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Hanford Site Baseline Risk Assessment Methodology

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HANFORD SITE BASELINE RISK ASSESSMENT METHODOLOGY

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PREFACE

The U.S. Department of Energy (DOE) Field Office, Richland (RL) has prepared the Hanford Site Baseline Risk Assessment Methodology (HSBRAM, DOE/RL-91-45) to satisfy the M-29-03 Milestone of the Hanford Federal Facility Agreement and Consent Order (Ecology et al. 1990), referred to as the Tri-Party Agreement (TPA). The following statements provide the position of DOE on issues related to the preparation of risk assessments at the Hanford Site. They are provided to limit the use of qualifying statements within the HSBRAM and to allow the disposition of regulatory comments in order to finalize the risk assessment methodology document and meet the March 1992 milestone date.

- Land Use - RL's position on current and future applicable land use scenarios is that the Hanford Site will remain under institutional control and RL will reserve control over beneficial use of the land and over any uses of groundwater for the foreseeable future. Use of the land is expected to remain industrial, to include waste disposal. However, the risk assessment methodology, at the request of the U.S. Environmental Protection Agency (EPA) and the Washington State Department of Ecology (Ecology), provides other scenarios of land use. Assessments of other on-site scenarios may be done on a case-by-case basis. However, DOE reserves the right to qualify the use of these assessments based on the position stated above.

- Model Toxics Control Act (MTCA) - Information in MTCA relating to risk assessment was incorporated into the risk assessment methodology. Cleanup standards set forth in MTCA may be relevant and appropriate standards and thus may constitute Applicable or Relevant and Appropriate Requirements (ARARs) as defined in Section 121 of the Comprehensive Environmental Response, Compensation, and Liability Act (CERCLA) (42 USC 6901 et seq.) not only for the Hanford Site but any National Priorities List (NPL) site in the State of Washington. However, a decision on what constitutes an ARAR is not finalized until a Record of Decision (ROD) is issued for an operable unit. As no RODs have been issued, RL will reserve the right to challenge MTCA as an ARAR. The NCP requires a baseline risk assessment be performed "to characterize the current and potential threats to human health and the environment" with the results used to "help establish acceptable exposure levels for use in developing remedial alternatives" (40 CFR 300.430(d)(4)). Developing the risk assessment methodology in the manner presented provides estimates of risks that are meaningful should they be evaluated in the context of requirements of the NCP or MTCA. Although the risk assessment methodology incorporates MTCA, the methodology is consistent with the NCP.

- Points of Compliance - RL's position is that points of compliance shall be based upon maintenance of institutional control of the site. For example, groundwater use is and will continue to be controlled or

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restricted on the Hanford Site, making the river the point of compliance for groundwater contaminants. For soil contaminants outside of disposal sites which will remain under institutional control, a depth of 15 feet, consistent with MTCA, will be the point of compliance. The point of compliance for air contaminants shall be at the edge of each operable unit.

- Time of Application of Future Scenarios - Risk assessments evaluate potential threats to human health and the environment according to future land use scenarios. Probable future land use scenarios are based on available information and professional judgement (OSWER Directive 9285.7-01a, Risk Assessment Guidance for Superfund, Vol. 1, September 1989). However, guidance on the time of application of future land use scenarios is not specified. Radionuclide contaminants at the Hanford Site will be significant risk drivers in many risk assessments. Since radionuclides decay with time, the date when future land use is assessed will be very important in determining cleanup strategies because risks for sites will decrease as the radionuclides decay. The time of compliance has been established as 2018 based on the cleanup schedule specified in the Tri-Party Agreement. However, DOE Order 5820.2A allows for 100 years of active institutional controls. Therefore, RL maintains that the application of future scenarios, if any, should occur in 2118. This time requirement would be extended at sites where longer periods of institutional control or monitoring are specified in the RODs.

LIST OF ACRONYMS

ARAR	legally applicable, or relevant and appropriate federal and state requirement
ATSDR	Agency for Toxic Substances Disease Registry
AWQA	EPA ambient water quality advisory
AWQC	EPA or state ambient water quality criterion
CEDE	committed effective dose equivalent
CERCLA	Comprehensive Environmental Response, Compensation, and Liability Act of 1980
CFR	Code of Federal Regulations
DCF	dose conversion factor
DNR	Washington State Department of Natural Resources
DOE	U.S. Department of Energy
DOE-RL	U.S. Department of Energy-Richland Operations Office
DOW	Washington State Department of Wildlife
DRF	dose rate (conversion) factor
EC_{50}	effective concentration at which 50% of experimental organisms exhibit a nonlethal effect
ED_{50}	effective dose at which 50% of experimental organisms exhibit a nonlethal effect
ECAO	Environmental Criteria and Assessment Office
Ecology	Washington State Department of Ecology
EPA	U.S. Environmental Protection Agency
ESA	Endangered Species Act
FI	Facility Investigation
FS	feasibility study
HEAST	Health Effects Assessment Summary Table
HHE	Human Health Evaluation
HMS	Hanford Meteorological Station
HMWA	Hazardous Waste Management Act
HRS	Hazard Ranking System
HSBRAM	Hanford Site Baseline Risk Assessment Methodology
ICRP	International Commission on Radiological Protection
IRIS	Integrated Risk Information System
IRM	interim remedial measure
LC_{50}	lethal concentration at which 50% of experimental organisms die
LD_{50}	lethal dose at which 50% of experimental organisms die
LOAEL	lowest observed adverse effects level
LOEL	lowest observed effects level
MRL	minimum risk level
MTCA	Model Toxics Control Act
MTCACR	Model Toxics Control Act Cleanup Regulations
NCP	National Oil and Hazardous Substances Pollution Contingency Plan
NCRP	National Council on Radiation Protection and Measurements
NOAEL	no observed adverse effects level
NOEL	no observed effects level
NPL	National Priorities List
ORP	Office of Radiation Programs
RAGS	Risk Assessment Guidance for Superfund Volume I

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LIST OF ACRONYMS (cont.)

RCRA	Resource Conservation and Recovery Act of 1976
RfC	reference concentration
RfD	reference dose
RFI	RCRA facility investigation
RI	remedial investigation
RME	Reasonable Maximum Exposure
ROD	Record of Decision
SQL	sample quantitation limit
TAL	target analyte list
TCL	target compound list
TDL_o	lowest observed toxic dose
TPA	Tri-Party Agreement
WAC	Washington Administrative Code
WHC	Westinghouse Hanford Company

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1.0 INTRODUCTION

This first chapter of the report provides introductory information on the Hanford Site baseline risk assessment methodology. The purpose and objectives of the methodology are presented in Section 1.1. The organization of the methodology and relevant statutes, regulations, and guidelines are described in Sections 1.2 and 1.3, respectively. Section 1.4 discusses the role of risk assessment in the activities of the Hanford Site Environmental Restoration Program and the evolving Hanford Site Past-Practice Investigation Strategy. Finally, the methodology approach to the human health evaluation and environmental evaluation is described in Section 1.5

1.1 PURPOSE OF THE METHODOLOGY

This methodology has been developed to prepare human health and environmental evaluations of risk as part of the Comprehensive Environmental Response, Compensation, and Liability Act (CERCLA, 42 USC 9601 et seq.) remedial investigations (RIs) and the Resource Conservation and Recovery Act (RCRA, 42 USC 6901 et seq.) facility investigations (FIs) performed at the Hanford Site pursuant to the Hanford Federal Facility Agreement and Consent Order (Ecology et al. 1990), referred to as the Tri-Party Agreement (TPA). Development of the methodology has been undertaken so that Hanford Site risk assessments are consistent with current regulations and guidance, while providing direction on flexible, ambiguous, or undefined aspects of the guidance. The methodology identifies Site-specific risk assessment considerations and integrates them with approaches for evaluating human and environmental risk that can be factored into the risk assessment program supporting the Hanford Site cleanup mission. Consequently, the methodology will enhance the preparation and review of individual risk assessments at the Hanford Site.

This report and the risk assessment methodology have been prepared to satisfy TPA Milestone M-29-03. The objective of this milestone is to "submit risk assessment methodology document (primary document) and include in Appendix F" (i.e., the most recent version of the TPA). Technical representatives from the U.S. Department of Energy - Field Office Richland (DOE-RL), the State of Washington Department of Ecology (Ecology), the U.S. Environmental Protection Agency (EPA), Westinghouse Hanford Company (WHC), and the U.S. Army Corps of Engineers (COE) participated in a risk assessment methodology committee to support this TPA milestone. The purpose of the committee was to provide input into the development of a Hanford Site Baseline Risk Assessment Methodology.

1.2 METHODOLOGY ORGANIZATION

This section assists the reader in understanding the organization of the methodology presented. Chapter 1 provides background information on the development of the methodology including applicable environmental statutes, regulations, and regulatory guidelines. The role of risk assessment in the Hanford Site environmental restoration process is discussed and the approach to the development of the risk assessment methodology is described.

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Chapter 2 describes the methodology for the human health evaluation and Chapter 3 describes the methodology for the environmental evaluation. Potential applications of the risk assessment methodology at the Hanford Site are discussed in Chapter 4. References for the methodology report are provided in Chapter 5. Appendices provide supporting information. Appendix A provides an overview of the exposure scenarios and exposure parameters for those scenarios. A Hanford modeling standard identifying computer codes for use in support of risk assessment at the Hanford Site is presented in Appendix B.

1.3 APPLICABLE ENVIRONMENTAL STATUTES, REGULATIONS, AND REGULATORY GUIDELINES

The Hanford Site environmental restoration activities are being conducted pursuant to multiple federal statutes, regulations, and guidelines. The primary federal statutes relevant to the risk assessment process include CERCLA and RCRA. The primary Washington State statutes that are potential applicable or relevant and appropriate requirements (ARARs) for these activities include the Model Toxics Control Act (MTCA, Ch. 70.105D RCW) and the Hazardous Waste Management Act (HWMA, Ch. 70.105 RCW).

The regulations corresponding to the above statutes are the National Oil and Hazardous Substances Pollution Contingency Plan (NCP, 40 CFR §300) and the Model Toxics Control Act Cleanup Regulations (MTCACR, Ch. 173-340 WAC). RCRA regulations pertaining to risk assessment have yet to be promulgated; however, proposed rules (55 FR 30798) indicate an intention to maintain a high degree of consistency with the NCP. Similarly, the existing state HWMA regulations do not address risk assessment.

The overall mandate of CERCLA, the NCP, MTCA, and MTCACR is to protect human health and the environment from current and potential threats posed by hazardous substance releases. Considerable guidance on risk assessment has been published to support CERCLA RI and RCRA FI activities. The primary sources of general guidance on the risk assessment process utilized in preparation of the methodology include:

- Risk Assessment Guidance for Superfund, Volume I: Human Health Evaluation Manual (RAGS, EPA 1989a),
- Risk Assessment Guidance for Superfund, Volume II: Environmental Evaluation Manual (EPA 1989b),
- Standard Default Exposure Factors (EPA 1991a),
- EPA Region 10, Supplemental Risk Assessment Guidance for Superfund, August 16, 1991 (EPA-10 1991),
- Statement of Work for the RI/FS Environmental Evaluation for Superfund Sites (EPA-10 1989),
- RCRA Corrective Action Interim Measures Guidance, Interim Final, (EPA 1988a), and
- RCRA Facility Investigations (RFI) Guidance (EPA 1989c).

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Additional guidance documents specific to various technical areas of the risk assessment process have also been used and are cited throughout the methodology, as appropriate.

The baseline risk assessment methodology report is a living document. Since the guidance for both human health and environmental risk assessment is evolving, new guidance should be periodically reviewed for its relevance to this document and inclusion, as appropriate. Similarly, as Hanford-site specific information is collected during investigations, the application of the methodology should also be reviewed and refinements made in the methodology if necessary.

1.4 ROLE OF RISK ASSESSMENT IN THE HANFORD SITE ENVIRONMENTAL RESTORATION PROGRAM

As indicated above, the assessment of health and environmental risk plays an essential role in the RI/FS process. The NCP calls for a site-specific baseline risk assessment to be conducted, as appropriate, as part of the RI [40 CFR 300.430 (d)(1)] and MTCACR has provisions for a risk assessment as part of a state RI/FS (WAC 173-340-350). Recently, EPA, Ecology, and DOE-RL have agreed on a strategy, Hanford Past-Practice Investigation Strategy (DOE-RL 1991), for conducting past-practice site investigations and cleanups to maximize efficiency, maintain project schedules, and achieve earlier remedial action. This approach to RI/FS activities with a bias for action has resulted in additional applications of risk assessment other than the baseline risk assessment to support the strategy. Specifically, the baseline risk assessment methodology may be used to:

- Assist in determining the need for interim remedial measures (IRMs) and establishing acceptable exposure levels, and
- Determine the residual baseline risk associated with a single waste unit, an operable unit, or an aggregate area.

Such uses of the methodology represent, to a degree, unique applications of baseline risk assessment techniques. The Hanford Past-Practice Investigation Strategy emphasizes implementation of early IRMs. IRMs may be undertaken at some waste units prior to investigation at others within the same operable unit. These applications are consistent with recent guidance on the "Role of the Baseline Risk Assessment in Superfund Remedy Selection Decisions" (EPA 1991b).

1.5 RISK ASSESSMENT METHODOLOGY APPROACH

This section assists the reader in understanding the approach used to develop the risk assessment methodology. The approach to the methodology for the human health evaluation is discussed in subsection 1.5.1. The environmental evaluation approach is described in subsection 1.5.2.

1.5.1 Human Health Evaluation

The methodology for the Human Health Evaluation (HHE) is based on the process set forth in the NCP with incorporation of the requirements set forth in the MTCACR cleanup standard development process. The MTCACR approach has been utilized, in conjunction with the NCP process, because it is considered a potentially applicable or relevant and appropriate requirement for NPL sites in the State of Washington. A decision on what constitutes an ARAR is finalized when the Record of Decision (ROD) for an operable unit is issued. The integration of exposure parameters and other risk assessment aspects of MTCACR into the Hanford Site Baseline Risk Assessment Methodology will provide estimates of risk that are meaningful should they be evaluated within the context of the NCP or the MTCACR.

Although the MTCACR provides for risk assessment procedures (WAC 173-340-708), the resulting cleanup standards are developed using risk-based calculations which are generic rather than site-specific. All sites within the state are regarded as either being generically residential or generically industrial with specific exposure assumptions defined. Other land uses are recognized, such as agricultural or recreational, but all cleanup standards must be at least as stringent as Method C cleanup levels [e.g., WAC 173-340-740 (1)(d)], which are generally applied to industrial sites.

The HHE methodology that is presented in Chapter 2 incorporates the requirements of MTCACR and CERCLA risk assessment guidance, including EPA Region 10 supplemental risk assessment guidance, to provide a risk assessment approach tailored to the Hanford Site that conservatively focuses on probable human health impacts. The methodology provides procedures for focusing on major contaminants, environmental media, pathways, receptors, and exposures to identify the significant risk drivers without compromising human health concerns. This HHE methodology is only one tool to be used within the overall site evaluation.

1.5.2 Environmental Evaluation

Unlike the human health evaluation, the current MTCACR cleanup standard development process provides no specific procedures for the environmental evaluation component of a baseline risk assessment other than the requirement that cleanup standards be protective of the environment (WAC 173-340-100). However, the Hanford Site Environmental Restoration Program is being conducted in accordance with CERCLA and RCRA, as well as with potential state requirements. As pertinent RCRA regulations have yet to be promulgated (and proposed rules indicate that EPA plans to maintain a high degree of consistency with CERCLA regulations), CERCLA baseline risk assessment requirements set forth within the NCP can be consulted for procedural guidance.

The NCP states that at an NPL site [40 CFR § 300.430(d)(4)]:

> . . . the lead agency shall conduct a site-specific baseline risk assessment to characterize the current and potential threats to human health and the environment that may be posed by contaminants migrating to ground water or surface water, releasing to air, leaching through soil, remaining in the soil, and bioaccumulating in the food chain. The results of the baseline risk

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> assessment will help establish acceptable exposure levels for use in developing remedial alternatives. . . .

The overall goal of the environmental evaluation is therefore to characterize current and likely future non-human ecological risks attributable to releases of contaminants from the site to determine cleanup levels. The NCP goes on to focus the scope of the environmental evaluation by stating that [40 CFR § 300.430(e)(2)(i)(G)]:

> Environmental evaluations shall be performed to assess threats to the environment, especially sensitive habitats and critical habitats of species protected under the Endangered Species Act (ESA).

These general requirements of the NCP form the basis for the environmental evaluation methodology presented in Chapter 3. By satisfying the NCP requirements, the general requirement of RCRA and the state to protect the environment will also be met. The methodology presented also incorporates supplemental EPA guidelines on ecological risk assessment that have been developed under various programs.

The field of ecology, like any other science, utilizes a vocabulary that at times requires clarification. "Ecological risk assessment" and "ecorisk assessment" are phrases currently used to describe the process referred to within this methodology as an environmental evaluation. The term environmental evaluation is employed to maintain consistency with the NCP and CERCLA program guidance documents. "Ecology" refers to the study of the biological organisms in relation to each other and to their environment. Within this methodology, the term is applied to all organisms except humans, as humans are addressed separately within the human health evaluation component of the baseline risk assessment.

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2.0 HUMAN HEALTH EVALUATION METHODOLOGY

The human health evaluation consists of four elements. These elements are:

- Identification of contaminants of potential concern
- Exposure assessment,
- Toxicity assessment, and
- Risk characterization.

The following sections present a detailed discussion on these four elements relative to the Hanford Site baseline risk assessment methodology. Section 2.1 discusses the identification of contaminants of potential concern and the preliminary risk-based screening process. The exposure assessment and Hanford Site-specific details are provided in Section 2.2 and a discussion of the toxicity assessment is provided in Section 2.3. The integration of exposure information and toxicity information to develop the risk characterization is discussed in Section 2.4. A summary of the human health evaluation is provided in Section 2.5.

The risk assessor is referred to Exhibit 9-1 of RAGS (EPA 1989a) for a suggested outline of the baseline risk assessment based on the four elements above. The first element listed above has been modified from the outline by the replacement of the term "chemical" with the term "contaminant". Potential contamination at the Hanford Site is not limited to chemicals, but also includes radionuclides. Therefore, the more generic term is used. The outline is provided as a guide and should be modified appropriately for use at the Hanford Site. The baseline risk assessment at the Hanford Site is usually part of a more complex investigation and report (e.g., the RI/FS) and some information may be addressed in other portions of such a report.

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2.1 IDENTIFICATION OF CONTAMINANTS OF POTENTIAL CONCERN

This section discusses the process for using data that has been collected and evaluated as part of the site characterization process to identify contaminants of potential concern at a site.

2.1.1 Pareto Principle

At any hazardous substance release site there is the potential that a large number of hazardous substances are present, as thousands of chemical products have been and are being produced and used in the United States. It is not feasible to analyze for all these substances at any given site; fortunately, it is not necessary to do so. Multivariate data sets (e.g., data sets consisting of multiple hazardous substance parameters) typically exhibit a phenomenon described by the Pareto principle (Wadsworth 1990). The Pareto principle states that a relatively large number of problems (e.g., a large proportion of site-attributable risks) in any given situation will found to be caused by only a few factors (e.g., only a few hazardous substances).

The EPA applied this principle when it developed lists of approximately 150 hazardous substances that have been most commonly encountered in the course of implementing the nation's clean water, clean air, and hazardous substances programs. The substances in these lists — which are referred to as the target analyte list (TAL) and the target compound list (TCL), and which consist of inorganic and organic substances, respectively — are among those that were manufactured and used in the greatest amounts and that are the most toxic.

The TAL/TCL typically forms the initial set of hazardous substances considered during Hanford site investigations. With appropriate information on the history or site operations and previous environmental investigation data, the initial set can be tailored to site conditions by adding site-specific hazardous substances and indicator parameters that may prove to be of interest and deleting those TAL/TCL parameters that are likely to be absent in any significant quantities. Given the need to augment the TAL/TCL with radiological parameters to take specific operations into account, it is thus easy to see that most remedial, facility, or limited field investigations at the Hanford Site will generate data sets consisting of 100 to 200 hazardous substance parameters.

Once such a data set is validated and provided to the risk assessor, he or she must capitalize on the Pareto principle further to focus the set on a manageable number of parameters to carry through the detailed baseline risk assessment. The need for this step in the baseline risk assessment process is well documented (EPA 1989a):

> Carrying a large number of chemicals through a quantitative risk assessment may be complex, and it may consume significant amounts of time and resources. The resulting risk assessment report, with its large, unwieldy tables and text, may be difficult to read and understand, and it may distract from the dominant risks presented by [releases from] the site.

This step of the process is especially critical at the Hanford Site given the complexities of the Site and the importance of the Tri-Party Agreement milestones. However, many of the procedures previously set forth by EPA (1989a) are technically flawed (are non-conservative), do not objectively interpret analytical results (are biased), and are inefficient (save little, if any, time and resources).

2.1.2 Contaminant-of-Potential-Concern Identification Process Overview

The contaminant-of-potential-concern identification process described below is logical (based on science and statutory requirements), effective (focuses without sacrificing conservatism), and easy to employ (saves time and resources); it also incorporates recent modifications employed by EPA regional risk assessment personnel (EPA-10 1991). The process involves two stages:

- control (or project-specific background) screening (i.e., determination of which of the initial set of hazardous substances and indicator parameters are actual contaminants, or, more simply, definition of contaminants); and

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- preliminary risk screening (i.e., determination of which of the substances shown to be actual contaminants in the first step are the potential dominant risk-drivers, or, more simply, definition of contaminants of potential concern).

The overall process is graphically depicted in Figure 2-1, and the two stages are described in Subsections 2.1.3 and 2.1.4, respectively. A summary of the process is provided in 2.1.5. According to Figure 2-1, validated investigation data are required to initiate the contaminant-of-potential-concern identification process. Data validation procedures used at the Hanford Site, that have been adapted from EPA protocols, are documented by and available from Westinghouse Hanford Company. Other data evaluation steps, as recommended in Section 5 of RAGS (EPA 1989a) are conducted as part of the RI/FS or RFI/CMS tasks implemented during the site investigation and characterization process. The risk assessor is referred to Section 2.4.3.1 for additional discussion.

2.1.3 Definition of Contaminants

The selection of controls and the selection of control distributions, including detected and non-detected parameters and the handling of non-detected parameters are discussed in the following subsections.

2.1.3.1 Selection of Controls. In order to maintain objectiveness, this stage of the process is statistically based. Site investigation data will (or should) include a characterization of control conditions for each parameter in the initial site-tailored data set. It must be noted that control conditions do not refer to pristine or pre-industrial conditions. Such conditions no longer exist. The presence of pesticides in remote sections of polar icecaps attributable to world-wide applications, of elevated lead levels in surface soils attributable to internal combustion engine exhaust, and of man-made radionuclides in precipitation from atmospheric nuclear weapons testing prove this. In addition, federal and state hazardous substance and waste statutes apply only to releases from specific sites, not to wide-spread anthropogenic sources.

The control conditions must be defined on a project-specific basis. A sampling station appropriate for use as a Hanford-Site-specific control, for example, may or may not be suitable for use as a control for a given aggregate area (i.e., NPL site), operable unit, or waste unit. Specific guidelines can not be given, but, in general, the further a control or background station is from the project area, the less likely the data from that station are representative of project-specific control or background conditions. Background conditions are currently being compiled for the Hanford Site as a whole under Tri-Party Agreement Milestone M-28. Once the deliverable for this milestone becomes available, however, the above caveats must be heeded, and the application of Site-wide background values to specific projects must be evaluated on a case-by-case basis.

The risk assessor must ensure that proper control or background data sets are employed in the contaminant identification process. As many substances defined as hazardous by regulation are naturally occurring, failure to consider control data could lead, for example, to the error of attributing risks to a given site when the risks are, in fact, of natural origin. If such an error were not recognized at the Hanford Site, taxpayers could be burdened with the costs of remediation of the natural environment. The purpose of a

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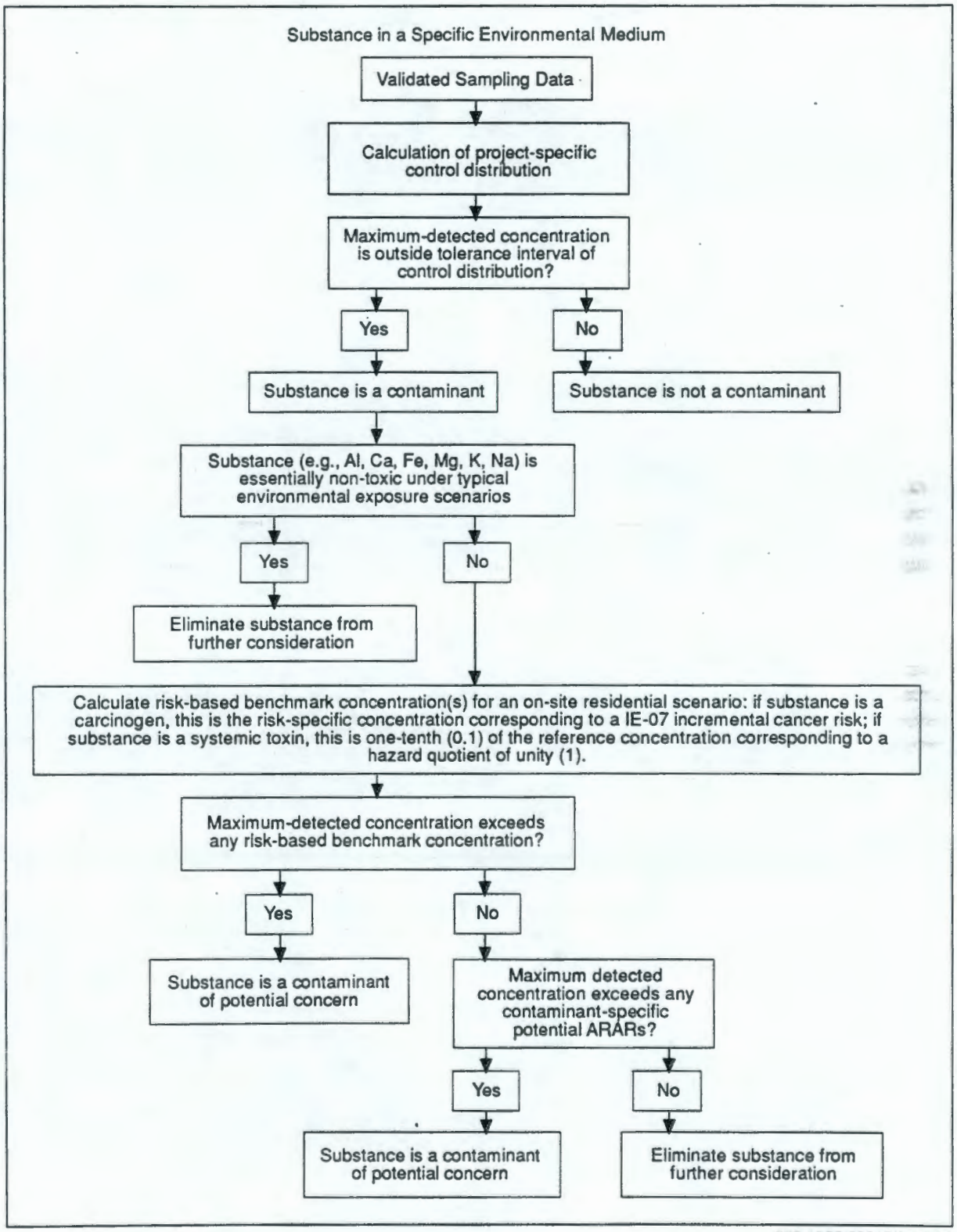

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Figure 2-1. Overview of Contaminant Identification Process.

baseline risk assessment is to characterize the risks posed by the release of hazardous substances from a facility. Thus, the presence of a naturally occurring hazardous substance is not germane to the process, just as other risk-producing components of the natural environment — such as sunshine, cosmic radiation, and extreme weather — are not germane to the process.

Appropriate control data must also be used to ensure that cleanups of widespread anthropogenic releases are not funded. Appropriate control data for mobile environmental media (e.g., air, ground water, surface water) are especially important to ensure that site characterization, evaluation, and remediation efforts are directed to the waste units actually releasing a given hazardous substance. Contamination can incorrectly be attributed to a waste unit, when the release is actually occurring from an upwind, upgradient, or upstream facility. Such data are necessary for the successful implementation of the IRM strategy which is focusing on priority waste sites.

Proper project-specific control data are critical to any environmental investigation effort, and the absence of such data indicates a deficiency in the investigation planning phase. In the absence of appropriate data, control data available from another project in a similar environmental setting — or those available from a Site-wide, regional, national, or global basis — may be able to be used, with documented caution, if one can be reasonably certain that the environmental media associated with the facility, operable unit, or site in question have not been impacted by other operations, and reasonably certain that the environmental media are representative of Site, regional, national, or global conditions.

In general, neither of the above assumptions are often valid, but they could apply to some project locations in the relatively remote portions of the Hanford Site. More likely, certain control data for specific non-mobile media (e.g., soils) from nearby projects can be extrapolated with less concern. However, because obtaining proper control data is a critical data quality objective, an absence of such data may result in the need to expand the scope of a project investigation, and delay the investigation documentation and reporting process, so that such data may be properly obtained and evaluated.

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2.1.3.2 Definition of Control Distributions

Detected Parameters

Once the risk assessor determines that the validated control data available are appropriate to the project, control conditions for each parameter are defined by means of a tolerance interval. In virtually all instances, where one is concerned only about elevated concentrations of a substance in the environment, a one-sided upper tolerance limit is calculated; the 95th percentile of the control distribution is estimated at the default confidence rate of 95% (i.e., Type I error rate or false positive error rate, $\alpha = 0.05$). In the case of pH, where one is concerned about both elevated and depressed values, the 2.5th and 97.5th percentiles are estimated (also with $\alpha = 0.05$). The method for the calculation of an upper tolerance limit is provided in EPA (1989d).

The tolerance limit (or limits) statistically defines the normal range within which one could expect to find the substance occurring 95% of the time (with 95% certainty) under control conditions. Any departure from this range is interpreted as evidence of contamination attributable to the waste unit (or operable unit or aggregate area) in

question. As any control data set used at the Hanford Site will be multivariate (i.e., contain perhaps 100 to 200 single parameter variables), the experiment-wise false positive error rate will be much higher than 0.05. The actual error rate will be:

$$\alpha = 1 - (0.95)^n.$$

where n is the number of variables in the data set. Using the same control data set repeatedly will have a similar effect on increasing the error rate (Green 1979). As a result, many more substances will be defined as contaminants. For the sake of conservatism and simplicity, however, this bias, even though substantial, can be ignored.

Implicit in this tolerance interval approach is the assumption that the parameters are distributed normally under the control conditions. This assumption, as it is being applied to the distribution of a parameter, as opposed to the distribution of the mean of a parameter distribution (where the Central Limit Theorem supports a normal assumption), is not always met.

However, an up-front assumption of normality is objective, and, for all practical purposes, sufficiently robust to justify its use in contaminant identification. In addition, funding and schedule constraints are such that the control data sets are not anticipated to be large enough to conduct a meaningful test for normality; furthermore, the control data sets are expected to be far below the numbers necessary to conduct a nonparametric or distribution-free tolerance interval evaluation.

Care should be taken in combining data sets, when defining control distributions. When practicable, statistical screening should be employed to ensure that data from more than one distribution are not inadvertently combined into a single distribution. Examples include: differences in surface-soil quality and subsurface soil quality; differences in soil or ground-water quality in different pedons or aquifers; and differences in ground-water quality data from different time frames in situations where control conditions are temporally variable (e.g., where an upgradient contaminant plume, unrelated to the facility, operable unit, or site in question, is impacting a project-specific control location). An example of an appropriate statistical screening procedure for these situations is an analysis of variance (ANOVA); the reader is advised to consult a statistical methods reference for the specifics on this and other potentially appropriate procedures.

Non-Detected Parameters

As contaminant data sets contain considerable amounts of censored data, a process must be established for estimating tolerance limits when a given parameter is never detected or only sometimes detected in the control data set. Computerized methods, such as iterative maximum likelihood calculations, are available for estimating the true means and variances of censored data sets. However, none of these methods are of use if the data set contains no detections of a given parameter, and in a typical control data set of 100 to 200 parameters, this situation arises frequently.

Therefore, for the sake of consistency, objectivity, and simplicity, one-half the sample quantitation limit (SQL) reported whenever a parameter is not detected will be used as a surrogate value in the calculation of the tolerance limit. For example:

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Reported Validated Value	Surrogate Value
10 U (i.e., < 10) mg/kg	5 mg/kg

Substituting one-half the SQL in such cases does bias the variance estimate of the parameter, but it does not bias the estimate of the mean. Consistent use of this procedure is objective, quick, and simple, does not require special equipment, and allows calculations to be easily duplicated or checked. It will not work, however, in those cases where a given parameter is never detected in the control data set. In these cases, use of 0.5 SQL gives an unbiased estimate of the mean, but the variance data is entirely artificial.

Therefore, when a parameter is never detected in the control data set the highest reported SQL for that parameter is used as a surrogate upper tolerance limit. This simple procedure objectively interprets the validated data. For example, one can not distinguish 10 U mg/kg from 6 J mg/kg (where J denotes that the value is estimated, usually at a concentration below the laboratory-contract-specific detection limit), and this procedure acknowledges this. It must be noted that 10 mg/kg is greater than 10 U mg/kg, as 10 U denotes < 10. Therefore, a concentration equivalent to a surrogate tolerance limit would be regarded as evidence of contamination, whereas for a calculated tolerance limit, a concentration must first lie outside that limit.

2.1.4 Preliminary Risk-Based Screening

Once actual contaminants attributable to a particular waste unit, operable unit, or aggregate area have been defined, the risk assessor can focus the relevant data set further by implementing a preliminary risk screening procedure. Such a procedure takes advantage of the Pareto principle by identifying which of the actual site contaminants are the dominant risk drivers. Those contaminants that have the potential to dominate the risk attributable to the release at the site are referred to as contaminants of potential concern. This stage of the contaminant identification process, when performed effectively, is highly useful.

The preliminary risk screening procedure must be simple and conservative. Simplicity is necessary so that the procedure is quick, easy to use, and applications are readily verifiable. With relative procedural simplicity, one can save significant time and resources in the implementation and review of the risk assessment; one can also produce a more focused, more easily understood characterization of risks attributable to the release of hazardous substances at a given site.

Conservatism can be viewed within the context of the preliminary risk screening as meaning less than 100% efficiency. For example, only a handful of contaminants may actually dominate the risk at a given site. However, one would not know this until after the entire risk assessment process has been completed. For the sake of conservatism, or to ensure that all dominant risk driving contaminants are carried through the entire process, several non-dominant contaminants should be retained by the preliminary risk screening procedure.

Given the above constraints, the EPA-10 procedure for risk-based screening of contaminants (EPA-10 1991) has been adopted and modified for use in this stage of the

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contaminant identification process. This process consists of four steps:

1. Tabulate the maximum concentration of each contaminant in each environmental medium.

2. Calculate the risk-based benchmark concentrations.

3. Compare the maximum concentration to risk-based benchmark concentrations or other appropriate benchmark concentrations (e.g., reference concentrations, incremental probabilities of cancer risk, contaminant-specific potential ARARs). Eliminate contaminants if they do not exceed any of their respective risk-based benchmark concentrations.

4. Designate all contaminants not eliminated as contaminants of potential concern and carry these through the remainder of the risk assessment process.

The following paragraphs discuss these steps in more detail.

2.1.4.1 Maximum Contaminant Concentration. The maximum concentration of a contaminant in a given medium should be identified. The use of the maximum concentration enhances the simplicity of the screening procedure and adds an element of conservatism, as well. The maximum concentration should be used regardless of location with respect to potential receptor exposure points. For example, the maximum concentration of a contaminant detected in soil would be used regardless of depth. This is not only easier to do, but is also conservative in that it assumes that human exposure is possible, whether or not such exposure really is possible.

2.1.4.2 Risk-Based Benchmark Calculations. For conservatism and simplicity, all risk-based benchmark concentrations are calculated using residential exposure assumptions. The residential exposure parameters provided in Appendix A and Tables A-5 and A-6 should be used for this purpose. This is a conservative exposure scenario for the Hanford Site as there is currently no on-site residential use of the Site, nor is there likely to be such use in the foreseeable future. Standard residential exposure assumptions are higher than the standard industrial exposure assumptions that are most appropriate for the Hanford Site. In addition, as discussed below, the risk-based benchmark calculations are calculated at risk levels less than those used to determine acceptable exposure levels for the NCP or MTCACR.

Toxicity information, including slope factors and reference doses should be identified for all contaminants for use in calculating the risk-based benchmark calculations.

The calculation of the risk-based benchmark concentrations should consider both carcinogenic and noncarcinogenic effects. Therefore, concentrations should be calculated using exposure parameters for evaluating carcinogens and separate risk-based benchmark concentrations should be calculated based on noncarcinogenic systemic effects.

One or more of three general exposure pathways — soil ingestion, water ingestion, and air inhalation — occur at most sites and potentially will drive risks at most sites. This is

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why EPA and Ecology have devoted most of their efforts in defining default parameters for these pathways. As a result, the preliminary risk screening will utilize all three of these pathways.

Soil Ingestion

The maximum concentration of a contaminant detected in soil should be evaluated using the soil ingestion pathway. This concentration is used regardless of the depth of the maximum concentration. As noted in EPA-10 1991, soil contaminants with known inhalation toxicities (eg. chromium, carcinogenic forms of nickel, etc.) must not be eliminated based only on the soil ingestion route. Since all three general exposure pathways are used in the screening, this concern is addressed.

Water Ingestion

The water ingestion pathway should conservatively assume, at this time, direct exposure to ground water. Surface water exposure will not be assessed during this step at many sites due to lack of empirical surface water quality data, or the fact that realistic extrapolation of ground water concentrations to surface water concentrations would be time consuming and would usually result in more diluted contaminant concentrations. At this point in the process, the combination of simplicity and conservatism outweighs the desire for realism. Realism becomes important in the exposure assessment stage of the baseline risk assessment process (see Section 2.2).

Air Inhalation

Empirical air monitoring data will often not be available, or if data are, it will usually be inadequate to characterize contaminant concentrations or the variety of potential contaminants in this medium. As a result, soil concentrations will need to be extrapolated to the atmosphere. Soil gas concentrations can be conservatively extrapolated for the purposes of screening by assuming that the concentration in the ambient atmosphere is the same as that in the soil gas. For particulate concentrations, conservative extrapolations can be made by hypothetically placing sufficient amounts of soil in the atmosphere at the annual average national ambient air quality standard for respirable particulates — 50 μg/m3 [40 CFR §50.6(b)].

Radionuclide Considerations

For radionuclides, additional pathways may be required for the preliminary screening. The hazard posed by radiation exposure depends to a large extent on the combination of radiation quality (alpha, beta, gamma) and exposure pathway. External exposure to radionuclides is only a concern from gamma-emitters due to their ability to penetrate tissue and deliver a deep dose. Alpha and beta-emitting radionuclides only pose a significant health hazard when inhaled or ingested, as their energy is deposited locally in tissues and cells.

An additional consideration for radionuclides is decay chains. Often the decay of a radionuclide results in a radioactive daughter product which itself poses a health hazard (e.g., the gamma-emitting daughter of Cs-137, Ba-137m). While many radionuclides of interest have only one radioactive daughter (e.g., Cs-137, Ru-106, Sr-90), others (i.e.,

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transuranics) have complex decay chains which require special consideration in determining the importance and risks associated with these daughter products. For this reason, among others, the evaluation of risk from radionuclides should be performed by persons with training and experience in health physics or radioecology.

Screening Criteria

The EPA-10 suggests elimination of contaminants if the maximum concentration detected does not exceed, in the case of a carcinogen, a risk-specific concentration corresponding to either a 1E-06 or a 1E-07 incremental cancer risk, depending on certain circumstances. In the case of a systemic toxin, EPA-10 suggests elimination of the contaminant if its maximum detected concentration does not exceed one-tenth (0.1) of its reference concentration, the concentration that corresponds to a hazard quotient of unity (1) (i.e., the concentration above which a toxic effect could reasonably be anticipated).

The above EPA-10 elimination criteria are adopted for use at the Hanford Site with one exception: a risk-specific concentration corresponding to a 1E-07 incremental cancer risk will be used in all circumstances. This provides sufficient conservatism to allow for the overall MTCACR cleanup standard requiring no more than a 1E-05 incremental cancer risk for all carcinogens over all pathways at any given site.

In addition to calculating risk-based benchmark concentrations as note above, other benchmark concentrations, such as contaminant-specific potential ARARs, should also be compiled for use in the screening process.

2.1.4.3 Comparison of Contaminants to Benchmarks. As indicated above, the maximum concentration of a contaminant should be compared to the risk-based benchmark screening concentrations. If the maximum-detected concentration exceeds any risk-based benchmark concentration, the contaminant should be retained for more detailed evaluation in the remainder of the baseline risk assessment.

One additional modification to the EPA-10 procedure is necessary. If a contaminant qualifies for elimination on the basis of the above criteria, it will be retained if it exceeds any contaminant-specific potential ARAR (e.g., federal or state ambient water quality criteria). This provides not only for additional conservatism in the human health evaluation, it also makes the screening procedure relevant to the environmental evaluation (see Section 3.2).

For some contaminants that are essentially non-toxic under typical environmental exposure scenarios (e.g., aluminum, calcium, iron, magnesium, potassium, and sodium), there is no need to apply the risk screening procedure in a quantitative sense. Such contaminants can be eliminated from further risk analysis qualitatively on the basis of professional judgement, as noted in EPA-10 (1991) and EPA (1989a).

2.1.3 Contaminant-of-Potential-Concern Summary

The use of the contaminant identification procedure and the preliminary risk screening procedure results in a site-specific list of contaminants of potential concern. These contaminants are then subjected to the remainder of the data evaluation

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(contaminant fate and transport analysis) and baseline risk assessment process (exposure assessment, toxicity assessment, and risk characterization).

A simplified example of how this process works is summarized below. Certain substances — for example, calcium and plutonium — may be encountered in the site environment at levels elevated above the statistically defined control conditions. The first stage of the contaminant-of-potential-concern-identification process would result in both calcium and plutonium being defined as contaminants. With the substances chosen for this example, no formal exposure or toxicity assessments are required for most individuals to understand that while plutonium is highly toxic, calcium is an essential micronutrient that is virtually non-toxic from the perspective of typical environmental exposure scenarios.

The second stage of the process would therefore define plutonium as a contaminant of potential concern, retaining the substance for detailed contaminant fate and transport analysis, exposure and toxicity assessment, and risk characterization; calcium would be dropped from further consideration, thus streamlining the entire assessment by precluding the substance from additional costly, time-consuming, and, in this instance, unnecessary analysis.

Once the entire risk assessment process is completed for a given site, Pareto analysis can be used to test whether or not all dominant risk driving contaminants have been evaluated. The Pareto analysis consists of ranking each contaminant with respect to incremental risk, then plotting incremental risks on both a contaminant-specific and cumulative basis. (Note: When referring to contaminant-specific issues, EPA uses the term "chemical-specific"; contaminant-specific is the term used throughout this document as it is more appropriate to Hanford Site conditions, where radiological, as well as chemical, contamination exists.) If the dominant risk driving contaminants have been evaluated, the cumulative risk curve should flatten, showing that the non-dominant contaminants evaluated, and thus all those eliminated in the preliminary risk screening, add little or nothing to the overall understanding of site risk.

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2.2 EXPOSURE ASSESSMENT

The objective of the exposure assessment is to estimate the type and magnitude of exposures to chemicals and radionuclides of potential concern that are present at or migrating from a site. This information is then integrated with the toxicity information to characterize the potential risks associated with exposure to contaminants at a site.

The elements of the exposure assessment are discussed in the following subsections. The characterization of the exposure setting and the general Hanford Site exposure setting are discussed in subsection 2.2.1, potentially exposed populations are identified in subsection 2.2.2, and exposure pathways are presented in subsection 2.2.3. This information is integrated to develop exposure scenarios for use in assessing risk at the Hanford Site. A brief discussion of the exposure scenarios is presented in subsection 2.2.4, with a more extensive discussion and a summary of exposure parameters provided in Appendix A. Subsection 2.2.5 provides a discussion on the calculation of intakes and equations for quantifying chemical exposures. The methodology for quantification of radionuclide exposure is also presented in this subsection. The exposure assessment uncertainty evaluation is discussed in 2.2.6.

Extensive information is available to individuals preparing exposure assessments for Hanford Site risk assessments. The risk assessor is specifically referred to the annual environmental reports such as "Hanford Site Environmental Report for Calendar Year 1989" (Jaquish and Bryce 1990), the Hanford Site Development Plan (DOE-RL 1990), the Hanford defense waste environmental impact statement (DOE 1987), and site-specific work plans for additional Hanford Site information and references. The EPA Region-10 library also publishes the "Hanford Bibliography" that lists materials in the Region-10 library related to the Hanford Site. Several other sources are referenced in this section on exposure assessment.

2.2.1 Characterization of Exposure Setting

Characterization of the physical setting of a specific site is essential in developing the exposure assessment. Section 6.2.1 of RAGS (EPA 1989a) identifies important site characteristics that should be considered when preparing an exposure assessment. Because of the size of the Hanford Site, characterization of the physical setting in the exposure assessment should include both general information on the physical setting of the Hanford Site and site-specific information. Site-specific information is especially important because of the number of operable units and waste disposal locations, and because local differences in characteristics such as ground-water hydrology, soil type, meteorology, and vegetation may influence potential exposures.

General information on the physical setting of the Hanford Site has been used to support and develop the methodology. This information, including information on climate, temperature, and surface waters, is summarized in the following paragraphs to provide background for the methodology and the exposure scenarios discussed in subsection 2.2.4 and Appendix A.

The Hanford Site is a 150,000 ha (560 mi^2) reservation owned by the United States government. It is located in the Pasco Basin along the Columbia River in southeastern Washington and covers portions of Benton, Grant, Franklin, and Adams counties (Figure 2-2). The primary mission of the Hanford Site has been plutonium production for military use and nuclear energy research and development. Designated areas of the Hanford Site (100, 200, 300, 400, 600, 700, 1100, and 3000) are shown in Figure 2-2.

The general climate of the Hanford Site is arid to semiarid due to a rain shadow created by the Cascade Range located approximately 130 km (80 mi) west of the Hanford Site. Data collected at the Hanford Meteorological Station (HMS), a large meteorology facility located on a plateau in the center of the Hanford Site, and compiled by Stone et al. (1983) indicate that the total annual precipitation is approximately 18 cm (7 in), usually in the form of rain. Snowfall regularly occurs during the winter, but accumulations are generally limited to depths of less than 15 cm (6 in). Little surface runoff occurs because of the generally flat topography and the limited precipitation. The estimated annual rate of actual evapotranspiration is approximately 18 cm (7 in) (USWB and USDOA 1962).

The summer months at the Hanford Site are typically hot and dry, and winters are moderately cold. Air temperatures for HMS on average reach highs of 37°C (100°F) in the summers and lows of -5°C (23°F) in winter. Wind speed and direction vary throughout the Hanford Site because of the local influence of mountain ridges and river valleys. Average

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wind speeds at HMS are 10 to 12 km/h (6 to 7 mi/h) in winter and 13 to 17 km/h (8 to 10 mi/h) in summer (Stone et al. 1983).

The major surface-water bodies in the Pasco Basin are shown in Figure 2-3. The Columbia River, the major river in the area, crosses the northern portion of the Hanford Site, then turns southward to form the site's eastern boundary. The Columbia is an important source of water for domestic, agricultural, industrial, and recreational users in the Pasco Basin (DOE 1987; Jaquish and Bryce 1990).

The Hanford Reach of the Columbia River extends from Priest Rapids Dam, the first dam upriver from the Hanford Site (approximately 8.5 km or 5.3 mi above the Site boundary), to the head of Lake Wallula (approximately at the southeastern Site boundary), which is created by McNary Dam, the nearest dam downstream. The Hanford Reach, which is approximately 100 km (60 mi) in length, is the only substantial remaining stretch of the Columbia River within the United States that is not impounded by a dam (Jaquish and Mitchell 1988).

Protection of the reach is the current focus of efforts by the Hanford Reach Study Task Force under the Hanford Reach Study Legislation (PL 100-605). A draft environmental impact statement/study report is scheduled to be released for public review Spring 1992. Under consideration for the Hanford Reach, including area within approximately one-forth mile of the river on the Hanford Site, is designation as a Wild and Scenic river, a National River, a National Wildlife Refuge, or a National Conservation Area.

The arid-to-semiarid climate of the Pasco Basin does not support any perennial streams; however, two ephemeral streams, Cold Creek and Dry Creek, are located on the southwestern boundary of the Hanford Site (Figure 2-3). The only naturally-occurring surface water on the Hanford Site is West Lake, near the 200 East Area. The source of recharge for the pond is ground water, which is locally mounded due to infiltration resulting from 200 Areas operations. Some wastewater disposal ditches and ponds are found in the 200 Area plateau.

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2.2.2 Characterization of Potentially Exposed Populations

The receptor populations evaluated in the exposure assessment should be determined on a site-specific basis for each risk assessment. Section 6.2.2 of RAGS (EPA 1989a) provides information on identifying potentially exposed populations. Current on-site receptors are primarily Hanford Site workers. Other receptors, such as recreational populations or those that may be located off-site, will be determined based on the location of a specific site, contaminants detected at a site, the potential for contaminant transport off-site, and other relevant factors. It is important to note that off-site receptors may not always be the closest receptors based on physical location. For example, for some sites downwind receptors may be more distant than upwind receptors, but because of the potential for contaminant transport the downwind receptors would be a more likely exposed population.

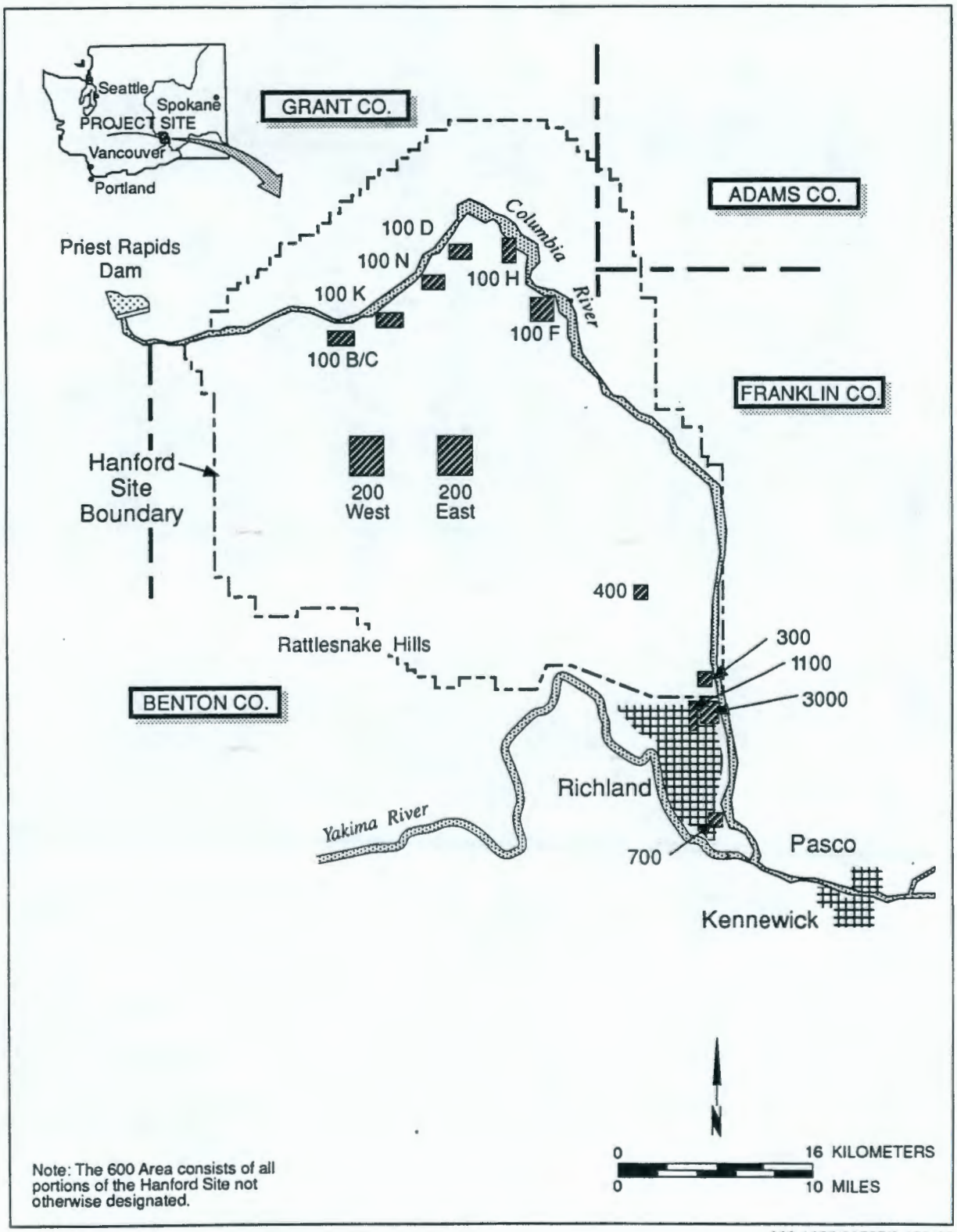

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Figure 2-2. Hanford Site and Area Designations.

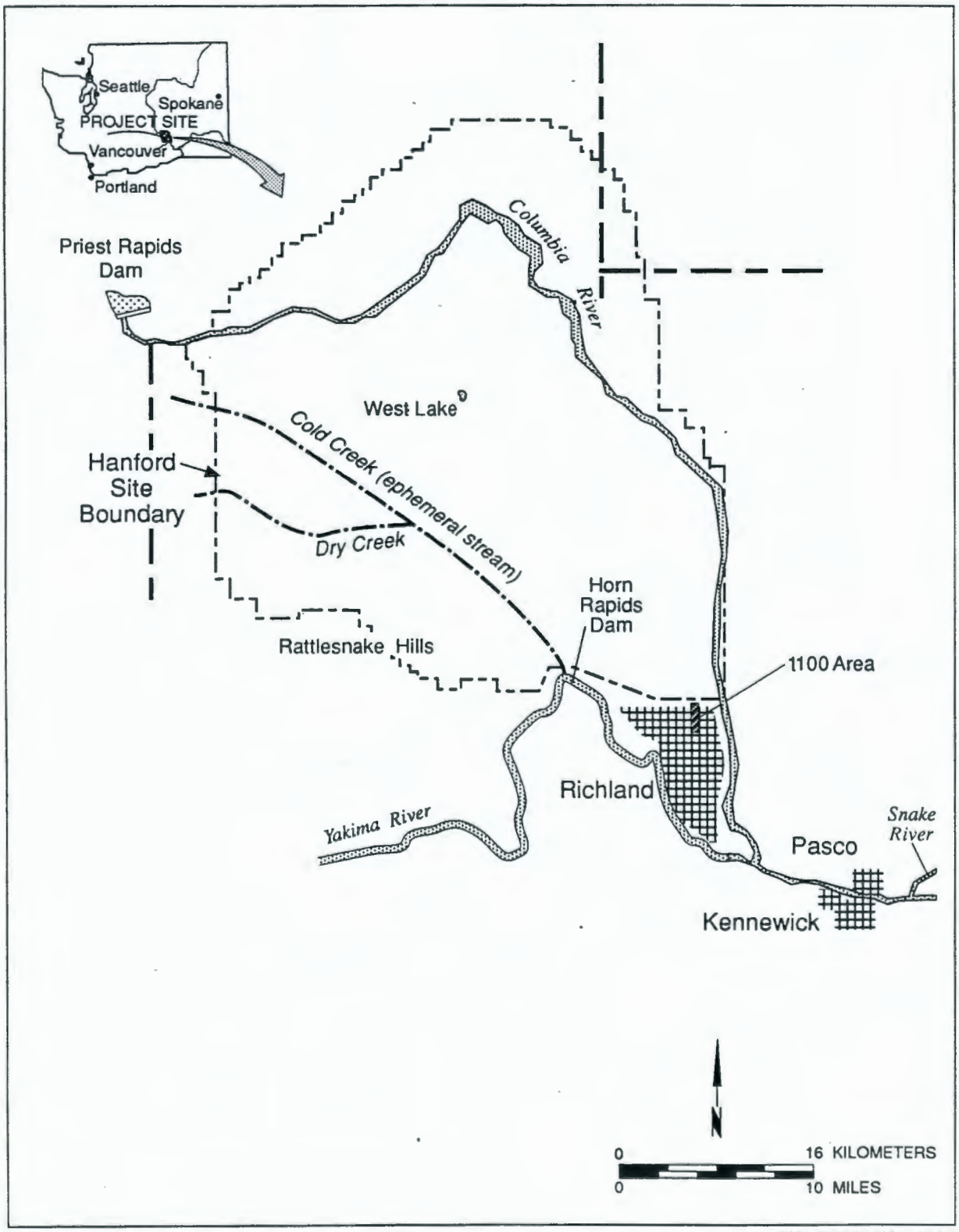

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Figure 2-3. Major Surface Water Features of the Hanford Site.

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Approximately 12,000 people were employed in DOE activities on the Hanford Site in 1989. The greatest number of these people (about 5,200 or 43%) worked within or immediately adjacent to the Richland city limits in the 700, 1100, and 3000 Areas. The 200 and 300 Areas employed about 3,500 (29%) and 1,900 (15%), respectively. Another 750 (6%) were located in the 100 Area with the remaining individuals (about 7%) working in other areas. Washington Public Power Supply System employed about 1,400 people. A number of other work areas exist on the Hanford Site, such as the Solid Waste Landfill, the Fast Flux Test Facility, and U.S. Ecology.

The population in the area surrounding the Hanford Site is predominantly rural, with the exception of the cities of Kennewick, Pasco, and Richland. Using the HMS tower as a reference point that is approximately in the center of the site and 1980 census data, the total population 80 km (50 mi) from the tower was 340,943 in 1980. The number who resided in incorporated cities was 210,999 (Jaquish and Bryce 1990).

Recreational activities associated with the Columbia River include hunting, fishing, boating, water skiing, and swimming. Agricultural activities near the Hanford Site include irrigated and dryland farming, and livestock grazing. About one-third of the crop acreage is irrigated, one third in dryland production, and the remaining third is idle or in summer fallow (Watson et al. 1991).

The Hanford Site is located in lands ceded to the United States in 1855 under treaties with the Yakima Indian Nation Reservation. Under both treaties the Native American signatories retained the right to fish at usual and accustomed places, and retained the privileges of pasturing horses, hunting and gathering roots and berries on open and unclaimed lands within the ceded areas. The protection of these resources for potential future use by the Native Americans, if areas of the Hanford Site were to become open and unclaimed, has been an issue in connection with activities at the Hanford Site.

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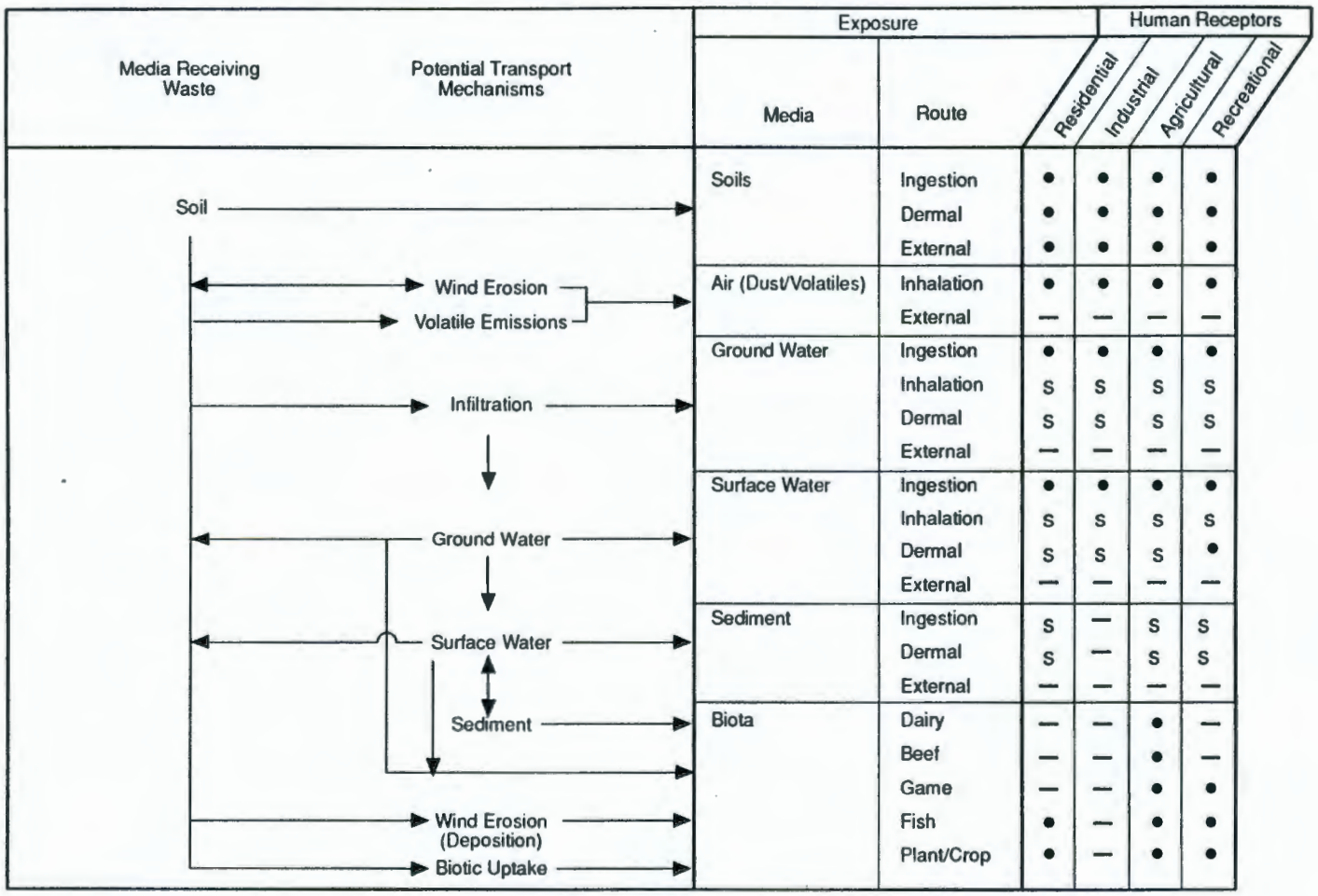
2.2.3 Identification of Exposure Pathways

An exposure pathway describes a unique mechanism by which an individual or population is exposed to chemical or physical agents at or originating from a site. This subsection discusses the identification of exposure pathways including a conceptual model of the exposure assessment.

2.2.3.1 Conceptual Model for Human Exposure Assessment. A conceptual model for human exposures has been prepared as part of this methodology to identify potential human exposure pathways that should be considered in risk assessments prepared for the Hanford Site. The conceptual model, presented in Figure 2-4, summarizes paths that hazardous substances may take to reach potential receptors. A discussion of the model is provided below. It is important to note that although many pathways are possible, the conceptual model focuses on those pathways that are likely to contribute significantly to overall risks and that can be assessed with well-defined parameters. More extensive discussion on potential exposures, exposure pathways, and the selection of pathways for assessment (i.e., primary pathway, secondary pathway, or pathway not assessed) is presented below in 2.2.3.2 and in Appendix A as part of the discussion of scenarios.



Exposure		Human Receptors			
Media	Route	Residential	Industrial	Agricultural	Recreational
Soils	Ingestion	●	●	●	●
	Dermal	●	●	●	●
	External	●	●	●	●
Air (Dust/Volatiles)	Inhalation	●	●	●	●
	External	—	—	—	—
Ground Water	Ingestion	●	●	●	●
	Inhalation	s	s	s	s
	Dermal	s	s	s	s
	External	—	—	—	—
Surface Water	Ingestion	●	●	●	●
	Inhalation	s	s	s	s
	Dermal	s	s	s	●
	External	—	—	—	—
Sediment	Ingestion	s	—	s	s
	Dermal	s	—	s	s
	External	—	—	—	—
Biota	Dairy	—	—	●	—
	Beef	—	—	●	—
	Game	—	—	●	●
	Fish	●	—	●	●
	Plant/Crop	●	—	●	●

● Primary Pathway
s Secondary Pathway
— Pathway not assessed

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Figure 2-4. Conceptual Model for Human Exposure Assessment

The elements necessary for a completed exposure pathway are represented in the conceptual model. These elements are:

- a source and mechanism for hazardous substance release,
- transport mechanisms/media,
- exposure media or point,
- exposure routes, and
- receptors.

All elements must be present for an exposure pathway to be complete. However, the importance of individual pathways to the overall exposure assessment may vary because of the physical characteristics of a site, the physical, chemical, and toxicological characteristics of the hazardous substances present, the probability that a pathway will be completed, and receptor characteristics.

The soil is the medium that received most of the hazardous waste at the Hanford Site through direct disposal of liquids to the soil in cribs, trenches, retention basins, etc., burial of wastes in landfills or burial grounds, and spills and leaks from storage tanks (Jaquish and Bryce 1990). For most of the operable units at the Hanford Site, soil currently represents the primary source of hazardous substances that can potentially be transported to other media where human contact may occur.

As indicated in the conceptual model, potential exposure to hazardous substances in the soil can occur through several exposure routes (i.e., ways for a receptor to come in contact with a hazardous substance such as ingestion, inhalation, etc.) and transport mechanisms. Direct receptor contact with the soil can result in incidental ingestion, dermal exposure, and external radionuclide exposures. Hazardous substances may be transported from the soil to the air in particulate form (dust) through wind erosion or as a volatile emission from the soil. Receptors located near the site or downwind from a site could potentially be exposed through the inhalation of contaminated dust or volatiles.

Hazardous substances in soil can also be transported to groundwater. Hanford Site groundwater monitoring has detected radionuclides, including tritium and uranium, and chemicals such as carbon tetrachloride, chromium, trichloroethylene, and nitrate in the groundwater under the site (Jaquish and Bryce 1990). Although previous liquid disposal practices may have contributed more to the migration of contaminants into the groundwater in the past, infiltration due to precipitation is the probable mechanism for transport currently. Columbia River-groundwater interactions may also influence contaminant transport from the soil.

Once in the groundwater, hazardous substances can be directly ingested by receptors or they may be exposed through dermal contact with the water during showering, bathing, and other domestic or commercial water use if the groundwater is used for such purposes. Inhalation of volatile substances in groundwater may also occur during groundwater use in homes, businesses, or agriculture, and from volatile substances diffusing through the soil to the ambient air from the groundwater. Groundwater used as an irrigation source may reintroduce hazardous substances to the soil. Hazardous substances could also be transported to livestock if groundwater is used as a water source.

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Hazardous substances that have migrated to groundwater may also be transported to surface water. Groundwater flowing under the Hanford Site enters the Columbia River. As discussed above, the Columbia River is used as a source of water for domestic, industrial, agricultural, and recreational purposes. For example, the City of Richland uses river water to artificially recharge the unconfined aquifer to provide treatment of turbid Columbia River water and enhance the city well field capacity. Thus, hazardous substances transported to the Columbia River could potentially be ingested, dermally absorbed during water use, swimming, showering, or bathing, and inhaled (volatile substances) during recreational, residential, industrial, or agricultural uses.

Hazardous substances in surface water may directly impact biota consumed by human receptors (e.g, bioaccumulation in fish, livestock watered with Columbia River water, etc.). These substances can also settle into sediments where contact during recreational use may occur or from which Columbia River biota can be impacted. Surface water used as an irrigation source could reintroduce hazardous substances to the soil.

Biotic uptake of hazardous substances from the soil can also occur resulting in the transport of contaminants from soil to humans. For example, domestic animals and wildlife, while grazing, can ingest soil containing hazardous substances. This would be in addition to ingesting plants that may have taken up hazardous substances directly from the soil or from the deposition of hazardous substances on the plants from dust as a result of wind erosion at a site. Crops or garden produce grown in soils containing hazardous substances are another source of potential exposure for humans.

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2.2.3.2 Exposure Pathways. As noted above, the conceptual model summarizes pathways that are likely to contribute significantly to overall risks and that can be assessed with relatively well-defined parameters. These pathways are considered either primary or secondary pathways as discussed below.

Primary pathways are presented in the conceptual model as those pathways, exposure media, and routes that should be quantatively evaluated for a specific scenario if contaminants are present in a medium. The primary pathways are those pathways that are routinely evaluated in risk assessments at most hazardous waste sites (EPA-10 1991 and EPA 1991a) and frequently are risk-driving pathways at hazardous waste site (see Figure 2-4). These pathways include:

- Direct ingestion of soil,
- Inhalation of fugitive dust and/or volatiles, and
- Ingestion of water (surface water or groundwater).

In addition to these pathways, two other pathways have been identified as primary pathways for all scenarios. These are:

- Dermal exposure to soil contaminants, and
- External exposure from radionuclides in soil.

Dermal exposure from soil has been selected as a primary pathway because the soil was the receiving medium of most waste disposal at the Hanford Site. External exposure to radionuclides is a Hanford Site-specific pathway because of radionuclide use and production at the site. Soil contaminated by photon-emitters is the only exposure media

that should be routinely evaluated for the external exposure pathway.

Several biota pathways have also been selected as primary exposure pathways for specific scenarios. Again, because of the residential and agricultural areas in the vicinity of the Hanford Site, the specific concern with radionuclides, the location of the Columbia River and the potential for contaminated groundwater to reach the river, and the recreational hunting that occurs, biota pathways have been selected as primary pathways for the recreational, residential, and agricultural scenarios, as appropriate.

Secondary pathways are pathways that should be qualitatively evaluated, at a minimum, but may be quantitatively evaluated based on site characterization, contaminant characteristics, contaminant migration, and availability of pathway-specific toxicity information. Any pathway qualitatively evaluated should be discussed in the uncertainty sections of the exposure assessment and the risk characterization. Secondary pathways are indicated in the conceptual model presented in Figure 2-4.

Several pathways have been selected as secondary pathways because they potentially represent exposure routes that contribute less to the overall risk because of dependence on contaminant-specific parameters, frequency and duration of exposures, or likelihood of occurrence. For example, a sediment exposure pathway in the river is an example of a pathway selected as a secondary pathway because the frequency of contact is limited by such factors as the weather and receptor activity patterns (e.g., hunters normally wear clothing that protects from such exposures, water skiers have less contact with sediment, swimmers and people wading represent a population much smaller than those drinking the water and exposures occur on a less regular basis at the same exact location). In addition, the size of the Hanford Site may preclude the impacts to sediments from many sites that are located away from the river and for which groundwater transport would be the only mechanism of transport for contaminants.

Dermal exposures to water are also considered a secondary pathway because significant exposures may rely on chemical-specific factors like dermal permeability. The faster penetrating contaminants (dermal permeability constant >0.6 cm/hr) may pose a hazard similar to direct consumption, but environmental contaminants in this range appear to be a minority (EPA 1991c). Dermal uptake is generally not an important route for radionuclides, most of which have small dermal permeability constants.

Two potential air exposure routes are not presented in the conceptual model and are not recommended for quantitative evaluation under any scenario. These pathways are:

- Ingestion of contaminated particulates or volatiles secondary to inhalation of contaminated air, and
- dermal exposure to airborne contaminants.

Ingestion of contaminants is adequately evaluated by the soil ingestion pathway, especially for children who are considered to have the same potential ingestion intake during the entire year from either playing outside in contaminated soil or inside on dirty/dusty floors. Ingestion of radioactive contaminants secondary to inhalation is already accounted for by the EPA inhalation dosimetry model (EPA 1989e). Dermal exposures to airborne chemicals and radionuclides are considered to be lower than inhalation intakes and are generally not considered in Superfund Risk Assessments as noted in Sections 6.6.3

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and 10.5.5 of RAGS (EPA 1989a). Qualitative discussion of such pathways may be appropriate in the uncertainty section of the risk assessment.

Three pathways associated with external exposure to radionuclides are also not recommended for quantitative or qualitative evaluation unless site-specific information suggests the need to consider these pathways. The pathways are:

- Air immersion pathway,
- External radiation exposure from sediment, and
- Water submersion.

Exclusion of the air immersion pathway is suggested due to the transient nature of such exposures and the current lack of contaminant waste streams (e.g., the production of gaseous effluents) that would provide an air immersion exposure. External radiation exposure from contaminants in sediment or due to water submersion may also be excluded from evaluation due to the generally short durations of exposure and the shielding effects of water, as stated in Section 10.5.5 of RAGS (EPA 1989a).

If site-specific information suggests that these may be important pathways, radiation doses and risks incurred during such events may be evaluated as follows. The first step is to use dose rate conversion factors (DRFs), combined with estimates of exposure duration, to calculate committed effective doses due to radioactively contaminated media. Current sources of DRFs include DOE 1988 and EPA 1988b. The resulting dose (in units of rems) can then be multiplied by a cancer incidence risk factor to yield a cancer risk estimate. For the purposes of this methodology, a cancer induction risk factor of 6.2E-04/rem should be used, as this is the factor currently employed by EPA (1989e) to calculate radionuclide slope factors. Such an evaluation should be performed by health physicists familiar with dosimetric concepts. Justification for such an evaluation may include the detection of external radiation fields with field survey instruments, or the presence of high-energy gamma emitters in media of concern (e.g., water and sediment).

Although the conceptual model and scenarios presented in this methodology have indicated primary and secondary pathways for evaluation in risk assessments, this does not preclude the evaluation of other pathways, if in the judgement of the risk assessor, contaminants at a specific site warrant additional evaluation. Pathways that are not quantitatively evaluated should be addressed with respect to their contributions to uncertainty in exposure assessment and risk characterization, focusing on their potential impacts on overall risk.

2.2.4 Exposure Scenarios

The Hanford Site information found in subsection 2.2.1, the receptor information discussed in 2.2.2, and the pathway information and conceptual model presented in 2.2.3 are integrated to develop exposure scenarios for use in assessing risk at a site. Both current and potential future exposures are considered in accordance with 40 CFR 300.430(d)(4) and WAC 173-340-708. Current exposure estimates are used to determine the risk based on existing exposure conditions as the site. Future exposure estimates provide decision-makers with an understanding of likely future risks.

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At the request of EPA and Ecology, four scenarios have been developed for the methodology. These scenarios are a commercial/industrial scenario, a recreational scenario, a residential scenario, and an agricultural scenario. Almost all current use of the Hanford Site, except for that described under the recreational scenario, is commercial/industrial and as indicated by DOE-RL in the preface, this commercial/industrial use is expected to continue. A recreational scenario is appropriate because of the location of the Columbia River adjacent to the Hanford Site. The Columbia River is an important recreational area for fishing and other water sports, including use of the river bank along the Hanford Site up to the high water mark for waterfowl hunting. Residential and agricultural land use also occur near the Hanford Site.

Although several scenarios have been developed, the application of a specific scenario in the risk assessment should be based on site-specific information and characterization of exposed populations as discussed in Section 6.2.2 of RAGS (EPA 1989a). This is also in accordance with WAC 173-340-708. The rationale for selection of current and future scenarios should be well documented.

The scenarios are briefly discussed below. The risk assessor is referred to Appendix A for more extensive information on each scenario including exposure parameters that will be required for preparing the baseline risk assessments.

Commercial/Industrial Scenario: Because the current use of the Hanford Site is commercial/industrial, this scenario will be assessed at most sites. A site-specific commercial/industrial scenario should be developed on a case-by-case basis for use as a current scenario if industrial activities are currently conducted at a site. Site-specific exposure parameters related to type of activities (e.g., office workers, maintenance workers, etc.), frequency and duration of activities (e.g., daily, monthly, etc.), and media contact during the activities (e.g., drinking water, soil, etc.) should be applied. All site-specific data and values must be justified and documented in the risk assessment report as recommended in EPA 1991a.

A commercial/industrial scenario, presented in Appendix A, has been developed for use as a future scenario. This scenario may also be used, wherever appropriate, for evaluating current exposures when site-specific activities are similar to those represented by this scenario. The commercial/industrial scenario represents exposures that may occur to a person working at a site whose job is primarily indoors, but who would have some outside activities that could result in exposure to the soil sufficient to incur soil ingestion and dermal contact exposures. In addition, the scenario considers other commercial/industrial exposures (e.g., ingestion of potable water and inhalation of contaminated air) generally used to assess exposures associated with commercial/industrial land use, as recommended in EPA 1991a, and exposures specific to the Hanford Site and its contaminants (e.g., external exposure to radionuclides). Specific exposure parameters and factors are summarized in Tables A-1 and A-2 of Appendix A.

Recreational Scenario: A recreational scenario is provided because current recreational activities associated with the Columbia River could potentially result in exposure to hazardous substances released from the Hanford Site. As discussed above, these recreational activities include hunting, fishing, boating, water skiing, and swimming. The recreational scenario presented in Appendix A considers these current activities and incorporates additional activities, as appropriate, for a future recreational scenario. Specific

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exposure parameters and factors are summarized in Tables A-3 and A-4 of Appendix A. Revisions in the recreational scenario may be required when options under consideration for the Hanford Reach are finalized. Among the options the Hanford Reach Task Force is evaluating for the Reach are designation as a Wild and Scenic river, a National River, a National Wildlife Refuge, or a National Conservation Area.

Residential Scenario: As discussed above, there is no current residential use of Hanford Site land. However, residents are located in areas adjacent to, downwind, and down river from the site. Therefore, a residential scenario is provided in Appendix A for evaluating residential populations. Specific exposure parameters and factors for the residential scenario are summarized in Tables A-5 and A-6.

Agricultural Scenario: An agricultural scenario is also provided in Appendix A. As discussed in subsection 2.2.2, agricultural land use occurs in the vicinity of the Hanford Site. The agricultural scenario includes a farm residence. Specific exposure parameters and factors for the agricultural scenario are summarized in Tables A-7 and A-8, with qualifications.

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2.2.5 Quantification of Exposures

The exposure assessment includes a quantification of exposures for various pathways and receptors. Exposure is defined as the contact of a receptor with a chemical or physical agent. An exposure concentration (i.e., a concentration that is contacted over the exposure period) is estimated that is used with population variables (e.g., exposure parameters) and assessment variables (e.g., averaging times) to determine an intake. This subsection describes the methodology for calculating intakes that are integrated with toxicity values in the risk characterization to calculate risk.

2.2.5.1 Reasonable Maximum Exposure. The NCP and RAGS (EPA 1989a) recommend the evaluation of exposures based on reasonable maximum exposures (RME). Similarily, the MTCACR states that cleanup levels shall be based on estimates of current and future resource uses and reasonable maximum exposures expected to occur under both current and potential future site use conditions (WAC 173-340-708). The goal of calculating an RME is a result that represents an exposure scenario that is both protective and reasonable; not the worst possible case (EPA 1991a). The exposure parameters and pathways presented in this methodology are based on the RME concept presented in RAGS and the MTCACR.

2.2.5.2 Exposure Concentrations. Contaminant concentration data collected during the investigation of a site are used to estimate the exposure concentrations of contaminants in various media. The determination of an exposure concentration for use in calculating intakes is described in Section 6.4.1 of RAGS with additional information provided in Section 6.5 (EPA 1989a). Exposure concentrations are based on monitoring data using the 95 percent upper confidence limit on the arithmetic average, or the maximum concentration detected, if the upper confidence limit exceeds the maximum concentration detected. See subsection 2.2.5.4 for additional information on exposure concentrations for radioactive contaminants.

Modeled exposure concentrations may be required to provide estimates of exposure concentrations at points remote from the sources, concentrations at a future time, or

concentrations in biota. Two TPA milestones were established to support the modeling efforts that may be required for baseline risk assessments at the Hanford Site (Ecology et al. 1990). Appendix B of this document provides information on the codes and models to be used in Hanford Site risk assessments (Milestone M-29-01). These codes and models are related to the physical environment of soil and subsurface soil, groundwater, surface water and sediment, and air. They have been selected, reviewed, and recommended by a committee of Tri-Party representatives for use in support of the baseline risk assessment methodology.

A plan for development of area-wide groundwater models to support risk assessment and to evaluate impacts of changing groundwater flow fields has also been prepared. This plan was developed to support the M-29-02 TPA milestone. The groundwater is considered a major medium for the transport of contaminants to both human and ecological receptors.

Models or methods for determining exposure concentrations in food-chain biota, such as edible plants and animals, may also be required depending on the exposure scenario selected for evaluation. The methodology does not recommend any specific method or model for evaluating uptake or bioaccumulation of non-radioactive contaminants in the food chain. Food chains are very complex exposure pathways. Contaminant-specific physical and chemical information is required to estimate contaminant transport through a food chain. The scenarios presented in Appendix A provide standard exposure parameters for consumption of food-chain products such as beef, dairy products, fruit, and garden vegetables, that should be used in the exposure assessment. Chemical-specific biotransfer factors should be developed, as necessary and appropriate, on a site- and chemical-specific basis to be used in estimating concentrations of contaminants in biota.

Several sources of information are recommended by EPA Region-10 for this purpose including:

- Estimating Exposures to 2,3,7,8-TCDD (EPA 1988c),
- Methodology for Assessing Health Risks Associated with Indirect Exposure to Combustor Emissions (EPA 1990), and
- Development of a Risk Assessment Methodology for Land Application and Distribution and Marketing of Municipal Sludge (EPA 1989f).

Additional sources of useful information for evaluating the transfer of non-radioactive substances in a food chain are:

- Land Application and Distribution and Marketing of Sewage Sludge: Technical Support Document (EPA 1986a),
- Model of Organic Chemical Uptake and Clearance by Fish from Food and Water (Clark et al. 1990),

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- Plant Uptake of Non-Ionic Organic Chemicals from Soils (Ryan et al. 1988),
- Disposition of Toxic Metals in the Agricultural Food Chain. 1. Steady-State Bovine Milk Biotransfer Factors (Stevens 1991),
- Bioconcentration of Organics in Beef, Milk, and Vegetation (Travis and Arms 1988),
- The Risk Assessment of Environmental and Human Hazards: A Textbook of Case Studies (Paustenbach 1989), and
- Assessing Human Health Risks from Chemically Contaminated Fish and Shellfish: A Guidance Manual (EPA 1989g).

An example of a code that may be used for evaluating transport and biotic uptake of radioactive contaminants is the GENII computer code (Napier et al. 1988). GENII contains models to estimate biotic transport of radionuclides and to estimate dose from terrestrial exposure pathways such as aquatic food ingestion, ingestion of crops from farmlands contaminated by air transport and deposition or irrigation by contaminated water, or animal product ingestion (i.e., animals fed contaminated crops or water). The GENII code presents an analysis in terms of radiation dose, not risk. Doses can be converted into risk estimates through the proper application of risk factors as described in Section 2.2.3.2.

Any estimations of radionuclide concentrations in biota that are prepared for a risk assessment should be reviewed in relation to the results of the ongoing Hanford Site environmental monitoring program. Currently, food and farm product surveillance for radionuclides is performed for milk, vegetables (cabbage, broccoli, etc.), fruits (apples, cherries, grapes, and melons), wheat, alfalfa, beef, chicken, eggs, and wine grown in the vicinity of the Hanford Site (Jaquish and Bryce 1990). There is also sampling conducted for radionuclides in wildlife including deer, fish, upland game birds, waterfowl, and rabbits.

Another issue related to the estimation of the exposure concentration that should be considered is when the future scenario will occur. The application of future scenarios to estimate the risk associated with a site is especially important at the Hanford Site because of the presence of radionuclides. The exposure concentration will change because of the radioactive decay that occurs resulting in different estimations of contaminant activities and activities of decay daughters. The time in the future when future scenarios are applied may also affect concentrations of organic chemicals since many of these volatilize or biodegrade resulting in depletion of the contaminant source. Currently, the determination of when future scenarios occur has been be referred to TPA representatives. The position of DOE-RL on the application of furture scenarios is provided in the preface to this document.

2.2.5.3 Calculation of Non-radioactive Contaminant Intakes. Standard EPA equations for exposure and risk assessment, as provided in RAGS (EPA 1989a) and MTCACR, are used as a basis for all calculations with appropriate conversion factors, as necessary.

The basic equation for calculating intakes via ingestion (soil, water, or biota) or inhalation

is:

$$Intake = \frac{C \times IR \times EF \times ED \times CF}{BW \times AT}$$

where :

C =	Concentration of chemical in the medium
IR =	Contact rate
EF =	Exposure frequency (d/yr)
ED =	Exposure duration (yr)
CF =	Conversion factor (as appropriate)
BW =	Body weight (kg)
AT =	Averaging time (yr x 365 d/yr)
Intake =	Contaminant-specific intake (mg/kg-d)

The basic intake equation is modified to provide an equation for calculating the absorbed dose resulting from dermal exposure to contaminated water as follows:

$$Dermally\ absorbed\ dose = \frac{CW \times SA \times Kp \times ET \times EF \times ED \times CF}{BW \times AT}$$

where :

CW =	Concentration of chemical in water
SA =	Skin surface area available for contact (cm^2)
Kp =	Chemical-specific permeability coefficient (cm/hr)
ET =	Event time (hr/d)
EF =	Exposure frequency (d/yr)
ED =	Exposure duration (yr)
CF =	Conversion factor (1 L/1000 cm^3)
BW =	Body weight (kg)
AT =	Averaging time (yr x 365 d/yr)
Dermally absorbed dose =	(mg/kg-d)

The intake equation is also modified to provide the absorbed dose equation for dermal exposures to contaminated soil:

$$Dermally\ absorbed\ dose = \frac{CS \times SA \times AF \times ABS \times EF \times ED \times CF}{BW \times AT}$$

where :

CS =	Concentration of chemical in soil
SA =	Skin surface area available for contact (cm^2)
AF =	Soil-to-skin adherence factor (mg/cm^2/event)
ABS =	Chemical-specific absorption factor (unitless)
EF =	Event frequency (events/yr)

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ED =	Exposure duration (yr)
CF =	Conversion factor (1E-06 kg/mg)
BW =	Body weight (kg)
AT =	Averaging time (yr x 365 d/yr)
Dermally absorbed dose =	(mg/kg-d)

The exposure intakes for the contaminants of potential concern should be presented in tabular form in the baseline risk assessment. Exhibit 6-22 in RAGS (EPA 1989a) provides an example of how these may be presented.

2.2.5.4 Calculation of Radioactive Contaminant Intakes. The quantification of exposures to radioactive contaminants requires a separate treatment. The units used to express environmental concentrations of radioactive and non-radioactive contaminants are different. Unlike non-radioactive contaminants, intake estimates for radionuclides should not be divided by body weight or averaging time. The calculated intakes, therefore, represent radionuclide activities inhaled or ingested over a lifetime, or the lifetime-averaged contact with contaminated soils.

This baseline risk assessment methodology recommends the use of HEAST (EPA 1991d) or subsequent editions as the method for calculating lifetime cancer induction risks from radioactive contaminant exposures. Standard EPA equations for exposure and risk assessment, as provided in EPA 1991c, are used as the basis for all HEAST methodology calculations with appropriate conversion factors, as necessary.

The basic equation for calculating intakes via ingestion (water, soil, biota) or inhalation is:

$$\text{Intake} = C \times IR \times EF \times ED$$

where:

Intake =	Radionuclide-Specific lifetime intake (pCi)
C =	Concentration of radionuclide in media
IR =	Contact rate
EF =	Exposure frequency (d/yr)
ED =	Exposure duration (yr)

Typical units for radionuclide concentrations and contact rates are pCi/m^3 and m^3/d for air, and pCi/L and L/d for water. The potentially transient nature of some ingested media (e.g., surface water) requires that an evaluation be made as to the average radionuclide concentration during the exposure duration in question.

The above equation may also be used for the evaluation of external exposures. In this case, the "intake" would have the units of pCi-yr/g, environmental (i.e., soil) concentrations would be measured in units of pCi/g, and the "contact rate" would be determined as follows:

$$IR_{ext} = ET \times RF \times CF$$

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where:

IR_{ext}	=	External exposure contact rate (yr/d)
ET	=	Exposure Time (hr/d)
RF	=	Dose reduction factor (0.6, unitless)
CF	=	Conversion factor (1.14E-04 yr/hr)

An external exposure pathway must be evaluated for photon (gamma and x-ray) emitting radioactive contaminants, due to their ability to penetrate and impart a dose to internal tissues. Radionuclides that have negligible photon emissions do not need to be considered in the external exposure pathway. External exposure slope factors are currently based on the assumption of uniform radionuclide distribution in soil. Therefore, choosing the maximum detected soil concentration to represent the soil contamination profile is likely to introduce conservatism which should be noted in the uncertainty analysis.

If soil contamination is deep enough, the soil may provide sufficient shielding to effectively eliminate an external exposure. This shielding effect is a function of photon energy and source activity. However, it may be generally assumed that all low energy (<400 keV) photons are effectively shielded if the source is at least 1 meter deep.

Consideration must be made for the loss of radioactive contaminants either through decay or migration into or away from the site in order to determine a time-averaged radionuclide concentration for calculation of external exposures. This information will be provided by appropriate modeling of contaminants through the environment (see 2.2.5.2 and Appendix B). This external exposure analysis does not include any contributions from radioactive daughter products, which are evaluated as any other radioactive contaminant.

A dose reduction factor is used to obtain a more realistic estimate of external exposures by taking into account the effects of shielding while indoors, ground roughness, and time spent indoors. For the time assumed to be spent outdoors (50%), a factor of 0.7 is used to account for the shielding provided by ground roughness. For the time assumed to be spent indoors (50%), a factor of 0.5 is used to account for the shielding provided by building structures. Guidance regarding the choice and use of these factors is provided in DOE 1988. The resulting effective dose reduction factor is:

$$RF = (0.5 \times 0.7) + (0.5 \times 0.5) = 0.6$$

This dose reduction factor is most appropriate for land use scenarios in which the individual is mostly outdoors (agricultural and recreational scenarios), but would be conservative for those who spend most of their time indoors, as in residential and industrial scenarios. However, this factor is not very sensitive to time spent indoors vs. outdoors, so the use of a single value (0.6) is recommended.

Once the total intake and time-averaged soil concentrations have been determined as above, lifetime cancer incidence risks associated with these exposure pathways are calculated using HEAST radionuclide slope factors (see Section 2.3.3.2).

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2.2.6 Uncertainty Evaluation For Exposure Assessment

The exposure assessment requires multiple assumptions that can significantly impact the outcome of a risk assessment. Key factors contributing to uncertainty in the exposure assessment are related to definition of the physical setting, applicability of models and assumptions, and uncertainty in the value of parameters used in the exposure assessment. Consideration in the uncertainty section may include discussion on:

- Adequate identification of current land use,
- Likelihood of future land use actually occurring,
- Exclusion of chemicals from the quantitative risk assessment,
- Exclusion of pathways from the quantitative risk assessment,
- Model assumptions that impact exposure point concentrations,
- Use of the upper 95% confidence limit as the exposure point concentration,
- Use of standard default parameters,
- Uncertainty related to site-specific parameters, and
- Uncertainty related to biotransfer factors.

Discussion of these factors and other assumptions, as appropriate for a site, should include the potential for over estimating or under estimating exposures based on the possible uncertainties stated above.

2.3 TOXICITY ASSESSMENT

The purpose of the toxicity assessment is to identify the potential adverse effects associated with exposure to site-related substances and to estimate, using numerical toxicity values, the likelihood that these adverse effects may occur based on the extent of the exposure. The toxicity assessment for Hanford Site risk assessments is conducted in accordance with EPA 1989a and EPA-10 1991. Supplemental information is provided on the toxicity assessment for radionuclides. Chapter 7 of RAGS (EPA 1989a) and the regional guidance (EPA-10 1991) should be consulted for additional information on the toxicity assessment.

The toxicity assessment component of the human health evaluation is presented below. Sources of toxicity information are discussed in subsection 2.3.1. Subsections 2.3.2 and 2.3.3 present information on noncarcinogenic and carcinogenic toxicity values, respectively, for both non-radioactive and radioactive substances. Substances without toxicity values or without route-specific toxicity values are discussed in subsection 2.3.4. A discussion of the toxicity profile is presented in subsection 2.3.5 and the evaluation of uncertainty in the toxicity assessment is discussed in subsection 2.3.6.

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Extensive toxicological and radiological information has been published and toxicity values are readily available. However, it is recommended that this portion of the risk assessment be performed by individuals with training, experience, and expertise in evaluating toxicological and radiological data.

2.3.1 Sources of Toxicity Information

The preparation of the toxicity assessment relies primarily on existing toxicity information, and does not usually involve development of toxicity information or dose-response relationships. Toxicological and radiological information that is already evaluated and summarized is available in a number of documents, databases, or other sources. Information on general sources for toxicity information is provided in subsection 2.3.1.1 and the hierarchy of sources for numerical toxicity values is discussed in subsection 2.3.1.2.

2.3.1.1 General Toxicity Information. The toxicity assessment should include information regarding the toxic effects of contaminants identified at a waste site. Sources of general toxicity information for risk assessment may include, but are not limited to, the following:

- Agency for Toxic Substances Disease Registry (ATSDR) Toxicological Profiles
- Casarett and Doull's Toxicology: The Basic Science of Poisons (Amdur et al. 1991) - [includes a comprehensive chapter on radiation and radionuclides]
- Chemical Hazards in the Workplace (Proctor et al. 1988)
- Clinical Toxicology of Commercial Products (Gosselin et al. 1984)
- Dangerous Properties of Industrial Materials (Sax 1984)
- Disposition of Toxic Drugs and Chemicals in Man (Baselt et al. 1989)
- EPA contaminant-specific documents including: Health Assessment Documents, Health Effects Assessments, Health and Environmental Effects Assessments, and Health and Environmental Effects Profiles,
- Hazardous Substances Database
- The Merck Index (Windholz et al. 1983)
- Patty's Industrial Hygiene and Toxicology (Clayton and Clayton 1981)
- Toxicological Chemistry (Manahan 1989)
- Health Effects of Exposure to Low Levels of Ionizing Radiation, BEIR V (NRC 1990)
- NESHAPs for Radionuclides - Background Information Document

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Volume 1 (EPA 1989e)

- International Commission on Radiological Protection, Publication 30 (ICRP 1979-1988)

2.3.1.2 Numerical Toxicity Values. The hierarchy of sources for numerical toxicity values used in Hanford Site risk assessments is provided below.

The Integrated Risk Information System (IRIS), the EPA's on-line database, is the preferred source for numerical toxicity values and toxicity information. This system provides chemical-specific slope factors, weight-of-evidence classifications, reference doses, and supporting discussion and references for this information. The slope factors and reference doses have been reviewed and verified by agency-wide work groups.

If toxicity information for a chemical or radionuclide is not available in IRIS, the risk assessor should consult the HEAST (EPA 1991b) or subsequent editions. The HEAST tables provide a summary of all currently available toxicity factors developed by the Environmental Criteria and Assessment Office (ECAO) for the Office of Emergency and Remedial Response, many of which are not yet available in IRIS. However, certain toxicity values available in HEAST may not be verified and may still be undergoing work-group review within EPA. These tables also provide pathway specific slope factors for many radionuclides.

When numerical toxicity values are not available from IRIS or HEAST, alternative methods for developing toxicity values or for evaluating compounds may be used when appropriate. Subsection 2.3.4 below discusses how to evaluate substances without IRIS or HEAST toxicity values.

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2.3.2 Toxicity Values for Noncarcinogenic Substances

Systemic, toxic effects other than cancer can be associated with exposures to both chemicals and radionuclides. Subsections 2.3.2.1 and 2.3.2.1 discuss the toxicity values and the approach for evaluating noncarcinogenic effects that may occur from exposure to non-radioactive substances and radioactive substances, respectively.

2.3.2.1 Non-radioactive Substances. The reference dose (RfD) is the toxicity value used to evaluate noncarcinogenic effects resulting from exposures to chemicals or radionuclides. The RfD has been developed based on the concept that protective mechanisms exist that must be overcome before an adverse effect is manifested (i.e., there is a threshold which must be reached before adverse effects occur). The RfD is developed to reflect the duration of exposure (e.g., subchronic exposures - 2 weeks to 7 years and chronic exposures - 7 years to a lifetime) and the route of exposure (e.g., inhalation, oral , etc.). In addition, RfDs are currently being developed, as appropriate, to evaluate specific critical effects such as developmental effects that may occur because of exposure to certain chemicals.

The subchronic RfD is utilized to evaluate potential noncarcinogenic effects from exposures that occur because activities are performed for a limited amount of time (e.g., during remediation activities) or when a substance with a short half-life degrades to negligible concentrations within several months. For longer exposures, the chronic RfD is

utilized to evaluate potential noncarcinogenic effects. The chronic RfD is a daily exposure level that is likely to be without an appreciable risk of deleterious effects to the general population, including sensitive subpopulations, during a lifetime.

The toxicity reference values for the inhalation pathway are currently being verified as reference concentrations (RfCs). The RfC can be converted to an RfD for use in risk characterization using the following conversion (EPA-10-1991):

$$RfD = \frac{RfC\ (mg/m^3) \times 20\ m^3/day}{70\ kg}$$

Because carcinogens can also have systemic effects other than cancer, carcinogenic substances should be evaluated for noncarcinogenic adverse effects. However, carcinogenic effects usually occur at levels significantly lower than those associated with systemic toxic effects; thus, cancer risk is usually the predominant adverse effect for carcinogens.

2.3.2.2 Radioactive Substances. Radiation-induced health effects can be classified as stochastic or non-stochastic (i.e, acute toxicity). Stochastic effects are those for which induction is probabilistic, and that probability is a function of the absorbed dose. In addition, there is generally believed to be no threshold dose below which a stochastic health effect cannot be induced, nor is there a dose above which such an effect is guaranteed. Examples of stochastic health effects include carcinogenesis, mutagenesis, teratogenesis, and life shortening.

Acute toxicity effects are those which have a threshold dose, and will occur if that threshold is exceeded. Examples include hematological changes, cataracts of the lens of the eye, erythema, and acute radiation syndromes. As stated in RAGS (EPA 1989a), Section 10.5.3:

> In general, radiation exposure assessments need not consider acute toxicity effects. Acute exposures are of less concern for radionuclides than for chemicals because the quantities of radionuclides required to cause adverse effects from acute exposure are extremely large, and such levels are not normally encountered at Superfund sites.

Stochastic but noncarcinogenic health effects include genetic mutations, birth defects, and life shortening. Several current references (NRC 1990; EPA 1989e) provide risk factors for these effects. BEIR V (NRC 1990) considers that limiting exposure to reduce cancer risk also limits genetically significant exposure. RAGS (EPA 1989a) states that the risk of cancer appears to be limiting, and may be used as the sole basis for assessing the radiation related human health risks of a site contaminated with radionuclides. Thus, for Hanford baseline risk assessments, it is recommended that only carcinogenic effects be routinely evaluated for radionuclides, as carcinogenesis is the predominant adverse human health effect.

It is important to distinguish the carcinogenic potential of radiation (in which a radionuclide is only the delivering agent) from the chemical toxicity of these elements and their compounds (e.g., Sr-90 vs. strontium salts or Pb-210 vs. lead). The internally committed quantities that pose a significant radiation-induced cancer risk generally have an

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insignificant chemical toxicity. Some exceptions may occur (e.g., the nephrotoxic effects of uranium) and will be evaluated on a case-by-case basis.

2.3.3 Toxicity Values For Carcinogenic Substances

Toxicity values have also been developed for evaluating potential human carcinogenic effects from exposures to chemicals and radionuclides. Subsection 2.3.3.1 discusses the toxicity values utilized in evaluating non-radioactive substances. Subsection 2.3.3.2 discusses the evaluation of carcinogenic effects from exposure to radioactive substances.

2.3.3.1 Non-radioactive Substances. Potential human carcinogenic effects are evaluated using the chemical-specific slope factor and accompanying EPA weight-of evidence determination. The toxicity values (i.e, slope factors) for carcinogens have been derived based on the concept that for any exposure to a carcinogenic chemical there is always a carcinogenic response (i.e., there is no threshold). The slope factor is used in risk assessment to estimate an upper-bound lifetime probability of an individual developing cancer as a result of exposure to a particular level of a potential carcinogen. In addition to the slope factor, the likelihood that a substance is a human carcinogen is also considered. A weight-of-evidence classification is assigned to each substance based on the strength of human and animal evidence of carcinogenicity. The EPA weight-of evidence classifications are:

- Group A — Human Carcinogen
- Group B — Probable Human Carcinogen
- Group C — Possible Human Carcinogen
- Group D — Not Classifiable as to Human Carcinogenicity
- Group E — Evidence of Non-Carcinogenicity in Humans

Toxicity values for carcinogens can also be expressed in terms of risk per unit concentration of the substance in the medium where contact occurs. Unit risk values may be found in IRIS and HEAST and are medium-specific. As discussed above for RfDs, slope factors and unit risks are specific for the route of exposure. For non-radioactive substances, oral or inhalation slope factors and unit risks are used as appropriate for the pathway.

Currently, the toxicity reference values for the inhalation pathway are being verified in units of concentration in air. These unit risks must be converted to inhalation slope factors for use in risk characterization. The recommended conversion (EPA-10 1991) is:

$$\textit{Slope factor} = \frac{\textit{Unit Risk } (\mu g/m^3)^{-1} \times 70\ kg \times 1E\text{-}03\ mg/\mu g}{20\ m^3/day}$$

The toxicity values and supporting information for carcinogenic substances carried through the risk assessment should be summarized in tabular form in the toxicity assessment. Examples of table formats that may be used are presented in Section 7 of RAGS (EPA 1989a).

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2.3.3.2 Radioactive Substances. Since all radionuclides are classified by EPA as Group A - Human Carcinogens, further consideration of weight-of-evidence for radionuclides is not necessary. In addition, cancer risk may be used as the sole basis for assessing the radiation related human health risks of a site contaminated with radionuclides (EPA 1989a).

HEAST provides slope factors for radionuclides for ingestion, inhalation, and external exposure pathways. These values are used to determine the lifetime incremental cancer induction risks associated with environmental concentrations of radionuclides (see Section 2.4.1). Currently, each radionuclide slope factor is calculated for a single default lung class and gastro-intestinal absorption factor. These factors were chosen to reflect those chemical forms of radionuclides that a receptor would expect to encounter through environmental contamination. Determination of the specific chemical form of a radionuclide may indicate that the slope factor is biased, while lack of such information is a source of uncertainty. HEAST users should consult EPA 1989e for a more detailed discussion of current EPA radiation risk assessment methodology. Included in this document are the risk factors used to calculate radionuclide slope factors presented in HEAST. The nominal lifetime cancer induction risk factor is 6.2E-04/rem.

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2.3.4 Evaluating Substances Without Toxicity Values

EPA-derived toxicity values may not be available for all substances and all routes of exposure. In these cases, the type of substance, the extent of contamination detected, the contaminated media, the potential mobility, persistence, and toxicity of the substance, and implications of not quantitatively evaluating the substance should be reviewed. Professional judgement may be used to select an alternative method to evaluate the substance. However, the rationale for selecting an alternative should always be well documented. Examples of several alternatives that can be considered for evaluating substances without specific toxicity values are provided below.

Toxicity values may be developed by, or in consultation with, the Superfund Technical Support Center at the ECAO office in Cincinnati on a case-by-case basis. However, EPA Region 10 risk assessment staff should be consulted prior to contacting ECAO (EPA-10 1991). More current information may be available or similar derivations may be available from other risk assessments conducted in Region 10.

ATSDR minimum risk levels (MRLs) are available for some substances. The MRLs have been developed consistent with reference dose methodology and may be useful for evaluating short-term exposures (EPA-10 1991). Use of specific ATSDR MRLs should also be discussed with the EPA Region 10 risk assessment group.

Substances with established toxicity values may be considered for use as surrogates for substances that do not have published toxicity values. The use of a surrogate may be appropriate for evaluating substances that have similar documented toxic effects in the same target organ or similar mechanisms of toxic action.

Toxicity values for specific substances may not be available for potential routes of exposure that are evaluated (e.g., dermal exposure), but may be available from another exposure route (e.g., oral or inhalation). In general, route-to-route extrapolation of toxicity values is not recommended except for using oral toxicity values, adjusted for absorbed

dose, to evaluate dermal exposures (EPA 1989a). Any extrapolation between routes of exposure should consider toxic effects that are localized or are dependent on the route of exposure.

Substances may also be qualitatively evaluated in place of a quantitative evaluation. The potential impact of either a qualitative evaluation of a substance or the use of a derived toxicity value in the absence of a published value, should always be discussed in the uncertainty section of the toxicity assessment. This discussion should address the implications of the absence of the quantitative evaluation on the risk estimate. It should also address, as appropriate, the impact from the utilization of non-verified toxicity values.

2.3.5 Toxicity Profiles

The baseline risk assessment should provide toxicity information for each contaminant of potential concern carried through the full assessment. A short toxicity profile may be provided in the toxicity assessment section or in an appendix, but should provide toxicity information that is concise and directed toward the non-technical reader. It is recommended that acute and chronic toxic effects should be distinguished, and the level of confidence in the studies providing the toxicity information stated. A brief summary of the supporting information for the toxicity values used in the assessment should be included as part of the toxicity profile. This information includes critical effects and target organs, uncertainty factors, and other relevant information.
IRIS provides valid and useful data for the toxicity profiles. The ATSDR toxicological profiles are good examples of informative yet readable toxicity discussions.

2.3.6 Uncertainty Evaluation

Uncertainty is associated with the toxicity values and toxicity information available to assess potential adverse effects. This uncertainty in the information and the lack of specific toxicity information contribute to uncertainty in the toxicity assessment. The types of uncertainty that should be discussed in the toxicity assessment are provided in the following subsections. Additional information on uncertainty in toxicity assessments can be found in Chapters 7 and 8 of EPA 1989a.

2.3.6.1 Uncertainty in Toxicity Values and Information. An understanding of the degree of uncertainty associated with toxicity values is an important part of interpreting and using those values. A high degree of uncertainty in the information used to derive a toxicity value contributes to less confidence in the assessment of risk associated with exposures to a substance. Sources of uncertainty associated with published toxicity values may include:

- Use of dose-response information from effects observed at high doses to predict the adverse health effects that may occur following exposure to the low levels expected from human contact with the agent in the environment,

- Use of data from short-term exposure studies to extrapolate to long term exposures or vice-versa,

- Use of data from animal studies to predict human effects, and
- Use of data from homogenous animal populations or healthy human populations to predict effects in the general population.

For noncarcinogenic substances, the toxicity values (RfDs) have uncertainty factors calculated into the value for each of the sources of uncertainty listed above. A factor of 10 is usually used for each of the sources of uncertainty listed above.

In addition, the quality of the studies, the consistency of results between different studies, the biological plausibility of the toxic effect, and the completeness of the supporting data base contribute to confidence in the toxicity value and the likelihood that a substance will cause an adverse effect. Such information should be included in the discussion of uncertainty regarding the toxic effects of substances carried through the risk assessment.

Radiation exposure data has a distinct advantage over most chemical exposure data in that it is largely derived from human epidemiological studies, eliminating the uncertainty that arises when relating data gathered from animal studies to humans. However, the use of this human radiation exposure data has its own particular and significant sources of uncertainty. The most important of these sources include: the extrapolation of risks observed in populations exposed to relatively high doses, delivered acutely, to populations receiving relatively low dose chronic exposures; selection of an appropriate risk projection model; application of cancer risks derived for one population (Japanese) to another (U.S.); and statistical uncertainties. The last of these, introduced through sample variation, is possibly the most significant. In accounting for all these sources of uncertainty, risk factors for cancer incidence span an order of magnitude.

In addition, the chemical form of a radionuclide has significant implications regarding uptake after inhalation or ingestion. As is often the case, such information is not available, and it is not possible to know whether a radionuclide slope factor is conservatively or liberally biased. The importance of this issue depends on the radionuclide in question, and such a determination should be made by persons with training in dosimetric modeling.

2.3.6.2 Uncertainty in the Toxicity Assessment. Uncertainty is also present in the overall toxicity assessment because of

- Uncertainty in the toxicity information of individual substances;
- Potential synergistic or antagonistic interactions of substances;
- Evaluation of substances that do not have toxicity values through qualitative discussion, use of surrogates, route-to-route extrapolation of toxicity values, etc.

The overall confidence in the toxicity assessment should be discussed regarding these uncertainties and other site- or substance-specific considerations.

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2.4 RISK CHARACTERIZATION

The information from the exposure assessment and the toxicity assessment is integrated to form the basis for the characterization of risks and human health hazards. The risk characterization presents quantitative and qualitative descriptions of risk. As stated in RAGS (EPA 1989a), "A risk characterization cannot be considered complete unless the numerical expressions of risk are accompanied by explanatory text interpreting and qualifying the results". Thus, the risk characterization serves as the bridge between risk assessment and risk management and is a key step in the ultimate decision making process.

The following subsections describe the risk characterization methodology. Carcinogenic risk characterization is presented in subsection 2.4.1, noncarcinogenic risk characterization is presented in 2.4.2, and assessment and presentation of uncertainty is discussed in 2.4.3. Use of site-specific human studies is discussed in 2.4.4 and a risk characterization summary is presented in 2.4.5. Additional information on risk characterization can be found in Section 8 of RAGS (EPA 1989a) and in Region 10 guidance (EPA-10 1991).

2.4.1 Quantification of Carcinogenic Risk

For carcinogens, risks are estimated as the likelihood of an individual developing cancer over a lifetime as a result of exposure to a potential carcinogen (i.e., incremental or excess individual lifetime cancer risk). The slope factor converts daily intakes averaged over a lifetime of exposure, as derived in the exposure assessment, to the estimated incremental lifetime risk of an individual developing cancer. The equation for risk estimation is:

$$\text{Risk} = (\text{Chronic Daily Intake})\ (\text{Slope Factor}).$$

This linear equation is only valid at low risk levels (i.e., below estimated risks of 1E-02), and is an upperbound estimate based on the upper 95th percent confidence limit of the slope of the dose-response curve. Thus, one can be reasonably confident that the actual risk is likely to be less than that predicted. Cancer risk estimates should be expressed using one significant figure only.

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Absorption adjustments may be required in the risk characterization to ensure that the site exposure estimates and the slope factors are both expressed as absorbed doses or as intakes. Appendix A of RAGS (EPA 1989a) describes how adjustments for absorption efficiency should be made.

Estimations of carcinogenic risk for individual chemicals are calculated as described above. Cancer risk is assumed to be additive, and risks from different chemicals and pathways can be summed, as appropriate, to evaluate the overall cancer risk posed by chemicals at a site (see discussion below relating to the summation of estimated chemical cancer risks and estimated radionuclide cancer risks). In addition to the chemical-specific risk, the pathway-specific risk and the total site risk should be calculated for chemical carcinogens present at a site that impact the same receptor. Because scenarios have been defined for Hanford Site risk assessments, pathways that should be summed have already been defined.

When using HEAST methodology, carcinogenic risk associated with radionuclides is calculated as above for chemical carcinogens, except that intakes and slope factors are defined to represent lifetime (not daily) exposures. The environmental concentrations and intake and exposure factors are combined, as in Section 2.2.5.4, to provide intake values and time-averaged concentrations (in units of pCi, and (pCi-yr/g), respectively). These values are then multiplied by the appropriate slope factors provided in HEAST to yield the lifetime incremental cancer incidence risk.

In spite of the fact that risks from exposure to chemicals and radionuclides can be quantified, the models, assumptions, and data used to estimate these risks are very different. As stated in Section 10.7.3 of RAGS (EPA 1989a):

> One important difference of this nature is how the cancer toxicity values (i.e., slope factors) were developed. For both radionuclides and chemicals, cancer toxicity values are obtained by extrapolation from experimental and epidemiological data. For radionuclides, however, human epidemiological data form the basis of the extrapolation, while for many chemical carcinogens, laboratory experiments are the primary basis for the extrapolation. Another even more fundamental difference between the two is that slope factors for chemical carcinogens generally represent an upper bound or 95th percent confidence limit value, while radionuclide slope factors are best estimate values. In light of these limitations, the two sets of risk estimates should be tabulated separately in the final baseline risk assessment.

2.4.2 Quantification of Noncarcinogenic Risk

Potential human health hazards associated with exposure to noncarcinogenic substances, or carcinogenic substances with systemic toxicities other than cancer, are evaluated differently from carcinogenic risks. The daily intake over a specified time period (e.g., lifetime or some shorter time period) is compared to an RfD for a similar time period (e.g., chronic RfD or subchronic RfD) to determine a ratio called the hazard quotient. Estimates of intakes for Hanford Site baseline risk assessments will usually be determined for chronic exposures. The nature of the contaminant sources and the potential for release

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of contaminants from a site preclude short-term fluctuations in contaminant concentrations that might produce acute or subchronic effects. However, the risk assessor should be aware of site-specific information that would suggest acute or sub-chronic exposures may occur and risks should be appropriately evaluated using the chronic or subchronic RfD.

The formula for estimation of the hazard quotient is:

$$\text{Hazard Quotient} = \frac{\text{Daily Intake}}{\text{RfD}}$$

Absorption adjustments may be required in the risk characterization to ensure that the site exposure estimates and RfDs are both expressed as absorbed doses or as intakes. Appendix A of RAGS (EPA 1989a) describes how adjustments for absorption efficiency should be made.

If the hazard quotient exceeds unity, the possibility exists for systemic toxic effects. The hazard quotient is not a mathematical prediction of the severity or incidence of the effects, but rather is an indication that effects may occur, especially in sensitive subpopulations. The chemical-specific hazard quotients can be summed to determine a hazard index for a pathway or a site (based on the same scenario). If a hazard index exceeds unity, an evaluation of the specific substances should be performed so that only substances with similar systemic toxic effects (i.e., similar effects in the same target organs via the same mechanism) are summed. Section 8.2.2 of RAGS (EPA 1989a) discusses this segregation of substances by effect and mechanism of action more extensively.

In general, noncarcinogenic risks posed by radionuclides do not need to be evaluated (see Section 2.3.2.2). However, the chemical toxicity of some radionuclides (e.g., uranium) may be significant when compared to the hazard posed by their radioactive characteristics, and will be evaluated on a case-by-case basis.

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2.4.3 Assessment and Presentation of Uncertainty

The risks, both carcinogenic and noncarcinogenic, presented in a risk assessment are not fully probabilistic estimates of risks, but rather are conditional estimates given multiple assumptions about exposures, toxicity, and other variables. Therefore, at a minimum, a qualitative discussion of uncertainty should be provided in all risk assessments performed for the Hanford Site to place the risk estimates in proper perspective. The uncertainty discussion should consider both uncertainties inherent in the risk assessment process (e.g., toxicity values, default exposure parameters) and uncertainties specific to a project (e.g., data evaluation, contaminant identification). Specific considerations in evaluating uncertainty are discussed above in 2.2.6 and 2.3.6 and in the following subsections. The risk assessor should also consult EPA 1989a and EPA-10 1991 for additional discussion on uncertainty considerations.

2.4.3.1 Site-specific Uncertainty Factors. Uncertainty related to the likelihood that site contaminants and concentrations of those contaminants detected are representative of the site should be discussed. Data collection (e.g., sampling plans, sample quality control, analytical limits, etc.) and data evaluation factors (e.g., data validation considerations, tentatively identified compounds (TICs), etc.) that can influence the risk assessment results

should also be included. Consideration of specific site characteristics, availability of information on site-specific environmental conditions, and uncertainties in model application to the site should also be evaluated for their impact on over- or underestimating the risk associated with a site.

2.4.3.2 Exposure Assessment Uncertainty Factors. A discussion of important assumptions used in the exposure assessment should be included as part of the uncertainty discussion of the risk characterization. Subsection 2.2.6 provides information on the uncertainty evaluation that should be summarized in the overall uncertainty in the risk associated with a site. The multiple assumptions made in the exposure assessment can significantly impact the risk assessment results.

2.4.3.3 Toxicity Assessment Uncertainty Factors. Factors related to uncertainty in the toxicity assessment are presented above in subsection 2.3.6 and should be summarized as part of the uncertainty section of the risk characterization. The uncertainty related to the toxicity information and values used in the risk assessment is especially important for those substances carried through the quantitative assessment that contribute most of the estimated risk. The weight of evidence for the carcinogenic substances and the confidence in the database supporting noncarcinogenic effects should be included in the uncertainty discussion.

The uncertainty contributed by not quantitatively evaluating substances in the risk assessment because of inadequate toxicity information should also be presented in the uncertainty section of the risk characterization. The possible consequences of excluding substances and impacts on the overall estimate of risk for a site should also be evaluated.

2.4.3.4 Risk Characterization Uncertainty Factors. The summation of cancer risks across pathways or for multiple pathways may make the total cancer risk estimate artificially more conservative. This is because each slope factor is an upper 95th percentile estimate and such probability distributions are not strictly additive. Also, summing risks from all carcinogens gives equal weight to slope factors derived from animal data and slope factors derived from human data.

Also, summing risks of various carcinogens does not account for relative uncertainty between slope factors derived from animal data and slope factors derived from human data.

Similarly, for noncarcinogenic substances, the assumption of dose additivity is not always appropriate because substances may have different effects in different target organs. In addition, the confidence in the RfD databases and the severity of effects associated with exposures greater than the RfD may vary. Summing all hazard quotients gives equal weight to critical effects of varying toxicological significance.

Consideration of these uncertainty factors in the risk characterization should be included in discussing the uncertainty of the final risk estimates for a site.

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2.4.4 Consideration of Other Site-Specific Human Studies

RAGS (EPA 1989a) recommends consideration of site-specific human studies that may be available to aid in evaluating the estimates of risk associated with a site. This aspect of the risk characterization is especially important for the Hanford Site where epidemiological studies have been conducted to evaluate potential exposures to radionuclides. Little information is available with respect to site-specific human health studies of potential exposures to non-radioactive contaminants.

Consideration should also be given to a comparison of information presented in the risk assessment with respect to the on-going surveillance that occurs at the Hanford Site. Radionuclide concentrations are routinely measured in food and farm products and wildlife. The Columbia River is monitored for water quality parameters and radionuclides. Also, air monitoring and radiation surveys are conducted. Information on radioactive dose exposures based on surveillance information is also available (Jaquish and Bryce 1990). Any use of on-going surveillance studies or results should be carefully evaluated for its applicability to the baseline risk assessment process and specific risk assessments.

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2.4.5 Risk Characterization Summary

The risk characterization should include a final discussion to place the numerical estimates of risk in the context of what is known and what is not known about the site. This discussion should include:

- confidence that key site-related contaminants have been identified,
- description of known or predicted health risks (cancer and noncarcinogenic effects),
- confidence in the toxicity information supporting the risk estimations,
- confidence in the exposure assessment estimates,
- magnitude of the cancer risks relative to the site-remediation goals (i.e., NCP or MTCACR),
- major factors driving the risks including contaminants, pathways, and scenarios, and
- the uncertainty associated with the results.

The risk characterization summary should include tables to display risk information in addition to the text. Examples of table format are provided in Section 8 of RAGS (EPA 1989a).

The risk characterization provides information to aid remedial project managers in making decisions regarding a site. It is the responsibility of the risk assessment team members, who are familiar with all of the steps involved in the site risk assessment, to highlight the major conclusions of the risk assessment. References to risks as significant or

insignificant, acceptable or unacceptable, should not be made unless the use of these terms is specifically defined. The results of the human health evaluation and the environmental evaluation are provided separately. However, an overall summary of both may be provided in the final risk characterization summary for the site risk assessment.

2.5 SUMMARY OF HUMAN HEALTH EVALUATION

The human health evaluation methdology presented in Section 2.0 integrates federal, regional, and state requirements with Hanford Site-specific information to provide a framework for conducting baseline risk assessments at the Hanford Site. The human health evaluation of the baseline risk assessment should be conducted by individuals with experience in the technical and regulatory aspects and limitations of risk assessment.

The process to identify contaminants of potential concern for Hanford Site baseline risk assessments utilizes the numerous site-specific procedures for data collection, sampling and analysis, data evaluation and validation, and site characterization to focus the efforts of the risk assessor in this step. The identification of contaminants of potential concern also includes the use of a conservative preliminary risk-based screening, as recommended by EPA Region-10, to focus the overall baseline risk assessment process.

The exposure assessment provides four potential exposure scenarios as requested by EPA and Ecology: a commercial/industrial scenario, a recreational scenario, a residential scenario, and an agricultural scenario. A general conceptual model for exposure assessment is provided in Figure 2-4 and includes the primary and secondary pathways that should be considered for each scenario. The exposure parameters incorporate those recommended in a potential state ARAR, the MTCACR, with site-specific and EPA standard default exposure parameters. Appendix A provides the main body of information on the assumptions for each scenario and the exposure parameters that should be used. The exposure assessment also presents information on evaluating exposures to radionuclides because of the use and disposal of radioactive materials at the Hanford Site.

The toxicity assessment provides general information for conducting a toxicity assessment and provides supplemental information on evaluating toxicity associated with radioactive contaminants.

The risk characterization is the final step in the human health evaluation. For Hanford Site risk assessments, this step is conducted as recommended in RAGS (EPA 1989a). Toxicity information and exposure assessment information is integrated to quantify the cancer risks and hazard quotients/indices. The risk characterization for Hanford Site baseline risk assessments includes consideration of carcinogenic risks related to radioactive contaminants and non-radioactive contaminants. An essential element of the risk characterization is a discussion of the uncertainty associated with the results of the risk assessment.

The human health evaluation methodology provides guidance for conducting baseline risk assessments at the Hanford Site. Within the methodology, there is also a recognition that risk assessment is a dynamic, evolving process that may require the use of professional judgement during the preparation of a site-specific risk assessment.

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3.0 ENVIRONMENTAL EVALUATION METHODOLOGY

A general methodology for baseline environmental evaluations is provided below. Given the relatively recent regulatory emphasis on environmental evaluations (as opposed to human health evaluations), a discussion of the overall framework is presented in Section 3.1. The three major elements of the process:

- Problem definition,
- Analysis, and
- Risk characterization,

are then presented in Sections 3.2, 3.3, and 3.4, respectively.

3.1 FRAMEWORK FOR CONDUCTING BASELINE ENVIRONMENTAL EVALUATIONS

The regulatory goal for the environmental evaluation component of a baseline risk assessment, as presented in Subsection 1.5.2, is to characterize current and likely future site-specific ecological risks, with a focus on critical habitats and sensitive habitats, that are attributable to releases of contaminants from a site. The purpose of such risk characterization is to assist in determining site cleanup levels.

The EPA has developed a draft framework for ecological risk assessment. While this document was not available during the development of this methodology, an EPA-sponsored draft peer review of the document (Fava et al. 1991) was provided to the risk assessment committee. The ecological risk assessment framework proposed by Fava et al. is based on the assumption that such a framework should be applicable to the entire spectrum of potential ecological problems. As such, the Fave et al. recommends a three-step approach to ecological risk assessment:

1. Problem Definition/Scoping
2. Analysis
   - 2a. Characterization of Stress
   - 2b. Characterization of Ecological Effects
3. Risk Characterization

The recommendation of Fava et al. places particular emphasis on the evaluation of a broad range of potential stressors, and on a need to evaluate action alternatives for a given project. However, the application of the environmental evaluation component of the Hanford Site baseline risk assessment methodology presented herein is much more focused. Per the NCP, the methodology addresses only the releases of site contaminants and evaluates only the baseline condition(s) (see Subsection 1.5.2).

The baseline environmental evaluation methodology for the Hanford Site follows the conceptual recommendations of Fava et al., but the three steps to the process have been given different names that are more appropriate to the focused objectives and demonstrate a parallel to the human health evaluation process outlined in Chapter 2. These three steps, discussed in detail in the following subsections, are:

1. Problem Definition
2. Analysis
   - 2a. Exposure Assessment
   - 2b. Toxicity Assessment
3. Risk Characterization

The interrelationship of the steps of this process is displayed graphically in Figure 3-1.

Although the analogy to the human health evaluation process is evident, several differences remain. The first step, problem definition (the term "scoping" is deleted as this term has a specific, although not unrelated, meaning under CERCLA), includes contaminant identification, which forms the initial step of the human health evaluation process. The terminology is broader for the environmental process as there is a need to identify, in addition to contaminants of potential concern, habitats of potential concern (e.g., sensitive or critical habitats) and biological species of potential concern (e.g., endangered or threatened, or structurally or functionally important, species) within such habitats.

While human health evaluations focus on sensitive individuals within a single species, environmental evaluations are much more complex. During the problem definition stage, the ecological risk assessor is faced with a site at which there are numerous individuals of many species, the populations of which are organized into a community which is the living part of the ecosystem. The initial challenge to the ecological risk assessor is to focus, in a logical manner, the scope of the environmental evaluation on a manageable and meaningful level of effort. That is, the focus of ecological importance is usually not on individuals, but populations.

A second departure from the human health evaluation process entails grouping exposure and toxicity assessment into the single step of analysis. This is done because the two assessments are often inseparable in many environmental evaluations. While biological species or assemblages of species are the ultimate assessment endpoints of any baseline environmental evaluation conducted at a site of hazardous substance release, it is often more convenient to assess these species indirectly. In conducting such indirect assessments, the boundaries of exposure and toxicity often become blurred. An example that is quite applicable to the Hanford Site is evaluating the risk to the species comprising the freshwater aquatic community of the Hanford Reach of the Columbia River. Documented or projected contaminant levels in the water column can be compared to water quality criteria established by EPA and Ecology to provide for the protection of freshwater aquatic life.

Another significant difference between environmental and human health evaluations is that the analysis stage of the environmental evaluation process must take potential indirect effects into account. It is possible to have a significant adverse effect, attributable to a release of a hazardous substance, on a species of potential concern without a direct exposure or direct toxic effect on that species. An example, although not necessarily applicable to the Hanford Site, is the elimination of a migratory predator's prey species due to the toxic effects of a spill occurring during the predator's season of absence from the site.

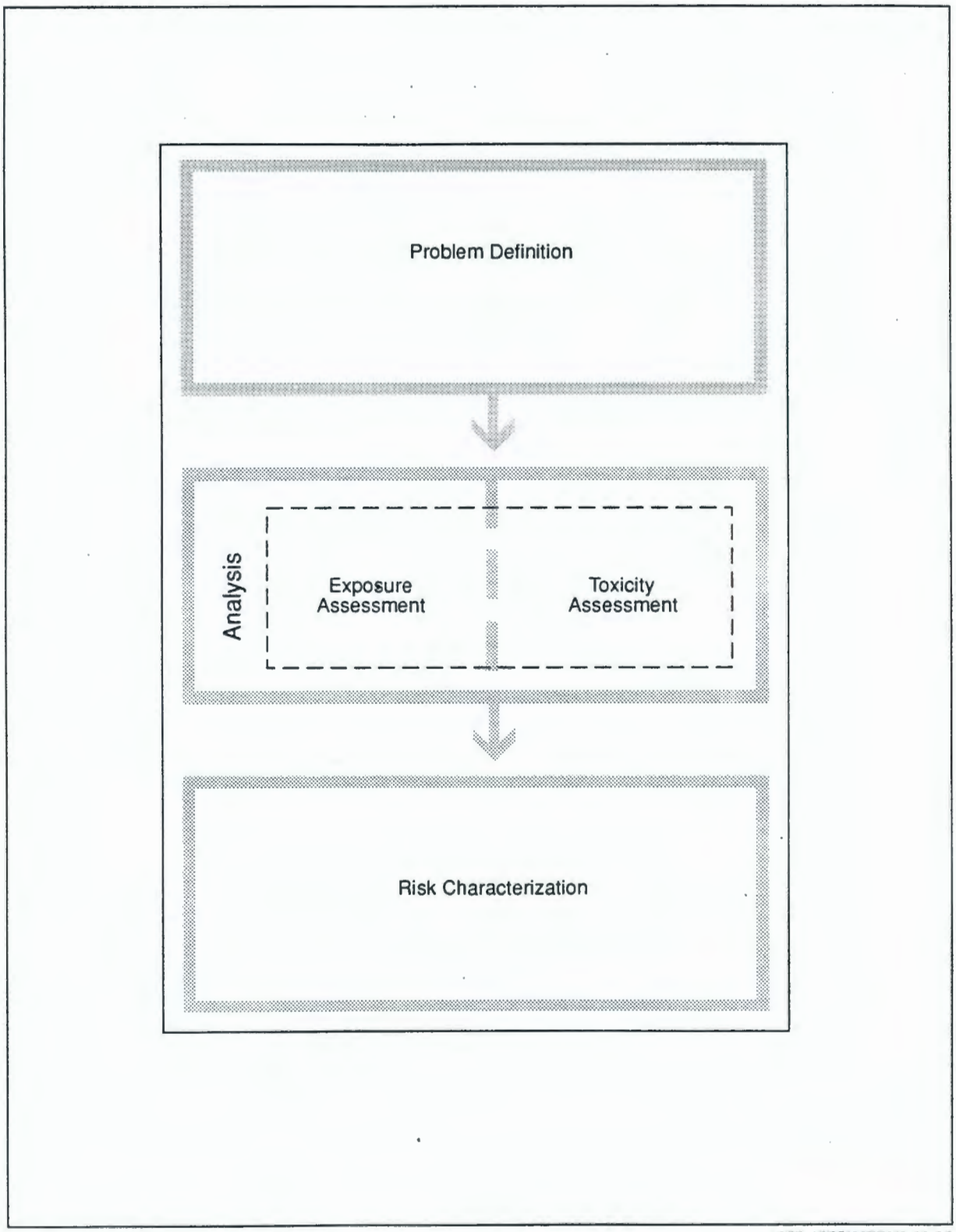

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Figure 3-1. Environmental Evaluation Framework.

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Finally, a distinction needs to be made between the environmental evaluation and the biotic pathways component of the human health evaluation. These evaluations, while having the potential to share significant portions of data and much of the same methods of analysis, are separate. Species forming a major portion of the human food chain would be considered of interest from the perspective of the biotic pathways component of the human health evaluation, but not necessarily of interest from the perspective of the environmental evaluation. The environmental evaluation addresses ecological concerns which have been defined as those that exclude the concerns involving human ecology. These latter concerns are properly addressed within the context of the human health evaluation.

The magnitude of the challenges faced by the environmental evaluator, especially the complexity of the various levels of ecological organization, combined with a lack of both the more specific regulatory guidelines and the relatively ample exposure parameter and toxicity data bases enjoyed by the human health evaluator, points out the critical need to have all Hanford Site environmental evaluations conducted and reviewed under the supervision of qualified ecologists.

Qualified ecologists are those that are familiar with the ecology and biogeochemistry of the Hanford Site ecosystem, are familiar with the regulatory requirements applicable to the Hanford Site Environmental Restoration Program, have proven experience in supervising multidisciplinary teams of the nature required for a baseline environmental evaluation, have proven experience in exercising professional ecological judgement, and have proven experience in documenting and defending logical assumptions used in ecological analyses.

The lack of specific guidelines and adequate data bases, the complexity — relative to the human environment — of the Hanford Site ecosystem, and the need for site-specific evaluations result in the methodology presented below in being more generic than its human health counterpart presented in Chapter 2. However, these factors also point to a need for the environmental evaluation process to be flexible. (As a result, the environmental evaluation process set forth below is more accurately characterized as guidance, as opposed to a methodology.)

The environmental evaluator must rely on professional experience and judgement to compensate for the lack of specific guidelines and data bases. The methodology must be equally applicable to the aquatic, riparian, and terrestrial habitats found at the Hanford Site and the diversity of resident species utilizing them, and the methodology must be applied, at various points in time, at the waste-unit-, operable-unit-, NPL-site- (i.e., aggregate-area-), or Hanford-Site-level of project organization.

The three stages of the environmental evaluation methodology — problem definition, analysis, and risk characterization — are outlined below in Sections 3.2, 3.3, and 3.4, respectively.

3.2 PROBLEM DEFINITION

According to Fava et al. (1991), the objective of the problem definition stage of a baseline environmental evaluation at a hazardous substance release site is to initially

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examine the nature of the contamination problem at hand and then to develop and document a conceptual understanding of the exposures and resulting toxicological effects for the potentially impacted habitats. The problem definition process, depicted in Figure 3-2, can be viewed as a rather complex, interrelated, three-tier operation which demonstrates particular need for process flexibility and for the involvement of a qualified supervising ecologist at this stage.

The top tier represents the need for general and site-specific ecological knowledge and program- and project-specific regulatory goals. The second tier represents the need for a preliminary evaluation in the form of expert examination of such information to condense ecological reality into a manageable and testable conceptual ecological model. The third tier represents the documentation of the conceptual understanding, usually in the form of a simple word or graphic model. Uncertainties inherent in the overall problem definition process should be noted for incorporation into the uncertainty assessment component of the risk characterization process (see Section 3.4).

3.2.1 Integration of Knowledge and Goals

Relative to most NPL sites, many components of the ecology of the Hanford Site have been extensively studied over the years of operation, and many are still being studied. In addition, there have been some ecological studies of the Columbia Basin shrub-steppe vegetational zone, the zone within which the Hanford Site is situated. This knowledge base, in terms of both literature and personnel, will be an invaluable resource in the problem definition stage of each environmental evaluation. Site ecologists and personnel intimately familiar with the abundant available literature must be utilized in the initial and review phases of each evaluation.

Documents that provide good starting points for understanding the available Hanford Site ecological literature are currently in preparation. One is a literature review on the biological effects of the 100 Area of the Hanford Site, and the other is an impact assessment for the Hanford Reach of the Columbia River. Both documents will contain extensive bibliographies and should become available sometime during calendar year 1992.

In addition to Site-specific ecological knowledge, a broad understanding of general environmental science is necessary, as such factors as contaminant fate and transport, or receptor exposure potential, are often controlled by a broad range of physical, chemical, and biological environmental conditions or processes. Thus, the required environmental knowledge base must include not only ecology, but other relevant environmental disciplines such as biology, chemistry, geology, hydrology, meteorology, pedology, and toxicology. Access to such a scientific team allows the evaluation to integrate the professional judgement of a diversity of disciplines.

A knowledge of regulatory goals is essential. This will require an environmental evaluation staff that is well versed in CERCLA, RCRA, MTCA, ESA and other pertinent federal and state environmental statutes and their associated regulations and guidance documents. It will also require a working knowledge of the site-specific requirements of the Hanford Federal Facility Agreement and Consent Order and the various DOE-RL orders. An understanding of such regulatory goals will help to ensure that environmental evaluations are kept focused on pertinent issues, that the evaluations produce useful

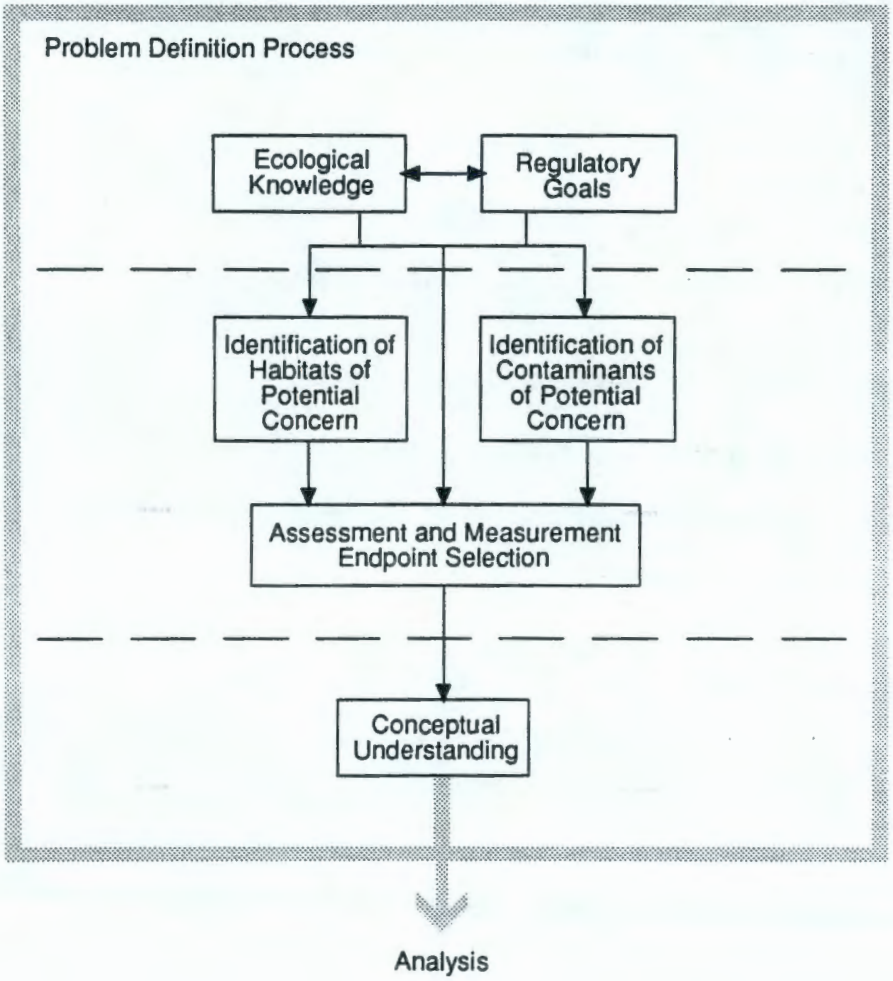

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Figure 3-2. Problem Definition Process.

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results, and that schedule commitments are met.

It should be noted that natural resource trustees (e.g., the United States Fish and Wildlife Service) have the responsibility and authority under CERCLA to identify the need for and, as necessary, to conduct natural resource damage assessments (NRDAs) at NPL sites. The need for a NRDA at the Hanford Site has not been identified. Given the extensive environmental monitoring that has occurred at Hanford (see Section 3.2.1), the need for a NRDA is unlikely; however, this situation could change, pending the outcomes of the various baseline environmental evaluations that have been, are, and will be conducted.

If a natural resource trustee should identify the need for a NRDA at the Hanford Site, the appropriate baseline environmental evaluations should then be scoped (or reviewed, if already planned) with close participation on the part of trustee agency officials. In doing so, certain information valuable to the NRDA process may be able to be obtained more efficiently and cost effectively through the baseline environmental evaluation process.

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3.2.2 Preliminary Evaluation

The preliminary evaluation performed in the problem definition stage consists of three steps:

- Identification of habitats of potential concern,
- Identification of contaminants of potential concern, and
- Selection of assessment and measurement endpoints.

These steps are discussed below.

3.2.2.1 Identification of Habitats of Potential Concern. Habitat identification consists of recognizing and documenting the critical and sensitive habitats located on, adjacent to, or near the hazardous substance release site of interest. Critical habitats are defined pursuant to ESA regulations [50 CFR § 424.02(d)] to be:

> (1) the specific areas within the geographical area currently occupied by a species, at the time it is listed in accordance with the [ESA], on which are found those physical or biological features
>
> > (i) essential to the conservation of the species and
> > (ii) that may require special management considerations or protection, and
>
> (2) specific areas outside the geographical area occupied by a species at the time it is listed upon a determination by the Secretary [of the Interior] that such areas are essential for the conservation of the species.

A list of state-designated endangered or threatened plant and animal species that could potentially occur at the Hanford Site is provided in Table 3-1. As State of Washington designations, in all cases, are as strict or stricter than the corresponding federal designations, this list, updated appropriately, may be used as a tool to identify potential

Table 3-1. Endangered and Threatened Species Potentially Associated with the Hanford Site.
(Sheet 1 of 2)

Endangered Vascular Plants persistentsepal yellowcress (*Rorippa columbiae*): Known to have a scattered distribution due to specialized habitat requirements or habitat loss; generally occurs in moist to marshy places and is known to inhabit the wetted shoreline of the Hanford reach of the Columbia River in Benton County.
Threatened Vascular Plants Columbia milk-vetch (*Astragalus columbianus*): Locally endemic to the area in the immediate vicinity of Priest Rapids Dam, including a portion of Benton County; could potentially occur along the Columbia River in the northwestern portion of the Hanford Site. eatonella (*Eatonella nivea*): Known to occur along the Columbia River in Grant County; could potentially occur along the river in the northern portion of the Hanford Site. Hoover's desert parsley (*Lomatium tuberosum*): Locally endemic to southcentral Washington, including Benton County; known to inhabit rocky hillsides.
Endangered Birds Aleutian Canada goose (*Branta canadensis leucopareia*): Nests in the Aleutian Islands of Alaska and winters in California; has been occasionally sighted, as a migrant, in Benton County; a potential seasonal user of the Columbia River valley, feeding on grasses, sedges, and berries. American white pelican (*Pelecanus erythrorhynchus*): Winters along the southern Pacific Coast and the Gulf Coast and nests in northern prairie and intermontane lakes; no longer nests in Washington; migrates through eastern Washington; flocks are common in the Columbia Basin during the summer; known to occasionally winter on the Columbia River, foraging on fish, amphibians, and crustaceans and roosting on islands. peregrine falcon (*Falco peregrinus*): Breeds and winters in eastern Washington, inhabiting open marshes, river shorelines, wide meadows and farmlands; nests on undisturbed cliff faces; an erratic visitor at the Hanford Site, feeding on songbirds, shorebirds, and waterfowl. sandhill crane (*Grus canadensis*): Inhabits open prairies, grainfields, shallow lakes, marshes, and ponds, nesting in drier grassy and marshy areas; common migrant during the spring and fall in Washington; some known and suspected nesting sites in eastern Washington; occasional visitor at the Hanford Site.

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Table 3-1. Endangered and Threatened Species Potentially Associated with the Hanford Site.
(Sheet 2 of 2)

Threatened Birds bald eagle (*Haliaeetus leucocephalus*): A regular winter visitor to the Columbia River, feeding on spawning salmon and perhaps waterfowl and small mammals; roosting areas are known to exist in the 100 Areas of the Hanford Site (roost sites and winter feeding areas constitute critical habitats for this species). ferruginous hawk (*Buteo regalis*): Inhabits open prairies and sagebrush plains, usually with rocky outcrops or scattered trees, located well away from human disturbance; known to nest in Benton and Franklin counties, with Franklin County possessing the majority of the nests within Washington; known to nest on the Hanford Site; rarely winters in Washington; known to occasionally forage on small mammals, birds, and reptiles on sagebrush plains in the Hanford Site.
Threatened Mammals pygmy rabbit (*Sylvilagus idahoensis*): Presumed extirpated from Washington since 1984 wildfire; inhabits undisturbed areas of sagebrush having soils soft enough to dig burrows in; once known to exist on the Hanford Site near springs in the Snively Basin, west of the 200 Area plateau.
[a]State designations are as strict or stricter than federal designations. Adapted from DOE 1987, DOW 1987, DNR 1990, and Hitchcock and Cronquist 1973.

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critical habitats. (While state laws do not provide for a level of protection for endangered and threatened species that is comparable to that provided for under federal laws, this distinction will be disregarded for the purposes of identifying critical and sensitive habitats, for the purposes of conducting baseline environmental evaluations, at the Hanford Site.)

Another resource for identifying critical habitats is the State of Washington Natural Heritage Program. The Department of Natural Resources (DNR) manages the plant aspects of the program, while the Department of Wildlife (DOW) manages the animal aspects. Each department will search its computerized data base upon receipt of a written request with project location (section, township, and range) and a nominal fee (approximately $15).

Each department also provides any information it has on the occurrence of other species, although not endangered or threatened, that it considers sensitive, or of interest. Such species have no protected status under state or federal regulations, but some are candidates for listing. In any event, the fact that DNR or DOW has deemed a species to be of interest should be regarded as significant, and any habitats of such species in the project vicinity meeting the requirements of the above definition should be regarded as critical for the purposes of the environmental evaluation.

A regulatory definition of sensitive habitat does not exist. However, guidance can be derived from the CERCLA hazard ranking system (HRS), which is promulgated as an appendix to the NCP (40 CFR Part 300, Appendix A). Table 4-23 of the HRS provides sensitive environments rating values. Table 3-2 is a tailored list of potential Hanford Site sensitive habitats derived from the HRS. The list was kept deliberately broad to allow for potential future classifications, but it is expected that the list, upon closer examination, will be shortened substantially.

Once habitats of potential concern have been identified, their relevant characteristics should be evaluated. For example, species of potential interest should be identified for each habitat of interest that has been identified.

Those species classified as endangered, threatened, or sensitive, or those regarded as being of interest to DNR or DOW will already be identified through the habitat identification process. Using primarily data obtained from literature searches or field reconnaissances, one should determine which species are structurally or functionally important within each identified critical or sensitive habitat, and any species that provide an environmental service to man. All such species are of potential interest for further use in the evaluation as either assessment or measurement endpoints.

Structurally or functionally important species are those which are dominant in the community in terms of productivity, relative abundance, or biomass. Keystone species, those whose removal from the habitat would result in a drastic change in the characteristics of that habitat, are also considered to be structurally or functionally important. Species that provide a service to man are those that are of commercial or recreational importance, or those that perform miscellaneous environmental services (e.g., pest control).

Once the species of potential interest are identified, the feeding relationships of these species should be considered. A knowledge of feeding relationships is important in understanding potentially operative biocontamination pathways within habitats. Within

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Table 3-2. Sensitive Habitats Potentially Occurring at the Hanford Site. Derived from 40 CFR Part 300, Appendix A.

Areas identified under the Coastal Zone Management Act in the state coastal zone management plan as requiring protection because of ecological value

National monument

Habitat known to be used by designated or proposed endangered or threatened species

National preserve

National or state wildlife refuge

Federal land designated for protection of natural ecosystems

Spawning areas critical for the maintenance of fish/shellfish species with river waters (i.e., those areas used for intense or concentrated spawning by a given species)

Migratory pathways and feeding areas critical for maintenance of anadromous fish species within river reaches in which the fish spend extended periods of time

Terrestrial areas utilized for breeding by large or dense aggregations of animals

National river reach designated as recreational

Federal- or state-designated scenic or wild river

State land designated for wildlife or game management

State-designated natural areas

Particular areas, relatively small in size, important to maintenance of unique biotic communities

State-designated areas, pursuant to CWA, for protection or maintenance of aquatic life

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the habitat characteristics component of the problem definition process, readily available information should be considered, as necessary, to assist in the endpoint selection component, which is described below.

3.2.2.2 Identification of Contaminants of Potential Concern. The processes to identify contaminants and determine their characteristics are identical to those described for the human health evaluation methodology within Section 2.1. An implicit assumption in using this process is that the preliminary risk screening methodology, which is based primarily on human health protection, is also protective with regard to ecological exposures. This is probably a good assumption for several reasons.

First, the contaminant identification process involves multiple univariate comparisons of multivariate contaminant data sets against corresponding background data sets. Multiple univariate comparisons within multivariate data sets increase the experiment-wise Type I error rate (i.e., the false positive error rate) dramatically, as does multiple use of a single background data set (Green 1979). A high degree of conservatism is thus built into this step of the process.

Second, the preliminary risk screening component also provides a great deal of conservatism. The use of maximum detected contaminant concentration values, regardless of distance to potential points of exposure, is conservative. The use of carcinogenicity in humans as toxicity measure can be regarded as generally conservative, because it is based on the upper-bound lifetime probability of an individual developing cancer. The toxicity values are usually derived by extrapolation from high dose animal exposures. Most organisms at the Hanford Site will probably be exposed to much lower environmental exposures.

Use of the human risk screening procedure is environmentally protective due to the built-in conservatism of the reasonable maximum exposure assumptions and safety factors applied to toxicological constants (which are typically derived from toxicological studies performed on non-human [i.e., environmental] receptors). This is especially true given that the human health preliminary risk screen is designed to protect sensitive individuals, whereas the environmental evaluation is generally designed to protect populations of species.

Third, the preliminary risk screening component has an ecological safety net which consists of consideration of environmental protection ARARs. For example, the maximum copper concentration in ground water may be well below any level of concern with respect to human health. But, if this concentration exceeds EPA and state water quality criteria established for the protection of freshwater aquatic life, even if the monitoring well is located far from the Columbia River, copper would be retained for both the human health and the environmental evaluations. (It should be further noted that both CERCLA and MTCA require site cleanup to attain such contaminant-specific ARARs whether or not any actual risk is posed by a release of contaminants from a site.)

Finally, the contaminant identification process is designed with a sufficient level of conservatism to retain any contaminant which has even a remote chance of posing an unacceptable exposure risk. This inevitably results in a multivariate data set, comprised of those contaminants of the highest potential concern, which must be carried through the human health and environmental evaluations. Multivariate environmental data sets

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typically exhibit a phenomenon accounted for by the Pareto principle (Wadsworth 1990).

The Pareto principle states that a relatively large number of problems (in this case, a large proportion of site-attributable risk) in any given situation will be caused by only a few factors (in this case, only a few contaminants). The preliminary risk screen is a useful application of the Pareto principle in that it focuses the risk assessor and reviewers on the critical few contaminants rather than the trivial many.

3.2.2.3 Selection of Assessment and Measurement Endpoints. Using information considered under habitat identification, and with an understanding of the characteristics of the various contaminants of potential concern, the next step in the problem definition process is the selection of appropriate endpoints for the environmental evaluation. Initially, assessment endpoints, or ecological endpoints, are identified. Fava et al. (1991) define these as the specific properties of each habitat of interest used to evaluate the state or change in the state of the ecological system. Such endpoints must be of ecological importance and of direct management relevance: they must be tied to specific regulatory or public concerns. Assessment endpoint identification can be viewed as being analogous to receptor identification in the human health evaluation process.

Once assessment endpoints are defined, appropriate measurement endpoints, or indicators, are selected. Fava et al. (1991) defines these as the items actually measured in a monitoring scheme or, more likely the case at the Hanford Site, are data sought from an historical data base. In some cases the assessment and measurement endpoints are identical; in others they may not be.

For example, two general types of indicator species could potentially be used for measurement endpoints. The first type are those species or groups of species which could be used to evaluate prevailing environmental conditions at a site. It could be more cost effective to monitor this type of indicator species rather than monitoring the actual release within nonbiotic environmental transport media. Examples of classes of this first type are (Beanland and Duinker 1983):

- sentinel species — introduced species that are sensitive to the environmental perturbation (in the case of the 100 Area, this will generally be a contaminant release);
- detector species — indigenous species that are sensitive to the environmental perturbation;
- exploiter species — species that display a competitive advantage under conditions of the environmental perturbation;
- accumulator species — species that bioaccumulate substances of interest; and
- bioassay species — species sensitive to the environmental perturbation that are suitable for laboratory tests.

The need for the second type of indicator species arises when an endangered or threatened species requires further investigation which could prove harmful to the species.

Rather than jeopardizing such a protected species, it may be possible to identify an analogue species that can be investigated and extrapolate the results using professional judgement. Another possible approach to take in this circumstance would be to find an appropriate species that constitutes a significant component of the protected species food chain and investigate it (resulting in an indirect investigation on the protected species).

Ecological indicators could also find use as measurement endpoints. An ecological indicator is a parameter that is not species-specific, but that measures an integrated biological community process or characteristic (e.g., species richness or primary productivity).

Endpoint selections are highly site-specific. Two EPA documents, EPA 1989h and EPA 1988d provide useful guidance on selection of appropriate endpoints, but the actual selection is far from an exact science, and most projects are highly constrained by schedule and budget. Table 3-3 provides some characteristics of good assessment and measurement endpoints.

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3.2.3 Documentation of Conceptual Understanding

The final component of the problem definition process is documentation of the conceptual understanding. As mentioned earlier, the documentation is usually accomplished in the form of either a simple word or graphic model. Such models are used to establish a working concept how contaminant exposure pathways may be or may become operative, and how toxic effects may be imposed on components of the various habitats of interest.

This understanding serves to document assumptions made in the problem definition process and to allow for meaningful review of this critical stage. It also serves as the basis for initiating the analysis stage of the environmental evaluation.

The problem definition process is the most critical stage of the environmental evaluation methodology. It involves distilling a complex site into a manageable model that can be tested during the analysis stage of the evaluation. Like all models, the resulting conceptual understanding is an abstraction of reality, and in the course of developing the model, the environmental evaluator will inevitably run up against many of the limitations of our current ecological knowledge base.

Such limitations place significant demands on both the evaluators and the reviewers. The evaluators must be qualified and experienced, be familiar with regulatory goals, exercise good judgement, document assumptions, and communicate effectively. Reviewers must have the same traits and, in addition, have realistic expectations and understand the limitations inherent in the methodology.

3.3 ANALYSIS

Fava et al. (1991) noted that toxicity and exposure assessments are not separate processes that can proceed independently of one another, but are interrelated as shown in Figure 3-3. In some forms of environmental evaluation, such as assessing the risk of

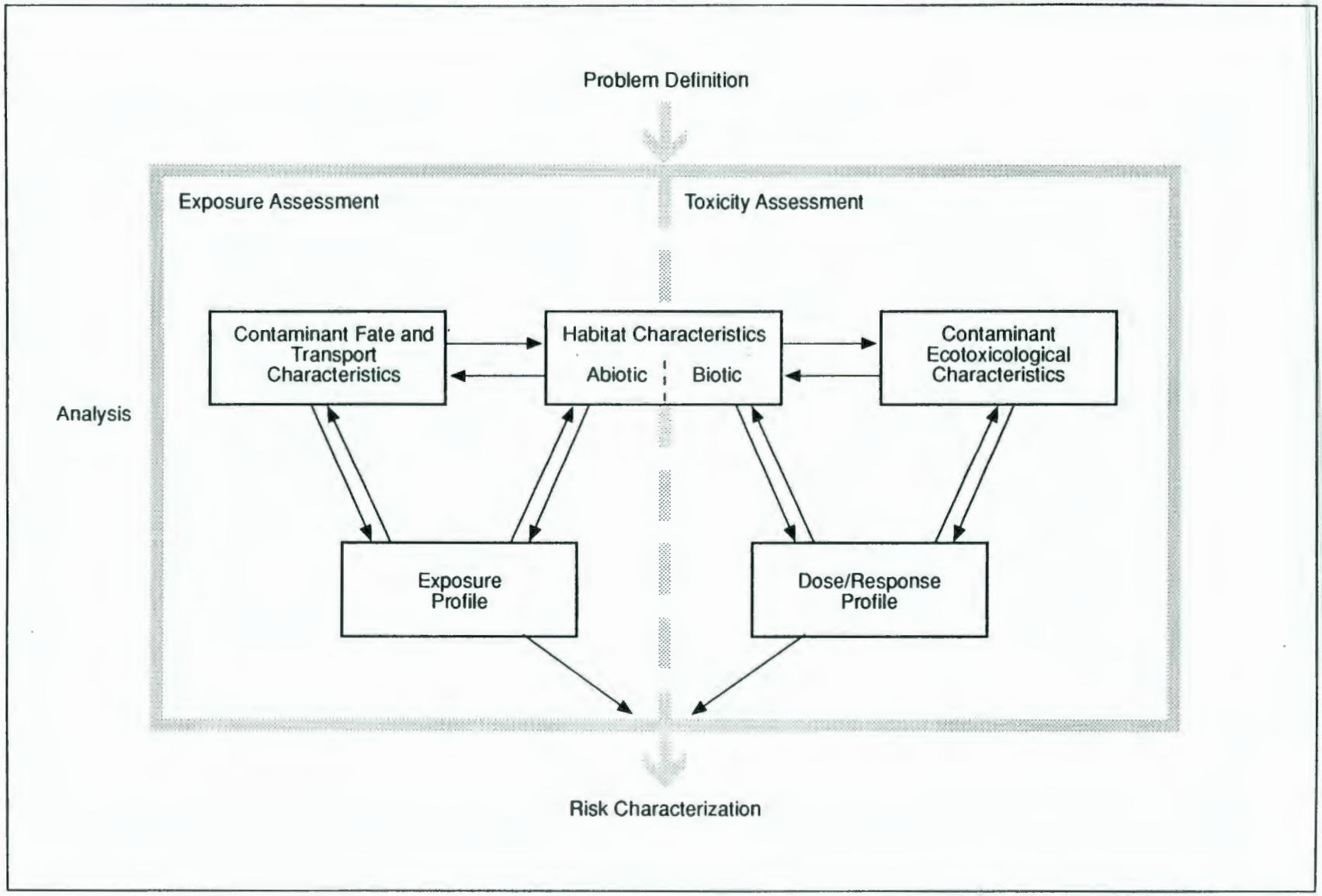

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Figure 3-3. Analysis Process.

Table 3-3. Characteristics of Good Assessment and Measurement Endpoints. Derived from EPA (1989e).

Assessment Endpoints

- Socially relevant
- Ecologically relevant
- Unambiguous operational definition
- Measurable or predictable
- Susceptible to the contaminant
- Logically relevant to the project decision

Measurement Endpoints

- Correlative to or predictive of an assessment endpoint
- Easily measured
- Cost-effectively measured
- Appropriate to the scale of the site
- Appropriate to the exposure pathway
- Appropriate temporal dynamics
- Low natural variability
- Diagnostic
- Broadly applicable
- Standard
- Existent data series

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contaminant inputs to a body of surface water with water quality criteria, the two subprocesses are indistinguishable as they were effectively integrated during development of the criteria.

Both subprocesses share the need to take habitat characteristics, both abiotic and biotic, into account. For example, physical and chemical habitat factors and the biological species present must be taken into consideration in determining contaminant fate and transport and the form of ecotoxicological information to be employed. Such information is iteratively processed to obtain the dose/response and exposure profiles needed to proceed to the risk characterization stage.

Evaluation of the two subprocesses makes it necessary to consider the spatial, temporal, and ecological boundaries of the project. Spatial considerations include such factors as whether contaminants are or could be present at a potential point of exposure and lateral and vertical distribution of contamination in either a soil or water column. Temporal considerations include determining whether chronic or acute exposures are most appropriately assessed and whether a species is migratory or present throughout the year. Ecological boundary considerations include such factors as behavior, as it relates to exposure duration, and life cycles, as they relate to exposure duration and sensitivity.

Given the high degree of uncertainty associated with many exposure and toxicological parameters employed in environmental evaluations, the reasonable maximum exposure concepts mentioned in Chapter 2 should be used here, as well. This entails use of a conservatively biased estimate of mean values, where possible, in developing exposure and ecotoxicological profiles. In some instances, however, the environmental evaluator must be aware that such conservatism has already been employed. Examples include water quality criteria.

The elements of the toxicity assessment and of the exposure assessment are outlined in Subsections 3.3.1 and 3.3.2, respectively.

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3.3.1 Toxicity Assessment

The elements of the toxicity assessment stage of the environmental evaluation are graphically depicted in the right half of the analysis process presented in Figure 3-3. The central form of analysis occurring during the toxicity assessment is the compilation and evaluation of ecotoxicological data.

As mentioned above, the appropriate type of ecotoxicological data will depend on project boundaries and habitat characteristics. As many of these boundaries and characteristics are highly site-specific, and as there are no relatively comprehensive standardized ecotoxicological data bases equivalent to those EPA develops and maintains for use in human health evaluations, it is not possible to provide detailed guidance. The evaluator must exercise professional judgement in selecting the appropriate ecotoxicological data, and in adapting these data to the site at hand.

Potential sources for identifying toxicity information that may be useful in the environmental evaluation are:

- Contaminant Hazard Series, Fish and Wildlife Service, U.S. Department of the Interior,
- Dangerous Properties of Industrial Materials (Sax 1984),
- Handbook of Environmental Data on Organic Chemicals (Vershueren 1983),
- Superfund Public Health Evaluation Manual (1986b), and
- Trace Elements in Soils and Plants (Kabata-Pendias and Pendias 1984),

It will often be necessary to make dose duration adjustments, as well as extrapolations between species (often between species within differing genera, families, etc.). If lethal doses are employed in the assessment, they will need to be converted to a more protective ecotoxicological measure from the perspective of the environmental evaluation, such as an effective dose or a lowest observed effects level. All such modifications of the ecotoxicological data should be well documented and justified, and uncertainties must be noted and addressed in the uncertainty assessment component of the risk characterization process (see Section 3.4). Table 3-4 provides a list of some of the types of ecotoxicological data available that may be used in the toxicity assessment.

The final output of the toxicity assessment is a dose/response profile. In most instances this will not be in the form of an elegant dose/response curve. Even if such data were initially available, it would be inappropriate to imply a high degree of certainty to the assessment by carrying the entire curve through the many manipulations that are often required to apply the data to the endpoint of interest. Most frequently, the dose/response profile will consist of a conservatively derived value that estimates either a reference dose or concentration of contaminant that is expected to result in no significant impact to the endpoint in question.

The environmental evaluator must keep in mind that a particular species of interest, as noted earlier, could potentially be subjected to an indirect toxicological effect. A well developed conceptual understanding output of the problem definition process will assist in identifying such a circumstance.

3.3.2 Exposure Assessment

The elements of the exposure assessment stage of the environmental evaluation are graphically depicted in the left half of the analysis process presented in Figure 3-3. The key element of the exposure assessment is compiling and processing all relevant information with respect to fate and transport of the contaminants of potential concern to allow for the development of an exposure profile for each species of interest. Uncertainties identified in this stage of the process should be noted and documented in the uncertainty assessment component of the risk characterization (see Section 3.4).

Physical and chemical contaminant fate and transport data are usually readily available from the CERCLA remedial investigation, RCRA facility investigation, or limited

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field investigation data gathered and evaluated prior to the exposure assessment. Quite frequently, however, the environmental evaluator is responsible for the characterization of biological fate of contaminants and transport of contaminants through biotic pathways.

Biological fate can be altered primarily through various types of microbial processes and through bioaccumulation. Biological transport must take such things into account as feeding relationships, body weights, uptake factors, bioconcentration factors, environmental setting, home range, and ethology. Exposure pathway evaluations should focus on only those pathways that have a reasonable potential to be operative, however, consideration of the potential for indirect effects of contaminant exposure should not be overlooked. In evaluating these pathways, as with all other pathways, the fundamental concepts applicable to the development of a reasonable maximum exposure scenario should be applied.

Figures 3-4, 3-5, and 3-6 depict generic potentially operative Hanford Site ecological exposure pathways for the aquatic, riparian, and terrestrial habitats, respectively. These pathway diagrams are provided for preliminary conceptual consideration only, and they in no way obviate the need to assess exposure pathways on a site-specific basis for each environmental evaluation.

The final output of the exposure assessment, an exposure profile, parallels that of the toxicity assessment, the dose/response profile. The profile will often simply consist of an estimated exposure dose, but, depending on the habitat type and species involved, it could consist of a concentration. The exposure and toxicity assessors need to ensure unit consistency for these two assessments to allow for subsequent integration and comparison in the risk characterization stage of the evaluation.

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3.4 RISK CHARACTERIZATION

The risk characterization process of the environmental evaluation is depicted in Figure 3-7. In this stage of the evaluation exposure and toxicity profiles are integrated, uncertainties in the overall evaluation are analyzed, and risk predictions and ecological interpretations of the baseline condition(s) are documented.

The exposure profiles and dose/response profiles, for each combination of contaminant of potential concern and measurement endpoint within habitat type, are integrated by comparison, and the results extrapolated, to determine whether or not a significant potential exists for an adverse impact to the corresponding assessment endpoints. An indication of such an impact exists when the exposure profile exceeds the dose/response profile. Those contaminants of potential concern that are associated with such an impact potential are then regarded to be actual contaminants of concern.

Given the relative lack of ecotoxicological and ecological exposure data, it should be anticipated that some environmental evaluations for Hanford Site projects will, by necessity, be strictly qualitative. A qualitative risk characterization has a basis in best professional judgement and on-site observation. For example, if knowledge about a given substance is limited to knowing that it is phytotoxic to some degree, an ecologist could develop a qualitative risk characterization based on knowledge of length of environmental

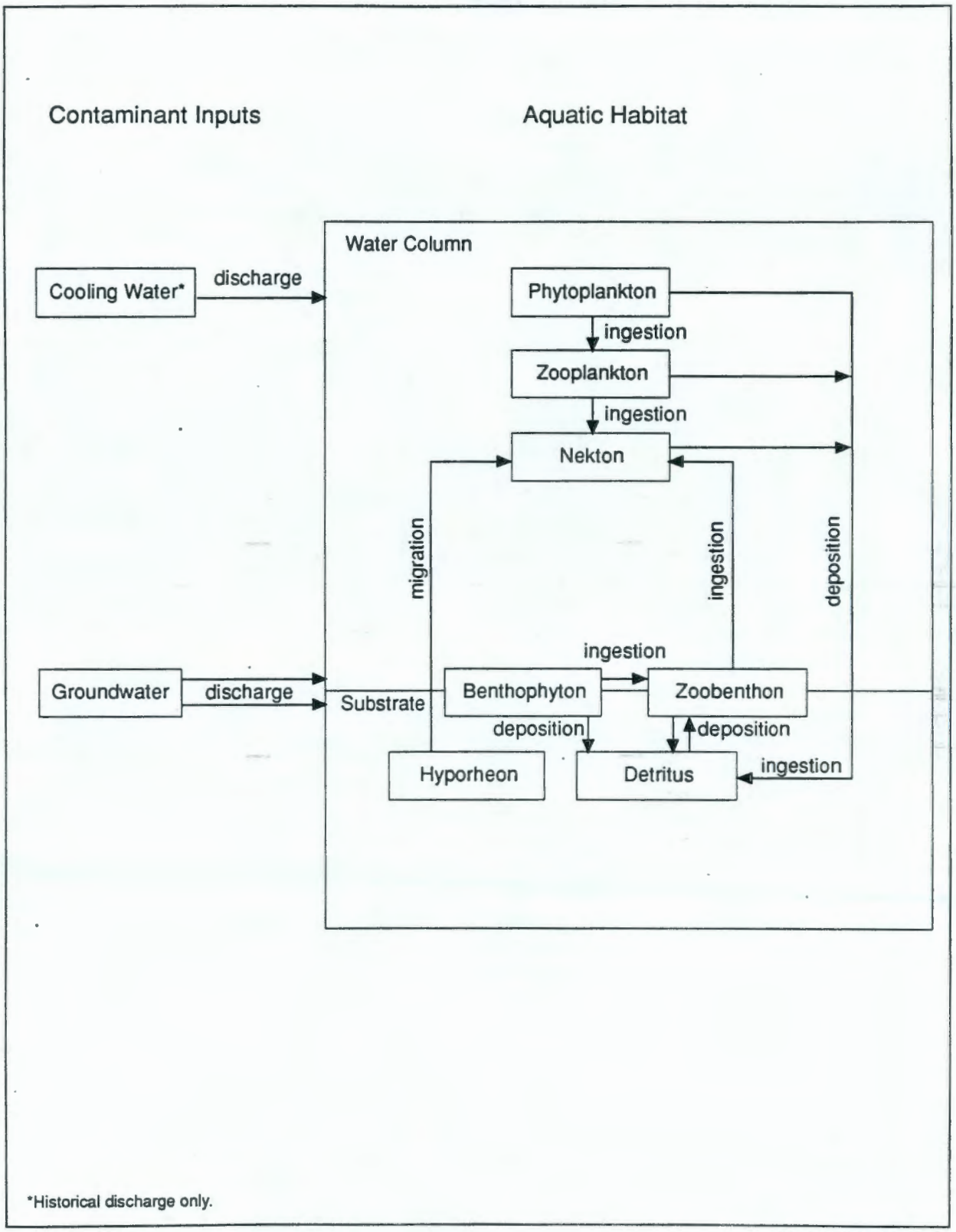

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Figure 3-4. Potential Aquatic Ecological Exposure Pathways at the Hanford Site.

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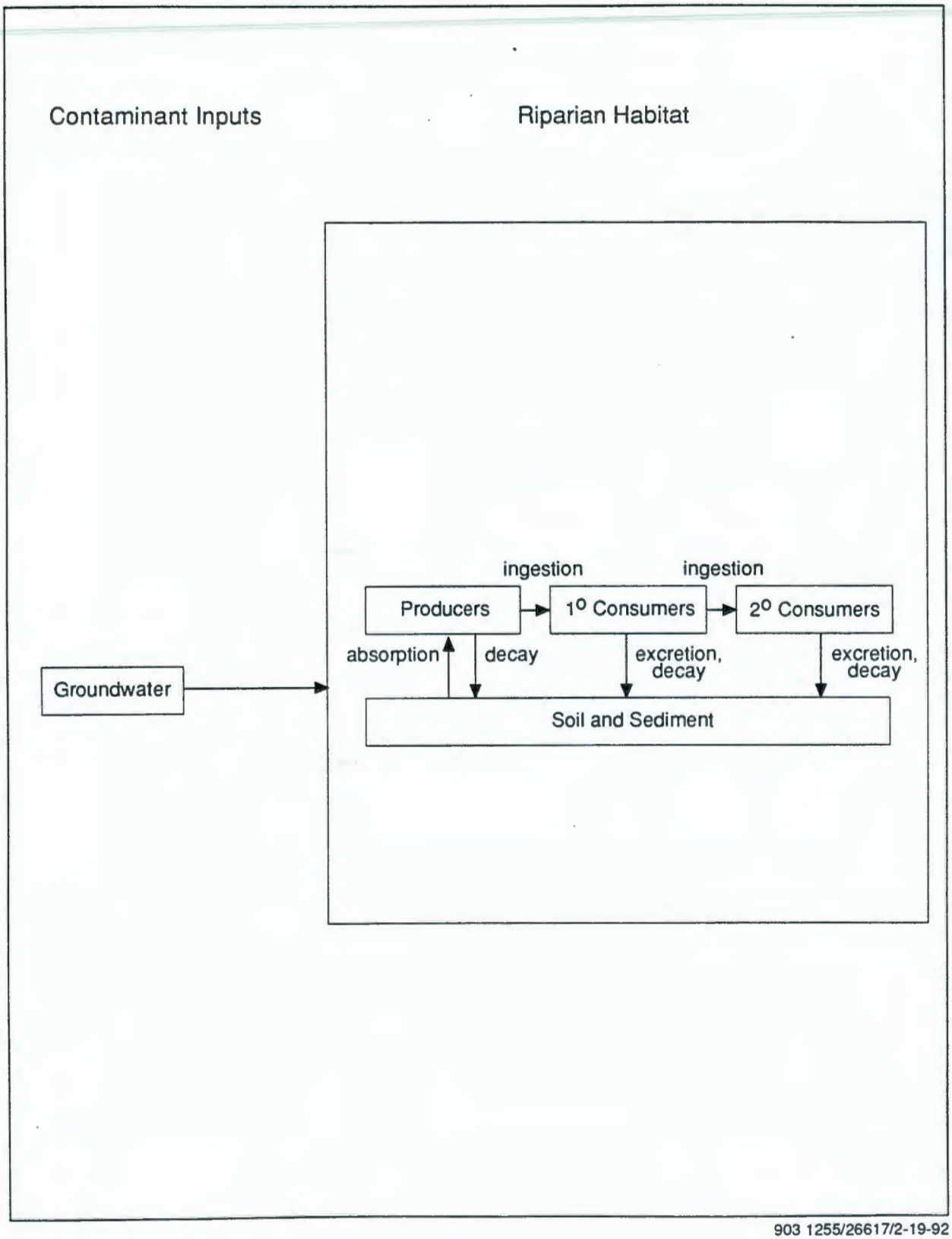

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Figure 3-5. Potential Riparian Ecological Exposure Pathways at the Hanford Site.

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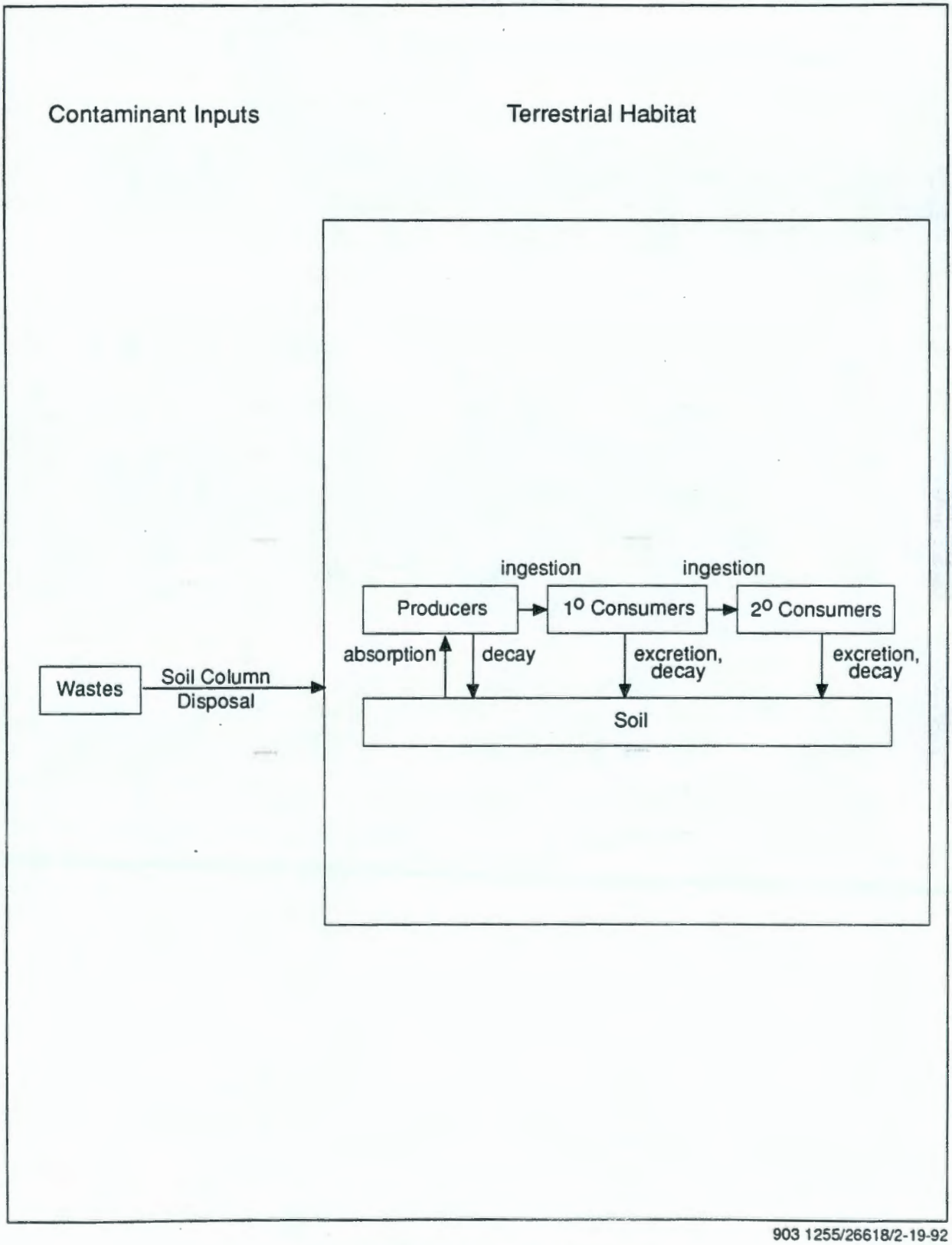

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Figure 3-6. Potential Terrestrial Ecological Exposure Pathways at the Hanford Site.

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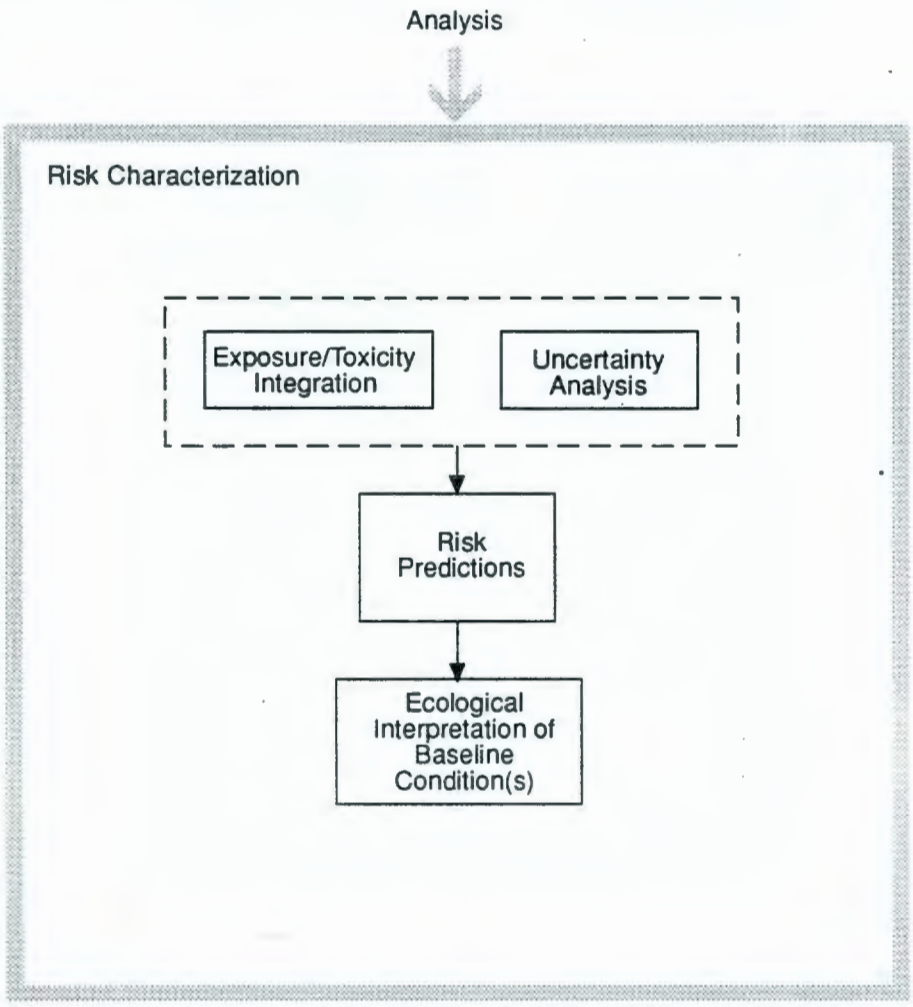

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Figure 3-7. Risk Characterization Process.

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Table 3-4. Example Ecotoxicological Values.

AWQA — EPA ambient water quality advisory

AWQC — EPA or state ambient water quality criterion

EC_{50} — effective concentration at which 50% of experimental organisms exhibit a nonlethal effect

ED_{50} — effective dose at which 50% of experimental organisms exhibit a nonlethal effect

LC_{50} — lethal concentration at which 50% of experimental organisms die

LD_{50} — lethal dose at which 50% of experimental organisms die

LOAEL — lowest observed adverse effects level

LOEL — lowest observed effects level

NOAEL — no observed adverse effects level

NOEL — no observed effects level

TDL_o — lowest observed toxic dose

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exposure (based on operational history of the facility in question), and observed impacts or lack of impacts to the vegetation community. Assumptions used in such qualitative characterizations, including lack of quantitative ecotoxicological or exposure data, must be explicitly noted.

Impact potential can be displayed quantitatively in the form of a contaminant-specific environmental hazard quotient (EHQ), which equals the exposure profile divided by the dose/response profile. If the EHQ exceeds unity (> 1), then the potential for an environmental impact, to the assessment endpoint, is deemed to exist. The degree of impact must then be evaluated by taking into consideration such factors as the degree of habitat contamination (e.g., what areal proportion of a given sensitive habitat is, or is expected to become, contaminated to a level of concern [EHQ > 1]?) or the degree of species utilization of a particular contaminated habitat (e.g., how many threatened species are, or are expected to be, utilizing the significantly contaminated portion [EHQ > 1] of a given critical habitat?).

The risk characterization process should also address the potential for cumulative toxic effects. Each contaminant of potential concern should be evaluated with respect to its toxicity mechanism. For example, if two substances are known to affect an assessment endpoint in a similar manner, these effects of these substances should (in the absence of substance-specific information) be assumed to be additive. The substance-specific EHQs can then be added together to form an environmental hazard index (EHI), and the risk characterization can proceed in the manner described above with regard to EHQs.

An important element of the risk characterization process is the uncertainty analysis. This is particularly true for environmental evaluations where methods and data bases are not accepted as being standardized to the extent the human health evaluation methods and data bases are. The uncertainty assessment for an environmental evaluation can range from a purely qualitative assessment, based on best professional judgement, to a detailed quantitative assessment (e.g., in the form of a subjective probabilistic evaluation). The type of assessment best suited for a particular environmental evaluation will depend on the degree of detail and the outcome of the evaluation itself.

The EPA (1989b) recommends that biotic pathway assessments (within the context of both the human health and the environmental evaluation) be regarded, at best, as semiquantitative due to a current lack of understanding of the general theoretical relationships among contaminants, ecosystems, and biological species; the EPA goes on to note that biotic pathway assessment errors of up to three to four orders of magnitude should be anticipated. In spite of the broad range of uncertainty associated with risk assessment, the use of reasonable maximum exposure concepts provides ample conservatism to most likely render such degrees of uncertainty irrelevant from a regulatory perspective. Examples of such conservatism include:

- the control statistical screening procedure;
- the preliminary risk screening procedure, including the use of human carcinogenicity as an environmental toxicity screening measure and added safety factors contained in overall human toxicity data bases and the use of protective environmental benchmarks (e.g., water quality standards); and

- the use of conservatively biased estimates of environmental exposure and ecotoxicological values.

Several approaches can be taken to the uncertainty analysis. The first is a simple qualitative approach that provides only a general statement of the degree of uncertainty. A second would consist of an entirely independent analysis following the problem definition stage of the methodology. In this case, if the two independent analyses agree with one another sufficiently, the overall evaluation is deemed to have an acceptable low degree of uncertainty.

EPA (1989b) recommends that the uncertainty assessment include the following:

- variance estimates for all statistics;
- assumptions underlying use of statistics, indices, and models;
- the range of conditions under which models or indices are applicable; and
- narrative explanations of other sources of potential error in the data.

In addition to the above, any data quality deficiencies that are known to be associated with the data sets and any faulty assumptions employed in the evaluation should also be documented and addressed in the uncertainty assessment.

The above options for uncertainty analysis all leave much to be desired. The best, most scientific approach to the problem would be to conduct a probabilistic risk assessment from the start of the evaluation. In this manner, the resulting risk distribution curve not only elegantly provides variance information, it also provides for a much more reasonable estimate of risk or likely future risk. For most environmental evaluations at the Hanford Site, however, a semiquantitative uncertainty assessment is expected to suffice.

The final portion of the risk characterization process involves documenting evaluation results to the project decision maker to allow him to decide relevant issues in a risk-based manner. Risk predictions that are testable are documented. Environmental risk endpoints are expressed in terms of assessment endpoints and their direct management relevance. (If the assessment endpoints have no direct management relevance, they should not have been selected during the problem definition phase).

Explaining the ecological implications of the baseline condition(s) at a site to the decision maker is also necessary. In addition to the potential for toxicological effects, ecologically relevant topics such as ecosystem and habitat recovery following any impact should be explained.

3.5 SUMMARY OF ENVIRONMENTAL EVALUATION

The methodology presented in Section 3.0 provides a general framework for conducting environmental evaluations as part of the baseline risk assessment. The current process for performing environmental evaluations is not as well developed as the human

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health evaluation. Consequently, evnvironmental evaluations may rely on qualitative assessments more than quantitative, and often rely considerably on professional judgement.

The first phase of the environmental evaluation is the problem definition. Problem definition requires general and site-specific ecological knowledge, and regulatory goals. Three steps are identified in the problem definition:

- Identification of habitats of potential concern,
- Identification of contaminants of potential concern, and
- Selection of assessment and measurement endpoints.

Many components of the Hanford Site ecology have been extensively studied, or are under study. The environmental evaluation methodology recommends that this valuable resource be utilized in defining the problems associated with contaminant releases.

The second phase of the environmental evaluation is analysis. Analysis in the environmental evaluation encompasses the elements of the toxicity assessment and the exposure assessment. These two subprocesses are interrelated and involve an understanding of the spatial, temporal, and ecological boundaries of the project. Potentially operative ecological exposure pathways are provided for the aquatic, riparian, and terrestrial habitats at the Hanford Site.

The third and final phase of the environmental evaluation is the risk characterization. The risk characterization addresses the integration of exposure and toxicity information, the discussion of uncertainty in the environmental evaluation process, and the assessment of baseline ecological conditions.

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4.0 APPLICATIONS OF METHODOLOGY AT THE HANFORD SITE

This methodology has been developed to prepare baseline human health and environmental risk assessments to support RIs and FIs at the Hanford Site. Although the primary application of the methodology is performance of a baseline risk assessment, other applications of the methodology may be appropriate. The potential applications of the methodology, as part of the "Hanford Past Practice Investigation Strategy", are briefly presented in this section. Further refinement of the methodology uses, other than the baseline risk assessment, is beyond the scope of this document.

4.1 BASELINE RISK ASSESSMENT

The baseline risk assessment, as part of the CERCLA process, provides an evaluation of the potential threat to human health and the environment in the absence of remedial action. The NCP calls for a site-specific baseline risk assessment [40 CFR §300.430(d)(4)]. As indicated in the Preamble to the NCP (FR Vol.55, No.46, p.8709), this assessment provides a basis for determining whether or not remedial action is necessary and the justification for performing remedial actions.

For carcinogenic substances, acceptable exposure levels are generally concentration levels that represent an excess upper bound lifetime cancer risk to an individual of between 1E-04 to 1E-06, with 1E-06 the point of departure [i.e., starting point or initial "protectiveness goal" (FR Vol.55, No.46, p.8718)] for determining remediation goals when ARARs are not available or are not sufficiently protective because of the presence of multiple contaminants or multiple exposure pathways [40 CFR §300.430(e)(2)(i)(A)(2)]. For systemic toxins, acceptable exposure levels are concentration levels to which human populations, including sensitive subgroups, may be exposed without adverse effects during a lifetime or part of a lifetime, incorporating an adequate margin of safety (e.g., a hazard quotient of one) [40 CFR §300.430(e)(2)(i)(A)(1)].

The MTCACR is a potential state ARAR that could impact the use of the baseline risk assessment results in determining remediation goals at CERCLA sites as discussed above. The MTCACR defines a risk assessment framework that can be utilized to establish cleanup levels (WAC 173-340-708). Within the MTCACR framework, three methods are described for establishing cleanup levels at a site. Method A incorporates a table of numeric standards and is intended for use at sites undergoing routine cleanup with relatively few hazardous substances (WAC 173-340-704). Thus, method A may not be applicable to the Hanford Site. Method B is used for developing cleanup levels unless conditions for using another method (A or C) are demonstrated (WAC 173-340-705). Method C cleanup levels represent concentrations which are protective of human health and the environment for specified uses as discussed in WAC 173-340-706. Although the applicability of Method C must be determined on a site-by-site basis, many areas at the Hanford Site, for example, may meet the criteria established in WAC 173-340-745 for establishing soil cleanup levels which are:

- The site is zoned or has been otherwise officially designated for industrial use,

- The site is currently used for industrial purposes or has a history of use for industrial purposes,

- Adjacent properties are currently used or designated for use for industrial purposes,

- The site is expected to be used for industrial purposes for the foreseeable future due to site zoning, statutory or regulatory restrictions, comprehensive plans, adjacent land use, and other relevant factors, and

- The cleanup action provides for institutional controls implemented in accordance with WAC 173-340-440.

By comparison to the NCP, the human health risk assessment procedures of the MTCACR calculate cleanup levels based on a carcinogenic risk less than or equal to 1E-06 and a hazard index less than or equal to one for Method B. For individual substances under Method C, the cleanup levels are calculated based on a carcinogenic risk less than or equal to 1E-05 and a hazard index less than or equal to one. For multiple hazardous substances and/or pathways, the carcinogenic risk shall not exceed 1E-05 and the hazard index shall not exceed one (WAC 173-340-708). These limits are applicable to both Method B or Method C.

It is noted that an acceptable exposure level associated with a lifetime cancer risk of 1E-04 or less is sufficiently conservative that satisfaction of such a goal ensures that any current radiation protection standards (e.g., DOE Order 5400.5) pertinent to the Hanford Site will be met. These standards have two components: a radiation dose limit to individuals; and an "as low a reasonably achievable" (ALARA) principle. The lower limit for application of the ALARA process is a fatal health risk of 1E-05 due to radiation exposure (NCRP 1987). Therefore, adoption of NCP acceptable exposure levels or MTCACR cleanup levels ensures automatic compliance with current radiation protection standards.

The methodology presented in this document has been developed to provide estimates of baseline risks that are meaningful should they be evaluated in the context of the requirements of the NCP or the MTCACR. For example, the preliminary screening discussed in Section 2.1 is used to identify contaminants at a conservative level so that focused risks assessments can be conducted to efficiently, yet conservatively, identify those contaminants that contribute the most risk. This screening has been designed to meet the constraints of the MTCACR risk levels of 1E-05 and the hazard index of 1 for multiple substances. The exposure assessment, exposure scenarios, and exposure parameters are based on the MTCACR risk assessment framework, most of which have been derived from EPA exposure assessment methodologies. The toxicity assessment and risk characterization also are consistent with both the NCP and MTCACR.

The HSBRAM has been developed to support the Hanford Past-Practices Strategy. The four NPL sites at the Hanford Site have multiple waste units and multiple operable units. Therefore, by necessity, investigations and remedial actions at some sites may precede investigations and actions at other sites. Consequently, baseline conditions may be in flux. The baseline risk assessment methodology can be used to conduct baseline risk assessments for evaluating individual waste units, operable units, or aggregate areas as the environmental restoration activities proceed at the Hanford Site.

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4.2 QUALITATIVE RISK ASSESSMENT TO SUPPORT THE IRM PATH

The baseline risk assessment methodology can also be used to support the Hanford Past-Practice Investigation Strategy IRM path. The IRM path is used where existing data are sufficient to indicate that a site poses a risk through one or more pathways and additional investigations are not needed to screen the likely range of remedial alternatives for interim actions. The IRM path calls for a qualitative risk assessment. A qualitative risk assessment is defined in the Hanford Past-Practices Strategy as "a judgement not based solely on quantification, agreed to by the parties, based upon available site data regarding the threat posed by site contamination" (DOE-RL 1991). Thus, the qualitative risk assessment includes the calculation of site risks, along with evaluation of other site information, that TPA representatives will use to determine that an IRM is appropriate. This strategy is a bias for action to begin cleanup at priority sites.

The IRM path is based on a strategy of two passes to evaluate a site. If an IRM is justified at a site, remedial actions are begun (i.e., a first pass) with additional sampling and characterization conducted during the remedial action. This additional data collection is used to evaluate the site after the IRM is completed (e.g., a baseline risk assessment with more extensive information) to determine if additional site cleanup (i.e., a second pass) may be required.

The risk assessment methodology can be used to estimate the risk associated with a potential IRM site. This risk would then be compared to a trigger risk level, as agreed to by the TPA representatives, to determine if the estimated risk for that site exceeds the trigger level. Consistent with the NCP, and as recommended in EPA 1991e, the trigger risk level for a waste unit could be based on a risk of 1E-04. Remedial action is generally warranted at a site when the site risk exceeds 1E-04 (EPA 1991b).

The IRM path would also require that acceptable exposure levels be determined for the contaminants identified at the site so that there are remediation goals for the waste site. The acceptable exposure levels are based on exposure and toxicity information combined with a target risk level.

At the Hanford Site, it is recommended that the commercial/industrial scenario be used for estimating risk associated with a site and for calculating acceptable exposure levels for the IRM path. The industrial scenario is recommended because the current land use at the Hanford Site is industrial. Based on the Hanford Site Development Plan (DOE-RL 1990) and the position of DOE-RL, as stated in the preface, this will be the land use in the foreseeable future. In addition, the industrial scenario has a set of well-defined exposure pathways and parameters associated with it. The application of other scenarios, such as a residential, would screen fewer sites (i.e., most sites would be expected to have a high risk associated with them) making prioritization of sites and justification of remedial action at this time on specific sites more difficult.

For individual carcinogens, for example, the acceptable exposure level could be based on an incremental lifetime cancer risk of 1E-06 as the target risk level. An incremental lifetime cancer risk of 1E-06 or less is generally considered insignificant for regulatory purposes as the NCP considers this risk level as its point of departure (40 CFR 300.430). Also, 1E-06 is an order of magnitude lower than the MTCACR 1E-05 target risk for sites with multiple substances and pathways.

For systemic toxins, the NCP requires that "...acceptable exposure levels shall represent concentration levels to which the human population, including sensitive subgroups, may be exposed without adverse effect during a lifetime or part of a lifetime, incorporating an adequate margin of safety" [§300.430(e)(2)(i)(A)(1)]. The MTCACR require that the hazard index (i.e., the sum of multiple hazard quotients) for multiple substances with similar effects shall not exceed one (WAC 173-340-708). Based on contamination information found in the current work plans, it is anticipated that multiple substances will be present at most sites. Therefore, something less than one should be selected for the acceptable exposure level for an individual substance in a specific media.

It is suggested that the acceptable exposure level in a medium for an individual substance would be based on a hazard quotient of 0.3. A hazard quotient of 0.3 would allow for the presence of the same contaminant in multiple media and would provide, to a certain extent, for multiple chemicals with similar effects. Furthermore, little is known about the additive or synergist effects of multiple chemicals so this is a somewhat conservative approach. However, it is also recommended that before determining an acceptable exposure level based on a hazard quotient of 0.3, the types of contaminants and their toxic effects should be evaluated to determine if this concentration is appropriate. In reality, it will probably be the rare situation at the Hanford Site where the noncarcinogenic effects in humans would actually drive final remediation at a site.

Targeting acceptable exposure levels to these levels during the IRM would result in cleanup of the contamination posing the greatest risk. It would also provide for multiple substances, multiple pathways, and multiple sites that may contribute to the overall risk at an operable unit or aggregate area. A second pass including an additional risk assessment to identify residual risk and final cleanup would follow.

The following general information is presented to show how modifications are made to the basic intake equations provided in 2.2.5.3 to determine acceptable exposure levels based on target risks. The result is a reduction of the multiple steps that are usually performed in the risk assessment process. The formulas are rearranged to identify concentrations in a medium associated with a given risk rather than vice versa which is the usual case. This is referred to as the risk-based acceptable exposure level.

For nonradioactive carcinogens, the basic equation to calculate the risk-based acceptable exposure level in a medium is:

$$C = \frac{TR \times BW \times AT}{SF \times IR \times EF \times ED}$$

where:

C =	Risk-based acceptable exposure level
TR =	Target excess individual lifetime cancer risk (1E-06)
BW =	Body weight (kg)
AT =	Averaging time (yr x 365 d/yr)
SF =	Chemical specific cancer slope factor $(mg/kg/d)^{-1}$
IR =	Contact rate (variable)
EF =	Exposure frequency (d/yr)
ED =	Exposure duration (yr)

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Typical units for acceptable exposure levels are mg/kg, mg/L, and mg/m^3 for non-radioactive contaminants in soil, water, and air, respectively.

For radioactive contaminants, the general equation to calculate risk-based acceptable exposure levels is:

$$C = \frac{TR}{SF \times IR \times EF \times ED}$$

where:

C =	Risk-based acceptable exposure level
SF =	Radionuclide-specific slope factor
IR =	Contact rate (variable)
EF =	Exposure frequency (d/yr)
ED =	Exposure duration (yr)

Typical units for acceptable exposure levels are pCi/g, pCi/L, and pCi/m^3 for radioactive contaminants in soil, water, and air respectively.

Risk-based acceptable exposure levels should also be calculated for noncarcinogenic substances as discussed above. For noncarcinogens the equation is:

$$C = \frac{THQ \times RfD \times BW \times AT}{IR \times EF \times ED}$$

where:

C =	Risk-based acceptable exposure level
THQ =	Target Hazard Quotient (0.3)
RfD =	Chemical specific chronic reference dose (mg/kg/d)
BW =	Body weight (kg)
AT =	Averaging time (yr x 365 d/yr)
IR =	Contact rate (variable)
EF =	Exposure frequency (d/yr)
ED =	Exposure duration (yr)

Typical units for acceptable exposure levels are mg/kg, mg/L, and mg/m^3 for non-radioactive contaminants in soil, water, and air, respectively.

The remaining parameters in both equations would be based on the industrial scenario, but parameters from other scenarios could be used.

Two important considerations in the application of the methodology to the qualitative risk assessment discussed above should be noted. First, the qualitative assessment is intended to be only part of the justification for an IRM. Secondly, the acceptable exposure levels are only estimates. Final cleanup levels would be based on multiple considerations and would be documented in the record of decision. The application of the methodology discussed above is a recommendation of how the baseline risk assessment methodology can be used within the current strategy.

4.3 OTHER RISK ASSESSMENTS

Other applications of the methodology may include assessing risks potentially associated with remedial action alternatives. Although the general baseline risk assessment methodology is applicable for human health evaluations, site-specific modifications would be required based on the location of receptors with respect to the site, remedial activities selected, exposure durations, or other site-specific factors. The methodology should not be directly applied without appropriate modifications. Modifications would also be necessary to evaluate more extensive ecological concerns that would extend beyond the scope of the baseline risk assessment which focuses on contaminant impacts only. Evaluation of remedial alternatives with respect to ecological receptors would need to consider such effects as habitat destruction from implementation of remedial alternatives, impacts to nesting areas because of increased human activities, and other more broad ecological impacts.

The methodology may also be used for assessing the residual risk on a waste unit, operable unit, or aggregate area after IRMs or other cleanups have been completed. The use of the methodology for such purposes would require minimal modifications. A key consideration would be the selection of exposure pathways and media for receptors with access to large areas.

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APPENDIX A

EXPOSURE SCENARIOS

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EXPOSURE SCENARIOS

This appendix provides four exposure scenarios for use in Hanford Site risk assessments. The four scenarios are: commercial/industrial, recreational, residential, and agricultural. Application of the scenarios in individual risk assessments should be based on site-specific information and characterization of exposed populations as discussed in Section 6.2.2 of RAGS (EPA 1989a) and in accordance with WAC 173-340-708. Additional information on pathways is provided in subsection 2.3.3.

The scenarios provided below include discussions of the exposure assumptions and parameters used to develop each scenario. The exposure parameters are based on a blend of conservative exposure parameters from MTCACR (a potential state ARAR), site-specific parameters, standard EPA default exposure parameters, and professional judgement. Based on the concept of reasonable maximum exposure as recommended by RAGS (EPA 1989a) and MTCACR (WAC 173-340-708), the most conservative parameter is not always used. The rationale for the selection of specific exposure parameters is presented under each exposure scenario.

All references cited in this appendix are provided in Section 5.0 of the Hanford Site Baseline Risk Assessment Methodology.

COMMERCIAL/INDUSTRIAL SCENARIO

A site-specific industrial scenario should be developed and used as a current scenario if industrial activities are currently conducted at the site. Site-specific exposure parameters related to type of activities (e.g., office workers, maintenance workers, etc.), frequency and duration of activities (e.g., daily, monthly, etc.), and media contact (e.g., source of drinking water) should be applied and the rationale for their use justified and documented in the risk assessment.

A default commercial/industrial scenario has also been developed for use as a future scenario or wherever appropriate for current site-specific activities. This scenario represents exposures that may occur to a person whose job at a site is primarily indoors, but would include some outside activities, for example building and grounds maintenance, that could result in exposure to the soil sufficient to incur soil ingestion and dermal contact exposures on a less than daily basis. If changes in the current use of the Hanford Site occur, such a scenario could represent future commercial/industrial workers that would have combination indoor/outdoor work responsibilities such as facility maintenance, hardware/lumber sales, or farm equipment sales. The scenarios conservatively assume that workers do not wear protective clothing while working.

A discussion of the pathways and assumptions used to evaluate the risk associated with the commercial/industrial scenario is presented below. The pathways represent exposure pathways recommended in EPA 1991a, EPA Region-10, MTCACR, and Hanford Site-specific pathways (e.g., radionuclide exposures), as appropriate. Exposure parameters and factors for the commercial/industrial scenario are summarized in Tables A-1 and A-2.

Several assumptions are common to all exposure pathways. Since adults are the only receptor population, an exposure duration of 20 years (WAC 173-340-745) and body

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weight of 70 kg (MTCACR, EPA 1991a, EPA-10 1991) are used to evaluate carcinogenic and noncarcinogenic contaminants. The averaging time for noncarcinogens is always equal to the exposure duration, converted to days, while the averaging time for carcinogens is 70 yr (x 365 d/yr), in accordance with EPA 1991a. Although MTCACR uses a 75 yr averaging time, the derivation of EPA slope factors for estimating lifetime cancer risks is based on a 70 yr averaging time. The use of the 70 yr value results in a more conservative estimate of the incremental lifetime cancer risk because the intake is averaged over a shorter time period.

Direct soil exposure pathways: Three primary exposure pathways have been identified that should be evaluated as part of the commercial/industrial scenario. These pathways include:

- ingestion of contaminated soil,
- dermal contact with the soil, and
- external exposure from radionuclides in the soil.

The MTCACR provides standard exposure parameters for exposure to soil at industrial sites for the ingestion pathway (WAC 173-340-745). These parameters are used in evaluating soil ingestion. For purposes of the methodology, the same exposure frequency, exposure duration, body weight, and averaging time are also applied to dermal contact with soil. Additional dermal exposure parameters, as required, are based on the "Interim Guidance for Dermal Exposure Assessment" (EPA 1991c). All parameters are presented in Tables A-1 and A-2 with appropriate references to the source of the parameter.

The MTCACR assumes a frequency of contact of 0.4 to represent a reasonable maximum soil exposure. This parameter has been retained for the commercial/industrial scenario. The climate at the Hanford Site (hot summers and cold winters) supports the assumption that outside activities would not be likely for most workers on a daily basis.

Air Exposure Pathways: The potential air exposure pathways include:

- inhalation of fugitive dust, and
- inhalation of volatile emissions from the soil.

The MTCACR Method C provides parameters for evaluating industrial/commercial exposures to airborne contaminants under WAC 173-340-750. These parameters are used for evaluating exposures under the commercial/industrial scenario and are presented in Tables A-1 and A-2. The exposure frequency of 250 d/yr, recommended by EPA 1991a, is used to represent the number of working days per year.

Groundwater Exposure Pathways: The potential groundwater exposure pathways are:

- direct ingestion of groundwater,
- inhalation of volatile contaminants from groundwater use at work, and
- dermal contact with groundwater during showering.

For any of these pathways to be operable, the risk assessor should evaluate both the potential for groundwater use and whether site-specific conditions or modeling indicate that contaminants from the site impact the groundwater.

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Ingestion exposures of groundwater should be evaluated for all sites where groundwater use is likely. Inhalation of volatiles and dermal contact with the groundwater should be quantitatively evaluated when site contamination suggests these may be important routes of exposure (e.g., volatile or organic chemicals are present in the groundwater). The risk assessor should consult EPA 1991c and EPA-10 1991 for additional guidance on evaluating dermal exposures.

Specific parameters for evaluating exposure through ingestion of groundwater in the commercial/industrial scenario are based on EPA 1991a. These parameters have been selected because the water intake parameters for Method B of MTCACR (e.g., included exposure of a child) and the exposure parameters for Method C (2 L/d with institutional controls) are not typical of work-place water consumptions. Standard default parameters for dermal contact with water and inhalation of volatiles from water use as provided in EPA 1991a and EPA-10 1991 are used for evaluating the dermal contact and volatile inhalation pathway. Modification has been made to use the MTCACR WAC 173-340-745 commercial/industrial exposure duration of 20 yr for all exposure pathways in this scenario.

Surface Water Exposure Pathways: The potential surface water exposure pathways are the same as those listed above for groundwater. As with the groundwater pathway, the surface water pathways should only be considered if site-specific conditions indicate that site contaminants will impact the surface water and surface water is used at a site. Furthermore, the surface water pathway would generally be evaluated in place of the groundwater pathway. Appropriate modifications would be required if both groundwater and surface water were used at a site. The parameters for evaluating the surface water pathways are the same as those used for evaluating groundwater exposure pathways.

RECREATIONAL SCENARIO

A recreational scenario is provided because recreational activities associated with the Columbia River could potentially result in exposure to hazardous substances released from the Hanford Site. As discussed above, these recreational activities currently include activities such as hunting, fishing, boating, water skiing, and swimming. The recreational scenario presented considers pathways related to these current activities and incorporates additional pathways, as appropriate, that may occur in the future should recreational use of the Hanford Site be expanded. For use in current scenarios, only those exposure pathways that are directly related to a site should be evaluated. For most risk assessments, this will be limited to the surface water, air, sediment, and biota exposures since direct access to the Hanford Site for recreational purposes is limited to the bank of the Columbia River up to the high water mark. Future scenarios could include on-site exposures pathways related to soil and groundwater. Therefore, these pathways have been included in the recreational scenario.

A discussion of the pathways used to evaluate the risk associated with the recreational scenario is provided below. The MTCACR, although acknowledging that recreational activities may occur at a site, does not provide parameters for evaluating recreational exposures. Similarly, EPA does not currently provide standard default parameters for exposures that may occur during recreational activities other than for swimming (EPA 1991a). Therefore, exposure parameters are derived based on information contained in the "Exposure Factors Handbook" (EPA 1989i), EPA 1991a, EPA-10 1991, EPA

1991c, and residential exposure parameters of MTCACR. The rationale for the parameters used is described in the pathway discussions. The exposure parameters are summarized in Tables A-3 and A-4 with appropriate references to the sources of the parameters.

Several assumptions are common to all exposure pathways. These are the exposure frequency, the exposure duration, body weights, and the averaging times. Since much of the recreation is centered around the Columbia River, the exposure frequency for swimming activities [7 d/yr (EPA 1989a, EPA 1991a)] is considered representative of time spent in outdoor activities during good weather. The mean time that men and women spend in active sports and outdoors (i.e., activity categories 80 and 81 which includes hiking, fishing, hunting, swimming, picnicking, etc.) is 1.70 hr/week (EPA 1989i). During the warmer weather months (e.g., approximately 26 weeks/yr) this would correspond to about 6 days total (8 hr/d) in outdoor recreational activities. Therefore, 7 days is considered appropriate for use as an exposure frequency for evaluation of recreational exposures, in the absence of site-specific data.

If location-specific data becomes available that is representative of the actual frequency of various outdoor activities in the Tri-Cities area, then this information should be used in place of these estimations. A review of exposure parameters used in radionuclide dose surveillance is now being conducted and may provide additional data for estimating recreational frequencies for the Tri-Cities area. The one exception to this exposure frequency is for the ingestion of game and fish, for which daily exposure is assumed.

The exposure duration is based on whether the receptors are exclusively children, exclusively adults, or both adults and children. For example, the MTCACR recommendation of children as the receptor population for noncarcinogens is used for inhalation and soil and water ingestion routes. Dermal exposures are evaluated for the RME combination, in accordance with EPA-10 1991. Adults are assumed to be the only receptors ingesting game and fish, as is recommended for biota exposures (EPA-10 1991 and EPA 1991a).

Body weights are largely determined by the choice of receptor (i.e., child or adult). The child body weight (16 kg) is consistent with MTCACR recommendations. The 70 kg adult weight is recommended by MTCACR, EPA 1991a, and EPA-10 1991.

For noncarcinogens, the averaging time is equal to the exposure duration converted to days [i.e. yr x (365 d/yr)] Although MTCACR uses a 75 yr averaging time for carcinogens, the derivation of EPA slope factors for estimating lifetime cancer risks is based on a 70 yr averaging time. Therefore, the averaging time for evaluating carcinogens is 70 yr (converted to days), in accordance with EPA 1991a and consistent with the assumptions used in the development of cancer slope factors. The use of the 70 yr value results in a more conservative estimate of the incremental lifetime cancer risk because the intake is averaged over a shorter time period.

Direct Soil Exposure Pathways: These pathways include:

- ingestion of soil,
- dermal contact with soil, and
- external exposure to radionuclides.

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These exposure pathways are those that would occur during outside recreational activities such as picnicking, fishing, hunting, or hiking. All of these pathways are considered primary pathways that should be evaluated for recreational exposures on the Hanford Site.

Other than the exposure parameters discussed above, the remaining parameters for recreational soil exposures are based on MTCACR Method B residential soil exposures (WAC 173-340-740) with modification as indicated below. For noncarcinogens, the parameters are a soil contact rate of 200 mg/d, an average body weight of 16 kg, and an exposure duration of 6 yr. The typical child exposure values are used because these are representative of a potentially sensitive subpopulation.

For carcinogens, the MTCACR Method B parameters have been modified to reflect the parameters recommended by EPA 1991a and EPA-10 1991 except for the child body weight which for consistency is 16 kg as recommended throughout the MTCACR. These modifications to MTCACR recommendations have been made because the exposure to carcinogens at an earlier age is potentially more toxicologically significant and should be considered. The use of these modified factors is more conservative than the use of MTCACR parameters. The exposure parameters for carcinogens are child contact rate of 200 mg/d, average body weight of 16 kg, exposure duration of 6 years in addition to exposure of an adult at 100 mg/d, average body weight of 70 kg, and exposure duration of 24 years.

Dermal contact with soil by children and adults is also assumed to occur with the same frequency (7 d/yr) and duration as the soil ingestion pathways. The remaining dermal exposure parameters are standard exposure factors provided in EPA 1991c or EPA-10 1991, as noted in the tables.

Air Exposure Pathways: The only air pathways considered in the recreational scenario are the inhalation of fugitive dust and the inhalation of volatile emissions that may be associated with a site. These are considered primary exposure pathways and should be evaluated in the recreational scenario. The frequency of contact is 7 d/yr, consistent with the soil exposure pathways. The remainder of the parameters are the same as those provided in the MTCACR (WAC 173-340-750).

Groundwater Exposure Pathways: The direct ingestion of groundwater is provided as a potential exposure pathway only if groundwater is likely to be a source of drinking water. The contact rates and other exposure factors are the same as provided in Method B of MTCACR (WAC 173-340-720) except for the exposure frequency (7 d/yr). Under current recreational exposure scenarios, groundwater is not accessible for consumption except at some springs along the river.

The potential exposure to groundwater contaminants via other pathways such as dermal absorption from contact with the groundwater or the inhalation of volatiles is not recommended for quantitative assessment. If contaminants at a site suggest that this may be a significant pathway (i.e., extensive groundwater contamination with organics or volatiles), then the risk can be assessed quantitatively. For most cases, it is assumed that a qualitative evaluation will be appropriate.

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Surface Water Exposure Pathways: The direct ingestion of surface groundwater is provided as a potential exposure pathway only if surface water is likely to be a source of drinking water. The contact rates and other exposure factors are the same as provided in Method B of MTCACR for ingestion of groundwater except for the exposure frequency (7 d/yr). The consumption of surface water as a primary drinking source would be in place of groundwater as a drinking source.

Dermal contact with surface water in the Columbia River during swimming should be assessed if site contaminants impact the Columbia River. The exposure parameters provided are standard default parameters for swimming (EPA 1991c). Dermal absorption should only be quantitatively assessed if sufficient information is available to derive an appropriate chemical-specific absorption factor. If absorption information is unavailable, this pathway may be qualitatively evaluated.

Sediment Exposure Pathways for the Columbia River: These pathways are considered secondary pathways that should be assessed only if there is sufficient information to identify sediment impacts from a specific site. Direct ingestion of sediment that may occur during swimming events should be evaluated using soil ingestion parameters. Dermal contact with sediment may also occur during swimming or wading events. Exposure by the dermal route would be limited because sediment would continually be washed off by contact with the water.

Biota Exposure Pathways: Several potential recreational exposure pathways related to the ingestion of biota should be considered for the recreational scenario. It is important that a plausible connection be made between the contamination at a site and the likelihood that biota are impacted by contaminants from that site. All analysis of biota pathways should be well-documented in the risk assessment. The potential exposure pathways include:

- Ingestion of fish from the Columbia River,
- Ingestion of game (deer) foraging on contaminated sites,
- Ingestion of waterfowl, and
- Ingestion of native plants.

Parameters for evaluating these exposure pathways are provided in Tables A-3 and A-4. Parameters for evaluating risk associated with consumption of waterfowl (e.g., geese and duck) have not been developed by any of the regulatory agencies. These parameters should be developed on a case-by-case basis using professional judgement. The Washington Department of Wildlife provides annual small game harvest summaries that can be used in determining hunter success rates in the region. Modeling information on radionuclide concentrations in biota should be compared to on-going data collected in surveillance of wildlife as discussed in subsection 2.2.5.2.

The MTCACR specifies that soil cleanup levels for other nonresidential site uses such as recreational or agricultural uses shall be established on a case-by-case basis, and these cleanup levels shall be at least as stringent as method C cleanup levels (WAC 173-340-740). Although a recreational scenario is presented here, it is very likely that under current land use other scenarios, such as the commercial/industrial scenario, may dominate the estimation of risk associated with a site. In many cases, the exposures resulting from recreational use will be less than an industrial exposure because the frequency, duration, and magnitude of the recreational exposure is less. An exception to this would be if

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contamination in the food chain (not evaluated under the commercial/industrial scenario) resulted in high exposures.

Revisions in the recreational scenario may be required when options under consideration for the Hanford Reach are finalized. The consideration of the Hanford Reach for protection may change potential recreational uses along the river and on the Hanford Site bordering the river.

RESIDENTIAL SCENARIO

Residential land use of the Hanford Site does not currently occur, and it does not appear likely in the foreseeable future (DOE-RL 1990). Residences are currently located downwind, down river, and in the vicinity of the site. The current residential exposures are primarily limited to contaminants in mobile media, specifically air, water, and some biota such as fish or wildlife. If residential use occurred in the future, on-site receptors would also have the potential for exposure to soil contaminants.

A residential scenario has been developed for use in assessing residential exposures. This scenario evaluates the risks associated with common residential activities that could result in exposure to hazardous substances found on a site.

Several assumptions regarding exposure parameters are common to all pathways. The exposure duration is based on whether the receptors are exclusively children, exclusively adults, or both adults and children. For example, the MTCACR recommendation of children as the receptor population for noncarcinogens is used for inhalation and soil and water ingestion routes. Dermal exposures are evaluated for the RME combination, in accordance with EPA-10 1991. Adults are assumed to be the only receptors ingesting fish, as is recommended for biota exposures (EPA-10 1991 and EPA 1991a).

Body weights are largely determined by the choice of receptor (i.e., child or adult). The child body weight (16 kg) is consistent with MTCACR recommendations. The 70 kg adult weight is recommended by MTCACR, EPA 1991a, and EPA-10 1991.

Fon noncarcinogens, the averaging time is equal to the exposure duration converted to days [i.e. yr x (365 d/yr)] Although MTCACR uses a 75 yr averaging time for carcinogens, the derivation of EPA slope factors for estimating lifetime cancer risks is based on a 70 yr averaging time. Therefore, the averaging time for evaluating carcinogens is 70 yr (converted to days), in accordance with EPA 1991a and consistent with the assumptions used in the development of cancer slope factors. The use of the 70 yr value results in a more conservative estimate of the incremental lifetime cancer risk because the intake is averaged over a shorter time period.

A discussion of the pathways used to evaluate the risk associated with the residential scenario is presented below. The pathways represent exposure pathways recommended by MTCACR, EPA 1991a, EPA -10 1991, and also include Hanford Site-specific pathways (e.g., radionuclide exposures), as appropriate. Exposure parameters and factors for the residential scenario are summarized in Tables A-5 and A-6.

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Direct soil exposure pathways: These pathways include:

- ingestion of contaminated soil,
- dermal contact with the soil, and
- external exposure from radionuclides.

These are typical exposures that may occur while children are playing outside, crawling on the floor, or while adults are working around the yard. All of these pathways are considered primary pathways for residential scenarios. The MTCACR, Method B, provides standard exposure parameters for exposure to soil at residential sites for the ingestion pathway (WAC 173-340-740). These parameters are used in evaluating soil ingestion. For purposes of the methodology. Dermal exposure parameters are based on the "Interim Guidance for Dermal Exposure Assessment" (EPA 1991c) and EPA-10 1991.

For noncarcinogens, the soil ingestion parameters are directly taken from Method B (i.e., a soil contact rate of 200 mg/d, an average body weight of 16 kg, and an exposure duration of 6 yr.) These typical child exposure values are used because these are representative of a potentially sensitive subpopulation.

For carcinogens, however, the parameters have been modified to reflect the parameters recommended by EPA 1991a and EPA-10 1991 except for the child body weight, which for consistency is 16 kg as recommended throughout the MTCACR. These modifications to MTCACR recommendations have been made because the exposure to carcinogens at an earlier age is potentially more toxicologically significant and should be considered. The use of these modified factors is more conservative than the use of MTCACR parameters. The exposure parameters for carcinogens are child contact rate of 200 mg/d, average body weight of 16 kg, exposure duration of 6 years in addition to exposure of an adult at 100 mg/d, average body weight of 70 kg, and exposure duration of 24 years.

Dermal exposures are assumed to occur less frequently than potential soil ingestion for both adults and children. Given that the climate in the vicinity of the Hanford Site is cool to cold for approximately half of the year, the dermal exposure to soil and dirt carried into the house is only assumed to occur at a frequency of 180 days/year as recommended in EPA 1991c. During the period from mid-October to mid April, receptors would wear more clothing that would limit the potential for dermal exposures. Therefore, the frequency of exposure is considered a reasonable maximum exposure.

Air Exposure Pathways: The potential air exposure pathways include inhalation of fugitive dust and inhalation of volatile emissions from the soil. The MTCACR provides parameters for evaluating exposures to airborne contaminants (WAC 173-340-750). These parameters are used for the evaluation of residential air exposures. In addition, the MTCACR [WAC 173-340-740(3)(a) and (4)(b)(iv)] requires that soil concentrations ensure that the release of hazardous substances shall not result in ambient air concentrations which exceed cleanup levels established under WAC 173-340-750.

Groundwater Exposure Pathways: The potential groundwater exposure pathways are:

- direct ingestion of groundwater,
- inhalation of volatile contaminants from groundwater used in the home (e.g.,

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volatilization from dishwashers, showers, or washing), and

- dermal contact with groundwater during showering, kids running through the sprinkler, etc. are considered secondary pathways.

For any of these pathways to be operable, the risk assessor should evaluate both the potential for groundwater use and whether site-specific conditions or modeling indicate that contaminants from the site impact the groundwater.

Inhalation of volatiles and dermal contact with the groundwater should be quantitatively evaluated when site contamination suggests these may be important routes of exposure (e.g., volatile or organic chemicals are present in the groundwater). Evaluation of dermal exposures through showering or bathing is considered representative of the reasonable exposure that could occur through the dermal pathway since this is an ongoing and common exposure for most receptors. Other dermal exposures to water, such as running through the sprinkler, washing cars, watering the lawn, would be much less than exposures evaluated through showering and water ingestion. The risk assessor should consult EPA 1991c and EPA-10 1991 for additional guidance on evaluating dermal exposures and identifying situations for potentially important exposures.

Surface Water Exposure Pathways: The potential surface water exposure pathways are the same as those listed above for groundwater. As with the groundwater pathway, the surface water pathways should only be considered if site-specific conditions indicate that site contaminants will impact the surface water and surface water use is likely. Furthermore, the surface water pathway would be evaluated in place of the groundwater pathway. Appropriate modifications would be required if both groundwater and surface water were used at a site. The parameters for evaluating the surface water pathways are based on the Method B of MTCACR for ingestion of groundwater (WAC 173-340-720) and standard default parameters for dermal contact with water and inhalation of volatiles from water use as provided in EPA 1991a and EPA-10 1991.

In addition to the above surface water pathways, dermal contact with surface water in the Columbia River during swimming is considered a secondary pathway and should be assessed only if site contaminants will impact the Columbia River. The exposure parameters provided in Tables A-5 and A-6 are standard default parameters for swimming (EPA 1991c). Dermal absorption should only be quantitatively assessed if sufficient information is available to derive an appropriate chemical-specific absorption factor. If absorption information is unavailable, this pathway may be qualitatively evaluated.

Sediment Exposure Pathways for the Columbia River: These pathways are considered secondary pathways that should be assessed only if there is sufficient information to identify sediment impacts from a specific site. Direct ingestion of sediment that may occur during swimming events should be evaluated using soil ingestion parameters. Dermal contact with sediment may also occur during swimming or wading events. Exposure by the dermal route would be limited because sediment would continually be washed off by contact with the water.

Biota Exposure Pathways: Potential residential exposure pathways related to the ingestion of biota should be considered for this scenario. However, it is important that a plausible connection can be made between the contamination at a site and the likelihood that biota are impacted by contaminants from that site. All analysis of biota pathways should be

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well-documented in the risk assessment. The potential exposure pathways include:

- Ingestion of fish from the Columbia River,
- Ingestion of garden produce, and
- Ingestion of home-grown fruit.

Parameters for evaluating these exposure pathways are provided in Tables A-5 and A-6. Hunting is not allowed in residential areas, therefore, the ingestion of waterfowl or game are not evaluated for the residential scenario. For the biota pathways, modeling information on radionuclide concentrations in biota should be compared to on-going data collected in surveillance of plants and wildlife as discussed in subsection 2.2.5.2.

AGRICULTURAL SCENARIO

An agricultural scenario has been developed for evaluation of potential risks associated with such land use. Farmland and farm residences are located downwind, down river, and in the vicinity of the site. The current agricultural exposures are primarily limited to contaminants in mobile media, specifically air, water, and biota that may be impacted by transport of airborne or surface-water contaminants. Agricultural land use of the Hanford Site does not currently occur, and does not appear likely in the foreseeable future (DOE-RL 1990). If on-site agricultural use were to occur in the future, this scenario could be used to evaluate the risks from potential exposures to hazardous substances that would be associated with a farm residence on land affected by contamination.

A discussion of the pathways used to evaluate the risk associated with the agricultural scenario is presented below. This scenario expands the residential scenario to include potential exposures via consumption of beef and dairy products from animals grazing on a contaminated site. In addition, consumption of deer is included because of the rural setting.

The MTCACR, although acknowledging that agricultural activities may occur at a site, does not provide parameters for evaluating an agricultural scenario. Parameters associated with residential exposure pathways and animal product consumption factors, as recommended by EPA 1991a and EPA-10 1991, are used with appropriate parameters for Hanford Site-specific pathways (e.g., radionuclide exposures), to evaluate an agricultural farm family scenario.

Neither EPA-10 1991 nor EPA 1991a provide exposure parameters for farm workers. Many farm activities could result in potentially greater exposures to soil (e.g., through direct ingestion, airborne particulate from plowing and harrowing) and dermal contact with water (e.g., from working with irrigation) than would be expected with residential exposures. Although information is available on types of crops grown in the vicinity of the Hanford Site (Watson et al. 1991), the local extension service (as recommended in EPA 1991a) has been unable to provide sufficient information that can be used to develop defensible farm worker exposure factors.

Several assumptions regarding exposure parameters are common to all pathways. The exposure duration is based on whether the receptors are exclusively children, exclusively adults, or both adults and children. For example, the MTCACR

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recommendation of children as the receptor population for noncarcinogens is used for inhalation and soil and water ingestion routes. Dermal exposures are evaluated for the RME combination, in accordance with EPA-10 1991. Adults are assumed to be the only receptors ingesting fish and other biota, as is recommended for biota exposures (EPA-10 1991 and EPA 1991a).

Body weights are largely determined by the choice of receptor (i.e., child or adult). The child body weight (16 kg) is consistent with MTCACR recommendations. The 70 kg adult weight is recommended by MTCACR, EPA 1991a, and EPA-10 1991.

Fon noncarcinogens, the averaging time is equal to the exposure duration converted to days [i.e. yr x (365 d/yr)] Although MTCACR uses a 75 yr averaging time for carcinogens, the derivation of EPA slope factors for estimating lifetime cancer risks is based on a 70 yr averaging time. Therefore, the averaging time for evaluating carcinogens is 70 yr (converted to days), in accordance with EPA 1991a and consistent with the assumptions used in the development of cancer slope factors. The use of the 70 yr value results in a more conservative estimate of the incremental lifetime cancer risk because the intake is averaged over a shorter time period.

Exposure parameters and factors for the agricultural farm family scenario are summarized in Tables A-7 and A-8. Those parameters that may not be representative of reasonable maximum exposures for a farm worker living on a site are noted in the tables. It is recommended that if future agricultural use of specific areas of the Hanford Site is considered likely, then farm worker exposure parameters should be developed through research and documentation of farm practices in the vicinity of the Hanford Site.

Direct soil exposure pathways: These pathways include ingestion of contaminated soil, dermal contact with the soil, and external exposure from radionuclides that may occur while children are playing outside, crawling on the floor, or while adults are working in a garden or field. All of these pathways are considered primary pathways for residential scenarios and would also apply to residences on a farm. The MTCACR, Method B, provides standard exposure parameters for exposure to soil at residential sites for the ingestion pathway (WAC 173-340-740). These parameters are used in evaluating soil ingestion. For purposes of the methodology. Dermal exposure parameters are based on the "Interim Guidance for Dermal Exposure Assessment" (EPA 1991c) and EPA-10 1991.

For noncarcinogens, the soil ingestion parameters are directly taken from Method B (i.e., a soil contact rate of 200 mg/d, an average body weight of 16 kg, and an exposure duration of 6 yr.) These typical child exposure values are used because these are representative of a potentially sensitive subpopulation.

For carcinogens, however, the parameters have been modified to reflect the parameters recommended by EPA 1991a and EPA-10 1991 except for the child body weight, which for consistency is 16 kg as recommended throughout the MTCACR. These modifications to MTCACR recommendations have been made because the exposure to carcinogens at an earlier age is potentially more toxicologically significant and should be considered. The use of these modified factors is more conservative than the use of MTCACR parameters. The exposure parameters for carcinogens are child contact rate of 200 mg/d, average body weight of 16 kg, exposure duration of 6 years in addition to exposure of an adult at 100 mg/d, average body weight of 70 kg, and exposure duration of

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24 years. Although the parameters recommended above are appropriate for residential land use, they may underestimate the exposures occurring for a farm family resident and may be more typical of average exposures than reasonable maximum exposures.

Air Exposure Pathways: The potential air exposure pathways include inhalation of fugitive dust and inhalation of volatile emissions from the soil. The MTCACR provides parameters for evaluating exposures to airborne contaminants (WAC 173-340-750). These parameters are used for the evaluation of air exposures for farm families, but may not be representative of exposure for individuals living on a farm and working in the fields.

Groundwater Exposure Pathways: The potential groundwater exposure pathways are:

- direct ingestion of groundwater,
- inhalation of volatile contaminants from groundwater used in the home (e.g., volatilization from dishwashers, showers, or washing), and
- dermal contact with groundwater during showering, kids running through the sprinkler, irrigation activities, etc.

For any of these pathways to be operable, the risk assessor should evaluate both the potential for groundwater use and whether site-specific conditions or modeling indicate that contaminants from the site impact the groundwater.

Inhalation of volatiles and dermal contact with the groundwater should be quantitatively evaluated when site contamination suggests these may be important routes of exposure (e.g., volatile or organic chemicals are present in the groundwater). Evaluation of dermal exposures through showering or bathing is considered representative of the reasonable exposure that could occur through the dermal pathway since this is an ongoing and common exposure for most receptors. Other dermal exposures to water, such as running through the sprinkler, washing cars, and watering the lawn do not occur on a regular basis. Dermal exposure that would occur during irrigation activities would not be on a regular basis, and given that only one-third of the current crop production is irrigated and the type of mechanized irrigation systems used, it is suggested that this potential exposure be addressed qualitatively. The risk assessor should consult EPA 1991c and EPA-10 1991 for additional guidance on evaluating dermal exposures and identifying situations for potentially important exposures.

Surface Water Exposure Pathways: The potential surface water exposure pathways are the same as those listed above for groundwater. As with the groundwater pathway, the surface water pathways should only be considered if site-specific conditions indicate that site contaminants will impact the surface water and surface water use is likely. Furthermore, the surface water pathway would be evaluated in place of the groundwater pathway. Appropriate modifications would be required if both groundwater and surface water were used at a site. The parameters for evaluating the surface water pathways are based on the Method B of MTCACR for ingestion of groundwater (WAC 173-340-720) and standard default parameters for dermal contact with water and inhalation of volatiles from water use as provided in EPA 1991a and EPA-10 1991.

Dermal contact with surface water in the Columbia River during swimming is considered a secondary pathway and should be assessed only if site contaminants will impact the Columbia River. The exposure parameters provided in Tables A-7 and A-8 are

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standard default parameters for swimming (EPA 1991c). Dermal absorption should only be quantitatively assessed if sufficient information is available to derive an appropriate chemical-specific absorption factor. If absorption information is unavailable, this pathway may be qualitatively evaluated.

Sediment Exposure Pathways for the Columbia River: These pathways are considered secondary pathways that should be assessed only if there is sufficient information to identify sediment impacts from a specific site. Direct ingestion of sediment that may occur during swimming events should be evaluated using soil ingestion parameters. Dermal contact with sediment may also occur during swimming or wading events. Exposure by this route would be limited because sediment would continually be washed off by contact with the water.

Biota Exposure Pathways: Potential exposure pathways related to the ingestion of biota should be considered for the agricultural scenario. It is important that a plausible connection can be made between the contamination at a site and the likelihood that biota are impacted by contaminants from that site. All analysis of biota pathways should be well-documented in the risk assessment. The potential exposure pathways include:

- Ingestion of fish from the Columbia River,
- Ingestion of garden produce,
- Ingestion of home-grown fruit,
- Ingestion of dairy products from animals grazing on contaminated areas,
- Ingestion of beef from animals grazing on contaminated areas, and
- Ingestion of game (deer).

Parameters for evaluating these exposure pathways are provided in Tables A-7 and A-8. For the biota pathways, information on radionuclide concentrations in biota should be compared to on-going data collected in surveillance of plants and wildlife as discussed in subsection 2.2.5.2.

The MTCACR specifies that soil cleanup levels for other nonresidential site uses such as recreational or agricultural uses shall be established on a case-by-case basis, and these cleanup levels shall be at least as stringent as method C cleanup levels (WAC 173-340-740). It is very likely that agricultural exposures could dominate the estimation of risk associated with a site. The exposures associated with agricultural use include residential exposures, which occur with a greater frequency, duration, and magnitude than industrial exposures. In addition, food chain exposures are considered under the agricultural scenario.

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Table A-1

Exposure Factors Used for Hanford Site Human Risk Assessment:
Noncarcinogens

Scenario	Pathway		Exposure Parameters							Summary Intake Factor
	Media	Route	Daily Intake Rate[a]	Exposure Frequency[a] (d/yr)	Exposure Duration[b] (yr)	Body Weight (kg)	Averaging Time[b] (yr x d/yr)	Conversion Factors	Other Factors	
Industrial	Soil	Ingestion	50 mg[b]	146[c]	20	70	20 x 365	1E-06 kg/mg	--	2.9E-07
		Dermal	1 mg/cm^2[d,e,f]	146[c]	20	70	20 x 365	1E-06 kg/mg	ABS[g]	varies
	Air	Inhalation	20m^3[h]	250	20	70	20 x 365	--	--	2E-01
	Ground Water	Ingestion	1L	250	20	70	20 x 365	--	--	9.8E-03
		Inhalation	20 m^3	250	20	70	20 x 365	--	0.5 L/m^3[h]	9.8E-02
		Dermal	10 min[d,i]	250	20	70	20 x 365	1.7E-02 hr/min 1L/1000 cm^3	K_p[j]	varies
	Surface Water	Ingestion	1L	250	20	70	20 x 365	--	--	9.8E-03
		Inhalation	20 m^3	250	20	70	20 x 365	--	0.5 L/m^3[h]	9.8E-02
		Dermal	10 min[d,i]	250	20	70	20 x 365	1.7E-02 hr/min 1L/1000 cm^3	K_p[j]	varies

[a]Parameters recommended in EPA 1991a, except as noted.
[b]Parameters based on WAC 173-340-745 or WAC 173-340-750, Method C.
[c]Derived from frequency of exposure = 0.4 (i.e., 365 d/yr x 0.4 = 146 d/yr) based on WAC 173-350-745.
[d]Interim Guidance for Dermal Exposure Assessment (EPA 1991c)
[e]Skin surface area: 5,000 cm^2 (adult)
[f]Adherence rate/event
[g]Chemical-specific absorption factor
[h]0.0005 x 1,000 L/m^3 (Andelman 1990)
[i]Skin surface area: 20,000 cm^2 (adult)
[j]Chemical-specific permeability coefficient (cm/hr)

Table A-2

Exposure Factors Used for Hanford Site Human Risk Assessment:
Carcinogens

Scenario	Pathway		Exposure Parameters							Summary Intake Factor
	Media	Route	Daily Intake Rate[a]	Exposure Frequency[a] (d/yr)	Exposure Duration[b] (yr)	Body Weight (kg)	Averaging Time (yr x d/yr)	Conversion Factors	Other Factors	
Industrial	Soil	Ingestion	50 mg[b]	146[c]	20	70	70 x 365	1E-06 kg/mg	--	8.2E-08
		Dermal	1 mg/cm^2[d,e,f]	146[c]	20	70	70 x 365	1E-06 kg/mg	ABS[g]	varies
		External	8 hr	250	20	--	--	--	--	--
	Air	Inhalation	20m^3[h]	250	20	70	70 x 365	--	--	5.6E-02
	Ground Water	Ingestion	1 L	250	20	70	70 x 365	--	--	2.8E-03
		Inhalation	20 m^3	250	20	70	70 x 365	--	0.5 L/m^3[h]	2.8E-02
		Dermal	10 min[d,j]	250	20	70	70 x 365	1.7E-02 hr/min 1L/1000 cm^3	K_p[j]	varies
	Surface Water	Ingestion	1L	250	20	70	70 x 365	--	--	2.8E-03
		Inhalation	20 m^3	250	20	70	70 x 365	--	0.5 L/m^3[h]	2.8E-02
		Dermal	10 min[d,j]	250	20	70	70 x 365	1.7E-02 hr/min 1L/1000 cm^3	K_p[j]	varies

[a]Parameters recommended in EPA 1991a, except as noted.
[b]Parameters based on WAC 173-340-745 or WAC 173-340-750, Method C.
[c]Derived from frequency of exposure = 0.4 (i.e., 365 d/yr x 0.4 = 146 d/yr) based on WAC 173-350-745.
[d]Interim Guidance for Dermal Exposure Assessment (EPA 1991c)
[e]Skin surface area: 5,000 cm^2 (adult)
[f]Adherence rate/event
[g]Chemical-specific absorption factor
[h]0.0005 x 1,000 L/m^3 (Andelman 1990)
[i]Skin surface area: 20,000 cm^2 (adult)
[j]Chemical-specific permeability coefficient (cm/hr)

Table A-3.

Exposure Factors Used for Hanford Site Human Risk Assessment:
Noncarcinogens

Scenario	Pathway		Exposure Parameters							Summary Intake Factor
	Media	Route	Daily Intake Rate[a]	Exposure Frequency[b] (d/yr)	Exposure Duration[a] (yr)	Body Weight[a] (kg)	Averaging Time (yr x d/yr)	Conversion Factors	Other Factors	
Recreational	Soil	Ingestion	200 mg	7	6	16	6 x 365	1E-06 kg/mg	--	2.4E-07
		Dermal	1 mg/cm²[c,d,e]	7	6 (C) 24 (A)[f]	16 (C) 70 (A)[f]	30 x 365	1E-06 kg/mg	ABS[g]	varies
	Air	Inhalation	10 m³	7	6	16	6 x 365	--	--	1.2E-02
	Ground Water	Ingestion	1 L	7	6	16	6 x 365	--	--	1.2E-03
		Inhalation	--	--	--	--	--	--	--	--
		Dermal	--	--	--	--	--	--	--	--
	Surface Water	Ingestion	1 L	7	6	16	6 x 365	--	--	1.2E-03
		Inhalation	--	--	--	--	--	--	--	--
		Dermal	2.6 hr[a,e,i]	7	6 (C) 24 (A)[f]	16 (C) 70 (A)[f]	30 x 365	--	K_p[h]	varies
	Sediment[j]	Ingestion	200 mg	7	6	16	6 x 365	1E-06 kg/mg	--	2.4E-07
		Dermal	1 mg/cm²[c,d,e]	7	6 (C) 24 (A)[f]	16 (C) 70 (A)[f]	30 x 365	1E-06 kg/mg	ABS[g]	varies
	Biota	Waterfowl[k]	--	--	--	--	--	--	--	--
		Game	1g[l]	365	30	70	30 x 365	--	0.19[m]	2.7E-03
		Fish	54 g[n]	365	30	70	30 x 365	--	0.5[n]	3.9E-01
		Plant	--	--	--	--	--	--	--	

[a]Parameters recommended in WAC 173-340-720, WAC 173-340-740, or WAC 173-340-750, Method B, except as noted C = Child
A = Adult
[b]Site specific parameter; see text for additional information
[c]Interim Guidance for Dermal Exposure Assessment (EPA 1991c)
[d]Skin surface area: 2,500 cm² (child); 5,000 cm² (adult)
[e]Adherence rate/event
[f]EPA-10 1991
[g]Chemical-specific absorption factor
[h]Chemical-specific permeability coefficient (cm/hr)
[i]Skin surface area: 10,000 cm² (child); 20,000 cm² (adult)
[j]Exposure parameters correspond to swimming events and utilize default parameters for ingestion and dermal exposure to soil
[k]Develop parameters on a site-by-site basis; see text for additional information
[l]Venison fat consumption rate based on 45-kg deer per family per year (Paustenback 1989)
[m]Intake adjusted for upperbound mean hunter success rate of 19% for game management unit 370
[n]WAC 173-340-730

Table A-4

Exposure Factors Used for Hanford Site Human Risk Assessment:
Carcinogens

Scenario	Pathway		Exposure Parameters							Summary Intake Factor
	Media	Route	Daily Intake Rate[a]	Exposure Frequency[b] (d/yr)	Exposure Duration[a] (yr)	Body Weight[a] (kg)	Averaging Time (yr x d/yr)	Conversion Factors	Other Factors	
Recreational	Soil	Ingestion	200 mg (C) 100 mg (A)	7	6 (C) 24 (A)	16 70	70 x 365	1E-06 kg/mg	--	3.0E-08
		Dermal	1 mg/cm²[d,e]	7	6 (C) 24 (A)	16 (C) 70 (A)	70 x 365	1E-06 kg/mg	ABS[f]	varies
		External	8 hr	7	30	--	--	--	--	--
	Air	Inhalation	20 m³	7	30	70	70 x 365	--	--	2.3E-03
	Ground Water	Ingestion	2 L	7	30	70	70 x 365	--	--	2.3E-04
		Inhalation	--	--	--	--	--	--	--	--
		Dermal	--	--	--	--	--	--	--	varies
	Surface Water	Ingestion	2 L	7	30	70	70 x 365	--	--	2.3E-04
		Inhalation	--	--	--	--	--	--	--	--
		Dermal	2.6 hr[d,e]	7	6 (C) 24 (A)	16 (C) 70 (A)	70 x 365	1L/1000 cm³	K_p[h]	varies
	Sediment[i]	Ingestion	200 mg (C) 100 mg (A)	7	6 (C) 24 (A)	16 (C) 70 (A)	70 x 365	1E-06 kg/mg	--	3.0E-08
		Dermal	1 mg/cm²[d,e]	7	6 (C) 24 (A)	16 (C) 70 (A)	70 x 365	1E-06 kg/mg	ABS[f]	varies
	Biota	Waterfowl[j]	--	--	--	--	--	--	--	--
		Game	1g[k]	365	30	70	70 x 365	--	0.19[l]	2.7E-03
		Fish	54 g[m]	365	30	70	70 x 365	--	0.5	1.7E-01
		Plant	--	--	--	--	--	--	--	

[a]Parameters recommended in EPA 1991a, EPA-10 1991, except as noted
[b]Site specific parameters; see text for additional information
[c]Parameters recommended in WAC 173-340-720, WAC 173-340-740, or WAC 173-340-750
[d]Interim Guidance for Dermal Exposure Assessment (EPA 1991c)
[e]Skin surface area: 2,500 cm² (child); 5,000 cm² (adult)
[f]Chemical-specific absorption factor
[g]Skin surface area: 10,000 cm² (child); 20,000 cm² (adult)
[h]Chemical-specific permeability coefficient (cm/hr)
[i]Exposure parameters correspond to swimming events and utilize default parameters for ingestion and dermal exposure to soil
[j]Develop parameters on a site-by-site basis; see text for additional information
[k]Venison fat consumption rate based on 45 kg deer per family per year (Paustenback 1989)
[l]Intake adjusted for upperbound mean hunter success rate of 19% for game management unit 370
[m]WAC 173-340-730

C = Child
A = Adult

Table A-5
Exposure Factors Used for Hanford Site Human Risk Assessment:
Noncarcinogens

Scenario	Pathway		Exposure Parameters							Summary Intake Factor
	Media	Route	Daily Intake Rate[a]	Exposure Frequency[a] (d/yr)	Exposure Duration[a] (yr)	Body Weight[a] (kg)	Averaging Time (yr x d/yr)	Conversion Factors	Other Factors	
Residential	Soil	Ingestion	200mg	365	6	16	6 x 365	1E-06 kg/mg	-	1.3E-05
		Dermal	1 mg/cm[2][b,c,d]	180[e]	6 (C) 24 (A)[f]	16 (C) 70 (A)[f]	30 x 365	1E-06 kg/mg	ABS[g]	varies
	Air	Inhalation	10m[3]	365	6	16	6 x 365	-	-	6.3E-01
	Ground Water	Ingestion	1L	365	6	16	6 x 365	-	-	6.3E-02
		Inhalation	15m[3][h]	365	30	70	30 x 365	-	0.5 L/m[3][i]	1.1E-01
		Dermal	10 min[h,j,k]	365	30	70	30 x 365	1L/1000 cm[3] 1.7E-02 hr/min[k]	K_p[l]	varies
	Surface Water	Ingestion	1L	365	6	16	6 x 365	-	-	6.3E-02
		Inhalation	15m[3][h]	365	30	70	30 x 365	-	0.5L/m[3][i]	1.1E-01
		Dermal	10 min[h,j,k] 2.6 hr[h,k,m]	365 7[h,o]	30	70	30 x 365	1L/1000 cm[3] 1.7E-02 hr/min[k]	K_p[l]	varies
	Sediment	Ingestion	200 mg	7[p]	6	16	30 x 365	1E-06 kg/mg	-	4.8E-08
		Dermal	1 mg/cm[2][b,c,d]	7[p]	6 (C) 24 (A)[f]	16 (C) 70 (A)[f]	30 x 365	1E-06 kg/mg	ABS[g]	varies
	Biota	Fruit	42g[q]	365	30	70	30 x 365	-	--	6.0E-01
		Fish	54g[q]	365	30	70	30 x 365	-	0.5	3.9E-01
		Vegetables	80g[q]	365	30	70	30 x 365	-	--	1.1

[a]Parameters recommended in WAC 173-340-720, WAC 173-340-740, or WAC 173-340-750, Method B, except as noted.
[b]Interim Guidance for Dermal Exposure Assessment (EPA 1991c)
[c]Skin surface area: 2,500 cm2 (child); 5,000 cm[2] (adult)
[d]Adherence rate/event
[e]Site-specific parameter; see text for additional information
[f]EPA-10 1991
[g]Chemical-specific absorption factor
[h]Inhalation rate (EPA 1991a): 20 m[3]/d (total); 15 m[3]/d (indoor)
[i]0.0005 x 1,000 L/m[3] (Andelman 1990)
[j]Skin surface area: 20,000 cm[2] (adult)
[k]Default value for showering
[l]Chemical-specific permeability coefficient (cm/hr)
[m]Default value for swimming
[n]Exposure parameters correspond to swimming events and utilize default parameters for ingestion and dermal exposures to soil
[o]EPA 1991a
[p]WAC 173-340-730

C = Child
A = Adult

Table A-6
Exposure Factors Used for Hanford Site Human Risk Assessment:
Carcinogens

Scenario	Pathway		Exposure Parameters							Summary Intake Factor
	Media	Route	Daily Intake Rate	Exposure Frequency[a] (d/yr)	Exposure Duration[a] (yr)	Body Weight[a] (kg)	Averaging Time (yr x d/yr)	Conversion Factors	Other Factors	
Residential	Soil	Ingestion	200mg (C)[b] 100mg (A)[b]	365	6 (C) 24 (A)	16 (C) 70 (A)	70 x 365	1E-06 kg/mg	-	1.6E-06
		Dermal	1 mg/cm^2[c,d,e]	180[f]	6 (C)[g] 24 (A)[g]	16 (C)[g] 70 (A)[g]	70 x 365	1E-06 kg/mg	ABS[h]	varies
		External	24 hr[i]	365	30	-	-	-	-	-
	Air	Inhalation	20m^3[j]	365	30	70	70 x 365	-	-	1.2E-01
	Ground Water	Ingestion	2L[a]	365	30	70	70 x 365	-	-	1.2E-02
		Inhalation	15m^3[j]	365	30	70	70 x 365	-	0.5 L/m^3[k]	4.6E-02
		Dermal	10 min[a,l,m]	365	30	70	70 x 365	1L/1000 cm^3 1.7E-02hr/min	K_p[n]	varies
	Surface Water	Ingestion	2L[a]	365	30	70	70 x 365	-	-	1.2E-02
		Inhalation	15m^3[j]	365	30	70	70 x 365	-	0.5L/m^3[k]	4.6E-02
		Dermal	10 min[a,l,m] 2.6 hr[a,l,o]	365 7[i]	30	70	70 x 365	1L/1000 cm^3 1.7E-02hr/min	K_p[n]	varies
	Sediment[p]	Ingestion	200 mg (C)[b] 100 mg (A)[b]	7[i]	6 (C) 24 (A)	16 (C) 70 (A)	70 x 365	1E-06 kg/mg	-	3.0E-08
		Dermal	1 mg/cm^2[c,d,e]	7[i]	6 (C)[g] 24 (A)[g]	16 (C)[g] 70 (A)[g]	70 x 365	1E-06 kg/mg	ABS[h]	varies
	Biota	Fruit	42g[b]	365	30	70	70 x 365	-	-	2.6E-01
		Fish	54g[q]	365	30	70	70 x 365	-	0.5[q]	1.7E-01
		Vegetable	80g[b]	365	30	70	70 x 365	-	-	4.9E-01

[a]Parameters recommended in WAC 173-340-720, WAC 173-340-740, or WAC 173-340-750, Method B, except as noted.
[b]EPA 1991a.
[c]Interim Guidance for Dermal Exposure Assessment (EPA 1991c)
[d]Skin surface area: 2,500 cm^2 (child); 5,000 cm^2 (adult)
[e]Adherence rate/event
[f]Site-specific parameter; see text for additional information
[g]EPA-10 1991
[h]Chemical-specific absorption factor
[i]Radionuclides only, as appropriate
[j]Inhalation rate (EPA 1991a): 20 m^3/d (total); 15 m^3/d (indoor)
[k]0.0005 x 1,000 L/m^3 (Andelman 1990)
[l]Skin surface area: 20,000 cm^2 (adult)
[m]Default value for showering
[n]Chemical-specific permeability coefficient (cm/hr)
[o]Default value for swimming
[p]Exposure parameters correspond to swimming events and utilize default parameters for ingestion and dermal exposures to soil

C = Child
A = Adult

[q]WAC 173-340-730

Table A-7

Exposure Factors Used for Hanford Site Human Risk Assessment:
Noncarcinogens

Scenario	Pathway		Exposure Parameters[a]							Summary Intake Factor
	Media	Route	Daily Intake Rate	Exposure Frequency[a] (d/yr)	Exposure Duration (yr)	Body Weight[b] (kg)	Averaging Time (yr x d/yr)	Conversion Factors	Other Factors	
Agricultural	Soil	Ingestion[c,d]	200 mg	365	6	16	6 x 365	1E-06 kg/mg	--	1.3E-05
		Dermal[d,e]	1 mg/cm²[d]	180[g]	6 (C) 24 (A)	16 (C) 70 (A)	30 x 365	1E-06 kg/mg	ABS[h]	varies
	Air	Inhalation[i]	10 m³	365	6	16	6 x 365	--	--	6.3E-01
	Ground Water	Ingestion[j]	1 L	365	6	16	6 x 365	--	--	6.3E-02
		Inhalation[k]	15 m³	365	30	70	30 x 365	--	0.5 L/m³[m]	1.1E-01
		Dermal[d,n]	10 min[o]	365	30	70	30 x 365	1 L/1000 cm³	K_p[p]	varies
	Surface Water	Ingestion[j]	1 L	365	6	16	6 x 365	--	--	6E-02
		Inhalation[k]	15 m³	365	30	70	30 x 365	--	0.5 L/m³[m]	1.1E-01
		Dermal[d,n]	10 min[o] 2.6 hr[q]	365 7[q]	30	70	30 x 365	1 L/1000 cm³	K_p[p]	varies
	Sediment[r]	Ingestion	200 mg	7	6	16	6 x 365	1E-06 kg/mg	--	2.4E-07
		Dermal	1 mg/cm²[d]	7	6 (C) 24 (A)	16 (C) 24 (A)	30 x 365	1E-06 kg/mg	ABS[h]	varies
	Biota[s]	Dairy	300 g	365	30	70	30 x 365	--	--	4.3
		Beef	75 g	365	30	70	30 x 365	--	--	1.1
		Fruit	42g	365	30	70	30 x 365	--	--	6.0E-01
		Game	1 g[t]	365	30	70	30 x 365	--	0.19[u]	2.7E-03
		Fish	54	365	30	70	30 x 365	--	0.5	3.9E-01
		Vegetable	80g	365	30	70	30 x 365	--	--	1.1

[a]WAC 173-340-740; Exposure frequencies modified to 365 d/yr for consistency with WAC 173-340.
[b]Default body weights are those used in WAC 173-340 (i.e., Adult = 70 kg, Child = 16 kg)
[c]Ingestion rate may be representative of average exposures; see text for additional information
[d]Interim Guidance for Dermal Exposure Assessment (EPA 1991c)
[e]Skin surface area child: 2,500 cm² (EPA 1991c); Skin surface area adult: 5,000 cm² (EPA 1991c)
[f]Adherence rate/event
[g]Site-specific parameter
[h]Chemical-specific absorption factor
[i]WAC 173-340-750, Method B
[j]WAC 173-340-720, Method B
[k]EPA 1991a
[l]Outdoor inhalation rate - 20 m³/d; indoor inhalation rate = 15 m³/d
[m]0.0005 x 1,000 L/m³ (Andelman 1990)
[n]Skin surface area: 20,000 cm² (adult)
[o]Default parameter for showering
[p]Chemical-specific permeability coefficient (cm/m)
[q]Default parameter for swimming
[r]Exposure parameters correspond to swimming events and utilize default parameters for ingestion and dermal exposure to soil
[s]Venison fat consumption rate based on 45-kg deer per family per year (Paustenbach 1989)
[t]Intake adjusted factor based on upperbound hunter success rate of 19% for game management unit 370

Table A-8

Exposure Factors Used for Hanford Site Human Risk Assessment:
Carcinogens

Scenario	Pathway		Exposure Parameters							Summary Intake Factor
	Media	Route	Daily Intake Rate	Exposure Frequency[a] (d/yr)	Exposure Duration (yr)	Body Weight[b] (kg)	Averaging Time (yr x d/yr)	Conversion Factors	Other Factors	
Agricultural	Soil	Ingestion[c,d]	200 mg (C) 100 mg (A)	365	6 (C) 24 (A)	16 (C) 70 (A)	70 x 365	1E-06 kg/mg	--	1.6E-06
		Dermal[d,e]	1 mg/cm²[f]	180[g]	6 (C) 24 (A)	16 (C) 70 (A)	70 x 365	1E-06 kg/mg	ABS[h]	varies
		External[i]	24 hr	--	30	--	--	--	--	--
	Air	Inhalation[j]	20m³	365	30	70	70 x 365	--	--	1.2E-01
	Ground Water	Ingestion[j]	2L	365	30	70	70 x 365	--	--	1.2E-02
		Inhalation[k]	15m³	365	30	70	70 x 365	--	0.5 L/m³[m]	4.6E-02
		Dermal[d,n]	10 min[o]	365	30	70	70 x 365	1.7E-02 hr/min[o] 1 L/1000 cm³	K_p[p]	varies
	Surface Water	Ingestion[j]	2L	365	30	70	70 x 365	--	--	1.2E-02
		Inhalation[k]	15m³	365	30	70	70 x 365	--	0.5 L/m³[m]	4.6E-02
		Dermal[d,n]	10 min[o] 2.6 hr[q]	365 7	30	70	70 x 365	1.7E-02 hr/min[o] 1L/1000 cm³	K_p[p]	varies
	Sediment[r]	Ingestion	200 mg (C) 100 mg (A)	7	6 (C) 24 (A)	70	70 x 365	1E-06 kg/mg	--	3.0E-08
		Dermal	1 mg/cm²[f]	7	6 (C) 24 (A)	16 (C) 70 (A)	70 x 365	1E-06 kg/mg	ABS[h]	varies
	Biota[s]	Dairy	300g	365	30	70	70 x 365	--	--	1.8
		Beef	75g	365	30	70	70 x 365	--	--	4.6E-01
		Fruit	42g	365	30	70	70 x 365	--	--	2.6E-01
		Game	1g[t]	365	30	70	70 x 365	--	0.19[u]	1.2E-03
		Fish	54g	365	30	70	70 x 365	--	0.5	1.7E-01
		Vegetable	80g	365	30	70	70 x 365	--	--	4.9E-01

[a]WAC 173-340-740; Exposure frequencies modified to 365 d/yr for consistency with WAC 173-340.
[b]Default body weights are those used in WAC 173-340 (i.e., Adult = 70 kg, Child = 16 kg)
[c]Ingestion rate may be representative of average exposures; see text for additional information
[d]Interim Guidance for Dermal Exposure Assessment (EPA 1991c)
[e]Skin surface area child: 2,500 cm² (EPA 1991c); Skin surface area adult: 5,000 cm² (EPA 1991c)
[f]Adherence rate/event
[g]Site-specific parameter
[h]Chemical-specific absorption factor
[i]WAC 173-340-750, Method B
[j]WAC 173-340-720, Method B
[k]EPA 1991a
[l]Outdoor inhalation rate = 20 m³/d; indoor inhalation rate = 15 m³/d
[m]0.0005 x 1,000 L/m³ (Andelman 1990)
[n]Skin surface area: 20,000 cm² (adult)
[o]Default parameter for showering
[p]Chemical-specific permeability coefficient (cm/hr)
[q]Default parameter for swimming
[r]Exposure parameters correspond to swimming events and utilize default parameters for ingestion and dermal exposure to soil
[s]Venison fat consumption rate based on 45-kg deer per family per year (Paustenbach 1989)
[t]Intake adjusted factor based on upperbound hunter success rate of 19% for game management unit 370
[u]Radionuclides only
C = Child
A = Adult

APPENDIX B

DESCRIPTION OF CODES AND MODELS TO BE USED IN RISK ASSESSMENT

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DOE/RL-91-44

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Description of Codes and Models to be Used in Risk Assessment

Date Published
September 1991

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United States
Department of Energy
P.O. Box 550
Richland, Washington 99352

Approved for Public Release

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FOREWORD

A Risk Assessment Modeling Committee was formed to exchange experiences and opinions relating to the use of numerical models for risk assessment. The committee included representatives from the U.S. Environmental Protection Agency, the Washington State Department of Ecology, the U.S. Department of Energy, and their contractors. In general, the committee meetings have enhanced understanding between the involved parties and hopefully improved the decision-making process. Continuation of the committee is recommended for completion of future milestones related to modeling, selection of additional computer codes, and to address computer code and modeling issues that arise during implementation of remedial investigation/feasibility study activities.

While opinions expressed by all members of the committee have been considered, this document does not necessarily reflect the views of individual committee members. Final selection of the computer codes recommended for use was performed by the Hanford Site Operations Contractor, Westinghouse Hanford Company. This recommendation is contained herein and provided to the U.S. Department of Energy to fulfill the M-29-01 Milestone: "Descriptions of Codes and Models to be Used in Risk Assessment."

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1.0 INTRODUCTION

1.1 PURPOSE

Human health and environmental risk assessments will be performed as part of the Comprehensive Environmental Response, Compensation, and Liability Act of 1980 (CERCLA) remedial investigation/feasibility study (RI/FS) activities at the Hanford Site. Analytical and computer encoded numerical models are commonly used during both the remedial investigation (RI) and feasibility study (FS) to predict or estimate the concentration of contaminants at the point of exposure to humans and/or the environment. For the purposes of this discussion, the term "computer code" or "software" will refer to the list of computer commands that perform mathematical calculations and manipulate data, while the term "model" will refer to the combination of data and computer code that represents or describes a physical system. This document has been prepared to identify the computer codes that will be used in support of RI/FS human health and environmental risk assessments at the Hanford Site. In addition to the CERCLA RI/FS process, it is recommended that these computer codes be used when fate and transport analyses is required for other activities. Additional computer codes may be used for other purposes (e.g., design of tracer tests, location of observation wells, etc.).

This document provides guidance for unit managers in charge of RI/FS activities. Use of the same computer codes for all analytical activities at the Hanford Site will promote consistency, reduce the effort required to develop, validate, and implement models to simulate Hanford Site conditions, and expedite regulatory review. Although creating guidelines for computer codes at the Hanford Site is intended to limit the number of codes used at the Hanford Site, it should not discourage advancements in modeling capability or use of alternative software when warranted. It is recognized that software development is a dynamic process and periodic upgrading will be necessary as better computer codes are developed. Furthermore, unique situations may arise that could be better modeled using software not included in these guidelines.

This document is divided into four section: (1) Introduction, (2) Discussion, (3) Selection Criteria, and (4) Recommendations. The "Discussion" section provides a description of how models will likely be developed and utilized at the Hanford Site. It is intended to summarize previous environmental-related modeling at the Hanford Site and provide background for future model development. The "Selection Criteria" section lists the modeling capabilities that are desirable for the Hanford Site and compares the codes that were proposed for consideration. Only those codes that have been used at the Hanford Site were evaluated. The "Recommendations" section lists the codes proposed to support future risk assessment modeling at the Hanford Site, and provides the rational for the codes selected.

1.2 SCOPE

The specific objective of this document is to satisfy the M-29-01 Milestone of the Hanford Federal Facility Agreement and Consent Order, referred to as the Tri-Party Agreement (Ecology et al. 1989). The direction of this milestone was to "Identify and Submit Descriptions of Codes and Models to be Used in Risk Assessment." A follow-up document will satisfy the requirements of the M-29-02 Milestone, which requires "a plan for development of area wide groundwater models to support risk assessment and to evaluate impacts of changing groundwater flow fields." The third and final milestone (M-29-03) requires a preparation of a risk assessment methodology document. A risk

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assessment committee is guiding the completion of the third milestone. The first two milestones are support documents for the third milestone.

A model is defined as a simplified description of a physical system. When considering human health and environmental risk assessments, the physical system is defined to include the waste site, the environmental setting, and the pathway to the potential receptors. This document is limited to modeling of the release and transport of contaminants from the waste site to the receptor via the air, surface water, and groundwater. Milestone M-29-03 will address the development of a risk assessment methodology that includes modeling of dose and response of the potential receptors. A variety of models, ranging from simplified analytical models to complex computer-encoded numerical models, are available for modeling the fate and transport of contaminants, i.e., prediction of contaminant concentrations in air, surface water, and groundwater. The selection of appropriate models depends on several site-specific factors, including (but not limited to) the nature and extent of contamination, spatial geometry, complexity of the physical system, presence of an exposed population, points of compliance, space and time scales, and extent of site characterization. The primary use of these models will be to predict the concentrations of various contaminants in the air, soil, groundwater, and surface water.

This document reflects an emphasis on the subsurface pathway, specifically unsaturated and saturated groundwater transport. The subsurface pathway was considered more important at this point in time because the vast majority of contaminants at the Hanford Site are found in the soil and groundwater and transport in the subsurface will require evaluation in all cases beginning with the baseline risk assessment or no-action alternative. All proposed remedial actions will be compared to the results from this analysis. Other pathways, including air and surface water, may require more focused consideration in the future depending on the method and level of remediation considered.

Moreover, this document does not address the use of analytical models, waste package models, or geochemical models. Although analytical models are expected to play an important role for preliminary evaluation, they are not included here because they are generally abundant, require little development, and are easy to use, review, and test. Although both waste package and geochemical models may be important for confirmation of field or laboratory observations, it is believed that their use will be infrequent and will be addressed on a case-by-case basis.

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2.0 DISCUSSION

This section sets the stage for how models will likely be developed and utilized in support of risk assessments. Model development will be specifically addressed in the M-29-02 Milestone. The discussion provides the background for Section 3.0 (Selection Criteria) and Section 4.0 (Recommendations).

2.1 MODELING FRAMEWORK

A framework for screening and defining the need for contaminant fate and transport modeling in air, surface water, and groundwater has been prepared by the U.S. Environmental Protection Agency (EPA [EPA 1988]) and is shown in Figure 2-1 for air, Figure 2-2 for surface water, and Figure 2-3 for soils and groundwater. It is recommended that these decision networks be used during the planning process to help structure the RI/FS process and determine the need for and nature of contaminant transport modeling. The following guidelines are proposed:

1) The complexity of the model should be consistent with the objectives of the risk assessment.

   Calculations using simple analytical models may be sufficient for preliminary evaluation, while more complex numerical models may be required for determining the final Record of Decision (ROD). It is expected that detailed numerical modeling will be performed when simpler models reveal the potential for violating standards of safe exposure or health risk. When contaminant inventory is small, the waste form is extremely stable and/or the constituents are relatively benign, the amount of risk may be many orders of magnitude less than allowable standards. Alternatively, in situations where large quantities of relatively toxic constituents are free to migrate, the risk may be clearly unacceptable. Simple analytical models will be relied upon to identify these situations, thereby significantly reducing the time and resources that would be expended if extensive numerical modeling were performed for all situations. This screening approach is analogous to the multi-tired approach recommended by the EPA (EPA 1988). More sophisticated modeling may be necessary to compare remedial alternatives at high risk sites.

   Furthermore, if it is anticipated that detailed modeling will eventually be required, it may be more efficient to begin development of a more powerful numerical model during the early screening stages of risk assessment. The decision between using the initial simple analytical codes or the more powerful numerical codes will be carefully weighed on a case-by-case basis. Input from the regulators is encouraged during screening assessments to help identify the appropriate level of modeling for use in analyses supporting the anticipated ROD.

2) Use of models will be factored into the RI/FS process during the planning stages and considered throughout the RI/FS process.

   During the initial planning process, numerical modeling will be useful to help structure the conceptual model of the physical system, identify potential migration pathways and points of exposure, and to define data needs. During the investigation phase of the RI/FS, modeling will provide a means for interpreting data, revising the conceptual model, and determining if sufficient data have been collected. Additionally, models will be used to provide information for the baseline risk assessment. The FS process will rely on models to estimate the effectiveness, efficiency, economy, and risk posed by the various remediation and mitigation approaches. It is, therefore, important that the proper model be selected and appropriate data is collected in the RI/FS.

(Adapted from EPA, 1988)

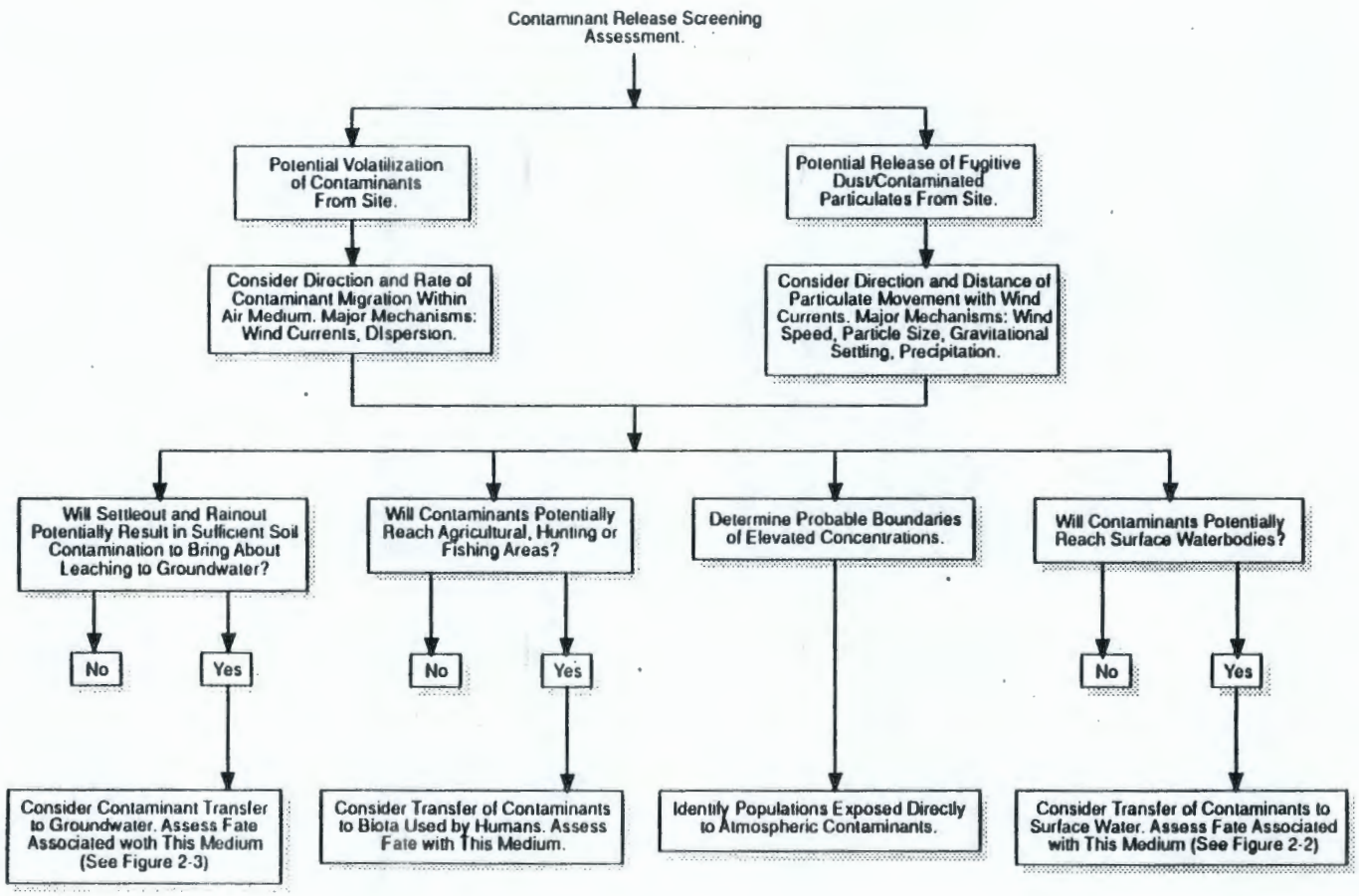

Figure 2-1. Environmental Fate Screening Assessment Decision Network: Atmosphere.

(Adapted from EPA, 1988)

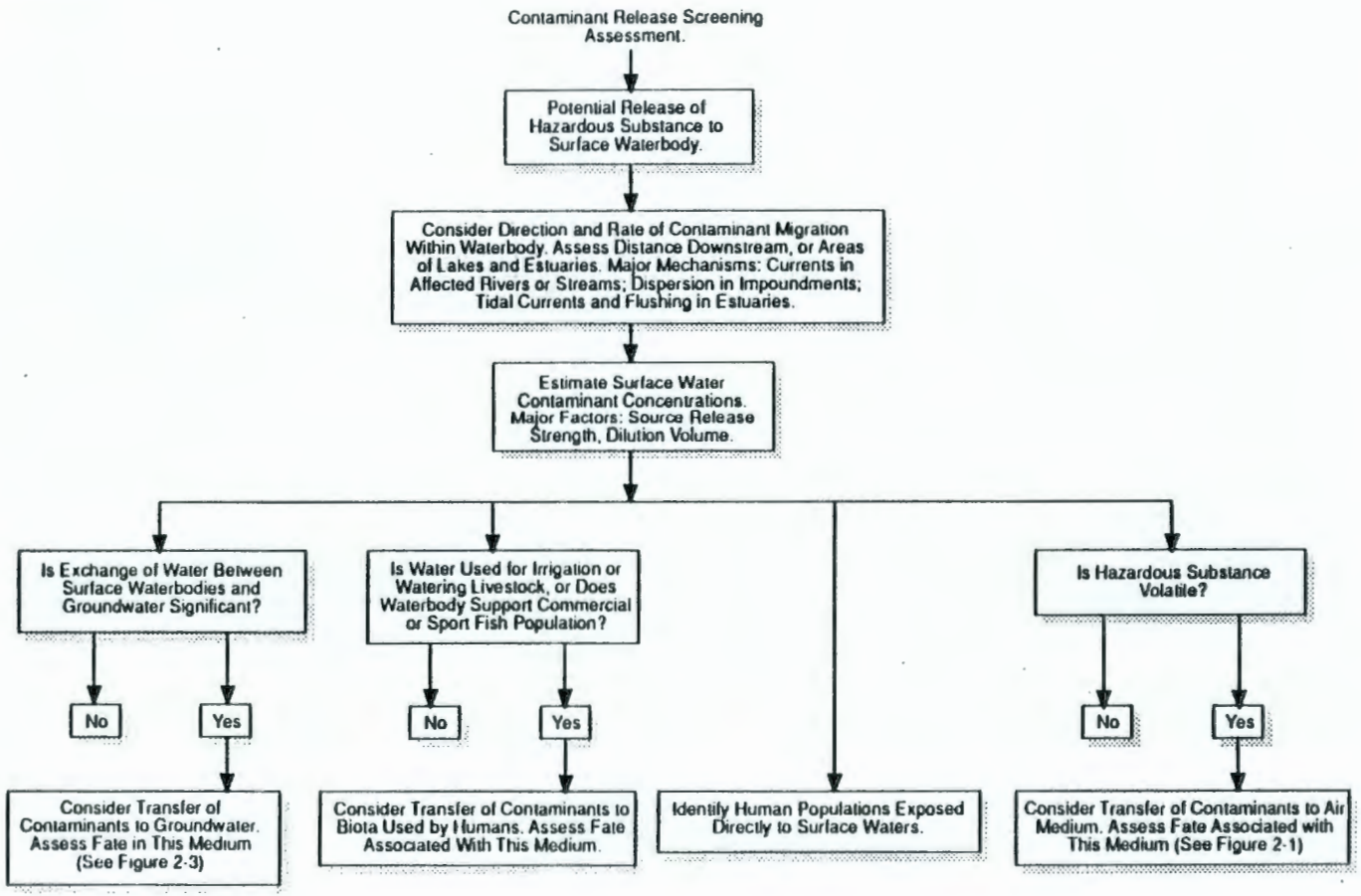

Figure 2-2. Environmental Fate Screening Assessment Decision Network: Surface Water.

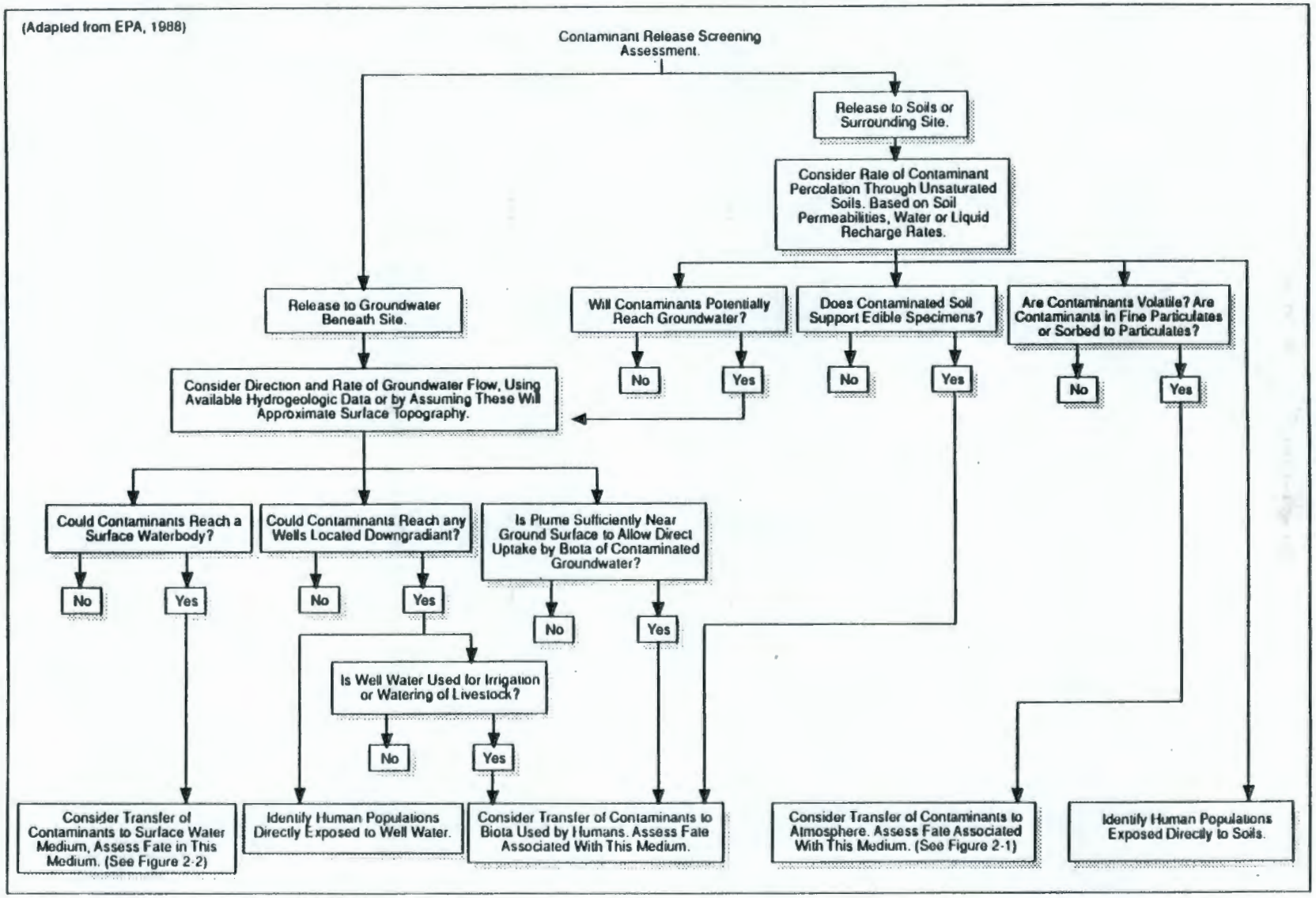

Figure 2-3. Environmental Fate Screening Assessment Decision Network: Soils and Groundwater.

3) Modeling efforts associated with remediation of various waste units at the Hanford Site (waste sites, operable units, aggregate areas) will be coordinated to ensure consistency and transferability of data and results, thereby minimizing total effort.

It is likely that different RI/FS efforts will utilize overlapping or similar models. The characteristics of these models, including general conceptual elements, flow and transport parameters, boundary conditions, and level of complexity should be consistent. Encouragement of such consistency begins with the selection of a standard set of codes and the coordinated development and application of models that use these codes.

4) Improvements in modeling capabilities will be encouraged.

The future may bring improvements in modeling capabilities, and the list of Hanford Site software should evolve to incorporate these technical advances. Changes to the list of Hanford Site software will be based on demonstrated need and undertaken with the consensus of both the technical and regulatory communities.

5) Use of software for risk assessments not included in this document will be allowed given sufficient technical justification.

It is conceivable that situations will arise requiring software capabilities not included in the Hanford Site list of codes. If this occurs, it may be technically justifiable to utilize a computer code that includes the necessary capability even if it is not on the Hanford Site list. Suggested guidelines for approval of new software are provided in Section 4.5.

6) Uncertainty and parameter sensitivity will be qualified with nonprobabilistic approaches.

An evaluation that includes the quantification of uncertainty will be required in most situations. Complete understanding and description of natural hydrogeologic systems is not possible; therefore, model uncertainty is unavoidable given limitations in data collection, modeling capability, and theoretical simplifications. Furthermore, in most situations it is important to know the sensitivity of model results to variations in model parameters. Although uncertainty and parameter sensitivity could be quantified with probabilistic approaches, sufficient data may not be available to provide statistically defensible results. In such cases, nonprobabilistic approaches (such as manual variation of parameters using deterministic models) will allow qualitative assessment of prediction uncertainty. The data needed to quantify estimates of uncertainty and parameter sensitivity will be determined as part of the RI/FS through the establishment of data quality objectives.

2.2 MODEL DEVELOPMENT PROCESS

Model development should continue throughout the RI/FS process in response to new data, improved data interpretation, changing exposure assessment needs, and other factors. A more complete description of the model development process will be provided in the milestone (M-29-02) that will address the development of area-wide models. However, the initial stages of model development are summarized below to illustrate how the needs of the project might affect the selection and application of computer codes.

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1) Identify objectives

All modeling activities should begin with a clear definition of the project objectives. This definition is important because it affects the choice of computer codes, the data needs, the density of the spatial grid, and the effort involved. Future objectives should also be considered during model development.

2) Evaluate existing data

Initial model development will rely on readily available data. Attention will be focused on parameters that have the most impact on predicted concentrations. If data are lacking, it may be necessary to assume values and provide rationale for these assumptions.

3) Define the appropriate conceptual model

Available data will be synthesized into a coherent depiction of the physical system, referred to as a conceptual model. The conceptual model may address a single component or multiple components, including the waste source, the engineered barriers, the surrounding hydrogeologic system, and the potential exposure pathways. Determining which components to include in the model depends on the modeling objectives and the existing understanding of the physical system.

4) Select the appropriate analytical or numerical model

The modeling effort should utilize a level of sophistication that is appropriate considering the modeling objectives, the available data, the complexity of the conceptual model, and the required accuracy of the results. If a numerical model is used, the spatial grid must also be defined at this stage. The ability to address future modeling needs and incorporate future data may affect the choice of models.

5) Incorporate data into the mathematical model and identify additional data needs

This stage of model development requires representation of actual or estimated data as parameters or boundary conditions in the mathematical model. For groundwater modeling, aquifer geometry and stratigraphy from borehole logs and geophysical surveys will be used to define the model boundaries, and measurements of hydraulic characteristics from aquifer tests and laboratory analyses will help determine parameters used in the model. Additional data needs should be identified during this stage.

6) Model calibration

Model calibration involves adjusting hydrogeologic structure, boundary conditions, and aquifer hydraulic parameters until simulated results compare well with observed conditions. For groundwater flow models, calibration may include comparison of observed hydraulic head and gradient conditions with simulated results and comparison of observed plume velocities with simulated velocities.

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Generally, the model should be calibrated using data that supports the primary process of interest. For example, if the model is used primarily to simulate flow and transport, then the model should be calibrated using information on flow (i.e., velocities from tracer studies, etc.) as opposed to an indirect measure such as the calibration against piezometric head variation.

Observations of actual conditions and behavior may not be readily available for characteristically long-term processes, such as vadose zone flow, diffusion from vitrified blocks, and transport of strongly sorbed constituents. For these processes, when no transport data exists, calibration may be difficult or impossible. In such case, it may be possible to bound the behavior, e.g., generally, it is accepted that areal recharge across the Hanford Site averages less than several inches per year.

2.3 PREVIOUS ENVIRONMENTAL TRANSPORT MODELING AT HANFORD

This section provides a summary of computer code development and use at the Hanford Site that supports environmental fate and transport modeling. For the most part, discussion is limited to recent experience that is important with regards to risk assessments and waste isolation performance assessments. This discussion provides a basis for the selection of computer codes and the recommendations found in Section 4.0.

2.3.1 Air

The development of air transport models for use at the Hanford Site was initiated in the 1940's. This activity was originally supported by the Atomic Energy Commission (AEC). The release of airborne contamination (gaseous and particulate emissions) from the stacks of various production facilities has long been recognized as a potential threat to human health and the environment. The basic models to predict transport and dispersion of airborne contaminants that are commonly in use today at the Hanford Site and across the country were derived from pioneering efforts performed under the auspices of the AEC. As a result, there is considerable confidence in these models for predicting the fate and transport of airborne contaminants at the Hanford Site.

Routinely used computer codes supporting Hanford Site operations include GENII and AIRDOS. Both models are used to calculate dose from the interaction of receptors and airborne radioactivity. The air transport model included in GENII is an atmospheric dispersion model that does not take into consideration depletion of air concentrations through deposition or scavenging (Napier et al. 1988). The atmospheric transport model included in GENII is an analytical solution to the multi-dimensional Gaussian diffusion model for continuous release. This analytical methodology is similar to the approach described in the EPA "Superfund Exposure Assessment Manual" (EPA 1988). In contrast to GENII, the atmospheric transport model contained in AIRDOS-PC allows for depletion resulting from the deposition and scavenging of radioactive contaminants (EPA 1989).

Although other more specialized computer codes, such as the Industrial Source Complex (ISC) model, have been used and are available for future use in support of health risk assessments, they have not been used routinely at the Hanford Site. The ISC is designed to address health hazards associated with hazardous chemicals from multiple sources.

2.3.2 Surface Water

The development of surface water models for application at the Hanford Site was initiated during the mid 1960's. Over the years, large quantities of heated effluent from the production reactors in the 100 Areas were discharged directly into the Columbia River. The COLHEAT computer code was developed to predict the fate of heated effluent discharged into the Columbia River (HEDL 1972). Thermographs were installed along the Columbia River beginning in the late 1960's and the temperature records from this network of thermographs were used routinely to calibrate the COLHEAT model. The COLHEAT computer code was used and maintained through the mid-1970's.

A mathematical model to simulate the transport of sediment and radioactivity in the Columbia River was developed in the mid-1970's (Onishi 1977). The resulting computer code (SERATRA) was used in a pilot-scale study to model the longitudinal and vertical distribution of sediments in the Columbia River between Priest Rapids and McNary dams. Sediment and radionuclide interactions and transport were investigated for three sediment fractions, (sand, silt, and clay). Although the preliminary results from the pilot-scale study were encouraging, the model was never used on a routine basis to simulate the transport of sediments and sediment-contaminant interactions in the Columbia River. Although the SERATRA computer code was not used to simulate sediment-contaminant transport in support of Columbia River studies, the code has been used successfully at other locations (Onishi et al. 1982).

The DWOPER computer code has been applied by the U.S. Army Corps of Engineers and the U.S. Bureau of Reclamation to simulate river stage variation in the Hanford Reach of the Columbia River (Fread 1973). The DWOPER code does not address contaminant transport.

2.3.3 Soil and Groundwater

A generic model of the mechanisms that generally influence the modeling of flow and transport of contaminants in the soil and groundwater system is shown in Figure 2-3. These mechanisms include; the release of contaminants to the soils and groundwater that surround the waste site, infiltration of groundwater beyond the root zone, migration of contaminants in partially saturated sediments, migration of contaminants in saturated sediments, multi-phase flow, and geochemistry. A brief description of the history of modeling these processes at the Hanford Site is discussed in the following sections.

2.3.3.1 Release Models. Contaminants find their way into the soil column through planned or accidental releases (spills or leaks), or through waste form degradation. The release of contaminants from specific waste forms rely on knowledge of the chemical and physical processes that govern degradation. With the great variety of wastes and waste containment systems that exist at the Hanford Site, a corresponding range of releases is envisioned. As a result, modeling of waste form release can be achieved by either of two methods: (1) simple, yet conservative models can be used in an attempt to bound the release, or (2) release can be quantified empirically through direct measurement. In support of the Hanford Defense Waste-Environmental Impact Statement (HDW-EIS) (DOE 1987), relatively simple conservative models were used to estimate the release from the various waste forms. Simulated waste forms have been studied in the laboratory to quantify the release from large monolithic grouted waste vaults proposed for use at the Hanford Site (Serne 1990). In either case, it is assumed that the release can be characterized and quantified as a boundary condition (contaminant concentration or mass flux) or initial condition for inclusion into the transport model. As such, it is

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proposed that contaminant release be addressed on a case-by-case basis depending on the specifics of the waste and waste site being assessed.

2.3.3.2 Infiltration Model. The ROD issued for the HDW-EIS (DOE 1987) identified the need for a better understanding on the mechanisms governing the rate of surface infiltration and percolation of water in the partially saturated sediments. Since that time, considerable emphasis has been placed on the quantification and development of analytical and numerical methods that can be used to predict the infiltration of water through partially saturated sediments at the Hanford Site. UNSAT-H has been developed for use at the Hanford Site and reflects the current state-of-the-art understanding of Hanford Site conditions (Fayer and Jones 1990). This computer code simulates the one-dimensional, non-isothermal, dynamic processes of infiltration, drainage, moisture redistribution, evaporation, and plant uptake of water. To date, calibration of the model has been limited to application of results from controlled lysimeter studies and experiments involving bare (nonvegetated) soils. Therefore, the model has not been uniformly calibrated to all conditions that exist across the Hanford Site.

2.3.3.3 Unsaturated Flow and Transport Model. Modeling of groundwater flow in the partially saturated sediments began in the mid 1960's. Over the years, a number of computer codes were developed and applied at the Hanford Site. The primary motivation stemmed from interest in studying single-shell tank releases, and the potential migration of contaminants through the thick zone of partially saturated sediments beneath the 200 Areas. The vadose zone is between 60 and 80 meters thick in these areas. To support the HDW-EIS, a simplified methodology for vadose zone flow simulation was described that relied on the assumption of unit hydraulic gradient conditions and application of the steady-state solution to the Richards' equation (DOE 1987). Recently, vadose zone analyses have been supported through the use of more sophisticated models, including PORFLO-3 (Sager and Runchal 1990), VAM2DH (Huyakorn et al. 1988), VAM3DCG (Huyakorn and Panday 1990) and TRACR3D (Travis 1984). PORFLO-3 has been used on a number of projects, including: (1) modeling the flow of liquid effluent from the 1324 and 1325 cribs in the 100-N Area to the Columbia River, (2) simulation of groundwater flow in operable unit 300-FF-5, (3) analysis of the T-106 single-shell tank release, and (4) preliminary analyses of liquid-effluent sites requested by EPA and the Washington State Department of Ecology (Ecology). VAM2DH has been used in support of solid waste disposal facility siting, and the purge water discharge analysis. TRACR3D has been used for unsaturated zone analysis in support of the grout facility. The actual transport modeling for this application was performed using S301 (Wikramaratna and Farmer 1987), a transport code that is designed for advective dominated transport applications; the code uses the velocity vectors from TRACR3D.

2.3.3.4 Saturated Zone Flow and Transport Model. Modeling of flow in the saturated sediments beneath the Hanford Site was initiated in the mid-1960's. During the late 1960's and 1970's the Hanford Site standard was represented by Variable Thickness Transient (VTT), a two-dimensional finite-difference groundwater flow computer code (Reisenauer 1979). Transport codes that used velocity vector output from VTT have also been developed. The TRANSS code (Simmons et al. 1986) has been applied to assess the potential transport of contaminants at various waste sites over the years on the Hanford Site. Results obtained using the VTT/TRANSS model were used in support of assessing the health risks associated with various Hanford Site defense waste scenarios evaluated in the HDW-EIS (DOE 1987).

During the early 1980's, the CFEST (Gupta et al. 1982) computer code was developed for use at the Hanford Site. For detailed combined flow and transport analyses, the CFEST computer code has replaced the VTT/TRANSS computer code. More recently, the MODFLO (USGS 1988),

SLAEM (Strack 1989), and GGWP (GAI 1987) computer codes have been used to support various applications at the Hanford Site.

2.3.3.5 Multi-Phase Modeling. Development of multi-phase fluid flow and transport models was pioneered in the petroleum industry. Cases involving the disposal of volatile organic compounds that could migrate as separate fluid phases to the subsurface environment exist at locations on the Hanford Site. Experience in characterizing and modeling the fate and transport of these substances at the Hanford Site is limited. During fiscal year 1991, an investigation of a disposal site in the 200 Areas where large quantities of carbon tetrachloride have been disposed was initiated. Although the primary focus of this activity is to develop and test alternative methods for the purpose of characterizing and recovering large quantities of the carbon tetrachloride under the direction of an "expedited response action," an effort to apply existing computer codes and models to assist this effort was included in the scope of work. To date, emphasis has been placed on the use of PORFLO-3 to assist in this effort. Results from these preliminary analyses are not available. In addition to PORFLO-3, TRACR3D allows simulation of some aspects of multiphase flow. However, the use of TRACR3D in support of multi-phase modeling activities at the Hanford Site in unknown.

2.3.3.6 Geochemistry Modeling. Because of the importance of understanding and interpreting the geochemistry of natural waters, a number of chemical equilibrium computer codes have been developed in the last 20 years. Although these programs were originally research tools, they have become widely available and are commonly applied to a variety of hydrogeological problems. Even more so than the hydrogeological codes presented in this document, however, geochemical equilibria codes require the user to be quite knowledgeable. The user not only must be familiar with the specific details of these complex computer codes, but should also have a thorough understanding of the chemical processes that are being represented and the quality of the input data available.

Although a variety of geochemical codes have been used at the Hanford Site, only a small number have become mainstays for practical applications. Hanford Site experience with geochemical codes is related to their application in a wide variety of programs involving radioactive waste and hazardous chemicals. Some of the most common and widely accepted codes in use include: PHREEQE (Parkhurst et al. 1980); MINTEQ (Brown and Allison, 1987); EQ3, EQ6 (Wolery et al. 1990); and WATEQ (Ball et al. 1987). These codes have been modified to various extents over the last 10 years and a number of program versions are in existence. WATEQ has basic speciation of aqueous solutes capability, whereas MINTEQ, EQ3/EQ6, and PHREEQE have speciation and geochemical reaction sequencing capabilities. All of these computer codes are available for use in support of the RI/FS process.

3.0 SELECTION CRITERIA

3.1 ADMINISTRATIVE CRITERIA

Administrative criteria include availability, user support, useability, portability, modifiable, and reliability. It is believed that all the codes considered in this report satisfy the administrative criteria to some extent. Fulfillment of these administrative criteria will require support throughout the lifetime of the projects that rely on the selected computer codes. An explanation of each of these criteria is provided in the following sections.

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3.1.1 Availability

Computer codes will be made available to all users for confirming modeling results. Generally speaking, public domain codes will be favored over proprietary software. However, the modeling committee believes that some proprietary software provided enhanced technical capability not available in existing nonproprietary software. The proprietary software included in the Hanford Site list of computer codes are required to have a licensing agreement that includes a mechanism for providing access to outside users wishing to examine the source code or run the executable code to confirm the Hanford Site modeling results. It is recognized that an excessive financial burden for such a licensing agreement could disqualify use of a code.

3.1.2 User Support

The primary criteria for selection will be that sufficient technical support will be available throughout the lifetime of the project from the software developer or distributor.

3.1.3 Useability

Useability refers to factors such as the ease of grid definition, parameter input, calibration, graphical capabilities, and the effectiveness of output presentation. Code documentation must be readily available and computer codes should be generally "user friendly." Computer codes currently in use at the Hanford Site have an established user community and are preferred over computer codes that are unfamiliar to Hanford Site users.

3.1.4 Portability

The software should operate on a variety of different hardware systems. A personal computer (PC) version is particularly desirable because PCs are more accessible.

3.1.5 Modifiable

Software modifications will likely be required to expand capabilities and allow inclusion of technological improvements. All modifications will be documented and controlled under computing software quality assurance guidelines.

3.1.6 Reliability

Quality assurance guidelines will include a testing program to verify all computer codes and validate models. Additionally, computer codes should have a history of effective use, with emphasis on Hanford Site usage.

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3.2 TECHNICAL CRITERIA FOR CONCEPTUAL MODELS OF THE HANFORD SITE

This section describes the physical features and processes that are currently considered part of the Hanford Site conceptual transport models for air, surface water, and groundwater for conducting risk assessments. These conditions help define the technical capabilities considered in Section 3.3 and ultimately determine the criteria for the recommendations provided in Section 4.0.

3.2.1 Air Transport

During Hanford Site remediation, it is anticipated that various contaminants will become airborne. These contaminants could be released (continuous or instantaneous) in either gaseous or particulate form, or both. As contaminants are transported downwind, the concentration will be modified by three-dimensional dispersion, radioactive decay and chemical transformation, and gravitational deposition. The governing parameters describing these processes tend to be location and weather dependent; therefore, computer codes and models that have been demonstrated under Hanford Site conditions are considered most desirable. Based on current understanding, contaminant fate and transport analyses required to support risk assessments at the Hanford Site will likely be limited to individual sources located at ground level or a specified elevation. Multiple sources could be quantified using superposition. Additional requirements are likely if air transport becomes a major issue and more detailed analyses are required.

3.2.2 Surface Water Flow and Transport

Contamination could enter the Columbia River through diffusion, groundwater, influx, or direct discharge from seeps and springs. In either case, potential contamination is considered to be more of a localized problem than a regional problem due to the massive dilution capacity of the Columbia River. The average flow rate of the Columbia River in the Hanford Reach is approximately 3,000 cubic meters per second (DOE 1987), compared with an estimate of influx to the river over the entire Hanford Reach of approximately 1 cubic meter per second. This rate of influx is less than 0.04% of the Columbia River average flow rate.

Two specific needs for surface water flow and transport modeling have been identified, including: (1) prediction of river stage variation and its effect on contaminant migration near the Columbia River, and (2) downstream mixing of contaminants discharging from groundwater, springs, and seeps into the Columbia River. Modeling river stage variation will require quantification of the transient hydraulic behavior of the Columbia River in response to natural and man-made changes to the flow rate. Important factors may include the hydraulic profile of the river, bank storage, groundwater interactions, stream bed configuration, etc. Mixing of contaminants from groundwater, springs, and seeps discharging into the Columbia River will likely require modeling of point and distributed sources, advection, turbulent mixing (combining mass and momentum), and chemical partitioning between water and sediments. Additional factors may become important if surface water transport becomes a major issue and more detailed analyses are required.

3.2.3 Groundwater Transport

The transport of contaminants through the soil and groundwater of the Hanford Site sediments will require consideration of: (1) infiltration processes, (2) groundwater flow and transport of

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contaminants under partially saturated (vadose zone) conditions, and (3) groundwater flow and contaminant transport under saturated conditions.

3.2.3.1 Infiltration. Most of the waste at the Hanford Site is, and will be, contained in the vadose zone. Infiltration of water through these partially saturated sediments is considered the primary mechanism for release of waste to the accessible environment. As such, considerable emphasis has been placed on the study and quantification of the infiltration rate (i.e., the flux of water past the root zone). The physical processes that effect the infiltration rate include; precipitation, evaporation, transpiration, and drainage. Modeling infiltration at the Hanford Site requires the capability to simulate the following characteristics:

- Semiarid climate with average annual precipitation of 0.16 meters
- Temperatures in excess of 40°C, and extended periods of freezing temperatures
- Snow cover and snowmelt
- Evapotranspiration with little or no vegetation and variable rooting depths
- Layered soils with lithologies ranging from sand and gravel to sandy loam
- Simulation of groundwater flow under variably saturated conditions
- Soil heterogeneity; i.e., variations in hydraulic conductivity, storativity, effective and total porosity
- Soil heating and cooling.

3.2.3.2 Vadose Zone Flow and Transport. Once the water drains below the root zone, it is redistributed in the subsurface sediments. Drainage of water through these sediments is estimated to range from zero to 10 centimeters per year under natural conditions at the Hanford Site. Characteristics considered important for modeling vadose zone flow and transport at the Hanford Site are listed below:

- Moisture-dependent hydraulic conductivity relationships (characteristic curves) that differ for different soil types
- Hysteresis (characteristic curves that are dependent on the recent wetting and drying history of the soil)
- Vadose zone thickness ranging from several meters near the Columbia River to more than 20 meters beneath the 200 Area plateau
- Layered soils, including relatively impermeable caliche layers that may cause lateral spreading or perched water table conditions
- Discontinuous stratigraphic layers that are tilting in places
- First-order, linear sorption/desorption processes, using an effective distribution or retardation coefficient

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- Radioactive decay.

Additional capabilities that may become important include: (1) heat transport, and (2) contaminant volatilization and vapor transport. In summary, a multi-dimensional, transient, partially saturated flow and transport modeling capability is required to simulate the behavior of contaminants in the vadose zone.

3.2.3.3 Saturated Flow and Transport. Contaminants transported through the vadose zone will become mixed with the groundwater in the unconfined aquifer. These contaminants will move with groundwater and could eventually reach downgradient pumping wells or the Columbia River. The flow velocity through the saturated sediments at the Hanford Site is estimated to range from several centimeters to several meters per day. Simulation of saturated groundwater flow and transport will be required to predict contaminant concentrations for use in support of risk assessments. Based on current understanding, saturated flow and transport modeling of Hanford Site conditions should account for the following conditions:

- Heterogeneous and isotropic porous media aquifer properties
- Layered soils with tilting beds in places
- Transient flow and transport behavior
- Confined and unconfined conditions
- Up to 70-feet variations in water table elevations with time, due to changes in waste-disposal practices at the Hanford Site, and future irrigation scenarios on or adjacent to the site
- Contaminant advection and dispersion
- Radiological and biological decay
- Contaminant retardation using an equilibrium sorption model with linear and completely reversible isotherms
- Point or distributed sources
- Aquifer/river interactions.

3.3 COMPARISON OF TECHNICAL CAPABILITIES

This section presents a matrix (Table 3-1) showing the modeling capabilities for each of the groundwater flow and transport computer codes considered for inclusion in the list of Hanford Site software. Only computer codes previously used at the Hanford Site were included in the matrix. The purpose of this matrix is to facilitate side-by-side comparison of the candidate software. Although air and surface water transport software were discussed in this report, none of these computer codes were eliminated from the list of Hanford Site software. Consequently, a matrix comparison of air and surface water transport software is not provided.

Table 3-1. Modeling Capabilities for Groundwater Flow and Transport Software.

Characteristic	UNSATH	PORFLO3	VAM2D	VAM3D	TRACR3D	S301	VTT	TRANSS	CFEST	GGWP	MODFLO	SLAEM
Confined Aquifer		X	X	X	X		X		X	X	X	X
Phreatic Aquifer[1]		X	X	X	X				X	X	X	
Heterogeneous Aquifer	X	X	X	X	X		X	X	X	X	X	X
Anisotropic K		X	X	X	X				X	X	X	
Single Fluid	X	X	X	X	X		X	X	X	X	X	X
Multiple Fluids		X			X							
Saturated		X	X	X	X		X	X	X	X	X	X
Unsaturated	X	X	X	X	X							
Transient Flow		X	X	X	X		X		X	X	X	X
Hysteresis			X									
Aquifer/River Interactions[2]											X	
Non-Orthogonal Grid				X					X	X		X
2-Dimensional (Horizontal)		X	X		X		X		X	X	X	X
2-Dimensional (Vertical)		X	X	X	X				X	X	X	
Pseudo 3-Dimensional[3]							X			X	X	X
3-Dimensional		X		X	X	X			X			
Advection/Dispersion[4]		X	X	X	X	X		X	X	X		
Linear Sorption		X	X	X	X	X		X	X	X		
Non-Linear Sorption												
Radioactive Decay		X	X	X	X	X		X	X	X		
Heat Transport	X	X							X			
Evapotranspiration	X		X	X								
Hanford Vegetation	X											

[1]Phreatic aquifer capability signifies that the aquifer transmissivity automatically adjusts to accommodate changes in the saturated thickness of the aquifer.

[2]Aquifer/river interactions refers to linking of a groundwater flow model and a streamflow routing model to simulate the interaction between groundwater and a dynamic river stage.

[3]Pseudo 3-dimensional refers to the capability to model layered aquifer systems without solving the fully three-dimensional system of equations, thereby reducing computational effort.

[4]TRANSS is a streamtube travel time model that does not actually solve the advection/dispersion equation.

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4.0 RECOMMENDATIONS

Specific recommendations for computer codes included in the list of Hanford Site software are primarily dependent on the capability of the chosen software to simulate the majority of processes governing contaminant transport at the Hanford Site. A description of the most relevant processes, and comparison of the candidate groundwater codes to simulate these processes, were provided in the previous sections. The recommended codes and the rationale for their selection are provided in the following sections.

4.1 AIR

As stated in Section 2.4.1, it is recommended that simplified conservative analytical models be used whenever possible. These models have been encoded into several existing radiological safety codes used routinely at the Hanford Site (e.g., GENII [Napier et al. 1988]). With regard to GENII, two atmospheric transport models have been encoded; a straight line Gaussian model is used to compute acute maximum exposure based on an assumed maximum plume "centerline" concentration, and a chronic exposure model that assumes a sector-averaged concentration. The chronic exposure model also employs the use of the straight line Gaussian model for computing plume centerline concentrations. The source can either be released at ground level or at some elevation. Hanford Site meteorological conditions are programmed into GENII. Application of AIRDOS-PC will account for decreases in contaminant concentration resulting from deposition and scavenging. Similarly, if additional detail is required in the modeling of hazardous chemicals, it is recommended that the ISC computer code be applied.

4.2 SURFACE WATER

It is recommended that simplified conservative analytical models be used whenever possible. Guidance on the use of a one-dimension completely mixed model assuming the existence of a mixing zone is provided in EPA's Superfund Exposure Assessment Manual (EPA 1988). A solution to a quasi-two dimensional advective-dispersion model appropriate for estimating the decrease in contamination concentration resulting from lateral and longitudinal mixing is contained in GENII. Application of this analytical model should be more accurate but less conservative than application of the analysis methodology outlined in the EPA guidance document. If river stage variations resulting from hydropeaking and annual flooding is required, it is recommended that the DWOPER computer code, or equivalent, be applied. The DWOPER computer code has been applied by both the U.S. Army Corps of Engineers and the U.S. Bureau of Reclamation to simulate river stage variation in the Hanford Reach of the Columbia River (Fread 1973). Since the DWOPER computer code does not address sediment and sediment-related contaminant transport, if detailed analyses of these parameters are required to support future risk assessments based on information contained herein, it is recommended that the SERATRA (Onishi 1977) computer code or equivalent be applied. However, since the SERATRA computer code has not been used for several years, a review of currently available and maintained computer codes should be conducted prior to updating and implementing the SERATRA computer code.

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4.3 GROUNDWATER FLOW AND TRANSPORT CODES

The recommended codes for the subsurface pathway include one infiltration code (UNSAT-H), two unsaturated zone codes (PORFLO-3 and VAM3D), and one saturated zone code (CFEST). The capabilities of these computer codes are outlined in the following sections, followed by a discussion of the rationale for choosing this set of codes.

4.3.1 UNSAT-H

UNSAT-H has been developed at the Hanford Site and is designed to simulate infiltration under typical Hanford Site conditions (Fayer and Jones 1990). UNSAT-H is a one-dimensional finite-difference code that accounts for precipitation, drainage, redistribution, evaporation, soil heating, and plant uptake of water. UNSAT-H allows specification of site-specific vegetation and soil conditions, and includes four different relationships between hydraulic conductivity and moisture content. The computer code will be used to establish moisture flux for the upper boundary condition in vadose zone flow and transport models. The code was selected because it best represents the current understanding on the quantification of those processes that govern infiltration at the Hanford Site.

4.3.2 PORFLO-3

PORFLO-3 (Sagar and Runchal 1990) is a fully three-dimensional, integrated finite-difference, flow and solute transport code with a wide variety of capabilities, including coupled unsaturated/saturated analysis, retardation, radioactive decay, and conductive heat transport. The geologic media may be heterogeneous and anisotropic and may contain linear and planar features such as boreholes and fractures. The computer code includes four different numerical solution techniques, each having certain advantages under differing conditions. Three options are available for specifying the relationship between hydraulic conductivity and moisture content. The computer code does not allow for hysteresis and is limited to grids with orthogonal geometry. PORFLO-3 has been applied at the Hanford Site, in addition to being developed (partially) at the Hanford Site. Although PORFLO-3 is proprietary, all use of the code in support of Hanford Site work is specifically excluded from the copyright limitation. Stochastic and multi-phase versions of PORFLO-3 are available.

PORFLO-3 was included in the list of Hanford Site software to simulate near-field unsaturated and saturated flow and transport in three dimensions. Although similar capabilities are available with VAM3D, PORFLO-3 was selected as the primary unsaturated flow and transport model for the following reasons:

- Westinghouse Hanford supported the development of this computer code for several years with the specific intention of using this computer code in support of environmental restoration activities
- The computer code has undergone extensive testing and peer review
- The code has been tailored to address the specific needs of the Hanford Site
- Hanford Site personnel have considerable experience in using this computer code

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4.3.3 VAM3D

VAM3D (Huyakorn and Panday 1990) is a finite-element flow and solute transport code capable of coupled unsaturated/saturated analysis. Many of the features included in PORFLO-3 are included in VAM3D, although the VAM3D computer code cannot model heat flow. The code includes a routine for simulation of surface infiltration similar to UNSAT-H, although the infiltration routine is not specifically designed for the arid conditions found at the Hanford Site. Also, the aspect of hysteresis available in VAM2D can be easily incorporated into VAM3D. The code is proprietary; a licensing agreement will be modified to satisfy availability criteria.

VAM3D was included in the list of Hanford Site software because it utilizes a finite-element approach that will facilitate simulation of tilting and discontinuous bedding in the unsaturated zone. The computer code will also serve as a benchmark computer code for evaluating results obtained through the use of PORFLO-3. These intercode comparisons are considered extremely important during future testing of these computer codes.

In addition, the VAM3D computer code will be considered for area-wide saturated flow and transport analyses. As such, the VAM3D computer code has several capabilities that can be used in supporting environmental restoration activities.

4.3.4 CFEST

CFEST (Gupta et al. 1982) is a fully three-dimensional, finite-element, saturated flow and transport code developed at Pacific Northwest Laboratory. Capabilities include retardation and radioactive decay. The code was developed at Hanford for the Office of Nuclear Waste Isolation and the Seasonal Thermal Energy programs. As such, it was developed for confined aquifer systems and does not readily allow for changes in water table variation or changes in transmissivity resulting from water table variation. Recent proprietary versions have extended the capabilities of the CFEST computer code.

As stated previously, in the mid-1980's CFEST replaced the VTT/TRANSS computer code for performing detailed large-scale flow and transport analyses in the saturated sediments on the Hanford Site. As such, an operating version of the CFEST computer code is available for Hanford Site-wide application. However, this model has not been updated or applied fully at the Hanford Site for several years; therefore, user application of CFEST is unknown at this time. Based on discussions to date, it appears that the features of most importance, e.g., improved solver, adjustment of the water table, and transmissivity coefficients, have been developed and incorporated into the proprietary versions of the computer code. A licensing agreement does not exist for these versions in support of Hanford Site risk assessments.

It is recommended that CFEST be evaluated to support Hanford Site-wide risk assessments. This recommendation is based on the following considerations:

- An operational model of the Hanford Site saturated sediments that employs the use of CFEST exists
- The intrinsic value of using an integrated finite-element flow and transport computer code to model potential contaminant movement in the saturated sediments.

4.3.5 Summary

The four computer codes that have been selected for assessing contaminant fate and transport in the subsurface pathway (UNSAT-H, PORFLO-3, VAM3D, and CFEST) provide a broad base of analytical capability. Although some questions and concerns remain, it is believed that these computer codes, when appropriately implemented, will satisfy the administrative and technical criteria discussed in Section 3.0. In addition, this set of computer codes provides a level of technical redundancy considered prudent at this time based on current uncertainty. Although other codes are available for use in support of risk assessment, the selection reflects a "bias for action." As such, the computer codes that are currently is use in support of the Hanford Site remediation had a definite advantage.

4.4 SOFTWARE VERIFICATION AND BENCHMARKING

It is recommended that the computer codes in the list of Hanford Site software be verified and benchmarked against each other (when possible). Verification would involve inspection of the analytical formulation to confirm proper performance of the mathematical calculations and comparison of results against analytical solutions. In addition, the computer codes that have similar capabilities should be benchmarked under conditions typical of the Hanford Site to allow direct comparison. This process would develop an understanding of which codes are most appropriate for certain conditions and perhaps result in elimination of codes from the list of Hanford Site software. It is recommended that verification and benchmarking be initiated as soon as possible. If available, the results will be included in the second milestone (M-29-02).

4.5 FUTURE SOFTWARE APPROVAL PROCESS

As stated previously, the development of computer codes for modeling environmental pathways will continue to evolve. In the event that new or revised software is found to offer significant advantages, then its use in support of Hanford Site remediation activities will be considered. Two prerequisites for consideration of new software will be: (1) evidence of peer review and general acceptance by the technical community, and (2) recognition of the need for the additional software by the Hanford Site technical and regulatory communities. It is recommended that the software approval process will proceed in a manner similar to the selection of the computer codes contained herein, in that it would involve a committee of technical experts representing Ecology, EPA, and DOE. Since significant expansion of the list of Hanford Site codes is undesirable, additions of new codes to the list may require deletion of old codes. Alternatively, computer codes not included on the Hanford Site list may be approved for limited use in specialized applications.

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