Table 4.28. Number of Aquatic Species for Which the Deterministic EHQ Exceeded 1 (Counts are displayed by contaminant, river segment, and species grouping.)[(a)]

No. of Species with EHQ >1		Contaminant																				
Segment	Species Category	ammonia	benzene	cesium-137	chromium	cobalt-60	copper	cyanide	europium-152	europium-154	Iodine-129	lead	mercury	nickel	nitrate	strontium-90	Technetium-99	Tritium	Uraniuim-234	Uranium-238	xylene	zinc
1	Invertebrate	0	0	0	0	0	6	0	0	0	0	1	1	0	0	0	0	0	0	0	0	7
	Vegetation	0	0	0	0	0	3	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	Vertebrate adult	0	0	0	0	0	0	0	0	0	0	2	0	0	0	0	0	0	0	0	0	5
	Vertebrate eggs	0	0	0	2	0	2	0	0	0	0	0	0	0	0	0	0	0	0	0	0	2
	Vertebrate larvae	0	0	0	0	0	3	0	0	0	0	3	3	0	0	0	0	0	0	0	0	4
1 Total		0	0	0	2	0	14	0	0	0	0	6	4	0	0	0	0	0	0	0	0	18
2	Invertebrate	5	0	0	1	0	1	0	0	0	0	1	1	0	0	0	0	0	0	0	0	3
	Vegetation	0	0	0	1	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	Vertebrate adult	2	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	3
	Vertebrate eggs	1	0	0	2	0	2	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1
	Vertebrate larvae	0	0	0	0	0	1	0	0	0	0	3	3	0	0	0	0	0	0	0	0	4
2 Total		8	0	0	4	0	5	0	0	0	0	5	4	0	0	0	0	0	0	0	0	11
3	Invertebrate	0	0	0	0	0	0	0	0	0	0	1	1	0	0	0	0	0	0	0	0	4
	Vegetation	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	Vertebrate adult	0	0	0	0	0	0	0	0	0	0	1	2	0	0	0	0	0	0	0	0	0
	Vertebrate eggs	0	0	0	2	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	Vertebrate larvae	0	0	0	0	0	0	0	0	0	0	3	3	0	0	0	0	0	0	0	0	0
3 Total		0	0	0	2	0	0	0	0	0	0	5	6	0	0	0	0	0	0	0	0	4
4	Invertebrate	4	0	0	0	0	4	5	0	0	0	1	2	0	0	0	0	0	0	0	0	7
	Vegetation	0	0	0	0	0	2	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1
	Vertebrate adult	2	0	0	0	0	1	0	0	0	0	2	2	0	0	0	0	0	0	0	0	5
	Vertebrate eggs	1	0	0	2	0	2	1	0	0	0	0	0	0	0	0	0	0	0	0	0	2
	Vertebrate larvae	0	0	0	0	0	4	2	0	0	0	4	3	0	0	0	0	0	0	0	0	4
4 Total		7	0	0	2	0	13	8	0	0	0	7	7	0	0	0	0	0	0	0	0	19
5	Invertebrate	0	0	0	0	0	2	0	0	0	0	1	1	0	0	0	0	0	0	0	0	3
	Vegetation	0	0	0	1	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	Vertebrate adult	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	2
	Vertebrate eggs	1	0	0	2	0	2	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	Vertebrate larvae	0	0	0	0	0	1	1	0	0	0	3	3	0	0	0	0	0	0	0	0	4
5 Total		1	0	0	3	0	6	2	0	0	0	5	4	0	0	0	0	0	0	0	0	9
6	Invertebrate	5	0	0	1	5	5	0	0	0	0	4	1	0	0	3	0	0	0	0	0	7
	Vegetation	0	0	0	1	2	1	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0
	Vertebrate adult	2	0	0	0	3	0	0	0	0	0	3	2	0	0	0	0	0	0	0	0	0
	Vertebrate eggs	1	0	0	0	2	0	0	0	0	0	2	0	0	0	0	0	0	0	0	0	0
	Vertebrate larvae	2	0	0	0	2	2	0	0	0	0	4	3	0	0	0	0	0	0	0	0	4
6 Total		10	0	0	2	14	8	0	0	0	0	14	6	0	0	3	0	0	0	0	0	11
7	Invertebrate	0	0	0	0	0	6	0	0	0	0	1	1	2	0	0	0	0	0	0	0	7
	Vegetation	0	0	0	0	0	3	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0
	Vertebrate adult	0	0	0	0	0	2	0	0	0	0	0	0	0	0	0	0	0	0	0	0	4
	Vertebrate eggs	0	0	0	2	0	2	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	Vertebrate larvae	0	0	0	0	0	4	0	0	0	0	3	3	0	0	0	0	0	0	0	0	4
7 Total		0	0	0	2	0	17	0	0	0	0	4	4	3	0	0	0	0	0	0	0	15
8	Invertebrate	5	0	0	0	0	6	0	0	0	0	1	2	0	0	0	0	0	0	0	0	7
	Vegetation	0	0	0	1	0	3	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	Vertebrate adult	4	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	3
	Vertebrate eggs	1	0	0	2	0	2	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	Vertebrate larvae	3	0	0	1	0	4	0	0	0	0	3	3	0	0	0	0	0	0	0	0	4
8 Total		13	0	0	4	0	15	0	0	0	0	4	6	0	0	0	0	0	0	0	0	14
9	Invertebrate	0	0	0	0	0	0	0	0	0	0	1	2	0	0	0	0	0	0	0	0	7
	Vegetation	0	0	0	1	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0
	Vertebrate adult	0	0	0	0	0	0	0	0	0	0	2	1	0	0	0	0	0	0	0	0	5
	Vertebrate eggs	0	0	0	2	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	2
	Vertebrate larvae	0	0	0	0	0	0	0	0	0	0	4	3	0	0	0	0	0	0	0	0	4
9 Total		0	0	0	3	0	0	0	0	0	0	8	6	0	0	0	0	0	0	0	0	18
10	Invertebrate	0	0	2	0	0	2	0	0	0	0	1	2	0	0	0	0	0	0	0	0	7
	Vegetation	0	0	0	1	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	Vertebrate adult	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	2
	Vertebrate eggs	0	0	2	2	0	2	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	Vertebrate larvae	0	0	1	0	0	1	0	0	0	0	3	3	0	0	0	0	0	0	0	0	4
10 Total		0	0	5	3	0	6	0	0	0	0	4	6	0	0	0	0	0	0	0	0	13
12	Invertebrate	0	0	0	0	0	5	0	0	0	0	0	0	0	0	0	0	0	0	0	0	7
	Vegetation	0	0	0	0	0	2	0	0	0	0	0	0	0	0	0	0	0	0	0	0	2
	Vertebrate adult	0	0	0	0	0	2	0	0	0	0	0	0	0	0	0	0	0	0	0	0	7
	Vertebrate eggs	0	0	0	2	0	2	0	0	0	0	0	0	0	0	0	0	0	0	0	0	2
	Vertebrate larvae	0	0	0	0	0	4	0	0	0	0	0	0	0	0	0	0	0	0	0	0	4
12 Total		0	0	0	2	0	15	0	0	0	0	0	0	0	0	0	0	0	0	0	0	22
13	Invertebrate	5	0	0	1	0	5	0	0	0	0	1	2	0	0	0	0	0	0	0	0	7
	Vegetation	0	0	0	1	1	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	Vertebrate adult	2	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	2
	Vertebrate eggs	1	0	0	0	2	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	Vertebrate larvae	0	0	0	0	1	2	0	0	0	0	0	3	0	0	0	0	0	0	0	0	4
13 Total		8	0	0	2	4	8	0	0	0	0	1	6	0	0	0	0	0	0	0	0	13
14	Invertebrate	0	0	0	0	0	2	0	0	0	0	1	2	0	0	0	0	0	0	0	0	5
	Vegetation	0	0	0	0	0	2	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	Vertebrate adult	0	0	0	0	0	0	0	0	0	0	2	2	0	0	0	0	0	0	0	0	5
	Vertebrate eggs	0	0	0	2	0	2	0	0	0	0	0	0	0	0	0	0	0	0	0	0	2
	Vertebrate larvae	0	0	0	0	0	1	0	0	0	0	4	3	0	0	0	0	0	0	0	0	4
14 Total		0	0	0	2	0	7	0	0	0	0	7	7	0	0	0	0	0	0	0	0	16
15	Invertebrate	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	7
	Vegetation	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	Vertebrate eggs	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	Vertebrate adult	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	2
	Vertebrate larvae	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	4
15 Total		0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	13
16	Invertebrate	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	7
	Vegetation	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	Vertebrate eggs	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	Vertebrate adult	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	2
	Vertebrate larvae	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	4
16 Total		0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	13
17	Invertebrate	0	0	0	0	0	5	0	0	0	0	3	0	0	0	0	0	0	0	0	0	7
	Vegetation	0	0	0	0	0	2	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0
	Vertebrate adult	0	0	0	0	0	0	0	0	0	0	2	0	0	0	0	0	0	0	0	0	3
	Vertebrate eggs	0	0	0	0	0	2	0	0	0	0	2	0	0	0	0	0	0	0	0	0	0
	Vertebrate larvae	0	0	0	0	0	3	0	0	0	0	4	0	0	0	0	0	0	0	0	0	4
17 Total		0	0	0	0	0	12	0	0	0	0	12	0	0	0	0	0	0	0	0	0	14
19	Invertebrate	4	0	0	0	0	2	0	0	0	0	3	0	2	0	0	0	0	0	0	0	7
	Vegetation	0	0	0	1	0	1	0	0	0	0	0	0	2	0	0	0	0	0	0	0	0
	Vertebrate adult	2	0	0	0	0	0	0	0	0	0	2	2	0	0	0	0	0	0	0	0	5
	Vertebrate eggs	1	0	0	2	0	2	0	0	0	0	0	0	0	0	0	0	0	0	0	0	2
	Vertebrate larvae	0	0	0	0	0	1	0	0	0	0	4	3	0	0	0	0	0	0	0	0	4
19 Total		7	0	0	3	0	6	0	0	0	0	9	5	4	0	0	0	0	0	0	0	18
20	Invertebrate	7	0	0	0	0	4	5	0	0	0	1	0	0	0	0	0	0	0	0	0	7
	Vegetation	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0
	Vertebrate adult	5	0	0	0	0	0	0	0	0	0	1	2	0	0	0	0	0	0	0	0	3
	Vertebrate eggs	1	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	Vertebrate larvae	1	0	0	0	0	1	2	0	0	0	4	3	0	0	0	0	0	0	0	0	4
20 Total		14	0	0	0	0	5	8	0	0	0	6	5	1	0	0	0	0	0	0	0	14
21	Invertebrate	7	0	0	0	0	5	0	0	0	0	3	1	2	0	0	0	0	0	0	0	7
	Vegetation	0	0	0	1	0	1	0	0	0	0	0	0	3	0	0	0	0	0	0	0	0
	Vertebrate adult	4	0	0	0	0	0	0	0	0	0	2	0	0	0	0	0	0	0	0	0	5
	Vertebrate eggs	0	0	0	2	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	2
	Vertebrate larvae	1	0	0	0	0	2	1	0	0	0	4	3	1	0	0	0	0	0	0	0	4
21 Total		12	0	0	3	0	8	2	0	0	0	9	4	6	0	0	0	0	0	0	0	18

(a) Algae EHQs were less than 1 everywhere, so they were not included in the table.

Table 4.29. Classification Criteria for Stochastic Risk Results

Risk Category	Nominal	Low	Medium	High
Risk Score	0	1	2	3
Proportion of simulations exceeding LOEL or LC/D_{50}	<25%	25-50%	51-75%	>75%

actual risk to the extent that the supporting data are representative and the contaminants conform to the assumptions of the exposure model, especially regarding the monotonic relationship between environmental concentration and body burden. These issues are discussed in more detail in the following section.

4.2.10 Uncertainty Analysis

As stated earlier, we use the term "uncertainty" to mean the likelihood of a certain amount of variability in model parameters or dose estimates. In this section, we discuss the uncertainty to be found in the exposure model, data, exposure scenarios, and toxicological response, as well as the effect of this uncertainty on the risk assessment results.

The large number of parameters (about 5500) produces two types of uncertainties in the exposure model: those resulting from compounding errors in a model that has so many parameters and those resulting from varying levels of confidence in the data used in the model.

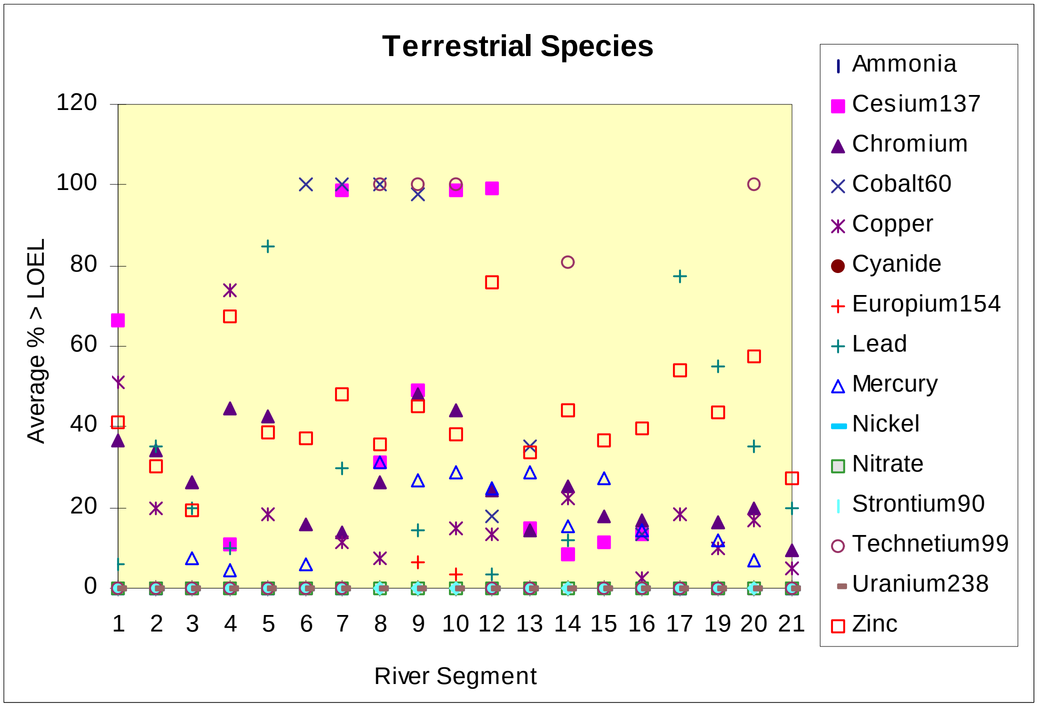
Uncertainties in the media data include natural variability in the concentrations and uncertainties in knowledge that stem from using surrogates, such as groundwater and/or spring/seep water as a surrogate for pore water. Uncertainties in the exposure scenarios include uncertainty in the media data as well as the lack of specific knowledge about which media the organism contacts for how long. One example is the conservative assumption that pore water extends into the river in an undiluted form for some distance. Uncertainties in the toxicological response references result from extrapolating between data from one species to another and from one endpoint to another.

This section discusses the uncertainty in the exposure model, the media data, the exposure scenarios, and in the toxological response, as well as the effect of this uncertainty on the risk assessment results.

4.2.10.1 Uncertainties Intrinsic to the Exposure Model

The exposure model contains about 140 parameters for each contaminant, about 400 parameters for the species, and a 45-by-45 matrix of predation fractions. Additionally, five site-specific parameters were necessary to evaluate exposures. Collectively, calculating exposures of all species to all contaminants required estimating about 5500 parameters. This produces two types of uncertainties: those arising from compounding errors in a parameter-rich model, and those arising from varying levels of confidence in the data used to parameterize the model.

The parameter-rich nature of the exposure model clearly contributes a large portion of uncertainty to the overall exposure result. Although not all parameters were required to estimate exposure for every species (for example, coyote body size was irrelevant to estimating tissue concentrations of uranium in periphyton), the uncertainty in the species-specific and species-by-contaminant parameters that are a part of an exposure pathway act together to enhance the uncertainty in the resulting estimate. A less



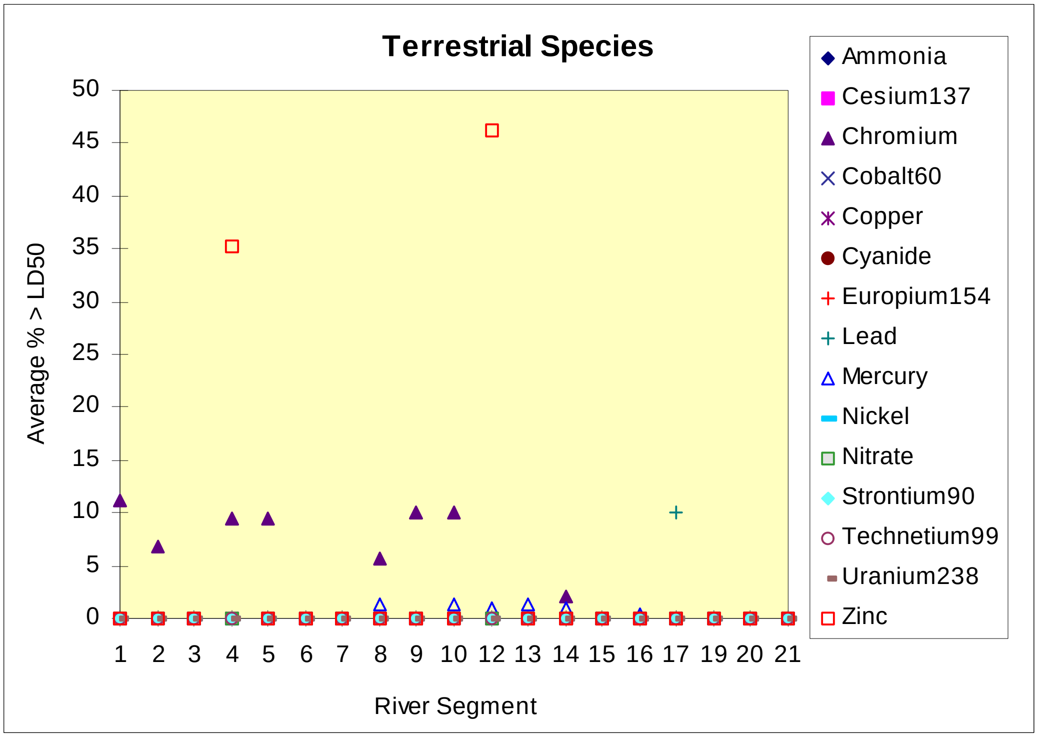


Figure 4.13. Relative Risk Indices for Terrestrial Species Based on the Stochastic Exposure Model

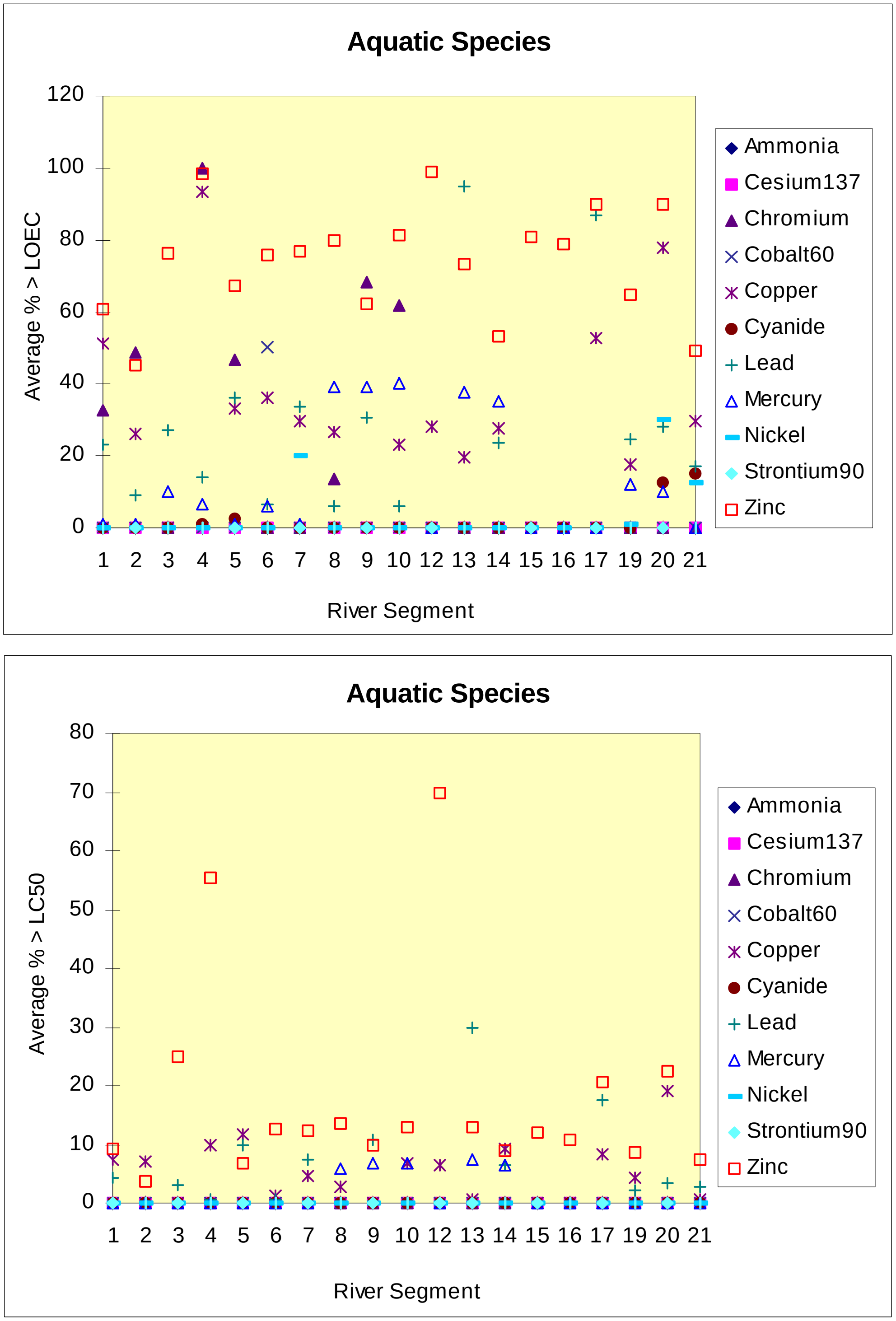


Figure 4.14. Relative Risk Indices for Aquatic Species Based on the Stochastic Exposure Model

parameter-rich model could be expected to produce a less-variable exposure estimate, although some may place less confidence in this estimate because of the more simplistic nature of the model used. This illustrates an interesting paradox: increasing model complexity may lead to increased uncertainty in the output, but also may lead to increased perception of confidence. This paradox may be resolved somewhat by validating the model against tissue concentration data from the Hanford Site and elsewhere (Table 4.23 and Figure 4.15).

Parameterization uncertainty can be divided into three classes: uncertainty due to cross-species extrapolations required by the lack of data, uncertainty caused by extrapolating conditions reported in the literature to those present in the Columbia River study area, and uncertainty due to variability among the species and studies comprising the various references used. Although a large body of literature was reviewed to obtain estimates for each parameter, emphasizing Hanford Site-specific data as much as possible, the review was not exhaustive. The sources reviewed varied both in degree of relevance and in the quality of results. This area could be improved by conducting a more thorough review of the literature and a detailed analysis of data quality and applicability to conditions within the study area.

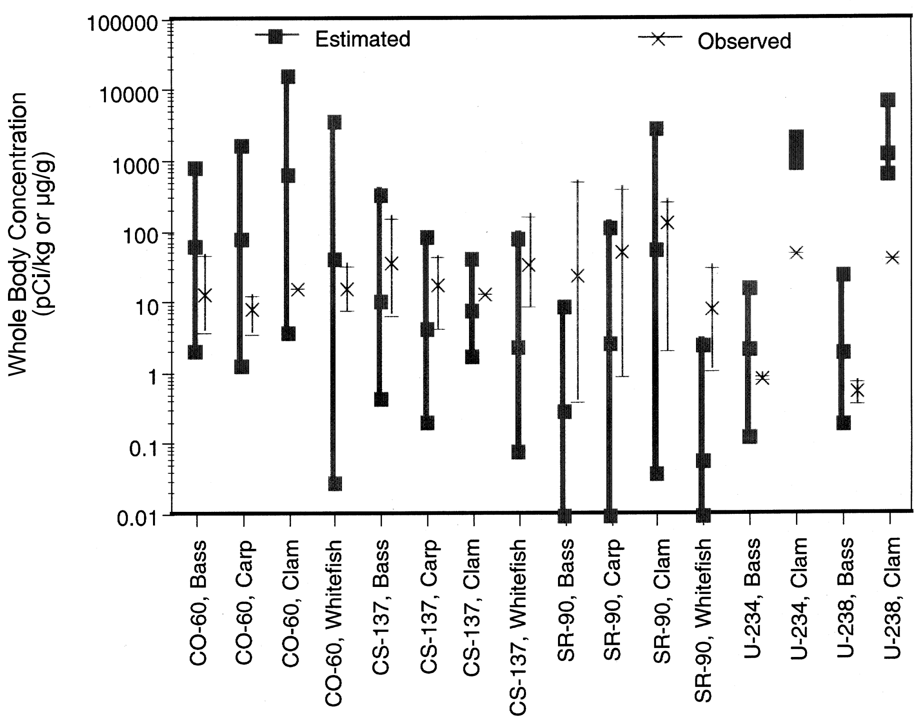


Figure 4.15. Measured Versus Estimated (mean and range) Whole Body Concentrations of Radionuclides in Biota (Measured values were available for biota. Comparisons are simulation results for a given segment and species.)

Because data for Tier II species were often lacking for many contaminant-by-species parameters, data from other species had to be used. Although this uncertainty was represented in the wide parameter bounds for the stochastic analysis, there is no established basis for accurately estimating the uncertainty introduced by such extrapolation. In some cases, such as iodine-129 depuration in terrestrial species, relatively little variation occurred between such widely diverse taxonomic groups as chickens and mammals. In other cases, such as bioconcentration factors for zinc, variation both within and between taxonomic groups covered several orders of magnitude. EPA, DOE, and other agencies have funded a great deal of research into establishing methodologies for extrapolating toxicological data across taxonomic groups, with less attention to extrapolating parameters needed to estimate exposures across taxonomic groups. The toxicological databases used for most of the input values varied in the amount of detail they provided. In most cases, toxicity values (for example, LC_{50} or LOELs) were provided without detail on the species life stage or exposure conditions. Knowledge about test conditions is particularly important for metals such as copper and zinc because water hardness and pH dramatically affect both bioavailability and toxicity of the compounds to aquatic organisms. The effect of including this uncertainty in the stochastic simulations was to inflate the range of estimated potential exposures beyond that actually present.

An estimate of this inflated range is demonstrated by comparing tissue concentrations estimated from the exposure model with actual measurements from animals collected during routine and special DOE environmental monitoring programs within the study area. Biota data were obtained from PNNL's Surface Environmental Surveillance Project and HEIS databases from the past 5 years for the 100-N Area (corresponding to Segment 6), F-Slough (corresponding to Segment 14), and the 300 Area (corresponding to Segment 20). These data consist of radionuclide concentrations in muscle tissues. To use these data, reported values that were less than the total analytical error were substituted with half the total analytical error. Reported muscle values were converted to approximate whole body concentrations minus gut contents by applying body-fraction and radionuclide activity ratios contained in Ney and Van Hassel (1983) and Dauble and Poston (1994). The observed versus estimated ranges are compared in Figure 4.15.

The available data include the key radionuclides in the risk assessment, but none of the non-radioactive metals. As illustrated in Figure 4.15, the exposure model estimates contained and exceeded the entire range of observed body burdens for all radionuclides and species except for strontium-90 in fish, cesium-137 in whitefish, and uranium in clams. Tissue levels of strontium-90 were measured in biota at the 100-N Area, at F-Slough (only in smallmouth bass), and at the 300 Area (only in clams). The range estimated for clam tissue contained that of the observed tissue data, but the estimated ranges for the fish species were farther out of alignment with observed data. This deviation may be from the bioavailability conditions prevailing in the literature studies that were used to parameterize the model, which underestimate those in the study area; to environmental media data for strontium-90 that underestimate levels experienced by the species in the 100-N Area and F-Slough; or to exposure conditions in the model that underestimate exposure conditions in the field. The latter is the least likely explanation, since estimates for all contaminant exposures were based on the same exposure data set (for example, fractional exposure to pore water), and estimates of tissue concentrations for the other contaminants were not underestimated so severely. However, as Figure 4.6 and its related discussion show, strontium-90 behavior in aquatic systems varies widely (transfer factors range over 3 orders of magnitude). Consequently, the present model may be underestimating strontium-90 in at least some aquatic organisms.

Given these data, the CRCIA Team elected not to calibrate the exposure model to better align predicted with observed body burdens. This option was rejected in part because the team recognized that the exposure scenarios were probably more conservative than reality. Thus, actual data on tissue concentrations would not be expected to match simulated results. The measured concentrations were to be used for comparative purposes only.

The influence of uncertainties in particular parameters can, to some degree, be quantified through an analysis of model sensitivity. Example results are presented in Figure 4.16 for mercury body burdens in coyotes (a top-level terrestrial predator/scavenger) and cobalt-60 body burdens in white sturgeon (a top-level aquatic predator) using data from Segment 7 in 100 simulations of exposure. This figure shows the percentage variation in the estimated body burden that was due to variation in the input parameter set. For both metals, the largest single influence on body burden was pore water concentration, accounting for over 25 percent of the variability in the resulting exposure estimate. Coyote and sturgeon parameter variation accounted for less than 3 percent of the variance in the resulting estimates. Variation in the remaining species parameters accounted for up to 29 percent of the variance in exposure. Site-specific parameter variance accounted for less than 5 percent of the exposure variance, with all contaminant-by-species variation accounting for the remaining 30 to 34 percent.

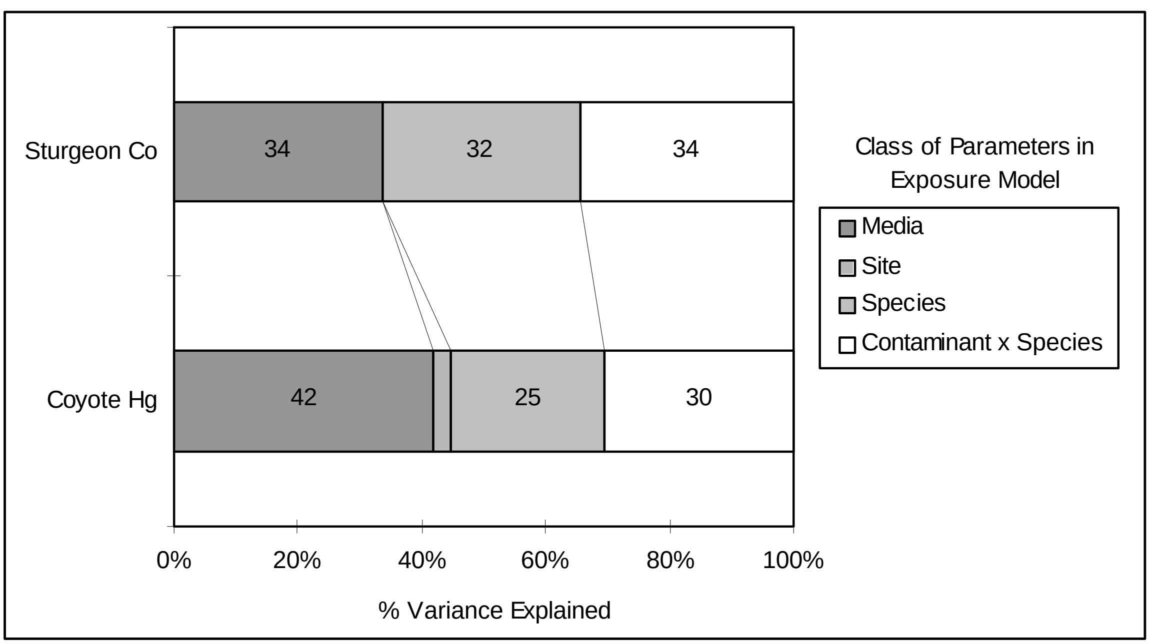


Figure 4.16. Sensitivity of Exposure Estimates to Model Input Parameters

Because the contaminant-by-species parameters comprised a key source of uncertainty in the exposure estimates, this source will be explored further. For a given environmental concentration of metals, including radioactive metals, the expected body burdens for exposed organisms are primarily functions of the relative magnitude of uptake/assimilation rates and loss/depuration rates. Literature-derived values for both types of rates varied widely among the various contaminants, with the most variable rates occurring for zinc, chromium, mercury, uranium, and cobalt (Table 4.30). Uptake/loss parameters varied by three orders of magnitude for these metals. This variation has several causes, including differences in uptake and metabolism of essential versus non-essential metals (Newman and Heagler 1991), differences among metals in their range of bioavailability (Förstner and Wittmann 1981), and differences in the state of the organisms and environments among the various studies reviewed. For example, inorganic mercury is poorly absorbed by animal tissues, whereas methyl mercury is rapidly and almost completely absorbed.

Table 4.30. Average Ranges of Uptake and Metabolism Parameters Obtained from Literature Reviewed and Used in the Exposure Model[(a)] (Ranges are expressed as the reported maximum for the parameter in fresh water fish divided by the reported minimum for fresh water fish.)

	Metal											
Parameter	**Zn**	**Cr**	**Hg**	**U**	**Cs**	**Co**	**Ni**	**Np**	**Sr**	**Cu**	**Pb**	**Tc**
Bioconcentration Factor	127	20	450	8	30	250	4	35	20	4	1	2
Depuration	56	28	3	2	3	23	4	◷	56	◷	3	4
Fractional Assimilation	18	370	1	200	2	60	39	26	8	9	10	◷

◷ = only 1 value available
(a) References for values are provided in Appendix I-D.

Zinc is one of the most problematic metals in the data set because it presents a wide range of uncertainty in uptake (bioconcentration factor) rates as well as the highest geomean bioconcentration factor of all the contaminants. Thus, most of the variance in the estimated body burdens for zinc, as well as the extremely high upper tail of the distributions, is entirely due to uncertainty regarding parameterization for local conditions, not the result of a high level of zinc in the environment. As noted earlier, the concentrations of zinc in the affected portion of the study area are within the range of concentrations found in the upstream sediment. Therefore, less support can be given for the conclusion that the present estimates reflect a potential biotic risk from this metal without obtaining data on tissue levels in biota within the study area and uptake/loss rates for the species evaluated in this study.

The high uncertainties in mercury and uranium are not due to their resemblance to nutrient metals. The uncertainties reflect widely different behaviors of their various forms. For example, mercury occurs primarily in an inorganic form in abiotic media, except where organic matter is relatively abundant. Inorganic mercury is poorly absorbed and readily excreted. The methyl form readily crosses tissue membranes and is depurated extremely slowly. It is also the form most common in biological tissues.

4.2.10.2 Uncertainties in Media Data

One of the primary assumptions with using the media data was that groundwater and/or spring/seep water is an appropriate surrogate for pore water. This assumption's validity may depend on the amount and type of sediment present in the various portions of the study area and on the relative contributions of groundwater versus sediment. In areas of net deposition, such as the sloughs and McNary Pool, sediment concentrations may drive pore water concentrations of metals and other contaminants. In erosional areas, such as the majority of the study area segments, the lack of sediment will enhance the importance of groundwater as the major determinant of pore water contaminant concentrations.

Hope and Peterson (1996) measured aquifer and adjacent pore waters in the 100-D Area for hexavalent chromium content. Their results demonstrated that the mean chromium concentration measured in pore waters was 1.5 to 23 times lower than the values used for surrogated pore water in the exposure model. As noted earlier, the variable driving body burden estimates for both aquatic organisms and terrestrial omnivores/predators were pore water exposure. Thus, the exposure model likely overestimated exposures in all cases. This issue is explored further in Section 4.2.11.

Using a single spatial scale as a frame of reference for describing the distributions of all contaminant data contributed to the uncertainty in the estimated (simulated) body burdens in some segments. Problematic segments comprise areas where the spatial scale of the contaminant plume(s) entering the segment, or the scale of the sediment contamination area, is less than the scale of the segment itself. In such cases, the resulting data distribution will not be unimodal, and the assumption that the data within the entire segment are lognormally distributed will be invalid. Bi- or multi-modal distributions simulated as a unimodal lognormal will produce estimates of concentrations well beyond those observed, or likely to be observed, in the segment.

In other cases, multimodality in the input data can be traced to differences in detection limits used by the various studies that provide data for this study and to changes in sampling strategies implemented during the time period encompassed by the data search (for example, some sampling programs changed the location of sampling efforts from one year to another). The effects on the simulated concentrations are the same as described earlier: geometric standard deviations are inflated, causing body burden distributions from the stochastic simulations to have an artificially high upper tail.

Finally, sampling programs generating the input data were primarily directed at obtaining estimates of contaminant concentrations in areas where such levels are likely to be high. They were not directed at obtaining average estimates or estimates of spatial variability. Consequently, data averages for a segment may exceed reality in those segments where such a sampling regime was used. This is certainly the case for seep/spring data, where sampling focused on locations near known plumes and ignored the remaining seeps in the area. The resulting average estimates of exposure may therefore be overestimated in such cases.

4.2.10.3 Uncertainties in the Exposure Scenarios

One of the assumptions used in the exposure model was that pore water extends into the river in an undiluted form for some distance (10-15 centimeters, more or less) (about 4 to 6 inches). This assumption was considered conservative in the absence of other data to indicate the dilution distance. The study by Hope and Peterson (1996), described earlier, was published after the exposure assumption was made. Their analysis of chromium and electrical conductivity in the aquifer, pore water, and surface water at the 100-D Area was unable to detect evidence of pore water at distances more than 2.5 centimeters (about 1 inch) above the substrate. If this study were valid for the rest of the study area, the exposure estimates for many aquatic and terrestrial biota would be overestimated by a factor of 10 to several thousand, depending on the difference between groundwater and surface water contaminant concentrations in the various segments.

Another assumption in the exposure model was that terrestrial species consumed spring/seep water when it was available or groundwater (pore water) when springs/seeps were unavailable. As described above, the use of groundwater may be untenable if the Hope and Peterson (1996) results are valid throughout the study area. If the assumption is not tenable, exposures of terrestrial organisms from ingestion of surface water are overestimated in areas lacking springs.

For equilibrium modeling, terrestrial and aquatic animals were assumed not to travel out of a modeled segment during the animal's period of residence at the Hanford Site. For example, for exposure modeling, mule deer were assumed to reside 100 percent of the time within the riparian portion of a given segment and not enter the adjacent terrestrial habitat. For some species, this assumption is clearly invalid.

The assumption that a species spends all of its life in one segment or near a contaminant source may be valid for sessile aquatic organisms such as benthic insects and clams. However, mobile aquatic species such as resident and anadromous fish and other aquatic organisms that occur in the drift, for example, zooplankton and insects, will experience highly variable contaminant exposures. Spatial distribution of these mobile species will vary vertically, as well as laterally, within the river. Thus, their behavior will contribute to their exposure variability. Hydrologic conditions are also highly variable across different habitats. Backwater sloughs have limited circulation, nearshore areas typically are low velocity, and midchannel areas have higher velocity. These conditions will cause contaminant concentrations to vary in time and space within a given segment.

Because most of the terrestrial environment surrounding the study area is less burdened with Hanford Site contaminants, the actual risk experienced by most animals with large home ranges will be less than that estimated here. Where significant terrestrial contamination does exist within the terrestrial operable units at the Hanford Site, risk to biota is estimated by the Environmental Restoration Contractor using the assumption that the organisms reside entirely within the contaminated area, as is the case in the present assessment. Hence, no exposure or risk could be greater than those evaluated in the two studies simply by assuming an organism divides its time between a contaminated terrestrial and a contaminated aquatic environment.

One of the primary starting assumptions for this study was that all species evaluated occur throughout the study area and use each segment of the study area in the same manner. Although not true for all species, this assumption was made to demonstrate the risk a species might encounter if it did use a segment of the study area. A risk assessment that only addressed areas where the species was known to occur would miss areas where the species could occur but was not present because of high levels of contaminants. For example, the distributions of salmon spawning within the study area have been mapped from annual aerial surveys conducted by PNNL. Salmon redds are known not to occur below the Richland pumphouse (Segment 21) or in certain other portions of the Hanford Reach. The absence of salmon redds from these locations should not be assumed to be from contamination. Many, if not all, of these areas may not contain suitable nesting habitat, especially Segments 21-27, which are part of the McNary Pool.

In the deterministic analyses of exposure, species were assumed to be located in the most heavily contaminated spot in time and space. For stochastic simulations, individuals of each species were effectively assumed to be located in fixed portions of each river segment where concentrations were held constant, but were simulated throughout and beyond the range of observed data. In other words, the results of the equilibrium model stochastic simulations can be viewed as the estimated distribution of body burdens if some member of the population simulated lived its entire Hanford Site existence in an area represented by the simulated media concentrations. Thus, some individuals will live in the less polluted portions of the segment and others will live in the most polluted portions of the segment. Thus, the model runs should encompass all possibilities of exposure to all levels of environmental contamination currently present within a segment, assuming the media data are representative concentrations. This will result in an overestimate for organisms with large home ranges modeled in higher concentration areas.

As noted earlier, the exposure model used in this assessment is an equilibrium model rather than a dynamic model. Tissue concentrations of an organism's prey have equilibrated with their environments and do not vary. The primary effect of this difference is that the upper tails of the exposure distributions from equilibrium models will be above those from dynamic models for animals that are mobile and can move in and out of contaminated segments. However, dynamic models are not practical for food-web exposure modeling that involves a number of species simultaneously because of the lengthy computational requirements.

4.2.10.4 Uncertainties in the Toxicological Response References

The exposure model for aquatic species estimates equilibrium body burdens through water and food uptake, but toxicological data are generally given in terms of water concentrations producing a given effect. To convert the benchmark into a body burden, the water concentration was multiplied by the bioconcentration factor that was used in the deterministic calculations for the contaminant and species.