The first level of relevance was defined primarily on the bases of major life style and taxonomic similarities (Table 4.19). Under this scheme, when data were unavailable for a species of interest, values for closely related species at the immediately more general level of similarity were used instead. For example, if bioconcentration factor data for suckers were unavailable, data from the Cyprinoideae family were used (see Table 4.19). If no data were available for that family, then the average value for bony fishes was used instead. This rule was not required for parameterization of species-specific parameters, but was used for several chemical-by-species parameters.

The organisms dominating the phytoplankton and periphyton groups in the study area are diatoms (Neitzel et al. 1982). Therefore, diatom data were used for these groups if such data were available.

Data were generally rounded to one or two significant digits, depending on average data quality among taxonomic groups.

In parameterizing the uncertainties for the stochastic simulations of exposure, we generally followed the recommendations of MacIntosh et al. (1994) with regard to known versus unknown properties. Table 4.20 shows the general distributions and their parameters used for each model parameter. Stochastic parameters are those that are expected to vary in the populations being modeled. For example, body weights of bald eagles within the study area vary among individuals, and lipid content of salmon varies among individuals. Uncertain parameters are those that may or may not vary in the environment, but for which the necessary information to fully characterize them is lacking. For example, contaminant uptake by plants from soil varies among species that have been studied, but data are unavailable on this parameter for the species evaluated in this risk assessment. Additional research on these species could reduce the uncertainty for this parameter. Other uncertain parameters were assumed not to vary significantly with respect to the measurement endpoints (for example, $K_{ow}$).

The following rules were used to establish distributions for the variable parameters in the exposure model. Where data were available to provide estimates of geometric means and standard deviations, lognormal distributions were used. For certain components, uniform or triangular distributions were used with upper and lower limits set using a consistent fraction of the mean of the available data. These limits were a factor of 2 about the mean or geomean for data-rich parameters, a factor of 5 for data-poor parameters, and a factor of 10 for parameters extrapolated from widely different taxonomic groups or for which higher variability was expected. These values reflect the increasing (but unknown) uncertainty involved in extrapolating between less-related taxonomic groups. These uncertainties are shown in Table 4.21.

The uniform and triangular distributions are commonly used in stochastic modeling when the shape of the actual distribution is unknown (Kirchner 1994). When extremes and a modal or mean value of a distribution are known, the triangular distribution is the least biased assumed distribution. When only the extremes can be estimated, the uniform distribution is the least biased assumption. Kirchner (1994) notes that in many cases the type of distribution chosen in complex models such as this one has little effect on the form of the output distribution.

**Table 4.19**. Hierarchy of Substitution for Species-by-Chemical Parameterization

- Aquatic Vegetation
  - Phytoplankton
  - Periphyton
    - Diatoms
  - Macrophytes
    - Water milfoil

- Benthic Invertebrates
  - Crustaceans
    - Crayfish
  - Insect larvae
    - Mayfly (as larvae)
  - Molluscs
    - Snails
    - Bivalves
      - Clams
      - Mussels

- Zooplankton
  - Crustaceans (planktonic forms)
    - Branchiopods
      - Anostracans (e.g., fairy shrimp)
        - *Hyallela*
      - Cladocerans
        - *Daphnia*

- Aquatic Vertebrates
  - Amphibians
    - Anurans (frogs and toads)
      - Woodhouse’s toad tadpoles
  - Fishes
    - Agnathans (jawless fishes)
      - Lamprey
    - Bony fishes
      - Sturgeon
      - Salmoniformes
        - Whitefish
        - Salmon
        - Trout
          - Rainbow trout
      - Cyprinoideae
        - Carp
        - Suckers
      - Catfish
      - Percoids
        - Bass

- Fungi

- Terrestrial Vegetation
  - Trees
  - Grasses
  - Forbs
  - Ferns

- Terrestrial Animals
  - Nonhomeotherms
    - Terrestrial arthropods
    - Reptiles
      - Snakes
      - Lizards
    - Toads (note tadpoles are evaluated in aquatic portion of the model)
  - Homeotherms
    - Mammals
      - Rodentia
        - Beaver
        - Muskrat
        - Western harvest mouse
      - Mustelidae (weasels, mink, otter)
        - Weasels
      - Procionidae (raccoons, skunks)
        - Raccoon
      - Artiodactyla (cloven-hoofed mammals)
        - Deer
          - Mule deer
    - Birds
      - Pelicaniformes (pelican order)
        - American white pelican
      - Ciconiiformes (heron order)
        - Great blue heron
      - Anseriformes (swan, duck, and goose order)
        - Geese
          - Canada goose
        - Ducks
          - Bufflehead
          - Mallard
      - Gruiformes (rail, gallinule, and coot order)
        - American coot
      - Charadriiformes (gull, tern, and snipe order)
        - Forster’s tern
        - Common snipe
      - Falconiformes (hawk, eagle, and falcon order)
        - Accipitridae (kites, hawks, and eagles)
          - Bald eagle
          - Northern harrier
        - Falconidae (falcons)
          - American kestrel
      - Galliformes (chicken, pheasant, and quail order)
        - California quail
      - Passeriformes
        - Cliff swallow

**Table 4.20.** Uncertainty Classification of Exposure Model Parameters and Distributions Used in the Stochastic Simulations

| Model Parameter Varies with Respect to Prediction Endpoint? | Distribution Characterized | Distribution Type Used in Simulation |
|---|---|---|
| Yes | | |
| Stochastic component (e.g., body weight) | Population variability and mean | Normal, lognormal (uniform when data were scarce) |
| Uncertain component (e.g., plant uptake rates) | Component parameters (mean, maximum, minimum) | Uniform, triangular |
| No | | |
| Uncertain only (e.g., $K_{ow}$) | Likeliest value for deterministic model<br>Component parameters (mean, maximum, minimum) for stochastic simulations | Uniform, triangular |

### 4.2.5 Exposure Scenarios

**We estimated risk for each species as if it were found in each segment even though the habitat of some segments may be unfit for certain species. For example, we have 27 estimates for the risk to spawning salmon, 1 estimate for each of the 27 segments. Then within each segment we estimated risk to spawning salmon from each type of contaminant: carcinogenic chemicals, radionuclides, and toxic chemicals.**

Exposure conditions were selected to be conservative. All animals were assumed to spend their entire time at the Hanford Site within a single river study segment. Their entire exposure to Hanford Site contaminants was assumed to be from a single segment. No foraging outside of the Columbia River and its associated riparian zone was included in the model. All Tier II species were assumed potentially to occur throughout the study area, although some segments do not contain appropriate habitat (for example, gravel beds for spawning salmon). Note that this residency assumption is more conservative than that used in the species screening described in Section 4.1.

Aquatic organisms were assumed to spend some fraction of their lives in contact with sediment and/or pore water (see Figure 3.1 for usage and see Appendix I-D, diskette file "paramtrs.xls"). Pore water was assumed to be represented by seep/spring data where such data were available; otherwise groundwater values were used. Pore water was further assumed to extend at least 15 centimeters (5.9 inches) up from the river bed into the water column, in the absence of data on the actual mixing zone.

Terrestrial animals were assumed to consume water from within the study segment in amounts consistent with their taxonomic group and metabolic demands. The water they consumed was assumed to be seep/spring water, where such was available, or surface water where no seeps or springs occurred.

**Table 4.21.** Parameterization of Uncertainties Used in the Stochastic Simulations

| **Parameter**[a] | **Uncertainty Representation (distribution type: distribution parameters)** |
|---|---|
| BW (species wet body weight) | Homeotherms: (normal: mean = observed mean, std. dev. = mean/10)<br>Nonhomeotherms: (uniform: limits = mean ± mean/5) |
| $f_L$ (species lipid fraction) | (uniform: limits = mean ± mean/5) |
| awd (species average weight dry) | (uniform: limits = mean ± 50%) |
| $f_{oc}$ (soil fraction organic carbon) | (uniform: limits = mean ± mean/10) |
| GE (gross energy from prey) | (uniform: limits = mean ± mean/10) |
| AE (prey assimilation efficiency) | (uniform: limits = mean ± mean/10) |
| BCF (bioconcentration factor) | Observed range <50: (triangular: observed upper and lower range, with observed mean as most likely, or range of mean ± 50% if range data absent)<br>Observed range > 50: (logtriangular: observed upper and lower range, with observed geomean as most likely) |
| $K_{ps1}$ (plant-soil partition coefficient— rainsplash) | (normal: mean = 0.0034, std. dev. = 0.0034) |
| $K_{pa2}$ (plant-air partition coefficient—particulates) | (normal: mean = 3300, std. dev. = 4950) |
| $B_v$ (plant bioconcentration factor) | Tritium (hydrogen-3): (triangular: lower limit = 0.7, most likely value = 0.9, upper limit = 0.95)<br>Other contaminants: (triangular: 10th and 90th percentiles as range, observed median as most likely, or ± 50% of mean if range is unknown) |
| Contaminant absorption fractions ($\alpha_{d,s,v,p}$ and $K_p$) | (triangular: observed upper and lower range, with observed mean as most likely, or range of mean ± 50% if range data absent, with a maximum absorption of 1 and a minimum of 0) |
| $K_{ei}$ (contaminant depuration rate) | (triangular: observed upper and lower range, with observed mean as most likely; range = mean ± 50% if range data absent and taxonomic extrapolation is within the same level; otherwise range = mean ± 100%) |
| Environmental Concentrations | Lognormal: (observed geometric mean and geometric standard deviation) |

(a) Parameters are defined in the "Exposure Model Description and Parameters" section in Appendix I-D.

Contaminants can become airborne (and therefore available for inhalation) through wind erosion from within the segment's riparian zone, which was assumed to have a grass cover. Where trees actually occur in the study area, this assumption will overestimate inhalation exposures. The average distances above the substrate where animals were assumed to be found (diffusion height) are shown in Appendix I-D (diskette file "paramtrs.xls"). Contaminant uptake by plant roots was assumed to be from sediment, the data for which were obtained from samples of the riverbed (see Section 3.0). An exception was for tritium (hydrogen-3), which was assumed to be taken up by plants from groundwater. The sediment samples were assumed to represent the soil of the riparian zone.

### 4.2.6 Validation Results

**In this section, we describe how we evaluated the degree to which the modeled exposures were accurate and give the results of our tests.**

To determine whether the model produced reasonable results, output from the exposure model was evaluated against several data sets obtained from the literature. The data sets used in the validation were not used in setting the parameters for the model. The basis used for comparison was the ratio of the reported body burden to that of the organism's food, or to the water concentration in the case of fish. These ratios are generally termed "transfer factors" or "concentration factors" (Peterson 1983). The exposure model was run to obtain transfer factors for contaminants for each segment where a complete media concentration file was available. Figure 4.6 shows the results of this comparison, and Table 4.22 shows the alpha code references for Figure 4.6. The CRCIA Team elected to use these results as a source of information and not to change the model to better match reported results.

Because the exposure conditions were set conservatively, the exposure model was expected to produce transfer factors somewhat greater than those referenced in the literature. This was the case for most contaminants. Mercury was underestimated in fish (by 15 times), and tritium (hydrogen-3) (by 2 times) in herbivorous mammals (Table 4.23).

The mercury underestimate is likely due to a deficiency in the media data input for the model. Virtually all of the mercury data were reported at the instrument detection limits (see Section 3.0). In general, no values exceeded detection limits for sediment. Consequently, the input data did not reflect equilibrium between the abiotic compartments. The literature values for strontium-90 varied widely, indicating a great deal of uncertainty in factors controlling the movement of this contaminant in terrestrial and aquatic biota.

Constituents for which exposures were overestimated in fish were technetium-99 (170 times), tritium (hydrogen-3) (10 times), cobalt-60 (5 times), zinc (2.5 times), and uranium-238 (2 times). The behavior of technetium-99 in aquatic biota is poorly studied, in part because of difficulties in chemical analysis (Driver 1994). Based on limited data sets, recommended transfer factors range from 15 to 30 (surface water to fish muscle, Driver 1994). A wide range in transfer factors was estimated for different species: rainbow trout had a mean transfer factor of 6, as did salmon.

Underestimated constituent concentrations in herbivorous mammals were limited to tritium (hydrogen-3) (2 times).

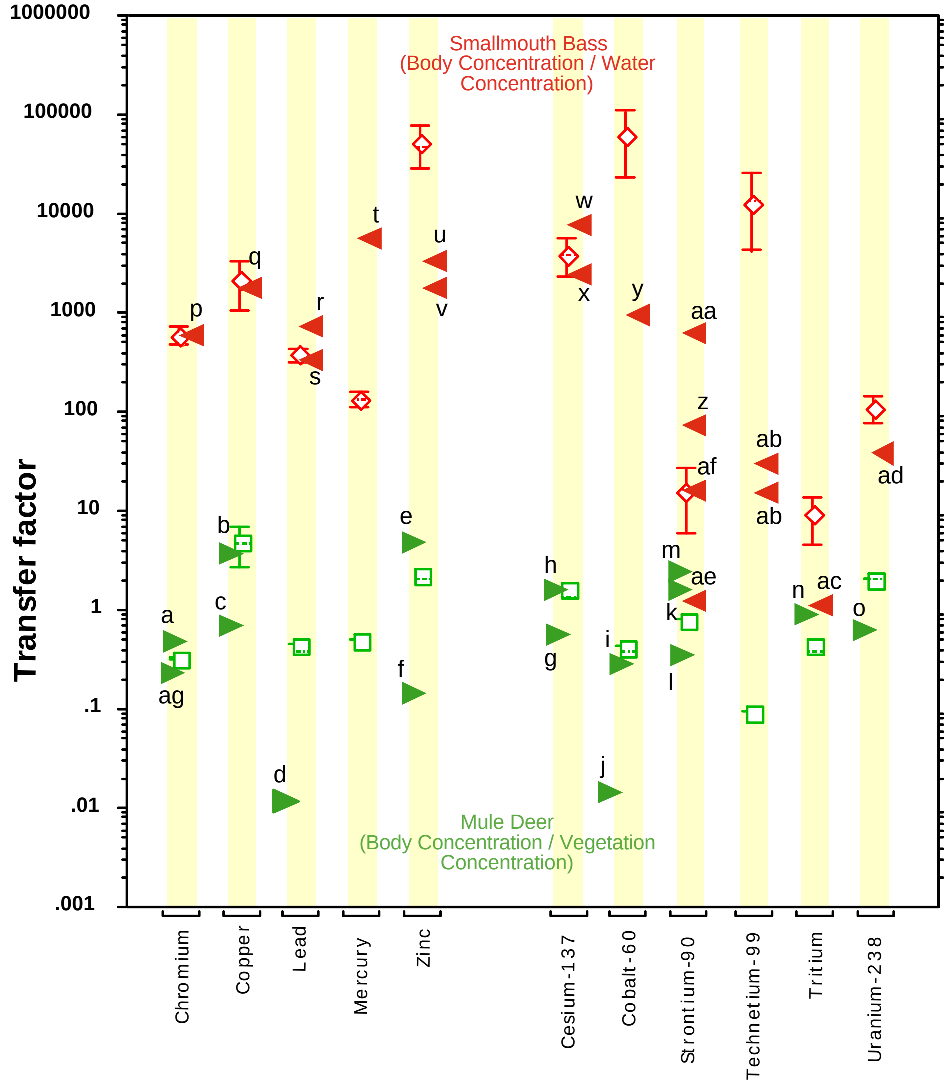


**Figure 4.6**. Comparison of Transfer Factors Estimated from the Exposure Model (⌖1 standard error bars) and Those Reported in the Literature (arrows with alpha code references in Table 4.22) (squares = means for mule deer, diamonds = means for smallmouth bass)

For the mammalian results, uranium-238 concentrations were overestimated by 2.5 times. The uranium-238 reference (Driver 1994) is based on food-only transfer, whereas much of the exposure in the present analysis is due to drinking the equivalent of groundwater. Zinc values reported in the literature ranged over 1.5 orders of magnitude. The model's estimate was within the upper portion of that range. The model's output for mollusc (clams and mussels) body burden as a fraction of sediment concentration versus published benchmarks (Thomann et al. 1995) are compared in Figure 4.7. The model output matched benchmarks for chromium, copper, and nickel. Lead was slightly overestimated, and mercury and

**Table 4.22**. References for Literature-Derived Transfer Factors in Figure 4.6

| Table Key | Element or Isotope | Species | Reference Medium | Factor (fresh weight basis) | Reference |
|---|---|---|---|---|---|
| a | Chromium | Mammals-mice | feed | 0.5 (heart & liver) | Driver 1994 |
| b | Copper | Mammals-mice | feed | 4 | Beyer et al. 1990 |
| c | Copper | Mammals-herbivores | feed | 0.7 | Pascoe et al. 1996 |
| d | Lead | Cattle-muscle | feed | 0.013 | Stevens 1992 |
| e | Zinc | Mammals-mice | feed | 9 | Beyer et al. 1990 |
| f | Zinc | Mammals-herbivores | feed | 0.15 | Pascoe et al. 1996 |
| g | Cesium-137 | Golden mouse | feed | 0.19 | Kaye & Dunaway 1962 |
| h | Cesium-137 | Mammals-herbivores | feed | 2 | Driver 1994 |
| i | Cobalt-60 | Mammals-herbivores | feed | 0.3 | Driver 1994 |
| j | Cobalt-60 | White-footed mouse | feed | 0.013 | Kaye & Dunaway 1962 |
| k | Strontium-90 | White-footed mouse | feed | 1.8 | Kaye & Dunaway 1962 |
| l | Strontium-90 | Golden mouse | feed | 0.39 | Kaye & Dunaway 1962 |
| m | Strontium-90 | Mammals-herbivores | feed | 2.5 (midrange) | Driver 1994 |
| n | Tritium (Hydrogen-3) | Mammals-rabbits | feed/water | 0.97 | Driver 1994 |
| o | Uranium | Mammals-deer | feed | 0.7 | Driver 1994 |
| p | Chromium | Fish-Salmo | water | 560 | Dallinger & Kautzky 1985 |
| q | Copper | Fish-Salmo | water | 1800 | Dallinger & Kautzky 1985 |
| r | Lead | Fish | water | 720 | Van Hassel et al. 1980 |
| s | Lead | Fish-Salmo | water | 360 | Dallinger & Kautzky 1985 |
| t | Mercury (total) | Fish | water | 6000 | Hendriks 1995 |
| u | Zinc | Fish-Salmo | water | 3600 | Dallinger & Kautzky 1985 |
| v | Zinc | Fish | water | 1920 | Van Hassel et al. 1980 |
| w | Cesium-137 | Smallmouth bass | water | 7800 | Whicker et al. 1990 |
| x | Cesium-137 | Fish-walleye | water | 2500 | Driver 1994 |
| y | Cobalt-60 | Fish-smelt | water | 1000 (largest) | Driver 1994 |
| z | Strontium-90 | Fish-carnivore | water | 75 (midrange) | Driver 1994 |
| aa | Strontium-90 | Smallmouth bass | water | 680 | Whicker et al. 1990 |
| ab | Technetium-99 | Fish | water | 15 (default) | Driver 1994 |
| ac | Tritium (Hydrogen-3) | Fish | water | ~1 | Driver 1994 |
| ad | Uranium | Fish | water | 38 | Driver 1994 |
| ae | Strontium-90 | Fish-bass | water | 1.15 | Peterson 1983 |
| af | Strontium-90 | Fish-trout | water | 20 | Peterson 1983 |
| ag | Chromium | Mammals-cattle muscle | feed | 0.26 (calculated for muscle) | Peterson 1983 |

**Table 4.23**. Performance of Exposure Model Versus Published Transfer Factor References

| Species Group | Model < Reference | Model = Reference (within 95% CI) | Model > Reference |
|---|---|---|---|
| Herbivorous mammals | $^{3}H$ | Cr, $^{137}Cs$, $^{60}Co$, Cu, Pb, $^{90}Sr$, Zn | $^{238}U$ |
| Predatory fish | Hg | $^{137}Cs$, Cr, Cu, Pb, $^{90}Sr$, Zn | $^{60}Co$, $^{99}Tc$, $^{3}H$, $^{238}U$ |

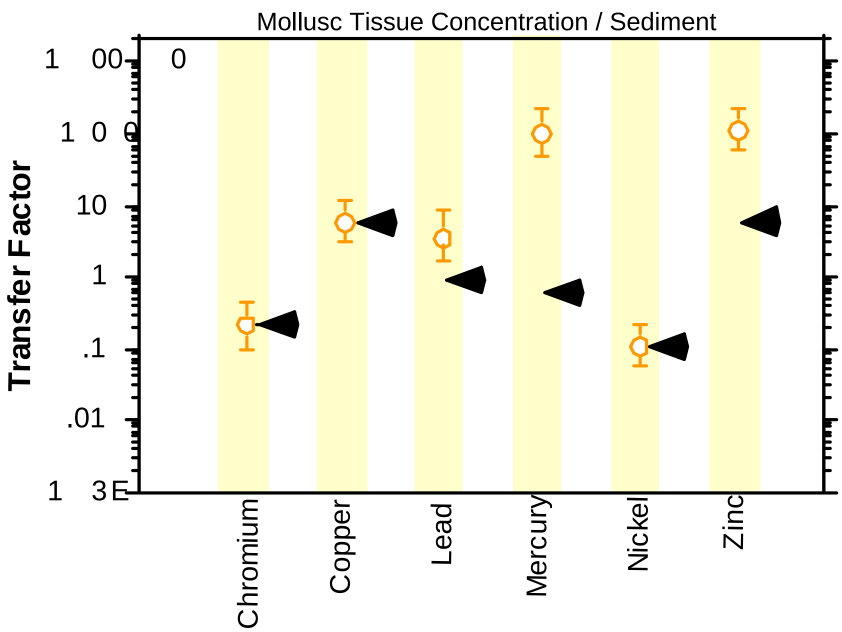


**Figure 4.7**. Comparison of Sediment-to-Mollusc Transfer Factors Estimated from the Exposure Model (mean ±1 standard error from deterministic analyses for all segments for which data were available) and Those Reported in the Literature (arrows, Thomann et al. 1995)

zinc were greatly overestimated. The overestimated mercury was due to the lack of data above the detection limit for sediment as noted above. The model overestimated zinc by a factor of 10 because the published bioconcentration factors vary much more widely than for any other metal except mercury. This factor is discussed later in Section 4.3.

In general, except for tritium (hydrogen-3) in terrestrial herbivores and technetium-99 and mercury in predatory fish, the model performed adequately. The model met the operational criteria for the risk assessment: accurate representation of exposure as much as possible with errors favoring a conservative estimate of exposure.

### 4.2.7 Adjusting Exposure Estimates for Nutrient/Micronutrient Metals

> **Our computer models calculated the concentration of a contaminant in the tissue of a species in direct proportion to the concentration of that contaminant in the environment. However, nutrient metals do not behave in this fashion. Therefore, we needed to adjust the calculation for certain contaminants. In this section, we explain which contaminants were adjusted and how.**

Models of contaminant movement in biological systems generally are linear at equilibrium, in that the equilibrium tissue concentration is directly proportional to the environmental concentration (for example, Hope 1995; Thomann et al. 1995; EPA 1991, 1992b, 1993; Peterson 1983). This is probably appropriate for elements that are not essential to biological function, are not analogs of such metals, or are not taken up by organisms via nutrient pathways. However, several of the metals in this assessment do not meet these conditions.

Organisms adjust uptake and loss of essential elements to keep internal concentrations within certain tolerance limits. The tissue concentrations of the element are maintained within a relatively narrow band despite some variation in environmental concentrations (Newman and Heagler 1991). For example, absorption of ingested copper by vertebrates is a function of the copper concentration in the body: more is absorbed when the body concentration is low, and less is absorbed when the body concentration is high (Piscator 1979). A classification of metals by their nutrient/non-nutrient status is presented in Table 4.24.

**Table 4.24**. Biological Classification of Metal Contaminants (Beeby 1991) (Shaded cells are non-nutrient analogs. Bolded contaminant names are toxic heavy metals.)

| **Period**[a] | **Macronutrient** | | **Micronutrient** | | | | | **Non-Essential** | |
|---|---|---|---|---|---|---|---|---|---|
| 3 | Na | Mg | | | | | | | |
| 4 | K | Ca | Cr | Mn | Fe, Co, Ni | Cu | Zn | | |
| 5 | | Sr | | Tc | | | Cd | | |
| 6 | Cs | | | | | | | Hg, Pb | |
| 7 | | | | | | | | U, Np, Eu | |
| (a) Period is from the Periodic Chart of the Elements. | | | | | | | | | |

Nutrient elements include a number of the contaminants in this risk assessment, especially copper and zinc. Therefore, the degree to which the estimated body burdens of these elements reflect the true body burdens depends on how closely the organism is able to regulate the element and how far the environmental concentrations exceed the range over which the organism is able to maintain homeostasis.

The CRCIA Team determined that risk to organisms from nutrient metals should be evaluated as an increment to the estimated risk posed in Segment 1. The CRCIA Team also decided that risk should be evaluated where media concentrations statistically exceeded those in Segment 1. Otherwise, the upstream value should be taken as the baseline for these contaminants.

Non-parametric statistics (Mann-Whitney U test) were used to compare sediment concentrations in Segment 1 with concentrations in the downstream segments. Nickel, lead, and zinc were in highest average concentrations in Segment 1 (Table 4.25). Copper was significantly elevated in Segment 14. Chromium was significantly elevated in Segments 2 and 4. Cobalt-60 was elevated in a number of segments, while cesium-137 was significantly elevated only in Segment 12.

Because exposure models (including the models adapted for this assessment) assume a monotonic relationship between environmental concentration and an organism's body burden, the presence of a homeostatic zone, such as Figure 4.8 shows, will result in the model overestimating tissue concentrations over a portion of the range of environmental concentrations (Chapman et al. 1996). This overestimation will be maximal at and beyond the upper portion of the homeostatic zone. To address the issue of how this information can be used to correct the model for micronutrient metals, the following was assumed:

1. The upper portion of the body concentration curve for nutrient metals parallels the non-nutrient curve.

2. The environmental concentrations of nutrient metals in the region immediately upriver from Hanford Site-derived inputs to the Columbia River are within the homeostatic zone.

**Table 4.25**. Segments Where Contaminant Concentrations in Sediment Significantly Exceeded Concentrations in Segment 1 (Mann-Whitney U test, significance level of $P<0.1$)

| Analyte | Segment | U | P |
|---|---|---|---|
| Cr | 2 | 9.45 | 0.0021 |
| Cr | 4 | 2.91 | 0.088 |
| Cu | 14 | 3.33 | 0.068 |
| Ni | none | | |
| Pb | none | | |
| Zn | none | | |
| Co-60 | 6 | 13.2 | 0.0003 |
| Co-60 | 8 | 27.2 | 0.0001 |
| Co-60 | 9 | 28.11 | 0.0001 |
| Co-60 | 10 | 9.9 | 0.0017 |
| Co-60 | 12 | 23.5 | 0.0001 |
| Co-60 | 13 | 8.1 | 0.0044 |
| Co-60 | 14 | 22.7 | 0.0001 |
| Co-60 | 15 | 15.2 | 0.0001 |
| Co-60 | 16 | 22.6 | 0.0001 |
| Co-60 | 17 | 9.8 | 0.0018 |
| Co-60 | 21 | 11.8 | 0.0006 |
| Cs-137 | 12 | 6.6 | 0.01 |

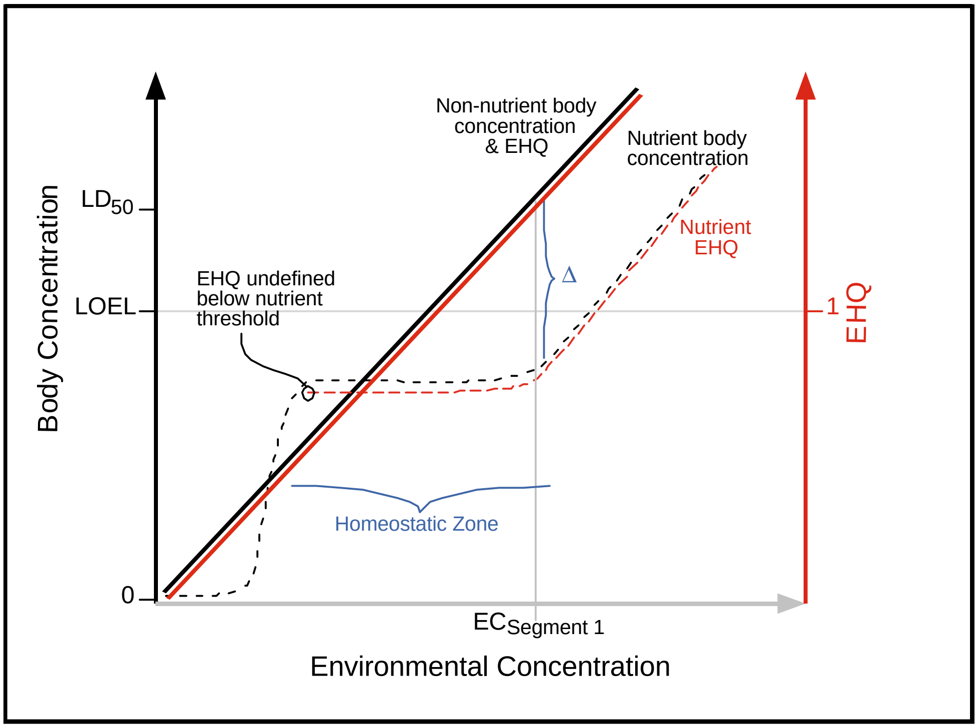


**Figure 4.8**. Equilibrium Relationships Between Environmental and Tissue Concentrations for Nutrient and Non-Nutrient Metals and Their Relationships to EHQ (terms are described in text)

Assumption 1 is conservative in the following derivation if the slope of the curve for nutrient metals is no greater than the slope of the curve for non-nutrients (see discussion below). Assumption 2 will be made conservative by further assuming the environmental concentration is at the upper end of the homeostatic zone. The validity of assumption 2 will be evaluated by comparing media and fish tissue concentrations from samples obtained from uncontaminated areas.

As stated earlier, the fundamental comparison in this assessment is the ratio of the body concentration or its analog, such as a water or ingestion daily exposure to the LOEL benchmark for a given species, which is termed an EHQ. This is shown in Figure 4.8. By definition, the EHQ within the homeostatic range should be less than 1, but will increase above 1. Thus, the actual EHQ within this region should be:

$$EHQ_{\text{segment 1}} = BC_{\text{nutrient}} / LOEL, \tag{4.2}$$

where $BC_{nutrient}$ refers to the whole-body concentration of the nutrient metal.

As shown in Figure 4.8, the non-nutrient body concentration estimate exceeds the nutrient body concentration estimate by an amount $\Delta$ at the upper inflection point, which was set to be at the upriver Segment 1:

$$BC_{\text{nutrient, 1}} = BC_{\text{non-nutrient, 1}} - \Delta \tag{4.3}$$

If a species-specific body concentration for species from uncontaminated areas can be obtained from the literature, the value for $\Delta$ can be derived by subtraction:

$$\Delta = BC_{\text{non-nutrient, 1}} - BC_{\text{nutrient, 1}}. \tag{4.4}$$

Then,

$$EHQ_{\text{segment 1}} = (BC_{\text{non-nutrient, 1}} - \Delta) / LOEL, \tag{4.5}$$

which, assuming assumption 1 above holds true for any segment where $EHQ_{\text{segment x}} \geq EHQ_{\text{segment 1}}$, then

$$EHQ_{\text{segment x}} = (BC_{\text{non-nutrient, x}} - \Delta) / LOEL \tag{4.6}$$

Note that if the non-nutrient curve diverges from the nutrient curve above the homeostatic zone, use of equation (4.6) will overestimate $BC_{\text{non-nutrient, x}}$; thus, its use is considered conservative under these conditions. If the curves converge, the model will underestimate tissue concentrations above the homeostatic zone, and the resulting EHQ will be underestimated. The curves are expected to be approximately parallel, however, because the depuration and BCF data upon which the linear model is based were usually obtained from experimental exposures to relatively high, fluctuating concentrations of metal. Such conditions would produce the upper tail of the non-linear curve.

The validity of assumption 2 (regarding environmental concentrations in Segment 1 being within the homeostatic zone for copper, chromium, and zinc) was evaluated using data collected from sediment and fish within the Columbia River upstream from the study area by Washington State Department of Ecology and others (Fuhrer et al. 1996; Munn et al. 1995; Serdar et al. 1994; Serdar 1993). The sediment concentration data for copper and zinc from Segment 1 fall within the lower range of samples obtained from Franklin D. Roosevelt Lake and downstream (Figure 4.9). Tissue data for largescale suckers collected from these sites show no apparent relationship to sediment concentrations within this range (Figure 4.9). Similarly, benthic insect monitoring data from the Yakima River also show no relationship between tissue concentrations and sediment concentrations within the ranges found in Segment 1 (Figure 4.10). There is thus tentative support for the validity of assumption 2. Note that the sediment concentrations in the North Port area exceeded those in Segment 1, and the fish tissue concentrations in North Port also are higher than from other areas, supporting assumption 2.

The model was therefore calibrated to produce approximate average tissue concentrations for copper and zinc within Segment 1. The calibration procedure focused on selecting ranges for bioconcentration

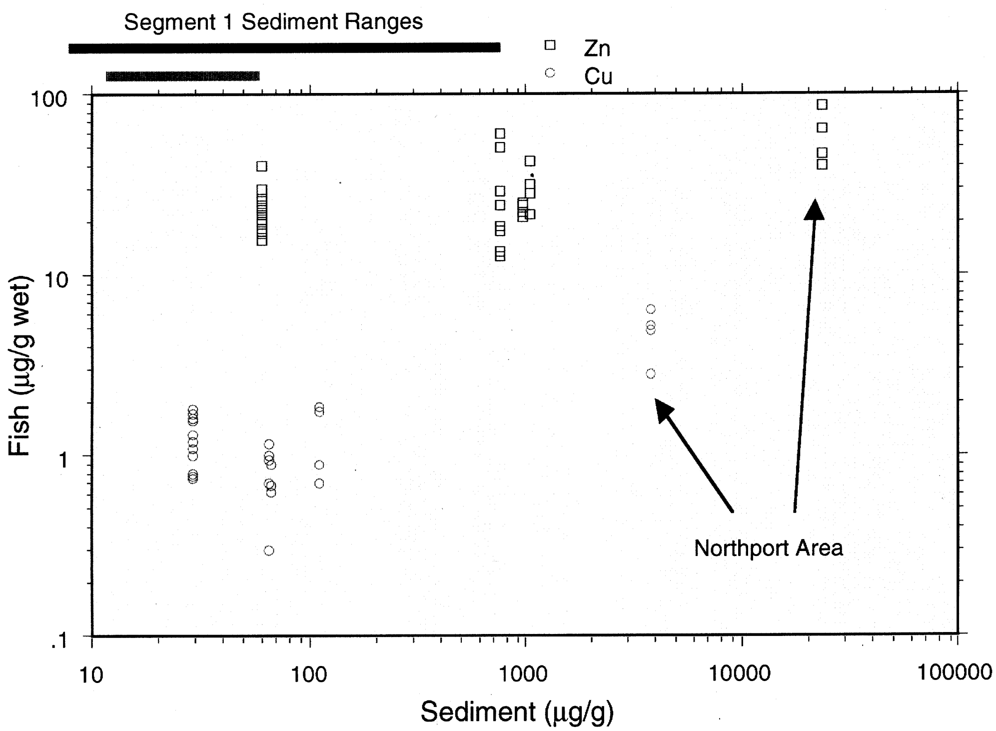


**Figure 4.9**. Sediment and Tissue Largescale Sucker Concentrations of Copper and Zinc from Lake Roosevelt and Downstream Areas (data sources given in text)

factors, ingestion assimilation fractions, and depuration rates from within the published ranges for parameters that would produce estimates between one-half and five times the average tissue concentrations for species obtained from uncontaminated areas. These objective concentrations are given in Table 4.26. Calibration parameters are given in the "Calibration Parameters for Copper and Zinc" section of Appendix I-D.

### 4.2.8 Deterministic Analyses Using Media Concentration Maxima

**We used the deterministic method in the computer model to identify the species, segments, and contaminants that posed a potential hazard and therefore needed to be evaluated further using the stochastic method. All segments contained contaminants at levels posing a potential hazard, including Segment 1, which is upstream from the Hanford Site and therefore considered to be indicative of contamination not resulting from the Hanford Site. Contaminants requiring further evaluation were ammonia, cesium-137, chromium, cobalt-60, copper, cyanide, europium-154, lead, mercury, nickel, nitrate, strontium-90, tritium (hydrogen-3), uranium-234, uranium-238, and zinc.**

The exposure model was run in deterministic mode to identify river segments, contaminants, and species that should be evaluated further in the stochastic evaluation. This objective was addressed by identifying areas where the maximum observed concentration of contaminants would pose a potential hazard to any Tier II species if that species were exposed continuously to those concentrations. Model settings used in the

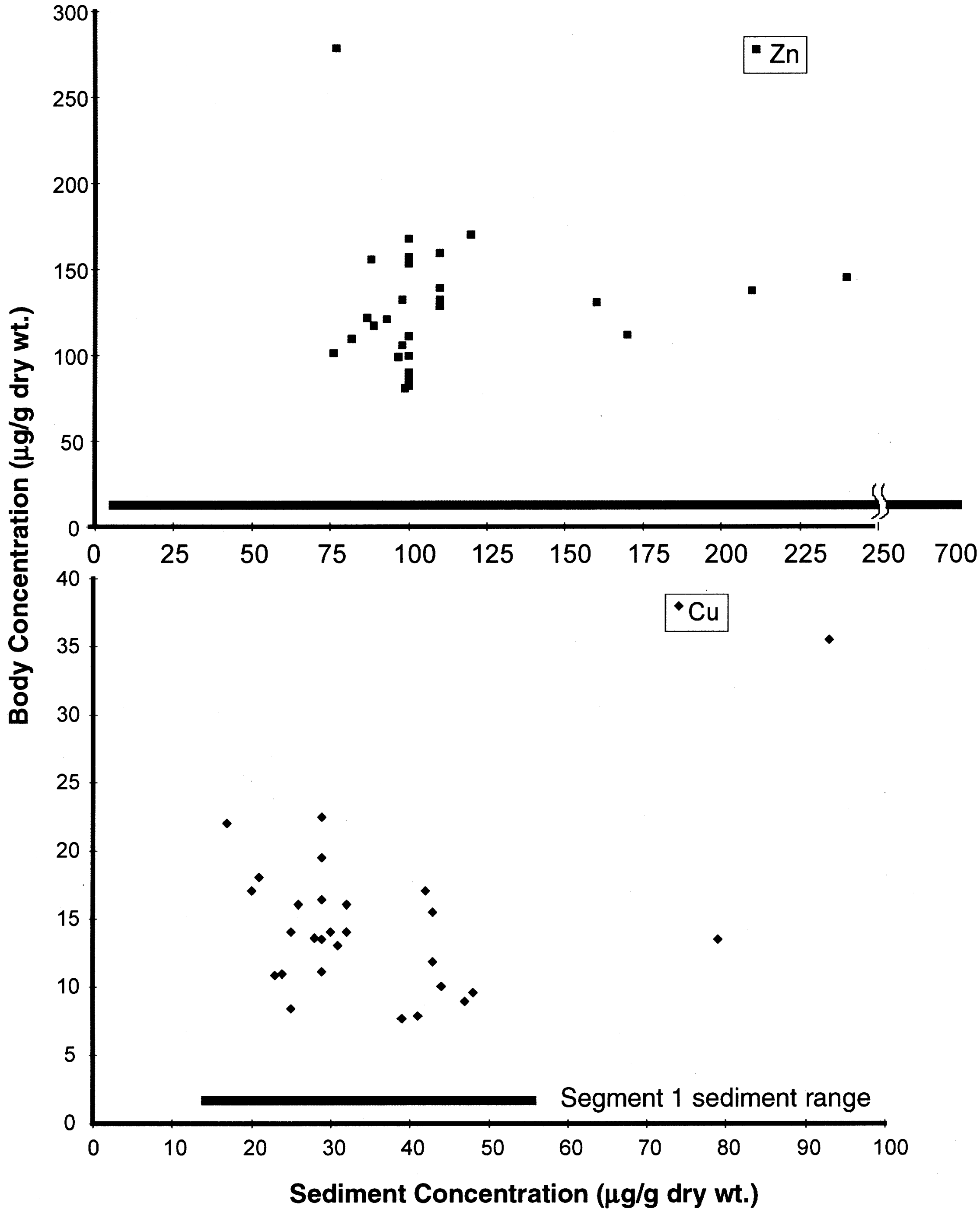


**Figure 4.10**. Sediment and Tissue (Caddisfly larvae) Concentrations of Copper and Zinc from the Yakima River