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Model of the PCB and Mercury Exposure of Mink and Great Blue Heron Inhabiting the Off-Site Environment Downstream from the U.S. Department of Energy Oak Ridge Reservation

D. L. MacIntosh
G. W. Suter II
F. O. Hoffman

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Environmental Restoration Program
Clinch River Environmental Restoration Program

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EXECUTIVE SUMMARY

This report presents a pair of wildlife exposure models developed for use in investigating the risks to wildlife of releases of mercury and PCBs. The species modeled are the great blue heron and mink. The models may be used to estimate the exposures experienced by mink and herons, to help establish remedial action goals, and to identify research needs. Because mercury and PCBs bioaccumulate through dietary uptake, the models simulate the food webs supporting the two species. Sources of contaminants include surface water, sediment, sediment pore water, and soil. The models are stochastic equilibrium models. Two types of variance in the input parameters are distinguished: stochastic variance among individual mink and herons and ignorance concerning true parameter values. The variance in the output due to stochastic parameters indicates the expected variance among the receptors. The variance due to ignorance indicates the extent to which the model outputs could be improved by additional sampling and measurement. The results of the models were compared to concentrations measured in great blue heron eggs and nestlings from colonies on the Clinch and Tennessee Rivers. The predicted concentrations agreed well with the measured concentrations. In addition, the variances in measured values among individuals was approximately equal to the total stochastic variance predicted by the models.

1. INTRODUCTION

In support of the Resource Conservation and Recovery Act (RCRA) Facility Investigation of the Clinch River, two wildlife toxicant exposure models were developed to investigate the fate of environmental contamination off-site of the U.S. Department of Energy (DOE) Oak Ridge Reservation (ORR). The models focus on two contaminants of primary concern, polychlorinated biphenyls (PCBs) and mercury, identified as important by a screening-level risk assessment for off-site ecological effects (Suter 1990). The mercury model considers both organic mercury in its methylated form and inorganic mercury (Hg^{2+}).

The approach of this investigation was to develop models capable of providing probabilistic statements regarding PCBs and mercury exposure to piscivorous wildlife inhabiting the Watts Bar Reservoir–Clinch River system downstream from the ORR. The modeling effort had three primary objectives. The first objective was to provide a tool capable of projecting responses in PCBs and mercury concentrations in fish and the subsequent dietary exposure to piscivores, following a change in ambient contaminant concentrations. The second objective was to identify parameters that contribute most to prediction uncertainty, in order to provide guidance for future research and data collection efforts. The final objective was to estimate, with a high level of confidence, the safe concentrations of PCBs and mercury in water and sediment to ensure the protection of piscivorous wildlife.

The mink (*Mustela vison*) and the great blue heron (*Ardea herodias*) were chosen as the target species. Both species are primarily piscivorous and at the top of their respective food webs. The mink was selected because it is resident and is highly sensitive to both PCBs and mercury. The great blue heron was selected because it is resident and may be monitored through food, egg, and carcass samples collected from its colonial nesting sites. Furthermore, both mink and great blue heron are societally valued species.

A steady-state compartmental model was selected because effects from PCBs and mercury at the ambient levels measured around the ORR are believed to be a function of cumulative dose rather than a response to acute exposure. Thus, we focused on equilibrium or near-equilibrium dietary doses and whole-body concentrations in mink and great blue heron. A further consideration is that the species of interest are resident and may be expected to reach equilibrium with PCBs and mercury during their lifetimes. The equilibrium assumption was relaxed for invertebrate and fish compartments to account for the short lifespans of invertebrates and forage fish relative to the biological half-life of PCBs and mercury. In reference to the fact that some compartments do not attain steady state, the models are referred to as lifetime models.

The model structure is based on work by Thomann and Connolly (1984) regarding the transfer of PCBs through a five-compartment aquatic food chain. Their structure has been modified to include (1) three contaminant sources in addition to water (sediment, interstitial or pore water, and soil); (2) aquatic plants; and (3) a terrestrial component consisting of invertebrates, waterfowl, and small mammals.

An integral element of this analysis is the propagation of parameter uncertainty through the model. Two types of uncertainty are considered: (1) stochastic variability with respect to the predicted model endpoints (e.g., the distribution of doses and whole-body concentrations among individual herons and minks) and (2) knowledge uncertainty about either the true value or the true distribution of each model input parameter. As a result, probabilistic statements concerning the distribution of PCBs and mercury exposure within a given population of heron or mink are made, as well as statements regarding uncertainty about the true but unknown distributions and associated percentiles.

Details of the model structures and sources of the parameter distributions are documented in Sects. 2 and 3. The results of the modeling effort are presented as the 90% confidence interval (CI) about the expected distribution of

1. chronic dietary dose and
2. corresponding heron egg concentrations

of PCBs and mercury in individual adult females comprising a mink and great blue heron population Also in this section are the mean predictions of the lifetime models compared with those from early versions in which each compartment was assumed to reach total equilibrium with each contaminant. A comparison between model output and observed PCB and mercury concentrations in heron eggs collected from colonies near the ORR is presented in Sect. 4.6. The results of the uncertainty analysis are outlined in Sect. 5, which is followed by conclusions and recommendations for future research.

2. MODEL STRUCTURE AND ASSUMPTIONS

The PCB and mercury wildlife exposure models are probabilistic quasi-equilibrium compartment models, *quasi* in the sense that the fraction of equilibrium attained by a compartment is calculated as a function of lifespan. The model endpoints are the lifetime average chronic dietary dose of PCBs and methylmercury to individual minks and great blue herons feeding from the Clinch River–Watts Bar Reservoir system and the steady-state PCB and total mercury concentrations in individual great blue heron eggs.

The exposure models consider contaminant uptake from three pathways: direct uptake from water for aquatic compartments, ingestion of contaminated food and sediment for all compartments other than the primary producers, and uptake from drinking water for terrestrial compartments.

Values calculated for direct uptake will be referred to as bioconcentration factors (BCFs), where a BCF represents simple partitioning between a biotic compartment and water. Values calculated for contaminant uptake through food will be referred to as biomagnification factors (BMFs), where the BMF for compartment i is the product of a magnification term and the bioaccumulation factor in its prey. Values calculated for total contaminant uptake will be referred to as bioaccumulation factors (BAFs), where a compartment-specific BAF is the sum of the BCF and the BMF for that compartment.

The contaminant concentration in compartment i (C_i) is calculated as the product of BAF_i and the contaminant concentration in column water (C_{H_2O}) (all BCFs and BMFs are normalized to column water ambient concentrations).

Model inputs are ambient contaminant concentrations in column water, sediment, and soil and a variety of contaminant uptake and loss parameters. The contaminant BMF and concentration are calculated for each food web compartment. Additional outputs are the dietary doses of PCBs and methylmercury among mink and great blue heron populations and the PCB and total mercury concentration in individual heron eggs. Methylmercury is the most toxic form of mercury to nearly all animal species (Eisler 1987), and as such, most tests of mercury toxicity are performed with this root form. Thus the mercury model dose output is reported for methylmercury only for ease of comparison with readily available toxicological benchmarks. The methylmercury and Hg^{2+} estimates of egg concentrations are summed for comparison with measured total mercury concentrations in eggs.

2.1 FOOD WEB

The PCB and mercury exposure models each contain 30 food web compartments, 15 representative of aquatic and 15 representative of terrestrial species (Fig. 1). The compartments include 4 primary producers, 1 secondary organic source (detritus), 21 primary/secondary consumers, and 4 secondary or greater consumers.

The compartments were selected to group all mink and great blue heron food web species (Collazo 1985; Hamilton 1959; R. H. Halbrook, ORAU personal communication, 1991) in the Watts Bar Reservoir ecosystem into categories that are reasonably homogeneous

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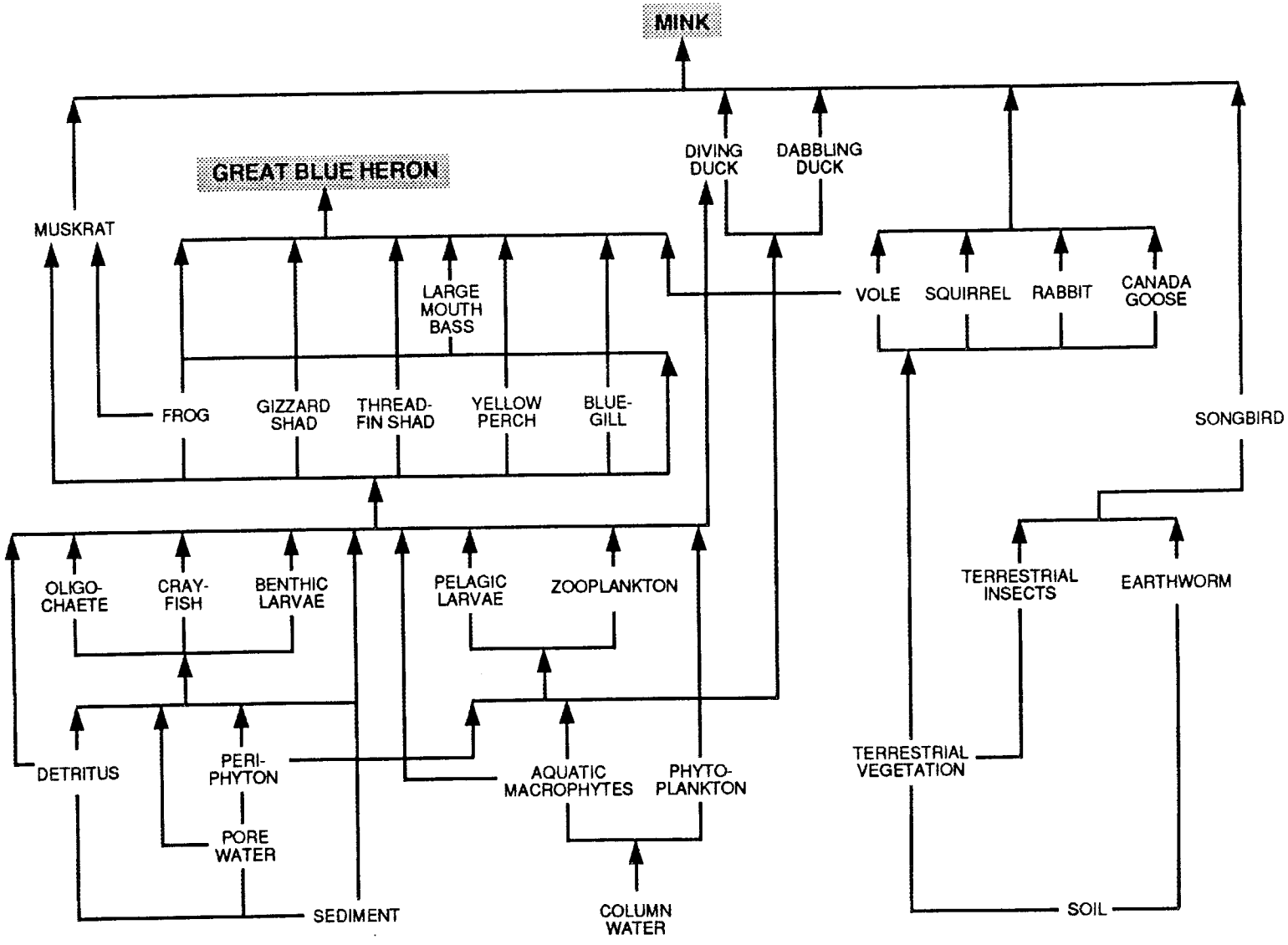

Fig. 1. Mink and great blue heron exposure model food web.

with respect to the properties that determine contaminant concentrations in the food web. The properties used to define compartments were organism anatomy and morphology, primary exposure medium, diet, general life history, and local abundance.

The fish compartments provide an example of the selection rationale. Relative lipid content is used as a morphological criterion because PCBs are highly lipophilic and believed to partition between lipids and water, much as they do between *n*-octanol and water (EPA 1980). Thus, high-lipid fish species are expected to accumulate a greater portion of PCBs than low-lipid species. Diet was considered because benthic organisms are exposed primarily to chemicals in the aqueous phase of sediments (OWRS 1989), an area assumed to have higher concentrations of neutral organics than column water. As a result, benthic feeding planktivorous fish may be exposed to higher dietary concentrations of PCBs than pelagic planktivores. Finally, piscivorous fish are expected to receive a higher dose of biomagnified chemicals than do fish that feed on invertebrates.

Tennessee Valley Authority (TVA) rotenone records for Watts Bar Reservoir cove from 1978 through 1986 indicate that gizzard and threadfin shad are the dominant forage species in the lake (70% and 4% of all fish collected), bluegill is the most abundant primarily planktivorous species (16%), and largemouth bass is one of the most abundant piscivorous species (~0.5%). Yellow perch is not a principal species in Watts Bar Lake (~1%) but was chosen because of its relative leanness and predominately planktivorous, benthic feeding habits (Dugal 1962; Ploskey and Jenkins 1982). Figure 2 summarizes the respective compartment filled by each fish species.

FEEDING REGIME	LIPID CONTENT: Low	LIPID CONTENT: High
Benthic	Yellow perch	Gizzard shad
Pelagic	Bluegill	Threadfin shad
Piscivorous	Largemouth Bass	

Fig. 2. Fish food web compartments.

2.2 PARAMETERS

Each food web compartment is described by a maximum of ten parameters that are divided into two types, toxicant-dependent parameters (TDPs) and toxicant-independent parameters (TIPs) (Table 1). The TDPs are specific to each toxicant-compartment

combination and describe the concentration and kinetics of the contaminant relative to the compartment. The TIPs are specific to each food web compartment and principally describe the dietary habits of the compartment.

Bioconcentration factor. As mentioned, the BCF for a given biotic compartment represents simple partitioning of PCBs, methylmercury, and Hg^{2+} between the compartment and the contaminant in water. The contaminant concentration in a given compartment associated with direct uptake from water is equal to the product of the BCF and the water concentration.

Unlike the other biotic compartments, total plant uptake is represented by simple partitioning alone ($BMF_{plants} = BCF_{plants}$). The aquatic vascular plant and phytoplankton compartments are in equilibrium with the contaminant concentrations in column water. The periphyton and terrestrial vegetation compartments are in equilibrium with the contaminant concentrations in sediment pore water and soil, respectively. The periphyton compartment is designed to solely represent benthic periphyton. Epiphytic periphyton is considered to be the same as phytoplankton.

The species making up the benthic aquatic insect larvae, aquatic oligochaete, and crayfish compartments are assumed to be primarily benthic dwelling. Consequently, the BCFs for these compartments are relative to pore water concentrations. The remaining aquatic invertebrate compartment BCFs are relative to column water concentrations. The earthworm BCF is relative to ambient soil concentrations.

Table 1. Exposure model parameters

TDP	Bioconcentration factor	BCF	mL/g
	Assimilation efficiency	A_{eff}	
	Depuration rate	K_{el}	day^{-1}
	Body:egg coefficient	E_p	$g \cdot g^{-1}$
TIP	Food intake rate	I_p	$g \cdot g^{-1} \cdot day^{-1}$
	Water intake rate	I_w	$g \cdot g^{-1} \cdot day^{-1}$
	Body weight	W	grams
	Percent lipid	L	g lipid$\cdot W^{-1}$
	Dietary fraction	f_{ij}	
	Prey size	S	mm
	Lifespan	t	days

The primary environmental medium for all five fish compartments is assumed to be column water. Many studies have suggested that fish BCFs are positively correlated with lipid content (as referred to in Hamdy and Gooch 1986). Thus, the BCFs for the fish compartments are given on a percentage lipid basis; BCFs are converted to a whole-body factor by multiplying by L, the percentage lipid value.

Assimilation efficiency (A_{eff}). Assimilation efficiency is the fraction of ingested contaminant that is absorbed across the gut lining of an organism (Thomann and Connolly 1984).

Depuration rate (K_{el}). Contaminants that have been fully absorbed are assumed to deposit in storage sites determined by the selective preference of the contaminant (lipids for PCBs and proteins for mercury); depletion from these sites is a function of the contaminant-specific elimination rate. Depuration is assumed to be first-order for each model compartment and contaminant.

Female-to-egg partition coefficient (E_p). E_p is the ratio of whole-body contaminant concentration in an adult female great blue heron to the concentration in the associated eggs. The PCB E_p is expressed on a percentage fat basis for the adult. Thus, whole-body concentrations are converted to fat basis assuming that percentage fat for adult herons ranges uniformly from 10 to 20% (Griminger 1986).

Food intake rate (I_p). I_p is the amount of food ingested by an organism per day, expressed as a percentage of its body weight. I_p times the contaminant concentration in the prey is the contaminant intake rate. The product of I_p, contaminant concentration in prey, and A_{eff} is the net contaminant intake rate.

Water intake rate (I_w). I_w is the amount of water ingested by an organism per day, expressed as a percentage of body weight. I_w is zero for aquatic species and is a function of body weight for birds and mammals. I_w is included to account for toxicant uptake through ingestion of contaminated drinking water.

Body weight (W). W is used as the independent variable in the determination of food and water intake rates for several compartments as well as in the calculation of compartment-specific contaminant concentrations.

Percent lipid (L). L is the lipid or adipose tissue fraction of an organism, expressed as a percentage of body weight. It is used in the determination of direct contaminant uptake from column water for the fish and frog compartments as described in Sect. 2.2.

Dietary fractions (f_{ij}). The dietary fraction for each predator (i) and prey (j) combination in the model represents the fraction of predator i diet consisting of prey j. Thus, for each predator, the sum of dietary factors for all prey is equal to 1.

Prey size (S). S is the length of fish prey for five partially or exclusively piscivorous model compartments: bluegill, yellow perch, largemouth bass, great blue heron, and mink. Prey size is included to prevent implausible model behavior such as a predator consuming prey that is near its own size.

Lifespan (t). Lifespan is the average lifespan of individuals comprising a compartment. It is used to determine the fraction of contaminant equilibrium attained in the invertebrate and fish compartments by the formula

$$\text{Fraction of equilibrium with contaminant } \phi \text{ in compartment i} = 1 - \exp(-K_{el_\phi} * t_i) . \quad (1)$$

Average lifespans in all other compartments are assumed to be great enough for practical equilibrium to be reached, where practical equilibrium is defined as 90% of theoretical

equilibrium or 3.32 biological half-lives (one biological half-life for a given contaminant = $0.693/K_{el}$).

2.3 ALGORITHMS

Bioaccumulation factor. As outlined in the beginning of this section, total uptake of contaminant ϕ by each compartment i (BAF_i) is the sum of the BCF and the BMF for that compartment multiplied by the fraction of equilibrium attained (t = age of organism in days).

$$BAF_i = [BCF_i + BMF_i][1 - \exp(-K_{el_\phi} * t_i)] . \quad (2)$$

Fraction of equilibrium. As mentioned, lifespan is assumed to be a factor for only the invertebrate and fish compartments. For invertebrate compartments, the average age at time of consumption is sampled from an estimated average minimum and maximum, as documented in the following section.

For fish, the age at time of consumption is calculated from length based on a look-up table of maximum average length by age class generated from age and length data reported by Carlander (1969). The following example illustrates how the model performs the age computation.

Assume that the average length of gizzard shad consumed by a given great blue heron throughout its life is 190 mm. On the look-up table (see Table A.5) the length is determined to be greater than that for a 1-year-old (178 mm in length) but less than that for a 2-year-old (240 mm in length) shad. The age (t) of this average gizzard shad is then computed as

$$\begin{aligned} t &= [(178/178) + (190 - 178)/(240 - 178)] * 365 \text{ days} \\ &= 436 \text{ days} . \end{aligned}$$

Biomagnification factor. The BMF is the product of a magnification term (M) and the weighted average BAF among all prey j of predator i.

$$BMF_i = M_i * \Sigma f_{ij} * BAF_j , \quad (3)$$

where

$$M_i = I_{p_i} * A_{eff_i} * K_{el_i}^{-1} . \quad (4)$$

Equations 3 and 4 show that the toxicant-specific BMF for compartment i is a function of (1) the uptake and elimination rates for that toxicant by compartment i and (2) the degree of contamination in the prey of compartment i. A magnification term much less than 1 will result in a BAF for compartment i approximately equal to the BCF_i and virtually no amplification of contaminant concentrations through a food chain. A magnification term much greater than 1 generally will result in $BAF_i > BCF_i$, and depending on the relative magnitude of M_i and BCF_i, significant amplification of contaminant concentrations through a food web may occur (Thomann 1981).

Contaminant concentration. The contaminant concentration in a given compartment (C_i) is calculated as the product of the BAF for that compartment and the ambient column water concentration.

$$C_i = BAF_i * C_{H_2O} \tag{5}$$

Although the primary environmental medium for many compartments is not column water, Eq. 5 reflects the normalization of all BCFs relative to column water concentrations.

Chronic dietary dose. The chronic dietary dose of PCBs and methylmercury to the mink and great blue heron compartments is calculated as the toxicant-specific weighted average BAF among all prey j for each compartment multiplied by the ambient column water concentration.

$$Dose_i = C_{H_2O} * \Sigma f_{ij} * BAF_j . \tag{6}$$

Ambient water criteria (AWC). The algorithm used to determine ambient water criteria (AWC) for protection of mink and great blue heron was adapted from the equation recommended by the U.S. Environmental Protection Agency for generating screening-level wildlife criteria (EPA 1989). AWC was calculated as follows:

$$AWC = \text{toxicity value}_{ep}/BAF_{ep} , \tag{7}$$

where

AWC	= ambient water criteria (mg/L),
toxicity value$_{ep}$	= toxicological benchmark for the endpoint (ppm),
BAF$_{ep}$	= bioaccumulation factor for the endpoint.

3. UNCERTAINTY ANALYSIS AND PARAMETERIZATION

Following recommendations made by International Atomic Energy Agency Safety Series No. 100 (IAEA 1989), this uncertainty analysis distinguishes stochastic variability (uncertainty that is assigned to the natural variability of the environment) from knowledge uncertainty (uncertainty that is the result of incomplete measurement of properties of the environment). This section explains the application of these concepts in the analysis.

3.1 UNCERTAINTY ANALYSIS ASSUMPTIONS AND METHODOLOGY

3.1.1 Stochastic Variability vs Knowledge Uncertainty

The prediction endpoints of the current assessment are characteristics of individual adult minks and great blue herons and of great blue heron eggs. The distributions of individuals are of interest because the effects on populations are determined by the proportion of individuals affected rather than the effects on the average individual and because field measurements consist of measurements of individual organisms. Thus, the TDPs and TIPs specific to these compartments must represent interindividual variability within populations. The variability in parameter values among individuals in a population is assumed to be adequately modeled as a set of stochastic outcomes characterized by normal or lognormal distributions. Thus, we assume the effect of the covariance structure, which is characteristic of interindividual behavior on model predictions, is negligible relative to other uncertainties.

The stochastic parameters for the mink and great blue heron compartments are body weight, daily food consumption, prey size, contaminant elimination rate, percentage body fat, and whole-body:egg transfer coefficient. The lifetime weighted-average contaminant concentrations in ambient media are also assumed to vary among individuals within the endpoint species. This variability represents differential feeding site preferences among herons and spatial distribution of individual mink home ranges.

In addition to stochastic variablility, knowledge uncertainty exists about the true distribution of each stochastic parameter because of the lack of perfect information. This uncertainty is largely the result of interspecies extrapolation from laboratory animals to the wildlife species of interest; extrapolation from PCB congener-specific data to parameters representative of a mixture of congeners such as Aroclor 1254; and a lack of knowledge about PCB and mercury concentrations in column water, sediments, and soil around the ORR.

The knowledge uncertainty about the mink and heron parameters is modeled by varying the means and standard deviations of the stochastic distributions within uniform, loguniform, triangular, and logtriangular distributions. The methodology and rationale employed in determining the parameters of these distributions is discussed in Sect. 3.3.

This parameterization strategy allows for the prediction of the distribution of exposure among individuals within a population (henceforth termed the population distribution) as well as estimation of the effect of knowledge uncertainty on that prediction. In short, one is able to establish levels of confidence about each percentile of the population distribution.

All other parameters are assumed not to vary among individuals within mink and heron populations. For example, it is assumed that a heron that selectively preys on gizzard shad of a given size range will over its lifetime have ingested the average shad of that population (i.e., the average value for shad for each of the relevant model parameters). For any population in a particular time interval, there is only one true mean (and variance, median, mode, etc.). Hence, the relevant gizzard shad parameters show no stochastic variability, only knowledge uncertainty.

Therefore, the parameter distributions for all compartments other than those described in the preceding four paragraphs describe knowledge uncertainty about their true means. As in the case of the stochastic distributions, knowledge uncertainty is characterized by uniform, loguniform, triangular, and logtriangular distributions. For allometric relationships derived by bivariate linear regression, uncertainty about the mean is normally distributed.

The uncertainty analysis was performed using @RISK Version 1.55 (Palisade Corporation, Newfield, New York) added in to Lotus 1-2-3 Release 2.2 (Lotus Development Corporation, Cambridge, Massachusetts). Simulations of stochastic variability were made using Latin Hypercube sampling. Simulations of knowledge uncertainty were made using simple random sampling to establish distribution-free statistical tolerance limits when interpreting the output.

Simulations of stochastic variability and knowledge uncertainty are made alternately under the uncertainty analysis strategy employed. First, one sample is taken from all distributions representing knowledge uncertainty. These include the distributions of the unknown mean and standard deviation of the parameters exhibiting stochastic variability. Second, all the stochastic parameters are sampled repeatedly, holding their means and standard deviations constant and holding the previously sampled value of all the knowledge uncertainty parameters constant. In this way, a distribution of stochastic variability is generated, given a randomly selected set of values drawn from the distributions that represent the knowledge uncertainty.

Simulation of stochastic variability was repeated 59 times for both the PCB and mercury model to determine a 90% confidence interval based on the distribution-free statistical tolerance limit about all quantiles of the model endpoints (IAEA 1989). The confidence intervals established are subjective in the sense that they are a function of the uncertainty determined by the authors' professional judgement for each model input parameter rather than from functions fit to distributions of observed parameter values. Methods used to determine parameter uncertainty are presented in Sect. 3.2.

Individual parameters are ranked with respect to their contribution to uncertainty in model predictions by regressing the predictions against the individual parameters. The resulting r^2 per regression is a measure of the amount of variability in the population distribution explained by variability in the parameter. This ranking provides guidance for further research efforts if the uncertainty in the model predictions is to be reduced efficiently (IAEA 1989).

3.1.2 Sources of Knowledge Uncertainty

Several sources of knowledge uncertainty were identified as relevant to parameterizing the model. The first source, aggregation uncertainty, is associated with the use of the available species-specific data to determine the true mean and variance among all species comprising a food web compartment. Aggregation uncertainty also includes variability in reported data because of life stage, age, and size differences among test animals. For example, the distribution of PCB BCFs for the aquatic insect larvae compartment is based on literature values for only four species, represented by individual organisms at different stages of maturity. These values serve only as indicators of the true mean aquatic insect larvae BCF.

The second source, surrogate species uncertainty, is associated with extrapolation of $compartment_a$ or laboratory animal data to $compartment_b$ parameters. For example, in the absence of species-specific data, the PCB assimilation efficiency distributions for mammalian compartments are skewed toward published values for the Long-Evans laboratory rat.

The third source of uncertainty is variance among mixtures of PCB congeners. It results from using values generated from studies done with specific PCB congeners or mixtures that are not representative of the PCB mixture in the Watts Bar Reservoir-Clinch River system. Variability in the kinetics of PCB congeners may be expected because of the wide range of congener-specific octanol-water partition coefficients, $\log K_{ow} = 4.46$ to 8.18 (Hawker and Connell 1988); surface areas, 195.45×10^{-20} to 304.45×10^{-20} m^2 (Hawker and Connell 1988); and molecular weights, approximately 188 to 494 g/mol. Lutz and Dedrick (1987) demonstrate this variability by outlining the disparity in kinetics between four PCB congeners in the laboratory rat.

The PCB congener profile in many natural systems is characteristic of more highly chlorinated mixtures (Hansen 1987) such as Aroclor 1254 and 1260. This is primarily because of the general tendency for the degree of congener chlorination to be positively correlated with resistance to biodegradation and metabolism (Hansen 1987; Safe et al. 1987; Tanabe et al. 1982). Thus, literature-derived parameter values from experiments performed with Aroclor 1254 or 1260 were considered free of congener diversity uncertainty.

Finally, all other sources of uncertainty regarding application of the published parameter values to the assessment questions addressed by the exposure models were considered. These sources include measurement error, experimental design, laboratory conditions, and differences among reporting methods.

3.2 PARAMETERIZATION

Implied in the following informal decision criteria used for parameterizing the models is the assumption that species for which data are reported in the literature are reasonable estimators of the mean among all the species in the associated model compartment. It was recognized that the validity of this assumption is suspect in some cases, and attempts were made to identify and account for this fact in those situations.

If the range of values of a parameter was determined to be a factor of 10 or greater, the data were log-transformed. Parameter distributions representing knowledge uncertainty for

which at least one piece of species-specific information was found were represented as trianglular distributions, with the mode equal to the mean or point value of the available data. Parameters representing knowledge uncertainty for which no species-specific information was found were represented as uniform distributions. Thus, the distributions reflecting knowledge uncertainty were either uniform, loguniform, triangular, or logtriangular.

Methods used to bound the uncertainty about each parameter were determined by the authors' professional judgement after reviewing available relevant information. For metabolic-based parameters such as ingestion and elimination rate, the variability in weight, respiration, and production among species of a given model compartment was considered relevant. For example, the geometric standard error in weight among 54 zooplankters was considered to be a good indicator of the range of uncertainty about ingestion by the average forage zooplankter. Uncertainty bounds for other parameters for which very few data were found were often determined by examining variability in similar compartments for another contaminant. For example, the only avian Hg^{2+} elimination rate (K_{el}) found was for the domestic chicken. Conversely, six species-specific methylmercury K_{el} values were found. Thus, the variability in methylmercury K_{el} among birds was used as an indicator of uncertainty about the average inorganic mercury K_{el} for the avian compartments, given the single-chicken Hg^{2+} K_{el}.

Parameters for which an adequate amount of data were found ($n \geq \sim 5$) were bounded by the 95% confidence interval about the geometric mean or a factor of 2 about the geometric mean, whichever yielded the greater range. Parameters for which less than three values were found were bounded by the geometric standard error about similar or relevant parameters not included in the model, uncertainty ranges determined for similar model parameters for which more data were available, or at least a factor of 3 about the geometric mean or point value. Parameters for which (1) no data were found and (2) there was intrinsic high uncertainty, such as the Hg^{2+} K_{el} for any invertebrate, were bounded conservatively (a factor of 10 about an estimated central value, for example).

4. RESULTS

Model results for the PCB and methylmercury chronic dietary dose (fresh weight basis) to the mink and great blue heron and respective heron egg concentrations are presented in Figs. 3 through 8. The lines in each figure are complementary cumulative distribution functions, where each ordinate value (Y) represents the fraction of the target population estimated to be exposed to a concentration X greater than the corresponding abscissa concentration x.

The central line in each figure represents the expected distribution generated from the "best estimate" values of the uncertain parameters. This line is our best estimate of the distribution among individuals for each endpoint parameter. The outer lines represent uncertainty about the stochastic distribution as a result of knowledge uncertainty about the input parameters and their distributions. These lines represent bounds on the 5th and 95th fractiles obtained from the family of 59 distributions generated from the two-phase uncertainty analysis methodology.

The uncertainty about the population distribution is, of course, highly dependent on the uncertainty established for the model parameters. The expressed uncertainty assumes that all remaining uncertainties that have not been quantified are negligible (IAEA 1989). Parameters explaining more than 10% of the uncertainty in the model predictions are listed in Table 2.

Table 2. Ranking of parameters contributing to model prediction uncertainty. Values are coefficient of determination (r^2) between column output and row input

Model	Rank	GBH dose	GBH egg concentration	Mink dose
PCB	1	C_{H_2O} (0.81)	C_{H2O} (0.55)	C_{H2O} (0.20)
	2	GShad L (0.14)	GBH K_{el} (0.27)	Phyto BCF(0.16)
	3	C_{sed} (0.09)	< 0.10	< 0.10
Hg	1	C_{H_2O} (0.48)	GBH K_{el} (0.43)	C^{H_2O} (0.28)
	2	C_{sed} (0.150)	E_p (0.40)	Mink size(0.17)
	3	GShad I_p (0.13)	C_{H2O} (0.14)	< 0.10Hg

4.1 MINK EXPOSURE AND SENSITIVITY

PCBs. A dose of 0.64 ppm PCB as Aroclor 1254 in the food of female mink for 160 days prior to whelping resulted in 100% kit mortality 2 weeks after birth (Platonow and Karstad 1973). A combined dose of 0.5 ppm Aroclor 1254 and 0.5 ppm methylmercury in the food of female mink for 8 months resulted in reduced kit survival to weaning (Wren et al. 1987b). Figure 3 shows our best estimate (the central line) of the proportion of the mink population receiving a chronic total PCB dose greater than 0.64 ppm and 0.5 ppm (the lowest observed effect level—LOEL) to be 57% and 52%, respectively.

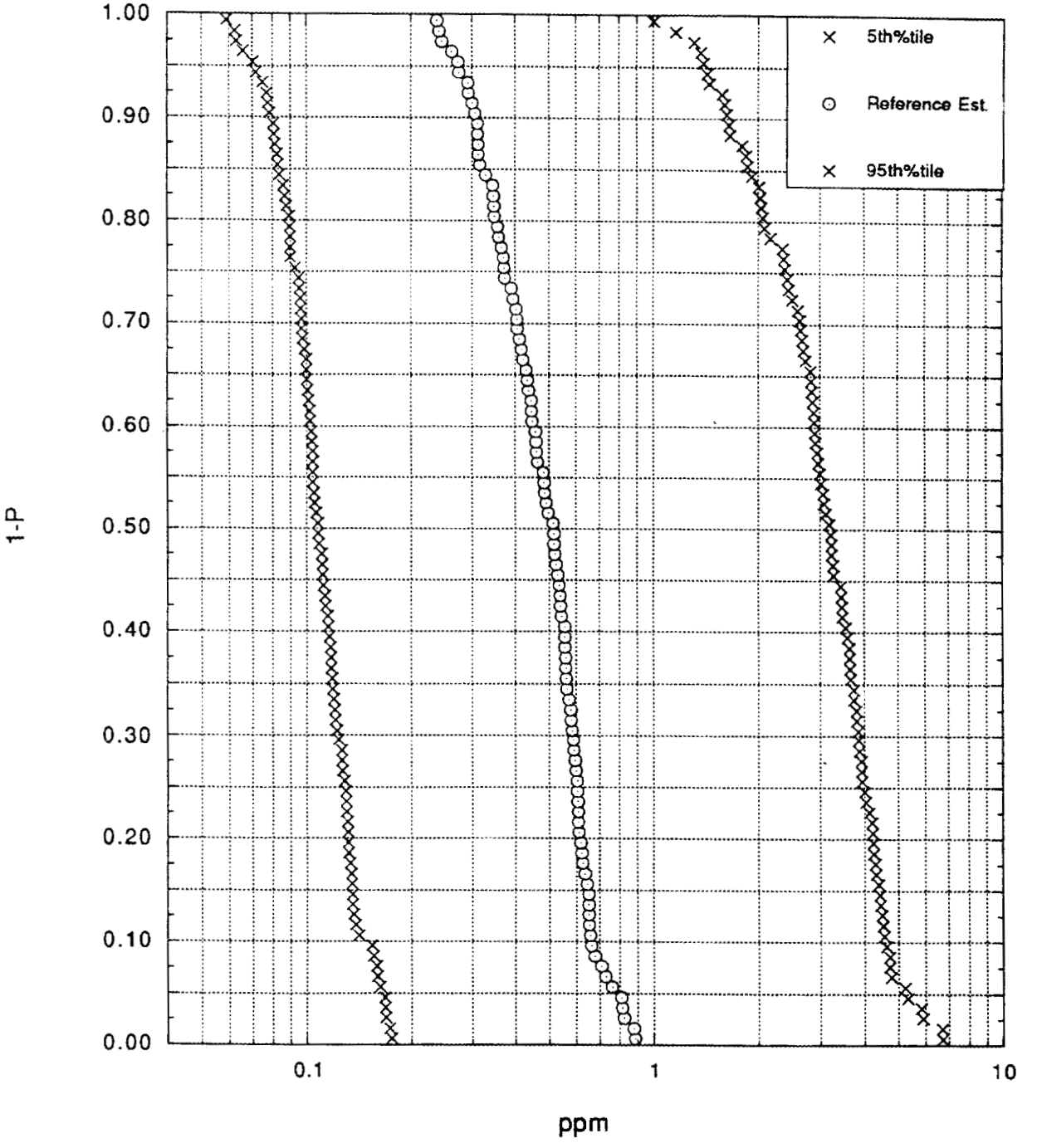

Fig. 3. Complementary cumulative distribution functions for PCBs in diet of mink.

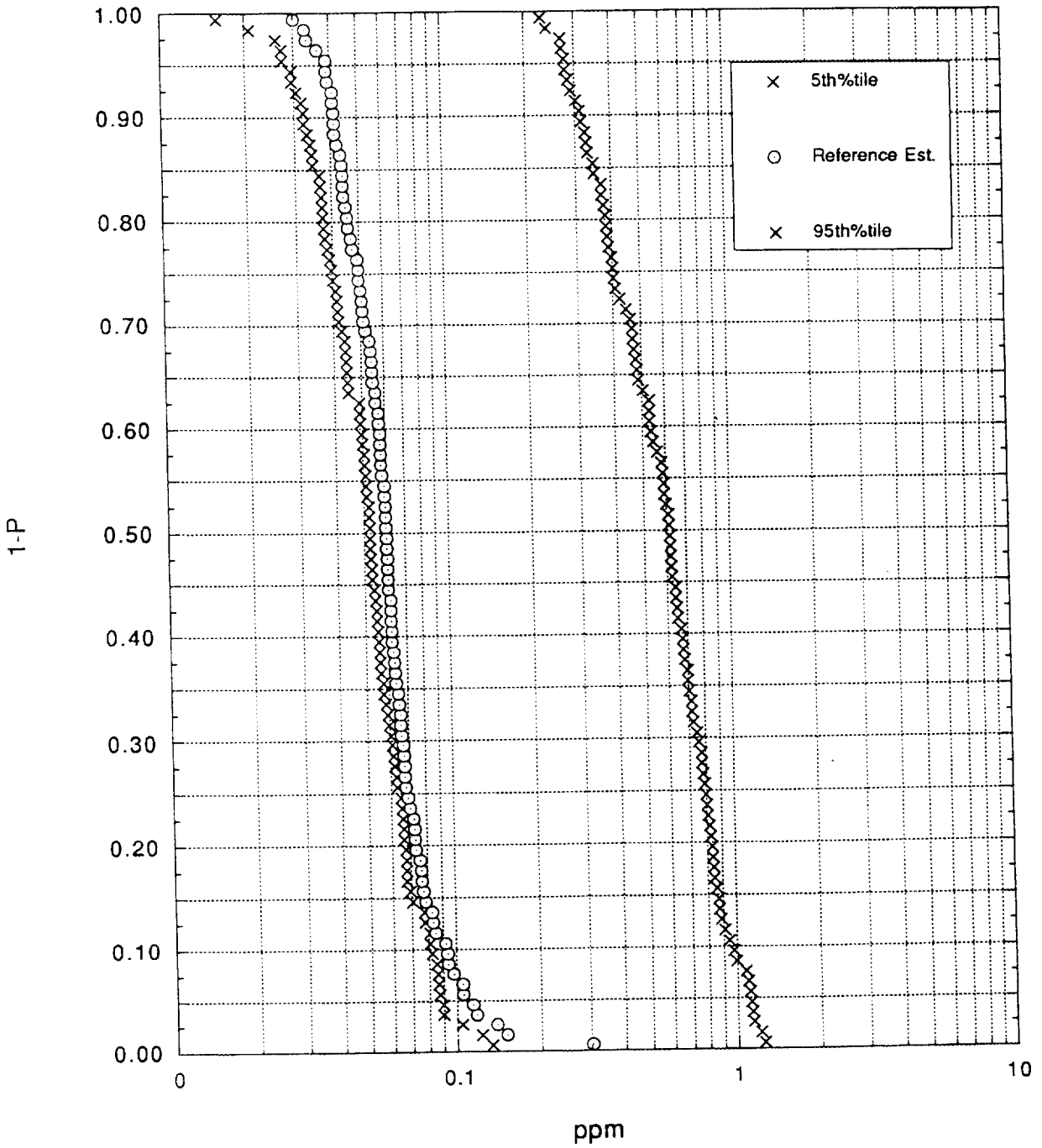

Fig. 4. Complementary cumulative distribution functions for methylmercury in diet of mink.

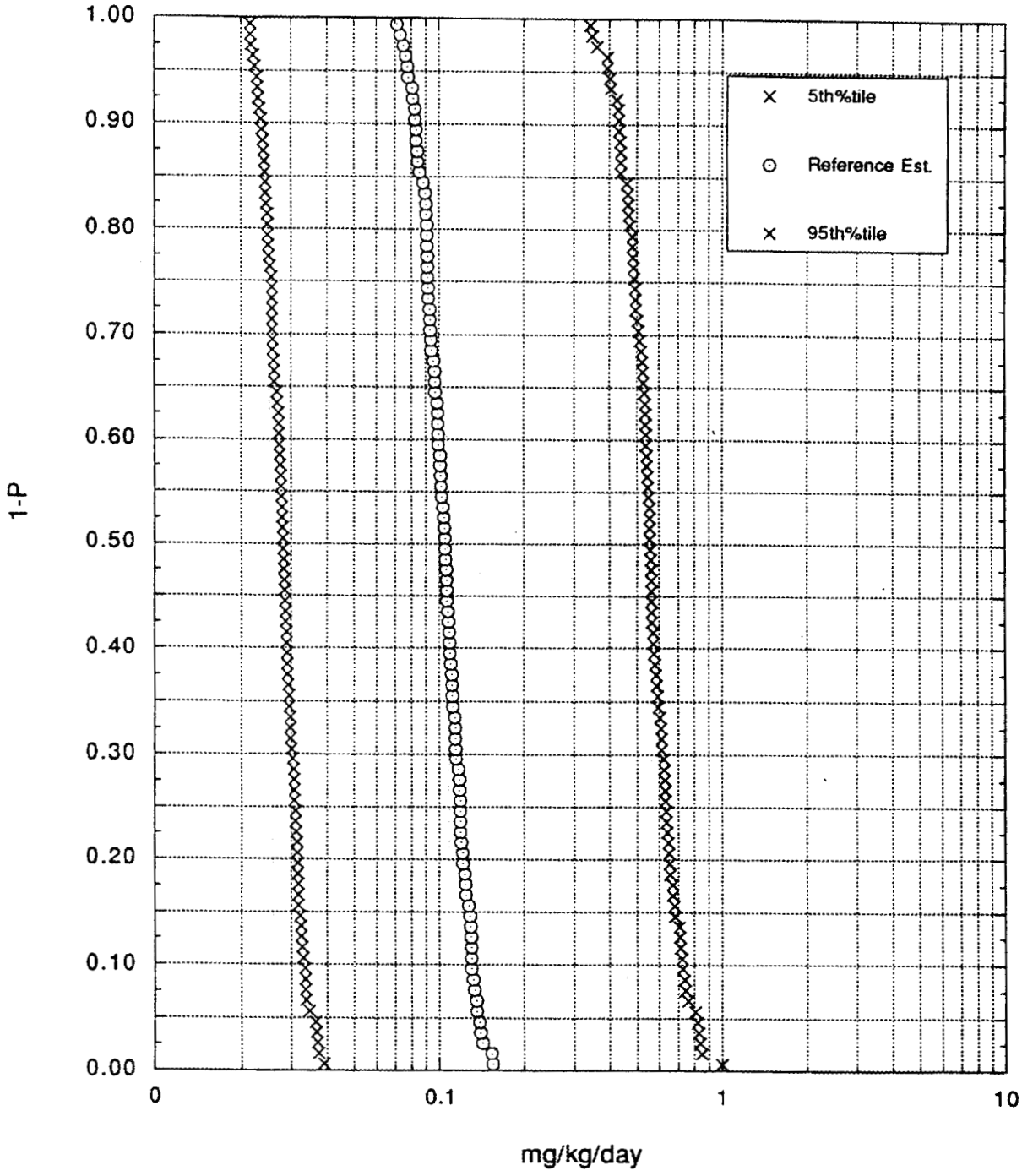

Fig. 5. Complementary cumulative distribution functions for PCBs in diet of great blue heron.

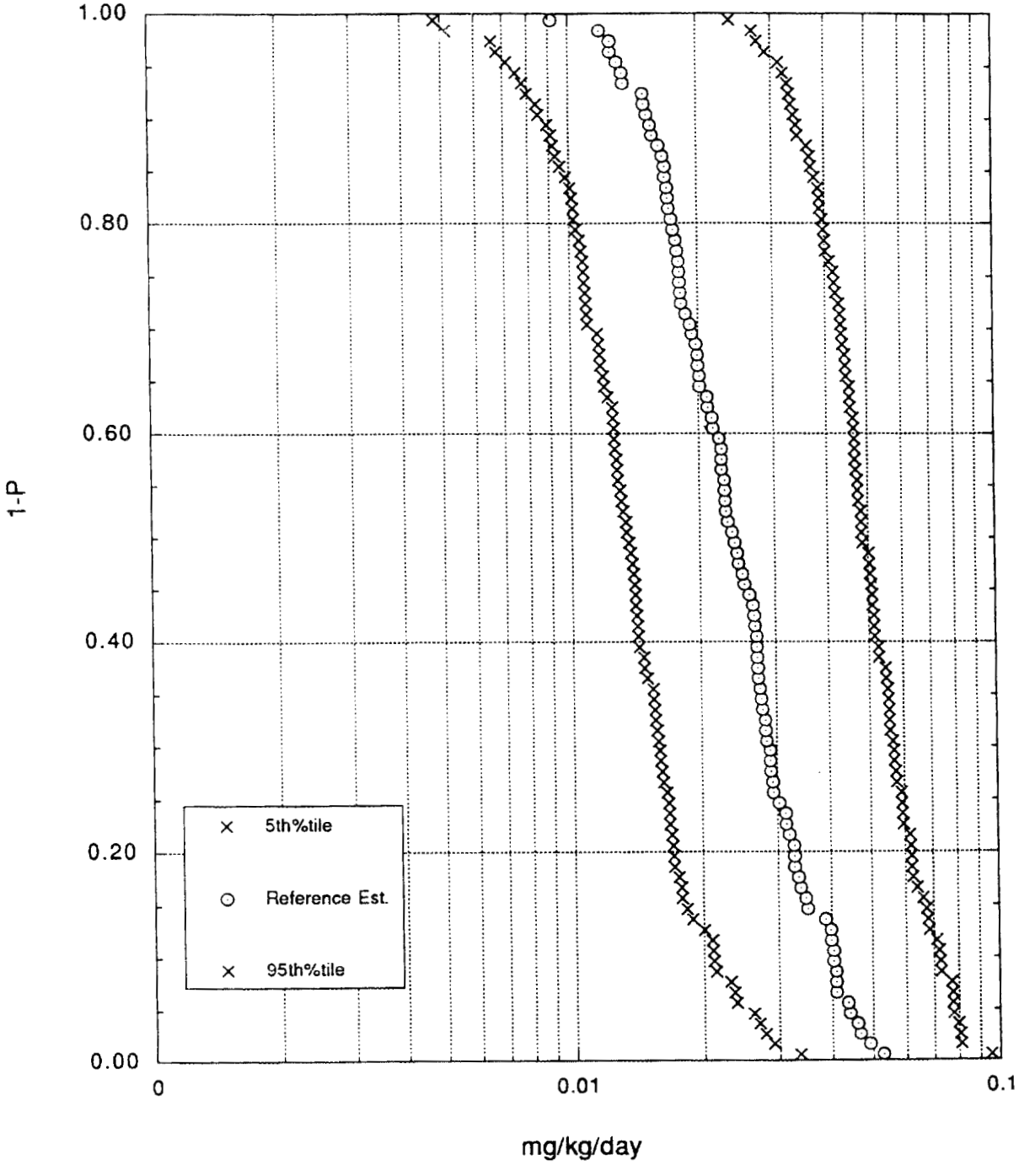

Fig. 6. Complementary cumulative distribution functions for methylmercury in diet of great blue heron.

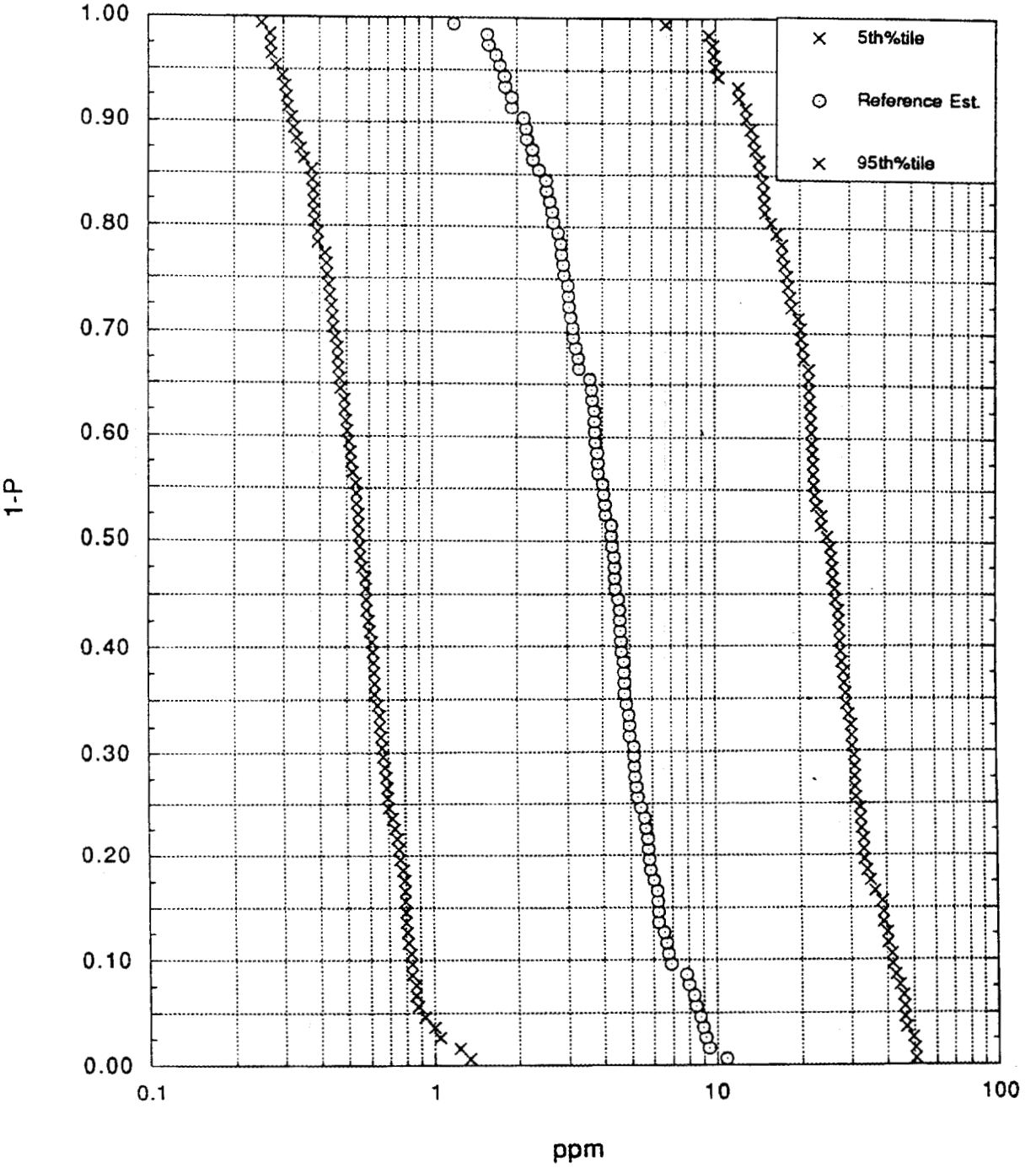

Fig. 7. Complementary cumulative distribution functions for PCBs in great blue heron eggs.

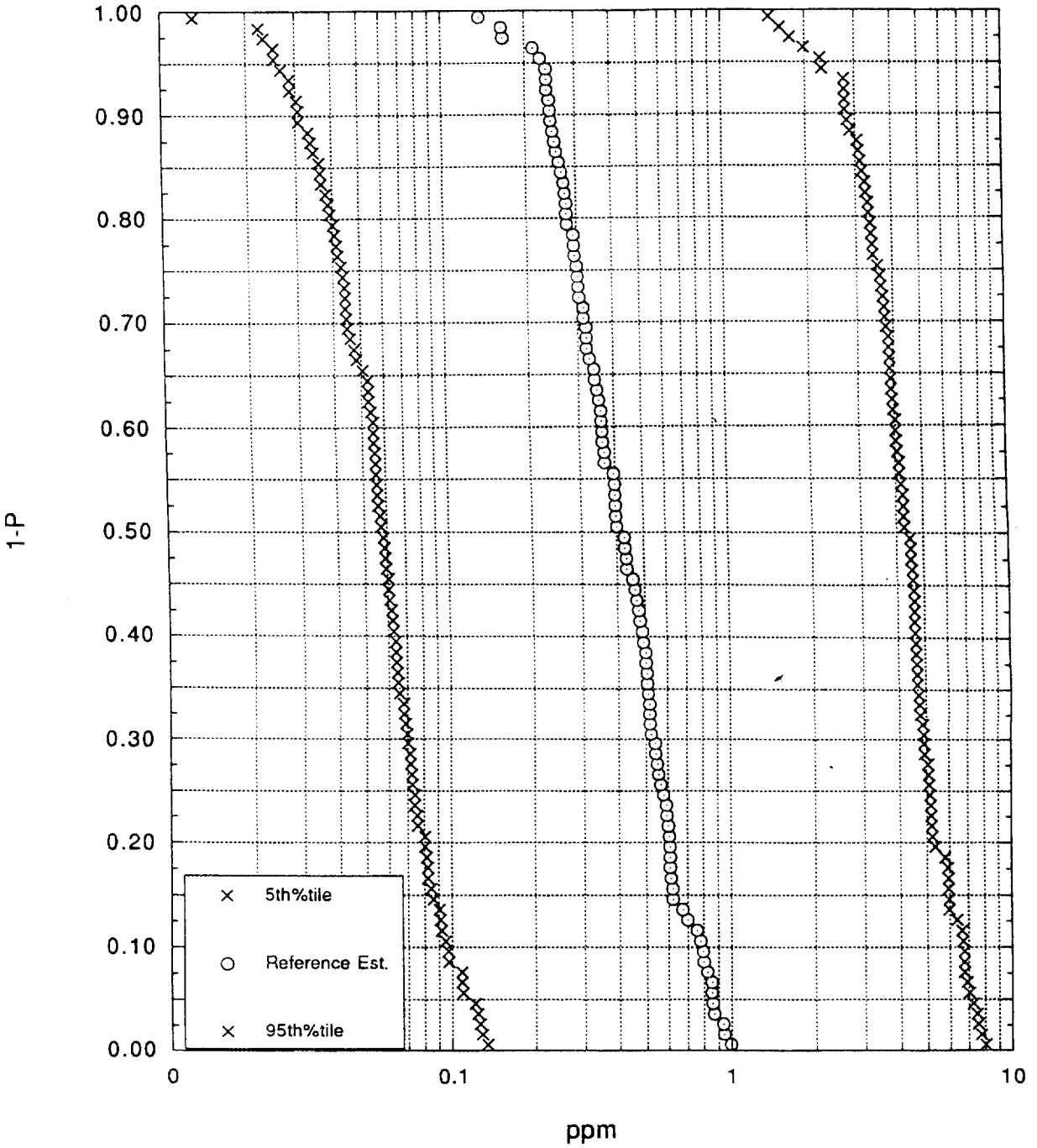

Fig. 8. Complementary cumulative distribution functions for total mercury in great blue heron eggs.

The curves can be further interpreted as follows: although our population distribution for mink indicates that doses to all individuals are expected to fall between 0.2 and 0.9 ppm, we are reasonably (95%) confident that all of the population receives a dose more than 0.06 ppm and less than 7.8 ppm. Parameters contributing most to prediction uncertainty are ambient water concentration, terrestrial vegetation BCF, and phytoplankton BCF (Table 2).

Mercury. One part per million methylmercury administered to adult mink via their food for a period of about 2 months resulted in 100% mortality (Eisler 1987). As stated previously, PCBs and methylmercury appear to behave synergistically in mink, producing reduced fecundity at a dietary level of 0.5 ppm of each.

Figure 4 shows our best estimate is that none of the mink population receives a chronic methylmercury dose greater than 0.5 ppm, the LOEL. In addition, we are 95% confident that no more than 63% of the population receives a chronic dose greater than 0.5 ppm. Similarly, we are 95% confident that no more than 8% of the population receives a dose greater than 1.0 ppm. Parameters contributing most to prediction uncertainty are ambient water concentration and the size of individual mink (Table 2).

4.2 GREAT BLUE HERON EXPOSURE AND SENSITIVITY

PCBs. The total PCB dietary ingestion criterion for protection of birds has been proposed as 3 ppm (Eisler 1986). The criterion was based on the results of an experiment in which screech owls (*Otus asio*) fed 3 ppm of Aroclor 1248 for two breeding seasons experienced no reproductive effects as measured by number of eggs per clutch, hatchability, chick malformations and survival, and shell thickness (McLane and Hughes 1980 as referred to in Eisler 1986). The dietary dose was converted to a body weight basis (0.45 mg/kg per day) by assuming that the owls consumed 15% of their body weight per day in food.

Figure 5 shows that, based on our estimate of the population distribution, the great blue heron population receives a chronic total PCB dose greater than the no observed effect level. However, at a confidence level of 95%, no more than 85% of the heron population is exposed to a chronic PCB dose of >0.45 mg/kg per day and all of the population receives doses of >0.012 mg/kg per day. Parameters contributing most to prediction uncertainty are ambient water concentration; lipid content of its primary prey, gizzard shad; and ambient sediment concentration (Table 2).

Mercury. The mercury dietary ingestion criterion for methylmercyry for protection of birds has been proposed as 0.64 mg/kg per day (Eisler 1987). Three generations of mallard ducks fed a methylmercury dose of 0.64 mg/kg per day laid fewer eggs and produced fewer ducklings than did control groups (Heinz 1979). Additionally, the ducklings produced from the experimental groups were less responsive to tape-recorded maternal calls and were hyperresponsive to a frightening stimulus in avoidance tests (Heinz 1979).

Figure 6 shows that at a 95% level of confidence, none of the great blue heron population receives a chronic dose of methylmercury >0.64 mg/kg per day but all of the population receives doses >0.0047 mg/kg per day. Parameters contributing most to prediction uncertainty are ambient water concentration, ambient sediment concentration, and the gizzard shad food ingestion rate (Table 2).

4.3 EGG CONCENTRATION

PCBs. Embryo weights of black-crowned night heron (*Nycticorax nycticorax*) eggs containing a geometric mean PCB concentration of 4.1 ppm were 15% lower than those of control embryos (Hoffmann et al. 1986). Also, crown-to-rump length and femur length in the field-collected eggs were shorter than those for the control eggs. We use the 4.1 ppm as the LOEL criterion for PCBs in great blue heron eggs.

Figure 7 shows that our best estimate is that 52% of the great blue heron egg population contains PCB concentrations greater than the LOEL. However, we are 95% confident that all eggs have concentrations >0.25 ppm and <50 ppm. Parameters contributing most to prediction uncertainty are ambient water concentration and PCB elimination rate for adult herons (Table 2).

Mercury. The total mercury egg concentration criterion of 0.9 ppm fresh weight has been proposed for protection of birds (Eisler 1986). This value is the LOEL because eggs of mallards (*Anas platyrhynchos*) and ring-necked pheasant (*Phasianus colchicus*) containing 0.9 ppm total mercury showed impaired development (Heinz 1979 and Spann et al. 1972 as cited in Eisler 1987). Furthermore, decreased hatching success was observed in the eggs of white-tailed sea eagles (*Haliaeetus albicilla*), the common loon (*Gavia immer*), and several seed-eating species that had mercury concentrations between 1.3 and 2.0 ppm (Fimreite 1979).

Figure 8 shows that 7% is our best estimate of the great blue heron egg population that contains mercury concentrations >0.9 ppm. At a 95% level of confidence, 100% of the egg population contains concentrations <8 ppm and >0.004 ppm. Parameters contributing most to prediction uncertainty are methylmercury elimination rate for adult herons, female:egg partition coefficient, and total mercury ambient water concentration (Table 2).

4.4 AMBIENT WATER AND SEDIMENT CRITERIA

The AWC for total PCBs and mercury as calculated by Eq. (7) are presented in Figs. 9 and 10. The AWCs are concentrations in water that are estimated to result in doses to herons and mink and concentrations in heron eggs equal to reported toxicological thresholds. PCB AWC were generated from the LOEL of 0.64 ppm in the diet of mink discussed in Sect. 4.1. Mercury AWC were generated from the LOEL of 0.9 ppm in bird eggs discussed in Sect. 4.3.

Recall that the toxicity values used in this analysis are LOELs. These values are anticonservative because effects at the LOEL may be quite high (e.g., 100% welp mortality). However, no observed effect levels (NOELs) were unavailable for wildlife species. In addition, one might wish to allow for differences in sensitivity between tested species and endpoint species or between penned and free-ranging animals. However, those considerations are beyond the scope of this report.

As with the other plots, the cumulative distributions are based on the estimated variance among members of the endpoint populations. The left-most uncertainty band

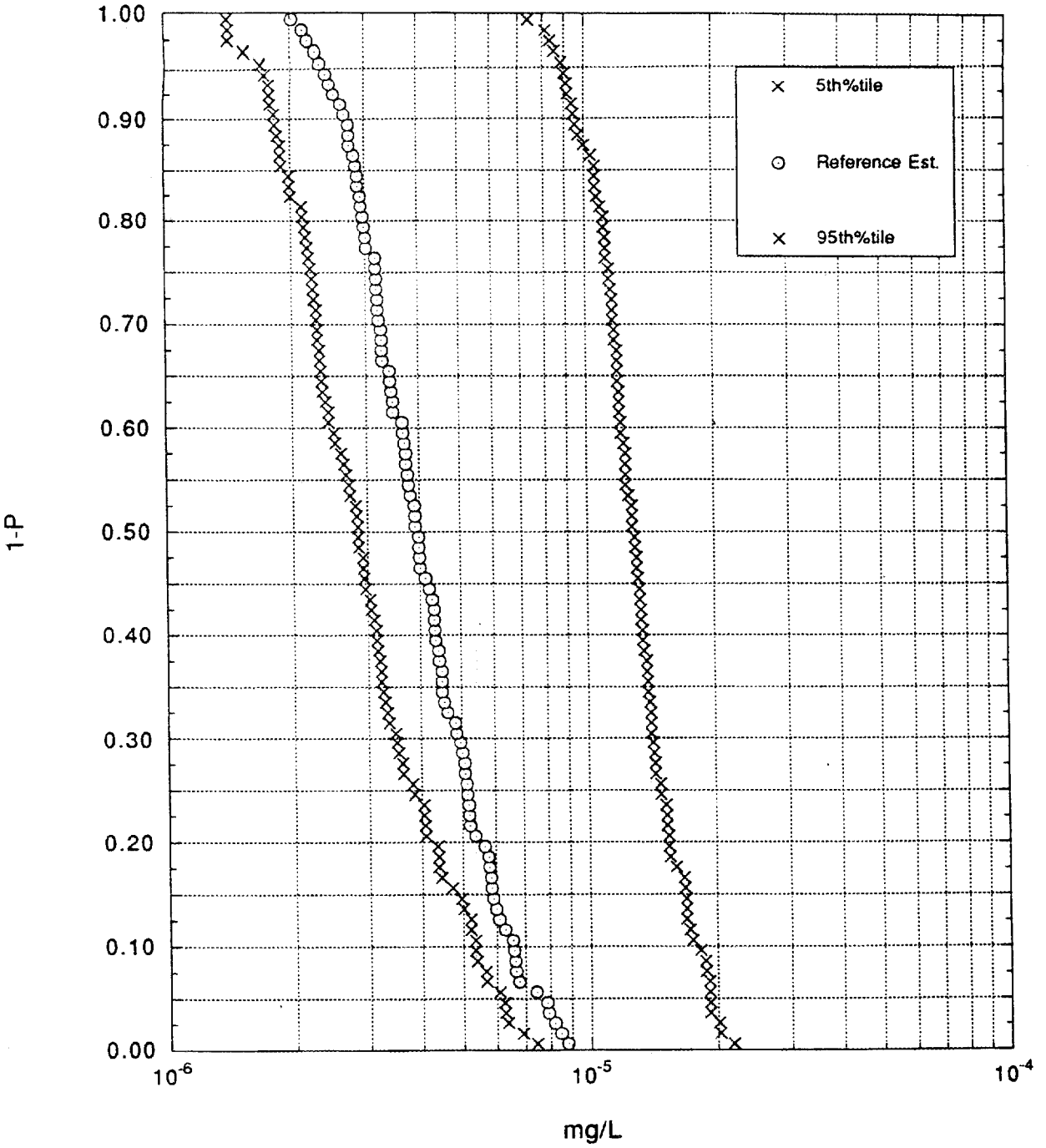

Fig. 9. Complementary cumulative distribution functions for PCB ambient water criteria.

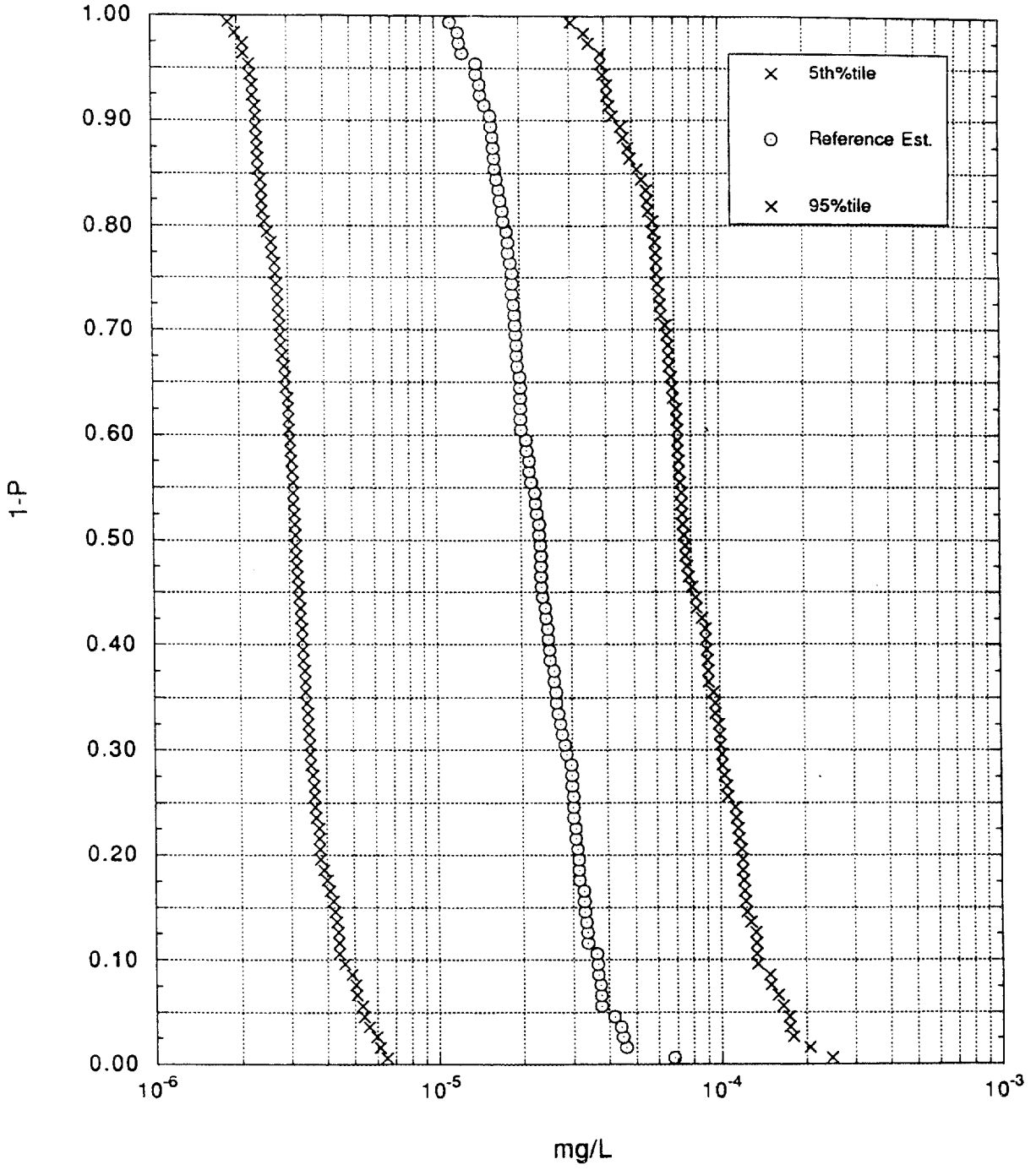

Fig. 10. Complementary cumulative distribution functions for total mercury ambient water criteria.

in Fig. 9 represents water concentrations that we are 95% confident will result in dietary concentrations less than the LOEL for a given fraction of the reference mink population. For example, we are 95% sure that a PCB water concentration of 2.8×10^{-6} mg/L will result in 50% of the mink population receiving dietary concentrations of PCBs less than the LOEL. Alternatively, our best estimate is that the same water concentration will yield dietary concentrations less than the LOEL for only 15% of the population. Likewise, the right-most uncertainty band represents water concentrations that we are 95% confident will result in dietary concentrations greater than the LOEL for a given fraction of the reference mink population.

Figure 10 is interpreted in a similar manner. We are 95% confident that a total mercury concentration in water of $<3.0 \times 10^{-6}$ mg/L will result in concentrations less than the LOEL (0.9 ppm) for 50% of the reference great blue heron egg population. Our best estimate is that at the same water concentration, no eggs will have mercury concentrations greater than the LOEL.

Ambient sediment criteria (ASC) for PCBs and mercury can be determined by assuming equilibrium partitioning between sediment and column water. The average K_{oc} among PCB congeners is approximately 10^5 (Lyman et al. 1982) and K_d for mercury is reported to be 10^5 (Lyman et al. 1982). The equations for estimating sediment concentrations from aqueous concentrations for PCBs and mercury are:

$$C_{sed,\,PCBs} = C_{H_2O,\,PCBs} \times K_{oc} \times f_{oc} \qquad \text{(OWRS 1989)}$$

$$C_{sed,\,Hg} = C_{H_2O,\,Hg} \times K_d$$

Thus, the ASC required to achieve 95% confidence that only the maximally exposed mink and great blue heron eggs will receive a chronic PCB dose or contain a mercury concentration equal to or greater than the LOELs are 1.5 and 0.2 mg/kg, respectively.

4.5 TOTAL EQUILIBRIUM VS LIFETIME MODELS

The mean prediction for five output parameters as calculated by the lifetime models, and similar, yet simpler, total equilibrium versions of the PCB and mercury models are presented in Table 3. PCB lifetime model predictions for the endpoint parameters are ~75% of those from the total equilibrium model. Likewise, mercury lifetime model predictions for the endpoint parameters are approximately 85% of those from the total equilibrium model. Given the similarity of predicted dietary doses and egg concentrations to those that induce toxic effects, this difference in model output is significant and justifies the additional complexity of the lifetime models.

4.6 VALIDATION

Six great blue heron eggs were collected from separate nests from each of two heron nesting colonies during March and April of 1991. One colony, designated K-25, is on the ORR. The other colony, designated Long Island, is on an island in the Tennessee River

approximately 12 km south of the ORR and 7 km above the confluence with the Clinch River. The egg contents were analyzed for total PCB and mercury residue. The distribution of the measured concentrations in eggs from each colony are plotted against model predictions in Figs. 11 and 12 (R.H. Halbrook, unpublished data).

Table 3. Mean estimates of endpoint parameters by the lifetime and total equilibrium exposure models

		Endpoint parameter (ppm)		
Model	Version	Mink dose	Great blue heron dose	Heron egg concentration
PCB	Lifetime	0.49	0.59	4.27
	Total	0.60	0.85	6.08
Hg	Lifetime	0.10	0.28	0.85
	Total	0.12	0.35	1.06

A comparison of the magnitude of the observed and predicted egg concentrations shows that all but one of the observations lie within the 90% confidence interval of prediction values. The 50th percentile of the K-25 and Long Island mercury observations are within factors of 3 and 7 of the predicted reference distribution, respectively. The predicted reference distribution of PCBs in heron eggs is nearly identical to that of the Long Island measured concentrations and within a factor of 5 of the measured K-25 egg distribution. The uncertainty in the source terms of the models (i.e., ambient water and sediment concentrations) was designed to include the true concentrations throughout the Clinch River-Watts Bar Reservoir system.

A comparison of the slope of the measured and predicted values provides insight to the interindividual predictive capability of the model. In general, the model slopes approximate those generated from the measured values. Both predicted and measured distribution slopes are rather steep, which represents relatively little variation among individuals in a population.

Although the initial results are encouraging, it must be kept in mind that the model predictions are based on several simplifying assumptions. Most relevant to the slope comparison is the assumption that individuals in a population are all resident and over their lifetimes forage from approximately the same areas. This assumption may be quite suspect in reality. During the nonbreeding season, individual great blue heron are rather mobile and thus may or may not feed in similarly contaminated waters throughout a year or lifetime. Evidence of this may be found in the two outliers among the Long Island egg mercury concentrations (0.229 and 0.656 ppm) and the lone outlier among the K-25 egg PCB concentrations (2.53 ppm). It must also be kept in mind that the number of observed values is rather small, and therefore the sample distributions may not be representative of the respective population distributions.

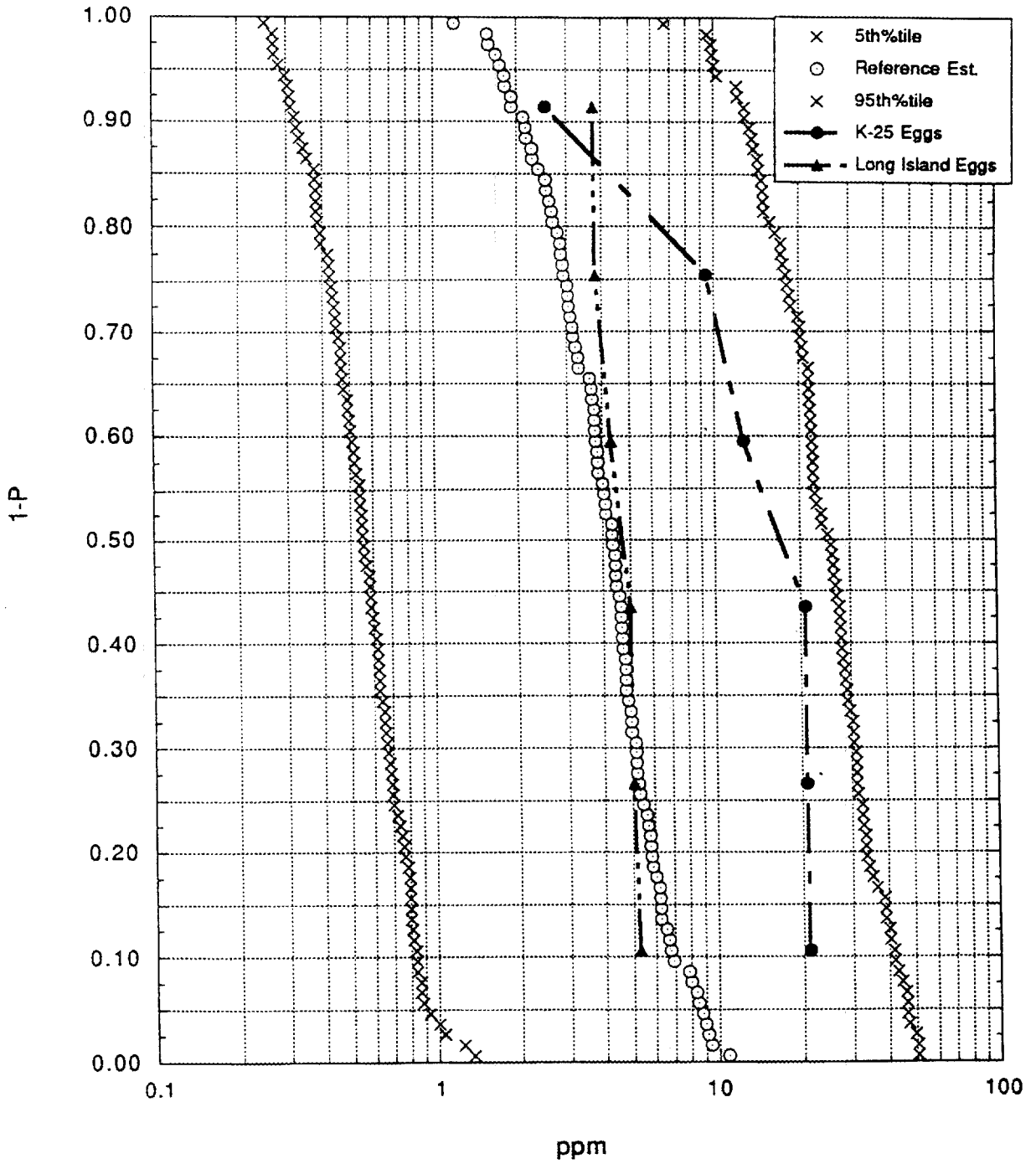

Fig. 11. Observed PCB concentrations in great blue heron eggs vs predicted.

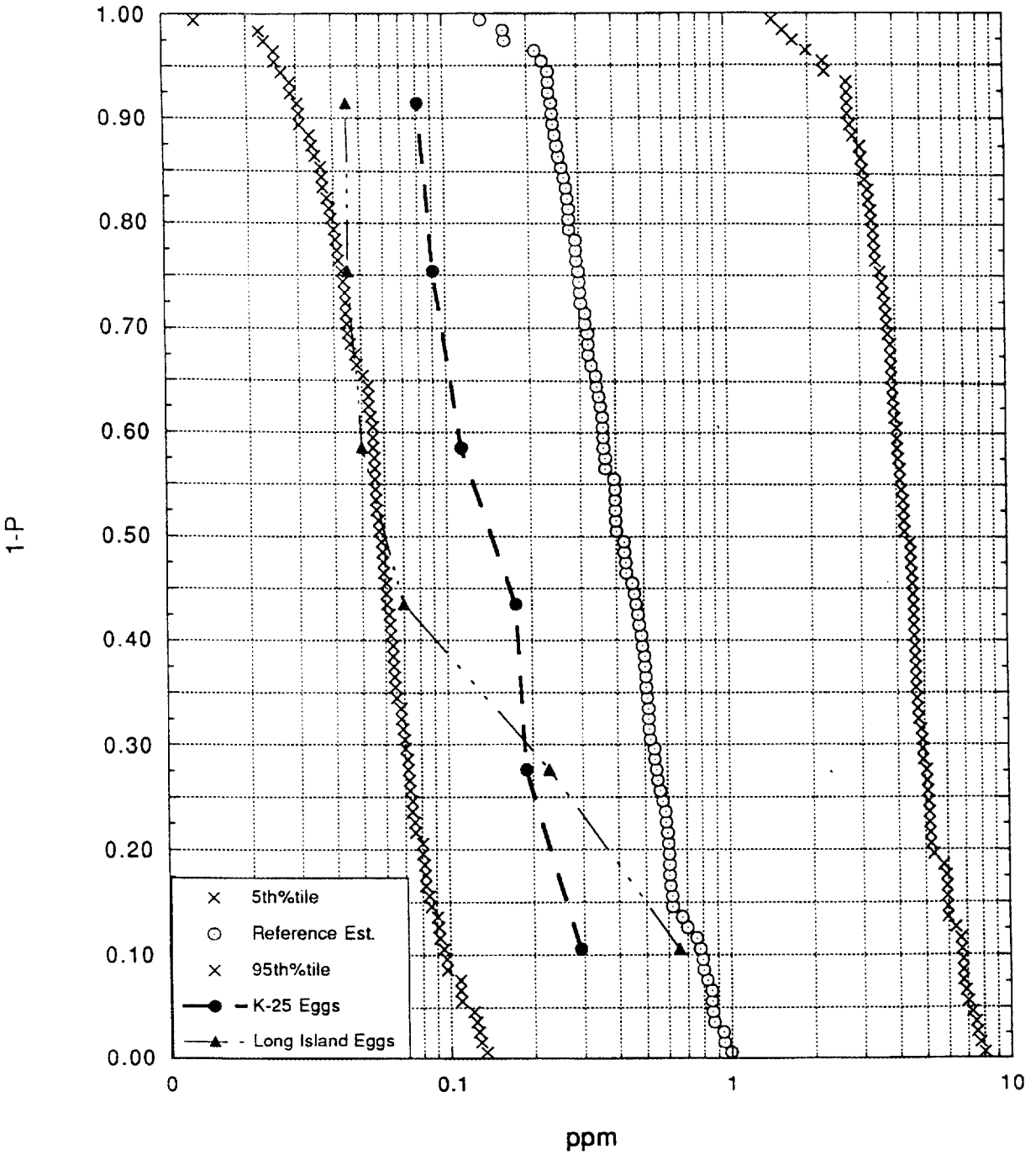

Fig. 12. Observed total mercury concentrations in great blue heron eggs vs predicted.

5. DISCUSSION

5.1 UNCERTAINTY ESTIMATION AND REDUCTION

As shown in Figs. 3 through 8, current model predictions exhibit a wide range of uncertainty. The ratio of the upper 95th fractile of confidence to the best estimate of the 50th percentile among predictions ranges from a low of 1.82 for methylmercury in the great blue heron diet to a high of 8.66 for total mercury in great blue heron eggs. The trend of increasing uncertainty from dose predictions to egg concentration predictions is expected because the latter contain the propagated uncertainty of an additional ten parameters relative to dose estimates.

A second observation is that uncertainty about PCB doses to mink and heron is greater than that about methylmercury doses. This indicates that greater uncertainty exists about the PCB parameters than about the methylmercury parameters for food web compartments. The opposite is true for the predictions of heron egg PCB and total mercury concentrations, which indicates that overall uncertainty about the uptake, loss, and transfer of mercury in the heron compartment is greater than that for PCBs.

The ranking of input parameters in terms of their contribution to endpoint parameter uncertainty is a straightforward and valuable exercise. However, the results of such a ranking do not quantify the benefit-to-cost ratio associated with a reduction in uncertainty about any given input parameter. This is important because, in terms of efficiently reducing overall model uncertainty, the value of additional information (generally in the form of more-specific measurements) depends on the purpose for which the model is being used, model characteristics, and the feasibility and cost of obtaining the information relative to the expected gain in precision. The ranking of parameters as presented in the following paragraphs relates solely to their contribution to prediction uncertainty. Feasibility, cost, and similar attributes were not considered for this report.

Table 2 shows that uncertainty about ambient water concentrations accounts for most of the uncertainty about all model predictions, except for total mercury concentrations in heron eggs. This indicates that dose and egg concentration predictions could be most improved by expending resources to put more-sensitive analytical methods in place to determine ambient media concentrations around the ORR. This result is counterintuitive because measured environmental concentrations are usually specified with greater precision than are the biological parameters of food web transfer. However, this model implementation uses existing water monitoring data rather than data gathered for assessment. Those existing data are sparce, and nearly all specify that concentrations are below detection limits. This ranking is useful in terms of improving the models ability to accurately predict future exposure because of declines in contamination following a remedial action.

The effect of reduced uncertainty about ambient water concentrations on model predictions can be examined by considering the ambient water criteria estimates. Estimates of the 50th percentile for total mercury and PCB AWC are uncertain by a factor of 4.8 and 3.2, respectively. Recall that the mercury AWC was based on heron egg toxicity, and the PCB AWC was based on mink dietary toxicity. Hence, the mercury AWC was based on predictions generated from a greater number of parameters than was the PCB AWC.

Therefore, uncertainty about the mercury AWC represents the worst case (highest uncertainty), given that the distribution of ambient water concentration relative to mink and heron exposure is known. From this it can be inferred that if ambient PCB and mercury water concentrations are known, uncertainty in predictions (represented by the 90% CI about the best estimate) for a given endpoint will be less than or equal to a factor of 5 about the best estimate of the stochastic distribution.

If one is most interested in improving estimates of AWC rather than dietary doses and egg concentrations, the parameters other than ambient water and sediment concentrations that contribute most to prediction uncertainty must be considered. These parameters are discussed in the following paragraphs.

For AWC based on toxicity to heron eggs, a better knowledge of the PCB and mercury elimination rate in adults would most reduce uncertainty in the respective criterion (Table 2). Improved estimates of factors contributing to PCB and mercury concentrations in gizzard shad would most reduce uncertainty about AWC generated from a great blue heron dietary toxicity value.

Finally, the results of the parameter ranking are unclear with regard to the most effective way to reduce uncertainty about AWC based on a mink dietary dose toxicity. It appears that the complexity and diversity of the mink food web, as characterized by this model, results in no single parameter or set of parameters dominating prediction uncertainty. Hence, it is suggested that improved knowledge about the diet of mink (f_{ij}s) would be the most effective way to reduce uncertainty about AWC generated from a mink dietary dose toxicity value. However, it is recognized that the feasibility of significantly improving estimates of mink f_{ij}s is low and that the associated cost is high. Therefore, it is unlikely that investigations of this kind would ever be undertaken. Future consideration will be given to examining the effect of aggregated parameters on prediction uncertainty.

5.2 MODEL COMPLEXITY

The comparison between the lifetime and full equilibrium versions of the exposure models was a test of whether additional model complexity changes and/or enhances model output. The comparison demonstrated that their respective predictions are biologically significantly different. Thus, the additional complexity did have an effect on model output.

Lifespans and age classes were included in the lifetime models to incorporate what were believed to be more realistic assumptions. In the event that the inclusion of more realistic assumptions in any model results in no significant change in output, they are likely to be considered not worth the associated additional complexity and possibly longer run time. For the PCB and mercury exposure models, the consideration of time in the form of lifespan and age classes has the benefit of improved realism and thus more defensible predictions. These benefits outweigh the costs of the associated complexity.

Consideration must then be given to the adoption of further complexity and realism in the form of a dynamic or time-dependent model. The utility of such a model rests primarily in the nature of the assessment question. For example, if one is interested in estimating the time required for contaminant concentrations in biota to reach acceptable levels following a

remedial action, a dynamic model that includes the cycling of those contaminants from biota to abiotic media is essential. Alternatively, if one is interested in the answer to questions such as (1) by what percentage must ambient media concentrations be reduced in order to attain acceptable concentrations in biota or (2) what ambient media concentrations will yield acceptable concentrations in biota, a model based on equilibrium solutions may be adequate. For the assessment questions we have asked, the lifetime models provide satisfactory results.

6. CONCLUSIONS AND RECOMMENDATIONS

The structure, parameters, and initial results of a pair of equilibrium-solution based wildlife exposure models have been presented. The models have been shown to provide probabilistic statements regarding PCB and mercury exposure to mink and great blue heron populations inhabiting the Clinch River-Watts Bar Reservoir system downstream from the ORR. Model output allows for estimates of the fraction of a mink or heron population exposed to a given dietary concentration of contaminant as well as characterization of the uncertainty about that fractile estimate.

Model predictions of PCB and mercury concentrations in the eggs of great blue heron were compared with those measured in eggs collected from two colonies near the ORR. The comparison was favorable, indicating that the models are capable of providing reasonable estimates of actual heron and mink exposure to these contaminants.

Ambient water and sediment criteria for the protection of great blue heron and mink were derived from the distributions of biomagnification factors calculated for each the reference compartments. The results suggest that water and sediment concentrations of ~1 pg/L and 0.1 mg/kg for both PCBs and mercury can be expected (with a high degree of confidence) to yield mink dietary doses and heron egg concentrations below the LOELs for those measures for all but the maximally exposed (top 1%) individuals.

The results of a comparison between the lifetime and full equilibrium versions of the models were presented. It was shown that the respective model outputs are significantly different. Given that the lifespan and age-class considerations of the lifetime models are assumed to be truer to reality, the cost of additional complexity associated with their inclusion was determined to be less than the benefit gained in the form of improved and more defensible results.

Uncertainty about ambient water concentrations was identified as contributing most to uncertainty about dietary dose and egg concentration estimates. Thus, further research efforts to improve this knowledge base would yield the most benefit in terms of reduced uncertainty about these specific model predictions. Estimates of ambient water and sediment criteria are most sensitive to the PCB and mercury elimination rate in adult herons and the diet of mink. Thus, improved knowledge of these parameters would effectively reduce the uncertainty about water and sediment criteria estimates.

Development of a dynamic time-dependent model should be considered as a long-term goal of future exposure modeling efforts in order to better tie model output to observations of individual organisms and to estimate the ecosystem recovery time associated with a remedial action. A model of this sort should include effects of organism growth, age, and range, as well as spatial and temporal changes in ambient concentrations, water temperature, and prey availability. An important additional consideration should be the cycling of contaminants between biota and abiotic media because, in the event that contaminated sediments are removed, the largest remaining contaminant sink is likely to be the living organisms. In light of the uncertainty related to the relatively modest data requirements of the current models, the value of the results of a dynamic assessment model should be considered before committing substantial resources to development of a time-dependent exposure model.

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Appendix

DATA SOURCES AND DISTRIBUTION OF EACH PARAMETER IN PCB AND MERCURY EXPOSURE MODELS

APPENDIX

The following sections document the data sources and distribution of each parameter in the PCB and mercury exposure models. Each paragraph first presents the applicable data found in the literature and ends with the model parameter distribution. Uniform distributions are presented as Unif(minimum value, maximum value), triangular distributions are presented as Triang(minimum, most likely, maximum), normal distributions are presented as Normal(mean, standard deviation), and lognormal distributions are presented as Lognorm(mean, standard deviation). Parameters composed of both variability with respect to the reference unit and knowledge uncertainty are presented as nested combinations of the preceeding notation. For example, Lognormal{Unif(minimum, maximum), Unif(minimum, maximum)} denotes a lognormal distribution with both an uncertain mean and standard deviation.

A.1 AMBIENT CONCENTRATIONS

Ambient concentrations are expressed as

$$\frac{\text{contaminant (g)}}{\text{medium (g)}}$$

A.1.1 Polychlorinated Biphenyls

Total PCB ambient concentrations in the sediment and water of the Clinch River-Watts Bar Reservoir system are below the detection limits of Comprehensive Environmental Response, Compensation, and Liability Act (CERCLA) mandated analytical chemistry methods employed to date. Thus, surface sediment and column water concentrations are primarily reported as <0.1 mg/kg and 0.5 μg/L, respectively (Suter 1990).

The distributions of uncertainty about mean water and surface sediment concentrations listed below were generated by examining observed concentrations in bluegill collected from the local Clinch River system and assuming a BMF of $\sim 1 \times 10^5$. Because of the temporal scale of the models, variability in exposure of great blue heron and mink to contaminated media concentrations was assumed to be relatively small [GSD = Unif(1.05, 1.3)]. Assuming that all herons and mink are resident nearby the ORR throughout their lifespans, individuals will ultimately be exposed to similar environmental contaminant concentrations.

C_{H_2O}: lognorm{loguniform(2×10^{-12}, 2×10^{-11}), uniform(0.05,0.3)*mean}

$C_{sediment}$: lognorm{loguniform(4×10^{-10}, 4×10^{-9}), uniform(0.05,0.3)*mean}

C_{pw}: Pore water concentrations of nonionic organic chemicals can be estimated by assuming equilibrium partitioning between the aqueous phase of sediments and the organic matter fraction of the sediment (OWRS 1989; Suter 1990). The pore water concentration (C_{pw}) is the quotient of the sediment concentration divided by the product of the organic matter/partitioning coefficient (K_{oc}) and the fraction of sediment as organic matter (f_{oc}).

$$C_{pw} = C_{sed}*(K_{oc}*f_{oc})^{-1} .$$

The average PCB log K_{oc} was estimated from the equation

$$\log K_{oc} = -0.557*\log S + 4.277 \quad r^2 = 0.99 \quad \text{(Lyman et al. 1982)}$$

where S is aqueous solubility (mg/L). Log K_{oc} was calculated for fourteen PCB congeners (1 di-, 2 tri-, 7 tetra-, 2 penta-, 1 hexa-, and 1 octa-) for which solubility data were found (Hutzinger et al. 1974) and then averaged to arrive at a normally distributed estimate of the mean log K_{oc} [normal(4.68, 0.342)]. The fraction of sediment as organic matter is estimated to be 1%.

C_{soil}. Ambient surface soil PCB concentrations near Bear Creek on the ORR were found to range from 0.05 to 11.3 ppm, with a geometric mean of 0.51 ppm (R. Turner, ORNL, personal communication, 1991). Because the locations of interest for this project are outside the ORR (a local source of PCBs), it was assumed that soil concentrations downstream are less than those observed on-site by a factor of 5 or 1×10^{-7} g/g.

A.1.2 Mercury

C_{H_2O}. Total mercury ambient water column concentrations in the Clinch River-Watts Bar Reservoir system are below the detection limits of CERCLA-mandated analytical chemistry methods employed to date. Thus, column water concentrations are primarily reported as <0.2 μg/L (Suter 1990). The distribution of uncertainty about mean water and surface sediment concentrations was generated by examining observed concentrations in bluegill collected from the local Clinch River system and assuming a BMF of $\sim 1 \times 10^4$. The total mercury water column concentration distribution is lognorm[loguniform(2×10^{-12}, 2×10^{-11}), uniform(0.05, 0.30*mean}. The portion of total mercury as methylmercury in column water is generally on the order of 10% in the dissolved fraction, although in anoxic systems the methylmercury proportion may be much higher (Saroff 1990; R. Turner personal communication). The fraction of total mercury in the water column as methylmercury distribution is loguniform(0.05, 0.5).

$C_{sediment}$. The mean total mercury concentration in the local Clinch River surface sediments is reported as $\sim 5.7 \times 10^{-6}$ ppm (Suter 1990). Uncertainty about the mean concentration was designated as a loguniform distribution; lognorm{loguniform(2×10^{-6}, 2×10^{-5}), uniform(0.05, 0.3)*mean}. Concentrations of methylmercury in natural nonspiked sediments are generally <1% of total mercury, although the proportion of methylmercury may be slightly higher in freshwater sediments than in marine or estuarine systems (Saroff 1990; R. Turner, ORNL, personal communication, 1991). Thus, the fraction of total mercury in sediments as methylmercury distribution is loguniform(0.001, 0.10).

C_{pw}. No data regarding concentrations of total or speciated mercury in interstitial water were found. Solicitation of expert judgement yielded the assumption that methylmercury and Hg^{2+} pore water concentrations are equal to those in the water column (R. Turner, ORNL, personal communication, 1991).

C_{soil}. Ambient surface soil mercury concentrations on the ORR were found to range from 0.09 to 7.3 ppm, with a geometric mean of 0.75 ppm (Talmage and Walton 1990). In

the absence of expert opinion, the fraction of total mercury in soil as methylmercury was assumed to be similar to that in sediment.

A.2 PCB TOXICANT-DEPENDENT PARAMETERS

A.2.1 Bioconcentration Factors

Aquatic vascular plants. Three values, ranging from 1000 to 1500, were found in the literature for the bioconcentration of 2,5,4′-trichlorobiphenyl in *Potamogeton natans* and *Callitriche sp.* (Crossland et al. 1987). The bioconcentration of nine nonreactive, hydrophobic organic substances, including 2,2′,5,5′-tetra-, 2,2′,4,4′,6,6′-hexa-, 2,2′,3,3′,4,4′,5,5′-octa-, and decachlorobiphenyl, was measured in *Myriophyllum spicatum* and fit to the equation log BCF = 0.98*log K_{ow} - 2.24 (r^2 = 0.97) (Gobas et al. 1991). Log K_{ow} for Aroclor 1254 is assumed to be 6.4 (Hutzinger et al. 1974), yielding a BCF of 10750 in *M. spicatum.* Applying the formula to 2,5,4′-trichlorobiphenyl with log K_{ow} 5.67 (Hawker and Connell 1988) yields a BCF of 2100, suggesting the *M. spicatum* derived relationship may be extrapolated to other macrophyte species. Furthermore, caloric content is reported to range from 4207 to 5140 cal/g among emergent and submergent macrophytes (Jorgensen 1979), indicating relatively little variation in lipid/protein ratios among species of this vegetation type. From this and the preceeding information, the true weighted-average BCF among aquatic vascular macrophytes is assumed to lie within plus or minus a factor of 2 from the Aroclor 1254 based BCF of 10,750. Triangular(5375; 10,750; 21,500).

Periphyton. The mean bioconcentration of Clophen A50 in *Cladophora glomerata* was calculated as 3×10^4 from six reported values (Larsson 1987). Because other periphyton data are lacking, the mean is allowed to range plus or minus three times the literature-derived value. Triangular(10,000; 30,000; 90,000).

Phytoplankton. Mean BCF among three composite samples of Lake Ontario plankton, consisting primarily of phytoplankton, was calculated as 4.5×10^4 from reported water and plankton total PCB concentrations (Oliver and Niimi 1988). Minimum and maximum plankton BCF values for the same data were estimated to be 26,500 and 78,000, respectively. Extrapolation from Lake Ontario plankton to the site in question yields a true site-specific phytoplankton BCF estimate within plus or minus two times the Lake Ontario mean. Triangular(22,500; 45,000; 90,000).

Zooplankton. Two values (*Daphnia magna* = 3800, *Gammarus pseudolimnaeus* = 6200) for bioconcentration of Aroclor 1254 in zooplankton were found in the literature (Mayer et al. 1977). Assuming lipid-based partitioning between PCBs in water and zooplankton, the geomentric standard error of 1.52 about the mean dry weight among 54 zooplankton species (Jorgensen 1979) was used as a proxy for variability about the average BCF between the daphnid and scud of 5000. Triangular(2000; 5000; 12,500).

Aquatic insect larvae. Four values (stonefly naiad = 740, dobsonfly larvae = 1500, phantom midge larvae = 2700, mosquito larvae = 3500) for the bioconcentration of Aroclor 1254 in aquatic insect larvae (geometric mean = 1800) were found in the literature (Mayer et al. 1977). Uncertainty about the true weighted mean determined as plus or minus two

times the calculated mean based on zooplankton uncertainty and guidelines discussed in Sect. 3.2. Triangular(900, 1800, 4000).

Aquatic oligochaete. Two species were reported to have average lipid content of 1% of whole-body wet weight (Oliver 1987, 1984). These studies assumed lipid-based partitioning between PCBs in pore water and benthic worms and used the PCB lipid/water ratio determined from available fish data (see below). Lognormal(10,400; 4400) * 1%.

Crayfish. One value (750) for the bioconcentration of Aroclor 1254 in crayfish was found in the literature. Uncertainty was assigned to be plus or minus three times the reported value based on guidelines discussed in Sect. 3.2. Triangular(250, 750, 2250).

Fish. The BCF for each fish compartment is assumed to be the same on a percentage lipid basis. Five values for the bioconcentration of Aroclor 1016 and 1254 in fish on a percentage lipid basis were found in the literature (EPA 1980) with a geometric mean and standard error of 10,400 and 1.42, respectively. Lognormal(10,400; 4400) * $\%lipid_i$.

Frog. No data were found for the bioconcentration of PCBs in amphibia. Frogs respire through the skin (Duellman and Trueb 1986) and are assumed to bioconcentrate PCBs and and mercury from water through this process, much like fish do through their gills. Thus, the frog BCF is analogous to and based on the fish distribution. Lognormal(10,400, 4400) * $\%lipid_{frog}$.

Terrestrial vegetation. The bioconcentration of Aroclor 1254 from soil in eight plant types was found to range from 0.004 to 0.05, with a geometric mean of 0.009 and standard error of 1.39 (Strek and Weber 1980). Lognormal(0.009, 0.0035).

Detritus. PCB concentrations in detritus are assumed to be determined by partitioning between the pore water PCBs and the organic matter in the detritus according to the following equation.

$$C_{detritus} = C_{pw} * K_{oc} * f_{oc}$$

and

$$BCF_{detritus} = C_{detritus}/C_{pw} = K_{oc} * f_{oc} .$$

Carbon content data were available in the literature and assumed to be representative of that in detritus. The fraction of carbon among 19 aquatic macrophyte species on a dry weight basis was determined to be approximately normally distributed with a mean of 0.31 and standard error of 0.0072 (Jorgensen 1979). Macrophytes are assumed to be 80% water. The resulting uncertainty about the bioconcentration of PCBs in detritus is expressed as normal(0.31,0.0072) * 0.2 * K_{oc}.

A.2.2 Assimilation Efficiency

Zooplankton. One A_{eff} value (0.2 for *Acartia tonsia*) was found in the literature (Thomann and Connolly 1984). Distribution bounds are based on other aquatic invertebrate A_{eff} data. Logtriangular(0.05, 0.2, 0.8).

Aquatic insect larvae. Two values (0.7 and 0.79) were found for the assimilation of 2,4,5,2′,4′,5′-hexachlorobiphenyl by *Chironomus tentans* larvae (Muir et al. 1983). Distribution bounds are based on other aquatic invertebrate A_{eff} data. Triangular(0.2, 0.7, 0.9).

Aquatic oligochaete. One value was found for the assimilation of 2,4,5,2′,4′,5′-hexachlorobiphenyl from each of four sediment particle size groups with a mean of 0.08 (Klump et al. 1987). Distribution bounds are minimum and maximum reported values. Triangular(0.03, 0.08, 0.25).

Crayfish and terrestrial insect. No A_{eff} data were found for these three compartments. Distribution is based on other aquatic invertebrate data. Loguniform(0.08, 0.8).

Earthworm. Distribution is based on aquatic oligochaete data expanded by a factor of 2 for extrapolation among compartments. Loguniform(0.03, 0.5).

Fish. The mean A_{eff} of 2,5,2′,5′-tetrachlorobiphenyl in rainbow trout and brook trout is reported as 0.8 and 0.72, respectively (McKim et al. 1983). Assimilation of the same congener was later reported to have an approximate mean of 0.55 among another cohort of rainbow trout (McKim et al. 1985). The average A_{eff} among 31 different congeners by rainbow trout is reported to be 0.75 (Niimi and Oliver 1983). PCB assimilation efficiency is reported to be 0.8 in yellow perch (Nortsrom et al. 1976), 0.67 to 0.93 in carp (Tanabe et al. 1982) and 0.65 to 0.8 in lake trout (Thomann and Connolly 1984). This sample of reported values is approximately normally distributed with mean 0.74 and standard error 0.039. Normal(0.74, 0.039).

Frog. No A_{eff} data were found for amphibia. Distribution boundaries were set to the bounds of the 99% CI on the mean A_{eff} for the fish compartments because of the gastrointestinal tract morphology and diet of fish and frogs (Storer et al. 1957). Uniform(0.61, 0.87).

Birds. The assimilation efficiency of 7 Aroclor mixtures from food by chickens was found to range from 0.892 to 0.962, with a geometric mean of 0.92 (Fries et al. 1977). Consistency of chicken data led to lower bound assumption that A_{eff} for all birds is >0.75. Triangular(0.75, 0.92, 0.99).

Mammals. A_{eff} of 20 congeners administered to male CD strain rats at three dose levels was found to range from 0.9 to 0.99, with a geometric mean of 0.96 (Albro and Fishbein 1972). Consistency of the rat data led to an assumption bound that A_{eff} for all mammals is >0.75. Triangular(0.75, 0.96, 0.99).

Great blue heron and mink. Interindividual variability in A_{eff} was assumed to be negligible.

A.2.3 Elimination Rate

Aquatic insect larvae. The first-order elimination rate (per day) of 2,4,5,2′,4′,5′-hexachlorobiphenyl by *Chironomus tentans* larvae has been reported to be 0.097 (Muir et al.

1983). The K_{el} geometric mean of 11 tetra- and pentachlorobiphenyls is reported to be 0.73 for the same species (Novak et al. 1990).

Variability in PCB K_{el} among species is assumed to be related to variability in respiration rate among species. The geometric standard deviation (GSD) and error (GSE) for the respiration rate (mg $O_2 \cdot mg^{-1} \cdot day^{-1}$) of 15 aquatic insect species is 25 and 2.1, respectively (Jorgensen 1979).

The lack of Aroclor-specific K_{el} data and the high variability in respiration rates led to a range about the true mean for this compartment of 100×, approximately centered on the mean between the two available data points. Loguniform(0.01, 1.00).

Zooplankton. 2,4,5,2′,4′,5′-hexachlorobiphenyl elimination in *Mysis relicta* and *Pontoporeia hoyi* is reported to be 0.003 and 0.019 per day, respectively (Evans and Landrum 1989).

The assumption was again made that PCB K_{el} variability among species is related to that for respiration rate. The GSD and GSE for respiration rate (mg $O_2 \cdot mg^{-1} \cdot day^{-1}$) among 14 zooplankton families and species were calculated to be 6.7 and 1.34, respectively (Jorgensen 1979).

Again, the lack of Aroclor specific K_{el} data and the high variability in respiration rates led to a range about the true mean for this compartment of 100×, approximately centered on the mean between the two available data points. Loguniform(0.001, 0.1).

Aquatic oligochaete. Mean K_{el} among 12 congeners (representative of Aroclor 1254 and 1260) in at least two species of *Tubifex* worms in 8°C water is reported to be 0.0099 per day (Oliver 1987). Because average annual temperatures in East Tennessee are warmer than 8°C, the upper uncertainty bound was set at three times the reported mean and the lower bound at 0.5 the reported mean. Triangular(0.005, 0.01, 0.03).

Other invertebrates. The range is bounded by minimum and maximum K_{el} values among the above invertebrate compartments. Loguniform(0.001, 1.00).

Fish. Elimination of total PCBs by fish has been reported to be related to fish size as described in the following equation (Thomann 1978):

$$K_{el} = 80 * L^{-0.8},$$

where L = fish length in micrometers. The elimination of 31 PCB congeners reflecting the approximate mix of congeners in Aroclor 1254 and 1260 was measured in mature rainbow trout (estimated length = 400 mm) averaging 900 g in fresh weight (Niimi and Oliver 1983). The geometric mean and standard error of elimination rates among the 31 congeners were calculated to be 0.002 and 1.73 per day, respectively. The mean and upper and lower 95% CI values of distribution of the mean correspond to exponent values in the Thomann equation of -0.82, -0.77, and -0.86, respectively. Thus, the uncertainty about the average K_{el} for a fish of a given size is expressed as:

$$K_{el} = 80 * L^{\text{Triangular}(-0.86,\ -0.82,\ -0.77)}$$

Birds other than great blue heron. Aroclor 1254 elimination rate has been calculated to be 0.006 from reported quail and pigeon adipose tissue retention data (Bailey and Bunyan 1972). The ratio of the maximum to minimum methylmercury elimination rate among six bird species is 2.5. From this information, it was assumed that PCB K_{el} for any of the six bird compartments is within plus or minus three times the quail and pigeon rate. Logtriangular(0.002, 0.006, 0.018).

Mammals other than mink. Geometric mean K_{el} of three congeners among a laboratory dog, mouse, and rat was calculated to be 0.079 per day from reported tissue retention data (Lutz 1987). The lack of Aroclor-specific data and the interspecies extrapolation led to a range of 100× centered on the mean of the available data. Loguniform(0.0079, 0.079, 0.79).

Great blue heron. Lognorm{logtriang(0.002,0.006,0.018), uniform(0.2, 0.5)*mean}. The magnitude and uncertainty about variability among individual herons is based on a maximum:minimum ratio in heron weights of 1.33 (Bull and Farrand 1977; Kushlan 1978).

Mink. Mean Aroclor 1254 K_{el} was reported to be 0.007 per day among individual adult mink (Hornshaw et al. 1983). Lower bound is assumed by the authors to be an overestimate because the study was terminated before second-order elimination could occur. Therefore, the average K_{el} for mink is assumed to lie between 0.0007 and 0.007 per day. The distribution of uncertainty about the standard deviation among mink was based on variability in weight among adult mink (EPA 1987). Lognorm{loguniform(0.0007,0.007), uniform(0.06,0.1)*mean}.

A.2.4 Great Blue Heron Whole Body:Egg Partition Coefficient

The PCB content of chicken eggs relative to laying hen adipose tissue concentrations was reported to be 0.084 (Fries et al. 1977). The mean partitioning coefficient for herons was assumed to lie within plus or minus two times the chicken mean. GSD among individual eggs was assumed to range from 1.1 to 1.25.

A.3 METHYLMERCURY TOXICANT-DEPENDENT PARAMETERS

A.3.1 Bioconcentration Factors

Aquatic vascular plants. Bioconcentration of methylmercury from water in the submergent and emergent portions of six species of aquatic vascular plants is reported to range from 34 to 3500 and 6 to 32, respectively (ESB 1978). Emergent—loguniform(6, 32); Submergent—loguniform (34, 3500).

Phytoplankton and periphyton. The methylmercury BCF for *Scedesmus obliqus* and *Microcystis incerta* is reported to range from 761 to 2100 and 461 to 990, respectively (EPA 1985). The geometric mean of the available data was calculated to be 925. Uncertainty bounds were estimated to be plus or minus three times the mean. Logtriangular(300, 925, 2800).

Aquatic insect larvae. The methylmercury BCF for *Chironomus riparius* larvae is reported to range from 3000 to 5000 (Blaylock et al. 1973). Uncertainty bounds were estimated to be plus or minus three times the mean. Logtriangular(1400; 4000; 12,000).

Zooplankton. The methylmercury BCF for *Gammarus sp.* is reported to be approximately 8000 (EPA 1985). Uncertainty bounds were estimated to be plus or minus three times the mean. Logtriangular(2700; 8000; 24,000).

Other invertebrates. No methylmercury BCF data were found for the other aquatic invertebrate compartments. The distribution for these compartments is bounded by the minimum and maximum values from the distributions of the preceding two compartments. Loguniform(1400; 24,000).

Fish. Three methylmercury BCF values for brook trout (10,000; 12,000; 23,000) and one for rainbow trout (11,000) were found in the literature (geometric mean 13,200) (EPA 1985). The BCF among fish species is assumed to vary little. The GSE among the available values was calculated to be 1.21, and the bounds of the distribution were set to the bounds of the 95% CI on the mean. Logtriangular(9050; 13,200; 19,250).

Frog. Distribution bounds were based on those of the fish distribution because of the physiological similarity (Storer et al. 1957). Uniform(9000; 20,000).

Detritus. The methylmercury BCF for detritus was calculated to be 1.05 from reported uptake and loss rates (Herrick et al. 1982). Uncertainty bounds were estimated to be plus or minus three times the mean. Logtriangular(0.35, 1.05, 3.15).

Terrestrial vegetation. The geometric mean mercury BCF from soil was calculated to be 0.04 for alfalfa grown in mercury contaminated soil (Lindberg et al. 1979). No distinction was made between organic and inorganic mercury in the alfalfa study. The model terrestrial vegetation BCFs for methylmercury and Hg^{2+} are the same. Uncertainty bounds were estimated to be plus or minus three times the mean. Logtriangular(0.013, 0.04, 0.12).

A.3.2 Assimilation Efficiency

Zooplankton. The assimilation efficiency of methylmercury by *Daphnia pulex* was estimated to be between 0.4 and 0.6 from reported retention data (Huckabee et al. 1975). Distribution bounds of 0.2 and 0.8 were based on high uncertainty resulting from lack of data among the invertebrate compartments. Triangular(0.2, 0.5, 0.8).

Aquatic insect larvae. A_{eff} for *Chironomus riparius* was reported to be 0.76 (Blaylock et al. 1973). Distribution bounds of 0.1 and 0.9 were based on high uncertainty resulting from lack of data among the invertebrate compartments. Triangular(0.2, 0.76, 0.9).

Other invertebrates. Distribution was bounded by the values approaching the maximum possible range (0.0 to 1.0) because of lack of data among the invertebrate compartments. Loguniform(0.1, 0.9).

Fish. Methylmercury assimilation efficiency is reported to be 0.67 to 0.94 in yellow perch (Norstrom et al. 1976), 0.815 in mosquito fish (Blaylock et al. 1973), and 0.15 in predators

(Jernelov and Lann 1971). The geometric mean of the four values is 0.53. Conventional wisdom holds that the methylmercury A_{eff} for fish is at least 0.7, thus this value is used as the mode of the triangular distribution bounded by the available minimum and maximum values. Triangular(0.15, 0.7, 0.94).

Frog. No A_{eff} data were found for amphibia. The distribution bounds are set to equal to those of fish. Uniform(0.15, 0.94).

Birds. Assimilation of methylmercury by Leghorn chickens is reported to be 0.61, 0.82, 0.91, and 0.95 (geometric mean = 0.81), depending on dose size (March et al. 1983). The data indicate that A_{eff} in birds is likely to be at least 0.50. Triangular(0.50, 0.81, 0.95).

Mammals. The assimilation of methylmercury from fescue by the cotton rat (*Sigmodon hipidus*) is reported to be 0.99 (Huckabee et al. 1981). Methylmercury A_{eff} is reported to be >0.8 in the cow and goat (Sell and Davison 1975). Data indicate methylmercury A_{eff} in mammals may be about 0.9 on average and is likely to be greater than 0.75 for all model mammalian compartments. Triangular(0.75, 0.9, 0.95).

Great blue heron and mink. Interindividual variability in A_{eff} was assumed to be negligible.

A.3.3 Elimination Rate

Aquatic insect larvae. The methylmercury elimination rate (per day) for *Chironomus riparius* larvae is reported to be 0.099 (Blaylock et al. 1973). Uncertainty bounds were set at plus or minus three times the chironomid value. Logtriangular(0.033, 0.099, 0.3).

Zooplankton. The methylmercury elimination rate for *Daphnia pulex* is reported to range from 0.188 to 0.217 (Huckabee et al. 1975). Uncertainty bounds were set at plus or minus three times the daphnid value. Logtriangular(0.067, 0.099, 0.60).

Crayfish. The methylmercury elimination rate for crayfish is reported to range from 0.0552 to 0.1152 (ESB 1978). Uncertainty bounds were set at plus or minus two times the mean of the literature values. Triangular(0.0426, 0.0852, 0.1704).

Other invertebrates. Loguniform distribution bounded by reported minimum and maximum values among invertebrate compartments. Loguniform(0.03, 0.60).

Fish. Weight and elimination rate data are reported for five species of different sizes (total n = 8) in temperatures ranging from 12 to 15°C (De Freitas et al. 1974). Simple correlation coefficient between weight (g) and K_{el} was calculated to be 0.98 for the following equation. $K_{el} = 0.044 * W^{Normal(-0.54,\ 0.04)}$.

Frog. No methylmercury elimination rate data were found for amphibia. Frog K_{el} distribution is assumed to be the same as fish because of the similarity in gastrointestinal tract and diet (Storer et al. 1957).

Birds. Methylmercury K_{el} was reported to be 0.00825 in mallards (Stickel et al. 1977), 0.0099 in chickens, 0.0113 in osprey (Fimreite 1979), 0.025 in ducks, 0.054 in pheasant, and

0.05862 in chickens (Gardiner 1972). The geometric mean and standard deviation of the preceding data are 0.02 and 2.4, respectively. The bird species and compartment-specific values of K_{el} are likely to lie within the distribution among birds described by the data. Thus, the K_{el} mean for each bird compartment is bounded by the 95% CI bounds of that distribution. Logtriangular (0.0034, 0.02, 0.1152). The GSD for the great blue heron compartment ranges uniformly from 1.04 to 1.08, based on the range in size for adults.

Mammals. Methylmercury K_{el} is reported to be 0.064 and 0.077 in male and female Long-Evans laboratory rats (Thomas et al. 1986) and 0.073 in the cotton rat (Huckabee et al. 1981). The variability among mammalian species is probably at least as great as that among birds; so, K_{el} uncertainty is bounded by plus or minus six times the mean of the rat data. Logtriangular (0.0012, 0.071, 0.426). The GSD for the mink compartment ranges uniformly from 1.06 t 1.1, based on the standard deviation in weight for adults.

A.3.4 Great Blue Heron Whole Body:Egg Partition Coefficient

The methylmercury content of mallard eggs relative to laying hen whole-body concentrations was estimated from reported data to be 0.465 (Heinz 1979). The mean partitioning coefficient for herons was assumed to lie within approximately plus or minus two times the mallard mean. GSD among individual eggs was assumed to range from 1.1 to 1.25, based on reported variability among mallard eggs within the test cohort.

A.4 INORGANIC MERCURY TOXICANT-DEPENDENT PARAMETERS

A.4.1 Bioconcentration Factors

Aquatic vascular plants. Inorganic mercury bioconcentration factors are reported to range from 3 to 77 in the emergent portion and 4 to 264 in the submergent portion of six species of aquatic vascular plants (ESB 1978). Distributions were based on the mean of the respective plant portions among species plus or minus three times. Emergent - Logtriangular (5, 15, 45); Submergent - Logtriangular (10, 32, 100).

Phytoplankton and periphyton. One value for each of four algae types ranging from 853 to 10,920 (geometric mean 3900, GSE = 1.91) is reported in the literature (EPA 1985). Distribution bounded by the 95% CI on the mean. Logtriangular (1075; 3900; 14,200).

Zooplankton. Three zooplankton values are reported in the literature, 2500 for *Gammarus sp.* and mercuric chloride, 2500 for *Gammarus sp.* and mercuric nitrate, and 7500 for a copepod and mercuric chloride (EPA 1985). The geometric mean of the three preceding values is the mode of a logtriangular distribution bounded by plus or minus three times the mode. Logtriangular (1400; 4300; 13,000).

Crayfish. Three values have been reported for the crayfish, *Procambarus clarkii*: 121, 158, and 216 (geometric mean = 161) (Del Ramo et al. 1988). Distribution bounded by bounds of the 95% CI on the mean. Logtriangular (106, 161, 244).

Aquatic insect larvae. One value for the bioconcentration of $HgCl_2$ in *Chironomus riparius* of 12,600 was found in the literature (Rossaro et al. 1986). Distributions are based on the literature value plus or minus three times. Loguniform (4200; 12,600; 37,800).

Other invertebrates. A range of 500 to 50,000 was based on estimated extreme values for zooplankton, insect larvae, and crayfish, with a ratio of 100× for uncertainty. Loguniform (100, 38,000).

Fish. Inorganic mercury BCF values of 1800 and 4994 are reported for rainbow trout and fathead minnow, respectively (EPA 1984b). Distributions were based on the mean of the literature values plus or minus three times. Loguniform (1000, 3000, 9000).

Frog. No BCF data for amphibia were found in the literature. A range of 1000 to 9000 was based on fish data. Loguniform (1000, 9000).

Detritus. The inorganic mercury bioconcentration factor for detritus is reported to be ~1100 (Herrick et al. 1982). Distributions were based on the mean of the literature values plus or minus three times. Loguniform (350, 1100, 3300).

A.4.2 Assimilation Efficiency

Invertebrates. One Hg^{2+} A_{eff} value was found in the literature for invertebrates, 0.6 in *Chironomus riparius* larvae (Blaylock et al. 1973). Distribution is bounded by the values approaching the maximum possible range (0.0 to 1.0) because data among the invertebrate compartments is lacking. Triangular (0.1, 0.6, 0.9).

Fish. Inorganic mercury assimilation efficiency is reported to be between 0.05 and 0.25 for perch, pike, bullhead, ling, and goldfish (De Freitas et al. 1974) and 0.4 for largemouth bass (Blaylock et al. 1973). Geometric mean and standard error of the preceding data are 0.172 and 1.24, respectively. Distribution bounds were based on the bounds of 95% CI of the mean. Triangular (0.112, 0.172, 0.264).

Frog. No A_{eff} data were found for amphibia. The loguniform distribution bounds are based on those of fish. Uniform (0.1, 0.3).

Birds. Inorganic mercury assimilation efficiency range was estimated from Hg^{2+} retention curves in chickens (Fimreite 1979) and was assumed to be <0.5, based on fish and mammalian data. Uniform (0.1, 0.5).

Mammals. Inorganic mercury A_{eff} was reported to be <0.2 in the goat and cow (Sell and Davison 1975). Upper bound was based on fish data. Triangular (0.1, 0.2, 0.5).

Great blue heron and mink. Interindividual variability in A_{eff} was assumed to be negligible.

A.4.3 Elimination Rate

Invertebrates. One Hg^{2+} K_{el} value was found in the literature for invertebrates, 0.126 in *Chironomus riparius* larvae (Blaylock et al. 1973). Distribution bounds set at a ratio of

100× centered on the chironomid value because of the high uncertainty (see PCB invertebrate K_{el} discussions). Logtriangular (0.0126, 0.126, 1.26).

Fish. Weight and elimination rate data are reported for five species in temperatures ranging from 12 to 15°C (De Freitas et al. 1974). A simple correlation coefficient between weight and K_{el} was calculated to be 0.86 for the equation $K_{el} = 0.111 * W^{normal(0.46, 0.16)}$.

Frog. No inorganic mercury elimination rate data were found for amphibia. Frog K_{el} distribution is the same as fish because of the similarity in gastrointestinal tract and diet (Storer et al. 1957).

Birds. Inorganic mercury elimination rate was estimated to be ~0.82 per day from reported Leghorn cock retention data (Fimreite 1979). The ratio of maximum to minimum methylmercury K_{el} values for birds was 34, and the value for chicken was approximately equal to the mean of the available data. Thus, distribution bounds are based on plus or minus six times the Hg^{2+} chicken value. Logtriangular (0.137, 0.82, 4.92). The GSD for the great blue heron compartment ranges uniformly from 1.04 to 1.08, based on the range in size for adults.

Mammals. Inorganic methylmercury K_{el} was reported to be 0.098 and 0.029 in laboratory rats (Thomas et al. 1986 and Jugo 1979) and 0.009 in the goat (Sell and Davison 1975). Variability among mammals is assumed to be as great as that among birds; thus, bounds are equal to plus or minus six times the mean of the rat and goat values. Logtriangular (0.005, 0.03, 0.18). The GSD for the mink compartment ranges uniformly from 1.06 to 1.1, based on the standard deviation in weight for adults.

A.5 TOXICANT-INDEPENDENT PARAMETERS

A.5.1 Body Weight

Zooplankton. The geometric mean and standard error of dry body weight (in grams) for 54 zooplankton species and families was calculated to be 6.55×10^{-5} g and 1.52, respectively, from data tabulated by Jorgensen (1979). Zooplankton were assumed to be 80% water. Distribution bounds are the bounds of 95% CI on the mean. Logtriangular (1.4×10^{-5}, 3.3×10^{-5}, 7.6×10^{-5}).

Larvae. The geometric mean and standard error of dry body weight for 21 aquatic insect species and families were calculated to be 1.25×10^{-4} g and 1.25, respectively, from data tabulated by Jorgensen (1979). Aquatic insects were assumed to be 80% water. Distribution bounds are bounds of the 95% CI on the mean. Logtriangular (4×10^{-4}, 6.25×10^{-4}, 9.8×10^{-4}).

Crayfish. Body weight range is based on a length of 1 to 5 cm and length to width ratio of 5:1 (estimated from Pennak 1978); the true shape of crayfish is to be approximated by a cylinder with density equal to 1 g/cm^3. Uniform (0.1, 4).

Aquatic oligochaete. Body weight range is based on reported fresh weights for *Tubifex tubifex* (Oliver 1984). Uniform (0.0023, 0.019).

Frog. Body weight range is estimated from reported leopard frog lengths (Behler and King 1979), shape was approximated by a sphere, and density was estimated to be 1 g/cm^3. Uniform (2, 15).

Terrestrial insect. Prey-size insects are assumed to be of length 0.5 to 1.5 cm and width and height equal to 1/5 of length. Insect shape and mass is assumed to be approximated by a three-dimensional rectangle with density equal to 1 g/cm^3. Uniform (0.005, 0.135).

Earthworm. Distribution is assumed to be the same as aquatic oligochaete distribution.

Gizzard shad. Body weight is calculated from prey length (L) based on relationship derived from reported Southeastern United States length/weight data (Carlander 1969). $W = 1.12 \times 10^{-5} * L^{3.02}$, $r^2 = 0.993$.

Threadfin shad. Body weight is calculated from prey length (L) based on relationship derived from reported Southeastern United States threadfin shad length/weight data (Carlander 1969). $W = 2.11 \times 10^{-4} * L^{2.67}$, $r^2 = 0.991$.

Yellow Perch. Body weight is calculated from prey length (L) by using reported length/weight relationship for bluegill (Carlander 1969). $W = 5.62 \times 10^{-4} * L^{3.01}$.

Bluegill. Body weight is calculated from prey length (L) by using reported length/weight equation for bluegill (Carlander 1977). $W = 5.62 \times 10^{-4} * L^{3.01}$.

Largemouth bass. Body weight is calculated from prey length (L) by using reported length/weight equation for bluegill (Carlander 1977). $W = 1.33 \times 10^{-6} * (L/1.205)^{3.115}$.

Great blue heron. Heron weights are based on allometric relationships derived from wood stork body length and weight data (Bull and Farrand 1977; Kushlan 1978) and adult great blue heron length data (Bull and Farrand 1977). Assumed mean weight of female herons lies within the central 50% of the range derived from minimum and maximum lengths of adult herons. Triangular{2901, uniform(3143, 3627), 3868}.

Songbird. Distribution parameters are equal to mean weight among twelve species of common songbirds, and bounds are those of the 95% CI on the mean (Kenaga 1973). Triangular (16.3, 23, 32.5).

Canada goose. Canada geese are reported to reach up to 4.5 kg in weight (Bellrose 1976), presumably much too large for the average adult mink (~1.5 kg) to prey upon. Distribution is based upon the assumption that small geese are selectively preyed upon. Uniform (50, 75% of mink body weight).

Diving and dabbling duck. The range is the same as that for geese. The mammalian body weight distributions are all based on reported ranges of adult weights for species common to east Tennessee (Whitaker 1980). For animals that may approach a mink in size, the maximum possible weight is truncated to 75% of the mink weight.

Squirrel. Uniform (400, 710).

Rabbit. Uniform (10% of mink weight, 75% of mink weight).

Vole. Uniform (20, 70).

Muskrat. Same as that for rabbit.

Mink. Distribution parameters calculated from a cohort of 24 female mink were weighed every 2 weeks from 90 through 438 days of age (EPA 1987). Normal (953,80). Because of the relatively large sample size, uncertainty about the interindividual distribution parameters was assumed to be negligible.

A.5.2 Food Intake Rate

Zooplankton. Ingestion of algae by *Daphnia pulex* is reported to be normally distributed, with mean of 3.1×10^7 and standard deviation of 6.1×10^6 μm^3 of algae·mg predator^{-1}·h^{-1} (Vanderploeg et al. 1986). The weight of algae in picograms is reported as 0.47*Volume$^{0.99}$ (Reynolds 1984). The preceding two equations are combined with the appropriate unit conversion factors to arrive at an ingestion rate of 26.6% per day. Distribution bounds are set at plus or minus three times the calculated value. Logtriangular (0.09,0.266,0.80)

Aquatic insect larvae. Daily intake rate is based on reported values for *Chaoborus* larvae of 6 to 12.5% of body weight (Pastorok 1980). Distribution upper bound is equal to three times the reported maximum value. Loguniform(0.06, 0.37).

Crayfish. Distribution is based on aquatic insect larvae intake data. (Uniform (0.01, 0.5).

Aquatic oligochaete. Intake rate is based on reported oligochaete sediment ingestion rates (Klump et al. 1987). Normal (1.79, 0.36).

Frog. No ingestion rate data were found for amphibians. Distribution was based on fish data. Uniform (0.01, 0.04).

Terrestrial insect. Herbivorous insect ingestion rate has been estimated to be 13.6% of body weight per day (Crossley and Howden 1961). Distribution bounds are equal to plus or minus two times the reported value. Uniform (0.07, 0.27).

Earthworm. Same as that for aquatic oligochaete distribution.

Largemouth bass. Largemouth bass daily ingestion rate in grams (C) may be calculated from the following equation:

$$C = 0.33 * W^{-0.325} * r_c * P \quad \text{(Rice et al. 1983)}$$

where

r_c = temperature dependent scaling factor,

P = proportionality constant indicating fraction of C_{max} ingested.

Mean annual temperatures in surface waters on and around the ORR range from ~17 to 19°C which yields r_c values from 0.4 to 0.5 (Rice et al. 1983). The scaling factor r_c is modeled by a uniform distribution bounded by 0.4 and 0.5; P was estimated to lie between 0.4 and 0.6 and is also modeled by a uniform distribution.

Yellow perch. Yellow perch daily ingestion rate in grams (C) may be calculated from the following equation:

$$C = 0.25 * W^{-0.27} * r_c * P \quad \text{(Kitchell et al. 1977)}$$

where r_c and P are the same as that for largemouth bass.

Bluegill, threadfin shad, gizzard shad. No species-specific feeding rate equations were found for these compartments. The equation used to describe bass and perch ingestion are used for the bluegill and shad compartments. Ingestion equation parameters are assumed to lie within the values reported for bass and perch.

$$C = \text{uniform}(0.25,0.33) * W^{\text{uniform}(-0.325,-0.27)} * r_c * P .$$

Great blue heron. Wading bird daily ingestion rate as a function of body weight is reported as log Food (g/day) = 0.966*log W - 0.64, $r^2 = 0.95$, n = 7 (Kushlan 1978). The result of this equation is divided by the GBH body weight to arrive at the food intake rate as a fraction of body weight per day.

Song bird and woodcock. Food ingestion rates (mass basis) for eight common songbirds, as measured in the laboratory and estimated from metabolism-based feeding rate equations, are reported in the literature (Kenaga 1973). Linear equations were fit to both the experimental and metabolic rate data, with body weight as the independent variable by least-squares regression. The estimated experimental values were observed to be lower than the metabolic rate based estimates for each possible body weight. Thus, the model songbird intake rate is set to fall between the results of the two equations for each model body weight. {Uniform(2.1+0.067*W, 2.25+0.21*W)}/W.

Canada goose, diving duck, and dabbling duck. These values are based on reported intake rate (mass basis) equation derived from chicken data (EPA 1987). Intake (g/day) = normal (0.75, 0.043)*$W^{\text{normal}(0.85,0.04)}$, $r^2 = 0.97$.

Squirrel, vole, and deer. These values are based on reported intake rate (mass basis) equation derived from multispecies data, for species that have primarily dry diets (EPA 1987). Intake (g/day) = normal (0.049, 0.002)*$W^{\text{normal}(0.79,0.01)}$, $r^2 = 0.95$.

Rabbit. This values is based on reported intake rate (mass basis) equation derived from rabbit data (EPA 1987). Intake (g/day) = normal (0.041, 0.004)*$W^{\text{Normal}(0.79,0.09)}$, $r^2 = 0.73$.

Muskrat and mink. These values are based on reported intake rate (mass basis) equation derived from multispecies data, for species that have primarily wet diets (EPA 1987). Intake (g/day) = normal (0.054, 0.014)*$W^{\text{normal}(0.945,0.06)}$, $r^2 = 0.975$.

A.5.3 Water Intake Rate

Terrestrial invertebrates. Assumed to be negligible.

Song bird, woodcock, Canada goose, squirrel, vole, rabbit, and deer. These values are based on the reported intake rate (mass basis) equation for species that have dry-food diets (EPA 1987). Intake (liters/day) = normal (0.093, 0.005)*$W^{normal(0.758,0.021)}$, $r^2 = 0.98$.

Other terrestrial compartments. This value is based on reported intake rate (mass basis) equation for species that have a wet-food diet (EPA 1987). Intake (liters/day) = normal (0.009, 0.004)*$W^{normal(1.204,0.096)}$, $r^2 = 0.96$.

A.5.4 Dietary Fractions

The degree of uncertainty associated with the true lifetime weighted-average diet for the model predators resulted in each predator-prey percentage (f_{ij}) being expressed as a uniform distribution. The midpoint of each distribution is shown in Tables A.1 through A.3. In general, the upper and lower bounds of each f_{ij} distribution are plus and minus one-half the best estimate of the mean. In cases where the f_{ij} sum among all prey for a given predator is greater than 1 because of high uncertainty, the individual values are normalized relative to their sum to result in a total of 1.

Table A.1. Dietary fractions for consumer species—nonvertebrates

Food	Zooplankton	Oligochaete	Earthworm	Insect larvae	Terrestrial insect	Crayfish	Snail
Sediment		0.90					
Vascular plants				0.20		0.25	
Detritus	0.20	0.05		0.40		0.75	0.22
Phytoplankton	0.80						
Periphyton		0.05		0.20			0.78
Zooplankton				0.20			
Soil			1.00				
Terrestrial vegetation					1.00		

No quantitative measures of invertebrate dietary habits were found in the literature. Therefore, the invertebrate compartment f_{ij} values are based on qualitative discussions found in the literature (Pennak 1978).

Table A.2. Dietary fractions for consumer species—fish

Food	Bluegill YOY	Bluegill	Large-mouth bass YOY	Large-mouth bass	Great blue heron	Gizzard shad	Gizzard shad YOY	Threadfin shad	Threadfin shad YOY	Yellow perch	Yellow perch YOY
Sediment		0.03				0.03	0.03	0.03	0.04	0.03	0.03
Vascular plants	0.04	0.10									
Detritus	0.03					0.60	0.28	0.07	0.04	0.03	
Phytoplankton	0.04	0.10				0.08	0.09	0.33	0.12		
Periphyton						0.12	0.12	0.05	0.12		
Zooplankton	0.13	0.15	0.46			0.04	0.38	0.47	0.60	0.19	0.67
Larvae (benthic)	0.20	0.15	0.10	0.05		0.05	0.03		0.02	0.19	0.02
Larvae (pelagic)	0.14	0.14				0.04	0.04	0.03	0.04	0.01	0.03
Crayfish	0.20	0.15	0.10	0.03						0.19	0.10
Snail											
Oligochaete	0.20	0.15	0.10	0.03		0.04	0.03	0.02	0.02	0.24	0.15
Frog					0.05						
Gizzard shad	0.005	0.01	0.15	0.52	0.69					0.06	
Threadfin shad	0.005	0.01	0.05	0.31	0.11					0.06	
Yellow perch	0.005	0.01	0.02	0.03	0.05						
Bluegill			0.02	0.03	0.05						
Largemouth bass					0.05						

Table A.3. Dietary fractions for consumer species—frog, birds, and mammals

Food	Frog	Song bird	Canada goose	Diving duck	Dabbling duck	Squirrel	Vole	Rabbit	Muskrat	Mink
Vascular plants				0.42	0.42				0.85	
Phytoplankton					0.04					
Periphyton				0.11						
Zooplankton					0.04					
Larvae (benthic)					0.04					
Larvae (pelagic)	0.25			0.16	0.18				0.05	0.01
Crayfish	0.25			0.16	0.08				0.05	0.08
Snail	0.25			0.16	0.18				0.05	
Oligochaete	0.25				0.02					
Frog										0.09
Gizzard shad										0.05
Threadfin shad										0.05
Yellow perch										0.05
Bluegill										0.05
Largemouth bass										0.04
Terrestrial vegetation		0.12	0.95			0.95	0.50			
Tubers							0.50			
Terrestrial insect		0.20	0.05			0.05				
Earthworm		0.68								
Song bird										0.04
Canada goose										0.10
Diving duck										0.04
Dabbling duck										0.04
Squirrel										0.03
Vole										0.21
Rabbit										0.08
Muskrat										0.04

The fish dietary percentages are based on data generated from stomach analyses of fish collected from an Arkansas reservoir (Ploskey and Jenkins 1982). Stomach content analyses of largemouth bass collected from Watts Bar Reservoir show that these fish selectively prey on gizzard and threadfin shad (Adams et al. 1982). Fish less than 1-year old (young-of-the-year, YOY) are reported to have different diets than their more mature counterparts (Ploskey and Jenkins 1982). These differences are accounted for by including a set of f_{ij} values for both YOY and adults for each fish compartment.

The waterfowl f_{ij} values are based on several sources, each containing qualitative descriptions of geese, diving duck, and dabbling duck foraging habits (Martin et al. 1951; Driver et al. 1974; Swanson et al. 1974; Bellrose 1976; Kaminski and Prince 1981). In general, the goose compartment consumes primarily terrestrial vegetation, the dabbling duck compartment consumes primarily aquatic vegetation, and the diving duck compartment primarily consumes aquatic vegetation and benthic invertebrates.

The great blue heron compartment is assumed to be primarily piscivorous, with frogs and voles accounting for the balance of its diet. Fish remains collected on the floor of two colonies near the ORR were exclusively shad, with all identifiable remains being gizzard shad.

As with the waterfowl compartments, no reliable quantitative data regarding the diets of terrestrial mammals (other than mink) were found. The mammalian dietary percentages are based on imprecise graphical information and general trends for representative species of each compartment (Martin et al. 1951).

The mink dietary distributions are based on the results of stomach analyses performed on mink collected from trappers in Missouri (Korschgen 1958). Korschgen's results indicate that midwestern mink prey primarily upon mammals (40% of stomach content volume on average) during the winter months, with fish accounting for ~20% of the average diet. There are likely to be seasonal trends in the diet of mink as a result of prey availability. Further research into this question is recommended.

A.5.5 Percentage Lipid

Frog. The lipid fraction of total body weight is reported to vary between 5 and 20% throughout the year in frogs, with high values observed prior to the onset of winter and low values observed in the spring following the winter stasis period (Duellman and Trueb 1986). Lifetime average lipid fraction is assumed to be approximated by the midpoint of seasonal variation. Uniform (0.10, 0.15).

Gizzard shad. The lipid fraction of total body weight on a dry-weight basis is reported to vary between 5 and 30% throughout the year in mature gizzard shad, with high values observed during the spring and summer months and low values observed in the fall and winter (Strange and Pelton 1987). Lifetime average lipid fraction is assumed to be approximated by the midpoint of seasonal variation. Gizzard shad are assumed to be 75% water. Uniform (0.03, 0.06).

Threadfin shad. The lipid fraction of total body weight on a dry-weight basis is reported to vary between 3.8 and 19% throughout the year in YOY threadfin shad, with high values observed during the spring and summer months and low values observed in the fall and winter

(Strange and Pelton 1987). Lifetime average lipid fraction is assumed to be approximated by the midpoint of seasonal variation. The ratio of lipid fraction in adult to YOY gizzard shad is 2.63. Threadfin shad are assumed to be 75% water. Uniform (0.05, 0.09).

Yellow perch. The lipid fraction of total body weight is reported to vary between 1 and 5.2% in Lake Erie yellow perch (Dugal 1962). Lifetime average lipid fraction is assumed to be approximated by the midpoint of seasonal variation. Uniform (0.02, 0.04).

Bluegill. The average lipid fraction of total body weight in mature bluegill collected from White Oak Creek embayment on the ORR ranges from 1.5 to 3% (M. Adams, ORNL, unpublished data). Uniform (0.015, 0.03).

Largemouth bass. No data for the lipid content of mature largemouth bass were found. Model distribution is based on the assumption that bass are relatively lean and thus have lipid fractions similar to perch and bluegill. Uniform (0.01, 0.05).

Great blue heron. The fraction of body weight as fat for the heron compartment was based on reported fat concentrations for chickens and geese, ranging from 10 to 20% (Sturkie 1986). Loguniform{uniform(0.1, 0.2), uniform(0.2, 0.5)*mean}.

A.5.6 Prey Size

Fish. The calculated mean prey size of 441 largemouth bass collected in Watts Bar Reservoir over a 2-year period, as a fraction of predator length, fits a normal distribution with a mean of 0.346 (standard deviation = 0.028, standard error = 0.012) (Adams et al. 1982). Distribution bounds were set to bounds of the 95% CI on the mean. Triangular (0.322, 0.346, 0.370)

Great blue heron. The mean length of 23 fish carcasses collected from the floor of two great blue heron nesting colonies near the ORR was determined to be normally distributed with 169 mm (standard deviation = 48, standard error = 10.25). Normal{uniform(148, 190), uniform(20,40)}.

Mink. The average length of mink fish prey has been reported to be approximately 76 to 101 mm (Hamilton 1940), 178 mm (Hamilton 1959), and 190 cm (Wise 1981). Mean length of prey fish for mink is assumed to vary uniformly from 75 to 200 mm. The standard deviation was subjectively determined. Normal{uniform(75, 203), uniform(0.1, 0.2)*mean}.

A.5.7 Lifespan

Invertebrates. The mean lifespan (days) of organisms comprising the invertebrate model compartments was estimated from both qualitative and quantitative information contained in Pennak (1978) and Zaika (1973). The distribution and bounds for each compartment are listed in Table A.4.

Fish. Estimates of the age of a given fish species and size were generated from age class and length data for the appropriate species reported by Carlander (1969). Examination of the data indicated that age class and size relationships vary with geographic region; thus only data reported from Southeastern United States lakes and reservoirs were considered.

Table A.5 is used in model determination of the age of a prey fish for the mink, heron, and largemouth bass compartments. No data were found for yellow perch; therefore, the age-to-length relationship for this compartment was assumed to be the same as that for bluegill.

Table A.4. Expressions of uncertainty about mean lifespan in days for invertebrate compartments

Compartment	Distribution	Min	Max
Zooplankton	Uniform	10	20
Aqatic insect larvae	Loguniform	30	360
Crayfish	Loguniform	30	360
Oligochaete	Loguniform	30	360
Terrestrial insect	Uniform	10	20
Earthworm	Loguniform	30	360

Table A.5. Look-up table used to estimate fish age in years based on length (mm)

	Age class						
Species	0	1	2	3	4	5	6
Gizzard shad	178	240	279	333	353	na	na
Threadfin shad	100	200	na	na	na	na	na
Bluegill	120	155	168	186	201	218	na
Largemouth bass	157	292	331	380	419	445	488

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