



4.2.1 Endpoints for Ecological Species of Interest

Following EPA usage, we use the term "endpoint" to indicate the biological resources and their attributes that are of concern for this assessment. "Assessment endpoints" indicates the attributes of interest for the species. "Measurement endpoint values" or "measurement endpoints" indicate the toxicological response used to represent the assessment endpoint. For all chemicals, the measurement endpoints are the levels known to be lethal to 50 percent of an exposed population (expressed as LD₅₀ or LC₅₀) and the lowest levels known to produce a toxic response in any member of a population (expressed as LOEL). We were not always able to find measurement endpoints for each species. In those cases, we used the measurement endpoint value of a similar species.

Assessment endpoints indicate the biological resources and attributes that are to be protected and maintained within the ecosystems potentially at risk (EPA 1992a). Consequently, these endpoints are defined by CRCIA Team concerns for the study area. The species evaluated as assessment endpoints under this risk assessment are described in Section 4.1. The CRCIA Team specified that the resource attributes to be protected center on the long-term survival and health of the populations of these species within the study area and throughout the Columbia River system. Consequently, the measurement endpoints (the measurable ecological characteristics related to the ecological values to be

protected, EPA 1992a) selected for this assessment include the concentrations of contaminants that are known to be lethal to 50 percent of an exposed population (LD₅₀ or LC₅₀), and the lowest concentrations that are known to produce clinically toxic responses in any member of a population (the lowest observed effects level or LOEL). These acute and chronic measurement endpoints are alternative measures of effect (not risk). These benchmarks serve as reference values in the denominator of the EHQ ratio (environmental concentration/benchmark concentration), further detailed in Section 4.2.3. The potential hazard (or impact) is characterized by the EHQ, typically using an EHQ threshold of 1.0 as a decision screen.

The magnitude of risk posed to Tier II species was assessed by comparing chemical and radiological toxicity endpoints (chemical concentrations or doses of ionizing radiation that produce harmful effects on test organisms in a toxicity test) with estimated exposures (quantification of an organism's contact with contaminants or contaminated media expressed as a chemical concentration or dose of ionizing radiation). EPA refers to this method as the quotient method (EPA 1996a).

This section presents the methodology used in selecting chemical and radiological toxicity measurement endpoints for each of the contaminants of interest for the Tier II species.

4.2.1.1 Toxicological Database and Literature Review

Chemical toxicities of the contaminants of interest to Tier II species were obtained by searching five toxicological databases as well as primary and secondary literature sources. Information in three toxicological databases (AQUIRE - Aquatic Information Retrieval Database, TERRATOX - Terrestrial Toxicological Effects Database, and PHYTOTOX - Phytotoxicological Effects Database), was accessed simultaneously through a central database retrieval system (ECOTOX - Ecotoxicology Database System,



EPA 1996b). The other two toxicological databases searched were the HSDB (Hazardous Substances Databank, NLM 1996) and OHM-TADS (Oil and Hazardous Materials Technical Assistance Data System Databank, EPA 1996c).

Effects of radionuclides on Tier II species were obtained from primary and secondary literature sources. Radiological effects are a function of the energy deposited in the receiving biological tissues and the relative biological effectiveness of the radiation. The National Research Council has determined that the relative biological effectiveness of beta and gamma radiations may be assumed to be the same (NRC 1990). All the radionuclides in this risk assessment are beta and gamma emitters, except for uranium, which emits alpha particles. Alpha particles have a higher ionizing potential than beta or gamma particles per unit distance traveled. However, the energy of alpha particles is rapidly absorbed by any moderately dense material such as water.

Chemical and radiological endpoint data were seldom available for the Tier II species. Consequently, endpoint data were collected initially for all species for which effects data were available for the contaminants of interest. From these species, one or more species were selected as surrogates for the Tier II species based on similarities of taxonomic group, life style, and/or toxicological response. Endpoint values for surrogate species were then extrapolated to Tier II species.

4.2.1.2 Selection of Endpoints for Tier II Species

The endpoint values collected from the literature and databases were generally derived from chemical toxicity tests on 1) aquatic animals and plants for which uptake was from water, 2) terrestrial animals for which uptake was by ingestion, and 3) terrestrial and emergent plants and fungi for which the chemical was introduced either in soil or in a hydroponic growth chamber. The "Measurement Endpoint Values Used in the Risk Assessment" section of Appendix I-D identifies the benchmark species and toxicological responses used in determining toxicological values for the species of interest.

4.2.1.2.1 Terrestrial and Emergent Plants. No chemical LD₅₀ or LC₅₀ values were available for terrestrial or emergent plants in the databases or literature that was reviewed. One radiological LD₅₀ value was available for terrestrial plants. No radiological LD₅₀ values were available for emergent plants.

Phytotoxic effects are seldom reported in terms of lethality. Of the many types of sublethal phytotoxic effects reported, reduction in growth and yield parameters (for example, production and growth of shoots and roots, and production and viability of seeds) were assumed to be the most ecologically relevant in both the plant populations themselves and their use as forage. An environmental or tissue concentration that produced a 20 percent reduction in a growth or yield parameter was selected as the LOEL value for that plant/contaminant combination. The use of 20 percent is consistent with other screening benchmarks for ecological risk assessment and with current regulatory practice. Most regulatory phytotoxicity criteria are based on concentrations that produce effects significantly different from controls. On average, those concentrations correspond to greater than 20 percent effects (Will and Suter 1995).



Consequently, the phytotoxicity databases were screened to obtain toxicity values for the contaminants of interest that corresponded to approximately 20 percent reduction in growth and/or yield (or a parameter that directly contributes to growth and yield, such as photosynthesis). Where available, both tissue and soil LOEL concentrations were collected for terrestrial plants, and both tissue and water concentrations were collected for emergent plants.

Of the databases and literature sources reviewed, Will and Suter (1995) provided the most comprehensive and conservative summary of chemical toxicities for terrestrial plants. Will and Suter's screening-level LOEL benchmarks are based on numerous toxicity tests performed on a relatively wide variety of terrestrial plant species. Will and Suter provide three categories of confidence in their LOEL benchmarks: low confidence benchmarks were derived from 10 or fewer literature values, moderate confidence benchmarks were derived from 11 to 20 literature values, and high confidence benchmarks were derived from more than 20 literature values. Moderate-to-high confidence benchmarks were derived by selecting the tenth percentile of the reported values. Low confidence benchmarks were derived by selecting the lowest reported value.

In this risk assessment, toxicity LOEL values were selected on the basis of similarity of the test species to the Tier II species, on the reliability of the reported toxicity value, and the similarity of the toxic response to the selected LOEL responses. The order of preference for identifying suitable species was as follows:

- ◆ Tier II species
- ◆ closely-related species in riparian or riverine habitats
- ◆ all other species

Toxicity values were selected in the following order of preference:

- ◆ Will and Suter's (1995) high or moderate confidence LOEL benchmarks, whichever was reported for the contaminant of interest in question
- ◆ Will and Suter's (1995) low confidence LOEL benchmarks or other values that were based on a greater than 20 percent reduction in growth or yield, whichever was lowest
- ◆ the lowest of the values based on unspecified levels of adverse effects in growth or yield
- ◆ the lowest of the reported values based on unspecified toxic effects

For example, if only a low-reliability toxicity value was available for a Tier II species and a higher-reliability value was available for another related species, the low-reliability value for the Tier II species was selected. Where toxicity values were available only for distantly related species, as was commonly the case, values were selected based solely on the four reliability criteria, irrespective of the species used in the toxicity test.



4.2.1.2.2 Fungi. Several chemical LC₅₀ values were available for fungi. Because fungi were treated as a broad taxonomic group in this risk assessment (no individual Tier II fungal species are identified), there was no selection of surrogate species per se. Instead, the fungal LC₅₀ values used in this risk assessment were those associated with the lowest reported values for the taxonomic group. One radiological LD₅₀ and two radiological LD₆₃ (dose of ionizing radiation that produces mortality in 63 percent of the test organisms in an acute exposure test) values were available. The lowest of these values was selected.

Available chemical toxicity values were screened to select those reporting a reduction in growth or yield of approximately 20 percent. These values were classified as LOELs (after Will and Suter 1995), and the lowest of these were selected for each contaminant of interest. Several radiological LOEL values based on lethality were available. The lowest of these were selected where no growth or yield effects were reported.

4.2.1.2.3 Terrestrial Animals. The databases and literature sources reviewed yielded relatively few data on the toxicity of most chemicals to wildlife. Most of the data obtained from these sources apply to laboratory animals such as rats, house mice, guinea pigs, mallards, and quail. As a result, many toxicological values for the species of interest were derived by extrapolating across taxonomic groups. In the few cases where a toxicological value was available for a species of interest, it was used without any extrapolation. Details on the toxicological responses are contained in Sample et al. (1996).

Toxicological values were extrapolated from a benchmark species to a species of interest separately for each contaminant. A hierarchical decision making process was used to determine which benchmark toxicological values would be used for a given species of interest. Values in decreasing order of importance by species, genus, family, order, class, and superclass were sought. Where values were available for several different species, the species closest in life style to the species of interest was used. The lowest LD₅₀ value found was used to extrapolate to an LD₅₀ for the species of interest by using an adjustment factor for the differences in body size. LD₅₀ values for mammalian species of interest were calculated using the following equation (based on Sample et al. 1996):

$$LD_{50soc} = LD_{50b} (bw_b/bw_{soc})^{1/4} \quad (4.1)$$

where LD_{50soc} = LD₅₀ value for the species of interest
 LD_{50b} = LD₅₀ value for the benchmark species
 bw_b = body weight of the benchmark species
 bw_{soc} = body weight of the species of interest

For avian and herpetilian species, a 1:1 conversion was used (Sample et al. 1996).

As with LD₅₀ values, where a LOEL value was available for a species of interest, it was used without any extrapolation. In most instances, however, LOEL values were derived from the calculated LD₅₀ value



for a given species of interest. A range of $LOEL_{Hi}$ and $LOEL_{Lo}$ values was calculated by taking 1/15th and 1/30th of the LD_{50} values, respectively, which has been found to be in good agreement with the data (Urban and Cook 1986; Tucker and Lietzke 1979).

4.2.1.2.4 Aquatic Species. One endpoint selected for evaluating potential aquatic effects was LC_{50} . The length of exposure for the reported toxicity tests varied, usually ranging from 24 to 96 hours for an acute exposure. The longest exposure time available for LC_{50} was selected because exposures within the study area are assumed to be chronic. Endpoints for aquatic plants and phytoplankton were usually EC_{50s} (an effective concentration resulting in a 50 percent reduction in cell growth rates) determined for a 5- to 14-day exposure period.

Besides differences in exposure duration, the toxicity studies also used test organisms of differing sizes and life stages. To limit the effects of this variability on the chosen measurement endpoints for the risk assessment, data were grouped only from tests conducted from aquatic organisms of similar size and age. The form of the metal used in the test solutions also varied among reported tests (for example, using $NiSO_4$ versus $NiCl_2$); and in those cases, the lowest toxicity value for the exposure series was used.

The process used to select measurement endpoint values for each species and contaminant from the data review was as follows: where two or fewer toxicological studies for a given endpoint and species of interest were available, the lower (most conservative) reported value was selected. Where more than two references for the same endpoint were available, the mean reported value was used.

The other endpoint review in the toxicological literature was $LOEL$. However, because of the wide variation in reported data, a standard endpoint for these values was not established. For example, $LOELs$ included such endpoints as lethality, immobilization, reduced growth, and biochemical alterations. Where such variation occurred within a single species for a chemical, the lowest value consistent with a toxic response was used. Where $LOELs$ were unavailable, they were estimated using 1/15th the LC_{50} (Suter 1993; Urban and Cook 1986; Tucker and Lietzke 1979). Note that this process uses only the lowest concentration, equivalent to the $LOEL_{Lo}$ used for terrestrial species.

Toxicity data were not available for all species. Therefore, toxicological data from taxonomically related species were used as needed. However, the closeness of the taxonomic relationship varied according to contaminant and species. For example, toxicological data for some contaminants in primary producers were available only for green algae. These data were then used for phytoplankton, periphyton, and water milfoil. While the relationship between unicellular algae and phytoplankton is relatively close, the relationship to a vascular plant is more remote. Frequently, only general terms such as crustacea and mollusca were included in the toxicological references in the toxicological databases that were searched. In the absence of species-specific data, values for crustacea were used for crayfish, *Daphnia magna*, and *Hyallela*, and values for mollusca were used for clams, Columbia pebblesnail, and mussels. Similarly, values for *Daphnia magna* were used for other crustacea when additional data were not available. Uncertainties in the parameters and their use in the model are discussed in Section 4.2.4.



More species-specific data were available for fish than for any other taxonomic group. Our criteria for surrogate species selection were based first on similarities in feeding habits, then on taxonomic closeness. For example, bluegill might be taxonomically closer to smallmouth bass than squawfish, but squawfish have a similar predatory life style and feed on the same food organisms in the Hanford Reach. Thus, toxicological data from squawfish were used for smallmouth bass when data for the latter were unavailable. In the absence of a taxonomic or life style match for a Tier II fish, toxicological data from the most commonly tested species such as rainbow trout, fathead minnow, and goldfish were substituted.

4.2.2 Estimating Biotic Exposure to Contaminants of Interest

Exposures of individuals and populations to contaminants had to be estimated for each Tier II species and their primary prey within the study area. The amount of each contaminant that an individual might encounter via all exposure pathways had to be quantified. A preliminary, qualitative estimate of exposure was performed earlier to assist in identifying Tier II species (see Section 4.1). That estimate was based on evaluating the degree to which the average life styles of individuals of a particular species brought them into contact with potentially contaminated media (air, food, groundwater, pore water within sediment, sediment, and surface water).

To estimate the contaminant levels in each species, we had to take several factors into consideration:

- ◆ the pathways by which each species is exposed
- ◆ the degree of contact the species had with potentially contaminated media (air, food, groundwater, pore water within sediment, sediment, and surface water)
- ◆ the uptake rate by each species of each contaminant
- ◆ the availability and applicability of computer models that address the diverse types of contaminants studied

The contaminants addressed by this assessment include organic chemicals, metals (both radioactive and non-radioactive), tritium (hydrogen-3), and non-metallic ions. These diverse contaminants vary widely in their chemical and physical properties, which makes modeling exposures problematic. A review of available exposure models found none capable of addressing the full range of contaminants in both aquatic and terrestrial systems simultaneously. However, a set of aquatic exposure models developed by Thomann et al. (1992, 1995) presented sufficient generality that they could be adapted and used in the aquatic system. Similarly, the EPA modeling system for human health exposures contained in the following also could be adapted to address the terrestrial system: *Risk Assessment Guidance for Superfund* (EPA 1991), *Dermal Exposure Assessment: Principles and Applications* manual (EPA 1992b), and the terrestrial exposure methodology contained in the *Wildlife Exposure Factors Handbook* (EPA 1993). These submodels could be combined to incorporate biomass transfers from the aquatic to the terrestrial component to allow foraging by air-breathing species on aquatic components of the ecosystem.

The CRCIA Team chose to adapt the Thomann/EPA systems into a single spreadsheet-based model, the components of which are described below. A detailed description of the model and its parameters is provided in the "Exposure Model Description and Parameters" section of Appendix I-D. Dynamic versus steady-state models and their relevance to the findings of this assessment are discussed in Section 4.2.10.3.



Aquatic animals may be exposed to environmental contaminants via two pathways: dietary (ingestion) and dermal/gill exposures (Figure 4.4). For many species and contaminants, dietary exposure constitutes the primary exposure to pathway (Thomann et al. 1995; Thomann 1989). Exposure of primary producers occurs entirely by dermal (or surficial) exposure. As shown in Figure 4.4, uptake of a contaminant into the body of an organism is a function of both the amount of contaminant present in the media (food, sediment, or water) contacted by the organism and the rate or fraction of the contaminant crossing the organism's integument (for example, gill membrane) or gut membrane. A standard parameter used to describe this fractional uptake from water is the species- and chemical-specific bioconcentration factor, which is the ratio of the equilibrium body concentration of a chemical in a species versus the water concentration when uptake is limited to the water phase only (meaning no contaminated food is eaten by the species in question). Because an organism can acquire a body burden by ingesting a contaminant as well as from respiring the contaminant, the most appropriate bioconcentration factor values from the literature are those where estimates have been obtained from controlled laboratory studies rather than from field-derived observations because the latter reflect exposures to both water and contaminant-bearing food.

Exposure of terrestrial or air-respiring species may occur through ingestion of contaminated food or water, dermal exposure to contaminated soil or water, and/or inhalation of airborne contaminants (EPA 1993). These various routes of exposure and the key information required to estimate these exposures are shown in Figure 4.5.

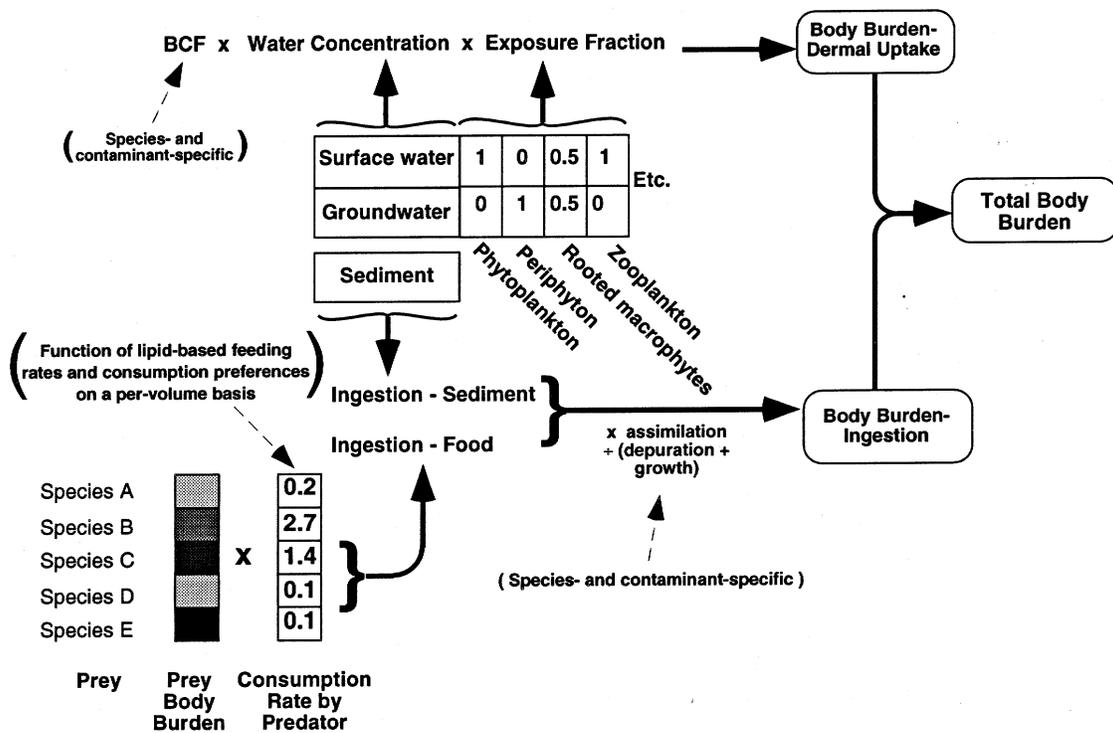


Figure 4.4. Conceptual Model of Exposure for Water-Respiring Species

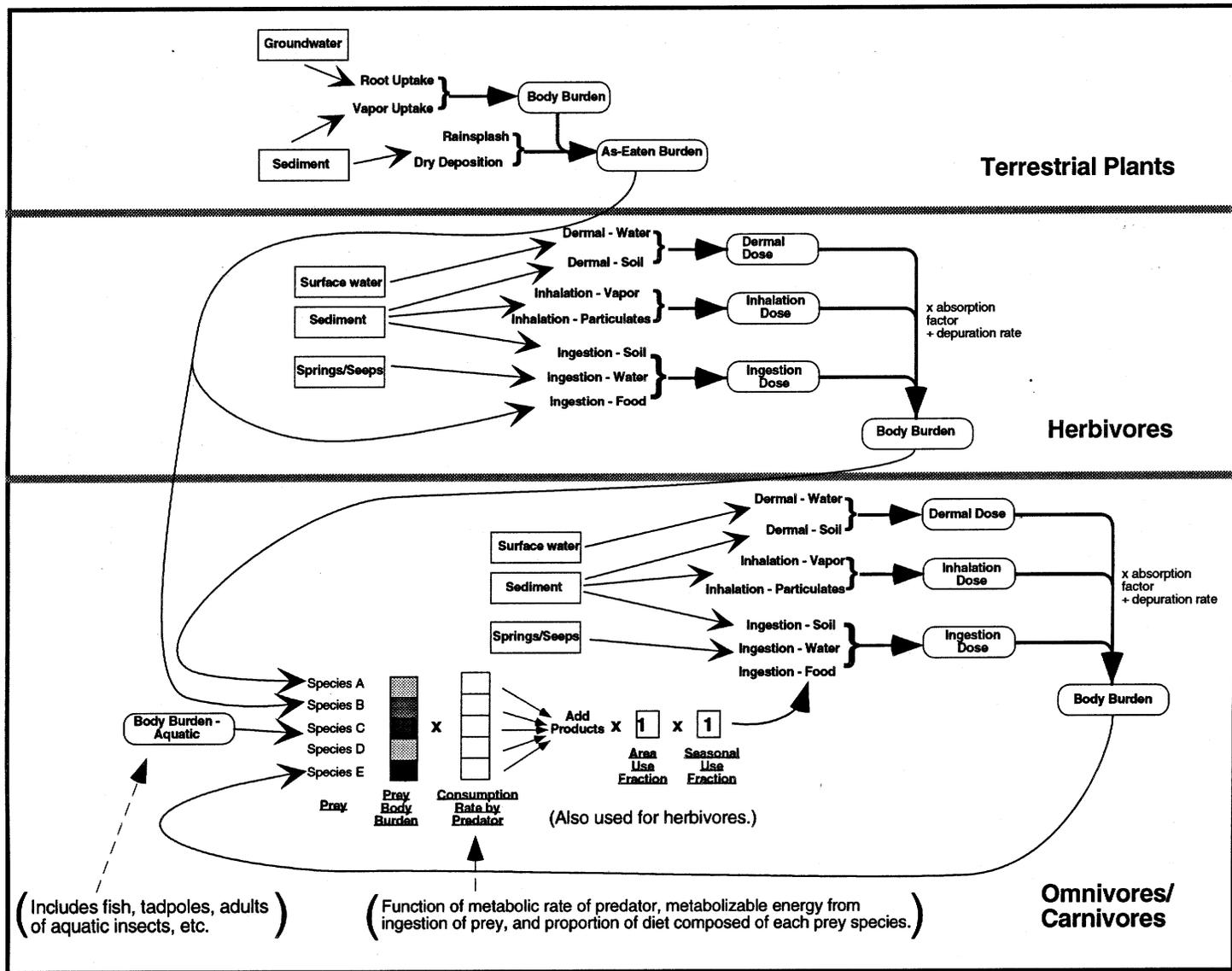


Figure 4.5. Conceptual Model of Exposure for Air-Respiring Species



The hierarchical nature of exposure is depicted in Figure 4.5 by three levels of foraging life styles. The body burden of plants is from uptake of contaminants from air, soil, pore water, and groundwater. Uptake may be either through the roots or through transport across aboveground membranes containing aerial deposits of vapor-phase contaminants. Herbivores and omnivores consume this plant material along with the contaminants that have been deposited on the plant tissues as particulate matter. They may also ingest soil directly, and all are assumed to consume some amount of water, which may itself contain contaminants. Omnivores and carnivores also consume animal prey that have integrated the various contaminants encountered throughout the prey's lifetime. Besides the level of contamination present in the various pathways of exposure, the fractional absorption of these contaminants controls both the resulting concentration in the organism and its toxicological response to those absorbed concentrations.

In this risk assessment, the contaminant body burdens of the prey are assumed to be at equilibrium with the environment. The contaminant concentrations in the prey are those most likely to be encountered under long-term, steady-state exposure and depuration/loss conditions. Also, the amount of prey consumed by a predator is assumed to be a function of its field metabolic rate, the amount of energy contained in each prey type consumed, and the proportion of each prey type consumed (EPA 1993).

The model and its parameters are described in detail in the “Exposure Model Description and Parameters” section of Appendix I-D. Dynamic versus steady-state models and their relevance to the findings of this assessment are discussed in Section 4.2.10.3.

4.2.3 Approach for CRCIA Ecological Risk Assessment

The approach used in this screening assessment of ecological risk was discussed with and approved by the CRCIA Team. The approach was selected to make the best use of available data given the time and funding constraints of the study, to use assumptions that the team determined were reasonable, to follow an internal screening approach that would further reduce the amount of area subject to detailed analyses, and to incorporate Technical Peer Review comments that the team deemed important.

The following key set of assumptions was used in this analysis. For estimating the level of hazard posed by contaminants, the ratio of estimated exposure to measurement endpoints (often termed an environmental hazard quotient or

In this section we describe the key techniques and assumptions we used to estimate ecological risk:

- ◆ The level of hazard is indicated by the ratio of the level of estimated exposure to a contaminant for a species to the measurement endpoint. That ratio is the environmental hazard quotient (EHQ). Any EHQ over 1 indicates a potential hazard.
- ◆ To do the exposure calculations we used computer models and put three types of data into the models: 1) parameters describing the attributes of the species (see Section 3.0 and the parameters described in Section 4.2.2), 2) contaminants and their behavior in biological systems (data from literature referenced in text), and 3) locations (segments along the Columbia River). To ensure the models provided reasonable estimates of exposure, we compared the estimated movement of contaminants using data with known results.
- ◆ We first analyzed the data (see Section 3.0) using the deterministic method. If the EHQ for any species-contaminant-location combination was more than 1, we then further analyzed that combination using the stochastic method.



EHQ) was used. EHQs greater than 1 represent an exposure that meets the level determined to pose a hazard (LOEL or LD/LC₅₀). EHQs were estimated separately for each species but were combined for similar acting contaminants (for example, radionuclides).

Each parameter in the exposure model needed a defined value (Jorgensen 1990). For these values, Hanford Site data were used as much as possible. Preliminary model validation and calibration were performed first using data sets from other locations. Data sets from Hanford Site studies that were not used as parameters in the model were used to evaluate model output but were not used to adjust the model. This constraint was chosen because the CRCIA Team agreed that the conditions defined for the exposure model may not be well represented by biological samples obtained under the various Hanford Site monitoring programs, which have focused on protecting human rather than ecological health.

The media data were first used in a deterministic exposure analysis to identify location-species-contaminant sets that indicate a potential hazard. This analysis used observed maxima for media concentrations and mean or geometric parameters for the exposure models. Measurement endpoints used for the EHQs were LOELs since this is more conservative than the acute exposure endpoint (LD/LC₅₀). Species-contaminant-location combinations that met or exceeded an EHQ of 1 were further evaluated using stochastic exposure modeling. The others were dropped from further consideration.

Stochastic modeling used media geometric means and standard deviations and known or estimated uncertainties in the exposure model parameters. Measurement endpoints used were both LOELs and LD/LC₅₀. Finally, results of exposure modeling were compared with measured tissue concentrations obtained from biota samples within the study area to provide a basis for comparing the risk assessment results.

4.2.4 Setting Input Values and Uncertainties for Exposure Modeling

Model parameters included chemical attributes (for example, K_{ow} , molecular diffusivity), species attributes (for example, body weights, fraction of organic carbon), site attributes (for example, average air temperature, average wind speed), species-by-chemical attributes (for example, bioconcentration parameters, depuration rates), and chemical-by-site attributes (for example, chemical concentration in sediment). Many of these parameter values were defined using literature references because site-specific measurements were unavailable. All chemical-by-site attributes, however, were defined using values measured within the study area (see Section 2.0). Chemical-specific parameter values were obtained from standard references with little ambiguity in the values. Parameterization of species and chemical-by-species parameters, however, incorporated more ambiguity. Much of the ambiguity resulted from lack of information for the specific species evaluated in this risk assessment, which required values to be selected based on degree of relevance to the species and/or chemicals being evaluated.

In this section, we give the rules we followed when we had to use substitute values, and we define the uncertainty of the values used in the stochastic analyses. We use the term "uncertainty" to mean the likelihood of a certain amount of variability in model parameters or dose estimates.