SECTION

2 OF 4

Table 4-28. Groundwater Summary Statistics for 100-BC

Analyte Name	Filtered	Units	Number of Results	Number of Detects	Frequency of Detects (%)	Min Non-Detect	Max Non-Detect	Min Detect	Max Detect	Mean Detect	Median Detect	Background 90th Percentile	No. of Detects > Background	Action Level	No. of Detects > Action Level	Action Level Basis
Iron	Y	ug/L	127	46	36	9.0	38	9.5	71	26	24	570	--	300	--	40 CFR 141 - Federal MCL
Lead	N	ug/L	54	4	7	0.1	0.2	0.2	0.3	0.25	0.24	0.92	--	2.1	--	WAC 173-201A
Lead	Y	ug/L	54	0	0	0.1	0.2	--	--	--	--	0.92	--	2.1	--	WAC 173-201A
Manganese	N	ug/L	109	15	14	0.96	6.0	4.0	41	12	9	39	1	50	--	40 CFR 141 - Federal MCL
Manganese	Y	ug/L	127	19	15	0.84	6.0	1.8	39	9	6	39	1	50	--	40 CFR 141 - Federal MCL
Mercury	N	ug/L	54	0	0	0.050	0.10	--	--	--	--	0.003	--	0.012	--	40 CFR 131 -- Freshwater CCC
Mercury	Y	ug/L	54	0	0	0.016	0.10	--	--	--	--	0.003	--	0.012	--	40 CFR 131 -- Freshwater CCC
Nickel	N	ug/L	109	19	17	4.0	13	4.0	12	5.4	4.7	1.6	19	52	--	Clean Water Act -- Freshwater CCC
Nickel	Y	ug/L	127	7	5.5	4.0	13	4.2	7.1	5.7	5.2	1.6	7	52	--	Clean Water Act -- Freshwater CCC
Selenium	N	ug/L	54	44	81	0.60	0.60	0.58	4.2	1.4	1.2	11	--	5.0	--	Clean Water Act -- Freshwater CCC
Selenium	Y	ug/L	54	49	91	0.60	0.60	0.40	8.3	1.5	1.2	11	--	5.0	1	Clean Water Act -- Freshwater CCC
Thallium	N	ug/L	54	1	1.9	0.050	0.10	1.0	1.0	1.00	1.00	1.7	--	0.24	1	Clean Water Act -- Human Health Water + Organism
Thallium	Y	ug/L	54	1	1.9	0.050	0.10	1.2	1.2	1.2	1.2	1.7	--	0.24	1	Clean Water Act -- Human Health Water + Organism
Uranium	N	ug/L	54	54	100	--	--	1.1	7.4	2.1	1.6	9.9	--	30	--	40 CFR 141 - Federal MCL
Zinc	N	ug/L	109	24	22	4.0	9.0	4	23	11	9	21.8	1	91	--	WAC 173-201A
Zinc	Y	ug/L	127	31	24	4.0	9.0	1.7	22	8	6	22	1	91	--	WAC 173-201A
Other																
Total petroleum hydrocarbons - diesel range	N	ug/L	54	2	3.7	17	70	180	220	200	200	--	--	500	--	WAC 173-340-900, Table 720-1

Sources:

40 CFR 131, "Water Quality Standards."

40 CFR 141, "National Primary Drinking Water Regulations."

40 CFR 141.66, "National Primary Drinking Water Regulations," "Maximum Contaminant Levels for Radionuclides."

WAC 173-201A, "Water Quality Standards for Groundwaters of the State of Washington."

WAC 173-340-720, "Model Toxics Control Act—Cleanup," "Groundwater Cleanup Standards."

WAC 173-340-730, "Model Toxics Control Act—Cleanup," "Surface Water Cleanup Standards."

MCL = maximum contaminant level

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4.3.1.1 Final COPCs Warranting Evaluation in the FS

Section 6.3 identifies the COPCs that warrant further evaluation in the FS for each of the exposure areas evaluated in the supplemental groundwater risk evaluation. Cr(VI), strontium-90 and tritium are identified as COPCs in the 100-BC-5 Groundwater OU that warrant further evaluation in the FS.

Concentrations of Cr(VI) are widely distributed and consistently present at concentrations above the AWQC. Strontium-90and tritium are identified as COPCs because groundwater concentrations above the drinking water standards are present in localized areas. The following paragraphs provide a summary for each COPC. Additional information regarding trend plots and contours are provided in Section 4.3.2.

Strontium-90 was identified as a COPC in the 100-BC Work Plan (DOE/RL-2008-46-ADD3) because it was detected above the drinking water standard. Strontium-90 was detected in 50 of 105 unfiltered groundwater samples (48 percent) and was not analyzed for in filtered groundwater samples. Strontium-90 concentrations ranged between 0.52 and 49 pCi/L in unfiltered groundwater samples. Strontium-90 was reported above the drinking water standard of 8 pCi/L in 76 percent of the unfiltered detected results. Strontium-90 concentrations greater than the drinking water standard were reported at seven wells:

- One of five sample results from 199-B2-14 (undetected to 8.6 pCi/L)
- All nine sample results from 199-B3-1 (20 to 42 pCi/L)
- All eight sample results from 199-B3-46 (15 to 49 pCi /L)
- All 10 sample results from 199-B3-47 (14 to 37 pCi /L)
- Two of four sample results from 199-B4-1 (4.2 to 21.2 pCi /L)
- Three of five sample results from 199-B4-4 (undetected to 18.6 pCi /L)
- Five of six sample results from 199-B5-2 (7.4 to 15 pCi /L)

The results of this evaluation indicate that strontium-90 has been detected historically in groundwater samples at concentrations greater than the drinking water standard of 8 pCi/L. Concentrations of strontium-90 have been reported historically above the Hanford Site 90th percentile (filtered) background level of 0.001 pCi/L.

Tritium was identified as a COPC in the 100-BC Work Plan (DOE/RL-2008-46-ADD3) because it was detected above the drinking water standard. Tritium was detected in 137 of 154 unfiltered groundwater samples (89 percent) and was not analyzed for in filtered groundwater samples. Tritium concentrations ranged between 190 and 125,000 pCi/L. Tritium was reported above the drinking water standard of 20,000 pCi/L in 31 percent of the unfiltered detected results. Tritium concentrations greater than the drinking water standard were reported at six wells:

- Nine of 10 sample results from 199-B3-47 (20,000 to 47,000 pCi/L)
- Three of six sample results from 199-B4-1 (13,000 to 37,900 pCi/L)
- Two of eight sample results from 199-B5-2 (12,000 to 125,000 pCi /L)
- Four of eight sample results from 199-B8-6 (4,600 to 31,000 pCi /L)
- Eleven of 13 sample results from 199-B8-7 (16,000 to 35,600 pCi /L)
- All 13 sample results from 199-B8-8 (28,000 to 59,000 pCi /L)

The results of this evaluation indicate that tritium has been detected historically in groundwater at concentrations greater than the drinking water standard. Tritium has been reported historically at concentrations above the Hanford Site 90th percentile concentration of 119 pCi/L.

Cr(VI) was identified as a COPC in the 100-BC Work Plan (DOE/RL-2008-46-ADD3) because it was detected above the AWQC. Cr(VI) was detected in 124 of 129 unfiltered samples (96 percent) and 87 of 94 filtered groundwater samples (93 percent). Cr(VI) concentrations range between 2.4 and 64 µg/L in unfiltered groundwater samples and between 2.4 and 58 µg/L in filtered groundwater samples[1]. Cr(VI) was reported above the AWQC of 10 µg/L in 79 percent of the unfiltered detected results and 84 percent of the filtered detected results. Cr(VI) concentrations greater than the AWQC were reported at 18 well locations. MDLs ranged between 2.0 and 5.0 µg/L, which are less than the AWQC of 10 µg /L. The results of this evaluation indicate the Cr(VI) has been detected historically in groundwater samples at concentrations consistently above the AWQC of 10 µg /L at 18 of the 24 monitoring locations included in the dataset.

4.3.1.2 Analytes of Interest

Analytes of interest are either those analytes that were identified as COPCs in the 100-BC SAP (DOE/RL-2009-44) or those analytes for which a maximum concentration exceeding an action limit was reported during the spatial and temporal sampling (Section 6.3). The following descriptions include radionuclides, VOCs, anions, and metals.

Radionuclides. Carbon-14 was identified as a COPC in the 100-BC Work Plan (DOE/RL-2008-46-ADD3). Although carbon-14 was detected at concentrations below the , there were limited analytical results available for evaluation to determine its presence or absence in the 100-BC-5 Groundwater OU. Carbon-14 was not detected in any of the unfiltered groundwater samples analyzed (54 samples) and was not analyzed for in filtered groundwater samples. All samples included in this evaluation were collected as part of the spatial and temporal sampling activities of the RI. The results of this evaluation indicate that carbon-14 was not detected in RI groundwater samples.

Cesium-137 was identified as a COPC in the 100-BC Work Plan (DOE/RL-2008-46-ADD3). Although it was detected at concentrations below the drinking water standard, there were limited analytical results available for evaluation to determine its presence or absence in the 100-BC-5 Groundwater OU. Cesium-137 was detected in one of 54 unfiltered groundwater samples (1.9 percent) and was not analyzed for in filtered groundwater samples. The single detection was reported at a concentration of 2 pCi/L which is less than the drinking water standard of 200 pCi/L. All samples included in this evaluation were collected as part of the spatial and temporal sampling activities of the RI. The results of this evaluation indicate that cesium-137 has been detected in RI groundwater samples at a low frequency (1.9 percent) at concentrations less than the drinking water standard of 200 pCi/L. The single detection of cesium-137 was reported at a concentration less than the Hanford Site 90th percentile (filtered) background level of 8.6 pCi/L.

Cobalt-60 was identified as a COPC in the 100-BC Work Plan (DOE/RL-2008-46-ADD3). Although it was detected at concentrations below the drinking water standard, there were limited analytical results available for evaluation to determine its presence or absence in the 100-BC-5 Groundwater OU. Cobalt-60 was not detected in any unfiltered groundwater samples (54 samples) and was not analyzed for in filtered groundwater samples. All samples included in this evaluation were collected as part of the spatial and temporal sampling activities of the RI. The results of this evaluation indicate that cobalt-60 was not detected in RI groundwater samples.

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[1] This section discusses groundwater results from January 5, 2006 through January 26, 2011. A total chromium result from February 2012 also exceeded the action level (Section 4.3.2).

Europium-155 was identified as a COPC in the 100-BC Work Plan (DOE/RL-2008-46-ADD3). Although it was not detected there were limited analytical results available to determine its presence or absence in the 100-BC-5 Groundwater OU. Europium-155 was not detected in any unfiltered groundwater sample (54 samples) and was not analyzed for in filtered samples. All samples included in this evaluation were collected as part of the spatial and temporal sampling activities of the RI. The results of this evaluation indicate that europium-155 was not detected in RI groundwater samples.

Gross alpha was identified as a COPC in the 100-BC Work Plan (DOE/RL-2008-46-ADD3) as an indicator parameter. Gross alpha was detected in 26 of 117 unfiltered groundwater samples (22 percent) and was not analyzed for in filtered groundwater samples. Gross alpha concentrations ranged between 1.0 and 9.6 pCi/L in unfiltered samples, all of which are below the drinking water standard of 15 pCi/L. The results of this evaluation indicate that gross alpha has been detected historically in groundwater samples at concentrations less than the drinking water standard of 15 pCi/L.

Gross beta was identified as a COPC in the 100-BC Work Plan (DOE/RL-2008-46-ADD3) as an indicator parameter. Gross beta was detected in 115 of 117 unfiltered (98 percent) and was not analyzed for in filtered groundwater samples. Gross beta concentrations ranged between 3.6 and 110 pCi/L in unfiltered samples. It should be noted that the presence of elevated gross beta concentrations is generally co-located with the presence of elevated technetium-99 and strontium-90 concentrations.

Iodine-129 was identified as a COPC in the 100-BC Work Plan (DOE/RL-2008-46-ADD3). Although it was detected at concentrations below the drinking water standard, there were limited analytical results available for evaluation to determine its presence or absence in the 100-BC-5 Groundwater OU. Iodine-129 was not detected in any of the unfiltered groundwater samples (54 samples) and was not analyzed for in filtered groundwater samples. All samples included in this evaluation were collected as part of the spatial and temporal sampling activities of the RI. The results of this evaluation indicate that iodine-129 was not detected in RI groundwater samples.

Nickel-63 was identified as a COPC in the 100-BC Work Plan (DOE/RL-2008-46-ADD3). Although it was not detected there were limited analytical results available to determine its presence or absence in the 100-BC-5 OU. Nickel-63 was not detected in any of the unfiltered groundwater samples (54 samples) analyzed. All samples included in this evaluation were collected as part of the spatial and temporal sampling activities of the RI. The results of this evaluation indicated that nickel-63 was not detected in RI groundwater samples.

Radium-228 was identified as a COPC in the 100-BC Work Plan (DOE/RL-2008-46-ADD3). There were limited analytical data available to determine the presence or absence of it in the 100-BC-5 Groundwater OU. Radium-228 was not detected in any unfiltered groundwater samples (54 samples) and was not analyzed for in filtered groundwater samples. All samples included in this evaluation were collected as part of the spatial and temporal sampling activities of the RI. The results of this evaluation indicate that radium-228 was not detected in RI groundwater samples.

Technetium-99 was identified as a COPC in the 100-BC Work Plan (DOE/RL-2008-46-ADD3). Although it was detected at concentrations below the drinking water standard, there were limited analytical results available for evaluation. Technetium-99 was detected in 31 of 69 unfiltered groundwater samples (45 percent) and was not analyzed for in filtered groundwater samples. Technetium-99 concentrations range between 5.9 to 26 pCi/L. The results of this evaluation indicate that technetium-99 has been detected historically in groundwater samples at concentrations below the drinking water standard of 900 pCi/L. Techetium-99 concentrations have been reported historically above the Hanford Site 90th percentile (unfiltered) background level of 0.83 pCi/L.

Volatile Organic Compounds. 1,1,2,2-Tetrachloroethane was identified as a COPC in the 100-BC Work Plan (DOE/RL-2008-46-ADD3) because it was detected infrequently above the action level and all MDLs were greater than the action level. 1,1,2,2-Tetrachloroethane was not detected in any unfiltered groundwater samples (54 samples) and was not analyzed for in filtered groundwater samples. The action level for 1,1,2,2-tetrachloroethane is 0.17 µg/L based on the *National Recommended Water Quality Criteria* (EPA, 2009b), "Human Health Water + Organism" value; however, it defaults to the estimated quantitation limit (EQL) of 1.5 µg/L listed in the 100-BC SAP (DOE/RL-2009-44) when the analytical method cannot achieve the action level. MDLs were reported as 0.1 µg/L which is less than the EQL of 1.5 µg/L. All samples included in this evaluation were collected as part of the spatial and temporal sampling activities of the RI. The results of this evaluation indicate that 1,1,2,2,-tetrachloroethane was not detected in RI groundwater samples.

1,1-Dichloroethene was identified as a COPC in the 100-BC Work Plan (DOE/RL-2008-46-ADD3) because all MDLs were greater than the action level. 1,1-Dichloroethene was not detected in any unfiltered groundwater samples (54 samples) and was not analyzed for in filtered groundwater samples. The action level for 1,1-dichloroethene is 0.057 µg/L based on the "Water Quality Standards" (40 CFR 131) Human Health Water + Organism value; however, it defaults to the EQL of 2 µg/L listed in the 100-BC SAP (DOE/RL-2009-44) when the analytical method cannot achieve the action level. MDLs were reported as 0.08 µg/L which is less than the EQL of 2 µg/L. The results of this evaluation indicate that 1,1-dichloroethene has not been detected in RI groundwater samples.

Benzene was identified as a COPC in the 100-BC Work Plan (DOE/RL-2008-46-ADD3) because it was detected above the action level and because MDLs were greater than the action level. Benzene was not detected in unfiltered groundwater samples (54 samples) and was not analyzed for in filtered groundwater samples. The action level for benzene is 0.80 µg/L based on the MTCA "Groundwater Cleanup Standards" (WAC 173-340-720); however, it defaults to the EQL of 1.5 µg/L reported in the 100-BC SAP (DOE/RL-2009-44) when the analytical method cannot achieve the action level. All MDLs were reported as 0.06 µg/L which is less than the EQL of 1.5 µg/L. All samples included in this evaluation were collected as part of the spatial and temporal sampling activities of the RI. The results of this evaluation indicate that benzene was not detected in RI groundwater samples.

Carbon tetrachloride was identified as a COPC in the 100-BC Work Plan (DOE/RL-2008-46-ADD3) because it was detected above the action level and most MDLs were greater than the action level. Carbon tetrachloride was not detected in any unfiltered groundwater samples (54 samples) and was not analyzed for in filtered groundwater samples. 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-BC SAP (DOE/RL-2009-44) when the analytical method cannot achieve the action level. All MDLs were reported as 0.12 µg/L which is less than the EQL of 1 µg/L. All samples included in this evaluation were collected as part of the spatial and temporal sampling activities of the RI. The results of this evaluation indicate that carbon tetrachloride was not detected in RI groundwater samples.

Chloroform was identified as a COPC in the 100-BC Work Plan (DOE/RL-2008-46-ADD3) because it was detected in groundwater and most MDLs were greater than the action level. Chloroform was detected in 42 of 54 unfiltered groundwater samples (78 percent) and was not analyzed for in filtered groundwater samples. The action level for chloroform is 1.4 µg/L based on the MTCA "Groundwater Cleanup Standards" (WAC 173-340-720); however, it defaults to the EQL of 5 µg/L reported in the 100-BC SAP (DOE/RL-2009-44) when the analytical method cannot achieve the action level. Chloroform concentrations range between 0.1 and 3.0 µg/L in unfiltered samples. All MDLs were reported as

0.1 µg/L which are less than the EQL of 5 µg/L. All samples included in this evaluation were collected as part of the spatial and temporal sampling activities of the RI. The results of this evaluation indicate that chloroform was detected in RI groundwater samples at concentrations below the EQL.

Tetrachloroethene was identified as a COPC in the 100-BC Work Plan (DOE/RL-2008-46-ADD3) because it was detected infrequently above the action level and most MDLs were greater than the action level. Tetrachloroethene was not detected in any unfiltered groundwater samples (54 samples) and was not analyzed for in filtered groundwater samples. The action level for tetrachloroethene is 0.081 µg/L based on the MTCA "Groundwater Cleanup Standards" (WAC 173-340-720); however, it defaults to the EQL of 5 µg/L reported in the 100-BC SAP (DOE/RL-2009-44) when the analytical method cannot achieve the action level. All MDLs were reported as 0.18 µg/L which are less than the EQL of 5 µg/L. All samples included in this evaluation were collected as part of the spatial and temporal sampling activities of the RI. The results of this evaluation indicate that tetrachloroethene was not detected in RI groundwater samples.

Trichloroethene was identified as a COPC in the 100-BC Work Plan (DOE/RL-2008-46-ADD3) because it was detected above the action level and MDLs were greater than the action level. Trichloroethene was detected in 40 of 54 unfiltered groundwater samples (74 percent) and was not analyzed for in filtered groundwater samples. The action level for trichloroethene is 0.49 µg/L based on the MTCA "Groundwater Cleanup Standards" (WAC 173-340-720); however, it defaults to the EQL of 1 µg/L reported in the 100-BC SAP (DOE/RL-2009-44) when the analytical method cannot achieve the action level. Trichloroethene concentrations ranged between 0.23 µg/L and 3.3 µg/L in unfiltered samples. MDLs range between 0.21 and 0.25 µg/L, which are less than the EQL of 1 µg/L. Trichloroethene concentrations greater than the EQL were reported at seven wells:

- One of three sample results from 199-B3-46 (0.9 to 1.2 µg/L)
- Two of three sample results from 199-B3-50 (0.9 to 1.2 µg/L)
- One of three sample results from 199-B4-4 (0.7 to 1.2 µg/L)
- All three sample results from 199-B5-5 (1.8 to 2.2 µg/L)
- All three sample results from 199-B5-6 (2.8 to 3.3 µg/L)
- All three sample results from 199-B9-3 (1.2 to 2.0 µg/L)
- Two of three sample results from 699-65-72 (0.99 to 1.3 µg/L)

All samples included in this evaluation were collected as part of the spatial and temporal sampling activities of the RI. The results of the evaluation indicate that trichloroethene has been detected in RI groundwater samples at concentrations above the EQL of 1 µg/L.

Vinyl chloride was identified as a COPC in the 100-BC Work Plan (DOE/RL-2008-46-ADD3) because all MDLs were greater than the action level. Vinyl chloride was not detected in any unfiltered groundwater samples (54 samples) and was not analyzed for in filtered groundwater samples. The action level for vinyl chloride is 0.025 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 listed in the 100-BC SAP (DOE/RL-2009-44) when the analytical method cannot achieve the action level. All MDLs were reported as 0.08 µg/L which is less than the EQL of 5 µg/L. All samples included in this evaluation were collected as part of the spatial and temporal sampling activities of the RI. The results of this evaluation indicate that vinyl chloride was not detected in RI groundwater samples.

Anions. Nitrate was identified as a COPC in the 100-BC Work Plan (DOE/RL-2008-46-ADD3) because it was detected above the drinking water standard. Nitrate was detected in all unfiltered groundwater samples (127 samples) and was not analyzed for in filtered groundwater samples. Nitrate concentrations ranged between 0.708 and 44.2 mg/L. All nitrate concentrations were less than the drinking water standard of 45 mg/L. Unfiltered nitrate concentrations were reported above the Hanford Site 90th percentile (unfiltered) background level of 26.9 mg/L.

Metals. Aluminum was detected above the secondary drinking water standard of 50 µg/L in samples collected for the RI. Aluminum is included in the nature and extent evaluation; however, it was not identified as a COPC in the 100-BC Work Plan (DOE/RL-2008-46-ADD3). Aluminum was detected in 23 of 54 unfiltered groundwater samples (43 percent) and was detected in 5 of 54 filtered groundwater samples (9 percent). The action level for aluminum is based on the secondary drinking water standard. Aluminum affects aesthetic qualities relating to public acceptance of drinking water. These regulations are not federally enforceable, but are intended as guidelines for states. Because the action level is based on a secondary drinking water standard, aluminum concentrations in groundwater are compared to the AWQC of 87 µg/L. Aluminum concentrations range between 7.1 and 72 µg/L in unfiltered samples and between 12 and 30 µg/L in filtered samples. MDLs ranged from 5 to 10 µg/L which are less than the action level. All samples included in this evaluation were collected as part of the spatial and temporal sampling activities of the RI. The results of this evaluation indicate that aluminum has been detected in RI groundwater samples at concentrations below the AWQC. Aluminum concentrations in unfiltered and filtered samples are above the Hanford Site 90th percentile (filtered) background level of 7.1 µg/L.

Antimony was identified as a COPC in the 100-BC Work Plan (DOE/RL-2008-46-ADD3) because it was detected infrequently above the action level and most MDLs were greater than the action level. Antimony was not detected in any unfiltered groundwater sample (109 samples) or filtered groundwater sample (127 samples). Samples collected for the RI were analyzed using trace methods identified in the 100-BC SAP (DOE/RL-2009-44). MDLs for these RI samples range between 0.3 and 1.1 µg/L, which are less than the action level of 5.6 µg/L. Unfiltered and filtered samples collected for purposes other than the RI were analyzed by Method 6010. MDLs for the Method 6010 results range between 4 and 60 µg/L, almost all of which were above the action limit of 5.6 µg/L. The results of this evaluation indicate that antimony has not been detected historically in groundwater samples.

Arsenic was identified as a COPC in the 100-BC Work Plan (DOE/RL-2008-46-ADD3) because it was detected above the action level and all MDLs were greater than the action level. Arsenic was detected in 51 of 54 unfiltered (94 percent) and 50 of 54 filtered groundwater samples (93 percent). The action level for arsenic is 0.018 µg/L is based on the *National Recommended Water Quality Criteria* (EPA, 2009b), "Human Health Water + Organism" value; however, the action level defaults to the EQL of 4 µg/L reported in the 100-BC SAP (DOE/RL-2009-44) when the analytical method cannot achieve the action level. Arsenic concentrations in unfiltered groundwater samples range between 0.82 and 3.9 µg/L, which are less than the EQL. In filtered groundwater samples, concentrations range between 0.84 and 18 µg/L. Minimum, maximum, and 90th percentile values for Hanford Site background (filtered) concentrations of arsenic are 0.5, 8.8, and 7.85 µg/L, respectively. With the exception of a single arsenic detection, arsenic concentrations in all filtered and unfiltered samples are less than the 90th percentile Hanford site background value of 7.85 µg/L. Arsenic was detected in one filtered groundwater sample from 199-B3-50 (July 2010) at a concentration of 18 µg/L, however the corresponding unfiltered sample reported an arsenic concentration of 3.1 µg/L. Arsenic concentrations from the previous and subsequent sampling rounds were less than the arsenic background value. For these reasons the high arsenic result was flagged with a "Y" review qualifier indicating that the result is suspect. All samples included in this evaluation were collected as part of the spatial and temporal sampling activities of the RI. The results of this evaluation indicate that arsenic was detected in RI groundwater samples. Arsenic concentrations in

unfiltered and filtered samples are less than the Hanford Site 90th percentile (filtered) background level of 7.9 μg/L with the exception of a single, suspect detection, which is not associated with a trend at that location.

Beryllium was identified as a COPC in the 100-BC Work Plan (DOE/RL-2008-46-ADD3) because it was detected above the action level. Beryllium was detected in 1 of 109 unfiltered groundwater samples (0.9 percent) and in 3 of 127 filtered groundwater samples (2.4 percent). The detected result was 0.13 μg/L in the unfiltered sample and ranged between 0.06 and 0.24 μg/L in filtered samples, all of which are less than the drinking water standard of 4 μg/L. MDLs ranged from 0.05 to 4 μg/L, which are less than the drinking water standard of 4 μg/L. The results of this evaluation indicate that beryllium has been detected infrequently in groundwater at concentrations less than the drinking water standard of 4 μg/L. Beryllium concentrations in unfiltered and filtered samples were reported below the Hanford Site 90th percentile (filtered) background level of 2.3 μg/L.

Cadmium was identified as a COPC in the 100-BC Work Plan (DOE/RL-2008-46-ADD3) because it was detected above the AWQC and most MDLs were greater than the AWQC. Cadmium was not detected in unfiltered groundwater samples (109 samples) and was detected in one of 127 filtered groundwater samples (0.8 percent). The cadmium concentration in the filtered groundwater sample was 0.06 μg/L, which is less than the AWQC of 0.25 μg/L. Samples collected for purposes other than the RI were analyzed by Method 6010. MDLs for these results range between 0.5 and 4 μg/L, which are higher than the AWQC of 0.25 μg/L. No detected results were reported by Method 6010. Samples collected for the RI used trace methods identified in the 100-BC SAP (DOE/RL-2009-44). MDLs for these samples range between 0.06 and 0.2 μg/L and with the exception of the single filtered result, cadmium was not detected in samples collected for the RI. The results of this evaluation indicate that cadmium has been detected historically in groundwater samples at a low frequency (less than 1 percent in filtered samples) at a concentration below the AWQC and the Hanford Site 90th percentile (filtered) background level of 0.92 μg/L.

Chromium was identified as a COPC in the 100-BC Work Plan (DOE/RL-2008-46-ADD3) because it was detected above the AWQC. Chromium was detected in 95 of 109 unfiltered groundwater samples (87 percent) and 111 of 127 filtered groundwater samples (87 percent). Chromium concentrations range between 4.5 and 69 μg/L in unfiltered groundwater samples and between 3.1 and 56 μg/L in filtered groundwater samples. MDLs ranged from 1.9 to 14 μg/L, which are lower than the AWQC of 65 μg/L. A single chromium concentration above the AWQC of 65 μg/L was reported in an unfiltered sample at Well 199-B3-47. The remaining 11 chromium results at this location were reported below the AWQC and ranged from 38 to 56 μg/L. The results of this evaluation indicate that chromium has been detected historically in groundwater. A chromium concentration above the AWQC was reported once in an unfiltered sample (69 μg/L)[2]. All MDLs were reported below the AWQC. Chromium concentrations in filtered and unfiltered samples are greater than the Hanford Site 90th percentile (filtered) background level of 2.4 μg/L. The presence of total chromium in 100-BC groundwater reflects the presence of Cr(VI) (Section 4.3.1.1).

Cobalt was identified as a COPC in the 100-BC Work Plan (DOE/RL-2008-46-ADD3) because it was detected above the action level. Cobalt was detected in eight of 109 unfiltered groundwater samples (7.3 percent) and 21 of 127 filtered groundwater samples (17 percent). Cobalt concentrations in unfiltered and filtered groundwater samples range between 0.05 and 9.4 μg/L. Samples collected for purposes other than the RI were analyzed by Method 6010. MDLs for these results range between 1.7 and 5 μg/L,

[2] This section discusses groundwater results from January 5, 2006 through January 26, 2011. A total chromium result from February 2012 also exceeded the action level (Section 4.3.2).

unfiltered concentrations (two detects) ranged between 5.5 and 7.9 μg/L, and filtered concentrations (six detects) ranged between 3.0 and 9.4 μg/L. Most MDLs and the eight detected concentrations reported by Method 6010 are above the action level of 2.6 μg/L. Six of the eight detected results reported by Method 6010 were flagged with a "B" qualifier (indicating the analyte was detected at a concentration less than the contract required detection limit, but greater than or equal to the instrument or method detection limit) or a "C" qualifier (indicating the analyte was detected in both the sample and the associated QC blank). Cobalt concentrations for unfiltered and filtered samples flagged with a "B" ranged between 3.0 and 5.0 μg/L. Cobalt concentrations for unfiltered and filtered samples flagged with a "C" ranged between 5.5 and 9.4 μg/L. Samples collected for the RI used trace methods identified in the 100-BC SAP (DOE/RL-2009-44). MDLs for these samples ranged between 0.05 and 0.22 μg/L; unfiltered concentrations ranged between 0.05 and 0.67 μg/L; and filtered concentrations ranged between 0.06 and 1.2 μg/L. All MDLs and detected results reported by the trace methods are below the action level of 2.6 μg/L. The results of this evaluation indicate that cobalt has been detected historically in groundwater samples (7.3 percent of unfiltered samples and 17 percent of filtered samples) at concentrations up to 3.6 times greater than the action level. However, all cobalt concentration above the action level are reported by Method 6010 and almost all are flagged with either a "B" or C" qualifier. Cobalt concentrations and MDLs for samples collected for the RI are less than the action level. Cobalt concentrations in filtered samples are above the Hanford Site 90th percentile (filtered) background level of 0.92 μg/L.

Copper was identified as a COPC in the 100-BC Work Plan (DOE/RL-2008-46-ADD3) because it was detected above the AWQC. Copper was detected in 24 of 109 unfiltered groundwater samples (22 percent) and 21 of 127 filtered groundwater samples (17 percent). Copper concentrations in unfiltered and filtered groundwater samples range between 0.11 and 5.0 μg/L and the MDLs range between 0.1 and 6 μg/L. All MDLs and concentrations were reported below the AWQC of 9 μg/L. The results of this evaluation indicate that copper has been detected historically in groundwater samples (22 percent of unfiltered samples and 17 percent of filtered samples) at concentrations below the AWQC of 9 μg/L. Copper concentrations in filtered samples are above the Hanford Site 90th percentile (filtered) background level of 0.81 μg/L.

Iron was detected above the secondary drinking water standard of 300 μg/L in samples collected for the RI. Iron is included in the nature and extent evaluation; however, it was not identified as a COPC in the 100-BC Work Plan (DOE/RL-2008-46-ADD3). The action level for iron is based on the secondary drinking water standard. Iron affects aesthetic qualities relating to public acceptance of drinking water. These regulations are not federally enforceable, but are intended as guidelines for states. Because the action level is based on a secondary drinking water standard, iron concentrations in groundwater are compared to the AWQC of 300 μg/L. Iron was detected in 52 of 109 unfiltered groundwater samples (48 percent) and 46 of 127 filtered groundwater samples (36 percent). Iron concentrations range between 9.2 and 347 μg/L in unfiltered samples and between 9.5 and 71 μg/L in filtered samples. MDLs ranged between 9 and 38 μg/L. All MDLs and concentrations were reported below the action level of 300 μg/L. The results of this evaluation indicate that iron has been detected historically in groundwater (48 percent in unfiltered samples and 36 percent in filtered samples) at concentrations below the AWQC of 300 μg/L. Iron concentrations in filtered and unfiltered samples are less than the Hanford Site 90th percentile (filtered) background level of 570 μg/L.

Lead was identified as a COPC in the 100-BC Work Plan (DOE/RL-2008-46-ADD3) because it was detected above the AWQC. Lead was detected in four of 54 unfiltered (7 percent) and was not detected in filtered groundwater samples (54 samples). Lead concentrations in unfiltered groundwater samples range between 0.2 and 0.3 μg/L and MDLs range from 0.1 to 0.2 μg/L. All MDLs and concentrations were reported below the AWQC of 2.1 μg/L. All samples included in this evaluation were collected as part of the spatial and temporal sampling activities of the RI. The results of this evaluation indicate that lead has

been detected in RI groundwater samples (7 percent of unfiltered samples) at concentrations less than the AWQC of 2.1 µg/L. Unfiltered lead concentrations are less than the Hanford Site 90th percentile (filtered) background level of 0.92 µg/L.

Manganese was identified as a COPC in the 100-BC Work Plan (DOE/RL-2008-46-ADD3) because it was detected above the secondary drinking water standard. Manganese was detected in 15 of 109 unfiltered groundwater samples (14 percent) and 19 of 127 filtered groundwater samples (15 percent). The action level for manganese is based on the secondary drinking water standard. Manganese affects aesthetic qualities relating to public acceptance of drinking water. These regulations are not federally enforceable, but are intended as guidelines for states. Because the action level is based on a secondary drinking water standard, manganese concentrations in groundwater are compared to the MTCA "Surface Water Cleanup Standards" (WAC 173-340-730) of 907 µg/L. Manganese concentrations in unfiltered and filtered groundwater samples range between 1.8 and 41 µg/L and MDLs ranged between 0.8 and 6 µg/L. All MDLs and concentrations were reported below the secondary drinking water standard of 50 µg/L and the surface water cleanup level of 907 µg/L. The results of this evaluation indicate that manganese has been detected historically in groundwater samples (14 percent of unfiltered samples and 15 percent of filtered samples) at concentrations below the surface water cleanup level of 907 µg/L. A single manganese concentration in a filtered sample was reported above the Hanford Site 90th percentile (filtered) background level of 39 µg/L.

Mercury was identified as a COPC in the 100-BC Work Plan (DOE/RL-2008-46-ADD3) because it was detected above the AWQC and all MDLs were greater than the AWQC. Mercury was not detected in unfiltered groundwater samples (54 samples) or in filtered groundwater samples (54 samples). The AWQC for mercury is 0.012 µg/L based on the AWQC; however, it defaults to the EQL of 0.5 µg/L reported in the 100-BC SAP (DOE/RL-2009-44) when the analytical method cannot achieve the AWQC. MDLs range from 0.016 to 0.1 µg/L, which are less than the EQL of 0.5 µg/L. All samples included in this evaluation were collected as part of the spatial and temporal sampling activities of the RI. The results of this evaluation indicated that mercury has not been detected in RI groundwater samples. All MDLs are reported above the Hanford Site 90th percentile (filtered) background level of 0.003 µg/L.

Nickel was identified as a COPC in the 100-BC Work Plan (DOE/RL-2008-46-ADD3) because it was detected above the AWQC. Nickel was detected in 19 of 109 unfiltered groundwater samples (17 percent) and 7 of 127 filtered groundwater samples (5.5 percent). Nickel concentrations in unfiltered and filtered groundwater samples range between 4 and 12 µg/L; MDLs range from 4.0 to 13 µg/L. All results and MDLs are less than the AWQC of 52 µg/L. The results of this evaluation indicate that nickel has been detected historically in groundwater samples (17 percent of unfiltered samples and 5.5 percent of filtered samples) below the AWQC. Nickel concentrations in filtered samples are above the Hanford Site 90th percentile (filtered) background level of 1.6 µg/L.

Selenium was identified as a COPC in the 100-BC Work Plan (DOE/RL-2008-46-ADD3) because it was detected above the AWQC and MDLs were greater than the AWQC. Selenium was detected in 44 of 54 unfiltered groundwater samples (81 percent) and 49 of 54 filtered groundwater samples (91 percent). Selenium concentrations in unfiltered and filtered samples range between 0.40 and 8.3 µg/L. All MDLs were reported as 0.6 µg/L. All samples included in this evaluation were collected as part of the spatial and temporal sampling activities of the RI. A single filtered result was reported greater than the AWQC of 5 µg/L (8.3 µg/L at 199-B3-50) and was flagged with a "B" qualifier (indicating the analyte was detected at a concentration less than the contract required detection limit, but greater than or equal to the instrument or method detection limit). The corresponding unfiltered result was less than the AWQC. The results of this evaluation indicate that selenium has been detected in RI groundwater samples (81 percent of unfiltered samples and 91 percent of filtered samples). A single detections greater than the

AWQC is not associated with a trend or a location. All selenium concentrations are less than the Hanford Site 90th percentile (filtered) background level of 11 μg/L.

Thallium was identified as a COPC in the 100-BC Work Plan (DOE/RL-2008-46-ADD3) because it was detected above the action level and all MDLs were greater than the action level. The action level for thallium is 0.24 μg/L based on the AWQC; however, it defaults to the EQL of 2 μg/L identified in the 100-BC SAP (DOE/RL-2009-44) when the analytical method cannot achieve the action level. Thallium was detected in one of 54 of unfiltered groundwater samples (1.9 percent) and one of 54 filtered groundwater samples (1.9 percent). Thallium concentrations in unfiltered and filtered samples range between 1.0 and 1.2 μg/L. MDLs range from 0.05 to 0.1 μg/L. All MDLs and concentrations were reported below the EQL. All samples included in this evaluation were collected as part of the spatial and temporal sampling activities of the RI. The results of this evaluation indicate that thallium was detected at a low frequency in RI groundwater samples (1.9 percent of unfiltered samples and 1.9 percent of filtered samples) at concentrations below the EQL. Thallium concentrations in unfiltered and filtered samples are below the Hanford Site 90th percentile (filtered) background level of 1.7 μg/L.

Uranium was identified as a COPC in the 100-BC Work Plan (DOE/RL-2008-46-ADD3) because additional data are needed to determine is its concentration in groundwater. Uranium was detected in all unfiltered samples (54 samples) and was not analyzed for in filtered groundwater samples. Uranium concentrations ranged between 1.1 and 7.4 μg/L. The results of this evaluation indicated that uranium was reported below the drinking water standard of 30 μg/L and the Hanford Site 90th percentile (filtered) background level of 9.9 μg/L in RI groundwater samples.

Zinc was identified as a COPC in the 100-BC Work Plan (DOE/RL-2008-46-ADD3) because it was detected above the action level. Zinc was detected in 24 of 109 unfiltered groundwater samples (22 percent) and 31 of 127 filtered groundwater samples (24 percent). Detected concentrations ranged between 4 and 23 μg/L for unfiltered samples and between 1.7 and 22 μg/L for filtered samples. MDLs ranged between 4 and 9 μg/L. All MDLs and concentrations were reported below the action level of 91 μg/L. The results of this evaluation indicate that zinc has been detected historically in groundwater samples (22 percent of unfiltered samples and 24 percent of filtered samples) at concentrations below the action level. A single zinc concentration in a filtered sample is above the Hanford Site 90th percentile (filtered) background level of 22 μg/L.

Other Analytes. Total petroleum hydrocarbons – diesel range was identified as a COPC in the 100-BC Work Plan (DOE/RL-2008-46-ADD3) because it was analyzed infrequently and an MDL was greater than the action level. Total petroleum hydrocarbons – diesel range was detected in two of 54 unfiltered groundwater samples (3.7 percent). Total petroleum hydrocarbons – diesel range concentrations were reported as 180 and 220 μg/L. Both detected concentrations and all MDLs were reported below the action level of 500 μg/L. All samples included in this evaluation were collected as part of the spatial and temporal sampling activities of the RI. The results of this evaluation indicate that total petroleum has been detected at low frequency in RI groundwater samples at concentrations below the action level.

4.3.1.3 Conclusions from Analysis

Uncertainties identified for the COPCs in the 100-BC Work Plan (DOE/RL-2008-46-ADD3) were generally associated with limitations in the analytical methods selected for analysis. Action levels for many of the metals and VOCs require methods that can measure the analyte at low levels and require the use of analytical methods that can detect analytes at trace levels. The analytical data set evaluated for this evaluation represented five years of data from all of the monitoring and compliance wells in use between January 5, 2006 and January 25, 2010. The analysis provided above identified when uncertainties were associated with data quality issues or limitations associated with the analytical method used.

Table 4-29 summarizes the outcome of the analysis Cr(VI), strontium-90, and tritium are groundwater COPCs. Most of the analytes of interest (32 of 37) were not detected above levels of concern (that is, action level, background, or EQL). Trichloroethene was detected at concentrations above action levels but concentrations were all below the drinking water standard and the 1×10^{-5} for chemicals based on the MTCA "Human Health Risk Assessment Procedures" (WAC 173-340-708(5)) cumulative risk threshold. Gross beta was detected as an indicator of strontium-90 or tecnetium-99. Three metals were detected above action levels but circumstances indicate they are not of continuing concern:

- Arsenic: A single, filtered sample had a concentration above Hanford Site background. The unfiltered sample had a much lower concentration and the high result was outside the trend for the well and is considered unrepresentative.
- Chromium: A single, unfiltered sample had a concentration above the AWQC for total chromium. Total chromium in filtered samples represents Cr(VI), which is a COPC.
- Cobalt was detected at concentrations greater than the action level. However, all of the exceedances were for samples analyzed by Method 6010, which has a higher detection limit and is less accurate at low concentrations than Method 200.8. Consequently most of the detections are flagged with a "B" qualifier. Others are flagged "C," indicating associated blank contamination. Samples collected for the RI were analyzed for cobalt using Method 200.8, which has a lower detection limit, and those results were less than the action level. Thus cobalt is not a groundwater contaminant of concern.

Table 4-29. Summary of Groundwater Contaminant Evaluation

Category	Constituent
Contaminant of potential concern (contaminants that warrant further evaluation in FS)	Cr(VI); strontium-90; tritium
Detected at levels above action level and background	trichloroethene[a]; gross beta[b]
Detected above action level, but isolated instances and/or suspect data	Arsenic; total chromium; cobalt
Detected in groundwater but below action level, EQL, or background levels	Cs-137; Tc-99; gross alpha; chloroform; nitrate; aluminum; beryllium; cadmium; copper; iron; lead; manganese; nickel; selenium; thallium; uranium; zinc; TPH-diesel
Not detected in groundwater	C-14; Co-60; Eu-155; I-129; Nickel-63; Ra-228; 1,1,2,2-Tetrachloroethane; 1,1-Dichloroethene; benzene; carbon tetrachloride; tetrachloroethene; vinyl chloride; antimony; mercury

Based on evaluation of data collected January 2006 through January 2010.

[a]See discussion in Section 4.3.2

[b]Gross beta indicates presence of strontium-90 or Tc-99. See results for those radionuclides.

4.3.2 Groundwater Contaminant Plumes

Defining contaminant plumes was part of the work scope designed to fill Data Gap 4. Data from new wells were combined with data from existing wells to refine knowledge of the nature and extent of groundwater contamination in 100-BC.

An evaluation of the nature and extent of contamination in 100-BC groundwater concluded that hexavalent chromium, strontium-90, tritium, and trichloroethene exceeded action levels (Section 4.3.1). Table 4-30 lists estimated areas of the plumes. The following sections discuss the current distribution of these contaminants in groundwater.

Table 4-30. 100-BC Plume Areas (2011)

Constituent	Contour Concentration	Area (km^2)
Cr(VI)	10 µg/L	2.2
Strontium-90	8 pCi/L	0.38
Tritium	20,000 pCi/L	0.16
Trichloroethene*	0.5	1.67

* This plume is not well defined because concentrations are near detection limits. Area is for fall 2010 plumes because map not prepared for 2011.

Contaminant plume maps presented in this section include:

- Cr(VI), strontium-90, tritium, and trichloroethene maps for two of the three periods sampled for RI spatial and temporal assessment: spring (May) and fall (September) 2010. The spring event was intended to represent high river stage conditions. However, as illustrated in Figure 4-28, the major increase in river stage occurred after the sampling event. The fall sampling event represented low river stage conditions.
- Cr(VI), strontium-90, and tritium plume maps representing average concentrations for 2011, as prepared for the Hanford Site groundwater annual report. The comprehensive, annual sampling event for 2011 was spread across several months (Figure 4-28) because of safety-related work stoppages. Selected wells were sampled quarterly in 2011.
- Cr(VI) for February 2012. The annual event for 2012 occurred in February. Plume maps for strontium-90 and tritium are not provided for this time period because their distribution did not change significantly from 2011.

Most of the monitoring wells in 100-BC are screened in the upper 5 m (16 ft) of the unconfined aquifer. New Wells 199-B2-16, 199-B3-51, 199-B5-5, and 199-B5-6 monitor intervals near the bottom of the unconfined aquifer (Table 3-5). Two of these deep wells are paired with wells screened at the top of the aquifer. In these cases, data from both wells are posted on the maps, and the higher concentration was honored in contouring the plume maps. Data from the two wells screened in the Ringold confined aquifer were excluded from the maps.

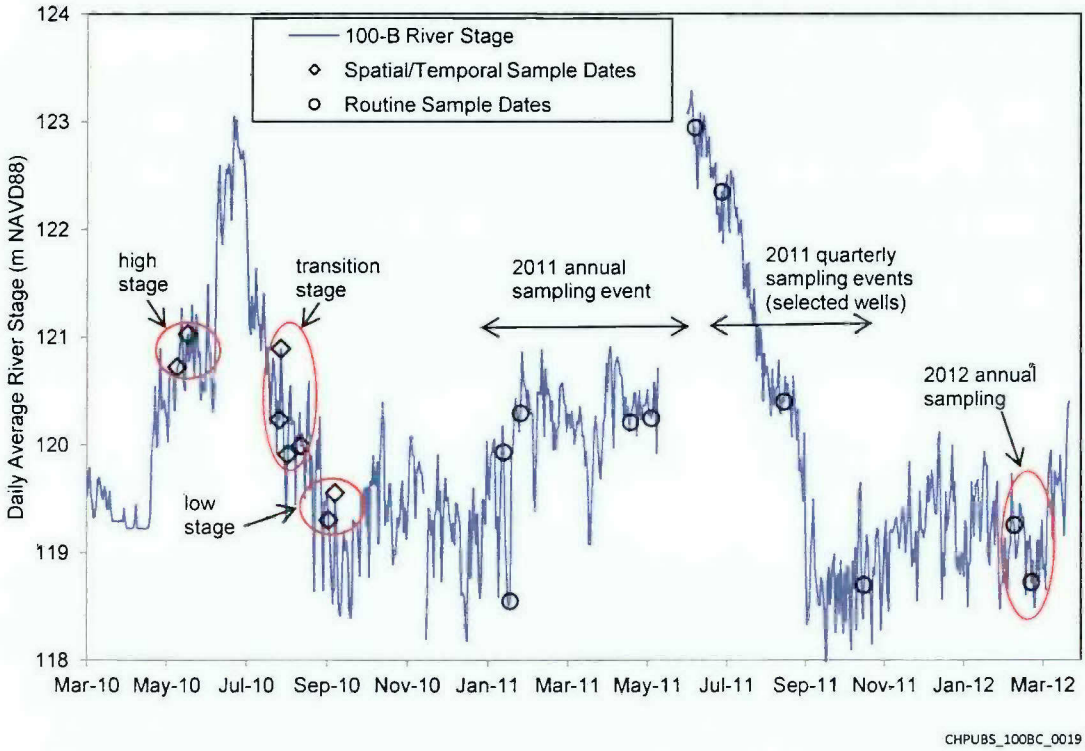


Figure 4-28. Spatial/Temporal Sampling Times as Related to River Stage at 100-BC

4.3.2.1 Cr(VI) in Groundwater

Groundwater samples may be analyzed specifically for Cr(VI) or for total chromium, which includes hexavalent and trivalent. Dissolved chromium in Hanford Site groundwater is virtually all hexavalent (*Speciation and Transport Characteristics of Chromium in the 100D/H Areas of the Hanford Site* [WHC-SD-EN-TI-302]). Filtered, total chromium data effectively represent Cr(VI) (Appendix C of *Hanford Site Groundwater Monitoring for Fiscal Year 2007* [DOE/RL-2008-01]). Trend plots in the section include filtered, total chromium, and Cr(VI) data because Cr(VI) data were not available in the past.

Cr(VI) exceeds the 10 µg/L AWQC in groundwater beneath a large portion of 100-BC, as illustrated in Figures 4-29 through 4-32. The plume changed little between spring and fall 2010, indicating little seasonal variation. Data from new wells helped refine the plume outline from previous estimates, but the boundaries of the plume at the 10 µg/L contour are still not well defined. The western boundary lies east of Well 199-B3-13 near the river but the boundary farther south, west of 199-B5-5, is not defined. The eastern boundary of the plume lies between new Well 199-B3-50 and older well 699-71-77. Interpreting the southern boundary at 10 µg/L is difficult because concentrations in wells 199-B9-3 and 199-B5-8 fluctuate above and below that level. The plume is much better defined at the 20 µg/L contour.

Data from 2011 indicate the western edge of the plume has contracted (Figure 4-31). Declining concentrations in wells in this region suggest that clean water is migrating in from the west.

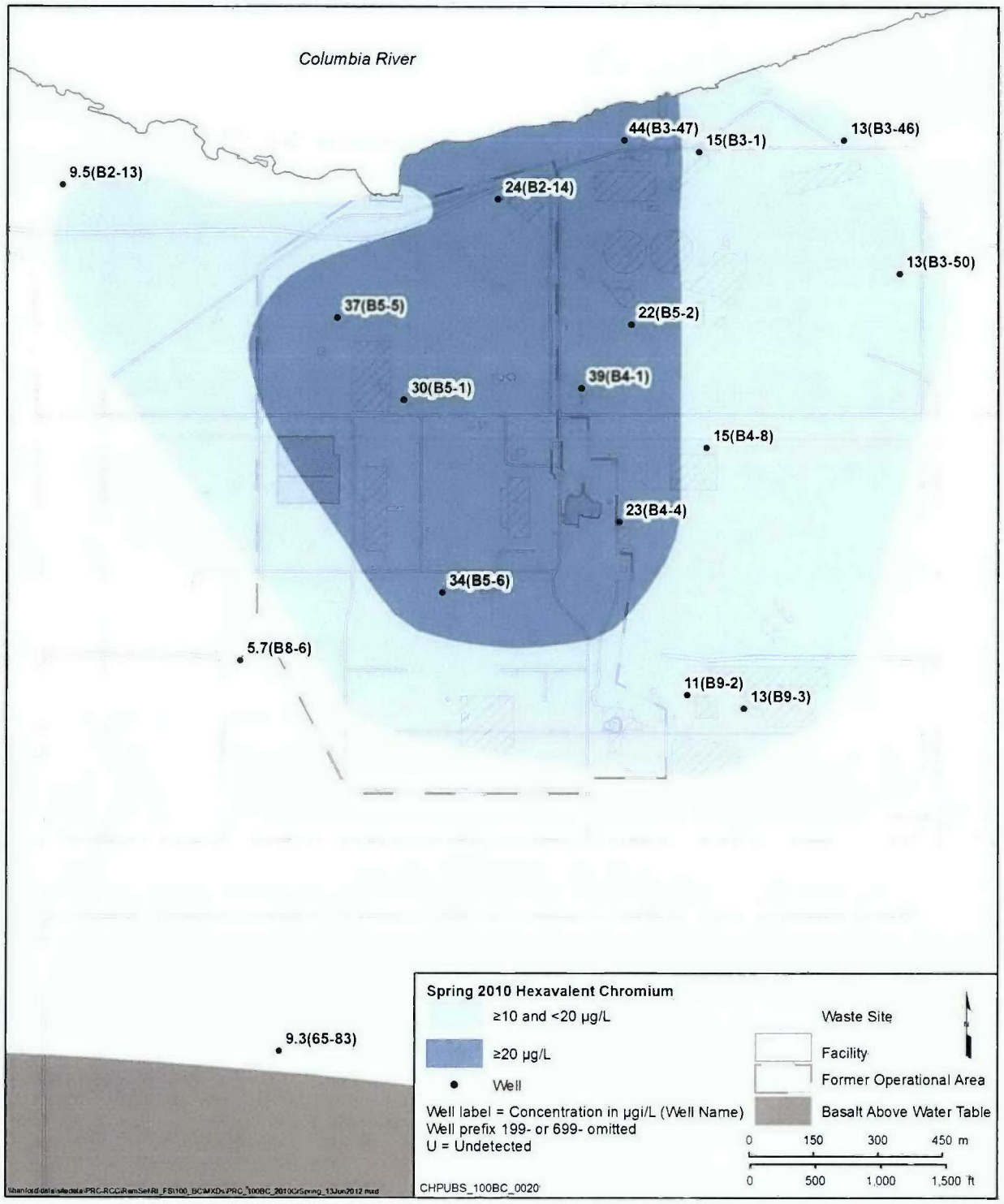


Figure 4-29. Cr(VI) in 100-BC Unconfined Aquifer, Spring 2010

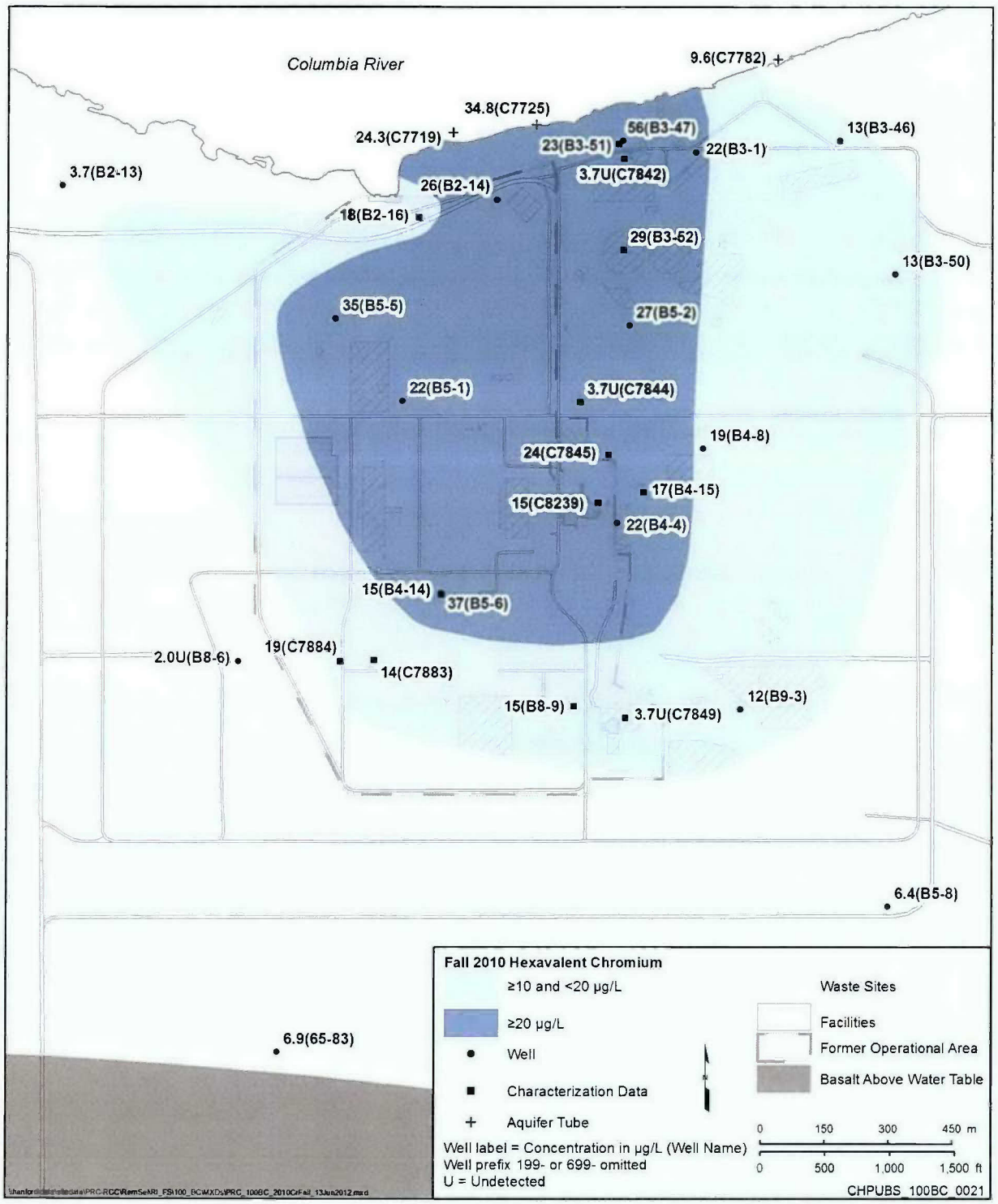


Figure 4-30. Cr(VI) in 100-BC Unconfined Aquifer, Fall 2010

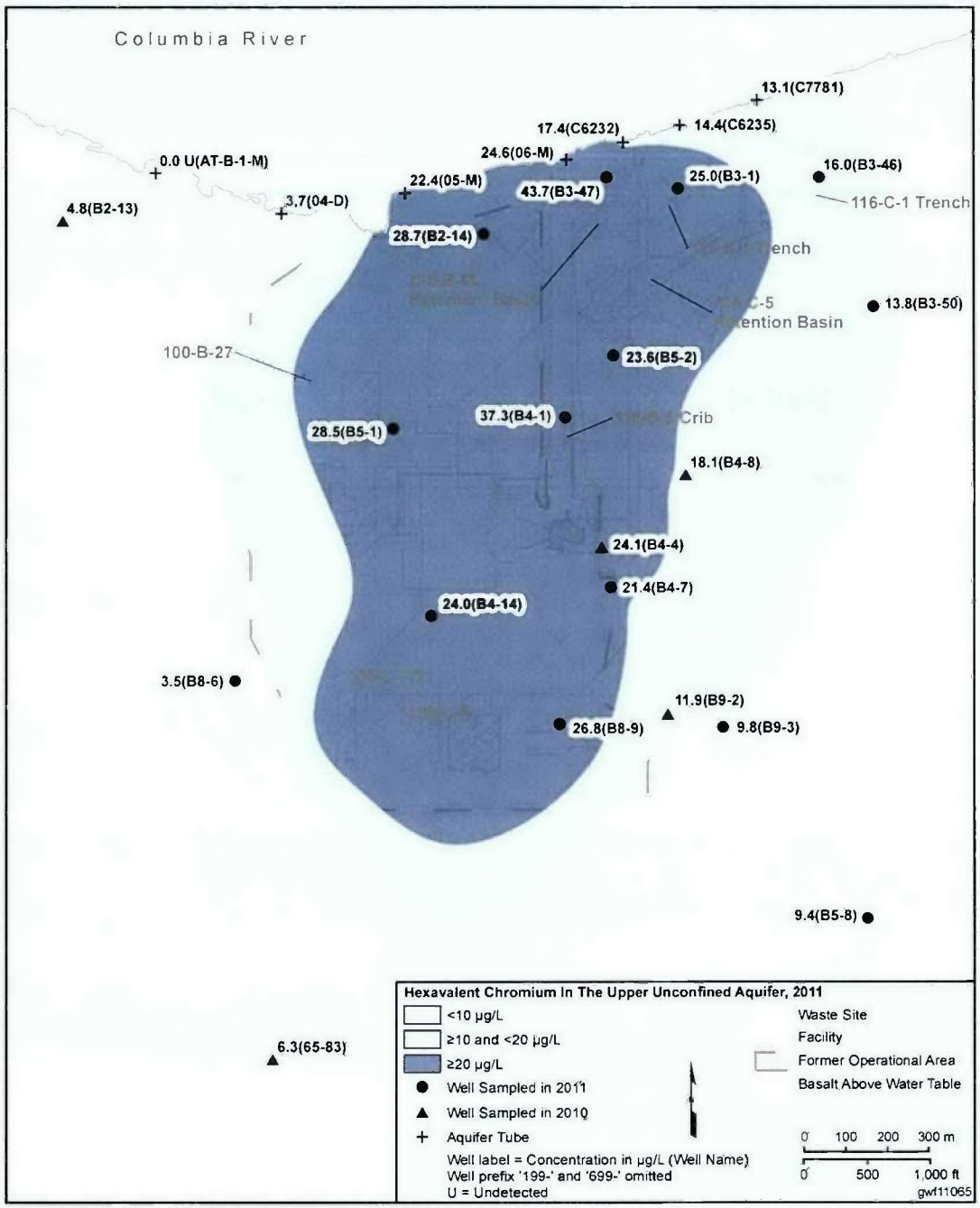


Figure 4-31. Cr(VI) in 100-BC Unconfined Aquifer, 2011

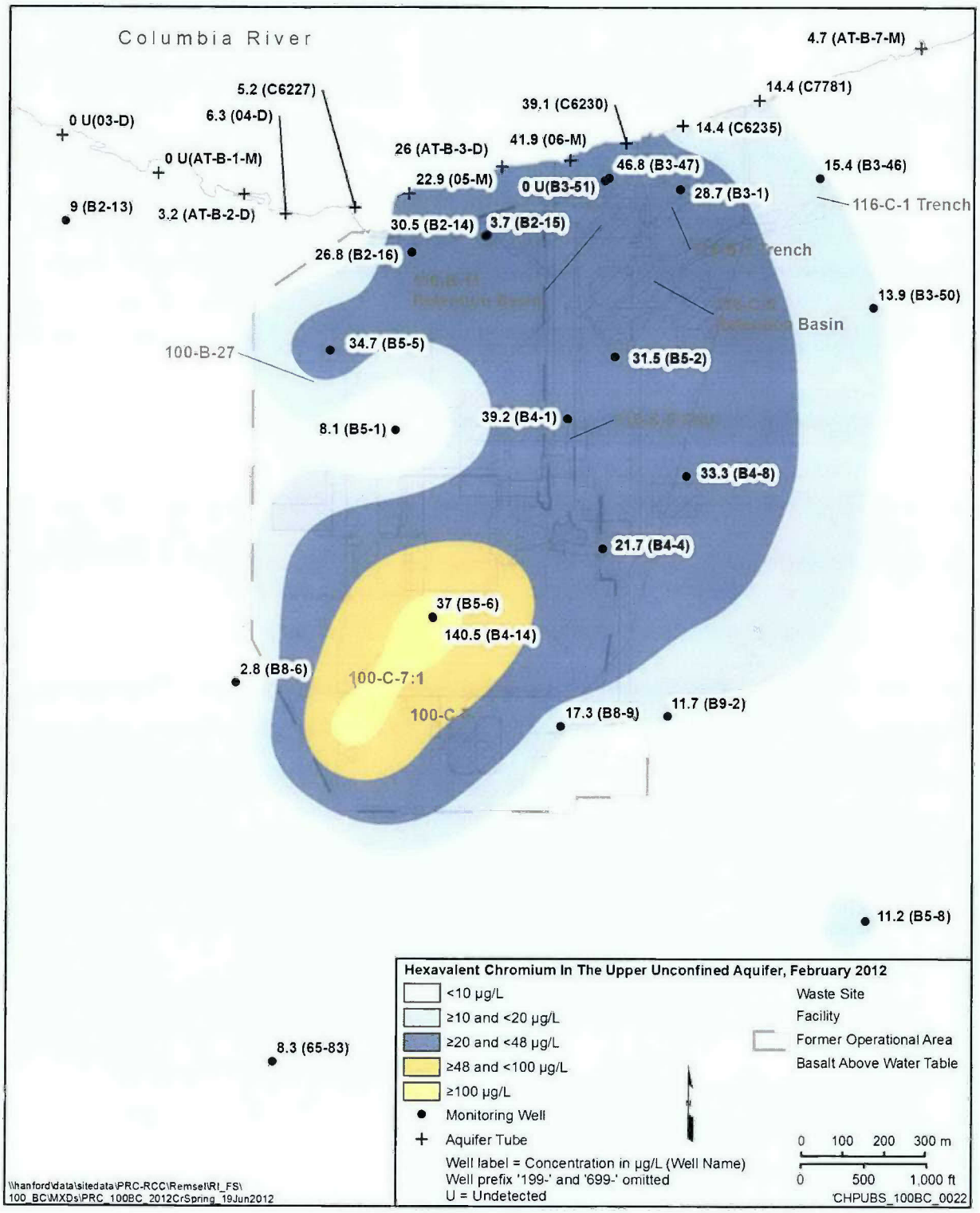


Figure 4-32. Cr(VI) in 100-BC Unconfined Aquifer, February 2012

The February 2012 map (Figure 4-32) shows significant changes from the 2010 and 2011 maps. Groundwater samples from Well 199-B4-14, screened in the Hanford formation, showed a sharp increase in Cr(VI) concentration to levels above 100 µg/L (Figure 4-33). The timing of the increase and the estimated direction of groundwater flow (Section 3.6) suggest that the source of contamination was remediation of the 100-C-7:1 waste site. This site has significant Cr(VI) contamination in soil at the water table (Section 4.2.1.11) that may have been mobilized by remediation activities. DOE is conducting additional studies of groundwater beneath the 100-C-7:1 waste site in 2012. Well 199-B5-6, adjacent to 199-B4-14 but screened at the bottom of the unconfined aquifer in Ringold unit E, did not show a similar Cr(VI) increase in February 2012 because Ringold unit E is less permeable than the Hanford formation. Farther east, the Cr(VI) concentration in Well 199-B8-9 rose to 50 µg/L in June 2010 and subsequently declined again. The increase also may be related to remediation activities at the 100-C-7:1 or 100-C-7 waste sites.

Another noteworthy change in February 2012 was a decrease in Cr(VI) concentration to 8 µg/L in Well 199-B5-1 (Figure 4-34). A previous decline (2003 to 2006) was accompanied by low specific conductance, indicating dilution. A leaking water line was subsequently identified and repaired. The 2012 decline in Cr(VI) was not accompanied by low specific conductance, indicating dilution is not a factor. If the low concentrations persist, it will indicate cleaner groundwater entering the area from the west. The upper portion of the aquifer in the vicinity of Well 199-B5-1 is believed to include a portion of the permeable Hanford formation. This would explain why Well 199-B5-1 responds to changes more rapidly than does nearby Well 199-B5-5, which is screened in the lower, less permeable part of the aquifer (Ringold unit E).

Trends in Cr(VI) concentrations in different parts of 100-BC indicate that the plume is gradually migrating to the northeast. This reflects movement of clean groundwater from the west and is consistent with the estimated directions of groundwater flow (Section 3.6). However, in most locations, the Cr(VI) plume is not attenuating rapidly. Specific observations that support these interpretations include the following.

- Concentrations have declined in wells 199-B2-13, 199-B8-6, and 199-B5-1 in western 100-BC (Figure 4-34). These wells show larger percent declines from peak levels than do wells in other parts of 100-BC, suggesting eastward movement of the plume.
- Many of the wells in central or northern 100-BC show similar trends in chromium concentrations with time, with rising levels in the 1990s, and then remained stable or very gradually declined (Figure 4-35).
- Concentrations are increasing gradually in wells 199-B3-1 and 199-B3-46 in northeastern 100-BC (Figure 4-36) indicating down-river movement of the plume.

Cr(VI) is detected in 100-BC aquifer tubes at concentrations similar to those in groundwater monitoring wells. Figure 4-37 illustrates chromium concentrations in a cross section through aquifer tubes and near river wells. Concentrations are typically highest in the mid-depth or deepest tubes, and lower in the shallow tubes. The shallowest tubes tend to have lower specific conductance, indicating greater mixing with river water.

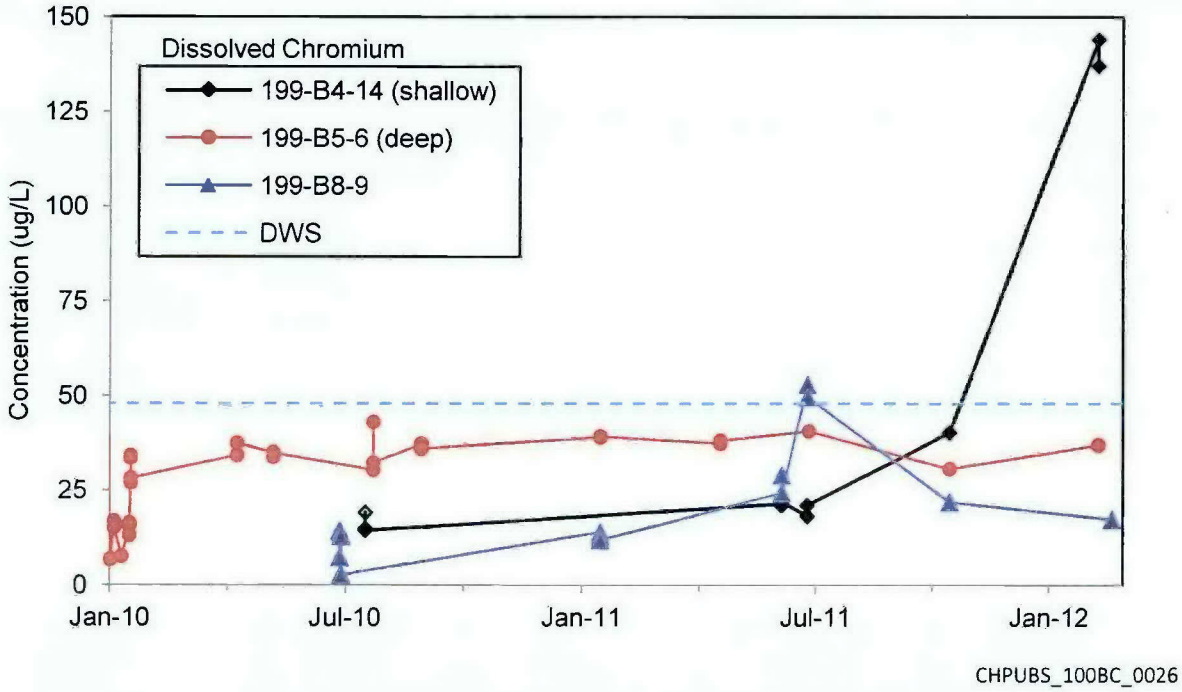


Figure 4-33. Dissolved Chromium in Wells in Southern 100-BC

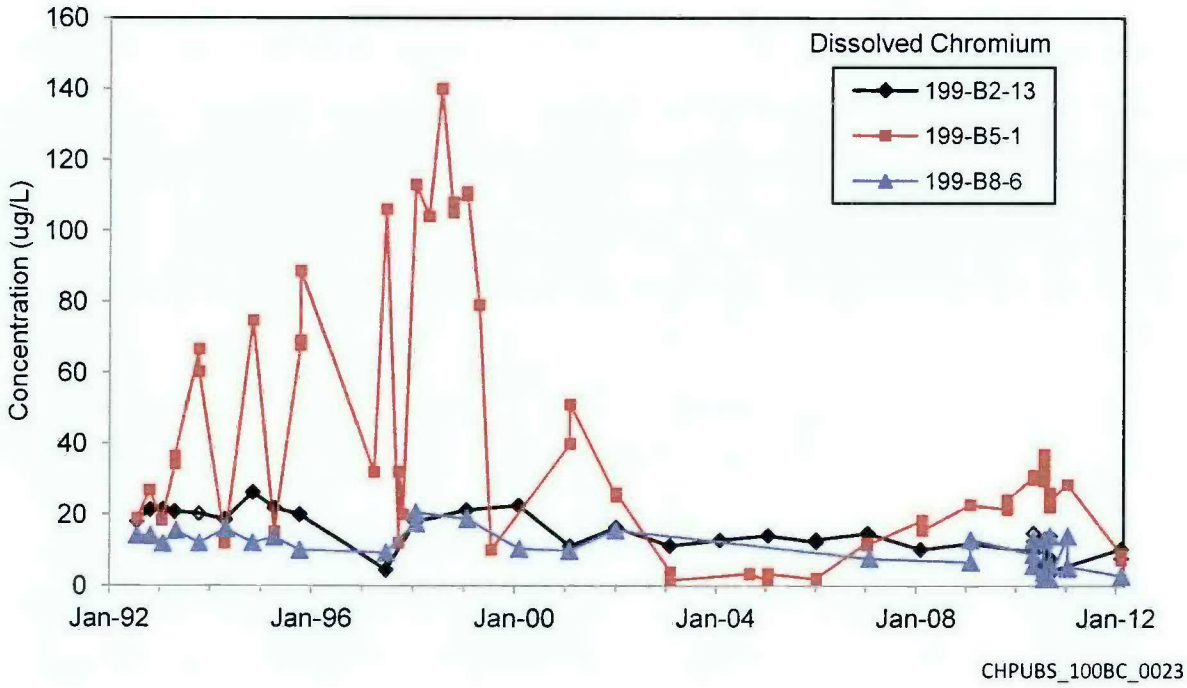


Figure 4-34. Dissolved Chromium in Wells in Western 100-BC

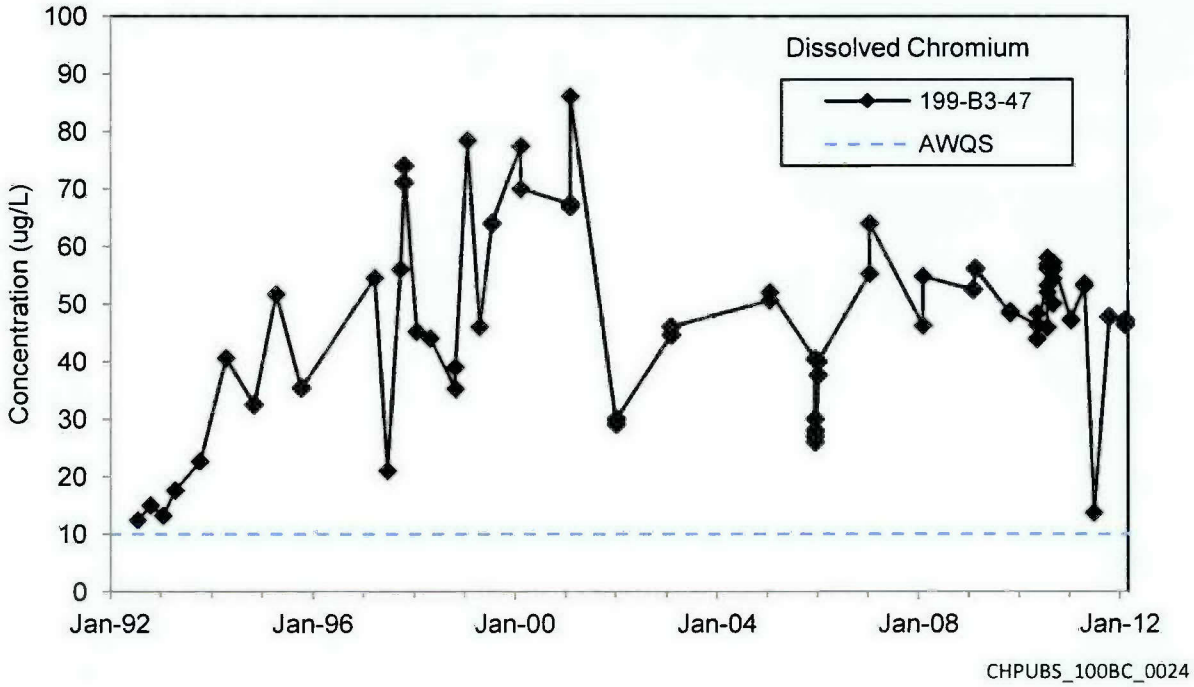


Figure 4-35. Dissolved Chromium in Well 199-B3-47

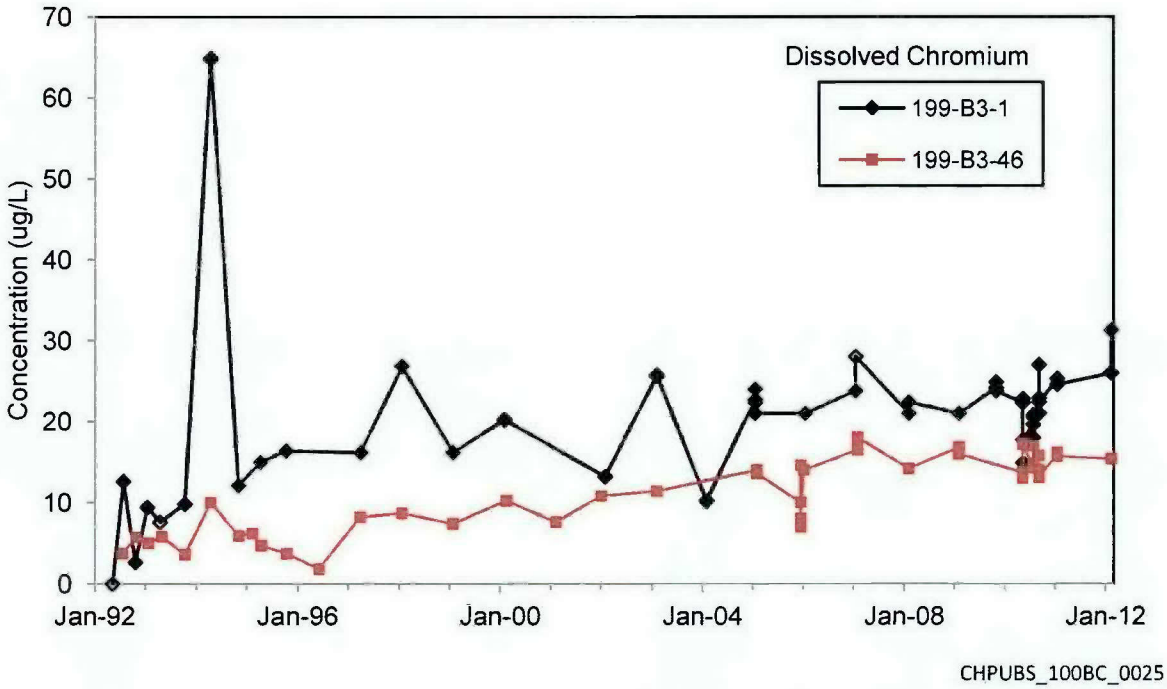


Figure 4-36. Dissolved Chromium in Wells 199-B3-1 and 199-B3-46 in Northeastern 100-BC

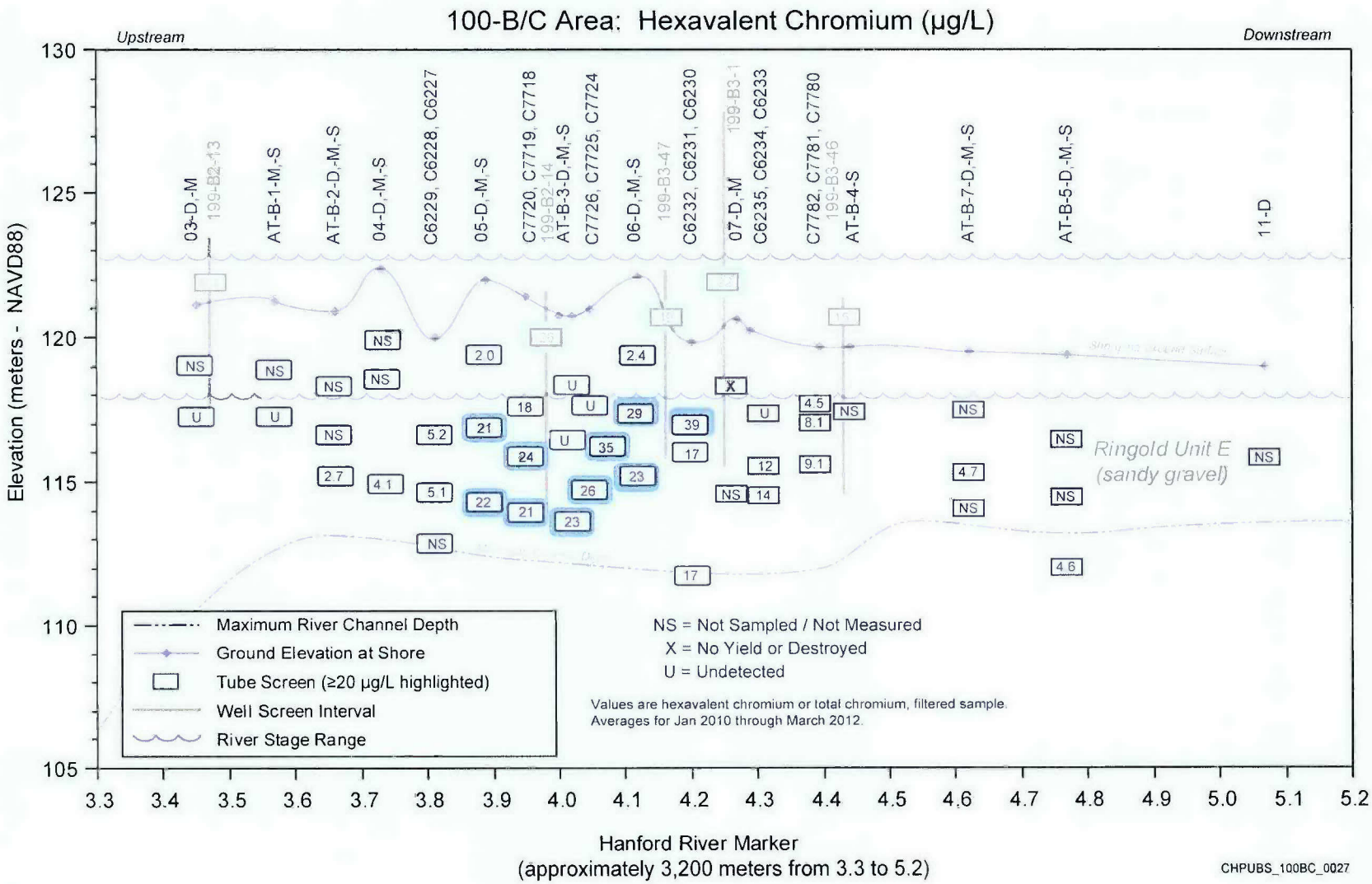


Figure 4-37. Dissolved Chromium in Aquifer Tubes and Near River Wells, 100-BC

4.3.2.2 Strontium-90 in Groundwater

The strontium-90 plume in 100-BC groundwater had sources near the B Reactor and also near the Columbia River. Figures 4-38 and 4-39 illustrates the plume based on data from spring (May) and fall (September) 2010. Figure 4-40 shows the plume in 2011. New Wells 199-B2-14, 199-B5-5, 199-B4-14, and 199-B3-50 helped define the extent of the plume.

The fall 2010 plume map includes data from groundwater grab samples from RI vadose boreholes. These data affected the interpretation of the plume near its southern tip. A characterization sample of groundwater from vadose borehole C7845, collected 1.5 m (4.8 ft) below the water table, contained 72.9 pCi/L strontium-90 (Figure 4-39). This concentration was higher than those recently detected in monitoring wells because of the borehole location at a waste site. The higher concentration also may reflect the fact that in 100-BC groundwater, strontium-90 concentrations are highest at the top of the aquifer. Monitoring wells typically have screened intervals of 4.6 m (15 ft). Strontium-90 concentrations continued to decline in Well 199-B4-4, located south of borehole C7845 (Figure 4-41).

Based on 2011 data, the plume is interpreted to have split into two portions (Figure 4-40). Although few wells near B Reactor had concentrations above the drinking water standard in 2011, data from 2010 support the interpretation that a plume remains in this region. Concentrations in wells 199-B4-1 and 199-B5-2 were atypically low in 2011. Concentrations in wells near the river remained above the drinking water standard.

The highest concentrations of strontium-90 are typically in Wells 199-B3-1 and 199-B3-46 (Figure 4-42). The maximum concentration in 2010 or 2011 was 49 pCi/L in Well 199-B3-46. Concentrations are declining overall.

Strontium-90 concentrations in shallow aquifer tubes are typically the same or lower than in nearby monitoring wells. Trends are declining overall.

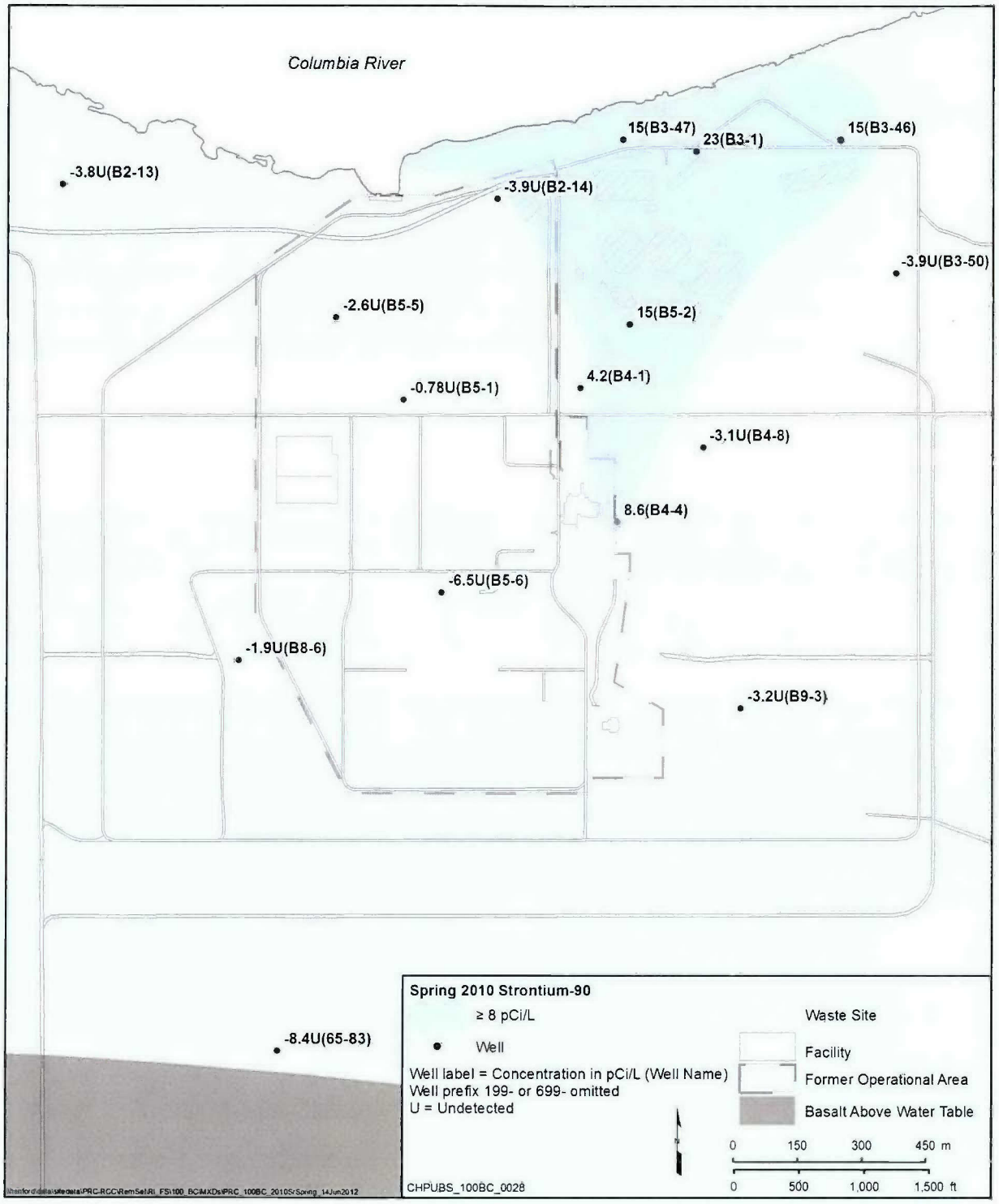


Figure 4-38. Strontium-90 in 100-BC Unconfined Aquifer, Spring 2010

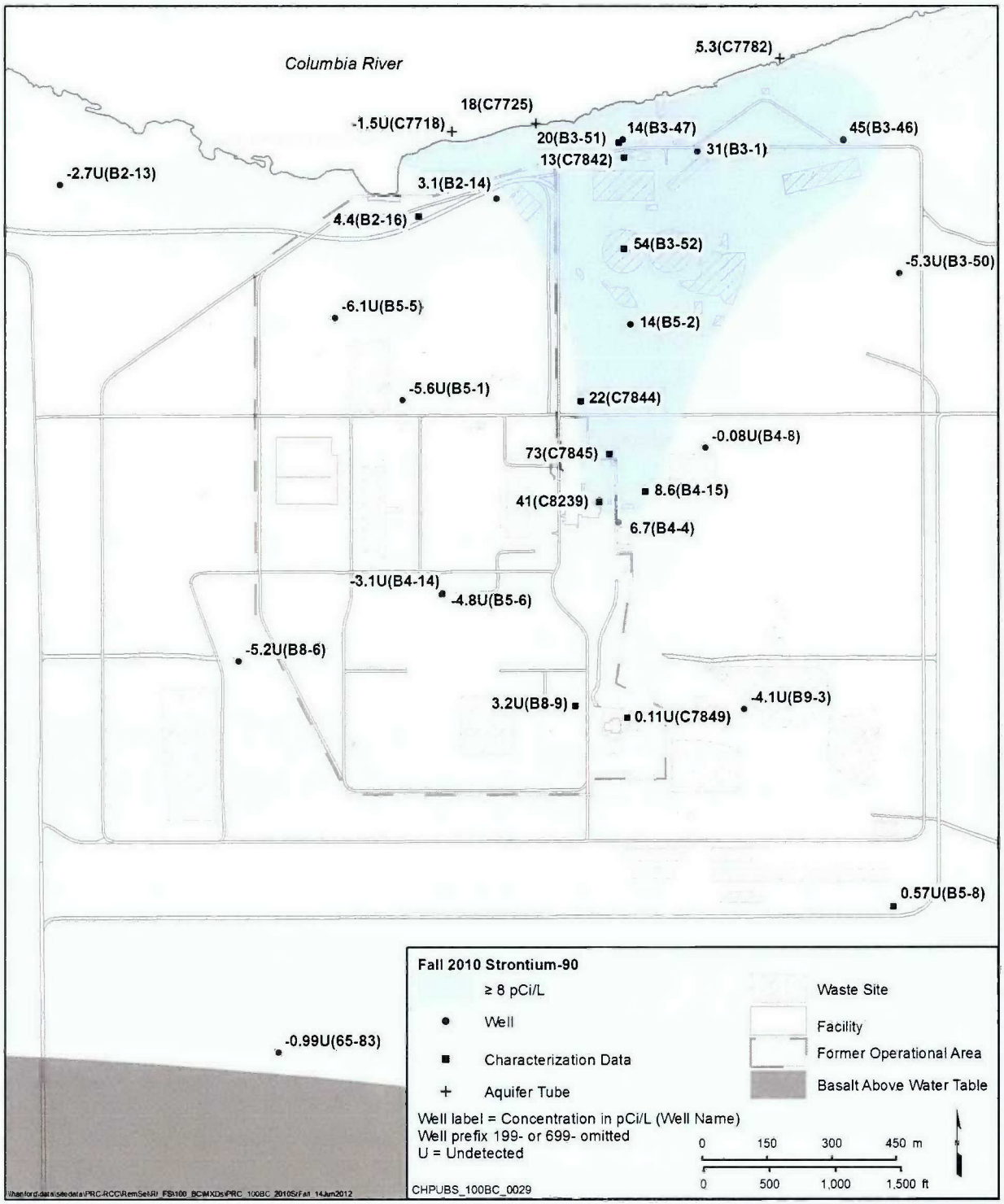


Figure 4-39. Strontium-90 in 100-BC Unconfined Aquifer, Fall 2010

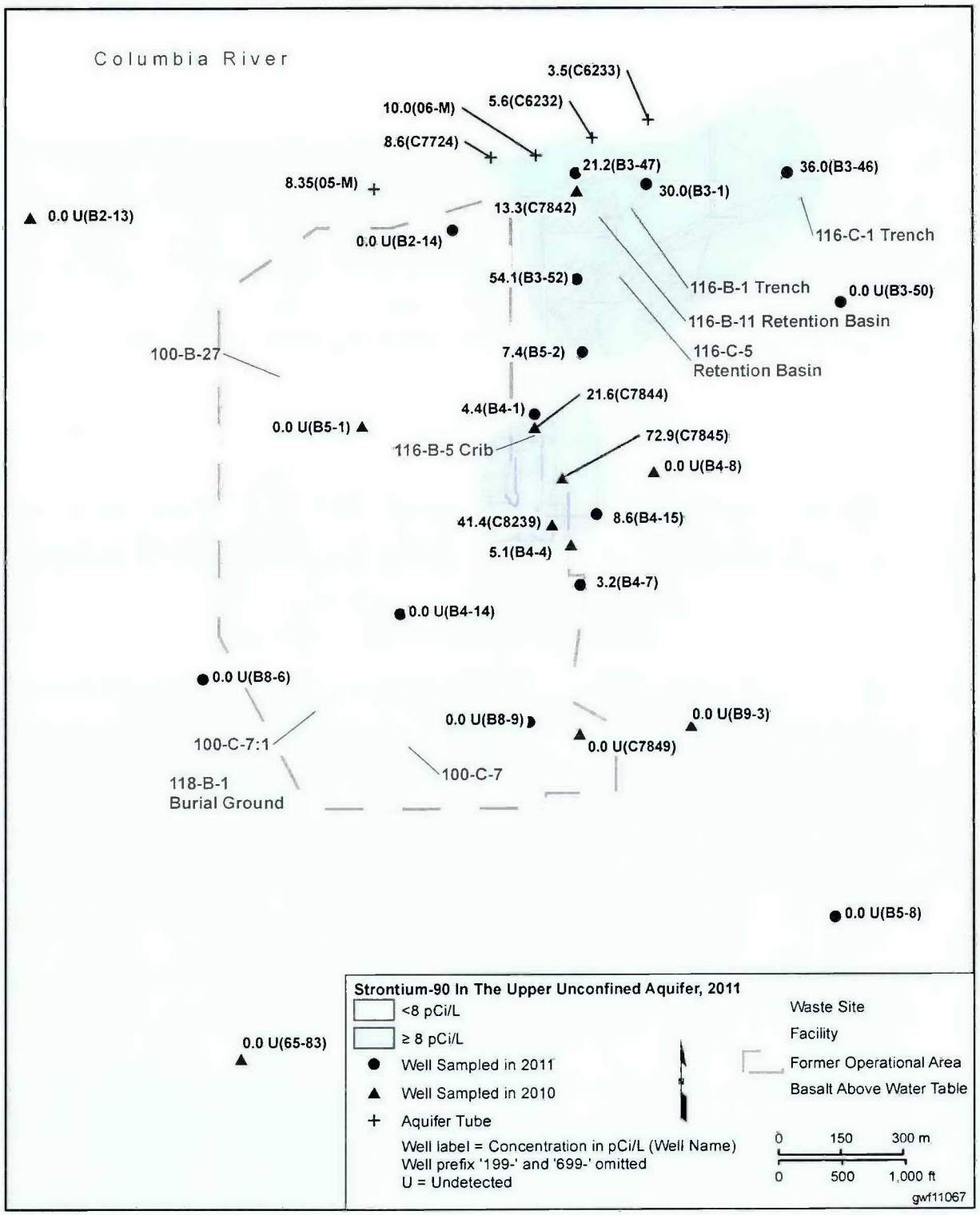


Figure 4-40. Strontium-90 in 100-BC Unconfined Aquifer, 2011

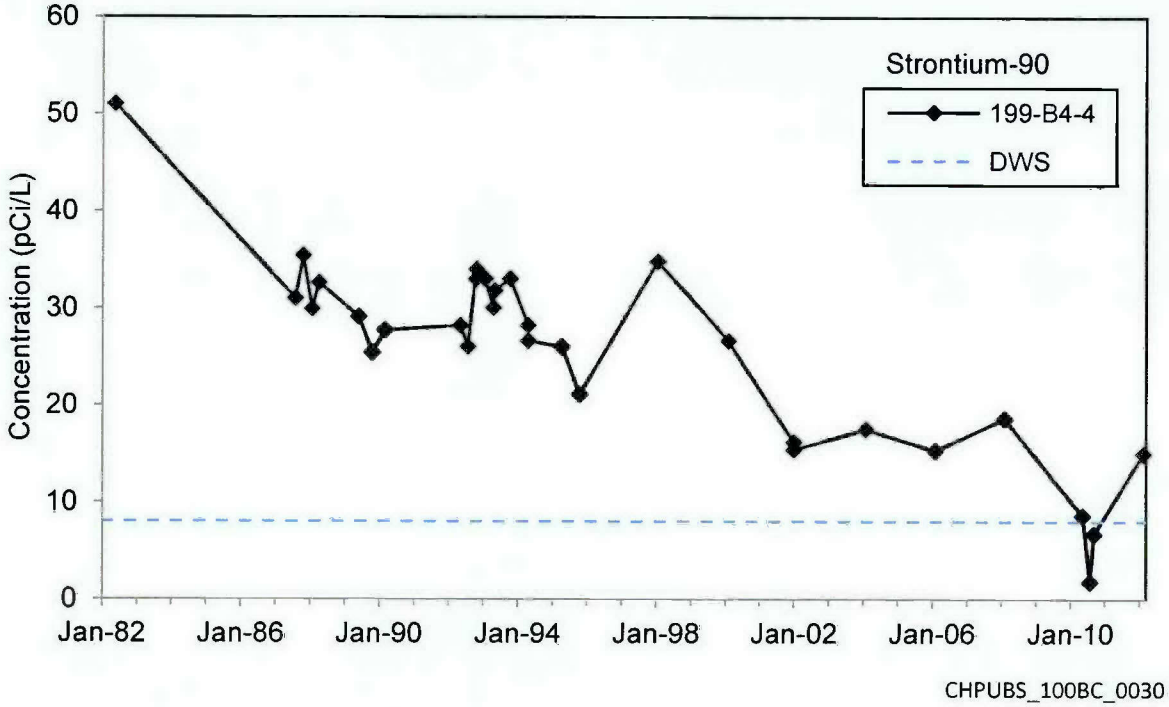


Figure 4-41. Strontium-90 in Well 199-B4-4

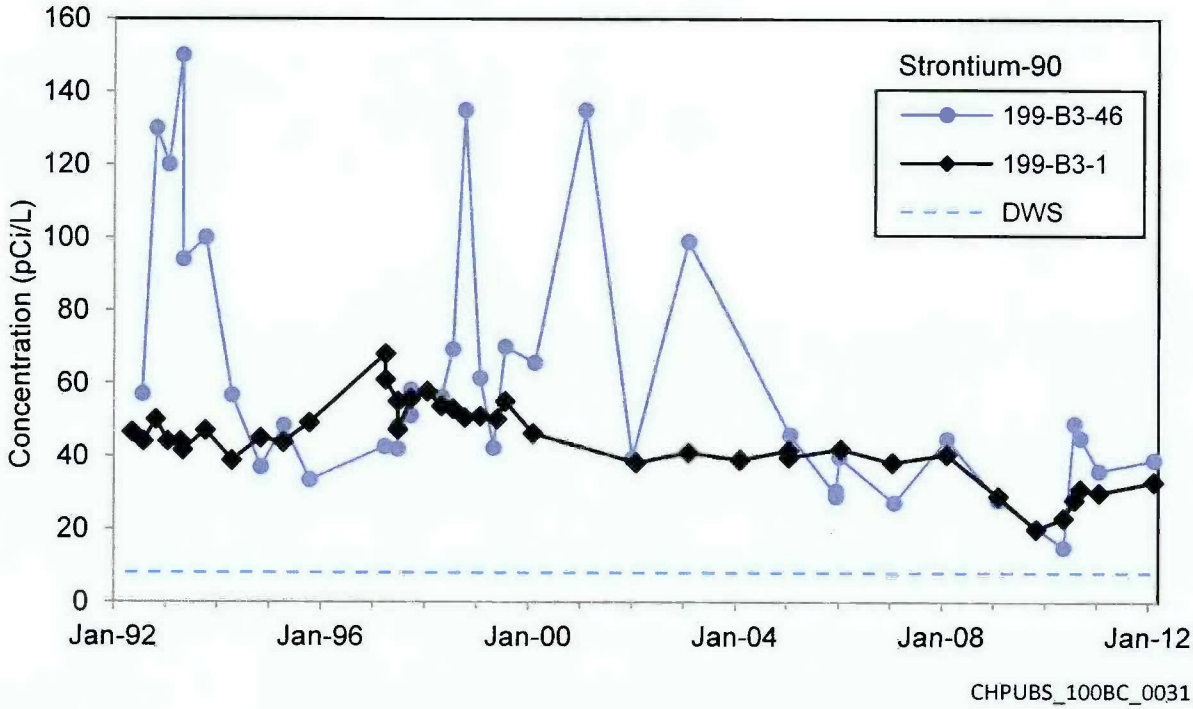


Figure 4-42. Strontium-90 Wells 199-B3-1 and 199-B3-46

4.3.2.3 Tritium in Groundwater

Figures 4-43 and 4-44 depict the tritium plume in 100-BC groundwater in May and September 2010, and Figure 4-45 shows the plume in 2011. Concentrations above 2,000 pCi/L are observed beneath much of the area, and two portions of the plume have concentrations above the 20,000 pCi/L drinking water standard. Differences in the maps are explained by variability in concentrations in individual wells, as explained below. Changing river stage between spring and fall appeared to have little influence on plume configuration.

Tritium concentrations are declining in many parts of 100-BC because of migration, dispersion, and radioactive decay. This pattern is especially evident in wells in western 100-BC and in the 600 Area (Figure 4-46). Tritium in well 699-72-77, located southeast of 100-BC, originated in the 200 Areas. Peak concentrations have passed this well and the plume is attenuating.

Tritium concentrations have spiked several times in Well 199-B5-2 in the northern plume since the 1990s (Figure 4-47). Each spike has been smaller than the previous one; in 2010, the maximum concentration was 69,000 pCi/L in September. Other wells in northeastern 100-BC (for example, 199-B3-47) also have variable tritium concentrations.

Increases in tritium concentrations in eastern 100-BC between 2008 and early 2012 (Figure 4-48) indicate movement of the southern plume to the northeast. This is consistent with a hydraulic gradient sloping to the northeast or east in southern 100-BC (Section 3.6) and with movement of the chromium plume.

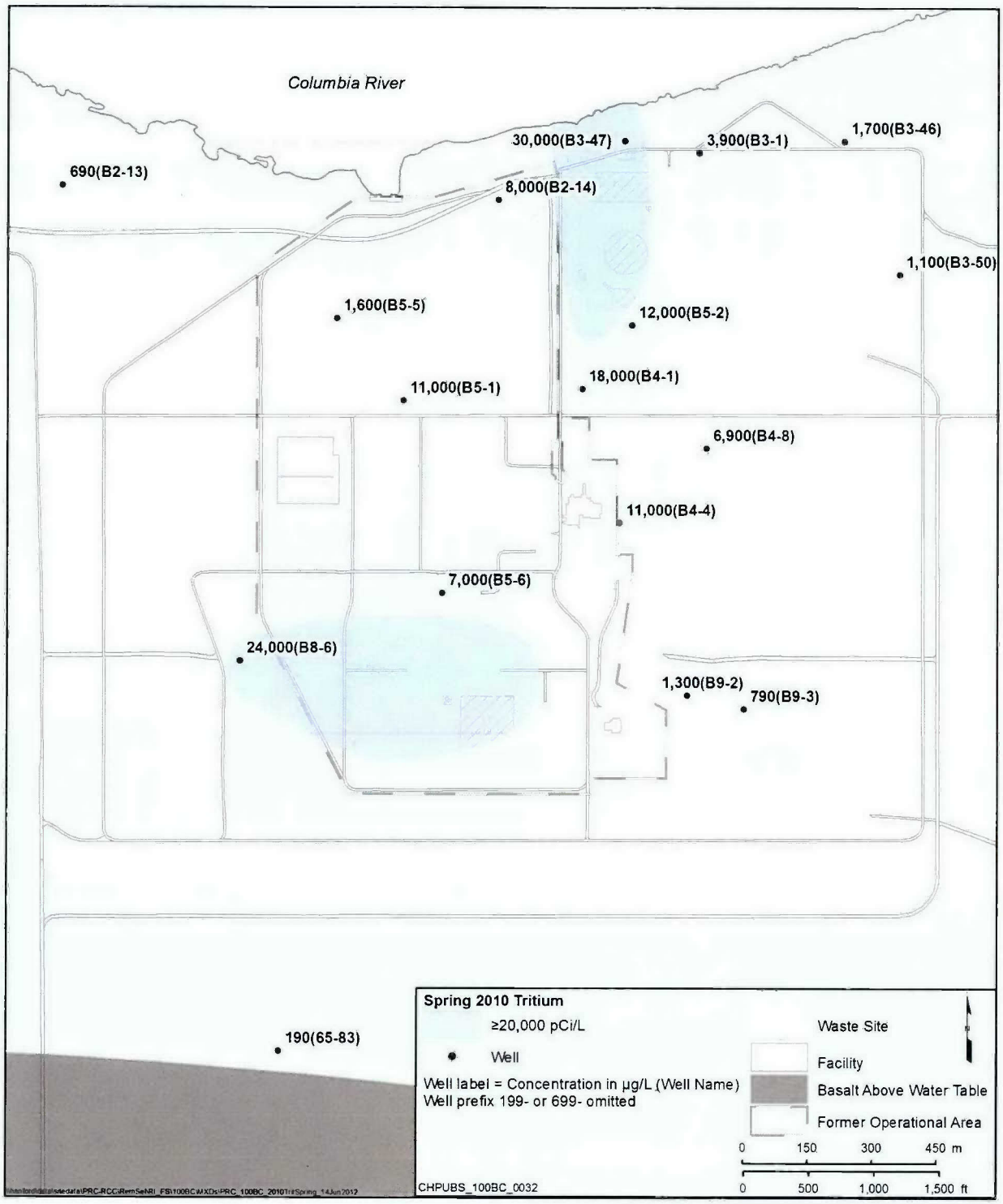


Figure 4-43. Tritium in 100-BC Unconfined Aquifer, Spring 2010

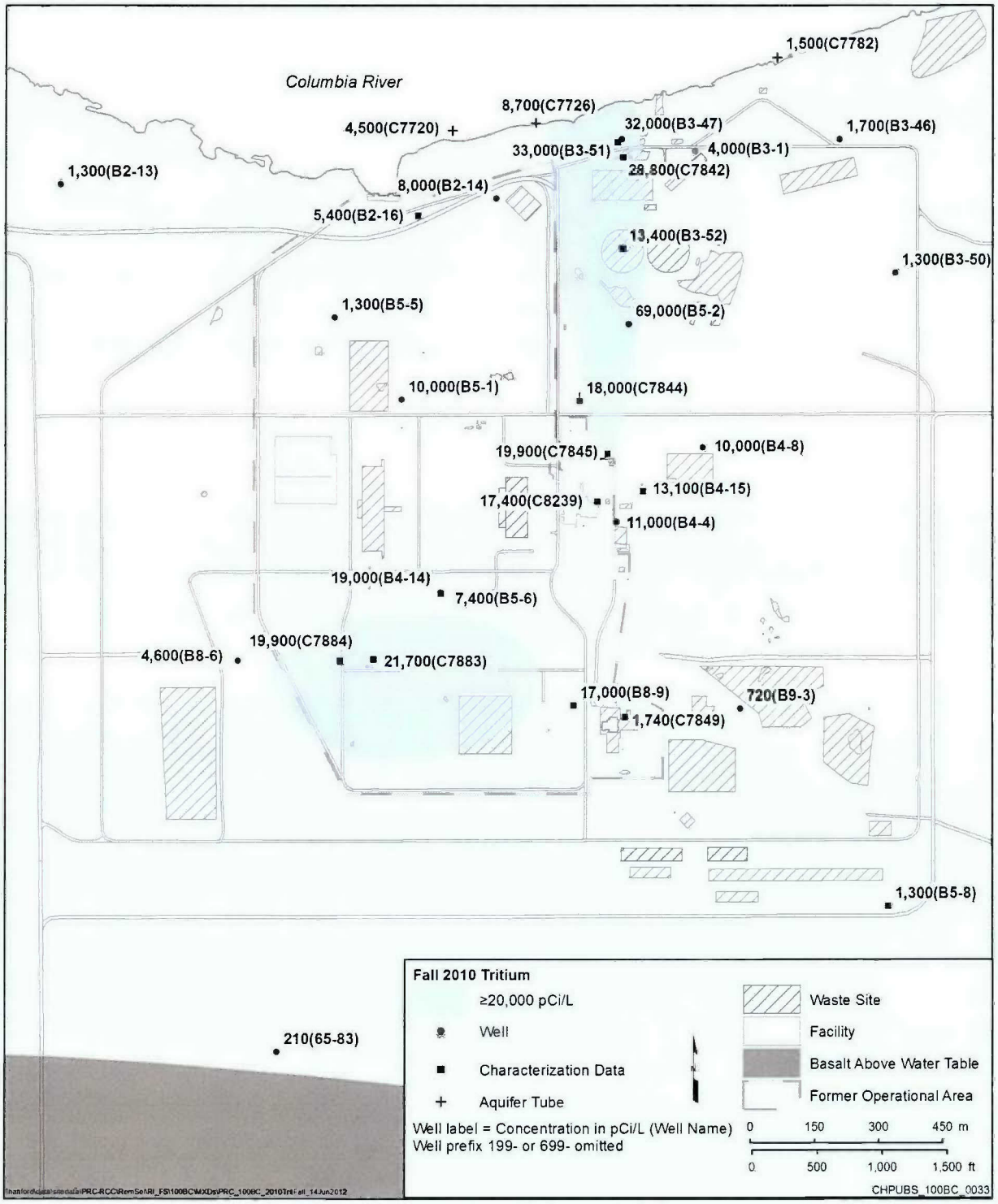


Figure 4-44. Tritium in 100-BC Unconfined Aquifer, Fall 2010

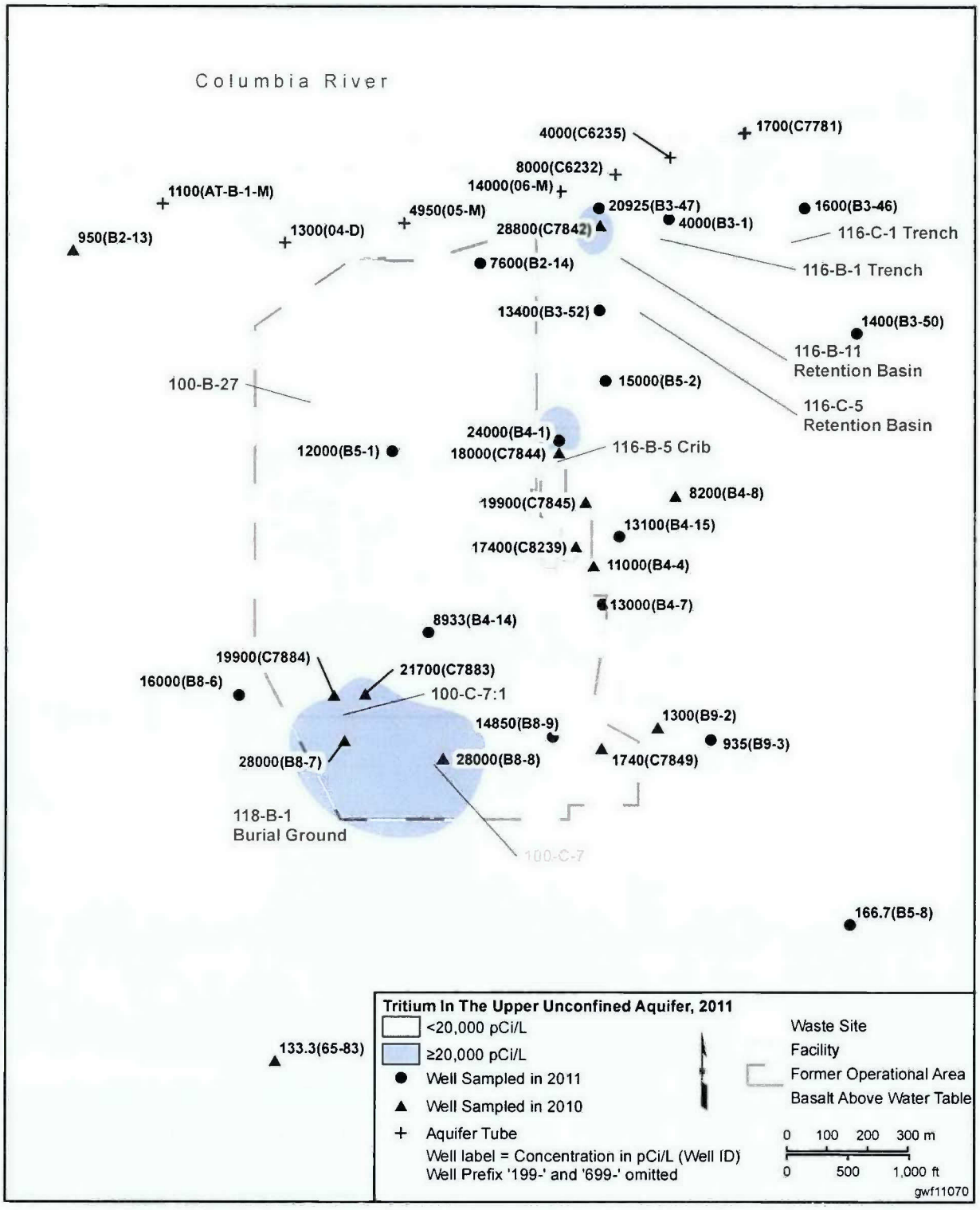


Figure 4-45. Tritium in 100-BC Unconfined Aquifer, 2011

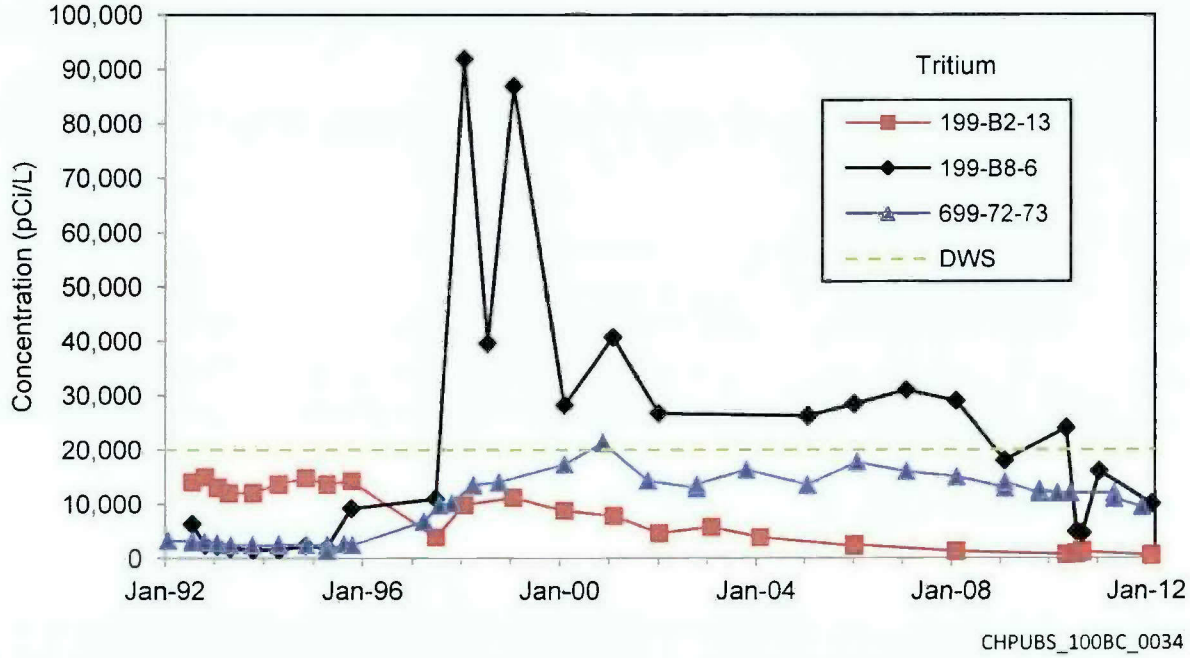


Figure 4-46. Declining Tritium Levels in Wells in and Near 100-BC.

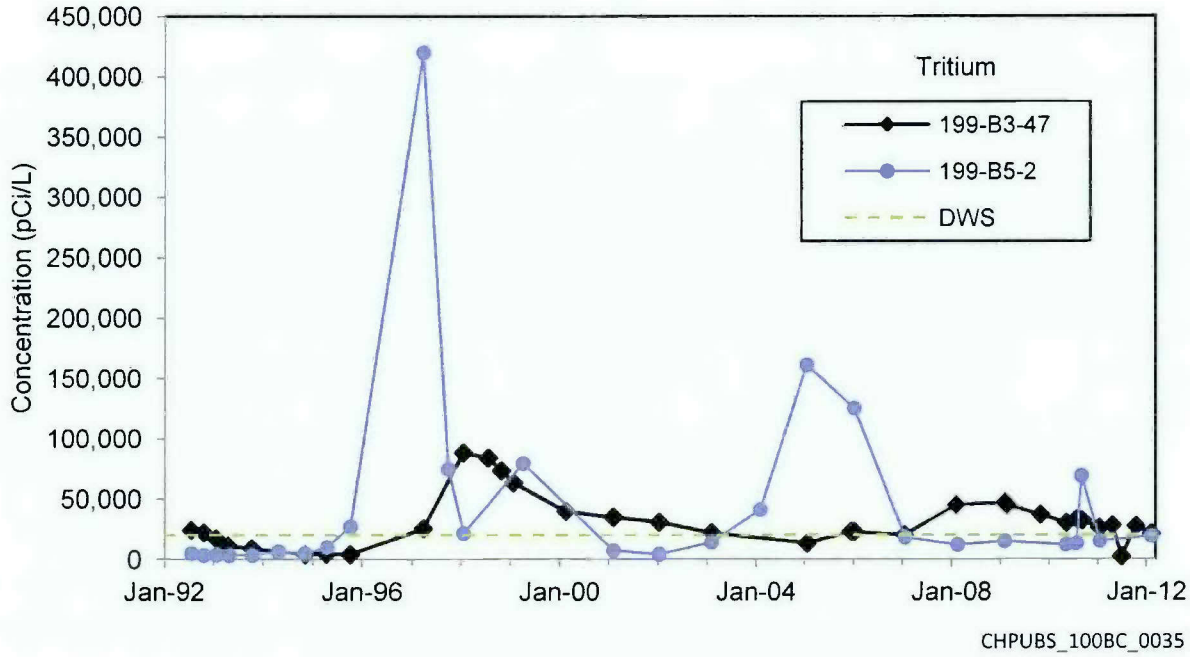


Figure 4-47. Tritium in Wells 199-B3-47 and 199-B5-2

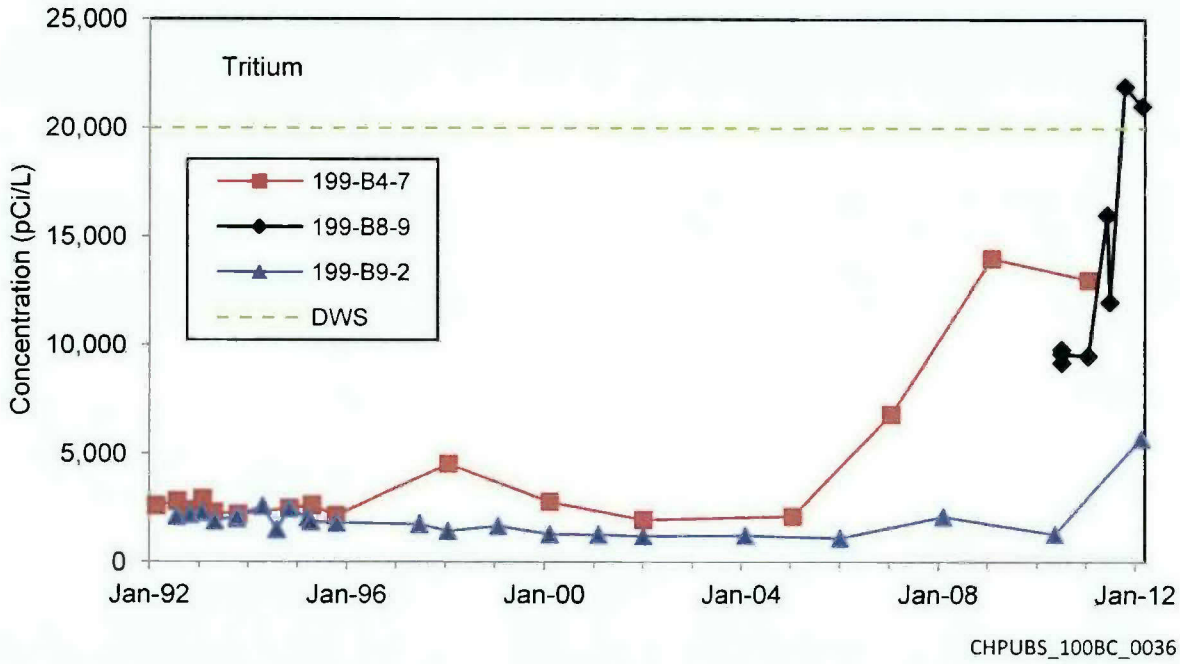


Figure 4-48. Increasing Tritium Levels in Wells in Eastern 100-BC

4.3.2.4 Nitrate in Groundwater

Nitrate concentrations range from 4 to 14 mg/L (as NO_3^-) beneath most of 100-BC. Concentrations are higher in two wells near the Columbia River: 199-B3-1 (20 mg/L in July 2010) and 199-B3-47 (43.8 mg/L in July 2010). Both of these wells had nitrate concentrations above the drinking water standard in the past (Figure 4-49). Concentrations have recently increased to near the drinking water standard in Well 199-B3-47, perhaps because of movement of the plume. Concentrations in aquifer tube 06-M also have been near the drinking water standard. Plume maps are not provided for nitrate because concentrations are below the drinking water standard.

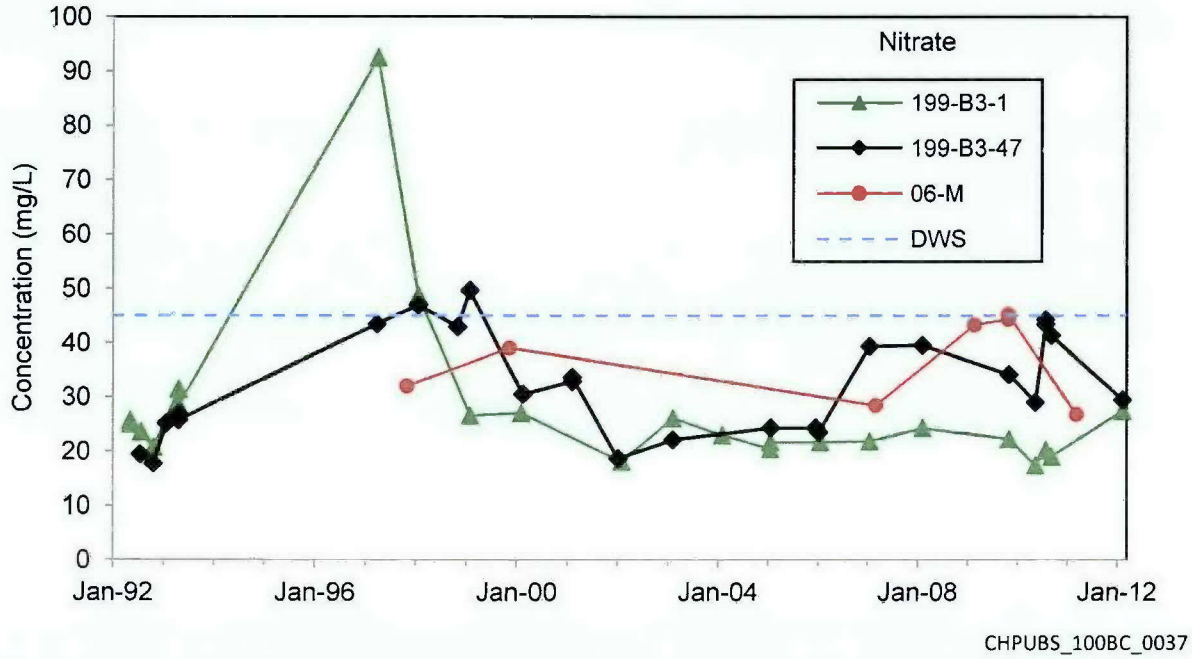


Figure 4-49 Nitrate in Wells 199-B3-1, 199-B3-47 and Aquifer Tube 06-M

4.3.2.5 Trichloroethene in Groundwater

Volatile organic compounds are not routinely monitored in 100-BC wells, but were analyzed for the RI in 2010. Based on the 2010 results, VOCs were added to routine monitoring list for selected wells in 2011 and 2012. Trichloroethene concentrations exceed the MTCA "Groundwater Cleanup Standards" (WAC 173-340-720) groundwater cleanup level of 0.49 µg/L and the EQL (1 µg/L) in many wells throughout 100-BC (Figure 4-50), Most of these results are flagged J, indicating they are estimated concentrations that are above the method detection limit but below the contractually required detection limit. All concentrations were below the 5 µg/L drinking water standard and 1×10^{-5} for chemicals based on the MTCA "Human Health Risk Assessment Procedures" (WAC 173-340-708(5)) cumulative risk threshold. Therefore trichloroethene is not considered a final COPC for groundwater in 100-BC. Sources of trichloroethene contamination are unknown. The 100-BC waste sites cannot solely be responsible because trichloroethene contamination occurs in upgradient and 600 Area wells at levels similar to those in 100-BC wells.

Figure 4-50 illustrates the maximum reported trichloroethene detections in 100-BC groundwater between January 2010 and February 2012. The data are not contoured because concentrations are so close to detection limits and most are flagged J," and because there is no obvious pattern to the distribution. Concentrations > 1 µg/L are detected throughout 100-BC, and also in wells 199-B5-8 and 699-65-72, located southeast of 100-BC.

Samples from Wells 199-B5-5, 199-B5-6, and 199-B5-8 contained slightly higher concentrations of trichloroethene (up to 3.3 µg/L; Figure 4-51) than other wells. Wells 199-B5-5 and 199-B5-6 are screened in the lower portion of the unconfined aquifer and 199-B5-8 is screened at the top of the aquifer.

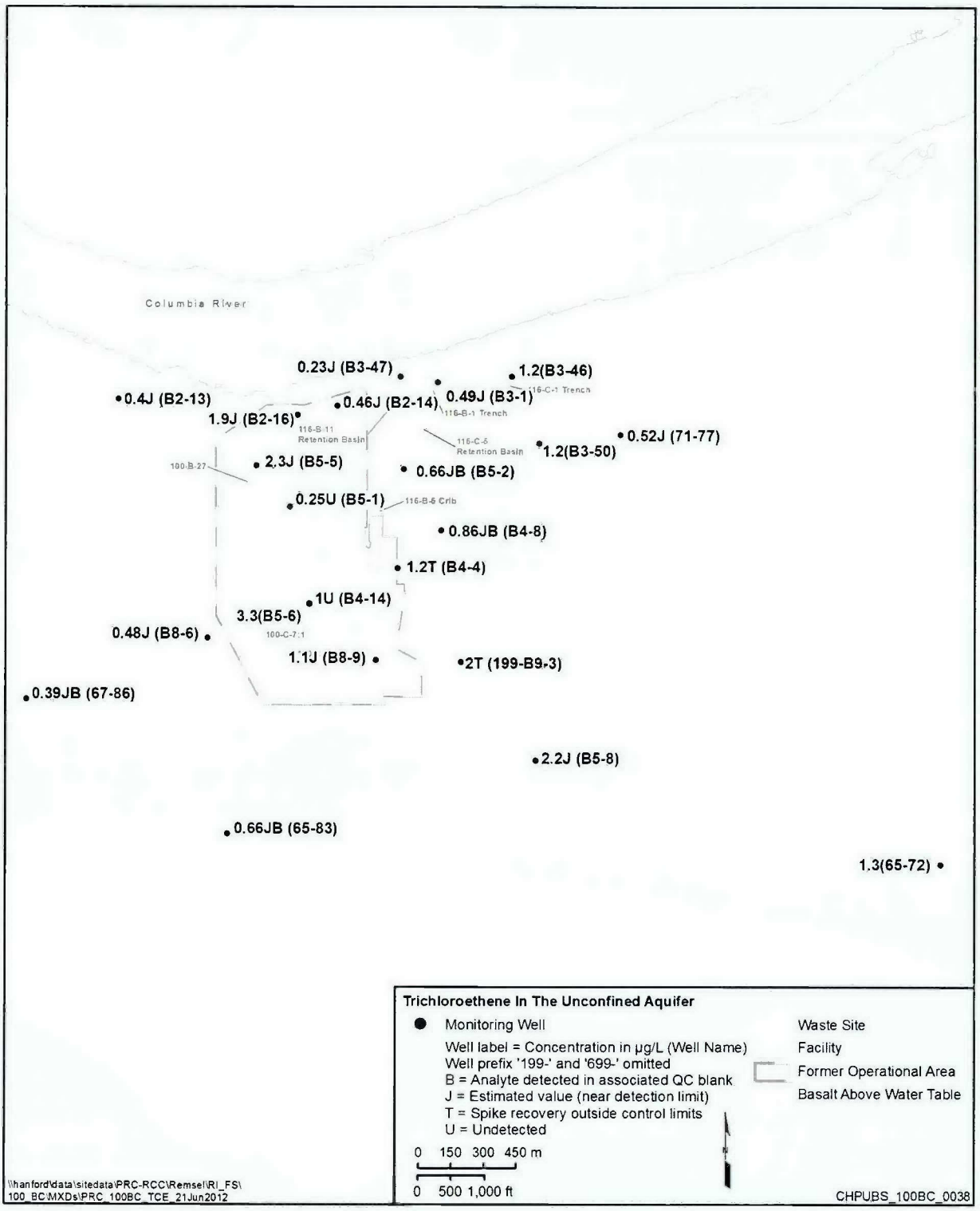


Figure 4-50. Trichloroethene in 100-BC Unconfined Aquifer (Maximum Concentration, January 2010 through February 2012)

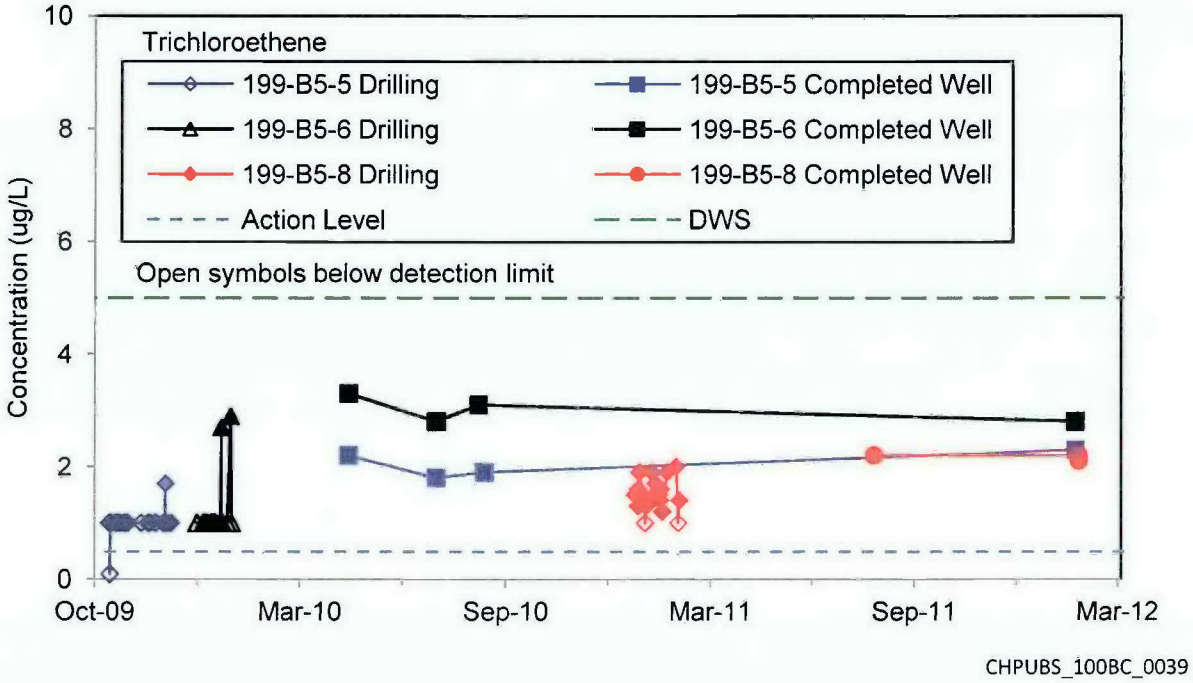


Figure 4-51. Trichloroethene in Wells 199-B5-5, 199-B5-6, and 199-B5-8

4.3.2.6 Carbon Tetrachloride in Characterization Samples

Groundwater samples collected during drilling of new wells were analyzed for volatile organic compounds. Carbon tetrachloride was detected in a percentage of the samples at low levels, but it appears the detections were not representative of aquifer conditions. Samples collected from completed wells for the RI did not detect carbon tetrachloride (Section 4.3.1).

The detection limit for carbon tetrachloride (1 µg/L) is higher than the MTCA "Groundwater Cleanup Standards" (WAC 173-340-720) groundwater cleanup level. Thus, any detection of carbon tetrachloride was an exceedance. This contaminant was detected in 13 of 191 100-BC characterization samples (6.8 percent). All but one of the results were flagged "J" (near detection limit). The maximum (unflagged) was 6 µg/L in upgradient Well 199-B5-8. The detections were not associated with specific instances of field blank contamination. An evaluation of the entire Hanford Site groundwater monitoring program showed detections of carbon tetrachloride in 6.7 percent of field blanks in 2010 and early 2011, at concentrations as higher or higher than observed in 100-BC groundwater samples (Figures 3 and 4 of SGW-52194, *VOC Contamination in Groundwater Samples*). The contamination was attributed to contamination of the deionized water used to generate field blanks, which does not explain their occurrence in 100-BC groundwater samples. None of the samples collected from completed wells had carbon tetrachloride detections, including results of analyses with lower detection limits (0.12 µg/L). These data included samples from Wells 199-B2-14, 199-B3-50, 199-B5-5, and 199-B5-8, all of which had detections in characterization samples. The cause of the false detections in the 100-BC characterization samples is unknown.

4.3.3 Vertical Distribution of Contaminants in Groundwater

Data Gaps 4 and 6 include tasks to provide additional information about vertical contaminant distribution in groundwater. Before recent RI studies, little information was available about vertical distribution in the thick, unconfined aquifer beneath 100-BC. In addition, only one well was screened beneath the

unconfined aquifer. Sources of information about vertical distribution of groundwater contaminants include the following:

- Characterization data collected during installation of wells in 2009 through 2011
- Monitoring data from well clusters
  - 199-B2-12 (confined Ringold), 199-B3-51 (bottom of unconfined aquifer), and 199-B3-47 (top of unconfined aquifer)
  - 199-B5-6 (bottom of unconfined aquifer) and 199-B4-14 (top of unconfined aquifer)
  - 199-B2-15 (confined Ringold) and 199-B2-14 (top of unconfined aquifer)
- Aquifer tube clusters screened at various depths in the unconfined aquifer

During drilling of new monitoring wells, water samples were collected every 1.5 m (5 ft) through the thickness of the unconfined aquifer and analyzed for COPCs. These were the first comprehensive data collected to define vertical distribution of contaminants through the entire aquifer thickness.

The drilling process can create chemically reducing conditions in the borehole. This has the effect of lowering concentrations of Cr(VI) and dissolved total chromium, and increasing concentrations of some other metals like manganese. The effect can be minimized by purging the borehole until the dissolved oxygen of the water increases and indicates an oxidizing environment. During drilling of the first few wells, relatively high-flow pumps were used, and sufficient flow could not always be maintained to achieve an adequate purge. Later, lower-flow pumps were used and most intervals could be purged adequately. Project staff analyzed purge rates/volumes, dissolved oxygen, chromium, and manganese concentrations from the wells installed in 2009 and early 2010, and determined a general rule that dissolved oxygen above 6 mg/L (preferably above 7 mg/L) appeared to indicate representative samples. Chromium data from samples with lower dissolved oxygen (or other factors indicating insufficient purging) were subsequently flagged as suspect in HEIS using the project's data review process. Figures in this section indicate suspect data (for this or other reasons) with an X.

4.3.3.1 Vertical Distribution of Groundwater Contamination in Wells 199-B2-14 and 199-B2-15

This well pair is located near the Columbia River within an area contaminated by Cr(VI) and strontium-90. Well 199-B2-14 is screened in the upper 5 m (16 ft) of the aquifer and Well 199-B2-15 is screened in the RUM. The unconfined aquifer is entirely in Ringold unit E at this location.

In general, contaminant concentrations decreased with depth in the aquifer at Well 199-B2-14 Figure 4-52. Chromium concentrations greater than 15 μg/L were observed only in the upper half of the aquifer. The maximum concentrations were 22 to 23 μg/L. Cr(VI) concentrations declined steadily below 26 m (85 ft) depth and were below or near detection limits at the bottom of the aquifer. Cr(VI) concentrations of routine samples collected after well completion were comparable to characterization data from the same interval.

Tritium and nitrate concentrations were below their drinking water standards in all samples from Well 199-B2-14, and decreased with depth. Strontium-90 concentrations were below detection limits in all but three characterization samples. The maximum concentration was 5.3 ± 4.4 pCi/L at 4.3 m (14 ft) below the water table. The other two detections were much deeper in the aquifer, but were barely above detection limits. Routine samples collected after completion of the well at the top of the aquifer had strontium-90 concentrations ranging from undetected to 8.6 ± 2.8 pCi/L.

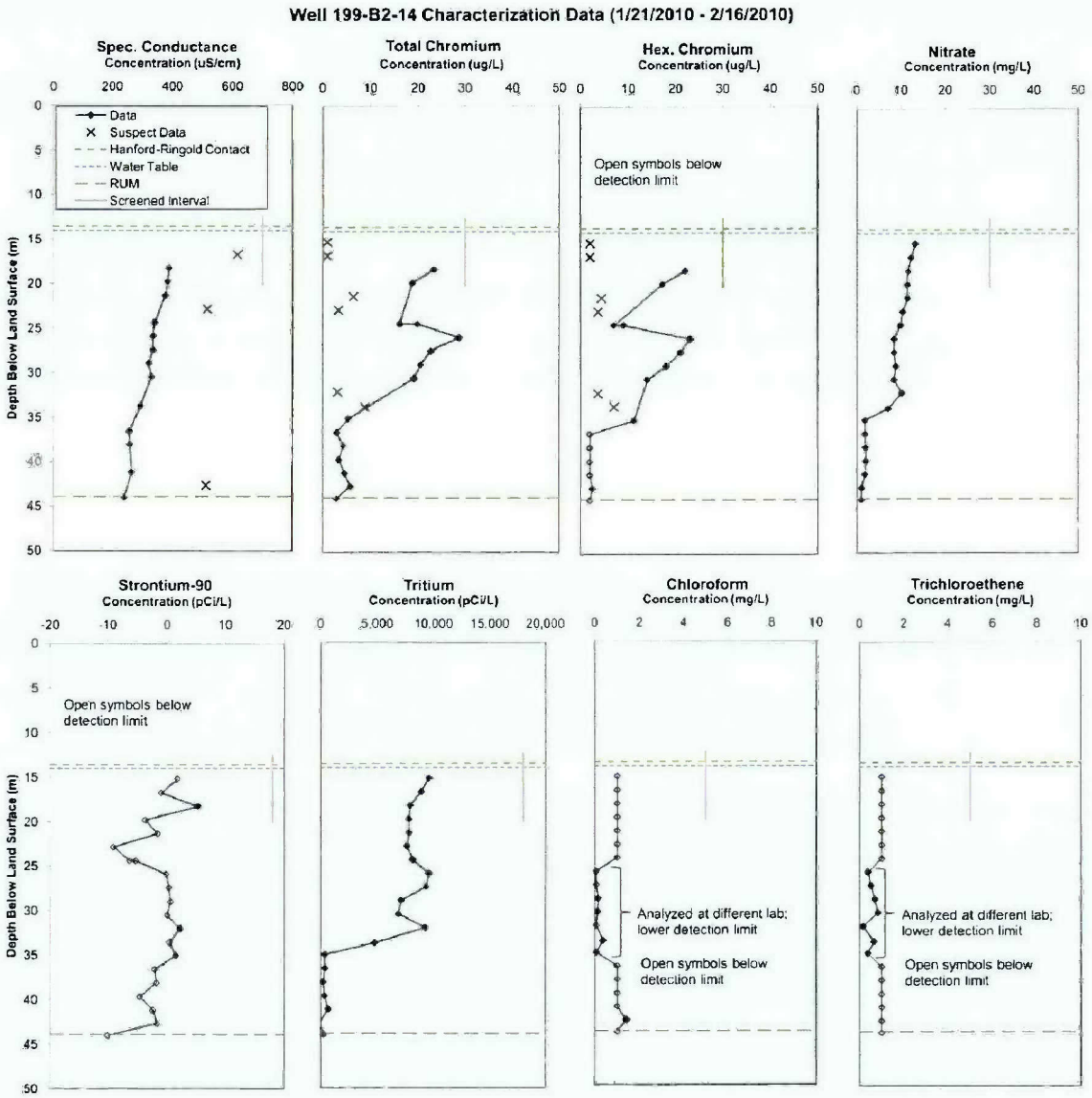


Figure 4-52. Vertical Distribution of Contaminants in Well 199-B2-14

Chloroform and trichloroethene concentrations in characterization samples from Well 199-B2-14 were <1 µg/L.

Because characterization data were available from Well 199-B2-14, characterization samples were not collected in the unconfined aquifer during drilling of Well 199-B2-15. Characterization samples were collected in the confined Ringold, where chloroform, Cr(VI), strontium-90, trichloroethene, and tritium were undetected. Nitrate concentrations were lower than in the unconfined aquifer.

4.3.3.2 Vertical Distribution of Groundwater Contamination in Well 199-B2-16

This well, which is located near the Columbia River 100-BC water intake structure, was screened in the lower part of the aquifer. The aquifer is entirely in Ringold unit E at this location.

Cr(VI) was detected throughout the thickness of the aquifer. Concentrations ranged from 5 to 22 µg/L, increasing slightly with depth (Figure 4-53). Strontium-90 was detected at levels below the drinking water standard in two samples, both in the upper portion of the aquifer. Nitrate and tritium concentrations were far below the drinking water standard and declined with depth. Chloroform was detected at the bottom of the aquifer at concentrations up to 4.8 µg/L. Trichloroethene concentrations were above detection limits in three samples in the lower half of the aquifer (maximum of 2.1 µg/L). After well completion, chromium concentrations increased from 15 to approximately 25 µg/L. Strontium-90 and tritium concentrations in routine samples were comparable to those in characterization samples.

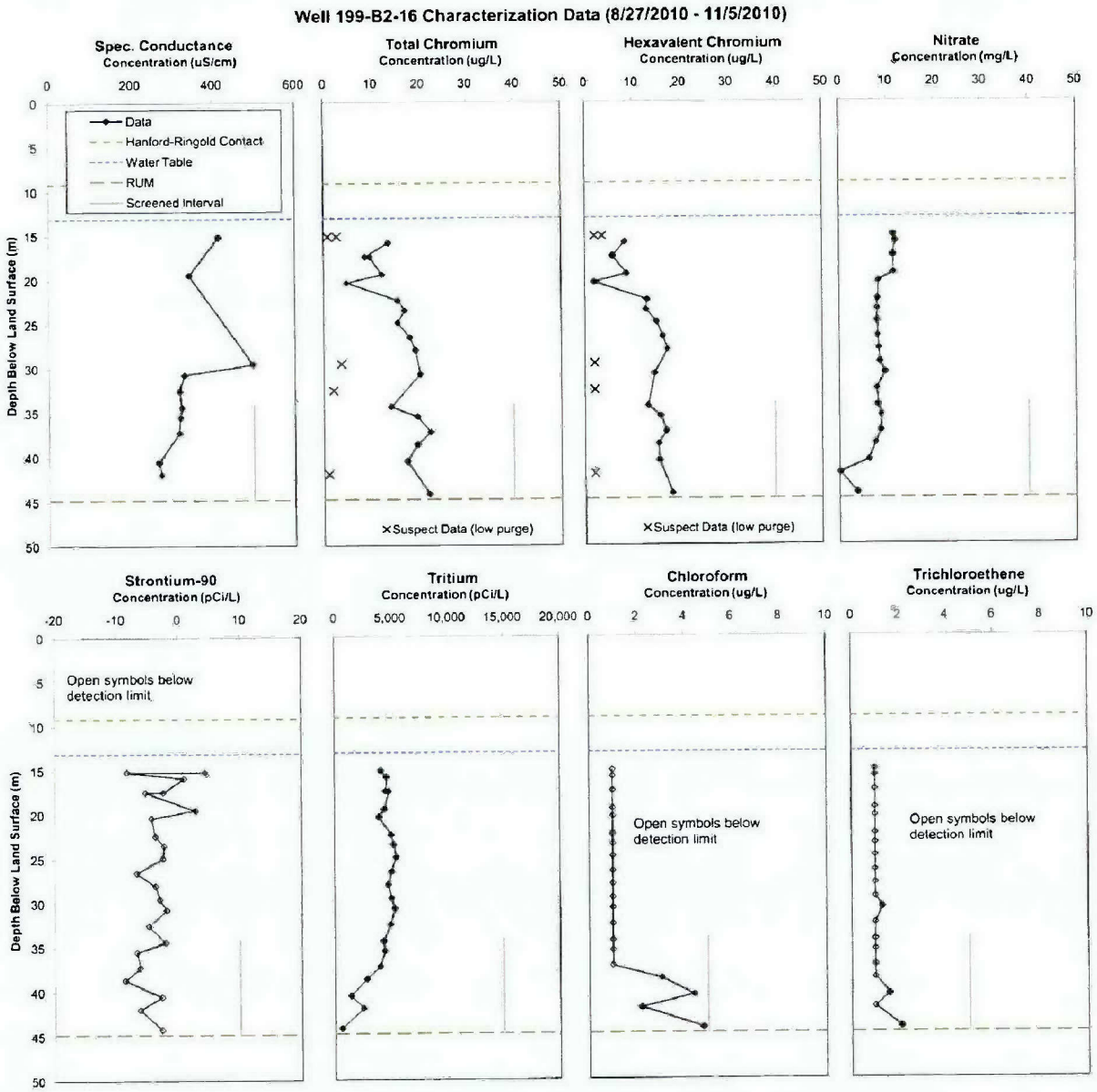


Figure 4-53. Vertical Distribution of Contaminants in Well 199-B2-16

4.3.3.3 Vertical Distribution of Groundwater Contamination in Well 199-B3-50

This well, which is located in northeastern 100-BC, was intended to delimit the eastern extent of the Cr(VI) and strontium-90 plumes. It is screened at the top of the unconfined aquifer in the Hanford formation. Well 199-B3-50 has only low levels of groundwater contaminants (Figure 4-54).

The highest Cr(VI) concentration during drilling was 11 µg/L at a depth of 31 m (102 ft) below land surface (8.7 m [29 ft] below the water table). Concentrations in deeper samples were all less than 10 µg/L. Chromium concentrations in routine samples collected after well completion were slightly higher than characterization samples (15 to 21 µg/L).

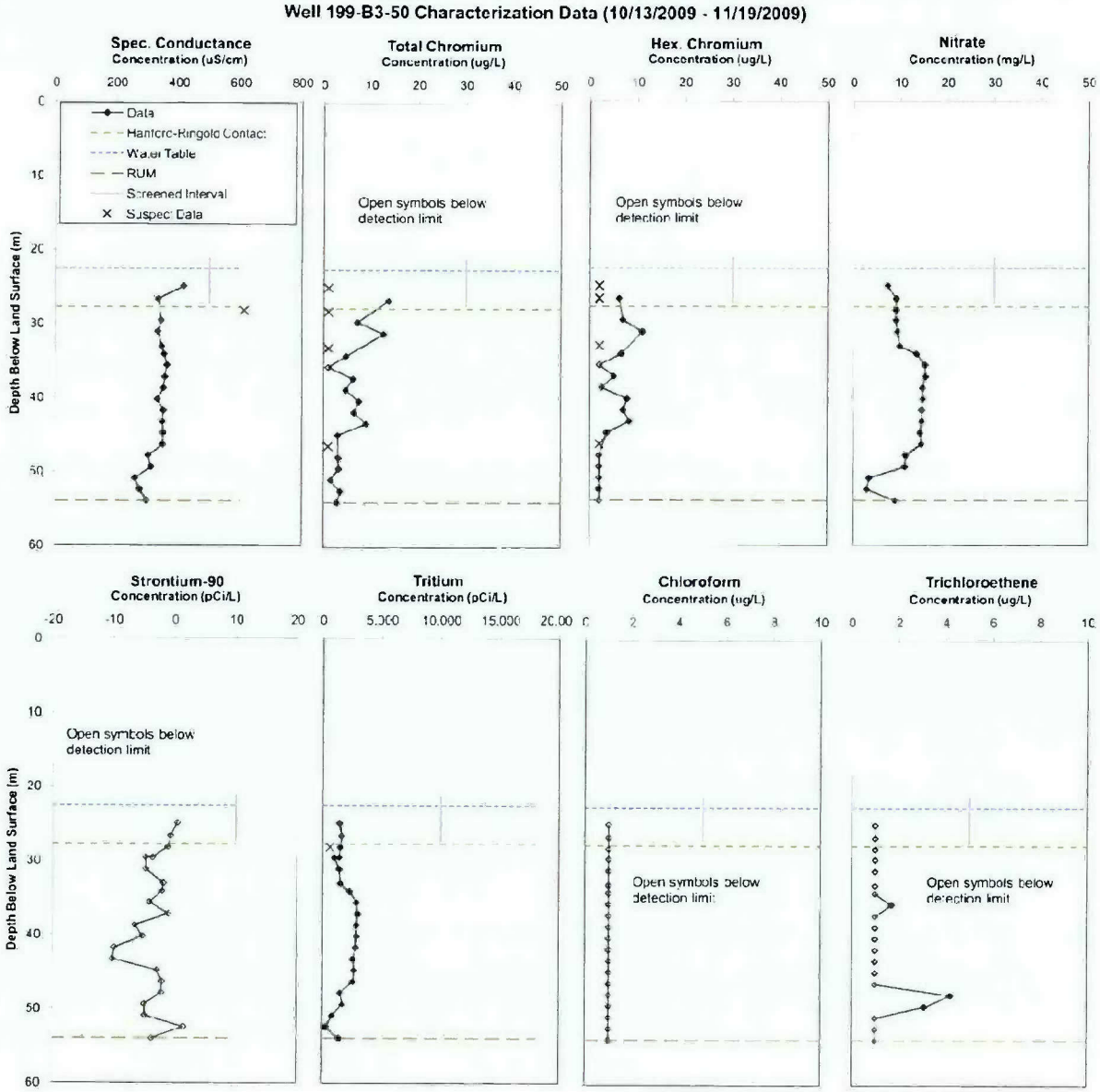


Figure 4-54. Vertical Distribution of Contaminants in Well 199-B3-50

Tritium and nitrate concentrations were all far below their drinking water standards and were highest in the middle portion of the aquifer. Strontium-90 was undetected in all samples. Concentrations of these contaminants in the completed well were comparable to those from characterization samples at equivalent depths.

Chloroform was undetected at all depths. Trichloroethene was detected in three samples, with the maximum of 4.2 μg/L. The highest concentrations were near the bottom of the aquifer.

4.3.3.4 Vertical Distribution of Groundwater Contamination in Wells 199-B3-51, 199-B2-12, and 199-B3-47

New Well 199-B3-51 was installed adjacent to two older wells to provide data on contaminant and head distribution with depth. Well 199-B3-47 is screened at the top of the unconfined aquifer, Well 199-B3-51 at the base of the unconfined aquifer, and Well 199-B2-12 in the RUM. Characterization data were unavailable for the older wells, so samples were collected during drilling of the new well. The water table is in Ringold unit E at this location.

Well 199-B3-47, screened at the top of the unconfined aquifer, has some of the highest levels of contaminants in 100-BC. Tritium and strontium-90 are above their drinking water standards; Cr(VI) is above the AWQC; and nitrate is near the drinking water standard.

Characterization samples from new Well 199-B3-51 showed a general decrease in contamination with depth (Figure 4-55). Cr(VI) concentrations were near 20 μg/L at the top of the aquifer and declined to below the detection limit in the bottom half of the aquifer. Strontium-90 and tritium concentrations were above the drinking water standard near the top of the aquifer and declined to lower levels at 8 m (26 ft) below the water table. Nitrate concentrations were below the drinking water standard in all samples and declined with depth. No chloroform or trichloroethene were detected in any samples.

Contaminant concentrations are low in Well 199-B2-12, screened in the RUM. Chromium concentrations have been near or below detection limits since the well was installed in 1992. Nitrate concentrations are less than 2 mg/L, much lower than in the unconfined aquifer. Strontium-90 and tritium are undetected.

4.3.3.5 Vertical Distribution of Groundwater Contamination in Wells 199-B4-14 and 199-B5-6

This well pair is located in central 100-BC with an objective of defining contaminant plumes. Well 199-B4-14 is screened at the top of the unconfined aquifer in the Hanford formation. Well 199-B5-6 is screened in the lower portion of the unconfined aquifer, in Ringold unit E. Characterization samples were collected during drilling of the deeper well.

Many of the Cr(VI) characterization results from Well 199-B5-6 were unrepresentative because recharge was slow and dissolved oxygen of the water remained low. However, the remaining data were sufficient to define an apparent bimodal distribution in the aquifer, with maximum concentrations (>30 μg/L) near the water table and near the base of the aquifer (Figure 4-56). Concentrations in routine samples collected after completion of Well 199-B5-6 in 2010 were comparable. Concentrations increased in late 2011 and early 2012 (Section 4.3.2).

The vertical profile for tritium differed from Cr(VI). Tritium concentrations exceeded the drinking water standard in the upper portion of the aquifer and decreased with depth. Nitrate concentrations were low throughout the aquifer, and strontium-90 was undetected. Tritium, nitrate, and strontium-90 concentrations in the completed wells initially were consistent with characterization results. Tritium concentrations later declined. Chloroform was detected in two samples near the bottom of the aquifer, with a maximum concentration of 2.2 µg/L. Trichloroethene also was detected in two samples. However, a duplicate of one of the samples showed no detectable trichloroethene. The maximum concentration was 2.9 µg/L. After completion of the well, trichloroethene concentrations ranged from 2.2 to 3.3 µg/L, and chloroform from 1.5 to 1.6 µg/L (Figure 4-51).

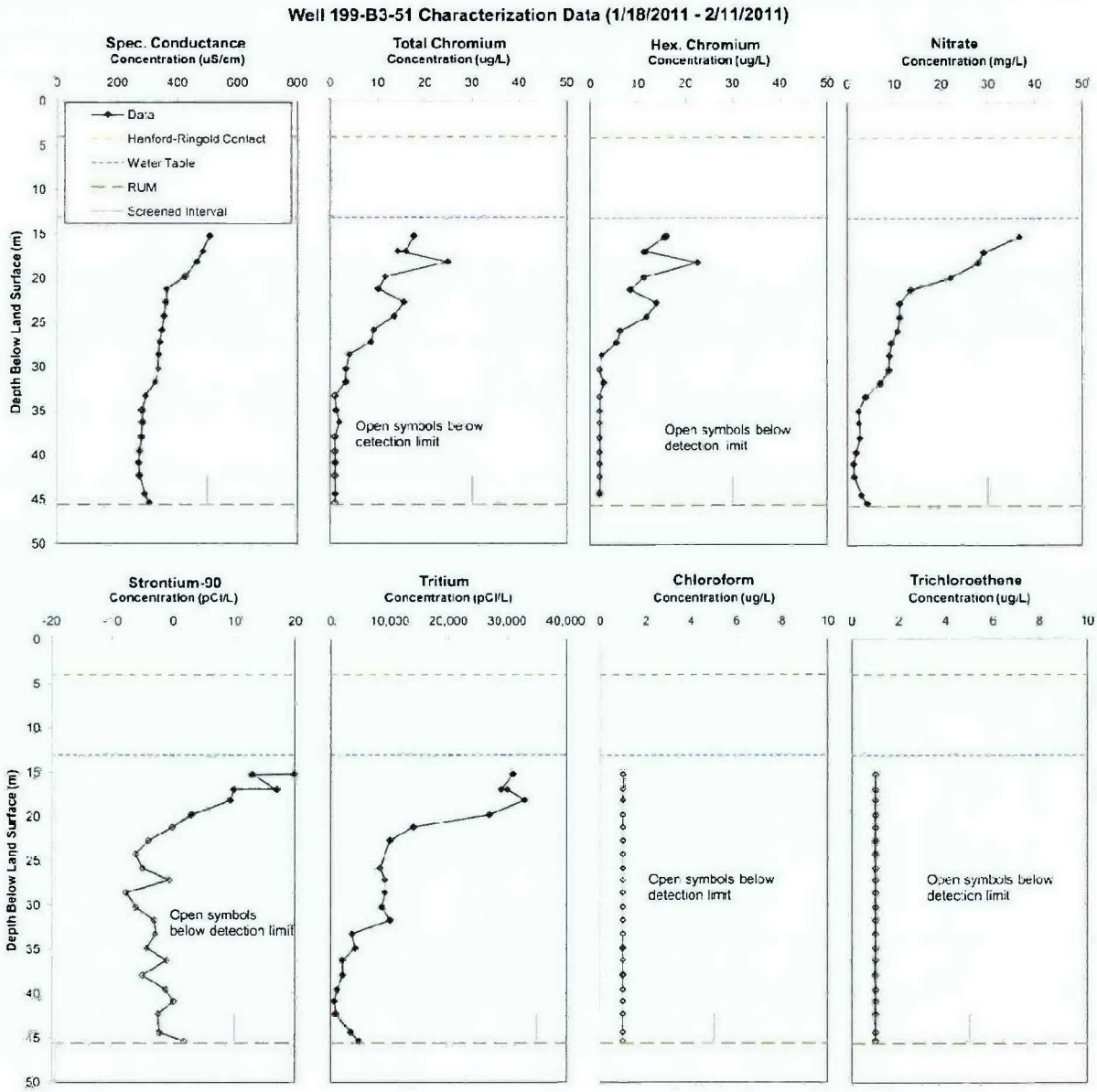


Figure 4-55. Vertical Distribution of Contaminants in Well 199-B3-51

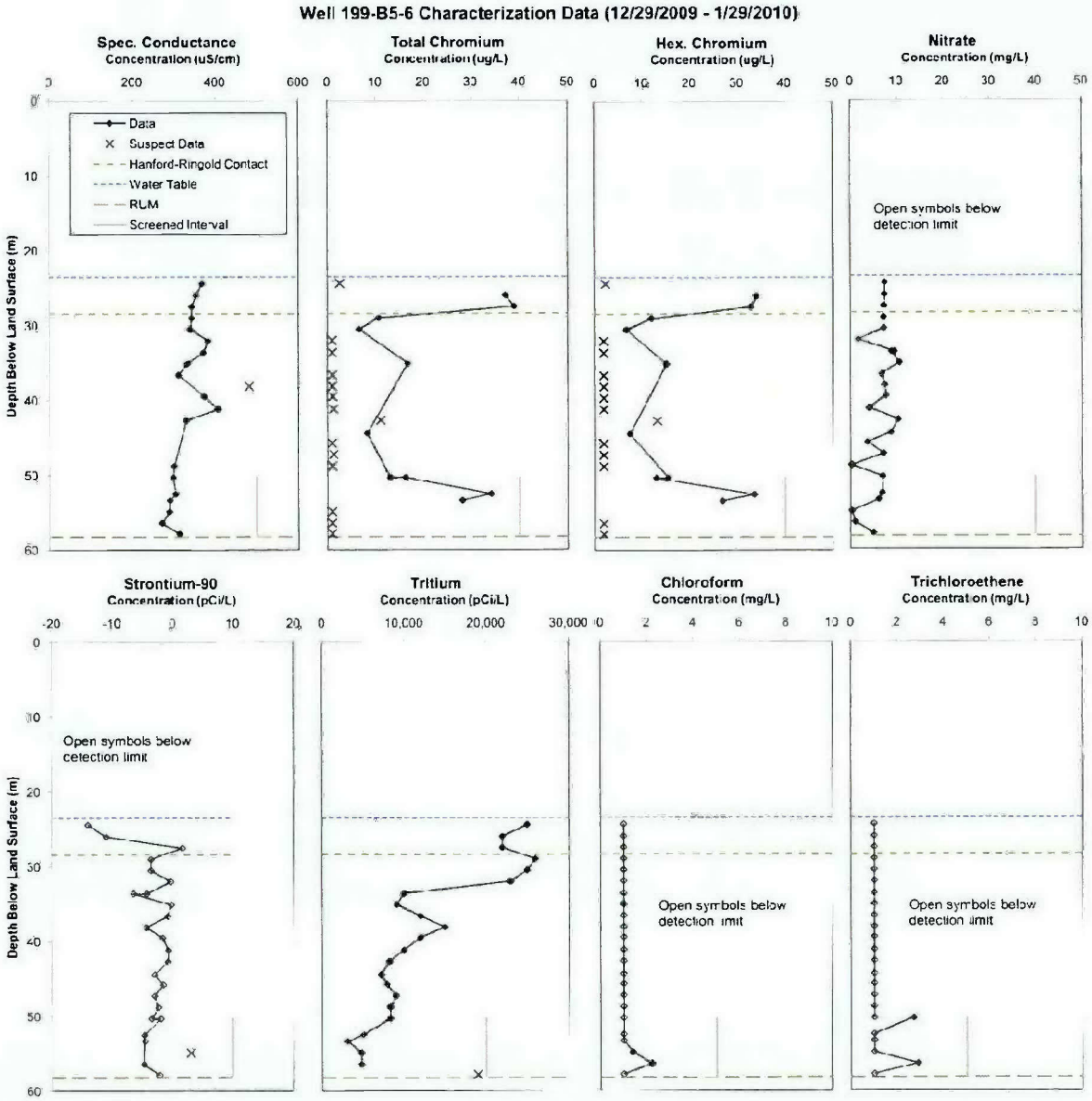


Figure 4-56. Vertical Distribution of Contaminants in Well 199-B5-6

4.3.3.6 Vertical Distribution of Groundwater Contamination in Well 199-B5-5

This well is located in northwestern 100-BC, downgradient of the 100-B-27 waste site, which was excavated to groundwater in two phases. The initial excavation was in 2007. This was followed by additional excavation to groundwater in 2009. The well is screened in the lower portion of the unconfined aquifer, in Ringold unit E.

Chromium concentrations in Well 199-B5-5 were elevated in a thin layer at the top of the unconfined aquifer (Hanford formation) and at a depth of 37 to 56 m (121 to 184 ft) bgs (bottom portion of Ringold unit E; Figure 4-57). The highest concentration was 31.5 µg/L at a depth of 45 m (148 ft) bgs (30 m [98 ft] below the water table). Concentrations gradually decreased below this depth. Cr(VI)

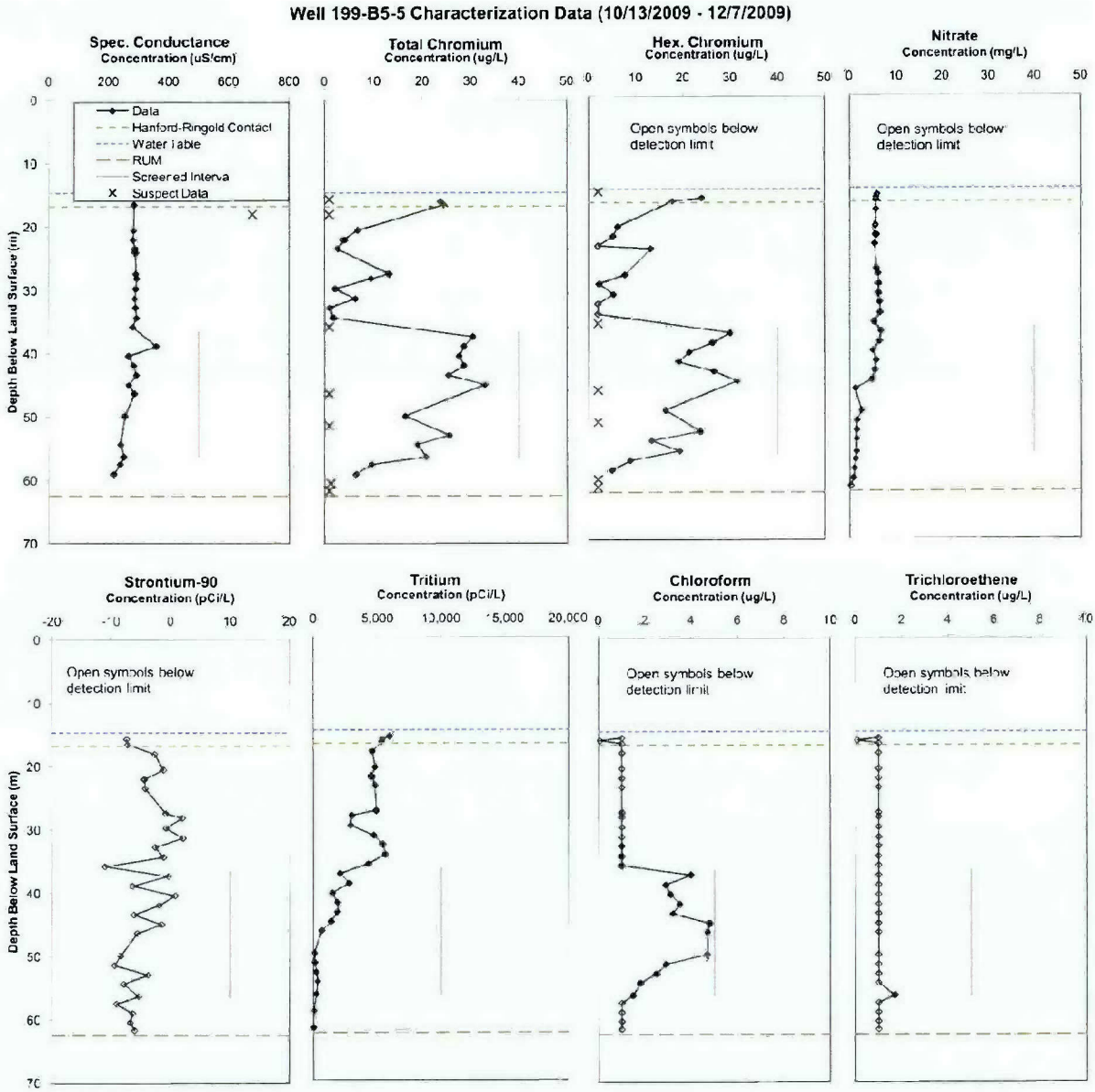


Figure 4-57. Vertical Distribution of Contaminants in Well 199-B5-5

concentrations in routine samples collected after well completion (32 to 38 µg/L) were comparable to characterizations samples from the same depths.

Nitrate and tritium concentrations were far below their drinking water standards and, unlike Cr(VI), decreased with depth in the aquifer. Strontium-90 was undetected in all samples. Tritium, nitrate, and strontium-90 concentrations in the completed well were consistent with characterization results.

Chloroform was detected at concentrations up to 4.8 µg/L 37 to 56 m (121 to 184 ft) bgs. The well was screened across this depth, and samples collected after well completion had concentrations ranging from 2.2 to 3.0 µg/L. Trichloroethene was detected in one deep characterization sample, and ranged from 1.8 to 2.2 µg/L after well completion (Figure 4-51).

4.3.3.7 Vertical Distribution of Groundwater Contamination in Well 199-B5-8

This well is located southeast of 100-BC, with an objective of defining the southern extent of contamination. It is screened in the upper portion of the unconfined aquifer, in the Hanford formation.

Cr(VI) concentrations were near or below detection limits throughout the thickness of the unconfined aquifer (Figure 4-58). Nitrate and tritium concentrations were low and did not show vertical stratification. No strontium-90 or chloroform was detected. Trichloroethene was detected at low concentrations (≤2 µg/ L) throughout the aquifer. Cr(VI) levels in the completed well (screened at the top of the aquifer) were higher than in characterization samples, ranging from 6 to 13 µg/L.

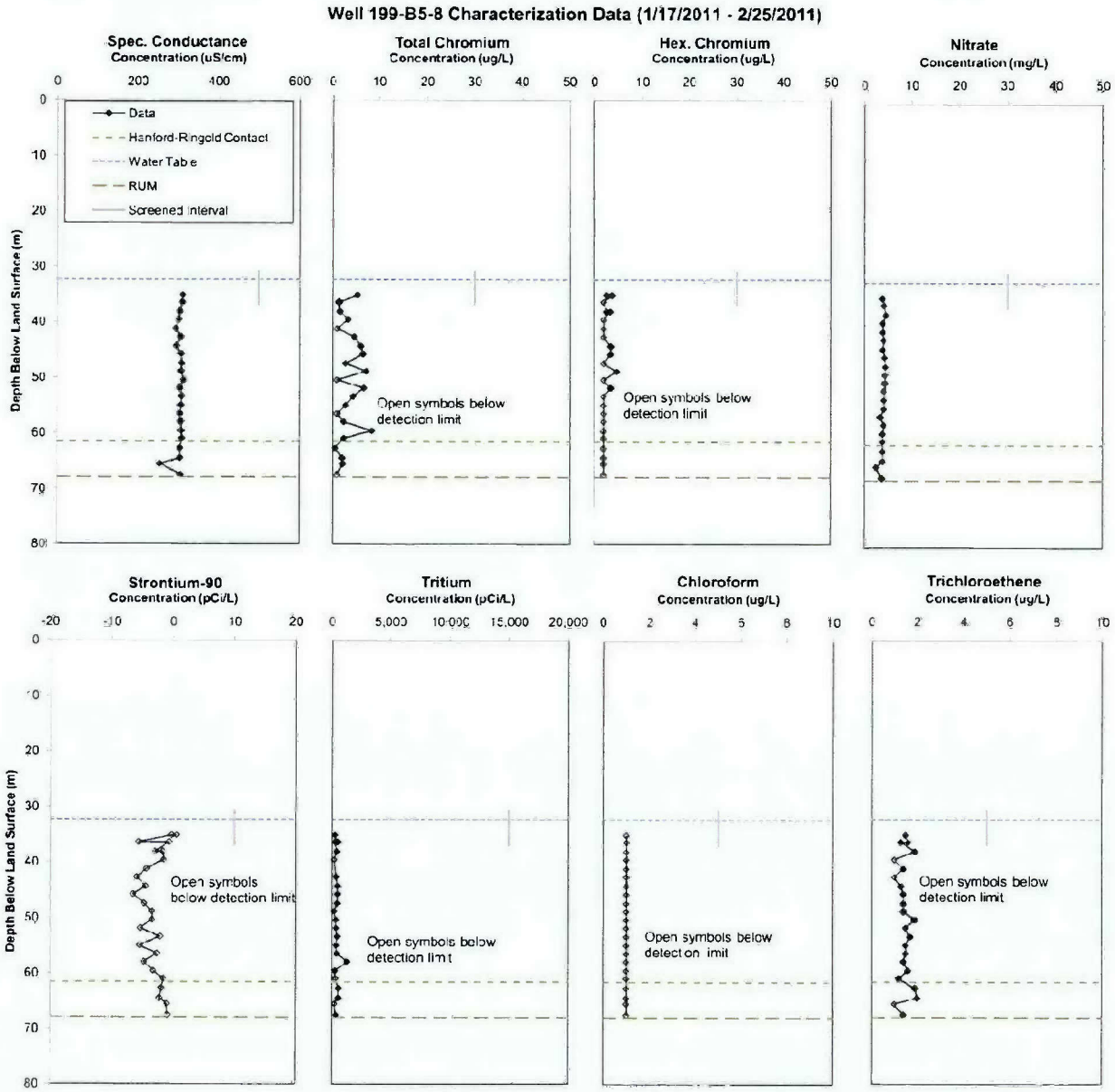


Figure 4-58. Vertical Distribution of Contaminants in Well 199-B5-8

4.3.3.8 Vertical Distribution of Groundwater Contamination in Well 199-B8-9

This well is located in southern 100-BC near C Reactor. Its objective is to define the extent of contamination. It is screened in the upper portion of the unconfined aquifer in the Hanford formation.

Cr(VI) concentrations in Well 199-B8-9 were below 10 µg/L except for a single sample at a depth of 50.3 m (165 ft) bgs, which had a Cr(VI) concentration of 15 µg/L (Figure 4-59). Total chromium ranged from 2 to 19 µg/L. Concentrations were lowest at the bottom of the aquifer.

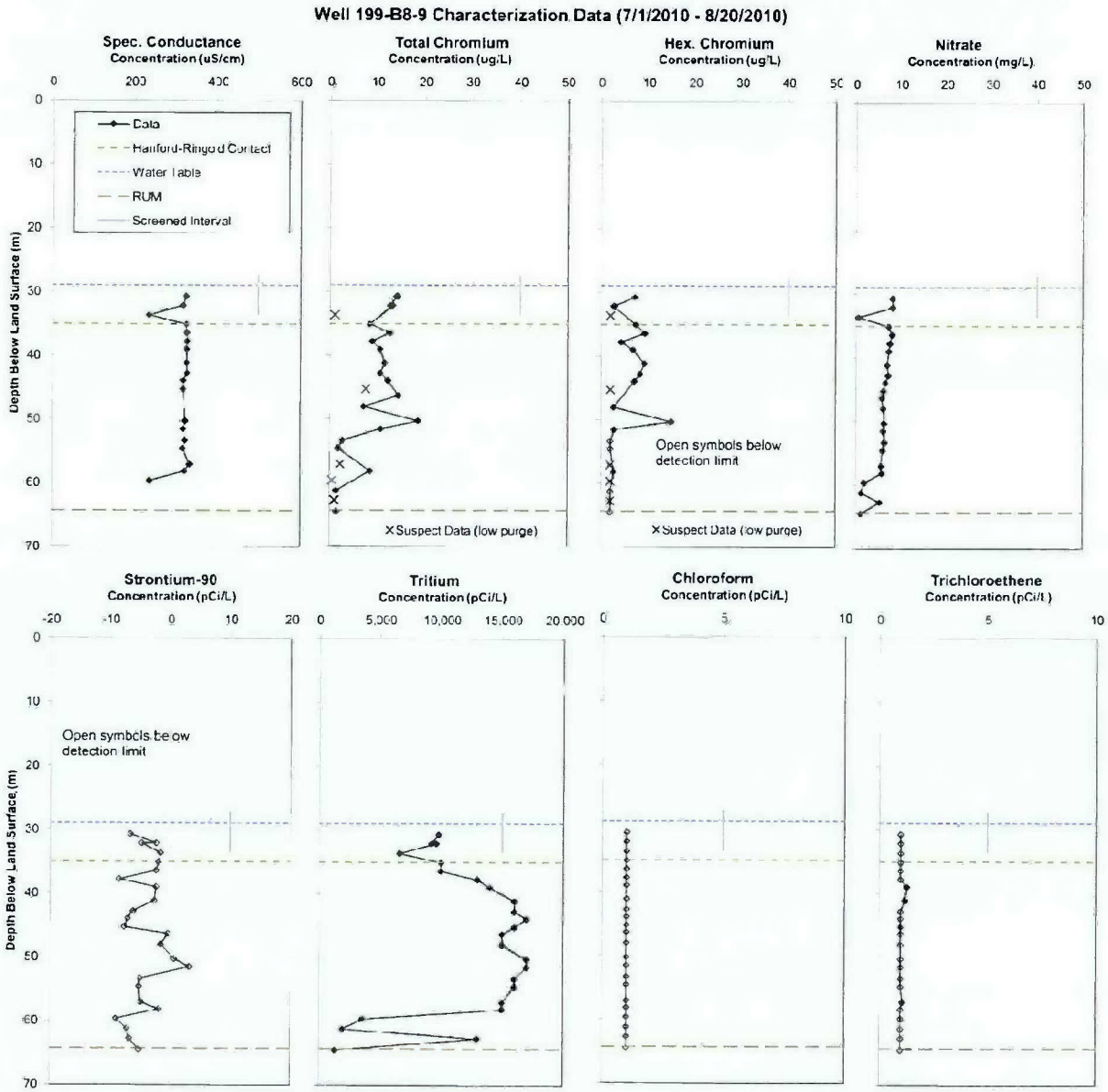


Figure 4-59. Vertical Distribution of Contaminants in Well 199-B8-9

Tritium concentrations all were below the drinking water standard. The highest concentrations (15,000 to 17,000 pCi/L) were in the middle of the aquifer. Concentrations were lower near the top of the aquifer in the Hanford formation. Nitrate concentrations all were far below the drinking water standard and strontium-90 was undetected in all samples. Chloroform was undetected. Trichloroethene was detected in only three samples, at 1.1 to 1.3 μg/L.

4.3.3.9 Vertical Distribution of Groundwater Contamination in Aquifer Tubes

Aquifer tubes in 100-BC have screen depths ranging from 2 to 8 m (6 to 27 ft) bgs (Figure 4-37). The elevations of these screens all fall within the upper half of the unconfined aquifer, which is in Ringold unit E in northern 100-BC. Concentrations of most contaminants (such as, Cr(VI) and tritium) tend to be lowest in the shallowest tubes, which are most affected by mixing with river water. This is evident from lower specific conductance in these tubes. In 100-BC, Cr(VI) concentrations in mid-depth and deep tubes are fairly consistent. Unlike other contaminants, strontium-90 concentrations are higher in shallow or mid-depth tubes than in deep tubes. This reflects its distribution in the aquifer and its lower mobility in the subsurface.

4.3.3.10 Summary of Vertical Distribution of Groundwater Contamination

The vertical distribution of Cr(VI) varies with location in 100-BC. In western 100-BC (Figures 4-60 and 4-61), chromium has an apparently bimodal distribution, with the highest concentrations at the top of the aquifer and in the bottom third of the aquifer. In Well 199-B2-16, Cr(VI) concentrations were low at the top of the aquifer and increased slightly with depth. In eastern 100-BC, Cr(VI) concentrations decreased with depth and were undetectable at the bottom of the unconfined aquifer (Figures 4-60 and 4-62).

Wells 199-B5-5 and 199-B5-6 both showed a thin layer of relatively high Cr(VI) concentrations near the top of the aquifer. This layer appears to be within the Hanford formation (Figures 4-56 and 4-57). The Cr(VI) concentration declined in the upper portion of Ringold unit E, and increased again deeper in the unit. There is no obvious geologic explanation for the distribution within Ringold unit E.

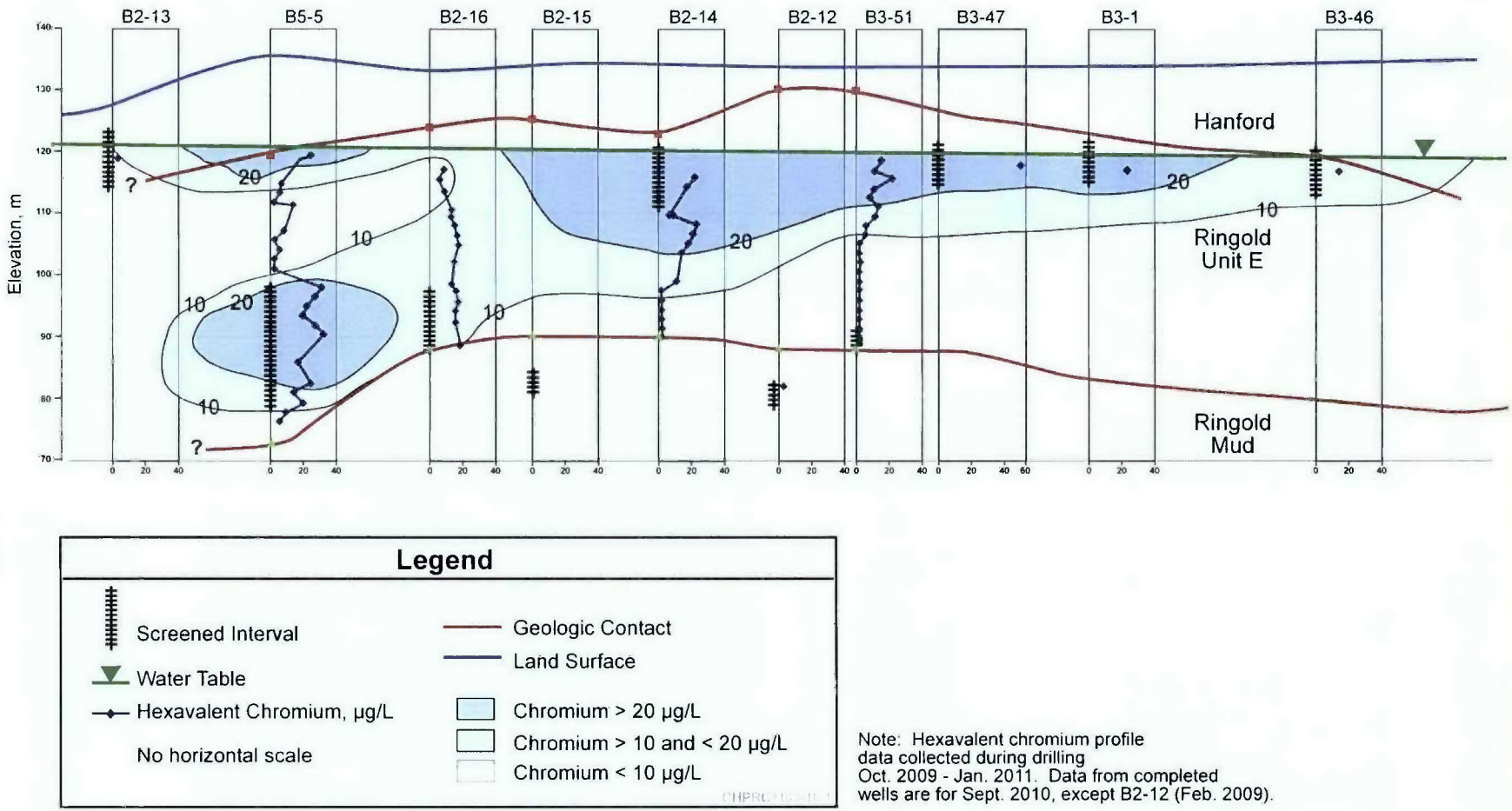
The bimodal vertical distribution of Cr(VI) in western 100-BC may be an expression of different periods of waste releases. Waste released when groundwater mounds were present would have been driven deep into the aquifer. Subsequent recharge with less contaminated water would account for the portion of the aquifer with lower Cr(VI) concentrations. The higher concentrations in the Hanford formation at the top of the aquifer (for example, 199-B4-14 February 2012 result) may represent more recent releases.

Tritium concentrations decreased with depth in most locations. One exception was Well 199-B8-9, near C Reactor, where maximum concentrations at the time of drilling were in the middle of the aquifer. Tritium concentrations in this well are higher in Ringold unit E than in the overlying Hanford formation. However, concentrations increased in the completed well (screened in Hanford formation) in late 2011.

Strontium-90 was detected in very few characterization samples, except in Well 199-B3-51. In that well, its concentration is highest near the top of the aquifer, and was undetectable at 8 m (26 ft) below the water table. This observation is consistent with strontium-90 concentrations in multi-depth aquifer tube clusters in 100-BC, and in monitoring wells elsewhere in the 100 Area.

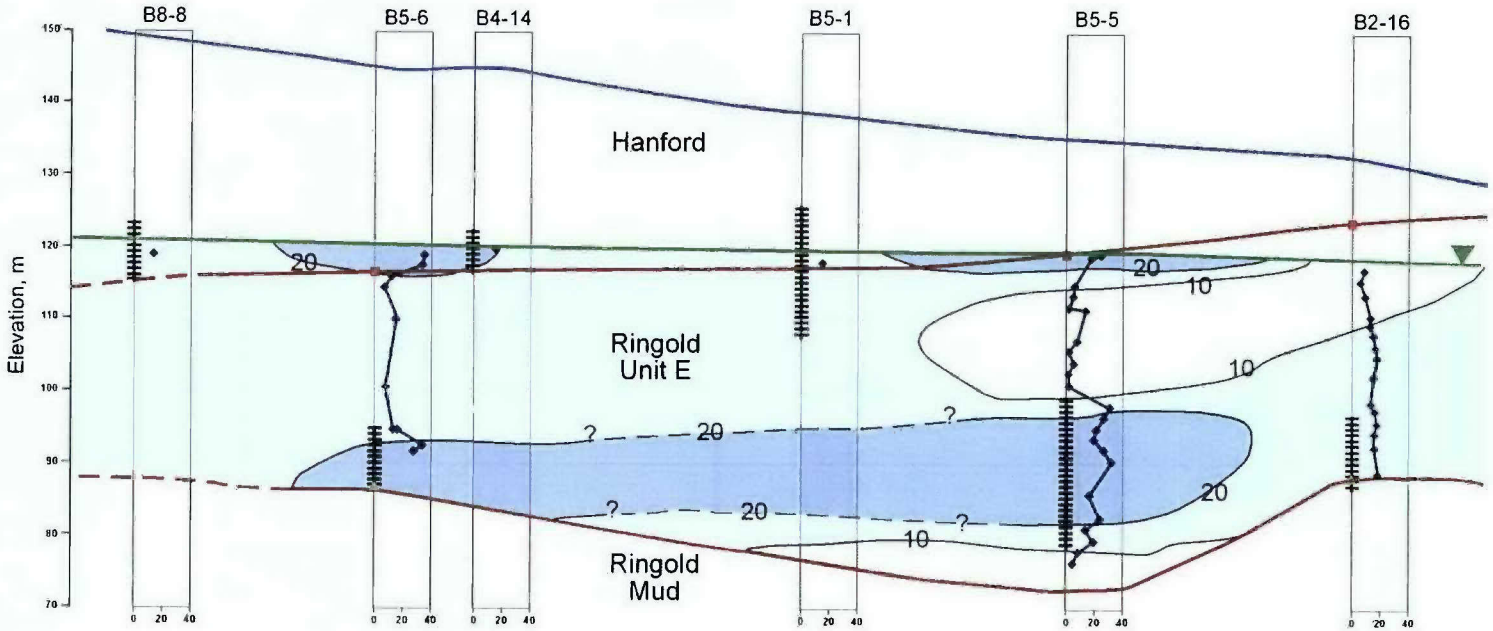
Low levels of chloroform and trichloroethene (<5 μg/L) were detected in some wells, generally in the lower portion of the aquifer. The highest concentrations were in wells in western 100-BC.

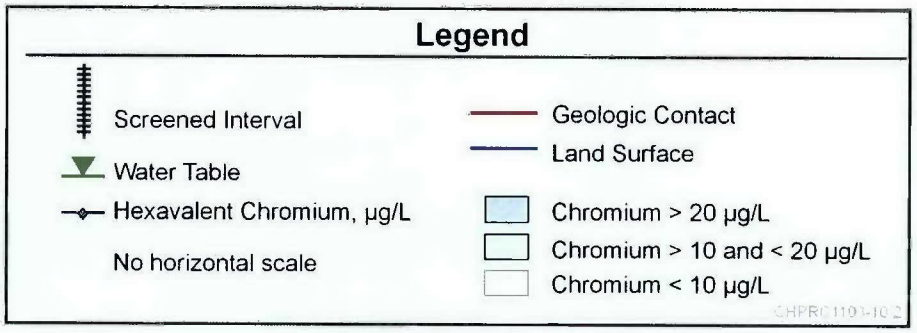
Groundwater in the confined Ringold (that is, beneath the RUM) is uncontaminated.



Note: Hexavalent chromium profile data collected during drilling Oct. 2009 - Jan. 2011. Data from completed wells are for Sept. 2010, except B2-12 (Feb. 2009).

Figure 4-60. Cross Section of 100-BC Chromium Plume, West to East Near River





Note: Hexavalent chromium profile data collected during drilling Oct. 2009 - Jan. 2011. Data from B5-1 from Sept. 2010 and B8-8 from Jan. 2010.

Figure 4-61. Cross Section of Chromium Plume, Western 100-BC, South to North

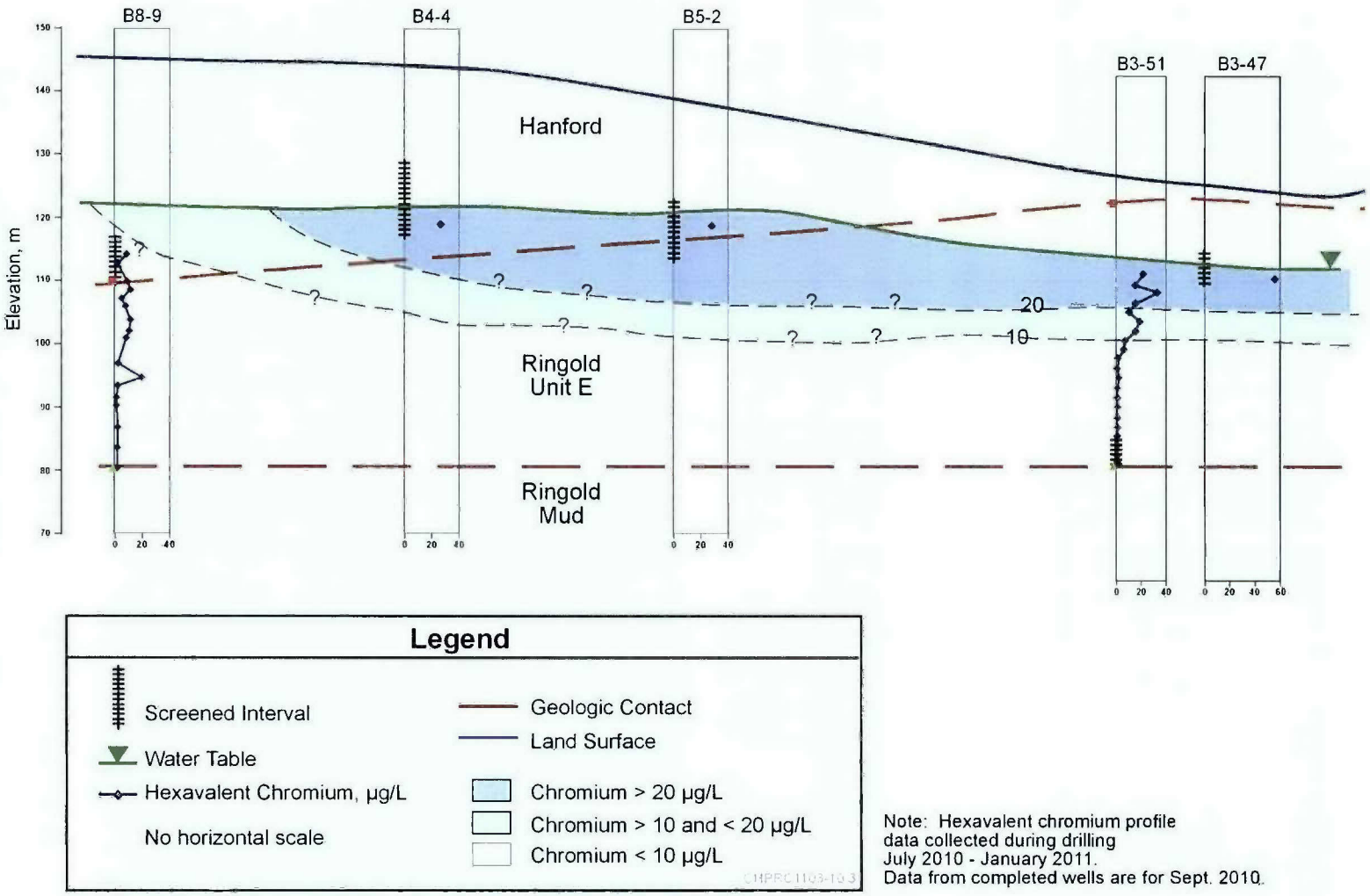


Figure 4-62. Cross Section of Chromium Plume, Eastern 100-BC, South to North

4.3.4 Soil Chemistry in Aquifer and Aquitard

Section 4.2 summarized analytical results for sediment above the water table in groundwater monitoring wells. Samples were also collected at 1.5 m (5 ft) below the water table, at the bottom of the unconfined aquifer, and 1.5 m (5 ft) into the RUM. Additional samples were collected at depths relative to the Hanford/Ringold geologic contact, and some of these fell below the water table. Table 4-31 summarizes soil chemistry data from these samples collected below the water table, compared to Hanford Site soil background (Non-Rad Soil Background document [DOE/RL-92-24]). The background values represent soils in the vadose zone, so they are not directly applicable to soils from below the water table. Background values are used here only for general perspective on concentrations measured in sediment samples from the 100-BC wells.

Table 4-31. Summary of Soil Chemistry Data from New Monitoring Wells, Samples Collected below the Water Table

Well	Constituent[a]	Number Samples	Number above Background	Background (µg/kg or pCi/g)[b]	Max. Conc. (µg/kg or pCi/g)	Depth (m)	Geologic Unit Where Maximum Detected
199-B2-14	Arsenic	3	1	6,470	6,530	44.68	RUM
	Nickel	3	1	19,100	19,500	44.68	
	Selenium	3	2	780	1,780	44.68	
	Zinc	3	1	67,800	85,000	44.68	
199-B2-15	Chromium	11	5	18,500	53,400	51.33	RUM
	Lead	11	1	10,200	10,400	44.90	
	Nickel	11	2	19,100	22,200	44.90	
	Selenium	11	2[c]	780	1,280	44.90	
199-B2-16	None	3	0	—	—	—	Nothing above background
199-B3-50	Chromium	4	1	18,500	60,400	23.74	Hanford formation
	Lead	4	1	10,200	128,000	26.79	
	Zinc	4	1	67,800	71,200	23.74	
199-B3-51	Chromium	6	2	18,500	37,300	47.60	Ringold unit E
	Nickel	6	1	19,100	23,900	14.51	
	Selenium	6	1[c]	780	800	47.82	
	strontium-90	6	3	0.178	0.396± 0.17	14.51	
199-B5-5	Chromium	5	1	18,500	23,900[d]	15.76	Hanford formation
	Selenium	5	1[c]	780	830	62.48	RUM

Table 4-31. Summary of Soil Chemistry Data from New Monitoring Wells, Samples Collected below the Water Table

Well	Constituent[a]	Number Samples	Number above Background	Background (µg/kg or pCi/g)[b]	Max. Conc. (µg/kg or pCi/g)	Depth (m)	Geologic Unit Where Maximum Detected
199-B5-6	Arsenic	4	1	6,470	6,780	58.83	RUM
	Lead	4	1	10,200	12,200	58.83	
	Nickel	4	1	19,100	25,700	23.59	Hanford formation
	Selenium	4	2	780	1,360	23.59	
	Zinc	4	4	67,800	93,800	23.59	
199-B5-8	Chromium	2	2	18,500	31,400	68.28	Bottom of Ringold unit E
	Lead	2	1	10,200	11,600	70.29	RUM
	Manganese	2	1	512,000	910,000	70.29	
	Nickel	2	1	19,100	24,500	70.29	
	Selenium	2	2[c]	780	1,740	70.29	
199-B8-9	Arsenic	12	1	6,470	9,340	66.90	RUM
	Chromium	12	2	18,500	53,100	31.39	Hanford formation
	Selenium	12	2[c]	780	1,660	66.90	RUM

a. Constituents evaluated were antimony, arsenic, beryllium, cadmium, cesium-137, chromium, cobalt-60, copper, europium-152, europium-154, Cr(VI), lead, manganese, nickel, selenium, silver, strontium-90, thallium, vanadium, and zinc (100-BC SAP [DOE/RL-2009-44], Table 2-12). Only those constituents with a result over background (90th percentile; Table 4-1) are listed here.

b. Background concentrations are 90th percentile from the Non-Rad Soil Background document (DOE/RL-92-24), as listed in Table 4-1 of this chapter. Background concentration not available for Cr(VI); undetected except in few samples from Wells 199-B3-51 (maximum = 300 µg/kg) and 199-B8-9 (maximum = 280 µg/L); results near detection limit.

c. Results flagged "B" (near detection limit).

d. Split sample had lower concentration (9,410 µg/kg).

e. Results flagged "C" (associated with blank contamination).

f. Duplicate samples had lower values (6,030 to 16,100 µg/kg).

The only radionuclide exceeding a background concentration was strontium-90 in Well 199-B3-51. The other exceedances were all for metals. Results for individual wells are discussed in the following text.

No soil background has been established for Cr(VI). Concentrations in most samples were below detection limits. Wells 199-B3-51 and 199-B8-9 each had three Cr(VI) detections in the unconfined aquifer. The maximum concentration was 300 µg/kg, which is near the detection limit.

Well 199-B2-14. Three soil samples were collected below the water table: one from the top of the unconfined aquifer and two from the RUM. Arsenic, nickel, and zinc exceeded background in one sample from the RUM. Selenium exceeded background in two samples from the RUM.

Well 199-B2-15. Eleven soil samples were collected below the water table, all from the RUM. Shallower sampling was not required because adjacent Well 199-B2-14 had previously been characterized. Total chromium concentrations in five samples (four depths) exceeded background. Lead, nickel, and selenium concentrations exceeded background in one or two samples.

Well 199-B2-16. Three soil samples were collected below the water table and concentrations were all below background.

Well 199-B3-50. Three soil samples were collected below the water table: one near the top of the unconfined aquifer and two in the RUM. Chromium, lead, and zinc concentrations each exceeded background in one Hanford formation sample.

Well 199-B3-51. Six samples were collected below the water table: three (two depths) near the top of the unconfined aquifer, one at the bottom of the aquifer, and two in the RUM. Chromium, nickel, selenium, and strontium-90 concentrations exceeded background in one or more sample from the top of the aquifer. Strontium-90 was detected in three samples near the top of the aquifer; concentrations were undetectable in the RUM. This was the only monitoring well with detectable strontium-90 in soil below the water table.

Well 199-B5-5. Five samples were collected below the water table: three (two depths) in the unconfined aquifer and two in the RUM. Chromium concentration exceeded background in one of the Hanford formation samples. However, a split sample had a much lower concentration that did not exceed background (Table 4-31). The selenium concentration in one of the RUM samples exceeded background.

Well 199-B5-6. Four samples were collected below the water table: two in the unconfined aquifer and two in the RUM. Concentrations of arsenic and lead exceeded background in one RUM sample. Nickel concentrations exceeded background in one sample from the top of the unconfined aquifer. Selenium concentrations exceeded background in both unconfined aquifer samples. Zinc concentrations exceeded background in all four samples.

Well 199-B5-8. Metals results are available from two soil samples below the water table, both in the RUM. Radionuclides were analyzed in four samples. Chromium, lead, manganese, nickel, and selenium concentrations exceeded background in the RUM.

Well 199-B8-9. Four depths below the water table were sampled: two in the unconfined and two in the RUM. Multiple results are available for some of the metals, for 12 samples. Arsenic and selenium concentrations exceeded background in the RUM. Chromium concentrations exceeded background in one sample from the unconfined aquifer, but results of duplicates were highly variable (Table 4-31). One RUM sample also exceeded background for chromium.

4.4 Columbia River Surface Water and Sediments

Hanford Site surface water and sediment investigations include annual environmental monitoring (2010 Sitewide Environmental Report [PNNL-20548]), and recent remedial investigations (Columbia River RI Report [WCH-380]; Hanford Site Releases Data Summary [WCH-398]). This section summarizes results from those reports, and presents results of additional pore water sampling conducted as part of this 100-BC RI.

In 2004, DOE established a process to compile, classify, and manage environmental data (for example, surface water and sediment) associated with the Columbia River in the *Columbia River Component of the River Corridor Baseline Risk Assessment: Basis and Assumptions on Project Scope* (DOE/RL-2004-49). The Columbia River Component (CRC) database was created as a result of these efforts and was documented in the *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 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 data set. The *Columbia River Component Data Gap Analysis* (WCH-201) presented the results of that analysis and formed the foundation for the sampling plan that was documented in the Columbia River RI Work Plan (DOE/RL-2008-11).

During this time, sampling was underway supporting the *Risk Assessment Work Plan for the 100 Area and 300 Area Component of the RCBRA* (DOE/RL-2004-37). Samples of surface water, groundwater, nearshore sediment, soils, and biota were collected and analyzed to support the RCBRA. Evaluation of that data is summarized in the RCBRA Volume II (DOE/RL-2007-21).

Sampling to fulfill the needs defined in the Columbia River RI Work Plan (DOE/RL-2008-11) took place between October 2008 and June 2010. Surface water, pore water, sediment (shoreline, shallow, cores), island soils, and six species of fish were sampled. The RI field activities associated with the collection of sediment, river water, and island soil in the Columbia River adjacent to and downstream of the Hanford Site and in nearby tributaries are documented in the *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 Soils* (WCH-352). That document describes the sampling locations, identifies samples collected, and describes modifications and additions made to the SAP. Groundwater upwelling field activities and data collection are documented in the Columbia River RI Report (WCH-380).

Groundwater beneath the Hanford Site discharges to the Columbia River via seeps and upwelling to the river bed. This flow path for groundwater provides a means for transporting Hanford Site-associated contaminants in groundwater to the Columbia River. 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.

Sampling locations and methods were described in Chapter 2. Sediment 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 2-3. Additional sediment, island soils, 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).

Further discussion of the sediment, surface water, and pore water samples collected during the CRC RI are discussed in the following sections.

4.4.1 Groundwater Upwelling Investigations at 100-BC

Data were collected near 100-BC 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).

Results of Phase IIa of the study, which mapped locations where groundwater discharged through the riverbed, are summarized in Section 3.6.4.

During Phase IIb of the study, 30 100-BC locations were sampled during August and September 2009 to collect pore water for Cr(VI) analyses. All but three sample locations (J100BC5, T100BC4A, and J100BC2) were sampled during relatively low and stable river levels.

Seventeen of 29 (57 percent) pore water Cr(VI) results were reported above the practical quantitation limit (PQL) of 3.7 µg/L (Table 4-32). Fourteen results exceeded the AWQC (10 µg/L), and results ranged between 15 and 112 µg/L. Four of the five highest Cr(VI) pore water concentrations (range 56 to 112 µg/L) were collected from offshore locations, including the deepest channel of river near 100-BC. The Cr(VI) results generally corresponded to the groundwater plume estimates, but one sample location (J100BC47) was located nearly 0.6 km (0.37 mi) downstream of known groundwater plume and proximal to an active remediation site. This site also contained a significant temperature anomaly value during the Phase IIa mapping effort.

Table 4-32. Concentrations of Chromium in Columbia River Pore Water at 100-BC

Station No.	August–September 2009 Phase IIb		January–February 2010 Phase III				November 2010			
	Cr(VI) (µg/L)		Cr(VI) (µg/L)		Tot. Cr (µg/L)		Cr(VI) (µg/L)		Tot. Cr (µg/L)	
2A-A	24		10		6.7		4.4		11.4	b
J100BC10	5									
J100BC13	3.7	U								
J100BC19	15									
J100BC2	16									
J100BC21	73									
J100BC23	91		3.7	U	3.17		2	U	5.2	
J100BC25	5									
J100BC33	3.7	U								
J100BC42	3.7	U								
J100BC47	28		13		8.8		13.6		14.5	b
J100BC47 SPLIT							10		15.1	b
J100BC5	3.7	U								
J100BC9	15									
T100BC1J1	18		5		2	U	2	U	1	U
T100BC1J5	23	J	3.7	U	1.29	B	2	U	2.21	
T100BC2A	3.7	U								
T100BC2B	27									
T100BC2C	3.7	UJ								
T100BC3C	112	J	22		23.6		2.3		8.99	b
T100BC3C DUP							6.9		15.8	b

Table 4-32. Concentrations of Chromium in Columbia River Pore Water at 100-BC

Station No.	August–September 2009 Phase IIb		January–February 2010 Phase III				November 2010			
	Cr(VI) (µg/L)		Cr(VI) (µg/L)		Tot. Cr (µg/L)		Cr(VI) (µg/L)		Tot. Cr (µg/L)	
T100BC3D	3.7	U								
T100BC3E	3.7	U								
T100BC4A	80		46	a	20	a	12.6		13.7	
T100BC4B	9	J								
T100BC5C	57		3.7	U	11.1		2	U	5.01	b
T100BC6A	3.7	U								
T100BC6J10	26		10		10.2		12.6		14.3	
T100BC6J2	3.7	U								
T100BC6J4	3.7	U								
T100BC6J5	3.7	U								
Upstream1							2	U		
Upstream2							2	U		

a. Note that Cr(VI) result much higher than total chromium, suggesting a possible data quality problem.

b. Unfiltered, total Cr. Other total Cr results are for filtered samples.

B = value is less than contractually required detection limit but above method detection limit

J = estimated value (for example, quality control sample results out of range)

U = undetected

Phase IIb pore water specific conductance values were generally higher than Phase IIa results obtained during January through March 2009, and supported the premise that low- and stable river levels were important factors to consider during the Phase IIb and III sampling events. However, higher specific conductance levels observed at all three mid-river sample locations during Phase IIa compared to Phase IIb may indicate changes in the groundwater upwelling patterns as seasons and hydraulic gradients change.

High pore water specific conductance values indicated the presence of groundwater, but high specific conductance was not linearly consistent with the level of Cr(VI) contaminated groundwater present near 100-BC. For example, a 91 µg/L concentration of Cr(VI) was found where specific conductance was only 160 µS/cm, a specific conductance value just marginally higher than surface water (approximately 140 µS/cm). The Cr(VI) pore water patterns seen at 100-BC suggested a non-uniform pattern of contaminated pore water might exist.

Phase III sample locations were selected as a subset of the previous sample locations for characterization sampling and analysis of pore water, surface water (defined as water 0.3 m [1 ft] above the riverbed), and collocated sediment for a broad range of analyses as defined in the Columbia River RI Work Plan (DOE/RL-2008-11). Sediment samples were attained as close to the pore water sample location as reasonably possible, with a preference given to locations with sediment deposits. Additional discussion of

these results and all other analyses performed can be found in the Hanford Site Releases Data Summary (WCH-398).

Ten sample locations were selected for Phase III sampling near 100-BC, six primary sites and four secondary sites. Pore water and surface water samples were collected at nine of the ten 100-BC sites. One site (J100BC21) was not sampled because a groundwater signal greater than 160 μS/cm could not be found. Pore water specific conductance values ranged between 163 and 344 μS/cm. Site J100BC47 yielded the maximum pore water specific conductance (344 μS/cm) value during Phase III near 100-BC. This site was found to have 28 μg/L of Cr(VI) during Phase IIb and was located just offshore of an active remediation site. Pore water specific conductance dropped more than 10 percent on one sample near 100-BC (T100BC6J10), but the final specific conductance value obtained during Phase III was similar to that obtained during Phase IIb.

Laboratory results for Cr(VI) and total uranium in surface water were all at nondetect levels during Phase III. Only one detection of strontium-90 at 0.6 pCi/L was seen in surface water at Station T100BC3C.

Phase III pore water sample results for Cr(VI) ranged from 5 to 46 μg/L, but the latter value is questionable because the total chromium result was lower. At station T100BC3C where the Cr(VI) result was 112 μg/L during Phase IIb, a detection of only 22 μg/L was observed in Phase III. In general, Phase III chromium results were lower than Phase IIb. Strontium-90 was detected once in pore water at station T100BC4A at a concentration of 6.1 pCi/L. Tritium was detected over a range of 1,400 to 12,100 pCi/L with the maximum found at station T100BC4A.

This study was followed up by additional pore water sampling as part of the RI/FS (100-BC SAP [DOE/RL-2009-44]; *Columbia River Pore Water Sampling in 100-BC Area, November 2010* [SGW-49368]). Figure 4-63 and Table 4-32 list Cr(VI) data from pore water samples collected during Phase IIb (August/September 2009), Phase III (January/February 2010), and RI sampling (November 2010). Concentrations in the RI sampling of November 2010 were the lowest of the three sampling campaigns: up to 13.6 μg/L. It appears that Cr(VI) concentrations in Columbia River pore water declined between fall 2009 and fall 2010. This may represent the passage of a historical groundwater plume that was not fully detected by monitoring wells and aquifer tubes.

Another explanation of the apparent decline in Cr(VI) concentrations is that an increase in river stage suppressed groundwater upwelling and caused a decrease in specific conductance and Cr(VI) concentrations. Staff evaluated river stage, specific conductance, and Cr(VI) data to determine possible relationships. In some cases, an increase in river stage or decrease in specific conductance was accompanied by a decrease in Cr(VI). However, this relationship was determined to be statistically insignificant. *Columbia River Pore Water Sampling in 100-BC Area, November 2010* (SGW-49368) describes the November 2010 sampling task and presents the results of statistical evaluations.

The average river stage during the November 2010 sampling event (119.5 m) was slightly higher than during Phase IIb (119.2 m) and river stage was more variable in November 2010 (Figure 4-64). This could have decreased the ratio of groundwater to river water in the pores. However, this would have been evident from the specific conductance of the samples. As explained previously, the change in conductance was not statistically significant.

Co-contaminants were analyzed only in the Hanford Site releases study, Phase III. Strontium-90 was detected in only one pore water sample (6.1 pCi/L). Tritium was detected over a range of 1,400 to 12,100 pCi/L.

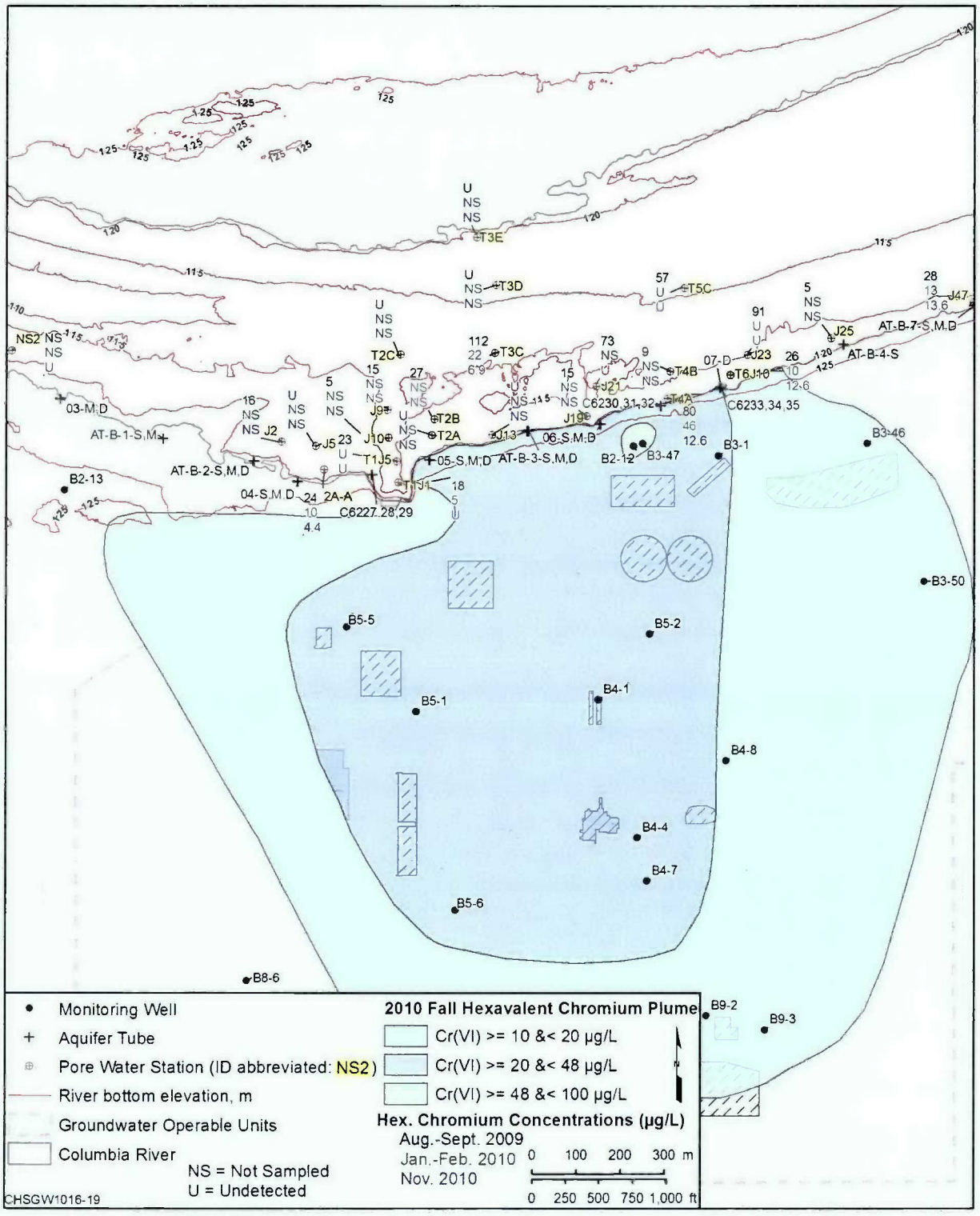


Figure 4-63. Pore Water Cr(VI) Concentrations at 100-BC

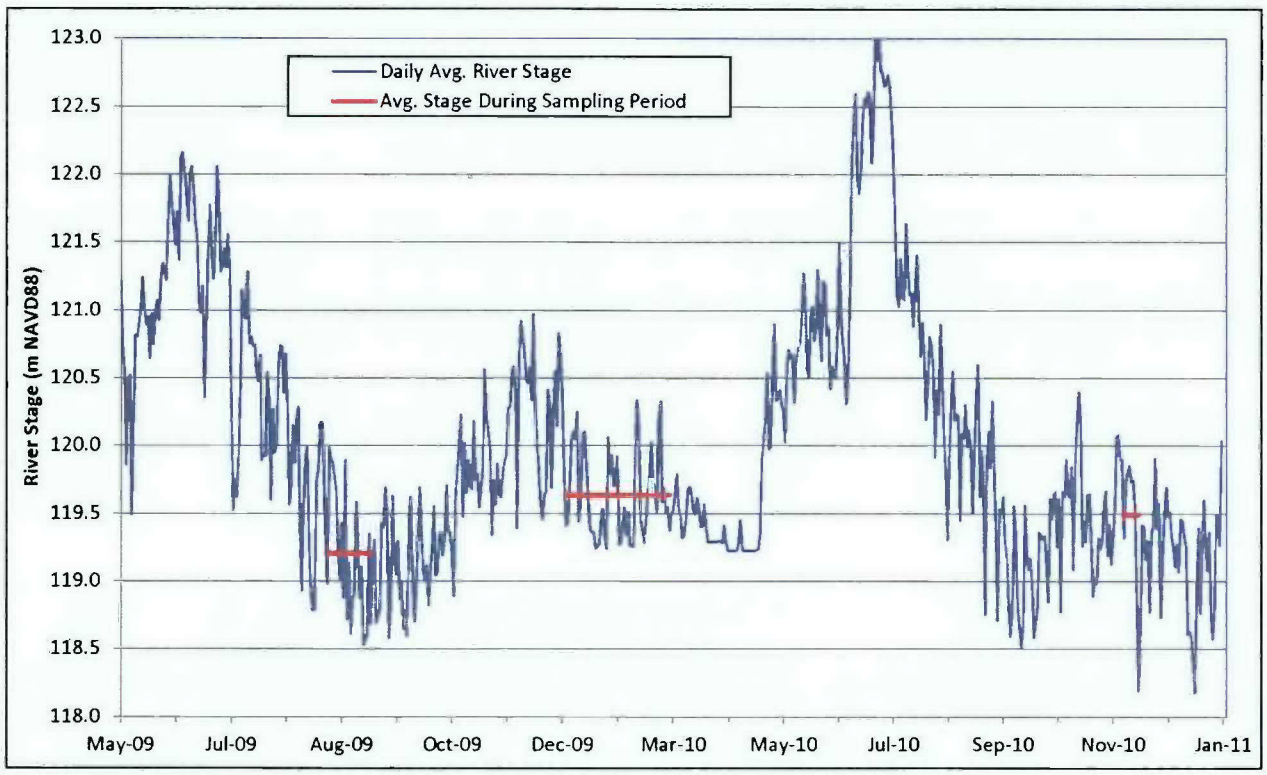


Figure 4-64. Daily Average River Stage during 100-BC Pore Water Sampling

4.4.2 Sitewide Environmental Monitoring of Surface Water and Sediments

DOE conducts environmental monitoring on the Hanford Site, including surface water and associated sediment. The following information for 2010 is summarized from 2010 Sitewide Environmental Report (PNNL-20548).

Samples of surface water and sediment are collected upstream of the Hanford Site at Priest Rapids Dam, downstream of the site at the City of Richland, and at several locations on the site, but not at 100-BC. Constituents of interest in Columbia River water samples collected at Priest Rapids Dam and the City of Richland include gamma-emitting radionuclides, tritium, strontium-90, technetium-99, uranium isotopes, and plutonium isotopes.

4.4.2.1 Columbia River Water Monitoring

Radionuclide concentrations monitored in Columbia River water were low throughout 2010 and similar to previous years. Statistical analyses were performed to determine if concentrations downstream of the Hanford Site were greater than those upstream of the site.

Statistical analyses indicated that downstream tritium concentrations were higher than upstream concentrations. The average tritium concentration in Columbia River water collected at the City of Richland (39 ± 5.2 pCi/L) was higher than at Priest Rapids Dam (25 ± 8.9 pCi/L), but was only 0.2 percent of the Washington State ambient surface water quality criterion of 20,000 pCi/L. Average total uranium concentration measured at the City of Richland (0.66 ±0.099 pCi/L) was higher than at Priest Rapids Dam (0.43 ±0.078 pCi/L). The primary source of uranium discharging to the Columbia River at the Hanford Site is the 300 Area, a short distance upstream from the City of Richland. Statistical comparisons for plutonium, strontium-90, gross alpha, and gross beta concentrations at Priest Rapids Dam and the City of Richland were not performed because most of the concentrations were less than detection limits.

All metal and anion concentrations in river water, including dissolved chromium, were less than the AWQC in 2010.

The U.S. Geological Survey sampled Columbia River water at Vernita Bridge (upstream of the Hanford Site) and the City of Richland in 2010. These data provided no indication of any deterioration of water quality along the Hanford Reach of the Columbia River. Median concentrations of dissolved chromium were similar for water samples collected near Vernita Bridge and the City of Richland and were well below the ambient water quality criterion.

4.4.2.2 Columbia River Sediment Monitoring

Some constituents in liquid effluents previously discharged to the Columbia River on the Hanford Site may have become associated with particles that accumulated in riverbed sediment, particularly in slackwater areas and in the reservoirs upstream of dams. Fluctuations in the river flow may redistribute these contaminated sediments. Upper-layer sediment in the Columbia River downstream of the Hanford Site contains low concentrations of radionuclides and metals of Hanford origin, as well as contaminants from mining, agriculture, and atmospheric fallout.

Sediment samples were collected upstream of the Hanford Site at the Priest Rapids Dam reservoir and downstream of the site at McNary Dam. Sediment samples were also collected within the Hanford Site from slackwater areas where fine-grained material is known to deposit (for example, the White Bluffs, 100-F, and Hanford Sloughs), and from the publicly accessible City of Richland shoreline. Radionuclides consistently detected in river sediment adjacent to and downstream of the Hanford Site during 2010 included beryllium-7, potassium-40, cesium-137, uranium isotopes, and plutonium isotopes. The concentrations of all other radionuclides, including strontium-90, were below detection limits for most samples. Cesium-137 and plutonium isotopes exist in worldwide fallout as well as in effluent from Hanford Site facilities. Beryllium-7, potassium-40 and uranium isotopes occur naturally in the environment, and uranium isotopes are also present in Hanford Site effluent. There were no obvious differences between locations.

Detectable amounts of most metals were found in all river sediment samples. Maximum and average concentrations of most metals were higher for sediment collected in the reservoir upstream of Priest Rapids Dam than in sediment from either the Hanford Reach or McNary Dam. The concentrations of cadmium, lead, nickel, and zinc differed the most between locations and may be associated with upstream mining activity.

4.4.2.3 Columbia River Shoreline Spring Monitoring

Shoreline springs represent areas where groundwater discharges to the Columbia River. DOE routinely samples springs and associated sediments along the Hanford Reach. This section summarizes results of monitoring springs at 100-BC.

Table 4-33 lists concentration maxima for radionuclides, metals, and nitrate in 100-BC spring water over the period 2004 through 2010. Figure 4-65 shows chromium concentrations at a 100-BC spring (located near aquifer tube site 04) since 1996. Concentrations typically range from 10 to 20 μg/L.

Concentrations of radionuclides in sediment associated with a 100-BC shoreline spring are very low. Concentrations of cobalt-60, strontium-90, europium-152, and europium-154 have all been below detection limits from 2004 through 2009. Cesium-137 was detected at a maximum concentration of 0.055 ±0.012 pCi/g.

Table 4-33. Sitewide Environmental Monitoring Results for Shoreline Springs in 100-BC

Constituent, Units	No. Samples	Concentration Maximum, 2004-2010
Radionuclides		
Gross alpha, pCi/L	10	14 ± 5.6
Gross beta, pCi/L	10	23 ± 5.1
Strontium-90, pCi/L	10	2.8 ± 0.42
Technetium-99, pCi/L	10	7.8 ± 1.0
Tritium, pCi/L	10	2,800 ± 180
Dissolved Metals		
Antimony, µg/L	12	0.25
Arsenic, µg/L	12	1.3
Cadmium, µg/L	12	0.029
Chromium[a], µg/L	12	18
Copper, µg/L	12	1.6
Lead, µg/L	12	1.4
Nickel, µg/L	12	2.1
Silver, µg/L	12	0.0050
Thallium, µg/L	12	0.024
Zinc, µg/L	12	17
Anions		
Nitrate (as NO_3), mg/L	10	9.9[b]

Source:, *Hanford Site Environmental Report for Calendar Year 2010* (PNNL-20548), Tables 8.5.3 and C.10

a. Dissolved chromium is considered to represent Cr(VI).

b. Converted from nitrate as N.

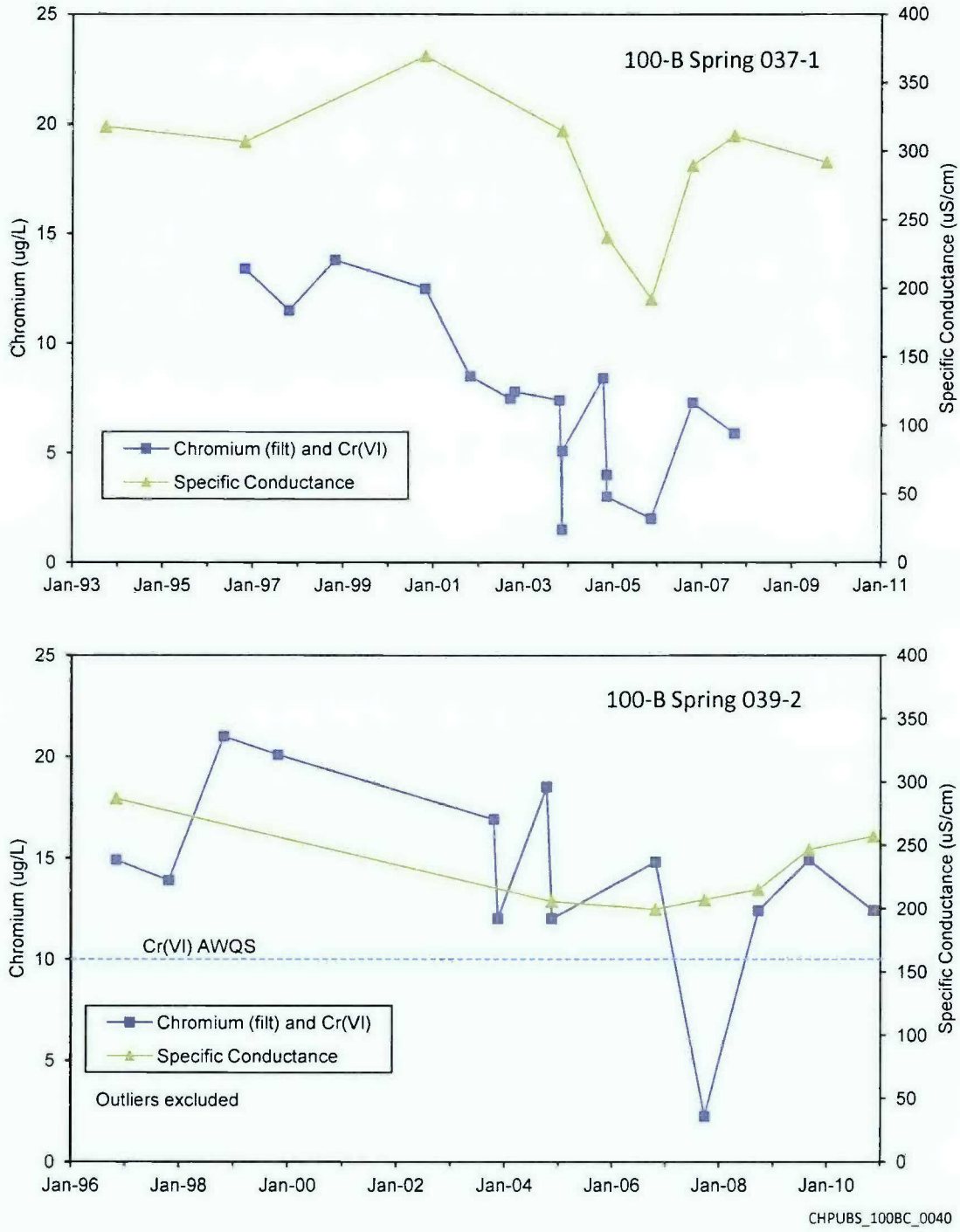


Figure 4-65. Dissolved Chromium Concentrations in 100-BC Shoreline Springs

4.5 Biota

This section summarizes ecological sampling or biological monitoring data that have been collected for 100-BC. Biota data are useful to understand biological receptors, which are evaluated in Chapter 7.

Biota data from three main environmental sampling projects conducted at Hanford were reviewed and summarized for this section. The Surface Environmental Surveillance Project (SESP) is a multimedia environmental surveillance project. 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 assess environmental impacts. Project personnel annually collect selected samples of ambient air, surface water, agricultural products, fish, wildlife, and sediments. Soil and vegetation samples are collected approximately every five years. Analytical capabilities include the measurement of radionuclides at environmental concentrations, and in selected media, nonradiological constituents including metals, anions, volatile organic compounds, and total organic carbon. The SESP sampling design is described in the Hanford Site *Environmental Monitoring Plan United States Department of Energy Richland Operations Office* (DOE/RL-91-50).

Some of the biota data are summarized from ecological samples collected to support 100-BC Ecological Risk Assessment (BC Pilot). The BC Pilot project was a small-scale ecological risk assessment conducted in 100-BC. The primary goal of the BC Pilot ecological risk assessment was to evaluate current and potential future risks to the environment posed by releases of hazardous substances and to serve as a test project for a larger, River Corridor risk assessment. The BC Pilot project appraised relevant sources of contamination, exposure pathways, and contaminants for several environmental media and receptors including surface soil, vegetation, soil invertebrates, and small mammals. Analytical capabilities include the measurement of radionuclides, metals, anions, semivolatile organic compounds, herbicides, and pesticides at environmental concentrations as well as physical properties (such as, pH, moisture, and particle size) in selected media. The BC Pilot sampling and analytical specifications are documented in the *100-B/C Area Ecological Risk Assessment Sampling and Analysis Plan* (DOE/RL-2003-08).

Finally, biota data are summarized from ecological samples collected to support the RCBRA. The primary goal of the RCBRA ecological risk assessment is to evaluate current and potential future risks to the environment posed by releases of hazardous substances. The 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. Analytical capabilities include the measurement of radionuclides, metals, anions, semivolatile organic compounds, herbicides, and pesticides at environmental concentrations as well as physical properties (such as, pH, moisture, and particle size) in selected media. The RCBRA sampling and analytical specifications are documented in the RCBRA SAP (DOE/RL-2005-42).

The location of the SESP, BC Pilot, and RCBRA biota samples summarized in this section are shown in Figure 4-66. The river level is shown at high water stage. Samples collected at lower river stages may appear to have been collected in the river, when in fact, the area would have been exposed at the time. The collection dates for the samples collected as a part of the SESP ranged from 1994 to 2008. The collection dates for the samples collected as a part of the BC Pilot Project ranged from 2002 to 2004. The majority of the RCBRA environmental samples were collected in 2006 and 2007. The various terrestrial species of plants and animals collected and the tissues analyzed are summarized below.

- Perennial vegetation:
  - Unique local plants: mulberry leaves and shoots
  - Dominant shrub: current year's growth, stems and leaves (combined)
  - Dominant grass: current year's growth, stems and leaves (combined)
  - Balsam root: leaves, roots
- Terrestrial invertebrate: whole-body composites
- Mouse: whole-body composites, kidney and liver (combined)

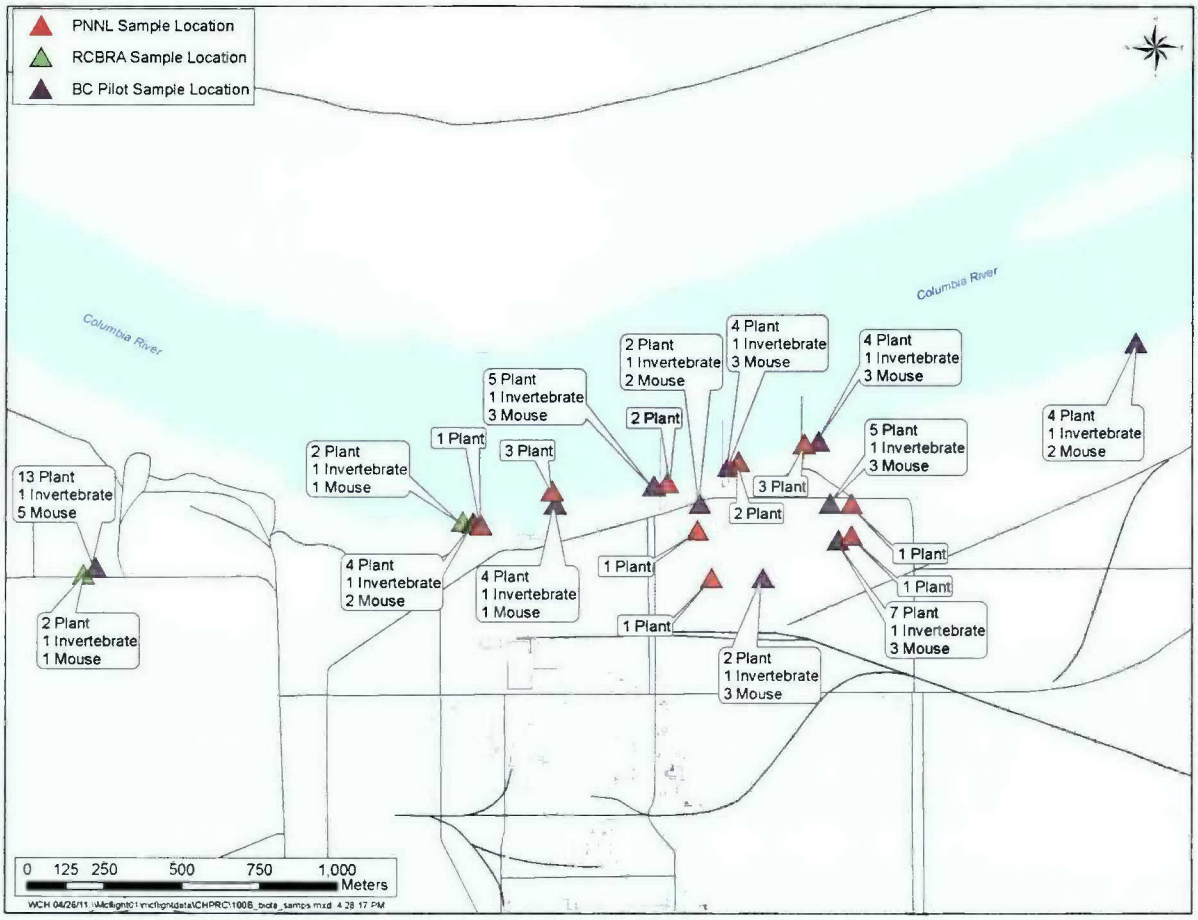


Figure 4-66. Terrestrial Upland Biota Sampling Locations in 100-BC

Although 100-BC is adjacent to the river, only samples of terrestrial plants and animals are summarized for this discussion. Water, sediment, and aquatic organisms from the Columbia River areas adjacent to 100-BC are addressed in the Columbia River RI Work Plan (DOE/RL-2008-11).

Appendix H, Attachment H-1 shows a summary of plant tissue samples collected for the SESP, BC Pilot, and RCBRA projects in 100-BC. Samples of leaves, stems, and roots from several varieties of plants were analyzed for metals and radionuclides. Samples collected for the RCBRA project were also analyzed for organic compounds. The results for the organics and radionuclides are below or very near the detection levels. Although metals were detected in many of the samples, no unusual trends were observed for the analytes detected.

Appendix H, Attachment H-1 shows a summary of invertebrate tissue samples collected for the BC Pilot and RCBRA projects in 100-BC. The samples were analyzed for metals. Samples collected for the RCBRA project were also analyzed for selected radionuclides. Results for the radionuclides are below or very near the detection levels. Although metals were detected in many of the samples, no unusual trends were observed for the analytes detected.

Appendix H, Attachment H-1 shows a summary of mouse muscle and kidney tissue samples collected for the BC Pilot and RCBRA projects in 100-BC. The samples were analyzed for metals and radionuclides. Samples collected for the RCBRA project were also analyzed for organic compounds. The results for the organics and radionuclides are below or very near the detection levels. Although metals were detected in many of the samples, no unusual trends were observed for the analytes detected.

4.6 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 the air are monitored onsite near facilities and operations, at site-wide locations away from facilities, and offsite around the Hanford Site perimeter as well as in nearby and distant communities. As discussed in Section 2.1.8, 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 Site locations away from facilities, around the perimeter of the Site, and offsite in nearby and distant communities. Remediation projects under CERCLA may also establish additional monitoring stations specific to their needs. 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. No point source releases are currently associated with the 100-BC. 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. Additional description of the ambient air sampling activities is available in the 2010 Sitewide Environmental Report (PNNL-20548).

4.6.1 Site-Wide and Offsite Ambient-Air Monitoring

Airborne radioactivity across Hanford is sampled by a network of continuously operating ambient air samplers. The samplers were monitored biweekly for gross alpha and gross beta, and composite samples were collected for gamma energy analysis, strontium-90, plutonium-238, plutonium-239/240, and uranium-234, uranium-235, and uranium-238.

Overall, the effective dose equivalent (EDE) to the Hanford Site maximally exposed individual (MEI) resulting from routine and nonroutine emissions in 2010 from Hanford Site point sources was 0.053 mrem. The EDE from fugitive emissions at the Hanford Site in 2010 was 0.0081 mrem. The contribution from radon emissions in 2010 was 0.014 mrem. The total radiological dose for 2010 to the MEI from all Hanford Site radionuclide emissions, including radon, was 0.075 mrem, or 0.75 percent of the federal and state standard of 10 mrem/yr, to which the Hanford Site was in compliance (*Radionuclide Air Emissions Report for the Hanford Site, Calendar Year 2010* [DOE/RL-2011-12]).

4.6.2 Ambient Air Monitoring Near Facilities and Operations

At 100-BC, sources of air emissions include environmental restoration activities. These activities were conducted at 100-BC in recent years, with contaminated soil and debris from inactive waste sites, such as 100-C-7, being excavated, transported, and disposed of at ERDF and other appropriate locations. Air emissions at 100-BC are limited to nonpoint sources such as fugitive dust.

As discussed in Section 2.1.8, near facility air sampling monitors measure the effectiveness of waste management, environmental remediation controls, and effluent treatment systems in reducing effluents and emissions. These air samplers also monitor diffuse source emissions. Air radioactivity was sampled by a network of continuously operating samplers (N466, N496, and N497) at three locations in 100-BC in 2008. Sampling locations are shown in Figure 2-2, Potential Air Sampling Locations at 100-BC. Samples have not been collected since 2008.

The Hanford Site Air Operating Permit (Federal Facility License FF-01) requires regulatory notification for composite (isotopic) air sample results that exceed 10 percent of EPA Table 2 ("National Emission Standards for Hazardous Air Pollutants," [40 CFR 61] Appendix E, "Compliance Procedures Methods for Determining Compliance with Subpart I," Table 2) values. At the three 100-BC stations monitored in

2008, uranium-234 (N466, N496, and N497), uranium-238 (N466), and plutonium-239/240 (N497) were detected, but none of the air results exceeded 10 percent of EPA Table 2, so Washington State Department of Health (WDOH) notifications were not required.

4.7 Summary of Nature and Extent of Contamination

This section summarizes the main components of a conceptual site model describing the nature and extent of contamination in 100-BC. Data collected for the RI helped delimit contaminant concentrations in the vadose zone, groundwater, and pore water of the Columbia River.

4.7.1 Operational Period

Liquid and solid waste discharged during the reactor operational period represented the primary contaminant sources. These included cooling water conditioning and handling facilities, underground piping, liquid waste and solid waste storage and disposal sites, and unplanned releases (surface spills). Vadose zone soil and aquifer sediments affected by releases represent potential secondary contaminant sources. Secondary sources can release contaminants to the environment long after discharges from primary contaminant sources have stopped.

In the vadose zone, the downward movement of infiltrating water was perpendicular to the orientation of the sedimentary layers and lenses in the Hanford formation and Ringold Formation unit E. Transient saturated conditions were present during reactor operations throughout the vadose zone beneath high volume liquid effluent waste sites. Pressure head and vertical saturated hydraulic conductivity would have been the dominant factors controlling the downward movement of water and contaminants during this period.

Low-mobility contaminants, including many metals and radionuclides, sorbed to sediment grains in the vadose zone. These contaminants were found at the greatest concentrations within and near the areas of discharge. When little or no liquid effluent was discharged to a waste site, soil contamination remained in the shallow sediment. Most of this shallow contamination has been removed during remediation activities. Sufficiently high volumes of liquid discharged into a waste site modestly expanded the depth of low-mobility contamination in the vadose zone.

Strontium-90 is a slightly mobile contaminant in the subsurface and was present in several 100-BC waste sites. The largest estimated inventories were at the 118-B-1 and 118-C-1 Solid Waste Burial Grounds, but this contamination was less mobile than that carried in liquid effluent. Strontium-90 was also disposed at liquid waste sites, principally from decontamination solutions and, particularly, contaminated reactor coolant or fuel storage basin liquid. This contaminant migrated vertically and horizontally in groundwater but is not as widespread as the more mobile contaminants. During the operational period, some of it sorbed to sediment grains in the vadose zone and aquifer.

Mobile or moderately mobile contaminants include tritium and Cr(VI). During the period of reactor operations, large volumes of water containing Cr(VI) and radionuclides were discharged to the soil via trenches, cribs, and leaks from pipelines and retention basins, primarily in eastern 100-BC. Liquid effluent also was released through outfall piping to the Columbia River. During the operational period, large groundwater mounds helped spread these contaminants in groundwater in a radial pattern.

Some contaminants were released in relatively low volumes, but at higher concentrations. Examples of this type of waste distribution are the 100-B-27 and 100-C-7 sites in western 100-BC where Cr(VI) remained in vadose zone soils, rather than being washed through by high volumes of water.

4.7.2 Post-Operations, Pre-Interim Remediation

After reactor operations and liquid effluent disposal ceased, the driving force for infiltration decreased. For an undetermined period, water in the vadose zone continued to drain beneath the sites, but the volume was minor compared to the operational period. Infiltration of precipitation through contaminated vadose zone sediments carried some additional contamination to groundwater. Net infiltration of 17 to 52 mm/yr (0.67 to 2.0 in./yr) likely continued at nonvegetated waste sites. During this period, short-lived radionuclides continued to decay. The groundwater mounds dissipated and groundwater began flowing northward toward the Columbia River.

4.7.3 Post-Interim Remediation

DOE began interim remediation of 100-BC waste sites in the mid-1990s. Remediation generally included excavation to about 4.6 m (15 ft) bgs, which removed the most heavily contaminated sediments. Dust suppression water was used during demolition and remediation of waste sites, which may have temporarily mobilized some contaminants in the vadose zone. Once waste sites are revegetated, the plants consume the natural precipitation, limiting infiltration deep into the vadose zone. Some contaminants may remain dissolved in the unsaturated water content of the vadose zone, but the mass in this phase is likely to be low, given the relatively low moisture content of the vadose zone (Chapter 3).

The 100-C-7 and 100-C-7:1 waste sites are currently undergoing remediation. Characterization and excavation continue to reveal Cr(VI) throughout the thick vadose zone at these sites. Sharp increases in Cr(VI) concentrations in wells north and east of the site in 2011 and 2012 indicate mobilization of chromium from the waste site into groundwater.

Remedial investigation sampling of the vadose zone identified no substantive quantities of residual contaminant mass in the vadose zone. Notable detections in the vadose zone included the following.

- Low concentrations of strontium-90 were detected through the vadose zone to groundwater beneath several waste sites.
- Low concentrations of tritium were detected through the vadose to groundwater beneath the 118-B-6 Burial Ground. Tritium concentrations measured during the deep zone closeout sampling were elevated.
- Cr(VI) concentrations beneath remediated waste sites were generally low. The most elevated result was 1.05 mg/kg beneath the 118-B-6 burial ground. Low level detections of Cr(VI) in the rewetted zone and within the water table were present at 100-BC groundwater characterization wells.

Based on these multiple lines of evidence, waste sites that have undergone interim remedial action are not a significant, ongoing source of contaminants to groundwater.

Contaminants of potential concern in 100-BC groundwater include Cr(VI), strontium-90, and tritium. Data from wells installed for the RI helped define the extent of these contaminants horizontally and vertically. Key elements of the nature and extent of groundwater contamination include the following.

- Cr(VI) exceeds the 10 µg/L AWQC beneath much of 100-BC, with an estimated plume area of 2.2 km^2 (0.85 mi^2). Three wells had concentrations above the MTCA "Groundwater Cleanup Standards" (WAC 173-340-720) groundwater cleanup level of 48 µg/L at least once in 2010 through early 2012.
- Data from a new groundwater well in northwestern 100-BC show that a portion of the Cr(VI) plume extends farther west than previously known.

- Cr(VI) concentrations in groundwater decrease with depth in eastern 100-BC. In western 100-BC, concentrations are highest at the top and bottom of the aquifer, and lower in between.
- Strontium-90 exceeds the 8 pCi/L drinking water standard in a plume beneath northeastern 100-BC with an area of 0.38 km^2 (0.15 mi^2). Concentrations decline with depth in the aquifer.
- Tritium concentrations exceed the 20,000 pCi/L drinking water standard in a narrow plume in northern 100-BC and a larger, less-defined plume in southern 100-BC. Total area of the plumes is estimated to be 0.16 km^2 (0.06 mi^2). Concentrations generally decline with depth.
- Groundwater contains trichloroethene at concentrations above action levels, but below drinking water standards. trichloroethene is detected in wells in the 600 Area, as well as in 100-BC.
- Nitrate concentrations are below the 45 mg/L drinking water standard (expressed as NO_3), but are above background.
- Trends in Cr(VI) and tritium concentrations indicate the plumes are migrating from the south part of 100-BC to the northeast.
- Groundwater contamination is limited to the unconfined aquifer. Wells screened in a confined aquifer of the Ringold Formation are uncontaminated.

Groundwater continues to flow towards the north-northeast, discharging to the Columbia River. The groundwater contaminant plumes persist despite the length of time since waste discharges ceased, and despite removal of contaminated sediment from the waste sites. Factors that contribute to the persistence of the plumes include the following.

- The horizontal hydraulic gradient is low in southern 100-BC.
- Low hydraulic conductivity of Ringold unit E compared to Hanford formation—The top of the aquifer is in Ringold unit E near the river, which slows flow.
- Recent presence of secondary sources—Interim remediation of some waste sites has only recently been completed (for example, 100-B-27) or is still going on (100-C-7). Data collected during excavation of 100-C-7 indicate that vadose zone contamination from this site has recently leached Cr(VI) to groundwater.
- Contaminant sorption in the aquifer—Strontium-90 is known to sorb to sediment grains in the vadose zone and aquifer. This behavior also has been observed for a relatively small percentage of Cr(VI).

As groundwater approaches the Columbia River, it enters a zone of interaction with river water. Flow paths in the zone of interaction vary with daily and seasonal fluctuations in river stage. River water infiltrates the banks when river stage is high, moves inland, and then reverses flow as the river stage subsides. Modeling suggests that there is a significant back-and-forth motion in the groundwater, effectively lengthening the flow path through the aquifer. A water molecule will experience numerous reversals in flow direction before it eventually reaches the water column in the river. This mechanism delays contamination from reaching the river and provides additional opportunity for contaminant attenuation by chemical reaction or radioactive decay.

The water in the zone of interaction is a mixture of groundwater and river water, as evidenced by lower specific conductance in shallow aquifer tubes and, seasonally, in near-river monitoring wells. Mixing continues in the pore water of shallow river sediments. Recent studies show that specific conductance rapidly declines as the river rises.

Groundwater from the unconfined aquifer eventually discharges to the Columbia River near the 100-BC shoreline and farther out in the channel. Groundwater flow under the river channel is prevented by higher hydraulic heads on the north side of the river. Columbia River pore water was sampled during periods of low river flow three times at 100-BC. Cr(VI) concentrations were far above the aquatic standard in some locations in 2009. Concentrations declined in two subsequent sampling campaigns, but remained slightly above the standard at some locations. Co-contaminants were below applicable standards in pore water. All contaminants were below standards in river water.

4.7.4 Human and Ecological Receptors/Exposure Pathways

There are currently no unacceptable human health risks from contaminants in the 100-BC groundwater, primarily because exposure is precluded by DOE site controls. Groundwater is a potential, albeit not planned, drinking water source at 100-BC, and it discharges into the Columbia River, which is used for drinking water.

A qualitative ecological risk assessment concluded that the Cr(VI) concentrations in groundwater discharging to surface water cause exceedances of the AWQC of 10 μg/L for protection of freshwater aquatic life. However, there are no data to support that there is significant affect on receptors. Of particular concern is the potential for groundwater to enter pore water in the gravel river bottom habitat used by salmon eggs, alevins, and fry.

Looking Forward in this Document

Chapter 4 described the contaminants resulting from 100-BC operations and their current extent in the environment. Contaminants can be harmful to human health and the environment if there is contact with sufficient concentrations, mass, or radioactivity.

Chapter 5 describes and predicts fate and transport, that is, how these contaminants will migrate through the environment. The potential harm depends on specific receptors as well as exposure times and patterns that might bring receptors and contaminants into contact. The ways that the contaminants could come into contact with humans and the environment are called pathways. Chapter 6 addresses the human health pathway, and evaluates potential uses. Chapter 7 addresses the biological receptor pathway and evaluates how plant, animal, bird, or invertebrate species might be affected.

Chapter 8 identifies technologies that could remove contaminants from the setting or interrupt these pathways. Chapter 9 develops and evaluates remedial alternatives using these technologies. Chapter 10 compares the alternatives that can best address the problem. This evaluation and comparison will support a remedial decision to implement actions to protect human health and the environment.

5 Contaminant Fate and Transport

This chapter presents an evaluation of the anticipated behavior of contaminants present in vadose zone soil and groundwater at 100-BC. An approach is presented for the assessment of the anticipated future behavior of vadose zone contaminants that may function as secondary groundwater contaminant sources. The approach describes how contaminants released to the environment may affect the underlying groundwater.

An approach is also presented to simulate the future behavior of selected COPCs already present in 100-BC groundwater. The conditions affecting contaminant behavior, modeling methods and results, and uncertainties are also discussed, concluding with a summary of the chapter as a whole.

The purpose of the fate and transport information presented in this chapter is to describe:

1. The development of soil screening levels (SSLs) and PRGs for COPCs present in vadose zone soil at 100-BC and to describe the application of SSLs and PRGs to observed soil conditions to support the assessment of potential threats to groundwater and surface water.
2. How SSLs and PRGs are used to evaluate whether COPCs present in the vadose zone at waste sites characterized during the RI and during the preceding LFI, as well as previously remediated waste sites, may act as secondary sources of groundwater contamination.
3. The process that will be used to evaluate ongoing and future vadose zone remediation activities.
4. How existing Cr(VI), strontium-90, and tritium groundwater contaminant plumes will behave in the future if no further action is taken.

Understanding contaminant fate and transport in the environment is an important part of the RI/FS process. Simulations 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 waste 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 were simulated using a one-dimensional computer model for the vadose zone and a three-dimensional computer model for groundwater.

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-BC.
- Contaminant migration rates are currently much slower because liquid effluent discharges stopped in 1974.
- Eighty-one previously remediated waste sites were evaluated to assess the potential for groundwater and surface water quality impacts under native vegetation and irrigation conditions.
- Tritium at the 118-B-1 and 118-B-6 remediated waste sites was found to pose a threat to groundwater or surface water quality under native vegetation conditions.
- Pesticides, antimony, and carbon tetrachloride present in soil at the 100-B-1, 100-B-14:2, 100-B-18, 116-B-4, 116-B-14, and 118-C-1 waste sites may pose a threat to groundwater or surface water quality under future irrigation conditions.
- Localized tritium and strontium-90 groundwater plumes are present in 100-BC. Tritium decays to concentrations below its 20,000 pCi/L MCL between 5 and 15 years. Strontium-90 decays to concentrations below its 8 pCi/L MCL in a timeframe between 52 and 72 years.
- Contaminant transport simulations indicate that Cr(VI) concentrations in groundwater along the river shoreline persist at concentrations above the 10 µg/L state AQWC for timeframes between 104 and 108 years, and above the 48 µg/L groundwater CUL for up to 11 years.
- The persistence of Cr(VI) in 100-BC groundwater is attributed to slow transport and degradation rates.

This chapter describes the key processes affecting the fate and transport of COPCs present in 100-BC environmental media, and the effect these processes may have on the future distribution of COPCs. The information presented in this chapter was used to calculate SSLs and PRGs that are protective of

groundwater and surface water quality for the conditions simulated. Remediated waste site COPC concentrations are compared to the SSLs and PRGs to identify waste sites requiring consideration in the FS for groundwater and surface water protection. A saturated zone contaminant transport model was used to simulate the fate and transport of selected COPCs in groundwater for the period between 2010 and 2160.

The assumptions and model input parameters described in this chapter are important for future waste site remediation efforts and assessing achievement of remedial goals. The cleanup verification process, including demonstration of how remedial goals are achieved, involves comparing the waste site CSM to the generic CSM used to develop the SSLs and PRGs. To the extent a significant deviation in the two CSMs is observed, site-specific conditions can be used to evaluate the potential for a waste site to act as a source of groundwater contamination.

Eighty-one previously remediated waste sites representing 176 decision units were evaluated to determine whether further action might be needed for groundwater and surface water quality protection. Based on the results of these evaluations, tritium present in soil at the 118-B-1 and 118-B-6 waste sites was determined to pose a potential threat. The 100-C-7:1 side slope is also expected to represent a secondary Cr(VI) contaminant source. Remediation of this waste site is ongoing and scheduled for completion in early 2013. If irrigation of the land surface overlying the waste site were to occur, pesticides, antimony and carbon tetrachloride present at the 100-B-1, 100-B-14:2, 100-B-18, 116-B-14, 116-B-4, and 118-C-1 waste sites may pose a potential threat to groundwater or surface water quality.

5.1 Evaluation Process for Assessment of Groundwater and Surface Water Protection

The evaluation process used to assess whether there is potential for vadose zone contaminants to affect groundwater and/or surface water quality followed a specific set of logical steps as shown on Figure 5-1. This process is intended to determine whether the potential for soil contaminants to migrate to the underlying groundwater and, subsequently, be discharged to surface water at concentrations that would pose a threat to human or ecological receptors exists (this evaluation does not include assessment of potential for effects due to direct contact exposures to shallow or surface contamination). The activities associated with the evaluation process included the following:

- The available data describing the nature and extent of residual vadose zone soil contamination at a particular waste site are identified and assembled. This may include laboratory analysis of soil samples collected from the vadose zone, 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 or particle size distribution), and field observations made during drilling and/or excavation. These data have been generated from process knowledge and operating history, specific waste site characterization activities (for example, LFIs and RI activities), or completion and verification measurements associated with vadose zone remedial activities.

- The data that provide a description of residual contaminant distribution, including concentrations and horizontal and vertical distribution, for each waste site were evaluated to develop the waste site CSM.

- The waste site CSM is then compared to the generic CSM, upon which the SSL and PRG values are based. If the known waste site conditions are similar to those used for the SSL and PRG simulations (that is, 100:0 or 70:30 contaminant distributions, depending on individual contaminant soil:water partition coefficients), then the evaluation may follow the SSL and PRG comparison pathway.

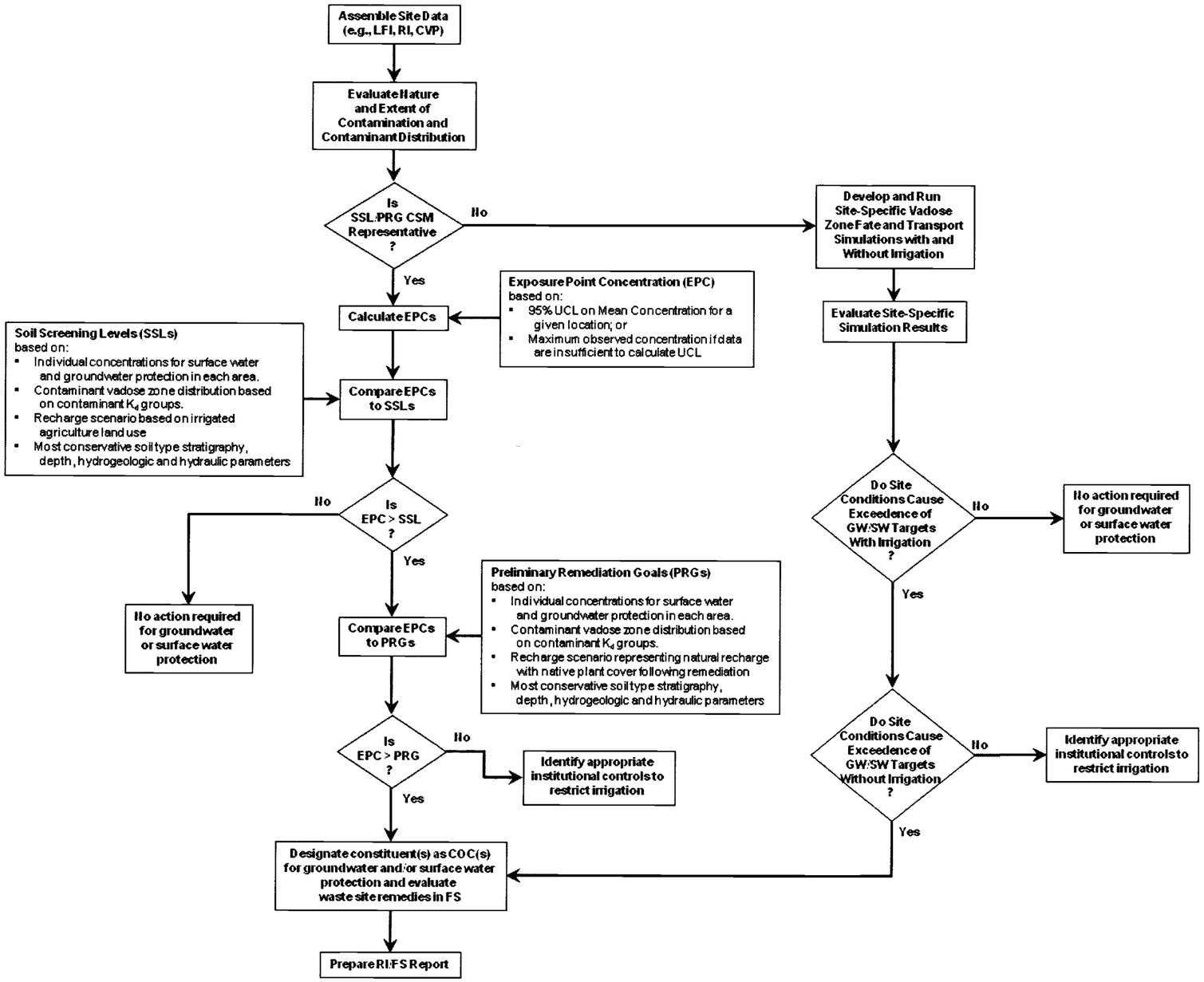


Figure 5-1. Screening Process for Groundwater/Surface Water Protection at Sites with Measurement Data Describing Site Conditions

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- If known waste site conditions differ from the generic CSM, then the waste site is evaluated using a site-specific contaminant transport simulation. Conditions that may indicate that the generic CSM is not representative include: presence of historical or persistent groundwater plumes associated with a specific waste site or operating area or contaminant distribution within the vadose zone that is not consistent with the default distribution (that is, 100:0 or 70:30) for that contaminant. Waste sites that are not represented by the default simulation scenarios are subsequently evaluated individually; that analysis is discussed later in this chapter.

- Exposure point concentrations (EPCs) for each COPC are calculated based on the site-specific data at hand. The EPC is calculated using either a 95 percent upper confidence level (UCL) on the mean or a maximum observed concentration (if insufficient data are available to derive a UCL).

- The EPCs for each contaminant at a waste site are then compared to the SSL. The SSLs represent a conservative groundwater and surface water protection value based on the assumption of a long-term irrigation recharge scenario. If the EPC is less than the SSL, then that contaminant is identified for no further action and the assessment moves to the next contaminant.

- If the site-specific contaminant EPC is greater than the SSL, then the EPC is subsequently compared to the PRG for that contaminant. The PRG represents a groundwater and surface water protection value based on the expected land use in the 100 Area (that is, conservation activities that do not include intensive irrigated agriculture). If the EPC is greater than the SSL, but less than the PRG, then the affected waste site is identified for application of institutional controls that will prevent irrigation in the future at the waste site. If the EPC is greater than the PRG, then the contaminant is identified as a contaminant of concern (COC) for groundwater/surface water protection, and the waste site carried forward into the feasibility study for identification of appropriate remedial alternatives to mitigate risks to groundwater and surface water posed by the vadose zone contamination.

If the condition described above (that is, waste site conditions are not adequately represented by the generic CSM used for SSL and/or PRG development simulations), then the waste site and its affected contaminants are evaluated using a site-specific vadose zone transport simulation. This simulation utilizes the same general fate and transport modeling approach used for the SSL and PRG development, except that site-specific conditions are substituted where appropriate. The site-specific simulation also uses the *Subsurface Transport Over Multiple Phases* (STOMP) transport computer code. The simulations include recharge estimates with and without the assumed application of irrigation. When the simulations are completed, the results were evaluated as follows:

- The site-specific transport simulation results for the specified contaminant are prepared and evaluated.

- The site-specific results under the irrigation recharge scenario are evaluated to determine whether the waste site conditions result in an exceedance of the contaminant-specific groundwater or surface water protection criteria (for example, maximum contaminant levels [MCLs] or AWQC). If the waste site conditions do not cause an exceedance of any of the criteria, then the site is identified for no further action. If site conditions cause an exceedance of the groundwater or surface water protection criteria, then the waste site is identified for application of institutional controls that prevent future irrigation at the site.

- The site-specific results under the expected land use (native vegetation recharge scenario) are evaluated next to determine whether waste site conditions result in exceedance of the groundwater or surface water protection criteria. If the site conditions do not cause an exceedance of any of the criteria, then the site is identified for no further action. If site conditions cause an exceedance of the

groundwater or surface water protection criteria, then the affected waste site is carried forward into the FS for identification of appropriate remedial alternatives to mitigate risks to groundwater and surface water posed by the vadose zone contamination.

Site-specific modeling can be done in either of two ways:

- Developing site-specific SSLs and PRGs in a manner similar to development of the generic CSM based SSLs/PRGs, but modifying the model input parameters to reflect the site-specific conditions that vary at the selected waste site, or
- Performing a site-specific fate and transport simulation to assess the site-specific effects to groundwater

The result of the site-specific evaluation is an assessment of the overall site conditions and if those conditions result in an unacceptable threat to groundwater or surface water. Site-specific evaluations were not performed for any 100-BC waste sites.

5.2 Overview of the 100-BC Conceptual Site Model

The components of a CSM include contaminant sources, release mechanisms, transport processes, exposure pathways, and receptors. The following subsections identify the primary sources of contamination in 100-BC, describe probable release mechanisms, and summarize the means by which contaminants are transported in the environment. Processes that influence contaminant fate are also described. The information presented in the following subsections lays the groundwork for the vadose zone contaminant transport assessment presented in Section 5.7 and the groundwater contaminant transport modeling presented in Section 5.8.

Components of the CSM that are essential to the evaluation of contaminant fate and transport under current and future conditions at 100-BC include the following:

- The disposal of contaminants creating the primary sources ceased when 100-BC reactor operations stopped in 1974.
- Contaminants associated with primary sources were transported to various depths in the vadose zone and to groundwater according to their relative mobility and volume of liquid effluent discharged. Moderately mobile to highly mobile constituents such as Cr(VI) and tritium were transported to the water table while slightly mobile constituents, such as strontium-90, moved more slowly.
- Contaminants that reached groundwater were transported at varying rates that are generally determined by their soil:water distribution coefficient (K_d). Mobile contaminants were transported by groundwater flow where they may have entered surface water by upwelling though the river bottom or by direct discharge via springs or seeps that, in turn, may flow overland across the riparian zone into the river. River bottom upwelling and springs and seeps represent ongoing sources of contaminant release in 100-BC.
- Secondary contaminant sources may occur at lower depths within the vadose zone, beneath remediated and unremediated waste sites, because of COPC sorption to soil. Residual contaminant concentrations at remediated sites were evaluated in cleanup verification package reports and reported to be below the applicable interim action RAGs. Residual contaminant concentrations are evaluated further in Section 5.7 to determine whether they pose a future threat to groundwater and surface water quality.

Groundwater in 100-BC contains Cr(VI) at concentrations above the 10 µg/L state AWQC. Cr(VI) concentrations above the 48 µg/L state MTCA groundwater CUL occur at several inland monitoring well locations. Strontium-90 and tritium occur in localized areas at concentrations above their respective federal drinking water standards.

5.2.1 Contaminant Sources

The identified sources of contamination at 100-BC fall into two types: primary and secondary. Primary sources are the process chemicals, working solutions, and radioactive and nonradioactive wastes that were released into the environment during the reactor operations period. Releases from the primary 100-BC sources ceased in 1974.

Secondary sources consist of environmental media (for example, soil, sediment, and groundwater) impacted by releases from the primary sources. These media can retain sufficient levels of contaminants that can act as a reservoir for continuing releases to adjacent soil, surface water, groundwater, or air. Current and future remedial actions will focus on the control of secondary contaminant sources that may result in either direct contact exposure to identified receptors, or be released and transported to groundwater or surface water or air, where potential exposures may occur.

Historical releases of various liquid and solid wastes resulted in contamination of the vadose zone and underlying groundwater. Contaminated groundwater flowed north towards the Columbia River and discharged into the river through surface springs and direct interaction of groundwater with surface water or the river's hyporheic zone. Unremediated waste sites continue to represent potential sources of contamination. The sites are evaluated in this chapter to determine if they pose a threat to human health and the environment.

5.2.1.1 Primary Sources

The contaminants detected at 100-BC arose from the historical operation of the two water-cooled nuclear reactors (105-B and 105-C) and their support infrastructure. The primary source was the liquid waste generated during reactor operation (that is, reactor cooling water, fuel storage basin water, and decontamination solutions). Reactor cooling water, obtained from the Columbia River, was conditioned before passing through the reactors. The conditioning process included solids removal and sodium dichromate addition for corrosion protection. Contaminants potentially introduced into the cooling water as it passed through the reactors consisted of fuel materials, fission and activation products, and residual Cr(VI).

Reactor cooling water was by far the largest volume of primary source material at 100-BC. Other liquid primary sources included concentrated water treatment chemical solutions (for example, high-concentration sodium dichromate solutions). Solid waste associated with reactor operations included sludge from the fuel storage basin, solids generated from cooling water pretreatment (for example, crystalline sodium dichromate dihydrate), reactor components, and various other contaminated items. Waste generated from reactor operations may have been contaminated with radionuclides, chemicals, or both (mixed waste).

5.2.1.2 Secondary Sources

The release of contaminants from primary sources to the environment resulted in contaminated vadose zone soil beneath facilities and waste sites. Some of this contamination may be mobile and subject to leaching to groundwater, transport by surface water run-on or run-off, or transport by wind as particulates or vapors. Left unremediated, this material may allow the further release of contaminants to the environment and potential exposure to human and ecological receptors. The main secondary source at 100-BC is vadose zone soil contaminated with Cr(VI).

In 100-BC, nearly all waste sites have undergone remediation, significantly reducing the potential for a remaining source. Other potential secondary sources that may still be present include the 10-C-7:1 west side slope (Cr(VI) contamination) and residual contamination within the vadose zone, periodically rewetted zone (PRZ) or the unconfined aquifer beneath remediated and unremediated waste sites. The C-7:1 side slope remediation excavation is expected to be complete by early 2013.

The evaluation of risks posed by the identified secondary sources to human health and the environment through direct exposure are discussed in Chapters 6 and 7, respectively. The potential for secondary sources to provide a significant ongoing source of contamination is evaluated through comparison of contaminant EPCs present in waste site vadose zone soil to the SSLs and PRGs (Section 5.7) for groundwater and surface water quality protection.

The continued presence of a Cr(VI) plume in 100-BC groundwater is largely due to the low horizontal hydraulic gradient and periodic gradient reversals that occur along the river shoreline. This, in turn, slows plume migration and the natural flushing of Cr(VI) from the unconfined aquifer. In general, the RI characterization dataset did not identify any significant remaining sources of Cr(VI) in the vadose zone or the PRZ except for the 100-C-7 and 100-C-7:1 subsites. Cr(VI) concentrations beneath remediated waste sites were generally low with the highest concentration of 1.05 mg/kg occurring beneath 118-B-6. At the 100-C-7 and 100-C-7:1 subsites, Cr(VI) concentrations up to 1,970 mg/kg have been detected. Interim remedial actions are ongoing at both sites.

5.2.2 Release Mechanisms

Primary release mechanisms are the processes during reactor operations that resulted in the initial distribution of contaminants to the environment. Secondary release mechanisms are the processes that redistribute secondary source contaminants to other environmental media.

5.2.2.1 Primary Liquid Waste Release Mechanisms

The primary release mechanism for liquid waste at 100-BC falls into two general categories: intentional or the planned discharge of liquid waste, and UPRs. Planned releases fall into two groups:

- High-volume, low-concentration liquids (for example, reactor cooling water) that were typically released directly to the Columbia River but were also sometimes discharged to engineered trenches.
- Lower-volume, higher concentration liquids (for example, contaminated reactor cooling water during upset conditions) that were released directly to the land surface and allowed to infiltrate or were discharged to engineered subsurface waste infiltration structures (for example, cribs and covered trenches).

In addition, acids, solvents, and cleaning solutions used in reactor maintenance were discharged to cribs and trenches.

During 100-BC reactor operations, high volumes of process effluent (which includes reactor cooling water), steam condensate, and fuel storage basin cooling/shielding water) were intentionally released via pipelines through retention basins to either outfalls to the Columbia River or trenches where the effluent was allowed to infiltrate into the vadose zone. In addition, liquid waste was intentionally released to the subsurface via cribs, trenches, French drains, and sewage disposal systems. Although trenches are surface features, their purpose was to facilitate rapid infiltration of wastewater into the vadose zone. Unplanned releases occurred during various material handling transfers. Cooling water and other liquid waste was routinely transferred using underground pipelines. Releases through leaks at joints or material failure between joints occurred because of corrosion, thermal, or other physical damage. Other UPRs at 100-BC included leakage of cooling water from pipelines and retention basins, and spillage of water

treatment chemicals (for example, sodium dichromate at various solution concentrations and as a crystalline solid).

5.2.2.2 Primary Dry Waste Release Mechanisms

Contaminants associated with dry solid waste were released to the environment through intentional disposal at waste sites or through UPRs of particulate material. Various solid wastes, including contaminated reactor hardware and components, were placed directly in burial grounds. Once the material was made available to interact with water or air, contaminants may have transferred to other media through leaching or dissolution. Dry granular or crystalline chemical materials (for example, sodium dichromate dihydrate) or contaminated soil particulates may also have become windborne, suspended in surface run-off, or transferred to soil through physical contact with a contaminated surface. Intentional/planned releases of solid waste are believed to account for most of the historical dry waste releases to the environment.

5.2.2.3 Secondary Release Mechanisms

Contaminated material in the vadose zone or aquifer is considered an ongoing secondary source of some contaminants. Secondary sources remaining in pipelines and control structures in the form of pipe scaling, corrosion products, sludge, or sediment may be released through structural failure of the pipeline and exposure to net infiltration. Other contaminant release mechanisms include the following:

- Volatilization to the atmosphere or soil gas (applicable to VOCs and tritium)
- Resuspension of particulates in air (applicable to all contaminants located at the soil surface)
- Transport in surface water run-off, both as dissolved constituents and suspended particles (applicable to all contaminants)
- Dissolution, desorption, and transport with infiltrating precipitation (applicable to soluble contaminants)
- Biotic uptake (applicable to soluble contaminants)
- Groundwater discharge to surface water and to the riparian zone in seeps and springs
- Release of reactor-related chemicals that could occur during facility demolition

5.2.3 Potential Routes of Migration

Contaminants released during 100-BC reactor operations or currently present as potential secondary sources have the ability to migrate through the air, vadose zone, groundwater, surface water, and biota uptake pathways. Each of these migration routes is discussed in the following subsections.

5.2.3.1 Air

Contaminants can potentially migrate to the air as vapor or solid particles. The meteorological conditions summarized here are described in more detail in Chapter 3. In the 100 Area, and along the Columbia River, local winds are strongly influenced by near river topography. At 100-BC, the prevailing wind direction is from the west (NEPA Characterization Report [PNNL-6415]).

During historical operations, contaminants may have been released with stack emissions. No stack release events have been identified for the 100-BC Reactors, and such events, if they occurred, would have likely distributed contaminants over a broad portion of the 100-BC Area. Without appropriate mitigation measures, strong winds can disperse contaminated surface soil while waste sites are exposed during

excavation or demolition. Methods employed to minimize this hazard include applying dust suppression water and soluble adhesives during remedial activities. Air monitoring is conducted during waste site remediation under approved site-specific air monitoring plans. Air monitoring programs conducted Sitewide are summarized in Chapter 2.

5.2.3.2 Vadose Zone

The stratigraphic units relevant to contaminant transport at 100-BC are the Hanford formation and Ringold unit E. The RUM interface is well below the groundwater table and is, therefore, not significant to contaminant transport within the vadose zone. Following are the characteristics of these two geologic units:

- The Hanford formation consists of gravel with sandy interbeds. Cobble-size clasts are common and boulders may be present.
- Ringold unit E consists of fluvial gravel with minor amounts of silt and sand. Ringold unit E is more inhibitive of water and contaminant movement and generally less permeable than the Hanford formation, although migration can still occur.

In the vadose zone, the downward movement of infiltrating water is generally perpendicular to the orientation of the sedimentary bedding present in the Hanford formation and Ringold unit E. Transient water saturated conditions were present during reactor operations throughout the vadose zone beneath high-volume, liquid effluent waste sites. Pressure head and vertical saturated hydraulic conductivity would have been the dominant factors controlling the downward movement of water and contaminants during this period.

The local saturation of the vadose zone due to creation of recharge mounds (see Figures 5-2 and 3-39) during the reactor operations period, resulted in the modification of local groundwater gradients, with water flowing radially away from the recharge mounds in all directions, with a preference towards the west (*Status of the Ground Water Beneath Hanford Reactor Areas April 1, 1962 through January 1, 1963* [HW-77170]). During this period, isotopic analysis of well and spring waters was used to calculate the average horizontal groundwater flow velocity. These evaluations revealed velocities ranging from 3.05 to 9.14 m/day (10 to 30 ft/day) versus a natural groundwater flow velocity of 0.07 to 0.7 m/day (0.23 to 2.3 ft/day).

In the Hanford Site's low rainfall, semi-arid climate, the soil water holding capacity, or matric potential, largely controls the movement of water through the soil and the corresponding rate of migration of mobile contaminants. Migration of contaminants through the vadose zone is currently slower than during reactor operations due to elimination of local liquid effluent discharges at the end of reactor operations and reestablishment of surface vegetation following remediation. Natural interbedding of gravel layers with fine grained sand and silt layers in the vadose zone produces capillary barriers that inhibit the vertical movement of water under natural water infiltration rates. Vertical migration of contaminants within the soil pore spaces is therefore slowed under these conditions.

5.2.3.3 Groundwater

As described previously in Section 3.6, the uppermost aquifer at 100-BC occurs within Ringold unit E, and in the lower portion of the Hanford formation.

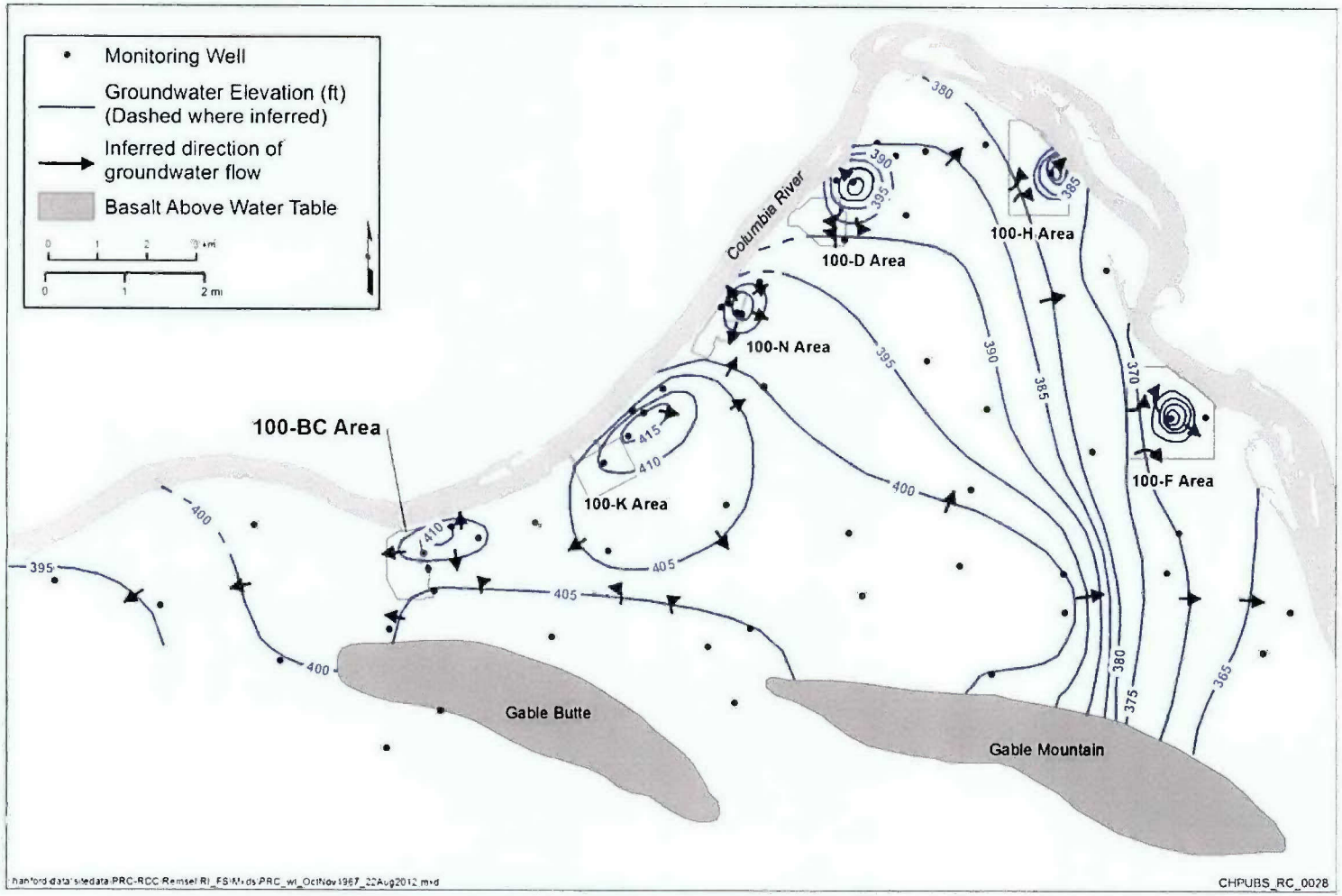

Figure 5-2. 100-BC Groundwater Mound during Operations (1967)

The groundwater flow system beneath 100-BC is the principal contaminant transport pathway between historical primary sources, secondary sources, and the river. Characterization of the 100 Area hydrogeology requires an understanding of surface water recharge sources, vadose zone characteristics, groundwater flow, and groundwater/surface water interactions. Both natural and anthropogenic recharge sources have influenced vadose zone water and unconfined aquifer groundwater flow patterns and contaminant distribution in the subsurface underlying 100-BC. The effects of natural processes on contaminant migration are ongoing, while the effects of anthropogenic driven processes have diminished with the end of reactor operations.

Generally, natural groundwater flow patterns transport contaminants north toward the Columbia River. Groundwater flow directions close to the Columbia River are influenced by the river stage. During the fall, when the river stage is relatively low, groundwater flow is toward the river. In spring, when the river stage is high, the water table near the river flattens and river water may flow a limited distance inland into the 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 Columbia River 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.

5.2.3.4 Surface Water

The Columbia River is the only natural surface water feature in 100-BC, except for the overland flow of seasonal seepage in the riparian zone. The routes of migration to the river include groundwater upwelling to the riverbed, previous discharges to the river during operations, and overland flow of water discharged from seeps. Previous groundwater discharges to the Columbia River were observed as seeps on the riverbank that may have transported mobile contaminants (*Status of the Ground Water Beneath Hanford Reactor Areas April 1, 1962 through January 1963* [HW-77170]). The Columbia River is the dominant aquatic ecosystem on the Hanford Site and supports a large and diverse population of plankton, benthic and lotic invertebrates, fish, and other ecological communities.

5.2.3.5 Biotic Uptake

Plants may absorb contaminants through their roots. Animals may also uptake contaminants from surface deposits, which could accumulate in their tissues. This migration pathway is discussed further in Chapters 6 and 7. Impacts from biotic uptake of Hanford Site contaminants is of particular concern in the riparian zone where contaminants may be found at the ground surface, within plants growing in contaminated soil, and where contaminated groundwater may discharge seasonally to the ground surface and flow overland to the river.

5.3 Contaminant Persistence

The persistence of various contaminants determines how long they remain in the environment and how long they are available for transport to potential receptors. If a contaminant remains in the environment for a long time, and is mobile, it is more likely to be transported from the vadose zone to the groundwater, and eventually to the surface water. Persistence also determines 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 leave the affected area. Radionuclides undergo radioactive decay at varying rates specific to the individual nuclide. Chemicals may also degrade (abiotically or through microbial mediated oxidation-reduction reactions) in the subsurface or be dispersed in a manner that reduces the mass and/or the concentration of the contaminant available for transport or direct exposure. The following paragraphs discuss the persistence of selected COPCs detected in 100-BC environmental media.

5.3.1 Nonradionuclide Chemical Constituents

The persistence or decay of chemical constituents at 100-BC is primarily driven by biological and geochemical oxidation/reduction (redox) processes, potential biological uptake, and physical processes (for example, volatilization and water solubility). The chemical constituents identified for this discussion include Cr(VI), which is generally present as an oxyanion, VOCs (trichloroethene and chloroform in groundwater), and PAHs. These constituents are subject to a variety of transformational processes.

Cr(VI) is both relatively stable and persistent in the vadose zone and groundwater environments at 100-BC. Chromium is typically present in the environment in one of two oxidation states, trivalent (chromium(III)) or hexavalent (Cr(VI)). Chromium(III) is typically precipitated in the environment as a low-solubility hydroxide molecule, $Cr(OH)_3$ and, as such, has low mobility and exhibits low mammalian toxicity. Cr(VI), however, is acutely toxic and is typically present under ambient conditions at 100-BC as a soluble oxyanion, $Cr_2O_7^{-2}$ or CrO_4^{-2}, depending primarily on pH. The ionic forms of Cr(VI) are relatively stable under the redox conditions typically found in soil and unconfined groundwater at 100-BC, and as such, Cr(VI) tends to have moderate mobility under present conditions.

The primary source of Cr(VI) present in the subsurface environment was the sodium dichromate dihydrate 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 lower solubility. These compounds can include potassium dichromate (which is about one tenth as soluble as sodium dichromate dihydrate) and lead chromate (which is relatively insoluble). The Cr(VI) ion can also undergo chemical reduction under moderately reducing conditions, or upon reaction with reducing agents such as ferrous iron. Ferrous iron is effective at reducing Cr(VI) to Chromium(III), producing a very low solubility hydroxide molecule (*Monitored Natural Attenuation of Inorganic Contaminants in Ground Water* [EPA/600/R-07/140]).

Trichloroethene and chloroform, both found at relatively low concentrations in 100-BC groundwater, degrade very slowly under the redox and dissolved oxygen conditions currently present in groundwater (that is, moderately aerobic). These compounds can be reductively dechlorinated by facultative[1] and obligate[2] anaerobic microorganisms under anoxic conditions or undergo abiotic transformation (*Identification and Characterization Methods for Reactive Minerals Responsible for Natural Attenuation of Chlorinated Organic Compounds in Ground Water* [EPA 600/R-09/115]). Additionally, trichloroethene and chloroform may volatilize from the land surface or surface water directly to the atmosphere. Trichloroethene 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 zone (for example, below a few meters bgs) or from groundwater accounts for only a small potential loss. Once in the atmosphere, these compounds can be destroyed through photodegradation (sunlight). The potential for volatilization, abiotic or biologically mediated degradation is dependent upon the specific physical and chemical properties of the constituent, the diversity and population of microorgansims, and the geochemical characteristics of the subsurface environment. The chlorinated VOCs present at 100-BC are expected to persist in soil and groundwater, however, due to the low concentrations present they pose limited threats to human health and the environment.

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[1] Can survive in both aerobic and anaerobic conditions.

[2] Can survive only in anaerobic conditions.

Microbial metabolism is the primary process for transformation and degradation of PAHs in soil environments. Photolysis, hydrolysis, and oxidation generally are not considered to be important processes for PAH degradation in soil ("Fate of Polynuclear Compounds (PNAs) in Soil-Plant Systems" [Sims and Overcash, 1983]). A study performed to assess abiotic degradation (photolysis and oxidation) reported mass losses of 13 percent, 8.3 percent, and 15.8 percent for naphthalene, anthracene, and phenanthrene, respectively. No significant abiotic-driven mass reduction was observed for other PAHs (Fate of PAH Compounds in Two Soil Types: Influence of Volatilization, Abiotic Loss and Biological Activity. Environ. Toxicol. Chem. 9:187-195, Park et al., 1990).

Environmental factors that may influence PAH microbial degradation rates in soil include temperature, pH, oxygen concentration, initial PAH concentrations, soil contamination history, soil type, moisture content, nutrients, and other substances that may act as substrate co-metabolites ("Fate of Polynuclear Compounds (PNAs) in Soil-Plant Systems" [Sims and Overcash, 1983]). The size and makeup of microbial populations, in turn, can be affected by these factors. For example, in low pH soil, fungi are dominant over bacteria, and thereby control microbial degradation in these environments. Sorption of PAHs to organic matter and soil particulates also influences bioavailability and, hence, biotransformation potential. Sorption of PAHs by soil organic matter may limit biodegradation of compounds that would otherwise rapidly undergo metabolism (Factors affecting the microbial degradation of phenanthrene in soil, Appl. Microbiol. Biotechnol. 35: 401-405. Manila and Alexander, 1991; Adsorption of Polycyclic Aromatic Hydrocarbons (PAHs) by Soil Particles: Influence on Biodegradability and Biotoxicity. Appl. Microbiol. Biotechnol.36: 689-696, Weissenfels et al., 1992).

Based on laboratory experiments, observed biodegradation half-lives have ranged from as little as 2.1 days for naphthalene to as much as 420 days for dibenz(a,h)anthracene (Fate of PAH Compounds in Two Soil Types: Influence of Volatilization, Abiotic Loss and Biological Activity. Environ. Toxicol. Chem. 9:187-195, Park et al., 1990). However, these experiments were likely conducted under favorable oxygen, nutrient, and temperature conditions. Actual PAH half-lives are expected to be much greater because of the less favorable oxygen and nutrient conditions present in most subsurface environments.

5.3.2 Radionuclide Constituents

Radionuclide persistence is controlled by the radioactive decay process that transforms the original radioisotope either into another isotope of the same element or into another element. The daughter product of radioactive decay may be a radionuclide or a stable isotope. Exclusive of their relative mobility in the environment, radionuclides with relatively long half-lives ($T_{1/2}$)[3] are of more environmental concern than radionuclides with shorter $T_{1/2}$. This is due to the potential for constituents with longer half-lives to remain in the environment after release and to present a potential for exposure to human and ecological receptors either through direct exposure at or near the point of release, or through migration to distant exposure points. The following radionuclides were identified as COPCs for 100-BC:

- **Tritium in soil and groundwater.** $T_{1/2}$ = 12.3 years; decays by beta emission to stable helium.
- **Stronium-90 in soil and groundwater.** $T_{1/2}$ = 28.8 years; decays beta emission to yttrium-90, which decays by beta emission with a $T_{1/2}$ of 64 hours to stable zirconium.
- **Cesium-137 in soil.** $T_{1/2}$ = 30.2 years; decays by beta emission to barium-137.
- **Carbon-14 in soil.** $T_{1/2}$ = 5,730 years; decays by beta emission to stable nitrogen.

[3] For this RI/FS, half-life data were obtained from the Radiochemistry Web site in September 2011.

- **Europium-152 in soil.** $T_{1/2}$ = 13.5 years; decays electron capture to stable samarium-152 (branch ratio 72.1 percent) and by beta emission to gadolinium-152, a long-lived alpha emitter (branch ratio 27.9 percent).
- **Europium-154 in soil.** $T_{1/2}$ = 8.6 years; decays by beta to stable gadolinium-154 (branch ratio 99.98 percent) and by electron capture to stable samarium-154 (branch ratio 0.02 percent).
- **Cobalt-60 in soil.** $T_{1/2}$ = 5.27 years; decays by beta emission to stable nickel-60.
- **Nickel-63 in soil.** $T_{1/2}$ = 100.1 years; decays by beta emission to stable copper-63.
- **Plutonium-238 in soil.** $T_{1/2}$ = 88 years; decays via alpha to uranium-234.
- **Plutonium-239/240 in soil.** There are 15 radioisotopes of plutonium. The three most common are plutonium-239 (~ 94 percent), plutonium-240 (~ 6 percent), and plutonium-241 (0.4 percent). The $T_{1/2}$ of plutonium-239 is 24,100 years, and it decays to uranium-235 with the emission of an alpha particle. The $T_{1/2}$ of plutonium-240 is 6,540 years, producing uranium-236 via alpha decay.
- **Americium-241 in soil.** $T_{1/2}$ = 432 years; decays by alpha emission to neptunium-237.

In addition to radioactive decay, the persistence of radionuclides in groundwater is affected by their individual chemical and physical behaviors. For example, strontium-90 will also behave as a metallic strontium molecule, with the exception being that the isotope fraction will decay. In the environment, strontium-90 commonly remains as an exchangeable divalent cation. As such, it is not readily mobile and tends to be sorbed on soil particles near its point of release. Conversely, tritium is found in the environment most commonly as tritiated water. Tritium typically replaces one of the hydrogen ions in a water molecule. In the environment, tritium behaves as water because it is most commonly a part of a water molecule. Tritium is transported at the same rate as other water molecules.

5.4 Vadose Zone Contaminant Migration Assessment

In addition to posing direct contact risk to future human or ecological receptors, which are discussed in Chapters 6 and 7 respectively, COPCs present in vadose zone soil may pose a threat to groundwater and surface water quality. Leaching of COPCs from secondary sources (contaminated soil) with subsequent transport by infiltrating water (rainfall, snowmelt, export water system [EWS] leaks, dust suppression, irrigation, or septic drain field return flow) represents the primary vadose zone contaminant transport process for nonvolatile COPCs.

Contaminants released from the 100-BC primary and secondary sources were transported through the vadose zone and, in some cases, reached the water table. This section describes the factors affecting contaminant transport through the vadose zone. The most significant factors affecting 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.

Once contaminants reached groundwater, mobile contaminants traveled with the groundwater in the direction of groundwater flow. Contaminated groundwater may flow downgradient to upwell into the Columbia River. In addition, contaminated groundwater may be seasonally discharged through seeps or springs to flow overland across the riparian zone to the river. Seasonal seeps and springs may be an ongoing contaminant source for the riparian zone.

The vadose zone contaminant migration assessment presented in this section focuses on evaluation of waste sites characterized during the LFI, the current RI, and sites where soil remediation is complete and CVP data describing post-remediation conditions are available (that is, the "previously remediated sites"). This assessment provides a basis for confirming that completed remedial actions at these sites do not pose a threat to groundwater or surface water.

5.4.1 Factors Affecting Contaminant Migration in the Vadose Zone

Contaminant migration from 100-BC waste sites through the vadose zone to the underlying aquifer is controlled by the driving forces and the interactions between the contaminants and sediments specific to 100-BC. The driving forces include gravity, matric potential gradients, recharge, and artificial discharges such as those from septic tank leach fields, pipe leaks, and tank leaks. The types, thicknesses, and properties of the sediments affect the rate and direction of solute and water movement in the vadose zone. A contaminant's concentration in the groundwater and its concentration in Columbia River surface water, including the peak concentration, are dependent on the solute flux from the vadose zone, vadose zone thickness and properties, groundwater flux rates, travel distance, groundwater and river water mixing, and the sample location. Each contaminant's decay rate and propensity to sorb to vadose zone or aquifer materials are important factors that control the peak concentration from which the SSL and PRG are calculated. Specific factors that affect contaminant transport in the vadose zone are discussed further in the following subsections.

5.4.1.1 Infiltration and Recharge

Net infiltration into the vadose zone is driven by competition between the processes of precipitation, evaporation, transpiration, run-off, and run-on. In a semiarid or arid climate, downward water fluxes resulting from this competition are episodic and usually infrequent. Many 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 the Vadose Zone Hydrogeology Package (PNNL-14702) for the 100 Area.

The 100 Area-specific recharge rates described in the Vadose Zone Hydrogeology Package (PNNL-14702) vary with surface soil type and provide an estimate of the range of possible recharge rates for various land uses. The four surface soil types identified in the 100 Area include the Ephrata Sandy Loam, Ephrata Stony Loam, Burbank Loamy Sand, and Rupert Sand. Recharge rates for the Ephrata Sandy Loam and the Ephrata Stony Loam are described as being identical (Vadose Zone Hydrogeology Package [PNNL-14702]). The Ephrata soil series and Burbank Loamy Sand are the primary soil series present in 100-BC. Two other soil series identified as the Kiona Silt Loam and Pasco Silt Loam also occur in remote portions of 100-BC. The Kiona Silt Loam occupies steep slopes and ridges whereas the Pasco Silt Loam occurs in low areas adjacent to the Columbia River. These latter two soil series were not considered part of the vadose zone contaminant migration assessment.

The long-term natural driving force for contaminant 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 through the vadose zone to groundwater.

Direct measurement of naturally occurring recharge attributed to surface infiltration at the Hanford Site is not practical. Therefore, the measurement is made indirectly because the thickness of the vadose zone and the time scale required for water to travel from the surface to the water table would require long periods of observation. 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 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 indicates the sagebrush 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 *Artemisia tridentata* (sagebrush) on the surface may continue to reduce the species richness on the surface (Figure 5-3).

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 infest the barrier surface, which indicates that the restored barrier surface is beginning to function like a recovering ecosystem.

Numerous studies have estimated recharge rates for the vadose zone 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 soils, similar to those present at 100-BC, where a 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.

The recharge rate affects the flow velocity/volume of water through the vadose zone. The flow velocity in the vadose zone is expected to have been greatest beneath the retention basin, French drains, trenches, and cribs during the operational period when percolation was greatest. 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 (bare soil) surface conditions. After 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 and reduce the net infiltration rate to the vadose zone.

The recharge input values to the STOMP model for SSL (irrigation) and PRG (native vegetation conditions) development were based on the vadose zone data package compiled in the Vadose Zone Hydrogeology Package (PNNL-14702). These data provided the basis for the recharge rates input to the sequential models used to calculate 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 the post-2010 model, which simulated the migration of water and contaminants to the underlying aquifer. The recharge rates, summarized in the following paragraphs, are discussed in detail in *STOMP 1-D Modeling for Determination of Soil Screening Levels and Preliminary Remediation Goals for 100 Area B and C Source Areas* (ECF-HANFORD-12-0003) (Appendix F).

Source: *200-BP-1 Prototype Hanford Barrier Annual Monitoring Report for Fiscal Years 2005 Through 2007* (PNNL-17176).

Figure 5-3. Prototype Hanford Barrier Cover in 2007 Dominated by Tall Sage (*Artemisia tridentata*) Covering Most of the Soil Surface, 13 Years after Plant Community Establishment

For the pre-2010 simulations, land use and recharge rates were assumed to change from a native shrub-steppe plant community (preoperational) to bare soil (operational). Recharge rates for each type of land cover, and for each soil type, were applied to the top boundary of the model for the year 0 to year 1944 (preoperational) period. The preoperational period is an arbitrarily long period used simply to establish a steady state moisture profile for the specified recharge rate. The second period, which extends from 1944 to 2010, corresponds to the operational period (Table 5-1) and accounts for surface cover changes arising during the operational era and their influence on soil moisture conditions in the soil column.

Table 5-1. Recharge Rate for Different Time Periods in STOMP Model Simulations

Soil Type	Recharge Scenario	Recharge Rate (mm/yr)				
		0 to 1944	1944-2010	2010-2015 (Period 1)	2010-2045 (Period 2	2045-5010 (Period 3)
Ephrata Sandy and Stony Loam	Native Vegetation	1.5	17.0	17.0	3.0	1.5
	Irrigation	1.5	17.0	17.0	71.4	69.9
Burbank Sandy Loam	Native Vegetation	3.0	52.0	52.0	6.0	3.0
	Irrigation	3.0	52.0	52.0	74.4	71.4

Notes:

Period 1: Disturbed surface and no vegetation

Period 2: Developing young shrub steppe vegetation progressing to mature shrub steppe vegetation

Period 3: Mature shrub-steppe vegetation

Three recharge periods were specified for the post-2010 simulations to account for evolutionary changes in the surface cover (Table 5-1). Bare soil was assumed to continue to be the land cover above the waste site during the first recharge period spanning 2010 to 2015. For the native vegetation scenario, the second recharge period of 30 years (*Regulatory Basis and Implementation of a Graded Approach to Evaluation of Groundwater Protection* [DOE/RL-2011-50]) represents a period where grasses and shrubs cover 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 decrease with time as the vegetation cover transitions from bare soil to 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. The recharge rates (Table 5-1) used in the model were adjusted to reflect the two primary soil types present in 100-BC (see Figure 3-31) and the different vegetation covers that evolve over time. The first recharge period (Period 1) of 5 years begins in 2010 with an initial disturbed soil surface and no vegetation. This is followed by a 30-year period (Period 2) with a cover of developing young shrub-steppe vegetation progressing to mature shrub-steppe vegetation (Period 3). The recharge rate to the aquifer during Period 2 increases from that present in Period 1 for both soil types. For a future irrigation land use scenario, Periods 2 and 3 assume an additional 68.4 mm/yr of recharge for each soil type that assumes total recharge is a combination of irrigation and non-irrigation recharge.

As the recharge rates used in the RESRAD simulations were different 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 non-irrigation total recharge rate was 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 was added to the native soil recharge rate to determine a recharge rate for the irrigation scenario for each soil type in the SSL and PRG simulations. The resulting recharge rates for native soil and irrigation scenarios for each soil type are shown in Table 5-1.

The final SSL and PRG values were calculated assuming the most conservative set of input values. For the SSL calculation, the maximum recharge rate (irrigation scenario) was applied for each soil type (Table 5-1) at 100-BC. For the PRG value calculation, the base case recharge rate was applied for each soil type.

5.4.1.2 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 the 100-BC SSL and PRG values were determined by running STOMP simulations for a number of stratigraphic columns that represent the range of conditions present at 100-BC.

Representative 100-BC stratigraphic columns for the STOMP model simulations were developed from existing borehole logs. The two lithologic units present in the 100-BC vadose zone include the gravel-dominated Hanford formation and Ringold Formation unit E. Due to its coarse texture and higher permeability, the Hanford formation transmits water and dissolved or particulate bound contaminants more rapidly than the underlying Ringold Formation. The borehole geologic logs were divided into seven stratigraphic column types (Figure 5-4) to correspond with the range of vadose zone thicknesses (based on June 2008 [high groundwater elevation] conditions) and lithologic composition observed in 100-BC. Each column was assumed to contain a clean backfill layer to represent conditions following interim remedial action completion. Backfill was assumed to replace the uppermost 4.6 m (15 ft) of native material in each column.

5.4.1.3 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 cover types discussed in the preceding section will cause variations in the moisture and matric potential, in accordance with the net infiltration allowed by each cover type.

Like pressure head, matric potential can be measured in the field and the laboratory. In situ measurements of matric potential in the shallow Hanford Site vadose zone have been made using tensiometers and heat dissipation sensors installed 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 van Genuchten ("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-BC 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 Area B and C Source Areas* (ECF-HANFORD-12-0003) (Appendix F).

5.4.1.4 Sorption

One of the single most important properties influencing contaminant mobility is the soil:water distribution coefficient or K_d value. This parameter is dependent on the relative abundance of different cations and anions in soil, soil pH, reduction oxidation potential, cation exchange capacity, and the organic carbon content of the soil matrix. In general, organic constituents with lower molecular weights have lower K_d values than those with higher molecular weights. K_d values for metals and metallic radionuclides are influenced by soil pH, cation exchange capacity, and the metal's oxidation state.

Several metals of environmental concern, such as arsenic, chromium, and mercury, may occur in more than one oxidation state: trivalent (III) and pentavalent (V) arsenic; trivalent (III) and hexavalent (VI) chromium; and monovalent (I) and divalent (II) mercury represent a few common examples. The oxidation state and mineral speciation of these metals determines their relative mobility and toxicity. Cr(VI), originally released as high-solubility sodium dichromate dihydrate, is moderately mobile in soil and, therefore, moderately sorbed. However, Cr(VI) may also be present in soil as relatively low-solubility mineral species such as potassium dichromate or lead chromate. Batch leach testing performed during the RI indicates that Cr(VI) present in vadose zone soil is moderately mobile (K_d = 0.8 mL/g) suggesting different mineral phases may be present. Chromium(III) is relatively immobile, being generally present as relatively insoluble precipitate such as chromium hydroxide $Cr(OH)_3$ (*Ground Water Issue: Behavior of Metals in Soils* [EPA/540/S-92/018]).

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]).

Strontium-90 is assumed to exist as cationic species in the 100-BC subsurface environment. Cationic species, which have higher K_d values than anions, are typically moderately mobile to relatively immobile. Cations are absorbed by clay minerals, oxides, and organic matter. Adsorption is pH dependent, increasing with increasing pH. Stronium-90 may also become immobilized by forming precipitates with phosphate, carbonate, and hydroxide (*Ground Water Issue: Behavior of Metals in Soils* [EPA/540/S-92/018]).

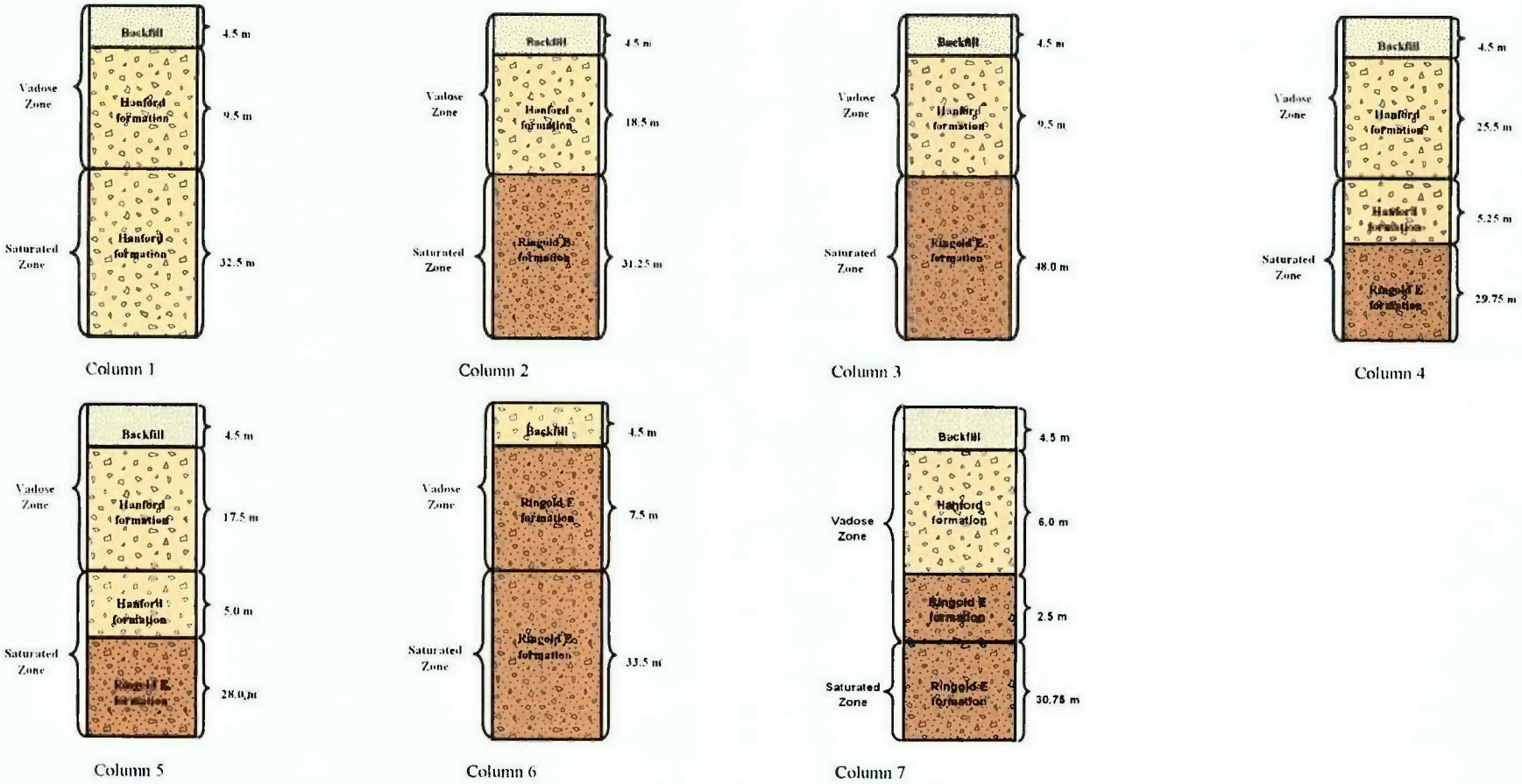

Figure 5-4. Representative Stratigraphic Columns for 100-BC STOMP Model Simulation

Table 5-2 summarizes the K_d values and relative mobility for selected 100-BC COPCs. These contaminants are grouped by their relative mobility and K_d values. The K_d values used for the STOMP simulations were taken from *Calculation of Nonradiological Preliminary Remediation Goals Using the Fixed Parameter 3-Phase Equilibrium Partitioning Equation for the Protection of Groundwater for the 100 Areas and 300 Area Remedial Investigation/Feasibility Study Reports* (ECF-HANFORD-10-0442) (Appendix F).The K_d value for strontium-90 was obtained from *Corrective Measures Study for 100-NR-1 and 100-NR-2 Operable Units* (DOE/RL-95-111). As described further in Section 5.5, batch leach testing was performed during the RI to develop a site-specific K_d value for Cr(VI).

Table 5-2. Relative Mobility of Selected 100-BC Contaminants of Potential Concern

Relative Mobility	Inorganics	Radionuclides	Organics
High	None	Tritium (K_d=0 mL/g) Carbon-14 (K_d=0 mL/g)	None
Moderate	Cr(VI) (K_d=0.8 mL/g)	None	Trichloroethene (K_d = 0.094 mL/g)
Slight	None	Strontium-90 (K_d=25 mL/g)	PAHs- Naphthalene (K_d = 1.19 mL/g)
Low	Chromium (K_d = 200 mL/g)	Cesium-137 (K_d=50 mL/g) Cobalt-60 (K_d=50 mL/g) Nickel-63 (K_d=30 mL/g) Europium-152 (K_d=200 mL/g) Europium-154 (K_d=200 mL/g)	PAHs – Benzo(a)pyrene (K_d = 5500 mL/g)

Source: *Calculation of Nonradiological Preliminary Remediation Goals Using the Fixed Parameter 3-Phase Equilibrium Partitioning Equation for the Protection of Groundwater for the 100 Areas and 300 Area Remedial Investigation/Feasibility Study Reports* (ECF-HANFORD-10-0442).

Highly Mobile COPCs. Contaminants that are considered highly mobile move freely with the water in which they are dissolved, exhibiting limited interaction with the vadose zone soil or aquifer matrix that would retard their movement. High mobility contaminants are those that exhibit a K_d of zero (no retardation) such as tritium and carbon-14.

Moderately Mobile COPCs. Moderate mobility contaminants move readily with infiltrating water or groundwater but also exhibit a moderate degree of interaction with vadose zone soil and aquifer sediments. Sorptive processes generally tend to slow the rate of migration of these contaminants. A reduction is observed in the contaminant's concentration with increasing depth in the vadose zone and in groundwater with increasing distance downgradient from the point of entry. The definition of moderately mobile is somewhat subjective; for the 100 Area, moderate mobility contaminants are identified as those exhibiting K_d values greater than 0 but less than 1. Hexavalent chromium falls in the moderately mobile group.

Slightly Mobile COPCs. Contaminants in the slight mobility group exhibit a high degree of interaction with vadose zone soil and aquifer sediments and, as a result, migrate slowly through the vadose zone and aquifer. The concentration in soil decreases rapidly with increasing vadose zone depth, while the concentration in groundwater decreases with increasing distance from the point of entry. The decrease

in concentration is due to the relatively large fraction of the contaminant that interacts with soil. This contaminant becomes sorbed to the vadose zone and aquifer solids. The slight mobility group includes contaminants that exhibit K_d values greater than one but less than 30 such as strontium-90.

Low Mobility COPCs. Contaminants with low mobility sorb strongly to vadose zone soil such that no apparent migration occurs under natural recharge rates and infiltrating water near neutral pH conditions. Liquid waste sources with highly acidic or basic pH values, or those that contained complexing agents, may have transported these contaminants deeper into the vadose zone at the time of disposal, but their migration has since decreased as the liquid waste equilibrated with the vadose zone soil. These constituents are not expected to reach the water table except at waste sites that discharge directly to the aquifer (deep injection/reverse wells) or where the vadose zone is very thin, or where the effluent had acidic or caustic pH. Those that may have reached groundwater would not be expected to migrate much further in the aquifer. The low mobility contaminants are identified as those that exhibit K_d values greater than 30 and include chromium(III) and many radionuclides.

5.5 Batch Leach Tests

As described in the 100-BC Work Plan (DOE/RL-2008-46-ADD3), batch leach tests were conducted on samples from 12 borings and 3 test pits at 100-BC to estimate K_d values for selected metals to support the vadose zone modeling assessment. Contaminants present in pore water within the bulk soil matrix were not analyzed or accounted for separately, as the associated contaminant mass is included within the bulk leachate concentration. 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.

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 United States Department of Agriculture soil grain-size classification scheme. 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 digestions was performed using Method 6010, 6020, or 200.8 for ICP metals, as applicable (bulk soil was digested using Method 3050B or 3051 for metals and Method 3060A for Cr(VI) to prepare for analysis). Separate aliquots of material were used for the bulk soil analysis and leaching tests.

The K_d was calculated as the ratio of the contaminant sorbed to soil 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)

The boring number, formation identification, sample depth intervals, and calculated K_d values for the metals identified above are presented in Table 5-3. Calculated K_d values for selenium and silver are not shown in the table because the soil samples had consistently non-detect concentrations. The complete datasets for vadose zone samples are provided in Appendix D.

For each vadose zone soil sample, four replicate samples were analyzed for total 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 the 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 in Table 5-3. In the common case, where an average soil concentration was calculated but the leachate water concentration was below reporting limit, that reporting limit was used in the calculation of a minimum K_d value, and a greater than (>) sign was placed before the calculated K_d value shown in Table 5-3.

5.5.1 Batch Leach Testing Data Evaluation

The batch leach test results were further evaluated to provide a basis for estimating a K_d value to use in the vadose zone transport estimates used to calculate the SSLs and PRGs. This data analysis includes evaluation of uncertainty and a focused statistical analysis to recommend an area-wide conservative estimate for the Cr(VI) K_d value.

In calculating K_d using the equation above, it was assumed that each soil sample was 100 g, and the volumes of water used in the ratios were 100, 250, and 500 mL. Exact quantities of soil and water were not available from the laboratory, but the K_d value is not very sensitive to slight variances from these assumed values. Given these uncertainties, along with laboratory analytical uncertainty, the reported K_d values are considered accurate within approximately 30 percent.

Because of the nature of the procedure, these K_d values are to be viewed as *desorption* partition coefficients, as opposed to adsorption coefficients. It is common to observe differences in K_d between adsorption and desorption reactions, termed hysteresis ("Nonreversible Adsorption of Divalent Metal Ions (Mn^{II}, Co^{II}, Ni^{II}, Cu^{II}, and Pb^{II}) onto Goethite: Effects of Acidification, Fe^{II} Addition, and Picolinic Acid Addition" [Coughlin and Stone, 1995]), with the desorption K_d usually greater than the adsorption value.

Except for silver and selenium, many of the metals were detected at some frequency in the soil samples. Chromium (total) was detected at concentrations between 2.75 and 246 mg/kg. Cr(VI) was at concentrations between 0.4 and 0.80 mg/kg.

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Table 5-3. Calculated Soil-Water Partition Coefficients (K_d)

Waste Site	Formation	Interval	Depth (ft bgs)	Arsenic K_d (L/kg)			Barium K_d (L/kg)			Cadmium K_d (L/kg)			Chromium K_d (L/kg)			Cr(VI) K_d (L/kg)			Lead K_d (L/kg)		
				1:1	1:2.5	1:5	1:1	1:2.5	1:5	1:1	1:2.5	1:5	1:1	1:2.5	1:5	1:1	1:2.5	1:5	1:1	1:2.5	1:5
Well 199-B8-9 (C7508)	Hanford	Interval I-001	80.1 – 82.6	>25	>24	>17	545	900	1,480	>6	>4	>1	3,810	>913	>758	N/A	N/A	N/A	121	>39	>30
	Hanford	Interval I-002	85.8 – 88.3	>24	>23	>20	466	654	1,120	>5	>4	>1	>421	>419	>417	N/A	N/A	N/A	>35	>33	>31
	Hanford	Interval I-003	89.5 – 92.0	>23	>18	>15	302	610	1,190	>6	>3	>1	676	>381	>379	N/A	N/A	N/A	>40	>31	>29
	Hanford	Interval I-004	93.0 – 95.5	>17	>16	>13	298	284	397	>5	>3	>1	451	277	415	N/A	N/A	N/A	>31	>30	>27
	Hanford	Interval I-005	95.0 – 97.5	>18	>16	>14	451	429	502	>5	>3	>1	386	329	628	N/A	N/A	N/A	105	145	>39
	Hanford	Interval I-011	100.5 – 107.6	>17	>15	>13	207	348	616	>4	>2	>0	679	1,270	2,540	N/A	N/A	N/A	17.6	15.1	126
Well 199-B2-16 (C7784)	Hanford	Interval I-001	23.6 – 26.1	>29	>28	>20	393	241	190	>7	>6	>2	1,290	921	771	N/A	N/A	N/A	>54	182	>41
	Hanford	Interval I-002	28.5 – 31.0	>31	>24	>22	1,140	1,860	501	>7	>4	>2	5,550	>1,290	4,310	N/A	N/A	N/A	>76	>61	>59
	Ringold unit E	Interval I-004	36.4 – 38.9	>22	>21	>18	603	7,250	10,200	>3	>2	>0	1,760	>492	>489	N/A	N/A	N/A	>66	>65	>62
	Ringold unit E	Interval I-005	38.4 – 40.9	>23	>21	>19	5,300	14,600	9,710	>4	>2	>0	826	>759	>756	N/A	N/A	N/A	>55	>54	>51
	Ringold unit E	Interval I-013	47.3 – 48.3	>15	>17	>14	241	632	774	N/A	N/A	N/A	227	1,400	>333	N/A	N/A	N/A	478	>180	>177
Well 199-B3-51 (C7785)	Ringold unit E	Interval I-001	29.8 – 30.8	>15	>76	>73	>2,930	24,400	36,600	N/A	N/A	N/A	>1,370	>6,830	>6,830	N/A	N/A	N/A	>21	>106	>103
	Ringold unit E	Interval I-002	34.7 – 36.5	>19	>98	>95	10,900	27,200	36,200	N/A	N/A	N/A	>1,360	>6,780	>6,780	>22	>20	>18	>22	>20	>18
	Ringold unit E	Interval I-003	40.4 – 42.2	>11	>57	>55	6,260	>13,800	>13,800	N/A	N/A	N/A	>1,260	>6,310	>6,310	N/A	N/A	N/A	>16	>84	>82
	Ringold unit E	Interval I-004	43.5 – 44.8	>15	>75	>73	>3,050	19,100	25,400	N/A	N/A	N/A	>1,200	>5,980	>5,980	N/A	N/A	N/A	>14	>74	>72
	Ringold unit E	Interval I-005	45.1 – 47.6	N/A	N/A	N/A	4,490	11,700	19,500	N/A	N/A	N/A	>1,240	>6,210	>6,210	N/A	N/A	N/A	>18	>93	>90
	Ringold unit E	Interval I-011	49.0 – 51.5	N/A	N/A	N/A	>1,740	10,900	21,800	N/A	N/A	N/A	>799	>4,000	>4,000	N/A	N/A	N/A	>15	>78	>75
Well 199-B5-8 (C8244)	Hanford	Interval I-001	80.7 – 83.2	>10	>53	>50	>2,260	11,300	9,440	N/A	N/A	N/A	>1,080	>5,410	>5,400	N/A	N/A	N/A	>32	>161	>158
	Hanford	Interval I-002	85.8 – 88.3	>7	>38	>36	>2,240	18,600	27,900	N/A	N/A	N/A	>509	>2,550	>2,540	N/A	N/A	N/A	>48	>240	>238
	Hanford	Interval I-003	90.0 – 92.5	>15	>77	>74	4,240	15,600	9,330	N/A	N/A	N/A	>348	>1,740	2,620	N/A	N/A	N/A	>14	>71	>69
	Hanford	Interval I-004	92.5 – 95.0	>14	>73	>70	>2,250	14,100	14,100	N/A	N/A	N/A	>316	>1,580	>1,580	N/A	N/A	N/A	>12	>64	>62
	Hanford	Interval I-005	95.0 – 97.5	>14	>75	>72	2,460	9,430	11,300	N/A	N/A	N/A	492	>3,450	>3,440	N/A	N/A	N/A	>13	>68	>65
	Hanford	Interval I-006	97.8 – 100.3	>11	>59	>56	>1,920	16,000	16,000	N/A	N/A	N/A	145	>1,020	>1,020	N/A	N/A	N/A	>9	>50	>47
	Hanford	Interval I-007	100.4 – 102.9	>13	>65	>63	3,830	9,970	12,500	N/A	N/A	N/A	>549	>2,750	>2,740	N/A	N/A	N/A	>11	>58	>56
	Hanford	Interval I-008	102.3 – 104.8	>11	>56	>53	4,700	11,800	15,700	N/A	N/A	N/A	>198	>993	>990	N/A	N/A	N/A	>12	>64	>61
	Hanford	Interval I-010	104.5 – 107.0	>10	>51	>49	>1,860	11,600	11,600	N/A	N/A	N/A	>389	>1,950	>1,940	N/A	N/A	N/A	>11	>55	>53
	Hanford	Interval I-012	110.8 – 113.3	>16	>82	>80	2,850	12,100	16,200	N/A	N/A	N/A	>386	>1,930	>1,930	N/A	N/A	N/A	>12	>65	>62

Table 5-3. Calculated Soil-Water Partition Coefficients (K_d)

Waste Site	Formation	Interval	Depth (ft bgs)	Arsenic K_d (L/kg)			Barium K_d (L/kg)			Cadmium K_d (L/kg)			Chromium K_d (L/kg)			Cr(VI) K_d (L/kg)			Lead K_d (L/kg)		
				1:1	1:2.5	1:5	1:1	1:2.5	1:5	1:1	1:2.5	1:5	1:1	1:2.5	1:5	1:1	1:2.5	1:5	1:1	1:2.5	1:5
Borehole C7842	Hanford	Interval I-001	19.8 – 22.3	>35	>33	>31	4,220	7,590	>15,200	>5	>4	>1	>680	>679	1,890	N/A	N/A	N/A	>68	>66	>64
	Hanford	Interval I-003	29.8 – 32.3	>24	>27	>25	2,040	5,720	7,800	>3	>2	>0	>771	>924	>921	N/A	N/A	N/A	>66	>78	>76
	Hanford	Interval I-004	32.7 – 35.2	>22	>26	>23	1,500	2,560	4,930	>2	>2	>0	>841	4,210	>1,000	N/A	N/A	N/A	>68	>80	>78
	Hanford	Interval I-005	34.8 – 37.3	>18	>20	>17	1,980	5,770	9,880	>3	>2	>0	>601	>720	>717	N/A	N/A	N/A	>57	>67	>64
	Hanford	Interval I-006	37.3 – 39.8	>17	>19	>17	4,390	8,780	17,600	>2	>1	>0	>906	>1,090	>1,080	N/A	N/A	N/A	>65	>76	>74
	Hanford	Interval I-008	44.8 – 47.3	>14	>15	>12	5,260	7,370	7,360	N/A	N/A	N/A	>862	>1,030	>1,030	N/A	N/A	N/A	>46	>54	>52
	Hanford	Interval I-009	42.3 – 44.8	>21	333	>107	5,200	11,700	23,400	>2	>11	>9	>1,140	>5,720	>5,720	N/A	N/A	N/A	>61	1,030	>305
	Hanford	Interval I-010	50.3 – 52.8	>9	>9	>7	2,940	4,780	5,460	>2	>1	>0	>4,280	>5,140	>5,130	N/A	N/A	N/A	>137	>163	>160
Borehole C7843 (199-B3-52)	Hanford	Interval I-002	15.5 – 18.0	>20	>15	>12	666	1,180	1,200	>5	>3	>0	1,680	>447	>445	N/A	N/A	N/A	>64	>51	>49
	Ringold unit E	Interval I-004	29.9 – 32.4	>11	>21	>18	>9,810	24,500	>19,600	>9,810	>2	>0	1,580	3,620	4,220	N/A	N/A	N/A	69	208	>58
	Ringold unit E	Interval I-005	34.9 – 37.4	>26	>25	>22	18,600	37,200	>22,300	>4	>2	>0	8,200	5,220	9,560	N/A	N/A	N/A	173	180	>64
	Ringold unit E	Interval I-006	37.8 – 40.3	>39	>38	>35	14,600	116,500	>23,300	>4	>3	>0	6,780	7,630	6,780	N/A	N/A	N/A	169	189	165
	Ringold unit E	Interval I-007	39.7 – 42.4	>22	>20	>18	56,660	8,800	15,800	>5	>3	>1	>1,900	>1,900	>1,900	>11	>9	>7	>56	>55	>52
	Ringold unit E	Interval I-008	42.5 – 45.3	>25	>24	>21	5,130	9,530	33,400	>4	>3	>0	>920	>919	>916	N/A	N/A	N/A	>105	>104	>101
	Ringold unit E	Interval I-009	45.0 – 47.5	>23	>22	>19	6,840	10,300	13,700	>3	>2	>0	>854	>853	>850	N/A	N/A	N/A	>62	>61	>58
	Ringold unit E	Interval I-010	47.2 – 49.7	>26	>24	>22	4,930	9,160	9,160	>3	>2	>0	>1,030	>1,030	>1,030	N/A	N/A	N/A	>113	>111	>109
	Ringold unit E	Interval I-012	54.9 – 57.2	>24	>22	>118	3,820	7,150	9,540	>3	>1	>15	>781	>780	>3,900	N/A	N/A	N/A	>52	>51	>262
Borehole C7844	Hanford	Interval I-001	15.2 – 17.7	>20	>103	>100	1,820	4,540	7,270	>22	>114	>112	>477	>2,390	>2,380	N/A	N/A	N/A	>72	>365	>362
	Hanford	Interval I-002	19.5 – 22.0	>20	>100	>98	2,740	5,020	8,610	>9	>48	>46	>666	>3,330	>3,330	N/A	N/A	N/A	>27	>140	>137
	Hanford	Interval I-003	25.0 – 27.0	>9	>47	>44	2,100	4,900	2,940	N/A	N/A	N/A	>268	>1,340	>1,340	N/A	N/A	N/A	>19	>97	>95
	Hanford	Interval I-004	29.2 – 31.7	>9	>49	>47	1,830	4,390	9,410	N/A	N/A	N/A	>353	>1,770	>1,760	N/A	N/A	N/A	>25	>126	>123
	Hanford	Interval I-005	19.5 – 22.0	>10	>41	>49	2,410	5,540	9,230	N/A	N/A	N/A	>386	>1,930	>1,930	N/A	N/A	N/A	>29	>147	>145
	Hanford	Interval I-006	39.5 – 42.0	>10	>51	>48	3,710	5,930	14,800	N/A	N/A	N/A	>2,530	>12,600	>12,600	N/A	N/A	N/A	>23	>119	>116
	Hanford	Interval I-007	46.1 – 48.6	>10	>52	>50	>2,100	17,500	>10,500	N/A	N/A	N/A	>1,210	>6,030	>6,030	N/A	N/A	N/A	>25	>128	>126
	Hanford	Interval I-008	50.5 – 53.0	>12	>61	>58	>3,020	>15,100	>15,100	>4	>22	>19	>297	>1,490	>1,480	N/A	N/A	N/A	>24	>124	>122
	Hanford	Interval I-009	56.0 – 58.5	>16	>81	>78	1,490	4,910	11,500	N/A	N/A	N/A	>1,390	>6,940	>6,940	N/A	N/A	N/A	>31	>155	>153
	Hanford	Interval I-010	57.3 – 59.8	>10	>50	>48	>2,660	5,540	11,100	>3	>20	>17	>3,090	>15,400	>15,400	N/A	N/A	N/A	>28	>141	>138
	Hanford	Interval I-011	60.5 – 63.0	>10	>53	>50	>3,700	23,100	46,300	>5	>30	>27	>2,320	>11,600	>11,600	N/A	N/A	N/A	>39	>200	>197
	Hanford	Interval I-012	63.0 – 65.5	>9	>48	>45	6,790	13,600	15,500	>6	>33	>31	>992	>4,960	>4,960	N/A	N/A	N/A	>36	>182	>179
	Hanford	Interval I-015	65.5 – 68.0	>18	>92	>89	>4,020	<20,100	<20,100	N/A	N/A	N/A	>2,350	>11,700	>11,700	N/A	N/A	N/A	>28	>143	>140

Table 5-3. Calculated Soil-Water Partition Coefficients (K_d)

Waste Site	Formation	Interval	Depth (ft bgs)	Arsenic K_d (L/kg)			Barium K_d (L/kg)			Cadmium K_d (L/kg)			Chromium K_d (L/kg)			Cr(VI) K_d (L/kg)			Lead K_d (L/kg)		
				1:1	1:2.5	1:5	1:1	1:2.5	1:5	1:1	1:2.5	1:5	1:1	1:2.5	1:5	1:1	1:2.5	1:5	1:1	1:2.5	1:5
	Hanford	Interval I-016	71.6 – 73.1	N/A	N/A	N/A	1,850	17,500	23,400	N/A	N/A	N/A	>2,990	>14,900	>14,900	N/A	N/A	N/A	>24	>123	>120
Borehole C7845	Hanford	Interval I-003	31.1 – 32.9	>19	>17	>15	13,500	18,000	27,100	>7	>6	>3	>208	>206	>204	N/A	N/A	N/A	>49	>48	>45
	Hanford	Interval I-004	36.0 – 37.5	>19	>18	>15	27,100	13,600	1,800	>5	>4	>1	>415	>413	>411	N/A	N/A	N/A	>42	>41	>38
	Hanford	Interval I-006	45.8 – 47.2	>15	>79	>76	7,380	12,300	14,800	>8	>44	>41	>2,370	>11,800	>11,800	N/A	N/A	N/A	>44	447	556
	Hanford	Interval I-007	50.6 – 53.1	>21	>20	>17	11,200	18,700	2,550	>8	>6	>4	>731	>730	>727	N/A	N/A	N/A	>41	>39	>37
	Hanford	Interval I-008	56.0 – 58.5	>15	>17	>14	8,370	8,360	14,600	>5	>4	>2	>157	>187	>185	N/A	N/A	N/A	>33	>38	>35
	Hanford	Interval I-009	51.5 – 63.5	>17	>19	>17	12,400	18,600	24,800	>5	>5	>2	>582	>698	>695	N/A	N/A	N/A	>43	>50	>47
	Hanford	Interval I-010	63.9 – 66.4	>22	>20	>18	4,350	16,300	65,200	>7	>5	>3	>2,380	>2,380	>2,380	N/A	N/A	N/A	>92	>91	>88
	Hanford	Interval I-011	66.4 – 68.9	106	128	>44	12,700	15,200	25,300	>8	>6	>4	>759	>758	>755	N/A	N/A	N/A	>61	>60	>57
	Hanford	Interval I-012	68.4 – 70.9	>28	>27	>24	6,660	21,600	21,600	>8	>6	>4	>893	>892	>889	N/A	N/A	N/A	>65	>63	>61
	Hanford	Interval I-015	71.0 – 73.5	>19	>98	>96	>2,990	24,900	24,900	>4	>23	>20	>1,480	>7,410	>7,400	>18	>16	>14	>21	>110	>107
	Hanford	Interval I-016	76.4 – 78.9	>54	>53	>50	11,900	19,900	19,900	>19	>18	>15	>4,460	>4,460	>4,450	N/A	N/A	N/A	>124	>123	>120
Borehole C7846 199-B4-15)	Hanford	Interval I-001	27.9 – 30.4	>26	>25	>22	5,140	9,640	12,900	>13	>11	>9	>1,760	>1,760	>1,760	N/A	N/A	N/A	>94	>93	>90
	Hanford	Interval I-002	32.8 – 35.3	>18	>17	>14	6,150	14,300	21,500	>10	>8	>6	21,000	>5,870	>5,860	N/A	N/A	N/A	>57	>55	>53
	Hanford	Interval I-003	38.0 – 40.5	>20	>15	>16	7,980	16,000	>12,800	>5	>2	>1	>536	>445	>532	N/A	N/A	N/A	>46	>36	>42
	Hanford	Interval I-004	42.8 – 45.3	>14	>13	>10	1,430	1,830	1,900	>5	>3	>1	>503	>502	>499	N/A	N/A	N/A	>41	>40	>37
	Hanford	Interval I-005	47.8 – 50.6	N/A	>15	>12	N/A	7,733	27,100	N/A	>3	>0	N/A	>278	>276	N/A	N/A	N/A	N/A	>38	>38
	Hanford	Interval I-006	53.1 – 55.6	>20	>19	>16	1,530	1,850	2,070	>4	>3	>0	>665	>664	>661	N/A	N/A	N/A	>52	>50	>48
	Hanford	Interval I-007	58.2 – 60.7	>19	>18	>15	5,430	8,530	59,700	>6	>4	>2	>212	>211	>208	N/A	N/A	N/A	>36	>35	>32
	Hanford	Interval I-008	63.2 – 65.7	>21	>19	>17	1,720	11,500	17,200	>6	>5	>2	>478	>476	>474	N/A	N/A	N/A	>77	>76	>73
	Hanford	Interval I-009	68.0 – 70.5	>20	>19	>16	5,700	34,200	17,100	>6	>5	>2	5,000	>998	>995	N/A	N/A	N/A	>64	>63	>60
	Hanford	Interval I-010	72.9 – 75.4	>18	>16	>14	9,540	8,350	16,700	>6	>5	>2	>901	>900	>897	N/A	N/A	N/A	>51	>50	>47
	Hanford	Interval I-011	75.4 – 77.9	>7	>13	>10	3,580	7,160	8,600	>2	>4	>1	>291	>581	>578	N/A	N/A	N/A	>19	>37	>35
	Hanford	Interval I-015	80.2 – 82.8	>22	>21	>18	6,640	14,400	14,400	>4	>2	>0	>372	>371	>368	N/A	N/A	N/A	>47	>46	>43
Borehole C7847	Hanford	Interval I-001	0.0 – 2.5	>26	>24	>22	5,320	8,870	13,300	>5	>3	>1	>343	>341	>339	N/A	N/A	N/A	>78	>76	>74
	Hanford	Interval I-002	5.0 – 7.5	>45	>43	>41	6,800	13,600	13,600	>6	>5	>2	1,740	>1,320	>1,320	N/A	N/A	N/A	>96	>95	>92
	Hanford	Interval I-003	10.0 – 12.5	>26	211	306	>2,980	14,900	18,600	>7	>36	>33	>816	>4,080	>4,080	N/A	N/A	N/A	>68	>345	>342
	Hanford	Interval I-004	12.5 – 13.4	>30	>152	458	>3,120	15,600	26,000	>9	>49	>47	>1,080	8,140	16,300	N/A	N/A	N/A	>103	>518	>515

Table 5-3. Calculated Soil-Water Partition Coefficients (K_d)

Waste Site	Formation	Interval	Depth (ft bgs)	Arsenic K_d (L/kg)			Barium K_d (L/kg)			Cadmium K_d (L/kg)			Chromium K_d (L/kg)			Cr(VI) K_d (L/kg)			Lead K_d (L/kg)		
				1:1	1:2.5	1:5	1:1	1:2.5	1:5	1:1	1:2.5	1:5	1:1	1:2.5	1:5	1:1	1:2.5	1:5	1:1	1:2.5	1:5
Borehole C7849	Hanford	Interval I-001	15.1 – 17.6	>24	245	332	>2,320	11,600	19,400	>6	>31	>29	>689	>3,440	>3,440	N/A	N/A	N/A	>28	>144	>141
	Hanford	Interval I-002	19.5 – 22.0	>160	2,100	2,010	6,650	8,310	8,310	>10	>52	>49	1,450	>1,320	5,810	N/A	N/A	N/A	>0	>4	>1
	Hanford	Interval I-003	24.2 – 26.7	>20	>18	>16	4,100	7,780	11,100	>7	>5	>3	>1,290	>1,290	>1,280	N/A	N/A	N/A	>44	>42	>40
	Hanford	Interval I-004	30.2 – 32.7	>21	>19	>17	1,720	7,860	10,100	>7	>5	>3	>893	>892	>889	N/A	N/A	N/A	>46	>45	>42
	Hanford	Interval I-005	34.7 – 37.2	>33	>32	>29	11,000	21,900	3,650	>3	>2	>0	>660	>659	>656	N/A	N/A	N/A	>171	>170	>167
	Hanford	Interval I-006	40.6 – 43.1	>24	>22	>20	5,550	18,500	18,500	>3	>1	>0	>402	>401	>398	N/A	N/A	N/A	>82	>81	>78
	Hanford	Interval I-007	45.3 – 47.8	>17	>16	>13	7,610	26,600	>10,700	>3	>2	>0	>264	>263	>260	N/A	N/A	N/A	>55	>53	>51
	Hanford	Interval I-008	49.5 – 52.0	>17	>15	>13	29,500	>11,800	>11,800	>4	>2	>0	>332	>330	>328	N/A	N/A	N/A	>85	>84	>81
	Hanford	Interval I-009	54.5 – 57.0	>14	>13	>10	19,700	>11,800	>11,800	>4	>2	>0	>321	>319	>317	N/A	N/A	N/A	>62	>61	>58
	Hanford	Interval I-010	60.6 – 63.1	>22	>20	>18	4,240	14,800	29,700	>5	>3	>1	>310	>309	>306	N/A	N/A	N/A	>59	>57	>55
	Hanford	Interval I-011	65.2 – 67.6	>24	>23	>20	14,400	28,800	11,500	>4	>2	>0	>331	>330	>327	N/A	N/A	N/A	>171	>169	>167
	Hanford	Interval I-012	70.6 – 73.1	>22	>114	>111	12,400	31,100	31,100	>4	>22	>20	>292	>1,460	>1,460	N/A	N/A	N/A	>344	>1,720	>1,720
	Hanford	Interval I-013	75.2 – 77.7	>20	>103	391	7,390	14,800	24,600	>3	>20	>17	>1,190	>5,970	<5,970	N/A	N/A	N/A	>199	>997	2,490
	Hanford	Interval I-014	80.1 – 82.6	>19	>95	>93	4,960	14,900	20,000	>4	>21	>18	<267	>1,340	>1,330	N/A	N/A	N/A	>267	>1,340	>1,340
	Hanford	Interval I-015	84.5 – 87.0	>21	>105	>103	19,200	14,400	28,800	>3	>18	>16	1270	>1,530	>1,530	N/A	N/A	N/A	>305	>1,520	>1,520
	Hanford	Interval I-016	87.5 – 90.0	>13	>66	>63	>2,020	25,300	25,300	>3	>17	>15	>345	>1,730	>1,730	N/A	N/A	N/A	>46	>234	>232
	Hanford	Interval I-017	90.2 – 92.7	>14	>71	>68	4,600	13,800	18,400	>3	>18	>15	>489	>2,450	>2,440	N/A	N/A	N/A	>171	>859	>857
	Hanford	Interval I-018	92.0 – 94.5	>19	>18	>15	7,560	10,600	17,600	N/A	N/A	N/A	927	>294	>292	N/A	N/A	N/A	>43	>42	>39
	Hanford	Interval I-019	94.6 – 97.1	>17	>15	>13	9,380	23,500	46,900	>3	>1	>0	>384	1,200	>380	N/A	N/A	N/A	>38	>36	>34
	Hanford	Interval I-020	97.0 – 99.5	>16	>14	>12	9,380	23,500	>9,380	>3	>1	>0	>162	>160	>158	N/A	N/A	N/A	>34	>32	>30
	Hanford	Interval I-026	99.7 – 102.2	>8	>42	>39	>1,770	>8,860	>8,860	N/A	N/A	N/A	>264	>1,320	>1,320	N/A	N/A	N/A	>12	>63	>60
Borehole C8239	Hanford	Interval I-001	0.0 – 2.5	>24	>124	>121	>3,240	11,600	16,200	N/A	N/A	N/A	>726	10,900	>3,630	N/A	N/A	N/A	>54	>275	>272
	Hanford	Interval I-002	4.8 – 7.3	>31	>156	>153	>3,550	22,200	22,200	N/A	N/A	N/A	>802	12,000	>4,010	N/A	N/A	N/A	>59	>298	>295
	Hanford	Interval I-003	10.0 – 12.5	>53	571	798	6,330	12,700	19,000	N/A	N/A	N/A	>420	>2,100	>2,100	N/A	N/A	N/A	>140	2,350	>701
	Hanford	Interval I-004	12.4 – 14.9	>57	868	>285	5,220	14,600	18,300	N/A	N/A	N/A	>455	>2,280	>2,270	N/A	N/A	N/A	>96	>482	>479
	Hanford	Interval I-005	17.4 – 19.9	>43	>217	>215	14,200	17,800	23,700	>7	>37	>35	>632	>3,160	>3,160	N/A	N/A	N/A	>179	>898	3,000
	Hanford	Interval I-006	22.3 – 24.8	>15	>77	>75	>2,680	22,400	>13,400	>6	>34	>31	>772	>3,860	11,600	N/A	N/A	N/A	>36	>184	>181
	Hanford	Interval I-007	27.7 – 30.2	>13	>65	>63	>2,220	11,100	>11,100	>5	>28	>25	>418	>2,090	>2,090	N/A	N/A	N/A	>24	>123	>121
	Hanford	Interval I-008	32.3 – 34.8	>21	>109	>107	7,190	19,200	28,800	N/A	N/A	N/A	>225	>1,130	>1,120	N/A	N/A	N/A	>60	>302	>300
	Hanford	Interval I-009	37.4 – 39.9	>34	>174	>171	6,640	9,300	23,200	N/A	N/A	N/A	>235	>1,180	>1,180	N/A	N/A	N/A	>42	>212	530

Table 5-3. Calculated Soil-Water Partition Coefficients (K_d)

Waste Site	Formation	Interval	Depth (ft bgs)	Arsenic K_d (L/kg)			Barium K_d (L/kg)			Cadmium K_d (L/kg)			Chromium K_d (L/kg)			Cr(VI) K_d (L/kg)			Lead K_d (L/kg)		
				1:1	1:2.5	1:5	1:1	1:2.5	1:5	1:1	1:2.5	1:5	1:1	1:2.5	1:5	1:1	1:2.5	1:5	1:1	1:2.5	1:5
	Hanford	Interval I-010	42.3 – 44.9	>35	>176	>174	5,640	12,400	20,700	N/A	N/A	N/A	>288	>1,440	>1,440	N/A	N/A	N/A	>50	629	837
	Hanford	Interval I-011	47.3 – 49.8	N/A	N/A	N/A	2,370	5,930	7,900	N/A	N/A	N/A	>558	>2,790	>2,790	N/A	N/A	N/A	>14	>73	>71
	Hanford	Interval I-012	52.2 – 54.7	>18	>90	>88	>1,890	9,450	7,870	N/A	N/A	N/A	>265	>1,320	>1,320	N/A	N/A	N/A	>21	>108	>105
	Hanford	Interval I-013	55.3 – 57.2	>21	>109	>107	>3,820	31,900	>19,100	N/A	N/A	N/A	>346	>1,730	>1,730	N/A	N/A	N/A	>17	>87	>85
	Hanford	Interval I-014	57.2 – 59.7	>6	>31	>28	1,810	5,090	7,000	N/A	N/A	N/A	>454	>2,270	>2,270	N/A	N/A	N/A	>21	>106	>104
	Hanford	Interval I-015	62.3 – 64.8	>20	>101	>98	>3,030	9,480	15,200	N/A	N/A	N/A	>824	>4,120	>4,120	N/A	N/A	N/A	>23	>116	>114
	Hanford	Interval I-016	65.0 – 67.5	>12	>62	>60	2,860	8,570	11,000	N/A	N/A	N/A	>4,480	>22,400	>22,400	N/A	N/A	N/A	>17	>86	>83
	Hanford	Interval I-017	67.3 – 69.8	>9	>48	>45	4,090	10,200	15,300	N/A	N/A	N/A	>1,910	9,570	>9,570	N/A	N/A	N/A	>18	>90	>89
	Hanford	Interval I-018	69.8 – 72.3	>19	>100	>97	6,460	11,800	17,800	N/A	N/A	N/A	>1,080	>5,410	>5,400	N/A	N/A	N/A	>51	>259	>257
	Hanford	Interval I-020	72.5 – 75.0	>11	>56	>53	2,650	7,580	13,300	N/A	N/A	N/A	>1,060	>5,320	>5,320	>20	>19	>16	>17	>87	>84
Test Pit 116-B-6B	Hanford	16.4 - 18.4 ft	16.4-18.4	>31	213	>153	>2,910	>14,600	>14,600	N/A	N/A	N/A	>501	>2,500	>2,500	N/A	N/A	N/A	>64	>320	>318
	Hanford	19.4 - 21.4 ft	19.4-21.4	>16	150	>82	>1,800	>8,990	>8,980	N/A	N/A	N/A	>20	3,000	>998	N/A	N/A	N/A	>11	>60	>57
Test Pit 116-B-9	Hanford	8 - 10 ft	8-10	>25	276	>125	>2,550	21,300	5,800	N/A	N/A	N/A	>743	856	11,200	N/A	N/A	N/A	>40	>201	>198
	Hanford	11 - 13 ft	11-13	>30	216	301	>2,880	14,400	>14,400	N/A	N/A	N/A	>830	>4,150	>4,150	N/A	N/A	N/A	>39	>198	>195
Test Pit 118-B-8:3	NA	Pipe Sediment	N/A	>24	>124	>121	4,600	6,440	7,430	>107	>539	>537	2,170	3,150	6,930	N/A	N/A	N/A	>385	>1,930	>1,920
	Hanford	N/A	23	>15	213	>74	>3,050	25,400	>15,300	N/A	N/A	N/A	>443	6,660	>2,220	N/A	N/A	N/A	>23	>118	>115
	Hanford	N/A	22	>12	140	>61	>2,530	21,100	21,100	N/A	N/A	N/A	>708	5,310	3,540	N/A	N/A	N/A	>30	>150	>148

Note: 1:1, 1:2.5, 1:5—ratios of soil mass (g) to leaching solution (mL).

N/A = At least one of the four replicate soil samples was below detection limit, so K_d was not calculated.

>-1 = Kd values with greater than (>) signs indicate that the constituent was below reporting limit in the leaching solution, and the minimum Kd was calculated using that reporting limit.

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In the leachate samples, most metals, except for barium, were infrequently detected. Cr(VI) was detected in only five (includes samples flagged with B) of the 1:1 batch leachate samples at concentrations between 0.01 and 0.03 mg/L and in four of the 1:2.5 and 1:5 dilution leachate samples. The relative vertical distribution of total chromium and Cr(VI) observed in the bulk soil samples is illustrated on Figures 5-5 through 5-12. Laboratory analysis results for the saturated soil samples are presented (*100-BC-5 Operable Unit, Batch Leach Analyses and Report for Sediments at RI/FS Wells C7508, C7783, C7784, C7785, and C7787* [PNNL-20352]). The calculation of K_d values for the 100-BC samples is presented in *100-BC Remedial Investigation Distribution Coefficient Calculations* (0100X-CA-V0058).

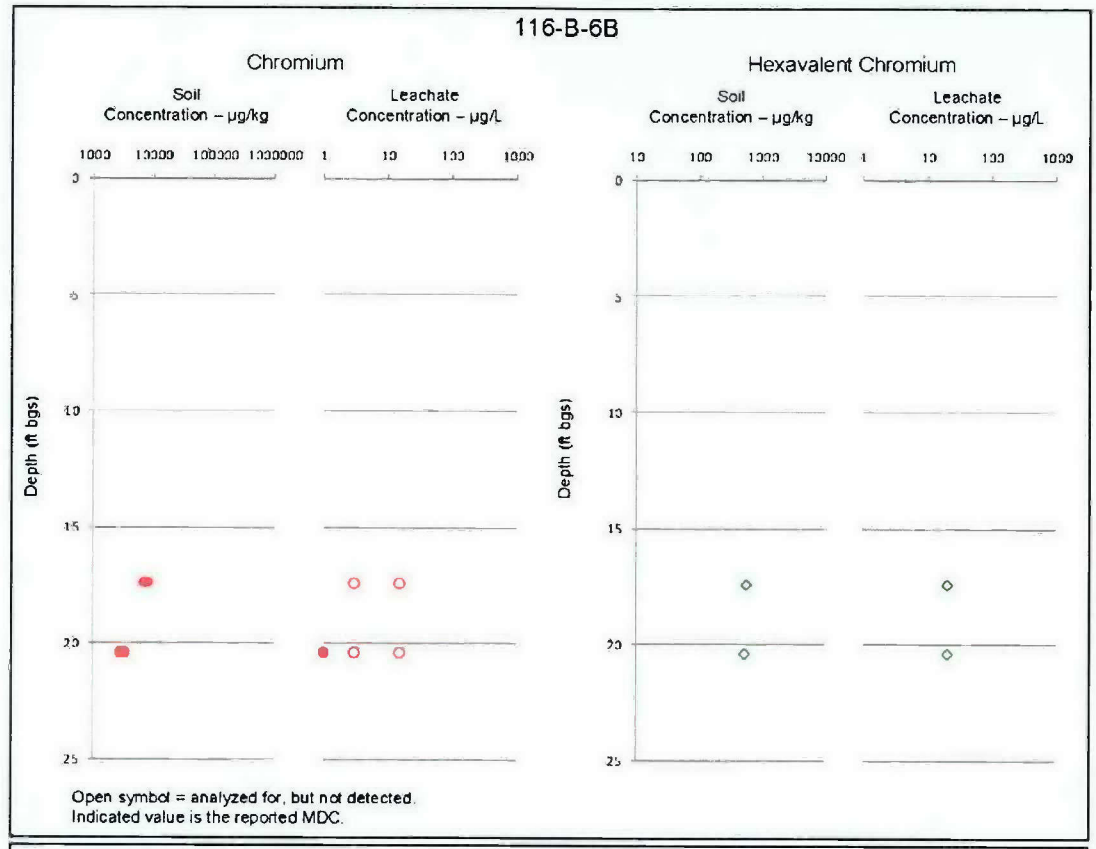
The calculated K_d value for Cr(VI) ranged from >7 to >22 suggesting that most chromium in soil is in trivalent (chromium(III)) form. Chromium(III) is relatively insoluble and more strongly bound to soil than Cr(VI). The calculated K_d vales for total chromium of 145 to 21,000 mL/g, with a median value of 2,580, are consistent with what is known about its subsurface behavior. Only 56 of the 132 samples could be quantified for total chromium K_d due to the low solubility of chromium(III) in the leachate.

Cr(VI) is generally presumed to have moderate sorption potential with nominal solubility in aqueous solutions of neutral pH. The infrequent and low concentrations observed in the soil and leachate samples make quantification of K_d highly uncertain in the batch leach test dataset. The low measured concentrations observed in the dataset, and as reported in other literature sources, suggest a K_d close to 0. It is important to note that the Cr(VI) sample extraction method used to prepare the solid soil samples for Cr(VI) analysis is intended to extract low water-solubility Cr(VI) compounds for measurement. Although mineralogical analysis to identify specific Cr(VI) compounds in the soil samples was not performed, some of these compounds (for example, potassium dichromate, lead chromate) likely can be found in 100-BC soil as a result of simple ionic reactions between the sodium dichromate in reactor cooling water and other naturally occurring metal ions. The batch leach solution used in this test is intended to approximate weakly acidic rainfall, which should solubilize Cr(VI) present in vadose zone soil under future expected land use conditions.

5.5.2 Uncertainty in Batch Leach Testing Results

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 used in calculating K_d. Specific areas of uncertainty identified during the data evaluation apply to the derivation of contaminant-specific K_d as described in *Evaluation of Hexavalent Chromium Leach Test Data Conducted in Vadose Zone Sediment Samples from the 100 Area* (ECF-HANFORD-11-0165) (Appendix F). The following general conditions 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 distribution coefficients
- Effects of coarse material (that is, gravel fraction) on the K_d in the geologic formation
- Potential dilution effects on the samples resulting from addition of potable water to boreholes during drilling



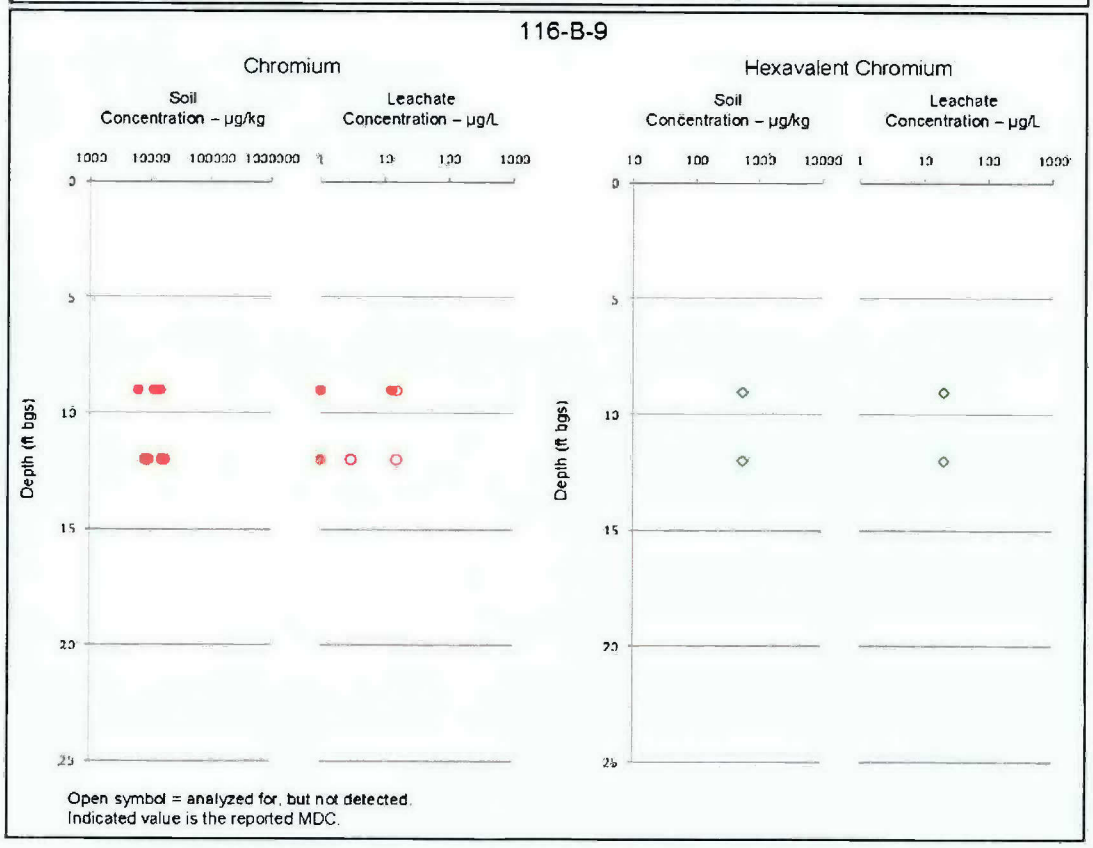
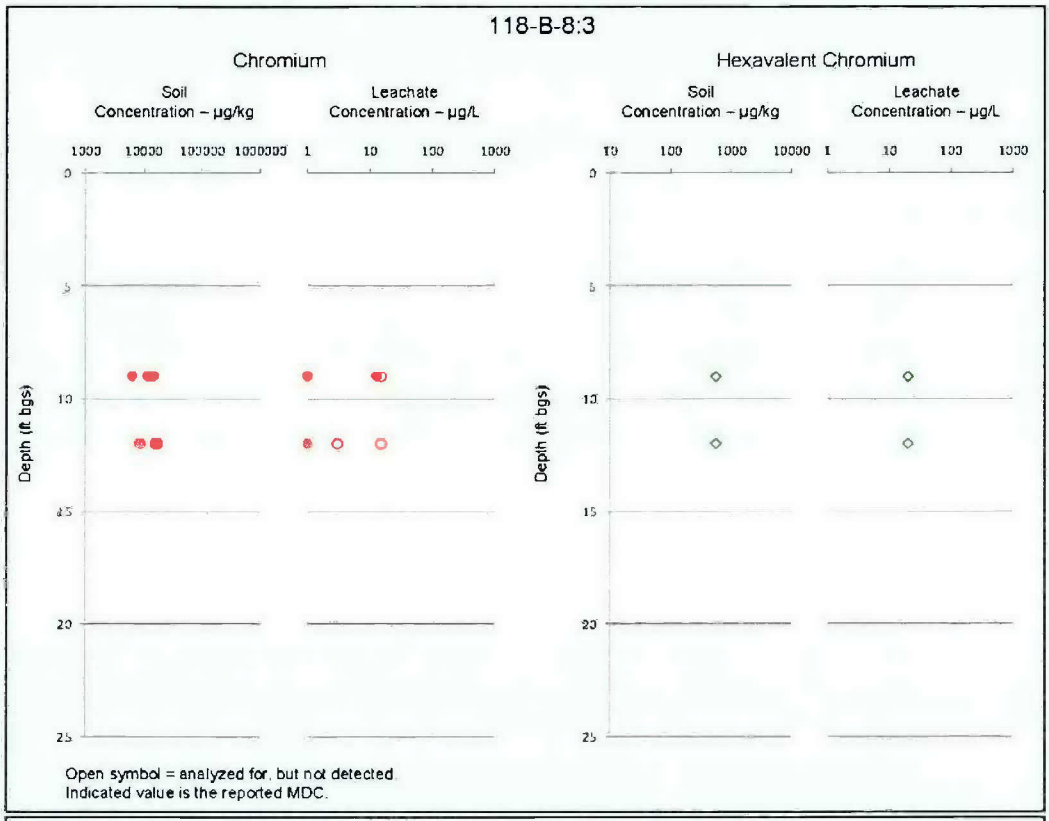


Figure 5-5. Soil Batch Leach Vertical Profile for Test Pits 116-B-6B and 116-B-9



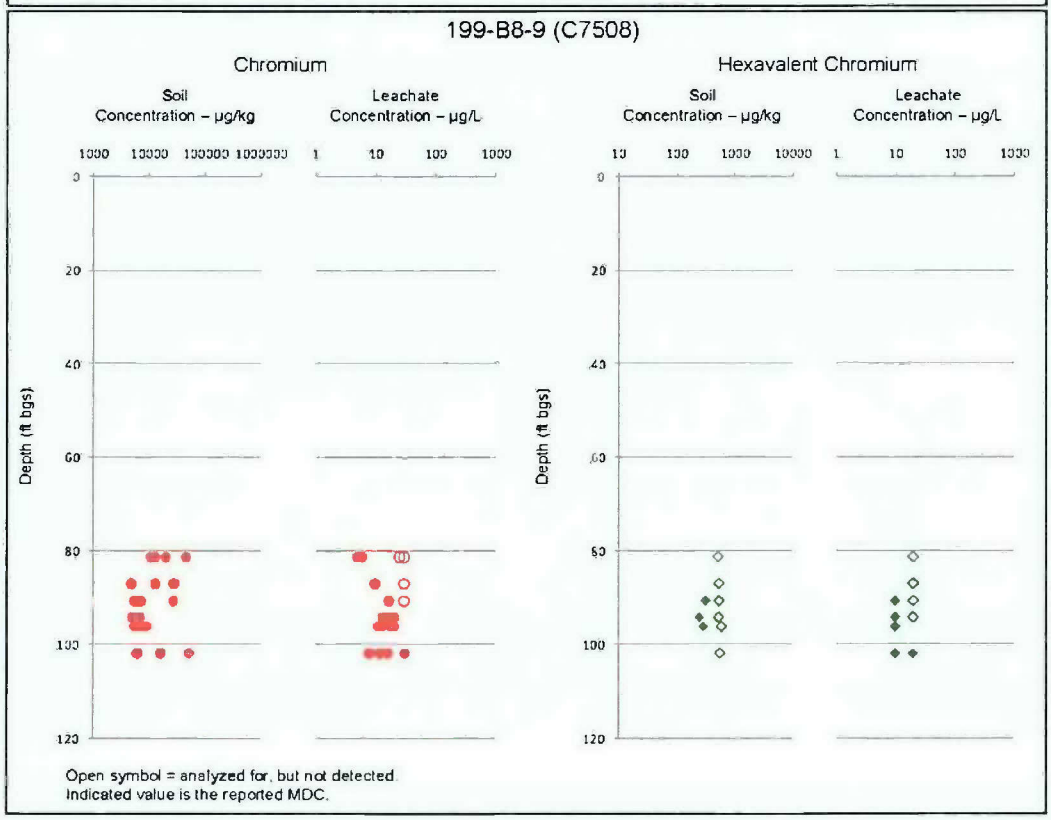


Figure 5-6. Soil Batch Leach Vertical Profile for Test Pit 118-B-8:3 and Well 199-B8-9 (C7508)

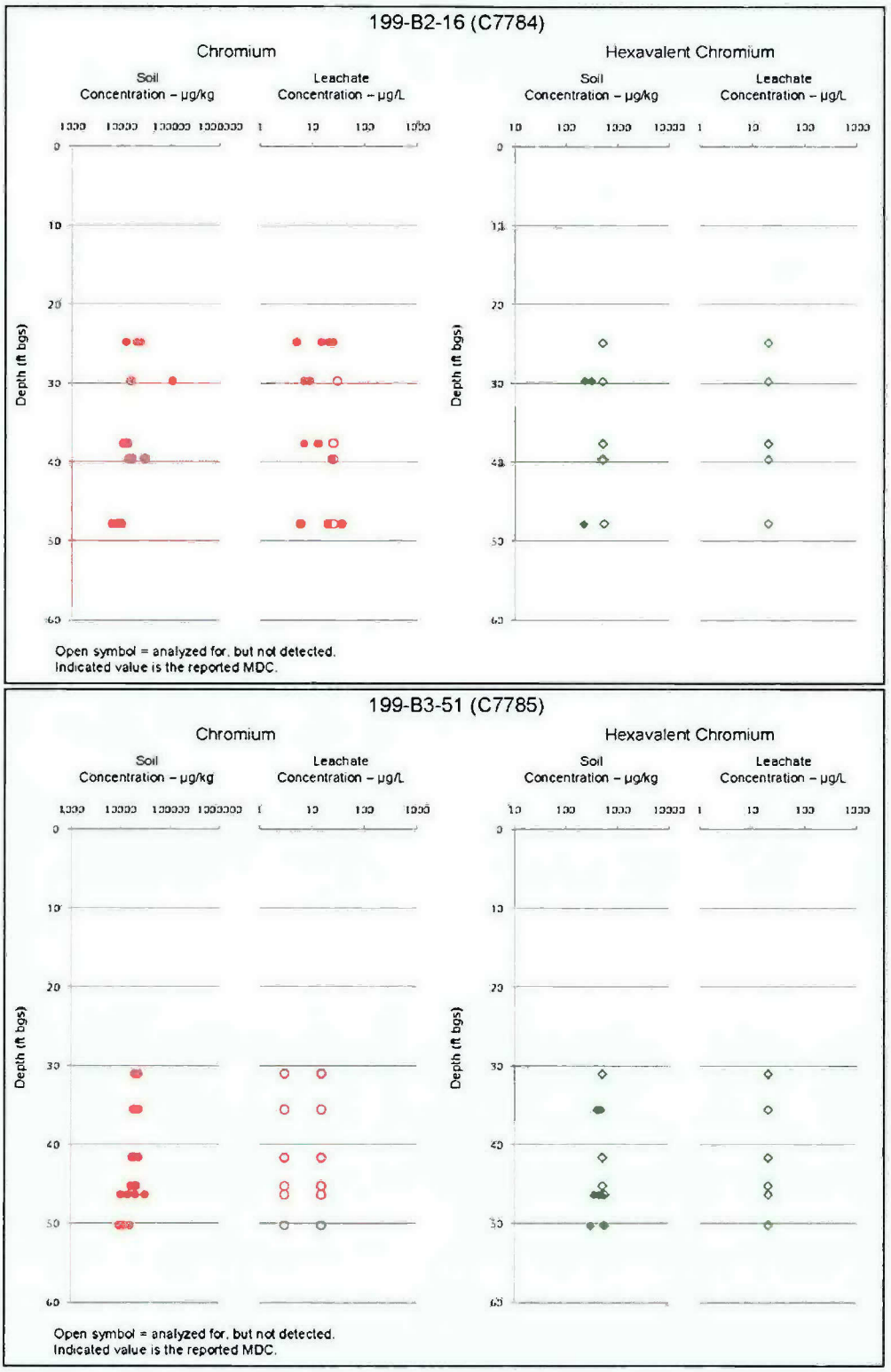


Figure 5-7. Soil Batch Leach Vertical Profile for Wells 199-B2-16 (C7784) and 199-B3-51 (C7785)

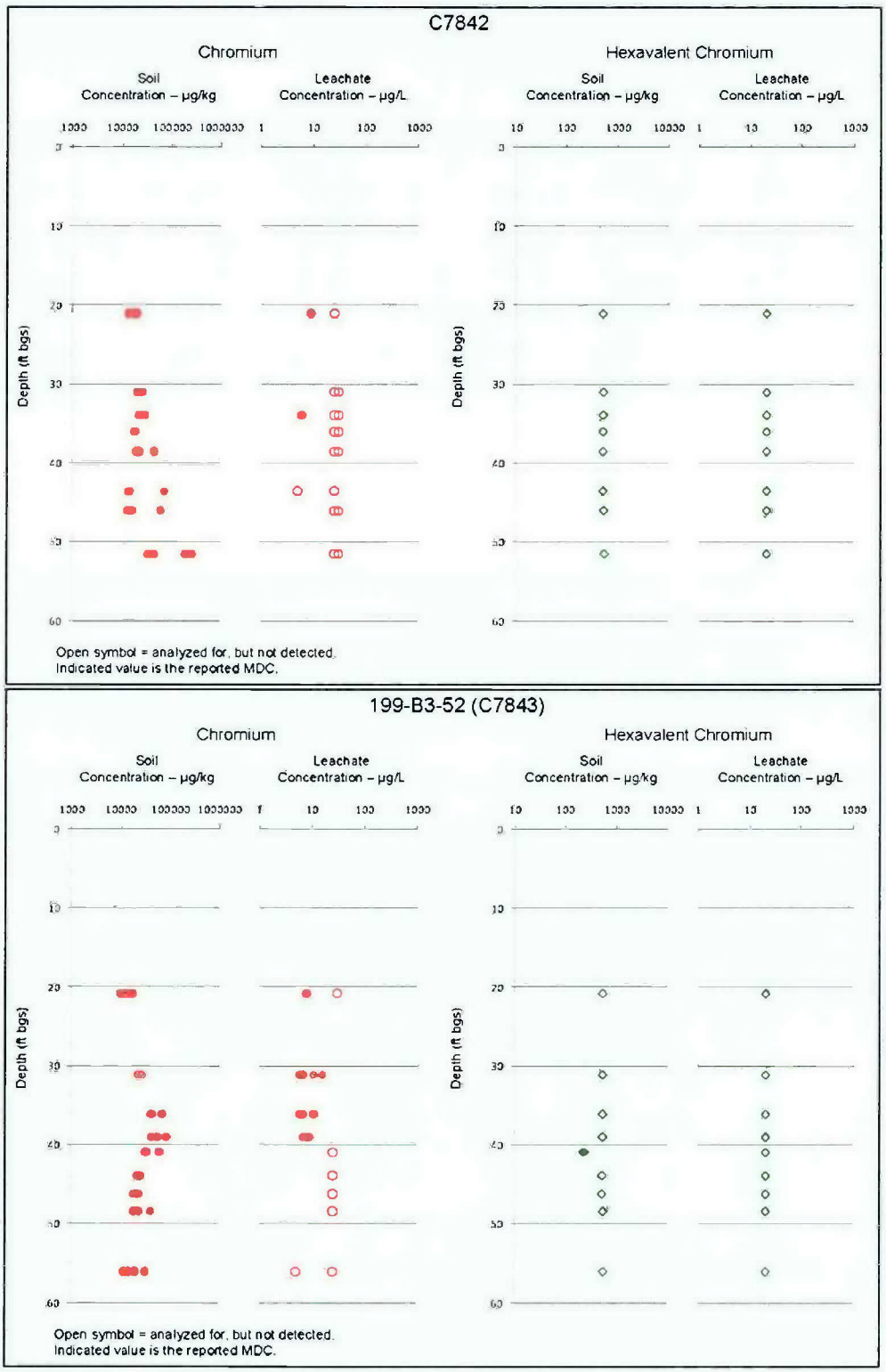
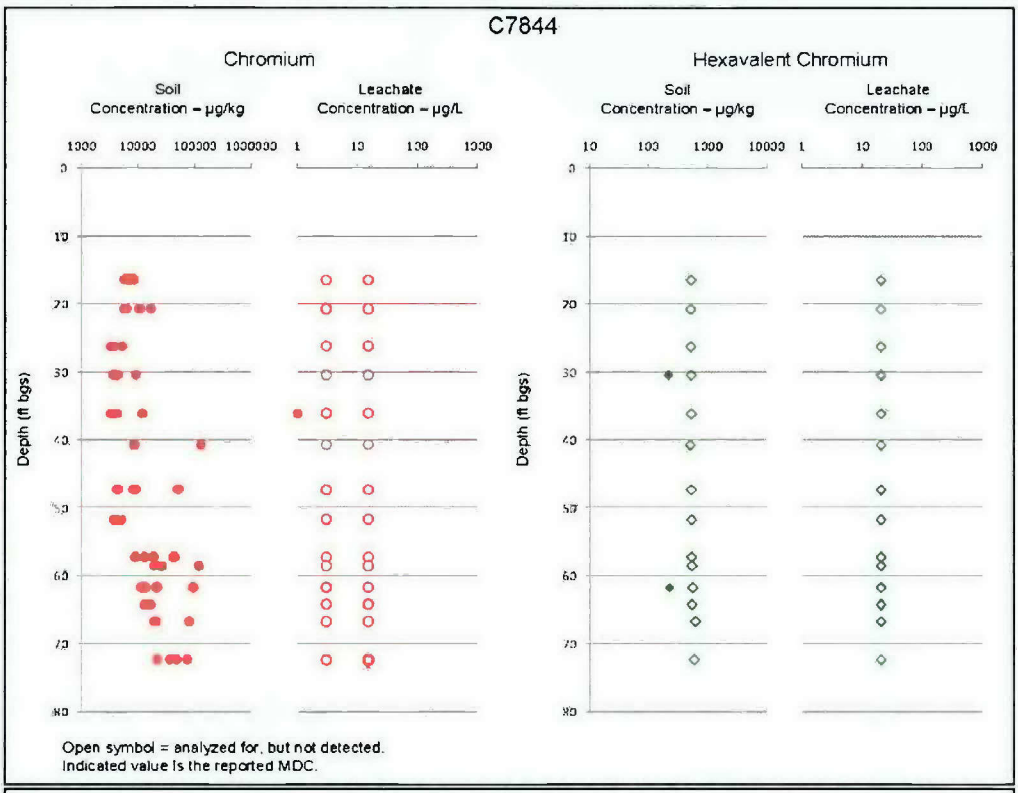


Figure 5-8. Soil Batch Leach Vertical Profile for Borehole C7842 and Well 199-B3-52 (C7843)



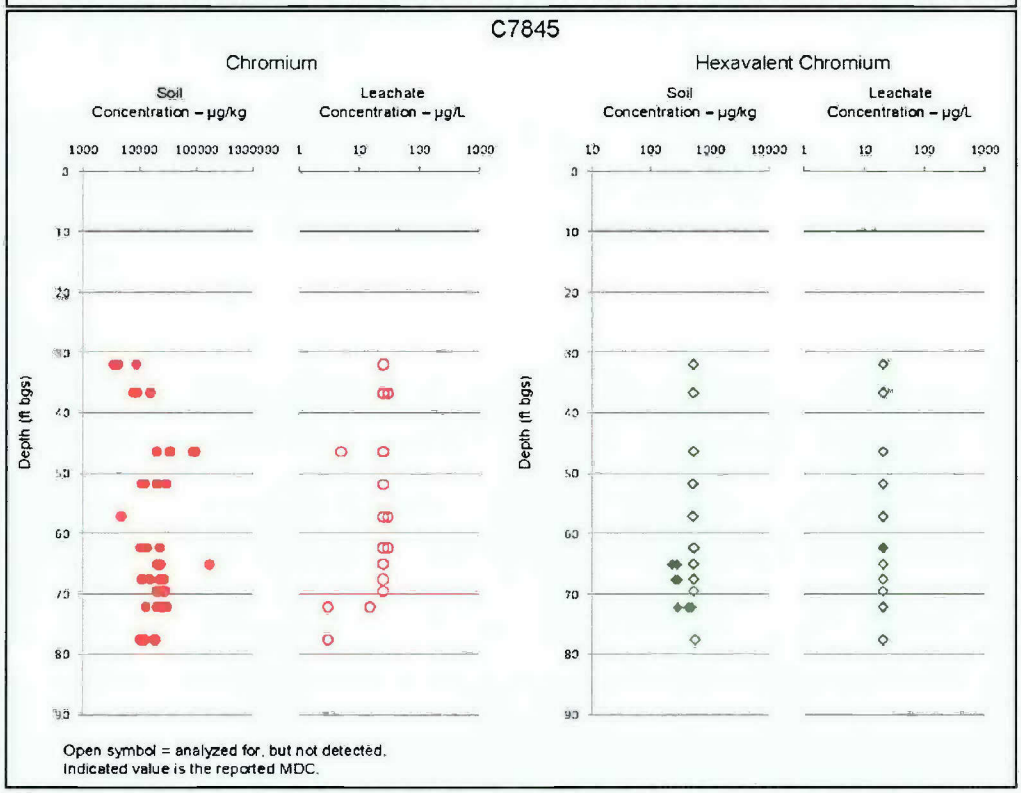


Figure 5-9. Soil Batch Leach Vertical Profile for Boreholes C7844 and C7845

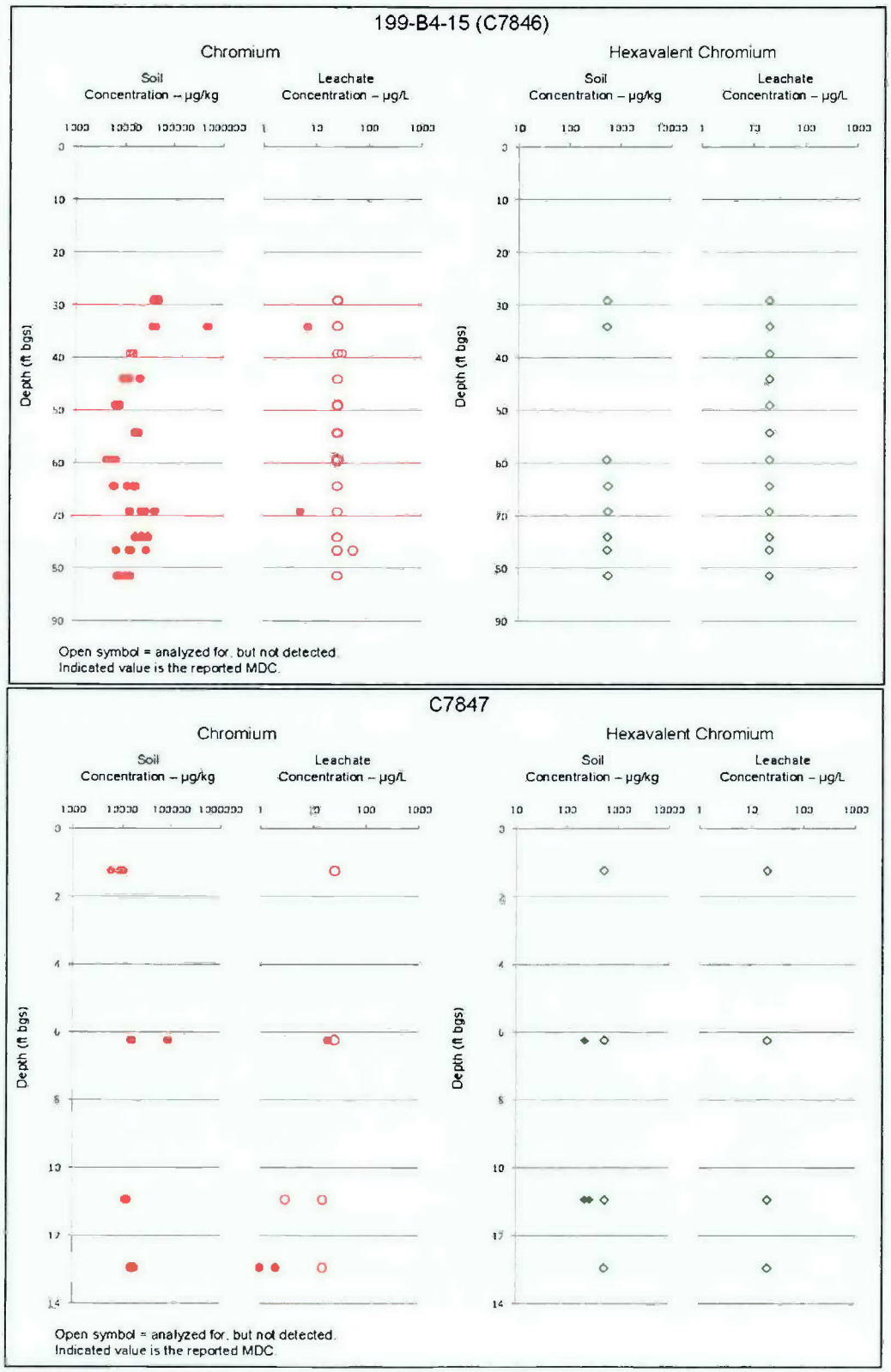


Figure 5-10. Soil Batch Leach Vertical Profile for Well 199-B4-15 (C7846) and Borehole C7847

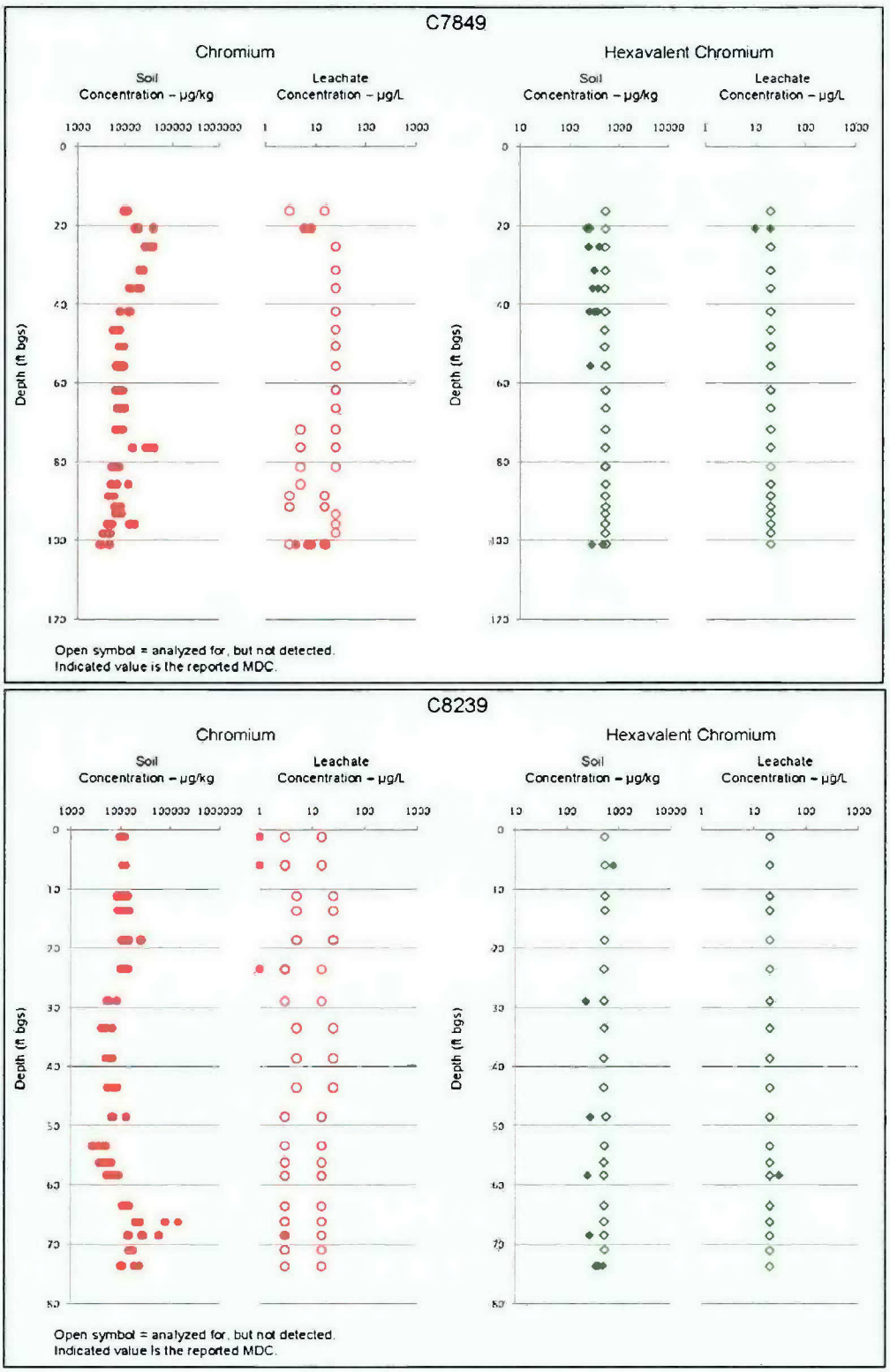


Figure 5-11. Soil Batch Leach Vertical Profile for Boreholes C7849 and C8239

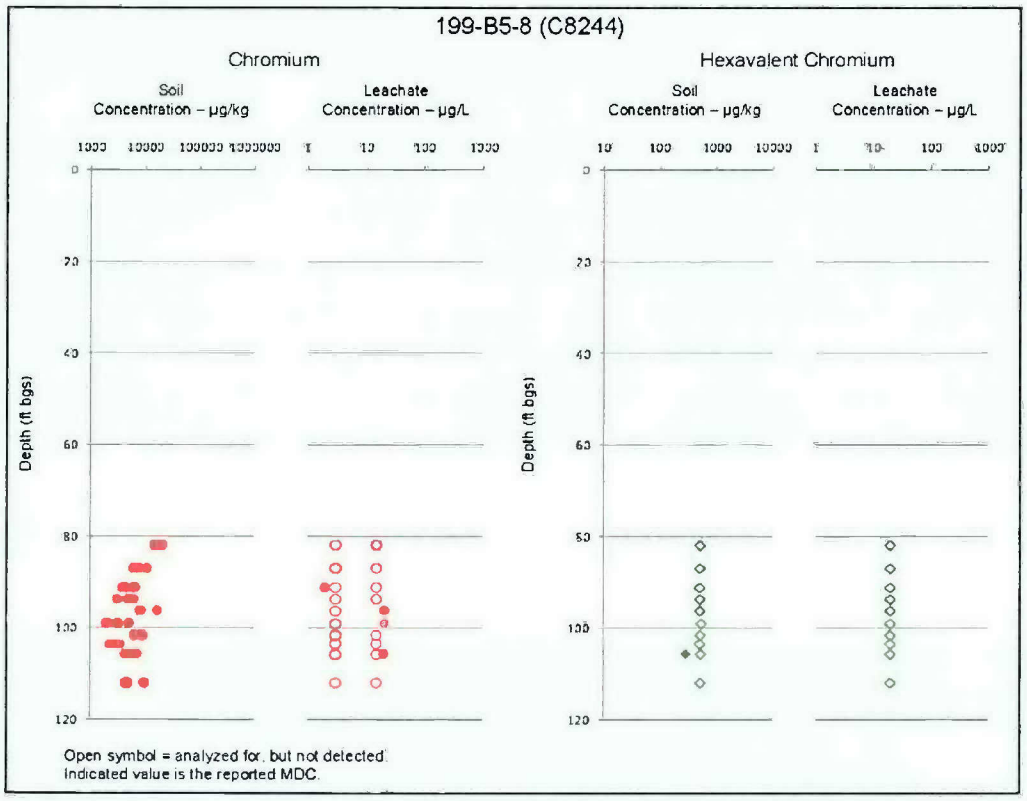


Figure 5-12. Soil Batch Leach Vertical Profile for Well 199-B5-8 (C8244)

5.5.3 Development of a Hexavalent Chromium K_d for Vadose Zone Simulations from Batch Leach Testing Results

The results of the batch leach testing for Cr(VI) were further evaluated to identify a single K_d value to represent Cr(VI) behavior in the vadose zone model used for SSL and PRG development.

The results of leach tests described in Calculations 0100K-CA-V0081 (100-K), 0100X-CA-V0058 (100-BC), 0100X-CA-V0059 (100-D/H), and 0100X-CA-V0060 (100-F) were analyzed to estimate a linear isotherm (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-BC SAP (DOE/RL-2009-44) to recommend a K_d value for use in the 100 Area. All methods used to calculate a K_d value were outlined in the SAP for each respective River Corridor OU. The objective for this evaluation was to recommend a single K_d value for use in the River Corridor, including 100-BC. Details of the evaluation, including a historical evaluation of 100-BC column leaching tests, is described in *Evaluation of Hexavalent Chromium Leach Test Data Conducted in Vadose Zone Sediment Samples from the 100 Area* (ECF-HANFORD-11-0165).

The large number of K_d measurements and the lack of calculated K_d correlation with possible explanatory parameters require use of a conservative value across the River Corridor for evaluation of future fate and transport of residual Cr(VI) after interim remedial actions have been implemented for waste sites in the vadose zone. The evaluation of K_d for the soil samples indicates that more than 90 percent of the values are higher than 1.2 mL/g and more than 95 percent of the values are higher than 0.65 mL/g. If the K_d values are adjusted for water used during the tests (normalizing the values to the smallest soil:water extract ratio), the 90th percentile K_d value drops to about 0.8 mL/g.

Based on the batch leach results for the 100 Area soil samples, a K_d value of 0.8 mL/g was used as a conservative estimate for calculating the Cr(VI) SSLs and PRGs in the vadose zone contaminant migration assessment. This value is subject to the uncertainties described in Section 5.5.2.

5.6 Vadose Zone Modeling Methods and Results

Vadose zone transport simulations were performed using the STOMP computer code (*STOMP: Subsurface Transport Over Multiple Phases Version 2.0: Theory Guide* [PNNL-12030]). The STOMP code was selected to perform the simulations because it can adequately simulate the vadose zone features, events, and processes relevant to the SSL and PRG calculations for the 100 Area while satisfying the criteria for numerical model code selection described in *Regulatory Basis and Implementation of a Graded Approach to Evaluation of Groundwater Protection* (DOE/RL-2011-50). The model development approach used to support this RI is 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 *STOMP 1-D Modeling for Determination of Soil Screening Levels and Preliminary Remediation Goals for 100 Area B and C Source Areas* (ECF-Hanford-12-0003) included in Appendix F.

A range of model input values (Table 5-4) were used for the simulations to account, to the extent possible, for the range of conditions observed or measured at representative locations within 100-BC. One-dimensional numerical models were constructed to represent the key facets of the conceptual model and were solved using STOMP. 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 simulations predict contaminant concentration and the time to reach the peak concentration for the recharge rates and sediment types, thicknesses, and properties appropriate to 100-BC.

Conceptually, the model simulation represents a column of sediments that comprise the vadose zone underlain by an aquifer. Recharge-driven flow moves downward through the vadose zone, where it encounters contaminated soil that releases soluble contaminants for transport to the underlying aquifer, across which a hydraulic gradient drives horizontal groundwater 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 thickness of 5 m (16.4 ft), so that contaminant concentrations in a monitoring well with a 5 m (16.4 ft) long screen interval are simulated. Within 100-BC (Figure 5-4), the vadose zone is composed of backfill, Hanford formation and Ringold Formation unit E sediments while the saturated zone is comprised of Hanford formation and Ringold Formation unit E sediments.

5.6.1 Initial Contaminant Distribution

Based on observations of contaminant distribution made using laboratory analysis of soil samples collected from RI borings and wells, the spatial distribution of contaminants was identified for use in the initial flow and transport simulations. Numerous contaminants were found to be distributed across the full vadose zone thickness while others exhibited limited vertical distribution. Therefore, contaminants were grouped into two categories. The first category included contaminants with low K_d (<2 mL/g) values while the second category included contaminants with high K_d (≥2 mL/g) values. The process used for identifying low K_d and high K_d range contaminants is presented in *Conceptual Basis for Distribution of Highly Sorbed Contaminants in 100 Areas Vadose Zone* (SGW-51818).

Table 5-4. 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-BC-1 and 100-BC-2 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
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)					
Hydraulic gradient (m/m)	0.0019				
Hydraulic Parameters					
	Vadose Zone			**Saturated Zone**	
	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	0.009	0.015	0.009	0.015

Table 5-4. 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-BC-1 and 100-BC-2 Operable Units

Input Parameter (units)	Input Parameter Value				
n van Genuchten water retention function exponential fitting parameter (dimensionless)	1.4	1.369	1.516	1.369	1.516
s_r residual saturation (dimensionless)	0.162	0.024	0.050	0.024	0.050
$K_{s,h}$ saturated horizontal hydraulic conductivity (m/d)	0.517	0.853	0.968	47.4	6.2
$K_{s,v}$ saturated vertical hydraulic conductivity (m/d)	0.517	0.0853	0.0968	4.74	0.62
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	1.91	1.90	1.91	1.90
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)				
α_T/α_L dispersivity anisotropy ratio (dimensionless)	Not applicable (one-dimensional model)				
K_d partition coefficient (mL/g)	27 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 simulated K_d values are: [0, 0.00001, 0.0001, 0.001, 0.01, 0.02, 0.04, 0.06, 0.08, 0.10, 0.20, 0.30, 0.40, 0.50, 0.60, 0.70, 0.80, 0.90, 1, 2, 4, 6, 8, 10, 12, 14, 16]. The specific contaminant K_d values for all contaminants evaluated (191 non-radionuclides and 28 radionuclides) are listed in ECF-Hanford-12-0003 (found in this Appendix) in Attachment A, Tables A-1, A-2, A-3, and Attachment B, Tables B-1, B-2, and B-3.				

Table 5-4. 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-BC-1 and 100-BC-2 Operable Units

Input Parameter (units)	Input Parameter Value
Sources for constitutive relationships: • 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." Details on the basis for all parameters in this table are found in this Appendix (*STOMP 1-D Modeling for Determination of Soil Screening Levels and Preliminary Remediation Goals for 100 Area B and C Source Areas* [ECF-Hanford-12-0003]).	

For low K_d contaminants, a uniform unit-source concentration of 1 mg/kg was applied across the entire vadose zone below the clean backfill interval to a depth 0.5 m (1.64 ft) above the water table (that is, the capillary fringe). This is referred to as the 100:0 initial source distribution. The unit-source concentration in the 0.5 m (1.64 ft) zone above the water table was not applied because capillary fringe and water table fluctuation effects across the periodically rewetted zone would result in boundary effects and extreme-concentration gradients. For the high K_d contaminants, a uniform concentration unit-source of 1 mg/kg was applied in the upper 70 percent of the vadose zone below the clean backfill interval. This is referred to as the 70:30 initial source distribution.

The use of a unit-concentration source, typically 1 mg/kg, is a common practice in the simulation of multi-contaminant transport in a system where transport is controlled by K_d. The transport simulation results for individual contaminants 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 for each. The SSL simulation uses a conservative recharge scenario based on an assumed irrigated agriculture land use whereas the PRG simulation uses a recharge scenario based on a recharge rate that accounts for re-establishment of a native plant population across the surface.

The initial simulation configuration assumes that interim remedial actions have been undertaken and that 4.6 m (15 ft) of clean backfill is present starting at the ground surface. This clean fill assumption is a default condition to the simulations and is not meant to represent conditions at any particular waste site, where interim action excavations may have proceeded to greater or lesser depths.

5.6.2 Simulation Duration

Per regulatory agreement, the STOMP simulations were limited to a maximum duration of 1,000 years. The peak contaminant concentration calculated within the 1,000-year simulation duration defines the SSL and PRG value.

The vadose zone unit-source peak concentration breakthrough curves for the simulated entry of contaminants into underlying groundwater for a range of K_d values in a typical soil column are shown on Figure 5-13. These breakthrough curves reveal that for $K_d < 2$ mL/g, peak concentration occurs within 1,000 years, while for $K_d \geq 2$ mL/g, the peak concentration occurs after 1,000 years at concentrations above 0.001 mg/m^3. This observation provided the basis for grouping contaminants into the low K_d (<2 mL/g) category for use with the 100:0 initial source distribution, and the second high K_d (≥2 mL/g) category for use with the 70:30 initial source distribution.

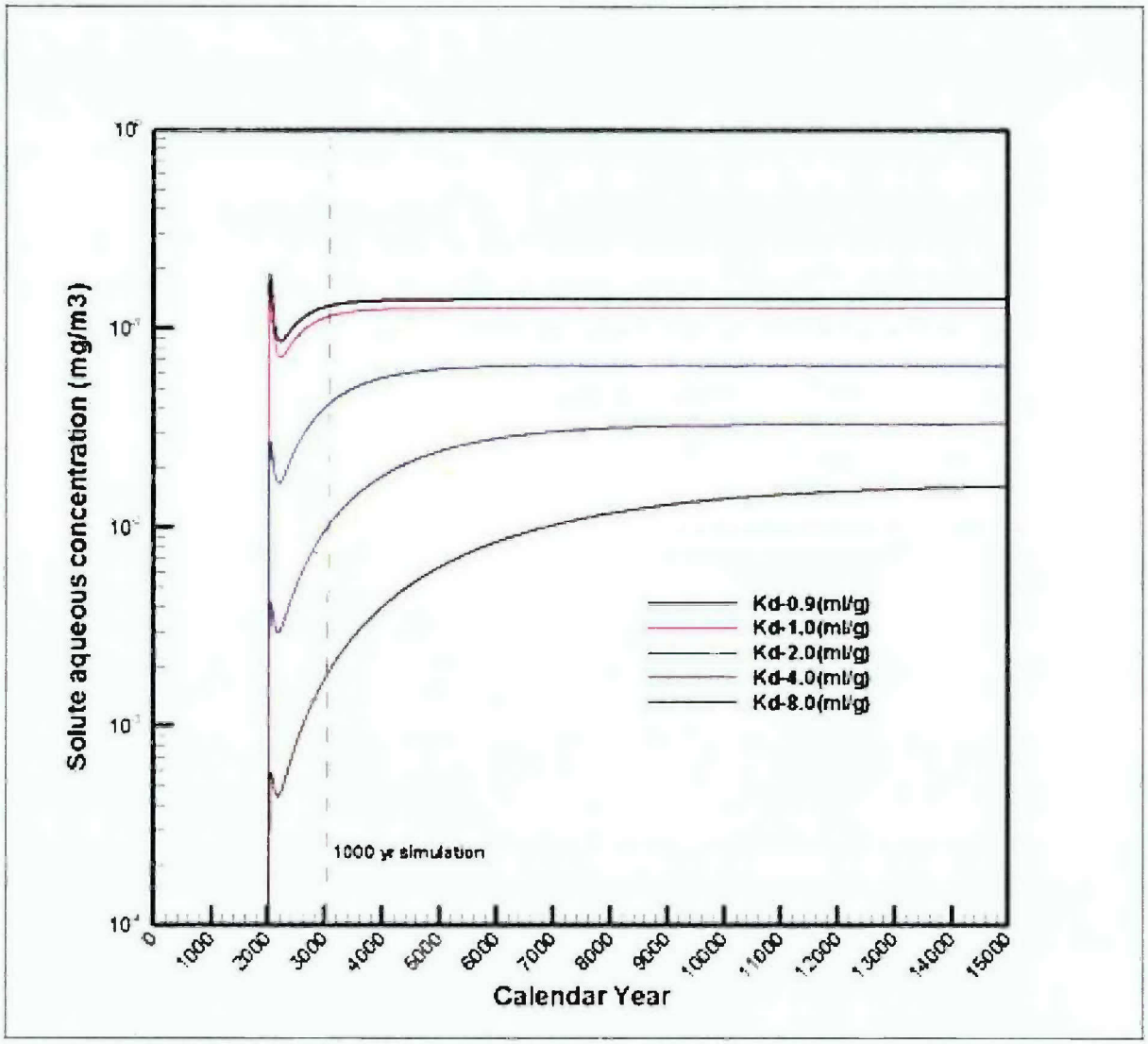


Figure 5-13. Breakthrough Simulated in Representative Soil Column for Range of K_d Values based on 100:0 Initial Source Distribution

5.7 Groundwater/Surface Water Protection Soil Screening Level and Preliminary Remediation Goal Development

Soil screening levels and PRGs represent a data evaluation tool that provides a technical basis for identifying whether a previously remediated waste site or waste site to be remediated in the future could pose a threat to groundwater or surface water quality. The SSL and PRG development process and results are described further in *STOMP 1-D Modeling for Determination of Soil Screening Levels and Preliminary Remediation Goals for 100 Area B and C Source Areas* (ECF-Hanford-12-0003) (Appendix F).

5.7.1 Identification of Peak Groundwater Concentrations

Peak groundwater concentrations were calculated along a portion of the model domain's downgradient boundary, corresponding to the top 5 m (16.4 ft) of the aquifer. The 5 m (16.4 ft) aquifer thickness corresponds to a 5 m (16.4 ft) long monitoring well screen straddling the water table. The concentration was estimated conservatively by calculating it at the aquifer edge beneath the downgradient edge of the waste site footprint, from which the peak concentration and breakthrough time was determined for each contaminant.

Peak concentrations were calculated by running multiple STOMP simulations using a defined set of K_d values. The K_d dataset included 26 values ranging from 0 to 16 mL/g. Peak concentrations greater than 16 mL/g were estimated using regression analysis. No breakthrough was assumed if the estimated peak concentration within 1,000 years was calculated at less than 0.0001 (1E-04) µg/L or 0.0001 (1E-04) pCi/L.

A "NR" result was used to designate non-representative results. The "NR" symbol was also used when no breakthrough was observed in all of the representative stratigraphic columns except one. This condition was observed for the PRG calculation (conservation land use recharge factor) for analytes with K_d values greater than 1 mL/g and for the SSL calculation (irrigation land use recharge factor) for analytes with a K_d value greater than 16 mL/g. Therefore, for these conditions, the SSL and PRG values are defined by a NR.

Chemical SSLs and PRGs protective of groundwater and surface water are presented in Table 5-5. Radionuclide SSLs and PRGs protective of groundwater are presented in Table 5-6. Radionuclide SSLs and PRGs for protection of surface water were not calculated because AWQC for radionuclides have not been established. The final SSL and PRG value for a specific chemical or radionuclide constituent was selected from an array of values calculated for each of the seven different 100-BC soil stratigraphies (Figure 5-4). The final SSL and PRG values correspond to the minimum value calculated from the seven soil stratigraphies. If the minimum value calculated is less than the estimated quantitation limit (EQL), then the EQL was selected as the final SSL or PRG.

5.7.2 Comparison of Waste Site EPCs to SSLs

Soil screening levels were developed to identify COPCs that may pose a threat to groundwater and surface water quality at previously remediated waste sites or waste sites to be remediated in the future. The SSLs were derived using conservative assumptions that include an irrigation land use and a 100:0 waste profile for COPCs with K_d values less than 2 and a 70:30 waste profile for COPCs with K_d values greater than 2. The SSL represents the maximum concentration of a COPC in soil that is not expected to produce a concentration in groundwater at the downgradient boundary of the waste site exceeding the drinking water standard, the MTCA groundwater CUL, or surface water AWQC within 1,000 years.

The waste site evaluation process also included calculation of an EPC, based on a 95 percent UCL, for each COPC present at the waste sites listed in Table 5-7 and comparing the EPC to the SSL. The following exclusion criteria were applied when comparing waste site EPCs to SSLs to eliminate contaminants that are unlikely to pose a threat because:

- The contaminant is an essential nutrient (for example, calcium, magnesium, potassium, or sodium) that is not excessively elevated above background or is not associated with the waste at a waste site.

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Table 5-5. Summary of Chemical SSLs and PRGs Protective of Groundwater and Surface Water Calculated with STOMP

CAS No.	Analyte	K_d Value used to Calculate SSL and PRG (mL/g)	Groundwater Standard (µg/L)	GW SSL (mg/kg)[a,b,c]	GW PRG (mg/kg)[a,b,c]	Surface Water Standard (µg/L)	SW SSL (mg/kg)[a,b,c]	SW PRG (mg/kg)[a,b,c]	EQL[d] (mg/kg)
71-55-6	1,1,1-Trichloroethane	0.14	2.00E+02	1.19E+01	2.51E+01	9.30E+05	5.49E+04	1.16E+05	5.00E-03
79-34-5	1,1,2,2-Tetrachloroethane	0.079	2.20E-01	9.78E-03	1.93E-02	1.70E-01	7.60E-03	1.50E-02	5.00E-03
79-00-5	1,1,2-Trichloroethane	0.075	7.70E-01	3.33E-02	6.55E-02	5.90E-01	2.56E-02	5.03E-02	5.00E-03
75-34-3	1,1-Dichloroethane	0.053	1.60E+03	5.93E+01	1.14E+02	7.40E+04	2.73E+03	5.26E+03	1.00E-02
75-35-4	1,1-Dichloroethene	0.065	7.00E+00	2.84E-01	5.53E-01	5.70E-02	1.00E-02	1.00E-02	1.00E-02
120-82-1	1,2,4-Trichlorobenzene	1.66	1.50E+00	7.21E-01	NR	2.00E+00	9.37E-01	NR	3.30E-01
95-50-1	1,2-Dichlorobenzene	0.379	6.00E+02	7.87E+01	2.58E+02	4.20E+02	5.51E+01	1.80E+02	3.30E-01
107-06-2	1,2-Dichloroethane	0.038	4.80E-01	1.58E-02	3.03E-02	3.80E-01	1.25E-02	2.39E-02	5.00E-03
540-59-0	1,2-Dichloroethene (Total)	0.0396	7.20E+01	2.41E+00	4.61E+00	2.10E+03	7.03E+01	1.35E+02	5.00E-03
78-87-5	1,2-Dichloropropane	0.047	1.20E+00	4.30E-02	8.27E-02	5.00E-01	1.77E-02	3.40E-02	5.00E-03
541-73-1	1,3-Dichlorobenzene	0.434	No Value	NA	NA	3.20E+02	4.73E+01	1.71E+02	3.30E-01
106-46-7	1,4-Dichlorobenzene	0.616	8.10E+00	1.67E+00	8.31E+00	2.20E+01	4.45E+00	2.21E+01	5.00E-03
93-76-5	2,4,5-T	0.049	160	5.75E+00	1.11E+01	No Value	NA	NA	---
93-72-1	2,4,5-TP	0.08	50	2.25E+00	4.44E+00	10	4.50E-01	8.88E-01	---
95-95-4	2,4,5-Trichlorophenol	1.6	8.00E+02	3.67E+02	NR	4.90E+02	2.23E+02	NR	3.30E-01
88-06-2	2,4,6-Trichlorophenol	0.381	4.00E+00	5.24E-01	1.72E+00	1.40E+00	3.30E-01	6.07E-01	3.30E-01
94-75-7	2,4-D	0.029	70	2.11E+00	4.03E+00	100	3.02E+00	5.76E+00	---
94-82-6	2,4-DB	0.1	128	6.50E+00	1.31E+01	6564	3.33E+02	6.72E+02	---
120-83-2	2,4-Dichlorophenol	0.147	2.40E+01	1.47E+00	3.13E+00	7.70E+01	4.71E+00	1.00E+01	3.30E-01
105-67-9	2,4-Dimethylphenol	0.209	1.60E+02	1.30E+01	3.12E+01	3.80E+02	3.09E+01	7.42E+01	3.30E-01
51-28-5	2,4-Dinitrophenol	0.00001	3.20E+01	8.25E-01	1.35E+00	6.90E+01	1.51E+00	2.92E+00	8.25E-01
121-14-2	2,4-Dinitrotoluene	0.0955	2.80E-01	3.30E-01	3.30E-01	1.10E-01	3.30E-01	3.30E-01	3.30E-01
606-20-2	2,6-Dinitrotoluene	0.0692	1.60E+01	6.68E-01	1.30E+00	1.20E+02	4.92E+00	9.61E+00	3.30E-01
78-93-3	2-Butanone	0.00451	4.80E+03	1.11E+02	2.14E+02	4.90E+05	1.14E+04	2.20E+04	1.00E-02
111-76-2	2-Butoxyethanol	0.001	800	1.77E+01	3.43E+01	82044	1.82E+03	3.52E+03	---
91-58-7	2-Chloronaphthalene	2.98	6.40E+02	5.30E+02	NR	1.00E+03	8.29E+02	NR	3.30E-01
95-57-8	2-Chlorophenol	0.388	4.00E+01	5.37E+00	1.80E+01	8.10E+01	1.09E+01	3.64E+01	3.30E-01

Table 5-5. Summary of Chemical SSLs and PRGs Protective of Groundwater and Surface Water Calculated with STOMP

CAS No.	Analyte	K_d Value used to Calculate SSL and PRG (mL/g)	Groundwater Standard (µg/L)	GW SSL (mg/kg)[a,b,c]	GW PRG (mg/kg)[a,b,c]	Surface Water Standard (µg/L)	SW SSL (mg/kg)[a,b,c]	SW PRG (mg/kg)[a,b,c]	EQL[d] (mg/kg)
591-78-6	2-Hexanone	0.01498	4.00E+01	1.05E+00	2.01E+00	3.40E+03	9.02E+01	1.72E+02	2.00E-02
91-57-6	2-Methylnaphthalene	2.98	3.20E+01	2.65E+01	NR	1.40E+02	1.15E+02	NR	3.30E-01
95-48-7	2-Methylphenol (cresol, o-)	0.434	4.00E+02	5.91E+01	2.14E+02	1.20E+04	1.79E+03	6.49E+03	3.30E-01
88-74-4	2-Nitroaniline	0.0527	1.60E+02	5.92E+00	1.14E+01	2.60E+03	9.59E+01	1.85E+02	3.30E-01
88-75-5	2-Nitrophenol	0.297	No Value	NA	NA	No Value	NA	NA	6.60E-01
91-94-1	3,3'-Dichlorobenzidine	0.724	1.90E-01	3.30E-01	3.30E-01	2.10E-02	3.30E-01	3.30E-01	3.30E-01
65794-96-9	3+4 Methylphenol (cresol, m+p)	No Value	No Value	NA	NA	No Value	NA	NA	3.30E-01
99-09-2	3-Nitroaniline	0.0516	4.20E+00	3.30E-01	3.30E-01	1.00E+02	3.77E+00	7.27E+00	3.30E-01
72-54-8	4,4'-DDD (Dichlorodiphenyldichloroethane)	45.8	3.60E-01	NR	NR	6.40E-05	NR	NR	1.65E-02
72-55-9	4,4'-DDE (Dichlorodiphenyldichloroethylene)	86.4	2.60E-01	NR	NR	3.90E-05	NR	NR	1.65E-03
50-29-3	4,4'-DDT (Dichlorodiphenyltrichloroethane)	678	2.60E-01	NR	NR	2.80E-03	NR	NR	1.50E-02
534-52-1	4,6-Dinitro-2-methylphenol	0.6015	1.30E+00	3.30E-01	1.26E+00	1.30E+01	2.62E+00	1.28E+01	3.30E-01
1918-02-1	4-Amino-3,5,6-trichloropicolinic acid	0.039	500	1.66E+01	3.18E+01	57431	1.91E+03	3.66E+03	---
101-55-3	4-Bromophenylphenyl ether	No Value	No Value	NA	NA	No Value	NA	NA	3.30E-01
59-50-7	4-Chloro-3-methylphenol	0.4918	1.60E+03	2.66E+02	1.08E+03	3.10E+04	5.08E+03	2.05E+04	3.30E-01
106-47-8	4-Chloroaniline	0.0725	2.20E-01	3.30E-01	3.30E-01	4.60E+00	3.30E-01	3.88E-01	3.30E-01
7005-72-3	4-Chlorophenylphenyl ether	3.08	No Value	NA	NA	No Value	NA	NA	2.00E+00
108-10-1	4-Methyl-2-pentanone	0.0126	6.40E+02	1.64E+01	3.13E+01	6.10E+04	1.56E+03	2.99E+03	1.00E-02
106-44-5	4-Methylphenol (cresol, p-)	0.434	4.00E+01	5.91E+00	2.14E+01	1.50E+03	2.17E+02	7.84E+02	---
100-01-6	4-Nitroaniline	0.0516	4.40E+00	3.30E-01	3.30E-01	8.10E+01	2.97E+00	5.73E+00	3.30E-01
100-02-7	4-Nitrophenol	0.309	1.30E+02	1.42E+01	4.10E+01	4.00E+03	4.48E+02	1.29E+03	6.60E-01
83-32-9	Acenaphthene	6.12	9.60E+02	1.80E+03	NR	No Value	1.21E+03	NR	1.00E-01
208-96-8	Acenaphthylene	5.03	No Value	NA	NA	5.00E+00	NA	NA	1.00E+00
67-64-1	Acetone	0.0006	7.20E+03	1.59E+02	3.07E+02	7.40E+05	1.63E+04	3.15E+04	2.00E-02
309-00-2	Aldrin	48.7	2.60E-03	NR	NR	3.10E-04	NR	NR	3.30E-03
319-84-6	Alpha-BHC	1.76	1.40E-02	7.14E-03	NR	2.60E-03	1.65E-03	NR	1.65E-03

Table 5-5. Summary of Chemical SSLs and PRGs Protective of Groundwater and Surface Water Calculated with STOMP

CAS No.	Analyte	K_d Value used to Calculate SSL and PRG (mL/g)	Groundwater Standard (µg/L)	GW SSL (mg/kg)[a,b,c]	GW PRG (mg/kg)[a,b,c]	Surface Water Standard (µg/L)	SW SSL (mg/kg)[a,b,c]	SW PRG (mg/kg)[a,b,c]	EQL[d] (mg/kg)
5103-71-9	Alpha-Chlordane	51	2.50E-01	NR	NR	5.00E+01	NR	NR	5.00E+00
7429-90-5	Aluminum	1500	5.00E+01	NR	NR	2.80E-03	NR	NR	1.50E-02
120-12-7	Anthracene	23.5	4.80E+03	NR	NR	9.00E+00	NR	NR	1.00E+00
7440-36-0	Antimony	3.76	6.00E+00	6.66E+00	NR	No Value	6.22E+00	NR	---
12674-11-2	Aroclor-1016	107	5.00E-01	NR	NR	2.00E-04	NR	NR	1.65E-01
11104-28-2	Aroclor-1221	10.3	4.40E-02	1.53E-01	NR	5.20E+00	1.65E-02	NR	---
11141-16-5	Aroclor-1232	10.3	4.40E-02	1.53E-01	NR	6.40E-05	1.65E-02	NR	1.65E-02
53469-21-9	Aroclor-1242	44.8	4.40E-02	NR	NR	6.40E-05	NR	NR	1.65E-02
12672-29-6	Aroclor-1248	43.9	4.40E-02	NR	NR	9.10E+01	NR	NR	1.00E+00
11097-69-1	Aroclor-1254	75.6	4.40E-02	NR	NR	2.40E-01	NR	NR	5.00E-01
11096-82-5	Aroclor-1260	822	4.40E-02	NR	NR	2.80E-03	NR	NR	1.50E-02
7440-38-2	Arsenic	29	5.80E-02	NR	NR		NR	NR	1.00+00
7440-39-3	Barium	25	2.00E+03	NR	NR	8.30E+03	NR	NR	5.00E-02
71-43-2	Benzene	0.062	8.00E-01	3.17E-02	6.15E-02	1.20E+00	4.78E-02	9.27E-02	5.00E-03
56-55-3	Benzo(a)anthracene	360	1.20E-01	NR	NR	4.00E+01	NR	NR	3.30E-01
50-32-8	Benzo(a)pyrene	5500	1.20E-02	NR	NR	2.80E-03	NR	NR	3.00E-02
205-99-2	Benzo(b)fluoranthene	803	1.20E-01	NR	NR	2.70E+02	NR	NR	2.00E-01
191-24-2	Benzo(ghi)perylene	1950	No Value	NA	NA	2.80E-03	NA	NA	3.00E-02
207-08-9	Benzo(k)fluoranthene	1230	1.20E-01	NR	NR	No Value	NR	NR	2.50E+00
7440-41-7	Beryllium	790	4.00E+00	NR	NR	2.20E-04	NR	NR	3.30E-03
319-85-7	beta-1,2,3,4,5,6-Hexachlorocyclohexane (beta-BHC)	2.14	4.90E-02	3.16E-02	NR	9.10E-03	5.92E-03	NR	1.65E-03
108-60-1	bis(2-chloro-1-methylethyl)ether	0.0392	6.30E-01	3.30E-01	3.30E-01	3.70E+01	1.25E+00	2.39E+00	3.30E-01
111-91-1	bis(2-Chloroethoxy)methane	0.00277	4.80E+01	1.09E+00	2.10E+00	2.30E+03	5.31E+01	1.02E+02	3.30E-01
111-44-4	bis(2-chloroethyl) ether	0.076	4.00E-02	3.30E-01	3.30E-01	3.00E-02	3.30E-01	3.30E-01	3.30E-01
117-81-7	bis(2-ethylhexyl) phthalate	110	6.00E+00	NR	NR	6.40E-05	NR	NR	1.65E-02
7440-69-9	Bismuth	No Value	No Value	NA	NA	No Value	NA	NA	3.30E-01

Table 5-5. Summary of Chemical SSLs and PRGs Protective of Groundwater and Surface Water Calculated with STOMP

CAS No.	Analyte	K_d Value used to Calculate SSL and PRG (mL/g)	Groundwater Standard (µg/L)	GW SSL (mg/kg)[a,b,c]	GW PRG (mg/kg)[a,b,c]	Surface Water Standard (µg/L)	SW SSL (mg/kg)[a,b,c]	SW PRG (mg/kg)[a,b,c]	EQL[d] (mg/kg)
7440-42-8	Boron	3	3.20E+03	2.67E+03	NR	1.80E-02	NA	NA	1.00E+00
24959-67-9	Bromide	No Value	No Value	NA	NA	No Value	NA	NA	1.00E+01
75-27-4	Bromodichloromethane	0.055	7.10E-01	2.66E-02	5.14E-02	2.70E-01	1.02E-02	1.97E-02	5.00E-03
75-25-2	Bromoform	0.13	5.50E+00	3.15E-01	6.57E-01	4.30E+00	2.45E-01	5.10E-01	5.00E-03
74-83-9	Bromomethane	0.009	1.10E+01	2.75E-01	5.27E-01	4.70E+01	1.15E+00	2.21E+00	1.00E-02
85-68-7	Butylbenzylphthalate	13.8	4.60E+01	2.93E+02	NR	1.70E+00	5.25E+01	NR	3.30E-01
7440-43-9	Cadmium	30	5.00E+00	NR	NR	5.20E-05	NR	NR	3.30E-03
7440-70-2	Calcium	No Value	No Value	NA	NA	No Value	NA	NA	2.50E+00
86-74-8	Carbazole	3.39	4.40E+00	4.18E+00	NR	No Value	NA	NA	3.30E-01
75-15-0	Carbon disulfide	0.046	8.00E+02	2.81E+01	5.40E+01	1.30E+04	4.67E+02	8.97E+02	5.00E-03
56-23-5	Carbon tetrachloride	0.152	3.40E-01	2.10E-02	4.52E-02	2.30E-01	1.44E-02	3.09E-02	5.00E-03
57-74-9	Chlordane	51	2.50E-01	NR	NR	5.70E-04	NR	NR	1.65E-02
16887-00-6	Chloride	0	2.50E+05	5.46E+03	1.06E+04	2.30E+05	5.03E+03	9.73E+03	2.00E+00
108-90-7	Chlorobenzene	0.22	1.00E+02	8.40E+00	2.04E+01	1.30E+02	1.09E+01	2.66E+01	5.00E-03
75-00-3	Chloroethane	0.0217	No Value	NA	NA	No Value	NA	NA	1.00E-02
67-66-3	Chloroform	0.053	1.40E+00	5.23E-02	1.01E-01	5.70E+00	2.11E-01	4.08E-01	5.00E-03
74-87-3	Chloromethane	0.006	No Value	NA	NA	No Value	NA	NA	1.00E-02
7440-47-3	Chromium	200	1.00E+02	NR	NR	No Value	NR	NA	5.00E+00
218-01-9	Chrysene	200	1.20E+00	NR	NR	6.50E+01	NR	NR	2.00E-01
156-59-2	cis-1,2-Dichloroethylene	0.036	7.00E+01	2.26E+00	4.32E+00	2.30E+03	7.53E+01	1.44E+02	5.00E-03
10061-01-5	cis-1,3-Dichloropropene	0.027	4.40E-01	1.30E-02	2.47E-02	3.40E-01	1.01E-02	1.92E-02	5.00E-03
7440-48-4	Cobalt	50	4.80E+00	NR	NR	9.00E+01	NR	NR	5.00E-02
PCB1242/1016	Co-elution of Aroclor-1242 and Aroclor-1016	No Value	No Value	NA	NA	No Value	NA	NA	1.00E+02
7440-50-8	Copper	22	6.40E+02	1.66E+04	NR	1.30E+03	2.34E+02	NR	2.00E+00
57-12-5	Cyanide	9.9	2.00E+02	6.53E+02	NR	No Value	1.70E+01	NR	3.30E-03
75-99-0	Dalapon	0.0027	200	4.53E+00	8.74E+00	10773	2.44E+02	4.71E+02	---

Table 5-5. Summary of Chemical SSLs and PRGs Protective of Groundwater and Surface Water Calculated with STOMP

CAS No.	Analyte	K_d Value used to Calculate SSL and PRG (mL/g)	Groundwater Standard (µg/L)	GW SSL (mg/kg)[a,b,c]	GW PRG (mg/kg)[a,b,c]	Surface Water Standard (µg/L)	SW SSL (mg/kg)[a,b,c]	SW PRG (mg/kg)[a,b,c]	EQL[d] (mg/kg)
319-86-8	Delta-BHC	2.81	No Value	NA	NA	No Value	NA	NA	1.65E-03
53-70-3	Dibenz[a,h]anthracene	1790	1.20E-01	NR	NR	8.70E+01	NR	NR	5.00E+00
132-64-9	Dibenzofuran	11.3	1.60E+01	6.64E+01	NR	2.90E-01	7.08E+00	NR	3.30E-03
124-48-1	Dibromochloromethane	0.063	5.20E-01	2.09E-02	4.05E-02	4.00E-01	1.60E-02	3.11E-02	5.00E-03
1918-00-9	Dicamba	0.029	480	1.45E+01	2.77E+01	24613	7.42E+02	1.42E+03	---
60-57-1	Dieldrin	25.6	5.50E-03	NR	NR	2.60E+04	NR	NR	1.00E+00
60-29-7	Diethylether	0.009699	1600	3.96E+01	7.60E+01	96000	2.38E+03	4.56E+03	---
84-66-2	Diethylphthalate	0.082	1.30E+04	5.83E+02	1.15E+03	1.70E+04	7.74E+02	1.53E+03	3.30E-01
131-11-3	Dimethyl phthalate	0.0316	No Value	NA	NA	2.70E+05	8.34E+03	1.60E+04	3.30E-01
84-74-2	Di-n-butylphthalate	1.57	1.60E+03	7.19E+02	NR	2.00E+03	8.99E+02	NR	3.30E-01
117-84-0	Di-n-octylphthalate	83000	No Value	NA	NA	2.80E-03	NR	NA	1.50E-02
88-85-7	Dinoseb	3.5	7	6.98E+00	NR	No Value	NA	NA	5.00E+01
959-98-8	Endosulfan I	2.04	9.60E+01	6.09E+01	NR	5.60E-02	3.55E-02	NR	1.65E-03
33213-65-9	Endosulfan II	2.04	9.60E+01	6.09E+01	NR	5.60E-02	3.55E-02	NR	3.30E-03
1031-07-8	Endosulfan sulfate	2.04	No Value	NA	NA	9.30E-01	5.90E-01	NR	3.30E-03
72-20-8	Endrin	10.8	2.00E+00	7.61E+00	NR	6.40E-05	8.76E-03	NR	1.65E-02
7421-93-4	Endrin aldehyde	10.8	No Value	NA	NA	2.30E-03	1.10E+00	NR	3.30E-03
53494-70-5	Endrin ketone	9.72	No Value	NA	NA	7.90E-05	NA	NA	1.65E-03
100-41-4	Ethylbenzene	0.2	4.00E+00	3.16E-01	7.49E-01	1.60E+01	1.25E+00	2.96E+00	5.00E-03
107-21-1	Ethylene Glycol	0.001	16,000	3.55E+02	6.86E+02	1,640,000	3.63E+04	7.03E+04	---
206-44-0	Fluoranthene	49.1	6.40E+02	NR	NR	4.90E-05	NR	NR	1.65E-03
86-73-7	Fluorene	7.71	6.40E+02	1.51E+03	NR	6.40E+02	2.60E+03	NR	1.00E-01
16984-48-8	Fluoride	150	4.80E+02	NR	NR	5.20E+02	NR	NR	1.00E+01
58-89-9	Gamma-BHC (Lindane)	1.35	8.00E-02	3.12E-02	NR	1.90E-02	7.45E-03	NR	1.65E-03
76-44-8	Heptachlor	9.53	1.90E-02	5.96E-02	NR	1.10E+03	1.65E-03	NR	3.00E-02
1024-57-3	Heptachlor epoxide	83.2	4.80E-03	NR	NR	3.00E-02	NR	NR	1.65E-02
118-74-1	Hexachlorobenzene	80	5.50E-02	NR	NR	6.40E-05	NR	NR	1.65E-02

Table 5-5. Summary of Chemical SSLs and PRGs Protective of Groundwater and Surface Water Calculated with STOMP

CAS No.	Analyte	K_d Value used to Calculate SSL and PRG (mL/g)	Groundwater Standard (µg/L)	GW SSL (mg/kg)[a,b,c]	GW PRG (mg/kg)[a,b,c]	Surface Water Standard (µg/L)	SW SSL (mg/kg)[a,b,c]	SW PRG (mg/kg)[a,b,c]	EQL[d] (mg/kg)
87-68-3	Hexachlorobutadiene	53.7	5.60E-01	NR	NR	5.70E-04	NR	NR	1.65E-02
77-47-4	Hexachlorocyclopentadiene	200	4.80E+01	NR	NR	2.80E-03	NR	NR	1.00E-01
67-72-1	Hexachloroethane	1.78	3.10E+00	1.63E+00	NR	1.40E+00	7.31E-01	NR	3.30E-01
18540-29-9	Hexavalent Chromium	0.8	4.80E+01	6[e]	6[e]	1.00E+01	2.65E+00	6[e]	---
193-39-5	Indeno(1,2,3-cd)pyrene	3470	1.20E-01	NR	NR	No Value	NR	NR	3.00E-02
7439-89-6	Iron	25	3.00E+02	NR	NR	1.00E+03	NR	NR	5.00E-01
78-59-1	Isophorone	0.0468	4.60E+01	1.63E+00	3.13E+00	8.40E+00	3.30E-01	5.70E-01	3.30E-01
7439-92-1	Lead	30	1.50E+01	NR	NR	2.50E-01	NR	NR	2.00E-01
7439-93-2	Lithium	50	3.20E+01	NR	NR	2.60E+00	NA	NA	2.00E+00
7439-95-4	Magnesium	No Value	No Value	NA	NA	No Value	NA	NA	---
7439-96-5	Manganese	50	5.00E+01	NR	NR	No Value	NR	NR	2.50E+00
7439-97-6	Mercury	30	2.00E+00	NR	NR	2.10E+00	NR	NR	5.00E-01
72-43-5	Methoxychlor	80	4.00E+01	NR	NR	2.80E-04	NR	NR	3.30E-01
75-09-2	Methylene chloride	0.01	5.00E+00	1.24E-01	2.38E-01	4.60E+00	1.14E-01	2.19E-01	5.00E-03
7439-98-7	Molybdenum	20	8.00E+01	1.47E+03	NR	No Value	2.39E+04	NR	5.00E-02
108-38-3	m-Xylene	0.2	1.60E+03	1.27E+02	3.01E+02	No Value	NA	NA	---
91-20-3	Naphthalene	1.19	1.60E+02	5.74E+01	NR	4.90E+03	1.77E+03	NR	1.00E-01
7440-02-0	Nickel	65	1.00E+02	NR	NR	4.40E-01	NR	NR	3.30E-01
14797-55-8	Nitrate	0	4.50E+04	9.84E+02	1.90E+03	4.50E+04	9.84E+02	1.90E+03	2.50E+00
14797-65-0	Nitrite	0	3.30E+03	7.21E+01	1.40E+02	No Value	NA	NA	2.50E+00
98-95-3	Nitrobenzene	0.119	1.60E+01	8.72E-01	1.79E+00	1.70E+01	9.26E-01	1.91E+00	3.30E-01
NO3-N	Nitrogen in Nitrate	0	1.00E+04	2.19E+02	4.23E+02	1.00E+04	2.19E+02	4.23E+02	7.50E-01
NO2-N	Nitrogen in Nitrite	0	1.00E+03	2.19E+01	4.23E+01	No Value	NA	NA	7.50E-01
NO2+NO3-N	Nitrogen in Nitrite and Nitrate	0	1.00E+04	2.19E+02	4.23E+02	1.00E+04	2.19E+02	4.23E+02	---
621-64-7	n-Nitrosodi-n-dipropylamine	0.024	1.30E-02	3.30E-01	3.30E-01	5.00E-03	3.30E-01	3.30E-01	3.30E-01
86-30-6	n-Nitrosodiphenylamine	1.29	1.80E+01	6.77E+00	NR	3.30E+00	1.25E+00	NR	3.30E-01
95-47-6	o-Xylene	0.24	1.60E+03	1.42E+02	3.57E+02	No Value	NA	NA	---

Table 5-5. Summary of Chemical SSLs and PRGs Protective of Groundwater and Surface Water Calculated with STOMP

CAS No.	Analyte	K_d Value used to Calculate SSL and PRG (mL/g)	Groundwater Standard (µg/L)	GW SSL (mg/kg)[a,b,c]	GW PRG (mg/kg)[a,b,c]	Surface Water Standard (µg/L)	SW SSL (mg/kg)[a,b,c]	SW PRG (mg/kg)[a,b,c]	EQL[d] (mg/kg)
87-86-5	Pentachlorophenol	0.592	7.30E-01	3.30E-01	6.94E-01	2.70E-01	3.30E-01	3.30E-01	3.30E-01
85-01-8	Phenanthrene	16.7	No Value	NA	NA	8.20E+00	NA	NA	3.30E-01
108-95-2	Phenol	0.0288	2.40E+03	7.23E+01	1.38E+02	1.00E+04	3.01E+02	5.75E+02	3.30E-01
14265-44-2	Phosphate	No Value	No Value	NA	NA	No Value	NA	NA	7.50E+01
7723-14-0	Phosphorus	3.5	1.60E-01	5.00E+01	NR	No Value	NA	NA	3.30E-01
PO4-P	Phosphorus in phosphate	No Value	No Value	NA	NA	No Value	NA	NA	5.00E+00
7440-09-7	Potassium	No Value	No Value	NA	NA	No Value	NA	NA	---
129-00-0	Pyrene	68	4.80E+02	NR	NR	5.20E+01	NR	NR	4.00E+00
7782-49-2	Selenium	5	5.00E+01	7.42E+01	NR	5.60E+00	7.42E+00	NR	6.00E-01
7440-21-3	Silicon	No Value	No Value	NA	NA	No Value	NA	NA	4.00E+02
7440-22-4	Silver	90	8.00E+01	NR	NR	2.20E-04	NR	NR	3.30E-03
7440-23-5	Sodium	No Value	No Value	NA	NA	No Value	NA	NA	2.00E+00
7440-24-6	Strontium	25	9.60E+03	NR	NR	3.00E+02	NR	NR	5.00E+00
100-42-5	Styrene	0.91	1.00E+02	2.98E+01	2.35E+02	3.80E+04	1.15E+04	9.01E+04	5.00E-03
14808-79-8	Sulfate	0	2.50E+05	5.46E+03	1.06E+04	No Value	NA	NA	5.00E+00
127-18-4	Tetrachloroethene	0.27	8.10E-02	7.93E-03	2.11E-02	3.90E-01	3.79E-02	1.01E-01	5.00E-03
7440-28-0	Thallium	71	5.00E-01	NR	NR	8.30E+02	NR	NR	5.00E-02
7440-31-5	Tin	130	9.60E+03	NR	NR	1.20E+00	NR	NR	3.30E-01
108-88-3	Toluene	0.14	6.40E+02	3.80E+01	8.02E+01	1.30E+03	7.71E+01	1.63E+02	5.00E-03
TPHDIESEL	Total petroleum hydrocarbons - diesel range	No Value	5.00E+02	NA	NA	No Value	NA	NA	5.00E+01
TPH/OILH	Total petroleum hydrocarbons - motor oil (high boiling)	No Value	5.00E+02	NA	NA	No Value	NA	NA	---
8001-35-2	Toxaphene	95.8	8.00E-02	NR	NR	2.60E+00	NR	NR	2.00E-01
156-60-5	trans-1,2-Dichloroethylene	0.038	1.00E+02	3.29E+00	6.30E+00	1.40E+02	4.61E+00	8.82E+00	5.00E-03
10061-02-6	trans-1,3-Dichloropropene	0.027	4.40E-01	1.30E-02	2.47E-02	3.40E-01	1.01E-02	1.92E-02	5.00E-03
126-73-8	Tributyl phosphate	1.89	9.51	5.42E+00	NR	24.3	1.39E+01	NR	---
79-01-6	Trichloroethene	0.094	4.90E-01	2.40E-02	4.81E-02	2.50E+00	1.22E-01	2.45E-01	5.00E-03

Table 5-5. Summary of Chemical SSLs and PRGs Protective of Groundwater and Surface Water Calculated with STOMP

CAS No.	Analyte	K_d Value used to Calculate SSL and PRG (mL/g)	Groundwater Standard (µg/L)	GW SSL (mg/kg)[a,b,c]	GW PRG (mg/kg)[a,b,c]	Surface Water Standard (µg/L)	SW SSL (mg/kg)[a,b,c]	SW PRG (mg/kg)[a,b,c]	EQL[d] (mg/kg)
75-69-4	Trichloromonofluoromethane	0.04389	2400	8.29E+01	1.59E+02	35800	1.24E+03	2.37E+03	---
7440-61-1	Uranium	2	3.00E+01	1.89E+01	NR	7.80E+02	4.89E+02	NR	---
7440-62-2	Vanadium	1000	8.00E+01	NR	NR	6.40E-05	NA	NA	1.65E-02
75-01-4	Vinyl chloride	0.019	6.10E-02	5.00E-03	5.00E-03	2.50E-02	5.00E-03	5.00E-03	5.00E-03
1330-20-7	Xylenes (total)	0.23	1.60E+03	1.38E+02	3.41E+02	No Value	NA	NA	1.00E-02
7440-66-6	Zinc	30	4.80E+03	NR	NR	1.20E-02	NR	NR	---

Notes:

a. All analytes screening value defaults to the EQL when the calculated value is less than the EQL.

b. NR is applied to indicate non-representative results when there is breakthrough simulated within 1,000 years for only one soil column while all other soil columns (most) show no breakthrough (defined as concentrations below 1E-04 µg/L or activity below 1E-04 pCi/L).

c. NA is applied when applicable water quality standard is not available to calculate the SSL or PRG values.

d. The SSL and/or PRG are set equal to the EQL if the calculated value is below the EQL.

e. The PRG value was constrained to a maximum value of 6 mg/kg because the K_d value used in the model was derived from experiments with soil concentrations less than 6 mg/kg. A PRG level of 2 mg/kg will be used in the FS to align with the remedial goal specified in the interim action ROD.

Table 5-6. Summary of Radionuclide SSL and PRGs Protective of Groundwater Calculated with STOMP

Radionuclide	Distribution Coefficient Kd Value (mL/g)	Maximum Contaminant Level (pCi/L)	Half-life (years)	Groundwater SSL (pCi/g)[a,b,c]	Groundwater PRG (pCi/g)[a,b,c]	EQL[f] (mg/kg)
Americium-241	200	15	4.32E+02	NR	NR	1.00E+00
Carbon-14[e]	200	2,000	5.73E+03	NR	NR	---
Carbon-14[d]	0	2,000	5.73E+03	4.38E+01	8.44E+01	---
Cesium-137	50	200	3.00E+01	NR	NR	1.00E-01
Curiumm-243	200	15	2.85E+01	NR	NR	---
Cobalt-60	50	100	5.72E+00	NR	NR	5.00E-02
Europium-152	200	200	1.33E+01	NR	NR	1.00E-01
Europium-154	200	60	8.80E+00	NR	NR	1.00E-01
Europium-155	200	600	4.96E+00	NR	NR	1.00E-01
Iodine-129	1	1	1.57E+07	3.26E-01	NR	---
Niobium-94	200	No Value	2.03E+04	NA	NA	---
Neptunium-237	15	15	2.14E+06	1.18E+02	NR	---
Nickel-63	30	50	9.60E+01	NR	NR	---
Plutonium-238	200	15	8.77E+01	NR	NR	1.00E+00
Plutonium-239	200	15	2.41E+04	NR	NR	1.00E+00
Plutonium-240	200	15	6.54E+03	NR	NR	1.00E+00
Plutonium-241	200	300	1.40E+01	NR	NR	---
Radium-226	200	5	1.60E+03	NR	NR	---

Table 5-6. Summary of Radionuclide SSL and PRGs Protective of Groundwater Calculated with STOMP

Radionuclide	Distribution Coefficient Kd Value (mL/g)	Maximum Contaminant Level (pCi/L)	Half-life (years)	Groundwater SSL (pCi/g)[a,b,c]	Groundwater PRG (pCi/g)[a,b,c]	EQL[f] (mg/kg)
Radium-228	200	5	5.75E+00	NR	NR	2.00E-01
Strontium-90	25	8	2.91E+01	NR	NR	---
Technetium-99	0	900	2.13E+05	1.97E+01	3.80E+01	---
Thorium-228	200	15	1.91E+00	NR	NR	---
Thorium-230	200	15	7.70E+04	NR	NR	---
Thorium-232	200	15	1.41E+10	NR	NR	---
Tritium	0	20,000	1.24E+01	8.68E+02	1.22E+03	---
Uranium-233/234	2	20	2.45E+05	NA	NA	1.00E+00
Uranium-235	2	20	7.04E+08	NA	NA	5.00E-01
Uranium-238	2	20	4.47E+09	NA	NA	1.00E+00

a. All analytes screening value defaults to the EQL when the calculated value is less than the EQL.

b. NR is applied to indicate non-representative results when there is breakthrough simulated within 1,000 years for only one soil column while all other soil columns (most) show no breakthrough (defined as concentrations below 1E-04 µg/L or activity below 1E-04 pCi/L).

c. NA is applied when applicable water quality standard is not available to calculate the SSL or PRG values.

d. Carbon-14 in liquid form, typically associated with reactor gas condensate.

e. Carbon-14 in solid form, typically associated with graphite.

f. The SSL and/or PRG are set equal to the EQL if the calculated value is below the EQL.

Table 5-7. Interim Remediated Waste Sites in 100-BC

Waste Site Identification			
100-B-1_Shallow _1	100-B-28_Shallow_1	116-B-5_Deep_Focused	118-B-4_Shallow
100-B-1_Shallow _2	100-B-28_Shallow_3	116-B-5_Shallow_Focused	118-B-6_Deep
100-B-1_Shallow _Focused	100-B-28_Shallow_5	116-B-6A_Deep	118-B-6_Overburden
100-B-11_Shallow_Focused	100-B-28_Shallow_Focused	116-B-6A_Shallow	118-B-6_Shallow
100-B-14:1_Deep_Focused	100-B-28_Staging pile area_2	116-B-6B_Shallow	118-B-6_Staging Pile Area
100-B-14:1_Overburden_2	100-B-28_Staging Pile Area_4	116-B-7, 132-B-6, 132-C-2_Deep	118-B-7_Shallow_Focused
100-B-14:1_Shallow	100-B-31_Shallow	116-B-7, 132-B-6, 132-C-2_Shallow	118-B-9_Shallow_Focused
100-B-14:2_Overburden_Focused	100-B-33_Shallow_Focused	116-B-9_Shallow	118-C-1_Overburden
100-B-14:2_Shallow_1	100-B-33_Staging Pile Area_Focused	116-C-1_Deep	118-C-1_Shallow_1
100-B-14:2_Shallow_2	100-B-5_Deep	116-C-1_Deep_Focused	118-C-1_Shallow_2
100-B-14:2_Shallow_3	100-B-5_Shallow	116-C-1_Overburden	118-C-1_Shallow_3
100-B-14:2_Shallow_Focused	100-B-8:1_Deep	116-C-1_Shallow	118-C-1_Shallow_4
100-B-14:3_Deep_Focused	100-B-8:1_Overburden	116-C-2A_Deep	118-C-1_Shallow_Focused
100-B-14:5_Shallow_Focused	100-B-8:1_Shallow	116-C-2A_Overburden	118-C-1_Staging Pile Area
100-B-14:6_Shallow_Focused	100-B-8:2_Deep	116-C-2A_Shallow	118-C-2_Shallow
100-B-14:7_Shallow_Focused	100-B-8:2_Overburden	116-C-3_Overburden_Focused	118-C-2_Staging Pile Area
100-B-16_Shallow_Focused	100-B-8:2_Shallow_1	116-C-3_Shallow	118-C-3:2_Deep_Focused
100-B-18_Shallow_Focused	100-B-8:2_Shallow_3	116-C-3_Shallow_Focused	118-C-3:3_Shallow_Focused
100-B-19_Shallow_1	100-C-3_Shallow	116-C-3_Staging pile area_Focused	118-C-4_Shallow
100-B-19_Shallow_2	100-C-9:1_Deep_Focused	116-C-5_Deep	120-B-1_Shallow_Focused
100-B-19_Shallow_4	100-C-9:1_Overburden_Focused	116-C-5_Overburden	126-B-3_Shallow

Table 5-7. Interim Remediated Waste Sites in 100-BC

Waste Site Identification			
100-B-19_Shallow_5	100-C-9:1_Shallow_1	116-C-5_Shallow	126-B-3_Staging pile area_2
100-B-19_Shallow_Focused	100-C-9:1_Shallow_2	116-C-6_Shallow_Focused	126-B-3_Staging pile area_3
100-B-19_Staging pile area	100-C-9:1_Shallow_Focused	118-B-1_Overburden_1	126-B-3_Staging Pile Area_Focused
100-B-19_Staging Pile Area_Focused	100-C-9:2_Overburden_Focused	118-B-1_Overburden_8	128-B-2_Shallow
100-B-20_Shallow_Focused	100-C-9:2_Shallow	118-B-1_Overburden_9	128-B-3_Shallow_1
100-B-21:2_Overburden_Focused	100-C-9:2_Shallow_Focused	118-B-1_Shallow_1	128-B-3_Shallow_2
100-B-21:2_Shallow	100-C-9:3_Deep_Focused	118-B-1_Shallow_2	128-B-3_Shallow_3
100-B-21:3_Overburden_Focused	116-B-1_Deep	118-B-1_Shallow_3	128-B-3_Staging pile area
100-B-21:3_Stsging Pile Area_Focused	116-B-1_Shallow	118-B-1_Shallow_4	128-B-3_Staging pile area_Focused
100-B-21:4_Overbuden_Focused	116-B-10_Shallow	118-B-1_Shallow_5	128-C-1_Shallow
100-B-21:4_Shallow	116-B-11_Deep	118-B-1_Shallow_6	128-C-1_Shallow_Focused
100-B-21:4_Staging Pile Area	116-B-11_Shallow	118-B-1_Shallow_7	1607_B-10_Shallow
100-B-22:2_Shallow_Focused	116-B-12_Deep	118-B-1_Shallow_Focused	1607_B-11_Shallow
100-B-22:2_Staging Pile Area_Focused	116-B-13_Shallow	118-B-1_Staging pile area	1607_B-7_Shallow
100-B-23_Shallow_Focused	116-B-14_Deep	118-B-1_Staging Pile Area_Focused	1607_B-8_Shallow
100-B-25_Overbuden_Focused	116-B-14_Shallow	118-B-10_Shallow	1607_B-9_Shallow
100-B-25_Shallow	116-B-15_Shallow_Focused	118-B-10_Shallow_Focused	1607-B1_Shallow_Focused
100-B-25_Staging Pile Area	116-B-2_Deep	118-B-10_Staging Pile Area	1607-B2:1_Overburden_Focused
100-B-26_Shallow_Focused	116-B-2_Shallow	118-B-3_Overburden_2	1607-B2:1_Shallow
100-B-27_Deep	116-B-3_Deep	118-B-3_Overburden_4	1607-B2:2_Overburden_Focused
100-B-27_Overburden_Focused	116-B-3_Shallow	118-B-3_Shallow	1607-B2:2_Shallow

Table 5-7. Interim Remediated Waste Sites in 100-BC

Waste Site Identification			
100-B-27_Staging Pile Area	116-B-4_Deep	118-B-3_Shallow_Focused	600-232-Shallow
100-B-28_Overburden_Focused	116-B-4_Shallow	118-B-3_Staging pile area	600-233_Shallow_Focused

- The radionuclides are associated with background conditions and not attributable to waste site practices (for example, potassium-40, radium-224, radium-226, radium-228, thorium-228, thorium-230, or thorium-232).
- The radionuclides have half-lives less than 3 years and that, upon decay, produce no significant daughter products.
- The maximum concentration of an inorganic analyte is less than or equal to the 90th percentile value of the background concentration.

If a waste site COPC EPC exceeded the SSL for groundwater or surface water protection, and was not eliminated by one of the exclusion criteria described above, it was carried forward to the PRG evaluation step. Waste sites with COPC EPCs greater than SSL will be identified for a non-irrigation land use institutional control in the FS. The information presented in Table 5-8 presents the comparisons for the COPCs that exceeded a SSL only. *Comparison of 100-BC-1 and 100-BC-2 Source Operable Unit Exposure Point Concentrations to Soil Screening Levels Protective of Groundwater and Soil Screening Levels Protective of Surface Water* (ECF-100BC1-11-0082 [Appendix F]) presents a summary of the for all the waste site EPCs to SSLs.

The following waste sites and COPCs had EPCs above their respective SSL (Table 5-8):

- **100-B-1.** Beta-BHC at 6.8 µg/kg in the Shallow 2 decision unit exceeded its surface water protection SSL of 5.9 µg/kg. Heptachlor at 5.5 µg/kg in the Shallow 2 decision unit exceeded its surface water protection SSL of 1.7 µg/kg.
- **100-B-14:2.** Alpha-BHC at 2.0 µg/kg in the Overburden Focused decision unit and pentachlorophenol at 1,900 µg/kg in the Shallow 2 decision unit exceeded their respective surface water protection SSLs. Pentachlorophenol also exceeded its groundwater protection SSL.
- **100-B-18 (Shallow Focused decision unit), 116-B-14 (Deep decision unit), and 116-B-4 (Deep decision unit).** Antimony concentrations ranging between 6,600 and 9,300 µg/kg exceeded the groundwater protection and/or the surface water protection SSL.
- **118-B-1 (Shallow 4 decision unit) and 118-C-1 (Shallow Focused decision unit).** Carbon tetrachloride concentrations of 17 µg/kg and 39 µg/kg exceeded the groundwater and/or surface water protection SSL.
- **118-B-6 (Deep decision unit).** Tritium at 2,780 pCi/L exceeded the groundwater and surface water protection SSL of 868 pCi/L.

Based on the waste site EPC to SSL comparisons presented in Table 5-8, as summarized above, will be identified in the FS for application of a no irrigation institutional control.

5.7.3 Comparison of Waste Site EPCs to PRGs

PRGs represent the maximum concentration of COPCs that can remain in the vadose zone without causing an exceedance of an applicable groundwater or surface water quality standard. As described previously, the primary factor that leads to the difference between the SSL and PRG values is the magnitude of the infiltration/recharge term used in their derivation. The STOMP model derived SSL presumes an irrigation land use whereas the PRG derivation uses the natural infiltration rate.

The waste site with COPC EPCs greater than the groundwater protection (Table 5-9) and surface water (Table 5-9) protection PRG was 118-B-6 Deep for tritium. The EPC of 2,780 pCi/g exceeded the groundwater and surface water protection PRG of 1,217 pCi/g. To assess how well tritium, which

Table 5-8. Waste Sites with EPCs Greater than SSLs Protective of Groundwater or Surface Water

Waste Site/Decision Unit	Analyte Name	CAS No.	Units	Exposure Point Concentration	STOMP 1D 70:30/100:0 Contaminant Source Model SSL for Groundwater Protection*	Is EPC > SSL Protective of Groundwater?	STOMP 1D 70:30/100:0 Contaminant Source Model SSL for Surface Water Protection*	Is EPC > SSL Protective of Surface Water?
100-B-1_Shallow_2	beta-1,2,3,4,5,6-Hexachlorocyclohexane (beta-BHC)	319-85-7	µg/kg	6.8	32	No	5.9	Yes
100-B-1_Shallow_2	Heptachlor	76-44-8	µg/kg	5.5	60	No	1.7	Yes
118-B-1_Shallow_4	beta-1,2,3,4,5,6-Hexachlorocyclohexane (beta-BHC)	319-85-7	µg/kg	7.8	32	No	5.9	Yes
100-B-14:2_Overburden_Focused	Alpha-BHC	319-84-6	µg/kg	2.0	7.1	No	1.7	Yes
100-B-14:2_Shallow_2	Pentachlorophenol	87-86-5	µg/kg	1,900	330	Yes	330	Yes
100-B-18_Shallow_Focused	Antimony	7440-36-0	µg/kg	9,300	6,661	Yes	6,217	Yes
116-B-14_Deep	Antimony	7440-36-0	µg/kg	6,600	6,661	No	6,217	Yes
116-B-4_Deep	Antimony	7440-36-0	µg/kg	7,900	6,661	Yes	6,217	Yes
118-B-1_Shallow_4	Carbon tetrachloride	56-23-5	µg/kg	17	21	No	14	Yes
118-B-6_Deep	Tritium	10028-17-8	pCi/g	2,780	868	Yes	868	Yes
118-C-1_Shallow_Focused	Carbon tetrachloride	56-23-5	µg/kg	39	21	Yes	14	Yes

** *STOMP 1-D Modeling for Determination of Soil Screening Levels and Preliminary Remediation Goals for 100 Area B and C Source Areas* (ECF-Hanford-12-0003) A 70:30 source distribution is used for analytes with $K_d \geq 2$ mL/g; a 100:0 source distribution is used for analytes with $K_d < 2$ mL/g.

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Table 5-9. Waste Sites with EPCs Greater than PRGs Protective of Groundwater

Waste Site/Decision Unit	COPC	CAS No.	Units	EPC	STOMP 1D 70:30/100:0 Groundwater Protection PRG*	Is EPC > PRG?
118-B-6_Deep	Tritium	10028-17-8	pCi/g	2,780	1,217	Yes
COPCs Eliminated through Further Evaluation						
100-B-14:2_Shallow_2	Pentachlorophenol	87-86-5	µg/kg	1,900	694	Yes

* * *STOMP 1-D Modeling for Determination of Soil Screening Levels and Preliminary Remediation Goals for 100 Area B and C Source Areas* (ECF-Hanford-12-0003). A 70:30 source distribution is used for analytes with $K_d \geq 2$ mL/g; a 100:0 source distribution is used for analytes with $K_d < 2$ mL/g.

Table 5-10. Waste Sites with EPCs Greater than PRGs Protective of Surface Water

Waste Site/Decision Unit	COPC	CAS No.	Units	EPC	STOMP 1D 70:30/100:0 Surface Water Protection PRG*	Is EPC > PRG?
118-B-6_Deep	Tritium	10028-17-8	pCi/g	2,780	1,217	Yes
COPCs Eliminated Through Further Evaluation						
100-B-14:2_Shallow_2	Pentachlorophenol	87-86-5	µg/kg	1,900	330	Yes
118-C-1_Shallow_Focused	Carbon tetrachloride	56-23-5	µg/kg	39	31	Yes

* *STOMP 1-D Modeling for Determination of Soil Screening Levels and Preliminary Remediation Goals for 100 Area B and C Source Areas* (ECF-Hanford-12-0003). A 70:30 source distribution is used for analytes with Kd ≥ 2 mL/g; a 100:0 source distribution is used for analytes with Kd < 2 mL/g.

has a K_d value of 0, conforms with the 100:0 waste distribution model upon which the 1,217 pCi/L groundwater and surface water protection PRGs are based, the vertical distribution of tritium in the vadose zone was reviewed. As shown on Figure 4-12, the distribution of tritium at the 118-B-6 waste site drops off sharply with depth. Therefore, this PRG is expected to be conservative and a site-specific vadose zone assessment would be warranted for the 118-B-6 waste site.

Based on the current results of the EPC and PRG comparison, and vertical distribution of tritium in the vadose zone, the 118-B-6 waste site will be carried forward into the FS for groundwater surface water protection remedial action alternatives development and evaluation.

5.7.3.1 Waste Sites with COPC Concentrations Greater than PRGs – Identified via Other Means

Tritium concentrations between 1 and 39,900 pCi/g were reported in characterization pothole and borehole subsurface soil samples collected in 2007 at the 118-B-1 (*Cleanup Verification Package for the 118-B-1, 105-B Solid Waste Burial Ground* [CVP-2007-00006], Appendix D, Tritium Plume Characterization Sampling Results) waste site. Although the tritium characterization data from 118-B-1 were not evaluated using the SSL and PRG approach described in Section 5.2,this waste site is located immediately upgradient and appears to be a likely source for the tritium plume described further in Section 5.8. Therefore, 118-B-1 is also carried forward into the FS based on an expected EPC that exceeds the 1,217 pCi/g groundwater and surface water protection PRG.

5.7.3.2 Waste Sites with COPC Concentrations Greater than PRGs – Not Carried Forward

As shown in Tables 5-9 and 5-10, the EPC for pentachlorophenol and carbon tetrachloride was greater than the groundwater and/or surface water protection PRG at two waste sites. The rationale for excluding these two waste sites is described below.

100-B-14:2 Shallow 2. Pentachlorophenol present at this waste site decision unit at an EPC of 1,900 µg/kg exceeded the groundwater protection PRG of 694 µg/kg and the surface protection PRG of 330 µg/kg. The EPC for pentachlorophenol is based on the use of a maximum detected value from the interim action verification sampling results. Pentachlorophenol was detected in 1 of 55 statistical verification samples collected from the combined 100-B-14:2 and 1607-B2 excavations. This single detection is not an accurate representation of a waste site leaching based EPC and as such does not represent a significant potential risk to groundwater or surface water at this waste site.

118-C-1 Shallow Focused. The EPC for carbon tetrachloride of 39 µ/kg exceeded the surface water protection PRG of 31 µg/kg. The EPC for this waste site is based on the maximum detected concentration of two focused verification samples with reported concentrations of 10 µg/kg and 39 µg/kg. Because the maximum detected concentration represents a single sample at a point source, the total mass of carbon tetrachloride present at this waste site does not provide a large enough reservoir to generate a leachate concentration greater than the PRG value. Recall, that the 100:0 waste distribution profile used to calculate the PRG for carbon tetrachloride presumes a 1 mg/kg concentration distributed across the full vadose zone thickness. Because the maximum detected carbon tetrachloride concentration of 39 µg/kg is significantly less than 1 mg/kg, residual carbon tetrachloride concentrations at the 118-C-1 waste site should not pose a surface water quality threat.

Groundwater and surface water protection PRGs for the pesticides alpha-BHC, beta-BHC, and heptachlor, which were detected at several waste sites above groundwater and/or surface water protection SSLs, were not developed. These three constituents have K_d values of 1.76, 2.14 and 9.53, respectively. Although they can be transported to groundwater within 1,000 years under the irrigation recharge rate used for SSL development, they do not reach groundwater within 1,000 years under natural recharge conditions. Therefore, these constituents and their associated waste sites do not pose a threat to groundwater or surface water quality under the land use conditions planned for the River Corridor. However, as described in Section 5.7.2, these three COPCs will be addressed in the FS through application of a non-irrigation institutional control.

5.8 Saturated Zone Modeling Methods and Results

A groundwater flow and contaminant transport model was constructed for the Hanford Site 100 Area to simulate future contaminant migration and to assist with developing remedial action alternatives for the individual groundwater OUs. The model development and calibration process are documented in *Conceptual Framework and Numerical Implementation of 100 Areas Groundwater Flow and Transport Model*, (SGW-46279). Hydrogeologic model inputs for the simulations were derived from *Geohydrologic Data Package in Support of 100-BC-5 Modeling* (SGW-44022). K_d values for each COPC simulated, except Cr(VI), were obtained from *Selection and Traceability of Parameters to Support Hanford-Specific RESRAD Analyses: Fiscal Year 2008 Status Report* (PNNL-18564).

5.8.1 Simulation Tools

The model domain and groundwater flow element were constructed using the U.S. Geological Survey's (USGS) MODFLOW ("A Modular Three-Dimensional Finite-Difference Ground-Water Flow Model,"[McDonald and Harbaugh, 1988]) code. Model construction was supported by *User's Documentation for MODFLOW-96, an Update to the U.S. Geological Survey Modular Finite-Difference Ground-Water Flow Model* (Harbaugh and McDonald, 1996); *MODFLOW-2000, the U.S. Geological Survey Modular Ground-Water Model – User Guide to Modularization Concepts and the Ground-Water Flow Process* (Harbaugh et al., 2000); and "*MODFLOW-2005, The U.S. Geological Survey Modular Ground-Water Model – The Ground-Water Flow Process*" (Harbaugh, 2005).

Contaminant plume migration was simulated using *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). This was supplemented with information from *MT3DMS v5.2: Supplemental User's Guide* (Zheng, 2006).

Particle tracking was performed using *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 MODFLOW code (*User's Documentation for MODFLOW-96, an Update to the U.S. Geological Survey Modular Finite-Difference Ground-Water Flow Model* [Harbaugh and McDonald, 1996]) was selected because it has the capabilities, is relatively simple to use, and can be executed on a variety of computers and operating systems without modification. MODFLOW (*User's Documentation for MODFLOW-96, an Update to the U.S. Geological Survey Modular Finite-Difference Ground-Water Flow Model* [Harbaugh and McDonald, 1996]) simulates groundwater flow using a block-centered, finite-difference grid. A finite-difference grid can simulate three-dimensional groundwater flow using one or more model layers that correspond to individual aquifers or aquitards, or that provide vertical contaminant discretization across thick aquifers and aquitards. Individual or multiple layers

corresponding to aquifers can be simulated as unconfined (for example, water table aquifers), confined, or convertible between unconfined and confined conditions.

This desktop software complies with standard methods and distinguishes between safety software and support software based on whether the software managed calculates reportable results or provides run support, visualization, or other similar functions. The following sections summarize the model development and deployment; additional details are presented in Appendix F.

5.8.2 Model Construction

The groundwater flow model grid encompasses all 100 Area OUs. The model finite-difference grid is constructed so 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 node-grid spacing is relatively coarse (about 100 m [328 ft]) throughout much of the domain, but more refined (15 m [49.2 ft]) within the 100 Area OU boundaries to provide for more detailed simulations.

5.8.2.1 Layers

The model is composed of four layers corresponding to the Hanford formation (always present in Layer 1) and Ringold 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 model domain (including 100-K and 100-D), the water table lies within the Ringold unit E sands, whereas toward the east and north of the model domain (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 (Layer 4) corresponds with 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 compiled as part of the Model Data Packages (*100-HR-3 Remedial Process Optimization Modeling Data Package* [SGW-40781]; 100-KR-4 RPO Mod Data Package [SGW-41213]; *Geohydrologic Data Package in Support of 100-BC-5 Modeling* [SGW-44022]; and *Geohydrologic Data Package in Support of 100-FR-3 Modeling* [SGW-47040]) depicts the reasonably abrupt lateral transition from the Ringold 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.

5.8.2.2 Hydraulic Conductivity

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. Following are the mean values for the aquifer hydraulic conductivity that resulted from the model calibration process:

- 6 m/day for Ringold unit E in the 100-BC region of the aquifer
- 40 m/day for the Hanford formation
- 1,000 m/day for the highly transmissive channel that runs parallel to the basalt outcrop inland of 100-BC and connects the Gable Gap with the Columbia River

Saturated zone model input parameters are summarized in Table 5-11.

Table 5-11. 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-BC-5 Operable Unit

Input Parameter (units)	Input Parameter Value
	Simulation Duration
Simulation of historic conditions (used for model calibration)	5 years – January 2006 through December 2010 Monthly stress periods
Simulation of future conditions	77 years – January 2011 through December 2077 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 Rev. 1, 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 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 2010 for following systems included: DR-5, HR-3, KX, KR4, and KW.
	Hydraulic Parameters
	Full Model Domain
Specific yield (unitless)	0.10
Specific storage (1/day)	0.000005

Table 5-11. 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-BC-5 Operable Unit

Input Parameter (units)	Input Parameter Value		
	Hanford formation	**Channel [(a)]**	**Ringold Formation**
K_h saturated horizontal hydraulic conductivity (m/d)	Spatially variable; mean zonal value in 100-BC-5 Operable Unit: 40*	1000	Spatially variable; mean zonal value in 100-BC-5 Operable Unit: 6*
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 partition coefficient (mL/g)	Contaminant-dependent and modeled using dual-domain approach; see SGW-46279 and ECF-100BC5-11-0115 for details.		

(a) Highly transmissive channel parallel to basalt outcrop inland of 100-BC-5 and connecting Gable Cap with the Columbia River

* denotes calibrated value

Note: Details on the basis for all parameters in this table are found in this Appendix (*Conceptual Framework and Numerical Implementation of 100 Areas Groundwater Flow and Transport Model* [SGW-46279]).

5.8.2.3 Recharge

An aerial recharge was specified based on information included in the *Groundwater Data Package for Hanford Assessments* (PNNL-14753). An electronic version of the recharge package developed in the PNNL report was obtained, and the data spatially distributed across the model domain. Recharge rates were subsequently adjusted during the model calibration process. Based on the results of the model calibration, the recharge value was set equal to 12 mm/yr throughout much of the model domain. Local recharge conditions (for example, surface reservoirs) were assigned recharge rates based on reported values (*In Situ Redox Manipulation (ISRM) Annual Report Fiscal Year 2007* [DOE/RL-2008-10]).

5.8.2.4 Effective Porosity, Specific Yield, and River Conductance

Initial values for the aquifer layers were identified from published sources and subsequently revised during model calibration. The final values determined through the calibration process were 18 and 10 percent, respectively, for the Hanford formation and Ringold unit E layers. 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* [PNL-10886]). Riverbed conductance values were also determined during calibration,

separately for the stretches of the Columbia River within each OU, to reflect the variability in geologic conditions.

5.8.2.5 Model Calibration

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 the use of PEST (*PEST Model-Independent Parameter Estimation: User Manual* [Doherty, 2010]) and post-processing programs that calculate water level responses to stresses. The model was calibrated to data from January 2006 to June 2009 and was validated with data from July 2009 to December 2010. The calibration step focused on the transient response of water levels to changing stresses and how they compare to values measured at wells in each OU. In addition, maps of simulated water level contours were compared to actual contoured data included in published reports. The manual comparison step was performed to ensure that the simulated hydraulic gradient magnitude and direction agree with prior independent interpretations.

5.8.3 Contaminant Transport/Plume Simulations

Contaminant transport simulations, based on current groundwater flow patterns, were run for Cr(VI) (Figures 5-14 through 5-19), strontium-90 (Figures 5-20 through 5-24), and tritium (Figures 5-25 through 5-28) for each of the four layers (Layer 1 to Layer 4) comprising the unconfined aquifer. Layer 1 corresponds to the saturated portion of the Hanford formation, and Layers 2 to 4 to Ringold unit E. The model simulations indicate that, in the absence of remedial action, Cr(VI) may persist in 100-BC groundwater for between 100 and 150 years, strontium-90 for between 50 and 100 years, and tritium between 10 and 15 years.

The modeling results are consistent with observed migration patterns of Cr(VI), which suggest very slow movement in groundwater and persistent concentrations for nearly 40 years since reactor operations ceased. Recent studies by PNNL (*Geochemical Characterization of Chromate Contamination in the 100 Area Vadose Zone at the Hanford Site* [PNNL-17674]) suggest that Cr(VI) within soil of the 100 Area exhibits migration characteristics that may be more complex than can be represented using simple advection. According to these tests, although most of the mass is moderately mobile and migrates by advection, Cr(VI) mass can be held in heterogeneous parts of the aquifer of low hydraulic conductivity.

Hexavalent chromium held within the low hydraulic conductivity portion of the aquifer represents a continuing source of low-level chromium release to the higher hydraulic conductivity portion of the aquifer. Mass transfer from the low to high hydraulic conductivity domain occurs primarily through diffusion. 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 a mobile and low mobility domain. Advective transport occurs predominantly in the mobile domain, while mass transfer occurs between the mobile and low mobility domains. This mass transfer process is the primary cause for Cr(VI) longevity at 100-BC.

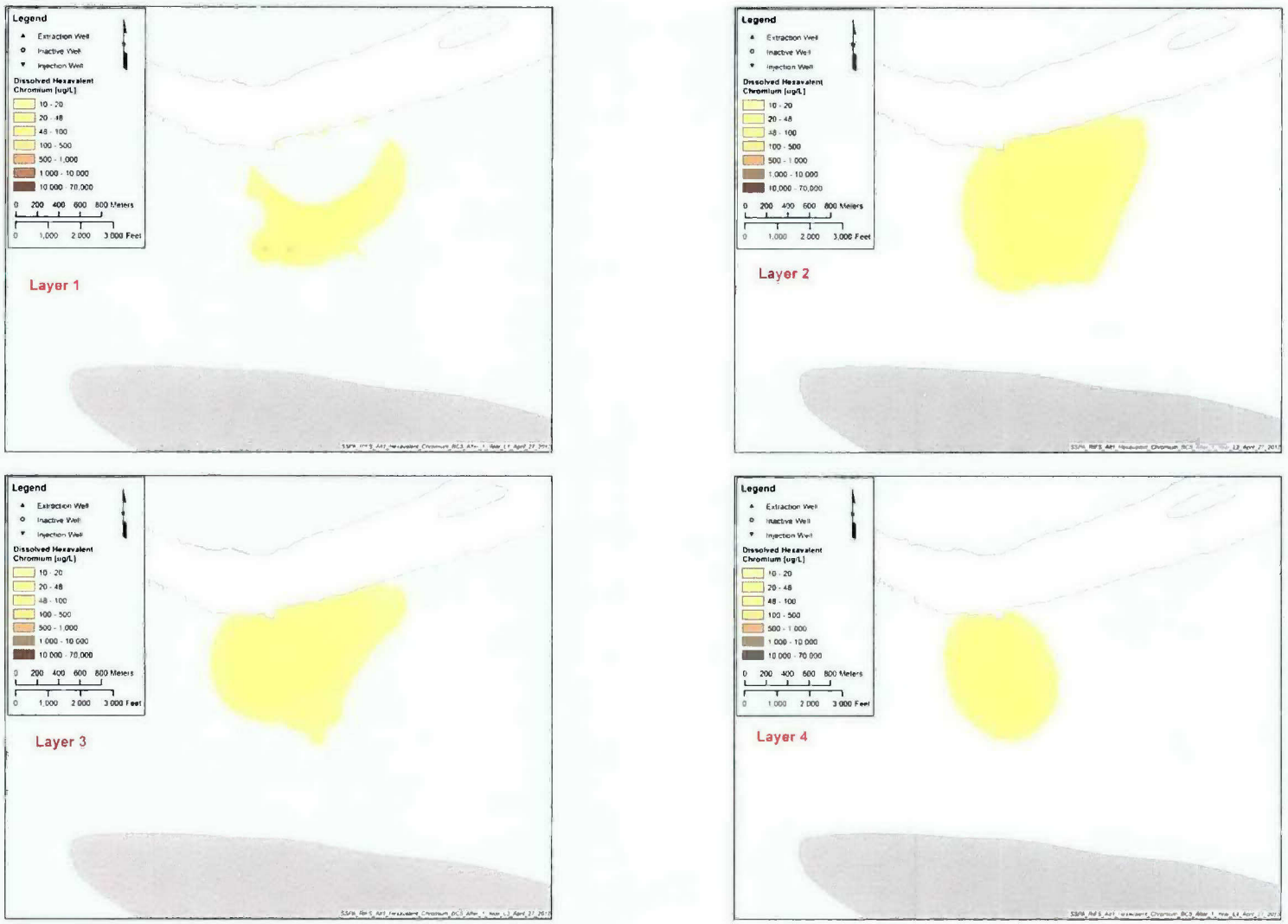


Figure 5-14. Model Simulated Cr(VI) Distribution in Unconfined Aquifer Groundwater – Year 2010

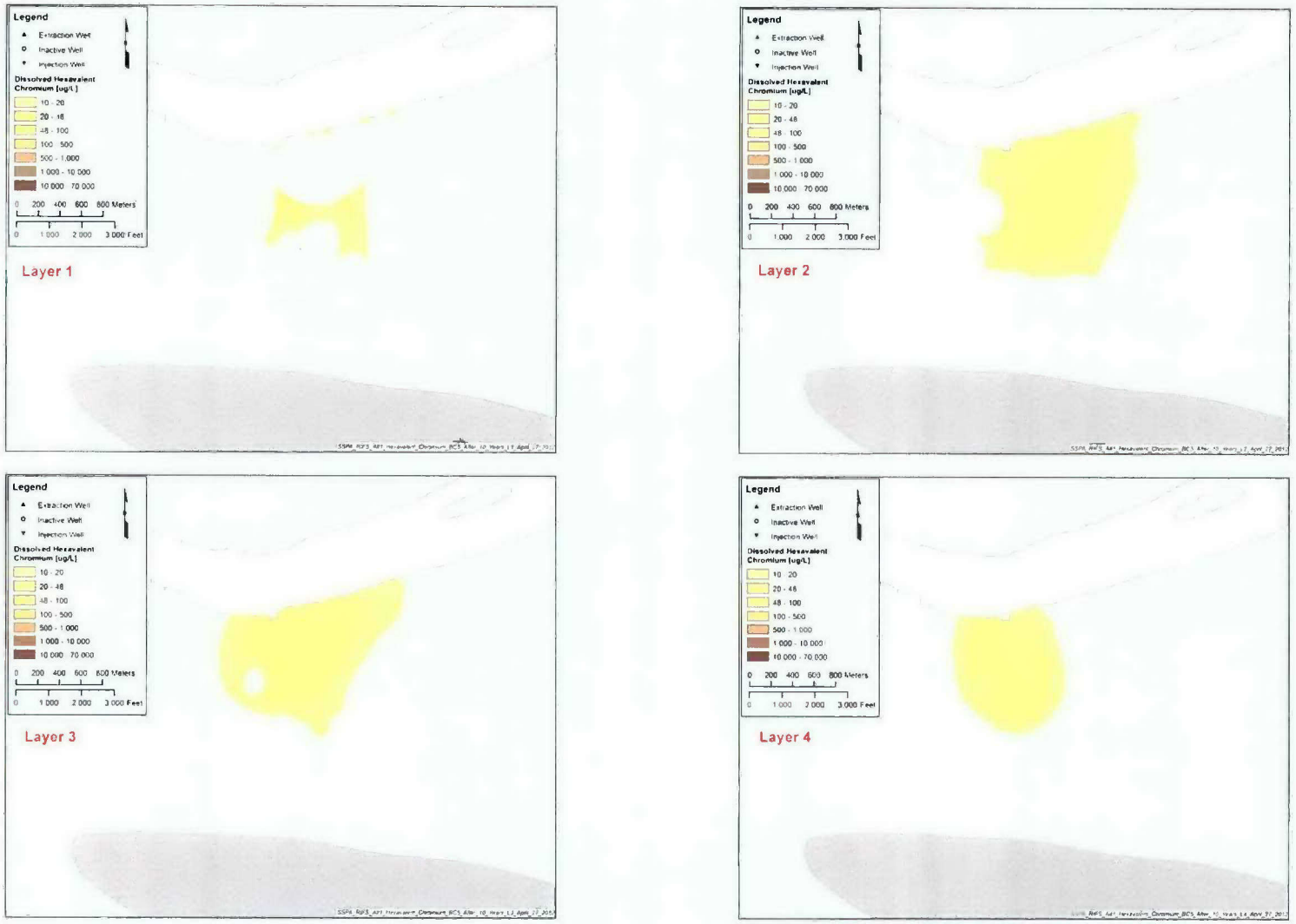


Figure 5-15. Model Simulated Cr(VI) Distribution in Unconfined Aquifer Groundwater – Approximate Year 2020

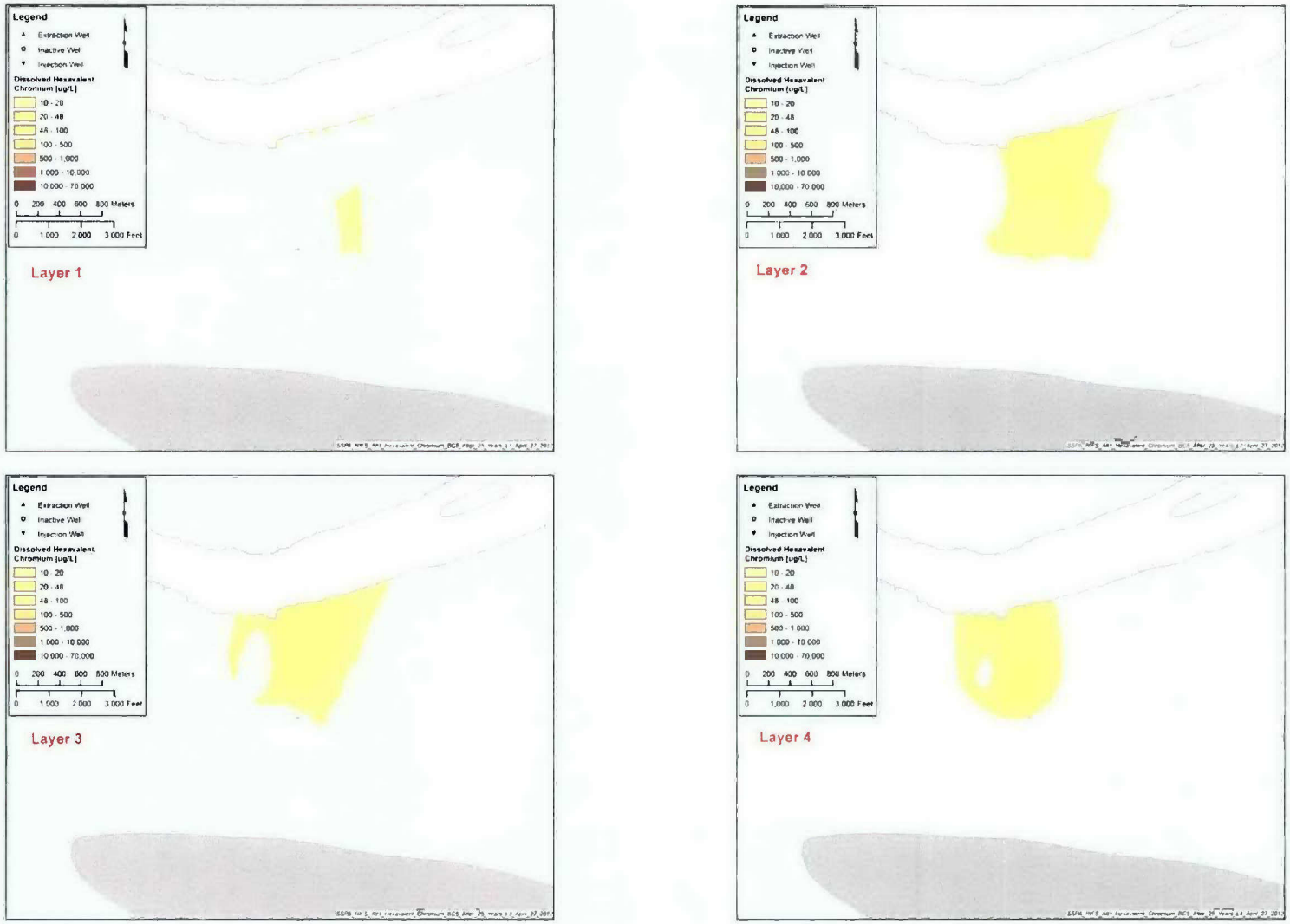


Figure 5-16. Model Simulated Cr(VI) Distribution in Unconfined Aquifer Groundwater - Approximate Year 2035

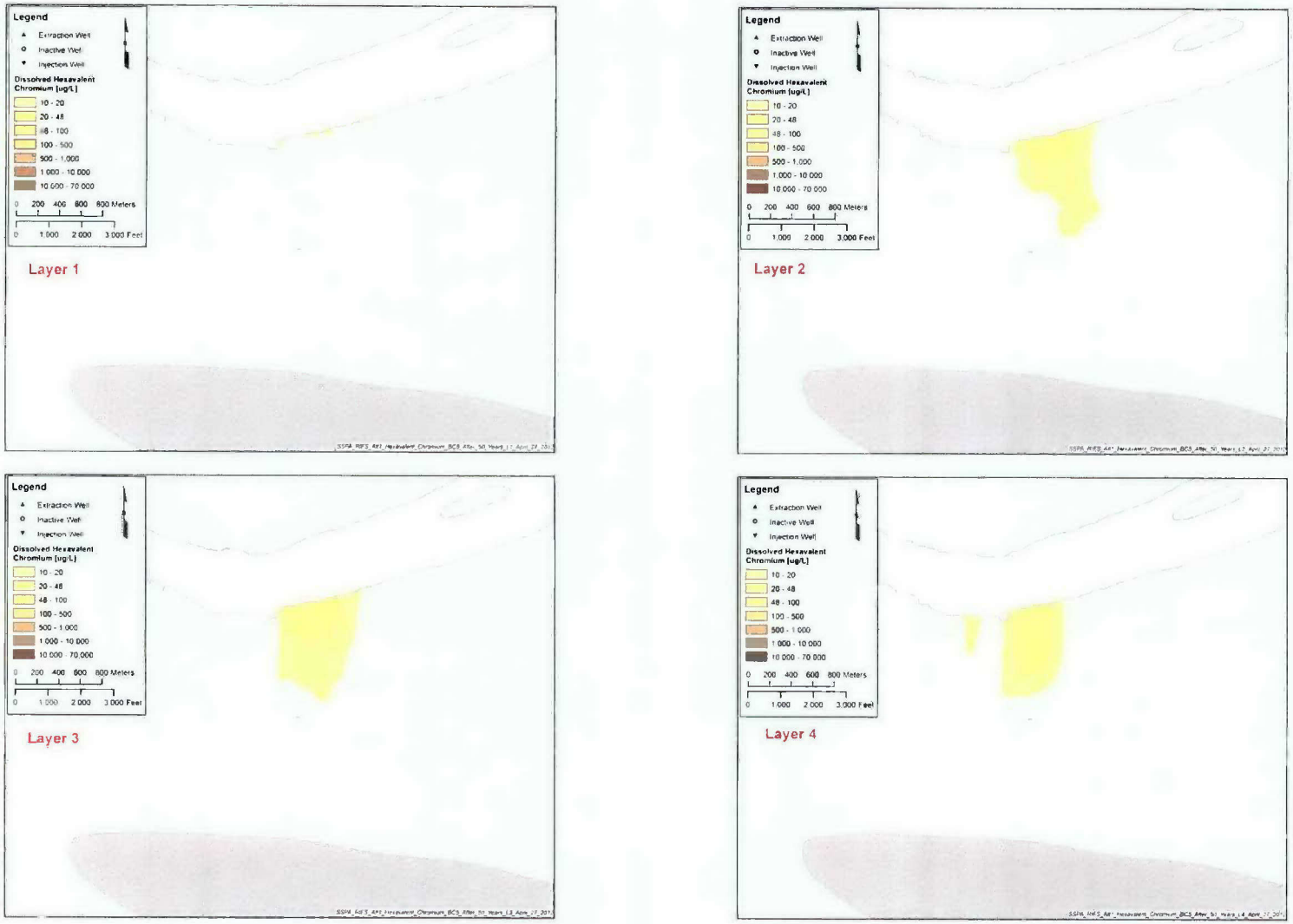


Figure 5-17. Model Simulated Cr(VI) Distribution in Unconfined Aquifer Groundwater – Approximate Year 2060

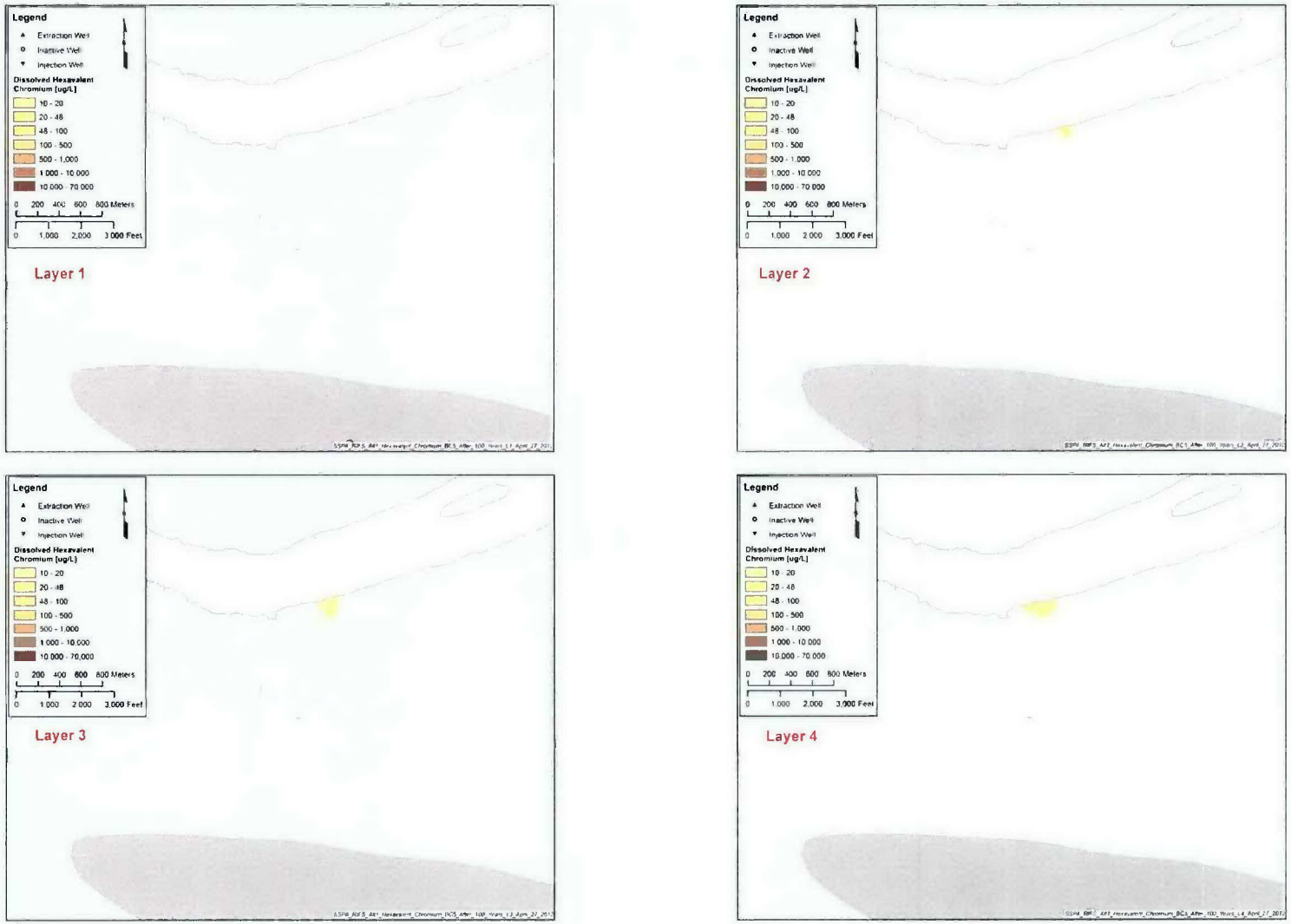


Figure 5-18. Model Simulated Cr(VI) Distribution
in Unconfined Aquifer Groundwater – Approximate Year 2110

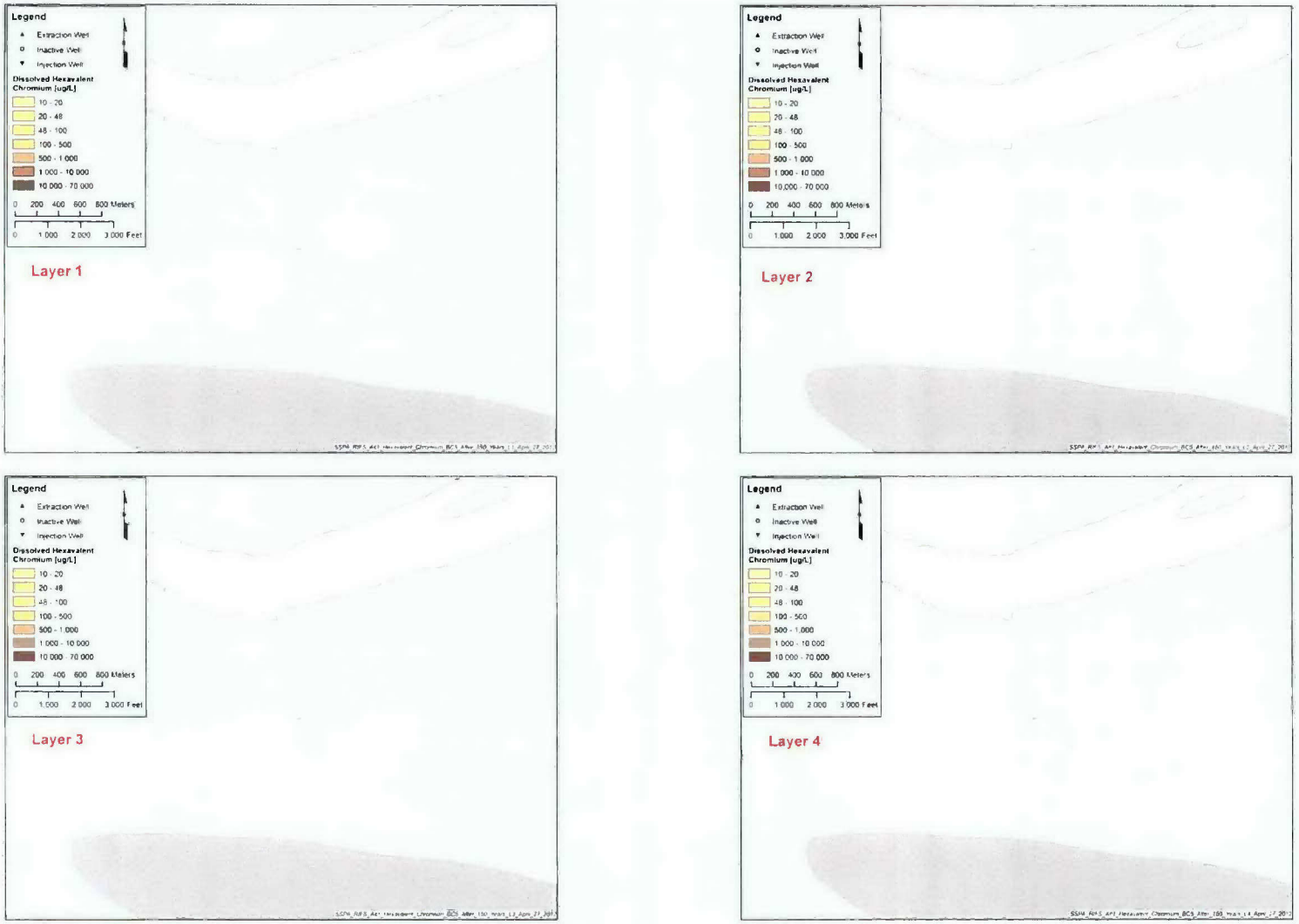


Figure 5-19. Model Simulated Cr(VI) Distribution in Unconfined Aquifer Groundwater – Approximate Year 2160

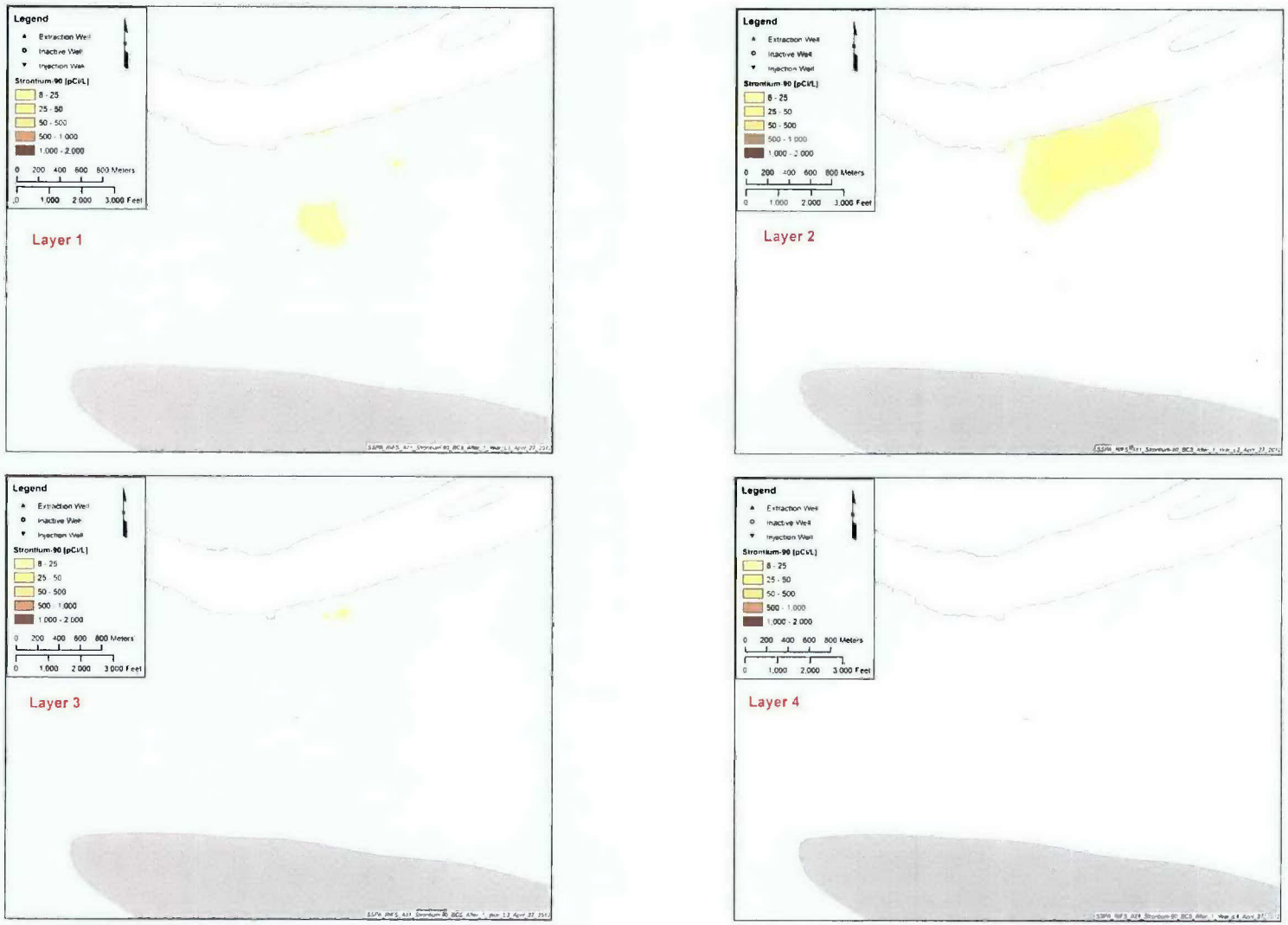


Figure 5-20. Model Simulation of Strontium-90 Distribution in Unconfined Aquifer Groundwater—Approximate Year 2010

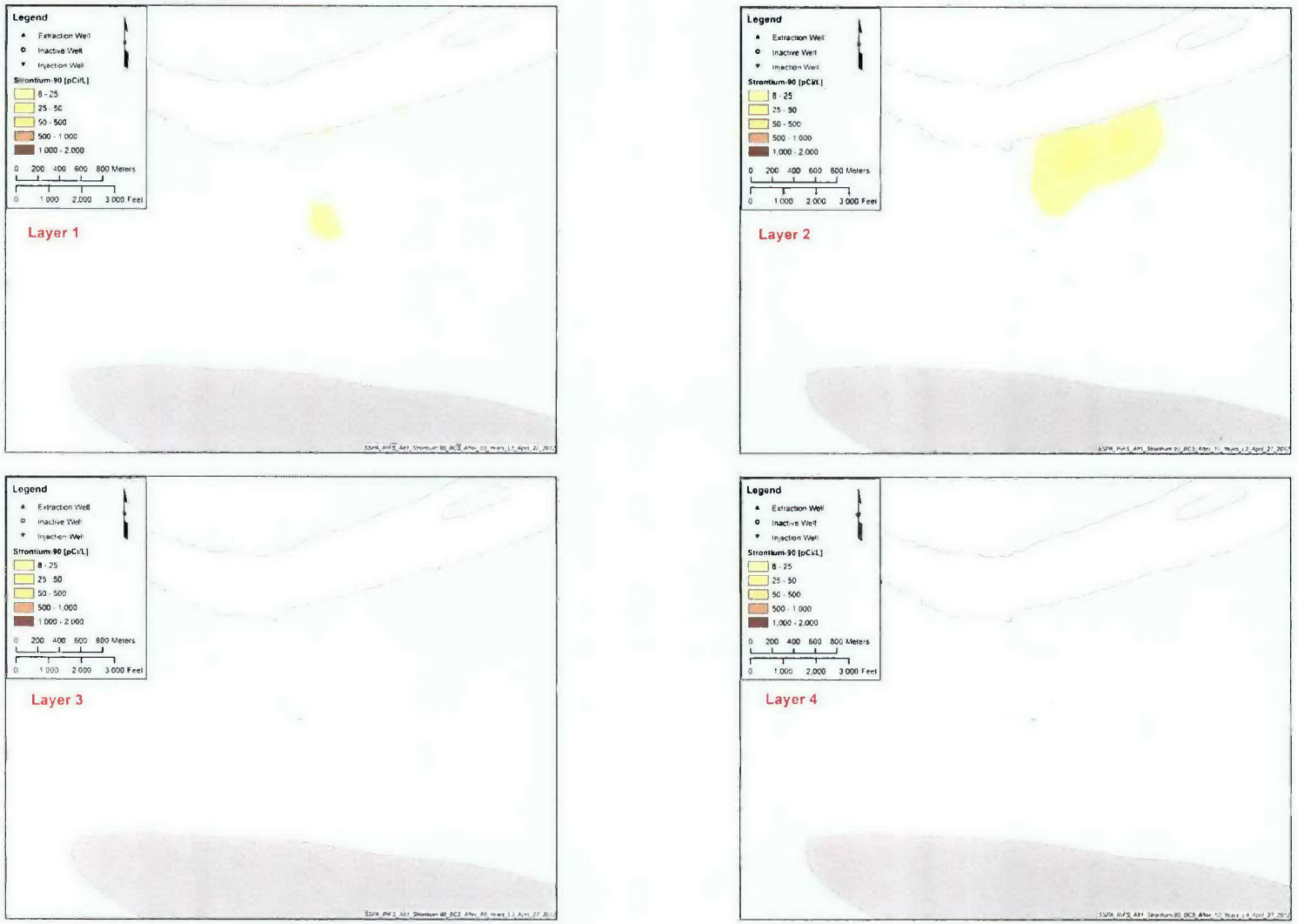


Figure 5-21. Model Simulation of Strontium-90 Distribution in Unconfined Aquifer Groundwater—Approximate Year 2020

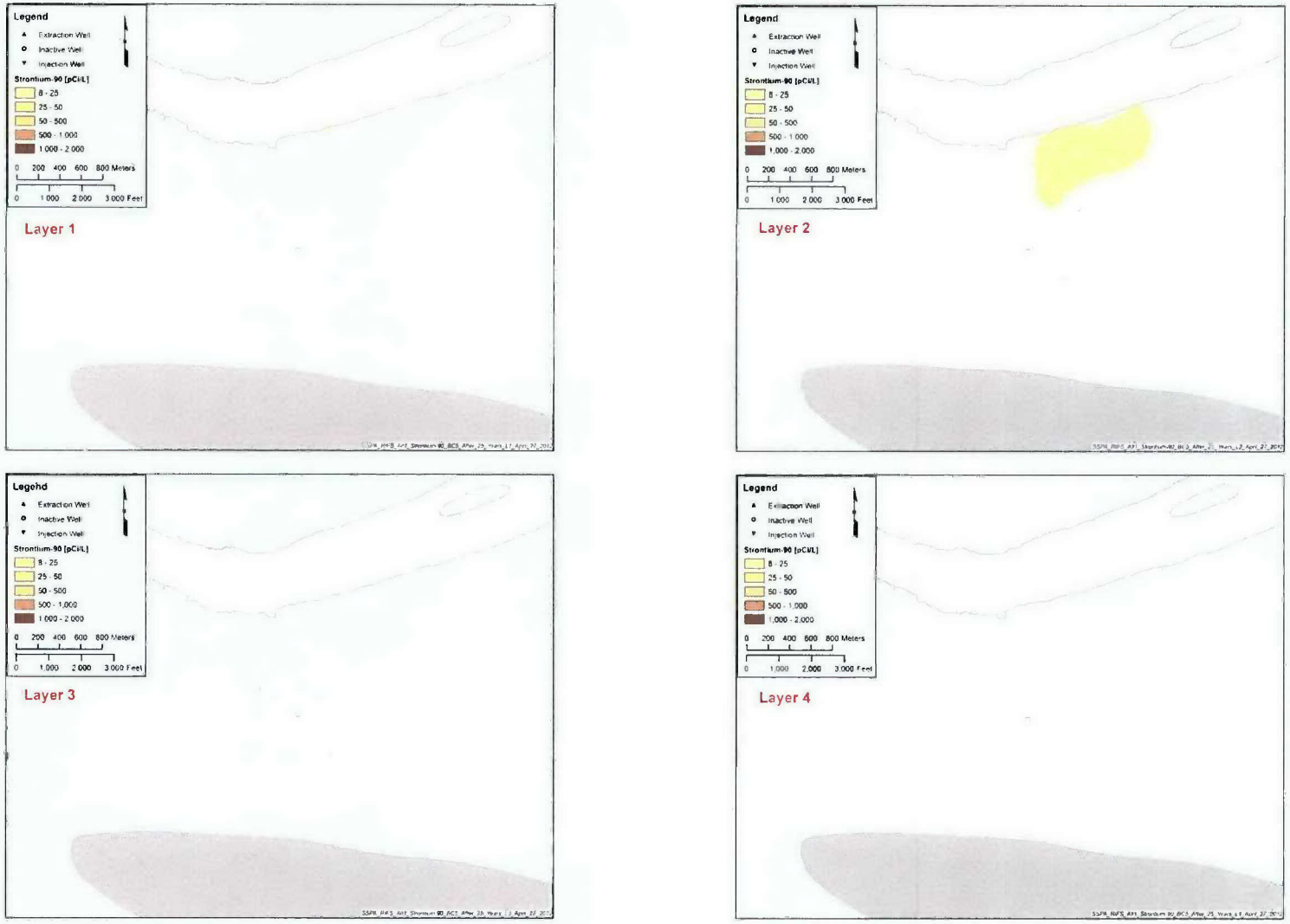
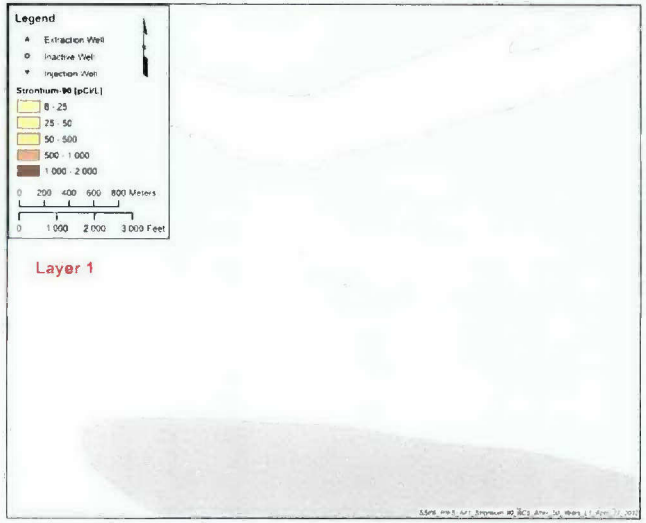
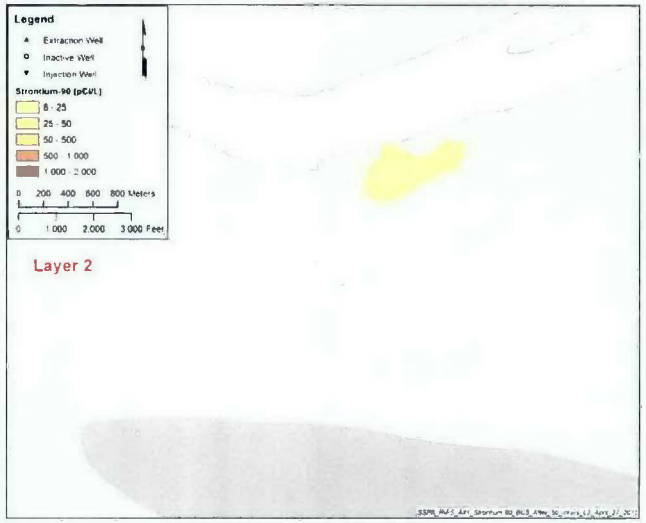
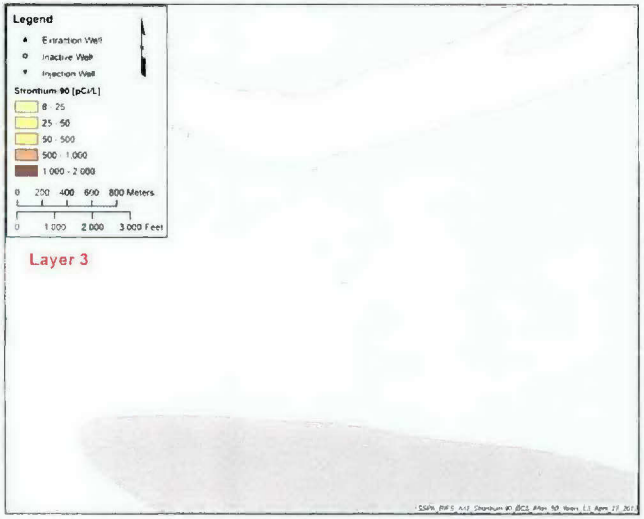


Figure 5-22. Model Simulation of Strontium-90 Distribution in Unconfined Aquifer Groundwater—Approximate Year 2035







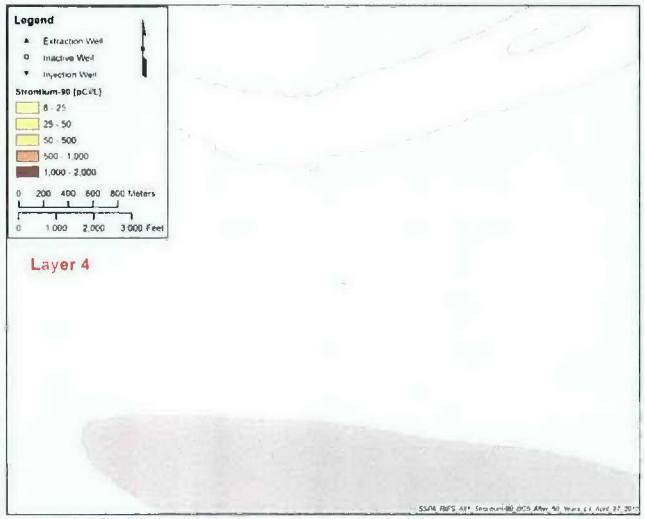


Figure 5-23. Model Simulation of Strontium-90 Distribution in Unconfined Aquifer Groundwater—Approximate Year 2060

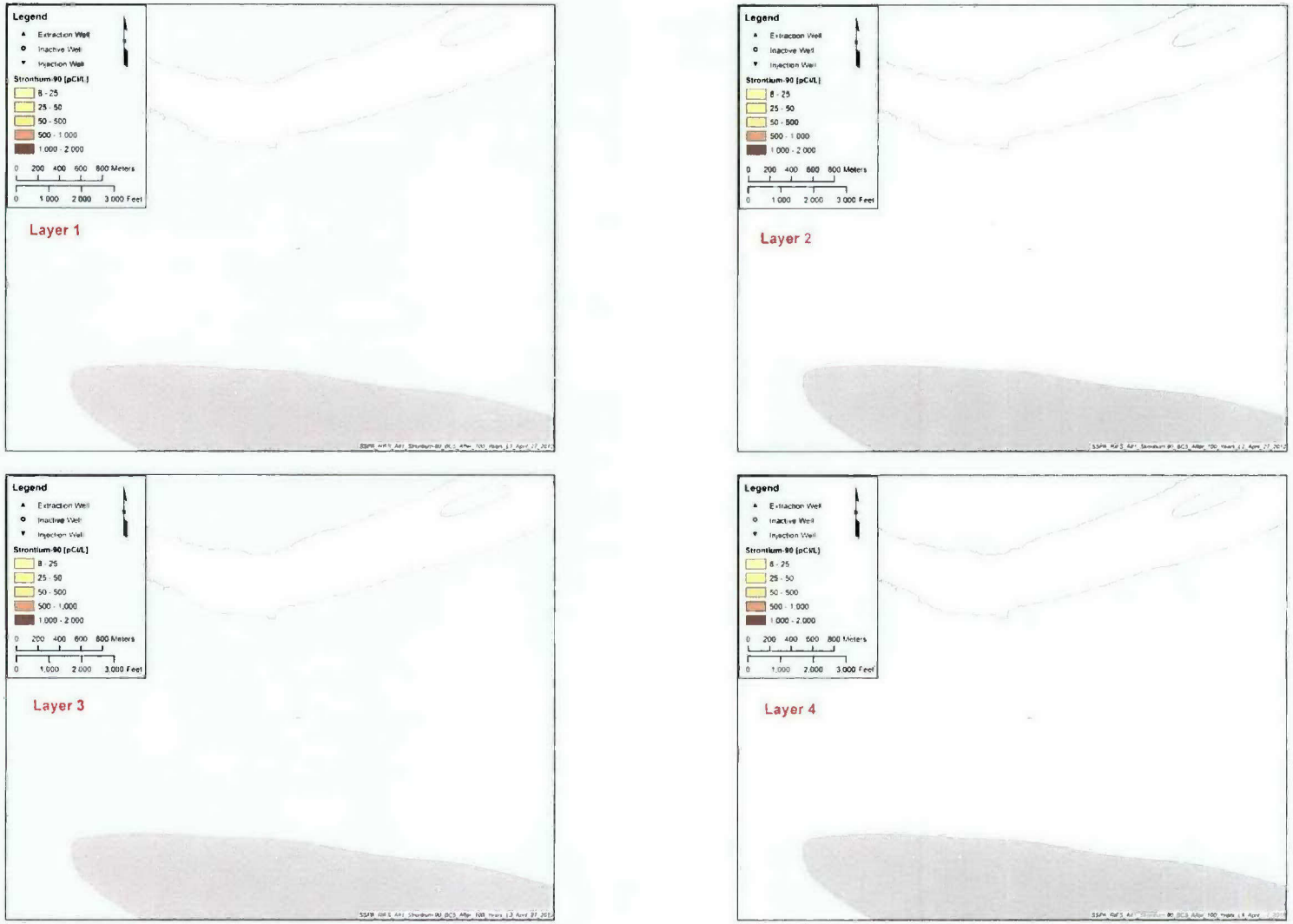


Figure 5-24. Model Simulation of Strontium-90 Distribution in Unconfined Aquifer Groundwater—Approximate Year 2110

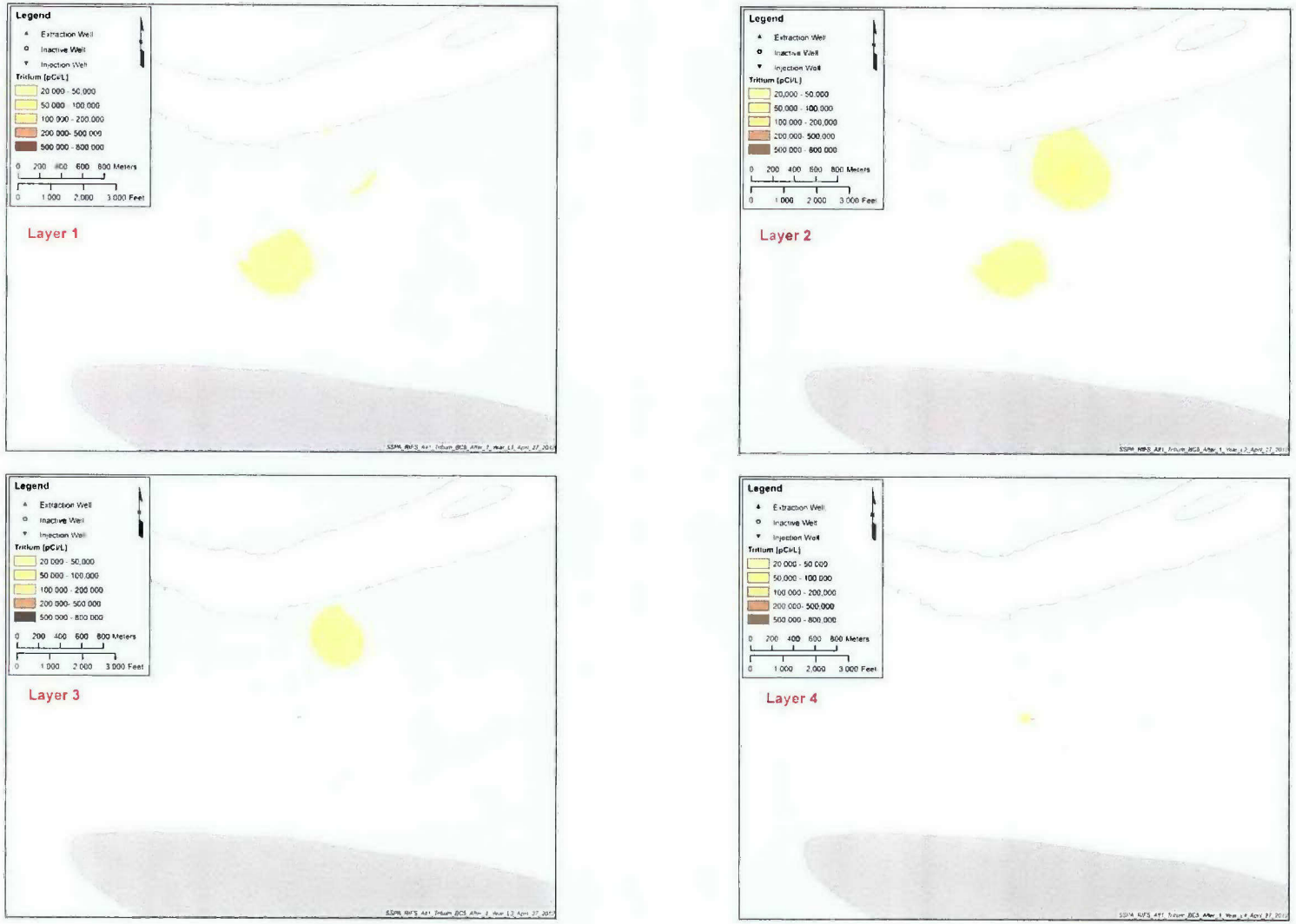


Figure 5-25. Model Simulation of Tritium Distribution in Unconfined Aquifer Groundwater—Approximate Year 2010

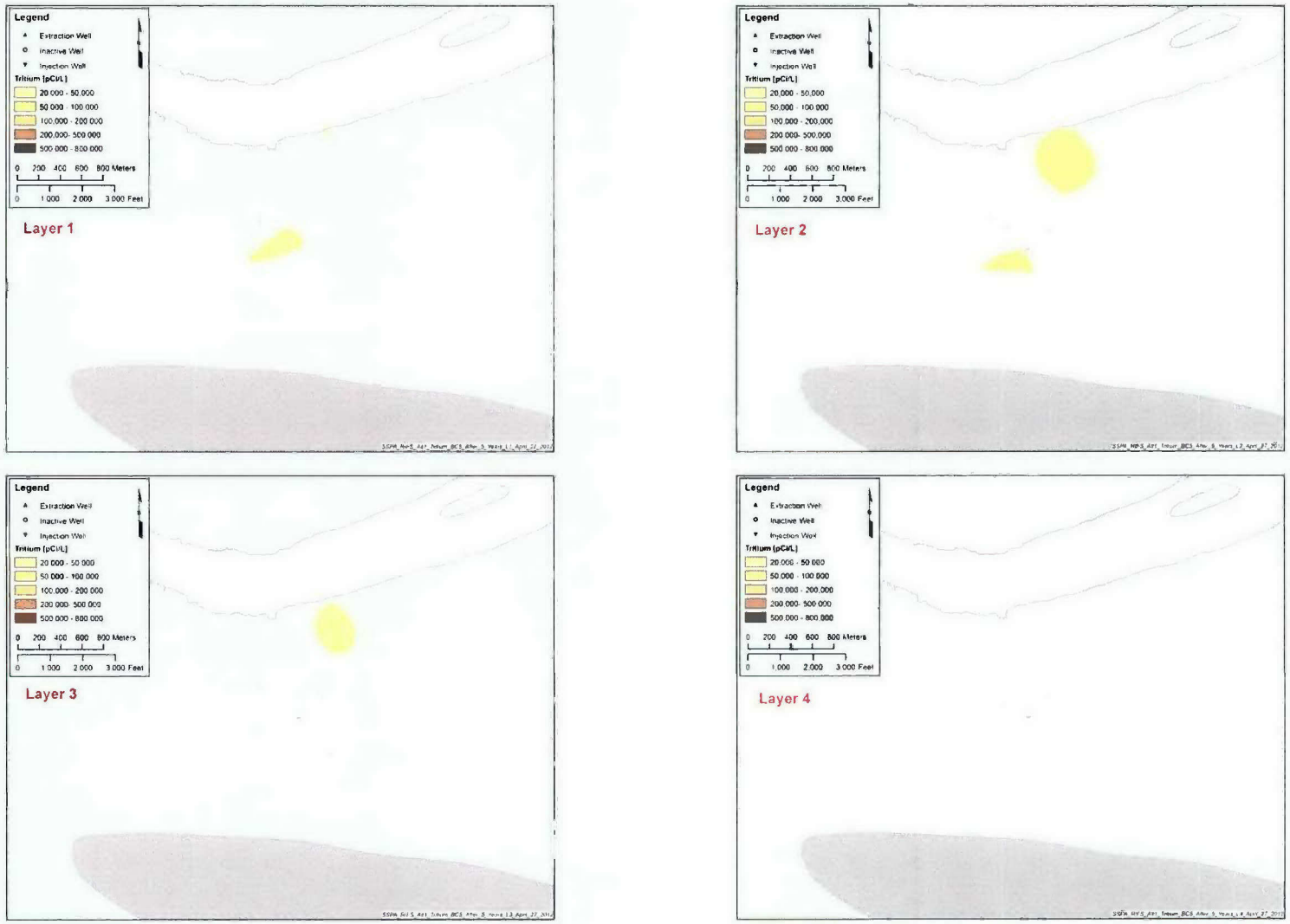


Figure 5-26. Model Simulation of Tritium Distribution in Unconfined Aquifer Groundwater—Approximate Year 2015

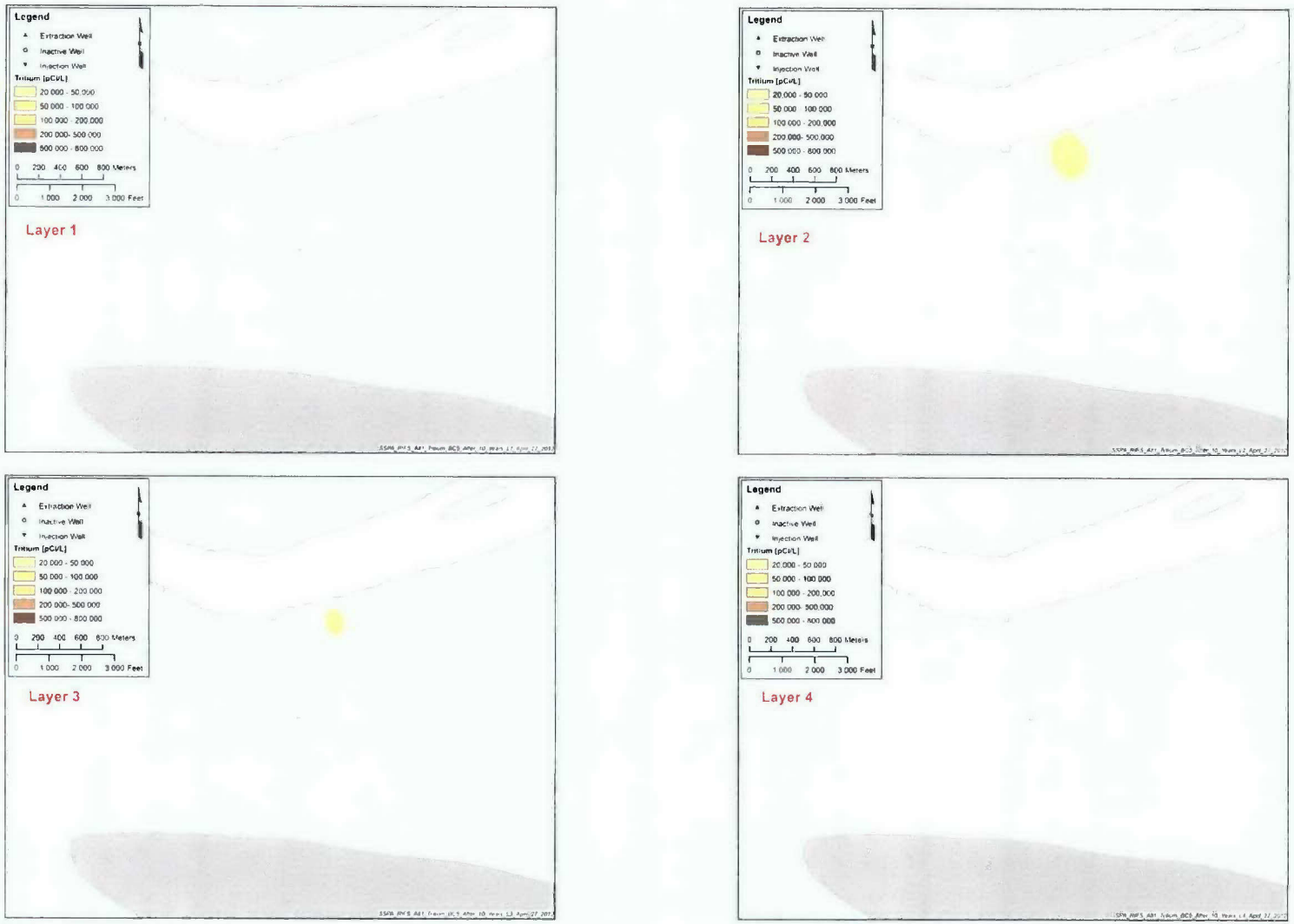


Figure 5-27. Model Simulation of Tritium Distribution in Unconfined Aquifer Groundwater—Approximate Year 2020

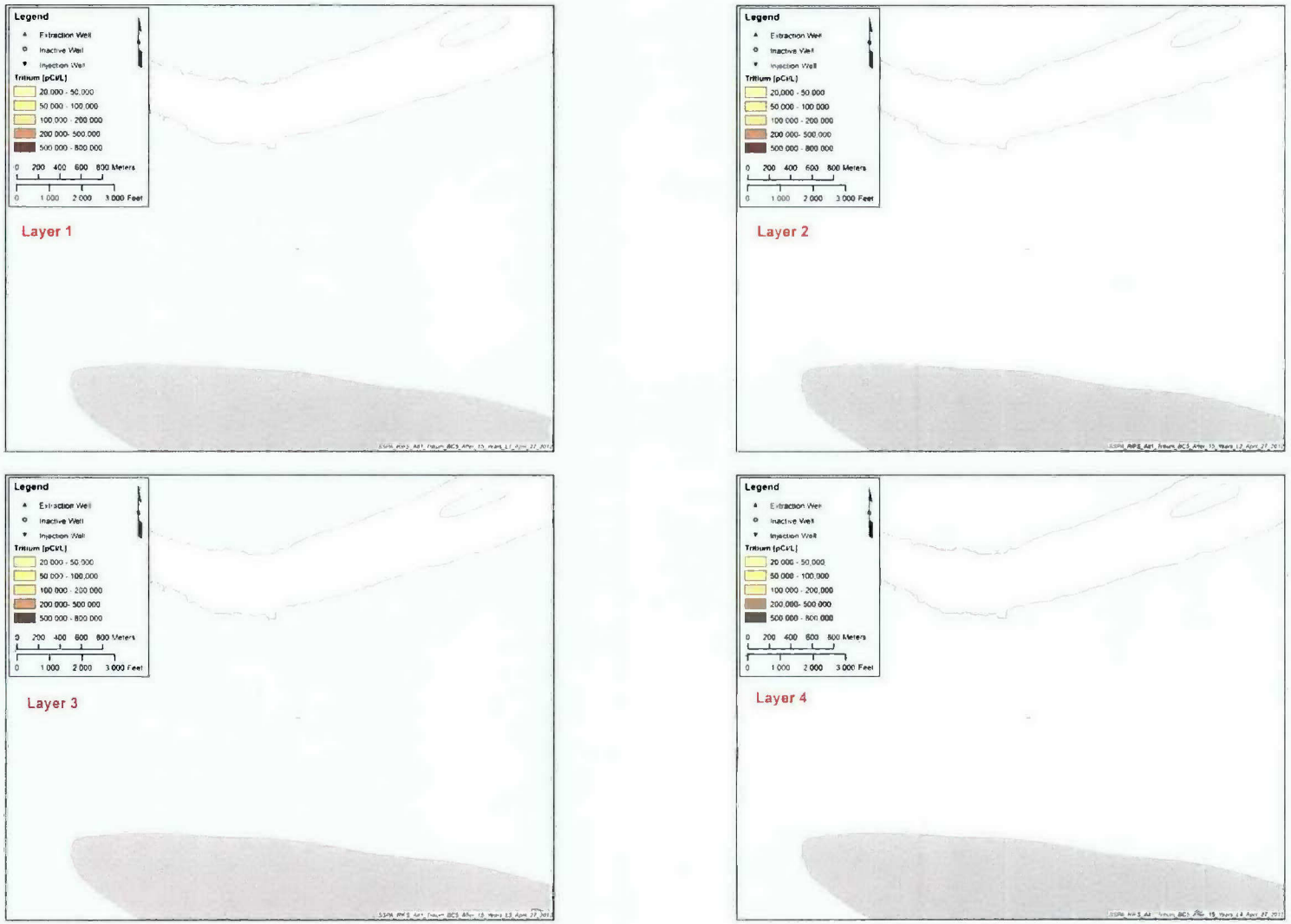


Figure 5-28. Model Simulation of Tritium Distribution in Unconfined Aquifer Groundwater—Approximate Year 2025

Contaminant transport in 100-BC groundwater could also be attributed to the relatively low groundwater velocities that are calculated from continuous transducer monitoring data across the OU in Ringold unit E where Cr(VI) is primarily found. Low groundwater velocities are the result of a nearly flat horizontal hydraulic gradient across 100-BC and low hydraulic conductivities that characterize the aquifer in this area. In addition, the aquifer geometry and corresponding hydraulic properties exert significant influence on plume migration patterns and corresponding travel times for the portions of the plume that are present in the more transmissive Hanford formation compared to the less transmissive Ringold unit E. Although the water table fluctuates between the more transmissive sands and gravels of the Hanford formation and Ringold unit E, the exact location of the contact between the two formations is the subject of an ongoing characterization process. The groundwater model implements the current interpretation of the Hanford formation/Ringold unit E contact as included in the Model Data Package for 100-BC-5, with the addition of a highly transmissive paleochannel that appears to be present inland of the reactor area and abuts the basalt outcrop, connecting the Gable Gap with the Columbia River upstream from 100-BC.

5.9 Uncertainties in Simulating Contaminant Fate and Transport

EPA guidance to remedial program managers states that "...the objective of the RI/FS is not to remove all uncertainty, but to gather information sufficient to support an informed risk management decision regarding what remedy appears to be most appropriate given what is known about the site" (*Scoper's Notes, An RI/FS Costing Guide, Bringing in A Quality RI/FS On Time and Within Budget* [EPA/540/G-90/002]). 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 remediation scenarios. Although these uncertainties must be considered, conservative assumptions employed in developing the vadose zone and saturated zone models are expected to lessen the effects of uncertainty in remedial action decision-making.

5.9.1 Uncertainty in the Conceptual Site Model

When using models to predict future contaminant fate and transport, the main uncertainty is often in the CSM. 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 obtain best-estimate predictions, 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 actual contaminant distributions that vary from the simulations. For example, the assumption of constant effective porosity and hydraulic conductivity for a given stratigraphic unit is assumed in calculating SSLs and PRGs protective of groundwater and surface, and converting water flux (Darcy Velocity) calculated by MODFLOW (*User's Documentation for MODFLOW-96, an Update to the U.S. Geological Survey Modular Finite-Difference Ground-Water Flow Model* [Harbaugh and McDonald, 1996]) to average linear groundwater velocity as used in MODPATH and MT3DMS for fate and transport calculations. Heterogeneity in the form of lenses, bar structures, and overbank deposits is common at a scale smaller than the model grid size employed for the 100-BC groundwater flow model and is not accounted for in the STOMP SSL and PRG model simulations. 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. This is one reason why models make better predictions of bulk-averaged quantities than they make of values at specific point locations and times.

The effects of these local-scale uncertainties on predictions of groundwater and surface water protection SSLs and PRGs are minimized to the practical extent possible. The effects are minimized through conservatism such as selecting the lowest SSL and PRG resulting from the STOMP simulations. In addition, the effects are minimized by using predictions that the models are better suited to simulating (such as area-wide average values and area wide-changes in concentrations) than predictions that, by their nature, are greatly impacted by such uncertainty or variability (such as point concentrations at specific locations and times). Inferences drawn from the model simulations should also focus on these types of model outputs.

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 due to 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, are likely the greatest contributors to overall uncertainty in the initial contaminant distribution.

5.9.3 Uncertainty in Contaminant Transport Parameters

Parameters that affect contaminant transport include the K_d value of the contaminant and soil porosity. Soil porosity is variable throughout the soil column based on compaction and heterogeneity. The more porous the soil, the faster the contaminant can potentially move. Porosity values used in modeling assume the highest values for the soil types present, thereby over-representing the rate of movement through the vadose zone.

The K_d value represents the propensity for an analyte to move through the soil instead of binding to it. A high K_d value is typically found in contaminants such as PAHs, 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 (ionic strength) 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. The K_d values tend to represent the more mobile K_d conditions for a particular contaminant.

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 present in the 100 Area.

Assumptions within the model input parameters have an effect on the simulation outcomes. The key assumptions used for 100-BC are as follows, with other assumptions presented in Appendix F:

- The vadose zone is considered homogeneous in nature, without accounting for the presence of thin, finer-grained material, which can retard the downward migration of contaminants.
- 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 five years, with bare soil present for the first five years. This assumption results in more water infiltrating to the vadose zone than may actually occur.

- Groundwater is assumed to have negligible mixing with the Columbia River. In calculating the values for surface water protection, the point of compliance is assumed to occur in groundwater at the downgradient boundary of the waste site. No attenuation or decay of contaminants is assumed to occur between the downgradient boundary of the waste site and the river.

- The surface cover used in development of SSLs and PRGs includes a progression from bare ground through a developing shrub-steppe plant community to a long-term mature shrub-steppe community. This surface cover may be subject to specific uncertainty due to 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. The effect 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 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, which necessarily incorporate the effects of the history of all land surface changes at the measurement sites, including past wild fires.

- 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.

- The median hydraulic gradient value for each source area may be too large by many times for waste sites near the Columbia River and may be several times too large for waste sites that are far inland from the river.

- The assumption of a 5 m (16.4 ft) thick aquifer is conservative as most locations within 100-BC have an aquifer thicknesses greater than 5 m (16.4 ft).

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 [49.2 ft]) in 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-BC are as follows, with other assumptions and the specific input parameters presented in Appendix F:

- Predictive simulations were based on transient state (that is, time varying) conditions in the aquifer that reflect water level changes due to river stage variation. The modeling period corresponds to 150 years (approximately 2010 to 2160). For the period 2010 to 2035, 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 from these conditions.

- Groundwater flow is simulated as three-dimensional using four layers to represent the Hanford formation (always present in Layer 1) and Ringold 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. At 100-BC, the water table fluctuates between Ringold unit E and the Hanford formation.

- 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.

- The mean values for the aquifer hydraulic conductivity that resulted from the model calibration process are 6 m/day for Ringold unit E in the 100-BC-5 region of the aquifer and 63 m/day for the Hanford formation. The mean is 1,000 m/day for the highly transmissive channel that runs parallel to the basalt outcrop inland of the 100-BC-5 OU and connects the Gable Gap with the Columbia River.

- Areal recharge from precipitation was specified based on information included in the *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 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.

- 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 COPC in groundwater within the 100-BC-5 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 liquid and solid wastes associated with nuclear material production represent the primary contaminant sources at 100-BC. These releases ceased with the end of active reactor operations in 1974 but resulted in the presence of secondary contaminant sources in the vadose zone and created contaminant plumes in unconfined aquifer groundwater.

The waste site EPCs for each remediated waste site and COPC were compared to the PRGs under the 100:0 and 70:30 waste site distribution scenarios. After excluding COPCs with peak concentration times greater than 1,000 years, only two waste sites (118-B-1 and 118-B-6 [both tritium]) were identified with EPCs greater than groundwater and/or surface water protection PRGs. Further evaluation of the vertical distribution of tritium at these two waste sites indicates that most of the mass occurs within the upper portion of the vadose zone and, therefore, the 100:0-based PRG may be overly conservative for these two waste sites. However, based on the level of PRG exceedance observed, and presence of a tritium plume within the unconfined aquifer, these two waste sites will be carried forward into the FS based on groundwater and/or surface water protection PRG exceedance.

Groundwater contaminant plume simulations indicate that Cr(VI) will persist in groundwater for between 100 and 150 years; strontium-90 will decay to concentrations below its 8 pCi/L drinking water standard within 50 to 100 years; and tritium in less than 15 years.

Chapter 5 describes how contaminants migrate and predicts their potential pace 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 contaminants into contact. The ways the contaminants could come into contact with and impact human health and the environment are called pathways. Chapter 6 addresses the human health pathway and evaluates scenarios of how humans might come into contact with contaminants in the setting with resultant health impacts. Chapter 7 addresses the biological receptor pathway and evaluates scenarios of how plant, animal, bird, or invertebrate species might come into contact with contaminants in the setting and the impact.

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6 Human Health Risk Assessment

The integration of past and ongoing HHRAs supports the development of remedial alternatives for waste sites and contaminated groundwater in the 100-BC 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, and polycyclic aromatic hydrocarbons.
- The principal contaminants in groundwater are Cr(VI), strontium-90, tritium, and trichloroethene.
- The baseline risk assessment identified Cr(VI), strontium-90, and tritium as 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 provides the basis for action.

As described in the previous chapters, the remedial actions completed to date in the River Corridor were implemented primarily under interim action RODs. Under CERCLA, a baseline risk assessment (BRA) is required to characterize current and potential threats to HHE before final action RODs for final remedies can be issued. The RCBRA Report (DOE/RL-2007-21) was prepared to address the regulatory requirement that a BRA be performed. The RCBRA Report (DOE/RL-2007-21) 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 Report (DOE/RL-2007-21), as described in Section 1.1 of the RCBRA:

> *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 Guidance for Superfund Volume I Human Health Evaluation Manual (Part A): Interim Final* (hereinafter called the Risk Assessment Guide; EPA/540/1-89/002), a BRA 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 (i.e., under an assumption of no action)."

The BRA 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 BRA provides information to assist in the development, evaluation, and selection of appropriate response alternatives. The results of the BRA are used for the following:

- Determine whether additional response action is necessary at a site.
- Support development of PRGs.

- Support selection of the "no action" remedial alternative where it is appropriate.
- Document the magnitude of risk and primary contributors (e.g., 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 more rapidly. However, final remedy selection (development of Final Action RODs) must be completed in order for the NCP (40 CFR 300, Appendix B, "National Priorities List") 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 BRA (*Risk Assessment Work Plan for the 100 Area and 300 Area Component of the RCBRA* [DOE/RL-2004-37]). The RCBRA Report (DOE/RL-2007-21) HHRA and the companion ecological risk assessment 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. The site-specific risk information provided by the RCBRA Report (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 Report (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 RAGs for the interim action ROD cleanups in the 100 Areas
- Assessment of risks for several yet-to-be remediated waste sites using a broad range of exposure scenario
- 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 Report (DOE/RL-2007-21). However, for the 100-BC RI/FS, the unrestricted land use PRGs were developed in the RCBRA Report using the approach presented in the interim action ROD. This approach was brought up to date using current risk assessment guidance and methodologies. The following issues are addressed in this chapter as part of the integration of RCBRA Report (DOE/RL-2007-21) 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 Report (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 (i.e., waste sites cleaned up after the analysis conducted in the RCBRA Report [DOE/RL-2007-21])

[1] The term "broad area" is used in the RCBRA (DOE/RL-2007-21) 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.

- Use of EPCs consistent with the waste site decision units (e.g., shallow zone, deep zone) and based on current EPA guidance

The following sections discuss the integration of the RCBRA Report (DOE/RL-2007-21) and the RI/FS risk assessment:

- Section 6.1.1 summarizes the evaluation of residual risks performed in the RCBRA Report (DOE/RL-2007-21) for waste sites cleaned up under the interim action ROD. The results from this soil risk assessment have been compared with the PRGs developed in the RCBRA Report (DOE/RL-2007-21) for use in the RI/FS.

- Sections 6.1.2 and 6.1.3 describe unrestricted use and other scenarios used in the River Corridor, their associated uncertainties, and how they have been incorporated into the RI/FS.

The human health risk assessment 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 groundwater risk assessment.

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 Report (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 data set. The groundwater 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 chemical-specific ARARs (Section 6.3.6), risk characterization results of the EPA tap water scenario (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 nearshore environment from the RCBRA (Section 6.4.1), and conclusions from the CRC (Section 6.4.2). 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 Report (DOE/RL-2007-21). It also compares the results from screening level evaluation to the methodology used to develop the interim action RAGs and describes how analytical data from CVP/RSVP were used in the screening evaluation. Finally, the screening level risk results from the RCBRA) are compared to the results of the supplemental soil risk evaluation. The risk results from the evaluation are based on guidance and exposure assumptions that have been updated since the interim action RAGs were published. The methods used in the supplemental risk evaluation are described in Sections 6.1.2 and 6.1.3.

Chapter 2 of the RCBRA Report (DOE/RL-2007-21) presents a screening-level assessment of residual cancer risks and noncancer hazards for the remediated wastes sites using the exposure scenarios that were the basis of the residential RAGs for the Interim Action ROD cleanups in the 100 Areas. 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 cleanups for the 100 Areas were based on an unrestricted scenario that was the basis for the RAGs. The interim action ROD residential scenario for radionuclides is a Rural Residential scenario that, in addition to direct contact, includes food chain exposure pathways (e.g., 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 RAG for arsenic was based on the MTCA Method A soil cleanup level (WAC 173-340-740). The RAG for lead was calculated using the *Guidance Manual for the Integrated Exposure Uptake Biokinetic Model for Lead in Children* (EPA/540/R-93/081). It should be noted that PRGs for the residential scenario used in the supplemental soil risk evaluation 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 Report (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.

Forty-five waste sites from the 100-BC Source OU were evaluated in the RCBRA Report (DOE/RL-2007-21). Thirty-six additional waste sites with closeout verification data have been remediated at the 100-BC Source OU since 2005, and are not addressed in the RCBRA. An additional eight sites in the 100-BC Source OU have been remediated, but are included in another waste site's sampling and closeout documentation. Residual cumulative cancer risks from chemicals evaluated in the RCBRA Report (DOE/RL-2007-21) are less than 1×10^{-5} using the Interim Action ROD residential scenario (i.e., MTCA [WAC 173-340] Method B Unrestricted Use scenario). 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:

- 116-B-11
- 116-B-14
- 116-B-6A
- 116-C-1
- 116-C-2A
- 116-C-5
- 116-C-6
- 132-B-6

The noncancer hazard indexes (HIs) for chemicals do not exceed a threshold of 1.0 at the 45 remediated waste sites. A summary of the risk assessment results for a residential scenario using approaches from both RCBRA Report (DOE/RL-2007-21) and the RI/FS is provided in Table 6-1 through Table 6-3.

[3] Note that for beryllium, cadmium, and Cr(VI), the Interim Action RAG 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-B-11	Shallow Focused	--	--	--	--
100-B-14:5	Shallow Focused	--	--	--	--
100-B-14:6	Shallow Focused	--	--	--	--
100-B-14:7	Shallow Focused	--	--	--	--
100-B-16	Shallow Focused	--	--	3.1×10^{-7}	None
100-B-5	Shallow	--	--	--	--
100-B-8:1	Shallow	--	--	--	--
100-B-8:2	Shallow_1	--	--	--	--
	Shallow_3	--	--	--	--
100-C-3	Shallow	--	--	3.9×10^{-9}	None
116-B-1	Shallow	--	--	--	--
116-B-10	Shallow	--	--	--	--
116-B-11	Shallow	--	--	--	--
116-B-12	Shallow	--	--	--	--
116-B-13	Shallow	--	--	--	--
116-B-14	Shallow	--	--	--	--
116-B-15	Shallow Focused	--	--	1.0×10^{-5}	Arsenic (1.0×10^{-5})
116-B-2	Shallow	--	--	--	--
116-B-3	Shallow	--	--	--	--
116-B-4	Shallow	--	--	--	--
116-B-6A	Shallow	--	--	--	--
116-B-6B	Shallow	--	--	--	--
116-B-7	Shallow	--	--	--	--
116-B-9	Shallow	--	--	--	--
116-C-1	Shallow	--	--	--	--
116-C-2A	Shallow	--	--	--	--
116-C-5	Shallow	--	--	--	--

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
116-C-6	Shallow Focused	--	--	--	--
118-B-10	Shallow	--	--	--	--
	Shallow Focused	--	--	--	--
118-B-3	Shallow	--	--	--	--
	Shallow Focused	--	--	8.0×10^{-10}	None
118-B-4	Shallow	--	--	--	--
118-B-5	Shallow	--	--	--	--
	Shallow Focused	--	--	--	--
118-B-9	Shallow Focused	--	--	--	--
118-C-2	Shallow	--	--	--	--
118-C-4	Shallow	--	--	4.4×10^{-7}	None
128-B-2	Shallow	2.0×10^{-7}	None	1.5×10^{-7}	None
128-C-1	Shallow	1.0×10^{-6}	None	6.3×10^{-7}	None
	Shallow Focused	--	--	5.0×10^{-10}	None
132-B-6	Shallow	--	--	--	--
132-C-2	Shallow	--	--	--	--
1607-B10	Shallow	1.0×10^{-7}	None	1.1×10^{-7}	None
1607-B11	Shallow	--	--	9.5×10^{-9}	None
1607-B7	Shallow	--	--	4.2×10^{-9}	None
1607-B8	Shallow	8.0×10^{-7}	None	4.8×10^{-7}	None
1607-B9	Shallow	--	--	9.7×10^{-7}	None
600-232	Shallow	5.0×10^{-8}	None	1.1×10^{-6}	None
600-233	Shallow Focused	--	--	2.0×10^{-6}	Benzo(a)pyrene (1.3×10^{-6})

Source:

RCBRA data: DOE/RL-2007-21, *River Corridor Baseline Risk Assessment, Volume II: Human Health Risk Assessment*, Part 2, Rev. 0, Table 2-10

RI/FS data: Table G-26 (Appendix G).

Note: Chemical drivers shown have an associated risk greater than 1 × 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

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.

— = carcinogenic COPCs were not identified

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-B-11	Shallow Focused	0.02	None	0.05	None
100-B-14:5	Shallow Focused	0.02	None	0.01	None
100-B-14:6	Shallow Focused	0.08	None	0.10	None
100-B-14:7	Shallow Focused	0.03	None	0.04	None
100-B-16	Shallow Focused	0.01	None	0.02	None
100-B-5	Shallow	0.01	None	<0.01	None
100-B-8:1	Shallow	0.01	None	<0.01	None
100-B-8:2	Shallow_1	0.01	None	<0.01	None
	Shallow_3	--	--	--	--
100-C-3	Shallow	0.02	None	<0.01	None
116-B-1	Shallow	0.02	None	<0.01	None
116-B-10	Shallow	0.06	None	0.05	None
116-B-11	Shallow	0.03	None	0.02	None
116-B-12	Shallow	0.01	None	--	--
116-B-13	Shallow	0.01	None	<0.01	None
116-B-14	Shallow	0.02	None	0.02	None
116-B-15	Shallow Focused	0.01	None	0.28	None
116-B-2	Shallow	0.01	None	--	--
116-B-3	Shallow	0.01	None	<0.01	None

Table 6-2. Summary of Residential Scenario Risk Assessment Results for Noncarcinogens

		Noncancer Hazard Index and Noncancer Hazard Drivers			
Waste Site Name	**RI/FS Decision Unit**	**RCBRA Hazard Index**	**RCBRA Chemical Hazard Driver**	**RI/FS Hazard Index**	**RI/FS Chemical Hazard Driver**
116-B-4	Shallow	0.01	None	--	--
116-B-6A	Shallow	0.01	None	<0.01	None
116-B-6B	Shallow	0.01	None	<0.01	None
116-B-7	Shallow	0.02	None	<0.01	None
116-B-9	Shallow	0.02	None	0.02	None
116-C-1	Shallow	0.01	None	--	--
116-C-2A	Shallow	0.01	None	<0.01	None
116-C-5	Shallow	0.01	None	<0.01	None
116-C-6	Shallow Focused	0.01	None	--	--
118-B-10	Shallow	0.01	None	--	--
	Shallow Focused	--	--	0.01	None
118-B-3	Shallow	0.01	None	<0.01	None
	Shallow Focused	--	--	<0.01	None
118-B-4	Shallow	--	--	<0.01	--None
118-B-5	Shallow	--	--	--	--
	Shallow Focused	--	--	--	--
118-B-9	Shallow Focused	0.04	None	0.05	None
118-C-2	Shallow	--	--	--	--
118-C-4	Shallow	0.01	None	<0.01	None
128-B-2	Shallow	0.08	None	0.05	None
128-C-1	Shallow	0.39	None	0.17	None
	Shallow Focused	--	--	0.05	None
132-B-6	Shallow	0.02	None	<0.01	None
132-C-2	Shallow	0.01	None	<0.01	None
1607-B10	Shallow	0.06	None	0.05	None
1607-B11	Shallow	<0.01	None	<0.01	None
1607-B7	Shallow	0.01	None	<0.01	None

Table 6-2. Summary of Residential Scenario Risk Assessment Results for Noncarcinogens

		Noncancer Hazard Index and Noncancer Hazard Drivers			
Waste Site Name	**RI/FS Decision Unit**	**RCBRA Hazard Index**	**RCBRA Chemical Hazard Driver**	**RI/FS Hazard Index**	**RI/FS Chemical Hazard Driver**
1607-B8	Shallow	0.25	None	0.24	None
1607-B9	Shallow	0.02	None	<0.01	None
600-232	Shallow	0.02	None	0.02	None
600-233	Shallow Focused	0.01	None	<0.01	None

Source:

RCBRA data: DOE/RL-2007-21, *River Corridor Baseline Risk Assessment, Volume II: Human Health Risk Assessment*, Part 2, Rev. 0, Table 2-10

RI/FS data: Table G-26 (Appendix G).

Note: Chemical drivers shown have an associated hazard quotient (HQ) greater than 1.

The HQ value for the individual drivers is shown in parentheses after the name of the risk driver chemical.

Hazard indices are based on reasonable maximum EPCs.

— = 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-B-11	Shallow Focused	1.0×10^{-5}	None	1.1×10^{-5}	None
100-B-14:5	Shallow Focused	--	--	--	--
100-B-14:6	Shallow Focused	--	--	--	--
100-B-14:7	Shallow Focused	--	--	--	--
100-B-16	Shallow Focused	--	--	--	--
100-B-5	Shallow	7.0×10^{-6}	None	2.5×10^{-7}	None
100-B-8:1	Shallow	6.0×10^{-5}	None	2.2×10^{-5}	None
100-B-8:2	Shallow_1	8.0×10^{-5}	None	9.3×10^{-5}	None
	Shallow_3	--	--	9.4×10^{-5}	None
100-C-3	Shallow	1.0×10^{-5}	None	--	--
116-B-1	Shallow	7.0×10^{-5}	None	2.8×10^{-5}	None
116-B-10	Shallow	1.0×10^{-5}	None	4.1×10^{-6}	None
116-B-11	Shallow	3.0×10^{-4}	Eu-152 (2×10^{-4})	7.9×10^{-5}	None
116-B-12	Shallow	5.0×10^{-6}	None	--	--
116-B-13	Shallow	4.0×10^{-5}	None	1.8×10^{-5}	None
116-B-14	Shallow	4.0×10^{-4}	Eu-152 (2×10^{-4})	1.9×10^{-4}	Eu-152 (1.2×10^{-4})
116-B-15	Shallow Focused	7.0×10^{-6}	None	3.4×10^{-6}	None
116-B-2	Shallow	6.0×10^{-5}	None	4.2×10^{-5}	None
116-B-3	Shallow	8.0×10^{-6}	None	--	--
116-B-4	Shallow	7.0×10^{-6}	None	3.3×10^{-7}	None
116-B-6A	Shallow	5.0×10^{-4}	Cs-137 (3×10^{-4}) Sr-90 (2×10^{-4})	2.1×10^{-4}	Sr-90 (1.1×10^{-4})
116-B-6B	Shallow	5.0×10^{-6}	None	--	--
116-B-7	Shallow	3.0×10^{-5}	None	6.5×10^{-5}	None
116-B-9	Shallow	7.0×10^{-6}	None	--	--
116-C-1	Shallow	2.0×10^{-4}	None	5.6×10^{-5}	None
116-C-2A	Shallow	1.0×10^{-4}	None	9.7×10^{-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-C-5	Shallow	2.0×10^{-4}	None	1.0×10^{-4}	None
116-C-6	Shallow Focused	1.0×10^{-4}	None	7.2×10^{-5}	None
118-B-10	Shallow	4.0×10^{-5}	None	4.1×10^{-5}	None
	Shallow Focused	--	--	--	--
118-B-3	Shallow	3.0×10^{-5}	None	1.3×10^{-5}	None
	Shallow Focused	--	--	3.7×10^{-5}	None
118-B-4	Shallow	3.0×10^{-5}	None	9.5×10^{-6}	None
118-B-5	Shallow	2.0×10^{-6}	None	--	--
	Shallow Focused	--	--	--	--
118-B-9	Shallow Focused	--	--	--	--
118-C-2	Shallow	2.0×10^{-5}	None	1.3×10^{-5}	None
118-C-4	Shallow	3.0×10^{-6}	None	--	--
128-B-2	Shallow	2.0×10^{-5}	None	--	--
128-C-1	Shallow	--	--	--	--
	Shallow Focused	--	--	--	--
132-B-6	Shallow	2.0×10^{-4}	None	6.5×10^{-5}	None
132-C-2	Shallow	3.0×10^{-5}	None	6.5×10^{-5}	None
1607-B10	Shallow	2.0×10^{-6}	None	--	--
1607-B11	Shallow	--	--	--	--
1607-B7	Shallow	--	--	--	--
1607-B8	Shallow	--	--	--	--
1607-B9	Shallow	--	--	--	--
600-232	Shallow	--	--	--	--
600-233	Shallow Focused	--	--	--	--

Source:

RCBRA data: DOE/RL-2007-21, *River Corridor Baseline Risk Assessment, Volume II: Human Health Risk Assessment*, Part 2, Rev. 0, Table 2-10

RI/FS data: Table G-26 (Appendix G).

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**

Note: 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.

Soil samples collected from the waste site were not analyzed for radiological analytes.

— = radionuclide COPCs were not identified

6.1.2 RI/FS Soil Risk Assessment (Unrestricted Land Use)

As shown in Table 6-1 through Table 6-3, the risk assessment results are similar between the RCBRA Report (DOE/RL-2007-21) and the RI/FS for the residential scenario. Differences in results are attributed generally 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 Report (DOE/RL-2007-21) because there are several key differences between the scope and purpose of the RCBRA and the scope and purpose of the RI/FS. Differences between the RCBRA Report (DOE/RL-2007-21) 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 RAGs 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.

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 RAGs were calculated based on a dose threshold of 15 mrem/yr.	Radionuclide cancer risk is evaluated using the interim action ROD rural residential exposure scenario reported in DOE/RL-96-17. Radionuclide RAGs were calculated based on a dose threshold of 15 mrem/yr. In the RCBRA (DOE/RL-2007-21), these RAGs were converted to RBSLs based on a risk threshold of 1×10^{-4}. (RCBRA Report [DOE/RL-2007-21], pg. 2-41). 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 below.	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 RAGs reported in the closeout documentation, RCBRA Report (DOE/RL-2007-21), 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 RAGs reported in closeout document and in the RCBRA Report (DOE/RL-2007-21). Risk-based PRG values reported in the RI/FS for some alpha emitters are greater than the RAGs reported in closeout document and in the RCBRA Report (DOE/RL-2007-21).
Updates to EPA guidance for residential PRG	External gamma shielding factor is 0.8 (*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]). Outdoor time fraction is 0.2 (5 hours per day over 350 days per year) (*Hanford Guidance for Radiological Cleanup* [WDOH/320-015]). Annual dose rate is 15 mrem/yr ("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 per day over 350 days per year) (WDOH/320-015). Target cancer risk value is 1×10^{-4} (*Radiation Risk Assessment at CERCLA Sites: Q & A* [EPA/540/R/99/006]).	External gamma shielding factor is 0.4 (*Soil Screening Guidance for Radionuclides: User's Guide* [EPA/540-R-00-007]). Outdoor time fraction is 0.12 (3 hours per day over 350 days per year) (*Exposure Factors Handbook Volume II – Food Ingestion Factors* [EPA/600/P-95/002Fb]). Target cancer risk value is 1×10^{-4} (*Radiation Risk Assessment At CERCLA Sites: Q & A* [EPA/540/R/99/006]).	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. Use of the updated assumption results in slightly less exposure and a less conservative PRG value (higher). Outdoor time fraction was revised from 0.2 to 0.12. The current assumption assumes the resident spends 3 hours per day outside rather than 5 hours per 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 (WAC 173-340) Method B Soil Cleanup Levels for unrestricted land use	Separate MTCA (WAC 173-340) Method B levels were calculated for incidental soil ingestion and inhalation.	MTCA (WAC 173-340) Method B levels are based solely on incidental soil ingestion.	Separate MTCA (WAC 173-340) 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 Report (DOE/RL-2007-21) evaluation (beryllium, cadmium, cobalt, Cr(VI), and nickel). RAGs 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 hazard indices for both incidental soil ingestion and inhalation exposure routes.

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
MTCA (WAC 173-340) 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 RAGs for beryllium, cadmium, Cr(VI) based on the inhalation exposure pathway, based on WAC 173-340-750 (3), "Cleanup Standards to Protect Air Quality," 1996. A particulate emission factor (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 (WAC 173-340) Method B inhalation cleanup levels were not evaluated in the RCBRA Report (DOE/RL-2007-21).	MTCA (WAC 173-340) 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 EPAs 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) that those cleanup levels that are based on EPA Methodology.
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 is described in the *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 Report (DOE/RL-2007-21) 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 May 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 risk assessment is intended to supplement the analysis in Chapter 2 of the RCBRA Report (DOE/RL-2007-21). The RI/FS risk assessment results will be used to identify waste sites that warrant evaluation of remedial alternatives in the FS. The RI/FS 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 remedial action objectives.	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 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 risk assessment results will be used to identify waste sites that warrant evaluation of remedial alternatives in the FS. The RI/FS 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.

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
COPC Identification	Closeout documentation did not incorporate a COPC identification step. All detected analytes with RAGs reported in DOE/RL-96-17 were evaluated in the closeout documentation. It should be noted that the RAGs listed in DOE/RL-96-17 do not include analytes that meet exclusion criteria.	COPC refinement process includes a number of complementary steps and criteria, including a pre-selected 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 Report (DOE/RL-2007-21) 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 Report (DOE/RL-2007-21) 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 Report (DOE/RL-2007-21) and the RI/FS, similar risk drivers were identified in the risk characterization step of the analysis as shown in Tables 6-1 through 6-3.
EPCs	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 Managers* (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 Report (DOE/RL-2007-21) follows EPA guidance as provided in the *ProUCL Version 4.00.05 User Guide (Draft)* (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 data sets without nondetects and on data sets 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-BC Source OU.	EPA/600/R-07/038 draws from guidance documented in OSWER 9285.6-10. Methodologies for calculating 95 UCLs are similar between the RCBRA Report (DOE/RL-2007-21) and the RI/FS. The methodology used in the closeout documentation address only two data distributions for the 95 UCL calculation and implemented the substitution of one-half the detection limit value for nondetected results.
Waste Site Specific Information				
116-B-7, 132-B-6, 132-C-2	Evaluated as a single waste site.	Evaluated as three separate waste sites.	Evaluated as a single waste site.	

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RAOs are narrative statements that define the extent to which waste sites require cleanup to protect HHE. Further, PRGs (also used as risk based screening levels [RBSLs]) are the numeric values that represent the RAOs presented in Chapter 8. The 100-BC Area PRGs are developed in the RCBRA Report (DOE/RL-2007-21) and presented in this chapter.

For the 100-BC Source OU, the results of the soil risk assessment presented in this chapter will be used to 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. 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 assessments and the RCBRA focus on the protection of HHE 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 HHE:

- Are residual conditions for cleanup actions completed under the interim action RODs protective of HHE 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 out using RAGs related to direct contact soil exposure by human receptors. These RAGs are reported in the 100 Area RDR/RAWP (DOE/RL-96-17). The RAGs for radionuclides have not been revised since originally published in 1996. Interim RAGs 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 (e.g., 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 (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. 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 RAGs based on the inhalation exposure route.

In addition to performing the risk-based screening evaluation, another purpose for updating the PRGs is to determine if the RAGs 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 RAGs reported in the 100 Area RDR/RAWP (DOE/RL-96-17) 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 Report (DOE/RL-2007-21) evaluated risks for exposure scenarios that represent a range of upper bound and reasonably anticipated receptors and activities. When soil cleanup levels were established initially 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 interim cleanup actions did not limit future use of the site. The resident Monument worker and the casual user scenario represent reasonably anticipated future land use.

The resident Monument worker and the casual user scenario represent reasonably anticipated land use for the 100 Areas. Although land use is conservation (mining) in the 100 Areas, DOE has elected to clean up these areas to a more protective land use standard (unrestricted). Unrestricted land use criteria are used for the preparation of RAOs.

PRGs are presented in this section for both scenarios (resident Monument worker and casual user) that represent reasonably anticipated future land use, as well as residential PRGs, for use in the risk-based screening evaluation. CVP and RSVP data are compared to these PRGs to determine if additional remediation is required. 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 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 achieved. When the total risk for a waste site is greater than 1×10^{-4} for radionuclides based on the residential scenario or 1×10^{-5} for chemicals based on the *MTCA Method B Soil Cleanup Levels* ("Unrestricted Land Use Soil Cleanup Standards" [WAC 173-340-740]), then the waste site is compared to the resident Monument worker and casual user PRGs. The results of these comparisons are used to determine the need for remedial action. The results of these comparisons (presented in Section 6.2.5.5) show that the total risk calculated for the resident and the resident Monument worker scenarios are essentially identical.

The resident Monument worker scenario was evaluated in the RCBRA Report (DOE/RL-2007-21) as an occupational scenario and was applied on a local and broad area scale. In the RCBRA Report (DOE/RL-2007-21), the resident Monument worker spent a fraction of the day on the waste site as his residence (local area) and spent a fraction of the same day in a region as large as an individual interim action ROD decision area 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 reasonable maximum exposure (RME) broad area upland surface soil concentration reported in the RCBRA Report (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* <u>*Supplemental Guidance*</u> *"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, particulate emission factors (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 numeric value 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.

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 he spends at each area. The RCBRA Report (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 Report (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 Report (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 Report (DOE/RL-2007-21) assumed an exposure time of 13 hours/day spent at the residence (local area), 8 hours spent at 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 per day.
Indoor and outdoor exposure time	The RCBRA Report (DOE/RL-2007-21) assumed that the resident spent 13 hours/day indoors, 8 hours/day outdoors, and 3 hours per 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 Report (DOE/RL-2007-21) used an external gamma shielding factor is 0.7.	The RI/FS used an external gamma shielding factor is 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.

The casual user scenario was evaluated in the RCBRA Volume II (DOE/RL-2007-21, Volume II) as a recreational scenario and was applied on a broad area scale. In the RCBRA Report (DOE/RL-2007-21), 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 numeric value to be developed to confirm that cleanup actions at the waste site are protective.

Some exposure assumptions for the casual 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, 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 numeric value to be developed to confirm that cleanup actions at the waste site are protective of HHE. Table 6-6 summarizes the modifications made to the casual user exposure scenario for use as a PRG.

Table 6-6. Summary of Differences in Exposure Assumptions for the Casual User between the RCBRA and RI/FS Risk Evaluation

Parameter	RCBRA Casual User	RI/FS Casual 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 Report (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 Report (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 Report (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 Report (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.

6.1.4 Other Residential Land Use Scenarios in RCBRA

The RCBRA Report (DOE/RL-2007-21) 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 in order to protectively assign all soil-related exposures to that site.

PRGs were not calculated in the RCBRA Report (DOE/RL-2007-21) for these additional residential scenarios. Direct contact and food chain exposure associated with radiological contaminants for unrestricted land use are represented by the 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" [Brockman, 2007]), 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. 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 FS.

The results of the local area risk assessment for the Residential scenarios indicate that present-day RME cancer risk is frequently greater than 1×10^{-4} (10 of 45 remediated sites for the Subsistence Farmer scenario) and that RME chemical HI frequently exceeds the threshold of 1.0 (11 of the 45 remediated sites for the Subsistence Farmer scenario). A summary of cancer risks and noncancer hazards associated with the Subsistence Farmer 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, Cs-137, and Co-60
- Exposure to arsenic from ingestion of garden produce
- Exposure to the short-lived radionuclide strontium-90 from ingestion of produce and livestock products

As evaluated and explained in the RCBRA Report (DOE/RL-2007-21), by the year 2075, 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) site-raised food ingestion rates, strontium-90 still plays a significant role in food-related exposures at year 2075 for these scenarios. By 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 Subsistence Farmer cancer risk and chemical HI results were frequently above threshold criteria. The two major differences between the risk assessment methods used in the RCBRA Report (DOE/RL-2007-21) and the basis of the interim action RAGs. These differences largely explain why some waste sites remediated to meet the interim action RAGs still appear to present high levels of residual risk under the Subsistence Farmer scenario:

1. Residential Interim Action RAGs for chemicals are the MTCA (WAC 173-340) Method B Soil Cleanup Levels ("Unrestricted Land Use Soil Cleanup Standards" [WAC 173-340-740]), which is a RME scenario based on incidental soil ingestion and does not address the food exposure pathways historically evaluated for radionuclides.

2. The interim action RAG 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]).

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 Report (DOE/RL-2007-21). As discussed in Section 5.9.4.2 of the RCBRA Report (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 Report (DOE/RL-2007-21), 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 Report (DOE/RL-2007-21) to assess exposure through food pathways, may result in a scenario that provides exposures to nonradionuclide contaminants higher than an RME. Therefore, these food chain pathways have not been incorporated into the development of PRGs for nonradiological constituents.

Table 6-7. Summary of Total Risks and Noncancer Hazards for the Subsistence Farmer Scenario

Waste Site Name	Present Day Total Risk	COPC	Pathway	Present Day Hazard Index	COPC	Pathway
100-B-11	3×10^{-5}	None	--	1.6	None	--
100-B-14:5	2×10^{-8}	None	--	1.6	None	--
100-B-14:6	2×10^{-8}	None	--	19	Mercury	Beef Ingestion
					Mercury	Produce Ingestion
100-B-14:7	2×10^{-8}	None	--	3.1	Mercury	Beef Ingestion
					Cadmium	Produce Ingestion
					Zinc	Beef Ingestion
					Zinc	Milk Ingestion
100-B-16	2×10^{-8}	None	--	0.75	None	--
100-B-5	2×10^{-5}	None	--	0.4	None	--
100-B-8:1	1×10^{-4}	None	--	0.76	None	--
100-B-8:2	2×10^{-4}	Europium-154	External Irradiation	0.39	None	--
		Cobalt-60	External Irradiation			
		Europium-152	External Irradiation			
100-C-3	2×10^{-5}	None	--	0.5	None	--
116-B-1	2×10^{-4}	Europium-152	External Irradiation	0.52	None	--
		Cobalt-60	External Irradiation			
		Europium-154	External Irradiation			
116-B-10	4×10^{-5}	None	--	16	Mercury	Beef Ingestion
					Mercury	Produce Ingestion
116-B-11	7×10^{-4}	Europium-152	External Irradiation	1.8	None	--
		Cobalt-60	External Irradiation			
116-B-12	1×10^{-5}	None	--	0.16	None	--

Table 6-7. Summary of Total Risks and Noncancer Hazards for the Subsistence Farmer Scenario

Waste Site Name	Present Day Total Risk	COPC	Pathway	Present Day Hazard Index	COPC	Pathway
116-B-13	1×10^{-4}	None	--	0.52	None	--
116-B-14	8×10^{-4}	Europium-152	External Irradiation	0.81	None	--
		Strontium-90	Milk Ingestion			
116-B-15	2×10^{-5}	None	--	0.2	None	--
116-B-2	1×10^{-4}	None	--	0.14	None	--
116-B-3	2×10^{-5}	None	--	0.42	None	--
116-B-4	2×10^{-5}	None	--	0.19	None	--
116-B-6A	1×10^{-3}	Cesium-137	External Irradiation	0.41	None	--
		Strontium-90	Milk Ingestion			
		Strontium-90	Produce Ingestion			
		Strontium-90	Beef Ingestion			
116-B-6B	1×10^{-5}	None	--	0.69	None	--
116-B-7	6×10^{-5}	None	--	0.32	None	--
116-B-9	2×10^{-5}	None	--	5.0	Mercury	Beef Ingestion
					Mercury	Produce Ingestion
116-C-1	4×10^{-4}	Europium-152	External Irradiation	0.61	None	--
		Cesium-137	External Irradiation			
116-C-2A	4×10^{-4}	Strontium-90	Milk Ingestion	1.6	None	--
		Strontium-90	Produce Ingestion			
		Cobalt-60	External Irradiation			
		Europium-152	External Irradiation			
		Strontium-90	Beef Ingestion			

Table 6-7. Summary of Total Risks and Noncancer Hazards for the Subsistence Farmer Scenario

Waste Site Name	Present Day Total Risk	COPC	Pathway	Present Day Hazard Index	COPC	Pathway
116-C-5	4×10^{-4}	Europium-152	External Irradiation	0.79	None	--
		Cesium-137	External Irradiation			
		Cobalt-60	External Irradiation			
116-C-6	2×10^{-4}	Cesium-137	External Irradiation	0.15	None	--
		Strontium-90	Milk Ingestion			
		Europium-152	External Irradiation			
		Strontium-90	Produce Ingestion			
118-B-10	1×10^{-4}	None	--	2.9	Mercury	Beef Ingestion
					Mercury	Produce Ingestion
118-B-3	8×10^{-5}	None	--	0.46	None	--
118-B-4	6×10^{-5}	None	--	--*	None	--
118-B-5	3×10^{-6}	None	--	0.0007	None	--
118-B-9	2×10^{-8}	None	--	6.9	Mercury	Beef Ingestion
					Zinc	Beef Ingestion
					Zinc	Milk Ingestion
					Zinc	Produce Ingestion
					Mercury	Produce Ingestion
118-C-2	4×10^{-5}	None	--	--*	None	--
118-C-4	7×10^{-6}	None	--	0.18	None	--
128-B-2	3×10^{-5}	None	--	1.8	None	--
128-C-1	1×10^{-5}	None	--	5.0	Cadmium	Produce Ingestion
					Mercury	Beef Ingestion

Table 6-7. Summary of Total Risks and Noncancer Hazards for the Subsistence Farmer Scenario

Waste Site Name	Present Day Total Risk	COPC	Pathway	Present Day Hazard Index	COPC	Pathway
132-B-6	4×10^{-4}	Europium-152	External Irradiation	3.2	Mercury	Beef Ingestion
		Cesium-137	External Irradiation		Mercury	Produce Ingestion
132-C-2	9×10^{-5}	None	--	0.2	None	--
1607-B10	5×10^{-6}	None	--	5.0	Mercury	Beef Ingestion
					Mercury	Produce Ingestion
1607-B11	--*	None	--	0.38	None	--
1607-B7	2×10^{-8}	None	--	0.24	None	--
1607-B8	8×10^{-6}	None	--	2.1	Mercury	Beef Ingestion
					Mercury	Produce Ingestion
					Aroclor-1254	Soil Ingestion
1607-B9	2×10^{-8}	None	--	2.0	Mercury	Beef Ingestion
					Mercury	Produce Ingestion
					Cadmium	Produce Ingestion
600-232	5×10^{-7}	None	--	0.4	None	--
600-233	2×10^{-8}	None	--	1	None	--

Source:

RCBRA data: *River Corridor Baseline Risk Assessment, Volume II: Human Health Risk Assessment* (DOE/RL-2007-21), Part 2, Tables 5-57 and 5-59.

Notes: Risk drivers shown have an associated risk greater than 1×10^{-4}.

* No contaminants of potential concern were identified for the health assessment.

6.2 Soil Risk Assessment

Section 6.1.1 summarizes the evaluation of residual risks performed in the RCBRA Report (DOE/RL-2007-21) for waste sites cleaned up under the interim action ROD. Section 6.1.2 describes how elements of the RCBRA 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 100-BC Source OU soil risk assessment followed the approach described below:

- 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 May 2011.[4]
- Compute EPCs for each detected analyte measured at a waste site using the EPA's ProUCL version 4.00.05 software.
- Compare EPCs to direct contact RBSLs selected to represent baseline conditions and reasonably anticipated future 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 if the "no action" or "interim closed" waste site should be carried forward into the FS to select remedial alternatives.

This soil risk assessment follows Risk Assessment Guide (EPA/540/1-89/002). The following subsections describe the four-step process. Because this soil risk assessment is intended to complement the analysis performed in the RCBRA Report (DOE/RL-2007-21), 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-BC-1and 100-BC-2 Source OUs. Waste sites where remediation and verification sampling and analysis were assessed by the end of May 2011 are included in the soil risk assessment.

All samples were collected in accordance with the requirements stated in the 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:

- EPA/ROD/R10-99/039, 100 Area Remaining Sites ROD
- EPA/AMD/R10-97/044, *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*

[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 May 2011.

- EPA/541/R-00/121, *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*

Remediation of waste sites in the 100-BC Source OU began in 1996. Analytical results for each waste site are included in the associated closeout documentation, which are listed in Appendix B, Table B-1, of the 100-BC Work Plan (DOE/RL-2008-46-ADD3). The 100-BC Work Plan (DOE/RL-2008-46-ADD3) and the 100 Area RDR/RAWP (DOE/RL-96-17) were reviewed and approved by the Tri-Parties.

Eighty-one waste sites in the 100-BC Source OU have verification sampling and analysis data and are included in this soil risk assessment. Forty-five of the 81 waste sites from the 100-BC Source OU were evaluated in the RCBRA Report (DOE/RL-2007-21). An additional eight sites in the 100-BC Source OU 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-BC Source OU is provided in Table G-1. Waste site decision units are defined in Section 6.2.2.2. The waste sites listed in Table G-1 are a subset of the waste sites that were listed in Appendix B, Table B-1, of the 100-BC Work Plan (DOE/RL-2008-46-ADD3). A summary of the remediated waste sites and consolidated waste sites for the 100-BC-1 and 100-BC-2 OUs is provided in Table 6-8.

The following sources of analytical data were used in the soil risk assessment:

- All verification sampling and analysis data reside in the HEIS database.
- All closeout verification data used in this risk evaluation are included in Appendix D of this report.

Table 6-8. Summary of Waste Sites with Verification Data in the 100-BC-1 and 100-BC-2 OUs

100-BC-1 Waste Sites			
100-B-11	100-B-25	116-B-14	118-B-7
100-B-14:1	100-B-26	116-B-15	118-B-9
100-B-14:2	100-B-27	116-B-16[c]	120-B-1
100-B-14:3	100-B-28	116-B-2	126-B-3
100-B-14:5	100-B-31	116-B-3	128-B-2
100-B-14:6	100-B-32	116-B-4	128-B-3
100-B-14:7	100-B-33	116-B-5	132-B-6[d]
100-B-16	100-B-5	116-B-6A[c]	132-C-2[d]
100-B-18	100-B-8:1[a]	116-B-6B	1607-B1
100-B-19	100-B-8:2[b]	116-B-7[d]	1607-B2:1
100-B-20	116-B-1	116-B-9	1607-B2:2
100-B-21:2	116-B-10	116-C-1	1607-B7
100-B-21:3	116-B-11	116-C-5	--
100-B-21:4	116-B-12	118-B-10	--
100-B-22:2	116-B-13	118-B-5	--

Table 6-8. Summary of Waste Sites with Verification Data in the 100-BC-1 and 100-BC-2 OUs

100-BC-1 Subtotal		**57**	
100-BC-2 Waste Sites			
100-B-1	100-C-9:2	118-B-2[f]	118-C-4
100-B-23	100-C-9:3	118-B-3[f]	128-C-1
100-C-3	116-C-2A[e]	118-B-4	1607-B10
100-C-6:1[a]	116-C-2B[e]	118-B-6	1607-B11
100-C-6:2[b]	116-C-2C[e]	118-C-1	1607-B8
100-C-6:3[b]	116-C-3	118-C-2	1607-B9
100-C-6:4[b]	116-C-6	118-C-3:2	600-232
100-C-9:1	118-B-1	118-C-3:3	600-233
100-BC-2 Subtotal		**32**	
100-BC Area Total		**89**	

Notes:

The following are those sites for which remediation and closeout documentation were associated with another remediated waste site:

a. Associated with 100-B-8:1 (100-BC-1) remediated waste site.

b. Associated with 100-B-8:2 (100-BC-1) remediated waste site.

c. Associated sites 116-B-16 and 116-B-6A.

d. Samples results are associated for the waste sites 116-B-7,132-B-6, and 132-C-2.

e. Associated sites 116-C-2A, 116-C-2B, 116-C-2C.

f. Associated site 118-B-2 and 118-B-3.

6.2.1.2 Data Quality Evaluation 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 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 site verification. The cleanup verification sample analytical data and detailed DQA are summarized in the appendices associated with the cleanup verification packages. The results of each DQA are incorporated by reference and no further DQA was performed as part of this risk assessment.

All 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 if 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 if 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 HHE, 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 81 waste sites included in the risk assessment are identified as COPCs. As described below (Section 6.2.2.2), the floor and sidewalls of an excavated waste site are divided into one or more decision units (e.g., shallow zone, deep zone, overburden, or staging pile area footprint). Verification sampling and analysis data are collected according to sample design requirements for the type of decision unit. For the purpose of this risk evaluation, 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* (hereinafter called CERCLA Soil Background Comparisons Guidance; EPA 540-R-01-003). The risk characterization will discuss elevated background concentrations and their contribution to site risks, as well as naturally occurring elements that are not CERCLA hazardous substances, pollutants, and contaminants, but exceed the RBSLs.

The RCBRA Report (DOE/RL-2007-21) identifies a subset of analytes that are 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 Report (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, Table G-3:

- 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 (e.g., grain size for soils or 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 Report (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 Report (DOE/RL-2007-21) to identify COPCs, more analytes are identified as COPCs in this soil risk assessment than were identified in the RCBRA. 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 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 the 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 data set 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). While 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 data sets (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-BC 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-BC-1, and 100-BC-2 Source Operable Units* (ECF-100KR4-11-0007), 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" or "no action" through May 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 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 soil risk assessment. A list of the sample numbers associated with each waste site decision unit is provided in Table G-2 (Appendix G). Table G-2 also lists the date the sample was collected, the type of sample design used, and the Washington state plane coordinates of the sample location.

Waste Site Decision Units. Verification sampling and analysis data that are associated with the samples listed in Table G-2 (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 paragraphs describe 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 and a sample design is developed for the decision unit. Sample design requirements for each decision unit are described in 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 (i.e., 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 remedial action objectives 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 6-9:

Table 6-9. Summary and Definition of Decision Unit Types

Decision Unit Name	Depth	Sampling Design Description
Shallow	0 to 4.6 m (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	
Overburden_Focused	Not applicable	
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-BC-1 and 100-BC-2 Source Operable Units* (ECF-100BC1-11-0012), which is provided in Appendix G. The purpose of *Computation of Exposure Point Concentrations for the 100-BC-1 and 100-BC-2 Source Operable Units* (ECF-100BC1-11-0012) is to document the data processing and reduction steps, methodology, decision logic, assumptions, input files, and output files used to determine the EPCs.

Data Processing and Reduction. This section describes the data processing and reduction steps that are taken prior to the calculation of UCLs. Figure 6-1 shows each of the data processing and data reduction steps, and the number of records associated with each step for the 100-BC Source OU.

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" data 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.

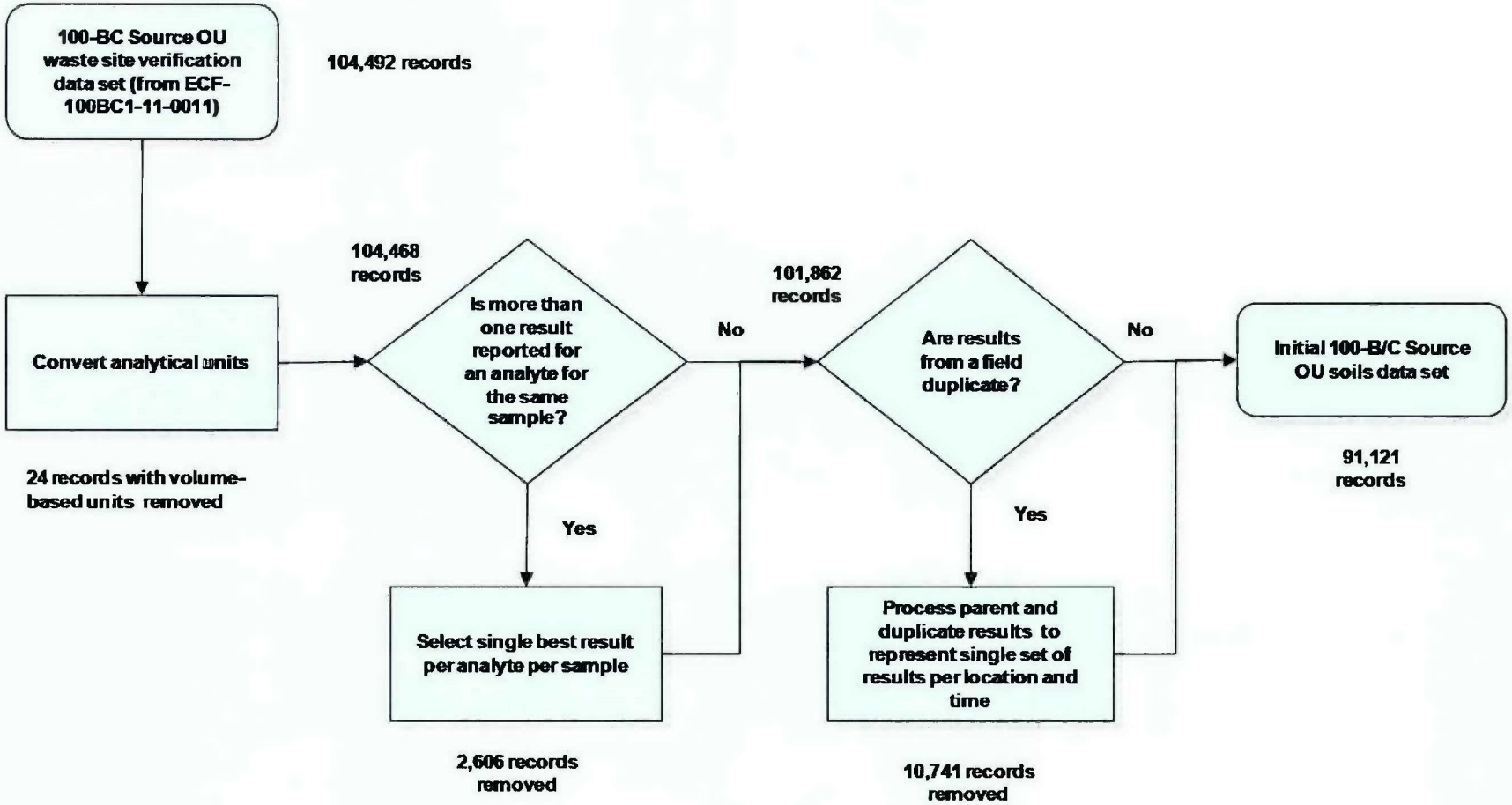

Figure 6-1. Data Processing and Reduction Steps for the 100-BC Source OU

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 (i.e., 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*, hereinafter called SW-846 [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) because of the more sensitive detection limit.

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 (i.e., 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 (i.e., 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 (i.e., 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.

Identify Analytes for 95 percent UCL Calculation. After extracting and processing the data set, 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 were not detected in any of the samples analyzed with the 100-BC 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 in Figure 6-2 for the 100-BC Source OU.

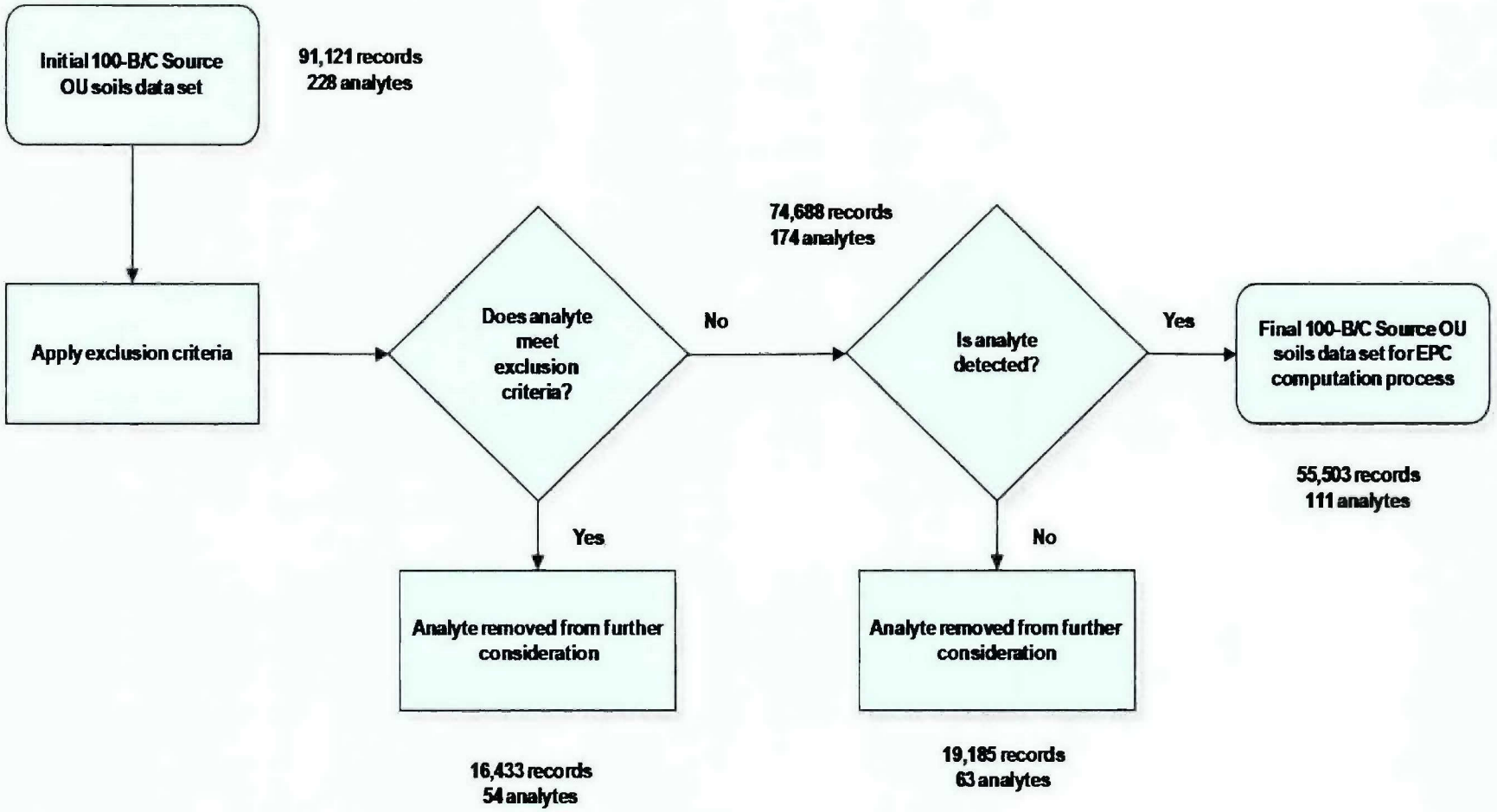

Figure 6-2. Analyte Identification Steps for the 100-BC Source OU Data Set

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 54 analytes for the 100-BC Source OU meet the exclusion criteria and are listed in Table G-3 (Appendix G). Sampling dates, minimum and maximum detected concentrations, minimum and maximum method detection limits, and the basis for their exclusion are provided in this table.

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 are eliminated from further consideration. Any analyte that is detected at least once in the 100-BC Source OU is carried forward to the next step of the process.

A total of 63 analytes were not detected in the 100-BC Source OU samples and are listed in Table G-4 (Appendix G). The table also provides sampling dates, total number of samples, and minimum and maximum MDLs.

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 likely 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 data set and subsequently formatted so they can be directly imported into ProUCL where 95 percent UCL calculations and summary statistics are performed.

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

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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.

[6] Minimum and maximum detection limits are summarized in the ProUCL output only when a valid UCL can be calculated.

- Coefficient of variation for each analyte
- The UCL value, the UCL basis, and comments and/or warning statements for each analyte

For most ta sets, 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 data sets. The most conservative (i.e., health-protective) result 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. More than one UCL was recommended and at least one of the UCLs was greater than the maximum observed concentration for 21 analytes.

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.

Selection of the EPC value using the above decision logic is presented on Figure 6-3. A summary of the EPCs for each detected analyte in a given waste site decision unit is provided in Table G-5 (Appendix G) for the 100-BC Source OU.

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

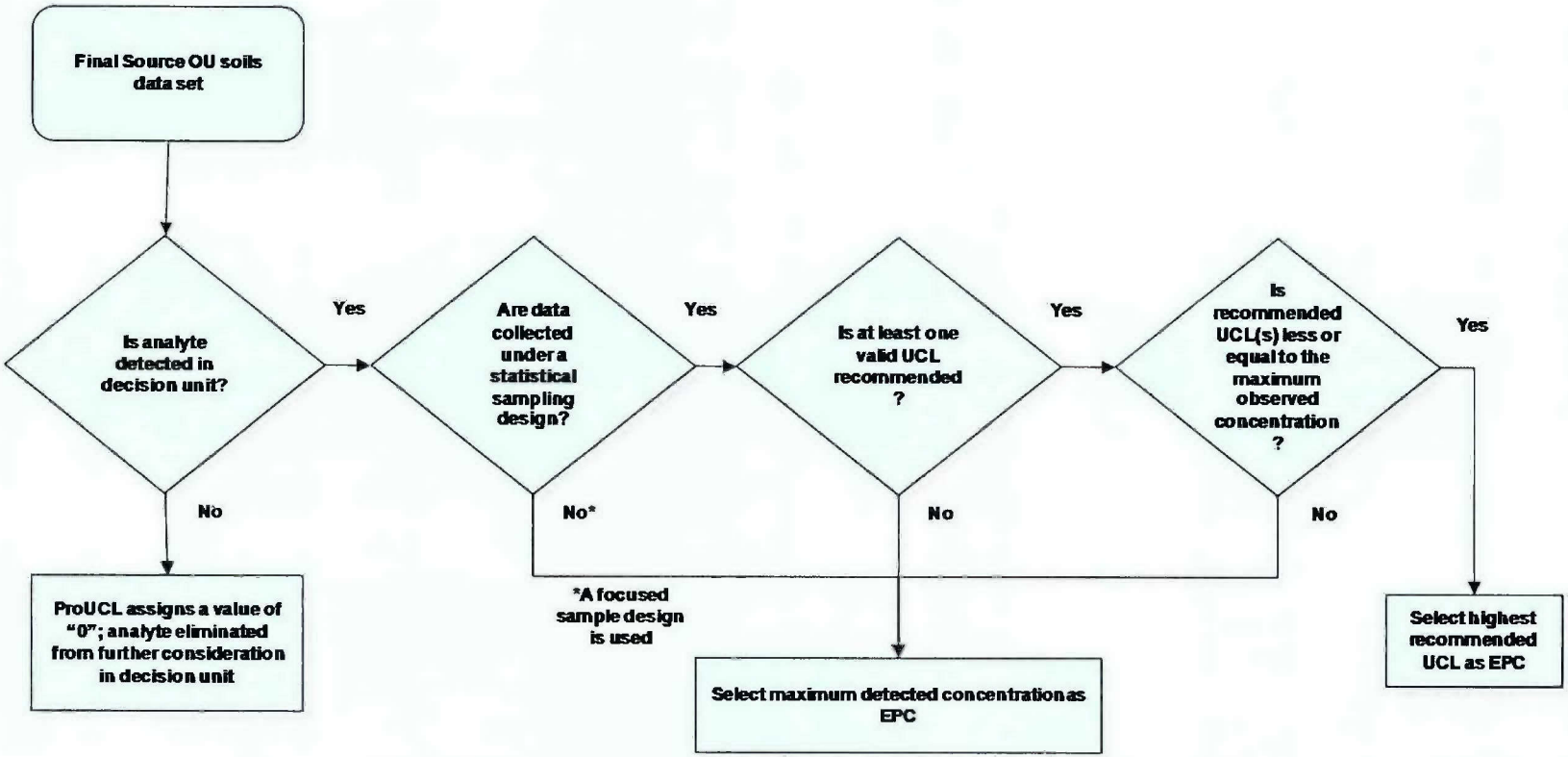

Figure 6-3. Decisions Logic for Computing 100-BC Exposure Point Concentrations

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. Guidance provided by 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 (i.e., 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 if the 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 (less than 5), 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 data set. For small datasets with very few detected values (less than 5), where a valid UCL cannot be calculated, the EPC defaults to the maximum (single) concentration for the 100-BC Source OU.

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. When the recommended UCL exceeds the maximum detected concentration, ProUCL, however, advises that an alternative UCL (i.e., 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-BC Source OU. ProUCL displays a warning message when the recommended 95 percent UCL of the mean exceeds the observed maximum concentration.

6.2.2.3 Methodology Used to Calculate Total Uranium Concentrations from Isotopic Uranium Concentrations

Uranium analytical data are reported for most of the 100-BC Area 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-BC 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 non-detects, the individual values are not summed, but the maximum non-detect value is retained and put in the ProUCL file, flagged as a nondetect.

If both converted isotopic and analytical total uranium EPCs are available for a decision unit, then the EPC based on analytical total uranium concentrations is carried through as the final EPC for that decision unit.

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 6-10. As mentioned previously, only detected concentrations are included in the summations. In the Table 6-10 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 6-10. 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	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 for uranium-234; uranium-234 is assumed to be the dominant isotope in undifferentiated uranium-233/234.

ND = nondetect

D = detect

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 site contaminants, and provides the methodology for calculating the RBSLs for direct contact, based on currently available 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-BC Source OU. This results in a set of exposure pathways that reflect a reasonable maximum exposure.

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 and, therefore, creates no risk or hazard.

6.2.3.1 Contaminant Sources

The primary sources of contamination in the 100-BC Source OU are two water-cooled nuclear reactors (105-B and 105-C Reactors) and the structures (e.g., fuel storage basins) and processes (e.g., sodium dichromate process) associated with reactor operations. A complete discussion of primary and secondary contaminant sources is provided in Section 4.2.1 and Section 5.2 of this report.

6.2.3.2 Release Mechanisms and Environmental Transport Media

The primary COPC release mechanisms and transport pathways at the 100-BC are discussed in Sections 5.3 and 5.4, 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-BC Source OU
- Volatilization of COPCs emanating from shallow vadose zone material to ambient air at the 100-BC 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-BC Source OU, the most plausible exposure pathways for calculating PRGs and characterizing the human health risks have been 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 6.2.1.2, Table 6-9). 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 (e.g., 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 below are consistent with the exposure scenario and ARARs used to develop the interim action RAGs for soil presented in the 100 Area RDR/RAWP (DOE/RL-96-17). This exposure scenario is also evaluated in the RCBRA to determine if cleanup actions completed under the interim action RODs are protective of HHE 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).

For radiological PRG development, a subsistence farming scenario is used. This assumes that each interim remediated waste site decision unit (1) has the potential to be developed into a residence with a basement, (2) vegetable and fruit crops are grown in a backyard garden, and (3) a pasture 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 (i.e., 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 the established land uses and the proclamation of "Establishment of the Hanford Reach National Monument" (65 FR 37253), it is unlikely that land within the 100-BC 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 irrigation of 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 RAGs for soil presented in the 100 Area RDR/RAWP (DOE/RL-96-17). The exposure scenario for protection of human health is based on the MTCA "Standard 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 a child and adult receptor, includes inhalation of vapors and dust in ambient air, and assumes residential exposure frequency and duration assumptions. For arsenic and lead, the MTCA "Tables" (WAC 173-340-900), Table 740-1 Method A, soil cleanup levels for unrestricted land use of 20 and 250 mg/kg were used.

Groundwater. Groundwater within the 100-BC-5 OU is currently contaminated, and withdrawal is prohibited as a result of ICs placed on it by DOE through the interim action ROD; however, institutional controls will be evaluated as part of the final remedy. Under current 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 RAGs documented in the interim action RODs and in the 100 Area RDR/RAWP (DOE/RL-96-17). Groundwater concentrations are compared to current 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 radium-226, but excluding uranium and radon), and 5 pCi/L combined for radium-226 and radium-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 a child and adult receptor, includes drinking water ingestion and inhalation of vapors, and assumes residential exposure frequency and duration assumptions.

Resident Monument Worker Scenario. Land use within the River Corridor's 100 and 600 Areas is predominantly conservation/preservation. In 2000, a Presidential Proclamation 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 for protecting 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]).

For the purposes of the RI/FS, the resident Monument worker represents reasonably anticipated future land use. PRGs are developed for this scenario for use in the risk-based screening evaluation. Resident Monument worker PRG values are developed for radiological contaminants. For chemical contaminants, 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. 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 "Human Health Risk Assessment Procedures" (WAC 173-340-708(5)) cumulative risk threshold, then protection of the resident Monument worker is achieved. When the total risk for a waste site exceeds 1×10^{-4} based on the residential scenario, then the waste site is compared to the PRGs developed for the resident Monument worker. If the total risk for the resident Monument worker is greater than 1×10^{-4} for radionuclides or 1×10^{-5} for chemicals based on the MTCA "Human Health Risk Assessment Procedures" (WAC 173-340-708(5)) cumulative risk threshold, then the results of these comparisons are used to determine the need for remedial action. The results of these comparisons are used to determine the need for remedy selection relative to reasonably anticipated land use.

In addition to residential PRGs, CVP and RSVP data are also compared to the resident Monument worker PRGs to confirm that cleanup actions are protective of the reasonably anticipated future land uses scenarios. The resident Monument worker is selected as the receptor to represent potential exposures from occupational use along the River Corridor. This exposure scenario was included in the subset of occupational scenarios presented in the RCBRA Report (DOE/RL-2007-21). 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-BC-5 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 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.

Casual Recreational User Scenario. As discussed previously, reasonably anticipated future land use within the River Corridor's 100 and 600 Areas is predominantly conservation/preservation. The casual recreational user scenario is also used in the evaluation of balancing criteria in the FS. The casual recreational user is selected as the 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 Report (DOE/RL-2007-21). 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.

For the purposes of the RI/FS, the casual user represents reasonably anticipated future land use. PRGs are developed for this scenario for use in the risk-based screening evaluation. Casual user PRG values are developed for radiological and nonradiological contaminants. When the total risk for a waste site exceeds 1×10^{-4} based on the residential scenario or 1×10^{-5} for chemicals based on the MTCA "Human Health Risk Assessment Procedures" (hereinafter called HHRA Procedures [WAC 173-340-708(5)]) cumulative risk threshold, then protection of the casual recreational user is achieved. When the total risk for a waste site exceeds 1×10^{-4} based on the residential scenario, then the waste site is compared to the PRGs

developed for the casual recreational user. If the total risk for the casual recreational user is greater than 1×10^{-4} for radionuclides or chemicals, then the waste site is compared to the PRGs developed for the casual user. The results of these comparisons are used to determine the need for remedial action

In addition to comparison to residential PRGs, CVP and RSVP data are also compared to the casual user PRGs to confirm that cleanup actions are protective of the reasonably anticipated future land use scenarios. 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 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* (hereinafter called Risk Assessment Guidance Volume I, Part B [EPA/540/R-92/003]) 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, MTCA [WAC 173-340]) 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 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 EPA's Superfund Radionuclide PRG download and calculation web site (EPA, 2010). 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 "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 6-11 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, 2009) 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 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 list of the site-specific RESRAD input parameters is provided in Table G-6 (Appendix G). 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) (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 (i.e., 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. Default exposure parameters for calculating the PRGs are defined in Table G-7 (Appendix G). Standard Method B soil cleanup levels for unrestricted land use are based on an acceptable cancer risk level of 1×10^{-6} for carcinogens or a hazard quotient (HQ) of 1 for noncarcinogens.

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Table 6-11. Summary of Preliminary Remediation Goals for the 100-B/C Source OU

Analyte	90th Percentile Background	Method A Soil PRG	Residential PRG	Method B Direct Contact Soil PRG (carcinogen)	Method B Direct Contact Soil PRG (noncarcinogen)	Method B Inhalation Soil PRG (carcinogen)	Method B Inhalation Soil PRG (noncarcinogen)	Casual User PRG (carcinogen)	Casual User PRG (noncarcinogen)	Resident National Monument Worker PRG (carcinogen)
Radionuclides (pCi/g)										
Americium-241	--	--	155	--	--	--	--	2,570	--	275
Carbon-14	--	--	81	--	--	--	--	327,610	--	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	--	--	--	--	5,869	--	354
Nickel-63	--	--	608	--	--	--	--	575,308	--	91,600
Niobium-94	--	--	1.4	--	--	--	--	26	--	1.7
Plutonium-238	0.0038	--	236	--	--	--	--	3,818	--	605
Plutonium-239/240	0.025	--	203	--	--	--	--	3,342	--	539
Technetium-99	--	--	1.5	--	--	--	--	114,449	--	17,300
Total beta radiostrontium	0.18	--	2.3	--	--	--	--	5,064	--	518
Tritium	--	--	623	--	--	--	--	15,376	--	1,270,000
Uranium-233/234	1.1	--	133	--	--	--	--	5,808	--	931
Uranium-235	0.11	--	16	--	--	--	--	295	--	22
Uranium-238	1.1	--	54	--	--	--	--	1,093	--	93
Metals (mg/kg)										
Aluminum	11,800	--	--	--	80,000	--	>1,000,000	--	912,453	--
Antimony	0.13	--	--	--	32	--	--	--	365	--
Arsenic	6.5	20	--	0.67	24	42,414	500,240	4.5	253	--
Barium	132	--	--	--	16,000	--	>1,000,000	--	182,481	--
Beryllium	1.5	--	--	--	160	75,991	666,986	>1,000,000	1,825	--
Boron	3.9	--	--	--	16,000	--	>1,000,000	--	182,500	--
Cadmium	0.56	--	--	--	40	101,322	333,493	>1,000,000	821	--
Chromium	19	--	--	--	120,000	--	--	--	>1,000,000	--
Cobalt	16	--	--	--	24	20,264	200,096	920,451	274	--
Copper	22	--	--	--	3,200	--	--	--	36,500	--
Hexavalent Chromium	--	--	--	--	240	2,171	>1,000,000	98,620	2,737	--
Iron	32,600	--	--	--	56,000	--	--	--	638,750	--
Lead	10	250	--	--	--	--	--	--	--	--
Lithium	13	--	--	--	160	--	--	--	1,825	--

Table 6-11. Summary of Preliminary Remediation Goals for the 100-B/C Source OU

Analyte	90th Percentile Background	Method A Soil PRG	Residential PRG	Method B Direct Contact Soil PRG (carcinogen)	Method B Direct Contact Soil PRG (noncarcinogen)	Method B Inhalation Soil PRG (carcinogen)	Method B Inhalation Soil PRG (noncarcinogen)	Casual User PRG (carcinogen)	Casual User PRG (noncarcinogen)	Resident National Monument Worker PRG (carcinogen)
Manganese	512	--	--	--	11,200	--	>1,000,000	--	127,658	--
Mercury	0.013	--	--	--	24	--	>1,000,000	--	274	--
Molybdenum	0.47	--	--	--	400	--	--	--	4,563	--
Nickel	19	--	--	--	1,600	701,458	>1,000,000	>1,000,000	18,249	--
Selenium	0.78	--	--	--	400	--	>1,000,000	--	4,562	--
Silver	0.17	--	--	--	400	--	--	--	4,563	--
Strontium	--	--	--	--	48,000	--	--	--	547,500	--
Tin	--	--	--	--	48,000	--	--	--	547,500	--
Uranium	3.2	--	--	--	240	--	>1,000,000	--	2,737	--
Vanadium	85	--	--	--	400	--	--	--	4,563	--
Zinc	68	--	--	--	24,000	--	--	--	273,750	--
Polyaromatic Hydrocarbons (PAHs) (mg/kg)										
Benzo(a)pyrene	--	--	--	0.14	--	165,799	--	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	--	--
Indeno(1,2,3-cd)pyrene	--	--	--	1.4	--	>1,000,000	--	1.7	--	--
Pyrene	--	--	--	--	2,400	--	--	--	20,070	--
Polychlorinated Biphenyls (PCBs) (mg/kg)										
Aroclor-1221	--	--	--	0.50	--	319,963		2.0	--	--
Aroclor-1242	--	--	--	0.50	--	319,963	--	2.6	--	--
Aroclor-1254	--	--	--	0.50	1.6	319,963	--	2.6	13	--
Aroclor-1260	--	--	--	0.50	--	319,963	--	2.6	--	--
Other Inorganics (mg/kg)										
Fluoride	2.8	--	--	--	4,800	--	>1,000,000	--	54,750	--
Nitrate	52	--	--	--	568,000	--	--	--	>1,000,000	--
Nitrogen in Nitrate	--	--	--	--	128,000	--	--	--	>1,000,000	--
Nitrogen in Nitrite and Nitrate	--	--	--	--	128,000	--	--	--	>1,000,000	--
Other Organics (mg/kg)										
1,2,4-Trichlorobenzene	--	--	--	34	800	--	14	257	1,296	--
2,4,5-T(2,4,5-Trichlorophenoxyacetic acid)	--	--	--	--	800	--	--	--	7,129	--
2,4,5-TP(2-(2,4,5-Trichlorophenoxy)propionic acid)Silvex	--	--	--	--	640	--	--	--	5,703	--

Table 6-11. Summary of Preliminary Remediation Goals for the 100-B/C Source OU

Analyte	90th Percentile Background	Method A Soil PRG	Residential PRG	Method B Direct Contact Soil PRG (carcinogen)	Method B Direct Contact Soil PRG (noncarcinogen)	Method B Inhalation Soil PRG (carcinogen)	Method B Inhalation Soil PRG (noncarcinogen)	Casual User PRG (carcinogen)	Casual User PRG (noncarcinogen)	Resident National Monument Worker PRG (carcinogen)
2,4-D(2,4-Dichlorophenoxyacetic acid)	--	--	--	--	800	--	--	--	8,004	--
2,4-DB(4-(2,4-Dichlorophenoxy)butanoic acid)	--	--	--	--	640	--	--	--	5,703	--
2-Butanone	--	--	--	--	48,000	--	28,673	--	464,234	--
2-Methylnaphthalene	--	--	--	--	320	--	--	--	2,676	--
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	--
4-Amino-3,5,6-trichloropicolinic acid	--	--	--	--	5,600	--	--	--	49,902	--
4-Methyl-2-Pentanone	--	--	--	--	6,400	--	13,103	--	69,370	--
Acenaphthene	--	--	--	--	4,800	--	--	--	40,139	--
Acetone	--	--	--	--	72,000	--	189,926	--	789,195	--
Aldrin	--	--	--	0.059	2.4	37,220	--	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	--
Anthracene	--	--	--	--	24,000	--	--	--	200,696	--
Benzene	--	--	--	18	320	0.57	24	22	1,513	--
Benzo(a)anthracene	--	--	--	1.4	--	>1,000,000	--	1.7	--	--
beta-1,2,3,4,5,6-Hexachlorocyclohexane (beta-BHC)	--	--	--	0.56	--	344,111	--	3.1	--	--
Bis(2-ethylhexyl) phthalate	--	--	--	71	1,600	>1,000,000	--	405	14,258	--
Butylbenzylphthalate	--	--	--	526	16,000	--	--	2,981	142,578	--
Carbazole	--	--	--	50	--	--	--	283	--	--
Carbon tetrachloride	--	--	--	14	320	0.24	127	9.2	610	--
Chlordane	--	--	--	2.9	40	>1,000,000	>1,000,000	19	410	--
Dalapon	--	--	--	--	2,400	--	--	--	21,387	--
Dibenz[a,h]anthracene	--	--	--	1.4	--	>1,000,000	--	1.7	--	--
Dibenzofuran	--	--	--	--	80	--	--	--	913	--
Dicamba	--	--	--	--	2,400	--	--	--	21,387	--
Dieldrin	--	--	--	0.063	4.0	39,648	--	0.35	36	--
Diethylphthalate	--	--	--	--	64,000	--	--	--	570,313	--
Di-n-butylphthalate	--	--	--	--	8,000	--	--	--	71,289	--
Dinoseb(2-secButyl-4,6-dinitrophenol)	--	--	--	--	80	--	--	--	713	--
Endosulfan I	--	--	--	--	480	--	--	--	4,277	--
Endosulfan II	--	--	--	--	480	--	--	--	4,277	--

Table 6-11. Summary of Preliminary Remediation Goals for the 100-B/C Source OU

Analyte	90th Percentile Background	Method A Soil PRG	Residential PRG	Method B Direct Contact Soil PRG (carcinogen)	Method B Direct Contact Soil PRG (noncarcinogen)	Method B Inhalation Soil PRG (carcinogen)	Method B Inhalation Soil PRG (noncarcinogen)	Casual User PRG (carcinogen)	Casual User PRG (noncarcinogen)	Resident National Monument Worker PRG (carcinogen)
Endrin	--	--	--	--	24	--	--	--	214	--
Fluoranthene	--	--	--	--	3,200	--	--	--	26,760	--
Fluorene	--	--	--	--	3,200	--	--	--	26,760	--
Gamma-BHC (Lindane)	--	--	--	0.91	24	588,319	--	6.0	246	--
Heptachlor	--	--	--	0.22	40	140,292	--	1.3	356	--
Heptachlor epoxide	--	--	--	0.11	1.0	70,146	--	0.62	9.3	--
Hexachloroethane	--	--	--	71	80	>1,000,000	--	405	713	--
Isophorone	--	--	--	1,053	16,000	--	50,482	5,962	142,578	--
Methoxychlor	--	--	--	--	400	--	--	--	3,564	--
Methylene chloride	--	--	--	133	4,800	11	965	337	35,713	--
Naphthalene	--	--	--	--	1,600	1.4	25	62	2,241	--
n-Nitrosodiphenylamine	--	--	--	204	--	>1,000,000	--	1,156	--	--
Pentachlorophenol	--	--	--	8.3	2,400	>1,000,000	--	35	16,103	--
Phenol	--	--	--	--	24,000	--	11,614	--	213,867	--
Styrene	--	--	--	--	16,000	--	2,950	--	115,430	--
Toluene	--	--	--	--	6,400	--	4,774	--	63,832	--
Total petroleum hydrocarbons	--	2,000	--	--	--	--	--	--	--	--
Trichloroethene	--	--	--	11	--	0.17	28	7.2	2,940	--
Xylenes (total)	--	--	--	--	16,000	--	103	--	10,346	--

Note:
-- = Not Applicable

Reference dose and carcinogenic potency factors are determined using the recommended reference hierarchy as described in "Human Health Toxicity Values in Superfund Risk Assessments" (hereinafter called Superfund HHT Risk Assessment Values [Cook, 2003]). A detailed description of methodology, inputs, assumptions and the results of the calculations are presented in *Documentation of Standard Method B Soil Cleanup Levels for Unrestricted Land Use* (ECF-HANFORD-10-0444) (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 were calculated for noncarcinogens and carcinogens using equations 750-1 and 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. Default exposure parameters for calculating the inhalation PRGs are defined in Table G-8 (Appendix G). 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 reference dose (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) (Appendix G).

Direct Contact. The following represents the Standard Method B direct contact soil PRG equations for noncarcinogens and carcinogens. The parameters used for calculating the soil cleanup levels are defined in Table G-7 (Appendix G).

Noncarcinogens. The equation below is obtained from MTCA "Unrestricted Land Use Soil Cleanup Standards" (WAC 173-340-740(3)(b)(iii)(B)(I)) Equation 740-1.

$$Soil\ PRG\left(\frac{mg}{kg}\right) = \frac{RfD \times ABW \times UCF \times HQ \times AT_{nc}}{SIR \times AB1 \times EF \times ED}$$

where:

RfD	=	reference dose (mg/kg-day)
ABW	=	average body weight (kg)
UCF	=	unit conversion factor (mg/kg)
HQ	=	hazard quotient (unitless)
AT_{nc}	=	averaging time- noncarcinogens (years)
SIR	=	soil ingestion rate (mg/day)
$AB1$	=	gastrointestinal absorption fraction (unitless)
EF	=	exposure frequency (unitless)
ED	=	exposure duration (years)

Carcinogens. The equation below is obtained from MTCA "Unrestricted Land Use Soil Cleanup Standards" (WAC 173-340-740(3)(b)(iii)(B)(II)) Equation 740-2.

$$Soil\ PRG\left(\frac{mg}{kg}\right) = \frac{RISK \times ABW \times AT_c \times UCF}{CPF \times SIR \times AB1 \times ED \times EF}$$

where:

$RISK$	=	acceptable cancer risk level (unitless)
ABW	=	average body weight (kg)
AT_c	=	averaging time-carcinogens (years)
UCF	=	unit conversion factor (mg/kg)
CPF	=	carcinogenic potency factor (kg-day/mg)
SIR	=	soil ingestion rate (mg/day)
$AB1$	=	gastrointestinal absorption fraction (unitless)
ED	=	exposure duration (years)
EF	=	exposure frequency (unitless)

Inhalation. The following represents the Standard Method B inhalation PRG equations for noncarcinogens and carcinogens. The parameters used for calculating the soil PRGs are defined in Table G-8 (Appendix G).

Noncarcinogens. The equation below is obtained from MTCA "Cleanup Standards to Protect Air Quality" (WAC 173-340-750(3)(b)(ii)(A)) Equation 750-1.

$$Air\ PRG\left(\frac{\mu g}{m^3}\right)=\frac{RfD\times ABW\times UCF\times HQ\times AT}{BR\times ABS\times ED\times EF}$$

where:

RfD	=	reference dose (mg/kg-day)
ABW	=	average body weight-noncarcinogen (kg)
UCF	=	unit conversion factor (µg/mg)
HQ	=	target hazard quotient (unitless)
AT	=	averaging time-noncarcinogen (years)
BR	=	breathing rate-noncarcinogen (m^3/day)
ABS	=	inhalation absorption fraction (unitless)
ED	=	exposure duration-noncarcinogen (years)
EF	=	exposure frequency (unitless)

Carcinogens. The equation below is obtained from MTCA "Cleanup Standards to Protect Air Quality" (WAC 173-340-750(3)(b)(ii)(B)) Equation 750-2.

$$Air\ PRG\left(\frac{\mu g}{m^3}\right)=\frac{RISK\times ABW\times AT\times UCF}{CPF\times BR\times ABS\times ED\times EF}$$

where:

$RISK$	=	target risk (unitless)
ABW	=	average body weight-carcinogen (kg)
AT	=	averaging time-carcinogen (years)
UCF	=	unit conversion factor (µg/mg)
CPF	=	cancer potency factor ([mg/kg-day]$^{-1}$)
BR	=	breathing rate-carcinogen (m^3/day)

ABS = inhalation absorption fraction (unitless)
ED = exposure duration-carcinogen (years)
EF = exposure frequency (unitless)

Air PRG to Soil PRG Conversion. The air PRGs calculated using the above equations are intended to protect air quality. However, concentrations of contaminants in air were not directly measured; therefore, emission and dispersion modeling are used to estimate a soil PRG protective of residential use. Conversion of air concentrations to soil PRGs is consistent with the methodology presented in *Supplemental Guidance for Developing Soil Screening Levels for Superfund Sites* (OSWER 9355.4-24) The air cleanup levels calculated using the above equation are converted to a soil PRG using the equation shown below.

$$Soil\ PRG\ (mg/kg) = \frac{Air\ PRG \times CF}{\left(\left(\frac{1}{PEF}\right)+\left(\frac{1}{VF}\right)\right)}$$

where:

UCF = unit conversion factor (µg/mg)
PEF = particulate emission factor (m^3/kg)
VF = volatilization factor (m^3/kg)—volatiles only

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. As described in Section 6.2.3.3, the resident Monument worker is a site-specific exposure scenario; site-specific exposure parameters for calculating the PRGs are defined in Table G-9 (Appendix G). 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 are 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).

PRG Equations for Incidental Soil Ingestion. The exposure assumptions used to calculate PRGs for the incidental soil ingestion route are presented and defined in Table G-9 (Appendix G). The following equations were used to calculate the risks from incidental ingestion of broad area soils for the occupational portion of the resident Monument worker exposure scenario:

$$CDI_{rmw_rad_ing_occ} = \frac{\left(1-e^{(-\lambda \times t_{rmw})}\right) \times C_s \times IRS_{rmw_occ_adj} \times EF_{rmw_occ} \times ED_{rmw} \times UCF1}{t_{rmw} \times \lambda}$$

where:

$$IRS_{rmw_occ_adj} = IRS_{rmw} \times \frac{EF_{rmw_occ}}{EF_{rmw_res}} \times \frac{ET_{rmw_occ}}{24hr/day}$$

and:

$CDI_{rmw_rad_ing_occ}$ = chronic daily intake-ingestion, occupational fraction (pCi)

λ = decay constant (years^{-1})
t_{rmw} = time (years)
C_s = concentration in soil-broad area (pCi/g)
$IRS_{rmw_occ_adj}$ = soil ingestion rate, occupational fraction, adjusted (mg/day)
IRS_{rmw} = soil ingestion rate (mg/day)
EF_{rmw_occ} = exposure frequency, occupation fraction (days/year)
EF_{rmw_res} = exposure frequency, residential fraction (days/year)
ED_{rmw} = exposure duration (years)
ET_{rmw_occ} = exposure time, occupational fraction (hours/day)
$UCF1$ = unit conversion factor (g/mg)

Radiological cancer risk is calculated using the following equation:

$$RISK_{rmw_rad_ing_occ} = CDI_{rmw_rad_ing_occ} \times SF_s$$

where:

$RISK_{rmw_rad_ing_occ}$ = cancer risk, broad area exposure, occupational fraction, incidental ingestion (unitless)
$CDI_{rmw_rad_ing_occ}$ = chronic daily intake, ingestion, occupational fraction (pCi)
SF_s = slope factor-soil ingestion (risk/pCi)

The following equations were used to calculate PRGs from incidental ingestion of local area soils for the residential portion of the resident Monument worker exposure scenario:

$$PRG_{rmw_rad_ing_res} = \frac{\left(TR_{_RAD} - RISK_{rmw_rad_ing_occ}\right) \times t_{rmw} \times \lambda}{\left(1 - e^{(-\lambda \times t_{rmw})}\right) \times SF_s \times IRS_{rmw_res_adj} \times EF_{rmw_res} \times ED_{rmw} \times UCF1}$$

where:

$$IRS_{rmw_res_adj} = IRS_{rmw} \times \left[\left(\frac{EF_{rmw_occ}}{EF_{rmw_res}} \times \frac{ET_{rmw_res}}{24 hrs / day}\right) + \left(\frac{EF_{rmw_res} - EF_{rmw_occ}}{EF_{rmw_res}}\right)\right]$$

and:

$PRG_{rmw_rad_ing_res}$ = preliminary remediation goal, ingestion, residential fraction (pCi/g)
$TR_{_rad}$ = target risk-radioisotopes (unitless)
$RISK_{rmw_rad_ing_occ}$ = cancer risk, broad area exposure, occupational fraction, incidental ingestion (unitless)
t_{rmw} = time (years)
λ = decay constant (years^{-1})
SF_s = slope factor-soil ingestion (risk/pCi)
$IRS_{rmw_res_adj}$ = soil ingestion rate, residential fraction, adjusted (mg/day)
EF_{rmw_res} = exposure frequency, residential fraction (days/year)
ED_{rmw} = exposure duration (years)
$UCF1$ = unit conversion factor (g/mg)
IRS_{rmw} = soil ingestion rate (mg/day)

EF_{rmw_occ}	=	exposure frequency, occupation fraction (days/year)
ET_{rmw_res}	=	exposure time, residential fraction (hours/day)
EF_{rmw_res}	=	exposure frequency, residential fraction (days/year)

PRG Equations for External Exposure to Ionizing Radiation. The exposure assumptions used to calculate PRGs for the external exposure to ionizing radiation route are presented and defined in Table G-9 (Appendix G). The following equations were used to calculate risks from external exposure to ionizing radiation from broad area soils for the occupational portion of the resident Monument worker exposure scenario:

$$CDI_{rmw_rad_ext_occ} = \frac{\left(1-e^{(-\lambda \times t_{rmw})}\right) \times C_s \times \left[\left(ET_{rmw_occ_out} \times GSF_{out}\right) + \left(ET_{rmw_occ_in} \times GSF_{in}\right)\right] \times EF_{rmw_occ} \times ED_{rmw} \times UCF3}{t_{rmw} \times \lambda}$$

where:

$CDI_{rmw_rad_ext_occ}$	=	chronic daily intake-external exposure, occupational fraction (pCi-year/g)
λ	=	decay constant (years^{-1})
t_{rmw}	=	time (years)
C_s	=	concentration in soil-broad area (pCi/g)
$ET_{rmw_occ_out}$	=	exposure time, occupational fraction, outdoors (hours/day)
GSF_{out}	=	gamma shielding factor, outdoors (unitless)
$ET_{rmw_occ_in}$	=	exposure time, occupational fraction, indoors (hours/day)
GSF_{in}	=	gamma shielding factor, indoors (unitless)
EF_{rmw_occ}	=	exposure frequency, occupation fraction (days/year)
ED_{rmw}	=	exposure duration (years)
$UCF3$	=	unit conversion factor (years/hour)

Radiological cancer risk is calculated using the following equation:

$$RISK_{rmw_rad_ext_occ} = CDI_{rmw_rad_ext_occ} \times SF_x$$

where:

$RISK_{rmw_rad_ext_occ}$	=	cancer risk, broad area exposure, ionizing radiation, occupational fraction (unitless)
$CDI_{rmw_rad_ext_occ}$	=	chronic daily intake, external exposure, occupational fraction (pCi-year/g)
SF_x	=	slope factor-external exposure (risk/pCi)

The following equation was used to calculate PRGs for the external exposure to ionizing radiation from local area soils for the residential portion of the resident Monument worker exposure scenario:

$$PRG_{rmw_rad_ext_res} = \frac{\left(TR_{_RAD} - RISK_{rmw_rad_ext_occ}\right) \times t_{rmw} \times \lambda}{\left(1-e^{(-\lambda \times t_{rmw})}\right) \times SF_s \times \left[\left(ET_{rmw_res_out} \times GSF_{out}\right) + \left(ET_{rmw_res_in} \times GSF_{in}\right)\right] \times EF_{rmw_res} \times ED_{rmw} \times UCF3}$$

and:

$PRG_{rmw_rad_ext_res}$	=	preliminary remediation goal, external exposure, residential fraction (pCi/g)
TR_rad	=	target risk-radioisotopes (unitless)

$RISK_{rmw_rad_ext_occ}$ = cancer risk, ionizing radiation, occupational fraction, (unitless)
t_{rmw} = time (years)
λ = decay constant (years^{-1})
SF_x = slope factor-external exposure ([risk/year]/[pCi/g])
$ET_{rmw_res_out}$ = exposure time, residential fraction, outdoors (hours/day)
GSF_{out} = gamma shielding factor, outdoors (unitless)
$ET_{rmw_res_in}$ = exposure time, residential fraction, indoors (hours/day)
GSF_{in} = gamma shielding factor, indoors (unitless)
EF_{rmw_res} = exposure frequency, residential fraction (days/year)
ED_{rmw} = exposure duration (years)
$UCF3$ = unit conversion factor (years/hour)

PRG Equations for the Inhalation of Dust in Soil. The exposure assumptions used to calculate PRGs for the inhalation of vapors and dust route are presented and defined in Table G-9 (Appendix G). The following equations were used to calculate risks from inhalation of dust from broad area soils for the occupational portion of the resident Monument worker exposure scenario:

$$CDI_{rmw_rad_inh_occ} = \frac{\left(1-e^{(-\lambda \times t_{rmw})}\right) \times C_s \times UCF2 \times \frac{1}{PEF_B} \times IRA_{rmw} \times ET_{rmw_occ} \times EF_{rmw_occ} \times ED_{rmw}}{t_{rmw} \times \lambda}$$

where:

$CDI_{rmw_rad_inh_occ}$ = chronic daily intake-inhalation, occupational fraction (pCi)
λ = decay constant (years^{-1})
t_{rmw} = time (years)
C_s = concentration in soil (broad area) (pCi/g)
$UCF2$ = unit conversion factor (g/kg)
PEF_B = particulate emission factor-broad area (m^3/kg)
IRA_{rmw} = inhalation rate (m^3/hour)
ET_{rmw_occ} = exposure time, occupational fraction (hours/day)
EF_{rmw_occ} = exposure frequency, occupation fraction (days/year)
ED_{rmw} = exposure duration (years)

Radiological cancer risk is calculated using the following equation:

$$RISK_{rmw_rad_inh_occ} = CDI_{rmw_rad_inh_occ} \times SF_i$$

where:

$RISK_{rmw_rad_inh_occ}$ = cancer risk, broad area exposure, inhalation, occupational fraction (unitless)
$CDI_{rmw_rad_inh_occ}$ = chronic daily intake, inhalation, occupational fraction (pCi)
SF_i = slope factor-inhalation (risk/pCi)

The following equation was used to calculate PRGs from inhalation of dust from local area soils for the residential portion of the resident Monument worker exposure scenario:

$$PRG_{rmw_rad_inh_res} = \frac{(TR_{_RAD} - RISK_{rmw_rad_inh_occ}) \times t_{rmw} \times \lambda}{(1 - e^{(-\lambda \times t_{rmw})}) \times SF_i \times UCF2 \times \frac{1}{PEF_L} \times IRA_{rmw} \times ET_{rmw_res} \times EF_{rmw_res} \times ED_{rmw}}$$

where:

$PRG_{rmw_rad_inh_res}$	=	preliminary remediation goal, inhalation residential fraction (pCi/g)
$TR_{_rad}$	=	target risk-radioisotopes (unitless)
$RISK_{rmw_rad_inh_occ}$	=	cancer risk, inhalation, occupational fraction, (unitless)
t_{rmw}	=	time (years)
λ	=	decay constant (years^{-1})
SF_i	=	slope factor-soil inhalation (risk/pCi)
$UCF2$	=	unit conversion factor (g/kg)
PEF_L	=	particulate emission factor-local area (m^3/kg)
IRA_{rmw}	=	inhalation rate (m^3/hour)
ET_{rmw_res}	=	exposure time, residential fraction (hours/day)
EF_{rmw_res}	=	exposure frequency, residential fraction (days/year)
ED_{rmw}	=	exposure duration (years)

PRG Equations Summing all Exposure Routes. The following presents the equations used to calculate the total PRGs for all exposure routes combined. The basis for the equation is provided in Risk Assessment Guidance Volume I, Part B (EPA/540/R-92/003).

$$PRG_{rmw_rad_total_res} = \frac{1}{\left(\left(\frac{1}{PRG_{rmw_rad_ing_res}}\right) + \left(\frac{1}{PRG_{rmw_rad_ext_res}}\right) + \left(\frac{1}{PRG_{rmw_rad_inh_res}}\right)\right)}$$

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 Preliminary Remediation Goals for Radionuclides tables (EPA, 2007b). As described in Section 6.2.3.3, the casual recreational user is a site-specific exposure scenario. Site-specific exposure parameters for calculating the PRGs are defined in Table G-10 (Appendix G). 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 are presented in *Calculation of Radiological Preliminary Remediation Goals in Soil for a Casual Recreational User Scenario for the 100 Areas and 300 Areas Remedial Investigation/ Feasibility Study (RI/FS) Reports* (ECF-HANFORD-10-0446).

PRG Equations for Incidental Soil Ingestion. The exposure assumptions used to calculate PRGs for the incidental soil ingestion route are presented and defined in Table G-10 (Appendix G). The following equations were used to calculate the PRG for the incidental soil ingestion route:

$$PRG_{cu_rad_ing} = \frac{TR_{_RAD} \times t_{cu} \times \lambda}{(1 - e^{(-\lambda t_{cu})}) \times SF_s \times IRS_{adj_cu_rad} \times EF_{cu} \times ED_{a+ccu} \times UCF3}$$

where:

$$IRS_{adj_cu_rad} = \frac{(ED_{ccu} \times IRS_{ccu}) + ((ED_{a+ccu} - ED_{ccu}) \times IRS_a)}{ED_{a+ccu}}$$

and:

$PRG_{cu_rad_ing}$	=	preliminary remediation goal, radionuclide, incidental soil ingestion (pCi/g)
$TR_{_rad}$	=	target risk-radioisotopes (unitless)
t_{cu}	=	time (years)
λ	=	decay constant (years^{-1})
SF_s	=	slope factor-soil ingestion (risk/pCi)
$IRS_{adj_cu_rad}$	=	soil ingestion rate, age-adjusted, radioisotopes (mg/day)
EF_{cu}	=	exposure frequency (days/year)
ED_{a+ccu}	=	exposure duration, adult +child (years)
$UCF3$	=	unit conversion factor (g/mg)
ED_{ccu}	=	exposure duration, child (years)
IRS_{ccu}	=	soil ingestion rate, child (mg/day)
IRS_a	=	soil ingestion rate, adult (mg/day)

PRG Equations for Exposure to Ionizing Radiation.The exposure assumptions used to calculate PRGs for the external exposure to ionizing radiation route are presented and defined in Table G-10. The following equation was used to calculate the PRG for the external exposure to ionizing radiation route:

$$PRG_{cu_rad_ext} = \frac{TR_{_RAD} \times t_{cu} \times \lambda}{\left(1 - e^{(-\lambda t_{cu})}\right) \times SF_x \times ACF \times EF_{cu} \times ED_{a+ccu} \times ET_{cu} \times \frac{1\,day}{24\,hours} \times UCF5}$$

where:

$PRG_{cu_rad_ext}$	=	preliminary remediation goal, radionuclide, external exposure (pCi/g)
$TR_{_rad}$	=	target risk-radioisotopes (unitless)
t_{cu}	=	time (years)
λ	=	decay constant (years^{-1})
SF_x	=	slope factor-external exposure ([risk/year]/[pCi/g])
ACF	=	area correction factor (unitless)
EF_{cu}	=	exposure frequency (days/year)
ED_{a+ccu}	=	exposure duration, adult +child (years)
ET_{cu}	=	exposure time (hours/day)
$UCF5$	=	unit conversion factor (years/day)

PRG Equations for the Inhalation of Vapors and Dust. The exposure assumptions used to calculate PRGs for the inhalation of vapors and dust route are presented and defined in Table G-10 (Appendix G). The following equations were used to calculate the PRG for the inhalation of vapors and dust route:

$$PRG_{cu_rad_inh} = \frac{TR_{_RAD} \times t_{cu} \times \lambda}{\left(1 - e^{(-\lambda t_{cu})}\right) \times SF_i \times IRA_{adj_cu} \times EF_{cu} \times ED_{a+ccu} \times ET_{cu} \times \left(\frac{1}{PEF} or \frac{1}{VF}\right) \times UCF4}$$

where:

$$IRA_{adj_cu} = \frac{(ED_{ccu} \times IRA_{ccu}) + ((ED_{a+ccu} - ED_{ccu}) \times IRA_a)}{ED_{a+ccu}}$$

and:

$PRG_{cu_rad_inh}$	=	preliminary remediation goal, radionuclide, inhalation (pCi/g)
$TR_{_rad}$	=	target risk-radioisotopes (unitless)
t_{cu}	=	time (years)
λ	=	decay constant (years^{-1})
SF_i	=	slope factor-inhalation (risk/pCi)
IRA_{adj_cu}	=	inhalation rate, age-adjusted (m^3/hour)
EF_{cu}	=	exposure frequency (days/year)
ED_{a+ccu}	=	exposure duration, adult +child (years)
ET_{cu}	=	exposure time (hours/day)
PEF	=	particulate emission factor (m^3/kg)
VF	=	volatilization factor (m^3/kg)
$UCF4$	=	unit conversion factor (g/kg)
ED_{ccu}	=	exposure duration, child (years)
IRA_{ccu}	=	inhalation rate, child (m^3/hour)
IRa_a	=	inhalation rate, adult (m^3/hour)

PRG Equation Summing all Exposure Routes. The following presents the equations used to calculate the total PRGs for all exposure routes combined. The basis for the equation is provided in Risk Assessment Guidance Volume I, Part B (EPA/540/R-92/003).

$$PRG_{cu_rad_total} = \frac{1}{\left(\left(\frac{1}{PRG_{cu_rad_ing}}\right)+\left(\frac{1}{PRG_{cu_rad_ext}}\right)+\left(\frac{1}{PRG_{cu_rad_inh}}\right)\right)}$$

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, 2009]). As described in Section 6.2.3.3, the casual recreational user is a site-specific exposure scenario. Exposure parameters for calculating the PRGs are defined in Table G-10 (Appendix G). 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. Reference dose 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 Areas Remedial Investigation/Feasibility Study (RI/FS) Reports* (ECF-HANFORD-10-0445).

PRG Equations for Incidental Soil Ingestion. The exposure assumptions used to calculate PRGs for the incidental soil ingestion route are presented and defined in Table G-10 (Appendix G).

Carcinogenic and Noncarcinogenic Effects. The following equations are used to calculate the carcinogenic PRG for incidental soil ingestion:

$$PRG_{cu_ca_ing} = \frac{TR \times AT_{ca}}{CSF_o \times IRS_{adj_cu} \times EF_{cu} \times UCF1}$$

where:

$$IRS_{adj_cu} = \left(\frac{ED_{ccu} \times IRS_{ccu}}{BW_{ccu}}\right) + \left(\frac{(ED_{a+ccu} - ED_{ccu}) \times IRS_a}{BW_a}\right)$$

and:

$PRG_{cu_ca_ing}$	=	preliminary remediation goal, carcinogen, incidental soil ingestion (mg/kg)
TR	=	target risk (unitless)
AT_{ca}	=	averaging time-carcinogen (days)
CSF_o	=	cancer slope factor-oral (mg/kg-day)$^{-1}$
IRS_{adj_cu}	=	soil ingestion rate, age-adjusted (mg-yr/kg-day)
EF_{cu}	=	exposure frequency (days/year)
$UCF1$	=	unit conversion factor (kg/mg)
ED_{ccu}	=	exposure duration, child (years)
ED_{a+ccu}	=	exposure duration, adult + child (years)
IRS_{cu}	=	soil ingestion rate, child (mg/day)
IRS_a	=	soil ingestion rate, adult (mg/day)
BW_{ccu}	=	body weight, child (kg)
BW_a	=	body weight, adults (kg)

The following equation is used to calculate the noncarcinogenic PRG for incidental soil ingestion:

$$PRG_{cu_nc_ing} = \frac{THQ \times AT_{nc_ccu} \times BW_{ccu}}{\frac{1}{RfD_o} \times IRS_{ccu} \times EF_{cu} \times ED_{ccu} \times UCF1}$$

where:

$PRG_{cu_nc_ing}$	=	PRG, noncarcinogen, incidental soil ingestion (mg/kg)
THQ	=	target hazard quotient (unitless)
AT_{nc_cu}	=	averaging time-noncarcinogen, child (days)
BW_{ccu}	=	body weight, child (kg)
RfD_o	=	reference dose-oral (mg/kg-day)
IRS_{ccu}	=	soil ingestion rate, child (mg/day)
EF_{cu}	=	exposure frequency (days/year)
ED_{ccu}	=	exposure duration, child (years)
$UCF1$	=	unit conversion factor (kg/mg)

Mutagenic Effects. The following equations are used to calculate the mutagenic mode of action PRG for incidental soil ingestion:

$$PRG_{cu_mu_ing} = \frac{TR \times AT_{ca}}{CSF_o \times IRSM_{adj_cu} \times EF_{cu} \times UCF1}$$

where:

$$IRSM_{adj_cu} = \frac{ED_{0-2} \times IRS_{ccu} \times 10}{BW_{ccu}} + \frac{ED_{2-6} \times IRS_{ccu} \times 3}{BW_{ccu}} + \frac{ED_{6-16} \times IRS_a \times 3}{BW_a} + \frac{ED_{16-30} \times IRS_a \times 1}{BW_a}$$

and:

$PRG_{cu_mu_ing}$	=	preliminary remediation goal, mutagen, incidental soil ingestion (mg/kg)
TR	=	target risk (unitless)
AT_{ca}	=	averaging time-carcinogen (days)
CSF_o	=	cancer slope factor, oral (mg/kg-day)$^{-1}$
$IRSM_{adj_cu}$	=	soil ingestion rate, age-adjusted-mutagens (mg-yr/kg-day)
BW_{ccu}	=	body weight, child (kg)
EF_{cu}	=	exposure frequency (days/year)
$UCF1$	=	unit conversion factor (kg/mg)
$ED_{0\text{-}2}$	=	exposure duration, 0-2 years (years)
IRS_{ccu}	=	soil ingestion rate, child (mg/day)
$ED_{2\text{-}6}$	=	exposure duration, 2-6 years (years)
$ED_{6\text{-}16}$	=	exposure duration, 6-16 years (years)
IRS_a	=	soil ingestion rate, adult (mg/day)
$ED_{16\text{-}30}$	=	exposure duration, 16-30 years (years)
BW_a	=	body weight, adult (kg)

Vinyl Chloride. The following equations are used to calculate the vinyl chloride PRG for incidental soil ingestion:

$$PRG_{cu_vc_ing} = \frac{TR}{\left(\frac{CSF_o \times IRS_{adj_cu} \times EF_{cu} \times UCF1}{AT_{ca}} + \frac{CSF_o \times IRS_{ccu} \times UCF1}{BW_{ccu}}\right)}$$

where:

$$IRS_{adj_cu} = \left(\frac{ED_{ccu} \times IRS_{ccu}}{BW_{ccu}}\right) + \left(\frac{(ED_{a+ccu} - ED_{ccu}) \times IRS_a}{BW_a}\right)$$

and:

$PRG_{cu_vc_ing}$	=	preliminary remediation goal, vinyl chloride, incidental soil ingestion (mg/kg)
TR	=	target risk (unitless)
CSF_o	=	cancer slope factor, oral (mg/kg-day)$^{-1}$
IRS_{adj_cu}	=	soil ingestion rate, age-adjusted (mg-yr/kg-day)
EF_{cu}	=	exposure frequency (days/year)
$UCF1$	=	unit conversion factor (kg/mg)
AT_{ca}	=	averaging time-carcinogen (days)
ED_{ccu}	=	exposure duration, child (years)
IRS_{ccu}	=	soil ingestion rate, child (mg/day)
BW_{ccu}	=	body weight, child (kg)
ED_{a+ccu}	=	exposure duration, adult+child (years)
IRS_a	=	soil ingestion rate, adult (mg/day)
BW_a	=	body weight, adult (kg)

PRG Equations for Dermal Contact with Soil. The exposure assumptions used to calculate PRGs for the dermal contact with soil route are presented and defined in Table G-10 (Appendix G). The following equations were used to calculate the PRG for the dermal contact with soil route:

Carcinogenic and Noncarcinogenic Effects. The following equations are used to calculate the carcinogenic PRG for the dermal contact route:

$$PRG_{cu_ca_dc} = \frac{TR \times AT_{ca}}{\left(\frac{CSF_o}{GIABS}\right) \times EF_{cu} \times DFS_{adj_cu} \times ABS_d \times UCF1}$$

where:

$$DFS_{adj_cu} = \left(\frac{ED_{ccu} \times SA_{ccu} \times AF_{ccu}}{BW_{ccu}}\right) + \left(\frac{(ED_{a+ccu} - ED_{ccu}) \times SA_a \times AF_a}{BW_a}\right)$$

and:

$PRG_{cu_vc_ing}$	=	preliminary remediation goal, vinyl chloride, incidental soil ingestion (mg/kg)
TR	=	target risk (unitless)
CSF_o	=	cancer slope factor, oral (mg/kg-day)$^{-1}$
IRS_{adj_cu}	=	soil ingestion rate, age-adjusted (mg-yr/kg-day)
EF_{cu}	=	exposure frequency (days/year)
$UCF1$	=	unit conversion factor (kg/mg)
AT_{ca}	=	averaging time-carcinogen (days)
ED_{ccu}	=	exposure duration, child (years)
IRS_{ccu}	=	soil ingestion rate, child (mg/day)
BW_{ccu}	=	body weight, child (kg)
ED_{a+ccu}	=	exposure duration, adult+child (years)
IRS_a	=	soil ingestion rate, adult (mg/day)
BW_a	=	body weight, adult (kg)

The following equation is used to calculate the noncarcinogenic PRG for the dermal contact route:

$$PRG_{cu_nc_dc} = \frac{THQ \times AT_{nc_cu} \times BW_{ccu}}{\left(\frac{1}{RfD_o} \times GIABS\right) \times SA_{ccu} \times AF_{ccu} \times EF_{cu} \times ED_{ccu} \times ABS_d \times UCF1}$$

where:

$PRG_{cu_nc_dc}$	=	preliminary remediation goal, noncarcinogen, dermal contact (mg/kg)
THQ	=	target hazard quotient (unitless)
AT_{nc_cu}	=	averaging time-noncarcinogen, child (days)
BW_{ccu}	=	body weight, child (kg)
RfD_o	=	reference dose, oral (mg/kg-day)
$GIABS$	=	gastro-intestinal absorption fraction (unitless)
SA_{ccu}	=	skin surface area, child (cm^2/day)
AF_{ccu}	=	skin adherence factor, child (mg/cm^2)
EF_{cu}	=	exposure frequency (days/year)
ED_{ccu}	=	exposure duration, child (years)
ABS_d	=	dermal absorption fraction (unitless)
$UCF1$	=	unit conversion factor (kg/mg)

Mutagenic Effects. The following equations are used to calculate the mutagenic mode of action PRG for the dermal contact route:

$$PRG_{cu_mu_dc} = \frac{TR \times AT_{ca}}{\frac{CSF_o}{GIABS} \times DFSM_{adj_cu} \times EF_{cu} \times ABS_d \times UCF1}$$

where:

$$DFSM_{adj_cu} = \frac{ED_{0-2} \times SA_{ccu} \times AF_{ccu} \times 10}{BW_{ccu}} + \frac{ED_{2-6} \times SA_{ccu} \times AF_{ccu} \times 3}{BW_{ccu}} + \frac{ED_{6-16} \times SA_a \times AF_a \times 3}{BW_a} + \frac{ED_{16-30} \times SA_a \times AF_a \times 1}{BW_a}$$

and:

$PRG_{cu_mu_dc}$	=	preliminary remediation goal, mutagen, dermal contact (mg/kg)
TR	=	target risk (unitless)
AT_{ca}	=	averaging time, carcinogen (days)
CSF_o	=	cancer slope factor, oral $(mg/kg\text{-}day)^{-1}$
$GIABS$	=	gastro-intestinal absorption fraction (unitless)
$DFSM_{adj_cu}$	=	dermal contact rate, age-adjusted-mutagens (mg-year/kg-day)
EF_{cu}	=	exposure frequency (days/year)
ABS_d	=	dermal absorption fraction (unitless)
$UCF1$	=	unit conversion factor (kg/mg)
$ED_{0\text{-}2}$	=	exposure duration, 0-2 years (years)
SA_{ccu}	=	skin surface area, child (cm^2/day)
AF_{ccu}	=	skin adherence factor, child (mg/cm^2)
BW_{ccu}	=	body weight, child (kg)
$ED_{2\text{-}6}$	=	exposure duration, 2-6 years (years)
$ED_{6\text{-}16}$	=	exposure duration, 6-16 years (years)
$ED_{16\text{-}30}$	=	exposure duration, 16-30 years (years)
SA_a	=	skin surface area, adult (cm^2/day)
AF_a	=	skin adherence factor, adult (mg/cm^2)
BW_a	=	body weight, adult (kg)

Vinyl Chloride. The following equations are used to calculate the vinyl chloride PRG for the dermal contact route:

$$PRG_{cu_vc_dc} = \frac{TR}{\left(\frac{\frac{CSF_o}{GIABS} \times DSF_{adj_cu} \times EF_{cu} \times ABS_d \times UCF1}{AT_{ca}} + \frac{\frac{CSF_o}{GIABS} \times SA_{ccu} \times AF_{ccu} \times ABS_d \times UCF1}{BW_{ccu}} \right)}$$

where:

$$DFS_{adj_cu} = \left(\frac{ED_{ccu} \times SA_{ccu} \times AF_{ccu}}{BW_{ccu}} \right) + \left(\frac{(ED_{a+ccu} - ED_{ccu}) \times SA_a \times AF_a}{BW_a} \right)$$

and:

$PRG_{cu_vc_dc}$	=	preliminary remediation goal, vinyl chloride, dermal contact (mg/kg)

TR	=	target risk (unitless)
CSF_o	=	cancer slope factor, oral $(mg/kg\text{-}day)^{-1}$
$GIABS$	=	gastro-intestinal absorption fraction (unitless)
DFS_{adj_cu}	=	dermal contact rate, age-adjusted (mg-year/kg-day)
EF_{cu}	=	exposure frequency (days/year)
ABS_d	=	dermal absorption fraction (unitless)
$UCF1$	=	unit conversion factor (kg/mg)
AT_{ca}	=	averaging time-carcinogenic (days)
SA_{ccu}	=	skin surface area, child (cm^2/day)
AF_{ccu}	=	skin adherence factor, child (mg/cm^2)
BW_{ccu}	=	body weight, child (kg)
ED_{a+ccu}	=	exposure duration, adult+child (years)
ED_{ccu}	=	exposure duration, child (years)
SA_a	=	skin surface area, adult (cm^2/day)
AF_a	=	skin adherence factor, adult (mg/cm^2)
BW_a	=	body weight, adult (kg)

PRG Equations for the Inhalation of Vapors and Dust in Soil. The exposure assumptions used to calculate PRGs for the inhalation of vapors and dust route are presented and defined in Table G-10. The following equations were used to calculate the PRG for the inhalation route:

Carcinogenic and Noncarcinogenic Effects. The following equation is used to calculate the carcinogenic PRG for the inhalation route:

$$PRG_{cu_ca_inh} = \frac{TR \times AT_{ca} \times 24\frac{hours}{day}}{IUR \times UCF2 \times EF_{cu} \times ED_{a+ccu} \times ET_{cu} \times \left(\frac{1}{VF} + \frac{1}{PEF}\right)}$$

where:

$PRG_{cu_ca_inh}$	=	preliminary remediation goal, carcinogen, inhalation (mg/kg)
TR	=	target risk (unitless)
AT_{ca}	=	averaging time-carcinogen (days)
IUR	=	inhalation unit risk $(\mu g/m^3)^{-1}$
$UCF2$	=	unit conversion factor (µg/mg)
EF_{cu}	=	exposure frequency (days/year)
ED_{a+ccu}	=	exposure duration, adult+child (years)
ET_{cu}	=	exposure time, (hours/day)
VF	=	volatilization factor (m^3/kg)
PEF	=	particulate emission factor (m^3/kg)

The following equation is used to calculate the noncarcinogenic PRG for the inhalation route:

$$PRG_{cu_nc_inh} = \frac{THQ \times AT_{nc_cu} \times 24\frac{hours}{day}}{EF_{cu} \times ED_{ccu} \times ET_{cu} \times \frac{1}{RfC} \times \left(\frac{1}{VF} + \frac{1}{PEF}\right)}$$

where:

$PRG_{cu_nc_inh}$	=	preliminary remediation goal, noncarcinogen, inhalation (mg/kg)
THQ	=	target hazard quotient (unitless)
AT_{nc_cu}	=	averaging time-noncarcinogen, child (days)
EF_{cu}	=	exposure frequency (days/year)
ED_{ccu}	=	exposure duration, child (years)
ET_{cu}	=	exposure time, (hours/day)
RfC	=	reference concentration (mg/m^3)
VF	=	volatilization factor (m^3/kg)
PEF	=	particulate emission factor (m^3/kg)

Mutagenic Effects. The following equation is used to calculate the mutagenic mode of action PRG for the inhalation route:

$$PRG_{cu_mu_inh} = \frac{TR \times AT_{ca} \times 24 \frac{hours}{day}}{UCF2 \times ET_{cu} \times EF_{cu} \times \left[\left(ED_{0-2} \times IUR \times 10\right) + \left(ED_{2-6} \times IUR \times 3\right) + \left(ED_{6-16} \times IUR \times 3\right) + \left(ED_{16-30} \times IUR \times 1\right)\right] \times \left(\frac{1}{VF} + \frac{1}{PEF}\right)}$$

where:

$PRG_{cu_mu_inh}$	=	preliminary remediation goal, mutagen, inhalation (mg/kg)
TR	=	target risk (unitless)
AT_{ca}	=	averaging time-carcinogen (days)
$UCF2$	=	unit conversion factor (µg/mg)
ET_{cu}	=	exposure time, (hours/day)
EF_{cu}	=	exposure frequency (days/year)
$ED_{0\text{-}2}$	=	exposure duration, 0-2 years (years)
IUR	=	inhalation unit risk (µg/m^3)$^{-1}$
$ED_{2\text{-}6}$	=	exposure duration, 2-6 years (years)
$ED_{6\text{-}16}$	=	exposure duration, 6-16 years (years)
$ED_{16\text{-}30}$	=	exposure duration, 16-30 years (years)
VF	=	volatilization factor (m^3/kg)
PEF	=	particulate emission factor (m^3/kg)

Vinyl Chloride. The following equation is used to calculate the vinyl chloride PRG for the inhalation route:

$$PRG_{cu_vc_inh} = \frac{TR}{\left(\frac{IUR \times ED_{a+ccu} \times EF_{cu} \times ET_{cu} \times UCF2}{AT_{ca} \times 24 \frac{hours}{day} \times VF}\right) + \left(\frac{IUR}{VF} \times UCF2\right)}$$

where:

$PRG_{cu_vc_inh}$	=	preliminary remediation goal, vinyl chloride, inhalation (mg/kg)
TR	=	target risk (unitless)
IUR	=	inhalation unit risk (µg/m^3)$^{-1}$
ED_{a+ccu}	=	exposure duration, adult+child (years)
EF_{cu}	=	exposure frequency (days/year)
ET_{cu}	=	exposure time (hour/day)
$UCF2$	=	unit conversion factor (µg/mg)

AT_{ca}	=	averaging time-carcinogen (days)
VF	=	volatilization factor (m^3/kg)

6.2.3.4.6 Calculation of Particulate Emission Factor and Volatilization Factors

The default input parameters used to calculate the particulate emission factor for the resident Monument worker are presented and defined in Table G-11 (Appendix G). The default input parameters used to calculate the particulate emission factor and volatilization factors for the casual recreational user are presented and defined in Table G-12 (Appendix G). The input parameters used to calculate chemical-specific volatilization factors are listed in Table G-13 (Appendix G). Volatilization factors are only calculated for VOCs. A VOC is defined by *Soil Screening Guidance: Users Guide* (EPA/540/R-96/018) as a chemical with a Henry's Law constant of 1×10^{-5} or greater and with a molecular weight of less than 200 g/mole. The volatilization factor for tritium is not a calculated value, but is a default value published in EPA Radiation Guidance for CERCLA: Risk Assessment Web Site (EPA, 2010b). The following presents the equations used to calculate the chemical-specific volatilization factors.

$$VF = \frac{\frac{Q}{C} \times \left(3.14 \times D_A \times T\right)^{\frac{1}{2}} \times 10^{-4} \left(\frac{m^2}{cm^2}\right)}{2 \times \rho_b \times D_A}$$

where:

$$\frac{Q}{C} = A \times \exp\left[\frac{\left(\ln\left[A_S\right] - B\right)^2}{C}\right]$$

and:

$$D_A = \left[\frac{\frac{\left(\theta_a^{\frac{10}{3}} \times D_i \times H' + \theta_w^{\frac{10}{3}} \times D_w\right)}{n^2}}{\rho_b \times k_{oc} \times f_{oc} + \theta_w + \theta_a \times H'}\right]$$

and:

VF	=	volatilization factor (mg^3/kg)
Q/C	=	inverse of the mean concentration at center of a source ([g/m^2-s]/[kg/m^3])
D_A	=	apparent diffusivity (cm^2/second)
T	=	exposure time (seconds)
ρ_b	=	dry soil bulk density (kg/liter)
A	=	constant (unitless)
A_s	=	site area (acres)
B	=	constant (unitless)
C	=	constant (unitless)

θ_a	=	air-filled porosity ($liters_{air}/liter_{soil}$)
D_i	=	diffusivity in air (cm^2/second)
H'	=	Henry's constant (dimensionless)
θ_w	=	water-filled porosity ($liters_{water}/liter_{soil}$)
D_w	=	diffusivity in water (cm^2/second)
n	=	soil porosity (unitless)
k_{oc}	=	soil organic carbon/water partition coefficient (liters/kg)
f_{oc}	=	fraction organic carbon in soil (gram/gram)

Particulate Emission Factor. The particulate emission factor is calculated using the following equations:

$$PEF = \frac{Q}{C} \times \frac{3{,}600}{0.036 \times (1-V) \times \left(\frac{u_m}{u_t}\right)^3 \times F(x)}$$

where:

$$\frac{Q}{C} = A \times \exp\left[\frac{(\ln(A_S) - B)^2}{C}\right]$$

and:

$$F(x) = 0.18(8x^3 + 12x)\exp-(x^2)$$

and:

$$x = 0.886\left(\frac{u_t}{u_m}\right)$$

and:

PEF	=	particulate emission factor (mg^3/kg)
Q/C	=	inverse of the mean concentration at center of a source ([g/m^2-s]/[kg/m^3])
V	=	fraction of vegetative cover (unitless)
u_m	=	annual average wind speed (m/second)
u_t	=	wind speed at anemometer height equivalent to corrected threshold friction velocity (m/second)
$F(x)$	=	wind speed distribution function (unitless)
A	=	constant (unitless)
A_s	=	site area (acres)
B	=	constant (unitless)
C	=	constant (unitless)
X	=	constant (unitless)

PRG Equations Summing all Exposure Routes. The following presents the equations used to calculate the total PRGs for all exposure routes combined.

The basis for the equation is provided in Risk Assessment Guidance Volume I, Part B (EPA/540/R-92/003).

Carcinogenic Effects

$$PRG_{cu_ca_total} = \frac{1}{\left(\left(\frac{1}{PRG_{cu_ca_ing}}\right)+\left(\frac{1}{PRG_{cu_ca_dc}}\right)+\left(\frac{1}{PRG_{cu_ca_inh}}\right)\right)}$$

Noncarcinogenic Effects

$$PRG_{cu_nc_total} = \frac{1}{\left(\left(\frac{1}{PRG_{cu_nc_ing}}\right)+\left(\frac{1}{PRG_{cu_nc_dc}}\right)+\left(\frac{1}{PRG_{cu_nc_inh}}\right)\right)}$$

Mutagenic Effects

$$PRG_{cu_mu_total} = \frac{1}{\left(\left(P\frac{1}{PRG_{cu_mu_ing}}\right)+\left(\frac{1}{PRG_{cu_mu_dc}}\right)+\left(\frac{1}{PRG_{cu_mu_inh}}\right)\right)}$$

Vinyl Chloride

$$PRG_{cu_vc_total} = \frac{1}{\left(\left(\frac{1}{PRG_{cu_vc_ing}}\right)+\left(\frac{1}{PRG_{cu_vc_dc}}\right)+\left(\frac{1}{PRG_{cu_vc_inh}}\right)\right)}$$

6.2.4 Toxicity Assessment

This toxicity assessment evaluates the relationship between the magnitude of exposure to a contaminant at the 100-BC 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 subsections.

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 (e.g., 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 (i.e., the most sensitive adverse effect following exposure). Table G-14 (Appendix G) lists the COPCs detected at the 100-BC 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 of time. 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 (i.e., 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-BC Source OU are summarized in Table G-14 (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 (i.e., 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-14 (Appendix G).

6.2.4.2 Toxicity Values

The analyte-specific toxicity values presented Table G-14 (Appendix G) are determined using the recommended reference hierarchy as described in Superfund HHT Risk Assessment Values (Cook, 2003). The hierarchy is summarized below.

- 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 used in preference 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) Toxicity Criteria Database contains toxicity values that are peer reviewed and address both cancer and noncancer effects.
- The Agency for Toxic Substances and Disease Registry (ATSDR) 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).

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 (RAIS) (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 NO3- = (14 g/mol)/(62 g/mol) = 0.226. The derived RfD for nitrate = (1.6 mg NO3-N/kg-day) × (1 mg NO3-/0.226 mg NO3-N) = 7.1 mg NO3-/kg-day.

A derived RfD for nitrite was calculated from the RfD reported in IRIS (0.1 mg/kg-day) for nitrite as nitrogen (NO2--N) using the mass fraction of nitrogen in nitrite. The mass fraction of nitrogen in nitrite = mol wt N/mol wt NO2- = (14 g/mol)/(46 g/mol) = 0.304. The derived RfD for nitrite = (0.1 mg NO2-N/kg-day) × (1 mg NO2-/0.304 mg NO2-N) = 0.3 mg NO2-/kg-day.

Toxic equivalence factors were used to calculate toxicity values for dioxins, furans, and carcinogenic PAHs as described in MTCA 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 differences in toxicity values are summarized below:

- For consistency with previous Hanford 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 Health Effects Assessment Summary Table (HEAST) are used for this assessment. HEAST 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 by EPA in the Regional Screening Levels (EPA, 2009a) are established by the CalEPA Office of 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 Chemicals in Drinking Water, Trichloroethylene* (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 values in this assessment has the potential to over-estimate cancer risk.

- For fluoride, the oral reference dose 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 reference dose currently implemented by EPA in the Regional Screening Levels (EPA, 2009a) is established by the CalEPA OEHHA. The oral reference dose 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 reference dose 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) soil cleanup 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 Ecology's *Cleanup Levels and Risk Calculations (CLARC) database* (hereinafter called the CLARC database; Ecology, 2010). In addition, the reference concentration of 0.7 (mg/m^3) published by HEAST 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) soil cleanup 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-14 (Appendix G) are determined using the recommended values from HEAST Radionuclides Table.

6.2.5 Risk Characterization

The risk characterization step is completed through the comparison of the EPC to the preliminary remediation goal. This comparison step is used to determine whether the post-remediation soil concentrations are protective of human health. It is also used to determine if 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 equation presented below is consistent with that published in Regional Screening Values (EPA, 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 (i.e., its RfD). The ratio of the chronic daily intake divided by RfD is termed 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 equation presented below is consistent with that published in Regional Screening Levels (EPA, 2009a).

$$HI_T = \sum_i \frac{EPC_{soil}}{PRG_{noncarcinogen}}$$

where:

HI_T	=	Total HI for all chemicals
EPC_{soil}	=	EPC in soil (μg/kg)
$PRG_{noncarcinogen}$ (μg/kg)	=	Preliminary remediation goal based on HQ=1 noncarcinogenic effects
i	=	The sum of the ratios for the ith chemical

6.2.5.2.1 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 above). 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 the EPA's Integrated Exposure Uptake Biokinetic (IEUBK) model, which is available on the EPA website. The IEUBK model is designed to predict probable blood-lead concentrations for children between 6 months and 84 months (i.e., up to 7 years) of age who have been exposed to lead through various sources (e.g., air, water, soil, dust, and in utero contributions from the mother).

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. The concentration of 20 mg/kg is based on the 90th percentile background value for Washington State documented in *Natural Background Soil Metals Concentrations in Washington State* (Ecology Publication 94-115).

6.2.5.2.2 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.

6.2.5.2.3 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 *Hanford Site Background: Part 1, Soil Background for Nonradioactive Analytes* (hereinafter called 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* (hereinafter called Review of Metal Concentrations [PNNL-18577]). 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 indicate 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 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) provided adequate analytical results.

Radionuclide background values (lognormal 90th percentile and maximum) are identified in *Hanford Site Background: Part 2, Soil Background for Radionuclides* (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), *Hanford Site Background: Part 2, Soil Background for Radionuclides* (DOE/RL-96-12), and Review of Metal Concentrations (PNNL-18577) is representative of the sedimentary facies in the vadose zone at the 100-BC 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-15 (Appendix G) lists the maximum and 90th percentile background concentration values for inorganic chemicals and radionuclides.

6.2.5.2.4 Comparison of Site and Background Risk Contributions.

Understanding the contribution to risk from naturally occurring elements is important because RAGs are not set at concentrations below natural background levels under CERCLA. Similarly, MTCA "Overview of Cleanup Standards", "Natural Background and Analytical Considerations" (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 for contaminants of concern that warrant remedial action.*

The 90th percentile value is used as a fixed benchmark concentration for the purpose of determining which contaminants should be evaluated for purposes of background risk. To assist risk managers make 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-15 (Appendix G). A comparison of EPCs to the lognormal 90th percentile value for each decision unit is provided in Table G-16 (Appendix G) for the 100-BC Source OU.
- 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.

6.2.5.3 Summary of Risk Estimates by Exposure Scenario

This section summarizes the risk estimates for each of the exposure scenarios considered for the 100-BC Source OU.

6.2.5.3.1 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. PRGs 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.

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 the background concentrations, are presented in Tables G-17 through G-25 (Appendix G).

Appendix G also includes risk estimates for each waste site decision unit, which include only those COPCs with EPCs greater than background values or do not have a background value. These risk estimates are presented in Tables G-26 through G-35 (Appendix G). Only the risk estimates without background contributions are discussed in the risk characterization because this information is used for decisions concerning appropriate remedial actions.

Risk estimates were calculated for each decision unit within a waste site including shallow vadose zone material, overburden material, staging pile area material, and deep vadose zone material. The results without background contribution for the residential scenario are summarized in Table G-26 through Table G-28 (Appendix G). An overall summary of the cumulative risk estimates for each of the waste sites evaluated is provided in Table 6-12 for the shallow zone and Table 6-13 for the deep zone.

Shallow Zone. A total of 75 waste sites are reported with CVP/RSVP data associated with the shallow zone in the 100-BC Source OU. The following sample designs were applied to the 75 waste sites evaluated:

- Twenty-one waste sites were sampled using a focused sampling design.
- Forty-one waste sites were sampled using a statistical sampling design (one site with two statistically distinct decision units and one site have three statistically distinct decision units).
- Thirteen waste sites were sampled using both a statistical and a focused sampling design (two waste sites each with two statistically distinct decision units; two waste sites each with three statistically distinct decision units; two waste sites each with four statistically distinct decision units; and one waste site with seven statistically distinct decision units).

The overall total risk summary for the residential scenario from all shallow zone waste sites evaluated is provided in Table 6-12. Table 6-12 lists the OU that each waste site resides in, the reclassification status, the remediated waste site, the consolidated 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).

As presented in Table 6-12, the potential cumulative total ELCR is greater than or equal to the upper risk threshold of 1×10^{-4} at seven remediated waste sites, is within the regulatory target risk range of 10^{-4} to 10^{-6} at 32 remediated waste sites, and is less than the lower risk threshold of 1×10^{-6} at 20 remediated waste sites. Risks were not reported at 16 remediated waste sites because there were no COPCs reported above background concentrations.

Seven waste sites report concentrations of site-related COPCs that are equal to or exceed the upper threshold of the regulatory target risk range. The cancer risk levels for the residential scenario are as follows:

- The 100-B-21:4 waste site (shallow decision unit) reports a cumulative total ELCR of 1.1×10^{-3}. The primary contributor to risk is cesium-137 (1.0×10^{-3}, 91 percent contribution). The EPC of cesium-137 is 45 pCi/g, which is greater than the residential RBSL of 4.4 pCi/g and is also greater than the current direct exposure RAG of 6.2 pCi/g published in the 100 Area RDR/RAWP (DOE/RL-96-17). Although the samples from this decision unit were assigned to the shallow zone, WSRF-2009-041 indicate theses samples were collected from a depth of 7.5 m (25 ft) bgs.

- The 116-B-5 waste site (shallow focused decision unit) reports a cumulative total ELCR of 4.0×10^{-4}. The primary contributors to risk include europium-152 (2.0×10^{-4}; 50 percent contribution) and tritium (1.1×10^{-4}; 27 percent contribution). The EPC of europium-152 is 7.3 pCi/g, which is greater than the residential RBSL of 3.7 pCi/g and is also greater than the current direct exposure RAG of 3.3 pCi/g published in the 100 Area RDR/RAWP (DOE/RL-96-17).The EPC of tritium is 680 pCi/g, which is greater than the residential RBSL of 623 pCi/g and is also greater than the current direct exposure RAG of 459 pCi/g published in the 100 Area RDR/RAWP (DOE/RL-96-17). WSRF-98-064 indicates that the final excavation depth at this site was 5 m (16 ft) bgs.

- The 116-B-6A waste site (shallow decision unit) reports a cumulative total ELCR of 2.1×10^{-4}. The primary contributors to risk include cesium-137 (9.5×10^{-5}; 45 percent contribution) and strontium-90 (1.1×10^{-4}; 54 percent contribution). The EPC of strontium-90 is 2.6 pCi/g, which is greater than the residential RBSL of 2.3 pCi/g. However, the EPC is less than the current direct exposure RAG of 4.5 pCi/G published in the 100 Area RDR/RAWP (DOE/RL-96-17). The EPC of cesium-137 is 4.2 pCi/g, which is less than the residential RBSL of 4.4 pCi/g and is also less than the current direct exposure RAG of 6.2 pCi/g published in the 100 Area RDR/RAWP (DOE/RL-96-17). CVP-99-00011 indicates that the final excavation depth at this site was 4.6 m (15 ft) bgs.

- 116-B-14 waste site (shallow decision unit) reports a cumulative total ELCR of 1.9×10^{-4}. The primary contributors to risk include europium-152 (1.2×10^{-4}; 63 percent contribution) and strontium-90 (6.8×10^{-5}; 35 percent contribution). The EPC of europium-152 is 4.4 pCi/g, which is greater than the residential RBSL of 3.7 pCi/g and is also greater than the current direct exposure RAG of 3.3 pCi/g published in the 100 Area RDR/RAWP (DOE/RL-96-17). The EPC of strontium-90 is 1.6 pCi/g, which is less than the residential RBSL of 2.3 pCi/g is also less than the current direct exposure RAG of 4.5 pCi/G published in the 100 Area RDR/RAWP (DOE/RL-96-17). CVP-99-00003 indicates that the final excavation depth at this site was approximately 6 m (19.7 ft) bgs.

Table 6-12. Summary of Total Risks for the 100-BC-1 and 100-BC-2 Shallow Zone Waste Sites for the Residential Scenario

Classification Status	Remediated Waste Site	Associated Waste Site[a]	Decision Unit with Exceedance	Total ELCR[b]	Risk Driver and % Contribution
100-BC-1					
Interim Closed	100-B-21:4	--	Shallow	1.1×10^{-3}	Cesium-137 (1.0×10^{-3} – 91%)
	116-B-5		Shallow Focused	4.0×10^{-4}	Europium -152 (2.0×10^{-4} – 50%) Tritium (1.1×10^{-4} – 27%)
	116-B-6A	116-B-16	Shallow	2.1×10^{-4}	Cesium-137 (9.5×10^{-5} – 45%) Strontium-90 (1.1×10^{-4} – 54%)
	116-B-14	--	Shallow	1.9×10^{-4}	Europium-152 (1.2×10^{-4} – 63%) Strontium-90 (6.8×10^{-5} – 35%)
	116-C-5		Shallow	1.0×10^{-4}	Cesium-137 (2.7×10^{-5} – 27%) Europium-152 (4.2×10^{-5} – 42%) Strontium-90 (1.7×10^{-5} – 17%)

Table 6-12. Summary of Total Risks for the 100-BC-1 and 100-BC-2 Shallow Zone Waste Sites for the Residential Scenario

Classification Status	Remediated Waste Site	Associated Waste Site[a]	Decision Unit with Exceedance	Total ELCR[b]	Risk Driver and % Contribution
	100-B-14:2				
	100-B-18				
	100-B-21:3				
	100-B-25				
	100-B-28				
	100-B-33				
	100-B-8:1				
	100-B-8:2				
	116-B-1				
	116-B-10		None	1×10^{-4} to 1×10^{-6}	None
	116-B-11				
	116-B-13				
	116-B-2				
	116-B-7, 132-B-6, 132-C-2				
	116-C-1				
	118-B-10				
	126-B-3				
	1607-B2:1				
	1607-B2:2				

Table 6-12. Summary of Total Risks for the 100-BC-1 and 100-BC-2 Shallow Zone Waste Sites for the Residential Scenario

Classification Status	Remediated Waste Site	Associated Waste Site[a]	Decision Unit with Exceedance	Total ELCR[b]	Risk Driver and % Contribution
	100-B-16 100-B-20 100-B-21:2 100-B-5 116-B-4 120-B-1 128-B-2 128-B-3 1607-B7	--	None	$< 1\times10^{-6}$	None
	100-B-14:1 100-B-19 100-B-22:2 100-B-32 116-B-12 116-B-3 116-B-6B 116-B-9 118-B-5 118-B-7	--	None	No COPCs reported above background	None
No Action	100-B-11 100-B-26 116-B-15	--	None	1×10^{-4} to 1×10^{-6}	None
	1607-B1	--	None	$< 1\times10^{-6}$	None

Table 6-12. Summary of Total Risks for the 100-BC-1 and 100-BC-2 Shallow Zone Waste Sites for the Residential Scenario

Classification Status	Remediated Waste Site	Associated Waste Site[a]	Decision Unit with Exceedance	Total ELCR[b]	Risk Driver and % Contribution
	100-B-14:5 100-B-14:6 100-B-14:7 118-B-9	--	None	No COPCs reported above background	None
100-BC-2					
Interim Closed	--	100-C-6:1	See 100-B-8:1 (100-BC-1)		
	--	100-C-6:2 100-C-6:3 100-C-6:4	See 100-B-8:2 (100-BC-1)		
	116-C-3	--	Shallow Focused	1.1×10^{-3}	Cesium-137 (3.2×10^{-4} – 29%) Strontium-90 (8.0×10^{-4} – 71%)
	118-B-1	--	Shallow	2.3×10^{-4}	Strontium-90 (2.0×10^{-4} – 84%) Cesium-137 (3.4×10^{-5} – 15%)
	116-C-2A	116-C-2B 116-C-2C	None	1×10^{-4} to 1×10^{-6}	None
	118-B-3	118-B-2			

Table 6-12. Summary of Total Risks for the 100-BC-1 and 100-BC-2 Shallow Zone Waste Sites for the Residential Scenario

Classification Status	Remediated Waste Site	Associated Waste Site[a]	Decision Unit with Exceedance	Total ELCR[b]	Risk Driver and % Contribution
	100-B-23 116-C-6 118-B-4 118-B-6 118-C-1 118-C-2 600-232 600-233	--			
	100-B-1 100-C-3 100-C-9:2 118-C-3:3 118-C-4 128-C-1 1607-B8 1607-B9 1607-B10 1607-B11	--	None	$< 1\times10^{-6}$	None
	100-B-31 100-C-9:1	--	None	No COPCs reported above background	None

Results summarized from Table G-26, Residential Scenario Risk Estimates and Noncancer Hazards for the 100-B/C Source OU Waste Site Decision Units Without Background Contribution.

a. Associated waste sites are those sites for which remediation and closeout documentation were consolidated with another remediated waste site.

b. Total ECLR represents risk contributions from both radiological and nonradiological COPCs.

- The 116-C-5 waste site (shallow decision unit) reports a cumulative total ELCR of 1.0×10^{-4}. The primary contributors to risk include cesium-137 (2.7×10^{-5}; 27 percent contribution), europium-152 (4.2×10^{-5}; 42 percent contribution), and strontium-90 (1.7×10^{-5}; 17 percent contribution). The EPC of cesium-137 is 1.2 pCi/g, which is less than the residential RBSL of 4.4 pCi/g and is also less than the current direct exposure RAG of 6.2 pCi/g published in the 100 Area RDR/RAWP (DOE/RL-96-17). The EPC of strontium-90 is 0.39 pCi/g, which is less than the residential RBSL of 2.3 pCi/g is also less than the current direct exposure RAG of 4.5 pCi/G published in the 100 Area RDR/RAWP (DOE/RL-96-17). The EPC for europium-152 is 1.5 pCi/g, which is less than the residential RBSL of 3.7 pCi/g and is also less than the current direct exposure RAG of 3.3 pCi/G published in the 100 Area RDR/RAWP (DOE/RL-96-17).

- The 116-C-3 waste site (shallow focused decision unit) reports a cumulative total ELCR of 1.1×10^{-3}. The primary contributors to risk include cesium-137 (3.2×10^{-4}; 29 percent contribution) and strontium-90 (8.0×10^{-4}; 71 percent contribution). The EPC of cesium-137 is 14 pCi/g, which is greater than the residential RBSL of 4.4 pCi/g and is also greater than the current direct exposure RAG of 6.2 pCi/g published in the 100 Area RDR/RAWP (DOE/RL-96-17). The EPC of strontium-90 is 18 pCi/g, which is greater than the residential RBSL of 2.3 pCi/g and is also greater than the current direct exposure RAG of 4.5 pCi/g published in the 100 Area RDR/RAWP (DOE/RL-96-17). WSRF-2008-002 indicates the maximum excavation depth at this site was approximately 8.5 m (28 ft) bgs.

- The 118-B-1 waste site (shallow 3 decision unit) reports a cumulative total ELCR of 2.3×10^{-4}. The primary contributors to risk include cesium-137 (3.4×10^{-5}; 15 percent contribution) and strontium-90 (2.0×10^{-4}; 84 percent contribution). The EPC of strontium-90 is 4.4 pCi/g, which is greater than the residential RBSL of 2.3 pCi/g. However, the EPC is less than the current direct exposure RAG of 4.5 pCi/g published in the 100 Area RDR/RAWP (DOE/RL-96-17). The EPC of cesium-137 is 1.5 pCi/g, which is less than the residential RBSL of 4.4 pCi/g and is also less than the current direct exposure RAG of 6.2 pCi/g published in the 100 Area RDR/RAWP (DOE/RL-96-17). CVP-2007-00006 indicates that the remediation footprint is partially in the deep zone (greater than 4.6 m [15 ft] bgs) with the deepest portion of the excavated area being approximately 10 m (33 ft) bgs.

WAC 173-340-740(6)(d) indicates that the point of compliance for human exposure via direct contact be established in the soils throughout the site from the ground surface to 15 feet bgs. Therefore, the direct contact exposure pathway is considered incomplete for samples collected at depths greater than 4.6 m (15 ft) bgs. Although samples collected from 100-B-21:4, 116-B-5, 116-B-14, 116-C-3, and 118-B-1 were assigned to the shallow zone for interim closure, closeout documentation indicates that verification samples are actually collected from depths greater than 4.6 m (15 ft) bgs. Comparison of EPCs to residential RBSLs overstates risk for direct contact exposure because sample collection depths are greater than 4.5 m (15 ft) bgs. The time required for radioisotope concentrations to decay to the residential RBSLs is provided in Section 6.2.6.4, and these sites are addressed with waste sites with radiological contamination present at depths greater than 4.6 m (15 ft) bgs.

The residential scenario results for shallow vadose zone material are summarized by decision unit in Table G-29 for the direct contact exposure pathway and Table G-30 for the inhalation pathway (Appendix G).

As presented in Table G-29 (Appendix G), the potential cumulative ELCR from direct contact for all nonradiological carcinogenic COPCs without background contribution ranges from 5.0×10^{-10} to 1.3×10^{-5} for 50 of the 109 shallow zone decision units. The potential cumulative ELCR is within the

regulatory target risk range of 10^{-4} to 10^{-6} for 12 decision units, is less than the lower regulatory risk threshold value of 1×10^{-6} for 38 decision units, and is greater than the MTCA HHRA Procedures (WAC 173-340-708(5)) cumulative risk threshold of 1×10^{-5} for four decision units. Nonradiological carcinogenic COPCs were not detected in 59 decision units.

Arsenic is a primary contributor to risk at four waste sites where the EPCs of arsenic range from 6.8 mg/kg to 8.0 mg/kg. Although the EPCs are greater than the lognormal 90th percentile of 6.5 mg/kg, they are consistent with naturally occurring levels at the Hanford Site (see Table G-16 [Appendix G]) and are considered naturally occurring and not the result of a site release. Additionally, all of the EPCs for arsenic are less than the direct exposure RAG of 20 mg/kg published in the 100 Area RDR/RAWP (DOE/RL-96-17).

In addition to arsenic, five waste sites report concentrations of site-related nonradiological carcinogenic COPCs in shallow zone material that result in cancer risk levels for individual carcinogens that exceed the MTCA "Unrestricted Land Use Soil Cleanup Standards" (WAC 173-340-740) acceptable cancer risk level of 1×10^{-6}. The cancer risk levels for the residential scenario are as follows:

- The 100-B-14:2 waste site (shallow 2 decision unit) reports benzo[a]pyrene (4.8×10^{-6}; 66 percent contribution) as an individual chemical risk driver.
- The 100-B-18 waste site (shallow focused decision unit) reports benzo[a]pyrene (2.2×10^{-6}; 71 percent contribution) as an individual chemical risk driver.
- 100-B-23 waste site (shallow focused decision unit) reports benzo[a]pyrene (1.6×10^{-6}; 61 percent contribution) as an individual chemical risk driver.
- The 126-B-3 waste site (shallow decision unit) reports benzo[a]pyrene (2.0×10^{-6}; 71 percent contribution) as an individual chemical risk driver.
- The 600-233 waste site (shallow focused decision unit) reports benzo[a]pyrene (1.3×10^{-6}; 67 percent contribution) as an individual risk driver.

The individual risk estimates for the PAHs are greater than the acceptable risk value of 1×10^{-6} for individual carcinogens but are less than the MTCA HHRA Procedures (WAC 173-340-708(5)) cumulative risk threshold of 1×10^{-5} for multiple contaminants and multiple pathways for these five waste sites.

As presented in Table G-29 (Appendix G), the potential HI from direct contact for noncancer effects without background contribution ranges from less than 1 to 1.7 for the shallow zone. The HI is greater 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 three waste sites. No individual COPCs were reported with an HQ greater than 1.

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 7.5×10^{-16} to 3.3×10^{-7} for 81 of the 109 shallow zone decision units. The potential cumulative ELCR is less than the MTCA "Cleanup Standards to Protect Air Quality" (WAC 173-340-750) risk threshold value of 1×10^{-6} for the shallow zone. Nonradiological carcinogenic COPCs were not reported at 28 decision units.

As presented in Table G-30 (Appendix G), the potential HI for the inhalation pathway from noncancer effects without background contribution is less than 1 for the shallow zone. 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) target HI of 1.

As shown in Table G-28 (Appendix G), all arsenic and lead EPCs are less than their respective Method A soil cleanup levels of 20 mg/kg and 250 mg/kg, respectively, for unrestricted land use.

Overburden. A total of 23 waste sites are reported with CVP/RSVP data associated with overburden material in the 100-BC Source OU. The following sample designs were applied to the 23 waste sites evaluated:

- Thirteen waste sites were sampled using a focused sampling design.
- Ten waste sites were sampled using a statistical sampling design (one waste site with two statistically distinct decision units, one waste site with three statistically distinct decision units, and one waste site with seven statistically distinct decision units).
- Six decision units from the 100-B-14 waste site reported all concentrations below detection limits, resulting in 26 overburden decision units for evaluation.

The residential scenario results for overburden materials are summarized by decision unit in Table G-31 and Table G-32 (Appendix G).

As presented in Table G-31 (Appendix G), potential cumulative ELCR from all radiological COPCs without background contribution ranges from 4.8×10^{-8} to 1.3×10^{-3} for 15 of the 26 overburden decision units. The potential cumulative ELCR is greater than the upper regulatory risk threshold value of 1×10^{-4} for three decision units; is within the regulatory target risk range of 10^{-4} to 10^{-6} for ten decision units; and is less than the lower regulatory risk threshold value of 1×10^{-6} for two decision units. Radiological COPCs were not reported at 11 decision units.

Three waste sites report concentrations of site-related radiological COPCs in overburden material that exceed the upper threshold of the regulatory target risk range. The cancer risk levels for the residential scenario are as follows:

- The 100-B-21:4 waste site (overburden focused decision unit) reports a cumulative ELCR of 2.7×10^{-4}. The primary risk driver is cesium-137 (1.8×10^{-4}; 69 percent contribution). The EPC of cesium-137 is 8.0 pCi/g, which is greater than the residential RBSL of 4.4 pCi/g and is also greater than the current direct exposure RAG of 6.2 pCi/g published in the 100 Area RDR/RAWP (DOE/RL-96-17). The WSRF 2009-041 reports that the portion of the overburden that exceeded the direct exposure criteria was backfilled to the deep zone of the excavation.
- The 100-B-8:2 waste site (overburden decision unit) reports a cumulative ELCR of 1.3×10^{-3}. The primary risk driver is strontium-90 (1.3×10^{-3}; 99 percent contribution). The EPC of strontium-90 is 29 pCi/g, which is greater than the residential RBSL of 2.3 pCi/g and is also greater than the current direct exposure RAG of 4.5 pCi/g published in the 100 Area RDR/RAWP (DOE/RL-96-17).
- The 116-C-2A waste site (overburden decision unit) reports a cumulative ELCR of 2.8×10^{-4}. The primary risk driver is cesium-137 (2.8×10^{-4}; 99 percent contribution). The EPC of cesium-137 is 12 pCi/g, which is greater than the residential RBSL of 4.4 pCi/g and is also greater than the current direct exposure RAG of 6.2 pCi/g published in the 100 Area RDR/RAWP (DOE/RL-96-17).

- As presented in Table G-31 (Appendix G), the potential cumulative ELCR for direct contact from all nonradiological carcinogenic COPCs without background contribution ranges from 2.7×10^{-10} to 7.4×10^{-6} for 11 of the 26 overburden decision units. The potential cumulative ELCR is within the regulatory target risk range of 10^{-4} to 10^{-6} for two decision units; is less than the lower regulatory risk threshold value of 1×10^{-6} for nine decision units, and is less than the MTCA HHRAProcedures (WAC 173-340-708(5)) cumulative risk threshold of 1×10^{-5} for the 11 decision units. Nonradiological carcinogenic COPCs were not reported in 15 decision units.

Two waste sites report concentrations of site-related nonradiological carcinogenic COPCs in overburden material that result in cancer risk levels for individual carcinogens that exceed the MTCA "Unrestricted Land Use Soil Cleanup Standards" (WAC 173-340-740) acceptable cancer risk level of 1×10^{-6}. The cancer risk levels for the residential scenario are as follows:

- The 100-B-14:2 waste site (overburden focused decision unit) reports benzo(a)pyrene (1.2×10^{-6}; 71 percent contribution) as an individual risk driver.
- The 100-C-9:1 waste site (overburden focused decision unit) reports benzo(a)pyrene (5.0×10^{-6}; 68 percent contribution) as an individual risk driver.

The individual risk estimates for the PAHs are greater than the acceptable risk value of 1×10^{-6} for individual carcinogens but are less than the MTCA HHRA Procedures (WAC 173-340-708(5)) cumulative risk threshold of 1×10^{-5} for multiple contaminants and multiple pathways for the two decision units.

As presented in Table G-31 (Appendix G), the potential HI for direct contact from noncancer effects without background contribution is less than 1 for all overburden decision units. The HI is less than EPA target HI of 1 and the MTCA "Unrestricted Land Use Soil Cleanup Standards" (WAC 173-340-740) target HI of 1.

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 2.5×10^{-16} to 7.2×10^{-8} for 23 of the 26 overburden decision units. 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 all overburden decision units. Nonradiological carcinogenic COPCs were not reported at three decision units.

As presented in Table G-32 (Appendix G), the potential HI for the inhalation pathway from noncancer effects without background contribution is less than 1 for all overburden decision units. 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) target HI of 1.

As shown in Table G-28 (Appendix G), all arsenic and lead EPCs are less than their Method A soil cleanup levels of 20 mg/kg and 250 mg/kg, respectively, for unrestricted land use.

Staging Pile Area. A total of 19 waste sites are reported with CVP/RSVP data associated with staging pile area material in the 100-BC Source OU. The following sample designs were applied to the 19 waste sites evaluated:

- Four waste sites were sampled using a focused sampling design.
- Eleven wastes sites were sampled using a statistical sampling design (one waste site with two statistically distinct decision units).

- Four waste sites were sampled using both a statistical and a focused sampling design (one waste with two statistically distinct decision units).

The residential scenario results for staging pile footprint materials are summarized by decision unit in Table G-33 and Table G-34 (Appendix G).

As presented in Table G-33 (Appendix G), the potential cumulative ELCR from all radiological COPCs without background contribution ranges from 1.0×10^{-6} to 2.8×10^{-4} for 10 of the 25 staging pile area decision units. The potential cumulative ELCR is greater than the upper regulatory risk threshold value of 1×10^{-4} for one decision unit and is within the regulatory target risk range of 10^{-4} to 10^{-6} for nine decision units. Radiological COPCs were not reported at 15 decision units.

One waste site reports a concentration of a site-related radiological COPC in staging pile area material that exceeds the upper threshold of the regulatory target risk range. The cancer risk level for the residential scenario is as follows:

- The 118-C-1 waste site (staging pile area decision unit) reports a cumulative ELCR of 2.8×10^{-4}. The primary contributor to risk is carbon-14 (2.3×10^{-4}; 82 percent contribution). The EPC for carbon-14 is 182 pCi/g, which is greater than the residential RBSL of 81 pCi/g and is also greater than the current direct exposure RAG of 8.7 pCi/g published in the 100 Area RDR/RAWP (DOE/RL-96-17). CVP-2006-00011 notes that material at this sample location was backfilled into deep zone portions of the waste site.

As presented in Table G-33 (Appendix G), the potential cumulative ELCR for direct contact from all nonradiological carcinogenic COPCs without background contribution ranges from 4.6×10^{-10} to 2.8×10^{-6} for seven of the 25 staging pile area decision units. The potential cumulative ELCR is within the regulatory target risk range of 10^{-4} to 10^{-6} for one decision unit; is less than or equal to the lower regulatory risk threshold value of 1×10^{-6} for six decision units; and is less than the MTCA HHRA Procedures (WAC 173-340-708(5)) cumulative risk threshold of 1×10^{-5} for the seven decision units. Nonradiological carcinogenic COPCs were not reported in 18 decision units.

One waste site reports a concentration of a site-related nonradiological carcinogenic COPC in staging pile area material that results in a cancer risk level for individual carcinogens that exceeds the "Unrestricted Land Use Soil Cleanup Standards" (WAC 173-340-740) acceptable cancer risk level of 1×10^{-6}. The cancer risk level for the residential scenario is as follows:

- The 118-C-1 waste site (staging pile area decision unit) reports benzo[a]pyrene (1.4×10^{-6}; 50 percent contribution) as an individual risk driver.

The individual risk estimates for this PAH is greater than the acceptable risk value of 1×10^{-6} for individual carcinogens but is less than the MTCA HHRA Procedures (WAC 173-340-708(5)) cumulative risk threshold of 1×10^{-5} for multiple contaminants and multiple pathways for the decision unit.

As presented in Table G-33 (Appendix G), the potential HI for direct contact from noncancer effects without background contribution is less than 1 for all staging pile area decision units. The potential HI is less than EPA target HI of 1 and the MTCA "Unrestricted Land Use Soil Cleanup Standards" (WAC 173-340-740) target HI of 1.

As presented in Table G-34 (Appendix G), the potential cumulative ELCR for the inhalation pathway from all nonradiological carcinogenic COPCs without background contribution ranges from 4.3×10^{-16} to 8.0×10^{-8} for 16 of the 25 staging pile area decision units. 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 all staging pile area decision units. Nonradiological carcinogenic COPCs were not reported at nine decision units.

As presented in Table G-34 (Appendix G), the potential HI for the inhalation pathway from noncancer effects without background contribution is less than 1 for all staging pile area decision units. 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) target HI of 1.

As shown in Table G-28 (Appendix G), all arsenic and lead EPCs are less than their respective Method A soil cleanup levels of 20 mg/kg and 250 mg/kg, respectively, for unrestricted land use.

Deep Zone. Deep zone soil samples are compared to RBSLs developed for the residential exposure scenario, although residents are unlikely to be exposed to contaminants in deep vadose zone. Deep vadose zone samples are collected from depths greater than 4.6 m (15 ft) bgs; as a result, direct contact within the point of compliance is incomplete. Additionally, the residential exposure scenario does not reflect reasonably anticipated future land use in the River Corridor. This comparison is included because it is the most conservative land use basis for the evaluation of waste sites and these results are only included to provide additional information for risk management decisions.

A total of 23 waste sites were reported with CVP/RSVP from deep zone decision units in the 100-BC Source OU. The remaining 52 waste sites were not excavated deeper than 4.6 m (15 ft) bgs and are not discussed in this section. The following sample designs were applied to the 23 waste sites evaluated:

- Five waste sites were sampled using a focused sampling design.
- Sixteen waste sites were sampled using a statistical sampling design.
- Two waste sites were sampled using both a statistical and a focused sampling design.

The overall risk summary for the residential scenario from all deep zone waste sites evaluated is provided in Table 6-13. Table 6-13 lists the OU that each waste site resides in, the reclassification status, the remediated waste site, the consolidated 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).

As presented in Table 6-13, the total ELCR is greater than the upper risk threshold of 1×10^{-4} at 18 remediated waste sites. Risks were not reported at three remediated waste sites because COPCs were not reported above background concentrations. Risks were also not reported at two remediated waste sites because COPCs were not detected. The cancer risk levels for the residential scenario are as follows:

- The 116-C-1 waste site (deep focused decision unit) reports a cumulative total ELCR of 1.7×10^{-1}. The primary contributors to risk include cesium-137 (1.3×10^{-1}; 76 percent contribution), cobalt-60 (3.7×10^{-3}; 2 percent contribution), europium-152 (3.1×10^{-2}; 18 percent contribution), europium-154 (3.3×10^{-3}; 2 percent contribution), nickel-63 (2.6×10^{-4}; 0.2 percent contribution) and strontium-90 (3.9×10^{-3}; 2 percent contribution).

- The 116-B-11 waste site (deep decision unit) reports a cumulative total ELCR of 1.5×10^{-2}. The primary contributors to risk include cesium-137 (2.8×10^{-3}; 19 percent contribution), cobalt-60 (1.3×10^{-3}; 9 percent contribution), europium-152 (8.9×10^{-3}; 61 percent contribution), europium-154 (9.6×10^{-4}; 7 percent contribution), nickel-63 (4.4×10^{-4}; 3 percent contribution), and strontium-90 (2.1×10^{-4}; 1 percent contribution).

Table 6-13. Summary of Total Risks for the 100-BC-1 and 100-BC-2 Deep Zone Waste Sites for the Residential Scenario

Classification Status	Remediated Waste Site	Associated Waste Site[a]	Decision Unit with Exceedance	Total ELCR[b]	Risk Driver and % Contribution
100-BC-1					
Interim Closed	116-C-1	--	Deep Focused	1.7×10^{-1}	Cesium-137 (1.3×10^{-1} – 76%) Cobalt-60 (3.7×10^{-3} – 2%) Europium-152 (3.1×10^{-2} – 18%) Europium-154 (3.3×10^{-3} – 2%) Nickel-63 (2.6×10^{-4} – 0.2%) Strontium-90 (3.9×10^{-3} – 2%)
	116-B-11		Deep	1.5×10^{-2}	Cesium-137 (2.8×10^{-3} – 19%) Cobalt-60 (1.3×10^{-3} – 9%) Europium-152 (8.9×10^{-3} – 61%) Europium-154 (9.6×10^{-4} – 7%) Nickel-63 (4.4×10^{-4} – 3%) Strontium-90 (2.1×10^{-4} – 1%)
	116-B-4		Deep	9.5×10^{-3}	Cesium-137 (2.5×10^{-3} – 27%) Cobalt-60 (5.9×10^{-4} – 6%) Europium-152 (5.3×10^{-3} – 56%) Europium-154 (1.0×10^{-3} – 11%)
	116-C-5		Deep	4.4×10^{-3}	Cesium-137 (1.2×10^{-3} – 27%) Cobalt-60 (5.9×10^{-4} – 13%) Europium-152 (2.0×10^{-3} – 45%) Europium-154 (3.6×10^{-4} – 8%) Nickel-63 (1.1×10^{-4} – 2%) Strontium-90 (1.9×10^{-4} – 4%)
	116-B-2		Deep	1.4×10^{-3}	Cesium-137 (1.1×10^{-3} – 76%) Strontium-90 (3.2×10^{-4} – 22%)

Table 6-13. Summary of Total Risks for the 100-BC-1 and 100-BC-2 Deep Zone Waste Sites for the Residential Scenario

Classification Status	Remediated Waste Site	Associated Waste Site[a]	Decision Unit with Exceedance	Total ELCR[b]	Risk Driver and % Contribution
	100-B-5		Deep	1.1×10^{-3}	Cesium-137 (5.1×10^{-4} – 47%) Europium-152 (4.2×10^{-4} – 38%) Strontium-90 (8.5×10^{-5} – 8%)
	116-B-6A	116-B-16	Deep	9.8×10^{-4}	Strontium-90 (9.3×10^{-4} – 95%)
	116-B-3		Deep	5.9×10^{-4}	Cesium-137 (4.5×10^{-4} – 76%) Strontium-90 (1.4×10^{-4} – 24%)
	100-B-8:2		Deep	5.9×10^{-4}	Cesium-137 (2.9×10^{-4} – 49%) Europium-152 (1.4×10^{-4} – 24%) Strontium-90 (1.2×10^{-4} – 20%)
	100-B-8:1		Deep	4.1×10^{-4}	Cesium -137 (2.2×10^{-4} – 54%) Europium -152 (7.5×10^{-5} – 18%) Strontium-90 (1.0×10^{-4} – 25%)
	100-B-14:1		Deep Focused	3.4×10^{-4}	Carbon-14 (3.4×10^{-4} – 100%)
	116-B-1	--	Deep	3.4×10^{-4}	Cesium-137 (7.1×10^{-5} – 21%) Europium-152 (1.9×10^{-4} – 57%) Strontium-90 (6.3×10^{-5} – 18%)
	116-B-5		Deep Focused	3.2×10^{-4}	Europium-152 (2.7×10^{-4} – 84%)
	116-B-14		Deep	2.4×10^{-4}	Cesium-137 (1.3×10^{-4} – 56%) Europium-152 (3.7×10^{-5} – 15%) Strontium-90 (6.7×10^{-5} – 28%)
	116-B-7, 132-B-6, 132-C-2		Deep	1.4×10^{-4}	Cesium-137 (8.0×10^{-5} – 56%) Europium-152 (3.8×10^{-5} – 27%) Strontium-90 (1.8×10^{-5} – 13%)
	100-B-27 116-B-12		None	No COPCs reported above background	None

Table 6-13. Summary of Total Risks for the 100-BC-1 and 100-BC-2 Deep Zone Waste Sites for the Residential Scenario

Classification Status	Remediated Waste Site	Associated Waste Site[a]	Decision Unit with Exceedance	Total ELCR[b]	Risk Driver and % Contribution
No Action	100-B-14:3	--	No COPCs detected		
100-BC-2					
Interim Closed	--	100-C-6:1	See 100-B-8:1 (100-BC-1)		
	--	100-C-6:2 100-C-6:3 100-C-6:4	See 100-B-8:2 (100-BC-1)		
	118-C-3:2	--	Deep Focused	6.0×10^{-3}	Cesium-137 (3.9×10^{-3} – 65%) Europium-152 (2.5×10^{-4} – 4%) Strontium-90 (1.7×10^{-3} – 27%)
	116-C-2A	116-C-2B 116-C-2C	Deep	2.2×10^{-3}	Cesium-137 (5.2×10^{-4} – 24%) Cobalt-60 (3.9×10^{-4} – 18%) Europium-152 (8.1×10^{-4} – 37%) Strontium-90 (2.7×10^{-4} – 13%)
	118-B-6	--	Deep	4.5×10^{-4}	Tritium (4.5×10^{-4} – 100%)
	100-C-9:1		None	No COPCs reported above background	None
No Action	100-C-9:3	--	None	No COPCs detected	None

Results summarized from Table G-26, Residential Scenario Risk Estimates and Noncancer Hazards for the 100-B/C Source OU Waste Site Decision Units Without Background Contribution.

a. Associated waste sites are those sites for which remediation and closeout documentation were consolidated with another remediated waste site.

b. Total ECLR represents risk contributions from radiological COPCs.

- The 116-B-4 waste site (deep decision unit) reports a cumulative total ELCR of 9.5×10^{-3}. The primary contributors to risk include cesium-137 (2.5×10^{-3}; 27 percent contribution), cobalt-60 (5.9×10^{-4}; 6 percent contribution), europium-152 (5.3×10^{-3}; 56 percent contribution), and europium-154 (1.0×10^{-3}; 11 percent contribution).

- The 116-C-5 waste site (deep decision unit) reports a cumulative total ELCR of 4.4×10^{-3}. The primary contributors to risk include cesium-137 (1.2×10^{-3}; 27 percent contribution), cobalt-60 (5.9×10^{-4}; 13 percent contribution), europium-152 (2.0×10^{-3}; 45 percent contribution), europium-154 (3.6×10^{-4}; 8 percent contribution), nickel-63 (1.1×10^{-4}; 2 percent contribution), and strontium-90 (1.9×10^{-4}; 4 percent contribution).

- The 116-B-2 waste site (deep decision unit) reports a cumulative total ELCR of 1.4×10^{-3}. The primary contributors to risk include cesium-137 (1.1×10^{-3}; 76 percent contribution) and strontium-90 (3.2×10^{-4}; 22 percent contribution).

- The 100-B-5 waste site (deep decision unit) reports a cumulative total ELCR of 1.1×10^{-3}. The primary contributors to risk include cesium-137 (5.1×10^{-4}; 47 percent contribution) and europium-152 (4.2×10^{-4}; 38 percent contribution), and strontium-90 (8.5×10^{-5}; 8 percent contribution).

- The 116-B-6A waste site (deep decision unit) reports a cumulative total ELCR of 9.8×10^{-4}. The primary contributor to risk is strontium-90 (9.3×10^{-4}; 95 percent contribution)

- The 116-B-3 waste site (deep decision unit) reports a cumulative total ELCR of 5.9×10^{-4}. The primary contributors to risk include cesium-137 (4.5×10^{-4}; 76 percent contribution) and strontium-90 (1.4×10^{-4}; 24 percent contribution).

- The 100-B-8:2 waste site (deep decision unit) reports a cumulative total ELCR of 5.9×10^{-4}. The primary contributors to risk include cesium-137 (2.9×10^{-4}; 49 percent contribution), europium-152 (1.4×10^{-4}; 24 percent contribution), and strontium-90 (1.2×10^{-4}; 20 percent contribution).

- The 100-B-8:1 waste site (deep decision unit) reports a cumulative total ELCR of 4.1×10^{-4}. The primary contributors to risk include cesium-137 (2.2×10^{-4}; 54 percent contribution), europium-152 (7.5×10^{-5}; 18 percent contribution), and strontium-90 (1.0×10^{-4}; 25 percent contribution).

- The 100-B-14:1 waste site (deep focused decision unit) reports a cumulative total ELCR of 3.4×10^{-4}. The primary contributor to risk is carbon-14 (3.4×10^{-4}; 100 percent contribution).

- The 116-B-1 waste site (deep decision unit) reports a cumulative total ELCR of 3.4×10^{-4}. The primary contributors to risk include cesium-137 (7.1×10^{-5}; 21 percent contribution), europium-152 (1.9×10^{-4}; 57 percent contribution), and strontium-90 (6.3×10^{-5}; 18 percent contribution).

- The 116-B-5 waste site (deep focused decision unit) reports a cumulative total ELCR of 3.2×10^{-4}. The primary contributor to risk is europium-152 (2.7×10^{-4}; 84 percent contribution).

- The 116-B-14 waste site (deep decision unit) reports a cumulative total ELCR of 2.4×10^{-4}. The primary contributors to risk include cesium-137 (1.3×10^{-4}; 56 percent contribution), europium-152 (3.7×10^{-5}; 15 percent contribution), and strontium-90 (6.7×10^{-5}; 28 percent contribution).

- The 116-B-7, 132-B-6, 132-C-2 waste site (deep decision unit) reports a cumulative total ELCR of 1.4×10^{-4}. The primary contributors to risk include cesium-137 (8.0×10^{-5}; 56 percent contribution), europium-152 (3.8×10^{-5}; 27 percent contribution), and strontium-90 (1.8×10^{-5}; 13 percent contribution).

- The 118-C-3:2 waste site (deep focused decision unit) reports a cumulative total ELCR of 6.0×10^{-3}. The primary contributors to risk include cesium-137 (3.9×10^{-3}; 65 percent contribution), europium-152 (2.5×10^{-4}; 4 percent contribution), and strontium-90 (1.7×10^{-3}; 27 percent contribution).

- T he 116-C-2A waste site (deep decision unit) reports a cumulative total ELCR of 2.2×10^{-3}. The primary contributors to risk include cesium-137 (5.2×10^{-4}; 24 percent contribution), cobalt-60 (3.9×10^{-4}; 18 percent contribution), europium-152 (8.1×10^{-4}; 37 percent contribution), and strontium-90 (2.7×10^{-4}; 13 percent contribution).

- The 118-B-6 waste site (deep decision unit) reports a cumulative total ELCR of 4.5×10^{-4}. The primary contributor to risk is tritium (4.5×10^{-4}; 100 percent contribution).

6.2.5.3.2 Resident Monument Worker Scenario

PRGs developed for the resident Monument worker scenario represents 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, risk estimates for each waste site decision unit are provided in Appendix G. All radiological COPCs, regardless of their EPCs relative to the background value, are presented in Table G-36 through Table G-39 (Appendix G).

Appendix G also includes risk estimates for each waste site decision unit, which includes only those radiological COPCs with EPCs greater than background values or that do not have a background value. he risk estimates are presented in Table G-40 through Table G-43 (Appendix G). Only these results are discussed in the risk characterization because this information is used for decisions concerning appropriate remedial actions.

Risk estimates were calculated for the shallow, overburden, and staging pile area decision units within a waste site. Risk estimates were not calculated for the deep zone decision units because the direct contact exposure pathway is incomplete (i.e., samples are collected from depths greater than 4.6 m [15 ft] bgs). The results without background contribution for the resident Monument worker scenario are presented in Table G-40 (Appendix G). An overall summary of the cumulative risk estimates for each of the waste sites evaluated is provided in Table 6-14 for the shallow zone.

Shallow Zone. A total of 75 waste sites are reported with CVP/RSVP data associated with the shallow zone in the 100-BC Source OU. The following sample designs were applied to the 75 waste sites evaluated:

- Twenty-one waste sites were sampled using a focused sampling design.

- Forty-one waste sites were sampled using a statistical sampling design (one site with two statistically distinct decision units and one site with three statistically distinct decision units).

- Thirteen waste sites were sampled using both a statistical and a focused sampling design (two waste sites each with two statistically distinct decision units, two waste sites each with three statistically

distinct decision units, two waste sites each with four statistically distinct decision units, and one waste site with seven statistically distinct decision units).

The overall total risk summary for the resident Monument worker scenario from all shallow zone waste sites evaluated is provided in Table 6-14. Table 6-14 lists the OU that each waste site resides in, the reclassification status, the remediated waste site, the consolidated 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). The resident Monument worker scenario results for shallow vadose zone materials are also summarized by decision unit in Table G-41 (Appendix G).

As presented in Table 6-14, the potential cumulative total ELCR is greater than the upper risk threshold value of 1×10^{-4} at four waste sites, is within the regulatory target risk range of 10^{-4} to 10^{-6} at 21 remediated waste sites and is less than the lower risk threshold value of 1×10^{-6} at nine remediated waste sites. Risks were not reported at 20 remediated waste sites because COPCs were not reported above background concentrations. Risks were also not reported at 21 waste sites because COPCs were not detected.

Four waste sites report concentrations of site-related COPCs that are equal to or exceed the upper threshold of the regulatory target risk range. The cancer risk levels for the resident Monument worker scenario are as follows:

- The 100-B-21:4 waste site (shallow decision unit) reports a cumulative total ELCR of 7.6×10^{-4}. The primary contributor to risk is cesium-137 (7.3×10^{-4}; 95 percent contribution). Although the samples from this decision unit were assigned to the shallow zone, WSRF-2009-041 indicate theses samples were collected from a depth of 7.5 m (25 ft) bgs.
- The 116-B-5 waste site (shallow focused decision unit) reports a cumulative total ELCR of 2.7×10^{-4}. The primary contributors to risk include cesium-137 (2.9×10^{-5}; 11 percent contribution), europium-152 (1.9×10^{-4}; 72 percent contribution), and cobalt-60 (3.1×10^{-5}; 12 percent contribution). WSRF-98-064 indicates that the final excavation depth at this site was 5 m (16 ft) bgs.
- The 116-B-14 waste site (shallow decision unit) reports a cumulative total ELCR of 1.2×10^{-4}. The primary contributor to risk is europium-152 (1.2×10^{-4}; 98 percent contribution). CVP-99-0003 indicates that the final excavation depth at this site was approximately 6 m (19.7 ft) bgs.
- 116-C-3 waste site (shallow focused decision unit) reports a cumulative total ELCR of 2.3×10^{-4}. The primary contributor to risk is cesium-137 (2.3×10^{-4}; 98 percent contribution).

Table 6-14. Summary of Total Risks for the 100-BC-1 and 100-BC-2 Shallow Zone Waste Sites for the Resident Monument Worker Scenario

Classification Status	Remediated Waste Site	Associated Waste Site[a]	Decision Unit with Exceedance	Total ELCR[b]	Risk Driver and % Contribution
100-BC-1					
Interim Closed	100-B-21:4	--	Shallow	7.6×10^{-4}	Cesium-137 (7.3×10^{-4} – 95%)
	116-B-5		Shallow Focused	2.7×10^{-4}	Cesium-137 (2.9×10^{-5} – 11%) Europium-152 (1.9×10^{-4} – 72%) Cobalt-60 (3.1×10^{-5} – 12%)
	116-B-14		Shallow	1.2×10^{-4}	Europium-152 (1.2×10^{-4} – 98%)
	116-B-6A	116-B-16	None	1×10^{-4} to 1×10^{-6}	None
	100-B-21:3 100-B-25 100-B-33 100-B-8:1 100-B-8:2 116-B-1 116-B-10 116-B-11 116-B-2 116-B-7, 132-B-6, 132-C-2 116-C-1 116-C-5 118-B-10	--			
	100-B-14:2 100-B-5 116-B-13 116-B-4 1607-B2:2		None	$< 1\times10^{-6}$	None

Table 6-14. Summary of Total Risks for the 100-BC-1 and 100-BC-2 Shallow Zone Waste Sites for the Resident Monument Worker Scenario

Classification Status	Remediated Waste Site	Associated Waste Site[a]	Decision Unit with Exceedance	Total ELCR[b]	Risk Driver and % Contribution
	100-B-14:1 100-B-21:2 100-B-32 116-B-12 116-B-3 116-B-6B 116-B-9 118-B-5 128-B-2		None	No COPCs reported above background	None
	100-B-16 100-B-18 100-B-19 100-B-20 100-B-22:2 100-B-28 118-B-7 120-B-1 126-B-3 128-B-3 1607-B2:1 1607-B7		None	No COPCs detected	None
No Action	100-B-26	--	None	1×10^{-4} to 1×10^{-6}	None
	100-B-11 116-B-15			$< 1\times10^{-6}$	

Table 6-14. Summary of Total Risks for the 100-BC-1 and 100-BC-2 Shallow Zone Waste Sites for the Resident Monument Worker Scenario

Classification Status	Remediated Waste Site	Associated Waste Site[a]	Decision Unit with Exceedance	Total ELCR[b]	Risk Driver and % Contribution
	100-B-14:5 100-B-14:6 100-B-14:7 1607-B1			No COPCs reported above background	
	118-B-9			No COPCs detected	
100-BC-2					
Interim Closed	--	100-C-6:1	See 100-B-8:1 (100-BC-1)		
	--	100-C-6:2 100-C-6:3 100-C-6:4	See 100-B-8:2 (100-BC-1)		
	116-C-3	--	Shallow Focused	2.3×10^{-4}	Cesium-137 (2.3×10^{-4} – 98%)
	116-C-2A	116-C-2B 116-C-2C	None	1×10^{-4} to 1×10^{-6}	None
	118-B-3	118-B-2			
	116-C-6 118-B-1 118-B-4 118-C-1	--			
	118-B-6 118-C-2	--	None	$< 1\times10^{-6}$	None

Table 6-14. Summary of Total Risks for the 100-BC-1 and 100-BC-2 Shallow Zone Waste Sites for the Resident Monument Worker Scenario

Classification Status	Remediated Waste Site	Associated Waste Site[a]	Decision Unit with Exceedance	Total ELCR[b]	Risk Driver and % Contribution
	100-B-1 100-C-3 100-C-9:2 118-C-3:3 118-C-4 1607-B10 1607-B8	--	None	No COPCs reported above background	None
	100-B-23 100-B-31 100-C-9:1 128-C-1 1607-B11 1607-B9 600-232 600-233	--	None	No COPCs detected	None

Results summarized from Table G-40, Resident Monument Worker Scenario Radiological Risk Estimates for 100-B/C Source OU Waste Site Decision Units Without Background Contribution.

a. Associated waste sites are those sites for which remediation and closeout documentation were consolidated with another remediated waste site.

b. Total ECLR represents risk contributions from radiological COPCs.

Overburden. A total of 23 waste sites are reported with CVP/RSVP data associated with overburden material in the 100-BC Source OU. The following sample designs were applied to the 23 waste sites evaluated:

- Thirteen waste sites were sampled using a focused sampling design.
- Ten waste sites were sampled using a statistical sampling design (one waste site with two statistically distinct decision units, one waste site with three statistically distinct decision units, and one waste site with seven statistically distinct decision units).
- Six decision units from 100-B-14 reported all concentrations below detection limits, resulting in 26 overburden decision units for evaluation.

The resident Monument worker scenario results for overburden materials are summarized by decision unit in Table G-42 (Appendix G). As presented in Table G-42 (Appendix G), potential cumulative ELCR from all radiological COPCs without background contribution ranges from 2.3×10^{-11} to 2.0×10^{-4} for 15 of the 26 overburden decision units. The potential cumulative ELCR is greater than the upper regulatory risk threshold value of 1×10^{-4} for two decision units; is within the regulatory target risk range of 10^{-4} to 10^{-6} for seven decision units; and is less than the lower regulatory risk threshold value of 1×10^{-6} for six decision units. Radiological COPCs were not reported at 11 decision units.

Two waste sites report concentrations of site-related radiological COPCs in overburden material that exceed the upper threshold of the regulatory target risk range. The cancer risk levels for the resident Monument worker scenario are as follows:

- The 100-B-21:4 waste site (overburden focused decision unit) reports a cumulative ELCR of 1.4×10^{-4}. The primary contributor to risk is cesium-137 (1.3×10^{-4}; 92 percent contribution). The WSRF 2009-041 reports that the portion of the overburden that exceeded the direct exposure criteria was backfilled to the deep zone of the excavation.
- The 116-C-2A waste site (overburden decision unit) reports a cumulative ELCR of 2.0×10^{-4}. The primary contributor to risk is cesium-137 (2.0×10^{-4}; 99 percent contribution).

Staging Pile Area. A total of 19 waste sites are reported with CVP/RSVP data associated with staging pile area material in the 100-BC Source OU. The following sample designs were applied to the 19 waste sites evaluated:

- Four waste sites were sampled using a focused sampling design.
- Eleven waste sites were sampled using a statistical sampling design (one waste site with two statistically distinct decision units).
- Four waste sites were sampled using both a statistical and a focused sampling design (one waste with two statistically distinct decision units).

The resident Monument worker scenario results for staging pile footprint materials are summarized by decision unit in Table G-43 (Appendix G). As presented in Table G-43 (Appendix G), the potential cumulative ELCR from all radiological COPCs without background contribution ranges from 3.3×10^{-9} to 4.4×10^{-5} for 10 of the 25 staging pile area decision units. The potential cumulative ELCR is within the regulatory target risk range of 10^{-4} to 10^{-6} for six decision units and is less than the lower regulatory risk threshold value of 1×10^{-6} for four decision units. Radiological COPCs were not reported at 15 decision units.

6.2.5.3.3 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 waste site decision unit are provided in Appendix G. The risk estimates, which include all COPCs regardless of their EPCs relative to the background value, are presented in Table G-44 through Table G-47 (Appendix G).

Appendix G also provides risk estimates for each waste site decision unit, which include only those COPCs with EPCs greater than background values or that do not have a background value. The risk estimates are presented in Table G-48 through Table G-51 (Appendix G). Only these results are discussed in the risk characterization because this information is used for decisions concerning appropriate remedial actions.

Risk estimates were calculated for the shallow, overburden, and staging pile area decision units within a waste site. Risk estimates were not calculated for the deep zone decision units because the direct contact exposure pathway is incomplete (i.e., samples are collected from depths greater than 4.6 m [15 ft] bgs). The results without background contribution for the casual recreational user scenario are presented in Table G-48 (Appendix G). An overall summary of the cumulative risk estimates for each of the waste sites evaluated for the shallow zone is provided in Table 6-15.

Shallow Zone. A total of 75 waste sites reported with CVP/RSVP data associated with the shallow zone in the 100-BC Source OU. The following sample designs were applied to the 75 waste sites evaluated:

- Twenty-one waste sites were sampled using a focused sampling design.
- Forty-one waste sites were sampled using a statistical sampling design (one site with two statistically distinct decision units and one site with three statistically distinct decision units).
- Thirteen waste sites were sampled using both a statistical and a focused sampling design (two waste sites each with two statistically distinct decision units; two waste sites each with three statistically distinct decision units; two waste sites each with four statistically distinct decision units; and one waste site with seven statistically distinct decision units).

The overall total risk summary for the casual recreational user from all shallow zone waste sites evaluated is provided in Table 6-15, which lists the OU that each waste site resides in the reclassification status, the remediated waste site, the consolidated 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).

As presented in Table 6-15, the potential cumulative total ELCR is within the regulatory target risk range of 10^{-4} to 10^{-6} at 27 remediated waste sites, is less than the lower risk threshold value of 1×10^{-6} at 39 remediated waste sites, and COPCs were not reported at concentrations above background at nine remediated waste sites.

The casual recreational user scenario results for shallow vadose zone material are summarized by decision unit in Table G-49 (Appendix G).

Table 6-15. Summary of Total Risks for the 100-BC-1 and 100-BC-2 Shallow Zone Waste Sites for the Casual Recreational User Scenario

Classification Status	Remediated Waste Site	Associated Waste Site[a]	Decision Unit with Exceedance	Total ELCR[b]	Risk Driver and % Contribution
100-BC-1					
Interim Closed	116-B-6A	116-B-16	None	1×10^{-4} to 1×10^{-6}	None
	100-B-14:2 100-B-18 100-B-21:4 100-B-25 100-B-28 100-B-8:2 116-B-1 116-B-11 116-B-14 116-B-2 116-B-5 116-B-7, 132-B-6, 132-C-2 116-C-1 116-C-5 126-B-3 1607-B2:1	--			

Table 6-15. Summary of Total Risks for the 100-BC-1 and 100-BC-2 Shallow Zone Waste Sites for the Casual Recreational User Scenario

Classification Status	Remediated Waste Site	Associated Waste Site[a]	Decision Unit with Exceedance	Total ELCR[b]	Risk Driver and % Contribution
	100-B-14:1 100-B-16 100-B-19 100-B-20 100-B-21:2 100-B-21:3 100-B-22:2 100-B-33 100-B-5 100-B-8:1 116-B-10 116-B-13 116-B-4 118-B-10 120-B-1 128-B-2 128-B-3 1607-B2:2 1607-B7	--	None	$< 1\times10^{-6}$	None
	100-B-32 116-B-12 116-B-3 116-B-6B 116-B-9 118-B-5 118-B-7	--	None	No COPCs reported above background	None

Table 6-15. Summary of Total Risks for the 100-BC-1 and 100-BC-2 Shallow Zone Waste Sites for the Casual Recreational User Scenario

Classification Status	Remediated Waste Site	Associated Waste Site[a]	Decision Unit with Exceedance	Total ELCR[b]	Risk Driver and % Contribution
No Action	100-B-26 116-B-15	--	None	1×10^{-4} to 1×10^{-6}	None
	100-B-11 100-B-14:6 100-B-14:7 1607-B1	--	None	$< 1\times10^{-6}$	None
	100-B-14:5 118-B-9	--	None	No COPCs reported above background	None
100-BC-2					
Interim Closed	--	100-C-6:1	See 100-B-8:1 (100-BC-1)		
	--	100-C-6:2 100-C-6:3 100-C-6:4	See 100-B-8:2 (100-BC-1)		
	118-B-3	118-B-2	None	1×10^{-4} to 1×10^{-6}	None
	100-B-23 116-C-3 116-C-6 118-B-1 118-B-6 118-C-1 600-233	--			
	116-C-2A	116-C-2B 116-C-2C	None	$< 1\times10^{-6}$	None

Table 6-15. Summary of Total Risks for the 100-BC-1 and 100-BC-2 Shallow Zone Waste Sites for the Casual Recreational User Scenario

Classification Status	Remediated Waste Site	Associated Waste Site[a]	Decision Unit with Exceedance	Total ELCR[b]	Risk Driver and % Contribution
	100-B-1				
	100-B-31				
	100-C-3				
	100-C-9:1				
	100-C-9:2				
	118-B-4				
	118-C-2				
	118-C-3:3	--			
	118-C-4				
	128-C-1				
	1607-B8				
	1607-B9				
	1607-B10				
	1607-B11				
	600-232				

Results summarized from Table G-48, Casual User Scenario Risk Estimates and Noncancer Hazards for the 100-B/C Source OU Waste Site Decision Units Without Background Contribution

a. Associated waste sites are those sites for which remediation and closeout documentation were consolidated with another remediated waste site.

b. Total ECLR represent risk contributions from both radiological and nonradiological COPCs.

As presented in Table G-49 (Appendix G), the potential cumulative ELCR from direct contact for all nonradiological carcinogenic COPCs without background contribution ranges from 4.3×10^{-13} to 5.5×10^{-6} for 81 of the 109 shallow zone decision units. The potential cumulative ELCR is within the regulatory target risk range of 10^{-4} to 10^{-6} for 10 decision units and less than the lower regulatory risk threshold value of 1×10^{-6} for 71 decision units. Nonradiological carcinogenic COPCs were not reported at 28 decision units.

As presented in Table G-49 (Appendix G), the potential HI from direct contact for noncancer effects without background contribution is less than 1 for all shallow zone decision units. The potential HI is less than the EPA target HI of 1.

Overburden. A total of 23 waste sites are reported with CVP/RSVP data associated with overburden material in the 100-BC Source OU. The following sample designs were applied to the 23 waste sites:

- Thirteen waste sites were sampled using a focused sampling design.
- Ten waste sites were sampled using a statistical sampling design (with one waste site with two statistically distinct decision units, one waste site with three statistically distinct decision units, and one waste site with seven statistically distinct decision units).
- Six decision units from 100-B-14 reported all concentrations below detection limits, resulting in 26 overburden decision units for evaluation.

The casual recreational user scenario results for overburden materials are summarized by decision unit in Table G-50 (Appendix G).

As presented in Table G-50 (Appendix G), the potential cumulative ELCR from all radiological COPCs without background contribution ranges from 1.9×10^{-9} to 1.2×10^{-5} for 15 of the 26 overburden decision units. The potential cumulative ELCR is within the regulatory target risk range of 10^{-4} to 10^{-6} for five decision units and is less than the lower regulatory risk threshold value of 1×10^{-6} for 10 decision units. Radiological COPCs were not reported at 11 decision units.

As presented in Table G-50 (Appendix G), the potential cumulative ELCR from direct contact for all nonradiological carcinogenic COPCs without background contribution ranges from 4.1×10^{-13} to 5.9×10^{-6} for 23 of the 26 overburden decision units. The potential cumulative ELCR is within the regulatory target risk range of 10^{-4} to 10^{-6} for two decision units and is less than the lower regulatory risk threshold value of 1×10^{-6} for 21 decision units. Nonradiological carcinogenic COPCs were not reported in three decision units.

As presented in Table G-50 (Appendix G), the potential HI from direct contact for noncancer effects without background contribution is less than 1 for all overburden decision units. The potential HI is less than the EPA target HI of 1.

Staging Pile Area. A total of 19 waste sites are reported with CVP/RSVP data associated with staging pile material in the 100-BC Source OU. The following sample designs were applied to the 19 waste sites evaluated:

- Four waste sites were sampled using a focused sampling design.
- Eleven waste sites were sampled using a statistical sampling design (one waste site with two statistically distinct decision units).

- Four waste sites were sampled using both a statistical and a focused sampling design (one waste site with two statistically distinct decision units).

The casual recreational user scenario results for staging pile footprint material are summarized by decision unit in Table G-51 (Appendix G).

As presented in Table G-51 (Appendix G), the potential cumulative ELCR from all radiological COPCs without background contribution ranges from 1.1×10^{-9} to 2.7×10^{-6} for 10 of the 25 staging pile area decision units. The potential cumulative ELCR is within the regulatory target risk range of 10^{-4} to 10^{-6} for three decision units and is less than the lower regulatory risk threshold value of 1×10^{-6} for seven decision units. Radiological COPCs were not reported at 15 decision units.

As presented in Table G-51 (Appendix G), the potential cumulative ELCR from direct contact for all nonradiological carcinogenic COPCs without background contribution ranges from 2.0×10^{-12} to 2.1×10^{-6} for 16 of the 25 staging pile area decision units. The potential cumulative ELCR is within the regulatory target risk range of 10^{-4} to 10^{-6} for 1 decision unit and is less than the lower regulatory risk threshold value of 1×10^{-6} for 15 decision units. Nonradiological carcinogenic COPCs were not reported at nine decision units.

As presented in Table G-50 (Appendix G), the potential HI from direct contact for noncancer effects without background contribution is less than 1 for all staging pile area decision units. The potential HI is less than the EPA 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, 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 if the remedial action processes implemented met the remedial action objectives 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 record of decision. 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 if remedial action objectives had been met report fewer analytes than those that have been remediated more recently. Waste sites that

are associated with the interim action ROD for the remaining sites tend to have verification samples analyzed using a method-based approach. These generally include burial grounds and waste sites identified during discovery process. Risks may be understated for waste sites that did not implement a method-based approach for sampling because fewer analytes are reported.

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 :

- 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 (<5)
- When 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 Uncertainties Associated with Sample Design and Depth of Collection

Soil samples collected from 100-B-21:4, 116-B-5, 116-B-14, 116-C-3, and 118-B-1 are assigned to the shallow zone decision unit for each of these waste sites for interim closure purposes. MTCA "Unrestricted Land Use Soil Cleanup Standards," "Point of compliance" (WAC 173-340-740(6)(d)) indicates that the point of compliance for human exposure via direct contact be established in the soils throughout the site from the ground surface to 4.6 m (15 ft) bgs. Closeout documentation for each of these waste sites indicate that the excavation depth was greater than 4.6 m (15 ft) bgs; therefore, verification samples are actually collected from depths greater than 4.6 m (15 ft) bgs. For these five waste sites, the direct contact exposure pathway is considered incomplete because samples are collected at depths greater than 4.6 m (15 ft) bgs. Comparison of EPCs to residential RBSLs overstates risk for direct contact exposure because sample collection depths are greater than 4.5 m (15 ft) bgs. Remedial action for these four sites should be similar to those that are considered for deep zone decision units. The time required for radioisotope concentrations to decay to the residential RBSLs is provided in Section 6.2.6.4.

6.2.6.4 Uncertainties in EPCs Associated with Decay of Radioisotopes

Section 6.2.5.5 provides a summary of the risk estimates by exposure scenario evaluated. The results of the soil risk assessment for the residential scenario identify a group of waste sites with concentrations of site-related COPCs that result in individual risks greater than the upper regulatory risk threshold value of 1×10^{-4}. Table G-52 (Appendix G) lists the 100-BC waste sites, the radioisotopes reported with concentrations greater than their respective residential RBSL for each waste site, 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 less than the residential RBSL. Deep zone soil samples are collected from depths greater than 4.6 m (15 ft) bgs; as a result, direct contact within the point of compliance is incomplete. Additionally, the residential exposure scenario does not reflect reasonably anticipated future land use in the River Corridor. This comparison is included because it is the most conservative land use basis for the

evaluation of waste sites and presentation of these results is included only to provide additional information for risk management decisions.

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)

Concentrations of radioisotopes currently measured in shallow decision units where the final excavation depth is less than 4.6 m (15 ft) bgs decay to activity levels less than residential RBSLs as follows:

- Strontium-90 concentrations at 116-B-6A (shallow decision unit) decayed to a level less than the residential RBSL in year 2004.

Concentrations of radioisotopes currently measured at depths greater than 4.6 m (15 ft) bgs or deep decision units decay to activity levels less than residential RBSLs as follows:

- Cesium-137 concentrations at 100-B-21:4 (overburden focused decision unit) decay to levels less than the residential RBSL by year 2035; 100-B-21:4 (shallow decision unit) decay to levels less than the residential RBSL by year 2110.
- Cesium-137 and europium-152 concentrations at 100-B-5 (deep decision unit) decay to levels less than residential RBSLs by 2073.
- Cesium-137 concentrations at 100-B-8:1 (deep decision unit) decay to levels less than the residential RBSL by year 2037.
- Cesium-137, europium-152, and strontium-90 concentrations at 100-B-8:2 (deep decision unit) decay to levels less than residential RBSLs by year 2048; strontium-90 concentrations at 100-B-8:2 (overburden decision unit) decayed to levels less than the residential RBSL by year 2111.
- Europium-152 concentrations at 116-B-1 (deep decision unit) decayed to a levels less than the residential RBSL by year 2012.
- Cesium-137, cobalt-60, europium-152, europium-154, nickel-63, and strontium-90 concentrations at 116-B-11 (deep decision unit) decay to levels less than residential RBSLs by year 2205.
- Europium-152 concentrations at 116-B-14 (shallow decision unit) decayed to levels less than the residential RBSL by year 2002; cesium-137 concentrations at 116-B-4 (deep decision unit) decayed to a level less than the residential RBSL in year 2011.
- Cesium-137 and strontium-90 concentrations at 116-B-2 (deep decision unit) decay to levels less than residential RBSLs by year 2102.

- Cesium-137 and strontium-90 concentrations at 116-B-3 (deep decision unit) decay to levels less than residential RBSLs by year 2064.
- Cobalt-60, europium-152, and europium-154 concentrations at 116-B-4 (deep decision unit) decay to levels less than residential RBSLs by year 2139.
- Europium-152 and tritium concentrations at 116-B-5 (shallow focused decision unit) decayed to levels less than residential RBSLs in year 2008; europium-152 concentrations at 116-B-5 (deep focused decision unit) decay to levels less than the residential RBSL by year 2014.
- Strontium-90 concentrations at 116-B-6A (deep decision unit) decay to levels less than the residential RBSL by year 2093.
- Cesium-137, cobalt-60, europium-152, europium-154, nickel-63, and strontium-90 concentrations at 116-C-1 (deep decision unit) decay to levels less than residential RBSLs by year 2262; cesium-137, cobalt-60, europium-152, europium-154, nickel-63, and strontium-90 concentrations at 116-C-1 (deep focused decision unit) decay to levels less than residential RBSLs by year 2308.
- Cesium-137, cobalt-60, europium-152, and strontium-90 concentrations at 116-C-2A (deep decision unit) decay to levels less than residential RBSLs by year 2071; cesium-137 concentrations at 116-C-2A (overburden decision unit) decay to level less than the residential RBSL by year 2043.
- Cesium-137 and strontium-90 concentrations at 116-C-3 (shallow focused decision unit) decay to levels less than residential RBSLs by year 2094.
- Cesium-137, cobalt-60, europium-152, europium-154, nickel-63, and strontium-90 concentrations at 116-C-5 (deep decision unit) decay to levels less than residential RBSLs by year 2106.
- Strontium-90 concentrations at 118-B-1 (shallow 3 decision unit) decay to levels less than the residential RBSL by year 2034; strontium-90 concentrations at 118-B-1 (shallow 5 decision unit) decayed to a level less than the residential RBSL in year 2009.
- Tritium concentrations at 118-B-6 (deep decision unit) decay to levels less than the residential RBSL by year 2033.
- Cesium-137, europium-152, and strontium-90 concentrations at 118-C-3:2 (deep focused decision unit) decay to levels less than residential RBSLs by year 2156.

Carbon-14 is detected at 100-B-14:1 deep focused decision unit and 118-C-1 staging pile area at concentrations that result in risks above 1×10^{-4}. It is also noted that 118-C-1 staging pile area material from this sample location with elevated carbon-14 concentrations was backfilled into deep zone portions of the site. Decay does not occur within a reasonable period for carbon-14 because the half-life is 5,730 years and is not included in previous calculations.

6.2.6.5 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.

6.2.6.6 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* [EPA/630/P-03/001F]),which modifies the agency's 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 reference dose 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 cleanup level would decrease from 240 mg/kg to 2.0 mg/kg. The use of the oral reference dose published by IRIS may result in underestimating risk.

6.2.6.7 Uncertainties Associated with Risk Characterization

In the risk characterization, the assumption was made that the total risk of developing cancer from exposure to 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.

6.2.6.7.1 Uncertainties in Risk Estimates Associated with Remedial Investigation and Limited Field Investigation Soil Data

In addition to the waste site remediation data (CVP/RSVP), two additional sources of data were considered for use in the risk assessment. : (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-BC OU. These data were collected for purposes other than fulfilling needs of the risk assessment. As such, they were not used to quantitatively evaluate risks; however, these data were evaluated qualitatively by comparing concentrations of analytes to risk-based screening levels to determine if the results could be useful for risk management decisions.

Remedial Investigation Data. Soil data identified as useful for informing risk management decisions include those collected to fill data gap No. 2, No. 3, and No. 4. Chapter 2, Table 2-1 lists the data gaps and the work conducted per the 100-BC RI/FS Work Plan (DOE/RL-2008-46 ADD3). The soil borings, test pits, and wells considered for use in the qualitative risk evaluation are listed in Table 6-16 for the 100-B/C Source OU. As shown in Table 6-16, eight boreholes, three test pits, and nine monitoring wells were drilled for the RI.

Limited Field Investigation Data. In the early 1990s, an LFI was performed in the 100-BC-1 and 100-BC-2 OUs. Results of the investigation are presented in DOE/RL-93-06 and DOE/RL-94-42.

Based on the work plan, the following waste sites were investigated in the 100-BC-1 and BC-2 OUs:

- 116-B-1 Trench
- 116-B-2 Trench
- 116-B-3 Crib
- 116-B-5 Crib
- 116-B-14 Trench
- 116-C-5 Retention Basin
- 116-C-2A Crib
- 116-C-1 Trench.

An LFI was also performed in the 100-BC-5 OU. Results of the investigation are presented in DOE/RL-93-37. Ten new wells were drilled for the LFI to determine the nature and extent of hazardous/radioactive materials present in the groundwater. Of these 10 wells, soil samples were collected from the following five wells and are included in the uncertainty evaluation:

- 199-B2-12
- 199-B4-8
- 199-B4-9

- 199-B9-2
- 199-B9-3

Comparison to Human Health RBSLs. All RI and LFI soil data from the soil borings and wells described in the previous section were compared to the human health RBSLs used in the soil risk assessment. A summary of the soil borings/wells and associated depth intervals for the 100-BC Source OUs is provided in Table G-53 (Appendix G).

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 6.2.2.2. 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 RBSLs developed for the residential, resident Monument worker, and the casual recreational user scenarios. Soil samples collected from depth intervals greater than 4.6 m (15 ft) bgs were combined and the maximum detected concentration was compared to the Hanford Site background concentration and the RBSLs for the residential scenario. EPCs are not calculated using the RI and LFI data.

6.2.6.7.2 Risk Results for Soil Samples Collected from 0 to 4.6 m (15 ft) bgs

Soil samples were collected from 0 to 4.6 m (15 ft) bgs from two RI soil borings, one RI test pit, one LFI text pit, and three LFI soil borings. A comparison of maximum detected concentrations to RBSLs from each of the well/test pit locations from 100-BC is provided in Table G-54 (Appendix G). A summary of the wells and test pits, which report a maximum concentration greater than the residential RBSLs for the 100-BC Source OUs, is provided in Table 6-16.

For the 100-B/C Source OU, three LFI soil borings (116-B-2 Fuel Storage Basin Trench, 116-B-3 Pluto Crib, and 116-C-5 Retention Basin) report soil concentrations greater than residential RBSLs. These three waste sites have been subsequently remediated under the interim action ROD. As shown in Table 6-16, risk results for these interim remediated waste sites are within acceptable EPA risk thresholds. The risk results for the remaining LFI soil boring with samples in the shallow zone (199-B5-4 at the 116-B-5 waste site) are within acceptable EPA risk thresholds

The RI soil boring collected from the 118-B-8:1 Fuel Storage Basin (B Reactor Subsite) (C7847) reports concentrations of cesium-137, europium-152, and strontium-90 greater than residential RBSLs at depths ranging between 3.8 m (12.5 ft) and 4.1 m (13.4 ft) bgs. Concentrations of cesium-137, europium-152, and strontium-90 in the RI borehole decay to levels less than residential RBSLs by year 2203 (Table G-56, Appendix G). These are results from the first attempted boring near the 105-B FSB and had to be abandoned following the collection of samples at four intervals because of refusal at 4.6 m (15 ft) bgs. These samples were collected to determine the nature and vertical extent of contamination in the vadose zone around the 105-B Reactor structure. The risk results for all remaining RI soil borings are within acceptable EPA risk thresholds. The 118-B-8:1 waste site is an accepted site and remediation has not occurred.

6.2.6.7.3 Risk Results for Soil Samples Collected from Depths Greater than 4.6 m (15 ft) bgs

Soil samples were collected from depths greater than 4.6 m (15 ft) bgs from seven RI soil borings, nine RI wells, two RI test pits, one LFI test pit, eight LFI soil borings, and five LFI wells. A comparison of maximum detected concentrations to RBSLs from each of these well/test pit locations from 100-BC is provided in Appendix G, Table G-55. A summary of the soil borings, wells and test pits which report a maximum concentration greater than the residential RBSLs for the 100-BC Source OUs is provided in Table 6-16. A discussion of waste sites that report concentrations greater than residential RBSLs is provided below.

RI soil boring/well samples from 100-B-5 Trench (199-B4-15/C7846), 116-C-5 Retention Basin (116-B3-52/C7843), 118-B-6 Burial Ground (C7845), 118-B-8:1 (C8239), and 199-B5-6 (C7507) report radionuclide concentrations greater than residential RBSLs. LFI soil boring samples from 199-B5-2 from the process effluent pipelines south of 116-C-5, 116-B-1 Trench (199-B3-48), 116-C-1 Liquid Waste Disposal Trench (199-B3-46), and 116-C-2A Pluto Crib (199-B9-4) 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 and are presented in Table G-56 (Appendix G). Table G-56 (Appendix G) lists the results from the above soil borings/wells, the radioisotopes reported with concentrations greater than their respective residential RBSL, the year the samples were collected, the detected concentration for the depth interval, the half-life for each radioisotope, and the year that each radioisotope decays to an activity level less than the residential RBSL.

At the 100-B-5 Trench, RI soil boring data (199-B4-15 [C7846]) and CVP/RSVP closeout data (CVP-2003-00014) are available (see Table 6-16). The RI data indicate that cesium-137 and strontium-90 are reported at concentrations greater than their respective RBSLs. These radioisotopes are present at depths ranging between 8.5 m (27.9 ft) and 18.5 m (60.7 ft) bgs. Concentrations of cesium-137 and strontium-90 in the RI borehole decay to activity levels less than residential RBSL by year 2071 (see Table G-56). 100-B-5 Trench was also evaluated in the soil risk assessment. The risk assessment for 100-B-5 Trench waste site (deep decision unit) reports a cumulative ELCR of 1.1×10^{-3}. The primary contributors to risk include cesium-137 (5.1×10^{-4}; 47 percent contribution), and europium-152 (4.2×10^{-4}; 38 percent contribution). Concentrations of cesium-137, europium-152, and strontium-90 decay to levels less than residential RBSLs by year 2073 (Table G-52, Appendix G).

At the 116-C-5 Retention Basin, RI soil boring data (199-B3-52 [C7843]), CVP/RSVP closeout data (CVP-99-00004), and LFI data (116-B-11 TP and 199-B5-2) are available (see Table 6-16). The RI data indicate that cesium-137 is reported at concentrations greater than its respective RBSL. Cesium-137 is present at depths ranging between 10.6 m (34.9 ft) and 12.3 m (40.3 ft) bgs. Concentrations of cesium-137 in the RI borehole decay to activity levels less than residential RBSL by year 2029 (Table G-56, Appendix G). The LFI data for 116-B-1 TP indicate that individual risks from all detected analytes are less than the risk threshold of 1×10^{-4}. The LFI data for 199-B5-2 indicate that strontium-90 is reported at concentrations greater than its respective RBSL. Strontium-90 is present at depths ranging between 16 m (53 ft) and 17 m (57 ft) bgs. Concentrations of strontium-90 in the LFI borehole have decayed to activity levels less than residential RBSL by year 2002 (see Table G-56). 116-C-5 Retention Basin was also evaluated in the soil risk assessment. The risk assessment for 116-C-5 Retention Basin waste site (deep decision unit) reports a cumulative ELCR of 4.4×10^{-3}. The primary contributors to risk include cesium-137 (1.2×10^{-3}; 27 percent contribution), cobalt-60 (5.9×10^{-4}; 13 percent contribution), europium-152 (2.0×10^{-3}; 45 percent contribution), europium-154 (3.6×10^{-4}; 8 percent contribution), Nickel-63 (1.1×10^{-4}; 2 percent contribution), and strontium-90 (1.9×10^{-4}; 4 percent contribution). Concentrations of cesium-137, cobalt-60, europium-152, europium-154, nickel-63, and strontium-90 decay to levels less than residential RBSLs by year 2106 (Table G-52, Appendix G).

At the 118-B-6 Burial Ground, RI soil boring data (C7845) and CVP/RSVP closeout data (CVP-2006-00002) are available (see Table 6-16). The RI data indicate that strontium-90 and tritium are reported at concentrations greater than their respective RBSLs. strontium-90 is present at depths ranging between 21.6 m (71 ft) and 24.0 m (78.9 ft) bgs and tritium is present at depths ranging between 9.5 m (31.1 ft) and 22.4 (73.5 ft). Concentrations of strontium-90 and tritium in the RI borehole decay to activity levels less than residential RBSLs by year 2042 (see Table G-56). 118-B-6 Burial Ground was also evaluated in the soil risk assessment. The risk assessment for 118-B-6 Burial Ground waste site (deep decision unit) reports a cumulative ELCR of 4.5×10^{-4}. The primary contributor to risk includes tritium

(4.5×10^{-4}; 100 percent contribution). Concentrations of tritium decay to levels less than residential RBSLs by year 2033 (Table G-52, Appendix G).

At the 118-B-8:1 Fuel Storage Basin (B Reactor subsite), RI soil boring data (C8239) are available (Table 6-16). The RI data indicate that carbon-14 and strontium-90 are reported at concentrations greater than their respective RBSLs. Carbon-14 is present at depths ranging between 5.3 m (17.4 ft) and 6.1 m (19.9 ft) bgs. Strontium-90 is present at depths ranging between 5.3 m (17.4 ft) and 25 m (82.2 ft) bgs. Concentrations of strontium-90 in the RI borehole decay to levels less than residential RBSL by year 2080 (Table G-56, Appendix G). Concentrations of carbon-14 do not decay within a reasonable time frame.

RI soil data are available at the 199-B5-6 (C7507) monitoring well (see Table 6-16). The RI data indicate that tin-126 is reported at a concentrations greater than its RBSL at a depth ranging between 22.8 m (74.9 ft) and 23.6 m (77.4 ft) bgs. Concentrations of tin-126 do not decay within a reasonable time frame. Eight soil samples were collected and analyzed for tin-126 from 199-B5-6. With the exception of the one depth interval, all tin-126 activity levels were not detectable. The single occurrence of tin-126 is likely an anomalous result because it is not known to be associated with releases at the Hanford site.

At the 116-B-1 Trench, CVP/RSVP closeout data (CVP-99-00012) and LFI data (199-B3-48) are available (see Table 6-16). The LFI data indicate that cesium-137, cobalt-60, europium-152, europium-154, and strontium-90 are reported at concentrations greater than their respective RBSLs. These radioisotopes are present at depths ranging between 4.6 m (15 ft) and 8.2 (27 ft) bgs. Concentrations of cesium-137, cobalt-60, europium-152, europium-154, and strontium-90 in the LFI borehole decay to levels less than residential RBSL by year 2092 (see Table G-56). 116-B-1 Trench was also evaluated in the soil risk assessment. The risk assessment for 116-B-1 Trench waste site (deep decision unit) reports a cumulative ELCR of 3.4×10^{-4}. The primary contributors to risk include cesium-137 (7.1×10^{-5}; 21 percent contribution), europium-152 (1.9×10^{-4}; 57 percent contribution), and strontium-90 (6.3×10^{-5}; 18 percent contribution). Concentrations of cesium-137, europium-152, and strontium-90 decayed to levels less than residential RBSLs in year 2012 (see Table G-52).

At the 116-C-1 Liquid Waste Disposal Trench, CVP/RSVP closeout data (CVP-98-00006) and LFI data (199-B3-46) are available (see Table 6-16). The LFI data indicate that strontium-90 is reported at concentrations greater than its respective RBSL. Strontium-90 is present at depths ranging between 10.7 m (35 ft) and 11.3 (37 ft) bgs. Concentrations of strontium-90 in the LFI borehole decay to levels less than residential RBSL by year 2044 (Table G-56, Appendix G). 116-C-1 Liquid Waste Disposal Trench was also evaluated in the soil risk assessment. The risk assessment for 116-C-1 Liquid Waste Disposal Trench (deep focused decision unit) reports a cumulative ELCR of 1.7×10^{-1}. The primary contributors to risk include cesium-137 (1.3×10^{-1}; 76 percent contribution), cobalt-60 (3.7×10^{-3}; 2.2 percent contribution), europium-152 (3.1×10^{-2}; 18 percent contribution), europium-154 (3.4×10^{-4}; 1.9 percent contribution), nickel-63 (2.6×10^{-4}; 0.15 percent contribution), and strontium-90 (3.9×10^{-3}; 2.3 percent contribution). Concentrations of cesium-137, cobalt-60, europium-152, europium-154, nickel-63, and strontium-90 decay to levels less than residential RBSLs by year 2308 (Table G-52, Appendix G).

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Table 6-16. Summary of Human Health Risk Comparisons for RI Data, CVP/RSVP Data, and LFI Data

Waste Site	RI Data	Shallow Zone (<4.6 m [<15 ft] bgs) Direct Contact Human Health Risks?	Deep Zone (> 4.6 m [>15 ft] bgs) Radiological Risks?	CVP/RSVP Data	Shallow Zone (<15 ft/bgs) Direct Contact Human Health Risks?	Deep Zone (>15 ft bgs) Radiological Risks?	LFI Data	Shallow Zone (<15 ft bgs) Direct Contact Human Health Risks?	Deep Zone (>15 ft bgs) Radiological Risks?
Soil Borings Installed to Characterize Nature and Vertical Extent of Contamination Beneath Select Waste Site									
100-B-5 Trench	199-B4-15 (C7846)	No samples collected from this depth range	Cesium-137 (27.9 to 35.3 ft bgs) and strontium-90 (27.9 to 60.7 ft bgs) > residential RBSL	CVP-2003-00014	$<1 \times 10^{-6}$	1.1×10^{-3} (Cs-137, Eu-152)	--	--	--
116-B-14 Trench	C7842	No samples collected from this depth range	No individual risks > thresholds	CVP-99-00003	1.9×10^{-4} (Eu-152)	2.4×10^{-4} (Cs-137)	199-B3-47	No samples collected from this depth range	No individual risks > thresholds
116-B-5 Crib	C7844	No samples collected from this depth range	No individual risks > thresholds	WSRF-98-064	4.0×10^{-4} (Eu-152, Tritium)	3.2×10^{-4} (Cs-137, Eu-152, Sr-90)	199-B5-4	No individual risks > thresholds	No individual risks > thresholds
116-B-6B Crib	TP1	No samples collected from this depth range	No COPCs detected	CVP-99-00017	No COPCs reported above background	--	--	--	--
116-B—9 French Drain	TP2	No individual risks > thresholds	No samples collected from this depth range.	WSRF-2004-004	No COPCs reported above background	--	--	--	--
116-C-5 Retention Basin	199-B3-52 (C7843)	No samples collected from this depth range	Cesium-137 > residential RBSL (34.9 to 40.3 ft bgs)	CVP-99-00004	1×10^{-4} (cesium-137, europium-152, strontium-90)	4.4×10^{-3} (Cs-137, Co-60, Eu-152, Eu-154, Ni-63, Sr-90)	116-B-11-TP1	Cs-137, Co-60, and Eu-152 > residential RBSL (5 to 8 ft bgs)	No individual risks > thresholds
Process effluent pipelines south of 116-C-5	--	--	--				199-B5-2	No samples collected from this depth range	Sr-90 > residential RBSL (53 to 57 ft bgs)
118-B-6 Burial Ground	C7845	No samples collected from this depth range	Strontium-90 (71 to 78.9 ft bgs) and Tritium (31.1 to 73.5 ft bgs) > residential RBSLs	CVP-2006-00002	1×10^{-4} to 1×10^{-6}	4.5×10^{-4} (Tritium)	--	--	--
118-B-8:1 Fuel Storage Basin (B Reactor subsite)	C7847	Cesium-137, europium-152, strontium-90 > residential RBSLs (12.5 – 13.4 ft bgs)	No samples collected from this depth range.	Accepted	--	--	--	--	--
	C8239	No individual risks > thresholds	Carbon-14 (17.4 to 19.9 ft bgs) and strontium-90 (17.4 to 82.2 ft > residential RBSLs)		--	--	--	--	--
118-B-8:3 Process Sewer (B Reactor subsite)	TP3	No samples collected from this depth range	No individual risks > thresholds	Accepted	--	--	--	--	--
118-C-3:2 Fuel Storage Basin (C Reactor Subsite)	C7849	No samples collected from this depth range	No individual risks > thresholds	CVP-98-00009	--	6.0×10^{-3} (Cs-137, Eu-152, Sr-90)	--	--	--
Wells Installed to Characterize Nature and Extent of Contamination in Rewetted Zone (not waste site-specific)									
--	199-B5-5 (C7505)	No samples collected from this depth range	No individual risks > thresholds	--	--	--	--	--	--
--	199-B3-50 (C7506)	No samples collected from this depth range	No individual risks > thresholds	--	--	--	--	--	--

Table 6-16. Summary of Human Health Risk Comparisons for RI Data, CVP/RSVP Data, and LFI Data

Waste Site	RI Data	Shallow Zone (<4.6 m [<15 ft] bgs) Direct Contact Human Health Risks?	Deep Zone (> 4.6 m [>15 ft] bgs) Radiological Risks?	CVP/RSVP Data	Shallow Zone (<15 ft/bgs) Direct Contact Human Health Risks?	Deep Zone (>15 ft bgs) Radiological Risks?	LFI Data	Shallow Zone (<15 ft bgs) Direct Contact Human Health Risks?	Deep Zone (>15 ft bgs) Radiological Risks?
--	199-B5-6 (C7507)	No samples collected from this depth range	Tin-126 > residential RBSL (74.9 to 77.4 ft bgs)	--	--	--	--	--	--
--	199-B8-9 (C7508)	No samples collected from this depth range	No COPCs detected	--	--	--	--	--	--
--	199-B2-14 (C7665)	No samples collected from this depth range	No individual risks > thresholds	--	--	--	--	--	--
--	199-B2-15 (C7783)	No samples collected from this depth range	No COPCs detected	--	--	--	--	--	--
--	199-B2-16 (C7784)	No samples collected from this depth range	No individual risks > thresholds	--	--	--	--	--	--
--	199-B3-51 (C7785)	No samples collected from this depth range	No individual risks > thresholds	--	--	--	--	--	--
--	199-B5-8 (C8244)	No samples collected from this depth range	No individual risks > thresholds	--	--	--	--	--	--
Soil Borings Installed during LFI to Characterize Priority Waste Sites in 100-BC-1 and 100-BC-2									
116-B-1 Trench	--	--	--	CVP-99-00012	1×10^{-4} to 1×10^{-6}	3.4×10^{-4} (Cs-137, Eu-152, Sr-90)	199-B3-48	No samples collected from this depth range.\	Cs-137, Co-60, Eu-152, Eu-154, and Sr-90 > residential RBSL (15 to 27 ft bgs)
116-B-2 Fuel Storage Basin Trench	--	--	--	CVP-99-00015	1×10^{-4} to 1×10^{-6}	1.4×10^{-3} (Cs-137, Sr-90)	199-B4-10	Cs-137. Eu-152, and Sr-90 > residential RBSLs (10 to 12 ft bgs)	No individual risks > thresholds
116-B-3 Pluto Crib	--	--	--	CVP-99-00013	No COPCs reported above background	5.9×10^{-4} (Cs-137, Sr-90)	199-B5-3	Cs-137 and Sr-90 > residential RBSLs (7.5 to 13 ft bgs)	No individual risks > thresholds
116-C-1 Liquid Waste Disposal Trench	--	--	--	CVP-98-00006	1×10^{-4} to 1×10^{-6}	1.7×10^{-1} (Cs-137, Co-60, Eu-152, Eu-154, Ni-63, Sr-90)	199-B3-46	No samples collected from this depth range	Sr-90 > residential RBSL (35 to 37 ft bgs)
116-C-2A Pluto Crib	--	--	--	CVP-99-00019	1×10^{-4} to 1×10^{-6}	2.2×10^{-3} (Cs-137, Co-60, Eu-152, Sr-90)	199-B9-4	No samples collected from this depth range	Ni-63 and Sr-90 > residential RBSL (22.9 to 26.9 ft bgs)
Monitoring Wells Installed for 100-BC-5 OU LFI									
--	--	--	--	--	--	--	199-B2-12	No samples collected from this depth range	No individual risks > thresholds
--	--	--	--	--	--	--	199-B4-8	No samples collected from this depth range	No individual risks > thresholds

Table 6-16. Summary of Human Health Risk Comparisons for RI Data, CVP/RSVP Data, and LFI Data

Waste Site	RI Data	Shallow Zone (<4.6 m [<15 ft] bgs) Direct Contact Human Health Risks?	Deep Zone (> 4.6 m [>15 ft] bgs) Radiological Risks?	CVP/RSVP Data	Shallow Zone (<15 ft/bgs) Direct Contact Human Health Risks?	Deep Zone (>15 ft bgs) Radiological Risks?	LFI Data	Shallow Zone (<15 ft bgs) Direct Contact Human Health Risks?	Deep Zone (>15 ft bgs) Radiological Risks?
--	--	--	--	--	--	--	199-B4-9	No samples collected from this depth range	Cs-137 (16 to 28 ft bgs) and Co-60 (16 to 18 ft bgs) > residential RBSLs
--	--	--	--	--	--	--	199-B9-2	No samples collected from this depth range	No individual risks > thresholds
--	--	--	--	--	--	--	199-B9-3	No samples collected from this depth range	No individual risks > thresholds

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At the 116-C-2A Pluto Crib, CVP/RSVP closeout data (CVP-99-00019) and LFI data (199-B9-4) are available (see Table 6-16). The LFI data indicate that nickel-63 and strontium-90 are reported at concentrations greater than their respective RBSLs. strontium-90 is present at depths ranging between 7.0 m (22.9 ft) and 8.2 m (26.9 ft) bgs. Concentrations of nickel-63 and strontium-90 in the LFI borehole decay to levels less than residential RBSL by year 2223 (Table G-56, Appendix G). 116-C-2A Pluto Crib was also evaluated in the soil risk assessment. The risk assessment for 116-C-2A Pluto Crib (deep decision unit) reports a cumulative ELCR of 2.2×10^{-3}. The primary contributors to risk include cesium-137 (5.2×10^{-4}; 24 percent contribution), cobalt-60 (3.9×10^{-4}; 18 percent contribution), europium-152 (8.1×10^{-4}; 37 percent contribution), and strontium-90 (2.7×10^{-4}; 13 percent contribution). Concentrations of cesium-137, cobalt-60, europium-152, europium-154, and strontium-90 decay to levels less than residential RBSLs by year 2071 (Table G-52, Appendix G).

6.3 Groundwater Risk Assessment

EPA guidance provided in "Summary of Key Existing EPA CERCLA Policies for Groundwater Restoration" (Woolford and Reeder, 2009) clarifies EPA's policies for determining whether a groundwater remedial action is warranted under CERCLA. In discussing the role of the BRA, "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) (page 5) clarifies when a CERCLA remedial action is appropriate:

> *A CERCLA remedial action generally is appropriate[7] in various circumstances, including: a regulatory 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;[8] 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 cause adverse environmental impacts.[9] 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 BRA to make risk management decisions such as determining whether remedial action under CERCLA Section 104 or Section 106 is necessary. "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 BRA 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.

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[7] See EPA 540-R-97-013, *Rules of Thumb for Superfund Remedy Selection.*

[8] See Clay, 1991, "Role of the Baseline Risk Assessment in Superfund Remedy Selection Decisions".

[9] See EPA 540-R-97-013, *Rules of Thumb for Superfund Remedy Selection,.*

- 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 HHE and whether remedial action is warranted.

Protectiveness of human health is evaluated by comparing groundwater concentrations to existing federal or state MCLs or nonzero MCLGs. Similarly, protectiveness of aquatic receptors is determined by the comparison of groundwater concentrations 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. Groundwater concentrations are compared to MTCA "Groundwater Cleanup Standards" (WAC 173-340-720) and MTCA "Surface Water Cleanup Standards" (WAC 173-340-730) to determine whether EPCs result in an 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 HHE 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 HHE.

The RCBRA Report (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 1 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 a 4-month period between May 10, 2010, and September 7, 2010. In this RI/FS, three analyses of groundwater data are presented. Section 4.3.1 uses all available information to describe the nature and extent of contamination in groundwater. Section 4.3.1 provides summary statistics for groundwater data collected over the last 5 years and 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-BC-5 Groundwater OU. The 100-BC-5 Groundwater OU groundwater 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 to establish a basis for screening analytes.
- Compare the detected contaminant concentrations to ARARs in order to identify analytes within the entire 100-BC-5 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 ARARs have been exceeded.

Results of this groundwater risk assessment indicate that concentrations of contaminants in the 100-BC-5 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 select remedial alternatives. The conceptual exposure model shows that exposure to groundwater contaminants is through direct contact and fish consumption, while other exposure pathways are considered incomplete or insignificant. The groundwater risk assessment identifies multiple contaminants within the 100-BC-5 Groundwater OU that exceed chemical-specific ARARs. MTCA HHRA Procedures "Multiple Hazardous Substances" (WAC 173-340-708(5)(a)) and "Multiple Pathways of Exposure" (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 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-BC-5 Groundwater Operable Unit* (ECF-100BC5-11-0017) in Appendix G. Attachment G-1 in Appendix G 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. The results of the Tap Water Risk Assessment are provided in *Tap Water Risk Assessment for the 100-BC-5 Groundwater Operable Unit* (ECF-100BC5-11-0018) in Appendix G.

6.3.1 Findings of the River Corridor Baseline Risk Assessment

The RCBRA Report (DOE/RL-2007-21) provides a screening level groundwater risk assessment for the 100-BC-5 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-BC-5 Groundwater OU. Noncancer chemical hazard results were also above the EPA's threshold value of 1.

Uncertainties associated with the groundwater data set were identified in the RCBRA Report (DOE/RL-2007-21). These uncertainties relate to the ability of the groundwater data set 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*. While the monitoring data can be used for risk assessment purposes, uncertainties are 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 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, and ensure that no contaminants were inadvertently overlooked based on the use of the existing data set. 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 Report (DOE/RL-2007-21) 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 Report (DOE/RL-2007-21) identified strontium-90, tritium, and trichloroethene in the 100-BC-5 Groundwater OU as the primary contributors 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 HHE. A preliminary COPC evaluation was conducted to support the 100-BC Work Plan (DOE/RL-2008-46-ADD3) and the 100-BC SAP (DOE/RL-2009-44). The work plan effort evaluated groundwater analytical data from the 100-BC-5 Groundwater OU collected over a 16-year period (1992 to 2008) and resulted in the identification of 36 COPCs, which are listed in Table 6-17.

Table 6-17. List of Historical Contaminants of Potential Concern in the 100-BC-5 Groundwater OU

Metals		
Antimony	Arsenic	Beryllium
Cadmium	Chromium	Cobalt
Copper	Hexavalent chromium	Lead
Manganese	Mercury	Nickel
Selenium	Thallium	Uranium
Zinc		

Table 6-17. List of Historical Contaminants of Potential Concern in the 100-BC-5 Groundwater OU

Volatile Organic Compounds		
1,1-Dichloroethene	1,1,2,2-Tetrachloroethane	Benzene
Carbon tetrachloride	Chloroform	Tetrachloroethene
Trichloroethene	Vinyl chloride	
Radiological		
Carbon-14	Cesium-137	Cobalt-60
Europium-155	Iodine-129	Nickel-63
Radium-228	Strontium-90	Technetium-99
Tritium		
Total Petroleum Hydrocarbons		
Total Petroleum Hydrocarbon (TPH)–Diesel Range		
Anions		
Nitrate		

Source: DOE/RL-2009-44, *Sampling and Analysis Plan for the 100-BC-1, 100-BC-2, and 100-BC-5 Operable Units Remedial Investigation/Feasibility Study*, Table 1-3

Note that although radium-228 was identified as a historical COPC, it is a naturally occurring radionuclide associated with background radiation.

The COPCs identified during the work plan phase were validated by using groundwater samples analyzed using the analytical methods documented in 100-BC SAP (DOE/RL-2009-44), Table 2-14. The groundwater data set used for COPC identification consists of sampling and analysis data collected from 18 monitoring wells within the 100-BC-5 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.

Identification of groundwater COPCs for the 100-BC-5 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 COPCs 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 6-4 presents a summary of the COPC identification process. The left side of Figure 6-4 illustrates the two steps used to identify COPCs. The right side of Figure 6-4 illustrates how the results of the tap water scenario analysis are used to confirm that COPCs are the major risk and hazard drivers.

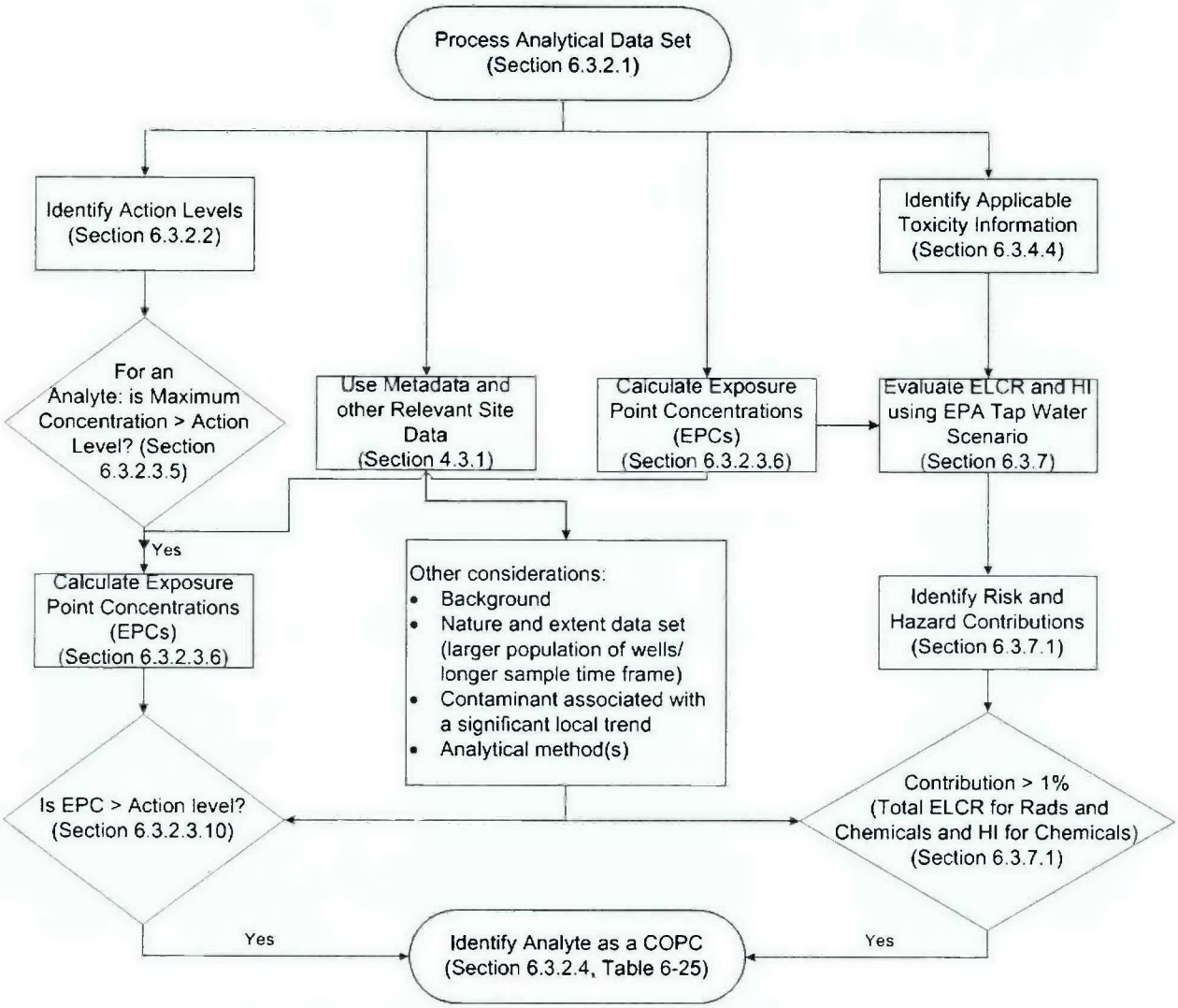

Figure 6-4. Overview of COPC Identification Process

The process used to identify data for COPC selection and the selection of action levels for this groundwater risk assessment is described in Sections 6.3.2.1 and 6.3.2.2. The process used to identify the COPCs and 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 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-BC-5 Groundwater OU.

6.3.2.1 Data Used to Identify Contaminants of Potential Concern

The sampling and analysis data were collected over a 4-month period between May 10, 2010 and September 7, 2010. Three sampling events were used to capture the effects that temporal fluctuations of river stage have on groundwater conditions. Samples collected in May 2010 represent the aquifer when the river stage is at its highest elevation. Samples collected in September 2010 represent the aquifer when the river is at its lowest elevation. Samples collected from July to August 2010 represent the aquifer conditions 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. Table 6-18 lists the wells used in the groundwater risk assessment for the 100-BC-5 Groundwater OU and Figure 6-5 shows their locations.

Table 6-18. Monitoring Wells Used in the Baseline Risk Assessment from the 100-BC-5 Groundwater OU

Well Name		
Near-River Exposure Area (less than 500 m [1,640 ft] from shoreline)		
199-B2-13	199-B2-14	199-B3-1
199-B3-46	199-B3-47	199-B5-1
199-B5-2	199-B5-5	--
Upland Exposure Area (greater than 500 m [1,640 ft] from shoreline)		
199-B3-50	199-B4-4	199-B4-8
199-B5-6	199-B8-6	199-B9-3
699-65-72	699-65-83	699-67-86
699-71-77	--	--

The analytical data set for the 100-BC-5 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 10,136 records were obtained from HEIS, and 120 analytes were included in the dataset prior to analytical data processing. After analytical data processing (as described in the next section), the final data set used for the COPC identification process contained 6,355 records, with 120 analytes included in the data set.

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

The analytical data were processed using the steps described below 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 6-6.

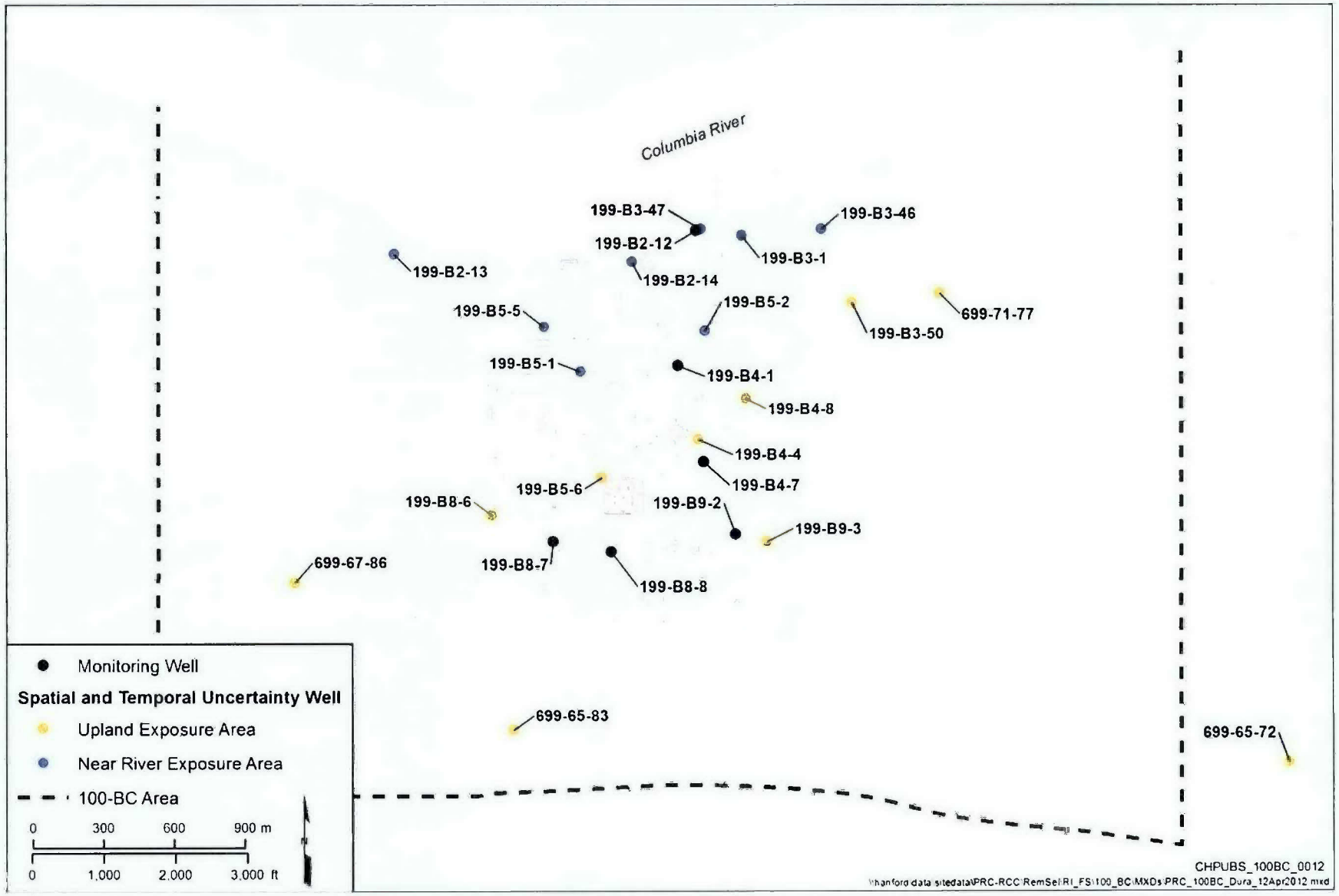

Figure 6-5. Monitoring Well Locations within the 100-BC-5 Groundwater OU

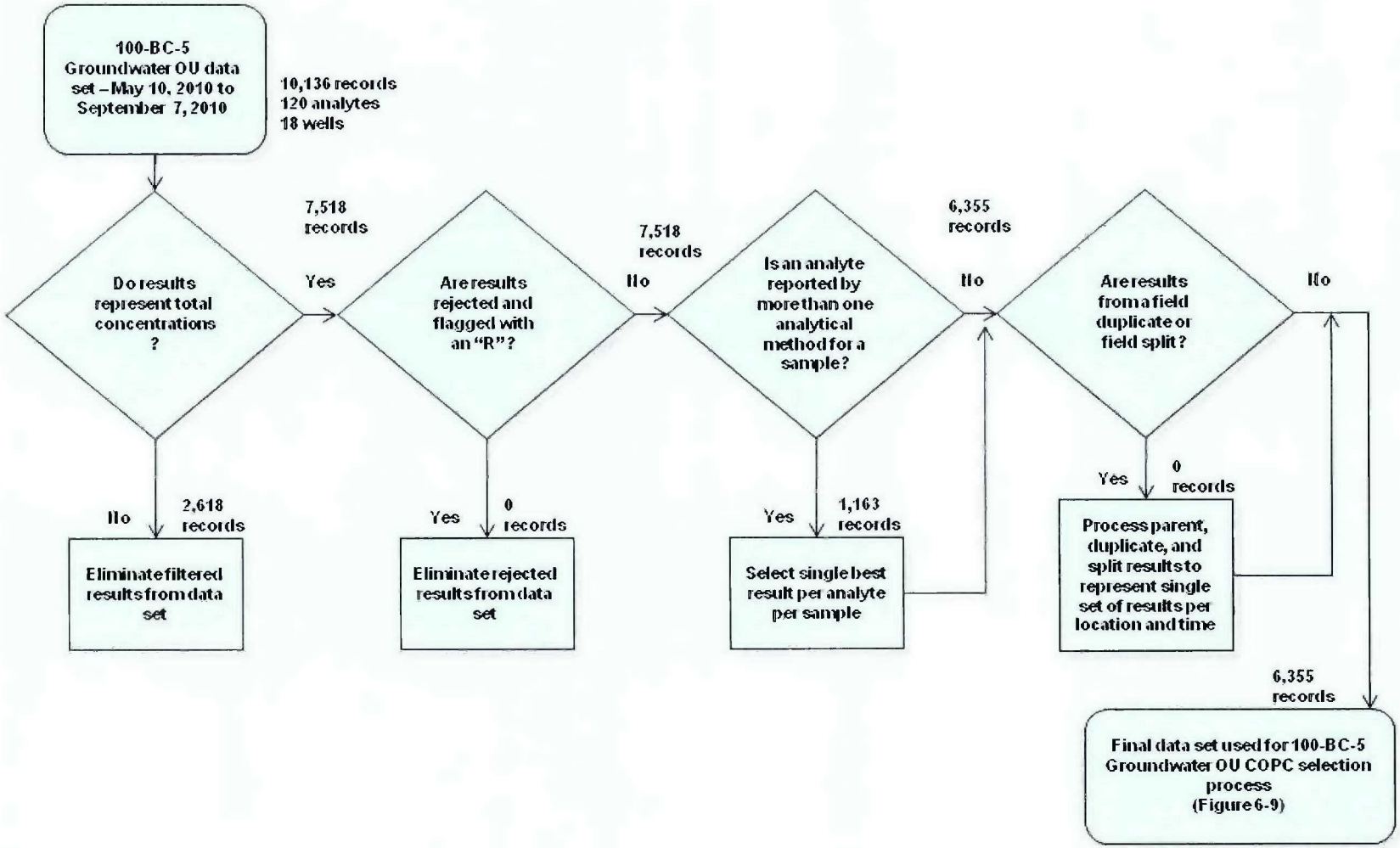

Figure 6-6. Data Processing and Reduction Steps for the 100-BC-5 Groundwater OU Dataset

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 (e.g., 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 ground-water 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.*

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 data 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 (i.e., 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 (i.e., 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 *Test Methods for Evaluating Solid Waste: Physical/Chemical Methods, Third Edition;*

Final Update IV-B [SW-846] with an EQL of 50 µg/L. For a sample with lead concentrations reported using both methods, the result reported by EPA Method 200.8 is selected over EPA Method 6010 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 (i.e., monitoring well) on the same date, resulting in more than one sample per location/date. The following criteria are used to reduce multiple sample results for an individual location/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

Action levels are derived from available sources of chemical-specific ARARs and default exposure assumptions. All sources of chemical-specific ARARs for each of the 120 analytes reported in the HEIS database for the 100-BC-5 OU are identified in Table 6-19.

For the COPC identification process, the action level is the lowest of the available chemical-specific ARARs for protection of human health and aquatic receptors.

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Table 6-19. Summary of Federal and State Water Quality Criteria Used as Action Levels for the 100-BC-5 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#	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)

Table 6-19. Summary of Federal and State Water Quality Criteria Used as Action Levels for the 100-BC-5 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#	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
78-87-5	1,2-Dichloropropane	--	µg/L	5.0	--	--	--	1.2	--	--	0.50	--	--	--	--	44	0.50	Clean Water Act -- Human Health Water + Organism
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)
75-05-8	Acetonitrile	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--
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 – Secondary 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

Table 6-19. Summary of Federal and State Water Quality Criteria Used as Action Levels for the 100-BC-5 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#	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
14234-35-6	Antimony-125	--	pCi/L	300	--	--	--	--	--	--	--	--	--	--	--	--	300	40 CFR 141 – Primary Federal MCL
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
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
13966-02-4	Beryllium-7	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--
7440-69-9	Bismuth	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--
7440-42-8	Boron	Boron And Borates Only	µg/L	--	--	--	--	3,200	--	--	--	--	--	--	--	--	3,200	WAC 173-340-720(4)(b)(iii)(A) and (B)
24959-67-9	Bromide	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--
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
7440-70-2	Calcium	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--

Table 6-19. Summary of Federal and State Water Quality Criteria Used as Action Levels for the 100-BC-5 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#	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
75-15-0	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
14762-75-5	Carbon-14	--	pCi/L	2,000	--	--	--	--	--	--	--	--	--	--	--	--	2,000	40 CFR 141 - Federal MCL
13967-70-9	Cesium-134	--	pCi/L	80	--	--	--	--	--	--	--	--	--	--	--	--	80	40 CFR 141 - Federal MCL
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)
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)

Table 6-19. Summary of Federal and State Water Quality Criteria Used as Action Levels for the 100-BC-5 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#	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
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)
107-12-0	Ethyl cyanide	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--
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
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	--	--	--	--	--	--	--	--	960	WAC 173-340-720(4)(b)(iii)(A) and (B)
12587-46-1	Gross alpha	--	pCi/L	15	--	--	--	--	--	--	--	--	--	--	--	--	15	40 CFR 141 - Federal MCL
12587-47-2	Gross beta	--	mrem/year	4.0	--	--	--	--	--	--	--	--	--	--	--	--	4.0	40 CFR 141 - Primary Federal MCL

Table 6-19. Summary of Federal and State Water Quality Criteria Used as Action Levels for the 100-BC-5 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#	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
18540-29-9	Hexavalent Chromium	Chromium (VI)	µg/L	--	--	--	--	48	16	11	--	10	15	10	--	486	10	40 CFR 131 -- Freshwater CCC
15046-84-1	Iodine-129	--	pCi/L	1.0	--	--	--	--	--	--	--	--	--	--	--	--	1.0	40 CFR 141 - Primary Federal MCL
74-88-4	Iodomethane	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--
7439-89-6	Iron	--	µg/L	300	--	300	--	11,200	--	1,000	300	--	--	--	--	9,074	300	40 CFR 141 - Secondary Federal 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)
7439-95-4	Magnesium	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--
7439-96-5	Manganese	Manganese (Water)	µg/L	50	--	50	--	2,240	--	--	50	--	--	--	--	907	50	40 CFR 141 – Seocndary Federal MCL
7439-97-6	Mercury	Mercurie 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)
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

Table 6-19. Summary of Federal and State Water Quality Criteria Used as Action Levels for the 100-BC-5 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#	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
13981-37-8	Nickel-63	--	pCi/L	50	--	--	--	--	--	--	--	--	--	--	--	--	50	40 CFR 141 - Primary Federal MCL
14681-63-1	Niobium-94	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--
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	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--
7440-09-7	Potassium	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--
13966-00-2	Potassium-40	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--
15262-20-1	Radium-228	--	pCi/L	5.0	--	--	--	--	--	--	--	--	--	--	--	--	5.0	40 CFR 141 - Primary Federal MCL
13967-48-1	Ruthenium-106	--	pCi/L	30	--	--	--	--	--	--	--	--	--	--	--	--	30	40 CFR 141 - Primary Federal MCL
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
7440-21-3	Silicon	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--
7440-22-4	Silver	--	µg/L	100	--	100	--	80	3.2	--	--	2.6	3.4	--	--	25,926	2.6	WAC 173-201A
7440-23-5	Sodium	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--
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

Table 6-19. Summary of Federal and State Water Quality Criteria Used as Action Levels for the 100-BC-5 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#	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
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)
109-99-9	Tetrahydrofuran	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--
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
7440-31-5	Tin	--	µg/L	--	--	--	--	9,600	--	--	--	--	--	--	--	519	519	WAC 173-340-730(3)(b)(iii)(A) and (B)
15832-50-5	Tin-126	--	pCi/L	--	--	--	--	--	--	--	--	--	--	--	--	--	--	
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)
ALPHA-RA	Total alpha energy emitted from Radium	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--
TPH-Diesel	Total petroleum hydrocarbons - diesel range	--	--	--	--	--	500	--	--	--	--	--	--	--	--	--	--	WAC 173-340-900, Table 720-1
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 (TCE)	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	Trichloromonofluoromethane	Trichlorofluoromethane	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 - Federal MCL

Table 6-19. Summary of Federal and State Water Quality Criteria Used as Action Levels for the 100-BC-5 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#	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
7440-61-1	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

Source: Ecology, 2007, The Model Toxics Control Act Cleanup Regulation Chapter 173-340 WAC, Publication No. 94-06, amended 1996, Washington State Department of Ecology, Olympia, Washington.

a. 40 CFR 141 National Recommended Water Quality Criteria. http://www.epa.gov/waterscience/criteria/wqctable/

b. Washington Department of Ecology, 2008, *Group A Public Water Supplies 246-290-310 WAC*, Publication No. 08-03-061

c. WAC 173-340-720(4)(b)(ii)(A) and (B), Ground water cleanup standards, Method B ground water cleanup levels, Noncarcinogens and Carcinogens

d. *Clean Water Act of 1977*

e. WAC 173-201A, "Water Quality Standards for Surface Waters of the State of Washington," *Washington Administrative Code*, Olympia, Washington. http://apps.leg.wa.gov/wac/default.aspx?cite=173-201A.

f. 40 Code of Federal Regulations 131 Water Quality Standards, http://ecfr.gpoaccess.gov

g. WAC 173-340-730(3)(b)(iii)(A) and (B). Surface water cleanup standards, Method B surface water cleanup levels, Noncarcinogens and Carcinogens

Note that 40 CFR 131, "Water Quality Standards", National Recommended Water Quality Criteria, 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 the Columbia River.

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6.3.2.2.1 ARAR-Based Remediation Goals.

The sources of the chemical-specific ARARs from federal regulations are:

- "National Primary Drinking Water Regulations" (40 CFR 141), MCLs, secondary MCLs, and nonzero MCLGs established under the SDWA
- *National Recommended Water Quality Criteria (EPA, 2009a)*, AWQC established under Section 304 of the *Clean Water Act of 1977*
- 40 CFR 131 water quality standards for states not complying with Section 303 of the *Clean Water Act of 1977*

The sources of the chemical-specific ARARs 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)

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 Report* [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 Report* [ECF-100NPL-10-0463]).

6.3.2.3 COPC Identification Process

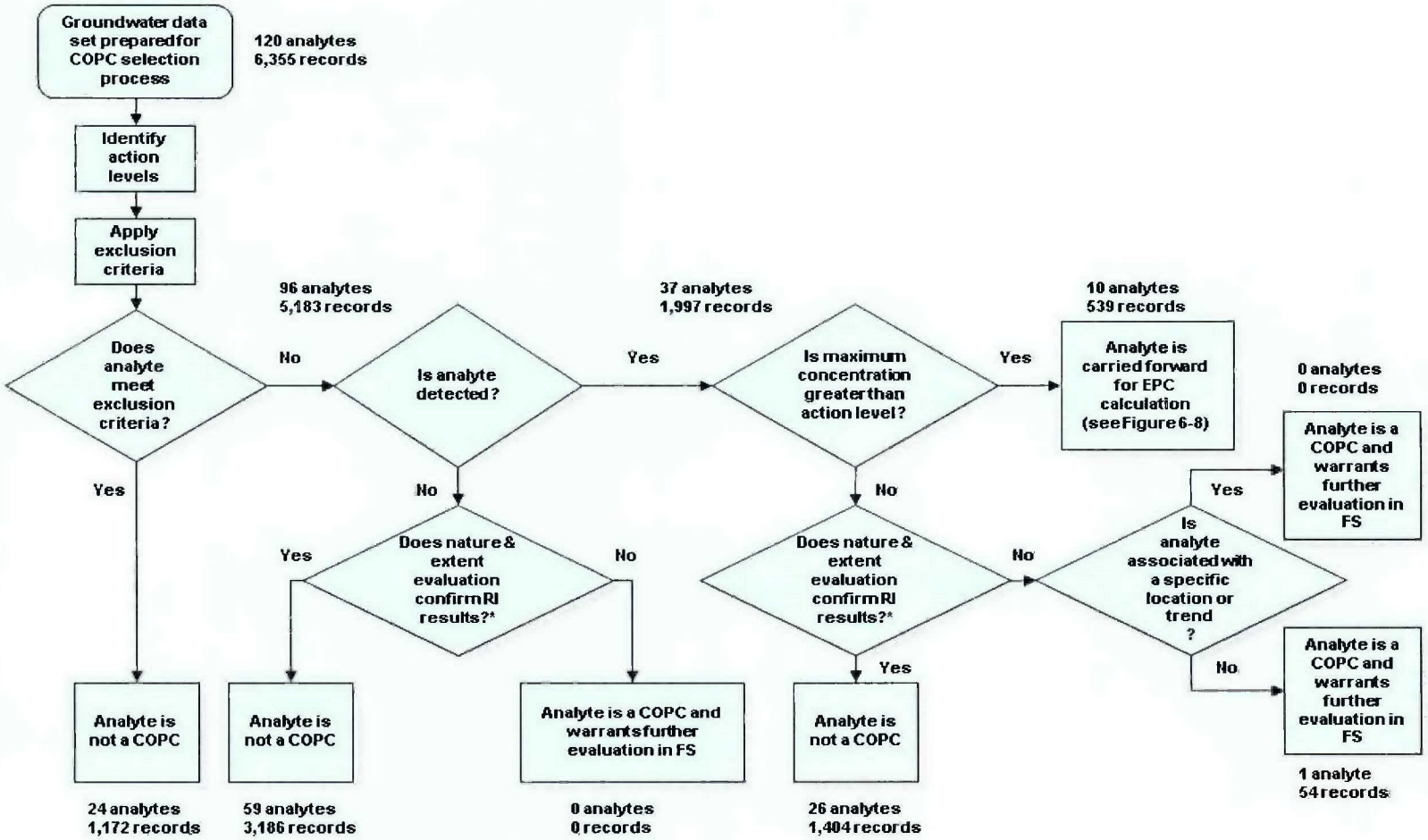
Section 6.3.2.1 defined the analytical data set and described the analytical data processing steps used in this section for identifying groundwater COPCs. Section 6.3.2.2 identified the action levels used in this section for identifying groundwater COPCs. The COPC identification process described in 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 18 monitoring wells in the 100-BC-5 Groundwater OU. The purpose of grouping all sampling and analysis data together from each exposure area (that is, the Near-River and Upland exposure areas) 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-BC-5 Groundwater Operable Unit* (ECF-100BC5-11-0015) in Appendix G. The COPC identification steps, number of records, and number of analytes associated with each step are depicted on Figure 6-7 and Figure 6-8 for the 100-BC-5 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 4.3.1 presents the nature and extent of contamination in groundwater based on the last five years of data (that is, samples collected between January 2006 and January 2011). All monitoring wells within the 100-BC-5 Groundwater OU that are screened in the unconfined aquifer were included in the nature and extent evaluation (see Figure 6-5). The nature and extent evaluation reviewed all analytes identified as historical COPCs. As described previously, historical COPCs (Table 6-17) were identified in the 100-BCWork Plan (DOE/RL-2008-46-ADD3) using data collected over a 16-year period (1992 to 2008). The nature and extent evaluation also includes the review of analytes that were not identified as historical COPCs, but were reported with 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 Report (DOE/RL-2007-21) and the 100-BC Work Plan (DOE/RL-2008-46-ADD3). 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 time frame.



* RI results = results from three rounds of spatial and temporal sampling from monitoring well network defined in DOE/RL-2009-44.

Figure 6-7. COPC Identification Process for the 100-BC-5 Groundwater OU: Part 1

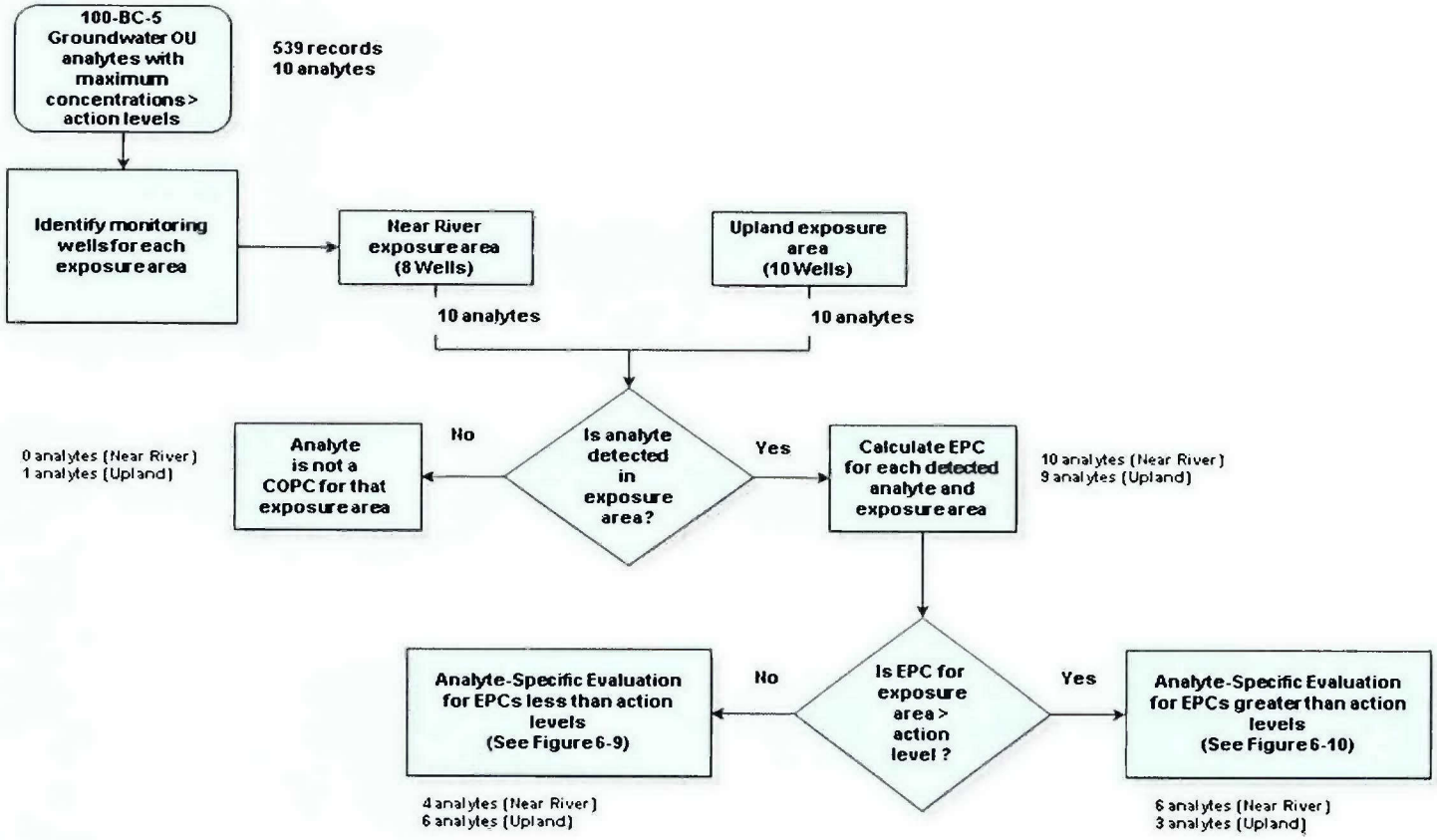


Figure 6-8. COPC Identification Process for the 100-BC-5 Groundwater OU: Part 2

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 6-20. Analytes that did not meet any of the exclusion criteria were carried forward into the next step. The exclusion criteria are:

- Naturally occurring radionuclides associated with background radiation
- Radionuclides that have half-lives of less than three 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-BC-5 Groundwater OU and is 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 activity remains after 30 years of decay. A total of four radioisotopes meet this exclusion criterion (antimony-125, beryllium-7, cesium-134, and ruthenium-106) and are 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-BC-5 OU, but are excluded from further consideration as COPCs.

Analytes without a promulgated chemical-specific ARAR were identified in Table 6-19. Because of the lack of promulgated standards, these analytes were not evaluated herein as 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. Fifteen analytes were eliminated from further consideration as COPCs because they do not have an action level nor do they have available toxicological information.

The federal MCL for gross beta is based on 4 mrem/yr annual dose and is used to indicate the presence of a group of beta-emitters. Although this standard is available, it requires a conversion from an activity level (pCi/L) to an annual dose rate (mrem/yr). Rather, beta-emitting radioisotopes such as strontium-90, technetium-99, and tritium are compared to their isotope specific standard, which is based on a 4 mrem/yr annual dose and is considered more protective than the overall standard for gross beta. It should be noted that the gross beta standard provides a measure for the daughter products of uranium, which are beta emitters. Although the 4 mrem/yr standard for gross beta is available, it is not carried further in the COPC identification process.

6.3.2.3.3 Identify Nondetected Analytes

The next step in the groundwater COPC identification process is 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 59 analytes were not detected in the 100-BC-5 OU groundwater samples collected for the RI. These analytes are listed in Table 6-21, each with sampling dates, minimum and maximum MDLs, the action level, the basis of the action level, and the level of exceedance.

1,1,2,2-Tetrachloroethane, 1,1-dichloroethene, antimony, benzene, cadmium, carbon tetrachloride, carbon-14, cobalt-60, europium-155, iodine-129, mercury, nickel-63, tetrachloroethene and vinyl chloride were identified as historical COPCs in the100-BC SAP (DOE/RL-2009-44), but are not detected in the samples collected for this RI. Antimony and cadmium are discussed below. 1,1,2,2-Tetrachloroethane, benzene, carbontetrachloride, carbon-14, cobalt-60, europium-155, iodine-129, mercury, nickel-63, tetrachloroethene, and vinyl chloride were not detected in samples collected specifically for this RI and data from the larger population of monitoring wells described previously in Section 4.3.1 and Section 6.3.2.3.1 were not available. 1,1-Dichloroethene was not detected in samples collected specifically for this RI, nor was it detected in the larger population of monitoring wells described previously in Section 4.3.1 and Section 6.3.2.3.1. With the exception of antimony, all MDLs associated with these analytes were less than the action level or the EQL (as applicable) listed in the 100-BC SAP (DOE/RL-2009-44).

Antimony was not detected in any of the unfiltered or filtered groundwater samples collected for the RI, nor was it detected in the larger population of monitoring wells described previously in Section 4.3.1 and Section 6.3.2.3.1.. Some MDLs greater than the action level were reported from the larger population of wells sampled in the past 5 years and reported by Method 6010 (SW 846 [SW-846]). Antimony results reported by Method 6010 (SW 846 [SW-846]) are considered uncertain because this method cannot accurately report trace concentrations. However, all MDLs associated with samples collected for the RI were less than the action level of 5.6 µg/L. The results of this evaluation indicate that antimony is not a COPC and will not be carried forward into the FS.

With the exception of one filtered cadmium result, cadmium was not detected in any samples collected for the RI, nor was it detected in the larger population of monitoring wells described previously in Section 4.3.1 and Section 6.3.2.3.1. Cadmium was detected once in 199-B3-47 at a concentration of 0.055 µg/L, which is less than the action level of 0.25 µg/L. Cadmium is not a COPC and will not be carried forward into the FS.

6.3.2.3.4 Identify Analytes with Maximum Detected Concentrations Less than Action Levels

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 in order 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 described below indicates otherwise.

Twenty-seven analytes were detected at least once and had maximum detected concentrations less than their respective action levels. Beryllium, cesium-137, cobalt, copper, lead, manganese, nickel, nitrate, selenium, technetium-99, total petroleum hydrocarbon (TPH)-diesel range, uranium, and zinc were identified as historic COPCs in the 100-BC Work Plan (DOE/RL-2008-46-ADD3). A list of these analytes is presented in Table 6-22, each with sampling dates, minimum and maximum MDLs, minimum and maximum detected concentrations, the action levels, and the basis for each action level.

Beryllium, copper, manganese, nickel, nitrate, and zinc were detected in groundwater samples collected for the RI at concentrations below their respective action level or EQL (as applicable) and in the larger population of monitoring wells described previously in Section 4.3.1 and Section 6.3.2.3. Cesium-137, lead, technetium-99, TPH-diesel range, and uranium were detected in groundwater samples collected for

the RI at concentrations below their respective action levels or EQLs (as applicable), and data from the larger population of monitoring wells described previously in Section 4.3.1 and Section 6.3.2.3.1 were not available. Therefore, beryllium, cesium-137, copper, lead, manganese, nickel, nitrate, technetium-99, TPH-diesel range, uranium, and zinc are not COPCs and will not be carried forward into the FS.

Detections of selenium above the action level were from the larger population of wells sampled in the past fiveyears as described previously in Section 4.3.1 and Section 6.3.2.3.1. Selenium was detected in a single filtered sample at a concentration greater than the action level of 5 µg/L (8.3 µg/L at 199-B3-50) and was flagged with a "B" qualifier. The corresponding unfiltered result was less than the action level. In addition, all selenium concentrations are less than the Hanford Site 90th percentile (filtered) background level of 11 µg/L. The results of this evaluation indicate that selenium is not identified as a COPC and will not be carried forward into the FS.

Detections of cobalt above the action level were from the larger population of wells sampled in the past fiveyears as described previously in Section 4.3.1 and Section 6.3.2.3.1.. Cobalt results (detected concentrations and MDLs) reported by Method 6010 (SW 846 [SW-846]) are generally greater than the action level in samples that were not collected for the RI. Groundwater samples analyzed by Method 6010 (SW 846 [SW-846]) generally report MDLs greater than the action level, resulting in nondetected concentrations greater than the action level. Cobalt results (detected concentrations and MDLs) reported by Method 6010 are considered uncertain because this method cannot accurately report trace concentrations. Six of the eight detected concentrations from Method 6010 were either 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. Although all cobalt results (detected concentrations and MDLs) were less than the action level in RI samples analyzed by trace methods, its historical presence with infrequent detections above the action level result in an uncertain status. Therefore, cobalt is considered a COPC and it warrants further evaluation in the FS.

6.3.2.3.5 Identify Analytes with Maximum Detected Concentrations Greater than Action Levels

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.

Ten 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 6-23, each with sampling dates, minimum and maximum MDLs, minimum and maximum detected concentrations, the action level, and the basis of the action level.

Table 6-20. Summary of Groundwater Analytes that Meet Exclusion Criteria in the 100-BC-5 Groundwater OU

Analyte Name	Analyte Class	Begin Sampling Date	End Sampling Date	Total Samples	Total Detects	Frequency of Detects	Units	Minimum Detection Limit	Maximum Detection Limit	Minimum Detected Result	Maximum Detected Result	Basis for Exclusion
Bromide	ANION	5/10/2010	9/7/2010	54	8	14.81%	µg/L	90	110	91	220	No Action Level/ No Toxicity Value
Phosphate	ANION	5/10/2010	9/7/2010	54	0	0.00%	µg/L	166	429	--	--	No Action Level/ No Toxicity Value
Bismuth	METAL	5/10/2010	9/7/2010	54	4	7.41%	µg/L	23	37	23	45	No Action Level/ No Toxicity Value
Calcium	METAL	5/10/2010	9/7/2010	54	54	100.00%	µg/L	--	--	28,900	79,900	Essential Nutrient
Magnesium	METAL	5/10/2010	9/7/2010	54	54	100.00%	µg/L	--	--	6,340	13,000	Essential Nutrient
Potassium	METAL	5/10/2010	9/7/2010	54	54	100.00%	µg/L	--	--	2,010	7,950	Essential Nutrient
Silicon	METAL	5/10/2010	9/7/2010	54	54	100.00%	µg/L	--	--	7,310	24,700	No Action Level/ No Toxicity Value
Sodium	METAL	5/10/2010	9/7/2010	54	54	100.00%	µg/L	--	--	7,270	31,000	Essential Nutrient
Antimony-125	RAD	5/10/2010	9/7/2010	54	0	0.00%	pCi/L	-4.73E+00	4.2	--	--	Half-Life less than 3 years
Beryllium-7	RAD	5/10/2010	9/7/2010	54	0	0.00%	pCi/L	-3.44E+01	49	--	--	Half-Life less than 3 years
Cesium-134	RAD	5/10/2010	9/7/2010	54	0	0.00%	pCi/L	-1.41E+00	2.4	--	--	Half-Life less than 3 years
Gross beta	RAD	5/10/2010	9/7/2010	54	52	96.30%	pCi/L	2.7	3.2	3.6	110	Action Level in units of mrem/yr

Table 6-20. Summary of Groundwater Analytes that Meet Exclusion Criteria in the 100-BC-5 Groundwater OU

Analyte Name	Analyte Class	Begin Sampling Date	End Sampling Date	Total Samples	Total Detects	Frequency of Detects	Units	Minimum Detection Limit	Maximum Detection Limit	Minimum Detected Result	Maximum Detected Result	Basis for Exclusion
Niobium-94	RAD	7/27/2010	7/27/2010	1	0	0.00%	pCi/L	--	--	--	--	No Action Level/ No Toxicity Value
Potassium-40	RAD	5/10/2010	9/7/2010	54	1	1.85%	pCi/L	-7.54E+01	29	40	40	Background Radiation
Ruthenium-106	RAD	5/10/2010	9/7/2010	54	0	0.00%	pCi/L	-1.18E+01	30	--	--	Half-Life less than 3 years
Tin-126	RAD	7/27/2010	7/27/2010	1	0	0.00%	pCi/L	17	17	--	--	No Action Level/ No Toxicity Value
Total alpha energy emitted from Radium	RAD	5/10/2010	8/12/2010	36	0	0.00%	pCi/L	-7.20E-02	0.13	--	--	No Action Level/ No Toxicity Value
Acetonitrile	VOC	5/10/2010	9/7/2010	54	0	0.00%	µg/L	2.0	2.0	--	--	No Action Level/ No Toxicity Value
Chloroethane	VOC	5/10/2010	9/7/2010	54	0	0.00%	µg/L	0.099	0.099	--	--	No Action Level/ No Toxicity Value
Chloromethane	VOC	5/10/2010	9/7/2010	54	6	11.11%	µg/L	0.077	0.077	0.10	0.31	No Action Level/ No Toxicity Value
Ethyl cyanide	VOC	5/10/2010	9/7/2010	54	0	0.00%	µg/L	1.4	1.4	--	--	No Action Level/ No Toxicity Value
Iodomethane	VOC	5/10/2010	9/7/2010	54	3	5.56%	µg/L	0.092	0.092	0.22	0.66	No Action Level/ No Toxicity Value

Table 6-20. Summary of Groundwater Analytes that Meet Exclusion Criteria in the 100-BC-5 Groundwater OU

Analyte Name	Analyte Class	Begin Sampling Date	End Sampling Date	Total Samples	Total Detects	Frequency of Detects	Units	Minimum Detection Limit	Maximum Detection Limit	Minimum Detected Result	Maximum Detected Result	Basis for Exclusion
Tetrahydrofuran	VOC	5/10/2010	9/7/2010	54	0	0.00%	µg/L	1.1	1.1	--	--	No Action Level/ No Toxicity Value
Trans-1,4-Dichloro-2-butene	VOC	5/10/2010	9/7/2010	54	0	0.00%	µg/L	0.29	0.29	--	--	No Action Level/ No Toxicity Value

Table 6-21. Summary of Analytes that Were Not Detected in the 100-BC-5 Groundwater OU

Analyte Name	Analyte Class	Begin Sample Date	End Sample Date	Total Samples	Total Detects	Frequency of Detects	Units	Minimum Detection Limit	Maximum Detection Limit	Action Level	Action Level Basis	Level of Exceedance
Nitrite	ANION	5/10/2010	9/7/2010	54	0	0%	µg/L	9.9	118	3,300	40 CFR 141 - Primary Federal MCL	2.98E-03
Antimony	METAL	5/10/2010	9/7/2010	54	0	0%	µg/L	0.30	1.1	5.6	Clean Water Act -- Human Health Water + Organism	5.36E-02
Cadmium	METAL	5/10/2010	9/7/2010	54	0	0%	µg/L	0.055	0.20	0.25	Clean Water Act -- Freshwater CCC	2.20E-01
Mercury	METAL	5/10/2010	9/7/2010	54	0	0%	µg/L	0.050	0.10	0.012	40 CFR 131 -- Freshwater CCC	4.17E+00
Silver	METAL	5/10/2010	9/7/2010	54	0	0%	µg/L	0.040	0.20	2.6	WAC 173-201A	1.53E-02
Carbon-14	RAD	5/10/2010	9/7/2010	54	0	0%	pCi/L	-4.97	7.6	2,000	40 CFR 141 - Primary Federal MCL	-2.49E-03
Cobalt-60	RAD	5/10/2010	9/7/2010	54	0	0%	pCi/L	-2.11	2.4	100	40 CFR 141 - Primary Federal MCL	-2.11E-02
Europium-152	RAD	5/10/2010	9/7/2010	54	0	0%	pCi/L	-7.10	5.2	200	40 CFR 141 - Primary Federal MCL	-3.55E-02
Europium-154	RAD	5/10/2010	9/7/2010	54	0	0%	pCi/L	-4.42	4.4	60	40 CFR 141 - Primary Federal MCL	-7.37E-02
Europium-155	RAD	5/10/2010	9/7/2010	54	0	0%	pCi/L	-3.15	2.5	600	40 CFR 141 - Primary Federal MCL	-5.25E-03
Iodine-129	RAD	5/10/2010	9/7/2010	54	0	0%	pCi/L	-0.33	0.24	1.0	40 CFR 141 - Primary Federal MCL	-3.25E-01
Nickel-63	RAD	5/10/2010	9/7/2010	54	0	0%	pCi/L	-3.42	3.2	50	40 CFR 141 - Primary Federal MCL	-6.84E-02
Radium-228	RAD	5/10/2010	9/7/2010	54	0	0%	pCi/L	-4.04	1.7	5.0	40 CFR 141 - Primary Federal MCL	-8.08E-01
1,4-Dichlorobenzene	SVOC	5/10/2010	9/7/2010	54	0	0%	µg/L	0.12	0.12	8.1	WAC 173-340-720(4)(b)(iii)(A) and (B)	1.48E-02
1,1,1,2-Tetrachloroethane	VOC	5/10/2010	9/7/2010	54	0	0%	µg/L	0.090	0.090	1.7	WAC 173-340-720(4)(b)(iii)(A) and (B)	5.36E-02
1,1,1-Trichloroethane	VOC	5/10/2010	9/7/2010	54	0	0%	µg/L	0.069	0.069	200	40 CFR 141 - Primary Federal MCL	3.45E-04
1,1,2,2-Tetrachloroethane	VOC	5/10/2010	9/7/2010	54	0	0%	µg/L	0.098	0.098	0.17	Clean Water Act -- Human Health Water + Organism	5.76E-01
1,1,2-Trichloroethane	VOC	5/10/2010	9/7/2010	54	0	0%	µg/L	0.15	0.15	0.59	Clean Water Act -- Human Health Water + Organism	2.54E-01
1,1-Dichloroethane	VOC	5/10/2010	9/7/2010	54	0	0%	µg/L	0.068	0.068	1,600	WAC 173-340-720(4)(b)(iii)(A) and (B)	4.25E-05
1,1-Dichloroethene	VOC	5/10/2010	9/7/2010	54	0	0%	µg/L	0.083	0.083	0.057	40 CFR 131 -- Human Health Water + Organism	1.46E+00
1,2,3-Trichloropropane	VOC	5/10/2010	9/7/2010	54	0	0%	µg/L	0.15	0.15	0.0015	WAC 173-340-720(4)(b)(iii)(A) and (B)	1.03E+02
1,2-Dibromo-3-chloropropane	VOC	5/10/2010	9/7/2010	54	0	0%	µg/L	0.41	0.41	0.055	WAC 173-340-720(4)(b)(iii)(A) and (B)	7.50E+00
1,2-Dibromoethane	VOC	5/10/2010	9/7/2010	54	0	0%	µg/L	0.13	0.13	0.022	WAC 173-340-720(4)(b)(iii)(A) and (B)	5.94E+00
1,2-Dichloroethane	VOC	5/10/2010	9/7/2010	54	0	0%	µg/L	0.10	0.10	0.38	Clean Water Act -- Human Health Water + Organism	2.63E-01
1,2-Dichloroethene (Total)	VOC	5/10/2010	9/7/2010	54	0	0%	µg/L	0.15	0.15	72	WAC 173-340-720(4)(b)(iii)(A) and (B)	2.08E-03
1,2-Dichloropropane	VOC	5/10/2010	9/7/2010	54	0	0%	µg/L	0.097	0.097	0.50	Clean Water Act -- Human Health Water + Organism	1.94E-01
1,4-Dioxane	VOC	5/10/2010	9/7/2010	54	0	0%	µg/L	7.6	7.6	4.0	WAC 173-340-720(4)(b)(iii)(A) and (B)	1.91E+00
1-Butanol	VOC	5/10/2010	9/7/2010	54	0	0%	µg/L	12	12	800	WAC 173-340-720(4)(b)(iii)(A) and (B)	1.50E-02

Table 6-21. Summary of Analytes that Were Not Detected in the 100-BC-5 Groundwater OU

Analyte Name	Analyte Class	Begin Sample Date	End Sample Date	Total Samples	Total Detects	Frequency of Detects	Units	Minimum Detection Limit	Maximum Detection Limit	Action Level	Action Level Basis	Level of Exceedance
2-Butanone	VOC	5/10/2010	9/7/2010	54	0	0%	µg/L	0.52	0.52	4,800	WAC 173-340-720(4)(b)(iii)(A) and (B)	1.08E-04
2-Hexanone	VOC	5/10/2010	9/7/2010	54	0	0%	µg/L	0.22	0.22	80	WAC 173-340-720(4)(b)(iii)(A) and (B)	2.75E-03
4-Methyl-2-pentanone	VOC	5/10/2010	9/7/2010	54	0	0%	µg/L	0.12	0.12	640	WAC 173-340-720(4)(b)(iii)(A) and (B)	1.88E-04
Acrolein	VOC	5/10/2010	9/7/2010	54	0	0%	µg/L	2.8	2.8	3.0	Clean Water Act -- Freshwater CCC	9.33E-01
Allyl chloride	VOC	5/10/2010	9/7/2010	54	0	0%	µg/L	0.11	0.11	2.1	WAC 173-340-720(4)(b)(iii)(A) and (B)	5.29E-02
Benzene	VOC	5/10/2010	9/7/2010	54	0	0%	µg/L	0.064	0.064	0.80	WAC 173-340-720(4)(b)(iii)(A) and (B)	8.04E-02
Bromodichloromethane	VOC	5/10/2010	9/7/2010	54	0	0%	µg/L	0.088	0.088	0.27	40 CFR 131 -- Human Health Water + Organism	3.26E-01
Bromoform	VOC	5/10/2010	9/7/2010	54	0	0%	µg/L	0.17	0.17	4.3	Clean Water Act -- Human Health Water + Organism	3.95E-02
Carbon tetrachloride	VOC	5/10/2010	9/7/2010	54	0	0%	µg/L	0.12	0.12	0.23	Clean Water Act -- Human Health Water + Organism	5.22E-01
Chlorobenzene	VOC	5/10/2010	9/7/2010	54	0	0%	µg/L	0.15	0.15	100	40 CFR 141 - Primary Federal MCL	1.50E-03
Chloroprene	VOC	5/10/2010	9/7/2010	54	0	0%	µg/L	0.097	0.097	160	WAC 173-340-720(4)(b)(iii)(A) and (B)	6.06E-04
cis-1,2-Dichloroethylene	VOC	5/10/2010	9/7/2010	54	0	0%	µg/L	0.087	0.087	70	40 CFR 141 - Primary Federal MCL	1.24E-03
cis-1,3-Dichloropropene	VOC	5/10/2010	9/7/2010	54	0	0%	µg/L	0.073	0.073	0.34	Clean Water Act -- Human Health Water + Organism	2.15E-01
Dibromochloromethane	VOC	5/10/2010	9/7/2010	54	0	0%	µg/L	0.13	0.13	0.40	Clean Water Act -- Human Health Water + Organism	3.25E-01
Dibromomethane	VOC	5/10/2010	9/7/2010	54	0	0%	µg/L	0.21	0.21	80	WAC 173-340-720(4)(b)(iii)(A) and (B)	2.63E-03
Dichlorodifluoromethane	VOC	5/10/2010	9/7/2010	54	0	0%	µg/L	0.084	0.084	1,600	WAC 173-340-720(4)(b)(iii)(A) and (B)	5.25E-05
Ethyl methacrylate	VOC	5/10/2010	9/7/2010	54	0	0%	µg/L	0.11	0.11	720	WAC 173-340-720(4)(b)(iii)(A) and (B)	1.53E-04
Ethylbenzene	VOC	5/10/2010	9/7/2010	54	0	0%	µg/L	0.086	0.086	4.0	WAC 173-340-720(4)(b)(iii)(A) and (B)	2.16E-02
Isobutyl alcohol	VOC	5/10/2010	9/7/2010	54	0	0%	µg/L	8.7	8.7	2,400	WAC 173-340-720(4)(b)(iii)(A) and (B)	3.63E-03
Methacrylonitrile	VOC	5/10/2010	9/7/2010	54	0	0%	µg/L	0.50	0.50	0.80	WAC 173-340-720(4)(b)(iii)(A) and (B)	6.25E-01
Methyl methacrylate	VOC	5/10/2010	9/7/2010	54	0	0%	µg/L	0.26	0.26	11,200	WAC 173-340-720(4)(b)(iii)(A) and (B)	2.32E-05
Methylene chloride	VOC	5/10/2010	9/7/2010	54	0	0%	µg/L	0.11	0.11	4.6	Clean Water Act -- Human Health Water + Organism	2.39E-02
Styrene	VOC	5/10/2010	9/7/2010	54	0	0%	µg/L	0.074	0.074	100	40 CFR 141 - Primary Federal MCL	7.40E-04
Tetrachloroethene	VOC	5/10/2010	9/7/2010	54	0	0%	µg/L	0.18	0.18	0.081	WAC 173-340-720(4)(b)(iii)(A) and (B)	2.22E+00
Toluene	VOC	5/10/2010	9/7/2010	54	0	0%	µg/L	0.072	0.072	640	WAC 173-340-720(4)(b)(iii)(A) and (B)	1.13E-04
trans-1,2-Dichloroethylene	VOC	5/10/2010	9/7/2010	54	0	0%	µg/L	0.083	0.083	100	40 CFR 141 - Primary Federal MCL	8.30E-04
trans-1,3-Dichloropropene	VOC	5/10/2010	9/7/2010	54	0	0%	µg/L	0.083	0.083	0.34	Clean Water Act -- Human Health Water + Organism	2.44E-01
Trichloromonofluoromethane	VOC	5/10/2010	9/7/2010	54	0	0%	µg/L	0.11	0.11	2,400	WAC 173-340-720(4)(b)(iii)(A) and (B)	4.58E-05

Table 6-21. Summary of Analytes that Were Not Detected in the 100-BC-5 Groundwater OU

Analyte Name	Analyte Class	Begin Sample Date	End Sample Date	Total Samples	Total Detects	Frequency of Detects	Units	Minimum Detection Limit	Maximum Detection Limit	Action Level	Action Level Basis	Level of Exceedance
Vinyl acetate	VOC	5/10/2010	9/7/2010	54	0	0%	µg/L	0.18	0.18	8,000	WAC 173-340-720(4)(b)(iii)(A) and (B)	2.25E-05
Vinyl chloride	VOC	5/10/2010	9/7/2010	54	0	0%	µg/L	0.084	0.084	0.025	Clean Water Act -- Human Health Water + Organism	3.36E+00
Xylenes (total)	VOC	5/10/2010	9/7/2010	54	0	0%	µg/L	0.20	0.20	1,600	WAC 173-340-720(4)(b)(iii)(A) and (B)	1.25E-04

Note: Shading indicates that an analyte is identified in the list of contaminants of potential concern in DOE/RL-2009-44, *Sampling and Analysis Plan for the 100-BC-1, 100-BC-2, 100-BC-5 OUs Remedial Investigation/Feasibility Study*.

Table 6-22. Summary of Groundwater Analytes that Do Not Exceed an Action Level in the 100-BC-5 Groundwater OU

Analyte Name	Analyte Class	Begin Sample Date	End Sample Date	Total Samples	Total Detects	Frequency of Detects	Units	Minimum Detection Limit	Maximum Detection Limit	Minimum Detected Result	Maximum Detected Result	Action Level	Action Level Basis
Chloride	ANION	5/10/2010	9/7/2010	54	54	100.00%	µg/L	--	--	2,350	25,700	230,000	Clean Water Act -- Freshwater CCC
Fluoride	ANION	5/10/2010	9/7/2010	54	42	77.78%	µg/L	60	88	66	290	960	WAC 173-340-720(4)(b)(iii)(A) and (B)
Nitrate	ANION	5/10/2010	9/7/2010	54	54	100.00%	µg/L	--	--	1,790	44,200	45,000	40 CFR 141 – Primary Federal MCL
Sulfate	ANION	5/10/2010	9/7/2010	54	54	100.00%	µg/L	--	--	19,700	70,500	250,000	40 CFR 141 – Primary Federal MCL
Barium	METAL	5/10/2010	9/7/2010	54	54	100.00%	µg/L	--	--	7.5	88	1,000	Clean Water Act -- Human Health Water + Organism
Beryllium	METAL	5/10/2010	9/7/2010	54	1	1.85%	µg/L	0.050	0.35	0.13	0.13	4.0	40 CFR 141 – Primary Federal MCL
Boron	METAL	5/10/2010	9/7/2010	54	2	3.70%	µg/L	19	41	19	131	3,200	WAC 173-340-720(4)(b)(iii)(A) and (B)
Cobalt	METAL	5/10/2010	9/7/2010	54	6	11.11%	µg/L	0.050	0.22	0.054	0.67	2.6	WAC 173-340-730(3)(b)(iii)(A) and (B)
Copper	METAL	5/10/2010	9/7/2010	54	23	42.59%	µg/L	0.20	0.20	0.12	0.92	9.0	Clean Water Act -- Freshwater CCC
Lead	METAL	5/10/2010	9/7/2010	54	4	7.41%	µg/L	0.10	0.20	0.18	0.32	2.1	WAC 173-201A
Lithium	METAL	5/10/2010	9/7/2010	54	37	68.52%	µg/L	4.0	4.0	4.0	10	32	WAC 173-340-720(4)(b)(iii)(A) and (B)
Manganese	METAL	5/10/2010	9/7/2010	54	7	12.96%	µg/L	3.3	6.0	4.0	41	50	40 CFR 141 – Secondary Federal MCL
Molybdenum	METAL	5/10/2010	9/7/2010	54	54	100.00%	µg/L	--	--	1.3	11	80	WAC 173-340-720(4)(b)(iii)(A) and (B)
Nickel	METAL	5/10/2010	9/7/2010	54	10	18.52%	µg/L	4.0	4.0	4.0	12	52	Clean Water Act -- Freshwater CCC
Selenium	METAL	5/10/2010	9/7/2010	54	44	81.48%	µg/L	0.60	0.60	0.58	4.2	5.0	Clean Water Act -- Freshwater CCC
Strontium	METAL	5/10/2010	9/7/2010	54	54	100.00%	µg/L	--	--	134	365	9,600	WAC 173-340-720(4)(b)(iii)(A) and (B)
Tin	METAL	5/10/2010	9/7/2010	54	2	3.70%	µg/L	0.050	0.10	0.066	0.25	519	WAC 173-340-730(3)(b)(iii)(A) and (B)
Uranium	METAL	5/10/2010	9/7/2010	54	54	100.00%	µg/L	--	--	1.1	7.3	30	40 CFR 141 – Primary Federal MCL
Vanadium	METAL	5/10/2010	9/7/2010	54	25	46.30%	µg/L	12	17	12	35	80	WAC 173-340-720(4)(b)(iii)(A) and (B)
Zinc	METAL	5/10/2010	9/7/2010	54	11	20.37%	µg/L	4.0	6.0	4.0	21	91	WAC 173-201A
Cesium-137	RAD	5/10/2010	9/7/2010	54	1	1.85%	pCi/L	-2.19E+00	1.7	2.0	2.0	200	40 CFR 141 – Primary Federal MCL
Gross alpha	RAD	5/10/2010	9/7/2010	54	8	14.81%	pCi/L	-3.20E+01	2.5	2.3	6.4	15	40 CFR 141 – Primary Federal MCL
Technetium-99	RAD	5/10/2010	9/7/2010	54	26	48.15%	pCi/L	-1.10E+01	6.1	6.2	26	900	40 CFR 141 – Primary Federal MCL
Total petroleum hydrocarbons - diesel range	TPH	5/10/2010	9/7/2010	54	2	3.70%	µg/L	17	70	180	220	500	WAC 173-340-900, Table 720-1
Acetone	VOC	5/10/2010	9/7/2010	54	1	1.85%	µg/L	0.34	0.34	1.0	1.0	7,200	WAC 173-340-720(4)(b)(iii)(A) and (B)
Bromomethane	VOC	5/10/2010	9/7/2010	54	16	29.63%	µg/L	0.13	0.25	0.15	0.74	11	WAC 173-340-720(4)(b)(iii)(A) and (B)
Carbon disulfide	VOC	5/10/2010	9/7/2010	54	1	1.85%	µg/L	0.051	0.051	0.074	0.074	800	WAC 173-340-720(4)(b)(iii)(A) and (B)

Table 6-22. Summary of Groundwater Analytes that Do Not Exceed an Action Level in the 100-BC-5 Groundwater OU

Analyte Name	Analyte Class	Begin Sample Date	End Sample Date	Total Samples	Total Detects	Frequency of Detects	Units	Minimum Detection Limit	Maximum Detection Limit	Minimum Detected Result	Maximum Detected Result	Action Level	Action Level Basis

Note: Shading indicates that an analyte is identified in the proposed list of contaminants of potential concern in DOE/RL-2009-44, *Sampling and Analysis Plan for the 100-BC-1, 100-BC-2, 100-BC-5 OUs Remedial Investigation/Feasibility Study*.

Table 6-23. Summary of Analytes that Exceed an Action Level in the 100-BC-5 Groundwater OU

Analyte Name	Analyte Class	Begin Sample Date	End Sample Date	Total Samples	Total Detects	Frequency of Detects	Units	Minimum Detection Limit	Maximum Detection Limit	Minimum Detected Result	Maximum Detected Result	Action Level	Action Level Basis
Aluminum	METAL	5/10/2010	9/7/2010	54	23	42.59%	µg/L	5.0	10	7.1	72	50	40 CFR 141 – Primary Federal MCL
Arsenic	METAL	5/10/2010	9/7/2010	54	51	94.44%	µg/L	0.80	0.80	0.82	3.9	0.018	Clean Water Act -- Human Health Water + Organism
Chromium	METAL	5/10/2010	9/7/2010	54	54	100.00%	µg/L	--	--	4.5	69	65	Clean Water Act -- Freshwater CCC
Hexavalent Chromium	METAL	5/10/2010	9/7/2010	53	48	90.57%	µg/L	2.0	2.0	2.4	57	10	40 CFR 131 -- Freshwater CCC
Iron	METAL	5/10/2010	9/7/2010	54	24	44.44%	µg/L	18	38	18	347	300	40 CFR 141 – Secondary Federal MCL
Thallium	METAL	5/10/2010	9/7/2010	54	1	1.85%	µg/L	0.050	0.10	1.0	1.0	0.24	Clean Water Act -- Human Health Water + Organism
Strontium-90	RAD	5/10/2010	9/7/2010	54	16	29.63%	pCi/L	-8.40E+00	1.8	3.1	49	8.0	40 CFR 141 – Primary Federal MCL
Tritium	RAD	5/10/2010	9/7/2010	54	47	87.04%	pCi/L	-1.20E+02	100	190	69,000	20,000	40 CFR 141 – Primary Federal MCL
Chloroform	VOC	5/10/2010	9/7/2010	54	42	77.78%	µg/L	0.10	0.10	0.11	3.0	1.4	WAC 173-340-720(4)(b)(iii)(A) and (B)
Trichloroethene	VOC	5/10/2010	9/7/2010	54	40	74.07%	µg/L	0.21	0.25	0.23	3.3	0.49	WAC 173-340-720(4)(b)(iii)(A) and (B)

Note: Shading indicates that an analyte is identified in the proposed list of contaminants of potential concern in DOE/RL-2009-44, *Sampling and Analysis Plan for the 100-BC-1, 100-BC-2, 100-BC-5 OUs Remedial Investigation/Feasibility Study*.

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6.3.2.3.6 Calculate EPCs for Each Analyte with Maximum Detected Concentrations Greater than Action Levels.

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 producing 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-BC-5 Groundwater Operable Unit* (ECF-100BC5-11-0016) (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-BC-5 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 below, 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 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; therefore, a 90th percentile concentration reflects an RME scenario.

Groundwater data sets 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 of the data set. The variability of the dataset is assessed in terms of the coefficient of variation (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 at the Hanford Site. Additional statistical evaluation of the 100-BC-5 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-BC-5 Groundwater Operable Unit* (ECF-100BC5-11-0016) (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 an (1) 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 6.3.9.2).

6.3.2.3.7 Identify Monitoring Wells in Each Exposure Area

Two exposure areas are identified for the 100-BC-5 Groundwater OU: (1) the Near-River exposure area (monitoring wells less than 500 m (1,640 ft) from the river shoreline) and (2) the Upland exposure area (monitoring wells greater than 500 m (1,640 ft) from the river shoreline). Table 6-18 lists the monitoring wells associated with each exposure area.

6.3.2.3.8 Identify Nondetected COPCs 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. All analytes detected at least once in an exposure area are carried forward to the next step of the process for that exposure area. All 10 analytes were detected at least once in the Near-River exposure area.
One analyte, thallium, was not detected in the Upland exposure area and therefore eliminated as a COPC.

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 chemical-specific ARARs for protection of human health and aquatic receptors (that is, action levels). Comparisons of EPCs to action levels for the Near-River exposure area and Upland exposure area are provided in Table 6-24 and Table 6-25, respectively. A flow chart depicting this comparison is provided on Figure 6-9.

Near-River Exposure Area. Four of the 10 analytes (aluminum, chromium, iron, and thallium) have been detected at least once in groundwater and have 90th percentile values less than their respective action levels (Table 6-24).

Two of the four analytes (chromium and thallium) were identified as historical COPCs in the 100-BC Work Plan (DOE/RL-2008-46-ADD3). A discussion of all analytes reported with an EPC less than the action level is provided below.

Aluminum was detected in groundwater samples collected for the RI and the EPC is less than the action level. Aluminum data from the larger population of monitoring wells sampled over the pastf iveyears, as described previously in Section 4.3.1 and Section 6.3.2.3.1, were not available. The action level for aluminum is based on the secondary MCL, which 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. All aluminum results (detected concentrations and MDLs) are less than the AWQC of 87 µg/L. Based on these results, aluminum is not a COPC and will not be carried forward into the risk characterization section.

Chromium was detected in groundwater samples collected for the RI and the EPC is less than the action level. Detections of chromium were less than the AWQC in the larger population of monitoring wells sampled over the past fiveyears as described previously in Section 4.3.1 and Section 6.3.2.3.1. Chromium was detected once (69 µg/L) above the AWQC at 199-B3-47; chromium concentrations between 2006 and 2010 ranged from 38 to 56 µg/L. It should be noted that Cr(VI) concentrations measured at 199-B3-47 are essentially the same as total chromium concentrations. As discussed below, Cr(VI) is identified as a COPC. Based on the results of this evaluation, chromium is not identified as a COPC and will not be carried forward into the risk characterization section.