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Great Lakes Water Quality Initiative Technical Support Document for Wildlife Criteria

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Office of Science and Technology
Office of Water
United States Environmental Protection Agency
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GREAT LAKES WATER QUALITY INITIATIVE TECHNICAL SUPPORT DOCUMENT FOR WILDLIFE CRITERIA

Note: This Technical Support document contains background material and material intended to clarify portions of the regulation. It does not establish any additional regulatory requirements.

I. INTRODUCTION

The purpose of this document is to provide technical information and the rationale for the procedure to derive chemical-specific water quality criteria to protect wildlife species. For the purposes of this document, "wildlife" are defined as non-domesticated species in the taxonomic classes Aves and Mammalia (birds and mammals).

Because the waters of the Great Lakes System not only support numerous human activities and habitat for aquatic organisms, but also sustain viable mammalian and avian wildlife populations, specific water quality criteria are derived to ensure the quality of the waters in the System are adequate to support these populations. The water quality criteria for wildlife are surface water concentrations of toxicants that will cause no significant reduction in the viability or usefulness (in a commercial or recreational sense) of a population of exposed animals utilizing waters of the Great Lakes System as a drinking and/or foraging source over several generations. For the purpose of the Great Lakes Water Quality Initiative (GLWQI) regulation, this concentration is called the Great Lakes Wildlife Criterion (GLWC).

This document contains a number of sections. In Section II a number of terms are defined which are used in Appendix D to Part 132 of the final GLWQI guidance, the *GLWQI Methodology for the Development of Wildlife Criteria,* as well as in this document. Section III presents the derivation of the equation used to determine wildlife values, and a description of the methods to be used to derive either Tier I wildlife criteria or Tier II wildlife values. Section IV describes U.S. EPA's intent in selecting representative wildlife species as well as summarizes the analyses carried out by U.S. EPA to determine the appropriate representative species for the final GLWQI regulation. Section V presents the minimum toxicity data requirements for the derivation of wildlife values and describes the scientific judgements required to select the most appropriate toxicity study to use in deriving wildlife values. Section VI discusses the theory for each of the uncertainty factors which are considered in deriving wildlife values, summarizes analyses carried out by U.S. EPA and others to support the recommended ranges for the uncertainty factors, and provides guidance for the selection of values for each of the uncertainty factors. Section VII describes the approach used to determine the trophic levels, body weights, and food and water ingestion rates for each of the representative species as well as describes general methods which can be used to estimate body weights and food and water ingestion rates for other wildlife species.

Great Lakes States and Tribes are required to adopt methodologies consistent with the method set out in appendix D to the GLWQI final to derive wildlife criteria only for the

bioaccumulative contaminants of concern (BCCs) as defined in Part 132.2. This is consistent with recommendations from the U.S. EPA's Science Advisory Board (U.S. EPA, 1992c and 1994) which endorsed the initial emphasis of the wildlife risk assessment program being on the direct effect of bioaccumulative chemicals on wildlife.

II. DEFINITIONS

Acceptable endpoints. For the purpose of wildlife criteria derivation, acceptable subchronic and chronic endpoints are those which affect reproductive or developmental success, organismal viability or growth, or any other endpoint which is, or is directly related to, parameters that influence population dynamics.

Assessment endpoint. An explicit expression of the environmental value that is to be protected. (As defined in U.S. EPA, 1992b.)

Measurement Endpoint. A measurable ecological characteristic that is related to the valued characteristic chosen as the assessment endpoint. Measurement endpoints are often expressed as the statistical or arithmetic summaries that comprise the measurement. (As defined in U.S. EPA, 1992b.)

Chronic effect. An adverse effect that is measured by assessing an acceptable endpoint, and results from continual exposure over several generations, or at least over a significant part of the test species' projected life span or life stage.

Subchronic effect. An adverse effect, measured by assessing an acceptable endpoint, resulting from continual exposure for a period of time less than that deemed necessary for a chronic test.

Lowest-observed-adverse-effect-level (LOAEL). The lowest tested dose or concentration of a substance which resulted in an observed adverse effect in exposed test organisms when all higher doses or concentrations resulted in the same or more severe effects.

Bounded LOAEL. A LOAEL, as defined above, that is obtained from a study where a NOAEL for the same endpoint is also determined, such that the dose-response threshold is bracketed or "bounded" by both the NOAEL and LOAEL.

Unbounded LOAEL. A LOAEL, as defined above, that is obtained from a study where the corresponding NOAEL for the same endpoint cannot be determined because all of the dose levels in the test were shown to cause adverse effects relative to the controls. Thus, only the upper bound of the dose-response threshold can be determined from the unbounded LOAEL.

No-observed-adverse-effect-level (NOAEL). The highest tested dose or concentration of a substance which resulted in no observed adverse effect in exposed test organisms where higher doses or concentrations resulted in an adverse effect.

Bounded NOAEL. A NOAEL, as defined above, that is obtained from a study where a corresponding LOAEL for the same endpoint is also determined, such that the dose-response threshold is bracketed or "bounded" by both the available NOAEL and LOAEL.

Unbounded NOAEL A NOAEL, as defined above, that is obtained from a study where the corresponding LOAEL for the same endpoint cannot be determined because none of the dose levels in the test were shown to cause adverse effects relative to the controls. Thus, only the lower bound of the dose-response threshold can be determined from the unbounded NOAEL.

Trophic Level. A functional classification of taxa within a community that is based on feeding relationships (e.g., aquatic green plants comprise the first trophic level and herbivores comprise the second). (As defined in U.S. EPA, 1992b.)

Bioaccumulation. The net accumulation of a substance by an organism as a result of uptake from all environmental sources. (As defined in Appendix B to Part 132 of the final GLWQI guidance, *GLWQI Methodology for Deriving Bioaccumulation Factors.*)

Bioaccumulation Factor (BAF). The ratio (in L/kg) of a substance's concentration in tissue of an aquatic organism to its concentration in the ambient water, in situations where both the organism and its food are exposed and the ratio does not change substantially over time. (As defined in Appendix B to Part 132 of the final GLWQI guidance, *GLWQI Methodology for Deriving Bioaccumulation Factors.*)

Biomagnification. The increase in tissue concentration of poorly depurated materials in organisms along a series of predator-prey associations, primarily through the mechanism of dietary accumulation. (As defined in Appendix B to Part 132 of the final GLWQI guidance, *GLWQI Methodology for Deriving Bioaccumulation Factors.*)

Biomagnification Factor (BMF). The ratio of a substance's concentration in the tissue of an animal that consumes aquatic organisms to its concentration in the aquatic organisms which it consumes (unitless). The BMF is used in the wildlife methodology to determine the concentration of a contaminant in gulls, which consume fish, and which, in turn, are consumed by eagles.

Representative Species. Wildlife species representative of avian and mammalian species resident in the Great Lakes basin which are likely to experience the highest exposures to bioaccumulative contaminants through the aquatic food web.

Population. An aggregate of individuals of a species within a specified location in space and time. (As defined in U.S. EPA, 1992b.)

Wildlife Value, species-specific. The value derived from applying Equation 5 (below) using exposure parameters for a representative species.

Wildlife Value, taxonomic class-specific. The value derived for a given taxonomic class by taking the geometric mean of wildlife values for the representative species within a given taxonomic class.

Great Lakes Wildlife Criterion. The lower of the two taxonomic class-specific wildlife values (one for birds and one for mammals).

III. CALCULATION OF TIER I WILDLIFE CRITERIA AND TIER II WILDLIFE VALUES

III.A. Derivation of the Equation

The equation used to calculate Wildlife Values (WV) has both an effect and an exposure component. The effect component is defined as the Test Dose (TD) which is either a LOAEL or NOAEL for milligrams of substance per kilogram body weight per day (mg/kg-d). The exposure routes considered in this derivation are food and water ingestion, and because the intake level is dependent on organism size, it is scaled to body weight. The total toxicant intake through these exposure routes is determined and then set equal to the TD as follows:

$$\text{Toxicant intake through drinking water} = (WV \times W)/Wt \quad (1)$$

$$\text{Toxicant intake through food} = [WV \times \Sigma (F_{Tli} \times BAF_{Tli}]/Wt \quad (2)$$

Where:

- WV = Species-specific wildlife value in milligrams of substance per liter (mg/L).
- W = Average daily volume of water consumed in liters per day (L/d) by the representative species.
- F_{Tli} = Average daily amount of food consumed from trophic level i in kilograms per day (kg/d) by the representative species.
- BAF_{Tli} = Bioaccumulation factor for wildlife food in trophic level i in liters per kilogram (L/kg). Developed using guidelines for wildlife presented in Apendix B of Part 132 of the final GLWQI guidance, *GLWQI Methodology for Deriving Bioaccumulation Factors*. For consumption of piscivorous birds by other birds, the BAF is derived by multiplying the trophic level three BAF for fish by a BMF for biomagnification of the chemical from fish to birds that consume these fish.
- Wt = Average weight in kilograms (kg) for the representative species.

Equations one and two are combined to yield Equation three.

$$TD > (WV \times W)/Wt + [WV \times \Sigma (F_{Tli} \times BAF_{Tli}]/Wt \quad (3)$$

Where:

- TD = Test Dose in milligrams of substance per kilogram body weight per day (mg/kg-d) for the test species (either a NOAEL or LOAEL derived from mammalian or avian toxicity studies).

Factoring and rearranging produces:

$$WV < \frac{TD \times Wt}{W + \Sigma [F_{Ti} \times BAF_{Ti}]} \quad (4)$$

To account for differences in toxicity among species and uncertainties in LOAEL to NOAEL extrapolations and subchronic to chronic extrapolations, the TD is divided by three uncertainty factors, UF_A, UF_S, and UF_L.
Where:

UF_A = Uncertainty Factor for extrapolating toxicity data across species (unitless). A species-specific uncertainty factor shall be selected for each representative species.

UF_S = Uncertainty Factor for extrapolating from subchronic to chronic exposures (unitless).

UF_L = Uncertainty Factor for LOAEL to NOAEL extrapolations (unitless).

The final equation for the WV is:

$$WV = \frac{\frac{TD}{UF_A \times UF_S \times UF_L} \times Wt}{W + \Sigma [F_{Ti} \times BAF_{Ti}]} \quad (5)$$

III.B. Derivation of the Final Tier I Wildlife Criterion

Under the final methodology, the wildlife values specific for each taxonomic class are derived by taking the geometric mean of the wildlife values across all of the representative species within each taxonomic class. The equation to do this is presented below (Equation 6).

$$WV \text{ (taxonomic class)} = e \text{ Exp } [\Sigma \ln WV_{(\text{repre. species } i)} / n] \quad (6)$$

Where:

n = The number of representative species in a given taxonomic class for which species-specific wildlife values were calculated.

As required in the final methodology, the Great Lakes Wildlife Criterion is then set equal to the lower of the two taxonomic class-specific wildlife values.

III.C. Derivation of a Tier II Wildlife Value

The equation to derive a Tier II wildlife value is the same as that presented above to derive the taxonomic class-specific Tier I wildlife values which are then used to determine the Tier I wildlife criterion. There are three differences in the derivation of a Tier I wildlife criterion and a Tier II wildlife value. The first is that for a Tier I wildlife criterion, a taxonomic class-specific wildlife value is derived for both taxonomic classes (i.e., Aves and Mammalia) while a Tier II wildlife value may be determined when a taxonomic class-

specific wildlife value is available for only one taxonomic class. The second difference is in the study duration requirements for both birds and mammals and these are described in Sections V.A and V.B below. The third difference is in the intent of the interspecies uncertainty factor (UF_A) and its associated value which is described in more detail in Section VI.A.7 below.

It is important to note that, based on the duration of the study from which the test dose was derived and the value selected for the interspecies uncertainty factor (UF_A) selected, it is possible to derive a Tier II wildlife value with as much scientific validity as a Tier I wildlife criterion. The only difference which could exist would be the assurance that the taxonomic class for which a wildlife value was not calculated would be protected by the Tier II wildlife value derived.

If a State or Tribe uses the methodology described in this document to derive a Tier II wildlife value for non-bioaccumulative chemical, it may be appropriate to select different representative species which are better examples of wildlife species with the greater exposure for a given chemical.

IV. DETERMINATION OF THE REPRESENTATIVE SPECIES

The wildlife criteria derived using the approach discussed in this document are intended to protect all avian and mammalian wildlife species in the Great Lakes basin. While it would be desirable also to consider reptiles and amphibians, toxicity data and approaches for assessing exposures of these two classes of vertebrates are not yet available.

To estimate water quality criteria to protect wildlife in the Great Lakes basin, the decision was made to establish an approach to ensure protection of those species likely to face the highest exposure levels to bioaccumulative chemicals as a consequence of their behaviors and dietary habits. Therefore, U.S. EPA identified species representative of avian and mammalian species resident in the Great Lakes basin which are likely to experience the highest exposures through the aquatic food web (i.e., piscivorous birds and mammals, as discussed below). Within each taxonomic class, the comparative toxicity data available for a given chemical is frequently very limited. This makes it difficult to determine the relative toxicological sensitivities of the numerous avian and mammalian wildlife species for which no data are available. The representative species concept intends to ensure that a wildlife criterion derived using this method would be protective of the wide distribution of species resident in the Great Lakes basin for which no toxicity data is available. While it is possible that a less exposed wildlife species (i.e., a non-piscivorous bird or mammal) may be more toxicologically sensitive than a representative species, that species will have a much lower exposure and therefore should still be protected by a criterion derived to protect the identified representative species. U.S. EPA also did not attempt to select species based on their toxicological sensitivities because such sensitivities will likely vary according to the chemical and its mode of action. The analysis described in this section was performed to determine which of the fish-eating avian and mammalian species of the Great Lakes basin are likely to experience the highest exposure to contaminants in the basin through aquatic food chains.

In general, smaller endotherms (i.e., warm-blooded animals, e.g., birds and mammals) have higher food ingestion rates relative to their body mass than do larger endotherms. This is because smaller animals generally have a larger surface area to volume ratio, and lose proportionately more energy to their environment as heat than do larger animals. This suggests that small animals would be exposed to a larger quantity of contaminants relative to their body size than larger animals. However, small piscivores (i.e., fish eating animals) are generally size-limited predators, and feed on smaller fish in lower trophic-levels than do larger piscivores. Because the concentration of bioaccumulative pollutants usually is lower in lower trophic level organisms, it is not clear that smaller animals will experience higher exposures than larger animals. Therefore, to identify species likely to experience the highest exposure levels, both relative food ingestion rates and the trophic level of the prey must be considered.

The identification of species of birds and mammals that are likely to experience the highest exposures to contaminants in aquatic food chains in the Great Lakes basin was based on both animal size and diet of piscivores in the region. In addition, a literature review was conducted to identify those species for which populations in the Great Lakes basin have already been adversely impacted by contaminants in aquatic systems at some time. Those species that have already experienced adverse impacts are likely to be more at risk in general than those species that have not yet been adversely affected by contaminants in the Great Lakes. A large part of the higher risk may result from higher exposure levels. The review and selection of representative mammalian species and representative avian species are summarized in Sections IV.A and IV.B below. The full analyses are presented in U.S. EPA (1995a).

IV.A. Selection of Mammalian Species

There are only three mammal species in the Great Lakes region and much of North America that are likely to be exposed to contaminants through aquatic prey: mink (*Mustela vison*), river otter (*Lutra canadensis*), and raccoon (*Procyon lotor*). Raccoons tend to be omnivorous, consuming both plant materials and animal matter; they rarely capture large fish (U.S. EPA, 1993d). In contrast, mink and otter can capture large fish, and river otter feed almost exclusively on aquatic prey (U.S. EPA, 1993c). These dietary habits suggest that the mink and otter are likely to be more exposed than raccoons. This suggestion is supported by comparative studies of contaminant levels in mammals. Studies of mercury contamination in furbearing mammals by Sheffy and Amant (1982) and by Wren et al. (1980) indicate that raccoons tend to exhibit lower concentrations of mercury in their tissues than both mink and otter captured in the same areas. Based on this information, the mink and otter were selected as representative of mammalian species most likely to experience the greatest exposures to bioaccumulative contaminants through the aquatic food web in the Great Lakes basin.

IV.B. Selection of Avian Species

In contrast to the situation with mammals, there are numerous species of fish-eating birds in the Great Lakes basin. Thus, selection of species likely to be most exposed required additional considerations. First, all species of birds that consume aquatic prey that could inhabit the Great Lakes region were identified on the basis of the overlap of their

habitat requirements and range with the Great Lakes, as indicated in the *National Geographic Field Guide to the Birds of North America* (NGS, 1987). In addition, a literature review was conducted to identify those avian species for which documented reports of adverse effects in the field have been attributed to toxic chemicals in their food. These species included bald eagles, osprey, herring gulls, ring-billed gulls, black-crowned night herons, common terns, Forster's tern, Caspian tern, and the double-crested cormorant (Colborn, 1991; Environment Canada, 1991; Gilbertson et al., 1991; Peakall, 1988; see U.S. EPA, 1995a). Most of these species are colonial nesters. For all species with documented adverse impacts in the past, a literature search was conducted, and body weights, food ingestion rates, dietary composition, and likely trophic level of prey were estimated. Of the impacted species, the ones most likely to suffer the highest exposures to contaminants that bioaccumulate in aquatic food chains were judged to be the bald eagle, herring gull, and common tern (U.S. EPA, 1995a).

In addition, several species for which adverse impacts have not been documented, but which appear to have diets and food ingestion rates similar to species that have been affected, were considered further. These included the belted kingfisher, common (and American) merganser, and the red-breasted merganser. These species in general are solitary nesters, and the lack of documented adverse effects may simply reflect the fact that these difficult-to-study species have not been studied in the Great Lakes basin. Of these, the species likely to suffer the highest exposure levels was judged to be the belted kingfisher (U.S. EPA, 1995a). Table 1 summarizes the exposure parameters for all of the wildlife species considered above. Supporting documentation for the values are in U.S. EPA (1995a, b).

The avian species selected as those likely to suffer the highest exposure levels to bioaccumulative contaminants in aquatic food chains were the bald eagle, herring gull, common tern, and belted kingfisher. Because the common tern and belted kingfisher are similar in size, and can be very similar in diet, the belted kingfisher was selected to represent the two species and to provide an example of an unstudied species that may be at risk.

It is important to remember that these species were selected as likely to be most exposed on the basis of food ingestion rates and diet. This selection did not cover some aspects of foraging behavior that can affect exposure. For example, the double-crested cormorant may capture more benthic fish than the other bird species because of its diving abilities. If the benthic fish have higher body burdens of contaminants than other fish because most bioaccumulative contaminants are in the sediments, the cormorants could be more exposed than birds feeding on higher trophic level fish that are not directly associated with the sediments. Although considered inappropriate for developing wildlife criteria applicable to the Great Lakes basin as a whole, this type of information may be worth considering for local areas with specific and well-defined sites of contaminated sediments.

Table 1. Exposure Parameter Values and Trophic Level of Prey for Species Potentially at Risk from Bioaccumulative Contaminants in the Great Lakes (U.S. EPA, 1995a).

Species	Adult Body Weight (kg)	Ingestion Rate (kg/kg-d)	Average Trophic Level of Aquatic Prey: Percent of Diet (wet weight)
mink	male: 1.0 female: 0.55 average: 0.80	male: 0.22 female: 0.24 average: 0.23	Terrestrial: 10% Aquatic TL 3: 90%
otter	male: 8.1 female: 6.7 average: 7.4	male: 0.16 female: 0.17 average: 0.17	TL 3: 80% TL 4: 20%
kingfisher	both: 0.15	both: 0.45	TL 3: 100%
osprey	male: 1.4 female: 1.8	both: 0.19	TL 3: 100%
bald eagle	male: 4.1 female: 5.2 average: 4.6	both: 0.12	Fish: 92% (TL 3: 80%) (TL 4: 20%) Birds: 8% (herring gull: 70%) (non-aquatic: 30%)
herring gull	male: 1.23 female: 1.00 average: 1.1	male: 0.25 female: 0.26	Fish: 90% TL 3: 80% TL 4: 20% Terrestrial: 10%
ring-billed gull	male: 0.566 female: 0.471	male: 0.31 female: 0.33	Terrestrial: 25% Aquatic TL 3: 75%
black-crowned night heron	both: 0.850	both: 0.22	Aquatic: 30-80% TL 2: 20% TL 3: 80% Terrestrial: 20-70%
common tern	both: 0.120	both: 0.49	TL 3: 100%
Forster's tern	both: 0.158	both: 0.45	TL 3: 100%
Caspian tern	both: 0.661	both: 0.30	TL 3: 100%
double-crested cormorant	male: 1.82 female: 1.54	male: 0.18 female: 0.19	TL 3: 100%
American merganser (common merganser)	both: 1.27	both: 0.24	TL 3: 100%
red-breasted merganser	male: 1.135 female: 0.908	male: 0.25 female: 0.27	TL 2: 10% TL 3: 90%

V. DETERMINATION OF THE MOST APPROPRIATE TEST DOSE FOR USE IN CALCULATING WILDLIFE VALUES

The selection of the TD for use in the calculation of individual species-specific WVs for each taxonomic class requires the application of best professional judgement in evaluating the results of all available studies.

Because the intent of wildlife criteria is to protect populations rather than individuals, the measurement endpoints assessed in the study from which a TD is derived are defined as a set of frank effects which could reasonably be expected to have implications at the population level. Therefore, for the purposes of wildlife criteria derivation (and as defined in the Definitions section of this document), acceptable endpoints are those which affect reproductive or developmental success, organismal viability or growth, or any other endpoint which is, or is directly related to, a parameter that influences population dynamics. In evaluating the studies which assess acceptable endpoints, preference should be given to studies which assess effects on developmental or reproductive endpoints because, in general, these are more important endpoints in ensuring that a population's productivity is maintained.

Another restriction placed on the study from which a TD is obtained is that it provide a defensible, chemical-specific, dose-response curve in which cause and effect are clearly established. In order to ensure this evaluation criterion is met, the methodology requires that any study used to obtain a TD be available in the peer-reviewed literature.

The duration of the study must also be considered in evaluating the available data. In terms of the duration of the study from which the TD is derived, the use of acute data with the application of an acute-to-chronic conversion ratio for wildlife criteria derivation was considered. However, the empirical relationship between acute endpoints and chronic endpoints is too uncertain to use in the derivation of wildlife values for application in this regulatory context.

Because this methodology is derived specifically for bioaccumulative pollutants, the duration of the study is required to be subchronic or chronic. General definitions of both subchronic effect and chronic effect are provided in the Definitions section of this document. More detailed definitions of subchronic and chronic, relating them to the lifespan of the test species are provided in *Appendix A: Uncertainty Factors* to the *GLWQI Technical Support Document for Human Health Criteria and Values*. This guidance indicates that, typically, continuous or repeated exposure for a period of approximately 10% of the lifespan of the test species is considered subchronic and exposure for approximately 50% of the lifespan of the test species is considered chronic. For the evaluation of studies to use in the determination of a TD for derivation of wildlife criteria, the definitions of subchronic and chronic effects are not more quantitative because of the wide variety of species, with their associated variety in life-spans, which may be tested, in addition to the variety of chemical characteristics and associated modes of toxic action. Minimum durations for the study from which a test dose is derived are discussed in Sections V.A. and V.B. below. Whether a given study which meets these minimum duration requirements is interpreted as subchronic or chronic is dependent on both characteristics of the chemical as well as lifespan and life-stage of the test species. The

minimum duration requirements are established mainly to ensure that the TD is not based on a study with an insufficient length of exposure which could underestimate the potency a compound.

If more than one study within a given taxonomic class meets the above qualifications (i.e., the study evaluates an acceptable endpoint over a subchronic or chronic duration for the test species and provides a defensible, chemical-specific, dose-response curve) additional evaluation is required. If studies of equal quality which assess equally significant endpoints are available for both "wildlife" species as well as traditional laboratory animals, the preference for obtaining the TD is for the study with the "wildlife" species. This is because obtaining a TD from study which tested a "wildlife" species may minimize the uncertainties associated with interspecies extrapolations, depending on both the test species and the representative species. In addition, many traditional laboratory species (and especially rats and mice) are bred from a fairly homogeneous gene-pool. Use of a TD derived from a "wildlife" species is thought to provide a more realistic representation of the dose-response relationship which may occur in the natural environment.

When evaluating studies for a given taxonomic class which meet the minimum requirements to derive a TD for calculation of taxonomic class-specific wildlife values, field studies are preferred over laboratory studies. It is recognized that there are very few field studies which meet the qualifications outlined above, but when available, this data would be most appropriate for derivation of a criterion because it provides an estimate of impacts within a natural ecosystem.

Another consideration in evaluating available laboratory studies for determination of the appropriate TD is the route of exposure used in the study. Studies involving exposure routes other than oral may be considered only when an equivalent oral daily dose can be estimated and technically justified. The need for oral exposures relates to the nature of the chemicals of concern, which are bioaccumulative. As a consequence, exposures to the representative species will be predominantly through the food chain and involve uptake across the gastrointestinal tract with first-pass metabolism by the liver. Because the toxicokinetics of bioaccumulative chemicals, and the resulting delivery of the chemical to the site of action, is critically related to this exposure route, it is imperative that use of a toxicity study involving a different route of exposure be carefully critiqued and an approach to convert a non-oral dose to an oral dose be presented. Use of an oral route of exposure in the study from which the TD is determined should reduce uncertainty in extrapolating toxicity results from the laboratory to the field.

V.A. Study Duration Requirements for the Mammalian Study from which a Test Dose is Derived

For mammals, the study from which the TD is derived should be of 90 days or greater in length. The 90-day study duration for mammals is consistent with the minimum requirements established in the 1980 Human Health National Guidelines (U.S. EPA, 1980) and in Appendix C to 40 CFR Part 132 of the final GLWQI guidance, *GLWQI Methodology for Development of Human Health Criteria and Values.* In the development of human health criteria, the 90-day duration is considered to be the minimal time-span for

subchronic effects to emerge based on the life-span of a rodent. Although the test species used in the study from which a TD is derived may have a very different life-span than a rodent, it is reasonable to use the minimum 90-day study duration for mammalian wildlife, keeping in mind the minimum duration requirements are established to ensure that the potency of a compound is not underestimated.

In order to derive a taxonomic class-specific wildlife value for a Tier II wildlife value, the minimum study duration for mammals is 28 days rather than 90 days.

V.B. Study Duration Requirements for the Avian Study from which a Test Dose is Derived

For avian species, the study from which the TD is derived should be of 70 days or greater in length. However, for tests which assess effects of a contaminant on growth or mortality endpoints to chicks *only* exposed post-hatch, 28 days of exposure may be considered adequate for determination of a TD.

The 70-day study duration for birds is derived from the *Hazard Evaluation Division: Standard Evaluation Procedure -- Avian Reproduction Test* (U.S. EPA, 1986). This Standard Evaluation Procedure explains the procedures used to evaluate effects data submitted to the Office of Pesticide Programs and is available to other program offices for the evaluation of studies and scientific data. This document specifies procedures specific for reproduction tests in birds and requires that the test chemical be administered for at least ten weeks prior to the onset of egg laying. It also specifies a number of specific reproductive parameters which may be assessed in a reproductive study. These are presented here to indicate some acceptable endpoints for the determination of an appropriate TD for wildlife value derivation, but this should not be seen as a comprehensive list. These effects are: the number of eggs laid per hen; the percentage of eggs cracked; the percentage of viable embryos of the egg set; the percentage of live three-week embryos; the percentage of normal hatchlings of live three-week embryos; the number of 14-day-old survivors per hen and the percentage of 14-day-old survivors of normal hatchlings.

In order to derive an avian class-specific wildlife value for a Tier II wildlife value, the minimum study duration is 28 days, regardless of the endpoint or life stage of the test species.

V.C. Dose Conversions for Calculating the Test Dose

As indicated in Section III, the wildlife criterion is estimated on the basis of a NOAEL, or the highest ingested dose at which no significant adverse effects occurred. The dose is expressed in terms of mg contaminant per kg body weight per day (i.e., mg/kg-day, wet-weight). Many reports on the results of toxicity tests describe exposure levels for the test animals only in terms of the concentration of the contaminant in their diet. Two common expressions of dietary concentration are mg of contaminant per kilogram of diet (mg/kg diet), or parts per million in the diet (ppm diet). These two values are equivalent when ppm is determined on the basis of weight.

To convert a toxicity value expressed as ppm in the diet (or drinking water) to ingested doses expressed as mg/kg-day to calculate a wildlife value, the following equations can be used:

D (mg/kg-day) = C in diet (mg/kg-diet) x F (kg/day) / Wt (kg), or
D (mg/kg-day) = C in water (mg/L-water) x W (kg/day) / Wt (kg),

where

D = dose in mg of contaminant per kg of body weight per day,
C = contaminant concentration as mg/kg in the diet or mg/L in water,
F = food ingestion rate in kg of diet/kg body weight per day,
W = water ingestion rate (L/day), and
Wt = body weight in kg wet weight.

Thus, to convert doses expressed as ppm in the diet or in drinking water to an intake of contaminant expressed as mg/kg body weight per day, the body weight and food or drinking water ingestion rates are required. One should always use the body weights and feeding or drinking rates reported for the experimental animals in the toxicity test. If these values are not reported by the investigators, one must estimate the values, if possible, from other sources of information. Information sources and methods for estimating food and water ingestion rates are described below for mammals (Section V.C.1) and birds (Section V.C.2).

V.C.1. Mammals

Common mammalian laboratory species include various strains of rats, mice, hamsters, guinea pigs, and for chemicals of particular concern for human health, rhesus monkeys. U.S. EPA has developed values that can be used as default exposure parameter values for these (and a few other) laboratory mammals in the absence of information in the toxicity test report. U.S. EPA recommends consulting the *Recommendations for and Documentation of Biological Values for Use in Risk Assessment* (U.S. EPA, 1988) to estimate body weights and food and water ingestion rates for the strain, sex, and ages covered during the exposure period that are appropriate for the animals exposed to the test chemical in the toxicity test.

U.S. EPA's (1988) *Recommendations* do not, however, cover all animals for which body weights and ingestion rates may be needed to perform dose conversions for wildlife toxicity tests. First, U.S. EPA has not developed recommended exposure parameter values for all strains of small mammal commonly used in toxicity tests. Second, not all investigators report the strain of the animals in their test. To estimate exposure parameters for unspecified strains, a review of the strains for which data are presented in U.S. EPA (1988) is appropriate. Table 2 presents body weights and food ingestion rates for mice, some strains of rats, hamsters, and rhesus monkeys from U.S. EPA (1988). Given the variation in body size among different strains of laboratory animals, it can be difficult to determine which body weight/food ingestion rate values might be most appropriate for an animal of unspecified weight in the toxicity test under consideration. On the other hand, food ingestion rates normalized to body weight are much more uniform across strains. The final column in Table 2 illustrates this point by presenting food

Table 2. Laboratory Mammal Body Weights and Food Ingestion Rates.

Species	Strain: sex	(a) Body Weight (kg) (U.S. EPA, 1988)	(b) Food Ingestion Rate (kg/day) (U.S. EPA, 1988)	(c) Food Ingestion Rate (kg/kg-day) (c) = (b)/(a)
Mice	BAF1 mice: males	0.035	0.0061	0.17
	(mature) females	0.030	0.0055	0.18
	B6C3F1 mice: males	0.040	0.0067	0.17
	(mature) females	0.035	0.0061	0.17
	BAF1 mice: males	0.0261	0.0050	0.19
	(chronic) females	0.0222	0.0045	0.20
	B6C3F1 mice: males	0.0373	0.0064	0.17
	(chronic) females	0.0353	0.0061	0.17
	Mouse Food Ingestion Rate (mature)			0.17
Rats	Sprague-Dawley: males	0.60	0.040	0.067
	(mature) females	0.35	0.028	0.080
	Wistar: males	0.50	0.035	0.070
	(mature) females	0.32	0.026	0.081
	Fisher: males	0.40	0.031	0.078
	(mature) females	0.25	0.022	0.088
	Sprague-Dawley: males	0.523	0.036	0.069
	(chronic) females	0.338	0.027	0.080
	Wistar: males	0.462	0.034	0.074
	(chronic) females	0.297	0.025	0.084
	Fisher: males	0.380	0.030	0.079
	(chronic) females	0.229	0.021	0.092
	Rat Food Ingestion Rate (mature or chronic)			0.080
Hamster	Golden Syrian males	0.090	0.014	0.16
	(mature) females	0.096	0.015	0.16
	Golden Syrian males	0.082	0.013	0.16
	(chronic) females	0.088	0.014	0.16
	Hamster Food Ingestion Rate (mature or chronic)			0.16
Rhesus Monkey	Rhesus monkey males	12	0.46	0.026
	(mature) females	9	0.37	0.041
	Rhesus monkey males	10.9	0.43	0.039
	(chronic) females	8.0	0.33	0.041
	Female Rhesus Monkey Food Ingestion Rate			0.041

ingestion rates expressed as kg food ingested/kg body weight per day (i.e., kg/kg-day). It is easier to determine an appropriate value to use when the food ingestion rate is expressed this way. Also, it is easy to convert toxicity values expressed as ppm in the diet to doses expressed as mg/kg-day using food ingestion rates normalized to body weight:

$$D \text{ (mg/kg-day)} = C \text{ (mg/kg diet)} \times F \text{ (kg diet/kg-day)}$$

where

D = dose in mg of contaminant per kg of body weight per day,
C = contaminant concentration as ppm or mg/kg in the diet, and
F = food ingestion rate in kg of diet/kg body weight per day.

When the toxicity test animals are mink (*Mustela vison*), they are assumed to be farm-bred or "ranch" mink unless otherwise stated. These tend to weigh more than free-living mink (U.S.EPA, 1993d). On the basis of the body weight data provided in Table 3, the female mink used in toxicity tests were assumed to weigh 1 kg and the males were assumed to weigh 1.8 kg.

Table 3. Body Weights of Farm-reared or "Ranch" Mink.

	Body Weight (kg)	Reference/Location
males females average	1.734 ± 350 SD (N=4) 0.974 ± 202 SD (N=12) 1.35	Hornshaw et al., 1983/Michigan
males females average	1.822 ± 0.095 SE (N=6) 0.872 ± 0.036 SE (N=6) 1.35	Bleavins and Aulerich, 1981/Michigan

Estimates of food ingestion rates of captive mink range from about 12 percent to 16 percent of the adult body weight per day (Aulerich et al., 1973; Bleavins and Aulerich, 1981). Thus, mink used in toxicity tests were assumed to consume about 15 percent of the adult body weight per day (Aulerich et al., 1973; Newell et al., 1987). For a female mink weighing 1 kg, this would equal 150 grams of food per day.

For other mammalian species used in toxicity tests that are not covered in U.S. EPA's (1988) *Recommendations*, the open literature can be used to identify body weights. Food and water ingestion rates can be estimated from the appropriate allometric equations presented in U.S. EPA (1993c), which include the allometric equations from Caulder and Braun (1983) and Nagy (1987) referenced in Appendix D to Part 132 of the Final GLWQI guidance. It is important to note the water content of the diet; dry laboratory chows tend to contain 10 percent water (Altman and Dittmer, 1972), whereas fresh fish and meat tend to contain approximately 75 percent water (U.S. EPA, 1993c).

V.C.2. Birds

The most common avian laboratory species include the mallard (*Anas platyrhynchas*), Japanese quail (*Coturnix japonica*), domestic chickens (often white leghorn) (*Gallus domesticus*), and ring-necked pheasants (*Phasianus colchicus*). U.S. EPA has not previously developed recommended exposure parameter values to use as default values for these birds in the absence of information in the toxicity test report. In this section, therefore, some guidance is provided on how to identify exposure parameter values for birds tested in the laboratory. In addition, the exposure parameter values that U.S. EPA has identified for the common avian laboratory species are provided. Table 4 presents body weights and food ingestion rates for adult mallards, Japanese quail, domestic chickens, and ring-necked pheasants.

Table 4. Body Weight and Food Ingestion Rates for Common Avian Laboratory Species.

Species	Body Weight (kg)	Reference	Food Ingestion Rate (kg/kg-day)[a]	Reference
mallard	1	Delnicki and Reinecke, 1986; U.S. EPA 1993c,d	0.060 (lab chow)	Nagy, 1987[b]
ring-necked pheasant	females: 0.95 males: 1.3 average: 1.1	Nelson and Martin, 1953	0.060 (seeds) 0.056 0.058	Nagy, 1987[b]
Japanese quail	0.12	Davison et al., 1976; Altman and Dittmer, 1972	0.090 (seeds)	Nagy, 1987[b]
chicken	2.0	Scott et al., 1976	0.067 (seeds)	Medway and Kare, 1959

[a] Expressed as kg fresh food/kg body weight per day (wet weight).
[b] Estimated from Nagy's (1987) allometric equation for estimating dry food ingestion rates for free-living non-passerine birds by assuming 10% water content of the laboratory feed (Altman and Dittmer, 1972).

If body weights or food or water ingestion rates are not reported for the toxicity test, data might be obtained from other sources; however, data from the toxicity test itself should always be used to the extent possible. It is important to identify the age of the bird over the experimental period. One can usually assume that adult body weights do not change (unless changes are reported by the investigators). Toxicity tests that begin with young birds require information on the size of the bird at the beginning and at the end of the test, if the rate of body weight increase is roughly linear. For growth rates that are nonlinear over the ages tested, a time-weighted average body weight would be most appropriate. Table 5 presents data on how body weight and food ingestion rates change with age for white leghorn chicks.

Table 5. Food Ingestion Rates of Growing White Leghorn Female Chickens (Diet Consists of 9% Water; Medway and Kare 1959).

Age (weeks)	N	Body Weight (g)		Food Ingestion Rate (g/chick-day)		Food Ingestion (g/g-day)
		Mean	± SD	Mean	± SD	
1	08	61.9	± 9.3	10.4	± 1.2	0.17
2	96	102.8	± 14.7	16.9	± 2.0	0.16
3	66	153.5	± 18.9	20.5	± 1.8	0.13
4	74	250.2	± 32.4	32.8	± 4.8	0.13
6	50	384.4	± 45.6	38.8	± 7.4	0.10
8	40	578.3	± 39.0	49.5	± 6.0	0.086
16	20	1,293.4	± 138.2	75.5	± 20.7	0.058
32	20	2,035.2	± 199.6	136.2	± 27.9	0.067

Body weights of the adults of many wild species of birds can be found in Dunning (1984), but most of these species are not used in toxicity tests. Information on additional species, if required, would require a search through the existing literature.

If measured values for food ingestion rates under captive conditions are not available in the open literature, one can estimate food ingestion rates on the basis of free living metabolic rate (FMR) or existence metabolic rate (EMR) (depending on the species) and information on the caloric content of the diet. For birds whose normal activities are significantly curtailed by captivity, (e.g., birds of prey, seabirds, passerines), EMR would be more appropriate than FMR. For birds that normally do not fly often, for example, most gallinaceous birds, FMR and EMR may not be significantly different, and either could be used. If necessary, EMR or FMR can be estimated on the basis of body weight from an allometric equation for the appropriate group of birds, as described in U.S. EPA (1993c, Chapter 4). When estimating food ingestion rates, it is important to note the water content of the diet. For example, seeds and dry laboratory chows generally contain only 9 to 10 percent water (U.S. EPA, 1993c, Table 4-2; Altman and Dittmer, 1972), while diets comprised of fresh meat can be 75 percent water (U.S. EPA, 1993c, Table 4-1).

VI. THEORY AND DETERMINATION OF APPROPRIATE UNCERTAINTY FACTORS FOR CALCULATION OF WILDLIFE VALUES

In applying Equation 5 above, uncertainty factors are applied to adjust the TD for interspecies differences in toxicological sensitivity (UF_A), subchronic to chronic extrapolations (UF_C), and LOAEL to NOAEL extrapolations (UF_L). The purpose of this section is to present the scientific basis for the ranges of the uncertainty factor values

recommended by U.S. EPA for use in deriving wildlife values. There is also discussion of an uncertainty factor to adjust for protection of individuals within a given population (UF_I) as discussed in Appendix F, Procedure 1.A: *Requirements for Site-specific Modifications to Criteria and Values* to 40 CFR Part 132 of the final GLWQI guidance.

VI.A. The Interspecies Uncertainty Factor (UF_A)

VI.A.1. Purpose and Recommended Range

In the derivation of wildlife values, an interspecies uncertainty factor ranging from 1 to 100 is applied to the NOAEL to account for uncertainties when extrapolating toxic effects across species (Appendix D of 40 CFR Part 132). This factor is typically used when adequate toxicity data are lacking for one or more of the representative species. The magnitude of the uncertainty factor chosen is based on the physicochemical, toxicokinetic, and toxicodynamic properties of the chemical of concern as well as the amount and quality of data available. The interspecies uncertainty factor is not intended to account for differences in ingested dose between the test and representative species since these factors (body weight, food consumption and water consumption rates) are incorporated into the derivation of the wildlife value.

VI.A.2. Theoretical Basis: Toxicokinetic and Toxicodynamic Differences

The toxicological basis of observed variability in species sensitivity to a given toxicant can be grouped into two broad categories: toxicological differences that are related to variability in *toxicokinetics* and toxicological differences that are related to variability in *toxicodynamics.* Toxicokinetics refers to processes that determine the delivery of a toxicant to the site of action. Such processes include absorption of the toxicant, its distribution to the target tissues, its metabolism, and its elimination from the organism. Toxicodynamics refers to factors that determine the magnitude of response of target tissue to a given degree of exposure at the site of toxic response. Toxicodynamic processes are specific to the mechanism of action of the chemical and can include factors such as binding of the toxicant (or metabolite) to enzymes, DNA sequences or other receptors that closely mediate the toxic response at the site of action. It is the combined effect of both toxicokinetics and toxicodynamics that determines the overall sensitivity of a species to a toxicant. Additional discussion of toxicokinetic and toxicodynamic differences across species is provided in U.S. EPA, 1995c.

VI.A.3. Allometric Scaling

The current state of the science indicates that some of the variability in sensitivity across species can be related to some rather simple and general quantitative patterns of anatomy and physiology of different sizes of mammalian species. It has been shown that the absolute *rates* of such processes, including basal metabolic rate, cardiac output, renal clearance, oxygen consumption, food consumption, water consumption, etc., tend to vary across species according to allometric scaling factors that can be expressed as a non-linear function of body weight (e.g., body weight$^{3/4}$ as endorsed in U.S. EPA, 1992a). The general form of these allometric equations is:

$$Y = a\ W^b \qquad (7)$$

where *b* is the power of body weight *(W)* to which attribute *Y* maintains a constant proportionality, *a*. A key point in this discussion of allometry is that the value of the allometric scaling factor, *b*, has been shown to be relatively constant for certain attributes relevant to toxicokinetics (Travis et al., 1990 and U.S. EPA, 1992a).

The scaling of physiological processes that relate to toxicokinetics can be drawn together in the concept of "physiological time". The concept of physiological time proposes that quantitative differences across mammalian species in physiological processes can be related to similar anatomical, physiological, and biochemical machinery operating at different rates in different size species. Thus, small species are considered to have faster "physiological clocks," whereby various processes stay in proportion to each other but are relatively sped up compared to larger species. This is consistent with the observation that life spans of mammals scale roughly by $W^{1/4}$. The relevance of the physiological time concept to toxicokinetic differences across species is that, in general, smaller species would be expected to process equal mg/kg body weight doses of toxicants at a faster rate than larger species, owing to faster metabolic, distribution, and elimination rates (when expressed on a per body weight basis) compared to larger species. This would imply, based solely on these generalized trends in toxicokinetic-related processes, that species of larger body mass would be more sensitive to the same mg/kg-body weight adjusted dose compared to smaller species, owing to slower metabolic, distribution, and elimination rates per gram body weight[1]. Clearly, chemical-specific and species-specific factors other than the generalized trends in anatomical and physiological differences represented by allometric scaling contribute to sensitivity differences across species. Therefore, when applicable data on toxicokinetic, toxicodynamic, and sensitivity differences between species are available, these data should be used to refine projections of allometric scaling when necessary.

The previous theoretical considerations of allometric scaling of toxicokinetically-related physiological processes, in addition to limited empirical evidence, summarized in U.S. EPA, 1992a, has led to the explicit use of allometric scaling in the estimation of toxicologically equivalent doses in U.S. EPA's human health cancer methodology and its implicit use in the human health noncancer methodology. When test results are expressed on a per body weight basis (i.e., mg/kg-d), the following equation can be used to assess an allometric scaling factor that can be used to address some of the general toxicokinetic differences that may occur between species. (The basis of this equation is discussed in detail in U.S. EPA, 1992a.)

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[1] It should be noted that toxicity data expressed in terms of concentrations of contaminant in environmental media (e.g., ppm in food) inherently adjust for differences predicted by allometry since food and water consumption rates also scale in approximate proportion to $W^{3/4}$.

$$TD_R = TD_T \times \left(\frac{Wt_T}{Wt_R}\right)^{\frac{1}{4}} \quad (8)$$

where:

TD_R = test dose scaled for the given representative species (mg/kg-d),
TD_T = test dose for the test species (mg/kg-d),
WT_T = body weight of test species (kg), and
WT_R = body weight of the given representative species (kg).

When applied to the derivation of human health criteria, this equation would yield an allometric scaling factor of 7 when extrapolating results from a mouse to a human *(i.e.,* the toxicokinetically equivalent dose would be 7-fold lower in humans than mice on a mg/kg-d basis). When applied to wildlife criteria, smaller factors typically would be derived, since the body weight differences among test and representative species are usually smaller compared to the differences between small laboratory mammals and human body weights. The largest allometric scaling factors that could be expected for the GLWQI wildlife criteria would involve potential extrapolation between the mouse and the river otter. In this example, a factor of 4 would account for allometric scaling from a mouse to a river otter (i.e., the toxicokinetically equivalent dose would be 4-fold lower in the otter than the TD in the mouse), assuming body weights of 0.03 kg and 7.4 kg, respectively.

It is very important to convey that allometric scaling according to body mass is based on broad differences in physiology between species that can mostly be related to general toxicokinetic processes. Allometric scaling does not necessarily account for all toxicokinetic and toxicodynamic differences that occur between species, owing to the many complexities of toxicokinetic and toxicodynamic processes that influence an organism's response to a toxicant. Significant departures from the predictions of allometric scaling have been noted for some individual chemicals, some up to two orders of magnitude in either direction. In addition, allometric scaling between typical test species and Great Lakes representative species, which have relatively similar body weights, may result in small allometric adjustment factors that are well within the error and uncertainty associated with the allometric scaling equations. Therefore, allometry should be used only as one part in the overall process of determining an interspecies UF. It should not supersede the use of available chemical-specific information on differences in sensitivity, toxicokinetics, and toxicodynamics across species.

VI.A.4. Empirical Basis: Variability in Acute Sensitivity

An analysis of the variability in acute sensitivity of birds and mammals (summarized in U.S. EPA, 1995c), was carried out because the available acute toxicity data represent the largest repository of information currently available for evaluating interspecies variability in sensitivity to toxicants.

Three analyses of avian acute toxicity data were carried out by analyzing the results of three studies (Schafer and Brunton, 1979; Hudson et al. 1984; and Hill et al. 1975) in which toxicity data for each study were collected from the same laboratory using

standardized experimental protocols. Each study tested a number of chemicals (21-48 chemicals in each study) and a number of species (4-13 species per chemical). Within each study and for each chemical, the variability in species sensitivity relative to the most sensitive species tested was illustrated by calculating "LD_{50} or LC_{50} ratios." These are the ratios of each species' LD_{50} or LC_{50} to the lowest LD_{50} or LC_{50} observed for that chemical. These analyses revealed that in 88 to 95 percent of the cases, the LD_{50} or LC_{50} ratios were less than a factor of 10, with a slightly smaller range for the dietary LC_{50}s.

To determine the variability in acute sensitivity of mammals, acute oral LD_{50}s were obtained and analyzed from the Registry of Toxic Effects of Chemical Substances (RTECS) database (NIOSH, 1993). Acute data for 58 chemicals with 5-8 species tested per chemical were analyzed in the same manner as described for birds. For all chemicals, greater than 50% of the LD_{50}s were within a factor of four of each other and 90% were within a factor of 20. A factor of 100 accounted for 96% of the LD_{50} ratios.

The analyses of variability in acute sensitivity provide support for the 1 to 100 recommended range for the interspecies uncertainty factor for both mammals and birds. However, in considering the results of the analyses of acute data presented above, several caveats apply. First, the number and type of avian and mammalian species tested in these analyses is fairly limited relative to the species that exist in nature. Also, the chemicals represented are heavily weighted by certain classes of pesticides with a similar mode of action. To the extent that chemicals and mechanisms of toxic action differ from those represented in this acute database, uncertainty exists in the applicability of these comparisons to other chemicals. Finally, these analyses are based on acute tests using high doses; therefore, the extent to which they reflect interspecies variability in responses from much lower, subchronic and chronic exposures is subject to uncertainty.

VI.A.5. Empirical Basis: Variability in Chronic Sensitivity

For this analysis, dietary, chronic and subchronic toxicity data were assembled from 174 separate toxicity studies on birds and mammals for four chemicals (cadmium, DDT and metabolites, dieldrin, and mercury). Specifics about the studies included in the database and the analyses that were carried out are provided in U.S. EPA, 1995c. In keeping with the wildlife criteria methodology, the endpoints assessed in the studies included in this analysis were reproductive, developmental, mortality, and growth. For this analysis, comparisons of chronic sensitivity were only made within the most specific endpoint, within specified exposure duration categories and when the exposure was to analogous life stages for each species. Chronic interspecies sensitivity ratios (like the LD_{50} and LC_{50} ratios in the acute analyses) were computed within each chemical-exposure-duration-life stage category.

A total of 122 interspecies comparisons of NOAELs for birds and mammals could be made within the various chemical-duration-endpoint-life stage categories. Because the database available for comparisons of species sensitivity is much smaller for chronic effects than it was for acute effects, the avian and mammalian data were combined in these analyses. Results from these analyses were expressed using three different approaches to gain the most information possible from the comparisons. These approaches are described in detail in U.S. EPA, 1995c. The results from the most

comprehensive analysis indicate an interspecies sensitivity ratio of 100 encompasses 84 percent of the ratios determined from this data set. There was also an indication of overall greater variability in the chronic sensitivity compared to the ranges observed in acute sensitivity, although other factors such as variability in chronic test designs likely contributed somewhat to the observed variability in chronic sensitivity.

VI.A.6. Guidance on Selecting the Interspecies Uncertainty Factor (UF_A) when Deriving Tier I Wildlife Criteria

The information on species sensitivity differences briefly summarized above and discussed in more detail in U.S. EPA (1995c), provides support for the 1-100 recommended range for the value of the interspecies uncertainty factor (UF_A). However, the actual selection of an interspecies UF for application to a particular situation must be made on a case-by-case basis and requires the use of best professional judgment. In determining the appropriate UF_A to apply in a given situation, consideration should be given to the physicochemical, toxicokinetic, and toxicodynamic properties of the chemical of concern and the amount and quality of available data. Generally, smaller values for UF_A (closer to 1) will be applied when toxicity data is available for a larger number of species within a given taxonomic class. Consideration should also be given to the taxonomic and physiological diversity of test species available and their relationship to the representative species. Comparative toxicity data on compounds which operate by the same mechanism of action should also be considered when determining the appropriate value for UF_A.

Allometric scaling of doses (expressed as mg/kg-d) can also be considered in the process of selecting an interspecies uncertainty factor based on toxicokinetic differences using equation 8 above. However, it should be recognized that allometric scaling may not accurately reflect the toxicokinetics of all chemicals nor encompass all the toxicodynamic differences among species. For example, based solely on predictions of allometric scaling, one would expect similar sensitivities between two taxonomically-related species of similar body weights, such as for starlings, *Sturnus vulgaris* (75g) and red-winged blackbirds, *Agelaius phoeniceus* (55g) which are both passerine species. However, when tested with acute, oral doses to 21 organophosphate and 9 carbamate pesticides, the red-winged blackbird was shown to be consistently more sensitive than the starling (Schafer and Brunton, 1979). For certain pesticides, acute oral LD_{50}s for red-winged blackbirds (mg/kg body weight) were more than 10-fold lower than LD_{50}s for starlings (e.g., coumaphos, bufencarb, carbofuran, mobam) and were at least 100-fold lower for diazinon, dichlofenthion, and trichloronat. Therefore, in determining an interspecies UF, allometrically derived TDs should be considered only as one component in the UF_A selection process and should be used in conjunction with chemical class-specific information on sensitivity, toxicokinetics, and toxicodynamics across species. This is consistent with the guidance provided in the U.S. EPA Science Advisory Board commentary (U.S. EPA, 1994) which stated that allometric relationships should not be the sole basis for selecting an interspecies UF. To see examples of how values for UF_As are determined, please refer to the *GLWQI Criteria Documents for the Protection of Wildlife: DDT; Mercury; 2,3,7,8-TCDD; PCBs* (U.S. EPA, 1995d).

To illustrate an approach to estimating the contribution of allometry to the value of UF_A, an example based on the avian PCB wildlife value is provided here. In this example,

the test species was a 1 kg pheasant and the three representative species were herring gull (1.1 kg), bald eagle (4.6 kg), and kingfisher (0.15 kg). The allometric adjustment of the TD between a pheasant and herring gull (1.0 kg vs. 1.1 kg) is negligible. The allometric adjustment factor of a TD between a pheasant and bald eagle would be:

$$\left(\frac{Wt_{pheasant}}{Wt_{bald\ eagle}}\right)^{0.25} = \left(\frac{1}{4.6}\right)^{0.25} = 0.68 \qquad (9)$$

The $UF_{A\ (bald\ eagle)}$ would be the inverse of this value (1 / 0.68), or 1.5. The allometric adjustment factor of a TD between a pheasant and kingfisher would be:

$$\left(\frac{Wt_{pheasant}}{Wt_{kingfisher}}\right)^{0.25} = \left(\frac{1}{0.15}\right)^{0.25} = 1.6 \qquad (10)$$

The $UF_{A\ (kingfisher)}$ would be the inverse of this value (1 / 1.6), or 0.65. These allometric adjustment factors were not applied in the derivation of the PCB avian wildlife values because their small magnitudes are well within the bounds of uncertainty and error associated with the allometric relationship.

As discussed in the PCB criterion document (in U.S. EPA, 1995d), toxicity data across a number of species were also examined, and based on best professional judgement, a UF_A of 3 was used to extrapolate from the pheasant to the eagle, to the herring gull and to the kingfisher. Because the body masses of the test species and representative species used in calculating the four GLWQI criteria (U.S. EPA, 1995d) are so similar, the largest allometric scaling factors that would be calculated are about two or less. Therefore, the use of allometric scaling in the derivation of these criteria is of limited utility. However, if the differences in body mass were large (e.g., extrapolating from a mouse to a moose), the use of allometry would provide much more significant insights.

VI.A.7. Guidance on Selecting the Interspecies Uncertainty Factor (UF_A) when Deriving Tier II Wildlife Values

When a Tier II wildlife value is derived, the value for UF_A may range from 1 to 1000. The larger range is provided to allow for extrapolation from the available data for one taxonomic class to a level thought to be protective of the taxonomic class for which the needed toxicity data was not available. However, at the discretion of the State or Tribe, the UF_A selected can be based solely on protection of the taxonomic class for which data was available (i.e., a UF_A between 1 and 100). In this case, a Tier II taxonomic class-specific wildlife value would be identical to a Tier I taxonomic class-specific wildlife value. In implementing this type of Tier II value, the State or Tribe should discuss the likelihood that a Tier II value of this type will or will not be protective of the other taxonomic class.

VI.B. The Subchronic-to-Chronic Uncertainty Factor (UF_s)

VI.B.1. Purpose and Recommended Range

Wildlife criteria that are derived according to the methodology presented in this rule (see Appendix D to 40 CFR Part 132 of the Final GLWQI guidance, *GLWQI Methodology for the Development of Wildlife Criteria*) are designed to protect wildlife from long-term exposures to toxic substances in the diet. However, in some situations, the NOAEL chosen to derive a wildlife value may be obtained from a study that tested organisms for a less than chronic (i.e., subchronic) exposure duration. In these cases, a subchronic-to-chronic uncertainty factor, UF_s, ranging between 1 and 10 is applied to the NOAEL to account for the possibility of greater toxicity of the substance to the test organisms had they actually been exposed over a chronic duration. The concept of a subchronic-to-chronic UF has previously been endorsed by U.S. EPA in the *Federal Register* for deriving human health criteria (U.S. EPA, 1985, and U.S. EPA, 1980). Since the conceptual basis of the UF_s is common to both wildlife and human health criteria, the reader is also referred to the discussion of the subchronic-to-chronic UF presented in Appendix A to the *GLWQI Technical Support Document Methodologies for Human Health Criteria and Values*. However, it should be noted that in contrast to the human health criteria methodology, the UF_s for wildlife criteria is *not* meant to be used to compensate for deficiencies in the study design not related to exposure duration.

VI.B.2. Technical Basis

It is well-recognized that subchronic toxicity studies may be conducted for exposure durations of insufficient length to measure adverse effects that would be observed in chronic tests of longer durations (National Academy of Sciences, 1977). Conceptually, there are several reasons why subchronic toxicity tests may be inadequate for detecting chronic adverse effects. First, exposure durations associated with subchronic studies may be too short to quantify adverse effects that result from long-term (chronic) accumulation of a toxicant at the site of action. Chronic exposure durations are particularly important for substances that require long time periods to reach equilibrium in the target tissues. Second, subchronic studies may not quantify adverse effects that result from long-term (chronic) failure of physiological compensation mechanisms that can mask adverse effects over shorter exposure durations associated with subchronic tests. Third, subchronic studies may not record latent adverse effects that can be manifested much later in an organism's lifespan. Finally, subchronic studies may simply fail to expose a particularly sensitive life stage of an organism that would be included in a chronic test.

Experimental support for the use of a subchronic-to-chronic uncertainty factor can be found in two published studies: Weil and McCollister (1963) and McNamara (1976). In the study by Weil and McCollister (1963), ratios of short-term (subchronic) to long-term (chronic) NOAELs were determined from tests on laboratory rats. Tests were performed with 33 different chemicals, including agricultural chemicals, food additives, chemicals used in water treatment processes, and chemical ingredients of food packaging materials. In these studies, subchronic tests typically lasted 90 days while chronic tests lasted approximately 2 years. The NOAELs used in the subchronic-to-chronic comparisons were determined from the most sensitive of a suite of whole organism, tissue-specific, and

biochemical endpoints for each chemical. For comparisons of subchronic and chronic NOAELs, Weil and McCollister determined that 50% of the 33 subchronic-to-chronic ratios had values of 2 or less. In addition, they reported that 97% of the ratios were 9 or below.

McNamara (1976) analyzed 82 studies (encompassing 126 compounds) along with data on the 33 compounds analyzed by Weil and McCollister (1963). The 41 additional comparisons performed by McNamara were taken mostly from studies of rats, with a much smaller number taken from studies of dogs and monkeys. In contrast to the analysis by Weil and McCollister (1963), McNamara's comparisons were based on tests performed by a wider array of researchers employing study designs of varying quality. Despite this difference, results from the two studies appear to agree reasonably well. McNamara reported that of the additional studies analyzed, 34 of the 41 subchronic-to-chronic ratios, 83% of the cases, were 1.0 or less. Some short-term effects were apparently reversible in the long-term, thus yielding ratios less than 1.0. McNamara further reported that 98% of the ratios for the 41 additional comparisons he made were less than 3, and all of the ratios were less than 7.

In an attempt to fill some gaps in the knowledge of subchronic-to-chronic extrapolations for avian species, additional analyses of avian chronic and subchronic toxicity data were evaluated for four chemicals (cadmium, DDT, dieldrin, and mercury), two of which are chemicals for which wildlife criteria were derived as part of the GLWQI (U.S. EPA, 1995d). As a starting point, an analysis of subchronic-to-chronic extrapolations were performed on mortality and growth endpoints for birds. Ratios of NOAELs from *short-term* (28-89 days), *intermediate* (90-180 days), and *long-term* (greater than 180 days) exposure durations were made within common chemical, species, endpoint and life stage categories. Further details of the analysis are provided in U.S. EPA (1995c). Based on the more comprehensive analysis of the data, results indicate about half of the long-term NOAELs were within a factor of 10 of the short-term NOAELs and 90% were within a factor of 20. As expected, smaller ratios were observed for intermediate-to-long-term NOAEL comparisons, where 50% of the ratios were within a factor of 1 and 90% were within a factor of 5. The somewhat higher variability observed in this analysis compared to those of Weil and McCollister (1963) and McNamara (1976) is likely due in part to greater variability in study designs evaluated in this analysis.

VI.B.3. Guidance on Selecting the Subchronic-to-Chronic Uncertainty Factor

While the preceding section provides support for the conceptual basis of a subchronic-to-chronic UF and suggest that a range of 1-10 is reasonable, two important issues remain that relate to when and how the UF_S is applied for deriving wildlife criteria. First and foremost is making a determination of when the application of a UF_S is necessary. This involves defining what exposure conditions constitute a reasonable definition of "chronic". As reviewed in *Appendix A: Uncertainty Factors* to the *GLWQI Technical Support Document for Human Health Criteria and Values*, a quantitative definition of chronic in terms of duration remains elusive. There have been various attempts to define chronic exposure in terms of a set duration of time, exposure across different life stages, or exposure to a given percentage of the expected lifespan of an organism with no clear standard emerging.

Qualitatively, chronic exposure can be defined as an exposure period of sufficient length to reveal most adverse effects that occur, or would be expected to occur, over the entire lifetime of an organism. However, the actual exposure duration required to detect chronic effects in a given test will depend on several factors, including the toxicokinetic properties of the substance, mechanism of toxic action, lifespan of the organism, indications of possible latent effects, whether critical life stages of the organism were exposed, etc. As discussed in *Appendix A: Uncertainty Factors* to the *GLWQI Technical Support Document for Human Health Criteria and Values*, a reasonable working definition of a minimum chronic exposure duration was defined as at least 50% of the lifespan of the organism. For rats and mice, this would correspond to about 52 and 45 weeks, respectively. For most bird species used in toxicology studies, this duration is also reasonable, given that their life expectancy in the wild (which is a function of their potential lifespan and other factors such as predation, disease, etc. that contribute to increased mortality rates) is in the range of 2 years. However, this working definition of a minimum chronic exposure duration should not be viewed as inflexible, and is subject to modification depending on chemical, species, and test-specific considerations as discussed above.

Once the determination has been made to apply a subchronic-to-chronic uncertainty factor, actual selection of the UF_S will require scientific judgement through the consideration of several factors, including length of the exposure, toxicokinetic properties of the chemical (e.g., bioaccumulation, metabolism, etc.), indications of latent effects, whether important life stages were tested, mechanism of toxic action, the lifespan of the organism, etc. Unless data indicate otherwise, larger values of the UF_S will generally be required for extrapolations from studies that are shorter relative to the lifespan of the organism compared to studies of durations that are longer relative to the lifespan of the organism. Thus, the concept of a "sliding scale" for selecting the UF_S based on the duration of exposure, as discussed in *Appendix A: Uncertainty Factors* to the *GLWQI Technical Support Document for Human Health Criteria and Values*, is appropriate unless other pertinent data indicate otherwise. Selection of the UF_S should include consideration of the amount of time required for the chemical to reach equilibrium in the tissues. All else being equal, chemicals that require longer time periods to reach steady-state will require a larger UF_S compared to chemicals that reach steady-state relatively quickly. To see examples of how values for UF_S are determined, please refer to *GLWQI Document for the Protection of Wildlife; DDT, Mercury, 2,3,7,8-TCDD, and PCBs* (U.S. EPA, 1995d).

VI.C. The LOAEL-TO-NOAEL Uncertainty Factor (UF_L)

VI.C.1. Purpose and Recommended Range

In some circumstances, a wildlife value may be calculated from a study that only provides a LOAEL without a corresponding NOAEL. This situation occurs when all the treatments of the chosen study caused a significant adverse effect relative to the control treatment. In these cases, U.S. EPA recommends dividing the available LOAEL (which is actually an unbounded LOAEL as defined in Section II) by a LOAEL-to-NOAEL uncertainty factor, UF_L, to reduce it to a dose range corresponding to the expected NOAEL. U.S. EPA's recommended range for the UF_L is from 1 to 10. Use of a LOAEL-to-NOAEL

uncertainty factor has been endorsed by U.S. EPA in the *Federal Register* for Water Quality Criteria Documents (U.S. EPA, 1980) and in the National Drinking Water Regulations (U.S. EPA, 1985).

VI.C.2. Technical Basis

Dourson and Stara (1983) conducted an analysis of chronic and subchronic rat data presented by Weil and McCollister (1963) that was designed to evaluate extrapolations from the LOAEL to the NOAEL. In their study, Dourson and Stara (1983) compared bounded LOAELs to their corresponding NOAELs for 33 different chemicals, including agricultural chemicals, food additives, chemicals used in water treatment processes, and chemical ingredients of food packaging materials. Of the 52 LOAEL-to-NOAEL comparisons made, 96% of the LOAEL-to-NOAEL ratios were 5 or less and all ratios were 10 or less.

An analysis similar to that conducted by Dourson and Stara (1983) was performed using additional chronic toxicity data for a variety of avian and mammalian species based on four chemicals (cadmium, DDT, dieldrin, and mercury), two of which are chemicals for which GLWQI wildlife criteria exist (more detail is provided in U.S. EPA, 1995c). Ratios of bounded LOAELs to their corresponding NOAELs were determined for a variety of growth, reproductive, developmental, and mortality endpoints. Of the 275 LOAEL-to-NOAEL ratios, more than half were less than or equal to 3, and 97% were less than or equal to 10. The ratios are slightly higher in this analysis because many of the studies evaluated used wider dose-spacings than did the studies in Dourson and Stara's analysis.

It should be recognized that the range of LOAEL-to-NOAEL ratios derived from these two analyses (*i.e.,* ratios of "bounded" LOAELs to NOAELs) are determined by the dose-spacing of the experiments from which they were derived. Since a LOAEL-to-NOAEL uncertainty factor is applied in situations where the available LOAEL is unbounded (for which a corresponding NOAEL could not be determined), it is implicitly assumed that the unbounded LOAEL is reasonably close to the dose-response threshold for the endpoint being evaluated. To the extent that an available unbounded LOAEL reflects a response level that is substantially greater than the dose-response threshold, additional uncertainty is introduced in the extrapolation down to the NOAEL. Subsequently, the magnitude of the LOAEL-to-NOAEL ratio required would increase and could conceivably extend beyond the 1 to 10 range derived from previous two analyses, depending on the slope of the dose-response curve for the endpoint being evaluated.

VI.C.3. Guidance on Selecting the LOAEL-to-NOAEL Uncertainty Factor

The previous analyses provide general support for the recommended range of 1 to 10 for the UF_L. In cases where a NOAEL cannot be quantified and only an unbounded LOAELs is available, determination of the appropriate value for the UF_L must be done on a chemical-specific and test-specific basis with the use of best professional judgement. In selecting the value of the UF_L, both the magnitude of the response observed at the unbounded LOAEL and the characteristics of the dose-response curve for the endpoint which the study evaluated should be considered.

In considering the magnitude of the response at the unbounded LOAEL, a smaller value for the UF_L (closer to 1) could be used for unbounded LOAELs that are judged to be at or near the dose-response threshold for the particular endpoint being evaluated. Since a value greater than one is applied for UF_L in cases where the dose-response threshold is unknown, estimates of the magnitude of the response associated with dose-response thresholds from other studies for the same endpoint--perhaps for a similar chemical or species--may be useful in evaluating the proximity of the unbounded LOAEL to a threshold response level.

In evaluating the characteristics of the dose-response curve, particular attention should be paid to the steepness of the slope. All else being equal, unbounded LOAELs from dose-response curves with steeper slopes would require smaller UF_L's compared to unbounded LOAELs derived from shallow slopes. However, this recommendation may not be applicable for a shallow dose-response curve if it is judged that an unbounded LOAEL is approaching a NOAEL threshold. To see examples of how values for UF_L are determined, please refer to *GLWQI Document for the Protection of Wildlife; DDT, Mercury, 2,3,7,8-TCDD, and PCBs* (U.S. EPA, 1995d).

It should also be noted that consideration of the severity of effects (e.g., more severe liver cell necrosis vs. less severe fatty infiltration of the liver) is incorporated into the selection of the LOAEL-to-NOAEL uncertainty factor for human health criteria, where protection of individuals is desired (see *Appendix A: Uncertainty Factors* to the *GLWQI Technical Support Document for Human Health Criteria and Values*). However, consideration of severity of effects is not to be considered when determining the value of UF_L for deriving wildlife values since a more narrowly defined set of frank effects (*e.g.,* growth, reproductive and developmental impairment) is used in the context of protecting populations.

VI.D. The Intraspecies Uncertainty Factor (UF_I)

VI.D.1. Purpose and Recommended Value

In situations where protection of sensitive individuals within a population of birds or mammals is required, U.S. EPA recommends an intraspecies uncertainty factor of 10 be applied to the NOAEL on a site-specific basis. Such protection is required for the protection of endangered species and discussed in Appendix F, Procedure 1.A: *Requirements for Site-specific Modifications to Criteria and Values* to 40 CFR Part 132 of the final GLWQI guidance. States and Tribes that choose to protect sensitive individuals for other reasons, may also use this intraspecies uncertainty factor. This uncertainty factor is intended to reduce the chosen NOAEL, which reflects the response of *average* individuals within a tested population, to a dose level that is protective of the more sensitive individuals of the population. As discussed in Appendix F, Procedure 1.A: *Requirements for Site-specific Modifications to Criteria and Values* to 40 CFR Part 132, the intraspecies uncertainty factor is intended for application to a NOAEL based on endpoints closely related to effects on populations (e.g., reproductive, growth, developmental, or lethal effects) rather than more subtle effects on individuals (e.g., biochemical responses, behavioral changes). The intraspecies uncertainty factor for wildlife is analogous to that

endorsed by U.S. EPA and the National Academy of Sciences (NAS, 1980) for deriving human health criteria.

VI.D.2. Technical Basis

Conceptually, the basis of intraspecies variability in sensitivity can be attributed to *individual* differences in toxicokinetics (factors affecting the magnitude and duration of toxicant exposure at target site) and toxicodynamics (factors affecting the degree of response at the target site to a given exposure level). It is the combined effect of individual toxicokinetic and toxicodynamic differences that determines the overall variability in susceptibility of individuals of a species to a toxicant. Sources of inter-individual variability in toxicokinetics and toxicodynamics include, among other factors, genetic heterogeneity, nutritional condition, changes in physiology with age, differences related to the sex of the organism, and disease. Examples of inter-individual variability in toxicokinetics and toxicodynamics that pertain to human subjects are reviewed in detail in U.S. EPA (1995c) and in *Appendix A: Uncertainty Factors* to the *GLWQI Technical Support Document for Human Health Criteria and Values*.

Experimental support for an intraspecies uncertainty factor for wildlife criteria is mostly limited to acute toxicity data extrapolations, where characterization of the dose-response curve has been relatively standardized. In a re-analysis of rat LD_{50} data presented by Weil (1972), Dourson and Stara (1983) analyzed 490 probit, log-dose slopes in an attempt to provide scientific support for the intraspecies uncertainty factor used to derive noncancer human health criteria. Although their analysis was originally intended to support an intraspecies UF of 10 for humans, its basis on rat data also make it applicable to mammalian wildlife criteria. Dourson and Stara (1983) report that approximately 92% of the probit, log-dose slopes were 3 or greater, which corresponds to intraspecies uncertainty factors of approximately 10 or less when extrapolating three probits below a median lethal response (e.g., from an LD_{50} to an $LD_{0.13}$). However, Dourson and Stara caution that their results may be under-protective for humans by acknowledging that the population on which their results are based (i.e., laboratory rats), may be more homogenous in their responses compared to those of the human population. Similarly, it seems reasonable to expect responses of laboratory rats to be more homogenous than responses of natural populations of wildlife, which wildlife criteria are designed to protect.

Intraspecies variability in sensitivity was also investigated for birds using a similar approach as described by Dourson and Stara (1983). Analysis of intraspecies variability in acute sensitivity was performed on dietary LC_{50} data reported by Hill et al. (1975) for four species of birds (bobwhite quail, Japanese quail, ring necked pheasant, and mallard) and 88 chemicals (mostly organophosphate, organochlorine pesticides and PCBs). Based on the reported log concentration-probit slopes for each species-chemical combination, extrapolations were made from the median lethal concentration (50% lethality) to a concentration corresponding to a response rate for the most sensitive individuals (e.g., a 1% lethality or LC_1). Of the 248 extrapolations made, about 75% of the estimated LC_1 values are within a factor of four of the median lethal response and 95% are within a factor of 10.

In an attempt to evaluate intraspecies variability in sensitivity from longer exposure durations that are more applicable to wildlife criteria, recent data were evaluated that characterize the dose-response-time surface from 28-day dietary tests on bobwhite quail (see Shirazi et al., 1994). In their study, Shirazi et al. exposed adult and juvenile quail to seven chemicals (two organochlorines, a carbamate, an organophosphate, and three rodenticides) in the diet for 28 days. Based on their model and reported parameter values, comparisons of reported 28-day LC_{50} values to estimated 28-day LC_1 values were performed. Of the 13 LC_{50} to LC_1 ratios calculated, nine were less than 10. The two highest LC_{50} to LC_1 ratios (from two rodenticides) were near 60.

The previous experimental analyses of intraspecies variability in sensitivity provide limited, indirect experimental support for an intraspecies UF and indicate that the majority of extrapolations to the more sensitive individuals are within a factor of 10 of the median response. However, several caveats to the empirical analysis described above should be noted. First, because quantitative analyses of the chronic dose-response curves are rare, much of the experimental evidence is based on acute exposures at high doses, all of which are for measures of mortality. Therefore, the relationship between intraspecies variability observed in these data to variability in other chronic responses (such as reproductive and developmental effects) at lower doses is unclear. Second, comparisons to extrapolated low response rates (e.g., $LC_{0.13}$ or LC_1) are inherently uncertain, and no attempt has been made in this evaluation to characterize this uncertainty. Finally, intraspecies variability in sensitivity has been characterized for relatively few species, and those are somewhat taxonomically removed from the GLWQI representative wildlife species. Therefore, these analyses suggest that using an intraspecies uncertainty factor of 10 is reasonable and is not likely to be unduly conservative.

VII. DETERMINATION OF EXPOSURE PARAMETER VALUES

Exposure parameters for deriving a wildlife value are of two types: (1) chemical-specific BAFs and (2) wildlife species-specific weights, and food and water ingestion rates. Section VII.A. describes the chemical-specific exposure parameters, Section VII.B. describes the species-specific trophic levels, Section VII.C. describes how to identify or estimate values for the exposure parameters, and Section VII.D. describes the derivation of the exposure parameter values for the representative wildlife species used to derive the wildlife criteria.

VII.A Chemical-specific Bioaccumulation Factors and Biomagnification Factors for DDT and metabolites; Mercury; 2,3,7,8-TCDD; and PCBs

A BAF is necessary to estimate the concentration of the chemical in the wildlife food source based on its concentration in the water source. Appendix G to the *GLWQI Technical Support Document for the Procedure to Determine Bioaccumulation Factors* presents the baseline BAFs from which BAFs for the derivation of wildlife values can be calculated. The procedure to derive these BAFs is specified in Appendix B to Part 132 of the final GLWQI guidance, *Great Lakes Water Quality Initiative Methodology for Deriving Bioaccumulation Factors.*

In addition to BAFs for aquatic prey of wildlife species, the eagle also consumes piscivorous birds. To estimate the concentration of contaminant in piscivorous birds (i.e., gulls) which are in turn consumed by eagles, a BMF is needed. The BMFs used for the derivation of wildlife criteria for the four chemicals for which wildlife criteria exist are presented in Appendix K to *GLWQI Technical Support Document for the Procedure to Determine Bioaccumulation Factors*, entitled *Determination of BAFs for DDT and Metabolites and Biomagnification Factors for the Derivation of Wildlife Criteria.* The BMFs are the ratio of the concentration of a contaminant in the gulls to the concentration in their prey fish.

VII.B Species-specific Trophic Levels

To determine trophic levels for the aquatic prey of the representative wildlife species, the dietary habits of both the wildlife species and their prey were investigated. The concept of numerical average trophic level was used to account for diets that represent prey from different trophic levels in a food web. For example, if half of an animal's diet came from trophic level 3 and half from trophic level 4, the numerical average trophic level for that animal's prey would be 3.5. Mearns et al. (1981) introduced the concept of numerical average trophic levels for marine predators that feed at more than one level, and Sanger (1987) has applied this concept to seabirds that feed at more than one trophic level (Hobson, 1990). Trophic levels were estimated first for the aquatic prey and then for the wildlife species that feed on them, as described below.

Trophic levels for aquatic prey species were determined from a literature search on their dietary habits and the dietary habits of their prey. For aquatic species that change their dietary habits as they grow in size (e.g., trout, perch), the species was categorized into two to five size-classes to reflect the change in dietary habits. For a single dietary study (i.e., one species at one location over a specified time frame), the weighted average trophic level for the species (by size class) is calculated as:

$$ATL = [(TL_{prey\text{-}1} * \%V_{prey\text{-}1}) + (TL_{prey\text{-}2} * \%V_{prey\text{-}2}) \ldots (TL_{prey\text{-}n} * \%V_{prey\text{-}n})] + 1$$

where

ATL = weighted average trophic level for the fish species of interest;

$TL_{prey\text{-}n}$ = trophic level for prey n; and

$\%V_{prey\text{-}n}$ = percent volume of prey n in the diet.

For aquatic species for which dietary studies from different locations were available, U.S. EPA developed an estimate of the range of trophic levels that might be exhibited by the species under different environmental conditions (U.S. EPA, 1995b).

To determine feeding habits of the piscivorous wildlife selected for evaluation, data in Volume II of U.S. EPA's (1993d) *Wildlife Exposure Factors Handbook* were reviewed. For the values in the *Handbook* cited from secondary sources, the original literature was retrieved and reviewed. To identify the size of prey captured by the wildlife species,

important for determining the trophic level of fish, all wildlife dietary studies that had been used to develop the *Handbook* were reviewed for information on prey size.

To estimate an average trophic level for a wildlife species in a given location, a similar approach was used to that described above for aquatic prey, except that the terrestrial and wetland components of the wildlife species' diet were separated from the aquatic components. Dietary studies from the Great Lakes region were preferred, but data from other locations were used to help define the wildlife species' range in trophic level of prey nationwide and the potential for variation within the Great Lakes region. The analyses are presented in detail in U.S. EPA's (1995a,b) *Trophic Level and Exposure Analyses for Selected Piscivorous Species*, Volumes I and III, Draft (Volume II looks at wildlife from a nationwide, instead of a Great Lakes, perspective).

As indicated above, the results of the analysis were numerical average trophic levels for each wildlife species. As indicated in Section VII.A., BAFs were developed for discrete integer trophic levels, i.e., for trophic levels 3 and 4, not for trophic level 3.2 for example. The non-integer average trophic level estimates were used to determine what proportion, on average, of trophic level 3 and trophic level 4 would be consumed by the wildlife species. For example, an average trophic level of 3.2 of the prey of an otter implies that the otter consumes on average 80 percent trophic level 3 and 20 percent trophic level 4 prey.

VII.B.1. Mink (*Mustela vison*)

The diet of mink consists primarily of prey linked to aquatic ecosystems, including fish, crayfish, frogs, muskrat, and waterfowl. Mink also consume terrestrial prey that are not associated with aquatic food chains, for example, shrews, mice, and voles (U.S. EPA 1993d). Most of the prey that mink consume from aquatic and wetland ecosystems nationwide represent trophic level 2 (muskrats, waterfowl in winter) to trophic level 3 (frogs, most fish in the size range captured, waterfowl in summer), with an average trophic level around 2.5 (including crayfish estimated to be trophic level 2.4), depending on prey availability. Alexander (1977) estimated dietary composition of mink in Michigan on the basis of percent wet weight of stomach contents from mink collected year round (U.S. EPA, 1995a). From this study, the average trophic level of the prey of mink was estimated to be 2.9 (for the size of fish captured by mink) and the proportion of the diet taken from strictly aquatic food webs ranged from 75 to 90 percent, depending on habitat. Sealander (1943) also studied mink in Michigan but during the winter only, and found a higher proportion of their diet to consist of prey from strictly terrestrial ecosystems (e.g., 20 to 30 percent) and a lower proportion to consist of strictly aquatic prey (about 10 percent), probably because of the winter ice cover. Other mink populations seem to obtain less of their food from aquatic and more from terrestrial sources (e.g., rabbits, voles, ground squirrels) year-round and during the summer than do mink in Michigan. However, most other studies have measured dietary composition by percent frequency of occurrence, which may not reflect percent biomass very well.

One difficulty for estimating aquatic trophic levels for mink is that the prey of most mink populations include a relatively high proportion of wetland animals, such as muskrat, waterfowl, and amphibians (U.S. EPA, 1995a, b). Ideally, one would estimate separate

bioaccumulation factors for contaminants in water relative to tissue concentrations in herbivorous muskrat, insectivorous waterfowl, and cold-blooded amphibians. In the absence of bioaccumulation data to support this level of detail, however, it is reasonable to assume that mink in the Great Lakes region are similar to mink in Michigan. It is assumed that mink obtain about 10 percent of their prey from terrestrial food webs and 90 percent from aquatic food webs on average for significant portions of the year. It is also reasonable to assume that their aquatic prey average trophic level 3 on the basis of Alexander's (1977) study.

VII.B.2 River Otter (*Lutra canadensis*)

In general, otters feed primarily on trophic level 3 fish, although they do catch some higher trophic level fish, including medium-sized piscivorous fish such as northern pike, walleye, and large trout. Knudsen and Hale (1968) estimated the dietary composition of otters in the Great Lakes region of Wisconsin, Michigan, and Minnesota on the basis of percent wet volume of the stomach contents of otters trapped by hunters. Based on this study, U.S. EPA estimated the average trophic level for the aquatic prey of these otters to be 2.9 and the average trophic level of the wetland prey (amphibians) to be trophic level 3.2 (U.S. EPA, 1995a). Lagler and Ostenson (1942) also estimated dietary composition of otters trapped in Michigan based on percent wet volume of stomach contents. Otters feeding in trout waters consumed on average trophic level 3.0 prey, while otters feeding in non-trout waters consumed on average trophic level 3.4 prey. Thus, it would be reasonable to assume an average trophic level of 3.2 for the aquatic prey of otter in the Great Lakes region. Depending on their availability, otters may consume waterbirds, but the amount of birds consumed is a minimal fraction of the diet and need not be quantified.

VII.B.3 Belted Kingfisher (*Ceryle alcyon*)

Belted kingfishers feed predominantly in shallow waters, capturing unusually large fish for their small body size. Although no feeding studies from the Great Lakes were identified, studies of belted kingfishers from almost any habitat feed exclusively on aquatic prey representing, on average, trophic level 2.7 to 3.0 (U.S. EPA, 1995a). Studies of belted kingfishers feeding on rivers and streams in Michigan indicate an average aquatic trophic level of 2.7 to 2.9 accounting for 70 to 99 percent of the diet, depending on the study. The remaining prey are primarily frogs. U.S. EPA (1995a,b) illustrated that many kingfisher populations are likely to feed on average at trophic level 3. Therefore, assuming 100 percent of the kingfisher's diet is trophic level 3 would be both reasonable and protective for populations in the Great Lakes basin.

As indicated earlier, the common tern is similar to the belted kingfisher in body weight. Courtney and Blokpoel (1980) found that common terns in one of two study locations in Lake Erie consumed almost 100 percent smelt and alewife in the spring. U.S. EPA (1995a) estimated that common terns feed on prey that average trophic level 2.9 in Lake Erie. Therefore, the trophic level assumed for the kingfisher is adequate to cover the feeding trophic level of the common tern as well.

VII.B.4 Herring Gull (*Larus argentatus*)

Fish usually considered trophic level 3 (e.g., small freshwater drum, alewife, smelt) comprise a large proportion of herring gull diets in the Great Lakes. Fox et al. (1990) and Ewins et al. (unpublished) found that during the breeding season, herring gulls feed primarily on alewife, and to a lesser extent on rainbow smelt. Based on these studies, U.S. EPA (1995a) estimated the average trophic level of the aquatic prey on which herring gulls feed in the Great Lakes to be 3.2.

Herring gull populations may feed on terrestrial organisms for some period of time. Ewins et al. (unpublished) found small mammals to account for between 1 and 98 percent of the diet for a given year, season, and site in the Great Lakes region. A diet consisting primarily of small herbivorous mammals (Trophic Level 2 in a terrestrial system) would effectively remove the gulls from exposure to aquatic food chains. On average, terrestrial sources may account for 10 percent of the herring gulls diet (U.S. EPA, 1995a,b).

VII.B.5. Bald Eagle (*Haliaeetus leucocephalus*)

Bald eagles prey on a variety of dead and living aquatic and terrestrial prey. They are opportunistic, and take whatever prey are readily available, although they appear to prefer live fish when possible. Thus, the trophic level at which bald eagles feed is likely to change with location, season, and year. One study of bald eagle dietary habits in the Great Lakes has been published (Kozie and Anderson, 1991; from Kozie, 1986) which assessed eagles nesting on islands and along the shore of Lake Superior in Wisconsin. This population fed both on fish from the Great Lakes and a wide diversity of birds. Data in U.S. EPA (1995a) indicate that the average trophic level of the fish component of the bald eagles' diet in Lake Superior is 3.2. Although bald eagles are large birds and can capture and carry large trophic level 4 fish, most of the time they capture the more abundant smaller fish (U.S. EPA, 1995a,b).

The herring gull comprises a portion of the bald eagles' diet on Lake Superior. In the Great Lakes, herring gulls feed primarily on alewife and smelt (see previous section). Most of the observations of the diet of bald eagles nesting on Lake Superior were from prey remains under the nest, which tend to overestimate the proportion of birds in the diet because the fish remains degrade faster than the bird remains. Data on both prey remains under the nest and observations of prey delivered to the nest on one island allow one to estimate the bias inherent in examining the prey remains only. As described in U.S. EPA (1995a), an adjustment factor was developed to estimate the relative number of birds and fish delivered to a nest on the basis of remains under the nest alone.

Using the adjustment factor described above, U.S. EPA estimated that on average, 92 percent of the diet of bald eagles in Lake Superior was comprised of fish, and 8 percent was comprised of birds or mammals. Of the birds/mammals component of the diet, herring gulls comprised 70 percent (on a wet-weight basis) of the biomass. Thus, herring gulls comprised 5.6 percent of the total bald eagle diet. For one bald eagle pair nesting near a gull colony, the proportion of herring gulls in the diet was higher, 12.5 percent of the total diet.

VII.B.6 Summary

Tables 6 and 7 provide a summary of the information on dietary composition and trophic level of the prey for the five representative wildlife species.

Table 6. Composition of Diet and Average Trophic Level of Aquatic Prey for Representative Wildlife Species.

Species	% Aquatic : % Terrestrial Prey	Average Trophic Level of Aquatic Prey	Of the Aquatic Prey, % TL3 : % TL4	Of the Terrestrial Prey, % Piscivorous Birds
mink	90 : 10	3.0	100%	0 %
river otter	100 : 0	3.2	80% : 20%	NA
belted kingfisher	100 : 0	3.0	100%	NA
herring gull	90 : 10	3.2	80% : 20%	0 %
bald eagle	92 : 8	3.2	80% : 20%	70 %

Table 7. Percent Composition of Diet by Wet Weight for the Representative Species.

Species	% TL 3 Fish	% TL 4 Fish	% Piscivorous Prey	% Non-piscivorous Prey	Total %
mink	90%	0%	0%	10%	100 %
river otter	80%	20%	0%	0%	100 %
belted kingfisher	100%	0%	0%	0%	100 %
herring gull	0.90 x 0.8 x 100 = 72%	0.90 x 0.2 x 100 = 18%	0%	10%	100 %
bald eagle	0.92 x 0.8 x 100 = 74%	0.92 x 0.2 x 100 = 18%	0.08 x 0.7 X 100 = 5.6%	0.08 x 0.3 x 100 = 2.4%	100 %

VII.C Species-specific Values for Body Weights and Water Ingestion Rates

As indicated in Section III, a body weight and food and water ingestion rates are required to estimate a wildlife value for each representative species in order to derive the wildlife criterion. This section explains how values can be identified for these exposure

parameters for wildlife in general. Section VII.D describes the selection of specific exposure parameter values for the five representative species for the Great Lakes basin.

VII.C.1 Body Weights

Adult body weights can vary by location, and often between sexes. In developing wildlife criteria, the body weights appropriate for the area or region in question should be used.

There are several sources that list body weight information for birds and mammals that might be of interest. For several species of both birds and mammals, a review of body weight data can be found in U.S. EPA's (1993c,d) *Wildlife Exposure Factors Handbook*. Dunning (1984) has compiled the body weights of 686 bird species from various sources. Body weights for mammalian species are included in Chapman and Feldhamer's (1982) *Wild Mammals of North America*, and in the *Special Publication* series of the American Society of Mammalogy. For species for which body weights cannot be found in these sources, a search through the open literature would be necessary.

VII.C.2 Drinking Water

Drinking water rates are quite variable among species and even within a species. The degree to which an animal must drink free water (e.g., from a pool) depends on the moisture content of its food as well as the animal's physiology (e.g., water economy). Many species in mesic habitats satisfy the bulk of their water requirements with the moisture in their food. In these habitats, drinking free water becomes more important during the higher temperatures of summer than at other times of the year.

There are few measures of the rate at which wildlife species drink free water. More information is available on the total water flux of wildlife species (e.g., Bartholomew and Cade, 1963; Nagy and Peterson, 1988), of which drinking free water is only one component. Calder and Braun (1983) reviewed available data on the drinking rates of wildlife species and developed allometric equations that can be used to estimate drinking water rates from body weights for birds and for mammals:

Birds: $W\ (L/day) = 0.059 Wt^{0.67}\ (kg)$, and

Mammals: $W\ (L/day) = 0.099\ Wt^{0.90}\ (kg)$.

where

W = rate of ingesting drinking water expressed as liters per day, and
Wt = the average weight in kilograms of the population or species.

VII.C.3 Food Ingestion Rates

Food ingestion rates are less variable within animals of the same age and species than are water ingestion rates. Food ingestion rates have been estimated for a few wildlife

species in the field (see U.S. EPA, 1993c). For the many wildlife species for which free-living food ingestion rates have not been measured or estimated, Chapter 4 of U.S. EPA's (1993c,d) *Wildlife Exposure Factors Handbook* explains how to estimate the rates.

Food ingestion rates can be calculated from the energetic needs of an animal, the composition and caloric content of its diet, and the efficiency with which the energy in the diet is assimilated. Basically, the first step is to identify or to estimate the average free-living metabolic rate (FMR) for the species (see Section VII.C.4). The FMR establishes the metabolizable energy (ME) needs of the species in the wild. ME is equal to the gross energy (GE) in the diet multiplied by the assimilation efficiency (AE) for the species of that diet:

ME (kcal/g diet) = GE (kcal/g diet) x AE.

The *Wildlife Exposure Factors Handbook* provides tables from which the GE of various prey can be estimated and from which the AE of different groups of animals consuming different types of prey can be estimated. Also note the water content of diet. Seeds tend to contain little water (9-10% water) whereas the tissues of fish, birds, and mammals tend consist of 75 percent water (U.S. EPA 1993c).

Examples of how food ingestion rates are estimated from body weight and dietary composition can be reviewed in Section VII.D, where the derivations of food ingestion rates for the five representative species selected for the GLWQI are described.

VII.C.4 Free-living Metabolic Rates

The free-living metabolic rates of several wildlife species have been estimated using doubly-labeled water measurements of CO_2 production. Using the data developed by the few laboratories that use the technique well, Nagy (1987) developed allometric equations relating free-living metabolic rate (FMR) to body weight for birds and mammals. Nagy (1987) includes four allometric equations for birds that include a reasonable number and taxonomic breadth of avian species:

All birds: $FMR\ (kcal/day) = 2.601\ Wt^{0.640}\ (g)$,

Passerine birds: $FMR\ (kcal/day) = 2.123\ Wt^{0.749}\ (g)$,

Seabirds: $FMR\ (kcal/day) = 1.916\ Wt^{0.704}\ (g)$, and

Non-passerines: $FMR\ (kcal/day) = 1.146\ Wt^{0.749}\ (g)$.

Passerines have higher metabolic rates for their body size than do other birds. For small birds, the difference between the passerine and non-passerine and between the passerine and all-birds allometric equations are relatively large. For large birds, all of these equations converge and provide similar estimates.

Nagy (1987) similarly provides several allometric equations for mammals. There are four equations that are likely to be useful for North American mammals:

Placental mammals: FMR (kcal/day) = $0.800\ Wt^{0.813}$ (g),

Herbivores: FMR (kcal/day) = $1.419\ Wt^{0.727}$ (g),

Non-herbivores: FMR (kcal/day) = $0.6167\ Wt^{0.862}$ (g), and

Rodents: FMR (kcal/day) = $2.514\ Wt^{0.507}$ (g).

Nagy (1987) and U.S. EPA (1993c) can be consulted to calculate the 95 percent confidence limits on an estimate of FMR for any single body weight estimate.

VII.D. Exposure Parameter Values for the Representative Wildlife Species

This section describes the sources of information for and derivation of exposure parameter values for the five representative wildlife species.

VII.D.1 Mink (*Mustela vison*)

Body weight. Mink body weights vary greatly throughout the species' range (adult males reaching 1.4 kg in the east and 2.3 kg in the west, according to Harding, 1934, cited in Linscombe et al., 1982). Males weigh markedly more than females (in some populations, almost twice as much). U.S. EPA's (1993c) *Wildlife Exposure Factors Handbook* had limited data on body weights of wild mink. From those data, body weights for mink from Montana during the summer are listed in Table 8. Thus, an average body weight for both sexes of 0.8 kg would be a reasonable value to assume for wild mink.

Table 8. Body Weights of Mink Populations.

Age/Sex/Season	Body Weight (kg)	Reference/Location
adult/male/summer adult/female/summer average	1.04 (N = 5) 0.550 (N = 25) 0.78	Mitchell 1961/ Montana

Water ingestion rate. No measured values for mink drinking water ingestion rates were identified. The moisture content of their diet of fish may satisfy much of their daily water requirements. In the absence of specific information, however, Calder and Braun's (1983) allometric equation for estimating drinking water ingestion rates predicts that mink averaging 0.8 kg drink 0.081 L/day:

Water Ingestion (L/day) = $0.099\ Wt^{0.90}$ (kg) = $0.099\ (0.8)^{0.90}$ = 0.081 L/day.

Food ingestion rate. No measured values for free-living mink food ingestion rates were found. Nagy's (1987) allometric equation for estimating free-living metabolic rate for non-herbivorous mammals predicts that mink weighing 0.80 kg require 200 kcal per day:

$$FMR\ (kcal/day) = 0.6167\ Wt^{0.862}\ (g) = 0.6167\ (800)^{0.862} = 196\ kcal/day.$$

Assuming that the fish consumed by mink provide 1.2 kcal gross energy (GE)/g wet weight and that the assimilation efficiency (AE) of mink consuming fish is 91 percent (U.S. EPA, 1993c, Tables 4-1 and 4-3), the metabolizable energy (ME) in fish for mink would be 1.1 kcal/g wet weight:

$$ME\ (kcal/g\ fish) = GE \times AE = 1.2\ (kcal/g) \times 0.91 = 1.09\ (kcal/g\ fish)$$

Assuming that the GE of birds and mammals consumed by mink is 1.8 kcal/g wet weight and that the AE of mink consuming birds and mammals is 84 percent (U.S. EPA, 1993c), the ME in birds and mammals for mink would be 1.5 kcal/g wet weight:

$$ME\ (kcal/g\ birds\text{-}mammals) = GE \times AE = 1.8\ (kcal/g) \times 0.84 = 1.51\ (kcal/g\ birds\text{-}mammals)$$

Thus, if mink consumed only fish, they would require 180 grams of fish daily (196 kcal/day divided by 1.09 kcal/g fish). If mink consumed only birds and mammals, they would require 130 grams of birds or mammals daily (196 kcal/day divided by 1.51 kcal/g birds or mammals). Given the assumption that mink consume 90 percent fish and 10 percent birds and mammals on a wet-weight basis, however, the total grams of each is estimated as follows:

If

Y = grams of birds or mammals consumed,
$9Y$ = grams of fish consumed,

then

$$Y\ (g) \times 1.51\ (kcal/g\ birds\text{-}mammals) + 9Y\ (g) \times 1.09\ (kcal/g\ fish) = 200\ kcal,$$

$$1.51\ Y\ (kcal) + 9.81\ Y\ (kcal) = 200\ kcal,$$
$$11.3\ Y = 200,$$

and

Y = 17.7 grams of birds-mammals consumed, and
$9Y$ = 159 grams of fish consumed.

As indicated in Table 5, all of the fish consumed are assumed to represent trophic level 3.

VII.D.2 River Otter (*Lutra canadensis*)

Body weight. Sexual dimorphism in size is seen among all subspecies of river otters (Toweill and Tabor, 1982), and adult males (5 to 10 kg) outweigh females (4 to 7 kg) by approximately 17 percent (Melquist and Hornocker, 1983; see Table 9). No body weights are available for otters in the vicinity of the Great Lakes. Given the large sample size and distributional data provided for the Alabama otter studied by Lauhachinda (1978), these data were used to estimate an average river otter body weight of 7.4 kg. If river otters tend to be larger at more northerly latitudes (Bergman's rule), this value may underestimate otter body weights in the Great Lakes region.

Table 9. Body Weights of River Otter Populations.

Age/Sex	Body Weight (kg)	Reference/ Location
adult female adult male average	6.7 ± 1.0 SD (N=71) (range 4.74 - 8.72) 8.1 ± 1.2 SD (N=153) (range 5.84 - 10.4) 7.40	Lauhachinda 1978/Alabama, Georgia
adult female adult male average	7.9 ± 0.2 SE (N=6) 9.2 ± 0.6 SE (N=4) 8.55	Melquist & Hornocker 1983/Idaho
adult female adult male average	7.0 (N=100) 8.3 (N=138) 7.65	Wilson 1959/ North Carolina

Water ingestion rate. No measured values for river otter drinking water ingestion rates were identified. The moisture content of their diet of fish may satisfy much of their daily water requirements. In the absence of specific information, however, Calder and Braun's (1983) allometric equation for estimating drinking water ingestion rates predicts that otter averaging 7.4 kg drink 0.60 L/day:

$$\text{Water Ingestion (L/day)} = 0.099\ Wt^{0.90}\ \text{(kg)} = 0.099\ (7.4)^{0.90} = 0.60\ \text{L/day}.$$

Food ingestion rate. No measured values for free-living river otter food ingestion rates were found. Nagy's (1987) allometric equation for estimating free-living metabolic rate for non-herbivorous mammals predicts that otters weighing 7.4 kg require 1,335 kcal per day:

$$\text{FMR (kcal/day)} = 0.6167\ Wt^{0.862}\ \text{(g)} = 0.6167\ (7400)^{0.862} = 1{,}335\ \text{kcal/day}.$$

Assuming that the GE of the prey (i.e., fish) of the otter be 1.2 kcal/g wet weight and the AE of otter consuming fish is 91 percent (U.S. EPA, 1993c), the ME in fish for otters would be 1.1 kcal/g wet weight:

$$\text{ME (kcal/g fish)} = \text{GE} \times \text{AE} = 1.2\ \text{kcal/g fish} \times 0.91 = 1.09\ \text{kcal/g fish}.$$

Thus, to satisfy the daily requirement for 1,335 kcal ME, the food ingestion rate (F) of an otter would need to be 1,220 g of fish per day:

$$\text{F (kg/day)} = (1{,}335\ \text{kcal/day})/(1.09\ \text{kcal/g wet weight fish}) = 1{,}220\ \text{g/day}.$$

As indicated in Table 6, the fish consumed by otter are assumed to consist of 80 percent trophic level 3 fish and 20 percent trophic level 4 fish. Thus, the trophic-level-specific fish ingestion rates for river otters can be estimated as:

F (kg/day) trophic level 3 fish = 1.22 x 0.80 = 0.976 kg TL3 fish/day, and
F (kg/day) trophic level 4 fish = 1.22 x 0.20 = 0.244 kg TL4 fish/day.

VII.D.3 Belted Kingfisher (*Ceryle alcyon*)

Body weight. The sexes are similar in size, and a value of 0.148, rounded to 0.15 kg would be a representative weight for this species (see Table 10).

Table 10. Body Weights of Belted Kingfisher Populations.

Age/Sex	Body Weight (kg)	Reference/Location
adults/both sexes	0.148 ± 0.0208 SD (N=29)	Powdermill Nature Center (unpub.) cited in Dunning, 1984/Pennsylvania
adults/both sexes	0.136 ± 0.156 SE (N=5)	Brooks & Davis, 1987/Pennsylvania
adults/both sexes	0.150 (N=98)	Alexander, 1977/Michigan
adults/both sexes	0.158 ± 0.115 SE (N=11)	Brooks & Davis, 1987/ Ohio

Water ingestion rate. No measured values for kingfisher drinking water ingestion rates were identified. The moisture content of their diet of fish may satisfy much of their daily water requirements. In the absence of specific information, however, Calder and Braun's (1983) allometric equation for estimating drinking water ingestion rates predicts that kingfishers averaging 0.15 kg drink 0.017 L/day:

$$\text{Water Ingestion (L/day)} = 0.059\ Wt^{0.67}\ \text{(kg)} = 0.059\ (0.148)^{0.67} = 0.017\ \text{L/day.}$$

Food ingestion rate. No measured values for free-living kingfisher food ingestion rates were found. Nagy's (1987) allometric equation for estimating free-living metabolic rate for all birds predicts that kingfishers weighing 0.15 kg require 64 kcal per day, or 430 kcal/kg kingfisher-day:

$$\text{FMR (kcal/day)} = 2.601\ Wt^{0.640}\ \text{(g)} = 2.601\ (148)^{0.640} = 63.7\ \text{kcal/day, or}$$
$$\text{FMR (kcal/kg-day)} = (63.7\ \text{kcal/day})/(0.148\ \text{kg}) = 430\ \text{kcal/kg-day.}$$

This might overestimate the food ingestion rate for kingfishers because they are not in the order Passeriformes, and therefore the non-passerine allometric equation would be more appropriate:

$$\text{FMR (kcal/day)} = 1.146\ Wt^{0.749}\ \text{(g)} = 1.146\ (148)^{0.749} = 48.4\ \text{kcal/day, or}$$
$$\text{FMR (kcal/kg-day)} = (49.4\ \text{kcal/day})/(0.148\ \text{kg}) = 334\ \text{kcal/kg-day.}$$

However, the exposure parameter values for the belted kingfisher are assumed to be representative of the likely exposures of common terns as well as belted kingfishers.

Nagy's (1987) allometric equation for estimating the free-living metabolic rate for seabirds predicts that common terns weighing 0.12 kg (LeCroy and LeCroy, 1974; Dunning, 1984) require 56 kcal/day, or 467 kcal/kg-day:

$$\text{FMR (kcal/day)} = 1.916\,Wt^{0.704}\text{ (g)} = 1.916\,(120)^{0.704} = 55.7\text{ kcal/day, or}$$
$$\text{FMR (kcal/kg-day)} = (55.7\text{ kcal/day})/(0.120\text{ kg}) = 467\text{ kcal/kg-day.}$$

Thus, the daily energy requirement of 63.7 kcal/day predicted from Nagy's (1987) allometric equation for all birds is used for the belted kingfisher and is intended to represent the caloric needs of the common tern relative to its body weight.

Assuming that the fish consumed by the kingfisher (or common tern) provide 1.2 kcal GE/g wet weight and that the AE of kingfishers consuming fish is 79 percent (U.S. EPA, 1993c), the ME in fish for belted kingfishers would be 0.948 kcal/g wet weight:

$$\text{ME (kcal/g fish)} = \text{GE} \times \text{AE} = 1.2\text{ kcal/g fish} \times 0.79 = 0.948\text{ kcal/g fish.}$$

Thus, to satisfy the daily requirement for 63.7 kcal ME, the fish ingestion rate (F) of a belted kingfisher would need to be 67.2 g of fish per day:

$$F\text{ (kg/day)} = (63.7\text{ kcal/day})/(0.948\text{ kcal/g wet weight fish}) = 67.2\text{ g/day.}$$

As indicated in Table 6, it is assumed that all of the 67.2 g of fish consumed by the kingfisher are trophic level 3 fish.

VII.D.4. Herring Gull (*Larus argentatus*)

Body weight. Adult herring gulls generally weigh between 1.0 and 1.2 kg, with the males being slightly heavier than the females (Table 11). An average weight for both sexes would be 1.1 kg. The data indicating slightly lower body weights from Lake Huron in Table 11 is not used for the GLWQI because of the small number of individuals of each sex in that sample.

Water ingestion rate. No measured values were found for the water ingestion rates of herring gulls. The moisture content of their diet may satisfy much of their daily water requirements. In the absence of specific information, however, Calder and Braun's (1983) allometric equation for estimating drinking water ingestion rates predicts that herring gulls averaging 1.1 kg in weight drink 0.063 L/day:

$$\text{Water Ingestion (L/day)} = 0.059\,Wt^{0.67}\text{ (kg)} = 0.059\,(1.1)^{0.67} = 0.063\text{ L/day.}$$

Table 11. Body Weights of Herring Gull Populations.

Age/Sex	Body Weight (kg)	Reference/Location
adult/female adult/male average	1.00 ± 0.090 SD (N=78) (range = 0.83 - 1.27) 1.23 ± 0.107 SD (N=180) (range = 1.01 - 1.62) 1.12	Threlfall and Jewer, 1978/ Newfoundland, Canada
adult/female adult/male average	0.920 ± 0.057 SD (N=10) 1.05 ± 0.058 SD (N=7) 0.99	Norstrom et al., 1986/ Lake Huron
adult/female adult/male average	1.04 (N=139; range = 0.717 - 1.39) 1.23 (N=220; range = 0.755 - 1.50) 1.14	Belopolskii, 1957 cited in Dunning, 1984/Barent Sea (arctic)

Food ingestion rate. No measured values for free-living herring gull food ingestion rates were found. Nagy's (1987) allometric equation for estimating free-living metabolic rate for seabirds predicts that herring gulls weighing 1.1 kg require 265 kcal per day:

$$FMR\ (kcal/day) = 1.916\ Wt^{0.704}\ (g) = 1.916\ (1100)^{0.704} = 265\ kcal/day.$$

Norstrom et al. (1986) have estimated an annual energy budget for free-living female herring gulls that breed in the Great Lakes. Between September and March, the non-breeding season, they estimate that adult females require 250 to 260 kcal/day. Following a dip in energy requirements when the male feeds the female during courtship, Norstrom et al. (1986) estimated that the female's needs increase to peak at 280 kcal/day for egg production, then fall to approximately 210 kcal/day during incubation. The average over this period is likely to be similar to the estimate for the non-breeding season. Thus, the estimate of 265 kcal/day from Nagy's (1987) allometric equation for seabirds is considered a reasonable estimate of the ME requirement of free-living herring gulls

Assuming that the GE of fish consumed by herring gulls is 1.2 kcal/g wet weight and that the AE of gulls consuming fish is 79 percent (U.S. EPA, 1993c), the ME in fish for gulls can be estimated:

$$ME\ (kcal/g\ fish) = GE \times AE = 1.2\ (kcal/g) \times 0.79 = 0.948\ (kcal/g\ fish).$$

Assuming that the birds and mammals consumed by herring gulls provide 1.8 kcal GE/g wet weight and that the AE of seabirds is the same as the AE of birds of prey consuming birds and mammals (i.e., 78 percent; U.S. EPA, 1993c), the ME in birds and mammals for the herring gull would be 1.40 kcal/g wet weight:

$$ME\ (kcal/g\ birds) = GE \times AE = 1.8\ (kcal/g) \times 0.78 = 1.404\ (kcal/g\ birds).$$

Thus, if herring gulls consumed only fish, they would require 280 grams of fish daily (265 kcal/day divided by 0.948 kcal/g fish). If herring gulls consumed only birds and small

mammals, they would require 189 grams of birds/mammals daily (265 kcal/day divided by 1.404 kcal/g birds). Given the assumption that herring gulls consume 90 percent fish and 10 percent birds and mammals on a wet-weight basis (Table 6), the total grams of each is estimated as follows:

If

Y = grams of birds-mammals consumed, and
9 Y = grams of fish consumed

then

Y (g) x 1.404 (kcal/g birds-mammals) + 9 Y (g) x 0.948 (kcal/g fish) = 265 kcal,

1.404 Y (kcal) + 8.532 Y (kcal) = 265 kcal,
9.936 Y = 265,

and

Y = 26.7 grams of terrestrial birds-mammals consumed, and
9 Y = 240 grams of fish consumed.

As indicated in Table 6, the fish consumed by herring gulls are assumed to consist of 80 percent trophic level 3 fish and 20 percent trophic level 4 fish. Thus, of the 240 grams of fish consumed daily, 80 percent, or 192 grams, are of trophic level 3 fish and 20 percent, or 48 grams, are of trophic level 4 fish.

VII.D.5 Bald Eagle (*Haliaeetus leucocephalus*)

Body weight. As for other birds of prey, female bald eagles are about 20 percent heavier than the males. Given the data provided in Table 12, 4.6 kg would be a reasonable value to use for the average weight of bald eagles.

Table 12. Body Weights of Bald Eagle Populations.

Age/Sex	Body Weight (kg)	Reference/Location
adult/female adult/male average	5.2 (N=37) 4.1 (N=35) 4.65	Synder & Wiley, 1976/ North America
juvenile/female (to 3 yrs old) juvenile/male (to 3 yrs old) average	5.1 (range = 4.3 - 5.8)[a] 4.0 (range = 3.5 - 4.6)[a] 4.55	Imler & Kalmbach, 1955/ Alaska

[a] N = 18 for both sexes combined.

Water ingestion rate. No measured values for bald eagle drinking water ingestion rates were identified. The moisture content of their diet may satisfy much of their daily water requirements. In the absence of specific information, however, Calder and Braun's (1983) allometric equation for estimating drinking water ingestion rates predicts that bald eagles averaging 4.6 kg in weight drink 0.16 L/day:

$$\text{Water Ingestion (L/day)} = 0.059\ Wt^{0.67}\ \text{(kg)} = 0.059\ (4.6)^{0.67} = 0.16\ \text{L/day}.$$

Food ingestion rate. Several investigators have estimated the rate of food consumption by bald eagles, but only some normalized their estimates to the body weight of the birds or reported the body weights so that others could normalize the ingestion rates. Two separate research teams provided similar estimates of food ingestion rates for free-flying bald eagles.

Stalmaster and Gessaman (1984) observed bald eagles taking pre-weighed salmon provided at artificial feeding stations in Washington State. Although the eagles may have fed elsewhere on occasion, Stalmaster and Gessaman (1984) felt that the feeding stations provided most of the birds' intake. Adults were observed to take 552 grams of fish per bird per day, while juveniles and subadults (up to three years old) were observed to take 410 and 459 grams of fish, respectively, per bird per day. Assuming that adult eagles weigh 4.5 kg on average, Stalmaster and Gessaman's (1984) observations indicate that the adult bald eagles ingest 12.3 percent of their body weight in fish daily. Assuming that the GE in fish consumed by bald eagles is 1.2 kcal/g wet weight and that the AE of birds of prey consuming fish is 79 percent (U.S. EPA, 1993c, Tables 4-1 and 4-3), the ME of the fish is estimated to be 0.948 kcal/g fish:

$$\text{ME (kcal/g fish)} = \text{GE} \times \text{AE} = 1.2 \text{ (kcal/g)} \times 0.79 = 0.948 \text{ (kcal/g fish)}.$$

Using an ME of 0.948 kcal/g fish, the total ME requirement for an adult bald eagle on a daily basis would be 523 kcal/bird per day (i.e., 552 g fish/bird-day x 0.948 kcal/g fish). This energy "requirement" is based on the amount of fish the bald eagles were observed to eat, which may have been more than they "needed" over the short term.

Craig et al. (1988) estimated gross energy consumption rates of bald eagles in Connecticut based the observed time spent feeding and the model of Stalmaster and Gessaman (1984). Assuming a body weight of 4.5 kg, they concluded that adult bald eagles require 448 kcal ME per day and that for adults and that subadults require 499 kcal ME per day.

Given the data presented above, it would be reasonable to assume that bald eagles require 500 kcal ME per bird per day.

As indicated in Table 6, bald eagles also include birds and mammals in their diet. Assuming that the birds consumed by bald eagles in the Lake Superior study provide 1.9 kcal GE/g wet weight (value for gulls and terns in Table 4-1 in U.S. EPA, 1993c) and that the AE of birds of prey consuming birds is 78 percent (Table 4-3 in U.S. EPA, 1993c), the ME in birds for the bald eagle would be 1.48 kcal/g birds:

$$\text{ME (kcal/g birds)} = \text{GE} \times \text{AE} = 1.9 \text{ (kcal/g)} \times 0.78 = 1.482 \text{ (kcal/g birds)}.$$

If bald eagles consumed only fish, to obtain 500 kcal of ME daily, they would need to ingest 527 grams of fish daily (i.e., 500 kcal/bird-day divided by 0.948 kcal/g fish). If bald eagles consumed only birds, to obtain 500 kcal of ME daily, they would need to ingest 338 grams of birds daily (500 kcal/day divided by 1.48 kcal/g birds). Given the assumption that bald eagles consume 92 percent fish and 8 percent birds on a wet-weight basis (see Section VII.B), the total grams of each type of prey is estimated as follows:

If

Y = grams of birds consumed, and
11.5 Y = grams of fish consumed (i.e., 92/8 = 11.5),

then

Y (g) x 1.482 (kcal/g birds) + 11.5 Y (g) x 0.948 (kcal/g fish) = 500 kcal,
1.482 Y (kcal) + 10.902 Y (kcal) = 500 kcal,
12.38 Y = 500,

and

Y = 40.4 grams of birds consumed, and
11.5 Y = 464 grams of fish consumed.

Of the 40.4 grams of birds consumed, 70 percent, or 28.3 grams are comprised of herring gulls, and the remaining 12.1 grams are comprised of other non-piscivorous birds. Of the 464 grams of fish consumed, 80 percent, or 371 grams, are of trophic level 3 fish, and 20 percent, or 92.8 grams, are of trophic level 4 fish.

For the sensitivity analyses carried out in the *Great Lakes Water Quality Initiative Criteria Documents for the Protection of Wildlife: DDT; Mercury; 2,3,7,8-TCDD; and PCBs* (U.S. EPA, 1995d), different values for the amount of gulls and fish in the bald eagles diet were considered. Kozie (1986) provided data for one pair of bald eagles nesting near a herring gull colony. For this pair, 13.75 percent of the diet consisted of birds and 91 percent of the wet mass of birds captured consisted of herring gulls (Exhibit 6-8 in U.S. EPA 1995a). Thus, 12.5 percent of the diet of this pair consisted of herring gulls on a wet weight basis. In contrast, considering all eight pairs studied on lake Superior, only 5.8 percent of the diet consisted of herring gulls on average (i.e., 8 percent of diet consists of birds x 70 percent of birds are comprised of herring gulls). To conduct the sensitivity analysis for the bald eagle related to its dietary composition, it was assumed that at one extreme the composition of the bald eagle's diet was the same as for the pair near the gull colony. The quantities of fish, gulls, and non-aquatic birds that would be needed to provide 500 kcal of ME are calculated as follows.

Still assuming that the ME in birds for the bald eagle is 1.48 kcal/g bird, and that the ME in fish for the bald eagles is 0.948 kcal.g fish, then assuming that bald eagles consume 86.3 percent fish and 13.8 percent birds on a wet-weight basis (see above), the total grams of each type of prey is estimated as follows:

If

Y = grams of birds consumed, and
6.27 = grams of fish consumed (i.e., 86.3/13.8 = 6.27)

then

Y (g) x 1.482 (kcal.g bird) + 6.27 Y (g) x 0.948 (kcal.g fish)
= 500 kcal.
1.482 Y (kcal) + 5.944 (kcal) = 500 kcal,
7.425 Y = 500 kcal,

and

Y = 67.33 grams of birds consumed, and
6.27 Y = 422.2 grams of fish consumed.

Of the 67.3 grams of birds consumed, 91 percent, or 61.3 grams are comprised of herring gulls, and the remaining 6.0 grams are comprised of other non-piscivorous birds. Of the 422 grams of fish consumed, 80 percent, or 338 grams, are of trophic level 3 fish, and 20 percent, or 84.5 grams, are of trophic level 4 fish.

Alternatively, for the sensitivity analyses, calculations assuming a 100 percent fish diet (with 80% trophic level 3 and 20% trophic level 4) were also performed. In that both of these assumptions (i.e., gull ingestion and no gull ingestion) assume no terrestrial component to the eagle diet, they are likely to provide slight overestimates of contaminant exposures (see Table 13 below).

VII.D.6 Summary of Exposure Parameter Values for the Representative Wildlife Species

Table 13 summarizes the exposure parameter values for the five representative wildlife species. This table also is presented as Table D-2 in Appendix D to Part 132.

Table 13. Exposure Parameter Values for the Five Representative Wildlife Species.

Species	Adult Body Weight (kg)	Water Ingestion Rate (L/day)	Food Ingestion Rate of Prey in Each Trophic Level (kg/day)	Trophic Level of Prey as Percent of Diet
Mink	0.80	0.081	TL3: 0.159 Other: 0.0177	TL3: 90% Other: 10%
Otter	7.4	0.60	TL3: 0.976 TL4: 0.244	TL3: 80% TL4: 20%
Kingfisher	0.15	0.017	TL3: 0.0672	TL3: 100%
Herring gull	1.1	0.063	TL3: 0.192 TL4: 0.0480 Other: 0.0267	Fish: 90% TL3: 80% TL4: 20% Other: 10%
Bald Eagle	4.6	0.16	TL3: 0.371 TL4: 0.0928 PB: 0.0283 Other: 0.0121	Fish: 92% TL3: 80% TL4: 20% Birds: 8% PB: 70% non-aquatic: 30%

Note: TL3 = trophic level three fish; TL4 = trophic level four fish; PB = piscivorous birds; Other = non-aquatic-based birds and mammals

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