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Hanford Site Risk Assessment Methodology

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LIST OF ACRONYMS

ALARA	as low as reasonably achievable
ANOVA	analysis of variance
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
BHI	Bechtel Hanford, Inc.
CEDE	committed effective dose equivalent
CERCLA	*Comprehensive Environmental Response, Compensation, and Liability Act of 1980*
CFR	*Code of Federal Regulations*
CMS	Corrective Measures Study
COC	contaminant of concern
COPC	contaminants of potential concern
DNR	Washington State Department of Natural Resources
DOE	U.S. Department of Energy
DOE-RL	U.S. Department of Energy, Richland Operations Office
DRF	dose rate conversion factors
EC_{50}	effective concentration at which 50% of experimental organisms exhibit a nonlethal effect
ED	effective dose
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
EHI	ecological hazard index
EHQ	environmental hazard quotient
EPA	U.S. Environmental Protection Agency
ERA	expedited response action
ESA	*Endangered Species Act*
FI	facility investigation
FR	*Federal Register*
FS	feasibility study
HEAST	Health Effects Assessment Summary Tables
HEDOP	Hanford Environmental Dose Overview Panel
HEIS	Hanford Environmental Information System
HHE	Human Health Evaluation
HMS	Hanford Meteorological Station
HWMA	Hazardous Waste Management Act
HRS	Hazard Ranking System
HSRAM	Hanford Site Risk Assessment Methodology
HFSUWG	Hanford Future Site Uses Working Group
HPPS	*Hanford Past-Practice Strategy*
HQ	hazard quotient
ICR	lifetime incremental cancer risk
IRIS	Integrated Risk Information System
IRM	interim remedial measure

LIST OF ACRONYMS (Cont.)

LC_{50}	lethal concentration at which 50% of experimental organisms die
LD_{50}	lethal dose at which 50% of experimental organisms die
LFI	limited field investigation
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
NEPA	*National Environmental Policy Act of 1969*
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
NPS	National Park Service
NRC	National Research Council
NRDA	natural resources damage assessment
ORP	Office of Radiation Programs
PNL	Pacific Northwest Laboratory
QA/QC	quality assurance/quality control
QRA	Qualitative Risk Assessment
QRAM	Qualitative Risk Assessment Methodology
RAC	Inter-Agency Working Group for Risk Assessment
RAGS	*Risk Assessment Guidance for Superfund Volume I*
RBC	risk-based concentration
RCRA	*Resource Conservation and Recovery Act of 1976*
RCW	*Revised Code of Washington*
RERA	Risk Evaluation of Remedial Alternatives
RfC	reference concentration
RfD	reference dose
RFI	RCRA facility investigation
RI	remedial investigation
RME	reasonable maximum exposure
ROD	Record of Decision
SF	slope factor
SQL	sample quantitation limit
TAL	target analyte list
TCL	target compound list
TDL	lowest observed toxic dose
TIC	tentatively identified compound
UCL	upper confidence limit
USACE	U.S. Army Corps of Engineers
UTL	upper tolerance limit
WAC	*Washington Administrative Code*
WDOW	Washington State Department of Wildlife
WHC	Westinghouse Hanford Company
VOC	volatile organic compound

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1.0 INTRODUCTION

This chapter provides introductory information on the Hanford Site Risk Assessment Methodology (HSRAM). The purpose and objectives of the methodology are presented in Section 1.1. The organization of the methodology is described in Section 1.2, while Section 1.3 discusses relevant statutes, regulations, and guidelines. Section 1.4 discusses the role and current applications of risk assessment in the activities of the Hanford Site Environmental Restoration Program and the *Hanford Past-Practice Strategy* [(HPPS) DOE-RL 1992a]. Finally, the methodology approach to the human health evaluation and ecological evaluation is described in Section 1.5.

1.1 PURPOSE OF THE METHODOLOGY

This methodology has been developed to prepare human health and ecological evaluations of risk as part of the *Comprehensive Environmental Response, Compensation, and Liability Act of 1980* (CERCLA) remedial investigations (RI) and the *Resource Conservation and Recovery Act of 1976* (RCRA) facility investigations (FI) performed at the Hanford Site pursuant to the *Hanford Federal Facility Agreement and Consent Order* (Ecology et al. 1994), referred to as the Tri-Party Agreement. 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 ecological 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.

Technical representatives from the U.S. Department of Energy, Richland Operations Office (DOE-RL), the Washington State Department of Ecology (Ecology), the U.S. Environmental Protection Agency Region 10 (EPA-10), and their respective contractors participated in an Inter-Agency Working Group for Risk Assessment (Risk Assessment Committee or RAC) to provide input into the development of the Hanford Site Risk Assessment Methodology.

1.2 METHODOLOGY ORGANIZATION

Section 2.0 describes the process of selecting and evaluating data as it relates to developing a list of contaminants to be evaluated in the risk assessment. The information presented in Section 2.0 is applicable to both human health and ecological evaluations. Section 3.0 describes the methodology for the human health evaluation and Section 4.0 describes the methodology for the ecological evaluation. The application of this methodology to qualitative risk assessments (QRA) is discussed in Section 5.0. References are provided in Section 6.0.

Appendices provide supporting information. Appendix A provides an overview of the human receptor exposure scenarios and exposure parameters for those scenarios. A Hanford Site modeling standard identifying computer codes for use in support of risk assessment at the Hanford Site is presented in Appendix B. Ecological data specific to the Hanford Site are presented in Appendix C. Appendices D and E provide example calculations for the human health and ecological evaluations, respectively.

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 (ARAR) for these activities include the Model Toxics Control Act [MTCA ,Ch. 70.105D *Revised Code of Washington* (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 *Code of Federal Regulations* (CFR) Section 300] and the "Model Toxics Control Act Cleanup Regulations" [MTCACR (Ch. 173-340 *Washington Administrative Code* (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 remedial investigation (RI) and RCRA FI activities. The primary sources of general guidance on the risk assessment process utilized in preparation of the methodology include the following:

- *Wildlife Exposure Factors Handbook* (EPA 1993)
- *Risk Assessment Guidance for Superfund, Volume I: Human Health Evaluation Manual* (RAGS) (Part A, 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)
- *RCRA Facility Investigations (RFI) Guidance* (EPA 1989c)
- *Final Guidance for Data Useability in Risk Assessment (Part A)* (EPA 1992a)
- *Framework for Ecological Risk Assessment* (EPA 1992b).

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. This revision does not incorporate the EPA final exposure assessment guidelines published in May 1992 [57 *Federal Register*

(FR) 22888-22938]. The final exposure assessment guidelines describe an analysis that can only be accomplished with the use of stochastic techniques (i.e., Monte Carlo sampling of distributions of input parameters). There has been no decision by EPA Region 10 and Ecology regarding the use of stochastic techniques in risk assessment.

The efforts of the Hanford Future Site Uses Working Group (HFSUWG) are acknowledged for potential impacts on evaluating risk at the Hanford Site. This group, comprised of representatives from federal, tribal, state, and local governments, and representatives from several constituencies with interests in the Hanford Site, prepared a document summarizing a range of possible uses for each major geographic area of the Hanford Site (HFSUWG 1992). The group cautions that this is not a land use report per se and that they did not intend to specify and delineate the exact future uses that would occur throughout the Hanford Site. The document, rather, presents conceptual, general future use options (e.g., agriculture, industry, wildlife) and levels of access that may be needed to permit these uses (e.g., unrestricted, restricted, exclusive, or buffer). A guiding principle provided by the group is that decisions made in the course of cleanup should result in decreased risk to public health and net benefits to the environment with both cleanup decisions and future development decisions guided by the principle "do no harm." Although the scenarios developed in HSRAM are analogous to the conceptual use options suggested by HFSUWG, the HSRAM scenarios permit quantification of exposures, while the HFSUWG scenarios are only conceptual. For this reason, it is difficult to provide an exact comparison of the individual scenarios of the two reports. The HSRAM will help identify if there are human health or environmental impacts, and is one of the tools used to help determine the need for cleanup with respect to various potential land uses.

The risk assessment methodology report is a living document. Because the guidance for both human health and ecological risk assessment is evolving, new guidance should be periodically reviewed for its relevance to this document and included, 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 RISK ASSESSMENT APPLICATIONS AT THE HANFORD SITE

Much of the available risk assessment guidance is appropriate for baseline risk assessments. Therefore, the methodology described in Sections 2.0 through 4.0 is generally applicable to baseline risk assessments. Other applications of the methodology are also appropriate. The potential applications of the methodology, as part of the HPPS (DOE-RL 1992a), are briefly presented in this section.

A separate document, *Risk Evaluation of Remedial Alternatives for the Hanford Site* (RERA) (DOE-RL 1994a) provides methodology for assessing human health and ecological risk associated with remediation activities during and after implementation. RERA addresses issues related to contaminant and noncontaminant stressors (e.g., transportation and construction accidents, heat stress, and habitat destruction), and short-term as well as long-term risk. In contrast, HSRAM focuses on long-term risk associated with environmental contaminants.

Other concerns exist for cultural resources on the Hanford Site. Radioactive and hazardous waste contamination may also pose risk to Hanford Site cultural resources and their associated spiritual values. It is necessary to characterize risk to human burial grounds and other archaeological sites, to the extent possible, so sensitive areas can be appropriately identified and addressed or protected. To this end, RERA will describe methodology for assessing risk to cultural resources so appropriate actions may be taken when planning characterization, construction, or remediation activities.

1.4.1 Baseline Risk Assessments

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 Section 300.430(d)(4)]. As indicated in the Preamble to the NCP (55 FR 46, p.8709), this assessment provides a basis for determining whether or not remedial action is necessary and 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" (55 FR 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 Section 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 Section 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 three methods for establishing cleanup levels at a site, including a risk assessment framework that can be utilized for this purpose (WAC 173-340-708). Cleanup levels are either tabulated (Method A) or calculated (Methods B and C) using target hazard quotients and cancer risk. Method B cleanup levels for individual substances are based on a carcinogenic risk less than or equal to 1E-06 and a hazard index less than or equal to one. 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 cumulative 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 and Method C.

The methodology presented in this document has been developed to provide estimates of exposure and risk 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 3.1 is used to identify contaminants at a conservative level so that focused risk 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.

It should be noted that the radiation exposure level associated with a lifetime cancer risk of 1E-04 or less is sufficiently protective that current radiation protection standards pertinent to the Hanford Site will be met (e.g., DOE Order 5400.5 [DOE 1990]). However, radiation protection standards have two components: a radiation dose limit to individuals; and an "as low as reasonably achievable" (ALARA) principle. According to the National Council on Radiation Protection and Measurements (NCRP), the lower limit for application of the ALARA process is a lifetime fatal health risk of 1E-05 due to radiation exposure (NCRP 1993).

The HSRAM has been developed in part to support the HPPS. The four National Priorities List (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 HSRAM 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.

1.4.2 Qualitative Risk Assessment to Support the Interim Remedial Measure Path

The signatories of the Tri-Party Agreement (EPA, Ecology, and DOE) developed a new strategy to manage and implement past-practice investigations. The *Hanford Past-Practice Strategy* (DOE-RL 1992a) was developed to enhance the efficiency of ongoing CERCLA RI/FS and RCRA facility investigation/corrective measures study (RFI/CMS) activities in the 100 Area of the Hanford Site. The objective of the HPPS is to expedite the ultimate goal of cleanup by initiating and completing waste site cleanup through interim cleanup actions.

The HPPS focuses on reaching early decisions to initiate and complete cleanup projects by maximizing the use of existing data that are consistent with the data quality objectives, together with short-time-frame investigations, where necessary. As more data become available on contaminant conditions and associated risk, the details for longer-term investigations and studies are better defined. The effective use of existing data along with better management of uncertainty should reduce the number of sampling episodes and expedite treatability studies, feasibility studies, and cleanup actions, including expedited response actions (ERA) and interim remedial measures (IRM).

The near-term strategy for decision-making in the HPPS and mitigating contamination problems at specific waste sites provides for three different pathways.

- The ERA pathway is used for abatement if conditions exist or are suspected that create an unacceptable current or future health or environmental risk and necessitate a rapid response to mitigate the problem.

- The IRM pathway without a limited field investigation (LFI) is appropriate if existing data are judged sufficient to develop a conceptual site model and perform a qualitative risk assessment. If necessary, a focused feasibility study (FS) will be conducted to select the IRM remedy.

- The IRM pathway with an LFI is used to identify and gather the minimum additional data needed to formulate a conceptual site model and perform a QRA that would support an IRM or other decisions. The LFI is limited in scope and generally is not intended to support a final record of decision. Regardless of scope, however, the LFI is part of the RI/FS (or RFI/CMS) process and not a substitute for it.

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 IRM path calls for a qualitative risk assessment. A qualitative risk assessment is defined in the HPPS as "a judgement not based solely on quantification, agreed to by the parties, based on available site data regarding the threat posed by site contamination" (DOE-RL 1992a). Thus, the QRA will provide the characterization of site risk that Tri-Party Agreement representatives will evaluate to determine whether an IRM is appropriate. Qualitative risk assessments were not intended to replace the need for a baseline risk assessment nor to serve as a basis for establishing preliminary remediation goals.

Such uses of the HSRAM represent, to a degree, unique applications of risk assessment techniques. The HPPS 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).

Although interim actions (ERAs and IRMs) may be used to mitigate specific contamination problems, the process of final remedy selection must be completed for the operable unit and the NPL site to reach closure. The information obtained from the LFIs and interim actions may be sufficient to perform a risk assessment and select a remedy for the operable unit. If the data are not sufficient, additional investigations and studies can be performed to the extent necessary to support the operable unit remedy selection. These investigations would be performed within the framework and process defined for RI/FS programs.

1.4.3 Other Risk Assessments

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.

It should be noted that natural resource trustees (e.g., the U.S. Fish and Wildlife Service and DOE) have the responsibility and authority under CERCLA to identify the need for and, as necessary, to conduct natural resources damage assessments (NRDA) at NPL sites. Therefore, DOE-RL, with the assistance of EPA-10 and Ecology, are soliciting the involvement of federal, state, and tribal natural resource trustees in the overall Hanford Site Environmental Restoration Program.

The appropriate baseline ecological evaluations should 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 obtained more efficiently and cost-effectively through the baseline ecological evaluation process. In fact, much of the information needed to conduct a NRDA preassessment screening may be obtained from various Hanford Site documents, including the extensive environmental monitoring studies, workplans and other reports encompassing baseline ecological evaluations.

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 Section 1.5.1. The ecological evaluation approach is described in Section 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 potential ARAR for NPL sites in Washington State. 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 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. Alternate exposure scenarios and cleanup levels for an appropriate site-specific set of exposure pathways may also be permitted by MTCACR if appropriate [WAC 173-340-708(3)(c)&(d)].

The HHE methodology that is presented in Section 3.0 incorporates the requirements of MTCACR and CERCLA risk assessment guidance, including EPA Region 10 *Supplemental Risk Assessment Guidance* (EPA-10 1991), 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 Ecological Evaluation

Unlike the human health evaluation, the current MTCACR cleanup standard development process provides no specific procedures for the ecological 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 are consulted for procedural guidance.

The NCP states that at an NPL site [40 CFR Section 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 assessment will help establish acceptable exposure levels for use in developing remedial alternatives. . . ."

The overall goal of the ecological evaluation is therefore to characterize current and likely future nonhuman ecological risk attributable to releases of contaminants from the site. The NCP goes on to focus the scope of this evaluation by stating that [40 CFR Section 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 ecological evaluation methodology presented in Section 4.0. By satisfying the NCP requirements, the general requirement of RCRA and the state to protect the environment may also be met. In addition, the EPA has developed a *Framework for Ecological Risk Assessment* (EPA 1992b) that is generally adopted as the methodology for conducting ecological risk assessments under HSRAM. The EPA framework is broadly written to apply to the entire spectrum of potential ecological problems. However, the application of the ecological evaluation component of the HSRAM presented herein is much more focused. Per the NCP, the methodology addresses only the releases (and subsequent fate and transport) of site contaminants (including indirect effects) within the context of the general framework. Methodology for the evaluation of other ecological effects, such as habitat destruction, are provided in RERA (DOE-RL 1994a).

1.6 INTENDED USERS OF HSRAM

Because of the variety of programs in which risk assessments play a role at the Hanford Site, risk assessments should be produced under the supervision of persons familiar with the purpose of the risk assessments within these programs to ensure that they maintain an appropriate level of effort and detail. Risk assessment also requires the talents of multiple scientific disciplines (described below). In addition, the authors must be able to provide a descriptive picture of risk, and not just a number. Only then can the risk assessment be accurately used as a decision-making tool by risk managers.

1.6.1 Qualifications of Human Health Risk Assessors

A risk assessment team should be made up of persons who can accurately calculate risk values according to current guidance, as well as interpret these values with respect to their inherent limitations, biases, and uncertainties. Such a team requires persons familiar with federal, regional, state, and site-specific regulations and guidance to ensure compliance with the objectives of the risk assessment. While the exposure assessment portion of some risk assessments may be predetermined, a fully quantitative analysis may require the development and use of probability density functions to adequately characterize the activities of a receptor population. Presentation of qualitative toxicity information and application of numerical toxicity values should be performed under the supervision of a qualified toxicologist who can ensure that the health effects of concern have been adequately identified. Sites in which radionuclides are the contaminants of concern should be evaluated by persons (such as qualified health physicists) who understand the assumptions inherent in radiation risk assessment methods, as well as are familiar with dose assessment techniques and dose-based standards.

1.6.2 Qualifications for Ecological Risk Assessors

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

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. A knowledge of pertinent federal and state environmental statutes and their associated regulations and guidance documents is also essential. Access to such a scientific team allows the integration of the professional judgement of a diversity of disciplines.

2.0 DATA EVALUATION AND CONTAMINANT IDENTIFICATION

Risk assessments conducted at the Hanford Site use data collected under a number of differing programs including site-specific RI/FS characterizations, historical studies, and routine environmental monitoring programs. The following sections describe the general process a risk assessor should use to evaluate data to determine its usability in the risk assessment and to determine if there are contaminants present.

2.1 DATA EVALUATION

This section discusses the process for using data that have been collected as part of the site characterization process to identify contaminants at a site. EPA (1989a) identifies nine steps that address the organization of data for the risk assessment that should be conducted prior to the risk assessment. These steps include the following:

1. Gather all data available from the site investigation and sort by medium
2. Evaluate the analytical methods used
3. Evaluate the quality of data with respect to sample quantitation limits and data uncertainty (EPA 1992a)
4. Evaluate the quality of data with respect to qualifiers and codes
5. Evaluate the quality of data with respect to blanks
6. Evaluate tentatively identified compounds (TIC)
7. Compare potential site-related contamination with background
8. Develop a set of data for use in the risk assessment
9. If appropriate, further limit the number of contaminants to be carried through the risk assessment.

At the Hanford Site, risk assessments may be part of the RI or LFI report for a specific site. Consequently, many of these nine steps are conducted as part of the overall data evaluation process that occurs for a report and may be addressed and documented in sections of the report other than within the risk assessment (e.g., step 2-5 as part of data validation).

The remainder of this chapter focuses on step 7 to provide specific procedures for Hanford Site risk assessments. The remaining steps (steps 8 and 9) are conducted differently for the human health and the ecological evaluations; these processes are described in Sections 3.1 (human health) and 4.1.1.1 (ecological). The risk assessor, however, should be aware of all nine steps provided above and how they have been conducted for the risk assessment. The quality of the data and confidence in the data based on these nine steps should be addressed in the uncertainty discussions for the risk assessment.

The contaminant identification process is depicted in Figure 2-1. As seen in this figure, analytical investigation sampling data are required to initiate the contaminant identification process. Risk

assessors are reminded that analytical results should not be accepted at face value and that data validation procedures used at the Hanford Site, which have been adapted from EPA protocols and documented by and available from Westinghouse Hanford Company (WHC), should be consulted. 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. It is the risk assessor's responsibility to confirm that all steps in the data evaluation have been performed.

2.2 CONTAMINANT IDENTIFICATION

The contaminant identification process provided below is defensible (based on science and statutory requirements), effective (focused without sacrificing conservatism), and easy to employ (saves time and resources). The process involves background (Hanford Site or project-specific background) screening to determine which of the initial set of hazardous substances and indicator parameters are site contaminants. A contaminant is then defined in the following process as any hazardous substance or indicator parameter that exceeds its background distribution. The overall process is graphically depicted in Figure 2-1.

In order to determine which substances are contaminants, the appropriate background and background distributions must be determined. The selection of background and background distributions, including detected and nondetected parameters and the handling of nondetected parameters and TICs are discussed in the following sections.

2.2.1 Selection of Background

The risk assessor must ensure that proper background data sets are employed in the contaminant identification process. Because many substances defined as hazardous by regulation are naturally occurring, failure to consider background data could lead, for example, to the error of attributing risk to a given site when the risk is, in fact, of natural origin. The purpose of a risk assessment is to characterize the risk posed by the release of hazardous substances from a facility.

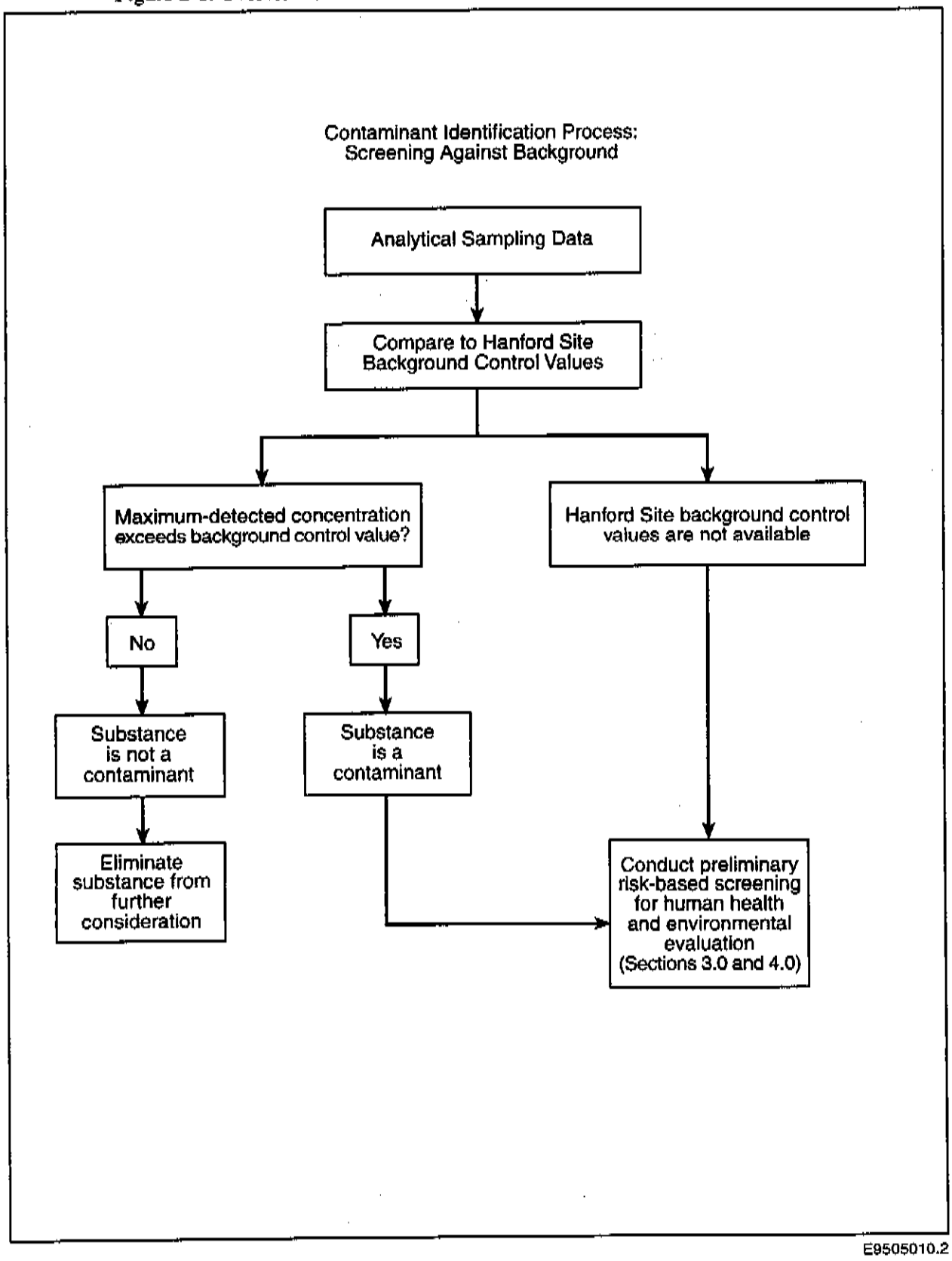
Background conditions do not refer to pristine or pre-industrial conditions because such conditions no longer exist. As stated by EPA in the final exposure assessment guidelines published in May 1992 (57 FR 22888 - 22938, Paragraph 3.5.2.4):

> "Background presence may be due to natural or anthropogenic sources. At some sites, it is significant and must be accounted for. The exposure assessor should try to determine local background concentrations by gathering data from nearby locations clearly unaffected by the site under investigation."

Furthermore, federal and state hazardous substance and waste statutes apply only to releases from specific sites as indicated above, not to wide-spread anthropogenic sources.

Background conditions are being compiled for the Hanford Site under Tri-Party Agreement Milestone M-28. Currently, background reports are available for nonradioactive soil analytes (DOE-RL 1993a) and nonradioactive groundwater analytes (DOE-RL 1992b). A report is also available for radioactive soil analytes (Petersen et al. 1994). Hanford Site background data will generally be used to identify contaminants at a waste site, as discussed below in Section 2.2.2.

Figure 2-1. Overview of Data Evaluation and Contaminant Identification Process.



Site investigation data should also include a characterization of background conditions for each parameter in the initial site-tailored data set in order to define background conditions on a project-specific basis. Appropriate project-specific background 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 releasing a given hazardous substance. Contamination can incorrectly be attributed to a waste unit, when the release is occurring from an upwind, upgradient, or upstream facility. Such data are necessary for the successful implementation of the IRM strategy, which is focused on priority waste sites. Although contamination may be attributed to a source other than a waste site, receptors are exposed to all contaminants in an exposure area regardless of their source.

In the absence of appropriate Hanford Site background data or project-specific data, background data available from another project in a similar environmental setting or those available from a regional, national, or global basis may be used, with documented caution. Acceptable types of background data would be data from locations one can be reasonably certain are not associated with contamination and are representative of site, regional, national, or global conditions. Generally, the further a background or background station is from the project area, the less likely the data from that station are representative of project-specific background or background conditions. A description of and rationale for the selection of background data should be provided in the risk assessment.

2.2.2 Definition of Background Distributions and Use of the Hanford Site Background Data

When the compilation of background conditions are finalized for the Hanford Site under Tri-Party Agreement Milestone M-28, these documents will provide the data necessary for evaluating sampling data for the risk assessments. Background reports for non-radioactive soil analytes (DOE-RL 1993a), non-radioactive groundwater analytes (DOE-RL 1992b), and radioactive soil analytes (Peterson et al. 1994) are currently available for determining which parameters detected at a site are contaminants. For the risk assessment, sampling data for a waste site will be compared to background data by following the methods described in the background data applications guide (DOE-RL 1994c). The application of the Site-wide background distributions should be evaluated on a parameter-by-parameter basis. Analytes that are demonstrated to exceed background values will be identified as contaminants and carried forward to the process for identification of contaminants of potential concern described in Section 3.1.

Hanford Site background data are not currently available for all potential waste site parameters that are detected during project sampling. Those waste site parameters for which background data are not available will be evaluated by performing the preliminary risk-based screening described in Section 3.1.2. Because organic chemicals are not naturally occurring at Superfund sites, organics are not compared to background concentrations (EPA 1989a, p. 5-19, sect. 5.7.3, 2nd para., 1st sentence).

In addition, the Hanford Site background report should be reviewed to ensure that it is appropriate for the project. Hanford Site background would not be used if the site-wide background information is too general for a specific natural condition of the project. For example, the Hanford Site background data report has identified soils in three terrestrial ecosystems that show distinctly higher concentrations for many analytes. These three soil association types are: (1) highly alkaline soils of playa and ephemeral drainages, (2) riparian ecosystem soils, and (3) the grassy soils on Rattlesnake Mountain (DOE-RL 1993a).

2.2.3 Project-Specific Background Data

Project-specific background distributions, if used for screening contaminants, will require review of the intended application by operable unit managers prior to use in the risk assessment process. Project-specific background distributions may also provide information for site characterization and site evaluation to ensure remedial efforts are directed to the source of the contamination. The procedures presented below are recommended for use in calculating project-specific background distributions from sampling data.

2.2.3.1 Detected Parameters

The screening algorithm described in the Section 2.2.2 is also used for detected parameters. Implicit in this tolerance interval approach is the assumption that the parameters (or a transform of them) are distributed normally under the background conditions. This assumption is not always met. An up-front assumption of normality, however, is objective, and for all practical purposes, sufficiently conservative and robust to justify its use in contaminant identification. [Conservative because virtually all environmental contaminant distributions are skewed to the right and are often best characterized by a lognormal distribution (Hahn and Meeker 1991)]. In addition, the project-specific background data sets are not anticipated to be large enough to conduct a meaningful test for normality; furthermore, the background data sets are expected to be far below the numbers necessary to conduct a nonparametric, or distribution-free, tolerance interval evaluation. However, if a data set is sufficiently large, the assumption of normality should be tested. If the assumption of normality is invalid, the data set should be appropriately transformed to approximate normal conditions (Green 1979; Hahn and Meeker 1991).

Care should be taken in combining data sets, when defining background distribution. 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 quality between soil types; differences in ground-water quality between aquifers; and differences in ground-water quality data from different time frames in situations where background 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 background 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.

2.2.3.2 Nondetected Parameters (Nonradioactive Constituents)

When measurements of nonradionuclides are less than the level of detection they are reported as nondetected and the data are referred to as censored. In contrast to nonradionuclides, when radionuclides are measured at less than the level of detection they should not be censored under current radioactivity measurement protocols. Thus the issue of censored data should not arise for radionuclides. In censored data sets, the number of nondetects is known. 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 background 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 background 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:

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

Substituting one-half of 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 and checked. It will not work, however, in those cases where a given parameter is never, not even once, detected in the background data set. In these cases, use of 0.5 SQL gives an unbiased estimate of the mean, but the variance estimate is entirely artificial.

Therefore, when a parameter is never detected in the background data set the highest reported SQL for that parameter is used as a surrogate upper tolerance limit to define the background distribution. This simple procedure objectively interprets the validated data. For example, one can not distinguish 10 U mg/kg from 6 mg/kg, and this procedure acknowledges this fact. 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 to qualify as evidence of contamination.

2.2.3.3 Tentatively Identified Compounds

Sample analyses for organic compounds may indicate the presence of additional organic compounds not on the target compound list (TCL). TICs should be reviewed by the risk assessor and discussed qualitatively in the uncertainty section of the risk assessment, as appropriate. Analytical specialists in the laboratory providing the analyses should assign a tentative identification and concentration based on established identification criteria or report organic chemicals as 'unknown.' Classification of the unknown compound should be given, if possible (e.g., unknown aromatic, unknown hydrocarbon). Quantitative assessment of the risk due to TICs is generally not feasible because there is limited availability of toxicity factors such as slope factors (SF) or reference dose (RfD) for chemicals not on the TCL. However, further evaluation of TICs may be necessary if site information indicates they are chemicals likely to be at the site, that the TICs may be highly toxic, or that there are many TICs present relative to analytes which are on the TCL.

3.0 HUMAN HEALTH EVALUATION METHODOLOGY

The following sections present a detailed discussion on the following four elements of a human health evaluation relative to the HSRAM:

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

Section 3.1 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 3.2 and a discussion of the toxicity assessment is provided in Section 3.3. The integration of exposure information and toxicity information to develop the risk characterization is discussed in Section 3.4. A summary of the human health evaluation is provided in Section 3.5.

The risk assessor is referred to Exhibit 9-1 of RAGS (EPA 1989a) for a suggested outline of the risk assessment report based on the four elements above. The first element has been modified from the RAGS outline by the replacement of the term "chemical" with the term "contaminant" because contamination at the Hanford Site includes radionuclides. Therefore, the more generic term is used. The RAGS outline is provided as a guide and should be modified appropriately for use at the Hanford Site. 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.

The CERCLA RI/FS process at the Hanford Site encourages active discussion by Tri-Party unit managers throughout the development of the work plan, site investigation, and preparation of risk assessments for RI/FS reports. Discussions on the status of the human health evaluation should be appropriately integrated into these activities, without compromising schedule constraints, to keep all parties informed of issues related to the four elements of the risk assessment.

3.1 IDENTIFICATION OF CONTAMINANTS OF POTENTIAL CONCERN

Identification of contaminants of potential concern (COPC) begins with the contaminant list developed as described in Section 2.2. Once 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-based screening procedure. Those contaminants that have the potential to contribute significantly to the risk at the site are referred to as COPC.

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 COPC identification process. This process consists of the following four steps.

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

2. Calculate the risk-based screening concentrations.

3. Compare the maximum concentration to risk-based screening 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 COPC and carry these through the remainder of the risk assessment process.

The following sections discuss these steps in more detail. The preliminary risk-based screening procedure has been developed to retain dominant contaminants and also some additional contaminants that will be determined to be nondominant when the entire risk assessment process has been completed. An overview of the preliminary risk-based screening is provided in Figure 3-1.

3.1.1 Maximum Contaminant Concentration

The maximum detected concentration of a contaminant in a given medium should be used in the risk-based screening regardless of location with respect to potential receptor exposure points. For example, for a baseline risk assessment the maximum concentration of a contaminant detected in soil would be used regardless of depth. The EPA methodology on which this is based does not address acute exposure. Risk of exposure to "specks" (discrete radioactive particles) is not addressed. All contaminants that exceed the background screen described in Section 2.2.2 should be evaluated for potential elimination from the risk assessment based on frequency of detection [as discussed in RAGS (EPA 1989a)].

3.1.2 Risk-Based Screening Calculations

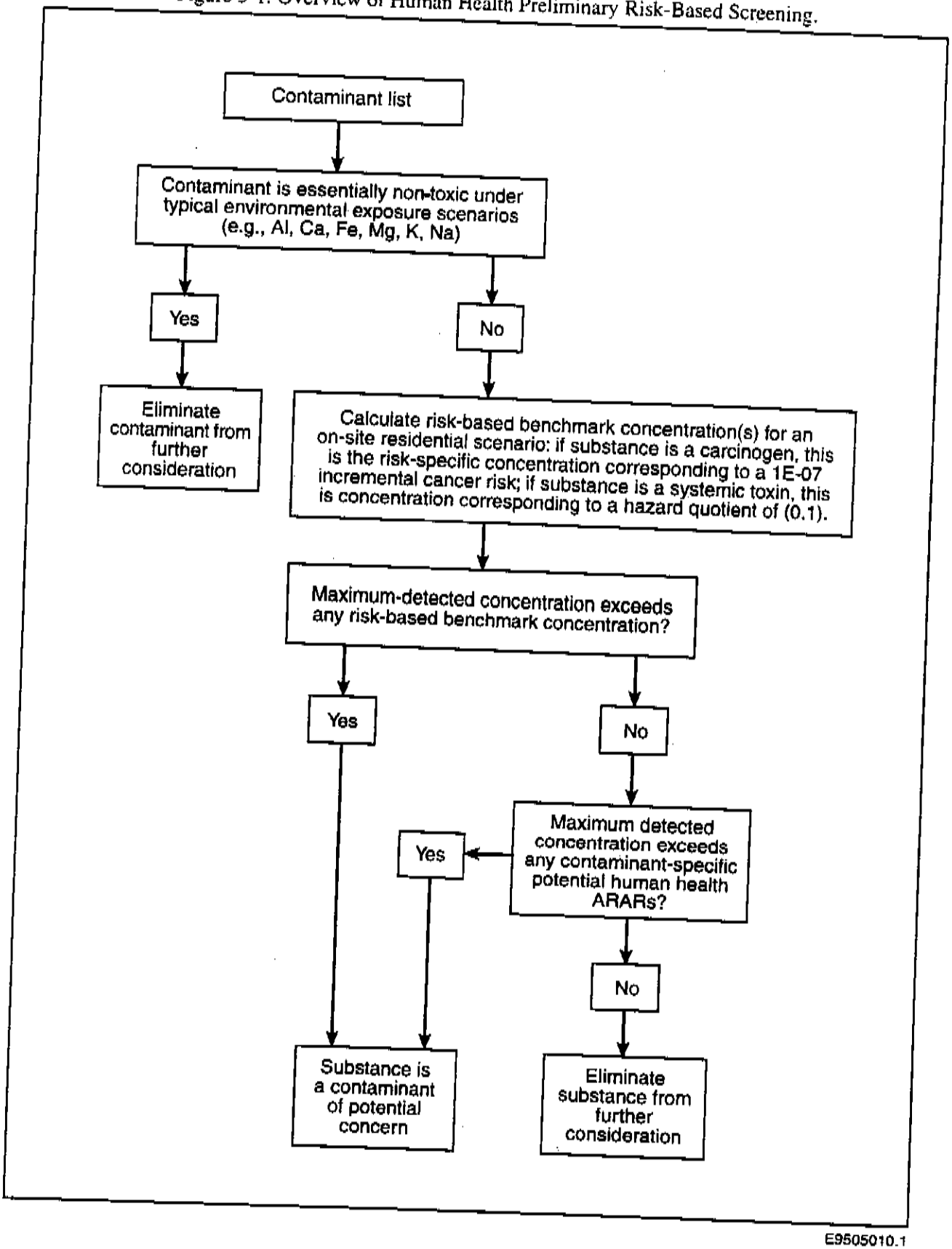
To ensure protectiveness and simplicity, all risk-based screening concentrations are calculated using residential exposure assumptions. The residential exposure parameters provided in Appendix A and Tables A-7, A-8, and A-9 should be used for this purpose.

Toxicity information, including slope factors and reference doses should be included in the evaluation for all contaminants for use in calculating the risk-based screening calculations.
Contaminants for which published toxicity values are unavailable should not be eliminated during screening without documented justification. Guidance for evaluating such contaminants in the risk assessment is provided in Section 3.3.4.

Risk-based screening concentrations should consider both carcinogenic and noncarcinogenic effects. Therefore, concentrations should be calculated using exposure parameters for evaluating carcinogens and separate risk-based screening concentrations should be calculated based on noncarcinogenic systemic effects. Section D-1.0 of Appendix D provides detailed equations for calculating risk-based screening concentrations. Example calculations are provided in Section D-4.0

One or more of four general exposure pathways — soil ingestion, water ingestion, air inhalation, and external exposure to radionuclides — potentially occur at most waste sites and are generally basic to any risk assessment at the Hanford Site. The preliminary risk screening should utilize pathways appropriate to the site under evaluation (e.g., soil ingestion and air inhalation for source risk assessments, water ingestion and air inhalation for groundwater risk assessments).

Figure 3-1. Overview of Human Health Preliminary Risk-Based Screening.



3.1.2.1 Soil Ingestion. As previously noted, the maximum concentration of a contaminant detected in soil should be evaluated in the risk-based screening using the soil ingestion pathway regardless of the depth. Soil contaminants with known inhalation toxicities (e.g., chromium, carcinogenic forms of nickel) must not be eliminated based only on the soil ingestion route [EPA-10 (1991)].

3.1.2.2 Water Ingestion. The water ingestion pathway should conservatively assume, at this time, direct exposure to groundwater. Surface water exposure will not be assessed during this step at many sites because of lack of empirical surface water quality data, or the fact that realistic extrapolation of groundwater 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 risk assessment process (see Section 3.2).

3.1.2.3 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 $\mu g/m^3$ [40 CFR Section 50.6(b)].

3.1.2.4 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 because of 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., transuranics) have complex decay chains that require special consideration in determining the importance and risk associated with these daughter products.

3.1.2.5 Screening Criteria. Screening is performed to eliminate from further concern those contaminants whose maximum detected concentration fall below the following risk-based criteria.

1. In the case of a carcinogen, a contaminant is eliminated if the maximum detected concentration does not exceed a risk-specific concentration corresponding to a 1E-07 incremental cancer risk.

2. In the case of a systemic toxin, a contaminant is eliminated if the maximum detected concentration does not exceed a hazard quotient of 0.1.

However, when the best-available/reasonably-achievable analytical limit of detection (or the sample quantitation limit) are above the concentration corresponding to a 1E-07 incremental cancer risk, the risk-based screening criteria do not apply. In this case, best professional judgement will be used to determine whether the analyte should be carried through the risk assessment and the decision appropriately documented.

3.1.3 Comparison of Contaminants to Benchmarks

In addition to calculating risk-based screening concentrations as noted above, other benchmark concentrations, such as contaminant-specific potential ARARs, should also be compiled for use in the screening process. Even though a contaminant qualifies for elimination on the basis of the above risk-based screening criteria, it will be retained if it exceeds any contaminant-specific potential ARAR (e.g., federal or state ambient water quality criteria).

Some analytes (e.g., aluminum, calcium, iron, magnesium, potassium, and sodium) are essential nutrients and are nontoxic under typical environmental exposure scenarios. These analytes can be eliminated from the human health evaluation as noted in EPA-10 (1991) and EPA (1989a).

3.1.4 Contaminant of Potential Concern Summary

The preliminary risk-based screening procedure results in a site-specific list of contaminants of potential concern for the human health risk assessment. These contaminants are then subjected to the remainder of the data evaluation and 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 mercury — may be encountered in the site environment at levels elevated above the statistically defined control conditions. The contaminant identification process would result in both calcium and mercury being defined as site contaminants. With the substances chosen for this example, no formal exposure or toxicity assessments are required for most individuals to understand that while mercury can be highly toxic, calcium is an essential micronutrient that is virtually nontoxic from the perspective of typical environmental exposure scenarios.

Depending on the maximum concentration detected, the risk-based screening process may define mercury 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.

3.2 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 risk associated with exposure to contaminants at a site.

The elements of the exposure assessment are discussed in the following sections. The characterization of the exposure setting and the general Hanford Site exposure setting are discussed in Section 3.2.1, potentially exposed populations are identified in Section 3.2.2, and exposure pathways are presented in Section 3.2.3. 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 Section 3.2.4, with a more extensive discussion and a summary of exposure parameters provided in Appendix A. Section 3.2.5 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 section. The exposure assessment uncertainty evaluation is discussed in 3.2.6.

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 1993* (Dirkes 1994), the *Hanford Site Development Plan* (DOE-RL 1990), the *Final Environmental Impact Statement - Disposal of Hanford Defense High-Level, Transuranic, and Tank Wastes: Hanford Site, Richland, Washington* (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.

3.2.1 Characterization of Exposure Setting

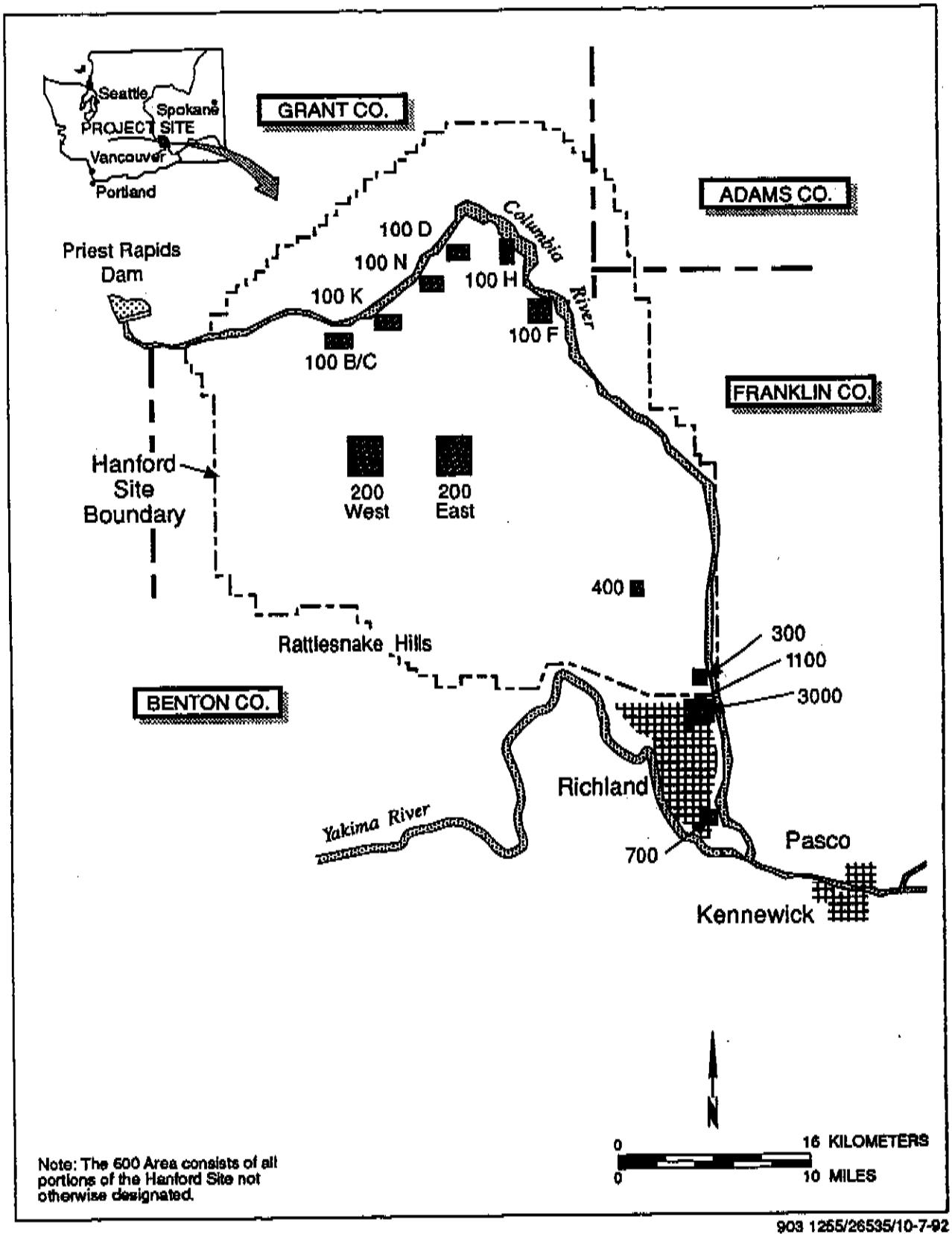
Characterization of the physical setting of a specific site is essential in developing the exposure assessment. Section 6.2.1 of RAGS (EPA 1989a) identifies important site characteristics that should be considered when preparing an exposure assessment. Because of the size of the Hanford Site, characterization 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 sections to provide background for the methodology and the exposure scenarios discussed in Section 3.2.4 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 3-2). 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 3-2.

The general climate of the Hanford Site is arid to semiarid because of 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 evaportranspiration is approximately 18 cm (7 in.) (Weather Bureau and SCS 1962). However, evaportranspiration levels can vary greatly with location at the Hanford Site because of differences in soil properties and the type and density of local vegetation. A study by Wallace in 1977 calculated evaportranspiration levels of from five to nine times the mean annual precipitation (DOE 1988a). Preliminary work has indicated that during significant precipitation events, water can move below the vegetation root zone and escape evaportranspiration.

Figure 3-2. Hanford Site and Area Designations.



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 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). High and low temperatures and maximum wind speed vary considerably from these averages.

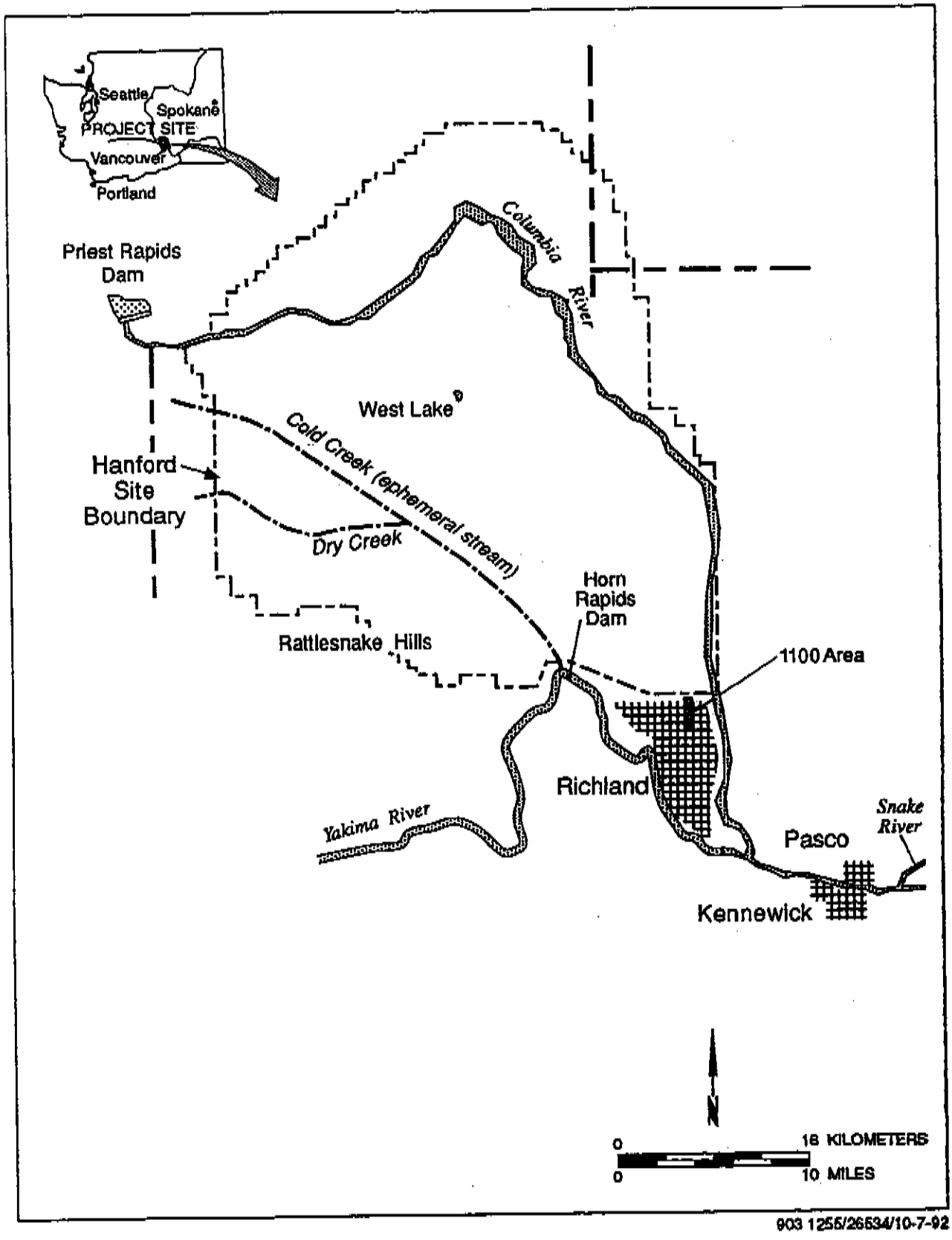
The major surface-water bodies in the Pasco Basin are shown in Figure 3-3. The Columbia River, the major river in the area, crosses the northern portion of the Hanford Site, then turns southward to form the site's eastern boundary. The Columbia is an important source of water for domestic, agricultural, 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. Except for the stretch of river at the mouth of the Columbia River, 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 final environmental impact statement/study report [National Park Service (NPS) 1994)] was released in 1994. Under consideration for the Hanford Reach, including area within approximately 0.40 km (0.25 mi) 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 task force has recommended the Hanford Reach and the entire Wahluke Slope be designated a National Wildlife Refuge with Wild and Scenic River overlay.

The arid-to-semiarid climate of the Pasco Basin does not support any perennial streams that reach the Columbia or Yakima Rivers; Rattlesnake Springs is a small perennial stream located within the Dry Creek Valley near the southwest corner of the Hanford Site. Other small streams arise in the Rattlesnake Hills and flow northward, but do not reach the Dry Creek/Cold Creek Valley. An intermittent perennial stream, Rattlesnake Springs, and two ephemeral streams, Cold Creek and Dry Creek, are located on the southwestern boundary of the Hanford Site (Figure 3-3). The only pond on the Hanford Site that is recharged by groundwater is West Lake, near the 200 East Area. The major source of recharge for the pond is locally mounded groundwater infiltrating from 200 Areas operations. Surface water is disposed to ditches and ponds in the 100, 200, and 300 Areas.

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



3.2.2 Characterization of Potentially Exposed Populations

Current on-site receptors are primarily Hanford Site workers. Approximately 12,000 people were employed in U.S. Department of Energy (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 the U.S. Ecology Low-Level Radioactive Waste Landfill.

The area surrounding the Hanford Site is predominantly rural farmland, with the exception of the cities of Kennewick, Pasco, and Richland. Using the Hanford Meteorological Station 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 and the Confederated Tribes of the Umatilla Indian 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.

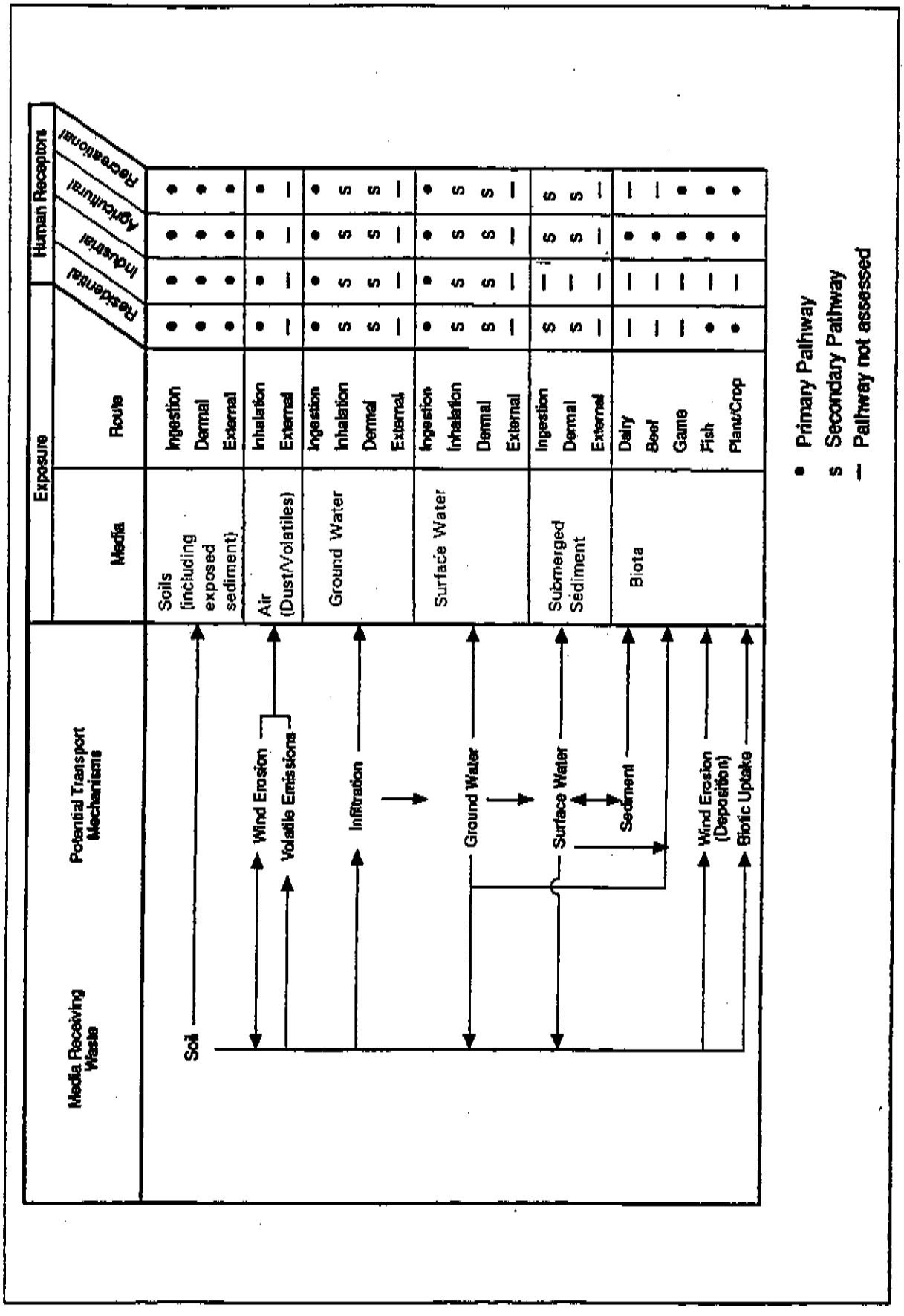
For the risk assessment, the receptor populations for a site are selected based on both the location and activities of current populations, as discussed above, and populations associated with potential future land use. Section 6.2.2 of RAGS (EPA 1989a) provides information on identifying potentially exposed populations. Specific current and future scenarios for Hanford Site risk assessments are discussed in Section 3.2.4.

3.2.3 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 section discusses the identification of exposure pathways including a conceptual model of the exposure assessment.

3.2.3.1 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 3-4, summarizes paths that hazardous substances may take to reach potential receptors. A discussion of the model is provided below. It is

Figure 3-4. Conceptual Model for Human Exposure Assessment.



important to note that although many pathways are possible, the conceptual model focuses on those pathways that are likely to contribute significantly to overall risk 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 3.2.3.2 and in Appendix A as part of the discussion of scenarios.

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

- A source and mechanism for hazardous substance release
- Transport mechanisms/media
- Exposure media or point
- Exposure routes
- 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, burial of waste 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) and transport mechanisms. Direct receptor contact with the soil can result in incidental ingestion, dermal exposure, and external radionuclide exposures. 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 receptors may be exposed through dermal contact with the water via wells during showering, bathing, and other domestic or commercial water use. Inhalation of volatile substances in groundwater may also occur during groundwater use 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.

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.

Hazardous substances in surface water may directly impact biota consumed by human receptors (e.g, bioaccumulation in fish, livestock watered with Columbia River water). 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.

3.2.3.2 Exposure Pathways. As noted above, the conceptual model (Figure 3-4) summarizes pathways that are likely to contribute significantly to overall risk and that can be assessed with relatively well-defined parameters. These pathways are considered either primary or secondary pathways for Hanford Site risk assessments as discussed below.

Primary pathways are presented in the conceptual model as those pathways, exposure media, and routes that should be quantitatively evaluated for a specific scenario if contaminants are present in a medium. The primary pathways listed below include many of those evaluated in risk assessments at most hazardous waste sites (EPA-10 1991 and EPA 1991a) and frequently are risk-driving pathways at hazardous waste sites (see Figure 3-4). These pathways should be evaluated for all scenarios and include the following:

- Ingestion of soil and dust
- Inhalation of fugitive dust and/or volatiles
- Ingestion of water (surface water or groundwater)
- Dermal contact with soil contaminants
- External exposure from radionuclides in soil.

External exposure to radionuclides is added as 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.

If radioactive contaminants are covered by clean soil, the soil may provide sufficient shielding to effectively eliminate the external exposure pathway. This shielding effect is a function of shielding thickness, photon energy, and source activity. However, it may be generally assumed that all low energy (< 100 keV) photons are effectively shielded if the source is covered by at least 1 meter of soil (Kocher and Sjoreen 1985). Three meters of soil will eliminate external exposure to all but the highest energy photon emitters (> 1 MeV).

Depending on land use, excavation may occur, thereby reducing or eliminating the clean soil cover. The effect of shielding on the reduction or elimination of external exposures should therefore be based on the thickness of clean soil cover assumed to exist following excavation.

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. The biota pathways are the following:

- Consumption of dairy products (milk)
- Consumption of beef
- Consumption of game (e.g., venison, upland game birds, waterfowl)
- Consumption of Columbia River fish
- Consumption of homegrown produce.

Secondary pathways are those 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. All pathways qualitatively evaluated should be discussed in the uncertainty sections of the exposure assessment and the risk characterization. The following secondary pathways are indicated in the conceptual model presented in Figure 3-4:

- Ingestion of sediment
- Dermal contact with sediment
- Inhalation of volatiles from water
- Dermal contact with water.

Pathways have been selected as secondary pathways because they represent exposure routes that depend on contaminant-specific parameters, frequency or duration of exposures, or likelihood of occurrence. These secondary pathways may also contribute less to the overall risk or may require qualitative evaluation because of limited availability of contaminant-specific information.

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 little, if any, direct 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).

Dermal exposures to water are also considered a secondary pathway because dermal exposures to water occur over relatively short timeframes, and significant exposures may rely on chemical-specific factors such as dermal permeability. The faster penetrating contaminants (dermal permeability constant > 0.1 cm/hr) may also pose a hazard similar to direct consumption, but environmental contaminants in this range appear to be a minority (EPA 1992c). Although risk values may indicate that dermal exposure is a dominant pathway, it is important to note that there is an extreme degree of uncertainty associated with its quantification. Dermal uptake is generally not an important pathway for exposure to radionuclides, most of which have small dermal permeability constants. However, there may be radionuclides at the Hanford Site, such as tritium, which have high dermal permeabilities and should be evaluated on a case-by-case basis.

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. 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 the following:

- Ingestion of contaminated particulates or volatiles secondary to inhalation of contaminated air
- Dermal exposure to airborne contaminants.

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 is comparable to dermal exposure to these analytes in soil. Therefore, the RAGS assertion that dermal uptake is generally not an important pathway for radionuclides and chemicals is as valid for airborne contaminants as it is for soil contaminants. Dermal exposures are considered to be lower than inhalation intakes and are generally not considered in Superfund risk assessments, as noted in Sections 6.3 and 10.5.5 of RAGS (EPA 1989a). Qualitative discussion of such pathways may be appropriate in the uncertainty section of a 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 them. The three pathways include the following:

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

Exclusion of the air immersion pathway is suggested because of 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 submerged sediment or due to water submersion may also be excluded from evaluation. As stated in Section 10.5.5 of RAGS (EPA 1989a), the shielding effects of water and the generally short duration of water submersion exposures typically make this a pathway of lesser significance.

If site-specific information suggests that these may be important pathways, radiation doses and risk incurred during such events may be evaluated as follows. This first step is to use dose rate conversion factors (DRF), combined with estimates of exposure duration, to calculate committed effective doses due to radioactively contaminated media. Current sources of DRF include DOE (1988b) and EPA (1988b). The resulting dose can then be multiplied by a cancer incidence risk factor to yield a cancer risk estimate. For the purposes of this methodology, a cancer incidence risk factor of 6.2E-04/rem should be used, as this is the nominal factor currently employed by EPA (1989e) and is the risk factor most consistent with EPA 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 risk methodology have indicated primary and secondary pathways for evaluation in risk assessments, this does not preclude the evaluation of other pathways. The risk assessor, lead regulatory unit manager, or the DOE unit manager may identify contaminants or pathways at a specific site that 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.

3.2.4 Exposure Scenarios

The Hanford Site information found in Section 3.2.1, the current receptor information discussed in Section 3.2.2, and the pathway information and conceptual model presented in Section 3.2.3 are integrated to develop current and potential future 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 at the site. Future exposure estimates provide decision-makers with an understanding of likely future risk.

As described previously, the scenarios developed in HSRAM are generally analogous to the conceptual use options described in the HFSUWG (1992) report. However, one potentially unique future use option is a Native American scenario. The specifics of such a scenario are currently under development, and may be incorporated in future revisions of HSRAM.

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. For the commercial/industrial scenario two variations are provided, limited action and no action. The limited action scenario is provided in the HSRAM because of recent DOE guidance on Use of Institutional Controls in a CERCLA Baseline Risk Assessment (DOE 1992). As indicated in Section 3.2.2, almost all current use of the Hanford Site, except for that described under the recreational scenario, is commercial/industrial. The recreational scenario has been developed 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 currently occur near the Hanford Site.

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 inclusion or exclusion of any of the four scenarios and evaluation of the limited action alternative should be determined in a timely manner by the regulatory and DOE unit managers with support from technical and risk assessment personnel. The rationale for inclusion or exclusion of current and future scenarios should be well documented. In addition to on-site scenarios, other receptors, such as recreational populations or those that may be located off-site, should also 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.

The scenarios are briefly discussed below. The risk assessor is referred to Appendix A for more extensive information in each scenario including exposure parameters that should be used for preparing risk assessments.

Limited Action - Commercial/Industrial Scenario: Because the current use of the Hanford Site is commercial/industrial, this scenario will be assessed at most sites. After analyzing the required reasonable maximum exposure (RME), additional exposure scenarios could be developed wherein the benefits of maintaining existing institutional controls or newly proposed controls are accounted for in a limited action - commercial/industrial scenario. A site-specific limited action commercial/industrial scenario may 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), frequency and duration of activities (e.g., daily, monthly), and media contact during the activities (e.g., drinking water, soil) should be applied. Development of a site-specific scenario will generally be the exception and will require agreement of the operable unit managers. All site-specific data and values must be justified and documented in the risk assessment report as recommended by EPA (1991a).

Commercial/Industrial Scenario: The commercial/industrial scenario is presented in Appendix A. The baseline risk assessment should assume no action and this scenario may be used, as appropriate, for evaluating current exposures when site-specific activities are similar to those represented by this scenario. It may also be used for evaluating future commercial/industrial scenarios. 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 by 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, A-2, and A-3 of Appendix A.

Recreational Scenario: 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 exposure parameters and factors are summarized in Tables A-4 and A-5, and A-6 of Appendix A.

Revisions in the recreational scenario may be required when options under consideration for the Hanford Reach are finalized. The *Hanford Reach of the Columbia River, Comprehensive River Conservation Study and Environmental Impact Statement* (NPS 1994) has proposed that land along the Hanford Reach be designated a National Wildlife Refuge with Wild and Scenic River overlay.

Residential Scenario: As discussed above, there is not current on-site residential use of Hanford Site land. However, residents are located in areas adjacent to, downwind, and down river from the site. 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-7, A-8, and A-9.

Agricultural Scenario: An agricultural scenario is also provided in Appendix A. As discussed in Section 3.2.2, 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-10, A-11, and A-12.

3.2.5 Quantification of Exposures

The exposure assessment includes a quantification of exposures for various pathways and receptors. Exposure is 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 section describes the methodology for calculating intakes that are integrated with toxicity values in the risk characterization to calculate risk.

3.2.5.1 Reasonable Maximum Exposure. The NCP and RAGS (EPA 1989a) recommend the evaluation of exposures based on RME. Similarly, 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.

3.2.5.2 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 (UCL) on the arithmetic average. RAGS (EPA 1989a) recommends using the maximum detected concentration when the 95% UCL exceeds the maximum concentration detected. Additional guidance on calculating the 95% UCL is provided by EPA (1992d). See Section 3.2.5.4 for additional information on exposure concentrations for radioactive contaminants.

Modeled exposure concentrations may also be required to provide estimates of exposure concentrations at points remote from the sources, concentrations at a future time, or concentrations in biota. Two Tri-Party Agreement milestones, M-29-01 and M-29-02, were established to support the modeling efforts that may be required for baseline risk assessments at the Hanford Site (Ecology et al. 1994). 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 Tri-Party Agreement 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 nonradioactive 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 the following:

- *Wildlife Exposure Factors Handbook* (EPA 1993)
- *Estimating Exposures to 2,3,7,8-TCDD* (EPA 1988c)
- *Methodology for Assessing Health Risks Associated with Indirect Exposure to Combustor Emissions* (EPA 1990)
- *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 nonradioactive substances in a food chain are:

- *Review and Analysis of Parameters for Assessment Transport of Environmentally Released Radionuclides through Agriculture* (Baes et al. 1984)
- *Development of a Risk Assessment Methodology for Land Application and Distribution and Marketing of Municipal Sludge* (EPA 1989f)
- "Model of Organic Chemical Uptake and Clearance by Fish from Food and Water" (Clark et al. 1990)
- "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 Health Hazards: A Textbook of Case Studies* (Paustenbach 1989)
- *Assessing Human Health Risks from Chemically Contaminated Fish and Shellfish: A Guidance Manual* (EPA 1989g)
- *Radioactive Contamination from Decommissioning* (Kennedy et al. 1992).

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 3.2.3.2.

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, wine, vegetables (e.g., cabbage, broccoli), fruits (e.g., apples, cherries, grapes, and melons), wheat, alfalfa, beef, chicken, and eggs 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 because many of these chemicals volatilize or biodegrade, resulting in depletion of the contaminant source. Future scenarios for the baseline risk assessment are the years 2018 and 2118. In practice, the application of these time frames is subject to agreement between operable unit managers, especially when land release or a change in land use sooner than 2018 is likely.

3.2.5.3 Calculation of Nonradioactive 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:

$$\text{Intake} = \frac{C \times IR \times EF \times ED \times CF}{BW \times AT} \qquad 1$$

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)

Typical concentration units, for example, are mg/kg, mg/L, mg/m^3 for soil, water, or air, respectively. Typical contact rate units for ingestion are mg/d, L/d, and g/d for soil, water, and biota ingestion, respectively; a typical contact rate for inhalation is m^3/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:

$$\text{Dermally absorbed dose} = \frac{CW \times SA \times Kp \times ET \times EF \times ED \times CF}{BW \times AT} \quad 2$$

where :

	CW	=	Concentration of chemical in water (mg/L)
	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:

$$\text{Dermally absorbed dose} = \frac{CS \times SA \times AF \times ABS \times EF \times ED \times CF}{BW \times AT} \quad 3$$

where :

	CS	=	Concentration of chemical in soil (mg/kg)
	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)
	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)

Section D-2.0 of Appendix D provides detailed equations for calculating intake values for nonradioactive contaminants for a variety of exposure pathways. Section D-5.0 provides examples of intake calculations.

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.

3.2.5.4 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 nonradioactive contaminants are different. Unlike nonradioactive 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 Health Effects Assessment Summary Tables (HEAST) (EPA 1994) or subsequent editions as the method for calculating lifetime cancer induction risk from radioactive contaminant exposures. Standard EPA equations for exposure and risk assessment, as provided in EPA (1989a), 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:

$$\text{Intake} = C \times IR \times EF \times ED \qquad 4$$

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 are pCi/m^3 for air, pCi/g for soil, and pCi/L 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. Typical contact rate units for ingestion are mg/d, L/d, and g/d for soil, water, and biota ingestion, respectively; a typical contact rate for inhalation is m^3/d.

The above equation may also be used for the evaluation of external exposures if geometric considerations are ignored and an infinite slab source is assumed. In this case, the "intake" is external exposure and 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:

$$IR_{ext} = ET \times RF \times CF \qquad 5$$

where:

IR_{ext}	=	External exposure contact rate (yr/d)
ET	=	Exposure Time (hr/d)
RF	=	Dose reduction factor (0.8, unitless)
CF	=	Conversion factor (1.14E-04 yr/hr)

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 3.2.5.2 and Appendix B).

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. EPA (1991c) recommends a default dose reduction factor of 0.8. The dose reduction factor may also be considered to account for ground roughness.

Section D-2.0 of Appendix D provides detailed equations for calculating intake values for radioactive contaminants. Section D-5.0 provides examples of these intake calculations.

3.2.6 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 the following:

- Adequate identification of current land use
- Likelihood of future land use occurring
- Exclusion of chemicals from the quantitative risk assessment
- Exclusion of pathways from the quantitative risk assessment
- Model assumptions that impact exposure point concentrations
- When a 95% UCL exceeds the maximum detected value, the contaminant will be characterized with the maximum value
- Use of standard default parameters
- Uncertainty related to site-specific parameters
- Uncertainty related to biotransfer factors
- Uncertainty related to site characterization and sample data.

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.

3.3 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 Region 10 (EPA-10 1991). Supplemental information is provided on the toxicity assessment for radionuclides. Section 7.0 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 Section 3.3.1. Sections 3.3.2 and 3.3.3 present information on noncarcinogenic and carcinogenic toxicity values, respectively, for both nonradioactive and radioactive substances. Substances without toxicity values or without route-specific toxicity values are discussed in Section 3.3.4. A discussion of the toxicity profile is presented in Section 3.3.5 and the evaluation of uncertainty in the toxicity assessment is discussed in Section 3.3.6.

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.

3.3.1 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 Section 3.3.1.1 and the hierarchy of sources for numerical toxicity values is discussed in Section 3.3.1.2. In addition, a companion document to the HSRAM is currently being developed to provide general toxicity information and numerical toxicity values for typical Hanford Site contaminants.

3.3.1.1 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 Poison* (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 Databases (e.g., IRIS and HEAST)
- *The Merck Index: An Encyclopedia of Chemicals, Drugs and Biologicals* (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)* [National Research Council (NRC) 1990)]

- *Risk Assessment Methodology: Environmental Impact Statement for NESHAPS Radionuclides, Volume I: Background Information Document* (EPA 1989e)

- *Limitation of Exposure to Ionizing Radiation* (NCRP 1993).

3.3.1.2 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 1994) or subsequent editions. HEAST 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. Section 3.3.4 below discusses how to evaluate substances without IRIS or HEAST toxicity values.

3.3.2 Toxicity Values for Noncarcinogenic Substances

Systemic, toxic effects other than cancer can be associated with exposures to both chemicals and radionuclides. Sections 3.3.2.1 and 3.3.2.2 discuss the toxicity values and the approach for evaluating noncarcinogenic effects that may occur from exposure to nonradioactive substances and radioactive substances, respectively.

3.3.2.1 Nonradioactive Substances. The 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 that 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). 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 (RfC). The RfC can be converted to an RfD for use in risk characterization using the following conversion (EPA-10 1991):

$$\text{RfD} = \frac{\text{RfC (mg/m}^3\text{) x 20 m}^3\text{/day}}{\text{70 kg}} \qquad 6$$

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.

3.3.2.2 Radioactive Substances. Radiation-induced health effects can be classified as stochastic or nonstochastic (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. The term acute is not exclusive to exposure duration but may also be used to characterize any short, or sudden, event and is used to refer to effects that manifest over a short time period. It does not refer to a length of exposure. 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 risk 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 insignificant chemical toxicity. Some exceptions may occur (e.g., the nephrotoxic effects of uranium) and will be evaluated on a case-by-case basis.

3.3.3 Toxicity Values For Carcinogenic Substances

Toxicity values have also been developed for evaluating potential human carcinogenic effects from exposures to chemicals and radionuclides. Section 3.3.3.1 discusses the toxicity values utilized in evaluating nonradioactive substances. Section 3.3.3.2 discusses the evaluation of carcinogenic effects from exposure to radioactive substances.

3.3.3.1 Nonradioactive 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 (1989a) weight-of evidence classifications are the following:

- 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 Noncarcinogenicity 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 risk are specific for the route of exposure. For nonradioactive substances, oral or inhalation slope factors and unit risk 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. The unit risk must be converted to inhalation slope factors for use in risk characterization. The recommended conversion (EPA-10 1991) is:

$$\text{Slope factor} = \frac{\text{Unit Risk } (\mu g/m^3)^{-1} \text{ x } 70 \text{ kg x } 1E{+}03 \ \mu g/mg}{20 \ m^3/day} \qquad 7$$

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).

3.3.3.2 Radioactive Substances. Because 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 risk 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 risk associated with environmental concentrations of radionuclides (see Section 3.4.1). 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.

3.3.4 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 (MRL) 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. 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 nonverified toxicity values.

3.3.5 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 nontechnical 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.

As indicated previously, a companion document to the HSRAM is currently being developed to provide toxicity profiles for typical Hanford Site contaminants. Profiles will include numerical toxicity values as well as discussion describing the health effects of concern and the extent of supporting data.

3.3.6 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 sections. Additional information on uncertainty in toxicity assessments can be found in Sections 7 and 8 of RAGS (EPA 1989a).

3.3.6.1 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 the following:

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

For noncarcinogenic substances, the toxicity values (RfD) 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. Use of uncertainty factors in the development of RfDs is directly related to the uncertainty associated with hazard quotients (HQ) calculated using these RfDs. More confidence can be placed in HQs calculated from RfDs with small uncertainty factors than HQs calculated from RfDs with large uncertainty factors.

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 database 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.

It is important to note that use of EPA toxicity values restricts the risk assessment to an evaluation of selected critical health effects, not a range of health effects that a receptor population might experience. This issue should be recognized in the uncertainty assessment, and the assessor should evaluate whether the health effects of concern have been adequately identified.

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 risk 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 risk derived for one population (Japanese) to another (U.S.); estimation of dose to target cells due to intakes of alpha-particle emitters; and statistical uncertainties (EPA 1989e).

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.

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

- Uncertainty in the toxicity information of individual substances
- Possible synergistic, antagonistic or potentiative 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.

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

3.4 RISK CHARACTERIZATION

The information from the exposure assessment and the toxicity assessment is integrated to form the basis for the characterization of risk 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 sections describe the risk characterization methodology. Carcinogenic risk characterization is presented in Section 3.4.1, noncarcinogenic risk characterization is presented in Section 3.4.2, and assessment and presentation of uncertainty is discussed in Section 3.4.3. Use of site-specific human studies is discussed in Section 3.4.4 and a risk characterization summary is presented in Section 3.4.5. Additional information on risk characterization can be found in Section 8 of RAGS (EPA 1989a) and in Region 10 guidance (EPA-10 1991).

3.4.1 Quantification of Carcinogenic Risk

For carcinogens, risk is 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:

$$\text{Risk} = (\text{Chronic Daily Intake})\ (\text{Slope Factor}) \qquad 8$$

This linear equation is only valid for intakes resulting in estimated risk below 1E-02. Cancer risk estimates should be expressed using one significant figure only. Section D-6.0 of Appendix D includes examples of risk calculations for nonradioactive and radioactive carcinogens.

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 risk 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 risk and estimated radionuclide cancer risk). 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 3.2.5.4, 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.

Radionuclide slope factors provided in HEAST for evaluating the external exposure pathway were developed under the assumption that radionuclides are uniformly distributed (e.g., geometric considerations are ignored and an infinite slab source is assumed). Therefore, use of these slope factors may not be appropriate if the extent of soil contamination is limited (e.g., within 1E+04 m^2), or if contaminated soil is covered with a clean soil cover. For such cases, shielding calculations may be performed, when approved by the unit managers, in order to provide a more accurate estimate of the external exposure risk. Such calculations usually provide a dose rate (not a lifetime risk) estimate. Dose rates can be combined with exposure parameters identified in Section 3.2.5.4 to

provide a lifetime dose, which can then be converted into a risk estimate by multiplying by the nominal cancer incidence risk factor (6.2E-04/rem) recommended by EPA (1989e).

A contaminant of concern for the human health evaluation is any contaminant that is retained in the screening process and whose pathway-specific risk or total incremental cancer risk exceeds 1E-06. This risk represents the total risk for a contaminant at the site when the 95% UCL is used as an exposure point concentration, without regard to background. For contaminants of concern, the risk associated with Hanford Site background concentrations should also be calculated and presented in the risk characterization. When contaminants of concern are present in mobile media (e.g., groundwater), the risk associated with project-specific background distribution may also be presented (see Section 2.2.3). The presentation of total site risk, background risk, and project-specific risk will provide Tri-Party risk managers important information to ensure that site characterization, evaluation, and remediation efforts are directed to the waste units releasing the contaminants of concern.

Although the risk from exposure to chemicals and radionuclides can be quantified, the models, assumptions, and data used to estimate chemical risk are very different from those used to estimate radionuclide risk. In spite of the differences between chemical and radiological incremental lifetime cancer risk estimation protocols, the NCP [40 CFR Section 300.432(e)(2)(i)(D)] and MTCACR (WAC 173-340-708) require the consideration of cumulative risk attributable to multiple contaminant or multiple pathway exposures. Therefore, to allow for the evaluation of such cumulative risk, chemical and radiological incremental lifetime cancer risk may be summed. Section 10.7.3 of RAGS (EPA 1989a) provides additional discussion on the summation of risk. When risk is summed, the differences in how risk is estimated for each category of substance should be thoroughly discussed in the uncertainty assessment. Cumulative risk estimates must be put into the proper perspective for each site.

3.4.2 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 risk. 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 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 risk should be appropriately evaluated using the chronic or subchronic RfD.

The formula for estimation of the hazard quotient is:

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

Section D-6.0 of Appendix D provides an example of the calculation of the hazard quotient for nonradioactive contaminants.

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 risk posed by radionuclides does not need to be evaluated (see Section 3.3.2.2). 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.

In addition to evaluation of carcinogenic risk, a contaminant of concern for the human health evaluation is any contaminant whose pathway-specific HQ or total hazard index exceeds 1. As with the evaluation of risk, the contribution of Hanford Site background and project-specific control distribution to the HQ or HI should be presented in the risk characterization for the contaminants of concern.

3.4.3 Assessment and Presentation of Uncertainty

The risk, both carcinogenic and noncarcinogenic, presented in a risk assessment is not a fully probabilistic estimate of risk, but rather a conditional estimate 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 3.2.6 and 3.3.6 and in the following sections. The risk assessor should also consult RAGS (EPA 1989a) and EPA Region 10 guidance (EPA-10 1991) for additional discussion on uncertainty considerations.

3.4.3.1 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) and data evaluation factors (e.g., data validation considerations, TIC) 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.

3.4.3.2 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. Section 3.2.6 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.

3.4.3.3 Toxicity Assessment Uncertainty Factors. Factors related to uncertainty in the toxicity assessment are presented above in Section 3.3.6 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.

3.4.3.4 Risk Characterization Uncertainty Factors. The summation of cancer risk across pathways or for multiple pathways may make the total cancer risk estimate more conservative. This is because each slope factor, for chemical carcinogens, is an upper 95th percentile estimate and such probability distributions are not strictly additive. Also, summing risk from all carcinogens gives equal weight to slope factors derived from animal data and slope factors derived from human data. As discussed in Section 3.4.1, most of the chemical carcinogenic data are derived from animal studies, whereas, radionuclide carcinogenic data have been derived primarily from human exposures.

The relative uncertainty between slope factors for various carcinogens should be fully discussed in the risk characterization. If estimates of carcinogenic risk from chemical exposures and estimates of carcinogenic risk from radionuclide exposures are combined to determine a cumulative risk, the relative significance of this cumulative risk should be put into perspective. Numerical risk estimates should always be accompanied by appropriate descriptive information to characterize the risk.

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.

Although a qualitative assessment of uncertainty in the overall risk estimate is required, at a minimum, the best and most scientific approach to the problem would be to conduct a probabilistic risk assessment from the start of the evaluation. A probabilistic assessment could utilize the range of variation in contaminant information, exposure parameters, and toxicity data to provide a risk distribution curve. This distribution of risk, rather than a point estimation of risk, would provide risk managers with a better understanding of the uncertainties in the risk and the protectiveness of potential risk reduction strategies.

3.4.4 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 nonradioactive 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 ground radiation surveys are conducted. Information on radioactive dose exposures based on surveillance information is published annually in a report available to the public (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.

3.4.5 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 the following:

- Confidence that key site-related contaminants have been identified
- Description of known or predicted health risk (cancer and noncarcinogenic effects)
- Confidence in the toxicity information supporting the risk estimations
- Confidence in the exposure assessment estimates
- Magnitude of the cancer risk relative to the site-remediation goals (i.e., NCP or MTCACR)
- Major factors driving the risk including contaminants, pathways, and scenarios
- Significance of cancer risk estimates from chemicals and radionuclides
- 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 risk 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.

3.5 SUMMARY OF HUMAN HEALTH EVALUATION

The human health evaluation methodology presented in Section 3.0 integrates federal, regional, and state requirements with Hanford Site-specific information to provide a framework for conducting baseline risk assessments at the Hanford Site. As such, it provides a conservative estimate of the risk associated with a 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: 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 3-4 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 risk and hazard quotients/indices. The risk characterization for Hanford Site baseline risk assessments includes consideration of carcinogenic risk related to radioactive contaminants and nonradioactive contaminants. An essential element of the risk characterization is a discussion of the uncertainty associated with the results of the risk assessment. Although a qualitative presentation of the uncertainty should be presented at a minimum, a probabilistic assessment to provide information on the range of the risk and the uncertainty in the risk analysis may be a useful tool in the decision-making process that follows 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 judgment during the preparation of a site-specific risk assessment.

4.0 ECOLOGICAL EVALUATION METHODOLOGY

This chapter provides general guidance and methods for conducting ecological evaluations for risk assessment at the Hanford Site. These evaluations consider risk and exposure scenarios developed to assess current and post-remediation conditions of the site.

The EPA framework (EPA 1992b) for ecological risk assessment provides a basic structure and starting principles for conducting an ecological evaluation. However, it does not provide specific methodology for the evaluation. The objective of this chapter is to provide guidelines and methodology for Hanford Site-specific ecological evaluations, based on the framework guidance and the Tri-Party Agreement. The general framework outline is presented, along with specifics on how the framework is implemented at the Hanford Site.

An ecological evaluation is different from a human health risk evaluation in that the ecological evaluation does the following:

- Estimates risk at several levels of ecological organization (individuals, populations, communities and ecosystems)
- Considers indirect effects to organisms (e.g., risk to a predator when contaminants reduce the number of prey)
- Addresses concerns independent from the human health evaluation
- Considers ecological as well as toxicological data.

The three phases of the ecological evaluation process are the following:

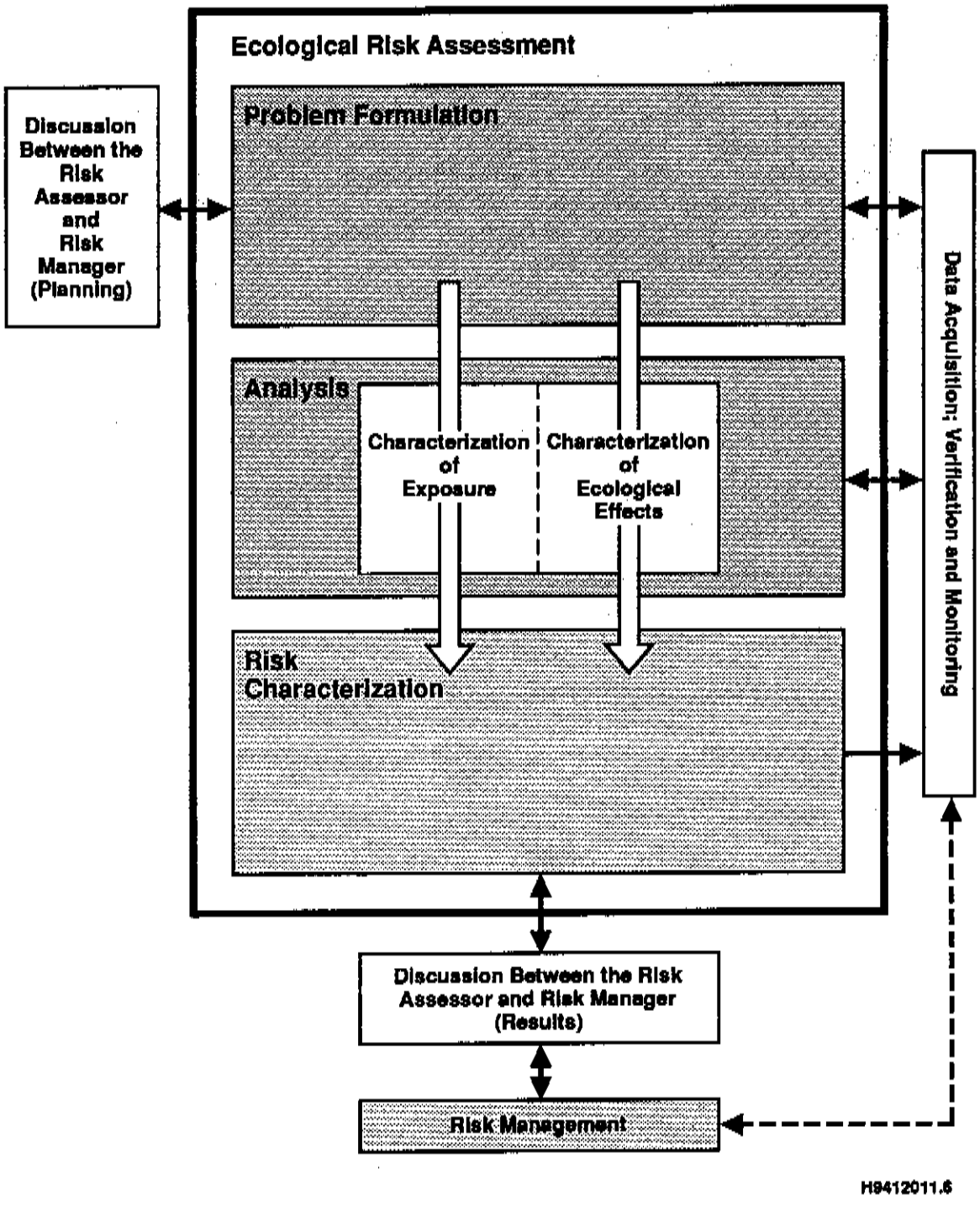
- Problem formulation
- Analysis
- Risk characterization.

The three-phased approach is displayed graphically in Figure 4-1. These phases are addressed in Sections 4.1, 4.2, and 4.3, respectively.

4.1 PROBLEM FORMULATION

The problem formulation phase of the ecological evaluation establishes the focus and extent of the evaluation. It is the site discovery and data-gathering phase. This phase identifies the objectives of the assessment and develops a conceptual model that illustrates the linkages between the ecological receptors at the site and the site contaminants. The conceptual model considers the toxic effects of the contaminants and the transport of contaminants via air, water, soil, and living organisms. This is the first step in identifying what data are required for the evaluation, and should involve reviewing Hanford Environmental Information System (HEIS) data, historical data, LFI data, and other sources. The basic question is, "what plants and animals exist at the site and how are they exposed to the contaminants?"

Figure 4-1. Ecological Evaluation Framework (Derived from EPA 1992b).



To effectively achieve study objectives, the risk assessor and the risk manager must establish communication during the problem formulation phase. The risk manager ensures the inclusion of societal values. The risk assessor ensures that the evaluation addresses the important ecological conditions and concerns. Both perspectives are necessary to appropriately utilize resources to produce technically sound evaluations that are based on a societal value system.

The problem formulation phase, depicted in Figure 4-2, collects data on three basic components to define how biological receptors are exposed to the contaminants, and what the ecological effects of that exposure might be. The basic components are the following.

- Identify and characterize stressors.
  - Are they inorganic or organic chemicals or radionuclides?
  - What are their distributions and concentrations?
  - How are they transported through the environment?
  - Will they bioaccumulate?

- Identify and characterize the potential biological receptors and habitats of concern.
  - What are the principal species at the site?
  - How abundant are they?
  - What is their spatial and temporal distribution?
  - How does energy flow through the ecosystem?

- Determine the possible toxicological effects of the contaminants.
  - What types of plants and animals will they affect?
  - How do the contaminants affect the plants and animals?
  - What concentrations are toxic?

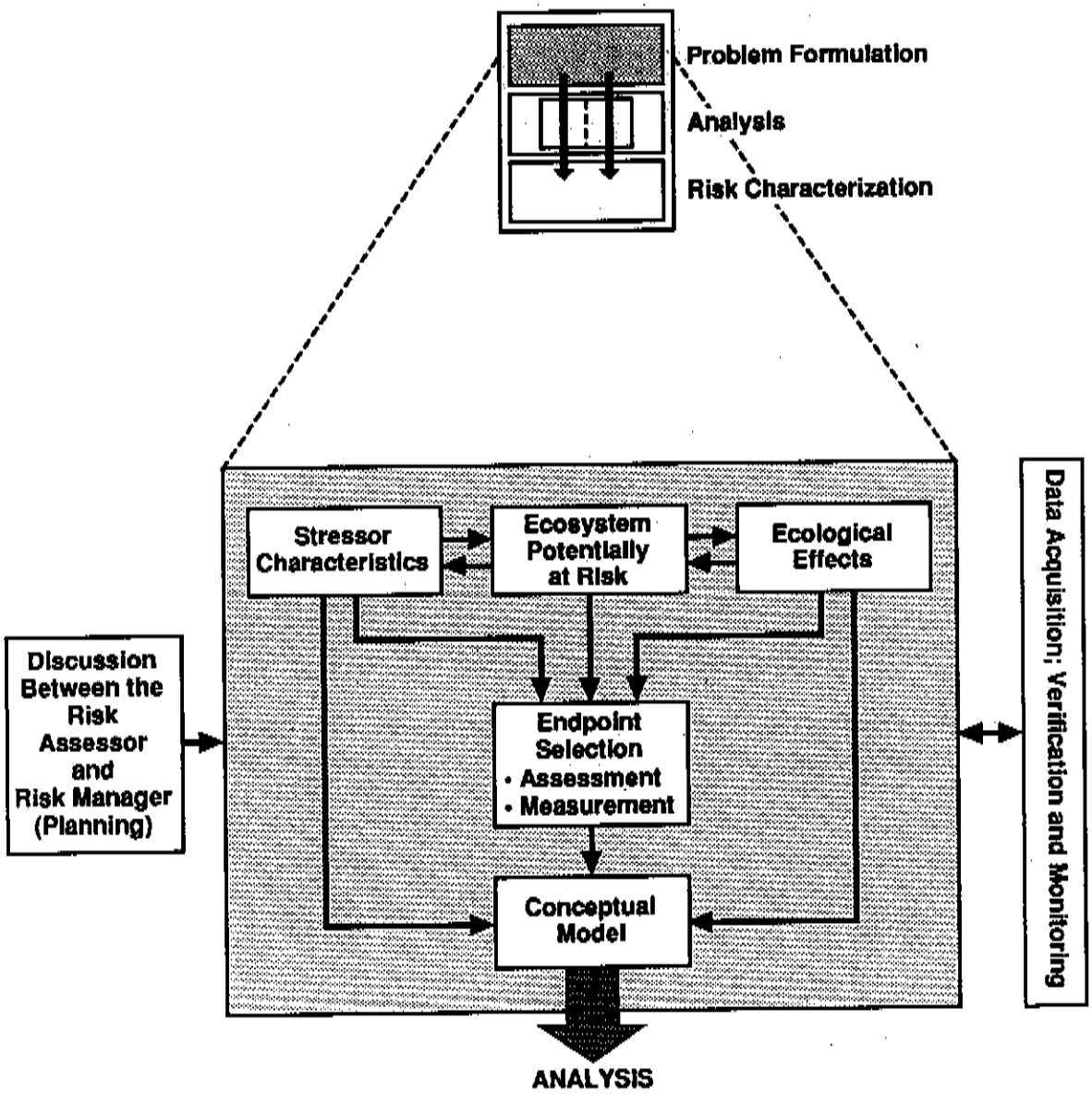
The information from the above basic components is used to determine how ecological effects can be measured and assessed (endpoints). An endpoint is defined as a quantitative expression of the environmental factors (ecological concerns) considered to be at risk (Suter 1993). These factors are reflected in the assessment and measurement endpoints. The information from the three components is integrated into a conceptual exposure model that will target the studies and identify data requirements to complete the evaluation (ecological risk assessment). The conceptual model illustrates how the principal receptors are exposed to the contaminants of concern at the site.

4.1.1 Characterization of Stressors

The characterization of stressors begins with the identification of chemical, physical, or biological entities that can cause an adverse response. Stressor characteristics include type (physical or chemical), intensity (concentration), duration, spatial and temporal distribution (occurrence relative to receptor distributions), and transport mechanisms. The stressors directly addressed by the HSRAM are radiological and chemical contaminants.

The identification of COPC begins with evaluation of site data. Historical information must be used to determine what was potentially disposed in a waste site. Historical data and field data from LFIs and HEIS define what contaminants have been measured in groundwater, surface water, soil and sediments, or biological samples. The quality of the data used in the risk assessment process must meet the data quality objectives set for the study, and be addressed in an uncertainty section of the risk assessment.

Figure 4-2. Problem Formulation for Ecological Evaluations (Derived from EPA 1992b).



Screening against background eliminates those constituents that are present below established background values for the various media. If no background data are available for a given constituent, then that constituent is included in the ecological risk assessment. The screening procedure can be done in conjunction with the human health evaluation or as an independent effort. The step-wise screening process must be consistent with EPA guidance (EPA 1992a) as discussed in Section 2.1. For specifics regarding background screening see Section 2.2. Those contaminants that remain after the background screen are defined as COPC for the ecological evaluation. The COPC are identified for each media (soil, sediments, groundwater, surface water, and biota) that represent a potential exposure route.

There is a difference between the human health and environmental screening process. The human health screening procedure includes, as a final step, a risk-based screening step. The ecological evaluation does not conduct any further routine or standard screening steps. However, the concentrations of the COPC present at the site should be evaluated based on their potential effects to biological receptors, to see if it is appropriate to further reduce the list of COPC. For example, some constituents toxic to humans at low levels may be essential nutrients for plants at the site and may not be toxic to the animals present. The COPC that are found to pose a risk to ecological receptors, at the completion of the environmental evaluation, are designated contaminants of concern (COC) and are carried forward through the ecological risk assessment process.

4.1.2 Ecosystems Potentially at Risk

To develop exposure scenarios and assess ecological risk for a site, the receptors that are potentially exposed to the contaminants must be identified. The receptors can be different species of plants and animals (big sagebrush, Great Basin pocket mouse, Swainson's hawk, coyote), different habitats (wetland, riparian, shrub-steppe), or other ecosystem components. Ecosystems potentially at risk at the Hanford Site include semi-arid terrestrial, riparian, lacustrine, and riverine systems. Major vegetation communities at the Hanford Site are presented in Section C.1.1.1 of Appendix C. Typically, for an ecological risk assessment, certain species of plants and animals from the habitats of interest are selected to represent the potential receptors that may be at risk at the site.

The biological receptors or ecosystem components are characterized to determine abundance, distribution, sensitive life stages, and other attributes that can be used to evaluate how the receptors are exposed to the contaminants. The species of interest are typically species that are considered most appropriate for determining if contaminants have a toxic effect (or negative impact) on the population of that species, or on a community comprised of several species. Species of interest may be species that are the following:

- Abundant at the site
- Structurally important in the ecosystem (dominant in terms of productivity, relative abundance, or biomass)
- Functionally important (major forage species, nesting species)
- Known to be sensitive to the contaminants
- Important as a commercial or recreational resource
- Threatened or endangered, or otherwise protected.

Maps of sensitive or important habitat for selected species of concern (e.g., wetlands, raptors) on the Hanford Site are presented by Downs et al. (1993) While species that use sensitive habitats are of special importance, the environmental risk assessment may include other species that are better indicators of adverse effects from exposure to the COPC.

4.1.2.1 Protected Species. The CERCLA process requires ecological risk assessments to evaluate potential impacts on threatened and endangered species, and crucial habitat that is designated as critical habitat for supporting those threatened or endangered species. Threatened and endangered species and designated critical habitats identified by the federal government are defined pursuant to the ESA. The Federally listed threatened and endangered species that are known to occur, or could occur, at the Hanford Site (and species that have been identified as candidates for future listing) are presented in Table C-1 of Appendix C. The ecological risk assessment will evaluate the potential effects of contaminants on the threatened and endangered species at the Hanford Site, and selected candidate species. There are no critical habitats, as defined pursuant to ESA regulations [50 CFR Section 424.02(d)], on the Hanford Site.

State-designated endangered, threatened, and sensitive plant and animal species that are known to occur, or could occur, at the Hanford Site are also shown in Table C-1. The Washington State Department of Natural Resources (DNR) and the Washington Department of Fish and Wildlife (WDFW) are the agencies responsible for protecting the state listed plant and animal species, respectively. Information on state identified species and sensitive habitats is also available through the Washington Natural Heritage Program. While state laws do not provide a level of protection equivalent to the ESA, the state listed species will be considered in the ecological risk assessment, and the state listed threatened and endangered species will be afforded the same level of protection as the federally listed species. The problem formulation phase of the risk assessment will include meeting with appropriate agencies and entities to determine specific requirements for evaluating protected species, and to assist in confirming which threatened, endangered, candidate, and sensitive species may occur within the area addressed by the risk assessment.

Threatened, endangered, and sensitive species at the Hanford Site are routinely monitored through DOE's Wildlife Resources Monitoring Project (Caldwell 1994). In addition, DOE's Wildlife Surveillance Program (Woodruff and Hanf 1992) monitors contaminant uptake in wildlife. Management Plans for the protection of threatened and endangered species at the Hanford Site have been written and implemented (Fitzner and Weiss 1994a, 1994b).

A regulatory definition of a 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-environment rating values. There is a list of potential Hanford Site sensitive habitats in Appendix C of this document.

4.1.2.2 Selection of Ecological Receptors. Because ecosystems are complex and the interaction of exposure and toxic effects is difficult to measure at the community or ecosystem level, individual species are frequently selected to assess the risk and extrapolate this risk to the population, community, or ecosystem. Factors to consider in selecting potential ecological receptors include the following:

Select species from several taxa

- Reptiles and Amphibians
- Invertebrates

- Mammals
- Birds
- Fish.

Select species from several different trophic levels

- Primary Producer
- Level 1 Consumer (herbivore, granivore)
- Level 2 Consumer (carnivore)
- Level 3 Consumer (carnivore/top predator).

Select species based on site use

- Common resident
- Seasonally abundant
- Nesting or spawning on site
- Migratory.

Select species based on legal/social status

- Federal Species of Concern
- State Species of Concern
- Tribal/Trustee Concerns and Interests.

The risk assessment should include receptors from several different trophic levels, assess various exposure routes, evaluate sensitive life stages, evaluate potential effects on threatened and endangered species, and address impacts to sensitive and valued habitats.

4.1.3 Ecological Effects

The potential ecological or toxicological effects on species or ecosystems from exposure to contaminants of potential concern are gathered from published literature and toxicological databases. These information sources can include field and/or laboratory test data. Identification of toxicological effects refers to the relationship between levels of exposure to the contaminants of potential concern and the degree and severity of the response (e.g., increased mortality, decreased growth rates, or reproductive failure). Ecological effects also include the indirect effects of the contaminants (e.g., nonchemical impact on a predator due to a decrease in the number of prey). The toxicological effects are related to the concentrations of the contaminants, in that different concentrations may have different effects (e.g., behavioral changes, reproductive effects, or mortality).

Many studies have been completed on Hanford Site wildlife and habitats. These studies were designed to assess habitat associations, food requirements, abundance, contaminant levels, and/or locations of species of interest. Some of the ecological sampling took place within individual operable units (e.g., Schmidt et al. 1993) while others addressed the entire site. These studies also provide information on organism weights, home range sizes, abundance, and distribution. This information is very useful for developing the conceptual exposure model, and for assessing risk later in the risk assessment process (see Appendix C). In addition, Driver (1994) reviewed ecotoxicological data for selected hazardous material found at the Hanford Site. The literature review summarized toxicological information for uranium, plutonium, cesium, strontium, cobalt, chromium, technetium, tritium, europium and nitrate.

In selected cases, quantitative ecological models may be developed to simulate contaminant transport through the ecosystem to the receptors, or to estimate what occurs at the community or ecosystem

level when a given population (single species) is impacted as a result of being exposed to one or more contaminants. When possible, these models will be verified by collecting data on the community or ecosystem responses to contaminants at the Hanford Site.

4.1.4 Endpoint Selection

Endpoints can be determined from information compiled in the first three steps of the problem formulation phase (Sections 4.1.1, 4.1.2, and 4.1.3).

> "An endpoint is a characteristic of an ecological component (e.g. increased mortality in fish) that may be affected by exposure to a stressor (Suter 1990). Two types of endpoints are relevant to an ecological evaluation. Assessment endpoints are explicit expressions of the environmental value that is to be protected. Measurement endpoints are measurable responses to a stressor that are related to the valued characteristics chosen as the assessment endpoints" (Suter 1990).

There are four levels of ecological organization: individual, population, community, and ecosystem. As shown in Table 4-1, each of these levels has attributes that could be selected as appropriate measurement or assessment endpoints. However, because of the complexity and inherent variations in natural ecosystems, measurement endpoints are usually selected from the individual and population levels. Changes in the number of species and shifts in the relative abundance of species within a community, however, are relatively easy to measure at the community level. The difficulty in this case may be in establishing a link between the measured community changes and the real cause of those changes.

Table 4-1 presents examples of assessment and measurement endpoints. EPA guidance has stated:

> "Sound professional judgement is necessary for proper assessment and measurement endpoint selection, and it is important that both the selection rationale and the linkages between the measurement endpoints, assessment endpoints, and policy goals can be clearly stated" (EPA 1992b). A public involvement process can be used to aid in the selection of assessment endpoints.

As noted earlier, ecological risk is the probability of an adverse effect on individuals, populations, communities, or ecosystems. Generally, adverse effects at higher levels of ecological organization are initially manifested through adverse effects to individual organisms. The selection of an endpoint (assessment and measurement) must be consistent with the purpose of the ecological risk assessment.

Once assessment endpoints are defined, appropriate measurement endpoints are selected. The EPA (1991d) defines measurement endpoints as those endpoints used to approximate, represent, or lead to an assessment endpoint. While assessment and measurement endpoints can be identical, they most often are not. The rationale for extrapolating from a measurement endpoint to an assessment endpoint must be explicitly stated. For example, in a closed population (e.g., pond) lower reproductive rates for a species could affect the population of the species. Table 4-2 summarizes characteristics of potential assessment and measurement endpoints.

It is likely that the use of individual-level effects will continue to be the basis for many risk assessments because of the difficulty and general lack of methods to extrapolate from measurement endpoints to assessment endpoints at the higher levels of ecological organization. In selecting endpoints for risk assessments, consideration should be given to the location of the waste site, the types of contaminants present, and the likely receptors located within or near the site.

Examples of endpoints that might be used for risk assessment are viability of the prey or predator population (an assessment endpoint), and individual mortality (a measurement endpoint). Other site conditions, such as the depauperate nature of many of the waste sites, occurrence of the species on and adjacent to the waste site, and seasonal use of the sites by the species should also be considered in selecting endpoints.

Table 4-1. Recommended Endpoints at Each Level of Ecological Organization[a].

Level of Organization	Assessment Endpoints	Measurement Endpoints
Individual	Organism health (Appropriate only for threatened and endangered species)	Death[b] Growth[b] Reproduction[b] Morbidity Behavior
Population	Viability	Birth rate Death rate Immigration/Emigration Age-Size-Class Structure[b] Distribution[b] Abundance[b]
Community	Deviation in structure and function from unimpaired community	Species shifts Numbers of species[b] Species dominance[b] Trophic shifts
Ecosystem	Deviation in structure and function from unimpaired system	Biomass[b] Productivity (P/R ratio)[b] Nutrient dynamics Materials and energy flow[b]

[a]Generic examples are shown in this table; in practice, an ecological component would also be specified (e.g., mortality in trout; flood retention by wetlands). In addition, the spatial scale of the endpoint is often specified.

[b]Depending on the goal of the assessment, these measurement endpoints may also serve as assessment endpoints.

Table 4-2. Characteristics of Assessment and Measurement Endpoints. Derived from EPA (1989b).

Assessment Endpoints	Measurement Endpoints
Socially relevant Biologically relevant Unambiguous operational definition Measurable or predictable Susceptible to the contaminant Logically relevant to the project decision	Corresponds to or is predictive of an assessment endpoint Readily 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

4.1.5 Conceptual Model

The conceptual model development involves reducing the study area into a practical ecological foodweb and contaminant transport and exposure model, that can be tested during the analysis phase of the evaluation. The model documents assumptions made in the problem formulation process and allows for a meaningful review of this first phase of the evaluation. It also serves as the basis for initiating the analysis phase of the ecological evaluation. The conceptual exposure scenarios for the risk assessment will be partly hypothetical because of institutional and engineering controls that currently limit contaminant transport from the waste sites. For example, contaminant transport from waste sites in the 100 Areas has been minimized by covering the sites with clean gravel or cobbles and treating the surface with nonselective herbicides on an annual basis. In the 200 Areas, most of the inactive waste sites have been re-vegetated with Siberian wheatgrass and are treated with broadleaf herbicides on a regular basis (Stegen 1994). A generic version of a conceptual model is depicted in Figure C-2, Appendix C.

> "The major focus of the conceptual model is the development of a series of working hypotheses regarding how the stressors might affect the ecological components of the natural environment (EPA 1992b)." Conceptual model development should identify the pathway(s) by which contaminants of potential concern may be transported to ecological receptors, and should assist in defining exposure scenarios. For example, an exposure model linking a chemical contaminant to a receptor might include processes such as partitioning of the chemical among various environmental media, chemical/biological transformation processes, and identification of potential routes of exposure (e.g., ingestion). Although many assumptions are included in the development of the conceptual model, only those assumptions and hypotheses that are most likely to contribute to the risk are usually evaluated in detail.

The intent of the ecological conceptual model is to provide a working model to illustrate ecological foodwebs and the abiotic and biotic transport pathways that will be assessed in the ecological risk assessment. Foodwebs may show energy flow in the ecosystems and provide guidance on selecting ecological receptors to meet the objectives of the risk assessment. A foodweb, in combination with an understanding of contaminant transport pathways, is used to identify potentially affected receptors.

Major habitat types at the Hanford Site include grasslands, shrublands, riparian zones, the Columbia River, and transition zones between the different habitat types. The wildlife present in each habitat type are dependent on the vegetation for food, cover, and nesting places. Many receptors are common to more than one habitat type, while others are confined to only one habitat.

Detailed foodwebs for grasslands and riparian habitats are shown in Figures C-3 and C-6 of Appendix C. The figure shows energy flow through various trophic levels. Foodwebs illustrate how a particular species may be exposed to contaminants. For example, a mule deer feeding on contaminated grass would be directly exposed to contaminants that have accumulated in vegetation. In contrast, a raptor would be exposed via the food chain to those same contaminants only after transport of the contaminant through two trophic levels (vegetation to herbivore to raptor). Both the deer and the raptor, however may be directly exposed to contaminants in drinking water.

Rooting depths of plant species and burrowing activities of animals should also be considered in the analysis of contaminant transport. Table C-2 in Appendix C shows the rooting depths of plants and burrowing depths of various insects and mammals found at the Hanford Site. This information can be used to develop site-specific exposure scenarios. Dominant plant species and plant cover must also be considered. For example, the dominant vegetation communities in the 100 B/C area are *Bromus tectorum/Salsola kali*, and *Chrysothamnus nauseosus/Bromus tectorum* (Stegen 1994). However, disturbed/nonvegetated areas with less than 5% cover are also present. These areas are routinely sprayed with herbicides. To develop a site specific exposure scenario, it is necessary to determine within the conceptual model what vegetation would be present if waste sites were not disturbed.

From the assessment of possible vegetation types, rooting depth scenarios and wildlife burrowing scenarios can be developed. These conditions can then be used to assess potential transport of contaminants through plants and animals. Exposure scenarios for the maximum exposed individual and for the average exposed individual should be developed. Another condition of the conceptual model is that the exposure scenarios should reflect current waste site conditions (limited vegetation, revegetated) as closely as practical.

The potential exposure of an animal to contaminants from a given waste site is related to the size of the animal's home range and its location relative to the location of the waste site. The degree of receptor exposure is assumed to be proportional to the amount of time the animal spends within the waste site, the amount of food consumed from within the waste site, the type of food available, and the concentration and bioavailability of the contaminants.

4.2 ANALYSIS PHASE

The analysis phase, as illustrated in Figure 4-3, consists of a technical evaluation of data concerning the receptors potential exposure to contaminants, and the subsequent effects of that exposure. This phase consists of two activities, characterization of exposure and characterization of ecological effects. The purpose of characterization of exposure is to predict or measure the spatial and temporal distribution of the stressors and their co-occurrence or contact with the ecological receptors of concern. The purpose of characterization of ecological effects is to identify and quantify the adverse effects elicited by a given concentration of a stressor and, to the extent possible, to evaluate cause-and-effect relationships (EPA 1992b).

The output of this analysis phase is an exposure profile and a stressor-response profile that are used in the next phase, the risk characterization phase. For the Hanford Site baseline risk assessment, the

scope of the analysis phase includes the impact of chemical and radionuclide contaminants on receptors, and the potential impact of changes in habitats because of contamination.

4.2.1 Characterization of Exposure

The elements of the characterization of exposure are graphically depicted in the left half of Figure 4-3. Key steps of the exposure assessment are 1) identifying the receptors and stressors that will be evaluated in the risk assessment, 2) compiling and processing all relevant information related to the distribution of the contaminants and the receptors in relation to space and time, and 3) developing an exposure profile for each receptor of interest.

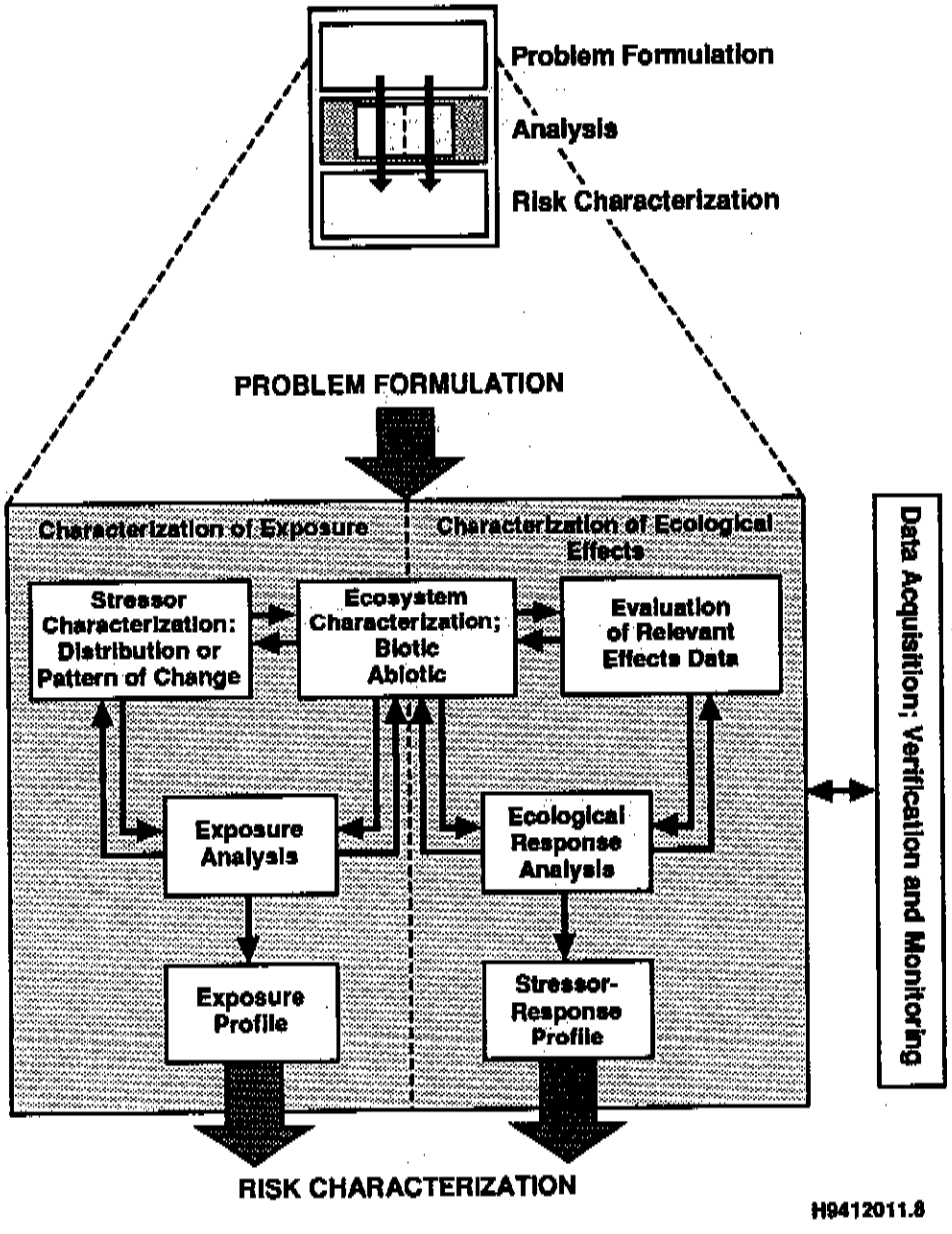
The process for characterization of exposure builds on the information developed in the Problem Formulation phase, and concludes by quantitatively estimating the exposure (or dose) of the receptors to each of the contaminants of potential concern. This exposure characterization process includes four steps; stressor characterization, ecosystem characterization, exposure analysis, and presentation of exposure profiles.

To characterize exposure, one or more exposure scenarios are developed to illustrate how the contaminants are transported from onsite media to the plants and animals, and how much of the contaminants are taken up by the receptors. The exposure scenarios, based on the conceptual model(s) developed in the Problem Formulation phase, identify the exposure routes used to estimate the risk to each receptor (e.g., breathing contaminated air, drinking contaminated water, eating contaminated food, and inadvertently consuming contaminated soil). The exposure scenarios also consider how frequently the receptors come in contact with the contaminants, and what quantities of the contaminants are taken up by the receptors. Separate exposure scenarios are developed for each receptor. Exposure scenarios should include reasonable maximum exposure scenarios and perhaps less conservative scenarios based on average contaminant concentrations and expected or known animal behavior and plant density at the site.

4.2.1.1 Stressor Characterization. Stressor characterization is the process of determining onsite concentrations of the COPC, and the scale of disturbance caused by nonchemical stressors (EPA 1992b). This process builds on the information developed in Section 4.1.1 of the Problem Formulation phase. For each operable unit, the distribution of stressors within the study area, the seasonal availability, and zones of contact between the stressors and receptors is evaluated. Also, because many of the COPC are radionuclides, the presence of shielding material between the stressor and the receptor must be considered. The solubility and transport characteristics of the specific chemicals and radionuclides are evaluated through various sources.

The concentrations or quantities of the contaminants at the waste site are determined based on field investigations (e.g., LFI), monitoring data (e.g., WHC's near-field monitoring program), or fate and transport modeling. Site-specific data will be used when available. The need for additional site-specific sampling should be determined during the Problem Formulation phase of the risk assessment, or earlier. The stressor characterization step must consider the spatial and temporal interactions between the COPC and the receptors of interest.

Figure 4-3. Analysis for Ecological Evaluations (Derived from EPA 1992b).



4.2.1.2 Ecosystem Characterization. This step in the analysis phase defines the spatial and temporal distribution of the receptors of interest, and describes the habitats that may be impacted by the contaminants. The receptors and habitats of interest are based on the conceptual model developed in the Problem Formulation phase, and food pathway diagrams for the receptors of interest. Examples of food pathway diagrams are shown in Appendix C, Figures C-2 through C-6. Site specific exposure models and food pathway diagrams will be developed for each ecological risk assessment.

The ecosystem characterization process identifies the food preferences, feeding habits, home range, reproductive cycles, and habitat requirements of each receptor of interest. This characterization process also considers if the stressor could alter normal behavior, that could in turn enhance or reduce the exposure of the receptor to the contaminant. Attributes of the ecosystem that might influence the distribution of the stressors in space or time are also considered. For example, contaminants that tend to bioaccumulate as they move through trophic levels can significantly increase the exposure of higher predators to those contaminants.

Several ecological studies have been conducted for the CERCLA program at the Hanford Site, and these studies are valuable data resources for characterizing ecosystems at the waste sites. For example, Landeen et al. (1993) and Weiss and Mitchell (1992) studied the ecosystems in the 100 Areas, Cushing (1993) provided information on the aquatic ecosystems at the 100-HR-3 and 100-NR-1 operable units, Brandt et al. (1993) reported on the biological resources at the 300-FF-5 operable unit, and Rogers and Richard (1977) studied the ecosystems at the 200 Area Plateau Waste Management Area. Ecological data are also available from the WHC near-facility monitoring program (both biotic and abiotic data) and the Pacific Northwest Laboratory site-wide environmental surveillance program (Schmidt et al. 1993; Woodruff and Hanf 1992).

4.2.1.3 Exposure Analysis. Exposure analysis considers the spatial and temporal distribution of the stressors and receptors and how they overlap. Spatial considerations include such factors as whether contaminants and receptors are present simultaneously at potential points of exposure. This requires information on both the horizontal and vertical distribution of contaminants and receptors. Temporal considerations include determining if the receptor is a seasonal inhabitant or permanent resident, and if the contaminants vary seasonally. The contaminant uptake by the species of interest is also considered in the exposure analysis.

For the Hanford Site, the spatial distribution of the receptors (including the home ranges of animals and distribution of vegetation) is evaluated to establish the point or zone of contact between contaminants and receptors. For example, if the home range for an animal of interest overlaps a waste site boundary, the animal is assumed to be exposed for at least part of the time to the site contaminants. The duration of exposure is assumed to be proportional to the fraction of the organisms home range that overlaps the waste site. For organisms whose home range falls wholly within a waste site it is assumed that 100% of their diet consists of contaminated foodstuffs. An organism also may spend time on more than one waste site. In this case, the proportional uptake of contaminants from all waste sites within the organism's home range should be calculated, and summed to estimate the total risk to the receptor.

Exposure to contaminants can occur through ingestion, by drinking water, through preening or burrowing activities, and through external exposure to radiation. For secondary consumers (predators), the major route of exposure is normally through ingestion. The dose of radiological or nonradiological chemicals received by a receptor can be calculated in many cases by using trophic level transfer coefficients. For radionuclides that emit gamma radiation, external radiation is the principal exposure route linking radionuclide contaminants to receptors. In this case, distance,

duration of exposure, and the presence of shielding material determine the amount of exposure. For alpha and beta emitting radionuclides, the principal exposure route is usually consumption of food, water, and/or soil. Example calculations for estimating the dose of radionuclides and chemicals received by receptors are given in Appendix E.

The importance of the drinking water pathway will vary, depending on the receptor. For some riparian species such as Canadian geese, water consumption is important and should be evaluated. For other receptors such as raptors, drinking water may be an insignificant exposure route. An assessment as to whether certain species are known to require drinking water is presented in Appendix C, Table C-2. Table C-2 in Appendix C also gives a summary of plant rooting depths for various species of Hanford Site plants. This information is used to determine if plants are exposed to contaminants that are located below the surface. Similarly, available information on burrowing depths of insects and animals is also provided in Table C-2.

4.2.1.4 Exposure Profile. The exposure profile quantifies the magnitude and spatial and temporal patterns of exposure for the scenarios developed during the Problem Formulation phase (Section 4.1) or the exposure analysis process (Section 4.2.1.3). Exposure profiles can be expressed using a variety of units. If chemical contaminant uptake by animals is being estimated, the exposure is usually expressed in dose units (e.g., mg/kg body wt./day). For expressing exposure at the ecosystem level, the exposure might be expressed in units of concentration/unit area/time. For habitat impacts, the exposure profile could be expressed as the percentage of habitat impacted per year, or the decrease in plant mass in kg/area/year.

The exposure profile step should also include a discussion of the uncertainties involved in estimating exposure. It is important that each of the various sources of uncertainty be evaluated and carried forward to the next phase of the risk assessment, the Risk Characterization phase.

4.2.2 Characterization of Ecological Effects

The characterization of ecological effects is based on what is known regarding the toxicity of contaminants. This assessment summarizes the data relating adverse effects in a given receptor to the concentrations of each contaminant of potential concern. The assessment quantifies the adverse effects caused by the stressor (contaminant), and relates these effects to the assessment endpoint established in the Problem Formulation phase. In general, the more complex the organism, the more sensitive the organism is to ionizing radiation. Table 4-3 presents minimum apparent safe threshold exposure levels of radiation for several classes of organisms, and ranks the general sensitivity of the types of organisms. Mammals, in general, are the class most sensitive to ionizing radiation, while crustaceans, invertebrates, and mollusks are least sensitive. No comparable relationship exists for inorganic and organic chemicals. The characterization of ecological effects (or toxicity assessment) consists of four steps; the evaluation of relevant effects data, ecosystem characterization, an ecological response analysis, and development of stressor-response profiles (Figure 4-3).

Table 4-3. Ionizing Radiation Sensitivities for Biological Organism Categories.

Organism	Source	Minimum Apparent Safe Threshold Exposure Level, mGy/d	Inferred Ordinal Class
Mammals	c	4-10	1 (most sensitive)
Reptiles	c	20	2
Fish (salmon smolts)	d	95	
Birds	c	200-1000	3
Deciduous trees (biomass)	c	240	
Shortgrass plains (biomass)	c	1700-3000	4[a]
Pond snail	d	240-1200-6000	
Old Fields (biomass)	c	5860	5[b]
Daphnia (freshwater invert.)	d	3940-4610	
Crustaceans	d	10,000-100,000	
Invertebrates[c]	c	500,000-1,000,000	6 (least sensitive)
Mollusks	d	100,000-1,000,000	

[a]It was assumed this represents monocot grasses; it was undefined in the source.
[b]It was assumed this represents dicot shrubs; it was undefined in the source.
[b]Except earthworms; found to be affected at 24 mGy/d.
[c]Terrestrial - IAEA 1992.
[d]Freshwater aquatic - NCRP 1991.
Dimensional units: 100 rad = 1 Gy (gray); 100 R = 1 Gy; 1 rad/d = 10 mGy/d.

4.2.2.1 Evaluation of Relevant Effects Data. The evaluation of relevant effects data relates the characteristics of the contaminants (chemicals and radionuclides) to ecotoxicological effects. This step includes obtaining information and data from the scientific literature and state and federal regulations, that present concentrations of contaminants, or doses, that are considered to be safe (levels that do not elicit measurable responses) to the receptors at the site. These "safe" levels are referred to as benchmarks, standards, or criteria. No observed adverse effects level (NOAEL) are a good example. These can be obtained from the literature or, if necessary, derived based on field and laboratory bioassay tests, or obtained from the literature. Guidance for calculating NOAELs are presented by Newell et al. (1987).

There are several benchmarks, standards, or criteria that have been developed for aquatic ecosystems. Benchmarks for radionuclides are given in NCRP (1991) and DOE Order 5400.5 (DOE 1990). The DOE benchmark for aquatic organisms is 1 rad/day. Benchmarks for inorganic and organic chemicals in aquatic ecosystems include EPA's *Quality Criteria for Water 1986* (EPA 1986b) and the state of Washington's Model Toxic Control Act (MTCA, 173-201A-040 Water Quality Standards-Surface Waters). The EPA criteria and the Washington State standards for inorganic contaminants applicable at the Hanford Site are shown in Tables 4-4 and 4-5. Because the toxicity of these contaminants (metals) change with water hardness, the toxicity values at water hardness of 63 mg/L (representative of the Columbia River) and 150 mg/L (representative of Hanford Site groundwater near the river) are given. Generally, the state standards are more restrictive than the EPA criteria, and the most conservative values are used in the ecological risk assessment.

There are few widely accepted benchmarks for terrestrial organisms. Terrestrial benchmarks will be obtained, when possible, from scientific reports documenting responses of receptors to contaminants based on laboratory or field experiments. Benchmarks that represent NOAELs or lowest observed adverse effects level (LOAEL) will be used when available, rather than lethal concentration at which 50% of experimental organisms die (LC_{50}) or other values that are based on more acute exposures. Also, whenever possible, data that minimize the need for extrapolations (to equate the laboratory data to the expected exposure conditions at the site) will be used. For example, results from acute toxicity tests should not be used to estimate risk under chronic exposure conditions, unless no other data are available. Opresko et al. (1993) developed benchmarks for some terrestrial animals, and Suter el al. (1993) developed benchmarks for plants.

Both of these publications, however, stipulated that the benchmarks were appropriate for screening potential contaminants to see if more studies were required, but were not necessarily appropriate for clean-up criteria or for establishing "safe" levels.

Table 4-4. Metal Criteria with Hardness Factors: Columbia River.

Metals		Hardness of Columbia River at Richland Pumphouse, ug/L[a]	Ecology: Criteria Concentrations ug/L[b]	EPA: Criteria Concentration ug/L[c]
Cadmium	acute	63	2	2
	chronic	63	1	1
Chromium III	acute	63	1189	1189
	chronic	63	142	145
Copper	acute	63	10	11
	chronic	63	7	8
Lead	acute	63	31	44
	chronic	63	1	2
Nickel	acute	63	982	967
	chronic	63	109	108
Zinc	acute	63	70	79
	chronic	63	64	72

[a] Woodruff and Hanf (1993).
[b] WAC 173-201A-040.
[c] EPA (1986a) and 57 FR 246, part 131, Water Quality Standards.

Table 4-5. Metal Criteria with Hardness Factors: Groundwater.

Metals		Hardness of Groundwater ug/L*	Ecology: Criteria Concentration ug/L	EPA: Criteria Concentration ug/L
Cadmium	acute	150	5	6
	chronic	150	1	2
Chromium III	acute	150	2420	2420
	chronic	150	288	294
Copper	acute	150	22	26
	chronic	150	14	17
Lead	acute	150	94	134
	chronic	150	4	5
Nickel	acute	150	2078	2019
	chronic	150	231	224
Zinc	acute	150	147	165
	chronic	150	133	149

*Dirkes (1990).

4.2.2.2 Ecosystem Characterization. The ecosystem characterization step in the ecological/toxicity assessment process is the same as the ecosystem characterization step conducted for the exposure assessment (see Figure 4-3 and Section 4.2.1.2). That is, the receptors of interest have to be identified and characterized in order to determine which species are used to determine relevant effects.

4.2.2.3 Ecological Response Analysis. The ecological response analysis is used to relate the magnitude, frequency, and duration of the exposure to the magnitude of the response. The ecological response analysis also describes the relationship between the measurement and assessment endpoints, and evaluates the strength of association between these two endpoints. This dose/response step also lists the assumptions used for the toxicity/ecological effects assessment, discusses the extrapolations used to estimate dose/response values, and presents cause-effect information when possible. Proof of causality, however, is not required in a CERCLA ecological risk assessment (EPA 1992b). The selection of appropriate experimental dose/response analyses used in toxicity tests are important for estimating responses of receptors at the site.

It will often be necessary to make dose-duration adjustments as well as extrapolations between species (often between species of differing genera, families). The EPA (1992b) identifies commonly used extrapolations as follows:

- Extrapolation between taxa
- Extrapolation between responses
- Extrapolation from laboratory to field
- Extrapolation from field to field.

The indirect effects of the contaminants, such as interspecies interactions (competition, disease), or trophic-level relationships (e.g., predation) are also considered in this analysis, when possible. However, these data are generally not available.

4.2.2.4 Stressor-Response Profile. The stressor-response profile describes the stressor-response relationship and summarizes the characterization of the toxicity/ecological effects assessment process (EPA 1992b). Because the ecological risk assessment addresses chemical and radionuclide stressors released from the site (i.e., contaminants of potential concern), the following discussion focuses on the dose/response profile of specific contaminants.

The output of the toxicity assessment is a contaminant dose/response profile. In most instances this will not be in the form of a 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 (e.g., magnitude, duration, frequency, and timing of exposure in the study setting should be related to the magnitude of effects) that are often required to apply the data to the endpoint of interest. Most frequently, the contaminant dose/response profile will consist of a conservatively-derived value that estimates either a reference dose, or a concentration of contaminant that is expected to result in no significant impact to the endpoint in question.

Similar to the exposure profile step (Section 4.2.1.4), the stressor-response profile step should include a discussion of the uncertainties involved in estimating the toxicological effects of the contaminants. This should include discussing the extrapolations used to estimate toxic benchmarks, and the assumptions used to develop the stressor-response profiles.

4.3 RISK CHARACTERIZATION

The risk characterization phase evaluates the likelihood of an adverse effect to receptors affected by the stressors identified. The risk characterization phase of the ecological evaluation is depicted in Figure 4-4. The risk characterization phase consists of two steps, risk estimation and risk description.

4.3.1 Risk Estimation

4.3.1.1 Integration of Stressor-Response and Exposure Profiles. The likelihood of an adverse effect to a receptor species is frequently estimated by integrating the results of the exposure profile (from the characterization of exposure step) and the stressor-response profile (from the characterization of ecological effects step). This can be accomplished by dividing the estimated dose received by the onsite receptor (the exposure profile) by the "safe" exposure level (the stressor-response profile) for that receptor. This ratio is referred to as the environmental hazard quotient (EHQ).

In this manner, the exposure profiles and contaminant dose/response profiles for COPC are compared, and the results extrapolated to determine whether or not a significant potential exists for an adverse impact to the receptor. An indication of such an impact exists when the exposure profile exceeds the dose/response profile (Barnthouse et al. 1986). Those contaminants of potential concern that are associated with such an impact potential are then regarded as contaminants of concern.

The EHQ is the exposure concentration (or dose) divided by a benchmark concentration (or dose) established by the dose/response profile. A benchmark is usually a toxicity value that is protective of the receptor(s) as discussed in the previous section. If the EHQ exceeds unity (> 1), then the

potential for an adverse impact to an individual organism is assumed to exist. The degree of potential impact is further evaluated by taking into consideration such factors as the degree of habitat contamination (e.g., the areal proportion of a given habitat that is, or is expected to be, contaminated) and the degree of confidence that the benchmark value is appropriate for the site.

In the case of ionizing radiation from a radionuclide, where the benchmark is 1.0 rad/day:

$$\text{EHQ} = \frac{\text{organism's dose}}{\text{benchmark}} \qquad 10$$

For nonradiological chemicals, an acute or chronic toxicity value, regulatory criteria, or the NOAEL is used to assess risk and serve as the benchmark for calculation of EHQs. An EHQ at or above 1 (exceeding or meeting the benchmark) would indicate a potentially measurable risk. For nonradiological chemicals, the EHQ is calculated by dividing the concentration of the contaminant at the exposure point by the benchmark value (expressed as a concentration). If the contaminant is taken up through ingestion of food or soil, then the dose rate rather than the concentration is used. For more information concerning the exposure equations see Appendix E.

$$\text{EHQ} = \frac{\text{contaminant concentration}}{\text{benchmark}} \qquad 11$$

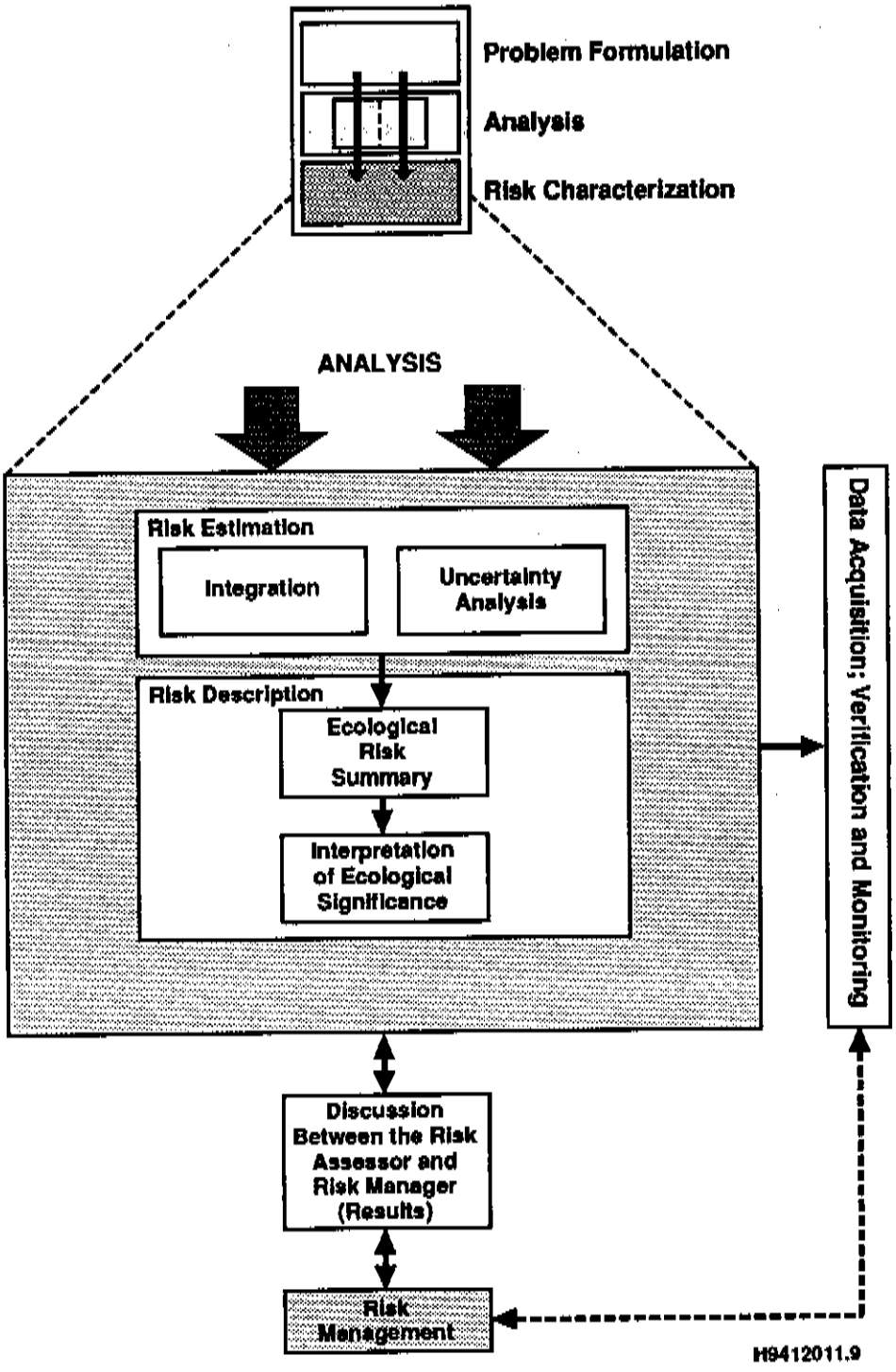
The risk characterization process should also address the potential for cumulative toxic effects. Each contaminant of potential concern should be evaluated with respect to its toxicity mechanism. For example, if two contaminants are known to affect an assessment endpoint in a similar manner, the effects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

Ecological risk associated with radionuclides is estimated by calculating the total dose from ionizing radiation. Each radionuclide in the environment contributes to the total dose. Therefore, the cumulative effect of all the radionuclides is accounted for in the initial calculation (Appendix E).

Risk estimations may also be made by comparing the distribution of an effect parameter to the distribution of an exposure parameter. The risk in this case is quantified by evaluating the degree of overlap (Barnthouse et al. 1986). This approach requires a larger database than the EHQ approach, and is a probabilistic approach.

Risk estimations can also be conducted using simulation modeling. Again, this approach requires a relatively large database, and, when possible, empirical field data should be used to validate the model. Simulation modeling can be used to estimate the transport of contaminants through a food web to a receptor, to estimate changes in the population of a receptor species that is under stress, or to predict plant succession in a contaminated area.

Figure 4-4. Risk Characterization for Environmental Evaluations (EPA 1992b).



4.3.1.2 Uncertainty. An important element of the risk characterization process is the uncertainty analysis. This is particularly true for ecological evaluations where methods and databases are not standardized to the extent of the human health evaluation methods and databases. The uncertainty assessment for an ecological evaluation can range from a purely qualitative assessment, based on best professional judgement, to a quantitative assessment employing probabilistic simulation modeling. The type of uncertainty assessment best suited for a particular ecological evaluation depends on the extent of the available data, the severity of the risk, and the outcome of the evaluation itself.

The EPA (1992b) points out four categories of uncertainty that may be relevant to an ecological risk assessment as follows:

- Conceptual model formulation — incorrect assumptions made during conceptual model development
- Information and data — incompleteness of data or information
- Stochasticity (natural variability) — inherent in stressor characteristics, ecological components and, the factors that influence their distribution
- Error — introduced through experimental design or the sampling/measurement procedures.

The EPA (1989a) recommends that biotic pathway assessments (within the context of both the human health and the ecological evaluation) be regarded, at best, as semiquantitative because of a current lack of understanding of the general theoretical relationships among contaminants, ecosystems, and biological species. The EPA also notes that biotic pathway assessment errors of up to three-to-four orders of magnitude should be anticipated. Two approaches are suggested for uncertainty analysis. The first would be a simple qualitative approach that provides an overall general statement of the degree of uncertainty based on an evaluation by the risk assessors performing the ecological assessment. The second would consist of an entirely independent analysis using available quantitative and qualitative information relevant to the four categories of uncertainty pointed out by the EPA (1992b), above. The two independent analyses should agree with one another if they have been performed correctly.

The EPA (1989a) 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
- 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 uncertainty analysis discussed above is a limited approach. The alternative is to conduct a probabilistic risk assessment from the start of the evaluation. In this manner, the resulting risk distribution curve not only provides variance information, it also provides for a much more reasonable estimate of risk or likely future risk. For most ecological evaluations at the Hanford Site a semi-quantitative uncertainty assessment is the most feasible approach.

4.3.2 Risk Description

The ecological risk description consists of two steps, the ecological risk summary component presented in Subsection 4.3.2.1, followed by the interpretation of ecological significance in Subsection 4.3.2.2.

4.3.2.1 Ecological Risk Summary. The final portion of the risk characterization process involves documenting the evaluation results for the project decision-makers to allow a decision on relevant issues in a risk-based manner. Risk estimation and uncertainty (through all phases of the evaluation) are summarized. Risk predictions that are testable are documented. Ecological risk endpoints are expressed in terms of assessment endpoints and their direct-management relevance.

A discussion of the weight-of-evidence provides additional input for determining the level of confidence in the assessment. Factors useful in the discussion of the weight-of-evidence are the completeness and quality of data, supplementary information relevant to the final conclusion, and the degree of correlation between the presence of a contaminant and some significant adverse effect. Identification of additional analyses or data that may reduce the uncertainty should also be discussed in the ecological risk summary.

4.3.2.2 Interpretation of Ecological Significance. This step explains the ecological implications of the condition(s) at a site to the decision-maker. In addition to the potential for toxicological effects, ecologically relevant topics such as physical alterations to the habitat from past activity (e.g., construction, farming) and the potential for ecosystem and habitat recovery following any impact, should be explained. Interpretation of ecological significance of the conclusion is necessary because it provides the critical link between the risk estimation and the communication of the results. Several aspects of ecological significance that potentially influence the interpretation of the results (EPA 1992b) are the following:

- The nature and magnitude of the effects
- Spatial and temporal patterns of the effects
- Recovery potential.

If significant changes occurred or are predicted in the assessment or measurement endpoints, it is necessary to discuss the nature and magnitude of the effects associated with each habitat and receptor. For example, the spatial distribution of the stressor can have devastating effects if the area affected is a critical resource for a certain species. With respect to temporal distribution, the persistence of the stressor and the seasonal occurrence of the receptor define the degree of exposure and potential risk. Finally, the nature of the contaminants, the duration of exposure, and the extent of the adverse impacts will effect the evaluation of recovery potential. Additional analyses may be required to investigate the potential recovery once the stressor is removed. The *Risk Evaluation of Remedial Alternatives for the Hanford Site* (DOE-RL 1994a) should be consulted for details regarding use of ecological evaluations for comparing remedial action alternatives.

5.0 QUALITATIVE RISK ASSESSMENT METHODOLOGY

This chapter of the HSRAM provides the general methodology to perform qualitative risk assessments in support of the interim remedial measures pathway of the *Hanford Past-Practice Strategy* (DOE-RL 1992a) as discussed in Section 1.4.2. It was intended that a qualitative risk assessment would provide a characterization of site risk to allow the Tri-Party Agreement representatives to evaluate the risk posed by selected waste sites identified in operable unit work plans and determine whether the risk warrant keeping the waste sites on the pathway for cleanup using interim remedial measures. Qualitative risk assessments were not intended to replace the need for a baseline risk assessment nor to serve as a basis for establishing preliminary remediation goals.

5.1 INTRODUCTION

A qualitative approach to risk assessment is a process to enable one to think usefully about a problem and associated remedies, to develop a robust argument for action, and to make use of quantitative information that may only partially characterize contamination at a site. The QRA is only one of the tools used in the HPPS. All sites, whether selected for an IRM or not, will be evaluated as part of a comprehensive baseline risk assessment prior to a final remedy selection for an operable unit or aggregate area.

The QRA evaluates risk for high-priority waste sites at an operable unit as identified in the work plan for that operable unit. The fundamental requirement for performing a QRA is that sufficient information be known from which a risk calculation and/or risk rating can be made.

When sampling data shows the concentration of contaminants at a site, the QRA provides a limited, quantitative evaluation of human health and ecological risk. Based on the results of this quantitative evaluation, a qualitative assessment of the potential risk associated with each site is made. A discussion of the uncertainties associated with the sampling data, exposures, and contaminant toxicity is also included. For some sites, site-specific sampling data may not be available. In these cases, the QRA presents a qualitative assessment of site risk based solely on an evaluation of historical site records, process information, and other available and pertinent information.

The approach for conducting a QRA includes the following general procedures.

- Collect, interpret, and review information for a site such as location, generation process, source characteristics, sampling data, and other pertinent information. In some cases, this may include information obtained from analogous sites.
- When data are available, perform a limited human health and ecological quantitative risk assessment for predefined exposure scenarios and indicator organism, respectively.
- When data are unavailable, a qualitative assessment should be made for ecological risk, as well as for human health risk.
- Provide project managers with a qualitative assessment of potential site risk, including the degree of confidence in the human health and ecological risk assessments.

Because of methodological differences between the human health and ecological portions of the QRA, the sections of this chapter do not directly correspond to these bulleted procedures. Section 5.2 pertains to both human health and ecological portions of the QRA because identical sources of data are used for both portions. The methodologies for the human health and ecological portions of the QRA are discussed separately in Sections 5.3 and 5.4, respectively.

5.2 DATA EVALUATION AND CONTAMINANT IDENTIFICATION

Site information should be reviewed to identify the nonradioactive and radioactive contaminants that may be present in the key media (e.g., soil, groundwater, surface water, air, or biota). This information includes process knowledge, disposal knowledge, records of inventory, historical information, information obtained during site reconnaissance, and LFI data. Historical and LFI data are generally the only data available with which to quantify environmental contaminant concentrations. The protocol for evaluating these data is described in Sections 5.2.1 and 5.2.2. Data collected from analogous sites or similar categories of sites (e.g., cribs associated with the reactor sites) may also be used to identify contaminants present at a site. Section 5.2.3 discusses how the various sources of information are combined to develop a site contaminant list.

5.2.1 Evaluation of Historical Data

Dorian and Richards (1978) is a primary source of historical radiological soil data for 100-Area operable units. Operable unit work plans often provide convenient summaries of these historical data for use in a QRA. These historical data, when available, provide waste site information that is compared to LFI data to support identification of possible contaminants for consideration in the QRA, and to support the characterization of the risk for the high-priority waste sites. Dorian and Richards data are considered of lower quality than LFI data. Although standard laboratory methods were used in the sample analyses, the historical data were not validated according to current EPA guidelines. Radionuclide concentrations in historical data must be corrected for radionuclide decay prior to use in the QRA.

5.2.2 Evaluation of LFI Data

The LFI data available for each operable unit commonly include information from samples collected from boreholes and test pits in areas both within and outside of the high priority waste sites specified in the work plan. Sampling and field activities for an operable unit are usually summarized in a "Description of Work" report for each specific operable unit.

Quantitative site information includes both historical data and data generated from LFI sampling activities. When historical and/or LFI sample data exist, these data are the primary source for identification of contaminants at a high-priority waste site. Qualitative site information (e.g., process knowledge, records of inventory) should still be evaluated even when sample data exist, particularly when data quality is not high or when the site is inadequately characterized. Discrepancies between the list of contaminants identified in limited site sampling and constituents expected to be present at a site based on historical and process knowledge may affect the level of confidence associated with the data.

When data are unavailable for a waste site, an attempt should be made to identify an analogous site. An analogous site is one for which identical types and similar concentrations of contaminants are

anticipated based on process knowledge and site conditions. If sample data from an analogous site have already been collected and evaluated in a previous QRA, the risk rating associated with the analogous site may be applied to the site in question (see Section 5.3.4.1). Contaminants identified at an analogous site may be presented as an indication of the contaminants and the associated risk expected at the site in question; however, the analogous site data itself should not be presented in the QRA.

5.2.3 Identification of Contaminants

For a QRA, contaminants are identified by evaluation of site information, historical data and LFI data. While the general concepts of the contaminant identification process described in Section 2.2 and depicted in Figure 2-1 are followed, much more latitude is allowed in selection of the information to be used in a qualitative evaluation of risk. When data are not available, a qualitative evaluation of risk may be presented based on analogous site information. The source and nature of all information used in a QRA must be clearly identified and its uncertainty discussed.

For a soil operable unit, sample data from within the depth range of 0 to 4.6 m (15 ft) from the surface is used for identification and characterization of contaminants. This represents a reasonable estimate of the depth from which soil may be excavated and re-distributed as accessible surface soil as a result of potential site development activities such as home construction. Contaminants in soil below 4.6 m (15 ft) are evaluated solely for potential threat to groundwater.

5.2.3.1 Consideration of Hanford Site Background Data. Currently, background reports are available for nonradioactive soil analytes (DOE-RL 1993a) and nonradioactive groundwater analytes (DOE-RL 1992b). A report of background for radioactive soil analytes is in preparation. Because organic chemicals are not naturally occurring at Superfund sites, organics are not compared to background concentrations (EPA 1989b). Section 2.2.2 provides a discussion on selection of an appropriate statistic to represent background constituent concentrations.

The Hanford Site background data should be reviewed to ensure that it is appropriate for the data being screened. Hanford Site background should not be used if the site-wide background information is too general for a specific natural condition of the project. For example, the Hanford Site background data report has identified soils in three terrestrial ecosystems that show distinctly higher concentrations for many analytes. These three soil association types are: (1) highly alkaline soils of playa and ephemeral drainages, (2) riparian ecosystem soils, and (3) the grassy soils on Rattlesnake Mountain (DOE-RL 1993a).

Hanford Site background data are not currently available for all potential waste site parameters that are detected during project sampling. In this case, the waste site parameters should be evaluated by performing the preliminary risk-based screening as described in Section 3.1.2 using the maximum detected concentration of those parameters.

Project-specific background distributions, if used for screening contaminants, must be reviewed for the intended application by operable unit managers prior to use in the risk assessment process. Project-specific background distributions can be valuable by providing information for site characterization and site evaluation to ensure remedial efforts are directed to the source of the contamination. Procedures for use in calculating project-specific background distributions from sampling data are discussed in Section 2.2.3.

5.3 HUMAN HEALTH EVALUATION FOR THE QRA

The human health evaluation for the QRA follows the same general methodology described in Section 3.0. What distinguishes the QRA is the scope of the risk assessment with regard to site characterization and the number of exposure scenarios and pathways evaluated. Because of these limitations, the risk estimates in a QRA are not presented directly but are used to assign a qualitative risk rating to each high-priority waste site. Sections 5.3.1 and 5.3.2 pertain exclusively to sites for which sample data are available.

5.3.1 Identification of Contaminants of Potential Concern and Exposure Concentrations

Following contaminant identification (discussed in Section 5.2.3), the same methodology presented in Section 3.1 is used to identify COPC in the QRA. In general, both the historical and LFI data are considered for identification of COPC. COPC are defined as contaminants that are retained for further evaluation in the QRA following both comparison to a background concentration value (see Section 5.2.3.1) and preliminary risk-based screening. The preliminary risk-based screening for the QRA is conducted using exposure parameters for the residential scenario at an lifetime incremental cancer risk (ICR) of 1E-07 and an HQ of 0.1.

The highest concentration of a contaminant from either the historical or LFI data sets is selected for comparison to risk-based concentrations in the preliminary risk-based screening. This maximum detected concentration is also used as the exposure concentration in the exposure assessment. The maximum detected concentration, rather than a statistic of the distribution of contaminant concentration, is used in the QRA because of the generally minimal number of site samples available for the risk evaluation. Radionuclide concentrations should be corrected for radioactive decay before comparison to risk-based screening concentrations.

5.3.2 Exposure Assessment

The exposure assessment for the QRA is conducted in a similar manner to that described in Section 3.2. The exposure assessment includes the determination of exposure scenarios, exposure pathways, exposure parameters, and the quantification of exposures. The scenarios and pathways for the QRA have been discussed and selected by the 100 Area Tri-Party Agreement unit managers. The components of the exposure assessment methodology are individually discussed in Sections 5.3.2.1 through 5.3.2.4.

5.3.2.1 Exposure Scenarios. Exposure scenarios define the conditions in which receptors are exposed to COPC. The QRA evaluates current human health risk for two exposure scenarios defined as frequent-use and occasional-use. These exposure scenarios use exposure parameters that are identical to those presented in Appendix A for the residential and recreational exposure scenarios. Within the context of the QRA, however, these exposure parameters do not define a particular land-use setting but are used to represent bounding estimates of potential site risk. The exposure scenarios are described as "frequent-use" and "occasional-use" to emphasize this bounding quality.

5.3.2.2 Exposure Pathways. The pathways that are evaluated in the human health portion of the QRA are a subset of those discussed in Section 3.2.3.2. The pathways that are evaluated in a QRA include the following:

- Soil ingestion
- Fugitive dust inhalation
- Inhalation of volatile organic compounds (VOC) from soil
- Water ingestion
- External radiation exposure from soil.

In general, evaluation of these pathways is sufficient for the purposes of a QRA. However, additional pathways, such as dermal absorption from water and soil, inhalation of VOCs from indoor water use, and ingestion of contaminated biota, may be also be evaluated if site information suggests that this is appropriate.

Not all of the exposure pathways identified in the bullets may be appropriate for a specific QRA. In particular, QRAs for a groundwater operable unit may include ingestion of water, as well as some of the additional pathways identified in the previous paragraph, but none of the other bulleted pathways. Conversely, QRAs for a source operable unit are unlikely to include ingestion of water.

Modeling of contaminant transport to an off-site receptor will generally not be conducted as part of the QRA for a site. Fate and transport modeling is more appropriate to the level of investigation associated with a baseline risk assessment in support of an RI/FS for a site, operable unit, or aggregate area. If the risk assessor chooses to conduct transport modeling in the QRA, 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 for the site, and the required accuracy of the results.

5.3.2.3 Exposure Parameters. As discussed in Section 5.3.2.1, the exposure parameters for the frequent- and occasional-use exposure scenarios are identical to those of the residential and recreational exposure scenarios, respectively. Exposure parameters for the residential and recreational exposure scenarios are provided in Appendix A.

5.3.2.4 Quantification of Exposure. Quantification of contaminant exposure should be conducted as described in Section 3.2.5. Detailed equations used for estimating contaminant intakes are provided in Appendix D. Evaluation of future scenarios requires that radionuclide concentrations be decay corrected before calculating intake values. It is not always necessary to evaluate future scenarios in a QRA because the IRM pathway focuses on current risk and prioritization of remediation efforts.

5.3.3 Toxicity Assessment

The general procedures for toxicity assessment are presented in Section 3.3. The toxicity assessment for the QRA should include identification of contaminant-specific toxicity factors and a brief discussion of the key toxicities associated with the COPC. The toxicity assessment, as with the remainder of the QRA, should include sufficient information on the COPC to assist project managers in reaching decisions on IRMs, but does not need to be an exhaustive evaluation of all potential toxicities.

5.3.4 Risk Characterization

The methodology for the risk characterization in the QRA differs depending on whether sample data were available for conducting a limited quantitative risk evaluation. Therefore, risk characterization is discussed separately for each situation in Sections 5.3.4.1 and 5.3.4.2. A discussion of uncertainty associated with risk characterization is provided in Section 5.3.4.3.

As described in Section 5.2.3, site risk may be evaluated for sites lacking sample data by comparison to an analogous site at which data has been collected. Although the sample data from the analogous site are not considered transferable to the site in question, the qualitative risk rating associated with the risk estimate is. Therefore, it is preferable to identify an analogous site that has already been evaluated in a previous QRA.

5.3.4.1 Risk Characterization When Quantitative Data Are Available. When contaminant concentrations can be quantified, risk characterization is conducted as described in Section 3.4. The risk characterization in a QRA should include the following:

- Calculation of contaminant-specific ICRs and HQs, as described in Section 3.4.1 and 3.4.2, respectively (equations and examples are provided in Appendix D)
- Calculation of total risk from contaminant-specific risk
- Qualitative discussion of the calculated contaminant-specific and total risk with respect to the following levels:

  Contaminant-specific ICR = 1E-06
  Contaminant-specific HQ = 1
  Total ICR = 1E-06
  HI (i.e., sum of HQs) = 1

- Categorization of site risk using the following relative risk ranking categories:

  High (ICR > 1E-02)
  Medium (ICR = 1E-02 to 1E-04)
  Low (ICR = 1E-04 to 1E-06)
  Very Low (ICR < 1E-06), and

- Qualitative discussion of the uncertainty associated with the risk estimates (see Section 5.3.4.3).

5.3.4.2 Risk Characterization When Quantitative Data Are Not Available. For sites where sampling data are not available, the risk characterization should focus on a qualitative discussion of the threat posed by the site and the confidence in the information available to assess the threat. All available sources of information for a high-priority waste site should be reviewed to identify chemicals or radionuclides that may impact the key exposure media (e.g., soil, groundwater, surface water, or air). Qualitative sources of information include process knowledge, disposal knowledge, records of inventory, historical information, and information obtained during site reconnaissance.

After compiling a list of potential site contaminants and likely exposure media, the risk assessor should calculate risk-based concentrations (RBC) for these contaminants and media for a frequent-use scenario and occasional-use scenario at a HQ of 1 and an ICR of 1E-06. The risk characterization is

performed by evaluating the likelihood that a potential contaminant is present at the site at concentrations exceeding its RBC. This comparison of estimated concentrations to RBCs is a means of putting qualitative site information into perspective where risk may be evaluated. For example, if historical data indicate that very high concentrations of contaminants may be present in the soil, the calculation of an RBC provides a basis for identifying concentrations that are of concern.

Exposure parameters for calculating RBCs for a frequent-use scenario or occasional-use scenario are provided in Appendix A (for residential scenario and recreational scenario, respectively). Detailed equations for calculating RBCs for exposure media and pathways of concern in a QRA are provided in Appendix D.

The risk characterization should include a qualitative discussion of the available site information, the results of the RBC calculations, and the potential threat posed by the site. The qualitative risk rating for a site where sample data are unavailable should be characterized as either high, medium, or low.

5.3.4.3 Uncertainty in the Risk Characterization. Uncertainty in the results of a QRA is expected to be high because of the methodology used in the risk evaluation. The relatively high degree of uncertainty is reflected in the use of bounding scenarios in the exposure assessment and the use of qualitative risk rankings to express the results of the QRA. The types of uncertainties associated with the QRA, however, are identical to those discussed in Section 3.4.3. These areas are as follows:

- Site-specific uncertainty
- Uncertainty in the exposure parameters
- Uncertainty in the toxicity values
- Uncertainty in the risk characterization.

For the QRA, a discussion of the uncertainty in the site-specific information is especially important. For sites where sample data are available, uncertainty associated with the nature and extent of contamination is critical. Data gaps should be identified and the potential impacts of the absent information on the QRA should be discussed. An evaluation of the sample data with regard to anticipated levels of contamination based on historical and process information should also be performed. Additional uncertainties associated with the use of an analogous site in a QRA must be addressed as well. For sites where sample data are unavailable, the discussion of site-specific uncertainty should include an evaluation of confidence associated with historical site information, process information, monitoring data, or other available information used in the QRA.

Uncertainty in the exposure assessment is addressed in part by the use of frequent- and occasional-use exposure scenarios as bounds. Discussion of uncertainties in the exposure assessment should include an evaluation of the likelihood that potential exposure conditions are in fact bounded by these scenarios. The risk assessor should utilize the discussion points presented in Section 3.2.6 to address the key factors contributing to uncertainty in the exposure assessment.

Uncertainty associated with contaminant toxicity can be divided into two basic categories: uncertainty associated with the accuracy of a particular toxicity value for a stated target effect, and uncertainty associated with toxic effects uncharacterized by the toxicity value. A discussion of the factors contributing to uncertainty in the toxicity values is presented in Section 3.3.6.

Uncertainty in the risk characterization is naturally a function of the uncertainties in each of the three areas discussed that contribute to the numerical risk estimate. An additional uncertainty is introduced during risk characterization by the evaluation of site-wide risk as an additive function of individual contaminant risk. Section 3.4.3.4 provides guidance for discussing uncertainty in the numerical

characterization of site risk. The relative risk rankings employed in the QRA are also affected by the uncertainties discussed in this section, particularly the risk rankings for sites where sample data exist because these risk rankings are directly linked to the numerical risk estimate.

Uncertainty in the qualitative risk rating for sites without sample data is expected to be higher than for sites where sample data exist. Uncertainties associated with the calculation of RBCs include uncertainties associated with exposure parameters and toxicity values as discussed above. The site-specific uncertainty, however, is subjective and cannot be readily described. Furthermore, the subjective nature of the risk classification increases the degree of uncertainty.

5.4 ECOLOGICAL EVALUATION FOR THE QRA

The ecological evaluation for the QRA is an abbreviated version of the baseline ecological risk assessment described in EPA's Framework Document for ecological risk assessments (EPA 1992b). The QRA includes the same three basic phases as a baseline ecological risk assessment; Problem Formulation, Analysis, and Risk Characterization. However, the number of biological receptors and the number of exposure routes to each receptor are limited to fewer than would be considered in a baseline risk assessment. This allows the QRA to be conducted with a more limited database, and with less extensive analyses than a baseline assessment. Because the main objective of the QRA is to compare the relative risk at several sites (to see which sites remain on an IRM pathway) rather than to estimate the potential risk posed by the site, the QRA does not require the full range of biological receptors and exposure routes that would normally be considered in a baseline risk assessment.

The Hanford waste sites have been placed into two categories of operable units for performing QRAs. The source operable units include the contaminated soil and solid waste sites that act as the "sources" for the contaminants that occur in the ecosystems at the Hanford Site. The source operable units are considered "terrestrial" ecosystems, and only terrestrial organisms and contaminants in soils and solid waste are considered in the QRA. The groundwater operable units (the second category) include the groundwater beneath the source operable units that can transport the contaminants to other areas of the site, or to the Columbia River. The groundwater operable units are considered "aquatic" ecosystems for the purpose of the QRA because the groundwater influences biological receptors only after it surfaces in springs, or enters creeks, or enters the Columbia River via springs or subsurface flows. Aquatic (e.g., fish and algae) and riparian (e.g., ducks and cattails) organisms and contaminants in groundwater are considered in the QRA. The biological receptors and exposure pathways used in the QRA for the source sites (terrestrial) are, therefore, different than the biological receptors and exposure pathways considered in the groundwater sites (aquatic).

5.4.1 Problem Formulation

The problem formulation phase of the QRA identifies the environmental contaminants (stressors), the ecosystems potentially at risk (receptors), and the potential ecological effects of the contaminants (Figure 4-2). The problem formulation phase also defines which biological endpoints will be used to assess risk, and presents a conceptual exposure model for the area being studied.

5.4.1.1 Stressor Identification. A stressor is any physical, chemical, or biological entity that can induce an adverse response. The stressors addressed in the QRAs at the Hanford Site are limited primarily to inorganic chemicals and radionuclides. Organic chemicals are considered if they are present, but have been found at relatively few waste sites. The stressors, or contaminants of potential concern, are identified using LFI sampling data, HEIS data, historical reports, and survey

reports (Sections 2.0 and 4.0). At many waste sites in the 100 Areas, the primary data sources for the QRA are LFI sampling data and historical data from the Dorian and Richards (1978) report on radionuclide contaminants at the Hanford Site.

For the source operable units, the contaminants of potential concern are identified using soils data from the waste sites and from the area immediately around the waste sites. For the groundwater operable units, the contaminants of potential concern are identified using data from near-river wells. Analyses of water from the near-river wells is used as a conservative indicator of the water quality of groundwater as it enters the Columbia River via springs or subsurface flows.

The concentrations of the inorganic chemicals and radionuclides in the soil and groundwater samples are evaluated to identify the maximum representative value for each of the constituents, in each media. The maximum representative value is defined in Section 2.0. The maximum representative value for each constituent is then compared to the Hanford Site background data, and if the maximum representative value exceeds the background range (95% upper tolerance level, see Section 2.0), the constituent is considered to be a contaminant of potential concern. In addition to the comparison to background, the soil and groundwater data are screened to identify those constituents that are essentially nontoxic to biological receptors under typical environmental exposure scenarios (e.g., constituents such as calcium and sodium). These may be removed from the list of COPC with appropriate documentation of this decision, based on their concentrations, the scientific literature, and site-specific conditions.

5.4.1.2 Receptor Identification. The Great Basin pocket mouse was selected as the representative receptor for the terrestrial waste sites (source operable units). This species is relatively common in the terrestrial ecosystems in the area, and has a home range that is comparable to the size of many of the waste sites. Therefore, an assumption was made for assessing risk, that this species lives within the waste site and obtains all of its food from within the waste site. The Great Basin pocket mouse lives in subsurface burrows and comes to the surface at night to feed primarily on plant seeds. This animal, therefore, theoretically lives in close proximity to the COPC at the waste sites. The risk to a Great Basin pocket mouse is estimated assuming that soil contaminants are taken up by plants, incorporated into plant tissue (including the seeds), and the seeds are subsequently eaten by the mouse. The food pathway is assumed to be the major exposure route. The mouse is also exposed to ionizing radiation from the radionuclides in the soil, but this is considered to be a minor pathway. The equations used for estimating the dose received by the mouse, for both radionuclides and inorganic chemicals, are presented in Appendix E. The Great Basin pocket mouse is the only biological receptor used in the QRA for the terrestrial sites.

To evaluate the potential risk of inorganic chemicals and radionuclides in groundwater operable units, several species were selected as representative receptors. These species were selected in order to assess impacts on species that live in the Columbia river, as well as species that live in the riparian habitats along the river (and eat plants and animals living in the aquatic ecosystem). The representative species used for estimating ecological risk in groundwater operable units include fish and aquatic invertebrates living in the river and ducks and herons living in the riparian habitats along the river. The food pathway is assumed to be the major exposure route. For the aquatic invertebrates and fish, a second principal pathway is the uptake of contaminants directly from their water environment across permeable membranes such as the gills. For assessing risk to the aquatic and riparian receptors, exposure to ionizing radiation was considered to be a minor pathway. The equations for estimating the doses received by fish, aquatic invertebrates, ducks, and herons from both radionuclides and inorganic chemicals are presented in Appendix E.

5.4.1.3 Endpoint Selection. For the purposes of an ecological risk assessment, an endpoint is an observed effect on a biological component of the ecosystem that may be affected by exposure to a stressor (Suter 1991). At the Hanford waste sites, the stressors are the inorganic chemicals and radionuclides present in the soil or water. For the QRAs, the endpoint is the toxicity of individual inorganic contaminants and/or the impact of the total radiation dose to the receptor. Endpoints for ecological risk assessments, and the distinction between assessment and measurement endpoints, are discussed by EPA (1989h).

For assessing the impact of radionuclides, the total dose received by a receptor (calculated using the equations in Appendix E), is compared to a dose of 1 rad/day, based on DOE Order 5400.5 (DOE 1990) and the International Atomic Energy Agency (1992). A total dose less than 1 rad/day is considered to be below the level that would cause measurable impacts to the receptors at the waste sites. For inorganic chemicals, the dose received by the receptor (estimated using the equations in Appendix E) is compared to the benchmark for the contaminant in question. If the estimated dose exceeds the benchmark, it is assumed that a toxic effect to the individual organism is possible. The benchmark values are usually acquired from the literature and represent concentrations or doses that are estimates of NOAEL, LOAEL, or other measures of chronic toxicity based on laboratory experiments or field studies. For aquatic organisms, EPA's ambient water quality criteria for the protection of aquatic organisms are frequently used (EPA 1986b). The state of Washington's MTCA surface water standards are similar benchmarks, specified at the state level.

5.4.1.4 The Conceptual Model. For the QRA, the conceptual model assumes the receptors are exposed to the contaminants via a limited number of pathways, essentially using only those pathways that are considered to be the major routes of exposure. In general, the conceptual models developed for the QRA assume that the primary route of exposure is via the food pathway. Contaminants in the soil or groundwater are taken up by plants (the primary producers) rooted in the soil or living in the water, and the contaminants are transferred through the food chain by animals that eat the plants (herbivores), and are subsequently by animals that prey on the plant-eating animals (see Appendix C for more information on the plants and animals in the area and the typical food webs).

The QRA for the source operable units evaluates the risk to a small herbivore, the Great Basin pocket mouse by estimating the transfer of soil contaminants from the soil to plants, and then estimating the daily food intake by the mouse. The conceptual model assumes that the mouse lives entirely within the waste site and obtains all of its food by eating contaminated plants from the waste site.

The QRA for the groundwater operable units evaluates the risk to fish and aquatic invertebrates (crustacea) living in the Columbia River, and to plant-eating ducks, fish-eating ducks, and herons living in the riparian areas and feeding primarily on aquatic organisms. The conceptual model assumes that fish and invertebrates live within the contaminated area, and reach a physiological equilibrium with the contaminants in the water. The conceptual model for the QRA assumes that the food pathway is the major exposure pathway for the ducks and heron. More detail on the calculations and input parameters used are presented in Appendix E.

5.4.2 Analysis

The analysis phase of the QRA includes two processes, a characterization of exposure and a characterization of ecological effects (Figure 4-3). The characterization of exposure process describes how the receptor comes in contact with and takes up the contaminants, and estimates the dose received by the receptor. The concentration and distribution of the contaminant is compared to the distribution and activities of the receptor to estimate the dose the receptor receives. For the QRA, the

dose is expressed in rads/day for the radionuclide contaminants, mg/kg body weight/day for the inorganic contaminants taken up via the food pathway, or simply the concentration of the contaminant for inorganic contaminants taken up directly from soil or water.

The characterization of ecological effects process for the QRA is limited primarily to reviewing the scientific literature to obtain information regarding the toxicity of the contaminants of potential concern at the Hanford Site to the plants and animals selected as receptors for the QRA.

5.4.2.1 Characterization of Exposure. Several assumptions regarding the concentrations and distributions of the contaminants and the distribution and activities of the receptors are made for the QRA, to simplify the process of estimating the dose to the receptor. Because the objective of the QRA is to compare the relative risk of different waste sites, an abbreviated risk assessment protocol can be used, as long as the same protocol is used at all the waste sites being compared. Therefore, one protocol was developed to compare the waste sites within the source operable units and one protocol was developed to compare the waste sites within the groundwater operable units.

The first set of assumptions made for the QRA concerns the types of receptors used to assess risk. Only one receptor, the Great Basin pocket mouse, is used to assess the potential ecological risk at the source operable units. The mouse is common in the area, has a home range comparable to the area of a typical waste site, and lives in and on the soils where the contaminants are located. For the groundwater operable units, several aquatic and riparian species are used to estimate ecological risk. These include fish, aquatic invertebrates, and aquatic plants living in the river; and plant-eating ducks, fish-eating ducks, and herons living in the riparian habitat.

The second set of assumptions concerns the distribution and concentrations of the contaminants at the operable units. The concentration of the contaminant being considered is assumed to be the maximum representative concentration for that contaminant, based on the LFI and historical data sets. For the source operable units, the soil concentrations from samples at or near waste sites are used. For the groundwater operable units, the concentrations in near-river wells are used. The maximum representative value is defined in Section 2.0. Also, the maximum representative concentration is assumed to be uniformly distributed over the waste site (that is, the contaminant concentration at the point of exposure is the maximum representative concentration). The contaminants are assumed to be 100% bioavailable.

A third set of assumptions concerns the activities of the receptors. The pocket mouse is assumed to spend its entire lifetime within the confines of the waste site, to get all of its food from contaminated plants, and to be exposed to the contaminants 365 days per year. The fish and aquatic invertebrates are assumed to live their entire lifetimes in contaminated water, to reach an equilibrium with the concentrations of contaminants in the water media where they live, and to be exposed for 365 days per year. The ducks and heron are assumed to get all of their food from contaminated prey (plants, fish), and to live within the contaminated area. The exposure durations and uptake rates for the ducks and heron are described in more detail in Appendix E. The transfer of contaminants in soils to the plants is estimated using soil-to-plant transfer coefficients and equations adapted from EPA's *Human Health Evaluation Manual* (EPA 1989a). The transfer of contaminants from water to the fish and aquatic invertebrates is estimated using bioconcentration factors, which assume that these aquatic animals reach an equilibrium condition with the water in which they live.

To estimate the total dose from ionizing radiation to organisms in the aquatic and riparian ecosystems, the CRITR2 computer code developed by Baker and Soldat (1992) is used. The steady-state model embodied in CRITR2 assumes that the aquatic organisms reach an equilibrium with the concentrations of the contaminants in the water where they live. Selected receptors are evaluated at various levels of

the aquatic foodweb. The organisms evaluated using CRITR2 are generic aquatic plants, fish, aquatic invertebrates, a plant-eating duck, a fish-eating duck, and a heron. The CRITR2 model is used to estimate radiation doses to plants and animals that are truly aquatic (e.g., fish and algae that reside within the water), and also to estimate total doses to animals that feed primarily on aquatic species (e.g., fish-eating ducks and herons).

For the source operable units, two scenarios were evaluated with respect to the soil depth. In one case the mouse is assumed to be exposed only to those contaminants found in the upper 1.8 m (6 ft) of soil, and in the second case exposure is assumed to include contaminants within the upper 4.6 m (15 ft) of soil. In the former case the exposure scenario assumes that most plant roots are within the upper 1.8 m of soil, and that there is no excavation of the soils in the area for construction of buildings or industrial/agricultural purposes. Therefore, the pocket mouse is exposed only to contaminants in the soil zone where it lives (at the ground surface and in burrows, plus the external exposure to radionuclides in soils both above and below its burrows within the upper 1.8 m of soil). In the latter case, it is assumed that the mouse feeds on deeper rooted plants whose roots penetrate more than 1.8m, and that the ground may be excavated for buildings. Therefore, the deeper contaminants could be physically moved to shallower depths and be located in areas frequented by the mouse.

5.4.2.2 Characterization of Ecological Effects. During the process for characterization of ecological effects information is gathered from a variety of resources to estimate acceptable levels of exposure for each receptor and for each contaminant. The acceptable exposure levels are those doses or concentrations of the contaminants that are below levels that cause a measurable adverse response in the receptor. The DOE benchmark of 1 rad/day, based on DOE Order 5400.5 (DOE 1990) and the International Atomic Energy Agency (1992), is used as the acceptable total dose to ionizing radiation, for both aquatic and terrestrial organisms. Appendix E describes the techniques and formulas for estimating total dose from exposure to radionuclides, both external and internal. For fish and aquatic invertebrates, the acceptable exposure levels to inorganic and organic chemicals are represented by EPA's *Quality Criteria for Water 1986* for the protection of aquatic organisms, or the Washington MTCA surface water standards. The acceptable exposure levels for the mouse, ducks, and heron, with respect to inorganic and organic chemicals, are obtained from the scientific literature that presents NOAEL, LOAEL, or other chronic toxicity values. The available literature frequently does not have values for the exact receptor species being considered (i.e., the Great Basin pocket mouse), and in these cases toxicity data on closely related species are used, sometimes with an appropriate adjustment factor.

The use of a limited number of receptors, a limited number of exposure pathways, the conservative assumptions regarding exposure, and the assumption of 100% bioavailability of the contaminants introduces uncertainty into the risk assessment process. The possible influence of these and other assumptions and other sources of uncertainty must be discussed in the risk assessment reports along with the conclusions of the results of the assessment.

5.4.3 Risk Characterization

In the risk characterization phase, exposure information and toxicity data are integrated to estimate risk to the riparian, aquatic and terrestrial receptors selected for the QRAs (Figure 4-4). For the QRAs at the Hanford Site, a quotient method is used to determine whether or not a specified level of environmental contamination might be of concern. In this method, a benchmark concentration or dose is identified and used as a "safe threshold" or measure of protection for a given receptor. These benchmark values are then compared with the contaminant concentrations at the waste sites (or the

dose received by the receptor), and those concentrations (or doses) that exceed the benchmark values are considered to have potential adverse effects (see Section 4.0). The likelihood of an adverse effect on a receptor, therefore, is expressed in the form of an EHQ. The EHQ is defined as the ratio of the dose received by the receptor (or concentration of the contaminant at the waste site) to the benchmark (safe threshold) value.

The EHQ for determining effects of ionizing radiation from radionuclides is calculated using the equation:

$$EHQ = \frac{\text{organism's dose}}{\text{benchmark}} \qquad 12$$

where the benchmark is one rad/day (DOE Order 5400.5 [DOE 1990]), and where the total dose to the receptor is determined using the CRITR2 computer model (Baker and Soldat 1992) for aquatic and riparian species, or the equations for calculating internal radiation dose to terrestrial receptors (Section E-1.1.1 in Appendix E) for the mouse.

The EHQ for determining the effect of an inorganic (or organic) contaminant to an aquatic receptor such as fish is calculated using the equation:

$$EHQ = \frac{\text{contaminant concentration}}{\text{benchmark}} \qquad 13$$

where the benchmark is EPA's chronic ambient water quality criterion or MTCA's chronic water quality standard for that contaminant, and where the exposure to the fish is expressed as the concentration of the contaminant in the water. An EHQ at or exceeding one is assumed to indicate potential risk to the receptor.

The EHQ for determining the effect of an inorganic (or organic) contaminant on an aquatic or terrestrial receptor that is exposed to the contaminant through the food chain is calculated using the equation:

$$EHQ = \frac{\text{organism's chemical dose}}{\text{benchmark}} \qquad 14$$

where the benchmark is the NOAEL, LOAEL, or other indicator of a chronic toxic dose obtained from the literature, and where the organism's chemical dose is calculated using the equation for calculating internal chemical doses to receptors (Section E-1.2.1 and E-1.2.2 in Appendix E).

5.4.3.1 Ecological Risk Summary. An ecological risk summary must be presented for the QRA, summarizing the risk estimation calculations, discussing the potential additive effects of the contaminants present at waste sites, and discussing the uncertainties involved in deriving the results and conclusions of the QRA. The risk summary process should integrate all of the information developed in the QRA, and present it in a format appropriate for the risk managers.

5.4.3.2 Uncertainty Evaluation. Significant uncertainty exists in the ecological QRA because of the simplifying assumptions used, and because the science of ecological risk assessments is relatively new. Most available information on the effects of ionizing radiation is based on acute dose situations, not from low-dose exposure and chronic effect conditions (Rose 1992). The use of acute data extrapolated to chronic levels is not always appropriate and must be viewed with caution. For example, under chronic exposure conditions, there is a point at which damage caused by very low concentrations of contaminants can be counteracted by a living organisms ability to repair damage.

At this point, there is no adverse biological effect (Ophel et al. 1976) but this condition is not considered in evaluation of the risk of radiological dose.

Toxicity information also presents uncertainties for both human and ecological risk assessments. Animal toxicity values for many contaminants are based on acute-exposure animal studies, with the effects extrapolated to estimate chronic dose levels for those animals. Also, in many cases, plant or animal toxicity values are not available for the receptors of interest, so available data from related species are used.

The EHQ approach is a frequently used technique for estimating the potential for a toxic effect of a contaminant on an individual organism. The NOAELs, LOAELs, or other dose benchmarks used to calculate the EHQ usually represent responses to contaminants by individuals, and not responses by populations or communities. Therefore, an EHQ of one (or greater than one) frequently represents a potential risk to individual plants or animals (e.g., a mouse or a duck), but may have very little relevance to the risk to a population of mice or ducks in a natural ecosystem.

5.4.3.3 Uncertainty for Source Operable Unit QRAs. The QRA for terrestrial ecosystems models the potential exposure of mice assuming they live their entire life at the waste site. The following conservative assumptions lead the QRA to conclusions that overestimate the risk at the waste site.

- Vegetation and physical conditions at the waste sites are suitable habitat for mice.
- The mouse remains within the confines of the waste site for its entire life.
- The mouse eats only contaminated food.
- The concentrations of the contaminants at the point of exposure are the maximum representative concentrations, not average or median concentrations.
- The contaminants are 100% bioavailable, and uniformly distributed within the abiotic and biotic media.
- Plant-to-soil transfer coefficients, and other trophic level input parameters are usually not specific to the Hanford Site.
- Only the food pathway is considered in the QRA to estimate risk to the Great Basin pocket mouse (this assumption would tend to underestimate risk).

5.4.3.4 Uncertainty for Groundwater Operable Unit QRAs. The QRA for aquatic (and riparian) ecosystems models the potential exposure of site contaminants to several aquatic and riparian species. Similar to the QRA approach for terrestrial ecosystems, the approach for aquatic ecosystems is generally conservative. The QRA results will tend to overestimate the risk from groundwater contamination. The following assumptions bias the QRA towards overestimating risk:

- The receptor is assumed to remain within the contaminated zone, and obtain all its food by eating contaminated vegetation or prey.
- The concentrations of the contaminants at the point of exposure are the maximum representative concentrations, not average or median concentrations.

- The groundwater flows (represented by near-river wells) are not diluted by bank storage or as the groundwater enters the river.

- The contaminants are 100% bioavailable, and uniformly distributed within the abiotic and biotic media.

- Only the food pathway, and/or direct absorption of contaminants from the water (for fish and aquatic invertebrates) are considered in the QRA (this assumption would tend to underestimate risk).

- Toxicity benchmark values (NOAELs, LOAELs) are generally not available for the native plant and wildlife species present at the site, so data from other related species are used. (NOTE: this may overestimate or underestimate the risk).

The conservative approach used in the QRA for estimating ecological risk does not have a significant effect on the conclusions of the QRA because the conclusions are used to compare the relative risk at the different waste sites and all waste sites are assessed using the same conservative approach. The results of the QRA, however, should not be used as an estimate of the risk at the waste sites, or even as an approximate estimate of the contaminant concentrations that should be attained by remedial actions at the site.

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APPENDIX A

EXPOSURE SCENARIOS

CONTENTS

A-1.0 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 1989) and in accordance with WAC 173-340-708. Additional information on scenario selection is provided in Section 3.2.4.

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 judgment. Based on the concept of reasonable maximum exposure as recommended by RAGS (EPA 1989) 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.

A-2.0 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 1991, 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, A-2, and A-3.

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 weight of 70 kg (MTCACR, EPA 1991, 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 1991. 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. Body weight and averaging time apply only to nonradioactive contaminants, and are not used in the calculation of radionuclide intakes.

A-2.1 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
- 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 "Dermal Exposure Assessment: Principles and Applications" (EPA 1992). All parameters are presented in Tables A-1, A-2, and A-3 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.

A-2.2 AIR EXPOSURE PATHWAYS

The potential air exposure pathways include:

- Inhalation of fugitive dust
- 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, A-2, and A-3. The exposure frequency of 250 d/yr, recommended by EPA 1991, is used to represent the number of working days per year.

A-2.3 GROUNDWATER EXPOSURE PATHWAYS

The potential groundwater exposure pathways (via wells) are:

- Direct ingestion of groundwater
- Inhalation of volatile contaminants from groundwater use at work
- 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.

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 1992 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 1991. 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 workplace water consumptions. Standard default parameters for dermal contact with water and inhalation of volatiles from water use as provided in EPA 1991 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.

A-2.4 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.

A-3.0 RECREATIONAL SCENARIO

A recreational scenario is provided because recreational activities associated with the Columbia River could 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 1991). Therefore, exposure parameters are derived based on information contained in the "Exposure Factors Handbook" (EPA 1990), EPA 1991, EPA-10 1991, EPA 1992, 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-4, A-5, and A-6 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 1989, EPA 1991)] 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 that include hiking, fishing, hunting, swimming, picnicking, etc.) is 1.70 hr/week (EPA 1990). 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. Then, for exposure pathways such as soil ingestion or inhalation where the exposure parameters are "daily intake rates," (see Tables A-5, A-6, A-7) the exposure frequency becomes 7 days per year, 24 hrs per day, while exposure during events such as swimming is limited by site-specific parameters and the time actually spent performing the event.

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. Exposure to contaminants in biota is evaluated for adult receptors because consumption rates referenced in EPA (1991) and WAC 173-340-730 are adult-specific.

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 1991, 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 1991 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.

Body weight and averaging time apply only to nonradioactive contaminants, and are not used in the calculation of radionuclide intakes (see Section 3.2.5.4).

A-3.1 DIRECT SOIL EXPOSURE PATHWAYS

These pathways include:

- Ingestion of soil
- dermal contact with soil
- external exposure to radionuclides.

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 1991 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 as follows: for children, ingestion rate of 200 mg/d, body weight of 16 kg, and exposure duration of 6 years; for adults, ingestion rate of 100 mg/d, 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 1992 or EPA-10 1991, as noted in the tables.

A-3.2 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).

A-3.3 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 possibly 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.

A-3.4 SURFACE WATER EXPOSURE PATHWAYS

The direct ingestion of surface water 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 1992). 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.

A-3.5 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.

A-3.6 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
- Ingestion of native plants.

Parameters for evaluating these exposure pathways are provided in Tables A-4, A-5, and A-6. 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 judgment. The Washington Department of Wildlife (WDOW) provides annual small game harvest summaries that can be used in determining hunter success rates in the region.

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 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.

A-4.0 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. The Hanford Future Site Uses Working Group (HFSUWG 1992) identified eight future use options and two cleanup scenarios for the Hanford Site. While none of these include specific future use of the Hanford Site for residences the "unrestricted" cleanup scenario does not preclude any human uses.

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. First, 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. Exposure to contaminants in biota is evaluated for adult receptors because consumption rates referenced in EPA (1991) and WAC 173-340-730 are adult-specific.

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

Third, 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 1991 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.

Body weight and averaging time apply only to nonradioactive contaminants, and are not used in the calculation of radionuclide intakes.

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 1991, 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-7, A-8, and A-9.

A-4.1 DIRECT SOIL EXPOSURE PATHWAYS

These pathways include:

- Ingestion of contaminated soil
- Dermal contact with the soil
- 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. Dermal exposure parameters are based on the "Dermal Exposure Assessment: Principles and Applications" (EPA 1992) 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 1991 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 1992. 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.

A-4.2 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, 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 that exceed cleanup levels established under WAC 173-340-750.

A-4.3 GROUNDWATER EXPOSURE PATHWAYS

The potential groundwater exposure pathways (via wells) are:

- Direct ingestion of groundwater
- Inhalation of volatile contaminants from groundwater used in the home (e.g., volatilization from dishwashers, showers, or washing)

- Dermal contact with groundwater during showering, kids running through the sprinkler, 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, watering the lawn, would be much less than exposures evaluated through showering and water ingestion. The risk assessor should consult EPA 1992 and EPA-10 1991 for additional guidance on evaluating dermal exposures and identifying situations for potentially important exposures.

A-4.4 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 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 1991 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-7 and A-8 are standard default parameters for swimming (EPA 1992). 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.

A-4.5 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.

A-4.6 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 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.

Parameters for evaluating these exposure pathways are provided in Tables A-7, A-8, and A-9. Care should be taken that exposure parameters, uptake factors and biota contaminant concentrations are utilizing comparable factors (e.g., dry weight to dry weight, or wet weight to wet weight). Hunting is not allowed in residential areas, therefore, the ingestion of waterfowl or game are not evaluated for the residential scenario.

A-5.0 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 1991 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 1991 provides 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 1991) has been unable to provide sufficient information that can be used to develop defensible farm worker exposure factors. Therefore, the exposure parameters are primarily based on residential exposure parameters. 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.

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. Exposure to contaminants in biota is evaluated for adult receptors because consumption rates referenced in EPA (1991) and WAC 173-340-730 are adult-specific.

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 1991, 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 1991 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. Body weight and averaging time apply only to nonradioactive contaminants, and are not used in the calculation of radionuclide intakes.

Exposure parameters and factors for the agricultural farm family scenario are summarized in Tables A-10, A-11, and A-12.

A-5.1 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. Dermal exposure parameters are based on the "Dermal Exposure Assessment: Principles and Applications" (EPA 1992) and EPA-10 1991.

For noncarcinogens, the soil ingestion parameters are taken directly from Method B (i.e., a soil ingestion 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 1991 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. 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.

A-5.2 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 RME exposures for individuals living on a farm and working in the fields.

A-5.3 GROUNDWATER EXPOSURE PATHWAYS

The potential groundwater exposure pathways (via wells) are:

- Direct ingestion of groundwater
- Inhalation of volatile contaminants from groundwater used in the home (e.g., volatilization from dishwashers, showers, or washing)
- 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 1992 and EPA-10 1991 for additional guidance on evaluating dermal exposures and identifying situations for potentially important exposures.

A-5.4 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 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 1991 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-10 and A-11 are standard default parameters for swimming (EPA 1992). 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.

A-5.5 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.

A-5.6 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
- Ingestion of game (deer).

Parameters for evaluating these exposure pathways are provided in Tables A-10, A-11, and A-12. Care should be taken that exposure parameters, uptake factors, and biota contaminant concentrations are utilizing comparable factors (e.g., dry weight to dry weight, or wet weight to wet weight).

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.

A-6.0 REFERENCES

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Table A-1. Industrial Scenario Exposure Factors - Noncarcinogens.

Pathway		Exposure Parameters							Summary Intake Factor
Media	Exposure Route	Intake Rate[a]	Exposure Frequency[b] (d/yr)	Exposure Duration[a] (yr)	Body Weight (kg)	Averaging Time (yr x d/yr)	Conversion Factors	Other Factors	
Soil	Ingestion	50 mg/d	146[c]	20	70	20 x 365	1E-06 kg/mg	--	2.9E-07 $(d)^{-1}$
	Dermal	0.2 mg/cm^2-d[d,e]	146[c]	20	70	20 x 365	1E-06 kg/mg	5,000 cm^{2f} ABS[g]	ABS x 5.7E-06 $(d)^{-1}$
Air	Inhalation	20 m^3/d	250	20	70	20 x 365	--	--	2.0E-01 m^3/kg-d
Groundwater	Ingestion	1 L/d[b]	250	20	70	20 x 365	--	--	9.8E-03 L/kg-d
	Inhalation[h]	20 m^3/d	250	20	70	20 x 365	--	0.5 L/m^{3i}	9.8E-02 L/kg-d
	Dermal	0.17 hr/d[j]	250	20	70	20 x 365	1E-03 L/cm^3	20,000 cm^{2k} Kp[l]	Kp(cm/hr) x 3.3E-02 L-hr/kg-cm-d
Surface Water	Ingestion	1 L/d[b]	250	20	70	20 x 365	--	--	9.8E-03 L/kg-d
	Inhalation[h]	20 m^3/d	250	20	70	20 x 365	--	0.5 L/m^{3i}	9.8E-02 L/kg-d
	Dermal	0.17 hr/d[j]	250	20	70	20 x 365	1E-03 L/cm^3	20,000 cm^{2k} K_p[l]	K_p(cm/hr) x 3.3E-02 L-hr/kg-cm-d

[a]Parameters based on WAC 173-340-745 or WAC 173-340-750, Method C, except as noted.
[b]Parameters recommended in EPA 1991, except as noted.
[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]*Dermal Exposure Assessment: Principles and Applications* (EPA 1992).
[e]Adherence rate/event.
[f]Skin surface area for soil contact (adult).
[g]Chemical-specific absorption factor (unitless).
[h]Evaluated only for volatile organic contaminants.
[i]0.0005 x 1,000 L/m^3 (Andelman 1990).
[j]EPA-10 (EPA 1991).
[k]Skin surface area for water contact (adult).
[l]Chemical-specific permeability coefficient (cm/hr).

Table A-2. Industrial Scenario Exposure Factors - Carcinogens (Nonradioactive).

Pathway		Exposure Parameters							Summary Intake Factor
Media	Exposure Route	Intake Rate[a]	Exposure Frequency[b] (d/yr)	Exposure Duration[a] (yr)	Body Weight (kg)	Averaging Time (yr x d/yr)	Conversion Factors	Other Factors	
Soil	Ingestion	50 mg/d	146[c]	20	70	70 x 365	1E-06 kg/mg	--	8.2E-08 $(d)^{-1}$
	Dermal	0.2 mg/cm^2-d[d,e]	146[c]	20	70	70 x 365	1E-06 kg/mg	5,000 cm^2[f] ABS[g]	ABS x 1.6E-06 $(d)^{-1}$
Air	Inhalation	20 m^3/d	250	20	70	70 x 365	--	--	5.6E-02 m^3/kg-d
Groundwater	Ingestion	1 L/d[b]	250	20	70	70 x 365	--	--	2.8E-03 L/kg-d
	Inhalation[h]	20 m^3/d	250	20	70	70 x 365	--	0.5 L/m^3[i]	2.8E-02 L/kg-d
	Dermal	0.17 hr/d[j]	250	20	70	70 x 365	1E-03 L/cm^3	20,000 cm^2[k] K_p[l]	K_p(cm/hr) x 9.5E-03 L-hr/kg-cm-d
Surface Water	Ingestion	1 L/d[b]	250	20	70	70 x 365	--	--	2.8E-03 L/kg-d
	Inhalation[h]	20 m^3/d	250	20	70	70 x 365	--	0.5 L/m^3[i]	2.8E-02 L/kg-d
	Dermal	0.17 hr/d[j]	250	20	70	70 x 365	1E-03 L/cm^3	20,000 cm^2[k] K_p[l]	K_p(cm/hr) x 9.5E-03 L-hr/kg-cm-d

[a]Parameters based on WAC 173-340-745 or WAC 173-340-750, Method C, except as noted.
[b]Parameters recommended in EPA 1991.
[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]*Dermal Exposure Assessment: Principles and Applications* (EPA 1992).
[e]Adherence rate/event.
[f]Skin surface area for soil contact (adult).
[g]Chemical-specific absorption factor (unitless).
[h]Evaluated only for volatile organic contaminants.
[i]0.0005 x 1,000 L/m^3 (Andelman 1990).
[j]EPA-10 (EPA 1991).
[k]Skin surface area for water contact (adult).
[l]Chemical-specific permeability coefficient (cm/hr).

Table A-3. Industrial Scenario Exposure Factors - Carcinogens (Radioactive).

Pathway		Exposure Parameters					Summary Intake Factor
Media	Exposure Route	Intake Rate[a]	Exposure Frequency[b] (d/yr)	Exposure Duration[a] (yr)	Conversion Factors	Other Factors	
Soil	Ingestion	50 mg/d	146[c]	20	1E-03 g/mg	--	1.5E+02 g
	External	8 hr/d	146[c]	20	1.14E-04 yr/hr	0.8[d]	2.1E+00 yr
Air	Inhalation	20 m^3/d	250	20	--	--	1.0E+05 m^3
Groundwater	Ingestion	1 L/d[b]	250	20	--	--	5.0E+03 L
	Inhalation[e]	20 m^3/d	250	20	--	0.1 L/m^3[f]	1.0E+04 L
Surface Water	Ingestion	1 L/d[b]	250	20	--	--	5.0E+03 L
	Inhalation[e]	20 m^3/d	250	20	--	0.1 L/m^3[f]	1.0E+04 L

[a]Parameters based on WAC 173-3409-745 or WAC 173-340-750, Method C, except as noted.
[b]Parameters recommended in EPA 1991, except as noted.
[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]Dose reduction factor (unitless; EPA 1991).
[e]Evaluated only for radon-222.
[f]0.0001 x 1,000 L/m^3 (Andelman 1990).

Table A-4. Recreational Scenario Exposure Factors - Noncarcinogens. (2 sheets)

Pathway		Exposure Parameters							Summary Intake Factor
Media	Exposure Route	Intake Rate[a]	Exposure Frequency[b] (d/yr)	Exposure Duration[a] (yr)	Body Weight[c] (kg)	Averaging Time (yr x d/yr)	Conversion Factors	Other Factors	
Soil	Ingestion	200 mg/d	7	6	16	6 x 365	1E-06 kg/mg	--	2.4E-07 $(d)^{-1}$
	Dermal	0.2 mg/cm^2-d[c,d]	7	6 (C)[e] 24 (A)[e]	16 (C) 70 (A)	30 x 365	1E-06 kg/mg	2,500 cm^2[f] 5,000 cm^2[f] ABS[g]	ABS x 3.4E-07 $(d)^{-1}$
Air	Inhalation	10 m^3/d	7	6	16	6 x 365	--	--	1.2E-02 m^3/kg-d
Groundwater	Ingestion	1 L/d	7	6	16	6 x 365	--	--	1.2E-03 L/kg-d
	Dermal	0.17 hr/d[e]	7	30[e]	70[e]	30 x 365	1E-03 L/cm^3	20,000 cm^2[h] K_p[1]	K_p(cm/hr) x 9.3E-04 L-hr/kg-cm-d
Surface Water	Ingestion	1 L/d	7	6	16	6 x 365	--	--	1.2E-03 L/kg-d
	Dermal	2.6 hr/d[e]	7	30[e]	70[e]	30 x 365	1E-03 L/cm^3	20,000 cm^2[h] K_p[1]	K_p(cm/hr) x 6.1E-03 L-hr/kg-cm-d
Sediment[j]	Ingestion	200 mg/d	7	6	16	6 x 365	1E-06 kg/mg	--	2.4E-07 $(d)^{-1}$
	Dermal	0.2 mg/cm^2-d[c,d]	7	6 (C)[e] 24 (A)[e]	16 (C) 70 (A)	30 x 365	1E-06 kg/mg	2,500 cm^2(C)[f] 5,000 cm^2(A)[f] ABS[g]	ABS x 3.4E-07 $(d)^{-1}$
Biota	Waterfowl[k]	--	--	--	--	--	--	--	--
	Game	1 g/d[l]	365	30	70	30 x 365	1E-03 kg/g	0.19[m]	2.7E-06 $(d)^{-1}$
	Fish	54 g/d[n]	365	30	70	30 x 365	1E-03 kg/g	0.5[n]	3.9E-04 $(d)^{-1}$
	Plant	--	--	--	--	--	--	--	--

Table A-4. Recreational Scenario Exposure Factors - Noncarcinogens. (2 sheets)

(C) = Child
(A) = Adult
[a]Parameters recommended in EPA 1991, 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]*Dermal Exposure Assessment: Principles and Applications* (EPA 1992).
[e]Adherence rate/event.
[f]Skin surface area for soil contact (adult).
[g]Chemical-specific absorption factor (unitless).
[h]Skin surface area for water contact (adult).
[i]Ch-specific permeability coefficient (cm/hr).
[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 (Paustenbach 1989).
[m]Intake adjusted for upperbound mean hunter success rate of 19% for game management unit 370.
[n]WAC 173-340-730.

Table A-5. Recreational Scenario Exposure Factors Carcinogens (Nonradioactive). (2 sheets)

Pathway		Exposure Parameters							Summary Intake Factor
Media	Exposure Route	Intake Rate[a]	Exposure Frequency[b] (d/yr)	Exposure Duration[a] (yr)	Body Weight[c] (kg)	Averaging Time (yr x d/yr)	Conversion Factors	Other Factors	
Soil	Ingestion	200 mg/d (C) 100 mg/d (A)	7	6 (C) 24 (A)	16 70	70 x 365	1E-06 kg/mg	--	3.0E-08 $(d)^{-1}$
	Dermal	0.2 mg/cm^2-d[d,e]	7	6 (C) 24 (A)	16 (C) 70 (A)	70 x 365	1E-06 kg/mg	2,500 cm^2[f] 5,000 cm^2[f] ABS[g]	ABS x 1.5E-07 $(d)^{-1}$
Air	Inhalation	20 m^3/d	7	30	70	70 x 365	--	--	2.3E-03 m^3/kg-d
Groundwater	Ingestion	2 L/d	7	30	70	70 x 365	--	--	2.3E-04 L/kg-d
	Dermal	0.17 $(d)^{-1}$	7	30	70[a]	70 x 365	1E-03 L/cm^3	20,000 cm^2[h] K_p[i]	K_p(cm/hr) x 4.0E-04 L-hr/kg-cm-d
Surface Water	Ingestion	2 L/d	7	30	70	70 x 365	--	--	2.3E-04 L/kg-d
	Dermal	2.6 hr/d	7	30	70[a]	70 x 365	1E-03 L/cm^3	20,000 cm^2[h] K_p[i]	K_p(cm/hr) x 6.1E-03 L-hr/kg-cm-d
Sediment[j]	Ingestion	200 mg/d (C) 100 mg/d (A)	7	6 (C) 24 (A)	16 (C) 70 (A)	70 x 365	1E-06 kg/mg	--	3.0E-08 $(d)^{-1}$
	Dermal	0.2 mg/cm^2-d[d,e]	7	6 (C) 24 (A)	16 (C) 70 (A)	70 x 365	1E-06 kg/mg	2,500 cm^2[f] 5,000 cm^2[f] ABS[g]	ABS x 1.5E-07 $(d)^{-1}$
Biota	Waterfowl[k]	--	--	--	--	--	--	--	--
	Game	1 g/d[l]	365	30	70	70 x 365	1E-03 kg/g	0.19[m]	1.2E-06 $(d)^{-1}$
	Fish	54 g/d[n]	365	30	70	70 x 365	1E-03 kg/g	0.5[n]	1.7E-04 $(d)^{-1}$
	Plant	--	--	--	--	--	--	--	--

Table A-5. Recreational Scenario Exposure Factors Carcinogens (Nonradioactive). (2 sheets)

[a]Parameters recommended in EPA 1991, 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]*Dermal Exposure Assessment: Principles and Applications* (EPA 1992).
[e]Adherence rate/event.
[f]Skin surface area for soil contact (adult).
[g]Chemical-specific absorption factor (unitless).
[h]Skin surface area for water contact (adult).
[i]Chemical-specific permeability coefficient (cm/hr).
[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 (Paustenbach 1989).
[m]Intake adjusted for upperbound mean hunter success rate of 19% for game management unit 370.
[n]WAC 173-340-730.

Table A-6. Recreational Scenario Exposure Factors - Carcinogens (Radioactive).

Pathway		Exposure Parameters					Summary Intake Factor
Media	Exposure Route	Intake Rate[a]	Exposure Frequency[b] (d/yr)	Exposure Duration[a] (yr)	Conversion Factors	Other Factors	
Soil	Ingestion	200 mg/d (C) 100 mg/d (A)	7	6 (C) 24 (A)	1E-03 g/mg	--	2.5E+01 g
	External	8 hr/d[b]	7	30	1.14E-04 yr/hr	0.8[c]	1.5E-01 yr
Air	Inhalation	20 m^3/d	7	30	--	--	4.2E+03 m^3
Groundwater	Ingestion	2 L/d	7	30	--	--	4.2E+02 L
Surface Water	Ingestion	2 L/d	7	30	--	--	4.2E+02 L
Sediment[d]	Ingestion	200 mg/d (C) 100 mg/d (A)	7	6 (C) 24 (A)	1E-03 g/mg	--	2.5E+01 g
Biota	Waterfowl[b]	--	--	--	--	--	--
	Game	1 g/d[e]	365	30	--	0.19[f]	2.1E+03 g
	Fish	54 g/d[g]	365	30	--	0.5[g]	3.0E+05 g
	Plant	--	--	--	--	--	--

(C) = Child
(A) = Adult
[a]Parameters recommended in EPA 1991, EPA-10 1991, except as noted.
[b]Site-specific parameters; see text for additional information.
[c]Dose reduction factor (unitless; EPA 1991).
[d]Exposure parameters correspond to swimming events and utilize default parameters for soil ingestion.
[e]Venison fat consumption rate based on 45 kg deer per family per year (Paustenbach 1989).
[f]Intake adjusted for upperbound mean hunter success rate of 19% for game management unit 370.
[g]WAC 173-340-730.

Table A-7. Residential Scenario Exposure Factors - Noncarcinogens. (2 sheets)

Pathway		Exposure Parameters							Summary Intake Factor
Media	Exposure Route	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	
Soil	Ingestion	200 mg/d	365	6	16	6 x 365	1E-06 kg/mg	--	1.3E-05 $(d)^{-1}$
	Dermal	0.2 mg/cm^2-d[b,c]	180[e]	6 (C)[f] 24 (A)[f]	16 (C) 70 (A)	30 x 365	1E-06 kg/mg	2,500 cm^2(C)[d] 5,000 cm^2(A)[d] ABS[g]	ABS x 8.7E-06 $(d)^{-1}$
Air	Inhalation	10 m^3/d	365	6	16	6 x 365	--	--	6.3E-01 m^3/kg-d
Groundwater	Ingestion	1 L/d	365	6	16	6 x 365	--	--	6.3E-02 L/kg-d
	Inhalation	15 m^3/d[h]	365	30	70	30 x 365	--	0.5 L/m^3[i]	1.1E-01 L/kg-d
	Dermal	0.17 hr/d[f,j]	365	30[f]	70[f]	30 x 365	1E-03 L/cm^3	20,000 cm^2[k] K_p[f]	K_p(cm/hr) x 4.9E-02 L-hr/kg-cm-d
Surface Water	Ingestion	1 L/d	365	6	16	6 x 365	--	--	6.3E-02 L/kg-d
	Inhalation	15 m^3/d[b]	365	30	70	30 x 365	--	0.5 L/m^3[i]	1.1E-01 L/kg-d
	Dermal	0.17 hr/d[f,j] 2.6 hr/d[f,m]	365 7[e]	30[f]	70[f]	30 x 365	1E-03 L/cm^3	20,000 cm^2[k] K_p[l]	K_p(cm/hr) x 4.9E-02[j] L-hr/kg-cm-d K_p(cm/hr) x 1.4E-02[m] L-hr/kg-cm-d
Sediment[n]	Ingestion	200 mg/d	7[e]	6	16	6 x 365	1E-06 kg/mg	--	2.4E-07 $(d)^{-1}$
	Dermal	0.2 mg/cm^2-d[b,c]	7[e]	6 (C)[f] 24 (A)[f]	16 (C) 70 (A)	30 x 365	1E-06 kg/mg	2,500 cm^2(C)[d] 5,000 cm^2(A)[d] ABS[g]	ABS x 3.4E-07 $(d)^{-1}$
Biota	Fish	54 g/d[o]	365	30	70	30 x 365	1E-03 kg/g	0.5[o]	3.9E-04 $(d)^{-1}$
	Fruit	42 g/d[p]	365	30	70	30 x 365	1E-03 kg/g	--	6.0E-04 $(d)^{-1}$
	Vegetables	80 g/d[p]	365	30	70	30 x 365	1E-03 kg/g	--	1.1E-03 $(d)^{-1}$

Table A-7. Residential Scenario Exposure Factors - Noncarcinogens. (2 sheets)

[a]Parameters recommended in WAC 173-340-720, WAC 173-340-740, or WAC 173-340-750, Method B, except as noted.
[b]*Dermal Exposure Assessment: Principles and Applications* (EPA 1992).
[c]Adherence rate/event.
[d]Skin surface area for soil contact (adult).
[e]Site-specific parameter; see text for additional information.
[f]EPA-10 1991.
[g]Chemical-specific absorption factor (unitless).
[h]Indoor inhalation rate (EPA 1991); evaluated only for volatile organic contaminants.
[i]0.0005 x 1,000 L/m3 (Andelman 1990).
[j]Default value for showering.
[k]Skin surface area for water contact (adult).
[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]WAC 173-340-730.
[p]EPA 1991; based on wet weight.

Table A-8. Residential Scenario Exposure Factors - Carcinogens (Non-Radioactive). (2 sheets)

Pathway		Exposure Parameters							Summary Intake Factor
Media	Exposure Route	Intake Rate[a]	Exposure Frequency[b] (d/yr)	Exposure Duration[a] (yr)	Body Weight[b] (kg)	Averaging Time (yr x d/yr)	Conversion Factors	Other Factors	
Soil	Ingestion	200 mg/d (C) 100 mg/d (A)	365	6 (C) 24 (A)	16 (C) 70 (A)	70 x 365	1E-06 kg/mg	--	1.6E-06 $(d)^{-1}$
	Dermal	0.2 mg/cm^2-d[c,d]	180[f]	6 (C)[g] 24 (A)[g]	16 (C) 70 (A)	70 x 365	1E-06 kg/mg	2,500 cm^2(C)[e] 5,000 cm^2(A)[e] ABS[h]	ABS x 3.7E-06 $(d)^{-1}$
Air	Inhalation	20 m^3/d	365	30	70	70 x 365	--	--	1.2E-01 m^3/kg-d
Groundwater	Ingestion	2 L/d	365	30	70	70 x 365	--	--	1.2E-02 L/kg-d
	Inhalation	15 m^3/d[i]	365	30	70	70 x 365	--	0.5 L/m^3[j]	4.6E-02 L/kg-d
	Dermal	0.17 hr/d[g,k]	365	30[g]	70[g]	70 x 365	1E-03 L/cm^3	20,000 cm^2[l] K_p[m]	K_p(cm/hr) x 2.1E-02 L-hr/kg-cm-d
Surface Water	Ingestion	2 L/d	365	30	70	70 x 365	--	--	1.2E-02 L/kg-d
	Inhalation	15 m^3/d[i]	365	30	70	70 x 365	--	0.5 L/m^3[j]	4.6E-02 L/kg-d
	Dermal	0.17 hr-d[g,k] 2.6 hr/d[n]	365 7[f]	30[g]	70[g]	70 x 365	1E-03 L/cm^3	20,000 cm^2[l] K_p[m]	K_p(cm/hr) x 2.1E-02[k] L-hr/kg-cm-d K_p(cm/hr) x 6.1E-03[n] L-hr/kg-cm-d
Sediment[o]	Ingestion	200 mg/d (C) 100 mg/d (A)	7[f]	6 (C) 24 (A)	16 (C) 70 (A)	70 x 365	1E-06 kg/mg	--	3.0E-08 $(d)^{-1}$
	Dermal	0.2 mg/cm^2-d[c,d]	7[f]	6 (C)[g] 24 (A)[g]	16 (C) 70 (A)	70 x 365	1E-06 kg/mg	2,500 cm^2 (C)[e] 5,000 cm^2 (A)[e] ABS[h]	ABS x 1.5E-07 $(d)^{-1}$
Biota	Fish	54 g/d[p]	365	30	70	70 x 365	1E-03 kg/g	0.5[p]	1.7E-04 $(d)^{-1}$
	Fruit	42 g/d[q]	365	30	70	70 x 365	1E-03 kg/g	--	2.6E-04 $(d)^{-1}$
	Vegetables	80 g/d[q]	365	30	70	70 x 365	1E-03 kg/g	--	4.9E-04 $(d)^{-1}$

Table A-8. Residential Scenario Exposure Factors - Carcinogens (Non-Radioactive). (2 sheets)

(C) = Child
(A) = Adult
[a]Parameters recommended in EPA 1991, except as noted.
[b]Parameters recommended in WAC 173-340-720, WAC 173-340-740, or WAC 173-340-750, Method B, except as noted.
[c]Dermal Exposure Assessment: Principles and Applications (EPA 1992).
[d]Adherence rate/event.
[e]Skin surface area for soil contact (adult).
[f]Site-specific parameter; see text for additional information.
[g]EPA-10 1991.
[h]Chemical-specific absorption factor (unitless).
[i]Indoor inhalation rate (EPA 1991); evaluated only for volatile organic contaminants.
[j]0.0005 x 1,000 L/m3 (Andelman 1990).
[k]Default value for showering.
[l]Skin surface area for water contact (adult).
[m]Chemical-specific permeability coefficient (cm/hr).
[n]Default value for swimming.
[o]Exposure parameters correspond to swimming events and utilize default parameters for ingestion and dermal exposures to soil.
[p]WAC 173-340-730.
[q]EPA 1991; based on wet weight.

Table A-9. Residential Scenario Exposure Factors - Carcinogens (Radioactive).

Pathway		Exposure Parameters					Summary Intake Factor
Media	Exposure Route	Intake Rate[a]	Exposure Frequency[b] (d/yr)	Exposure Duration[a] (yr)	Conversion Factors	Other Factors	
Soil	Ingestion	200 mg/d (C) 100 mg/d (A)	365	6 (C) 24 (A)	1E-03 g/mg	--	1.3E+03 g
	External	24 hr/d[c]	365	30	1.14E-04 yr/hr	0.8[d]	2.4E+01 yr
Air	Inhalation	20 m^3/d	365	30	--	--	2.2E+05 m^3
Groundwater	Ingestion	2 L/d	365	30	--	--	2.2E+04 L
	Inhalation	15 m^3/d[e]	365	30	--	0.1 L/m^3[f]	1.6E+04 L
Surface Water	Ingestion	2 L/d	365	30	--	--	2.2E+04 L
	Inhalation	15 m^3/d[e]	365	30	--	0.1 L/m^3[f]	1.6E+04 L
Sediment[g]	Ingestion	200 mg/d (C) 100 mg/d (A)	7[c]	6 (C) 24 (A)	1E-03 g/mg	--	2.5E+01 g
Biota	Fish	54 g/d[h]	365	30	--	0.5[h]	3.0E+05 g
	Fruit	42 g/d[i]	365	30	--	--	4.6E+05 g
	Vegetable	80 g/d[i]	365	30	--	--	8.8E+05 g

(C) = Child
(A) = Adult
[a]Parameters recommended in EPA 1991, except as noted.
[b]Parameters recommended in WAC 173-340-720, WAC 173-340-740, or WAC 173-340-750, Method B, except as noted.
[c]Site-specific parameter; see text for additional information.
[d]Dose reduction factor (unitless; EPA 1991).
[e]Indoor inhalation rate (EPA 1991); evaluated only for radon-222.
[f]0.0001 x 1,000 L/m3 (Andelman 1990).
[g]Exposure parameters correspond to swimming events and utilize default parameters for soil ingestion.
[h]WAC 173-340-730.
[i]EPA 1991; based on wet weight.

Table A-10. Agricultural Scenario Exposure Factors - Noncarcinogens. (2 sheets)

Pathway		Exposure Parameters[a]							Summary Intake Factor
Media	Exposure Route	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	
Soil	Ingestion	200 mg/d	365	6	16	6 x 365	1E-06 kg/mg	--	1.3E-05 $(d)^{-1}$
	Dermal	0.2 mg/cm^2-d[b,c]	180[e]	6 (C)[f] 24 (A)[f]	16 (C)[f] 70 (A)[f]	30 x 365	1E-06 kg/mg	2,500 cm^2(C)[d] 5,000 cm^2(A)[d] ABS[g]	ABS x 8.7E-06 $(d)^{-1}$
Air	Inhalation	10 m^3/d	365	6	16	6 x 365	--	--	6.3E-01 m^3/kg-d
Groundwater	Ingestion	1 L/d	365	6	16	6 x 365	--	--	6.3E-02 L/kg-d
	Inhalation	15 m^3/d[h]	365	30	70	30 x 365	--	0.5 L/m^3[i]	1.1E-01 L/kg-d
	Dermal	0.17 hr/d[h,j]	365	30[f]	70[f]	30 x 365	1E-03 L/cm^3	20,000 cm^2[k] K_p[l]	K_p(cm/hr) x 4.9E-02 L-hr/kg-cm-d
Surface Water	Ingestion	1 L/d	365	6	16	6 x 365	--	--	6.2E-02 L/kg-d
	Inhalation	15 m^3/d[h]	365	30	70	30 x 365	--	0.5 L/m^3[i]	1.1E-01 L/kg-d
	Dermal	0.17 hr/d[h,j] 2.6 hr/d[f,m]	365 7[e]	30[f]	70[f]	30 x 365	1E-03 L/cm^3	20,000 cm^2[k] K_p[l]	K_p(cm/hr) x 4.9E-02[j] L-hr/kg-cm-d K_p(cm/hr) x 1.4E-02[m] L-hr/kg-cm-d
Sediment[n]	Ingestion	200 mg/d	7[e]	6	16	6 x 365	1E-06 kg/mg	--	2.4E-07 $(d)^{-1}$
	Dermal	0.2 mg/cm-d[b,c]	7[e]	6 (C)[f] 70 (A)[f]	16 (C)[f] 24 (A)[f]	30 x 365	1E-06 kg/mg	2,500 cm^2(C)[d] 5,000 cm^2(A)[d] ABS[g]	ABS x 3.4E-07 $(d)^{-1}$
Biota[k]	Dairy	300 g/d[o]	365	30	70	30 x 365	1E-03 kg/g	--	4.3E-03 $(d)^{-1}$
	Beef	75 g/d[o]	365	30	70	30 x 365	1E-03 kg/g	--	1.1E-03 $(d)^{-1}$
	Game	1 g/d[p]	365	30	70	30 x 365	1E-03 kg/g	0.19[q]	2.7E-06 $(d)^{-1}$
	Fish	54 g/d[r]	365	30	70	30 x 365	1E-03 kg/g	0.5[r]	3.9E-04 $(d)^{-1}$
	Fruit	42 g/d[o]	365	30	70	30 x 365	1E-03 kg/g	--	6.0E-04 $(d)^{-1}$
	Vegetable	80 g/d[o]	365	30	70	30 x 365	1E-03 kg/g	--	1.1E-03 $(d)^{-1}$

Table A-10. Agricultural Scenario Exposure Factors - Noncarcinogens. (2 sheets)

(C) = Child
(A) = Adult
[a]Parameters recommended in WAC 173-340-720, WAC 173-340-740, or WAC 173-340-750, Method B, except as noted.
[b]*Dermal Exposure Assessment: Principles and Applications* (EPA 1992).
[c]Adherence rate/event.
[d]Skin surface area for soil contact (adult).
[e]Site-specific parameter; see text for additional information.
[f]EPA-10 (EPA 1991).
[g]Chemical-specific absorption factor (unitless).
[h]Indoor inhalation rate (EPA 1991); evaluated only for volatile organic contaminants.
[i]0.0005 x 1,000 L/m3 (Andelman 1990).
[j]Default value for showering.
[k]Skin surface area for water contact (adult).
[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 1991; fruit and vegetable parameters based on wet weight.
[p]Venison fat consumption rate based on 45-kg deer per family per year (Paustenbach 1989).
[q]Intake adjusted for upperbound hunter success rate of 19% for game management unit 370.
[r]WAC 173-340-730.

Table A-11. Agricultural Scenario Exposure Factors
Carcinogens (Nonradioactive). (2 sheets)

Pathway		Exposure Parameters							Summary Intake Factor
Media	Exposure Route	Intake Rate[a]	Exposure Frequency[b] (d/yr)	Exposure Duration[a] (yr)	Body Weight[b] (kg)	Averaging Time (yr x d/yr)	Conversion Factors	Other Factors	
Soil	Ingestion	200 mg/d (C) 100 mg/d (A)	365	6 (C) 24 (A)	16 (C) 70 (A)	70 x 365	1E-06 kg/mg	--	1.6E-06 $(d)^{-1}$
	Dermal	0.2 mg/cm^2-d[c,d]	180[f]	6 (C)[g] 24 (A)[g]	16 (C) 70 (A)	70 x 365	1E-06 kg/mg	2,500 cm^2 (C)[e] 5,000 cm^2 (A)[e] ABS[h]	ABS x 3.7E-06 $(d)^{-1}$
Air	Inhalation	20 m^3/d	365	30	70	70 x 365	--	--	1.2E-01 m^3/kg-d
Groundwater	Ingestion	2 L/d[b]	365	30	70	70 x 365	--	--	1.2E-02 L/kg-d
	Inhalation	15 m^3/d[i]	365	30	70	70 x 365	--	0.5 L/m^3[j]	4.6E-02 L/kg-d
	Dermal	0.17 hr/d[g,k]	365	30[g]	70[g]	70 x 365	1E-03 L/cm^3	20,000 cm^2[l] K_p[m]	K_p(cm/hr) x 2.1E-02 L-hr/kg-cm-d
Surface Water	Ingestion	2 L/d[b]	365	30	70	70 x 365	--	--	1.2E-02 L/kg-d
	Inhalation	15 m^3/d[i]	365	30	70	70 x 365	--	0.5 L/m^3[j]	4.6E-02 L/kg-d
	Dermal	0.17 hr/d[g,k] 2.6 hr/d[g,n]	365 7[f]	30[g]	70[g]	70 x 365	1E-03 L/cm^3	20,000 cm^2[l] K_p[m]	K_p(cm/hr) x 2.1E-02[k] L-hr/kg-cm-d K_p(cm/hr) x 6.1E-03[n] L-hr/kg-cm-d
Sediment[p]	Ingestion	200 mg/d (C) 100 mg/d (A)	7[f]	6 (C) 24 (A)	16 (C) 70 (A)	70 x 365	1E-06 kg/mg	--	3.0E-08 $(d)^{-1}$
	Dermal	0.2 mg/cm^2-d[c,d]	7[f]	6 (C)[g] 24 (A)[g]	16 (C) 70 (A)	70 x 365	1E-06 kg/mg	2,500 cm^2(C)[e] 5,000 cm^2(A)[e] ABS[h]	ABS x 1.5E-07 $(d)^{-1}$
Biota	Dairy	300 g/d	365	30	70	70 x 365	1E-03 kg/g	--	1.8E-03 $(d)^{-1}$
	Beef	75 g/d	365	30	70	70 x 365	1E-03 kg/g	--	4.6E-04 $(d)^{-1}$
	Game	1 g/d[p]	365	30	70	70 x 365	1E-03 kg/g	0.19[q]	1.2E-06 $(d)^{-1}$
	Fish	54 g/d[r]	365	30	70	70 x 365	1E-03 kg/g	0.5[r]	1.7E-04 $(d)^{-1}$
	Fruit	42 g/d[s]	365	30	70	70 x 365	1E-03 kg/g	--	2.6E-04 $(d)^{-1}$
	Vegetable	80 g/d[s]	365	30	70	70 x 365	1E-03 kg/g	--	4.9E-04 $(d)^{-1}$

Table A-11. Agricultural Scenario Exposure Factors Carcinogens (Nonradioactive). (2 sheets)

(C) = Child
(A) = Adult
[a]Parameters recommended in EPA 1991, except as noted.
[b]Parameters recommended in WAC 173-340-720, WAC 173-340-740, or WAC 173-340-750, Method B, except as noted.
[c]*Dermal Exposure Assessment: Principles and Applications* (EPA 1992).
[d]Adherence rate/event.
[e]Skin surface area for soil contact (adult).
[f]Site-specific parameter; see text for additional information.
[g]EPA-10 (EPA 1991).
[h]Chemical-specific absorption factor (unitless).
[i]Indoor inhalation rate (EPA 1991); evaluated only for volatile organic contaminants.
[j]0.0005 x 1,000 L/m3 (Andelman 1990).
[k]Default value for showering.
[l]Skin surface area for water contact (adult).
[m]Chemical-specific permeability coefficient (cm/hr).
[n]Default value for swimming.
[o]Exposure parameters correspond to swimming events and utilize default parameters for ingestion and dermal exposures to soil.
[p]Venison fat consumption rate based on 45-kg deer per family per year (Paustenbach 1989).
[q]Intake adjusted for upperbound hunter success rate of 19% for game management unit 370.
[r]WAC 173-340-730.
[s]EPA 1991; based on wet weight.

Table A-12. Agricultural Scenario Exposure Factors - Carcinogens (Radioactive).

Pathway		Exposure Parameters					Summary Intake Factor
Media	Exposure Route	Intake Rate[a]	Exposure Frequency[b] (d/yr)	Exposure Duration[a] (yr)	Conversion Factors	Other Factors	
Soil	Ingestion	200 mg/d (C) 100 mg/d (A)	365	6 (C) 24 (A)	1E-03 g/mg	--	1.3E+03 g
	External	24 hr/d[c]	365	30	1.14E-04 yr/hr	0.8[d]	2.4E+01 yr
Air	Inhalation	20 m^3/d	365	30	--	--	2.2E+05 m^3
Groundwater	Ingestion	2 L/d[b]	365	30	--	--	2.2E+04 L
	Inhalation	15 m^3/d[e]	365	30	--	0.1 L/m^3[f]	1.6E+04 L
Surface Water	Ingestion	2 L/d[b]	365	30	--	--	2.2E+04 L
	Inhalation	15 m^3/d[e]	365	30	--	0.1 L/m^3[f]	1.6E+04 L
Sediment[g]	Ingestion	200 mg/d (C) 100 mg/d (A)	7[c]	6 (C) 24 (A)	1E-03 g/mg	--	2.5E+01 g
Biota	Dairy	300 g/d	365	30	--	--	3.3E+06 g
	Beef	75 g/d	365	30	--	--	8.2E+05 g
	Game	1 g/d[h]	365	30	--	0.19[i]	2.1E+03 g
	Fish	54 g/d[j]	365	30	--	0.5[j]	3.0E+05 g
	Fruit	42 g/d[k]	365	30	--	--	4.6E+05 g
	Vegetable	80 g/d[k]	365	30	--	--	8.8E+05 g

(C) = Child
(A) = Adult
[a]Parameters recommended in EPA 1991, except as noted.
[b]Parameters recommended in WAC 173-340-720, WAC 173-340-740, or WAC 173-340-750, Method B, except as noted.
[c]Site-specific parameter; see text for additional information.
[d]Dose reduction factor (unitless; EPA 1991).
[e]Indoor inhalation rate (EPA 1991); evaluated only for radon-222.
[f]0.0001 x 1,000 L/m3 (Andelman 1990).
[g]Parameter recommended in EPA-10 1991.
[h]Venison fat consumption rate based on 45-kg deer per family per year (Paustenbach 1989).
[i]Intake adjusted for upperbound hunter success rate of 19% for game management unit 370.
[j]WAC 173-340-730.
[k]EPA 1991; based on wet weight.

APPENDIX B

M-29-01 MILESTONE: DESCRIPTION OF CODES AND MODELS TO BE USED IN RISK ASSESSMENT

NOTE: Appendix B is currently published as DOE/RL-91-44 and has been reformatted for inclusion in this report.

FOREWORD

A Risk Assessment Modeling Committee was formed in late 1991 of representatives of DOE-RL, EPA, Ecology, and their support contractors to exchange experiences and opinions relating to the use of numerical models for risk assessment for the purpose of writing a document to fulfill Tri-Party Agreement Milestone M-29-01: "Identify and submit descriptions of codes and models to be used in risk assessment." In general, the committee meetings enhanced understanding between the involved parties and improved the decision-making process. It was recommended that meetings of the committee be continued 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. However, no further milestones for development of risk assessment methodology were imposed by the Tri-Parties and no update of this document has occurred since it was submitted in December, 1991.

CONTENTS

FIGURES

B-1.0 INTRODUCTION

B-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 are 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 sections: (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.

B-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

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.

B-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 B-3.0 (Selection Criteria) and Section B-4.0 (Recommendations).

B-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 B1-1 for air, Figure B1-2 for surface water, and Figure B1-3 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.

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 B-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.

B-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.

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.

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.

B-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 B-4.0.

B-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 multidimensional 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.

B-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.

B-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 B1-3. 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, multiphase flow, and geochemistry. A brief description of the history of modeling these processes at the Hanford Site is discussed in the following sections.

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

B-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.

B-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.

B-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.

B-2.3.3.5 Multiphase Modeling. Development of multiphase 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 multiphase modeling activities at the Hanford Site is unknown.

B-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.

B-3.0 SELECTION CRITERIA

B-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.

B-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.

B-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.

B-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.

B-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.

B-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.

B-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.

B-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 B-3.3 and ultimately determine the criteria for the recommendations provided in Section B-4.0.

B-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.

B-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.

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

B-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.

B-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
- Radioactive decay.

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

B-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.

B-3.3 COMPARISON OF TECHNICAL CAPABILITIES

This section presents a matrix (Table B-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.

B-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.

B-4.1 AIR

As stated in Section B-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.

B-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.

B-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.

B-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.

B-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 multiphase 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

B-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.

B-4.3.4 CFEST

CFEST (Gupta et al. 1987, 1982) is a finite element code for two- or three-dimensional analysis of hydrologic flow, heat transport, and single-constituent solute transport in subsurface confined environments at either the regional or local scale. Water table environments can be modeled with some extra effort. CFEST also has the capability for flow path and travel time analyses. The CFEST-SC code is linked to the ARC/INFO geographic information system (GIS) to provide pre- and post-processing capabilities.

CFEST was developed at the Hanford Site. The CFEST-SC computer code (Cole et al. 1988) is a proprietary code, but it is available on a royality free basis for all Hanford work. The program is available on several hardware platforms. CFEST-SC has been the subject of extensive verification efforts (Cole et al. 1988; Gupta et al. 1987).

CFEST-SC has been applied in water flow and contaminant transport studies of the unconfined and confined aquifers underlying the Hanford Site. CFEST-SC is presently being used for an area-wide study of the Hanford Site.

B-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 B-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 in use in support of the Hanford Site remediation had a definite advantage.

B-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).

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

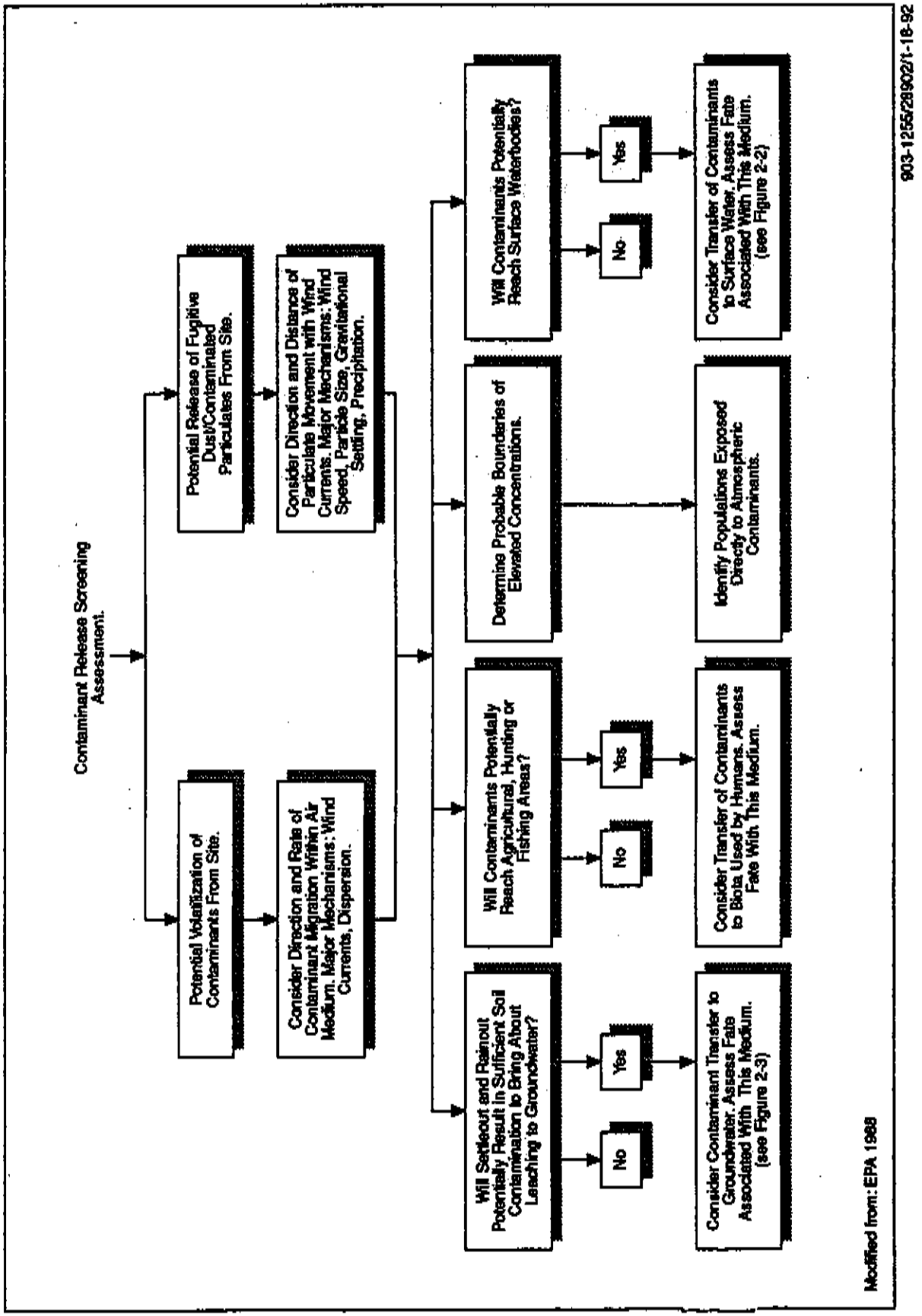


Figure B-2. Environmental Fate Screening Assessment Decision Network: Surface Water.

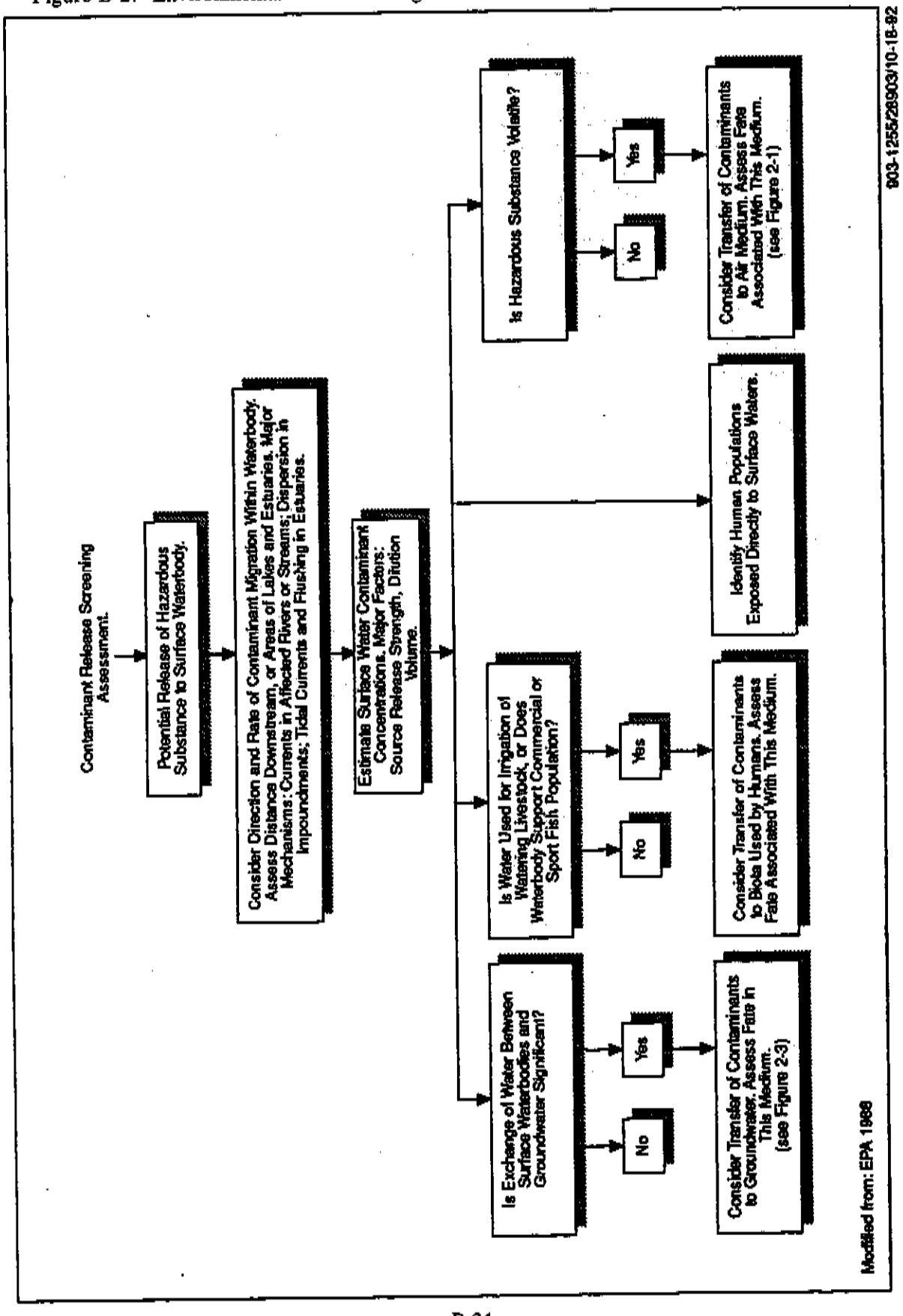
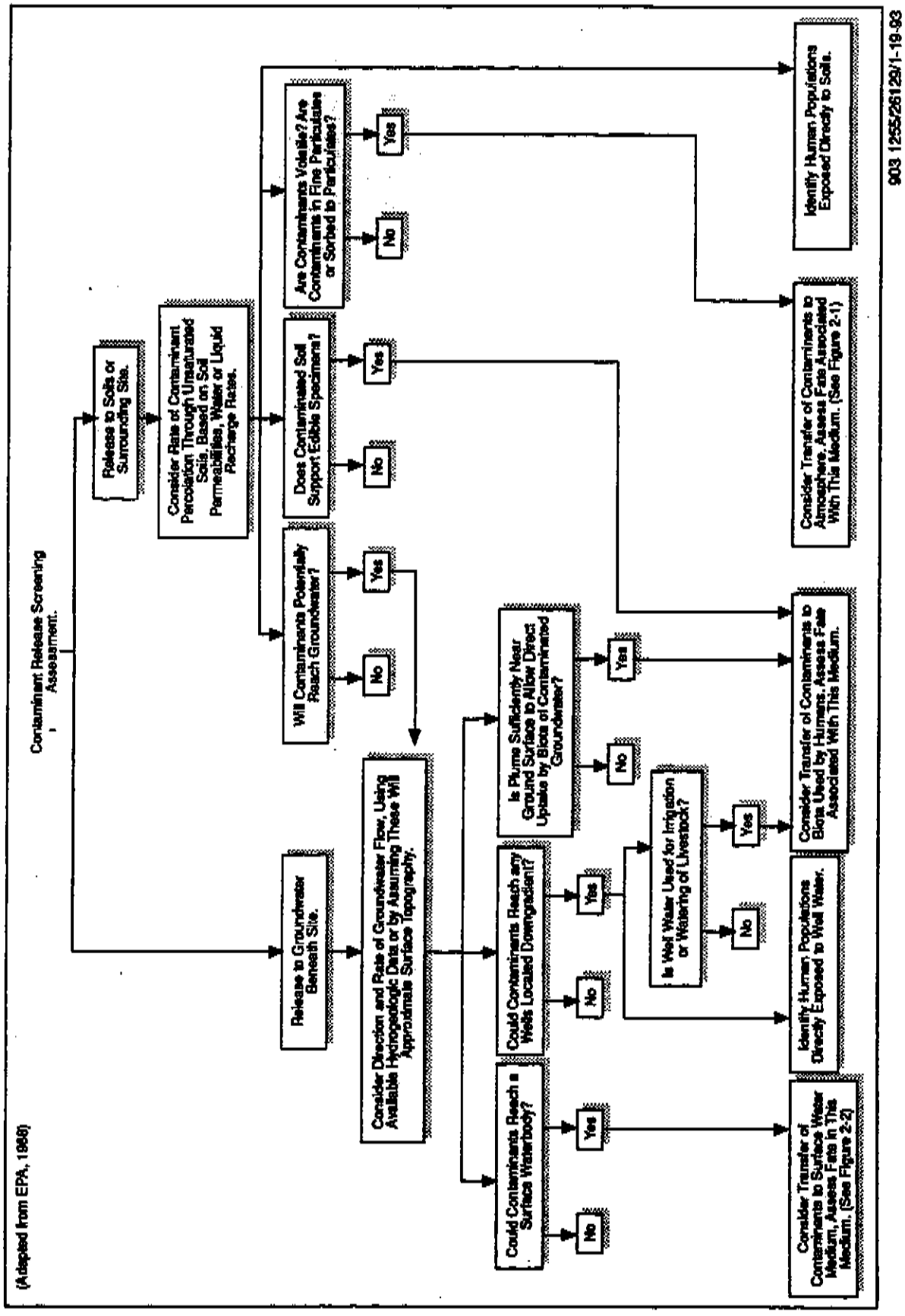


Figure B-3. Environmental Fate Screening Assessment Decision Network: Soils and Groundwater.



APPENDIX C

SUPPORTIVE ENVIRONMENTAL INFORMATION CONSIDERED IN THE PREPARATION OF THE HANFORD SITE RISK ASSESSMENT METHODOLOGY (HSRAM)

CONTENTS

FIGURES

C-1.0 INTRODUCTION

This appendix contains support material and detailed information for ecological risk assessment methodology, including an overview of Hanford ecological resources, parameters for the aquatic and terrestrial operable units, the conceptual model overview, and information concerning the toxicity values.

C-1.1 OVERVIEW OF HANFORD ECOLOGICAL RESOURCES

This section discusses ecological resources on the U.S. Department of Energy's (DOE) Hanford Site in southeastern Washington State.

C-1.1.1 Vegetation Communities at the Hanford Site

Ecosystems potentially at risk on the Hanford Site include the semi-arid terrestrial, riparian terrestrial, lacustine, and riverine systems. Terrestrial ecosystems are subdivided on the basis of plant community. Major communities are discussed below. A total of 590 species of vascular plants has been found at the Hanford Site. Of these, approximately 20% are not native to the area or have been introduced (Sackschewsky et al. 1992). Within the shrub-steppe ecosystem are four recognizable vegetation types. These are: 1) shrublands, 2) grasslands, 3) riparian areas, and 4) tree zones. Please see Figure C-1.

Stegen (1994) conducted a study to determine the plant communities and to estimate the vegetation cover in and directly adjacent to the 100 and 200 Areas, primarily in relation to waste sites. Much of the area in and around the 100 Area reactor sites and the 200 Areas was found to be highly disturbed. Additionally, many of the waste sites were nonvegetated (< 5 % cover) because these areas are routinely sprayed with herbicides as part of vegetation control programs.

Shrub-steppe/Bunchgrass Community

Many parts of the Hanford Site, including the Rattlesnake Hills and parts of the Yakima and Umtanum Ridges, support both shrub-steppe and bunchgrass communities. Vegetative composition varies, depending on soil type, available moisture, and elevation. Lower elevations typically support big sagebrush/Sandberg's bluegrass associations, although in most areas cheatgrass (*Bromos tectorum*) has become the dominant grass species (Downs et al. 1993). Needle-and-thread grass (*Stipa comata*) and Indian ricegrass (*Oryzopsis hymenoide*) replace Sandberg's bluegrass in coarser soils. Bluebunch wheatgrass dominates the understory in higher elevations, along with Cusick's bluegrass (*Poa cusickii*). Meadowlarks (Brandt and Rickard 1992), horned lark (*Eremphila alpestris*) (Landeen et al 1992), and sage sparrows (Fitzner and Rickard 1975) are common birds in the lowland shrub-steppe/bunchgrass community. The upland shrub-steppe community historically supported nesting sage grouse (*Centrocerus urophasianus*), a state threatened species. However, wildfires in the early 1980's destroyed much of the sagebrush on the Rattlesnake Hills. Currently, sage grouse are found only at upper elevations of Rattlesnake Mountain (Fitzner et al 1994). Black-tailed jackrabbits, deer mice, and Great Basin pocket mice are also common in the shrub-steppe/bunchgrass community. A herd of Rocky Mountain elk (Cervus elaphus) is also known to use this community in the Rattlesnake Hills (Cushing 1993).

Shrub-steppe communities also exist on the Columbia River plain. In these areas, big sagebrush and bitterbrush are the dominant shrubs, but cheatgrass has replaced or is interspersed with native grass species in the understory. Native bunchgrasses most likely to occur on sandy soils on the Columbia plain include squirreltail, Indian ricegrass, and needle-and-thread grass. Mature stands of sagebrush provide important habitat for loggerhead shrikes, a federal candidate species, as well as sage thrashers and sage sparrows, both state candidate species (Downs et al. 1993). Long-billed curlews (a federal candidate 3 species) are also commonly found on the Columbia River plain in areas devoid of shrub cover. Ferruginous hawks, red-tailed hawks, Swainson's hawks, and golden eagles use the Columbia River plain sagebrush community as hunting areas. Deer mice, Great Basin pocket mice, and black-tailed jackrabbits are common in this habitat type. Badger, small mammals, coyote, and mule deer also use this community (Downs et al. 1993).

Riparian Community

The shoreline of the Hanford Reach of the Columbia River is the most extensive riparian area on the site. The Hanford Reach is the only stretch of the river within the U.S. that has remained unimpounded and has experienced relatively little alteration by humans. Riparian/riverine habitat areas are characterized by a mixture of plant species that are tolerant to fluctuations in water level. These areas also support the few woody plants that occur on the Hanford Site. Willow are common along the shoreline, as are mulberry, Russian olive, and black cottonwood. Common herbaceous species include sedges and rushes (*Carex* and *Eleocharis* spp.), reed canarygrass (*Phalaris arundinacea*), and bulbous bluegrass (*Poa bulbosa*).

This area is vital to a variety of wildlife, providing food, nesting area, and cover during migration. Migrating waterfowl, such as Canada geese, as well as a number of shorebirds, such as killdeer (*Charadrius vociferus*) and spotted sandpiper (*Actitis macularia*), forage for food in willow thickets. Terrestrial and aquatic insects, abundant near the shoreline, are an important food source for fish and waterfowl. Bald eagles and osprey (*Pandion haliaetus*) fish in the Hanford Reach and perch in the shoreline tree zone. Mule deer and elk browse on mulberry trees and other shoreline plants. Beaver (*Castor canadensis*) and muskrat (*Ondatra zibethica*) den near the river. Raccoons (*Procyon lotor*), mink (*Mustela vison*) and coyotes are also common in riparian habitats (Downs et al. 1993). This area also contains a number of state and federally listed or protected plant species (Table C-1).

Abandoned Fields/Disturbed Habitat

Before 1943, portions of the Hanford Site were farmed using irrigation water from the Columbia River. When the Hanford Site became the scene of Federal nuclear activities farming and grazing of domestic animals ceased. Abandoned farm fields remain, and are characterized by a high density of alien annual species, primarily cheatgrass, jagged chickweed (*Holosteum umbellatum*) and tumble mustard. Some introduced tree species have also survived. Native plants have been very slow to re-establish. The trees provide nesting sites for Swainson's hawks, red-tailed hawks, ferruginous hawks, and kestrels (*Falco sparverius*). Small mammals, such as Great Basin pocket mice, occur in the old-field communities but at lower abundance than in surrounding undisturbed shrub-steppe communities (Gano and Rickard 1982). Mule deer and porcupines *(Erethizon dorsatum)* can be found near the trees in old-field communities. Even though these disturbed areas are used by Hanford Site wildlife, the numbers of species is small compared to those in the shrub-steppe habitat (Downs et al. 1993).

Springs/Streams

Small springs and streams exist on the Hanford Site, primarily on the slopes of the Rattlesnake Hills. Some streams are permanent, others are intermittent. Rattlesnake Springs, near the western edge of the Site, forms a surface stream that flows for approximately 2.5 km (1.6 mi) before disappearing due to seepage (Cushing 1992). Snively Creek, also located on the western edge of the Site, forms a permanent stream that flows for about 2 km along a small canyon. These streams support narrow corridors of shrubs and trees. The dominant tree species at Rattlesnake Springs is peachleaf willow (*Salix amygdaloides*), while at Snively Creek, black cottonwoods (*Populus trichocarpa*), shrub willows, and mock orange (*Philadelphis lewisii*) dominate (Downs et al. 1993). Other shrubs present at both springs include chokecherry (*Prunus virginiana*), golden currant (*Ribes aureum*), wild rose (*Rosa woodsii*), and blue elderberry (*Sambucus cerulea*). Watercress (*Rorippa nasturium aquatica*), rushes, and cattails are also abundant.

Springs and streams on the Hanford Site provide vital drinking water, and the associated riparian habitat provides important cover and forage for many terrestrial species. Mule deer and elk use these sites, as do a number of bird species that do not nest in the surrounding shrub-steppe habitat (Downs et al. 1993). Chukar partridges (*Alectoris chukar*), dark-eyed junco (*Junco hyemalis*), white-crown sparrows (*Zonotrichia leucophrys*), and American robin (*Turdus migratorius*) are commonly found in these areas.

C-1.1.2 Rooting Depths of Plants

Rooting depths of plants are important in evaluating their exposure to buried contaminants. Big sagebrush on the 200 Area plateau has been found to have an average maximum rooting depth of 200 cm (78.8 in.) (±12 cm [4.7 in.] std. error). Other shrubs in this habitat type ranged in depth from 153 to 195 cm (60.2 to 76.8 in.) (Klepper et al. 1985). Large perennial grasses of the big sagebrush/Sandberg's bluegrass habitat have been found with maximum rooting depths of 120 to 140 cm (47.2 to 55.1 in.) (Klepper et al. 1985). Sandberg's bluegrass roots extend to 35 cm (13.7 in.), while cheatgrass roots may extend to 45 cm (17.7 in.) (Link et al. 1990). Other plants of disturbed sites, particularly Russian thistle and ragweed (*Ambrosia acanthicarpa*), have maximum rooting depths of 172 cm (67.7 in.) (±11 cm [4.3 in.]) and 162 cm (63.8 in.) (±8 cm [3.1 in.]), respectively (Klepper et al. 1985). Rickard and Price (1990) found black locust trees to have rooting depths of up to 7.7 m (25 ft). See Table C-2.

C-1.1.3 Hanford Terrestrial Wildlife

Approximately 240 species of terrestrial vertebrates have been observed at Hanford. Included are 40 species of mammals, 187 species of birds, 3 species of amphibians, and 9 species of reptiles. An extensive overview of site-specific species for the Hanford Site is provided by Cushing (1992).

C-1.1.3.1 Mammals. The herbivorous/granivorous mammalian component on the 100 Area is dominated by small mammals, particularly Great Basin pocket mice (*Perognathus parvus*) and deer mice (*Peromyscus maniculatus*). Other abundant herbivores include Townsend's ground squirrels (*Spermophilus townsendii*) and blacktailed jackrabbits (*Lepus californicus*). Near buildings, Nuttall's cottontails (*Sylvilagus nuttallii*) are more abundant than are jackrabbits. The largest herbivore in 100 Area is the mule deer (*Odocoileus hemionus*). The predominant carnivorous mammals are coyotes (*Canis latrans*) and badgers (*Taxidea taxus*).

The most abundant small mammal on the 100 Area is the Great Basin pocket mouse (O'Farrell 1975). Although primarily a granivore, the pocket mouse also consumes insects early in the year before seed production (Kritzman 1974). Pocket mice constitute the principal prey items in the diets of burrowing owls (*Athene cunicularia*), great horned owls (*Bubo virginianus*), long-eared owls (*Asio otus*), and barn owls (*Tyto alba*) that forage on the Hanford Site (Fitzner 1980). Densities may range between 20 and 75 mice/ha (20 and 75 mice/2.47 acres) in April depending on habitat quality of the pocket mouse (Gano and Rickard 1982). Densities in cheatgrass habitat have been estimated at 30/ha (30/2.47 acres) (Hedlund et al. 1975).

The second most abundant mammal on the Hanford Site is the deer mouse. Deer mice are omnivorous, feeding primarily on green vegetation, especially tansy mustard and cheatgrass (Hedlund and Rogers 1976). Although nocturnal, they are found as occasional prey items in the diets of Swainson's hawks (*Buteo swainsonii*) and red-tailed hawks (*Buteo jamaicensis*) nesting on the Hanford Site. More frequently, they are consumed by great horned owls, long-eared owls, burrowing owls, and barn owls (Fitzner 1980).

Townsend's ground squirrels are locally abundant, with peak catches between one half and one fifth that of the Great Basin pocket mouse. Foraging preferences based on analyses of fecal samples include Sandberg's bluegrass and tumble mustard (Rogers and Gano 1980). Townsend's ground squirrels are the principal food item for red-tailed hawks and the second most important item in the diet of post-fledgling Swainson's hawks fledged on the Hanford Site (Fitzner 1980).

Blacktailed jackrabbits are found in nearly all habitats within the shrub-steppe region and are the most common lagomorph on the Hanford Site (Rickard et al. 1974). Blacktailed jackrabbits in the big sagebrush/Sandberg's bluegrass communities feed most heavily on needle-and-thread grass, yarrow (*Achillea millefolium*), turpentine cymopterus, and tumble mustard (*Sysimbrium altissimum*) (Uresk et al. 1975). Blacktailed jackrabbits are the principal prey of golden eagles (*Aquila chrysaetos*) wintering on the Hanford Site (Rickard et al. 1974) and are important constituents in the diets of great horned owls, long-eared owls, barn owls, ferruginous hawks (*Buteo regalis*), Swainson's hawks, and red-tailed hawks (Fitzner 1980).

Mule deer are widespread on the Hanford Site. They are usually dispersed throughout favorable habitats in small groups or singly. Mule deer of the Hanford Site are mainly forb and shrub consumers (Uresk and Uresk 1980). Natural mortality of mule deer fawns on the Hanford Site is relatively high, primarily as a result of coyote predation (Steigers and Flinders 1980). Home ranges vary about a mean of approximately 40 km^2 (15.6 mi^2) (Eberhardt et al. 1982), with densities near the Columbia River of approximately 1 deer/60 ha (Steigers and Flinders 1980).

Coyotes are the most abundant carnivores on the Hanford Site. They have not been studied to any great extent on the Hanford Site with the exception of the Arid Lands Ecology Reserve (ALE). Their diet is diverse, reflecting the availability of prey. Great Basin pocket mice are the primary dietary component for coyotes in areas of prime pocket mouse habitat (Stoel 1976). Other prey include leporids, voles, pocket gophers, ground squirrels, mule deer fawns, birds, reptiles, beetles, and grasshoppers (Steigers and Flinders 1980; Stoel 1976). Coyote density on ALE has been estimated at 1 coyote/2.5 km^2 (1 coyote/0.97 mi^2) (Crabtree 1989); Steigers and Flinders (1980) estimated coyote density near the Columbia River to be 1 coyote/4 km^2 (1 coyote/1.6 mi^2).

C-1.1.3.2 Birds. Bird species observed in the 100 Area are listed in Sackschewsky and Landeen (1992). A general review of birds observed on the Hanford Site along with their respective habitat associations was prepared by Fitzner and Gray (1991).

Bald eagles (*Haliaeetus leucocephalus*) do not nest on the Hanford Site, although nearly 40 bald eagles winter on the Site from November through March (Fitzner and Weiss 1992). Two nesting attempts have been observed near the 100-F Reactor in 1991 and 1992 (Downs et al. 1993). Overwintering eagles on the Hanford Reach of the Columbia River primarily forage on chinook salmon (*Oncorhynchus tschawytscha*) and mallard ducks (*Anas platyrhynchos*) (41 and 39% of dietary biomass respectively) (Fitzner and Hanson 1979). Other foods eaten include other waterfowl and fish species associated with the river (Fitzner and Hanson 1979).

The peregrine falcon (*Falco peregrinus*) is an infrequent visitor to the Hanford Site. This species is not resident, and peregrine falcons do not nest on the Site. Peregrine falcons are occasionally noted during the winter months during migration and may winter in the area, particularly near the Yakima Delta. Peregrine falcons elsewhere are known to feed primarily on birds, although their food habits on the Hanford Site are not known. Peregrine falcons apparently use the Hanford Site primarily as a stopover during migration. Because of their extreme rarity on the Site, and the very short duration of their residence here (days), Hanford Site contaminants are not considered a significant risk to their existence.

Approximately 50 pairs of ferruginous hawks nested in the state of Washington as of 1991 (Fitzner et al. 1992). Ten active ferruginous hawk nests were found on the Hanford Site as of 1991, with eight located in high-tension electric transmission towers and two in trees (Fitzner et al. 1992). These hawks feed primarily on small- to medium-sized mammals such as rabbits and ground squirrels (Fitzner et al. 1981; Howard and Wolfe 1976).

Approximately 15 to 20 pairs of Swainson's hawks nest on the Hanford Site (Fitzner et al. 1981). The birds nest in trees on the Site from April to September. Swainson's hawks feed primarily on snakes, medium-sized mammals, and insects, with yellow-bellied racers (*Coluber constrictor*) being the most important prey (Fitzner 1980).

Red-tailed hawks are the most common hawks nesting on the Hanford Site. At least 20 nesting pairs have occupied the Site. Most nesting occurs in utility towers, on Gable Butte, and in larger trees (Fitzner 1980). Red-tailed hawks on the Hanford Site primarily feed on medium-sized mammals such as black-tailed jackrabbits and Townsend's ground squirrels, and on snakes (Fitzner 1980).

Burrowing owls are widely distributed on the Hanford Site. The nesting population during the mid-1970s was estimated at 20 to 26 pairs (Fitzner et al. 1981). Most nest sites are found in abandoned badger and coyote burrows. No systematic survey of the Hanford Site has been conducted to determine the nesting locations of these birds (Rickard and Poole 1989). These small owls are primarily insect and small mammal predators. Insects represent the majority of prey captures, but Great Basin pocket mice form the major part of their diet in terms of biomass (Fitzner et al. 1981).

Loggerhead shrikes (*Lanius ludovicianus*) are year-round residents on the Hanford Site, although they occur at relatively low densities (Fitzner et al. 1981). They nest from March through August in undisturbed portions of the big sagebrush/Sandberg's bluegrass community, where they average 3.5 pairs/km^2 (3.5 pairs/0.39 mi^2) in the 200 Area plateau (Poole 1992). These medium-sized passerines feed on insects, small mammals, and birds (Fitzner and Rickard 1975).

Great Blue herons nest in the trees along the Columbia River in the 100 Areas. The herons that nest on the Hanford Site feed mostly on Columbia River fish and can serve as biological indicators of chemical contamination in the riparian environment (Weiss and Mitchell 1992). Toxic metals, such as lead, cadmium, and mercury, have been measured in the nest debris (feces and food scraps) at one

Hanford Site heron rookery. The levels of these metals found in herons on the Hanford Site are lower than these reported elsewhere in the Northwest (Fitzner et al 1982). Heavy metal concentrations have also been examined in eggs and in young herons from the Hanford Site. No elevated levels were detected for lead, copper, zinc, or mercury (Weiss and Mitchell 1992).

Sage sparrows (*Amphispiza belli*) are common summer residents of the 200 Area plateau (Fitzner and Rickard 1975). These small passerines are restricted in their distribution almost entirely to sagebrush stands (Schuler et al. 1988). Sage sparrow abundance on the 200 Area plateau has been shown to be related to sagebrush density (Schuler et al. 1988), although abundance may vary widely between years because of natural environmental variation (Rotenberry 1980). Sage sparrows are the second most abundant bird in the undisturbed areas of the 200 Area plateau, reaching densities of 7.5 birds/km^2 (7.5 birds/0.39 mi^2) (Schuler et al. 1988). They forage primarily on phytophagous (plant-eating) beetles and other arthropods, with seeds composing less than 5% of their diet (Rotenberry 1980).

The sage thrasher (*Oreoscoptes montanus*) is confined to areas of big sagebrush cover, where it consumes primarily insects and spiders on the ground rather than in the canopy (Terres 1980). Sage thrashers are resident on Hanford from spring into the fall (Fitzner and Gray 1991), although at very low densities (Schuler et al. 1988).

The most abundant bird found in the shrub-steppe habitat is the western meadowlark (*Sturnella neglecta*) (Brandt and Rickard 1992). Western meadowlarks are present on the Hanford Site throughout the year (Fitzner and Rickard 1975). These passerines nest on the ground from April through July (Brandt and Rickard 1992). Their diet is composed almost entirely of phytophagous insects (Rotenberry 1980). Meadowlark abundance in sagebrush habitat on the Hanford Site was estimated to be approximately 11 birds/km (6.8 birds/mi) (Schuler et al. 1988).

Great Basin Canada geese (*Branta canadensis*) nest on the islands in the Columbia River and forage on the grasses on the Hanford Site (Gano and Rickard 1982; Rickard et al. 1990). Game bird species present of the Hanford Site include the mourning dove (*Zenaida maroura*), California quail (*Callipepla californica*), ring-necked pheasant (*Phasianus colchicus*), Hungarian partridge (*Perdix perdix*), and chukar partridge (*Alectoris chukar*). California quail often can be found near riparian areas and in abandoned orchards near the Columbia River shoreline. The ring-necked pheasant is most often found in association with riparian vegetation near a water source. Hungarian partridge and chukar partridge are common inhabitants of sagebrush-steppe areas above 250 m, but are also found near areas where water is easily accessible (Downs et al 1993). Chukars are concentrated in the vicinity of West Lake and along the spring streams in the Rattlesnake Hills. Rock doves (*Columbia livia*) have been found nesting at abandoned buildings in the reactor areas.

In 1977, approximately 60 pairs of long-billed curlews (*Numenius americanus*) were estimated to have been nesting in the 600 Area of the Hanford Site (Allen 1980). Curlews nest from April through June in relatively flat areas dominated by cheatgrass. They feed primarily on beetles and subterranean insect larvae (Allen 1980). No systematic surveys of the Site have been conducted since Allen's study (Fitzner et al. 1992).

C-1.1.3.3 Reptiles and Amphibians. Reptiles that occur on the Hanford Site include the western yellow-bellied racer (*Coluber constrictor*), the Great Basin gopher snake (*Pituophis melanoleucus*), the northern Pacific rattlesnake (*Crotalus viridus*), the desert nightsnake (*Hypsiglena torquata*), a state monitor species, the striped whipsnake (*Masticophis taeniatus*, a state candidate species), the sagebrush lizard (*Sceloporus graciosus*), the side-blotched lizard (*Uta stansburiana*), and the pygmy short-horned lizard (*Phrynosoma douglassi*) (Rogers and Rickard 1977). The most common reptiles

found in the big sagebrush/Sandberg's bluegrass habitat are side-blotched lizards and yellow-bellied racers (Marr et al. 1988). Side-blotched lizards were found in approximate densities of 15 lizards/ha (15 lizards/2.47 acres) in the 100-DR-1 Operable Unit crib area in the 1970s (Rogers and Rickard 1977).

C-1.1.3.4 Insects. Insects and spiders are an important component of the plant communities in terms of biomass and ecological role. Invertebrate densities in sagebrush/bunchgrass habitat on ALE ranged from 450 to nearly 2,000 individuals/m^2 (10.7 ft^2), with a biomass of up to 0.5 g/m^2 (Rogers 1977). The predominant taxa include ground-dwelling darkling beetles (Family Tenebrionidae), and shrub-dwelling bugs (Order Homoptera), grasshoppers (Order Orthoptera), true bugs (Order Hemiptera), and spiders (Order Araneida) (Rogers 1979). Harvester ants, a potentially significant component in the exposure of buried contaminants (Fitzner et al. 1979), were observed on the waste sites within the 100 Area (Sackschewsky and Landeen 1992).

C-1.1.4 Hanford Aquatic Resources

The Columbia River is the principal aquatic ecosystem found at Hanford and supports a diverse community of plankton, benthic invertebrates, and fish species. It is the fifth largest river in North America, traveling approximately 1,240 miles from the source in Canada to its mouth at the Pacific Ocean (Cushing 1991). The Hanford Reach is controlled by seven upstream dams, the nearest of which is Priest Rapids, about 12 river miles (Rmi) [19 river kilometers (Rkm)] upstream.

Other perennial water sources that provide aquatic habitat include Rattlesnake Springs, Snively Springs, and several artificial water sources including ponds and ditches. Ponds located north of the Columbia River are the result of irrigation runoff and support typical pond vegetation such as cattails (*Typha lattifolia*). Other ponds located onsite are more temporary and depend on Site waste-water as a source.

C-1.1.4.1 Aquatic Plants. Some of the typical aquatic vascular plant species include water milfoil (*Myriophyllum* spp.), water smartweed (*Polygonum amphibium*), pondweed (*Potamogeton* spp.), persistent sepal yellowcress (*Rorippa columbiae*), watercress (*Rorippa nasturium-aquatic*), and duckweed (*Lemna* spp.) (Cushing 1992). Three groups of plants constitute the primary producers of the aquatic ecosystem: phytoplankton, periphyton, and macrophytes.

Phytoplankton are floating, free-living algae drifting with the current in the water column, and periphyton are predominately algae colonizing solid substrata, such as rocks. Although the dominant phytoplankton species in the Columbia River are true lentic (lake) forms, many species in the water column are detached periphytic forms that have been washed off of rocks. The periphyton mat commonly found on solid substrata is made up of algae and other organisms; these include microcrustaceans, rotifers, fungi, bacteria, and detritus. These communities are more abundant at the margins of the river near the 100 Area where conditions are suitable. Phytoplankton and periphyton are present year-round in the Columbia River; populations are highest in spring and summer and lowest in winter.

Macrophytes can be found rooted to the bottom of the river where the current slackens and fine sediments accumulate in sufficient amounts. Macrophytes are sparse in the Columbia River because of the strong currents, rocky bottom, and frequently fluctuating water levels. Rushes and sedges occur along the shorelines of the slack-water areas such as the White Bluffs Slough below the 100-K Area, the slough area downstream of the 100-F Area, and the Hanford Slough. Macrophytes are also

present along gently sloping shorelines that are subject to flooding during the spring freshet and daily fluctuating river levels. Macrophytes are present during the warmer months and usually die in the winter.

Commonly found macrophytes include *Lemna, Potamogeton, Elodea*, and *Myriophyllum*, and they have considerable ecological value. Macrophytes are most important as food after they die and decompose into fine particulate detritus. These macrophytes provide food and shelter for juvenile fish and spawning areas for some species of warm-water game fish. However, if some of the exotic macrophytes increase to nuisance levels, they may encourage increased sedimentation of fine particulate matter. This could negatively effect the spawning of salmonids but could increase the range for American shad (*Alosa sapidissima*) by providing more suitable spawning habitat. These changes could significantly impact the trophic relationships of the Columbia River.

C-1.1.4.2 Herbivores. Organisms that directly feed on the primary producers (usually periphytes) are herbivores. The common herbivores in the area near the 100 Area include zooplankton, immature insects, molluscs, and herbivorous fish. Zooplankton, insects, molluscs, and herbivorous fish are present at all times. The zooplankton are not abundant in this reach of the river. Molluscs are neither abundant nor important in terms of energy flow in the ecosystem. However, two species found in the Columbia River are listed as candidates for protection under the Endangered Species Act. These are the shortface lanx (*Fisherola nuttalli*), which is a state candidate, and the Columbia pebblesnail (*Fluminicolla columbiana*), which is both a federal and state candidate. Herbivorous fish, such as some species of suckers, actively graze on the periphyton; Dauble (1986) reported that large-scale suckers (*Castostomus macrocheilus*) in the Columbia River feed predominantly on periphyton and insect larvae.

C-1.1.4.3 Primary carnivores. Primary carnivores feed on the herbivores. Dominant groups found in the Columbia River include several species of forage fish, mountain whitefish (*Prosopium williamsoni*), and juvenile salmonids. The carnivores in this group use several different sources of food. This group includes several species of primary concern from an economic, sport, and protected species viewpoint. These are the salmonids, including steelhead (*Oncorhynchus mykiss*) and the various species of salmon. The steelhead provides major sport fishing in and above the 100-KR-4 Operable Unit. Because the Hanford Reach is the last mainstream spawning area for both steelhead and salmon, the potential impacts to these migrating populations must be considered.

C-1.1.4.4 Secondary Carnivores. Secondary carnivores feed on a variety of sources, but mainly the primary carnivores. This category includes species such as smallmouth bass (*Micropterus dolomieui*), and other organisms in the vicinity of the river, such as bald eagles (*Haliaeetus leucocephalus*), hawks, and swallows.

C-1.1.4.5 Omnivores. Omnivores feed on both animals and plants. Crayfish are omnivorous and feed on decaying animal and plant tissue. Waterfowl are also omnivores, feeding on macrophytes and primary carnivores. Immature aquatic insects are one of the basic food items and consist of the larvae and nymphs of several orders of insects. The aquatic insects are usually most abundant during fall and winter where they mature until they emerge as adults in spring and summer. Immature insects are most important as a food source in the aquatic system but are also important as adults for insectivorous birds, such as swallows. Insects also enter the aquatic food web after they die if they fall back into the river.

Forty-four species of fish are known to occur in the Columbia River (Cushing 1992). Among these are five species of anadromous fish, warm-water game species such as bass (*Micropterus* spp.),

crappie (*Pomoxis* spp.), and perch (*Perca* spp.), and large populations of rough fish such as carp (*Cyprinus carpio*) and squawfish (*Ptychocheilus oregonensis*). Economically important salmonids like chinook salmon (*Oncorhynchus tshawytscha*), sockeye salmon (*Onchorynchus nerka*), coho salmon (*Oncorhynchus kisutch*), and steelhead use the Columbia River as a migration route to and from spawning areas in and near the Hanford Reach of the Columbia River (Cushing 1992).

Fall chinook salmon and steelhead spawn in the Hanford Reach. The relative contribution of upper river stocks to fall chinook salmon runs in the Columbia River increased from about 24% of the total in the early 1980s to 50% to 60% of the total by 1988 (Dauble and Watson 1990). The destruction of other mainstream Columbia River spawning grounds by dams has increased the importance of the Hanford Reach spawning (Watson et al. 1970; Watson 1973). The upper estimates of the annual average Hanford Reach steelhead spawning population based on dam counts from 1962 to 1971 were about 10,000 fish. The estimated annual sport catch from 1963 to 1968 in the reach of the river from Ringold to the mouth of the Snake River was approximately 2,700 fish (Watson 1973).

The American shad, an anadromous species, may also spawn in the Hanford Reach. The upstream range of the shad has been increasing since 1956 when fewer than 10 adult shad ascended McNary Dam. Since then, the number ascending Priest Rapids Dam, immediately upstream from the Hanford Reach, has risen to many thousands per year and the young-of-the-year have been collected in the Hanford Reach. The shad is not dependent on specific current and bottom conditions required by the salmonids for spawning and has apparently found favorable conditions for reproduction throughout much of the Columbia and Snake rivers.

Other fish of importance to sport fishers are the mountain whitefish (*Prosopium williamosi*), lake whitefish (*Lampetra ayresi*), white sturgeon (*Acipenser transmontanus*), smallmouth bass, white and black crappie (*Pomoxis annularis* and *nigromaculatus*), channel catfish (*Ictalurus punctatus*), walleye (*Stizostedion vitreum*), and yellow perch (*Perca flavescens*). White sturgeon are a long-lived scavenger and as adults do not move above or below dams. Extensive work has been done to determine the radionuclide uptake of strugeon (Dauble et al. 1993).

C-1.1.5 Threatened and Endangered Species

Species of concern include federally listed threatened or endangered species, state-listed species, and other categories of sensitive species of plants and animals. Sixty-three animal species of concern are known to occur or potentially occur at Hanford (Cadwell 1994). These include 12 species of mammals, 41 species of birds, 4 species of fish, 2 species of molluscs, 1 species of amphibian, and 2 species of reptiles.

The pygmy rabbit *(Sylvilagus idahoensis)* is currently a state endangered and federal candidate species (Cushing 1992). It requires dense stands of sagebrush found on the shrublands for food and cover. This species has not been sighted at Hanford since 1984 when wildfires destroyed large areas of sagebrush habitat.

Federally listed raptor species include the bald eagle and peregrine falcon. These species are the only federally listed animals known to occur on the Hanford Site. The bald eagle is federally and state listed as threatened. It is a common winter migrant that uses the tree zone along the Columbia for perching and roosting. The peregrine falcon is federally and state listed as endangered, and occurs at Hanford as a migrant. Occurrence of the peregrine falcon at Hanford is rare, with the last documented sighting reported in 1993 (Fitzner et. al 1994). The peregrine prefers open country for hunting prey, such as that found along the Columbia River.

Other sensitive raptors at Hanford include the ferruginous hawk, Swainson's hawk, and red-tailed hawk. The ferruginous hawk is a federal candidate for listing and state threatened species, and the Swainson's hawk is a state candidate species. Red-tailed hawks, although not a special status-species, are currently being monitored at Hanford because of a decline in nesting.

The sandhill crane (*Grus canadensis*) and American white pelican (*Pelecanus erythrorhnchos*) are listed as state endangered species. Both species occur at Hanford as migrants and are generally associated with aquatic habitat. Pelicans at Hanford utilize sloughs and protected waters around islands in the Columbia River. Sandhill cranes have mainly been observed migrating over Hanford, with few sightings of cranes actually landing onsite.

All six aquatic species of concern at Hanford occur in the Columbia River. The Columbia pebblesnail (*Fluminicola columbiana*) is both a federal and state candidate species. The shortface lanx (*Fisherola nuttalli*) is a state candidate species. Four species of nongame fish known to occur in the Hanford Reach of the Columbia are state monitor species. Included are the mountain sucker (*Catostomus platyrhynchus*), piute sculpin (*Cottus beldingi*), reticulate sculpin (*Cottus perplexus*), and sand roller (*Percopsis transmontana*).

At the present time, 25 species of plants found at Hanford are listed as either state threatened or endangered species, federal candidates for listing, or state sensitive or monitor species (Cadwell 1994). None are currently listed as federally threatened or endangered species.

Wetland plant species of concern include five species known to occur along the Columbia River. Included in this group are the Columbia yellowcress (*Rorippa columbiae*), a state endangered and federal candidate species, and four state sensitive species: southern mudwort (*Limosella aquatica*), dense sedge (*Carex densa*), false pimpernel (*Lindernia anagallidea*), and shining flatsedge (*Cyperus rivularis*).

Other plant species of concern include six plants that may occur on upland areas. The northern wormwood (*Artemisia campestris* spp. *borealis* var. *wormskioldii*), a state endangered and federal candidate species has not been found on the Hanford Site, but is found near the Columbia River just north of Beverly, Washington (Sackschewsky et. al 1992). Additional plant species of concern include the Columbia milk-vetch (*Astragalus columbianus*) and Hoover's desert parsley (*Lomatium tuberosum*), both state threatened and Federal candidate species. Piper's daisy (*Erigeron piperianus*), gray cryptantha (*Cryptantha leucophaea*), and dwarf desertprimrose (*Camissonia pygmaea*) are state sensitive species.

C-2.0 SPECIFIC PARAMETERS FOR RECEPTOR SPECIES

This section provides specific information on selected receptor species analyzed for the Hanford Site ecological evaluations.

C-2.1 TERRESTRIAL RECEPTOR: THE GREAT BASIN POCKET MOUSE

The most abundant and frequently studied small mammal on the Hanford Site is the Great Basin pocket mouse (*Parognathus parvus*) (O'Farrell 1975). For these reasons, the pocket mouse has been

selected as a receptor for the terrestrial operable units. The pocket mouse is found throughout the shrub-steppe habitat, but prefers open, shrub-dominated areas with understories of cheatgrass (*Bromus tectorum*) or Sandberg's bluegrass (*Poa sandbergii*). Pocket mice are primarily granivorous, feeding on cheatgrass and other grass seeds. Pocket mice are also known to feed on shoots of Russian thistle (*Salsola kali*) and on leaves and stems of *Descurainia* (Kritzman 1974). In early spring, before grass seeds have ripened, pocket mice feed on arthropods (Kritzman 1974). Pocket mice can sustain themselves on only the water found in their food supply; they do not require free water (Rickard 1974). Average body weight of pocket mice is 23 g (Burt and Grossenheider 1976).

Pocket mice are nocturnal, burrowing mammals. Most burrows are between 35 and 193 cm (1.2 and 6.3 ft) deep, depending on soil texture and the soil frost line (Gano and Rickard 1982). Pocket mice are torpid from December to February, and remain in the deepest areas of the burrows during these months (O'Farrell 1974). Males generally emerge in March, followed by females one month later in April. Evidence suggests that emergence patterns are based on seasonal changes in soil temperature rather than on a response to photoperiod (O'Farrell 1974). Pocket mice are active aboveground for an average of 60 to 90 days in the spring, and return to periodic states of torpor underground throughout the summer months (O'Farrell 1974). The length of time spent active depends upon the adequacy of the food supply.

Home ranges for pocket mice vary, from 1,560 to 4,005 m^2 for males, and 508 to 2301 m^2 for females. Fluctuation in home range sizes is due to population density and food availability (O'Farrell 1974). Pocket mice breed in the spring; young are born in midsummer. In years with good moisture availability, two litters can be produced (Rickard 1974). Densities range from 80 to 115 individuals per hectare in April (O'Farrell 1974).

C-2.2 AQUATIC RECEPTOR

There are multiple receptor species found in the Columbia River environment that have been chosen for evaluation within aquatic operable units. These receptors represent a generic sampling of different trophic levels. Using LOELs and a dose modeling code (CRITR2), evaluations for potential ecological risk within the aquatic foodweb can be conducted. The sampling includes a generic aquatic plant, a generic fish, a crustacean, a plant-eating duck, a fish-eating duck, and a heron, rather that one specific species. In this matter, the input into the CRITR2 model is averaged for all species at that trophic level. For information on dose calculations, please see Appendix E.

C-3.0 THE CONCEPTUAL MODEL OVERVIEW

The intent of the site-wide ecological conceptual model is to provide a working model to illustrate ecological foodwebs at Hanford to be used to perform ecological evaluations. These foodwebs are intended to show energy flow in the terrestrial and riparian ecosystems. The purpose of the conceptual model is to identify important foodwebs and linkages that can be utilized in selecting ecological receptors for ecological risk assessments. The conceptual models display ecosystems potentially at risk at the Hanford Site. No attempt has been made to describe exposure scenarios because they are dependent on site-specific stressors. Key ecological receptors are not identified because they are dependent on the objective of the ecological risk assessment. Foodwebs are presented to provide guidance on selecting ecological receptors to meet the objective of particular risk assessment. The foodwebs also provide background information for selection of indicator species that can be linked to a guild or particular species of interest.

Foodwebs have been developed based upon identification of potential ecological receptors by habitat type. For example, the major habitat types at Hanford are grasslands, shrublands, riparian and transition zones between the different habitat types. The type of wildlife present in each habitat type is dependent on the vegetation present. Many receptors are common to more than one habitat type, while many are confined to only one habitat.

The potential exposure of an ecological receptor to contaminants within a waste site is related to the size of the animal's home range. The degree of exposure is proportional to an area of the waste site relative to the size of the receptor's home range, amount of food consumed from within the waste site, the type of food, and the concentration and bioavailability of contamination present.

A generic foodweb is presented in Figure C-2. Shown is transfer of a contaminant from soil to primary producers to herbivore to carnivore. This foodweb is provided to give a frame of reference for developing stressor exposure scenarios. A more specific foodweb is shown in the Figure C-3. The ecological foodweb is centered on Hanford grasslands. One of the herbivores of the grassland community is a grasshopper. Figure C-4 shows the relationships of the grasshopper to other inhabitants in the grassland. Figure C-5 shows the same relationships for the chukar in the grassland. The figures show energy flow from grass/shrubs to various trophic levels. Many species are common to both grass and shrublands. An important aspect of Figure C-3 is how a particular species may be contaminated by one or more stressors. For example, a mule deer feeding on contaminated grass would receive direct exposure to hazardous materials that have accumulated in vegetation. In contrast, a raptor would receive exposure to contaminants after transport of a contaminant through two trophic levels.

A series of foodwebs centered on the Columbia River riparian zone are shown in Figure C-6. This figure shows a number of different foodwebs that bridge both terrestrial and riparian habitats besides foodwebs that are uniquely terrestrial or riparian.

C-3.1 APPLICATION OF THE FOODWEBS

The foodwebs provide a means to select receptors for use in ecological risk assessment. The selection of appropriate receptors should lead to identification of assessment and measurement endpoints. Listed below is specific information regarding stressor transport pathways and sensitivity of potential receptors to ionizing radiation.

C-3.1.1 Ingestion Pathway

Various foodwebs are presented to assist in the preparation of ecological risk assessments by identifying those species and trophic levels considered most important. The selection of ecological receptors will depend upon habitat type. Once the required habitat type has been identified, the exposure pathway can be completed for key ecological receptors. For both the terrestrial and riparian ecosystems, the major pathways of contamination are soil to primary producers and related ingestion pathways.

C-3.1.2 Water Consumption Pathway

The importance of the drinking water pathway for exposure to site contaminants has yet to be resolved. There are few areas of open water on Hanford, except for the Columbia River, West Lake, D-pond, a very few sites in the 200 areas, and springs in the ALE; the sandy soils and low precipitation preclude the formation of ponds near waste sites. To date, no studies have been completed to evaluate the importance of drinking water as a pathway of receptor contamination. For some riparian species such as the Canada geese, water consumption is important and should be evaluated, for other receptors such as raptors, the drinking water requirements are not known. An assessment as to whether certain species require drinking water is presented in Table C-2.

C-3.1.3 Rooting and Burrowing Depths

Table C-1 presents a summary of plant rooting depths for various species of Hanford plants. This information should provide a basis for modeling uptake of radionuclides and hazardous chemicals by specific species of plants. Similarly, available information on burrowing depths of insects and animals is also provided in Table C-2.

C-3.1.4 Receptor Sensitivity to Ionizing Radiations

A summary of sensitivities of various organisms to ionization radiation is shown in Table C-3. Presented are classes of organisms, safe exposure ranges and ranking of general sensitivity. In general, the more complex the organism the more sensitive it is to ionizing radiation.

C-3.1.5 Ecological Parameter Database

Currently several databases are being developed to contain ecological parameters for specific organisms, such as home range, weight, food consumption rates, etc. This database will contain information on mammals, birds and plants. In addition to providing organism specific factors for risk assessment the database will be useful in identifying those species specific data gaps.

C-3.1.6 Factors to Consider in Selecting Receptors

Listed below are factors to be considered when selecting receptors for ecological risk assessment. For example, depending upon the objectives of the assessment one or more receptors would be appropriate. In a comprehensive risk assessment, several taxa should be considered. Other factors that should be considered in the selection of a receptor species include trophic level position, frequency of occurrence, legal status, availability of species-specific data, size of the animal's home range, contaminant sensitivity, and likelihood of exposure.

Taxa

- Reptiles
- Invertebrates
- Mammals
- Birds
- Fish

Trophic Level

- Primary Producer
- Level 1 Consumer (herbivore, granivore)
- Level 2 Consumer (carnivore)
- Level 3 Consumer (carnivore/top predator)

Frequency of Occurrence

- Common
- Occasional
- Rare
- Migratory

Legal Status

- Federal
- State
- Other

Availability of Species-Specific Data

- Food habits
- Home range size
- Population density

C-3.1.7 Other Conceptual Model Considerations

Current Waste Site Conditions. Most waste sites are under institutional control and measures have been taken to prevent vegetation growth within the waste sites. Vegetated areas are, however, located between waste sites. The risk to any receptor will therefore be a function of the exposure scenario assumptions. For example, if it is assumed that a waste site is vegetated, the pocket mouse may be assumed to ingest contaminated plant matter.

C-4.0 LOELs and NOELs and FACTORS INVOLVED IN ESTIMATES OF ENVIRONMENTAL TOXICITY

A Lowest Observed Effects Level (LOEL) refers to the lowest dose of a contaminant found, under a given set of parameters (amount per unit time, age and sex of test animal, duration of testing), to promote an observable effect (i.e., death, injury, or behavior modification).

Those concentrations or doses which do not produce effects within the animals are referred to as No Observable Effects Levels (NOEL). NOELs are lower than the LOELs. The difference between a LOEL and NOEL is dependent on the dose intervals utilized in the toxicity test.

The geometric mean between the NOEL and the LOEL is sometimes referred to as the Maximum Acceptable Toxicant Concentration (MATC). Any of these three levels can be used depending on the conservatism desired.

The geometric mean between the NOEL and the LOEL is sometimes referred to as the Maximum Acceptable Toxicant Concentration (MATC). Any of these three levels can be used depending on the conservatism desired.

C-4.1 OTHER TOXICOLOGICAL TERMS

The definition of several terms and abbreviations used for qualitative or quantitative risk assessments are fairly straight forward although the exact application is often subjective in nature. Selected toxicity terms are discussed below. It should be noted that "concentration" refers to the toxicant in the environment (air or water) whereas, "dose" refers to the introduction of the toxicant into the test species.

- EC_{50} - Median Effective Concentration. The concentration of material in water to which test organisms are exposed that is estimated to be effective in producing some sublethal response in 50% of the test organisms (Rand and Petrocelli 1985)

- ED_{50} - Median Effective Dose. The dose of material estimated to be effective in producing sublethal response in 50% of the test organisms (Rand and Petrocelli 1985).

- NOAEL - No Observed Adverse Effect Level. The highest concentration of a toxicant that does not elicit an observable adverse effect on the test species.

- LOAEL - Lowest Observed Adverse Effect Level. The lowest concentration of a toxicant that elicits an adverse effect on the test species.

- Radiation Dose Rate. In the context of the ecological risk assessments, this refers to the radiation absorbed by the receptor species per unit time.

- LD_{50} - Lethal Dose Fifty. Is the estimated dose that, when the toxicant is administered directly to experimental test animals, results in the death of 50% of the population so exposed under the defined conditions of the test (Hodgson and Levi 1987).

- LC_{50} - Median Lethal Concentration. Is the estimated concentration, in the environment to which animals are exposed, that will kill 50% of the population so exposed under the defined conditions of the test (Hodgson and Levi 1987).

C-4.2 ECOLOGICAL RISK FACTORS

The ultimate objective of ecological risk assessments is usually to determine not the "safe' dosage for laboratory animals, but the "safe" dosage for actual species found in an environmental setting. Wildlife NOELs estimated from laboratory animals must consider factors such as physiological, biochemical, and behavioral differences between differing genuses and families, and even within the same species. Historically, the standard approach has been to reduce the NOEL by a safety or uncertainty factor that considers both intraspecies and interspecies differences (Klaassen and Eaton 1991).

A general method for extrapolations to wildlife from laboratory values that is itself based on EPA methodology for deriving human toxicity values from animal data is found in Opresko et al. (1993). Their methodology included:

- Identification of known NOAEL and LOAEL data on a dose per unit body <weight (e.g., mg/kg)
- Scaling of the dose to that of the chosen "wild" species bases on a weight/body surface area ratio. This employs time-weighted average body weights for the entire life span of a species for a chronic exposure scenario.
- Calculation of dietary levels resulting in a dose equivalent to the NOAEL from various food consumption rates. The food consumption rates were estimated from allometric regression models based on metabolic rates (Nagy 1987) and varied for such types as placental mammals, rodents, herbivores, marsupials, birds, and passerine birds.
- Calculations of dose rates equivalent to NOAELs for drinking water based on daily water consumption values and average body weights for each species. Water consumption rates can be estimated from allometric regression models derived from experimental data for mamallian wildlife.
- Dietary calculations for wildlife species (such as mink and otter) which feed primarily on aquatic organisms and the concentration of the contaminant in the food are proportional to the concentration in the water. In this case the food consumption rate is modified by a bioaccumulation factor derived from the ratio of the concentration in tissue to the concentration in water.
- If unknown, bioaccumulation factors were predicted through multiplication of the bioconcentration factor (ratio of concentration in food to concentration in water) by an appropriate food chain multiplying factor. For most inorganic compounds the bioconcentration factors and the bioaccumulation factors are assumed to be equal.

This approach greatly improves on the calculation of acceptable risk factors to wildlife and then ultimately to humans themselves. However, much of the basic information needed for the calculations are often unavailable or is itself the product of estimates. Therefore EPA has often recommended uncertainty factors and included additional "modifying factors" that allow for "professional judgment" in the estimation of allowable levels. Most regulatory agencies consider "safety factor" or uncertainty factors of at least 100 or 1,000 adequate if the data upon which the NOEL was calculated is inadequate.

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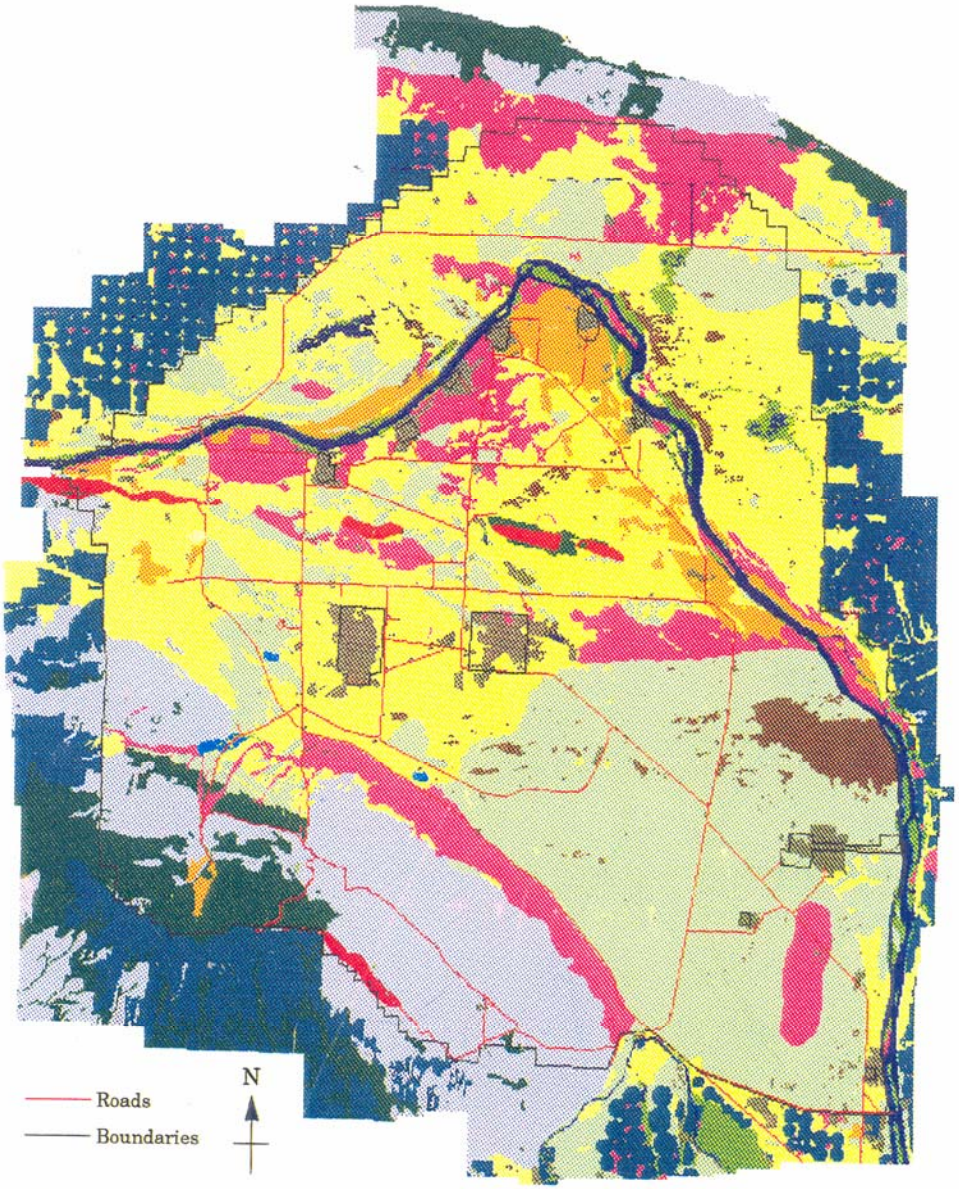
Figure C-1. Vegetation/Land Use Cover Map for the Hanford Site.

(November 1, 1993, Rev. 0)*



*Based on 1987 and 1991 aerial photography.
Map subject to revision as additional survey data become available and as cover types change in response to natural and human-caused events.

Figure C-2. Generic Conceptual Model for Terrestrial/Riparian Foodwebs.

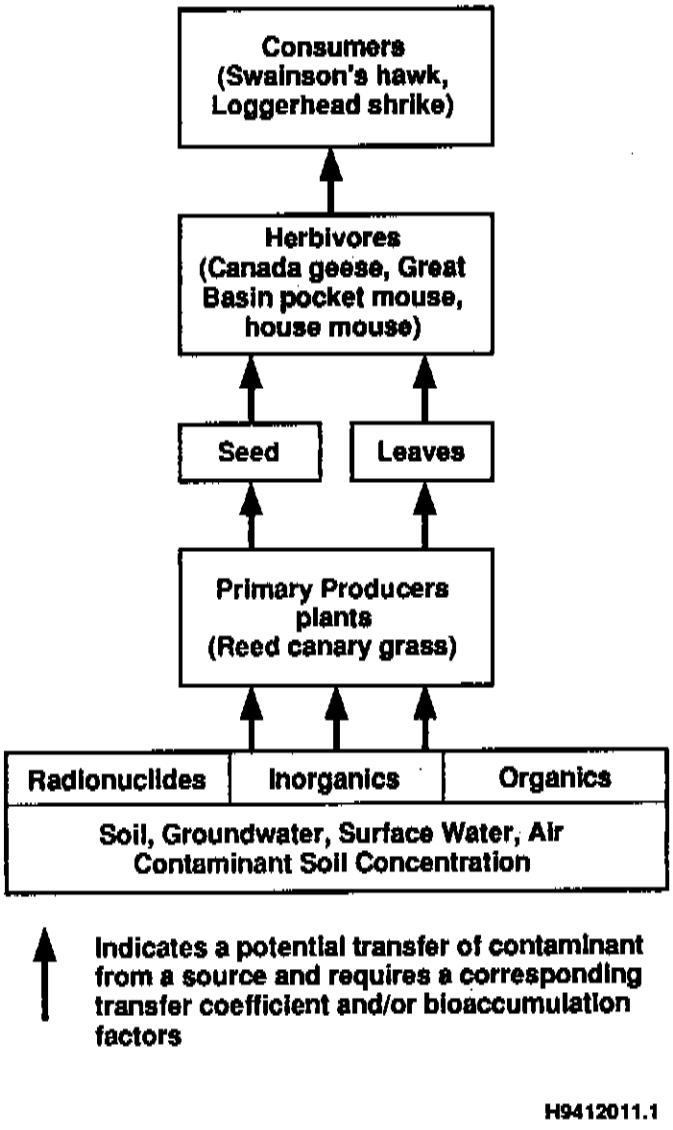


Figure C-3. Foodweb for Grassland Habitat Type.

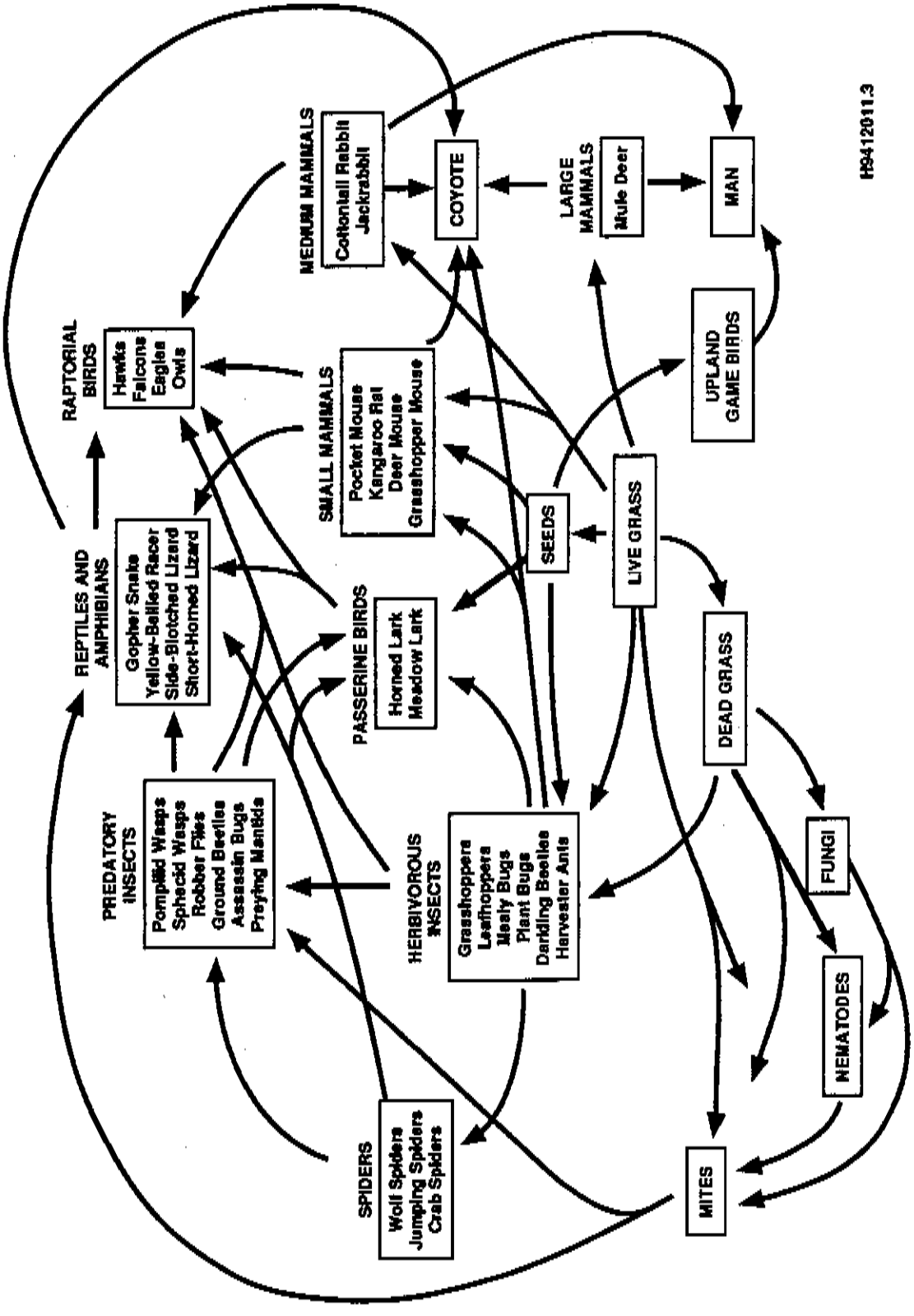


Figure C-4. Grassland Foodweb: Grasshopper.

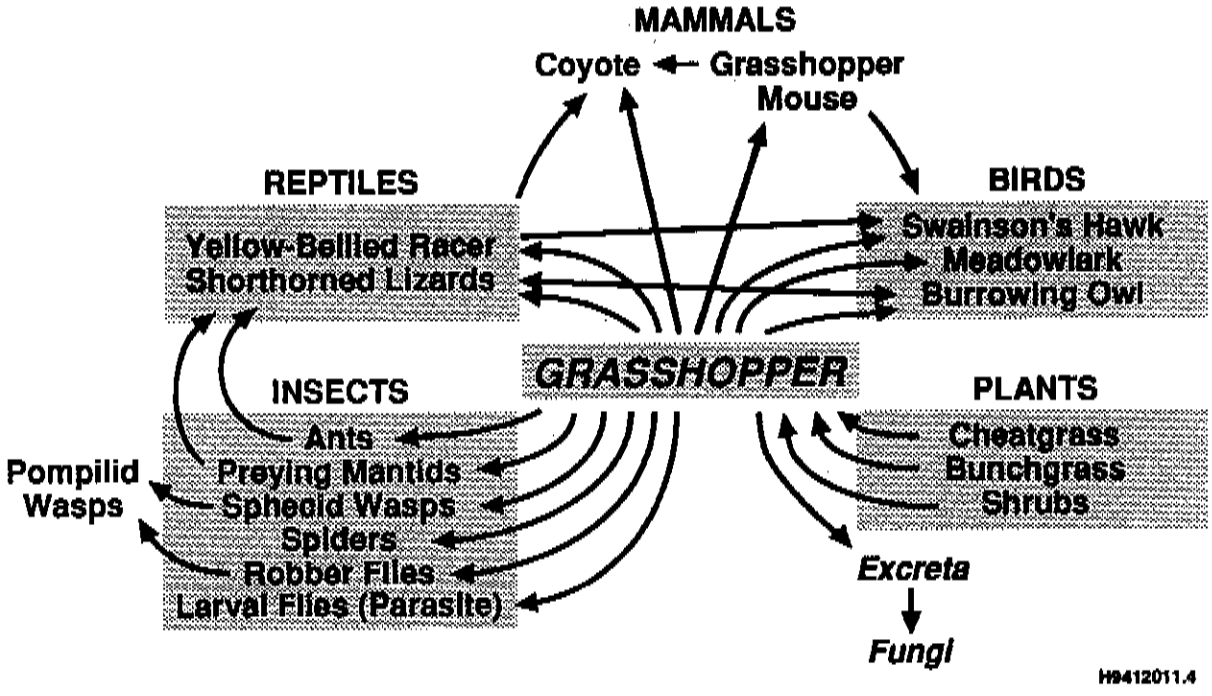


Figure C-5. Grassland Foodweb: Chukar.

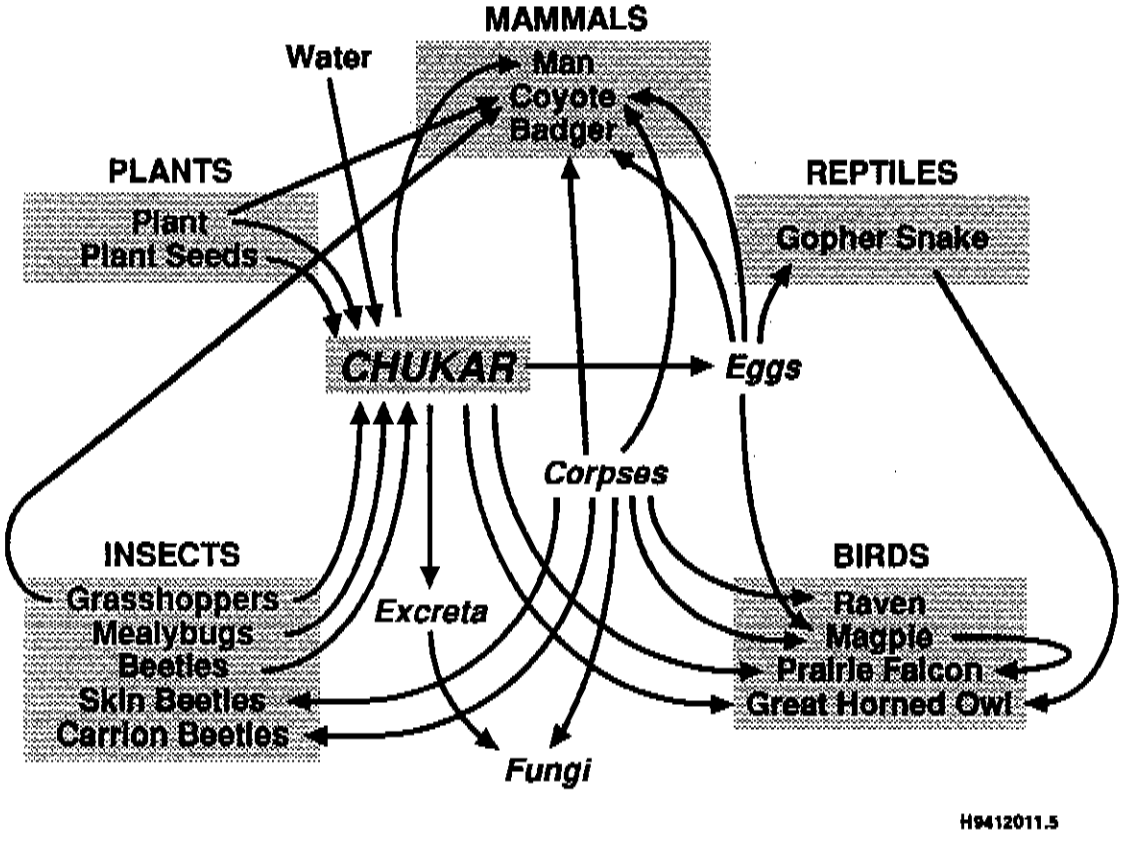


Figure C-6. Foodweb for Riparian and Aquatic Ecosystems.

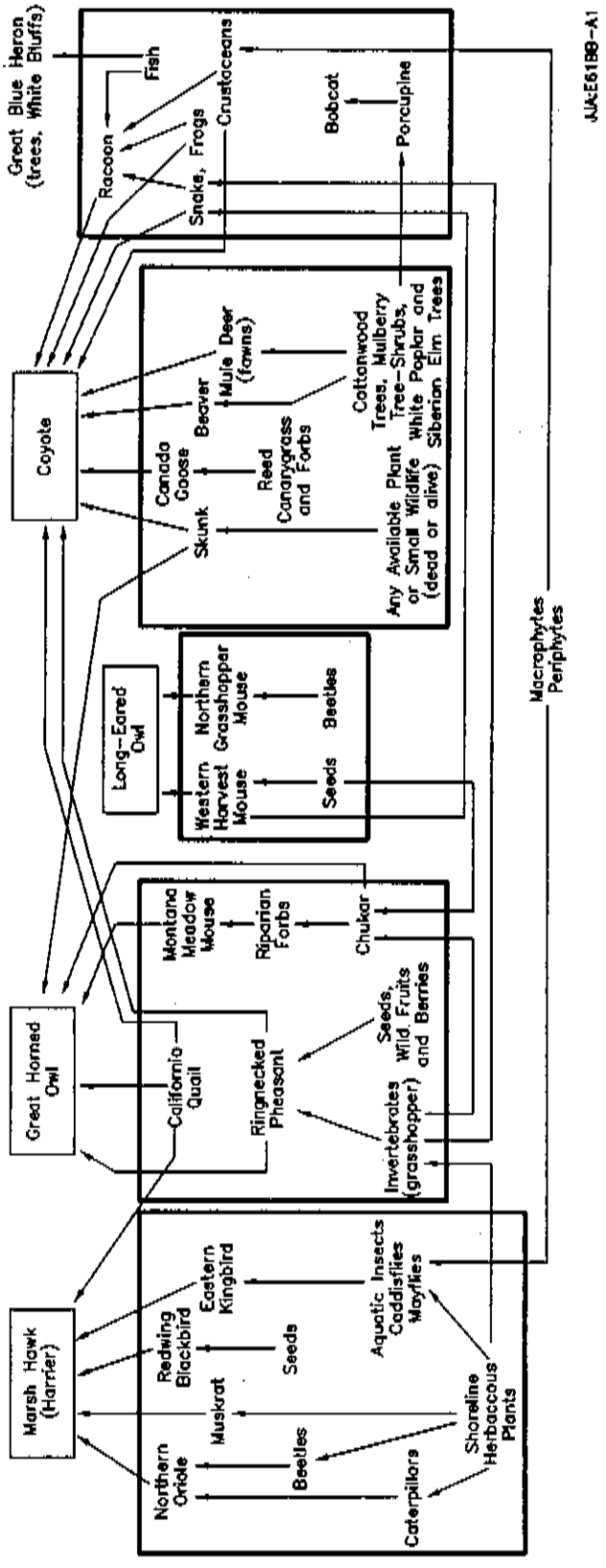


Table C-1. Hanford Site Sensitive Species. (4 sheets)

Common Name	Latin Name	Federal Status	State Status	Abundance[a]	Habitat Assn.
BIRDS					
Sandhill crane	Grus canadensis		Endangered	UC	Wetlands/Meadows
American white pelican	Pelecanus erythrorhynchos		Endangered	C	Riparian
American peregrine falcon	Falco peregrinus anatum	Endangered	Endangered	R	Entire Site
Aleutian Canada goose	Branta canadensis leucopareia	Threatened	Endangered	R	Columbia River
Bald eagle	Haliaeetus leucocephalus	Threatened	Threatened	C	Riparian
Ferruginous hawk	Buteo regalis	Candidate 2	Threatened	UC	Shrubsteppe
Sage sparrow	Amphispiza belli		Candidate	C	Shrubsteppe
Golden eagle	Aquila chrysaetos		Candidate	R	Entire Site
Burrowing owl	Athene cunicularia	Candidate 2	Candidate	C	Shrubsteppe/Open Shrubstepp
Swainson's hawk	Buteo swainsoni	(b)	Candidate	C	Trees/Shrubsteppe
Common loon	Gavia immer		Candidate	UC	Columbia River
Lewis' woodpecker	Melanerpes lewis		Candidate	R	Trees
Sage thrasher	Oreoscoptes montanus		Candidate	UC	Shrubsteppe
Flammulated owl	Otus flammeolus		Candidate	R	Shrubsteppe
Western bluebird	Sialia mexicana		Candidate	R	Entire Site
Northern goshawk	Accipiter gentilis	Candidate 2	Candidate	R	Shrubsteppe
Western sage grouse	Centrocercus urophasianus phaios	Candidate 2	Candidate	R	Shrubsteppe
Loggerhead shrike	Lanius ludovicianus	(b)	Candidate	C	Shrubsteppe
Clark's grebe	Aechmophorus clarkii		Monitor	R	Columbia River
Western grebe	Aechmophorus occidentalis		Monitor	UC	Columbia River
Grasshopper sparrow	Ammodramus savannarum		Monitor	UC	Shrubsteppe
Great blue heron	Ardea herodias		Monitor	C	Riparian
Great egret	Casmerodius albus		Monitor	R	Riparian
Turkey vulture	Cathartes aura		Monitor	R	Entire Site
Merlin	Falco columbarius		Monitor	R	Entire Site
Prairie falcon	Falco mexicanus		Monitor	UC	Shrubsteppe
Gyrfalcon	Falco rusticola		Monitor	R	Entire Site
Black-necked stilt	Himantopus mexicanus		Monitor	R	Riparian
Ash-throated flycatcher	Myiarchus cinerascens		Monitor	R	Entire Site
Snowy owl	Nyctea scandiaca		Monitor	R	Entire Site
Black-crowned night heron	Nycticorax nycticorax		Monitor	UC	Riparian

Table C-1. Hanford Site Sensitive Species. (4 sheets)

Common Name	Latin Name	Federal Status	State Status	Abundance[a]	Habitat Assn.
Osprey	Pandion haliaetus		Monitor	UC	Riparian
Horned grebe	Podiceps auritus		Monitor	UC	Columbia River
Red-necked grebe	Podiceps grisegena		Monitor	R	Columbia River
Caspian tern	Sterna caspia		Monitor	C	Columbia River
Forster's tern	Sterna forsteri		Monitor	C	Columbia River
Arctic tern	Sterna paradisaea		Monitor	R	Columbia River
Barred owl	Strix varia		Monitor	R	Riparian/Trees
Black tern	Chlidonias niger	Candidate 2	Monitor	R	Columbia River
Long-billed curlew	Numenius americanus	(b)	Monitor	C	Open shrubsteppe
Red-tailed hawk	Buteo jamaicensis		Protected	C	Trees/Shrubsteppe
FISH					
Mountain sucker	Catostomus platyrhynchus		Monitor	R	Columbia River
Piute sculpin	Cottus beldingi		Monitor	C	Columbia River
Reticulate sculpin	Cottus perplexus		Monitor	R	Columbia River
Sand roller	Percopsis transmontana		Monitor	R	Columbia River
INVERTEBRATES					
Columbia River tiger beetle	Cicindela columbica	(b)	Candidate	UC	Riparian
Columbia pebblesnail	Fulminicola columbianus	Candidate 2	Candidate	UC	Columbia River
Shortface lanx	Fisherola nuttalli	(b)	Candidate	UC	Columbia River
MAMMALS					
Pygmy rabbit	Brachylagus idahoensis	Candidate 2	Endangered	R	Oldgrowth Shrubsteppe
Merriam's shrew	Sorex merriami		Candidate	UC	Shrubsteppe
Townsend's big-eared bat	Plecotus townsendii townsendii	Candidate 2	Candidate	R	Buildings/Riparian
Pallid bat	Antrozous pallidus		Monitor	C	Buildings/Riparian
Sagebrush vole	Lagurus curtatus		Monitor	UC	Shrubsteppe
Small-footed myotis	Myotis ciliolabrum	Candidate 2	Monitor	UN	Buildings/Riparian
Long-eared myotis	Myotis evotis		Monitor	UN	Buildings/Riperian
Washington ground squirrel	Spermophilus washingtoni	Candidate 2	Monitor	UN	Shrubsteppe
Ord's Kangaroo rat	Dipodomys ordii		Monitor	UN	Shrubsteppe
Long-legged myotis	Myotis volans	Candidate 2	Monitor	UN	Buildings/Riparian
Northern grasshopper mouse	Onychomys leucogaster		Monitor	C	Shrubsteppe
White-tailed jackrabbit	Lepus townsendii		Game	UC	Shrubsteppe

Table C-1. Hanford Site Sensitive Species. (4 sheets)

Common Name	Latin Name	Federal Status	State Status	Abundance[a]	Habitat Assn.
REPTILES/AMPHIBIANS					
Northern sagebrush lizard	Sceloporus graciosus graciosus	Candidate 2		C	Shrubsteppe
Striped whipsnake	Masticophis taeniatus		Candidate	R	Shrubsteppe
Woodhouse's toad	Bufo woodhousei		Monitor	C	Riverine
Night snake	Hypsiglena torquata		Monitor	UC	Shrubsteppe
PLANTS					
Northern wormwood	Artemisia campestris var. wormskioldii	Candidate 1	Endangered	R	Rocky riparian
Columbia yellowcress	Rorippa columbiae	Candidate 2	Endangered	UC	Riparian
Dwarf desert primrose	Oenothera pygmaea		Threatened	R	Sand/Shrubsteppe
Columbia milkvetch	Astragalus columbianus	Candidate 1	Threatened	R	Sand/Shrubsteppe
Hoover's desert parsley	Lomatium tuberosum	Candidate 2	Threatened	R	Talus
Palouse milkvetch	Astragalus arrectus		Sensitive	R	Shrubsteppe
Dense sedge	Carex densa		Sensitive	R	Riparian
Gray cryptantha	Cryptantha leucophaea		Sensitive	UC	Sand/Shrubsteppe
Shining flatsedge	Cyperus bipartitus		Sensitive	UC	Riparian
Piper's daisy	Erigeron piperianus		Sensitive	UC	Disturbances/Sand/ Shrubsteppe
Southern mudwort	Limosella acaulis		Sensitive	UC	Riparian
False pimpernel	Lindernia dubia var. anagallidea		Sensitive	UC	Riparian
Coyote tobacco	Nicotiana attenuata		Sensitive	R	Open/Sandy
Desert evening primrose	Oenothera cespitosa		Sensitive	R	Riparian/Islands
Desert dodder	Cuscuta denticulata		Monitor 1	R	Shrubsteppe
Thompson's sandwort	Arenaria franklinii var. thompsonii		Monitor 2	C	Sand/Shrubsteppe
Bristly cryptantha	Cryptantha interrupta		Monitor 2	UC	Bluffs
Robinson's onion	Allium robinsonii		Monitor 3	C	Shrubsteppe
Squill onion	Allium scillioides		Monitor 3	C	Shrubsteppe
Columbia River mugwort	Artemisia lindleyana		Monitor 3	C	Riparian
Stalked-pod milkvetch	Astragalus sclerocarpus		Monitor 3	C	Sand/Shrubsteppe
Medick milkvetch	Astragalus speirocarpus		Monitor 3	R	Drainages/Shrubsteppe
Crouching milkvetch	Astragalus succumbens		Monitor 3	C	Shrubsteppe
Rosy balsamroot	Balsamorhiza rosea		Monitor 3	C	Rattlesnake Ridge
Palouse thistle	Cirsium brevifolium		Monitor 3	R	Shrubsteppe

Table C-1. Hanford Site Sensitive Species. (4 sheets)

Common Name	Latin Name	Federal Status	State Status	Abundance[a]	Habitat Assn.
Smooth cliffbrake	Pellaea glabella var. slimpex		Monitor 3	R	Umtanum Ridge moist canyons
Fuzzy beardtongue	Penstemon eriantherus var. whitedii		Monitor 3	R	White Bluffs

(a) C = Common
R = Rare
UC = Uncommon
UN = Unknown

(b) These taxa were once considered for listing, however they have proven to be more abundant or widespread than previously believed and/or were not subject to any identifiable threat.

Federal status for plant species taken from U.S. Department of Interior, U.S. Fish and Wildlife Service: 50 CFR Part 17, Endangered and Threatened Wildlife and Plants, Review of Plant Taxa for Listing Endangered or Threatened Species, Proposed Rules, September 30, 1993. Federal status for animal species taken from U.S. Department of Interior, U.S. Fish and Wildlife Service: 50 CFR 17 Endangered and Threatened Wildlife and Plants, Animal Candidate Review for Listing Endangered or Threatened Species, Proposed Rule, November 21, 1991.

State status for plant species taken from Washington Department of Natural Resources: Endangered, Threatened, and Sensitive Vascular Plants of Washington, 1994. State status for animal species taken from Washington Department of Wildlife; Species of Special Concern in Washington State - April 1994. Federal Status for animal species taken from 50 CFR Part 17 "Endangered and Threatened Wildlife and Plants; Animal Candidate Review for Listing as Endangered or Threatened Species; Proposed Rule", November 15, 1994.

Critical habitat for animal species taken from Washington Department of Wildlife; Priority Habitats and Species, November 1993.

Washington state monitor level 3 species are those that are more abundant and/or less threatened than previously assumed.

The abundance status of wildlife species was adapted from Fitzner and Gray, 1990 and Ennor, 1991.

Table C-2. Selected Hanford Site Foodweb Biota (2 sheets)

Area Description	Soil Penetration or Burrowing Depth			Drinking Water Required
	Average or Min., meters	Max., meters	Ref. No.	
GRASSLAND				
Cheatgrass	0.70	1.20	3, 5	N/A
Gray rabbitbrush	1.83	2.50	2	N/A
Green rabbitbrush	1.53	DG	2	N/A
Tumble mustard	1.00	2.00	Est.	N/A
Russian thistle	1.72	3.00	2	N/A
Northern grasshopper mice	0.40	DG	Est.	DG
Deer mice	0.40	DG	Est.	DG
Side-blotched lizards	0.30	1.0	6	DG
BOTH GRASSLAND & SHRUBLAND				
Sandberg's bluegrass	DG	0.35	11	N/A
Bluebunch wheatgrass	1.00	2.00	5	N/A
Needle-and-thread grass	1.39	1.83	2, 7	N/A
Indian ricegrass	1.19	1.22	2, 3	N/A
Thick-spike wheatgrass	0.50	DG	Est.	N/A
Sand dropseed	0.99	1.22	3	N/A
Bursage	1.62	1.80	2	N/A
Great Basin pocket mice	0.9	2.0	10, 8	No
Ground squirrels	2.0	DG	8	DG
North American badgers	2.5	DG	8	DG
GRASSLAND, SHRUBLAND & RIPARIAN				
Harvester ants (colony)	2.3	2.7	4	No
Darkling beetles	0.15	DG	9	No
Ground beetles	0.15	DG	Est.	No
Grasshoppers	0.05	DG	Est.	No
Coyotes	DG	DG		Yes
SHRUBLAND				
Big sagebrush	2.0	9.14	2, 3	N/A
Antelope bitterbrush	2.96	DG	2	N/A
Spiny hopsage	1.95	DG	2	N/A
Buckwheat	1.50	3.05	2, 3	N/A
Siberian elm trees (exotic)	GW	DG		N/A
Loggerhead shrikes	N/A	N/A	N/A	DG
Pygmy rabbits	1.0	1.50	Est.	DG
RIPARIAN				
White popular trees	GW	10.0	1	N/A
Marsh hawks (harriers)	N/A	N/A	N/A	DG

Table C-2. Selected Hanford Site Foodweb Biota (2 sheets)

Area Description	Soil Penetration or Burrowing Depth			Drinking Water Required
	Average or Min., meters	Max., meters	Ref. No.	
BOTH RIPARIAN AND REACTOR				
Violet-green swallows	N/A	N/A	N/A	DG
Bank swallows	.7 -.9 horizontal	DG	12	DG
REACTOR AREA				
House mice	DG	DG		DG
Bats	N/A	N/A	N/A	DG
Pigeons	N/A	N/A	N/A	DG

DG = Data gap
Est. = Parameters Estimated
GW = Groundwater
N/A = Not applicable

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3. Foxx, T.S., G.D. Tierney, and J.M. Williams, 1984, *Rooting Depths of Plants on Low-Level Waste Disposal Sites*, LA-10253-MS, Los Alamos National Laboratory, Los Alamos, New Mexico.

4. Rogers, L.E., R.E. Fitzner, L.L. Cadwell, and B.E. Vaughan, 1988, *Terrestrial Animal Habitats and Population Responses*, pp.182-256, Ln: Shrub-Steppe: Balance and Change in a Semi-Arid Terrestrial Ecosystem, W.H. Rickard et al. (eds). Elsevier, NY.

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6. Rickard, W.H., 1967, "Onset of Winter Dormancy in Lizards and Beetles," *Northwest Science*, 41(2):92-95.

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8. McKenzie, D. H., L.L. Cadwell, L.E. Eberhardt, W.E. Kennedy, Jr., R.A. Peloquin, and M.A. Siminona, 1982, *Relevance of Biotic Pathways to the Long-term Regulation of Nuclear Waste Disposal*, NUREG/CR-2675, PNL-4241, Vol 2., Pacific Northwest Laboratoroes, Richland, Washington.

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10. O'Farrell, T.P., R.J. Olson, R.O. Gilbert, and J.D. Hedlund, 1975, "A Population of Great Basin Pocket Mice, Perognathus Parvus, in the Shrub-Steppe of South-Central Washington, " *Ecol. Monogr.*, 45:1-28.

11. Lind, S.O., O.W. Gee, and J.L. Downs, 1990, "The Effect of Water Stress on Phenological and Ecophysiological Characteristics of Cheatgrass and Sandberg's Bluegrass," *J. Range Management*, 43(6):506-512.

APPENDIX D

EQUATIONS AND EXAMPLE CALCULATIONS FOR HUMAN HEALTH RISK ASSESSMENT

CONTENTS

FIGURES:

INTRODUCTION

This appendix provides general equations and example calculations for performing preliminary risk-based screening and risk-based calculations, calculating contaminant intakes, and calculating human health impacts associated with contaminant intakes. The appendix consists of six sections; Sections D-1.0 through D-3.0 provide the general equations, while Sections D-4.0 through D-6.0 provide example calculations.

Within each section, equations are organized first by exposure route (ingestion, inhalation of fugitive dust, inhalation of volatiles, dermal exposure, and external exposure), then by environmental source media (soil/sediment, surface and groundwater, and biota), and lastly by contaminant type (noncarcinogen, nonradioactive carcinogen, and radioactive carcinogen). Flowcharts are provided in Figures D-1 and D-2 for assistance in locating the appropriate equation or example calculation for a selected exposure route, environmental media, and type of contaminant. Table D-1 provides summary screening factors for calculating preliminary risk-based screening levels using default exposure parameters. Table D-2 provides default parameters for calculating a particulate emission factor and a volatilization factor, which are applied in equations for the inhalation exposure route for contaminants in soil media.

Although equations and example calculations are provided here for human health risk evaluation, a knowledgeable toxicologist or risk assessor should be consulted when applying these equations for a particular case. For example, oral slope factors and reference doses should be adjusted appropriately when used for dermal exposures to account for differences in absorbed dose via the gastrointestinal tract and the skin. In addition, averaging time for adverse effects should be set equal to 70 years, 365 days per year, for nonradioactive carcinogens, and equal to exposure duration, 365 days per year, for noncarcinogens.

Equations presented in this appendix were prepared following methodology described in Risk Assessment Guidance for Superfund, Part A (EPA 1989) and Supplemental Risk Assessment Guidance for Superfund (EPA-10 1991).

D-1.0 EQUATIONS FOR PRELIMINARY RISK-BASED SCREENING AND RISK-BASED CONCENTRATIONS

In Section D-1.0, general equations for performing preliminary risk-based screening and calculating risk-based concentrations are presented. The equations are used to calculate a contaminant concentration in a chosen media, which corresponds with a target level of human health risk. Example calculations corresponding to each equation in this section, using appropriate default exposure factors and target risk levels for preliminary risk-based screening, may be found in the identical subsection in Section D-4.0.

D-1.1 INGESTION

D-1.1.1 Soil and Sediment

D-1.1.1.1 Noncarcinogenic

$$C = \frac{THQ \times RfD_o \times BW \times AT \times CF}{IR \times EF \times ED} \qquad \text{D-1}$$

where:

C	=	risk-based soil concentration (mg/kg)
THQ	=	target hazard quotient
RfD_o	=	oral reference dose (mg/kg-d)
BW	=	body weight (kg)
AT	=	averaging time (yr x 365 d/yr)
CF	=	conversion factor (1E+06 mg/kg)
IR	=	ingestion rate (mg/d)
EF	=	exposure frequency (d/yr)
ED	=	exposure duration (yr)

D-1.1.1.2 Carcinogenic - Nonradioactive

$$C = \frac{TR \times AT \times CF}{SF_o \times \left[\left(\frac{IR \times EF \times ED}{BW} \right)_{child} + \left(\frac{IR \times EF \times ED}{BW} \right)_{adult} \right]} \qquad \text{D-2}$$

where:

C	=	risk-based soil concentration (mg/kg)
TR	=	target risk level
AT	=	averaging time (yr x 365 d/yr)
CF	=	conversion factor (1E+06 mg/kg)
SF_o	=	oral slope factor $(mg/kg\text{-}d)^{-1}$
IR	=	ingestion rate (mg/d)
EF	=	exposure frequency (d/yr)
ED	=	exposure duration (yr)
BW	=	body weight (kg)

D-1.1.1.3 Carcinogenic - Radioactive

$$C = \frac{TR \times CF}{SF_o \times \left((IR \times EF \times ED)_{child} + (IR \times EF \times ED)_{adult}\right)} \quad \text{D-3}$$

where:

C	=	risk-based soil concentration (pCi/g)
TR	=	target risk level
CF	=	conversion factor (1E+03 mg/g)
SF_o	=	oral slope factor $(pCi)^{-1}$
IR	=	ingestion rate (mg/d)
EF	=	exposure frequency (d/yr)
ED	=	exposure duration (yr)

D-1.1.2 Surface and Groundwater

D-1.1.2.1 Noncarcinogenic

$$C = \frac{THQ \times RfD_o \times BW \times AT}{IR \times EF \times ED} \quad \text{D-4}$$

where:

C	=	risk-based water concentration (mg/L)
THQ	=	target hazard quotient
RfD_o	=	oral reference dose (mg/kg-d)
BW	=	body weight (kg)
AT	=	averaging time (yr x 365 d/yr)
IR	=	ingestion rate (L/d)
EF	=	exposure frequency (d/yr)
ED	=	exposure duration (yr)

D-1.1.2.2 Carcinogenic - Nonradioactive

$$C = \frac{TR \times BW \times AT}{SF_o \times IR \times EF \times ED} \quad \text{D-5}$$

where:

C	=	risk-based water concentration (mg/L)
TR	=	target risk level
BW	=	body weight (kg)
AT	=	averaging time (yr x 365 d/yr)
SF_o	=	oral slope factor $(mg/kg\text{-}d)^{-1}$
IR	=	ingestion rate (L/d)
EF	=	exposure frequency (d/yr)
ED	=	exposure duration (yr)

D-1.1.2.3 Carcinogenic - Radioactive

$$C = \frac{TR}{SF_o \times IR \times EF \times ED} \qquad \text{D-6}$$

where:

C	=	risk-based water concentration (pCi/L)
TR	=	target risk level
SF_o	=	oral slope factor $(pCi)^{-1}$
IR	=	ingestion rate (L/d)
EF	=	exposure frequency (d/yr)
ED	=	exposure duration (yr)

D-1.2 INHALATION (Fugitive Dust)

D-1.2.1 Soil

D-1.2.1.1 Noncarcinogenic

$$C = \frac{THQ \times RfD_i \times BW \times AT \times PEF}{IR \times EF \times ED} \qquad \text{D-7}$$

where:

C	=	risk based soil concentration (mg/kg)
THQ	=	target hazard quotient
RfD_i	=	inhalation reference dose (mg/kg-d)
BW	=	body weight (kg)
AT	=	averaging time (yr x 365 d/yr)
PEF	=	particulate emission factor (m^3/kg)
IR	=	inhalation rate (m^3/d)
EF	=	exposure frequency (d/yr)
ED	=	exposure duration (yr)

and, for preliminary risk-based screening purposes:

$$PEF = \frac{1}{NAAQS} \times CF \qquad \text{D-8}$$

where:

PEF	=	particulate emission factor (m^3/kg)
NAAQS	=	National Ambient Air Quality Standard for PM_{10} ($\mu g/m^3$)
CF	=	conversion factor (1E+09 μg/kg)

while, for site-specific RBCs:

$$PEF = \frac{LS \times V \times MH \times CF_1 \times CF_2}{A \times E_{10}} \qquad \text{D-9}$$

where:

PEF	=	particulate emission factor (m^3/kg)
LS	=	length of side of contaminated area (m)
V	=	wind speed in breathing zone (m/s)
MH	=	mixing height (m)
CF_1	=	conversion factor (3.6E+03 s/hr)
CF_2	=	conversion factor (1E+03 g/kg)
A	=	area of contamination (m^2)
E_{10}	=	annual average PM_{10} emission rate per unit area of contaminated surface (g/m^2-hr)

and,

$$E_{10} = RESP \times (1 - G) \times (U_m/U_t)^3 \times F_x \qquad \text{D-10}$$

where:

E_{10}	=	annual average PM_{10} emission rate per unit area of contaminated surface (g/m^2-hr)
RESP	=	PM_{10} fraction emission per unit area (g/m^2-hr)
G	=	fraction of vegetative cover (unitless)
U_m	=	mean annual wind speed (m/s)
U_t	=	equivalent threshold value of wind speed at 10 m (m/s)
F_x	=	function dependent on U_m/U_t (unitless)

(Table D-2 provides EPA default parameters for calculating a PEF).

D-1.2.1.2 Carcinogenic - Nonradioactive

$$C = \frac{TR \times BW \times AT \times PEF}{SF_i \times IR \times EF \times ED} \qquad \text{D-11}$$

where:

C	=	risk-based soil concentration (mg/kg)
TR	=	target risk level
BW	=	body weight (kg)
AT	=	averaging time (yr x 365 d/yr)
PEF	=	particulate emission factor (m^3/kg) - see equations D8 and D9
SF_i	=	inhalation slope factor $(mg/kg\text{-}d)^{-1}$
IR	=	inhalation rate (m^3/d)
EF	=	exposure frequency (d/yr)
ED	=	exposure duration (yr)

D-1.2.1.3 Carcinogenic - Radioactive

$$C = \frac{TR \times PEF \times CF}{SF_i \times IR \times EF \times ED} \qquad \text{D-12}$$

where:

C	=	risk-based soil concentration (pCi/g)
TR	=	target risk level
PEF	=	particulate emission factor (m^3/kg) - see equations D8 and D9
CF	=	conversion factor (1E-03 kg/g)
SF_i	=	inhalation slope factor $(pCi)^{-1}$
IR	=	inhalation rate (m^3/d)
EF	=	exposure frequency (d/yr)
ED	=	exposure duration (yr)

D-1.3 INHALATION (Volatile Compounds)

D-1.3.1 Soil

D-1.3.1.1 Noncarcinogenic

$$C = \frac{THQ \times RfD_i \times BW \times AT \times VF_s}{IR \times EF \times ED} \qquad \text{D-13}$$

where:

C	=	risk based soil concentration (mg/kg)
THQ	=	target hazard quotient
RfD_i	=	inhalation reference dose (mg/kg-d)
BW	=	body weight (kg)
AT	=	averaging time (yr x 365 d/yr)
VF_s	=	soil volatilization factor (m^3/kg)
IR	=	inhalation rate (m^3/d)
EF	=	exposure frequency (d/yr)
ED	=	exposure duration (yr)

and,

$$VF_s = \frac{LS \times V \times DH}{A} \times \frac{(3.14 \times \alpha \times T)^{1/2}}{2 \times D_{ei} \times E \times K_{as} \times CF} \qquad \text{D-14}$$

where:

LS	=	length of side of contaminated area (m)
V	=	wind speed in breathing zone (m/s)
DH	=	diffusion height (m)
A	=	area of contamination (cm^2)
T	=	exposure interval (s)
D_{ei}	=	effective diffusivity (cm^2/s) = $D_i \times E^{0.33}$
D_i	=	molecular diffusivity (cm^2/s)
E	=	true soil porosity (unitless)

CF = conversion factor (1E-03 kg/g)

and,

$$K_{as} = \left(\frac{H}{K_d}\right) \times 41 \qquad \text{D-15}$$

where:

K_{as} = soil-air partition coefficient (g soil/cm^3 air)
H = Henry's constant (atm-m^3/mol)
K_d = soil-water partition coefficient (cm^3/g)

If a K_d value is not available, it may be calculated as follows:

$$K_d = K_{oc} \times oc \qquad \text{D-16}$$

where:

K_d = soil-water partition coefficient (cm^3/g)
K_{oc} = organic carbon partition coefficient (cm^3/g)
oc = organic carbon fraction of soil (unitless)

and:

$$\alpha = \frac{D_{ei} \times E}{E + \left((P_s)(1-E)/K_{as}\right)} \qquad \text{D-17}$$

where:

P_S = particulate density (g/cm^3)

(Table D-2 provides default parameters for calculating a VF_s).

D-1.3.1.2 Carcinogenic - Nonradioactive

$$C = \frac{TR \times BW \times AT \times VF_s}{SF_i \times IR \times EF \times ED} \qquad \text{D-18}$$

where:

C = risk-based soil concentration (mg/kg)
TR = target risk level
BW = body weight (kg)
AT = averaging time (yr x 365 d/yr)
VF_s = soil volatilization factor (m^3/kg) - see equation D-14
SF_i = inhalation slope factor (mg/kg-d)$^{-1}$
IR = inhalation rate (m^3/d)
EF = exposure frequency (d/yr)
ED = exposure duration (yr)

D-1.3.1.3 Carcinogenic - Radioactive

Not applicable.

D-1.3.2 Surface and Groundwater

D-1.3.2.1 Noncarcinogenic

$$C = \frac{THQ \times RfD_i \times BW \times AT}{IR \times EF \times ED \times VF_{wvoc}} \qquad \text{D-19}$$

where:

C	=	risk-based water concentration (mg/L)
THQ	=	target hazard quotient
RfD_i	=	inhalation reference dose (mg/kg-d)
BW	=	body weight (kg)
AT	=	averaging time (yr x 365 d/yr)
IR	=	inhalation rate (m^3/d)
EF	=	exposure frequency (d/yr)
ED	=	exposure duration (yr)
VF_{wvoc}	=	water volatilization factor for VOCs (L/m^3)

D-1.3.2.2 Carcinogenic - Nonradioactive

$$C = \frac{TR \times BW \times AT}{SF_i \times IR \times EF \times ED \times VF_{wvoc}} \qquad \text{D-20}$$

where:

C	=	risk-based water concentration (mg/L)
TR	=	target risk level
BW	=	body weight (kg)
AT	=	averaging time (yr x 365 d/yr)
SF_i	=	inhalation slope factor $(mg/kg\text{-}d)^{-1}$
IR	=	inhalation rate (m^3/d)
EF	=	exposure frequency (d/yr)
ED	=	exposure duration (yr)
VF_{wvoc}	=	water volatilization factor for VOCs (L/m^3)

D-1.3.2.3 Carcinogenic - Radioactive (radon-222)

$$C = \frac{TR}{SF_i \times IR \times EF \times ED \times VF_{wr}} \qquad \text{D-21}$$

where:

C	=	risk-based water concentration (pCi/L)
TR	=	target risk level
SF_i	=	inhalation slope factor $(pCi)^{-1}$
IR	=	inhalation rate (m^3/d)
EF	=	exposure frequency (d/yr)

ED	=	exposure duration (yr)
VF_{wr}	=	water volatilization factor for radon (L/m^3)

D-1.4 EXTERNAL EXPOSURE TO RADIONUCLIDES

D-1.4.1 Soil

$$C = \frac{TR}{SF_e \times ET \times RF \times EF \times ED \times CF} \qquad \text{D-22}$$

where:

C	=	risk-based soil concentration (pCi/g)
TR	=	target risk level
SF_e	=	external slope factor $(pCi\text{-}yr/g)^{-1}$
ET	=	exposure time (hr/d)
RF	=	dose reduction factor (unitless)
EF	=	exposure frequency (d/yr)
ED	=	exposure duration (yr)
CF	=	conversion factor (1.14E-04 yr/hr)

D-2.0 INTAKE EQUATIONS

In Section D-2.0, general equations for calculating contaminant intake are presented. The equations are used to calculate daily (or lifetime, for radionuclides) contaminant intake for a chosen exposure scenario and known contaminant concentration. Default exposure parameters for a chosen exposure scenario are presented in Appendix A. Example calculations corresponding to each equation in this section may be found in the identical subsection in Section D-5.0.

D-2.1 INGESTION

D-2.1.1 Soil and Sediment

D-2.1.1.1 Noncarcinogenic

$$\text{Intake} = \frac{C \times IR \times EF \times ED \times CF}{BW \times AT} \qquad \text{D-23}$$

where:

Intake	=	chronic daily intake (mg/kg-d)
C	=	contaminant soil concentration (mg/kg)
IR	=	ingestion rate (mg/d)
EF	=	exposure frequency (d/yr)
ED	=	exposure duration (yr)
CF	=	conversion factor (1E-06 kg/mg)

BW = body weight (kg)
AT = averaging time (yr x 365 d/yr)

D-2.1.1.2 Carcinogenic - Nonradioactive

$$\text{Intake} = \frac{C \times \left[\left(\frac{IR \times ED}{BW} \right)_{child} + \left(\frac{IR \times ED}{BW} \right)_{adult} \right] \times EF \times CF}{AT} \qquad \text{D-24}$$

where:

Intake = chronic daily intake (mg/kg-d)
C = contaminant soil concentration (mg/kg)
IR = ingestion rate (mg/d)
ED = exposure duration (yr)
BW = body weight (kg)
EF = exposure frequency (d/yr)
CF = conversion factor (1E-06 kg/mg)
AT = averaging time (yr x 365 d/yr)

NOTE: Only adult exposure is evaluated for the industrial scenario.

D-2.1.1.3 Carcinogenic - Radioactive

$$\text{Intake} = C \times \left((IR \times ED)_{child} + (IR \times ED)_{adult} \right) \times EF \times CF \qquad \text{D-25}$$

where:

Intake = lifetime intake (pCi)
C = radionuclide concentration in soil (pCi/g)
IR = ingestion rate (mg/d)
ED = exposure duration (yr)
EF = exposure frequency (d/yr)
CF = conversion factor (1E-03 g/mg)

NOTE: Only adult exposure is evaluated for the industrial scenario.

D-2.1.2 Surface and Groundwater

D-2.1.2.1 Noncarcinogenic

$$\text{Intake} = \frac{C \times IR \times EF \times ED}{BW \times AT} \qquad \text{D-26}$$

where:

Intake = chronic daily intake (mg/kg-d)
C = contaminant concentration in water (mg/L)
IR = ingestion rate (L/d)
EF = exposure frequency (d/yr)
ED = exposure duration (yr)

BW	=	body weight (kg)
AT	=	averaging time (yr x 365 d/yr)

D-2.1.2.2 Carcinogenic - Nonradioactive

Same as for Noncarcinogenic intake in equation D-26.

D-2.1.2.3 Carcinogenic - Radioactive

$$\text{Intake} = C \times IR \times EF \times ED \qquad \text{D-27}$$

where:

Intake	=	lifetime intake (pCi)
C	=	radionuclide concentration in water (pCi/L)
IR	=	ingestion rate (L/d)
EF	=	exposure frequency (d/yr)
ED	=	exposure duration (yr)

D-2.1.3 Biota

D-2.1.3.1 Noncarcinogenic

$$\text{Intake} = \frac{C \times IR \times EF \times ED \times AF \times CF}{BW \times AT} \qquad \text{D-28}$$

where:

Intake	=	chronic daily intake (mg/kg-d)
C	=	contaminant concentration in biota (mg/kg)
IR	=	ingestion rate (g/d)
EF	=	exposure frequency (d/yr)
ED	=	exposure duration (yr)
AF	=	intake adjustment factor (unitless), for game and fish only
CF	=	conversion factor (1E-03 kg/g)
BW	=	body weight (kg)
AT	=	averaging time (yr x 365 d/yr)

D-2.1.3.2 Carcinogenic - Nonradioactive

Same as for Noncarcinogenic intake in equation D-28.

D-2.1.3.3 Carcinogenic - Radioactive

$$\text{Intake} = C \times IR \times EF \times ED \times AF \qquad \text{D-29}$$

where:

Intake	=	lifetime intake (pCi)
C	=	radionuclide concentration in biota (pCi/g)
IR	=	ingestion rate (g/d)
EF	=	exposure frequency (d/yr)

ED	=	exposure duration (yr)
AF	=	intake adjustment factor (unitless), for game and fish only.

D-2.2 INHALATION (Fugitive Dust)

D-2.2.1 Soil

D-2.2.1.1 Noncarcinogenic

$$\text{Intake} = \frac{C \times IR \times EF \times ED}{BW \times AT \times PEF} \qquad \text{D-30}$$

where:

Intake	=	chronic daily intake (mg/kg-d)
C	=	contaminant concentration in soil (mg/kg)
IR	=	inhalation rate (m^3/d)
EF	=	exposure frequency (d/yr)
ED	=	exposure duration (yr)
BW	=	body weight (kg)
AT	=	averaging time (yr x 365 d/yr)
PEF	=	site-specific particulate emission factor (m^3/kg) (see definition in Section 1.2.1.1)

D-2.2.1.2 Carcinogenic - Nonradioactive

Same as for Noncarcinogenic intake in equation D-30.

D-2.2.1.3 Carcinogenic - Radioactive

$$\text{Intake} = \frac{C \times IR \times EF \times ED \times CF}{PEF} \qquad \text{D-31}$$

where:

Intake	=	lifetime intake (pCi)
C	=	radionuclide concentration in soil (pCi/g)
IR	=	inhalation rate (m^3/d)
EF	=	exposure frequency (d/yr)
ED	=	exposure duration (yr)
CF	=	conversion factor (1E-03 kg/g)
PEF	=	site-specific particulate emission factor (m^3/kg) (see definition in Section 1.2.1.1)

D-2.3 INHALATION (Volatile Compounds)

D-2.3.1 Soil

D-2.3.1.1 Noncarcinogenic

$$\text{Intake} = \frac{C \times IR \times EF \times ED}{BW \times AT \times VF_s} \qquad \text{D-32}$$

where:

Intake	=	chronic daily intake (mg/kg-d)
C	=	contaminant concentration in soil (mg/kg)
IR	=	inhalation rate (m^3/d)
EF	=	exposure frequency (d/yr)
ED	=	exposure duration (yr)
BW	=	body weight (kg)
AT	=	averaging time (yr x 365 d/yr)
VF_s	=	soil volatilization factor (m^3/kg) (see definition in Section 1.3.1.1)

D-2.3.1.2 Carcinogenic - Nonradioactive

Same as for Noncarcinogenic intake in equation D-32.

D-2.3.1.3 Carcinogenic - Radioactive

Not applicable.

D-2.3.2 Surface and Groundwater

D-2.3.2.1 Noncarcinogenic

$$\text{Intake} = \frac{C \times IR \times EF \times ED \times VF_{wvoc}}{BW \times AT} \qquad \text{D-33}$$

where:

Intake	=	chronic daily intake (mg/kg-d)
C	=	contaminant concentration in water (mg/L)
IR	=	inhalation rate (m^3/d)
EF	=	exposure frequency (d/yr)
ED	=	exposure duration (yr)
VF_{wvoc}	=	water volatilization factor for VOCs (L/m^3)
BW	=	body weight (kg)
AT	=	averaging time (yr x 365 d/yr)

D-2.3.2.2 Carcinogenic - Nonradioactive

Same as for Noncarcinogenic intake in equation D-33.

D-2.3.2.3 Carcinogenic - Radioactive (radon-222)

$$\text{Intake} = C \times IR \times EF \times ED \times VF_{wr} \qquad \text{D-34}$$

where:

Intake	=	radionuclide-specific lifetime intake (pCi)
C	=	radionuclide concentration in water (pCi/L)
IR	=	inhalation rate (m^3/d)
EF	=	exposure frequency (d/yr)
ED	=	exposure duration (yr)
VF_{wr}	=	water volatilization factor for radon (L/m^3)

D-2.4 DERMAL EXPOSURE

D-2.4.1 Soil and Sediment

D-2.4.1.1 Noncarcinogenic

$$DAD = \frac{C \times ABS \times AF \times CF \times \left[\left(\frac{SA \times EF \times ED}{BW} \right)_{child} + \left(\frac{SA \times EF \times ED}{BW} \right)_{adult} \right]}{AT} \qquad \text{D-35}$$

where:

DAD	=	dermally absorbed dose (mg/kg-d)
C	=	contaminant soil concentration (mg/kg)
ABS	=	absorption factor (unitless)
AF	=	adherence factor (mg/cm^2-day)
CF	=	conversion factor (1E-06 kg/mg)
SA	=	surface area exposed (cm^2)
EF	=	exposure frequency (d/yr)
ED	=	exposure duration (yr)
BW	=	body weight (kg)
AT	=	averaging time (yr x 365 d/yr)

note: Only adult exposure is evaluated for the industrial scenario.

D-2.4.1.2 Carcinogenic - Nonradioactive

Same as for Noncarcinogenic intake in equation D-35.

NOTE: Only adult exposure is evaluated for the industrial scenario.

D-2.4.1.3 Carcinogenic - Radioactive

Not applicable.

D-2.4.2 Surface and Groundwater

D-2.4.2.1 Noncarcinogenic

$$DAD = \frac{C \times SA \times K_p \times ET \times EF \times ED \times CF}{BW \times AT} \qquad \text{D-36}$$

where:

DAD	=	dermally absorbed dose (mg/kg-d)
C	=	contaminant water concentration (mg/L)
SA	=	surface area exposed (cm^2)
K_p	=	permeability coefficient for a chemical in water through skin (cm/hr)
ET	=	exposure time (hr/d)
EF	=	exposure frequency (d/yr)
ED	=	exposure duration (yr)
CF	=	conversion factor (1E-03 L/cm^3)
BW	=	body weight (kg)
AT	=	averaging time (yr x 365 d/yr)

D-2.4.2.2 Carcinogenic - Nonradioactive

Same as for Noncarcinogenic intake in equation D-36.

D-2.4.2.3 Carcinogenic - Radioactive

Not applicable.

D-2.5 EXTERNAL EXPOSURE TO RADIONUCLIDES

D-2.5.1 Soil

$$Exposure = C \times ET \times RF \times EF \times ED \times CF \qquad \text{D-37}$$

where:

Exposure	=	lifetime external exposure to radionuclide (pCi-yr/g)
C	=	radionuclide concentration in soil (pCi/g)
ET	=	exposure time (hr/d)
RF	=	dose reduction factor (unitless)
EF	=	exposure frequency (d/yr)
ED	=	exposure duration (yr)
CF	=	conversion factor (1.14E-04 yr/hr)

D-3.0 RISK CALCULATION EQUATIONS

In Section D-3.0, general equations are presented for calculating human health risk using an intake factor (or, for external exposure to radionuclides, an exposure factor), and a toxicity value. General equations for calculating intake/exposure factors are presented in Section D-2.0. Toxicity values are discussed in Section 2.3. Example calculations corresponding to each equation in this section may be found in Section D-6.0.

D-3.1 NONCARCINOGENIC

$$HQ = \frac{Intake}{RfD} \qquad \text{D-38}$$

where:

HQ	=	hazard quotient (unitless)
Intake	=	chronic daily intake (mg/kg-d)
RfD	=	contaminant-specific reference dose (mg/kg-d)

D-3.2 CARCINOGENIC - Nonradioactive

$$ICR = Intake \times SF \qquad \text{D-39}$$

where:

ICR	=	lifetime incremental cancer risk (unitless)
Intake	=	chronic daily intake (mg/kg-d)
SF	=	contaminant-specific slope factor $(mg/kg\text{-}d)^{-1}$

D-3.3 CARCINOGENIC - Radioactive (internal)

$$ICR = Intake \times SF \qquad \text{D-40}$$

where:

ICR	=	lifetime incremental cancer risk (unitless)
Intake	=	lifetime intake (pCi)
SF	=	radionuclide-specific slope factor $(pCi)^{-1}$

D-3.4 CARCINOGENIC - Radioactive (external)

$$ICR = Exposure \times SF_e \qquad \text{D-41}$$

where:

ICR	=	lifetime incremental cancer risk (unitless)
Exposure	=	lifetime external exposure to radionuclide (pCi-yr/g)
SF_e	=	radionuclide-specific external slope factor $(pCi\text{-}yr/g)^{-1}$

D-4.0 EXAMPLE CALCULATIONS FOR PRELIMINARY RISK-BASED SCREENING CONCENTRATIONS

This appendix provides example calculations for performing preliminary risk-based screening using the general equations provided in Section D-1.0. For preliminary risk-based screening, a residential exposure scenario is evaluated with target risk levels equal to a hazard quotient of 0.1 and an incremental lifetime cancer risk of 1E-07. Default exposure parameters for a residential exposure scenario are presented in Appendix A. The example calculations use methylene chloride and ^{238}U as example contaminants for all equations excepting fugitive dust inhalation for nonradioactive contaminants. Because this pathway is not evaluated for volatile compounds, barium and arsenic are used for the noncarcinogenic and carcinogenic, nonradioactive examples, respectively. Toxicity values are obtained from IRIS (EPA 1993a) for all contaminants except ^{238}U, for which toxicity values are obtained from HEAST (EPA 1993b).

D-4.1 INGESTION

D-4.1.1 Soil

D-4.1.1.1 Noncarcinogenic

$$C = \frac{THQ \times RfD_o \times BW \times AT \times CF}{IR \times EF \times ED} \qquad \text{D-42}$$

Substituting the appropriate exposure factors from Table A-7, we have:

$$C\ (mg/kg) = \frac{(0.1)(RfD_o)(16\ kg)(6\ yr \times 365\ d/yr)(1E{+}06\ mg/kg)}{(200\ mg/d)(365\ d/yr)(6\ yr)} \qquad \text{D-43}$$

which reduces to a summary screening factor (see Table D-1) and a toxicity value:

$$C\ (mg/kg) = (8.0E{+}03\ d)(RfD_o) \qquad \text{D-44}$$

Using methylene chloride as an example, we can calculate:

$$C\ (mg/kg) = (8.0E{+}03\ d)(6E{-}02\ mg/kg{-}d) = 480\ mg/kg \qquad \text{D-45}$$

D-4.1.1.2 Carcinogenic, Nonradioactive

$$C = \frac{TR \times AT \times CF}{SF_o \times \left[\left(\frac{IR \times EF \times ED}{BW} \right)_{child} + \left(\frac{IR \times EF \times ED}{BW} \right)_{adult} \right]} \qquad \text{D-46}$$

Substituting the appropriate exposure factors from Table A-8, we have:

$$C\ (mg/kg) = \frac{(1E{-}07)(70\ yr \times 365\ d/yr)(1E{+}06\ mg/kg)}{SF_o \times \left[\left(\frac{(200\ mg/d)(365\ d/yr)(6\ yr)}{(16\ kg)} \right)_{child} + \left(\frac{(100\ mg/d)(365\ d/yr)(24\ yr)}{(70\ kg)} \right)_{adult} \right]} \qquad \text{D-47}$$

which reduces to a summary screening factor (see Table D-1) and a toxicity value:

$$C\ (mg/kg) = \frac{6.4E{-}02\ d}{SF_o} \qquad \text{D-48}$$

Using methylene chloride as an example, we can calculate:

$$C\ (mg/kg) = \frac{6.4E{-}02\ d}{7.5E{-}03\ (mg/kg{-}d)^{-1}} = 8.5\ mg/kg \qquad \text{D-49}$$

D-4.1.1.3 Carcinogenic, Radioactive

$$C = \frac{TR \times CF}{SF_o \times \left((IR \times EF \times ED)_{child} + (IR \times EF \times ED)_{adult} \right)} \qquad \text{D-50}$$

Substituting the appropriate exposure factors from Table A-9, we have:

$$C\ (pCi/g) = \frac{(1E{-}07)(1E{+}03\ mg/g)}{(SF_o) \left(((200\ mg/d)(365\ d/yr)(6\ yr))_{child} + ((100\ mg/d)(365\ d/yr)(24\ yr))_{adult} \right)} \qquad \text{D-51}$$

which reduces to a summary screening factor (see Table D-1) and a toxicity value:

$$C\ (pCi/g) = \frac{7.6E{-}11\ (g)^{-1}}{SF_o} \qquad \text{D-52}$$

Using ^{238}U as an example, we can calculate:

$$C\ (pCi/g) = \frac{7.6E-11\ (g)^{-1}}{2.8E-11\ (pCi)^{-1}} = 2.7\ pCi/g \qquad \text{D-53}$$

D-4.1.2 Surface and Groundwater

D-4.1.2.1 Noncarcinogenic

$$C = \frac{THQ \times RfD_o \times BW \times AT}{IR \times EF \times ED} \qquad \text{D-54}$$

Substituting the appropriate exposure factors from Table A-7, we have:

$$C\ (mg/L) = \frac{(0.1)(RfD_o)(16\ kg)(6\ yr \times 365\ d/yr)}{(1\ L/d)(365\ d/yr)(6\ yr)} \qquad \text{D-51}$$

which reduces to a summary screening factor (see Table D-1) and a toxicity value:

$$C\ (mg/L) = (1.6E+00\ kg-d/L)(RFD_o) \qquad \text{D-56}$$

Using methylene chloride as an example, we can calculate:

$$C\ (mg/L) = (1.6E+00\ kg-d/L)(6E-02\ mg/kg-d) = 9.6E-02\ mg/L \qquad \text{D-57}$$

D-4.1.2.2 Carcinogenic, Nonradioactive

$$C = \frac{TR \times BW \times AT}{SF_o \times IR \times EF \times ED} \qquad \text{D-58}$$

Substituting the appropriate exposure factors from Table A-8, we have:

$$C\ (mg/L) = \frac{(1E-07)(70\ kg)(70\ yr \times 365\ d/yr)}{(SF_o)(2\ L/d)(365\ d/yr)(30\ yr)} \qquad \text{D-59}$$

which reduces to a summary screening factor (see Table D-1) and a toxicity value:

$$C\ (mg/L) = \frac{8.2E-06\ kg-d/L}{SF_o} \qquad \text{D-60}$$

Using methylene chloride as an example, we can calculate:

$$C\ (mg/L) = \frac{8.2E-06\ kg-d/L}{7.5E-03\ (mg/kg-d)^{-1}} = 1.1E-03\ mg/L \qquad D-61$$

D-4.1.2.3 Carcinogenic, Radioactive

$$C = \frac{TR}{SF_o \times IR \times EF \times ED} \qquad D-62$$

Substituting the appropriate exposure factors from Table A-9, we have:

$$C\ (pCi/L) = \frac{1E-07}{(SF_o)(2\ L/d)(365\ d/yr)(30\ yr)} \qquad D-63$$

which reduces to a summary screening factor (see Table D-1) and a toxicity value:

$$C\ (pCi/L) = \frac{4.6E-12\ (L)^{-1}}{SF_o} \qquad D-64$$

Using ^{238}U as an example, we can calculate:

$$C\ (pCi/L) = \frac{4.6E-12\ (L)^{-1}}{2.8E-11\ (pCi)^{-1}} = 0.16\ pCi/L \qquad D-65$$

D-4.2 INHALATION (Fugitive Dust)

D-4.2.1 Soil

D-4.2.1.1 Noncarcinogenic

$$C = \frac{THQ \times RfD_i \times BW \times AT \times PEF}{IR \times EF \times ED} \qquad D-66$$

where:

$$PEF = \frac{1}{NAAQS} \times CF \qquad D-67$$

Substituting the appropriate exposure factors from Table A-7, we have:

$$C\ (mg/kg) = \frac{(0.1)(RfD_i)(16\ kg)(6\ yr \times 365\ d/yr)(PEF)}{(10\ m^3/d)(365\ d/yr)(6\ yr)} \quad \text{D-68}$$

where, substituting default parameters from Table D-2:

$$PEF\ (m^3/kg) = \frac{1}{(50\ \mu g/m^3)} \times (1E{+}09\ \mu g/kg) = 2E{+}07\ m^3/kg \quad \text{D-69}$$

This reduces to a summary screening factor (see Table D-1) and a toxicity value:

$$C\ (mg/kg) = (3.2E{+}06\ d)(RFD_o) \quad \text{D-70}$$

Using barium as an example, we can calculate:

$$C\ (mg/kg) = (3.2E{+}06\ d)(1E{-}04\ mg/kg{-}d) = 320\ mg/kg \quad \text{D-71}$$

D-4.2.1.2 Carcinogenic, Nonradioactive

$$C = \frac{TR \times BW \times AT \times PEF}{SF_i \times IR \times EF \times ED} \quad \text{D-72}$$

Substituting the appropriate exposure factors from Table A-8, we have:

$$C\ (mg/kg) = \frac{(1E{-}07)(70\ kg)(70\ yr \times 365\ d/yr)(PEF)}{(SF_i)(20\ m^3/d)(365\ d/yr)(30\ yr)} \quad \text{D-73}$$

Using the PEF calculated in equation D-69, this reduces to a summary screening factor (see Table D-1), and a toxicity value:

$$C\ (mg/kg) = \frac{(1.6E{+}01\ d)}{SF_i} \quad \text{D-74}$$

Using arsenic as an example, we can calculate:

$$C\ (mg/kg) = \frac{1.6E{+}01\ d}{1.5E{+}01\ (mg/kg{-}d)^{-1}} = 1.1\ mg/kg \quad \text{D-75}$$

D-4.2.1.3 Carcinogenic, Radioactive

$$C = \frac{TR \times PEF \times CF}{SF_i \times IR \times EF \times ED} \qquad \text{D-76}$$

Substituting the appropriate exposure factors from Table A-9, we have:

$$C\ (pCi/g) = \frac{(1E-07)(PEF)(1E-03\ kg/g)}{(SF_i)(20\ m^3/d)(365\ d/yr)(30\ yr)} \qquad \text{D-77}$$

Using the PEF calculated in equation D-69, this reduces to a summary screening factor (see Table D-1), and a toxicity value:

$$C\ (pCi/g) = \frac{(9.1E-09\ (g)^{-1})}{SF_i} \qquad \text{D-78}$$

Using ^{238}U as an example, we can calculate:

$$C\ (pCi/g) = \frac{9.1E-09\ (g)^{-1}}{5.2E-08\ (pCi)^{-1}} = 0.18\ (pCi/g) \qquad \text{D-79}$$

D-4.3 INHALATION (Volatile Compounds)

D-4.3.1 Soil

D-4.3.1.1 Noncarcinogenic

$$C = \frac{THQ \times RfD_i \times BW \times AT \times VF_s}{IR \times EF \times ED} \qquad \text{D-80}$$

where:

$$VF_s = \frac{LS \times V \times DH}{A} \times \frac{(3.14 \times \alpha \times T)^{1/2}}{2 \times D_{ei} \times E \times K_{as} \times CF} \qquad \text{D-81}$$

and,

$$K_{as} = \left[\frac{H}{K_d}\right] \times 41 \qquad \text{D-82}$$

If a K_d value is not available, it may be calculated as follows:

$$K_d = K_{oc} \times oc \qquad \text{D-83}$$

and,

$$\alpha = \frac{D_{ei} \times E}{E + \left((P_s)(1-E)/K_{as}\right)} \qquad \text{D-84}$$

Substituting the appropriate exposure factors from Table A-7 into equation D-80, we have:

$$C\ (mg/kg) = \frac{(0.1)(RfD_i)(16\ kg)(6\ yr \times 365\ d/yr)(VF_s)}{(10\ m^3/d)(365\ d/yr)(6\ yr)} \qquad \text{D-85}$$

Using methylene chloride as an example, (where D_{ei} = 6.5E-02 cm^2/s: K_{oc} = 8.8 cm^3/g: and H = 2.03E-03 atm-m^3/mol), and substituting default exposure factors from Table D-2, we can calculate a volatilization factor:

$$VF_s\ (m^3/kg) =$$

$$\frac{(45\ m)(2.25\ m/s)(2\ m)}{(20{,}250{,}000\ cm^2)} \times \frac{((3.14)(\alpha)(1.9E{+}08\ s))^{1/2}}{(2)(6.5E{-}02\ cm^2/s)(0.35)(K_{as})(1E{-}03\ kg/g)} = 860\ m^3/kg \qquad \text{D-86}$$

where:

$$K_{as}\ (g/cm^3) = \left[\frac{2.03E{-}03\ atm{-}m^3/mol}{K_d\ cm^3/g}\right] \times 41 = 0.47\ g\ soil/cm^3\ air \qquad \text{D-87}$$

and,

$$K_d\ (cm^3/g) = (8.8\ cm^3/g)(0.02) = 0.176\ cm^3/g \qquad \text{D-88}$$

and, where:

$$\alpha\ (cm^2/s) = \frac{(6.5E{-}02\ cm^2/s)(0.35)}{0.35 + \left((2.65\ g/cm^3)(1-0.35)/0.47\ g/cm^3\right)} = 5.65E{-}03\ cm^2/s \qquad \text{D-89}$$

Using this VF_s, we can reduce equation D-85 to a summary screening factor (see Table D-1) a VF_s, and a toxicity value for methylene chloride:

$$C\ (mg/kg) = (1.6E{-}01\ kg{-}d/m^3)(860\ m^3/kg)(9E{-}01\ mg/kg{-}d) = 124\ mg/kg \qquad \text{D-90}$$

D-4.3.1.2 Carcinogenic, Nonradioactive

$$C = \frac{TR \times BW \times AT \times VF_s}{SF_i \times IR \times EF \times ED} \qquad \text{D-91}$$

Substituting the appropriate exposure factors from Table A-8, we have:

$$C\ (mg/kg) = \frac{(1E{-}07)(70\ kg)(70\ yr \times 365\ d/yr)(VF_s)}{(SF_i)(20\ m^3/d)(365\ d/yr)(30\ yr)} \qquad \text{D-92}$$

Which reduces to a summary screening factor (see Table D-1), a VF_s (see D-4.3.1.1), and a toxicity value.

Using methylene chloride as an example, we can calculate:

$$C\ (mg/kg) = \frac{(8.2E{-}07\ kg{-}d/m^3)(860\ m^3/kg)}{1.6E{-}03\ (mg/kg{-}d)^{-1}} = 0.44\ mg/kg \qquad \text{D-93}$$

D-4.3.1.3 Carcinogenic, Radioactive

Not applicable

D-4.3.2 Surface and Groundwater

D-4.3.2.1 Noncarcinogenic

$$C = \frac{THQ \times RfD_i \times BW \times AT}{IR \times EF \times ED \times VF_{wvoc}} \qquad \text{D-94}$$

Substituting the appropriate exposure factors from Table A-7, we have:

$$C\ (mg/L) = \frac{(0.1)(RfD_i)(70\ kg)(30\ yr \times 365\ d/yr)}{(15\ m^3/d)(365\ d/yr)(30\ yr)(0.5\ L/m^3)} \qquad \text{D-95}$$

Which reduces to a summary screening factor (see Table D-1) and a toxicity value:

$$C\ (mg/L) = (9.3E{-}01\ kg{-}d/L)(RFD_i) \qquad \text{D-96}$$

Using methylene chloride as an example, we can calculate:

$$C\ (mg/L) = (9.3E{-}01\ kg{-}d/L)(9E{-}01\ mg/kg{-}d) = 0.84\ mg/L \qquad \text{D-97}$$

D-4.3.2.2 Carcinogenic, Nonradioactive

$$C = \frac{TR \times BW \times AT}{SF_i \times IR \times EF \times ED \times VF_{wvoc}} \qquad \text{D-98}$$

Substituting the appropriate exposure factors from Table A-8, we have:

$$C\ (mg/L) = \frac{(1E{-}07)(70\ kg)(70\ yr \times 365\ d/yr)}{(SF_i)(15\ m^3/d)(365\ d/yr)(30\ yr)(0.5\ L/m^3)} \qquad \text{D-99}$$

which reduces to a summary screening factor (see Table D-1) and a toxicity value:

$$C\ (mg/L) = \frac{(2.2E{-}06\ kg{-}d/L)}{(SF_i)} \qquad \text{D-100}$$

Using methylene chloride as an example, we can calculate:

$$C\ (mg/L) = \frac{2.2E{-}06\ kg{-}d/L}{1.6E{-}03\ (mg/kg{-}d)^{-1}} = 1.4E{-}03\ mg/L \qquad \text{D-101}$$

D-4.3.2.3 Carcinogenic, Radioactive (Radon-222)

$$C = \frac{TR}{SF_i \times IR \times EF \times ED \times VF_{wr}} \qquad \text{D-102}$$

Substituting the appropriate exposure factors from Table A-9, we have:

$$C\ (pCi/L) = \frac{(1E{-}07)}{(SF_i)(15\ m^3/d)(365\ d/yr)(30\ yr)(0.1\ L/m^3)} \qquad \text{D-103}$$

which reduces to a summary screening factor (see Table D-1) and a toxicity value:

$$C\ (pCi/L) = \frac{6.1E{-}12\ (L)^{-1}}{SF_i} \qquad \text{D-104}$$

For ^{222}Ra, we can calculate:

$$C\ (pCi/L) = \frac{6.1E{-}12\ (L)^{-1}}{7.7E{-}12\ (pCi)^{-1}} = 0.79\ pCi/L \qquad D\text{-}105$$

D-4.4 EXTERNAL EXPOSURE TO RADIONUCLIDES

D-4.4.1 Soil

$$C = \frac{TR}{SF_e \times ET \times RF \times EF \times ED \times CF} \qquad D\text{-}106$$

Substituting the appropriate exposure factors from Table A-9, we have:

$$C\ (pCi/g) = \frac{1E{-}07}{(SF_e)(24\ hr/d)(0.8)(365\ d/yr)(30\ yr)(1.14E{-}04\ yr/hr)} \qquad D\text{-}107$$

which reduces to a summary screening factor (see Table D-2) and a toxicity value:

$$C\ (pCi/g) = \frac{4.2E{-}09\ (yr)^{-1}}{SF_e} \qquad D\text{-}108$$

Using ^{238}U as an example, we can calculate:

$$C\ (pCi/g) = \frac{4.2E{-}09\ (yr)^{-1}}{3.6E{-}08\ (pCi{-}yr/g)^{-1}} = 0.12\ pCi/g \qquad D\text{-}109$$

D-5.0 EXAMPLE CALCULATIONS FOR INTAKES

This appendix provides examples for calculating contaminant intakes using the general equations provided in Section D-2.0. The equations are used to calculate daily (or lifetime, for radionuclides) contaminant intake for a chosen exposure scenario and known contaminant concentration. The examples provided in this section are for a residential exposure scenario. Intakes for other exposure scenarios may be calculated using the appropriate scenario-specific exposure parameters provided in Appendix A.

D-5.1 INGESTION

D-5.1.1 Soil and Sediment

D-5.1.1.1 Noncarcinogenic

$$\text{Intake} = \frac{C \times IR \times EF \times ED \times CF}{BW \times AT} \qquad \text{D-110}$$

Substituting the appropriate exposure factors from Table A-7, we have:

$$\text{Intake (mg/kg-d)} = \frac{(C \text{ mg/kg})(200 \text{ mg/d})(365 \text{ d/yr})(6 \text{ yr})(1E\text{-}06 \text{ kg/mg})}{(16 \text{ kg})(6 \text{ yr} \times 365 \text{ d/yr})} \qquad \text{D-111}$$

which reduces to a summary intake factor (see Table A-7) and a concentration:

$$\text{Intake (mg/kg-d)} = (C \text{ mg/kg})(1.3E\text{-}05 \ (d^{-1})) \qquad \text{D-112}$$

D-5.1.1.2 Carcinogenic, Nonradioactive

$$\text{Intake} = \frac{C \times \left[\left[\frac{IR \times ED}{BW} \right]_{child} + \left[\frac{IR \times ED}{BW} \right]_{adult} \right] \times EF \times CF}{AT} \qquad \text{D-113}$$

Substituting the appropriate exposure factors from Table A-8, we have:

$$\text{Intake (mg/kg-d)} =$$

$$\frac{C \text{ mg/kg} \times \left[\left[\frac{(200 \text{ mg/d})(6 \text{ yr})}{(16 \text{ kg})} \right]_{child} + \left[\frac{(100 \text{ mg/d})(24 \text{ yr})}{(70 \text{ kg})} \right]_{adult} \right] (365 \text{ d/yr})(1E\text{-}06 \text{ kg/mg})}{(70 \text{ yr} \times 365 \text{ d/yr})}$$

D-114

which reduces to a summary intake factor (see Table A-8) and a concentration:

$$\text{Intake (mg/kg-d)} = (C \text{ mg/kg})(1.6E\text{-}06 \ (d^{-1})) \qquad \text{D-115}$$

D-5.1.1.3 Carcinogenic, Radioactive

$$\text{Intake} = C \times \left((IR \times ED)_{child} + (IR \times ED)_{adult}\right) \times EF \times CF \qquad \text{D-116}$$

Substituting the appropriate exposure factors from Table A-9, we have:

$$\text{Intake (pCi)} = (C \text{ pCi/g}) \times \left(((200 \text{ mg/d})(6 \text{ yr}))_{child} + ((100 \text{ mg/d})(24 \text{ yr}))_{adult}\right) (365 \text{ d/yr})(1E{-}03 \text{ g/mg}) \qquad \text{D-117}$$

which reduces to a summary intake factor (see Table A-9) and a concentration:

$$\text{Intake (pCi)} = (C \text{ pCi/g})(1.3E{+}03 \text{ g}) \qquad \text{D-118}$$

D-5.1.2 Surface and Groundwater

D-5.1.2.1 Noncarcinogenic

$$\text{Intake} = \frac{C \times IR \times EF \times ED}{BW \times AT} \qquad \text{D-119}$$

Substituting the appropriate exposure factors from Table A-7, we have:

$$\text{Intake (mg/kg-d)} = \frac{(C \text{ mg/L})(1 \text{ L/d})(365 \text{ d/yr})(6 \text{ yr})}{(16 \text{ kg})(6 \text{ yr} \times 365 \text{ d/yr})} \qquad \text{D-120}$$

which reduces to a summary intake factor (see Table A-7) and a concentration:

$$\text{Intake (mg/kg-d)} = (C \text{ mg/L})(6.3E{-}02 \text{ L/kg-d}) \qquad \text{D-121}$$

D-5.1.2.2 Carcinogenic, Nonradioactive

$$\text{Intake} = \frac{C \times IR \times EF \times ED}{BW \times AT} \qquad \text{D-122}$$

Substituting the appropriate exposure factors from Table A-8, we have:

$$\text{Intake (mg/kg-d)} = \frac{(C \text{ mg/L})(2 \text{ L/d})(365 \text{ d/yr})(30 \text{ yr})}{(70 \text{ kg})(70 \text{ yr} \times 365 \text{ d/yr})} \qquad \text{D-123}$$

which reduces to a summary intake factor (see Table A-8) and a concentration:

$$\text{Intake (mg/kg-d)} = (\text{C mg/L})(1.2\text{E-}02\ \text{L/kg-d}) \qquad \text{D-124}$$

D-5.1.2.3 Carcinogenic, Radioactive

$$\text{Intake} = \text{C} \times \text{IR} \times \text{EF} \times \text{ED} \qquad \text{D-125}$$

Substituting the appropriate exposure factors from Table A-9, we have:

$$\text{Intake (pCi)} = (\text{C pCi/L})(2\ \text{L/d})(365\ \text{d/yr})(30\ \text{yr}) \qquad \text{D-126}$$

which reduces to a summary intake factor (see Table A-9) and a concentration:

$$\text{Intake (pCi)} = (\text{C pCi/L})(2.2\text{E+}04\ \text{L}) \qquad \text{D-127}$$

D-5.1.3 Biota (Fish)

D-5.1.3.1 Noncarcinogenic

$$\text{Intake} = \frac{\text{C} \times \text{IR} \times \text{EF} \times \text{ED} \times \text{AF} \times \text{CF}}{\text{BW} \times \text{AT}} \qquad \text{D-128}$$

Substituting the appropriate exposure factors from Table A-7, we have:

$$\text{Intake (mg/kg-d)} = \frac{(\text{C mg/kg})(54\ \text{g/d})(365\ \text{d/yr})(30\ \text{yr})(0.5)(1\text{E-}03\ \text{kg/g})}{(70\ \text{kg})(30\ \text{yr} \times 365\ \text{d/yr})} \qquad \text{D-129}$$

which reduces to a summary intake factor (see Table A-7) and a concentration:

$$\text{Intake (mg/kg-d)} = (\text{C mg/kg})(3.9\text{E-}04\ (\text{d}^{-1})) \qquad \text{D-130}$$

D-5.1.3.2 Carcinogenic, Nonradioactive

$$\text{Intake} = \frac{C \times IR \times EF \times ED \times AF \times CF}{BW \times AT} \qquad \text{D-131}$$

Substituting the appropriate exposure factors from Table A-8, we have:

$$\text{Intake (mg/kg-d)} = \frac{(C \text{ mg/kg})(54 \text{ g/d})(365 \text{ d/yr})(30 \text{ yr})(0.5)(1E-03 \text{ kg/g})}{(70 \text{ kg})(70 \text{ yr} \times 365 \text{ d/yr})} \qquad \text{D-132}$$

which reduces to a summary intake factor (see Table A-8) and a concentration:

$$\text{Intake (mg/kg-d)} = (C \text{ mg/kg})(1.7E-04 \ (d^{-1})) \qquad \text{D-133}$$

D-5.1.3.3 Carcinogenic, Radioactive

$$\text{Intake} = C \times IR \times EF \times ED \times AF \qquad \text{D-134}$$

Substituting the appropriate exposure factors from Table A-9, we have:

$$\text{Intake (pCi)} = (C \text{ pCi/g})(54 \text{ g/d})(365 \text{ d/yr})(30 \text{ yr})(0.5) \qquad \text{D-135}$$

which reduces to a summary intake factor (see Table A-9) and a concentration:

$$\text{Intake (pCi)} = (C \text{ pCi/g})(3.0E+05 \text{ g}) \qquad \text{D-136}$$

D-5.2 INHALATION - (Fugitive Dust)

D-5.2.1 Soil

D-5.2.1.1 Noncarcinogenic

$$\text{Intake} = \frac{C \times IR \times EF \times ED}{BW \times AT \times PEF} \qquad \text{D-137}$$

and,

$$PEF = \frac{LS \times V \times MH \times CF_1 \times CF_2}{A \times E_{10}} \qquad \text{D-138}$$

and,

Substituting the appropriate exposure factors from Table A-7, we have:

$$E_{10} = RESP \times (1 - G) \times (U_m/U_t)^3 \times F_x \qquad \text{D-139}$$

$$\text{Intake (mg/kg-d)} = \frac{(C \text{ mg/kg})(10 \text{ m}^3\text{/d})(365 \text{ d/yr})(6 \text{ yr})}{(16 \text{ kg})(6 \text{ yr} \times 365 \text{ d/yr})(PEF)} \qquad \text{D-140}$$

which reduces to a summary intake factor (see Table A-7) and a concentration, divided by a PEF:

$$\text{Intake (mg/kg-d)} = \frac{(C \text{ mg/kg})(6.3E\text{-}01 \text{ m}^3\text{/kg-d})}{(PEF)} \qquad \text{D-141}$$

By substituting default values from Table D-2:

$$\text{PEF (m}^3\text{/kg)} = \frac{(45 \text{ m})(2.25 \text{ m/s})(2 \text{ m})(3.6E{+}03 \text{ s/hr})(1E{+}03 \text{ g/kg})}{(2025 \text{ m}^2)(E_{10})} = 4.6E{+}09 \text{ m}^3\text{/kg}$$

D-142

and,

$$E_{10} \text{ (g/m}^2\text{-hr)} =$$

$$(0.036 \text{ g/m}^2\text{-hr})(1 - 0.0)\left((4.5 \text{ m/s}) / (12.8 \text{ m/s})\right)^3 (0.0497) = 7.77E\text{-}05 \text{ g/m}^2\text{-hr}$$

D-143

(As noted in Table D-2, the use of site-specific values, where available, is preferred when calculating a PEF).

D-5.2.1.2 Carcinogenic, Nonradioactive

$$\text{Intake} = \frac{C \times IR \times EF \times ED}{BW \times AT \times PEF} \qquad \text{D-144}$$

Substituting the appropriate exposure factors from Table A-8, we have:

$$\text{Intake (mg/kg-d)} = \frac{(C \text{ mg/kg})(20 \text{ m}^3\text{/d})(365 \text{ d/yr})(30 \text{ yr})}{(70 \text{ kg})(70 \text{ yr} \times 365 \text{ d/yr})(PEF)} \qquad \text{D-145}$$

which reduces to a summary intake factor (see Table A-8) and a concentration, divided by a PEF:

$$\text{Intake (mg/kg-d)} = \frac{(\text{C mg/kg})(1.2\text{E-}01\ \text{m}^3/\text{kg-d})}{(\text{PEF})} \qquad \text{D-146}$$

D-5.2.1.3 Carcinogenic, radioactive

$$\text{Intake} = \frac{\text{C x IR x EF x ED x CF}}{\text{PEF}} \qquad \text{D-147}$$

Substituting the appropriate exposure factors from Table A-9, we have:

$$\text{Intake (pCi)} = \frac{(\text{C pCi/g})(20\ \text{m}^3/\text{d})(365\ \text{d/yr})(30\ \text{yr})(1\text{E+}03\ \text{g/kg})}{(\text{PEF})} \qquad \text{D-148}$$

which reduces to a summary intake factor (see Table A-9) and a concentration, divided by a PEF:

$$\text{Intake (pCi)} = \frac{(\text{C pCi/g})(2.2\text{E+}05\ \text{g-m}^3/\text{kg})}{(\text{PEF})} \qquad \text{D-149}$$

D-5.3 INHALATION - (Volatile Compounds)

D-5.3.1 Soil

D-5.3.1.1 Noncarcinogenic

$$\text{Intake} = \frac{\text{C x IR x EF x ED}}{\text{BW x AT x VF}_s} \qquad \text{D-150}$$

Substituting the appropriate exposure factors from Table A-7, we have:

$$\text{Intake (mg/kg-d)} = \frac{(\text{C mg/kg})(10\ \text{m}^3/\text{d})(365\ \text{d/yr})(6\ \text{yr})}{(16\ \text{kg})(6\ \text{yr x }365\ \text{d/yr})(\text{VF}_s)} \qquad \text{D-151}$$

which reduces to a summary intake factor (see Table A-7) and a concentration, divided by a VF_s:

$$\text{Intake (mg/kg-d)} = \frac{(\text{C mg/kg})(6.3\text{E-}01\ \text{m}^3/\text{kg-d})}{\text{VF}_s\ (\text{m}^3/\text{kg})} \qquad \text{D-152}$$

(A contaminant-specific VF_s is calculated as demonstrated in Section D-4.3.1.1)

D-5.3.1.2 Carcinogenic, Nonradioactive

$$\text{Intake} = \frac{C \times IR \times EF \times ED}{BW \times AT \times VF_s} \qquad \text{D-153}$$

Substituting the appropriate exposure factors from Table A-8, we have:

$$\text{Intake (mg/kg-d)} = \frac{(C \text{ mg/kg})(20 \text{ m}^3/\text{d})(365 \text{ d/yr})(30 \text{ yr})}{(70 \text{ kg})(70 \text{ yr} \times 365 \text{ d/yr})(VF_s)} \qquad \text{D-154}$$

which reduces to a summary intake factor (see Table A-8) and a concentration, divided by a VF_s:

$$\text{Intake (mg/kg-d)} = \frac{(C \text{ mg/kg})(1.2E\text{-}01 \text{ m}^3/\text{kg-d})}{VF_s \text{ (m}^3/\text{kg)}} \qquad \text{D-155}$$

(A contaminant-specific VF_s is calculated as demonstrated in Section D-4.3.1.1)

D-5.3.1.3 Carcinogenic, Radioactive

Not applicable.

D-5.3.2 Surface and Groundwater

D-5.3.2.1 Noncarcinogenic

$$\text{Intake} = \frac{C \times IR \times EF \times ED \times VF_{wvoc}}{BW \times AT} \qquad \text{D-156}$$

Substituting the appropriate exposure factors from Table A-7, we have:

$$\text{Intake (mg/kg-d)} = \frac{(C \text{ mg/L})(15 \text{ m}^3/\text{d})(365 \text{ d/yr})(30 \text{ yr})(0.5 \text{ L/m}^3)}{(70 \text{ kg})(30 \text{ yr} \times 365 \text{ d/yr})} \qquad \text{D-157}$$

which reduces to a summary intake factor (see Table A-7) and a concentration:

$$\text{Intake (mg/kg-d)} = (C \text{ mg/L})(1.1E\text{-}01 \text{ L/kg-d}) \qquad \text{D-158}$$

D-5.3.2.2 Carcinogenic, Nonradioactive

$$\text{Intake} = \frac{C \times IR \times EF \times ED \times VF_{wvoc}}{BW \times AT} \qquad \text{D-159}$$

Substituting the appropriate exposure factors from Table A-8, we have:

$$\text{Intake (mg/kg-d)} = \frac{(C\ \text{mg/L})(15\ \text{m}^3\text{/d})(365\ \text{d/yr})(30\ \text{yr})(0.5\ \text{L/m}^3)}{(70\ \text{kg})(70\ \text{yr} \times 365\ \text{d/yr})} \qquad \text{D-160}$$

which reduces to a summary intake factor (see Table A-8) and a concentration:

$$\text{Intake (mg/kg-d)} = (C\ \text{mg/L})(4.6\text{E-}02\ \text{L/kg-d}) \qquad \text{D-161}$$

D-5.3.2.3 Carcinogenic, Radioactive (radon-222)

$$\text{Intake} = C \times IR \times EF \times ED \times VF_{wr} \qquad \text{D-162}$$

Substituting the appropriate exposure factors from Table A-9, we have:

$$\text{Intake (pCi)} = (C\ \text{pCi/L})(15\ \text{m}^3\text{/d})(365\ \text{d/yr})(30\ \text{yr})(0.1\ \text{L/m}^3) \qquad \text{D-163}$$

which reduces to a summary intake factor (see Table A-9) and a concentration:

$$\text{Intake (pCi)} = (C\ \text{pCi/L})(1.6\text{E+}04\ \text{L}) \qquad \text{D-164}$$

D-5.4 DERMAL EXPOSURE

D-5.4.1 Soil and Sediment

D-5.4.1.1 Noncarcinogenic

$$DAD = \frac{C \times ABS \times AF \times CF \times \left[\left[\frac{SA \times EF \times ED}{BW} \right]_{child} + \left[\frac{SA \times EF \times ED}{BW} \right]_{adult} \right]}{AT} \qquad \text{D-165}$$

Substituting the appropriate exposure factors from Table A-7, we have:

$$DAD\ (mg/kg\text{-}d) = (C\ mg/kg)(ABS)(0.2\ mg/cm^2\text{-}d)(1E\text{-}06\ kg/mg)\ x$$

$$\frac{\left[\left[\frac{(2{,}500\ cm^2)(180\ d/yr)(6\ yr)}{(16\ kg)}\right]_{child} + \left[\frac{(5{,}000\ cm^2)(180\ d/yr)(24\ yr)}{(70\ kg)}\right]_{adult}\right]}{(30\ yr\ x\ 365\ d/yr)}$$

D-166

which reduces to a summary intake factor (see Table A-7), a concentration, and an ABS:

$$DAD\ (mg/kg\text{-}d) = (C\ mg/kg)(8.7E\text{-}06\ (d)^{-1})(ABS) \qquad \text{D-167}$$

D-5.4.1.2 Carcinogenic, Nonradioactive

$$DAD = \frac{C\ x\ ABS\ x\ AF\ x\ CF\ x\left[\left[\frac{SA\ x\ EF\ x\ ED}{BW}\right]_{child} + \left[\frac{SA\ x\ EF\ x\ ED}{BW}\right]_{adult}\right]}{AT}$$

D-168

Substituting the appropriate exposure factors from Table A-8, we have:

$$DAD\ (mg/kg\text{-}d) = (C\ mg/kg)(ABS)(0.2\ mg/cm^2\text{-}d)(1E\text{-}06\ kg/mg)\ x$$

$$\frac{\left[\left[\frac{(2{,}500\ cm^2)(180\ d/yr)(6\ yr)}{(16\ kg)}\right]_{child} + \left[\frac{(5{,}000\ cm^2)(180\ d/yr)(24\ yr)}{(70\ kg)}\right]_{adult}\right]}{(70\ yr\ x\ 365\ d/yr)}$$

D-169

which reduces to a summary intake factor (see Table A-8), a concentration, and an ABS:

$$DAD\ (mg/kg\text{-}d) = (C\ mg/kg)(3.7E\text{-}06\ (d)^{-1})(ABS) \qquad \text{D-170}$$

D-5.4.1.3 Carcinogenic, Radioactive

Not applicable.

D-5.4.2 Surface and Groundwater (exposure via bathing)

D-5.4.2.1 Noncarcinogenic

$$DAD = \frac{C \times SA \times K_p \times ET \times EF \times ED \times CF}{BW \times AT} \qquad \text{D-171}$$

Substituting the appropriate exposure factors from Table A-7, we have:

$$DAD\ (mg/kg\text{-}d) = \frac{(C\ mg/L)(20{,}000\ cm^2)(K_p)(0.17\ hr/d)(365\ d/yr)(30\ yr)(1E\text{-}03\ L/cm^3)}{(70\ kg)(30\ yr \times 365\ d/yr)} \qquad \text{D-172}$$

which reduces to a summary intake factor (see Table A-7), a concentration, and a K_p:

$$DAD\ (mg/kg\text{-}d) = (C\ mg/L)(4.9E\text{-}02\ L\text{-}hr/kg\text{-}cm\text{-}d)(K_p\ cm/hr) \qquad \text{D-173}$$

D-5.4.2.2 Carcinogenic, Nonradioactive

$$DAD = \frac{C \times SA \times K_p \times ET \times EF \times ED \times CF}{BW \times AT} \qquad \text{D-174}$$

Substituting the appropriate exposure factors from Table A-8, we have:

$$DAD\ (mg/kg\text{-}d) = \frac{(C\ mg/L)(20{,}000\ cm^2)(K_p)(0.17\ hr/d)(365\ d/yr)(30\ yr)(1E\text{-}03\ L/cm^3)}{(70\ kg)(70\ yr \times 365\ d/yr)} \qquad \text{D-175}$$

which reduces to a summary intake factor (see Table A-8), a concentration, and a K_p:

$$DAD\ (mg/kg\text{-}d) = (C\ mg/L)(2.1E\text{-}02\ L\text{-}hr/kg\text{-}cm\text{-}d)(K_p\ cm/hr) \qquad \text{D-176}$$

D-5.4.2.3 Carcinogenic, Radioactive

Not applicable.

D-5.5 EXTERNAL EXPOSURE TO RADIONUCLIDES

D-5.5.1 Soil

$$Exposure = C \times ET \times RF \times EF \times ED \times CF \qquad \text{D-177}$$

Substituting the appropriate exposure factors from Table A-9, we have:

$$\text{Exposure (pCi-yr/g)} = (\text{C pCi/g})(24 \text{ hr/d})(0.8)(365 \text{ d/yr})(30 \text{ yr})(1.14\text{E-}04 \text{ yr/hr}) \quad \text{D-178}$$

which reduces to a summary intake factor (see Table A-9) and a concentration:

$$\text{Exposure (pCi-yr/g)} = (\text{C pCi/g})(2.4\text{E+}01 \text{ yr}) \quad \text{D-179}$$

D-6.0 EXAMPLE CALCULATIONS FOR RISK ASSESSMENT

This appendix provides example calculations for evaluating human health risk associated with contaminant intake (or external exposure to radionuclides) using a residential exposure scenario. Soil ingestion and external exposure are the exposure routes evaluated in the example calculations, which use summary intake factors identified in Sections D-5.1.1 and D-5.5.1. Contaminant soil concentrations of 200 mg/kg and 200 pCi/g have been arbitrarily chosen for these example calculations.

D-6.1 NONCARCINOGENIC

$$HQ = \frac{(C)(SIF)}{RfD_o} = \frac{\text{Intake}}{RfD_o} \quad \text{D-180}$$

Using methylene chloride at a soil concentration of 200 mg/kg as an example, we can calculate:

$$HQ = \frac{(200 \text{ mg/kg})(1.3\text{E-}05 \text{ (d)}^{-1})}{(6.0\text{E-}02 \text{ mg/kg-d})} = \frac{(2.6\text{E-}03 \text{ mg/kg-d})}{6.0\text{E-}02 \text{ mg/kg-d}} = 0.04 \quad \text{D-181}$$

D-6.2 CARCINOGENIC - Nonradioactive

$$ICR = (C)(SIF)(SF_o) = (Intake)(SF_o) \quad \text{D-182}$$

Using methylene chloride at a soil concentration of 200 mg/kg as an example, we can calculate:

$$ICR = (200 \text{ mg/kg})(1.6\text{E-}06 \text{ (d)}^{-1})(7.5\text{E-}03 \text{ (mg/kg-d)}^{-1}) = \quad \text{D-183}$$

$$(3.2\text{E-}04 \text{ mg/kg-d})(7.5\text{E-}03 \text{ (mg/kg-d)}^{-1}) = 2\text{E-}06$$

D-6.3 CARCINOGENIC - Radioactive (internal)

$$ICR = (C)(SIF)(SF_o) = (Intake)(SF_o) \quad \text{D-184}$$

Using ^{238}U at a soil concentration of 200 pCi/g as an example, we can calculate:

$$ICR = (200\ pCi/g)(1.3E{+}03\ g)(2.8E{-}11\ (pCi)^{-1}) = (2.6E{+}05\ pCi)(2.8E{-}11\ (pCi)^{-1}) = 7E{-}06 \quad \text{D-185}$$

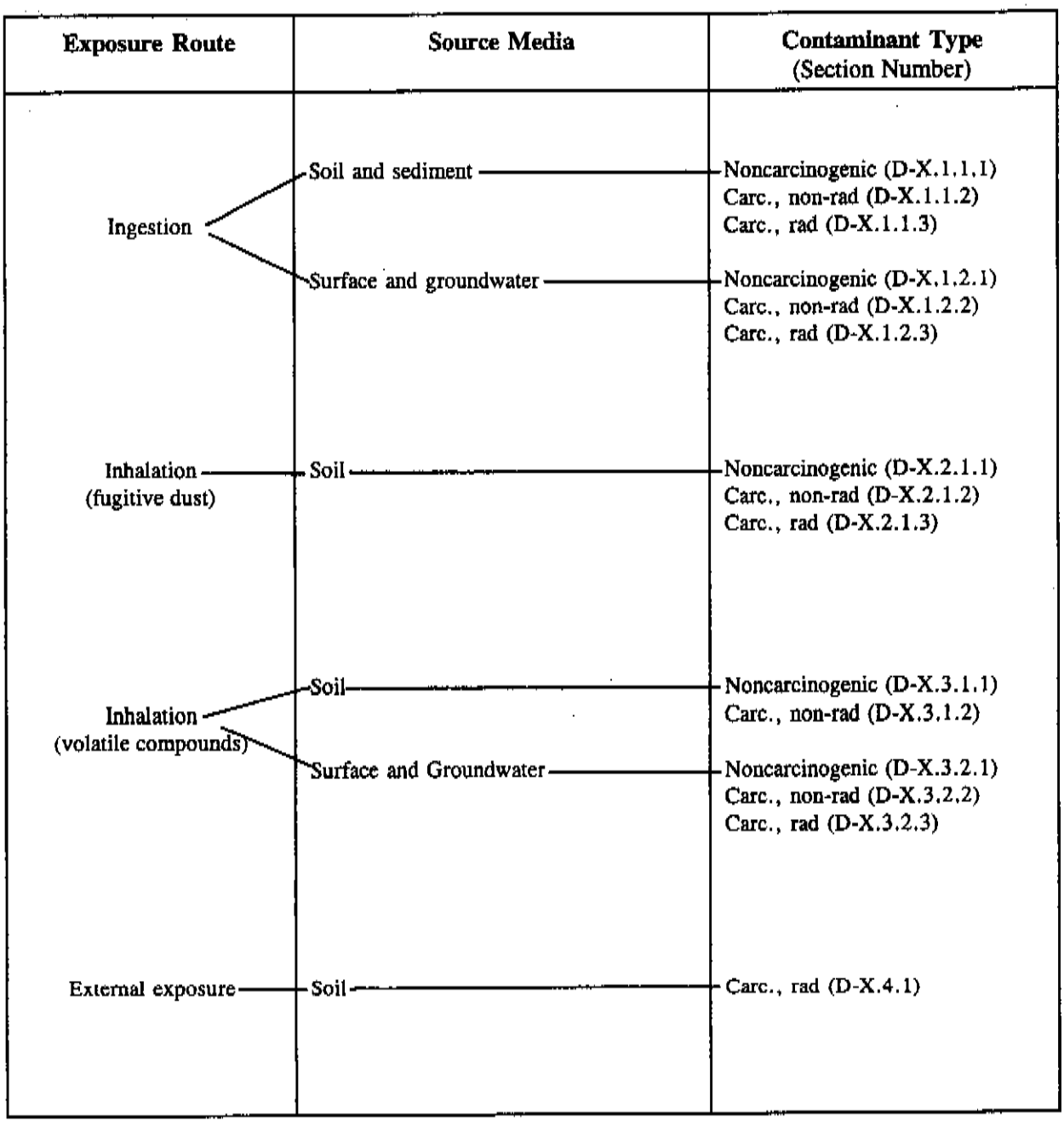
D-6.4 CARCINOGENIC - Radioactive (external)

$$ICR = (C)(SIF)(SF_e) = (Exposure)(SF_e) \quad \text{D-186}$$

Using ^{238}U at a soil concentration of 200 pCi/g as an example, we can calculate:

$$ICR = (200\ pCi/g)(2.4E{+}01\ yr)(3.6E{-}08\ (pCi{-}yr/g)^{-1}) = (4.8E{+}03\ pCi{-}yr/g)(3.6E{-}08\ (pCi{-}yr/g)^{-1}) = 2E{-}04 \quad \text{D-187}$$

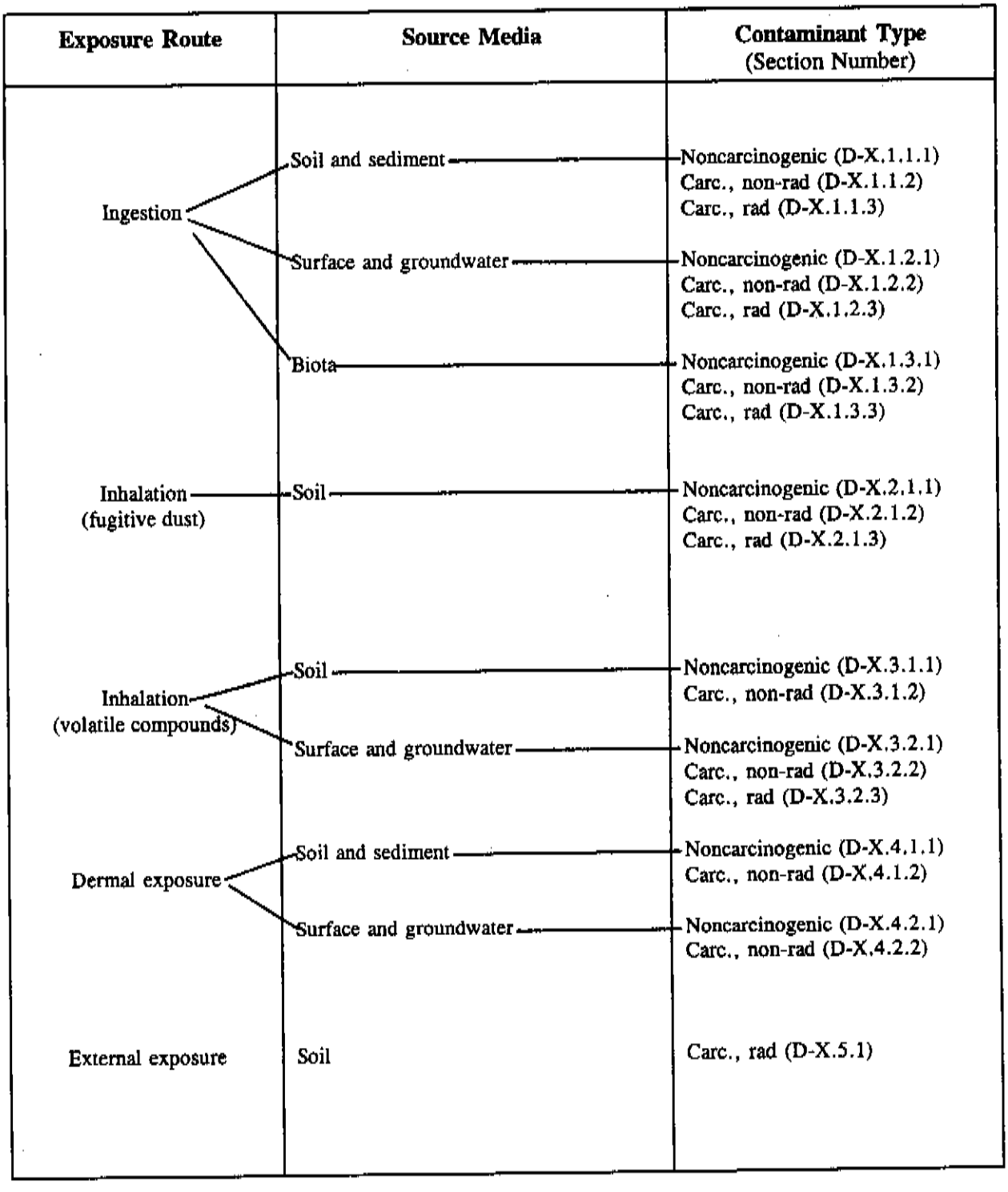
Figure D-1. Flowchart for Preliminary Risk-Based Screening Equations and Example Calculations.



For preliminary risk-based screening/risk-based calculation equations and parameter definitions, X = 1.

For example, preliminary risk-based screening calculations, X = 4.

Figure D-2. Flowchart for Intake Equations and Example Calculations.



For intake equations and parameter definitions, X = 2.

For example, intake calculations, X = 5.

References:

EPA, 1993a, *Integrated Risk Information System (IRIS)*, data file, U.S. Department of Health and Human Services, National Library of Medicine Toxicology Data Network (TOXNET), Bethesda, Maryland.

EPA, 1993b, *Health Effects Summary Tables: Annual FY-1993*, OHEA/ECAO-CIN-821, March 1993, U.S. Environmental Protection Agency, Office of Emergency and Remedial Response, Washington, D.C.

EPA, 1991, *Risk Assessment Guidance for Superfund: Volume 1, Human Health Evaluation Manual, Part B; Development of Risk-Based Remediation Goals; Review Draft*, OSWER 9285.7-01B, U.S. Environmental Protection Agency, Washington, D.C.

EPA, 1989, *Risk Assessment Guidance for Superfund: Volume 1, Human Health Evaluation Manual, Part A; Interim Final*, EPA/540/1-89/002, U.S. Environmental Protection Agency, Washington, D.C.

EPA-10, 1991, *Supplemental Risk Assessment Guidance for Superfund*, U.S. Environmental Protection Agency, Region X, Seattle, Washington.

Summary screening factors (SSF) are presented for preliminary risk-based screening calculations for a residential exposure scenario using a target incremental lifetime cancer risk of 1E-07 and a target hazard quotient of 0.1. Residential exposure factors used in calculating these SSFs can be found in Appendix A (Tables A-7, A-8, and A-9). Equations for calculating preliminary risk-based screening calculations can be found in Section D1.0. For carcinogens, a risk-based concentration can be determined by dividing the SSF by the contaminant-specific slope factor. For noncarcinogens, a risk-based concentration can be determined by multiplying the SSF by the contaminant-specific reference dose.

Table D-1. Summary Screening Factors for Preliminary Risk-Based Screening.

Exposure Route	Source Media	Summary Screening Factor by Contaminant Category		
		Noncarcinogenic	Carcinogenic - Nonradioactive	Carcinogenic - Radioactive
Ingestion	Soil	8.0E+03 d	6.4E-02 d	7.6E-11(g)$^{-1}$
	Surface and Groundwater	1.6E+00 kg-d/L	8.2E-06 kg-d/L	4.6E-12 (L)$^{-1}$
Inhalation[a] (Fugitive Dust)	Soil	3.2E+06 d	1.6E+01 d	9.1E-09 (g)$^{-1}$
Inhalation (Volatiles)	Soil	1.6E-01 kg-d/m^3 x VF_s (m^3/kg)	8.2E-07 kg-d/m^3 x VF_s (m^3/kg)	Not applicable
	Surface and Groundwater	9.3E-01 kg-d/L	2.2E-06 kg-d/L	6.1E-12 (L)$^{-1}$
External Exposure	Soil	Not applicable	Not applicable	4.2E-09 (yr)$^{-1}$

[a]Assuming PEF = 2E+07 m^3/kg.

Table D-2. Default Parameters for Calculating a Particulate Emission Factor and Soil Volatilization Factor[a].

Particulate Emission Factor		Volatilization Factor	
Parameter	Default Value[b]	Parameter	Default Value[b]
PEF (for Preliminary Risk-Based Screening)	2E+07 m^3/kg	LS	45 m
NAAQS	50 $\mu g/m^3$[c]	V	2.25 m/s
LS	45 m	DH	2 m
V	2.25 m/s	A	20,250,000 cm^2
MH	2 m	T	Equal to exposure duration, expressed in units of seconds
A	2025 m^2	D_i	Contaminant-specific
E_{10}	7.77E-05 g/m^2-hr	E	0.35
RESP	0.036 g/m^2-hr	H	Contaminant-specific
G	0.0	K_{oc}	Contaminant-specific
U_m	4.5 m/s	oc	0.02
U_t	12.8 m/s	P_s	2.65 g/cm^3
F_x	0.0497		

[a]Use of site-specific values is preferred when available.
[b]All values are taken from RAGS, Part B (EPA 1991) unless otherwise indicated.
[c]40 CFR §50.6(b).

APPENDIX E

ECOLOGICAL DOSE AND EXPOSURE CALCULATIONS

CONTENTS

TABLES

E-1.0 ECOLOGICAL DOSE AND EXPOSURE CALCULATIONS

E-1.1 RADIATION DOSE

The most recent and perhaps one of the most inclusive reviews on the effects of ionizing radiation on terrestrial organisms was completed by Rose (1992). Rose summarized the sensitivities of wildlife to ionizing radiation.

Rose 1992 reported the lower limits of lethal effects for chronic irradiation was 360 rad/year or roughly 1 rad/day for several American rodents (French et al. 1967, 1974). The lower limit for red pine was reported to be around 0.82 to 1.64 rad/day for continuous exposure (Sparrow et al. 1963, 1970). Semagin (1975) reported a dose of 0.008 rad/day as the lowest dose that produced an effect on the fetuses of laboratory rats irradiated during the third period of intrauterine life. It was found that body mass was reduced and brain mass increased at birth. The increase in brain mass was the result of nerve tissue and not oedema. The reported range for developmental and behavioral changes from chronic irradiation exposure was also summarized by Rose (1992). An exposure of 0.49 rad/day did not affect the growth rate of several American rodents, e.g., *Peromyscus leucopus* (Childs et al. 1966). Pocket mice (*Pergnathus formosus*) were reported unaffected at a dose of 0.96 rad/day (French et al. 1967). Mellinger and Schultz (1975) reviewed the literature on the effects of ionizing radiation on birds. The LD-50 values for birds ranged from 460 to 3,000 rad.

In another extensive review of the affects of ionizing radiation on terrestrial organisms, the International Atomic Energy Agency (IAEA) (1992), concluded that a "dose rate of approximately 10 mGy/d (1 rad/d) represents the threshold at which slight effects of radiation become apparent in those attributes, e.g., reproduction capacity, which are of importance for the maintenance of the population." IAEA concluded that "reproduction was the population attribute most sensitive to damage from chronic irradiation and also the attribute of greatest significance in the ecological context".

Ionizing radiation affects the cells of aquatic organisms at a biochemical level. The level of impact is controlled to some extent by environmental factors. Detailed information on the effects of ionizing radiation on aquatic organisms can be found in a report by the National Council of Radiation Protection (NCRP 1991). This report discusses the relationship between the response of aquatic organisms to differing levels of ionizing radiation.

The nonstochastic threshold for effects from acute exposure to ionizing radiation during early developmental stages of aquatic organisms is about 3 rad during the entire one-cell stage of development. Radiosensitivity decreases with increasing embryonic development (Frank 1971). All organisms during rapidly growing stages of their life cycle are more radiosensitive than mature organisms. Laboratory studies of Chinook salmon indicate that early life stages appear to be the most radiosensitive period during the entire salmon life cycle. Damage has been reported to occur when the dose reached 9.64 rad/day over the 81-day developmental period (Hyodo-Taguchi 1980). Other Chinook salmon studies indicate that an acute dose of 224 rad reduced female germ cells in Chinook salmon (Frank 1971). Frank (1971) has also shown that an acute dose of 600 rad reduced female germ cells in rainbow steelhead trout. The conclusions from research on salmon and trout can be extrapolated to all fish. Simpler life forms (e.g., algae) can be expected to be more resistant to ionizing radiation than fish (IAEA 1992).

Few studies have evaluated the effects of chronic exposure to ionizing radiation. A summary report by the NCRP (1991) stated that Chinook salmon chronically exposed to 5.1 rad/day for up to 69 days (as embryos and alevins up to release as smolts) produced no increase in mortality. Hershberger et al. (1978) reported lower return of spawning adult Chinook salmon after exposure of eggs and alevins to approximately 10 rad/day of gamma radiation. Gonadal development was retarded in Chinook salmon upon exposure to 10 rad/day (Bonham and Donaldson 1972).

Limits on radionuclide concentrations in water are based upon DOE Order 5400.5, which requires that maximum dose to aquatic animals should not exceed 1 rad/day. A review of current literature indicates that the dose limit promulgated in DOE Order 5400.5 is sufficiently conservative to protect most aquatic organisms. Because of its conservative nature, the concentration limits should also protect populations and the ecosystem in general. This assumption will be made unless additional data indicates otherwise.

An exception to the general findings of the literature reviewed regarding the acceptability of a 1 rad/day dose limit is found in a report by Erickson (1973). Erickson observed reduced male *Poecilia retculata* (guppy) courting activity when embryos had been exposed to 0.4 rad/day. However, little additional information exists with regards to behavioral changes in fish from exposure to ionizing radiation.

E-1.1.1 Internal Radiation Dose to Terrestrial Receptors

This section describes the methods used to estimate intake of radionuclides located in surface soils by the Great Basin pocket mouse. Radiological dose calculations are based upon a modified model developed for aquatic systems (Baker and Soldat 1992), human health equations (EPA 1991), and organic contaminants (DOE 1993).

Radionuclide concentrations in animals can be calculated directly from soil concentrations and soil-to-plant transfer factors (Table E-1). Thus, the radionuclide concentrations and consequently the dose rate for the pocket mouse can be calculated from their diet of plants.

The internal total-body dose rate to a pocket mouse for N radionuclides is given as

$$R_c = \sum_{i=1}^{N} \frac{(CS_i\ PS_i\ WW\ Qv\ FI\ EF\ ED\ FR\ B_i\ E_i\ Q_i)}{BW\ AT} \qquad \text{(E-1)}$$

The parameter "CS" is the radionuclide concentration in the soil in units of Ci/kg. Because the soil concentration is used in estimating dose to the pocket mouse, the parameters "WW" (wet weight) and "PS" are required. The parameter "PS" is the soil-to-plant conversion factor specific to a given radionuclide and chemical form in the soil. The values for PS were obtained from available literature. The highest reported values for PS were generally used in all dose calculations to provide conservative estimates. Specific PS values were used when Hanford-area data were available.

The parameter "QV" is the daily plant ingestion rate. The parameter FI accounts for the fraction of plants ingested per day from the contaminated source. The parameter "EF" accounts for the number of days per year the mouse eats contaminated plants. The parameter "ED" is the exposure duration (which is always one year in these dose calculations). Pocket mice are assumed to spend their entire

life in a waste site; hence, exposure duration (ED and exposure frequency (EF) are not important in these calculations.

The parameter "FR" refers to the fraction of the radionuclide initially consumed that is taken up by the body. This value was obtained from *ICRP 1959*. Note that since the radionuclides are taken up by plants from the soil, the chemical form taken up by the mouse is necessarily soluble. The solubility class affects dose because it impacts both the fraction that deposits and the biological half-life.

In the absence of specific data, the removal constants, $\lambda_{i,c}$, and uptake fractions, FR are taken to be that of Standard Man as derived from Publication 2 of the International Commission on Radiological Protection (ICRP 1959).

The parameter "B_i" refers to the effective decay constant of radionuclide "i" once taken up by the pocket mouse. This parameter accounts for removal by both biological and radiological decay processes. The equation for B_i is:

$$B_i = (1- e^{-\lambda i,cTe}) / \lambda_{i,c}, \qquad \text{(E-2)}$$

where $\lambda_{i,c}$ are removal constants obtained from equation 3:

$$\lambda_{i,c} = (\lambda_b + \lambda_r) \qquad \text{(E-3)}$$

the effective decay constant of radionuclide in the pocket mouse per day,

$$\text{where } \lambda_b = \ln(2)/T_b) = 0.693/T_b.$$

This is the biological removal rate constant for nuclide (i) in the pocket mouse; and

$$\lambda_r = \ln(2)/T_r = 0.693/T_r$$

This is the radiological removal rate for the nuclide (i).

Once values for r and b are obtained, equation E-2 is used to estimate the sum of the removal factors.

Effective Energy Absorbed (E_i)

$$E_i = 5.12E4\ \epsilon_i \qquad \text{(E-4)}$$

$$E_i = \epsilon_i \text{ MeV dis}^{-1} \text{ x } 3.70E10 \text{ dis sec}^{-1} \text{ Ci}^{-1} \text{ x } 86{,}400 \text{ s d}^{-1} \text{ x } 1.602E\text{-}11 \text{ kg rad MeV}^{-1}$$

The values 1.602E-11 kg rad/MeV and 86,400 s d^{-1} and 3.7E10 dis sec^{-1} are conversion constants.

The parameter "E_i" accounts for the effective energy absorbed since the radius of the mouse is small. Thus, some of the gamma rays emitted internally will not be absorbed by the organism. Not taking this affect into account would result in a biased high dose estimate. To determine the effective energy absorbed, the "effective energy" for the radionuclide and organism size is needed. The effective energy is a modification of a given radionuclide's gamma energy spectrum. This modification accounts for the escape of some gamma rays as a function of organism size (i.e., radius) so that absorption can be computed by equation 4. Effective energy values were determined for a series of radii and radionuclides and are compiled in Soldat et al. (1974). Because a given radionuclide may

emit gammas of different energies at varying frequencies (i.e., probability per decay), the effective energy values are based on a radionuclide-specific, weighted-average gamma energy. The values for effective absorbed energy were determined as a function of nuclide energies and diameter as described in ICRP-2 (1959). Among the assumptions in ICRP-2 include that a spherical geometry was assumed and all of the activity was assumed to be located in the center of the organism. All particulate radiation was assumed totally absorbed within the organ. Thus, only calculations were for the fraction of each gamma ray emitted that was actually absorbed. Soldat et al. (1974) computed the fraction absorbed by the standard equation (mu/density x thickness) to compute weighted averages for all the decay (energies) schemes for a given nuclide. The quality factor (rem/rad) was removed (Soldat et al. 1974) from the equations and from the effective energy values printed in ICRP 2. This is because the quality factor applies only to human health effects and all animal doses have to be reported in rad. These then are the values in Baker and Soldat (1992).

The exposure time, T_e, is usually assumed to be 1 year for regulatory purposes, and the concentration is averaged over 1 year. T_e, is usually assumed to be 1 year for regulatory purposes.

The parameter "BW" normalizes the dose to a given weight (i.e., kg) and "AT" normalizes the dose to that obtained in one day.

Example Internal Dose Calculation for Great Basin Pocket Mouse (Strontium-90 Scenario)

Given a site in which the soil is uniformly contaminated with 35 pCi/g strontium-90, the internal dose rate calculation for the Great Basin Pocket Mouse is calculated using Equation E-1. For the pocket mouse, the default parameters for the calculation are listed in Table E-1 and E-2.

Energy absorbed = 1.14 MeV dis^{-1}
Plant uptake factor for strontium = 19
Fraction retained (FR) = 0.3
Biological half-life T_b = 244 day
$\lambda_b = \ln(2)/T_b$ = 0.693/9.5 day = 2.84E-03/day
Radiological half-life (T_r) = 5.271 years x 365 days/year = 1924 day (Shleien 1992)
$\lambda_r = \ln(2)/T_r$ = 0.693/1924 day = 3.6E-04/day

Using equation E-3 we can compute the effective decay constant (λ) for strontium-90 from the pocket mouse.

$\lambda = (\lambda_b + \lambda_r)$ effective decay constant of radionuclide (i) in the pocket mouse
λ = 2.84E-03/day + 3.6E-04/day = 3.2E-03/day

And substituting the values for the effective decay into equation E-2 we get the sum of the removal factors (B_i).

$$B_i = (1- e^{-\lambda i,cTe}) / \lambda_{i,c} = [1- e^{(-3.2E-03/day)(365day)}]/0.073 = 9.5 \text{ day}$$

Effective absorbed energy (E_i) is computed by using equation E-4 and substituting the strontium-90 values for the radionuclide energy absorbed (ϵ_i) (Table E-2) in a 2-cm diameter mouse.

$$(E_i) = 5.12E4 \text{ x } 1.14 \text{ MeV/dis} = 58140 \text{ kg rad } Ci^{-1} d^{-1}$$

Finally, substituting all the values into equation E-1 we get:

$$R_c = (CS_i\ PS_i\ WW\ Q_v\ FI\ EF\ ED\ FR\ B_i\ E_i) / BW\ AT, \text{ and:}$$

$$R_c = [(35E\text{-}9\ Ci/kg)(19)(0.32)(0.0067\ kg/day)(1.0)(365\ day/yr)(1\ yr)$$
$$(0.3)(9.5\ day)(58140\ kg\ rad\ Ci^{-1}\ d^{-1})]/(0.023\ kg)(365\ day/yr)(1\ yr)$$

Conclusion: $R_c = 0.01$ rad/day

E-1.1.2 External Radiation Dose to Terrestrial Receptors

External radiation exposure to the pocket mouse is mainly from terrestrial sources of radiation (radionuclides in plants, soil, rocks) and from cosmic rays. The radionuclide inventory at many Hanford Site waste sites, induce a much higher dose from consumption than from external radiation. This is because alpha and beta particles do not contribute to the dose rate unless internally deposited. Moreover, alpha particles have a large linear energy transfer resulting in a high imparted dose to a small localized area once inside the body. Also, radionuclides deposit in target organs delivering a dose over a prolonged time. Depending on the chemical species, some radionuclides deposit essentially for the lifetime of the animal.

For mice, the external dose includes exposure from gamma and x-rays, but not alpha and beta particles. Because of their particulate nature, alpha and beta particles do not penetrate the skin surface. The external dose from gamma and x-ray energies to the pocket mouse is greatest while the mouse is underground in his burrow. While underground, the mouse can be considered in an infinite medium uniformly contaminated by the gamma emitter.

The following equation (Schleien 1992) provides an example of how external dose can be estimated while the mouse is underground in soil contaminated by a gamma emitter. Additional assumptions may also apply which may reduce or increase the potential dose while the pocket mouse is underground:

$$\text{External Dose Rate (D)} = 2.12\ EC/p, \tag{E-6}$$

where: E = the average gamma energy per disintegration in MeV
C = the concentration in uCi/cm^3
p = the density of the medium in g/cm^3
D = rad/hour.

Therefore, the sum of the dose from several radionuclides is equal to

$$(D_i) = 2.12\ E_iC_i/p, \tag{E-7}$$

D_i, (rad/h), is the sum of the doses for radionuclide (i)

where: E_i = average gamma energies per disintegration in MeV, and equals the sum of the probabilities per decay times its energy levels in MeV for each radionuclide (i)
C = the concentration in uCi/cm^3 for radionuclide (i)
p = density of the medium in g/cm^3 which the mouse resides (e.g., sand = 1.8 g/cm^3).

Each radionuclide has a number of decay schemes. Most decay modes emit gamma or X-rays with unique energies, and there is a probability associated with each potential mode of decay. Thus, for a given radionuclide, the summing of the product of each decay mode's energy and probability is required (note that the probabilities must be equal to or less than 1). The sum of these products is then multiplied by the conversion factor 2.12 (as indicated by equation E-6).

During the summer, a pocket mouse would spend about 1/3 of its time above ground foraging. However, because the mouse still receives an external dose when on the surface (although it is not surrounded by contaminated medium), it is conservative to assume an "infinite cloud" when computing external dose. Several radionuclides, if present at a waste site would generate no external dose to the mouse (Sr-90, C-14, N-63, H-3) since they do not emit penetrating gammas. An example of external dose calculated from radionuclides present for a hypothetical waste site (Table E-3) shows the total external dose rate is approximately equal to the total dose (internal + external). For this example waste site, the soil concentrations of europium are usually high and are not typical of many of the Hanford waste soils. However, this example illustrates a situation (soil concentrations) where the external dose rate to the pocket mouse as well as internal dose rate exceed the 1 rad/day benchmark.

When strong beta emitters such as strontium-90 are present at a waste site, the external dose portion of the total dose rate is usually minimal when compared to the total dose rate (from both internal and external). When strontium-90 is not present in soil, the total dose rate (sum of internal and external) is often less than the EHQ anyway. However, this is not always the case. Table E-4 shows the approximate contaminant soil concentrations that would be needed to reach a EHQ of 1 for individual radionuclides from either the external or internal dose rate. For this example, the assumptions are that the mouse is underground for a 24 hr period in uniformly contaminated soil and consumes 6.7 grams contaminated food stores during that same period. For radionuclides that contribute primarily external dose (cobalt-60, europium-154, europium-152, cesium-134, americium-241, thorium-228, and thorium-232) [see Table E-3], the concentration of the radionuclide in soil (pCi/g) needed (Table E-4) to reach individual EHQ levels of 1 rad/day (by either external or internal dose) is often quite large. Generally, this "threshold" concentration is not approached (except for the example in Table E-4) in Hanford soils for the external-dose driving radionuclides listed. See Table E-4 for the approximate soil concentration (pCi/g) needed to exceed individual EHQs of 1 rad/day dose rate for either external or internal dose by radionuclide. Also, note that the daily dose rate to the pocket mouse is determined by the sum of the doses from all radionuclides (equation E-1) and the "individual" thresholds listed in Table E-4 are intended as illustration of a potential screening process of dose-driving contaminants.

Radionuclides which primarily contribute to internal dose (see Table E-3) for pocket mice include strontium-90, technetium-99, plutonium-238, plutonium-239, radium-226, and tritium. When strontium-90 is present, the internal dose rate from strontium-90 is generally the dose driver for pocket mouse.

E-1.1.3 Dose Rate from Naturally Occurring Radionuclides

The internal doses to terrestrial organisms from K-40 (natural background dose) cannot be calculated from the formulas such as E-1. Since K-40 content of the body is controlled metabolically to a constant level, the normal K-40 content must be determined from a textbook or by analysis of the specific organism. Once determined, the concentration in picocuries per gram then can be used with the following formula to find the internal dose rate in rad per day. This formula is based on the dose

methodology of ICRP Publication 2, as are the equations in the CRITR2 code (Baker and Soldat 1992).

$$\text{rad/day} = 5.12\text{E-}05 \times \epsilon \times C \qquad \text{(E-11)}$$

where C = the concentration of K-40 measured in the organism, pCi/g
ϵ = MeV/dis for the appropriate "effective radius" of the organism.

Equation (E-11) was used to compute the dose to a pocket mouse from K-40. The information on K-40 is from the Standard Man (ICRP#23) data.

Calculation of dose to mouse:

$$\text{rad/day} = 5.12\text{E-}05 \times \epsilon \times C$$

where: C = the conc. of K-40 measured in the organism (pCi/g)
ϵ = MeV/dis for 2-cm effective radius (0.532 MeV/dis)

Using ICRP#23 for Standard Man, potassium (K) is 0.2% of total weight and the fraction of K-40 = 0.0118%.

SA (specific activity) = $(\lambda \times 6.022\text{E+}23)/A$ Bq/g = 7.163E+06 pCi/g,

where λ = $0.693/T$ = 1.72E-17
T = half-life (seconds) 1.277E+09 y = 4.0271E+16 seconds
A = atomic weight = 39.102
and a conversion factor of 27.03 pCi/Bq is used.

Assuming the percent potassium and specific activity of K-40 in the man and mouse to be equal, the percent of K in the mouse times the percent of K-40 times the specific activity of K-40 equals the concentration (C) of K-40 in the mouse.

$$C\ (\text{pCi/g}) = 0.002 \times 0.000118 \times 7.63\text{E+}06 = 1.69\ \text{pCi/g of K-40 in mouse}$$

Estimation of the radiation dose rate from K-40 equals:

rad/day = 5.12E-05 x E x C
= 5.12E-05 x 0.532 MeV/dis [from Table E-5] x 1.69 pCi/g
= 0.000046 rad/day
= 4.6E-05 rad/day dose rate from natural occurring K-40

The conclusion can also be checked by comparing the above results to the potassium content in the Reference Man. The dose to the mouse must be less than the dose to the Reference Man, because the smaller radius of the mouse allows more of the gamma energy to escape. The value for epsilon for standard man is 0.612 MeV/dis. The ratio of doses to mouse and man therefore is simply the ratio of 0.532 to 0.612. The dose to man from natural K-40 is only about 20 mrem/yr (EDE). Therefore, the mouse dose is (20 x 0.532)/0.612 = 17.4 mrad/yr. The daily dose would be 17.4/365.25 = 0.048 mrad/d.

The dose rate from natural occurring K-40 is much less than the EHQ of 1 rad/day, and therefore, can be ignored when computing the total dose rate to the small vertebrates. In addition, K-40 dose may not apply toward estimate of total dose, since K-40 is essentially background dose.

E-1.1.4 Radiation Dose to Aquatic Organisms

This section describes the methods used to estimate radiological dose to aquatic organisms. Doses to primary organisms from radionuclide uptake and immersion in contaminated water are calculated by the computer code CRTR2 developed by Baker and Soldat (1992). Primary aquatic organisms are those which reside continually in the water. Thus, ratios between radionuclide concentrations in the water and the primary organism (i.e., bioaccumulation factors) are available. CRTR2 also calculates dose to secondary organisms which feed upon contaminated primary organisms. Equations and parameters in CRTR2 necessary to compute dose to primary organisms (categories include fish, and algae) and to secondary organisms are summarized by Baker and Soldat (1992). These parameters include bioaccumulation factors, physical and biological half-lives, uptake fractions, the effective radionuclide energy for the organism's size, and immersion and sediment dose factors.

Bioaccumulation factors are obtained from a data library in the Hanford Environmental Radiation Dosimetry Software System (GENII) (Napier et al. 1988). The bioaccumulation factors relate the concentration of radionuclides in aquatic organisms to the concentration of radionuclides in the water.

The physical half-lives are from ICRP (1979-1988). These half-lives are the amount of time required for the activity of a given radionuclide to decrease by half because of radiological decay.

The biological half-lives are for the "standard man" and were obtained from (NRC 1977; ICRP-2 (1959); and ICRP-10 (1968). The biological half-life is the amount of time required for the human body to eliminate by natural biological means half of the material taken in. The biological half-lives and uptake fractions for the human body do not necessarily apply to aquatic organisms. However, because of a lack of specific data the values for humans are used in these dose assessments for aquatic organisms.

The uptake fractions are also for the "standard man" and were obtained from ICRP-30 Parts 1 through 4 (1979-1988). Uptake fractions refer to the fraction of radioactive materials initially retained in the total body of the secondary organism. The fraction that is initially retained will then be removed by a rate given by the biological half-life.

The parameter titled "effective energy absorbed" is needed since the radius of typical aquatic organisms of concern is small. Thus, some of the gamma rays emitted internally will not be absorbed by the organism. Not taking this affect into account would result in a biased high dose estimate. To determine the effective energy absorbed, the "effective energy" for the radionuclide and organism size is needed. The effective energy is a modification of a given radionuclide's gamma energy spectrum. This modification accounts for the escape of some gamma rays as a function of organism size (i.e., radius) so that absorption can be computed by equation E-2. Effective energy values were determined for a series of radii and radionuclides and are compiled in Soldat et al. (1974). Because a given radionuclide may emit gammas of different energies at varying frequencies (i.e., probability per decay), the effective energy values are based on a radionuclide-specific, weighted-average gamma energy. The values for effective absorbed energy were determined as a function of nuclide energies and diameter as described in ICRP-2 (1959). Among the assumptions in ICRP-2 include that a spherical geometry was assumed and all of the activity was assumed to be located in the center of the

organism. All particulate radiation was assumed totally absorbed within the organ. Thus, only calculations were for the fraction of each gamma ray emitted that was actually absorbed. Soldat et al. (1974) computed the fraction absorbed by the standard equation (mu/density x thickness) to compute weighted averages for all the decay (energies) schemes for a given nuclide. The quality factor (rem/rad) was removed (Soldat et al. 1974) from the equations and from the effective energy values printed in ICRP 2. This is because the quality factor applies only to human health effects and all animal doses must be reported in rad. These then are the values in Baker and Soldat (1992). The exposure time, T_e, is usually assumed to be 1 year for regulatory purposes.

E-1.1.4.1 Aquatic Internal Dose Calculations. Total daily doses to a primary organism are estimated as the sum of doses (based on a radionuclide-specific, weighted-average gamma energy) received from internal and external exposure to all radionuclides in the aquatic environment. CRITR2 is a steady state model which assumes dose rates of exposed organisms reach an equilibrium with the water concentration or food uptake.

The internal total-body dose rate (rad/day) to an organism for N radionuclides is given by equation E-12

$$R_c = \sum_{i=1}^{N} b_{i,c} E_{i,c} \qquad \text{(E-12)}$$

where: R_c = internal total-body dose rate of organism c (rad/day)
$b_{i,c}$ = specific body burden of nuclide i in organism c (Ci/kg)
$E_{i,c}$ = effective energy-absorbed for radionuclide (i) per unit activity in organism c (kg-rad/Ci-d).

$E_{i,c}$ is computed by equation E-13

$$E_{i,c} = \epsilon_{i,c} \text{ MeV dis}^{-1} \times 3.70\text{E}+10 \text{ dis s}^{-1} \text{ Ci}^{-1} \times 86{,}400 \text{ s d}^{-1} \times 1.602\text{E-}11 \text{ kg-rad MeV}^{-1}$$

$$E_{i,c} = 5.12\text{E}04 \times \epsilon_{i,c} \qquad \text{(E-13)}$$

where ϵ = effective radionuclide energy for diameter of aquatic organism for nuclide i in organism c.

For a primary organism the specific body burden of nuclide i in organism c (Ci/kg) for radionuclide i by equation E-14

$$b_{i,c} = C_{i,c} BF_{i,c} CF_{i,c} \qquad \text{(E-14)}$$

where $b_{i,c}$ = specific body burden of nuclide i in organism c (Ci/kg)
$C_{i,c}$ = concentration of radionuclide i in water to which organism c is exposed (Ci/L)
$BF_{i,c}$ = bioaccumulation factor for nuclide i and organism c (m^3/kg).
CF = conversion factor [0.001 (L/m^3)]

Combining equations (E-12) and (E-14) yields the dose rate (rad/day) to the primary organism, as given by equation E-15 and the parameters as defined previously.

$$R_c = \sum_{i=1}^{N} E_{i,c}\, C_{i,c}\, BF_{i,c}\, CF_{i,c} \qquad \text{(E-15)}$$

For secondary organism (such as carnivores) the change in body burden of a single radionuclide is a function of the uptake and removal rate of the radionuclide as expressed in E-16.

$$\frac{db^s}{dt} = \frac{P}{M} - \lambda b^s \qquad \text{(E-16)}$$

where: b^s = specific body burden of the secondary organism (Ci/kg)
M = mass of secondary organism (kg)
P = rate of uptake of radionuclide by body of secondary organism (Ci/d)

The uptake rate by a secondary organism is given by equation E-17

$$P_i = b_i\, U_i\, f_{1,i} \qquad \text{(E-17)}$$

where b = body burden of primary organism (Ci/kg)
U = intake rate of primary organism by secondary organism (kg/d)
f_1 = fraction of radionuclide initially retained in total body of secondary organism (unitless).

Solving E-16 when $b^s = 0$ and $t = 0$ results in equation E-18 for the computation of specific body burden:

$$b^s = P \times Bi/M \qquad \text{(E-18)}$$

where: B_i = effective decay constant of radionuclide i in the secondary organism (day),
$Bi = (1 - e^{-\lambda i,cTe}) / \lambda_{i,c}$,
$\lambda_{i,c} = (\lambda_b + \lambda_r)$ (day^{-1})

where: $\lambda_b = \ln(2)/T_b) = 0.693/T_b$

This is the biological removal rate constant for radionuclide i in organism c; and

$$\lambda_r = \ln(2)/T_r = 0.693/T_r$$

This is the radiological removal rate for the radionuclide i. The equation accounts for removal by both biological and radiological decay processes.

The dose rate to the secondary organism from the body burden of the primary organism for N radionuclides is given by equation E-19.

$$R_c = \sum_{i=1}^{N} b_i\, U_c\, f_{1,i}\, E_{i,c}\, B_{i,c} \qquad \text{(E-19)}$$

where U_c = intake rate of primary organism by secondary organism c (kg/d)

M_c = mass of secondary organism c (kg)

and all other parameters are as previously defined.

Example Internal Dose to a Fish (Co-60 scenario)

For an example of a primary organism, assume a fish in the Columbia River is exposed over the duration of a year to water having a concentration of 4E-7 Ci/m^3 of cobalt-60. The fish has an effective radius of 10 cm. The internal dose is computed by substitution into equation E-15

where: $C_{i,c}$ = 4E-7 Ci/m^3
$BF_{i,c}$ = 0.33 m^3/kg
CF = not applicable as concentration is given in Ci/m^3

$E_{i,c}$ is computed by substitution into equation E-13:
$E_{i,c} = \epsilon_{i,c}$ MeV dis^{-1} x 5.12E04
[For a fish with a radius of 5 cm, $\epsilon_{i,c}$ = 0.437 Mev/dis]
$E_{i,c}$ = 0.437 MeV/dis x 5.12E04 = 22,374 kg-rad/Ci-d

Therefore: R_c = 22,374 kg-rad/Ci-d x 4E-07 Ci/m^3 x 0.33 m^3/kg
R_c = 0.003 rad/day or 1.1 rad/yr

For an example of a secondary organism, assume a heron that eats 0.6 kg of fish per day. The heron has a mass of 5 kg with an effective radius of 10 cm. The internal dose is computed by substitution into equation E-19.

R_c = 1.3E-07 Ci/kg x 0.6 kg/d x 37,478 kg-rad/Ci-d x 0.3 x 13.7 d / 5 kg
= 0.024 rad/d or 0.88 rad/yr

Where: b_i = 0.33 m_3/kg x 4E-07 Ci/m_3 = 1.3E-07 Ci/kg
U_c = 0.6 kg/d)
f_i = 0.3
M_c = 5 kg
$\lambda_{i,c}$ = $(\lambda_b + \lambda_r)$ = 0.073/day
λ_b = 0.073/day
λ_r = 3.6E-04/day

and: $E_{i,c}$ is computed by substitution into equation E-13
$E_{i,c} = \epsilon_{i,c}$ MeV dis^{-1} x 5.12E04
[For a heron with a radius of 10 cm, $\epsilon_{i,c}$ = 0.732 Mev/dis]
$E_{i,c}$ = 0.732 MeV/dis x 5.12E04 = 37,478 kg-rad/Ci-d

E-1.1.4.2 Aquatic External Dose Calculations. Typically, primary organisms reside in the water continuously. They can be exposed externally from immersion in contaminated water and exposure to river bottom sediments. The main primary organisms of concern at Hanford (i.e., fish and algae) can be assumed to get 100% of their exposure from immersion in the water. Secondary organisms can be exposed externally from immersion in the water, and/or exposure to river bottom or shoreline

sediments. Thus, the external exposure is weighted by the fraction of time exposed by these various pathways.

Immersion and sediment dose rate factors are used to compute external dose rates. Immersion and sediment dose rate factors were obtained from DOE (1988). These radionuclide specific factors indicate the external dose rate received (i.e., rad/year) from exposure to contaminated water (immersion) or sediment.

The equation for computing exposure from immersion in water is given by equation E-20 (from Soldat et al. 1974).

$$R_c = \sum_{i=1}^{N} C_{i,c}\ DF_{im,i}\ F_{exp}\ CF \qquad \text{(E-20)}$$

where: RI = dose rate (rad/day) from immersion in water
$C_{i,c}$ = concentration of radionuclide i in water to which organism c is exposed (Ci/L))
DF_{im} = immersion dose rate factor for radionuclide i (rad-m^3/Ci-d)
Fexp = exposure fraction (unitless)
CF = conversion factor (0.001, in units of L/m^3)

Half of the immersion dose can be assumed for surface swimming animals (Baker and Soldat 1992).

The equation for computing dose from exposure to sediments is given by equation E-21

$$RS = (F_{sed}\ F_{ruf}\ F_{exp}\ CF)\ SUM\ C_{i,c}\ DF_{sed,i}\ (1-e^{\lambda_r T_S})/\lambda_r \qquad \text{(E-21)}$$

where: RS = dose rate (rad/day) from exposure to sediment
F_{sed} = sediment deposition transfer factor (m/d)
F_{ruf} = geometry roughness factor (unitless)
CF = conversion factor [0.001 (L/m^3)]
DF_{sed} = sediment irradiation dose rate factor for radionuclide i (rad-m^2/Ci-d)
T_s = time sediment is exposed to contaminated water, (day)
λ_r = ln(2)/T_r is the radiological decay constant for the radionuclide (day^{-1})

The exposure fraction accounts for the amount of time immersed in water (for equation C-20) or exposed to sediments (for equation C-21). The sediment deposition transfer factor (F_{sed}) is obtained from Soldat et al. (1974). A value of 0.07 Ci-m^3/m^2-d is used for the Columbia River at Hanford. Table 2 (Baker and Soldat 1992) provides recommended Hanford-specific sediment dose factor parameters for various radionuclides. The time that sediment is exposed to contaminated water (Ts) = 365 days even though the sediment would have been exposed to contaminated water for more than that time frame. Thus, build up in the sediment for over a year is not included.

The geometry roughness factor (F_{ruf}) modifies the "infinite plane" dose rate factor given in DOE (1988). The effective size of the contaminated area is not an "infinite plane" because of the organism's height above the surface, and scattering and attenuation of photons by uneven surfaces. The value used for Hanford sediments is 0.2 (Baker and Soldat 1992). This value was derived by assuming the same value as the shore-width factor for humans standing on the shore of a river. The geometry roughness factor is a "fudge" factor that the user can apply based on the specific conditions (animal and location). The geometry-roughness factor adjusts the external dose rate factors given in

DOE (1988) to account for organism size and/or lack of an infinite plane. The factor was included so that the user could account for the height of the organism above the ground and the geometry. The external dose rate factors in DOE (1988) represent exposure conditions at 1 meter above ground from an infinite plane, where dose rates are reduced because of gamma scattering from surface roughness. Dose rates to small organisms in direct contact with the ground may be higher than indicated by the factors in DOE (1988). They are arbitrarily multiplied by 2 for animals that crawl on the surface. At a river shore the geometry from sediments is not an infinite plain (plane). Location specific adjustment factors can be found in Reg Guide 1.109. Because of the lack of much specific data, the geometry- roughness factors in Baker and Soldat (1992) are based on scientific judgement.

In addition, an infinite plane may not represent the contaminated area. Reg Guide 1.109 provides factors to adjust for this lack of representativeness for specific types of locations (e.g., river shores). For example, the dose rate factor at a river shore should be adjusted by a factor of 0.2.

Example External Exposure Calculation (C0-60 scenario)

For an example of external exposure, assume a fish in the Columbia River is exposed over the duration of a year to water having a concentration of 4E-07 Ci/m^3 of cobalt-60. This species spends about a third of its time feeding on the river bottom.

Immersion dose is computed by substituting values into equation E-20.

$$R_o = \sum_{1-1}^{N} C_{i,c}\ DF_{im,i}\ F_{exp}\ CF$$

where: RI = dose rate (rad/day) from immersion in water
$C_{i,c}$ = 4E-07 Ci/m^3
DF_{im} = 77 rad-m^3/Ci-d
F_{exp} = 1
CF is not applicable as concentration is in Ci/m^3

RI = 77 rad-m^3/Ci-d x 4E-07 Ci/m^3 = 3E-05 rad/d or 0.01 rad/yr.

External dose from river bottom sediments is computed by substituting values into equation E-21.

where: RS = dose rate (rad/day) from exposure to sediment
Fsed = 0.07 m/d
F_{ruf} = 0.2
F_{exp} = 0.5
$C_{i,c}$ = 4E-07 Ci/m^3
CF = not applicable as concentration is in Ci/m
DF_{sed} = 622 rad-m^2/Ci-d
T_s = 365 d
λ_r = 3.4E-4/d

$$T_s \times \lambda_r = 3.4E\text{-}4/d \times 365\ d = 0.12$$

$$RS = 0.07\ m/d \times 0.2 \times 0.33 \times 622\ rad\text{-}m^2/Ci\text{-}d \times 4E\text{-}07\ Ci/m^3 \times [(1-e^{-0.12})/3.4E\text{-}4/d]$$

RS = 3.82E-4 rad/d or 0.14 rad/yr

Combining the immersion and sediment dose rates yields 0.15 rad/yr.

E-1.2 NONRADIONUCLIDE DOSE FOR TERRESTRIAL RECEPTORS

E-1.2.1 Internal Dose for Organics or Metals for Terrestrial Receptors

The intake of contaminants in soil through consumption of contaminants in vegetation was estimated by modifying Equation 6 of EPA's Human Health Evaluation Manual (EPA 1989) in which the dose rate (mg/kg-day):

$$Is = (CS\ PS\ WW\ Qv\ FI\ EF\ ED) / BW\ AT \qquad \text{(E-22)}$$

where: Is = dose rate (mg/kg-day)
CS = concentration of contaminant in soil
PS = soil-to-plant conversion factor
WW = dry weight to wet weight conversion factor
CV = concentration in vegetation (mg/kg)
QV = ingestion rate (kg/day)
FI = fraction ingested from contaminated source
EF = exposure frequency (days/year)
ED = exposure duration (years)
BW = body weight (kg)
AT = averaging time, (ED)(365 days/year)

For the organisms (e.g., pocket mice) that spend their entire life in a waste site, averaging time, exposure duration, and exposure frequency can be eliminated from all intake equations. The fraction ingested from a contaminated source was based on the animal's home range and the amount of food expected to be consumed from contaminated areas. Feeding rates are typically reported on a wet-weight basis, while contaminant concentrations in soil and biota are reported on a dry-weight basis.

The concentration factors from soil to the generic plant was obtained from available literature. The maximum reported transfer coefficients from soil to plants were used in all dose calculations. These values were used to model plants as a food source in successive trophic levels.

Example of Internal Dose Calculation for Great Basin Pocket Mouse (lead scenario)

Given a site in which the soil is uniformly contaminated with 15.8 mg/kg lead, the internal dose rate calculation for the Great Basin Pocket Mouse is calculated using Equation E-22. For the pocket mouse, the assumptions are listed in footnote (n) of Table E-1.

where: Is = (CS PS WW QV FI EF ED) / BW AT
CS = 15.8 mg/kg
PS = 9.0E-03 (Baes et al. 1984 for vegetative plant parts)
WW = 0.32 (Table E-1)

QV = 0.0067 kg/d
FI = 1
EF = 365 d/yr
ED = 1 year
BW = 0.0235 kg
AT = 365 days

The concentration in the vegetation (QV) is based on the soil concentration.

QV = CS PS WW
= (15.8 mg/kg soil) (0.009) (0.32)
= 0.046 mg/kg vegetation

The dose rate (Is) using equation E-22 is equal to:

$$Is = \frac{(15.8\ mg/kg)(0.009)(0.32)(0.0067\ kg/d)(1)(365\ d/yr)(1\ yr)}{(0.0235\ kg)(365\ d)}$$

Is = 0.013 mg/kg-d

The estimated wildlife NOEL value for lead adjusted for the body weight of the pocket mouse is 1.97 mg/kg-d (Opresko et al. 1993), so the environmental hazard quotient (EHQ) ratio of the dose rate to the NOEL value is:

EHQ = (1.3E-02 mg/day-kg) / (1.97 mg/kg-d)
EHQ = 6.6E-03

EHQ values exceeding unity would indicate potential adverse effects to the mouse following exposure to the specific contaminant at the given concentration in soil.

E-1.2.2 Internal Dose for Organics or Metals for Aquatic Receptors

The internal dose calculations for organics and metals for aquatic receptors is based on the EPA water quality criteria and standards. See Chapter 4.0 for the discussion concerning the comparison of water concentrations to LOEL values to determine the EHQ for aquatic organism.

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Table E-1. Examples of General Parameters Used for Ecological Dose (Radionuclide) Equations.

Contaminant	Biological half-life[d] (days)	Physical half-life (days)	Mev (absorbed energy for 2-cm diameter sphere)	Soil-to-Plant Transfer Factor[d]	Fraction Uptake
Radionuclides					
Americium-241	20000[a]	1.58E+05[a]	5.28[a,c]	0.01[g]	0.001[m]
Cerium-144	563[a]	2.84E+02[a]	1.32[a,c]	0.0025[c]	0.0003[m]
Cesium-134	7.5[f]	7.53E+02[a]	0.259[a]	0.62[h]	1[m]
Cesium-137	7.5[f]	1.10E+04[a]	0.267[a,c]	0.62[h]	1[m]
Cobalt-60	9.5[a]	1.92E+03[a]	0.237[a]	0.5[g]	0.3[m]
Europium-154	635[a]	3.21E+03[a]	0.311[a]	0.001[g]	0.001[m]
Europium-155	635[a]	1.81E+03[b]	0.061[a]	0.001[g]	0.001[m]
Manganese-54	17[a]	3.13E+02	0.0514[a]	10.6[k]	0.1[m]
Plutonium-238	65000[a]	3.20E+04[a]	5.51[a]	0.07[g]	0.001[m]
Plutonium-239	65000[a]	8.78E+06[a]	5.15[a]	0.07[g]	0.001[m]
Plutonium-240	65000[a]	2.39E+06[a]	5.16[a]	0.07[g]	0.001[m]
Radium-226	8100[a]	5.84E+05[b]	11[a]	0.1[g]	0.2[m]
Ruthenium-106	7.3[a]	3.68E+02[a]	1.44[a,c]	0.00005[i]	0.05[m]
Strontium-90	244[o]	1.06E+04[a]	1.14[a,c]	19[j]	0.3[m]
Technetium-99	1[a]	7.77E+07[a]	0.84[a]	421[j]	0.8[m]
Uranium-234	100[a]	5.79E+07[a]	4.9[a]	1[i]	0.05[m]
Uranium-235	100[a]	2.57E+11[a]	4.6[a,c]	1[i]	0.05[m]
Uranium-238	100[a]	1.63E+12[a]	4.3[a]	1[i]	0.05[m]

a Baker and Soldat (1992)
b Shleien (1992)
c includes the decay products in the energy absorbed.
d Parameter are continually revised with new information and are subject to change.
f value for Cesium calculated as $Y = 3.5\ (\text{mass,kg})^{0.24}$ (Digregorio et al. 1978)
g Coughtrey et al. (1985)
h Miller et al. (1977)
i Whicker and Schultz (1982)
j Rouston and Cataldo (1978)
k Cataldo and Wildung (1978)
m ICRP (1959) for standard man
n assumptions used in ecological dose equations:
assumes mouse consumption of 6.7 grams/day vegetation by using $0.157\ (\text{mass,kg})^{0.84}$ (Calder 1984)
assumes mouse weight of 23.5 grams (Burt and Grossenheider (1976)
assumes dry-to-wet plant conversion of 0.32 (Femp-SWCR-6 Final, 1993)
o value for Strontium calculated as $Y = 3.5\ (\text{mass in grams})^{0.26}$ (Reichle et al. 1970)

Table E-2. Factors Used for Computing the Dose Rate for Radionuclides in Pocket Mice.

RC	Internal total-body dose rate	rad/day	References[b]
PS[a]	soil-to-plant transfer coefficient	radionuclide specific	Baes et al. 1984
CS	concentration of radionuclide in the Soil (Ci/kg)	--	--
WW[a]	conversion from plant dry weight to wet weight	0.32	FEMP-SWCR-6 FINAL
QV	ingestion rate (kg/day)	0.0067	Calder 1984
FI	fraction ingested from contaminated source (unitless)	1.0	--
EF	exposure frequency (day/yr)	365	--
ED	exposure duration (years)	1.0	--
DF	decay factors (day^{-1})	radionuclide specific	--
BW	body weight (kg)	0.0235	Burt and Grossenheider 1976
AT	averaging time (yr), (ED x 365 day/yr)	365	--
B_i	sum of the removal factor for nuclide (i)	radionuclide specific	Eq. 2
r	radius of mouse (cm)	2	--
FR	fraction retained (unitless)	radionuclide specific	Baker and Soldat 1992
T_b	biological half-life (day)	radionuclide specific	Baker and Soldat 1992, Digregorio et al. 1978
T_r	radiological half-life (day)	radionuclide specific	Shleien 1992, Baker and Soldat 1992
λ_b	biological removal constant $day^{-0.00}$	radionuclide specific	Eq. 3.1
λ_r	radiological removal constant (day^{-1})	radionuclide specific	Eq. 3.2
E_i	effective energy-absorbed constant for radionuclide (i), (kg rad Ci^{-1} d^{-1})	radionuclide specific	Eq. 4
RC	Internal total-body dose rate	rad/day	References
ϵ_i	radionuclide energy for diameter of mouse (MeV/dis)	radionuclide specific	Baker and Soldat 1992
λ	effective decay (day^{-1})	radionuclide specific	Eq. 3

a These values are examples and are subject to change. Site-specific information is used as better factors become available.
b The references are not inclusive, but given as examples.

Table E-3. Example Comparison of External and Internal Dose Rate To a Pocket Mouse for a Hypothetical Waste Site.

Contaminant[c]	Soil Conc. (pCi/g)	External Dose Rate[a,d] (rad/d)	Internal Dose Rate[b] (rad/d)	Percent External of Total Dose
Americium-241	0.72	1.0E-06	6.7E-08	94
Carbon-14	33	0	9.7E-05	0
Cesium-134	5.5	4.3E-04	4.4E-05	91
Cobalt-60	2000	0.25	4.5E-03	98
Europium-152	17000	1	4.4E-05	100
Europium-154	5700	0.36	2.4E-06	100
Europium-155	660	2.0E-03	5.3E-08	100
Nickel-63	18000	0	2.8E-04	0
Plutonium-238	7	5.7E-07	4.6E-06	11
Plutonium-239	200	2.2E-05	1.2E-04	16
Radium-226	0.65	2.2E-07	2.4E-04	0.1
Strontium-90	240	0	1.6	0
Technetium-99	0.26	7.1E-12	6.7E-04	0
Thorium-228	0.81	1.3E-07	1.3E-10	100
Thorium-232	0.41	3.5E-08	5.6E-11	100
Tritium	150	0	3.5E-05	0
Zirconium-95	0.56	1.2E-05	1.8E-11	100.00
	TOTALS	1.6 rad/d	1.6 rad/d	50 % of total

(a) Calculated using Equation E-6 assuming 24 hr exposure.
(b) Calculated using Equation E-1 using assumptions listed in Table E-1.
(c) Note that this "hypothetical" waste site is not typical of most Hanford waste soils since the europium soil concentration is unusually high. This example illustrates a situation where the external dose rate exceeds the 1 rad/d limit.
(d) Values for some radionuclides ignore bremsstrahlung radiation, however the effects to the dose are insignificant.

Table E-4. Example Comparison of Soil Concentrations Needed To Exceed the Wildlife EHQ of 1 rad/day for the Great Basin Pocket Mouse for Either External or Internal Dose Rates for Various Radionuclides.

Soil Contaminant	Soil Conc. Needed to Exceed EHQ of 1 rad/day from External Dose (pCi/g soil)[a,c]	Soil Conc. Needed to Exceed EHQ of 1 rad/day from Internal Dose (pCi/g soil)[b,c]
Americium-241	70,000	11,000,000
Carbon-14	no dose	350,000
Cesium-134	13,000	130,000
Cobalt-60	8,000	450,000
Europium-152	17,000	400,000,000
Europium-154	16,000	23,000,000,000
Europium-155	33,000	12,000,000,000
Nickel-63	no dose	6,500,000
Plutonium-238	13,000,000	1,600,000
Plutonium-239	9,000,000	1,700,000
Radium-226	no dose	2,700
Strontium-90	no dose	148
Technetium-99	no dose	400
Thorium-228	6,500,000	no dose
Thorium-232	12,000,000	no dose
Tritium	no dose	4,300,000

[a]Calculated using external dose equation (Eq. E-6).
[b]Calculated using internal dose equation (Eq E-1), and assumptions listed in Table E-1).
[c]Exposure assumptions are that the 23.5 g mouse is underground for 24 hours and consumes 6.7 grams stored food during that period

Table E-5. MeV/dis of K-40 for Effective Radius of the Organism.

Effective Radius, cm	ϵ_a MeV/dis
1.4	0.529
2	0.532
3	0.536
5	0.544
7	0.551
10	0.561
20	0.590
30	0.612

NOTE: These values of ϵ vs the radii are from Baker and Soldat (1992).

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