may contribute to groundwater contamination include the following:

- D and DR Reactors where FSB leaks, disposal cribs, and trenches were historical sources of Cr(VI) and mixed fission product contamination
- High-concentration sodium dichromate transfer facility locations, including the vicinity of the 100-D-100 waste site and other related conveyance systems
- High-volume, low-concentration cooling water disposal areas
- 183-H Solar Evaporation Basin

8.2.1.3 Riparian Soil

Human health risks were assessed in areas outside the footprints of waste sites as part of the RCBRA (DOE/RL-2007-21, Volume II) and the CRC (DOE/RL-2010-117, Volume II) risk assessments. Based on the results from this analysis (summarized in Section 6.4), there are no COPCs in riparian soils, near-shore sediments, and surface water that warrant further evaluation in the FS.

Appendix L presents a CSM of the riparian and near shore environment along 100-D/H to supplement the analysis of River Corridor-wide ecological risks presented in the RCBRA (DOE/RL-2007-21). Results from this evaluation identifies that, with the exception of total chromium and Cr(VI), detected concentrations of contaminants in riparian or near shore groundwater, seeps, aquifer tubes, and pore water do not present an ecological concern, or are not associated with contaminated groundwater resulting from Hanford Site operations, and total chromium and Cr(VI) should be considered the only COECs for purposes of alternatives evaluation.

8.3 Identification and Screening of Technology Types and Process Options

This section presents remedial technologies and process options that are subsets of the selected GRAs, and that may potentially meet RAOs for contaminated waste sites and groundwater at 100-D/H. The potential remedial technologies are evaluated or screened for implementability, effectiveness in eliminating, reducing, or controlling risks to human health and the environment, and relative cost. The identified technologies are then combined into a range of remedial alternatives in Chapter 9.

8.3.1 Identification and Screening of Technologies

The discussion summarizes the technologies and process options considered as part of this evaluation. Although no action and institutional controls are not considered remedial technologies, they are important response actions to be considered as part of the remediation approach and are discussed herein.

Tables 8-6 and 8-7 present the identification and screening of technologies and remedial process options for 100-D/H in tabular form. Table 8-6 presents technologies for waste site treatment for radionuclides, Cr(VI) and other metals, and organic compounds. Table 8-7 presents GRAs and process options for groundwater impacted with Cr(VI) and other COPCs.

8.3.1.1 Identification and Screening of Technologies for Vadose Zone Contamination

No Action. The no action response means any further action to remove, remediate, monitor, or restrict access to contaminated waste sites is discontinued. Source areas and residual soil contaminants in the waste sites would be left untreated and current monitoring activities would cease. The CERCLA RI/FS Guidance (EPA/540/G-89/004) and the NCP (40 CFR 300) require this response to remain in the FS process, where it serves as a baseline against which to compare all other alternatives. Although generally considered unacceptable as a remedial alternative, no action would be an appropriate alternative component for waste sites where interim actions have been completed as dictated by the 100 Area Remaining Sites ROD (EPA/ROD/R10-99/039) (as presented in Table 1-2) and verification sampling data suggest the waste site does not present risks to human health and the environment.

Institutional Controls. Institutional controls are non-engineered instruments, such as administrative and/or legal controls, that minimize the potential for human exposure to contamination by limiting land or resource use. They are generally to be used in conjunction with, rather than in lieu of, engineering measures such as waste treatment or containment, and can be used during all stages of the cleanup process to accomplish various cleanup-related objectives. Institutional controls should be "layered" (that is, use multiple institutional controls) or implemented in a series to provide overlapping assurances of protection from contamination. These administrative controls are imposed on land use to prevent or reduce exposure to hazardous wastes or hazardous constituents and/or to protect the integrity of a remedy. The need for institutional controls is evaluated in the FS and recorded in CERCLA decision documents. The decision document is part of the Administrative Record for the selection of remedial actions. Institutional controls will be identified for each waste site following completion of the remedial action if the contamination left in place is greater than levels protective of unrestricted land use.

As they are identified, DOE will apply and implement institutional controls in an integrated manner such that mechanisms in place will ensure controls are effective, implemented as planned, properly maintained, inventoried, periodically re-evaluated, and modified as necessary to reflect changes in conditions, needs, or technological advancements. DOE will maintain the institutional controls as long as necessary to perform their intended protective purposes (*Use of Institutional Controls* [DOE P 454.1]).

The Site-Wide IC Plan (DOE/RL-2001-41, Rev. 5) developed by DOE-RL describes how OU-specific institutional controls specified in CERCLA decision documents will be implemented and maintained. The Site-Wide IC Plan is updated based on final CERCLA decision documents within 180 days after issuance of the final decision document. The Site-Wide IC Plan addresses the elements of *Institutional Controls: A Site Manager's Guide to Identifying, Evaluating and Selecting Institutional Controls at Superfund and RCRA Corrective Action Cleanups* (EPA 540-F-00-005). In addition, institutional controls are reviewed during the CERCLA 5-year review process.

Table 8-8 identifies DOE categories of institutional controls and the examples of institutional controls currently in use at the Hanford Site, including whether the institutional control will be retained for further

evaluation in the FS. Institutional controls will be identified for each waste site following completion of the remedial action if the contamination left in place is greater than levels protective of unrestricted land use.

In September 1993, DOE issued the Reactor Decommissioning ROD (58 FR 48509) that established a path forward for the Hanford Site reactors. The "Amended Record of Decision for the Decommissioning of Eight Surplus Production Reactors at the Hanford Site, Richland, WA" (75 FR 43158) was issued in July 2010. The Reactor Decommissioning ROD (58 FR 48509) provided options for immediate dismantlement for reactor decommissioning and one-piece disposal or dismantlement of the reactor cores after an ISS period of approximately 75 years, which allowed for the decay of the radionuclides that presented the major risk for the site workers. Until reactor removal is complete, DOE will continue to conduct routine maintenance, surveillance, and radiological monitoring activities to ensure continued protection of human health and the environment during the ISS period.

Removal. Removal technologies include excavation of contaminated materials. The engineering design is based on existing information. Existing information, including operational process knowledge, vadose zone data, groundwater data, and waste site remediation of similar sites, is used in determining the area for remediation. Excavation of sites with contaminated soil follows the observational approach, allowing waste characterization a to occur as excavation proceeds. The observational approach uses a variety of techniques including field screening, confirmation sampling, and potholes, soil borings, or test pits, as appropriate, to determine the extent of contaminant removal required until cleanup goals have been met. Excavation is coupled with additional characterization from the observational approach, analytical assessment of soil and groundwater to support design volumes, dust control, efficient transportation, treatment as required, and disposal. Excavated soil is segregated to determine disposal or treatment requirements.

Excavation can use conventional equipment and methods including excavators, bulldozers, and wheeled loaders. Earthmoving equipment removes clean overburden as appropriate, which can be staged for later use in backfilling, and contaminated media to stage for appropriate waste management activities. Contaminated media typically are removed in lifts (layers of uniform thickness) to allow screening for contamination. Field screening supports waste characterization and helps determine achievement of remedial goals.

Process options under the removal GRA include standard excavation (depths up to 6 m [20 ft]) and deep excavation (to depths greater than 6 m [20 ft]). The determination of 6 m (20 ft) for deep excavation is based on engineering considerations. At excavations exceeding 6 m (20 ft) bgs, implementation requires technologies that are more complex, such as large layback for open-pit type excavation or use of shoring. Given the increased complexity, deep excavations have an increased cost compared to standard excavation.

Ex Situ Treatment and Processing. Following excavation, soil can be treated with ex situ methods to reduce contaminant concentrations or toxicity, remove contaminants (transfer to different media), or reduce volume, and allow for less costly disposal. Treatment can be achieved by applying physical, chemical, biological, or thermal techniques.

Additional treatment that may be required to meet waste acceptance criteria at ERDF is not included in the costs for this process. This ex situ treatment process option only covers technologies that could be used to treat the soil so that part or all of the soil volume could be backfilled at the location from which it was removed.

Table 8-6. Screening Table—Technologies for Radionuclides, Hexavalent Chromium, Other Metals, and Organic Compounds in the Vadose Zone—100-D/H

General Response Actions	Remedial Technology	Process Option	COPC Applicability[a]	Depth Range[b]	Description	Relative Effectiveness	Relative Implementability	Relative Capital Cost	Relative O&M Cost	Sustainability[c]	Retained/ Not Retained[d]	Screening Comment
No Action	No Action	No Action	All	6 m (20 ft)/ greater than 6 m (20 ft)	No further actions to address contamination. Source areas and residual contaminants in vadose zone are left untreated.	Low/High No remedial actions are taken, but effectiveness could be high if risk is previously mitigated.	High No administrative or technical implementability challenges are associated with implementation of this option, since no actions are required.	Low No associated cost.	Low No associated cost.	Little impacts.	Retained	Retained per the NCP (40 CFR 300).
Removal	Excavation	Standard Excavation	All	6 m (20 ft)	Shallow soil is removed using conventional construction equipment. excavation limited to approximately 6 m (20 ft) bgs. Excavated soil is segregated (automated or laboratory based) to determine disposal or treatment requirements.	High Shallow contaminated soil removed.	High Standard excavation is typically straightforward. An Excavation Permit is required in the 100, 200, and 300 Areas and the Hanford Reach National Monument.	Moderate/High	Low No associated cost.	Waste generation if excavated soil is disposed of, GHG and energy for excavation equipment.	Retained	
		Deep Excavation	All	6 m (20 ft)/ greater than 6 m (20 ft)	Soil is removed to deeper depths. Deep excavation would require implementation of more complex technologies, for example, large layback for open pit type excavation or alternatively use of shoring. Excavated soil is segregated (automated or laboratory based) to determine disposal or treatment requirements.	High Locations of the deep contaminated soil will be difficult to identify, meaning large areas would have to be excavated to depth to ensure that the deep sources were removed.	Moderate Has been performed at Hanford Site using laybacks. Shoring may be difficult with cobbles and boulders. Increased safety challenges with very deep excavations. An Excavation Permit is required in the 100, 200, and 300 Areas and the Hanford Reach National Monument.	High	Low No associated cost.	Waste generation if excavated soil is disposed of, GHG and energy for excavation equipment.	Retained	
Ex Situ Treatment and Processing[e]	Ex Situ Treatment and Processing[e]	Solidification/Stabilization	Mobile to Semi-Mobile contaminants (Cr[VI], strontium-90)	Depends on excavation method	Contaminants are physically bound or enclosed within a stabilized mass (solidification), or chemical reactions are induced between the stabilizing agent and contaminants to reduce their mobility (stabilization). Agents include soluble phosphates, pozzolan/portland cement, polyethylene extrusion, etc. The stabilized mass is returned to its original location and capped to shed water and prevent weathering. The location is engineered to withstand seismic activity.	Low/Moderate Effective at immobilizing contaminants in excavated material. However, the stabilized mass must be protected from weathering and seismic activity for long-term durability.	Moderate Well established technology. Site-specific studies need to be completed to evaluate equipment required and appropriate solidification/ stabilization agents. Mechanically intense process; additional handling of the excavated soil could increase the potential for contaminant exposure, which could pose risk to workers.	High	Low	GHG and energy for production and delivery of reagent used, and for transport and mixing.	Not Retained	Screened out in favor of the safer alternative of disposal in ERDF, a centralized facility engineered to protect against weathering and seismic activity.

Table 8-6. Screening Table—Technologies for Radionuclides, Hexavalent Chromium, Other Metals, and Organic Compounds in the Vadose Zone—100-D/H

General Response Actions	Remedial Technology	Process Option	COPC Applicability[a]	Depth Range[b]	Description	Relative Effectiveness	Relative Implementability	Relative Capital Cost	Relative O&M Cost	Sustainability[c]	Retained/ Not Retained[d]	Screening Comment
Ex Situ Treatment and Processing[e] (cont.)	Ex Situ Treatment and Processing[e] (cont.)	Soil Washing	Cr(VI), nitrate	Depends on excavation method	Consists of size separation of highly contaminated soil fractions (fines) from minimally contaminated soil fractions (coarse), followed by mechanical abrasion or washing to remove surface contamination. Final contaminated fraction is typically treated by technologies such as solidification/stabilization before onsite or offsite disposal.	Low/Moderate Effectiveness is driven by the binding processes that exist between the contaminants and the soil particles (adsorbed or precipitated). Pilot testing at Hanford suggests a number of contaminants strongly sorb to all sizes of soil.	Low/Moderate Mechanically intense. Conventional aggregate washing and screening technology is used to separate soil particles by size fraction. Contaminated soils and water are disposed of, or further treated. Soils that meet cleanup criteria (remediated coarse soil) can be returned to the site. Rinsate will require treatment before disposal.	Moderate/High	Low No associated cost.	Additional resource impact (water used in process), GHG, and energy for process and additional treatment of contaminated fines and water.	Not Retained	Mechanically intense. Not proven for conditions similar to the Hanford Site.
		Vitrification	All	Depends on excavation method	Thermal treatment process that converts excavated soil and other materials into glass matrix. The thermal treatment process is typically performed inside a chamber using plasma torches or electric arc furnaces to melt the soil. Organic contaminants are typically destroyed during the process by pyrolysis, while metals and radionuclides are retained in the molten soil.	High Heavy metals and radionuclides are incorporated into the glass structure, which is generally resistant to leaching.	Low High complexity of equipment required. Ex situ joule heating vitrification uses furnaces that have evolved from the glass industry. Implementability is higher than for in situ application, given use of proven technology (furnaces).	High	Low No associated cost.	GHG and energy for heat generation. High energy requirements to sustain required heat.	Not Retained	Complex technology, safety concerns with implementation.
		Thermal Desorption	Organics	Depends on excavation method	Direct application of heat to soil piles to increase the temperature of soil and destroy or volatilize organic compounds. A vapor cover and vacuum system are needed to transport volatilized water and organics to the gas treatment system. Also completed using mechanical systems (for example, rotary drum).	High Technology can achieve rapid removal/destruction of a mix of volatile and semivolatile organics at low residual levels.	Low Equipment readily available and commonly used, but can be mechanically complex.	High	Low No associated cost.	GHG and energy for production of heat vapor treatment.	Not Retained	Complex and challenging to implement.
Disposal	Disposal	Backfill Treated Soil	All	6 m (20 ft)/ greater than 6 m (20 ft)	Excavation and ex situ treatment followed by onsite disposal (backfill).	High Contaminated material has been treated by ex situ technologies.	High Excavated and treated soil will need to be compared to cleanup criteria to verify backfill is appropriate.	Low/Moderate	Low No associated cost.	GHG and energy for backfill.	Not Retained	No ex situ treatment technologies are retained.
		Disposal to ERDF	All	6 m (20 ft)/ greater than 6 m (20 ft)	Disposal of excavated soil at onsite landfill (ERDF). Treatment performed at the facility as required to meet land disposal restrictions.	High	High Implementability limited by COPC concentrations and onsite landfill requirements.	Low/Moderate	Low	GHG and energy for transport.	Retained	
		Other EPA approved Landfill	All	6 m (20 ft)/ greater than 6 m (20 ft)	Disposal of excavated soil at offsite landfill.	High Contaminated material has been treated by ex situ technologies.	High Implementability limited by COPC concentrations and offsite landfill requirements.	Moderate	Low No associated cost.	GHG and energy for transport.	Retained	

Table 8-6. Screening Table—Technologies for Radionuclides, Hexavalent Chromium, Other Metals, and Organic Compounds in the Vadose Zone—100-D/H

General Response Actions	Remedial Technology		Process Option	COPC Applicability[a]	Depth Range[b]	Description	Relative Effectiveness	Relative Implementability	Relative Capital Cost	Relative O&M Cost	Sustainability[c]	Retained/ Not Retained[d]	Screening Comment
In Situ Treatment	In Situ Treatment via Reagent	Reagent Approach	Solidification	Mobile COCs to semimobile Radionuclides, Other Metals, and Organics	6 m (20 ft)	Contaminants are physically bound or enclosed within a stabilized mass. Agents include pozzolan/Portland cement and polyethylene extrusion, etc. With organics, typically only used for free phase to reduce mobility.	Low/Moderate There is debate about the long-term durability of the monolith and whether it is in fact permanent. Potential for exposure still exists if waste is shallow.	Moderate Depends on delivery method.	High	Low/Moderate Assuming monolith is permanent.	GHG and energy for production and delivery of substrate/reagent.	Retained	Potential for incomplete contact of grout in the targeted treatment zone, and uncertainty regarding the durability of shallow soil encapsulation.
			Stabilization/ Sequestration	Radionuclides and Metals	6 m (20 ft)/ greater than 6 m (20 ft)	Chemical reactions are induced between the stabilizing agent and contaminant to reduce mobility. Agents include soluble phosphates and polyphosphates.	Low/Moderate Potential for direct exposure still exists if waste is shallow.	Low/Moderate Depends on delivery method.	Moderate	Low/Moderate Assuming stabilized mass is permanent	GHG and energy for production and delivery of substrate/reagent.	Retained	Uncertainty with uniform phosphate delivery and adequacy in removing risk associated with strontium-90. However, retained for strontium-90 at locations where excavation is not implementable.
			Chemical Reduction	Cr(VI)	6 m (20 ft)/ greater than 6 m (20 ft)	Chemical reductant (for example, calcium polysulfide, dithionite, hydrogen sulfide gas, ferrous sulfate, zero valent iron) is applied to the subsurface to treat contaminants within vadose zone. Chemical can be combined with solidification/stabilization or other treatment mechanisms.	Low/Moderate Chemical reductants are instantly reactive, which requires overloading to maintain reactive strength at depth.	Moderate Depends on delivery method. Localized temporary generation of secondary byproducts may occur. May temporarily mobilize COPCs toward groundwater. Handling chemical reductants is a health and safety concern.	Moderate	Moderate/High	GHG and energy for production and delivery of chemical agent.	Not Retained	More challenging to implement and costly as compared to biological reduction.
			Chemical Oxidation	Organics	6 m (20 ft)/ greater than 6 m (20 ft)	Subsurface delivery of chemical oxidant (for example, hydrogen peroxide, ozone, permanganate, persulfate, percarbonate) to degrade organic COPCs. Oxidants cause chemical destruction of toxic organic chemicals. Petroleum hydrocarbons and PAHs can be treated with a variety of oxidants (including peroxide, percarbonate, persulfate, and ozone); however, there are limited case studies demonstrating the successful treatment of PCBs with in situ chemical oxidation. Ozone is the most likely oxidant.	Low/Moderate Effectiveness is a function of oxidant distribution and contact. Injection of ozone a possible alternative, but more complex than bioventing alone. Multiple applications may be required to achieve complete treatment.	Low/Moderate Chemical oxidants can be delivered using soil mixing, horizontal injections wells, or vertical injection wells.	Moderate	Low/High O&M costs would be low assuming complete treatment can be achieved with a single application or high if multiple applications are required to achieve treatment.	GHG and energy for production and delivery of substrate/reagent.	Not Retained	Limited case studies demonstrating the successful treatment of PCBs with in situ chemical oxidation.

Table 8-6. Screening Table—Technologies for Radionuclides, Hexavalent Chromium, Other Metals, and Organic Compounds in the Vadose Zone—100-D/H

General Response Actions	Remedial Technology		Process Option	COPC Applicability[a]	Depth Range[b]	Description	Relative Effectiveness	Relative Implementability	Relative Capital Cost	Relative O&M Cost	Sustainability[c]	Retained/ Not Retained[d]	Screening Comment
In Situ Treatment (cont.)	In Situ Treatment via Reagent (cont.)	Reagent Approach (cont.)	Biological Reduction	Cr(VI), nitrate	6 m (20 ft)/ greater than 6 m (20 ft)	Biological carbon source (for example, molasses, sodium lactate, emulsified oil, butane) is applied to the subsurface to treat contaminants within vadose zone.	Moderate/High Carbon source follows source release pathways. Biological reductants are activated by microbial activity, so reactive strength is maintained over relatively longer distances.	Moderate Depends on delivery method. Localized temporary generation of secondary byproducts may occur. May temporarily mobilize COPCs (in first pore volume) toward groundwater.	Low/Moderate	Moderate	GHG and energy for production and delivery of substrate. Depends on which substrate is used.	Retained	
			Combined Chemical/ Biological Reduction	Cr(VI), nitrate	6 m (20 ft)/ greater than 6 m (20 ft)	Chemical reductant (for example, calcium polysulfide, hydrogen sulfide gas, ferrous sulfate, zero valent iron) and biological carbon source (for example, molasses, sodium lactate, emulsified oil) are applied in combination to the subsurface to treat contaminants within vadose zone.	Moderate Amendments follow source release pathways. Combined chemical and biological might improve performance.	Moderate Depends on delivery method. Localized temporary generation of secondary byproducts may occur. May temporarily mobilize COPCs (in first pore volume) toward groundwater. Handling chemical reductants is a health and safety concern.	Low/Moderate	Moderate	GHG and energy for production and delivery of substrate/reagent. Depends on which substrate is used.	Not Retained	More challenging to implement and costly as compared to biological reduction.
			Gaseous Ammonia Injection	Mobile COPCs	6 m (20 ft)/ greater than 6 m (20 ft)	One of a number of possible gaseous reagents that are being investigated (along with in situ gaseous reduction). It involves the injection of ammonia gas to increase pH to dissolve silica. The pH naturally decreases to ambient conditions over time and aluminosilicate minerals precipitate and possibly coat and immobilize various contaminants.	Unknown Effectiveness is being studied as part of a laboratory-scale investigation.	Unknown Implementation is unknown at a full-scale level. Containment of injected gases in the shallow vadose zone may be an issue.	Unknown Technology evaluation has been limited to laboratory tests.	Unknown Technology evaluation has been limited to laboratory tests.	GHG emissions from injection activities.	Not Retained	Evaluation of results from the ongoing treatability study is needed before making a decision regarding its full-scale use at the Hanford Site. This technology could be evaluated as a remedial alternative later.
			Reductive Dechlorination Using Zero Valent metals and bioremediation	PCBs	6 m (20 ft)/ greater than 6 m (20 ft)	Zero valent metals have the potential to reductively dechlorinate PCBs. Metals include iron, palladium, and other combinations. The contaminated soil and the metals are mixed in some fashion to allow the reactions to occur. Bioremediation, via the addition of an organic substrate, is a very similar process and can be combined with zero-valent metal addition.	Unknown Very little published testing results are available.	Moderate Could be implemented by soil mixing with conventional excavation equipment if the contamination is shallow.	High	Low	No associated cost.	Not Retained	Reductive dechlorination using zero-valent metals and bioremediation are not proven technologies and were not retained for further consideration. More field studies must be conducted to test methods of bioaugmentation and biostimulation for PCB-dechlorinators.

Table 8-6. Screening Table—Technologies for Radionuclides, Hexavalent Chromium, Other Metals, and Organic Compounds in the Vadose Zone—100-D/H

General Response Actions	Remedial Technology		Process Option	COPC Applicability[a]	Depth Range[b]	Description	Relative Effectiveness	Relative Implementability	Relative Capital Cost	Relative O&M Cost	Sustainability[c]	Retained/ Not Retained[d]	Screening Comment
In Situ Treatment (cont.)	In Situ Treatment via Reagent (cont.)	Delivery Method	Gaseous Reduction with Chemical Reductant or Biological Substrate	Cr(VI), nitrate	6 m (20 ft)/ greater than 6 m (20 ft)	A gaseous mixture of chemical reductants (for example, hydrogen sulfide) or biological substrate (for example, butane) is injected into and drawn through the vadose zone to reduce Cr(VI). Research is underway to evaluate other reagents to immobilize contaminants.	Unknown Soil heterogeneity will result in preferential flow and limit treatment effectiveness of lower permeability soil.	Unknown Vapor extraction wells are installed around injection well at a radial spacing of approximately 4.6 m (15 ft). Large numbers of wells are required. Because of health and safety risks, monitoring and emergency response plan are required for transporting, storing, and handling.	Unknown	Unknown	GHG emissions from injection activities.	Not Retained	Evaluation of results from the ongoing treatability study is needed before making a decision regarding its full-scale use at the Hanford Site. This technology could be evaluated as a remedial alternative at a later date.
			Mixing with Conventional Excavation Equipment	Depends on type of reagent used	6 m (20 ft)	Use of conventional excavation equipment (backhoes, excavators, front-end loaders) to mix amendments into the soil.	High Agents are uniformly mixed with soil column, providing good contact and reaction between COPC and chemical.	Moderate Simple technology. Dust mitigation techniques will need to be implemented to control/prevent mechanical dispersion of contaminants.	Low/Moderate	Low No associated cost.	GHG emissions from machinery.	Not Retained	Not retained in favor of surface infiltration. Could be used if shallow mobile contaminants are identified in the future.
			Deep Soil Mixing (Vertical/Horizontal)	Depends on type of reagent used	6 m (20 ft)/ greater than 6 m (20 ft)	Large mixing augers (1.5 to 3 m [5 to 10 ft] in diameter) or horizontally rotating heads are used to blend and homogenize reactants with soil. The reactants may be chemical reductants, biological substrate, or solidification/stabilization agents.	High Chemical agents are uniformly mixed with soil column, providing good contact and reaction between COPC and chemical. Cement or clay can also be mixed with the chemical slurry to reduce the hydraulic conductivity and leachability of the soil.	Low/Moderate Implementation will be more challenging in gravelly/cobbly lithology. Although deep soil mixing has been performed to depths of 30 m (100 ft) bgs, most field applications have been limited to approximately 15 m (50 ft) bgs.	High	Low No associated cost.	GHG emissions from machinery	Retained	Deep soil mixing implementability will be limited by site conditions and required depth of treatment.
			Foam Delivery of Reagents	Depends on type of reagent used	6 m (20 ft)/ greater than 6 m (20 ft)	Injection of a foam into the vadose zone. The foam is a mixture of a surfactant solution and a reagent, such as phosphate or calcium polysulfide. The foam increases the horizontal migration of the reagent away from the injection well.	Unknown Technology evaluation has been limited to laboratory scale tests. The stability of the foam, which will dictate the well spacing, is unknown, as is the ability of the foam to sweep a large volume of the vadose zone.	Unknown Technology evaluation has been limited to laboratory scale tests.	Unknown Technology evaluation has been limited to laboratory scale tests.	Unknown Technology evaluation has been limited to laboratory scale tests.	GHG emissions from well installation, development, and injection activities; waste generation from soil cuttings.	Not Retained	Evaluation of results from the ongoing treatability study is needed before making a decision regarding its full-scale use at the Hanford Site. This technology could be evaluated as a remedial alternative later.

Table 8-6. Screening Table—Technologies for Radionuclides, Hexavalent Chromium, Other Metals, and Organic Compounds in the Vadose Zone—100-D/H

General Response Actions	Remedial Technology		Process Option	COPC Applicability[a]	Depth Range[b]	Description	Relative Effectiveness	Relative Implementability	Relative Capital Cost	Relative O&M Cost	Sustainability[c]	Retained/ Not Retained[d]	Screening Comment
In Situ Treatment (cont.)	In Situ Treatment via Reagent (cont.)	Delivery Method (cont.)	Gas Delivery of Reagents	Depends on type of reagent used	6 m (20 ft)/ greater than 6 m (20 ft)	A gaseous mixture of chemical reagent is injected into and drawn through the vadose zone to reduce mobile COPCs.	Unknown Soil heterogeneity will result in preferential flow and limit treatment effectiveness of lower permeability soil.	Unknown Vapor extraction wells are installed around injection well at a radial spacing of approximately 4.6 m (15 ft). Large numbers of wells are required. Because of health and safety risks, monitoring and emergency response plan are required for transporting, storing, and handling.	Unknown	Unknown	GHG emissions from well installation, development, and injection activities; waste generation from soil cuttings.	Not Retained	Evaluation of results from the ongoing treatability study is needed before making a decision regarding its full-scale use at the Hanford Site. This technology could be evaluated as a remedial alternative later.
			Injection Wells (Horizontal)	Depends on type of reagent used	6 m (20 ft)/ greater than 6 m (20 ft)	Delivery of amendments using horizontal wells. Wells are installed using horizontal drilling techniques.	Low/Moderate Effectiveness can be hindered by nonuniform amendment distribution. Soil heterogeneity will result in preferential flow and limit treatment effectiveness of lower permeability soil. Multiple injections could be required.	Low Implementation is challenging in gravelly/cobbly lithology. Lithology would also pose challenges with maintaining target depth and alignment with horizontal drilling. A pilot test of this technology encountered signification implementation challenges.	Moderate/High	Low	GHG emissions from well installation, development, and injection activities; waste generation from soil cuttings.	Not Retained	Testing at the Hanford Site has not been successful.
			Injection Wells (Vertical)	Depends on type of reagent used	6 m (20 ft)/ greater than 6 m (20 ft)	Delivery of amendments using conventional vertical wells.	Low/Moderate Effectiveness can be hindered by nonuniform amendment distribution. Distribution of liquid amendments is highly ineffective because of gravelly/cobbly lithology. Distribution in lower-permeability soil can be enhanced with the use of shear-thinning fluids.	Moderate Radius of influence likely to be low, requiring large number of injection wells.	Moderate/High	Low	GHG emissions from well installation, development, and injection activities; waste generation from soil cuttings.	Retained	Delivery of liquid substrates in vadose zone through surface infiltration.
			Jet Grouting	Depends on type of reagent used	6 m (20 ft)/ greater than 6 m (20 ft)	High-pressure injection of reactive slurry into soil to mix the soil hydraulically with the slurry. Fluidization of the soil is preferred.	Low/Moderate While jet grouting is capable of reaching the required treatment depth, jet grouting is not likely to achieve uniform distribution or a radius of influence greater about 1.5 m (5 ft). Jet grouting of apatite and phosphate was pilot tested at 100-N for shallow, limited application. Altered/ decreased permeability of soil resulted from amendment precipitation and/or liquefaction of fine-grained sediment fractures.	Low/Moderate Implementation will be more challenging in gravelly/cobbly lithology. Jet grouting has been performed to depths of up to 91 m (300 ft). Many closely spaced injection points (approximately 1.5 m [5 ft] spacing) will be required.	High Limited radius of influence would make jet grouting cost-prohibitive over a large area.	Low	GHG emissions from injection activities.	Retained	Could be considered in the future if technology develops. Currently, jet grouting has potentially limited effectiveness.

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General Response Actions	Remedial Technology		Process Option	COPC Applicability[a]	Depth Range[b]	Description	Relative Effectiveness	Relative Implementability	Relative Capital Cost	Relative O&M Cost	Sustainability[c]	Retained/ Not Retained[d]	Screening Comment
In Situ Treatment (cont.)	In Situ Treatment via Reagent (cont.)	Delivery Method (cont.)	Surface Infiltration	Depends on type of reagent used	6 m (20 ft)/ greater than 6 m (20 ft)	Reagent is applied to ground surface to treat contaminants within vadose zone. Surface infiltration can be done through drip irrigation and shallow basin systems. Systems are generally designed to be 30.5 cm (12 in.) below the surface and covered to be protected.	Moderate/High Amendments follow source release pathways. Distribution not likely to be uniform.	High Surface infiltration systems are simple to install and accessible for O&M.	Low	Low	Limited infrastructure. GHG emissions from production and delivery of substrate.	Retained	Retained for liquid substrates.
			Void Filling/Grouting	Dependent on type of reagent used	NA (Pipelines/ Structures)	Grouting for solidification of buried wastes. Void grouting is considered for filling large voids, specifically pipelines and structures.	High Established and commonly used technology for removing voids in pipelines and structures.	Moderate/High Established and commonly used technology for removing voids in pipelines and structures. Pipe branch lines/breaks need to be identified. Implementability can be more challenging and costly with long or large diameter pipelines.	Low	Low No associated cost.	GHG and energy for production and delivery of grout used.	Retained	Retained for structures and pipelines near river or groundwater monitoring wells.
	In Situ Treatment Other		Soil Blending	All except mobile COPCs	Depends on excavation method.	Contaminated soils are mechanically blended with clean soil or fill to reduce effective contaminant concentrations.	High	High Conventional equipment can be used.	Moderate/	Low No associated cost.	GHG and energy for tilling equipment.	Not Retained	Not effective since it relies on contaminant dilution.
			Desiccation	Mobile COPCs	greater than 6 m (20 ft)	Remediation by injecting hot dry air and withdrawing moist air from soil, immobilizing contaminants by preventing their aqueous-phase transport.	Unknown A treatability test for this technology will be conducted for waste sites in the Central Plateau contaminated with technetium-99. Theoretically, desiccation would reduce moisture content in the vadose zone. Reduction of COPC migration would be effective until the soil is re-wetted. The technology is not effective in the long term without concurrent infiltration control.	Unknown Implementation requires installation of injection and extraction wells, which are proven technology. However, there is uncertainty related with the number of wells, well spacing, and well configuration details required for optimal field/full-scale implementation. Would also require implementation of infiltration control.	Unknown	Unknown	GHG and energy for air injection. Waste generation from soil cuttings for well installation.	Not Retained	Evaluation of results from the ongoing treatability study is needed before making a decision regarding its full-scale use at the Hanford Site. This technology could be evaluated as a remedial alternative later.
			Thermal Desorption	Organics	6 m (20 ft)/ greater than 6 m (20 ft)	Direct application of heat (for example, using electrical heating elements, electrical resistive heating, injection of hot air, steam or hot water, radio frequency) to increase the temperature of soil and destroy or volatilize organic compounds. VOC capture required.	Moderate/High Technology can achieve rapid removal/destruction of a mix of volatile and semivolatile organics, and achieve low residual concentrations.	Low Technology is applied using vertical drilling methods, and requires a spacing of 1.5 to 3 m (5 to 10 ft). Recovery of COPC vapors will require soil vapor extraction network and vapor barrier over entire treatment area.	High	Low No associated cost.	GHG and energy for production of heat and vapor recovery; waste generation from soil cuttings.	Not Retained	Mechanically complex challenging to implement.

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In Situ Treatment (cont.)	In Situ Treatment Other (cont.)	Vitrification	All	6 m (20 ft)/ greater than 6 m (20 ft)	Thermal treatment process that converts soil and other materials to glass matrix. Contaminants are incorporated into the glass structure, which is generally strong, durable, and resistant to leaching.	High Metals and radionuclides are retained within the treated soil, which is generally resistant to leaching.	Low High complexity of equipment required. Process uses an electric current to melt soil or other earthen materials at extremely high temperatures (1,600 to 2,000 °C or 2,900 to 3,650 °F). It is also important to account for safety considerations from exposure to high heat.	High	Low No associated cost.	GHG and energy for heat generation. High energy requirements to sustain required heat.	Not Retained	Complex and challenging to implement.
		Soil Flushing – Vadose Zone, Water	Contaminants with high to moderate solubility (for example, Cr(VI) and nitrate)	6 m (20 ft)/ greater than 6 m (20 ft)	Clean or treated water is applied to the ground surface or in infiltration trenches to flush contaminants out of the vadose zone to the water table, where it would be captured and treated.	Moderate Water follows source release pathways, but contaminants that remain in adsorbed phase will not be treated. May create a larger groundwater problem if the groundwater capture is not effective.	High Drip irrigation system or trenches are simple to install and accessible for O&M.	Low/Moderate	Low	GHG and energy for installation.	Retained	
		Phytoremediation	Bio-available Metals and Organics	6 m (20 ft)	Phytoremediation uses plants and their associated rhizospheric microorganisms to remove, degrade, or contain contaminants.	Low Phytoremediation is only effective when plants are active, thus the technology is not effective during the winter. Phytoremediation only treats soils to the approximate depth of the plant roots, and is only appropriate for low concentrations of contaminants. It is a slow process that is applied over long periods, that is, years or decades. Many metals and radionuclides are only taken up by the plants and not transformed to innocuous forms.	Moderate Involves large land requirements, and considerable work would be required to make a plot of land at Hanford Site suitable for plant growth. If used to treat contaminants that are merely taken up and not transformed to innocuous forms, plants would need to be disposed of elsewhere to avoid ultimately returning the contaminants to the soils they came from. Concerns about contaminants in the plants entering the food chain may need to be addressed.	Low	Low	GHG and energy for installation, and potential disposal of harvested plants containing metals. Implementation of phytoremediation could lead to a CHG reduction credit.	Not Retained	Phytoremediation would only be effective for low concentrations of contaminants in shallow soils over long periods, and many metals and radionuclides would accumulate in the plants and would not actually be treated, posing risks to ecological receptors.

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General Response Actions	Remedial Technology	Process Option	COPC Applicability[a]	Depth Range[b]	Description	Relative Effectiveness	Relative Implementability	Relative Capital Cost	Relative O&M Cost	Sustainability[c]	Retained/ Not Retained[d]	Screening Comment
Containment	Surface Barrier	Surface Barriers (for example, Modified RCRA Subtitle C and/or D Barrier, Asphalt/Concrete barrier, Vegetative barrier [Evapotranspiration barrier], Hanford Barrier)	All	6 m (20 ft)/ greater than 6 m (20 ft)	Surface barriers are generally designed to be impermeable to prevent surface water infiltration through the vadose zone and limit contaminant leaching to groundwater. Surface barriers may also prevent direct contact to contaminants. Modified RCRA Subtitle C Barriers are designed for hazardous waste, category 3 and category 1 (mixed) low-level waste. Modified RCRA Subtitle D Barriers are designed for non-radiological and non-hazardous solid waste, or category 1 low level waste where hazardous constituents are not present. Evapotranspiration barriers consist of a fine-grained soil layer overlying a relatively coarse-grained soil layer designed to functionally increase the water-holding capacity. Asphalt/concrete barriers can be placed around structures to remain in place (for example, reactors) in the short term (75 years) to promote drainage, prevent infiltration into possible sources below the reactors, and prevent exposure to contaminated soil. The Hanford Barrier design was developed for sites containing low-level waste greater than Class C, and/or significant inventories of transuranic constituents.	Moderate/High Leaching of near-surface source COPCs will be controlled, but residual COPCs in capillary fringe and deeper vadose zone pore water will continue to impact groundwater because of water table fluctuation. Prevention of direct contact will depend on specific design. Effectiveness for asphalt caps is high in the short term; for increased effectiveness, barrier needs to be properly sealed, given that asphalt and concrete are permeable.	High No technical or administrative challenges are associated with implementing asphalt/concrete caps (high implementability). Modified RCRA Subtitle C and/or D Barrier and Evapotranspiration barriers are simple to install. Biointrusion may need to be considered as part of the barrier/cap design.	Low/High Hanford Barrier (High); Modified RCRA Subtitle C and/or D Barrier (Moderate); Asphalt/ Concrete Cap and Evapotranspiration Barrier (Low)	High Dependent on type of barrier and depth of contamination.	GHG and energy for installation. Continued impact to soil resources.	Retained	
	Subsurface Barriers	Jet grouting, soil freezing, or wire saw barriers	All	6 m (20 ft)/ greater than 6 m (20 ft)	Barriers placed beneath the contaminated zone to limit further migration. Jet grouting is as discussed above at one specific depth. Soil freezing involves placement of cooling media distribution systems into the subsurface to freeze a soil layer below the contamination. Wire saw barrier involves cutting a thin horizontal trench that is filled with grout using a diamond wire saw. The saw is placed in an excavation around the soil mass to be contained.	Low Significant uncertainty on the completeness of the barrier with all methods.	Low Would be difficult or impossible to implement at the Hanford Site because of presence of gravels and cobbles, and/or the depth of application.	High	Low	Large amount of wastes would be generated during installation and GHG and energy for installation.	Not Retained	Difficult to implement.
	Compaction	Dynamic compaction	All	6 m (20 ft)/ greater than 6 m (20 ft)	Dynamic compaction is used for consolidation of soils and buried wastes, and can be used to minimize the potential subsidence for a subsequent barrier. The process involves dropping a weight from a predetermined height onto the area to be compacted.	Moderate/High Effective at removing void spaces and compacting surface soils, where voids exist around buried waste. Not effective for native soils. Not effective for treatment of hazardous wastes.	Moderate Simple and widely used technology.	Low	Low No associated cost.	GHG and energy for installation.	Not Retained	Not effective for treatment of hazardous wastes.

Table 8-6. Screening Table—Technologies for Radionuclides, Hexavalent Chromium, Other Metals, and Organic Compounds in the Vadose Zone—100-D/H

General Response Actions	Remedial Technology	Process Option	COPC Applicability[a]	Depth Range[b]	Description	Relative Effectiveness	Relative Implementability	Relative Capital Cost	Relative O&M Cost	Sustainability[c]	Retained/ Not Retained[d]	Screening Comment

a. Indicates the contaminants that can be addressed by a technology based on geochemical properties. A COPC Applicability of "All" indicates implementation of a technology is not dependent on the nature of a contaminant.

b. Depth range is based on practical limitations of implementing the given technology.

c. Sustainability includes potential impacts to the environment that could arise from implementing this technology (for example, GHG emissions, waste generation, water use and resource impacts, and energy use).

d. Additional details on technologies not retained are provided in Appendix I.

e. Ex situ treatment does not include treatment done for disposal at ERDF or an approved offsite landfill. Treatment performed at ERDF or at the waste site as required to meet disposal restrictions is assumed to be part of the "disposal to ERDF" or "other approved EPA landfill" process options.

GHG = greenhouse gas

NA = not applicable

O&M = operations and maintenance

Table 8-7. Screening Table—Technologies for Hexavalent Chromium and Other COPCs in Groundwater—100-D/H

General Response Actions	Remedial Technology	Process Option	COPC Applicability[c]	Description	Relative Effectiveness		Relative Implementability	Relative Capital Cost	Relative O&M Cost	Sustainability[b]	Retained/Not Retained [a]	Screening Comment
No Action	No Action	No Action	All	No remedial actions taken.	Low/High		High	Low	Low	Little impacts	Retained	Retained per the NCP (40 CFR 300).
					No remedial actions are taken, but effectiveness could be high if risk is previously mitigated.							
MNA	MNA	MNA	All	Relies on natural attenuation processes to manage the contamination on site. Monitor groundwater plume to track natural attenuation processes until RAOs are achieved. Typically combined with other technologies that manage the source areas and mitigate exposure. Natural attenuation processes include: • Biological reduction – processes where naturally occurring microorganisms, such as yeast, fungi, and bacteria, break down target substances into less toxic or non-toxic substances. • Chemical reduction – geochemical process where natural reductants in sediments reduce contaminants into less toxic or non-toxic substances. • Adsorption - occurs in groundwater, as dissolved chemicals are removed from the solution and attach to soil particles. • Dispersion - the spreading of a chemical in groundwater outward from its expected path. As groundwater moves through different soil types and geological features, it travels at different velocities. This creates mechanical mixing, so groundwater spreads away from source areas into wider plumes. • Dilution – the decrease in the chemical concentration in a fluid caused by mixing with a fluid containing a lower concentration. • Radioactive decay – spontaneous disintegration of the nucleus of radionuclide resulting in reduction in radionuclide activity.	biological reduction	Low/Moderate	High	Low/Moderate	Low/Moderate	Little impacts	Retained	
					chemical reduction	Moderate/High						
					adsorption	Low/Moderate						
					dispersion	High						
					dilution	High						
					radioactive decay	Low/High depending on decay half-life of radioactive contaminant						

Table 8-7. Screening Table—Technologies for Hexavalent Chromium and Other COPCs in Groundwater—100-D/H

General Response Actions		Remedial Technology	Process Option	COPC Applicability[c]	Description	Relative Effectiveness	Relative Implementability	Relative Capital Cost	Relative O&M Cost	Sustainability[b]	Retained/Not Retained [a]	Screening Comment
Pump-and-Treat	Collection	Extraction	Groundwater Extraction System	All dissolved	Operation of existing and/or new groundwater extraction wells.	Moderate/High Pump-and-treat is a proven treatment technology for contaminants in groundwater, although there is some uncertainty as to its ability to achieve standards everywhere.	High System already in place for Cr(VI), but existing well field may not be appropriate to capture all of the COPCs.	Low System exists	Moderate/High	Energy consumption and GHG emissions from pumping systems.	Retained	
	Ex Situ Treatment	Chemical	Ion Exchange	Cr(VI), nitrate, strontium-90	Ions from the aqueous phase are removed by exchange with innocuous ions on the exchange medium.	Moderate/High Effective for Cr(VI) and nitrate treatment. Variable, depending on COPC.	High Vendors and equipment readily available. Currently used at the site.	Low/Moderate Systems exist, but may need to be expanded or upgraded for other COPCs	Moderate/High	Waste generation from ion exchange resin disposal or regeneration. Energy consumption from process equipment.	Retained	
			Chemical Reduction/ Softening and Precipitation	Cr(VI)	Dissolved COPCs are transformed into an insoluble solid, which is removed by flocculation, sedimentation, and filtration. COPCs are removed with the sludge.	Low/Moderate Effective for Cr(VI) treatment.	Moderate/High Vendors and equipment readily available, but no experience with the technology at Hanford. Large volume of sludge would be produced.	Moderate/High	Moderate	Waste generation from chemical precipitation. Energy consumption from process equipment.	Not Retained	For Cr(VI), implementability challenges given large sludge volume. In addition, ion exchange treatment systems are in place.
			Electrocoagulation	Cr(VI)	Relies on electrochemical generation of ferrous iron. The ferrous iron reduces metals that are susceptible to reduction and converts them to insoluble solids, which are removed by sedimentation and filtration.	Low/Moderate Not widely used for Cr(VI) removal. Pilot testing at the site had challenges.	Low/Moderate Additional development and testing would be required. Potential negative impacts on reinjection of water.	Moderate/High	Moderate	Waste generation from chemical precipitation. Energy consumption from process equipment.	Not Retained	Implementability challenges. In addition, the ion exchange treatment systems are in place.
		Biological	Wetlands	Cr(VI), nitrate	Extracted groundwater is pumped to a constructed wetland where contaminants are biologically reduced, or taken up by plants and algae. Petroleum aerobically degraded.	Moderate/High Effective for nitrate, but additional research/pilot testing is required to verify effectiveness for other COPCs.	Low/Moderate May require large surface area for extended period.	Moderate Depends on land and construction requirements	Low	Little impacts, except for land required	Not Retained	Performance uncertainty will require research to determine effectiveness. Not retained since ion exchange treatment system are in place.
			Subgrade Bioreactors	Cr(VI), nitrate	Extracted groundwater is pumped into a lined excavated area that has been backfilled with organic media (for example, wood mulch with zero valent iron). Cr(VI) and nitrate are biologically reduced as it passes through the media. A second stage aeration/filtration stage could be provided to remove any biological byproducts (for example, iron), petroleum, and solids before infiltrating or injecting back to groundwater.	Moderate/High Effective for nitrate, but treatability testing is required to verify effectiveness for other COPCs.	Moderate/High Excavation and backfilling is easy to implement. Piping can be incorporated into the design to facilitate future delivery of liquid carbon sources (for example, vegetable oil). Treatability testing required to verify implementability.	Low/Moderate Depends on land requirements	Low	Impacts include spent media disposal and land required	Not Retained	Has not been demonstrated on a full scale for Cr(VI) or nitrate remediation.

Table 8-7. Screening Table—Technologies for Hexavalent Chromium and Other COPCs in Groundwater—100-D/H

General Response Actions		Remedial Technology	Process Option	COPC Applicability[c]	Description	Relative Effectiveness	Relative Implementability	Relative Capital Cost	Relative O&M Cost	Sustainability[b]	Retained/Not Retained [a]	Screening Comment
Pump-and-Treat (cont.)	Ex Situ Treatment (cont.)	Biological (cont.)	Bioreactors	Cr(VI), nitrate	Groundwater is amended with electron donor (carbon source) and passes through a matrix (fixed bed, fluidized bed, or membranes) with microbial films, where contaminants are biologically reduced. Effluent is oxygenated, filtered, and amended before recharge into the ground.	Low/Moderate Bioreactors commonly used for nitrate removal, but less commonly for Cr(VI) reduction. Little experience with other COPCs.	Moderate/High Vendors and equipment readily available, but no current experience with the technology at the Hanford Site.	High	Moderate	Waste generation from biological sludge. Energy consumption from process equipment.	Not Retained	Performance uncertainty for Cr(VI). Considering large and complex system requirements for nitrate removal, and since in situ bioremediation could be used, ex situ bioreactors have not been retained.
			Phytoremediation	Cr(VI), nitrate, strontium-90	Use of plants and their associated rhizospheric microorganisms to remove, reduce/degrade, or contain chemical contaminants in soil or groundwater. Contaminants in groundwater can also be removed by applying it as irrigating water for plants.	Low/Moderate Low/Moderate for Cr(VI). Additional research/pilot testing is required to verify effectiveness for site conditions. Could be used as a barrier approach, but there would be challenges with the depth to the water table even close to the river. Commonly used for nitrate removal. Plants used for remediating radionuclides would require harvesting.	Low/Moderate Requires large surface area for plants. Potential cultural challenges with implementation near river.	Low	Low	Impacts include land required and potential disposal of harvested plants containing radionuclides.	Not Retained	Would only be effective for low concentrations of contaminants where groundwater is shallow over long periods, or when applied as irrigation water. Many metals and radionuclides would accumulate in the plants and not actually be treated, posing risks to ecological receptors.
		Physical	Membrane Separation (reverse osmosis)	All	Water pressure is used to force water molecules through a very fine membrane, leaving the contaminants behind. Purified water is collected from the "clean" or "permeate" side of the membrane, and water containing the concentrated contaminants is disposed.	High With the appropriate design, reverse osmosis can be effective for almost any compound.	Low/Moderate Vendors and equipment readily available, although additional site specific testing would be required. Pretreatment likely necessary, and a large volume of brine would be produced that would need to be treated and handled.	High	High	Waste generation in the form of brine and high energy use. Energy consumption from process equipment.	Not Retained	Implementability challenges from large volumes of brine produced that would require reduction and disposal.
	Discharge	Onsite Discharge	Groundwater Injection Wells	All	Treated groundwater is injected into onsite wells.	High Will enhance contaminant flushing, hydraulic control and capture of plume.	High Readily implementable at the site, currently used in existing pump-and-treat system. The wells may be subject to clogging because of the buildup of chemical precipitates or microbial biofouling.	Low/Moderate	Low/Moderate	Waste generation from soil cuttings for well installation.	Retained	
			Surface Infiltration	All	Treated groundwater is infiltrated into onsite trenches.	High Effective means of disposal and may enhance contaminant flushing, hydraulic control and capture of plume if trenches can be located appropriately.	Moderate/High Infiltration would be easy to engineer and implement.	Low Trenches are lower cost than wells	Low/Moderate	Little impacts	Retained	

Table 8-7. Screening Table—Technologies for Hexavalent Chromium and Other COPCs in Groundwater—100-D/H

General Response Actions		Remedial Technology	Process Option	COPC Applicability[c]	Description	Relative Effectiveness	Relative Implementability	Relative Capital Cost	Relative O&M Cost	Sustainability[b]	Retained/Not Retained [a]	Screening Comment
Pump-and-Treat (cont.)	Discharge (cont.)	Offsite Discharge	Discharge to Surface Water	All	Discharge of treated groundwater directly to the river at an outfall.	High	Low	Low	Low	Little impacts	Not Retained	Discharge to the river is not retained in favor of injection for hydraulic control in remediating the plume and minimize dewatering of the aquifer.
						Effective means of treated water disposal.	Although surface water discharge is commonly practiced for treated wastewater, no new outfalls are allowed on the Hanford Reach National Monument.		Little or no maintenance required			
In Situ Treatment	Reagent Approach	Chemical	Chemical Reduction	Cr(VI)	Subsurface delivery of chemical reductants (such as calcium polysulfide) within plume to stimulate reduction of contaminant.	Moderate	Moderate	Moderate/High	Moderate	Waste generation from soil cuttings for well installation. GHG and energy for production and delivery of chemicals	Not Retained	May be more challenging to implement and costly as compared to biological reduction.
						Chemical reductants instantly reactive, thus strongest reduction achieved near injection well, requiring tighter spacing of injection wells. Recirculation approach may increases size of reducing zone, and allows broader well spacing. Iron and sulfate reduction increases reductive capacity of subsurface, which makes the formation less sensitive to rebound.	May require large number of wells.	Dependent on number and type of wells. Likely higher capital cost compared to in situ biological				
			Chemical Stabilization	strontium-90	Subsurface delivery of chemical reagents (such phosphate) in a regular pattern of wells in the aquifer to sequester the contaminants. Chemical reactions are induced between the stabilizing agent and contaminant to reduce mobility.	Moderate/High	Moderate	Moderate/High	Moderate	Waste generation from soil cuttings for well installation. GHG and energy from chemical production and transport.	Retained	Retained for treatment of strrontium-90 at locations of continuing source to groundwater.
						Currently being implemented at 100-N in a barrier approach for strontium-90 with favorable results. Achieving even distribution may be difficult.	Requires large number of wells to cover a large area.	Function of number of injection wells required	Periodic reinjection may be required.			
		Biological	Biological Treatment (Anaerobic)	Cr(VI), nitrate	Subsurface delivery and recirculation of various organic substrates in a regular pattern of wells in the aquifer to stimulate anaerobic bioreduction of Cr(VI) and reduction of nitrate. Cr(VI) and nitrate in groundwater that is reinjected would be reduced in situ.	High	Moderate/High	Moderate/High	Moderate	Waste generation from soil cuttings for well installation. GHG and energy for production and delivery of substrate. Depends on which substrate is used.	Retained	
						Reactive life of biological electron donors is longer than chemical reductants so that reactive strength is maintained over relatively longer distances compared to in situ chemical treatment. Iron and sulfate reduction increases reductive capacity of subsurface, which makes the formation less sensitive to rebound.	Requires large number of wells to cover a large area.	Dependent on number and type of wells				

Table 8-7. Screening Table—Technologies for Hexavalent Chromium and Other COPCs in Groundwater—100-D/H

General Response Actions		Remedial Technology	Process Option	COPC Applicability[c]	Description	Relative Effectiveness	Relative Implementability	Relative Capital Cost	Relative O&M Cost	Sustainability[b]	Retained/Not Retained [a]	Screening Comment
In Situ Treatment (cont.)	Reagent Approach (cont.)	Chemical/ Biological	Hydrogen or other Organic Gas Sparging	Cr(VI)	Injection of biodegradable organic gasses (for example, propone or butane) or hydrogen into sparge wells that are screened below the water table.	Low Distribution of gasses likely to be poor under local heterogeneous geologic conditions. Has not been demonstrated for Cr(VI).	Low The radius of influence around each sparge well is likely to be low, so a large number of wells would be required. Safety challenges exist because of residual explosive gasses that may accumulate.	High Large number of wells would be required	Moderate	Waste generation from soil cuttings for well installation. GHG and energy for production and delivery of chemicals	Not Retained	Challenge in the distribution of the gases and safety risk associated with using explosive gases.
			Combination of Biological & Chemical Substrates	Cr(VI), nitrate	Subsurface delivery and recirculation of both chemical reductants and electron donors within plume to stimulate chemical and anaerobic biological reduction of Cr(VI).	High Chemical reductants could be used to treat smaller hot spot areas, while biological reductants could be used to sustain treatment over larger dilute plume areas. Recirculation approach increases the size of reducing zone, and allows broader well spacing. Iron and sulfate reduction increases reductive capacity of subsurface. Less sensitive to rebound from residual sources because of residual reactive phase.	Moderate Recirculation will likely be limited by extraction rate – addition of fresh water can be used to enhance coverage around injection wells. The formation of secondary byproducts may impact restoration to beneficial use.	Moderate/High Dependent on number and type of wells	Moderate/High	Waste generation from soil cuttings for well installation. GHG and energy for production and delivery of chemicals	Retained	May be more challenging to implement and costly as compared to biological reduction.
		Physical	Flushing – Saturated Zone, Water	Cr(VI), nitrate	Clean/treated water is injected to flush out contaminated groundwater to expedite remediation of plumes. Would be component of a pump-and-treat system.	Moderate/High The extraction wells system should be able to capture any contaminants mobilized. However, performance will depend on residual contamination in lower permeability layers.	High Standard wells or infiltration trenches used for injection.	Moderate Costs for wells and piping	Low	GHG and energy for installation. Waste generation from soil cuttings for well installation.	Retained	
	Delivery Method	Surface Infiltration	Surface Infiltration	NA	Trenches, French drains, or drip irrigations systems are used to apply water or reagents.	High Effective with appropriate design, installation, and maintenance.	Moderate Location of vadose zone contamination in relation to the water table needs to be known.	Low	Moderate	Less GHG and energy for installation	Retained	
		Groundwater circulation wells	Groundwater circulation wells	NA	Installation of wells with two screened zones. Groundwater is typically pumped out of the formation from lower screen zone, and injected back into the formation in the upper zone. A circulation pattern is created in the formation. The groundwater can be stripped inside the well to remove VOCs, or the wells can be used to deliver reagents.	Low/Moderate The establishment of a reasonable circulation pattern depends on the formation characteristics. The low permeability lenses present in some locations may be problematic. Very high permeability may result in a small radius of influence so more wells will be required.	Low/Moderate A large number of wells may be required.	Moderate Depends on the number of wells required	Moderate	Waste generation from soil cuttings for well installation. GHG and energy for operation.	Not Retained	Asymmetrical groundwater flow and groundwater flow short-circuiting, may limit the effectiveness of groundwater circulation wells.
		Vertical Wells	Vertical Wells	NA	Standard vertical wells are used to inject water or reagents.	High Effective with appropriate design, installation, and maintenance.	High Uses extensively at Hanford.	Moderate/High Wells at Hanford are generally expensive.	Moderate	Waste generation from soil cuttings, GHG and energy for installation	Retained	

Table 8-7. Screening Table—Technologies for Hexavalent Chromium and Other COPCs in Groundwater—100-D/H

General Response Actions		Remedial Technology	Process Option	COPC Applicability[c]	Description	Relative Effectiveness	Relative Implementability	Relative Capital Cost	Relative O&M Cost	Sustainability[b]	Retained/Not Retained [a]	Screening Comment
In Situ Treatment (cont.)	Delivery Method	Horizontal Wells	Horizontal Wells	NA	Horizontally drilled wells are used to inject water or reagents.	Low/Moderate Uncertain performance.	Low Pilot test was not successful.	Moderate/High Costs are high but fewer wells are required	Moderate	Waste generation from soil cuttings, GHG and energy for installation	Not Retained	Pilot test was not successful.
Containment		Physical	Containment Wall (slurry wall or grout wall)	All	Slurry or grout wall barriers consist of a vertical barrier perpendicular to the groundwater flow direction, partially filled with bentonite slurry, grout, or other low permeability material. The barrier is typically keyed into a lower permeability zone. The slurry/grout could be jet injected, mixed with the soils using large augers, or excavated.	Moderate Effectiveness is dependent on the continuity of the wall and the ability to key into the RUM, which will be difficult to achieve because of depth. Does not reduce toxicity or volume of contaminants by itself. This technology requires groundwater extraction to control groundwater pressures from building up behind the barrier and potentially damaging the barrier or causing groundwater to flow under, over, or around the barrier.	Low Installation of wall through cobbles and boulders to key into the RUM is very difficult and cost prohibitive. Driven sheet piles near the river have been attempted but failed because of the presence of cobbles.	High	Low/Moderate	GHG and energy for installation, waste from trench spoils.	Not Retained	Not required since there is an existing hydraulic containment system and not likely to be implementable.
		Chemical/ Biological	Reactive Chemical Barrier (ISRM)	Cr(VI)	Subsurface delivery and/or recirculation of chemical reductants along cross-gradient rows transecting plume. Residual reducing chemicals are retained in the aquifer matrix so Cr(VI) is passively removed as groundwater moves through the treatment zone barriers. Sodium dithionite or zero valent iron may be used as reductants. ISRM is currently in use at 100-D.	Moderate Effective if barrier treatment zone conditions are maintained. High flows of concentrated contaminants in groundwater and changing water levels may reduce effectiveness and require more frequent amendments. The ISRM at 100-D has experienced some breakthrough. Not effective in treating the bulk of the plume.	Moderate/High Can be implemented with injection wells or recirculation dipole wells. Broad zones of secondary byproduct generation within treatment area may occur.	Moderate/High Dependent on number and type of wells	Moderate	ISRM already exists, limited GHG and energy to augment	Retained	Existing ISRM will be allowed to function as it is, but amendments will not be considered as part of the FS evaluation since a barrier approach will not support the cleanup of the plume.
			Reactive Chemical Barrier (apatite, zero valent iron, zeolite, polyphosphate, etc.)	strontium-90	Subsurface injection or trenching in of reducing or sequestering chemicals along cross-gradient rows transecting plume. Chemicals are retained in the aquifer matrix so that contaminates are passively removed as groundwater moves through the treatment zone barriers.	Moderate Effective if barrier treatment zone conditions are maintained. High flows of highly aerobic groundwater and changing water levels are likely to necessitate more frequent amendments and/or reduce permeability of barrier (for zero valent iron). Not effective in treating the bulk of the plume.	Moderate Can be implemented with injection wells or trenching. However, both may be very challenging at this site due to the presence of cobbles/boulders.	High Dependent on number and type of wells	Moderate	GHG and energy for installation	Not Retained	Not retained in favor of existing hydraulic containment system.
			Reactive Biological Barrier	Cr(VI), nitrate	Subsurface delivery and recirculation of electron donors along cross-gradient rows transecting plume. Residual reducing byproducts and biomass are retained in the aquifer matrix so that contaminants are passively removed as groundwater moves through the treatment zone barriers.	Low/Moderate Effective if barrier treatment zone conditions are maintained. The aerobic groundwater conditions may require frequent amendment of the barrier. Not effective in treating the bulk of the plume.	Moderate/High Can be implemented with injection wells or recirculation dipole wells—latter option reduces number of wells required and is more cost effective. Broad zones of secondary byproduct generation within treatment area may occur—requires re-oxygenation of groundwater before discharge to the river.	Moderate/High Dependent on number and type of wells	Moderate	GHG and energy for installation	Not Retained	Not retained in favor of existing hydraulic containment system.

Table 8-7. Screening Table—Technologies for Hexavalent Chromium and Other COPCs in Groundwater—100-D/H

General Response Actions	Remedial Technology	Process Option	COPC Applicability[c]	Description	Relative Effectiveness	Relative Implementability	Relative Capital Cost	Relative O&M Cost	Sustainability[b]	Retained/Not Retained [a]	Screening Comment
Containment	Hydraulic Control	Hydraulic Containment via Extraction	All	Install extraction wells along downgradient edge of plumes to control migration of COPCs to the river.	Moderate Extraction should control plume migration to the river, but upgradient plumes and hot spots are left untreated.	High Compatible with existing infrastructure, and can be designed to work with other remedial technologies.	Low Facilities in place	Moderate	GHG and energy for operations	Retained	
		Hydraulic Containment via Injection	All	Injection of river water or groundwater parallel to the river. Manages hydraulic gradients to create conditions (for example, an inward gradient) throughout the year that mimic natural conditions of low plume discharge encountered during periods of high river stage. Barrier comprising closely spaced injection wells, infiltration trenches, and/or horizontal wells. Source of water from existing permitted Columbia River supply and/or groundwater.	Low/Moderate Should rapidly control plume migration to the river. However, some flushing and dilution of the contamination already close to the river may occur.	Moderate Can be accomplished using practically achievable injection rates. Injection only required 2 to 3 seasons (6 to 9 months). Infiltration trenches will be more cost effective than injection/horizontal wells but could cause seepage faces to develop along river cliff faces.	Moderate	Moderate	GHG and energy for operations	Not Retained	Not retained in favor of existing hydraulic containment system.

Note: COPCs include chromium (total), nitrate, and strontium-90.

a. Additional details on technologies not retained are provided in Appendix I.

b. Sustainability includes potential impacts to the environment that could arise from implementing this technology (for example, GHG emissions, waste generation, water use and resource impacts, energy use). Alternative design will dictate how sustainable an approach is.

c. Indicates the contaminants that can be addressed by a technology based on geochemical properties. A COPC Applicability of "All" indicates implementation of a technology is not dependent on the nature of a contaminant.

Table 8-8. Categories and Types of Current Hanford Institutional Controls

DOE Categories of Institutional Controls[a]	DOE Categorical Description	Types of Current Hanford Institutional Controls	Examples of Institutional Controls
Active/Passive Controls	These controls have long been understood to apply to the long-term management of radioactive waste. Active controls require clear institutional and human responsibilities and the active performance of responsibilities such as controlling access to a disposal site by means such as guards, performing maintenance operations or remedial actions at a site, controlling or cleaning up releases from a site, or monitoring parameters related to disposal system performance. Passive controls are defined by their dependence on the design of controls and structures such as permanent markers placed at a disposal site; public records and archives; government ownership and regulations regarding land or resource use; and other methods of preserving knowledge about the location design and contents of a disposal system.	**Warning Notices:** Provide visual identification and warning of hazardous or sensitive areas. A mechanism of warning notices includes signs that provide visual identification and warning of hazardous or sensitive areas. **Entry Restrictions:** Prevent or limit the access of humans to particular hazardous or sensitive areas. Procedural requirements for access warning signs (in conjunction with an engineering control such as fencing) can be implemented to provide entry restrictions.	**Warning Notices and Entry Restrictions** – Requirement for placement of permanent signs and/or markers at specific areas of the site. – Procedural requirements for access excavation/drilling permits. – Applies to all COPCs. – *Effectiveness*: Good. Reduces or eliminates the potential for direct contact with radiological contamination and contaminated groundwater for the duration of elevated risk period, and for preserving knowledge about a specific area or design. Protects integrity of active remedies. – *Implementability:* Very Good. Readily implemented, requires periodic surveillance and maintenance. – *Cost:* Low. *Retained*
Proprietary/Government Controls	This type of control is based on the legal authority of landowners to control the use of their land. Proprietary controls, such as easements, are based on the rights associated with ownership of an interest in land. Government controls rely on the powers of governments to protect the public health and safety through zoning, legislation, land ownership, or permit programs.	**Land Use Management:** Ensures that use of the land is compatible with any hazards that exist. As presented in *Sitewide Institutional Controls Plan for Hanford CERCLA Response Actions* (DOE/RL-2001-41), "DOE will restrict the use of land on waste sites and prohibit activities that would interfere with the remedial activity in accordance with the institutional controls requirements of the CERCLA decision documents and as described in applicable work plans." Implementation of land use management controls can ensure that any changes in use of the land are assessed before being allowed, and that institutional controls are maintained beyond change of ownership, as appropriate. Mechanisms include land use and real property controls (for example, proprietary controls including easements and covenants) and excavation permits. Land use and real property controls ensure that the use of land is in accordance with Hanford Site plans and CERCLA decision documents. Site evaluations are required before any land disturbance activity, and excavation permits are required for excavations on the Hanford Site to prevent unplanned disturbance or infiltration as prohibited by CERCLA decision documents. **Groundwater Use Management:** Ensures proper use of groundwater through groundwater controls. As described in *Sitewide Institutional Controls Plan for Hanford CERCLA Response Actions* (DOE/RL-2001-41), groundwater use on the Hanford Site is generally restricted, except for limited research purposes and for monitoring and treatment, as approved by EPA or Ecology, or as authorized in EPA- or Ecology-approved documents. Excavation permits and the land use process also control groundwater use.	**Land Use Management** – Land use and real property controls (for example, proprietary controls including easements and covenants). – Applies to all COPCs. – *Effectiveness:* Good. Reduces or eliminates the potential for direct contact with contaminated groundwater when well implemented and maintained for the duration of elevated risk period. Ensures compatible land use. – *Implementability:* Very Good. Readily implemented, must identify and comply with all necessary legal requirements. – *Cost:* Low. *Retained* **Groundwater Use Management** – Groundwater controls. – Applies to all COPCs. – *Effectiveness:* Good. Ensures no improper use of groundwater. – *Implementability:* Very Good. Readily implemented, but will likely require ongoing oversight and coordination with state water resource managers. – *Cost:* Low. *Retained*

Table 8-8. Categories and Types of Current Hanford Institutional Controls

DOE Categories of Institutional Controls[a]	DOE Categorical Description	Types of Current Hanford Institutional Controls	Examples of Institutional Controls
Informational Tools[b]	Provide information or notification about whether a remedy is operating as designed and/or that residual or contained contamination may remain onsite. Information devices include state registries, deed notices, and advisories.	**Waste Site Information Management:** This is an administrative mechanism implemented to maintain and provide access to information on the location and nature of contamination. The WIDS database identifies waste management units on the Hanford Site, their location, waste type, and status. Other descriptive information contained in WIDS includes size, extent, and appearance; testing or sampling efforts; regulatory information; bibliographic references; images; change history; and data validation. RL maintains the system in accordance with the WIDS change control system, which documents and traces additions, deletions, and/or other changes dealing with the status of waste management units.	**Waste Site Information Management** – Administrative – Applies to all COPCs. – *Effectiveness:* Good. Ensures access to information on the location and nature of contamination. – *Implementability:* Very Good. Readily implemented, but requires maintenance of the information management system. – *Cost:* Low. *Retained*

a. *Sitewide Institutional Controls Plan for Hanford CERCLA Response Actions* (DOE/RL-2001-41)

b. An "Informational Tool" is an EPA category of an institutional control that is used at the Hanford Site as discussed in *Sitewide Institutional Controls Plan for Hanford CERCLA Response Actions* (DOE/RL-2001-41).

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Ex situ treatment process options include the following:

- Ex situ solidification/stabilization
- Soil washing
- Ex situ vitrification
- Ex situ thermal desorption

Disposal. Following excavation, contaminated soil needs to be properly disposed, either at the onsite or an offsite landfill, or by backfilling treated soil. Prior to implementation of a disposal option, waste acceptance criteria must be evaluated. Treatment required to meet ERDF waste acceptance criteria is evaluated under the disposal GRA.

Backfilling treated soil involves excavation and ex situ treatment, followed by onsite disposal. Before implementation of this disposal option, treated soil will need to be compared to PRG criteria to verify backfilling is appropriate.

Disposal at the onsite landfill includes transport of excavated soil to EDRF. The waste acceptance criteria for ERDF are based on regulatory requirements (for example, RCRA Land Disposal Requirements) and risk-based considerations for long-term protection of human health and the environment. If waste cannot be accepted at ERDF, an EPA-approved offsite disposal facility will be used. Part of this process option is treatment required to meet ERDF waste acceptance criteria. Therefore, an ex situ treatment process option does not need to be evaluated if excavation and disposal at ERDF are selected as remedial options.

In Situ Treatment. In situ treatment consists of actions that treat contamination in place using physical, chemical, or biological treatment techniques. The main advantage of in situ treatment is that it allows soil to be treated without being excavated and transported, resulting in significantly reduced exposure to site workers relative to removal of contaminated media for disposal or ex situ treatment. Other advantages include reduced disturbances to vegetation and cultural resources relative to excavation. In situ treatment may also provide a larger areal zone treatment, and there is typically little secondary waste generated. For this evaluation, in situ process options were subdivided by technologies that require delivery of a reagent to the subsurface for treatment, and those that implement another technique. Within actions requiring delivery of a reagent, technologies can be further subdivided by the reagent approach (physical, chemical, or biological), and the method for delivering the reagent to the subsurface. For treatment of contaminated soil in 100-D/H, the following in situ remedial technologies and process options were evaluated:

- Reagent approach:
  - In situ solidification
  - In situ stabilization/sequestration
  - Chemical reduction
  - In situ chemical oxidation
  - Biological reduction
  - Combined chemical/biological reduction
  - Gaseous ammonia injection
  - Reductive dechlorination using zero-valent metals or bioremediation of PCBs
  - In situ gaseous reduction with chemical reductant or biological substrate

- Delivery method:
  - Mixing with conventional excavation equipment
  - Deep soil mixing (vertical/horizontal)
  - Foam delivery of reagents
  - Gas delivery of reagents
  - Horizontal injection wells
  - Vertical injection wells
  - Jet grouting
  - Surface infiltration
  - Void filling/grouting
- In situ treatment—other:
  - Soil blending
  - Desiccation
  - In situ thermal desorption
  - In situ vitrification
  - Soil flushing (vadose zone, water)
  - Phytoremediation

Containment. Containment actions consist of physical measures to restrict contaminant migration to groundwater, and/or break the direct contact exposure pathway. Remedial technologies evaluated under the containment GRA include surface barriers, horizontal subsurface barriers, and compaction.

Surface barrier technologies are constructed over contaminated waste sites to control the vertical entry of water into contaminated media, which in turn reduces leaching of contamination to groundwater. Surface barriers also provide a cover of contaminated waste sites to protect against direct contact exposure to minimize human and ecological risks. In addition to their hydrological performance, barriers can function as physical obstructions to prevent intrusion by human and ecological receptors, limit wind and water erosion, and attenuate radioactivity. Surface barriers include Hanford barrier, modified RCRA Subtitle C or Subtitle D barrier, asphalt/concrete cap, and vegetative cap (evapotranspiration cap). The Hanford barrier design was developed specifically for use at the Hanford Site for sites containing low-level waste greater than Class C and/or significant inventories of transuranic constituents.

Emplaced horizontal subsurface barriers are set beneath existing in situ contaminants. These bottom barriers have features similar to those of vertical barriers in that they minimize movement of contaminants, restrict infiltration of groundwater, and are constructed of similar materials with similar technologies. Horizontal barrier technologies can include jet grouting, soil freezing, and wire saw barriers.

Dynamic compaction can consolidate soil and buried wastes, and minimize the potential subsidence for a subsequent barrier. The process involves dropping a weight from a predetermined height onto the area to be compacted.

8.3.1.2 Identification and Screening of Technologies for Groundwater

No Action. The no action response means any further action to remove, remediate, monitor, or restrict access to contaminated groundwater is discontinued. CERCLA RI/FS Guidance (EPA/540/G-89/004) and NCP (40 CFR 300) guidance require this response to remain in the FS process for comparative purposes, where it is used as a baseline against which all other alternatives will be compared.

Institutional Controls. Institutional controls are administrative controls and legal restrictions imposed on land use to prevent or reduce exposure to hazardous wastes or hazardous constituents and/or to protect the integrity of a remedy. Section 8.3.1.1 and Table 8-8 describe institutional controls for the Hanford Site.

For groundwater, institutional controls include administrative controls, access, and drilling restrictions until achievement of RAOs. Groundwater use management controls are in place to ensure proper use of groundwater. Groundwater use on the Hanford Site is generally restricted, except for limited research purposes and for monitoring and treatment, as approved by EPA or Ecology, or as authorized in EPA- or Ecology-approved documents. Table 8-8 presents an evaluation of groundwater use management restrictions.

Monitored Natural Attenuation. MNA relies on natural attenuation processes such as biological and chemical reduction, adsorption, dilution, dispersion, and radioactive decay to manage the contamination onsite. MNA includes an evaluation of the natural attenuation mechanisms and implements source control and long-term monitoring to track progress toward complying with RAOs. When relying on natural attenuation processes for site remediation, EPA prefers processes that degrade or destroy contaminants (*Monitored Natural Attenuation of Inorganic Contaminants in Ground Water Volume 1 – Technical Basis for Assessment* [EPA/600/R-07/139]). Therefore, MNA can be an important component of the overall remedy, especially for waste sites with short-lived radionuclides.

As presented in *Use of Monitored Natural Attenuation at Superfund, RCRA Corrective Action, and Underground Storage Tank Sites* (OSWER Directive 9200.4-17P), MNA is an appropriate remedial response only where its use will be protective of human health and the environment, and when it will be capable of achieving site-specific RAOs within a timeframe that is reasonable compared with other alternatives. Largely because of the uncertainty associated with the potential effectiveness of MNA to meet remediation objectives that are protective of human health and the environment, EPA expects that source control and long-term performance monitoring will be fundamental components of any MNA remedy.

Evaluation of MNA as an appropriate response action for contaminated groundwater will be completed in accordance with the guidelines provided in *Use of Monitored Natural Attenuation at Superfund, RCRA Corrective Action, and Underground Storage Tank Sites* (OSWER Directive 9200.4-17P), in addition to *Decision-Making Framework Guide for the Evaluation and Selection of Monitored Natural Attenuation Remedies at U.S. Department of Energy Sites* (DOE, 1999a), *Technical Guidance for the Long-Term Monitoring of Natural Attenuation Remedies at U.S. Department of Energy Sites* (DOE, 1999b), and *Monitored Natural Attenuation of Inorganic Contaminants in Ground Water Volume 3: Assessment for Radionuclides Including Tritium, Radon, Strontium, Technetium, Uranium, Iodine, Radium, Thorium, Cesium, and Plutonium-Americium* (EPA/600/R-10/093).

MNA may be selected as appropriate technology for remediation of contaminated groundwater under certain circumstances. MNA may be considered as an individual remedial alternative, or it may be combined with other technologies to develop a compound alternative (Figure 8-14, provided later is this chapter, illustrates MNA of groundwater). Determining how MNA fits with other remediation technologies requires evaluation of the specific role that MNA will play in the alternative. Evaluation of an MNA technology application follows a logical sequence of assessment of the following four essential functional requirements of MNA:

- **The contamination condition does not currently present an actual risk to human or ecological receptors**. There must be an expectation that exposure mitigation can and will be maintained throughout the MNA period. Site monitoring must be adequate to confirm exposure mitigation.

- **The source of the observed contamination is no longer contributing to the plume**. The source may have been controlled previously through an engineered remedy or naturally ceased to contribute to the problem. In some cases, a source control element (for example, localized pump-and-treat or selected in situ remedy) may be combined with the MNA alternative to ensure adequate control of secondary sources (for example, residual mobile contamination in the vadose zone, or high-concentration plume segments in groundwater).

- **The target plume is static or retreating, or existing monitoring data otherwise confirm that attenuating processes are present and operating at the site. Effective monitoring either exists currently, or can be implemented**, that will provide confirmation that the attenuation is proceeding as expected and that remedial goals are achieved.

Development and evaluation of MNA either as a technology or as a standalone alternative for groundwater contaminant plumes requires thorough understanding and description of current site conditions, knowledge of contaminant characteristics, in addition to representative historical monitoring results to form the basis for evaluation of MNA as an appropriate alternative. The following conditions will be considered in evaluating MNA for COCs at 100 D/H:

- **Reduction of nitrate**, primarily facilitated by microbial populations in surface and groundwater, may occur when the dissolved oxygen content water becomes low and the water enters a reduced condition. Indigenous facultative and obligate anaerobic microbes may then use the oxygen atoms of the nitrate molecule in their metabolic processes, reducing the nitrate to other forms (for example, nitrite, diatomic nitrogen). While these processes may occur in some locations at 100-D/H, the aquifer is generally well-aerated and, as a result, nitrate tends to be quite stable and mobile in groundwater. Reduction of nitrate in an un-modified aquifer system is not considered an attenuating process at 100-D/H.

- **Reduction of Cr(VI)** may also occur in reducing conditions within the aquifer, or through chemical reaction with reducing compounds. Chemical reduction of Cr(VI) produces trivalent chromium, which is subject to subsequent precipitation of chromium oxide and hydroxide compounds that exhibit extremely low water solubility. In groundwater at 100-D/H, Cr(VI) reduction generally occurs only at locations where the aquifer has been modified to produce reducing conditions, either through application of some remedial process (examples include, the in situ biostimulation treatability test in the southwestern portion of the 100-D Area within the chromate and nitrate plumes, ISRM barrier installed in 100-D. and the calcium polysulfide treatability test at 100-K) or through some pollution-related process (for example, anaerobic conditions related to septic tank/leach field discharges, or historical releases of reducing constituents).

- **Diffusion and dispersion** within the aquifer are physical processes that reduce contaminant concentrations in groundwater over time and distance. Diffusion is a concentration-driven physical process that results in movement of dissolved constituents from areas of high concentration to adjacent areas of relatively low concentrations. Dispersion is a physical process that results in mixing of dissolved constituents within the aquifer water as the result of variations in groundwater flow velocity along varying flow paths within the aquifer. This mixing results in reduction in contaminant concentrations over distance. The 100-D/H groundwater plumes cover a relatively large area; however, the distance along flow paths is relatively short between inland areas of elevated contaminant concentration and locations of potential exposure to receptors at the groundwater/river interface. This indicates that the overall portion of the aquifer where diffusion and dispersion may provide substantial concentration reduction is relatively small. Diffusion and dispersion are, therefore,

not considered to be major contributors to attenuating processes at 100-D/H at higher concentrations of contamination, but may still be considered at concentrations near the cleanup levels.

Pump-and-Treat. The pump-and-treat GRA includes collection, ex situ treatment, and discharge. The following text details the remedial technologies and applicable process options.

- **Collection**—This process option involves collection of contaminated groundwater through operation of groundwater extraction wells. Groundwater is pumped to the surface through vertical wells and then transferred through pipes to a treatment facility. Two pump-and-treat systems currently operate to remediate groundwater in the 100-HR-3 OU. The DX pump-and-treat system and HX pump-and-treat systems remediate groundwater in the 100-D and 100-H areas, respectively. This process option includes expansion and/or modifications to the existing pump-and-treat systems.
- **Ex Situ Treatment**—Aboveground treatment may involve physical, biological, or chemical processes. Ex situ treatment process options include the following:
  - Ion exchange
  - Chemical reduction and precipitation
  - Electrocoagulation
  - Wetlands
  - Subgrade bioreactors
  - Bioreactors
  - Phytoremediation
  - Membrane separation (reverse osmosis)
- **Discharge**—Both onsite and offsite:
  - Onsite discharge includes groundwater injection wells and surface infiltration of treated water
  - Offsite discharge includes surface water discharge

In Situ Treatment. In situ treatment consists of actions that treat contamination in place. In situ treatment of contaminated groundwater generally includes methods to degrade contaminants, such as adding agents to groundwater (via injection wells or permeable barriers) that facilitate chemical or biological destruction or immobilization. For this evaluation, technologies are subdivided by the reagent approach (physical, chemical, or biological), and the method for delivering the reagent to the subsurface. For treatment of contaminated groundwater in 100-D/H, the following in situ remedial technologies and process options were evaluated:

- Reagent approach:
  - In situ chemical reduction
  - In situ chemical stabilization
  - In situ biological treatment (anaerobic)
  - Hydrogen or other organic gas sparging
  - In situ treatment using combination of biological and chemical substrates
  - Flushing (saturated zone, water)
- Delivery method:
  - Surface infiltration
  - Groundwater circulation wells

- Vertical wells
- Horizontal wells

Containment. Containment technologies assist in preventing or significantly reducing the migration of contaminants in groundwater through physical barriers or treatment barriers. For treatment of contaminated groundwater in 100-D/H, the following containment process options were evaluated:

- Containment wall (for example, slurry wall or grout wall)
- ISRM
- Reactive chemical barrier
- Reactive biological barrier
- Hydraulic containment via extraction
- Hydraulic containment via injection

8.3.2 Evaluation of Technologies and Selection of Representative Technologies

Tables 8-6 and 8-7 present the identification and screening of technologies and remedial process options for 100-D/H. Table 8-6 presents GRAs and process options for vadose zone soils, including waste sites, impacted with radionuclides, Cr(VI), and other metals, and organic compounds. Table 8-7 presents GRAs and process options for groundwater impacted with Cr(VI) and other COPCs.

The various technologies screened in the tables include demonstrated and proven processes, innovative technologies, and potential processes that have undergone laboratory trials or bench scale testing. Factors considered in this evaluation include the state of technology development, site conditions, waste characteristics, nature and extent of contamination, and presence of constituents that could limit the effectiveness of the technology. A qualitative comparison of implementability, effectiveness, and cost provided additional evaluation of technologies. The screening tables also present information pertaining to the sustainability of a process option. It is important to note, however, that sustainability was not considered as a criterion for the screening of process options.

Implementability refers to the relative degree of difficulty anticipated in implementing a particular process option under regulatory, technical, and schedule constraints posed by the site. As suggested by CERCLA RI/FS Guidance (EPA/540/G-89/004), process options and entire technology types can be eliminated from further consideration if a technology or process option cannot be effectively implemented at the site. As discussed in Section 4.2.5 of the CERCLA RI/FS Guidance (EPA/540/G-89/004), "technical implementability is used as an initial screen of technology types and process options to eliminate those that are clearly ineffective or unworkable at a site." Institutional or administrative implementability, which includes "the ability to obtain necessary permits for offsite actions, the availability of treatment, storage, and disposal services (including capacity), and the availability of necessary equipment and skilled workers to implement the technology," is also considered in the initial screening.

Effectiveness refers to the ability of the process option to perform as part of a comprehensive remediation plan to meet RAOs under the conditions and limitations present at site. Additionally, the NCP (40 CFR 300) defines effectiveness as the "degree to which an alternative reduces toxicity, mobility, or volume through treatment; minimizes residual risk; affords long-term protection; complies with ARARs; minimizes short-term impacts; and how quickly it achieves protection." This is a relative measure for comparison of process options that perform the same or similar functions. Section 4.2.5 of CERCLA RI/FS Guidance (EPA/540/G-89/004) states that the evaluation of process options with respect to effectiveness should focus on, "(1) the potential effectiveness of process options in handling the estimated areas or volumes of media and meeting the remediation goals identified in the RAOs; (2) the potential impacts to human health and the

environment during the construction and implementation phase; and (3) how proven and reliable the process is with respect to the contaminants and conditions at the site."

For the initial screening of technology types and process options, the cost criterion is relative. It compares processes and technologies that perform similar functions and have similar effectiveness. Section 4.2.5 of CERCLA RI/FS Guidance (EPA/540/G-89/004) states that, "cost plays a limited role in the screening of process options. Relative capital and O&M costs are used rather than detailed estimates. At this stage in the process, the cost analysis is made on the basis of engineering judgment, and each process is evaluated as to whether costs are high, low, or medium relative to other process options in the same technology type." For this evaluation, cost is used to screen out process options that have a high relative cost if there are other choices that perform similar functions with similar effectiveness. The cost criterion includes a cursory consideration of the rough order of magnitude costs of construction and any long-term costs to operate and maintain the technologies.

Technologies that are not technically feasible based on implementability, effectiveness, and cost were screened out. Technical implementability is the first screening criteria evaluated as part of this process, per EPA guidance. However, for technologies with significant technical implementability challenges, an evaluation of effectiveness and cost was still completed to allow for a more complete evaluation. Technologies that were considered technically impracticable based on unsuccessful case studies at the site, challenges associated with existing site conditions (lithology), a potential increased risk to worker safety, or of increased complexity as compared to other technologies of comparable effectiveness were screened out. Technologies were also removed from further consideration if they were considered to have limited treatment effectiveness for the specified COPC or performance uncertainties. Appendix I provides a thorough discussion of the technologies not retained, including a detailed screening rationale. Remedial technology types and process options considered viable for remediating contaminated soil at 100-D/H are carried forward into the development (Chapter 9) and detailed analysis of alternatives (Chapter 10).

8.3.2.1 Identification of Technologies and Selection of Representative Technologies for Vadose Zone Contamination

For remediation of vadose zone soils, including waste sites, at 100-D/H, the following response actions were retained (Table 8-6) to remove contaminants or interrupt the exposure pathway to receptors:

- Removal:
  - Standard excavation—Provides for removal of contaminants
  - Deep excavation—Provides for removal of contamination
- Disposal:
  - Disposal to ERDF—Provides for treatment at the facility (if necessary) and disposal of contaminants
  - Offsite disposal at an EPA approved landfill—Provides for disposal of contaminants
- In situ treatment via reagent:
  - Solidification—Provides for treatment of contaminants
  - Stabilization/sequestration—Provides for treatment of contaminants
  - Biological reduction—Provides for treatment of contaminants

- In situ treatment via reagent—delivery method:
  - Vertical injection wells—Used to deliver liquid reagent to the shallow or deep vadose zone
  - Jet grouting—Used to deliver liquid reagent to the shallow or deep vadose zone
  - Surface infiltration—Used to deliver liquid reagents
  - Deep soil mixing—Used to deliver reagent to the shallow or deep vadose zone
  - Void-fill grouting—Provides engineered barrier to interrupt the exposure pathway and immobilize contaminants
- In situ treatment—other:
  - Soil flushing (vadose zone, water)—Provides transfer of contaminants to groundwater during remediation; this in situ technology is coupled with groundwater hydraulic containment
- Containment:
  - Surface barrier—Provides engineered structure to interrupt the exposure pathway

Institutional controls as identified in Table 8-8 are also retained for controls during remediation to interrupt the exposure pathway. The "no action" GRA does not provide capability to remove contaminants or interrupt the exposure pathway to receptors but is also retained per the NCP (40 CFR 300).

8.3.2.2 Identification of Technologies and Selection of Representative Technologies for Groundwater Contamination

For treatment of contaminated groundwater (Table 8-7) at 100-D/H, the following response actions and were retained to remove contaminants or interrupt the exposure pathways to receptors:

- MNA—Treatment of contaminants through biological and chemical reduction, radioactive decay, adsorption, dilution, and dispersion
- Pump-and-treat—Provides for treatment of contaminants:
  - Collection through groundwater extraction system
  - Ex situ ion exchange (for Cr[VI], strontium-90, and nitrate)
  - Groundwater injection wells discharge
  - Discharge through surface infiltration
- In situ treatment—Provides for treatment of contaminants:
  - Chemical stabilization—Through subsurface delivery of chemical reagents
  - Biological treatment (anaerobic) —Using liquid substrate
  - Combined chemical and biological reagents—Through subsurface delivery of chemical reductants and electron donors
  - Physical treatment—Flushing the saturated zone with water to facilitate contaminant movement to allow for capture and treatment from the groundwater media (that is, pump-and-treat with hydraulic containment)

- In situ treatment—delivery methods:
  - Surface infiltration—Release of water or reagents at the surface or near surface
  - Vertical wells—Used to inject water or reagents to enhance contaminant flushing or promote biological treatment
  - Reactive chemical barrier—Existing ISRM
- Containment:
  - Hydraulic containment via extraction and injection - provides engineered system to interrupt the exposure pathways

Institutional controls as identified in Table 8-8 are also retained for controls during remediation to interrupt the exposure pathway. The "no action" GRA does not provide capability to remove contaminants or interrupt the exposure pathways to receptors but is also retained per the NCP (40 CFR 300).

Figures 8-7 through 8-23 present specific information on technologies that have been retained. For 100-D/H, Chapter 9 presents technologies that are applicable for each waste site type group and will be developed into alternatives. Appendix I provides a thorough discussion of the technologies not retained. This appendix describes the technology, followed by relevant case studies and the screening rationale.

Standard Excavation

Key Components

- Completed using standard earthmoving equipment.
- Conventional open pit (standard) excavation limited to approximately 20 feet below ground surface based on equipment constraints.
- Extent of excavation required will be determined using an observational approach which combines characterization and remediation steps to maximize use of resources.
- The observational approach includes design of remediation based on available data. Specific site characterization will be performed during the removal of the waste. Remedial actions are guided by the observational approach where various methods, including in-situ and ex-situ sampling, process knowledge, and field measurements, guide day-to-day excavation.
- Clean overburden soil is removed and stockpiled.
- Contaminated soil is removed and segregated to determine disposal or treatment requirement, or direct-loaded into containers for disposal.
- Verification sampling can be performed to demonstrate cleanup levels are achieved.
- Excavations are backfilled and compacted using clean overburden and borrow soil.

Shallow excavation at 100-B-19.

Examples of Relevant Experience

- Removal, treatment, and disposal (RTD) has been selected as a remedial alternative in previous 100 Area decision documents. Full-scale remediation in the 100 Areas using RTD began in July 1996. Over one million tons of contaminated soil and debris have been disposed of. *(EPA/ROD/R10-99/039)*
- Excavations completed at Trenches 216-B-26 and 216-B-53A and at 216-B-14 Crib for Sr-90 and Cs-137 bearing soils. *(HNF-36881)*
- Uranium-contaminated sediments at Process Trench 316-5 were also excavated. *(WHC-SA-2062-FP)*

Risk Reduction Mechanism

- Contaminated materials removed, eliminating source of exposure.
- Mitigates further migration of contaminants to groundwater.

Screening Criteria *(dashed line indicates range)*

	Low	Moderate	High
Relative Effectiveness			
Relative Implementability			
Relative Capital Cost			
Relative O&M Cost	*No associated costs.*		

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Figure 8-7. Standard Excavation

Deep Excavation

Key Components

- Excavation using standard equipment, requiring implementation of complex mechanisms such as shoring or lay backs to provide stability.
- Excavation complexity increases with greater depth.
- Extent of excavation required will be determined using an observation approach which combines characterization and remediation steps to maximize use of resources.
- The observational approach includes design of remediation based on available data. Additional site characterization can be performed during the removal of the waste. Remedial actions are guided by the observational approach where various methods, including in-situ and ex-situ sampling, process knowledge, and field measurements, guide day-to-day excavation.
- Requires careful evaluation of the side walls and anchoring systems selected to support the excavation, including stability calculations.
- Clean overburden soil is removed and stockpiled.
- Excavated soil is segregated to determine disposal or treatment requirements.
- A combination of in-process and verification sampling can be used to determine extent of excavation required and demonstrate cleanup levels are achieved.
- Excavations are backfilled and compacted using clean overburden and borrow soil.

Deep excavation at 100-B-27.

Examples of Relevant Experience

- Removal, treatment, and disposal (RTD) has been selected as a remedial alternative in previous 100 Area decision documents. Full-scale remediation in the 100 Areas using RTD began in July 1996. Over one million tons of contaminated soil and debris have been disposed of. *(EPA/ROD/R10-99/039)*
  - Excavation of contaminated soil was completed to groundwater (approximately 46 feet below ground surface) at waste site 100-B-27 at the Hanford 100-B/C Area with 10,190 cubic meters of contaminated soils removed *(RSVP-2009-040)*
  - Remediation of the 100-C-7 waste site at the Hanford 100-B/C Area has included two excavations to approximately 85 feet below ground surface with a combined total of 1.1 million cubic meters of soil and debris excavated. Excavation to remove contaminated soil is still ongoing.

Risk Reduction Mechanism

- Contaminant sources in deep vadose zone soils are physically removed.
- Mitigates further migration of contaminants to groundwater.

Screening Criteria *(dashed line indicates range)*

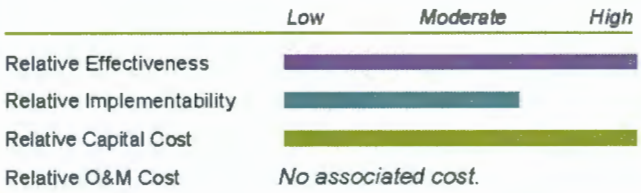

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Figure 8-8. Deep Excavation

On-Site Disposal: Environmental Restoration Disposal Facility (ERDF)

Key Components

- Contaminated soil and waste material transported from waste site to on-site disposal facility at Hanford—Environmental Restoration Disposal Facility (ERDF).
- Treatment (*e.g.*, macro-encapsulation) performed at the facility as required to meet land disposal restrictions (LDR).
- Engineered to meet appropriate performance standards under 10 CFR 61, "Licensing Requirements for Land Disposal of Radioactive Waste," and meet minimum technical requirements for landfills under WAC 173-303-665, "Landfills."
- Facility can accept the majority of remediation waste streams. Liquid wastes that cannot be solidified and certain LDR wastes that cannot be accepted would need to be sent off-site for disposal.
- ERDF consists of a series of disposal areas (cells). Each pair of cells is 70 feet deep, 500 feet by 1,000 feet at the base, and over 1,400 feet wide at the top.
- Cell pairs have a disposal capacity of 3 million tons. As of June 2010, over 11 million tons of contaminated material have been deposited into ERDF. *(www.handford.gov/page.cfm/ERDF)*

Hanford's Environmental Restoration Disposal Facility

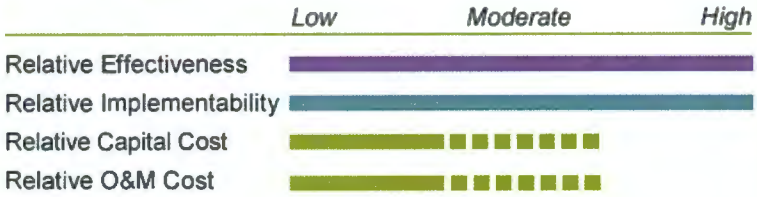
Examples of Relevant Experience

- Hanford's ERDF, in the 200 West Area, is a landfill regulated by USEPA and capable of receiving about 16,000,000 tons of waste.
- Accepts low-level radioactive, hazardous, and mixed wastes that are generated during the cleanup activities at the Hanford Site.
- First started operations in 1996. Over 11,000,000 tons of contaminated soil and debris have been disposed at the facility. *(RLI-D02-14)*

Risk Reduction Mechanism

- Waste material is placed in an engineered landfill with physical and regulatory controls to greatly restrict or eliminate environmental mobility.
- Waste material is consolidated at a single location. Risk reduction primarily achieved through excavation.

Screening Criteria *(dashed line indicates range)*



CHPUBS_RC_0016

Figure 8-9. Onsite Disposal: The Environmental Restoration Disposal Facility

In-Situ Biological Reduction (Vadose Zone)

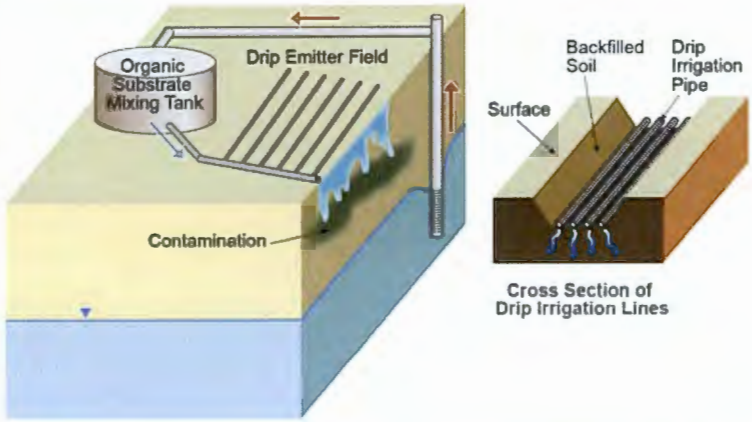
Key Components

- Uses native microorganisms to reduce contaminants to less- or non-toxic compounds, either directly by the microbes through dissimilatory or enzymatic reduction, or indirectly by a reduced electron acceptor (*e.g.*, ferrous iron or sulfide).
- Natural process are enhanced by adding organic substrates (a carbon source) to stimulate microorganisms in the subsurface and change the geochemistry to anaerobic conditions.
- Localized temporary generation of secondary byproducts (reduced manganese, iron, and arsenic) should be expected.
- Organic substrate application methods include surface infiltration (shown in conceptual schematic), aqueous injection using wells, gas injection using wells, and soil mixing using solid reagents.
- Components for surface infiltration include:
  - Reagent tank
  - Subsurface drip irrigation system
  - Infiltration basin

Risk Reduction Mechanism

- Biological treatment can reduce Cr(VI) to the less-toxic and less mobile Cr(III), and nitrate to nitrogen gas.
- Volume of Cr(III) will not change, but toxicity will be reduced.
- Can reduce contaminant volume by removing contaminants as they are completely degraded, and/or mobility and toxicity reduced by transforming contaminants to less-toxic and/or less soluble forms.

Conceptual Schematic



Examples of Relevant Experience

Reports for examples of applications using bio-remediation to convert Cr(VI) to less toxic (Cr(III)) include:

- Hinkley Remediation Semiannual Status Report (July through December 2009), Pacific Gas and Electric Company (PG&E) Groundwater Remediation Program, Hinkley, California (CH2M HILL, 2010)
- PNNL-18784, Hanford 100-D Area Biostimulation Treatability Test Results

Pilot studies have demonstrated the processes can be used for applications in the vadose zone:

- A trailer-mounted 10-gpm In Situ Delivery (ISD™) system was used at a former agricultural facility (chrome plating) in Walla Walla, Washington to treat Cr+6-contaminated soil and groundwater that was a result of a leaking UST. (ETEC, Case Study, AGGRESSIVE HEXAVALENT CHROMIUM REMEDIATION USING A 10-GPM IN SITU DELIVERY (ISD™) SYSTEM)
- Laboratory studies have demonstrated the processes, showing that adding water and organic nutrients to columns packed with vadose zone materials contaminated with Cr(VI) cause the effective conversion of Cr(VI) to Cr(III) (Tokunaga et al., 2003, "In-situ reduction of Cr(VI) in heavily contaminated soils through organic carbon amendment," and Oliver, 2001, Microbial Reduction of Hexavalent Chromium Under Vadose Zone Conditions)

Screening Criteria *(dashed line indicates range)*

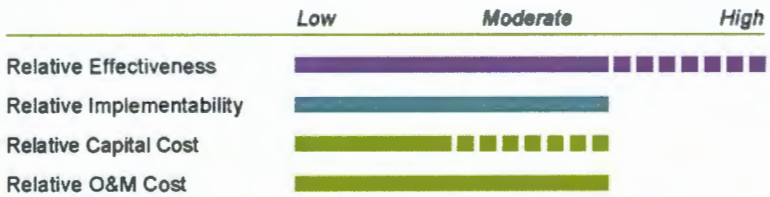


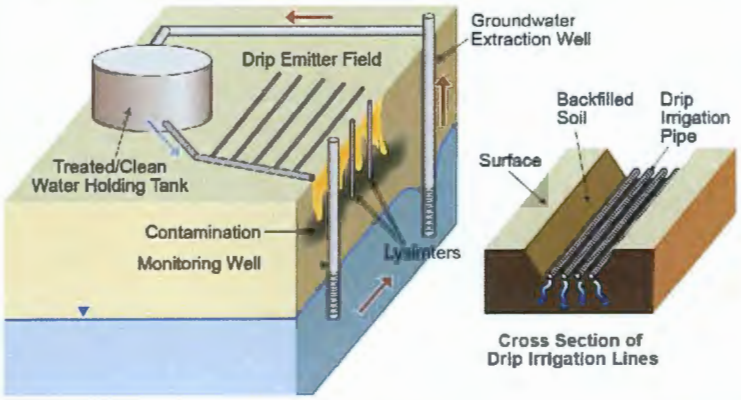
Figure 8-10. In Situ Biological Reduction (Vadose Zone)

Soil Flushing – Vadose Zone, Water

Key Components

- Involves the infiltration of clean or treated water into a zone of contaminated soil to flush contaminants out of the vadose zone to the water table.
- Applicable for media impacted with contaminants with high to moderate solubility (*e.g.*, Cr(VI), Tc-99, uranium, nitrate, and possible C-14).
- Solubility-enhancing solutions may be added to enhance mobility.
- Infiltrating water with desorbed contaminants need to be captured and treated to meet discharge standards.
- Contaminants flushed to the water table are captured by extraction wells coupled to pump-and-treat system(s) in place for groundwater remediation.
- Enhancement of conventional pump and treat with in-situ flushing of source area may speed site remediation and closure; however ineffective groundwater capture may create a larger groundwater plume.

Conceptual Schematic



Examples of Relevant Experience

- Soil flushing was used to treat soil and groundwater contaminated with Cr(VI) by United Chrome Products, Corvallis, OR. Delivery of solution was completed through two infiltration basins and one infiltration trench to flush Cr(VI) from the vadose zone to the water table. Extraction wells were used to recover the solution and extract groundwater. A 1998 report indicates 9.7 million gallons of impacted groundwater containing 26,732 pounds of Cr(VI) were removed in a 3-year period. *(TS-98-01)*

Risk Reduction Mechanism

- Toxicity and/or volume of source area in the vadose zone is reduced by mass transfer to groundwater.
- Reduction in toxicity, mobility, and volume of contaminants is achieved through a groundwater capture and treatment system.
- Treatment of impacted media in shallow soil may reduce direct contact risk and exposure to ecological receptors.

Screening Criteria *(dashed line indicates range)*

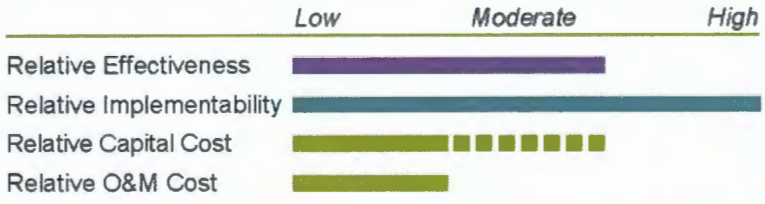


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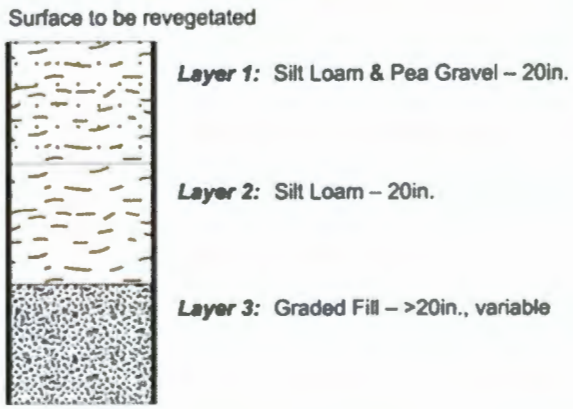
Figure 8-11. Soil Flushing—Vadose Zone, Water

Surface Barriers

Key Components

- Surface barriers are earthen and/or manufactured covers placed on the ground surface above contaminated media.
- Designed to prevent surface water infiltration through the vadose zone, and to limit contaminant leaching to groundwater. May also prevent direct contact to contaminants.
- Types of surface barriers include: Modified RCRA Subtitle C and/ or D, Asphalt/ Concrete, Evapotranspiration, Vegetative and the Hanford Barrier.
- Evapotranspiration (ET) barriers (shown in conceptual schematic) can be placed over structures to remain in place to promote drainage, prevent infiltration to possible sources below barrier, and prevent exposure to contaminated soil/debris.
- Excavation, handling, and transport of contaminated soil are reduced.
- Can also be implemented at the bottom of an excavation to limit infiltration through contaminated soil left in place. Implementation may require soil characterization and soil compaction tests.
- Periodic inspection and repair required.

Conceptual Schematic



Examples of Relevant Experience

- Examples of sites that have proposed, approved, or installed ET covers and the regulatory program they are operating under are given in EPA *Fact Sheet on Evapotranspiration Cover Systems for Waste Containment*, Appendix A. Details on these sites can be found in the alternative cover profiles database. *(http://cluin.org/products/altcovers)*

Risk Reduction Mechanism

- Prevents surface water infiltration and reduces contaminant migration through vadose zone, limiting potential leaching to groundwater.
- When coupled with Institutional Controls, may reduce direct contact and exposure to ecological receptors.
- Toxicity of contaminants is not reduced.

Screening Criteria *(dashed line indicates range)*

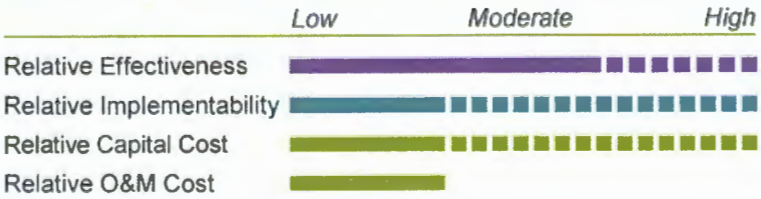


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Figure 8-12. Surface Barrier

Void Fill Grouting

Key Components

- In-situ treatment technology used to immobilize contaminants by solidification of wastes.
- Void grouting may be considered to fill large empty spaces (*e.g.*, pipelines, trenches, pits). Structure is then left in place.
- Grout can be cement-based (*e.g.*, Type I, II, III, IV, and V Portland cement) or chemical-based (*e.g.*, silicates, acrylics, lignosulfonates, phenoplasts, and aminoplasts).
- Portland cement-based grouts may offer an additional benefit to treat certain radionuclides and metals, since the increased pH from grouting may yield increasing precipitation and sorption of these compounds (*e.g.*, Sr-90).
- Grout can be mixed in batches or with a mobile continuous mixer, depending on the size of the grouting project.

Conceptual Schematic

Grouting of the 221-U Canyon Facility

Examples of Relevant Experience

- The *Interim Completion Report for the 221-U Facility* (DOE/RL-2011-80) provides status for void spaces of the 221-U Plant Canyon Facility filled with grout consistent with the remedial action identified in the 221-U Facility ROD. Void spaces included the process cell, hot pipe trench, piping and electrical galleries, drain header, process sewer, and ventilation and tunnel and ducts.

Risk Reduction Mechanism

- Immobilizes residual mobile contaminants that may be present within the structure.
- Reduces the potential of contaminant migration to groundwater.
- Immobilized contaminants left in place; however, volume of contaminated materials increases.

Screening Criteria *(dashed line indicates range)*

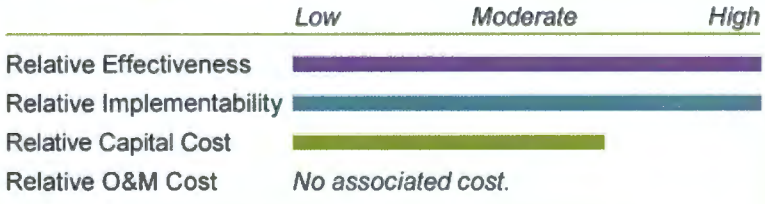

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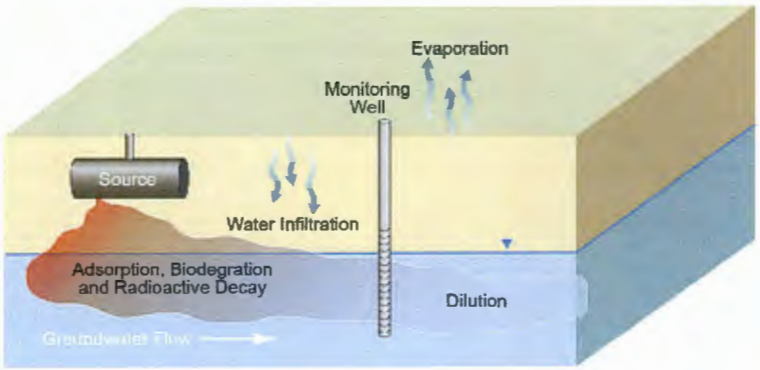
Figure 8-13. Void-fill Grouting

Monitored Natural Attenuation (MNA)

Key Components

- Relies on unaugmented natural, intrinsic processes (dilution, volatilization, flushing from surface water infiltration, sorption, microbial degradation, radioactive decay, and chemical reactions) to reduce contaminant concentrations and migration.
- Transport modeling and evaluation of intrinsic processes may be required to evaluate potential groundwater migration and time required to achieve cleanup criteria.
- Incorporates long-term monitoring to track progress towards compliance with cleanup objectives. Typically combined with other technologies that manage source areas and mitigate exposure.

Conceptual Schematic



Examples of Relevant Experience

- ROD for 200-ZP-1 indicates additional 100-yr period of MNA needed to reach groundwater cleanup goals. Response action for 200-PO-1 OU may include MNA of existing iodine-129, tritium, and nitrate in groundwater plume. *(DOE-RL-2009-10)*

Risk Reduction Mechanism

- Contaminant concentrations reduced by dilution, volatilization, adsorption, microbial degradation, radioactive decay, chemical reactions.
- Biodegradation can transform contaminants into benign compounds. Partial degradation may result in formation of more toxic compounds.
- Plume is diluted or dispersed as it moves through groundwater, reducing toxicity but possibly increasing volume to be treated.

Screening Criteria *(dashed line indicates range)*

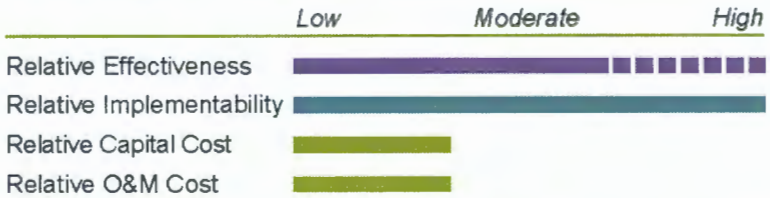


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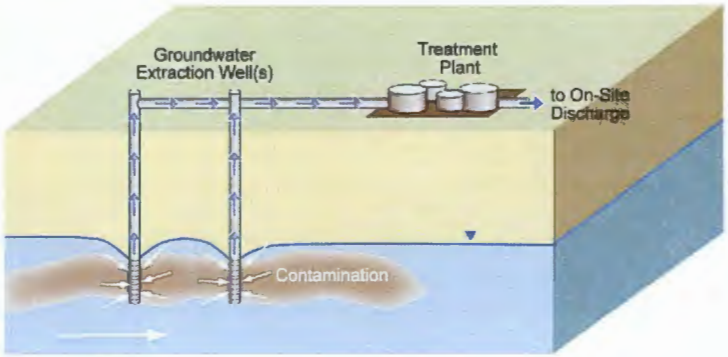
Figure 8-14. Monitored Natural Attenuation

Groundwater Extraction System and On-Site Discharge

Key Components

- Involves continued operation of existing groundwater extraction systems with the potential to expand the system configuration based on remediation goals.
- Treated water is discharged on site.
- Groundwater extraction and injection well network provides for hydraulic containment of contamination.
- Groundwater extraction and on-site discharge are components of a pump-and-treat system, where ex-situ treatment of extracted groundwater can include bioreactors, ion exchange, air stripping, etc.
- Incorporates long-term groundwater monitoring to evaluate system performance, effectiveness, and compliance with remedial action objectives.

Conceptual Schematic



Examples of Relevant Experience

Summary of Groundwater Pump-and-Treat Systems Operating at Hanford

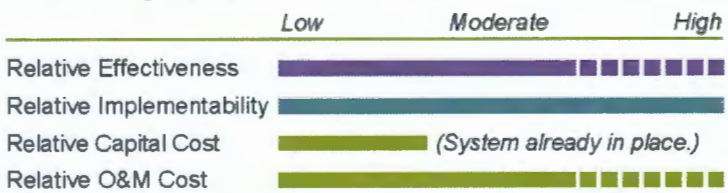
Pump and Treat System	Startup Date	Contaminant	Volume Treated (millions of liters)	Mass Removed (Kg)
100-DR-5[a]	2004	Cr(VI)	384	338
100-DX[a]	2010	Cr(VI)	974	461
100-HR-3[a]	1997	Cr(VI)	4,171	406
100-HX[a]	2011	Cr(VI)	303	11
100-KR-4[a]	1997	Cr(VI)	5,725	355
100-KW[a]	2007	Cr(VI)	1,410	163
100-KX[a]	2009	Cr(VI)	2,594	114
200-ZP-1[b]	1994	CCL4	5,833	13,503
200-UP-1[b]	1994	CCL4, Nitrate, Tc-99, and U	887	49,463

a) DOE/RL-2012-02, Calendar Year 2011 Annual Summary Report for the 100-HR-3 and 100-KR-4 Pump-and-Treat Operations, and 100-NR-2 Groundwater Remediation
b) DOE/RL-2013-03, Calendar Year 2011 Annual Summary Report for the 200-ZP-1 and 200-UP-1 Operable Unit Pump-and-Treat Operations

Risk Reduction Mechanism

- Extraction of groundwater removes contaminants from the subsurface and contains plume to prevent further migration.
- Contaminant volume, toxicity, and mobility are reduced through pump-and-treat process.

Screening Criteria *(dashed line indicates range)*



CHPUBS_RC_0007a

Figure 8-15. Groundwater Extraction System and Onsite Discharge

Ion Exchange

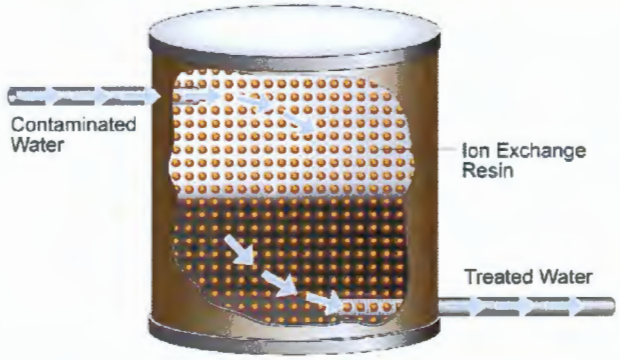
Key Components

- Ions are removed from an aqueous solution and replaced with innocuous ions from the exchange medium.
- Can remove dissolved metals and radionuclides from water.
- Exchange medium can be synthetic resins and inorganic or natural polymeric materials.
- Resins can be regenerated for reuse or disposed of.
- Ion exchange is a non-destructive technology (removal is achieved through mass transfer).

Risk Reduction Mechanism

- Contaminant is transferred to the ion exchange resin which ultimately requires disposal.
- Contaminant volume, toxicity, and migration are reduced through pump-and-treat process.

Conceptual Schematic



Examples of Relevant Experience

- Ion exchange (IX) is the current Hanford groundwater treatment for many pump-and-treat systems: *(DOE/RL-2012-02)*
  - 100-DR-5 system: Removed ~338 kg Cr(VI) since startup in 2004 through calendar year 2011 (CY11)
  - 100-HR-3 system: Removed ~406 kg Cr(VI) since startup in 1997 through CY11
  - 100-KR-4 system: Removed ~355 kg Cr(VI) since startup in 1997 through CY11
  - 100-KW system: Removed ~163 kg Cr(VI) since startup in 2007 through CY11
  - 100-KX system: Removed ~114 kg Cr(VI) since startup in 2009 through CY11
  - 100-DX system: Removed ~461 kg Cr(VI) since startup in December 2010 through CY11
  - 100-HX system: Removed ~11 kg Cr(VI) since startup in October 2011 through CY11

Screening Criteria *(dashed line indicates range)*

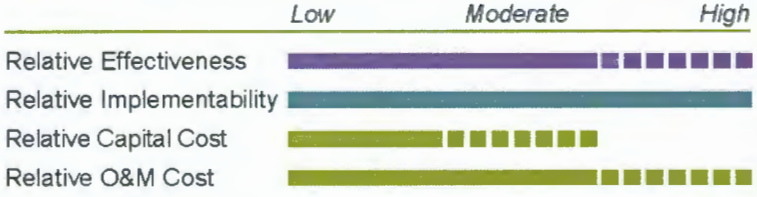


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Figure 8-16. Ion Exchange

In Situ Biological Treatment – Anaerobic (Groundwater)

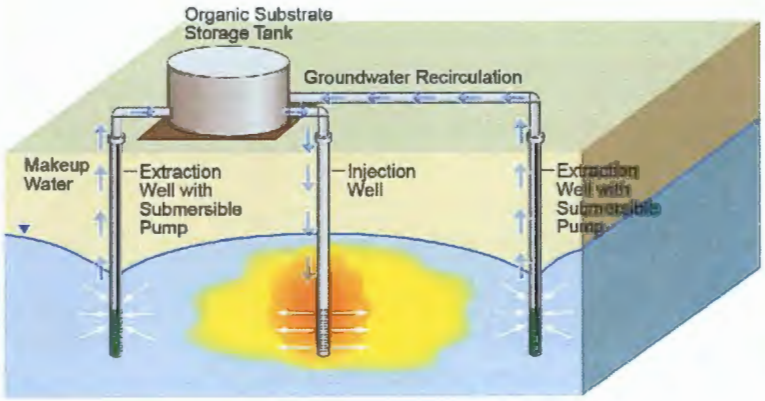
Key Components

- Uses native microorganisms to transform or break down contaminants into less- or non-toxic substances.
- Natural process enhanced by adding organic substrates to stimulate anaerobic micro-organisms in the subsurface.
- The addition of a recirculation system (extract and reinject groundwater) can enhance substrate delivery and increase the zone of influence.
- In-situ reduction of contaminants that are contained in the recycled groundwater reduces the need for more costly ex-situ treatment.
- Localized temporary generation of secondary byproducts (reduced manganese, iron, and arsenic) could be expected.

Risk-Reduction Mechanism

- Biological treatment can dechlorinate CVOCs to less toxic substances, and reduce nitrate to nitrogen gas.

Conceptual Schematic



Examples of Relevant Experience

- Reports for examples of applications using bio-remediation to convert Cr(VI) to less toxic (Cr(III)) include:
  - Hinkley Remediation Semiannual Status Report (July through December 2009), Pacific Gas and Electric Company (PG&E) Groundwater Remediation Program, Hinkley, California (CH2M HILL, 2010)
  - PNNL-18784, Hanford 100-D Area Biostimulation Treatability Test Results.
  - Faybishenko, B., 2009, In Situ Long-Term Reductive Bioimmobilization of Cr(VI) in Groundwater Using Hydrogen Release Compound, Lawrence Berkeley National Laboratory, Berkeley, California
- PG&E implemented two pilot studies at the site: an Upland reductive zone in situ pilot test (ISPT) to evaluate how well recirculation wells can distribute reductant (ethanol) throughout the aquifer to achieve treatment across a transect of the plume, and a Floodplain ISPT to evaluate the efficacy of using lactate to enhance the existing reducing environment in the floodplain adjacent to the Colorado River. List of the reports referenced regarding the two pilot studies implemented by PG&E are available at the California Department of Toxic Substances Control (DTSC) Project Website: http://dtsc-topock.com/.

Screening Criteria *(dashed line indicates range)*

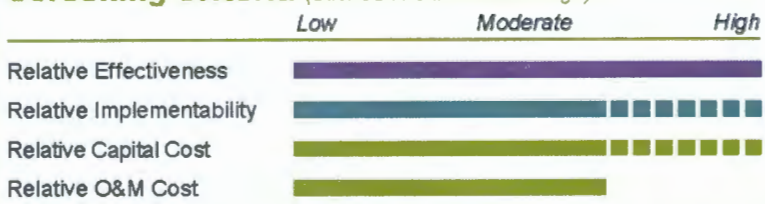


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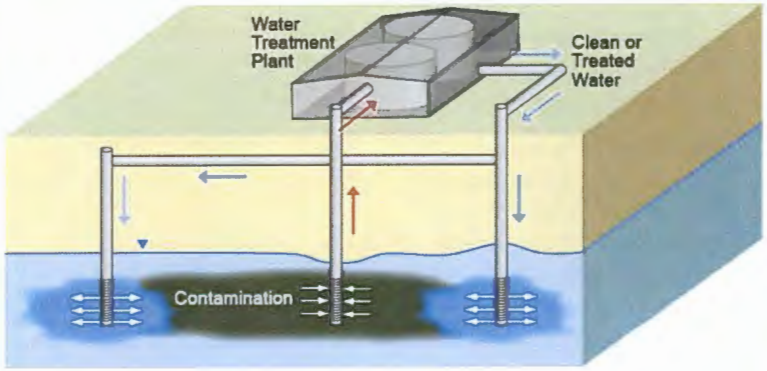
Figure 8-17. In Situ Biological Treatment—Anaerobic (Groundwater)

Flushing – Saturated Zone, Water

Key Components

- Involves the injection of clean or treated water into a zone of contaminated groundwater to expedite remediation of plume.
- Groundwater is captured and treated to meet discharge standards.
- Applicable for media impacted with contaminants with high to moderate solubility (*e.g.*, Cr(VI), Tc-99, uranium, nitrate, and possibly carbon-14).
- Effective groundwater capture is required to contain the plume.
- Groundwater flushing performance depends on residual contamination in lower-permeability layers, lenses, or sorbed to soil.

Conceptual Schematic



Examples of Relevant Experience

- At Topock Compressor Station, Needles, California, the U.S. Department of the Interior proposed water flushing as a preferred alternative for remediation of Cr(VI) in groundwater. This alternative involves injection of fresh and carbon-amended water to flush Cr(VI) and push the plume through in-situ biological treatment barriers located downgradient of the water injection wells. *(DOI060410A)*

Risk Reduction Mechanism

- Extraction of groundwater removes contaminants from the subsurface and contains plume to prevent further migration.
- Reduction in toxicity, mobility, and volume of contaminants is achieved through a groundwater capture and treatment system.

Screening Criteria *(dashed line indicates range)*

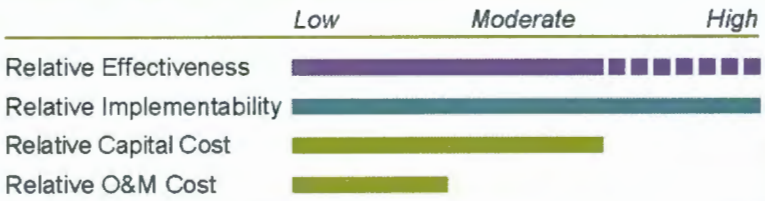


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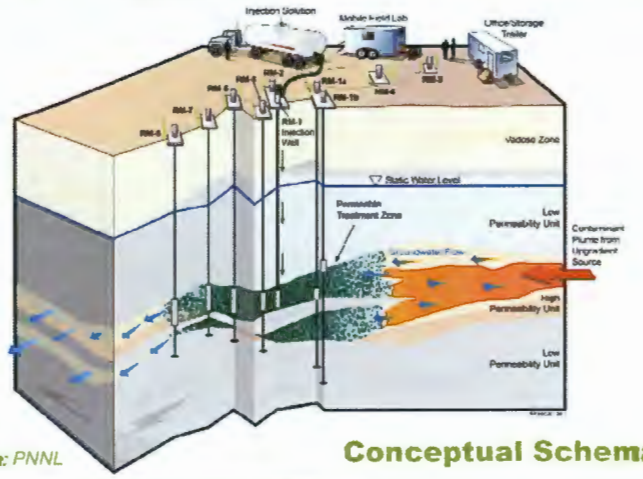
Figure 8-18. Flushing—Saturated Zone, Water

Reactive Chemical Barrier (In-Situ Redox Manipulation) (Groundwater)

Key Components

- Zone of chemically reactive material that transforms (reduces) contaminants in groundwater as it flows through.
- Reactive zone can be generated by a series of injection/recirculation wells or a trench that transects the groundwater flow pathway.
- Reactive material is a reducing chemical (*e.g.*, sodium dithionite or zero valent iron).
- Generation of secondary byproducts and/or break-through may occur.
- Occasional amendments/ applications may be necessary.
- Used to control migration; not effective in treating the bulk of the plume.
- Dispersing reactive material into the aquifer can make implementation complex. Varying hydraulic gradients, and varying water levels can reduce the effectiveness.



Source: PNNL

Conceptual Schematic

Examples of Relevant Experience

- Currently in use in Hanford 100 D; geochemical parameters indicate success in producing the desired Cr-reducing conditions; concentration reductions have been noted, but concentrations in downgradient wells have been variable (*i.e.*, some breakthrough has occurred). *(DOE/RL-2010-11)*

Risk Reduction Mechanism

- Risk reduction achieved through treatment. Risk reduction limited to zone of active treatment and further migration. Does not adequately reduce risk throughout the bulk of the plume.
- Chemical reagents transform (reduce) contaminant to non-or less-toxic compound [*e.g.*, Cr(VI) to Cr(III)]; generation of secondary byproducts may occur .
- ISRM acts as a barrier; when effective, reduces contaminant plume migration/mobility.

Screening Criteria *(dashed line indicates range)*

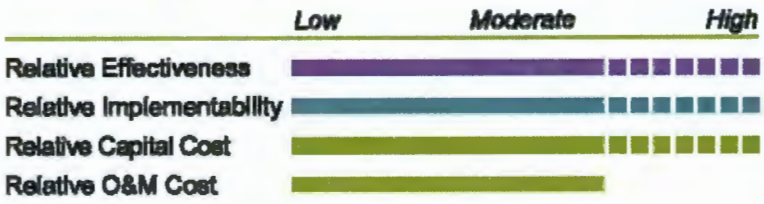


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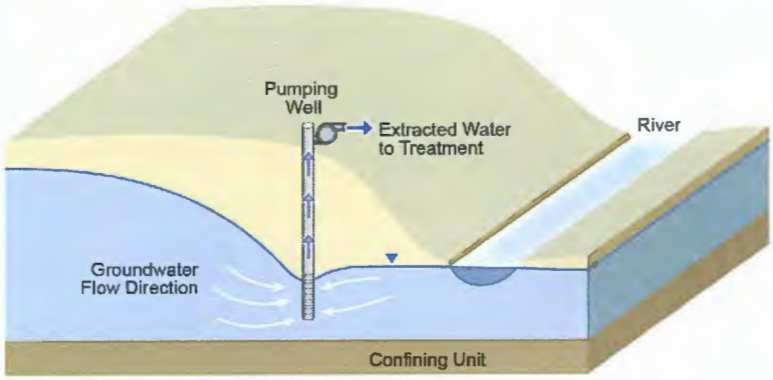
Figure 8-19. Reactive Chemical Barrier (In Situ Redox Manipulation) (Groundwater)

Hydraulic Containment via Extraction (Groundwater)

Key Components

- Extraction wells provide hydraulic containment by pumping groundwater from the plume edge to control contaminant migration.
- Changes groundwater flow characteristics and pulls contaminated groundwater towards the extraction wells.
- Removed groundwater will require treatment or proper disposal.

Conceptual Schematic



Examples of Relevant Experience

- At least eight pump-and-treat systems are successfully operating at Hanford to provide hydraulic containment in addition to removing contamination. Information regarding capture zone efficiency for each system can be found in *Hanford Site Groundwater Monitoring and Performance Report for 2009. (DOE/RL-2010-11)*

Risk Reduction Mechanism

- Reduces mobility by providing a barrier between the contaminated groundwater and the Columbia River.
- Reduces volume by removing dissolved phase contaminant mass; toxicity reduced by subsequent treatment at a temporary or permanent facility.

Screening Criteria *(dashed line indicates range)*

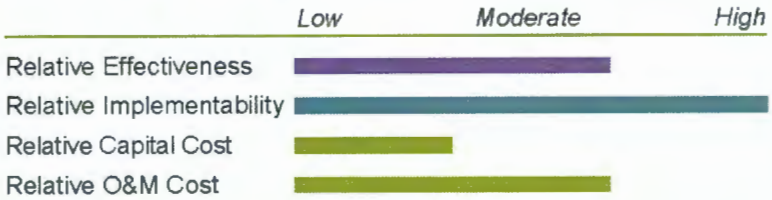


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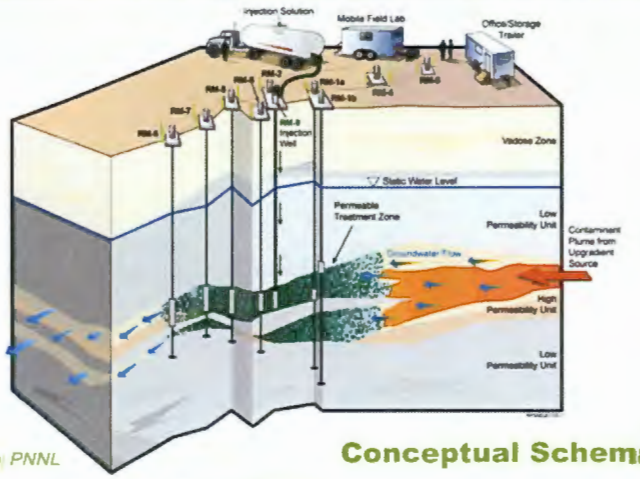
Figure 8-20. Hydraulic Containment via Extraction (Groundwater)

In Situ Chemical Stabilization

Key Components

- Zone of chemically reactive material provides for sequestration of contaminants (Sr-90) in groundwater.
- Reactive zone can be generated by a series of injection wells that transects the groundwater flow pathway.
- Calcium-citrate-phosphate solutions form apatite precipitate [Ca6(PO4)10(OH)2] for sequestration of Sr-90 by substitution of Sr for Ca.
- Occasional amendments (reinjection) may be necessary
- Amendments to injection wells are dependant on river stage and geologic formation for effective treatment



Source: PNNL

Conceptual Schematic

Examples of Relevant Experience

- In situ chemical stabilization is currently in use in Hanford 100-N. An initial 300 foot apatite Permeable Reactive Barrier (PRB) is in place along the 100-N Area shoreline. The PRB uses calcium-citrate-phosphate solution injections for in situ strontium-90 immobilization to reduce the strontium-90 flux to the Columbia River. Strontium-90 concentrations in the groundwater along the PRB have been reduced by 90 percent since injections began in 2006 (DOE/RL-2011-25).

Risk Reduction Mechanism

- Risk reduction achieved through treatment. Risk reduction limited to zone of active treatment.
- Apatite precipitate [Ca6(PO4)10(OH)2] from Ca-citrate-PO4 solutions injected in the PRB sequesters Sr-90 during initial precipitation and additionally slowly incorporates Sr-90 by solid phase substitution for Ca.
- Sufficient apatite needs to be emplaced in sediments to incorporate Sr and Sr-90 and the rate of incorporation needs to exceed the natural groundwater flux rate of strontium in the groundwater.

Screening Criteria *(dashed line indicates range)*

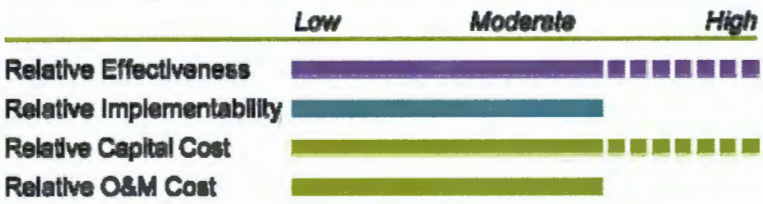


Figure 8-21. In Situ Chemical Stabilization

In Situ Solidification

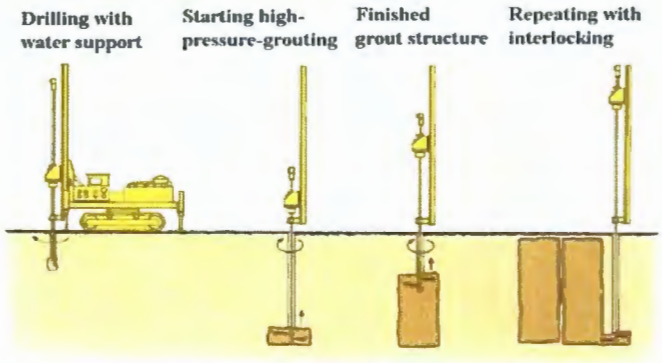
Key Components

- Solidification alters the physical and/or chemical characteristics of a soil through the addition of binders, including cements and chemical grouts, to immobilize contaminants.
- Multiple types of grouting/binding materials and emplacement techniques have been developed and demonstrated.
- The effectiveness of grout emplacement depends on the application method.
- Application for subsurface contaminant solidification in Hanford Site vadose zone sediments include jet grouting and permeation grouting.
- Two types of grout materials may be used: particulate (cement) grouts and chemical grouts. Cement grouts use Portland cement as the primary component. A chemical grout is a solution comprised of a binder (other than Portland cement) that reacts in place to form a gel or solid after injection into a porous subsurface soil, sediment, or rock volume.

Risk Reduction Mechanism

- The immobilization primarily works by coating and isolation processes that make the contaminants less prone to leaching or dissolution.
- The effectiveness of in situ solidification is a function of the distribution of the grout into the formation, the degree of encapsulation of contaminated sediment particles, and the long-term durability of shallow soil encapsulation when exposed to the elements.

Conceptual Schematic



Execution of a jet grout body

Examples of Relevant Experience

- In situ grouting is a component of the remedial actions implemented at the Old F-Area Seepage Basin (OFASB) at the Savannah River Site. The remedial action was completed on June 9, 2000. The grouting and soil cover remedy at OFASB is protective of human health and the environment for soil contamination and prevents external exposure to radiological contaminants as identified in the Second Five Year Review Report for the Savannah River Site (WSRC-RP-2001-4163).
- Treatability testing for in situ grouting of waste sites was completed at Idaho National Laboratory demonstrating the potential viability of grouting to stabilize waste (in situ solidification) in near surface sites (Final Results Report, In Situ Grouting Technology for Appication of Buried Transuranic Waste Sites, Volume 1: Technology Description and Treatability Study Results for OU 7-13/14 [INEEL/EXT-02-00233]).

Screening Criteria *(dashed line indicates range)*

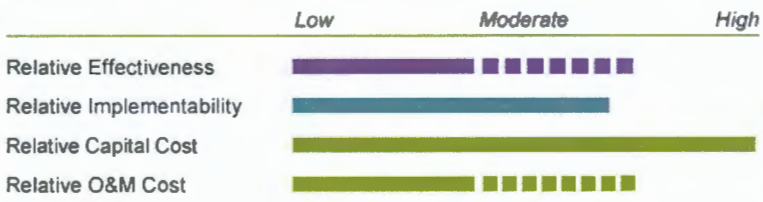


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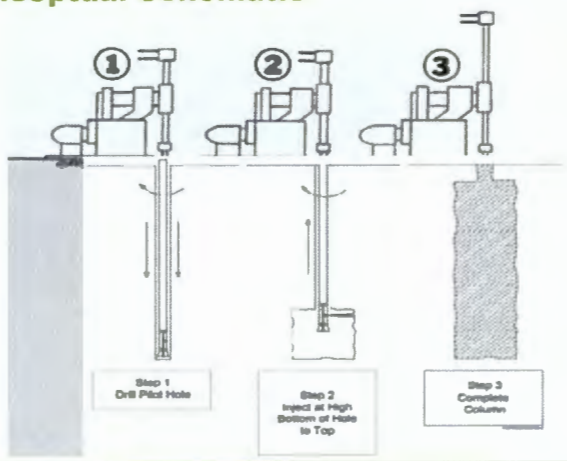
Figure 8-22. In Situ Solidification

In Situ Stabilization/Sequestration

Key Components

- Apatite minerals sequester elements into their molecular structures via isomorphic substitution, whereby elements of similar physical and chemical characteristics replace calcium, phosphate, or hydroxide in the hexagonal crystal structure.
- Phosphate solutions form apatite precipitate [$Ca_5(PO_4)_{10}(OH)_2$] for sequestration of Sr-90 by substitution of Sr for Ca.
- Methods of emplacing apatite in vadose zone soil include injection and infiltration of an aqueous solution containing phosphate.
- Jet injection is capable of delivering a specified amount of solid phase pre-farmed apatite into the vadose zone and upper unconfined aquifer.
- Jet injection delivery overcomes heterogeneity of formation that may impede application of apatite by infiltration.
- Use of heat-treated apatite source (e.g., fish bone or calcined cow bone) being investigated to reduce biomass generation in sediments.

Conceptual Schematic



Examples of Relevant Experience

- Field scale apatite jet injection has been demonstrated in the Hanford 100-N Area. Work was conducted under Strontium-90 Treatability Test Plan for 100-NR-2 Groundwater Operable Unit (DOE/RL-2005-96, Addendum 3). Results from collected sediment cores indicate that jet injection is a viable method for emplacement of phosphate and pre-formed apatite in the vadose zone. These cores also show that jet injection is a viable method for installing a PRB in the vadose zone (*Treatability Test Report for Field-Scale Apatite Jet Injection Demonstration for the 100-NR-2 Operable Unit*, SGW-47062, Rev. 0).

Risk Reduction Mechanism

- Apatite forming materials injected in the vadose zone sequesters Sr-90 during initial precipitation
- Over time, zones that received solid-phase apatite mass will also incorporate additional Sr-90 mass into the apatite.
- The potential effects of increased biomass in the sediments that received pre-formed apatite are still under investigation.

Screening Criteria *(dashed line indicates range)*

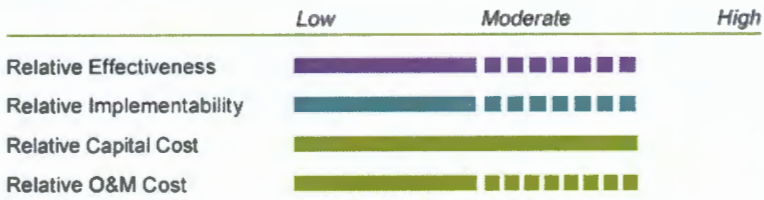


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Figure 8-23. In Situ Stabilization/Sequestration

9 Development and Screening of Alternatives

This chapter discusses the development of remedial action alternatives for 100-D/H. Primary inputs for this process were the physical characteristics of the site (Chapter 3); waste site characterization information, contaminant transport mechanisms, and the CSM (Chapters 4 and 5); the identified risks (Chapters 6 and 7); and the RAOs, target remediation areas, and the remedial technology screening results (Chapter 8).

In this chapter, remedial technologies retained from Chapter 8 are combined into remedial alternatives for 100-D/H that provide a range of technology groupings for integrated waste site and groundwater remediation. With the exception of the No Action Alternative, the remedial alternatives were developed to target achievement of the RAOs by considering the CERCLA program goals and expectations identified in the NCP (40 CFR 300). The remedial alternatives presented in this chapter are carried forward for detailed and comparative evaluation in Chapter 10.

Highlights

- Remedial action alternatives were developed for 100-D/H that provide a range of technology groupings for integrated waste sites and groundwater remediation.
- Alternatives evaluated include the following:
  - Alternative 1: No Action (as required by the NCP)
  - Alternative 2: RTD and Grouting for Waste Site and Pump-and-Treat with Biological Treatment for Groundwater
  - Alternative 3: RTD and Grouting of Waste Site and Increased Capacity Groundwater Pump-and-Treat for Groundwater
  - Alternative 4: RTD for Waste Sites and Pump-and-Treat for Groundwater
- Each waste site remaining for remedial action is evaluated for the alternatives.

9.1 Development of Remedial Alternatives

The NCP (under "Remedial Investigation/Feasibility Study and Selection of Remedy" [40 CFR 300.430(a)(1)(iii)]) sets the following expectations for remedial action alternatives development:

- To use treatment to address the principal threats posed by a site, wherever practicable. Principal threats for which treatment is most likely to be appropriate include liquids, areas contaminated with high concentrations of toxic compounds, and highly mobile materials.
- To use engineering controls, such as containment, for waste that poses a relatively low long-term threat or where treatment is impracticable.
- To use a combination of methods, as appropriate, to achieve protection of human health and the environment. In appropriate site situations, treatment of the principal threats posed by a site, with priority placed on treating waste that is liquid, highly toxic, or highly mobile, will be combined with engineering controls (such as containment) and institutional controls, as appropriate, for treatment of residuals and untreated waste.
- To use institutional controls such as water use and deed restrictions, to supplement engineering controls, as appropriate, for short- and long-term management to prevent or limit exposure to hazardous substances, pollutants, or contaminants. Institutional controls may be used during the conduct of the RI/FS, during implementation of the remedial action, and where necessary, as a component of the completed remedy. The use of institutional controls will not substitute for active response measures (for example, treatment or containment of source material or restoration of groundwater to beneficial use) as the sole remedy unless such active measures are determined not to be practicable.

- To consider using innovative technology when such technology offers the potential for comparable or superior treatment performance or implementability, fewer adverse impacts than other available approaches, or lower costs for similar levels of performance than demonstrated technologies.
- To return usable groundwater to its beneficial uses wherever practicable and within a time frame that is reasonable, given the particular circumstances of the site. When restoration of groundwater to beneficial uses is not practicable, EPA expects that further migration of the plume be prevented, exposure to the contaminated groundwater be prevented, and that further risk reduction be evaluated.
- For groundwater response actions, a limited number of remedial alternatives should be developed to achieve site-specific remediation levels within different restoration periods using one or more different technologies.
- The No Action Alternative (no further action if some removal or remedial action has already occurred at a site) will also be developed.

The purpose of the remedy selection process is to implement remedies that eliminate, reduce, or control risks to human health and the environment.

The remedial alternatives for 100-D/H have been developed to encompass all waste sites carried forward into the FS and groundwater plumes within 100-D/H. This section briefly summarizes the target remediation areas, so the alternative development can focus on the specific areas and COCs at 100-D/H and integrate the remedial alternatives for waste sites and groundwater.

The evaluated alternatives integrate DOE's CERCLA response obligations and RCRA corrective action obligations that relate to the release(s) of hazardous substances, hazardous wastes, pollutants, and contaminants. Therefore, the evaluated alternative is intended to achieve compliance with CERCLA remedial action requirements and satisfy the corrective action requirements of RCRA.

9.1.1 Waste Sites

As presented in Table 8-5 (Chapter 8), 291 waste sites have been included for evaluation in the FS. The COPCs for waste sites are listed in Table 8-1 (Chapter 8). The evaluation of remedial actions relies on the review of available data associated with the waste sites, including field data as available, radiological surveys, process history, analogous site information, personal interviews, engineering drawings and as-builts, and other information identified during the development of the RI/FS. The comprehensive review and evaluation of this information is provided in Chapters 4 through 7 of this RI/FS.

Of the 291 waste sites, 146 waste sites are listed in Chapter 8 as having no identified unacceptable risks and are slated for no further action (see Table 8-5 in Chapter 8). Remedial alternatives are not developed for these sites.

The remaining 145 waste sites are considered in the alternatives analysis. The alternatives are developed based on the risks and known or suspected contaminants for each of the waste sites summarized in Table J-1 (Appendix J). During implementation of remedial actions, should field conditions vary from those presented in the FS and indicate a need to re-evaluate the efficacy of the selected remedial action, the appropriate remedy modification will be used, consistent with CERCLA guidance.

Additional waste site groups considered in the alternatives analysis are described in the following sections.

9.1.1.1 Waste Sites to be Remediated Under Interim Action RODs

The 59 waste sites currently being remediated under the 100 Area Remaining Sites ROD (EPA/ROD/R10-99/039) or anticipated to be remediated by the time the ROD is issued are shown in Table 8-5 (Chapter 8). The cleanup levels in the ROD will be used to determine when remediation is complete for all waste sites.

9.1.1.2 Waste Sites for Institutional Controls

As discussed in Section 8.2.1.1 (Chapter 8), 32 waste sites have been interim or interim no action closed and had verification data with exceedances of human health protection criteria (residential scenario) for select radionuclide compounds (Chapter 6). Contamination was detected in deep zone verification soil samples collected from depths greater than 4.6 m (15 ft) bgs and, as a result, there is no direct exposure pathway. Deep excavation institutional controls will be implemented at the waste sites for remedial alternatives to limit exposure (see Section 9.2.2.1). Radionuclide contamination at the waste sites will continue to decay to below human health protection criteria within 2 to 185 years. Institutional controls will be maintained for these sites until unrestricted use is allowable and remedial action alternatives are not developed for these sites.

Two other waste sites associated with river effluent pipelines (100-D-60 and 100-H-34) were evaluated for risks, as discussed in Chapters 6 and 7. The risk assessments associated with river effluent pipeline investigation indicate no unacceptable human health risk, based on RME scenarios. No technology application is required and remedial action alternatives are not developed for the below river effluent pipeline waste sites in this FS. While no IC is required, an annual inspection under the RCRA permit (Permit Number WA7 89000 8967) is conducted along the shoreline to identify Hanford debris.

9.1.1.3 Waste Sites Remaining for Remedial Action

The identified risks and known or suspected contaminants identified in Table J-1 (Appendix J) for the 52 waste sites remaining for remedial action are used to develop the remedial alternatives and design data for cost estimating. The identification of the risk drivers is based on sample data, if available; on knowledge of the process that was performed at the sites; and on remediation results at similar sites in the River Corridor. One or more of the following risk drivers were identified for each site:

- Human health direct contact risk in shallow soil (less than 4.6 m [15 ft] bgs)
- Ecological risk in shallow soil (less than 4.6 m [15 ft] bgs)
- Groundwater/surface water protection risk for Cr(VI)
- Groundwater/surface water protection risk for other COCs/COPCs

9.1.2 Groundwater

The final COCs for groundwater at 100-D/H include chromium (total), Cr(VI), nitrate, and strontium-90, as identified in Section 8.1.4.5 (Chapter 8).

As part of the development and evaluation of the groundwater alternatives, a groundwater model has been used as an evaluation and design concept tool. Groundwater flow and transport simulations and particle tracking analyses were performed for each design concept to determine the feasibility of each design. The model was also used to perform a limited amount of optimization of well locations and pumping scenarios, including pumping scenarios that achieve groundwater remediation within the period to meet TPA (Ecology et al., 1989a) target milestones for remediation of groundwater. However, the design concepts developed for this FS are not final. They will be updated with additional modeling, including updating the conceptual site model and initial plume configurations, during the remedial design phase, which follows issuance of the ROD.

The groundwater flow model is constructed using the USGS modular groundwater flow model MODFLOW. Particle tracking was performed using the USGS program MODPATH. To simulate the contaminant plume migrations, the model MT3DMS was used. Model development and calibration are documented in a comprehensive modeling report contained in Appendix F (*Conceptual Framework and Numerical Implementation of 100 Areas Groundwater Flow and Transport Model* [SGW-46279]). The initial Cr(VI) plume distribution in the 100-HR-3 OU used in the groundwater model simulation is shown on Figure 9-1. Figures 9-2 and 9-3 show the initial distributions for strontium-90 and nitrate, respectively. The results of the groundwater alternative modeling are included in Appendix F (*Modeling of RI/FS Design Alternatives for 100-HR-3* [ECF-100HR3-11-0114]). These supplemental documents also discuss the uncertainty with the model results because of variability in subsurface conditions and other factors.

9.2 Description of Remedial Alternatives

As suggested by CERCLA RI/FS Guidance (EPA/540/G-89/004), alternatives were developed that incorporate process options and technologies retained (Chapter 8) and include an appropriate range of waste management options to ensure the protection of human health and the environment. In addition, according to EPA guidance, the alternatives address contamination for affected media at the entire 100-D/H Area (for example, waste sites and groundwater combined in each alternative).

Each alternative was developed based on the application of the retained technologies for waste site and groundwater remediation as identified in Sections 8.3.2.1 and 8.3.2.2, respectively. Four alternatives are evaluated:

- **Alternative 1 – (No Action [as required by the NCP]).** This alternative is required by the NCP ("Remedial Investigation/Feasibility Study and Selection of Remedy" [40 CFR 300.430(e)(6)]). Further description for this alternative is provided in Section 9.2.1.

- **Alternative 2 – RTD and Grouting for Waste Site and Pump-and-Treat with Biological Treatment for Groundwater.** This alternative uses RTD for removal of contamination to cleanup levels for waste sites. Void-fill grouting will be used for the box flume of waste site 100-H-36 where RTD would have large ecological impacts near the river. For groundwater, a pump-and-treat system and biological treatment of Cr(VI) will be used. Nitrate and strontium-90 contaminated groundwater plumes are within the treatment footprint for the Cr(VI) plume and will be co-extracted by the extraction well network used for the Cr(VI) plume remediation. The groundwater treatment system effluent is not expected to exceed MCLs and no treatment is proposed for strontium-90 and nitrate. Specific treatment would be provided if the extracted groundwater in the pump-and-treat effluent stream exceeds the MCLs for the respective COCs before reinjection. MNA and institutional controls will be used for strontium-90. Groundwater monitoring will confirm effectiveness of MNA for strontium-90 and determine the impact of pump-and-treat on the persistence of the COC within the aquifer over time. Institutional controls to restrict excavation will be applied at individual waste sites with residual risks associated with contamination at depths greater than 4.6 m (15 ft) bgs after remediation, but there is no direct exposure pathway. Additional institutional controls to waste sites may be added through closure reclassifications. Further description for this alternative is provided in Section 9.2.3.

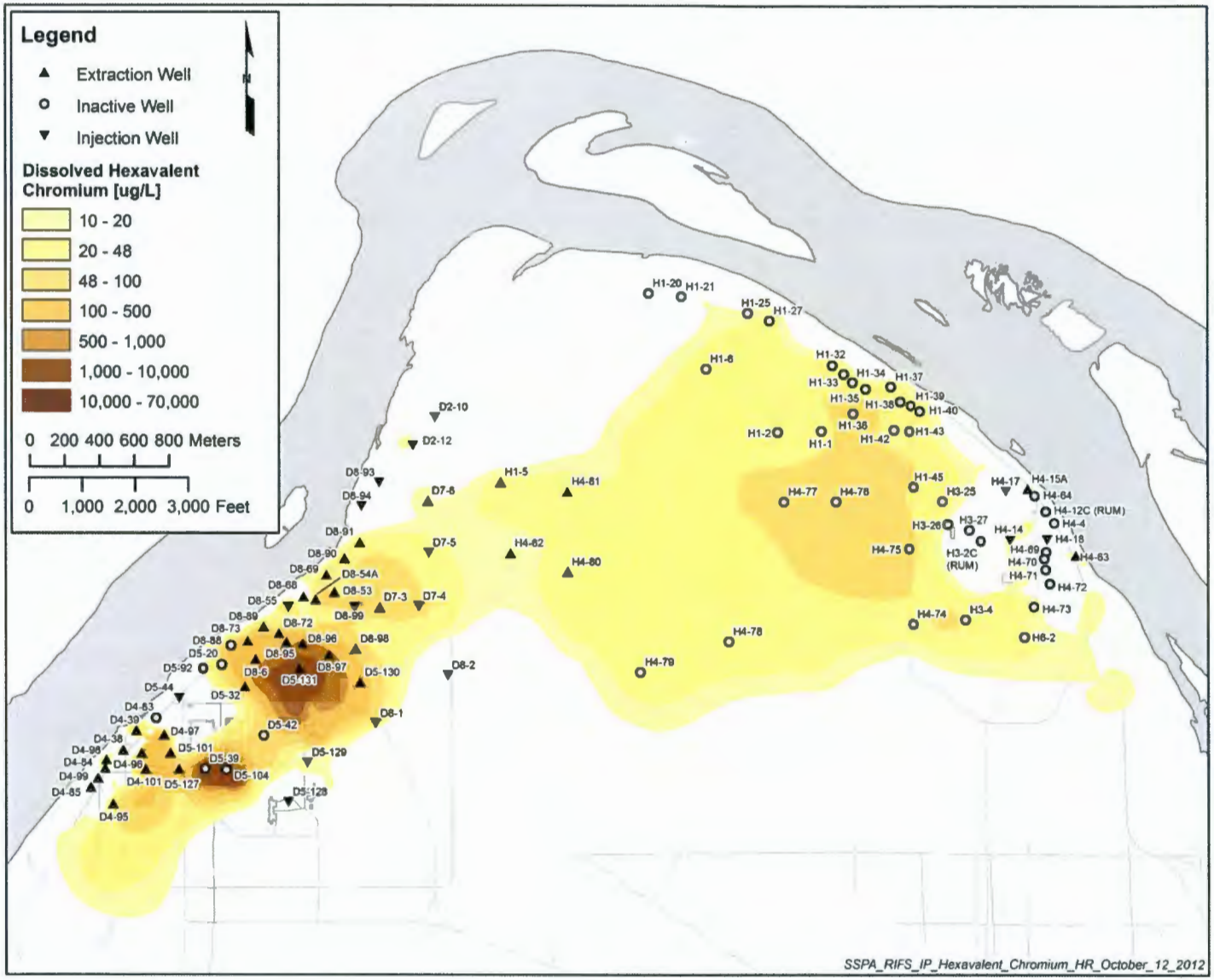

Figure 9-1. Initial Modeled Cr(VI) Plume

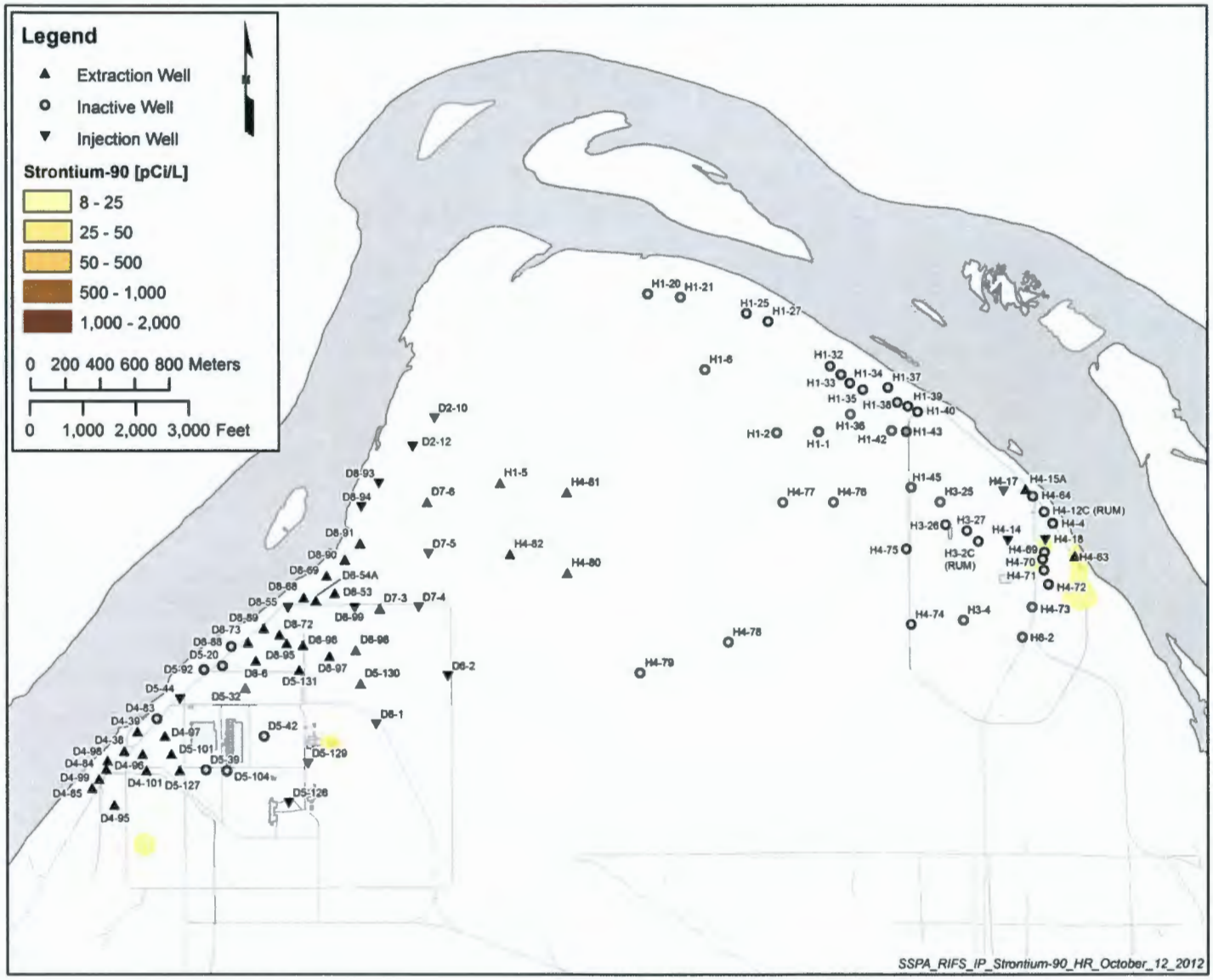

Figure 9-2. Initial Modeled Strontium-90 Plume

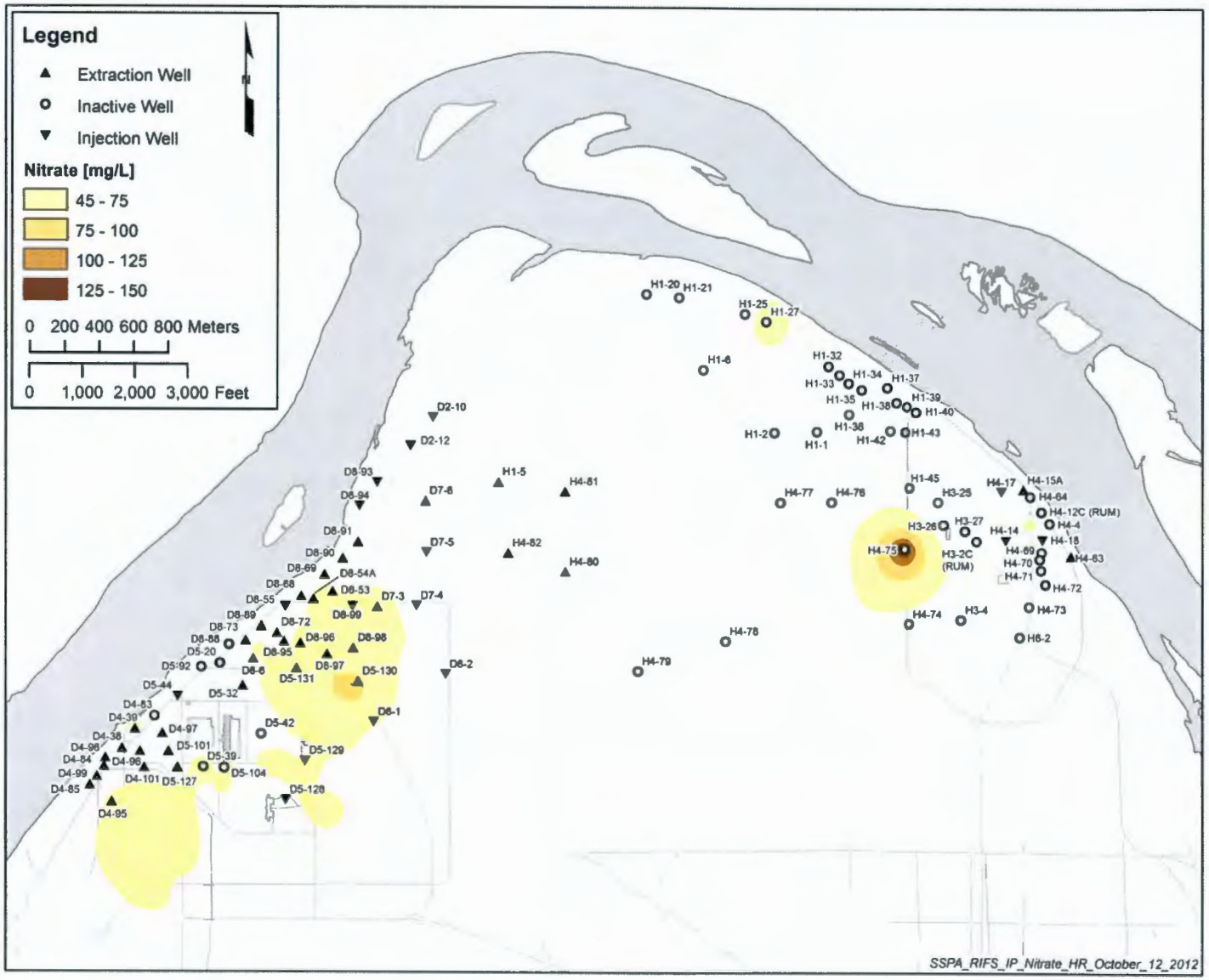

Figure 9-3. Initial Modeled Nitrate Plume

- **Alternative 3 – RTD and Grouting of Waste Site and Increased Capacity Groundwater Pump-and-Treat for Groundwater.** This alternative uses RTD for removal of contamination to cleanup levels for waste sites. Void-fill grouting will be used for the box flume of waste site 100-H-36 where RTD would have large ecological impacts near the river. For groundwater, an expanded pump-and-treat system for treatment of Cr(VI) will be used. Nitrate and strontium-90 contaminated groundwater plumes are within the treatment footprint for the Cr(VI) plume and will be co-extracted by the extraction well network used for the Cr(VI) plume remediation. The groundwater treatment system effluent is not expected to exceed MCLs and no treatment is proposed for strontium-90 and nitrate. Specific treatment would be provided if the extracted groundwater in the pump-and-treat effluent stream exceeds the MCLs for the respective COCs before reinjection. MNA and institutional controls will be used for strontium-90. Groundwater monitoring will confirm effectiveness of MNA for strontium-90 and determine the impact of pump-and-treat on the persistence of the COC within the aquifer over time. Institutional controls to restrict excavation will be applied at individual waste sites with residual risks associated with contamination at depths greater than 4.6 m (15 ft) bgs after remediation, but there is no direct exposure pathway. Additional institutional controls to waste sites may be added through closure reclassifications. Further description for this alternative is provided in Section 9.2.4.

- **Alternative 4 – RTD for Waste Sites and Pump-and-Treat for Groundwater.** This alternative uses RTD for removal of contamination to cleanup levels for waste sites. For groundwater, pump-and-treat system for treatment of Cr(VI) will be used. Nitrate and strontium-90 contaminated groundwater plumes are within the treatment footprint for the Cr(VI) plume and will be co-extracted by the extraction well network used for the Cr(VI) plume remediation. The groundwater treatment system effluent is not expected to exceed MCLs and no treatment is proposed for strontium-90 and nitrate. Specific treatment would be provided if the extracted groundwater in the pump-and-treat effluent stream exceeds the MCLs for the respective COCs before reinjection. MNA and institutional controls will be used for strontium-90. Groundwater monitoring will confirm effectiveness of MNA for strontium-90 and determine the impact of pump-and-treat on the persistence of the COC within the aquifer over time. Institutional controls to restrict excavation will be applied at individual waste sites with residual risks associated with contamination at depths greater than 4.6 m (15 ft) bgs after remediation, but there is no direct exposure pathway. Further description for this alternative is provided in Section 9.2.5.

The technology evaluation in Chapter 8 explored options for the treatment of vadose zone and groundwater contamination. A wide range of technologies, such as ISRM for groundwater treatment, have been tested and applied at Hanford with varying success. Waste site remediation using RTD has been the selected alternative for interim actions and has been used successfully over the past 16 years at Hanford. Similarly, treatment of Cr(VI) using pump-and-treat technology has been implemented at 100-D/H to meet cleanup goals. Additional retained technologies were not included in the alternatives development but may be considered for future remedial actions.

Table 9-1 summarizes the retained technologies identified in Chapter 8 and shows the application to remedy vadose zone soils, including waste sites, and groundwater for the remedial alternatives. Table 9-2 identifies the technologies applied for each alternative to each of the 291 waste sites that are carried into the FS. Institutional controls applied to the waste sites are discussed separately in Section 9.2.2.1, so institutional control components are not included in Table 9-2 and identified separately in Table 9-4.

Table 9-1. Retained Technologies Applied to Remedial Action Alternatives

Remedial Technology	Process Option	Alternative 1	Alternative 2	Alternative 3	Alternative 4
Vadose Zone					
No action	No Further Action	X	X	X	X
Excavation	Standard excavation		X	X	X
	Deep excavation		X	X	X
Disposal	Disposal to ERDF or other EPA-approved location		X	X	X
In situ treatment via reagent – delivery method	Void–fill grouting/grouting		X	X	
Groundwater					
No action	No Further Action	X			
Monitored natural attenuation	Monitored natural attenuation		X	X	X
In situ treatment – reagent approach	In situ biological treatment (anaerobic)[a]		X		
	Surface infiltration		X		
	Vertical wells[a,b]		X		
Pump-and-treat – collection	Groundwater extraction system[b]		X	X	X
Pump-and-treat – ex situ treatment	Ion exchange[b]		X	X	X
Pump-and-treat – discharge	Groundwater injection wells[b]		X	X	X
Containment	Hydraulic containment via extraction		X	X	X

a. In situ biological treatment and vertical wells when used together are called bioinjection.

b. Pump-and-treat includes the combination of groundwater extraction using vertical wells, ex situ ion exchange treatment, and groundwater injection using vertical wells.

Table 9-2. Components for Remedial Action Alternatives

Vadose Zone Soil Site	Alternative 1	Alternative 2	Alternative 3	Alternative 4
Waste Sites Remaining for Remedial Action				
100-D-10, 100-D-52, 100-D-59, 100-D-63, 100-D-75:1, 100-D-96, 100-D-98:1, 100-D-101, 100-D-102, 100-D-103, 118-DR-2:2, 100-H-5, 100-H-28:7, 100-H-38, 100-H-57, 100-H-58, 100-D-98:2, 100-D-105, 100-D-106, 100-H-56, 100-H-59, 100-H-51:6, 600-380, 600-381, 600-382:1, 600-382:2, 600-382:3, 600-382:4, 600-382:5, 600-383:1, 600-383:2, 600-383:3, 600-383:4, 600-383:5, 600-383:6, 600-383:7, 600-383:8, 600-383:9, 600-383:10, 600-384:1, 600-384:2, 600-384:3, 600-384:4, 600-384:5, 600-385, 100-D-107, 100-H-54	No Further Action	RTD to remove contamination using standard and/or deep excavation with disposal at ERDF		
100-D-50:2	No Further Action	Cap ends of pipe to contain contamination and maintain institutional control	Cap ends of pipe to contain contamination and maintain institutional control	RTD to remove contamination using standard and/or deep excavation with disposal at ERDF
100-H-36	No Further Action	Void-fill grouting of box flume RTD of spillway to ordinary high water mark	Void-fill grouting of box flume RTD of spillway to ordinary high water mark	RTD to remove contamination using standard and/or deep excavation with disposal at ERDF
116-DR-9, 100-D-25, 116-D-8	No Further Action	MNA/institutional control	MNA/institutional control	RTD to remove contamination using standard and/or deep excavation with disposal at ERDF
Waste Sites to be Remediated under Interim Action RODs				
100-D-8, 100-D-14, 100-D-31:11, 100-D-31:12, 100-D-50:1, 100-D-50:4, 100-D-50:6, 100-D-50:7, 100-D-50:8, 100-D-50:9, 100-D-62, 100-D-65, 100-D-66, 100-D-69, 100-D-71, 100-D-72, 100-D-73, 100-D-75:2, 100-D-76, 100-D-77, 100-D-78, 100-D-80:2, 100-D-81, 100-D-83:1, 100-D-83:2, 100-D-83:3, 100-D-83:5, 100-D-84:2, 100-D-85:2, 100-D-86:1, 100-D-86:3, 100-D-97, 100-D-99, 100-D-100, 100-D-104, 100-H-28:2, 100-H-28:3, 100-H-28:4, 100-H-28:5, 100-H-42, 100-H-43, 100-H-44, 100-H-46, 100-H-48, 100-H-49:1, 100-H-51:1, 100-H-51:2, 100-H-51:3, 100-H-52, 116-DR-3, 118-D-2:1, 118-D-2:2, 118-D-3:1, 118-D-3:2, 126-DR-1, 126-H-2, 132-H-3, 1607-D2:5, 100-D-30	No Further Action	RTD to remove contamination using standard and/or deep excavation with disposal at ERDF		

Table 9-2. Components for Remedial Action Alternatives

Vadose Zone Soil Site	Alternative 1	Alternative 2	Alternative 3	Alternative 4
Waste Sites Considered for No Further Action				
100-D-1, 100-D-2, 100-D-3, 100-D-4, 100-D-7, 100-D-9, 100-D-12, 100-D-13, 100-D-15, 100-D-20, 100-D-21, 100-D-22, 100-D-23, 100-D-24, 100-D-28:1, 100-D-29, 100-D-31:1, 100-D-31:2, 100-D-31:3, 100-D-31:4, 100-D-31:5, 100-D-31:6, 100-D-31:7, 100-D-31:8, 100-D-31:9, 100-D-31:10, 100-D-32, 100-D-42, 100-D-43, 100-D-45, 100-D-47, 100-D-48:4, 100-D-49:3, 100-D-50:5, 100-D-53, 100-D-54, 100-D-56:1, 100-D-56:2, 100-D-61, 100-D-64, 100-D-70, 100-D-74, 100-D-75:3, 100-D-80:1, 100-D-82, 100-D-83:4, 100-D-84:1, 100-D-85:1, 100-D-87, 100-D-88, 100-D-90, 100-D-94, 100-H-17, 100-H-2, 100-H-24, 100-H-28:1, 100-H-28:6, 100-H-3, 100-H-30, 100-H-35, 100-H-37, 100-H-4, 100-H-40, 100-H-41, 100-H-45, 100-H-49:2, 100-H-50, 100-H-51:4, 100-H-51:5, 100-H-53, 100-H-7, 100-H-8, 116-D-10, 116-D-2, 116-D-4, 116-D-5, 116-D-6, 116-D-9, 116-DR-10, 116-DR-4, 116-DR-5, 116-DR-7, 116-DR-8, 116-H-2, 116-H-5, 116-H-9, 118-D-1, 118-D-4, 118-D-5, 118-DR-1, 118-H-1:1, 118-H-1:2, 118-H-2, 118-H-3, 118-H-4, 118-H-5, 118-H-6:4, 118-H-6:5, 120-D-2, 126-D-2, 128-D-2, 128-H-1, 128-H-2, 128-H-3, 130-D-1, 132-D-1, 132-D-2, 132-D-3, 132-DR-1, 1607-D1, 1607-D2:1, 1607-D2:2, 1607-D2:3, 1607-D2:4, 1607-D4, 1607-D5, 1607-H1, 1607-H2, 1607-H3, 1607-H4, 600-30, 600-151, 600-152, 628-3, UPR-100-D-5, 100-D-50:3, 100-D-50:10, 100-D-67, 100-D-68, 100-D-86:2, UPR-100-D-1, 118-D-6:2, 132-D-4, 132-DR-2, 100-H-9, 100-H-10, 100-H-13, 100-H-31, 100-H-28:8, 100-H-33, 116-D-3, 116-H-4, 118-H-6:2, 128-D-1, 132-H-1, 132-H-2	No Further Action	No further action as discussed in Section 8.2.1.1 (Chapter 8)		

The *National Historic Preservation Act of 1966* is an ARAR for remedial actions where cultural resources are present. Remediation that has the potential to affect cultural resources will require an analysis of cultural resource impacts before any remedial action. Such an analysis is required by the ARARs discussed in Chapter 8. This will include an assessment of the cultural resources present at a site in accordance with the *Hanford Cultural Resources Management Plan* (DOE/RL-98-10). The guidelines and strategies have been developed based on the Hanford Site's unique history and cultural resources, and through recurring discussions with the State Historic Preservation Office and the Native American Tribes and Nations regarding the protective and mitigative measures that are needed. If during design or implementation of the remedy, culturally sensitive sites are identified for which mitigation activities to protect cultural resources would be inadequate, DOE and EPA will work with the Tribes to identify an alternative remediation strategy. This alternative remediation strategy would be implemented through a ROD change.

Table J-3 (Appendix J) provides additional information for each waste site evaluated in the development of alternatives. Details regarding the development of cost estimates are presented in *100-DH Cost Estimate Scoping Forms for Feasibility Study Alternative Costing* (ECF-100DR1-12-0022) in Appendix J.

Table 9-3 shows the remedial system components that are used for the cost estimates for the groundwater remediation alternatives. The cleanup period projections included in Table 9-3 are based on the fate and transport models. Estimated quantities for key groundwater remedial components of the selected approach will be refined in the RD/RAWP. The fate and transport model's details, assumptions, and implementation are included in Appendix F (*Modeling of RI/FS Design Alternatives for 100-HR-3* [ECF-100HR3-11-0114]), and the target milestones for groundwater remediation are as follows:

- M-016-110-T01, target date of December 31, 2012; take actions to contain or remediate Cr(VI) 100 Area groundwater plumes
- M-016-110-T04, target date of December 31, 2016; implement/start remedial actions in all 100 Area OU RODs for groundwater
- M-016-110-T02, target date of December 31, 2020; take actions such that Cr(VI) meets the MTCA (WAC 173-340) Method B cleanup level of 48 µg/L

Table 9-3. Groundwater Alternative Remedial Components

Groundwater Alternative Components	Alternative 1	Alternative 2	Alternative 3	Alternative 4
Pump-and-Treat				
Number of extraction wells operating during time frame	NA	70	126	97
Number of injection wells operating during time frame	NA	30	49	35
Number of new/converted extraction wells installed	NA	18	65	32
Number of new injection wells installed	NA	8	25	12
Operation time frame	NA	25 years	12 years	39 years
Extraction rate (L/min [gal/min])	NA	4,500 (1,200)	9,000 (2,400)	4,500 (1,200)
Ion exchange treatment (L/min [gal/min])	NA	4,500 (1,200)	9,000 (2,400)	4,500 (1,200)
Above ground piping (m [ft])	NA	47,000 (154,200)	458,700 (139,900)	211,200 (64,400)
Monitoring				
Number of new monitoring wells installed	NA	12	12	12
Number of groundwater samples collected	NA	720/annually	864/annually	728/annually

Table 9-3. Groundwater Alternative Remedial Components

Groundwater Alternative Components	Alternative 1	Alternative 2	Alternative 3	Alternative 4
Bioinjection for Hexavalent Chromium				
Number of bioinjection extraction wells	NA	13	N/A	N/A
Number of bioinjection injection wells	NA	5	N/A	N/A
Injection rate (L/min [gal/min])	NA	1,000 (270)	N/A	N/A
Biological injection volume (million L [million gal])*	NA	450 (120)	N/A	N/A
Number of bio node mixing facilities	NA	1	N/A	N/A
Number of biosubstrate mixing plants	NA	1	N/A	N/A
Above ground piping (m [ft])	NA	609 (2,000)	N/A	N/A

Note: The estimated quantities for key groundwater remedial components included above are projected and were developed for cost estimating purposes for this FS. Estimated quantities of the selected approach would be determined during the RD/RAWP. Extraction and injection rates include new and existing wells.

* Total volume of solution at 100 mg/L to 1,000 mg/L as carbon substrate

NA = not an applicable component of the alternative

The operating periods (Table 9-3) for Alternatives 2, 3, and 4 are based on achieving the groundwater cleanup levels for Cr(VI) throughout the aquifer as indicated by the predicted EPC. The EPC is estimated conservatively in the groundwater model as the maximum concentration predicted by the fate and transport simulation, but this will likely be overly conservative and does not take into account design considerations and actual operations of the pump and treat system. For future monitoring of the selected remedy performance, the EPC can be based on the 95th UCL on the mean groundwater concentration values from a specified group of wells and for a specified time window (e.g., annual or 2 years). These future EPCs can be used to demonstrate groundwater remediation to 48 µg/L (RAO 1) within the four different remediation areas specific to the Cr(VI) plumes (that is, 100-D northern plume, 100-D southern plume, 100-H, and horn area plumes), and 10 µg/L where groundwater has the potential to discharge to surface water (RAO 2). Compliance guidance from the state (MTCA, "Groundwater Cleanup Standards" [WAC 173-340-720(9)(d)(i)]; *Methods for Evaluating The Attainment Of Cleanup Standards Volume 2: Ground Water* [EPA 230-R-92-014]) will be used for these purposes. These activities will require a statistical evaluation of the monitoring well network data, which will be defined in the remedial design phase for this area.

Cr(VI) is the primary groundwater COC and has the largest contaminant plume area at 100-D/H. The other groundwater COCs lie largely within the footprint of the Cr(VI) plumes. The extraction well network installed for remediation of the Cr(VI) plumes is expected to capture the other groundwater COCs. Groundwater modeling results presented in Appendix F show that the strontium-90 and nitrate concentrations in the combined pump-and-treat influent are less than their respective MCLs and will remain below the MCLs upon injection of the pump-and-treat effluent to the aquifer. Specific treatment

would be provided if the combined extracted groundwater in the pump-and-treat effluent stream exceeds the MCLs for the respective COCs before reinjection.

Alternative development includes the following approaches to treat the COCs in groundwater:

- **Cr(VI):** Interim remedial actions using pump-and-treat systems with ion exchange treatment technology to remediate Cr(VI)-contaminated groundwater have been very effective in removing Cr(VI) mass from the aquifer. The remedial alternatives include expansion of the interim action pump-and-treat systems using ion exchange treatment technology. Alternative 2 also augments the treatment process with in situ biological treatment.

- **Nitrate:** Nitrate-contaminated groundwater will be co-extracted by the extraction well network used for the Cr(VI) plume remediation. Nitrate concentration in the combined pump-and-treat influent will be less than the MCL of 45,000 µg/L and will remain below the MCL upon injection of the pump-and-treat effluent to the aquifer. Specific treatment would be provided if the combined extracted groundwater in the pump-and-treat effluent stream exceeds the MCL before reinjection.

- **Total chromium:** Under the current geochemical conditions at the site, the majority of total chromium in groundwater exists as Cr(VI). Therefore, total chromium will not be specifically addressed in the alternatives but will be treated in conjunction with the selected Cr(VI) remediation alternative. Chromium(III) is the only other form of chromium likely to be at the site. The ISRM barrier and biostimulation treatability test area at 100-D formed treatment areas with reducing-type environment in the aquifer to reduce Cr(VI) to Cr(III) where the less toxic, immobile Cr(III) precipitates from solution. Cr(III) has very low solubility (K_d = 200 mL/g) and is not likely to be present in the groundwater at high concentrations. Samples from monitoring wells around the ISRM and biostimulation treatment areas with total chromium concentrations exceeding water quality criteria had comparable Cr(VI) concentrations that also exceeded water quality criteria, indicating the total chromium in groundwater is predominantly Cr(VI). Treatment of the chromium groundwater plumes to the Cr(VI) cleanup levels will also result in achievement of the total chromium standard because concentration limits for Cr(VI) are lower (total chromium PRG is 65 µg/L).

- **Strontium-90:** Strontium-90 found at concentrations above the MCL (8 pCi/L) in small, localized areas at 100-D and 100-H will be managed through MNA and institutional controls. Natural attenuation by radiological decay is another important component for managing the strontium-90 groundwater plume. Strontium-90 has low solubility and mobility, and has a half-life of 29.1 years. Appendix F presents groundwater modeling runs that were performed to evaluate the potential risk from strontium-90. Based on the groundwater modeling results presented in Appendix F, the strontium-90 contamination is likely to remain in small areas that can be monitored. Modeling predicts that the strontium-90 concentration will decrease by radiological decay to below the MCL within 25 years, based on the 90th percentile concentration of 14 pCi/L calculated in Section 6.3.2, and within 61 years based on the maximum nonsuspect concentration of 34 pCi/L. Strontium-90 contaminated groundwater co-extracted with Cr(VI) extraction wells will have concentration in the combined pump-and-treat influent less than the MCL of 8 pCi/L and will remain below the MCL upon injection of the pump-and-treat effluent to the aquifer. Specific treatment would be provided if the combined extracted groundwater in the pump-and-treat effluent stream exceeds the MCL before reinjection.

Remedy performance monitoring is conducted to evaluate effectiveness of the alternative to attain the cleanup levels as described in Section 9.2.2.7.

The design concepts presented for each alternative were developed to the level required to prepare a cost estimate that will allow comparison of the alternatives. The cost estimate accuracy recommended in

CERCLA RI/FS Guidance [EPA/540/G-89/004]) is a range of -30 to +50 percent. Significantly, more detail on the selected remedy for 100-D/H will be developed during the design phase, after the ROD is finalized. An RD/RAWP will be developed to discuss in detail the design of the specific components for each waste site and groundwater plume.

For each of the remedial action alternatives, the steps for establishing remedial action completion are as follows:

1. **Obtain waste site closure:** Once the waste sites have been remediated and verification sampling demonstrates acceptable levels of the COCs, closure will be obtained following the procedures in the ROD and RD/RAWP.

2. **Evaluate for groundwater remedial action completion:** Once groundwater cleanup levels are achieved, ongoing performance monitoring will be used to demonstrate that cleanup requirements have been achieved and maintained.

9.2.1 Alternative 1—No Action Alternative

The NCP ("Remedial Investigation/Feasibility Study and Selection of Remedy" [40 CFR 300.430(e)(6)]) requires consideration of a No Action Alternative. The No Action Alternative, which serves as a baseline for evaluating other remediation action alternatives, is retained throughout the FS process. No action means that remediation would not be implemented to alter the existing conditions. For this alternative, it is assumed that all site remedial activities and interim actions, with the possible exception of backfilling any unsafe open excavations, will be discontinued in December 2012. Operation of the existing DX and HX pump-and-treat systems and any other monitoring would cease. No design or cost estimates are prepared for Alternative 1 because no actions are proposed.

Figure 9-4 (a-d) presents the groundwater model prediction of Cr(VI) levels for this alternative 3, 18, 38, and 75 years after remedial actions are discontinued in 2012. The groundwater model simulations (*Modeling of RI/FS Design Alternatives for 100-HR-3* [ECF-100HR3-11-0114]) assume no continuing sources for groundwater contamination. Because the pump-and-treat systems are shut down after 2012, the containment of the plume along the river that is observed at 2012 is lost, as can be seen in the model prediction after 3 years of terminating interim actions. Some mass removal is predicted to occur through natural flushing, as can be seen in the changes in concentrations out through 75 years. However, relatively large areas with greater than 10 µg/L Cr(VI) are predicted to remain after 75 years. If waste site remediation is not complete, as assumed, then the area with greater than 10 µg/L Cr(VI) would be larger.

The concentration plume depictions shown on Figure 9-4 (a-d) reflect the maximum Cr(VI) concentrations calculated from the model simulation. The groundwater model simulation also provides predicted maximum, 95 UCL, 90th percentile, mean, and median concentrations and trends for the groundwater COCs. Results of the groundwater model simulation for the COCs are provided in *Modeling of RI/FS Design Alternatives for 100-HR-3* (ECF-100HR3-11-0114), with prediction of COC plumes and trends. The 95 UCL and 90th percentile concentration trends indicate that cleanup levels would not be achieved at all areas within the simulation period (75 years) and concentrations along the shoreline, entering the river, would exceed cleanup levels.

The shoreline concentration trends *Modeling of RI/FS Design Alternatives for 100-HR-3* (ECF-100HR3-11-0114) reflect transient state (that is, time varying) conditions in the aquifer from water level changes resulting from river stage variation. The simulations are discretized into 12 monthly stress periods over the first 25 years to reflect the seasonal variances in river stage. For the remaining simulation period, a single transient stress period is used with the river stage elevation remaining constant to reflect annual average conditions.

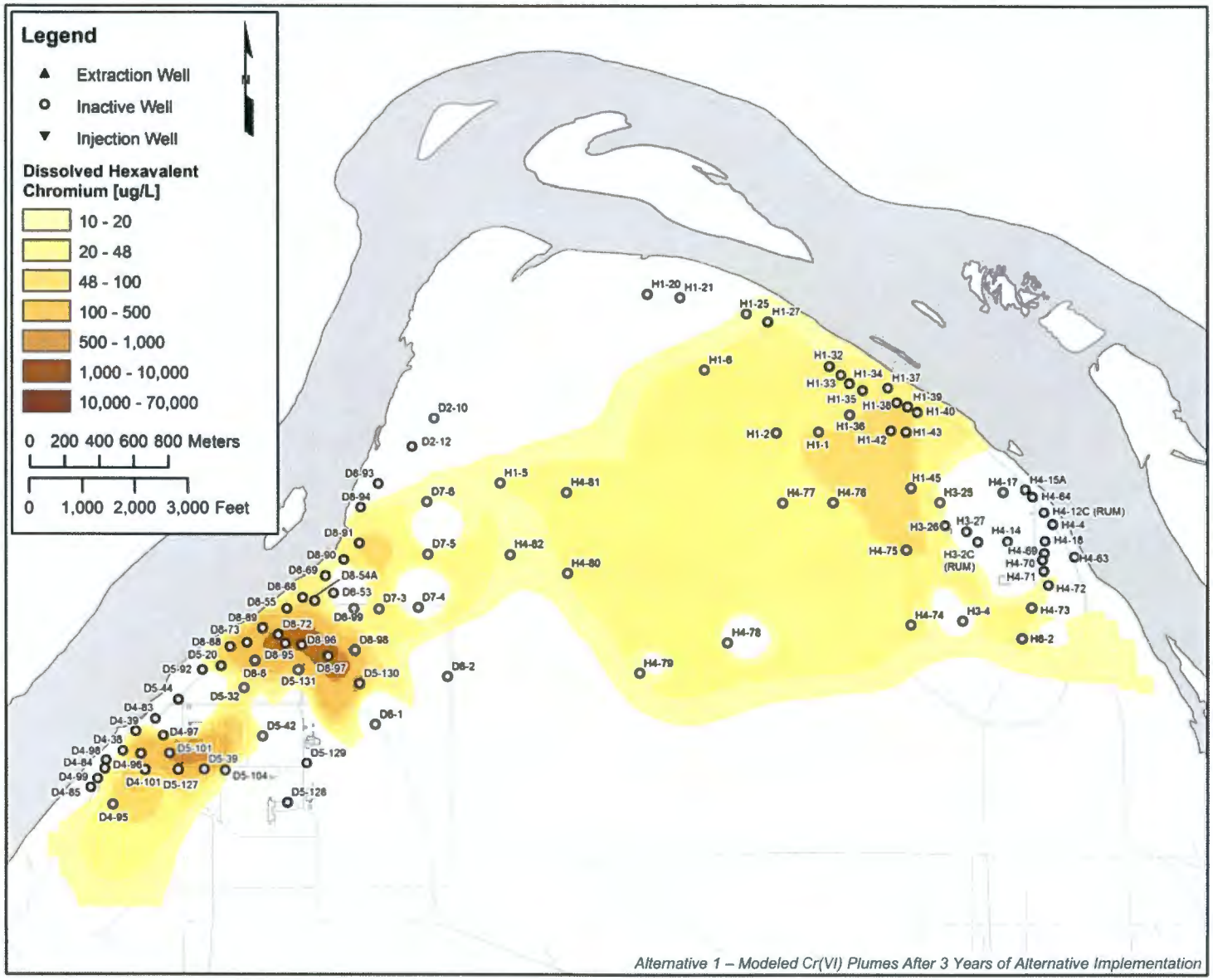

Figure 9-4a. Alternative 1—Modeled Cr(VI) Plumes After 3 Years

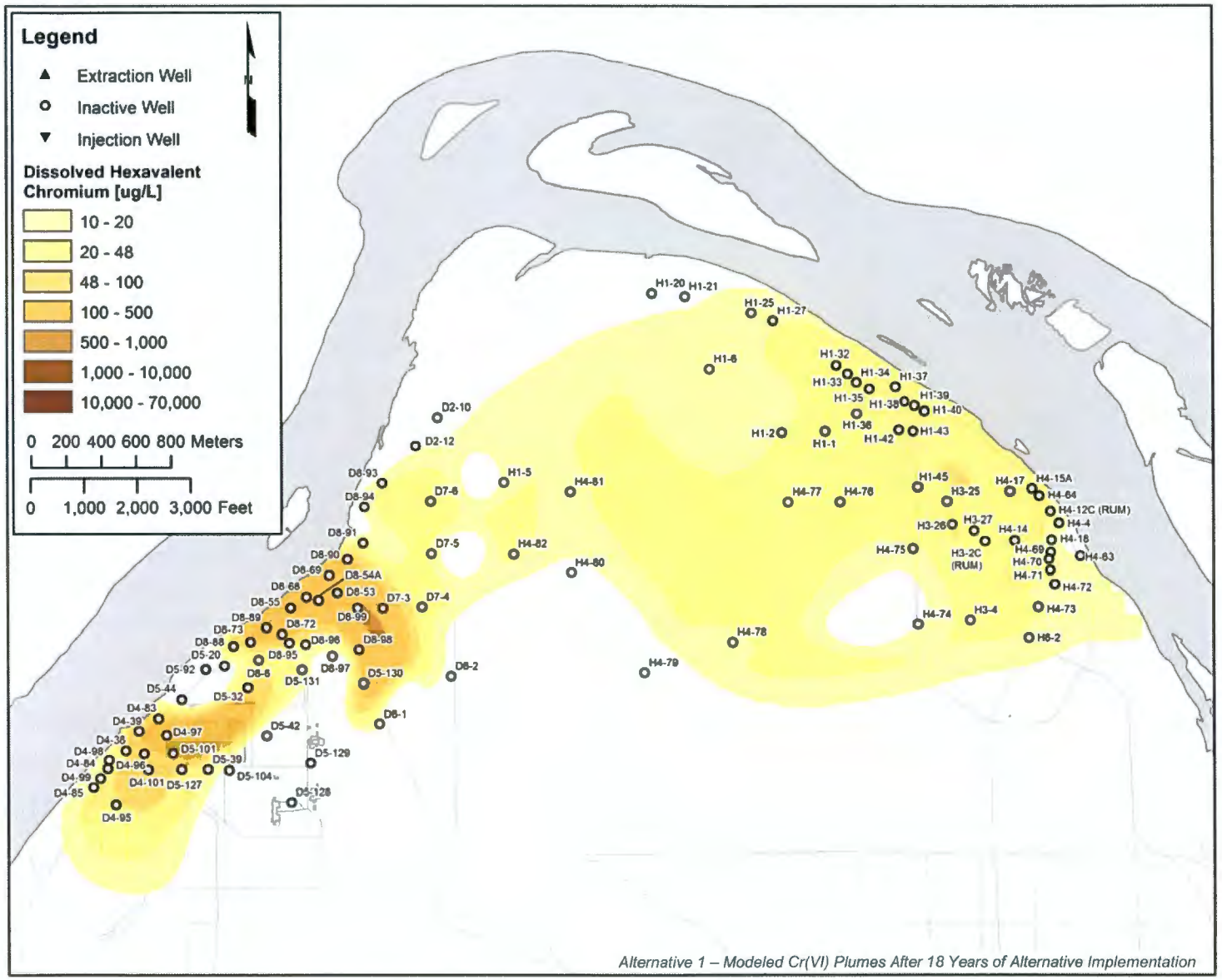

Figure 9-4b. Alternative 1—Modeled Cr(VI) Plumes After 18 Years

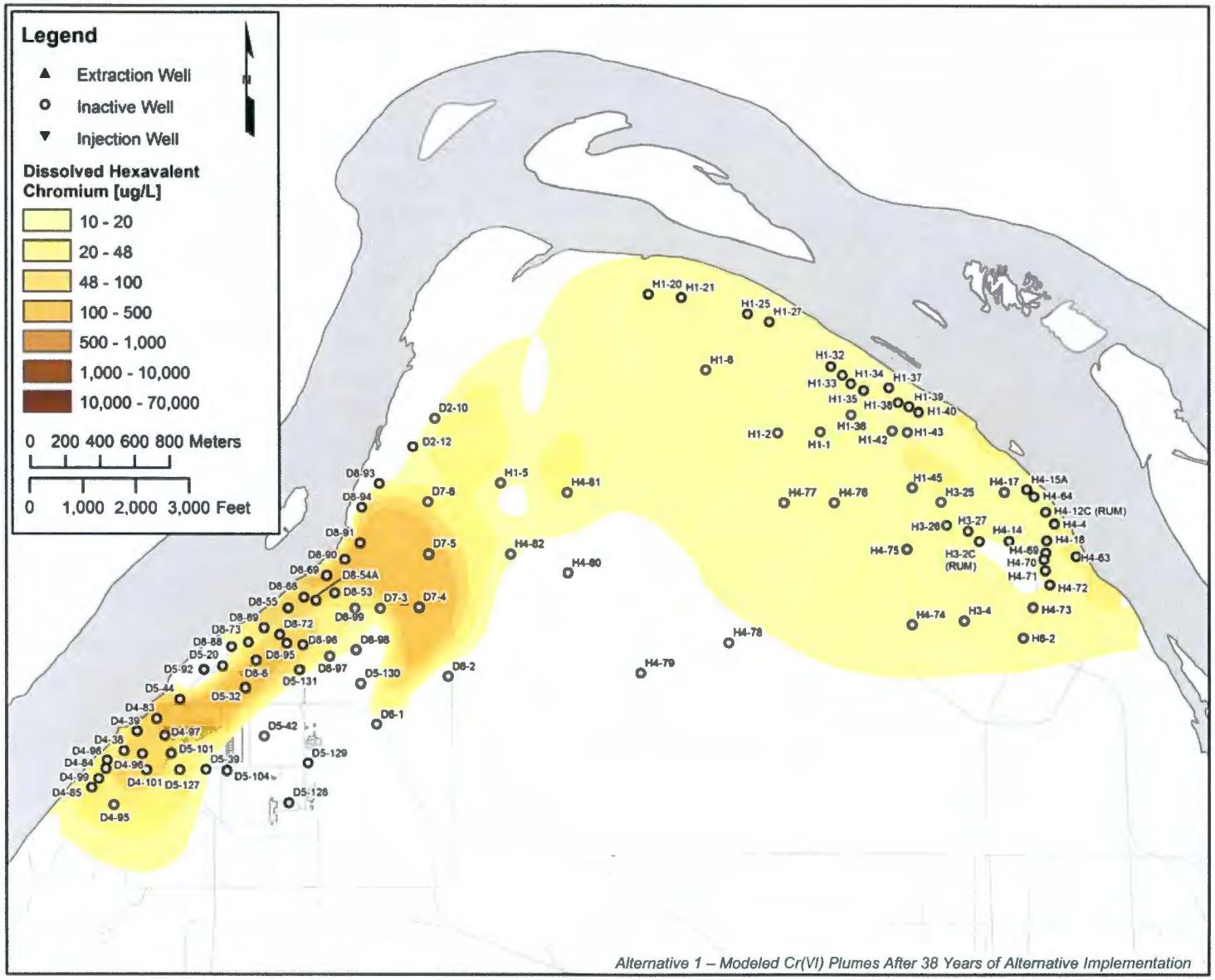

Figure 9-4c. Alternative 1—Modeled Cr(VI) Plumes After 38 Years

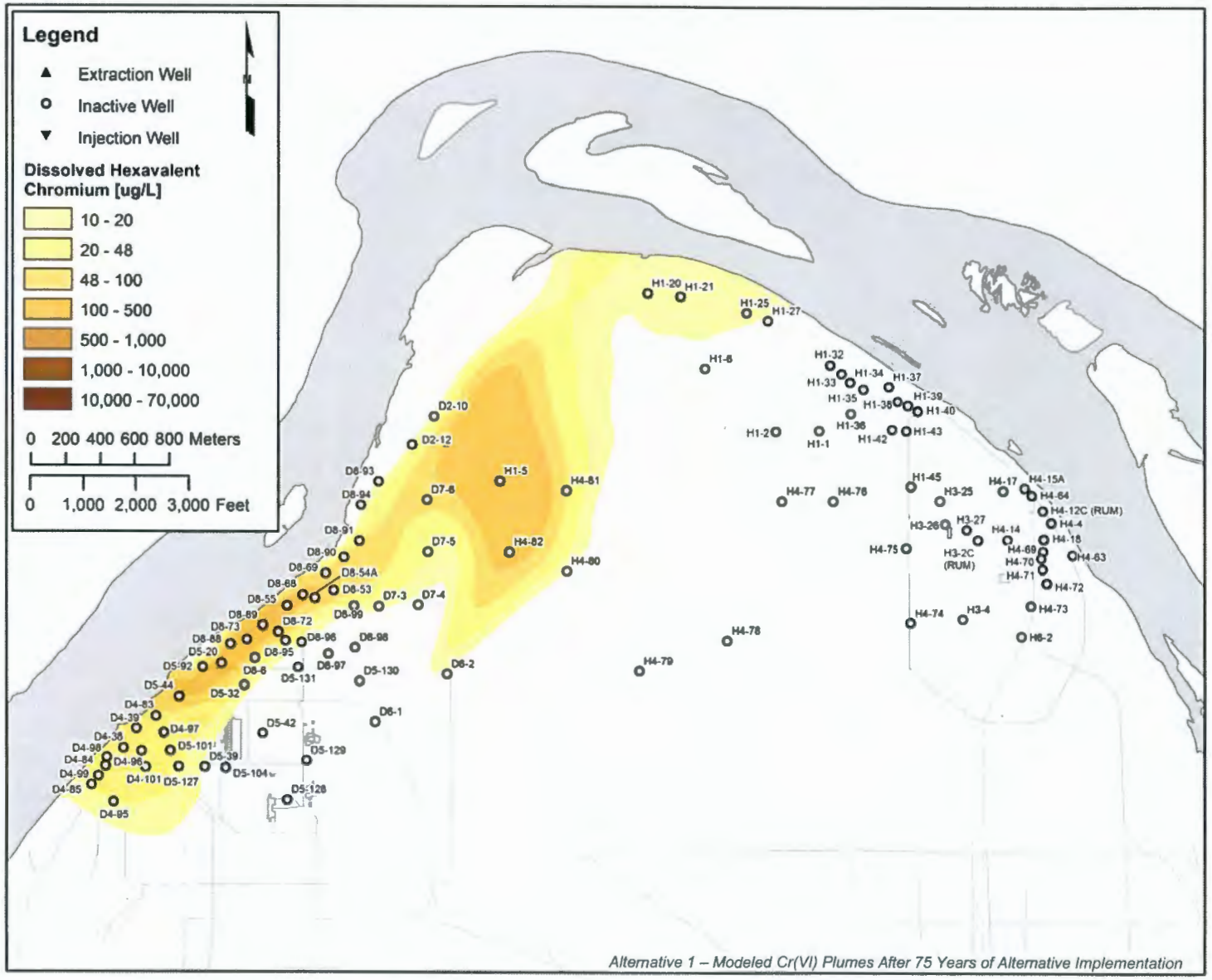

Figure 9-4d. Alternative 1—Modeled Cr(VI) Plumes After 75 Years

Predicted plume depictions for nitrate and strontium-90 COCs are provided on Figures 9-5 (a-d) and 9-6 (a-d).

9.2.2 Common Elements for Alternatives 2, 3, and 4

The remedial action alternatives developed for 100-D/H waste sites and groundwater contain elements that are common to multiple alternatives. To limit redundancy in the discussion, these common elements are described in this section.

9.2.2.1 Institutional Controls

While remediation is underway, institutional controls will be put in place to control access and to prevent exposure to contamination. Institutional controls for 100-D/H are expected to be implemented independently for each waste site or groundwater plume. Institutional controls are defined and discussed in more detail in Section 8.3.1.1 and Table 8-8 (Chapter 8). Institutional controls are currently in place to protect workers and control site access, and they will be continued during the period of remedial actions. Institutional controls that are in place to prevent exposure to contamination will remain in place until the waste site or groundwater plume is remediated.

Post-remediation institutional controls will be put in place to address waste site contamination using excavation and irrigation restrictions, as identified in Table 9-4. Additional waste sites may be added through closure reclassifications.

Programs are in place to control access onto and specific uses of the Hanford Site that, in addition to preservation of the national monument security and safety, also serve to protect human health and the environment by limiting potential exposure to hazardous substances. Many of these multi-purpose or programmatic controls are therefore institutional controls as required by each CERCLA ROD on the Hanford Site. The programmatic controls include site access; personnel badging; real estate and deeds; warning signs along the Columbia River bank and other access points; maintaining a current Sitewide institutional controls plan; and controls for excavating soil, accessing and using groundwater, and restricting irrigation. While these controls transcend any specific CERCLA ROD or even the overall CERCLA cleanup, DOE and EPA recognize the importance of maintaining these controls until unrestricted use is allowable.

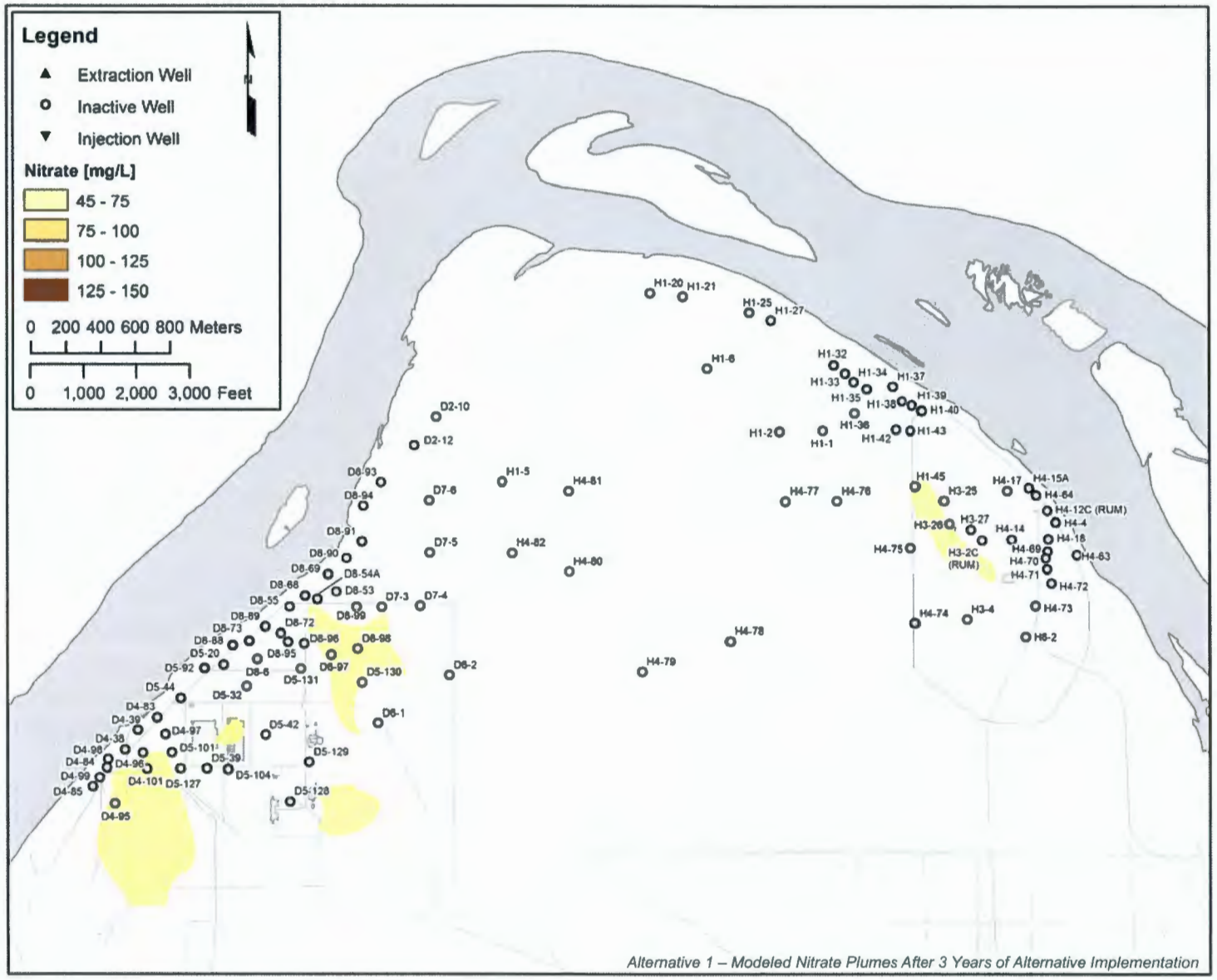

Figure 9-5a. Alternative 1—Modeled Nitrate Plumes After 3 Years

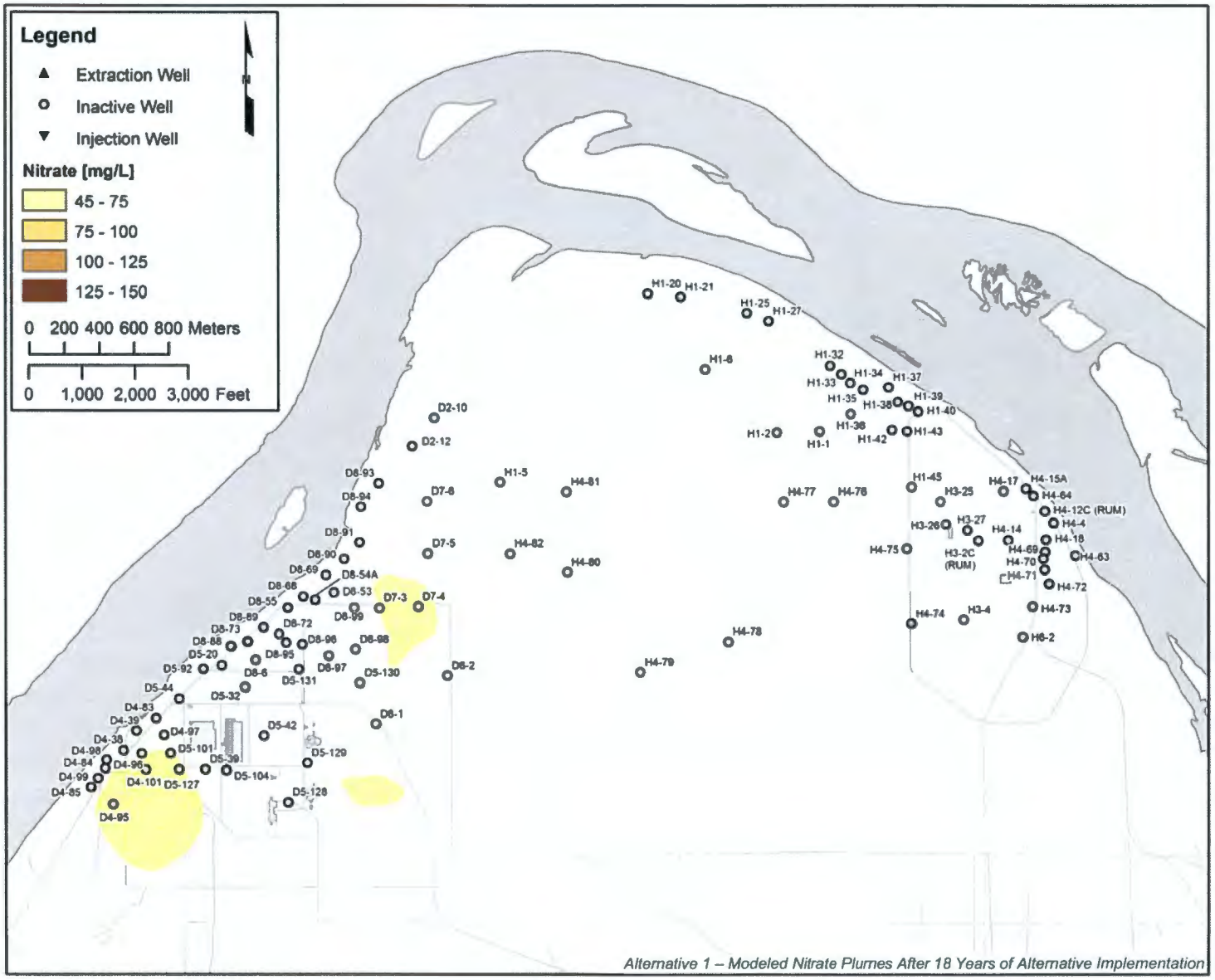

Figure 9-5b. Alternative 1—Modeled Nitrate Plumes After 18 Years

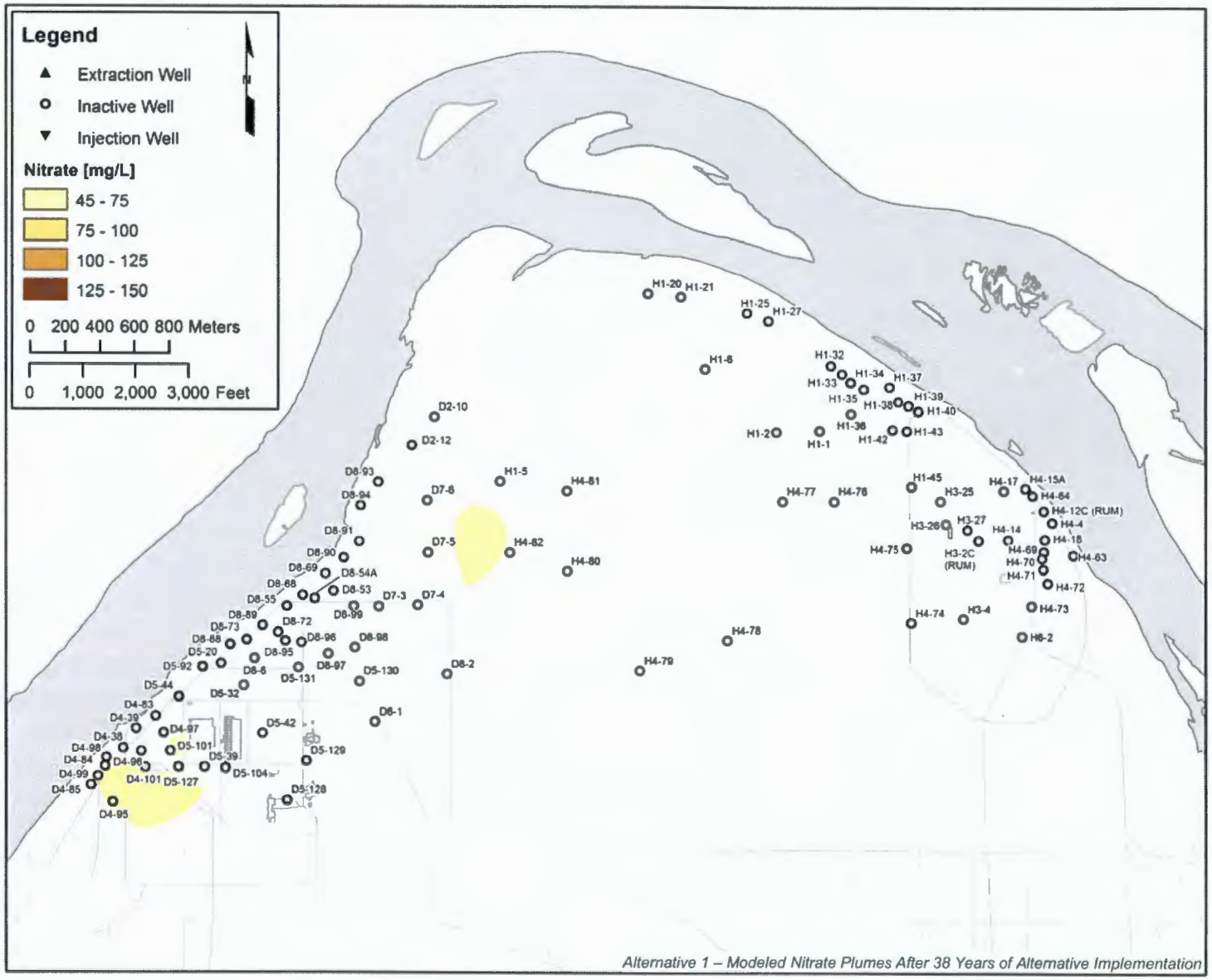

Figure 9-5c. Alternative 1—Modeled Nitrate Plumes After 38 Years

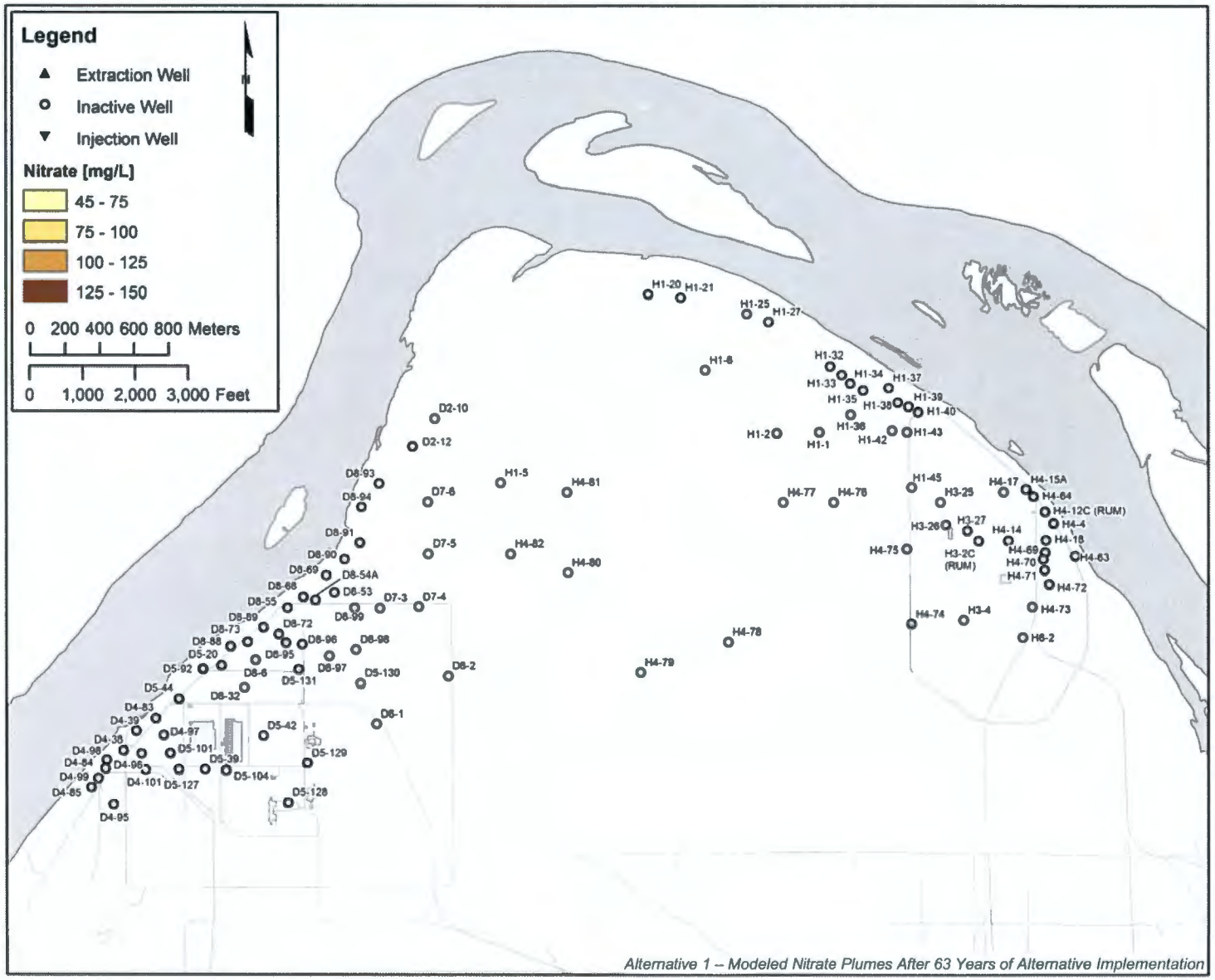

Figure 9-5d. Alternative 1—Modeled Nitrate Plumes After 63 Years

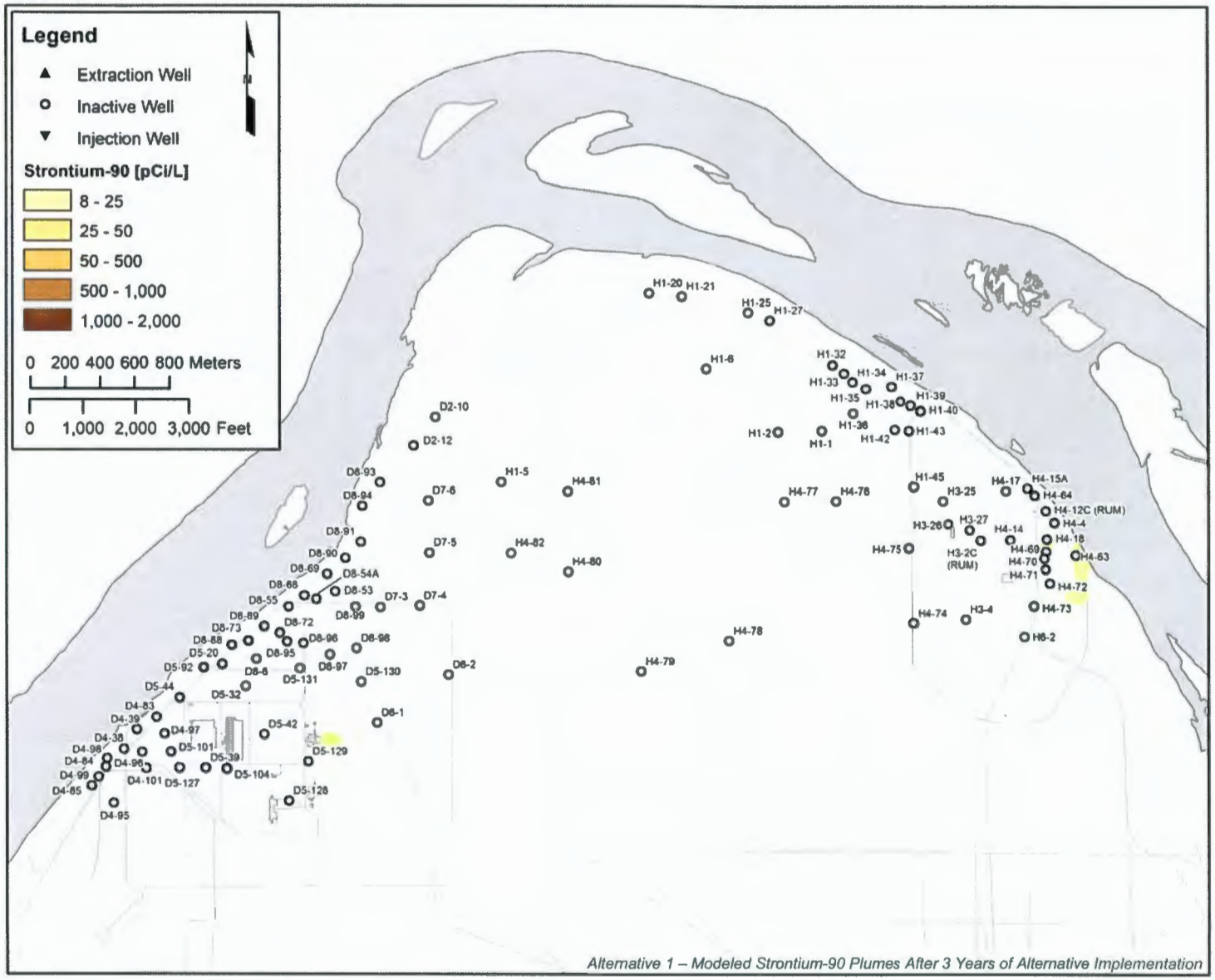

Figure 9-6a. Alternative 1—Modeled Strontium-90 Plume After 3 Years

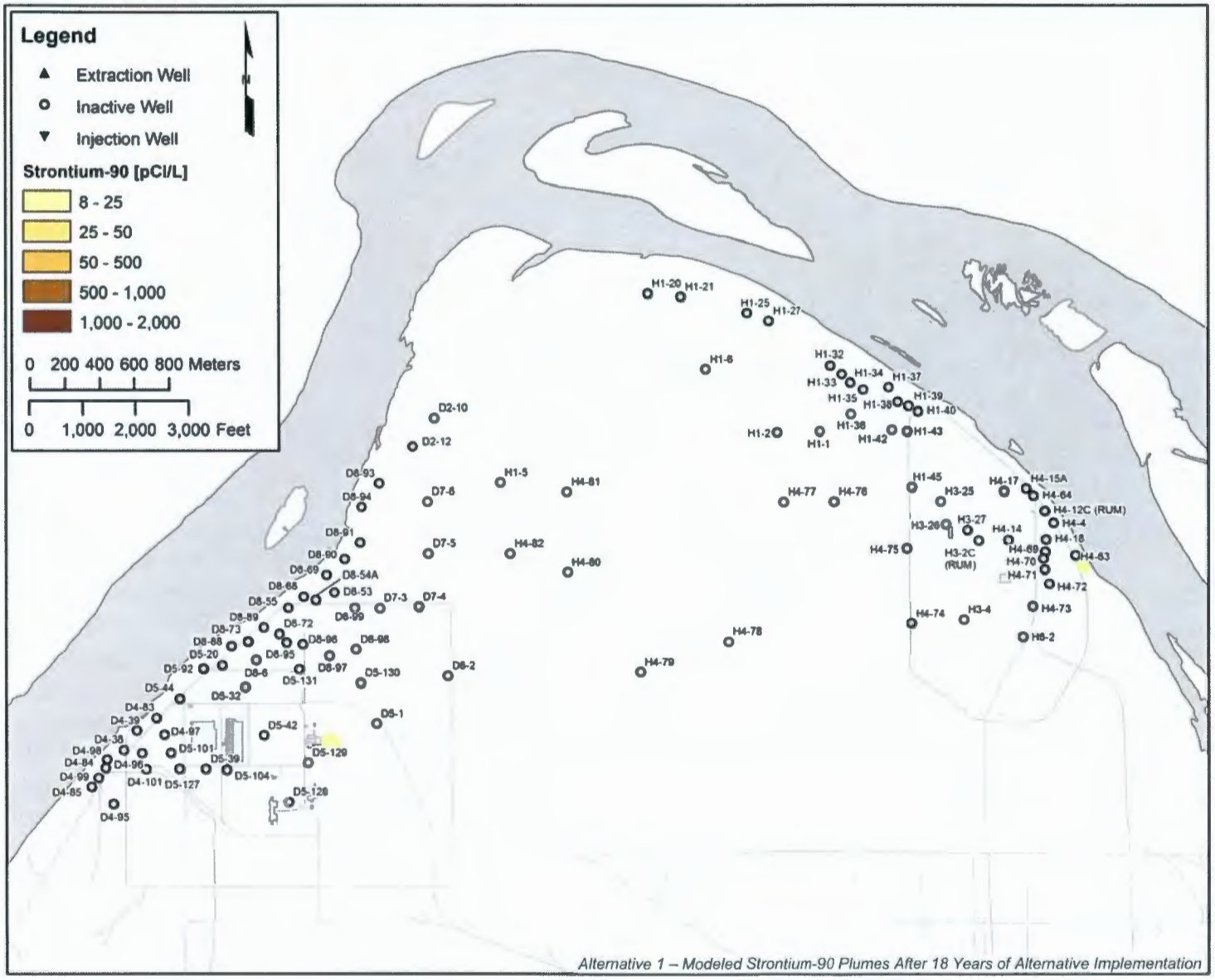

Figure 9-6b. Alternative 1—Modeled Strontium-90 Plume After 18 Years

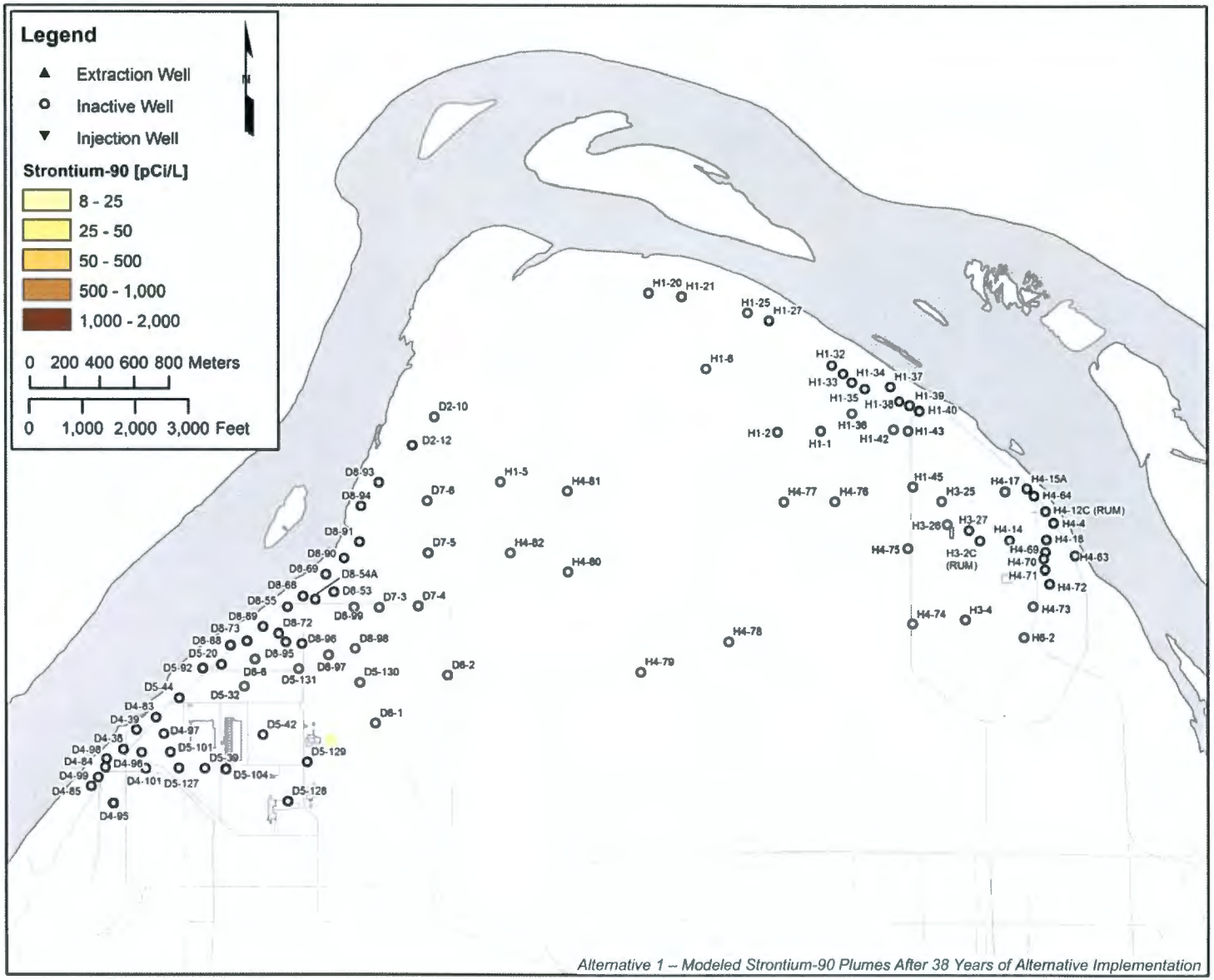

Figure 9-6c. Alternative 1—Modeled Strontium-90 Plume After 38 Years

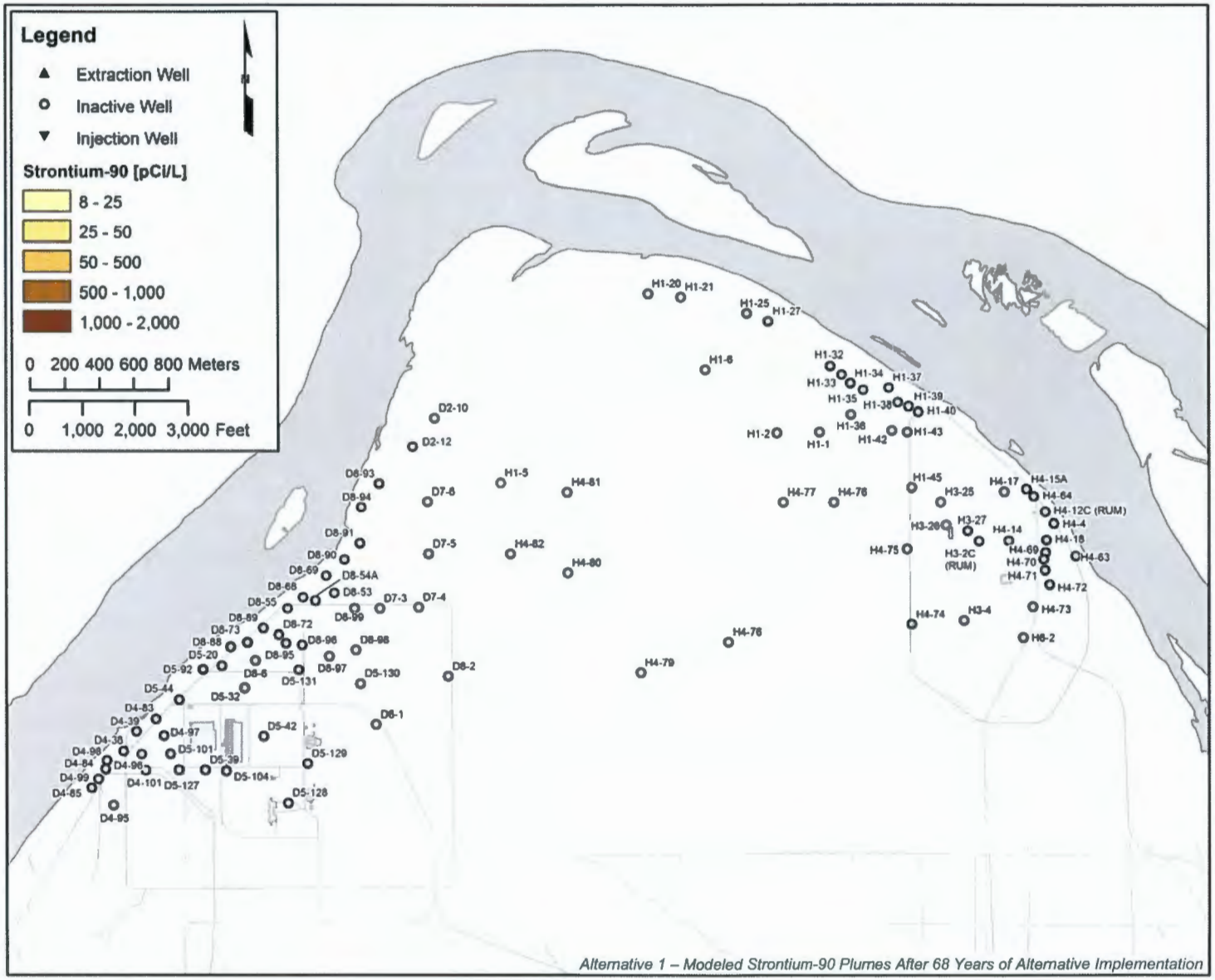

Figure 9-6d. Alternative 1—Model Strontium-90 Plume After 68 Years

Table 9-4. Institutional Controls Implemented at Waste Sites Post Remediation*

Waste Site Risk Driver	Institutional Controls	Alternative 2 Waste Sites	Alternative 3 Waste Sites	Alternative 4 Waste Sites
Waste sites with groundwater/surface water protection risk if irrigation were applied (vadose soil contaminant concentrations exceed SSL but are less than PRG)	Prohibit irrigation	None Identified	None Identified	None Identified
Waste sites with deep (greater than 4.6 m [15 ft] bgs) radiological contamination	Excavation restrictions (up to 185 years)	100-D-18, 100-D-19, 100-D-46, 100-D-48:1, 100-D-48:2, 100-D-48:3, 100-D-49:1, 100-D-49:2, 100-D-49:4, 100-D-5, 100-D-6, 100-H-1, 100-H-11, 100-H-12, 100-H-14, 100-H-21, 100-H-22, 100-H-36, 116-D-1A, 116-D-1B, 116-D-7, 116-DR-1 & 2, 116-DR-6, 116-DR-9, (100-D-25), 116-H-1, 116-H-3, 116-H-7, 118-D-6:3, 118-D-6:4, 118-H-6:3, 118-H-6:6, UPR-100-D-2, UPR-100-D-3, UPR-100-D-4, 100-D-60, 100-H-34	100-D-18, 100-D-19, 100-D-46, 100-D-48:1, 100-D-48:2, 100-D-48:3, 100-D-49:1, 100-D-49:2, 100-D-49:4, 100-D-5, 100-D-6, 100-H-1, 100-H-11, 100-H-12, 100-H-14, 100-H-21, 100-H-22, 100-H-36, 116-D-1A, 116-D-1B, 116-D-7, 116-DR-1 & 2, 116-DR-6, 116-H-1, 116-H-3, 116-H-7, 118-D-6:3, 118-D-6:4, 118-H-6:3, 118-H-6:6, UPR-100-D-2, UPR-100-D-3, UPR-100-D-4, 100-D-60, 100-H-34	100-D-18, 100-D-19, 100-D-46, 100-D-48:1, 100-D-48:2, 100-D-48:3, 100-D-49:1, 100-D-49:2, 100-D-49:4, 100-D-5, 100-D-6, 100-H-1, 100-H-11, 100-H-12, 100-H-14, 100-H-21, 100-H-22, 116-D-1A, 116-D-1B, 116-D-7, 116-DR-1 & 2, 116-DR-6, 116-DR-9, (100-D-25), 116-H-1, 116-H-3, 116-H-7, 118-D-6:3, 118-D-6:4, 118-H-6:3, 118-H-6:6, UPR-100-D-2, UPR-100-D-3, UPR-100-D-4, 100-D-60, 100-H-34
Waste site with contamination contained and left in place	Entry restrictions and Excavation restrictions	100-D-50:2	100-D-50:2	N/A

* Additional waste sites may be added through closure reclassifications.

9.2.2.2 Removal, Treatment, and Disposal

RTD, which can be used to eliminate the presence of contamination in soil, consists of the following actions:

- Collection of confirmatory samples based on the expected and actual risk drivers (media and COPCs). Confirmatory evaluation will determine the need for remediation and confirmation of COCs.
- Demolition of any surface structures, as required.
- Excavation of waste site structures and vadose zone soil where contaminant concentrations are above cleanup levels.
- Determination of the extent of excavation required uses an observational approach. Removal actions use in situ and ex situ sampling, process knowledge, and field measurements to guide day-to-day excavation.
- Excavation using best practices, which includes appropriately sloped sidewalls based on the type of the material being removed, benching, shoring, and proper placement of the stockpiled material according to Occupational Safety and Health Administration standards.
- Sampling and field screening during excavation to ensure that remediation meets the cleanup levels. If contamination above the cleanup levels is encountered beyond the planned limits of excavation, the extent of removal will be increased. The sampling design also identifies contaminant concentrations that pose a risk to groundwater or surface water because this risk contributes to additional uncertainty about the extent of contamination from potential lateral migration (discussed in the CSM, Chapter 4).
- Suppression of dust during excavation to ensure that contaminants are not spread by wind and do not drive mobile contamination toward groundwater.
- Disposal of excavated material (low-level waste) to ERDF as long as the material meets disposal criteria. Hazardous or mixed waste is treated to meet land disposal restrictions before disposal at ERDF or an EPA-approved offsite location.
- Verification sampling following excavation to demonstrate that soil remaining in the excavated area does not exceed the cleanup levels.
- Backfilling and contouring to blend the excavation with the surrounding ground surface and restore and revegetate the site. Sources for backfill material include local borrow pits and the excavated material determined to be clean (verified as clean by meeting cleanup levels). Sites are revegetated with native plant species after backfilling.

Figures 8-7, 8-8, and 8-9 (Chapter 8) illustrate the components of RTD, which has been the basic part of the interim actions performed for the waste sites at the Hanford Site.

9.2.2.3 Monitored Natural Attenuation

A screening level evaluation of potential application of MNA to the identified COCs in groundwater at 100-D/H indicates that MNA can play an important role in overall remediation of groundwater for strontium-90 (radioactive decay and sorption to aquifer materials). The following discussion provides a preliminary analysis of groundwater plume conditions at 100-D/H with respect to the requirements (bolded below) for application of MNA:

1. **The contamination condition does not currently present an actual risk to human or ecological receptors.** The groundwater contaminant plumes are generally well defined for 100-D/H, and current institutional controls (for example, prohibitions against use of groundwater as a source of drinking

water) prevent current exposure to human receptors. Existing groundwater pump-and-treat systems operating at 100-D/H are exerting groundwater capture forces that have reduced the discharge of contaminated groundwater into the Columbia River. This reduction in discharge mitigates exposure to ecological receptors and downstream human receptors.

2. **The source of the observed contamination is no longer contributing to the plume.** Remedial actions are planned, or have already been implemented, at known source areas that have contributed to groundwater COC plumes at 100-D/H. This is particularly important to supporting selection of MNA for groundwater remediation where unremediated source areas are still associated with persistent groundwater plumes. The expected efficacy of source area remedial alternatives at 100-D/H is considered in the overall assessment of MNA for groundwater plume remediation.

3. **The target plume is static or retreating, or existing monitoring data confirm that attenuating processes are present and operating at the site.** The presence and activity of attenuating processes within the affected aquifer system can be demonstrated by either of two methods: (1) monitoring history of the plume indicates that the plume is stable or shrinking, which means that sources are no longer contributing and that attenuating processes are working within the plume, or (2) if the plume is not stable or shrinking, then empirical measurements and observations of aquifer and plume conditions confirm that attenuating processes are operable within the aquifer. Within the 100-HR-3 Groundwater OU, the operating interim groundwater remedial actions have substantially influenced contaminant plumes, reducing the size of some plumes and reducing observed COC concentrations. Historical monitoring for specific contaminants does not indicate static or shrinking plumes at all locations. Observations and measurements of aquifer conditions, however, do indicate that some attenuating processes are at work within the system. Multiple attenuating processes may be effective on any one COC. The following processes are identified as potentially applicable within 100-HR-3 OU:

   - Radioactive decay is confirmed for radioactive COC. Strontium-90 exhibits a sufficiently short radioactive half-life (29.1 years), such that radioactive decay is a major attenuating element of an MNA alternative.

   - Sorption of constituents to the aquifer matrix reduces the relative groundwater concentration of contaminants that interact substantially with the matrix. The tendency of a constituent to sorb, or bind, to the aquifer matrix is generally described by its relative distribution coefficient (K_d). Constituents with higher K_d exhibit a stronger tendency to bind to the aquifer solid matrix and reduce the relative groundwater concentration. Alternatively, constituents that exhibit lower K_d exhibit a reduced tendency to bind to aquifer solids and, therefore, do not exhibit concentration reduction through sorption to the aquifer matrix. Some constituents exhibit no tendency to sorb to aquifer solids, so sorption does not provide any meaningful attenuation for those constituents. Strontium-90 contamination in groundwater at 100-D/H exhibits meaningful attenuation as a result of sorption effects.

4. **Effective monitoring either exists currently or can be implemented.** Groundwater monitoring at 100-D/H is currently sufficient to define COC plumes in groundwater and to evaluate and select remedial technologies. Implementation of an MNA technology at 100-D/H will require evaluation of groundwater monitoring systems to establish specific monitoring points for remediation performance. A site-specific data quality objective and SAP will identify specific data needs (that is, location and timing of monitoring activities) to support implementation of MNA as a remedial technology. Additionally, MNA performance monitoring should include an assessment and response plan that will be implemented to adjust the alternative if the findings are not as expected.

9.2.2.4 Groundwater Pump-and-Treat

Groundwater pump-and-treat systems can be used to contain groundwater plumes through hydraulic containment systems or to remediate the entire groundwater plume through extraction of the mass. A pump-and-treat system consists of an extraction well network, a treatment system, and an injection well network. Figures 8-15, 8-16, and 8-20 (in Chapter 8) illustrate some of the components of a pump-and-treat system.

Four pump-and-treat systems (DR-5, HR-3, DX, and HX) have been implemented within 100-D/H as part of the interim action. Currently, only the DX and HX pump-and-treat systems are in operation, which replaced DR-5 and HR-3, respectively. The objectives of the existing systems are to provide hydraulic containment of the Cr(VI)from reaching the river, and to begin remediation of the entire plume. The DX and HX pump-and-treat systems were installed as a component of the interim actions to meet TPA (Ecology et al., 1989a) Milestone M-016-110-T01 to take actions necessary to contain or remediate Cr(VI) groundwater plumes in each of the 100 Area NPL (40 CFR 300, Appendix B) OUs, such that ambient water quality standards for Cr(VI) are achieved in the hyporheic zone and river water column.

Figure 8-20 illustrates the concept of hydraulic containment through groundwater extraction. Flushing the saturated zone using treated water (Figure 8-18) is another component of pump-and-treat and can be achieved through reinjection of treated groundwater to the 100-D/H aquifer. Strontium-90 and nitrate-contaminated groundwater co-extracted during operation of the pump-and-treat systems may undergo treatment using ion exchange to remove these contaminants if the combined extracted groundwater in the pump-and-treat effluent stream is found to exceed the MCLs.

All the groundwater alternative components presented here, with the exception of the No Action Alternative, build upon these existing systems. The alternative specific enhancements of the system are described under each alternative in Sections 9.2.3, 9.2.4, and 9.2.5. Table 9-5 summarizes the pump-and-treat systems at 100-D/H as part of the interim action.

Table 9-5. Summary of the Existing Groundwater Pump-and-Treat Components

Groundwater Pump-and-Treat Systems	Components include the following: • A combined pump-and-treat capacity of 5,300 L/min (1,400 gal/min): – DX – 2,271 L/min (600 gal/min); replaced 189 L/min (50 gal/min) DR-5 pump-and-treat system – HX – 3,030 L/min (800 gal/min) ; replaced 1,136 L/min (300 gal/min) HR-3 pump-and-treat system • 82 extraction wells • 35 injection wells • Wellhead infrastructure • Cr(VI) treated by ion exchange • Water treated to achieve requirements before injecting into 100-D/H aquifer

Note: Pump-and-treat is being carried out under the *Record of Decision for the 100-HR-3 and 100-KR-4 Operable Units Interim Remedial Actions, Hanford Site, Benton County, Washington* (EPA/ROD/R10-96/134).

The pump-and-treat component of the remedial alternatives are designed to be robust and to provide for expansion to address the uncertainties identified in the CSM described in Section 4.9 associated with locations where groundwater monitoring indicates potential residual contamination in the vadose zone

that may contribute to groundwater contamination. RPO combined with remedy performance monitoring provides for robust pump-and-treat system performance. RPO will be applied to groundwater remedial components to monitor and evaluate remedy performance throughout the duration of the remedial action to assess and implement changes to the pump-and-treat systems to optimize system performance as cleanup progresses. Remedy performance monitoring, described in Section 9.2.2.7, will evaluate effectiveness of pump-and-treat at the locations with uncertainty for potential residual contamination in the vadose zone.

As discussed in Section 4.9.2.1 (Chapter 4), Cr(VI) contamination is present at concentrations above 10 µg/L in the first water-bearing unit of the RUM at 100-H. Two extraction wells, 199-H3-2C and 199-H4-12C, are screened and extract groundwater from the RUM water-bearing unit for treatment through the HX pump-and-treat system. A groundwater analysis conducted to evaluate the effectiveness of the two extraction wells to capture and remediate Cr(VI) contamination within the RUM water-bearing unit at 100-H is included in Appendix F (*Evaluation of Potential Hydraulic Capture and Plume Recovery from the Ringold Upper Mud (RUM) in the 100-HR-3 Operable Unit (OU)* [ECF-100HR3-12-0025]). The analysis presented in the ECF also evaluated use of additional extraction wells to capture the contamination plume in the RUM.

Sensitivity analysis was performed to evaluate bounding cases for flow through the aquifer, leakage from the overlying unit, and flow from the Columbia River. The evaluation indicated that in the case of high leakage from the overlying aquifer unit, pumping from the two existing wells may not be sufficient to provide capture of the plume in the remediation period. The addition of two extraction wells indicates that pump-and-treat is feasible for recovering contamination identified within the RUM, for the remedial action alternatives. Calculations suggest that the parameters of the confining unit and the degree of connection with the Columbia River play an important role in the effectiveness of pumping from the silty-sand RUM unit for contaminant recovery. For purposes of this FS, the four extraction well configuration is included in the remedial alternatives to remediate Cr(VI) contamination in the RUM water-bearing unit. Further calculations to provide information necessary for the design or scaling for groundwater extraction from the RUM water-bearing unit will be conducted as part of the remedial design phase.

The interim action groundwater remedies include the ISRM barrier (described in Chapter 1). This system will continue to operate as is (without enhancement), but it is not critical to the achievement of the plume cleanup. The ISRM barrier is a passive system, so there is no cost for its continued operation. Figure 8-19 illustrates the ISRM barrier.

9.2.2.5 Ion Exchange

Treatment of Cr(VI) contaminated groundwater use of ion exchange technology. The ion exchange process removes ions from the aqueous phase by the exchange of cations or anions between the contaminants and the exchange medium. The ion exchange materials may consist of resins made from synthetic organic materials that contain ionic functional groups to which exchangeable ions are attached. The materials may also be inorganic or natural polymeric. After the resin capacity has been exhausted, resins can be regenerated for reuse or disposed of at ERDF or other EPA-approved disposal facility.

Figure 8-16 illustrates the basic components of an ion exchange system.

9.2.2.6 Operations and Maintenance

O&M of each remedial alternative (except the No Action Alternative) is required to ensure that the remedy is operated and maintained in a manner that ensures long-term effectiveness and permanence. O&M requirements of the selected remedy will be described in an O&M plan, which details performance

monitoring needs, post-closure monitoring requirements, monitoring methods, analytes and intervals, maintenance activities and frequencies, and associated procedures.

The nature and scope of O&M activities vary by alternative component. For example, O&M activities for an MNA component primarily include inspection, maintenance, and periodic replacement of monitoring wells, whereas groundwater pump-and-treat components include routine and preventive maintenance programs and replacement of pump-and-treat system parts at the end of their design life (typically 15 years). Alternatives with longer durations include multiple replacements of system parts every 15 years.

The scope and cost of O&M activities are included in each of the remedy components.

9.2.2.7 Remedy Performance Monitoring

Remedy performance monitoring will be conducted to evaluate the effectiveness of the selected alternative to attain the cleanup levels that will be identified in the 100-D/H decision document. The nature and scope of the performance monitoring program will vary by alternative component, and will be developed during the remedial design process and included in a performance monitoring plan. Remedy performance monitoring applies to MNA actions as well as actively engineered remedies.

A groundwater monitoring plan has been developed under the interim actions via the 100 Area RD/RAWP (DOE/RL-96-17) and *Interim Action Monitoring Plan for the 100-HR-3 and 100-KR-4 Operable Units* (DOE/RL-96-90). The number of wells monitored and the frequency of remedy performance monitoring is anticipated to vary, depending on the phase of remediation. A geostatistical analysis will be conducted to determine the optimum spatial distribution for the performance monitoring network. For alternatives where active remediation is occurring, the frequency of monitoring is assumed to be quarterly, semiannually, or annually.

Sampling and analysis will also be conducted for analytes identified to have uncertain status (Section 8.1.1.2). To ensure protectiveness and confirm current understanding of the nature and extent of contamination and potential risks, these analytes will be analyzed as part of the performance monitoring. The CERCLA 5-year reviews will allow an evaluation of the approach taken with these compounds.

Identification of well locations, including identification of new wells for remedy performance monitoring, will be defined in the SAP and developed as part of the remedial design. The following assumptions were made for this FS:

- Monitoring is conducted to evaluate the effectiveness of the selected alternative to attain the cleanup levels (MNA actions are described in Section 9.2.2.3, and actively engineered remedies are described in Section 9.2).
- Nature and scope are specific to alternative components and would be developed during the remedial design process.
- Hydraulic and chemical monitoring of the monitoring wells, including extraction wells, is designed to evaluate contaminant mass removal and containment.
- Frequency is semiannual to annual for active remediation alternatives.
- Frequency is semiannual to annual for active remediation alternatives at selected locations to evaluate whether or not the analytes identified in Section 8.1.1.2 have uncertain status.

The strategy for implementation of the technologies for each alternative is presented in the following sections.

9.2.3 Alternative 2—RTD and Grouting for Waste Site and Pump-and-Treat with Biological Treatment for Groundwater

Table 9-6 presents the waste site and groundwater components of this alternative, and Figure 9-7 presents a pictorial summary of the alternative. The cost for this alternative is provided in Table 9-7. Details for the cost estimate are provided in Appendix J (*Environmental Cost Estimate for 100-D/H Vadose Zone and Groundwater RI/FS* [ECE-100HR311-00004]). Additional details on remedial components specific to this alternative are presented in sections identified in Table 9-6.

Table 9-6. Components for Alternative 2—RTD and Grouting for Waste Site and Pump-and-Treat with Biological Treatment for Groundwater

Waste Site Components	General Components	
	No Further Action	No additional remedial actions are taken for the 146 waste sites identified in Section 8.2.1.1 (Chapter 8).
	Institutional Controls	Institutional controls to be implemented within the 100-D, 100-H, and horn areas for land use management and waste site information management include the following: • Permits required for excavations on the Hanford Site to prevent unplanned disturbance or infiltration as prohibited by CERCLA decision documents. • Land use and real property controls (for example, proprietary controls including easements and covenants) ensure that the use of land is in accordance with Hanford Site plans and CERCLA decision documents. • Notices providing visual identification and warning of hazardous or sensitive areas. • Procedural requirements for access, warning signs, or fencing implemented to prevent or limit the access of humans to hazardous or sensitive areas. • Administrative mechanisms, such as the WIDS database, to maintain and provide access to information on the location and nature of contamination. Additional information on institutional controls is presented in Section 8.3.1 (Chapter 8). Post remediation institutional controls implemented at specific waste sites are identified in Table 9-4.

Table 9-6. Components for Alternative 2—RTD and Grouting for Waste Site and Pump-and-Treat with Biological Treatment for Groundwater

Waste Site Components (cont.)	For Waste Sites That Exceed Human Health, Environment, Surface Water, or Groundwater Protection PRGs	
	MNA/institutional control	MNA/institutional controls applied to waste sites 116-D-8, 116-DR-9, and 100-D-25. Waste site 100-D-25 is fully encompassed within waste site 116-DR-9. Risk evaluation of the verification data indicates that the shallow decision units for these sites contain cesium-137 at concentrations greater than the residential RBSL (Chapter 6). No ecological risks are identified for the sites (Chapter 7). Cesium-137 concentrations at the three sites will decay through natural attenuation to levels less than residential RBSLs in year 2035. Institutional controls will be in place through this period. Additional information on institutional controls is presented in Section 8.3.1 (Chapter 8).
	Void-fill Grouting	Void-fill grouting, as described in Section 9.2.3.1, will be used to remediate the box flumes of the 100-H-36 waste site to immobilize contaminants on interior surfaces of the underground concrete box flume sluiceway.
	RTD	RTD to depth of contamination exceeding cleanup levels, or until groundwater is encountered if contamination extends into groundwater, using standard and deep excavation technologies, as described in Section 9.2.2.2. RTD is applied to waste sites listed in Table 8-5 (Chapter 8) that are expected to be remediated under the 100 Area Remaining Sites ROD (EPA/ROD/R10-99/039). RTD is effective in meeting the human health, environment, surface water, and groundwater PRGs as reflected in the risk evaluation of completed waste sites with close out/verification data (Chapter 6). RTD also applied to waste sites listed in Table 8-5 (Chapter 8) that are yet to be remediated and are not remediated under one of the other waste site remedial components listed above. Table 9-2 identifies applicable waste sites remediated by RTD using standard and deep excavation technologies. Within the SMDP process, ecological PRGs will be considered at a population level for wildlife and at a community level for plants and invertebrates, to determine whether cleanup action is required to protect ecological receptors.

Table 9-6. Components for Alternative 2—RTD and Grouting for Waste Site and Pump-and-Treat with Biological Treatment for Groundwater

Groundwater Components	Pump-and-Treat System	Operation of existing pump-and-treat systems as described in Section 9.2.2.4, with additional extraction and injection wells to expand treatment coverage. Placement of additional extraction and injection wells, extraction and injection flow rates, and well operational periods were determined through groundwater model simulations described in *Modeling of RI/FS Design Alternatives for 100-HR-3* (ECF-100HR3-11-0114). The RPO process will be used to provide ongoing evaluations to ensure the system meets performance requirements that will be defined in the RD/RAWP. Designed with active remediation out to Cr(VI) 10 µg/L contour. Treatment of extracted water through ion exchange. Operation of pump-and-treat system and bioinjection components to meet AWQC at points of groundwater discharge to the river
	Bioinjection	Groundwater pump-and-treat system optimized with bioinjection for in situ treatment of Cr(VI) groundwater contamination. Components of bioinjection, described in Section 9.2.3.3, include the following: • Injection of biological substrate (for example, cheese whey or sodium lactate) into bioinjection wells with closed-loop recirculation from downgradient wells • Bionode mixing facilities • Biosubstrate mixing plants Placement of wells for bioinjection, flow rates, and well operational periods were determined through groundwater model simulations described in *Modeling of RI/FS Design Alternatives for 100-HR-3* (ECF-100HR3-11-0114).

Table 9-6. Components for Alternative 2—RTD and Grouting for Waste Site and Pump-and-Treat with Biological Treatment for Groundwater

Groundwater Components (cont.)	MNA and Institutional Controls	MNA as described in Section 9.2.2.3. Manage strontium-90 in groundwater through MNA and institutional controls until concentrations meet cleanup standards. Manage the plumes until concentrations are below the cleanup levels and after the pump-and-treat system is shut off. Monitoring of COCs to track the attenuation processes. Institutional controls to be implemented within 100-D/H include land use management and waste site information management. Additional institutional controls for groundwater include the following: • Permits required for excavations on the Hanford Site to prevent unplanned disturbance or infiltration as prohibited by CERCLA decision documents. • Land use and real property controls (for example, proprietary controls including irrigation restrictions, easements, and covenants) to ensure that the use of land is in accordance with Hanford Site plans and CERCLA decision documents. • Groundwater use management, as described in *Sitewide Institutional Controls Plan for Hanford CERCLA Response Actions* (DOE/RL-2001-41), to ensure proper use of groundwater through groundwater controls. • Administrative mechanisms, such as the WIDS database, to maintain and provide access to information on the location and nature of contamination. Additional information on institutional controls is presented in Section 8.3.1 (Chapter 8).
	Monitoring Requirements	Remedy performance monitoring will be defined as part of the remedial design. Monitoring evaluates the effectiveness of the selected alternative to attain the cleanup levels. The monitoring program will expand by adding a number of specific monitoring wells. For cost estimating purposes, it has been assumed that 12 new monitoring wells will be installed. Groundwater performance monitoring is presented in Section 9.2.2.7 and identifies constituents included in the monitoring program.

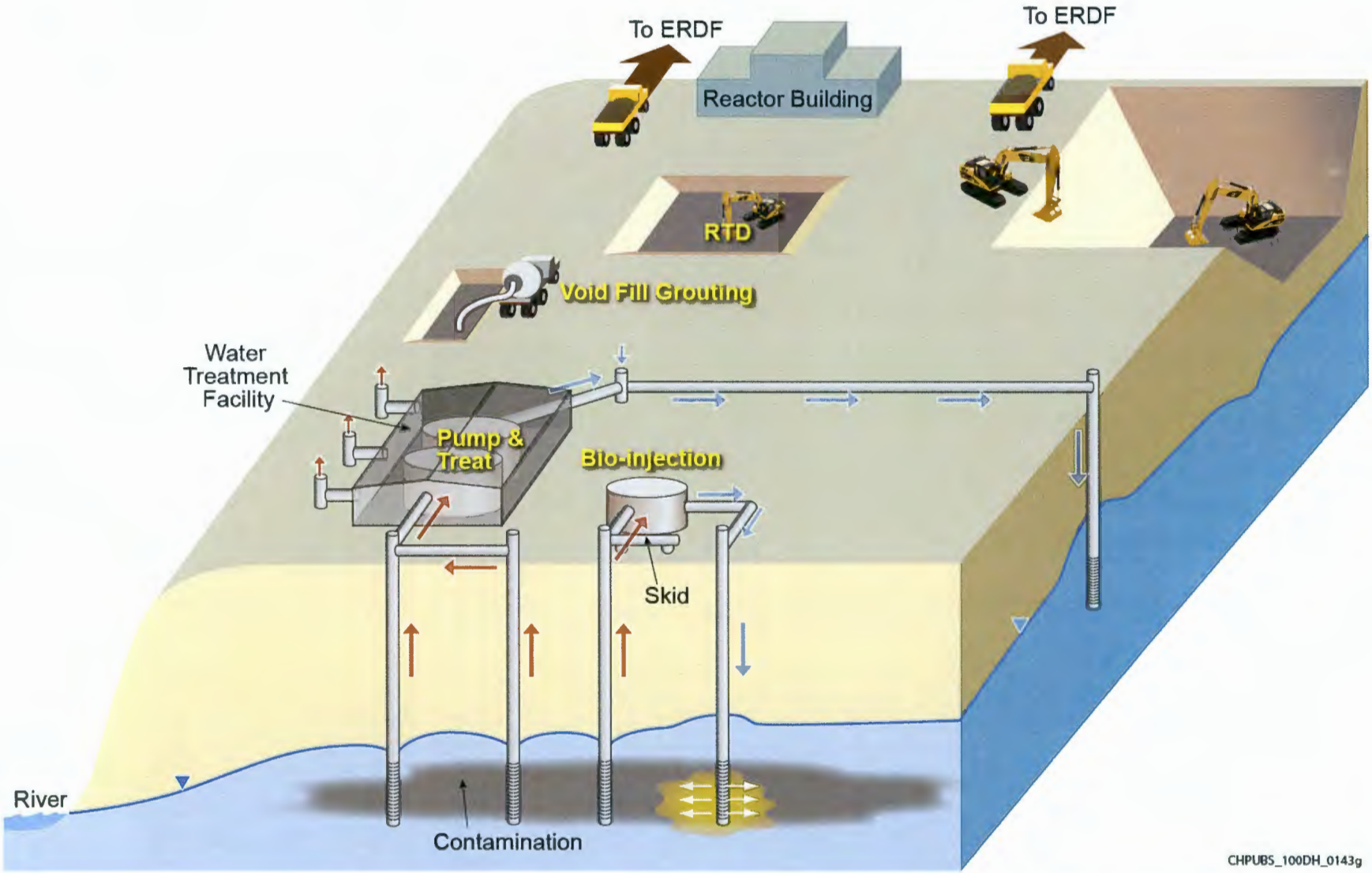

Figure 9-7. Alternative 2—RTD and Grouting for Waste Site and Pump-and-Treat with Biological Treatment for Groundwater

Table 9-7. Alternative 2—Cost Estimates (in millions)[a]

Waste Site Treatment	Capital	$132.4
	Annual O&M	$25.7
	Total Periodic[b]	$0.8
	Total Nondiscounted	$158.9
	Net Present Value	$142.9
Groundwater Treatment	Capital	$20.2
	Total O&M	$361.7
	Total Periodic[b]	$111.7
	Total Nondiscounted	$493.5
	Net Present Value	$363.1
Total Nondiscounted Cost of Alternative	$652.5	
Total Net Present Value (Discounted) of Alternative	$506	

a. Order of magnitude cost estimates and support information are presented in Appendix J. Costs for institutional controls are included under the costs for waste site treatment.

b. Periodic costs include O&M and/or construction activities, including costs to replace an installed remedy or components of an installed remedy, and services that are not included in initial capital costs or annual O&M costs. Periodic costs may be one-time costs or costs that recur at intervals over the life of the remedy.

This alternative optimizes the operation of the interim action pump-and-treat by including bioinjection for treatment of contaminated groundwater. Bioinjection will be used for specific well pairs or clusters in a closed-loop fashion separate from the ion exchange treatment plants. This reduces the likelihood of biodegradation byproducts reaching the ion exchange treatment systems. Bioinjection is implemented to augment groundwater remediation in the horn area to reduce the amount of groundwater that must be sent to the ion exchange plants, thereby increasing the total capacity of the system and reducing the O&M cost. Remediation components described in Table 9-6 that are specific to Alternative 2 are described in the following sections.

9.2.3.1 Void-fill Grouting

Void-fill grouting is an in situ treatment technology to immobilize contaminants by solidification of wastes. Void-fill grouting can be used to fill large empty spaces (for example, pits and trenches) where the structure would then be left in place. Specifically, waste site 100-H-36 will be remediated by void-fill grouting under this alternative. This waste site is an underground concrete box flume sluiceway that led from the 116-H-5 Outfall Structure to the river shoreline. The site is on the Columbia River shoreline, and void-fill grouting of the box flume presents fewer ecological impacts along the shoreline than remediation through RTD. The three side-by-side channels of box flumes, each with an approximately 1.2 × 2.1 m (4 × 7 ft) interior with 39.6 m (130 ft) of remaining length, will be grout-filled. The spillway run-off pad extending from the box flume to the river will be removed to the ordinary high water mark by RTD.

When void-fill grouting is done, thermal expansion effects of the interior grout fill on the surrounding structure need to be considered. For large areas, grout pours would be performed in intervals of small area and height, with a considerable time lapse between adjacent pours. This will allow the hardened grout to cool between pours, and the expansion effects should be easy to account for in the grout design. There are several ways to control expansion: proper mix design, fast- or slow-setting cement, or admixtures. The grout mixture would be designed with control of thermal expansion as one of the requirements, using grouting sequence and schedule requirements of the final design.

The void structure to be grout-filled would be prepared to provide access for grout pour and the controlled flow of grout. Flowable grout would be delivered into the structure to fill void spaces. Pressure grouting may also be used to fill voids not reached by previous grouting. Grout amendments, such as fly ash or zeolite clays, may be considered for grouting activities to reduce the potential for leaching of radioactive isotopes. Design components for void-fill grouting included in this FS are as follows:

- Design samples based on the expected and actual risk drivers (media and COPCs)
- Grout mix design and delivery system
- Structure preparation that provides access points for grout pour and controlled grout flow
- Grout pour in lifts to account for thermal expansion effects

Verification sampling is performed to demonstrate that grout matrix has been effective and remaining structure achieves RAOs.

Figure 8-13 illustrates the components of void-fill grouting.

9.2.3.2 Bioinjection

Bioinjection is included in Alternative 2 to enhance the pump-and-treat systems for Cr(VI) remediation. Bioinjection is the process of adding an organic substrate to water injected into the saturated zone to induce growth or activity of indigenous bacteria for reducing chromate. The injected water will be groundwater extracted from a downgradient well to create a closed-loop injection-extraction system. This will be done, in part, to keep groundwater affected with the organic substrate from reaching an ion exchange treatment plant and potentially fouling the ion exchange resin. Additionally, this approach will biologically reduce Cr(VI) in the extracted groundwater, which will reduce the flow rate to the ion exchange treatment plants, thereby reducing the O&M costs. Figure 8-17 illustrates bioinjection.

The extraction-injection well sets for bioinjection include the following design components:

- Organic substrates include soluble (miscible) substrates (for example, lactate and cheese whey) and immiscible substrates (for example, emulsified vegetable oil).
- Organic substrate will be injected at upgradient wells in pulsed operation to reduce well fouling.
- Groundwater from extraction wells will be used as the source water for the bioinjection and piped directly to the reagent mixing facility.
- After the source groundwater is mixed with a carbon source at the reagent mixing facility, a pipeline will carry fluid from the reagent mixing facility to the injection wells.

System components include the following:

- Reagent mixing facility, pipelines, injection wells, pumps, and valves
- Stand-alone injection wells

- Injection well components to allow the well to operate efficiently without aeration of the injection water:
  - A packer located 3 m (10 ft) from the top of casing to prevent injection well overtopping
  - A pressure transducer to measure pressure on the packer
  - A drop pipe and foot valve at the bottom of the drop pipe to maintain a standing column of water in the drop pipe
  - A water level indicator/transducer to monitor water levels in the injection wells
  - A cleanout for the pipeline from the mainline to the injection well (a tee in the line where cleaning tools can be inserted)
  - A sampling port that would allow sampling of the injection water
  - A pipeline to each injection well, installed on the ground surface (proposed injection rates ranging from 57 to 160 L/min [15 to 43 gal/min] for each well)
- A preventive well cleaning program to mitigate well fouling, which includes the following:
  - Flushing the wells with clean water on a regular basis
  - Physically cleaning the wells with appropriate cleaning solutions (for example, acid or bleach)

The specific extraction-injection well layouts for Alternative 2 are presented in *Modeling of RI/FS Design Alternatives for 100-HR-3* (ECF-100HR3-11-0114) and summarized in Section 9.2.3.3. It is likely that a pulsed/intermittent operation will be used to optimize the performance of the bioinjection system. The groundwater flow and transport model (*Modeling of RI/FS Design Alternatives for 100-HR-3* [ECF-100HR3-11-0114]) was used to simulate bioremediation of Cr(VI) through bioinjection of biological substrate reagents and identify locations and well spacing for bioinjection. Bioinjection is simulated in the groundwater flow and transport model (*Modeling of RI/FS Design Alternatives for 100-HR-3* [ECF-100HR3-11-0114]) as a first order decay term applied to the substrate to approximate the consumption of the substrate over time. Appendix I provides additional information on bioremediation as a potential remedial action for Cr(VI) in the groundwater and vadose zone of the 100 Area.

9.2.3.3 Groundwater Model for Alternative 2

This section provides a summary of the groundwater model for Alternative 2. The interim action pump-and-treat system is expanded in Alternative 2 to include additional extraction and injection wells, thus encompassing a larger area of the COC plumes to expedite hydraulic containment and recovery. In addition, in situ treatment is considered in the form of bioinjection at selected wells and periods to further enhance the reduction of dissolved COC concentrations in the aquifer and shorten cleanup times. This alternative expands the existing pump-and-treat system with 15 new or converted wells for extraction and injection associated with the DX pump-and-treat system, and 29 new or converted wells for extraction and injection at 100-H for the HX pump-and-treat system and bioinjection. Duration of well operation is detailed in *Modeling of RI/FS Design Alternatives for 100-HR-3* (ECF-100HR3-11-0114, Table 3-4).

The groundwater model assumes 85 percent availability of the pump-and-treat system capacity to account for scheduled or unforeseen shutdown periods. As a result, a total capacity of 1,930 L/min (510 gpm) was assumed for the DX pump-and-treat system and 2,575 L/min (680 gpm) for HX pump-and-treat system. During periods of in situ treatment, bio-amended water injected at the designated injection wells is recovered at downgradient extraction wells and recirculated back through the bioinjection system,

bypassing the ion exchange treatment system, as described in Section 9.2.3.2. This reduces the likelihood of biodegradation byproducts reaching the ion exchange treatment systems. The treatment rate capacity of the bio-remediation loop is 1,000 L/min (263 gpm).

Alternative 2 is designed to operate until the Cr(VI) groundwater plume's concentrations are substantially reduced to less than the groundwater cleanup level in MTCA ("Groundwater Cleanup Standards" [WAC 173-340-720]). Wells along the river will be used to contain concentrations in the plume that are above 10 µg/L, thereby reducing migration to the river, so the AWQC can be maintained at points of groundwater discharge to the river.

All waste sites will be remediated before final remediation of groundwater. Remedial action completion for waste site and groundwater remediation is described in Section 9.2.

Results from the groundwater flow and transport model *Modeling of RI/FS Design Alternatives for 100-HR-3* (ECF-100HR3-11-0114) in Appendix F predict COC plumes and trends over the remediation simulation period.

The groundwater model results predict Cr(VI) plumes will be remediated to meet cleanup levels to 48 µg/L DWS and 10 µg/L AWQC throughout the aquifer after 11 and 25 years, respectively, of implementing the groundwater remedy under Alternative 2. Figure 9-8 (a-e) show the modeled Cr(VI) groundwater plumes after 3, 8, 18, 23, and 25 years of operating under this alternative. RPO activities will be conducted throughout the life of the project to deal with the uncertainty in groundwater modeling simulations.

EPC calculations will be used to determine completion of groundwater remedial actions. Model predictions include statistical evaluations to estimate EPCs for the following concentration values: maximum encountered; 90th percentile; mean; and Cr(VI) median concentration trends for 100-D south, 100-D north, horn area, and 100-H, and along the river shoreline. The EPC trends for this alternative are included in *Modeling of RI/FS Design Alternatives for 100-HR-3* (ECF-100HR3-11-0114).

As described in Section 9.2, groundwater COCs co-extracted for treatment of the Cr(VI) plumes are managed through MNA and ICs to achieve cleanup levels. Predicted concentrations of the nitrate and strontium-90 COCs in the combined groundwater influent stream to the pump-and-treat facilities will be at concentrations below their respective cleanup levels. Predicted concentration trends for each COC in the combined influent to the pump-and-treat facilities are provided in *Modeling of RI/FS Design Alternatives for 100-HR-3* (ECF-100HR3-11-0114). Hexavalent chromium in the treatment system influent will be removed using ion exchange treatment technology. The other COCs in the resulting treatment system effluent injected into the aquifer remain below cleanup levels. Nitrate contaminated groundwater is within the capture footprint of the Cr(VI) pump-and-treat system well network. The groundwater model simulation for this alternative indicates that nitrate contamination is remediated to cleanup levels within the Cr(VI) remediation period. Nitrate plume depictions after 3, 5, 8, 12, and 13 years of operating under this alternative are provided in Figure 9-9 (a-e) showing nitrate concentrations are below cleanup levels after 13 years.

Strontium-90 has low solubility and mobility in groundwater and small pockets of strontium-90 contamination remain in the aquifer following completion of the Cr(VI) remediation. The pump-and-treat systems will be shut down following completion of Cr(VI) remediation. Remaining strontium-90 contamination will attenuate to cleanup levels through radiological decay in 31 years after Cr(VI) remediation (i.e., 56 years after implementation of the Alternative 2 remedy). Strontium-90 plume depictions after 3, 18, 48, and 56 years of operating under this alternative are provided in Figure 9-10 (a-d).

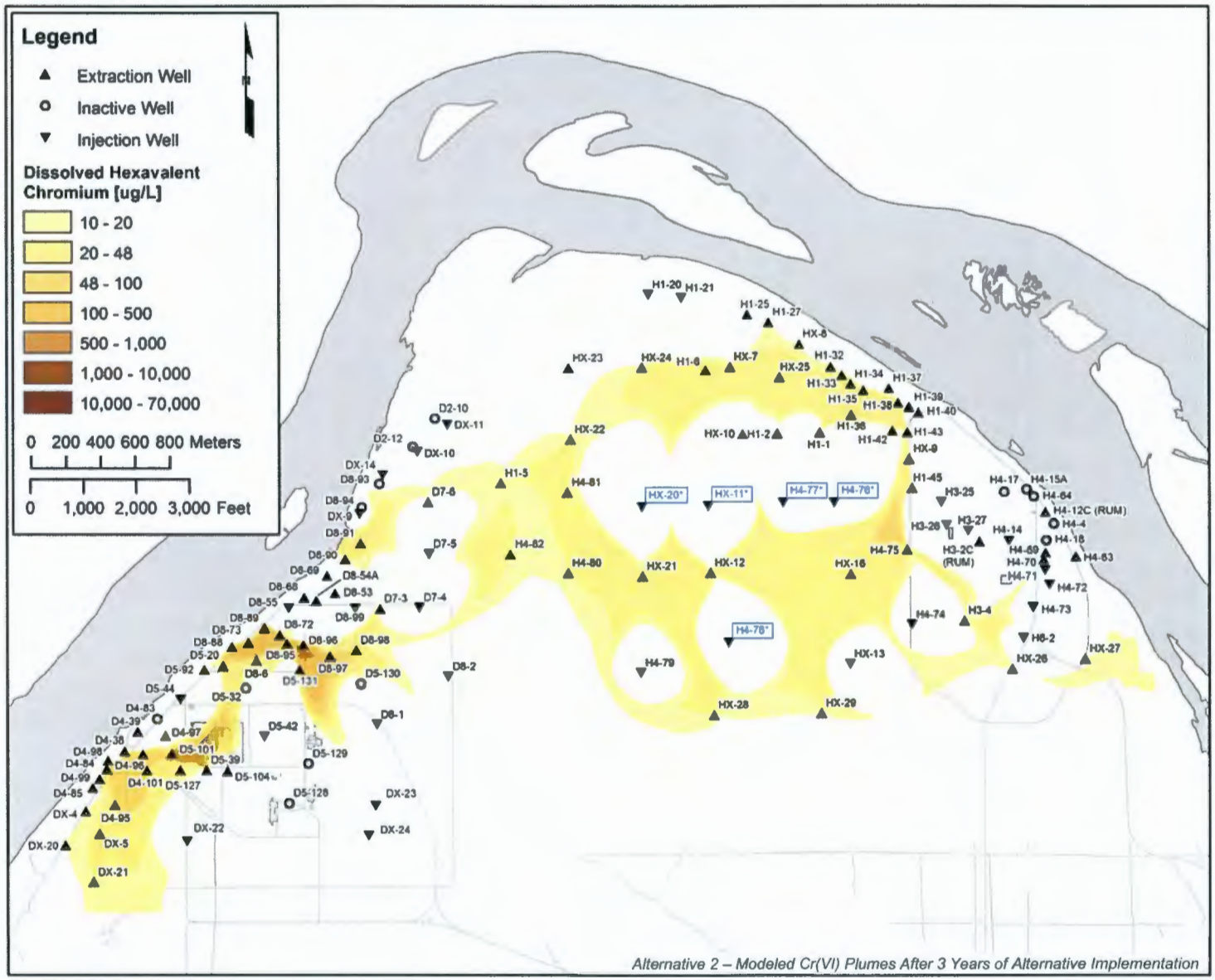

Figure 9-8a. Alternative 2—Modeled Cr(VI) Plumes After 3 Years

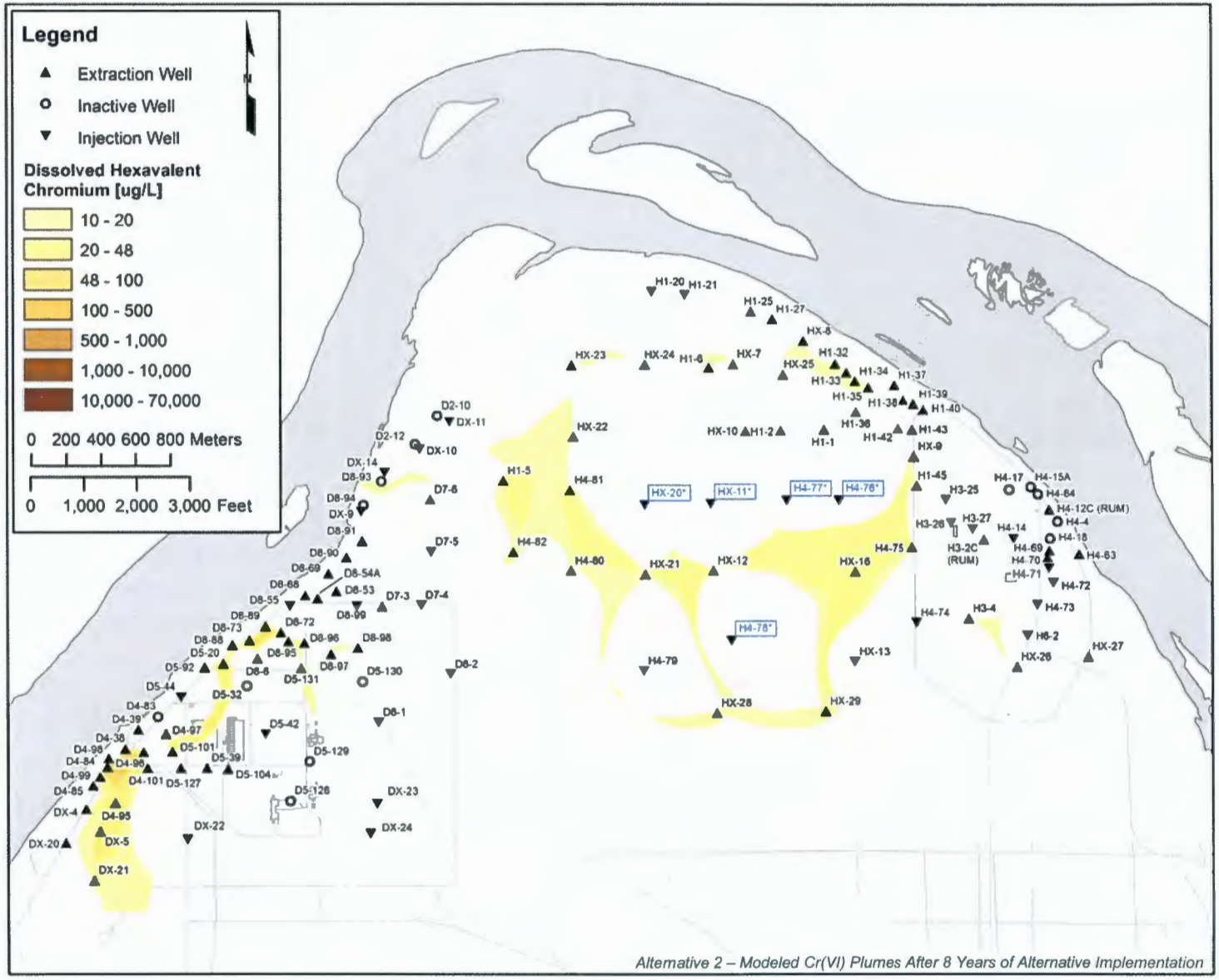

Figure 9-8b. Alternative 2—Modeled Cr(VI) Plumes After 8 Years

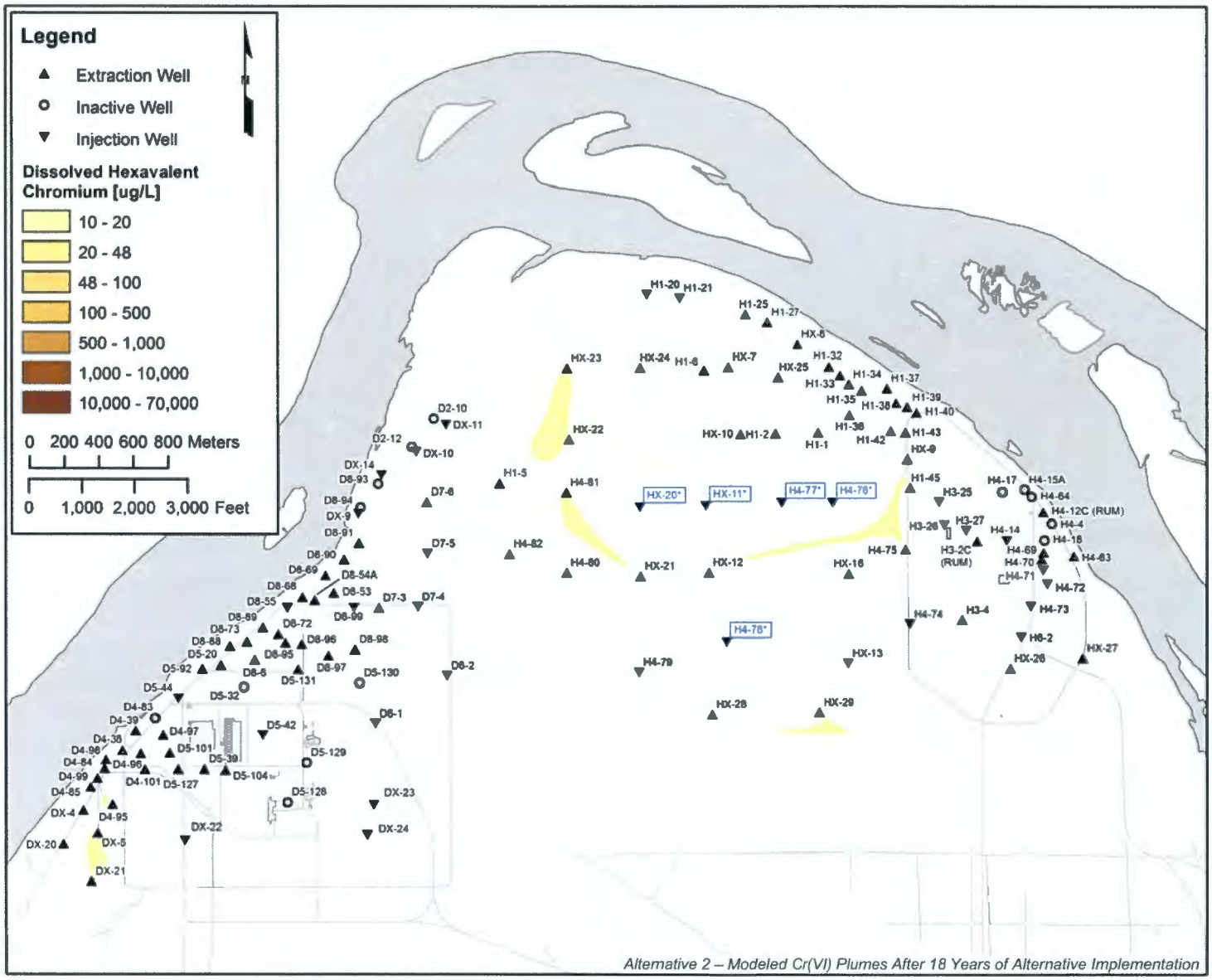

Figure 9-8c. Alternative 2—Modeled Cr(VI) Plumes After 18 Years

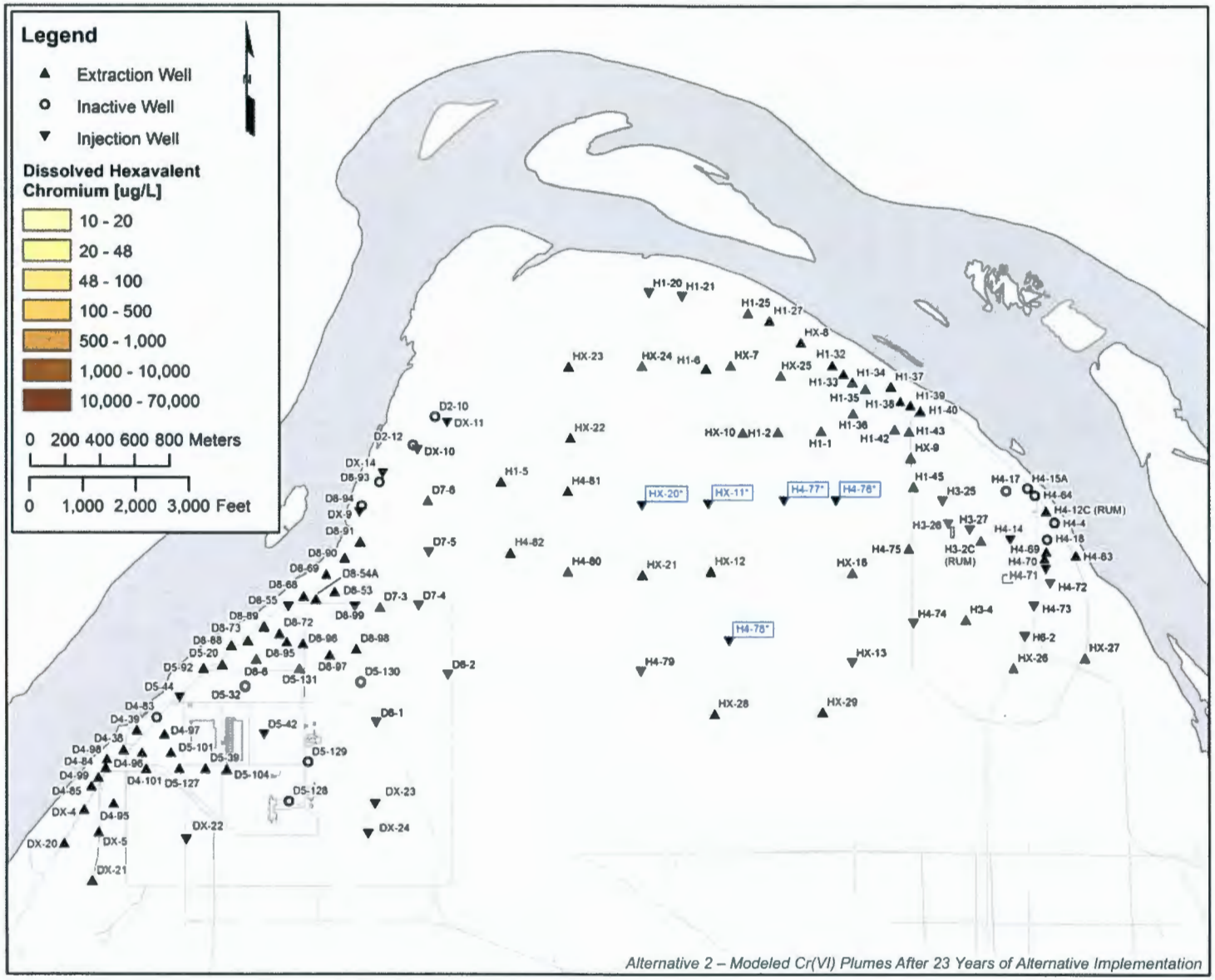

Figure 9-8d. Alternative 2—Modeled Cr(VI) Plumes After 23 Years

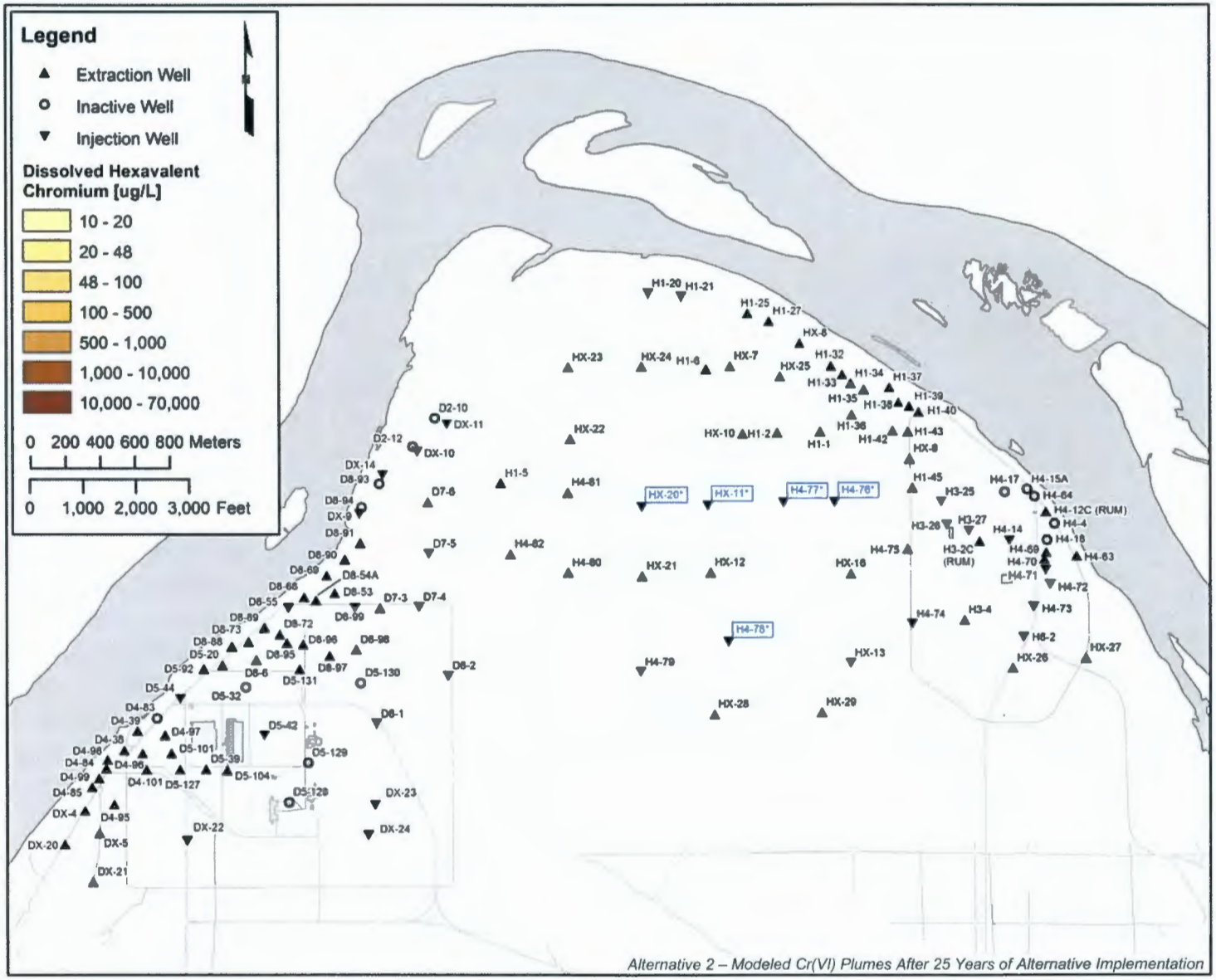

Figure 9-8e. Alternative 2—Modeled Cr(VI) Plumes After 25 Years

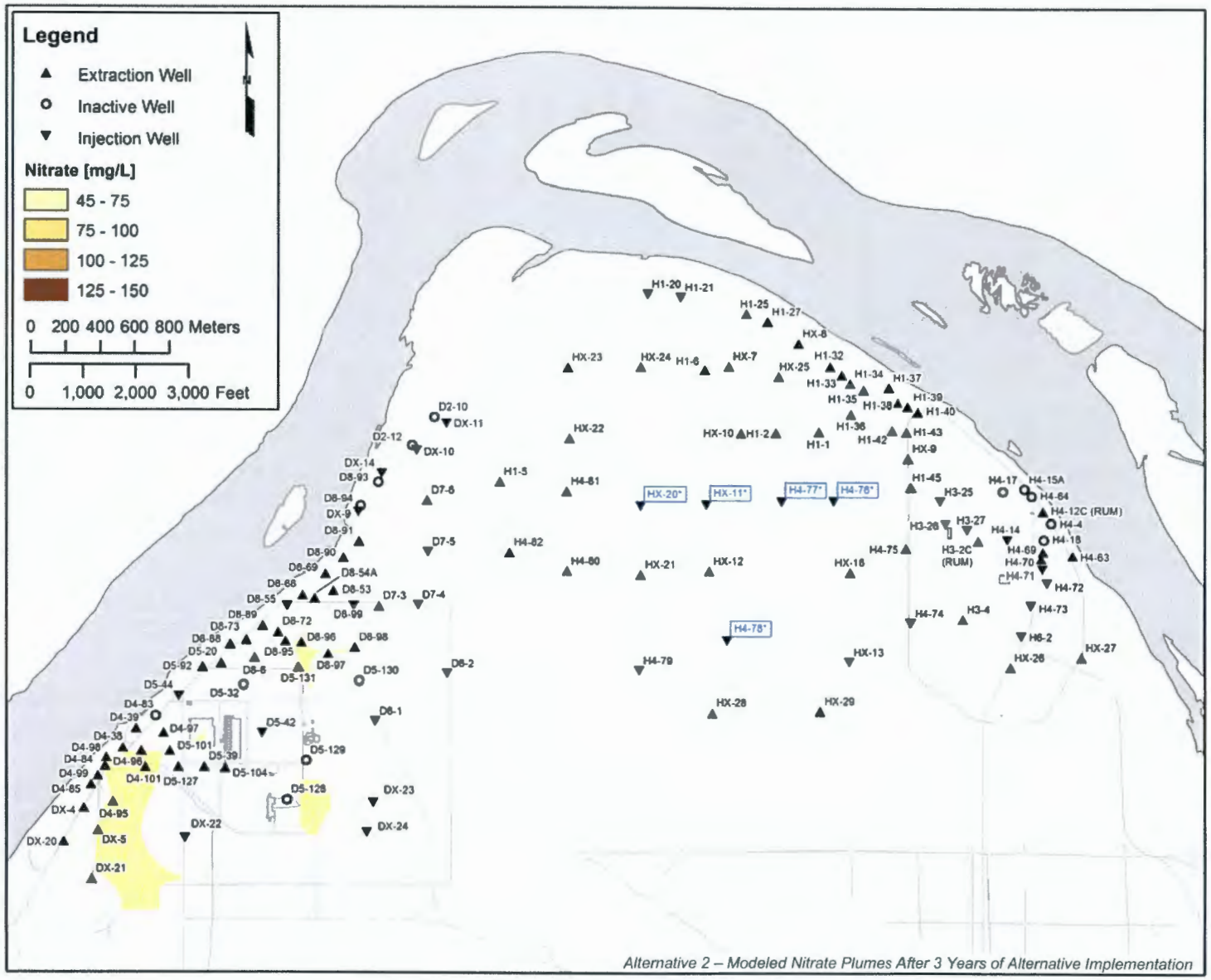

Figure 9-9a. Alternative 2—Modeled Nitrate Plumes After 3 Years

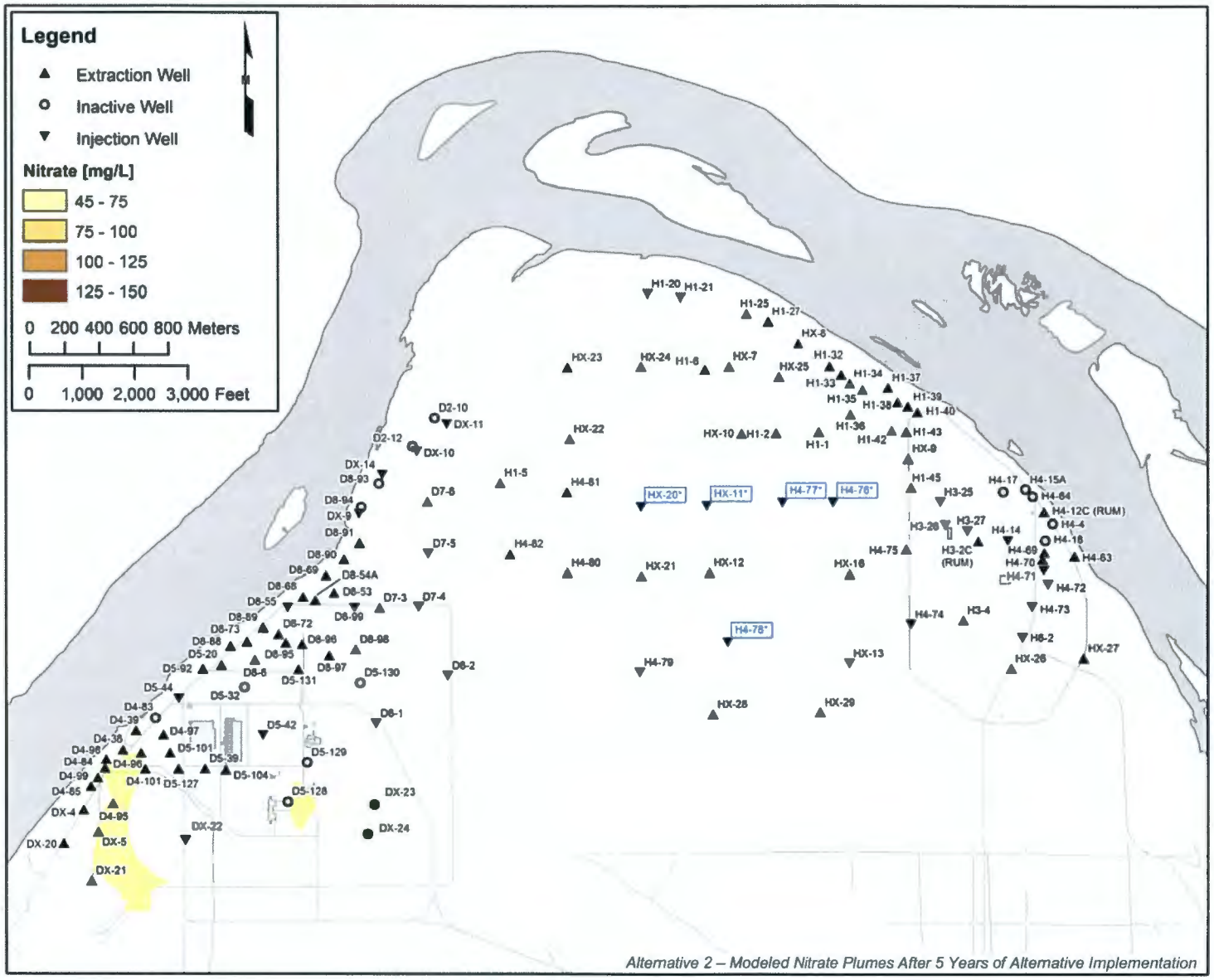

Figure 9-9b. Alternative 2—Modeled Nitrate Plumes After 5 Years

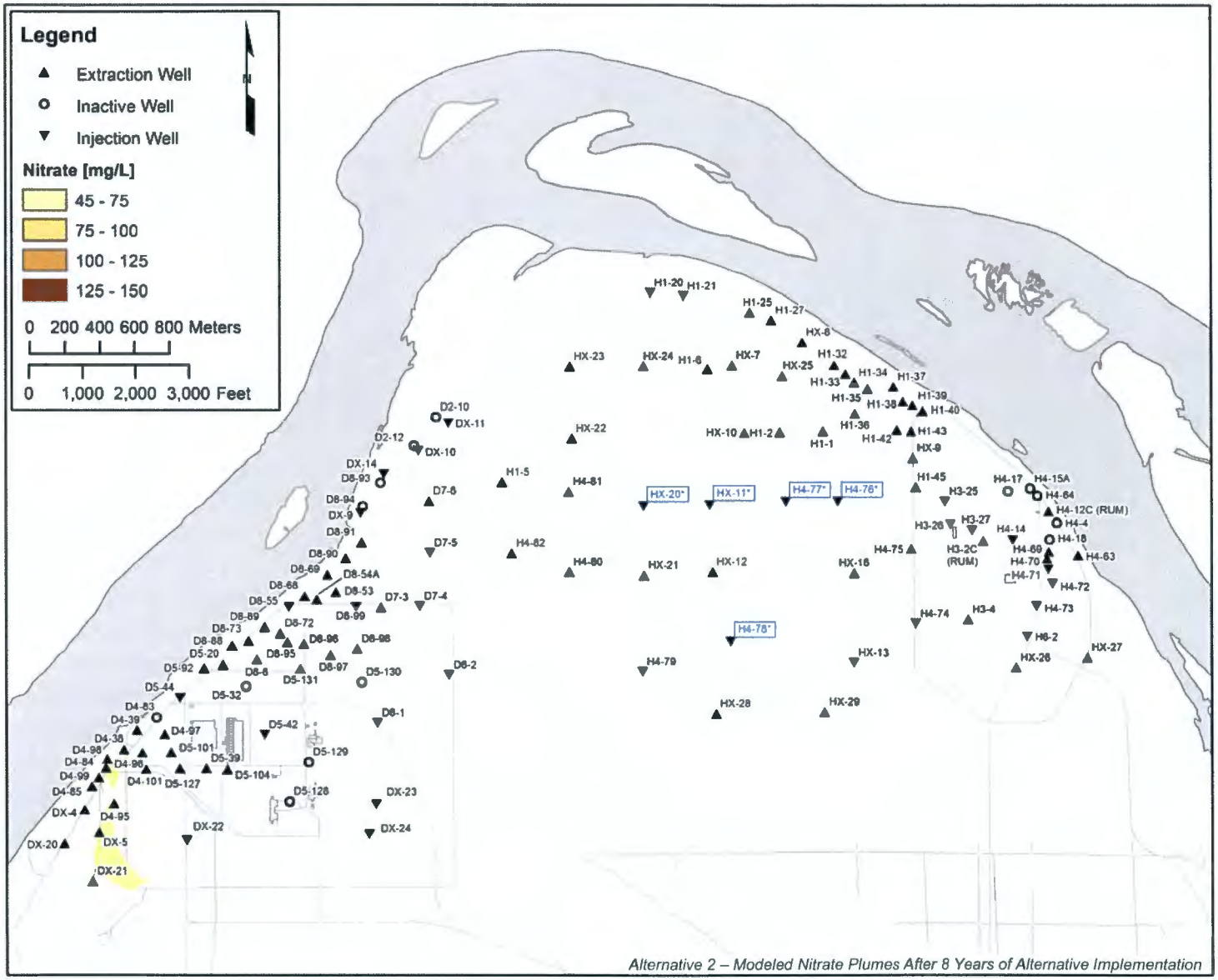

Figure 9-9c. Alternative 2—Modeled Nitrate Plumes After 8 Years

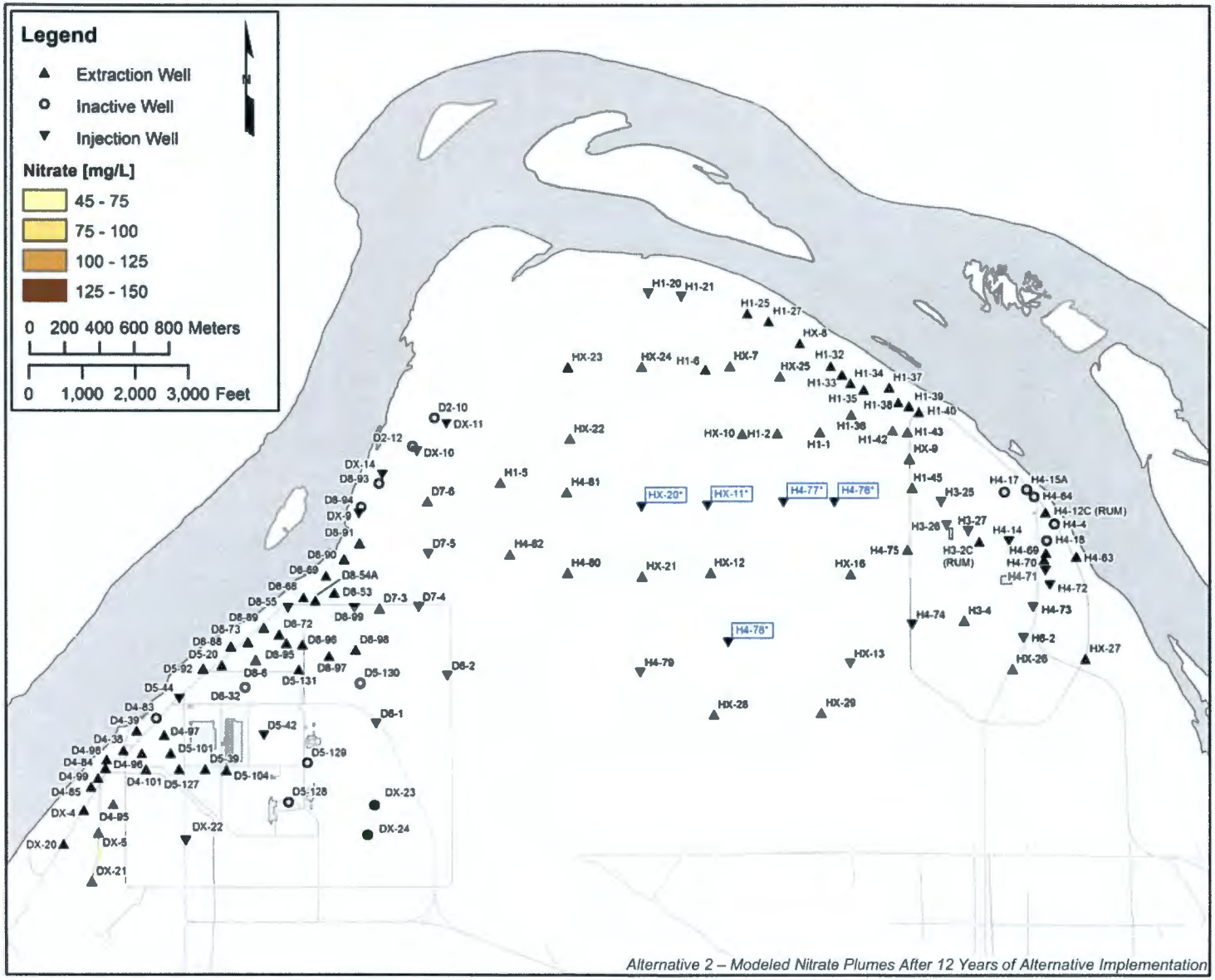

Figure 9-9d. Alternative 2—Modeled Nitrate Plumes After 12 Years

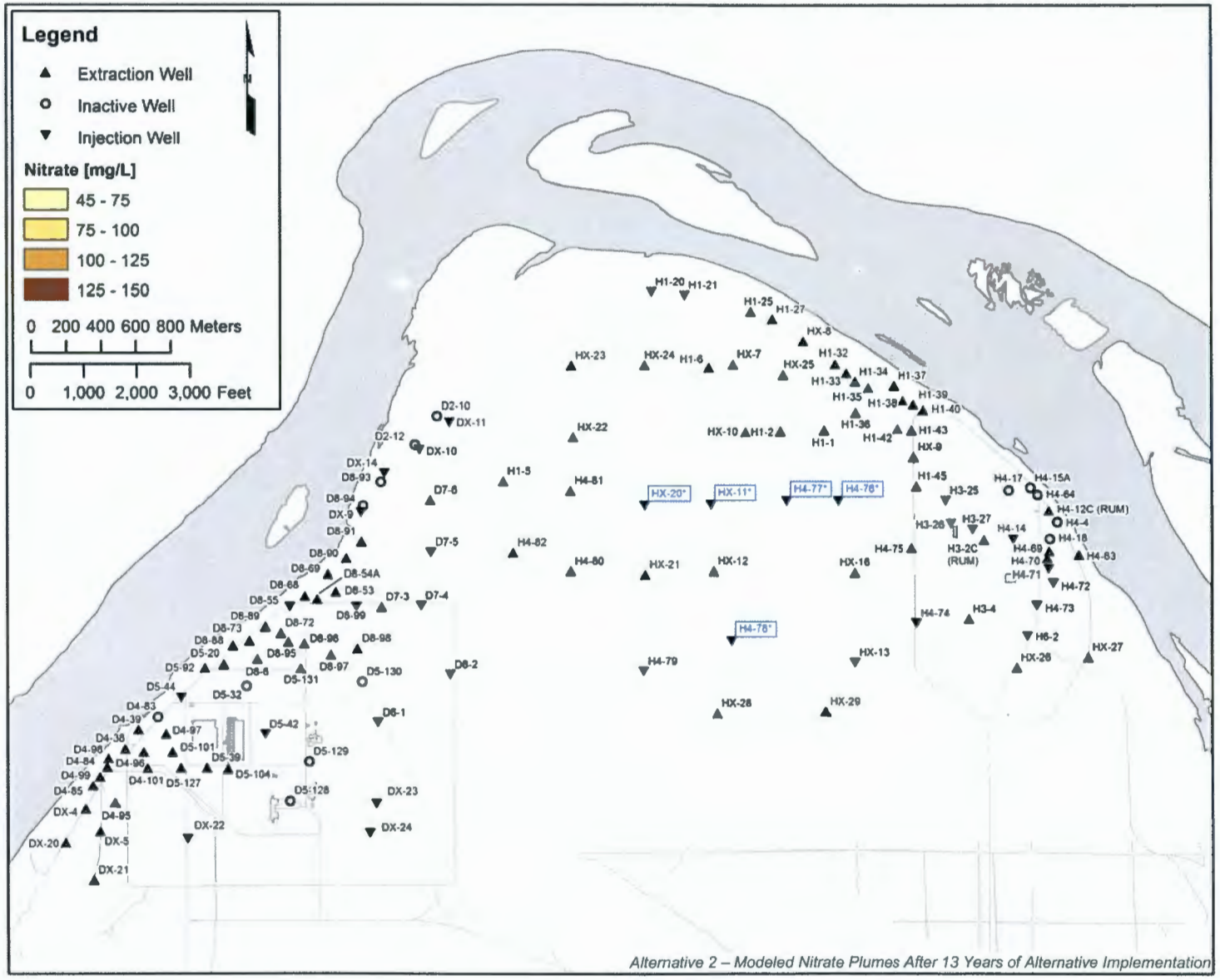

Figure 9-9e. Alternative 2—Modeled Nitrate Plumes After 13 Years

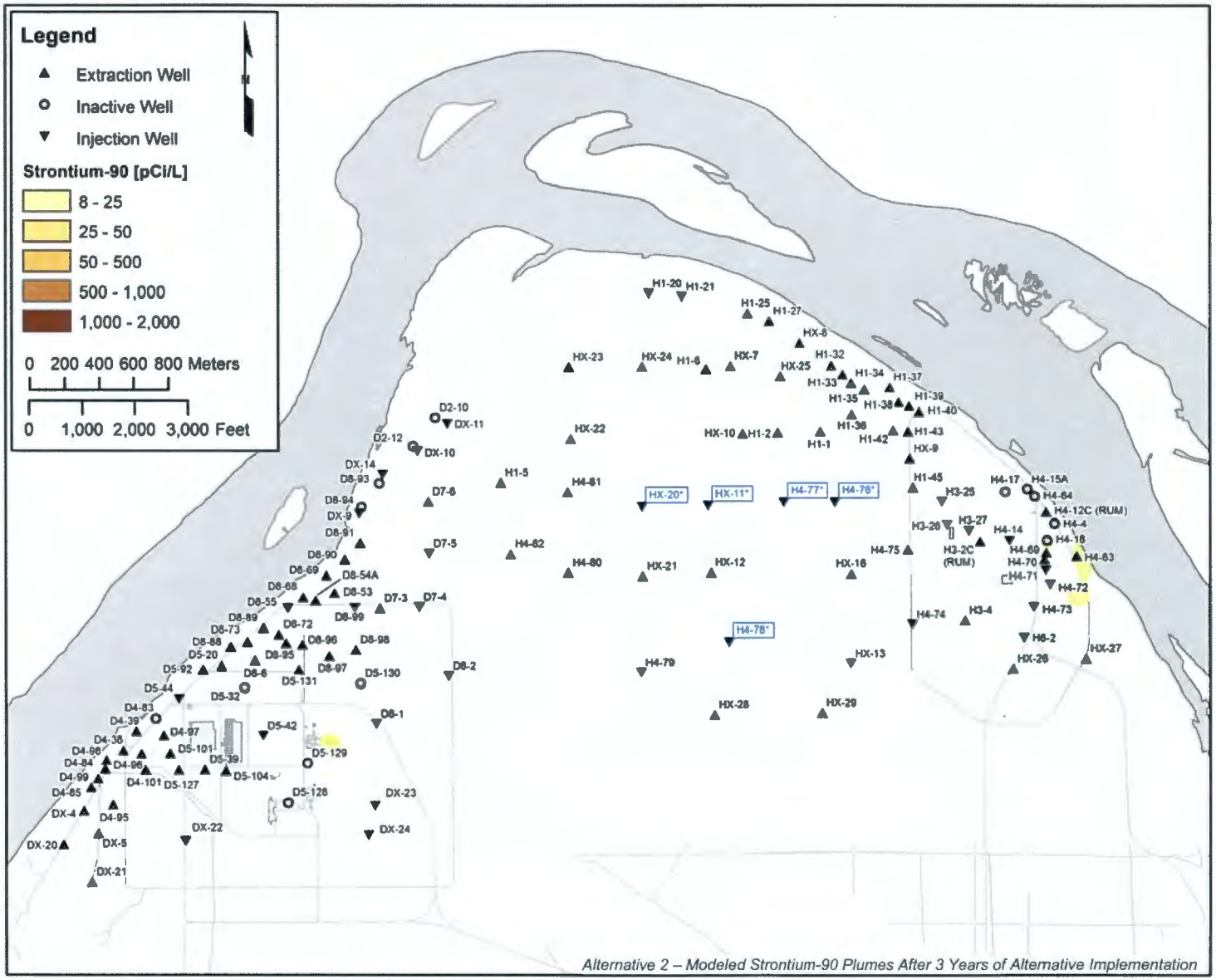

Figure 9-10a. Alternative 2—Modeled Strontium-90 Plumes After 3 Years

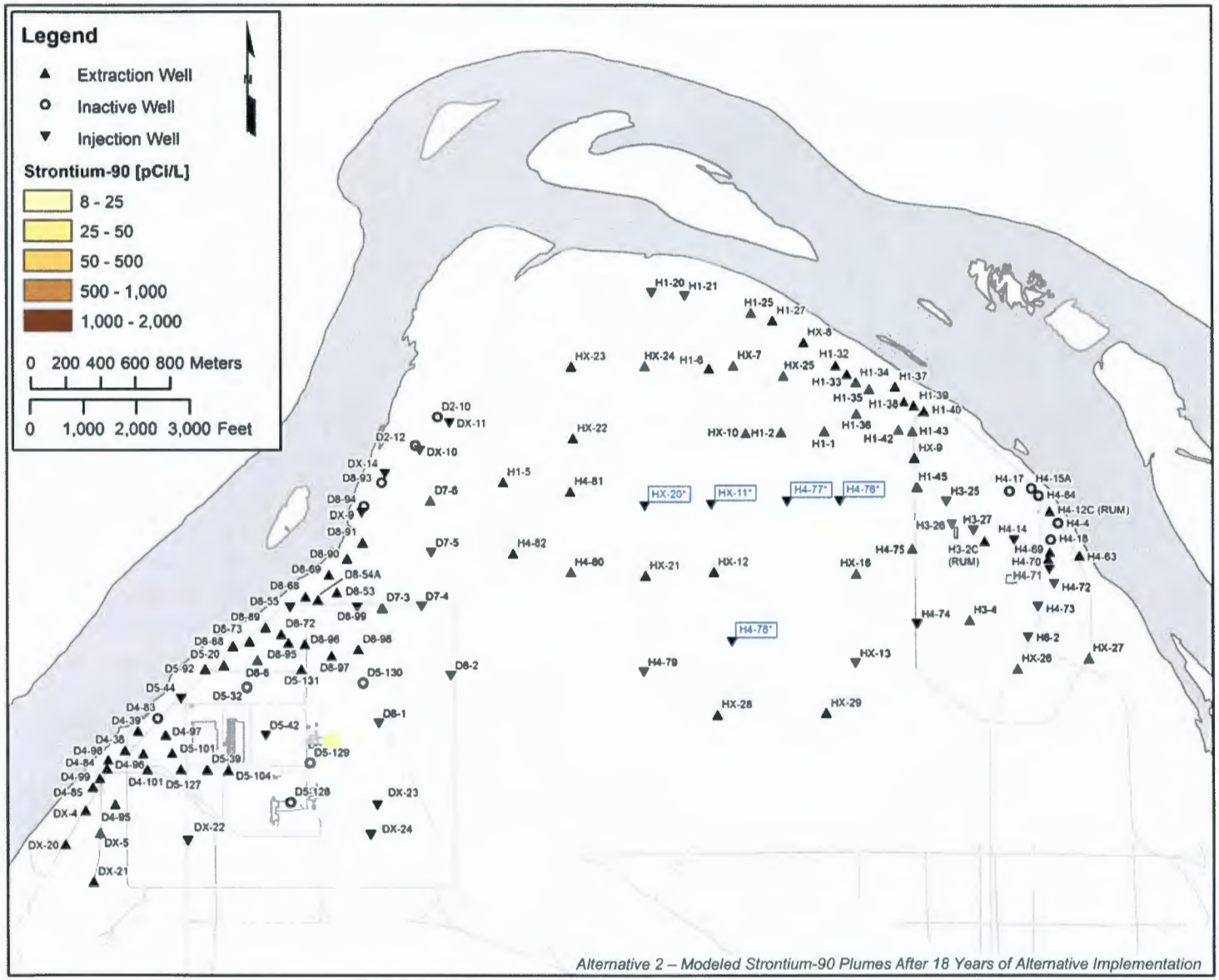

Figure 9-10b. Alternative 2—Modeled Strontium-90 Plumes After 18 Years

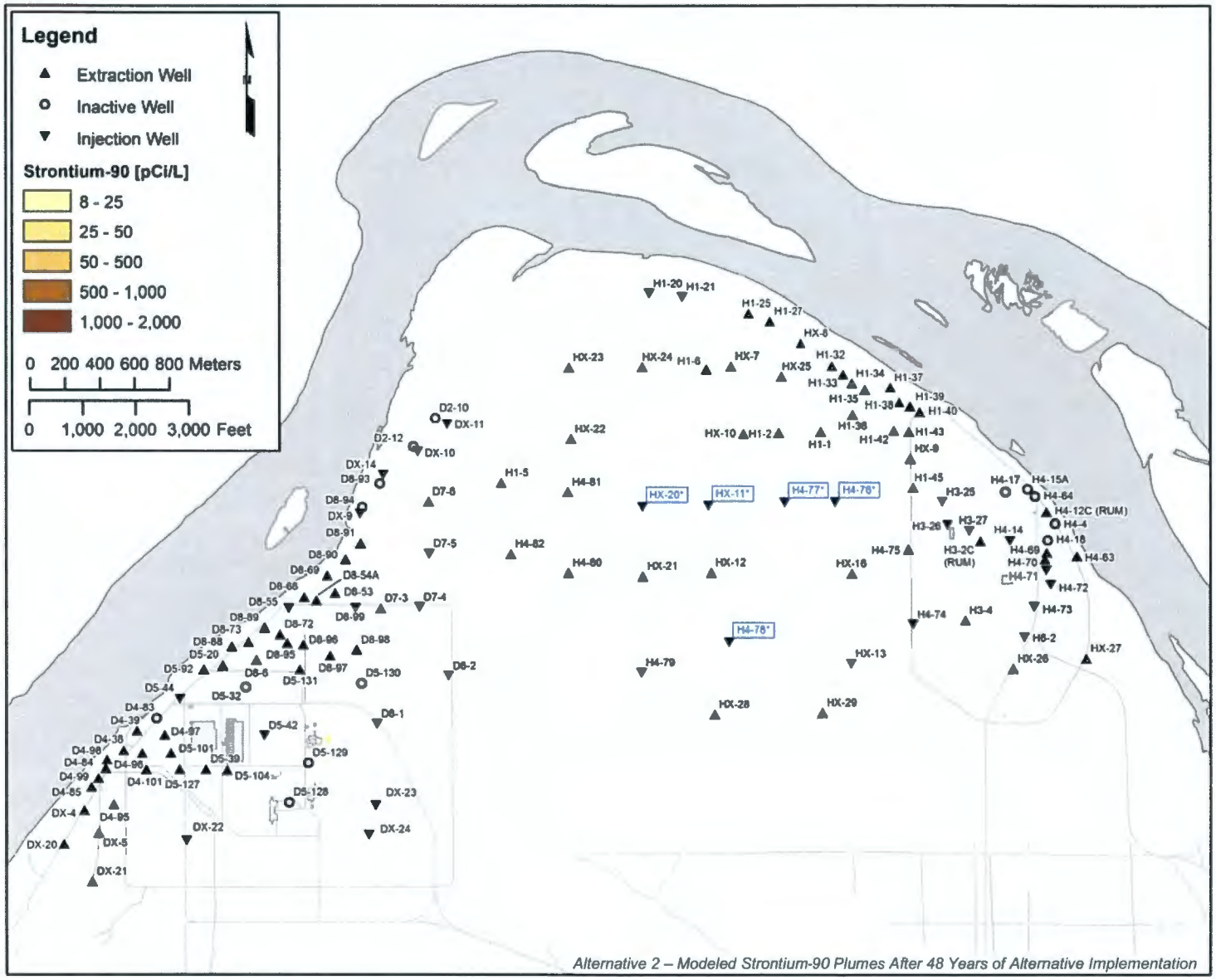

Figure 9-10c. Alternative 2—Modeled Strontium-90 Plumes After 48 Years

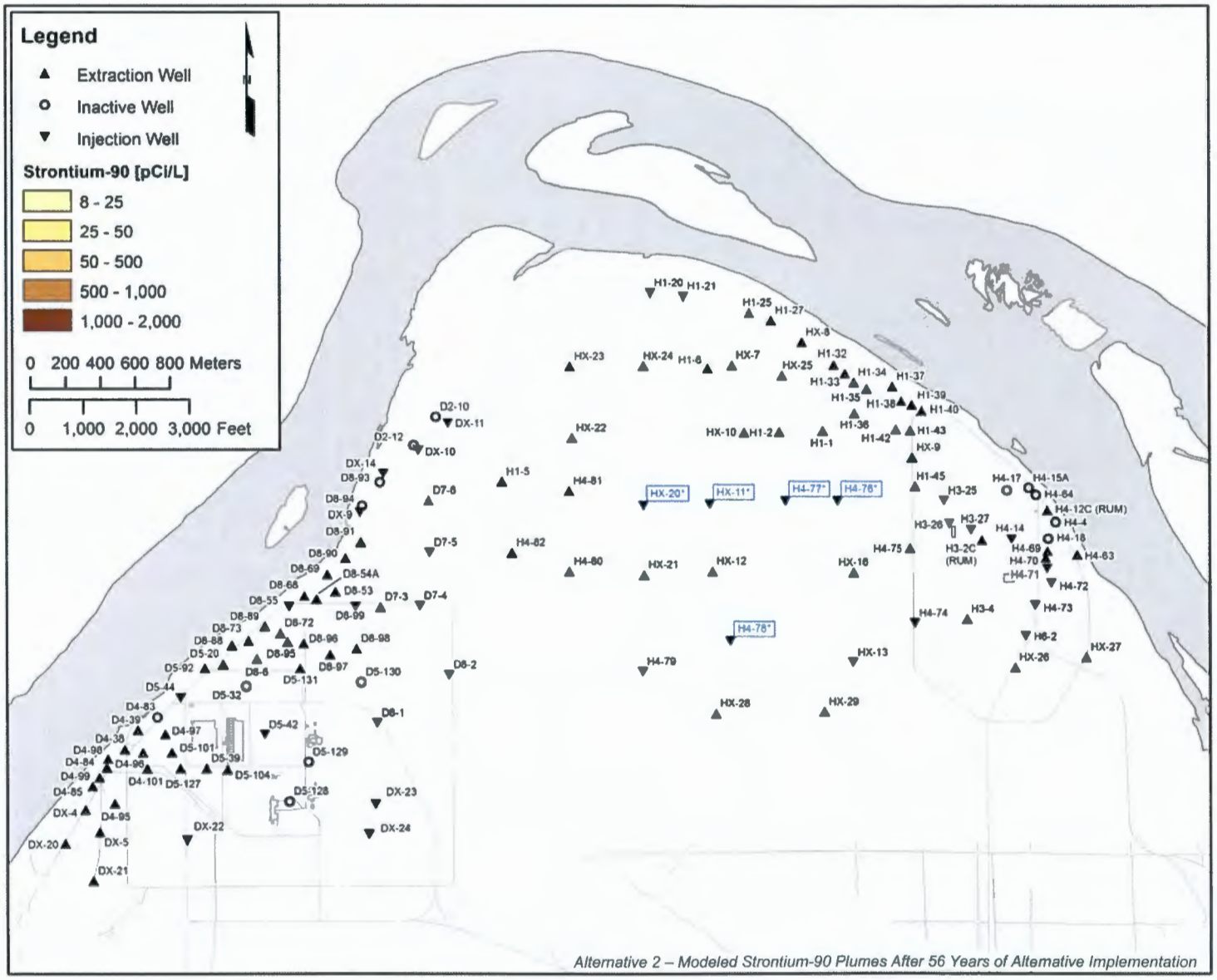

Figure 9-10d. Alternative 2—Modeled Strontium-90 Plumes After 56 Years

9.2.4 Alternative 3—RTD and Grouting of Waste Site and Increased Capacity Groundwater Pump-and-Treat for Groundwater

Alternative 3 uses a combination of RTD, void-fill grouting, containment, and MNA/Institutional Controls for waste site remediation to achieve the RAOs, and expanded pump-and-treat for groundwater. Table 9-8 presents the waste site and groundwater components of Alternative 3, and Figure 9-11 presents a pictorial summary of Alternative 3. The cost for Alternative 3 is provided in Table 9-9. Details for the cost estimate are provided in Appendix J (*Environmental Cost Estimate for 100-D/H Vadose Zone and Groundwater RI/FS* [ECE-100HR311-00004]). Additional details on remedial components specific to this alternative are presented in sections identified in Table 9-8.

Table 9-8. Components for Alternative 3—RTD and Grouting of Waste Site and Increased Capacity Groundwater Pump-and-Treat for Groundwater

Waste Site Components	General Components	
	No Further Action	No additional remedial actions are taken for the 146 waste sites identified in Section 8.2.1.1 (Chapter 8).
	Institutional Controls	Institutional controls to be implemented within the 100-D, 100-H, and horn areas for land use management and waste site information management include the following: • Permits required for excavations on the Hanford Site to prevent unplanned disturbance or infiltration as prohibited by CERCLA decision documents. • Land use and real property controls (for example, proprietary controls including easements and covenants) ensure that the use of land is in accordance with Hanford Site plans and CERCLA decision documents. • Notices providing visual identification and warning of hazardous or sensitive areas. • Procedural requirements for access, warning signs, or fencing to prevent or limit the access of humans to hazardous or sensitive areas. • Administrative mechanisms, such as the WIDS database, to maintain and provide access to information on the location and nature of contamination. Additional information on institutional controls is presented in Section 8.3.1 (Chapter 8). Post remediation institutional controls implemented at specific waste sites are identified in Table 9-4.

Table 9-8. Components for Alternative 3—RTD and Grouting of Waste Site and Increased Capacity Groundwater Pump-and-Treat for Groundwater

Waste Site Components (cont.)	For Waste Sites That Exceed Human Health, Environment, Surface Water, or Groundwater Protection PRGs	
	MNA/institutional control	MNA/institutional controls applied to waste sites 116-D-8, 116-DR-9 and 100-D-25. Waste site 100-D-25 is fully encompassed within waste site 116-DR-9. Risk evaluation of the verification data indicates that the shallow decision units for these sites contain Cs-137 at concentrations greater than the residential RBSL (Chapter 6). No ecological risks are identified for the sites (Chapter 7). Cesium-137 concentrations at the three sites will decay through natural attenuation to levels less than residential RBSLs in year 2035. Institutional controls will be in place through this period. Additional information on institutional controls is presented in Section 8.3.1 (Chapter 8).
	Void-fill Grouting	Void-fill grouting, as described in Section 9.2.3.1, will be used to remediate the box flumes of the 100-H-36 waste site to immobilize contaminants on interior surfaces of the underground concrete box flume sluiceway.
	RTD	RTD to depth of contamination exceeding surface water or groundwater PRGs, or until groundwater is encountered if contamination extends into groundwater, using standard and deep excavation technologies, as described in Section 9.2.2.2. RTD is applied to waste sites listed in Table 8-5 (Chapter 8) that are expected to be remediated under the 100 Area Remaining Sites ROD (EPA/ROD/R10-99/039). RTD is effective in meeting the human health, environment, surface water, and groundwater PRGs as reflected in the risk evaluation of completed waste sites with close out/verification data (Chapter 6). RTD also applied to waste sites listed in Table 8-5 (Chapter 8) that are yet to be remediated and are not remediated under one of the other waste site remedial components listed previously. Table 9-2 identifies applicable waste sites remediated by RTD using standard and deep excavation technologies. Within the SMDP process, ecological PRGs will be considered at a population level for wildlife and at a community level for plants and invertebrates, to determine whether cleanup action is required to protect ecological receptors.

Table 9-8. Components for Alternative 3—RTD and Grouting of Waste Site and Increased Capacity Groundwater Pump-and-Treat for Groundwater

Groundwater Components	Pump-and-Treat	Operation of existing pump-and-treat systems with additional 2,370 L/min (625 gal/min) treatment capacity at 100-DX and 3,030 L/min (800 gal/min) at100-HX pump-and-treat systems, and extraction and injection wells to expand treatment coverage. Placement of additional extraction and injection wells, extraction and injection flow rates, and well operational periods were determined through groundwater model simulations described in *Modeling of RI/FS Design Alternatives for 100-HR-3* (ECF-100HR3-11-0114). The RPO process will be used to provide ongoing evaluations to ensure the system meets performance requirements that will be defined in the RD/RAWP. Designed to treat Cr(VI) with active remediation out to Cr(VI) 10 µg/L contour. Treatment of extracted water through ion exchange. Operation of pump-and-treat system to meet AWQC at points of groundwater discharge to the river.
	MNA and Institutional Controls	MNA as described in Section 9.2.2.3. Manage strontium-90 in groundwater through MNA and institutional controls until concentrations meet cleanup standards. Manage the plumes until concentrations are below the cleanup levels in the plume and after the pump-and-treat system is shut off. Monitoring of COCs to track the attenuation processes. Institutional controls to be implemented within 100-D/H include land use management and waste site information management. Additional institutional controls implemented at specific waste sites include the following: • Permits required for excavations on the Hanford Site to prevent unplanned disturbance or infiltration as prohibited by CERCLA decision documents • Land use and real property controls (for example, proprietary controls including easements and covenants) ensure that the use of land is in accordance with Hanford Site plans and CERCLA decision documents • Groundwater use management, as described in *Sitewide Institutional Controls Plan for Hanford CERCLA Response Actions* (DOE/RL-2001-41), ensures proper use of groundwater through groundwater controls

Table 9-8. Components for Alternative 3—RTD and Grouting of Waste Site and Increased Capacity Groundwater Pump-and-Treat for Groundwater

Groundwater Components (cont.)	MNA and Institutional Controls (cont.)	• Administrative mechanisms, such as the WIDS database, to maintain and provide access to information on the location and nature of contamination Additional information on institutional controls is presented in Section 8.3.1 (Chapter 8).
	Monitoring Requirements	Remedy performance monitoring will be defined as part of the remedial design. Evaluates the effectiveness of the selected alternative to attain the cleanup levels. The existing monitoring program will be expanded when number of monitoring wells is added. For cost estimating purposes, it has been assumed that 12 new monitoring wells will be installed. Groundwater performance monitoring is presented in Section 9.2.2.7, which also identifies constituents included in the monitoring program.

Remediation components specific to Alternative 3 (described in Table 9-8) are described in Section 9.2.2 under common elements and Section 9.2.3.1 for void-fill grouting.

9.2.4.1 Groundwater Model for Alternative 3

This section provides a summary of the groundwater model for Alternative 3. Alternative 3 includes expansion to the interim action pump-and-treat facilities to provide an additional 5,400 L/min (1,425 gal/min) treatment capacity to the interim action 100-D/H pump-and-treat systems. New extraction and injection wells will be installed, along with increased treatment capacity to handle the additional flow. Alternative 3 expands the pump-and-treat systems with 41 new or converted extraction and injection wells to the 100-DX pump-and-treat system and 49 new extraction and injection wells to the 100-HX pump-and-treat system. Well use (that is, extraction or injection) and duration of well operation is detailed in *Modeling of RI/FS Design Alternatives for 100-HR-3* (ECF-100HR3-11-0114). Extraction and injection well locations, specified in *Modeling of RI/FS Design Alternatives for 100-HR-3* (ECF-100HR3-11-0114, Table 3-5), were identified to support alternatives evaluation in this FS. The final placement of wells will be determined in the RD/RAWP.

The model assumes 85 percent availability of the pump-and-treat system capacity to account for scheduled and unforeseen shutdown periods. As a result, a total capacity of 3,940 L/min (1,040 gpm) was assumed for the DX pump-and-treat system and 5,150 L/min (1,360 gpm) for HX pump-and-treat system.

All waste sites will be remediated before final remediation of groundwater. Remedial action completion for waste site and groundwater remediation is described in Section 9.2.

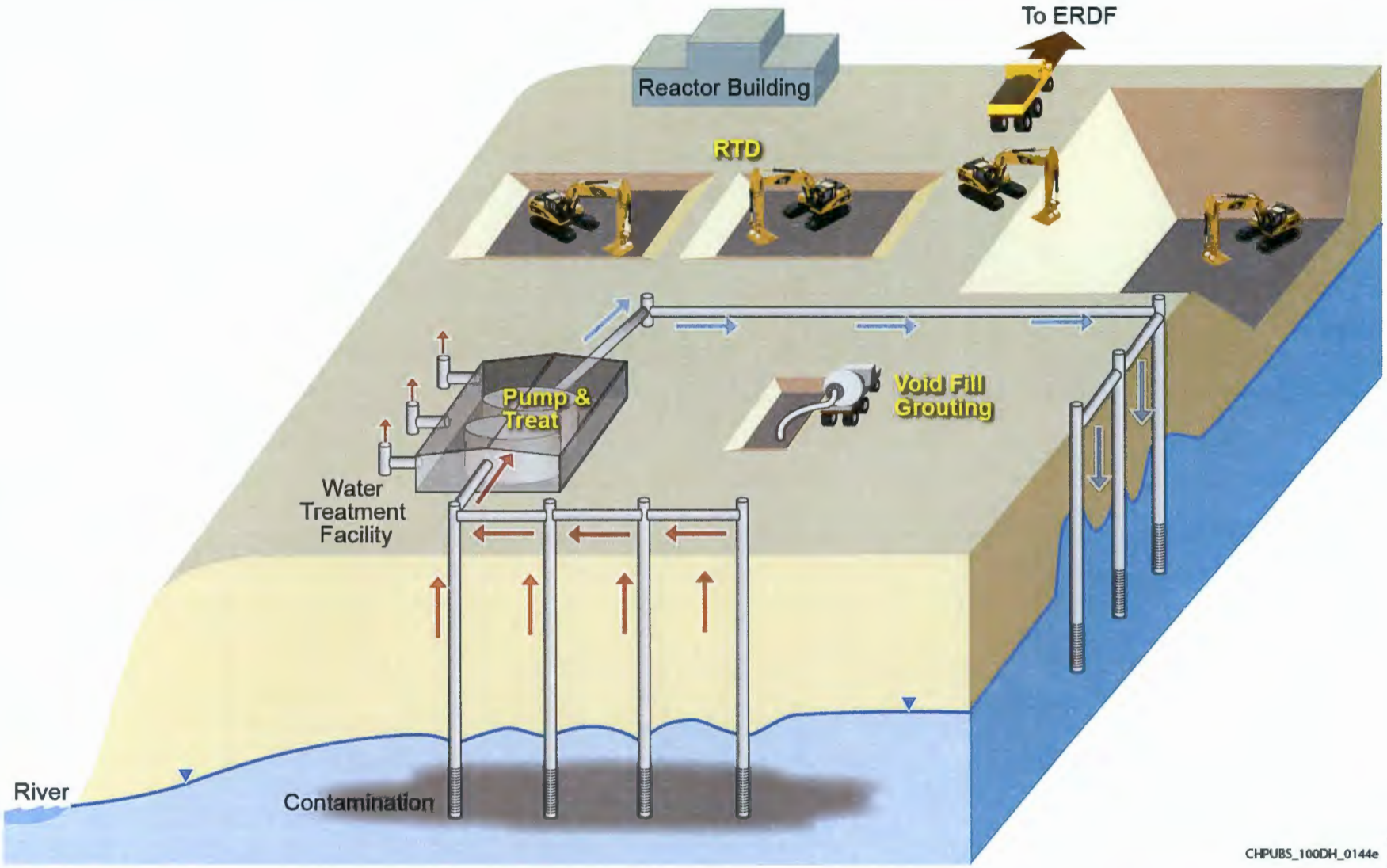

Figure 9-11. Alternative 3—RTD and Grouting of Waste Site and Increased Capacity Groundwater Pump-and-Treat for Groundwater

Table 9-9. Alternative 3—Cost Estimates (in Millions)[a]

Waste Site Treatment	Capital	$132.4
	Total O&M	$25.7
	Total Periodic[b]	$0.8
	Total Nondiscounted	$158.9
	Net Present Value	$142.9
Groundwater Treatment	Capital	$117.8
	Total O&M	$287.5
	Total Periodic[b]	$82.3
	Total Nondiscounted	$487.6
	Net Present Value	$427.2
Total Nondiscounted Cost of Alternative	$646.5	
Total Net Present Value (Discounted) of Alternative	$570.1	

a. Order of magnitude cost estimates and support information are presented in Appendix J. Costs for institutional controls are included under the costs for waste site treatment.

b. Periodic costs include O&M or construction activities, including costs to replace an installed remedy or components of an installed remedy, and services that are not included in initial capital costs or annual O&M costs. Periodic costs may be one-time costs or costs that recur at intervals over the life of the remedy.

Results of groundwater modeling (*Modeling of RI/FS Design Alternatives for 100-HR-3* [ECF-100HR3-11-0114]) predict COC plumes and trends over the remediation simulation period. The groundwater model results predict Cr(VI) plumes will be remediated to meet cleanup levels to 48 µg/L DWS and 10 µg/L AWQC throughout the aquifer after 6 and 12 years, respectively, of implementing the groundwater remedy under Alternative 3. Figure 9-12 (a-d) show the modeled Cr(VI) groundwater plumes after 3, 6, 10, and 12 years of operating under this alternative. RPO activities will be conducted throughout the life of the project to deal with the uncertainty in groundwater modeling simulations.

EPC calculations will be used to determine completion of groundwater remedial actions. Model predictions include statistical evaluations to estimate EPCs for the following concentration values: maximum encountered; 90th percentile; mean; and Cr(VI) median concentration trends for the 100-D south, 100-D north, horn area, and 100-H, and along the river shoreline. The EPC trends for this alternative are included in *Modeling of RI/FS Design Alternatives for 100-HR-3* (ECF-100HR3-11-0114).

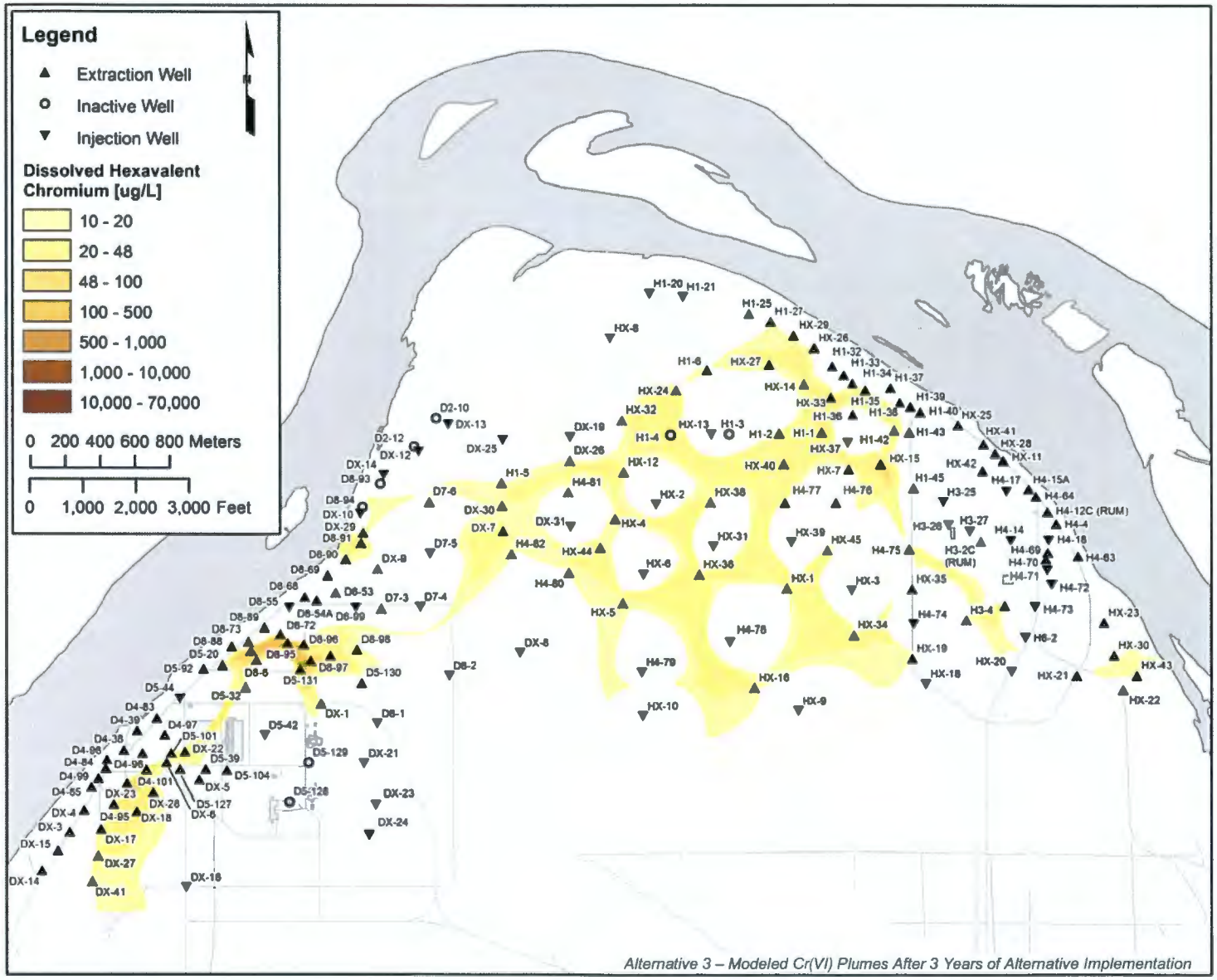

Figure 9-12a. Alternative 3—Modeled Cr(VI) Plumes After 3 Years

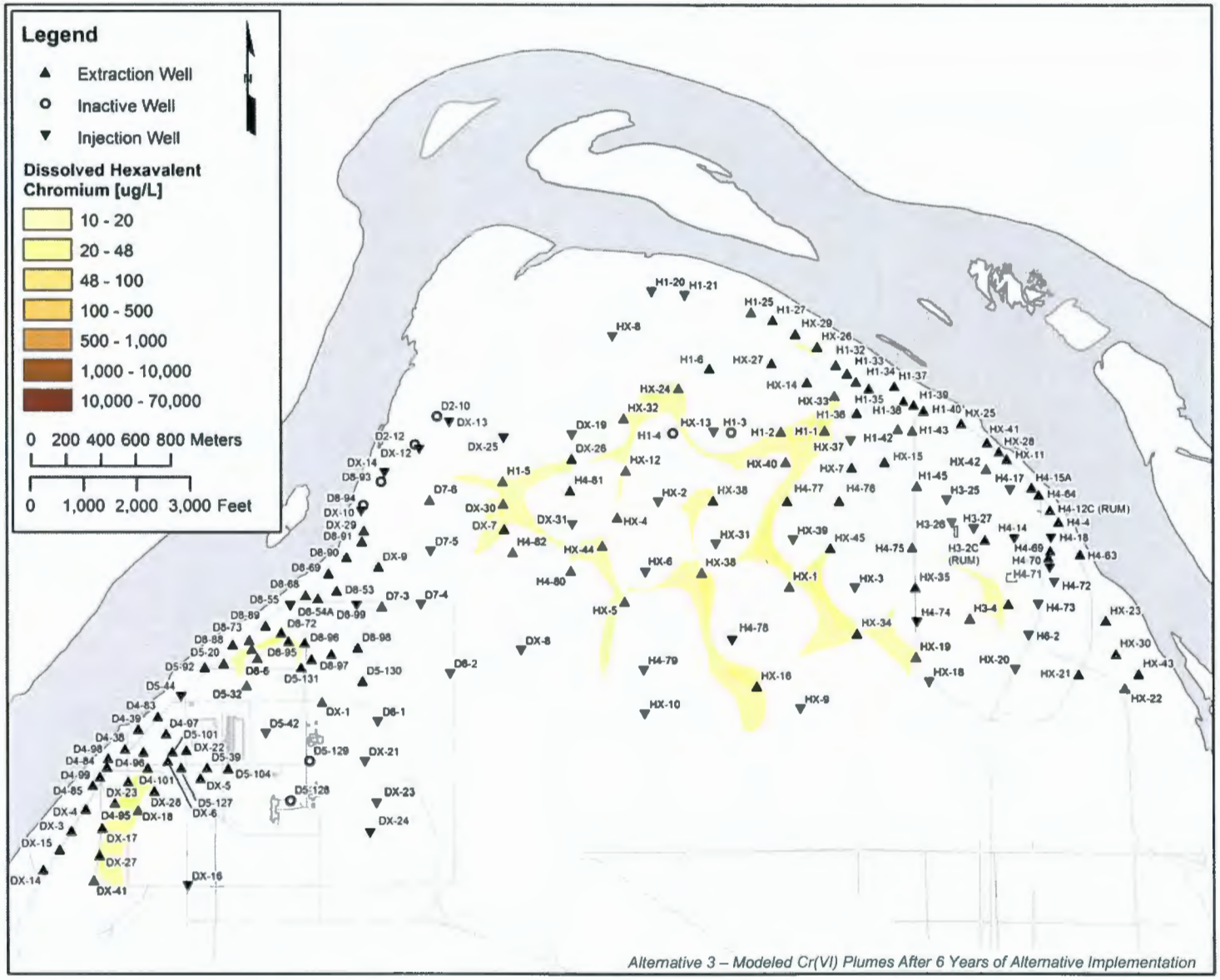

Figure 9-12b. Alternative 3—Modeled Cr(VI) Plumes After 6 Years

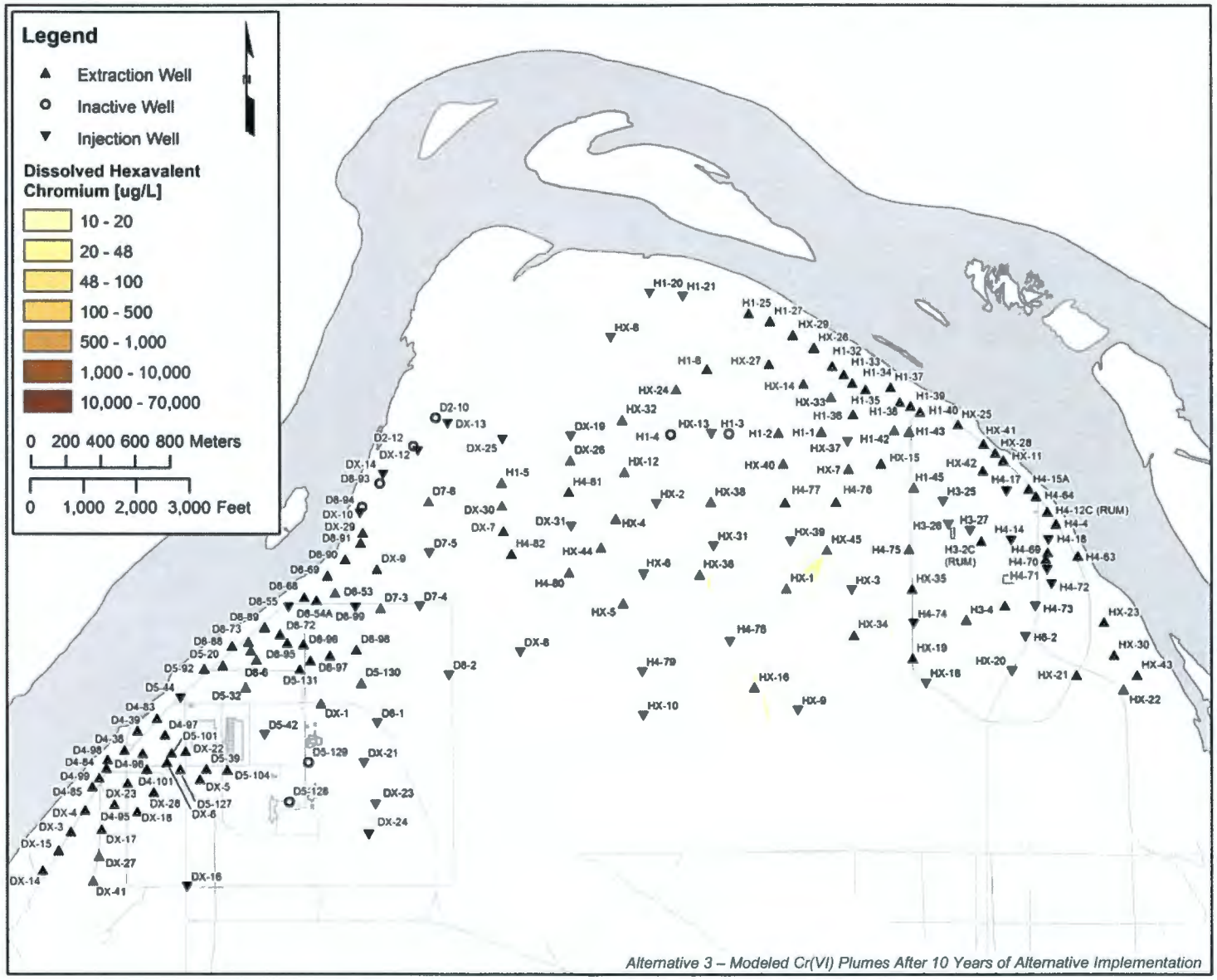

Figure 9-12c. Alternative 3—Modeled Cr(VI) Plumes After 10 Years

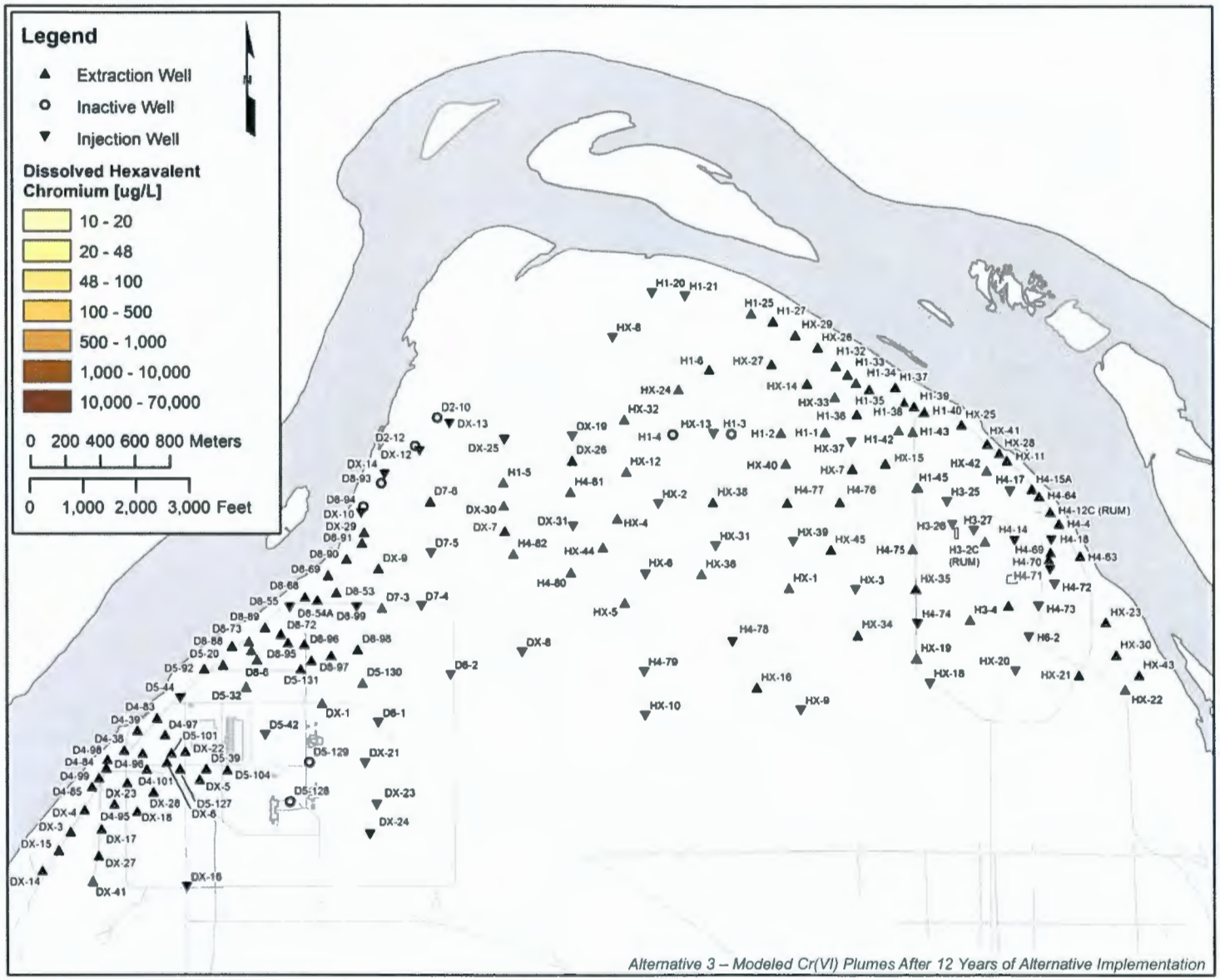

Figure 9-12d. Alternative 3—Modeled Cr(VI) Plumes After 12 Years

As described in Section 9.2, groundwater COCs co-extracted for treatment of the Cr(VI) plumes are managed through MNA and ICs to achieve cleanup levels. Predicted concentrations of the nitrate and strontium--90 COCs in the combined groundwater influent stream to the pump-and-treat facilities will be at concentrations below their respective cleanup levels. Predicted concentration trends for each COC in the combined influent to the pump-and-treat facilities are provided in *Modeling of RI/FS Design Alternatives for 100-HR-3* (ECF-100HR3-11-0114). Hexavalent chromium in the treatment system influent will be removed using ion exchange treatment technology. The other COCs in the resulting treatment system effluent injected into the aquifer remains below cleanup levels. Nitrate contaminated groundwater is within the capture footprint of the Cr(VI) pump-and-treat system well network. The groundwater model simulation for this alternative indicates that nitrate contamination is remediated to cleanup levels within the Cr(VI) remediation period. Nitrate plume depictions after 3, 5, and 6 years of operating under this alternative are provided in Figure 9-13 (a-c) showing nitrate concentrations are below cleanup levels after 6 years.

Strontium-90 has low solubility and mobility in groundwater and small pockets of strontium-90 contamination remain in the aquifer following completion of the Cr(VI) remediation. The pump-and-treat systems will be shut down following completion of Cr(VI) remediation. Remaining strontium-90 contamination will attenuate to cleanup levels through radiological decay in 32 years after Cr(VI) remediation (i.e., 44 years after implementation of the Alternative 3 remedy). Strontium-90 plume depictions after 3, 13, 38, and 44 years of operating under this alternative are provided in Figure 9-14 (a-d).

9.2.5 Alternative 4—RTD for Waste Sites and Pump-and-Treat for Groundwater

Alternative 4 uses RTD for waste site contamination removal to achieve the RAOs, with the greatest degree of certainty, and pump-and-treat for groundwater. Table 9-10 presents the waste site and groundwater components of this alternative, and Figure 9-15 presents a pictorial summary of Alternative 4. The cost for this alternative is provided in Table 9-11. Details for the cost estimate are provided in Appendix J (*Environmental Cost Estimate for 100-D/H Vadose Zone and Groundwater RI/FS* [ECE-100HR311-00004]). Additional details on remedial components specific to this alternative are presented in sections identified in Table 9-10.

9.2.5.1 Groundwater Model for Alternative 4

The groundwater model simulation for Alternative 4 expands the interim action pump-and-treat system to include additional extraction and injection wells, thus encompassing a larger area of the COC plumes to expedite hydraulic containment and recovery. This alternative expands the pump-and-treat system with 20 new or converted wells for extraction and injection associated with DX, and 24 new or converted wells for extraction and injection at 100-H. The duration of well operation is detailed in *Modeling of RI/FS Design Alternatives for 100-HR-3* (ECF-100HR3-11-0114, Table 3-6).

Alternative 4 is designed to operate until the Cr(VI) groundwater plume is substantially reduced to less than the groundwater cleanup level in MTCA ("Groundwater Cleanup Standards" [WAC 173-340-720]). Wells along the river will be used to contain concentrations in the plume that are above 10 µg/L, thereby reducing migration to the river, so the AWQC can be maintained at points of groundwater discharge to the river. The model assumes 85 percent availability of the pump-and-treat system capacity to account for scheduled or unforeseen shutdown periods. As a result, a total capacity of 1,930 L/min (510 gpm) was assumed for the DX pump-and-treat system and 2,575 L/min (680 gpm) for the HX pump-and-treat system.

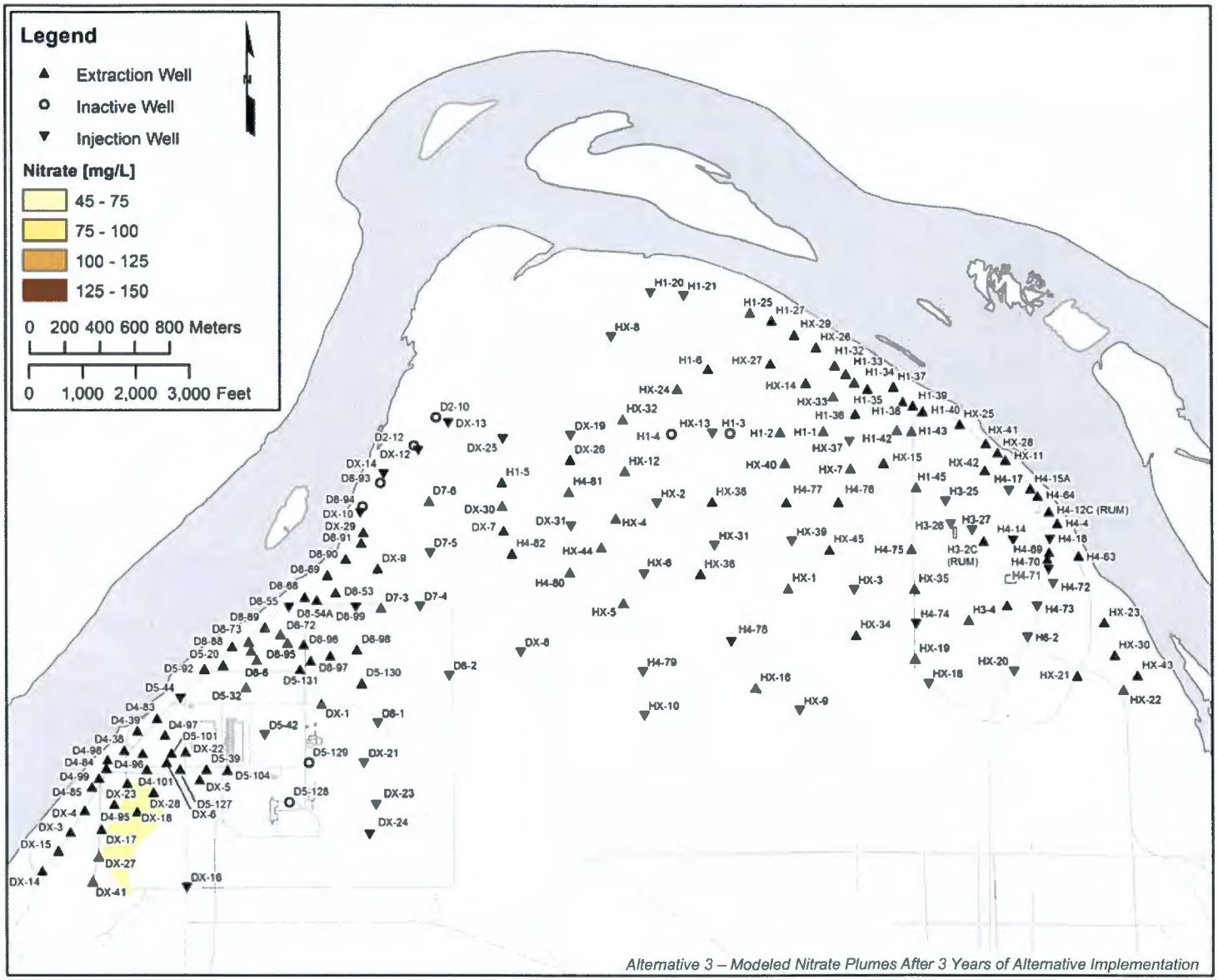

Figure 9-13a. Alternative 3—Modeled Nitrate Plumes After 3 Years

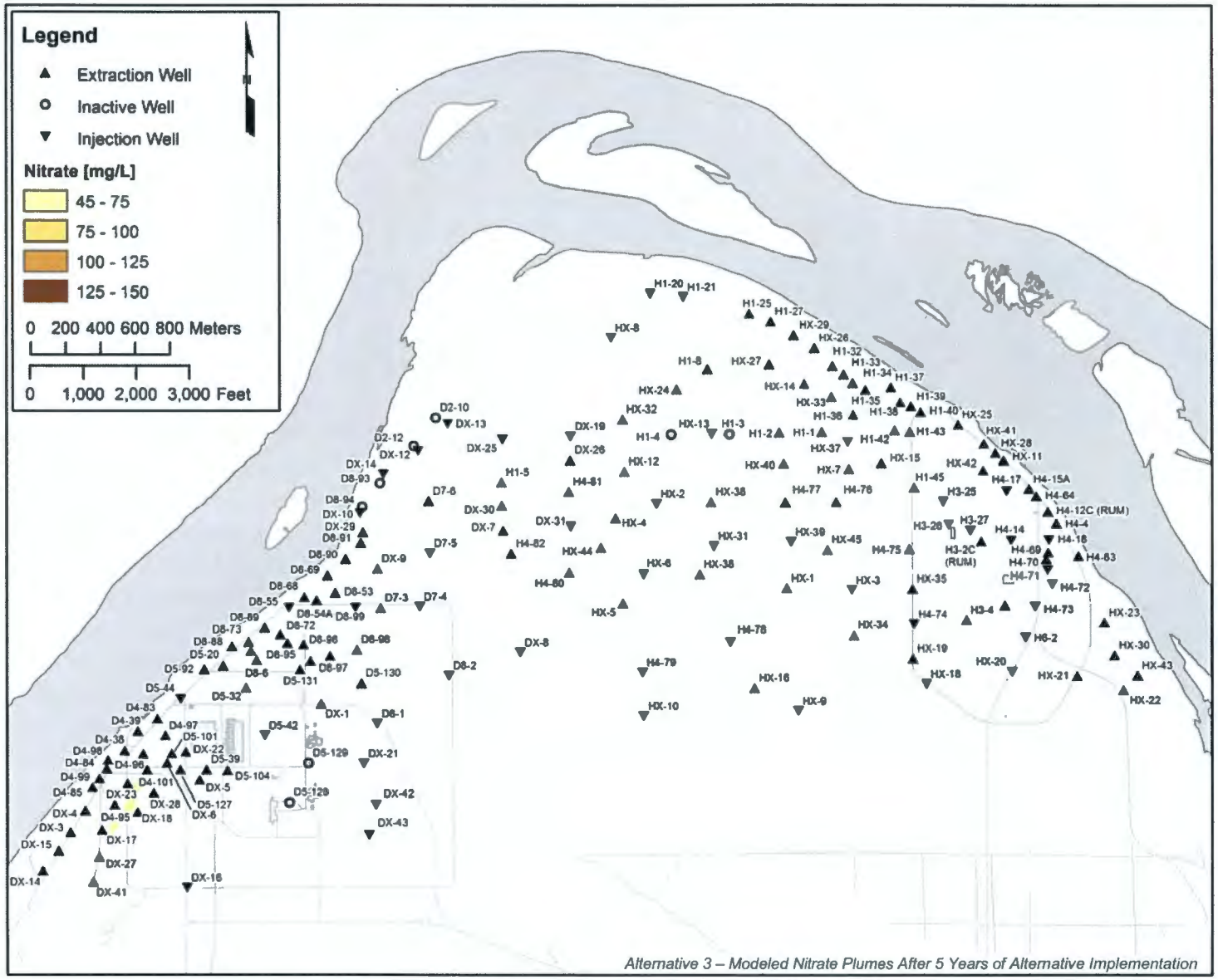

Figure 9-13b. Alternative 3—Modeled Nitrate Plumes After 5 Years

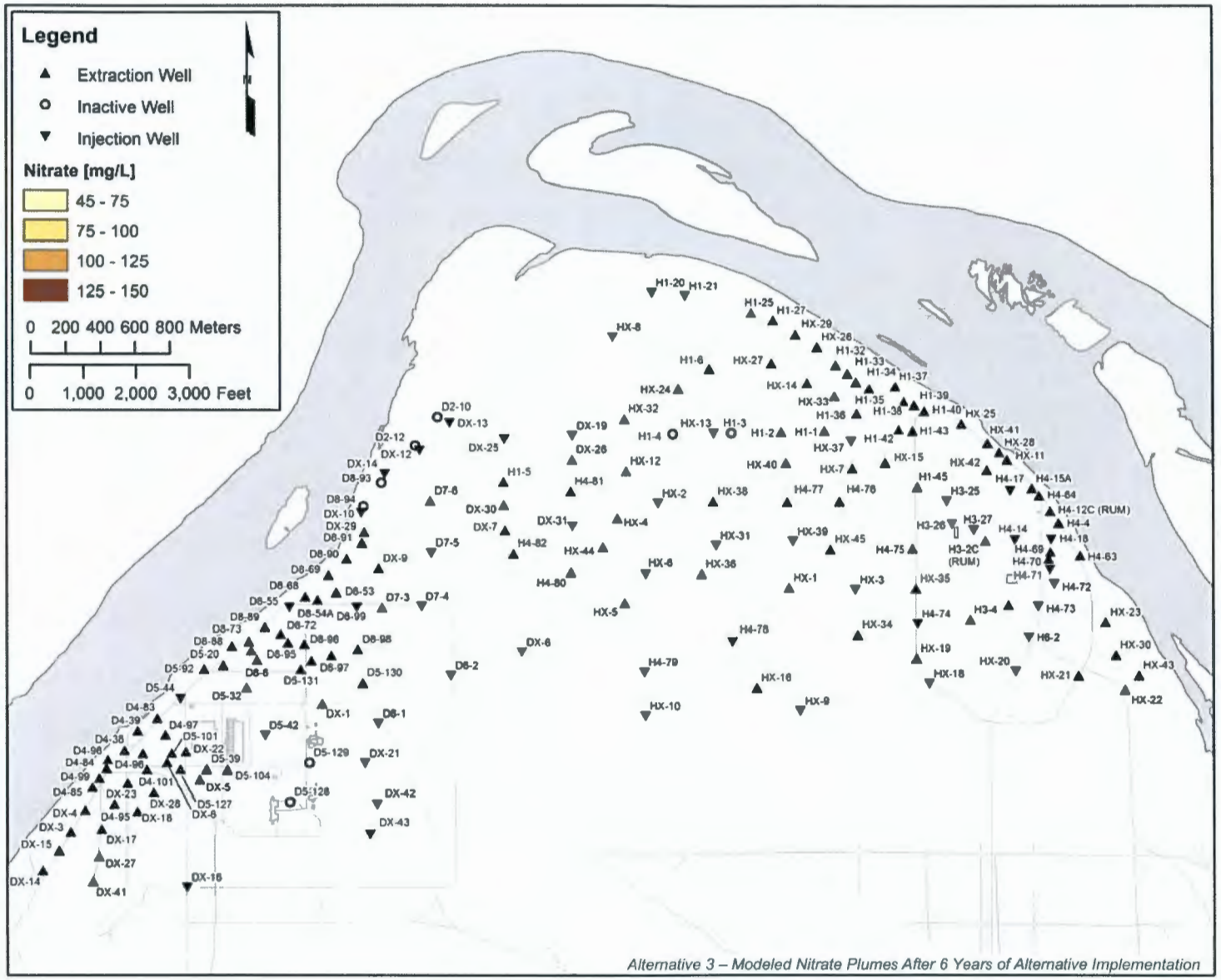

Figure 9-13c. Alternative 3—Modeled Nitrate Plumes After 6 Years

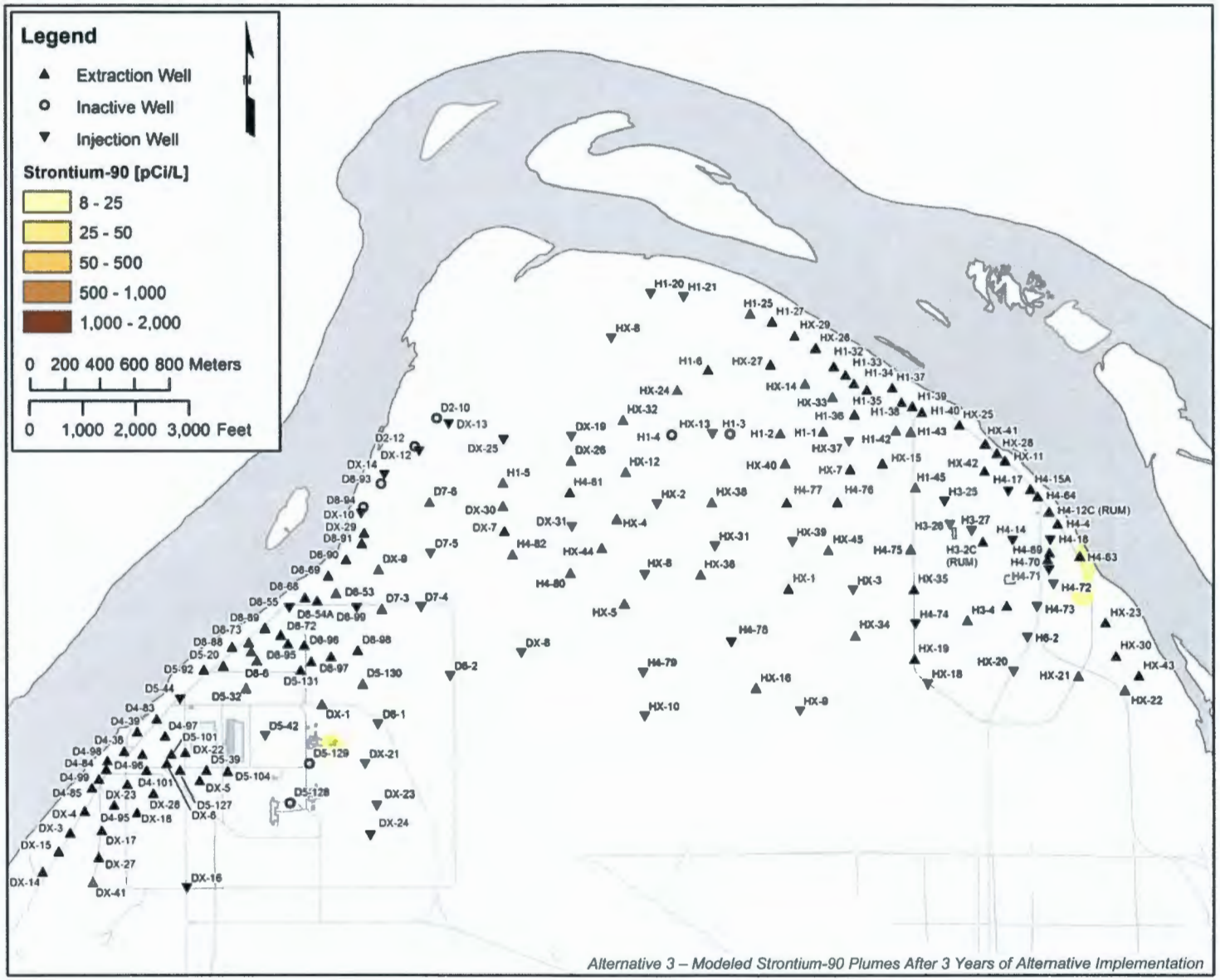

Figure 9-14a. Alternative 3—Modeled Strontium-90 Plumes After 3 Years

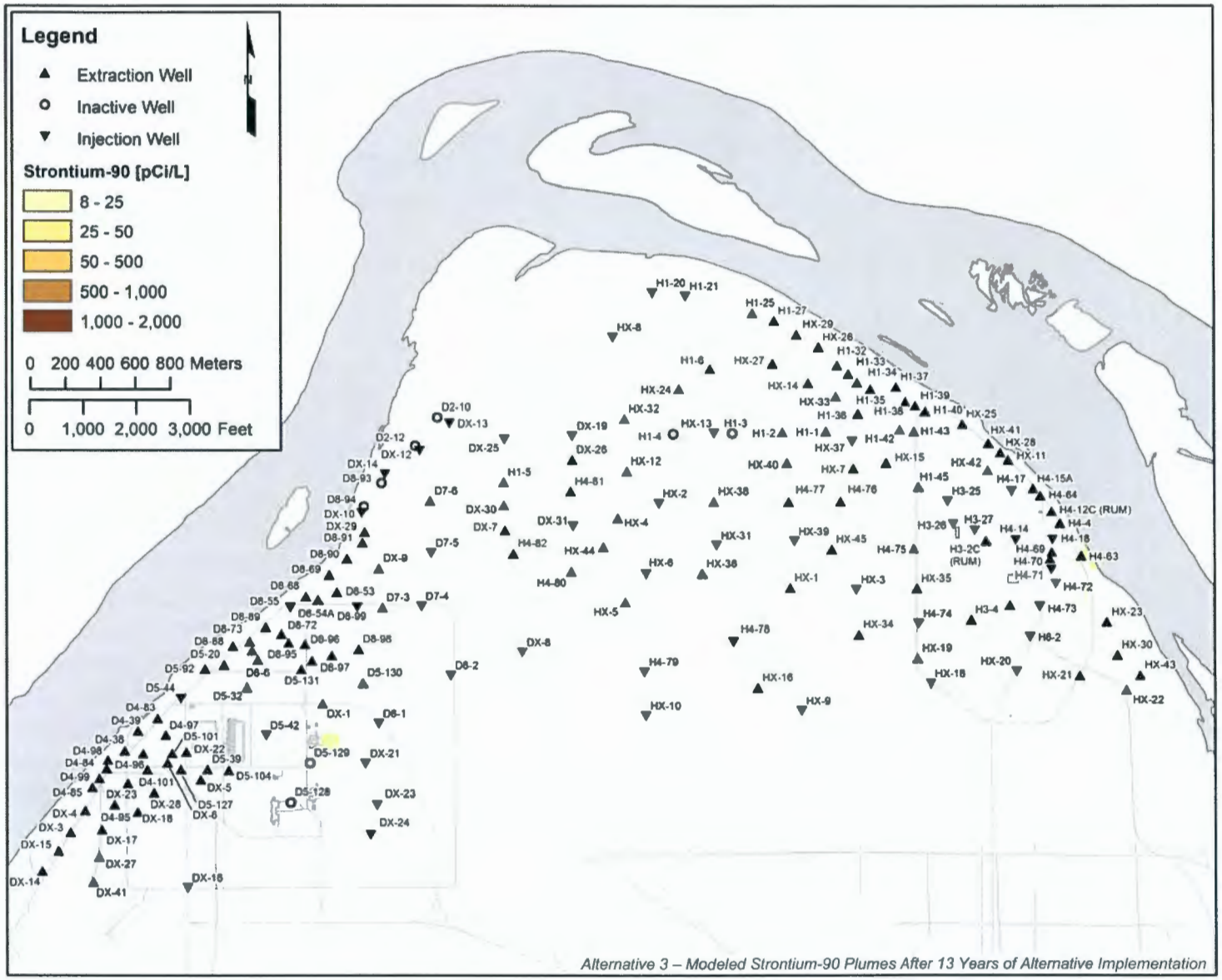

Figure 9-14b. Alternative 3—Modeled Strontium-90 Plumes After 13 Years

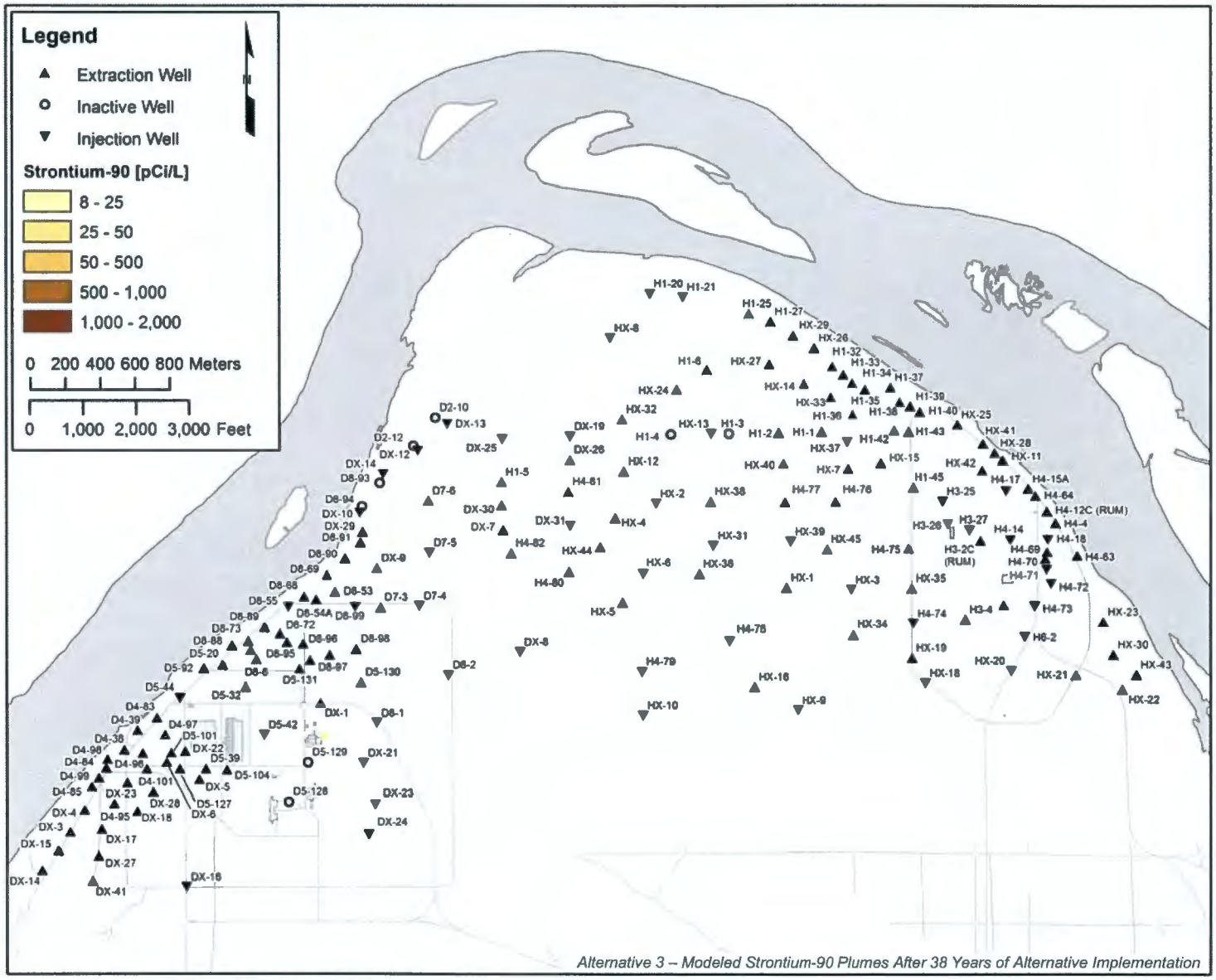

Figure 9-14c. Alternative 3—Modeled Strontium-90 Plumes After 38 Years

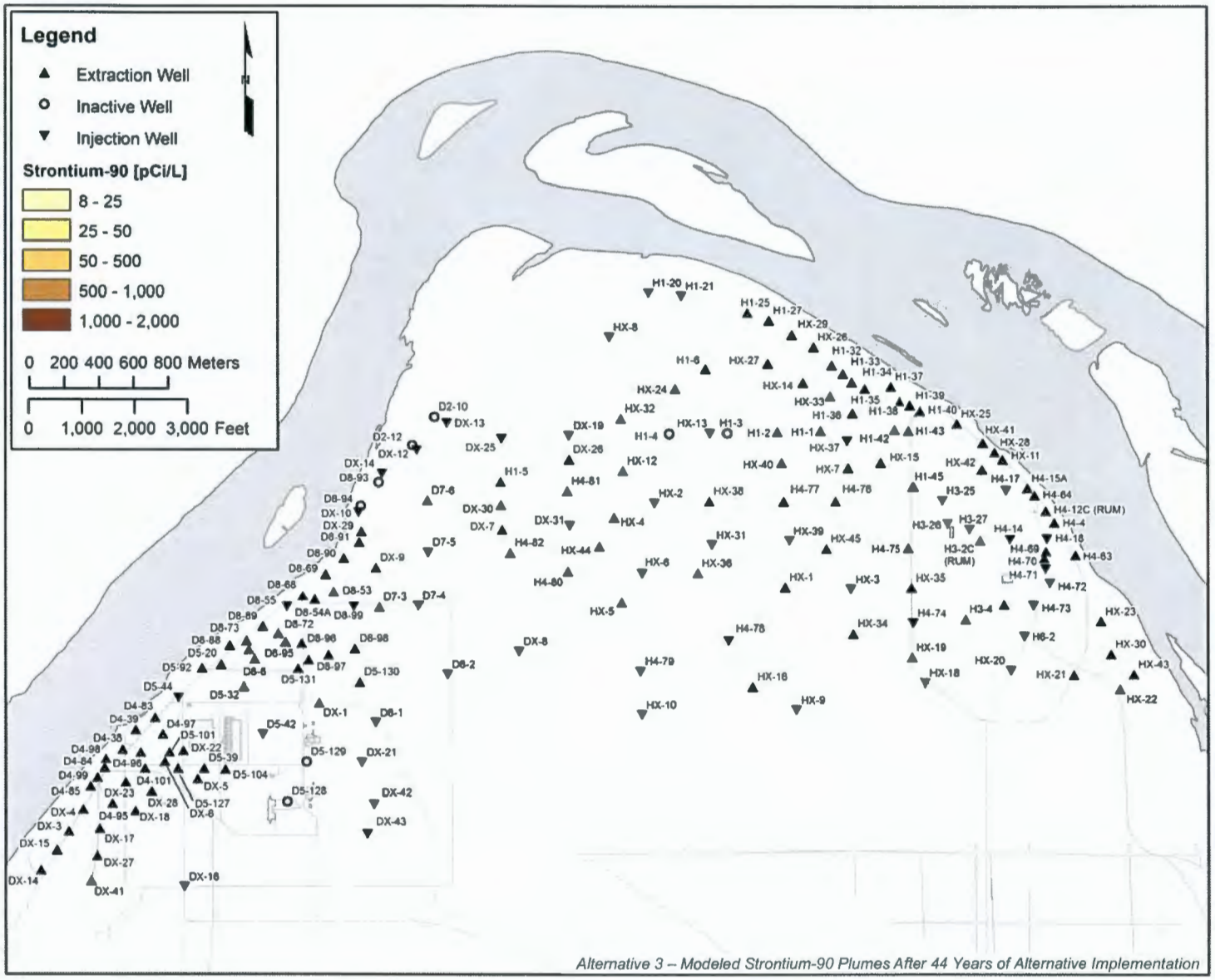

Figure 9-14d. Alternative 3—Modeled Strontium-90 Plumes After 44 Years

Table 9-10. Components for Alternative 4—RTD for Waste Sites and Pump-and-Treat for Groundwater

Waste Site Components	General Components	
	No Further Action	No additional remedial actions are taken for the 146 waste sites identified in Section 8.2.1.1 (Chapter 8).
	Institutional Controls	Institutional controls to be implemented within 100-D, 100-H, and horn areas for land use management and waste site information management include the following: • Permits required for excavations on the Hanford Site to prevent unplanned disturbance or infiltration as prohibited by CERCLA decision documents. • Land use and real property controls (for example, proprietary controls including easements and covenants) ensure that the use of land is in accordance with Hanford Site plans and CERCLA decision documents. • Notices providing visual identification and warning of hazardous or sensitive areas. • Procedural requirements for access, warning signs, or fencing implemented to prevent or limit the access of humans to hazardous or sensitive areas. • Administrative mechanisms, such as the WIDS database, to maintain and provide access to information on the location and nature of contamination. Additional information on institutional controls is presented in Section 8.3.1 (Chapter 8). Post-remediation institutional controls implemented at specific waste sites are identified in Table 9-4.
	For Waste Sites That Exceed Human Health, Environment, Surface Water, or Groundwater Protection PRGs	
	RTD	RTD to depth of contamination exceeding cleanup levels, or until groundwater is encountered if contamination extends into groundwater, using standard and deep excavation technologies, as described in Section 9.2.2.2. RTD is applied to waste sites listed in Table 8-5 (Chapter 8) that are expected to be remediated under the 100 Area Remaining Sites ROD (EPA/ROD/R10-99/039). RTD is effective in meeting the human health, environment, surface water, and groundwater PRGs as reflected in the risk evaluation of completed waste sites with closeout/ verification data (Chapter 6).

Table 9-10. Components for Alternative 4—RTD for Waste Sites and Pump-and-Treat for Groundwater

Waste Site Components (cont.)	RTD (cont.)	RTD also applied to waste sites listed in Table 8-5 (Chapter 8) that are yet to be remediated and not remediated under one of the other waste site remedial components listed above. Table 9-2 identifies applicable waste sites remediated by RTD using standard and deep excavation technologies. Within the SMDP process, ecological PRGs will be considered at a population level for wildlife and at a community level for plants and invertebrates, to determine whether cleanup action is required to protect ecological receptors.
Groundwater Components	Pump-and-Treat System	Operation of existing pump-and-treat systems as described in Section 9.2.2.4, with additional extraction and injection wells to expand treatment coverage. Placement of additional extraction and injection wells, extraction and injection flow rates, and well operational periods were determined through groundwater model simulations described in *Modeling of RI/FS Design Alternatives for 100-HR-3* (ECF-100HR3-11-0114). The RPO process will be used to provide ongoing evaluations to ensure the system meets performance requirements that will be defined in the RD/RAWP. Designed with active remediation out to Cr(VI) 10 µg/L contour. Treatment of extracted water through ion exchange. Operation of pump-and-treat system components to meet AWQC at points of groundwater discharge to the river.
	MNA and Institutional Controls	MNA as described in Section 9.2.2.3. Manage strontium-90 in groundwater through a combination of treatment, MNA, and institutional controls until concentrations meet cleanup standards. Manage the plumes until concentrations are below the action levels and after the pump-and-treat system is shut off. Monitoring of COCs to track the attenuation processes. Institutional controls to be implemented within 100-D/H include land use management and waste site information management. Additional institutional controls for groundwater include: • Permits required for excavations on the Hanford Site to prevent unplanned disturbance or infiltration as prohibited by CERCLA decision documents.

Table 9-10. Components for Alternative 4—RTD for Waste Sites and Pump-and-Treat for Groundwater

Groundwater Components (cont.)	MNA and Institutional Controls (cont.)	• Land use and real property controls (for example, proprietary controls including irrigation restrictions, easements, and covenants) ensure that the use of land is in accordance with Hanford Site plans and CERCLA decision documents. • Groundwater use management, as described in *Sitewide Institutional Controls Plan for Hanford CERCLA Response Actions* (DOE/RL-2001-41), ensures proper use of groundwater through groundwater controls. • Administrative mechanisms, such as the WIDS database, to maintain and provide access to information on the location and nature of contamination. Additional information on institutional controls is presented in Section 8.3.1 (Chapter 8).
	Monitoring Requirements	Remedy performance monitoring will be defined as part of the remedial design. Evaluates the effectiveness of the selected alternative to attain the cleanup levels. The existing monitoring program will expand when a number of monitoring wells are added. For cost estimating purposes, it has been assumed that 12 new monitoring wells will be installed. Groundwater performance monitoring is presented in Section 9.2.2.7 and identifies constituents included in the monitoring program.

All waste sites will be remediated before final remediation of groundwater. Remedial action completion for waste site and groundwater remediation is described in Section 9.2.

Results from the groundwater flow and transport model (*Modeling of RI/FS Design Alternatives for 100-HR-3* [ECF-100HR3-11-0114]) predict COC plumes and trends over the remediation simulation period. The groundwater model results predict Cr(VI) plumes will be remediated to meet cleanup levels to 48 µg/L DWS and 10 µg/L AWQC throughout the aquifer after 11 and 39 years, respectively, of implementing the groundwater remedy under Alternative 4. Figure 9-16 (a-f) shows the modeled Cr(VI) groundwater plumes after 3, 8, 18, 25, 33, and 39 years of operating under this alternative. Figure 9-16d shows that after 25 years, the majority of Cr(VI) contamination has been remediated and only one small area in the Horn remains above 10 µg/L. It is very likely this area would be remediated within the 25-year period with adjustments to the pumping rates. RPO activities will be conducted throughout the life of the project to deal with the uncertainty in groundwater modeling simulations.

EPC calculations will be used to determine completion of groundwater remedial actions. Model predictions include statistical evaluations to estimate EPCs for the following concentration values: maximum encountered; 90th percentile; mean; and Cr(VI) median concentration trends for the 100-D south, 100-D north, horn area, and 100-H, and along the river shoreline. The EPC trends for this alternative are included in *Modeling of RI/FS Design Alternatives for 100-HR-3* (ECF-100HR3-11-0114).

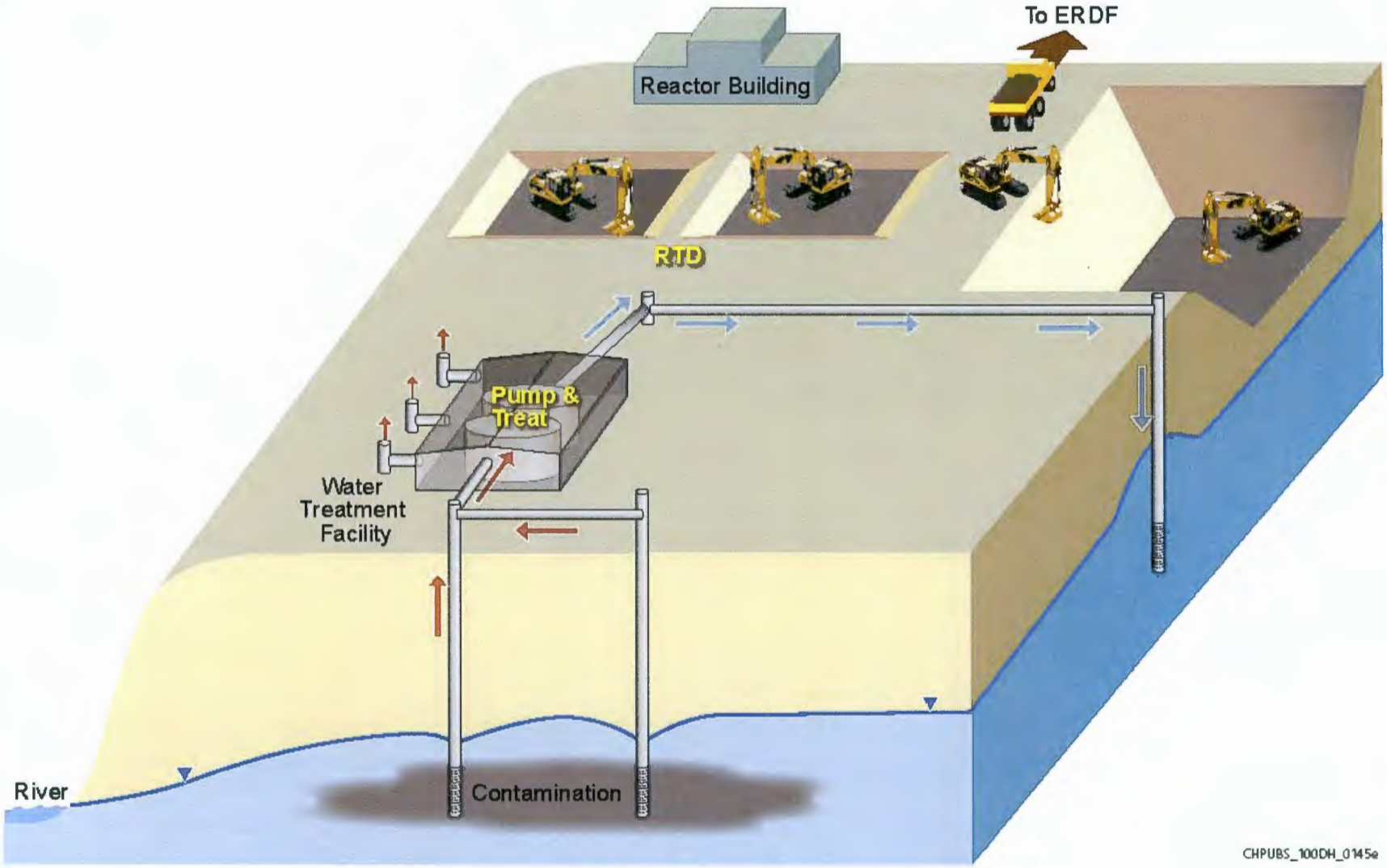

Figure 9-15. Alternative 4—RTD for Waste Sites and Pump-and-Treat for Groundwater

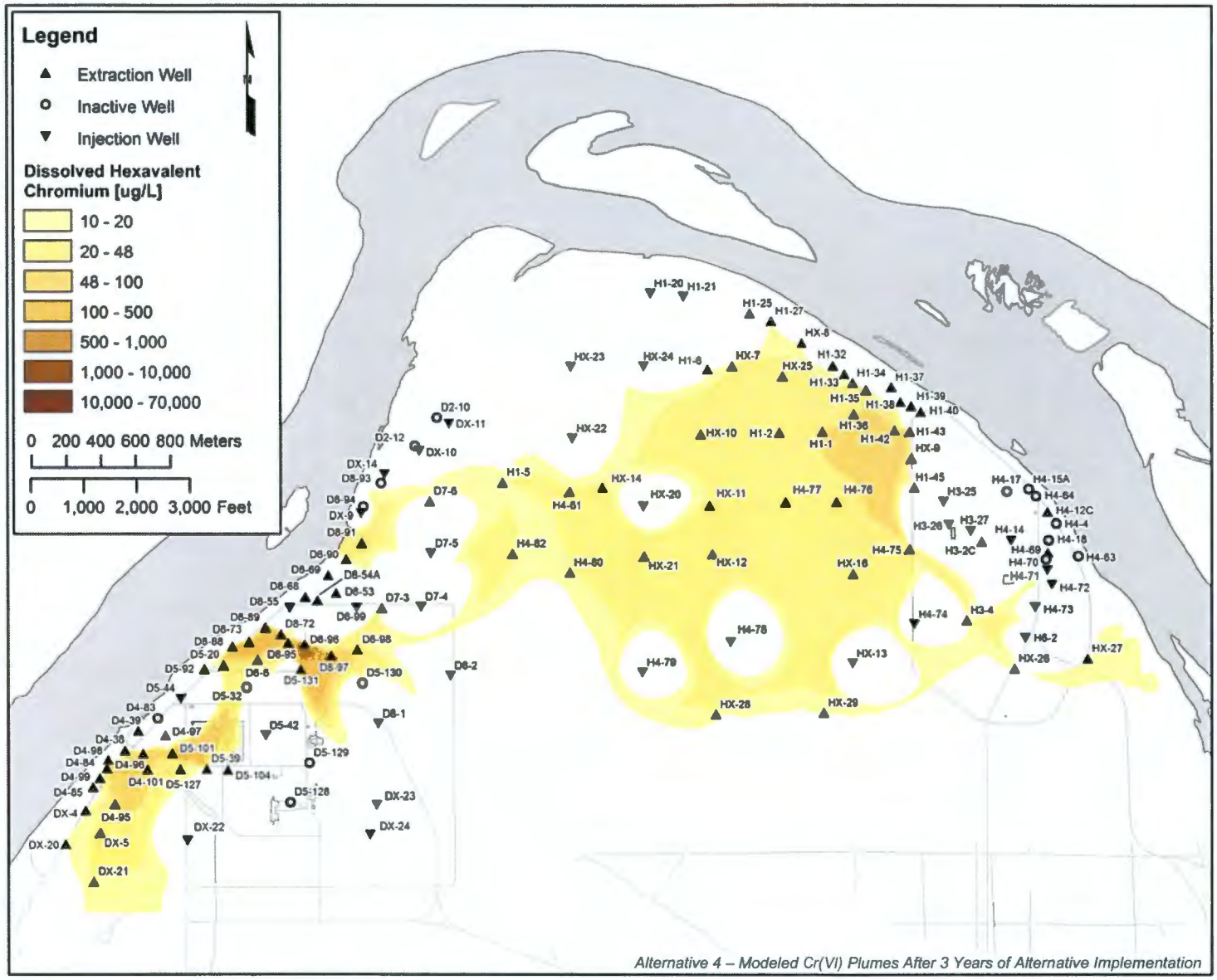

Figure 9-16a. Alternative 4—Modeled Cr(VI) Plume After 3 Years

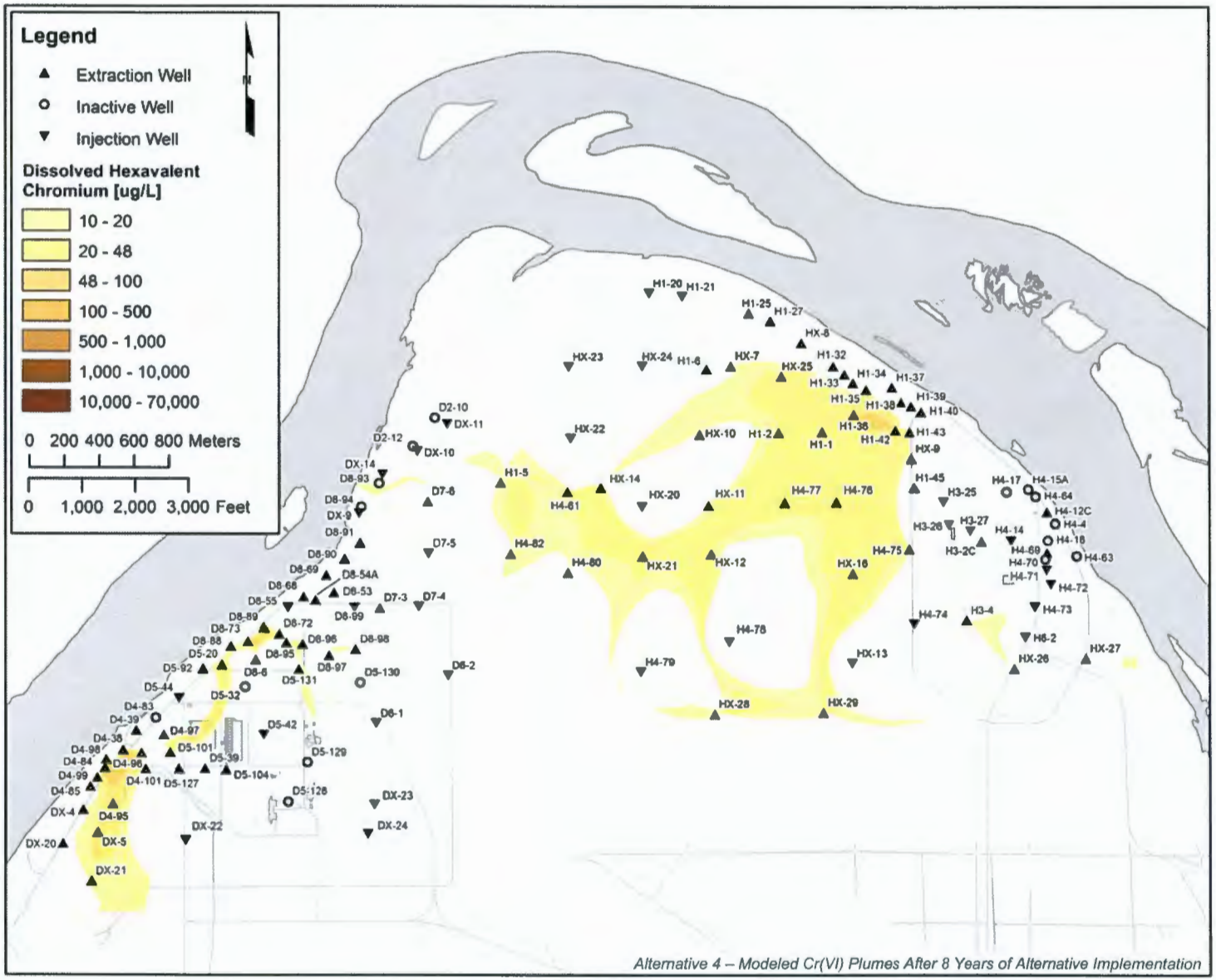

Figure 9-16b. Alternative 4—Modeled Cr(VI) Plume After 8 Years

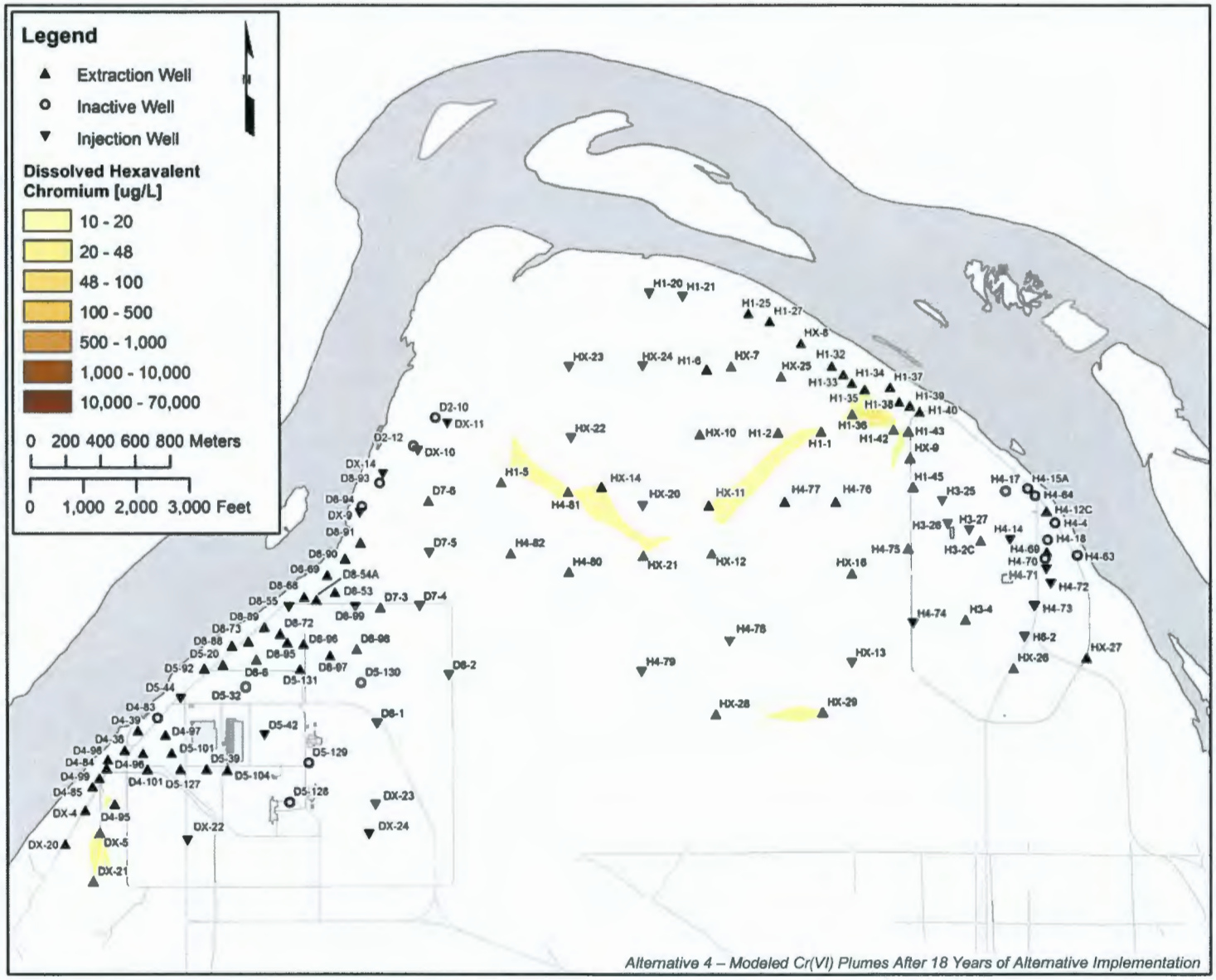

Figure 9-16c. Alternative 4—Modeled Cr(VI) Plume After 18 Years

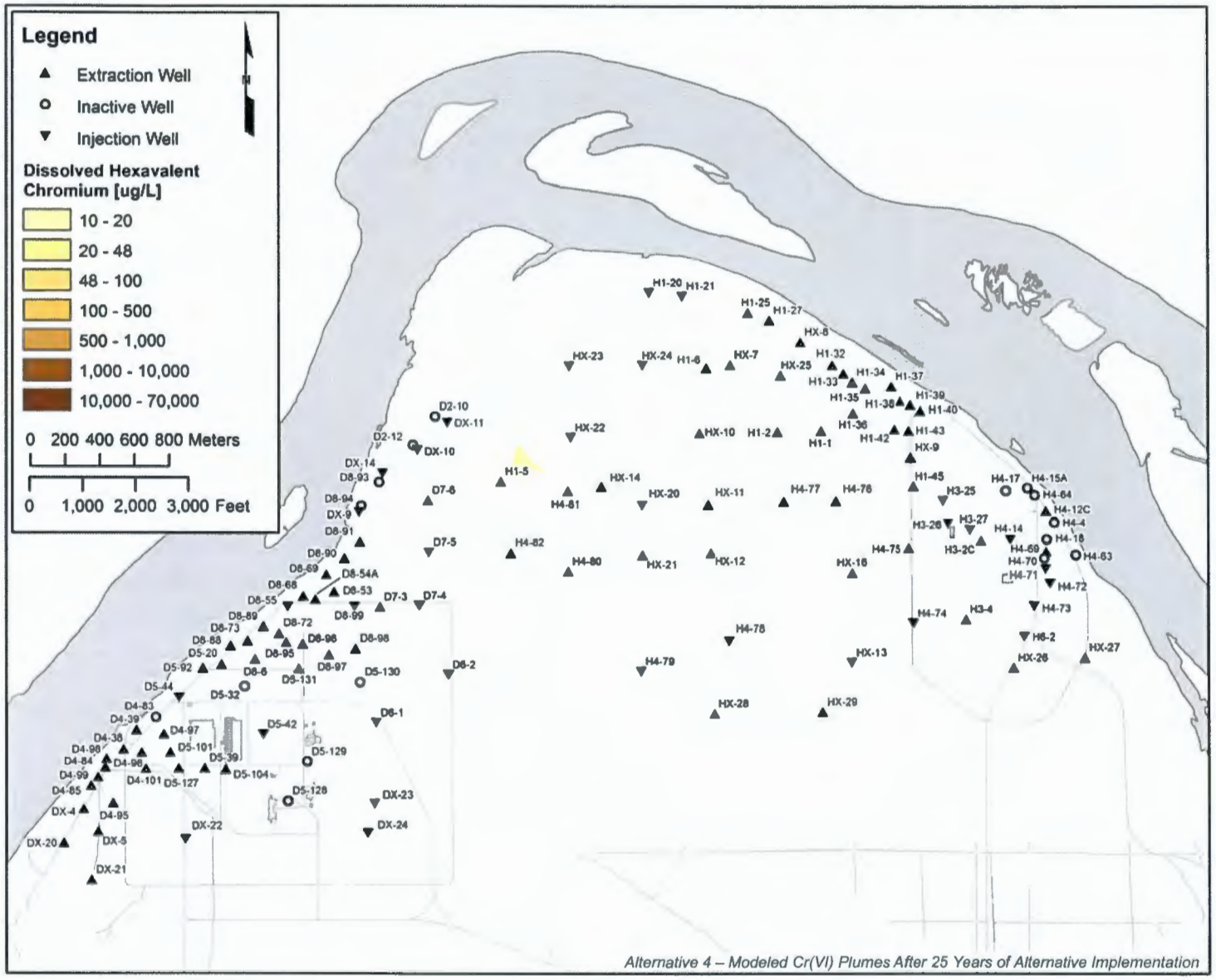

Figure 9-16d. Alternative 4—Modeled Cr(VI) Plume After 25 Years

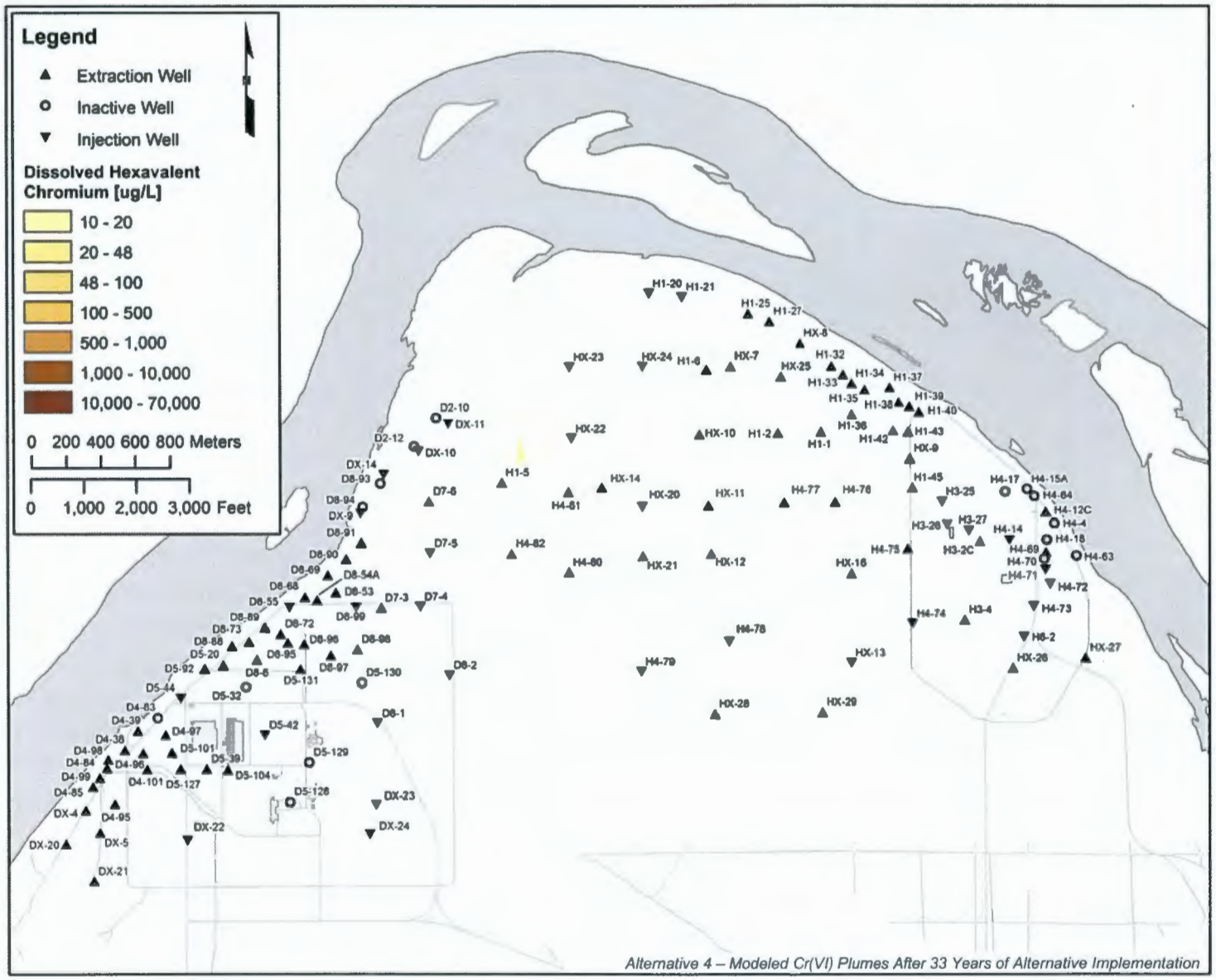

Figure 9-16e. Alternative 4—Modeled Cr(VI) Plume After 33 Years

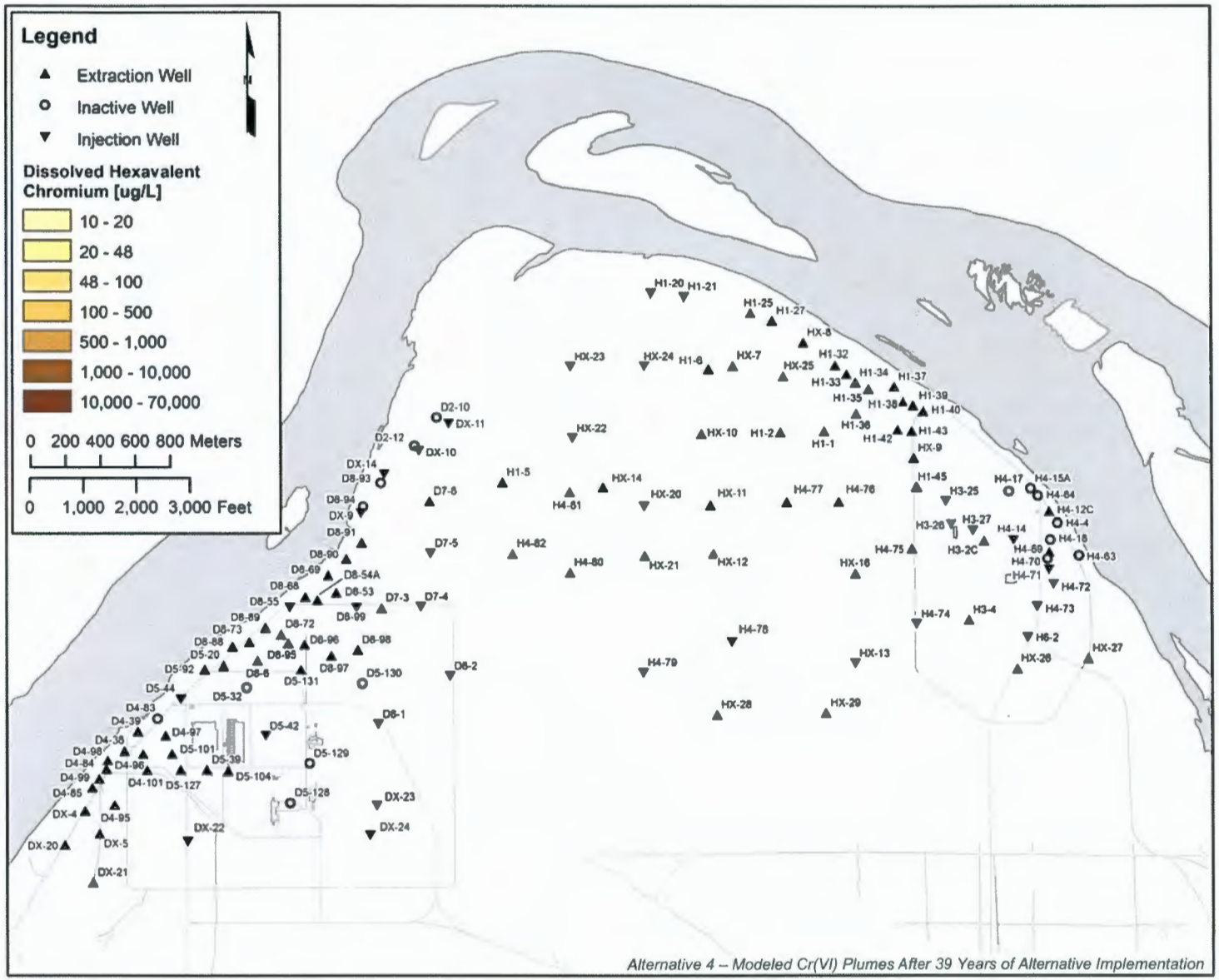

Figure 9-16f. Alternative 4—Modeled Cr(VI) Plume After 39 Years

As described in Section 9.2, groundwater COCs co-extracted for treatment of the Cr(VI) plumes are managed through MNA and ICs to achieve cleanup levels. Predicted concentrations of the nitrate and strontium-90 COCs in the combined groundwater influent stream to the pump-and-treat facilities will be at concentrations below their respective cleanup levels. Predicted concentration trends for each COC in the combined influent to the pump-and-treat facilities are provided in *Modeling of RI/FS Design Alternatives for 100-HR-3* (ECF-100HR3-11-0114). Hexavalent chromium in the treatment system influent will be removed using IX treatment technology. The other COCs in the resulting treatment system effluent injected into the aquifer remain below cleanup levels. Nitrate contaminated groundwater is within the capture footprint of the Cr(VI) pump-and-treat system well network. The groundwater model simulation for this alternative indicates that nitrate contamination is remediated to cleanup levels earlier than the Cr(VI) remediation period since nitrate contamination is not present in the horn. Nitrate plume depictions after 3, 8, 12, and 13 years of operating under this alternative are provided in Figure 9-17 (a-d) showing nitrate concentrations are below cleanup levels after 13 years.

Strontium-90 has low solubility and mobility in groundwater and small pockets of strontium-90 contamination remain in the aquifer following completion of the Cr(VI) remediation. The pump-and-treat systems will be shut down following completion of Cr(VI) remediation. Remaining strontium-90 contamination will attenuate to cleanup levels through radiological decay in 17 years after Cr(VI) remediation (i.e., 56 years after implementation of the Alternative 4 remedy). Strontium-90 plume depictions after 3, 18, 48, and 56 years of operating under this alternative are provided in Figure 9-18 (a-d).

9.3 Remedial Alternative Screening Evaluation

As discussed in EPA guidance (*The Feasibility Study: Development and Screening of Remedial Action Alternatives* [OSWER Directive 9355.3-01FS3]), screening of alternatives is not a required step because only three alternatives are being evaluated. Consequently, they are all carried into the detailed evaluation in Chapter 10.

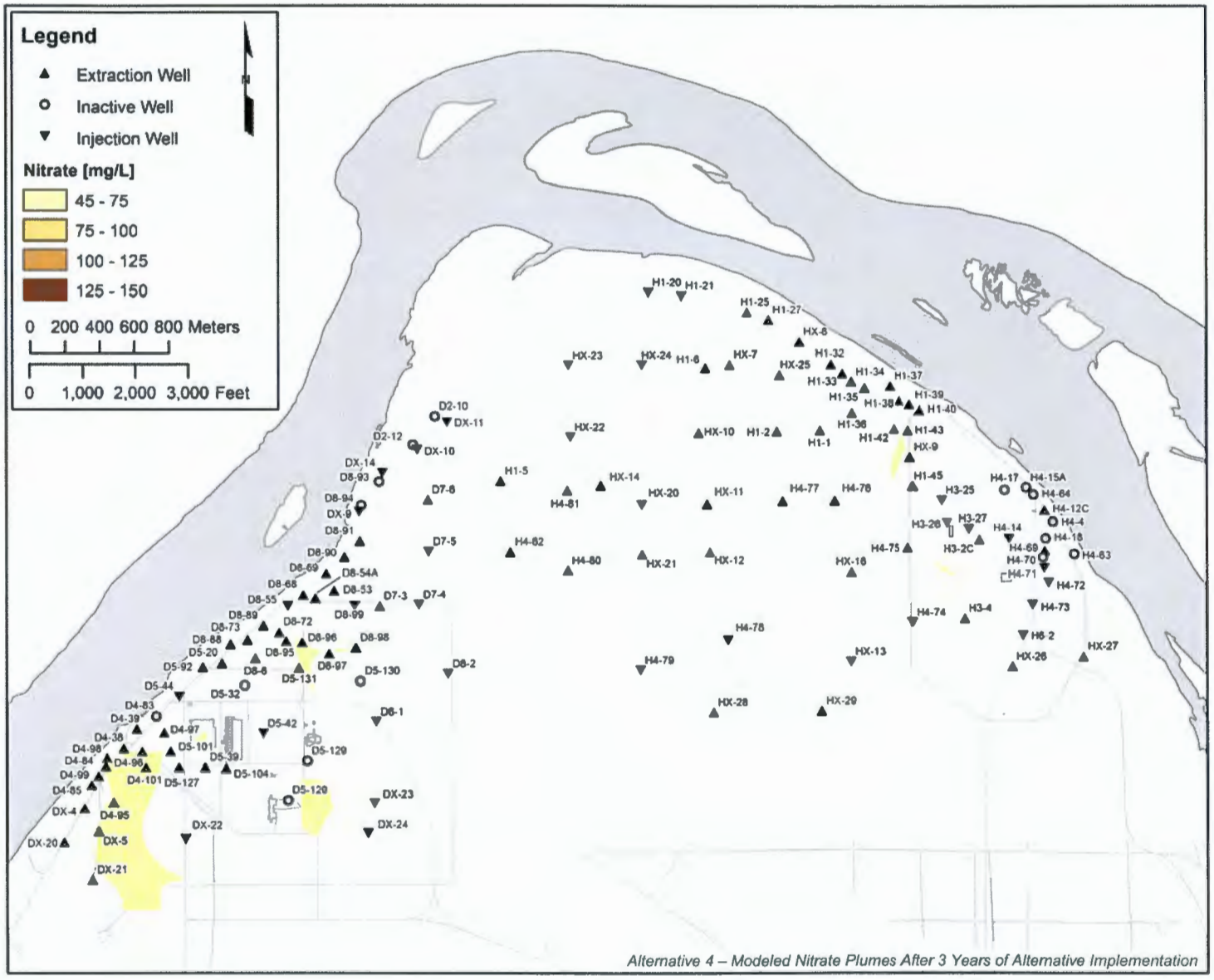

Figure 9-17a. Alternative 4—Modeled Nitrate Plume After 3 Years

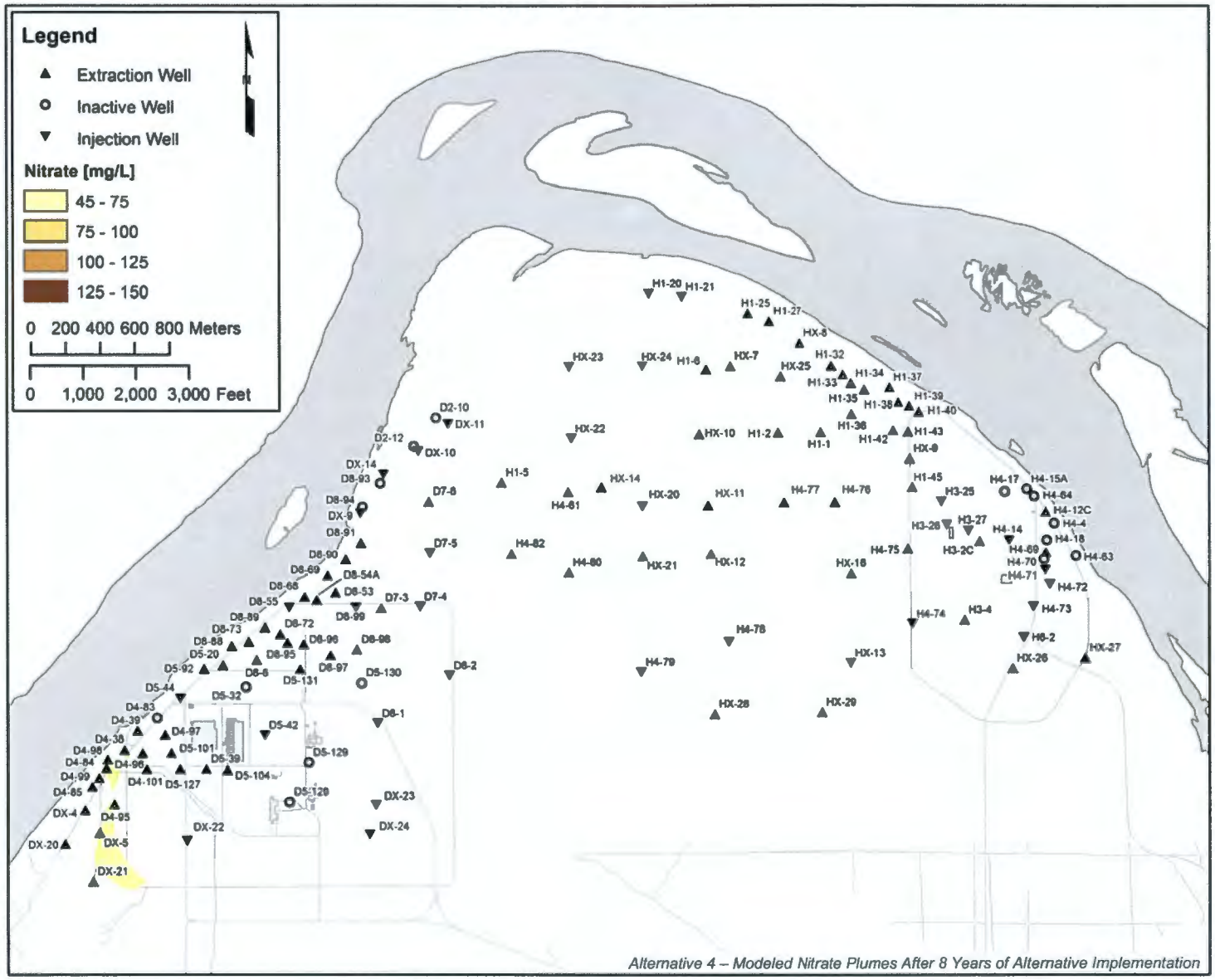

Figure 9-17b. Alternative 4—Modeled Nitrate Plume After 8 Years

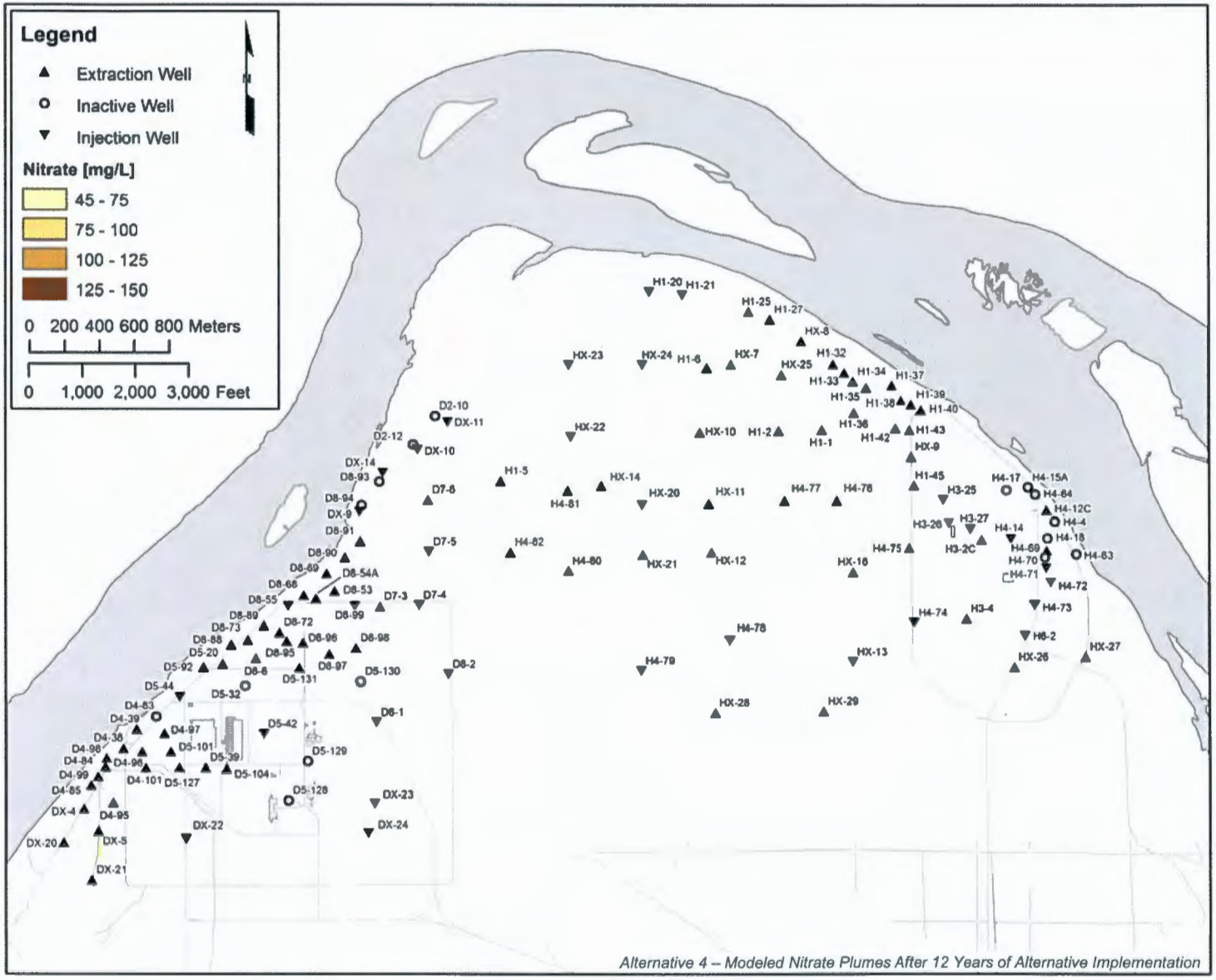

Figure 9-17c. Alternative 4—Modeled Nitrate Plume After 12 Years

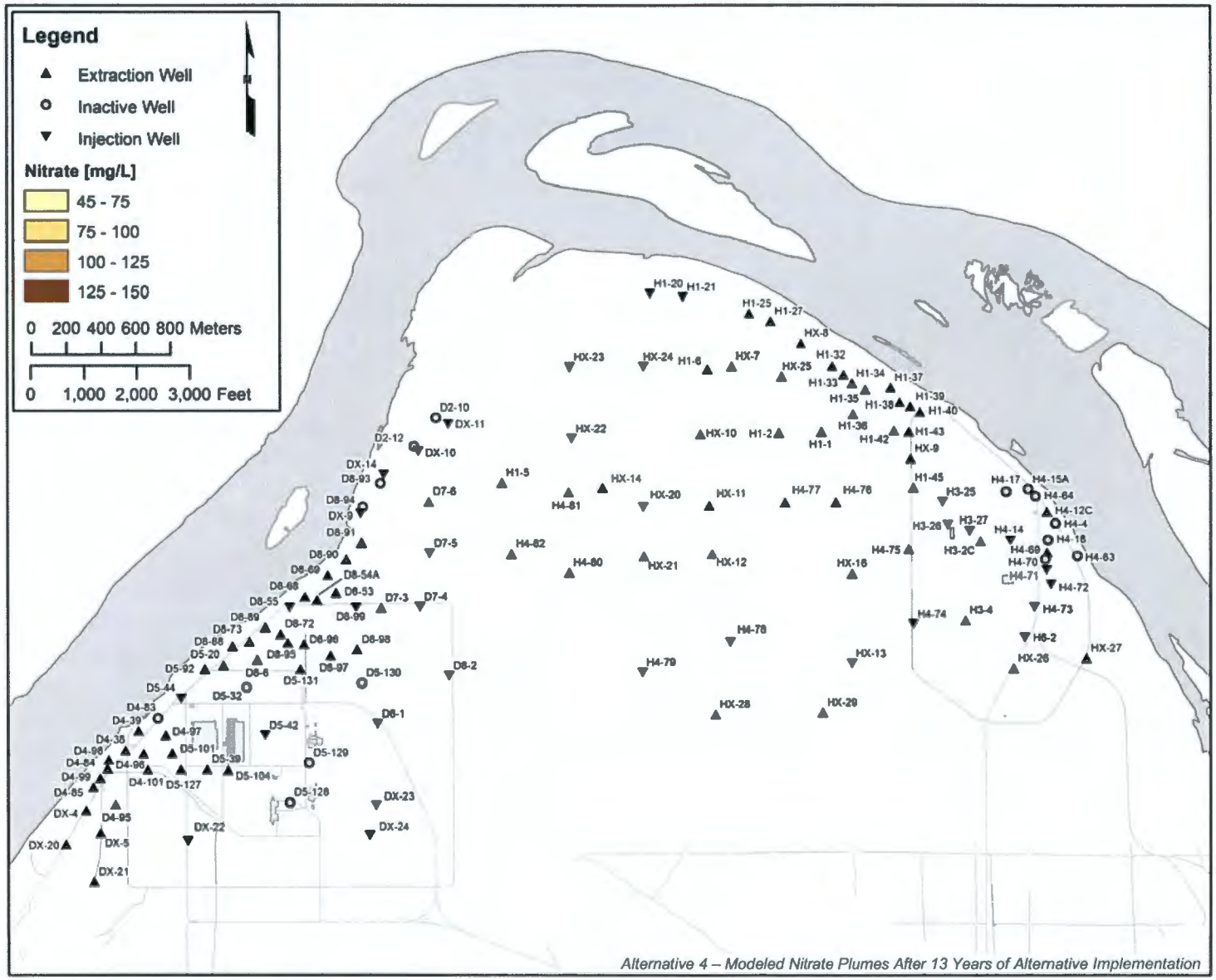

Figure 9-17d. Alternative 4—Modeled Nitrate Plume After 13 Years

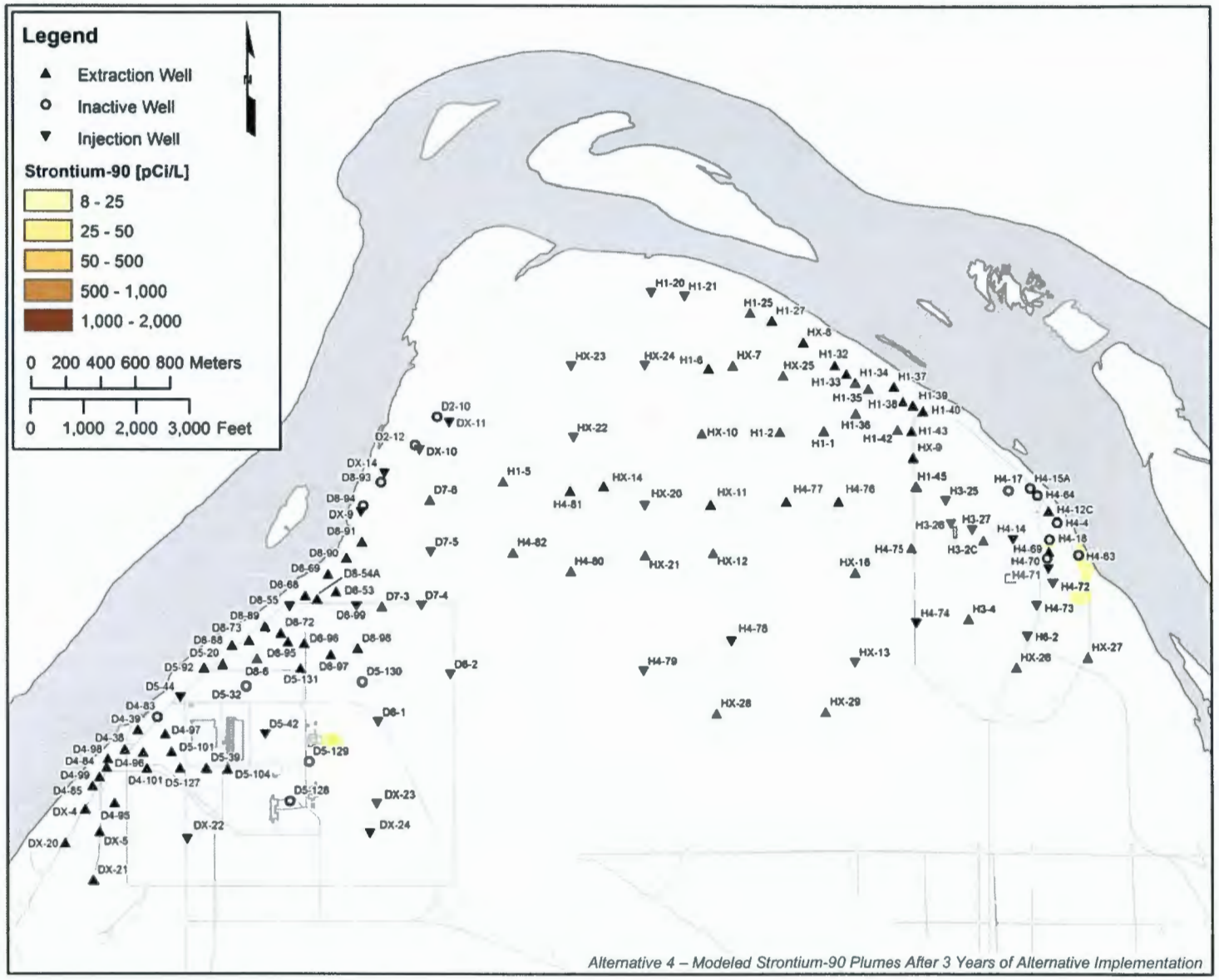

Figure 9-18a. Alternative 4—Modeled Strontium-90 Plume After 3 Years

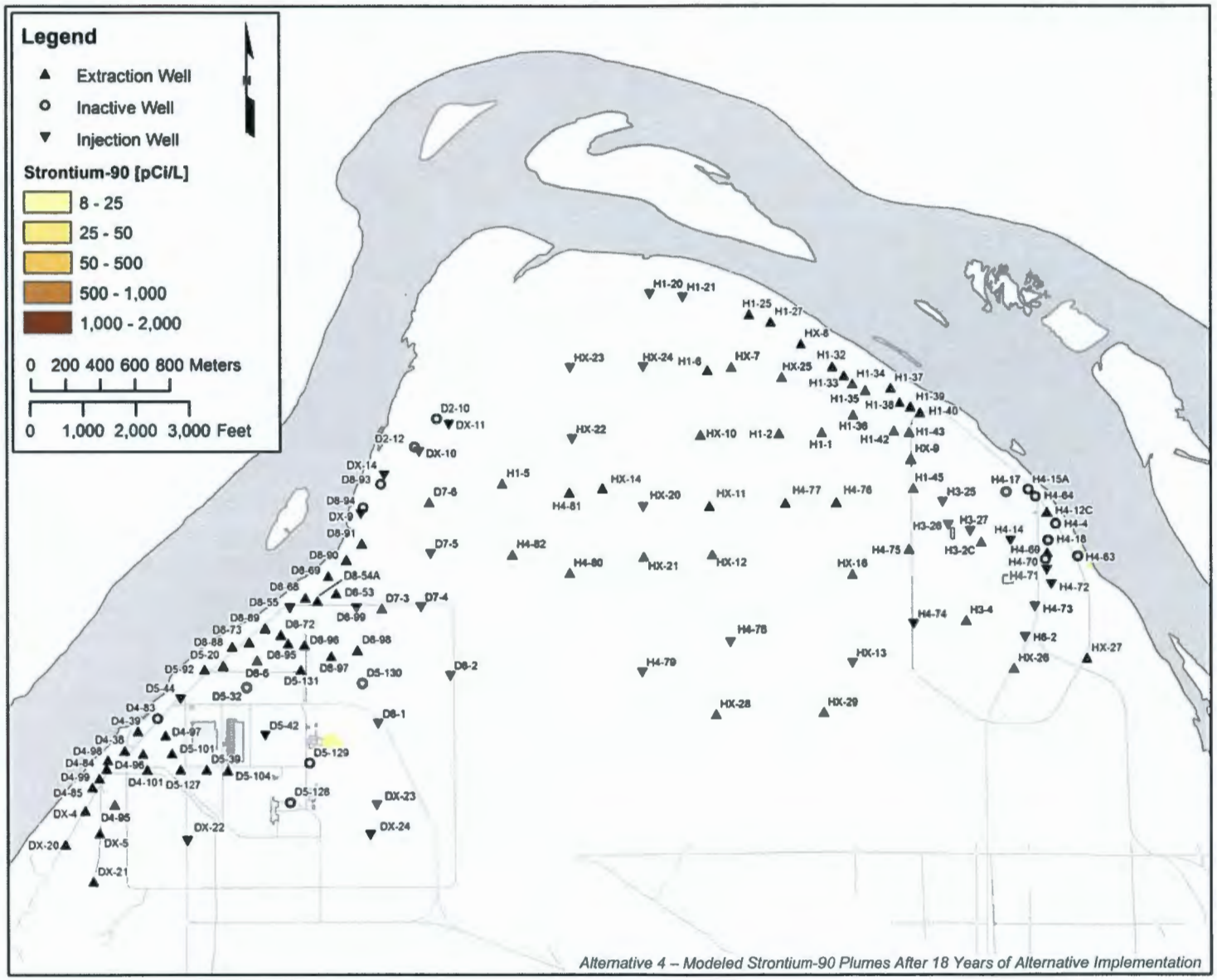

Figure 9-18b. Alternative 4—Modeled Strontium-90 Plume After 18 Years

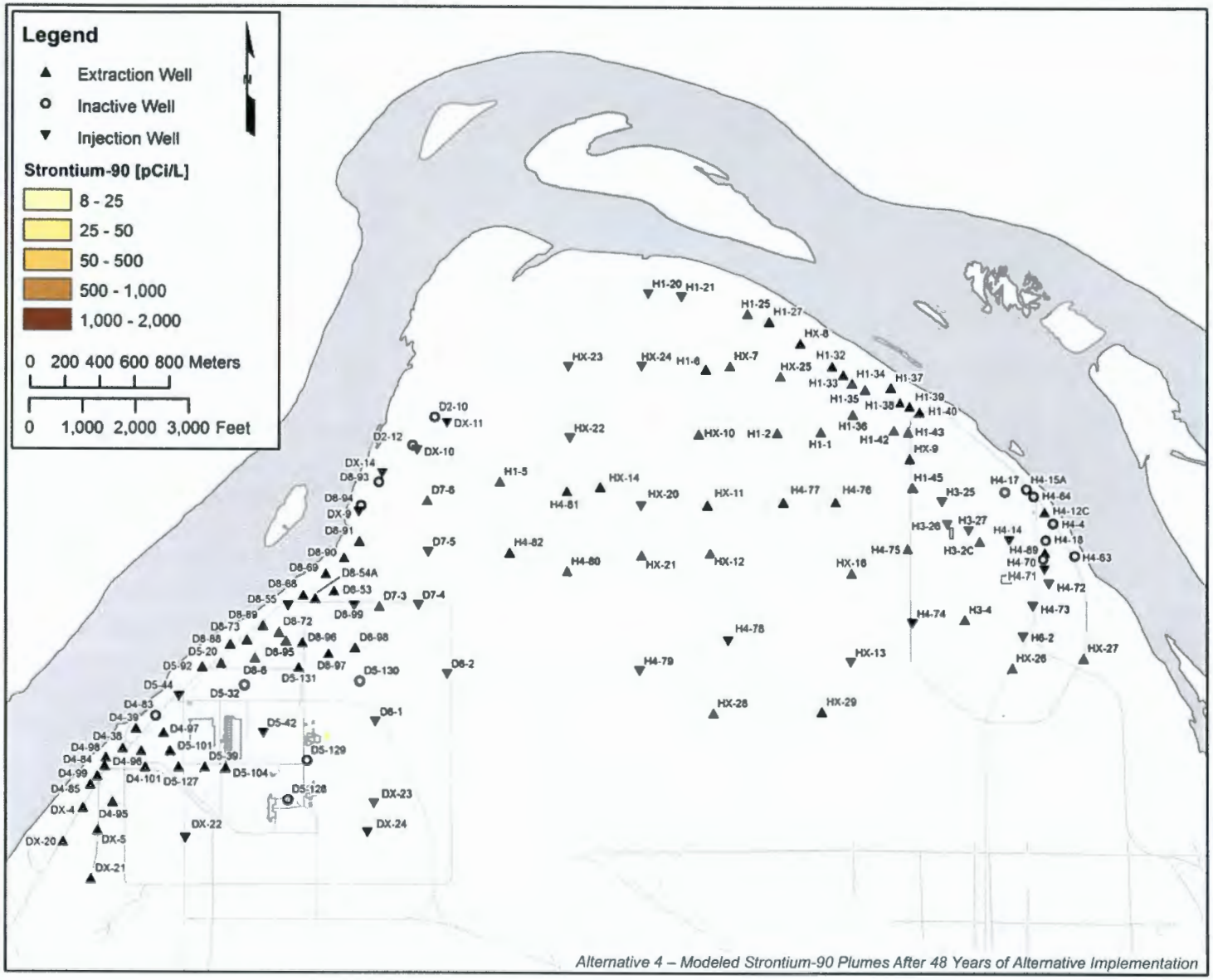

Figure 9-18c. Alternative 4—Modeled Strontium-90 Plume After 48 Years

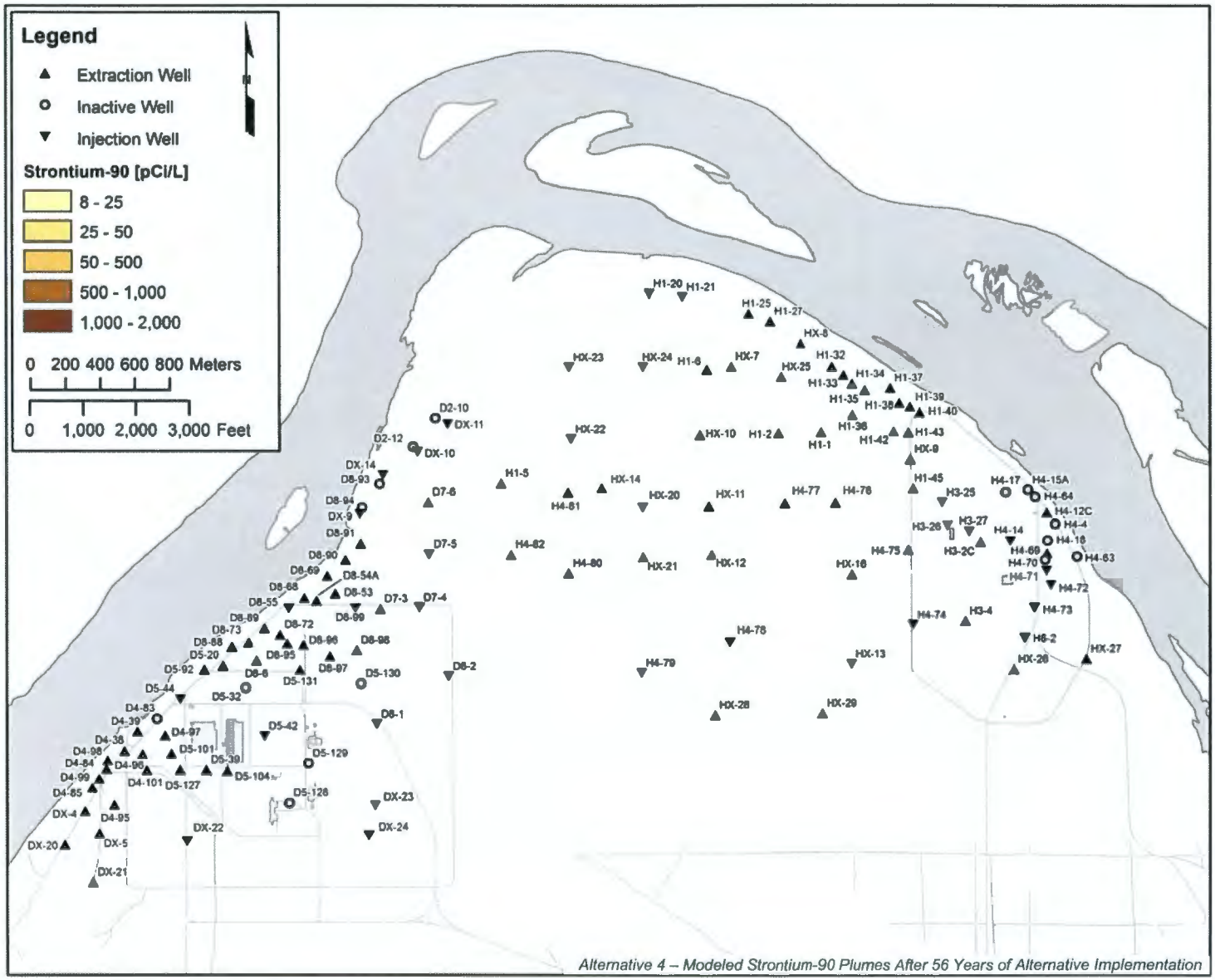

Figure 9-18d. Alternative 4—Modeled Strontium-90 Plume After 56 Years

Table 9-11. Alternative 4—Cost Estimates (in millions)[a]

Waste Site Treatment	Capital	$145
	Total O&M	$25.7
	Total Periodic[b]	$0.6
	Total Nondiscounted	$171.2
	Net Present Value	$155.3
Groundwater Treatment	Capital	$20.4
	Total O&M	$494.2
	Total Periodic[b]	$169
	Total Nondiscounted	$683.6
	Net Present Value	$445.5
Total Nondiscounted Cost of Alternative	$854.8	
Total Net Present Value (Discounted) of Alternative	$600.8	

a. Order of magnitude cost estimates and support information are presented in Appendix J. Costs for institutional controls are included under the costs for waste site treatment.

b. Periodic costs include O&M or construction activities, including costs to replace an installed remedy or components of an installed remedy, and services that are not included in initial capital costs or annual O&M costs. Periodic costs may be onetime costs or costs that recur at intervals over the life of the remedy.

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10 Detailed Analysis of Alternatives

This chapter provides a detailed analysis of the remedial alternatives developed for waste sites and groundwater at 100-D/H. This analysis follows the development of alternatives presented in Chapter 9 and precedes the Proposed Plan, which includes the identification of a preferred alternative.

This chapter evaluates the remedial alternatives defined in Chapter 9 for seven of the nine CERCLA criteria described in the NCP ("Remedial Investigation/Feasibility Study and Selection of Remedy" [40 CFR 300.430(e)(9)]).

The CERCLA evaluation criteria are presented in Section 10.1, and each of the remedial alternatives is evaluated individually and comparatively against the CERCLA criteria in Sections 10.2 and 10.3, respectively. The remaining two modifying criteria are formally assessed during preparation of the Proposed Plan (State Acceptance) and following review of public and stakeholder comments (Community Acceptance) on the Proposed Plan. The purpose of the detailed and comparative analysis is to develop the information necessary to recommend a preferred alternative in a Proposed Plan. Following public and stakeholder review of the Proposed Plan, the Tri-Parties will select a final remedial action alternative for 100-D/H that will lead to a ROD.

Highlights

- Alternative 1 (No Action [as required by the NCP]) does not meet threshold criteria for all sites.
- Alternative 2 (RTD and Grouting for Waste Site and Pump-and-Treat with Biological Treatment for Groundwater) meets threshold criteria, performs well for long-term effectiveness, and short-term effectiveness, and moderately for reduction of toxicity, mobility, and volume (TMV), and implementability. The NPV cost for this alternative is $506 million.
- Alternative 3 (RTD and Grouting of Waste Site and Increased Capacity Groundwater Pump-and-Treat for Groundwater) meets threshold criteria, performs well for short-term effectiveness, long-term effectiveness, and implementability, and moderately for reduction of TMV. The NPV cost for this alternative is $570 million.
- Alternative 4 (RTD for Waste Sites and Pump-and-Treat for Groundwater) meets threshold criteria, performs well for long-term effectiveness and implementability, and moderately for short-term effectiveness and reduction of TMV. The NPV cost for this alternative is $601 million.
- Alternative 3 performs better than Alternatives 2 and 4 for long-term effectiveness and short-term effectiveness.

10.1 Description of CERCLA Evaluation Criteria

This section describes the nine CERCLA evaluation criteria upon which the detailed and comparative evaluations are based. The nine criteria are designed to enable the analysis of each alternative to address the statutory, technical, and policy considerations necessary for selecting a final remedial alternative. These evaluation criteria (Table 10-1) provide the framework for conducting the detailed analysis of alternatives and selecting an appropriate remedial action. Table 10-1 provides the more detailed questions that CERCLA guidance suggests be used to address these criteria. The performance or acceptability of each alternative is first evaluated individually so relative strengths and weaknesses may be identified (Section 10.2). Section 10.3 compares the performance of each alternative to the others, relative to the CERCLA criteria.

The evaluation criteria are divided into three categories (threshold, balancing, and modifying) based on the function of each category in the remedy selection process. The NCP ("Remedial Investigation/Feasibility Study and Selection of Remedy" [40 CFR 300.430(f)]) states that the first two criteria, protection of HHE and compliance with ARARs are "threshold criteria" that must be met by the selected remedial action unless a waiver can be granted under CERCLA ("Degree of Cleanup" [Section 121(d)(4)]).

Table 10-1. Summary of CERCLA Criteria

Threshold Criteria	
Overall Protection of HHE	The assessment against this criterion describes how the alternative, as a whole, achieves and maintains protection of human health and the environment.
Compliance with ARARs	The assessment against this criterion describes how the alternative complies with ARARs, or if a waiver is required and how it is justified. The assessment also addresses other information from advisories, criteria, and guidance that the lead and support agencies have agreed is "to be considered."
Balancing Criteria	
Long-Term Effectiveness and Permanence	The assessment of alternatives against this criterion evaluates the long-term effectiveness of alternatives in maintaining protection of human health and the environment after response objectives have been met.
Reduction of TMV through Treatment	The assessment against this criterion evaluates the anticipated performance of the specific treatment technologies an alternative may employ.
Short-Term Effectiveness	The assessment against this criterion examines the effectiveness of alternatives in protecting human health and the environment during the construction and implementation of a remedy until response objectives have been met.
Implementability	This assessment evaluates the technical and administrative feasibility of alternatives and the availability of required goods and services.
Cost	This assessment evaluates the capital and O&M costs of each alternative.
Modifying Criteria	
State Acceptance*	This assessment reflects the state's (or support agency's) apparent preferences among or concerns about alternatives.
Community Acceptance*	This assessment reflects the community's apparent preferences among or concerns about alternatives.

* These criteria are not assessed in this report.

The five "balancing criteria" represent technical considerations upon which the detailed analysis is primarily based. These criteria include long-term effectiveness and permanence; reduction of toxicity, mobility, and volume [TMV] through treatment; short-term effectiveness; implementability; and cost. The preferred alternative will be the alternative that is protective of HHE, is ARAR-compliant, and provides the best combination of primary balancing attributes.

The final two criteria, State and Community Acceptance, are "modifying criteria." State Acceptance is formally assessed during preparation of the Proposed Plan, and Community Acceptance is formally assessed following review of public and stakeholder comments on the Proposed Plan. Community and State Acceptance are not addressed in the FS. Based on information from public participation, the Tri-Parties may modify some aspects of the preferred alternative or decide that another alternative is more appropriate.

10.1.1 Overall Protection of Human Health and the Environment

Overall protection of HHE is the primary requirement that remedial actions must meet under CERCLA. This evaluation criterion is an assessment of whether each alternative achieves and maintains adequate protection of HHE, in both the short term and the long term, from unacceptable risks posed by contaminants. Alternatives are protective by eliminating, reducing, or controlling exposure through applicable exposure pathways (NCP, "Remedial Investigation/Feasibility Study and Selection of Remedy" [40 CFR 300.430(e)(2)(i)]). Overall protection of HHE draws on the assessments of the other evaluation criteria, especially long-term effectiveness and permanence, short-term effectiveness, and compliance with ARARs.

10.1.2 Compliance with ARARs

Compliance with ARARs is the second threshold criterion of remedy selection. This evaluation criterion is used to determine whether an alternative meets the federal, state, and local ARARs identified for the site, as presented in Chapter 8. Alternatives are assessed to determine whether they meet ARARs and other requirements, or if a basis exists for invoking one of the waivers cited in NCP ("Remedial Investigation/Feasibility Study and Selection of Remedy" [40 CFR 300.430(f)(l)(ii)(C)]).

10.1.3 Long-Term Effectiveness and Permanence

Long-term effectiveness and permanence are criteria that evaluate the anticipated ability of an alternative to maintain reliable protection of HHE for the duration of time that risk is above allowable levels. Alternatives are assessed for the long-term effectiveness and permanence they afford, along with the degree of certainty that the alternative will prove successful in meeting the RAOs. The following factors are considered in this assessment:

- The magnitude of residual risk from untreated waste or treatment residuals remaining at the conclusion of the remedial action, including the TMV. Magnitude of residual risk is defined as the risk remaining from untreated waste or treatment residuals after remediation.
- The adequacy and reliability of controls that can be used to manage treatment residuals or residual contamination that remains at the site, such as containment systems or institutional controls. For example, this factor addresses uncertainties associated with land disposal for providing long-term protection from treatment residuals; the assessment of the potential need to replace technical components of the alternative such as a treatment system; and the potential exposure pathways and risks posed if the remedial action needs to be replaced.

10.1.4 Reduction of Toxicity, Mobility, and Volume through Treatment

This evaluation criterion concerns the statutory preference for remedies that employ treatment to reduce the TMV of the hazardous substances. This preference is satisfied when treatment is used to reduce the principal threats at a site through destruction of toxic contaminants, reduction in contaminant mobility, or reduction of the total mass or total volume of contaminated media. This criterion is specific to evaluating how the treatment reduces TMV.

The degree to which the alternative employs treatment that reduces TMV will be assessed. The following factors are considered for the evaluation:

- The treatment process and the materials treated
- The amount of hazardous substances, pollutants, or contaminants that will be destroyed or treated
- The degree of expected reduction in TMV of the waste as a percentage of reduction

- The degree to which the treatment is irreversible
- The type and quantity of residuals that will remain following treatment, taking into consideration the persistence, toxicity, mobility, and propensity of hazardous substances and their constituents to bioaccumulate
- The degree to which treatment reduces the inherent hazards posed by the principal threats

10.1.5 Short-Term Effectiveness

This criterion focuses on short-term effects of the remedial alternatives by examining the effectiveness of alternatives in protecting HHE during the construction and implementation phase until RAOs are met. As outlined by the CERCLA guidance, this criterion includes four analysis factors:

- Protection of the community during remedial actions (e.g., dust from excavations and transportation of hazardous materials)
- Protection of workers during remedial actions
- Potential adverse environmental impacts (e.g., waste and generation of greenhouse gas [GHG] emissions) and the effectiveness and reliability of mitigating measures
- Time until RAOs are achieved

10.1.6 Implementability

The implementability criterion relates to the technical and administrative feasibility of executing an alternative and the availability of various services and materials required during its implementation. The ease or difficulty of implementing the alternative is assessed by considering the following types of factors, as appropriate:

- Technical feasibility, including the technical difficulties and unknowns associated with constructing and operating the technology, the reliability of the technology, the ease of undertaking additional remedial actions, and the ability to monitor the effectiveness of the remedy
- Administrative feasibility, including activities requiring coordination with other agencies, and the ability and time needed to obtain any necessary approvals and permits for offsite actions
- Availability of required services, personnel, resources, technologies, and materials necessary to construct and operate the alternative

10.1.7 Cost

The cost estimate for each remedial alternative typically includes the following items:

- Remedial design costs including preparation of design drawings and specifications, construction bid documents, the RD/RA work plan, and the interim remedial action report; typically calculated as a percent of the capital cost
- Remedial alternative construction costs including construction management, capital equipment, general and administrative costs, and construction subcontract costs and fees
- Estimated operating, maintenance, and remedy performance monitoring and reporting costs for the duration of the remedial action

- Equipment replacement costs
- Project management

The evaluation of the cost criterion is based on a comparison of the estimated present worth of these costs for each alternative. Actual costs will depend on the final scope and design of the selected remedial action, the implementation schedule, competitive market conditions, and other variables. However, these factors, equally applicable to all alternatives, are not expected to affect the relative cost differences between alternatives.

Life-cycle costs are presented as net present worth values. The net present worth method establishes a common baseline for evaluating costs that occur during different periods, thus allowing for direct cost comparisons between different alternatives. The net present worth value represents the dollars that would need to be set aside today, at the defined interest rate, to ensure that funds would be available in the future, as they are needed to perform the remedial alternative.

Net present worth costs were estimated using the real discount rate published in Appendix C, "Discount Rates for Cost-Effectiveness, Lease Purchase, and Related Analyses" (Appendix C) of the "Guidelines and Discount Rates for Benefit-Cost Analysis of Federal Programs" (OMB Circular No. A-94, revised December 2011). This version of the appendix is valid for calendar year 2012.

The cost estimates are based on specific response action scenarios and assumptions. Sensitivity analyses were not performed to quantify the potential effect of changing key parametric assumptions.

The cost estimates were prepared in accordance with *A Guide to Developing and Documenting Cost Estimates During the Feasibility Study* (EPA 540-R-00-002), along with DOE's *Cost Estimating Guide* (DOE G 430.1-1). The cost estimates are for comparison purposes and are prepared to meet the -30 to +50 percent range of accuracy recommended in CERCLA RI/FS Guidance (EPA/540/G-89/004). The cost estimate details, uncertainties, and supporting information are included in *Environmental Cost Estimate for 100-D/H Vadose Zone and Groundwater RI/FS* (ECE-100HR311-00004) and *100-DH Cost Estimate Scoping Forms for Feasibility Study Alternative Costing* (ECF-100DR1-12-0022) in Appendix J.

The estimated costs for maintaining the site-wide or programmatic ICs including site access, personnel badging, real estate and deeds, warning signs along the Columbia River bank and other access points, maintaining a current Sitewide institutional controls plan, controls for excavating soil, accessing and using groundwater, and irrigation restrictions are also provided. These costs were assembled and, where appropriate, a 50 percent adjustment was made to represent CERCLA cleanup as a portion of the current Hanford Site mission. The TPA (Ecology et al., 1989a) currently identifies 22 CERCLA RODs, so each ROD would be allocated an equal portion of the CERCLA programmatic ICs costs.

The programmatic ICs costs are projected for the next 150 years. In 2068, ICs costs are reduced by 50 percent to reflect removal of the 100 Area reactors, as the more active programmatic controls, such as site access, would be likewise reduced.

The total non-discounted cost for the ICs for 150 years is estimated to be $563 million for the Hanford Site (about $26 million per ROD). The total discounted cost for the ICs at Hanford is estimated at $221 million (about $10 million per ROD).

The total non-discounted cost for the 5-year reviews for 150 years is estimated to be $14 million (about $0.6 million per ROD). The total discounted cost for the 5-year reviews for 150 years is estimated to be $4 million (about $190,000 per ROD).

The costs for maintaining programmatic ICs and the 5-year reviews conducted as oversight for the CERCLA actions on the Hanford Site are the same for each alternative (except No Action) and are included with the cost estimates. Detailed cost estimates for programmatic ICs and 5-year reviews are included in Appendix J.

10.2 Individual Analysis of Alternatives

This section evaluates each of the remedial action alternatives defined in Chapter 9 against the CERCLA threshold and balancing criteria described in Section 10.1. Criteria evaluation details for the remedial alternatives are documented in tabular form. The ratings provided below indicate the expected performance of each alternative relative to the CERCLA criteria:

◉= Performs very well against the criterion with minimal disadvantages or uncertainties

○ = Performs moderately well against the criterion but with some disadvantages or uncertainties

● = Performs poorly against the criterion and may have significant disadvantages or uncertainties

10.2.1 Alternative 1—No Action Alternative

The NCP ("Remedial Investigation/Feasibility Study and Selection of Remedy" [40 CFR 300.430(e)(6)]) requires consideration of a No Action Alternative to serve as a baseline for evaluating other remediation action alternatives and is retained throughout the FS process. As presented in Chapter 9, to evaluate the required No Action Alternative, all site remedial activities and interim actions (with the possible exception of backfilling any open excavations that are not safe) will be discontinued in December 2012, including ceasing operation of pump-and-treat systems and additional monitoring. Preliminary design details and cost estimates are not prepared for Alternative 1.

The individual analysis of this alternative is presented in Table 10-2. Given that No Action fails the threshold criteria established in NCP ("Remedial Investigation/Feasibility Study and Selection of Remedy" [40 CFR 300.430(f)]), information regarding the performance of this alternative with respect to the balancing criteria is not included.

Table 10-2. Individual Criteria Evaluation for Alternative 1—No Action

Criterion	Rating	Detailed Analysis
Overall Protection of Human Health and the Environment	-	***Not expected to be protective of HHE*** • RAOs will not be achieved • Allows unmonitored migration of contaminated groundwater to the Columbia River • Potential for exposure to human and ecological receptors and potential of contaminants to leach to groundwater may remain at the waste sites
Compliance with ARARs	-	***Is not compliant*** • Since there is no action, ARARs for waste sites will not be met • DWS and AWQC ARARs may not be achieved in a reasonable time since cessation of remedial actions allows for further migration of groundwater contaminants and exposure to groundwater
Long-term Effectiveness and Permanence	N/A	Alternative 1 fails threshold criteria. Therefore, an evaluation on balancing criteria is not provided.

Table 10-2. Individual Criteria Evaluation for Alternative 1—No Action

Criterion	Rating	Detailed Analysis
Reduction of TMV through Treatment	N/A	Alternative 1 fails threshold criteria. Therefore, an evaluation on balancing criteria is not provided.
Short-term Effectiveness	N/A	Alternative 1 fails threshold criteria. Therefore, an evaluation on balancing criteria is not provided.
Implementability	N/A	Alternative 1 fails threshold criteria. Therefore, an evaluation of balancing criteria is not provided.
Cost	N/A	A cost estimate for Alternative 1 is not provided because no action is proposed.

N/A = not applicable

10.2.2 Alternative 2—RTD and Grouting for Waste Site and Pump-and-Treat with Biological Treatment for Groundwater

The remedial strategy for Alternative 2 relies on optimizing the risk reduction and cost by using a mixture of RTD, void fill grouting, MNA and institutional controls, and groundwater pump-and-treat. For remediated waste sites that are evaluated in the FS because of PRG exceedances, this alternative builds off the interim actions previously completed. For waste sites, the actions include the following:

- RTD for removal of contamination to clean up levels for waste sites (47 waste sites), including demolition of structures (e.g., buildings)
- RTD and void filling at 100-H-36
- Capping of pipe ends to provide containment of contamination in pipelines associated with waste site 100-D-50:2
- MNA and institutional controls to mitigate exposure at waste sites 116-DR-9, 100-D-25, and 116-D-8 until radiological contaminants decay to less than residential RBSLs in the year 2035

For groundwater, the actions include optimization of the pump-and-treat with biological injection for treatment of Cr(VI). The optimized pump-and-treat system would operate for 25 years following implementation of the remedial alternative. Remedial process optimization activities will be conducted throughout the operational period to assess remedial action performance. Groundwater simulations are used to modify extraction/injection well fields and locations for bioremediation for remediation of remaining contamination. Groundwater actions include MNA and institutional controls for strontium-90 contamination.

Nitrate and strontium-90 contaminated groundwater plumes are within the treatment footprint for the Cr(VI) plume and will be co-extracted by the extraction well network used for the Cr(VI) plume remediation. The groundwater treatment system effluent is not expected to exceed MCLs, and no treatment is proposed for strontium-90 and nitrate. Specific treatment may be proposed if the combined extracted groundwater in the pump-and-treat effluent stream is found to exceed the MCLs.

Groundwater model simulation for this alternative indicates that nitrate contamination is remediated to cleanup levels within the Cr(VI) remediation period. Small pockets of strontium-90 contamination remain in the aquifer following completion of the Cr(VI); however, remaining strontium-90 contamination will

attenuate to cleanup levels through radiological decay in 31 years after Cr(VI) remediation (i.e., 56 years after implementation of the Alternative 2 remedy). Groundwater monitoring will confirm effectiveness of MNA for strontium-90 in the aquifer and determine the effect of pump-and-treat on the persistence of the strontium-90 and nitrate contamination within the aquifer over time.

Table 9-6 and Figure 9-7 provide additional information on this alternative. Appendix J, Table J-3, presents a summary of the remedial components, areas, and volumes that were assumed for each waste site in development of cost estimates.

Table 10-3 presents the individual analysis of Alternative 2 with respect to the seven CERCLA criteria. Alternative 2 is considered to pass the threshold criteria, since the actions as part of this alternative achieve RAOs and ARARs. Alternative 2 provides long-term effectiveness and permanence, and short-term effectiveness. Alternative 2 is expected to achieve reduction in TMV because it incorporates active treatment of groundwater and waste sites. However, contaminated soil removal by RTD is not considered treatment, so reduction by TMV was reduced to moderately effective. Alternative 2 is expected to be moderately implementable.

Table 10-3. Individual Criteria Evaluation for Alternative 2—RTD and Grouting for Waste Site and Pump-and-Treat with Biological Treatment for Groundwater

Criterion	Rating	Detailed Analysis
Overall Protection of Human Health and the Environment	O	***Protective of HHE*** • This alternative incorporates previous interim action remedies (i.e., soil RTD and groundwater extraction) that have been demonstrated to be effective, plus additional techniques to optimize remedial design. • Monitoring can track progress and achievement of RAOs. • RTD mitigates risk to human and ecological receptors and soil-to-groundwater/surface water risks encountered at the site through physical removal of contaminated soil. • Pump-and-treat provides for hydraulic containment to limit potential further migration of groundwater contaminants and treatment to remediate the Cr(VI) plume. Biological injection enhances the remediation of the Cr(VI) plume. Other COCs (strontium-90 and nitrate) will be co-extracted with groundwater extracted for Cr(VI) treatment, where the concentration of strontium-90 and nitrate in the combined effluent will be less than their respective MCLs. Following the cessation of active treatment for Cr(VI), MNA will be the sole mechanism to achieve cleanup levels for strontium-90. • The pump-and-treat enhanced with biological injection is projected to achieve risk standards in a period of about 56 years in the 100-D Area, 30 years in the 100-H Area, and 25 years in the horn area. • Institutional controls prevent exposure to affected soil and groundwater.
Compliance with ARARs	O	***Compliant*** • Remedial action/systems are designed to meet ARARs. • DWS and AWQC ARARs for Cr(VI) are projected to be achieved in approximately 11 and 25 years of operation, respectively.

Table 10-3. Individual Criteria Evaluation for Alternative 2—RTD and Grouting for Waste Site and Pump-and-Treat with Biological Treatment for Groundwater

Criterion	Rating	Detailed Analysis
Long-term Effectiveness and Permanence	●	***Expected to have long-term effectiveness and permanence*** *Factors for this alternative performing very well against the long-term effectiveness and permanence criterion:* • RTD, void fill grouting, containment (by capping of ends of pipelines), MNA and institutional controls, hydraulic containment, pump-and-treat, and biological injections achieve RAOs. • RTD using conventional equipment is reliable and practiced extensively at Hanford. ERDF disposal is effective and reliable. • Adequacy and reliability of pump-and-treat operations are proven at Hanford. • Groundwater biological treatment (bioinjection) provides additional treatment as pump-and-treat efficiency decreases with decreasing Cr(VI) concentration. • This alternative remediates soil site and groundwater COCs to PRG levels established to achieve the RAOs. Residual risks following cleanup to PRG levels will meet the 10^{-4} to 10^{-6} ELCR for radiological and carcinogenic COCs and will be less than or equal to an HI value of 1.0 for hazardous substances. • Long-term effectiveness and permanence will be evaluated in CERCLA 5-year review process. *Factors that may provide some disadvantages or uncertainty to long-term effectiveness and permanence:* • Requires long-term enforcement of institutional controls to mitigate risk. • Cr(VI) treatment systems will operate for more than 25 years to achieve AWQC and may require major system renovations or replacement.
Reduction of TMV by Treatment	○	***Expected to have moderate reduction of TMV by treatment*** *Factors for this alternative performing moderately well against the reduction of TMV by treatment criterion:* • Significant mass of Cr(VI) will be removed through pump-and-treat where ion exchange treatment removes Cr(VI) from the groundwater. When the ion exchange resin capacity is reached, the resin will be changed out with fresh resin. Removal of Cr(VI) is not reversible since the spent resin loaded with Cr(VI) will be disposed as solid waste at ERDF or another EPA-approved disposal facility. • Other reduction of TMV by treatment is achieved through biological reduction (bioinjection) converting Cr(VI) to less toxic Cr(III). The re-oxidation of Cr(III) compounds to Cr(VI) is not anticipated to be a significant issue after bioremediation is completed as identified in the bioremediation technology descriptions in Appendix I. • Void fill grouting reduces leaching (mobility) of contamination from the vadose zone to groundwater through containment of contaminated media. • Capping of ends of the 100-D-50:2 waste site pipelines provides containment of contaminated media (scale within the pipelines) to reduce direct contact human health risks. • Removed soil is treated at ERDF, if needed

Table 10-3. Individual Criteria Evaluation for Alternative 2—RTD and Grouting for Waste Site and Pump-and-Treat with Biological Treatment for Groundwater

Criterion	Rating	Detailed Analysis
		Factors that may provide some disadvantages or uncertainty to the reduction of TMV by treatment: • Design testing of biological injections is needed to fully develop the design parameters for these technologies, since they have not been applied full-scale at Hanford. • Contaminated soil removal by RTD is not considered treatment. RTD is the primary remediation technology implemented for most of the waste sites, and for this reason, this alternative performs moderately for this criterion.
Short-term Effectiveness	◐	***Expected to have short-term effectiveness*** *Factors for this alternative performing very well against the short-term effectiveness criterion:* • No adverse risks to the community from implementation of active treatment options because of the remote location and implementation of institutional controls. • Risks to workers from implementation of active treatment actions are minimized through a health and safety plan and proper PPE. • Nominal short-term risks to workers during installation of groundwater extraction and conveyance piping, system O&M activities, and monitoring. Risks are minimized through a health and safety plan and PPE. • Cr(VI) DWS and AWQC are achieved within 11 and 25 years of remedy implementation, respectively. *Factors that may provide some disadvantages or uncertainty to the short-term effectiveness:* • 56 years to achieve RAOs for strontium-90 in the unconfined aquifer. • Negative environmental impacts may include: – Generation of GHG from use of excavation equipment and disposal to ERDF; production and delivery of biological reagents (may mobilize other metal but will be captured and treated in pump-and-treat); well installation, development, and injection activities; and operation of pump-and-treat system – Waste generation from disposal of contaminated soil and spent ion exchange resin – Large amounts of backfill will be required; borrow material may have significant environmental impacts
Implementability	○	***Expected to be moderately implementable*** *Factors for this alternative performing moderately well against the implementability criterion:* • Conventional equipment and vendors for implementation of treatment options (RTD, void fill grouting, pump-and-treat, and biological injections) are readily available. • RTD of waste sites have been implemented and proven through interim actions at Hanford. • ERDF for waste disposal is proven, reliable, and accessible. • Grout void fill being implemented at the Hanford U Plant facility as part of the ROD selected remedy. Grout pours to fill voids in facility structures, larger than those identified for the 100-H-36 waste site at 100-H, have been successfully performed.

Table 10-3. Individual Criteria Evaluation for Alternative 2—RTD and Grouting for Waste Site and Pump-and-Treat with Biological Treatment for Groundwater

Criterion	Rating	Detailed Analysis	
		• Pump-and-treat construction and operations have been implemented through interim actions, and the ability to construct and operate is proven at Hanford. • Radioactive decay is a confirmed attenuating process for strontium-90, which exhibits a sufficiently short radioactive half-life (29.1 years). • Institutional controls have been approved and implemented, and are reliable/effective at Hanford. *Factors that may provide some disadvantages or uncertainty to implementability:* Design testing required for biological reducing zone parameters for design of in situ bioremediation treatment of groundwater.	
Cost (in millions)[a]			
Waste Site Treatment		Capital	$132.4
		Annual O&M	$25.7
		Total Periodic[b]	$0.8
		Total non-discounted	$158.9
		NPV	$142.9
Groundwater Treatment		Capital	$20.2
		Total O&M	$361.7
		Total Periodic[b]	$111.7
		Total nondiscounted	$493.5
		NPV	$363.1
Total Nondiscounted Cost of Alternative		$652.5	
Total Net Present Value (Discounted) of Alternative		$506	

a. Order-of-magnitude cost estimates and support information are presented in Appendix J. Costs for institutional controls are included under the costs for waste site treatment.

b. Periodic costs include O&M and/or construction activities, including costs to replace an installed remedy or components of an installed remedy, and services that are not included in initial capital costs or annual O&M costs. Periodic costs may be one-time costs or costs that recur at intervals over the life of the remedy.

10.2.3 Alternative 3—RTD and Grouting of Waste Site and Increased Capacity Groundwater Pump-and-Treat for Groundwater

The remedial strategy for Alternative 3 relies on optimizing the risk reduction and cost by using a mixture of RTD, void fill grouting, and MNA and institutional controls for waste site contamination to achieve the RAOs, as well as expanded pump-and-treat with increased treatment capacity for groundwater. For remediated waste sites that are evaluated in the FS because of PRG exceedances, this alternative builds off the interim actions previously completed. The remedial action will include the following:

- RTD for removal of contaminants to clean-up levels for waste sites (47 waste sites), including demolition of structures (e.g., buildings) when necessary
- RTD and void filling at 100-H-36
- Capping of pipe ends to provide containment of contamination in pipelines associated with waste site 100-D-50:2
- MNA and institutional controls to mitigate exposure at waste sites 116-DR-9, 100-D-25, and 116-D-8 until radiological contaminants decay to less than residential RBSLs in the year 2035
- For hexavalent chromium in groundwater, expanded pump-and-treat with increased treatment capacity for a period of about 12 years following implementation of the remedial alternative for treatment of Cr(VI). Groundwater actions include MNA and institutional controls for strontium-90 contamination.
- Nitrate and strontium-90 contaminated groundwater plumes are within the treatment footprint for the Cr(VI) plume and will be co-extracted by the extraction well network used for the Cr(VI) plume remediation. The groundwater treatment system effluent is not expected to exceed MCLs, and no treatment is proposed for strontium-90 and nitrate. Specific treatment would be provided if the combined extracted groundwater in the pump-and-treat effluent stream exceeds the MCLs for the respective COCs before reinjection.
- Groundwater model simulation for this alternative indicates that nitrate contamination is remediated to cleanup levels within the Cr(VI) remediation period. Small pockets of strontium-90 contamination remain in the aquifer following completion of the Cr(VI), however, remaining strontium-90 contamination will attenuate to cleanup levels through radiological decay in 32 years after Cr(VI) remediation (i.e., 44 years after implementation of the Alternative 3 remedy). Groundwater monitoring will confirm effectiveness of MNA for strontium-90 and nitrate and determine the effect of pump-and-treat on the persistence of these COCs within the aquifer over time.

Table 9-8 and Figure 9-11 provide additional information on this alternative. Appendix J, Table J-3, presents a summary of the remedial components, areas, and volumes that were assumed for each waste site in development of cost estimates.

Table 10-4 presents the individual analysis of Alternative 3 with respect to the seven CERCLA criteria. Alternative 3 was considered to pass the threshold criteria, since the actions as part of this alternative achieve RAOs and ARARs. Alternative 3 is considered to have good long-term effectiveness and permanence, short-term effectiveness, and reduction of TMV through RTD and void fill grouting of contaminated soil and pump-and-treat for groundwater. However, contaminated soil removal by RTD is not considered treatment, so reduction by TMV was reduced to moderately effective. Alternative 3 is expected to be implementable.

Table 10-4. Individual Criteria Evaluation for Alternative 3—RTD and Grouting of Waste Site and Increased Capacity Groundwater Pump-and-Treat for Groundwater

Criterion	Rating	Detailed Analysis
Overall Protection of Human Health and the Environment	O	***Protective of human health and the environment*** • This alternative incorporates previous interim action remedies (i.e., soil RTD and groundwater treatment) that have been demonstrated to be effective. • Monitoring can track progress and achievement of RAOs. • RTD mitigates risks to human and ecological receptors and soil-to-groundwater/surface water risks encountered at the site through physical removal of contaminated soil. • Pump-and-treat remediates the Cr(VI) plumes. Other COCs (strontium-90, and nitrate) will be co-extracted with groundwater extracted for Cr(VI) treatment, where the concentration of strontium-90 and nitrate in the combined effluent will be less than their respective MCLs. Following the cessation of active treatment for Cr(VI), MNA will be the sole mechanism to achieve cleanup levels for strontium-90. • The enhanced pump-and-treat achieves risk standards for groundwater within a period projected to be approximately 44 years of implementation at the 100-D Area, 28 years at the H Areas, and 12 years in the horn area. • Institutional controls prevent exposure to affected soil and groundwater.
Compliance with ARARs	O	***Compliant*** • Remedial action/systems meet ARARs. • DWS and AWQC ARARs for Cr(VI) are projected to be achieved in approximately 6 and 12 years of operation, respectively.
Long-term Effectiveness and Permanence	O	***Expected to have long-term effectiveness and permanence*** *Factors expected to perform very well against the long-term effectiveness and permanence criterion:* • RTD, void fill grouting, containment (by capping of ends of pipelines), MNA and institutional controls, hydraulic containment, and pump-and-treat achieve RAOs. • RTD using conventional equipment is reliable and practiced extensively at Hanford. ERDF disposal is effective and reliable. • Adequacy and reliability of pump-and-treat are proven at Hanford. • Monitoring and verification sampling, and long-term groundwater monitoring track progress toward achieving RAOs. • This alternative remediates soil site and groundwater COCs to PRG levels established to achieve the RAOs. Residual risks following cleanup to PRG levels will meet the 10^{-4} to 10^{-6} ELCR for radiological and carcinogenic COCs and will be less than or equal to an HI value of 1.0 for hazardous substances. • Long-term effectiveness and permanence will be evaluated in the CERCLA 5-year review process. *Factors that may provide some disadvantages or uncertainty to long-term effectiveness and permanence:* • Requires long-term enforcement of institutional controls to mitigate risk.

Table 10-4. Individual Criteria Evaluation for Alternative 3—RTD and Grouting of Waste Site and Increased Capacity Groundwater Pump-and-Treat for Groundwater

Criterion	Rating	Detailed Analysis
Reduction of TMV by Treatment	○	***Expected to have moderate reduction of TMV by treatment*** *Factors expected to perform moderately well against the reduction of TMV by treatment criterion:* • Significant mass of Cr(VI) will be removed through pump-and-treat where ion exchange treatment removes Cr(VI) from the groundwater. When the ion exchange resin capacity is reached, the resin will be changed out with fresh resin. Removal of Cr(VI) is not reversible since the spent resin loaded with Cr(VI) will be disposed as solid waste at ERDF or another EPA-approved disposal facility. • Void fill grouting reduces leaching (mobility) of contamination from the vadose zone to groundwater through containment of contaminated media. • Capping of ends of the 100-D-50:2 waste site pipelines provides containment of contaminated media (scale within the pipelines) to reduce direct contact human health risks. • Removed soil is treated at ERDF, if needed *Factors that may provide some disadvantage or uncertainty to the reduction of TMV by treatment:* Contaminated soil removal by RTD is not considered treatment. RTD is the primary remediation technology implemented for most of the waste sites; for this reason, this alternative performs moderately for this criterion.
Short-term Effectiveness	●	***Expected to have short-term effectiveness*** *Factors expected to perform very well against the short-term effectiveness criterion:* • No effects to the community associated with implementation of institutional controls. No adverse risks to the community from implementation of active treatment options because of the site's remote location. • Risks to workers from implementation of active treatment actions are minimized through an HSP and proper PPE. • Nominal short-term risks to workers during installation of groundwater extraction and conveyance piping, system O&M activities, and monitoring. Risks are minimized through HSP and PPE. • Cr(VI) DWS and AWQC are achieved within 6 years and 12 years of remedy implementation, respectively. *Factors that may provide some disadvantages or uncertainty to short-term effectiveness* • Negative environmental impacts may include the following: – Generation of GHG from use of excavation equipment and disposal to ERDF, operation of pump-and-treat system – Waste generation from disposal of contaminated soil and spent ion exchange resin – Energy consumption from process equipment • Large amounts of backfill will be required; borrow material may have significant environmental impacts
Implementability	●	***Expected to have Implementability*** *Factors considered for implementability* • Limited technical challenges are associated with implementation of this alternative. • Institutional controls have been approved and implemented, and are reliable/effective at Hanford.

Table 10-4. Individual Criteria Evaluation for Alternative 3—RTD and Grouting of Waste Site and Increased Capacity Groundwater Pump-and-Treat for Groundwater

Criterion	Rating	Detailed Analysis	
		• Conventional equipment and vendors for implementation of active treatment options (RTD, and pump-and-treat) are readily available. • ERDF is proven, reliable, and accessible. • Grout void fill being implemented at the Hanford U Plant facility as part of the ROD selected remedy. Grout pours to fill voids in facility structures, larger than those identified for the 100-H-36 waste site at 100-H, have been successfully performed. • Pump-and-treat construction and operations have been previously approved and the ability to construct and operate is proven at Hanford. • Radioactive decay is a confirmed attenuating process for strontium-90, which exhibits a sufficiently short radioactive half-life (29.1 years). *Factors considered low implementability:* Installation of the large number of new pump-and-treat wells may present logistics problems with waste site excavation.	
Cost (in millions)[a]			
Waste Site Treatment		Capital	$132.4
		Annual O&M	$25.7
		Total Periodic[b]	$0.8
		Total non-discounted	$158.9
		NPV	$142.9
Groundwater Treatment		Capital	$117.8
		Total O&M	$287.5
		Total Periodic[b]	$82.3
		Total non-discounted	$487.6
		NPV	$427.2
Total Nondiscounted Cost of Alternative		$646.5	
Total Net Present Value (Discounted) of Alternative		$570.1	

a. Order-of-magnitude cost estimates and support information are presented in Appendix J.

b. Periodic costs include O&M and/or construction activities, including costs to replace an installed remedy or components of an installed remedy, and services that are not included in initial capital costs or annual O&M costs. Periodic costs may be one-time costs or costs that recur at intervals over the life of the remedy.

10.2.4 Alternative 4— RTD for Waste Sites and Pump-and-Treat for Groundwater

Remedial action under Alternative 4 relies on RTD for waste site contamination, to achieve the RAOs with the greatest degree of certainty, and groundwater pump-and-treat. For remediated waste sites that are evaluated in the FS because of PRG exceedances, this alternative builds off the interim actions previously

completed. For waste sites, the actions include RTD for removal of contamination to clean-up levels for waste sites (52 waste sites), including demolition of structures (e.g., buildings).

For groundwater, the actions include optimization of the pump-and-treat for treatment of Cr(VI). The optimized pump-and-treat system would operate for 39 years following implementation of the remedial alternative. Remedial process optimization activities will be conducted throughout the operational period to assess remedial action performance. Groundwater modeling simulations are used to support design optimization. Actual performance data are used to update model parameters to reduce model uncertainty. Groundwater simulations are used to modify extraction/injection well field and locations for remediation of remaining contamination. Groundwater actions include MNA and institutional controls for strontium-90 contamination.

Nitrate and strontium-90 contaminated groundwater plumes are within the treatment footprint for the Cr(VI) plume and will be co-extracted by the extraction well network used for the Cr(VI) plume remediation. The groundwater treatment system effluent is not expected to exceed MCLs, and no treatment is proposed for strontium-90 and nitrate. Specific treatment would be provided if the combined extracted groundwater in the pump-and-treat effluent stream exceeds the MCLs for the respective COCs before reinjection.

Groundwater model simulation for this alternative indicates that nitrate contamination is remediated to cleanup levels within the Cr(VI) remediation period. Small pockets of strontium-90 contamination remain in the aquifer following completion of the Cr(VI), however, remaining strontium-90 contamination will attenuate to cleanup levels through radiological decay in 17 years after Cr(VI) remediation (i.e., 56 years after implementation of the Alternative 4 remedy). Groundwater monitoring will confirm effectiveness of MNA for strontium-90 and nitrate and determine the effect of pump-and-treat on the persistence of these COCs within the aquifer over time.

Table 9-10 and Figure 9-15 provide additional information on this alternative. Appendix J, Table J-3, presents a summary of the remedial components, areas, and volumes that were assumed for each waste site in development of cost estimates.

Table 10-5 presents the individual analysis of Alternative 4 with respect to the seven CERCLA criteria. Alternative 4 was considered to pass the threshold criteria, since the actions as part of this alternative achieve RAOs and ARARs. Alternative 4 is expected to have long-term effectiveness and permanence. Alternative 4 is expected to achieve reduction in TMV because it incorporates active treatment of groundwater and waste sites. However, contaminated soil removal by RTD is not considered treatment, so reduction by TMV was reduced to moderately effective. Short-term effectiveness is expected to be moderate since Cr(VI) AWQC is achieved following 39 years of remedy implementation. Alternative 4 is expected to be implementable.

Table 10-5. Individual Criteria Evaluation for Alternative 4— RTD for Waste Sites and Pump-and-Treat for Groundwater

Criterion	Rating	Detailed Analysis
Overall Protection of Human Health and the Environment	O	***Protective of human health and the environment*** • This alternative incorporates previous interim action remedies (i.e., soil RTD and groundwater extraction) that have been demonstrated to be effective, plus additional techniques to optimize remedial design. • Monitoring can track progress and achievement of RAOs. • RTD mitigates risk to human and ecological receptors and soil-to-groundwater/surface water risks encountered at the site through physical removal of contaminated soil. • Pump-and-treat remediates the Cr(VI) plumes. Other COCs (strontium-90, and nitrate) will be co-extracted with groundwater extracted for Cr(VI) treatment, where the concentration of strontium-90 and nitrate in the combined effluent will be less than their respective MCLs. Following the cessation of active treatment for Cr(VI), MNA will be the sole mechanism to achieve cleanup levels for strontium-90. • Pump-and-treat is projected to achieve risk standards in a period of about 56 years in the 100-D Area, 26 years in the 100-H Area, and 39 years in the horn area. • Institutional controls prevent exposure to affected soil and groundwater.
Compliance with ARARs	O	***Compliant*** • Remedial action/systems are designed to meet ARARs. • DWS and AWQC ARARs for Cr(VI) are projected be achieved in approximately 11 and 39 years of operation, respectively.
Long-term Effectiveness and Permanence	O	***Expected to have long-term effectiveness and permanence*** *Factors expected to perform very well against the long-term effectiveness and permanence criterion:* • RTD, hydraulic containment, and pump-and-treat achieve RAOs. • RTD using conventional equipment is reliable and practiced extensively at Hanford. ERDF disposal is effective and reliable. • Adequacy and reliability of pump-and-treat operations are proven at Hanford. • This alternative remediates soil site and groundwater COCs to PRG levels established to achieve the RAOs. Residual risks following cleanup to PRG levels will meet the 10^{-4} to 10^{-6} ELCR for radiological and carcinogenic COCs and will be less than or equal to an HI value of 1.0 for hazardous substances. • Long-term effectiveness and permanence will be evaluated in the CERCLA 5-year review process. *Factors that may provide some disadvantages or uncertainty to long-term effectiveness and permanence:* • Requires long-term enforcement of institutional controls to mitigate risk. • Cr(VI) treatment systems will operate for 39 years to achieve AWQC and may require major system renovations or replacement.

Table 10-5. Individual Criteria Evaluation for Alternative 4— RTD for Waste Sites and Pump-and-Treat for Groundwater

Criterion	Rating	Detailed Analysis
Reduction of TMV by Treatment	○	***Expected to have moderate reduction of TMV by treatment*** *Factors expected to perform moderately well against the reduction of TMV by treatment criterion:* • Significant mass of Cr(VI) will be removed through pump-and-treat where ion exchange treatment removes Cr(VI) from the groundwater. When the ion exchange resin capacity is reached, the resin will be changed out with fresh resin. Removal of Cr(VI) is not reversible since the spent resin loaded with Cr(VI) will be disposed as solid waste at ERDF or another EPA-approved disposal facility. • Removed soil is treated at ERDF, if needed *Factors that may provide some disadvantages or uncertainty to the reduction of TMV by treatment:* • Contaminated soil removal by RTD is not considered treatment. RTD is the primary remediation technology implemented for most of the waste sites; for this reason, this alternative performs moderately for this criterion.
Short-term Effectiveness	○	***Expected to have moderate short-term effectiveness*** *Factors expected to perform moderately well against the short-term effectiveness criterion:* • No adverse risks to the community from implementation of active treatment options because of the remote location and implementation of institutional controls. • Risks to workers from implementation of active treatment actions are minimized through an HSP and proper PPE. • Nominal short-term risks to workers during installation of groundwater extraction and conveyance piping, system O&M activities and monitoring. Risks are minimized through HSP and PPE. • Cr(VI) DWS and AWQC are achieved within 11 and 39 years of remedy implementation, respectively. *Factors that may provide some disadvantages or uncertainty to the short-term effectiveness:* • 56 years to achieve RAOs for strontium-90 in the unconfined aquifer. • Negative environmental impacts may include: - Generation of GHG from use of excavation equipment and disposal to ERDF; well installation, development; and operation of pump-and-treat system. - Waste generation from disposal of contaminated soil and spent ion exchange resin. • Large amounts of backfill will be required; borrow material may have significant environmental impacts.

Table 10-5. Individual Criteria Evaluation for Alternative 4— RTD for Waste Sites and Pump-and-Treat for Groundwater

Criterion	Rating	Detailed Analysis	
Implementability	○	***Expected to be implementable*** *Factors expected to perform very well against the implementability criterion:* • Conventional equipment and vendors for implementation of active treatment options (RTD, pump-and-treat) are readily available. • RTD of waste sites have been implemented and proven through interim actions at Hanford. • ERDF is proven, reliable, and accessible. • Pump-and-treat construction and operations have been implemented through interim actions, and the ability to construct and operate is proven at Hanford. • Radioactive decay is a confirmed attenuating process for strontium-90, which exhibits a sufficiently short radioactive half-life (29.1 years). • Institutional controls have been approved and implemented, and are reliable/effective at Hanford.	
Cost (in millions)[a]			
Waste Site Treatment		Capital	$145
		Annual O&M	$25.7
		Total Periodic[b]	$0.6
		Total non-discounted	$171.2
		NPV	$155.3
Groundwater Treatment		Capital	$20.4
		Annual O&M	$494.2
		Total Periodic[b]	$169
		Total nondiscounted	$683.6
		NPV	$445.5
Total Nondiscounted Cost of Alternative		$854.8	
Total Net Present Value (Discounted) of Alternative		$600.8	

a. Order-of-magnitude cost estimates and support information are presented in Appendix J. Costs for institutional controls are included under the costs for waste site treatment.

b. Periodic costs include O&M and/or construction activities, including costs to replace an installed remedy or components of an installed remedy, and services that are not included in initial capital costs or annual O&M costs. Periodic costs may be one-time costs or costs that recur at intervals over the life of the remedy.

10.3 Comparative Analysis of Remedial Alternatives

The comparative analysis of the proposed alternatives assists in identifying advantages and disadvantages of each alternative relative to the others to identify key tradeoffs that should be noted in the decision-making process. The previous sections presented individual evaluations for each alternative.

The remedial alternatives proposed for contaminated soil and groundwater at 100-D/H are compared in Table 10-6, and the analysis follows.

Table 10-6. Comparative Analysis of Alternatives

Criterion	Alternative 1 —No Action Alternative	Alternative 2—RTD and Grouting for Waste Site and Pump-and-Treat with Biological Treatment for Groundwater	Alternative 3—RTD and Grouting of Waste Site and Increased Capacity Groundwater Pump-and-Treat for Groundwater	Alternative 4—RTD for Waste Sites and Pump-and-Treat for Groundwater
	Rating	Rating	Rating	Rating
Overall Protection of HHE	No	Yes	Yes	Yes
Compliance with ARARs	No	Yes	Yes	Yes
Long-term Effectiveness and Permanence	N/A	★★☆	★★★	★★☆
Reduction of Toxicity, Mobility, or Volume by Treatment	N/A	★★☆	★★☆	★☆☆
Short-term Effectiveness	N/A	★★☆	★★★	★☆☆
Implementability	N/A	★☆☆	★★☆	★★★
Net Present Value of Alternative (Discounted)*	N/A	$506 million	$570.1 million	$600.8 million

* Detailed cost estimates are presented in Appendix J.

★☆☆ = Expected to perform less well with more disadvantages or uncertainty when compared to the other alternatives.
★★☆ = Expected to perform moderately well with some disadvantages or uncertainties when compared to the other alternatives
★★★ = Expected to perform best with less disadvantages or uncertainties when compared to the other alternatives.

The remedial alternatives for the vadose zone propose RTD for the waste sites. Alternatives 2 and 3 propose void filling with grout for waste site 100-H-36, containment by capping pipe ends and institutional controls for waste site 100-D-50:2, and MNA and institutional controls for waste sites 100-DR-9, 100-D-25, and 116-D-8. Alternative 3 proposes an expanded pump-and-treat system for remediation of groundwater contamination. Alternatives 2 and 4 use the capacity of the existing pump-and-treat systems, and Alternative 2 includes biological treatment through bioinjection to treat groundwater. Groundwater remediation to achieve RAOs for Alternatives 2 and 4 is projected to take longer than Alternative 3, with Alternative 4 taking longer than Alternative 2. Remedy performance time frames for Alternatives 2, 3, and 4 are based on successful source area remedial action.

10.3.1 Overall Protection of Human Health and the Environment

Alternative 1 is not protective of human health and the environment because RAOs and ARARs are not achieved in a reasonable period.

Alternatives 2, 3, and 4 will achieve the RAOs and meet this threshold criterion. Proposed actions for groundwater are projected to achieve cleanup criteria for Cr(VI) within 25, 12, and 39 years from the start of alternative implementation for Alternatives 2, 3, and 4, respectively. Cleanup levels for all groundwater COCs are projected to be achieved within 56, 44, and 56 years for Alternatives 2, 3, and 4, respectively. For affected waste sites, treatment alternatives such as RTD and void fill grouting effectively control or prevent significant risks. Unacceptable risks are also prevented or controlled through implementation of institutional controls as needed.

10.3.2 Compliance with ARARs

Alternative 1 will not comply with soil cleanup requirements or groundwater chemical-specific ARARs because of the potential for plume migration to the river. Since Alternative 1 does not comply with ARARs nor will it be protective of human health and the environment (the two threshold criteria), it was not evaluated for the balancing criteria.

Alternatives 2, 3, and 4 comply with ARARs and meet this threshold criterion. Remedial actions and treatment systems proposed under these alternatives would be designed to meet ARARs. For groundwater, proposed remedies will achieve DWS and AWQC ARARs in a reasonable time frame.

10.3.3 Long-Term Effectiveness and Permanence

Alternative 3 is evaluated to perform better than Alternatives 2 and 4 based on longer time to remediate groundwater in Alternatives 2 and 4, which potentially require need for replacement of pump-and-treat systems and components in the two alternatives. Proposed actions for groundwater are projected to achieve cleanup criteria for Cr(VI) within 25, 12, and 39 years from start of alternative implementation for Alternatives 2, 3, and 4, respectively. With projected pump-and-treat facility design life of 25 years, systems would need major renovations under Alternative 2, and Alternative 4, which uses existing systems that have been in place since 2010.

The alternatives remediate soil site and groundwater COCs to PRG levels established to achieve the RAOs. Residual risks following cleanup to PRG levels will meet the 10^{-4} to 10^{-6} ELCR for radiological and carcinogenic COCs and will be less than or equal to an HI value of 1.0 for hazardous substances.

For waste sites, Alternatives 2, 3, and 4 use RTD and disposal at ERDF to remove contamination, which are demonstrated to be adequate, effective, and reliable at the Hanford Site. Alternatives 2 and 3 also use void fill grouting of waste site 100-H-36, containment by capping pipe ends, and institutional controls for waste site 100-D-50:2, and MNA and institutional controls for waste sites 100-DR-9, 100-D-25, and 116-D-8. Alternative 4 remediates these sites with RTD and disposal at ERDF.

Pump-and-treat is used in Alternatives 2, 3, and 4 for treatment of groundwater. The adequacy and reliability of pump-and-treat is demonstrated at the Hanford Site through the interim actions. Alternative 3 remediates groundwater using expanded pump-and-treat. Alternative 2 uses pump-and-treat enhanced with biological treatment. Alternative 3 provides for greater pump-and-treat system robustness to address the uncertainties identified in the CSM described in Section 4.9 associated with locations where groundwater monitoring indicates potential residual contamination in the vadose zone that may contribute to groundwater contamination. Alternative 3 enables more flexibility for more responsive system adjustments through the RPO because of the expanded well field and increased system capacity.

Alternatives 2, 3, and 4 will achieve DWSs and AWQC ARARs in a reasonable time. The adequacy and reliability of institutional controls is proven at Hanford.

Alternatives 2, 3, and 4 propose implementation of monitoring and include the CERCLA 5-year review process, which would evaluate effectiveness of the proposed actions and track progress towards achievement of RAOs.

10.3.4 Reduction of Toxicity Mobility and Volume through Treatment

Alternatives 2 and 3 are evaluated to perform moderately well for reduction of TMV and better than Alternative 4. RTD is the primary technology implemented for Alternatives 2, 3, and 4 for remediation of waste sites. Contaminated soil removal by RTD is not considered treatment. Alternative 4 implements RTD exclusively for remediation of waste sites and, therefore, provides the least reduction in TMV by treatment among the alternatives.

Void fill grouting in Alternatives 2 and 3 provide for treatment of 4,550 m^3 (160,600 ft^3) of contaminated soil/debris.

For groundwater, Alternatives 2, 3, and 4 provide substantial reduction of TMV through pump-and-treat and MNA/institutional controls. Under Alternative 2, reduction of TMV is also achieved through treatment via biological reduction (bioinjection).

10.3.5 Short-Term Effectiveness

Alternatives 2, 3, and 4 are compared primarily against the time for the alternative to achieve RAOs. Alternative 3 achieves RAOs sooner than Alternative 2 and Alternative 4. Alternative 4 takes longer to achieve RAOs than both Alternative 2 and Alternative 3. Time to achieve RAOs following remedy implementation, to achieve AWQC in the unconfined aquifer, is 12, 25, and 39 years, respectively, for Alternatives 3, 2, and 4.

The alternatives all protect the community and workers during remedial actions. Potential effects to the community and workers from implementing any actions onsite would be controlled and mitigated through effective HSPs and adequate PPE.

Effects on the environment for all alternatives result from waste site areas exposed to the environment with machinery-generated dust, generation of remediation process waste, and GHG emissions (from excavation equipment and transportation of material to ERDF). All alternatives will have some negative environmental impacts as follows:

- Ecological disturbances and generation of GHG from use of excavation equipment and disposal to ERDF, installation of well pads and construction and operation of pump-and-treat system
- Waste generation from disposal of contaminated soil and spent ion exchange resin
- Energy consumption from process equipment
- Potential cultural resource effects

10.3.6 Implementability

Alternative 4 uses proven technologies that are currently being implemented for interim remedial actions (i.e., RTD and pump-and-treat) and is considered to have the best implementability compared to the other alternatives.

Alternatives 2 and 3 implement other technologies for waste site treatment (e.g., void fill grouting) that will likely require additional testing of grout material during remedial design. Alternative 2 also implements biological treatment that will likely require design testing to establish biological reducing zone parameters for design of in situ bioremediation treatment of groundwater.

Although biological treatment proposed in Alternative 2 has been proven, the additional design testing leads to comparatively lower implementability compared to the other alternatives.

10.3.7 Cost

Estimated design, construction, O&M, and decommissioning costs were developed for Alternatives 2, 3, and 4. The total estimated net present value is $506 million for Alternative 2, $570 million for Alternative 3, and $601 million for Alternative 4. The higher cost estimate for Alternative 4 is mainly due to the longer operational period of the pump-and-treat system.

Cost estimates for Alternative 3 include new facility construction to provide identified increased pump-and-treat capacity. The RD/RAWP will determine if new facilities are needed or if modifications to existing pump-and-treat facilities will provide needed capacity.

Details regarding cost estimates for Alternatives 2, 3, and 4 are provided in Appendix J. These cost estimates have been prepared to meet the -30 to +50 percent range of accuracy recommended in the CERCLA RI/FS Guidance (EPA/540/G-89/004). No capital or O&M costs are associated with Alternative 1.

Overall, Alternative 2 has the lowest cost estimate.

10.4 NEPA Values

This section addresses the incorporation of NEPA values into CERCLA documents. This is consistent with the *National Environmental Policy Act Compliance Program* (DOE O 451.1B, Change 2), which requires that CERCLA actions address and incorporate NEPA values such as socioeconomic, ecological, offsite, and cumulative effects in CERCLA documents to the extent practicable.

Alternatives to address contamination at 100-D/H are presented in Chapter 9, and include Alternative 1 (No Action [as required by the NCP]); Alternative 2 (RTD and Grouting for Waste Site and Pump-and-Treat with Biological Treatment for Groundwater); Alternative 3 (RTD and Grouting of Waste Site and Increased Capacity Groundwater Pump-and-Treat for Groundwater); and Alternative 4 (RTD for Waste Sites and Pump-and-Treat for Groundwater). The No Action Alternative would not mitigate the environmental impacts from contaminated waste sites and groundwater. All other alternatives could mitigate the effects associated with affected waste sites and groundwater present at 100-D/H.

NEPA values associated with remediation are based on the information presented in this RI/FS, including the area and site characteristics (Chapters 1, 2, and 3), final COPCs (Chapter 8), and identification and analysis of remedial actions (Chapter 9). Applying a "sliding scale" of NEPA analysis to 100-D/H (using *Recommendations for the Preparation of Environmental Assessments and Environmental Impact Statements: Second Edition* [DOE, 2004]) and considering the CERCLA ARARs (detailed in Section 8.1.2), the principle resource areas of concern include the contaminants in waste sites, contaminants in the groundwater and surface water, solid and liquid radioactive and hazardous waste management, air emissions, potential adverse effects to historic and cultural resources, ecological resources, socioeconomics (including environmental justice concerns), and transportation.

The net anticipated effect from implementation of Alternatives 2, 3, or 4 could be an overall positive contribution to cumulative environmental effects at the Hanford Site. For sites and alternatives with RTD

as the selected remedial action, DOE expects that the primary facility to receive contaminated soil will be the ERDF. NEPA values in the planning for the ERDF operation were explained in detail in the *NEPA Roadmap for the Environmental Restoration Disposal Facility Regulatory Package* (DOE/RL-94-41) for the *Remedial Investigation and Feasibility Study Report for the Environmental Restoration Disposal Facility* (DOE/RL-93-99) as described in the most recent ERDF ROD Amendment (*Amendment to the Record of Decision for the USDOE Hanford Environmental Restoration Disposal Facility* [EPA et al., 2007]).

In addition, DOE has included the combined effects anticipated from ongoing CERCLA/TPA (Ecology et al., 1989a) response actions as part of the cumulative impact analysis in the *Draft Tank Closure and Waste Management Environmental Impact Statement for the Hanford Site, Richland, Washington* (DOE/EIS-0391), which includes a Sitewide cumulative impact groundwater analysis. This has presented the public with a separate opportunity for comment as part of that NEPA process. The EIS is being used to inform the public concerning ongoing implementing cleanup actions on the Hanford Site. The NEPA values (i.e., resource area and relevant NEPA considerations) most relevant to and potentially affected by the actions taking place under this remedial action are described in Table 10-7.

10.5 Coordination of Interim and Final CERCLA Remedial Activities

A feature of each alternative is the ongoing implementation of interim action RODs, CERCLA removal actions, RCRA corrective actions, treatability tests, and other activities to remediate contaminated areas or to develop more effective methods that advance remediation.

Implementation of these interim action ROD activities is generating information that allows an improved understanding of site complexity, supports refinement of the CSM, and documents the effectiveness of the remedial actions.

Cleanup of waste sites, in accordance with the interim action RODs and focused FSs, is ongoing and expected to continue until final action RODs are in place. As remedial actions under interim action RODs are completed, verification sampling and laboratory analyses are performed to document the extent to which remedial action goals established under the interim action RODs have been met. This information will be essential to implementing final action RODs.

There are many buildings and structures in the 100 Area. The buildings and structures are evaluated for removal, usually using a CERCLA removal action. Once these structures are demolished and decommissioned under CERCLA non-time-critical removal actions, samples of the residual soil may be collected for analysis. If the analytical results indicate the area is contaminated, the area is considered a potential waste site. The area is then evaluated, and a remedy is selected in accordance with the interim action ROD.

To support a final action remedy at each OU, the current remedial actions under interim action RODs for the 100 Area OUs will continue until issuance of the final action ROD.

During the period between when the final action ROD is approved and when the required RD/RAWP is prepared and issued, DOE-RL plans to continue remedial activities such as waste site RTD and groundwater treatment. In order for these actions to be consistent with the final action remedy selection, the current interim action RD/RAWPs will be modified using the TPA (Ecology et al., 1989a) change notice process to include the cleanup levels specified in the final action ROD.

Table 10-7. NEPA Values Evaluation

NEPA Values	Description	Evaluation*
Transportation	Considers effects of the proposed action on local traffic (i.e., traffic at the Hanford Site) and traffic in the surrounding region.	Implementation of all action Alternatives (except No Action) would be expected to produce short-term effects on local traffic. A majority of the effect relates to increased truck traffic associated with the aforementioned Alternatives; they would involve transport of contaminated soil moved from a waste site(s) to ERDF. Transportation effects were considered in the *Remedial Investigation and Feasibility Study Report for the Environmental Restoration Disposal Facility* (DOE/RL-93-99) as part of the evaluation of short-term effectiveness and implementability. NEPA values in the planning for the ERDF operation were explained in detail in the *NEPA Roadmap for the Environmental Restoration Disposal Facility Regulatory Package* (DOE-RL-94-41). Transportation effects associated with a waste site under Alternatives 2, 3, and 4 are similar, though Alternative 2 and Alternative 3 effects are somewhat smaller than for Alternative 4, given the smaller volume of soil to be excavated and transported to ERDF. The discussion of cumulative effects provides a perspective of transportation to ERDF.
Water Quality	Considers potential water quality concerns associated with the groundwater below the site and the Columbia River.	All alternatives include source area remediation and implement P&T to restore groundwater to beneficial uses. Contaminants would be removed from groundwater in all alternatives, which would be expected to cause a net positive effect in water quality both for groundwater below the site and for the Columbia River.
Air Quality	Considers potential air quality concerns associated with emissions generated during the proposed action.	Airborne releases associated with Alternatives 2, 3, and 4 could occur because of dust generation during excavation. Any potential of airborne release of contaminants during alternative remedial actions would be controlled in accordance with DOE radiation control and air pollution control standards to minimize emissions of air pollutants at the Hanford Site and protect all communities outside the site boundaries. Operation of trucks and other diesel-powered equipment for these alternatives would be expected to introduce quantities of carbon dioxide, sulfur dioxide, nitrogen dioxide, particulates, and other pollutants to the atmosphere. Alternative 4 is expected to have a slightly greater extent of these emissions since the larger quantity of soil would be excavated and transported. As applicable, vehicular and equipment emissions would be controlled and mitigated in compliance with the substantive standards for air quality protection that apply to the Hanford Site.

Table 10-7. NEPA Values Evaluation

NEPA Values	Description	Evaluation*
Natural, Cultural, and Historic Resources	Considers effects of the proposed action on wildlife, wildlife habitat, archeological sites and artifacts, and historically significant properties.	Effects on ecological resources near the remedial actions could result from installation of groundwater wells and excavation of contaminated soil. Such effects would be mitigated in accordance with the *Hanford Site Biological Resources Management Plan* (DOE/RL-96-32) and *Biological Resources Mitigation Strategy* (DOE/RL-96-88), and applicable standards of all relevant biological species protection regulations. Effects to other cultural values will be minimized through implementation of *Hanford Cultural Resources Management Plan* (DOE/RL-98-10), *Revised Mitigation Action Plan for the Environmental Restoration Disposal Facility* (DOE/RL-2005-27), and consultation with area Tribes, as needed. This will help ensure appropriate mitigation to avoid or minimize any adverse effects to natural and cultural resources and address any other relevant concerns. Potential effects to cultural and historic resources that may be encountered during short-term construction activities associated with implementing the action would be mitigated through compliance with the appropriate substantive requirements of the *National Historic Preservation Act of 1966* and other ARARs related to cultural preservation.
Socioeconomic Effects	Considers effects pertaining to employment, income, other services (e.g., water and power utilities), and the effect of implementation of the proposed action on the availability of services and materials.	The proposed action is within the scope of current DOE-RL environmental restoration activities and would have minimal effect on the current availability of services and materials. This work would be expected to be accomplished largely using employees from the existing contractor workforce. Even if the remedial activities create additional service sector jobs, the total expected increase in employment would be expected to be less than 1 percent of the current employment levels. The socioeconomic effect of the project would contribute to the continuing overall positive employment and economic effects on eastern Washington communities from Hanford Site cleanup operations.

Table 10-7. NEPA Values Evaluation

NEPA Values	Description	Evaluation*
Environmental Justice	Considers if the proposed response actions would have inappropriately or disproportionately high and adverse human health or environmental effects on minority or low income populations.	Per *Federal Actions to Address Environmental Justice in Minority Populations and Low-Income Populations* (Executive Order 12898), DOE seeks to ensure that no group of people bears a disproportionate share of negative environmental consequences resulting from proposed federal actions. There are no effects associated with proposed activities associated with 100-D/H that could reasonably be determined to affect any member of the public; therefore, they would not have the potential for high and disproportional adverse effects on minority or low income groups.
Cumulative Effects (Direct and Indirect)	Considers if the proposed action could have cumulative effects on human health or the environment when considered together with other activities locally, at the Hanford Site, or in the region.	The environmental concern of 100-D/H is associated directly with the targeted area. Because of the temporary nature of the activities and their remote location, cumulative effects on air quality or noise with other Hanford Site or regional construction and cleanup projects would be minimal. When soil at a site in this area is found to be contaminated with hazardous substances in concentrations presenting a material threat to human health and the environment, that threat would be mitigated. The net anticipated effect could be a positive contribution to cumulative environmental effects at the Hanford Site through RTD of such hazardous substances and COCs into a facility that has been designed and legally authorized to contain such contaminants safely, like ERDF. Contaminated soil removed under any alternative would meet the ERDF waste acceptable criteria as described in the *Environmental Restoration Disposal Facility Waste Acceptance Criteria* (WCH-191). Wastes generated during implementation of the proposed alternatives would be manageable within the capacities of existing facilities. For perspective, ERDF received more than 700,000 tons of waste in CY 2008 and more than 430,000 tons in CY 2007). Radiological contamination is expected to be minimal. ERDF received approximately 22,500 Ci in CY 2008 and approximately 13,000 Ci in CY 2007.
Mitigation	Considers that, if adverse effects cannot be avoided, response action planning should minimize them to the extent practicable. This value identifies required mitigation activities.	Compliance with the substantive requirements of the ARARs would mitigate potential environmental impacts on the natural environment, including migratory birds and endangered species. DOE has established policies and procedures for the management of ecological and cultural resources when actions might affect such resources (*Hanford Site Biological Resources Management Plan* [DOE/RL-96-32], *Biological Resources Mitigation Strategy* [DOE/RL-96-88], and *Hanford Cultural Resources Management Plan* [DOE/RL-98-10]). Cultural resource and biological species reviews/surveys are undertaken that also provide suggested mitigation activities to assure adverse effects associated with implementing the actions are minimized or avoided. Health and safety procedures, documented in the HSPs established by site contractors, would mitigate risks to workers from the remedial activities.

Table 10-7. NEPA Values Evaluation

NEPA Values	Description	Evaluation*
Irreversible and Irretrievable Commitment of Resources	Considers the use of nonrenewable resources for the proposed response actions and the effects that resource consumption would have on future generations. (When a resource [e.g., energy minerals, water] or wetlands is used or destroyed and cannot be replaced within a reasonable amount of time, its use is considered irreversible.)	Materials that would be used to backfill waste sites could be taken from the surrounding area to contour the backfill to match the surrounding area. For Alternatives 2, 3, and 4, normal usage of resources during construction activities, such as fuel and water, would be irreversibly used. Potential effects would be expected to be greater for Alternative 4, because of the slightly larger extent of RTD. Restoration of formerly disturbed areas to a more natural state would be expected to result in a net benefit to the ecological and visual resources within the region.

* Includes the evaluation for each alternative.

10.6 CERCLA and RCRA Corrective Action and TSD Unit Closure

The TPA (Ecology et al., 1989a) states the intent of the Tri Parties' CERCLA remediation at the Hanford Site is to fulfill the corrective action requirements for the Site as a facility containing permitted TSD units. The TPA (Ecology et al., 1989a) guides integration and coordination of CERCLA and RCRA at the Hanford Site. The following articles explain the relations of CERCLA remedial actions and RCRA corrective actions:

- Article IV, Paragraph 17, which cites the Tri-Parties' intent "to integrate DOE's CERCLA response obligations and RCRA corrective action obligations that relate to the release(s) of hazardous substances, hazardous wastes, pollutants and contaminants" covered by the TPA (Ecology et al., 1989a).
- Article XIV applies to the performance of both CERCLA remedial action and RCRA corrective action.
- Article XXIII acknowledges the potential for overlap between CERCLA and RCRA cleanup.
- Article XXIV specifies the approach for regulatory oversight.

Section 5.4 of the TPA Action Plan (Ecology et al., 1989b) addresses the rationale and approach for past-practice cleanup. Two key objectives are to "ensure that only one past-practice program will be applied at each operable unit," and that "the process selected be sufficiently comprehensive to satisfy the technical requirements of both statutory authorities and the respective regulations."

DOE's corrective action obligation on the Hanford Site is addressed in the RCRA Hanford Facility Permit (*Hanford Facility Resource Conservation and Recovery Act Permit, Dangerous Waste Portion, Revision 8C, for the Treatment, Storage, and Disposal of Dangerous Waste* [WA7890008967]), Condition II.Y.2.a, which provides that DOE corrective action obligations are met through adherence to the TPA (Ecology et al., 1989a). In particular, "Overview of Cleanup Standards" (WAC 173-340-700) through "Sediment Cleanup Standards" (WAC 173-340-760) functions as ARAR standards for CERCLA remedial actions on the Hanford Site.

In addition, the final disposition of any remaining TSD units will address both CERCLA remedial action and RCRA TSD closure requirements. Coordinating RCRA TSD closure requirements and CERCLA remedial actions allows for the streamlining of cleanup options for disposal, closure, removal, and/or remedial actions. By allowing flexibility in final disposal options, the Tri-Parties intend to minimize disposal costs to the extent possible while remaining fully protective of human health and the environment. If necessary, a separate TSD unit closure plan that meets the requirements specified in "Dangerous Waste Regulations," "Closure and Post-Closure" (WAC 173-303-610) and the *Hanford Facility Resource Conservation and Recovery Act Permit, Dangerous Waste Portion, Revision 8C, for the Treatment, Storage, and Disposal of Dangerous Waste* (WA7890008967) will be prepared and submitted to Ecology.

The regulatory integration process will address each technical and procedural element of RCRA and CERCLA so that redundant work is not performed when remediating/closing the TSD unit. The CERCLA public involvement process, including public notice and opportunity to comment, will be adjusted as necessary to concurrently satisfy the public involvement requirements for the RCRA closure and corrective action processes.

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Appendices
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