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PHASE 2 REPORT- REVIEW COPY
FURTHER SITE CHARACTERIZATION AND ANALYSIS

VOLUME 2E-A BASELINE ECOLOGICAL RISK ASSESSMENT FOR FUTURE RISKS IN THE LOWER HUDSON RIVER

HUDSON RIVER PCBs REASSESSMENT RI/FS

DECEMBER 1999

UNITED STATES ENVIRONMENTAL PROTECTION AGENCY

For

U.S. Environmental Protection Agency
Region II
and
U.S. Army Corps of Engineers
Kansas City District

Book 1 of 1

TAMS Consultants, Inc.
Menzie-Cura & Associates, Inc.

UNITED STATES ENVIRONMENTAL PROTECTION AGENCY
REGION 2
290 BROADWAY
NEW YORK, NY 10007-1866

December 29, 1999

To All Interested Parties:

The U.S. Environmental Protection Agency (USEPA) is pleased to release the baseline Ecological Risk Assessment - Future Risks in the Lower Hudson River, which evaluates the future ecological risks in the Lower Hudson River (Federal Dam to the Battery in New York City) posed by PCBs in sediments at the Hudson River PCBs Superfund site, in the absence of remediation. This report, called the Ecological Risk Assessment (ERA) Addendum, is a companion volume to USEPA's August 1999 baseline Ecological Risk Assessment (ERA), which evaluated the current and future ecological risks in the Upper Hudson River and the current ecological risks in the Lower Hudson River. The ERA Addendum is posted on USEPA's website for the Hudson River PCBs Reassessment Remedial Investigation/Feasibility Study (Reassessment RI/FS) at www.epa.gov/hudson.

The ERA Addendum is part of Phase 2 of the Reassessment RI/FS for the Hudson River PCBs Superfund site. The ERA Addendum, together with the August 1999 ERA, will help establish acceptable exposure levels for use in developing remedial alternatives in the Feasibility Study, which is Phase 3 of the Reassessment RI/FS.

USEPA will accept comments on the ERA Addendum until January 28, 2000. Comments should be marked with the name of the report and should include the report section and page number for each comment. Comments should be sent to:

Alison A. Hess, C.P.G.
USEPA Region 2
290 Broadway - 19th Floor
New York, NY 10007-1866
Attn: Hudson River ERA Addendum Comments

USEPA will hold a Joint Liaison Group meeting to discuss the findings of the ERA Addendum on January 11, 2000, at 7:30 p.m. at the Sheraton Hotel, 40 Civic Center Plaza, Poughkeepsie, New York. The meeting is open to the general public. Notification of the meeting was sent to Liaison Group members, interested parties, and the press several weeks prior to the meeting.

During the public comment period, USEPA will hold an availability session to answer questions from the public regarding the ERA Addendum. The availability session will be held from 6:30 to 8:30 p.m. on January 18, 2000 at Sheraton Hotel, 40 Civic Center , Poughkeepsie, New York.

If you need additional information regarding the ERA Addendum or the Reassessment RI/FS in general, please contact Ann Rychlenski, the Community Relations Coordinator for this site, at (212) 637-3672.

Sincerely yours,

William McCabe

for Richard L. Caspe, Director
Emergency and Remedial Response Division

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ACRONYMS

ATSDR	Agency for Toxic Substances and Desease Registry
CDI	Chronic Daily Intake
CERCLA	Comprehensive Environmental Response, Compensation, and Liability Act
CSF	Carcinogenic Slope Factor
EPC	Exposure Point Concentration
GE	General Electric
HI	Hazard Index
HHRA	Human Health Risk Assessment
HHRASOW	Human Helath Risk Assessment Scope of Work
HQ	Hazard Quotient
NCP	National Oil and Hazardous Substances Pollution Contingency Plan
NPL	National Priorities List
NYSDEC	New York State Department of Environmental Conservation
NYSDOH	New York State Department of Health
PCB	Polychlorinated Biphenyl
RfD	References Dose
RI	Remedial Investigation
RI/FS	Remedial Investigation/Feasibility Study
ROD	Record of Decision
RM	River Mile
RI/FS	Remedial Investigation/Feasibility Study
SARA	Superfund Amendments and Reauthorization Act of 1986
TCDD	2,3,7,8-Tetrachlorodibenzo-p-dioxin
TEF	Toxicity Equivalency Factor
TSCA	Toxic Substances Control Act
UCL	Upper Confidence Limit
USEPA	United States Environmental Protection Agency

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Ecological Risk Assessment Addendum: Future Risks in the Lower Hudson River Executive Summary December 1999

This document presents the baseline Ecological Risk Assessment for Future Risks in the Lower Hudson River (ERA Addendum), which is a companion volume to the baseline Ecological Risk Assessment (ERA) that was released by the U.S. Environmental Protection Agency (USEPA) in August 1999. Together, the two risk assessments comprise the ecological risk assessment for Phase 2 of the Reassessment Remedial Investigation/Feasibility Study (Reassessment RI/FS) for the Hudson River PCBs site in New York.

The ERA Addendum quantitatively evaluates the future risks to the environment in the Lower Hudson River (Federal Dam at Troy, New York to the Battery in New York City) posed by polychlorinated biphenyls (PCBs) from the Upper Hudson River (Hudson Falls, New York to the Federal Dam at Troy, New York), in the absence of remediation. This report uses current USEPA policy and guidance as well as additional site data and analyses to update USEPA's 1991 risk assessment.

USEPA uses ecological risk assessments to evaluate the likelihood that adverse ecological effects are occurring or may occur as a result of exposure to one or more chemical or physical stressors. The Superfund ecological risk assessment process includes the following: 1) identification of contaminants of concern; 2) development of a conceptual model, which identifies complete exposure pathways for the ecosystem; 3) identification of assessment endpoints, which are ecological values to be protected; 4) development of measurement endpoints, which are the actual measurements used to assess risk to the assessment endpoints; 5) selection of receptors of concern; 6) the exposure assessment, which describes concentrations or dietary doses of contaminants of concern to which the selected receptors are or may be exposed; 7) the effects assessment, which describes toxicological effects due to chemical exposure and the methods used to characterize those effects to the receptors of concern; and 8) risk characterization, which compares the results of the exposure assessment with the effects assessment to evaluate the likelihood of adverse ecological effects associated with exposure to chemicals at a site.

The ERA Addendum indicates that, for some species, future concentrations of PCBs in the Lower Hudson River generally exceed levels that have been shown to cause adverse ecological effects through 2018 (the entire forecast period). The results of the ERA Addendum will help establish acceptable exposure levels for use in developing remedial alternatives for PCB-contaminated sediments in the Upper Hudson River, which is Phase 3 (Feasibility Study) of the Reassessment RI/FS.

Contaminants of Concern

The contaminants of concern identified for the site are PCBs. PCBs are a group of synthetic organic compounds consisting of 209 individual chlorinated biphenyls called congeners. Some PCB congeners are considered to be structurally similar to dioxin and are called dioxin-like PCBs. Toxic equivalency (TEQ) factors, based on the toxicity of dioxin, have been developed for the dioxin-like PCB congeners. PCBs have been shown to cause adverse reproductive and developmental effects in animals. Ecological exposure to PCBs is primarily an issue of bioaccumulation rather than direct toxicity. PCBs bioaccumulate in the environment by both bioconcentrating (being absorbed from water and accumulated in tissue to levels greater than those found in surrounding water) and biomagnifying (increasing in tissue concentrations as they go up the food chain through two or more trophic levels).

Site Conceptual Model

The Hudson River PCBs site is the 200 miles (322 km) of river from Hudson Falls, New York to the Battery in New York City. As defined in the ERA and ERA Addendum, the Lower Hudson River extends approximately 160 miles (258 km) from the Federal Dam at Troy (River Mile 153) to the Battery.

The Hudson River is home to a wide variety of ecosystems. The Lower Hudson River is tidal, does not have dams, and is freshwater in the vicinity of the Federal Dam, becoming brackish and increasingly more saline towards the Battery. Spring runoffs and major storms can push the salt front well below the Tappan Zee Bridge, and sometimes south to New York City. The Lower Hudson has deep water environments, shallow nearshore areas (shallows, mudflats, and shore communities), tidal marshes, and tidal swamps.

PCBs were released from two General Electric Company capacitor manufacturing facilities located in the Upper Hudson River at Hudson Falls and Fort Edward, New York. Many of these PCBs adhered to river sediments. As PCBs in the river sediments are released slowly into the river water, these contaminated sediments serve as a continuing source of PCBs. During high flow events, the sediments may be deposited on the floodplain and PCBs may thereby enter the terrestrial food chain. High flow events may also increase the bioavailability of PCBs to organisms in the river water.

Animals and plants living in or near the river, such as invertebrates, fish, amphibians, and water-dependent reptiles, birds, and mammals, may be directly exposed to the PCBs from contaminated sediments, river water, and air, and/or indirectly exposed through ingestion of food (*e.g.*, prey) containing PCBs.

Assessment Endpoints

Assessment endpoints are explicit expressions of actual environmental values (*i.e.*, ecological resources) that are to be protected. They focus a risk assessment on particular components of the ecosystem that could be adversely affected due to contaminants at the site. These endpoints are expressed in terms of individual organisms, populations, communities, ecosystems, or habitats with some common characteristics (e.g., feeding preferences, reproductive requirements). The assessment endpoints for the ERA Addendum were selected to include direct exposure to PCBs in Lower Hudson River sediments and river water through ingestion and indirect exposure to PCBs *via* the food chain. Because PCBs are known to bioaccumulate, an emphasis was placed on indirect exposure at various levels of the food chain to address PCB-related risks at higher trophic levels. The assessment endpoints that were selected for the Lower Hudson River are:

- Benthic community structure as a food source for local fish and wildlife
- Protection and maintenance (survival, growth, and reproduction) of local fish populations (forage, omnivorous, and piscivorous)
- Protection and maintenance (survival, growth, and reproduction) of local insectivorous bird populations
- Protection and maintenance (survival, growth, and reproduction) of local waterfowl populations
- Protection and maintenance (survival, growth, and reproduction) of local piscivorous birds populations
- Protection and maintenance (survival, growth, and reproduction) of local insectivorous wildlife populations
- Protection and maintenance (survival, growth, and reproduction) of local omnivorous wildlife populations
- Protection and maintenance (survival, growth, and reproduction) of local piscivorous wildlife populations
- Protection of threatened and endangered species
- Protection of significant habitats

Measurement Endpoints

Measurement endpoints provide the actual measurements used to evaluate ecological risk and are selected to represent mechanisms of toxicity and exposure pathways. Measurement endpoints for future risk generally include modeled concentrations of chemicals in water, sediment, fish, birds, and/or mammals, laboratory toxicity studies, and field observations. The measurement endpoints identified for the ERA Addendum are:

1) Modeled concentrations of PCBs in fish and invertebrates to evaluate food-chain exposure;

2) Modeled total PCB body burdens in receptors (including avian receptor eggs) to determine exceedance of effect-level thresholds based on toxicity reference values (TRVs);

3) Modeled TEQ-based PCB body burdens in receptors (including avian receptor eggs) to determine exceedance of effect-level thresholds based on TRVs;

4) Modeled concentration of PCBs in river water to determine exceedence of criteria for concentrations of PCBs in river water that are protective of benthic invertebrates, fish and wildlife;

5) Modeled concentrations of PCBs in sediment to determine exceedence of guidelines for concentrations of PCBs in sediments that are protective of aquatic health; and

6) Field observations.

Receptors of Concern

Risks to the environment were evaluated for individual receptors of concern that were selected to be representative of various feeding preferences, predatory levels, and habitats (aquatic, wetland, shoreline). The ERA Addendum does not characterize injury to, impact on, or threat to every species of plant or animal that lives in or adjacent to the Hudson River; such a characterization is beyond the scope of the Superfund ecological risk assessment. The following receptors of concern were selected for the ERA Addendum:

Aquatic Invertebrates

- Benthic macroinvertebrate community (e.g., aquatic worms, insect larvae, and isopods)

Fish Species

- Pumpkinseed (*Lepomis gibbosus*)
- Spottail shiner *(Notropis hudsonius)*

- Brown bullhead *(Ictalurus nebulosus)*
- White perch (*Morone americana)*
- Yellow perch (*Perca flavescens)*
- Largemouth bass (*Micropterus salmoides)*
- Striped bass (*Morone saxatilis)*
- Shortnose sturgeon (*Acipenser brevirostrum)*

Birds

- Tree swallow (*Tachycineta bicolor*)
- Mallard (*Anas platyrhychos*)
- Belted kingfisher (*Ceryle alcyon*)
- Great blue heron (*Ardea herodias*)
- Bald eagle (*Haliaeetus leucocephalus*)

Mammals

- Little brown bat *(Myotis lucifugus)*
- Raccoon (*Procyon lotor*)
- Mink (*Mustela vison*)
- River otter (*Lutra canadensis*)

Exposure Assessment

The Exposure Assessment describes complete exposure pathways and exposure parameters (*e.g.*, body weight, prey ingestion rate, home range) used to calculate the concentrations or dietary doses to which the receptors of concern may be exposed due to chemical exposure. USEPA previously released reports on the nature and extent of contamination in the Hudson River as part of the Reassessment RI/FS (*e.g.*, February 1997 Data Evaluation and Interpretation Report, July 1998 Low Resolution Sediment Coring Report, August 1998 Database for the Hudson River PCBs Reassessment RI/FS [Release 4.1], and May 1999 Baseline Modeling Report). The Reassessment RI/FS documents form the basis of the site data collection and analyses that were used in conducting the ERA Addendum. Future (*i.e.*, modeled) concentrations of PCBs in fish, sediments and river water are provided in the ERA Addendum, based on fate and bioaccumulation models by Farley *et al.* (1999) and USEPA's Revised Baseline Modeling Report (USEPA, 2000). Exposure parameters were obtained from USEPA references, the scientific literature, and directly from researchers as reported in the ERA.

Effects Assessment

The Effects Assessment describes the methods used to characterize particular toxicological effects of PCBs on aquatic and terrestrial organisms due to chemical exposure. These measures of toxicological effects, called TRVs, provide a basis for estimating whether the chemical exposure at a site is likely to result in adverse ecological effects.

In conducting the ERA Addendum, USEPA used the TRVs selected in the ERA based on Lowest Observed Adverse Effects Levels (LOAELs) and/or No Observed Adverse Effects Levels (NOAELs) from laboratory and/or field-based studies reported in the scientific literature. These TRVs examine the effects of PCBs and dioxin-like PCB congeners on the survival, growth, and reproduction of fish and wildlife species in the Lower Hudson River. Reproductive effects (*e.g.*, egg maturation, egg hatchability, and survival of juveniles) were generally the most sensitive endpoints for animals exposed to PCBs.

Risk Characterization

Risk Characterization examines the likelihood of adverse ecological effects occurring as a result of exposure to chemicals and discusses the qualitative and quantitative assessment of risks to ecological receptors with regard to toxic effects. Risks are estimated by comparing the results of the Exposure Assessment (*e.g.*, modeled concentrations of chemicals in receptors of concern) to the TRVs developed in the Effects Assessment. The ratio of these two numbers is called a Toxicity Quotient, or TQ.

TQs equal to or greater than one ($TQ \geq 1$) are typically considered to indicate potential risk to ecological receptors, for example reduced or impaired reproduction or recruitment of new individuals. The TQs provide insight into the potential for adverse effects upon individual animals in the local population resulting from chemical exposure. If a TQ suggests that effects are not expected to occur for the average individual, then they are probably insignificant at the population level. However, if a TQ indicates risks are present for the average individual, then risks may be present for the local population.

At each step of the risk assessment process there are sources of uncertainty. Measures were taken in the ERA to address and characterize the uncertainty. For example, in some cases uncertainty factors were applied in developing TRVs. The purpose of these uncertainty factors is to ensure that the calculated TRVs are protective of the receptor species of concern. Another source of uncertainty is associated with the future PCB concentrations in fish. The PCB concentrations in fish presented in the ERA Addendum (forecast from models in Farley *et al.* (1999) and the Revised Baseline Modeling Report (USEPA, 2000) may be significantly underestimated, which may underestimate risks to fish species. However, based on a comparison of measured concentrations of PCBs in fish to modeled concentrations, the forecasts presented in the ERA Addendum are not expected to overestimate future PCB concentration in fish, so that the risks to fish are not expected to be overestimated.

To integrate the various components of the ERA Addendum, the results of the risk characterization and associated uncertainties were evaluated using a weight-of-evidence approach to assess the risk of adverse effects in the receptors of concern as a result of exposure to PCBs in the Lower Hudson River. The weight-of-evidence approach considers both the results of the TQ analysis and field observations for each assessment endpoint. For the mammals and most birds, TQs for the dioxin-like PCBs were greater than the TQs for total PCBs.

Benthic Community Structure

Risks to local benthic invertebrate communities were examined using two lines of evidence. These lines of evidence are: 1) comparison of modeled water column concentrations of PCBs to criteria and 2) comparisons of modeled sediment concentrations to guidelines. Both suggest an adverse effect of PCBs on benthic invertebrate populations serving as a food source to local fish in the Lower Hudson River. Uncertainty in this analysis is considered low.

Local Fish (Forage, Omnivorous, Piscivorous and Semi-piscivorous)

Risks to local fish populations were examined using five lines of evidence. These lines of evidence are: 1) comparison of modeled total PCB fish body burdens to TRVs; 2) comparison of modeled TEQ fish body burdens to TRVs; 3) comparison of modeled water column concentrations of PCBs to criteria; 4) comparison of modeled sediment concentrations to guidelines; and 5) field-based observations. Multiple receptors were evaluated for forage and semi-piscivorous/piscivorous fish.

Collectively, the evidence indicates that future PCB exposures (predicted from 1993 to 2018) are not expected to be of a sufficient magnitude to prevent reproduction or recruitment of common fish species in the Lower Hudson River. However, based upon toxicity quotients, future exposure to PCBs may reduce or impair the survival, growth, and reproductive capability of some forage species (*e.g.*, pumpkinseed) and semi-piscivorous/piscivorus fish (*e.g.*, white perch, yellow perch, largemouth bass, and striped bass), particularly in the upper reaches of the Lower Hudson River.

There is a moderate degree of uncertainty in the modeled body burdens used to evaluate exposure, and at most an order of magnitude uncertainty in the TRVs (for the TEQ-based TRVs, no uncertainty factors were needed).

Modeled concentrations of PCBs in river water in the Lower Hudson River show exceedances of criteria developed for protection of fish and wildlife through the duration of the forecast period (1993 - 2018).

Insectivorous Birds

Risks to local insectivorous bird populations were examined using six lines of evidence. These lines of evidence are: 1) comparison of modeled total PCB dietary doses to TRVs; 2) comparison of modeled TEQ dietary doses to TRVs; 3) comparison of modeled total PCB egg concentrations to TRVs; 4) comparison of modeled TEQ egg concentrations to TRVs; 5) comparison of modeled water column concentrations of PCBs to criteria; and 6) field-based observations. The tree swallow was selected to represent insectivorous bird species.

Collectively, the evidence indicates that future PCB exposures (predicted from 1993 to 2018) are not expected to be of a sufficient magnitude to prevent reproduction or recruitment of common insectivorous bird species in the Lower Hudson River Valley. TQs are all below one for all locations for the entire forecast period (1993 to 2018). However, given that U.S. Fish and Wildlife Service field studies suggest PCBs may cause abnormal nest construction of Upper Hudson River tree swallows, it is possible that future exposure to PCBs in the Lower Hudson River may reduce or impair the reproductive capability of tree swallows, particularly in the upper reaches of the Lower Hudson River.

There is a moderate degree of uncertainty in the calculated modeled concentrations of PCBs in tree swallow diets and the concentrations of PCBs in eggs. There is a low degree of uncertainty associated with tree swallow TRVs, which were derived from field studies of Hudson River tree swallows.

Modeled concentrations of PCBs in river water in the Lower Hudson River show exceedances of criteria developed for the protection of wildlife through the duration of the forecast period (1993-2018).

Waterfowl

Risks to local waterfowl populations were examined using six lines of evidence. These lines of evidence are: 1) comparison of modeled total PCB dietary doses to TRVs; 2) comparison of modeled TEQ dietary doses to TRVs; 3) comparison of modeled total PCB egg concentrations to TRVs; 4) comparison of modeled TEQ egg concentrations to TRVs; 5) comparison of modeled water column concentrations of PCBs to criteria; and 6) field-based observations. The mallard was selected to represent waterfowl.

Collectively, the evidence indicates that future PCB exposures (predicted from 1993 to 2018) are not expected to be of a sufficient magnitude to prevent reproduction or recruitment of common waterfowl in the Lower Hudson River Valley. However, based upon toxicity quotients, future exposure to PCBs may reduce or impair the survival, growth, and reproductive capability of some waterfowl, particularly in the upper reaches of the lower river.

Calculated dietary doses of PCBs and concentrations of PCBs in eggs typically exceed their respective TRVs throughout the modeling period. Toxicity quotients for the TEQ-based (*i.e.*, dioxin-like) PCBs consistently show greater exceedances than for total (Tri+) PCBs. There is a moderate degree of uncertainty in the dietary dose and egg concentration estimates. Given the magnitude of the TEQ-based TQs, they would have to decrease by an order of magnitude or more to fall below one for waterfowl in the Lower Hudson River.

Modeled concentrations of PCBs in river water in the Lower Hudson River show exceedances of criteria developed for the protection of wildlife through the duration of the forecast period (1993-2018).

Piscivorous Birds

Risks to local semi-piscivorous/piscivorous bird populations were examined using six lines of evidence. These lines of evidence are: 1) comparison of modeled total PCB dietary doses to TRVs; 2) comparison of modeled TEQ dietary doses to TRVs; 3) comparison of modeled total PCB egg concentrations to TRVs; 4) comparison of modeled TEQ egg concentrations to TRVs; 5) comparison of modeled water column concentrations of PCBs to criteria; and 6) field-based observations. The belted kingfisher, great blue heron, and bald eagle were selected to represent piscivorous birds.

Collectively, the evidence indicates that future PCB exposures (predicted from 1993 to 2018) are not expected to be of a sufficient magnitude to prevent reproduction or recruitment of these piscivorous species. However, based upon toxicity quotients, future exposure to PCBs may reduce or impair the survival, growth, and reproductive capability of some piscivorous birds, particularly in the upper reaches of the Lower Hudson Rver. Calculated dietary doses of PCBs and concentrations of PCBs in eggs exceed all TRVs (*i.e.*, NOAELs and LOAELs) for the belted kingfisher and bald eagle throughout the modeling period, and exceed NOAELs for the great blue heron. Toxicity quotients for egg concentrations are generally higher than body burden TQs.

There is a moderate degree of uncertainty in the dietary dose and egg concentration estimates. Given the magnitude of the TQs, they would have to decrease by an order of magnitude or more to fall below one for piscivorous birds in the Lower Hudson River. In particular, the bald eagle TQs exceeded one by up to three orders of magnitude. Therefore, even if the factor of 2.5 to adjust from largemouth bass fillets to whole body burden and the subchronic-to-chronic uncertainty factor of 10 used for the body burden TRV are removed, the TQs would remain well over one. These results coupled with the lack of breeding success in Lower Hudson River bald eagles (USGS, 1999) indicate that reproductive effects may be present.

Modeled concentrations of PCBs in river water in the Lower Hudson River show exceedances of criteria developed for the protection of wildlife through the duration of the forecast period (1993-2018).

Insectivorous Mammals

Risks to local insectivorous mammal populations were examined using four lines of evidence. These lines of evidence are: 1) comparison of modeled total PCB dietary doses to TRVs;

2) comparison of modeled TEQ dietary doses to TRVs; 3) comparison of modeled water column concentrations of PCBs to criteria; and 4) field-based observations. The little brown bat was selected to represent insectivorous mammals.

Collectively, the evidence indicates that future PCB exposures (predicted from 1993 to 2018) are not expected to be of a sufficient magnitude to prevent reproduction or recruitment of common insectivorous mammals in the Lower Hudson River Valley. However, exposure to PCBs may reduce or impair the survival, growth, or reproductive capability of insectivorous mammals in the Lower Hudson River. Modeled dietary doses for the little brown bat exceed TRVs by up to two orders of magnitude at all locations modeled. There is a moderate degree of uncertainty in the calculated dietary doses.

Modeled concentrations of PCBs in river water in the Lower Hudson River show exceedances of criteria developed for the protection of wildlife through the duration of the forecast period (1993-2018).

<u>Omnivorous Mammals</u>

Risks to local omnivorous mammal populations were examined using four lines of evidence. These lines of evidence are: 1) comparison of modeled total PCB dietary doses to TRVs; 2) comparison of modeled TEQ dietary doses to TRVs; 3) comparison of modeled water column concentrations of PCBs to criteria; and 4) field-based observations. The raccoon was selected to represent omnivorous mammals.

Collectively, the evidence indicates that future PCB exposures (predicted from 1993 to 2018) are not expected to be of a sufficient magnitude to prevent reproduction or recruitment of common omnivorous mammals in the Lower Hudson River Valley. However, exposure to PCBs may reduce or impair the survival, growth, or reproductive capability of omnivorous mammals in the Lower Hudson River. Modeled dietary doses for the raccoon exceed dietary dose NOAELs on a total PCB (Tri+) basis and all TRVs on a TEQ-basis. There is a moderate degree of uncertainty in the calculated dietary doses.

Modeled concentrations of PCBs in river water in the Lower Hudson River show exceedances of criteria developed for the protection of wildlife through the duration of the forecast period (1993 - 2018).

<u>Piscivorous Mammals</u>

Risks to local semi-piscivorous/piscivorous mammal populations were examined using four lines of evidence. These lines of evidence are: 1) comparison of modeled total PCB dietary doses to TRVs; 2) comparison of modeled TEQ dietary doses to TRVs; 3) comparison of modeled water column concentrations of PCBs to criteria; and 4) field-based observations. The mink and river otter were selected to represent piscivorous mammals.

Collectively, the evidence indicates that future PCB exposures (predicted from 1993 to 2018) are not expected to be of a sufficient magnitude to prevent reproduction or recruitment of these piscivorous species. However, based upon toxicity quotients, future exposure to PCBs may reduce or impair the survival, growth, and reproductive capability of piscivorous mammals, particularly in the upper reaches of the Lower Hudson River. Calculated dietary doses of PCBs exceed the NOAEL on a total PCB basis for both the mink and river otter and exceed all TEQ-based TRVs by up to three orders of magnitude.

There is a moderate degree of uncertainty in the dietary dose estimates. However, given the magnitude of the TQs, they would have to decrease at least an order of magnitude to fall below one. In particular, the river otter TQs exceeded one by up to three orders of magnitude. Therefore, even if the factor of 2.5 to adjust from largemouth bass fillets to whole body burden is removed, the TQs would remain well over one.

Modeled concentrations of PCBs in river water in the Lower Hudson River show exceedances of criteria developed for the protection of wildlife through the duration of the forecast period (1993-2018). In addition, preliminary results from a NYSDEC study indicate that PCBs may have an adverse effect on the litter size and possibly kit survival of river otter in the Hudson River (Mayack, 1999b).

Threatened and Endangered Species

Risks to threatened and endangered species were examined using five lines of evidence. These lines of evidence are: 1) comparison of modeled total PCB dietary doses/egg concentrations to TRVs; 2) comparison of modeled TEQ dietary doses/egg concentrations to TRVs; 3) comparison of predicted modeled water column concentrations of PCBs to criteria; 4) comparison of modeled sediment concentrations of PCBs to guidelines; and 5) field-based observations. The shortnose sturgeon and bald eagle were selected to represent threatened and endangered species.

Collectively, the evidence indicates that future PCB exposures (predicted from 1993 to 2018) are not expected to be of a sufficient magnitude to prevent reproduction or recruitment of threatened or endangered species. However, using the TEQ-based toxicity quotients, potential for adverse reproductive effects in shortnose sturgeon exists, particularly when considering the long life expectancy of the sturgeon. Almost all TQs calculated for the bald eagle (across all locations) exceeded one, in some instances by more than three orders of magnitude. Both the dietary dose and egg-based results were consistent in this regard. Other threatened or endangered raptors, such as the peregrine falcon, osprey, northern harrier, and red-shouldered hawk may experience similar exposures.

There is a moderate degree of uncertainty in the dietary dose estimates. However, the bald eagle TQs exceeded one by up to three orders of magnitude. Therefore, even if the factor of 2.5 to adjust from largemouth bass fillets to whole body burden and the subchronic-to-chronic uncertainty factor of 10 used for the body burden TRV are removed, the TQs would remain well over one.

These results coupled with the lack of breeding success in Lower Hudson River bald eagles (USGS, 1999) indicate that reproductive effects may be present.

Modeled concentrations of PCBs in river water and sediment in the Lower Hudson River show exceedances of the majority of their respective criteria and guidelines through the duration of the forecast period (1993-2018).

Significant Habitats

Risks to significant habitats were examined using four lines of evidence. These lines of evidence are: 1) toxicity quotients calculated for receptors in this assessment; 2) comparison of modeled water column concentrations of PCBs to criteria; 3) comparison of modeled sediment concentrations of PCBs to guidelines; and 4) field-based observations.

Based on the toxicity quotients for receptors of concern, future PCB concentrations modeled for the Lower Hudson River exceed toxicity reference values for some fish, avian, and mammalian receptors. These comparisons indicate that animals feeding on Hudson River-based prey may be affected by the concentrations of PCBs found in the river on both a total PCB and TEQ basis. In addition, based on the ratios obtained in this evaluation, other taxononic groups not directly addressed in this evaluation (*e.g.*, amphibians and reptiles) may also be affected by PCBs in the Lower Hudson River. Many year-round and migrant species use the significant habitats along the Lower Hudson River for breeding or rearing their young. Therefore, exposure to PCBs may occur at a sensitive time in the life cycle (i.e., reproductive and development) and have a greater effect on populations than at other times of the year.

Modeled concentrations of PCBs in river water and sediment in the Lower Hudson River show exceedances of the majority of their respective criteria and guidelines through the duration of the forecast period (1993-2018).

Major Findings of the ERA Addendum

The results of the risk assessment indicate that receptors in close contact with the Lower Hudson River are at an increased ecological risk as a result of future exposure to PCBs in sediments, water, and/or prey. This conclusion is based on a TQ approach, in which modeled body burdens, dietary doses, and egg concentrations of PCBs were compared to TRVs, and on field observations. On the basis of these comparisons, all receptors of concern except the tree swallow are at risk. In summary, the major findings of the report are:

- Fish in the Lower Hudson River are at risk from future exposure to PCBs. Fish that eat other fish (*i.e.*, which are higher on the food chain), such as the largemouth bass and striped bass, are especially at risk. PCBs may adversely affect fish survival, growth, and reproduction.

- Mammals that feed on insects with an aquatic stage spent in the Lower Hudson River, such as the little brown bat, are at risk from future PCB exposure. PCBs may adversely affect the survival, growth, and reproduction of these species.

- Birds that feed on insects with an aquatic stage spent in the Lower Hudson, such as the tree swallow, are not expected to be at risk from future exposure to PCBs.

- Waterfowl feeding on animals and plants in the Lower Hudson River are at risk from PCB exposure. Future concentrations of PCBs may adversely affect avian survival, growth, and reproduction.

- Birds and mammals that eat PCB-contaminated fish from the Lower Hudson River, such as the bald eagle, belted kingfisher, great blue heron, mink, and river otter, are at risk. Future concentrations of PCBs may adversely affect the survival, growth, and reproduction of these species.

- Omnivorous animals, such as the raccoon, that derive some of their food from the Lower Hudson River are at risk from PCB exposure. Future concentrations of PCBs may adversely affect the survival, growth, and reproduction of these species.

- Fragile populations of threatened and endangered species in the Lower Hudson River, represented by the bald eagle and shortnose sturgeon, are particularly susceptible to adverse effects from future PCB exposure.

- Modeled PCB concentrations in water and sediments in the Lower Hudson River generally exceed standards, criteria and guidelines established to be protective of the environment. Animals that use areas along the Lower Hudson designated as significant habitats may be adversely affected by the PCBs.

- The future risks to fish and wildlife are greatest in the upper reaches of the Lower Hudson River and decrease in relation to decreasing PCB concentrations down river. Based on modeled PCB concentrations, many species are expected to be at risk through 2018 (the entire forecast period).

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Chapter 1

1.0 INTRODUCTION

1.1 Purpose of Report

This document presents the baseline Ecological Risk Assessment for Future Risks in the Lower Hudson River (ERA Addendum), which is a companion volume to the baseline Ecological Risk Assessment (ERA) that was released by the U.S. Environmental Protection Agency (USEPA) in August 1999. Together, the two risk assessments comprise the ecological risk assessment for Phase 2 of the Reassessment Remedial Investigation/Feasibility Study (Reassessment RI/FS) for the Hudson River PCBs site in New York.

The ERA Addendum quantitatively evaluates the future risks to the environment in the Lower Hudson River (Federal Dam at Troy, New York to the Battery in New York City) posed by polychlorinated biphenyls (PCBs) from the Upper Hudson River (Hudson Falls, New York to the Federal Dam at Troy, New York), in the absence of remediation. This report uses current USEPA policy and guidance as well as additional site data and analyses to update USEPA's 1991 risk assessment.

Consistent with USEPA guidance (USEPA, 1997b), the ERA addendum calculates the risk to individual receptor species of concern. The ERA addendum uses the same receptor species as the baseline ERA (USEPA, 1999c). The species were selected to represent various trophic levels, a variety of feeding types, and a diversity of habitats associated with the Hudson River. Receptor species were selected as surrogates for the range of species potentially exposed to PCBs in the Hudson River.

Because of the focused nature of the Reassessment RI/FS, a number of technical decisions were made to structure and focus the ERA, as described in the baseline ERA (USEPA, 1999c). The ERA and ERA Addendum focus on particular categories of PCBs that can be supported by the available data and are amenable to modeling. Selection of PCBs categories to measure, model, and assess was based on risk assessment considerations as well as on practical considerations related to modeling requirements. For the ecological risk assessment this led to a decision to evaluate total PCBs as represented by "tri and higher" chlorinated compounds, as well as select congeners. The "tri and higher" group includes the PCB compounds that are most toxic to fish and wildlife and therefore captures most of the toxicity associated with these compounds. Tri and higher totals for the Lower Hudson River that are compared to total PCBs (which include mono and dichlorinated PCBs) may underestimate risks in some instances.

1.2 Report Organization

This ERA follows *Ecological Risk Assessment Guidance for Superfund, Process for Designing and Conducting Ecological Risk Assessments* (ERAGS) (USEPA, 1997b), as detailed in the baseline ERA (USEPA, 1999c). The ERAGS guidance has of eight steps, as shown in Figure

1-2. This ERA Addendum covers Steps 6 and 7 of the ERAGS process (analysis of ecological exposures and effects and risk characterization) for the future risks in the Lower Hudson River. Steps 1-5 were completed in previous reports (*e.g.*, USEPA, 1999c). Step 8, Risk Management, occurs after the completion of the ERA and is the responsibility of the USEPA site risk manager, who balances risk reductions associated with cleanup of contaminants with potential impacts of the remedial actions themselves.

Much of the information used in this addendum was originally presented in the baseline ERA (USEPA, 1999c), where a detailed description of the assumptions and methodology that were used can be found. In keeping with ERAGS, the format of this ERA Addendum is as follows:

- Chapter 1, the introduction, provides an overview of purpose of the report.

- Chapter 2, problem formulation, summarizes the conceptual model, assessment and measurement endpoints, and the receptors of concern from the baseline ERA (USEPA, 1999c).

- Chapter 3, the exposure assessment, discusses modeled PCB concentrations forecast using the Farley *et al.* (1999) and FISHRAND models, identifies exposure pathways for receptors, and summarizes exposure parameters selected for avian and mammalian receptors in the baseline ERA (USEPA, 1999c).

- Chapter 4, the effects assessment, summarizes toxicity reference values (TRVs) selected for each receptor in the baseline ERA (USEPA, 1999c).

- Chapter 5, the risk characterization, uses the exposure and effects assessments to provide a quantitative estimate of risk to receptors. The results of the measurement endpoints are used to evaluate the assessment endpoints selected in the problem formulation phase of the assessment.

- Chapter 6, the uncertainty analysis, summarizes uncertainties associated with the assessment based on the baseline ERA (USEPA, 1999c).

- Chapter 7, conclusions, presents the conclusions of the risk assessment. This section integrates the results of the risk characterization with the uncertainty analysis to provide perspective on the overall confidence in the assessment.

Chapter 2

2.0 PROBLEM FORMULATION

Problem formulation establishes the goals, breadth, and focus of the assessment. It defines the questions and issues based on identifiable complete exposure pathways and ecological effects. A key aspect of problem formulation is the development of a conceptual model that illustrates the relationships among sources, pathways, and receptors.

2.1 Site Characterization

The Hudson River PCBs Site includes the 200 miles (322 km) of river from Hudson Falls, NY to the Battery in New York City, as described in the baseline ERA (USEPA, 1999c). The ERA Addendum covers future risks to the Lower Hudson River, which stretches from the Federal Dam to the Battery. Phase 2 ecological sampling locations are shown in Figure 2-1. The Lower Hudson River is tidal and includes freshwater, brackish, and estuarine habitats, as described below.

2.2 Contaminants of Concern

Consistant with the scopr of the Reassessment RI/FS, the contaminants of concern (COCs) are limited to PCBs. While there are other contaminants at various locations in the Hudson (*e.g.*, metals, polycyclic aromatic hydrocarbons), PCBs are the chemicals that are the basis for the 1984 ROD and the Reassessment RI/FS. Consistent with that focus, the evaluation examines risks posed by the presence of in-place PCBs in river sediments. PCBs can be described as individual congeners, Aroclors, and total PCBs. Total PCBs in this assessment are represented by the trichlorinated and higher congeners (designated Tri+) for the purposes of modeling (USEPA, 1999b), which approximate total PCBs in biota.

2.3 Conceptual Model

A site conceptual model identifies the source, media, pathway, and route of exposure evaluated in the ecological risk assessment, and the relationship of the measurement endpoints to the assessment endpoints (USEPA, 1997b). An integrated site conceptual model was developed for the Hudson River baseline ERA (Figure 2-2). In this model, the initial sources of PCBs are releases from the two GE capacitor manufacturing facilities located in Hudson Falls and Fort Edward, NY.

PCBs enter the Hudson River and adhere to sediments or are redistributed into the water column. Sediments may be deposited on the floodplain during high flow events and provide a pathway for PCBs to enter the terrestrial food chain.

Animals and plants living in or near the Hudson River, such as invertebrates, fish, amphibians, and water-dependent reptiles, birds, and mammals, are potentially exposed to the PCBs from contaminated sediments, surface water, and/or prey. Species representing various trophic levels living in or near the river were selected as receptor species for evaluating potential risks associated with PCBs. Exposure pathways by which these species could be exposed to PCBs were discussed in the baseline ERA (USEPA, 1999c) and are summarized in the following section.

2.3.1 Exposure Pathways in the Lower Hudson River Ecosystem

Ecological receptors may be exposed to PCBs *via* various pathways. A complete exposure pathway involves a potential for contact between the receptor and contaminant either through direct exposure to the media or indirectly through food. Pathways are evaluated by considering information on contaminant fate and transport, ecosystems at risk, and the magnitude and extent of contamination (USEPA, 1997b).

Contaminant fate and transport and the magnitude and extent of contamination have been discussed extensively in other Reassessment RI/FS reports, including the Baseline Modeling Report (USEPA, 1999b), Data Evaluation and Interpretation Report (USEPA, 1997a), Low Resolution Sediment Coring Report (USEPA, 1998a), and the baseline ERA (USEPA, 1999c). Exposure pathways considered in this assessment are: ingestion of contaminated prey, ingestion of contaminated sediments, and ingestion of contaminated surface water.

2.3.2 Ecosystems of the Lower Hudson River

The Lower Hudson River estuary is home to a wide variety of habitats. It is a valuable state and local resource (NYSDEC, 1998a). Many commercially valuable fish and shellfish species including striped bass, shad, Atlantic sturgeon, and blue crab use the estuary for spawning and as a nursery ground. Over 16,500 acres in the estuary have been inventoried and designated significant coastal fish and wildlife habitat. The NYS Natural Heritage Program has identified many areas along the Hudson River estuary where rare plants, animals, or natural communities are found (NYSDEC, 1999b). The estuary is also an important resting and feeding area for migratory birds, such as eagles, osprey, songbirds, and waterfowl (NYSDEC, 1998a).

A number of distinct ecological communities including deepwater; shallows, mudflats, and shore; tidal marsh; and tidal swamp communities are found in the Lower Hudson River. Brief descriptions of these communities are provided below based on a publication of the New York State Department of State and the Nature Conservancy (1990).

Deepwater- The deepwater community includes sections of the lower river with water depths greater than six feet at low tide. Vegetation is limited to phytoplankton in the upper layers of the water column, as light does not generally penetrate deep enough to support photosynthesis of rooted plants. The deepwater community is composed of abundant animal life supported by organic material originating in the watershed. Benthic invertebrates, fish, and fish eating predators (*e.g.*, birds, mammals) are found in this habitat. Fish found in the deepwater community include species such as American shad, blueback herring, alewife, striped bass, Atlantic tomcod, and Atlantic and shortnose sturgeon. Predators of deepwater fish can capture fish near the water's surface (*e.g.*, bald eagles, osprey) or below the surface of the water (*e.g.*, cormorants, loons, and diving ducks).

Shallows, Mudflats, and Shore- These communities include sections of the river found near the low tide mark. Shallows are always below the low tide mark, mudflats are barely exposed at low tide, and the shore is a zone largely exposed at low tide but inundated at high tide. The shallows support a variety of vascular plants rooted in the bottom (*e.g.*, waterweed, water celery, and various pondweeds) and free floating plants (either in the water column or on the surface). Mudflats support plants adapted to being submerged most of the day and then briefly exposed at low tide when they are typically found encrusted in mud. In addition to vascular species, mudflats support significant numbers of periphyton (attached algae) and bacteria that grow on mud or surfaces of vascular plants. Shore areas are found along rocky or gravelly banks. Vegetation may be limited in areas subject to waves, ice scour, and upland erosion.

Shallow waters support many zooplankton species and the animals that feed on them (*e.g.*, fish larvae and fish). Many adult fish found in the shallow water are year-round Hudson River residents including shiners, carp, white catfish, suckers, white and yellow perch, bass, sunfishes, and darters in freshwater regions. Bay anchovies, killifish, silversides, winter flounder, and hog chokers are found in more brackish sections of the river. Many anadromous (*i.e.*, migrating) fish of the deepwater community feed extensively in the shallows while preparing to return to the ocean. Many fish also use the shallows as spawning and nursery grounds.

Numerous upper trophic level bird species (*e.g.*, great blue heron, great egrets, least bittern) feed in shallows and mudflats. Waterfowl feeding on aquatic plants and small fish and sandpipers feeding on seeds, insects, and aquatic invertebrates are found in these communities.

Tidal Marsh- The tidal marsh community includes sections of the Hudson River where tidal waters inundate plants specifically adapted to daily flooding. Lower marsh plants, adapted to daily submersions, include broad-leaved plants such as spatterdock, pickerelweed, arrowhead, bulrushes, and plantains. Upper marsh vegetation consists of plants adapted to partial flooding, which are seldomly or never completely submerged. The upper marsh has a grassy appearance and is dominated by narrow-leaved cattail and common reed.

Tidal marshes provide important feeding and breeding areas for many resident and transient aquatic and terrestrial animals. Fish (*e.g.*, killifish, darters, mummichogs, sunfish, and carp) come into marshes at high tide to feed on invertebrates such as cladocerans, copepods, ostracods, and chironomids. A variety of amphibians, reptiles, birds, and mammals feed on the fish and invertebrates found in marshes. Hudson River tidal marshes support many bird species and large populations of nesting birds, which includes a high density of breeding marsh birds.

Tidal Swamp- The tidal swamp community includes land adjacent to the Hudson River that is regularly flooded by tidal waters. It is dominated by a closed canopy of trees (*e.g.*, green and black ash, red maple, and slippery elm). Below the canopy is a layer of shrubs and vines and at ground level there is a layer of herbs. Tidal swamps occur exclusively in freshwater, either near freshwater tributaries in brackish portions of the estuary or in upstream freshwater sections of the River.

The tidal swamp supports invertebrates and vertebrates feeding on plants, seeds, and organic materials found in the swamp. Terrestrial herbivores and granivores include pheasants, rabbits, squirrels, muskrats, beaver, and deer. Predators of invertebrates and vertebrates found in the swamp include salamanders, toads, snakes, turtles, shrews, foxes, weasels, and mink.

In addition to these communities, freshwater creek and upland forest communities are also ecologically linked to the Hudson River. Exposure to PCBs originating in the River may occur *via* the food chain or floodplain sediments.

Fish, amphibians, reptiles, birds, and mammals potentially found in or along the Hudson River are listed in Tables 2-1 and 2-3 to 2-6 of the baseline ERA (USEPA, 1999c), respectively.

2.3.3 Exposure Pathways

The aquatic and terrestrial pathways for the Lower Hudson River are outlined below and described in detail in Chapter 2 of the baseline ERA (USEPA, 1999c).

2.3.3.1 Aquatic Exposure Pathways

Aquatic and semi-aquatic organisms, such as fish, invertebrates, amphibians, and reptiles (*e.g.*, water snakes), are exposed to PCBs through:

- Direct uptake from water;
- Uptake from sediment; and
- Uptake *via* food.

2.3.3.2 Terrestrial Exposure Pathways

Terrestrial and semi-terrestrial animals, such as amphibians, reptiles, birds, and mammals, can be exposed to PCBs *via*:

- Food uptake;
- Surface water ingestion;
- Incidental sediment ingestion;
- Contact with floodplain sediments/soils; and
- Inhalation of air.

Food uptake of contaminated prey is considered to be the primary PCB exposure pathway (USEPA, 1999c).

2.4 Assessment Endpoints

Assessment endpoints are explicit expressions of actual environmental values (*e.g.*, ecological resources) that are to be protected (USEPA, 1992). They focus the risk assessment on particular components of the ecosystem that could be adversely affected by contaminants from the site (USEPA, 1997b). These endpoints are expressed in terms of individual organisms, populations, communities, ecosystems, or habitats with some common characteristics (*e.g.*, feeding preferences, reproductive requirements). In addition to protection of ecological values, assessment endpoints may also encompass a function or quality that is to be maintained or protected.

The assessment endpoints selected for the ERA Addendum focus on the protection and maintenance of local fish and wildlife populations exposed to PCBs in Hudson River sediments and water through sediment and surface water ingestion, uptake from water, and indirect exposure to PCBs *via* the food chain. Because PCBs are known to bioaccumulate, an emphasis was placed on exposure at various levels of the food chain to address PCB-related risks at higher trophic levels. The assessment endpoints selected to evaluate future risks in the Lower Hudson are:

- Benthic aquatic life as a food source for local fish and wildlife.

- Survival, growth, and reproduction of:
  - local forage fish populations;
  - local omnivorous fish populations; and
  - local piscivorous fish populations.
- Protection (*i.e.*, survival, growth, and reproduction) of local wildlife including:
  - insectivorous bird populations;
  - waterfowl populations;
  - semi-piscivorous/piscivorous bird populations;
  - insectivorous mammal populations;
  - omnivorous mammal populations; and
  - semi-piscivorous/piscivorous mammals populations.
- Protection of threatened and endangered species.
- Protection of significant habitats.

The selected assessment endpoints along with specific ecological receptors and measures of effect are listed in Table 2-1. These endpoints reflect a combination of values that have been identified by USEPA, New York State Department of Environmental Conservation (NYSDEC), US Fish and Wildlife Service (USFWS), and National Oceanic and Atmospheric Administration (NOAA) as being important, and/or habitats or species that have been identified as ecologically valuable.

2.5 Measurement Endpoints (Measures of Effect)

Measures of effect provide the actual measurements used to estimate risk, as described in the baseline ERA (USEPA, 1999c). Because of the complexity and inherent variability associated with ecosystems, there is always a certain amount of uncertainty associated with estimating risks. Measurement endpoints typically have specific strengths and weaknesses related to the data quality, study design and execution, and strength of association between the measurement and assessment endpoint. Therefore, it is common practice to use more than one measurement endpoint to evaluate an assessment endpoint, when possible.

Measures of effect used to evaluate each assessment endpoint in this addendum are the same as those used in the baseline ERA (USEPA, 1999c) and include:

- Modeled total PCB (*i.e.*, Tri+ congeners) body burdens in fish, birds, and mammals for 25 years (1993 to 2018) to determine exceedance of effect-level thresholds based on toxicity reference values (TRVs) derived in the baseline ERA (USEPA, 1999c).

- Modeled TEQ-based PCB body burdens in fish, birds, and mammals for 25 years (1993 to 2018) to determine exceedance of effect-level thresholds based on TRVs derived in the baseline ERA (USEPA, 1999c).

- Modeled total PCB egg concentrations in birds for 25 years (1993 to 2018) to determine exceedance of effect-level thresholds based on TRVs derived in the baseline ERA (USEPA, 1999c).

- Modeled TEQ-based PCB egg concentrations in birds for 25 years (1993 to 2018) to determine exceedance of effect-level thresholds based on TRVs derived in the baseline ERA (USEPA, 1999c).

- Modeled PCB concentrations in fresh water for 25 years (1993 to 2018) compared to NYS Ambient Water Quality Criteria (AWQC) for the protection of benthic aquatic life and protection of wildlife from toxic effects of bioaccumulation (NYSDEC, 1998b).

- Modeled PCB concentrations in sediment for 25 years (1993 to 2018) compared to applicable sediment benchmarks such as NOAA Sediment Effect Concentrations for PCBs in the Hudson River (NOAA, 1999), NYSDEC Technical Guidance for Screening Contaminated Sediments (1999a), Ontario sediment quality guideline (Persaud *et al.* 1993), and Washington Department of Ecology guidelines for protection of aquatic life (1997).

- Available field observations on the presence and relative abundance of Lower Hudson River fish and wildlife as an indication of the ability of the species to maintain populations.

- Available field observations on the presence and relative abundance of the wildlife species using significant habitats within the Lower Hudson River as an indication of the ability of the habitat to maintain populations.

Risk hypotheses posed as risk questions, along with specific measurement endpoints selected for each assessment endpoint, are provided in Table 2-2.

Effect-level concentrations are measured by TRVs. TRVs are exceeded when the modeled dose or concentration for the site is greater than the benchmark dose or concentration (*i.e.*, toxicity quotient [TQ] exceeds 1). Equations for estimating avian and mammalian dietary doses, avian egg concentrations, and fish body burdens are provided in Chapter 3 of the baseline ERA (USEPA, 1999c).

Population-level effects are determined for each receptor species by evaluating the species life-history and the magnitude of the TQ over time. TQs equal to or greater than one across the entire 25-year modeling period suggests sustained risk. If the life span of receptor covers only a fraction of the modeling period, then population level effects are more likely given the time trajectory. The results of all measurement endpoints, such as modeled total PCB dietary doses and/or egg concentrations, modeled TEQ-based PCB dietary doses and/or egg concentrations, exceedances of benchmarks and criteria, are used in a weight-of-evidence approach. For receptors with small populations (*e.g.*, threatened or endangered species), individual-level effects may place the population at risk.

2.6 Receptors of Concern

Potential adverse effects are evaluated for selected receptor species that represent various trophic levels living in or near the Lower Hudson River. These receptors are used to establish assessment endpoints for evaluation of risk. Receptors were selected to represent different trophic levels, a variety of feeding types, and a diversity of habitats (*e.g.*, aquatic, wetland, shoreline). Specific fish, avian, and mammalian species were selected for evaluation as surrogate species for the range of species likely to be exposed to PCBs in the Lower Hudson River. As described in the baseline ERA (USEPA, 1999c), species were selected based on species sensitivity to PCBs, societal relevance of selected species, discussions with agency representatives, and comments received on the ERA Scope of Work (USEPA, 1998c; USEPA, 1999a).

2.6.1 Fish Receptors

The Hudson River is home to over 200 species of fish (Stanne *et al.* 1996). The following eight fish species, representing a range of trophic levels were evaluated in the ERA and are also evaluated in the ERA Addendum:

- Spottail shiner (*Notropis hudsonius*) - forage fish;
- Pumpkinseed (*Lepomis gibbosus*) - forage fish;
- Brown bullhead (*Ictalurus nebulosus*) - omnivore;
- White perch (*Morone americana*) - semi-piscivore;
- Yellow perch (*Perca flavescens*) - semi-piscivore;
- Largemouth bass (*Micropterus salmoides*) - piscivore;
- Striped bass (*Morone saxatilis*) - piscivore; and,
- Shortnose sturgeon (*Acipenser brevirostrum*) - omnivore (evaluated only in the context of endangered and threatened species).

These forage fish, piscivorous/semi-piscivorous fish, and omnivorous fish provide a general estimate of PCB bioaccumulation potential according to trophic status and are designed to be protective of potential PCB exposures to other, less common species. Detailed profiles of the fish species are provided in Appendix D of the baseline ERA (USEPA, 1999c).

2.6.2 Avian Receptors

Five avian receptors were selected to represent various trophic levels and habitat use of the numerous year-round residents and migratory bird species found along the Hudson River.

- Tree swallow (*Tachycineta bicolor*)- insectivore;
- Mallard (*Anas platyrhychos*) - aquatic plants and animals;
- Belted kingfisher (*Ceryle alcyon*) - piscivore;
- Great blue heron (*Ardea herodias*) - piscivore; and
- Bald eagle (*Haliaeetus leucocephalus*) - piscivore.

Detailed life history profiles of the avian species listed below are provided in Appendix E of the baseline ERA (USEPA, 1999c).

2.6.3 Mammalian Receptors

The potential mammalian receptors found along the Hudson River also represent various trophic levels and habitats. The four mammals selected to serve as representative receptors in baseline ERA and the ERA Addendum are:

- Little brown bat (*Myotis* spp.) - insectivore;
- Raccoon (*Procyon lotor*) - omnivore;
- Mink (*Mustela vison*) - piscivore; and
- River Otter (*Lutra canadensis*) -piscivore.

Detailed profiles of these mammalian species are provided in Appendix F of the baseline ERA (USEPA, 1999c).

2.6.4 Threatened and Endangered Species

Federal and State threatened and endangered species found in the Lower Hudson Valley are:

- Karner blue butterfly (*Lycaeides melissa samuelis*) - federal- and State-listed endangered;
- Shortnose sturgeon (*Acipenser brevirostrum*) - federal- and State-listed endangered;
- Northern cricket frog (*Acris crepitans*)-State-listed endangered;
- Bog turtle (*Clemmys muhlenbergii*) - State-listed endangered;
- Blanding's turtle (*Emydoidea blandingii*) - State-listed threatened;
- Timber rattlesnake (*Crotalus horridus*)- State-listed threatened;
- Peregrine falcon (*Falco peregrinus*) - State-listed endangered;

• Bald eagle (*Haliaeetus leucocephalus*) - State-listed endangered and federal-listed threatened;
• Osprey (*Pandion haliaetus*) - State-listed threatened;
• Northern harrier (*Circus cyaneus*) - State-listed threatened;
• Red-shouldered hawk (*Buteo lineatus)* - State-listed threatened;
• Indiana bat (*Myotis sodalis*) - federal-listed endangered; and
• Eastern woodrat (*Neotoma magister*) - State-listed endangered.

Profiles of these threatened and endangered species are provided in Appendix G of the baseline ERA (USEPA, 1999c).

New York State avian species of concern found in the vicinity of the Hudson River include the least bittern (*Ixobrychus exilis*), Cooper's hawk (*Accipiter cooperii*), upland sandpiper (*Bartramia longicauda*), shorteared owl (*Asio flammeus*), common nighthawk (*Chordeiles minor*), eastern bluebird, (*Sialia sialis*), grasshopper sparrow (*Ammodramus savannarum*), and vesper sparrow (*Pooecetes gramineus).*

Amphibians of special concern listed by NYS potentially found along the Lower Hudson River include the Jefferson salamander (*Ambystoma jeffersonianum*), bluespotted salamander (*Ambystoma laterale*, and spotted salamander (*Ambystoma maculatum).* Reptiles of special concern include spotted turtle (*Clemmys guttata*), wood turtle (*Clemmys insculpta*), diamondback terrapin (*Malaclemys terrapin*), and worm snake (*Carphophis amoenus*).

The Hudson's tidal habitats support a number of rare plant species. A list of these species is provided in Appendix G of the baseline ERA (USEPA, 1999c).

This ERA Addendum evaluates risks to threatened and endangered species as represented by the bald eagle and shortnose sturgeon, consistent with the baseline ERA.

2.6.5 Significant Habitats

All portions of the Hudson River have value for plants and animals. However, 34 specific sites in the Lower Hudson River have been designated as Significant Coastal Fish and Wildlife Habitats under NYS' Coastal Management Program. Five additional sites have been identified as containing important plant and animal communities to bring the total number of sites to 39 (see Table 2-11 of the baseline ERA [USEPA, 1999c]). Four of these areas comprise the Hudson River National Estuarine Research Reserve (NERR), administered by NYS in partnership with NOAA.

Significant habitats contain areas that are unique, unusual, or necessary for continued propagation of key or rare and endangered species. Rare ecological communities and areas of concern often form part or all of the areas considered to be significant habitats. The community types, rare species, and valuable species found at each of these sites are summarized in Table 2-3 based on information provided in New York State Department of State and The Nature Conservancy (1990).

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3.0 EXPOSURE ASSESSMENT

The exposure assessment characterizes exposure concentrations or dietary doses for the selected receptors. Exposure concentrations are estimates of the PCB concentrations modeled under site-specific assumptions and are expressed as total PCBs (as Tri+) and dioxin-like toxic equivalencies (TEQs) to which selected receptors are exposed.

Several exposure models were developed to evaluate the potential risk of PCB exposures under baseline conditions. Sediment and water concentrations were estimated using the model developed by Farley *et al.* (1999) for the Hudson River Foundation (*i.e.,* independent of USEPA's Reassessment RI/FS), as described later in this section. The FISHRAND model (USEPA, 1999c and 2000) was used to calculate all fish body burdens from the sediment and water column concentrations forecast by the Farley model. The results of these models were used to estimate dietary doses to the avian and mammalian receptors for the period 1993-2018. Modeled fish body burdens were compared directly with the fish toxicity reference values to determine potential risk.

Egg concentrations in piscivorous receptors were estimated by applying a biomagnification factor from the literature (Giesy *et al.*, 1995) assumed to be 28 for total PCBs and 19 for TEQ-based concentrations. These factors were applied to both the observed and modeled fish concentrations to calculate egg concentrations in the bald eagle, great blue heron, and belted kingfisher. The USFWS data were used to determine a tree swallow egg to emergent aquatic insect (assumed as benthic invertebrate) biomagnification factor. The USFWS data were also used to establish a mallard duck egg to emergent aquatic insect biomagnification factor.

PCB exposures are evaluated using total PCB concentrations expressed in terms of the trichlorinated (Tri+) and higher PCB congeners in a series of body burden, dietary dose, and/or egg concentration models and using dioxin-like TEQ exposure concentrations based on toxic equivalency factors (TEFs) in a series of body burden, dietary dose and/or egg concentration models. As discussed in Appendix K of the baseline ERA (USEPA, 1999b), the Tri+ sum is nearly identical to the total PCB concentration in fish due to the lack of significant concentrations of monochloro or dichloro congeners in fish tissue.

These approaches involve the construction of a series of models to first estimate PCB concentrations in sediment, water and white perch *via* the Farley model (Farley *et al.,* 1999) with subsequent application of the FISHRAND model (USEPA, 1999c and 2000) to estimate concentrations in fish tissue, and finally the construction of exposure models to estimate body burdens, dietary doses, and/or egg concentrations in the various ecological receptors. These estimates were then compared to the toxicity reference values (TRVs) discussed later in this report.

3.1 Quantification of PCB Fate and Transport: Modeling Exposure Concentrations

The results of the sampling studies for the Reassessment RI/FS have been previously described in several Phase 2 reports, in particular the DEIR (USEPA, 1997) and the ERA (USEPA, 1999c). In this report, a model of Lower Hudson PCB transport developed by Farley *et al.* (1999), supplemented by two USEPA models (HUDTOX and FISHRAND; USEPA, 1999b and 2000), is applied to estimate current and future levels of PCB contamination in sediments, water and fish. The ERA Addendum uses a forecast of 25 years, from 1993 to 2018) while the Mid-Hudson Human Health Risk Assessment (USEPA, 1999d) uses up to a 41 year forecast (1999 to 2040). The forecast data are identical for the overlapping period (*i.e.*, 1999 to 2018).

The development and calibration of the model developed by Farley *et al.* is described in Farley *et al.*(1999) and is not repeated here. The model's calibration used USEPA sampling data from the Lower Hudson. The estimation of future PCB loads to the Lower Hudson from the Upper Hudson was based on results from the USEPA's Upper Hudson model (HUDTOX) (USEPA, 1999c and 2000). Estimation of fish body burdens was achieved through the use of the Farley *et al.* (1999) model as well as USEPA's FISHRAND model which was also developed as part of the Upper Hudson modeling effort (USEPA, 1999c and 2000).

This discussion of the modeling effort is comprised of three sections. The first, Section 3.1.1, describes the modeling approach used and provides details on how the fate, transport and bioaccumulation models were used. Because pre-existing models are used, no discussion of the construction and calibration of the models is presented and the reader is referred to the original modeling reports for additional information. Section 3.1.1 also provides a qualitative discussion on model verification by comparing the model output to previous modeling efforts as well as to sample data from the USEPA, NOAA and NYSDEC. Section 3.1.2 presents the model results which are used in the ERA Addendum and the Mid-Hudson HHRA (USEPA, 1999d). Section 3.1.3 provides a brief summary of the modeling analysis. Section 3.2 provides a summary of the exposure point concentrations used in the ERA Addendum.

3.1.1 Modeling Approach

Four separate models are used to calculate the exposure point concentrations in the Lower Hudson. The fate and transport model developed by USEPA for the Upper Hudson River (HUDTOX) provides the flux of PCBs over the Federal Dam into the Lower Hudson River (USEPA, 1999b). These results represent an external input to the Lower Hudson River fate and transport model (*i.e.*, the Farley *et al.*, 1999 model). The Farley *et al.* (1999) fate and transport model developed specifically for the Lower Hudson River is used to generate the water and sediment concentrations for the Lower Hudson River risk assessments. The water and sediment concentrations from the Farley fate and transport model are used as input for the USEPA bioaccumulation model (FISHRAND) to generate the PCB body burdens for all fish species examined in the Lower Hudson. The Farley bioaccumulation model was applied to yield PCB concentrations in white perch and striped bass for comparison purposes only.

3.1.1.1 Use of the Farley Models

The model segmentation for the Farley *et al.* (1999) fate and transport and bioaccumulation models is shown in Figure 3-1. Water column segments 1 to 14 correspond to the Lower Hudson between RM 153.5 and 14. There are 30 water column segments in all, which are combined into five food web regions. Food web regions 1 and 2 cover the spatial extent of the Lower Hudson River risk assessments. The sediment and dissolved water column concentrations of PCBs obtained for each of the segments of the fate and transport model are averaged by food web region utilized by the bioaccumulation model. Detailed descriptions of the models are given in Farley *et al.* (1999). Few changes were needed to make the models usable for the ERA Addendum and Mid-Hudson HHRA.

Unlike the HUDTOX model developed for the Upper Hudson, the Farley *et al.* (1999) model is based on five separate homologue groups (dichloro to hexachloro homologues) and requires external load estimates for each group. For comparison, the HUDTOX model uses the sum of the trichloro and higher homologues (Tri+), total PCBs and 5 individual congeners. In the original analysis by Farley *et al.* (1999), there were few bases on which to estimate future loads at the Federal Dam and so the original model was only run through the year 2001 (*i.e.*, to 2002).

For the ERA Addendum, the flux over the Federal Dam for each homologue is derived from the flux of Tri+ PCBs given by the HUDTOX model (USEPA, 1999c and 2000). In order to use the Tri+ flux given by the HUDTOX model, a basis for conversion of the Tri+ load to individual homologue loads was required. This was accomplished through the use of Tri+ to homologue conversion factor for each homologue group. These factors were determined by analyzing the available USEPA and General Electric Company water column data. Table 3-1 gives the means of conversion for each homologue during both the calibration and forecast periods. This conversion is described in Appendix A.

The Farley *et al.* (1999) models were originally designed to run for a 15 year period, 1987-2002. Because a 40 year forecast of concentrations is required for the Mid-Hudson HHRA, the models are run in 15 year increments with the final conditions in each model segment and each modeled species becoming the initial conditions for the next 15 years. The major external PCB load to the Lower Hudson, *i.e.*, the load from the Upper Hudson, was estimated using the 40-year forecast from the HUDTOX model, assuming a constant concentration of 10 ng/L at the upstream boundary of the HUDTOX model (USEPA, 2000). For the purposes of this ERA Addendum, only the model output from the period 1993 to 2018 was used.

Prior to using the forecast from the Farley *et al.* (1999) models in the risk assessments, an examination of the Farley model results was performed for the calibration period 1987 to 1997. In this examination, the original calibration curve developed by Farley *et al.*(1999) was compared with model results produced using the HUDTOX model PCB loads to the Lower Hudson. In this fashion, the effects of any differences in Upper Hudson load assumptions could be examined. The results of this comparison are discussed later in Section 3.1.1.3.

The Farley *et al.* (1999) models have been updated since the report was finalized in March 1999. In the fate and transport model, the suspended solids loads to Newark Bay were found to be too high and were corrected. This correction will have the greatest impact on food web region 3 and water column segments 15 and higher. Because these areas are not considered in the ERA Addendum and Mid-Hudson HHRA, the impact of these changes is minimal and this revision was not included in this Lower Hudson modeling analysis. In ignoring this correction, the maximum effect on food web region 2 (RM 14 to 60) would be slightly increased PCB concentrations, potentially yielding a slight overestimate of the risks for RM 14 to 60. Because the resulting risk estimate would still be protective of human health and the environment, no effort was made to update the Lower River fate and transport calculations to reflect the minor correction made to Farley *et al.* (1999).

The Farley *et al.* (1999) bioaccumulation model also underwent revisions after the original report was finalized. These revisions relate to the absorption efficiencies for PCBs across the fish digestive system and the estimation of lipid levels in fish. The July 1999 version of the Farley *et al.* (1999) bioaccumulation model incorporating these revisions (Cooney, 1999) is used in this report.

3.1.1.2 Use of FISHRAND

The FISHRAND model was used to model PCB concentrations in all of the fish receptors examined in the ERA Addendum except for striped bass. A full description of this model is given in USEPA (2000). The differences from the application of the FISHRAND model to the Upper Hudson River to the Lower Hudson River are:

- Water and sediment concentrations estimated from the Farley *et al.* (1999) fate and transport model are used;
- The percent lipid distribution is significantly different for the Lower Hudson River largemouth bass with an average lipid content of 2.5% in the Lower Hudson River versus 1.3% in the Upper Hudson River;
- The total organic carbon value for sediment segments used in the Farley *et al.* (1999) fate and transport model is used; and
- The K_{ow} values specified in USEPA (2000) for the Upper Hudson River below the Thompson Island Dam are applied to the Lower Hudson River.

Estimation of Striped Bass Body Burdens in the Lower Hudson

The Farley bioaccumulation model was used to estimate PCB levels for striped bass which migrate up to food web region 2 (*i.e.*, fish which remain downstream of the salt front, approximately RM 60). The model does not provide striped bass concentrations in food web region 1 (*i.e.*, the freshwater Lower Hudson). In order to estimate striped bass body burdens in food web region 1, the largemouth bass body burdens estimated from the FISHRAND model were multiplied by the ratio of striped bass to largemouth bass body burdens (MCA, 1999). Observed striped bass and largemouth bass concentrations from NYSDEC data were used to construct the ratio at RMs 152 and 113. The averaged concentrations for each year and species are shown in Table 3-2. Ratios for striped bass to white perch are also presented in the table for comparison.

Table 3-2a shows that the average ratio between measured striped bass and largemouth bass at RM 152 is approximately 2.5 (standard deviation = 1.6). In all instances, the data were restricted to fish larger than 25 cm to represent fish that would actually be caught and kept by an angler. This criterion was met by all largemouth bass samples but resulted in the exclusion of several striped bass samples. A similar ratio is obtained between striped bass and white perch, 3.43 (standard deviation of 4.1). Notably, if the year 1990 is eliminated from the white perch comparison, then the ratio becomes 1.62 (standard deviation of 0.4). However, elimination of an entire year of data given the small sample size is unjustified and was not considered.

The striped bass to largemouth bass ratio was also examined on a monthly basis at RM 152 as shown in Table 3-2b. All largemouth bass and white perch samples were collected in May and June at this location. Striped bass were collected in June, July, August, and October at RM 152. Three separate ratios were calculated, comparing the May-June largemouth bass with the June-August, June-July and June-only striped bass data. In all cases, the calculated ratios were essentially the same, ranging between 2.5 and 2.6. Based on these results, the ratio of 2.5 was used to approximate striped bass concentrations for 1998 to 2040 for RM 152. This is accomplished by simply multiplying the modeled concentrations in largemouth bass at this location by 2.5 to estimate the striped bass concentrations.

At RM 113, all of the largemouth bass and striped bass data were obtained in May and June sampling events, so a similar comparison could not be made. At RM 113, the striped bass to largemouth bass ratio is very different. The ratios in this region are much lower than at RM 152, with an average ratio of 0.52 and also exhibit less variability (standard deviation = 0.2). The striped bass concentrations are estimated in the same fashion as at RM 152, only with a multiplier of 0.52 instead of 2.5.

3.1.1.3 Comparison to the Farley *et al.* (1999) Model for the Period 1987 to 1997

In order to assess the impact to the Farley *et al.* (1999) model made by changing the Upper Hudson River PCB loads, the model inputs and outputs were compared. Specifically, the external load estimates (*i.e.*, an input to the Farley model) made by Farley *et al.*(1999) were first compared with the external loads estimated *via* HUDTOX for the calibration period 1987-1997. Differences in these load estimates should be evident in the model output because the Upper Hudson is such a major source of PCBs to the Lower Hudson.

Secondly, the Farley *et al.* (1999) model output in the form of white perch and striped bass body burdens were then compared between the March 1999 Farley *et al.* (1999) model results and the Farley *et al.* (1999) models rerun with the HUDTOX estimates of PCB flux over the Federal Dam.

The results of the Upper Hudson load comparison show the importance of the Upper Hudson in smoothing loads originating above Thompson Island (TI) Dam. Overall, both the Farley *et al.* (1999) and HUDTOX load estimates deliver approximately the same amount of PCBs to the Lower Hudson over the ten year calibration period (1987 - 1997). The comparison of the fish body burdens shows that the adjustments to the model made by Farley *et al.* (Cooney, 1999) are more important than any differences in the sequence of PCB loads assumed by Farley *et al.* (1999) and HUDTOX.

<u>Comparison of HUDTOX and Farley *et al.* (1999) PCB Load Estimates at the Federal Dam</u>

The revision of the flux of PCBs over the Federal Dam at Troy is the only modification made to the March 1999 Farley fate and transport model for the ERA Addendum and Mid-Hudson HHRA. The difference in magnitude between Farley's original flux estimate and that derived from the HUDTOX model can be seen in Table 3-3. This table shows the two estimates of the PCB homologue loads. The cumulative tri-through-hexa-load estimates over the Federal Dam from the

Farley model compare favorably with the estimates from HUDTOX for the period 1987-1997. The largest difference is 101 kg for the tri homologue, representing a cumulative difference of about 4 percent relative to the estimate by Farley *et al.* (1999) (see Table 3-3). Conversely, the estimates for the di homologue differ by a greater amount, 895 kg (76 percent relative to Farley *et al.* 1999). The Farley *et al.* (1999) model used the General Electric Company water column samples at TI Dam to estimate all homologue loads during the calibration period. As described in Appendix A and presented in Table A-2, the di homologue fraction based on HUDTOX was calculated from the Tri+ PCBs by applying a ratio developed from the USEPA Phase 2 water column data. Notably, the largest differences are for the homologue which matters least to Lower Hudson fish body burdens. It is noteworthy as well that the cumulative HUDTOX loads are closer to the load estimates made on a strictly statistical basis, as presented in the DEIR (USEPA, 1997).

The cumulative loads from both modeling estimates are plotted against time in Figure 3-2. Evident in all diagrams is a distinct difference in the timing of the loads to the Lower Hudson. Specifically, the loads estimated by Farley *et al.*(1999) show a distinct rise in the 1991-1993 period while those estimated from HUDTOX show a more gradual rise through the calibration period. This is a result of the assumptions used in creating the two estimates. In the estimate by Farley *et al.* (1999), the measured loads at TI Dam are directly translated to the Lower Hudson. In the HUDTOX-based estimates, loads at TI Dam are affected by the intervening 35 miles of the Upper Hudson, essentially buffering these loads and spreading them out over a longer time period. These assumptions bear directly on the Lower Hudson fish body burdens because the external load determines much of the fish exposure.

For tri through hexa homologues, the Farley *et al.* (1999) estimate is less than the HUDTOX estimate from 1987-1991 and greater than the HUDTOX estimate for 1992-1997, yielding cumulative loads which are quite similar. The Farley *et al.* (1999) estimate is always less than the HUDTOX estimate for the di homologue. This is attributed in part to the lower sensitivity of the General Electric Company data which was used by Farley *et al.* (1999) for this estimate, as discussed above. In addition, the Farley *et al.* (1999) model estimates for the period 1987-1991 were based on a total PCB load trajectory derived from an earlier modeling analysis prepared by Thomann (1989). The homologue distribution was assumed to be the same as that measured in 1991 by the General Electric Company. Conversely, the HUDTOX model is calibrated to the USGS data during this period. Lastly, it is unclear whether the General Electric Company data used by Farley *et al.* (1999) had been corrected for the BZ#4 bias as documented by QEA in O'Brien and Gere (1998). Overall, it is apparent that the assumptions made by Farley *et al.* and the loads derived from HUDTOX will yield different concentrations of PCBs on the Lower Hudson on a year-to-year basis. In the latter period of record, 1994-1998, the results appear to converge as upstream loads become more regular and predictable. (Note the parallel rates of increase in the cumulative curves.)

<u>Comparison of White Perch and Striped Bass Body Burdens</u>

Two changes in the Farley *et al.* (1999) bioaccumulation model are reflected in the comparisons described below. First, the timing and magnitude of the Upper Hudson loads to the Lower River have been changed as described above. Second, the bioaccumulation model itself has been modified by Farley *et al.* (1999), changing the response between the exposures and the fish body burdens. In this correction (Cooney, 1999), the lipid content of the modeled species was decreased to match the lipid content of fish sampled by NYSDEC in the 1990s. This serves to decrease the body burdens predicted by the application of the Farley *et al.* (1999) model regardless of the assumptions of the upstream loading.

The change in the body burden for white perch and striped bass resulting from these changes can be seen by plotting the model results from the March 1999 report (Farley *et al.*, 1999) and this analysis on the x and y axes, respectively, for each time step (approximately a 2 week period) over the entire calibration period (1987 to 1997). Tri+ PCBs (here defined as the sum of the tri through hexa homologues) are plotted because this fraction is most prominent in the fish body burdens (there is little contribution from the di fraction). This also minimizes the effect of the different bases used to estimate the di homologue fraction.

The results are shown in Figure 3-3 for the white perch and Figure 3-4 for the striped bass. The food web region 1 white perch values differ greatly, with the March 1999 values from Farley *et al.* (1999) being distinctly higher. The scatter in the data is attributed to the sensitivity of the white perch model in this food web region to the Upper Hudson River PCB loads. Nonetheless, the paired results do form a linear trend (although not a line), indicating a similar kind of response in both models. The displacement of the line away from the 1:1 line is largely attributed to the revisions to the bioaccumulation model made since the modeling report was released (Cooney, 1999 and Farley *et al.*, 1999). The scatter about the line is attributed to the loading differences, with the points falling above the line when the HUDTOX loading estimates are higher than those given by Farley *et al.* (1999). The points fall below the line when the converse is true. The plot of white perch estimates in food web region 2 is displaced from the 1:1 line by an amount similar to that for food web region 1 but the slope and the scatter in the data are much less as indicated by the difference in the R^2 values. The decreased scatter is attributed to a diminished sensitivity to the Upper Hudson loads in this region of the Hudson, with food web region 1 of the Hudson serving to buffer the variations in the Upper Hudson loads prior to their delivery to food web region 2.

The striped bass values (food web region 2 only) for both model runs is similar with slopes and regression coefficients near 1, showing that the modeled striped bass is not sensitive to this change in Upper Hudson River PCB loads.

3.1.1.4 Comparison Between Model Output and Sample Data

While the comparisons described in Section 3.1.1.3 are useful in examining the effects of model assumptions relative to the original model, it is also important to examine the correlation of the model output with the measurement results. Data from the Farley *et al.* (1999) model run with the Upper Hudson River loads determined by HUDTOX were compared to the water, sediment and fish samples taken from between 1987 and 1997 in order to test the accuracy of the Farley *et al.* (1999) model with the revised upstream loads. USEPA Phase 2 water and sediment samples and NYSDEC fish samples are available from the Lower Hudson River for this time period. Because the water and sediment samples from this portion of the river are relatively few and limited to one or two years, this comparison provides only a limited assessment of the fate and transport model approach.

The NYSDEC fish data represent a more extensive data set and, therefore, provide a better basis for assessing the overall modeling approach.

Dissolved Phase PCBs in Water Column

Modeled dissolved phase PCB concentrations are plotted by river mile for April and August 1993 against the USEPA Phase 2 water column samples in Figure 3-5. The dissolved phase data are especially important because it is the data input from the Farley fate and transport model into the bioaccumulation models. For April 1993, the model agrees reasonably well with the sampled data at RMs 77 and 125, but is 0.02 $\mu g/L$ lower than the sampled data at RM 152. For August 1993, the modeled results are from 0.01 to 0.02 $\mu g/L$ (or a factor of 2 to 3) lower than the sampled data. These results suggest that the Farley model may overestimate losses from the water column during the summer period. Nonetheless, the model trend is similar to the measured trend, with a gradual decline in concentration with RM, as would be expected in the absence of additional significant external sources of PCBs.

The dissolved-phase homologue patterns for August and September 1993 are shown in Figure 3-6. The homologue pattern derived from the Farley *et al.* (1999) model with the HUDTOX loads yields fairly good agreement with the sampled data based on the relative proportions of the homologues. Again, the modeled concentrations are lower for this period than the sampled concentrations, indicating that the possible overestimate of water column loss in the summer affects the entire pool of congeners and not just a single homologue.

Sediment Concentrations

Modeled surface sediment concentrations from 0-2.5 cm and 2.5-5 cm are plotted against the USEPA Phase 2 ecological samples (approximately 5 cm in depth). The modeled data fall within the range of the sampled concentrations for all RMs except for RM 47. At this location, the modeled values are about 0.1 ppm below the lowest sampled value. These results suggest that the model is able to represent the general level of sediment contamination in the river as a function of distance downstream.

Fish Body Burdens

The Farley bioaccumulation model yielded body burdens for white perch in regions 1 and 2 and striped bass in region 2 only. The modeled white perch and striped bass body burdens are plotted against sample data from NYSDEC in Figures 3-8 and 3-9. For white perch, the modeled data fall within the range of the sampled data for all years except 1990 in food web region 1. In addition, the model values fall within $\pm$ 50 percent of the mean value for all measurement years except 1990 (the mean is represented by the horizontal bars). This includes five of the six sampling events in food web region 1 and the one sampling event in food web region 2. In 1990, the modeled data are slightly higher in concentration then the maximum sampled value.

For striped bass (shown in Figure 3-9), the modeled data nearly always fall within the range of sampled values and are close to the mean sampled values, indicating a satisfactory level of agreement.

Although there is a relatively limited data set for PCBs in sediment, water and fish, the model is able to replicate the measurements fairly well, particularly for the fish data. This indicates that the use of the Farley *et al.* (1999) models with the HUDTOX Upper Hudson load estimate is consistent with the available data and should provide a reasonable basis for estimating future concentrations of PCBs in the Lower Hudson River.

3.1.1.5 Comparison of White Perch PCB Body Burden between the Farley Model (Using Upper River Loads from HUDTOX) and FISHRAND

White perch is the only species that is common to both the Farley *et al.* (1999) bioaccumulation model (as modified by Cooney, 1999) and the FISHRAND model, providing a point of comparison between the models. Similar results for both models would suggest a consistent basis on which to assess exposures and exposure-related risks to humans and the biota. As a basis for comparison, the results of the 70-year forecast for each model are compared for several locations.

White perch body burdens of Tri+ PCBs are plotted against time for each location modeled by FISHRAND in Figure 3-10. It is important to note that the Farley model predicts average fish body burden for the entire food web region 1 while FISHRAND has been applied separately to several locations within the region. In Region 1, the Farley model predicts lower concentrations than the FISHRAND model at RM 152. At RMs 113 and 90 the FISHRAND and Farley models agree fairly well, wherein FISHRAND results are only sometimes higher in concentration than the Farley model. In food web region 2, the Farley model predicts higher PCB concentrations than the FISHRAND model in the early portion of the forecast. Both models show a steady drop off in PCB concentration with time and appear to approach a similar asymptote.

The Farley model estimates for white perch body burdens from each region of the river are plotted against the corresponding FISHRAND estimates in Figure 3-11 for each time step in the forecast. The linear fits to the data are reasonable with regression coefficients ranging from 0.825 to 0.916. The difference in the magnitude of the concentrations are evident in the slopes. At RM 152, the slope is 1.27 where the FISHRAND concentrations are higher. At RM 50, the slope is 0.594 where the FISHRAND concentrations are lower. Overall, the agreement is considered good and indicates that both models provide a consistent basis for estimating future fish body burdens. This also indicates that it is reasonable to apply the FISHRAND outside its original calibration region (*i.e.*, the Upper Hudson River) and that the application of FISHRAND in the Lower Hudson will produce reasonable future estimates of the various fish body burdens. This conclusion is further supported by the comparisons to Lower Hudson data in the next subsection.

3.1.1.6 Comparison Between FISHRAND Output and Sample Data From NYSDEC and USEPA

Fish body burdens modeled using FISHRAND were compared to the NYSDEC, NOAA and USEPA sample data on both a wet weight basis and a lipid-normalized basis. This is shown in Figure 3-12a for the largemouth bass, white perch, brown bullhead and yellow perch at RM 152. Similarly, results for largemouth bass, white perch and yellow perch at RM 113 are shown in Figure 3-12b. These species plus striped bass represent the main human exposure routes. They are also important for the larger ecological receptors. These species also have larger data sets than other species and cover much of the Lower Hudson. In each diagram, the median fish body burden predicted by the FISHRAND model is compared with measured median fish body burden as reported by the various agencies. The error bars about each median represent the 95 percent confidence interval on the median. The error bars were calculated assuming the underlying distribution to be lognormal using the formulation given in Gilbert (1987). (Note that FISHRAND is a mechanistic model which also incorporates probability distributions for the various parameters. The model result is a probability distribution from which the mean, median or other statistical properties can be obtained.)

In general, the agreement between the modeled and sampled data is better on the wet weight basis than on the lipid normalized basis. For the wet weight data, the model results fall close to the median of the sampled data, in some cases mirroring the trend in the sample data. Nonetheless, the data show substantive year-to-year variations which are not reflected in the model output. Additionally, the model appears more accurate at RM 113 than at RM 152, falling within the confidence limits for nearly all years of measurement for the three species shown at RM 113. At both locations the model results reflect the general trend to lower PCB concentrations with time. On average, the model values tend to fall below the mean value for each species, location and year.

The difference between the measured and predicted values can be expressed as a relative percent difference (RPD). The RPD is calculated as follows:

$$\text{RPD} = \frac{(\text{Model Median Estimate} - \text{Median Measurement})}{\text{Median Measurement}}$$

Table 3-4 summarizes the RPDs calculated from the FISHRAND results and the 1987 to 1996 NYSDEC, USEPA and NOAA data. The RPDs are calculated using the wet weight median values from the model and the corresponding measurements. As was evident from the figures, the FISHRAND results tend to fall below the measurement medians, yielding negative RPDs. However, the measurements vary considerably so that both positive and negative deviations are obtained. Averaging by species and river mile, the mean RPD $\pm$ 2 standard errors rarely excludes zero, indicating a lack of statistical significance for the calculated differences. The mean RPD for the period 1986-1997 is –6 percent for all fish. For the potential game fish (largemouth bass, brown bullhead, white perch and yellow perch), the mean RPD for the latter years (1993-1997) throughout the Lower Hudson is –16 percent. Thus, while the model results tend to fall below the data (*i.e.*, model concentrations are less than measured concentrations), the difference tends to be within the uncertainty bounds of the measurements.

Figure 3-12c shows a comparison between model and measured fish body burdens for pumpkinseed. Here again, the model differs from the measurements for individual years but is able to reflect the overall trend. RPDs from these results are also included in Table 3-4. Pumpkinseed represent an intermediate trophic level in the food web and indicate that the model is relatively accurate at this level as well.

In 1993, USEPA in conjunction with NYSDEC and NOAA, collected and measured PCB concentrations in the spottail shiner in the Lower Hudson. These data exist only for the one year and are presented against the model results in Figure 3-12d. For this comparison, FISHRAND results were available for four locations and are summarized in the lower half of Table 3-4. These results again indicate that the model estimates are low with a mean RPD of -27 percent. It is important to note here, however, that the model appears to capture the spatial trend of the measurement values, that is, a gradual trend to lower PCB concentrations in fish with decreasing river mile.

The agreement between the FISHRAND results and the measurements is considered sufficiently good to support the use of FISHRAND in estimating fish body burdens in the Lower Hudson using the model output from the Farley *et al.* (1999) model. Although the agreement is not exact for each location examined with FISHRAND, the overall trends of food web region 1 appear to be captured, just as they were in the original model by Farley *et al.* (1999). On average, the FISHRAND model results tend to underpredict the measurements (by 16 percent in the most recent period), but are probably within measurement error. Additionally, model agreement is better at some locations than others but the differences appear to offset each other.

3.1.2 Model Results

The forecast results for the Farley fate and transport and bioaccumulation models and the FISHRAND model are presented for parameters which are used in ERA Addendum. Relevant examples of the model output are shown. This is appropriate because Section 3.1 serves as an explanation of the use of the models and not a report on the models themselves. Complete descriptions of the models are available in Farley *et al.* (1999) for the Farley model and USEPA (1999b and 2000) for the FISHRAND model. The Federal Dam flux is presented on each figure to show the effect of this parameter.

3.1.2.1 Farley Model Forecast Water Column and Sediment Concentrations

The averaged dissolved phase water column data for food web regions 1 and 2 are presented in Figure 3-13 for Tri+ PCBs. Food web region 1 particulate phase water column data for Tri+ PCBs and whole water data for total PCBs are shown in Figure 3-14. Sediment data from 0-2.5 cm model segments in the middle of the food web regions are plotted in Figure 3-15. Each of these diagrams shows the gradual decline of PCB concentrations in the region and their correspondence to the upstream loads. Additionally, the diagrams show that PCB levels appear to approach an asymptotic value, suggesting a long-term residual level of contamination in the system, presumably resulting from the continued upstream loads and the reworking of the existing sediment inventory.

3.1.2.2 Farley Model Forecast Fish Body Burdens

Modeled fish body burdens are plotted in Figures 3-16 and 3-17 for white perch and striped bass. The flux of Tri+ PCBs over the Federal Dam is also presented in these figures to show the correlation of this input with the fish body burden. Again, similar to the sediments and water, the fish results suggest a long-term residual level of PCBs.

3.1.2.3 FISHRAND Forecast Fish Body Burdens

The fish body burden forecasts for each receptor modeled using FISHRAND are shown in Figures 3-18 through 3-23. Modeled receptors are the largemouth bass, white perch, yellow perch, brown bullhead, pumpkinseed and spottail shiner. In these diagrams the mean PCB concentrations at each RM are shown with the 95% upper confidence level on the mean. These mean values were obtained based on the FISHRAND-predicted body burden distributions. The upper confidence level is calculated from these distributions as well, assuming a lognormal distribution and applying the calculation method given in Gilbert (1987). These confidence limits are based solely on the model output distributions. It is likely that these are underestimates of the true confidence limits given that the model is unable to capture the year-to-year variability evident in the data. Nonetheless, the model is expected to accurately represent the long-term behavior of the mean, as shown by the agreement between the model output and measurement medians presented previously.

3.1.3 Modeling Summary

This section describes the application of the model developed by Farley *et al.* (1999) to create a 70-year forecast for the Lower Hudson. For use in the ERA Addendum and Mid-Hudson HHRA, the Farley model was extensively supplemented by the USEPA models developed for the Upper Hudson, namely HUDTOX and FISHRAND. HUDTOX provides a reasonable basis for estimating future Upper Hudson loads to the lower river while FISHRAND provides estimates of PCB levels in fish species based on Farley *et al.* (1999) model output. Supplementing the Farley model in this manner provided acceptable agreement with the existing calibration data, particularly for fish and sediments. In general, fish body burdens estimated by the models tended to fall below the measurements by perhaps 16 percent. The model results were able to capture the general trend of decreasing PCB concentration with time and distance down river, but not the year-to-year variability. The agreement is considered sufficient for use in the ERA Addendum and Mid-Hudson HHRA.

3.2 Exposure Point Concentrations

Models have been developed to describe the fate, transport, and bioaccumulation potential of PCBs in the Upper Hudson River. The Farley *et al.* (1999) model provides sediment and water PCB concentrations and the FISHRAND model provides benthic invertebrate, water column invertebrate, macrophyte, and fish PCB concentrations (USEPA, 1999b). FISHRAND predicts probability distributions of expected concentrations of PCBs in fish based on mechanistic mass-balance principles and an understanding of the underlying biology.

FISHRAND is a mechanistic, fully time-varying model based on the Gobas (1993) modeling approach. The model relies on solutions of differential equations to describe the uptake of PCBs over time, and incorporates both sediment and water sources to predict the uptake of PCBs based on prey consumption and food web dynamics. The model provides expected fish species concentrations of PCBs in the form of distributions. These distributions can be interpreted as population-level concentrations; that is, at the 95^{th} percentile, 95% of the population is expected to experience the predicted concentration or less.

Concentrations of PCBs in the Lower Hudson River ecosystem were estimated for the period 1993 to 2018 for the four reaches comprising the lower river. These reaches are:

- River Mile (RM) 152 - encompassing RM 153.5 – 123.5;
- RM 113 - encompassing RM 123.5 – 93.5;
- RM 90 - encompassing RM 93.5 – 63.5; and
- RM 50 - encompassing RM 63.5 – 33.5.

3.2.1 Modeled Water Concentrations

The Farley model (Farley *et al.* 1999) was used to predict whole water and dissolved water concentrations of PCBs for four regions of the Lower Hudson River for the period of 1993 to 2018. Table 3-4 provides the predicted average and 95% UCL whole water concentrations on a Tri+ total PCB basis.

Table 3-5 also provides the predicted average and 95% UCL whole water concentrations expressed on a TEQ basis. These values were obtained by multiplying the Tri+ predictions in Table 3-5 by the toxic equivalency weighting factors developed to describe the proportion of the Tri+ total expressed as a TEQ (see USEPA, 1999c for details).

3.2.2 Modeled Sediment Concentrations

The Farley *et al.* (1999) model was also used to predict concentrations of PCBs in sediments for the period 1993 to 2018. Table 3-6 provides the predicted average and 95% UCL sediment concentrations on a Tri+ total PCB basis.

Table 3-7 provides total organic carbon (TOC) normalized predicted average and 95% UCL sediment concentrations. To estimate the TOC-normalized sediment concentrations the predicted dry weight was divided by the percent TOC, which was assumed to be 2.5% for the entire lower river (Farley *et al.*, 1999). TOC-normalized sediment concentrations are used for comparison to guidelines based on organic carbon normalization (*i.e.*, NYSDEC, 1999a and Persaud *et al.*, 1993).

These tables also provide the predicted average and 95% UCL sediment concentrations expressed on a TEQ basis. These values were obtained by multiplying the Tri+ predictions by the toxic equivalency weighting factors developed to describe the proportion of the Tri+ total expressed as a TEQ.

3.2.3 Modeled Benthic Invertebrate Concentrations

Benthic invertebrate concentrations of PCBs for the period 1993 to 2018 were predicted using the biota sediment accumulation factor (BSAF) developed for the baseline ERA (USEPA, 1999c). Table 3-8 provides the predicted average and 95% UCL benthic invertebrate concentrations expressed on a total PCB (Tri+) and a TEQ basis. The TEQ values were obtained by multiplying the predicted benthic invertebrate concentration by the TEF for that receptor species based on the analyses presented in subchapter 3.2 of the ERA (USEPA, 1999c).

3.2.4 Modeled Fish Concentrations

Concentrations of PCBs in spottail shiner, pumpkinseed, yellow perch, white perch, brown bullhead, and largemouth bass for the period 1993 to 2018 were predicted using the FISHRAND model (USEPA, 1999b).

Striped bass PCB concentrations were predicted *via* a ratio to largemouth bass from FISHRAND using the Farley model, as discussed in section 3.1.1.2. The average ratio between measured striped bass and largemouth bass at RM 152 is 2.5 (standard deviation = 1.6) and 0.52 (standard deviation = 0.2) at RM 113. Striped bass concentrations were not calculated for the lower regions because striped bass results for this region were already themselves averaged in the Farley model, and would have to be re-averaged to generate results (*i.e.*, taking the log of the already averaged age classes is not the same as taking the log of the original values and then taking the average). Using ratios to calculate the striped bass concentrations allows the population level risk, rather than the average risk, to be estimated.

Tables 3-9 through 3-15 provide the 25th and 95th percentile values as well as the median of the predicted distribution for the spottail shiner, pumpkinseed, yellow perch, white perch, brown bullhead, largemouth bass, and striped bass, respectively, expressed on a wet weight basis for Tri+ total PCBs.

Forecasts are not provided for the shortnose sturgeon, because a specific bioaccumulation model has not been developed for this species. For this analysis, brown bullhead results serve as an order-of-magnitude surrogate fish species to assess potential risks to shortnose sturgeon.

The observed fish PCB concentrations for all species except pumpkinseed and spottail shiner in both the USEPA Phase 2 and NYSDEC sampling programs are given as standard fillets. Because ecological receptors do not distinguish between standard fillets and whole fish, and TRVs for fish are typically based on whole body wet weight concentrations, the observed wet weight concentrations require an adjustment to reflect the difference between the standard fillet and the whole body. As PCBs are known to partition into lipid, the conversion was accomplished by evaluating whole body versus standard fillet lipid content to obtain a multiplier for those species for which data were available (USEPA, 1997c). For largemouth bass, this conversion factor is 2.5 and for brown bullhead, the conversion factor is 1.5. These values were discussed with NYSDEC and thought to be comparable to values for Hudson River fish (NYSDEC, 1999c). For those fish species for which the ratio of lipid in the whole fish relative to the standard fillet could not be obtained (*i.e.*, white perch and yellow perch), the observed and modeled body burdens expressed on a fillet basis were used and the calculated concentrations are likely to be underpredicted. Note that this is likely to underestimate wet weight concentrations in the whole body but has no effect on lipid-normalized concentrations. No conversion factors were required for the pumpkinseed and spottail shiner because they were modeled on a whole body basis.

3.3 Identification of Exposure Pathways

Potential PCB exposure pathways for aquatic and terrestrial receptors were identified in the baseline ERA (USEPA, 1999c), where the exposure equations can be found. The exposure pathways included in the quantitative exposure calculations in this assessment are:

- Benthic invertebrate exposure pathways (as prey of fish and wildlife receptors);
- Fish exposure pathways;
- Avian exposure pathways; and
- Mammalian exposure pathways.

3.3.1 Benthic Invertebrate Exposure Pathways

Benthic invertebrates accumulate PCBs from water, including sediment porewater and the overlying water, from ingestion of sediment particles, or from ingestion of particulate matter (phytoplankton and detrital material) in the overlying water at the sediment/water interface.

Predicted benthic invertebrate concentrations for 1993 to 2018 were estimated by multiplying the predicted sediment concentrations (from the Farley *et al.*, 1999 model) by a biota-sediment concentration factor, as described in the baseline ERA (USEPA, 1999c). These benthic invertebrate concentrations were used as prey concentrations for fish and wildlife receptors.

3.3.2 Fish Exposure Pathways

Fish are directly exposed to PCBs in water and sediments as well as indirectly through the food chain. Fish exposure to PCBs is described by a wet weight PCB tissue concentration. Concentrations of PCBs in spottail shiner, pumpkinseed, yellow perch, white perch, brown bullhead, and largemouth bass were predicted using the FISHRAND model, while striped bass PCB concentrations were predicted *via* a ratio to largemouth bass from FISHRAND using the Farley *et al.*, 1999 model as updated (Cooney, 1999).

3.3.3 Avian Exposure Pathways, Parameters, Daily Doses, and Egg Concentrations

Avian receptors along the Hudson River are exposed to PCBs primarily through ingestion of contaminated prey (*i.e.*, diet), surface water ingestion, and incidental ingestion of sediments (see USEPA, 1999c section 2.3.4). Intake is calculated as an average daily dosage (ADD) value, expressed as mg PCB/kg/day. The ADD from each of these three calculated exposure pathways is summed to develop the total ADD of PCBs from riverine sources. Exposure parameters for the tree swallow, mallard, belted kingfisher, great blue heron, and bald eagle are provided in Tables 3-16 to 3-20. The equations used to calculate intakes for each of the average daily doses are provided in Chapter 3 of the baseline ERA (USEPA, 1999c). All concentrations of PCBs in fish prey consumed by avian receptors were calculated using the FISHRAND model (USEPA, 2000).

3.3.3.1 Summary of $ADD_{Expected}$, $ADD_{95\%UCL}$, and Egg Concentrations for Avian Receptors

Tree Swallow

Tables 3-25 and 3-26 present the expected ADD and 95% UCL daily dose on a total PCB basis for the female tree swallow from water and dietary sources for the modeling period 1993 – 2018. Doses are based on the results from the Farley *et al.* (1999) model for water and FISHRAND (USEPA, 2000) for benthic invertebrates. Tables 3-35 and 3-36 present the expected ADD and 95% UCL daily dose on a TEQ PCB basis for the modeling period 1993 – 2018 using the same models. All tables also show the predicted egg concentrations using biomagnification factors based on the USFWS tree swallow data (2 for total PCBs and 7 on a TEQ basis).

Mallard Duck

Tables 3-27 and 3-28 present the expected ADD and 95% UCL daily dose on a total PCB basis for the female mallard from water, sediment, and dietary sources for the modeling period 1993 - 2018. Doses are based on the results from the Farley *et al.* (1999) model for water and sediment and FISHRAND (USEPA, 2000) for benthic invertebrates and macrophytes. Tables 3-37 and 3-38 present the expected ADD and 95% UCL daily dose on a TEQ PCB basis for the modeling period 1993 – 2018 using the same models. All tables show the predicted egg concentrations using biomagnification factors based on the USFWS mallard and wood duck data (3 for total PCBs and 28 on a TEQ basis).

Belted Kingfisher

Tables 3-29 and 3-30 present the expected ADD and 95% UCL daily dose on a total PCB basis for the female belted kingfisher from water, sediment, and dietary sources for the modeling period 1993 – 2018. Doses are based on the results from the Farley *et al.* (1999) model for water and sediment and FISHRAND (USEPA, 2000) for benthic invertebrates and forage fish. Tables 3-39 and 3-40 present the expected ADD and 95% UCL daily dose on a TEQ PCB basis for the modeling period 1993 – 2018 using the same models. All tables also show the predicted egg concentrations using biomagnification factors obtained from Giesy *et al.* (1995) for piscivorous birds (28 for total PCBs and 19 on a TEQ basis).

Great Blue Heron

Tables 3-31 and 3-32 present the expected ADD and 95% UCL daily dose on a total PCB basis for the female great blue heron from water, sediment, and dietary sources for the modeling period 1993 – 2018. Doses are based on the results from the Farley *et al.* (1999) model for water and sediment and FISHRAND for benthic invertebrates and forage fish. Tables 3-41 and 3-42 present the expected ADD and 95% UCL daily dose on a TEQ PCB basis for the modeling period 1993 – 2018 using the same models. All tables also show the predicted egg concentrations using biomagnification factors obtained from Giesy *et al.* (1995) for piscivorous birds (28 for total PCBs and 19 on a TEQ basis).

Bald Eagle

Tables 3-33 and 3-34 present the expected ADD and 95% UCL daily dose on a total PCB basis for the female bald eagle from water, sediment, and dietary sources for the modeling period 1993 – 2018. Doses are based on the results from the Farley *et al.* (1999) model for water and sediment and FISHRAND (USEPA, 2000) for piscivorous fish. Tables 3-43 and 3-44 present the expected ADD and 95% UCL daily dose on a TEQ PCB basis for the modeling period 1993 – 2018 using the same models. All tables also show the predicted egg concentrations using biomagnification factors obtained from Giesy *et al.* (1995) for piscivorous birds (28 for total PCBs and 19 on a TEQ basis).

3.3.4 Mammalian Exposure Pathways, Parameters, and Daily Doses

Terrestrial mammals living along the Hudson River are exposed to PCBs primarily *via* ingestion of contaminated prey (*i.e.*, diet), surface water ingestion, and incidental ingestion of sediments (see baseline ERA section 2.3.4). Intake is calculated as an ADD value expressed as mg PCB/kg/day. The ADDs from each of the three calculated exposure pathways are summed to develop the total ADD of PCBs from riverine sources. The equations and parameters used to calculate intakes for each of the ADDs are provided in Chapter 3 of the baseline ERA (USEPA, 1999c). Exposure parameters for the little brown bat, raccoon, mink, and river otter are provided in Tables 3-21 to 3-24. The equations used to calculate intakes for each of the ADD are provided in the baseline ERA (USEPA, 1999c). All concentrations of PCBs in fish prey consumed by mammalian receptors were calculated using the FISHRAND model (USEPA, 2000).

3.3.4.1 Summary of $ADD_{Expected}$ and $ADD_{95\%UCL}$ for Mammalian Receptors

Little Brown Bat

Tables 3-45 and 3-46 present the expected ADD and 95% UCL daily dose on a total PCB basis for the female little brown bat from water and dietary sources for the modeling period 1993 – 2018. Doses are based on the results from the Farley *et al.* (1999) model for water and FISHRAND (USEPA, 2000) for benthic invertebrates. Tables 3-53 and 3-54 present the expected ADD and 95% UCL daily dose on a TEQ PCB basis for the modeling period 1993 – 2018 using the same models.

Raccoon

Tables 3-47 and 3-48 present the expected ADD and 95% UCL daily dose on a total PCB basis for the female raccoon from water, sediment, and dietary sources for the modeling period 1993 – 2018. Doses are based on the results from the Farley *et al.* (1999) model for water and sediment and FISHRAND (USEPA, 2000) for benthic invertebrates and forage fish. Tables 3-55 and 3-56 present the expected ADD and 95% UCL daily dose on a TEQ PCB basis for the modeling period 1993 – 2018 using the same models.

Mink

Tables 3-49 and 3-50 present the expected ADD and 95% UCL daily dose on a total PCB basis for the female mink from water, sediment, and dietary sources for the modeling period 1993 – 2018. Doses are based on the results from the Farley *et al.* (1999) model for water and sediment and FISHRAND (USEPA, 2000) for benthic invertebrates and forage fish. Tables 3-57 and 3-58 present the expected ADD and 95% UCL daily dose on a TEQ PCB basis for the modeling period 1993 – 2018 using the same models.

River Otter

Tables 3-51 and 3-52 present the expected ADD and 95% UCL daily dose on a total PCB basis for the female river otter from water, sediment, and dietary sources for the modeling period 1993 – 2018. Doses are based on the results from the Farley *et al.* (1999) model for water and sediment and FISHRAND (USEPA, 2000) for forage fish and piscivorous fish. Tables 3-59 and 3-60 present the expected ADD and 95% UCL daily dose on a TEQ PCB basis for the modeling period 1993 – 2018 using the same models.

4.0 EFFECTS ASSESSMENT

This chapter provides a general overview of the toxicology of PCBs and provides a brief overview of the methods used to characterize particular toxicological effects of PCBs on aquatic and terrestrial organisms. Full details are provided in Appendix B. Toxicity reference values (TRVs) selected to estimate the potential risk to receptor species resulting from exposure to PCBs are presented following the background on PCB toxicology. TRVs are levels of exposure associated with either Lowest Observed Adverse Effects Levels (LOAELs) or No Observed Adverse Effects Levels (NOAELs). They provide a basis for judging the potential effects of measured or predicted exposures that are above or below these levels.

Use of both LOAELs and NOAELS provides perspective on the potential for risk as a result of exposure to PCBs originating from the site. LOAELs are values at which effects have been observed (in either laboratory or field studies), while the NOAEL represents the lowest dose or body burden at which an effect was not observed. Exceedance of a LOAEL indicates a greater potential for risk.

4.1 Selection of Measures of Effects

Many studies examined the effects of PCBs on aquatic and terrestrial organisms, and results of these studies are compiled and summarized in several reports and reviews (*e.g.*, Eisler and Belisle, 1996; Niimi, 1996; Hoffman *et al.*, 1998; ATSDR, 1996; Eisler, 1986; and NOAA, 1999b). For the present assessment, studies on the toxic effects of PCBs were identified by searching the National Library of Medicine (NLM) MEDLINE and TOXLINE databases. Other studies were identified from the reference section of papers that were identified by electronic search. Papers were reviewed to determine whether the study was relevant to the topic.

Many different approaches and methodologies are used in these studies, some of which are more relevant than others to the selection of TRVs for the ERA (USEPA, 1999c) and this ERA Addendum. TRVs are levels of exposure associated with either LOAELs or NOAELs. They provide a basis for judging the potential effects of measured or predicted exposures that are above or below these levels. Some studies express exposures as concentrations or doses of total PCBs, whereas other studies examine effects associated with individual congeners (*e.g.*, PCB 126) or as total dioxin equivalents (TEQs). This risk assessment develops separate TRVs for total PCBs and TEQs. This chapter briefly describes the rationale that was used to select TRVs for various ecological receptors of concern.

Some studies examine toxicity endpoints (such as lethality, growth, and reproduction) that are thought to have greater potential for adverse effects on populations of organisms than other studies. Other studies examine toxicity endpoints such as behavior, disease, cell structure, immunological responses, or biochemical changes that affect individual organisms, but may not result in adverse effects at the population level. For example, toxic effects such as enzyme induction may or may not result in adverse effects to individual animals or populations. For the ERA and ERA Addendum, TRVs were selected from studies that examine the effects of PCBs on lethality, growth or reproduction. Studies that examined the effects of PCBs on other sublethal endpoints are not used to select TRVs, although effects may occur at these concentrations. Lethality, growth, and reproductive-based endpoints typically present the greatest risk to the viability of the individual organism and therefore survival of the population. Thus, these are considered to be the measurement endpoints of greatest concern relative to the stated assessment endpoints.

When exposures are expected to be long-term, data from studies of chronic exposure are preferable to data from medium-term (subchronic), short-term (acute), or single-exposure studies (USEPA, 1997b). Because of the persistence of PCBs, exposure of ecological receptors to PCBs from the Hudson River is expected to be long-term, and therefore studies of chronic exposure are preferentially used to select the TRVs. Long-term studies are also preferred since reproductive effects of PCBs are typically studied and evaluated following long-term exposure.

Dose-response studies compare the response of organisms exposed to a range of doses to that of a control group. Ideally, doses that are below and above the threshold level that causes adverse effects are examined. Toxicity endpoints determined in dose-response and other studies include:

- NOAEL (No-Observed-Adverse-Effect-Level) is the highest exposure level shown to be without adverse effect in organisms exposed to a range of doses. NOAELs may be expressed as dietary doses (*e.g.*, mg PCBs consumed/kg body weight/day), as concentrations in external media (*e.g.*, mg PCBs/kg food), or as concentrations in tissue of the affected organisms (*e.g.*, mg chemical/kg egg).

- LOAEL (Lowest-Observed-Adverse-Effect-Level) is the lowest exposure level shown to produce adverse effect in organisms exposed to a range of doses. LOAELs may also be expressed as dietary doses (*e.g.*, mg PCBs consumed/kg body weight/day), as concentrations in external media (*e.g.*, mg PCBs/kg food), or as concentrations in tissue of the effected organisms (*e.g.*, mg chemical/kg egg). The LOAEL represents a concentration at which the particular effect has been observed and the occurrence of the effect is statistically significantly different from the control organisms.

- LD_{50} is the Lethal Dose that results in death of 50% of the exposed organisms. The LD_{50} is expressed in units of dose (*e.g.*, mg PCBs administered/kg body weight of test organism/day).

- LC_{50} is the Lethal Concentration in some external media (*e.g.* food, water, or sediment) that results in death of 50% of the exposed organisms. The LC_{50} is expressed in units of concentration (*e.g.*, mg PCBs/kg wet weight food).

- ED_{50} is the Effective Dose that results in a sublethal effect in 50% of the exposed organisms (mg/kg/day).

- EC_{50} is the Effective Concentration in some external media that results in a sublethal effect in 50% of the exposed organisms (mg/kg).

- CBR or Critical Body Residue is the concentration in the organism (*e.g.*, whole body, liver, or egg) that is associated with an adverse effect (mg PCBs/kg wet weight tissue).

- EL-effect is the effect level that results in an adverse effect in organisms exposed to a single dose, rather than a range of doses. Expressed in units of dose (mg/kg/day) or concentration (mg/kg).

- EL-no effect is the effect level that does not result in an adverse effect in organisms exposed to a single dose, rather than a range of doses. Expressed in units of dose (mg/kg/day) or concentration (mg/kg).

Most USEPA risk assessments typically estimate risk by comparing the exposure of receptors of concern to TRVs that are based on NOAELs. TRVs for the ERA (USEPA, 1999c) and ERA Addendum were developed on the basis of both NOAELs and LOAELs to provide perspective on the range of potential effects relative to measured or modeled PCB exposures. Because the LOAEL represents a concentration at which effects were definitely observed, this is a stronger indicator of the potential for risk. However, risk may occur at any concentration between the NOAEL and the LOAEL, so exceedance of the NOAEL also indicates the potential for risk.

Differences in the feeding behavior of aquatic and terrestrial organisms determine the type of toxicity endpoints that are most easily measured and most useful in assessing risk. For example, the dose consumed in food is more easily measured for terrestrial animals than for aquatic organisms because uneaten food can be difficult to collect and quantify in an aqueous environment. Therefore, for aquatic organisms, toxicity endpoints are more often expressed as concentrations in external media (*e.g.*, water) or as accumulated concentrations in the tissue of the exposed organism (also called a "body burden"). In some studies, doses are administered *via* gavage, intraperitoneal injection into an adult, or injection into a fish or bird egg. If appropriate studies are available, TRVs were selected on the basis of the most likely route of exposure, as described below:

- TRVs for fish are expressed as critical body residues (CBR) (*e.g.*, mg/kg whole body weight and mg/kg lipid in eggs).

- TRVs for terrestrial receptors (*e.g.*, birds and mammals) are expressed as daily dietary doses (*e.g.*, mg/kg whole body weight/day).

- TRVs for birds are also expressed as concentrations in eggs (*e.g.* mg/kg wet weight egg).

4.1.1 Methodology Used to Derive TRVs

The literature on toxic effects of PCBs to animals includes studies conducted solely in the laboratory, as well as studies including a field component. Each type of study has advantages and disadvantages for the purpose of deriving TRVs for a risk assessment. For example, a controlled laboratory study can be designed to test the effect of a single formulation or congener (*e.g.* Aroclor 1254 or PCB 126) on the test species in the absence of the effects of other co-occurring contaminants. This is an advantage because greater confidence can be placed in the conclusion that observed effects are related to exposure to the test compound. However, laboratory studies are often conducted on species that are easily maintained in the laboratory, rather than on wildlife species.

Therefore, laboratory studies may have the disadvantage of being conducted on species that are less closely related to a particular receptor of concern. Field studies have the advantage that organisms are exposed to a more realistic mixture of PCB congeners (with differences in toxic potencies), than, for example, laboratory tests that expose organisms to a commercial mixture, such as Aroclor 1254. Field studies have the disadvantage that organisms are usually exposed to other contaminants and observed effects may not be attributable solely to exposure to PCBs. Field studies can be used most successfully, however, to establish concentrations of PCBs or TEQs at which adverse effects are not observed (*e.g.*, a NOAEL). Because of the potential contribution of other contaminants (*e.g.* metals, pesticides, etc.) to observed effects in field studies, the ERA and ERA Addendum use field studies to establish NOAEL TRVs, but not LOAEL TRVs.

If appropriate field studies are available for species in the same taxonomic family as the receptor of concern, those field studies were used to derive NOAEL TRVs for receptors of concern. Appropriateness of a field study was based on the following considerations:

- whether the study examines sensitive endpoints, such as reproductive effects, in a species that is closely related (*e.g.* within the same taxonomic family) to the receptor of concern;

- whether measured exposure concentrations of PCBs or dioxin-like compounds are reported for dietary doses, whole organisms, or eggs;

- whether the study establishes a dose-response relationship between exposure concentrations of PCBs or dioxin-like contaminants and observed effects; and

- whether contributions of co-occurring contaminants are reported and considered to be negligible in comparison to contributions of PCBs or dioxin-like compounds.

If appropriate field studies are not available for a test species in the same taxonomic family as the receptor species of concern, laboratory studies were used to establish TRVs for the receptor species. The general methodology described in the following paragraphs was used to derive TRVs for receptors of concern from appropriate studies.

When appropriate chronic-exposure toxicity studies on the effects of PCBs on lethality, growth, or reproduction are not available for a species of concern, extrapolations from other studies were made in order to estimate appropriate TRVs. For example, if toxicity data are unavailable for a particular species of bird, toxicity data for a related species of bird were used if appropriate information was available. Several methodologies have been developed for deriving TRVs for wildlife species (*e.g.*, Sample *et al.*, 1996; California EPA, 1996; USEPA, 1996; and Menzie-Cura & Associates, 1997). The general methodology used to develop LOAEL and NOAEL TRVs is described below:

- If an appropriate NOAEL is unavailable for a phylogenetically similar species (*e.g.* within the same taxonomic family), NOAEL values for other species (as closely related as possible) were adjusted by dividing by an uncertainty factor of 10 to account for extrapolations between species. The lowest appropriate NOAEL was used whenever several studies are available. However, if the surrogate test species is known to be the most sensitive of all species tested in that taxonomic group (*e.g.* fish, birds, mammals), then an interspecies uncertainty factor was not applied

- In the absence of an appropriate NOAEL, if a LOAEL is available for a phylogenetically similar species, these may be divided by an uncertainty factor of 10 to account for a LOAEL to NOAEL conversion. The LOAEL to NOAEL conversion is similar to USEPA's derivation of human health RfD (Reference Dose) values, where LOAEL studies are adjusted by a factor of 10 to estimate NOAEL values.

- When calculating chronic dietary dose-based TRVs (*e.g.* mg/kg/day) from data for sub-chronic tests, the sub-chronic LOAEL or NOAEL values were divided by an additional uncertainty factor of 10 to estimate chronic TRVs. The use of an uncertainty factor of 10 is consistent with the methodology used to derive human health RfDs. These factors are applied to account for uncertainty in using an external dose (*e.g.*, mg/kg/day in diet) as a surrogate for the dose at the site of toxic action (*e.g.* mg/kg in tissue). Because organisms may attain a toxic dose at the site of toxic action (*e.g.* in tissues or organs) *via* a large dose administered over a short period, or *via* a smaller dose administered over a longer period, uncertainty factors are used to estimate the smallest dose that, if administered chronically, would result in a toxic dose at the site of action. USEPA has not established a definitive line between sub-chronic and chronic exposures for ecological receptors. The ERA and ERA Addendum follow recently developed guidance (Sample *et al.*, 1996) which considers 10 weeks to be the minimum time for chronic exposure of birds and 1 year for chronic exposure of mammals.

- For studies that actually measure the internal toxic dose (*e.g.*, mg PCBs/kg tissue), no sub-chronic to chronic uncertainty factor was applied. This is appropriate because effects are being compared to measured internal doses, rather than to external dietary doses that are used as surrogates for the internal dose.

- In cases where NOAELs are available as a dietary concentration (*e.g.*, mg contaminant per kg food), a daily dose for birds or mammals was calculated on the basis of standard estimates of food intake rates and body weights (*e.g.*, USEPA, 1993b).

Professional judgment is used to determine relevant endpoints for selecting TRVs. For example, hatching time in fish is considered less relevant than hatchability, which directly affects the viability of offspring. The implication of hatching time on the viability of the population is less clear than an effect such as hatchability. Specific endpoints relative to TRVs are provided in Appendix B.

The sensitivity of the risk estimates to the use of uncertainty factors and the selected TRVs will be examined in the uncertainty chapter (Chapter 6.0).

4.1.2 Selection of TRVs

TRVs selected for Hudson River receptors are provided in Tables 4-1 to 4-3 for fish, birds, and mammals, respectively. These tables provide both Total PCB (Tri+) TRVs and TEQ-based TRVs (discussed below). A complete description of the selection process for each receptor can be found in Appendix B.

As described in the baseline ERA (USEPA, 1999c), the Toxic Equivalency (TEQ)/Toxic Equivalency Factors (TEF) methodology (TEQ/TEF), quantifies the toxicities of PCB congeners relative to the toxicity of the potent dioxin 2,3,7,8-TCDD (see van den Berg *et al.*, 1998 for review). It is currently accepted that the carcinogenic potency of dioxin is affected by its ability to bind AhR and dioxin is considered to be the most potent known AhR ligand. It is also generally accepted that the dioxin-like toxicities of PCB congeners are directly correlated to their ability to bind the AhR. Thus, the TEQ/TEF methodology provides a toxicity measurement for all AhR-binding compounds based on their relative toxicity to dioxin. Since 2,3,7,8-TCDD has the greatest affinity for the AhR, it is assigned a TCDD-Toxicity Equivalent Factor of 1.0. PCB congeners are then assigned a TCDD-TEF relative to 2,3,7,8-TCDD, based on experimental evidence. For example, if the relative toxicity of a particular congener is one-thousandth that of TCDD, it would have a TEF of 0.001. The potency of a PCB congener is estimated by multiplying the tissue concentration of the congener in question by the TEF for that congener to yield the toxic equivalent (TEQ) of dioxin. A TEQ for the total PCB concentration can be determined from the sum of the calculated TEQs for each AhR-binding congener. The World Health Organization (WHO) has derived TEFs for a number of PCB congeners (van den Berg *et al.*, 1998). These values, which are used in this assessment, are presented in Table 4-4.

Chapter 5

5.0 RISK CHARACTERIZATION

Risk characterization is made up of two steps, risk estimation and risk description (USEPA, 1992a and 1997b). Risk estimation integrates stressor-response profiles (Chapter 4) with exposure profiles (Chapter 3) to provide an estimate of risk (Chapter 5) and related uncertainties (Chapter 6). The assessment endpoints and their associated measurement endpoints, selected during problem formulation (Chapter 2), are evaluated in this section.

In the toxicity quotient (TQ) approach, potential risks to ecological receptors are assessed by comparing measured or modeled concentrations (Chapter 3) to toxicity benchmarks developed in (Chapter 4). Future PCB concentrations are predicted on total PCBs (Tri+) and TEQ bases.

The TQ is the direct numerical comparison of a measured or modeled exposure concentration or dose to a benchmark dose or concentration. It is calculated as:

$$\text{Toxicity Quotient} = \frac{\text{Modeled Dose or Concentration}}{\text{Benchmark Dose or Concentration}}$$

TQs equal to or exceeding one are typically considered to indicate potential risk to ecological receptors. The TQ method provides insight into the potential for general effects upon individual animals in the local population resulting from exposure to PCBs. If effects are judged not to occur at the average individual level, they are probably insignificant at the population level. However, if risks are present at the individual level they may or may not be important at the population level.

The risk characterization in the Hudson River is based on the following assessment endpoints:

- Benthic community structure as a food source for local fish and wildlife (Section 5.1)

- Health and maintenance of local fish populations (Section 5.2) by evaluating survival, growth, and reproduction of:
  - local forage fish populations;
  - local omnivorous fish populations; and
  - local piscivorous/semi-piscivorous fish populations.

- Protection (*i.e.*, survival, growth, and reproduction) of local wildlife including:
  - insectivorous birds (Section 5.3);
  - waterfowl (Section 5.4);
  - semi-piscivorous/piscivorous birds (Section 5.5);
  - insectivorous mammals (Section 5.6);
  - omnivorous mammals (Section 5.7); and
  - semi-piscivorous/piscivorous mammals (Section 5.8)

- Protection of threatened and endangered species (Section 5.9).

- Protection of significant habitats (Section 5.10).

5.1 Evaluation of Assessment Endpoint: Benthic Community Structure as a Food Source for Local Fish and Wildlife

5.1.1 Do Modeled PCB Sediment Concentrations Exceed Appropriate Criteria and/or Guidelines for the Protection of Aquatic Life and Wildlife?

5.1.1.1 Measurement Endpoint: Comparisons of Modeled Sediment Concentrations to Guidelines For the Protection of Aquatic Life and Wildlife

Table 5-1 presents the ratios of forecast sediment concentrations to various sediment guidelines. Comparisons are made on total PCB (Tri+) sediment concentrations (*i.e.*, NOAA, 1999a; Persaud *et al.*, 1993; and Washington State, 1997) and TOC-normalized sediment concentrations (*i.e.*, NYSDEC, 1999a and Persaud *et al.*, 1993). A summary of sediment concentrations is provided in Table 3-2 and TOC-normalized sediment concentrations are shown in Table 3-3.

The NOAA (1999a) consensus-based sediment effect concentrations (SECs) for PCBs were developed to support an assessment to sediment-dwelling organisms living in the Hudson River Basin. They refer to all of the PCBs found in the Hudson River, plus the degradation products and metabolites of these chemicals. The Hudson River SECs provide a threshold effect concentration (TEC) of 0.04 mg/kg, a mid-range effect concentration (MEC) of 0.4 mg/kg, and an extreme effect concentration (EEC) of 1.7 mg/kg. The TEC is intended to identify the concentration of total PCBs below which adverse population-level effects (*e.g.*, mortality, decreased growth, reproductive failure) on sediment-dwelling organisms are unlikely to be observed (NOAA, 1999a). The MEC represents the concentration of total PCBs above which adverse effects on sediment-dwelling organisms are expected to be frequently observed. Adverse effects are expected to be usually or always observed at PCB concentrations exceeding the EEC.

Forecast sediment concentrations based on the Farley *et al.* (1999) model exceed the NOAA TEC at all four locations for both average and 95% UCL concentrations throughout the modeling period (Table 5-1). MEC consensus values are exceeded using 95% UCL concentrations at RMs 152, 113, and 90 throughout the modeling period and at RM 50 until 2006. The average forecast concentration at RM 152 exceeds the MEC throughout the modeling period and the average concentrations lower down river exceed the MEC for portions of the modeling period. None of the forecast concentrations exceed the EEC at any of the locations.

The NYSDEC has developed screening criteria concentrations that can be used to identify areas of sediment contamination and evaluate the potential risk that the contaminated sediment may pose to the environment (NYSDEC, 1999a). Criteria developed for the protection of aquatic life from chronic toxicity and protection of wildlife from toxic effects of bioaccumulation are examined in this addendum. Forecast sediment concentrations exceed the NYSDEC benthic aquatic life chronic toxicity criterion at RMs 152, 113, and 90 for the duration of the modeling period based on the 95% UCL. The benthic aquatic life criterion was exceeded until 2011 at RM 90 and until 1997 at RM 50 (Table 5-1). The average total PCB concentration exceeds the criterion for various portions of the modeling period at RMs 152, 113, and 90. The freshwater criterion value of 19.3 mg/kg OC was used, which based on the 2.5% OC assumption used in this assessment provides a dry weight value of 0.48 mg/kg.

Forecast sediment concentrations exceed the NYSDEC wildlife bioaccumulation criterion at all four locations for the duration of the modeling period using both average and 95th UCL results (Table 5-1). The NYSDEC wildlife criterion is 1.4 mg/kg OC, which based on the 2.5% OC assumption used in this assessment provides a dry weight value of 0.035 mg/kg.

The Ontario sediment quality guidelines for the protection and management of aquatic sediment quality (Persaud *et al.*, 1993) were developed to protect the aquatic environment by setting safe levels for metals, nutrients, and organic compounds. The no effect level (NEL) is the level at PCBs in the sediment that do not affect fish or the sediment-dwelling organism. The lowest effect level (LEL) indicates a level of contamination that has no effect on the majority of sediment dwelling organisms. At the severe effect level (SEL) sediments are likely to affect the health of sediment-dwelling organisms. Forecast sediment concentrations exceeded the total PCB NEL of 0.01 mg/kg at all locations for both the average and 95% UCL concentration for the duration of the sampling period (1993-2018) by up to two orders of magnitude (Table 5-1). The total PCB LEL of 0.07 mg/kg was also exceeded at all locations for both the average and 95% UCL concentration for the duration of the sampling period. The total PCB SEL of 530 mg/kg OC (equal to a dry weight value of 1.3 mg/kg using 2.5% OC) was not exceeded at any location for the duration of the modeling period.

Washington State has also derived chemical criteria to predict possible biological effects in sediments (Washington State, 1997). Bioassays for PCBs were conducted using both Microtox® (endpoint = luminescence reduction) and *Hyalella azteca* (endpoint = mortality). The Probable Apparent Effects Thresholds (PAET) for Microtox® was 0.021 mg/kg (total PCBS), while the PAET of *Hyalella azteca* was 0.45 mg/kg. The Microtox® PAET was exceeded at all locations for the duration of the modeling period (1993-2018) using both average and 95% UCL concentrations (Table 5-1). The PAET of *Hyalella azteca* was exceeded by predicted 95% UCL PCB concentrations at RMs 152 and 113 for the duration of the modeling period and at RMs 90 and 50 for portions of the modeling period. Using average PCB concentrations the *Hyalella azteca* PAET was exceeded for a portion of the modeling period at all stations.

Many of the ratios of modeled sediment concentrations to appropriate guidelines exceed 10 or occasionally even 100. Forecast total PCB concentrations are Tri+ values, and do not include mono or dichlorinated congeners that usually contribute a portion of the total PCB load. Thus, even in the unlikely event that forecast sediment concentrations were to decrease by an order of magnitude or more, comparisons to sediment guidelines would still show exceedances.

5.1.2 Do Modeled PCB Water Concentrations Exceed Appropriate Criteria and/or Guidelines for the Protection of Aquatic Life and Wildlife?

5.1.2.1 Measurement Endpoint: Comparison of Modeled Water Column Concentrations of PCBs to Criteria

Table 5-2 presents the results of the comparison between modeled whole water PCB concentrations and appropriate criteria and guidelines. All forecast water concentrations (*i.e.*, average and 95% UCL) exceed the NYSDEC wildlife bioaccumulation criterion of 0.001 μg/L and the USEPA wildlife criterion of 1.2 x 10^{-4} μg/L at all four locations throughout the modeling period. The whole water concentrations also exceed the USEPA/NYSDEC benthic aquatic life chronic toxicity criterion of 0.014 μg/L for a portion of the modeling period for both average and 95% UCL at all modeling locations. These comparisons are likely to underestimate the true risk, as concentrations are expressed as the sum of the Tri+ and higher congeners, while the criteria are based on total PCBs (the sum of all congeners).

5.2 Evaluation of Assessment Endpoint: Protection and Maintenance (*i.e.*, Survival, Growth, and Reproduction) of Local Fish Populations

5.2.1 Do Modeled Total PCB and TEQ-Based PCB Body Burdens in Local Fish Species Exceed Benchmarks for Adverse Effects on Forage Fish Reproduction?

5.2.1.1 Measurement Endpoint: Comparison of Modeled Total PCB Fish Body Burdens to Toxicity Reference Values for Forage Fish

Table 5-3 presents the results of the comparison between forecast PCB body burdens in pumpkinseed and spottail shiner to selected toxicity reference values on a total PCB basis (expressed as Tri+) under future conditions (1993 - 2018). The total PCB (Tri+) body burden in pumpkinseed exceeds a TQ of one using a field-based NOAEL at all four modeling locations (*i.e.*, RMs 152, 113, 90, and 50) for the 25th percentile, median, and 95th percentile. On a 95th percentile basis, the pumpkinseed exceeds one at RM 152 until the end of the modeling period (2018), at RM 133 until 2016, at RM 90 until 2007, and at RM 50 until 2005. This is interpreted to mean that 95% of individual pumpkinseed fish will experience the shown TQ or less for that year.

The spottail shiner did not exceed a TQ of one at any time or location using the laboratory-derived NOAEL and LOAEL (Tables 5-4 and 5-5). The TRV derived for the spottail shiner differ from the TRV derived for the pumpkinseed by more than an order of magnitude (0.5 mg/kg on a NOAEL basis for the pumpkinseed versus 15 mg/kg on a NOAEL basis for the spottail shiner). Consequently, spottail shiner TQs are much lower than pumpkinseed.

5.2.1.2 Measurement Endpoint: Comparison of Modeled PCB TEQs Fish Body Burdens to Toxicity Reference Values for Forage Fish

Tables 5-6 and 5-7 present the results of the comparison between forecast percentiles of pumpkinseed to laboratory-derived NOAEL and LOAEL on a TEQ basis under future conditions. The TRVs for TEQs in fish are mostly based on egg injection studies; however, Hudson River data are for concentrations in adult fish. These two numbers were not considered to be directly comparable since lipid concentrations in eggs and adults may differ substantially. The lipid-normalized egg concentration TRV (*e.g.*, ng TEQs/kg lipid) compared to the lipid-normalized concentration in adult fish (*e.g.*, ng TEQs/kg lipid) was considered to provide the most appropriate comparison.

On a NOAEL basis, the TQs exceed one on a 95th percentile basis at RM 152 until approximately 1999, at RM 113 until 1998, at RM 90 until 1995, and at RM 50 until 1994. On a LOAEL basis, all TQs fell below one.

Tables 5-8 and 5-9 presents the results for the spottail shiner. TQs for spottail shiners do not exceed one at any time or location during the modeling period on either a LOAEL or NOAEL basis.

5.2.1.3 Measurement Endpoint: Comparison of Modeled Total PCB Fish Body Burdens to Toxicity Reference Values for Brown Bullhead

Tables 5-10 and 5-11 present the results of the comparison between predicted percentiles of brown bullhead concentrations a total PCB basis to laboratory-derived NOAEL and LOAEL under future conditions (1993-2018). TQs for the brown bullhead exceed one at all locations during the entire modeling period on NOAEL basis. Using the laboratory-derived LOAEL, the 95th percentile concentration exceeds one at RMs 152 and 133 throughout the modeling period, at RM 90 until 2017, and at RM 50 until 2007. Because the FISHRAND model predicts standard fillet concentrations in fish, the wet weight model results were adjusted by a factor of 1.5 for the brown bullhead, as wildlife feeding on fish consumes them whole. Even without this adjustment, most of ratios would exceed one on a NOAEL basis.

5.2.1.4 Measurement Endpoint: Comparison of Modeled TEQ Basis Fish Body Burdens to Toxicity Reference Values for Brown Bullhead

Tables 5-12 and 5-13 present the results of the comparison between forecast percentiles of brown bullhead concentrations on a TEQ basis to a laboratory-derived NOAEL and LOAEL for TEQs under future conditions. TQs for the brown bullhead do not exceed one at any time or location during the modeling period on either a LOAEL or NOAEL basis.

5.2.1.5 Measurement Endpoint: Comparison of Modeled Total PCB Fish Body Burdens to Toxicity Reference Values for White and Yellow Perch

Table 5-14 presents the results of the comparison between forecast percentiles of white perch a total PCB basis to a field-based NOAEL for the period 1993 - 2018. The white perch exceeds a TQ of one at RM 152 in 1993. The remainder of the ratios fall below one at all locations.

The yellow perch exceeded a TQ of one at all locations during the entire modeling period using the laboratory-derived NOAEL (Table 5-15). All concentrations (*i.e.*, 25th , median, and 95th) were exceeded at all locations with the exception of the 25th percentile at RM 50 for 2016-2108. A TQ of one was not exceeded at any location using the laboratory-derived LOAEL (Table 5-16). The laboratory-based NOAEL TRV derived for the yellow perch is more than an order of magnitude lower than the field-based NOAEL TRV derived for the white perch (0.16 mg/kg on a NOAEL basis for yellow perch versus 3.1 mg/kg on a NOAEL basis for white perch).

Modeled concentrations are based on a standard fillet lipid content. Although an adjustment is required to estimate whole body tissue concentrations, there was not enough data available to make this adjustment. Thus, because the presented results are based on forecast standard fillet concentrations, true risks are likely underestimated for these two species.

5.2.1.6 Measurement Endpoint: Comparison of Modeled TEQ Basis Body Burdens to Toxicity Reference Values for White and Yellow Perch

Tables 5-17 and 5-18 present the results of the comparison between forecast percentiles of white perch TEQ–based PCB body burdens to laboratory-derived NOAEL and LOAEL under future conditions (1993-2018). The white perch exceeds a TQ of one on a TEQ basis at RMs 152, 113, and 90 for the 25th percentile, median, and 95th percentile and at RM 50 for the 95th percentile for a portion of the modeling period. On a 95th percentile basis, the white perch exceeds one at RMs 152 and RM 133 throughout the modeling period (2018), at RM 90 until 2014, and at RM 50 until 2005. The median-based TQs exceed one at RM 152 until 2008, at RM 113 until 2003, at RM 90 until 1997, and at RM 50 until 1994. On a LOAEL basis, the 95th percentile exceeds one at RM 152 until 2004, at RM 113 until 1999, and at RM 90 until 1995. All median-based ratios were below one at RM 50.

Results for yellow perch are shown in Tables 5-19 and 5-20. These tables show similar results to white perch, but yellow perch TQs fall below one a few years before white perch.

Because modeled TEQ concentrations are expressed on a lipid-normalized basis, an adjustment for standard fillet to whole body is not required for this analysis.

5.2.1.7 Measurement Endpoint: Comparison of Modeled Tri+ PCB Fish Body Burdens to Toxicity Reference Values for Largemouth Bass

Table 5-21 presents the results of the comparison between forecast percentiles of largemouth bass total PCB body burdens to a field-based NOAEL for the period 1993-2018. The largemouth bass total PCB tissue concentrations exceed the field-based NOAEL for all concentrations (*i.e.*, 25th percentile, median, and 95th percentile) at all RM s (*i.e.*, 152, 113, 90, and 50) for the duration of the modeling period (1993-2018) with the exceptions of the 25th percentile at RM 90 for 2017 and 2018 and at RM 50 for 2014-2108. As the FISHRAND model predicts standard fillet concentrations in fish, the wet weight model results were adjusted by a factor of 2.5 for the largemouth bass, because wildlife feeding on fish consumes them whole. The majority of the ratios would exceed one even without this adjustment.

5.2.1.8 Measurement Endpoint: Comparison of Modeled TEQ Based Fish Body Burdens to Toxicity Reference Values for Largemouth Bass

Tables 5-22 and 5-23 present the results of the comparison between modeled largemouth bass body burdens and laboratory-based NOAEL and LOAEL on a TEQ basis under future conditions (1993-2018). On a 95th percentile basis, concentrations on a TEQ basis exceed the NOAEL at RM 152 and RM 133 throughout the modeling period (2018), at RM 90 until 2014, and at RM 50 until 2009. Using the LOAEL, the 95th percentile exceed one at RM 152 until about 2005, at RM 133 until 2003, at RM 90 until 1999, and at RM 50 until 1998.

5.2.1.9 Measurement Endpoint: Comparison of Modeled Tri+ PCB Fish Body Burdens to Toxicity Reference Values for Striped Bass

Table 5-24 presents the results of the comparison between forecast percentiles of striped bass total PCB body burdens to a field-based NOAEL at RMs 152 and 113 for the period 1993- 2018. At RM 152, the striped bass Tri+ PCB tissue concentrations exceed the field-based NOAEL on 95th percentile, median, and 25th percentile bases throughout the entire modeling period (1993-2018). At RM 113, a ratio of one is exceeded on a 95th percentile basis until 2005, on a median basis until 1999, and on a 25th percentile basis until 1996.

5.2.1.10 Measurement Endpoint: Comparison of Modeled TEQ Based Fish Body Burdens to Toxicity Reference Values for Striped Bass

Table 5-24 presents the results of the comparisons between forecast percentiles of striped bass PCB egg concentrations and a TEQ-based laboratory-based NOAEL and LOAEL at RMs 152 and 113. At RM 152, the striped bass TEQ-based egg concentrations exceed the NOAEL on 95th percentile, median, and 25th percentile bases throughout the entire modeling period (1993-2018) and the LOAEL is exceeded on all three bases for almost the entire modeling period. At RM 113, a NOAEL ratio of one is exceeded on a 95th percentile basis until 2003, on a median basis until 1997, and on a 25th percentile basis until 1994. Using the LOAEL, the 95th percentile was only exceeded in 1993.

5.2.2 Do Modeled PCB Water Concentrations Exceed Appropriate Criteria and/or Guidelines for the Protection of Aquatic Life and Wildlife?

5.2.2.1 Measurement Endpoint: Comparison of Modeled Water Column Concentrations of PCBs to Criteria

Table 5-2 presents the results of the comparison between modeled whole water PCB concentrations and appropriate criteria and guidelines. All forecast water concentrations (*i.e.*, average and 95% UCL) exceed the NYSDEC wildlife bioaccumulation criterion of 0.001 μg/L and the USEPA wildlife criterion of 1.2×10^{-4} μg/L at all four locations throughout the modeling period. The whole water concentrations also exceed the USEPA/NYSDEC benthic aquatic life chronic toxicity criterion of 0.014 μg/L for a portion of the modeling period for both average and 95% UCL at all modeling locations. These comparisons are likely to underestimate the true risk, as concentrations are expressed as the sum of the Tri+ and higher congeners, while the criteria are based on total PCBs (the sum of all congeners).

5.2.3 Do Modeled PCB Sediment Concentrations Exceed Appropriate Criteria and/or Guidelines for the Protection of Aquatic Life and Wildlife?

5.2.3.1 Measurement Endpoint: Comparisons of Modeled Sediment Concentrations to Guidelines

Table 5-1 presents the ratios of forecast sediment concentrations to various sediment guidelines. Comparisons are made on total PCB (Tri+) sediment concentrations (*i.e.*, NOAA, 1999a; Persaud *et al.*, 1993; and Washington State, 1997) and TOC-normalized sediment concentrations (*i.e.*, NYSDEC, 1999a and Persaud *et al.* 1993) to NOAA sediment effect concentrations (NOAA, 1999a), NYSDEC criteria (NYSDEC, 1999a), Ontario sediment quality guidelines (Persaud *et al.*, 1993), and Washington State sediment quality values (Washington State, 1997), as described in subsection 5.1.1.1.

Forecast total PCB sediment concentrations exceeded the NOAA threshold effect concentration, NOAA mid-range effect concentration, NYSDEC criteria for the protection of aquatic life from chronic toxicity and wildlife from toxic effects of bioaccumulation, Ontario no effect and lowest effect levels, and Washington State Microtox® and *Hyalella azteca* probable effect levels.

Many of the ratios of modeled sediment concentrations to appropriate guidelines exceed 10 or occasionally even 100. Forecast total PCB concentrations are Tri+ values, and do not include mono or dichlorinated congeners that usually contribute a portion of the total PCB load. Thus, even in the unlikely event that forecast sediment concentrations were to decrease by an order of magnitude or more, comparisons to sediment guidelines would show exceedances.

5.2.4 What Do the Available Field-Based Observations Suggest About the Health of Local Fish Populations?

5.2.4.1 Measurement Endpoint: Evidence from Field Studies

Observational data for Hudson River fish are available for the Lower Hudson River (*e.g.*, see Klauda *et al.* 1988). The strengths and limitations of observational data have been previously described. Based on the available data, the following observations provide insights into the potential future risks associated with the presence of PCBs. Each insight is qualified to reflect the limitations inherent in using observational data. In particular, there are no wildlife field studies currently available that have directly addressed impacts associated with the presence of PCBs to Lower Hudson River fish and wildlife.

Monitoring studies in the Lower Hudson River indicate that the fish community composition is probably very similar to that which was present over the past few centuries. Beebe and Savidge (1988) note that, "Except for a few species that entered the estuary through direct introductions or through canals connecting other watersheds, the species composition of the Hudson River estuary has probably remained similar to what it was at the time the area was settled by Europeans. All but five species (barndoor skate, Atlantic salmon, cobia, nine-spine stickleback, and sharksucker) have been collected within the last 20 years." No obvious losses of species that have occurred over the past few decades during which PCB exposures have been greatest; however recommendation have been made to limit the consumption of fish from the Lower Hudson River and the striped bass fishery has been closed since February 1976. The qualitative data can not be used to provide insight into the possibility that PCBs have reduced or impaired reproduction or rates of recruitment. Risks to these endpoints could exist even if the fish species are able to maintain themselves in these areas. For this reason, the analysis presented in subsection 5.2.1 comparing forecast body burdens to TRV values is required to judge the possible magnitude of these risks.

The shortnose sturgeon has been on the federal endangered species list since 1967. Studies of the abundance of shortnose sturgeon indicate that this species is reproducing in the Lower Hudson River (below the Federal Dam) and that the population numbers are increasing (Bain, 1997). Increases in populations in the absence of fishing pressures have not been well documented. Ecological studies on the Hudson River during the 1970s suggest possible increases during that period, but those increases are at least partly an artifact of improved sampling (*e.g.*, Hoff *et al.*, 1988). The changing ratio of shortnose sturgeon: Atlantic sturgeon catches is also indicative of an increasing shortnose sturgeon population in the Hudson River. While there is evidence that populations of shortnose sturgeon are increasing following their demise at the turn of the century and following improvements in overall water quality, the growth of the species's populations is likely to be slow as a result of its biology. Measurable increases in shortnose sturgeon populations should not be expected over short time periods (*i.e.*, decades) as the species matures late (at about 7-10 years) and spawns infrequently. While available data indicate that the population growth of shortnose sturgeon in the Hudson is positive, it is not possible to quantify from these data the extent to which PCB exposures might impair or reduce these population growth rates.

Population data indicate that white perch, a semi-anadromous fish in the Lower Hudson River, has exhibited positive population growth during the 1970s and 1980s, a period when PCB exposures in the Lower Hudson River may have been highest. The data indicate that PCB exposures to this fish species are not sufficiently high to significantly reduce reproduction and recruitment rates. Wells *et al.* (1992) have reported on studies of the white perch during the 1970s and 1980s. This species is a permanent resident in the Hudson and, together with the shortnose sturgeon, one of two Hudson River species that are representative primarily of the Lower Hudson River. Wells *et al.* (1992) studied several sources of Hudson River data for the period 1975 through 1987 and concluded that the population of white perch has increased over this period. This positive population growth has occurred during a period when PCB exposures have been occurring. This indicates that PCB exposure to white perch has not been sufficient to prevent reproduction or recruitment. In fact, populations have increased in size during this period. However, as noted above, there are many factors that influence population size and it is possible that PCBs could influence rates of reproduction and recruitment to a degree that is not manifested in recent population trends. The analyses performed in this chapter provide insight into the degree to which PCB body burdens in Hudson River fish might pose a risk to their reproductive and recruitment rates.

5.3 Evaluation of Assessment Endpoint: Protection and Maintenance (*i.e.*, Survival, Growth, and Reproduction) of Lower Hudson River Insectivorous Bird Populations (as Represented by the Tree Swallow)

5.3.1 Do Modeled Total and TEQ-Based PCB Dietary Doses to Insectivorous Birds and Egg Concentrations Exceed Benchmarks for Adverse Effects on Reproduction?

5.3.1.1 Measurement Endpoint: Modeled Dietary Doses on a Tri+ PCB Basis to Insectivorous Birds (Tree Swallow)

Table 5-25 compares modeled dietary doses for the period 1993 – 2018 for the tree swallow to the field-based TRV derived in the baseline ERA (USEPA, 1999c). This TRV was derived from the USFWS data from the Hudson River. For the entire modeling period, the TQs for the tree swallow are below one at all locations.

5.3.1.2 Measurement Endpoint: Predicted Egg Concentrations on a Tri+ PCB Basis to Insectivorous Birds (Tree Swallow)

Table 5-26 compares predicted egg concentrations for the period 1993 – 2018 for the tree swallow to the field-based TRV derived in the baseline ERA (USEPA, 1999c) under future conditions. This TRV was derived from the USFWS data from the Hudson River, and the biomagnification factor from aquatic insects to eggs was also obtained from these data. The predicted egg concentrations used a biomagnification factor of 2 based on the USFWS tree swallow data. For the entire modeling period, the TQs for the tree swallow are below one at all locations.

5.3.1.3 Measurement Endpoint: Modeled Dietary Doses of PCBs Expressed on a TEQ Basis to Insectivorous Birds (Tree Swallow)

Table 5-27 compares the estimated TEQ-based dietary dose and predicted egg concentration to the piscivorous birds to the field-based TRV for TEQs derived from the Phase 2 database (USEPA, 1998b). For the entire modeling period (1993-2018), the TQs for the tree swallow are below one at all locations.

5.3.1.4 Measurement Endpoint: Predicted Egg Concentrations Expressed on a TEQ Basis to Insectivorous Birds (Tree Swallow)

Table 5-28 compares the estimated TEQ-based predicted egg concentrations for insectivorous birds to the field-based TRV for TEQs derived for egg concentrations. The predicted egg concentrations used a biomagnification factor of 7 based on the USFWS tree swallow data. For the entire modeling period, the TQs for the tree swallow are below one at all locations for the entire modeling period.

5.3.2 Do Modeled Water Concentrations Exceed Criteria for Protection of Wildlife?

5.3.2.1 Measurement Endpoint: Comparison of Modeled Water Column Concentrations to Criteria for the Protection of Wildlife

Table 5-2 presents the results of the comparison between modeled whole water PCB concentrations and appropriate criteria. All forecast water concentrations (*i.e.*, average and 95% UCL) exceed the NYSDEC wildlife bioaccumulation criterion of 0.001 μg/L and the USEPA wildlife criterion of 1.2×10^{-4} μg/L at all four locations throughout the modeling period. The whole water concentrations also exceed the USEPA/NYSDEC benthic aquatic life chronic toxicity criterion of 0.014 μg/L for a portion of the modeling period for both average and 95% UCL at all modeling locations. These comparisons are likely to underestimate the true risk, as concentrations are expressed as the sum of the Tri+ and higher congeners, while the criteria are based on total PCBs (the sum of all congeners).

5.3.3 What Do the Available Field-Based Observations Suggest About the Health of Local Insectivorous Bird Populations?

5.3.3.1 Measurement Endpoint: Evidence from Field Studies

A natural history study of the wildlife species known to forage and reproduce within the project site represents an important measurement endpoint. Whereas a species is not required to be currently using a site for inclusion in the ecological risk assessment (*i.e.*, the species may have been severely impacted by site contamination/conditions), evidence of past use is important in validating the endpoints and toxicity factors utilized in the analysis.

The last ten annual Audubon Society Christmas bird counts for Albany, Rensselear, Dutchess, Putnam, Southern and East Orange, Rockland, Catskill, Lower Hudson, and Bronx/Westchester count circles (Cornell University, 1999) were examined to determine whether any general inferences on insectivorous bird populations along the Hudson River could be made. Because many insectivorous bird species are migratory (*e.g.*, flycatchers, swallows, gnatcatchers), the Christmas count alone does not provide a good population estimate for these species.

Despite their migratory nature, tree swallows were observed in Christmas count circles along the Lower Hudson River. The Saw Mill Audubon Society provided year-round information on bird sightings at Croton Point Park in Westchester since January 1994 (Bickford, 1999). Tree swallows have been sighted from March to September, with the exception of during July. Lack of adequate nesting holes may account for the low numbers of summer sightings.

The Lower Hudson Valley Bird Line transcripts (sponsored by the Sullivan County, Saw Mill River, Rockland, Putnam Highlands, and Bedford Audubon Society chapters) from January 1998 to August 1999 (Audubon, 1999) were reviewed. Tree swallows were noted in the transcripts in the spring months (March, April, and May) and again in the fall and winter (October to January).

5.4 Evaluation of Assessment Endpoint: Protection and Maintenance (*i.e.*, Survival, Growth, and Reproduction) of Lower Hudson River Waterfowl Populations (as Represented by the Mallard)

5.4.1 Do Modeled Total and TEQ-Based PCB Dietary Doses to Waterfowl and Egg Concentrations Exceed Benchmarks for Adverse Effects on Reproduction?

5.4.1.1 Measurement Endpoint: Modeled Dietary Doses of Tri+ PCBs to Waterfowl (Mallard)

Table 5-29 provides the results of the comparison between predicted dietary doses of the female mallard based on predictions for the modeling period 1993 to 2018 to the laboratory-based NOAEL and LOAEL TRVs developed in the baseline ERA (USEPA, 1999c). On a NOAEL basis, the predicted TQs exceed one on both an average and 95% UCL period for a portion of the modeling period at all four locations. At RM152, the 95% UCL exceeds one until 2007, and the average until 2004. On a LOAEL basis, predicted TQs do not exceed one at any location.

5.4.1.2 Measurement Endpoint: Predicted Egg Concentrations of Tri+ PCBs to Waterfowl (Mallard)

Table 5-30 provides the results of the comparison between predicted egg concentrations and laboratory-based TRVs for the period 1993 to 2018. The predicted egg concentrations used a biomagnification factor of 3 based on the USFWS mallard and wood duck data. The TQs for mallard eggs exceed one for the duration of the modeling period on a NOAEL basis, for both the average and 95% UCL, at all four locations for the entire modeling period. LOAEL-based comparisons exceed one for both the average and 95% UCL at RM 152 for the entire modeling period and at RM 113 for most of the modeling period (until 2016). The LOAEL also exceeds one on an average and 95% UCL basis for a portion of the modeling period at RMs 90 and 50.

5.4.1.3 Measurement Endpoint: Modeled Dietary Doses of TEQ-Based PCBs to Waterfowl (Mallard)

Table 5-31 provides the results of the comparison between predicted dietary doses and female mallard PCB dietary doses on a TEQ basis to laboratory-based TRVs. The results presented in this table show that the NOAEL and LOAEL-based comparisons exceed one at all four locations for the duration of the modeling period (1993-2018), for both the average and the 95% UCL concentrations by up to two orders of magnitude.

5.4.1.4 Measurement Endpoint: Predicted Egg Concentrations of TEQ-Based PCBs to Waterfowl (Mallard)

Table 5-32 provides the results of the comparison between predicted concentrations of PCBs in mallard egg and the field-based TRV for TEQs derived in the baseline ERA (USEPA, 1999c), using a biomagnification factor of 28. These results show that predicted TQs exceed one for all locations, years, and concentrations. Predicted TQs exceed 100 on a NOAEL and LOAEL basis at RMs 152 and 113 locations for the duration of the modeling period and exceed 100 on a NOAEL basis at RMs 90 and 50. This suggests the potential for adverse reproductive effects to waterfowl species.

5.4.2 Do Modeled PCB Water Concentrations Exceed Criteria for the Protection of Wildlife?

5.4.2.1 Measurement Endpoint: Comparison of Modeled Water Concentrations to Criteria

Table 5-2 presents the results of the comparison between modeled whole water PCB concentrations and appropriate criteria. All predicted water concentrations (*i.e.*, average and 95% UCL) exceed the NYSDEC wildlife bioaccumulation criterion of 0.001 μg/L and the USEPA wildlife criterion of 1.2 x 10^{-4} μg/L at all four locations throughout the modeling period. The whole water concentrations also exceed the USEPA/NYSDEC benthic aquatic life chronic toxicity criterion of 0.014 μg/L for a portion of the modeling period for both average and 95% UCL at all modeling locations. These comparisons are likely to underestimate the true risk, as concentrations are expressed as the sum of the Tri+ and higher congeners, while the criteria are based on total PCBs (the sum of all congeners).

5.4.3 What Do the Available Field-Based Observations Suggest About the Health of Lower Hudson River Waterfowl Populations?

5.4.3.1 Measurement Endpoint: Observational Studies

The last ten annual Audubon Society Christmas bird counts for the Lower Hudson Valley count circles (Cornell University, 1999) were examined to determine whether any inferences on local waterfowl populations along the Hudson River could be made. Mallards were generally one of the most abundant species sighted during the Christmas count. Other waterfowl, including Canada geese, American black duck, ring-necked duck, ruddy duck, and common merganser are commonly seen in the Hudson River area. Mallards, Canada geese, and mute swans were sighted throughout the year in Croton Point Park (Bickford, 1999).

The Saw Mill Audubon Society provided information on bird sightings at Croton Point Park in Westchester since January 1994 (Bickford, 1999). Mallards are numerous at Croton Point Park, but nesting is probably limited due to lack of proper habitat. On the basis of breeding surveys, the mallard population using the Hudson River estuary is stable to increasing (NYSDEC, 1997).

Not all waterfowl are likely to be adversely impacted by PCBs (particularly in the less contaminated stretches), but PCB sensitive species may experience total reproductive failure nesting in more contaminated areas.

5.5 Evaluation of Assessment Endpoint: Protection and Maintenance (*i.e.*, Survival, Growth, and Reproduction) of Hudson River Piscivorous Bird Populations (as Represented by the Belted Kingfisher, Great Blue Heron, and Bald Eagle)

5.5.1 Do Modeled Total and TEQ-Based PCB Dietary Doses to Piscivorous Birds and Egg Concentrations Exceed Benchmarks for Adverse Effects on Reproduction?

5.5.1.1 Measurement Endpoint: Modeled Dietary Doses of Total PCBs for Piscivorous Birds (Belted Kingfisher, Great Blue Heron, Bald Eagle)

Tables 5-33 through 5-35 compare the estimated total PCB (*i.e.*, Tri+) dietary dose of the female belted kingfisher, great blue heron, and bald eagle to the laboratory-based TRVs presented in Table 4-2 and derived in the baseline ERA (USEPA, 1999c). The site-related doses are based on modeled concentrations in forage fish, piscivorous fish, benthic invertebrates, whole water, and sediment using the results from the FISHRAND (fish and invertebrates) and Farley *et al.* (1999) (water and sediment) models.

The ratio of the female belted kingfisher dietary doses to the TRVs exceed one at all four locations for the entire modeling period on both a NOAEL and LOAEL basis (Table 5-33).

The ratio of the female great blue heron dietary doses to the TRVs exceed one at all four locations for the entire modeling period on a NOAEL basis (Table 5-34). Estimated TQs exceed one on a LOAEL basis at all locations for portions of the modeling period.

Table 5-35 presents the results for the bald eagle. Again, all comparisons exceed one for the duration of the modeling period at all locations on both a NOAEL and LOAEL basis for both average and 95% UCL doses.

Reproductive effects TQs for great blue heron, belted kingfisher, and bald eagle using average and upper confidence limits all exceed one. This indicates that exposure to PCBs from the Hudson River via prey and water present a risk of reproductive effects to these species on the basis of modeled Tri+ PCB dietary doses as compared to appropriate toxicity reference values. These results suggest the possibility of population-level impacts, as these TQs are based on reproductive effects, and consistently exceed one over the course of the modeling period.

5.5.1.2 Measurement Endpoint: Predicted Egg Concentrations Expressed as Tri+ to Piscivorous Birds (Eagle, Great Blue Heron, Kingfisher)

Tables 5-36 through 5-38 compare the estimated total PCB (*i.e.*, Tri+) predicted egg concentrations for the belted kingfisher, great blue heron, and bald eagle to the toxicity benchmarks summarized in Table 4-2. Laboratory-based NOAELs and LOAELs were used for the belted kingfisher and the great blue heron, whereas a field-based NOAEL was selected for the bald eagle. Egg concentrations are estimated using a biomagnification factor of 28 from Giesy *et al.* (1995).

Table 5-36 presents the results for the modeled belted kingfisher egg concentrations. These results are similar to those shown for the dietary dose. All comparisons at all locations exceed one a NOAEL and LOAEL basis using both average and 95% UCL concentrations for the duration of the modeling period.

Table 5-37 presents the results for the great blue heron. Again, all comparisons at all four locations exceed one on both a NOAEL and LOAEL basis for the duration of the modeling period.

Table 5-38 presents the results for the bald eagle. These results are similar to those shown for the dietary dose. All comparisons at all locations exceed one for the duration of the modeling period.

All of the predicted TQs exceeded one on the basis of estimated egg concentrations. These results suggest that exposure of piscivorous birds to PCBs from the Hudson River may result in adverse reproductive effects. The elevated TQ over time for the modeling period 1993 to 2018 suggests that exposure to PCBs over the long term has the potential to impact piscivorous birds, as represented by these species, on a population level.

5.5.1.3 Measurement Endpoint: Modeled Dietary Doses of PCBs Expressed as TEQs to Piscivorous Birds (Belted Kingfisher, Great Blue Heron, Bald Eagle)

Tables 5-39 through 5-41 present the results of the comparison between modeled dietary doses expressed on a TEQ basis to piscivorous receptors over the modeling period (1993 – 2018). Dietary doses were estimated using modeled concentrations in forage fish, piscivorous fish, benthic invertebrates, whole water, and sediment using the results from the FISHRAND (fish and invertebrates) and Farley *et al.* (1999) (water and sediment) models. Model results were multiplied by the weighted TEF factors derived in the baseline ERA (USEPA, 1999c). Laboratory-based TRVs for TEQs were used for all species (Table 4-2).

The ratio of the female belted kingfisher PCB dietary doses on a TEQ-basis to the TRVs exceed one at all four locations for the entire modeling period on both a NOAEL and LOAEL basis (Table 5-39).

The ratio of the female great blue heron dietary doses to the TRVs exceed one at all four locations for the entire modeling period on a NOAEL basis using both average and 95%UCL doses (Table 5-40). Estimated TQs exceed one on a LOAEL basis at all locations for portions of the modeling period.

Table 5-41 presents the TEQ-basis ratios for the bald eagle. All comparisons exceed one for the duration of the modeling period at all locations on both a NOAEL and LOAEL basis, with the exception of the LOAEL ratios at RM 50 for 2106-2018.

Reproductive effects TQs for great blue heron, belted kingfisher, and bald eagle using the average and 95% upper confidence limit on a TEQ basis often exceed one, and in many cases exceed 100. This indicates that PCBs from the Hudson River in the diet and water are likely to result in adverse reproductive effects to these species on the basis of modeled TEQ-based PCB dietary doses as compared to appropriate toxicity reference values. These results suggest adverse population-level effects may occur, given the consistent exceedance of a reproductive-based endpoint.

5.5.1.4 Measurement Endpoint: Modeled Dietary Doses of PCBs Expressed as TEQs to Piscivorous Birds (Belted Kingfisher, Great Blue Heron, Bald Eagle)

Tables 5-42 through 5-45 present the results of the comparison between piscivorous bird egg concentrations expressed on a TEQ-basis to TRVs (laboratory-based for the kingfisher and eagle, field-based for the heron) for the period 1993-2018. Egg concentrations were estimated using modeled concentrations in forage fish and piscivorous fish from the FISHRAND. Model results were multiplied by the weighted TEF derived in the ERA (USEPA, 1999c) and then multiplied by a biomagnification factor of 19 (Giesy *et al.*, 1995).

The belted kingfisher ratios exceed one for at all four locations throughout the entire modeling period (Table 5-42).

The ratio of the female great blue heron egg concentration to the TEQ-based TRV egg concentration exceed one at all four locations for the entire modeling period on a NOAEL basis (Table 5-34). Estimated TQs also exceed one on a LOAEL basis at RMs 152 and 113 for all of the modeling period and at RMs 90 and 50 for most of the modeling period (*i.e.*, up to 2014 or later).

The bald eagle TQs exceed one for at all four locations throughout the entire modeling period (Table 5-45). Ratios are as high as three orders of magnitude above one.

TQs based on reproductive effects for the great blue heron, belted kingfisher, and bald eagle using average and upper confidence limits on a TEQ basis all exceed one, and in many cases exceed 100, and several of the bald eagle TQs exceed 1000. This indicates that PCBs from the Hudson River in fish as they translate to egg concentrations are likely to result in adverse reproductive effects to these species on the basis of modeled TEQ-based PCB egg concentrations as compared to appropriate TRVs. These results suggest adverse population-level effects may occur, given the consistent exceedance of a reproductive-based endpoint.

5.5.2 Do Modeled Water Concentrations Exceed Criteria for the Protection of Wildlife?

5.5.2.1 Measurement Endpoint: Comparison of Modeled Water Concentrations to Criteria

Table 5-2 presents the results of the comparison between modeled whole water PCB concentrations and appropriate criteria. All forecast water concentrations (*i.e.*, average and 95% UCL) exceed the NYSDEC wildlife bioaccumulation criterion of 0.001 μg/L and the USEPA wildlife criterion of 1.2 x 10^{-4} μg/L at all four locations throughout the modeling period. The whole water concentrations also exceed the USEPA/NYSDEC benthic aquatic life chronic toxicity criterion of 0.014 μg/L for a portion of the modeling period for both average and 95% UCL at all modeling locations. These comparisons are likely to underestimate the true risk, as concentrations are expressed as the sum of the Tri+ and higher congeners, while the criteria are based on total PCBs (the sum of all congeners).

5.5.3 What Do the Available Field-Based Observations Suggest About the Health of Local Piscivorous Bird Populations?

5.5.3.1 Measurement Endpoint: Observational Studies

Both the New York State Endangered Species Unit and The Atlas of Breeding Birds in New York (Andrle and Carroll, 1988) provide general information regarding the bird species using the Hudson River. The belted kingfisher (*Ceryle alcyon*) appears to breed along the Hudson River north of Westchester County in areas such as Oscawana and George's Island Parks. Belted kingfishers may also be found in the area year-round, as evidenced by sightings of it in the Christmas bird count (Cornell University, 1999).

The great blue heron (*Ardea herodias*) is found along the Lower Hudson River throughout the year. It has been observed in most count circles during the Christmas bird count (Cornell University, 1999). There is a breeding colony of herons in the freshwater portion of the Lower Hudson River (Rensselaer County).

Bald eagles are slowly returning to the Lower Hudson River Valley. Up to 40 eagles have wintered in the 30 miles between Danskammer Point (Orange County) and Croton Point (Westchester County) in the last few years (USGS, 1999). Releases of young eagles in the 1980's have resulted in two nesting pairs along the Hudson River. However, these two breeding pairs have been unsuccessful in producing offspring (USGS, 1999). Bald eagles have been sighted intermittently during Christmas counts conducted in the last 10 years (Cornell University, 1999).

5.6 Evaluation of Assessment Endpoint: Protection (*i.e.*, Survival and Reproduction) of Local Insectivorous Mammal Populations (as represented by the Little Brown Bat)

5.6.1 Do Modeled Total and TEQ-Based PCB Dietary Doses to Insectivorous Mammalian Receptors Exceed Benchmarks for Adverse Effects on Reproduction?

5.6.1.1 Measurement Endpoint: Modeled Dietary Doses of Tri+ to Insectivorous Mammalian Receptors (Little Brown Bat)

Modeled total PCB (Tri+) dietary dose comparisons to laboratory-based TRVs (Table 4-3) are presented for the female little brown bat in Table 5-45 for the period 1993 – 2018. Dietary doses are estimated by using forecast water concentrations from the Farley *et al.* (1999) model and predicted invertebrate (aquatic insect) concentrations derived from the FISHRAND model. These results show that all comparisons exceed one for at all four locations throughout the modeling period on both a NOAEL and LOAEL basis for both average and 95%UCL doses.

These results suggest the potential for adverse reproductive effects to insectivorous mammalian species at all locations in the Lower Hudson River based on using predicted future concentrations in the exposure models.

5.6.1.2 Measurement Endpoint: Modeled Dietary Doses on a TEQ Basis to Insectivorous Mammalian Receptors (Little Brown Bat)

Modeled PCB dietary dose on a TEQ basis comparisons to laboratory-based TRVs for TEQs (Table 4-3) are presented for the little brown bat in Table 5-46. These results show that all comparisons exceed one (by one or two orders of magnitude) at all locations during the entire modeling period on both a NOAEL and LOAEL basis.

These results suggest the potential for adverse reproductive effects to insectivorous mammalian species at all locations in the river based on using the results from the baseline modeling in the exposure models. Given the consistency of the results, the magnitude of the exceedances, and the duration of the exceedances, these results suggest the potential for population-level adverse reproductive effects.

5.6.2 Do Modeled Water Concentrations Exceed Criteria for Protection of Wildlife?

5.6.2.1 Measurement Endpoint: Comparison of Modeled Water Concentrations to Criteria for the Protection of Wildlife

Table 5-2 presents the results of the comparison between modeled whole water PCB concentrations and appropriate criteria. All forecast water concentrations (*i.e.*, average and 95% UCL) exceed the NYSDEC wildlife bioaccumulation criterion of 0.001 μg/L and the USEPA wildlife criterion of 1.2 x 10^{-4} μg/L at all four locations throughout the modeling period. The whole water concentrations also exceed the USEPA/NYSDEC benthic aquatic life chronic toxicity criterion of 0.014 μg/L for a portion of the modeling period for both average and 95% UCL at all modeling locations. These comparisons are likely to underestimate the true risk, as concentrations are expressed as the sum of the Tri+ and higher congeners, while the criteria are based on total PCBs (the sum of all congeners).

5.6.3 What Do the Available Field-Based Observations Suggest About the Health of Local Insectivorous Mammalian Populations?

5.6.3.1 Measurement Endpoint: Observational Studies

A limited amount of data is available on little brown bat populations in the Lower Hudson River, and only a small subset of that data is within a time frame relevant to this study. Therefore, field-based observations do not provide sufficient information to evaluate this measurement endpoint.

5.7 Evaluation of Assessment Endpoint: Protection (*i.e.*, Survival and Reproduction) of Local Omnivorous Mammal Populations (as represented by the Raccoon)

5.7.1 Do Modeled Total and TEQ-Based PCB Dietary Doses to Omnivorous Mammalian Receptors Exceed Benchmarks for Adverse Effects on Reproduction?

5.7.1.1 Measurement Endpoint: Modeled Dietary Doses of Tri+ to Omnivorous Mammalian Receptors (Raccoon)

Modeled total PCB (Tri+) dietary dose comparisons to laboratory based TRVs (Table 4-3) are presented for the female raccoon in Table 5-47 for the period 1993 – 2018. Dietary doses are estimated by using forecast water concentrations from the Farley *et al.* (1999) model and predicted forage fish and benthic invertebrate concentrations from the FISHRAND model.

Predicted TQs for RMs 152, 113, and 90 exceed one on a NOAEL basis for both the average and 95% UCL. At RM 50 TQs exceed one on using the 95% UCL concentration until 2011 and using the average concentration until 2007. TQs were below one at all locations on a LOAEL basis.

5.7.1.2 Measurement Endpoint: Modeled Dietary Doses on a TEQ Basis to Omnivorous Mammalian Receptors (Raccoon)

Modeled PCB dietary dose on a TEQ basis comparisons to laboratory-based TRVs for TEQs (Table 4-3) are presented for the female raccoon in Table 5-48 for the period 1993 – 2018. All comparisons exceed one at all four locations for the duration of the modeling period on both a NOAEL and LOAEL basis for both average and 95% UCL concentrations.

These results suggest the potential for adverse reproductive effects to omnivorous mammalian species in the Lower Hudson River. Given the consistency of the results, the magnitude of the exceedances, and the duration of the exceedances, these results suggest the potential for population-level adverse reproductive effects in the Lower Hudson River.

5.7.2 Do Modeled Water Concentrations Exceed Criteria for Protection of Wildlife?

5.7.2.1 Measurement Endpoint: Comparison of Modeled Water Concentrations to Criteria for the Protection of Wildlife

Table 5-2 presents the results of the comparison between modeled whole water PCB concentrations and appropriate criteria. All forecast water concentrations (*i.e.*, average and 95% UCL) exceed the NYSDEC wildlife bioaccumulation criterion of 0.001 μg/L and the USEPA wildlife criterion of 1.2×10^{-4} μg/L at all four locations throughout the modeling period. The whole water concentrations also exceed the USEPA/NYSDEC benthic aquatic life chronic toxicity criterion of 0.014 $\mu g/L$ for a portion of the modeling period for both average and 95% UCL at all modeling locations. These comparisons are likely to underestimate the true risk, as concentrations are expressed as the sum of the Tri+ and higher congeners, while the criteria are based on total PCBs (the sum of all congeners).

5.7.3 What Do the Available Field-Based Observations Suggest About the Health of Local Omnivorous Mammalian Populations?

5.7.3.1 Measurement Endpoint: Observational Studies

A limited amount of quantitative data is available on raccoon populations in the Lower Hudson River. However, casual observations imply that raccoons are abundant along the Lower Hudson River Valley. However, a large proportion of the raccoon population in the Lower Hudson River Valley is likely to be obtaining food from sources other than the Hudson River, as the raccoon is an opportunistic feeder. Therefore, only a small subset of the Lower Hudson River Valley raccoon population is likely to be experience the daily doses calculated in the ERA Addendum.

5.8 Evaluation of Assessment Endpoint: Protection (*i.e.*, Survival and Reproduction) of Local Piscivorous Mammal Populations (as represented by the Mink and River Otter)

5.8.1 Do Modeled Total and TEQ-Based PCB Dietary Doses to Piscivorous Mammalian Receptors Exceed Benchmarks for Adverse Effects on Reproduction?

5.8.1.1 Measurement Endpoint: Modeled Dietary Doses of Tri+ to Piscivorous Mammalian Receptors (Mink, River Otter)

Tables 5-49 and 5-50 present the results of the comparison between modeled dietary doses to female mink and river otter under future conditions (1993-2018). Field-based TRVs derived in the baseline ERA (Table 4-3) are used for both species. Modeled dietary doses are estimated by using Farley *et al.* (1999) model results for water and sediment, and FISHRAND results for forage fish and piscivorous fish concentrations.

On a dietary dose basis for total (Tri+) PCBs, predicted TQs for the female mink exceed one on a NOAEL basis at all four locations for both the average and 95% UCL (Table 5-49). TQs were below one at all locations on a LOAEL basis.

Table 5-50 shows the results for the female river otter. On a dietary dose basis for total (Tri+) PCBs, predicted TQs exceed one on both a NOAEL and LOAEL basis at RMs 152 and 113 for average and 95% UCL doses. At RMs 90 and 50, a ratio of one is exceeded for on a NOAEL basis (average and 95%UCL). On a LOAEL basis, one is exceeded until 2004 at RM 90 and until 2002 at RM 50. The river otter consumes a larger size range of fish than the mink and is likely to obtain fish from deeper in the river. Thus, the exposure of the river otter is greater than that of the mink.

These results suggest the potential for adverse reproductive effects to piscivorous mammalian species in the Hudson River based on using model results in the exposure models for dietary dose. Reproductive effects TQs for the mink and otter using average and upper confidence limits exceed one for the duration of the modeling period, often by more than two orders of magnitude. Given the consistency of the results, the magnitude of the exceedances, and the duration of the exceedances, these results suggest that PCBs from the Lower Hudson River in the diet and water are likely to present a significant risk of reproductive effects to the mink and river otter.

5.8.1.2 Measurement Endpoint: Modeled Dietary Doses on a TEQ Basis to Piscivorous Mammalian Receptors (Mink, River Otter)

Tables 5-51 and 5-52 present the results of the comparison between modeled dietary doses to mink and river otter under future conditions for the period 1993 - 2018 on a TEQ basis. Modeled mink dietary doses on a TEQ basis exceed the field-based NOAEL and LOAEL for TEQs (Table 4-3) at all four locations for the duration of the modeling period for both the average and 95% UCL (Table 5-51).

Table 5-52 shows the results for the female river otter. Modeled otter dietary doses on a TEQ basis exceed the field-based NOAEL and LOAEL for TEQs one at all four locations for the duration of the modeling period for both the average and 95% UCL by up to three orders of magnitude. The river otter, which consumes larger fish than the mink, demonstrates higher TQs than the mink, as seen by comparing Tables 5-51 and 5-52.

These results suggest the potential for adverse reproductive effects to piscivorous mammalian species in the Hudson River based on using Farley *et al.* (1999) and FISHRAND model results in the exposure models for dietary dose. Given the consistency of the results, the magnitude of the exceedances, and the duration of the exceedances, these results suggest the potential for population-level adverse reproductive effects for mink and river otter consuming fish from the Hudson River.

Reproductive effects TQs for the mink and river otter using average and upper confidence limits all exceed one on both a total PCB and TEQ basis, with generally higher TEQ based TQs. This indicates that PCBs from the Lower Hudson River in the diet and water are likely to present a significant risk of reproductive effects to the mink and river otter on the basis of modeled PCB dietary doses as compared to appropriate toxicity reference values.

5.8.2 Do Modeled Water Concentrations Exceed Criteria for the Protection of Piscivorous Mammals?

5.8.2.1 Measurement Endpoint: Comparison of Modeled Water Concentrations to Criteria for the Protection of Wildlife

Table 5-2 presents the results of the comparison between modeled whole water PCB concentrations and appropriate criteria. All forecast water concentrations (*i.e.*, average and 95% UCL) exceed the NYSDEC wildlife bioaccumulation criterion of 0.001 μg/L and the USEPA wildlife criterion of 1.2×10^{-4} μg/L at all four locations throughout the modeling period. The whole water concentrations also exceed the USEPA/NYSDEC benthic aquatic life chronic toxicity criterion of 0.014 μg/L for a portion of the modeling period for both average and 95% UCL at all modeling locations. These comparisons are likely to underestimate the true risk, as concentrations are expressed as the sum of the Tri+ and higher congeners, while the criteria are based on total PCBs (the sum of all congeners).

5.8.3 What Do the Available Field-Based Observations Suggest About the Health of Local Mammalian Populations?

5.8.3.1 Measurement Endpoint: Observational Studies

NYSDEC is currently performing a comprehensive study of three distinct aspects of injury to Hudson River semi-aquatic mammals (Mayack, 1999a). This study consists of:

- Measuring the levels and nature of contamination in mink, muskrat, and otter from within the Hudson River watershed.
- Measuring the population size and distribution of selected mammals throughout the Hudson River ecosystem.
- Comparing mammalian reproductive success in the Upper Hudson River with that in the Lower Hudson River.

A primary objective of the NYSDEC study is to evaluate the extent of PCB contamination in mink, river otter, and muskrat populations downstream of a major point source at Fort Edward, NY. Analysis of a small number of mink and otter collected from the Hudson River region (Foley *et al.*, 1988) suggests that concentrations of PCBs in mink may cause reproductive impairment and a consequent decease in wild populations. Contaminant levels in populations upstream of Fort Edward will be compared to levels in populations downstream. The study aims to establish a downstream limit of potential contaminant impact on mammal populations in the Hudson River ecosystem. A second objective is to determine if the abundance of mink can be related to the distribution of PCB contamination within the Hudson River drainage.

Preliminary results from this study indicate that PCBs may have an adverse effect on the litter size and possibly kit survival of river otter in the Hudson River (Mayack, 1999b). Mink appear to be accumulating PCBs to a lesser extent than river otter, possibly because their diet has a greater proportion of uncontaminated prey. However, given the variability in diet and opportunistic nature of mink foraging a portion of the population may be exposed to high dietary levels of PCBs if aquatic prey are available. Levels of PCBs in river otter may represent a diet more highly contaminated with PCBs than that of mink, because fish comprise the majority of the river otter diet.

Mink, river otter, and muskrats are found in several localized areas along the Lower Hudson River. The herbivorous/omnivorous muskrat has had low pup abundances up and down the Hudson River (Kiviat, 1999). The reason is unknown.

5.9 Evaluation of Assessment Endpoint: Protection of Threatened and Endangered Species

Two threatened and/or endangered species, the shortnose sturgeon and bald eagle, were selected as receptors in this assessment. The populations of other endangered, protected, and species of concern found along the Hudson River (Chapter 2.6.5) may also be affected by PCBs. The bald eagle is considered to be a representative surrogate for wildlife species, and the shortnose sturgeon a representative surrogate for fish.

5.9.1 Do Modeled Total and TEQ-Based PCB Body Burdens in Local Threatened or Endangered Fish Species Exceed Benchmarks for Adverse Effects on Fish Reproduction?

5.9.1.1 Measurement Endpoint: Inferences Regarding Shortnose Sturgeon Population

There are no experimental data available to assess uptake of PCBs by shortnose sturgeon. To evaluate the potential impact of PCBs on shortnose sturgeon, observed and modeled largemouth bass total and TEQ based PCB concentrations were compared to toxicity reference values.

The derived toxicity reference values (Table 4-1) are considered protective of this species. This analysis assumes that shortnose sturgeon are likely to experience patterns of uptake somewhere between a largemouth bass and a brown bullhead. Shortnose sturgeon are primarily omnivorous, but can live in excess of 30 years and thus might be expected to accumulate more PCBs than their diet alone would suggest.

For PCBs expressed as total PCBs, the comparison is no different from the results already presented for the brown bullhead for Tri+ PCBs (Tables 5-10 and 5-11) and largemouth bass on a TEQ basis (Tables 5-22 and 5-23), because the toxicity reference values are the same.

The analyses performed for both total (Tri+) and TEQ-based PCBs indicate the potential for adverse effects as compared to the NOAEL and LOAEL TRV values. Therefore, the potential for adverse reproductive effects in shortnose sturgeon exists, particularly in the upper reaches of the Lower Hudson River (*i.e.*, RMs 152 and 113).

5.9.2 Do Modeled Total and TEQ-Based PCB Body Burdens/Egg Concentrations in Local Threatened or Endangered Species Exceed Benchmarks for Adverse Effects on Avian Reproduction?

5.9.2.1 Measurement Endpoint: Inferences Regarding Bald Eagle and Other Threatened or Endangered Species Populations

The modeled results for the bald eagle were presented in Section 5.5. Almost all comparisons across all locations and on a total PCB and TEQ-basis exceeded one, in some instances by more than three orders of magnitude. Both the dietary dose and egg-based results were consistent in this regard. Other threatened or endangered raptors, such as the peregrine falcon, osprey, northern harrier, and red-shouldered hawk may experience similar exposures.

5.9.3 Do Modeled Water Concentrations Exceed Criteria for the Protection of Wildlife?

5.9.3.1 Measurement Endpoint: Comparisons of Modeled Water Concentrations to Criteria for the Protection of Wildlife

Table 5-2 presents the results of the comparison between modeled whole water PCB concentrations and appropriate criteria. All forecast water concentrations (*i.e.*, average and 95% UCL) exceed the NYSDEC wildlife bioaccumulation criterion of 0.001 μg/L and the USEPA wildlife criterion of 1.2×10^{-4} μg/L at all four locations throughout the modeling period. The whole water concentrations also exceed the USEPA/NYSDEC benthic aquatic life chronic toxicity criterion of 0.014 μg/L for a portion of the modeling period for both average and 95% UCL at all modeling locations. These comparisons are likely to underestimate the true risk, as concentrations are expressed as the sum of the Tri+ and higher congeners, while the criteria are based on total PCBs (the sum of all congeners).

5.9.4 Do Modeled Sediment Concentrations Exceed Guidelines for the Protection of Aquatic Health?

5.9.4.1 Measurement Endpoint: Comparisons of Modeled Sediment Concentrations to Guidelines

Table 5-1 presents the ratios of forecast sediment concentrations to various sediment guidelines. Comparisons are made on total PCB (Tri+) sediment concentrations (*i.e.*, NOAA, 1999a; Persaud *et al.*, 1993; and Washington State, 1997) and TOC-normalized sediment concentrations (*i.e.*, NYSDEC, 1999a and Persaud *et al.* 1993) to NOAA sediment effect concentrations (NOAA, 1999a), NYSDEC criteria (NYSDEC, 1999a), Ontario sediment quality guidelines (Persaud *et al.*, 1993), and Washington State sediment quality values (Washington State, 1997), as described in subchapter 5.1.1.1.

Forecast total PCB sediment concentrations exceeded the NOAA threshold effect concentration, NOAA mid-range effect concentration, NYSDEC criteria for the protection of aquatic life from chronic toxicity and wildlife from toxic effects of bioaccumulation, Ontario no effect and lowest effect levels, and Washington State Microtox® and *Hyalella azteca* probable effect levels.

Many of the ratios of modeled sediment concentrations to appropriate guidelines exceed 10 or occasionally even 100. Forecast total PCB concentrations are Tri+ values, and do not include mono or dichlorinated congeners that usually contribute a portion of the total PCB load. Thus, even in the unlikely event that forecast sediment concentrations were to decrease by an order of magnitude or more, comparisons to sediment guidelines would show exceedances.

5.9.5 What Do the Available Field-Based Observations Suggest About the Health of Local Threatened or Endangered Fish and Wildlife Species Populations?

5.9.5.1 Measurement Endpoint: Observational Studies

While available data indicate that the population growth of shortnose sturgeon in the Hudson is positive, it is not possible to quantify from these data the extent to which PCB exposures might impair or reduce these population growth rates. The kinds of effects expected in the field include reduced fecundity, decreased hatching success, and similar kinds of reproductive impairment indicators, which are often difficult to discern. These effects may be masked by populations increases due to protection from fishing pressures.

The bald eagle was discussed in subsection 5.5.3.1. Bald eagles are slowly returning to the Lower Hudson River Valley, however their long-term breeding success is unknown. Releases of young eagles in the 1980's have resulted in two nesting pairs along the Hudson River. However, these two breeding pairs have been unsuccessful in producing offspring (USGS, 1999). Part of the difficulty of assessing populations is that there are no reference data to measure abundance against, as bald eagles have not breed along the Hudson River for decades.

5.10 Evaluation of Assessment Endpoint: Protection of Significant Habitats

The significant habitats found along the Hudson River (Tables 2-3) are unique, unusual, or necessary for the propagation of key species. Various measurement endpoints developed throughout this risk assessment are used to determine the potential for adverse effects on significant habitats and the animals and plants associated with them, rather than performing a quantitative evaluation of risks to ecological communities.

5.10.1 Do Modeled Total and TEQ-Based PCB Body Burdens/Egg Concentrations in Receptors Found in Significant Habitats Exceed Benchmarks for Adverse Effects on Reproduction?

5.10.1.1 Measurement Endpoint: Inferences Regarding Receptor Populations

Based on the comparisons of observed and modeled body burdens to toxicity reference values presented in this chapter, current PCB concentrations found in the Lower Hudson River (*i.e.*, RMs 152, 113, 90, and 50) exceed toxicity reference values for some fish, avian, and mammalian receptors. These comparisons indicate that animals feeding on Lower Hudson River-based prey may be affected by the concentrations of PCBs found in the river on both a total PCB and TEQ basis. In addition, based on the ratios obtained in this evaluation, other taxononic groups not directly addressed in this evaluation (*e.g.*, amphibians and reptiles) may also be affected by exposure to PCBs in the Lower Hudson River.

Many year-round and migrant species use the significant habitats along the Lower Hudson River for breeding or rearing their young. Therefore, exposure to PCBs may occur at a sensitive time in the life cycle (*i.e.*, reproductive and development) and have a greater effect on populations than at other times of the year.

5.10.2 Do Modeled Water Column Concentrations Exceed Criteria for the Protection of Aquatic Wildlife?

5.10.2.1 Measurement Endpoint: Comparison of Modeled Water Concentrations to Criteria for the Protection of Wildlife

Table 5-2 presents the results of the comparison between modeled whole water PCB concentrations and appropriate criteria. All forecast water concentrations (*i.e.*, average and 95% UCL) exceed the NYSDEC wildlife bioaccumulation criterion of 0.001 μg/L and the USEPA wildlife criterion of 1.2 x 10^{-4} μg/L at all four locations throughout the modeling period. The whole water concentrations also exceed the USEPA/NYSDEC benthic aquatic life chronic toxicity criterion of 0.014 μg/L for a portion of the modeling period for both average and 95% UCL at all modeling locations. These comparisons are likely to underestimate the true risk, as concentrations are expressed as the sum of the Tri+ and higher congeners, while the criteria are based on total PCBs (the sum of all congeners).

5.10.3 Do Modeled Sediment Concentrations Exceed Guidelines for the Protection of Aquatic Health?

5.10.3.1 Measurement Endpoint: Comparison of Modeled Sediment Concentrations to Guidelines for the Protection of Aquatic Health

Table 5-1 presents the ratios of forecast sediment concentrations to various sediment guidelines. Comparisons are made on total PCB (Tri+) sediment concentrations (*i.e.*, NOAA, 1999; Persaud *et al.*, 1993; and Washington State, 1997) and TOC-normalized sediment concentrations (*i.e.*, NYSDEC, 1999a and Persaud *et al.* 1993) to NOAA sediment effect concentrations (NOAA, 1999a), NYSDEC criteria (NYSDEC, 1999a), Ontario sediment quality guidelines (Persaud *et al.*, 1993), and Washington State sediment quality values (Washington State, 1997), as described in subchapter 5.1.1.1.

Forecast total PCB sediment concentrations exceeded the NOAA threshold effect concentration, NOAA mid-range effect concentration, NYSDEC criteria for the protection of aquatic life from chronic toxicity and wildlife from toxic effects of bioaccumulation, Ontario no effect and lowest effect levels, and Washington State Microtox® and *Hyalella azteca* probable effect levels.

Many of the ratios of modeled sediment concentrations to appropriate guidelines exceed 10 or occasionally even 100. Predicted total PCB concentrations are Tri+ values, and do not include mono or dichlorinated congeners that usually contribute a portion of the total PCB load. Thus, even in the unlikely event that forecast sediment concentrations were to decrease by an order of magnitude or more, comparisons to sediment guidelines would show exceedances.

5.10.4 What Do the Available Field-Based Observations Suggest About the Health of Significant Habitat Populations?

5.10.4.1 Measurement Endpoint: Observational Studies

The Waterfront Revitalization and Coastal Resources Act (WRCR) of 1981 declares it to be the public policy of New York State to conserve, protect, and, where appropriate, promote commercial and recreational use of fish and wildlife resources and to conserve fish and wildlife habitats identified by NYSDEC as critical to the maintenance or re-establishment of species of fish and wildlife (Executive Law of New York, Article 42, Sections 910-920). The implementation of this policy required that significant coastal habitats be identified and designated for protection. It was not feasible to designate very large ecosystem, such as the Hudson River, even though they support significant fish and wildlife populations. This would diminish the ability of the area's fish and wildlife values to compete with other land uses. Therefore, only smaller, discrete communities that contribute to the overall significance of the large ecosystem were evaluated (NYSDEC, 1984).

Because the effort to designate significant habitats was undertaken in the early 1980s, it can be assumed that these areas support important biological resources although they have been exposed to PCBs since the 1940s. Information on species observed using significant habitats in the Lower Hudson River is of limited use because there are no data available for the comparison of biological resources prior to exposure to PCBs. In addition, many areas experience other effects (*e.g.*, development and habitat loss) at the same time as PCB exposure, so it would be difficult to segregate out the cause for changes in communities, even if data were available. However, based on the receptor analyses provided in the previous sections, some sensitive species may experience reproductive effects when attempting to breed in Lower Hudson River significant habitats.

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Chapter 6

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6.0 UNCERTAINTY ANALYSIS

A qualitative or quantitative assessment of risk is inherently uncertain. At each step of the risk assessment process there are sources of uncertainty. The sources of uncertainty in this ERA Addendum include:

- Sampling error and representativeness;
- Analysis and quantitation uncertainties;
- Conceptual model uncertainties;
- Toxicological study uncertainties; and,
- Exposure and modeling uncertainties.

The first two sources of uncertainty are discussed in greater detail in the baseline ERA (USEPA, 1999c). The remaining three sources of uncertainty are discussed in the following sections.

6.1 Conceptual Model Uncertainties

The conceptual model links PCB sources, likely exposure pathways, and potential ecological receptors. It is intended to provide broad linkages of various receptor groups found along the Hudson River to PCB contamination in Hudson River sediments and surface waters. However, because it is a generalized model, it is not intended to mimic actual individuals or species currently living in or around the Hudson River. The actual linkages between the biotic levels often depend on seasonal availability of various prey and food items. Specific uncertainties in the exposure and food web modeling are discussed in section 6.3.

The conceptual model used in the ERA Addendum is limited to animals exposed to Lower Hudson River sediment and water, either directly or *via* the food chain. Many animals may be exposed to PCBs from the Hudson River *via* floodplain soil pathways. These pathways are outside of the scope of the ERA and ERA Addendum. Inclusion of these pathways would increase the risks to the mink and raccoon, whose risks were calculated assuming 49.5% and 60% non-river related diet sources, respectively (see Tables 3-21 and 3-22). In addition, risks for terrestrial species (*e.g.*, shrews and moles) exposed to PCBs originating in the Hudson River are outside the scope of the Reassessment RI/FS and therefore were not quantified, but may be above acceptable levels.

6.2 Toxicological Uncertainties

PCB toxicological studies cover a wide range of test species, doses, exposures, instruments, and analytical methods. Toxicity can be measured in units of total PCBs, Aroclor mixtures, PCB-congeners, or normalized toxic equivalency factors. The results of typical toxicological studies can be reported based on doses by diet, doses per body weight, and as body burdens, as a total PCB concentration, or lipid normalized concentration. The TRVs that were selected in this assessment were based on best-available information and professional judgment. There are other TRVs which could have been selected which would result in higher or lower toxicity quotients.

Aquatic studies are further complicated by various exposure methods. The test species can be exposed to PCBs *via* water, sediment, or direct dosing either by food or injection. Given the insolubility of PCBs, they often partition/adhere to non-aqueous phase materials. Not all studies consider the effect of sediment or some other matrices (*e.g.*, glass, cotton) on the actual exposure concentration and availability to test organisms.

Most TRVs are based upon laboratory exposures. Laboratory experiments offer the advantage of being able to control exposure conditions, while field experiments may be are closer to actual exposure conditions. Some of the possible reasons for differences between laboratory and field studies include:

- Laboratory stress on the organisms;
- The lab does not create the actual environmental conditions experienced in the field;
- Contaminant concentration in the water at the study area may be below the instrument detection limit and therefore will not be reproduced accurately in a laboratory;
- Increases in concentrations along the food chain are not always reflected in the laboratory; and
- Confounding effects of other environmental contaminants associated with PCBs in the environmental media.

Furthermore, differences in species sensitivity between laboratory test populations and endemic populations are often unknown.

There are several uncertainties associated with the toxicological studies that were used to develop the TRVs for this ERA Addendum. Uncertainty Factors (UFs) may be applied to toxicity values to address interspecies uncertainty, intraspecies uncertainty, less-than-lifetime at steady state, acute toxicity to chronic NOAELs, LOAELs to NOAELs, and modifying factors (Calabrese and Baldwin, 1993).

When toxicological data are not available for specific receptor species, a species-to-species extrapolation must be made. Generally, the closest taxonomic linked TRV (e.g., species >genus >family >order >class) is preferred. Extrapolations can be made with a fair degree of certainty between aquatic species within genera and genera within families (USEPA, 1996). In contrast, uncertainties associated with extrapolating between orders, classes, and phyla tend to be very high and are not preferred over more taxonomically similar comparisons (Suter, 1993). Species level adjustments may be made to address specific developmental or reproductive endpoints or for application to an endangered species. Under such circumstances, an uncertainty factor (UF) can be used to account for species to species variation or for accounting for specific sensitive life stages.

A less-than-lifetime UF may be used if the test species is exposed to a contaminant for a fraction of its lifespan. The purpose of this factor is to ensure that growth, maintenance, and reproductive functions are accounted for within a protective range of uncertainty. Additional UF factors may be added for extrapolating acute toxicity to chronic studies and adapting a LOAEL to a NOAEL. An additional modifying factor may be added if there are aspects of the TRV study that are not covered by the other UFs.

Fish TRVs were expressed as a body burden. The pumpkinseed, largemouth bass, white perch, and striped bass field-based NOAEL TRVs did not require any uncertainty factors. The laboratory-based TRVs developed for yellow perch and brown bullhead required an interspecies uncertainty factor of 10. The laboratory-based TRV developed for the spottail shiner required no uncertainty factor.

For the avian receptors, the tree swallow and kingfisher dietary dose based TRVs required no uncertainty factors. The dietary dose TRV for the mallard duck, great blue heron, and the bald eagle all required a factor of 10 uncertainty to account for subchronic to chronic extrapolation. TRVs developed for the concentration in avian eggs required no uncertainty factors for any avian receptor.

Mammalian receptors all required a factor of 10 uncertainty on a total PCB basis except for the otter, which required no uncertainty factors. For the raccoon and bat, this value was for interspecies comparisons. For mink, this value was for extrapolation from a subchronic study to a chronic value.

There is also uncertainty in the manner in which TEQ concentrations are characterized in the original studies upon which the TEQ-based TRV was based. Some toxicity studies used slightly different TEFs when evaluating TEQ concentrations. Where available, a comparison of the difference in the result between using the TEF reported in the paper as compared to the TEF used in this analysis was conducted. This difference was no more than 30% and typically on the order of 13% - 20%.

For fish, the selected TRVs were based on egg concentrations in lake trout. Because lake trout are among the most sensitive species tested, and the concentration was in the egg rather than an estimated dose, the interspecies and subchronic-to-chronic uncertainty factors were not required. For the avian receptors, the TEQ-based TRV for the tree swallow was based on Hudson River data (USFWS), thus, no uncertainty factors were required. The egg-based TRVs for TEQ congeners for the avian receptors was based on a study in gallinaceous birds, among the most sensitive of receptors. For this reason, as with fish, no uncertainty factors were required. Dietary dose TRVs for the avian receptors incorporated a factor of 10 subchronic-to-chronic uncertainty factor. For the mammals, an uncertainty factor of 10 was applied in deriving the TEQ-based TRV to account for potential interspecies differences. In conclusion, at most a factor of 10 was applied to the TEQ-based TRV for mammals and for dietary-dose based TRVs for avian receptors. Fish and avian eggs did not require any uncertainty factors.

6.3 Exposure and Modeling Uncertainties

6.3.1 Natural Variation and Parameter Error

Parameter error includes both uncertainty in estimating specific parameters related to exposure or the specific exposure point concentrations being applied in the exposure models (*e.g.*, sediment and water concentrations) as well as variability (*e.g.*, ingestion rate and body weight). Some parameters can be both uncertain and variable. It is important to distinguish uncertainty from variability. Variability represents known variations in parameters based on observed heterogeneity in the characteristics of a particular endpoint species. Variability can be better understood by collecting additional data, although never eliminated. Uncertainty can be reduced directly through the confirmation of applied assumptions or inferences through direct measurement. Therefore, it is theoretically possible to eliminate uncertainty but not variability.

A detailed description of sources of uncertainty and variability in the exposure model parameters is presented in the baseline ERA (USEPA, 1999c).

6.3.2 Model Error

Model error is the uncertainty associated with how well a model approximates the true relationships between environmental components (*i.e.*, exposure sources and receptors). Model error includes: inappropriate selection or aggregation of variables, incorrect functional forms, and incorrect boundaries (Suter, 1993). This is the most difficult form of uncertainty to evaluate quantitatively. In the ERA Addendum, model error is not expected to be a significant source of uncertainty, for the reasons presented below. Relationships between trophic levels and food web components in the Hudson River are well understood.

6.3.2.1 Uncertainty in the Farley Model

Uncertainty in the application of the Farley *et al.*(1999) model for the purposes of the ERA Addendum and the Mid-Hudson HHRA arises from several sources. These sources of uncertainty can be classified as one of two types: uncertainties which originate from the parameterization of the model, and uncertainties concerning the assumptions of future conditions in the Hudson.

The uncertainties in model parameterization stem from the uncertainties in the individual parameter estimates. Because the model is mechanistic, the various parameters are independently obtained from the literature whenever possible. In this manner, the number of parameters which must be determined in the calibration is minimized and model uncertainty is minimized. Nonetheless, the data available for calibration are not sufficient to constrain the model completely and it is possible that more than one model solution would satisfy all the available constraints. In particular, data on sediment and water column PCB concentrations are very limited temporally. The more extensive fish data set provides an integrating constraint on model parameterization because it requires accuracy of both the fate and transport and the bioaccumulation models. However, its constraints on the fate and transport model are indirect and therefore limited. While the model uncertainty originating from parameterization is not known quantitatively, it is likely to be less than that associated with estimating future conditions. Indeed, the fact that the model is able to reproduce the general trends of the existing sediment, water and fish data suggests that the model uncertainty from parameterization is similar to the scale of the differences between the model calibration and the data themselves.

The second and probably greater source of uncertainty in the model is inherent in the assumption of future conditions. In order to estimate future PCB conditions, it is also necessary to estimate future hydrology, sediment loads, external PCB sources and other concerns. To some degree, hydrology and sediment loads can be estimated from historical records but the length of the forecast required adds great uncertainty. In particular, changes in land use, population density and other societal demands on the watershed are likely to change nature of water and sediment loads to the Lower Hudson relative to those assumed for the forecast. Similarly, assumptions of future PCB loads are also difficult to estimate and constrain. As demonstrated by the comparison of the HUDTOX and original Farley *et al.* (1999) model loads at the Federal Dam, the loads from the Upper Hudson have a significant effect on Lower Hudson fish body burdens. Thus, estimation of external PCB loads such as that at the Federal Dam represent a potentially large source of uncertainty. The use of HUDTOX model loads at Federal Dam is a direct attempt to minimize the uncertainty of the Federal Dam load. By using the HUDTOX forecast, loads from the sediments of the Upper Hudson, currently the most important external source to the Lower Hudson River, are relatively well constrained. However, the loads originating from the General Electric facilities at Hudson Falls and Fort Edward, NY remain an important source of long-term uncertainty to both Upper and Lower Hudson models of PCB contamination.

It is important to note that uncertainties associated with the estimation of future conditions affects any and all forecast models and is not unique to the models used by the USEPA. The reader is referred to the original work by Farley *et al.* (1999) for additional discussion of uncertainty associated with the Farley *et al.* (1999) fate and transport and bioaccumulation models.

6.3.2.2 Uncertainty in FISHRAND Model Predictions

A more detailed uncertainty and sensitivity analysis in the FISHRAND model is provided in the Baseline Modeling Report (USEPA, 1999b). Those results are summarized here.

Two approaches were used to evaluate the impact of small changes in user-specified input parameters (*e.g.*, lipid content in the organisms, weight of the organisms, water temperature, total organic carbon, sediment and water concentrations, and K_{ow}) and model constants on predicted fish body burdens.

In the first approach, a sensitivity analysis was conducted to evaluate the effect of varying the input parameters using a Monte Carlo methodology. In this method, combinations of values for the input parameters are generated randomly. Each parameter appears with the frequency suggested by its probability distribution. For each combination of input parameters, the output of the model is recorded. Each individually recorded input parameter is then plotted against the predicted body burden for that simulation. This is repeated many times to generate plots representing all possible combinations of input parameters leading to predicted body burdens.

The partial rank and Spearman rank regression techniques (Morgan and Henrion, 1990) are used as a formal method to find the most important parameters for the model performance. If the Spearman or partial rank regression coefficient (PRRC or SRRC) is close to 1 or -1 for a specific input model parameter, this parameter significantly influences model output. The percent lipid in fish is strongly negatively correlated with PCB body burden expressed on a lipid-normalized basis. This is because increases in lipid increase the PCB storage capacity of the fish, reducing the apparent concentration. As expected, the percent lipid in fish is positively associated for the wet weight results, but less so. This confirms that particularly on a lipid-normalized basis, the percent lipid distribution is very important. K_{ow} and benthic percent lipid are also important for some species on a wet weight basis. Feeding preferences are only weakly correlated with body burdens in terms of sensitivity to this parameter.

To evaluate changes in the model constants themselves, sensitivity to model constants was evaluated by approximating an analytical solution and then taking partial derivatives of all the model constants with respect to fish concentration. These partial derivatives were plotted to evaluate changes in magnitude and sign over time. The assimilation efficiency and growth rate were determined to be the most important parameters in terms of effect on predicted fish concentration.

The modeling results for this assessment show that the FISHRAND model tends to underpredict at specific locations and for specific years. On a median basis, FISHRAND does not overpredict. The FISHRAND calibration focused on optimizing wet weight concentrations, as described in the Baseline Modeling Report (USEPA, 1999b). This was done for three reasons. First, the model predicts a wet weight concentration in fish, and provides lipid normalized results by dividing the predicted wet weight concentration by a percent lipid. Second, the lipid content of any given fish is difficult to predict from first principles alone. Finally, potential target levels in fish are typically described as wet weight concentrations.

Optimizing the model for wet weight concentrations provides a reasonable basis upon which to make forecasts. In addition to forecasting fish responses to changes in sediment and water concentrations, it is also necessary to predict lipid content. By simply relying on the observed lipid for each year for which there are data, it is possible to obtain close to perfect agreement between hindcast and observed body burdens. This approach makes forecasts tenuous, however. Instead, the FISHRAND model forecasts wet weight concentrations by relying on a distribution of lipid values in each fish species that is representative of the observed variability in lipid content. This provides a more robust basis upon which to make predictions.

Focusing specifically on the wet weight results, largemouth bass hindcasts at RM 152 are within between 60% and 17% less than the observed medians, and fall within the lower bound of the error bars. This percentage represents 2 or 3 ppm on an absolute basis. At RM 113, hindcast largemouth bass concentrations of PCBs are between 3% and 50% less than the observed medians. For the period 1993 to 1996, the error between hindcast and observed is no more than 13%, representing less than 0.5 ppm PCBs on an absolute basis.

Brown bullhead concentrations of PCBs are typically within 6% and 30% less than the observed medians at RM 152, except for 1991. This difference represents less than one ppm on an absolute basis. White perch FISHRAND hindcasts at RM 152 are within 20% to 65% less than observed values for 1992 – 1994, but exceed the observed median by 20% for 1996. Hindcast concentrations of PCBs for 1993 and 1996 fall within the error bars of the observed median. These values range from less than one ppm to slightly more than a one ppm on an absolute basis. At RM 113, the hindcast white perch concentration in 1994 exceeds the observed median by 100%. However, for the remaining years, hindcast concentrations of PCBs fall below observed values by 40%, 6%, and 60% for 1993, 1995, and 1996, respectively. For 1996, this difference is 3 ppm PCBs on an absolute basis. Hindcasts for yellow perch exceed in 1991, but fall below for 1992 and 1993 (50% and 21%, respectively), although for 1993 the hindcast concentration is within the error bounds of the observed concentration. At RM 113, hindcast yellow perch concentrations of PCBs are 21% underpredicted for 1993 (but within the error bounds), and 36% overpredicted for 1994.

6.3.3 Sensitivity Analysis for Risk Models for Avian and Mammalian Receptors

Sensitivity analyses on the exposure and risk models were conducted by specifying distributions for key parameters. This allows the generation of a distribution of toxicity quotients to quantitatively evaluate the contribution of key parameters to the variance in the output based on the inputs. Distributions were described as triangular and were based on the ranges for exposure parameters presented in detail in Chapter 3 of the baseline ERA (USEPA, 1999c). Environmental concentrations were described as lognormal by a geometric mean and geometric standard deviation. Toxicity reference values were described as uniform and typically spanned an order of magnitude (see discussion above). Results showed that toxicity quotients were most sensitive to changes in concentrations in exposure media, followed by changes in the toxicity value, and finally by changes in exposure parameters (*e.g.*, ingestion rates and body weights). These results were consistent for all avian and mammalian receptors.

The output distributions of toxicity quotients generated by this Monte Carlo analysis represent population heterogeneity. Results are expressed as the ratio of selected percentiles to the expected toxicity quotient (based on the average) and show that the 95th percentile of toxicity quotients is typically 3.5 to 5 times the average, and the 99th percentile of toxicity quotients is typically at 10 to 15 times the average. Ninety-nine percent of the population is expected to experience the 99th percentile toxicity quotient or less, and which is estimated as between 10 and 15 times greater than the values shown in the tables for the average. These results were consistent for both avian and mammalian receptors.

Ratios of the 25^{th} percentile to the average typically range from 0.6 to 0.8 for the avian and mammalian receptors. This result suggests that even at the 25^{th} percentile, modeled dietary doses and/or egg concentrations exceed toxicity reference values for most of the receptors (with the exception of the tree swallow).

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Chapter 7

7.0 CONCLUSION

This chapter summarizes the results of the ERA Addendum. A summary of the results for each assessment endpoint is presented. The results of the risk characterization are evaluated in the context of uncertainties in a weight-of-evidence approach to assess the potential for adverse reproductive effects in the receptors of concern as a result of exposure to PCBs in the Lower Hudson River originating in the Upper Hudson River.

7.1 Assessment Endpoint: Benthic Community Structure as a Food Source for Local Fish and Wildlife

Risks to local benthic invertebrate communities were examined using two lines of evidence. These lines of evidence are: 1) comparison of modeled water column concentrations of PCBs to criteria and 2) comparisons of modeled sediment concentrations to guidelines.

Modeled concentrations of PCBs in river water and sediment in the Lower Hudson River show exceedances of the majority of their respective criteria and guidelines through the duration of the forecast period (1993 to 2018), indicating the potential for adverse effects on benthic invertebrate communities.

The uncertainty associated with the application of the Farley *et al.* (1999) model to estimate sediment and water concentrations is fairly low. The model is well constrained by the available sediment, water and fish data. Far greater uncertainty is associated with estimating future forcing conditions for the model (*i.e.*, external PCB loads, sediment loads and river hydrology). This uncertainty applies to all such forecasts and is not limited to the Farley *et al.* (1999) model. It is likely that the uncertainty in the model forecasts of sediment and water is on the order of a factor of two.

7.2 Assessment Endpoint: Protection and Maintenance (*i.e.*, Survival, Growth, and Reproduction) of Local Fish (Forage, Omnivorous, and Piscivorous) Populations

Risks to local fish populations were examined using five lines of evidence. These lines of evidence are: 1) comparison of modeled total PCB fish body burdens to TRVs; 2) comparison of modeled TEQ fish body burdens to TRVs; 3) comparison of modeled water column concentrations of PCBs to criteria; 4) comparisons of modeled sediment concentrations to guidelines; and 5) field-based observations. Multiple receptors were evaluated for forage and semi-piscivorous/piscivorous fish.

Collectively, the evidence indicates that future PCB exposures (predicted from 1993 to 2018) are not expected to be of a sufficient magnitude to prevent reproduction or recruitment of common fish species in the Lower Hudson River. However, based upon toxicity quotients, future exposure to PCBs may reduce or impair the survival, growth, and reproductive capability of some forage species (*e.g.*, pumpkinseed), omnivorous fish (*e.g.*, brown bullhead) and semi-piscivorous/piscivorus fish (*e.g.*, white perch, yellow perch, largemouth bass, and striped bass), particularly in the upper reaches of the Lower Hudson River.

There is a moderate degree of uncertainty in the modeled body burdens used to evaluate exposure, and at most an order of magnitude uncertainty in the TRVs (for the TEQ-based TRVs no uncertainty factors were needed).

Modeled concentrations of PCBs in river water and sediment in the Lower Hudson River show exceedances of the majority of their respective criteria and guidelines through the duration of the forecast period (1993-2018).

7.3 Assessment Endpoint: Protection and Maintenance (*i.e.*, Survival, Growth, and Reproduction) of Hudson River Insectivorous Bird Species (as Represented by the Tree Swallow)

Risks to local insectivorous bird populations were examined using six lines of evidence. These lines of evidence are: 1) comparison of modeled total PCB dietary doses to TRVs; 2) comparison of modeled TEQ dietary doses to TRVs; 3) comparison of modeled total PCB egg concentrations to TRVs; 4) comparison of modeled TEQ egg concentrations to TRVs; 5) comparison of modeled water column concentrations of PCBs to criteria; and 6) field-based observations. The tree swallow was selected to represent insectivorous bird species.

Collectively, the evidence indicates that future PCB exposures (predicted from 1993 to 2018) are not expected to be of a sufficient magnitude to prevent reproduction or recruitment of common insectivorous bird species in the Lower Hudson River Valley.

There is a moderate degree of uncertainty in the calculated modeled concentrations of PCBs in tree swallow diets and the concentrations of PCBs in eggs. There is a low degree of uncertainty associated with tree swallow TRVs, which were derived from field studies of Hudson River tree swallows.

Modeled concentrations of PCBs in river water in the Lower Hudson River show exceedances of criteria developed for the protection of wildlife through the duration of the forecast period (1993 to 2018).

7.4 Assessment Endpoint: Protection and Maintenance (*i.e.*, Survival, Growth and Reproduction) of Lower Hudson River Waterfowl (as Represented by the Mallard)

Risks to local waterfowl populations were examined using six lines of evidence. These lines of evidence are: 1) comparison of modeled total PCB dietary doses to TRVs; 2) comparison of modeled TEQ dietary doses to TRVs; 3) comparison of modeled total PCB egg concentrations to TRVs; 4) comparison of modeled TEQ egg concentrations to TRVs; 5) comparison of modeled water column concentrations of PCBs to criteria; and 6) field-based observations. The mallard was selected to represent waterfowl.

Collectively, the evidence indicates that future PCB exposures (predicted from 1993 to 2018) are not expected to be of a sufficient magnitude to prevent reproduction or recruitment of common waterfowl in the Lower Hudson River Valley. However, based upon toxicity quotients, future exposure to PCBs may reduce or impair the survival, growth, and reproductive capability of some waterfowl, particularly in the upper reaches of the Lower Hudson River.

Calculated dietary doses of PCBs and concentrations of PCBs in eggs typically exceed their respective TRVs throughout the modeling period. Toxicity quotients for the TEQ-based (*i.e.*, dioxin-like) PCBs consistently show greater exceedances than for total (Tri+) PCBs. There is a moderate degree of uncertainty in the dietary dose and egg concentrations estimates. Given the magnitude of the TEQ-based TQs, they would have to decrease by an order of magnitude or more to fall below one for waterfowl in the Lower Hudson River.

Modeled concentrations of PCBs in river water in the Lower Hudson River show exceedances of criteria developed for the protection of wildlife through the duration of the forecast period (1993 to 2018).

7.5 Assessment Endpoint: Protection and Maintenance (*i.e.*, Survival, Growth, and Reproduction) of Hudson River Piscivorous Bird Species (as Represented by the Belted Kingfisher, Great Blue Heron, and Bald Eagle)

Risks to local semi-piscivorous/piscivorous bird populations were examined using six lines of evidence. These lines of evidence are: 1) comparison of modeled total PCB dietary doses to TRVs; 2) comparison of modeled TEQ dietary doses to TRVs; 3) comparison of modeled total PCB egg concentrations to TRVs; 4) comparison of modeled TEQ egg concentrations to TRVs; 5) comparison of modeled water column concentrations of PCBs to criteria; and 6) field-based observations. The belted kingfisher, great blue heron, and bald eagle were selected to represent piscivorous birds.

Collectively, the evidence indicates that future PCB exposures (predicted from 1993 to 2018) are not expected to be of a sufficient magnitude to prevent reproduction or recruitment of these piscivorous species. However, based upon toxicity quotients, future exposure to PCBs may reduce or impair the survival, growth, and reproductive capability of some piscivorous birds, particularly in the upper reaches of the Lower Hudson River. Calculated dietary doses of PCBs and concentrations of PCBs in eggs exceed all TRVs (*i.e.*, NOAELs and LOAELs) for the belted kingfisher and bald eagle throughout the modeling period, and NOAELs for the great blue heron. Toxicity quotients for egg concentrations are generally higher than body burden TQs.

There is a moderate degree of uncertainty in the dietary dose and egg concentrations estimates. Given the magnitude of the TQs, they would have to decrease by an order of magnitude or more to fall below one for piscivorous birds in the Lower Hudson River. In particular, the bald eagle TQs exceeded one by up to three orders of magnitude. Therefore, even if the factor of 2.5 to adjust from largemouth bass fillets to whole body burden and the subchronic-to-chronic uncertainty factor of 10 used for the body burden TRV are removed, the TQs would remain well over one. These results, coupled with the lack of breeding success in Lower Hudson River bald eagles (USGS, 1999), indicate that reproductive effects may be present.

Modeled concentrations of PCBs in river water in the Lower Hudson River show exceedances of criteria developed for the protection of wildlife through the duration of the forecast period (1993 to 2018).

7.6 Assessment Endpoint: Protection (*i.e.*, Survival and Reproduction) of Insectivorous Mammals (as represented by the Little Brown Bat)

Risks to local insectivorous mammal populations were examined using four lines of evidence. These lines of evidence are: 1) comparison of modeled total PCB dietary doses to TRVs; 2) comparison of modeled TEQ mammal dietary doses to TRVs; 3) comparison of modeled water column concentrations of PCBs to criteria; and 4) field-based observations. The little brown bat was selected to represent insectivorous mammals.

Collectively, the evidence indicates that future PCB exposures (predicted from 1993 to 2018) are not expected to be of a sufficient magnitude to prevent reproduction or recruitment of common insectivorous mammals in the Lower Hudson River Valley. However, exposure to PCBs may reduce or impair the survival, growth, or reproduction capability of insectivorous mammals in the Lower Hudson River. Modeled dietary doses for the little brown bat exceed TRVs by up to two orders of magnitude at all locations modeled. There is a moderate degree of uncertainty in the calculated dietary doses.

Modeled concentrations of PCBs in river water in the Lower Hudson River show exceedances of criteria developed for the protection of wildlife through the duration of the forecast period (1993 to 2018).

7.7 Assessment Endpoint: Protection (*i.e.*, Survival and Reproduction) of Local Omnivorous Mammals (as represented by the Raccoon)

Risks to local omnivorous mammal populations were examined using four lines of evidence. These lines of evidence are: 1) comparison of modeled total PCB dietary doses to TRVs; 2) comparison of modeled TEQ dietary doses to TRVs; 3) comparison of water column concentrations of PCBs to criteria; and 4) field-based observations. The raccoon was selected to represent omnivorous mammals.

Collectively, the evidence indicates that future PCB exposures (predicted from 1993 to 2018) are not expected to be of a sufficient magnitude to prevent reproduction or recruitment of common omnivorous mammals in the Lower Hudson River Valley. However, exposure to PCBs may reduce or impair the survival, growth, or reproduction capability of omnivorous mammals in the Lower Hudson River. Modeled dietary doses for the raccoon exceed dietary dose NOAELs on a total PCB (Tri+) basis and all TRVs on a TEQ-basis. There is a moderate degree of uncertainty in the calculated dietary doses.

Modeled concentrations of PCBs in river water in the Lower Hudson River show exceedances of criteria developed for the protection of wildlife through the duration of the forecast period (1993 to 2018).

7.8 Assessment Endpoint: Protection (*i.e.*, Survival and Reproduction) of Local Piscivorous Mammals (as represented by the Mink and River Otter)

Risks to local semi-piscivorous/piscivorous mammal populations were examined using four lines of evidence. These lines of evidence are: 1) comparison of modeled total PCB dietary doses to TRVs; 2) comparison of modeled TEQ dietary doses to TRVs; 3) comparison of modeled water column concentrations of PCBs to criteria; and 4) field-based observations. The mink and river otter were selected to represent piscivorous mammals.

Collectively, the evidence indicates that future PCB exposures (predicted from 1993 to 2018) are not expected to be of a sufficient magnitude to prevent reproduction or recruitment of these piscivorous species. However, based upon toxicity quotients, future exposure to PCBs may reduce or impair the survival, growth, and reproductive capability of piscivorous mammals, particularly in the upper reaches of the Lower Hudson River. Calculated dietary doses of PCBs exceed the NOAEL on a total PCB basis for both species and exceed all TEQ-based TRVs by up to three orders of magnitude.

There is a moderate degree of uncertainty in the dietary dose estimates. However, given the magnitude of the TQs, they would have to decrease at least an order of magnitude to fall below one. In particular, the river otter TQs exceeded one by up to three orders of magnitude. Therefore, even if the factor of 2.5 to adjust from largemouth bass fillets to whole body burden is removed, the TQs would remain well over one. Preliminary results from a NYSDEC study indicate that PCBs may have an adverse effect on the litter size and possibly kit survival of river otter in the Hudson River (Mayack, 1999b), validating the TQ results.

Modeled concentrations of PCBs in river water in the Lower Hudson River show exceedances of criteria developed for the protection of wildlife through the duration of the forecast period (1993 to 2018).

7.9 Assessment Endpoint: Protection of Threatened and Endangered Species

Risks to threatened and endangered species were examined using five lines of evidence. These lines of evidence are: 1) comparison of modeled total PCB dietary doses/egg concentrations to TRVs; 2) comparison of modeled TEQ dietary doses/egg concentrations to TRVs; 3) comparison of modeled water column concentrations of PCBs to criteria; 4) comparison of modeled sediment concentrations of PCBs to guidelines; and 5) field-based observations. The shortnose sturgeon and bald eagle were selected to represent threatened and endangered species.

Collectively, the evidence indicates that future PCB exposures (predicted from 1993 to 2018) are not expected to be of a sufficient magnitude to prevent reproduction or recruitment of threatened or endangered species. However, using the TEQ-based toxicity quotients, potential for adverse reproductive effects in shortnose sturgeon exists, particularly when considering the long life expectancy of the sturgeon (30 years, [Bain, 1997]). Almost all TQs calculated for the bald eagle (across all locations) exceeded one, in some instances by more than three orders of magnitude. Both the dietary dose and egg-based results were consistent in this regard. Other threatened or endangered raptors, such as the peregrine falcon, osprey, northern harrier, and red-shouldered hawk may experience similar exposures.

There is a moderate degree of uncertainty in the dietary dose estimates. However, the bald eagle TQs exceeded one by up to three orders of magnitude. Therefore, even if the factor of 2.5 to adjust from largemouth bass fillets to whole body burden and the subchronic-to-chronic uncertainty factor of 10 used for the body burden TRV are removed, the TQs would remain well over one. These results, coupled with the lack of breeding success in Lower Hudson River bald eagles (USGS, 1999), indicate that reproductive effects may be present.

Modeled concentrations of PCBs in river water and sediment in the Lower Hudson River show exceedances of the majority of their respective criteria and guidelines through the duration of the forecast period (1993 to 2018).

7.10 Assessment Endpoint: Protection of Significant Habitats

Risks to significant habitats were examined using four lines of evidence. These lines of evidence are: 1) toxicity quotients calculated for receptors in this assessment; 2) comparison of modeled water column concentrations of PCBs to criteria; 3) comparison of modeled sediment concentrations of PCBs to guidelines; and 4) field-based observations.

Based on the toxicity quotients calculated in ERA Addendum, future PCB concentrations (predicted from 1993 to 2018) in the Lower Hudson River exceed toxicity reference values for some fish, avian, and mammalian receptors. These comparisons indicate that animals feeding on Lower Hudson River-based prey may be affected by the concentrations of PCBs found in the river on both a total PCB and TEQ basis. In addition, based on the TQs, other taxononic groups not directly addressed in the ERA and ERA Addendum (*e.g.*, amphibians and reptiles) may also be affected by PCBs in the river. Many year-round and migrant species use the significant habitats along the Hudson River for breeding or rearing their young. Therefore, exposure to PCBs may occur at a sensitive time in the life cycle (*i.e.*, reproductive and development) and have a greater effect on populations than at other times of the year.

Modeled concentrations of PCBs in river water and sediment in the Lower Hudson River show exceedances of the majority of their respective criteria and guidelines through the duration of the forecast period (1993 to 2018).

7.11 Summary

The results of the ERA Addendum indicate that receptors in close contact with the Lower Hudson River may experience adverse effects as a result of exposure to PCBs in prey, water, and sediments. Higher trophic level receptors, such as the bald eagle and the river otter, are considered to be particularly at risk. Risks are generally highest up river (*i.e.*, closer to the PCB source) and decrease in relation to PCB concentrations down river. Based on modeled PCB concentrations, many species are expected to be at considerable risk through the entire forecast period (1993 to 2018).

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TABLE 2-1

LOWER HUDSON ASSESSMENT ENDPOINTS, RECEPTORS, AND MEASURES

Assessment Endpoint	Specific Ecological Receptor ("Endpoint Species")	Measures	
		Exposure	Effect
Benthic aquatic life as a food source for local fish and wildlife.	· Benthic aquatic community	· Modeled PCB concentrations in sediments and water column	· Exceedance of AWQC and sediment guidelines
Survival, growth, and reproduction of local forage fish populations.	· Spottail shiner · Pumpkinseed	· Modeled PCB body burdens · Modeled PCB concentrations in sediments and water column	· Estimated exceedance of TRVs · Exceedance of AWQC and sediment guidelines · Field observations
Survival, growth, and reproduction of local piscivorous/semi-piscivorous fish populations.	· Yellow perch · White perch · Largemouth bass · Striped bass	· Modeled PCB body burdens · Modeled PCB concentrations in sediments and water column	· Estimated exceedance of TRVs · Exceedance of AWQC and sediment guidelines · Field observations
Survival, growth, and reproduction of local omnivorous fish populations.	· Shortnose sturgeon · Brown bullhead	· Modeled PCB body burdens · Modeled PCB concentrations in sediments and water column	· Estimated exceedance of TRVs · Exceedance of AWQC and sediment guidelines · Field observations
Protection (i.e., survival and reproduction) of insectivorous birds and mammals.	· Tree swallow · Little brown bat	· Modeled PCB concentrations in prey items (aquatic insects) · Modeled PCB concentrations in the water column	· Estimated exceedance of TRVs · Exceedance of AWQC for the protection of wildlife · Field observations
Protection (i.e., survival and reproduction) of waterfowl.	· Mallard	· Modeled PCB concentrations in prey (invertebrates, macrophytes) · Modeled PCB concentrations in the water column	· Estimated exceedance of TRVs · Exceedance of AWQC for the protection of wildlife · Field observations
Protection of piscivorous/semi-piscivorous birds and mammals.	· Belted kingfisher · Great blue heron · Mink · River Otter	· Modeled PCB concentrations in prey (forage fish, invertebrates) · Modeled PCB concentrations in sediments and water column	· Estimated exceedance of TRVs · Exceedance of AWQC for the protection of wildlife · Field observations
Protection of omnivorous mammals.	· Raccoon	· Modeled PCB concentrations in prey items (fish, invertebrates) · Modeled PCB concentrations in the water column	· Estimated exceedance of TRVs · Exceedance of AWQC for the protection of wildlife · Field observations
Protection of endangered and threatened species.	· Bald eagle · Shortnose sturgeon	· Modeled PCB body burdens (sturgeon) · Modeled PCB concentrations in prey (fish) · Modeled PCB concentrations in sediments and water column	· Estimated exceedance of TRVs · Exceedance of AWQC and sediment guidelines for the protection of wildlife · Field observations
Protection of significant habitats.	· Hudson River NERR · NYSDOS significant habitats	· Modeled PCB concentrations in sediments and water column	· Exceedance of federal and state AWQC and sediment guidelines · Field observations

Notes: Individual-level effects are considered to occur when the TQ is greater to or equal to one.
Receptor species are surrogates chosen to represent a wide range of species likely to use the Hudson River as habitat or foraging source.

302706

TABLE 2-2

LOWER HUDSON RIVER ENDPOINTS AND RISK HYPOTHESES

Assessment Endpoint: Benthic aquatic life as a food source for local fish and wildlife	
Do modeled total PCB water concentrations exceed criteria and/or guidelines for protection of aquatic health?	*Measurement Endpoint 1:* Modeled PCB concentrations in water (freshwater) compared to NYS Ambient Water Quality Criteria (AWQC) for the protection of benthic aquatic life (NYSDEC, 1998b).
Do modeled total PCB sediment concentrations exceed guidelines for protection of aquatic health?	*Measurement Endpoint 2:* Modeled PCB concentrations in sediment compared to applicable sediment benchmarks (e.g., NOAA Sediment Effect Concentrations for PCBs in the Hudson River [NOAA, 1999a], NYSDEC Technical Guidance for Screening Contaminated Sediments [1999a], etc.)
Assessment Endpoint: Sustainability (i.e., survival, growth, and reproduction) of Lower Hudson River Fish Populations (forage, omnivorous, piscivorous)	
Do modeled total PCB body burdens in local fish exceed benchmarks for adverse effects on fish reproduction?	*Measurement Endpoint 1:* Modeled total PCB body burdens in fish for each river segment over 25 years to determine exceedance of effect-level thresholds based on toxicity reference values (TRVs) derived in the baseline ERA (USEPA, 1999c).
Do modeled total PCB body burdens in local fish expressed on a TEQ basis exceed benchmarks for adverse effects on fish reproduction?	*Measurement Endpoint 2:* Modeled TEQ-based PCB body burdens in fish for each river segment over 25 years to determine exceedance of effect-level thresholds based on TRVs.
Do modeled total PCB water concentrations exceed criteria and/or guidelines for protection of aquatic health?	*Measurement Endpoint 3:* Modeled PCB concentrations in water (freshwater) compared to NYS Ambient Water Quality Criteria (AWQC) for the protection of benthic aquatic life (NYSDEC, 1998b).
Do modeled total PCB sediment concentrations exceed guidelines for protection of aquatic health?	*Measurement Endpoint 4:* Modeled PCB concentrations in sediment compared to applicable sediment benchmarks (e.g., NOAA Sediment Effect Concentrations for PCBs in the Hudson River [NOAA, 1999a], NYSDEC Technical Guidance for Screening Contaminated Sediments [1999a], etc.)
What do available field-based observations suggest about the health of local fish populations?	*Measurement Endpoint 5:* Available field observations on the presence and relative abundance of fish species within the Lower Hudson River as an indication of the ability of the species to maintain populations.
Assessment Endpoint: Sustainability (i.e., survival, growth, and reproduction) of Lower Hudson River Insectivorous Bird Populations (represented by the tree swallow)	
Do modeled total PCB dietary doses to insectivorous exceed benchmarks for adverse effects on reproduction?	*Measurement Endpoint 1:* Modeled total PCB body burdens in the tree swallow to determine exceedance of effect-level thresholds based on TRVs.
Do modeled TEQ-based dietary doses of PCBs to insectivorous birds exceed benchmarks for adverse effects on reproduction?	*Measurement Endpoint 2:* Modeled TEQ-based PCB body burdens in the tree swallow to determine exceedance of effect-level thresholds based on TRVs.
Do modeled total PCB concentrations in insectivorous bird eggs exceed benchmarks for adverse effects on reproduction?	*Measurement Endpoint 3:* Modeled total PCB egg concentrations in the tree swallow to determine exceedance of effect-level thresholds based on TRVs.
Do modeled TEQ-based PCB concentrations in insectivorous bird eggs exceed benchmarks for adverse effects on reproduction?	*Measurement Endpoint 4:* Modeled TEQ-based PCB egg concentrations in the trees swallow to determine exceedance of effect-level thresholds based on TRVs.

TABLE 2-2

LOWER HUDSON RIVER ENDPOINTS AND RISK HYPOTHESES

Do modeled whole water concentrations exceed criteria and/or guidelines for the protection of wildlife?	*Measurement Endpoint 5:* Modeled PCB concentrations in water (freshwater) compared to NYS AWQC for the protection of wildlife (NYSDEC, 1998b).
What do the available field-based observations suggest about the health of local insectivorous bird populations?	*Measurement Endpoint 6:* Available field observations on the presence and relative abundance of insectivorous bird species within the Lower Hudson River as an indication of the ability of the species to maintain populations.
Assessment Endpoint: Sustainability (i.e., survival, growth, and reproduction) of Lower Hudson River Waterfowl Populations (represented by the mallard)	
Do modeled total PCB dietary doses to waterfowl exceed benchmarks for adverse effects on reproduction?	*Measurement Endpoint 1:* Modeled total PCB body burdens in the mallard to determine exceedance of effect-level thresholds based on TRVs.
Do modeled TEQ-based dietary doses of PCBs to waterfowl exceed benchmarks for adverse effects on reproduction?	*Measurement Endpoint 2:* Modeled TEQ-based PCB body burdens in the mallard to determine exceedance of effect-level thresholds based on TRVs.
Do modeled total PCB concentrations in insectivorous bird eggs exceed benchmarks for adverse effects on reproduction?	*Measurement Endpoint 3:* Modeled total PCB egg concentrations in the tree swallow to determine exceedance of effect-level thresholds based on TRVs.
Do modeled TEQ-based PCB concentrations in waterfowl eggs exceed benchmarks for adverse effects on reproduction?	*Measurement Endpoint 4:* Modeled TEQ-based PCB egg concentrations in the mallard to determine exceedance of effect-level thresholds based on TRVs.
Do modeled whole water concentrations exceed criteria and/or guidelines for the protection of wildlife?	*Measurement Endpoint 5:* Modeled PCB concentrations in water (freshwater) compared to NYS AWQC for the protection of wildlife (NYSDEC, 1998b).
What do the available field-based observations suggest about the health of local waterfowl populations?	*Measurement Endpoint 6:* Available field observations on the presence and relative abundance of waterfowl along the Lower Hudson River as an indication of the ability of the species to maintain populations.
Assessment Endpoint: Sustainability (i.e., survival, growth, and reproduction) of Hudson River Piscivorous Bird Populations (represented by the bald eagle, great blue heron, and belted kingfisher)	
Do modeled total PCB dietary doses to piscivorous birds exceed benchmarks for adverse effects on reproduction?	*Measurement Endpoint 1:* Modeled total PCB body burdens in receptor species (i.e., bald eagle, great blue heron, and belted kingfisher) over 25 years to determine exceedance of effect-level thresholds based on TRVs.
Do modeled TEQ-based dietary doses of PCBs to piscivorous birds exceed benchmarks for adverse effects on reproduction?	*Measurement Endpoint 2:* Modeled TEQ-based PCB body burdens in receptor species for each river segment over 25 years to determine exceedance of effect-level thresholds based on TRVs.
Do modeled total PCB concentrations in piscivorous bird eggs exceed benchmarks for adverse effects on reproduction?	*Measurement Endpoint 3:* Modeled total PCB egg concentrations in receptor species to determine exceedance of effect-level thresholds based on TRVs.
Do modeled TEQ-based PCB concentrations in piscivorous bird eggs exceed benchmarks for adverse effects on reproduction?	*Measurement Endpoint 4:* Modeled TEQ-based PCB egg concentrations in receptor species to determine exceedance of effect-level thresholds based on TRVs.
Do modeled whole water concentrations exceed criteria and/or guidelines for the protection of wildlife?	*Measurement Endpoint 5:* Modeled PCB concentrations in water (freshwater and saline) compared to NYS AWQC for the protection of wildlife (NYSDEC, 1998b).
What do the available field-based observations suggest about the health of local piscivorous bird populations?	*Measurement Endpoint6:* Available field observations on the presence and relative abundance of piscivorous birds along the Lower Hudson River as an indication of the ability of the species to maintain populations.

TABLE 2-2

LOWER HUDSON RIVER ENDPOINTS AND RISK HYPOTHESES

Assessment Endpoint: Sustainability (i.e., survival, growth, and reproduction) of Lower Hudson River Insectivorous Mammals (as represented by the little brown bat)	
Do modeled total PCB dietary doses to local wildlife species exceed benchmarks for adverse effects on reproduction?	*Measurement Endpoint 1*: Modeled total PCB body burdens in the wildlife species to determine exceedance of effect-levels based on TRVs.
Do modeled TEQ-based PCB dietary doses to local wildlife species exceed benchmarks for adverse effects on reproduction?	*Measurement Endpoint 2*: Measured and modeled TEQ-based PCB body burdens in the little brown bat to determine exceedance of effect-level thresholds based on TRVs.
Do modeled whole water concentrations exceed criteria and/or guidelines for the protection of wildlife?	*Measurement Endpoint 3:* Modeled PCB concentrations in water (freshwater and saline) compared to NYS AWQC for the protection of wildlife (NYSDEC, 1999a).
What do the available field-based observations suggest about the health of local wildlife populations?	*Measurement Endpoint 4:* Available field observations on the presence and relative abundance of insectivorous species along the Lower Hudson River as an indication of the ability of the species to maintain populations.
Assessment Endpoint: Sustainability (i.e., survival, growth, and reproduction) of Hudson River Omnivorous Mammals (as represented by the raccoon)	
Do modeled total PCB dietary doses to local wildlife species exceed benchmarks for adverse effects on reproduction?	*Measurement Endpoint 1*: Modeled total PCB body burdens in the raccoon to determine exceedance of effect-levels based on TRVs.
Do modeled TEQ-based PCB dietary doses to local wildlife species exceed benchmarks for adverse effects on reproduction?	*Measurement Endpoint 2*: Measured and modeled TEQ-based PCB body burdens in the raccoon to determine exceedance of effect-level thresholds based on TRVs.
Do modeled whole water concentrations exceed criteria and/or guidelines for the protection of wildlife?	*Measurement Endpoint 3:* Modeled PCB concentrations in water (freshwater and saline) compared to NYS AWQC for the protection of wildlife (NYSDEC, 1999a).
What do the available field-based observations suggest about the health of local wildlife populations?	*Measurement Endpoint 4:* Available field observations on the presence and relative abundance of omnivorous mammals along the Lower Hudson River as an indication of the ability of the species to maintain populations.
Assessment Endpoint: Sustainability (i.e., survival, growth, and reproduction) of Lower Hudson River Piscivorous Wildlife (as represented by the mink and river otter)	
Do modeled total PCB dietary doses to local wildlife species exceed benchmarks for adverse effects on reproduction?	*Measurement Endpoint 1*: Modeled total PCB body burdens in the wildlife species to determine exceedance of effect-levels based on TRVs.
Do modeled TEQ-based PCB dietary doses to local wildlife species exceed benchmarks for adverse effects on reproduction?	*Measurement Endpoint 2*: Measured and modeled TEQ-based PCB body burdens in the wildlife species for each river segment over 25 years to determine exceedance of effect-level thresholds based on TRVs.
Do modeled whole water concentrations exceed criteria and/or guidelines for the protection of wildlife?	*Measurement Endpoint 3:* Modeled PCB concentrations in water (freshwater and saline) compared to NYS AWQC for the protection of wildlife (NYSDEC, 1999a).
What do the available field-based observations suggest about the health of local wildlife populations?	*Measurement Endpoint 4:* Available field observations on the presence and relative abundance of the wildlife species along the Hudson River as an indication of the ability of the species to maintain populations.
Assessment Endpoint: Protection of Threatened and Endangered Species	
Do modeled total PCB body burdens in local threatened or endangered species exceed benchmarks for adverse effects on reproduction?	Measurement Endpoint 1: Modeled total PCB body burdens in shortnose sturgeon (using surrogate upper trophic level fish species) and the bald eagle to determine exceedance of effect-level thresholds based on TRVs.

TABLE 2-2

LOWER HUDSON RIVER ENDPOINTS AND RISK HYPOTHESES

Do modeled TEQ-based PCB body burdens in local threatened or endangered species exceed benchmarks for adverse effects on reproduction?	*Measurement Endpoint 2:* Modeled TEQ-based PCB body burdens in shortnose sturgeon (using surrogate upper trophic level fish species) and the bald eagle to determine exceedance of effect-level thresholds based on TRVs.
Do modeled whole water concentrations exceed criteria and/or guidelines for the protection of wildlife?	*Measurement Endpoint 3:* Modeled PCB concentrations in water (freshwater and saline) compared to NYS AWQC for the protection of wildlife (NYSDEC, 1998b).
Do modeled sediment PCB concentrations exceed guidelines for the protection of aquatic health?	*Measurement Endpoint 4*: Modeled PCB concentrations in sediment compared to applicable sediment benchmarks (e.g., NOAA, 1999a, NYSDEC 1999, etc.)
What do the available field-based observations suggest about the health of local wildlife populations?	*Measurement Endpoint 5*: Available field observations on the presence and relative abundance of threatened and endangered species along the Lower Hudson River as an indication of the ability of the species to maintain populations.
Assessment Endpoint: Protection of Significant Habitats	
Do modeled toxicity quotients in local receptor species exceed benchmarks for adverse effects on reproduction?	*Measurement Endpoint 1*: Modeled total PCB and TEQ-based PCB body burdens in receptor species to determine exceedance of effect-level thresholds based on TRVs.
Do modeled whole water concentrations exceed criteria and/or guidelines for the protection of wildlife?	*Measurement Endpoint 2:* Modeled PCB concentrations in water (freshwater and saline) compared to NYS AWQC for the protection of benthic aquatic life (NYSDEC, 1998b) or wildlife (NYSDEC, 1998b).
Do modeled sediment PCB concentrations exceed guidelines for the protection of aquatic health?	*Measurement Endpoint 3:* Modeled PCB concentrations in sediment compared to applicable sediment benchmarks (e.g., NOAA, 1999a, NYSDEC 1999a, etc.).
What do the available field-based observations suggest about the health of local wildlife populations?	*Measurement Endpoint 4:* Available field observations on the presence and relative abundance of the wildlife species using significant habitats along the Hudson River as an indication of the ability of the habitat to maintain populations.
Note: Effect level-concentrations are measured by TRVs. Toxicity quotients are exceeded when the modeled dose or concentration is greater than the benchmark dose or concentration (i.e., toxicity quotient [TQ] exceeds 1). Calculation of the modeled dose and selection of the benchmark dose are covered in the baseline ERA (USEPA, 1999c).	

TABLE 2-3

LOWER HUDSON RIVER SIGNIFICANT HABITATS

Site Name	County	Community Types	Rare Species	Valuable Species
Freshwater Habitats				
Normans Kill	Albany	Freshwater creek with shallows associated with creek mouth.	None identified.	Spawning area for anadromous fish species including alewife, white perch, and blueback herring. Large resident smallmouth bass populations.
Shad and Schermerhorn Island	Albany	Largely comprised of shallows and mudflats with lesser amounts of lower marsh, upper marsh and freshwater creek.	Heart leaf plantain and estuary beggar ticks.	Large feeding areas for herons and other wading birds, furbearers, deer and other upland game, limited waterfowl usage, important spawning and nursery grounds for American shad, blueback herring, alewife, white perch, striped bass, and resident fish species.
Papascanee Marsh and Creek	Renssalear	Mainly upper marsh with lesser amounts of shallows, mudflats, lower marsh, and freshwater creek.	Least bittern nesting area; map turtles.	Waterfowl use during migrations. Breeding birds incl. green-backed heron, Virginia rail, several duck species, marsh wren, swamp swallow, and others. Spawning and nursery grounds for American shad, blueback herring, alewife, white catfish, black bass, white perch and other fish.
Schodack and Houghtaling Islands and Schodack Creek	Renssalear, Columbia, Greene	Predominantly shallows, mudflats, and sandy beach with lesser amounts of lower marsh and upper marsh.	Osprey roosting and feeding; possible use by shortnose sturgeon; heart leaf plantain.	Waterfowl use during migrations and limited nesting activity, nesting by other bird species. Furbearers present. Schodack Creek provides important spawning and nursery grounds for American Shad, white perch, alewife, and blueback herring, black bass and other species. Northmost concentration of shad spawning on the Hudson.

TABLE 2-3

LOWER HUDSON RIVER SIGNIFICANT HABITATS

Site Name	County	Community Types	Rare Species	Valuable Species
Coeymans Creek	Albany	Predominantly shallows with smaller amounts of mudflats, lower marsh, and swamp forest.	None.	Important spawning area for anadromous fish including alewife, blueback herring, white perch, and American Shad. Limited waterfowl during migrations.
Hannacroix Creek	Albany, Greene	Predominantly freshwater creek with shallows, mudflats, lower marsh, upper marsh and swamp forest.	None identified.	Important spawning area for alewife, blueback herring, white perch, American Shad, and other fish. Resting and feeding area for migratory waterfowl. Feeding area for herons, various birds, and furbearers.
Mill Creek Wetlands	Columbia	Swamp forest with some shallows, mudflats, sandy beach, lower marsh, and upper marsh.	Estuary beggar ticks.	Limited waterfowl use during migrations. Populations of breeding birds include green-backed herons, various ducks, and many passerines.
Stuyvesant Marshes*	Columbia	Roughly equal amounts of shallows, mudflats, sandy mudflats, sandy beach, rocky shore, lower marsh, and upper marsh.	Heart leaf plantain, kidney leaf mud plantain.	Limited use by migrating waterfowl, probable heavy use by various nesting bird species.
Coxsackie Creek	Greene	Principally freshwater creek with some shallows, mudflats, sandy beach, lower marsh, upper marsh, and freshwater creek.	Estuary beggar ticks.	Spawning habitat for alewife, blueback herring, white perch, and American shad. Feeding grounds for herons and other wading birds. Small mammal and furbearer foraging.

TABLE 2-3

LOWER HUDSON RIVER SIGNIFICANT HABITATS

Site Name	County	Community Types	Rare Species	Valuable Species
Coxsackie Island Backwater	Greene	Shallows with peripheral mud and sand flats, rocky shore, lower marsh, and upper marsh.	Heart leaf plantain, kidney leaf mud plantain.	Important spawning and nursery ground for resident fish including brown bullhead, largemouth bass, yellow perch, and redfin pickerel. Also feeding grounds for anadromous fish and wintering areas for largemouth bass.
Stockport Creek and Flats	Columbia	Shallows and mudflats with substantial areas of lower marsh, upper marsh, and woody swamp. Three miles of tidal and freshwater creek. Some deepwater and sandy beach associated with navigation channel and islands.	Heart leaf plantain, estuary beggar ticks, golden club; map turtle.	Very important spawning/nursery grounds for anadromous and freshwater fish including alewife, blueback herring, smelt, American shad, striped bass, and smallmouth bass. Very important feeding and resting habitat for migrating and overwintering waterfowl. Use by wading, shore, and passerine birds for feeding and breeding. Bank swallows nest in the vertical sand banks. Extensive stands of wild rice.
Vosburgh Swamp and Middle Ground Flats	Greene	Largely comprised of creek, deepwater, shallows, and mudflats with lesser amounts of sandy beach, lower marsh, upper marsh, and freshwater swamp.	Possible least bittern and mud turtle; heart leaf plantain, sublate arrowhead, estuary beggar ticks.	Important feeding and resting grounds for migrating waterfowl and wintering waterfowl (when open water is available). Extensive nesting area for ducks, green-backed herons, and other birds. Colony of bank swallows. Heavy use of shallows for American shad spawning and extensive spawning, nursery and feeding areas for striped bass, alewife, blueback herring and resident fish species.
Roger's Island	Columbia	Comprised of roughly equal amounts of shallows and mudflats with some sandy beach, lower marsh, upper marsh, and swamp forest.	Estuary beggar-ticks, goldenclub.	Extensive waterfowl use during migrations and overwintering, nesting sites for many birds, extensive spawning areas for anadromous fish including the American shad.

TABLE 2-3

LOWER HUDSON RIVER SIGNIFICANT HABITATS

Site Name	County	Community Types	Rare Species	Valuable Species
Catskill Creek	Greene	Predominantly creek with small amounts of shallows, mudflats, and lower marsh.	Wood turtle, probably in association with buffer area.	Important spawning and nursery grounds for anadromous and resident fishes including American shad, alewife, blueback herring, white perch, smallmouth and largemouth bass.
Ramshorn Marsh	Greene	Largely shallows, mudflats, lower marsh, upper marsh, and swamp forest with lesser amounts of sandy beach and rocky shore.	Least bittern nesting; estuary beggar-ticks, and heart leaf plantain.	Waterfowl use during migrations and overwintering, important heron feeding grounds, furbearer habitat, spawning and nursery grounds for American shad and black bass.
Inbocht Bay and Duck Cove	Greene	Principally shallows and mudflats with some lower marsh.	Estuary beggar-ticks.	Very extensive waterfowl concentrations during spring and fall migrations, some waterfowl overwintering, large muskrat and snapping turtle populations.
Roeliff-Jansen Kill	Columbia	Predominantly freshwater creek with limited shallows, mudflats, and lower marsh.	None identified.	Extensive use as a spawning/nursery ground for anadromous fish including American shad, blueback herring, white perch, and striped bass. Resident brown trout in upper reaches.
Smith's Landing Cementon*	Greene, Ulster	Limited mudflats, lower marsh, and upper marsh.	Heart leaf plantain, kidney leaf mud-plantain.	None identified.
Germantown/Clermont Flats	Columbia	Deepwater, shallows, mudflats, and limited lower marsh.	None identified.	Extremely important American shad spawning area, nursery areas for shad, striped bass, white perch, and resident fish. Extensive waterfowl feeding grounds during spring and fall migration periods. Some waterfowl overwintering.

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TABLE 2-3

LOWER HUDSON RIVER SIGNIFICANT HABITATS

Site Name	County	Community Types	Rare Species	Valuable Species
Esopus Estuary	Ulster, Dutchess	Comprised of freshwater creek, deepwater, shallows, mudflats, lower marsh, upper marsh, and a small amount of tidal swamp.	Shortnose sturgeon spawning and wintering area in deepwater; migrating osprey feeding grounds; heart leaf plantain, goldenclub.	Important spawning and nursery grounds for striped bass, white perch, American shad, alewife, blueback herring, rainbow smelt, and resident fish. Feeding and resting grounds for migrating waterfowl.
North and South Tivoli Bays	Dutchess	Comprised of shallows, lower marsh, and upper marsh, followed by tidal swamp forest, rocky shore and creeks.	Migrating osprey feeding and resting, least bittern nesting, king rail; map turtles; heart leaf plantain, estuary beggar-ticks, goldenclub and other rare plants.	Feeding, spawning and/or nursery areas for striped bass, alewife, blueback herring, largemouth and smallmouth bass, and other fishes. Large snapping turtle population. Extensive waterfowl use for feeding and resting during migrations. Many breeding birds. Furbearer habitat.
Mudder Kill*	Dutchess	Equal amounts of mudflats, lower marsh, upper marsh, and tidal swamp forest.	Goldenclub, hirsute sedge, Davis sedge, heavy sedge, kidney leaf mud-plantain, and spongy arrowhead.	None known.
The Flats	Ulster, Dutchess	Comprised entirely of shallows.	Potential shortnose sturgeon feeding and resting area.	Primary spawning grounds for American shad and spawning and nursery area for striped bass, white perch, and resident fishes. Feeding area during migration periods for diving ducks and resting areas for all duck species.

302715

TABLE 2-3

LOWER HUDSON RIVER SIGNIFICANT HABITATS

Site Name	County	Community Types	Rare Species	Valuable Species
Roundout Creek	Ulster	Predominantly creek with shallows, mudflats, rocky shore, lower marsh, and limited amounts of upper marsh in association with the creek mouth.	Osprey during migration; heart leaf plantain.	Important spawning area for anadromous fish including alewife, rainbow smelt, blueback herring, white perch, tomcod, striped bass, and American shad. Important for resident fish such as brown bullhead, yellow perch, sunfish, and black basses. Limited use by migrating waterfowl for resting and feeding, extensive feeding on mudflats by herons and other wading birds.
Kingston Deepwater Habitat	Dutchess, Ulster	Deepwater.	Shortnose sturgeon wintering area and possible spawning grounds.	Atlantic sturgeon wintering area, the northern extent of many marine fishes in the Hudson.
Vanderburgh Cove and Shallows	Dutchess	Largely shallows with smaller amounts of mudflats, lower marsh, upper marsh, tidal swamp, and freshwater creek.	Possible shortnose sturgeon feeding grounds, osprey feeding ground during migration, sharp-winged monkey flower.	Extensive waterfowl feeding and resting grounds during spring and fall migrations. Important spawning, nursery, and feeding grounds for anadromous fish (striped bass, American shad, white perch, rainbow smelt, alewife, blueback herring) and resident fish (largemouth bass, yellow perch, brown bullhead).
Esopus Meadows	Ulster	Shallows.	Important feeding area for shortnose sturgeon, especially in spring.	Spawning, nursery, and feeding grounds for anadromous fish (e.g., striped bass, American shad, and white perch) and resident fish (e.g., largemouth bass, yellow perch, brown bullhead, and shiners).
Poughkeepsie Deepwater Habitat	Dutchess, Ulster	Deepwater.	Shortnose sturgeon wintering area and possible nursery grounds.	Estuarine and marine fish including bay anchovies, silversides, bluefish, weakfish, and hogchokers.

TABLE 2-3

LOWER HUDSON RIVER SIGNIFICANT HABITATS

Site Name	County	Community Types	Rare Species	Valuable Species
Crum Elbow Marsh*	Dutchess	Small amount of shallows, lower marsh, upper marsh, and tidal swamp forest.	Map turtle population.	Waterfowl migration, value limited by size of the marsh.
Brackish Water Habitats				
Wappinger Creek	Dutchess	Predominantly creek with smaller amounts of shallows, mudflats, lower marsh, and upper marsh.	Osprey feeding during spring migrations. Grassleaf arrowhead, subulate arrowhead, kidney leaf mud plaintain and Maryland bur-marigold.	Important spawning areas for anadromous fish including alewife, blueback herring, white perch, tomcod, and striped bass. Resident fish include largemouth bass, bluegill, brown bullhead, and red-breasted sunfish. Productive area for herons, waterfowl, and turtles.
Fishkill Creek	Dutchess	Mostly shallows and wooded upland with smaller amounts of mudflats, lower marsh, and upper marsh.	Important feeding site for migrating osprey and a potential osprey nesting site. Least bittern breeding. Estuary beggar-ticks, subulate arrowhead, kidney leaf mud- plantain.	Important spawning areas for anadromous fish including alewife, blueback herring, white perch, tomcod, and striped bass. Resident fish include largemouth bass, bluegill, brown bullhead, and red-breasted sunfish. Also blue claw crabs, herons and turtles.
Moodna Creek	Orange	Predominantly freshwater creek with shallows, mudflats, lower marsh, and upper marsh associated with the creek mouth.	Major feeding and resting ground for bald eagles and osprey. Limited summer feeding ground for bald eagles. Least bittern breeding area.	Important spawning areas for anadromous fish including alewife, blueback herring, smelt, white perch, tomcod, and striped bass. Resident fish include largemouth bass, bluegill, brown bullhead, and pumpkinseed. Also many herons, snapping turtles, raccoons, and muskrats.

TABLE 2-3

LOWER HUDSON RIVER SIGNIFICANT HABITATS

Site Name	County	Community Types	Rare Species	Valuable Species
Hudson River Miles 44-56	Orange, Rockland, Putnam, Westchester	Deepwater, shallows, and forested uplands.	Bald eagle winter feeding grounds. Possible nursery area for shortnose sturgeon.	The major spawning area along the Hudson for striped bass and white perch (about 50% of northeast striped bass stocks come from the Hudson). Narrow migration corridor for all anadromous fish spawning upriver. Marine species (e.g., bluefish, bay anchovy) live here during periods of low freshwater flow (generally July through February).
Constitution Marsh	Putnam	Approximately equal amounts of shallows, mudflats, lower marsh, and upper marsh.	Least bittern nesting site. Osprey use during migrations.	Very important nesting habitat for a variety of bird species including green-backed heron, various waterfowl, and passerine birds. Important feeding grounds for herons and other wetland and shore birds. Significant spawning and feeding grounds for anadromous and resident fish. Muskrat population.
Iona Island Marsh	Rockland	Mainly upper marsh, followed by shallows and flats, with lesser amounts of woody tidal swamp and non-tidal freshwater marsh.	Least bittern nesting, adjacent bald eagle winter roosting. Walking fern and prickly pear cactus.	Extensive breeding for many birds. Muskrat and possibly other furbearers, amphibians, snapping turtle, and blue claw crab. Heron and shorebird feeding. Spawning and/or nursery for anadromous and resident fish.
Camp Smith Marsh and Annsville Creek*	Westchester	Largely shallows and creek with smaller amounts of mudflats and upper marsh.	Spongy arrowhead.	None identified.
Salt Water Habitats				

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TABLE 2-3

LOWER HUDSON RIVER SIGNIFICANT HABITATS

Site Name	County	Community Types	Rare Species	Valuable Species
Haverstraw Bay	Rockland, Westchester	Deepwater and shallows.	Shortnose sturgeon wintering area.	Extensive nursery for anadromous fish species. Nursery and feeding ground for marine species. Spawning and wintering grounds for Atlantic sturgeon. Waterfowl feeding and resting during migration.
Croton River and Bay	Westchester	Mostly shallows with lesser amounts of mudflats and brackish upper marsh.	Possible osprey feeding grounds during spring and fall migrations.	Productive nursery, foraging and resting area for anadromous and resident fish.
Piermont Marsh	Rockland	Predominantly shallows and brackish upper marsh with a broad transition area of mudflats.	Least bittern and sedgewren nesting. Diamondback turtle use. Osprey feeding during migration.	Extensive use of mudflats by herons and egrets. Large numbers of resident and breeding birds, blue claw crabs, resident fish, and lesser numbers of furbearers. Waterfowl, wading bird, and shorebird feeding during migration.

Notes: * Indicates areas recognized by the NYS Natural Heritage Program as containing rare/important species or communities, but not designated as significant habitats.
Source: NYSDOS and the Nature Conservancy, 1990.

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Table 3-1 Summary of Conversion for the Di through Hexa Homologues

Homologue	Period	Mean Mass Percent of Tri+ Using TID Data	Mean +2 Standard Errors	Mean -2 Standard Errors	Mean Mass Percent Ratio Waterford/TID	Corrected TID Mass Percent	Mass Percent of Tri+ at Waterford
Calibration Period							
Di-Hexa	1987-1990		Repeat the 1991 Distribution				
Di	High Flow 1991-1995	32.17	36.28	28.07	1.04	33.37	33.37
Di	Low Flow 1991-1995	48.40	53.02	43.78	0.52	25.41	25.41
Di	High Flow 1996-1998	70.64	76.69	64.60	1.04	73.27	73.27
Di	Low Flow 1996-1998	96.46	102.16	90.76	0.52	50.64	50.64
Tri-Hexa	Fall-winter 1991-1998		GE TID Data		Same as below by homologue.	Varies	Varies
Tri-Hexa	Spring 1991-1998		GE TID Data		"	Varies	Varies
Tri-Hexa	Summer 1991-1998		GE TID Data		"	Varies	Varies
Forecast Period							
Di	High Flow 1999+	70.64	76.69	64.60	1.04	73.27	73.27
Di	Low Flow 1999+	96.46	102.16	90.76	0.52	50.64	50.64
Tri	Fall-winter 1999+	47.21	48.82	45.60	0.98	46.11	44.97
Tri	Spring 1999+	45.90	47.71	44.09	0.98	44.83	44.06
Tri	Summer 1999+	54.30	55.12	53.48	0.91	49.18	48.08
Tetra	Fall-winter 1999+	29.66	30.51	28.81	0.97	28.76	28.05
Tetra	Spring 1999+	34.41	35.55	33.26	0.97	33.36	32.79
Tetra	Summer 1999+	30.12	30.55	29.69	1.09	32.81	32.08
Penta	Fall-winter 1999+	18.10	19.22	16.98	1.19	21.49	20.96
Penta	Spring 1999+	15.65	16.88	14.41	1.19	18.58	18.26
Penta	Summer 1999+	12.95	13.54	12.37	1.28	16.64	16.27
Hexa	Fall-winter 1999+	5.00	5.58	4.42	1.23	6.15	6.00
Hexa	Spring 1999+	4.04	4.61	3.48	1.23	4.97	4.89
Hexa	Summer 1999+	2.62	2.82	2.41	1.39	3.64	3.56
Tri-Hexa	Fall-winter 1999+	99.97				102.50	99.97
Tri-Hexa	Spring 1999+	100.00				101.74	100.00
Tri-Hexa	Summer 1999+	99.99				102.26	99.99

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Table 3-2
Ratio of Striped Bass to Largemouth Bass Concentrations

RM 152

Year	STB Tri + ppm	LMB Tri+ ppm	WP Tri+ ppm	STB/LMB	STB/WP
1990	9.02	3.53	0.84	2.56	10.68
1991	NA	NA	NA		
1992	15.32	3.24	8.64	4.73	1.77
1993	10.92	9.34	5.45	1.17	2
1995	NA	NA	NA		
1994	5.61	NA	4.81		1.16
1996	4.28	2.51	2.78	1.71	1.54
			Average --->>>	2.54	3.43

RM 152 Monthly Averages

	LMB	Striped Bass			STB/LMB		
Year	June	June-Aug	June-July	June Only	June-Aug	June-July	June Only
1990	3.53	9.02	9.39	4.95	3.55	3.70	1.95
1992	3.24	15.32	15.32	15.32	6.03	6.03	6.03
1993	9.34	11.38	11.38	11.37	4.48	4.48	4.47
1996	2.51	4.28	4.28	2.78	1.69	1.69	1.09
				Average	2.55	2.58	2.58

RM 113

Year	LMB Tri+ ppm	WP Tri+ ppm	STB Tri+ ppm	STB/LMB	STB/WP
1988	7.71	NA	6.31	0.82	
1989	NA	NA	NA		
1990	7.84	NA	4.64	0.59	
1991	NA	NA	NA		
1992	8.28	NA	2.94	0.35	
1993	4.45	3.25	3.27	0.74	1.01
1994	6.26	1.04	2.3	0.37	2.21
1995	3.27	1.86	1.11	0.34	0.6
1996	3.73	4.94	1.66	0.45	0.34
			Average --->>>	0.52	1.04

Note:

STB : Striped Bass; WP: White Perch; LMB: Large Mouth Bass.
NA: Data is not available.

Table 3-3
Sum of Monthly Average Loads Over the Troy Dam
(kg)

Homologue	Thomann/Farley Model	HUDTOX Converted According to Appendix A	Difference
Di	1182	2077	895
Tri	2320	2421	101
Tetra	1664	1599	-65
Penta	715	742	27
Hexa	270	251	-18
Total 1987-199	6151	7091	939

Homologue	Thomann/Farley Model	HUDTOX Converted According to Appendix A	DEIR
Di	857	566	540
Tri	1645	856	1180
Tetra	1081	593	860
Total 4/91-2/96	3583	2015	2580

Table 3-4a
Relative Percent Difference Between FISHRAND Results and Measured Fish Levels in the Lower Hudson

	Species										
	Largemouth Bass		Brown Bullhead	White Perch			Yellow Perch			Pumpkinseed	
River Mile	152	113	152	152	152 (seasonal)	113	152	152 (seasonal)	113	142	60
Year											
1987											18%
1988	67%	17%	-12%							-28%	25%
1989										-30%	-2%
1990	-5%	-29%									9%
1991				100%							-7%
1992	-21%	-39%	-31%	-67%			-62%				
1993	-64%	39%	-22%	-28%		-13%	-21%		-16%	100%	77%
1994		-10%		-41%		137%			43%	-38%	-40%
1995	-38%	12%	-35%	-50%		14%	-46%			-55%	
spring					-52%			-32%			
fall					-48%			-60%			
1996	-29%	-2%	-21%	20%		-46%				17%	-10%
Mean	-15%	-2%	-24%	-11%		23%	-43%		14%	-6%	9%
Std Deviation	45%	27%	9%	62%		80%	21%		42%	57%	34%
Std Error	18%	10%	4%	25%		40%	12%		30%	23%	12%
Mean + 2 std errors	21%	19%	-16%	40%		103%	-19%		73%	41%	33%
Mean - 2 std errors	-51%	-22%	-32%	-62%		-57%	-67%		-46%	-52%	-15%

Average RPD -6%

Note:
RPD = (Predicted Median Concentration - Observed Median Concentration)/Observed Median Concentration
Concentrations are all wet weight concentrations.

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Table 3-4b
Relative Percent Difference Between FISHRAND Results and Measured Spottail Shiner Levels in the Lower Hudson

Location (RM)		
Model	Measurement	RPD
60	58.7	-22%
90	88.9	-27%
113	113.8	-65%
152	143.5	5%
	Mean RPD	-27%

Note:
RPD = (Predicted Median Concentration - Observed Median Concentration)/Observed Median Concentration
Concentrations are all wet weight concentrations.

TABLE 3-5: SUMMARY OF TRI+ WHOLE WATER CONCENTRATIONS FROM THE FARLEY MODEL AND TEQ-BASED PREDICTIONS FOR 1993 - 2018

	Tri+ Average PCB Results				Tri+ 95% UCL Results				Average Avian TEF				95% Avian TEF				Average Mammalian TEF				95% UCL Mammalian TEF			
Year	152 Whole Water Conc mg/l	113 Whole Water Conc mg/l	90 Whole Water Conc mg/l	50 Whole Water Conc mg/l	152 Whole Water Conc mg/l	113 Whole Water Conc mg/l	90 Whole Water Conc mg/l	50 Whole Water Conc mg/l	152 Whole Water Conc mg/l	113 Whole Water Conc mg/l	90 Whole Water Conc mg/l	50 Whole Water Conc mg/l	152 Whole Water Conc mg/l	113 Whole Water Conc mg/l	90 Whole Water Conc mg/l	50 Whole Water Conc mg/l	152 Whole Water Conc mg/l	113 Whole Water Conc mg/l	90 Whole Water Conc mg/l	50 Whole Water Conc mg/l	152 Whole Water Conc mg/l	113 Whole Water Conc mg/l	90 Whole Water Conc mg/l	50 Whole Water Conc mg/l
1993	4.4E-05	3.0E-05	2.3E-05	1.8E-05	6.1E-05	3.8E-05	2.8E-05	2.2E-05	3.7E-08	2.6E-08	2.0E-08	1.6E-08	5.2E-08	3.2E-08	2.4E-08	1.9E-08	2.9E-08	2.0E-08	1.5E-08	1.2E-08	4.0E-08	2.5E-08	1.8E-08	1.4E-08
1994	4.0E-05	2.6E-05	2.0E-05	1.6E-05	4.9E-05	3.1E-05	2.4E-05	1.9E-05	3.4E-08	2.2E-08	1.7E-08	1.4E-08	4.2E-08	2.6E-08	2.0E-08	1.6E-08	2.6E-08	1.7E-08	1.3E-08	1.0E-08	3.2E-08	2.0E-08	1.6E-08	1.2E-08
1995	1.6E-05	1.6E-05	1.6E-05	1.4E-05	1.8E-05	1.9E-05	1.9E-05	1.6E-05	1.4E-08	1.4E-08	1.4E-08	1.2E-08	1.5E-08	1.6E-08	1.6E-08	1.4E-08	1.1E-08	1.1E-08	1.0E-08	9.0E-09	1.2E-08	1.2E-08	1.2E-08	1.1E-08
1996	4.7E-05	2.6E-05	1.8E-05	1.3E-05	6.9E-05	3.2E-05	2.1E-05	1.6E-05	4.0E-08	2.2E-08	1.5E-08	1.1E-08	5.9E-08	2.7E-08	1.8E-08	1.3E-08	3.1E-08	1.7E-08	1.2E-08	8.7E-09	4.5E-08	2.1E-08	1.4E-08	1.0E-08
1997	3.1E-05	2.1E-05	1.6E-05	1.2E-05	4.0E-05	2.5E-05	1.9E-05	1.5E-05	2.6E-08	1.8E-08	1.4E-08	1.1E-08	3.4E-08	2.2E-08	1.6E-08	1.3E-08	2.0E-08	1.4E-08	1.0E-08	8.1E-09	2.6E-08	1.7E-08	1.2E-08	9.7E-09
1998	1.8E-05	1.5E-05	1.3E-05	1.1E-05	2.0E-05	1.8E-05	1.6E-05	1.3E-05	1.6E-08	1.3E-08	1.1E-08	9.3E-09	1.7E-08	1.5E-08	1.4E-08	1.1E-08	1.2E-08	1.0E-08	8.8E-09	7.1E-09	1.3E-08	1.1E-08	1.0E-08	8.5E-09
1999	1.6E-05	1.3E-05	1.1E-05	9.7E-06	1.7E-05	1.5E-05	1.4E-05	1.1E-05	1.3E-08	1.1E-08	9.8E-09	8.2E-09	1.5E-08	1.2E-08	1.2E-08	9.8E-09	1.0E-08	8.3E-09	7.5E-09	6.3E-09	1.1E-08	9.5E-09	8.9E-09	7.5E-09
2000	2.6E-05	1.5E-05	1.1E-05	9.0E-06	3.1E-05	1.8E-05	1.3E-05	1.1E-05	2.2E-08	1.3E-08	9.7E-09	7.7E-09	2.6E-08	1.5E-08	1.1E-08	9.1E-09	1.7E-08	1.0E-08	7.4E-09	5.9E-09	2.0E-08	1.2E-08	8.7E-09	7.0E-09
2001	2.9E-05	1.7E-05	1.2E-05	8.7E-06	4.0E-05	2.1E-05	1.4E-05	1.0E-05	2.4E-08	1.5E-08	9.8E-09	7.4E-09	3.4E-08	1.8E-08	1.2E-08	8.8E-09	1.9E-08	1.1E-08	7.6E-09	5.7E-09	2.6E-08	1.4E-08	9.0E-09	6.8E-09
2002	1.7E-05	1.3E-05	1.0E-05	8.0E-06	2.0E-05	1.5E-05	1.2E-05	9.6E-06	1.4E-08	1.1E-08	8.7E-09	6.8E-09	1.7E-08	1.3E-08	1.0E-08	8.2E-09	1.1E-08	8.3E-09	6.6E-09	5.2E-09	1.3E-08	9.9E-09	8.0E-09	6.3E-09
2003	1.9E-05	1.3E-05	9.7E-06	7.5E-06	2.5E-05	1.5E-05	1.2E-05	9.0E-06	1.6E-08	1.1E-08	8.3E-09	6.4E-09	2.1E-08	1.3E-08	9.9E-09	7.6E-09	1.2E-08	8.3E-09	6.3E-09	4.9E-09	1.6E-08	1.0E-08	7.6E-09	5.9E-09
2004	1.0E-05	8.6E-06	7.8E-06	6.5E-06	1.1E-05	9.8E-06	9.3E-06	7.8E-06	8.6E-09	7.3E-09	6.7E-09	5.6E-09	9.5E-09	8.4E-09	7.9E-09	6.6E-09	6.6E-09	5.6E-09	5.1E-09	4.3E-09	7.3E-09	6.4E-09	6.1E-09	5.1E-09
2005	1.4E-05	9.1E-06	7.2E-06	6.0E-06	1.8E-05	1.1E-05	8.5E-06	7.0E-06	1.2E-08	7.7E-09	6.2E-09	5.1E-09	1.6E-08	8.9E-09	7.2E-09	6.0E-09	9.4E-09	5.9E-09	4.7E-09	3.9E-09	1.2E-08	6.9E-09	5.5E-09	4.6E-09
2006	1.9E-05	1.1E-05	7.5E-06	5.8E-06	2.6E-05	1.3E-05	8.8E-06	6.8E-06	1.6E-08	9.1E-09	6.4E-09	4.9E-09	2.2E-08	1.1E-08	7.5E-09	5.8E-09	1.2E-08	7.0E-09	4.9E-09	3.8E-09	1.7E-08	8.4E-09	5.8E-09	4.5E-09
2007	1.9E-05	1.1E-05	7.4E-06	5.5E-06	3.2E-05	1.4E-05	8.7E-06	6.5E-06	1.6E-08	9.3E-09	6.3E-09	4.7E-09	2.7E-08	1.2E-08	7.4E-09	5.6E-09	1.3E-08	7.2E-09	4.8E-09	3.6E-09	2.1E-08	9.1E-09	5.7E-09	4.3E-09
2008	7.9E-06	7.0E-06	6.1E-06	5.0E-06	8.7E-06	8.0E-06	7.2E-06	5.9E-06	6.7E-09	5.9E-09	5.2E-09	4.2E-09	7.4E-09	6.8E-09	6.1E-09	5.0E-09	5.2E-09	4.5E-09	4.0E-09	3.3E-09	5.7E-09	5.2E-09	4.7E-09	3.9E-09
2009	8.5E-06	6.5E-06	5.6E-06	4.6E-06	1.0E-05	7.6E-06	6.6E-06	5.5E-06	7.2E-09	5.5E-09	4.7E-09	3.9E-09	8.6E-09	6.5E-09	5.6E-09	4.7E-09	5.6E-09	4.2E-09	3.6E-09	3.0E-09	6.6E-09	5.0E-09	4.3E-09	3.6E-09
2010	1.5E-05	8.8E-06	6.1E-06	4.6E-06	2.3E-05	1.1E-05	7.2E-06	5.5E-06	1.3E-08	7.5E-09	5.2E-09	3.9E-09	1.9E-08	9.3E-09	6.1E-09	4.7E-09	9.9E-09	5.7E-09	4.0E-09	3.0E-09	1.5E-08	7.2E-09	4.7E-09	3.6E-09
2011	1.5E-05	9.1E-06	6.2E-06	4.6E-06	2.5E-05	1.2E-05	7.3E-06	5.4E-06	1.3E-08	7.8E-09	5.2E-09	3.9E-09	2.1E-08	9.8E-09	6.2E-09	4.6E-09	1.0E-08	6.0E-09	4.0E-09	3.0E-09	1.6E-08	7.5E-09	4.8E-09	3.5E-09
2012	1.0E-05	7.7E-06	5.9E-06	4.5E-06	1.3E-05	9.2E-06	7.1E-06	5.4E-06	8.9E-09	6.5E-09	5.0E-09	3.8E-09	1.1E-08	7.8E-09	6.0E-09	4.6E-09	6.8E-09	5.0E-09	3.8E-09	2.9E-09	8.5E-09	6.0E-09	4.6E-09	3.5E-09
2013	1.4E-05	8.6E-06	6.0E-06	4.4E-06	2.0E-05	1.0E-05	7.1E-06	5.2E-06	1.2E-08	7.3E-09	5.1E-09	3.8E-09	1.7E-08	8.9E-09	6.0E-09	4.5E-09	8.9E-09	5.6E-09	3.9E-09	2.9E-09	1.3E-08	6.8E-09	4.6E-09	3.4E-09
2014	1.1E-05	7.5E-06	5.7E-06	4.3E-06	1.3E-05	8.6E-06	6.7E-06	5.1E-06	9.2E-09	6.4E-09	4.8E-09	3.6E-09	1.1E-08	7.3E-09	5.7E-09	4.3E-09	7.1E-09	4.9E-09	3.7E-09	2.8E-09	8.3E-09	5.6E-09	4.4E-09	3.3E-09
2015	1.0E-05	7.1E-06	5.4E-06	4.1E-06	1.2E-05	8.1E-06	6.4E-06	4.9E-06	8.8E-09	6.0E-09	4.6E-09	3.5E-09	1.0E-08	6.9E-09	5.4E-09	4.2E-09	6.8E-09	4.6E-09	3.5E-09	2.7E-09	8.0E-09	5.3E-09	4.2E-09	3.2E-09
2016	5.4E-06	5.0E-06	4.6E-06	3.8E-06	5.9E-06	5.7E-06	5.4E-06	4.5E-06	4.6E-09	4.2E-09	3.9E-09	3.2E-09	5.0E-09	4.9E-09	4.6E-09	3.8E-09	3.5E-09	3.3E-09	3.0E-09	2.5E-09	3.9E-09	3.7E-09	3.5E-09	2.9E-09
2017	5.1E-06	4.4E-06	4.1E-06	3.5E-06	5.7E-06	5.0E-06	4.8E-06	4.1E-06	4.4E-09	3.7E-09	3.5E-09	3.0E-09	4.8E-09	4.3E-09	4.1E-09	3.5E-09	3.3E-09	2.9E-09	2.7E-09	2.3E-09	3.7E-09	3.3E-09	3.2E-09	2.7E-09
2018	7.6E-06	5.4E-06	4.3E-06	3.4E-06	1.1E-05	6.8E-06	5.2E-06	4.1E-06	6.5E-09	4.6E-09	3.6E-09	2.9E-09	9.0E-09	5.8E-09	4.4E-09	3.5E-09	5.0E-09	3.5E-09	2.8E-09	2.2E-09	6.9E-09	4.4E-09	3.4E-09	2.7E-09

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TABLE 3-6: SUMMARY OF TRI+ SEDIMENT CONCENTRATIONS FROM THE FARLEY MODEL AND TEQ-BASED PREDICTIONS FOR 1993 - 2018

	Tri+ Average PCB Results				Tri+ 95% UCL Results				Average Avian TEF				95% Avian TEF				Average Mammalian TEF				95% UCL Mammalian TEF			
Year	152 Total Sed Conc mg/kg	113 Total Sed Conc mg/kg	90 Total Sed Conc mg/kg	50 Total Sed Conc mg/kg	152 Total Sed Conc mg/kg	113 Total Sed Conc mg/kg	90 Total Sed Conc mg/kg	50 Total Sed Conc mg/kg	152 Total Sed Conc mg/kg	113 Total Sed Conc mg/kg	90 Total Sed Conc mg/kg	50 Total Sed Conc mg/kg	152 Total Sed Conc mg/kg	113 Total Sed Conc mg/kg	90 Total Sed Conc mg/kg	50 Total Sed Conc mg/kg	152 Total Sed Conc mg/kg	113 Total Sed Conc mg/kg	90 Total Sed Conc mg/kg	50 Total Sed Conc mg/kg	152 Total Sed Conc mg/kg	113 Total Sed Conc mg/kg	90 Total Sed Conc mg/kg	50 Total Sed Conc mg/kg
1993	0.967	0.757	0.610	0.449	1.072	0.860	0.677	0.505	8.2E-04	6.4E-04	5.2E-04	3.8E-04	9.1E-04	7.3E-04	5.8E-04	4.3E-04	6.3E-04	4.9E-04	4.0E-04	2.9E-04	7.0E-04	5.6E-04	4.4E-04	3.3E-04
1994	0.882	0.720	0.581	0.426	1.023	0.838	0.656	0.490	7.5E-04	6.1E-04	4.9E-04	3.6E-04	8.7E-04	7.1E-04	5.6E-04	4.2E-04	5.8E-04	4.7E-04	3.8E-04	2.8E-04	6.7E-04	5.5E-04	4.3E-04	3.2E-04
1995	0.806	0.676	2.181	0.406	0.999	0.817	0.652	0.474	6.9E-04	5.7E-04	1.9E-03	3.4E-04	8.5E-04	6.9E-04	5.5E-04	4.0E-04	5.3E-04	4.4E-04	1.4E-03	2.6E-04	6.5E-04	5.3E-04	4.3E-04	3.1E-04
1996	0.809	0.649	2.179	0.387	0.977	0.795	0.634	0.460	6.9E-04	5.5E-04	1.9E-03	3.3E-04	8.3E-04	6.8E-04	5.4E-04	3.9E-04	5.3E-04	4.2E-04	1.4E-03	2.5E-04	6.4E-04	5.2E-04	4.1E-04	3.0E-04
1997	0.787	0.630	0.503	0.370	0.954	0.777	0.606	0.450	6.7E-04	5.4E-04	4.3E-04	3.1E-04	8.1E-04	6.6E-04	5.2E-04	3.8E-04	5.1E-04	4.1E-04	3.3E-04	2.4E-04	6.2E-04	5.1E-04	4.0E-04	2.9E-04
1998	0.728	0.600	0.482	0.355	0.942	0.766	0.590	0.438	6.2E-04	5.1E-04	4.1E-04	3.0E-04	8.0E-04	6.5E-04	5.0E-04	3.7E-04	4.8E-04	3.9E-04	3.1E-04	2.3E-04	6.1E-04	5.0E-04	3.8E-04	2.9E-04
1999	0.680	0.568	0.460	0.341	0.938	0.761	0.574	0.431	5.8E-04	4.8E-04	3.9E-04	2.9E-04	8.0E-04	6.5E-04	4.9E-04	3.7E-04	4.4E-04	3.7E-04	3.0E-04	2.2E-04	6.1E-04	5.0E-04	3.7E-04	2.8E-04
2000	0.666	0.547	0.440	0.327	0.910	0.745	0.566	0.421	5.7E-04	4.6E-04	3.7E-04	2.8E-04	7.7E-04	6.3E-04	4.8E-04	3.6E-04	4.3E-04	3.6E-04	2.9E-04	2.1E-04	5.9E-04	4.9E-04	3.7E-04	2.7E-04
2001	0.672	0.537	0.425	0.315	0.870	0.726	0.552	0.411	5.7E-04	4.6E-04	3.6E-04	2.7E-04	7.4E-04	6.2E-04	4.7E-04	3.5E-04	4.4E-04	3.5E-04	2.8E-04	2.1E-04	5.7E-04	4.7E-04	3.6E-04	2.7E-04
2002	0.646	0.524	0.415	0.306	0.866	0.709	0.540	0.401	5.5E-04	4.5E-04	3.5E-04	2.6E-04	7.4E-04	6.0E-04	4.6E-04	3.4E-04	4.2E-04	3.4E-04	2.7E-04	2.0E-04	5.7E-04	4.6E-04	3.5E-04	2.6E-04
2003	0.616	0.506	0.401	0.296	0.848	0.695	0.528	0.398	5.2E-04	4.3E-04	3.4E-04	2.5E-04	7.2E-04	5.9E-04	4.5E-04	3.4E-04	4.0E-04	3.3E-04	2.6E-04	1.9E-04	5.5E-04	4.5E-04	3.4E-04	2.6E-04
2004	0.586	0.486	0.387	0.286	0.872	0.700	0.524	0.389	5.0E-04	4.1E-04	3.3E-04	2.4E-04	7.4E-04	5.9E-04	4.5E-04	3.3E-04	3.8E-04	3.2E-04	2.5E-04	1.9E-04	5.7E-04	4.6E-04	3.4E-04	2.5E-04
2005	0.566	0.468	0.372	0.276	0.875	0.693	0.513	0.380	4.8E-04	4.0E-04	3.2E-04	2.3E-04	7.4E-04	5.9E-04	4.4E-04	3.2E-04	3.7E-04	3.1E-04	2.4E-04	1.8E-04	5.7E-04	4.5E-04	3.4E-04	2.5E-04
2006	0.561	0.457	0.360	0.267	0.811	0.675	0.503	0.372	4.8E-04	3.9E-04	3.1E-04	2.3E-04	6.9E-04	5.7E-04	4.3E-04	3.2E-04	3.7E-04	3.0E-04	2.4E-04	1.7E-04	5.3E-04	4.4E-04	3.3E-04	2.4E-04
2007	0.549	0.446	0.350	0.259	0.789	0.658	0.500	0.371	4.7E-04	3.8E-04	3.0E-04	2.2E-04	6.7E-04	5.6E-04	4.3E-04	3.2E-04	3.6E-04	2.9E-04	2.3E-04	1.7E-04	5.1E-04	4.3E-04	3.3E-04	2.4E-04
2008	0.528	0.434	0.340	0.251	0.809	0.646	0.489	0.363	4.5E-04	3.7E-04	2.9E-04	2.1E-04	6.9E-04	5.5E-04	4.2E-04	3.1E-04	3.4E-04	2.8E-04	2.2E-04	1.6E-04	5.3E-04	4.2E-04	3.2E-04	2.4E-04
2009	0.508	0.421	0.329	0.244	0.839	0.656	0.480	0.355	4.3E-04	3.6E-04	2.8E-04	2.1E-04	7.1E-04	5.6E-04	4.1E-04	3.0E-04	3.3E-04	2.7E-04	2.2E-04	1.6E-04	5.5E-04	4.3E-04	3.1E-04	2.3E-04
2010	0.501	0.411	0.320	0.237	0.770	0.639	0.469	0.348	4.3E-04	3.5E-04	2.7E-04	2.0E-04	6.5E-04	5.4E-04	4.0E-04	3.0E-04	3.3E-04	2.7E-04	2.1E-04	1.5E-04	5.0E-04	4.2E-04	3.1E-04	2.3E-04
2011	0.494	0.403	0.312	0.230	0.714	0.617	0.457	0.340	4.2E-04	3.4E-04	2.7E-04	2.0E-04	6.1E-04	5.2E-04	3.9E-04	2.9E-04	3.2E-04	2.6E-04	2.0E-04	1.5E-04	4.7E-04	4.0E-04	3.0E-04	2.2E-04
2012	0.480	0.394	0.305	0.225	0.699	0.586	0.445	0.332	4.1E-04	3.4E-04	2.6E-04	1.9E-04	5.9E-04	5.0E-04	3.8E-04	2.8E-04	3.1E-04	2.6E-04	2.0E-04	1.5E-04	4.6E-04	3.8E-04	2.9E-04	2.2E-04
2013	0.471	0.386	0.298	0.219	0.679	0.571	0.433	0.323	4.0E-04	3.3E-04	2.5E-04	1.9E-04	5.8E-04	4.9E-04	3.7E-04	2.7E-04	3.1E-04	2.5E-04	1.9E-04	1.4E-04	4.4E-04	3.7E-04	2.8E-04	2.1E-04
2014	0.457	0.377	0.291	0.214	0.668	0.558	0.421	0.315	3.9E-04	3.2E-04	2.5E-04	1.8E-04	5.7E-04	4.7E-04	3.6E-04	2.7E-04	3.0E-04	2.5E-04	1.9E-04	1.4E-04	4.4E-04	3.6E-04	2.8E-04	2.1E-04
2015	0.443	0.367	0.284	0.208	0.659	0.560	0.411	0.307	3.8E-04	3.1E-04	2.4E-04	1.8E-04	5.6E-04	4.8E-04	3.5E-04	2.6E-04	2.9E-04	2.4E-04	1.9E-04	1.4E-04	4.3E-04	3.7E-04	2.7E-04	2.0E-04
2016	0.429	0.357	0.276	0.203	0.706	0.557	0.403	0.300	3.6E-04	3.0E-04	2.3E-04	1.7E-04	6.0E-04	4.7E-04	3.4E-04	2.6E-04	2.8E-04	2.3E-04	1.8E-04	1.3E-04	4.6E-04	3.6E-04	2.6E-04	2.0E-04
2017	0.418	0.348	0.269	0.198	0.714	0.556	0.395	0.293	3.6E-04	3.0E-04	2.3E-04	1.7E-04	6.1E-04	4.7E-04	3.4E-04	2.5E-04	2.7E-04	2.3E-04	1.8E-04	1.3E-04	4.7E-04	3.6E-04	2.6E-04	1.9E-04
2018	0.407	0.339	0.261	0.193	0.679	0.561	0.388	0.287	3.5E-04	2.9E-04	2.2E-04	1.6E-04	5.8E-04	4.8E-04	3.3E-04	2.4E-04	2.7E-04	2.2E-04	1.7E-04	1.3E-04	4.4E-04	3.7E-04	2.5E-04	1.9E-04

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TABLE 3-7: ORGANIC CARBON NORMALIZED SEDIMENT CONCENTRATIONS BASED ON USEPA PHASE 2 DATASET

	Tri+ Average PCB Results				Tri+ 95% UCL Results			
Year	152 Total Sed Conc mg/kg	113 Total Sed Conc mg/kg	90 Total Sed Conc mg/kg	50 Total Sed Conc mg/kg	152 Total Sed Conc mg/kg	113 Total Sed Conc mg/kg	90 Total Sed Conc mg/kg	50 Total Sed Conc mg/kg
1993	38.67	30.29	24.39	17.97	42.90	34.40	27.09	20.18
1994	35.29	28.81	23.23	17.05	40.94	33.51	26.22	19.60
1995	32.25	27.04	87.23	16.22	39.96	32.67	26.08	18.97
1996	32.38	25.97	87.14	15.47	39.06	31.78	25.34	18.39
1997	31.47	25.19	20.14	14.82	38.17	31.06	24.23	18.02
1998	29.13	24.00	19.29	14.21	37.68	30.64	23.58	17.53
1999	27.20	22.73	18.40	13.62	37.53	30.42	22.95	17.26
2000	26.66	21.87	17.59	13.07	36.39	29.78	22.62	16.83
2001	26.88	21.47	16.99	12.58	34.79	29.04	22.08	16.42
2002	25.85	20.97	16.60	12.23	34.66	28.37	21.61	16.05
2003	24.64	20.26	16.06	11.82	33.94	27.80	21.11	15.91
2004	23.42	19.45	15.49	11.43	34.89	27.99	20.95	15.56
2005	22.66	18.74	14.90	11.04	35.00	27.70	20.54	15.21
2006	22.42	18.27	14.40	10.67	32.42	26.98	20.10	14.89
2007	21.96	17.86	13.98	10.35	31.55	26.30	20.00	14.84
2008	21.12	17.37	13.59	10.05	32.35	25.85	19.56	14.52
2009	20.31	16.82	13.18	9.75	33.55	26.25	19.18	14.22
2010	20.05	16.43	12.80	9.47	30.80	25.58	18.77	13.92
2011	19.76	16.11	12.48	9.22	28.57	24.67	18.29	13.60
2012	19.20	15.77	12.19	8.98	27.98	23.45	17.79	13.27
2013	18.85	15.44	11.91	8.76	27.16	22.84	17.31	12.94
2014	18.28	15.08	11.63	8.54	26.74	22.33	16.86	12.61
2015	17.71	14.70	11.34	8.34	26.38	22.42	16.45	12.29
2016	17.16	14.29	11.03	8.12	28.25	22.30	16.11	12.00
2017	16.73	13.91	10.74	7.93	28.54	22.23	15.80	11.71
2018	16.26	13.58	10.44	7.71	27.16	22.43	15.53	11.48

average TOC from Farley model 2.5%

TABLE 3-8: SUMMARY OF TRI+ BENTHIC INVERTEBRATE CONCENTRATIONS FROM THE FISHRAND MODEL AND TEQ-BASED PREDICTIONS FOR 1993 - 2018

	Tri+ Average PCB Results				Tri+ 95% UCL Results				Average Avian TEF				95% Avian TEF				Average Mammalian TEF				95% UCL Mammalian TEF			
Year	152 Total Benthic Conc mg/kg	113 Total Benthic Conc mg/kg	90 Total Benthic Conc mg/kg	50 Total Benthic Conc mg/kg	152 Total Benthic Conc mg/kg	113 Total Benthic Conc mg/kg	90 Total Benthic Conc mg/kg	50 Total Benthic Conc mg/kg	152 Total Benthic Conc mg/kg	113 Total Benthic Conc mg/kg	90 Total Benthic Conc mg/kg	50 Total Benthic Conc mg/kg	152 Total Benthic Conc mg/kg	113 Total Benthic Conc mg/kg	90 Total Benthic Conc mg/kg	50 Total Benthic Conc mg/kg	152 Total Benthic Conc mg/kg	113 Total Benthic Conc mg/kg	90 Total Benthic Conc mg/kg	50 Total Benthic Conc mg/kg	152 Total Benthic Conc mg/kg	113 Total Benthic Conc mg/kg	90 Total Benthic Conc mg/kg	50 Total Benthic Conc mg/kg
1993	1.754	1.393	1.131	0.831	1.885	1.495	1.215	0.893	2.4E-04	1.9E-04	1.6E-04	1.2E-04	2.6E-04	2.1E-04	1.7E-04	1.2E-04	1.9E-04	1.5E-04	1.2E-04	9.0E-05	2.0E-04	1.6E-04	1.3E-04	9.6E-05
1994	1.573	1.304	1.073	0.780	1.686	1.398	1.151	0.837	2.2E-04	1.8E-04	1.5E-04	1.1E-04	2.3E-04	1.9E-04	1.6E-04	1.2E-04	1.7E-04	1.4E-04	1.2E-04	8.4E-05	1.8E-04	1.5E-04	1.2E-04	9.0E-05
1995	1.522	1.252	1.006	0.741	1.632	1.341	1.079	0.794	2.1E-04	1.7E-04	1.4E-04	1.0E-04	2.3E-04	1.9E-04	1.5E-04	1.1E-04	1.6E-04	1.4E-04	1.1E-04	8.0E-05	1.8E-04	1.4E-04	1.2E-04	8.6E-05
1996	1.502	1.202	0.958	0.713	1.610	1.289	1.026	0.764	2.1E-04	1.7E-04	1.3E-04	9.9E-05	2.2E-04	1.8E-04	1.4E-04	1.1E-04	1.6E-04	1.3E-04	1.0E-04	7.7E-05	1.7E-04	1.4E-04	1.1E-04	8.2E-05
1997	1.422	1.153	0.928	0.690	1.524	1.235	0.994	0.739	2.0E-04	1.6E-04	1.3E-04	9.6E-05	2.1E-04	1.7E-04	1.4E-04	1.0E-04	1.5E-04	1.2E-04	1.0E-04	7.4E-05	1.6E-04	1.3E-04	1.1E-04	8.0E-05
1998	1.362	1.121	0.884	0.652	1.460	1.200	0.947	0.699	1.9E-04	1.6E-04	1.2E-04	9.0E-05	2.0E-04	1.7E-04	1.3E-04	9.7E-05	1.5E-04	1.2E-04	9.5E-05	7.0E-05	1.6E-04	1.3E-04	1.0E-04	7.5E-05
1999	1.291	1.087	0.852	0.633	1.386	1.166	0.912	0.678	1.8E-04	1.5E-04	1.2E-04	8.8E-05	1.9E-04	1.6E-04	1.3E-04	9.4E-05	1.4E-04	1.2E-04	9.2E-05	6.8E-05	1.5E-04	1.3E-04	9.8E-05	7.3E-05
2000	1.298	1.042	0.829	0.614	1.393	1.119	0.887	0.658	1.8E-04	1.4E-04	1.1E-04	8.5E-05	1.9E-04	1.6E-04	1.2E-04	9.1E-05	1.4E-04	1.1E-04	8.9E-05	6.6E-05	1.5E-04	1.2E-04	9.6E-05	7.1E-05
2001	1.269	1.027	0.804	0.595	1.360	1.103	0.861	0.637	1.8E-04	1.4E-04	1.1E-04	8.2E-05	1.9E-04	1.5E-04	1.2E-04	8.8E-05	1.4E-04	1.1E-04	8.7E-05	6.4E-05	1.5E-04	1.2E-04	9.3E-05	6.9E-05
2002	1.213	0.991	0.784	0.585	1.303	1.065	0.840	0.628	1.7E-04	1.4E-04	1.1E-04	8.1E-05	1.8E-04	1.5E-04	1.2E-04	8.7E-05	1.3E-04	1.1E-04	8.5E-05	6.3E-05	1.4E-04	1.1E-04	9.1E-05	6.8E-05
2003	1.140	0.946	0.767	0.564	1.225	1.016	0.823	0.606	1.6E-04	1.3E-04	1.1E-04	7.8E-05	1.7E-04	1.4E-04	1.1E-04	8.4E-05	1.2E-04	1.0E-04	8.3E-05	6.1E-05	1.3E-04	1.1E-04	8.9E-05	6.5E-05
2004	1.122	0.912	0.727	0.539	1.208	0.981	0.781	0.579	1.6E-04	1.3E-04	1.0E-04	7.5E-05	1.7E-04	1.4E-04	1.1E-04	8.0E-05	1.2E-04	9.8E-05	7.8E-05	5.8E-05	1.3E-04	1.1E-04	8.4E-05	6.2E-05
2005	1.091	0.904	0.700	0.519	1.174	0.972	0.752	0.557	1.5E-04	1.3E-04	9.7E-05	7.2E-05	1.6E-04	1.3E-04	1.0E-04	7.7E-05	1.2E-04	9.8E-05	7.6E-05	5.6E-05	1.3E-04	1.0E-04	8.1E-05	6.0E-05
2006	1.049	0.877	0.669	0.496	1.127	0.943	0.720	0.533	1.5E-04	1.2E-04	9.3E-05	6.9E-05	1.6E-04	1.3E-04	1.0E-04	7.4E-05	1.1E-04	9.5E-05	7.2E-05	5.3E-05	1.2E-04	1.0E-04	7.8E-05	5.7E-05
2007	1.035	0.859	0.652	0.482	1.113	0.924	0.701	0.518	1.4E-04	1.2E-04	9.0E-05	6.7E-05	1.5E-04	1.3E-04	9.7E-05	7.2E-05	1.1E-04	9.3E-05	7.0E-05	5.2E-05	1.2E-04	1.0E-04	7.6E-05	5.6E-05
2008	0.999	0.827	0.633	0.469	1.077	0.890	0.680	0.504	1.4E-04	1.1E-04	8.8E-05	6.5E-05	1.5E-04	1.2E-04	9.4E-05	7.0E-05	1.1E-04	8.9E-05	6.8E-05	5.1E-05	1.2E-04	9.6E-05	7.3E-05	5.4E-05
2009	0.978	0.802	0.619	0.459	1.055	0.864	0.665	0.494	1.4E-04	1.1E-04	8.6E-05	6.4E-05	1.5E-04	1.2E-04	9.2E-05	6.8E-05	1.1E-04	8.7E-05	6.7E-05	5.0E-05	1.1E-04	9.3E-05	7.2E-05	5.3E-05
2010	0.962	0.786	0.608	0.450	1.034	0.846	0.653	0.484	1.3E-04	1.1E-04	8.4E-05	6.2E-05	1.4E-04	1.2E-04	9.1E-05	6.7E-05	1.0E-04	8.5E-05	6.6E-05	4.9E-05	1.1E-04	9.1E-05	7.0E-05	5.2E-05
2011	0.922	0.779	0.587	0.443	0.991	0.838	0.631	0.477	1.3E-04	1.1E-04	8.1E-05	6.1E-05	1.4E-04	1.2E-04	8.7E-05	6.6E-05	9.9E-05	8.4E-05	6.3E-05	4.8E-05	1.1E-04	9.0E-05	6.8E-05	5.1E-05
2012	0.899	0.762	0.573	0.433	0.966	0.820	0.616	0.466	1.2E-04	1.1E-04	7.9E-05	6.0E-05	1.3E-04	1.1E-04	8.5E-05	6.5E-05	9.7E-05	8.2E-05	6.2E-05	4.7E-05	1.0E-04	8.8E-05	6.7E-05	5.0E-05
2013	0.879	0.745	0.556	0.420	0.945	0.802	0.598	0.452	1.2E-04	1.0E-04	7.7E-05	5.8E-05	1.3E-04	1.1E-04	8.3E-05	6.3E-05	9.5E-05	8.0E-05	6.0E-05	4.5E-05	1.0E-04	8.6E-05	6.4E-05	4.9E-05
2014	0.870	0.727	0.543	0.410	0.935	0.782	0.583	0.441	1.2E-04	1.0E-04	7.5E-05	5.7E-05	1.3E-04	1.1E-04	8.1E-05	6.1E-05	9.4E-05	7.8E-05	5.9E-05	4.4E-05	1.0E-04	8.4E-05	6.3E-05	4.8E-05
2015	0.845	0.700	0.532	0.400	0.911	0.754	0.572	0.430	1.2E-04	9.7E-05	7.4E-05	5.5E-05	1.3E-04	1.0E-04	7.9E-05	6.0E-05	9.1E-05	7.6E-05	5.7E-05	4.3E-05	9.8E-05	8.1E-05	6.2E-05	4.6E-05
2016	0.853	0.681	0.521	0.392	0.923	0.734	0.560	0.422	1.2E-04	9.4E-05	7.2E-05	5.4E-05	1.3E-04	1.0E-04	7.8E-05	5.8E-05	9.2E-05	7.3E-05	5.6E-05	4.2E-05	1.0E-04	7.9E-05	6.0E-05	4.5E-05
2017	0.842	0.675	0.515	0.382	0.912	0.729	0.553	0.411	1.2E-04	9.4E-05	7.1E-05	5.3E-05	1.3E-04	1.0E-04	7.7E-05	5.7E-05	9.1E-05	7.3E-05	5.6E-05	4.1E-05	9.8E-05	7.9E-05	6.0E-05	4.4E-05
2018	0.822	0.673	0.505	0.373	0.890	0.728	0.543	0.402	1.1E-04	9.3E-05	7.0E-05	5.2E-05	1.2E-04	1.0E-04	7.5E-05	5.6E-05	8.9E-05	7.3E-05	5.4E-05	4.0E-05	9.6E-05	7.9E-05	5.9E-05	4.3E-05

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TABLE 3-9: SPOTTAIL SHINER PREDICTED TRI+ CONCENTRATIONS FOR 1993 - 2018

	River Mile 152			River Mile 113			River Mile 90			River Mile 50		
Year	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)
1993	0.36	0.46	0.76	0.25	0.33	0.49	0.21	0.27	0.39	0.20	0.26	0.38
1994	0.28	0.41	0.63	0.23	0.31	0.45	0.19	0.24	0.35	0.18	0.23	0.33
1995	0.22	0.29	0.51	0.18	0.23	0.35	0.16	0.21	0.31	0.16	0.20	0.29
1996	0.29	0.40	0.66	0.20	0.27	0.40	0.15	0.20	0.29	0.15	0.18	0.27
1997	0.25	0.32	0.51	0.17	0.23	0.34	0.14	0.18	0.27	0.13	0.17	0.25
1998	0.18	0.22	0.34	0.14	0.19	0.28	0.12	0.16	0.24	0.12	0.15	0.22
1999	0.15	0.20	0.31	0.12	0.16	0.25	0.11	0.14	0.21	0.11	0.14	0.20
2000	0.16	0.22	0.35	0.12	0.17	0.25	0.10	0.13	0.20	0.10	0.13	0.19
2001	0.19	0.24	0.39	0.12	0.17	0.26	0.09	0.13	0.19	0.09	0.12	0.18
2002	0.15	0.19	0.30	0.12	0.15	0.23	0.09	0.12	0.19	0.09	0.11	0.17
2003	0.13	0.18	0.30	0.11	0.14	0.22	0.09	0.12	0.18	0.08	0.11	0.16
2004	0.10	0.14	0.22	0.09	0.12	0.18	0.08	0.10	0.16	0.08	0.10	0.15
2005	0.11	0.15	0.23	0.08	0.11	0.18	0.07	0.10	0.15	0.07	0.09	0.14
2006	0.12	0.17	0.29	0.08	0.12	0.18	0.06	0.09	0.14	0.06	0.09	0.13
2007	0.10	0.14	0.22	0.08	0.11	0.18	0.06	0.09	0.13	0.06	0.08	0.12
2008	0.09	0.11	0.19	0.07	0.10	0.16	0.06	0.08	0.13	0.06	0.08	0.12
2009	0.07	0.11	0.18	0.06	0.09	0.15	0.06	0.08	0.12	0.05	0.07	0.11
2010	0.10	0.14	0.21	0.07	0.10	0.15	0.05	0.08	0.12	0.05	0.07	0.11
2011	0.09	0.13	0.21	0.07	0.10	0.16	0.05	0.08	0.12	0.05	0.07	0.10
2012	0.09	0.13	0.20	0.07	0.10	0.16	0.05	0.08	0.12	0.05	0.07	0.10
2013	0.10	0.13	0.22	0.07	0.10	0.15	0.05	0.07	0.11	0.05	0.07	0.10
2014	0.09	0.12	0.20	0.07	0.10	0.15	0.05	0.07	0.11	0.05	0.06	0.10
2015	0.08	0.11	0.19	0.06	0.09	0.14	0.05	0.07	0.11	0.05	0.06	0.09
2016	0.06	0.09	0.14	0.05	0.08	0.12	0.05	0.07	0.10	0.04	0.06	0.09
2017	0.06	0.08	0.13	0.05	0.07	0.11	0.04	0.06	0.09	0.04	0.06	0.09
2018	0.07	0.09	0.14	0.05	0.07	0.12	0.04	0.06	0.10	0.04	0.06	0.09

TABLE 3-10: PUMPKINSEED PREDICTED TRI+ CONCENTRATIONS FOR 1993 - 2018

	River Mile 152			River Mile 113			River Mile 90			River Mile 50		
Year	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)
1993	1.16	1.57	2.54	0.76	1.05	1.73	0.58	0.84	1.37	0.57	0.79	1.31
1994	0.86	1.17	1.87	0.67	0.95	1.54	0.53	0.75	1.25	0.50	0.71	1.17
1995	0.74	1.03	1.71	0.53	0.77	1.28	0.46	0.66	1.09	0.45	0.63	1.04
1996	0.92	1.26	2.03	0.59	0.81	1.33	0.43	0.62	1.02	0.40	0.58	0.94
1997	0.78	1.06	1.72	0.51	0.74	1.24	0.39	0.57	0.95	0.37	0.53	0.86
1998	0.53	0.77	1.28	0.42	0.61	1.02	0.36	0.53	0.87	0.34	0.49	0.79
1999	0.47	0.68	1.13	0.37	0.54	0.90	0.32	0.46	0.77	0.30	0.44	0.72
2000	0.49	0.67	1.10	0.36	0.50	0.84	0.29	0.42	0.70	0.28	0.40	0.65
2001	0.55	0.75	1.22	0.37	0.52	0.87	0.28	0.40	0.66	0.26	0.37	0.60
2002	0.45	0.65	1.10	0.34	0.50	0.85	0.27	0.39	0.65	0.25	0.35	0.58
2003	0.43	0.60	1.00	0.32	0.46	0.77	0.25	0.36	0.61	0.23	0.33	0.55
2004	0.32	0.46	0.78	0.27	0.39	0.67	0.23	0.33	0.56	0.21	0.31	0.51
2005	0.33	0.46	0.77	0.26	0.36	0.62	0.21	0.30	0.52	0.20	0.28	0.47
2006	0.40	0.55	0.91	0.26	0.37	0.63	0.20	0.29	0.49	0.19	0.27	0.44
2007	0.32	0.45	0.75	0.26	0.36	0.61	0.20	0.28	0.47	0.18	0.25	0.42
2008	0.28	0.41	0.70	0.23	0.34	0.57	0.18	0.27	0.45	0.17	0.24	0.40
2009	0.26	0.37	0.64	0.21	0.30	0.52	0.17	0.25	0.43	0.16	0.23	0.38
2010	0.29	0.41	0.70	0.21	0.30	0.52	0.17	0.24	0.40	0.15	0.22	0.36
2011	0.32	0.45	0.75	0.23	0.32	0.54	0.17	0.24	0.40	0.15	0.21	0.35
2012	0.29	0.42	0.71	0.22	0.31	0.53	0.17	0.24	0.41	0.15	0.21	0.35
2013	0.32	0.45	0.76	0.22	0.32	0.54	0.17	0.24	0.40	0.15	0.21	0.35
2014	0.29	0.42	0.70	0.21	0.30	0.52	0.16	0.23	0.39	0.14	0.20	0.33
2015	0.26	0.37	0.62	0.20	0.29	0.48	0.15	0.22	0.38	0.14	0.20	0.32
2016	0.20	0.30	0.52	0.18	0.26	0.44	0.14	0.21	0.36	0.13	0.19	0.32
2017	0.19	0.29	0.50	0.16	0.24	0.41	0.14	0.20	0.34	0.13	0.18	0.30
2018	0.20	0.29	0.51	0.16	0.23	0.40	0.13	0.19	0.33	0.12	0.17	0.29

TABLE 3-11: YELLOW PERCH PREDICTED TRI+ CONCENTRATIONS FOR 1993 - 2018

	River Mile 152			River Mile 113			River Mile 90			River Mile 50		
Year	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)
1993	0.85	0.99	1.28	0.64	0.75	0.98	0.51	0.60	0.78	0.41	0.47	0.61
1994	0.71	0.85	1.11	0.58	0.69	0.90	0.47	0.56	0.73	0.38	0.44	0.57
1995	0.67	0.80	1.04	0.54	0.64	0.84	0.44	0.53	0.69	0.35	0.41	0.53
1996	0.70	0.83	1.06	0.52	0.61	0.81	0.42	0.49	0.65	0.33	0.39	0.50
1997	0.66	0.78	1.01	0.50	0.59	0.78	0.39	0.47	0.62	0.31	0.36	0.47
1998	0.58	0.71	0.92	0.47	0.56	0.73	0.37	0.45	0.59	0.29	0.35	0.45
1999	0.52	0.63	0.83	0.43	0.51	0.68	0.35	0.42	0.55	0.27	0.33	0.43
2000	0.50	0.60	0.79	0.40	0.49	0.64	0.33	0.40	0.52	0.26	0.31	0.40
2001	0.51	0.62	0.81	0.40	0.48	0.63	0.32	0.38	0.50	0.25	0.29	0.39
2002	0.50	0.60	0.78	0.39	0.47	0.62	0.31	0.37	0.49	0.24	0.28	0.37
2003	0.46	0.55	0.73	0.37	0.45	0.59	0.30	0.36	0.47	0.23	0.27	0.36
2004	0.42	0.50	0.67	0.35	0.42	0.56	0.28	0.34	0.45	0.22	0.26	0.34
2005	0.40	0.48	0.64	0.33	0.40	0.53	0.27	0.32	0.43	0.21	0.25	0.33
2006	0.42	0.50	0.66	0.32	0.39	0.52	0.26	0.31	0.41	0.20	0.24	0.32
2007	0.40	0.47	0.63	0.31	0.38	0.51	0.25	0.30	0.40	0.19	0.23	0.31
2008	0.38	0.46	0.60	0.31	0.37	0.49	0.24	0.29	0.39	0.19	0.22	0.30
2009	0.35	0.42	0.57	0.29	0.35	0.47	0.23	0.28	0.38	0.18	0.22	0.29
2010	0.35	0.42	0.56	0.28	0.34	0.46	0.22	0.27	0.36	0.17	0.21	0.28
2011	0.36	0.43	0.57	0.28	0.34	0.46	0.22	0.27	0.35	0.17	0.20	0.27
2012	0.35	0.42	0.55	0.28	0.34	0.45	0.21	0.26	0.35	0.17	0.20	0.26
2013	0.35	0.42	0.55	0.27	0.33	0.44	0.21	0.26	0.34	0.16	0.20	0.26
2014	0.33	0.40	0.53	0.27	0.32	0.43	0.21	0.25	0.33	0.16	0.19	0.25
2015	0.31	0.38	0.51	0.26	0.31	0.42	0.20	0.24	0.32	0.15	0.19	0.25
2016	0.30	0.36	0.48	0.25	0.30	0.40	0.19	0.24	0.31	0.15	0.18	0.24
2017	0.29	0.35	0.47	0.24	0.29	0.39	0.19	0.23	0.30	0.15	0.18	0.23
2018	0.28	0.34	0.45	0.23	0.28	0.37	0.18	0.22	0.29	0.14	0.17	0.23

TABLE 3-12: WHITE PERCH PREDICTED TRI+ CONCENTRATIONS FOR 1993 - 2018

	River Mile 152			River Mile 113			River Mile 90			River Mile 50		
Year	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)
1993	2.69	2.86	3.30	2.08	2.21	2.55	1.65	1.75	2.03	1.32	1.39	1.58
1994	2.32	2.47	2.88	1.91	2.03	2.37	1.54	1.64	1.92	1.23	1.29	1.47
1995	2.16	2.32	2.70	1.76	1.88	2.21	1.43	1.53	1.81	1.14	1.20	1.38
1996	2.32	2.45	2.77	1.70	1.80	2.10	1.35	1.44	1.69	1.07	1.13	1.29
1997	2.10	2.24	2.61	1.62	1.73	2.04	1.28	1.37	1.62	1.01	1.07	1.23
1998	1.86	2.01	2.40	1.54	1.63	1.91	1.21	1.30	1.55	0.95	1.02	1.18
1999	1.72	1.84	2.17	1.39	1.49	1.78	1.13	1.22	1.45	0.89	0.96	1.11
2000	1.66	1.77	2.11	1.31	1.41	1.69	1.07	1.15	1.37	0.85	0.90	1.05
2001	1.72	1.82	2.12	1.29	1.39	1.66	1.02	1.10	1.32	0.81	0.86	1.01
2002	1.65	1.76	2.06	1.27	1.37	1.63	1.00	1.07	1.28	0.78	0.83	0.97
2003	1.51	1.62	1.92	1.21	1.30	1.56	0.96	1.03	1.24	0.75	0.80	0.94
2004	1.36	1.47	1.78	1.13	1.23	1.48	0.91	0.99	1.19	0.71	0.76	0.90
2005	1.31	1.42	1.72	1.07	1.16	1.41	0.87	0.94	1.13	0.68	0.73	0.86
2006	1.36	1.45	1.73	1.05	1.14	1.38	0.83	0.90	1.09	0.65	0.70	0.83
2007	1.30	1.40	1.66	1.02	1.11	1.34	0.80	0.87	1.06	0.63	0.67	0.80
2008	1.23	1.33	1.61	1.00	1.08	1.31	0.78	0.85	1.03	0.61	0.65	0.78
2009	1.15	1.24	1.51	0.95	1.03	1.25	0.75	0.82	0.99	0.58	0.63	0.75
2010	1.17	1.26	1.52	0.92	1.01	1.23	0.72	0.79	0.96	0.56	0.61	0.73
2011	1.19	1.28	1.52	0.92	1.00	1.21	0.71	0.77	0.94	0.55	0.59	0.71
2012	1.14	1.23	1.48	0.91	0.99	1.20	0.71	0.76	0.93	0.54	0.58	0.69
2013	1.15	1.24	1.47	0.90	0.97	1.17	0.69	0.75	0.90	0.53	0.57	0.67
2014	1.09	1.17	1.40	0.87	0.94	1.14	0.67	0.72	0.88	0.51	0.55	0.66
2015	1.03	1.11	1.34	0.84	0.91	1.10	0.65	0.70	0.86	0.50	0.53	0.64
2016	0.98	1.06	1.29	0.81	0.88	1.07	0.63	0.68	0.83	0.48	0.52	0.62
2017	0.94	1.02	1.25	0.77	0.84	1.03	0.61	0.66	0.81	0.47	0.51	0.61
2018	0.92	1.01	1.23	0.76	0.83	1.02	0.59	0.65	0.80	0.46	0.50	0.60

TABLE 3-13: BROWN BULLHEAD PREDICTED TRI+ CONCENTRATIONS FOR 1993 - 2018

	River Mile 152			River Mile 113			River Mile 90			River Mile 50		
Year	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)
1993	2.34	3.32	5.48	1.78	2.55	4.28	1.43	2.05	3.44	1.10	1.57	2.59
1994	2.04	2.94	4.90	1.66	2.39	4.00	1.35	1.93	3.25	1.03	1.47	2.44
1995	1.90	2.74	4.56	1.54	2.23	3.75	1.26	1.82	3.06	0.97	1.39	2.31
1996	1.93	2.77	4.61	1.49	2.14	3.60	1.19	1.72	2.90	0.91	1.31	2.18
1997	1.83	2.63	4.38	1.43	2.07	3.45	1.14	1.64	2.77	0.87	1.24	2.08
1998	1.69	2.43	4.06	1.34	1.95	3.28	1.09	1.57	2.64	0.83	1.18	1.97
1999	1.52	2.20	3.70	1.25	1.81	3.05	1.02	1.48	2.50	0.78	1.13	1.88
2000	1.48	2.16	3.63	1.20	1.75	2.93	0.97	1.41	2.36	0.74	1.07	1.79
2001	1.50	2.17	3.62	1.18	1.72	2.87	0.93	1.36	2.28	0.71	1.03	1.71
2002	1.44	2.09	3.49	1.15	1.67	2.80	0.91	1.32	2.21	0.69	0.99	1.66
2003	1.35	1.96	3.29	1.09	1.60	2.69	0.87	1.27	2.14	0.66	0.96	1.60
2004	1.26	1.83	3.08	1.04	1.52	2.57	0.83	1.22	2.06	0.63	0.92	1.54
2005	1.21	1.78	2.99	1.00	1.46	2.46	0.80	1.17	1.97	0.61	0.89	1.48
2006	1.23	1.78	2.98	0.98	1.43	2.40	0.77	1.13	1.90	0.59	0.85	1.43
2007	1.17	1.71	2.88	0.95	1.39	2.34	0.75	1.10	1.84	0.57	0.82	1.38
2008	1.13	1.64	2.77	0.93	1.35	2.27	0.73	1.06	1.78	0.55	0.80	1.34
2009	1.08	1.57	2.65	0.89	1.30	2.19	0.70	1.03	1.72	0.53	0.77	1.29
2010	1.06	1.57	2.64	0.87	1.27	2.14	0.68	1.00	1.67	0.52	0.75	1.25
2011	1.07	1.55	2.62	0.86	1.26	2.11	0.66	0.97	1.64	0.50	0.73	1.22
2012	1.04	1.52	2.55	0.84	1.24	2.07	0.65	0.96	1.61	0.49	0.72	1.20
2013	1.02	1.49	2.51	0.83	1.21	2.03	0.64	0.93	1.57	0.48	0.70	1.16
2014	0.99	1.44	2.42	0.81	1.18	1.98	0.62	0.91	1.53	0.47	0.68	1.13
2015	0.95	1.38	2.33	0.78	1.14	1.92	0.61	0.89	1.49	0.46	0.66	1.11
2016	0.90	1.32	2.24	0.76	1.10	1.86	0.59	0.86	1.44	0.44	0.64	1.08
2017	0.88	1.28	2.16	0.73	1.07	1.80	0.57	0.83	1.40	0.43	0.63	1.05
2018	0.85	1.25	2.12	0.71	1.04	1.77	0.55	0.81	1.37	0.42	0.61	1.03

TABLE 3-14: LARGEMOUTH BASS PREDICTED TRI+ CONCENTRATIONS FOR 1993 - 2018

	River Mile 152			River Mile 113			River Mile 90			River Mile 50		
Year	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)
1993	11.28	14.33	21.56	7.50	9.58	14.39	1.84	2.23	3.05	1.75	2.11	2.86
1994	8.05	10.38	15.44	6.55	8.37	12.63	1.69	2.03	2.78	1.57	1.89	2.57
1995	7.10	8.92	13.51	5.89	7.45	11.24	1.52	1.83	2.53	1.41	1.70	2.30
1996	8.25	10.58	15.79	5.39	6.94	10.40	1.37	1.67	2.30	1.28	1.53	2.08
1997	7.62	9.63	14.45	5.26	6.71	10.08	1.29	1.56	2.14	1.17	1.42	1.92
1998	6.05	7.56	11.61	4.73	6.10	9.19	1.20	1.44	1.98	1.09	1.30	1.78
1999	5.06	6.53	9.76	3.96	5.10	7.73	1.07	1.29	1.78	0.98	1.18	1.62
2000	4.78	6.12	9.25	3.57	4.64	7.04	0.96	1.17	1.63	0.89	1.08	1.48
2001	5.34	6.96	10.34	3.64	4.70	7.11	0.90	1.11	1.55	0.83	1.01	1.39
2002	5.07	6.37	9.66	3.62	4.65	7.07	0.88	1.08	1.50	0.79	0.97	1.32
2003	4.34	5.66	8.54	3.31	4.27	6.52	0.84	1.03	1.43	0.75	0.92	1.26
2004	3.59	4.57	7.01	2.96	3.79	5.81	0.78	0.95	1.33	0.70	0.86	1.18
2005	3.35	4.35	6.61	2.68	3.48	5.31	0.72	0.88	1.23	0.65	0.80	1.10
2006	3.83	4.90	7.49	2.65	3.44	5.23	0.67	0.83	1.16	0.61	0.75	1.03
2007	3.48	4.52	6.79	2.60	3.37	5.10	0.65	0.80	1.13	0.58	0.71	0.98
2008	3.32	4.21	6.41	2.53	3.24	4.96	0.63	0.77	1.09	0.55	0.68	0.94
2009	2.81	3.64	5.57	2.29	2.96	4.54	0.59	0.73	1.03	0.53	0.65	0.90
2010	2.99	3.84	5.80	2.18	2.83	4.31	0.56	0.69	0.98	0.50	0.62	0.86
2011	3.28	4.29	6.49	2.31	3.01	4.57	0.56	0.69	0.97	0.48	0.60	0.83
2012	2.99	3.84	5.81	2.27	2.94	4.49	0.56	0.68	0.96	0.48	0.58	0.82
2013	3.19	4.18	6.30	2.33	3.03	4.62	0.57	0.70	0.98	0.49	0.60	0.82
2014	2.94	3.80	5.80	2.22	2.87	4.38	0.53	0.66	0.93	0.46	0.56	0.79
2015	2.70	3.51	5.36	2.11	2.74	4.17	0.52	0.64	0.90	0.45	0.55	0.77
2016	2.56	3.22	4.97	1.99	2.55	3.91	0.50	0.61	0.86	0.43	0.53	0.74
2017	2.27	2.90	4.44	1.82	2.35	3.59	0.47	0.58	0.81	0.42	0.52	0.72
2018	2.16	2.82	4.30	1.71	2.23	3.42	0.44	0.55	0.78	0.40	0.49	0.68

TABLE 3-15: STRIPED BASS PREDICTED TRI+ CONCENTRATIONS FOR 1993 - 2018

	River Mile 152			River Mile 113		
Year	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)
1993	28.66	36.41	54.77	3.90	4.98	7.48
1994	20.43	26.37	39.23	3.40	4.35	6.57
1995	18.03	22.65	34.33	3.06	3.88	5.85
1996	20.95	26.88	40.12	2.81	3.61	5.41
1997	19.34	24.47	36.70	2.73	3.49	5.24
1998	15.36	19.19	29.49	2.46	3.17	4.78
1999	12.85	16.58	24.80	2.06	2.65	4.02
2000	12.15	15.55	23.50	1.86	2.41	3.66
2001	13.57	17.67	26.26	1.89	2.44	3.69
2002	12.87	16.19	24.54	1.88	2.42	3.68
2003	11.02	14.37	21.69	1.72	2.22	3.39
2004	9.12	11.61	17.80	1.54	1.97	3.02
2005	8.50	11.04	16.80	1.39	1.81	2.76
2006	9.72	12.45	19.03	1.38	1.79	2.72
2007	8.85	11.49	17.26	1.35	1.75	2.65
2008	8.43	10.69	16.27	1.32	1.69	2.58
2009	7.14	9.25	14.16	1.19	1.54	2.36
2010	7.59	9.74	14.73	1.14	1.47	2.24
2011	8.33	10.89	16.50	1.20	1.56	2.38
2012	7.58	9.75	14.75	1.18	1.53	2.33
2013	8.11	10.62	15.99	1.21	1.58	2.40
2014	7.47	9.66	14.72	1.15	1.49	2.28
2015	6.87	8.92	13.60	1.09	1.42	2.17
2016	6.51	8.17	12.62	1.03	1.33	2.03
2017	5.77	7.36	11.27	0.95	1.22	1.87
2018	5.50	7.16	10.92	0.89	1.16	1.78

TABLE 3-16
EXPOSURE PARAMETERS FOR THE TREE SWALLOW (*Tachycineta bicolor*)

	Exposure Parameters		Range Reported for Species
Common Name	Tree Swallow		-
Genus	*Tachycineta*		-
Species	*bicolor*		-
Sex (M/F)	Female	Male	-
Age (Adult/Juv.)	Adult, Breeding		-
Male/Female Body Weight (kg) [1]	0.0210	0.0206	0.017-0.0255 (M and F)
Total Daily Dietary Ingestion (kg/day wet wt.) [2]	0.018	0.018	0.016-0.020
Total Daily Dietary Ingestion (kg/day dry wt.) [3]	0.005	-	No Contact with Sediments
General Dietary Characterization	Insectivore		-
Percent Diet Composition (% wet wt.) [4]			
Fish (Total Component)	0%		0%
Aquatic Invertebrates (Total Component)[5]	100%		95.0% - 100.0%
Non-river Related Diet Sources	0%		0%
Water Consumption Rate (L/day) [6]	0.0044		0.0038-0.0050
Percent Incidental Sediment Ingestion in Diet[7]	0.00%		No Contact with Sediments
Foraging Territory (km) [8]	0.1		0.1-0.2
Behavioral Modification Factors in the Exposure Assessment[9]			
Temporal Migration CorrectionFactor (1-%Annual Temporal Displaceme	1		-
Temporal Hibernation/Asetivation Correction Factor (1-%Temporal Hib/	1		-
Habitat Use Factor (Temporal use factor %)	1		Feeds over open water habitats
Temporal Reproductive Period (Mating/Gestation/Birth)[10, 11]	April - June		April - June

Notes: [1] Secord and McCarty (1997), Robertson et al. (1992); [2] Estimated from Nagy (1987) and USEPA (December, 1993); [3] No contact with sediments; [4] Secord and McCarty (1997), McCarty and Winkler (In Press); [5] Emergent forms of insects with partial aquatic life histories; [6] Calder and Braun (1983 In USE December 1993), Davis (1982); [7] Robertson et al. (1992); [8] McCarty and Winkler (In Press); [9] Robertson et al. (1992), see text for rationale; [10] Bull (1998), And (1988).

302735

TABLE 3-17
EXPOSURE PARAMETERS FOR THE MALLARD (*Anas platyrhynchos*)

	Exposure Parameters		Range Reported for Species
Common Name	Mallard		-
Genus	*Anas*		-
Species	*platyrhychos*		-
Sex (M/F)	Female	Male	-
Age (Adult/Juv.)	Adult, Breeding		-
Male/Female Body Weight (kg) [1]	1.06	1.24	1.01 - 1.11 F/M 1.21 - 1.27
Total Daily Dietary Ingestion (kg/day wet wt.) [2]	0.292	0.322	0.270-0.279 F/0.317-0.326 M
Total Daily Dietary Ingestion (kg/day dry wt.) [3]	0.061	0.067	0.058-0.063 F/ 0.066-0.068 M
General Dietary Characterization	Opportunistic Omnivore		-
Percent Diet Composition (% wet wt.) [4]			
Fish (Total Component)	0%		0%
Aquatic Invertebrates (Total Component)	50%		10 - 100%
Aquatic Vegetation/Seeds	48%		8 - 90 %
Water Consumption Rate (L/day) [5]	0.061	0.068	0.059-0.063 F/ 0.067 - 0.069 M
Percent Incidental Sediment Ingestion in Diet [6]	2.00%		2.00%
Foraging Territory (km) [7]	540.0	620.0	40.0-1440.0 Ha
Behavioral Modification Factors in the Exposure Assessment [8]			
Temporal Migration CorrectionFactor (1-%Annual Temporal Displaceme	1		Resident
Temporal Hibernation/Asetivation Correction Factor (1-%Temporal Hib/	1		Active Year Round
Habitat Use Factor (Temporal use factor %)	1		Riparian habitats preferred
Temporal Reproductive Period (Mating/Gestation/Birth)[9,10]	February -May		February -May

[1] Dunning (1993), USEPA (December 1993); [2] Estimated from Nagy (1987) and USEPA (December 1993); [3] Estimated from USEPA (December 1993); [4] Average of diet study summaries presented in USEPA (December 1993); [5] Calder and Braun (1983 In USEPA, December 1993); [6] Beyer et al. (1994); [7] Kirby et al. (1985 In USEPA, December 1993); [8] Bull (1998), USEPA (December 1993); [9 10] Bull (1998), Andrle and Carroll (1988).

302736

TABLE 3-18
EXPOSURE PARAMETERS FOR BELTED KINGFISHER (*Ceryle alcyon*)

	Exposure Parameters		Range Reported for Species
Common Name	Belted Kingfisher		-
Genus	*Ceryle*		-
Species	*alcyon*		-
Sex (M/F)	Female	Male	-
Age (Adult/Juv.)	Adult, Breeding		-
Male/Female Body Weight (kg) [1]	0.147	0.147	0.136-0.158 M and F
Total Daily Dietary Ingestion (kg/day wet wt.) [2]	0.058	0.058	0.055-0.060 M and F
Total Daily Dietary Ingestion (kg/day dry wt.) [3]	0.017	0.017	-
General Dietary Characterization	Opportunistic Piscivore		-
Percent Diet Composition (% wet wt.) [4]			
Fish (Total Component)	78%		46% - 100%
Aquatic Invertebrates (Total Component)	22%		5% - 41%
Non-river Related Diet Sources	0%		0-4.3%
Water Consumption Rate (L/day) [5]	0.016		0.015-0.017
Percent Incidental Sediment Ingestion in Diet [6]	1.00%		nests in banks, grooming
Foraging Territory (km) [7]	0.70		0.389-1.03
Behavioral Modification Factors in the Exposure Assessment [8]			
Temporal Migration CorrectionFactor (1-%Annual TemporalDisplacement)	1		Resident
Temporal Hibernation/Asetivation Correction Factor (1-%Temporal Hib/Aset.)	1		Active Year Round
Habitat Use Factor (Temporal use factor %)	1		Riparian habitats preferred
Temporal Reproductive Period (Mating/Gestation/Hatching) [9,10]	April - June		April - June

[1] Brooks and Davis (1987), Poole (1932); [2] Estimated from Nagy (1987) and USEPA (December 1993); [3] No contact with sediments;
[4] Gould unpublished data (In USEPA, December 1993), Davis (1982); [5] Calder and Braun (1983 In USEPA December 1993); [6] Best Professional Judgment based on Davis (1982); [7] Davis (1982); [8] Bull (1998), USEPA (December 1993); [9, 10] Bull (1998), Andrle and Carroll (1988).

302737

TABLE 3-19
EXPOSURE PARAMETERS FOR GREAT BLUE HERON (*Ardea herodias*)

	Exposure Parameters		Range Reported for Species
Common Name	Great Blue Heron		-
Genus	*Ardea*		-
Species	*herodias*		-
Sex (M/F)	Female	Male	-
Age (Adult/Juvenile)	Adult, Breeding		-
Male/Female Body Weight (kg) [1]	2.20	2.58	1.87-2.54 F/ 2.28-2.88 M
Total Daily Dietary Ingestion (kg/day wet wt.) [2]	0.352	0.390	0.284-0.431 F/ 0.331-0.455 M
Total Daily Dietary Ingestion (kg/day dry wt.) [3]	0.097	0.108	
General Dietary Characterization	Opportunistic Piscivore		-
Percent Diet Composition (% wet wt.) [4]			
Fish (Total Component)	98%		72-98%
Aquatic Invertebrates (Total Component)	1%		1-18%
Non-river Related Diet Sources	1%		0-4.3%
Water Consumption Rate (L/day) [5]	0.100	0.111	0.089-0.110 F/ 0.102-0.119 M
Percent Incidental Sediment Ingestion in Diet [6]	2.00%		-
Foraging Territory (km) [7]	0.98		0.6-1.37
Behavioral Modification Factors in the Exposure Assessment [8]			
Temporal Migration CorrectionFactor (1-%Annual Temporal Displacement)	1		Resident
Temporal Hibernation/Asetivation Correction Factor (1-%Temporal Hib/Aset.)	1		Active Year Round
Habitat Use Factor (Temporal use factor %)	1		Riparian habitats preferred
Temporal Reproductive Period (Mating/Gestation/Birth)[9,10]	March - June		March -June

Notes: [1] Dunning (1993) ; [2] Estimated from Nagy (1987) and USEPA (December 1993); [4] Alexander (1977 In USEPA, December 1993), Cotaam and Uhler (1945); [5] Calder and Braun (1983 In USEPA, December 1993); [6] Best Professional Judgement based on Eckert and Karalus (1988); 7 Peifer (1979 In USEPA (December, 1993); [8] USEPA (December, 1993); [9, 10] Bull (1998) and Andrle and Carroll (1988).

302738

TABLE 3-20
EXPOSURE PARAMETERS FOR BALD EAGLE (*Haliaeetus leucocephalus*)

	Exposure Parameters		Range Reported for Species
Common Name	Bald Eagle		-
Genus	*Haliaeetus*		-
Species	*leucocephalus*		-
Sex (M/F)	Female	Male	-
Age (Adult/Juvenile)	Adult, Breeding		-
Male/Female Body Weight (kg) [1]	5.10	3.20	4.5-5.6 F/M 3.0-3.4
Total Daily Dietary Ingestion (kg/day wet wt.) [2]	0.65	0.46	0.60-0.69 F/0.46-0.49 M
Total Daily Dietary Ingestion (kg/day dry wt.) [3]	-	-	-
General Dietary Characterization [4]	Opportunistic Piscivore		-
Percent Diet Composition (% wet wt.) [4]			
Fish (Total Component)	100%		70-100%
Aquatic Invertebrates (Total Component)	0%		0-18%
Non-river Related Diet Sources	0%		0-4.3%
Water Consumption Rate (L/day) [5]	0.175	0.129	0.162-0.187 F/0.123-0.134 M
Percent Incidental Sediment Ingestion in Diet [6]	0.00%		0.00%
Foraging Territory (km) [7]	5.0		3.0-7.0 Km
Behavioral Modification Factors in the Exposure Assessment [8]			
Temporal Migration CorrectionFactor (1-%Annual Temporal Displacement)	1		Resident
Temporal Hibernation/Asetivation Correction Factor (1-%Temporal Hib/Aset.)	1		Active Year Round
Habitat Use Factor (Temporal use factor %)	1		Riparian habitats preferred
Temporal Reproductive Period (Mating/Gestation/Birth)[9,10]	February - May		February - May

[1] Bopp (1999), USEPA (December 1993), Dunning (1993); [2], [3] Estimated from Nagy (1987) and USEPA (December 1993);
[4] Nye (1999), Bull (1998), USEPA (December 1993), Nye and Suring (1978); [5] Caluder and Braun (1983 In USEPA December 1993);
[6] Best Professional Judgement - USEPA (December 1993);
[7] Craig et al. (1988 In USEPA, December 1993); [8] Nye (1999), USEPA (December 1993); [9] [10] Nye (1999), Andrle and Carroll (1988).

302739

TABLE 3-21
EXPOSURE PARAMETERS FOR LITTLE BROWN BAT (*Myotis lucifugus*)

	Exposure Parameters		Proximal Range Reported for Species
Common Name	Little Brown Bat		-
Genus	*Myotis*		-
Species	*lucifugus*		-
Sex (M/F)	Female	Male	-
Age (Adult/Juv.)	Adult, Breeding		-
Male/Female Body Weight (kg) [1]	0.0071	0.0069	0.0042-0.0094 /0.0055-0.0077
Total Daily Dietary Ingestion (kg/day wet wt.) [2]	0.0025	0.0025	0.0025-0.0037 F/ No Male Data
Total Daily Dietary Ingestion (kg/day dry wt.) [3]	-	-	-
General Dietary Characterization [4]	Insectivore		-
Percent Diet Composition (% wet wt.) [4]			
Fish (Total Component)	*0.0%*		0%
Aquatic Invertebrates (Total Component)	*100.0%*		87.0 % - 100.0%
Non-river Related Diet Sources	*0.0%*		0 % - 13.0 %
Water Consumption Rate (L/day) [5]	0.0011	0.0011	Based upon 0.007 Kg
Percent Incidental Sediment Ingestion in Diet [6]	0.00%	0.00%	0.00%
Home Range (km) [7]	0.1	>0.1	0.1 - >0.1
Behavioral Modification Factors in the Exposure Assessment [8]			
Temporal Migration CorrectionFactor (1-%Annual Temporal Displacement)	1		Resident
Temporal Hibernation/Asetivation Correction Factor (1-%Temporal Hib/Aset.)	1		See text
Habitat Use Factor (Temporal use factor %)	1		Feeds over waterbody
Temporal Reproductive Period (Mating/Gestation/Birth) [9, 10]	April to July	-	April to July

[1] Bopp (1999); [2] Fenton and Barclay (1980); [3] Dry weight basis of ingestion not required;
[4] Anthony and Kunz (1977), Belwood and Fenton (1976), Buchler (1976); [5] Farrell and Wood (1968c In USEPA, December 1993); [6] No contact with sediments; [7] Bulcher (1976); [8] Davis and Hitchcock (1965); [9, 10] Belwood and Fenton (1976), Wimbatt (1945).

302740

TABLE 3-22
EXPOSURE PARAMETERS FOR RACCOON (*Proycon lotor*)

	Exposure Parameters		Proximal Range Reported for Species
Common Name	Raccoon		-
Genus	*Procyon*		-
Species	*lotor*		-
Sex (M/F)	Female	Male	-
Age (Adult/Juv.)	Adult, Breeding		-
Male/Female Body Weight (kg) [1]	6.400	7.600	5.6-7.1 F/7.0-8.3 M
Total Daily Dietary Ingestion (kg/day wet wt.) [2]	0.99	1.20	0.866-1.1 F/1.1-1.30 M
Total Daily Dietary Ingestion (kg/day dry wt.) [3]	0.316	0.364	0.283-0.344 F/0.340-0.391 M
General Dietary Characterization [4]	Opportunistic Omnivore		-
Percent Diet Composition (% wet wt.) [4]			
Fish (Total Component)	*3.0%*		0-3%
Aquatic Invertebrates (Total Component)	*37.0%*		1.4-37.0%
Non-river Related Diet Sources	*60.0%*		0-1.5%
Water Consumption Rate (L/day) [5]	0.526	0.614	0.467-0.578 F/0.571-0.665 M
Percent Incidental Sediment Ingestion in Diet [6]	9.4%	9.4%	9.40%
Home Range (hectare) [7]	48.0	48.0	5.3-376 F/18.2-814 M
Behavioral Modification Factors in the Exposure Assessment [8]			
Temporal Migration CorrectionFactor (1-%Annual TemporalDisplacement)	1		Resident
Temporal Hibernation/Asetivation Correction Factor (1-%Temporal Hib/Aset.)	1		Active Year Round
Habitat Use Factor (Temporal use factor %)	1		Riparian habitats preferred
Temporal Reproductive Period (Mating/Gestation/Birth) [9, 10]	January to May	-	January to May

[1] Bopp (1999), Sanderson (1984), USEPA (December 1993); [2], [3] Estimated from NFMR and ME in USEPA (December 1993) and Nagy (1987);
[4] Tabatabai and Kennedy)1988), Newell et al. (1987), [illegible] and Uhler (1952), Hamilton (1951); [5] Farrell and Wood (1968c In USEPA, 1993a);
[6] Beyer et al. (1994); [7] Urban (1970), Stuewer (1943); [8] USEPA (December, 1993), Hamilton (1951); [9 10] USEPA (December, 1993), Stuewer (1943).

302741

TABLE 3-23
EXPOSURE PARAMETERS FOR MINK (*Mustela vison*)

	Exposure Parameters		Proximal Range Reported for Species
Common Name	Mink		-
Genus	*Mustela*		-
Species	*vision*		-
Sex (M/F)	Female	Male	-
Age (Adult/Juv.)	Adult, Breeding		-
Male/Female Body Weight (kg) [1]	0.83	1.02	0.550-1.101 F/0.681-1.362 M
Total Daily Dietary Ingestion (kg/day wet wt.) [2]	0.132	0.132	0.145 F/ 0.119 M
Total Daily Dietary Ingestion (kg/day dry wt.) [3]	0.059	0.069	0.042-1.013 F/0.050-0.089 M
General Dietary Characterization [4]	Opportunistic Piscivore/Carnivore		-
Percent Diet Composition (% wet wt.) [4]			
Fish (Total Component)	34.0%		18.8-34.0%
Aquatic Invertebrates (Total Component)	16.5%		13.9-16.5%
Non-river Related Diet Sources	49.5%		49.5 % - 67.0 %
Water Consumption Rate (L/day) [5]	0.084	0.101	0.052-0.107 F/0.070-0.131 M
Percent Incidental Sediment Ingestion in Diet [6]	1.0%		1.0%
Home Range (km) [7]	1.9	3.4	1.0-2.8 km F/1.8-5.0 km M
Behavioral Modification Factors in the Exposure Assessment [8]			
Temporal Migration CorrectionFactor (1-%Annual TemporalDisplacement)	1		Resident
Temporal Hibernation/Asetivation Correction Factor (1-%Temporal Hib/Aset.)	1		Active Year Round
Habitat Use Factor (Temporal use factor %)	1		Riparian habitats preferred
Temporal Reproductive Period (Mating/Gestation/Birth)[8]	March to June		March to June

[1] Mitchell (1961); J. Bopp (1999), [2] Bleavins and Aulerich (1981); [3] Estimated from Nagy (1987) and USEPA (December, 1993); [4] Hamilton (1951), Hamilton (1940), Hamilton (1936); [5] Farrell and Wood (1968c In USEPA, December 1993); [6] Best Professional Judgement - based upon observations in Hamilton (1940); [7] Gerell (1970), Mitchell (1961); [8] Allen (1986).

302742

TABLE 3-24
EXPOSURE PARAMETERS FOR RIVER OTTER (*Lutra canadensis*)

	Exposure Parameters		Proximal Range Reported for Species
Common Name	River Otter		-
Genus	*Lutra*		-
Species	*canadensis*		-
Sex (M/F)	Female	Male	-
Age (Adult/Juv.)	Adult, Breeding		-
Male/Female Body Weight (kg) [1]	7.32	10.9	6.73-7.90 F/9.20-12.7 M
Total Daily Dietary Ingestion (kg/day wet wt.) [2]	0.900	0.900	0.7-1.1
Total Daily Dietary Ingestion (kg/day dry wt.) [3]	0.353	0.491	0.329-0.376 F/0.425-0.555 M
General Dietary Characterization [4]	Opportunistic Piscivore		-
Percent Diet Composition (% wet wt.) [4]			
Fish (Total Component)	*100%*		70-100%
Aquatic Invertebrates (Total Component)	*0.0%*		5-15%
Non-river Related Diet Sources	0.0%		0-25%
Water Consumption Rate (L/day) [5]	0.594	0.853	0.551-0.636 F/0.730-0.975 M
Percent Incidental Sediment Ingestion in Diet [6]	1.0%		1.0%
Home Range (km) [7]	10.0		1.5-22.3 Km
Behavioral Modification Factors in the Exposure Assessment [8]			
Temporal Migration CorrectionFactor (1-%Annual TemporalDisplacement)	1		Resident
Temporal Hibernation/Asetivation Correction Factor (1-%Temporal Hib/Aset.)	1		Active Year Round
Habitat Use Factor (Temporal use factor %)	1		Riparian habitats preferred
Temporal Reproductive Period (Mating/Gestation/Birth)[9]	March to March [10]		March to March

[1] Spinola et al., (undated), Bopp (1999), USEPA (December 1993); [2], [3] Harris (1968 In USEPA, December 1993), Penrod (1999); [4] Spinola (1999), Newell et al. (1987), Hamilton (1961); [5] Farrell and Wood (1968c In USEPA, December 1993); [6] Best Professional Judgement - based upon Liers (1951) In USEPA, 1993); [7] Spinola et al. (undated); [8] USEPA (December 1993a); [9] Hamilton and Eadie (1964); [10] Period between mating and birth extends for one full year due to delayed implantation of zygote.

302743

TABLE 3-25: SUMMARY OF $ADD_{Expected}$ AND EGG CONCENTRATIONS FOR FEMALE SWALLOW BASED ON TRI+ CONGENERS FOR PERIOD 1993 - 2018

Year	Average Dietary Dose (mg/Kg/day)				Average Egg Concentration (mg/Kg)			
	152	113	90	50	152	113	90	50
1993	1.50E+00	1.19E+00	9.69E-01	7.13E-01	3.51E+00	2.79E+00	2.26E+00	1.66E+00
1994	1.35E+00	1.12E+00	9.20E-01	6.68E-01	3.15E+00	2.61E+00	2.15E+00	1.56E+00
1995	1.30E+00	1.07E+00	8.62E-01	6.35E-01	3.04E+00	2.50E+00	2.01E+00	1.48E+00
Sex (M/F)	1.29E+00	1.03E+00	8.21E-01	6.11E-01	3.00E+00	2.40E+00	1.92E+00	1.43E+00
1997	1.22E+00	9.88E-01	7.95E-01	5.91E-01	2.84E+00	2.31E+00	1.86E+00	1.38E+00
1998	1.17E+00	9.61E-01	7.58E-01	5.59E-01	2.72E+00	2.24E+00	1.77E+00	1.30E+00
1999	1.11E+00	9.31E-01	7.30E-01	5.43E-01	2.58E+00	2.17E+00	1.70E+00	1.27E+00
2000	1.11E+00	8.93E-01	7.10E-01	5.27E-01	2.60E+00	2.08E+00	1.66E+00	1.23E+00
2001	1.09E+00	8.81E-01	6.89E-01	5.10E-01	2.54E+00	2.05E+00	1.61E+00	1.19E+00
2002	1.04E+00	8.50E-01	6.72E-01	5.01E-01	2.43E+00	1.98E+00	1.57E+00	1.17E+00
2003	9.77E-01	8.11E-01	6.57E-01	4.84E-01	2.28E+00	1.89E+00	1.53E+00	1.13E+00
2004	9.62E-01	7.82E-01	6.23E-01	4.62E-01	2.24E+00	1.82E+00	1.45E+00	1.08E+00
2005	9.36E-01	7.75E-01	6.00E-01	4.44E-01	2.18E+00	1.81E+00	1.40E+00	1.04E+00
2006	8.99E-01	7.52E-01	5.74E-01	4.25E-01	2.10E+00	1.75E+00	1.34E+00	9.91E-01
2007	8.87E-01	7.36E-01	5.59E-01	4.13E-01	2.07E+00	1.72E+00	1.30E+00	9.64E-01
2008	8.56E-01	7.09E-01	5.42E-01	4.02E-01	2.00E+00	1.65E+00	1.27E+00	9.38E-01
2009	8.38E-01	6.87E-01	5.30E-01	3.94E-01	1.96E+00	1.60E+00	1.24E+00	9.18E-01
2010	8.25E-01	6.74E-01	5.21E-01	3.86E-01	1.92E+00	1.57E+00	1.22E+00	9.01E-01
2011	7.90E-01	6.68E-01	5.03E-01	3.80E-01	1.84E+00	1.56E+00	1.17E+00	8.86E-01
2012	7.70E-01	6.53E-01	4.92E-01	3.71E-01	1.80E+00	1.52E+00	1.15E+00	8.66E-01
2013	7.54E-01	6.39E-01	4.77E-01	3.60E-01	1.76E+00	1.49E+00	1.11E+00	8.40E-01
2014	7.46E-01	6.23E-01	4.65E-01	3.51E-01	1.74E+00	1.45E+00	1.09E+00	8.19E-01
2015	7.24E-01	6.00E-01	4.56E-01	3.42E-01	1.69E+00	1.40E+00	1.06E+00	7.99E-01
2016	7.31E-01	5.84E-01	4.46E-01	3.36E-01	1.71E+00	1.36E+00	1.04E+00	7.84E-01
2017	7.22E-01	5.79E-01	4.41E-01	3.27E-01	1.68E+00	1.35E+00	1.03E+00	7.63E-01
2018	7.05E-01	5.76E-01	4.33E-01	3.20E-01	1.64E+00	1.35E+00	1.01E+00	7.46E-01

TABLE 3-26: SUMMARY OF $ADD_{95\%UCL}$ AND EGG CONCENTRATIONS FOR FEMALE SWALLOW BASED ON TRI+ CONGENERS FOR PERIOD 1993 - 2018

Year	95% UCL Dietary Dose (mg/Kg/day)				95% UCL Egg Concentration (mg/Kg)			
	152	113	90	50	152	113	90	50
1993	1.62E+00	1.28E+00	1.04E+00	7.65E-01	3.77E+00	2.99E+00	2.43E+00	1.79E+00
1994	1.45E+00	1.20E+00	9.87E-01	7.17E-01	3.37E+00	2.80E+00	2.30E+00	1.67E+00
1995	1.40E+00	1.15E+00	9.25E-01	6.81E-01	3.26E+00	2.68E+00	2.16E+00	1.59E+00
1996	1.38E+00	1.10E+00	8.80E-01	6.55E-01	3.22E+00	2.58E+00	2.05E+00	1.53E+00
1997	1.31E+00	1.06E+00	8.52E-01	6.33E-01	3.05E+00	2.47E+00	1.99E+00	1.48E+00
1998	1.25E+00	1.03E+00	8.12E-01	5.99E-01	2.92E+00	2.40E+00	1.89E+00	1.40E+00
1999	1.19E+00	9.99E-01	7.82E-01	5.81E-01	2.77E+00	2.33E+00	1.82E+00	1.36E+00
2000	1.19E+00	9.59E-01	7.60E-01	5.64E-01	2.79E+00	2.24E+00	1.77E+00	1.32E+00
2001	1.17E+00	9.46E-01	7.38E-01	5.46E-01	2.72E+00	2.21E+00	1.72E+00	1.27E+00
2002	1.12E+00	9.13E-01	7.20E-01	5.38E-01	2.61E+00	2.13E+00	1.68E+00	1.26E+00
2003	1.05E+00	8.71E-01	7.05E-01	5.19E-01	2.45E+00	2.03E+00	1.65E+00	1.21E+00
2004	1.04E+00	8.41E-01	6.69E-01	4.96E-01	2.42E+00	1.96E+00	1.56E+00	1.16E+00
2005	1.01E+00	8.33E-01	6.44E-01	4.77E-01	2.35E+00	1.94E+00	1.50E+00	1.11E+00
2006	9.66E-01	8.09E-01	6.17E-01	4.57E-01	2.25E+00	1.89E+00	1.44E+00	1.07E+00
2007	9.54E-01	7.92E-01	6.01E-01	4.44E-01	2.23E+00	1.85E+00	1.40E+00	1.04E+00
2008	9.23E-01	7.63E-01	5.83E-01	4.32E-01	2.15E+00	1.78E+00	1.36E+00	1.01E+00
2009	9.04E-01	7.40E-01	5.70E-01	4.23E-01	2.11E+00	1.73E+00	1.33E+00	9.87E-01
2010	8.87E-01	7.25E-01	5.60E-01	4.15E-01	2.07E+00	1.69E+00	1.31E+00	9.68E-01
2011	8.49E-01	7.18E-01	5.41E-01	4.09E-01	1.98E+00	1.68E+00	1.26E+00	9.53E-01
2012	8.28E-01	7.02E-01	5.28E-01	3.99E-01	1.93E+00	1.64E+00	1.23E+00	9.32E-01
2013	8.10E-01	6.87E-01	5.12E-01	3.87E-01	1.89E+00	1.60E+00	1.20E+00	9.04E-01
2014	8.02E-01	6.70E-01	5.00E-01	3.78E-01	1.87E+00	1.56E+00	1.17E+00	8.82E-01
2015	7.81E-01	6.46E-01	4.90E-01	3.68E-01	1.82E+00	1.51E+00	1.14E+00	8.60E-01
2016	7.91E-01	6.29E-01	4.80E-01	3.61E-01	1.85E+00	1.47E+00	1.12E+00	8.43E-01
2017	7.82E-01	6.25E-01	4.74E-01	3.52E-01	1.82E+00	1.46E+00	1.11E+00	8.21E-01
2018	7.63E-01	6.24E-01	4.66E-01	3.44E-01	1.78E+00	1.46E+00	1.09E+00	8.03E-01

TABLE 3-27: SUMMARY OF $ADD_{Expected}$ AND EGG CONCENTRATIONS FOR FEMALE MALLARD BASED ON TRI+ CONGENERS FOR PERIOD 1993 - 2018

	Average Dietary Dose (mg/Kg/day)				Average Egg Concentration (mg/Kg)			
Year	152	113	90	50	152	113	90	50
1993	5.69E-01	4.55E-01	3.67E-01	3.30E-01	5.26E+00	4.18E+00	3.39E+00	2.49E+00
1994	4.99E-01	4.12E-01	3.36E-01	2.96E-01	4.72E+00	3.91E+00	3.22E+00	2.34E+00
1995	4.21E-01	3.47E-01	2.95E-01	2.68E-01	4.57E+00	3.75E+00	3.02E+00	2.22E+00
1996	5.19E-01	3.58E-01	2.82E-01	2.46E-01	4.51E+00	3.61E+00	2.87E+00	2.14E+00
1997	4.37E-01	3.33E-01	2.63E-01	2.28E-01	4.27E+00	3.46E+00	2.78E+00	2.07E+00
1998	3.54E-01	2.87E-01	2.36E-01	2.08E-01	4.09E+00	3.36E+00	2.65E+00	1.96E+00
1999	3.13E-01	2.58E-01	2.15E-01	1.94E-01	3.87E+00	3.26E+00	2.56E+00	1.90E+00
2000	3.37E-01	2.52E-01	2.04E-01	1.79E-01	3.89E+00	3.13E+00	2.49E+00	1.84E+00
2001	3.56E-01	2.56E-01	1.97E-01	1.69E-01	3.81E+00	3.08E+00	2.41E+00	1.79E+00
2002	3.10E-01	2.40E-01	1.90E-01	1.63E-01	3.64E+00	2.97E+00	2.35E+00	1.75E+00
2003	2.75E-01	2.26E-01	1.83E-01	1.55E-01	3.42E+00	2.84E+00	2.30E+00	1.69E+00
2004	2.43E-01	1.98E-01	1.65E-01	1.45E-01	3.37E+00	2.74E+00	2.18E+00	1.62E+00
2005	2.39E-01	1.92E-01	1.56E-01	1.35E-01	3.27E+00	2.71E+00	2.10E+00	1.56E+00
2006	2.42E-01	1.91E-01	1.50E-01	1.28E-01	3.15E+00	2.63E+00	2.01E+00	1.49E+00
2007	2.25E-01	1.86E-01	1.45E-01	1.22E-01	3.10E+00	2.58E+00	1.96E+00	1.45E+00
2008	2.13E-01	1.73E-01	1.38E-01	1.17E-01	3.00E+00	2.48E+00	1.90E+00	1.41E+00
2009	1.90E-01	1.61E-01	1.31E-01	1.13E-01	2.93E+00	2.41E+00	1.86E+00	1.38E+00
2010	2.14E-01	1.66E-01	1.29E-01	1.09E-01	2.89E+00	2.36E+00	1.82E+00	1.35E+00
2011	1.96E-01	1.66E-01	1.27E-01	1.07E-01	2.77E+00	2.34E+00	1.76E+00	1.33E+00
2012	2.00E-01	1.65E-01	1.25E-01	1.04E-01	2.70E+00	2.29E+00	1.72E+00	1.30E+00
2013	2.18E-01	1.66E-01	1.23E-01	1.02E-01	2.64E+00	2.24E+00	1.67E+00	1.26E+00
2014	1.95E-01	1.58E-01	1.20E-01	9.98E-02	2.61E+00	2.18E+00	1.63E+00	1.23E+00
2015	1.88E-01	1.51E-01	1.16E-01	9.73E-02	2.53E+00	2.10E+00	1.60E+00	1.20E+00
2016	1.69E-01	1.36E-01	1.10E-01	9.44E-02	2.56E+00	2.04E+00	1.56E+00	1.18E+00
2017	1.63E-01	1.30E-01	1.06E-01	9.10E-02	2.53E+00	2.02E+00	1.54E+00	1.15E+00
2018	1.71E-01	1.33E-01	1.04E-01	8.77E-02	2.47E+00	2.02E+00	1.51E+00	1.12E+00

TABLE 3-28: SUMMARY OF $ADD_{95\%UCL}$ AND EGG CONCENTRATIONS FOR FEMALE MALLARD BASED ON TRI+ CONGENERS FOR PERIOD 1993 - 2018

Year	95% UCL Dietary Dose (mg/Kg/day)				95% UCL Egg Concentration (mg/Kg)			
	152	113	90	50	152	113	90	50
1993	6.10E-01	4.88E-01	3.94E-01	3.54E-01	5.65E+00	4.48E+00	3.65E+00	2.68E+00
1994	5.34E-01	4.42E-01	3.60E-01	3.17E-01	5.06E+00	4.19E+00	3.45E+00	2.51E+00
1995	4.51E-01	3.72E-01	3.14E-01	2.87E-01	4.89E+00	4.02E+00	3.24E+00	2.38E+00
1996	5.57E-01	3.84E-01	3.01E-01	2.64E-01	4.83E+00	3.87E+00	3.08E+00	2.29E+00
1997	4.68E-01	3.57E-01	2.82E-01	2.45E-01	4.57E+00	3.71E+00	2.98E+00	2.22E+00
1998	3.80E-01	3.07E-01	2.53E-01	2.23E-01	4.38E+00	3.60E+00	2.84E+00	2.10E+00
1999	3.36E-01	2.76E-01	2.30E-01	2.08E-01	4.16E+00	3.50E+00	2.74E+00	2.03E+00
2000	3.62E-01	2.70E-01	2.19E-01	1.92E-01	4.18E+00	3.36E+00	2.66E+00	1.97E+00
2001	3.82E-01	2.75E-01	2.11E-01	1.81E-01	4.08E+00	3.31E+00	2.58E+00	1.91E+00
2002	3.33E-01	2.58E-01	2.04E-01	1.75E-01	3.91E+00	3.19E+00	2.52E+00	1.88E+00
2003	2.95E-01	2.42E-01	1.96E-01	1.66E-01	3.68E+00	3.05E+00	2.47E+00	1.82E+00
2004	2.62E-01	2.12E-01	1.78E-01	1.55E-01	3.62E+00	2.94E+00	2.34E+00	1.74E+00
2005	2.57E-01	2.07E-01	1.67E-01	1.45E-01	3.52E+00	2.92E+00	2.25E+00	1.67E+00
2006	2.60E-01	2.06E-01	1.61E-01	1.37E-01	3.38E+00	2.83E+00	2.16E+00	1.60E+00
2007	2.42E-01	2.00E-01	1.56E-01	1.31E-01	3.34E+00	2.77E+00	2.10E+00	1.55E+00
2008	2.30E-01	1.87E-01	1.48E-01	1.26E-01	3.23E+00	2.67E+00	2.04E+00	1.51E+00
2009	2.04E-01	1.74E-01	1.41E-01	1.21E-01	3.16E+00	2.59E+00	2.00E+00	1.48E+00
2010	2.30E-01	1.79E-01	1.39E-01	1.17E-01	3.10E+00	2.54E+00	1.96E+00	1.45E+00
2011	2.11E-01	1.78E-01	1.36E-01	1.14E-01	2.97E+00	2.51E+00	1.89E+00	1.43E+00
2012	2.15E-01	1.77E-01	1.35E-01	1.12E-01	2.90E+00	2.46E+00	1.85E+00	1.40E+00
2013	2.34E-01	1.78E-01	1.32E-01	1.09E-01	2.83E+00	2.40E+00	1.79E+00	1.36E+00
2014	2.09E-01	1.70E-01	1.29E-01	1.07E-01	2.81E+00	2.35E+00	1.75E+00	1.32E+00
2015	2.02E-01	1.62E-01	1.25E-01	1.05E-01	2.73E+00	2.26E+00	1.72E+00	1.29E+00
2016	1.82E-01	1.47E-01	1.18E-01	1.01E-01	2.77E+00	2.20E+00	1.68E+00	1.26E+00
2017	1.77E-01	1.40E-01	1.14E-01	9.78E-02	2.74E+00	2.19E+00	1.66E+00	1.23E+00
2018	1.84E-01	1.44E-01	1.11E-01	9.43E-02	2.67E+00	2.18E+00	1.63E+00	1.20E+00

TABLE 3-29: SUMMARY OF $ADD_{Expected}$ AND EGG CONCENTRATIONS FOR FEMALE BELTED KINGFISHER BASED ON TRI+ CONGENERS FOR PERIOD 1993 - 2018

	Average Dietary Dose (mg/Kg/day)				Average Egg Concentration (mg/Kg)			
Year	152	113	90	50	152	113	90	50
1993	6.67E-01	4.68E-01	3.76E-01	3.34E-01	5.05E+01	3.54E+01	2.84E+01	2.53E+01
1994	5.22E-01	4.28E-01	3.45E-01	3.02E-01	3.94E+01	3.23E+01	2.60E+01	2.28E+01
1995	4.74E-01	3.66E-01	3.15E-01	2.73E-01	3.58E+01	2.76E+01	2.33E+01	2.06E+01
1996	5.44E-01	3.72E-01	2.96E-01	2.53E-01	4.11E+01	2.81E+01	2.18E+01	1.91E+01
1997	4.73E-01	3.49E-01	2.72E-01	2.35E-01	3.57E+01	2.63E+01	2.05E+01	1.77E+01
1998	3.77E-01	3.03E-01	2.53E-01	2.19E-01	2.84E+01	2.28E+01	1.91E+01	1.66E+01
1999	3.41E-01	2.76E-01	2.27E-01	2.00E-01	2.57E+01	2.08E+01	1.71E+01	1.51E+01
2000	3.35E-01	2.58E-01	2.13E-01	1.86E-01	2.53E+01	1.95E+01	1.60E+01	1.40E+01
2001	3.58E-01	2.64E-01	2.03E-01	1.74E-01	2.71E+01	1.99E+01	1.53E+01	1.32E+01
2002	3.24E-01	2.56E-01	1.98E-01	1.68E-01	2.45E+01	1.93E+01	1.49E+01	1.26E+01
2003	3.01E-01	2.37E-01	1.89E-01	1.60E-01	2.27E+01	1.78E+01	1.43E+01	1.21E+01
2004	2.55E-01	2.12E-01	1.75E-01	1.50E-01	1.92E+01	1.60E+01	1.32E+01	1.13E+01
2005	2.49E-01	2.02E-01	1.64E-01	1.40E-01	1.87E+01	1.52E+01	1.24E+01	1.06E+01
2006	2.75E-01	2.03E-01	1.56E-01	1.32E-01	2.07E+01	1.53E+01	1.18E+01	9.97E+00
2007	2.40E-01	1.96E-01	1.51E-01	1.27E-01	1.81E+01	1.48E+01	1.13E+01	9.55E+00
2008	2.26E-01	1.86E-01	1.46E-01	1.22E-01	1.70E+01	1.40E+01	1.10E+01	9.19E+00
2009	2.12E-01	1.74E-01	1.39E-01	1.17E-01	1.60E+01	1.31E+01	1.05E+01	8.81E+00
2010	2.23E-01	1.71E-01	1.33E-01	1.12E-01	1.68E+01	1.29E+01	9.99E+00	8.46E+00
2011	2.32E-01	1.76E-01	1.31E-01	1.10E-01	1.75E+01	1.33E+01	9.90E+00	8.26E+00
2012	2.19E-01	1.73E-01	1.31E-01	1.08E-01	1.65E+01	1.30E+01	9.85E+00	8.17E+00
2013	2.28E-01	1.73E-01	1.29E-01	1.06E-01	1.72E+01	1.30E+01	9.73E+00	8.02E+00
2014	2.17E-01	1.67E-01	1.25E-01	1.03E-01	1.64E+01	1.26E+01	9.41E+00	7.76E+00
2015	1.97E-01	1.58E-01	1.21E-01	1.00E-01	1.49E+01	1.19E+01	9.15E+00	7.57E+00
2016	1.78E-01	1.47E-01	1.17E-01	9.74E-02	1.34E+01	1.11E+01	8.79E+00	7.34E+00
2017	1.73E-01	1.40E-01	1.12E-01	9.37E-02	1.30E+01	1.05E+01	8.41E+00	7.06E+00
2018	1.72E-01	1.38E-01	1.08E-01	9.10E-02	1.30E+01	1.04E+01	8.16E+00	6.86E+00

TABLE 3-30: SUMMARY OF $ADD_{95\%UCL}$ AND EGG CONCENTRATIONS FOR FEMALE BELTED KINGFISHER BASED ON TRI+ CONGENERS FOR PERIOD 1993 - 2018

Year	95% UCL Dietary Dose (mg/Kg/day)				95% UCL Egg Concentration (mg/Kg)			
	152	113	90	50	152	113	90	50
1993	6.93E-01	4.87E-01	3.92E-01	3.47E-01	5.24E+01	3.68E+01	2.96E+01	2.63E+01
1994	5.42E-01	4.45E-01	3.59E-01	3.14E-01	4.09E+01	3.36E+01	2.71E+01	2.37E+01
1995	4.95E-01	3.82E-01	3.21E-01	2.84E-01	3.73E+01	2.88E+01	2.42E+01	2.14E+01
1996	5.65E-01	3.87E-01	3.02E-01	2.63E-01	4.27E+01	2.92E+01	2.28E+01	1.99E+01
1997	4.92E-01	3.64E-01	2.84E-01	2.44E-01	3.71E+01	2.75E+01	2.14E+01	1.84E+01
1998	3.94E-01	3.17E-01	2.65E-01	2.28E-01	2.97E+01	2.38E+01	1.99E+01	1.72E+01
1999	3.57E-01	2.89E-01	2.38E-01	2.08E-01	2.69E+01	2.18E+01	1.79E+01	1.57E+01
2000	3.51E-01	2.70E-01	2.23E-01	1.94E-01	2.64E+01	2.03E+01	1.68E+01	1.46E+01
2001	3.74E-01	2.76E-01	2.13E-01	1.82E-01	2.82E+01	2.08E+01	1.60E+01	1.37E+01
2002	3.40E-01	2.68E-01	2.07E-01	1.75E-01	2.56E+01	2.02E+01	1.56E+01	1.32E+01
2003	3.15E-01	2.48E-01	1.98E-01	1.67E-01	2.37E+01	1.87E+01	1.49E+01	1.26E+01
2004	2.68E-01	2.23E-01	1.84E-01	1.57E-01	2.01E+01	1.67E+01	1.38E+01	1.18E+01
2005	2.62E-01	2.13E-01	1.72E-01	1.47E-01	1.96E+01	1.60E+01	1.30E+01	1.11E+01
2006	2.88E-01	2.13E-01	1.64E-01	1.38E-01	2.17E+01	1.60E+01	1.23E+01	1.04E+01
2007	2.53E-01	2.06E-01	1.58E-01	1.33E-01	1.90E+01	1.55E+01	1.19E+01	9.98E+00
2008	2.38E-01	1.96E-01	1.53E-01	1.28E-01	1.79E+01	1.47E+01	1.15E+01	9.60E+00
2009	2.24E-01	1.84E-01	1.46E-01	1.23E-01	1.68E+01	1.38E+01	1.10E+01	9.21E+00
2010	2.35E-01	1.80E-01	1.40E-01	1.18E-01	1.76E+01	1.35E+01	1.05E+01	8.84E+00
2011	2.43E-01	1.85E-01	1.38E-01	1.15E-01	1.83E+01	1.39E+01	1.04E+01	8.64E+00
2012	2.30E-01	1.82E-01	1.38E-01	1.14E-01	1.73E+01	1.36E+01	1.03E+01	8.55E+00
2013	2.39E-01	1.82E-01	1.36E-01	1.12E-01	1.80E+01	1.37E+01	1.02E+01	8.39E+00
2014	2.28E-01	1.75E-01	1.31E-01	1.08E-01	1.71E+01	1.32E+01	9.86E+00	8.11E+00
2015	2.08E-01	1.66E-01	1.28E-01	1.05E-01	1.56E+01	1.25E+01	9.59E+00	7.92E+00
2016	1.89E-01	1.55E-01	1.23E-01	1.02E-01	1.42E+01	1.16E+01	9.22E+00	7.68E+00
2017	1.83E-01	1.48E-01	1.18E-01	9.84E-02	1.37E+01	1.11E+01	8.83E+00	7.39E+00
2018	1.83E-01	1.46E-01	1.14E-01	9.56E-02	1.37E+01	1.09E+01	8.56E+00	7.18E+00

TABLE 3-31: SUMMARY OF $ADD_{Expected}$ AND EGG CONCENTRATIONS FOR FEMALE GREAT BLUE HERON BASED ON TRI+ CONGENERS FOR PERIOD 1993 - 2018

	Average Dietary Dose (mg/Kg/day)				Average Egg Concentration (mg/Kg)			
Year	152	113	90	50	152	113	90	50
1993	2.61E-01	1.75E-01	1.40E-01	1.33E-01	4.98E+01	3.35E+01	2.68E+01	2.54E+01
1994	1.94E-01	1.59E-01	1.27E-01	1.19E-01	3.71E+01	3.03E+01	2.42E+01	2.27E+01
1995	1.73E-01	1.29E-01	1.13E-01	1.06E-01	3.29E+01	2.47E+01	2.12E+01	2.01E+01
1996	2.09E-01	1.35E-01	1.05E-01	9.66E-02	4.00E+01	2.57E+01	1.98E+01	1.84E+01
1997	1.76E-01	1.25E-01	9.65E-02	8.84E-02	3.37E+01	2.39E+01	1.84E+01	1.69E+01
1998	1.30E-01	1.03E-01	8.88E-02	8.22E-02	2.48E+01	1.97E+01	1.69E+01	1.57E+01
1999	1.15E-01	9.11E-02	7.69E-02	7.31E-02	2.19E+01	1.74E+01	1.47E+01	1.39E+01
2000	1.12E-01	8.39E-02	7.06E-02	6.67E-02	2.13E+01	1.60E+01	1.34E+01	1.27E+01
2001	1.25E-01	8.75E-02	6.68E-02	6.17E-02	2.38E+01	1.67E+01	1.27E+01	1.18E+01
2002	1.10E-01	8.52E-02	6.51E-02	5.88E-02	2.10E+01	1.62E+01	1.24E+01	1.12E+01
2003	1.01E-01	7.74E-02	6.13E-02	5.57E-02	1.93E+01	1.47E+01	1.17E+01	1.06E+01
2004	7.85E-02	6.62E-02	5.62E-02	5.17E-02	1.49E+01	1.26E+01	1.07E+01	9.86E+00
2005	7.67E-02	6.16E-02	5.16E-02	4.79E-02	1.46E+01	1.17E+01	9.82E+00	9.13E+00
2006	9.23E-02	6.32E-02	4.88E-02	4.47E-02	1.76E+01	1.20E+01	9.29E+00	8.52E+00
2007	7.51E-02	6.07E-02	4.69E-02	4.25E-02	1.43E+01	1.15E+01	8.93E+00	8.11E+00
2008	6.94E-02	5.70E-02	4.53E-02	4.07E-02	1.32E+01	1.08E+01	8.62E+00	7.75E+00
2009	6.31E-02	5.19E-02	4.24E-02	3.86E-02	1.20E+01	9.87E+00	8.06E+00	7.35E+00
2010	6.97E-02	5.12E-02	3.97E-02	3.66E-02	1.33E+01	9.73E+00	7.56E+00	6.96E+00
2011	7.59E-02	5.41E-02	4.01E-02	3.56E-02	1.45E+01	1.03E+01	7.63E+00	6.78E+00
2012	7.07E-02	5.31E-02	4.04E-02	3.54E-02	1.35E+01	1.01E+01	7.69E+00	6.75E+00
2013	7.62E-02	5.41E-02	4.04E-02	3.50E-02	1.45E+01	1.03E+01	7.69E+00	6.67E+00
2014	7.10E-02	5.17E-02	3.88E-02	3.37E-02	1.35E+01	9.84E+00	7.39E+00	6.42E+00
2015	6.19E-02	4.84E-02	3.75E-02	3.29E-02	1.18E+01	9.20E+00	7.14E+00	6.27E+00
2016	5.19E-02	4.36E-02	3.56E-02	3.17E-02	9.86E+00	8.29E+00	6.77E+00	6.04E+00
2017	4.94E-02	4.04E-02	3.33E-02	3.03E-02	9.39E+00	7.68E+00	6.34E+00	5.77E+00
2018	5.01E-02	3.93E-02	3.21E-02	2.93E-02	9.54E+00	7.47E+00	6.10E+00	5.59E+00

TABLE 3-32: SUMMARY OF $ADD_{95\%UCL}$ AND EGG CONCENTRATIONS FOR FEMALE GREAT BLUE HERON BASED ON TRI+ CONGENERS FOR PERIOD 1993 - 2018

Year	95% UCL Dietary Dose (mg/Kg/day)				95% UCL Egg Concentration (mg/Kg)			
	152	113	90	50	152	113	90	50
1993	2.68E-01	1.81E-01	1.45E-01	1.37E-01	5.11E+01	3.44E+01	2.76E+01	2.61E+01
1994	2.00E-01	1.64E-01	1.31E-01	1.22E-01	3.81E+01	3.12E+01	2.49E+01	2.33E+01
1995	1.78E-01	1.34E-01	1.15E-01	1.09E-01	3.39E+01	2.54E+01	2.19E+01	2.07E+01
1996	2.15E-01	1.39E-01	1.08E-01	9.96E-02	4.10E+01	2.64E+01	2.04E+01	1.90E+01
1997	1.82E-01	1.30E-01	1.00E-01	9.12E-02	3.46E+01	2.46E+01	1.89E+01	1.74E+01
1998	1.35E-01	1.07E-01	9.21E-02	8.48E-02	2.55E+01	2.02E+01	1.74E+01	1.61E+01
1999	1.19E-01	9.49E-02	8.00E-02	7.56E-02	2.25E+01	1.79E+01	1.51E+01	1.44E+01
2000	1.16E-01	8.74E-02	7.34E-02	6.90E-02	2.19E+01	1.64E+01	1.38E+01	1.31E+01
2001	1.29E-01	9.10E-02	6.95E-02	6.39E-02	2.45E+01	1.71E+01	1.31E+01	1.21E+01
2002	1.14E-01	8.87E-02	6.78E-02	6.09E-02	2.16E+01	1.67E+01	1.28E+01	1.15E+01
2003	1.05E-01	8.06E-02	6.39E-02	5.77E-02	1.98E+01	1.52E+01	1.20E+01	1.09E+01
2004	8.22E-02	6.93E-02	5.86E-02	5.37E-02	1.54E+01	1.30E+01	1.10E+01	1.02E+01
2005	8.03E-02	6.45E-02	5.40E-02	4.98E-02	1.50E+01	1.21E+01	1.01E+01	9.40E+00
2006	9.59E-02	6.61E-02	5.10E-02	4.65E-02	1.81E+01	1.24E+01	9.57E+00	8.77E+00
2007	7.85E-02	6.35E-02	4.90E-02	4.42E-02	1.47E+01	1.19E+01	9.19E+00	8.34E+00
2008	7.28E-02	5.97E-02	4.74E-02	4.23E-02	1.36E+01	1.12E+01	8.89E+00	7.98E+00
2009	6.63E-02	5.46E-02	4.44E-02	4.02E-02	1.24E+01	1.02E+01	8.31E+00	7.56E+00
2010	7.29E-02	5.37E-02	4.17E-02	3.81E-02	1.37E+01	1.00E+01	7.79E+00	7.17E+00
2011	7.90E-02	5.67E-02	4.20E-02	3.71E-02	1.49E+01	1.06E+01	7.85E+00	6.98E+00
2012	7.38E-02	5.56E-02	4.23E-02	3.69E-02	1.39E+01	1.04E+01	7.92E+00	6.94E+00
2013	7.92E-02	5.66E-02	4.22E-02	3.65E-02	1.49E+01	1.06E+01	7.92E+00	6.87E+00
2014	7.40E-02	5.41E-02	4.06E-02	3.51E-02	1.39E+01	1.01E+01	7.61E+00	6.61E+00
2015	6.47E-02	5.06E-02	3.93E-02	3.43E-02	1.21E+01	9.48E+00	7.35E+00	6.45E+00
2016	5.47E-02	4.58E-02	3.73E-02	3.31E-02	1.02E+01	8.55E+00	6.98E+00	6.22E+00
2017	5.21E-02	4.26E-02	3.50E-02	3.16E-02	9.69E+00	7.92E+00	6.54E+00	5.94E+00
2018	5.28E-02	4.14E-02	3.37E-02	3.06E-02	9.84E+00	7.70E+00	6.28E+00	5.75E+00

TABLE 3-33: SUMMARY OF $ADD_{Expected}$ AND EGG CONCENTRATIONS FOR FEMALE EAGLE BASED ON TRI+ CONGENERS FOR PERIOD 1993 - 2018

	Average Dietary Dose (mg/Kg/day)				Average Egg Concentration (mg/Kg)			
Year	152	113	90	50	152	113	90	50
1993	1.90E+00	1.27E+00	2.91E-01	2.74E-01	4.17E+02	2.79E+02	6.40E+01	6.03E+01
1994	1.38E+00	1.11E+00	2.65E-01	2.46E-01	3.02E+02	2.44E+02	5.82E+01	5.40E+01
1995	1.18E+00	9.86E-01	2.39E-01	2.21E-01	2.59E+02	2.17E+02	5.25E+01	4.86E+01
1996	1.40E+00	9.20E-01	2.18E-01	2.00E-01	3.08E+02	2.02E+02	4.78E+01	4.39E+01
1997	1.27E+00	8.89E-01	2.03E-01	1.85E-01	2.80E+02	1.95E+02	4.47E+01	4.06E+01
1998	1.00E+00	8.09E-01	1.87E-01	1.69E-01	2.20E+02	1.78E+02	4.12E+01	3.72E+01
1999	8.65E-01	6.77E-01	1.69E-01	1.54E-01	1.90E+02	1.49E+02	3.71E+01	3.39E+01
2000	8.12E-01	6.16E-01	1.53E-01	1.42E-01	1.78E+02	1.35E+02	3.36E+01	3.11E+01
2001	9.22E-01	6.24E-01	1.45E-01	1.32E-01	2.03E+02	1.37E+02	3.19E+01	2.90E+01
2002	8.43E-01	6.17E-01	1.41E-01	1.26E-01	1.85E+02	1.36E+02	3.10E+01	2.77E+01
2003	7.52E-01	5.67E-01	1.34E-01	1.20E-01	1.65E+02	1.25E+02	2.95E+01	2.63E+01
2004	6.06E-01	5.03E-01	1.25E-01	1.12E-01	1.33E+02	1.11E+02	2.74E+01	2.46E+01
2005	5.78E-01	4.63E-01	1.15E-01	1.04E-01	1.27E+02	1.02E+02	2.53E+01	2.28E+01
2006	6.51E-01	4.57E-01	1.09E-01	9.77E-02	1.43E+02	1.00E+02	2.39E+01	2.15E+01
2007	6.00E-01	4.47E-01	1.05E-01	9.30E-02	1.32E+02	9.83E+01	2.31E+01	2.04E+01
2008	5.58E-01	4.30E-01	1.01E-01	8.91E-02	1.23E+02	9.45E+01	2.23E+01	1.96E+01
2009	4.84E-01	3.93E-01	9.58E-02	8.47E-02	1.06E+02	8.64E+01	2.10E+01	1.86E+01
2010	5.09E-01	3.76E-01	9.09E-02	8.06E-02	1.12E+02	8.25E+01	2.00E+01	1.77E+01
2011	5.70E-01	3.99E-01	9.01E-02	7.82E-02	1.25E+02	8.78E+01	1.98E+01	1.72E+01
2012	5.09E-01	3.90E-01	8.93E-02	7.65E-02	1.12E+02	8.57E+01	1.96E+01	1.68E+01
2013	5.56E-01	4.03E-01	9.17E-02	7.82E-02	1.22E+02	8.85E+01	2.01E+01	1.72E+01
2014	5.05E-01	3.82E-01	8.67E-02	7.39E-02	1.11E+02	8.39E+01	1.90E+01	1.62E+01
2015	4.67E-01	3.64E-01	8.41E-02	7.20E-02	1.03E+02	8.00E+01	1.85E+01	1.58E+01
2016	4.27E-01	3.39E-01	8.05E-02	6.98E-02	9.38E+01	7.45E+01	1.77E+01	1.53E+01
2017	3.84E-01	3.13E-01	7.63E-02	6.77E-02	8.44E+01	6.87E+01	1.68E+01	1.49E+01
2018	3.75E-01	2.97E-01	7.23E-02	6.39E-02	8.24E+01	6.52E+01	1.59E+01	1.40E+01

TABLE 3-34: SUMMARY OF $ADD_{95\%UCL}$ AND EGG CONCENTRATIONS FOR FEMALE EAGLE BASED ON TRI+ CONGENERS FOR PERIOD 1993 - 2018

Year	95% UCL Dietary Dose (mg/Kg/day)				95% UCL Egg Concentration (mg/Kg)			
	152	113	90	50	152	113	90	50
1993	1.94E+00	1.30E+00	2.96E-01	2.79E-01	4.26E+02	2.85E+02	6.50E+01	6.12E+01
1994	1.40E+00	1.13E+00	2.69E-01	2.50E-01	3.09E+02	2.49E+02	5.92E+01	5.48E+01
1995	1.21E+00	1.01E+00	2.43E-01	2.25E-01	2.65E+02	2.21E+02	5.34E+01	4.93E+01
1996	1.43E+00	9.40E-01	2.21E-01	2.03E-01	3.15E+02	2.06E+02	4.86E+01	4.46E+01
1997	1.30E+00	9.08E-01	2.07E-01	1.88E-01	2.86E+02	1.99E+02	4.54E+01	4.12E+01
1998	1.02E+00	8.27E-01	1.90E-01	1.72E-01	2.25E+02	1.82E+02	4.18E+01	3.78E+01
1999	8.84E-01	6.92E-01	1.72E-01	1.57E-01	1.94E+02	1.52E+02	3.77E+01	3.45E+01
2000	8.29E-01	6.30E-01	1.56E-01	1.44E-01	1.82E+02	1.38E+02	3.42E+01	3.16E+01
2001	9.42E-01	6.38E-01	1.48E-01	1.34E-01	2.07E+02	1.40E+02	3.25E+01	2.95E+01
2002	8.62E-01	6.31E-01	1.44E-01	1.28E-01	1.89E+02	1.39E+02	3.16E+01	2.82E+01
2003	7.68E-01	5.80E-01	1.37E-01	1.22E-01	1.69E+02	1.27E+02	3.00E+01	2.67E+01
2004	6.20E-01	5.14E-01	1.27E-01	1.14E-01	1.36E+02	1.13E+02	2.79E+01	2.50E+01
2005	5.91E-01	4.73E-01	1.17E-01	1.06E-01	1.30E+02	1.04E+02	2.57E+01	2.32E+01
2006	6.65E-01	4.67E-01	1.11E-01	9.94E-02	1.46E+02	1.03E+02	2.44E+01	2.18E+01
2007	6.14E-01	4.57E-01	1.07E-01	9.46E-02	1.35E+02	1.00E+02	2.35E+01	2.08E+01
2008	5.70E-01	4.40E-01	1.03E-01	9.06E-02	1.25E+02	9.66E+01	2.27E+01	1.99E+01
2009	4.95E-01	4.02E-01	9.75E-02	8.61E-02	1.09E+02	8.83E+01	2.14E+01	1.89E+01
2010	5.20E-01	3.84E-01	9.25E-02	8.21E-02	1.14E+02	8.44E+01	2.03E+01	1.80E+01
2011	5.83E-01	4.09E-01	9.18E-02	7.96E-02	1.28E+02	8.97E+01	2.02E+01	1.75E+01
2012	5.21E-01	3.99E-01	9.09E-02	7.78E-02	1.14E+02	8.77E+01	2.00E+01	1.71E+01
2013	5.68E-01	4.12E-01	9.33E-02	7.95E-02	1.25E+02	9.05E+01	2.05E+01	1.75E+01
2014	5.17E-01	3.91E-01	8.82E-02	7.53E-02	1.14E+02	8.58E+01	1.94E+01	1.65E+01
2015	4.78E-01	3.72E-01	8.56E-02	7.32E-02	1.05E+02	8.18E+01	1.88E+01	1.61E+01
2016	4.36E-01	3.47E-01	8.19E-02	7.11E-02	9.58E+01	7.62E+01	1.80E+01	1.56E+01
2017	3.93E-01	3.20E-01	7.76E-02	6.89E-02	8.63E+01	7.02E+01	1.71E+01	1.51E+01
2018	3.84E-01	3.04E-01	7.36E-02	6.50E-02	8.43E+01	6.67E+01	1.62E+01	1.43E+01

TABLE 3-35: SUMMARY OF $ADD_{Expected}$ AND EGG CONCENTRATIONS FOR FEMALE TREE SWALLOW FOR THE PERIOD 1993 - 2018 ON TEQ BASIS

Year	Total Average Dietary Dose (mg/Kg/day)				Average Egg Concentration (mg/Kg)			
	152	113	90	50	152	113	90	50
1993	2.08E-04	1.65E-04	1.34E-04	9.88E-05	1.70E-03	1.35E-03	1.10E-03	8.07E-04
1994	1.87E-04	1.55E-04	1.27E-04	9.26E-05	1.53E-03	1.27E-03	1.04E-03	7.56E-04
1995	1.81E-04	1.49E-04	1.19E-04	8.80E-05	1.48E-03	1.21E-03	9.75E-04	7.19E-04
1996	1.78E-04	1.43E-04	1.14E-04	8.47E-05	1.46E-03	1.17E-03	9.29E-04	6.91E-04
1997	1.69E-04	1.37E-04	1.10E-04	8.19E-05	1.38E-03	1.12E-03	9.00E-04	6.69E-04
1998	1.62E-04	1.33E-04	1.05E-04	7.74E-05	1.32E-03	1.09E-03	8.57E-04	6.32E-04
1999	1.53E-04	1.29E-04	1.01E-04	7.52E-05	1.25E-03	1.05E-03	8.26E-04	6.14E-04
2000	1.54E-04	1.24E-04	9.84E-05	7.30E-05	1.26E-03	1.01E-03	8.04E-04	5.96E-04
2001	1.51E-04	1.22E-04	9.55E-05	7.07E-05	1.23E-03	9.97E-04	7.80E-04	5.77E-04
2002	1.44E-04	1.18E-04	9.32E-05	6.95E-05	1.18E-03	9.62E-04	7.61E-04	5.67E-04
2003	1.35E-04	1.12E-04	9.11E-05	6.70E-05	1.11E-03	9.17E-04	7.44E-04	5.47E-04
2004	1.33E-04	1.08E-04	8.63E-05	6.40E-05	1.09E-03	8.85E-04	7.05E-04	5.23E-04
2005	1.30E-04	1.07E-04	8.31E-05	6.16E-05	1.06E-03	8.77E-04	6.79E-04	5.03E-04
2006	1.25E-04	1.04E-04	7.95E-05	5.89E-05	1.02E-03	8.51E-04	6.49E-04	4.81E-04
2007	1.23E-04	1.02E-04	7.74E-05	5.72E-05	1.00E-03	8.33E-04	6.32E-04	4.67E-04
2008	1.19E-04	9.82E-05	7.51E-05	5.57E-05	9.69E-04	8.02E-04	6.14E-04	4.55E-04
2009	1.16E-04	9.53E-05	7.35E-05	5.46E-05	9.49E-04	7.78E-04	6.00E-04	4.45E-04
2010	1.14E-04	9.34E-05	7.22E-05	5.35E-05	9.33E-04	7.63E-04	5.90E-04	4.37E-04
2011	1.10E-04	9.26E-05	6.98E-05	5.26E-05	8.94E-04	7.56E-04	5.70E-04	4.30E-04
2012	1.07E-04	9.05E-05	6.81E-05	5.15E-05	8.72E-04	7.39E-04	5.56E-04	4.20E-04
2013	1.04E-04	8.85E-05	6.61E-05	4.99E-05	8.53E-04	7.23E-04	5.39E-04	4.07E-04
2014	1.03E-04	8.63E-05	6.45E-05	4.87E-05	8.44E-04	7.05E-04	5.26E-04	3.97E-04
2015	1.00E-04	8.32E-05	6.32E-05	4.75E-05	8.19E-04	6.79E-04	5.16E-04	3.88E-04
2016	1.01E-04	8.09E-05	6.19E-05	4.65E-05	8.27E-04	6.60E-04	5.05E-04	3.80E-04
2017	1.00E-04	8.02E-05	6.11E-05	4.53E-05	8.17E-04	6.55E-04	4.99E-04	3.70E-04
2018	9.77E-05	7.99E-05	6.00E-05	4.43E-05	7.98E-04	6.52E-04	4.90E-04	3.62E-04

TABLE 3-36: SUMMARY OF $ADD_{95\%UCL}$ AND EGG CONCENTRATIONS FOR FEMALE TREE SWALLOW FOR THE PERIOD 1993 - 2018 ON TEQ BASIS

Year	Total 95% UCL Dietary Dose (mg/Kg/day)				95% UCL Egg Concentration (mg/Kg)			
	152	113	90	50	152	113	90	50
1993	2.24E-04	1.78E-04	1.44E-04	1.06E-04	1.83E-03	1.45E-03	1.18E-03	8.66E-04
1994	2.00E-04	1.66E-04	1.37E-04	9.94E-05	1.64E-03	1.36E-03	1.12E-03	8.12E-04
1995	1.94E-04	1.59E-04	1.28E-04	9.43E-05	1.58E-03	1.30E-03	1.05E-03	7.70E-04
1996	1.91E-04	1.53E-04	1.22E-04	9.07E-05	1.56E-03	1.25E-03	9.96E-04	7.41E-04
1997	1.81E-04	1.47E-04	1.18E-04	8.78E-05	1.48E-03	1.20E-03	9.64E-04	7.17E-04
1998	1.73E-04	1.43E-04	1.13E-04	8.30E-05	1.42E-03	1.16E-03	9.19E-04	6.78E-04
1999	1.65E-04	1.38E-04	1.08E-04	8.06E-05	1.34E-03	1.13E-03	8.85E-04	6.58E-04
2000	1.65E-04	1.33E-04	1.05E-04	7.81E-05	1.35E-03	1.09E-03	8.60E-04	6.38E-04
2001	1.62E-04	1.31E-04	1.02E-04	7.57E-05	1.32E-03	1.07E-03	8.35E-04	6.18E-04
2002	1.55E-04	1.26E-04	9.98E-05	7.45E-05	1.26E-03	1.03E-03	8.15E-04	6.09E-04
2003	1.46E-04	1.21E-04	9.77E-05	7.20E-05	1.19E-03	9.85E-04	7.98E-04	5.88E-04
2004	1.43E-04	1.16E-04	9.27E-05	6.87E-05	1.17E-03	9.51E-04	7.57E-04	5.61E-04
2005	1.40E-04	1.15E-04	8.93E-05	6.61E-05	1.14E-03	9.43E-04	7.29E-04	5.40E-04
2006	1.34E-04	1.12E-04	8.55E-05	6.33E-05	1.09E-03	9.15E-04	6.98E-04	5.17E-04
2007	1.32E-04	1.10E-04	8.32E-05	6.15E-05	1.08E-03	8.96E-04	6.80E-04	5.03E-04
2008	1.28E-04	1.06E-04	8.08E-05	5.99E-05	1.05E-03	8.64E-04	6.60E-04	4.89E-04
2009	1.25E-04	1.03E-04	7.90E-05	5.86E-05	1.02E-03	8.38E-04	6.45E-04	4.79E-04
2010	1.23E-04	1.01E-04	7.76E-05	5.75E-05	1.00E-03	8.21E-04	6.34E-04	4.70E-04
2011	1.18E-04	9.96E-05	7.50E-05	5.66E-05	9.61E-04	8.13E-04	6.12E-04	4.62E-04
2012	1.15E-04	9.74E-05	7.32E-05	5.54E-05	9.37E-04	7.95E-04	5.98E-04	4.52E-04
2013	1.12E-04	9.52E-05	7.10E-05	5.37E-05	9.17E-04	7.78E-04	5.80E-04	4.38E-04
2014	1.11E-04	9.29E-05	6.93E-05	5.24E-05	9.07E-04	7.58E-04	5.66E-04	4.28E-04
2015	1.08E-04	8.96E-05	6.79E-05	5.11E-05	8.84E-04	7.31E-04	5.55E-04	4.17E-04
2016	1.10E-04	8.72E-05	6.65E-05	5.01E-05	8.96E-04	7.12E-04	5.43E-04	4.09E-04
2017	1.08E-04	8.66E-05	6.57E-05	4.88E-05	8.85E-04	7.07E-04	5.37E-04	3.98E-04
2018	1.06E-04	8.65E-05	6.45E-05	4.77E-05	8.64E-04	7.06E-04	5.27E-04	3.90E-04

TABLE 3-37: SUMMARY OF $ADD_{Expected}$ AND EGG CONCENTRATIONS FOR FEMALE MALLARD ON A TEQ BASIS FOR PERIOD 1993 - 2018

	Average Dietary Dose (mg/Kg/day)				Average Egg Concentration (mg/Kg)			
Year	152	113	90	50	152	113	90	50
1993	2.27E-04	1.82E-04	1.49E-04	2.23E-04	6.81E-03	5.40E-03	4.39E-03	3.23E-03
1994	1.97E-04	1.62E-04	1.32E-04	1.93E-04	6.11E-03	5.06E-03	4.16E-03	3.03E-03
1995	1.54E-04	1.27E-04	1.19E-04	1.48E-04	5.91E-03	4.86E-03	3.90E-03	2.87E-03
1996	2.13E-04	1.37E-04	1.07E-04	2.11E-04	5.83E-03	4.66E-03	3.72E-03	2.77E-03
1997	1.70E-04	1.25E-04	9.66E-05	1.65E-04	5.52E-03	4.47E-03	3.60E-03	2.68E-03
1998	1.24E-04	9.96E-05	8.70E-05	1.18E-04	5.28E-03	4.35E-03	3.43E-03	2.53E-03
1999	1.05E-04	8.46E-05	7.95E-05	9.75E-05	5.01E-03	4.22E-03	3.31E-03	2.46E-03
2000	1.19E-04	8.38E-05	7.20E-05	1.12E-04	5.04E-03	4.04E-03	3.22E-03	2.38E-03
2001	1.31E-04	8.76E-05	6.70E-05	1.26E-04	4.93E-03	3.99E-03	3.12E-03	2.31E-03
2002	1.07E-04	8.01E-05	6.40E-05	1.01E-04	4.71E-03	3.85E-03	3.04E-03	2.27E-03
2003	9.14E-05	7.45E-05	6.02E-05	8.50E-05	4.42E-03	3.67E-03	2.98E-03	2.19E-03
2004	7.39E-05	6.00E-05	5.58E-05	6.61E-05	4.35E-03	3.54E-03	2.82E-03	2.09E-03
2005	7.32E-05	5.74E-05	5.13E-05	6.57E-05	4.24E-03	3.51E-03	2.72E-03	2.01E-03
2006	7.77E-05	5.85E-05	4.80E-05	7.10E-05	4.07E-03	3.40E-03	2.60E-03	1.92E-03
2007	6.87E-05	5.65E-05	4.56E-05	6.14E-05	4.02E-03	3.33E-03	2.53E-03	1.87E-03
2008	6.40E-05	5.10E-05	4.34E-05	5.68E-05	3.88E-03	3.21E-03	2.45E-03	1.82E-03
2009	5.13E-05	4.55E-05	4.11E-05	4.34E-05	3.80E-03	3.11E-03	2.40E-03	1.78E-03
2010	6.63E-05	4.92E-05	3.92E-05	5.97E-05	3.73E-03	3.05E-03	2.36E-03	1.75E-03
2011	5.85E-05	4.94E-05	3.82E-05	5.21E-05	3.58E-03	3.02E-03	2.28E-03	1.72E-03
2012	6.25E-05	4.99E-05	3.74E-05	5.66E-05	3.49E-03	2.96E-03	2.23E-03	1.68E-03
2013	7.39E-05	5.15E-05	3.67E-05	6.90E-05	3.41E-03	2.89E-03	2.16E-03	1.63E-03
2014	6.09E-05	4.80E-05	3.61E-05	5.51E-05	3.38E-03	2.82E-03	2.11E-03	1.59E-03
2015	5.85E-05	4.55E-05	3.52E-05	5.28E-05	3.28E-03	2.72E-03	2.06E-03	1.55E-03
2016	4.66E-05	3.83E-05	3.39E-05	3.97E-05	3.31E-03	2.64E-03	2.02E-03	1.52E-03
2017	4.42E-05	3.49E-05	3.26E-05	3.72E-05	3.27E-03	2.62E-03	2.00E-03	1.48E-03
2018	4.97E-05	3.66E-05	3.12E-05	4.33E-05	3.19E-03	2.61E-03	1.96E-03	1.45E-03

TABLE 3-38: SUMMARY OF $ADD_{95\%UCL}$ AND EGG CONCENTRATIONS FOR FEMALE MALLARD ON A TEQ BASIS FOR PERIOD 1993 - 2018

Year	95% UCL Dietary Dose (mg/Kg/day)				95% UCL Egg Concentration (mg/Kg)			
	152	113	90	50	152	113	90	50
1993	2.43E-04	1.95E-04	1.57E-04	1.54E-04	7.31E-03	5.80E-03	4.72E-03	3.46E-03
1994	2.11E-04	1.74E-04	1.41E-04	1.35E-04	6.54E-03	5.43E-03	4.47E-03	3.25E-03
1995	1.65E-04	1.36E-04	1.19E-04	1.20E-04	6.33E-03	5.20E-03	4.19E-03	3.08E-03
1996	2.29E-04	1.47E-04	1.14E-04	1.08E-04	6.25E-03	5.00E-03	3.98E-03	2.96E-03
1997	1.82E-04	1.34E-04	1.05E-04	9.85E-05	5.92E-03	4.79E-03	3.86E-03	2.87E-03
1998	1.34E-04	1.07E-04	9.06E-05	8.84E-05	5.67E-03	4.66E-03	3.68E-03	2.71E-03
1999	1.12E-04	9.08E-05	7.92E-05	8.06E-05	5.38E-03	4.52E-03	3.54E-03	2.63E-03
2000	1.27E-04	9.02E-05	7.40E-05	7.27E-05	5.40E-03	4.34E-03	3.44E-03	2.55E-03
2001	1.41E-04	9.42E-05	7.13E-05	6.75E-05	5.28E-03	4.28E-03	3.34E-03	2.47E-03
2002	1.15E-04	8.61E-05	6.82E-05	6.44E-05	5.06E-03	4.13E-03	3.26E-03	2.44E-03
2003	9.82E-05	8.01E-05	6.49E-05	6.04E-05	4.76E-03	3.94E-03	3.19E-03	2.35E-03
2004	7.95E-05	6.45E-05	5.64E-05	5.59E-05	4.69E-03	3.81E-03	3.03E-03	2.25E-03
2005	7.88E-05	6.18E-05	5.22E-05	5.13E-05	4.56E-03	3.77E-03	2.92E-03	2.16E-03
2006	8.36E-05	6.31E-05	5.05E-05	4.79E-05	4.37E-03	3.66E-03	2.79E-03	2.07E-03
2007	7.39E-05	6.08E-05	4.84E-05	4.54E-05	4.32E-03	3.59E-03	2.72E-03	2.01E-03
2008	6.89E-05	5.50E-05	4.54E-05	4.32E-05	4.18E-03	3.45E-03	2.64E-03	1.96E-03
2009	5.53E-05	4.90E-05	4.18E-05	4.08E-05	4.09E-03	3.35E-03	2.58E-03	1.92E-03
2010	7.14E-05	5.30E-05	4.14E-05	3.88E-05	4.01E-03	3.28E-03	2.53E-03	1.88E-03
2011	6.30E-05	5.32E-05	4.12E-05	3.80E-05	3.85E-03	3.25E-03	2.45E-03	1.85E-03
2012	6.73E-05	5.37E-05	4.13E-05	3.73E-05	3.75E-03	3.18E-03	2.39E-03	1.81E-03
2013	7.95E-05	5.55E-05	4.09E-05	3.66E-05	3.67E-03	3.11E-03	2.32E-03	1.75E-03
2014	6.55E-05	5.17E-05	3.98E-05	3.60E-05	3.63E-03	3.03E-03	2.26E-03	1.71E-03
2015	6.30E-05	4.91E-05	3.84E-05	3.51E-05	3.53E-03	2.93E-03	2.22E-03	1.67E-03
2016	5.04E-05	4.12E-05	3.52E-05	3.37E-05	3.58E-03	2.85E-03	2.17E-03	1.64E-03
2017	4.78E-05	3.77E-05	3.27E-05	3.23E-05	3.54E-03	2.83E-03	2.15E-03	1.59E-03
2018	5.37E-05	3.96E-05	3.20E-05	3.08E-05	3.45E-03	2.83E-03	2.11E-03	1.56E-03

TABLE 3-39: SUMMARY OF $ADD_{Expected}$ AND EGG CONCENTRATIONS FOR FEMALE BELTED KINGFISHER FOR THE PERIOD 1993 - 2018 ON TEQ BASIS

	Average Dietary Dose (mg/Kg/day)				Average Egg Concentration (mg/Kg)			
Year	152	113	90	50	152	113	90	50
1993	1.20E-04	8.37E-05	6.71E-05	6.03E-05	5.63E-03	3.91E-03	3.13E-03	2.83E-03
1994	9.32E-05	7.64E-05	6.14E-05	5.44E-05	4.34E-03	3.56E-03	2.86E-03	2.55E-03
1995	8.43E-05	6.47E-05	6.02E-05	4.90E-05	3.93E-03	3.01E-03	2.55E-03	2.29E-03
1996	9.76E-05	6.61E-05	5.69E-05	4.53E-05	4.57E-03	3.08E-03	2.39E-03	2.12E-03
1997	8.44E-05	6.19E-05	4.82E-05	4.20E-05	3.94E-03	2.88E-03	2.24E-03	1.96E-03
1998	6.64E-05	5.32E-05	4.48E-05	3.92E-05	3.08E-03	2.47E-03	2.08E-03	1.83E-03
1999	5.98E-05	4.83E-05	3.99E-05	3.56E-05	2.77E-03	2.23E-03	1.85E-03	1.66E-03
2000	5.87E-05	4.50E-05	3.72E-05	3.29E-05	2.72E-03	2.08E-03	1.72E-03	1.53E-03
2001	6.32E-05	4.62E-05	3.55E-05	3.08E-05	2.94E-03	2.14E-03	1.64E-03	1.43E-03
2002	5.70E-05	4.48E-05	3.46E-05	2.96E-05	2.64E-03	2.07E-03	1.60E-03	1.38E-03
2003	5.28E-05	4.14E-05	3.30E-05	2.82E-05	2.44E-03	1.91E-03	1.52E-03	1.31E-03
2004	4.41E-05	3.68E-05	3.06E-05	2.64E-05	2.03E-03	1.69E-03	1.41E-03	1.22E-03
2005	4.30E-05	3.49E-05	2.85E-05	2.47E-05	1.98E-03	1.61E-03	1.31E-03	1.14E-03
2006	4.83E-05	3.51E-05	2.71E-05	2.32E-05	2.23E-03	1.62E-03	1.25E-03	1.07E-03
2007	4.17E-05	3.40E-05	2.61E-05	2.22E-05	1.92E-03	1.56E-03	1.20E-03	1.03E-03
2008	3.91E-05	3.22E-05	2.53E-05	2.14E-05	1.80E-03	1.48E-03	1.16E-03	9.88E-04
2009	3.65E-05	3.00E-05	2.40E-05	2.04E-05	1.67E-03	1.38E-03	1.10E-03	9.45E-04
2010	3.87E-05	2.95E-05	2.29E-05	1.96E-05	1.78E-03	1.35E-03	1.05E-03	9.04E-04
2011	4.05E-05	3.05E-05	2.27E-05	1.91E-05	1.87E-03	1.40E-03	1.04E-03	8.83E-04
2012	3.83E-05	2.99E-05	2.27E-05	1.89E-05	1.76E-03	1.37E-03	1.04E-03	8.75E-04
2013	4.00E-05	3.00E-05	2.24E-05	1.86E-05	1.85E-03	1.38E-03	1.03E-03	8.60E-04
2014	3.79E-05	2.89E-05	2.17E-05	1.80E-05	1.75E-03	1.33E-03	9.97E-04	8.31E-04
2015	3.43E-05	2.73E-05	2.10E-05	1.75E-05	1.58E-03	1.26E-03	9.67E-04	8.11E-04
2016	3.06E-05	2.53E-05	2.02E-05	1.70E-05	1.40E-03	1.16E-03	9.27E-04	7.85E-04
2017	2.95E-05	2.40E-05	1.92E-05	1.63E-05	1.35E-03	1.10E-03	8.83E-04	7.54E-04
2018	2.95E-05	2.35E-05	1.86E-05	1.59E-05	1.36E-03	1.08E-03	8.55E-04	7.32E-04

TABLE 3-40: SUMMARY OF $ADD_{95\%UCL}$ AND EGG CONCENTRATIONS FOR FEMALE BELTED KINGFISHER FOR THE PERIOD 1993 - 2018 ON TEQ BASIS

Year	95% UCL Dietary Dose (mg/Kg/day)				95% UCL Egg Concentration (mg/Kg)			
	152	113	90	50	152	113	90	50
1993	1.25E-04	8.70E-05	1.69E-04	1.42E-04	5.82E-03	4.05E-03	3.25E-03	2.93E-03
1994	9.68E-05	7.95E-05	1.59E-04	1.34E-04	4.50E-03	3.69E-03	2.97E-03	2.64E-03
1995	8.80E-05	6.77E-05	1.50E-04	1.27E-04	4.08E-03	3.13E-03	2.65E-03	2.38E-03
1996	1.02E-04	6.90E-05	1.44E-04	1.21E-04	4.73E-03	3.19E-03	2.48E-03	2.20E-03
1997	8.80E-05	6.48E-05	1.39E-04	1.16E-04	4.08E-03	2.99E-03	2.33E-03	2.03E-03
1998	6.97E-05	5.59E-05	1.34E-04	1.12E-04	3.21E-03	2.57E-03	2.16E-03	1.90E-03
1999	6.31E-05	5.09E-05	1.29E-04	1.08E-04	2.89E-03	2.33E-03	1.92E-03	1.72E-03
2000	6.18E-05	4.75E-05	1.23E-04	1.03E-04	2.83E-03	2.17E-03	1.79E-03	1.59E-03
2001	6.62E-05	4.86E-05	1.18E-04	9.95E-05	3.05E-03	2.22E-03	1.71E-03	1.49E-03
2002	6.01E-05	4.73E-05	1.17E-04	9.66E-05	2.75E-03	2.16E-03	1.67E-03	1.43E-03
2003	5.56E-05	4.37E-05	1.13E-04	9.39E-05	2.54E-03	1.99E-03	1.59E-03	1.36E-03
2004	4.70E-05	3.91E-05	1.13E-04	9.25E-05	2.12E-03	1.77E-03	1.47E-03	1.27E-03
2005	4.59E-05	3.71E-05	1.11E-04	9.01E-05	2.07E-03	1.68E-03	1.37E-03	1.19E-03
2006	5.10E-05	3.74E-05	1.04E-04	8.69E-05	2.32E-03	1.69E-03	1.30E-03	1.12E-03
2007	4.43E-05	3.61E-05	1.01E-04	8.43E-05	2.00E-03	1.63E-03	1.25E-03	1.07E-03
2008	4.18E-05	3.43E-05	1.02E-04	8.24E-05	1.88E-03	1.55E-03	1.21E-03	1.03E-03
2009	3.92E-05	3.21E-05	1.03E-04	8.24E-05	1.75E-03	1.44E-03	1.15E-03	9.85E-04
2010	4.12E-05	3.15E-05	9.56E-05	7.99E-05	1.86E-03	1.42E-03	1.10E-03	9.43E-04
2011	4.29E-05	3.25E-05	9.03E-05	7.73E-05	1.95E-03	1.46E-03	1.09E-03	9.21E-04
2012	4.06E-05	3.18E-05	8.88E-05	7.43E-05	1.84E-03	1.44E-03	1.09E-03	9.12E-04
2013	4.23E-05	3.19E-05	8.67E-05	7.25E-05	1.93E-03	1.45E-03	1.08E-03	8.97E-04
2014	4.02E-05	3.07E-05	8.49E-05	7.07E-05	1.83E-03	1.39E-03	1.04E-03	8.66E-04
2015	3.64E-05	2.91E-05	8.34E-05	7.04E-05	1.65E-03	1.31E-03	1.01E-03	8.45E-04
2016	3.30E-05	2.71E-05	8.68E-05	6.96E-05	1.47E-03	1.21E-03	9.69E-04	8.19E-04
2017	3.19E-05	2.58E-05	8.65E-05	6.87E-05	1.42E-03	1.15E-03	9.24E-04	7.86E-04
2018	3.18E-05	2.54E-05	8.27E-05	6.87E-05	1.42E-03	1.13E-03	8.94E-04	7.64E-04

TABLE 3-41: SUMMARY OF $ADD_{Expected}$ AND EGG CONCENTRATIONS FOR FEMALE GREAT BLUE HERON FOR THE PERIOD 1993 - 2018 ON TEQ BASIS

	Average Dietary Dose (mg/Kg/day)				Average Egg Concentration (mg/Kg)			
Year	152	113	90	50	152	113	90	50
1993	3.09E-05	2.11E-05	1.69E-05	1.59E-05	3.68E-03	2.48E-03	1.98E-03	1.88E-03
1994	2.32E-05	1.92E-05	1.53E-05	1.42E-05	2.74E-03	2.24E-03	1.79E-03	1.68E-03
1995	2.06E-05	1.57E-05	1.47E-05	1.27E-05	2.43E-03	1.82E-03	1.57E-03	1.49E-03
1996	2.48E-05	1.63E-05	1.38E-05	1.16E-05	2.95E-03	1.90E-03	1.46E-03	1.36E-03
1997	2.10E-05	1.52E-05	1.17E-05	1.07E-05	2.49E-03	1.77E-03	1.36E-03	1.25E-03
1998	1.56E-05	1.26E-05	1.08E-05	9.91E-06	1.83E-03	1.45E-03	1.25E-03	1.16E-03
1999	1.38E-05	1.12E-05	9.42E-06	8.85E-06	1.62E-03	1.28E-03	1.08E-03	1.03E-03
2000	1.34E-05	1.03E-05	8.66E-06	8.10E-06	1.57E-03	1.18E-03	9.94E-04	9.40E-04
2001	1.49E-05	1.07E-05	8.22E-06	7.51E-06	1.76E-03	1.23E-03	9.41E-04	8.70E-04
2002	1.32E-05	1.04E-05	8.00E-06	7.16E-06	1.55E-03	1.20E-03	9.17E-04	8.28E-04
2003	1.21E-05	9.51E-06	7.55E-06	6.79E-06	1.42E-03	1.09E-03	8.63E-04	7.85E-04
2004	9.50E-06	8.21E-06	6.94E-06	6.32E-06	1.10E-03	9.31E-04	7.90E-04	7.29E-04
2005	9.29E-06	7.65E-06	6.40E-06	5.86E-06	1.08E-03	8.66E-04	7.26E-04	6.75E-04
2006	1.11E-05	7.82E-06	6.06E-06	5.48E-06	1.30E-03	8.89E-04	6.87E-04	6.30E-04
2007	9.08E-06	7.52E-06	5.83E-06	5.22E-06	1.06E-03	8.54E-04	6.60E-04	5.99E-04
2008	8.41E-06	7.08E-06	5.63E-06	5.00E-06	9.76E-04	8.01E-04	6.37E-04	5.73E-04
2009	7.66E-06	6.48E-06	5.28E-06	4.75E-06	8.87E-04	7.30E-04	5.96E-04	5.43E-04
2010	8.41E-06	6.39E-06	4.97E-06	4.51E-06	9.80E-04	7.19E-04	5.59E-04	5.15E-04
2011	9.13E-06	6.71E-06	5.00E-06	4.39E-06	1.07E-03	7.61E-04	5.64E-04	5.01E-04
2012	8.52E-06	6.59E-06	5.03E-06	4.36E-06	9.95E-04	7.47E-04	5.68E-04	4.99E-04
2013	9.14E-06	6.69E-06	5.02E-06	4.31E-06	1.07E-03	7.62E-04	5.68E-04	4.93E-04
2014	8.53E-06	6.40E-06	4.83E-06	4.15E-06	9.99E-04	7.27E-04	5.46E-04	4.74E-04
2015	7.48E-06	6.01E-06	4.67E-06	4.05E-06	8.71E-04	6.80E-04	5.27E-04	4.63E-04
2016	6.30E-06	5.45E-06	4.44E-06	3.91E-06	7.29E-04	6.13E-04	5.00E-04	4.46E-04
2017	6.01E-06	5.08E-06	4.17E-06	3.74E-06	6.94E-04	5.68E-04	4.68E-04	4.26E-04
2018	6.09E-06	4.94E-06	4.02E-06	3.62E-06	7.05E-04	5.52E-04	4.51E-04	4.13E-04

TABLE 3-42: SUMMARY OF $ADD_{95\%UCL}$ AND EGG CONCENTRATIONS FOR FEMALE GREAT BLUE HERON FOR THE PERIOD 1993 - 2018 ON TEQ BASIS

Year	95% UCL Dietary Dose (mg/Kg/day)				95% UCL Egg Concentration (mg/Kg)			
	152	113	90	50	152	113	90	50
1993	3.20E-05	2.17E-05	1.74E-05	1.63E-05	3.78E-03	2.54E-03	2.04E-03	1.93E-03
1994	2.41E-05	1.97E-05	1.58E-05	1.46E-05	2.81E-03	2.30E-03	1.84E-03	1.72E-03
1995	2.15E-05	1.62E-05	1.39E-05	1.30E-05	2.50E-03	1.88E-03	1.61E-03	1.53E-03
1996	2.58E-05	1.68E-05	1.30E-05	1.19E-05	3.03E-03	1.95E-03	1.51E-03	1.40E-03
1997	2.19E-05	1.57E-05	1.21E-05	1.10E-05	2.55E-03	1.82E-03	1.40E-03	1.28E-03
1998	1.64E-05	1.30E-05	1.12E-05	1.02E-05	1.89E-03	1.50E-03	1.29E-03	1.19E-03
1999	1.46E-05	1.16E-05	9.73E-06	9.12E-06	1.67E-03	1.32E-03	1.12E-03	1.06E-03
2000	1.42E-05	1.07E-05	8.96E-06	8.35E-06	1.62E-03	1.22E-03	1.02E-03	9.68E-04
2001	1.57E-05	1.11E-05	8.50E-06	7.75E-06	1.81E-03	1.27E-03	9.68E-04	8.95E-04
2002	1.40E-05	1.08E-05	8.29E-06	7.39E-06	1.60E-03	1.24E-03	9.44E-04	8.52E-04
2003	1.29E-05	9.90E-06	7.83E-06	7.02E-06	1.47E-03	1.12E-03	8.89E-04	8.07E-04
2004	1.02E-05	8.57E-06	7.21E-06	6.54E-06	1.14E-03	9.60E-04	8.15E-04	7.50E-04
2005	9.97E-06	8.01E-06	6.66E-06	6.07E-06	1.11E-03	8.92E-04	7.48E-04	6.95E-04
2006	1.18E-05	8.18E-06	6.31E-06	5.68E-06	1.34E-03	9.16E-04	7.07E-04	6.48E-04
2007	9.71E-06	7.86E-06	6.07E-06	5.42E-06	1.09E-03	8.79E-04	6.79E-04	6.16E-04
2008	9.06E-06	7.42E-06	5.88E-06	5.19E-06	1.01E-03	8.26E-04	6.57E-04	5.90E-04
2009	8.32E-06	6.82E-06	5.52E-06	4.93E-06	9.14E-04	7.53E-04	6.14E-04	5.59E-04
2010	9.04E-06	6.71E-06	5.19E-06	4.69E-06	1.01E-03	7.41E-04	5.75E-04	5.30E-04
2011	9.72E-06	7.04E-06	5.22E-06	4.57E-06	1.10E-03	7.84E-04	5.80E-04	5.16E-04
2012	9.10E-06	6.89E-06	5.24E-06	4.54E-06	1.02E-03	7.69E-04	5.85E-04	5.13E-04
2013	9.72E-06	7.01E-06	5.23E-06	4.48E-06	1.10E-03	7.85E-04	5.85E-04	5.08E-04
2014	9.10E-06	6.71E-06	5.03E-06	4.32E-06	1.03E-03	7.50E-04	5.63E-04	4.88E-04
2015	8.01E-06	6.30E-06	4.87E-06	4.21E-06	8.97E-04	7.00E-04	5.43E-04	4.77E-04
2016	6.87E-06	5.73E-06	4.63E-06	4.07E-06	7.52E-04	6.32E-04	5.16E-04	4.60E-04
2017	6.58E-06	5.35E-06	4.36E-06	3.89E-06	7.16E-04	5.86E-04	4.83E-04	4.39E-04
2018	6.63E-06	5.22E-06	4.20E-06	3.77E-06	7.27E-04	5.69E-04	4.64E-04	4.25E-04

TABLE 3-43: SUMMARY OF $ADD_{Expected}$ AND EGG CONCENTRATIONS FOR FEMALE EAGLE FOR THE PERIOD 1993 - 2018 ON TEQ BASIS

	Average Dietary Dose (mg/Kg/day)				Average Egg Concentration (mg/Kg)			
Year	152	113	90	50	152	113	90	50
1993	3.60E-04	2.41E-04	5.52E-05	5.21E-05	5.37E-02	3.59E-02	8.23E-03	7.76E-03
1994	2.61E-04	2.10E-04	5.03E-05	4.66E-05	3.89E-02	3.14E-02	7.49E-03	6.95E-03
1995	2.24E-04	1.87E-04	4.53E-05	4.19E-05	3.34E-02	2.79E-02	6.76E-03	6.25E-03
1996	2.66E-04	1.75E-04	4.13E-05	3.79E-05	3.96E-02	2.60E-02	6.15E-03	5.65E-03
1997	2.42E-04	1.69E-04	3.86E-05	3.50E-05	3.60E-02	2.51E-02	5.75E-03	5.22E-03
1998	1.90E-04	1.53E-04	3.55E-05	3.22E-05	2.83E-02	2.29E-02	5.30E-03	4.79E-03
1999	1.64E-04	1.28E-04	3.20E-05	2.93E-05	2.45E-02	1.92E-02	4.77E-03	4.37E-03
2000	1.54E-04	1.17E-04	2.90E-05	2.69E-05	2.30E-02	1.74E-02	4.33E-03	4.00E-03
2001	1.75E-04	1.18E-04	2.76E-05	2.51E-05	2.61E-02	1.76E-02	4.11E-03	3.74E-03
2002	1.60E-04	1.17E-04	2.68E-05	2.39E-05	2.39E-02	1.75E-02	4.00E-03	3.57E-03
2003	1.43E-04	1.08E-04	2.55E-05	2.27E-05	2.13E-02	1.60E-02	3.80E-03	3.38E-03
2004	1.15E-04	9.54E-05	2.36E-05	2.12E-05	1.72E-02	1.42E-02	3.53E-03	3.16E-03
2005	1.10E-04	8.78E-05	2.18E-05	1.97E-05	1.63E-02	1.31E-02	3.25E-03	2.94E-03
2006	1.23E-04	8.67E-05	2.07E-05	1.85E-05	1.84E-02	1.29E-02	3.08E-03	2.76E-03
2007	1.14E-04	8.49E-05	1.99E-05	1.76E-05	1.70E-02	1.27E-02	2.97E-03	2.63E-03
2008	1.06E-04	8.16E-05	1.92E-05	1.69E-05	1.58E-02	1.22E-02	2.87E-03	2.52E-03
2009	9.19E-05	7.46E-05	1.82E-05	1.61E-05	1.37E-02	1.11E-02	2.71E-03	2.39E-03
2010	9.65E-05	7.13E-05	1.72E-05	1.53E-05	1.44E-02	1.06E-02	2.57E-03	2.28E-03
2011	1.08E-04	7.58E-05	1.71E-05	1.48E-05	1.61E-02	1.13E-02	2.55E-03	2.21E-03
2012	9.66E-05	7.40E-05	1.69E-05	1.45E-05	1.44E-02	1.10E-02	2.53E-03	2.16E-03
2013	1.05E-04	7.64E-05	1.74E-05	1.48E-05	1.57E-02	1.14E-02	2.59E-03	2.21E-03
2014	9.59E-05	7.24E-05	1.64E-05	1.40E-05	1.43E-02	1.08E-02	2.45E-03	2.09E-03
2015	8.86E-05	6.91E-05	1.60E-05	1.37E-05	1.32E-02	1.03E-02	2.38E-03	2.04E-03
2016	8.10E-05	6.43E-05	1.53E-05	1.33E-05	1.21E-02	9.59E-03	2.28E-03	1.98E-03
2017	7.29E-05	5.93E-05	1.45E-05	1.28E-05	1.09E-02	8.84E-03	2.16E-03	1.91E-03
2018	7.11E-05	5.63E-05	1.37E-05	1.21E-05	1.06E-02	8.39E-03	2.04E-03	1.81E-03

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TABLE 3-44: SUMMARY OF $ADD_{95\%UCL}$ AND EGG CONCENTRATIONS FOR FEMALE EAGLE FOR THE PERIOD 1993 - 2018 ON TEQ BASIS

Year	95% UCL Dietary Dose (mg/Kg/day)				95% UCL Egg Concentration (mg/Kg)			
	152	113	90	50	152	113	90	50
1993	3.68E-04	2.46E-04	5.61E-05	5.29E-05	5.48E-02	3.67E-02	8.37E-03	7.88E-03
1994	2.67E-04	2.15E-04	5.11E-05	4.74E-05	3.97E-02	3.21E-02	7.62E-03	7.06E-03
1995	2.29E-04	1.91E-04	4.61E-05	4.26E-05	3.41E-02	2.85E-02	6.87E-03	6.35E-03
1996	2.72E-04	1.78E-04	4.20E-05	3.85E-05	4.05E-02	2.66E-02	6.26E-03	5.74E-03
1997	2.47E-04	1.72E-04	3.92E-05	3.56E-05	3.68E-02	2.57E-02	5.85E-03	5.31E-03
1998	1.94E-04	1.57E-04	3.61E-05	3.27E-05	2.89E-02	2.34E-02	5.39E-03	4.87E-03
1999	1.68E-04	1.31E-04	3.26E-05	2.98E-05	2.50E-02	1.96E-02	4.85E-03	4.44E-03
2000	1.57E-04	1.20E-04	2.96E-05	2.73E-05	2.35E-02	1.78E-02	4.41E-03	4.07E-03
2001	1.79E-04	1.21E-04	2.81E-05	2.55E-05	2.67E-02	1.80E-02	4.18E-03	3.80E-03
2002	1.63E-04	1.20E-04	2.73E-05	2.43E-05	2.44E-02	1.78E-02	4.07E-03	3.63E-03
2003	1.46E-04	1.10E-04	2.59E-05	2.31E-05	2.17E-02	1.64E-02	3.86E-03	3.44E-03
2004	1.18E-04	9.76E-05	2.41E-05	2.16E-05	1.75E-02	1.45E-02	3.59E-03	3.22E-03
2005	1.12E-04	8.98E-05	2.22E-05	2.01E-05	1.67E-02	1.34E-02	3.31E-03	2.99E-03
2006	1.26E-04	8.87E-05	2.10E-05	1.89E-05	1.88E-02	1.32E-02	3.14E-03	2.81E-03
2007	1.16E-04	8.68E-05	2.03E-05	1.80E-05	1.74E-02	1.29E-02	3.03E-03	2.68E-03
2008	1.08E-04	8.34E-05	1.96E-05	1.72E-05	1.61E-02	1.24E-02	2.92E-03	2.56E-03
2009	9.40E-05	7.63E-05	1.85E-05	1.63E-05	1.40E-02	1.14E-02	2.76E-03	2.44E-03
2010	9.86E-05	7.29E-05	1.76E-05	1.56E-05	1.47E-02	1.09E-02	2.62E-03	2.32E-03
2011	1.11E-04	7.75E-05	1.74E-05	1.51E-05	1.65E-02	1.16E-02	2.60E-03	2.25E-03
2012	9.88E-05	7.57E-05	1.72E-05	1.48E-05	1.47E-02	1.13E-02	2.57E-03	2.20E-03
2013	1.08E-04	7.82E-05	1.77E-05	1.51E-05	1.61E-02	1.17E-02	2.64E-03	2.25E-03
2014	9.81E-05	7.41E-05	1.67E-05	1.43E-05	1.46E-02	1.10E-02	2.50E-03	2.13E-03
2015	9.06E-05	7.06E-05	1.62E-05	1.39E-05	1.35E-02	1.05E-02	2.42E-03	2.07E-03
2016	8.28E-05	6.58E-05	1.55E-05	1.35E-05	1.23E-02	9.81E-03	2.32E-03	2.01E-03
2017	7.45E-05	6.06E-05	1.47E-05	1.31E-05	1.11E-02	9.04E-03	2.20E-03	1.95E-03
2018	7.28E-05	5.76E-05	1.40E-05	1.23E-05	1.08E-02	8.59E-03	2.08E-03	1.84E-03

TABLE 3-45: SUMMARY OF $ADD_{Expected}$ FOR FEMALE BAT BASED ON TRI+ PREDICTIONS FOR THE PERIOD 1993 - 2018

	Total Average Dietary Dose (mg/Kg/day)			
Year	152	113	90	50
1993	6.18E-01	4.90E-01	3.98E-01	2.93E-01
1994	5.54E-01	4.59E-01	3.78E-01	2.75E-01
1995	5.36E-01	4.41E-01	3.54E-01	2.61E-01
1996	5.29E-01	4.23E-01	3.37E-01	2.51E-01
1997	5.01E-01	4.06E-01	3.27E-01	2.43E-01
1998	4.79E-01	3.95E-01	3.11E-01	2.30E-01
1999	4.55E-01	3.83E-01	3.00E-01	2.23E-01
2000	4.57E-01	3.67E-01	2.92E-01	2.16E-01
2001	4.47E-01	3.62E-01	2.83E-01	2.10E-01
2002	4.27E-01	3.49E-01	2.76E-01	2.06E-01
2003	4.01E-01	3.33E-01	2.70E-01	1.99E-01
2004	3.95E-01	3.21E-01	2.56E-01	1.90E-01
2005	3.84E-01	3.18E-01	2.46E-01	1.83E-01
2006	3.69E-01	3.09E-01	2.36E-01	1.75E-01
2007	3.64E-01	3.03E-01	2.29E-01	1.70E-01
2008	3.52E-01	2.91E-01	2.23E-01	1.65E-01
2009	3.44E-01	2.82E-01	2.18E-01	1.62E-01
2010	3.39E-01	2.77E-01	2.14E-01	1.59E-01
2011	3.25E-01	2.74E-01	2.07E-01	1.56E-01
2012	3.16E-01	2.68E-01	2.02E-01	1.53E-01
2013	3.10E-01	2.62E-01	1.96E-01	1.48E-01
2014	3.06E-01	2.56E-01	1.91E-01	1.44E-01
2015	2.97E-01	2.47E-01	1.87E-01	1.41E-01
2016	3.00E-01	2.40E-01	1.83E-01	1.38E-01
2017	2.97E-01	2.38E-01	1.81E-01	1.34E-01
2018	2.89E-01	2.37E-01	1.78E-01	1.31E-01

TABLE 3-46: SUMMARY OF $ADD_{95\%UCL}$ FOR FEMALE BAT BASED ON TRI+ PREDICTIONS FOR THE PERIOD 1993 - 2018

Year	Total 95% UCL Dietary Dose (mg/Kg/day)			
	152	113	90	50
1993	6.64E-01	5.26E-01	4.28E-01	3.14E-01
1994	5.94E-01	4.92E-01	4.05E-01	2.95E-01
1995	5.74E-01	4.72E-01	3.80E-01	2.80E-01
1996	5.67E-01	4.54E-01	3.61E-01	2.69E-01
1997	5.37E-01	4.35E-01	3.50E-01	2.60E-01
1998	5.14E-01	4.23E-01	3.34E-01	2.46E-01
1999	4.88E-01	4.10E-01	3.21E-01	2.39E-01
2000	4.90E-01	3.94E-01	3.12E-01	2.32E-01
2001	4.79E-01	3.89E-01	3.03E-01	2.24E-01
2002	4.59E-01	3.75E-01	2.96E-01	2.21E-01
2003	4.32E-01	3.58E-01	2.90E-01	2.13E-01
2004	4.25E-01	3.45E-01	2.75E-01	2.04E-01
2005	4.14E-01	3.42E-01	2.65E-01	1.96E-01
2006	3.97E-01	3.32E-01	2.53E-01	1.88E-01
2007	3.92E-01	3.25E-01	2.47E-01	1.82E-01
2008	3.79E-01	3.14E-01	2.39E-01	1.77E-01
2009	3.71E-01	3.04E-01	2.34E-01	1.74E-01
2010	3.64E-01	2.98E-01	2.30E-01	1.70E-01
2011	3.49E-01	2.95E-01	2.22E-01	1.68E-01
2012	3.40E-01	2.89E-01	2.17E-01	1.64E-01
2013	3.33E-01	2.82E-01	2.10E-01	1.59E-01
2014	3.29E-01	2.75E-01	2.05E-01	1.55E-01
2015	3.21E-01	2.65E-01	2.01E-01	1.51E-01
2016	3.25E-01	2.59E-01	1.97E-01	1.48E-01
2017	3.21E-01	2.57E-01	1.95E-01	1.45E-01
2018	3.13E-01	2.56E-01	1.91E-01	1.41E-01

TABLE 3-47: SUMMARY OF $ADD_{Expected}$ FOR FEMALE RACCOON BASED ON TRI+ PREDICTIONS FOR THE PERIOD 1993 - 2018

	Average Dietary Dose (mg/Kg/day)			
Year	152	113	90	50
1993	1.13E-01	8.84E-02	7.17E-02	5.36E-02
1994	9.99E-02	8.27E-02	6.78E-02	5.01E-02
1995	9.60E-02	7.86E-02	7.10E-02	4.74E-02
1996	9.59E-02	7.58E-02	6.80E-02	4.54E-02
1997	9.03E-02	7.26E-02	5.83E-02	4.38E-02
1998	8.52E-02	7.00E-02	5.54E-02	4.14E-02
1999	8.04E-02	6.75E-02	5.32E-02	4.00E-02
2000	8.07E-02	6.46E-02	5.15E-02	3.87E-02
2001	7.95E-02	6.39E-02	5.00E-02	3.73E-02
2002	7.57E-02	6.17E-02	4.87E-02	3.66E-02
2003	7.11E-02	5.88E-02	4.76E-02	3.53E-02
2004	6.93E-02	5.64E-02	4.51E-02	3.37E-02
2005	6.74E-02	5.57E-02	4.33E-02	3.24E-02
2006	6.53E-02	5.42E-02	4.14E-02	3.09E-02
2007	6.40E-02	5.30E-02	4.03E-02	3.00E-02
2008	6.17E-02	5.10E-02	3.91E-02	2.92E-02
2009	6.02E-02	4.94E-02	3.82E-02	2.86E-02
2010	5.94E-02	4.84E-02	3.74E-02	2.80E-02
2011	5.73E-02	4.81E-02	3.62E-02	2.75E-02
2012	5.57E-02	4.70E-02	3.54E-02	2.69E-02
2013	5.48E-02	4.60E-02	3.44E-02	2.61E-02
2014	5.40E-02	4.49E-02	3.36E-02	2.54E-02
2015	5.22E-02	4.32E-02	3.29E-02	2.48E-02
2016	5.23E-02	4.19E-02	3.21E-02	2.43E-02
2017	5.16E-02	4.14E-02	3.17E-02	2.37E-02
2018	5.04E-02	4.12E-02	3.11E-02	2.31E-02

TABLE 3-48: SUMMARY OF $ADD_{95\%UCL}$ FOR FEMALE RACCOON BASED ON TRI+ PREDICTIONS FOR THE PERIOD 1993 - 2018

Year	95% UCL Dietary Dose (mg/Kg/day)			
	152	113	90	50
1993	1.21E-01	9.49E-02	7.70E-02	5.75E-02
1994	1.07E-01	8.87E-02	7.28E-02	5.38E-02
1995	1.03E-01	8.45E-02	6.81E-02	5.09E-02
1996	1.03E-01	8.15E-02	6.48E-02	4.88E-02
1997	9.70E-02	7.81E-02	6.26E-02	4.71E-02
1998	9.19E-02	7.54E-02	5.97E-02	4.45E-02
1999	8.72E-02	7.30E-02	5.72E-02	4.30E-02
2000	8.73E-02	7.00E-02	5.55E-02	4.16E-02
2001	8.56E-02	6.92E-02	5.39E-02	4.03E-02
2002	8.19E-02	6.68E-02	5.26E-02	3.96E-02
2003	7.71E-02	6.37E-02	5.14E-02	3.82E-02
2004	7.55E-02	6.14E-02	4.88E-02	3.65E-02
2005	7.36E-02	6.07E-02	4.70E-02	3.51E-02
2006	7.11E-02	5.90E-02	4.50E-02	3.36E-02
2007	6.96E-02	5.78E-02	4.38E-02	3.27E-02
2008	6.75E-02	5.57E-02	4.26E-02	3.18E-02
2009	6.62E-02	5.41E-02	4.16E-02	3.11E-02
2010	6.49E-02	5.30E-02	4.08E-02	3.04E-02
2011	6.23E-02	5.25E-02	3.95E-02	2.99E-02
2012	6.07E-02	5.12E-02	3.86E-02	2.93E-02
2013	5.95E-02	5.02E-02	3.74E-02	2.84E-02
2014	5.88E-02	4.89E-02	3.65E-02	2.77E-02
2015	5.71E-02	4.72E-02	3.58E-02	2.70E-02
2016	5.77E-02	4.59E-02	3.50E-02	2.65E-02
2017	5.70E-02	4.55E-02	3.45E-02	2.58E-02
2018	5.56E-02	4.55E-02	3.39E-02	2.52E-02

TABLE 3-49: SUMMARY OF $ADD_{Expected}$ FOR FEMALE MINK BASED ON TRI+ PREDICTIONS FOR THE PERIOD 1993 - 2018

	Average Dietary Dose (mg/Kg/day)			
Year	152	113	90	50
1993	1.37E-01	9.75E-02	7.84E-02	6.79E-02
1994	1.09E-01	8.94E-02	7.22E-02	6.16E-02
1995	9.99E-02	7.78E-02	6.62E-02	5.60E-02
1996	1.12E-01	7.84E-02	6.24E-02	5.22E-02
1997	9.86E-02	7.38E-02	5.79E-02	4.88E-02
1998	8.09E-02	6.53E-02	5.41E-02	4.56E-02
1999	7.38E-02	6.02E-02	4.91E-02	4.20E-02
2000	7.29E-02	5.65E-02	4.63E-02	3.93E-02
2001	7.67E-02	5.74E-02	4.44E-02	3.71E-02
2002	7.01E-02	5.56E-02	4.32E-02	3.58E-02
2003	6.51E-02	5.17E-02	4.15E-02	3.42E-02
2004	5.68E-02	4.70E-02	3.86E-02	3.21E-02
2005	5.54E-02	4.52E-02	3.63E-02	3.03E-02
2006	5.96E-02	4.50E-02	3.46E-02	2.86E-02
2007	5.33E-02	4.37E-02	3.34E-02	2.74E-02
2008	5.04E-02	4.15E-02	3.24E-02	2.65E-02
2009	4.77E-02	3.91E-02	3.10E-02	2.55E-02
2010	4.95E-02	3.85E-02	2.98E-02	2.45E-02
2011	5.06E-02	3.93E-02	2.94E-02	2.40E-02
2012	4.82E-02	3.85E-02	2.91E-02	2.37E-02
2013	4.96E-02	3.84E-02	2.87E-02	2.32E-02
2014	4.75E-02	3.71E-02	2.78E-02	2.25E-02
2015	4.37E-02	3.52E-02	2.70E-02	2.19E-02
2016	4.05E-02	3.31E-02	2.61E-02	2.13E-02
2017	3.93E-02	3.18E-02	2.51E-02	2.06E-02
2018	3.91E-02	3.14E-02	2.44E-02	2.00E-02

TABLE 3-50: SUMMARY OF $ADD_{95\%UCL}$ FOR FEMALE MINK BASED ON TRI+ PREDICTIONS FOR THE PERIOD 1993 - 2018

Year	95% UCL Dietary Dose (mg/Kg/day)			
	152	113	90	50
1993	1.42E-01	1.02E-01	8.21E-02	7.08E-02
1994	1.14E-01	9.34E-02	7.56E-02	6.44E-02
1995	1.05E-01	8.16E-02	6.82E-02	5.85E-02
1996	1.17E-01	8.20E-02	6.41E-02	5.45E-02
1997	1.03E-01	7.73E-02	6.07E-02	5.10E-02
1998	8.50E-02	6.85E-02	5.67E-02	4.77E-02
1999	7.77E-02	6.34E-02	5.16E-02	4.40E-02
2000	7.66E-02	5.95E-02	4.86E-02	4.12E-02
2001	8.04E-02	6.04E-02	4.66E-02	3.88E-02
2002	7.38E-02	5.86E-02	4.55E-02	3.75E-02
2003	6.85E-02	5.45E-02	4.37E-02	3.59E-02
2004	6.00E-02	4.97E-02	4.07E-02	3.38E-02
2005	5.85E-02	4.77E-02	3.83E-02	3.18E-02
2006	6.27E-02	4.76E-02	3.65E-02	3.00E-02
2007	5.63E-02	4.61E-02	3.53E-02	2.89E-02
2008	5.34E-02	4.40E-02	3.42E-02	2.79E-02
2009	5.06E-02	4.15E-02	3.28E-02	2.68E-02
2010	5.23E-02	4.07E-02	3.15E-02	2.59E-02
2011	5.33E-02	4.16E-02	3.11E-02	2.53E-02
2012	5.08E-02	4.07E-02	3.08E-02	2.50E-02
2013	5.22E-02	4.06E-02	3.03E-02	2.45E-02
2014	5.01E-02	3.92E-02	2.93E-02	2.37E-02
2015	4.62E-02	3.73E-02	2.86E-02	2.31E-02
2016	4.31E-02	3.51E-02	2.76E-02	2.25E-02
2017	4.19E-02	3.38E-02	2.66E-02	2.17E-02
2018	4.16E-02	3.34E-02	2.59E-02	2.11E-02

TABLE 3-51: SUMMARY OF $ADD_{Expected}$ FOR FEMALE OTTER BASED ON TRI+ PREDICTIONS FOR THE PERIOD 1993 - 2018

	Average Dietary Dose (mg/Kg/day)			
Year	152	113	90	50
1993	1.83E+00	1.23E+00	2.81E-01	2.65E-01
1994	1.33E+00	1.07E+00	2.56E-01	2.37E-01
1995	1.14E+00	9.52E-01	2.31E-01	2.13E-01
1996	1.35E+00	8.88E-01	2.11E-01	1.93E-01
1997	1.23E+00	8.57E-01	1.96E-01	1.78E-01
1998	9.66E-01	7.81E-01	1.81E-01	1.64E-01
1999	8.35E-01	6.54E-01	1.63E-01	1.49E-01
2000	7.83E-01	5.95E-01	1.48E-01	1.37E-01
2001	8.90E-01	6.02E-01	1.40E-01	1.28E-01
2002	8.14E-01	5.96E-01	1.37E-01	1.22E-01
2003	7.25E-01	5.48E-01	1.30E-01	1.15E-01
2004	5.85E-01	4.86E-01	1.20E-01	1.08E-01
2005	5.57E-01	4.47E-01	1.11E-01	1.00E-01
2006	6.28E-01	4.41E-01	1.05E-01	9.44E-02
2007	5.79E-01	4.32E-01	1.02E-01	8.98E-02
2008	5.38E-01	4.15E-01	9.80E-02	8.60E-02
2009	4.68E-01	3.79E-01	9.26E-02	8.18E-02
2010	4.91E-01	3.63E-01	8.78E-02	7.79E-02
2011	5.50E-01	3.86E-01	8.71E-02	7.56E-02
2012	4.92E-01	3.77E-01	8.63E-02	7.39E-02
2013	5.36E-01	3.89E-01	8.86E-02	7.55E-02
2014	4.88E-01	3.69E-01	8.38E-02	7.14E-02
2015	4.51E-01	3.51E-01	8.13E-02	6.95E-02
2016	4.12E-01	3.27E-01	7.78E-02	6.75E-02
2017	3.71E-01	3.02E-01	7.37E-02	6.54E-02
2018	3.62E-01	2.86E-01	6.98E-02	6.17E-02

TABLE 3-52: SUMMARY OF $ADD_{95\%UCL}$ FOR FEMALE OTTER BASED ON TRI+ PREDICTIONS FOR THE PERIOD 1993 - 2018

Year	95% UCL Dietary Dose (mg/Kg/day)			
	152	113	90	50
1993	1.87E+00	1.25E+00	2.86E-01	2.69E-01
1994	1.36E+00	1.09E+00	2.60E-01	2.41E-01
1995	1.16E+00	9.72E-01	2.35E-01	2.17E-01
1996	1.38E+00	9.07E-01	2.14E-01	1.96E-01
1997	1.26E+00	8.76E-01	2.00E-01	1.81E-01
1998	9.87E-01	7.98E-01	1.84E-01	1.66E-01
1999	8.53E-01	6.68E-01	1.66E-01	1.52E-01
2000	8.01E-01	6.08E-01	1.51E-01	1.39E-01
2001	9.10E-01	6.16E-01	1.43E-01	1.30E-01
2002	8.32E-01	6.09E-01	1.39E-01	1.24E-01
2003	7.42E-01	5.60E-01	1.32E-01	1.17E-01
2004	5.98E-01	4.97E-01	1.23E-01	1.10E-01
2005	5.70E-01	4.57E-01	1.13E-01	1.02E-01
2006	6.42E-01	4.51E-01	1.07E-01	9.61E-02
2007	5.92E-01	4.41E-01	1.03E-01	9.15E-02
2008	5.50E-01	4.25E-01	9.98E-02	8.76E-02
2009	4.78E-01	3.88E-01	9.43E-02	8.33E-02
2010	5.02E-01	3.71E-01	8.95E-02	7.93E-02
2011	5.62E-01	3.94E-01	8.88E-02	7.70E-02
2012	5.03E-01	3.85E-01	8.79E-02	7.52E-02
2013	5.48E-01	3.98E-01	9.02E-02	7.68E-02
2014	4.99E-01	3.77E-01	8.53E-02	7.27E-02
2015	4.61E-01	3.59E-01	8.28E-02	7.08E-02
2016	4.21E-01	3.35E-01	7.92E-02	6.87E-02
2017	3.79E-01	3.09E-01	7.51E-02	6.66E-02
2018	3.70E-01	2.93E-01	7.12E-02	6.29E-02

TABLE 3-53: SUMMARY OF $ADD_{Expected}$ FOR FEMALE BAT ON A TEQ BASIS FOR THE PERIOD 1993 - 2018

	Total Average Dietary Dose (mg/Kg/day)			
Year	152	113	90	50
1993	6.67E-05	5.29E-05	4.30E-05	3.16E-05
1994	5.98E-05	4.96E-05	4.08E-05	2.96E-05
1995	5.78E-05	4.75E-05	3.82E-05	2.81E-05
1996	5.71E-05	4.57E-05	3.64E-05	2.71E-05
1997	5.40E-05	4.38E-05	3.53E-05	2.62E-05
1998	5.17E-05	4.26E-05	3.36E-05	2.48E-05
1999	4.90E-05	4.13E-05	3.24E-05	2.41E-05
2000	4.93E-05	3.96E-05	3.15E-05	2.33E-05
2001	4.82E-05	3.90E-05	3.06E-05	2.26E-05
2002	4.61E-05	3.77E-05	2.98E-05	2.22E-05
2003	4.33E-05	3.59E-05	2.91E-05	2.14E-05
2004	4.26E-05	3.47E-05	2.76E-05	2.05E-05
2005	4.15E-05	3.43E-05	2.66E-05	1.97E-05
2006	3.98E-05	3.33E-05	2.54E-05	1.88E-05
2007	3.93E-05	3.26E-05	2.48E-05	1.83E-05
2008	3.79E-05	3.14E-05	2.40E-05	1.78E-05
2009	3.72E-05	3.05E-05	2.35E-05	1.74E-05
2010	3.65E-05	2.99E-05	2.31E-05	1.71E-05
2011	3.50E-05	2.96E-05	2.23E-05	1.68E-05
2012	3.41E-05	2.89E-05	2.18E-05	1.65E-05
2013	3.34E-05	2.83E-05	2.11E-05	1.60E-05
2014	3.31E-05	2.76E-05	2.06E-05	1.56E-05
2015	3.21E-05	2.66E-05	2.02E-05	1.52E-05
2016	3.24E-05	2.59E-05	1.98E-05	1.49E-05
2017	3.20E-05	2.56E-05	1.95E-05	1.45E-05
2018	3.12E-05	2.56E-05	1.92E-05	1.42E-05

TABLE 3-54: SUMMARY OF $ADD_{95\%UCL}$ FOR FEMALE BAT ON A TEQ BASIS FOR THE PERIOD 1993 - 2018

Year	Total 95% UCL Dietary Dose (mg/Kg/day)			
	152	113	90	50
1993	7.16E-05	5.68E-05	4.62E-05	3.39E-05
1994	6.41E-05	5.31E-05	4.37E-05	3.18E-05
1995	6.20E-05	5.10E-05	4.10E-05	3.02E-05
1996	6.12E-05	4.90E-05	3.90E-05	2.90E-05
1997	5.79E-05	4.69E-05	3.78E-05	2.81E-05
1998	5.55E-05	4.56E-05	3.60E-05	2.65E-05
1999	5.27E-05	4.43E-05	3.47E-05	2.58E-05
2000	5.29E-05	4.25E-05	3.37E-05	2.50E-05
2001	5.17E-05	4.19E-05	3.27E-05	2.42E-05
2002	4.95E-05	4.04E-05	3.19E-05	2.38E-05
2003	4.66E-05	3.86E-05	3.13E-05	2.30E-05
2004	4.59E-05	3.73E-05	2.97E-05	2.20E-05
2005	4.46E-05	3.69E-05	2.86E-05	2.12E-05
2006	4.28E-05	3.58E-05	2.73E-05	2.02E-05
2007	4.23E-05	3.51E-05	2.66E-05	1.97E-05
2008	4.09E-05	3.38E-05	2.58E-05	1.91E-05
2009	4.01E-05	3.28E-05	2.53E-05	1.88E-05
2010	3.93E-05	3.22E-05	2.48E-05	1.84E-05
2011	3.76E-05	3.18E-05	2.40E-05	1.81E-05
2012	3.67E-05	3.11E-05	2.34E-05	1.77E-05
2013	3.59E-05	3.05E-05	2.27E-05	1.72E-05
2014	3.55E-05	2.97E-05	2.22E-05	1.67E-05
2015	3.46E-05	2.86E-05	2.17E-05	1.63E-05
2016	3.51E-05	2.79E-05	2.13E-05	1.60E-05
2017	3.47E-05	2.77E-05	2.10E-05	1.56E-05
2018	3.38E-05	2.77E-05	2.06E-05	1.53E-05

TABLE 3-55: SUMMARY OF $ADD_{Expected}$ FOR FEMALE RACCOON ON A TEQ BASIS FOR THE PERIOD 1993 - 2018

	Total Average Dietary Dose (mg/Kg/day)			
Year	152	113	90	50
1993	1.47E-05	1.15E-05	9.31E-06	6.95E-06
1994	1.31E-05	1.08E-05	8.82E-06	6.52E-06
1995	1.24E-05	1.02E-05	1.32E-05	6.17E-06
1996	1.25E-05	9.85E-06	1.29E-05	5.91E-06
1997	1.18E-05	9.46E-06	7.59E-06	5.69E-06
1998	1.11E-05	9.09E-06	7.22E-06	5.39E-06
1999	1.04E-05	8.74E-06	6.92E-06	5.20E-06
2000	1.04E-05	8.38E-06	6.69E-06	5.01E-06
2001	1.03E-05	8.27E-06	6.48E-06	4.84E-06
2002	9.83E-06	8.00E-06	6.33E-06	4.74E-06
2003	9.26E-06	7.64E-06	6.16E-06	4.57E-06
2004	8.97E-06	7.33E-06	5.85E-06	4.37E-06
2005	8.72E-06	7.21E-06	5.63E-06	4.20E-06
2006	8.49E-06	7.02E-06	5.39E-06	4.02E-06
2007	8.31E-06	6.87E-06	5.25E-06	3.91E-06
2008	8.01E-06	6.62E-06	5.09E-06	3.80E-06
2009	7.80E-06	6.40E-06	4.97E-06	3.71E-06
2010	7.70E-06	6.27E-06	4.86E-06	3.62E-06
2011	7.45E-06	6.22E-06	4.71E-06	3.56E-06
2012	7.25E-06	6.08E-06	4.60E-06	3.48E-06
2013	7.12E-06	5.96E-06	4.47E-06	3.38E-06
2014	7.00E-06	5.81E-06	4.37E-06	3.29E-06
2015	6.77E-06	5.60E-06	4.27E-06	3.21E-06
2016	6.74E-06	5.44E-06	4.17E-06	3.14E-06
2017	6.64E-06	5.36E-06	4.11E-06	3.06E-06
2018	6.48E-06	5.32E-06	4.02E-06	2.99E-06

TABLE 3-56: SUMMARY OF $ADD_{95\%UCL}$ FOR FEMALE RACCOON ON A TEQ BASIS FOR THE PERIOD 1993 - 2018

Year	Total 95% UCL Dietary Dose (mg/Kg/day)			
	152	113	90	50
1993	1.58E-05	1.25E-05	1.01E-05	7.51E-06
1994	1.42E-05	1.17E-05	9.55E-06	7.07E-06
1995	1.37E-05	1.12E-05	9.03E-06	6.72E-06
1996	1.36E-05	1.08E-05	8.63E-06	6.45E-06
1997	1.29E-05	1.04E-05	8.32E-06	6.24E-06
1998	1.23E-05	1.01E-05	7.95E-06	5.93E-06
1999	1.18E-05	9.83E-06	7.64E-06	5.75E-06
2000	1.18E-05	9.46E-06	7.44E-06	5.57E-06
2001	1.15E-05	9.32E-06	7.23E-06	5.40E-06
2002	1.11E-05	9.03E-06	7.06E-06	5.30E-06
2003	1.05E-05	8.65E-06	6.90E-06	5.14E-06
2004	1.04E-05	8.41E-06	6.61E-06	4.93E-06
2005	1.02E-05	8.32E-06	6.38E-06	4.76E-06
2006	9.74E-06	8.09E-06	6.14E-06	4.58E-06
2007	9.53E-06	7.91E-06	6.01E-06	4.47E-06
2008	9.35E-06	7.66E-06	5.84E-06	4.36E-06
2009	9.28E-06	7.51E-06	5.71E-06	4.26E-06
2010	8.97E-06	7.34E-06	5.60E-06	4.17E-06
2011	8.55E-06	7.24E-06	5.43E-06	4.10E-06
2012	8.34E-06	7.03E-06	5.30E-06	4.01E-06
2013	8.16E-06	6.87E-06	5.15E-06	3.89E-06
2014	8.05E-06	6.70E-06	5.02E-06	3.80E-06
2015	7.84E-06	6.53E-06	4.91E-06	3.70E-06
2016	8.02E-06	6.38E-06	4.80E-06	3.62E-06
2017	7.97E-06	6.33E-06	4.73E-06	3.53E-06
2018	7.73E-06	6.34E-06	4.64E-06	3.45E-06

TABLE 3-57: SUMMARY OF $ADD_{Expected}$ FOR FEMALE MINK ON A TEQ BASIS FOR THE PERIOD 1993 - 2018

	Average Dietary Dose (mg/Kg/day)			
Year	152	113	90	50
1993	1.59E-05	1.13E-05	9.12E-06	7.90E-06
1994	1.27E-05	1.04E-05	8.40E-06	7.18E-06
1995	1.16E-05	9.05E-06	8.33E-06	6.52E-06
1996	1.30E-05	9.12E-06	7.89E-06	6.08E-06
1997	1.15E-05	8.58E-06	6.73E-06	5.67E-06
1998	9.41E-06	7.59E-06	6.29E-06	5.31E-06
1999	8.57E-06	6.99E-06	5.71E-06	4.89E-06
2000	8.46E-06	6.57E-06	5.38E-06	4.57E-06
2001	8.91E-06	6.66E-06	5.15E-06	4.31E-06
2002	8.14E-06	6.46E-06	5.02E-06	4.16E-06
2003	7.57E-06	6.01E-06	4.81E-06	3.97E-06
2004	6.59E-06	5.46E-06	4.49E-06	3.73E-06
2005	6.43E-06	5.24E-06	4.22E-06	3.52E-06
2006	6.93E-06	5.22E-06	4.02E-06	3.32E-06
2007	6.19E-06	5.07E-06	3.89E-06	3.19E-06
2008	5.85E-06	4.82E-06	3.76E-06	3.07E-06
2009	5.53E-06	4.54E-06	3.60E-06	2.96E-06
2010	5.75E-06	4.46E-06	3.46E-06	2.85E-06
2011	5.88E-06	4.56E-06	3.41E-06	2.79E-06
2012	5.60E-06	4.47E-06	3.38E-06	2.75E-06
2013	5.76E-06	4.46E-06	3.33E-06	2.69E-06
2014	5.52E-06	4.30E-06	3.23E-06	2.61E-06
2015	5.08E-06	4.09E-06	3.14E-06	2.55E-06
2016	4.69E-06	3.84E-06	3.03E-06	2.47E-06
2017	4.55E-06	3.69E-06	2.91E-06	2.39E-06
2018	4.52E-06	3.63E-06	2.83E-06	2.32E-06

TABLE 3-58: SUMMARY OF $ADD_{95\%UCL}$ FOR FEMALE MINK ON A TEQ BASIS FOR THE PERIOD 1993 - 2018

Year	95% UCL Dietary Dose (mg/Kg/day)			
	152	113	90	50
1993	1.66E-05	1.19E-05	9.55E-06	8.25E-06
1994	1.33E-05	1.09E-05	8.81E-06	7.50E-06
1995	1.22E-05	9.53E-06	7.95E-06	6.83E-06
1996	1.36E-05	9.58E-06	7.49E-06	6.37E-06
1997	1.20E-05	9.04E-06	7.08E-06	5.95E-06
1998	9.94E-06	8.01E-06	6.63E-06	5.58E-06
1999	9.10E-06	7.42E-06	6.03E-06	5.14E-06
2000	8.96E-06	6.97E-06	5.69E-06	4.81E-06
2001	9.40E-06	7.07E-06	5.45E-06	4.54E-06
2002	8.64E-06	6.86E-06	5.32E-06	4.39E-06
2003	8.03E-06	6.39E-06	5.11E-06	4.20E-06
2004	7.06E-06	5.83E-06	4.77E-06	3.95E-06
2005	6.89E-06	5.61E-06	4.50E-06	3.73E-06
2006	7.37E-06	5.59E-06	4.28E-06	3.53E-06
2007	6.61E-06	5.42E-06	4.15E-06	3.39E-06
2008	6.29E-06	5.17E-06	4.02E-06	3.27E-06
2009	5.98E-06	4.89E-06	3.86E-06	3.15E-06
2010	6.16E-06	4.80E-06	3.70E-06	3.04E-06
2011	6.27E-06	4.89E-06	3.65E-06	2.98E-06
2012	5.98E-06	4.78E-06	3.62E-06	2.93E-06
2013	6.13E-06	4.77E-06	3.56E-06	2.87E-06
2014	5.89E-06	4.61E-06	3.45E-06	2.78E-06
2015	5.44E-06	4.39E-06	3.36E-06	2.72E-06
2016	5.08E-06	4.14E-06	3.24E-06	2.64E-06
2017	4.95E-06	3.99E-06	3.12E-06	2.55E-06
2018	4.91E-06	3.94E-06	3.04E-06	2.48E-06

TABLE 3-59: SUMMARY OF $ADD_{Expected}$ FOR FEMALE OTTER ON A TEQ BASIS FOR THE PERIOD 1993 - 2018

	Total Average Dietary Dose (mg/Kg/day)			
Year	152	113	90	50
1993	2.14E-04	1.43E-04	3.30E-05	3.10E-05
1994	1.55E-04	1.25E-04	3.00E-05	2.78E-05
1995	1.33E-04	1.11E-04	2.76E-05	2.50E-05
1996	1.58E-04	1.04E-04	2.52E-05	2.26E-05
1997	1.44E-04	1.00E-04	2.31E-05	2.09E-05
1998	1.13E-04	9.13E-05	2.12E-05	1.92E-05
1999	9.76E-05	7.64E-05	1.91E-05	1.75E-05
2000	9.16E-05	6.95E-05	1.74E-05	1.60E-05
2001	1.04E-04	7.04E-05	1.65E-05	1.50E-05
2002	9.52E-05	6.97E-05	1.60E-05	1.43E-05
2003	8.48E-05	6.40E-05	1.52E-05	1.35E-05
2004	6.85E-05	5.68E-05	1.42E-05	1.27E-05
2005	6.52E-05	5.22E-05	1.31E-05	1.18E-05
2006	7.34E-05	5.16E-05	1.24E-05	1.11E-05
2007	6.77E-05	5.05E-05	1.19E-05	1.06E-05
2008	6.30E-05	4.86E-05	1.15E-05	1.01E-05
2009	5.47E-05	4.44E-05	1.09E-05	9.61E-06
2010	5.74E-05	4.24E-05	1.03E-05	9.15E-06
2011	6.43E-05	4.51E-05	1.02E-05	8.88E-06
2012	5.75E-05	4.41E-05	1.02E-05	8.68E-06
2013	6.27E-05	4.55E-05	1.04E-05	8.87E-06
2014	5.70E-05	4.31E-05	9.85E-06	8.39E-06
2015	5.27E-05	4.11E-05	9.56E-06	8.17E-06
2016	4.82E-05	3.83E-05	9.14E-06	7.93E-06
2017	4.34E-05	3.53E-05	8.67E-06	7.68E-06
2018	4.23E-05	3.35E-05	8.22E-06	7.25E-06

TABLE 3-60: SUMMARY OF $ADD_{95\%UCL}$ FOR FEMALE OTTER ON A TEQ BASIS FOR THE PERIOD 1993 - 2018

Year	Total 95% UCL Dietary Dose (mg/Kg/day)			
	152	113	90	50
1993	2.19E-04	1.46E-04	3.35E-05	3.15E-05
1994	1.58E-04	1.28E-04	3.05E-05	2.83E-05
1995	1.36E-04	1.14E-04	2.75E-05	2.54E-05
1996	1.61E-04	1.06E-04	2.51E-05	2.30E-05
1997	1.47E-04	1.02E-04	2.35E-05	2.13E-05
1998	1.15E-04	9.33E-05	2.16E-05	1.95E-05
1999	9.98E-05	7.82E-05	1.95E-05	1.78E-05
2000	9.36E-05	7.12E-05	1.77E-05	1.63E-05
2001	1.06E-04	7.20E-05	1.68E-05	1.53E-05
2002	9.73E-05	7.13E-05	1.64E-05	1.46E-05
2003	8.68E-05	6.55E-05	1.55E-05	1.38E-05
2004	7.01E-05	5.81E-05	1.44E-05	1.29E-05
2005	6.68E-05	5.35E-05	1.33E-05	1.20E-05
2006	7.52E-05	5.28E-05	1.26E-05	1.13E-05
2007	6.93E-05	5.17E-05	1.22E-05	1.08E-05
2008	6.44E-05	4.97E-05	1.18E-05	1.03E-05
2009	5.60E-05	4.55E-05	1.11E-05	9.81E-06
2010	5.88E-05	4.34E-05	1.06E-05	9.35E-06
2011	6.58E-05	4.62E-05	1.05E-05	9.07E-06
2012	5.88E-05	4.51E-05	1.04E-05	8.87E-06
2013	6.42E-05	4.66E-05	1.06E-05	9.05E-06
2014	5.84E-05	4.41E-05	1.01E-05	8.57E-06
2015	5.40E-05	4.21E-05	9.77E-06	8.34E-06
2016	4.93E-05	3.92E-05	9.35E-06	8.10E-06
2017	4.45E-05	3.62E-05	8.87E-06	7.84E-06
2018	4.34E-05	3.44E-05	8.41E-06	7.41E-06

TABLE 4-1
TOXICITY REFERENCE VALUES FOR FISH
DIETARY DOSES AND EGG CONCENTRATIONS OF TOTAL PCBs AND DIOXIN TOXIC EQUIVALENTS (TEQs)

TRVs		Pumpkinseed (*Lepomis gibbosus*)	Spottail Shiner (*Notropis hudsonius*)	Brown Bullhead (*Ictalurus nebulosus*)	Yellow Perch (*Perca flavescens*)	White Perch (*Morone americana*)	Largemouth Bass (*Micropterus salmoides*)	Striped Bass (*Morone saxatilus*)	Shortnose Sturgeon (*Acipenser brevirostrum*)	References
Tissue Concentration										
Lab-based TRVs for PCBs (mg/kg wet wt.)	LOAEL	1.5	***15***	***1.5***	***1.5***	1.5	1.5	1.5	***1.5***	Bengtsson (1980)
	NOAEL	0.16	***1.6***	***0.16***	***0.16***	0.16	0.16	0.16	***0.16***	
Field-based TRVs for PCBs (mg/kg wet wt.)	LOAEL	NA	NA	NA	NA	NA	NA	NA	NA	White perch and striped bass: Westin et al. (1983)
	NOAEL	***0.5***	NA	NA	NA	***3.1***	***0.5***	***3.1***	NA	Pumpkinseed and Largemouth bass: Adams et al. (1989, 1990, 1992)
Egg Concentration										
Lab-based TRV for TEQs (ug/kg lipid) from salmonids	LOAEL	***0.6***	Not derived	***18***	***0.6***	***0.6***	***0.6***	***0.6***	***0.6***	Brown Bullhead: Elonen et al. (1998)
	NOAEL	***0.29***	Not derived	***8.0***	***0.29***	***0.29***	***0.29***	***0.29***	***0.29***	All others: Walker et al. (1994)
Lab-based TRV for TEQs (ug/kg lipid) from non-salmonids	LOAEL	10.3	***103***	Not derived	10.3	10.3	10.3	10.3	10.3	Oliveri and Cooper (1997)
	NOAEL	0.54	***5.4***	Not derived	0.54	0.54	0.54	0.54	0.54	
Field-based TRVs for TEQs (ug/kg lipid)	LOAEL	NA	NA	NA	NA	NA	NA	NA	NA	
	NOAEL	NA	NA	NA	NA	NA	NA	NA	NA	

Note:

[a] Pumpkinseed (*Lepomis gibbosus*) and spottail shiner (*Notropis hudsonius*)

Units vary for PCBs and TEQ.

NA = Not available

Selected TRVs are ***bolded and italicized***.

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TABLE 4-2
TOXICITY REFERENCE VALUES FOR BIRDS
DIETARY DOSES AND EGG CONCENTRATIONS OF TOTAL PCBs AND DIOXIN TOXIC EQUIVALENTS (TEQs)

TRVs		Tree Swallow (*Tachycineta bicolor*)	Mallard Duck (*Anas platyrhychos*)	Belted Kingfisher (*Ceryle alcyon*)	Great Blue Heron (*Ardea herodias*)	Bald Eagle (*Haliaeetus leucocephalus*)	References
Dietary Dose							
Lab-based TRVs for PCBs (mg/kg/day)	LOAEL	0.07	***2.6***	***0.07***	***0.07***	***0.07***	Mallard: Custer and Heinz (1980)
	NOAEL	0.01	***0.26***	***0.01***	***0.01***	***0.01***	All others: Scott (1977)
Field-based TRVs for PCBs (mg/kd/day)	LOAEL	NA	NA	NA	NA	NA	Tree Swallow: US EPA Phase 2 Database (1998)
	NOAEL	***16.1***	NA	NA	NA	NA	
Lab-based TRVs for TEQs (ug/kg/day)	LOAEL	0.014	***0.014***	***0.014***	***0.014***	***0.014***	Nosek et al. (1992)
	NOAEL	0.0014	***0.0014***	***0.0014***	***0.0014***	***0.0014***	
Field-based TRVs for TEQs (ug/kg/day)	LOAEL	NA	NA	NA	NA	NA	US EPA Phase 2 Database (1998)
	NOAEL	***4.9***	NA	NA	NA	NA	
Egg Concentration							
Lab-based TRVs for PCBs (mg/kg egg)	LOAEL	2.21	***2.21***	***2.21***	***2.21***	2.21	Scott (1977)
	NOAEL	0.33	***0.33***	***0.33***	***0.33***	0.33	
Field-based TRVs for PCBs (mg/kg egg)	LOAEL	NA	NA	NA	NA	NA	Bald Eagle: Wiemeyer (1984, 1993)
	NOAEL	***26.7***	NA	NA	NA	***3.0***	Tree Swallow: US EPA Phase 2 Database (1998)
Lab-based TRVs for TEQs (ug/kg egg)	LOAEL	0.02	0.02	***0.02***	NA	***0.02***	Great Blue Heron: Janz and Bellward (1996)
	NOAEL	0.01	0.01	***0.01***	2	***0.01***	Others: Powell et al. (1996a)
Field-based TRVs for TEQs (ug/kg egg)	LOAEL	NA	*0.02*	NA	*0.5*	NA	Mallard: White and Segniak (1994); White and Hoffman (1995)
	NOAEL	***13***	***0.005***	NA	***0.3***	NA	Great Blue Heron: Sanderson et al. (1994)
							Tree Swallow: US EPA Phase 2 Database (1998)

Note: Units vary for PCBs and TEQ.

NA = Not Available

Selected TRVs are ***bolded and italicized.***

TABLE 4-3
TOXICITY REFERENCE VALUES FOR MAMMALS
DIETARY DOSES OF TOTAL PCBs AND DIOXIN TOXIC EQUIVALENTS (TEQs)

TRVs		**Little Brown Bat** (*Myotis lucifugus*)	**Raccoon** (*Procyon lotor*)	**Mink** (*Mustela vison*)	**River Otter** (*Lutra canadensis*)	**References**
Lab-based TRVs for PCBs (mg/kg/day)	LOAEL	***0.15***	***0.15***	0.07	0.07	Mink and otter: Aulerich and Ringer (1977)
	NOAEL	***0.032***	***0.032***	0.01	0.01	Raccoon and bat: Linder et al. (1984)
Field-based TRVs for PCBs (mg/kg/day)	LOAEL	NA	NA	***0.13***	***0.13***	Heaton et al. (1995)
	NOAEL	NA	NA	***0.004***	***0.004***	
Lab-based TRVs for TEQs (ug/kg/day)	LOAEL	***0.001***	***0.001***	0.001	0.001	Murray et al. (1979)
	NOAEL	***0.0001***	***0.0001***	0.0001	0.0001	
Field-based TRVs for TEQs (ug/kg/day)	LOAEL	NA	NA	***0.00224***	***0.00224***	Tillitt et al. (1996)
	NOAEL	NA	NA	***0.00008***	***0.00008***	

Note: Units vary for PCBs and TEQ.
Note: TRVs for raccoon and bat are based on mulit-generational studies to which interspecies uncertainty factors are applied.
NA = Not Available
Final selected TRVs are ***bolded and italicized***.

TABLE 4-4
WORLD HEALTH ORGANIZATION - TOXIC EQUIVALENCY FACTORS (TEFs)
FOR HUMANS, MAMMALS, FISH, AND BIRDS

Congener	Toxic Equivalency Factor		
	Humans/Mammals	Fish	Birds
Non-*ortho* PCBs			
3,4,4',5-TetraCB (81)	0.0001	0.0005	0.1
3,3',4,4'-TetraCB (77)	0.0001	0.0001	0.05
3,3',4,4',5-PentaCB (126)	0.1	0.005	0.1
3,3',4,4',5,5'-HexaCB (169)	0.01	0.00005	0.001
Mono-*ortho* PCBs			
2,3,3',4,4'-PentaCB (105)	0.0001	<0.000005	0.0001
2,3,4,4',5-PentaCB (114)	0.0005	<0.000005	0.0001
2,3',4,4',5-PentaCB (118)	0.0001	<0.000005	0.00001
2',3,4,4',5-PentaCB (123)	0.0001	<0.000005	0.00001
2,3,3',4,4',5-HexaCB (156)	0.0005	<0.000005	0.0001
2,3,3',4,4',5'-HexaCB (157)	0.0005	<0.000005	0.0001
2,3',4,4',5,5'-HexaCB (167)	0.00001	<0.000005	0.00001
2,3,3',4,4',5,5'-HeptaCB (189)	0.0001	<0.000005	0.00001

Notes: CB = chlorinated biphenyls
Reference: van den Berg et al. 1998. Toxic Equivalency Factors (TEFs) for PCBs, PCDDs, PCDFs for Humans and Wildlife. *Environmental Health Perspectives*, 106:12, 775-791.

TABLE 5-1: RATIO OF PREDICTED SEDIMENT CONCENTRATIONS TO SEDIMENT GUIDELINES

Year	Average PCB Results				Tri+ 95% UCL Results				Average PCB Results				Tri+ 95% UCL Results			
	152 Total Sed Conc	113 Total Sed Conc	90 Total Sed Conc	50 Total Sed Conc	152 Total Sed Conc	113 Total Sed Conc	90 Total Sed Conc	50 Total Sed Conc	152 Total Sed Conc	113 Total Sed Conc	90 Total Sed Conc	50 Total Sed Conc	152 Total Sed Conc	113 Total Sed Conc	90 Total Sed Conc	50 Total Sed Conc
	NOAA Consensus-Based Total PCB TEC: 0.04 mg/kg dry weight								NOAA Consensus-Based Total PCB MEC: 0.4 mg/kg dry weight							
1993	24	19	15	11	27	21	17	13	2.4	1.9	1.5	1.1	2.7	2.1	1.7	1.3
1994	22	18	15	11	26	21	16	12	2.2	1.8	1.5	1.1	2.6	2.1	1.6	1.2
1995	20	17	55	10	25	20	16	12	2.0	1.7	5.5	1.0	2.5	2.0	1.6	1.2
1996	20	16	54	9.7	24	20	16	11	2.0	1.6	5.4	1.0	2.4	2.0	1.6	1.1
1997	20	16	13	9.3	24	19	15	11	2.0	1.6	1.3	0.9	2.4	1.9	1.5	1.1
1998	18	15	12	8.9	24	19	15	11	1.8	1.5	1.2	0.9	2.4	1.9	1.5	1.1
1999	17	14	11	8.5	23	19	14	11	1.7	1.4	1.1	0.9	2.3	1.9	1.4	1.1
2000	17	14	11	8.2	23	19	14	11	1.7	1.4	1.1	0.8	2.3	1.9	1.4	1.1
2001	17	13	11	7.9	22	18	14	10	1.7	1.3	1.1	0.8	2.2	1.8	1.4	1.0
2002	16	13	10	7.6	22	18	14	10	1.6	1.3	1.0	0.8	2.2	1.8	1.4	1.0
2003	15	13	10	7.4	21	17	13	9.9	1.5	1.3	1.0	0.7	2.1	1.7	1.3	1.0
2004	15	12	9.7	7.1	22	17	13	9.7	1.5	1.2	1.0	0.7	2.2	1.7	1.3	1.0
2005	14	12	9.3	6.9	22	17	13	9.5	1.4	1.2	0.9	0.7	2.2	1.7	1.3	1.0
2006	14	11	9.0	6.7	20	17	13	9.3	1.4	1.1	0.9	0.7	2.0	1.7	1.3	0.9
2007	14	11	8.7	6.5	20	16	13	9.3	1.4	1.1	0.9	0.6	2.0	1.6	1.3	0.9
2008	13	11	8.5	6.3	20	16	12	9.1	1.3	1.1	0.8	0.6	2.0	1.6	1.2	0.9
2009	13	11	8.2	6.1	21	16	12	8.9	1.3	1.1	0.8	0.6	2.1	1.6	1.2	0.9
2010	13	10	8.0	5.9	19	16	12	8.7	1.3	1.0	0.8	0.6	1.9	1.6	1.2	0.9
2011	12	10	7.8	5.8	18	15	11	8.5	1.2	1.0	0.8	0.6	1.8	1.5	1.1	0.9
2012	12	9.9	7.6	5.6	17	15	11	8.3	1.2	1.0	0.8	0.6	1.7	1.5	1.1	0.8
2013	12	9.7	7.4	5.5	17	14	11	8.1	1.2	1.0	0.7	0.5	1.7	1.4	1.1	0.8
2014	11	9.4	7.3	5.3	17	14	11	7.9	1.1	0.9	0.7	0.5	1.7	1.4	1.1	0.8
2015	11	9.2	7.1	5.2	16	14	10	7.7	1.1	0.9	0.7	0.5	1.6	1.4	1.0	0.8
2016	11	8.9	6.9	5.1	18	14	10	7.5	1.1	0.9	0.7	0.5	1.8	1.4	1.0	0.8
2017	10	8.7	6.7	5.0	18	14	9.9	7.3	1.0	0.9	0.7	0.5	1.8	1.4	1.0	0.7
2018	10	8.5	6.5	4.8	17	14	9.7	7.2	1.0	0.8	0.7	0.5	1.7	1.4	1.0	0.7

exceedances are bolded

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TABLE 5-1: RATIO OF PREDICTED SEDIMENT CONCENTRATIONS TO SEDIMENT GUIDELINES (CONT.)

	Average PCB Results				Tri+ 95% UCL Results				Average PCB Results				Tri+ 95% UCL Results			
Year	152 Total Sed Conc	113 Total Sed Conc	90 Total Sed Conc	50 Total Sed Conc	152 Total Sed Conc	113 Total Sed Conc	90 Total Sed Conc	50 Total Sed Conc	152 Total Sed Conc	113 Total Sed Conc	90 Total Sed Conc	50 Total Sed Conc	152 Total Sed Conc	113 Total Sed Conc	90 Total Sed Conc	50 Total Sed Conc
	NOAA Consensus-Based Total PCB EEC: 1.7 mg/kg dry weight								NYSDEC Benthic Chronic Total PCB 19.3 mg/Kg OC (0.482 mg/kg using 2.5% OC							
1993	0.6	0.4	0.4	0.3	0.6	0.5	0.4	0.3	**2.0**	**1.6**	**1.3**	0.9	**2.2**	**1.8**	**1.4**	**1.0**
1994	0.5	0.4	0.3	0.3	0.6	0.5	0.4	0.3	**1.8**	**1.5**	**1.2**	0.9	**2.1**	**1.7**	**1.4**	**1.0**
1995	0.5	0.4	1.3	0.2	0.6	0.5	0.4	0.3	**1.7**	**1.4**	**4.5**	0.8	**2.1**	**1.7**	**1.4**	**1.0**
1996	0.5	0.4	1.3	0.2	0.6	0.5	0.4	0.3	**1.7**	**1.3**	**4.5**	0.8	**2.0**	**1.6**	**1.3**	**1.0**
1997	0.5	0.4	0.3	0.2	0.6	0.5	0.4	0.3	**1.6**	**1.3**	**1.0**	0.8	**2.0**	**1.6**	**1.3**	0.9
1998	0.4	0.4	0.3	0.2	0.6	0.5	0.3	0.3	**1.5**	**1.2**	**1.0**	0.7	**2.0**	**1.6**	**1.2**	0.9
1999	0.4	0.3	0.3	0.2	0.6	0.4	0.3	0.3	**1.4**	**1.2**	**1.0**	0.7	**1.9**	**1.6**	**1.2**	0.9
2000	0.4	0.3	0.3	0.2	0.5	0.4	0.3	0.2	**1.4**	**1.1**	0.9	0.7	**1.9**	**1.5**	**1.2**	0.9
2001	0.4	0.3	0.2	0.2	0.5	0.4	0.3	0.2	**1.4**	**1.1**	0.9	0.7	**1.8**	**1.5**	**1.1**	0.9
2002	0.4	0.3	0.2	0.2	0.5	0.4	0.3	0.2	**1.3**	**1.1**	0.9	0.6	**1.8**	**1.5**	**1.1**	0.8
2003	0.4	0.3	0.2	0.2	0.5	0.4	0.3	0.2	**1.3**	**1.0**	0.8	0.6	**1.8**	**1.4**	**1.1**	0.8
2004	0.3	0.3	0.2	0.2	0.5	0.4	0.3	0.2	**1.2**	**1.0**	0.8	0.6	**1.8**	**1.5**	**1.1**	0.8
2005	0.3	0.3	0.2	0.2	0.5	0.4	0.3	0.2	**1.2**	**1.0**	0.8	0.6	**1.8**	**1.4**	**1.1**	0.8
2006	0.3	0.3	0.2	0.2	0.5	0.4	0.3	0.2	**1.2**	0.9	0.7	0.6	**1.7**	**1.4**	**1.0**	0.8
2007	0.3	0.3	0.2	0.2	0.5	0.4	0.3	0.2	**1.1**	0.9	0.7	0.5	**1.6**	**1.4**	**1.0**	0.8
2008	0.3	0.3	0.2	0.1	0.5	0.4	0.3	0.2	**1.1**	0.9	0.7	0.5	**1.7**	**1.3**	**1.0**	0.8
2009	0.3	0.2	0.2	0.1	0.5	0.4	0.3	0.2	**1.1**	0.9	0.7	0.5	**1.7**	**1.4**	**1.0**	0.7
2010	0.3	0.2	0.2	0.1	0.5	0.4	0.3	0.2	**1.0**	0.9	0.7	0.5	**1.6**	**1.3**	**1.0**	0.7
2011	0.3	0.2	0.2	0.1	0.4	0.4	0.3	0.2	**1.0**	0.8	0.6	0.5	**1.5**	**1.3**	0.9	0.7
2012	0.3	0.2	0.2	0.1	0.4	0.3	0.3	0.2	**1.0**	0.8	0.6	0.5	**1.4**	**1.2**	0.9	0.7
2013	0.3	0.2	0.2	0.1	0.4	0.3	0.3	0.2	**1.0**	0.8	0.6	0.5	**1.4**	**1.2**	0.9	0.7
2014	0.3	0.2	0.2	0.1	0.4	0.3	0.2	0.2	0.9	0.8	0.6	0.4	**1.4**	**1.2**	0.9	0.7
2015	0.3	0.2	0.2	0.1	0.4	0.3	0.2	0.2	0.9	0.8	0.6	0.4	**1.4**	**1.2**	0.9	0.6
2016	0.3	0.2	0.2	0.1	0.4	0.3	0.2	0.2	0.9	0.7	0.6	0.4	**1.5**	**1.2**	0.8	0.6
2017	0.2	0.2	0.2	0.1	0.4	0.3	0.2	0.2	0.9	0.7	0.6	0.4	**1.5**	**1.2**	0.8	0.6
2018	0.2	0.2	0.2	0.1	0.4	0.3	0.2	0.2	0.8	0.7	0.5	0.4	**1.4**	**1.2**	0.8	0.6

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TABLE 5-1: RATIO OF PREDICTED SEDIMENT CONCENTRATIONS TO SEDIMENT GUIDELINES (CONT.)

	Average PCB Results				Tri+ 95% UCL Results				Average PCB Results				Tri+ 95% UCL Results			
Year	152 Total Sed Conc	113 Total Sed Conc	90 Total Sed Conc	50 Total Sed Conc	152 Total Sed Conc	113 Total Sed Conc	90 Total Sed Conc	50 Total Sed Conc	152 Total Sed Conc	113 Total Sed Conc	90 Total Sed Conc	50 Total Sed Conc	152 Total Sed Conc	113 Total Sed Conc	90 Total Sed Conc	50 Total Sed Conc
	NYSDEC Wildlife Total PCB 1.4 mg/Kg OC (0.035 mg/kg using 2.5% OC)								Persaud Total PCB NEL 0.01 mg/Kg dry weight							
1993	28	22	17	13	31	25	19	14	97	76	61	45	107	86	68	50
1994	25	21	17	12	29	24	19	14	88	72	58	43	102	84	66	49
1995	23	19	62	12	29	23	19	14	81	68	218	41	100	82	65	47
1996	23	19	62	11	28	23	18	13	81	65	218	39	98	79	63	46
1997	22	18	14	11	27	22	17	13	79	63	50	37	95	78	61	45
1998	21	17	14	10	27	22	17	13	73	60	48	36	94	77	59	44
1999	19	16	13	9.7	27	22	16	12	68	57	46	34	94	76	57	43
2000	19	16	13	9.3	26	21	16	12	67	55	44	33	91	74	57	42
2001	19	15	12	9.0	25	21	16	12	67	54	42	31	87	73	55	41
2002	18	15	12	8.7	25	20	15	11	65	52	41	31	87	71	54	40
2003	18	14	11	8.4	24	20	15	11	62	51	40	30	85	70	53	40
2004	17	14	11	8.2	25	20	15	11	59	49	39	29	87	70	52	39
2005	16	13	11	7.9	25	20	15	11	57	47	37	28	87	69	51	38
2006	16	13	10	7.6	23	19	14	11	56	46	36	27	81	67	50	37
2007	16	13	10	7.4	23	19	14	11	55	45	35	26	79	66	50	37
2008	15	12	9.7	7.2	23	18	14	10	53	43	34	25	81	65	49	36
2009	15	12	9.4	7.0	24	19	14	10	51	42	33	24	84	66	48	36
2010	14	12	9.1	6.8	22	18	13	9.9	50	41	32	24	77	64	47	35
2011	14	12	8.9	6.6	20	18	13	9.7	49	40	31	23	71	62	46	34
2012	14	11	8.7	6.4	20	17	13	9.5	48	39	30	22	70	59	44	33
2013	13	11	8.5	6.3	19	16	12	9.2	47	39	30	22	68	57	43	32
2014	13	11	8.3	6.1	19	16	12	9.0	46	38	29	21	67	56	42	32
2015	13	10	8.1	6.0	19	16	12	8.8	44	37	28	21	66	56	41	31
2016	12	10	7.9	5.8	20	16	12	8.6	43	36	28	20	71	56	40	30
2017	12	9.9	7.7	5.7	20	16	11	8.4	42	35	27	20	71	56	39	29
2018	12	9.7	7.5	5.5	19	16	11	8.2	41	34	26	19	68	56	39	29

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TABLE 5-1: RATIO OF PREDICTED SEDIMENT CONCENTRATIONS TO SEDIMENT GUIDELINES (CONT.)

Year	Average PCB Results				Tri+ 95% UCL Results				Average PCB Results				Tri+ 95% UCL Results			
	152 Total Sed Conc	113 Total Sed Conc	90 Total Sed Conc	50 Total Sed Conc	152 Total Sed Conc	113 Total Sed Conc	90 Total Sed Conc	50 Total Sed Conc	152 Total Sed Conc	113 Total Sed Conc	90 Total Sed Conc	50 Total Sed Conc	152 Total Sed Conc	113 Total Sed Conc	90 Total Sed Conc	50 Total Sed Conc
	Persaud Total PCB LEL 0.07 mg/Kg dry weight								Persaud Total PCB SEL 530 mg/Kg OC (1.3 mg/kg using 2.5% OC)							
1993	**14**	**11**	**9**	**6**	**15**	**12**	**10**	**7**	0.1	0.1	0.0	0.0	0.1	0.1	0.1	0.0
1994	**13**	**10**	**8**	**6**	**15**	**12**	**9**	**7**	0.1	0.1	0.0	0.0	0.1	0.1	0.0	0.0
1995	**12**	**10**	**31**	**6**	**14**	**12**	**9**	**7**	0.1	0.1	0.2	0.0	0.1	0.1	0.0	0.0
1996	**12**	**9**	**31**	**6**	**14**	**11**	**9**	**7**	0.1	0.0	0.2	0.0	0.1	0.1	0.0	0.0
1997	**11**	**9**	**7**	**5**	**14**	**11**	**9**	**6**	0.1	0.0	0.0	0.0	0.1	0.1	0.0	0.0
1998	**10**	**9**	**7**	**5**	**13**	**11**	**8**	**6**	0.1	0.0	0.0	0.0	0.1	0.1	0.0	0.0
1999	**10**	**8**	**7**	**5**	**13**	**11**	**8**	**6**	0.1	0.0	0.0	0.0	0.1	0.1	0.0	0.0
2000	**10**	**8**	**6**	**5**	**13**	**11**	**8**	**6**	0.1	0.0	0.0	0.0	0.1	0.1	0.0	0.0
2001	**10**	**8**	**6**	**4**	**12**	**10**	**8**	**6**	0.1	0.0	0.0	0.0	0.1	0.1	0.0	0.0
2002	**9**	**7**	**6**	**4**	**12**	**10**	**8**	**6**	0.0	0.0	0.0	0.0	0.1	0.1	0.0	0.0
2003	**9**	**7**	**6**	**4**	**12**	**10**	**8**	**6**	0.0	0.0	0.0	0.0	0.1	0.1	0.0	0.0
2004	**8**	**7**	**6**	**4**	**12**	**10**	**7**	**6**	0.0	0.0	0.0	0.0	0.1	0.1	0.0	0.0
2005	**8**	**7**	**5**	**4**	**12**	**10**	**7**	**5**	0.0	0.0	0.0	0.0	0.1	0.1	0.0	0.0
2006	**8**	**7**	**5**	**4**	**12**	**10**	**7**	**5**	0.0	0.0	0.0	0.0	0.1	0.1	0.0	0.0
2007	**8**	**6**	**5**	**4**	**11**	**9**	**7**	**5**	0.0	0.0	0.0	0.0	0.1	0.0	0.0	0.0
2008	**8**	**6**	**5**	**4**	**12**	**9**	**7**	**5**	0.0	0.0	0.0	0.0	0.1	0.0	0.0	0.0
2009	**7**	**6**	**5**	**3**	**12**	**9**	**7**	**5**	0.0	0.0	0.0	0.0	0.1	0.0	0.0	0.0
2010	**7**	**6**	**5**	**3**	**11**	**9**	**7**	**5**	0.0	0.0	0.0	0.0	0.1	0.0	0.0	0.0
2011	**7**	**6**	**4**	**3**	**10**	**9**	**7**	**5**	0.0	0.0	0.0	0.0	0.1	0.0	0.0	0.0
2012	**7**	**6**	**4**	**3**	**10**	**8**	**6**	**5**	0.0	0.0	0.0	0.0	0.1	0.0	0.0	0.0
2013	**7**	**6**	**4**	**3**	**10**	**8**	**6**	**5**	0.0	0.0	0.0	0.0	0.1	0.0	0.0	0.0
2014	**7**	**5**	**4**	**3**	**10**	**8**	**6**	**5**	0.0	0.0	0.0	0.0	0.1	0.0	0.0	0.0
2015	**6**	**5**	**4**	**3**	**9**	**8**	**6**	**4**	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
2016	**6**	**5**	**4**	**3**	**10**	**8**	**6**	**4**	0.0	0.0	0.0	0.0	0.1	0.0	0.0	0.0
2017	**6**	**5**	**4**	**3**	**10**	**8**	**6**	**4**	0.0	0.0	0.0	0.0	0.1	0.0	0.0	0.0
2018	**6**	**5**	**4**	**3**	**10**	**8**	**6**	**4**	0.0	0.0	0.0	0.0	0.1	0.0	0.0	0.0

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TABLE 5-1: RATIO OF PREDICTED SEDIMENT CONCENTRATIONS TO SEDIMENT GUIDELINES (CONT.)

	Average PCB Results				Tri+ 95% UCL Results				Average PCB Results				Tri+ 95% UCL Results			
Year	152 Total Sed Conc	113 Total Sed Conc	90 Total Sed Conc	50 Total Sed Conc	152 Total Sed Conc	113 Total Sed Conc	90 Total Sed Conc	50 Total Sed Conc	152 Total Sed Conc	113 Total Sed Conc	90 Total Sed Conc	50 Total Sed Conc	152 Total Sed Conc	113 Total Sed Conc	90 Total Sed Conc	50 Total Sed Conc
	Washington State Total PCB PAET *Hyalella azteca* 0.45 mg/Kg dry weight								Washington Total PCB PAET Microtox 0.021 mg/Kg dry weight							
1993	2.1	1.7	1.4	1.0	2.4	1.9	1.5	1.1	46	36	29	21	51	41	32	32
1994	2.0	1.6	1.3	0.9	2.3	1.9	1.5	1.1	42	34	28	20	49	40	31	31
1995	1.8	1.5	4.8	0.9	2.2	1.8	1.4	1.1	38	32	104	19	48	39	31	31
1996	1.8	1.4	4.8	0.9	2.2	1.8	1.4	1.0	39	31	104	18	47	38	30	30
1997	1.7	1.4	1.1	0.8	2.1	1.7	1.3	1.0	37	30	24	18	45	37	29	29
1998	1.6	1.3	1.1	0.8	2.1	1.7	1.3	1.0	35	29	23	17	45	36	28	28
1999	1.5	1.3	1.0	0.8	2.1	1.7	1.3	1.0	32	27	22	16	45	36	27	27
2000	1.5	1.2	1.0	0.7	2.0	1.7	1.3	0.9	32	26	21	16	43	35	27	27
2001	1.5	1.2	0.9	0.7	1.9	1.6	1.2	0.9	32	26	20	15	41	35	26	26
2002	1.4	1.2	0.9	0.7	1.9	1.6	1.2	0.9	31	25	20	15	41	34	26	26
2003	1.4	1.1	0.9	0.7	1.9	1.5	1.2	0.9	29	24	19	14	40	33	25	25
2004	1.3	1.1	0.9	0.6	1.9	1.6	1.2	0.9	28	23	18	14	42	33	25	25
2005	1.3	1.0	0.8	0.6	1.9	1.5	1.1	0.8	27	22	18	13	42	33	24	24
2006	1.2	1.0	0.8	0.6	1.8	1.5	1.1	0.8	27	22	17	13	39	32	24	24
2007	1.2	1.0	0.8	0.6	1.8	1.5	1.1	0.8	26	21	17	12	38	31	24	24
2008	1.2	1.0	0.8	0.6	1.8	1.4	1.1	0.8	25	21	16	12	39	31	23	23
2009	1.1	0.9	0.7	0.5	1.9	1.5	1.1	0.8	24	20	16	12	40	31	23	23
2010	1.1	0.9	0.7	0.5	1.7	1.4	1.0	0.8	24	20	15	11	37	30	22	22
2011	1.1	0.9	0.7	0.5	1.6	1.4	1.0	0.8	24	19	15	11	34	29	22	22
2012	1.1	0.9	0.7	0.5	1.6	1.3	1.0	0.7	23	19	15	11	33	28	21	21
2013	1.0	0.9	0.7	0.5	1.5	1.3	1.0	0.7	22	18	14	10	32	27	21	21
2014	1.0	0.8	0.6	0.5	1.5	1.2	0.9	0.7	22	18	14	10	32	27	20	20
2015	1.0	0.8	0.6	0.5	1.5	1.2	0.9	0.7	21	17	14	9.9	31	27	20	20
2016	1.0	0.8	0.6	0.5	1.6	1.2	0.9	0.7	20	17	13	9.7	34	27	19	19
2017	0.9	0.8	0.6	0.4	1.6	1.2	0.9	0.7	20	17	13	9.4	34	26	19	19
2018	0.9	0.8	0.6	0.4	1.5	1.2	0.9	0.6	19	16	12	9.2	32	27	18	18

TABLE 5-2: RATIO OF PREDICTED WHOLE WATER CONCENTRATIONS TO CRITERIA AND BENCHMARKS

Year	Tri+ Average PCB Results: 152 Whole Water Conc	Tri+ Average PCB Results: 113 Whole Water Conc	Tri+ Average PCB Results: 90 Whole Water Conc	Tri+ Average PCB Results: 50 Whole Water Conc	Tri+ 95% UCL Results: 152 Whole Water Conc	Tri+ 95% UCL Results: 113 Whole Water Conc	Tri+ 95% UCL Results: 90 Whole Water Conc	Tri+ 95% UCL Results: 50 Whole Water Conc
	USEPA/NYSDEC Total PCB - FW Benthic Aquatic Life 0.014 ug/L							
1993	3.1	2.2	1.6	1.3	4.4	2.7	2.0	1.6
1994	2.9	1.9	1.4	1.1	3.5	2.2	1.7	1.4
1995	1.2	1.2	1.1	1.0	1.3	1.3	1.4	1.2
1996	3.4	1.9	1.3	1.0	5.0	2.3	1.5	1.1
1997	2.2	1.5	1.1	0.9	2.9	1.8	1.4	1.1
1998	1.3	1.1	1.0	0.8	1.4	1.3	1.1	0.9
1999	1.1	0.9	0.8	0.7	1.2	1.0	1.0	0.8
2000	1.8	1.1	0.8	0.6	2.2	1.3	1.0	0.8
2001	2.0	1.2	0.8	0.6	2.9	1.5	1.0	0.7
2002	1.2	0.9	0.7	0.6	1.5	1.1	0.9	0.7
2003	1.3	0.9	0.7	0.5	1.8	1.1	0.8	0.6
2004	0.7	0.6	0.6	0.5	0.8	0.7	0.7	0.6
2005	1.0	0.6	0.5	0.4	1.3	0.8	0.6	0.5
2006	1.3	0.8	0.5	0.4	1.8	0.9	0.6	0.5
2007	1.4	0.8	0.5	0.4	2.3	1.0	0.6	0.5
2008	0.6	0.5	0.4	0.4	0.6	0.6	0.5	0.4
2009	0.6	0.5	0.4	0.3	0.7	0.5	0.5	0.4
2010	1.1	0.6	0.4	0.3	1.6	0.8	0.5	0.4
2011	1.1	0.7	0.4	0.3	1.8	0.8	0.5	0.4
2012	0.7	0.5	0.4	0.3	0.9	0.7	0.5	0.4
2013	1.0	0.6	0.4	0.3	1.4	0.7	0.5	0.4
2014	0.8	0.5	0.4	0.3	0.9	0.6	0.5	0.4
2015	0.7	0.5	0.4	0.3	0.9	0.6	0.5	0.4
2016	0.4	0.4	0.3	0.3	0.4	0.4	0.4	0.3
2017	0.4	0.3	0.3	0.2	0.4	0.4	0.3	0.3
2018	0.5	0.4	0.3	0.2	0.8	0.5	0.4	0.3

Year	Tri+ Average PCB Results: 152 Whole Water Conc	Tri+ Average PCB Results: 113 Whole Water Conc	Tri+ Average PCB Results: 90 Whole Water Conc	Tri+ Average PCB Results: 50 Whole Water Conc	Tri+ 95% UCL Results: 152 Whole Water Conc	Tri+ 95% UCL Results: 113 Whole Water Conc	Tri+ 95% UCL Results: 90 Whole Water Conc	Tri+ 95% UCL Results: 50 Whole Water Conc
	NYSDEC Total PCB - Wildlife Bioaccumulation 0.001 ug/L							
1993	44	30	23	18	61	38	28	22
1994	40	26	20	16	49	31	24	19
1995	16	16	16	14	18	19	19	16
1996	47	26	18	13	69	32	21	16
1997	31	21	16	12	40	25	19	15
1998	18	15	13	11	20	18	16	13
1999	16	13	11	10	17	15	14	11
2000	26	15	11	9.0	31	18	13	11
2001	29	17	12	8.7	40	21	14	10
2002	17	13	10	8.0	20	15	12	10
2003	19	13	10	7.5	25	15	12	9.0
2004	10	8.6	7.8	6.5	11	10	9.3	7.8
2005	14	9.1	7.2	6.0	18	11	8.5	7.0
2006	19	11	7.5	5.8	26	13	8.8	6.8
2007	19	11	7.4	5.5	32	14	8.7	6.5
2008	7.9	7.0	6.1	5.0	8.7	8.0	7.2	5.9
2009	8.5	6.5	5.6	4.6	10	7.6	6.6	5.5
2010	15	8.8	6.1	4.6	23	11	7.2	5.5
2011	15	9.1	6.2	4.6	25	12	7.3	5.4
2012	10	7.7	5.9	4.5	13	9.2	7.1	5.4
2013	14	8.6	6.0	4.4	20	10	7.1	5.2
2014	11	7.5	5.7	4.3	13	8.6	6.7	5.1
2015	10	7.1	5.4	4.1	12	8.1	6.4	4.9
2016	5.4	5.0	4.6	3.8	5.9	5.7	5.4	4.5
2017	5.1	4.4	4.1	3.5	5.7	5.0	4.8	4.1
2018	7.6	5.4	4.3	3.4	11	6.8	5.2	4.1

Year	Tri+ Average PCB Results: 152 Whole Water Conc	Tri+ Average PCB Results: 113 Whole Water Conc	Tri+ Average PCB Results: 90 Whole Water Conc	Tri+ Average PCB Results: 50 Whole Water Conc	Tri+ 95% UCL Results: 152 Whole Water Conc	Tri+ 95% UCL Results: 113 Whole Water Conc	Tri+ 95% UCL Results: 90 Whole Water Conc	Tri+ 95% UCL Results: 50 Whole Water Conc
	USEPA Total PCB - Wildlife Criterion 1.2E-04 ug/l							
1993	37	25	19	15	51	32	23	18
1994	33	22	17	13	41	26	20	16
1995	13	14	13	11	15	16	16	14
1996	39	22	15	11	58	27	17	13
1997	26	18	13	10	34	21	16	12
1998	15	13	11	9.1	17	15	13	11
1999	13	11	10	8.0	14	12	11	9.6
2000	21	13	9.5	7.5	26	15	11	8.9
2001	24	14	9.6	7.3	34	17	11	8.7
2002	14	11	8.5	6.7	17	13	10	8.0
2003	16	11	8.1	6.3	21	13	10	7.5
2004	8.4	7.2	6.5	5.4	9.3	8.2	7.7	6.5
2005	12	7.6	6.0	5.0	15	8.8	7.1	5.9
2006	16	8.9	6.2	4.8	22	11	7.3	5.7
2007	16	9.1	6.1	4.6	26	12	7.3	5.5
2008	6.6	5.8	5.1	4.2	7.2	7	6.0	4.9
2009	7.1	5.4	4.6	3.8	8.5	6.3	5.5	4.6
2010	13	7.3	5.1	3.9	19	9.1	6.0	4.6
2011	13	7.6	5.1	3.8	20	9.6	6.1	4.5
2012	8.7	6.4	4.9	3.7	11	7.7	5.9	4.5
2013	11	7.2	5.0	3.7	17	8.7	5.9	4.4
2014	9.0	6.3	4.7	3.6	11	7.2	5.6	4.3
2015	8.7	5.9	4.5	3.5	10.2	6.7	5.3	4.1
2016	4.5	4.2	3.8	3.1	4.9	4.8	4.5	3.7
2017	4.3	3.7	3.4	2.9	4.7	4.2	4.0	3.4
2018	6.4	4.5	3.6	2.9	8.8	5.7	4.3	3.4

exceedances are bolded

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TABLE 5-3: RATIO OF PREDICTED PUMPKINSEED CONCENTRATIONS TO FIELD-BASED NOAEL FOR TRI+ PCBS

	River Mile 152			River Mile 113			River Mile 90			River Mile 50		
Year	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)
1993	**2.3**	**3.1**	**5.1**	**1.5**	**2.1**	**3.5**	**1.2**	**1.7**	**2.7**	**1.1**	**1.6**	**2.6**
1994	**1.7**	**2.3**	**3.7**	**1.3**	**1.9**	**3.1**	**1.1**	**1.5**	**2.5**	**1.0**	**1.4**	**2.3**
1995	**1.5**	**2.1**	**3.4**	**1.1**	**1.5**	**2.6**	0.9	**1.3**	**2.2**	0.9	**1.3**	**2.1**
1996	**1.8**	**2.5**	**4.1**	**1.2**	**1.6**	**2.7**	0.9	**1.2**	**2.0**	0.8	**1.2**	**1.9**
1997	**1.6**	**2.1**	**3.4**	**1.0**	**1.5**	**2.5**	0.8	**1.1**	**1.9**	0.7	**1.1**	**1.7**
1998	**1.1**	**1.5**	**2.6**	0.8	**1.2**	**2.0**	0.7	**1.1**	**1.7**	0.7	**1.0**	**1.6**
1999	0.9	**1.4**	**2.3**	0.7	**1.1**	**1.8**	0.6	0.9	**1.5**	0.6	0.9	**1.4**
2000	**1.0**	**1.3**	**2.2**	0.7	**1.0**	**1.7**	0.6	0.8	**1.4**	0.6	0.8	**1.3**
2001	**1.1**	**1.5**	**2.4**	0.7	**1.0**	**1.7**	0.6	0.8	**1.3**	0.5	0.7	**1.2**
2002	0.9	**1.3**	**2.2**	0.7	**1.0**	**1.7**	0.5	0.8	**1.3**	0.5	0.7	**1.2**
2003	0.9	**1.2**	**2.0**	0.6	0.9	**1.5**	0.5	0.7	**1.2**	0.5	0.7	**1.1**
2004	0.6	0.9	**1.6**	0.5	0.8	**1.3**	0.5	0.7	**1.1**	0.4	0.6	**1.0**
2005	0.7	0.9	**1.5**	0.5	0.7	**1.2**	0.4	0.6	**1.0**	0.4	0.6	0.9
2006	0.8	**1.1**	**1.8**	0.5	0.7	**1.3**	0.4	0.6	**1.0**	0.4	0.5	0.9
2007	0.6	0.9	**1.5**	0.5	0.7	**1.2**	0.4	0.6	0.9	0.4	0.5	0.8
2008	0.6	0.8	**1.4**	0.5	0.7	**1.1**	0.4	0.5	0.9	0.3	0.5	0.8
2009	0.5	0.7	**1.3**	0.4	0.6	**1.0**	0.3	0.5	0.9	0.3	0.5	0.8
2010	0.6	0.8	**1.4**	0.4	0.6	**1.0**	0.3	0.5	0.8	0.3	0.4	0.7
2011	0.6	0.9	**1.5**	0.5	0.6	**1.1**	0.3	0.5	0.8	0.3	0.4	0.7
2012	0.6	0.8	**1.4**	0.4	0.6	**1.1**	0.3	0.5	0.8	0.3	0.4	0.7
2013	0.6	0.9	**1.5**	0.4	0.6	**1.1**	0.3	0.5	0.8	0.3	0.4	0.7
2014	0.6	0.8	**1.4**	0.4	0.6	**1.0**	0.3	0.5	0.8	0.3	0.4	0.7
2015	0.5	0.7	**1.2**	0.4	0.6	**1.0**	0.3	0.4	0.8	0.3	0.4	0.6
2016	0.4	0.6	**1.0**	0.4	0.5	0.9	0.3	0.4	0.7	0.3	0.4	0.6
2017	0.4	0.6	**1.0**	0.3	0.5	0.8	0.3	0.4	0.7	0.3	0.4	0.6
2018	0.4	0.6	**1.0**	0.3	0.5	0.8	0.3	0.4	0.7	0.2	0.3	0.6

Bold values indicate exceedances

TABLE 5-4: RATIO OF PREDICTED SPOTTAIL SHINER CONCENTRATIONS TO LABORATORY-DERIVED NOAEL FOR TRI+ PCBS

	River Mile 152			River Mile 113			River Mile 90			River Mile 50		
Year	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)
1993	0.22	0.29	0.48	0.16	0.21	0.31	0.13	0.17	0.24	0.13	0.16	0.23
1994	0.18	0.25	0.39	0.14	0.19	0.28	0.12	0.15	0.22	0.11	0.14	0.21
1995	0.14	0.18	0.32	0.11	0.15	0.22	0.10	0.13	0.19	0.10	0.13	0.18
1996	0.18	0.25	0.41	0.12	0.17	0.25	0.10	0.12	0.18	0.09	0.11	0.17
1997	0.16	0.20	0.32	0.11	0.14	0.21	0.09	0.11	0.17	0.08	0.11	0.15
1998	0.11	0.14	0.22	0.09	0.12	0.18	0.08	0.10	0.15	0.07	0.09	0.14
1999	0.10	0.12	0.20	0.08	0.10	0.16	0.07	0.09	0.13	0.07	0.09	0.13
2000	0.10	0.14	0.22	0.07	0.10	0.16	0.06	0.08	0.13	0.06	0.08	0.12
2001	0.12	0.15	0.24	0.08	0.11	0.16	0.06	0.08	0.12	0.06	0.07	0.11
2002	0.09	0.12	0.19	0.07	0.09	0.14	0.06	0.08	0.12	0.05	0.07	0.10
2003	0.08	0.11	0.18	0.07	0.09	0.14	0.05	0.07	0.11	0.05	0.07	0.10
2004	0.07	0.09	0.14	0.05	0.07	0.12	0.05	0.06	0.10	0.05	0.06	0.09
2005	0.07	0.09	0.15	0.05	0.07	0.11	0.04	0.06	0.09	0.04	0.06	0.08
2006	0.08	0.11	0.18	0.05	0.07	0.11	0.04	0.06	0.09	0.04	0.05	0.08
2007	0.06	0.09	0.14	0.05	0.07	0.11	0.04	0.05	0.08	0.04	0.05	0.08
2008	0.06	0.07	0.12	0.05	0.06	0.10	0.04	0.05	0.08	0.04	0.05	0.07
2009	0.05	0.07	0.11	0.04	0.06	0.09	0.03	0.05	0.07	0.03	0.05	0.07
2010	0.07	0.09	0.13	0.04	0.06	0.09	0.03	0.05	0.07	0.03	0.04	0.07
2011	0.06	0.08	0.13	0.05	0.06	0.10	0.03	0.05	0.07	0.03	0.04	0.06
2012	0.06	0.08	0.13	0.04	0.06	0.10	0.03	0.05	0.07	0.03	0.04	0.06
2013	0.06	0.08	0.14	0.04	0.06	0.10	0.03	0.05	0.07	0.03	0.04	0.06
2014	0.05	0.07	0.12	0.04	0.06	0.09	0.03	0.05	0.07	0.03	0.04	0.06
2015	0.05	0.07	0.12	0.04	0.06	0.09	0.03	0.04	0.07	0.03	0.04	0.06
2016	0.04	0.05	0.09	0.03	0.05	0.08	0.03	0.04	0.06	0.03	0.04	0.06
2017	0.04	0.05	0.08	0.03	0.04	0.07	0.03	0.04	0.06	0.03	0.04	0.05
2018	0.04	0.05	0.09	0.03	0.04	0.07	0.03	0.04	0.06	0.03	0.04	0.06

TABLE 5-5: RATIO OF PREDICTED SPOTTAIL SHINER CONCENTRATIONS TO LABORATORY-DERIVED LOAEL FOR TRI+ PCBS

	River Mile 152			River Mile 113			River Mile 90			River Mile 50		
Year	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)
1993	0.002	0.031	0.051	0.017	0.022	0.033	0.014	0.018	0.026	0.014	0.017	0.025
1994	0.002	0.027	0.042	0.015	0.021	0.030	0.013	0.016	0.024	0.012	0.015	0.022
1995	0.001	0.019	0.034	0.012	0.016	0.023	0.011	0.014	0.020	0.011	0.013	0.020
1996	0.002	0.027	0.044	0.013	0.018	0.027	0.010	0.013	0.019	0.010	0.012	0.018
1997	0.001	0.021	0.034	0.011	0.015	0.022	0.009	0.012	0.018	0.009	0.011	0.016
1998	0.001	0.015	0.023	0.009	0.013	0.019	0.008	0.010	0.016	0.008	0.010	0.015
1999	0.0009	0.013	0.021	0.008	0.011	0.017	0.007	0.010	0.014	0.007	0.009	0.014
2000	0.0009	0.015	0.023	0.008	0.011	0.017	0.007	0.009	0.013	0.007	0.008	0.012
2001	0.001	0.016	0.026	0.008	0.011	0.017	0.006	0.009	0.013	0.006	0.008	0.012
2002	0.0009	0.013	0.020	0.008	0.010	0.015	0.006	0.008	0.012	0.006	0.008	0.011
2003	0.0008	0.012	0.020	0.007	0.010	0.014	0.006	0.008	0.012	0.005	0.007	0.010
2004	0.0006	0.009	0.015	0.006	0.008	0.012	0.005	0.007	0.011	0.005	0.007	0.010
2005	0.0006	0.010	0.015	0.005	0.008	0.012	0.005	0.006	0.010	0.005	0.006	0.009
2006	0.0007	0.012	0.019	0.006	0.008	0.012	0.004	0.006	0.009	0.004	0.006	0.009
2007	0.0006	0.010	0.015	0.005	0.008	0.012	0.004	0.006	0.009	0.004	0.005	0.008
2008	0.0005	0.008	0.013	0.005	0.007	0.010	0.004	0.006	0.009	0.004	0.005	0.008
2009	0.0004	0.007	0.012	0.004	0.006	0.010	0.004	0.005	0.008	0.004	0.005	0.007
2010	0.0006	0.009	0.014	0.005	0.006	0.010	0.003	0.005	0.008	0.003	0.005	0.007
2011	0.0005	0.009	0.014	0.005	0.007	0.010	0.004	0.005	0.008	0.003	0.005	0.007
2012	0.0005	0.008	0.014	0.005	0.007	0.011	0.004	0.005	0.008	0.003	0.005	0.007
2013	0.0006	0.009	0.015	0.005	0.007	0.010	0.003	0.005	0.008	0.003	0.004	0.007
2014	0.0005	0.008	0.013	0.004	0.006	0.010	0.003	0.005	0.007	0.003	0.004	0.006
2015	0.0005	0.008	0.012	0.004	0.006	0.009	0.003	0.005	0.007	0.003	0.004	0.006
2016	0.0004	0.006	0.009	0.004	0.005	0.008	0.003	0.004	0.007	0.003	0.004	0.006
2017	0.0004	0.005	0.009	0.003	0.005	0.008	0.003	0.004	0.006	0.003	0.004	0.006
2018	0.0004	0.006	0.009	0.003	0.005	0.008	0.003	0.004	0.006	0.003	0.004	0.006

TABLE 5-6: RATIO OF PREDICTED PUMPKINSEED CONCENTRATIONS TO LABORATORY-DERIVED NOAEL ON A TEQ BASIS

	River Mile 152			River Mile 113			River Mile 90			River Mile 50		
Year	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)
1993	0.8	**1.1**	**1.9**	0.6	0.8	**1.4**	0.4	0.6	**1.1**	0.4	0.6	**1.0**
1994	0.6	0.9	**1.5**	0.5	0.7	**1.2**	0.4	0.6	**1.0**	0.4	0.5	0.9
1995	0.6	0.8	**1.4**	0.4	0.6	**1.0**	0.3	0.5	0.9	0.3	0.5	0.8
1996	0.7	**1.0**	**1.7**	0.4	0.6	**1.1**	0.3	0.5	0.8	0.3	0.4	0.8
1997	0.6	0.8	**1.4**	0.4	0.5	**1.0**	0.3	0.4	0.8	0.3	0.4	0.7
1998	0.4	0.6	**1.0**	0.3	0.5	0.8	0.3	0.4	0.7	0.3	0.4	0.6
1999	0.3	0.5	0.9	0.3	0.4	0.7	0.2	0.3	0.6	0.2	0.3	0.6
2000	0.4	0.5	0.9	0.3	0.4	0.7	0.2	0.3	0.6	0.2	0.3	0.5
2001	0.4	0.6	**1.0**	0.3	0.4	0.7	0.2	0.3	0.5	0.2	0.3	0.5
2002	0.3	0.5	0.8	0.3	0.4	0.7	0.2	0.3	0.5	0.2	0.3	0.5
2003	0.3	0.4	0.8	0.2	0.3	0.6	0.2	0.3	0.5	0.2	0.3	0.4
2004	0.2	0.4	0.6	0.2	0.3	0.5	0.2	0.3	0.5	0.2	0.2	0.4
2005	0.2	0.4	0.6	0.2	0.3	0.5	0.2	0.2	0.4	0.1	0.2	0.4
2006	0.3	0.4	0.7	0.2	0.3	0.5	0.2	0.2	0.4	0.1	0.2	0.4
2007	0.2	0.3	0.6	0.2	0.3	0.5	0.1	0.2	0.4	0.1	0.2	0.3
2008	0.2	0.3	0.6	0.2	0.3	0.5	0.1	0.2	0.4	0.1	0.2	0.3
2009	0.2	0.3	0.5	0.2	0.2	0.4	0.1	0.2	0.3	0.1	0.2	0.3
2010	0.2	0.3	0.6	0.2	0.2	0.4	0.1	0.2	0.3	0.1	0.2	0.3
2011	0.2	0.3	0.6	0.2	0.2	0.4	0.1	0.2	0.3	0.1	0.2	0.3
2012	0.2	0.3	0.6	0.2	0.2	0.4	0.1	0.2	0.3	0.1	0.2	0.3
2013	0.2	0.3	0.6	0.2	0.2	0.4	0.1	0.2	0.3	0.1	0.2	0.3
2014	0.2	0.3	0.6	0.2	0.2	0.4	0.1	0.2	0.3	0.1	0.2	0.3
2015	0.2	0.3	0.5	0.2	0.2	0.4	0.1	0.2	0.3	0.1	0.1	0.3
2016	0.2	0.2	0.4	0.1	0.2	0.4	0.1	0.2	0.3	0.1	0.1	0.3
2017	0.1	0.2	0.4	0.1	0.2	0.3	0.1	0.1	0.3	0.09	0.1	0.2
2018	0.1	0.2	0.4	0.1	0.2	0.3	0.1	0.1	0.3	0.09	0.1	0.2

Bold values indicate exceedances

302792

TABLE 5-7: RATIO OF PREDICTED PUMPKINSEED CONCENTRATIONS TO LABORATORY-DERIVED LOAEL ON A TEQ BASIS

	River Mile 152			River Mile 113			River Mile 90			River Mile 50		
Year	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)
1993	0.4	0.5	0.9	0.3	0.4	0.7	0.2	0.3	0.5	0.2	0.3	0.5
1994	0.3	0.4	0.7	0.2	0.3	0.6	0.2	0.3	0.5	0.2	0.3	0.5
1995	0.3	0.4	0.7	0.2	0.3	0.5	0.2	0.2	0.4	0.2	0.2	0.4
1996	0.3	0.5	0.8	0.2	0.3	0.5	0.2	0.2	0.4	0.1	0.2	0.4
1997	0.3	0.4	0.7	0.2	0.3	0.5	0.1	0.2	0.4	0.1	0.2	0.3
1998	0.2	0.3	0.5	0.2	0.2	0.4	0.1	0.2	0.3	0.1	0.2	0.3
1999	0.2	0.2	0.4	0.1	0.2	0.3	0.1	0.2	0.3	0.1	0.2	0.3
2000	0.2	0.2	0.4	0.1	0.2	0.3	0.1	0.2	0.3	0.1	0.1	0.3
2001	0.2	0.3	0.5	0.1	0.2	0.3	0.1	0.1	0.3	0.09	0.1	0.2
2002	0.2	0.2	0.4	0.1	0.2	0.3	0.1	0.1	0.3	0.09	0.1	0.2
2003	0.2	0.2	0.4	0.1	0.2	0.3	0.09	0.1	0.2	0.08	0.1	0.2
2004	0.1	0.2	0.3	0.1	0.1	0.3	0.08	0.1	0.2	0.08	0.1	0.2
2005	0.1	0.2	0.3	0.09	0.1	0.2	0.08	0.1	0.2	0.07	0.1	0.2
2006	0.1	0.2	0.3	0.10	0.1	0.2	0.07	0.1	0.2	0.07	0.1	0.2
2007	0.1	0.2	0.3	0.09	0.1	0.2	0.07	0.1	0.2	0.06	0.09	0.2
2008	0.1	0.1	0.3	0.08	0.1	0.2	0.07	0.1	0.2	0.06	0.09	0.2
2009	0.1	0.1	0.2	0.08	0.1	0.2	0.06	0.09	0.2	0.06	0.08	0.1
2010	0.1	0.2	0.3	0.08	0.1	0.2	0.06	0.09	0.2	0.06	0.08	0.1
2011	0.1	0.2	0.3	0.08	0.1	0.2	0.06	0.09	0.2	0.05	0.08	0.1
2012	0.1	0.2	0.3	0.08	0.1	0.2	0.06	0.09	0.2	0.05	0.08	0.1
2013	0.1	0.2	0.3	0.08	0.1	0.2	0.06	0.09	0.2	0.05	0.08	0.1
2014	0.1	0.2	0.3	0.08	0.1	0.2	0.06	0.08	0.2	0.05	0.07	0.1
2015	0.09	0.1	0.2	0.07	0.1	0.2	0.06	0.08	0.1	0.05	0.07	0.1
2016	0.07	0.1	0.2	0.06	0.09	0.2	0.05	0.08	0.1	0.05	0.07	0.1
2017	0.07	0.1	0.2	0.06	0.09	0.2	0.05	0.07	0.1	0.05	0.07	0.1
2018	0.07	0.1	0.2	0.06	0.08	0.2	0.05	0.07	0.1	0.04	0.06	0.1

Bold values indicate exceedances

TABLE 5-8: RATIO OF PREDICTED SPOTTAIL SHINER CONCENTRATIONS TO LABORATORY-DERIVED NOAEL ON A TEQ BASIS

Year	River Mile 152			River Mile 113			River Mile 90			River Mile 50		
	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)
1993	0.05	0.07	0.1	0.04	0.05	0.1	0.03	0.04	0.08	0.03	0.04	0.07
1994	0.05	0.07	0.1	0.04	0.05	0.09	0.03	0.04	0.07	0.03	0.04	0.07
1995	0.03	0.05	0.09	0.03	0.04	0.07	0.03	0.03	0.06	0.03	0.03	0.06
1996	0.05	0.07	0.1	0.03	0.04	0.08	0.02	0.03	0.06	0.02	0.03	0.05
1997	0.04	0.05	0.09	0.03	0.04	0.07	0.02	0.03	0.05	0.02	0.03	0.05
1998	0.03	0.04	0.07	0.02	0.03	0.06	0.02	0.03	0.05	0.02	0.02	0.04
1999	0.02	0.03	0.06	0.02	0.03	0.05	0.02	0.02	0.04	0.02	0.02	0.04
2000	0.03	0.04	0.06	0.02	0.03	0.05	0.02	0.02	0.04	0.02	0.02	0.04
2001	0.03	0.04	0.07	0.02	0.03	0.05	0.02	0.02	0.04	0.01	0.02	0.03
2002	0.02	0.03	0.06	0.02	0.03	0.05	0.02	0.02	0.04	0.01	0.02	0.03
2003	0.02	0.03	0.05	0.02	0.02	0.04	0.01	0.02	0.03	0.01	0.02	0.03
2004	0.02	0.02	0.04	0.01	0.02	0.04	0.01	0.02	0.03	0.01	0.02	0.03
2005	0.02	0.02	0.04	0.01	0.02	0.03	0.01	0.02	0.03	0.01	0.01	0.03
2006	0.02	0.03	0.05	0.01	0.02	0.03	0.01	0.02	0.03	0.01	0.01	0.03
2007	0.02	0.02	0.04	0.01	0.02	0.03	0.01	0.01	0.03	0.01	0.01	0.02
2008	0.01	0.02	0.03	0.01	0.02	0.03	0.01	0.01	0.03	0.01	0.01	0.02
2009	0.01	0.02	0.03	0.01	0.02	0.03	0.01	0.01	0.02	0.009	0.01	0.02
2010	0.02	0.02	0.04	0.01	0.02	0.03	0.009	0.01	0.02	0.009	0.01	0.02
2011	0.02	0.02	0.04	0.01	0.02	0.03	0.009	0.01	0.02	0.009	0.01	0.02
2012	0.01	0.02	0.04	0.01	0.02	0.03	0.009	0.01	0.02	0.009	0.01	0.02
2013	0.02	0.02	0.04	0.01	0.02	0.03	0.009	0.01	0.02	0.008	0.01	0.02
2014	0.01	0.02	0.04	0.01	0.02	0.03	0.009	0.01	0.02	0.008	0.01	0.02
2015	0.01	0.02	0.03	0.01	0.02	0.03	0.009	0.01	0.02	0.008	0.01	0.02
2016	0.01	0.01	0.03	0.009	0.01	0.02	0.008	0.01	0.02	0.008	0.01	0.02
2017	0.01	0.01	0.03	0.009	0.01	0.02	0.008	0.01	0.02	0.007	0.01	0.02
2018	0.01	0.01	0.03	0.009	0.01	0.02	0.007	0.01	0.02	0.007	0.009	0.02

Bold values indicate exceedances

302794

TABLE 5-9: RATIO OF PREDICTED SPOTTAIL SHINER CONCENTRATIONS TO LABORATORY-DERIVED LOAEL ON A TEQ BASIS

Year	River Mile 152			River Mile 113			River Mile 90			River Mile 50		
	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)
1993	0.003	0.004	0.007	0.002	0.003	0.005	0.002	0.002	0.004	0.002	0.002	0.004
1994	0.003	0.004	0.006	0.002	0.003	0.005	0.002	0.002	0.004	0.002	0.002	0.003
1995	0.002	0.003	0.005	0.002	0.002	0.004	0.001	0.002	0.003	0.001	0.002	0.003
1996	0.003	0.003	0.006	0.002	0.002	0.004	0.001	0.002	0.003	0.001	0.002	0.003
1997	0.002	0.003	0.005	0.002	0.002	0.004	0.001	0.002	0.003	0.001	0.001	0.003
1998	0.001	0.002	0.004	0.001	0.002	0.003	0.001	0.001	0.003	0.001	0.001	0.002
1999	0.001	0.002	0.003	0.001	0.001	0.003	0.001	0.001	0.002	0.001	0.001	0.002
2000	0.001	0.002	0.003	0.001	0.001	0.002	0.001	0.001	0.002	0.001	0.001	0.002
2001	0.002	0.002	0.004	0.001	0.001	0.003	0.001	0.001	0.002	0.001	0.001	0.002
2002	0.001	0.002	0.003	0.001	0.001	0.002	0.001	0.001	0.002	0.001	0.001	0.002
2003	0.001	0.002	0.003	0.001	0.001	0.002	0.001	0.001	0.002	0.001	0.001	0.002
2004	0.001	0.001	0.002	0.001	0.001	0.002	0.001	0.001	0.002	0.001	0.001	0.002
2005	0.001	0.001	0.002	0.001	0.001	0.002	0.001	0.001	0.002	0.001	0.001	0.001
2006	0.001	0.001	0.003	0.001	0.001	0.002	0.001	0.001	0.001	0.001	0.001	0.001
2007	0.001	0.001	0.002	0.001	0.001	0.002	0.001	0.001	0.001	0.001	0.001	0.001
2008	0.001	0.001	0.002	0.001	0.001	0.002	0.001	0.001	0.001	0.001	0.001	0.001
2009	0.001	0.001	0.002	0.001	0.001	0.001	0.001	0.001	0.001	0.000	0.001	0.001
2010	0.001	0.001	0.002	0.001	0.001	0.001	0.000	0.001	0.001	0.000	0.001	0.001
2011	0.001	0.001	0.002	0.001	0.001	0.002	0.000	0.001	0.001	0.000	0.001	0.001
2012	0.001	0.001	0.002	0.001	0.001	0.002	0.000	0.001	0.001	0.000	0.001	0.001
2013	0.001	0.001	0.002	0.001	0.001	0.002	0.000	0.001	0.001	0.000	0.001	0.001
2014	0.001	0.001	0.002	0.001	0.001	0.001	0.000	0.001	0.001	0.000	0.001	0.001
2015	0.001	0.001	0.002	0.001	0.001	0.001	0.000	0.001	0.001	0.000	0.001	0.001
2016	0.001	0.001	0.001	0.000	0.001	0.001	0.000	0.001	0.001	0.000	0.001	0.001
2017	0.001	0.001	0.001	0.000	0.001	0.001	0.000	0.001	0.001	0.000	0.000	0.001
2018	0.001	0.001	0.001	0.000	0.001	0.001	0.000	0.001	0.001	0.000	0.000	0.001

Bold values indicate exceedances

302795

TABLE 5-10: RATIO OF PREDICTED BROWN BULLHEAD CONCENTRATIONS TO LABORATORY-DERIVED NOAEL FOR TRI+ PCBS

	River Mile 152			River Mile 113			River Mile 90			River Mile 50		
Year	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)
1993	15	21	34	11	16	27	8.9	13	21	6.9	9.8	16
1994	13	18	31	10.4	15	25	8.4	12	20	6.5	9.2	15
1995	12	17	28	9.7	14	23	7.9	11	19	6.1	8.7	14
1996	12	17	29	9.3	13	22	7.5	11	18	5.7	8.2	14
1997	11	16	27	8.9	13	22	7.1	10	17	5.4	7.8	13
1998	11	15	25	8.4	12	20	6.8	9.8	16	5.2	7.4	12
1999	9.5	14	23	7.8	11	19	6.4	9.3	16	4.9	7.0	12
2000	9.3	14	23	7.5	11	18	6.0	8.8	15	4.6	6.7	11
2001	9.4	14	23	7.4	11	18	5.8	8.5	14	4.4	6.4	11
2002	9.0	13	22	7.2	10	18	5.7	8.3	14	4.3	6.2	10
2003	8.4	12	21	6.8	10	17	5.4	8.0	13	4.1	6.0	10
2004	7.8	11	19	6.5	9.5	16	5.2	7.6	13	4.0	5.8	9.6
2005	7.6	11	19	6.2	9.1	15	5.0	7.3	12	3.8	5.5	9.3
2006	7.7	11	19	6.1	8.9	15	4.8	7.1	12	3.7	5.3	8.9
2007	7.3	11	18	6.0	8.7	15	4.7	6.9	11	3.5	5.2	8.6
2008	7.0	10	17	5.8	8.4	14	4.5	6.6	11	3.4	5.0	8.4
2009	6.7	9.8	17	5.6	8.1	14	4.4	6.4	11	3.3	4.8	8.1
2010	6.6	9.8	17	5.4	8.0	13	4.3	6.2	10	3.2	4.7	7.8
2011	6.7	9.7	16	5.4	7.8	13	4.2	6.1	10	3.1	4.6	7.6
2012	6.5	9.5	16	5.3	7.7	13	4.1	6.0	10	3.1	4.5	7.5
2013	6.4	9.3	16	5.2	7.6	13	4.0	5.8	9.8	3.0	4.4	7.3
2014	6.2	9.0	15	5.0	7.4	12	3.9	5.7	9.5	2.9	4.2	7.1
2015	5.9	8.6	15	4.9	7.1	12	3.8	5.5	9.3	2.8	4.1	6.9
2016	5.6	8.3	14	4.7	6.9	12	3.7	5.4	9.0	2.8	4.0	6.7
2017	5.5	8.0	13	4.6	6.7	11	3.6	5.2	8.8	2.7	3.9	6.5
2018	5.3	7.8	13	4.4	6.5	11	3.4	5.1	8.6	2.6	3.8	6.4

TABLE 5-11: RATIO OF PREDICTED BROWN BULLHEAD CONCENTRATIONS TO LABORATORY-DERIVED LOAEL FOR TRI+ PCBS

	River Mile 152			River Mile 113			River Mile 90			River Mile 50		
Year	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)
1993	**1.6**	**2.2**	**3.7**	**1.2**	**1.7**	**2.9**	1.0	**1.4**	**2.3**	0.7	**1.0**	**1.7**
1994	**1.4**	**2.0**	**3.3**	**1.1**	**1.6**	**2.7**	0.9	**1.3**	**2.2**	0.7	**1.0**	**1.6**
1995	**1.3**	**1.8**	**3.0**	**1.0**	**1.5**	**2.5**	0.8	**1.2**	**2.0**	0.6	0.9	**1.5**
1996	**1.3**	**1.8**	**3.1**	**1.0**	**1.4**	**2.4**	0.8	**1.1**	**1.9**	0.6	0.9	**1.5**
1997	**1.2**	**1.8**	**2.9**	**1.0**	**1.4**	**2.3**	0.8	**1.1**	**1.8**	0.6	0.8	**1.4**
1998	**1.1**	**1.6**	**2.7**	0.9	**1.3**	**2.2**	0.7	**1.0**	**1.8**	0.6	0.8	**1.3**
1999	**1.0**	**1.5**	**2.5**	0.8	**1.2**	**2.0**	0.7	**1.0**	**1.7**	0.5	0.8	**1.3**
2000	**1.0**	**1.4**	**2.4**	0.8	**1.2**	**2.0**	0.6	0.9	**1.6**	0.5	0.7	**1.2**
2001	**1.0**	**1.4**	**2.4**	0.8	**1.1**	**1.9**	0.6	0.9	**1.5**	0.5	0.7	**1.1**
2002	**1.0**	**1.4**	**2.3**	0.8	**1.1**	**1.9**	0.6	0.9	**1.5**	0.5	0.7	**1.1**
2003	0.9	**1.3**	**2.2**	0.7	**1.1**	**1.8**	0.6	0.8	**1.4**	0.4	0.6	**1.1**
2004	0.8	**1.2**	**2.1**	0.7	**1.0**	**1.7**	0.6	0.8	**1.4**	0.4	0.6	**1.0**
2005	0.8	**1.2**	**2.0**	0.7	**1.0**	**1.6**	0.5	0.8	**1.3**	0.4	0.6	**1.0**
2006	0.8	**1.2**	**2.0**	0.7	**1.0**	**1.6**	0.5	0.8	**1.3**	0.4	0.6	**1.0**
2007	0.8	**1.1**	**1.9**	0.6	0.9	**1.6**	0.5	0.7	**1.2**	0.4	0.5	0.9
2008	0.8	**1.1**	**1.8**	0.6	0.9	**1.5**	0.5	0.7	**1.2**	0.4	0.5	0.9
2009	0.7	**1.0**	**1.8**	0.6	0.9	**1.5**	0.5	0.7	**1.1**	0.4	0.5	0.9
2010	0.7	**1.0**	**1.8**	0.6	0.8	**1.4**	0.5	0.7	**1.1**	0.3	0.5	0.8
2011	0.7	**1.0**	**1.7**	0.6	0.8	**1.4**	0.4	0.6	**1.1**	0.3	0.5	0.8
2012	0.7	**1.0**	**1.7**	0.6	0.8	**1.4**	0.4	0.6	**1.1**	0.3	0.5	0.8
2013	0.7	**1.0**	**1.7**	0.6	0.8	**1.4**	0.4	0.6	**1.0**	0.3	0.5	0.8
2014	0.7	**1.0**	**1.6**	0.5	0.8	**1.3**	0.4	0.6	**1.0**	0.3	0.5	0.8
2015	0.6	0.9	**1.6**	0.5	0.8	**1.3**	0.4	0.6	**1.0**	0.3	0.4	0.7
2016	0.6	0.9	**1.5**	0.5	0.7	**1.2**	0.4	0.6	**1.0**	0.3	0.4	0.7
2017	0.6	0.9	**1.4**	0.5	0.7	**1.2**	0.4	0.6	0.9	0.3	0.4	0.7
2018	0.6	0.8	**1.4**	0.5	0.7	**1.2**	0.4	0.5	0.9	0.3	0.4	0.7

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TABLE 5-12: RATIO OF PREDICTED BROWN BULLHEAD CONCENTRATIONS TO LABORATORY-DERIVED NOAEL ON A TEQ BASIS

Year	River Mile 152 25th (mg/kg wet weight)	River Mile 152 Median (mg/kg wet weight)	River Mile 152 95th Percentile (mg/kg wet weight)	River Mile 113 25th (mg/kg wet weight)	River Mile 113 Median (mg/kg wet weight)	River Mile 113 95th Percentile (mg/kg wet weight)	River Mile 90 25th (mg/kg wet weight)	River Mile 90 Median (mg/kg wet weight)	River Mile 90 95th Percentile (mg/kg wet weight)	River Mile 50 25th (mg/kg wet weight)	River Mile 50 Median (mg/kg wet weight)	River Mile 50 95th Percentile (mg/kg wet weight)
1993	0.04	0.05	0.09	0.03	0.04	0.07	0.02	0.03	0.06	0.02	0.02	0.04
1994	0.03	0.05	0.08	0.03	0.04	0.07	0.02	0.03	0.05	0.02	0.02	0.04
1995	0.03	0.04	0.07	0.02	0.04	0.06	0.02	0.03	0.05	0.01	0.02	0.04
1996	0.03	0.04	0.08	0.02	0.03	0.06	0.02	0.03	0.05	0.01	0.02	0.04
1997	0.03	0.04	0.07	0.02	0.03	0.06	0.02	0.03	0.05	0.01	0.02	0.03
1998	0.03	0.04	0.07	0.02	0.03	0.05	0.02	0.02	0.04	0.01	0.02	0.03
1999	0.02	0.03	0.06	0.02	0.03	0.05	0.02	0.02	0.04	0.01	0.02	0.03
2000	0.02	0.03	0.06	0.02	0.03	0.05	0.01	0.02	0.04	0.01	0.02	0.03
2001	0.02	0.03	0.06	0.02	0.03	0.05	0.01	0.02	0.04	0.01	0.02	0.03
2002	0.02	0.03	0.06	0.02	0.03	0.05	0.01	0.02	0.04	0.01	0.02	0.03
2003	0.02	0.03	0.05	0.02	0.03	0.04	0.01	0.02	0.03	0.01	0.02	0.03
2004	0.02	0.03	0.05	0.02	0.02	0.04	0.01	0.02	0.03	0.01	0.01	0.03
2005	0.02	0.03	0.05	0.02	0.02	0.04	0.01	0.02	0.03	0.009	0.01	0.02
2006	0.02	0.03	0.05	0.01	0.02	0.04	0.01	0.02	0.03	0.009	0.01	0.02
2007	0.02	0.03	0.05	0.01	0.02	0.04	0.01	0.02	0.03	0.009	0.01	0.02
2008	0.02	0.03	0.05	0.01	0.02	0.04	0.01	0.02	0.03	0.008	0.01	0.02
2009	0.02	0.03	0.04	0.01	0.02	0.04	0.01	0.02	0.03	0.008	0.01	0.02
2010	0.02	0.02	0.04	0.01	0.02	0.04	0.01	0.02	0.03	0.008	0.01	0.02
2011	0.02	0.02	0.04	0.01	0.02	0.03	0.01	0.02	0.03	0.008	0.01	0.02
2012	0.02	0.02	0.04	0.01	0.02	0.03	0.01	0.02	0.03	0.007	0.01	0.02
2013	0.02	0.02	0.04	0.01	0.02	0.03	0.01	0.01	0.03	0.007	0.01	0.02
2014	0.01	0.02	0.04	0.01	0.02	0.03	0.009	0.01	0.03	0.007	0.01	0.02
2015	0.01	0.02	0.04	0.01	0.02	0.03	0.009	0.01	0.02	0.007	0.01	0.02
2016	0.01	0.02	0.04	0.01	0.02	0.03	0.009	0.01	0.02	0.007	0.01	0.02
2017	0.01	0.02	0.04	0.01	0.02	0.03	0.009	0.01	0.02	0.007	0.01	0.02
2018	0.01	0.02	0.03	0.01	0.02	0.03	0.01	0.02	0.03	0.006	0.01	0.02

Bold values indicate exceedances

302798

TABLE 5-13: RATIO OF PREDICTED BROWN BULLHEAD CONCENTRATIONS TO LABORATORY-DERIVED LOAEL ON A TEQ BASIS

	River Mile 152			River Mile 113			River Mile 90			River Mile 50		
Year	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)
1993	0.02	0.02	0.04	0.01	0.02	0.03	0.01	0.01	0.025	0.007	0.01	0.02
1994	0.01	0.02	0.04	0.01	0.02	0.03	0.01	0.01	0.024	0.007	0.01	0.02
1995	0.01	0.02	0.03	0.01	0.02	0.03	0.01	0.01	0.022	0.007	0.01	0.02
1996	0.01	0.02	0.03	0.01	0.02	0.03	0.008	0.01	0.021	0.006	0.009	0.02
1997	0.01	0.02	0.03	0.01	0.01	0.03	0.008	0.01	0.020	0.006	0.009	0.02
1998	0.01	0.02	0.03	0.009	0.01	0.02	0.007	0.01	0.02	0.006	0.008	0.01
1999	0.01	0.02	0.03	0.008	0.01	0.02	0.007	0.01	0.02	0.005	0.008	0.01
2000	0.01	0.02	0.03	0.008	0.01	0.02	0.006	0.01	0.02	0.005	0.008	0.01
2001	0.01	0.02	0.03	0.008	0.01	0.02	0.006	0.01	0.02	0.005	0.007	0.01
2002	0.01	0.01	0.03	0.008	0.01	0.02	0.006	0.009	0.02	0.005	0.007	0.01
2003	0.009	0.01	0.02	0.007	0.01	0.02	0.006	0.009	0.02	0.004	0.007	0.01
2004	0.008	0.01	0.02	0.007	0.01	0.02	0.006	0.009	0.01	0.004	0.006	0.01
2005	0.008	0.01	0.02	0.007	0.01	0.02	0.005	0.008	0.01	0.004	0.006	0.01
2006	0.008	0.01	0.02	0.007	0.01	0.02	0.005	0.008	0.01	0.004	0.006	0.01
2007	0.008	0.01	0.02	0.006	0.01	0.02	0.005	0.008	0.01	0.004	0.006	0.01
2008	0.008	0.01	0.02	0.006	0.01	0.02	0.005	0.007	0.01	0.004	0.006	0.01
2009	0.007	0.01	0.02	0.006	0.009	0.02	0.005	0.007	0.01	0.004	0.005	0.009
2010	0.007	0.01	0.02	0.006	0.009	0.02	0.005	0.007	0.01	0.003	0.005	0.009
2011	0.007	0.01	0.02	0.006	0.009	0.02	0.005	0.007	0.01	0.003	0.005	0.009
2012	0.007	0.01	0.02	0.006	0.009	0.02	0.004	0.007	0.01	0.003	0.005	0.009
2013	0.007	0.01	0.02	0.006	0.009	0.01	0.004	0.007	0.01	0.003	0.005	0.008
2014	0.007	0.01	0.02	0.005	0.008	0.01	0.004	0.006	0.01	0.003	0.005	0.008
2015	0.006	0.01	0.02	0.005	0.008	0.01	0.004	0.006	0.01	0.003	0.005	0.008
2016	0.006	0.009	0.02	0.005	0.008	0.01	0.004	0.006	0.01	0.003	0.005	0.008
2017	0.006	0.009	0.02	0.005	0.008	0.01	0.004	0.006	0.01	0.003	0.004	0.008
2018	0.006	0.009	0.02	0.005	0.007	0.01	0.005	0.007	0.01	0.003	0.004	0.007

Bold values indicate exceedances

302799

TABLE 5-14: RATIO OF PREDICTED WHITE PERCH CONCENTRATIONS TO FIELD-BASED NOAEL FOR TRI+ PCBS

	River Mile 152			River Mile 113			River Mile 90			River Mile 50		
Year	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)
1993	0.9	0.9	**1.1**	0.7	0.7	0.8	0.5	0.6	0.7	0.4	0.4	0.5
1994	0.7	0.8	0.9	0.6	0.7	0.8	0.5	0.5	0.6	0.4	0.4	0.5
1995	0.7	0.7	0.9	0.6	0.6	0.7	0.5	0.5	0.6	0.4	0.4	0.4
1996	0.7	0.8	0.9	0.5	0.6	0.7	0.4	0.5	0.5	0.3	0.4	0.4
1997	0.7	0.7	0.8	0.5	0.6	0.7	0.4	0.4	0.5	0.3	0.3	0.4
1998	0.6	0.6	0.8	0.5	0.5	0.6	0.4	0.4	0.5	0.3	0.3	0.4
1999	0.6	0.6	0.7	0.4	0.5	0.6	0.4	0.4	0.5	0.3	0.3	0.4
2000	0.5	0.6	0.7	0.4	0.5	0.5	0.3	0.4	0.4	0.3	0.3	0.3
2001	0.6	0.6	0.7	0.4	0.4	0.5	0.3	0.4	0.4	0.3	0.3	0.3
2002	0.5	0.6	0.7	0.4	0.4	0.5	0.3	0.3	0.4	0.3	0.3	0.3
2003	0.5	0.5	0.6	0.4	0.4	0.5	0.3	0.3	0.4	0.2	0.3	0.3
2004	0.4	0.5	0.6	0.4	0.4	0.5	0.3	0.3	0.4	0.2	0.2	0.3
2005	0.4	0.5	0.6	0.3	0.4	0.5	0.3	0.3	0.4	0.2	0.2	0.3
2006	0.4	0.5	0.6	0.3	0.4	0.4	0.3	0.3	0.4	0.2	0.2	0.3
2007	0.4	0.5	0.5	0.3	0.4	0.4	0.3	0.3	0.3	0.2	0.2	0.3
2008	0.4	0.4	0.5	0.3	0.3	0.4	0.3	0.3	0.3	0.2	0.2	0.3
2009	0.4	0.4	0.5	0.3	0.3	0.4	0.2	0.3	0.3	0.2	0.2	0.2
2010	0.4	0.4	0.5	0.3	0.3	0.4	0.2	0.3	0.3	0.2	0.2	0.2
2011	0.4	0.4	0.5	0.3	0.3	0.4	0.2	0.2	0.3	0.2	0.2	0.2
2012	0.4	0.4	0.5	0.3	0.3	0.4	0.2	0.2	0.3	0.2	0.2	0.2
2013	0.4	0.4	0.5	0.3	0.3	0.4	0.2	0.2	0.3	0.2	0.2	0.2
2014	0.4	0.4	0.5	0.3	0.3	0.4	0.2	0.2	0.3	0.2	0.2	0.2
2015	0.3	0.4	0.4	0.3	0.3	0.4	0.2	0.2	0.3	0.2	0.2	0.2
2016	0.3	0.3	0.4	0.3	0.3	0.3	0.2	0.2	0.3	0.2	0.2	0.2
2017	0.3	0.3	0.4	0.2	0.3	0.3	0.2	0.2	0.3	0.2	0.2	0.2
2018	0.3	0.3	0.4	0.2	0.3	0.3	0.2	0.2	0.3	0.1	0.2	0.2

Bold values indicate exceedances

TABLE 5-15: RATIO OF PREDICTED YELLOW PERCH CONCENTRATIONS TO LABORATORY-DERIVED NOAEL FOR TRI+ PCBS

	River Mile 152			River Mile 113			River Mile 90			River Mile 50		
Year	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)
1993	**5.3**	**6.2**	**8.0**	**4.0**	**4.7**	**6.1**	**3.2**	**3.7**	**4.9**	**2.5**	**3.0**	**3.8**
1994	**4.4**	**5.3**	**6.9**	**3.6**	**4.3**	**5.6**	**3.0**	**3.5**	**4.6**	**2.4**	**2.8**	**3.5**
1995	**4.2**	**5.0**	**6.5**	**3.4**	**4.0**	**5.3**	**2.8**	**3.3**	**4.3**	**2.2**	**2.6**	**3.3**
1996	**4.4**	**5.2**	**6.6**	**3.3**	**3.8**	**5.0**	**2.6**	**3.1**	**4.0**	**2.1**	**2.4**	**3.1**
1997	**4.1**	**4.9**	**6.3**	**3.1**	**3.7**	**4.9**	**2.5**	**2.9**	**3.9**	**1.9**	**2.3**	**3.0**
1998	**3.6**	**4.4**	**5.8**	**2.9**	**3.5**	**4.6**	**2.3**	**2.8**	**3.7**	**1.8**	**2.2**	**2.8**
1999	**3.3**	**3.9**	**5.2**	**2.7**	**3.2**	**4.2**	**2.2**	**2.6**	**3.4**	**1.7**	**2.0**	**2.7**
2000	**3.1**	**3.7**	**4.9**	**2.5**	**3.0**	**4.0**	**2.1**	**2.5**	**3.3**	**1.6**	**1.9**	**2.5**
2001	**3.2**	**3.9**	**5.1**	**2.5**	**3.0**	**4.0**	**2.0**	**2.4**	**3.1**	**1.5**	**1.8**	**2.4**
2002	**3.1**	**3.7**	**4.9**	**2.4**	**2.9**	**3.9**	**1.9**	**2.3**	**3.1**	**1.5**	**1.8**	**2.3**
2003	**2.9**	**3.4**	**4.5**	**2.3**	**2.8**	**3.7**	**1.9**	**2.2**	**3.0**	**1.4**	**1.7**	**2.2**
2004	**2.6**	**3.2**	**4.2**	**2.2**	**2.6**	**3.5**	**1.8**	**2.1**	**2.8**	**1.4**	**1.6**	**2.2**
2005	**2.5**	**3.0**	**4.0**	**2.1**	**2.5**	**3.3**	**1.7**	**2.0**	**2.7**	**1.3**	**1.6**	**2.1**
2006	**2.6**	**3.1**	**4.1**	**2.0**	**2.4**	**3.3**	**1.6**	**1.9**	**2.6**	**1.3**	**1.5**	**2.0**
2007	**2.5**	**2.9**	**3.9**	**2.0**	**2.4**	**3.2**	**1.5**	**1.9**	**2.5**	**1.2**	**1.5**	**1.9**
2008	**2.4**	**2.9**	**3.8**	**1.9**	**2.3**	**3.1**	**1.5**	**1.8**	**2.4**	**1.2**	**1.4**	**1.9**
2009	**2.2**	**2.7**	**3.5**	**1.8**	**2.2**	**2.9**	**1.4**	**1.8**	**2.3**	**1.1**	**1.4**	**1.8**
2010	**2.2**	**2.6**	**3.5**	**1.8**	**2.1**	**2.9**	**1.4**	**1.7**	**2.3**	**1.1**	**1.3**	**1.7**
2011	**2.3**	**2.7**	**3.6**	**1.8**	**2.1**	**2.9**	**1.4**	**1.7**	**2.2**	**1.1**	**1.3**	**1.7**
2012	**2.2**	**2.6**	**3.5**	**1.7**	**2.1**	**2.8**	**1.3**	**1.6**	**2.2**	**1.0**	**1.2**	**1.6**
2013	**2.2**	**2.6**	**3.4**	**1.7**	**2.1**	**2.8**	**1.3**	**1.6**	**2.1**	**1.0**	**1.2**	**1.6**
2014	**2.1**	**2.5**	**3.3**	**1.7**	**2.0**	**2.7**	**1.3**	**1.6**	**2.1**	**1.0**	**1.2**	**1.6**
2015	**2.0**	**2.4**	**3.2**	**1.6**	**2.0**	**2.6**	**1.3**	**1.5**	**2.0**	**1.0**	**1.2**	**1.5**
2016	**1.9**	**2.3**	**3.0**	**1.5**	**1.9**	**2.5**	**1.2**	**1.5**	**2.0**	0.9	**1.1**	**1.5**
2017	**1.8**	**2.2**	**2.9**	**1.5**	**1.8**	**2.4**	**1.2**	**1.4**	**1.9**	0.9	**1.1**	**1.4**
2018	**1.7**	**2.1**	**2.8**	**1.4**	**1.7**	**2.3**	**1.1**	**1.4**	**1.8**	0.9	**1.1**	**1.4**

Bold values indicate exceedances

TABLE 5-16: RATIO OF PREDICTED YELLOW PERCH CONCENTRATIONS TO LABORATORY-DERIVED LOAEL FOR TRI+ PCBS

	River Mile 152			River Mile 113			River Mile 90			River Mile 50		
Year	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)
1993	0.6	0.7	0.9	0.4	0.5	0.7	0.3	0.4	0.5	0.3	0.3	0.4
1994	0.5	0.6	0.7	0.4	0.5	0.6	0.3	0.4	0.5	0.3	0.3	0.4
1995	0.4	0.5	0.7	0.4	0.4	0.6	0.3	0.4	0.5	0.2	0.3	0.4
1996	0.5	0.6	0.7	0.3	0.4	0.5	0.3	0.3	0.4	0.2	0.3	0.3
1997	0.4	0.5	0.7	0.3	0.4	0.5	0.3	0.3	0.4	0.2	0.2	0.3
1998	0.4	0.5	0.6	0.3	0.4	0.5	0.2	0.3	0.4	0.2	0.2	0.3
1999	0.3	0.4	0.6	0.3	0.3	0.5	0.2	0.3	0.4	0.2	0.2	0.3
2000	0.3	0.4	0.5	0.3	0.3	0.4	0.2	0.3	0.3	0.2	0.2	0.3
2001	0.3	0.4	0.5	0.3	0.3	0.4	0.2	0.3	0.3	0.2	0.2	0.3
2002	0.3	0.4	0.5	0.3	0.3	0.4	0.2	0.2	0.3	0.2	0.2	0.2
2003	0.3	0.4	0.5	0.2	0.3	0.4	0.2	0.2	0.3	0.2	0.2	0.2
2004	0.3	0.3	0.4	0.2	0.3	0.4	0.2	0.2	0.3	0.1	0.2	0.2
2005	0.3	0.3	0.4	0.2	0.3	0.4	0.2	0.2	0.3	0.1	0.2	0.2
2006	0.3	0.3	0.4	0.2	0.3	0.3	0.2	0.2	0.3	0.1	0.2	0.2
2007	0.3	0.3	0.4	0.2	0.3	0.3	0.2	0.2	0.3	0.1	0.2	0.2
2008	0.3	0.3	0.4	0.2	0.2	0.3	0.2	0.2	0.3	0.1	0.1	0.2
2009	0.2	0.3	0.4	0.2	0.2	0.3	0.2	0.2	0.3	0.1	0.1	0.2
2010	0.2	0.3	0.4	0.2	0.2	0.3	0.1	0.2	0.2	0.1	0.1	0.2
2011	0.2	0.3	0.4	0.2	0.2	0.3	0.1	0.2	0.2	0.1	0.1	0.2
2012	0.2	0.3	0.4	0.2	0.2	0.3	0.1	0.2	0.2	0.1	0.1	0.2
2013	0.2	0.3	0.4	0.2	0.2	0.3	0.1	0.2	0.2	0.1	0.1	0.2
2014	0.2	0.3	0.4	0.2	0.2	0.3	0.1	0.2	0.2	0.1	0.1	0.2
2015	0.2	0.3	0.3	0.2	0.2	0.3	0.1	0.2	0.2	0.1	0.1	0.2
2016	0.2	0.2	0.3	0.2	0.2	0.3	0.1	0.2	0.2	0.1	0.1	0.2
2017	0.2	0.2	0.3	0.2	0.2	0.3	0.1	0.2	0.2	0.1	0.1	0.2
2018	0.2	0.2	0.3	0.2	0.2	0.2	0.1	0.1	0.2	0.1	0.1	0.2

Bold values indicate exceedances

TABLE 5-17: RATIO OF PREDICTED WHITE PERCH CONCENTRATIONS TO LABORATORY-DERIVED NOAEL ON A TEQ BASIS

Year	River Mile 152			River Mile 113			River Mile 90			River Mile 50		
	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)
1993	**1.5**	**2.0**	**3.7**	**1.2**	**1.5**	**2.8**	0.9	**1.2**	**2.2**	0.7	**1.0**	**1.7**
1994	**1.3**	**1.7**	**3.1**	**1.1**	**1.4**	**2.5**	0.9	**1.1**	**2.1**	0.7	0.9	**1.6**
1995	**1.2**	**1.6**	**3.0**	**1.0**	**1.3**	**2.4**	0.8	**1.1**	**1.9**	0.6	0.8	**1.5**
1996	**1.3**	**1.7**	**3.0**	**1.0**	**1.2**	**2.3**	0.8	**1.0**	**1.8**	0.6	0.8	**1.4**
1997	**1.2**	**1.6**	**2.9**	0.9	**1.2**	**2.2**	0.7	0.9	**1.7**	0.6	0.7	**1.3**
1998	**1.0**	**1.4**	**2.6**	0.9	**1.1**	**2.0**	0.7	0.9	**1.6**	0.5	0.7	**1.3**
1999	**1.0**	**1.3**	**2.3**	0.8	**1.0**	**1.9**	0.6	0.8	**1.5**	0.5	0.7	**1.2**
2000	0.9	**1.2**	**2.3**	0.7	**1.0**	**1.8**	0.6	0.8	**1.5**	0.5	0.6	**1.1**
2001	**1.0**	**1.3**	**2.3**	0.7	**1.0**	**1.8**	0.6	0.8	**1.4**	0.5	0.6	**1.1**
2002	0.9	**1.2**	**2.2**	0.7	**1.0**	**1.7**	0.6	0.7	**1.4**	0.4	0.6	**1.0**
2003	0.8	**1.1**	**2.0**	0.7	0.9	**1.6**	0.5	0.7	**1.3**	0.4	0.6	**1.0**
2004	0.8	**1.0**	**1.9**	0.6	0.9	**1.6**	0.5	0.7	**1.3**	0.4	0.5	**1.0**
2005	0.7	**1.0**	**1.8**	0.6	0.8	**1.5**	0.5	0.7	**1.2**	0.4	0.5	0.9
2006	0.8	**1.0**	**1.9**	0.6	0.8	**1.5**	0.5	0.6	**1.2**	0.4	0.5	0.9
2007	0.7	**1.0**	**1.8**	0.6	0.8	**1.4**	0.5	0.6	**1.1**	0.4	0.5	0.9
2008	0.7	0.9	**1.7**	0.6	0.8	**1.4**	0.4	0.6	**1.1**	0.3	0.5	0.8
2009	0.6	0.9	**1.6**	0.5	0.7	**1.3**	0.4	0.6	**1.1**	0.3	0.4	0.8
2010	0.7	0.9	**1.6**	0.5	0.7	**1.3**	0.4	0.6	**1.0**	0.3	0.4	0.8
2011	0.7	0.9	**1.6**	0.5	0.7	**1.3**	0.4	0.5	**1.0**	0.3	0.4	0.8
2012	0.6	0.9	**1.6**	0.5	0.7	**1.3**	0.4	0.5	**1.0**	0.3	0.4	0.7
2013	0.6	0.9	**1.6**	0.5	0.7	**1.2**	0.4	0.5	**1.0**	0.3	0.4	0.7
2014	0.6	0.8	**1.5**	0.5	0.7	**1.2**	0.4	0.5	0.9	0.3	0.4	0.7
2015	0.6	0.8	**1.4**	0.5	0.6	**1.2**	0.4	0.5	0.9	0.3	0.4	0.7
2016	0.6	0.7	**1.3**	0.5	0.6	**1.1**	0.4	0.5	0.9	0.3	0.4	0.7
2017	0.5	0.7	**1.3**	0.4	0.6	**1.1**	0.3	0.5	0.9	0.3	0.4	0.6
2018	0.5	0.7	**1.3**	0.4	0.6	**1.1**	0.3	0.4	0.8	0.3	0.3	0.6

Bold values indicate exceedances

302803

TABLE 5-18: RATIO OF PREDICTED WHITE PERCH CONCENTRATIONS TO LABORATORY-DERIVED LOAEL ON A TEQ BASIS

Year	River Mile 152			River Mile 113			River Mile 90			River Mile 50		
	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)
1993	0.7	**1.0**	**1.8**	0.6	0.7	**1.3**	0.5	0.6	**1.1**	0.4	0.5	0.8
1994	0.6	0.8	**1.5**	0.5	0.7	**1.2**	0.4	0.6	**1.0**	0.3	0.4	0.8
1995	0.6	0.8	**1.4**	0.5	0.6	**1.1**	0.4	0.5	0.9	0.3	0.4	0.7
1996	0.6	0.8	**1.5**	0.5	0.6	**1.1**	0.4	0.5	0.9	0.3	0.4	0.7
1997	0.6	0.8	**1.4**	0.4	0.6	**1.1**	0.4	0.5	0.8	0.3	0.4	0.6
1998	0.5	0.7	**1.2**	0.4	0.5	**1.0**	0.3	0.4	0.8	0.3	0.3	0.6
1999	0.5	0.6	**1.1**	0.4	0.5	0.9	0.3	0.4	0.7	0.2	0.3	0.6
2000	0.5	0.6	**1.1**	0.4	0.5	0.9	0.3	0.4	0.7	0.2	0.3	0.5
2001	0.5	0.6	**1.1**	0.4	0.5	0.9	0.3	0.4	0.7	0.2	0.3	0.5
2002	0.4	0.6	**1.1**	0.3	0.5	0.8	0.3	0.4	0.7	0.2	0.3	0.5
2003	0.4	0.5	**1.0**	0.3	0.4	0.8	0.3	0.3	0.6	0.2	0.3	0.5
2004	0.4	0.5	0.9	0.3	0.4	0.8	0.3	0.3	0.6	0.2	0.3	0.5
2005	0.4	0.5	0.9	0.3	0.4	0.7	0.2	0.3	0.6	0.2	0.2	0.4
2006	0.4	0.5	0.9	0.3	0.4	0.7	0.2	0.3	0.6	0.2	0.2	0.4
2007	0.4	0.5	0.9	0.3	0.4	0.7	0.2	0.3	0.5	0.2	0.2	0.4
2008	0.3	0.5	0.8	0.3	0.4	0.7	0.2	0.3	0.5	0.2	0.2	0.4
2009	0.3	0.4	0.8	0.3	0.3	0.6	0.2	0.3	0.5	0.2	0.2	0.4
2010	0.3	0.4	0.8	0.3	0.3	0.6	0.2	0.3	0.5	0.2	0.2	0.4
2011	0.3	0.4	0.8	0.3	0.3	0.6	0.2	0.3	0.5	0.1	0.2	0.4
2012	0.3	0.4	0.8	0.2	0.3	0.6	0.2	0.3	0.5	0.1	0.2	0.4
2013	0.3	0.4	0.8	0.2	0.3	0.6	0.2	0.3	0.5	0.1	0.2	0.3
2014	0.3	0.4	0.7	0.2	0.3	0.6	0.2	0.2	0.5	0.1	0.2	0.3
2015	0.3	0.4	0.7	0.2	0.3	0.6	0.2	0.2	0.4	0.1	0.2	0.3
2016	0.3	0.4	0.7	0.2	0.3	0.5	0.2	0.2	0.4	0.1	0.2	0.3
2017	0.3	0.3	0.6	0.2	0.3	0.5	0.2	0.2	0.4	0.1	0.2	0.3
2018	0.3	0.3	0.6	0.2	0.3	0.5	0.2	0.2	0.4	0.1	0.2	0.3

Bold values indicate exceedances

302804

TABLE 5-19: RATIO OF PREDICTED YELLOW PERCH CONCENTRATIONS TO LABORATORY-DERIVED NOAEL ON A TEQ BASIS

Year	River Mile 152			River Mile 113			River Mile 90			River Mile 50		
	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)
1993	**1.6**	**1.8**	**3.4**	**1.2**	**1.4**	**2.5**	0.9	**1.1**	**2.0**	0.7	0.9	**1.6**
1994	**1.3**	**1.6**	**2.8**	**1.1**	**1.3**	**2.3**	0.9	**1.1**	**1.9**	0.7	0.8	**1.5**
1995	**1.2**	**1.5**	**2.7**	**1.0**	**1.2**	**2.2**	0.8	**1.0**	**1.8**	0.6	0.8	**1.4**
1996	**1.3**	**1.6**	**2.7**	0.9	**1.2**	**2.1**	0.8	0.9	**1.7**	0.6	0.7	**1.3**
1997	**1.2**	**1.4**	**2.6**	0.9	**1.1**	**2.0**	0.7	0.9	**1.6**	0.6	0.7	**1.2**
1998	**1.1**	**1.3**	**2.3**	0.9	**1.0**	**1.9**	0.7	0.8	**1.5**	0.5	0.6	**1.2**
1999	**1.0**	**1.2**	**2.1**	0.8	**1.0**	**1.7**	0.6	0.8	**1.4**	0.5	0.6	**1.1**
2000	0.9	**1.1**	**2.0**	0.7	0.9	**1.6**	0.6	0.7	**1.3**	0.5	0.6	**1.0**
2001	0.9	**1.2**	**2.0**	0.7	0.9	**1.6**	0.6	0.7	**1.3**	0.5	0.6	**1.0**
2002	0.9	**1.1**	**2.0**	0.7	0.9	**1.6**	0.6	0.7	**1.2**	0.4	0.5	**1.0**
2003	0.8	**1.0**	**1.9**	0.7	0.8	**1.5**	0.5	0.7	**1.2**	0.4	0.5	0.9
2004	0.8	**1.0**	**1.7**	0.6	0.8	**1.4**	0.5	0.6	**1.1**	0.4	0.5	0.9
2005	0.7	0.9	**1.6**	0.6	0.8	**1.3**	0.5	0.6	**1.1**	0.4	0.5	0.8
2006	0.8	**1.0**	**1.7**	0.6	0.8	**1.3**	0.5	0.6	**1.0**	0.4	0.5	0.8
2007	0.7	0.9	**1.6**	0.6	0.7	**1.3**	0.5	0.6	**1.0**	0.4	0.4	0.8
2008	0.7	0.9	**1.5**	0.6	0.7	**1.3**	0.4	0.6	**1.0**	0.3	0.4	0.8
2009	0.6	0.8	**1.4**	0.5	0.7	**1.2**	0.4	0.5	0.9	0.3	0.4	0.7
2010	0.7	0.8	**1.4**	0.5	0.7	**1.2**	0.4	0.5	0.9	0.3	0.4	0.7
2011	0.7	0.8	**1.5**	0.5	0.7	**1.2**	0.4	0.5	0.9	0.3	0.4	0.7
2012	0.6	0.8	**1.4**	0.5	0.6	**1.1**	0.4	0.5	0.9	0.3	0.4	0.7
2013	0.6	0.8	**1.4**	0.5	0.6	**1.1**	0.4	0.5	0.9	0.3	0.4	0.7
2014	0.6	0.8	**1.4**	0.5	0.6	**1.1**	0.4	0.5	0.8	0.3	0.4	0.6
2015	0.6	0.7	**1.3**	0.5	0.6	**1.1**	0.4	0.5	0.8	0.3	0.3	0.6
2016	0.6	0.7	**1.2**	0.5	0.6	**1.0**	0.4	0.4	0.8	0.3	0.3	0.6
2017	0.5	0.7	**1.2**	0.4	0.5	**1.0**	0.3	0.4	0.8	0.3	0.3	0.6
2018	0.5	0.6	**1.1**	0.4	0.5	0.9	0.3	0.4	0.7	0.3	0.3	0.6

Bold values indicate exceedances

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TABLE 5-20: RATIO OF PREDICTED YELLOW PERCH CONCENTRATIONS TO LABORATORY-DERIVED LOAEL ON A TEQ BASIS

Year	River Mile 152			River Mile 113			River Mile 90			River Mile 50		
	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)
1993	0.7	0.9	**1.6**	0.6	0.7	**1.2**	0.4	0.5	**1.0**	0.4	0.4	0.8
1994	0.6	0.8	**1.4**	0.5	0.6	**1.1**	0.4	0.5	0.9	0.3	0.4	0.7
1995	0.6	0.7	**1.3**	0.5	0.6	**1.1**	0.4	0.5	0.9	0.3	0.4	0.7
1996	0.6	0.8	**1.3**	0.5	0.6	**1.0**	0.4	0.4	0.8	0.3	0.4	0.6
1997	0.6	0.7	**1.3**	0.4	0.5	**1.0**	0.4	0.4	0.8	0.3	0.3	0.6
1998	0.5	0.6	**1.1**	0.4	0.5	0.9	0.3	0.4	0.7	0.3	0.3	0.6
1999	0.5	0.6	**1.0**	0.4	0.5	0.8	0.3	0.4	0.7	0.2	0.3	0.5
2000	0.4	0.6	**1.0**	0.4	0.4	0.8	0.3	0.4	0.6	0.2	0.3	0.5
2001	0.5	0.6	**1.0**	0.4	0.4	0.8	0.3	0.3	0.6	0.2	0.3	0.5
2002	0.4	0.5	**1.0**	0.3	0.4	0.8	0.3	0.3	0.6	0.2	0.3	0.5
2003	0.4	0.5	0.9	0.3	0.4	0.7	0.3	0.3	0.6	0.2	0.2	0.4
2004	0.4	0.5	0.8	0.3	0.4	0.7	0.3	0.3	0.6	0.2	0.2	0.4
2005	0.4	0.4	0.8	0.3	0.4	0.7	0.2	0.3	0.5	0.2	0.2	0.4
2006	0.4	0.5	0.8	0.3	0.4	0.6	0.2	0.3	0.5	0.2	0.2	0.4
2007	0.4	0.4	0.8	0.3	0.4	0.6	0.2	0.3	0.5	0.2	0.2	0.4
2008	0.3	0.4	0.7	0.3	0.3	0.6	0.2	0.3	0.5	0.2	0.2	0.4
2009	0.3	0.4	0.7	0.3	0.3	0.6	0.2	0.3	0.5	0.2	0.2	0.4
2010	0.3	0.4	0.7	0.3	0.3	0.6	0.2	0.2	0.4	0.2	0.2	0.3
2011	0.3	0.4	0.7	0.3	0.3	0.6	0.2	0.2	0.4	0.2	0.2	0.3
2012	0.3	0.4	0.7	0.2	0.3	0.6	0.2	0.2	0.4	0.1	0.2	0.3
2013	0.3	0.4	0.7	0.2	0.3	0.5	0.2	0.2	0.4	0.1	0.2	0.3
2014	0.3	0.4	0.7	0.2	0.3	0.5	0.2	0.2	0.4	0.1	0.2	0.3
2015	0.3	0.3	0.6	0.2	0.3	0.5	0.2	0.2	0.4	0.1	0.2	0.3
2016	0.3	0.3	0.6	0.2	0.3	0.5	0.2	0.2	0.4	0.1	0.2	0.3
2017	0.3	0.3	0.6	0.2	0.3	0.5	0.2	0.2	0.4	0.1	0.2	0.3
2018	0.2	0.3	0.6	0.2	0.3	0.5	0.2	0.2	0.4	0.1	0.2	0.3

Bold values indicate exceedances

TABLE 5-21: RATIO OF PREDICTED LARGEMOUTH BASS CONCENTRATIONS TO FIELD-BASED NOAEL FOR TRI+ PCBS

	River Mile 152			River Mile 113			River Mile 90			River Mile 50		
Year	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)
1993	**23**	**29**	**43**	**15**	**19**	**29**	**3.7**	**4.5**	**6.1**	**3.5**	**4.2**	**5.7**
1994	**16**	**21**	**31**	**13**	**17**	**25**	**3.4**	**4.1**	**5.6**	**3.1**	**3.8**	**5.1**
1995	**14**	**18**	**27**	**12**	**15**	**22**	**3.0**	**3.7**	**5.1**	**2.8**	**3.4**	**4.6**
1996	**16**	**21**	**32**	**11**	**14**	**21**	**2.7**	**3.3**	**4.6**	**2.6**	**3.1**	**4.2**
1997	**15**	**19**	**29**	**11**	**13**	**20**	**2.6**	**3.1**	**4.3**	**2.3**	**2.8**	**3.8**
1998	**12**	**15**	**23**	**9**	**12**	**18**	**2.4**	**2.9**	**4.0**	**2.2**	**2.6**	**3.6**
1999	**10**	**13**	**20**	**8**	**10**	**15**	**2.1**	**2.6**	**3.6**	**2.0**	**2.4**	**3.2**
2000	**10**	**12**	**19**	**7.1**	**9.3**	**14**	**1.9**	**2.3**	**3.3**	**1.8**	**2.2**	**3.0**
2001	**11**	**14**	**21**	**7.3**	**9.4**	**14**	**1.8**	**2.2**	**3.1**	**1.7**	**2.0**	**2.8**
2002	**10**	**13**	**19**	**7.2**	**9.3**	**14**	**1.8**	**2.2**	**3.0**	**1.6**	**1.9**	**2.6**
2003	**8.7**	**11**	**17**	**6.6**	**8.5**	**13**	**1.7**	**2.1**	**2.9**	**1.5**	**1.8**	**2.5**
2004	**7.2**	**9.1**	**14**	**5.9**	**7.6**	**12**	**1.6**	**1.9**	**2.7**	**1.4**	**1.7**	**2.4**
2005	**6.7**	**8.7**	**13**	**5.4**	**7.0**	**11**	**1.4**	**1.8**	**2.5**	**1.3**	**1.6**	**2.2**
2006	**7.7**	**9.8**	**15**	**5.3**	**6.9**	**10**	**1.3**	**1.7**	**2.3**	**1.2**	**1.5**	**2.1**
2007	**7.0**	**9.0**	**14**	**5.2**	**6.7**	**10**	**1.3**	**1.6**	**2.3**	**1.2**	**1.4**	**2.0**
2008	**6.6**	**8.4**	**13**	**5.1**	**6.5**	**9.9**	**1.3**	**1.5**	**2.2**	**1.1**	**1.4**	**1.9**
2009	**5.6**	**7.3**	**11**	**4.6**	**5.9**	**9.1**	**1.2**	**1.5**	**2.1**	**1.1**	**1.3**	**1.8**
2010	**6.0**	**7.7**	**12**	**4.4**	**5.7**	**8.6**	**1.1**	**1.4**	**2.0**	**1.0**	**1.2**	**1.7**
2011	**6.6**	**8.6**	**13**	**4.6**	**6.0**	**9.1**	**1.1**	**1.4**	**1.9**	**1.0**	**1.2**	**1.7**
2012	**6.0**	**7.7**	**12**	**4.5**	**5.9**	**9.0**	**1.1**	**1.4**	**1.9**	**1.0**	**1.2**	**1.6**
2013	**6.4**	**8.4**	**13**	**4.7**	**6.1**	**9.2**	**1.1**	**1.4**	**2.0**	**1.0**	**1.2**	**1.6**
2014	**5.9**	**7.6**	**12**	**4.4**	**5.7**	**8.8**	**1.1**	**1.3**	**1.9**	0.9	**1.1**	**1.6**
2015	**5.4**	**7.0**	**11**	**4.2**	**5.5**	**8.3**	**1.0**	**1.3**	**1.8**	0.9	**1.1**	**1.5**
2016	**5.1**	**6.4**	**10**	**4.0**	**5.1**	**7.8**	**1.0**	**1.2**	**1.7**	0.9	**1.1**	**1.5**
2017	**4.5**	**5.8**	**8.9**	**3.6**	**4.7**	**7.2**	0.9	**1.2**	**1.6**	0.8	**1.0**	**1.4**
2018	**4.3**	**5.6**	**8.6**	**3.4**	**4.5**	**6.8**	0.9	**1.1**	**1.6**	0.8	**1.0**	**1.4**

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TABLE 5-22: RATIO OF PREDICTED LARGEMOUTH BASS CONCENTRATIONS TO LABORATORY-DERIVED NOAEL ON A TEQ BASIS

	River Mile 152			River Mile 113			River Mile 90			River Mile 50		
Year	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)
1993	**3.1**	**3.9**	**6.0**	**2.1**	**2.7**	**4.1**	**1.6**	**2.1**	**3.1**	**1.6**	**2.0**	**3.0**
1994	**2.3**	**3.0**	**4.4**	**1.9**	**2.4**	**3.6**	**1.5**	**1.9**	**2.9**	**1.4**	**1.8**	**2.7**
1995	**2.0**	**2.5**	**3.8**	**1.6**	**2.1**	**3.2**	**1.3**	**1.7**	**2.6**	**1.3**	**1.6**	**2.4**
1996	**2.3**	**3.0**	**4.5**	**1.5**	**2.0**	**3.0**	**1.2**	**1.6**	**2.4**	**1.1**	**1.4**	**2.2**
1997	**2.1**	**2.7**	**4.0**	**1.5**	**1.9**	**2.9**	**1.1**	**1.5**	**2.2**	**1.0**	**1.3**	**2.0**
1998	**1.6**	**2.1**	**3.2**	**1.3**	**1.7**	**2.5**	**1.1**	**1.3**	**2.0**	**1.0**	**1.2**	**1.8**
1999	**1.4**	**1.8**	**2.7**	**1.1**	**1.4**	**2.2**	0.9	**1.2**	**1.8**	0.9	**1.1**	**1.7**
2000	**1.4**	**1.8**	**2.7**	**1.0**	**1.3**	**2.0**	0.9	**1.1**	**1.7**	0.8	**1.0**	**1.5**
2001	**1.5**	**2.0**	**2.9**	**1.0**	**1.3**	**2.0**	0.8	**1.0**	**1.6**	0.7	0.9	**1.4**
2002	**1.4**	**1.7**	**2.6**	**1.0**	**1.3**	**2.0**	0.8	**1.0**	**1.5**	0.7	0.9	**1.4**
2003	**1.2**	**1.6**	**2.4**	0.9	**1.2**	**1.9**	0.7	**1.0**	**1.5**	0.7	0.9	**1.3**
2004	**1.0**	**1.3**	**2.0**	0.8	**1.1**	**1.7**	0.7	0.9	**1.4**	0.6	0.8	**1.2**
2005	**1.0**	**1.3**	**1.9**	0.8	**1.0**	**1.5**	0.6	0.8	**1.3**	0.6	0.7	**1.1**
2006	**1.1**	**1.4**	**2.2**	0.8	**1.0**	**1.5**	0.6	0.8	**1.2**	0.6	0.7	**1.1**
2007	**1.0**	**1.3**	**1.9**	0.7	**1.0**	**1.5**	0.6	0.8	**1.1**	0.5	0.7	**1.0**
2008	0.9	**1.2**	**1.8**	0.7	0.9	**1.4**	0.6	0.7	**1.1**	0.5	0.6	**1.0**
2009	0.8	**1.0**	**1.6**	0.7	0.8	**1.3**	0.5	0.7	**1.0**	0.5	0.6	0.9
2010	0.9	**1.1**	**1.7**	0.6	0.8	**1.2**	0.5	0.7	**1.0**	0.4	0.6	0.9
2011	0.9	**1.2**	**1.8**	0.7	0.9	**1.3**	0.5	0.6	**1.0**	0.4	0.6	0.9
2012	0.9	**1.1**	**1.7**	0.6	0.8	**1.3**	0.5	0.6	**1.0**	0.4	0.6	0.8
2013	0.9	**1.2**	**1.8**	0.7	0.9	**1.3**	0.5	0.7	**1.0**	0.4	0.6	0.9
2014	0.8	**1.1**	**1.6**	0.6	0.8	**1.2**	0.5	0.6	0.9	0.4	0.5	0.8
2015	0.8	**1.0**	**1.5**	0.6	0.8	**1.2**	0.5	0.6	0.9	0.4	0.5	0.8
2016	0.7	0.9	**1.4**	0.6	0.7	**1.1**	0.4	0.6	0.9	0.4	0.5	0.8
2017	0.6	0.8	**1.3**	0.5	0.7	**1.0**	0.4	0.5	0.8	0.4	0.5	0.7
2018	0.6	0.8	**1.2**	0.5	0.6	**1.0**	0.5	0.6	1.0	0.4	0.5	0.7

Bold values indicate exceedances

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TABLE 5-23: RATIO OF PREDICTED LARGEMOUTH BASS CONCENTRATIONS TO LABORATORY-DERIVED LOAEL ON A TEQ BASIS

Year	River Mile 152			River Mile 113			River Mile 90			River Mile 50		
	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)
1993	**1.5**	**1.9**	**2.9**	**1.0**	**1.3**	**2.0**	0.8	**1.0**	**1.5**	0.8	**1.0**	**1.4**
1994	**1.1**	**1.4**	**2.1**	0.9	**1.1**	**1.7**	0.7	0.9	**1.4**	0.7	0.9	**1.3**
1995	**1.0**	**1.2**	**1.9**	0.8	**1.0**	**1.5**	0.7	0.8	**1.2**	0.6	0.8	**1.2**
1996	**1.1**	**1.5**	**2.2**	0.7	**1.0**	**1.4**	0.6	0.8	**1.1**	0.5	0.7	**1.0**
1997	**1.0**	**1.3**	**1.9**	0.7	0.9	**1.4**	0.6	0.7	**1.1**	0.5	0.6	**1.0**
1998	0.8	**1.0**	**1.5**	0.6	0.8	**1.2**	0.5	0.6	**1.0**	0.5	0.6	0.9
1999	0.7	**0.9**	**1.3**	0.5	0.7	**1.1**	0.5	0.6	0.9	0.4	0.5	0.8
2000	0.7	**0.9**	**1.3**	0.5	0.6	**1.0**	0.4	0.5	0.8	0.4	0.5	0.7
2001	0.7	**1.0**	**1.4**	0.5	0.6	**1.0**	0.4	0.5	0.8	0.4	0.5	0.7
2002	0.7	0.8	**1.3**	0.5	0.6	**1.0**	0.4	0.5	0.7	0.3	0.4	0.7
2003	0.6	0.8	**1.2**	0.5	0.6	0.9	0.4	0.5	0.7	0.3	0.4	0.6
2004	0.5	0.6	**1.0**	0.4	0.5	0.8	0.3	0.4	0.7	0.3	0.4	0.6
2005	0.5	0.6	0.9	0.4	0.5	0.7	0.3	0.4	0.6	0.3	0.4	0.5
2006	0.5	0.7	**1.0**	0.4	0.5	0.7	0.3	0.4	0.6	0.3	0.3	0.5
2007	0.5	0.6	0.9	0.4	0.5	0.7	0.3	0.4	0.6	0.2	0.3	0.5
2008	0.4	0.6	0.9	0.3	0.4	0.7	0.3	0.4	0.5	0.2	0.3	0.5
2009	0.4	0.5	0.8	0.3	0.4	0.6	0.3	0.3	0.5	0.2	0.3	0.4
2010	0.4	0.6	0.8	0.3	0.4	0.6	0.2	0.3	0.5	0.2	0.3	0.4
2011	0.5	0.6	0.9	0.3	0.4	0.6	0.2	0.3	0.5	0.2	0.3	0.4
2012	0.4	0.5	0.8	0.3	0.4	0.6	0.2	0.3	0.5	0.2	0.3	0.4
2013	0.4	0.6	0.9	0.3	0.4	0.6	0.2	0.3	0.5	0.2	0.3	0.4
2014	0.4	0.5	0.8	0.3	0.4	0.6	0.2	0.3	0.5	0.2	0.3	0.4
2015	0.4	0.5	0.7	0.3	0.4	0.6	0.2	0.3	0.4	0.2	0.2	0.4
2016	0.3	0.4	0.7	0.3	0.4	0.5	0.2	0.3	0.4	0.2	0.2	0.4
2017	0.3	0.4	0.6	0.3	0.3	0.5	0.2	0.3	0.4	0.2	0.2	0.4
2018	0.3	0.4	0.6	0.2	0.3	0.5	0.2	0.3	0.5	0.2	0.2	0.3

Bold values indicate exceedances

TABLE 5-24: RATIO OF PREDICTED STRIPED BASS CONCENTRATIONS TO TRI+ AND TEQ PCB-BASED TRVs

	River Mile 152									River Mile 113								
	Tri+-based Field-derived TRV			TEQ-based Laboratory-derived TRV						Tri+-based Field-derived TRV			TEQ-based Laboratory-derived TRV					
	NOAEL			LOAEL			NOAEL			NOAEL			LOAEL			NOAEL		
Year	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)
1993	**9.2**	**12**	**18**	**3.7**	**4.7**	**7.1**	**7.7**	**9.8**	**15**	**1.3**	**1.6**	**2.4**	0.5	0.6	**1.0**	**1.0**	**1.3**	**2.0**
1994	**6.6**	**8.5**	**13**	**2.7**	**3.4**	**5.1**	**5.5**	**7.1**	**11**	**1.1**	**1.4**	**2.1**	0.4	0.6	0.9	0.9	**1.2**	**1.8**
1995	**5.8**	**7.3**	**11**	**2.3**	**2.9**	**4.5**	**4.8**	**6.1**	**9.2**	**1.0**	**1.3**	**1.9**	0.4	0.5	0.8	0.8	**1.0**	**1.6**
1996	**6.8**	**8.7**	**13**	**2.7**	**3.5**	**5.2**	**5.6**	**7.2**	**11**	0.9	**1.2**	**1.7**	0.4	0.5	0.7	0.8	**1.0**	**1.5**
1997	**6.2**	**7.9**	**12**	**2.5**	**3.2**	**4.8**	**5.2**	**6.6**	**9.9**	0.9	**1.1**	**1.7**	0.4	0.5	0.7	0.7	0.9	**1.4**
1998	**5.0**	**6.2**	**9.5**	**2.0**	**2.5**	**3.8**	**4.1**	**5.2**	**7.9**	0.8	**1.0**	**1.5**	0.3	0.4	0.6	0.7	0.9	**1.3**
1999	**4.1**	**5.3**	**8.0**	**1.7**	**2.2**	**3.2**	**3.5**	**4.5**	**6.7**	0.7	0.9	**1.3**	0.3	0.3	0.5	0.6	0.7	**1.1**
2000	**3.9**	**5.0**	**7.6**	**1.6**	**2.0**	**3.1**	**3.3**	**4.2**	**6.3**	0.6	0.8	**1.2**	0.2	0.3	0.5	0.5	0.6	**1.0**
2001	**4.4**	**5.7**	**8.5**	**1.8**	**2.3**	**3.4**	**3.6**	**4.8**	**7.1**	0.6	0.8	**1.2**	0.2	0.3	0.5	0.5	0.7	**1.0**
2002	**4.2**	**5.2**	**7.9**	**1.7**	**2.1**	**3.2**	**3.5**	**4.4**	**6.6**	0.6	0.8	**1.2**	0.2	0.3	0.5	0.5	0.7	**1.0**
2003	**3.6**	**4.6**	**7.0**	**1.4**	**1.9**	**2.8**	**3.0**	**3.9**	**5.8**	0.6	0.7	**1.1**	0.2	0.3	0.4	0.5	0.6	0.9
2004	**2.9**	**3.7**	**5.7**	**1.2**	**1.5**	**2.3**	**2.5**	**3.1**	**4.8**	0.5	0.6	**1.0**	0.2	0.3	0.4	0.4	0.5	0.8
2005	**2.7**	**3.6**	**5.4**	**1.1**	**1.4**	**2.2**	**2.3**	**3.0**	**4.5**	0.4	0.6	0.9	0.2	0.2	0.4	0.4	0.5	0.7
2006	**3.1**	**4.0**	**6.1**	**1.3**	**1.6**	**2.5**	**2.6**	**3.3**	**5.1**	0.4	0.6	0.9	0.2	0.2	0.4	0.4	0.5	0.7
2007	**2.9**	**3.7**	**5.6**	**1.2**	**1.5**	**2.2**	**2.4**	**3.1**	**4.6**	0.4	0.6	0.9	0.2	0.2	0.3	0.4	0.5	0.7
2008	**2.7**	**3.4**	**5.2**	**1.1**	**1.4**	**2.1**	**2.3**	**2.9**	**4.4**	0.4	0.5	0.8	0.2	0.2	0.3	0.4	0.5	0.7
2009	**2.3**	**3.0**	**4.6**	0.9	**1.2**	**1.8**	**1.9**	**2.5**	**3.8**	0.4	0.5	0.8	0.2	0.2	0.3	0.3	0.4	0.6
2010	**2.4**	**3.1**	**4.8**	**1.0**	**1.3**	**1.9**	**2.0**	**2.6**	**4.0**	0.4	0.5	0.7	0.1	0.2	0.3	0.3	0.4	0.6
2011	**2.7**	**3.5**	**5.3**	**1.1**	**1.4**	**2.1**	**2.2**	**2.9**	**4.4**	0.4	0.5	0.8	0.2	0.2	0.3	0.3	0.4	0.6
2012	**2.4**	**3.1**	**4.8**	**1.0**	**1.3**	**1.9**	**2.0**	**2.6**	**4.0**	0.4	0.5	0.8	0.2	0.2	0.3	0.3	0.4	0.6
2013	**2.6**	**3.4**	**5.2**	**1.1**	**1.4**	**2.1**	**2.2**	**2.9**	**4.3**	0.4	0.5	0.8	0.2	0.2	0.3	0.3	0.4	0.6
2014	**2.4**	**3.1**	**4.7**	**1.0**	**1.3**	**1.9**	**2.0**	**2.6**	**4.0**	0.4	0.5	0.7	0.1	0.2	0.3	0.3	0.4	0.6
2015	**2.2**	**2.9**	**4.4**	0.9	**1.2**	**1.8**	**1.8**	**2.4**	**3.7**	0.4	0.5	0.7	0.1	0.2	0.3	0.3	0.4	0.6
2016	**2.1**	**2.6**	**4.1**	0.8	**1.1**	**1.6**	**1.8**	**2.2**	**3.4**	0.3	0.4	0.7	0.1	0.2	0.3	0.3	0.4	0.5
2017	**1.9**	**2.4**	**3.6**	0.8	**1.0**	**1.5**	**1.6**	**2.0**	**3.0**	0.3	0.4	0.6	0.1	0.2	0.2	0.3	0.3	0.5
2018	**1.8**	**2.3**	**3.5**	0.7	0.9	**1.4**	**1.5**	**1.9**	**2.9**	0.3	0.4	0.6	0.1	0.2	0.2	0.2	0.3	0.5

Note a Tri+ LOAEL was not determined
Bold values indicate exceedances

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TABLE 5-25: RATIO OF MODELED DIETARY DOSE BASED ON FISHRAND FOR FEMALE TREE SWALLOWS BASED ON THE SUM OF TRI+ CONGENERS FOR THE PERIOD 1993 - 2018

Year	LOAEL 152 Average	LOAEL 152 95% UCL	NOAEL 152 Average	NOAEL 152 95% UCL	LOAEL 113 Average	LOAEL 113 95% UCL	NOAEL 113 Average	NOAEL 113 95% UCL	LOAEL 90 Average	LOAEL 90 95% UCL	NOAEL 90 Average	NOAEL 90 95% UCL	LOAEL 50 Average	LOAEL 50 95% UCL	NOAEL 50 Average	NOAEL 50 95% UCL
1993	NA	NA	0.09	0.1	NA	NA	0.07	0.08	NA	NA	0.06	0.06	NA	NA	0.04	0.05
1994	NA	NA	0.08	0.09	NA	NA	0.07	0.07	NA	NA	0.06	0.06	NA	NA	0.04	0.04
1995	NA	NA	0.08	0.09	NA	NA	0.07	0.07	NA	NA	0.05	0.06	NA	NA	0.04	0.04
1996	NA	NA	0.08	0.09	NA	NA	0.06	0.07	NA	NA	0.05	0.05	NA	NA	0.04	0.04
1997	NA	NA	0.08	0.08	NA	NA	0.06	0.07	NA	NA	0.05	0.05	NA	NA	0.04	0.04
1998	NA	NA	0.07	0.08	NA	NA	0.06	0.06	NA	NA	0.05	0.05	NA	NA	0.03	0.04
1999	NA	NA	0.07	0.07	NA	NA	0.06	0.06	NA	NA	0.05	0.05	NA	NA	0.03	0.04
2000	NA	NA	0.07	0.07	NA	NA	0.06	0.06	NA	NA	0.04	0.05	NA	NA	0.03	0.04
2001	NA	NA	0.07	0.07	NA	NA	0.05	0.06	NA	NA	0.04	0.05	NA	NA	0.03	0.03
2002	NA	NA	0.06	0.07	NA	NA	0.05	0.06	NA	NA	0.04	0.04	NA	NA	0.03	0.03
2003	NA	NA	0.06	0.07	NA	NA	0.05	0.05	NA	NA	0.04	0.04	NA	NA	0.03	0.03
2004	NA	NA	0.06	0.06	NA	NA	0.05	0.05	NA	NA	0.04	0.04	NA	NA	0.03	0.03
2005	NA	NA	0.06	0.06	NA	NA	0.05	0.05	NA	NA	0.04	0.04	NA	NA	0.03	0.03
2006	NA	NA	0.06	0.06	NA	NA	0.05	0.05	NA	NA	0.04	0.04	NA	NA	0.03	0.03
2007	NA	NA	0.06	0.06	NA	NA	0.05	0.05	NA	NA	0.03	0.04	NA	NA	0.03	0.03
2008	NA	NA	0.05	0.06	NA	NA	0.04	0.05	NA	NA	0.03	0.04	NA	NA	0.02	0.03
2009	NA	NA	0.05	0.06	NA	NA	0.04	0.05	NA	NA	0.03	0.04	NA	NA	0.02	0.03
2010	NA	NA	0.05	0.06	NA	NA	0.04	0.05	NA	NA	0.03	0.03	NA	NA	0.02	0.03
2011	NA	NA	0.05	0.05	NA	NA	0.04	0.04	NA	NA	0.03	0.03	NA	NA	0.02	0.03
2012	NA	NA	0.05	0.05	NA	NA	0.04	0.04	NA	NA	0.03	0.03	NA	NA	0.02	0.02
2013	NA	NA	0.05	0.05	NA	NA	0.04	0.04	NA	NA	0.03	0.03	NA	NA	0.02	0.02
2014	NA	NA	0.05	0.05	NA	NA	0.04	0.04	NA	NA	0.03	0.03	NA	NA	0.02	0.02
2015	NA	NA	0.04	0.05	NA	NA	0.04	0.04	NA	NA	0.03	0.03	NA	NA	0.02	0.02
2016	NA	NA	0.05	0.05	NA	NA	0.04	0.04	NA	NA	0.03	0.03	NA	NA	0.02	0.02
2017	NA	NA	0.04	0.05	NA	NA	0.04	0.04	NA	NA	0.03	0.03	NA	NA	0.02	0.02
2018	NA	NA	0.04	0.05	NA	NA	0.04	0.04	NA	NA	0.03	0.03	NA	NA	0.02	0.02

Bold value indicates exceedances

TABLE 5-26 : RATIO OF MODELED EGG CONCENTRATIONS TO BENCHMARKS FOR FEMALE TREE SWALLOWS BASED ON THE SUM OF TRI+ CONGENERS FOR THE PERIOD 1993 - 2018

Year	LOAEL 152 Average	LOAEL 152 95% UCL	NOAEL 152 Average	NOAEL 152 95% UCL	LOAEL 113 Average	LOAEL 113 95% UCL	NOAEL 113 Average	NOAEL 113 95% UCL	LOAEL 90 Average	LOAEL 90 95% UCL	NOAEL 90 Average	NOAEL 90 95% UCL	LOAEL 50 Average	LOAEL 50 95% UCL	NOAEL 50 Average	NOAEL 50 95% UCL
1993	NA	NA	0.1	0.1	NA	NA	0.1	0.1	NA	NA	0.08	0.09	NA	NA	0.06	0.07
1994	NA	NA	0.1	0.1	NA	NA	0.1	0.1	NA	NA	0.08	0.09	NA	NA	0.06	0.06
1995	NA	NA	0.1	0.1	NA	NA	0.09	0.10	NA	NA	0.08	0.08	NA	NA	0.06	0.06
1996	NA	NA	0.1	0.1	NA	NA	0.09	0.10	NA	NA	0.07	0.08	NA	NA	0.05	0.06
1997	NA	NA	0.1	0.1	NA	NA	0.09	0.09	NA	NA	0.07	0.07	NA	NA	0.05	0.06
1998	NA	NA	0.1	0.1	NA	NA	0.08	0.09	NA	NA	0.07	0.07	NA	NA	0.05	0.05
1999	NA	NA	0.1	0.1	NA	NA	0.08	0.09	NA	NA	0.06	0.07	NA	NA	0.05	0.05
2000	NA	NA	0.1	0.1	NA	NA	0.08	0.08	NA	NA	0.06	0.07	NA	NA	0.05	0.05
2001	NA	NA	0.1	0.1	NA	NA	0.08	0.08	NA	NA	0.06	0.06	NA	NA	0.04	0.05
2002	NA	NA	0.09	0.1	NA	NA	0.07	0.08	NA	NA	0.06	0.06	NA	NA	0.04	0.05
2003	NA	NA	0.09	0.09	NA	NA	0.07	0.08	NA	NA	0.06	0.06	NA	NA	0.04	0.05
2004	NA	NA	0.08	0.09	NA	NA	0.07	0.07	NA	NA	0.05	0.06	NA	NA	0.04	0.04
2005	NA	NA	0.08	0.09	NA	NA	0.07	0.07	NA	NA	0.05	0.06	NA	NA	0.04	0.04
2006	NA	NA	0.08	0.08	NA	NA	0.07	0.07	NA	NA	0.05	0.05	NA	NA	0.04	0.04
2007	NA	NA	0.08	0.08	NA	NA	0.06	0.07	NA	NA	0.05	0.05	NA	NA	0.04	0.04
2008	NA	NA	0.07	0.08	NA	NA	0.06	0.07	NA	NA	0.05	0.05	NA	NA	0.04	0.04
2009	NA	NA	0.07	0.08	NA	NA	0.06	0.06	NA	NA	0.05	0.05	NA	NA	0.03	0.04
2010	NA	NA	0.07	0.08	NA	NA	0.06	0.06	NA	NA	0.05	0.05	NA	NA	0.03	0.04
2011	NA	NA	0.07	0.07	NA	NA	0.06	0.06	NA	NA	0.04	0.05	NA	NA	0.03	0.04
2012	NA	NA	0.07	0.07	NA	NA	0.06	0.06	NA	NA	0.04	0.05	NA	NA	0.03	0.03
2013	NA	NA	0.07	0.07	NA	NA	0.06	0.06	NA	NA	0.04	0.04	NA	NA	0.03	0.03
2014	NA	NA	0.07	0.07	NA	NA	0.05	0.06	NA	NA	0.04	0.04	NA	NA	0.03	0.03
2015	NA	NA	0.06	0.07	NA	NA	0.05	0.06	NA	NA	0.04	0.04	NA	NA	0.03	0.03
2016	NA	NA	0.06	0.07	NA	NA	0.05	0.06	NA	NA	0.04	0.04	NA	NA	0.03	0.03
2017	NA	NA	0.06	0.07	NA	NA	0.05	0.05	NA	NA	0.04	0.04	NA	NA	0.03	0.03
2018	NA	NA	0.06	0.07	NA	NA	0.05	0.05	NA	NA	0.04	0.04	NA	NA	0.03	0.03

Bold value indicates exceedances

TABLE 5-27: RATIO OF MODELED DIETARY DOSE BASED ON FISHRAND FOR FEMALE TREE SWALLOW USING TEQ FOR THE PERIOD 1993 - 2018

Year	LOAEL 152 Average	LOAEL 152 95% UCL	NOAEL 152 Average	NOAEL 152 95% UCL	LOAEL 113 Average	LOAEL 113 95% UCL	NOAEL 113 Average	NOAEL 113 95% UCL	LOAEL 90 Average	LOAEL 90 95% UCL	NOAEL 90 Average	NOAEL 90 95% UCL	LOAEL 50 Average	LOAEL 50 95% UCL	NOAEL 50 Average	NOAEL 50 95% UCL
1993	NA	NA	0.04	0.05	NA	NA	0.03	0.04	NA	NA	0.03	0.03	NA	NA	0.02	0.02
1994	NA	NA	0.04	0.04	NA	NA	0.03	0.03	NA	NA	0.03	0.03	NA	NA	0.02	0.02
1995	NA	NA	0.04	0.04	NA	NA	0.03	0.03	NA	NA	0.02	0.03	NA	NA	0.02	0.02
1996	NA	NA	0.04	0.04	NA	NA	0.03	0.03	NA	NA	0.02	0.02	NA	NA	0.02	0.02
1997	NA	NA	0.03	0.04	NA	NA	0.03	0.03	NA	NA	0.02	0.02	NA	NA	0.02	0.02
1998	NA	NA	0.03	0.04	NA	NA	0.03	0.03	NA	NA	0.02	0.02	NA	NA	0.02	0.02
1999	NA	NA	0.03	0.03	NA	NA	0.03	0.03	NA	NA	0.02	0.02	NA	NA	0.02	0.02
2000	NA	NA	0.03	0.03	NA	NA	0.03	0.03	NA	NA	0.02	0.02	NA	NA	0.01	0.02
2001	NA	NA	0.03	0.03	NA	NA	0.02	0.03	NA	NA	0.02	0.02	NA	NA	0.01	0.02
2002	NA	NA	0.03	0.03	NA	NA	0.02	0.03	NA	NA	0.02	0.02	NA	NA	0.01	0.02
2003	NA	NA	0.03	0.03	NA	NA	0.02	0.02	NA	NA	0.02	0.02	NA	NA	0.01	0.01
2004	NA	NA	0.03	0.03	NA	NA	0.02	0.02	NA	NA	0.02	0.02	NA	NA	0.01	0.01
2005	NA	NA	0.03	0.03	NA	NA	0.02	0.02	NA	NA	0.02	0.02	NA	NA	0.01	0.01
2006	NA	NA	0.03	0.03	NA	NA	0.02	0.02	NA	NA	0.02	0.02	NA	NA	0.01	0.01
2007	NA	NA	0.03	0.03	NA	NA	0.02	0.02	NA	NA	0.02	0.02	NA	NA	0.01	0.01
2008	NA	NA	0.02	0.03	NA	NA	0.02	0.02	NA	NA	0.02	0.02	NA	NA	0.01	0.01
2009	NA	NA	0.02	0.03	NA	NA	0.02	0.02	NA	NA	0.02	0.02	NA	NA	0.01	0.01
2010	NA	NA	0.02	0.03	NA	NA	0.02	0.02	NA	NA	0.01	0.02	NA	NA	0.01	0.01
2011	NA	NA	0.02	0.02	NA	NA	0.02	0.02	NA	NA	0.01	0.02	NA	NA	0.01	0.01
2012	NA	NA	0.02	0.02	NA	NA	0.02	0.02	NA	NA	0.01	0.01	NA	NA	0.01	0.01
2013	NA	NA	0.02	0.02	NA	NA	0.02	0.02	NA	NA	0.01	0.01	NA	NA	0.01	0.01
2014	NA	NA	0.02	0.02	NA	NA	0.02	0.02	NA	NA	0.01	0.01	NA	NA	0.01	0.01
2015	NA	NA	0.02	0.02	NA	NA	0.02	0.02	NA	NA	0.01	0.01	NA	NA	0.01	0.01
2016	NA	NA	0.02	0.02	NA	NA	0.02	0.02	NA	NA	0.01	0.01	NA	NA	0.01	0.01
2017	NA	NA	0.02	0.02	NA	NA	0.02	0.02	NA	NA	0.01	0.01	NA	NA	0.01	0.01
2018	NA	NA	0.02	0.02	NA	NA	0.02	0.02	NA	NA	0.01	0.01	NA	NA	0.01	0.01

TABLE 5-28: RATIO OF MODELED EGG CONCENTRATIONS BASED ON FISHRAND FOR FEMALE TREE SWALLOW USING TEQ FOR THE PERIOD 1993 - 2018

Year	LOAEL 152 Average	LOAEL 152 95% UCL	NOAEL 152 Average	NOAEL 152 95% UCL	LOAEL 113 Average	LOAEL 113 95% UCL	NOAEL 113 Average	NOAEL 113 95% UCL	LOAEL 90 Average	LOAEL 90 95% UCL	NOAEL 90 Average	NOAEL 90 95% UCL	LOAEL 50 Average	LOAEL 50 95% UCL	NOAEL 50 Average	NOAEL 50 95% UCL
1993	NA	NA	0.1	0.1	NA	NA	0.1	0.1	NA	NA	0.08	0.09	NA	NA	0.06	0.07
1994	NA	NA	0.1	0.1	NA	NA	0.1	0.1	NA	NA	0.08	0.09	NA	NA	0.06	0.06
1995	NA	NA	0.1	0.1	NA	NA	0.09	0.1	NA	NA	0.08	0.08	NA	NA	0.06	0.06
1996	NA	NA	0.1	0.1	NA	NA	0.09	0.1	NA	NA	0.07	0.08	NA	NA	0.05	0.06
1997	NA	NA	0.1	0.1	NA	NA	0.09	0.09	NA	NA	0.07	0.07	NA	NA	0.05	0.06
1998	NA	NA	0.1	0.1	NA	NA	0.08	0.09	NA	NA	0.07	0.07	NA	NA	0.05	0.05
1999	NA	NA	0.1	0.1	NA	NA	0.08	0.09	NA	NA	0.06	0.07	NA	NA	0.05	0.05
2000	NA	NA	0.1	0.1	NA	NA	0.08	0.08	NA	NA	0.06	0.07	NA	NA	0.05	0.05
2001	NA	NA	0.09	0.1	NA	NA	0.08	0.08	NA	NA	0.06	0.06	NA	NA	0.04	0.05
2002	NA	NA	0.09	0.1	NA	NA	0.07	0.08	NA	NA	0.06	0.06	NA	NA	0.04	0.05
2003	NA	NA	0.09	0.09	NA	NA	0.07	0.08	NA	NA	0.06	0.06	NA	NA	0.04	0.05
2004	NA	NA	0.08	0.09	NA	NA	0.07	0.07	NA	NA	0.05	0.06	NA	NA	0.04	0.04
2005	NA	NA	0.08	0.09	NA	NA	0.07	0.07	NA	NA	0.05	0.06	NA	NA	0.04	0.04
2006	NA	NA	0.08	0.08	NA	NA	0.07	0.07	NA	NA	0.05	0.05	NA	NA	0.04	0.04
2007	NA	NA	0.08	0.08	NA	NA	0.06	0.07	NA	NA	0.05	0.05	NA	NA	0.04	0.04
2008	NA	NA	0.07	0.08	NA	NA	0.06	0.07	NA	NA	0.05	0.05	NA	NA	0.03	0.04
2009	NA	NA	0.07	0.08	NA	NA	0.06	0.06	NA	NA	0.05	0.05	NA	NA	0.03	0.04
2010	NA	NA	0.07	0.08	NA	NA	0.06	0.06	NA	NA	0.05	0.05	NA	NA	0.03	0.04
2011	NA	NA	0.07	0.07	NA	NA	0.06	0.06	NA	NA	0.04	0.05	NA	NA	0.03	0.04
2012	NA	NA	0.07	0.07	NA	NA	0.06	0.06	NA	NA	0.04	0.05	NA	NA	0.03	0.03
2013	NA	NA	0.07	0.07	NA	NA	0.06	0.06	NA	NA	0.04	0.04	NA	NA	0.03	0.03
2014	NA	NA	0.06	0.07	NA	NA	0.05	0.06	NA	NA	0.04	0.04	NA	NA	0.03	0.03
2015	NA	NA	0.06	0.07	NA	NA	0.05	0.06	NA	NA	0.04	0.04	NA	NA	0.03	0.03
2016	NA	NA	0.06	0.07	NA	NA	0.05	0.05	NA	NA	0.04	0.04	NA	NA	0.03	0.03
2017	NA	NA	0.06	0.07	NA	NA	0.05	0.05	NA	NA	0.04	0.04	NA	NA	0.03	0.03
2018	NA	NA	0.06	0.07	NA	NA	0.05	0.05	NA	NA	0.04	0.04	NA	NA	0.03	0.03

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TABLE 5-29: RATIO OF MODELED DIETARY DOSE FOR FEMALE MALLARD BASED ON FISHRAND RESULTS FOR THE TRI+ CONGENERS

Year	LOAEL 152 Average	LOAEL 152 95% UCL	NOAEL 152 Average	NOAEL 152 95% UCL	LOAEL 113 Average	LOAEL 113 95% UCL	NOAEL 113 Average	NOAEL 113 95% UCL	LOAEL 90 Average	LOAEL 90 95% UCL	NOAEL 90 Average	NOAEL 90 95% UCL	LOAEL 50 Average	LOAEL 50 95% UCL	NOAEL 50 Average	NOAEL 50 95% UCL
1993	0.2	0.2	**2.2**	**2.3**	0.2	0.2	**1.7**	**1.9**	0.1	0.2	**1.4**	**1.5**	0.1	0.1	**1.3**	**1.4**
1994	0.2	0.2	**1.9**	**2.1**	0.2	0.2	**1.6**	**1.7**	0.1	0.1	**1.3**	**1.4**	0.1	0.1	**1.1**	**1.2**
1995	0.2	0.2	**1.6**	**1.7**	0.1	0.1	**1.3**	**1.4**	0.1	0.1	**1.1**	**1.2**	0.1	0.1	**1.0**	**1.1**
1996	0.2	0.2	**2.0**	**2.1**	0.1	0.1	**1.4**	**1.5**	0.1	0.1	**1.1**	**1.2**	0.09	0.10	0.9	**1.0**
1997	0.2	0.2	**1.7**	**1.8**	0.1	0.1	**1.3**	**1.4**	0.1	0.1	**1.0**	**1.1**	0.09	0.09	0.9	0.9
1998	0.1	0.1	**1.4**	**1.5**	0.1	0.1	**1.1**	**1.2**	0.09	0.1	0.9	**1.0**	0.08	0.09	0.8	0.9
1999	0.1	0.1	**1.2**	**1.3**	0.1	0.1	**1.0**	**1.1**	0.08	0.09	0.8	0.9	0.07	0.08	0.7	0.8
2000	0.1	0.1	**1.3**	**1.4**	0.1	0.1	**1.0**	**1.0**	0.08	0.08	0.8	0.8	0.07	0.07	0.7	0.7
2001	0.1	0.1	**1.4**	**1.5**	0.1	0.1	**1.0**	**1.1**	0.08	0.08	0.8	0.8	0.07	0.07	0.7	0.7
2002	0.1	0.1	**1.2**	**1.3**	0.09	0.1	0.9	**1.0**	0.07	0.08	0.7	0.8	0.06	0.07	0.6	0.7
2003	0.1	0.1	**1.1**	**1.1**	0.09	0.09	0.9	0.9	0.07	0.08	0.7	0.8	0.06	0.06	0.6	0.6
2004	0.09	0.1	0.9	**1.0**	0.08	0.08	0.8	0.8	0.06	0.07	0.6	0.7	0.06	0.06	0.6	0.6
2005	0.09	0.1	0.9	**1.0**	0.07	0.08	0.7	0.8	0.06	0.06	0.6	0.6	0.05	0.06	0.5	0.6
2006	0.09	0.1	0.9	**1.0**	0.07	0.08	0.7	0.8	0.06	0.06	0.6	0.6	0.05	0.05	0.5	0.5
2007	0.09	0.09	0.9	0.9	0.07	0.08	0.7	0.8	0.06	0.06	0.6	0.6	0.05	0.05	0.5	0.5
2008	0.08	0.09	0.8	0.9	0.07	0.07	0.7	0.7	0.05	0.06	0.5	0.6	0.05	0.05	0.5	0.5
2009	0.07	0.08	0.7	0.8	0.06	0.07	0.6	0.7	0.05	0.05	0.5	0.5	0.04	0.05	0.4	0.5
2010	0.08	0.09	0.8	0.9	0.06	0.07	0.6	0.7	0.05	0.05	0.5	0.5	0.04	0.04	0.4	0.4
2011	0.08	0.08	0.8	0.8	0.06	0.07	0.6	0.7	0.05	0.05	0.5	0.5	0.04	0.04	0.4	0.4
2012	0.08	0.08	0.8	0.8	0.06	0.07	0.6	0.7	0.05	0.05	0.5	0.5	0.04	0.04	0.4	0.4
2013	0.08	0.09	0.8	0.9	0.06	0.07	0.6	0.7	0.05	0.1	0.5	0.5	0.04	0.04	0.4	0.4
2014	0.07	0.08	0.7	0.8	0.06	0.07	0.6	0.7	0.05	0.0	0.5	0.5	0.04	0.04	0.4	0.4
2015	0.07	0.08	0.7	0.8	0.06	0.06	0.6	0.6	0.04	0.0	0.4	0.5	0.04	0.04	0.4	0.4
2016	0.06	0.07	0.6	0.7	0.05	0.06	0.5	0.6	0.04	0.0	0.4	0.5	0.04	0.04	0.4	0.4
2017	0.06	0.07	0.6	0.7	0.05	0.05	0.5	0.5	0.04	0.0	0.4	0.4	0.04	0.04	0.4	0.4
2018	0.07	0.07	0.7	0.7	0.05	0.06	0.5	0.6	0.04	0.0	0.4	0.4	0.03	0.04	0.3	0.4

Bold values indicate exceedances

302815

TABLE 5-30: RATIO OF EGG CONCENTRATIONS FOR FEMALE MALLARD BASED ON FISHRAND RESULTS FOR THE TRI+ CONGENERS

Year	LOAEL 152 Average	LOAEL 152 95% UCL	NOAEL 152 Average	NOAEL 152 95% UCL	LOAEL 113 Average	LOAEL 113 95% UCL	NOAEL 113 Average	NOAEL 113 95% UCL	LOAEL 90 Average	LOAEL 90 95% UCL	NOAEL 90 Average	NOAEL 90 95% UCL	LOAEL 50 Average	LOAEL 50 95% UCL	NOAEL 50 Average	NOAEL 50 95% UCL
1993	**2.4**	**2.6**	**15.9**	**17.1**	**1.9**	**2.0**	**12.7**	**13.6**	**1.5**	**1.6**	**10.3**	**11.0**	**1.1**	**1.2**	**7.6**	**8.1**
1994	**2.1**	**2.3**	**14.3**	**15.3**	**1.8**	**1.9**	**11.9**	**12.7**	**1.5**	**1.6**	**9.8**	**10.5**	**1.1**	**1.1**	**7.1**	**7.6**
1995	**2.1**	**2.2**	**13.8**	**14.8**	**1.7**	**1.8**	**11.4**	**12.2**	**1.4**	**1.5**	**9.1**	**9.8**	**1.0**	**1.1**	**6.7**	**7.2**
1996	**2.0**	**2.2**	**13.7**	**14.6**	**1.6**	**1.7**	**10.9**	**11.7**	**1.3**	**1.4**	**8.7**	**9.3**	**1.0**	**1.0**	**6.5**	**6.9**
1997	**1.9**	**2.1**	**12.9**	**13.9**	**1.6**	**1.7**	**10.5**	**11.2**	**1.3**	**1.3**	**8.4**	**9.0**	0.9	**1.0**	**6.3**	**6.7**
1998	**1.8**	**2.0**	**12.4**	**13.3**	**1.5**	**1.6**	**10.2**	**10.9**	**1.2**	**1.3**	**8.0**	**8.6**	0.9	0.9	**5.9**	**6.4**
1999	**1.8**	**1.9**	**11.7**	**12.6**	**1.5**	**1.6**	**9.9**	**10.6**	**1.2**	**1.2**	**7.7**	**8.3**	0.9	0.9	**5.8**	**6.2**
2000	**1.8**	**1.9**	**11.8**	**12.7**	**1.4**	**1.5**	**9.5**	**10.2**	**1.1**	**1.2**	**7.5**	**8.1**	0.8	0.9	**5.6**	**6.0**
2001	**1.7**	**1.8**	**11.5**	**12.4**	**1.4**	**1.5**	**9.3**	**10.0**	**1.1**	**1.2**	**7.3**	**7.8**	0.8	0.9	**5.4**	**5.8**
2002	**1.6**	**1.8**	**11.0**	**11.8**	**1.3**	**1.4**	**9.0**	**9.7**	**1.1**	**1.1**	**7.1**	**7.6**	0.8	0.9	**5.3**	**5.7**
2003	**1.5**	**1.7**	**10.4**	**11.1**	**1.3**	**1.4**	**8.6**	**9.2**	**1.0**	**1.1**	**7.0**	**7.5**	0.8	0.8	**5.1**	**5.5**
2004	**1.5**	**1.6**	**10.2**	**11.0**	**1.2**	**1.3**	**8.3**	**8.9**	**1.0**	**1.1**	**6.6**	**7.1**	0.7	0.8	**4.9**	**5.3**
2005	**1.5**	**1.6**	**9.9**	**10.7**	**1.2**	**1.3**	**8.2**	**8.8**	**1.0**	**1.0**	**6.4**	**6.8**	0.7	0.8	**4.7**	**5.1**
2006	**1.4**	**1.5**	**9.5**	**10.2**	**1.2**	**1.3**	**8.0**	**8.6**	0.9	**1.0**	**6.1**	**6.5**	0.7	0.7	**4.5**	**4.8**
2007	**1.4**	**1.5**	**9.4**	**10.1**	**1.2**	**1.3**	**7.8**	**8.4**	0.9	**1.0**	**5.9**	**6.4**	0.7	0.7	**4.4**	**4.7**
2008	**1.4**	**1.5**	**9.1**	**9.8**	**1.1**	**1.2**	**7.5**	**8.1**	0.9	0.9	**5.8**	**6.2**	0.6	0.7	**4.3**	**4.6**
2009	**1.3**	**1.4**	**8.9**	**9.6**	**1.1**	**1.2**	**7.3**	**7.9**	0.8	0.9	**5.6**	**6.0**	0.6	0.7	**4.2**	**4.5**
2010	**1.3**	**1.4**	**8.7**	**9.4**	**1.1**	**1.1**	**7.1**	**7.7**	0.8	0.9	**5.5**	**5.9**	0.6	0.7	**4.1**	**4.4**
2011	**1.3**	**1.3**	**8.4**	**9.0**	**1.1**	**1.1**	**7.1**	**7.6**	0.8	0.9	**5.3**	**5.7**	0.6	0.6	**4.0**	**4.3**
2012	**1.2**	**1.3**	**8.2**	**8.8**	**1.0**	**1.1**	**6.9**	**7.5**	0.8	0.8	**5.2**	**5.6**	0.6	0.6	**3.9**	**4.2**
2013	**1.2**	**1.3**	**8.0**	**8.6**	**1.0**	**1.1**	**6.8**	**7.3**	0.8	0.8	**5.1**	**5.4**	0.6	0.6	**3.8**	**4.1**
2014	**1.2**	**1.3**	**7.9**	**8.5**	**1.0**	**1.1**	**6.6**	**7.1**	0.7	0.8	**4.9**	**5.3**	0.6	0.6	**3.7**	**4.0**
2015	**1.1**	**1.2**	**7.7**	**8.3**	**1.0**	**1.0**	**6.4**	**6.9**	0.7	0.8	**4.8**	**5.2**	0.5	0.6	**3.6**	**3.9**
2016	**1.2**	**1.3**	**7.8**	**8.4**	0.9	**1.0**	**6.2**	**6.7**	0.7	0.8	**4.7**	**5.1**	0.5	0.6	**3.6**	**3.8**
2017	**1.1**	**1.2**	**7.7**	**8.3**	0.9	**1.0**	**6.1**	**6.6**	0.7	0.8	**4.7**	**5.0**	0.5	0.6	**3.5**	**3.7**
2018	**1.1**	**1.2**	**7.5**	**8.1**	0.9	**1.0**	**6.1**	**6.6**	0.7	0.7	**4.6**	**4.9**	0.5	0.5	**3.4**	**3.7**

Bold values indicate exceedances

TABLE 5-31: RATIO OF MODELED DIETARY DOSE TO BENCHMARKS FOR FEMALE MALLARD FOR PERIOD 1993 - 2018 ON A TEQ BASIS

Year	LOAEL 152 Average	LOAEL 152 95% UCL	NOAEL 152 Average	NOAEL 152 95% UCL	LOAEL 113 Average	LOAEL 113 95% UCL	NOAEL 113 Average	NOAEL 113 95% UCL	LOAEL 90 Average	LOAEL 90 95% UCL	NOAEL 90 Average	NOAEL 90 95%UCL	LOAEL 50 Average	LOAEL 50 95% UCL	NOAEL 50 Average	NOAEL 50 95% UCL
1993	**16**	**17.4**	**162**	**174**	**13**	**13.9**	**130**	**139**	**11**	**11**	**107**	**112**	**16**	**11**	**159**	**110**
1994	**14**	**15.0**	**140**	**150**	**12**	**12.4**	**116**	**124**	**9.4**	**10**	**94**	**101**	**14**	**9.7**	**138**	**97**
1995	**11**	**11.8**	**110**	**118**	**9**	**9.7**	**91**	**97**	**8.5**	**8.5**	**85**	**85**	**11**	**8.6**	**105**	**86**
1996	**15**	**16.3**	**152**	**163**	**10**	**10.5**	**98**	**105**	**7.7**	**8.1**	**77**	**81**	**15**	**7.7**	**151**	**77**
1997	**12**	**13.0**	**121**	**130**	**8.9**	**9.6**	**89**	**96**	**6.9**	**7.5**	**69**	**75**	**12**	**7.0**	**118**	**70**
1998	**8.9**	**9.6**	**89**	**96**	**7.1**	**7.6**	**71**	**76**	**6.2**	**6.5**	**62**	**65**	**8.4**	**6.3**	**84**	**63**
1999	**7.5**	**8.0**	**75**	**80**	**6.0**	**6.5**	**60**	**65**	**5.7**	**5.7**	**57**	**57**	**7.0**	**5.8**	**70**	**58**
2000	**8.5**	**9.1**	**85**	**91**	**6.0**	**6.4**	**60**	**64**	**5.1**	**5.3**	**51**	**53**	**8.0**	**5.2**	**80**	**52**
2001	**9.4**	**10.1**	**94**	**101**	**6.3**	**6.7**	**63**	**67**	**4.8**	**5.1**	**48**	**51**	**9.0**	**4.8**	**90**	**48**
2002	**7.7**	**8.2**	**77**	**82**	**5.7**	**6.2**	**57**	**62**	**4.6**	**4.9**	**46**	**49**	**7.2**	**4.6**	**72**	**46**
2003	**6.5**	**7.0**	**65**	**70**	**5.3**	**5.7**	**53**	**57**	**4.3**	**4.6**	**43**	**46**	**6.1**	**4.3**	**61**	**43**
2004	**5.3**	**5.7**	**53**	**57**	**4.3**	**4.6**	**43**	**46**	**4.0**	**4.0**	**40**	**40**	**4.7**	**4.0**	**47**	**40**
2005	**5.2**	**5.6**	**52**	**56**	**4.1**	**4.4**	**41**	**44**	**3.7**	**3.7**	**37**	**37**	**4.7**	**3.7**	**47**	**37**
2006	**5.5**	**6.0**	**55**	**60**	**4.2**	**4.5**	**42**	**45**	**3.4**	**3.6**	**34**	**36**	**5.1**	**3.4**	**51**	**34**
2007	**4.9**	**5.3**	**49**	**53**	**4.0**	**4.3**	**40**	**43**	**3.3**	**3.5**	**33**	**35**	**4.4**	**3.2**	**44**	**32**
2008	**4.6**	**4.9**	**46**	**49**	**3.6**	**3.9**	**36**	**39**	**3.1**	**3.2**	**31**	**32**	**4.1**	**3.1**	**41**	**31**
2009	**3.7**	**4.0**	**37**	**40**	**3.2**	**3.5**	**32**	**35**	**2.9**	**3.0**	**29**	**30**	**3.1**	**2.9**	**31**	**29**
2010	**4.7**	**5.1**	**47**	**51**	**3.5**	**3.8**	**35**	**38**	**2.8**	**3.0**	**28**	**30**	**4.3**	**2.8**	**43**	**28**
2011	**4.2**	**4.5**	**42**	**45**	**3.5**	**3.8**	**35**	**38**	**2.7**	**2.9**	**27**	**29**	**3.7**	**2.7**	**37**	**27**
2012	**4.5**	**4.8**	**45**	**48**	**3.6**	**3.8**	**36**	**38**	**2.7**	**3.0**	**27**	**30**	**4.0**	**2.7**	**40**	**27**
2013	**5.3**	**5.7**	**53**	**57**	**3.7**	**4.0**	**37**	**40**	**2.6**	**2.9**	**26**	**29**	**4.9**	**2.6**	**49**	**26**
2014	**4.4**	**4.7**	**44**	**47**	**3.4**	**3.7**	**34**	**37**	**2.6**	**2.8**	**26**	**28**	**3.9**	**2.6**	**39**	**26**
2015	**4.2**	**4.5**	**42**	**45**	**3.3**	**3.5**	**33**	**35**	**2.5**	**2.7**	**25**	**27**	**3.8**	**2.5**	**38**	**25**
2016	**3.3**	**3.6**	**33**	**36**	**2.7**	**2.9**	**27**	**29**	**2.4**	**2.5**	**24**	**25**	**2.8**	**2.4**	**28**	**24**
2017	**3.2**	**3.4**	**32**	**34**	**2.5**	**2.7**	**25**	**27**	**2.3**	**2.3**	**23**	**23**	**2.7**	**2.3**	**27**	**23**
2018	**3.5**	**3.8**	**35**	**38**	**2.6**	**2.8**	**26**	**28**	**2.2**	**2.3**	**22**	**23**	**3.1**	**2.2**	**31**	**22**

Bold values indicate exceedances

TABLE 5-32: RATIO OF MODELED EGG CONCENTRATION TO BENCHMARKS FOR FEMALE MALLARD FOR PERIOD 1993 - 2018 ON A TEQ BASIS

Year	LOAEL 152 Average	LOAEL 152 95% UCL	NOAEL 152 Average	NOAEL 152 95% UCL	LOAEL 113 Average	LOAEL 113 95% UCL	NOAEL 113 Average	NOAEL 113 95% UCL	LOAEL 90 Average	LOAEL 90 95% UCL	NOAEL 90 Average	NOAEL 90 95%UCL	LOAEL 50 Average	LOAEL 50 95% UCL	NOAEL 50 Average	NOAEL 50 95% UCL
1993	340	366	1362	1463	270	290	1081	1160	219	236	878	943	161	173	645	693
1994	305	327	1221	1308	253	271	1012	1085	208	223	833	894	151	162	605	649
1995	295	317	1181	1266	243	260	971	1041	195	209	780	837	144	154	575	616
1996	291	312	1166	1249	233	250	933	1000	186	199	743	796	138	148	553	593
1997	276	296	1104	1183	224	240	895	959	180	193	720	772	134	143	535	573
1998	264	283	1057	1133	217	233	870	932	171	184	686	735	126	136	506	542
1999	250	269	1002	1076	211	226	843	905	165	177	661	708	123	132	492	526
2000	252	270	1007	1081	202	217	808	868	161	172	643	688	119	128	477	510
2001	246	264	985	1055	199	214	797	856	156	167	624	668	115	124	462	495
2002	235	253	941	1011	192	207	769	826	152	163	609	652	113	122	454	487
2003	221	238	885	951	183	197	734	788	149	160	595	639	109	118	438	470
2004	218	234	871	937	177	190	708	761	141	151	564	606	105	112	418	449
2005	212	228	847	911	175	189	701	754	136	146	543	583	101	108	402	432
2006	203	219	814	875	170	183	681	732	130	140	520	558	96	103	385	413
2007	201	216	803	864	167	179	667	717	126	136	506	544	93	101	374	402
2008	194	209	775	836	160	173	642	691	123	132	491	528	91	98	364	391
2009	190	205	759	819	156	168	622	670	120	129	480	516	89	96	356	383
2010	187	201	747	803	153	164	610	657	118	127	472	507	87	94	350	376
2011	179	192	715	769	151	163	605	651	114	122	456	490	86	92	344	370
2012	174	187	697	750	148	159	591	636	111	120	445	478	84	90	336	362
2013	171	183	682	733	145	156	578	622	108	116	432	464	81	88	326	351
2014	169	181	675	726	141	152	564	607	105	113	421	453	79	86	318	342
2015	164	177	655	707	136	146	543	585	103	111	413	444	78	83	310	334
2016	165	179	662	716	132	142	528	570	101	109	404	434	76	82	304	327
2017	163	177	654	708	131	141	524	566	100	107	399	429	74	80	296	319
2018	160	173	638	691	130	141	522	565	98	105	392	422	72	78	290	312

Bold values indicate exceedances

302818

TABLE 5-33: RATIO OF MODELED DIETARY DOSE TO BENCHMARKS BASED ON FISHRAND FOR FEMALE KINGFISHER BASED ON THE SUM OF TRI+ CONGENERS FOR THE PERIOD 1993 - 2018

Year	LOAEL 152 Average	LOAEL 152 95% UCL	NOAEL 152 Average	NOAEL 152 95% UCL	LOAEL 113 Average	LOAEL 113 95% UCL	NOAEL 113 Average	NOAEL 113 95% UCL	LOAEL 90 Average	LOAEL 90 95% UCL	NOAEL 90 Average	NOAEL 90 95% UCL	LOAEL 50 Average	LOAEL 50 95% UCL	NOAEL 50 Average	NOAEL 50 95% UCL
1993	10	10	67	69	6.7	7.0	47	49	5.4	5.6	38	39	4.8	5.0	33	35
1994	7.5	7.7	52	54	6.1	6.4	43	45	4.9	5.1	34	36	4.3	4.5	30	31
1995	6.8	7.1	47	49	5.2	5.5	37	38	4.5	4.6	31	32	3.9	4.1	27	28
1996	7.8	8.1	54	57	5.3	5.5	37	39	4.2	4.3	30	30	3.6	3.8	25	26
1997	6.8	7.0	47	49	5.0	5.2	35	36	3.9	4.1	27	28	3.4	3.5	23	24
1998	5.4	5.6	38	39	4.3	4.5	30	32	3.6	3.8	25	26	3.1	3.3	22	23
1999	4.9	5.1	34	36	3.9	4.1	28	29	3.2	3.4	23	24	2.9	3.0	20	21
2000	4.8	5.0	34	35	3.7	3.9	26	27	3.0	3.2	21	22	2.7	2.8	19	19
2001	5.1	5.3	36	37	3.8	3.9	26	28	2.9	3.0	20	21	2.5	2.6	17	18
2002	4.6	4.9	32	34	3.7	3.8	26	27	2.8	3.0	20	21	2.4	2.5	17	18
2003	4.3	4.5	30	31	3.4	3.5	24	25	2.7	2.8	19	20	2.3	2.4	16	17
2004	3.6	3.8	25	27	3.0	3.2	21	22	2.5	2.6	18	18	2.1	2.2	15	16
2005	3.6	3.7	25	26	2.9	3.0	20	21	2.3	2.5	16	17	2.0	2.1	14	15
2006	3.9	4.1	28	29	2.9	3.0	20	21	2.2	2.3	16	16	1.9	2.0	13	14
2007	3.4	3.6	24	25	2.8	2.9	20	21	2.2	2.3	15	16	1.8	1.9	13	13
2008	3.2	3.4	23	24	2.7	2.8	19	20	2.1	2.2	15	15	1.7	1.8	12	13
2009	3.0	3.2	21	22	2.5	2.6	17	18	2.0	2.1	14	15	1.7	1.8	12	12
2010	3.2	3.4	22	23	2.4	2.6	17	18	1.9	2.0	13	14	1.6	1.7	11	12
2011	3.3	3.5	23	24	2.5	2.6	18	19	1.9	2.0	13	14	1.6	1.6	11	12
2012	3.1	3.3	22	23	2.5	2.6	17	18	1.9	2.0	13	14	1.5	1.6	11	11
2013	3.3	3.4	23	24	2.5	2.6	17	18	1.8	1.9	13	14	1.5	1.6	11	11
2014	3.1	3.3	22	23	2.4	2.5	17	18	1.8	1.9	12	13	1.5	1.5	10	11
2015	2.8	3.0	20	21	2.3	2.4	16	17	1.7	1.8	12	13	1.4	1.5	10	11
2016	2.5	2.7	18	19	2.1	2.2	15	16	1.7	1.8	12	12	1.4	1.5	10	10
2017	2.5	2.6	17	18	2.0	2.1	14	15	1.6	1.7	11	12	1.3	1.4	9.4	10
2018	2.5	2.6	17	18	2.0	2.1	14	15	1.5	1.6	11	11	1.3	1.4	9.1	10

Bold values indicate exceedances

302819

TABLE 5-34: RATIO OF MODELED DIETARY DOSE TO BENCHMARKS BASED ON FISHRAND FOR FEMALE BLUE HERON BASED ON THE SUM OF TRI+ CONGENERS FOR THE PERIOD 1993 - 2018

Year	LOAEL 152 Average	LOAEL 152 95% UCL	NOAEL 152 Average	NOAEL 152 95% UCL	LOAEL 113 Average	LOAEL 113 95% UCL	NOAEL 113 Average	NOAEL 113 95% UCL	LOAEL 90 Average	LOAEL 90 95% UCL	NOAEL 90 Average	NOAEL 90 95% UCL	LOAEL 50 Average	LOAEL 50 95% UCL	NOAEL 50 Average	NOAEL 50 95% UCL
1993	**3.7**	**3.8**	**26**	**27**	**2.5**	**2.6**	**18**	**18**	**2.0**	**2.1**	**14**	**14**	**1.9**	**2.0**	**13**	**14**
1994	**2.8**	**2.9**	**19**	**20**	**2.3**	**2.3**	**16**	**16**	**1.8**	**1.9**	**13**	**13**	**1.7**	**1.7**	**12**	**12**
1995	**2.5**	**2.5**	**17**	**18**	**1.8**	**1.9**	**13**	**13**	**2**	**1.6**	**11**	**12**	**1.5**	**1.6**	**11**	**11**
1996	**3.0**	**3.1**	**21**	**22**	**1.9**	**2.0**	**13**	**14**	**1.5**	**1.5**	**11**	**11**	**1.4**	**1.4**	**9.7**	**10**
1997	**2.5**	**2.6**	**18**	**18**	**1.8**	**1.9**	**13**	**13**	**1.4**	**1.4**	**10**	**10**	**1.3**	**1.3**	**8.8**	**9.1**
1998	**1.9**	**1.9**	**13**	**13**	**1.5**	**1.5**	**10**	**11**	**1.3**	**1.3**	**8.9**	**9.2**	**1.2**	**1.2**	**8.2**	**8.5**
1999	**1.6**	**1.7**	**11**	**12**	**1.3**	**1.4**	**9.1**	**9.5**	**1.1**	**1.1**	**7.7**	**8.0**	**1.0**	**1.1**	**7.3**	**7.6**
2000	**1.6**	**1.7**	**11**	**12**	**1.2**	**1.2**	**8.4**	**8.7**	**1.0**	**1.0**	**7.1**	**7.3**	**1.0**	**1.0**	**6.7**	**6.9**
2001	**1.8**	**1.8**	**12**	**13**	**1.2**	**1.3**	**8.7**	**9.1**	**1.0**	**1.0**	**6.7**	**7.0**	0.9	0.9	**6.2**	**6.4**
2002	**1.6**	**1.6**	**11**	**11**	**1.2**	**1.3**	**8.5**	**8.9**	0.9	**1.0**	**6.5**	**6.8**	0.8	0.9	**5.9**	**6.1**
2003	**1.4**	**1.5**	**10**	**11**	**1.1**	**1.2**	**7.7**	**8.1**	0.9	0.9	**6.1**	**6.4**	0.8	0.8	**5.6**	**5.8**
2004	**1.1**	**1.2**	**7.8**	**8.2**	0.9	**1.0**	**6.6**	**6.9**	0.8	0.8	**5.6**	**5.9**	0.7	0.8	**5.2**	**5.4**
2005	**1.1**	**1.1**	**7.7**	**8.0**	0.9	0.9	**6.2**	**6.4**	0.7	0.8	**5.2**	**5.4**	0.7	0.7	**4.8**	**5.0**
2006	**1.3**	**1.4**	**9.2**	**10**	0.9	0.9	**6.3**	**6.6**	0.7	0.7	**4.9**	**5.1**	0.6	0.7	**4.5**	**4.6**
2007	**1.1**	**1.1**	**7.5**	**7.8**	0.9	0.9	**6.1**	**6.3**	0.7	0.7	**4.7**	**4.9**	0.6	0.6	**4.3**	**4.4**
2008	**1.0**	**1.0**	**6.9**	**7.3**	0.8	0.9	**5.7**	**6.0**	0.6	0.7	**4.5**	**4.7**	0.6	0.6	**4.1**	**4.2**
2009	0.9	**0.9**	**6.3**	**6.6**	0.7	0.8	**5.2**	**5.5**	0.6	0.6	**4.2**	**4.4**	0.6	0.6	**3.9**	**4.0**
2010	**1.0**	**1.0**	**7.0**	**7.3**	0.7	0.8	**5.1**	**5.4**	0.6	0.6	**4.0**	**4.2**	0.5	0.5	**3.7**	**3.8**
2011	**1.1**	**1.1**	**7.6**	**7.9**	0.8	0.8	**5.4**	**5.7**	0.6	0.6	**4.0**	**4.2**	0.5	0.5	**3.6**	**3.7**
2012	**1.0**	**1.1**	**7.1**	**7.4**	0.8	0.8	**5.3**	**5.6**	0.6	0.6	**4.0**	**4.2**	0.5	0.5	**3.5**	**3.7**
2013	**1.1**	**1.1**	**7.6**	**7.9**	0.8	0.8	**5.4**	**5.7**	0.6	0.6	**4.0**	**4.2**	0.5	0.5	**3.5**	**3.6**
2014	**1.0**	**1.1**	**7.1**	**7.4**	0.7	0.8	**5.2**	**5.4**	0.6	0.6	**3.9**	**4.1**	0.5	0.5	**3.4**	**3.5**
2015	0.9	0.9	**6.2**	**6.5**	0.7	0.7	**4.8**	**5.1**	0.5	0.6	**3.8**	**3.9**	0.5	0.5	**3.3**	**3.4**
2016	0.7	0.8	**5.2**	**5.5**	0.6	0.7	**4.4**	**4.6**	0.5	0.5	**3.6**	**3.7**	0.5	0.5	**3.2**	**3.3**
2017	0.7	0.7	**4.9**	**5.2**	0.6	0.6	**4.0**	**4.3**	0.5	0.5	**3.3**	**3.5**	0.4	0.5	**3.0**	**3.2**
2018	0.7	0.8	**5.0**	**5.3**	0.6	0.6	**3.9**	**4.1**	0.5	0.5	**3.2**	**3.4**	0.4	0.4	**2.9**	**3.1**

Bold values indicate exceedances

302820

TABLE 5-35: RATIO OF MODELED DIETARY DOSE TO BENCHMARKS BASED ON FISHRAND FOR FEMALE BALD EAGLE BASED ON THE SUM OF TRI+ CONGENERS FOR THE PERIOD 1993 - 2018

Year	LOAEL 152 Average	LOAEL 152 95% UCL	NOAEL 152 Average	NOAEL 152 95% UCL	LOAEL 113 Average	LOAEL 113 95% UCL	NOAEL 113 Average	NOAEL 113 95% UCL	LOAEL 90 Average	LOAEL 90 95% UCL	NOAEL 90 Average	NOAEL 90 95% UCL	LOAEL 50 Average	LOAEL 50 95% UCL	NOAEL 50 Average	NOAEL 50 95% UCL
1993	27	28	190	194	18	19	127	130	4.2	4.2	29	30	3.9	4.0	27	28
1994	20	20	138	140	16	16	111	113	3.8	3.8	26	27	3.5	3.6	25	25
1995	17	17	118	121	14	14	99	101	3.4	3.5	24	24	3.2	3.2	22	22
1996	20	20	140	143	13	13	92	94	3.1	3.2	22	22	2.9	2.9	20	20
1997	18	19	127	130	13	13	89	91	2.9	3.0	20	21	2.6	2.7	18	19
1998	14	15	100	102	12	12	81	83	2.7	2.7	19	19	2.4	2.5	17	17
1999	12	13	86	88	10	10	68	69	2.4	2.5	17	17	2.2	2.2	15	16
2000	12	12	81	83	8.8	9.0	62	63	2.2	2.2	15	16	2.0	2.1	14	14
2001	13	13	92	94	8.9	9.1	62	64	2.1	2.1	15	15	1.9	1.9	13	13
2002	12	12	84	86	8.8	9.0	62	63	2.0	2.1	14	14	1.8	1.8	13	13
2003	11	11	75	77	8.1	8.3	57	58	1.9	2.0	13	14	1.7	1.7	12	12
2004	8.7	8.9	61	62	7.2	7.3	50	51	1.8	1.8	12	13	1.6	1.6	11	11
2005	8.3	8.4	58	59	6.6	6.8	46	47	1.6	1.7	11	12	1.5	1.5	10	11
2006	9.3	10	65	67	6.5	6.7	46	47	1.6	1.6	11	11	1.4	1.4	9.8	9.9
2007	8.6	8.8	60	61	6.4	6.5	45	46	1.5	1.5	11	11	1.3	1.4	9.3	9.5
2008	8.0	8.1	56	57	6.1	6.3	43	44	1.4	1.5	10	10	1.3	1.3	8.9	9.1
2009	6.9	7.1	48	50	5.6	5.7	39	40	1.4	1.4	10	10	1.2	1.2	8.5	8.6
2010	7.3	7.4	51	52	5.4	5.5	38	38	1.3	1.3	9.1	9.3	1.2	1.2	8.1	8.2
2011	8.1	8.3	57	58	5.7	5.8	40	41	1.3	1.3	9.0	9.2	1.1	1.1	7.8	8.0
2012	7.3	7.4	51	52	5.6	5.7	39	40	1.3	1.3	8.9	9.1	1.1	1.1	7.6	7.8
2013	7.9	8.1	56	57	5.8	5.9	40	41	1.3	1.3	9.2	9.3	1.1	1.1	7.8	7.9
2014	7.2	7.4	51	52	5.5	5.6	38	39	1.2	1.3	8.7	8.8	1.1	1.1	7.4	7.5
2015	6.7	6.8	47	48	5.2	5.3	36	37	1.2	1.2	8.4	8.6	1.0	1.0	7.2	7.3
2016	6.1	6.2	43	44	4.8	5.0	34	35	1.1	1.2	8.0	8.2	1.0	1.0	7.0	7.1
2017	5.5	5.6	38	39	4.5	4.6	31	32	1.1	1.1	7.6	7.8	1.0	1.0	6.8	6.9
2018	5.4	5.5	37	38	4.2	4.3	30	30	1.0	1.1	7.2	7.4	0.9	0.9	6.4	6.5

Bold values indicate exceedances

302821

TABLE 5-36: RATIO OF MODELED EGG CONCENTRATIONS TO BENCHMARKS FOR FEMALE KINGFISHER BASED ON THE SUM OF TRI+ CONGENERS FOR THE PERIOD 1993 - 2018

Year	LOAEL 152 Average	LOAEL 152 95% UCL	NOAEL 152 Average	NOAEL 152 95% UCL	LOAEL 113 Average	LOAEL 113 95% UCL	NOAEL 113 Average	NOAEL 113 95% UCL	LOAEL 90 Average	LOAEL 90 95% UCL	NOAEL 90 Average	NOAEL 90 95% UCL	LOAEL 50 Average	LOAEL 50 95% UCL	NOAEL 50 Average	NOAEL 50 95% UCL
1993	23	24	153	159	16	17	107	111	13	13	86	90	11	12	77	80
1994	18	19	119	124	15	15	98	102	12	12	79	82	10	11	69	72
1995	16	17	109	113	12	13	84	87	11	11	71	73	9.3	10	62	65
1996	19	19	125	129	13	13	85	89	10	10	66	69	8.6	9.0	58	60
1997	16	17	108	112	12	12	80	83	9.3	10	62	65	8.0	8.3	54	56
1998	13	13	86	90	10	11	69	72	8.6	9.0	58	60	7.5	7.8	50	52
1999	12	12	78	81	9.4	10	63	66	7.8	8.1	52	54	6.8	7.1	46	48
2000	11	12	77	80	8.8	9.2	59	62	7.3	7.6	49	51	6.3	6.6	42	44
2001	12	13	82	85	9.0	9.4	60	63	6.9	7.2	46	48	6.0	6.2	40	42
2002	11	12	74	78	8.7	9.1	58	61	6.8	7.1	45	47	5.7	6.0	38	40
2003	10	11	69	72	8.1	8.4	54	57	6.5	6.7	43	45	5.5	5.7	37	38
2004	8.7	9.1	58	61	7.2	7.6	48	51	6.0	6.3	40	42	5.1	5.3	34	36
2005	8.5	8.9	57	59	6.9	7.2	46	48	5.6	5.9	37	39	4.8	5.0	32	34
2006	9.4	10	63	66	6.9	7.2	46	49	5.3	5.6	36	37	4.5	4.7	30	32
2007	8.2	8.6	55	57	6.7	7.0	45	47	5.1	5.4	34	36	4.3	4.5	29	30
2008	7.7	8.1	52	54	6.3	6.7	42	45	5.0	5.2	33	35	4.2	4.3	28	29
2009	7.2	7.6	48	51	5.9	6.2	40	42	4.7	5.0	32	33	4.0	4.2	27	28
2010	7.6	8.0	51	53	5.8	6.1	39	41	4.5	4.7	30	32	3.8	4.0	26	27
2011	7.9	8.3	53	55	6.0	6.3	40	42	4.5	4.7	30	31	3.7	3.9	25	26
2012	7.5	7.8	50	52	5.9	6.2	39	41	4.5	4.7	30	31	3.7	3.9	25	26
2013	7.8	8.1	52	55	5.9	6.2	40	41	4.4	4.6	29	31	3.6	3.8	24	25
2014	7.4	7.8	50	52	5.7	6.0	38	40	4.3	4.5	29	30	3.5	3.7	24	25
2015	6.7	7.1	45	47	5.4	5.6	36	38	4.1	4.3	28	29	3.4	3.6	23	24
2016	6.1	6.4	41	43	5.0	5.3	34	35	4.0	4.2	27	28	3.3	3.5	22	23
2017	5.9	6.2	39	42	4.8	5.0	32	34	3.8	4.0	25	27	3.2	3.3	21	22
2018	5.9	6.2	39	41	4.7	4.9	31	33	3.7	3.9	25	26	3.1	3.3	21	22

Bold values indicate exceedances

TABLE 5-37: RATIO OF MODELED EGG CONCENTRATIONS TO BENCHMARKS FOR FEMALE BLUE HERON BASED ON THE SUM OF TRI+ CONGENERS FOR THE PERIOD 1993 - 2018

Year	LOAEL 152 Average	LOAEL 152 95% UCL	NOAEL 152 Average	NOAEL 152 95% UCL	LOAEL 113 Average	LOAEL 113 95% UCL	NOAEL 113 Average	NOAEL 113 95% UCL	LOAEL 90 Average	LOAEL 90 95% UCL	NOAEL 90 Average	NOAEL 90 95% UCL	LOAEL 50 Average	LOAEL 50 95% UCL	NOAEL 50 Average	NOAEL 50 95% UCL
1993	23	23	151	155	15	16	101	104	12	12	81	84	11	12	77	79
1994	17	17	112	115	14	14	92	94	11	11	73	76	10	11	69	71
1995	15	15	100	103	11	12	75	77	10	10	64	66	9.1	9.4	61	63
1996	18	19	121	124	12	12	78	80	9.0	9.2	60	62	8.3	8.6	56	57
1997	15	16	102	105	11	11	72	75	8.3	8.6	56	57	7.6	7.9	51	53
1998	11	12	75	77	8.9	9.2	60	61	7.7	7.9	51	53	7.1	7.3	48	49
1999	10	10	66	68	7.9	8.1	53	54	6.6	6.8	44	46	6.3	6.5	42	43
2000	10	10	64	66	7.2	7.4	48	50	6.1	6.3	41	42	5.8	5.9	39	40
2001	11	11	72	74	7.5	7.8	51	52	5.8	5.9	39	40	5.3	5.5	36	37
2002	9.5	10	64	65	7.3	7.6	49	51	5.6	5.8	38	39	5.1	5.2	34	35
2003	8.7	9.0	58	60	6.7	6.9	45	46	5.3	5.4	35	36	4.8	4.9	32	33
2004	6.8	7.0	45	47	5.7	5.9	38	39	4.8	5.0	32	33	4.5	4.6	30	31
2005	6.6	6.8	44	46	5.3	5.5	35	37	4.4	4.6	30	31	4.1	4.3	28	28
2006	8.0	8.2	53	55	5.4	5.6	36	38	4.2	4.3	28	29	3.9	4.0	26	27
2007	6.5	6.7	43	45	5.2	5.4	35	36	4.0	4.2	27	28	3.7	3.8	25	25
2008	6.0	6.2	40	41	4.9	5.1	33	34	3.9	4.0	26	27	3.5	3.6	23	24
2009	5.4	5.6	36	37	4.5	4.6	30	31	3.6	3.8	24	25	3.3	3.4	22	23
2010	6.0	6.2	40	41	4.4	4.5	29	30	3.4	3.5	23	24	3.2	3.2	21	22
2011	6.5	6.7	44	45	4.7	4.8	31	32	3.5	3.6	23	24	3.1	3.2	21	21
2012	6.1	6.3	41	42	4.6	4.7	31	32	3.5	3.6	23	24	3.1	3.1	20	21
2013	6.6	6.8	44	45	4.7	4.8	31	32	3.5	3.6	23	24	3.0	3.1	20	21
2014	6.1	6.3	41	42	4.5	4.6	30	31	3.3	3.4	22	23	2.9	3.0	19	20
2015	5.3	5.5	36	37	4.2	4.3	28	29	3.2	3.3	22	22	2.8	2.9	19	20
2016	4.5	4.6	30	31	3.8	3.9	25	26	3.1	3.2	21	21	2.7	2.8	18	19
2017	4.2	4.4	28	29	3.5	3.6	23	24	2.9	3.0	19	20	2.6	2.7	17	18
2018	4.3	4.5	29	30	3.4	3.5	23	23	2.8	2.8	18	19	2.5	2.6	17	17

Bold values indicate exceedances

302823

TABLE 5-38: RATIO OF MODELED EGG CONCENTRATIONS TO BENCHMARKS FOR FEMALE BALD EAGLES BASED ON THE SUM OF TRI+ CONGENERS FOR THE PERIOD 1993 - 2018

Year	LOAEL 152 Average	LOAEL 152 95% UCL	NOAEL 152 Average	NOAEL 152 95% UCL	LOAEL 113 Average	LOAEL 113 95% UCL	NOAEL 113 Average	NOAEL 113 95% UCL	LOAEL 90 Average	LOAEL 90 95% UCL	NOAEL 90 Average	NOAEL 90 95% UCL	LOAEL 50 Average	LOAEL 50 95% UCL	NOAEL 50 Average	NOAEL 50 95% UCL
1993	NA	NA	**139**	**142**	NA	NA	**93**	**95**	NA	NA	**20**	**20**	NA	NA	**20**	**20**
1994	NA	NA	**101**	**103**	NA	NA	**81**	**83**	NA	NA	**18**	**18**	NA	NA	**18**	**18**
1995	NA	NA	**86**	**88**	NA	NA	**72**	**74**	NA	NA	**16**	**16**	NA	NA	**16**	**16**
1996	NA	NA	**103**	**105**	NA	NA	**67**	**69**	NA	NA	**15**	**15**	NA	NA	**15**	**15**
1997	NA	NA	**93**	**95**	NA	NA	**65**	**66**	NA	NA	**14**	**14**	NA	NA	**14**	**14**
1998	NA	NA	**73**	**75**	NA	NA	**59**	**61**	NA	NA	**12**	**13**	NA	NA	**12**	**13**
1999	NA	NA	**63**	**65**	NA	NA	**50**	**51**	NA	NA	**11**	**11**	NA	NA	**11**	**11**
2000	NA	NA	**59**	**61**	NA	NA	**45**	**46**	NA	NA	**10**	**11**	NA	NA	**10**	**11**
2001	NA	NA	**68**	**69**	NA	NA	**46**	**47**	NA	NA	**10**	**10**	NA	NA	**9.7**	**9.8**
2002	NA	NA	**62**	**63**	NA	NA	**45**	**46**	NA	NA	**9.2**	**9.4**	NA	NA	**9.2**	**9.4**
2003	NA	NA	**55**	**56**	NA	NA	**42**	**42**	NA	NA	**8.8**	**8.9**	NA	NA	**8.8**	**8.9**
2004	NA	NA	**44**	**45**	NA	NA	**37**	**38**	NA	NA	**8.2**	**8.3**	NA	NA	**8.2**	**8.3**
2005	NA	NA	**42**	**43**	NA	NA	**34**	**35**	NA	NA	**7.6**	**7.7**	NA	NA	**7.6**	**7.7**
2006	NA	NA	**48**	**49**	NA	NA	**33**	**34**	NA	NA	**7.2**	**7.3**	NA	NA	**7.2**	**7.3**
2007	NA	NA	**44**	**45**	NA	NA	**33**	**33**	NA	NA	**6.8**	**6.9**	NA	NA	**6.8**	**6.9**
2008	NA	NA	**41**	**42**	NA	NA	**32**	**32**	NA	NA	**6.5**	**6.6**	NA	NA	**6.5**	**6.6**
2009	NA	NA	**35**	**36**	NA	NA	**29**	**29**	NA	NA	**6.2**	**6.3**	NA	NA	**6.2**	**6.3**
2010	NA	NA	**37**	**38**	NA	NA	**28**	**28**	NA	NA	**5.9**	**6.0**	NA	NA	**5.9**	**6.0**
2011	NA	NA	**42**	**43**	NA	NA	**29**	**30**	NA	NA	**5.7**	**5.8**	NA	NA	**5.7**	**5.8**
2012	NA	NA	**37**	**38**	NA	NA	**29**	**29**	NA	NA	**5.6**	**5.7**	NA	NA	**5.6**	**5.7**
2013	NA	NA	**41**	**42**	NA	NA	**29**	**30**	NA	NA	**5.7**	**5.8**	NA	NA	**5.7**	**5.8**
2014	NA	NA	**37**	**38**	NA	NA	**28**	**29**	NA	NA	**5.4**	**5.5**	NA	NA	**5.4**	**5.5**
2015	NA	NA	**34**	**35**	NA	NA	**27**	**27**	NA	NA	**5.3**	**5.4**	NA	NA	**5.3**	**5.4**
2016	NA	NA	**31**	**32**	NA	NA	**25**	**25**	NA	NA	**5.1**	**5.2**	NA	NA	**5.1**	**5.2**
2017	NA	NA	**28**	**29**	NA	NA	**23**	**23**	NA	NA	**5.0**	**5.0**	NA	NA	**5.0**	**5.0**
2018	NA	NA	**27**	**28**	NA	NA	**22**	**22**	NA	NA	**4.7**	**4.8**	NA	NA	**4.7**	**4.8**

Bold values indicate exceedances

302824

TABLE 5-39: RATIO OF MODELED DIETARY DOSE BASED ON FISHRAND FOR FEMALE BELTED KINGFISHER USING TEQ FOR THE PERIOD 1993 - 2018

Year	LOAEL 152 Average	LOAEL 152 95% UCL	NOAEL 152 Average	NOAEL 152 95% UCL	LOAEL 113 Average	LOAEL 113 95% UCL	NOAEL 113 Average	NOAEL 113 95% UCL	LOAEL 90 Average	LOAEL 90 95% UCL	NOAEL 90 Average	NOAEL 90 95% UCL	LOAEL 50 Average	LOAEL 50 95% UCL	NOAEL 50 Average	NOAEL 50 95% UCL
1993	8.6	8.9	86	89	6.0	6.2	60	62	4.8	12	48	121	4.3	10	43	102
1994	6.7	6.9	67	69	5.5	5.7	55	57	4.4	11	44	113	3.9	10	39	96
1995	6.0	6.3	60	63	4.6	4.8	46	48	4.3	11	43	107	3.5	9.0	35	90
1996	7.0	7.3	70	73	4.7	4.9	47	49	4.1	10.3	41	103	3.2	8.6	32	86
1997	6.0	6.3	60	63	4.4	4.6	44	46	3.4	9.9	34	99	3.0	8.3	30	83
1998	4.7	5.0	47	50	3.8	4.0	38	40	3.2	9.6	32	96	2.8	8.0	28	80
1999	4.3	4.5	43	45	3.4	3.6	34	36	2.9	9.2	29	92	2.5	7.7	25	77
2000	4.2	4.4	42	44	3.2	3.4	32	34	2.7	8.8	27	88	2.4	7.4	24	74
2001	4.5	4.7	45	47	3.3	3.5	33	35	2.5	8.4	25	84	2.2	7.1	22	71
2002	4.1	4.3	41	43	3.2	3.4	32	34	2.5	8.3	25	83	2.1	6.9	21	69
2003	3.8	4.0	38	40	3.0	3.1	30	31	2.4	8.1	24	81	2.0	6.7	20	67
2004	3.2	3.4	32	34	2.6	2.8	26	28	2.2	8.1	22	81	1.9	6.6	19	66
2005	3.1	3.3	31	33	2.5	2.7	25	27	2.0	7.9	20	79	1.8	6.4	18	64
2006	3.4	3.6	34	36	2.5	2.7	25	27	1.9	7.4	19	74	1.7	6.2	17	62
2007	3.0	3.2	30	32	2.4	2.6	24	26	1.9	7.2	19	72	1.6	6.0	16	60
2008	2.8	3.0	28	30	2.3	2.5	23	25	1.8	7.3	18	73	1.5	5.9	15	59
2009	2.6	2.8	26	28	2.1	2.3	21	23	1.7	7.4	17	74	1.5	5.9	15	59
2010	2.8	2.9	28	29	2.1	2.3	21	23	1.6	6.8	16	68	1.4	5.7	14	57
2011	2.9	3.1	29	31	2.2	2.3	22	23	1.6	6.4	16	64	1.4	5.5	14	55
2012	2.7	2.9	27	29	2.1	2.3	21	23	1.6	6.3	16	63	1.4	5.3	14	53
2013	2.9	3.0	29	30	2.1	2.3	21	23	1.6	6.2	16	62	1.3	5.2	13	52
2014	2.7	2.9	27	29	2.1	2.2	21	22	1.5	6.1	15	61	1.3	5.0	13	50
2015	2.4	2.6	24	26	1.9	2.1	19	21	1.5	6.0	15	60	1.3	5.0	13	50
2016	2.2	2.4	22	24	1.8	1.9	18	19	1.4	6.2	14	62	1.2	5.0	12	50
2017	2.1	2.3	21	23	1.7	1.8	17	18	1.4	6.2	14	62	1.2	4.9	12	49
2018	2.1	2.3	21	23	1.7	1.8	17	18	1.3	5.9	13	59	1.1	4.9	11	49

Bold values indicate exceedances

TABLE 5-40: RATIO OF MODELED DIETARY DOSE BASED ON FISHRAND FOR FEMALE GREAT BLUE HERON USING TEQ FOR THE PERIOD 1993 - 2018

Year	LOAEL 152 Average	LOAEL 152 95% UCL	NOAEL 152 Average	NOAEL 152 95% UCL	LOAEL 113 Average	LOAEL 113 95% UCL	NOAEL 113 Average	NOAEL 113 95% UCL	LOAEL 90 Average	LOAEL 90 95% UCL	NOAEL 90 Average	NOAEL 90 95% UCL	LOAEL 50 Average	LOAEL 50 95% UCL	NOAEL 50 Average	NOAEL 50 95% UCL
1993	**2.2**	**2.3**	**22**	**23**	**1.5**	**1.6**	**15**	**16**	**1.2**	**1.2**	**12**	**12**	**1.1**	**1.2**	**11**	**12**
1994	**1.7**	**1.7**	**17**	**17**	**1.4**	**1.4**	**14**	**14**	**1.1**	**1.1**	**11**	**11**	**1.0**	**1.0**	**10**	**10**
1995	**1.5**	**1.5**	**15**	**15**	**1.1**	**1.2**	**11**	**12**	**1.1**	**1.0**	**11**	**9.9**	0.9	0.9	**9.0**	**9.3**
1996	**1.8**	**1.8**	**18**	**18**	**1.2**	**1.2**	**12**	**12**	**1.0**	0.9	**9.9**	**9.3**	0.8	0.9	**8.3**	**8.5**
1997	**1.5**	**1.6**	**15**	**16**	**1.1**	**1.1**	**11**	**11**	0.8	0.9	**8.4**	**8.6**	0.8	0.8	**7.6**	**7.8**
1998	**1.1**	**1.2**	**11**	**12**	0.9	0.9	**9.0**	**9.3**	0.8	0.8	**7.7**	**8.0**	0.7	0.7	**7.1**	**7.3**
1999	**1.0**	**1.0**	**10**	**10**	0.8	0.8	**8.0**	**8.3**	0.7	0.7	**6.7**	**7.0**	0.6	0.7	**6.3**	**6.5**
2000	**1.0**	**1.0**	**10**	**10**	0.7	0.8	**7.4**	**7.7**	0.6	0.6	**6.2**	**6.4**	0.6	0.6	**5.8**	**6.0**
2001	**1.1**	**1.1**	**11**	**11**	0.8	0.8	**7.7**	**7.9**	0.6	0.6	**5.9**	**6.1**	0.5	0.6	**5.4**	**5.5**
2002	0.9	**1.0**	**9.4**	**10**	0.7	0.8	**7.5**	**7.7**	0.6	0.6	**5.7**	**5.9**	0.5	0.5	**5.1**	**5.3**
2003	0.9	0.9	**8.7**	**9.2**	0.7	0.7	**6.8**	**7.1**	0.5	0.6	**5.4**	**5.6**	0.5	0.5	**4.9**	**5.0**
2004	0.7	0.7	**6.8**	**7.3**	0.6	0.6	**5.9**	**6.1**	0.5	0.5	**5.0**	**5.1**	0.5	0.5	**4.5**	**4.7**
2005	0.7	0.7	**6.6**	**7.1**	0.5	0.6	**5.5**	**5.7**	0.5	0.5	**4.6**	**4.8**	0.4	0.4	**4.2**	**4.3**
2006	0.8	0.8	**7.9**	**8.4**	0.6	0.6	**5.6**	**5.8**	0.4	0.5	**4.3**	**4.5**	0.4	0.4	**3.9**	**4.1**
2007	0.6	0.7	**6.5**	**6.9**	0.5	0.6	**5.4**	**5.6**	0.4	0.4	**4.2**	**4.3**	0.4	0.4	**3.7**	**3.9**
2008	0.6	0.6	**6.0**	**6.5**	0.5	0.5	**5.1**	**5.3**	0.4	0.4	**4.0**	**4.2**	0.4	0.4	**3.6**	**3.7**
2009	0.5	0.6	**5.5**	**5.9**	0.5	0.5	**4.6**	**4.9**	0.4	0.4	**3.8**	**3.9**	0.3	0.4	**3.4**	**3.5**
2010	0.6	0.6	**6.0**	**6.5**	0.5	0.5	**4.6**	**4.8**	0.4	0.4	**3.6**	**3.7**	0.3	0.3	**3.2**	**3.3**
2011	0.7	0.7	**6.5**	**6.9**	0.5	0.5	**4.8**	**5.0**	0.4	0.4	**3.6**	**3.7**	0.3	0.3	**3.1**	**3.3**
2012	0.6	0.6	**6.1**	**6.5**	0.5	0.5	**4.7**	**4.9**	0.4	0.4	**3.6**	**3.7**	0.3	0.3	**3.1**	**3.2**
2013	0.7	0.7	**6.5**	**6.9**	0.5	0.5	**4.8**	**5.0**	0.4	0.4	**3.6**	**3.7**	0.3	0.3	**3.1**	**3.2**
2014	0.6	0.6	**6.1**	**6.5**	0.5	0.5	**4.6**	**4.8**	0.3	0.4	**3.5**	**3.6**	0.3	0.3	**3.0**	**3.1**
2015	0.5	0.6	**5.3**	**5.7**	0.4	0.4	**4.3**	**4.5**	0.3	0.3	**3.3**	**3.5**	0.3	0.3	**2.9**	**3.0**
2016	0.5	0.5	**4.5**	**4.9**	0.4	0.4	**3.9**	**4.1**	0.3	0.3	**3.2**	**3.3**	0.3	0.3	**2.8**	**2.9**
2017	0.4	0.5	**4.3**	**4.7**	0.4	0.4	**3.6**	**3.8**	0.3	0.3	**3.0**	**3.1**	0.3	0.3	**2.7**	**2.8**
2018	0.4	0.5	**4.3**	**4.7**	0.4	0.4	**3.5**	**3.7**	0.3	0.3	**2.9**	**3.0**	0.3	0.3	**2.6**	**2.7**

Bold values indicate exceedances

TABLE 5-41: RATIO OF MODELED DIETARY DOSE BASED ON FISHRAND FOR FEMALE BALD EAGLE USING TEQ FOR THE PERIOD 1993 - 2018

Year	LOAEL 152 Average	LOAEL 152 95% UCL	NOAEL 152 Average	NOAEL 152 95% UCL	LOAEL 113 Average	LOAEL 113 95% UCL	NOAEL 113 Average	NOAEL 113 95% UCL	LOAEL 90 Average	LOAEL 90 95% UCL	NOAEL 90 Average	NOAEL 90 95% UCL	LOAEL 50 Average	LOAEL 50 95% UCL	NOAEL 50 Average	NOAEL 50 95% UCL
1993	**26**	**26**	**257**	**263**	**17**	**18**	**172**	**176**	**3.9**	**4.0**	**39**	**40**	**3.7**	**3.8**	**37**	**38**
1994	**19**	**19**	**186**	**190**	**15**	**15**	**150**	**154**	**3.6**	**3.6**	**36**	**36**	**3.3**	**3.4**	**33**	**34**
1995	**16**	**16**	**160**	**163**	**13**	**14**	**134**	**137**	**3.2**	**3.3**	**32**	**33**	**3.0**	**3.0**	**30**	**30**
1996	**19**	**19**	**190**	**194**	**12**	**13**	**125**	**127**	**2.9**	**3.0**	**29**	**30**	**2.7**	**2.8**	**27**	**28**
1997	**17**	**18**	**173**	**176**	**12**	**12**	**120**	**123**	**2.8**	**2.8**	**28**	**28**	**2.5**	**2.5**	**25**	**25**
1998	**14**	**14**	**136**	**139**	**11**	**11**	**110**	**112**	**2.5**	**2.6**	**25**	**26**	**2.3**	**2.3**	**23**	**23**
1999	**12**	**12**	**117**	**120**	**9.2**	**9.4**	**92**	**94**	**2.3**	**2.3**	**23**	**23**	**2.1**	**2.1**	**21**	**21**
2000	**11**	**11**	**110**	**112**	**8.3**	**8.5**	**83**	**85**	**2.1**	**2.1**	**21**	**21**	**1.9**	**1.9**	**19**	**19**
2001	**12**	**13**	**125**	**128**	**8.5**	**8.6**	**85**	**86**	**2.0**	**2.0**	**20**	**20**	**1.8**	**1.8**	**18**	**18**
2002	**11**	**12**	**114**	**117**	**8.4**	**8.6**	**84**	**86**	**1.9**	**1.9**	**19**	**19**	**1.7**	**1.7**	**17**	**17**
2003	**10**	**10**	**102**	**104**	**7.7**	**7.9**	**77**	**79**	**1.8**	**1.9**	**18**	**19**	**1.6**	**1.6**	**16**	**16**
2004	**8.2**	**8.4**	**82**	**84**	**6.8**	**7.0**	**68**	**70**	**1.7**	**1.7**	**17**	**17**	**1.5**	**1.5**	**15**	**15**
2005	**7.8**	**8.0**	**78**	**80**	**6.3**	**6.4**	**63**	**64**	**1.6**	**1.6**	**16**	**16**	**1.4**	**1.4**	**14**	**14**
2006	**8.8**	**9.0**	**88**	**90**	**6.2**	**6.3**	**62**	**63**	**1.5**	**1.5**	**15**	**15**	**1.3**	**1.3**	**13**	**13**
2007	**8.1**	**8.3**	**81**	**83**	**6.1**	**6.2**	**61**	**62**	**1.4**	**1.4**	**14**	**14**	**1.3**	**1.3**	**13**	**13**
2008	**7.6**	**7.7**	**76**	**77**	**5.8**	**6.0**	**58**	**60**	**1.4**	**1.4**	**14**	**14**	**1.2**	**1.2**	**12**	**12**
2009	**6.6**	**6.7**	**66**	**67**	**5.3**	**5.4**	**53**	**54**	**1.3**	**1.3**	**13**	**13**	**1.1**	**1.2**	**11**	**12**
2010	**6.9**	**7.0**	**69**	**70**	**5.1**	**5.2**	**51**	**52**	**1.2**	**1.3**	**12**	**13**	**1.1**	**1.1**	**11**	**11**
2011	**7.7**	**7.9**	**77**	**79**	**5.4**	**5.5**	**54**	**55**	**1.2**	**1.2**	**12**	**12**	**1.1**	**1.1**	**11**	**11**
2012	**6.9**	**7.1**	**69**	**71**	**5.3**	**5.4**	**53**	**54**	**1.2**	**1.2**	**12**	**12**	**1.0**	**1.1**	**10**	**11**
2013	**7.5**	**7.7**	**75**	**77**	**5.5**	**5.6**	**55**	**56**	**1.2**	**1.3**	**12**	**13**	**1.1**	**1.1**	**11**	**11**
2014	**6.8**	**7.0**	**68**	**70**	**5.2**	**5.3**	**52**	**53**	**1.2**	**1.2**	**12**	**12**	**1.0**	**1.0**	**10**	**10**
2015	**6.3**	**6.5**	**63**	**65**	**4.9**	**5.0**	**49**	**50**	**1.1**	**1.2**	**11**	**12**	**1.0**	**1.0**	**10**	**10**
2016	**5.8**	**5.9**	**58**	**59**	**4.6**	**4.7**	**46**	**47**	**1.1**	**1.1**	**11**	**11**	0.9	**1.0**	**9.5**	**10**
2017	**5.2**	**5.3**	**52**	**53**	**4.2**	**4.3**	**42**	**43**	**1.0**	**1.1**	**10**	**11**	0.9	0.9	**9.2**	**9.3**
2018	**5.1**	**5.2**	**51**	**52**	**4.0**	**4.1**	**40**	**41**	**1.0**	**1.0**	**10**	**10**	0.9	0.9	**8.7**	**8.8**

Bold values indicate exceedances

302827

TABLE 5-42: RATIO OF MODELED EGG CONCENTRATIONS BASED ON FISHRAND FOR FEMALE BELTED KINGFISHER USING TEQ FOR THE PERIOD 1993 - 2018

Year	LOAEL 152 Average	LOAEL 152 95% UCL	NOAEL 152 Average	NOAEL 152 95% UCL	LOAEL 113 Average	LOAEL 113 95% UCL	NOAEL 113 Average	NOAEL 113 95% UCL	LOAEL 90 Average	LOAEL 90 95% UCL	NOAEL 90 Average	NOAEL 90 95% UCL	LOAEL 50 Average	LOAEL 50 95% UCL	NOAEL 50 Average	NOAEL 50 95% UCL
1993	281	291	563	582	195	203	391	405	142	147	283	293	142	147	283	293
1994	217	225	434	450	178	185	356	369	128	132	255	264	128	132	255	264
1995	196	204	393	408	150	156	301	313	115	119	229	238	115	119	229	238
1996	229	237	457	473	154	160	308	319	106	110	212	220	106	110	212	220
1997	197	204	394	408	144	150	288	299	98	102	196	203	98	102	196	203
1998	154	160	308	321	123	128	247	257	91	95	183	190	91	95	183	190
1999	138	144	277	289	112	116	223	233	83	86	166	172	83	86	166	172
2000	136	141	272	283	104	108	208	217	77	80	153	159	77	80	153	159
2001	147	152	294	305	107	111	214	222	72	74	143	149	72	74	143	149
2002	132	138	264	275	104	108	207	216	69	71	138	143	69	71	138	143
2003	122	127	244	254	95	100	191	199	65	68	131	136	65	68	131	136
2004	101	106	203	212	85	88	169	177	61	64	122	127	61	64	122	127
2005	99	103	198	207	80	84	161	168	57	60	114	119	57	60	114	119
2006	112	116	223	232	81	85	162	169	54	56	107	112	54	56	107	112
2007	96	100	192	200	78	82	156	163	51	54	103	107	51	54	103	107
2008	90	94	180	188	74	77	148	155	49	51	99	103	49	51	99	103
2009	84	88	167	175	69	72	138	144	47	49	94	98	47	49	94	98
2010	89	93	178	186	68	71	135	142	45	47	90	94	45	47	90	94
2011	94	97	187	195	70	73	140	146	44	46	88	92	44	46	88	92
2012	88	92	176	184	69	72	137	144	44	46	87	91	44	46	87	91
2013	93	96	185	193	69	72	138	145	43	45	86	90	43	45	86	90
2014	88	91	175	183	66	69	133	139	42	43	83	87	42	43	83	87
2015	79	82	158	165	63	66	126	131	41	42	81	85	41	42	81	85
2016	70	74	140	147	58	61	116	121	39	41	79	82	39	41	79	82
2017	68	71	135	142	55	58	110	115	38	39	75	79	38	39	75	79
2018	68	71	136	142	54	57	108	113	37	38	73	76	37	38	73	76

Bold values indicate exceedances

302828

TABLE 5-43: RATIO OF MODELED EGG CONCENTRATIONS BASED ON FISHRAND FOR FEMALE GREAT BLUE HERON USING TEQ FOR THE PERIOD 1993 - 2018

Year	LOAEL 152 Average	LOAEL 152 95% UCL	NOAEL 152 Average	NOAEL 152 95% UCL	LOAEL 113 Average	LOAEL 113 95% UCL	NOAEL 113 Average	NOAEL 113 95% UCL	LOAEL 90 Average	LOAEL 90 95% UCL	NOAEL 90 Average	NOAEL 90 95% UCL	LOAEL 50 Average	LOAEL 50 95% UCL	NOAEL 50 Average	NOAEL 50 95% UCL
1993	**7.4**	**7.6**	**12**	**13**	**5.0**	**5.1**	**8.3**	**8.5**	**4.0**	**4.1**	**6.6**	**6.8**	**3.8**	**3.9**	**6.3**	**6.4**
1994	**5.5**	**5.6**	**9.1**	**9.4**	**4.5**	**4.6**	**7.5**	**7.7**	**3.6**	**3.7**	**6.0**	**6.1**	**3.4**	**3.4**	**5.6**	**5.7**
1995	**4.9**	**5.0**	**8.1**	**8.3**	**3.6**	**3.8**	**6.1**	**6.3**	**3.1**	**3.2**	**5.2**	**5.4**	**3.0**	**3.1**	**5.0**	**5.1**
1996	**5.9**	**6.1**	**10**	**10**	**3.8**	**3.9**	**6.3**	**6.5**	**2.9**	**3.0**	**4.9**	**5.0**	**2.7**	**2.8**	**4.5**	**4.7**
1997	**5.0**	**5.1**	**8.3**	**8.5**	**3.5**	**3.6**	**5.9**	**6.1**	**2.7**	**2.8**	**4.5**	**4.7**	**2.5**	**2.6**	**4.2**	**4.3**
1998	**3.7**	**3.8**	**6.1**	**6.3**	**2.9**	**3.0**	**4.8**	**5.0**	**2.5**	**2.6**	**4.2**	**4.3**	**2.3**	**2.4**	**3.9**	**4.0**
1999	**3.2**	**3.3**	**5.4**	**5.6**	**2.6**	**2.6**	**4.3**	**4.4**	**2.2**	**2.2**	**3.6**	**3.7**	**2.1**	**2.1**	**3.4**	**3.5**
2000	**3.1**	**3.2**	**5.2**	**5.4**	**2.4**	**2.4**	**3.9**	**4.1**	**2.0**	**2.0**	**3.3**	**3.4**	**1.9**	**1.9**	**3.1**	**3.2**
2001	**3.5**	**3.6**	**5.9**	**6.0**	**2.5**	**2.5**	**4.1**	**4.2**	**1.9**	**1.9**	**3.1**	**3.2**	**1.7**	**1.8**	**2.9**	**3.0**
2002	**3.1**	**3.2**	**5.2**	**5.3**	**2.4**	**2.5**	**4.0**	**4.1**	**1.8**	**1.9**	**3.1**	**3.1**	**1.7**	**1.7**	**2.8**	**2.8**
2003	**2.8**	**2.9**	**4.7**	**4.9**	**2.2**	**2.2**	**3.6**	**3.7**	**1.7**	**1.8**	**2.9**	**3.0**	**1.6**	**1.6**	**2.6**	**2.7**
2004	**2.2**	**2.3**	**3.7**	**3.8**	**1.9**	**1.9**	**3.1**	**3.2**	**1.6**	**1.6**	**2.6**	**2.7**	**1.5**	**1.5**	**2.4**	**2.5**
2005	**2.2**	**2.2**	**3.6**	**3.7**	**1.7**	**1.8**	**2.9**	**3.0**	**1.5**	**1.5**	**2.4**	**2.5**	**1.3**	**1.4**	**2.2**	**2.3**
2006	**2.6**	**2.7**	**4.3**	**4.5**	**1.8**	**1.8**	**3.0**	**3.1**	**1.4**	**1.4**	**2.3**	**2.4**	**1.3**	**1.3**	**2.1**	**2.2**
2007	**2.1**	**2.2**	**3.5**	**3.6**	**1.7**	**1.8**	**2.8**	**2.9**	**1.3**	**1.4**	**2.2**	**2.3**	**1.2**	**1.2**	**2.0**	**2.1**
2008	**2.0**	**2.0**	**3.3**	**3.4**	**1.6**	**1.7**	**2.7**	**2.8**	**1.3**	**1.3**	**2.1**	**2.2**	**1.1**	**1.2**	**1.9**	**2.0**
2009	**1.8**	**1.8**	**3.0**	**3.0**	**1.5**	**1.5**	**2.4**	**2.5**	**1.2**	**1.2**	**2.0**	**2.0**	**1.1**	**1.1**	**1.8**	**1.9**
2010	**2.0**	**2.0**	**3.3**	**3.4**	**1.4**	**1.5**	**2.4**	**2.5**	**1.1**	**1.2**	**1.9**	**1.9**	**1.0**	**1.1**	**1.7**	**1.8**
2011	**2.1**	**2.2**	**3.6**	**3.7**	**1.5**	**1.6**	**2.5**	**2.6**	**1.1**	**1.2**	**1.9**	**1.9**	**1.0**	**1.0**	**1.7**	**1.7**
2012	**2.0**	**2.0**	**3.3**	**3.4**	**1.5**	**1.5**	**2.5**	**2.6**	**1.1**	**1.2**	**1.9**	**2.0**	**1.0**	**1.0**	**1.7**	**1.7**
2013	**2.1**	**2.2**	**3.6**	**3.7**	**1.5**	**1.6**	**2.5**	**2.6**	**1.1**	**1.2**	**1.9**	**2.0**	**1.0**	**1.0**	**1.6**	**1.7**
2014	**2.0**	**2.1**	**3.3**	**3.4**	**1.5**	**1.5**	**2.4**	**2.5**	**1.1**	**1.1**	**1.8**	**1.9**	0.9	**1.0**	**1.6**	**1.6**
2015	**1.7**	**1.8**	**2.9**	**3.0**	**1.4**	**1.4**	**2.3**	**2.3**	**1.1**	**1.1**	**1.8**	**1.8**	0.9	**1.0**	**1.5**	**1.6**
2016	**1.5**	**1.5**	**2.4**	**2.5**	**1.2**	**1.3**	**2.0**	**2.1**	**1.0**	**1.0**	**1.7**	**1.7**	0.9	0.9	**1.5**	**1.5**
2017	**1.4**	**1.4**	**2.3**	**2.4**	**1.1**	**1.2**	**1.9**	**2.0**	0.9	**1.0**	**1.6**	**1.6**	0.9	0.9	**1.4**	**1.5**
2018	**1.4**	**1.5**	**2.3**	**2.4**	**1.1**	**1.1**	**1.8**	**1.9**	0.9	0.9	**1.5**	**1.5**	0.8	0.8	**1.4**	**1.4**

Bold values indicate exceedances

302829

TABLE 5-44: RATIO OF MODELED EGG CONCENTRATIONS BASED ON FISHRAND FOR FEMALE BALD EAGLE USING TEQ FOR THE PERIOD 1993 - 2018

Year	LOAEL 152 Average	LOAEL 152 95% UCL	NOAEL 152 Average	NOAEL 152 95% UCL	LOAEL 113 Average	LOAEL 113 95% UCL	NOAEL 113 Average	NOAEL 113 95% UCL	LOAEL 90 Average	LOAEL 90 95% UCL	NOAEL 90 Average	NOAEL 90 95% UCL	LOAEL 50 Average	LOAEL 50 95% UCL	NOAEL 50 Average	NOAEL 50 95% UCL
1993	2683	2741	5367	5482	1795	1834	3590	3668	412	418	823	837	388	394	776	788
1994	1944	1986	3889	3973	1569	1603	3138	3206	375	381	749	762	347	353	695	706
1995	1669	1705	3338	3409	1395	1425	2790	2850	338	343	676	687	313	318	625	635
1996	1982	2024	3963	4049	1301	1329	2602	2658	308	313	615	626	283	287	565	574
1997	1802	1840	3604	3680	1256	1284	2513	2567	288	292	575	585	261	265	522	531
1998	1416	1447	2832	2893	1144	1169	2288	2339	265	269	530	539	240	243	479	487
1999	1223	1250	2446	2499	958	979	1916	1958	239	243	477	485	218	222	437	444
2000	1148	1173	2295	2346	871	891	1743	1782	217	220	433	441	200	203	400	407
2001	1304	1333	2608	2665	882	902	1765	1804	206	209	411	418	187	190	374	380
2002	1193	1218	2386	2437	873	892	1746	1785	200	203	400	407	178	181	357	363
2003	1063	1087	2125	2173	802	820	1604	1641	190	193	380	386	169	172	338	344
2004	858	877	1715	1753	711	727	1423	1455	176	179	353	359	158	161	316	322
2005	817	835	1633	1670	654	669	1309	1339	163	165	325	331	147	150	294	299
2006	920	941	1841	1882	646	661	1293	1322	154	157	308	314	138	141	276	281
2007	849	868	1698	1735	633	647	1265	1293	149	151	297	303	132	134	263	268
2008	789	806	1577	1612	608	622	1217	1244	143	146	287	292	126	128	252	256
2009	685	701	1370	1401	556	568	1112	1137	135	138	271	276	120	122	239	244
2010	719	735	1439	1471	531	543	1062	1086	129	131	257	262	114	116	228	232
2011	806	824	1611	1648	565	578	1130	1155	127	130	255	260	111	113	221	225
2012	720	736	1440	1472	552	564	1104	1129	126	129	253	257	108	110	216	220
2013	786	803	1572	1607	570	583	1139	1165	130	132	259	264	111	112	221	225
2014	715	731	1430	1462	540	552	1080	1104	123	125	245	250	105	106	209	213
2015	660	675	1320	1351	515	526	1029	1053	119	121	238	242	102	104	204	207
2016	604	617	1207	1234	480	490	959	981	114	116	228	232	99	101	198	201
2017	544	556	1087	1111	442	452	884	904	108	110	216	220	96	97	191	195
2018	530	542	1060	1085	420	429	839	859	102	104	204	208	90	92	181	184

Bold values indicate exceedances

302830

TABLE 5-45: RATIO OF MODELED DIETARY DOSES TO TOXICITY BENCHMARKS
FOR FEMALE BAT FOR TRI+ CONGENERS FOR THE PERIOD 1993 - 2018

Year	LOAEL 152 Average	LOAEL 152 95% UCL	NOAEL 152 Average	NOAEL 152 95% UCL	LOAEL 113 Average	LOAEL 113 95% UCL	NOAEL 113 Average	NOAEL 113 95% UCL	LOAEL 90 Average	LOAEL 90 95% UCL	NOAEL 90 Average	NOAEL 90 95%UCL	LOAEL 50 Average	LOAEL 50 95% UCL	NOAEL 50 Average	NOAEL 50 95% UCL
1993	**4.1**	**4.4**	**19**	**21**	**3.3**	**3.5**	**15**	**16**	**2.7**	**2.9**	**12**	**13**	**20**	**21**	**9.1**	**10**
1994	**3.7**	**4.0**	**17**	**19**	**3.1**	**3.3**	**14**	**15**	**2.5**	**2.7**	**12**	**13**	**18**	**20**	**8.6**	**9.2**
1995	**3.6**	**3.8**	**17**	**18**	**2.9**	**3.1**	**14**	**15**	**2.4**	**2.5**	**11**	**12**	**17**	**19**	**8.2**	**8.7**
1996	**3.5**	**3.8**	**17**	**18**	**2.8**	**3.0**	**13**	**14**	**2.2**	**2.4**	**11**	**11**	**17**	**18**	**7.8**	**8.4**
1997	**3.3**	**3.6**	**16**	**17**	**2.7**	**2.9**	**13**	**14**	**2.2**	**2.3**	**10**	**11**	**16**	**17**	**7.6**	**8.1**
1998	**3.2**	**3.4**	**15**	**16**	**2.6**	**2.8**	**12**	**13**	**2.1**	**2.2**	**10**	**10**	**15**	**16**	**7.2**	**7.7**
1999	**3.0**	**3.3**	**14**	**15**	**2.6**	**2.7**	**12**	**13**	**2.0**	**2.1**	**9.4**	**10**	**15**	**16**	**7.0**	**7.5**
2000	**3.0**	**3.3**	**14**	**15**	**2.4**	**2.6**	**11**	**12**	**1.9**	**2.1**	**9.1**	**10**	**14**	**15**	**6.8**	**7.2**
2001	**3.0**	**3.2**	**14**	**15**	**2.4**	**2.6**	**11**	**12**	**1.9**	**2.0**	**8.9**	**9.5**	**14**	**15**	**6.5**	**7.0**
2002	**2.8**	**3.1**	**13**	**14**	**2.3**	**2.5**	**11**	**12**	**1.8**	**2.0**	**8.6**	**9.2**	**14**	**15**	**6.4**	**6.9**
2003	**2.7**	**2.9**	**13**	**13**	**2.2**	**2.4**	**10**	**11**	**1.8**	**1.9**	**8.4**	**9.1**	**13**	**14**	**6.2**	**6.7**
2004	**2.6**	**2.8**	**12**	**13**	**2.1**	**2.3**	**10**	**11**	**1.7**	**1.8**	**8.0**	**8.6**	**13**	**14**	**5.9**	**6.4**
2005	**2.6**	**2.8**	**12**	**13**	**2.1**	**2.3**	**10**	**11**	**1.6**	**1.8**	**7.7**	**8.3**	**12**	**13**	**5.7**	**6.1**
2006	**2.5**	**2.6**	**12**	**12**	**2.1**	**2.2**	**10**	**10**	**1.6**	**1.7**	**7.4**	**7.9**	**12**	**13**	**5.5**	**5.9**
2007	**2.4**	**2.6**	**11**	**12**	**2.0**	**2.2**	**9.5**	**10**	**1.5**	**1.6**	**7.2**	**7.7**	**11**	**12**	**5.3**	**5.7**
2008	**2.3**	**2.5**	**11**	**12**	**1.9**	**2.1**	**9.1**	**10**	**1.5**	**1.6**	**7.0**	**7.5**	**11**	**12**	**5.2**	**5.5**
2009	**2.3**	**2.5**	**11**	**12**	**1.9**	**2.0**	**8.8**	**10**	**1.5**	**1.6**	**6.8**	**7.3**	**11**	**12**	**5.1**	**5.4**
2010	**2.3**	**2.4**	**11**	**11**	**1.8**	**2.0**	**8.6**	**9.3**	**1.4**	**1.5**	**6.7**	**7.2**	**11**	**11**	**5.0**	**5.3**
2011	**2.2**	**2.3**	**10**	**11**	**1.8**	**2.0**	**8.6**	**9.2**	**1.4**	**1.5**	**6.5**	**6.9**	**10**	**11**	**4.9**	**5.2**
2012	**2.1**	**2.3**	**10**	**11**	**1.8**	**1.9**	**8.4**	**9.0**	**1.3**	**1.4**	**6.3**	**6.8**	**10**	**11**	**4.8**	**5.1**
2013	**2.1**	**2.2**	**10**	**10**	**1.7**	**1.9**	**8.2**	**8.8**	**1.3**	**1.4**	**6.1**	**6.6**	**10**	**11**	**4.6**	**5.0**
2014	**2.0**	**2.2**	**10**	**10**	**1.7**	**1.8**	**8.0**	**8.6**	**1.3**	**1.4**	**6.0**	**6.4**	**10**	**10**	**4.5**	**4.9**
2015	**2.0**	**2.1**	**9.3**	**10**	**1.6**	**1.8**	**7.7**	**8.3**	**1.2**	**1.3**	**5.9**	**6.3**	**9.4**	**10**	**4.4**	**4.7**
2016	**2.0**	**2.2**	**9.4**	**10**	**1.6**	**1.7**	**7.5**	**8.1**	**1.2**	**1.3**	**5.7**	**6.2**	**9.2**	**10**	**4.3**	**4.6**
2017	**2.0**	**2.1**	**9.3**	**10**	**1.6**	**1.7**	**7.4**	**8.0**	**1.2**	**1.3**	**5.7**	**6.1**	**9.0**	**10**	**4.2**	**4.5**
2018	**1.9**	**2.1**	**9.0**	**10**	**1.6**	**1.7**	**7.4**	**8.0**	**1.2**	**1.3**	**5.6**	**6.0**	**8.8**	**9.4**	**4.1**	**4.4**

Bold values indicate exceedances

302831

TABLE 5-46: RATIO OF MODELED DIETARY DOSES TO TOXICITY BENCHMARKS FOR FEMALE BAT ON A TEQ BASIS FOR THE PERIOD 1993 - 2018

Year	LOAEL 152 Average	LOAEL 152 95% UCL	NOAEL 152 Average	NOAEL 152 95% UCL	LOAEL 113 Average	LOAEL 113 95% UCL	NOAEL 113 Average	NOAEL 113 95% UCL	LOAEL 90 Average	LOAEL 90 95% UCL	NOAEL 90 Average	NOAEL 90 95%UCL	LOAEL 50 Average	LOAEL 50 95% UCL	NOAEL 50 Average	NOAEL 50 95% UCL
1993	67	72	667	716	53	57	529	568	43	46	430	462	32	34	316	339
1994	60	64	598	641	50	53	496	531	41	44	408	437	30	32	296	318
1995	58	62	578	620	48	51	475	510	38	41	382	410	28	30	281	302
1996	57	61	571	612	46	49	457	490	36	39	364	390	27	29	271	290
1997	54	58	540	579	44	47	438	469	35	38	353	378	26	28	262	281
1998	52	55	517	555	43	46	426	456	34	36	336	360	25	27	248	265
1999	49	53	490	527	41	44	413	443	32	35	324	347	24	26	241	258
2000	49	53	493	529	40	43	396	425	31	34	315	337	23	25	233	250
2001	48	52	482	517	39	42	390	419	31	33	306	327	23	24	226	242
2002	46	49	461	495	38	40	377	404	30	32	298	319	22	24	222	238
2003	43	47	433	466	36	39	359	386	29	31	291	313	21	23	214	230
2004	43	46	426	459	35	37	347	373	28	30	276	297	20	22	205	220
2005	41	45	415	446	34	37	343	369	27	29	266	286	20	21	197	212
2006	40	43	398	428	33	36	333	358	25	27	254	273	19	20	188	202
2007	39	42	393	423	33	35	326	351	25	27	248	266	18	20	183	197
2008	38	41	379	409	31	34	314	338	24	26	240	258	18	19	178	191
2009	37	40	372	401	30	33	305	328	24	25	235	253	17	19	174	188
2010	37	39	365	393	30	32	299	322	23	25	231	248	17	18	171	184
2011	35	38	350	376	30	32	296	318	22	24	223	240	17	18	168	181
2012	34	37	341	367	29	31	289	311	22	23	218	234	16	18	165	177
2013	33	36	334	359	28	30	283	305	21	23	211	227	16	17	160	172
2014	33	36	331	355	28	30	276	297	21	22	206	222	16	17	156	167
2015	32	35	321	346	27	29	266	286	20	22	202	217	15	16	152	163
2016	32	35	324	351	26	28	259	279	20	21	198	213	15	16	149	160
2017	32	35	320	347	26	28	256	277	20	21	195	210	15	16	145	156
2018	31	34	312	338	26	28	256	277	19	21	192	206	14	15	142	153

Bold values indicate exceedances

302832

TABLE 5-47: RATIO OF MODELED DIETARY DOSES TO TOXICITY BENCHMARKS FOR FEMALE RACCOON FOR TRI+ CONGENERS FOR THE PERIOD 1993 - 2018

Year	LOAEL 152 Average	LOAEL 152 95% UCL	NOAEL 152 Average	NOAEL 152 95% UCL	LOAEL 113 Average	LOAEL 113 95% UCL	NOAEL 113 Average	NOAEL 113 95% UCL	LOAEL 90 Average	LOAEL 90 95% UCL	NOAEL 90 Average	NOAEL 90 95% UCL	LOAEL 50 Average	LOAEL 50 95% UCL	NOAEL 50 Average	NOAEL 50 95% UCL
1993	0.8	0.8	**3.5**	**3.8**	0.6	0.6	**2.8**	**3.0**	0.5	0.5	**2.2**	**2.4**	0.4	0.4	**1.7**	**1.8**
1994	0.7	0.7	**3.1**	**3.3**	0.6	0.6	**2.6**	**2.8**	0.5	0.5	**2.1**	**2.3**	0.3	0.4	**1.6**	**1.7**
1995	0.6	0.7	**3.0**	**3.2**	0.5	0.6	**2.5**	**2.6**	0.5	0.5	**2.2**	**2.1**	0.3	0.3	**1.5**	**1.6**
1996	0.6	0.7	**3.0**	**3.2**	0.5	0.5	**2.4**	**2.5**	0.5	0.4	**2.1**	**2.0**	0.3	0.3	**1.4**	**1.5**
1997	0.6	0.6	**2.8**	**3.0**	0.5	0.5	**2.3**	**2.4**	0.4	0.4	**1.8**	**2.0**	0.3	0.3	**1.4**	**1.5**
1998	0.6	0.6	**2.7**	**2.9**	0.5	0.5	**2.2**	**2.4**	0.4	0.4	**1.7**	**1.9**	0.3	0.3	**1.3**	**1.4**
1999	0.5	0.6	**2.5**	**2.7**	0.5	0.5	**2.1**	**2.3**	0.4	0.4	**1.7**	**1.8**	0.3	0.3	**1.2**	**1.3**
2000	0.5	0.6	**2.5**	**2.7**	0.4	0.5	**2.0**	**2.2**	0.3	0.4	**1.6**	**1.7**	0.3	0.3	**1.2**	**1.3**
2001	0.5	0.6	**2.5**	**2.7**	0.4	0.5	**2.0**	**2.2**	0.3	0.4	**1.6**	**1.7**	0.2	0.3	**1.2**	**1.3**
2002	0.5	0.5	**2.4**	**2.6**	0.4	0.4	**1.9**	**2.1**	0.3	0.4	**1.5**	**1.6**	0.2	0.3	**1.1**	**1.2**
2003	0.5	0.5	**2.2**	**2.4**	0.4	0.4	**1.8**	**2.0**	0.3	0.3	**1.5**	**1.6**	0.2	0.3	**1.1**	**1.2**
2004	0.5	0.5	**2.2**	**2.4**	0.4	0.4	**1.8**	**1.9**	0.3	0.3	**1.4**	**1.5**	0.2	0.2	**1.1**	**1.1**
2005	0.4	0.5	**2.1**	**2.3**	0.4	0.4	**1.7**	**1.9**	0.3	0.3	**1.4**	**1.5**	0.2	0.2	**1.0**	**1.1**
2006	0.4	0.5	**2.0**	**2.2**	0.4	0.4	**1.7**	**1.8**	0.3	0.3	**1.3**	**1.4**	0.2	0.2	**1.0**	**1.0**
2007	0.4	0.5	**2.0**	**2.2**	0.4	0.4	**1.7**	**1.8**	0.3	0.3	**1.3**	**1.4**	0.2	0.2	0.9	**1.0**
2008	0.4	0.5	**1.9**	**2.1**	0.3	0.4	**1.6**	**1.7**	0.3	0.3	**1.2**	**1.3**	0.2	0.2	0.9	**1.0**
2009	0.4	0.4	**1.9**	**2.1**	0.3	0.4	**1.5**	**1.7**	0.3	0.3	**1.2**	**1.3**	0.2	0.2	0.9	**1.0**
2010	0.4	0.4	**1.9**	**2.0**	0.3	0.4	**1.5**	**1.7**	0.2	0.3	**1.2**	**1.3**	0.2	0.2	0.9	**1.0**
2011	0.4	0.4	**1.8**	**1.9**	0.3	0.3	**1.5**	**1.6**	0.2	0.3	**1.1**	**1.2**	0.2	0.2	0.9	0.9
2012	0.4	0.4	**1.7**	**1.9**	0.3	0.3	**1.5**	**1.6**	0.2	0.3	**1.1**	**1.2**	0.2	0.2	0.8	0.9
2013	0.4	0.4	**1.7**	**1.9**	0.3	0.3	**1.4**	**1.6**	0.2	0.2	**1.1**	**1.2**	0.2	0.2	0.8	0.9
2014	0.4	0.4	**1.7**	**1.8**	0.3	0.3	**1.4**	**1.5**	0.2	0.2	**1.0**	**1.1**	0.2	0.2	0.8	0.9
2015	0.3	0.4	**1.6**	**1.8**	0.3	0.3	**1.4**	**1.5**	0.2	0.2	**1.0**	**1.1**	0.2	0.2	0.8	0.8
2016	0.3	0.4	**1.6**	**1.8**	0.3	0.3	**1.3**	**1.4**	0.2	0.2	**1.0**	**1.1**	0.2	0.2	0.8	0.8
2017	0.3	0.4	**1.6**	**1.8**	0.3	0.3	**1.3**	**1.4**	0.2	0.2	**1.0**	**1.1**	0.2	0.2	0.7	0.8
2018	0.3	0.4	**1.6**	**1.7**	0.3	0.3	**1.3**	**1.4**	0.2	0.2	**1.0**	**1.1**	0.2	0.2	0.7	0.8

Bold values indicate exceedances

302833

TABLE 5-48: RATIO OF MODELED DIETARY DOSES TO TOXICITY BENCHMARKS FOR FEMALE RACCOON ON A TEQ BASIS FOR THE PERIOD 1993 - 2018

Year	LOAEL 152 Average	LOAEL 152 95% UCL	NOAEL 152 Average	NOAEL 152 95% UCL	LOAEL 113 Average	LOAEL 113 95% UCL	NOAEL 113 Average	NOAEL 113 95% UCL	LOAEL 90 Average	LOAEL 90 95% UCL	NOAEL 90 Average	NOAEL 90 95% UCL	LOAEL 50 Average	LOAEL 50 95% UCL	NOAEL 50 Average	NOAEL 50 95% UCL
1993	15	16	147	158	12	12	115	125	9	10	93	101	7.0	7.5	70	75
1994	13	14	131	142	11	12	108	117	9	10	88	95	6.5	7.1	65	71
1995	12	14	124	137	10	11	102	112	13	9.0	132	90	6.2	6.7	62	67
1996	12	14	125	136	10	11	99	108	13	8.6	129	86	5.9	6.5	59	65
1997	12	13	118	129	9.5	10	95	104	7.6	8.3	76	83	5.7	6.2	57	62
1998	11	12	111	123	9.1	10	91	101	7.2	8.0	72	80	5.4	5.9	54	59
1999	10	12	104	118	8.7	10	87	98	6.9	7.6	69	76	5.2	5.8	52	58
2000	10	12	104	118	8.4	9.5	84	95	6.7	7.4	67	74	5.0	5.6	50	56
2001	10	11	103	115	8.3	9.3	83	93	6.5	7.2	65	72	4.8	5.4	48	54
2002	10	11	98	111	8.0	9.0	80	90	6.3	7.1	63	71	4.7	5.3	47	53
2003	9.3	10	93	105	7.6	8.7	76	87	6.2	6.9	62	69	4.6	5.1	46	51
2004	9.0	10	90	104	7.3	8.4	73	84	5.9	6.6	59	66	4.4	4.9	44	49
2005	8.7	10	87	102	7.2	8.3	72	83	5.6	6.4	56	64	4.2	4.8	42	48
2006	8.5	10	85	97	7.0	8.1	70	81	5.4	6.1	54	61	4.0	4.6	40	46
2007	8.3	10	83	95	6.9	7.9	69	79	5.2	6.0	52	60	3.9	4.5	39	45
2008	8.0	9.3	80	93	6.6	7.7	66	77	5.1	5.8	51	58	3.8	4.4	38	44
2009	7.8	9.3	78	93	6.4	7.5	64	75	5.0	5.7	50	57	3.7	4.3	37	43
2010	7.7	9.0	77	90	6.3	7.3	63	73	4.9	5.6	49	56	3.6	4.2	36	42
2011	7.5	8.6	75	86	6.2	7.2	62	72	4.7	5.4	47	54	3.6	4.1	36	41
2012	7.2	8.3	72	83	6.1	7.0	61	70	4.6	5.3	46	53	3.5	4.0	35	40
2013	7.1	8.2	71	82	6.0	6.9	60	69	4.5	5.1	45	51	3.4	3.9	34	39
2014	7.0	8.1	70	81	5.8	6.7	58	67	4.4	5.0	44	50	3.3	3.8	33	38
2015	6.8	7.8	68	78	5.6	6.5	56	65	4.3	4.9	43	49	3.2	3.7	32	37
2016	6.7	8.0	67	80	5.4	6.4	54	64	4.2	4.8	42	48	3.1	3.6	31	36
2017	6.6	8.0	66	80	5.4	6.3	54	63	4.1	4.7	41	47	3.1	3.5	31	35
2018	6.5	7.7	65	77	5.3	6.3	53	63	4.0	4.6	40	46	3.0	3.5	30	35

Bold values indicate exceedances

TABLE 5-49: RATIO OF MODELED DIETARY DOSES TO TOXICITY BENCHMARKS FOR FEMALE MINK FOR TRI+ CONGENERS FOR THE PERIOD 1993 - 2018

Year	LOAEL 152 Average	LOAEL 152 95% UCL	NOAEL 152 Average	NOAEL 152 95% UCL	LOAEL 113 Average	LOAEL 113 95% UCL	NOAEL 113 Average	NOAEL 113 95% UCL	LOAEL 90 Average	LOAEL 90 95% UCL	NOAEL 90 Average	NOAEL 90 95% UCL	LOAEL 50 Average	LOAEL 50 95% UCL	NOAEL 50 Average	NOAEL 50 95% UCL
1993	**1.1**	**1.1**	**34**	**36**	0.7	0.8	**24**	**25**	0.6	0.6	**20**	**21**	0.5	0.5	**17**	**18**
1994	0.8	0.9	**27**	**28**	0.7	0.7	**22**	**23**	0.6	0.6	**18**	**19**	0.5	0.5	**15**	**16**
1995	0.8	0.8	**25**	**26**	0.6	0.6	**19**	**20**	0.5	0.5	**17**	**17**	0.4	0.5	**14**	**15**
1996	0.9	0.9	**28**	**29**	0.6	0.6	**20**	**21**	0.5	0.5	**16**	**16**	0.4	0.4	**13**	**14**
1997	0.8	0.8	**25**	**26**	0.6	0.6	**18**	**19**	0.4	0.5	**14**	**15**	0.4	0.4	**12**	**13**
1998	0.6	0.7	**20**	**21**	0.5	0.5	**16**	**17**	0.4	0.4	**14**	**14**	0.4	0.4	**11**	**12**
1999	0.6	0.6	**18**	**19**	0.5	0.5	**15**	**16**	0.4	0.4	**12**	**13**	0.3	0.3	**11**	**11**
2000	0.6	0.6	**18**	**19**	0.4	0.5	**14**	**15**	0.4	0.4	**12**	**12**	0.3	0.3	**10**	**10**
2001	0.6	0.6	**19**	**20**	0.4	0.5	**14**	**15**	0.3	0.4	**11**	**12**	0.3	0.3	**9.3**	**10**
2002	0.5	0.6	**18**	**18**	0.4	0.5	**14**	**15**	0.3	0.3	**11**	**11**	0.3	0.3	**8.9**	**9.4**
2003	0.5	0.5	**16**	**17**	0.4	0.4	**13**	**14**	0.3	0.3	**10**	**11**	0.3	0.3	**8.5**	**9.0**
2004	0.4	0.5	**14**	**15**	0.4	0.4	**12**	**12**	0.3	0.3	**10**	**10**	0.2	0.3	**8.0**	**8.4**
2005	0.4	0.5	**14**	**15**	0.3	0.4	**11**	**12**	0.3	0.3	**9.1**	**10**	0.2	0.2	**7.6**	**8.0**
2006	0.5	0.5	**15**	**16**	0.3	0.4	**11**	**12**	0.3	0.3	**8.6**	**9.1**	0.2	0.2	**7.1**	**7.5**
2007	0.4	0.4	**13**	**14**	0.3	0.4	**11**	**12**	0.3	0.3	**8.4**	**8.8**	0.2	0.2	**6.9**	**7.2**
2008	0.4	0.4	**13**	**13**	0.3	0.3	**10**	**11**	0.2	0.3	**8.1**	**8.6**	0.2	0.2	**6.6**	**7.0**
2009	0.4	0.4	**12**	**13**	0.3	0.3	**10**	**10**	0.2	0.3	**7.8**	**8.2**	0.2	0.2	**6.4**	**6.7**
2010	0.4	0.4	**12**	**13**	0.3	0.3	**10**	**10**	0.2	0.2	**7.5**	**7.9**	0.2	0.2	**6.1**	**6.5**
2011	0.4	0.4	**13**	**13**	0.3	0.3	**10**	**10**	0.2	0.2	**7.3**	**7.8**	0.2	0.2	**6.0**	**6.3**
2012	0.4	0.4	**12**	**13**	0.3	0.3	**10**	**10**	0.2	0.2	**7.3**	**7.7**	0.2	0.2	**5.9**	**6.2**
2013	0.4	0.4	**12**	**13**	0.3	0.3	**10**	**10**	0.2	0.2	**7.2**	**7.6**	0.2	0.2	**5.8**	**6.1**
2014	0.4	0.4	**12**	**13**	0.3	0.3	**9.3**	**10**	0.2	0.2	**6.9**	**7.3**	0.2	0.2	**5.6**	**5.9**
2015	0.3	0.4	**11**	**12**	0.3	0.3	**8.8**	**9.3**	0.2	0.2	**6.8**	**7.1**	0.2	0.2	**5.5**	**5.8**
2016	0.3	0.3	**10**	**11**	0.3	0.3	**8.3**	**8.8**	0.2	0.2	**6.5**	**6.9**	0.2	0.2	**5.3**	**5.6**
2017	0.3	0.3	**10**	**10**	0.2	0.3	**8.0**	**8.4**	0.2	0.2	**6.3**	**6.6**	0.2	0.2	**5.1**	**5.4**
2018	0.3	0.3	**10**	**10**	0.2	0.3	**7.8**	**8.3**	0.2	0.2	**6.1**	**6.5**	0.2	0.2	**5.0**	**5.3**

Bold values indicate exceedances

302835

TABLE 5-50: RATIO OF MODELED DIETARY DOSE TO TOXICITY BENCHMARKS FOR FEMALE OTTER FOR TRI+ CONGENERS FOR THE PERIOD 1993 - 2018

Year	LOAEL 152 Average	LOAEL 152 95% UCL	NOAEL 152 Average	NOAEL 152 95% UCL	LOAEL 113 Average	LOAEL 113 95% UCL	NOAEL 113 Average	NOAEL 113 95% UCL	LOAEL 90 Average	LOAEL 90 95% UCL	NOAEL 90 Average	NOAEL 90 95% UCL	LOAEL 50 Average	LOAEL 50 95% UCL	NOAEL 50 Average	NOAEL 50 95% UCL
1993	**14**	**14**	**458**	**468**	**9.4**	**10**	**306**	**313**	**2.2**	**2.2**	**70**	**71**	**2.0**	**2.1**	**66**	**67**
1994	**10**	**10**	**332**	**339**	**8.2**	**8.4**	**268**	**273**	**2.0**	**2.0**	**64**	**65**	**1.8**	**1.9**	**59**	**60**
1995	**8.8**	**8.9**	**285**	**291**	**7.3**	**7.5**	**238**	**243**	**1.8**	**1.8**	**58**	**59**	**1.6**	**1.7**	**53**	**54**
1996	**10**	**11**	**338**	**345**	**6.8**	**7.0**	**222**	**227**	**1.6**	**1.6**	**53**	**53**	**1.5**	**1.5**	**48**	**49**
1997	**9.5**	**10**	**307**	**314**	**6.6**	**6.7**	**214**	**219**	**1.5**	**1.5**	**49**	**50**	**1.4**	**1.4**	**45**	**45**
1998	**7.4**	**7.6**	**242**	**247**	**6.0**	**6.1**	**195**	**200**	**1.4**	**1.4**	**45**	**46**	**1.3**	**1.3**	**41**	**42**
1999	**6.4**	**6.6**	**209**	**213**	**5.0**	**5.1**	**163**	**167**	**1.3**	**1.3**	**41**	**41**	**1.1**	**1.2**	**37**	**38**
2000	**6.0**	**6.2**	**196**	**200**	**4.6**	**4.7**	**149**	**152**	**1.1**	**1.2**	**37**	**38**	**1.1**	**1.1**	**34**	**35**
2001	**6.8**	**7.0**	**223**	**227**	**4.6**	**4.7**	**151**	**154**	**1.1**	**1.1**	**35**	**36**	**1.0**	**1.0**	**32**	**32**
2002	**6.3**	**6.4**	**204**	**208**	**4.6**	**4.7**	**149**	**152**	**1.1**	**1.1**	**34**	**35**	0.9	**1.0**	**30**	**31**
2003	**5.6**	**5.7**	**181**	**185**	**4.2**	**4.3**	**137**	**140**	**1.0**	**1.0**	**32**	**33**	0.9	0.9	**29**	**29**
2004	**4.5**	**4.6**	**146**	**150**	**3.7**	**3.8**	**121**	**124**	0.9	0.9	**30**	**31**	0.8	0.8	**27**	**27**
2005	**4.3**	**4.4**	**139**	**143**	**3.4**	**3.5**	**112**	**114**	0.9	0.9	**28**	**28**	0.8	0.8	**25**	**26**
2006	**4.8**	**4.9**	**157**	**161**	**3.4**	**3.5**	**110**	**113**	0.8	0.8	**26**	**27**	0.7	0.7	**24**	**24**
2007	**4.5**	**4.6**	**145**	**148**	**3.3**	**3.4**	**108**	**110**	0.8	0.8	**25**	**26**	0.7	0.7	**22**	**23**
2008	**4.1**	**4.2**	**135**	**138**	**3.2**	**3.3**	**104**	**106**	0.8	0.8	**25**	**25**	0.7	0.7	**22**	**22**
2009	**3.6**	**3.7**	**117**	**120**	**2.9**	**3.0**	**95**	**97**	0.7	0.7	**23**	**24**	0.6	0.6	**20**	**21**
2010	**3.8**	**3.9**	**123**	**125**	**2.8**	**2.9**	**91**	**93**	0.7	0.7	**22**	**22**	0.6	0.6	**19**	**20**
2011	**4.2**	**4.3**	**137**	**141**	**3.0**	**3.0**	**96**	**99**	0.7	0.7	**22**	**22**	0.6	0.6	**19**	**19**
2012	**3.8**	**3.9**	**123**	**126**	**2.9**	**3.0**	**94**	**96**	0.7	0.7	**22**	**22**	0.6	0.6	**18**	**19**
2013	**4.1**	**4.2**	**134**	**137**	**3.0**	**3.1**	**97**	**99**	0.7	0.7	**22**	**23**	0.6	0.6	**19**	**19**
2014	**3.8**	**3.8**	**122**	**125**	**2.8**	**2.9**	**92**	**94**	0.6	0.7	**21**	**21**	0.5	0.6	**18**	**18**
2015	**3.5**	**3.5**	**113**	**115**	**2.7**	**2.8**	**88**	**90**	0.6	0.6	**20**	**21**	0.5	0.5	**17**	**18**
2016	**3.2**	**3.2**	**103**	**105**	**2.5**	**2.6**	**82**	**84**	0.6	0.6	**19**	**20**	0.5	0.5	**17**	**17**
2017	**2.9**	**2.9**	**93**	**95**	**2.3**	**2.4**	**75**	**77**	0.6	0.6	**18**	**19**	0.5	0.5	**16**	**17**
2018	**2.8**	**2.8**	**90**	**93**	**2.2**	**2.3**	**72**	**73**	0.5	0.5	**17**	**18**	0.5	0.5	**15**	**16**

Bold values indicate exceedances

302836

TABLE 5-51: RATIO OF MODELED DIETARY DOSES TO TOXICITY BENCHMARKS FOR FEMALE MINK ON A TEQ BASIS FOR THE PERIOD 1993 - 2018

Year	LOAEL 152 Average	LOAEL 152 95% UCL	NOAEL 152 Average	NOAEL 152 95% UCL	LOAEL 113 Average	LOAEL 113 95% UCL	NOAEL 113 Average	NOAEL 113 95% UCL	LOAEL 90 Average	LOAEL 90 95% UCL	NOAEL 90 Average	NOAEL 90 95% UCL	LOAEL 50 Average	LOAEL 50 95% UCL	NOAEL 50 Average	NOAEL 50 95% UCL
1993	7.1	7.4	199	207	5.1	5.3	142	148	4.1	4.3	114	119	3.5	3.7	99	103
1994	5.7	5.9	158	166	4.6	4.9	130	136	3.8	3.9	105	110	3.2	3.3	90	94
1995	5.2	5.4	145	153	4.0	4.3	113	119	3.7	3.6	104	99	2.9	3.0	82	85
1996	5.8	6.1	163	171	4.1	4.3	114	120	3.5	3.3	99	94	2.7	2.8	76	80
1997	5.1	5.4	143	150	3.8	4.0	107	113	3.0	3.2	84	89	2.5	2.7	71	74
1998	4.2	4.4	118	124	3.4	3.6	95	100	2.8	3.0	79	83	2.4	2.5	66	70
1999	3.8	4.1	107	114	3.1	3.3	87	93	2.5	2.7	71	75	2.2	2.3	61	64
2000	3.8	4.0	106	112	2.9	3.1	82	87	2.4	2.5	67	71	2.0	2.1	57	60
2001	4.0	4.2	111	117	3.0	3.2	83	88	2.3	2.4	64	68	1.9	2.0	54	57
2002	3.6	3.9	102	108	2.9	3.1	81	86	2.2	2.4	63	66	1.9	2.0	52	55
2003	3.4	3.6	95	100	2.7	2.9	75	80	2.1	2.3	60	64	1.8	1.9	50	52
2004	2.9	3.2	82	88	2.4	2.6	68	73	2.0	2.1	56	60	1.7	1.8	47	49
2005	2.9	3.1	80	86	2.3	2.5	66	70	1.9	2.0	53	56	1.6	1.7	44	47
2006	3.1	3.3	87	92	2.3	2.5	65	70	1.8	1.9	50	54	1.5	1.6	41	44
2007	2.8	3.0	77	83	2.3	2.4	63	68	1.7	1.9	49	52	1.4	1.5	40	42
2008	2.6	2.8	73	79	2.2	2.3	60	65	1.7	1.8	47	50	1.4	1.5	38	41
2009	2.5	2.7	69	75	2.0	2.2	57	61	1.6	1.7	45	48	1.3	1.4	37	39
2010	2.6	2.7	72	77	2.0	2.1	56	60	1.5	1.7	43	46	1.3	1.4	36	38
2011	2.6	2.8	74	78	2.0	2.2	57	61	1.5	1.6	43	46	1.2	1.3	35	37
2012	2.5	2.7	70	75	2.0	2.1	56	60	1.5	1.6	42	45	1.2	1.3	34	37
2013	2.6	2.7	72	77	2.0	2.1	56	60	1.5	1.6	42	44	1.2	1.3	34	36
2014	2.5	2.6	69	74	1.9	2.1	54	58	1.4	1.5	40	43	1.2	1.2	33	35
2015	2.3	2.4	63	68	1.8	2.0	51	55	1.4	1.5	39	42	1.1	1.2	32	34
2016	2.1	2.3	59	64	1.7	1.8	48	52	1.4	1.4	38	41	1.1	1.2	31	33
2017	2.0	2.2	57	62	1.6	1.8	46	50	1.3	1.4	36	39	1.1	1.1	30	32
2018	2.0	2.2	57	61	1.6	1.8	45	49	1.3	1.4	35	38	1.0	1.1	29	31

Bold values indicate exceedances

302837

TABLE 5-52: RATIO OF MODELED DIETARY DOSES TO TOXICITY BENCHMARKS FOR FEMALE OTTER ON A TEQ BASIS FOR THE PERIOD 1993 - 2018

Year	LOAEL 152 Average	LOAEL 152 95% UCL	NOAEL 152 Average	NOAEL 152 95% UCL	LOAEL 113 Average	LOAEL 113 95% UCL	NOAEL 113 Average	NOAEL 113 95% UCL	LOAEL 90 Average	LOAEL 90 95% UCL	NOAEL 90 Average	NOAEL 90 95% UCL	LOAEL 50 Average	LOAEL 50 95% UCL	NOAEL 50 Average	NOAEL 50 95% UCL
1993	95	98	2674	2732	64	65	1789	1828	15	15	412	419	14	14	388	394
1994	69	71	1938	1981	56	57	1564	1598	13	14	375	382	12	13	347	353
1995	59	61	1664	1700	50	51	1391	1421	12	12	345	344	11	11	313	318
1996	71	72	1975	2018	46	47	1297	1326	11	11	315	314	10	10	283	288
1997	64	66	1796	1835	45	46	1253	1280	10	10	288	293	9.3	9.5	261	266
1998	50	52	1412	1443	41	42	1141	1167	9.5	10	266	270	8.6	8.7	240	244
1999	44	45	1220	1247	34	35	955	977	8.5	8.7	239	244	7.8	7.9	219	222
2000	41	42	1145	1171	31	32	869	890	7.8	7.9	217	221	7.2	7.3	200	204
2001	46	47	1300	1330	31	32	880	900	7.4	7.5	206	210	6.7	6.8	187	191
2002	42	43	1189	1216	31	32	871	891	7.2	7.3	200	204	6.4	6.5	179	182
2003	38	39	1060	1085	29	29	800	819	6.8	6.9	190	194	6.0	6.2	169	173
2004	31	31	856	876	25	26	710	727	6.3	6.4	177	180	5.7	5.8	159	162
2005	29	30	815	835	23	24	653	669	5.8	6.0	163	167	5.3	5.4	147	150
2006	33	34	918	939	23	24	645	660	5.5	5.6	155	158	4.9	5.0	139	141
2007	30	31	847	866	23	23	631	646	5.3	5.4	149	153	4.7	4.8	132	135
2008	28	29	787	805	22	22	607	621	5.1	5.3	144	147	4.5	4.6	126	129
2009	24	25	684	700	20	20	555	568	4.9	5.0	136	139	4.3	4.4	120	123
2010	26	26	718	735	19	19	530	543	4.6	4.7	129	132	4.1	4.2	114	117
2011	29	29	804	823	20	21	564	577	4.6	4.7	128	131	4.0	4.0	111	113
2012	26	26	719	735	20	20	551	564	4.5	4.6	127	130	3.9	4.0	109	111
2013	28	29	784	802	20	21	568	582	4.7	4.8	130	133	4.0	4.0	111	113
2014	25	26	713	730	19	20	539	552	4.4	4.5	123	126	3.7	3.8	105	107
2015	24	24	659	675	18	19	514	526	4.3	4.4	119	122	3.6	3.7	102	104
2016	22	22	602	617	17	18	479	490	4.1	4.2	114	117	3.5	3.6	99	101
2017	19	20	543	556	16	16	441	452	3.9	4.0	108	111	3.4	3.5	96	98
2018	19	19	529	542	15	15	419	429	3.7	3.8	103	105	3.2	3.3	91	93

Bold values indicate exceedances

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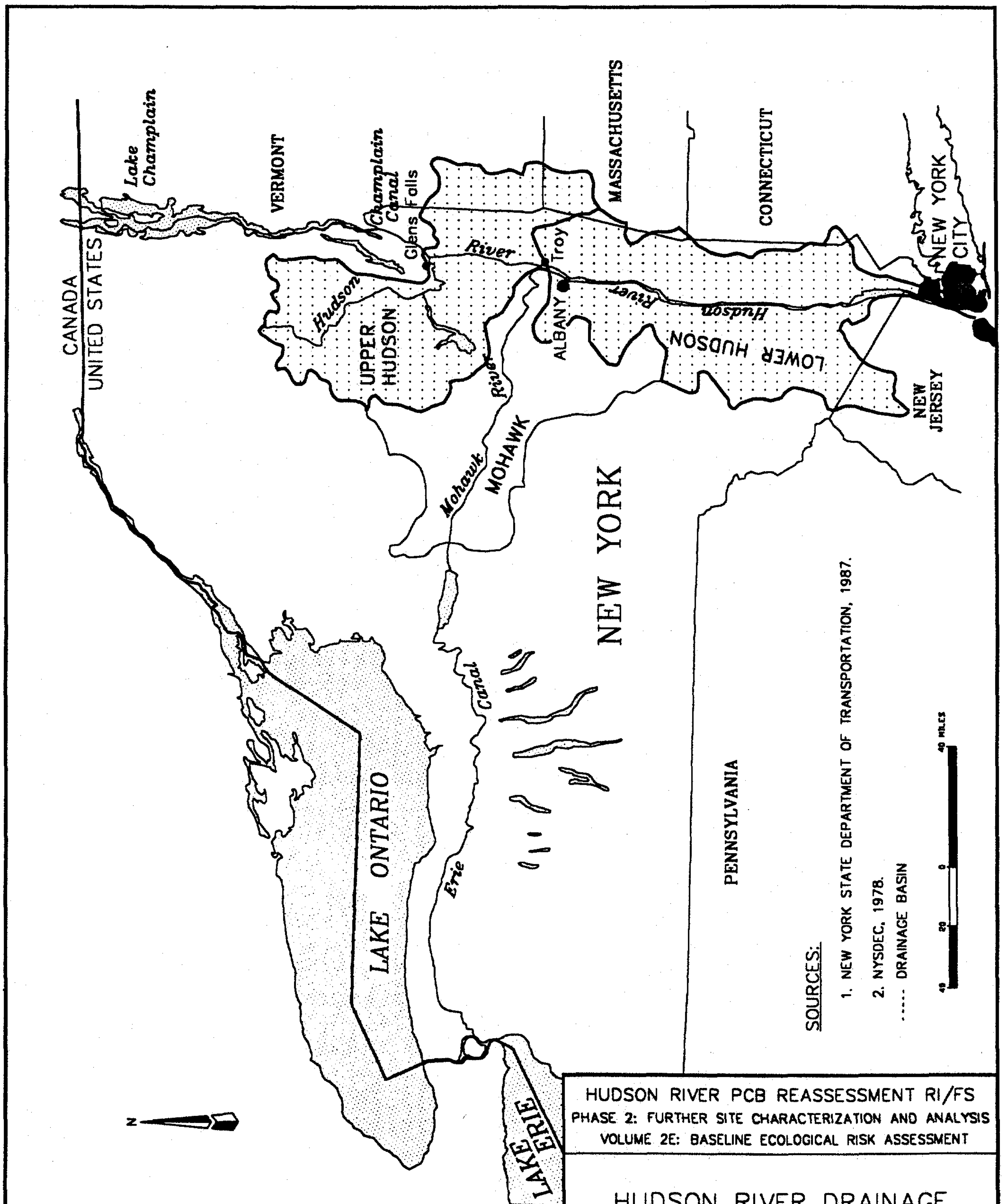
Figures

File No.: 48 File Name: U:\Project33\Hud-cadd\Reports\Ecosow\Fig1.1.dwg Date: 3-8-99 User: CM

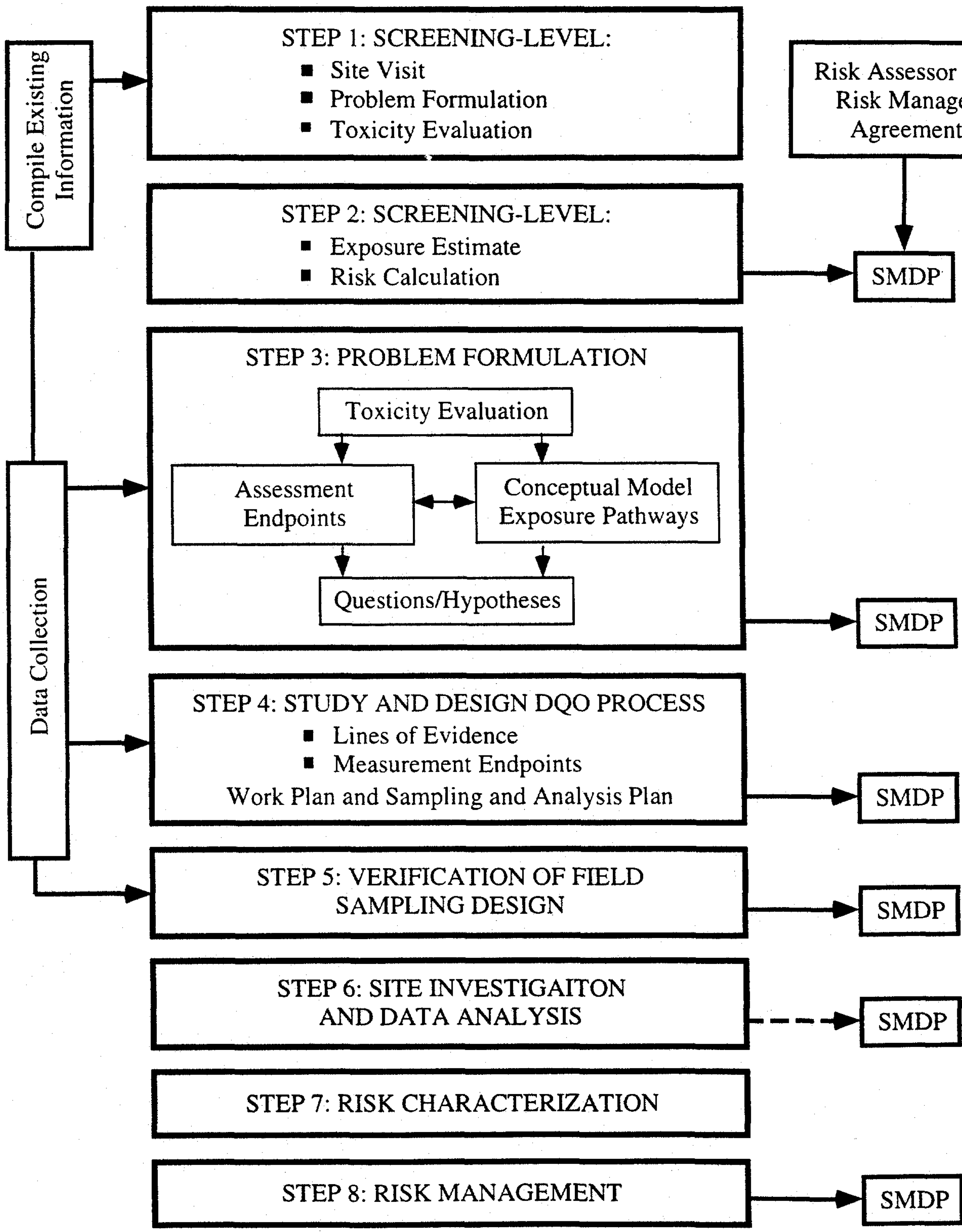


HUDSON RIVER PCB REASSESSMENT RI/FS
PHASE 2: FURTHER SITE CHARACTERIZATION AND ANALYSIS
VOLUME 2E: BASELINE ECOLOGICAL RISK ASSESSMENT

HUDSON RIVER DRAINAGE BASIN AND SITE LOCATION MAP

TAMS/MCA | FIGURE 1-1

Figure 1-2
Eight-Step Ecological Risk Assessment Process for Superfund
Hudson River PCB Reassessment
Ecological Risk Assessment



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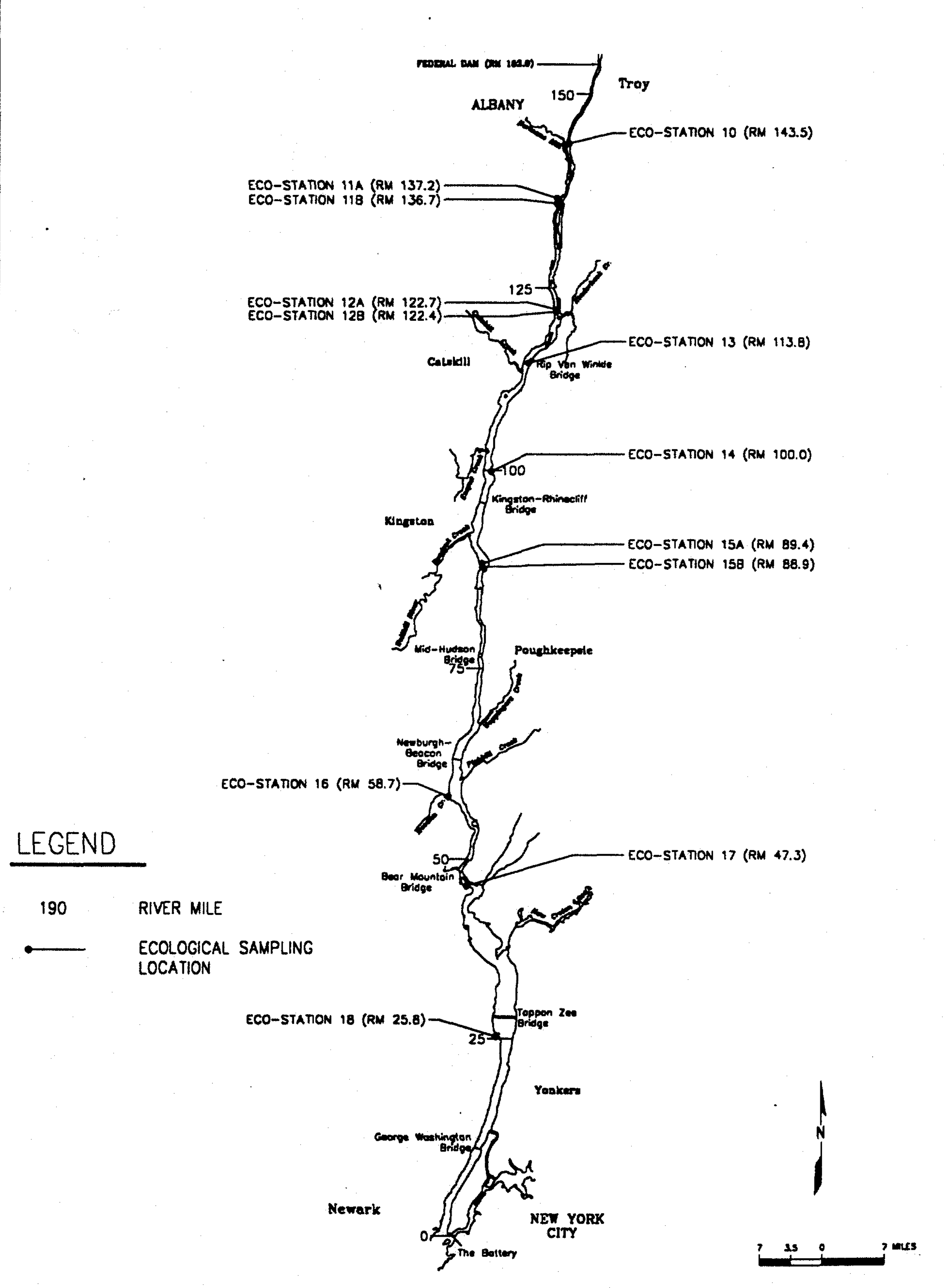
Filename: U:\PROJECT.33\HUD-CADD\ECOSOW\FIG2-2.dwg / Date: 3-8-99 / User: CM

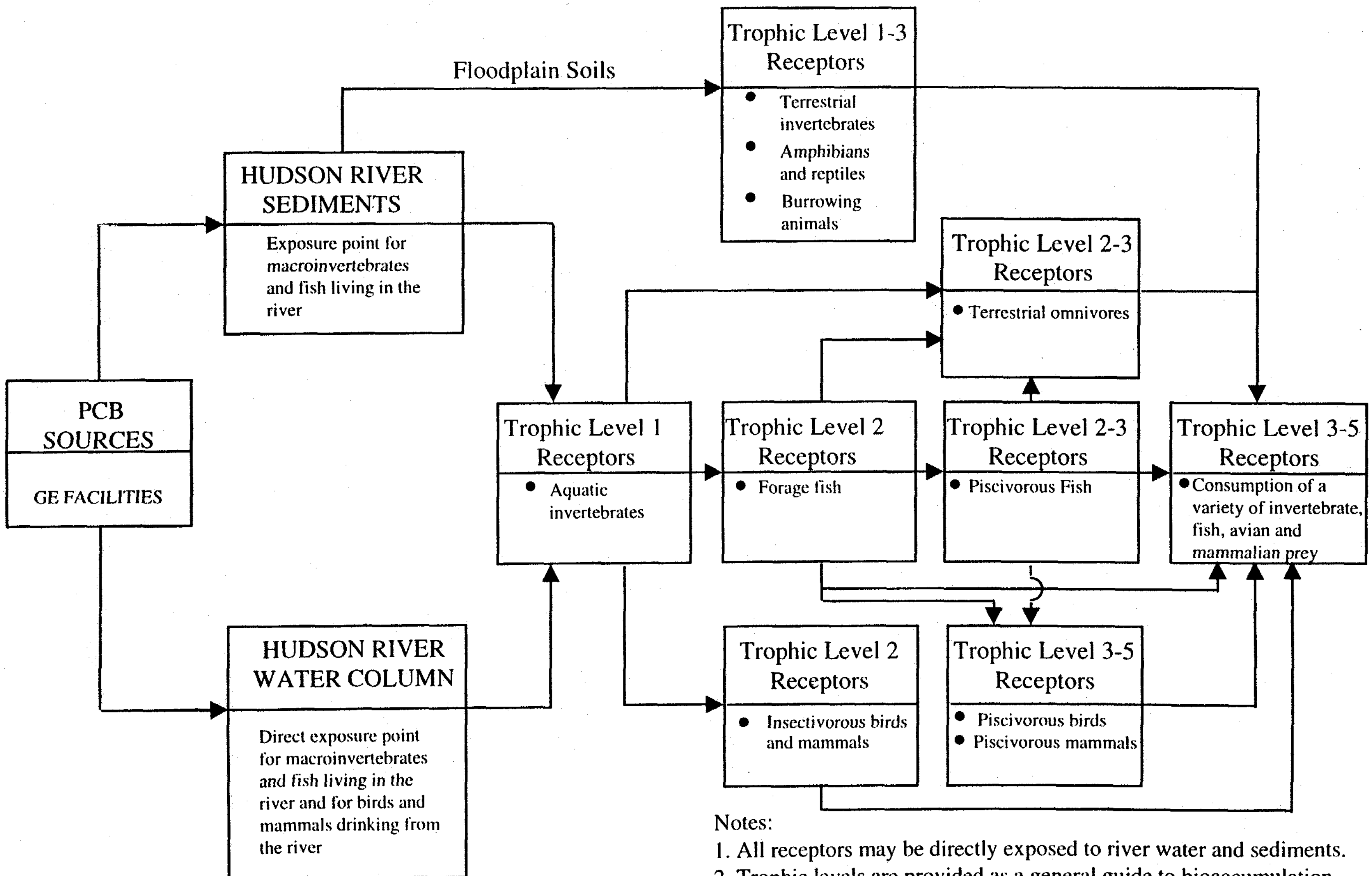
Figure 2-1
Baseline Ecological Risk Assessment
Lower Hudson River Sampling Stations



SOURCE: Reassessment Database

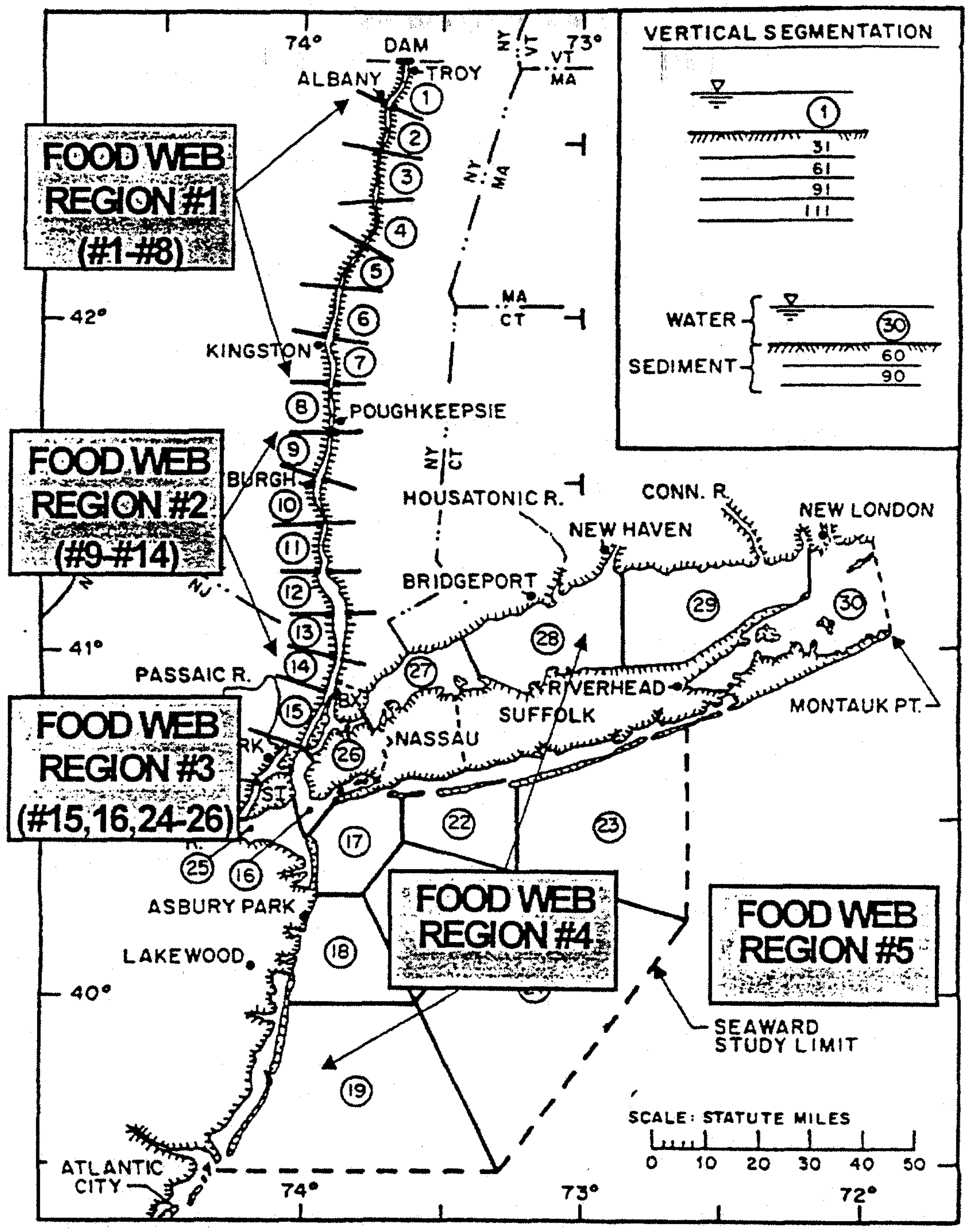
TAMS / MCA

Figure 2-2
Hudson River PCB Reassessment
Conceptual Model Diagram Including Floodplain Soils



Notes:
1. All receptors may be directly exposed to river water and sediments.
2. Trophic levels are provided as a general guide to bioaccumulation potential, but vary according to species and food availability.

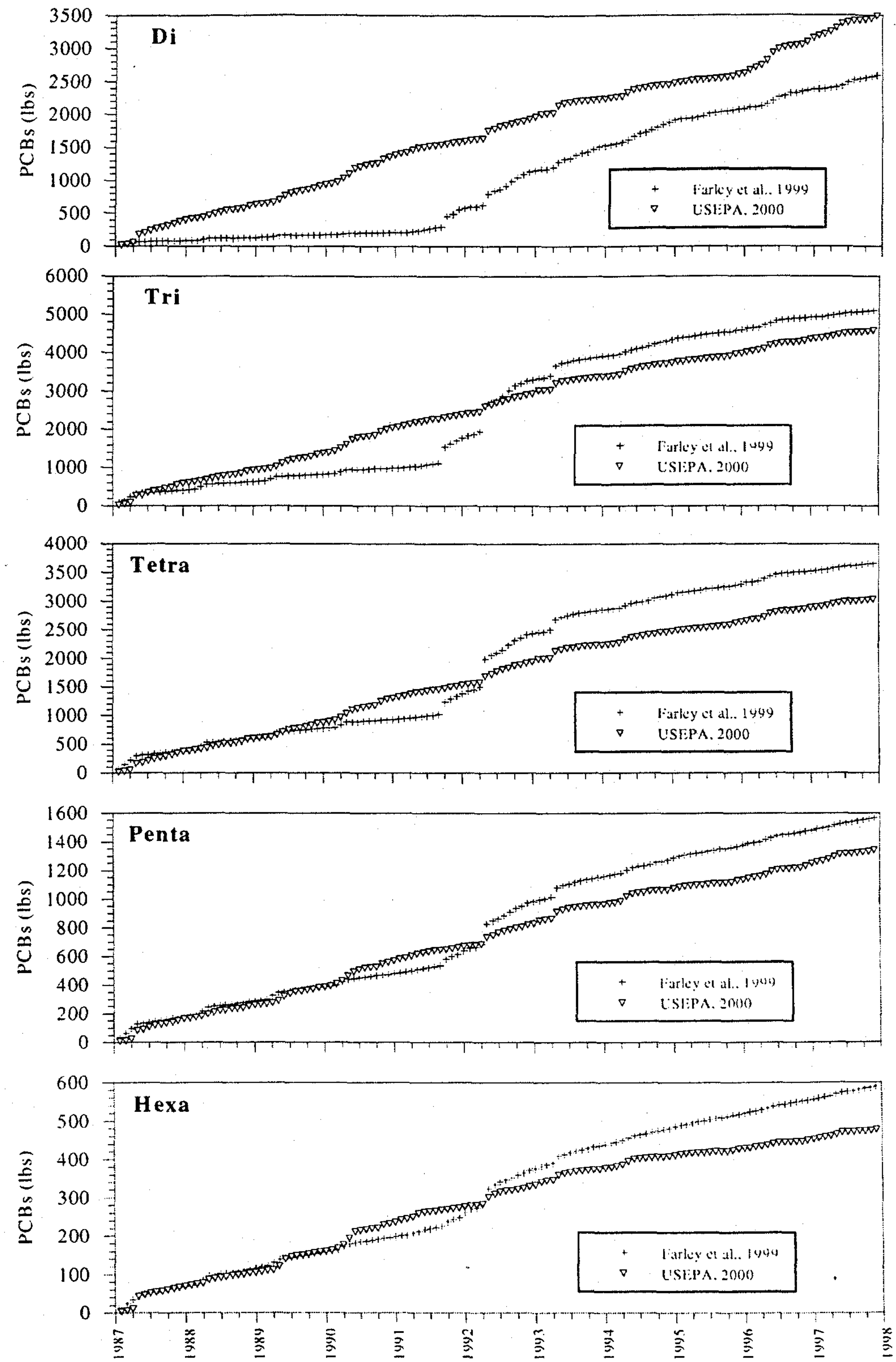
302844



Source: Farley et al., 1999

Note: Model segment numbers 1-30 pertain to the Fate and transport model. Model segments are combined into five food web regions for the bioaccumulation model calculations

Figure 3-1
Revised Segments and Regions of the Farley Model for PCBs in Hudson River Estuary and Surrounding Waters

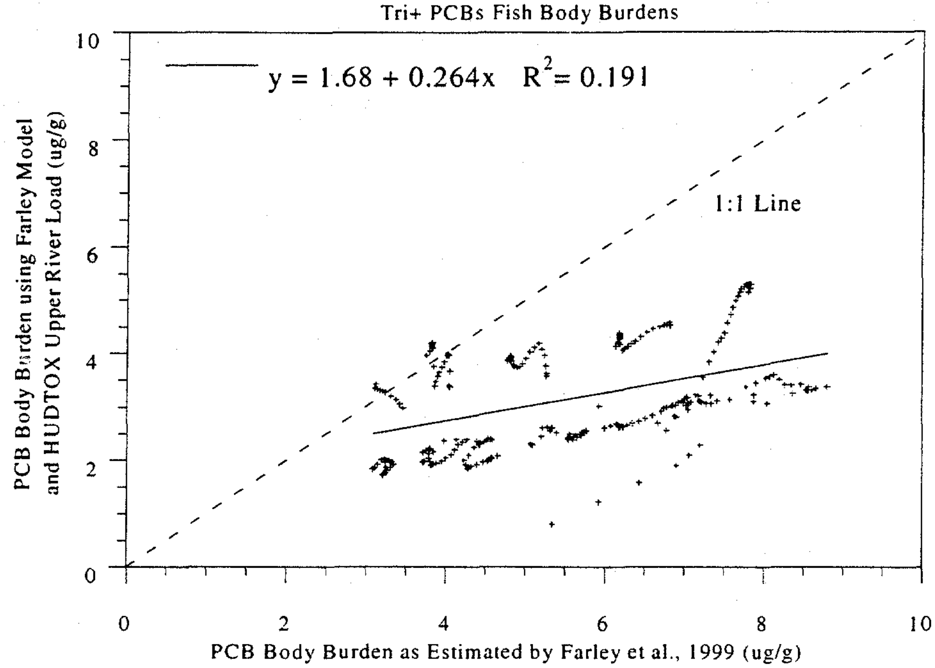


Sources: Farley et al., 1999 and USEPA, 2000

Figure 3-2
Comparison of Cumulative PCB Loads at WaterFord from Farley et al., 1999 and USEPA, 2000

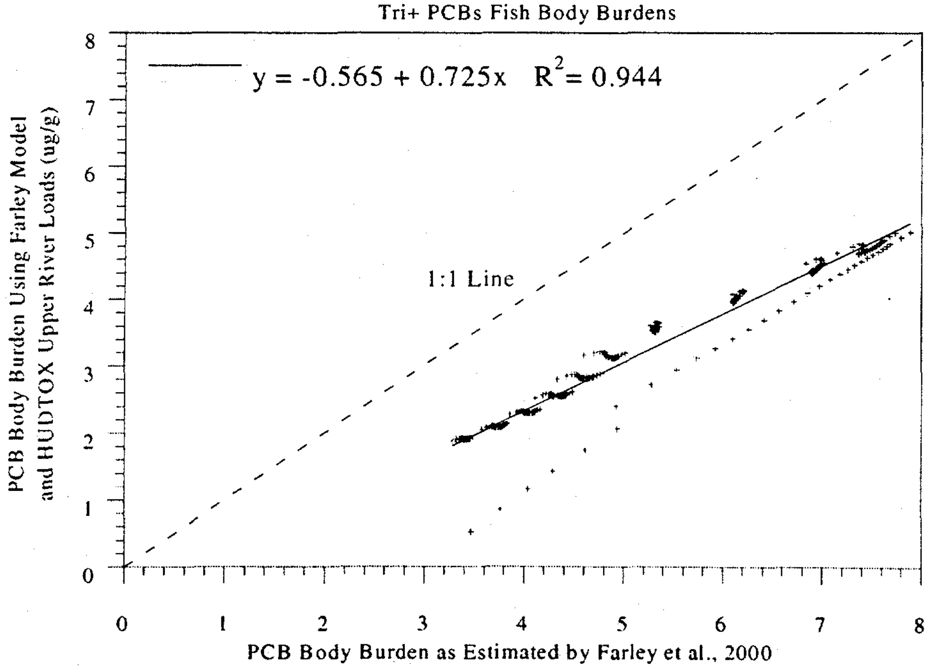
Region 1-White Perch (Age Class 1-7 Years)

Tri+ PCBs Fish Body Burdens



Region 2-White Perch (Age Class 1-7 Years)

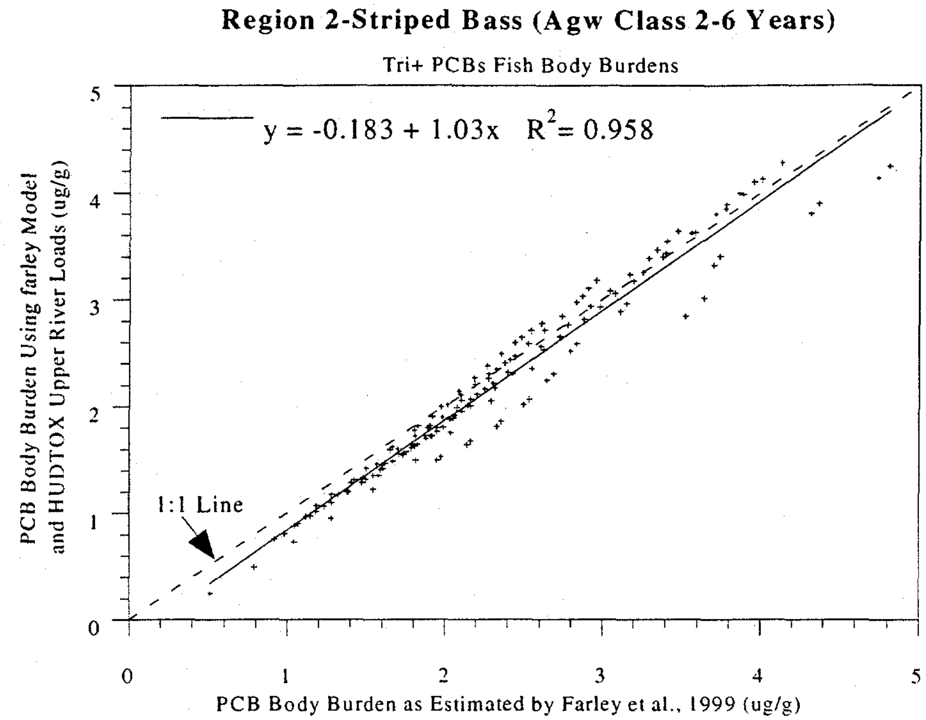
Tri+ PCBs Fish Body Burdens

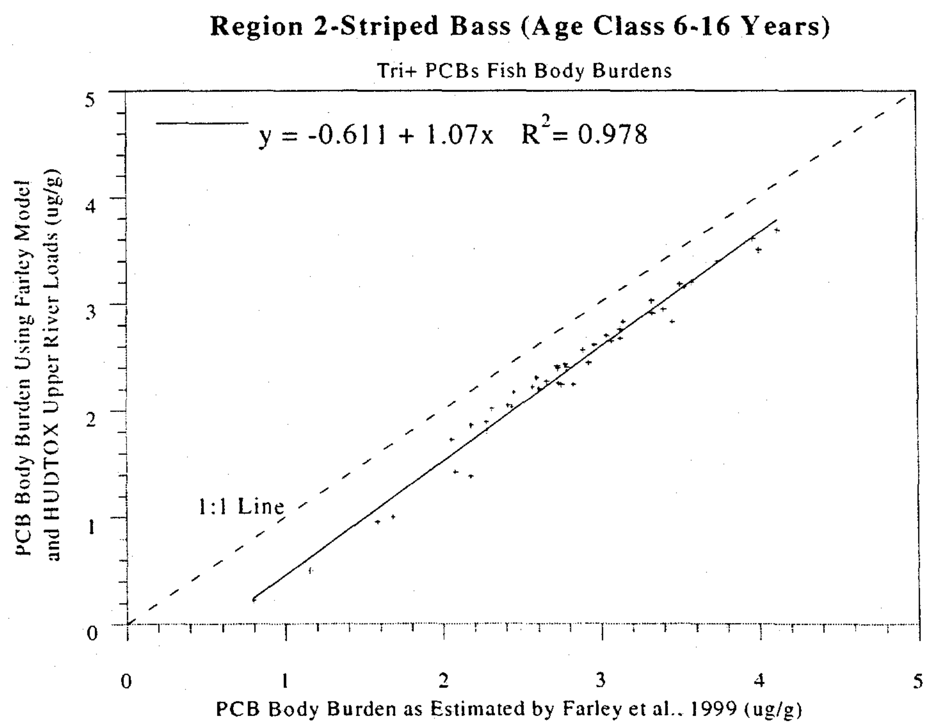


Sources: Farley et al., 1999 and USEPA, 2000

Figure 3-3
Comparison Between the White Perch Body Burdens Using the March, 1999 Model and the Farley Model Run with HUDTOX Upper River Loads (1987-1997)

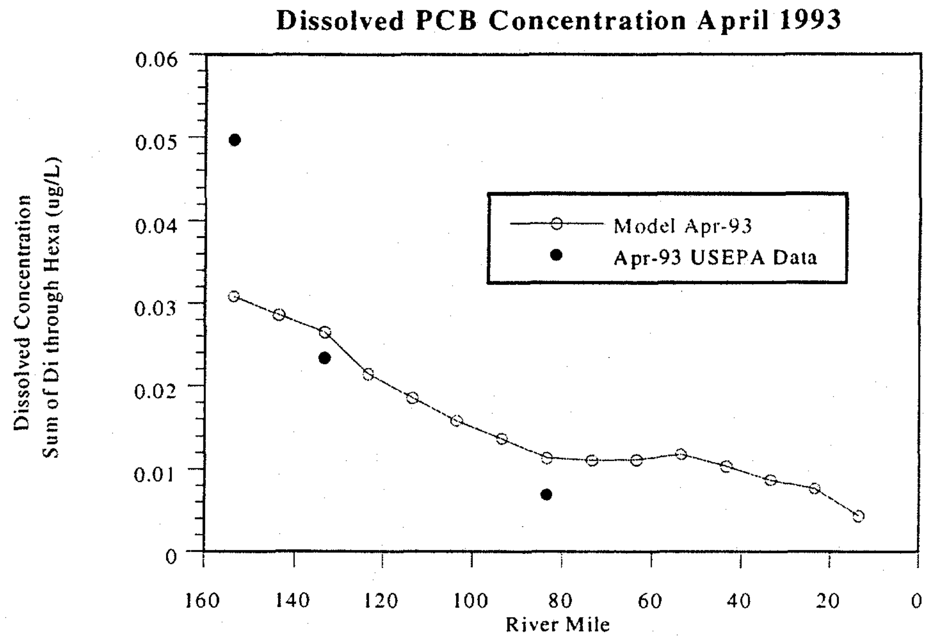
TAMS/MCA

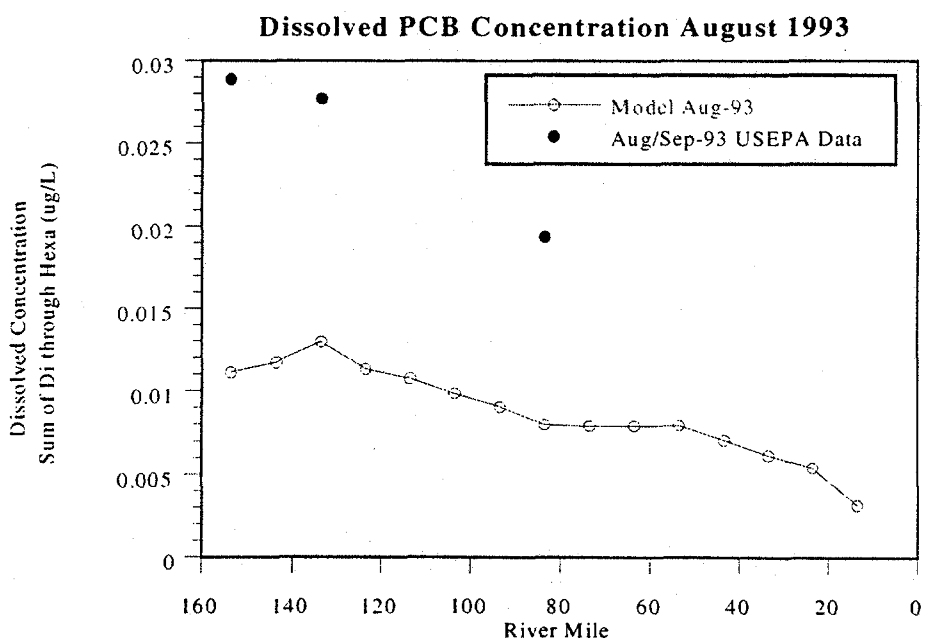




Sources: Farley et al., 1999 and USEPA, 2000

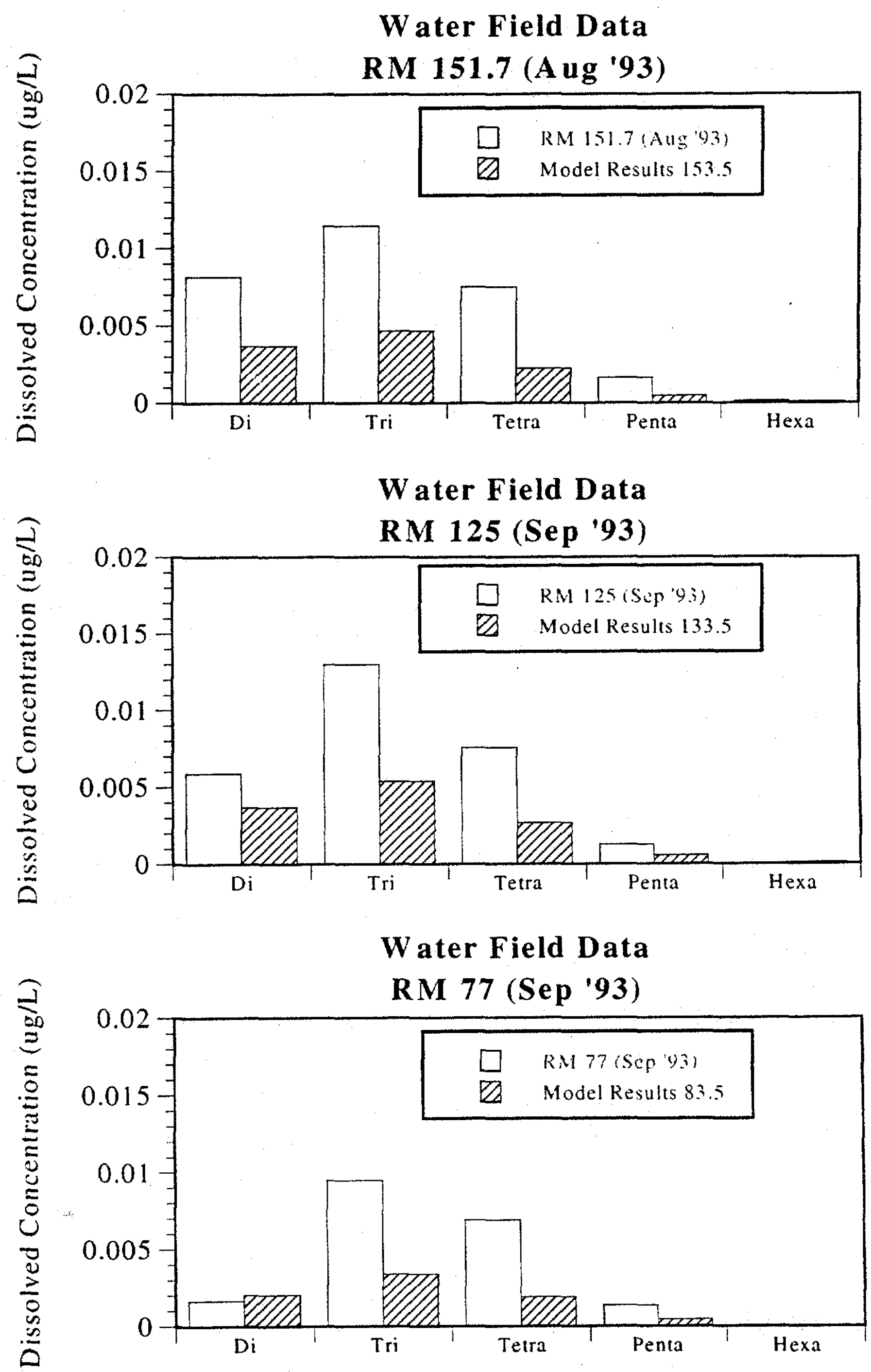
Figure 3-4
Comparison Between the Striped Bass Body Burdens Using the March, 1999 Model and the Farley Model Run with HUDTOX Upper River Loads





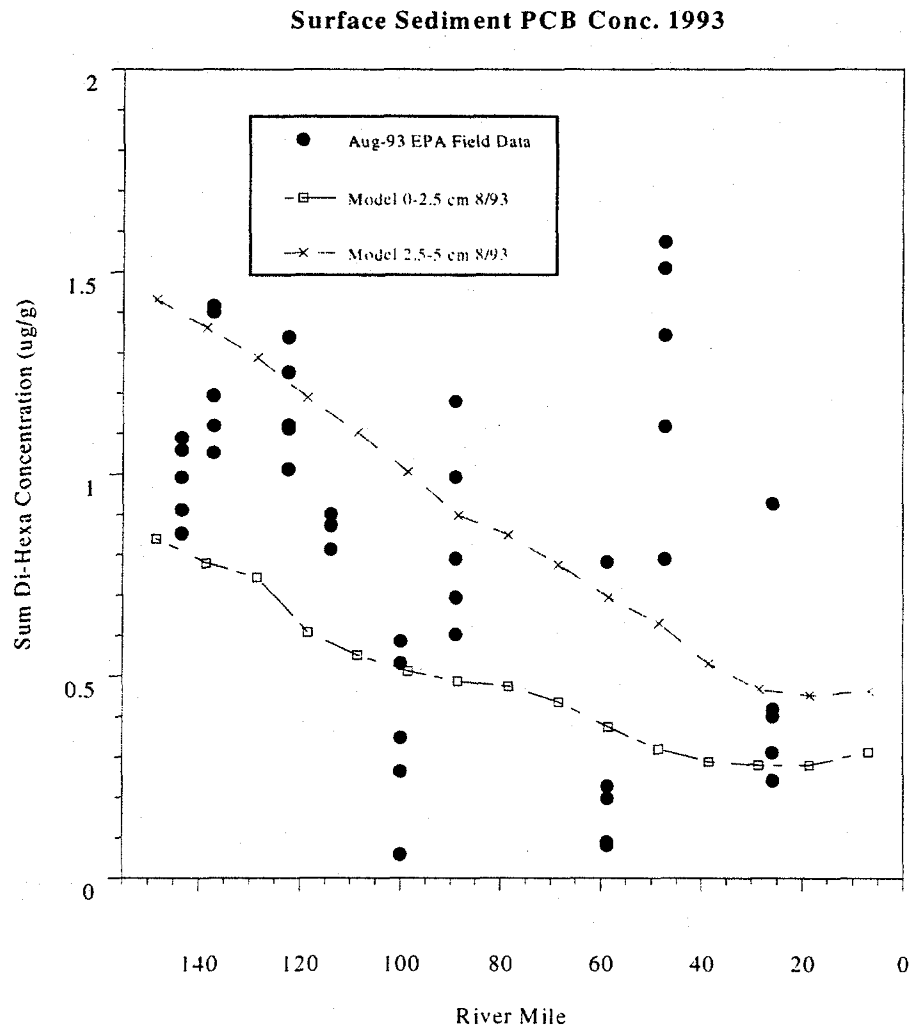
Sources: Farley et al., 1999, Hudson River Database Release 4.1 and USEPA, 2000

Figure 3-5
Comparison Between Field Data and Model Estimates for 1993 Dissolved PCB Concentrations (Farley Model with HUDTOX Upper River Loads)



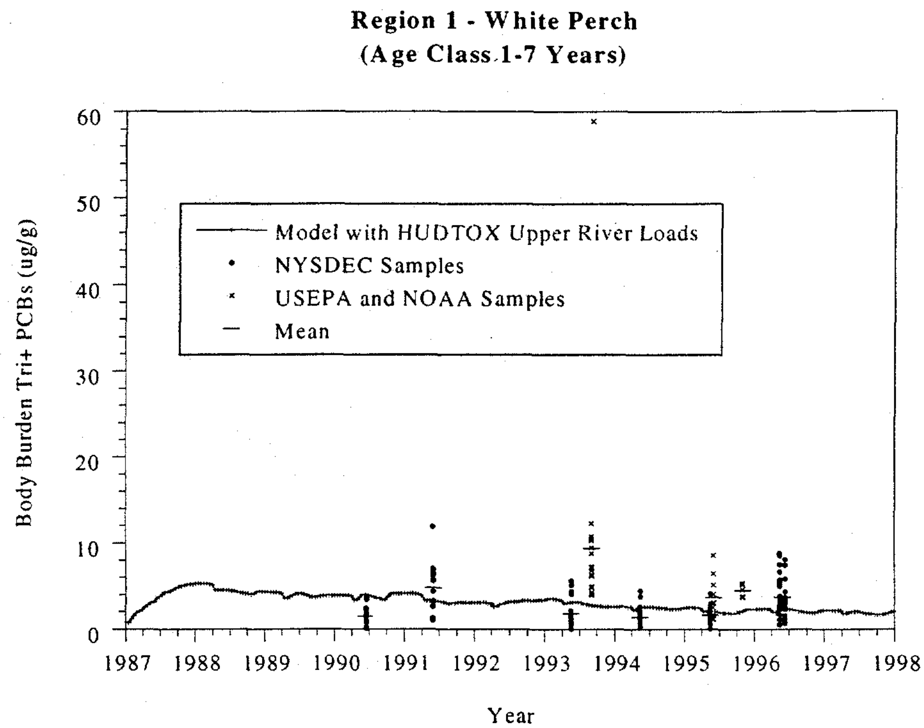
Sources: Farley et al., 1999, Hudson River Database Release 4.1 and USEPA, 2000

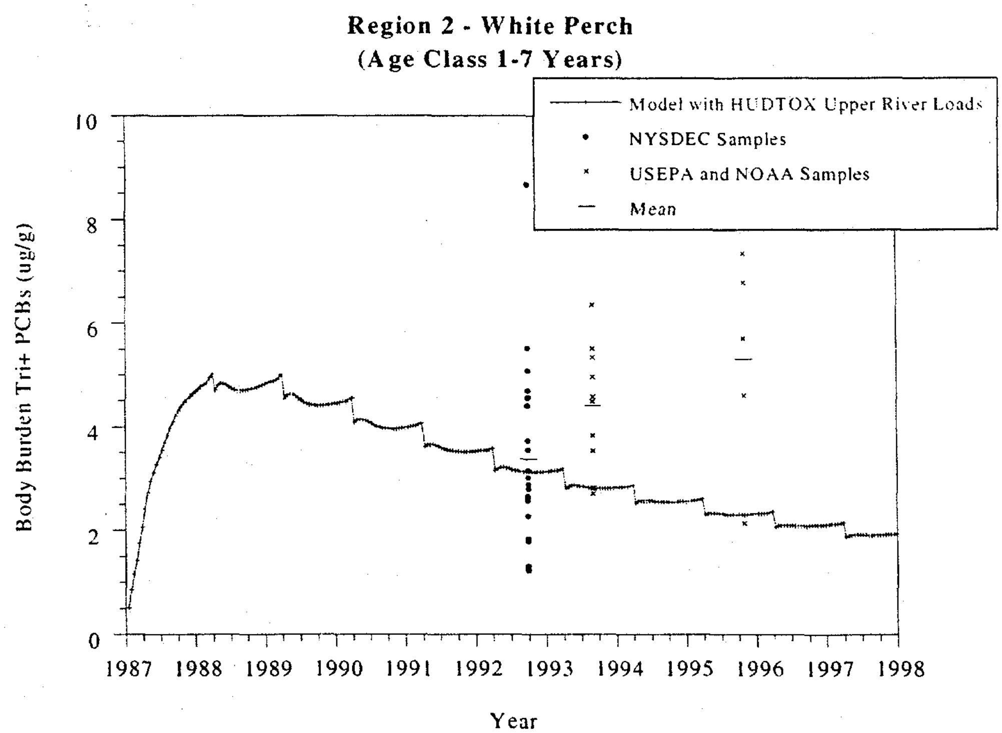
Figure 3-6
Comparison of Model and Measured Homologue Pattern for 1993 Dissolved Phase PCB Concentrations



Sources: Farley et al., 1999, Hudson River Database Release 4.1 and USEPA, 2000

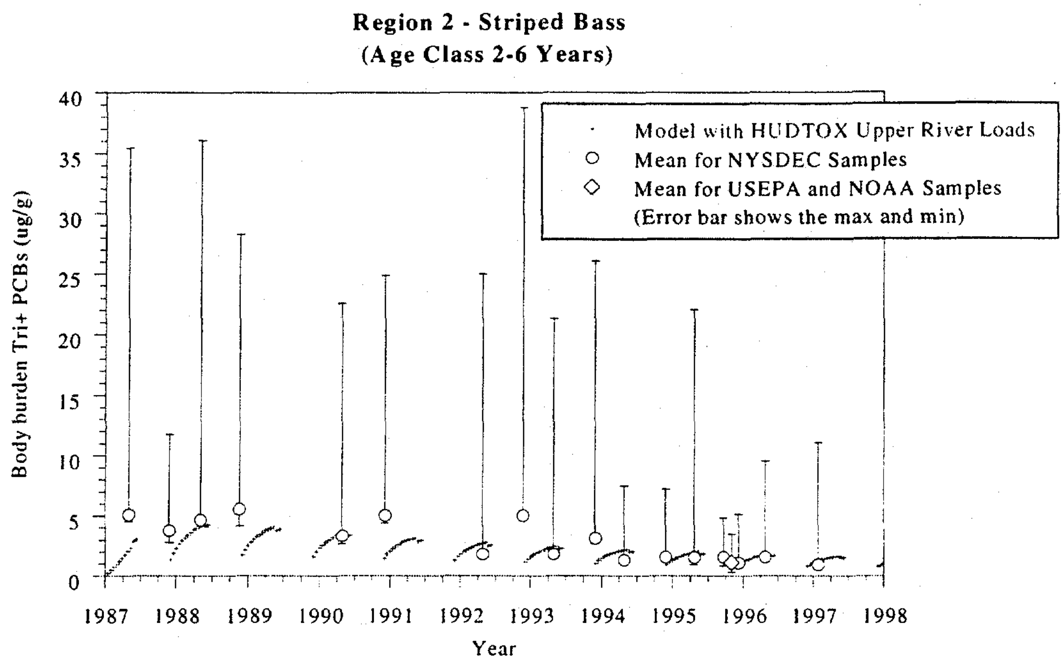
Figure 3-7
Comparison of Model and Measured PCB Surface Sediment Concentration for 1993

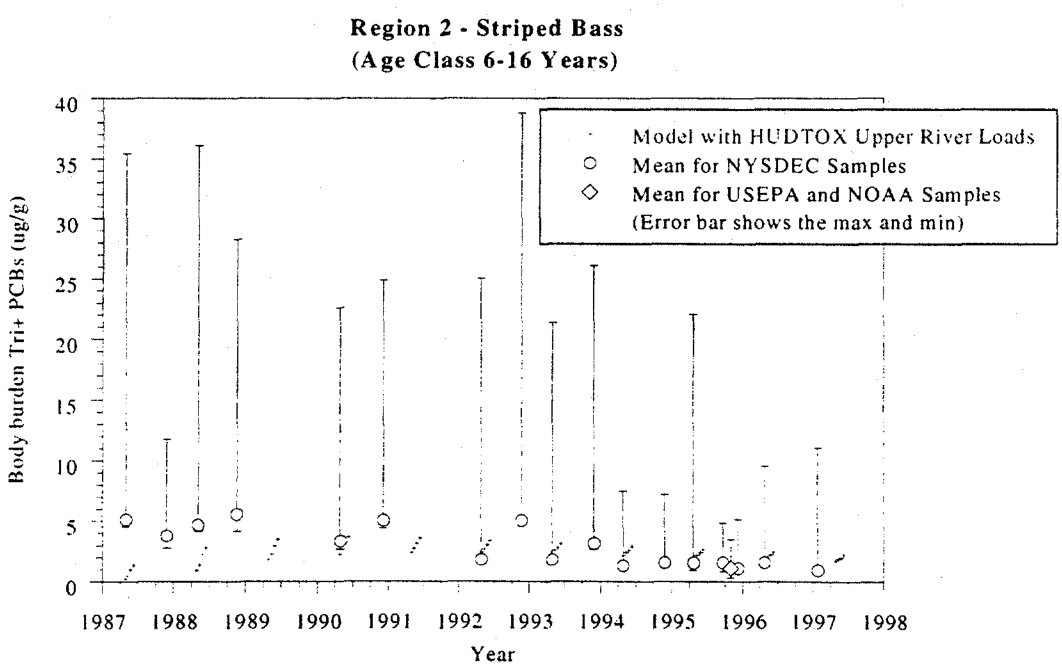




Sources: Farley et al., 1999, Hudson River Database Release 4.1 and USEPA, 2000

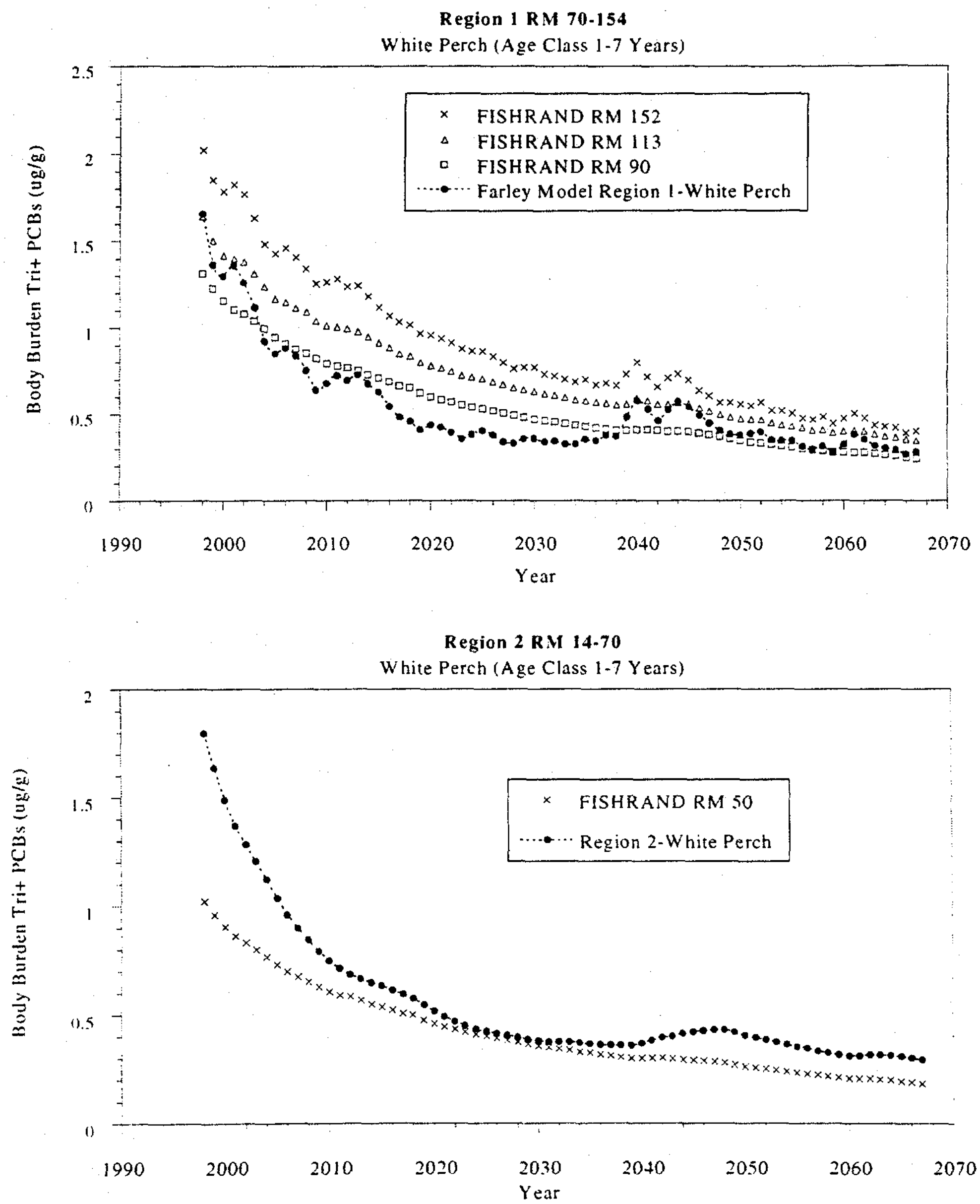
Figure 3-8
Comparison Between Model and Measured White Perch Body Burdens
NYSDEC Fish Samples *vs.* Farley Model with HUDTOX Upper River Loads





Sources: Farley et al., 1999, Hudson River Database Release 4.1 and USEPA, 2000

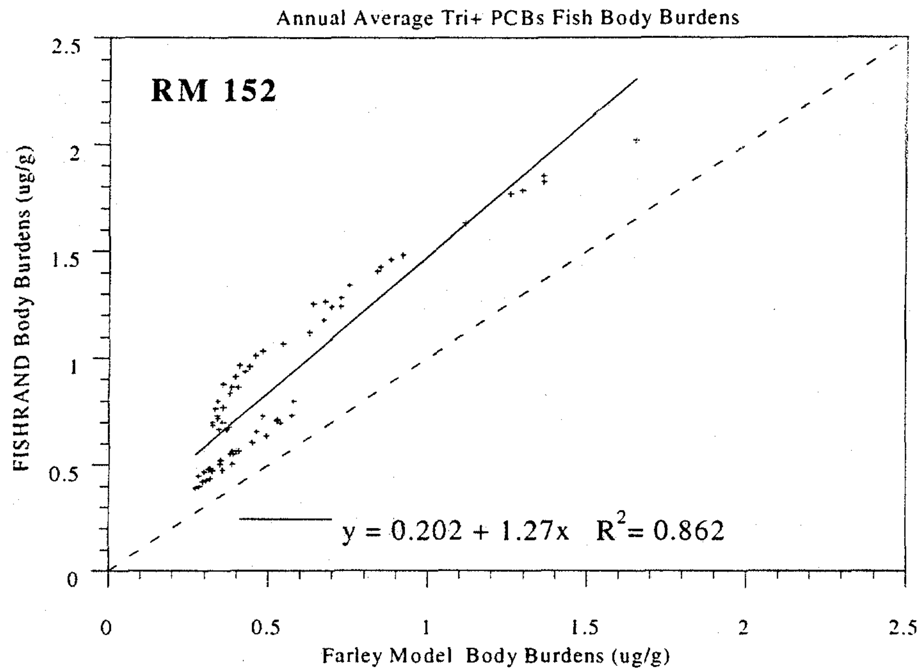
Figure 3-9
Comparison Between Model and Measured Striped Bass Body Burdens
NYSDEC Fish Samples *vs.* Farley Model with HUDTOX Upper River Loads



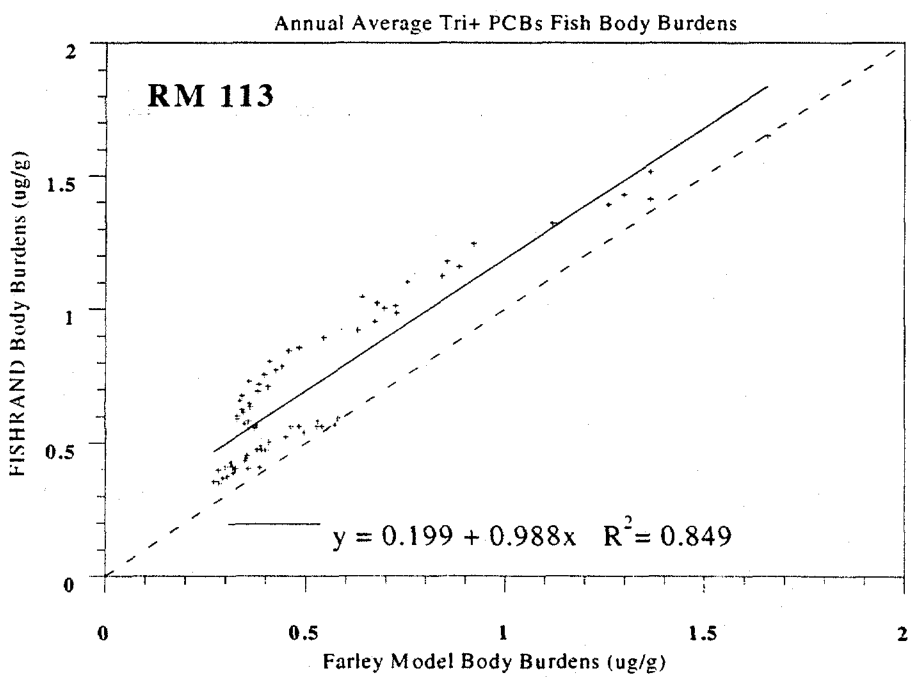
Sources: Farley et al., 1999, Hudson River Database Release 4.1 and USEPA, 2000

Figure 3-10
Comparison of Model Estimates for White Perch Body Burdens
Farley Model with HUDTOX Upper River Loads vs. FISHRAND in Food Web Regions 1 and 2

Region 1-White Perch (Age Class 1-7 Years)



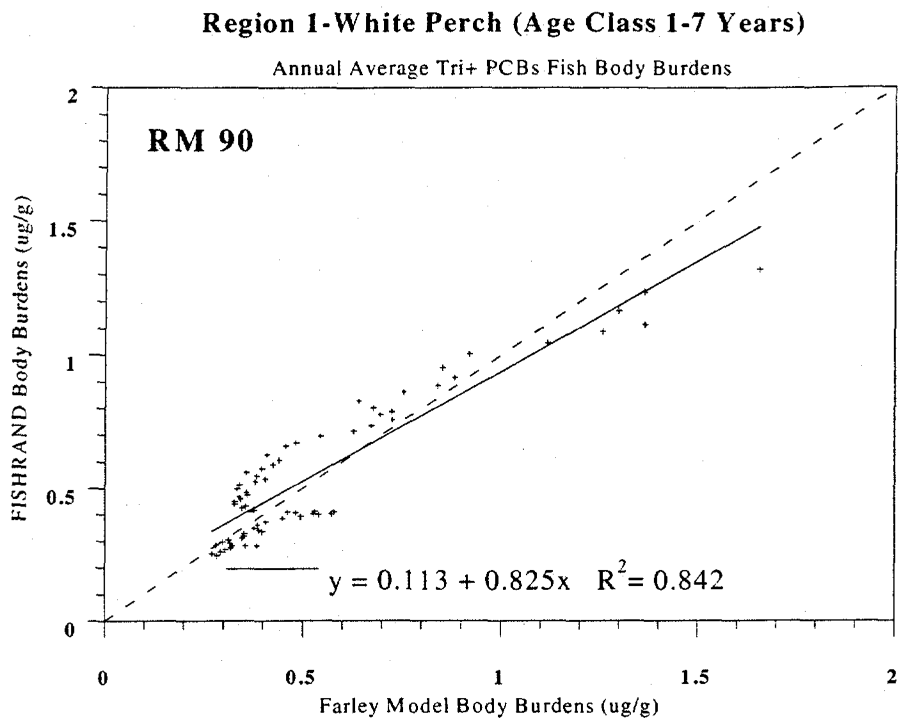
Region 1-White Perch (Age Class 1-7 years)

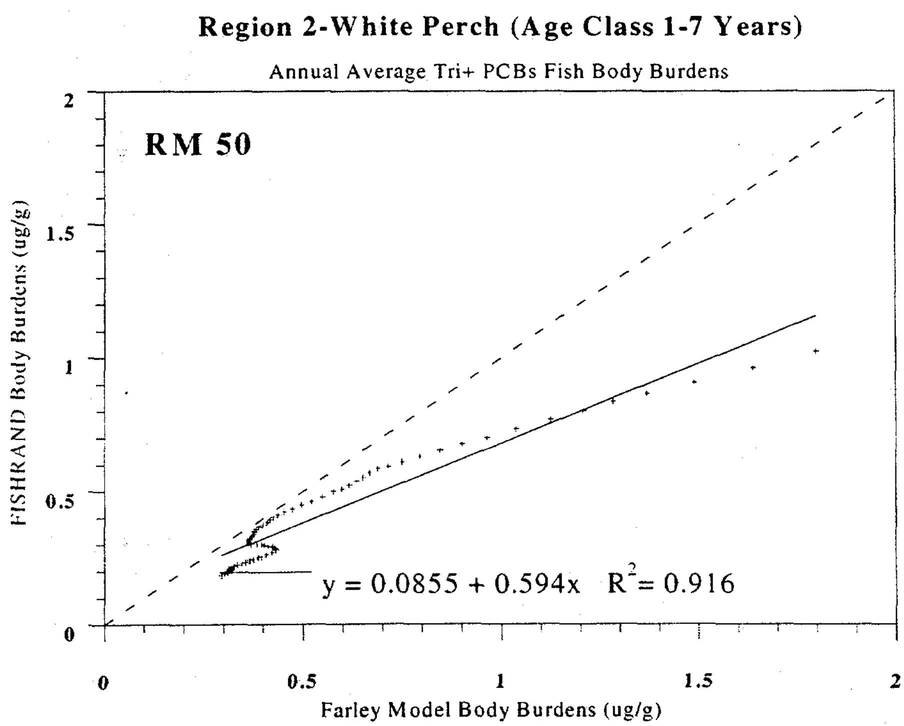


Sources: Farley et al., 1999, Hudson River Database Release 4.1 and USEPA, 2000

Notes: 1. Farley Model results represent the Farley Model with the upper river PCB loads from HUDTOX.
2.The dashed line represents a 1 to 1 line.

Figure 3-11
Comparison of White Perch Body Burdens
(Farley Model vs. FISHRAND)
(page 1 of 2)





Sources: Farley et al., 1999, Hudson River Database Release 4.1 and USEPA, 2000

Notes: 1. Farley Model results represent the Farley Model with the upper river PCB loads from HUDTOX.
2. The dashed line represents a 1 to 1 line.

Figure 3-11
Comparison of White Perch Body Burdens
(Farley Model vs. FISHRAND)
(page 2 of 2)

FIGURE 3-12a: Comparison Between FISHRAND Results and Measurements at RM 152

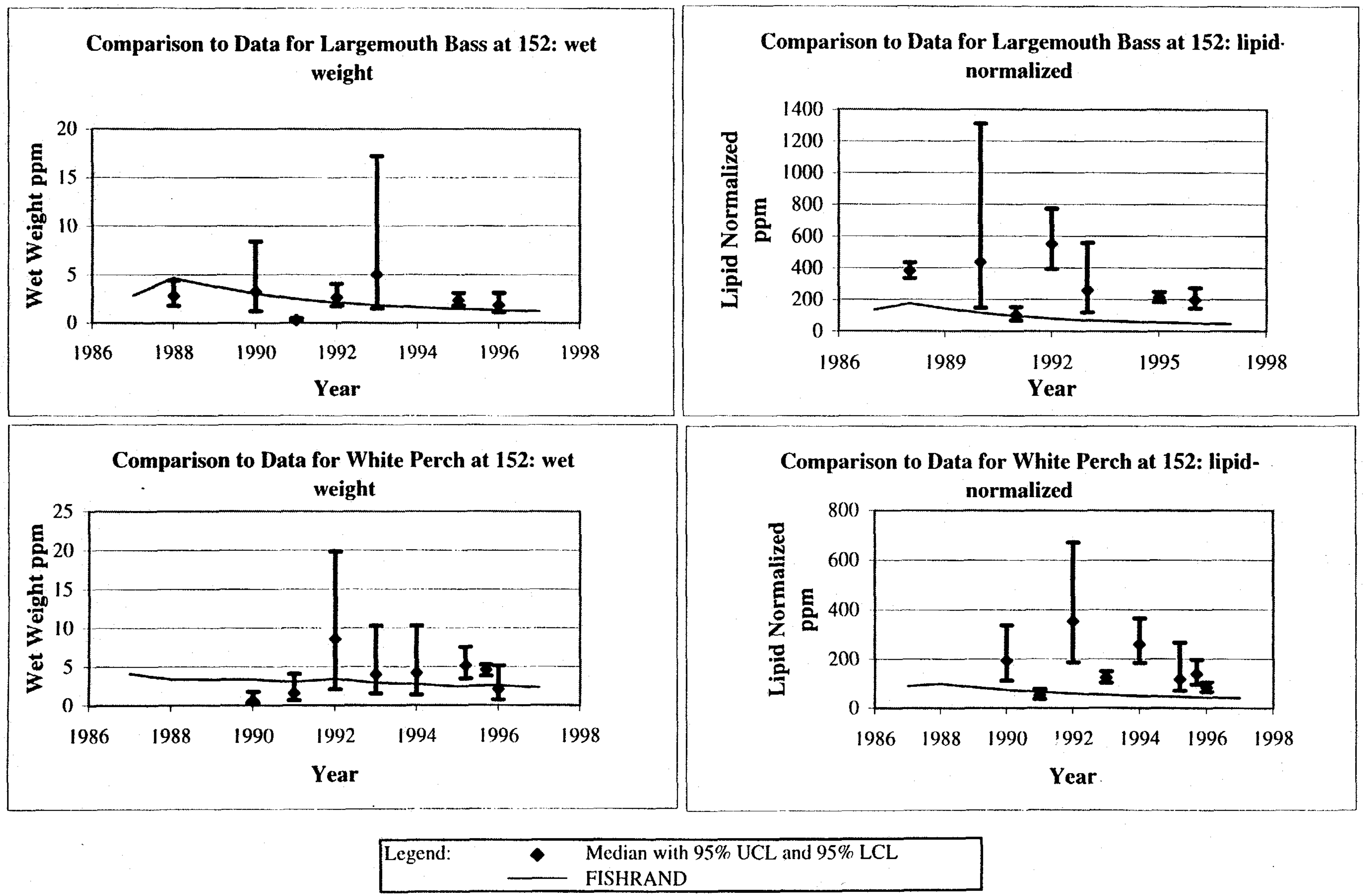


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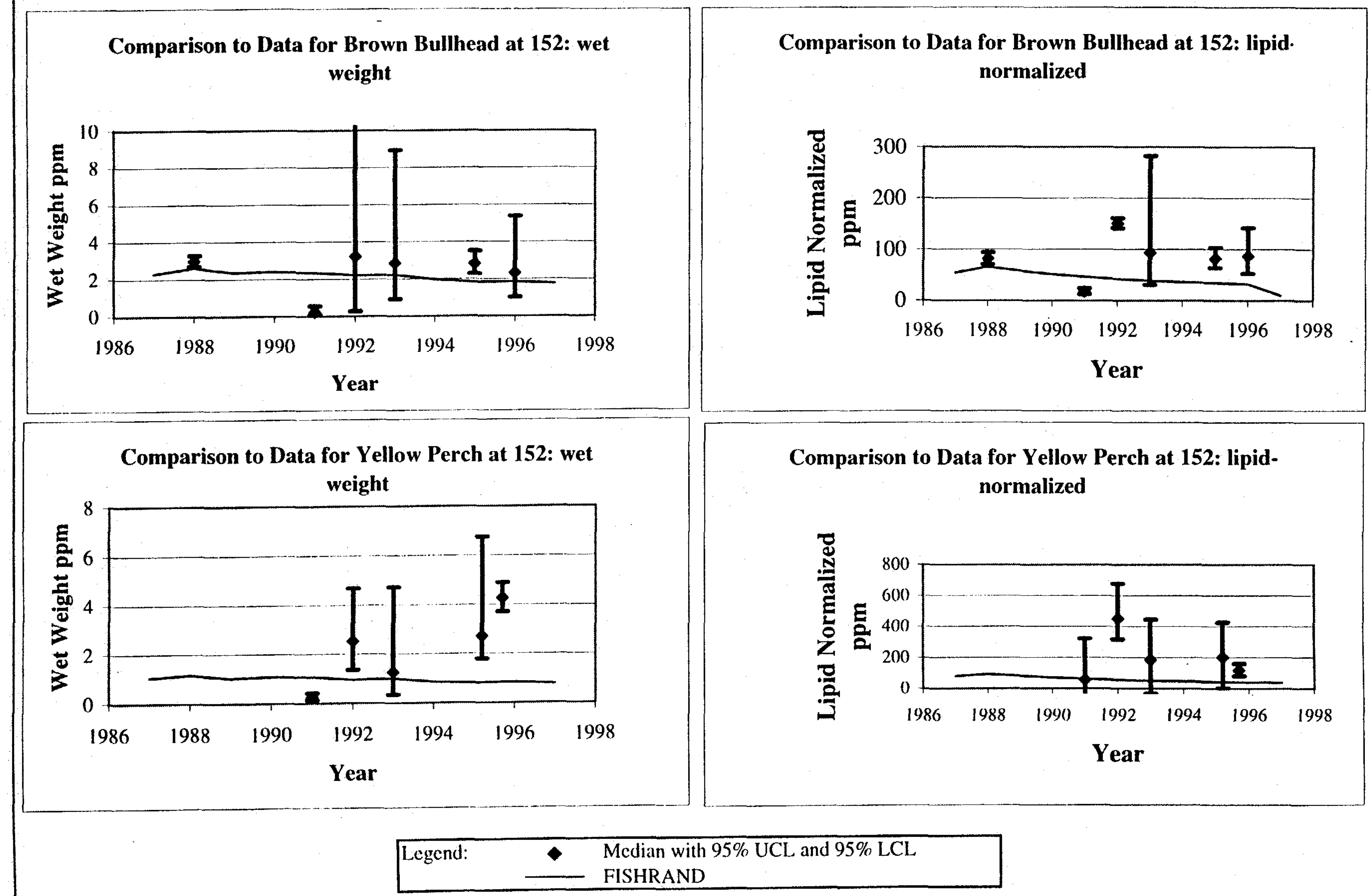
FIGURE 3-12a: Comparison Between FISHRAND Results and Measurements at RM 152



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TAMS/MCA

FIGURE 3-12b: Comparison Between FISHRAND Results and Measurements at RM 113

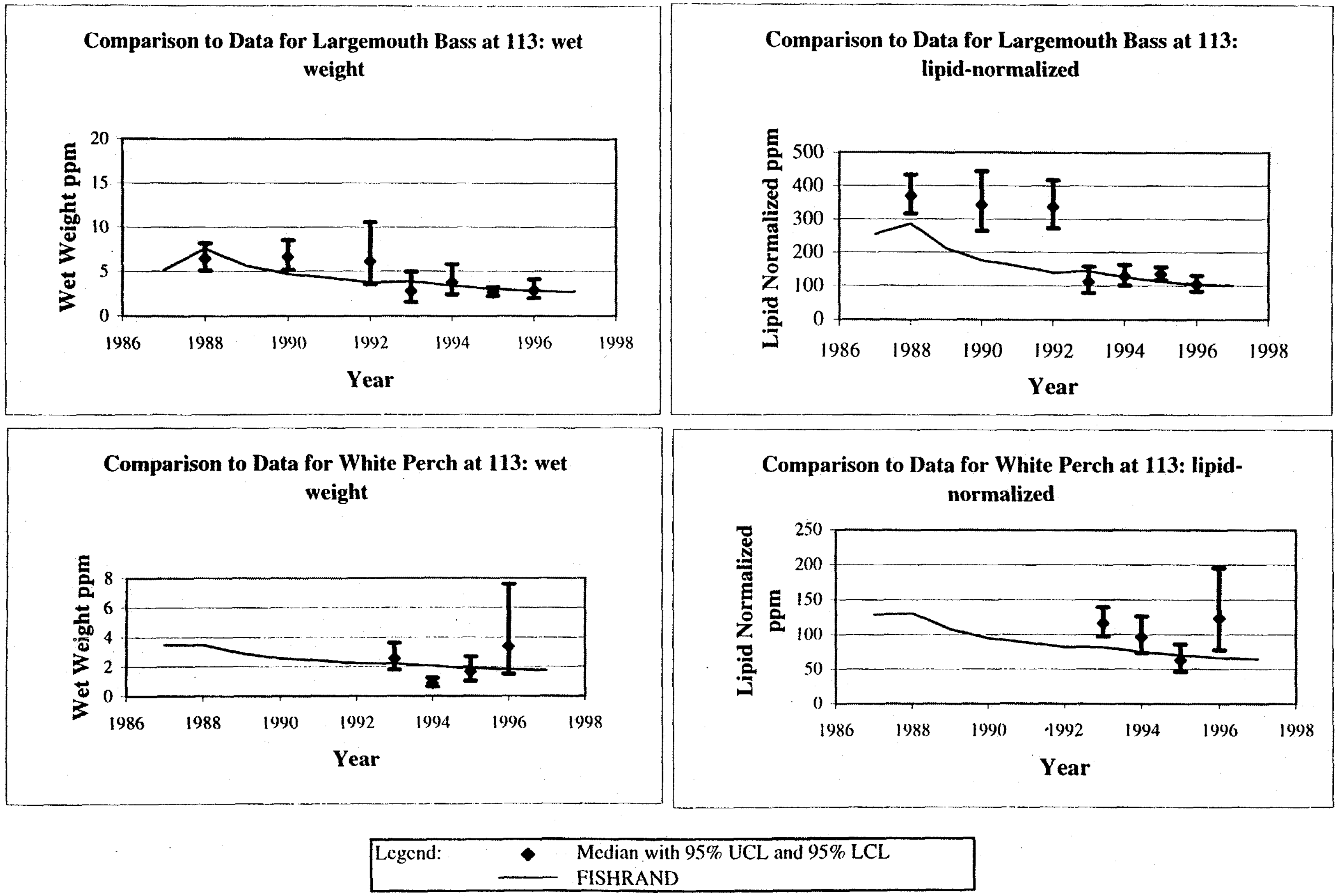

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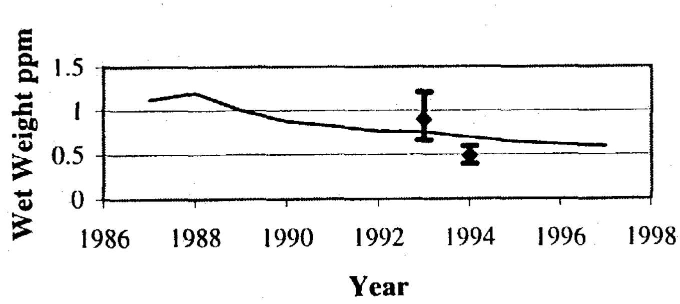
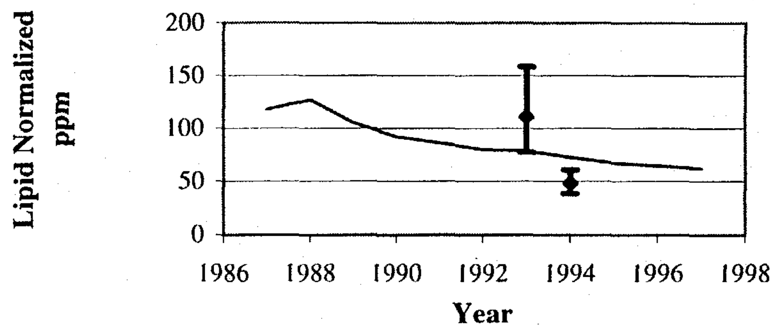
FIGURE 3-12b: Comparison Between FISHRAND Results and Measurements at RM 113





Legend: ◆ Median with 95% UCL and 95% LCL
—— FISHRAND

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FIGURE 3-12c: Comparison Between FISHRAND Results and Measurements of Pumpkinseed

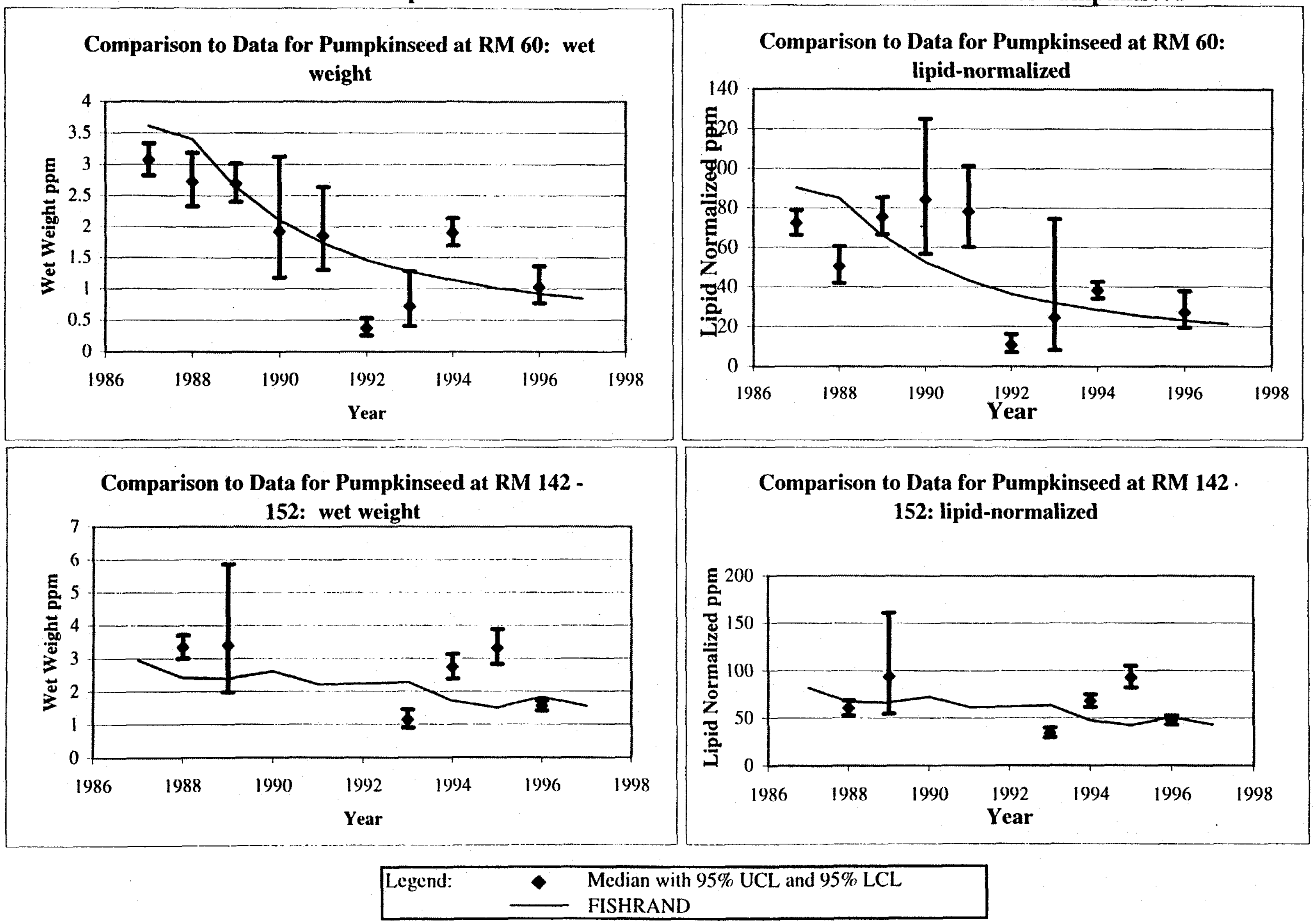

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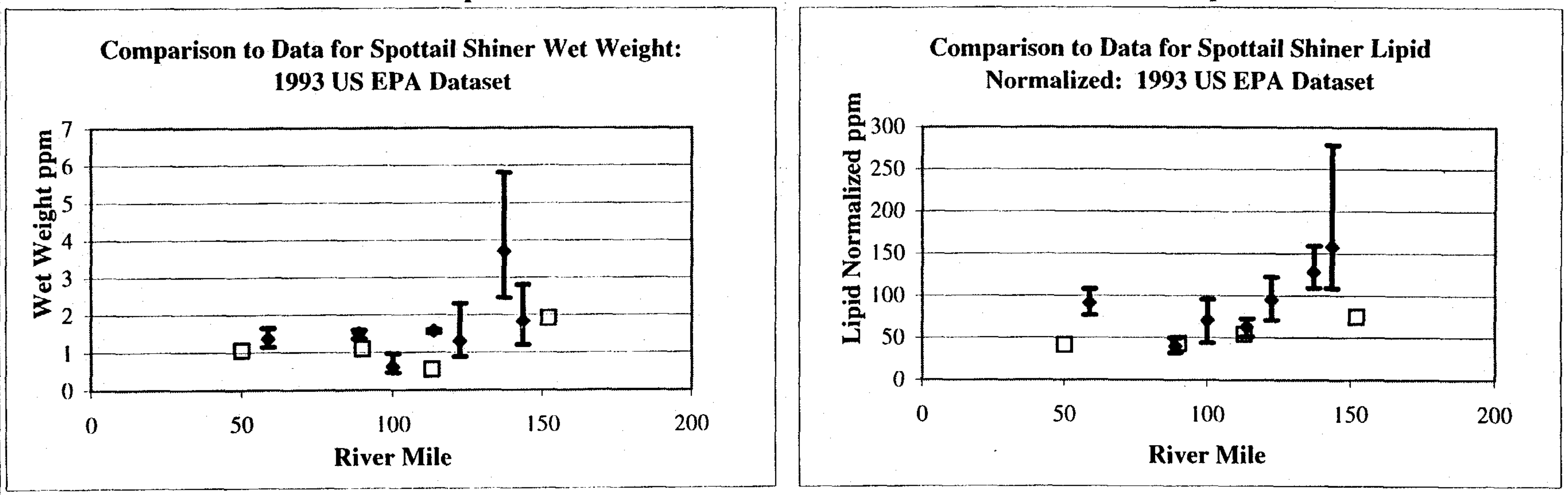
FIGURE 3-12d: Comparison Between FISHRAND Results and Measurements of Spottail Shiner



Legend: ◆ Median with 95% UCL and 95% LCL
□ FISHRAND

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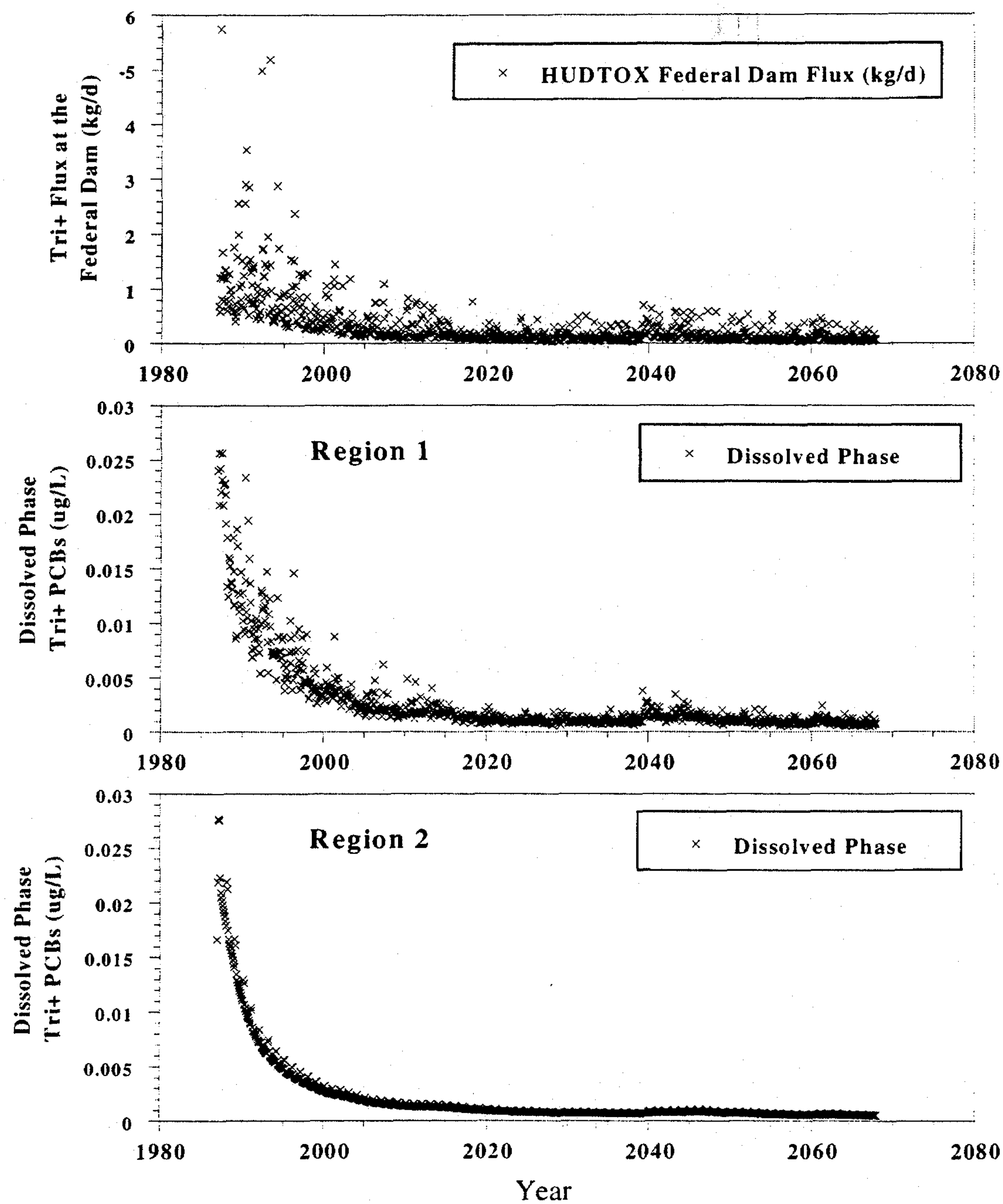
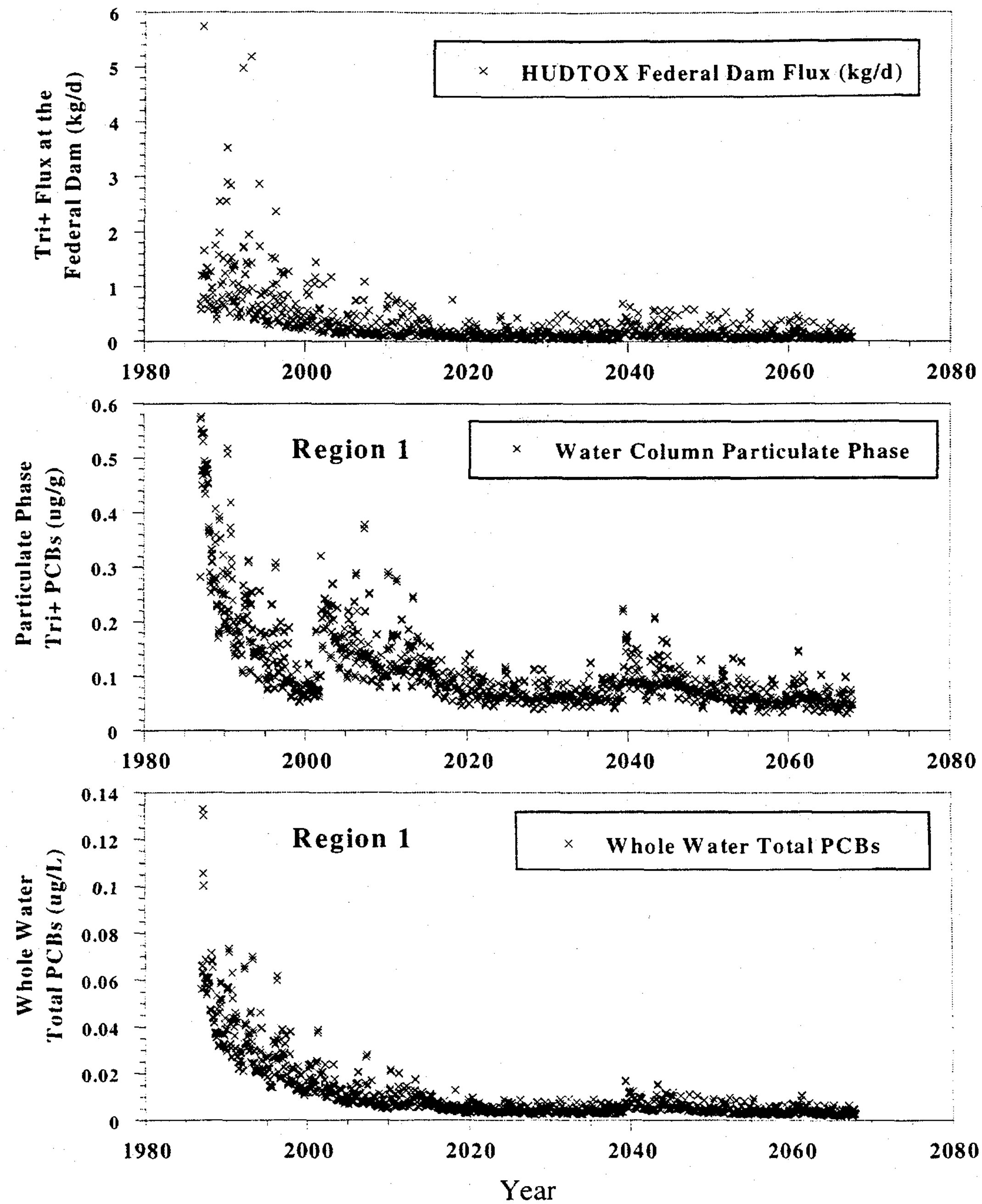


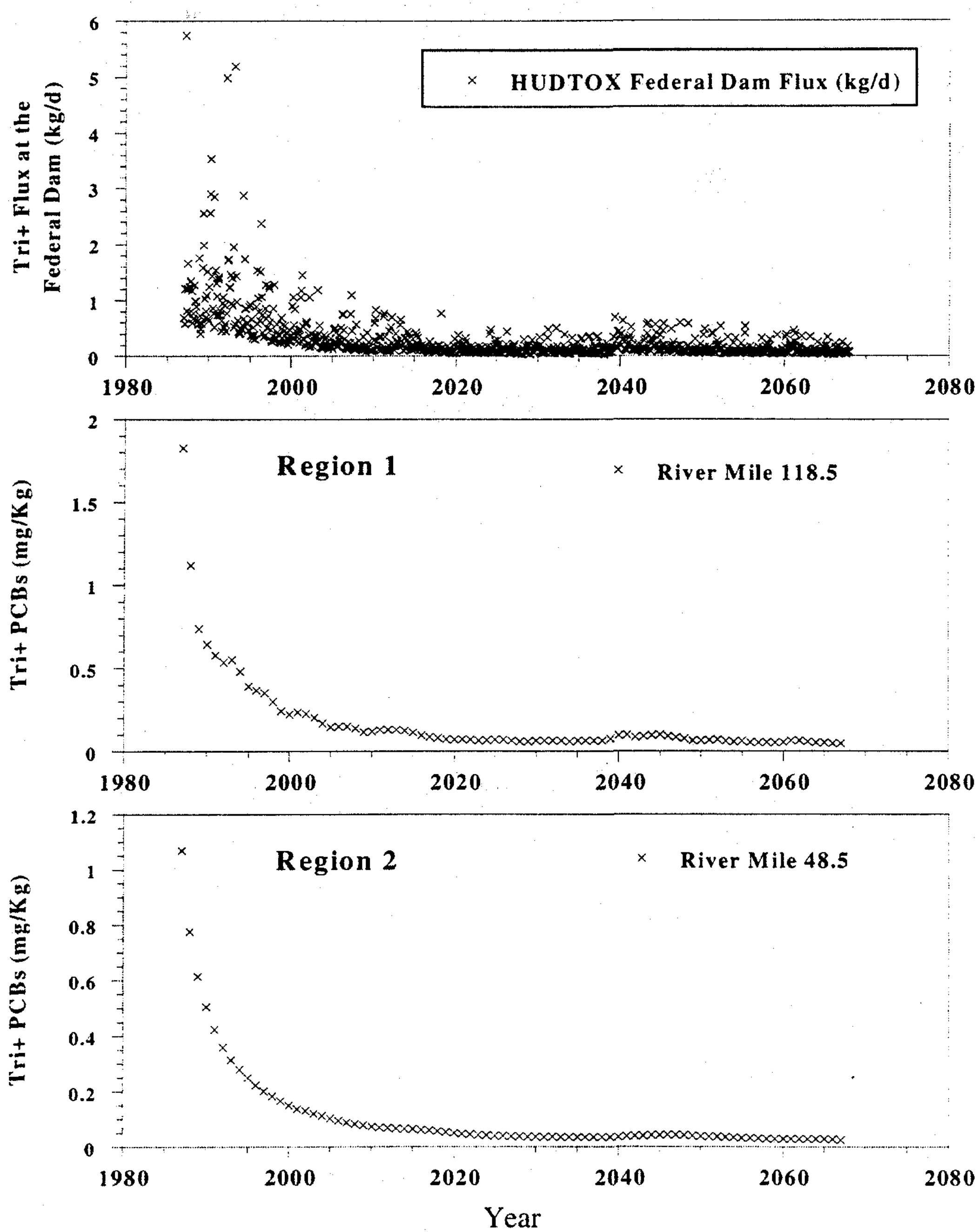
Souces: Farley et al., 1999, Hudson River Database Release 4.1 and USEPA, 2000

Figure 3-13
Comparison Among the HUDTOX Upper River Load and Farley Model Estimates of Dissolved Water Column Concentrations in Food Web Regions 1 and 2
(1987-2067)



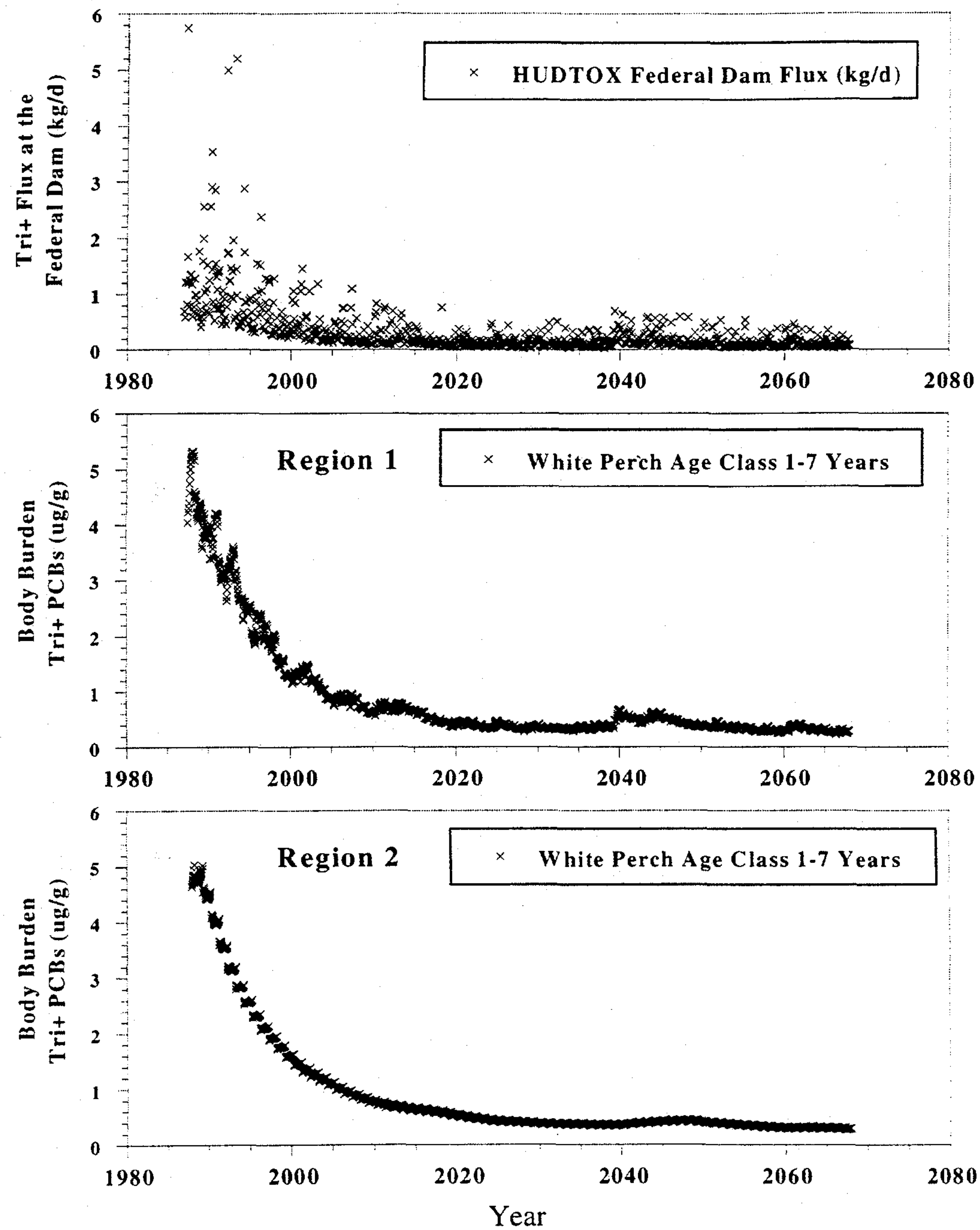
Souces: Farley et al., 1999, Hudson River Database Release 4.1 and USEPA, 2000

Figure 3-14
Comparison Among the HUDTOX Upper River Load and Farley Model Estimates of Particulate and Whole Water Column Concentrations in Food Web Region 1 (1987-2067)



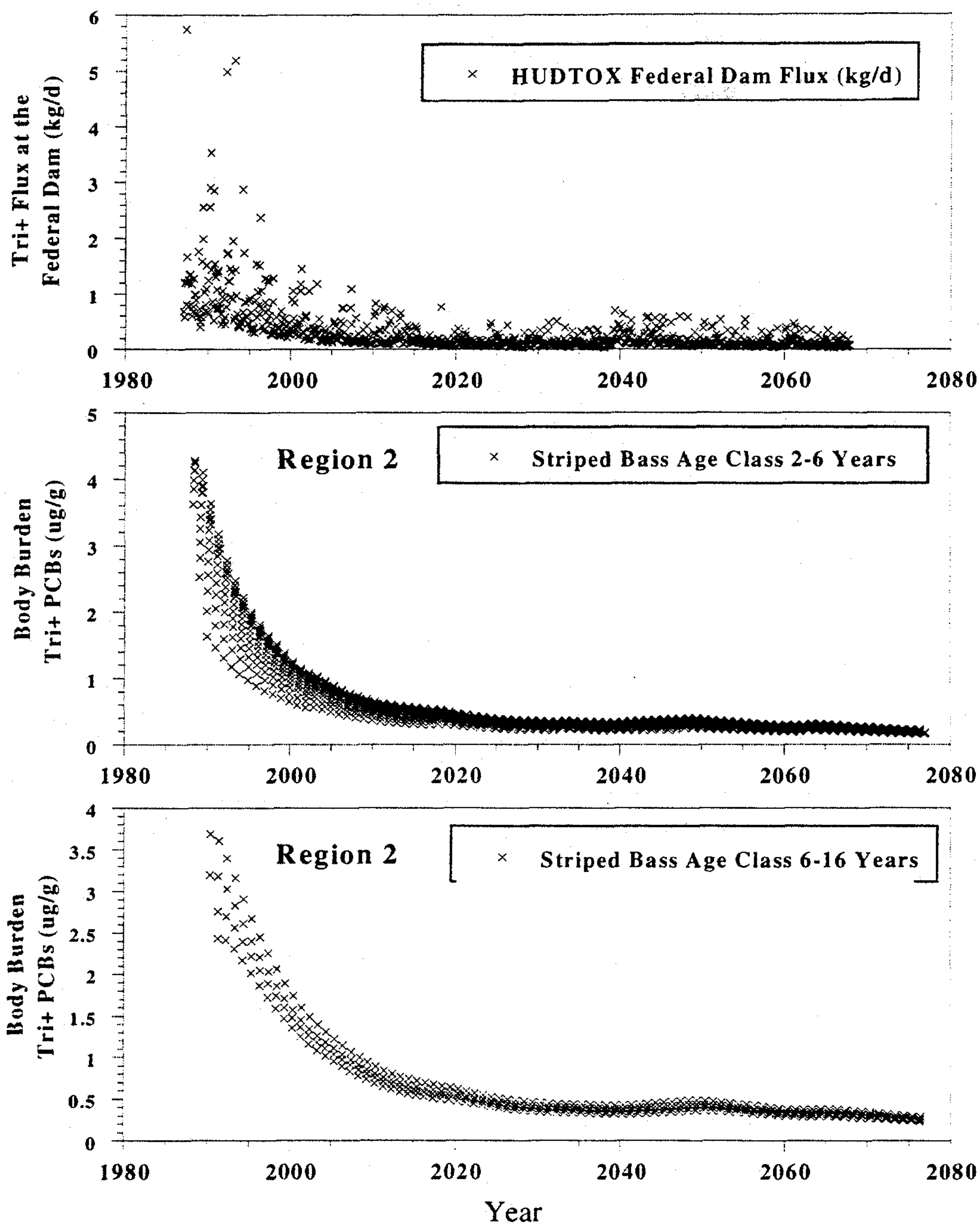
Souces: Farley et al., 1999, Hudson River Database Release 4.1 and USEPA, 2000

Figure 3-15
Comparison Among the HUDTOX Upper River Load and Farley Model Estimates of Surface Soil (0-2.5 cm) in Food Web Regions 1 and 2
(1987-2067)



Souces: Farley et al., 1999, Hudson River Database Release 4.1 and USEPA, 2000

Figure 3-16
Comparison Among the HUDTOX Upper River Load and Farley Model Estimates of White Perch Body Burdens in Food Web Regions 1 and 2
(1987-2067)



Souces: Farley et al., 1999, Hudson River Database Release 4.1 and USEPA, 2000

Figure 3-17
Comparison Among the HUDTOX Upper River Load and Farley Model Estimates
Striped Bass Body Burdens in Food Web Regions 1 and 2
(1987-2067)

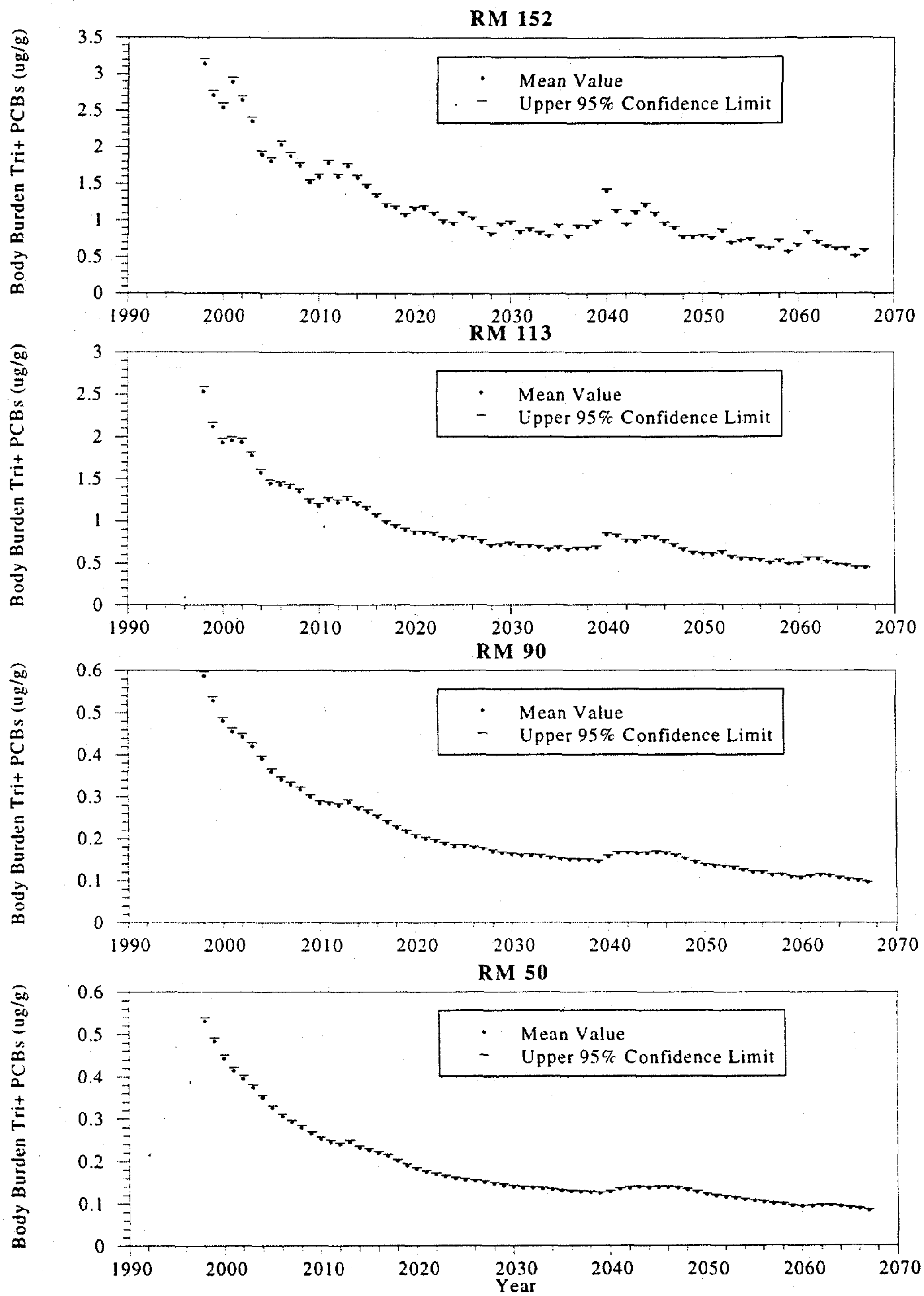


Figure 3-18
Forecasts of Large Mouth Bass Body Burdens from FISHRAND

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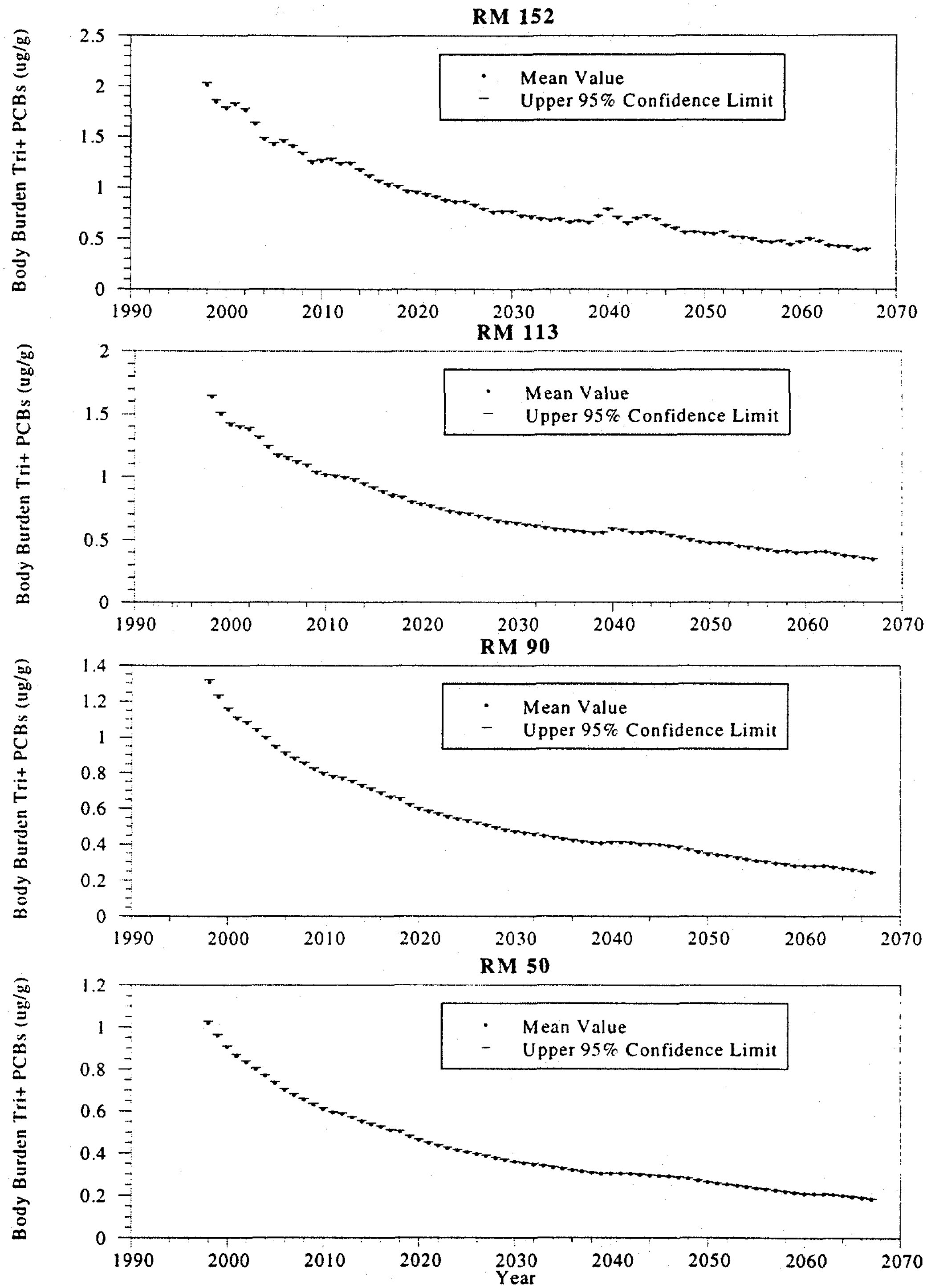

Figure 3-19
Forecasts of White Perch Body Burdens from FISHRAND

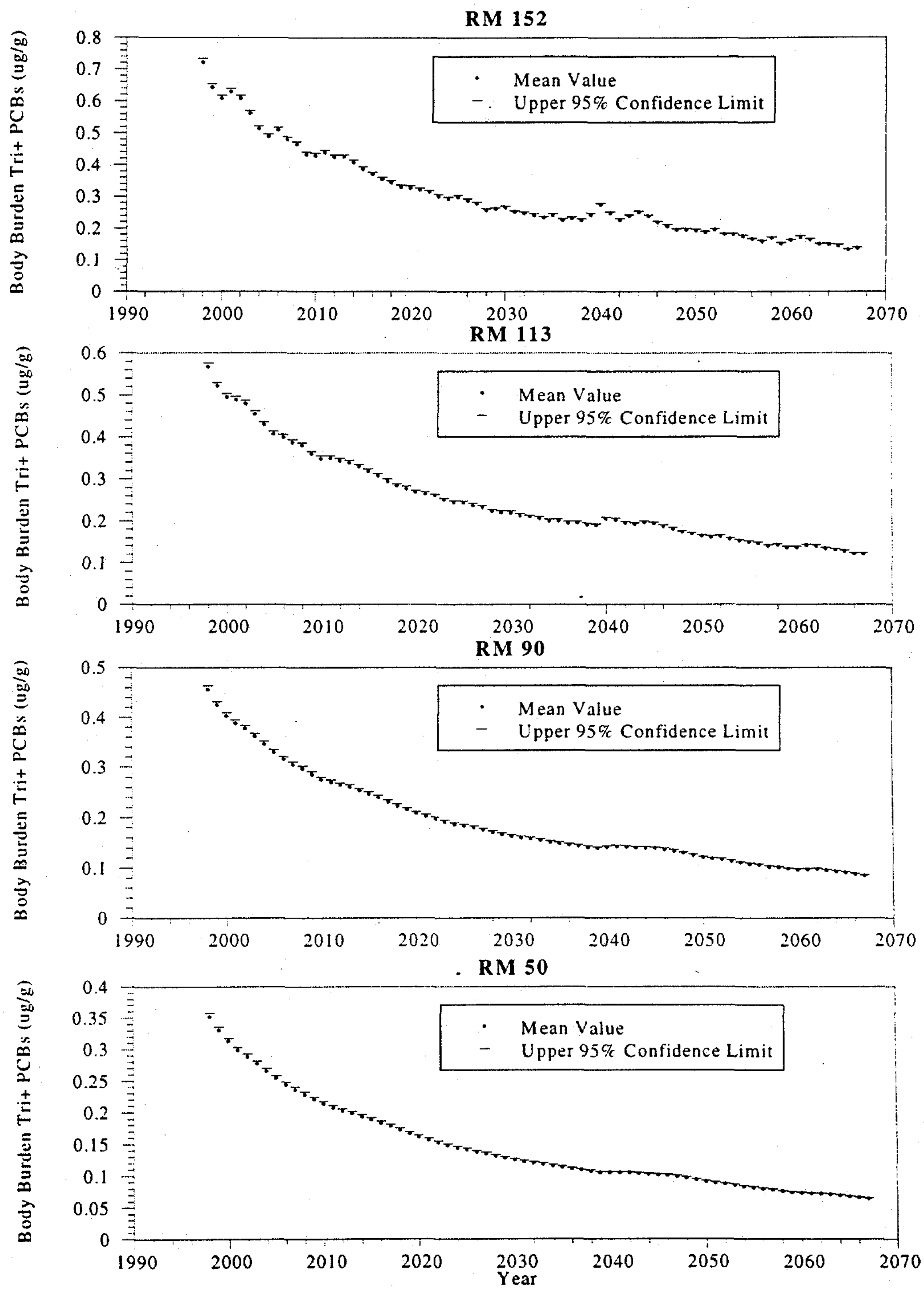

Figure 3-20
Forecasts of Yellow Perch Body Burdens from FISHRAND

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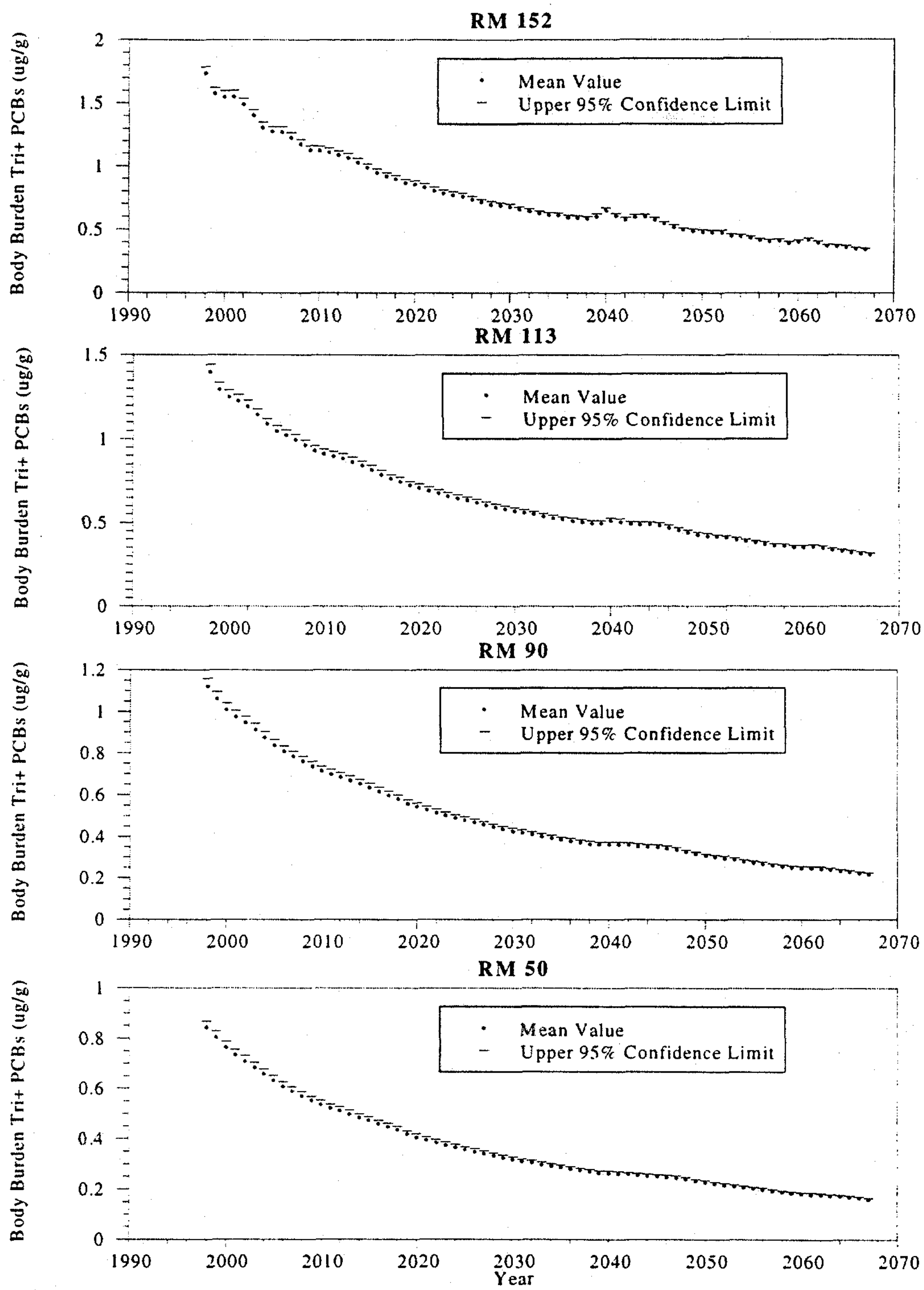

Figure 3-21
Forecasts of Brown Bullhead Body Burdens from FISHRAND

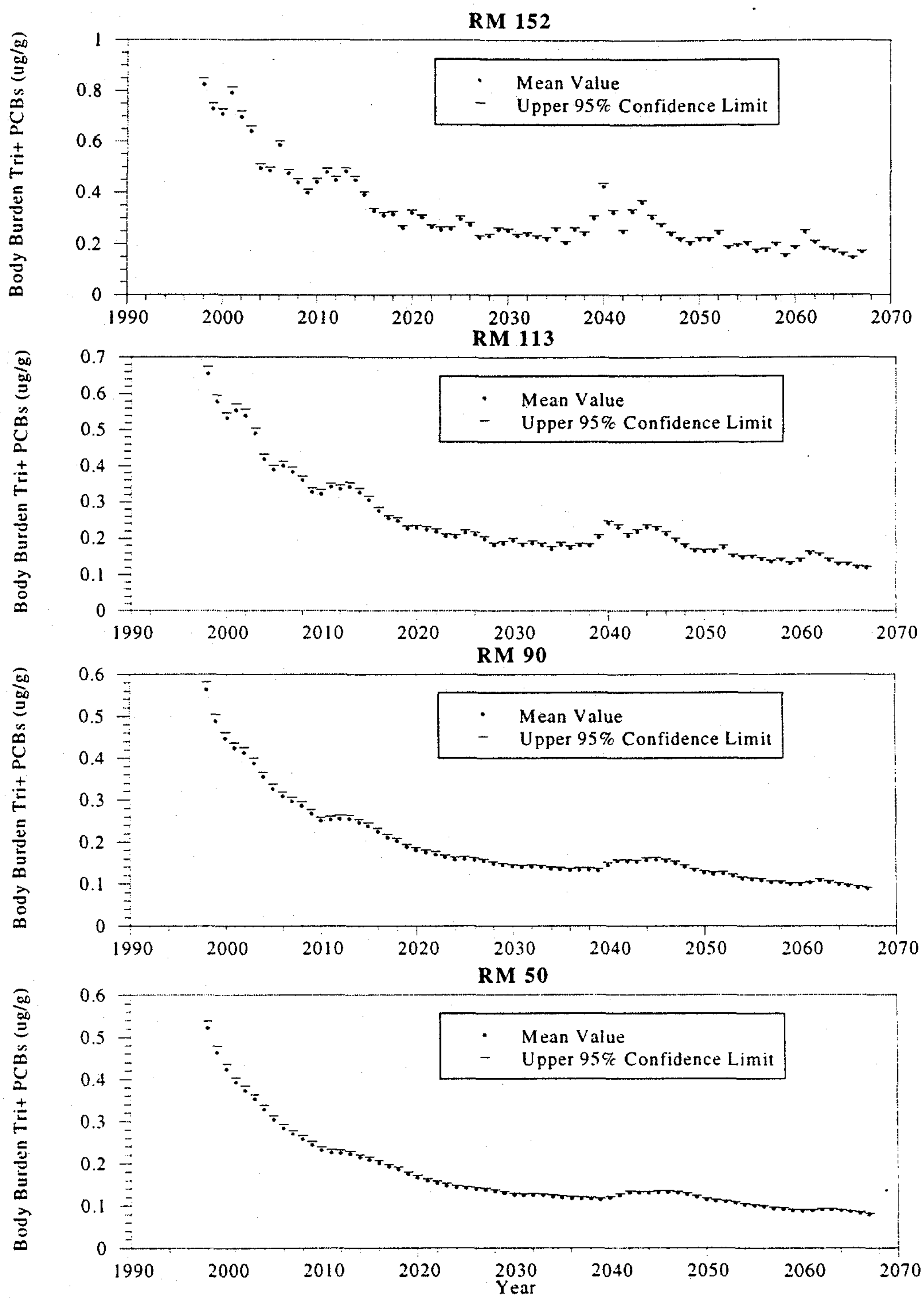

Figure 3-22
Forecasts of Pumpkinseed Body Burdens from FISHRAND

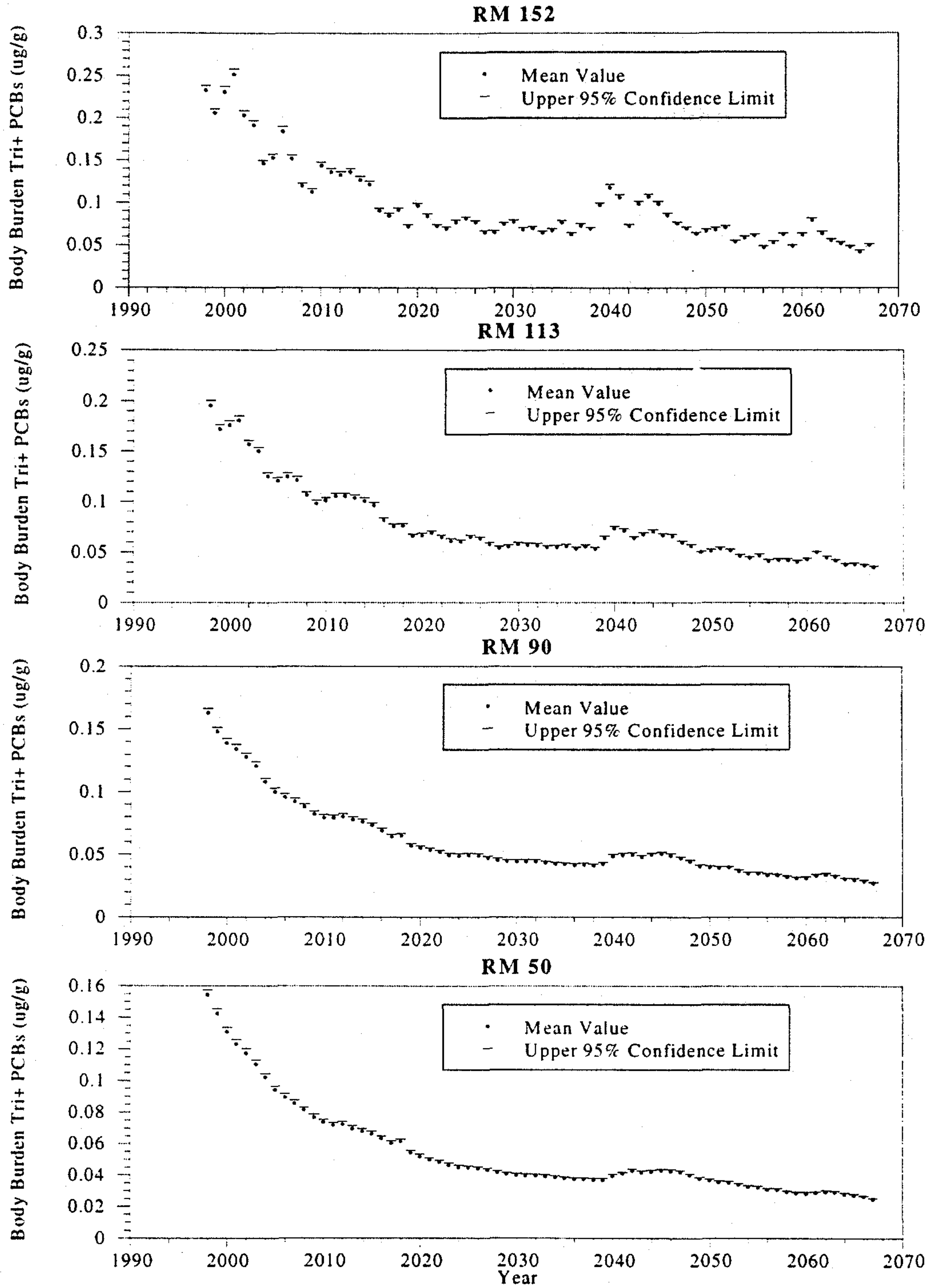

Figure 3-23
Forecasts of Spottail Shiner Body Burdens from FISHRAND

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Appendix A

CONVERSION FROM TRI+ PCB LOADS TO DICHLORO THROUGH HEXACHLORO HOMOLOGUE LOADS AT THE FEDERAL DAM

A.1 Introduction

The fate and transport and bioaccumulation models of PCBs described in Farley *et al.*(1999) (the Farley model) for the mid to lower regions of the Hudson River will be used to predict fish body burdens for the Mid-Hudson Human Health Risk Assessment and the ERA Addendum. As originally constructed, the Farley model relied on load estimates at Thompson Island (TI) Dam to directly represent the loads delivered to the Lower Hudson. Future loads were assumed to be identical to that measured at TI Dam in 1997. This assumption does not account for load variations between TI Dam and Waterford nor the anticipated Upper Hudson load decline over time. Indeed, the forecast prepared Farley *et al.* (1999) extended only to 2002. For the risk assessment requirements of the Phase 2 investigation, a forecast beyond 2002 is required and so the Upper Hudson loads must be adjusted to account for an expected decline in PCBs with time. Additionally, load estimates based on TI Dam measurements do not account for the influences of the intervening 35 miles of river between TI Dam and the Federal Dam at Troy, NY.

The preparation of the Upper Hudson model 70 year forecast also included estimates of Upper Hudson loads at Waterford. Results from the Upper Hudson River model, HUDTOX, developed by Limno-Tech, Inc. (LTI) will be used for the PCB loads coming over the Federal Dam at Troy, NY. The HUDTOX model accounts anticipated declines in water column loads over time as well as the riverine influences on these loads between TI Dam and Troy.

Dichloro through hexachloro PCB homologues are the state variables in the Farley model of the Lower Hudson River but HUDTOX simulates total PCB and the sum of trichloro through decachloro homologues (Tri+) for the Upper Hudson. Thus, a means of converting the data from total or Tri+ PCBs to individual dichloro through hexachloro homologues is required.

A conversion algorithm was developed based on the available data. An extensive number of samples are available from the TI Dam station, but relatively few are available from the lower station at Waterford, NY and even fewer from Troy, NY. In this analysis, homologue patterns at the TI Dam are compared to the patterns at Waterford to determine if a correction can be applied to the TI Dam data so as to yield conditions at Waterford. Mean homologue mass fractions are calculated using data collected at the TI Dam station and grouped to determine if the patterns should be adjusted for season or flow rate. Through this effort, a means of conversion of the HUDTOX Tri+ sum is developed. The conversion yields a daily load estimate of each of the homologue groups from dichloro to hexachlorobiphenyl. Referenced tables and figures relating to this analysis follow the text.

A.2 Data Preparation

The data used for this memo are whole water data from USEPA, 1993 and General Electric (QEA, 1999) from Waterford and TI Dam stations. The USEPA data are available in the Hudson River Database, Release 4.1 (USEPA, 1998a). The GE data is from the March 1999 update to the GE database. There are two important differences between the data sets, (1) homologue data from the two data sets do not represent the same exact suite of congeners and (2) the analytical methods are somewhat different. The USEPA homologue data is based on 126 congeners which are individually measured and calibrated . The GE homologue mass fractions are taken directly from the GE database file from March 1999 and are based on a smaller set of congeners and are calibrated to Aroclor standards. Some congeners are unique to each data set.

In compiling the sample results for interpretation, field duplicates collected by GE are not used. For the GE data, there are numerous instances of more than one sample per day per station, obtained for Quality Assurance purposes. The first sample listed per day per station in the GE database is used since the duplicate samples are equivalent. USEPA duplicates from the Phase 2 database were combined and averaged in the preparation of the database and were used as listed in the database tables.

Two USEPA samples (transect 2) were excluded because of data quality issues. Eight GE samples were excluded because the sum of the trichloro to hexachloro homologues was less 97 percent than the sum of the trichloro to decachloro homologues (Tri+). These samples were excluded because it was deemed unlikely that estimates of the true value of the mass percent of heptachloro through decachloro homologues would exceed a few percent of the Tri+ sum.

Samples are grouped by flow and season in several instances. High flow is defined as greater than or equal to 4000 cubic feet per second (cfs); low flow is less than 4000 cfs as measured at the USGS Fort Edward station. For the Waterford samples, flow data from the USGS Waterford station was used in preference over the Fort Edward data to determine the flow condition when available. The basis for defining the flowswith respect to 4000 cfs is discussed in the DEIR Responsiveness Summary (USEPA, 1998b). The seasons are defined as follows: spring, 3/16-5/15; summer, 5/16-10/31; and fall-winter, 11/1-3/15.

A.3 Dichloro Homologue

Optimally, to develop ratios to apply to the HUDTOX Tri+ sum, a long-term record of the homologue composition at Waterford is required. In this manner, a ratio could be developed for the existing period of record, enabling an examination of the results during the 1987-1997 calibration period. Similarly, the ratio could then be used to develop forecasts of Lower Hudson conditions. Unfortunately, this information does not exist but a long-term record does exist at TI Dam. From this information, an estimate of the homologue to Tri+ ratio at TI Dam could be obtained. This ratio is an estimate of the average loading condition at TI Dam. However, this analysis does not yield the homologue to Tri+ ratio at Waterford. Thus for each congener, the ratio at the TI Dam was examined relative to Waterford for the period where data were available. This second factor represents the effects of transport between TI Dam and Waterford. The first ratio would be expected to change with changes in loads originating above TI Dam, as might arise from remediation at the GE facilities. The second factor represents the impacts of water column transport and associated geochemical processes occurring between TI Dam and Waterford. This factor would not be expected to change with time because it is the cumulative result of geochemical processes (*e.g.*, gas exchange, sediment-water exchange, aerobic degradation) which should remain the same with time. This factor would be expected to vary seasonally, however, because temperature and flow rate changes will affect the rates of the various geochemical processes.

To determine the ratio of the dichloro homologue to the HUDTOX Tri+ load (di/Tri+) at Waterford, the following steps were taken:

- Comparison of the Waterford di/Tri+ ratio between the TI Dam and Waterford stations. Homologue data for Waterford are limited, but are available for the TI Dam from 1990-1998 using the GE data. A correction factor to relate these stations on either a seasonal or flow basis is needed in order to use the long record of data at the TI Dam. This factor represents the TI Dam-to-Waterford transport factor described above.

- Examination of the di/Tri+ PCBs ratio overtime to determined if the ratio has changed substantially overtime. Data were grouped to determine the mean values of the di/Tri+ ratio by period, season and flow. This represents the loading ratio described above.

The data set to establish the TI Dam to Waterford ratio is limited. In particular, the 1991 GE samples at TI Dam and Waterford were not timed to capture the same parcel of water as it traveled from the TI Dam to Waterford. Thus, these samples do not directly track the changes to the water column loads originating from the geochemical processes which occur enroute. Given the relatively low number of samples collected at the two stations that year, there are not enough samples to develop an average ratio to accurately represent the effects of the geochemical processes as a function of flow and season. Table A-1 lists the calculated time for each flow rate at Fort Edward for water to travel from TI Dam to Waterford and the hours between sampling at these stations. None of the travel times are similar to the sampling times, indicating that the sampling were not timed to capture the same parcel of data. Because of this aspect of the GE sampling method, only the USEPA Phase 2 samples, which were purposely timed to capture the same parcel of water, will be used to compare TI Dam to Waterford. As discussed below, all of the GE and Phase 2 samples at TI Dam will be used to examine the temporal changes in homologue percentages.

Figures A-1 through A-5 show the di/Tri+ ratio (expressed as a percentage of the Tri+ concentration) grouped by station, season and flow rate for the USEPA data only. Figure A-1 shows a statistically significant difference in the di/Tri+ ratio at the two stations for all Phase 2 results. The subsequent figures show how this difference correlates with flow and season. The grouping by flow shows a significant difference of the means during low flow (Figure A-4) and no difference during the high flow (Figure A-5). This suggests that during the typically low flow conditions of the warmer months, there is time for the PCBs in the water column to interact with the sediments, altering the homologue pattern. During the periods of high flow, the PCBs at TI Dam are translated to Waterford nearly unchanged. Flow was chosen as the main separation variable for this ratio because it yielded the greatest separation among groups at low flow and no separation at high flow, as might be expected.

To determine the loading ratio at TI Dam (the first factor discussed above), the di/Tri+ versus time at the TI Dam and Waterford stations is shown in Figures A-6 and A-7, respectively. These figures display both the USEPA and the GE data over the period 1991 to 1998. A change in the pattern of the di/Tri+ ratio is evident starting in mid-1996 in the TI Dam results. (No data are available for Waterford post-1993.) The range of di/Tri+ ratios is greater and the average value is higher at the TI Dam after 1995. This is coincident with a drop in total PCB concentration as shown in Figure A-8. This figure shows the total PCB concentration versus time at the TI Dam. The decrease in concentration in 1996 and later is attributed to the 1993-1995 remediation efforts above Rogers Island, which substantially reduced the Tri+ loading to the Hudson River. Little evidence of subsequent decline in loads is evident post-1995. As a result of the GE remedial efforts, the importance of the sediments to the water column loads was greatly increased while the sporadic, large-scale releases above Rogers Island largely disappear. Based on these results, the data from 1996-1998 should be used to predict future conditions. Figure A-9 shows the TI Dam di/Tri+ ratio grouped by years 1991-1995 and 1996-1998. The difference in means is clearly significant. Figures A-10 through A-13 show the same data further grouped by season and flow. Of these, the best separation of the means is seen using flow.

Table A-2 summarizes the basis for conversion for the di PCB homologue as well as the other homologue groups, which are discussed below. The table is separated into the calibration perio, (1987-1998) and the forecast period (1999 and later). The mean di/Tri+ ratios at the TI Dam are from Figures A-12 and A-13. For low flow, the correction from the TI Dam to Waterford is 0.52 which is the ratio of the means 45.5883/86.8350 given in Figure A-4. The correction during the high flow is small (1.04) because, as shown in Figure A-5, there is no significant difference between the means. Note that for the dichloro ratio only, the ratios developed here are applied throughout both the calibration and forecast periods, as appropriate. For the period prior to 1991 where no congener data exist, the ratios measured in 1991 are applied. In the forecast calculations, the ratios developed for the period 1996-1998 at TI Dam are applied along with the TI Dam to Waterford transport correction.

A.4 Trichloro through Hexachloro Homologues

Ratios for the trichloro to hexachloro homologues were developed in a fashion similar to that used for the dichloro homologue. These ratios has the additional constraint that they must sum to 100 percent, representing the entire Tri+ load. The fractions of trichloro through hexachloro homologues at Waterford are determined by two factors, as follows:

- TI Dam-to-Waterford Correction: Comparison of the fractions of trichloro through hexachloro homologues in Tri+ PCBs at Waterford to TI Dam. Because the number of samples is limited at Waterford, the extensive data from the TI Dam can be used with correction for the Waterford station. As was discussed in the DEIR (USEPA, 1997) and the LRC Responsiveness Summary (USEPA, 1999), the trichloro through hexachloro homologues appear to be translated from the TI Dam to lower river stations with little modification.

- TI Dam-Loading Factor: Development of this factor was based on two steps:

  - Principal components analysis to determine if the distribution of trichloro through hexachloro homologues in Tri+ PCBs is significantly affected by season, flow, etc.

  - Examination of the TI Dam Tri+ PCB ratios to determined if the ratios have changed substantially overtime. Data were grouped to determine the mean values of the ratios by period, season and flow.

As in the examination of TI Dam-to-Waterford transport for the di homologue, the GE samples were not timed to capture the same parcel of data (Table A-1). Thus, these samples were excluded from the determination of the TI Dam-to-Waterford correction for the heavier homologues as well.. Figures A-14 through A-21 show the USEPA data exclusively, grouped by season. The one fall-winter sample is grouped with the spring data. A significant difference in the means is only evident during the summer for the trichloro through pentachloro homologues. Notably, the fraction of tri/Tri+ decreases from TI Dam to Waterford while the remaining heavier homologues all increase relative to the TI Dam ratio. Mean ratios at TI Dam and Waterford are quite close during the remainder of the year. Nonetheless, the ratios developed from this analysis were applied to the data in order to represent the best estimate of the relative changes between TI Dam and Waterford. Use of the entire suite of ratios also serves to maintain conservation of mass (*i.e.,* one ratio cannot decrease without corresponding increases in the remaining ratios). These are summarized in Table A-2.

In the examination of the temporal variation of the homologue to Tri+ ratios, a principal components analysis was undertaken. In this examination the mass fractions of trichloro through hexachloro homologues were used as the primary variables. A principal components analysis using the GE and USEPA data is shown in Figure A-22. The results of the analysis are presented in five different ways, with indicators to denote sampling agency, season, flow, station and year (1991-1995 and 1996-1998). No significant separations among the data are seen using these groupings.

Although no evidence of the temporal variation was seen in the PCA analysis described above, an examination of the trends of the various ratios with time suggests the occurrence of a temporal change. A map of the GE TI Dam stations is shown in Figure A-23 with the coordinates provided in the GE database. Data from these stations along with the USEPA Phase 2 results are plotted against time as the mass fraction of trichloro through hexachloro homologues versus Tri+ PCBs in Figures A-24 through A-27. As with the di homologue fraction, a difference in the pattern is seen beginning in 1996. This change in pattern (particularly evident in the tri/Tri+ and penta/Tri+

ratios) coincides with the decline in total PCB concentration seen in Figure A-8. Based on these results, future conditions were predicted using the 1996 through 1998 data.

The TI Dam from 1996-1998 are grouped by season for each homologue of concern in Figures A-28 through A-31. The data are grouped by flow in Figures A-32 through A-35. The best separation (greatest distance between the Tukey-Kramer circles) of the means is given by grouping on season. It should be noted, however, that the ratio variations among these groups are relatively small, typically only a few percent of the total Tri+ mixture. The importance of these variations increases as the fraction of the homologue decreases, as would be expected. Thus, the summer to spring variation of 8 percent (54 - 46 percent) in the trichloro homologue percentage represents about 15 percent of the total trichloro mass. However, the 2.4 percent summer-to-spring change in the hexachloro homologue ratio represents nearly a 50 percent decline in the ratio from spring to summer. These results should be compared to the dichloro homologue results which show large changes on both absolute and relative scales.

The final conversion factors for the trichloro through hexachloro homologues are shown in Table A-2. The mean mass percent of trichloro to hexachloro homologues using the 1996-1998 TI Dam data was obtained from Figures 29 through 32. The correction for transport from TI Dam to Waterford is given as well. Before applying these two factors, a further step must be taken in order to conserve mass in the calculation. This is done by assuming that the concentration of a homologue at Waterford in 1996-98 is equal to the concentration at Fort Edward in 1996-98 times the ratio of the 1993 concentrations observed at Waterford and Fort Edward. The ratio of concentrations between Waterford and Fort Edward is assumed constant rather than the ratio of the mass percents. The proper way to calculate the mean mass percent at Waterford in 1996-98 for homologue i is:

$$P(WATR)_i = \frac{P(FE)_i \cdot K_i \cdot \sum_i \left[P(FE)_i\right]}{\sum_i \left[P(FE)_i \cdot K_i\right]}$$

where:

P(WATR) is the mass percent relative to Tri+ at Waterford;
P(FE) is the mass percent at Fort Edward; and,
K is the ratio of the 1993 mass percent at Waterford to the 1993 mass percent at Fort Edward.

In this manner, the sum of the tri/Tri+ to hexa/Tri+ ratios will sum to 100 percent in all instances, as it should. Without this correction, this last condition is not met.

A.5 Data Conversion Summary

Table A-2 provides a summary of the data conversion for all periods and flows. The distributions will be applied to the Federal Dam loads generated by the May 1999 HUDTOX model (both the calibration and forecast periods). For the period 1987-1990 where no homologue data are available, the dichloro through hexachloro distribution for 1991 will be applied without correction.

Although PCB releases from the Bakers Falls area may have occurred, this is not of concern because the 1987-1990 period will not be used in the ERA Addendum and Mid-Hudson HHRA and this period does not weigh strongly in the calibration. For the dichloro homologue, the mean mass percent of Tri+ PCBs calculated from the 1991-1995 TI Dam samples will be used for the Waterford distribution during high flow with the TI Dam to Waterford correction. Starting in 1996 and continuing for the remaining period of time to be modeled, the 1996-1998 mean mass percent of di/Tri+ at TI Dam will be used.

For the trichloro through hexachloro homologues during 1991-1998, the distribution defined by the mass percent of Tri+ PCBs from GE samples at the TI Dam was applied. For future predictions of the trichloro through hexachloro homologues, the mean distribution defined by the 1996-1998 data at the TI Dam was used. Each of the mass percent values were corrected for the measured difference between the TI Dam and Waterford to account for transport losses and then adjusted to conserve mass.

A.6 References

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USEPA, 1999. Responsiveness Summary for the Low Resolution Sediment Coring Report. USEPA, Region 2, New York. February, 1999.

Table A-1. Time Between General Electric TID and Waterford Samples in 1991

TID Sample			Waterford Sample			Fort Edward	Interval Between	Estimated Time from TID
Date	Hour	Minute	Date	Hour	Minute	Flow Rate	Samples (hours)	to Waterford (hours)
4/5/91	14	30	4/5/91	17	30	6240	3.0	48
4/12/91	16	0	4/12/91	18	15	12900	2.3	23
4/19/91	16	15	4/19/91	19	15	4750	3.0	63
4/26/91	13	0						
5/3/91	15	15	5/3/91	17	20	6820	2.1	44
5/10/91	16	0						
5/17/91	15	10	5/17/91	17	15	4000	2.1	74
5/24/91	16	15						
5/31/91	14	15	5/31/91	17	10	3310	2.9	90
6/7/91	16	0	6/7/91	18	0	2900	2.0	103
6/14/91	17	0	6/14/91	19	0	2210	2.0	135
7/11/91	16	0	7/11/91	18	10	2590	2.2	115
7/25/91	7	20	7/25/91	14	10	2210	6.8	135
8/7/91	12	0	8/7/91	14	30	2320	2.5	128
8/22/91	10	45	8/22/91	13	0	2450	2.3	122
9/5/91	11	15	9/5/91	15	25	2170	4.2	137
9/11/91	10	50	9/11/91	13	30	2890	2.7	103
9/18/91	10	15	9/18/91	12	45	3230	2.5	92
9/25/91	10	25	9/25/91	12	50	2710	2.4	110
10/2/91	10	40	10/2/91	13	30	2410	2.8	124
10/9/91	10	20	10/9/91	13	0	3340	2.7	89
10/16/91	10	0	10/16/91	12	45	3180	2.8	94
10/23/91	10	10	10/23/91	12	40	3110	2.5	96
10/30/91	9	35	10/30/91	12	15	2440	2.7	122
11/6/91	10	40	11/6/91	13	30	2590	2.8	115
11/13/91	9	20	11/13/91	12	0	3120	2.7	96
11/20/91	9	55	11/20/91	12	30	2870	2.6	104
11/26/91	10	50	11/26/91	13	30	3300	2.7	90
12/4/91	10	25	12/4/91	13	10	3700	2.8	81
12/11/91	11	5	12/11/91	14	20	4220	3.3	71
12/18/91	11	20	12/18/91	14	20	4200	3.0	71
12/26/91	10	45	12/26/91	14	10	3600	3.4	83

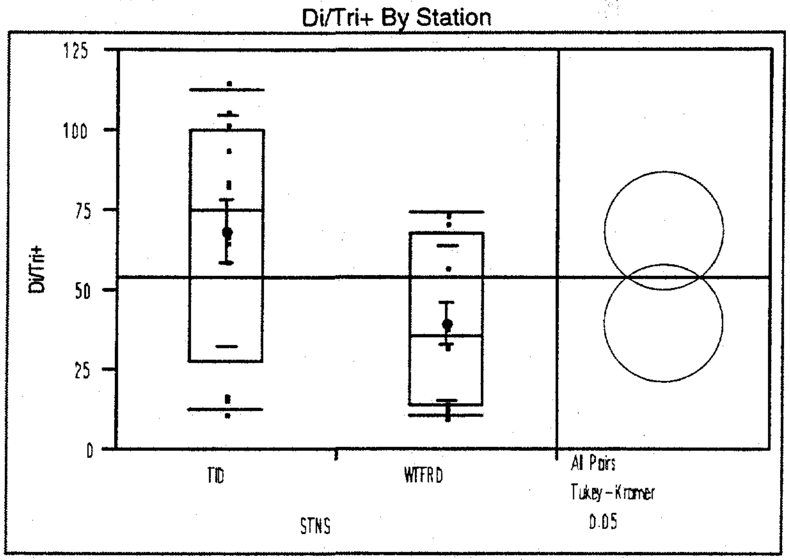
TAMS/MCA

302884

Table A-2. Summary of Conversion for the Di through Hexa Homologues

Homologue	Period	Mean Mass Percent of Tri+ Using TID Data	+2 Standard Errors	-2 Standard Errors	Mean Mass Percent Ratio Waterford/TID	Corrected TID Mass Percent	Mass Percent of Tri+ at Waterford
Calibration Period							
Di-Hexa	1987-1990			Repeat the 1991 Distribution			
Tri-Hexa	Fall-winter 1991-1998		GE TID Data		Same as below by homologue.	Varies	Varies
Tri-Hexa	Spring 1991-1998		GE TID Data		"	Varies	Varies
Tri-Hexa	Summer 1991-1998		GE TID Data		"	Varies	Varies
Forecast Period							
Di	High Flow 1991-1995	32.17	36.28	28.07	1.04	33.37	33.37
Di	Low Flow 1991-1995	48.40	53.02	43.78	0.52	25.41	25.41
Di	High Flow 1996-1998	70.64	76.69	64.60	1.04	73.27	73.27
Di	Low Flow 1996-1998	96.46	102.16	90.76	0.52	50.64	50.64
Di	High Flow 1999+	70.64	76.69	64.60	1.04	73.27	73.27
Di	Low Flow 1999+	96.46	102.16	90.76	0.52	50.64	50.64
Tri	Fall-winter 1999+	47.21	48.82	45.60	0.98	46.11	44.97
Tri	Spring 1999+	45.90	47.71	44.09	0.98	44.83	44.06
Tri	Summer 1999+	54.30	55.12	53.48	0.91	49.18	48.08
Tetra	Fall-winter 1999+	29.66	30.51	28.81	0.97	28.76	28.05
Tetra	Spring 1999+	34.41	35.55	33.26	0.97	33.36	32.79
Tetra	Summer 1999+	30.12	30.55	29.69	1.09	32.81	32.08
Penta	Fall-winter 1999+	18.10	19.22	16.98	1.19	21.49	20.96
Penta	Spring 1999+	15.65	16.88	14.41	1.19	18.58	18.26
Penta	Summer 1999+	12.95	13.54	12.37	1.28	16.64	16.27
Hexa	Fall-winter 1999+	5.00	5.58	4.42	1.23	6.15	6.00
Hexa	Spring 1999+	4.04	4.61	3.48	1.23	4.97	4.89
Hexa	Summer 1999+	2.62	2.82	2.41	1.39	3.64	3.56
Tri-Hexa	Fall-winter 1999+	99.97				102.50	99.97
Tri-Hexa	Spring 1999+	100.00				101.74	100.00
Tri-Hexa	Summer 1999+	99.99				102.26	99.99

TAMS/MCA



Quantiles

Level	minimum	10.0%	25.0%	median	75.0%	90.0%	maximum
TID	11.2	12.649	28.245	75.42	100.2925	112.793	115.58
WTFRD	10.12	10.984	14.355	35.765	67.7975	74.643	74.76

Means and Std Deviations

Level	Number	Mean	Std Dev	Std Err Mean
TID	12	68.6608	36.4512	10.523
WTFRD	12	39.5517	24.7008	7.130

Means Comparisons

Dif=Mean[i]-Mean[j]	TID	WTFRD
TID	0.0000	29.1092
WTFRD	-29.1092	0.0000

Alpha= 0.05

Comparisons for all pairs using Tukey-Kramer HSD

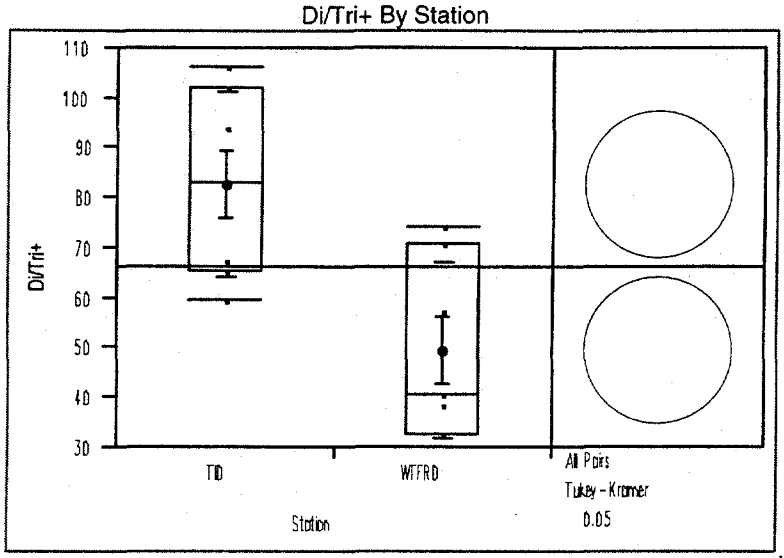
q*
2.07387

Abs(Dif)-LSD	TID	WTFRD
TID	-26.3609	2.7483
WTFRD	2.7483	-26.3609

Positive values show pairs of means that are significantly different.

Source: Hudson River Database Release 4.1

Figure A-1
Di/Tri+ Mass Ratio in USEPA Phase 2 Samples at the TI Dam and Waterford Stations



Quantiles

Level	minimum	10.0%	25.0%	median	75.0%	90.0%	maximum
TID	59.64	59.64	65.66	83.21	102.25	106.29	106.29
WTFRD	32.57	32.57	32.75	40.58	71.17	74.37	74.37

Means and Std Deviations

Level	Number	Mean	Std Dev	Std Err Mean
TID	7	82.7286	18.8217	7.1139
WTFRD	7	49.7000	17.8718	6.7549

Means Comparisons

Dif=Mean[i]-Mean[j]	TID	WTFRD
TID	0.0000	33.0286
WTFRD	-33.0286	0.0000

Alpha= 0.05

Comparisons for all pairs using Tukey-Kramer HSD

q*

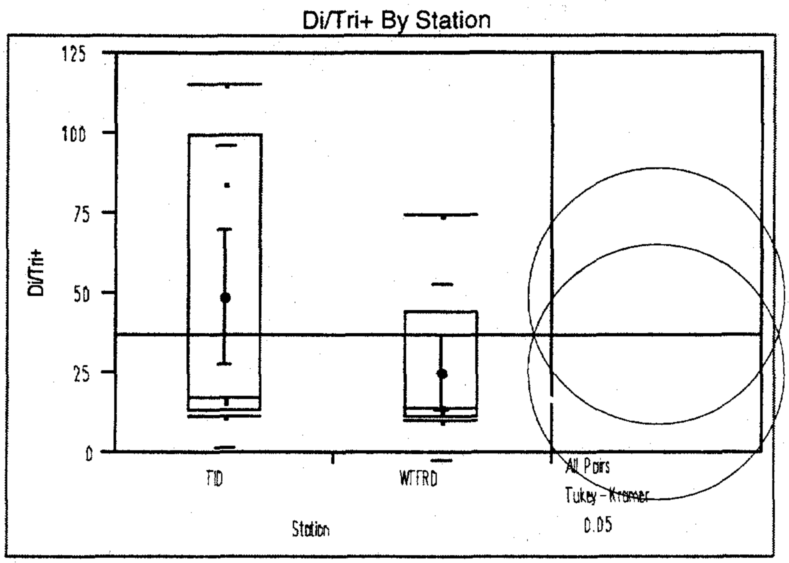
2.17880

Abs(Dif)-LSD	TID	WTFRD
TID	-21.3741	11.6545
WTFRD	11.6545	-21.3741

Positive values show pairs of means that are significantly different.

Source: Hudson River Database Release 4.1

Figure A-2
Di/Tri+ Mass Ratio in USEPA Phase 2 Samples at the TI Dam and Waterford Stations - Summer



Quantiles

Level	minimum	10.0%	25.0%	median	75.0%	90.0%	maximum
TID	11.2	11.2	13.615	17.78	99.91	115.58	115.58
WTFRD	10.12	10.12	11.56	14.29	44.655	74.76	74.76

Means and Std Deviations

Level	Number	Mean	Std Dev	Std Err Mean
TID	5	48.9660	47.8678	21.407
WTFRD	5	25.3440	27.6803	12.379

Means Comparisons

Dif=Mean[i]-Mean[j]	TID	WTFRD
TID	0.0000	23.6220
WTFRD	-23.6220	0.0000

Alpha= 0.05

Comparisons for all pairs using Tukey-Kramer HSD

q*

2.30593

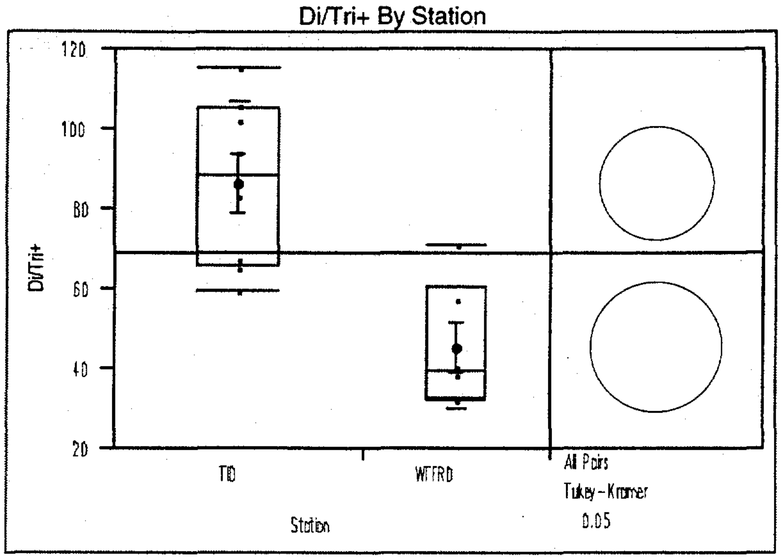
Abs(Dif)-LSD	TID	WTFRD
TID	-57.0224	-33.4004
WTFRD	-33.4004	-57.0224

Positive values show pairs of means that are significantly different.

Source: Hudson River Database Release 4.1

Figure A-3

Di/Tri+ Mass Ratio in USEPA Phase 2 Samples at the TI Dam and Waterford Stations - Fall, Winter and Spring



Quantiles

Level	minimum	10.0%	25.0%	median	75.0%	90.0%	maximum
TID	59.64	59.64	66.1525	88.815	105.28	115.58	115.58
WTFRD	32.57	32.57	32.705	39.68	61.0525	71.17	71.17

Means and Std Deviations

Level	Number	Mean	Std Dev	Std Err Mean
TID	8	86.8350	20.9416	7.4040
WTFRD	6	45.5883	15.5330	6.3413

Means Comparisons

Dif=Mean[i]-Mean[j]	TID	WTFRD
TID	0.0000	41.2467
WTFRD	-41.2467	0.0000

Alpha= 0.05

Comparisons for all pairs using Tukey-Kramer HSD

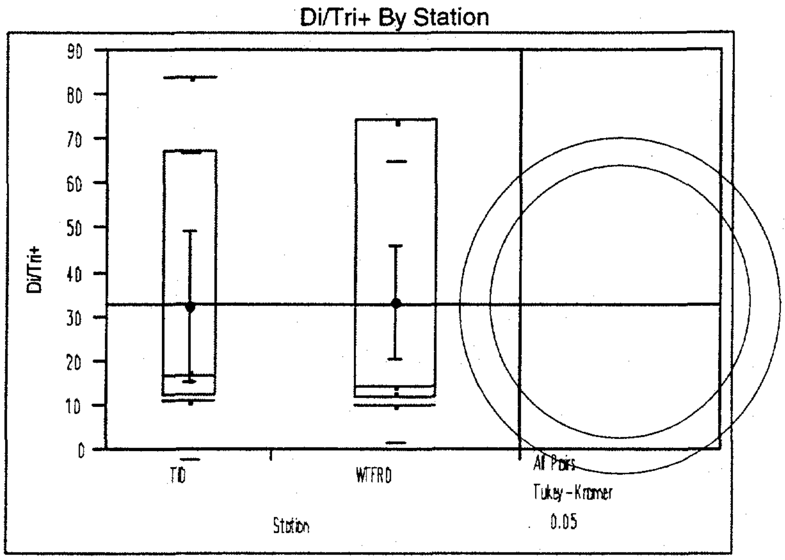
q*
2.17880

Abs(Dif)-LSD	TID	WTFRD
TID	-20.5649	19.0341
WTFRD	19.0341	-23.7463

Positive values show pairs of means that are significantly different.

Source: Hudson River Database Release 4.1

Figure A-4
Di/Tri+ Mass Ratio in USEPA Phase 2 Samples at the TI Dam and Waterford Stations - Low Flow



Quantiles

Level	minimum	10.0%	25.0%	median	75.0%	90.0%	maximum
TID	11.2	11.2	12.4075	16.905	67.625	84.24	84.24
WTFRD	10.12	10.12	12.28	14.42	74.4675	74.76	74.76

Means and Std Deviations

Level	Number	Mean	Std Dev	Std Err Mean
TID	4	32.3125	34.7300	17.365
WTFRD	6	33.5150	31.8363	12.997

Means Comparisons

Dif=Mean[i]-Mean[j]	WTFRD	TID
WTFRD	0.00000	1.20250
TID	-1.20250	0.00000

Alpha= 0.05

Comparisons for all pairs using Tukey-Kramer HSD

q*

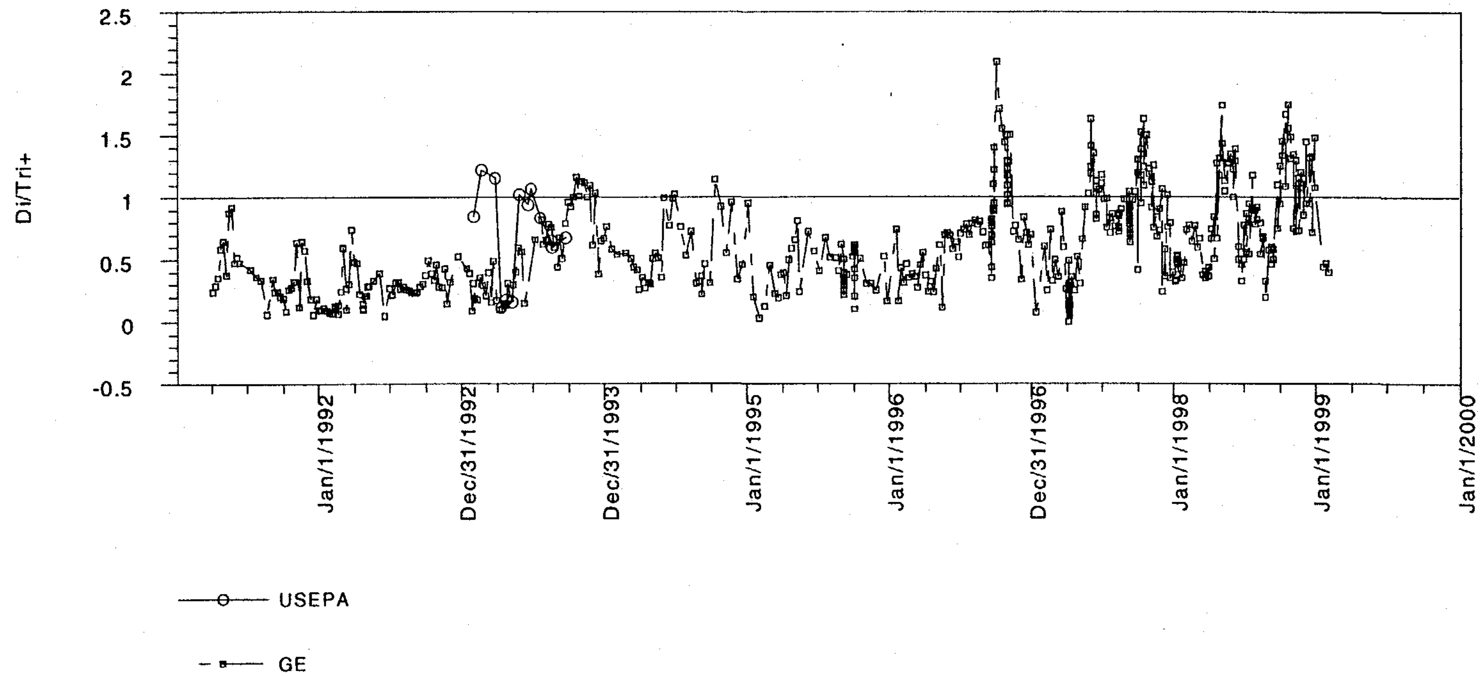
2.30593

Abs(Dif)-LSD	WTFRD	TID
WTFRD	-43.8689	-47.8444
TID	-47.8444	-53.7282

Positive values show pairs of means that are significantly different.

Source: Hudson River Database Release 4.1

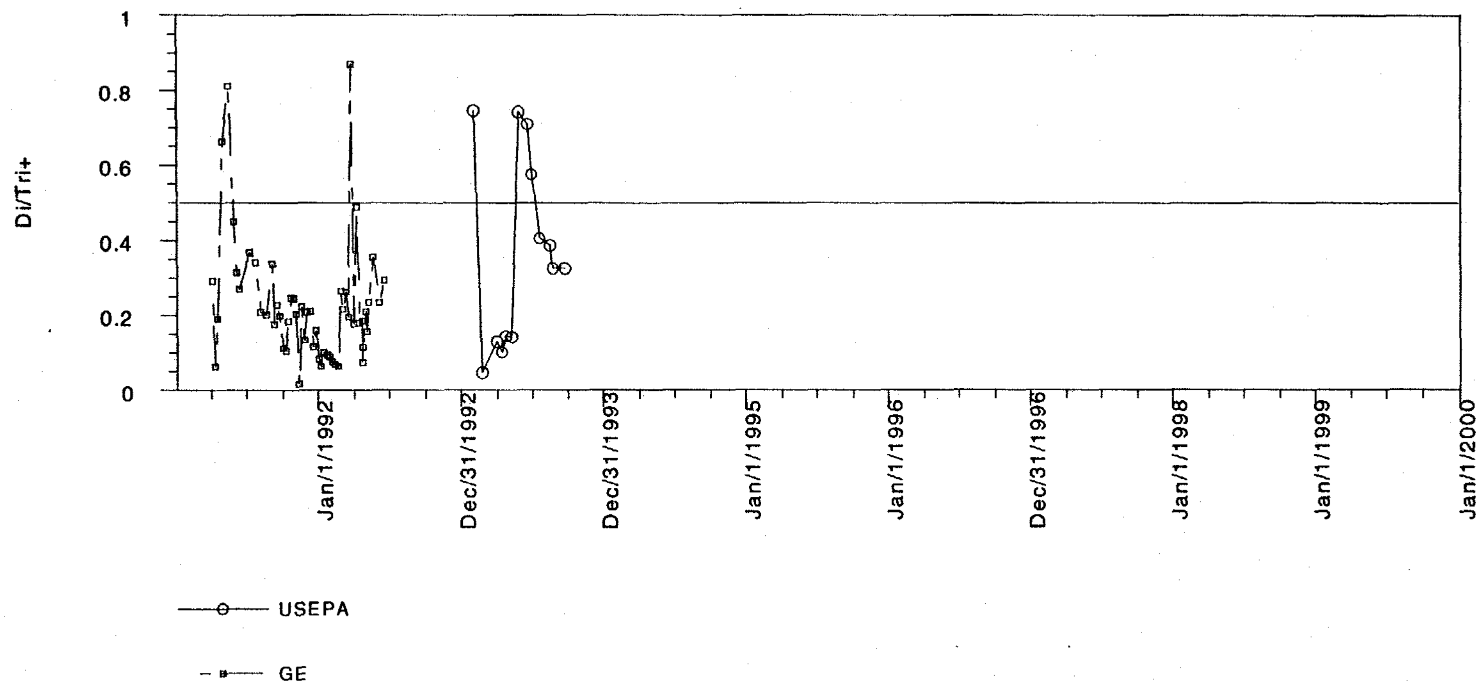
Figure A-5
Di/Tri+ Mass Ratio in USEPA Phase 2 Samples at the TI Dam and Waterford Stations – High Flow



Source: Hudson River Database Release 4.1

Figure A-6
Di/Tri+ Mass Ratio in USEPA and General Electric Water Column Samples
at the Thompson Island Dam

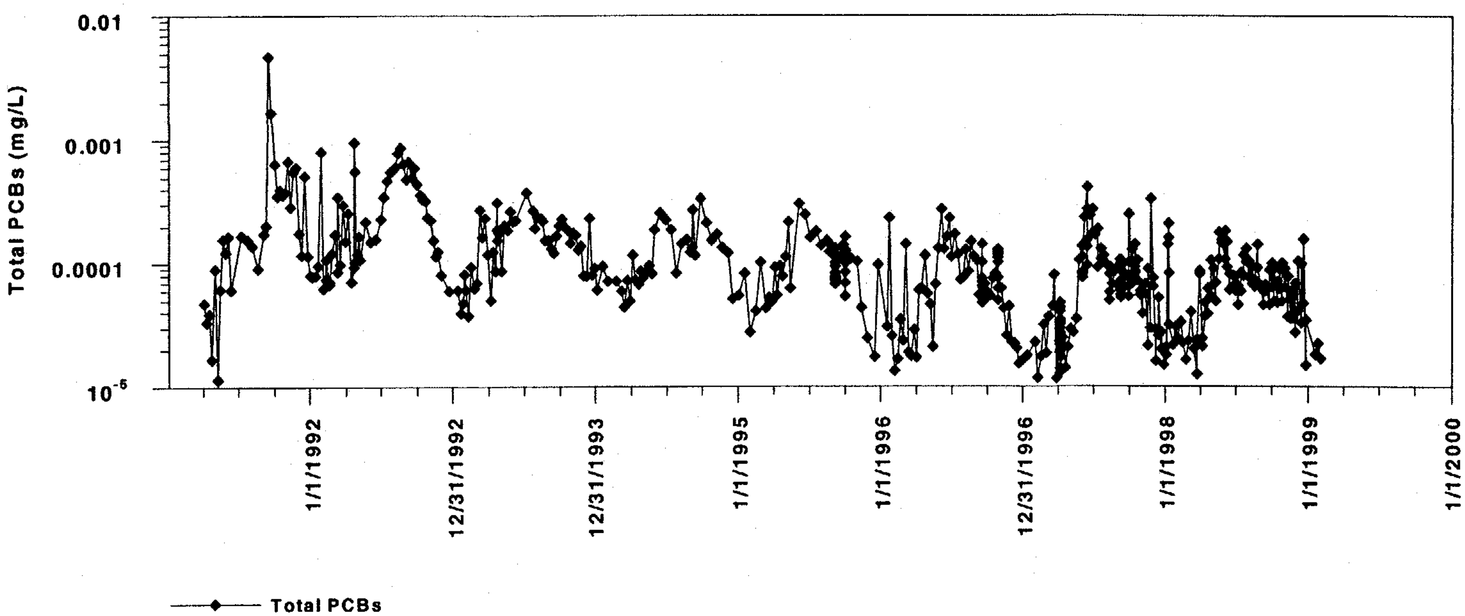
302891



Source: Hudson River Database Release 4.1

Figure A-7
Di/Tri+ Mass Ratio in USEPA and General Electric Water Column Samples
at Waterford

302892

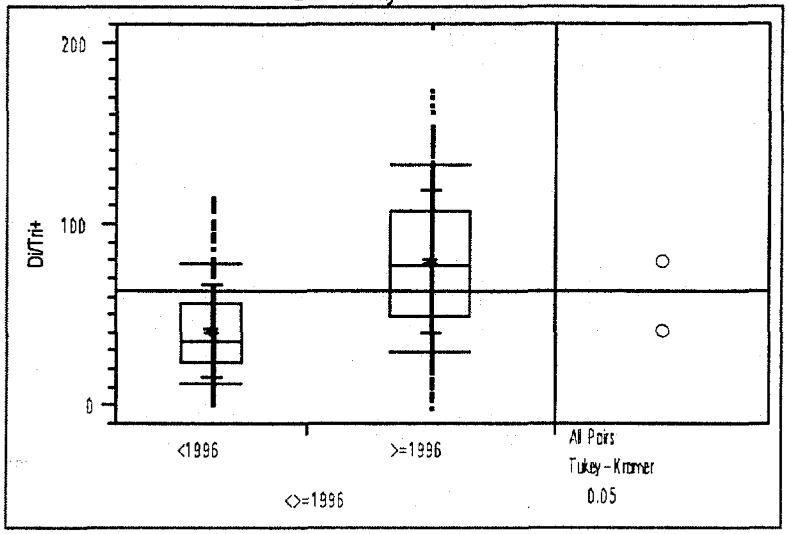


Source: Hudson River Database Release 4.1

Figure A-8
Total PCBs in General Electric Water Column Samples
at the Thompson Island Dam

302893

Di/Tri+ By <>=1996



Quantiles

Level	minimum	10.0%	25.0%	median	75.0%	90.0%	maximum
<1996	2.76	12.816	23.755	35.47	56.73	78.752	115.75
>=1996	0	29.514	50.05	77.49	107.76	133.368	209.28

Means and Std Deviations

Level	Number	Mean	Std Dev	Std Err Mean
<1996	225	42.1256	25.5812	1.7054
>=1996	293	79.9830	39.3904	2.3012

Means Comparisons

Dif=Mean[i]-Mean[j]	>=1996	<1996
>=1996	0.0000	37.8574
<1996	-37.8574	0.0000

Alpha= 0.05

Comparisons for all pairs using Tukey-Kramer HSD

q*

1.96457

Abs(Dif)-LSD	>=1996	<1996
>=1996	-5.5332	31.9209
<1996	31.9209	-6.3142

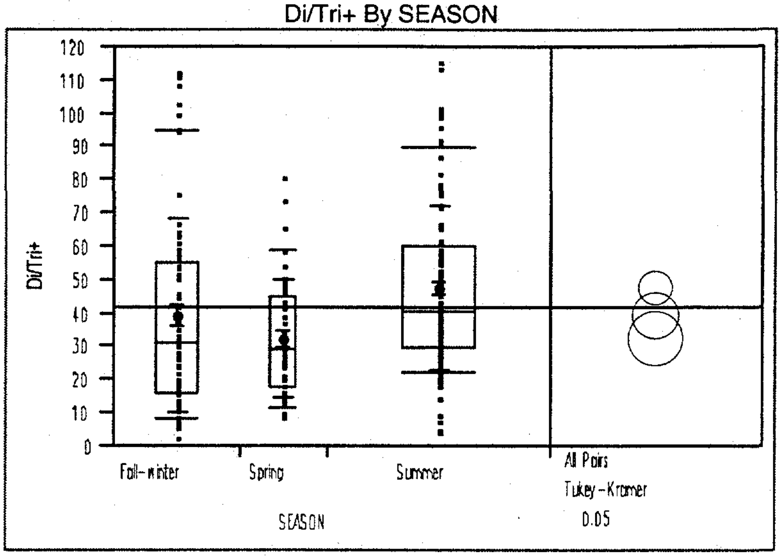
Positive values show pairs of means that are significantly different.

Source: Hudson River Database Release 4.1

Figure A-9

Di/Tri+ Mass Ratio in General Electric Samples at the TI Dam Grouped by Years

Di/Tri+ By SEASON



Quantiles

Level	minimum	10.0%	25.0%	median	75.0%	90.0%	maximum
Fall-winter	2.76	8.732	15.8275	31.31	55.4925	95.303	112.5
Spring	9.34	11.96	17.855	29.67	45.305	58.962	81.09
Summer	4.72	22.16	30.21	40.835	60.1625	89.935	115.75

Means and Std Deviations

Level	Number	Mean	Std Dev	Std Err Mean
Fall-winter	66	39.4198	29.1977	3.5940
Spring	45	32.3282	17.9895	2.6817
Summer	114	47.5595	24.6685	2.3104

Means Comparisons

Dif=Mean[i]-Mean[j]	Summer	Fall-winter	Spring
Summer	0.0000	8.1396	15.2313
Fall-winter	-8.1396	0.0000	7.0916
Spring	-15.2313	-7.0916	0.0000

Alpha= 0.05

Comparisons for all pairs using Tukey-Kramer HSD

q*

2.35960

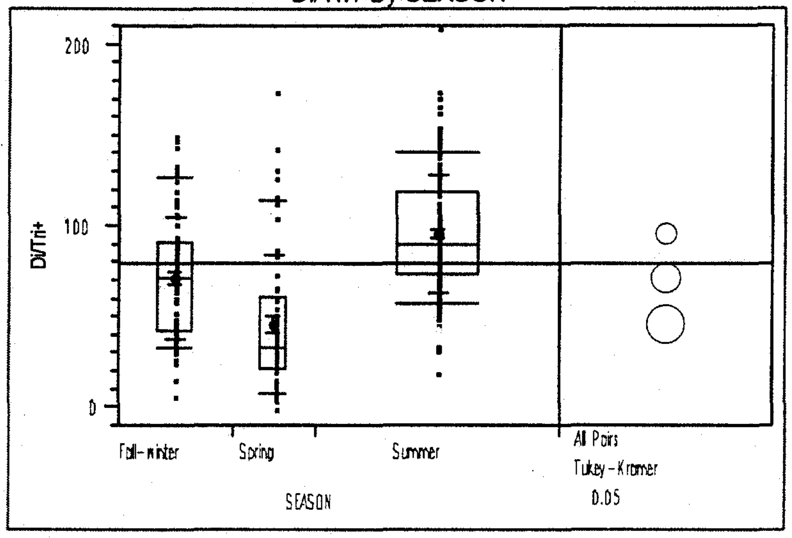
Abs(Dif)-LSD	Summer	Fall-winter	Spring
Summer	-7.8040	-0.9735	4.8584
Fall-winter	-0.9735	-10.2565	-4.2988
Spring	4.8584	-4.2988	-12.4212

Positive values show pairs of means that are significantly different.

Source: Hudson River Database Release 4.1

Figure A-10
Di/Tri+ Mass Ratio in General Electric Samples at the TI Dam
Grouped by Season (1991-1995)

Di/Tri+ By SEASON



Quantiles

Level	minimum	10.0%	25.0%	median	75.0%	90.0%	maximum
Fall-winter	7.03	33.186	42.4	71.555	91.6575	126.802	150.34
Spring	0	7.504	21.5025	33.395	60.9475	114.419	174.33
Summer	19.56	57.832	73.93	90.51	119.395	141.026	209.28

Means and Std Deviations

Level	Number	Mean	Std Dev	Std Err Mean
Fall-winter	76	71.4116	34.0453	3.9053
Spring	56	45.8050	38.7849	5.1828
Summer	161	95.9172	32.7418	2.5804

Means Comparisons

Dif=Mean[i]-Mean[j]	Summer	Fall-winter	Spring
Summer	0.0000	24.5056	50.1122
Fall-winter	-24.5056	0.0000	25.6066
Spring	-50.1122	-25.6066	0.0000

Alpha= 0.05

Comparisons for all pairs using Tukey-Kramer HSD

q*

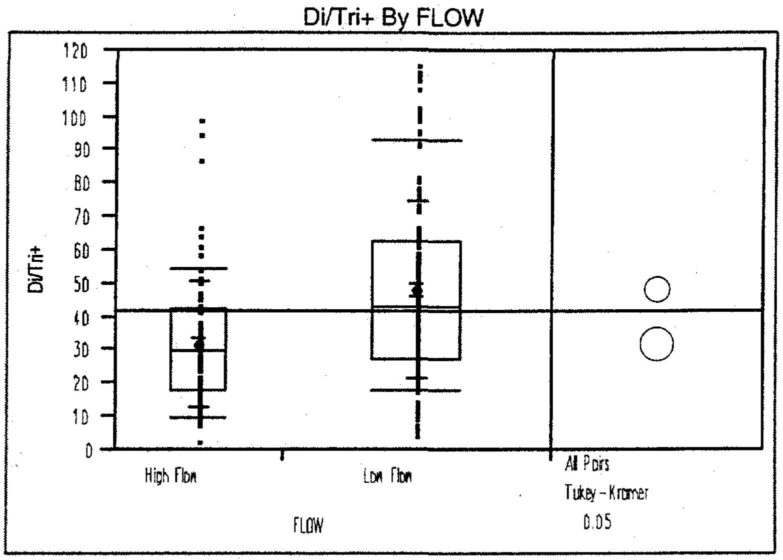
2.35588

Abs(Dif)-LSD	Summer	Fall-winter	Spring
Summer	-9.0065	13.2594	37.5757
Fall-winter	13.2594	-13.1087	11.3755
Spring	37.5757	11.3755	-15.2712

Positive values show pairs of means that are significantly different.

Source: Hudson River Database Release 4.1

Figure A-11
Di/Tri+ Mass Ratio in General Electric Samples at the TI Dam
Grouped by Season (1996-1998)



Quantiles

Level	minimum	10.0%	25.0%	median	75.0%	90.0%	maximum
High Flow	2.76	9.658	18.19	30.12	42.55	54.952	99.59
Low Flow	4.72	17.883	27.2525	43	62.5625	93.017	115.75

Means and Std Deviations

Level	Number	Mean	Std Dev	Std Err Mean
High Flow	87	32.1717	19.1418	2.0522
Low Flow	138	48.4009	27.1546	2.3116

Means Comparisons

Dif=Mean[i]-Mean[j]	Low Flow	High Flow
Low Flow	0.0000	16.2291
High Flow	-16.2291	0.0000

Alpha= 0.05

Comparisons for all pairs using Tukey-Kramer HSD

q*

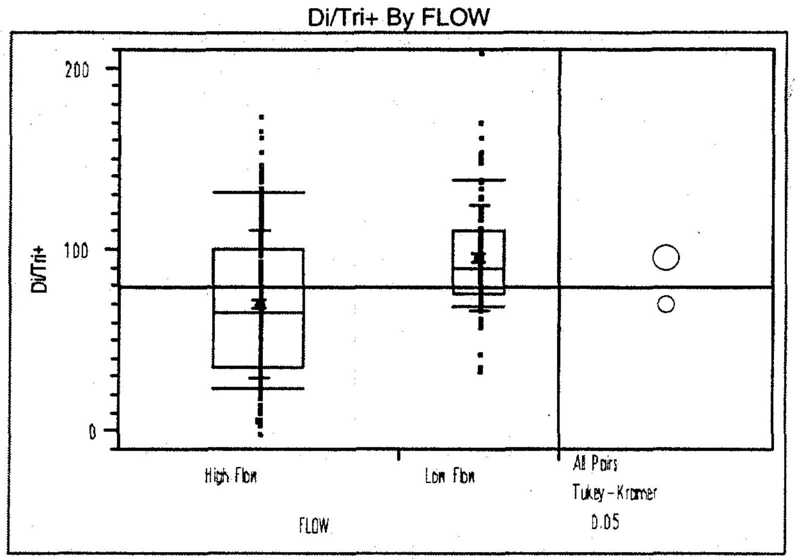
1.97066

Abs(Dif)-LSD	Low Flow	High Flow
Low Flow	-5.78354	9.65241
High Flow	9.65241	-7.28406

Positive values show pairs of means that are significantly different.

Source: Hudson River Database Release 4.1

Figure A-12
Di/Tri+ Mass Ratio in General Electric Samples at the TI Dam
Grouped by Flow (1991-1995)



Quantiles

Level	minimum	10.0%	25.0%	median	75.0%	90.0%	maximum
High Flow	0	23.702	36.07	66.1	100.16	131.4	174.44
Low Flow	34.84	69.141	76.41	89.6	111.5575	139.054	209.28

Means and Std Deviations

Level	Number	Mean	Std Dev	Std Err Mean
High Flow	187	70.6428	41.3259	3.0220
Low Flow	106	96.4606	29.3286	2.8486

Means Comparisons

Dif=Mean[i]-Mean[j]	Low Flow	High Flow
Low Flow	0.0000	25.8177
High Flow	-25.8177	0.0000

Alpha= 0.05

Comparisons for all pairs using Tukey-Kramer HSD

q*

1.96815

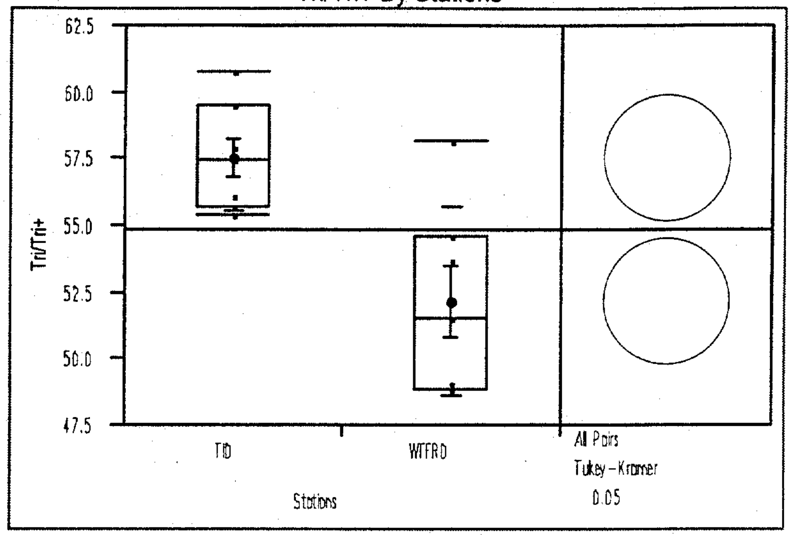
Abs(Dif)-LSD	Low Flow	High Flow
Low Flow	-10.1225	16.8581
High Flow	16.8581	-7.6212

Positive values show pairs of means that are significantly different.

Source: Hudson River Database Release 4.1

Figure A-13
Di/Tri+ Mass Ratio in General Electric Samples at the TI Dam
Grouped by Flow (1996-1998)

Tri/Tri+ By Stations



Quantiles

Level	minimum	10.0%	25.0%	median	75.0%	90.0%	maximum
TID	55.42	55.42	55.74	57.48	59.56	60.84	60.84
WTFRD	48.92	48.92	48.92	51.6	54.62	58.21	58.21

Means and Std Deviations

Level	Number	Mean	Std Dev	Std Err Mean
TID	7	57.5943	2.03177	0.7679
WTFRD	7	52.1600	3.55562	1.3439

Means Comparisons

Dif=Mean[i]-Mean[j]	TID	WTFRD
TID	0.00000	5.43429
WTFRD	-5.43429	0.00000

Alpha= 0.05

Comparisons for all pairs using Tukey-Kramer HSD

q*

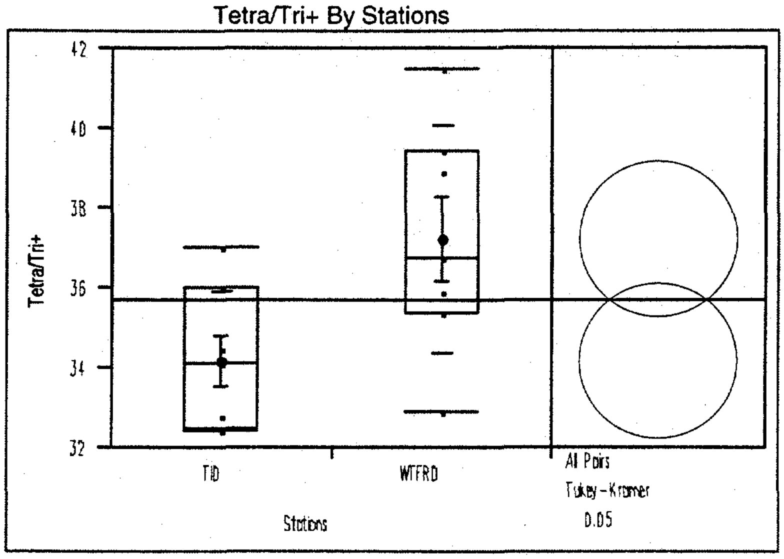
2.17880

Abs(Dif)-LSD	TID	WTFRD
TID	-3.37242	2.06187
WTFRD	2.06187	-3.37242

Positive values show pairs of means that are significantly different.

Source: Hudson River Database Release 4.1

Figure A-14
Tri/Tri+ Mass Ratio in USEPA Phase 2 Samples at the TI Dam and Waterford Stations - Summer



Quantiles

Level	minimum	10.0%	25.0%	median	75.0%	90.0%	maximum
TID	32.47	32.47	32.49	34.14	36.03	37.02	37.02
WTFRD	32.91	32.91	35.37	36.75	39.46	41.48	41.48

Means and Std Deviations

Level	Number	Mean	Std Dev	Std Err Mean
TID	7	34.2100	1.78649	0.6752
WTFRD	7	37.2629	2.88486	1.0904

Means Comparisons

Dif=Mean[i]-Mean[j]	WTFRD	TID
WTFRD	0.00000	3.05286
TID	-3.05286	0.00000

Alpha= 0.05

Comparisons for all pairs using Tukey-Kramer HSD

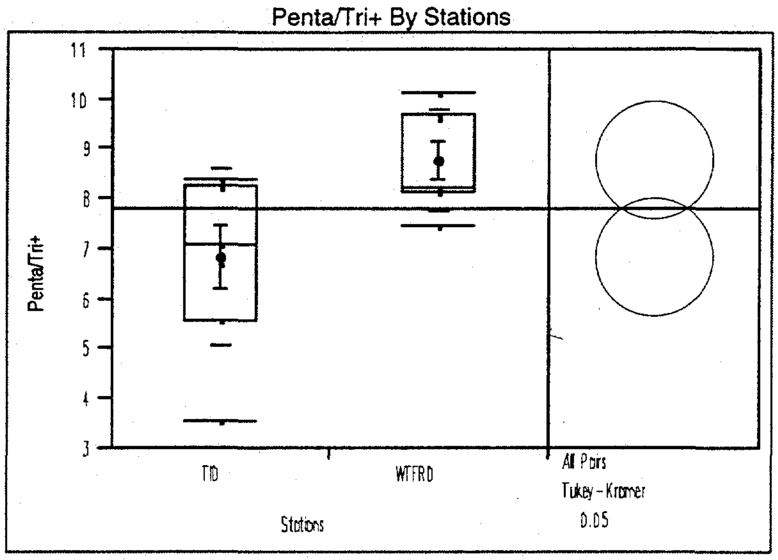
q*
2.17880

Abs(Dif)-LSD	WTFRD	TID
WTFRD	-2.79434	0.25852
TID	0.25852	-2.79434

Positive values show pairs of means that are significantly different.

Source: Hudson River Database Release 4.1

Figure A-15
Tetra/Tri+ Mass Ratio in USEPA Phase 2 Samples at the TI Dam and Waterford Stations - Summer



Quantiles

Level	minimum	10.0%	25.0%	median	75.0%	90.0%	maximum
TID	3.58	3.58	5.61	7.1	8.27	8.43	8.43
WTFRD	7.47	7.47	8.14	8.26	9.73	10.15	10.15

Means and Std Deviations

Level	Number	Mean	Std Dev	Std Err Mean
TID	7	6.85143	1.76931	0.66874
WTFRD	7	8.79857	1.01821	0.38485

Means Comparisons

Dif=Mean[i]-Mean[j]	WTFRD	TID
WTFRD	0.00000	1.94714
TID	-1.94714	0.00000

Alpha= 0.05

Comparisons for all pairs using Tukey-Kramer HSD

q*

2.17880

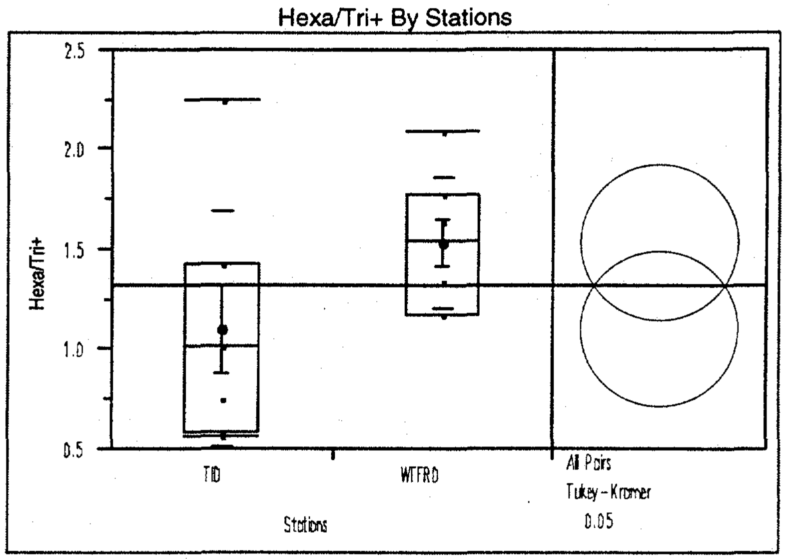
Abs(Dif)-LSD	WTFRD	TID
WTFRD	-1.68109	0.26606
TID	0.26606	-1.68109

Positive values show pairs of means that are significantly different.

Source: Hudson River Database Release 4.1

Figure A-16

Penta/Tri+ Mass Ratio in USEPA Phase 2 Samples at the TI Dam and Waterford Stations - Summer



Quantiles

Level	minimum	10.0%	25.0%	median	75.0%	90.0%	maximum
TID	0.57	0.57	0.59	1.02	1.43	2.25	2.25
WTFRD	1.18	1.18	1.18	1.55	1.78	2.09	2.09

Means and Std Deviations

Level	Number	Mean	Std Dev	Std Err Mean
TID	7	1.10429	0.590815	0.22331
WTFRD	7	1.53857	0.333038	0.12588

Means Comparisons

Dif=Mean[i]-Mean[j]	WTFRD	TID
WTFRD	0.000000	0.434286
TID	-0.43429	0.000000

Alpha= 0.05

Comparisons for all pairs using Tukey-Kramer HSD

q*

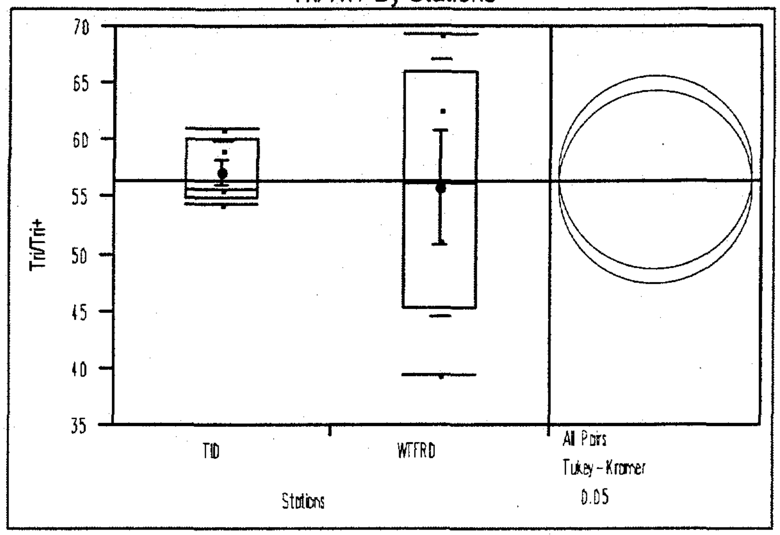
2.17880

Abs(Dif)-LSD	WTFRD	TID
WTFRD	-0.55852	-0.12423
TID	-0.12423	-0.55852

Positive values show pairs of means that are significantly different.

Source: Hudson River Database Release 4.1

Figure A-17
Hexa/Tri+ Mass Ratio in USEPA Phase 2 Samples at the TI Dam and Waterford Stations - Summer



Quantiles

Level	minimum	10.0%	25.0%	median	75.0%	90.0%	maximum
TID	54.44	54.44	55.04	55.71	60.09	61.01	61.01
WTFRD	39.57	39.57	45.48	56.27	66.025	69.39	69.39

Means and Std Deviations

Level	Number	Mean	Std Dev	Std Err Mean
TID	5	57.1940	2.7689	1.2383
WTFRD	5	55.8560	11.3448	5.0735

Means Comparisons

Dif=Mean[i]-Mean[j]	TID	WTFRD
TID	0.00000	1.33800
WTFRD	-1.33800	0.00000

Alpha= 0.05

Comparisons for all pairs using Tukey-Kramer HSD

q*

2.30593

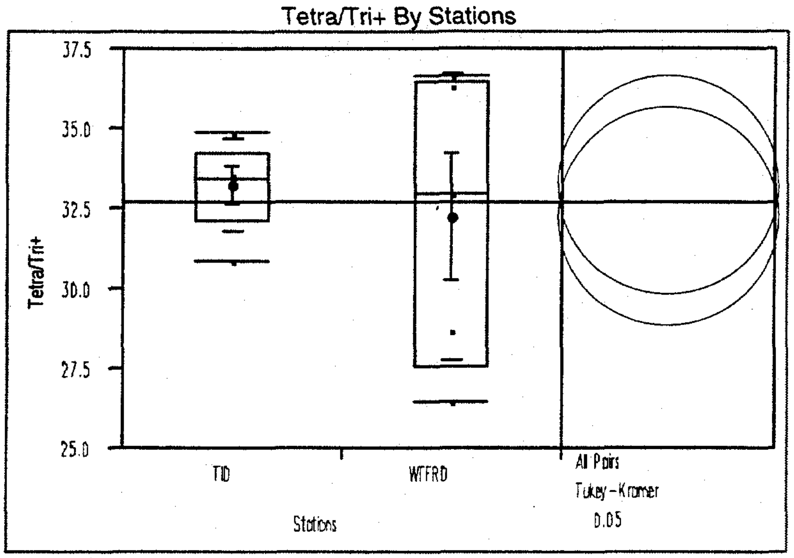
Abs(Dif)-LSD	TID	WTFRD
TID	-12.0426	-10.7046
WTFRD	-10.7046	-12.0426

Positive values show pairs of means that are significantly different.

Source: Hudson River Database Release 4.1

Figure A-18

Tri/Tri+ Mass Ratio in USEPA Phase 2 Samples at the TI Dam and Waterford Stations - Fall, Winter and Spring



Quantiles

Level	minimum	10.0%	25.0%	median	75.0%	90.0%	maximum
TID	30.89	30.89	32.155	33.45	34.235	34.87	34.87
WTFRD	26.5	26.5	27.62	32.98	36.49	36.66	36.66

Means and Std Deviations

Level	Number	Mean	Std Dev	Std Err Mean
TID	5	33.2460	1.44787	0.6475
WTFRD	5	32.2400	4.52570	2.0240

Means Comparisons

Dif=Mean[i]-Mean[j]	TID	WTFRD
TID	0.00000	1.00600
WTFRD	-1.00600	0.00000

Alpha= 0.05

Comparisons for all pairs using Tukey-Kramer HSD

q*

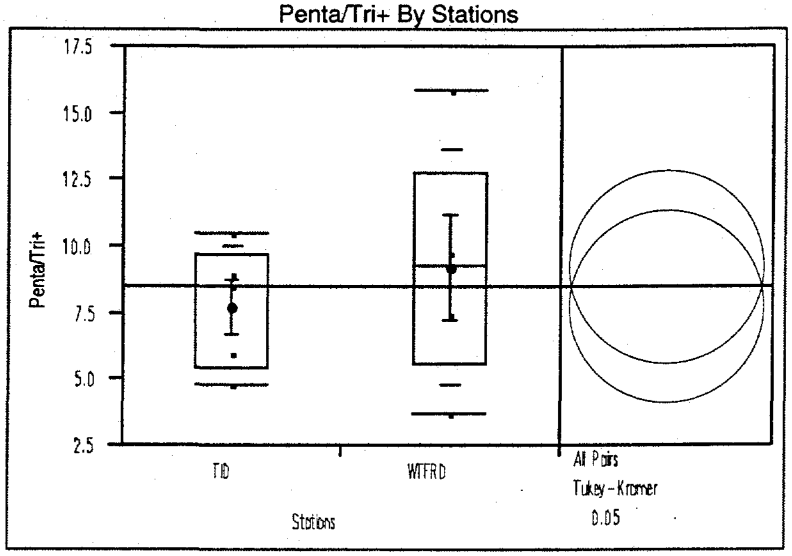
2.30593

Abs(Dif)-LSD	TID	WTFRD
TID	-4.90012	-3.89412
WTFRD	-3.89412	-4.90012

Positive values show pairs of means that are significantly different.

Source: Hudson River Database Release 4.1

Figure A-19
Tetra/Tri+ Mass Ratio in USEPA Phase 2 Samples at the TI Dam and Waterford Stations - Fall, Winter and Spring



Quantiles

Level	minimum	10.0%	25.0%	median	75.0%	90.0%	maximum
TID	4.84	4.84	5.435	8.54	9.765	10.53	10.53
WTFRD	3.71	3.71	5.6	9.34	12.84	15.91	15.91

Means and Std Deviations

Level	Number	Mean	Std Dev	Std Err Mean
TID	5	7.78800	2.30945	1.0328
WTFRD	5	9.24400	4.42784	1.9802

Means Comparisons

Dif=Mean[i]-Mean[j]	WTFRD	TID
WTFRD	0.00000	1.45600
TID	-1.45600	0.00000

Alpha= 0.05

Comparisons for all pairs using Tukey-Kramer HSD

q*

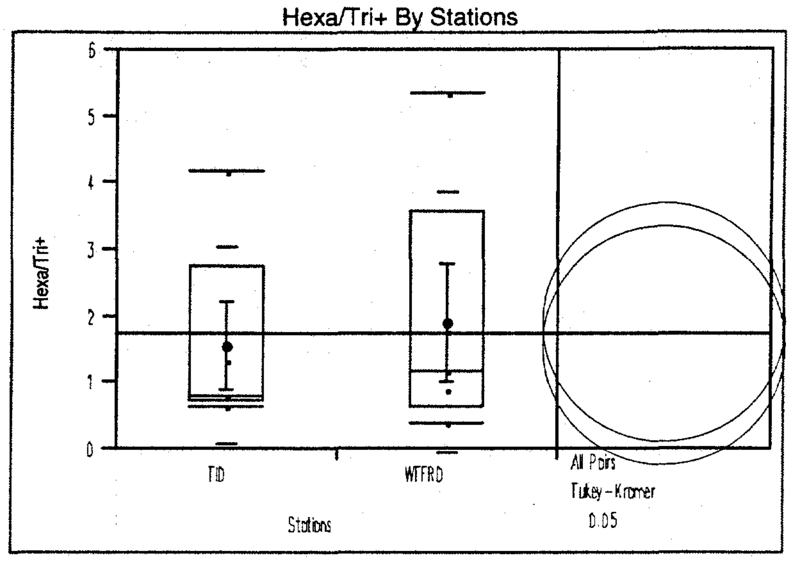
2.30593

Abs(Dif)-LSD	WTFRD	TID
WTFRD	-5.14996	-3.69396
TID	-3.69396	-5.14996

Positive values show pairs of means that are significantly different.

Source: Hudson River Database Release 4.1

Figure A-20
Penta/Tri+ Mass Ratio in USEPA Phase 2 Samples at the TI Dam and Waterford Stations - Fall, Winter and Spring



Quantiles

Level	minimum	10.0%	25.0%	median	75.0%	90.0%	maximum
TID	0.66	0.66	0.73	0.82	2.775	4.2	4.2
WTFRD	0.38	0.38	0.645	1.19	3.575	5.35	5.35

Means and Std Deviations

Level	Number	Mean	Std Dev	Std Err Mean
TID	5	1.56600	1.49572	0.66891
WTFRD	5	1.92600	1.98140	0.88611

Means Comparisons

Dif=Mean[i]-Mean[j]	WTFRD	TID
WTFRD	0.000000	0.360000
TID	-0.36	0.000000

Alpha= 0.05

Comparisons for all pairs using Tukey-Kramer HSD

q*

2.30593

Abs(Dif)-LSD	WTFRD	TID
WTFRD	-2.56012	-2.20012
TID	-2.20012	-2.56012

Positive values show pairs of means that are significantly different.

Source: Hudson River Database Release 4.1

Figure A-21
Hexa/Tri+ Mass Ratio in USEPA Phase 2 Samples at the TI Dam and Waterford Stations
- Fall, Winter and Spring

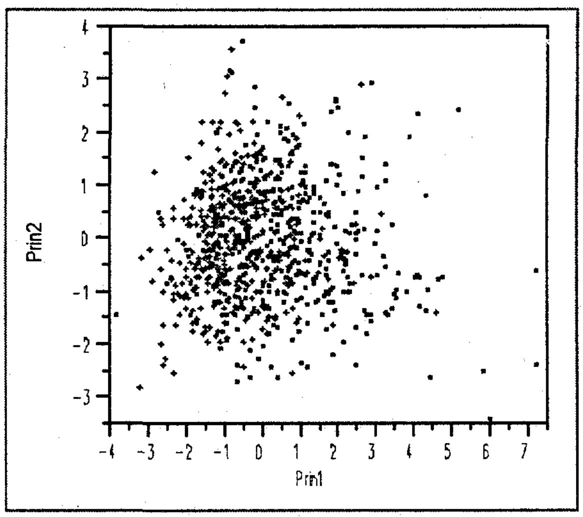
Correlations

Variable	Tri/Tri+	Tetra/Tri+	Penta/Tri+	Hexa/Tri+
Tri/Tri+	1.0000	-0.4212	-0.7396	-0.6588
Tetra/Tri+	-0.4212	1.0000	-0.2161	-0.1544
Penta/Tri+	-0.7396	-0.2161	1.0000	0.5716
Hexa/Tri+	-0.6588	-0.1544	0.5716	1.0000

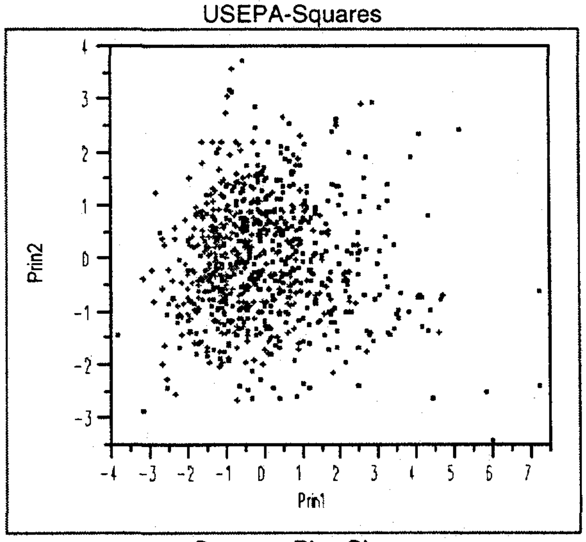
Prin. Components / Factor Analysis

Principal Components

EigenValue:	2.3172	1.2456	0.4360	0.0012
Percent:	57.9312	31.1402	10.8997	0.0290
CumPercent:	57.9312	89.0713	99.9710	100.0000



General Electric-Plus Sign
USEPA-Squares

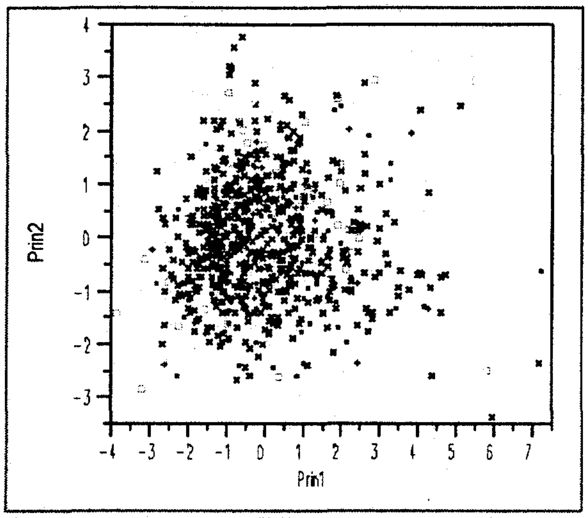


Summer-Plus Sign
Fall, Winter and Spring-Squares

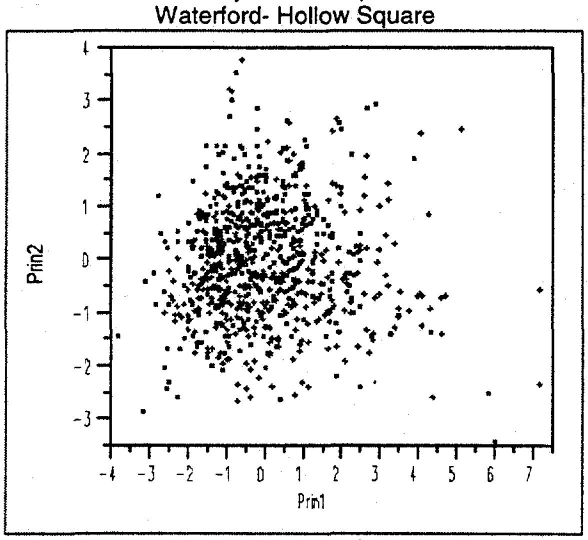
Figure A-22

Principal Components Analysis for USEPA and General Electric Water Column Samples at TI Dam, Schuylerville, Stillwater and Waterford 1991-1998

Page 1 of 3



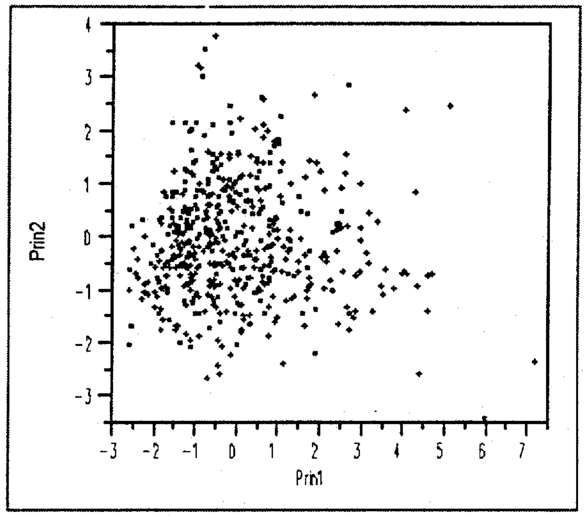
TID- X
Stillwater-Plus Sign
Schuylerville –Square
Waterford- Hollow Square



Low Flow-Plus Sign
HighFlow-Squares

Source: Hudson River Database Release 4.1

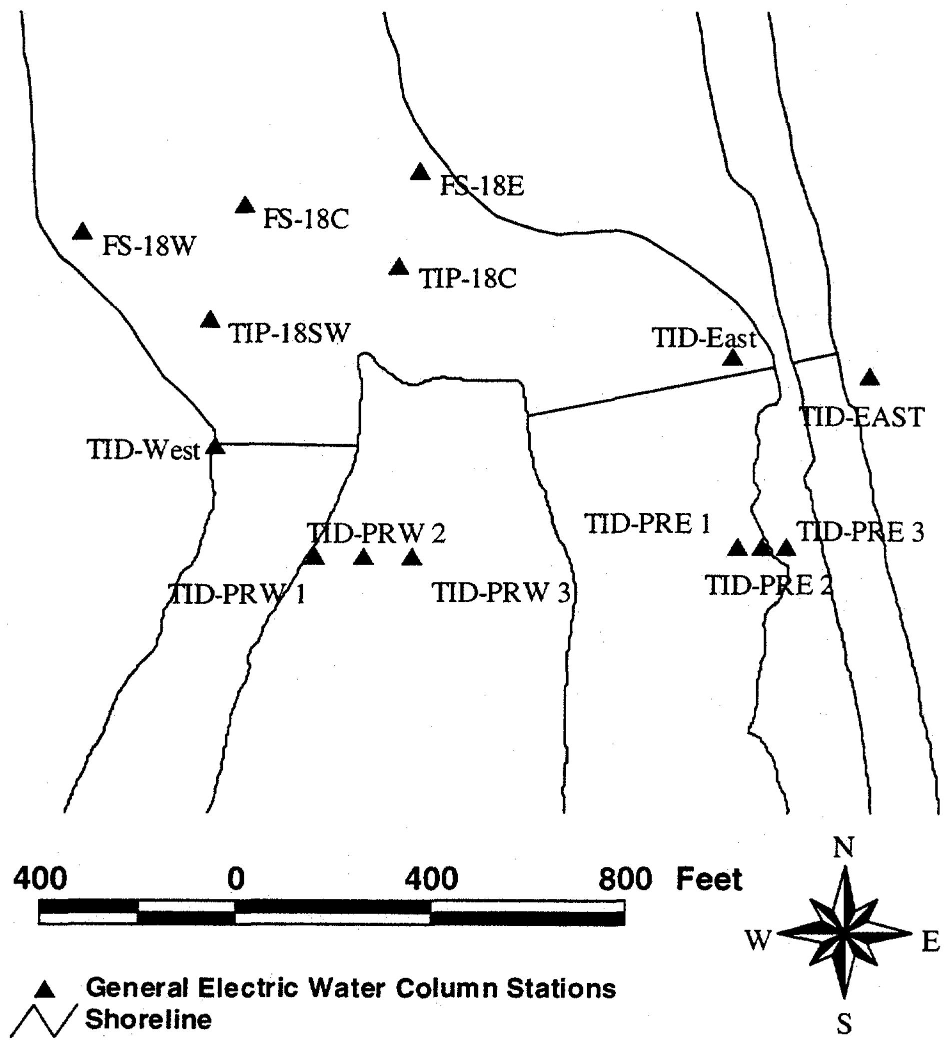
Figure A-22
Principal Components Analysis for USEPA and General Electric Water Column Samples at TI Dam, Schuylerville, Stillwater and Waterford 1991-1998
Page 2 of 3



1996-1998-Plus Sign
1991-1995-Squares

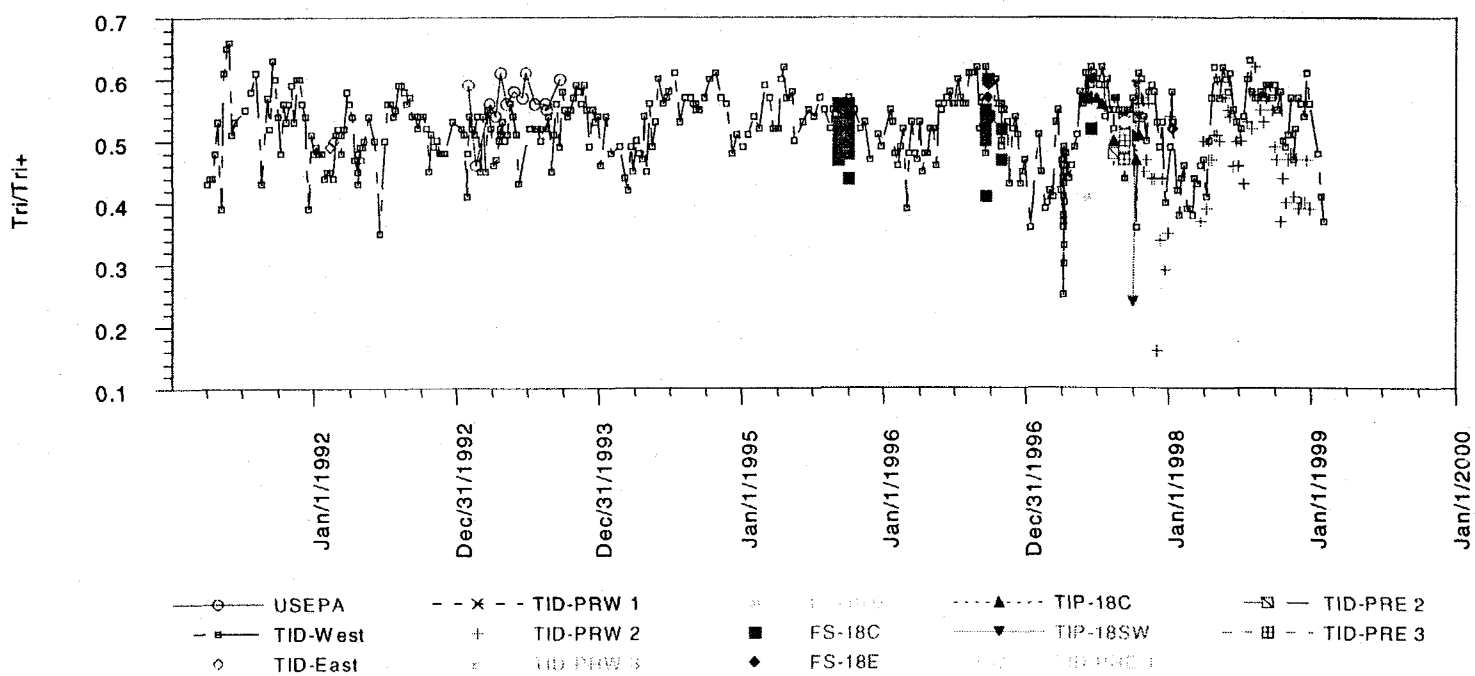
Source: Hudson River Database Release 4.1

Figure A-22
Principal Components Analysis for USEPA and General Electric Water Column Samples at TI Dam, Schuylerville, Stillwater and Waterford 1991-1998
Page 3 of 3



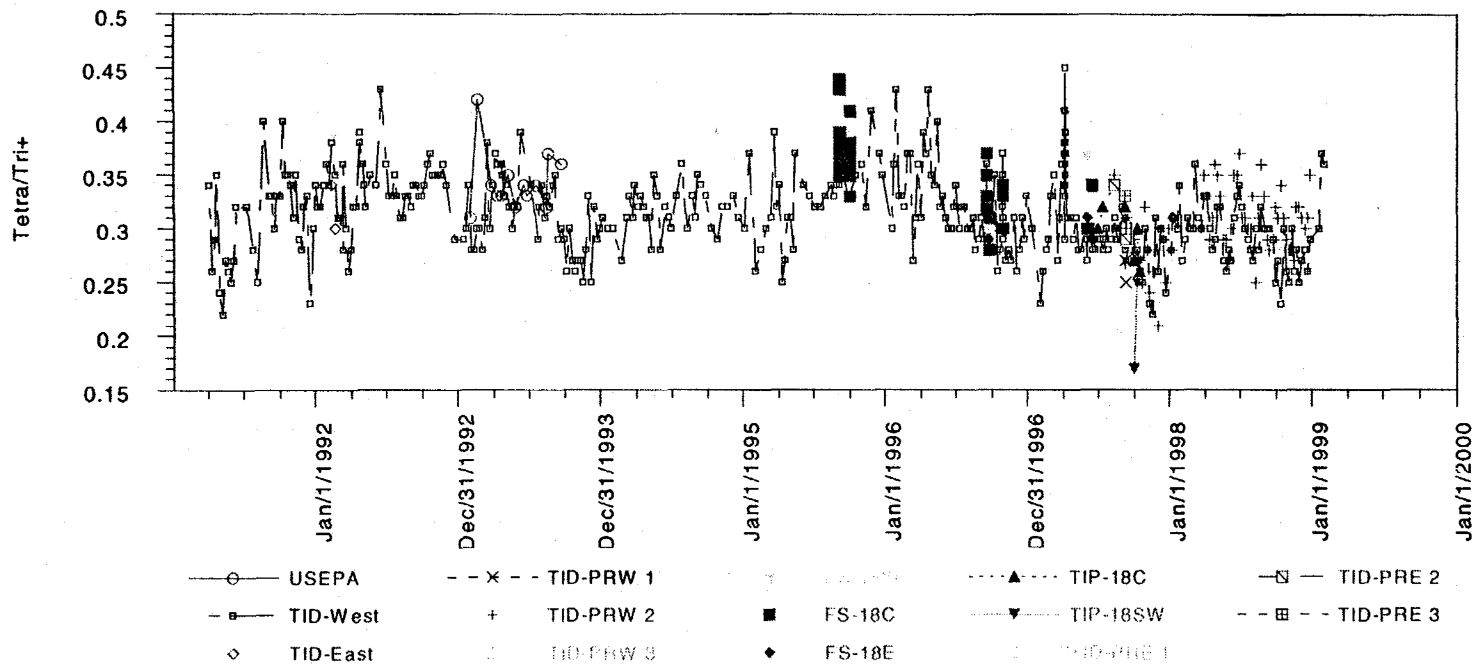
Source: Hudson River Database Release 4.1

Figure A-23
Location of General Electric Water Column Stations Near the Thompson Island Dam



Source: Hudson River Database Release 4.1

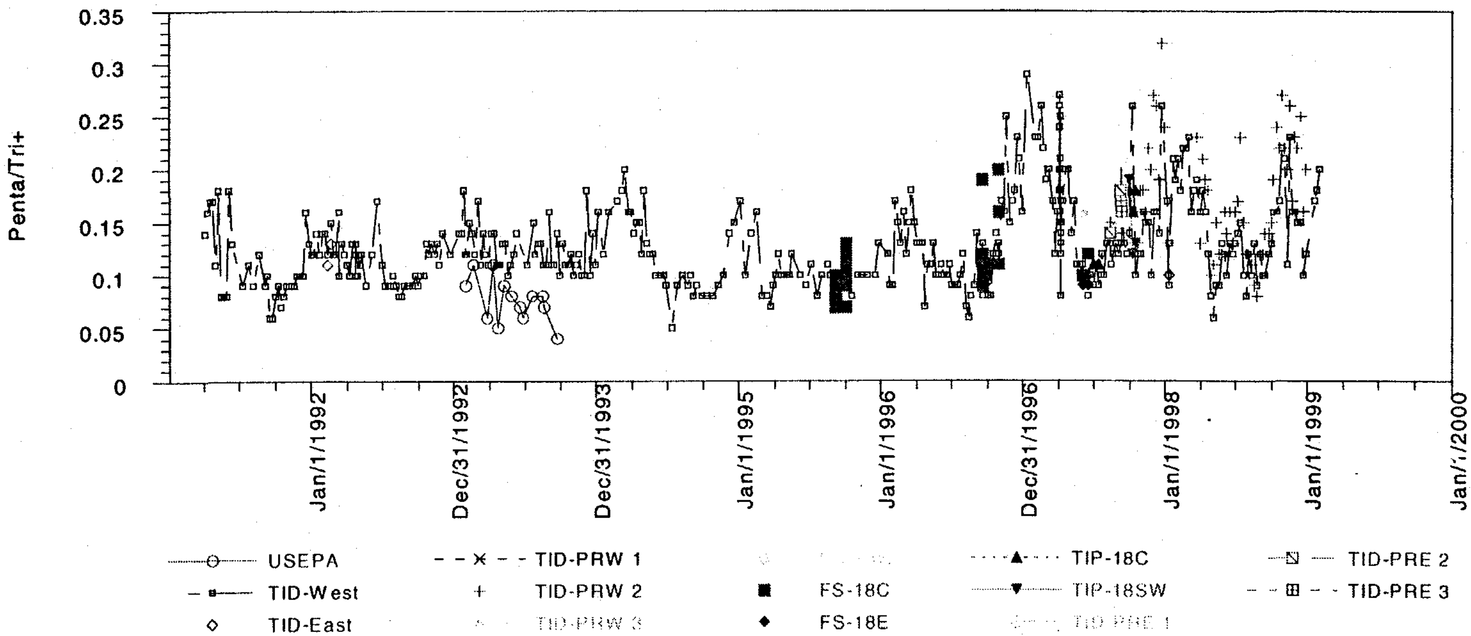
Figure A-24
Tri/Tri+ Mass Ratio in USEPA and General Electric Water Column Samples
at the Thompson Island Dam



Source: Hudson River Database Release 4.1

Figure A-25
Tetra/Tri+ Mass Ratio in USEPA and General Electric Water Column Samples
at the Thompson Island Dam

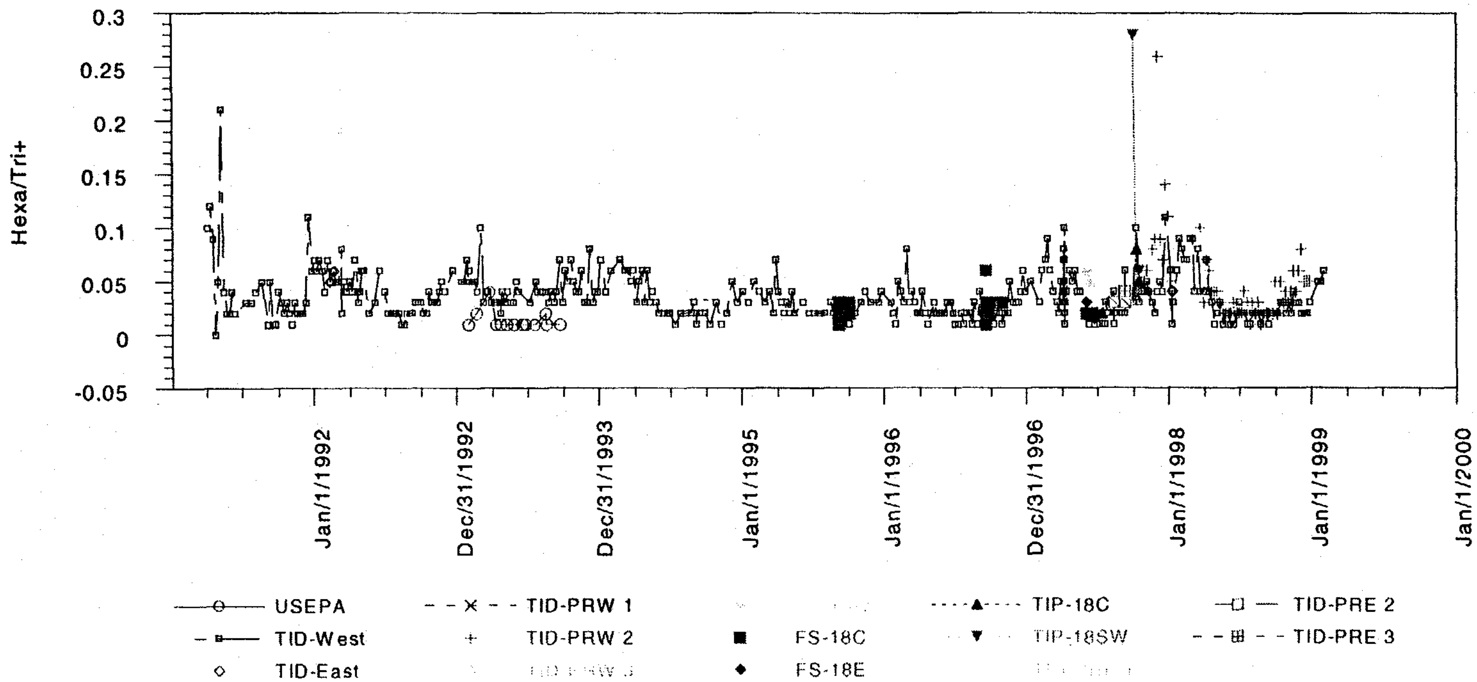
302912



Source: Hudson River Database Release 4.1

Figure A-26
Penta/Tri+ Mass Ratio in USEPA and General Electric Water Column Samples
at the Thompson Island Dam

302913

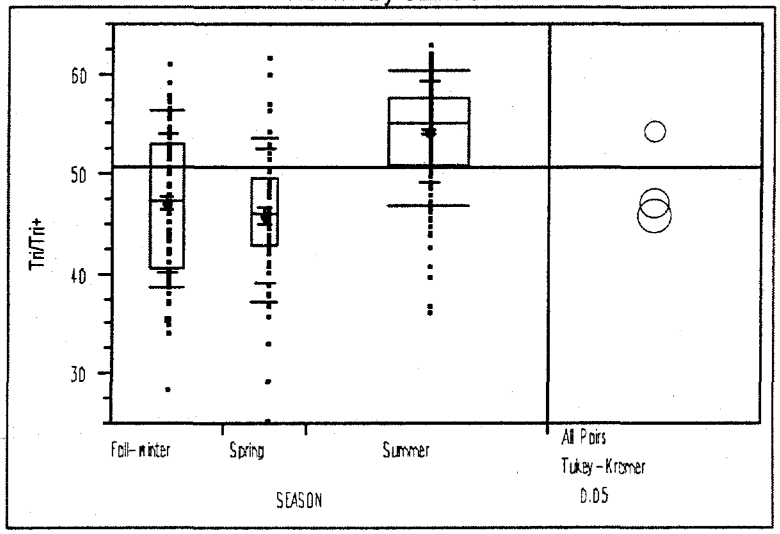


Source: Hudson River Database Release 4.1

Figure A-27
Hexa/Tri+ Mass Ratio in USEPA and General Electric Water Column Samples
at the Thompson Island Dam

302914

Tri/Tri+ By SEASON



Quantiles

Level	minimum	10.0%	25.0%	median	75.0%	90.0%	maximum
Fall-winter	28.78	38.734	40.69	47.355	53.04	56.456	61.36
Spring	25.47	37.28	42.935	46.265	49.725	53.652	61.99
Summer	36.45	46.986	50.92	55.24	57.665	60.39	63.3

Means and Std Deviations

Level	Number	Mean	Std Dev	Std Err Mean
Fall-winter	76	47.2103	7.02309	0.80560
Spring	56	45.9007	6.75424	0.90257
Summer	161	54.3002	5.19712	0.40959

Means Comparisons

Dif=Mean[i]-Mean[j]	Summer	Fall-winter	Spring
Summer	0.00000	7.08992	8.39947
Fall-winter	-7.08992	0.00000	1.30955
Spring	-8.39947	-1.30955	0.00000

Alpha= 0.05

Comparisons for all pairs using Tukey-Kramer HSD

q*
2.35588

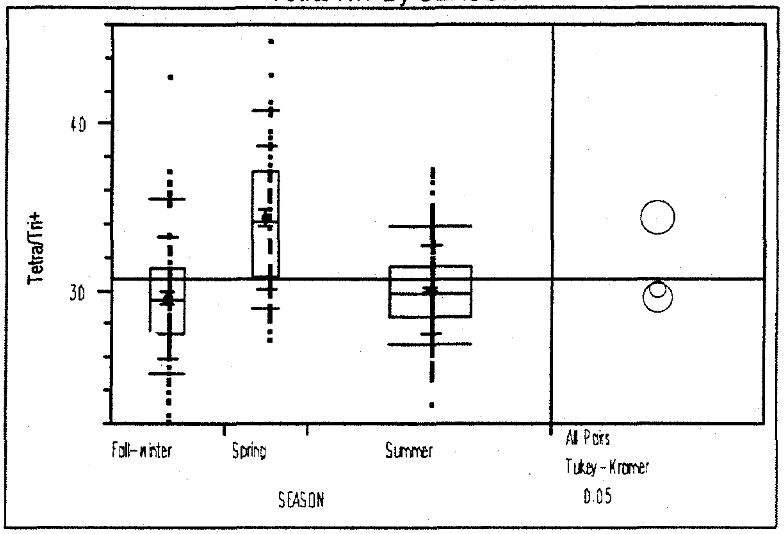
Abs(Dif)-LSD	Summer	Fall-winter	Spring
Summer	-1.58223	5.11422	6.19710
Fall-winter	5.11422	-2.30291	-1.19053
Spring	6.19710	-1.19053	-2.68281

Positive values show pairs of means that are significantly different.

Source: Hudson River Database Release 4.1

Figure A-28
Tri/Tri+ Mass Ratio in General Electric Samples at the TI Dam
Grouped by Season (1996-1998)

Tetra/Tri+ By SEASON



Quantiles

Level	minimum	10.0%	25.0%	median	75.0%	90.0%	maximum
Fall-winter	22.37	25.087	27.4375	29.57	31.3825	35.525	42.98
Spring	27.27	28.957	30.91	34.195	37.285	40.829	45.19
Summer	23.3	26.822	28.51	29.86	31.56	33.948	37.48

Means and Std Deviations

Level	Number	Mean	Std Dev	Std Err Mean
Fall-winter	76	29.6617	3.71522	0.42617
Spring	56	34.4057	4.29803	0.57435
Summer	161	30.1178	2.72116	0.21446

Means Comparisons

Dif=Mean[i]-Mean[j]	Spring	Summer	Fall-winter
Spring	0.00000	4.28789	4.74400
Summer	-4.28789	0.00000	0.45612
Fall-winter	-4.74400	-0.45612	0.00000

Alpha= 0.05

Comparisons for all pairs using Tukey-Kramer HSD

q*

2.35588

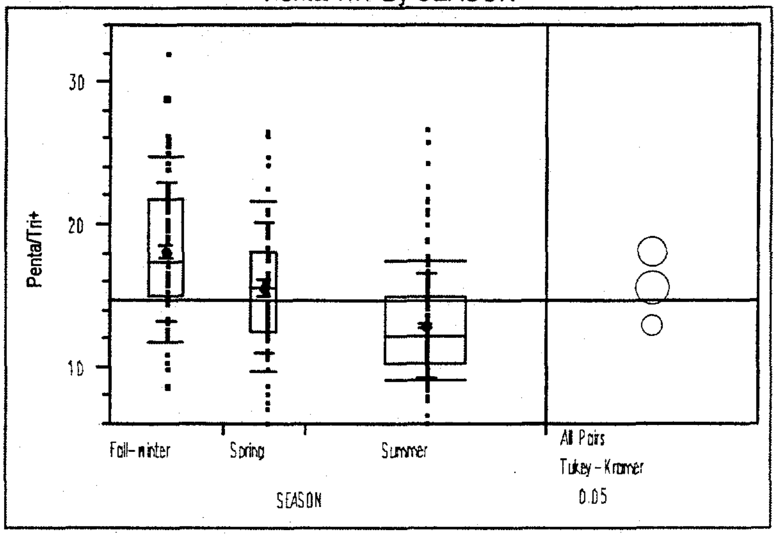
Abs(Dif)-LSD	Spring	Summer	Fall-winter
Spring	-1.48723	3.06698	3.35806
Summer	3.06698	-0.87712	-0.63913
Fall-winter	3.35806	-0.63913	-1.27663

Positive values show pairs of means that are significantly different.

Source: Hudson River Database Release 4.1

Figure A-29
Tetra/Tri+ Mass Ratio in General Electric Samples at the TI Dam
Grouped by Season (1996-1998)

Penta/Tri+ By SEASON



Quantiles

Level	minimum	10.0%	25.0%	median	75.0%	90.0%	maximum
Fall-winter	8.76	11.768	14.9975	17.35	21.815	24.75	32.1
Spring	6.3	9.708	12.5925	15.655	18.1875	21.728	26.68
Summer	6.26	9.138	10.38	12.31	15.125	17.508	26.82

Means and Std Deviations

Level	Number	Mean	Std Dev	Std Err Mean
Fall-winter	76	18.1020	4.89292	0.56126
Spring	56	15.6493	4.61842	0.61716
Summer	161	12.9545	3.70613	0.29208

Means Comparisons

Dif=Mean[i]-Mean[j]	Fall-winter	Spring	Summer
Fall-winter	0.00000	2.45269	5.14750
Spring	-2.45269	0.00000	2.69481
Summer	-5.14750	-2.69481	0.00000

Alpha= 0.05

Comparisons for all pairs using Tukey-Kramer HSD

q*
2.35588

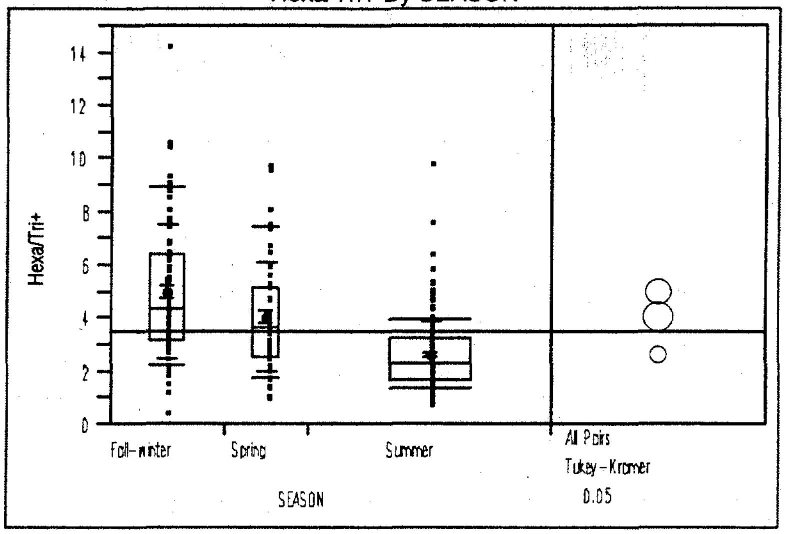
Abs(Dif)-LSD	Fall-winter	Spring	Summer
Fall-winter	-1.61308	0.70150	3.76361
Spring	0.70150	-1.87918	1.15216
Summer	3.76361	1.15216	-1.10828

Positive values show pairs of means that are significantly different.

Source: Hudson River Database Release 4.1

Figure A-30
Penta/Tri+ Mass Ratio in General Electric Samples at the TI Dam
Grouped by Season (1996-1998)

Hexa/Tri+ By SEASON



Quantiles

Level	minimum	10.0%	25.0%	median	75.0%	90.0%	maximum
Fall-winter	0.55	2.23	3.235	4.365	6.4325	8.952	14.32
Spring	1.08	1.805	2.5625	3.645	5.1575	7.485	9.85
Summer	0.84	1.402	1.7	2.33	3.265	3.96	9.92

Means and Std Deviations

Level	Number	Mean	Std Dev	Std Err Mean
Fall-winter	76	4.99842	2.53976	0.29133
Spring	56	4.04464	2.10051	0.28069
Summer	161	2.61528	1.30706	0.10301

Means Comparisons

Dif=Mean[i]-Mean[j]	Fall-winter	Spring	Summer
Fall-winter	0.00000	0.95378	2.38314
Spring	-0.95378	0.00000	1.42936
Summer	-2.38314	-1.42936	0.00000

Alpha= 0.05

Comparisons for all pairs using Tukey-Kramer HSD

q*

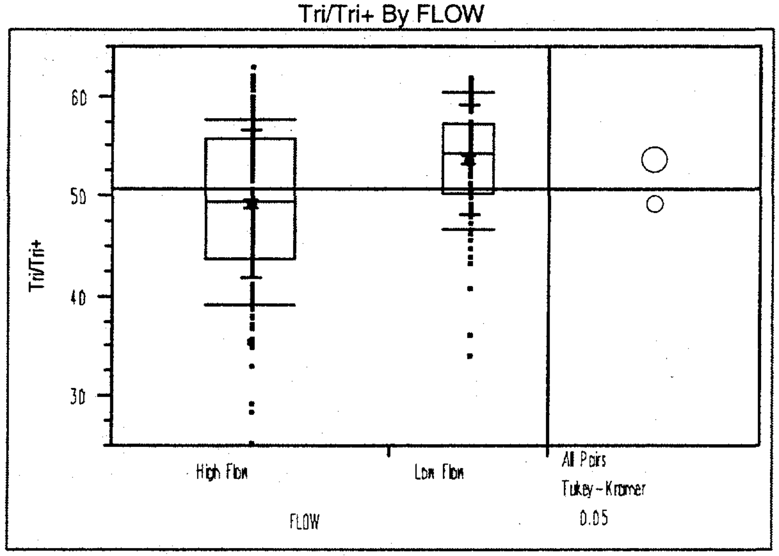
2.35588

Abs(Dif)-LSD	Fall-winter	Spring	Summer
Fall-winter	-0.70961	0.18342	1.77436
Spring	0.18342	-0.82667	0.75074
Summer	1.77436	0.75074	-0.48754

Positive values show pairs of means that are significantly different.

Source: Hudson River Database Release 4.1

Figure A-31
Hexa/Tri+ Mass Ratio in General Electric Samples at the TI Dam
Grouped by Season (1996-1998)



			Quantiles				
Level	minimum	10.0%	25.0%	median	75.0%	90.0%	maximum
High Flow	25.47	39.216	43.83	49.56	55.76	57.812	63.3
Low Flow	34.32	46.723	50.31	54.44	57.3375	60.507	62.11

Means and Std Deviations

Level	Number	Mean	Std Dev	Std Err Mean
High Flow	187	49.2525	7.45478	0.54515
Low Flow	106	53.6843	5.48550	0.53280

Means Comparisons

Dif=Mean[i]-Mean[j]	Low Flow	High Flow
Low Flow	0.00000	4.43188
High Flow	-4.43188	0.00000

Alpha= 0.05

Comparisons for all pairs using Tukey-Kramer HSD

q*

1.96815

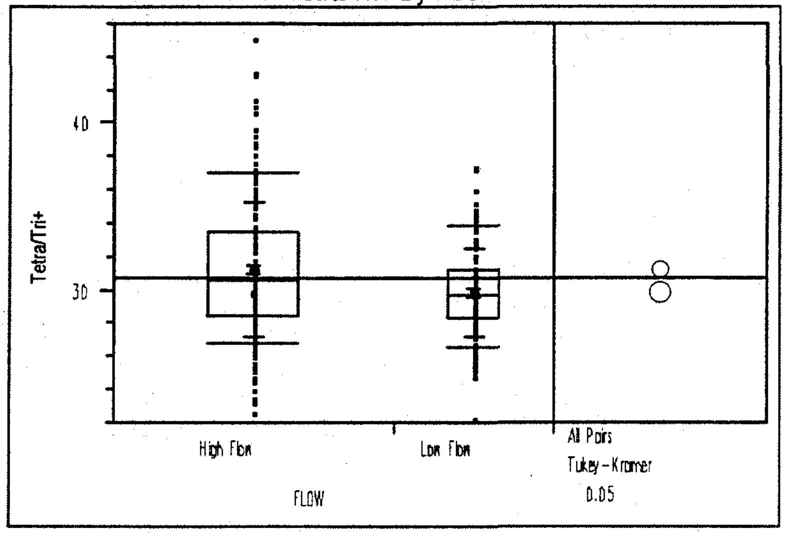
Abs(Dif)-LSD	Low Flow	High Flow
Low Flow	-1.84111	2.80229
High Flow	2.80229	-1.38616

Positive values show pairs of means that are significantly different.

Source: Hudson River Database Release 4.1

Figure A-32
Tri/Tri+ Mass Ratio in General Electric Samples at the TI Dam
Grouped by Flow (1996-1998)

Tetra/Tri+ By FLOW



Quantiles

Level	minimum	10.0%	25.0%	median	75.0%	90.0%	maximum
High Flow	22.74	26.808	28.46	30.6	33.5	37.146	45.19
Low Flow	22.37	26.64	28.44	29.76	31.28	33.962	37.48

Means and Std Deviations

Level	Number	Mean	Std Dev	Std Err Mean
High Flow	187	31.3062	4.16583	0.30464
Low Flow	106	29.9596	2.73741	0.26588

Means Comparisons

Dif=Mean[i]-Mean[j]	High Flow	Low Flow
High Flow	0.00000	1.34658
Low Flow	-1.34658	0.00000

Alpha= 0.05

Comparisons for all pairs using Tukey-Kramer HSD

q*

1.96815

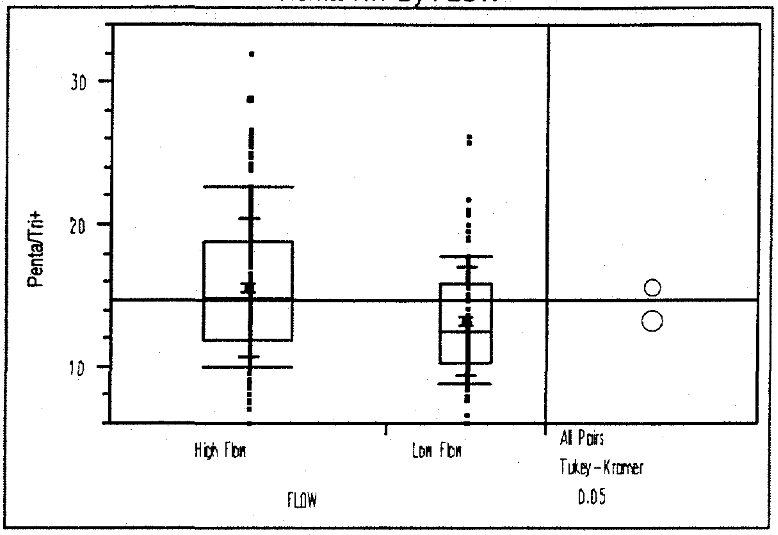
Abs(Dif)-LSD	High Flow	Low Flow
High Flow	-0.75602	0.45779
Low Flow	0.45779	-1.00415

Positive values show pairs of means that are significantly different.

Source: Hudson River Database Release 4.1

Figure A-33
Tetra/Tri+ Mass Ratio in General Electric Samples at the TI Dam
Grouped by Flow (1996-1998)

Penta/Tri+ By FLOW



Quantiles

Level	minimum	10.0%	25.0%	median	75.0%	90.0%	maximum
High Flow	6.3	10.028	11.98	14.99	18.87	22.76	32.1
Low Flow	6.26	8.874	10.3725	12.545	15.9125	17.912	26.41

Means and Std Deviations

Level	Number	Mean	Std Dev	Std Err Mean
High Flow	187	15.6719	4.97880	0.36409
Low Flow	106	13.2749	3.88450	0.37730

Means Comparisons

Dif=Mean[i]-Mean[j]	High Flow	Low Flow
High Flow	0.00000	2.39697
Low Flow	-2.39697	0.00000

Alpha= 0.05

Comparisons for all pairs using Tukey-Kramer HSD

q*

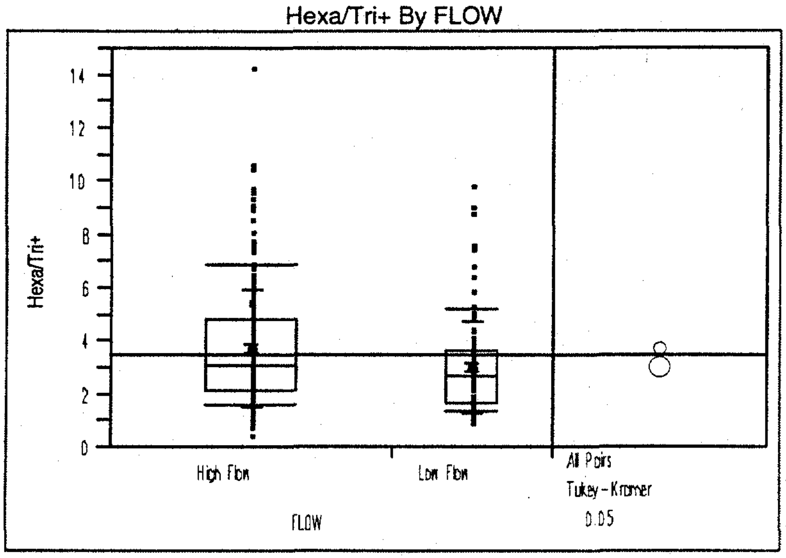
1.96815

Abs(Dif)-LSD	High Flow	Low Flow
High Flow	-0.93913	1.29290
Low Flow	1.29290	-1.24737

Positive values show pairs of means that are significantly different.

Source: Hudson River Database Release 4.1

Figure A-34
Penta/Tri+ Mass Ratio in General Electric Samples at the TI Dam
Grouped by Flow (1996-1998)



Quantiles

Level	minimum	10.0%	25.0%	median	75.0%	90.0%	maximum
High Flow	0.55	1.602	2.21	3.12	4.82	6.88	14.32
Low Flow	1.02	1.407	1.665	2.75	3.66	5.237	9.92

Means and Std Deviations

Level	Number	Mean	Std Dev	Std Err Mean
High Flow	187	3.75824	2.26079	0.16533
Low Flow	106	3.06274	1.77186	0.17210

Means Comparisons

Dif=Mean[i]-Mean[j]	High Flow	Low Flow
High Flow	0.000000	0.695499
Low Flow	-0.6955	0.000000

Alpha= 0.05

Comparisons for all pairs using Tukey-Kramer HSD

q*

1.96815

Abs(Dif)-LSD	High Flow	Low Flow
High Flow	-0.42694	0.193583
Low Flow	0.193583	-0.56707

Positive values show pairs of means that are significantly different.

Source: Hudson River Database Release 4.1

Figure A-35
Hexa/Tri+ Mass Ratio in General Electric Samples at the TI Dam
Grouped by Flow (1996-1998)

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Appendix B

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APPENDIX B

EFFECTS ASSESSMENT

This appendix provides a general overview of the toxicology of PCBs and describes the methods used to characterize particular toxicological effects of PCBs on aquatic and terrestrial organisms. Toxicity reference values (TRVs) used to estimate the potential risk to receptor species resulting from exposure to PCBs are presented following the background on PCB toxicology. TRVs are levels of exposure associated with either Lowest Observed Adverse Effects Levels (LOAELs) or No Observed Adverse Effects Levels (NOAELs). They provide a basis for judging the potential effects of measured or predicted exposures that are above or below these levels.

Use of both LOAELs and NOAELS provides perspective on the potential for risk as a result of exposure to PCBs originating from the site. LOAELs are values at which effects have been observed in either laboratory or field studies, while the NOAEL represents the lowest dose or body burden at which an effect was not observed. Exceedance of a LOAEL indicates a greater potential for risk.

B.1 Polychlorinated Biphenyl Structure and Toxicity

The toxicity of PCBs has been shown to manifest itself in many different ways, among various species of animals. Typical responses to PCB exposure in animals include wasting syndrome, hepatotoxicity, immunotoxicity, neurotoxicity, reproductive and developmental effects, gastrointestinal effects, respiratory effects, dermal toxicity, and mutagenic and carcinogenic effects. Some of these effects are manifested through endocrine disruption. Table B-1 provides a summary of the common effects documented to occur in animals as a result of PCB exposure.

PCBs are typically present in the environment as complex mixtures. These mixtures consist of discrete PCB molecules that are individually referred to as PCB congeners. PCB congeners are often introduced into the environment as commercial mixtures known as Aroclors. PCB toxicity varies significantly among different congeners and is dependent on a number of factors. Two significant factors relate to the chemical structure of the PCB congener (Figure B-1), including the degree of chlorination and the position of the chlorines on the biphenyl structure (Safe *et al.*, 1985a). In general, higher chlorine content typically results in higher toxicity, and PCB congeners that are chlorinated in the *ortho* position are typically less toxic than congeners chlorinated in the *meta* and *para* positions. These differences are discussed in more detail in the following sections with a focus on the metabolic processes involved in the activation of PCBs. Metabolic activation is believed to be the major process contributing to PCB toxicity.

B.1.1 Structure-Function Relationships of PCBs

PCB congeners have been shown to produce toxic effects similar to, although typically less potent than, 2,3,7,8-tetrachlorodibenzo-*p*-dioxin (2,3,7,8-TCDD), the most toxic member of all groups of halogenated aromatic hydrocarbons (Van den Berg *et al.* , 1998). The toxicity of these hydrocarbons is thought to be related to their ability to induce cytochrome P450-dependent aryl hydrocarbon metabolizing mixed-function oxidases (MFOs) (Safe *et al.* , 1985b; McFarland and Clarke, 1989). Similar to 2,3,7,8-TCDD, a number of PCB congeners have been shown to induce aryl hydrocarbon hydroxylase (AHH) activity, as well as ethoxyresorufin-O-deethylase (EROD) activity. The potency and specificity of MFO induction of individual PCB congeners is directly related to how closely they approach the molecular structure of 2,3,7,8-TCDD (Safe *et al.* , 1985b; McFarland and Clarke, 1989). The dioxin, 2,3,7,8-TCDD assumes a rigid coplanar configuration which facilitates its binding to the cytosolic *Ah* (aryl hydrocarbon) receptor (AhR). Translocation of the dioxin-*Ah*-receptor complex to the nuclear *Ah* locus is thought to initiate the synthesis of enzymes that exhibit AHH and EROD activity (Safe *et al.* , 1985a). The activation of these enzymes may be involved in biotransformation, conjugation and removal, or metabolic activation of aryl hydrocarbons to potentially toxic intermediates (McFarland and Clarke, 1989).

Studies of structure-function relationships for PCB congeners indicate that the location of the chlorine substitution determines the type and intensity of the toxicity that can be elicited (Safe *et al.* , 1985a). PCB congeners with substitutions at the *meta-* and *para-* positions as well as some mono-*ortho-* substituted congeners assume a coplanar conformation similar to 2,3,7,8-TCDD, and are typically more toxic than non-coplanar congeners with high *ortho*-substitution. The phenyl rings of PCB molecules are linked by a single carbon:carbon bond (Figure B-1), that, unlike the rigidly bound phenyl rings of dioxins, allows relatively unconstrained freedom of rotation of one ring relative to the other (Safe *et al.* , 1985a). When bulky chlorine atoms are substituted at certain positions on the biphenyl nucleus they inflict certain constraints on rotational freedom. The greatest effect is exerted by substitution of at least two opposing *ortho*-substitutions on opposite rings. The energetic cost of maintaining a coplanar configuration becomes increasingly high as *ortho* substitution increases. The release of steric hindrance, as a consequence of chlorine substitution in *ortho-* positions, yields a non-coplanar molecular configuration, making it less "dioxin-like". Moreover, since coplanarity facilitates binding to the AhR, which in turn effects the level of AHH activity, metabolic activation, and potential toxicity of certain PCB congeners, the toxicity of PCB congeners decreases as *ortho* substitution increases. PCB congeners with two chlorines in the *ortho* position (di-*ortho*), or other highly *ortho*-substituted congeners do not produce a strong, toxic, "dioxin-like" response (McFarland and Clarke, 1989; Safe, 1990). Table B-2 lists the coplanar non-*ortho* and mono-*ortho* congeners.

B.1.2 Metabolic Activation and Toxicity of PCBs

The toxicological effects of PCBs, as well as other halogenated aromatic hydrocarbons, including dioxins, are correlated with their ability to induce the cytochrome P450-dependent mixed function oxygenases (MFOs) (Safe *et al.* , 1985b; McFarland and Clarke, 1989). MFOs are a group of microsomal enzymes that catalyze oxidative biotransformation of aromatic ring-containing compounds to facilitate conjugation and removal. This metabolic activation occurs mainly in the liver and is a major mechanism of PCB metabolism and toxicity. The MFOs that are induced by PCBs have been divided into three general groups: 3-methylcholanthrene-type (3-MC-type); phenobarbital-type (PB-type); and mixed-type, possessing catalyzing properties of both. PB-induced

MFOs typically catalyze insertion of oxygen into conformationally nonhindered sites of non-coplanar lipophilic molecules, such as *ortho*-substituted PCBs, and 3-MC-induced MFOs typically catalyze insertion of oxygen into conformationally hindered sites of planar molecules, such as non-*ortho*-substituted PCBs (McFarland and Clarke, 1989). The intermediate transition products typically formed from these oxidations are reactive epoxides. Epoxide-derivatives of PCBs may be the carcinogenic, mutagenic, or teratogenic metabolites of the parent compounds (McFarland and Clarke, 1989). Ordinarily, reactions catalyzed by PB-induced MFOs go on to conjugation, which generally increases their water solubility, making them more easily excreted. On the other hand, the conformational hindrance of the oxygenated molecule subsequent to oxidation by 3-MC-induced MFOs, provides stability of the intermediate and tends to inhibit conjugation and detoxification (McFarland and Clarke, 1989). Thus, the potential for contributing to toxicity through bioactivation via an epoxide-intermediate is considered to be much greater with 3-MC induced enzymic reactions. This is reflected in the observed higher toxicity of the more "dioxin-like" coplanar PCBs, which are potent inducers of AHH, a 3-MC-type MFO (McFarland and Clarke, 1989).

There is significant variability in MFO activity among species. MFO activity generally decreases in the following order: mammals > birds and amphibians > fish (Walker *et al.* , 1984). The levels in aquatic invertebrates were found to be even lower. In addition, the levels can vary significantly even among closely related species (Knight and Walker, 1982). Low MFO activity may be a significant contributing factor in the bioaccumulation of organochlorines in many organisms (Fossi *et al.* , 1990).

B.1.3 Estimating the Ecological Effects of PCBs

This ecological risk assessment focuses on effects that relate to the survival, growth, and reproduction of individuals within the local populations of fish and wildlife species. Reproductive effects are defined broadly herein to include egg maturation, spawning, egg hatchability, and survival of fish larvae.

Reproductive effects tend to be the most sensitive endpoint for animals exposed to PCBs. Indeed, toxicity studies in vertebrates indicate a relationship between PCB exposure, as demonstrated by AHH induction, and functions that are mediated by the endocrine system, such as reproductive success. A possible explanation for the relationship between AHH activity and reproductive success may be due to a potential interference from the P450-dependent MFO with the ability of this class of P450 proteins to regulate sex steroids. In fact, the induction of cytochrome P450 isozymes from PCB exposure has been shown to alter patterns of steroid metabolism (Spies *et al.* , 1990). As another example, the maternal hepatic AHH activity of the flatfish, *Paralichthys stellatus*, at the time of spawning, was found to be inversely related to three reproductive functions: egg viability, fertilization success, and successful development from fertilization through hatching (Long and Buchman, 1990).

As discussed earlier, PCBs are often introduced into the environment as commercial PCB congener mixtures, known as Aroclors. Historically, the most common approach for assessing the ecological impact of PCBs has involved estimating exposure and effects in terms of totals or Aroclor mixtures. It is important to note that, since different PCB congeners may be metabolized at different rates through various enzymatic mechanisms, when subjected to processes of environmental degradation and mixing, the identity of Aroclor mixtures is altered (McFarland and Clarke, 1989). Therefore, depending on the extent of breakdown, the environmental composition of PCBs may be significantly different from the original Aroclor mixture. Furthermore, commercial Aroclor mixtures used in laboratory toxicity studies may not represent true environmental exposure to this Aroclor. Thus, there are some uncertainties associated with estimating the ecological effects of PCBs in terms of total PCBs or Aroclors. As a result, there has been a great emphasis on the development of techniques that provide an assessment of potential risk from exposure to individual PCB congeners.

A methodology has been established, known as Toxic Equivalency (TEQ) Toxic Equivalency Factors (TEF) methodology (TEQ/TEF), that quantifies the toxicities of PCB congeners relative to the toxicity of the potent dioxin 2,3,7,8-TCDD (see van den Berg *et al.* , 1998 for review). It is currently accepted that the carcinogenic potency of dioxin is effected by its ability to bind AhR. In fact, dioxin is thought to be the most potent known AhR ligand (NOAA, 1999b). It is also generally accepted that the dioxin-like toxicities of PCB congeners are directly correlated to their ability to bind the AhR. Thus, the TEQ/TEF methodology provides a toxicity measurement for all AhR-binding compounds based on their relative toxicity to dioxin. Since 2,3,7,8-TCDD has the greatest affinity for the AhR, it is assigned a TCDD-Toxicity Equivalent Factor of 1.0. PCB congeners are then assigned a TCDD-TEF relative to 2,3,7,8-TCDD, based on experimental evidence. For example, if the relative toxicity of a particular congener is one-thousandth that of TCDD, it would have a TEF of 0.001. The potency of a PCB congener is estimated by multiplying the tissue concentration of the congener in question by the TEF for that congener to yield the toxic equivalent (TEQ) of dioxin. Finally, a TEQ for the whole mixture can be determined from the sum of the calculated TEQs for each AhR-binding congener. The World Health Organization has derived TEFs for a number of PCB congeners (van den Berg *et al.* , 1998). These values are presented in Table B-2.

An advantage of the TEQ/TEF approach is that it provides a basis for determining the toxicity of a complex mixture of PCBs in media or tissues. The disadvantage of this approach is that only AhR-active PCBs, and AhR-mediated endpoints, are considered for TEF calculations. For this reason, it is useful to consider the TEQ/TEF method in concert with other methods for evaluating toxicity.

Recent data suggest that non-AhR mediated side effects may be important contributors to PCB toxicity. For example, Moore and Peterson (1996) suggest that PCBs may play a non-AhR mediated role in the induction of neurotoxicity, hormonal effects, estrogenic effects, and infertility in males. Although coplanar, "dioxin-like" congeners appear most toxic based on current evidence, other congeners may have important non-AhR mediated toxic effects. Thus it is becoming increasingly more important to examine the toxic effects of mixtures as well as individual congeners of PCBs when evaluating the total ecological impact of PCBs.

B.2 Selection of Measures of Effects

Many studies examine the effects of PCBs on aquatic and terrestrial organisms, and results of these studies are compiled and summarized in several reports and reviews (*e.g.*, Eisler and Belisle, 1996; Niimi, 1996; Hoffman *et al.* , 1998; ATSDR, 1996; Eisler, 1986; NOAA, 1999b). For the present assessment, studies on the toxic effects of PCBs were identified by searching the National Library of Medicine (NLM) MEDLINE and TOXLINE databases. Other studies were identified from the reference section of papers that were identified by electronic search. Papers were reviewed to determine whether the study was relevant to the topic.

Many different approaches and methodologies are used in these studies, some of which are more relevant than others to the selection of toxicity reference values (TRVs) for the present risk assessment. TRVs are levels of exposure associated with either LOAELs or NOAELs. They provide a basis for judging the potential effects of measured or predicted exposures that are above or below these levels. Some studies express exposures as concentrations or doses of total PCBs, whereas other studies examine effects associated with individual congeners (*e.g.* PCB 126) or as total dioxin equivalents (TEQs). This risk assessment develops separate TRVs for total PCBs and TEQs. This chapter briefly describes the rationale that was used to select TRVs for various ecological receptors of concern.

Some studies examine toxicity endpoints (such as lethality, growth, and reproduction) that are thought to have greater potential for adverse effects on populations of organisms than other studies. Other studies examine toxicity endpoints such as behavior, disease, cell structure, or biochemical changes that affect individual organisms, but may not result in adverse effects at the population level. For example, toxic effects such as enzyme induction may or may not result in adverse effects to individual animals or populations. The present risk assessment selects TRVs from studies that examine the effects of PCBs on lethality, growth or reproduction. Studies that examined the effects of PCBs on other sublethal endpoints are not used to select TRVs. Lethality, growth, and reproductive-based endpoints typically present the greatest risk to the viability of the individual organism and therefore of the population's survival. Thus, these are considered to be the endpoints of greatest concern relative to the stated assessment endpoints.

When exposures are expected to be long-term, data from studies of chronic exposure are preferable to data from medium-term (subchronic), short-term (acute), or single-exposure studies (USEPA, 1997). Because of the persistence of PCBs, exposure of ecological receptors to PCBs from the Hudson River is expected to be long-term, and therefore studies of chronic exposure are used to select TRVs for the present risk assessment. Long-term studies are also preferred because reproductive effects of PCBs are typically studied after long-term exposure.

Dose-response studies compare the response of organisms exposed to a range of doses to that of a control group. Ideally, doses that are below and above the threshold level that causes adverse effects are examined. Toxicity endpoints determined in dose-response and other studies include:

• NOAEL (No-Observed-Adverse-Effect-Level) is the highest exposure level shown to be without adverse effect in organisms exposed to a range of doses. NOAELs may be expressed as dietary doses (*e.g.*, mg PCBs consumed/kg body weight/d), as concentrations in external media (*e.g.*, mg PCBs/kg food), or as concentrations in tissue of the effected organisms (*e.g.*, mg chemical/kg egg).

• LOAEL (Lowest-Observed-Adverse-Effect-Level) is the lowest exposure level shown to produce adverse effect in organisms exposed to a range of doses. LOAELs may also be expressed as dietary doses (*e.g.*, mg PCBs consumed/kg body weight/d), as concentrations in external media (*e.g.*, mg PCBs/kg food), or as concentrations in tissue of the effected organisms (*e.g.*, mg chemical/kg egg).

• LD_{50} is the Lethal Dose that results in death of 50% of the exposed organisms. Expressed in units of dose (*e.g.*, mg PCBs administered/kg body weight of test organism/d).

• LC_{50} is the Lethal Concentration in some external media (*e.g.* food, water, or sediment) that results in death of 50% of the exposed organisms. Expressed in units of concentration (*e.g.*, mg PCBs/kg wet weight food).

• ED_{50} is the Effective Dose that results in a sublethal effect in 50% of the exposed organisms (mg/kg/d).

• EC_{50} is the Effective Concentration in some external media that results in a sublethal effect in 50% of the exposed organisms (mg/kg).

• CBR or Critical Body Residue is the concentration in the organism (*e.g.*, whole body, liver, or egg) that is associated with an adverse effect (mg PCBs/kg wet wt tissue).

• EL-effect is the effect level that results in an adverse effect in organisms exposed to a single dose, rather than a range of doses. Expressed in units of dose (mg/kg/d) or concentration (mg/kg).

• EL-no effect is the effect level that does not result in an adverse effect in organisms exposed to a single dose, rather than a range of doses. Expressed in units of dose (mg/kg/d) or concentration (mg/kg).

Most USEPA risk assessments typically estimate risk by comparing the exposure of receptors of concern to TRVs that are based on NOAELs. TRVs for the present baseline risk assessments are developed on the basis of both NOAELs and LOAELs to provide perspective on the range of potential effects relative to measured or modeled exposures.

Differences in the feeding behavior of aquatic and terrestrial organisms determine the type of toxicity endpoints that are most easily measured and most useful in assessing risk. For example, the dose consumed in food is more easily measured for terrestrial animals than for aquatic organisms since uneaten food can be difficult to collect and quantify in an aqueous environment. Therefore, for aquatic organisms, toxicity endpoints are more often expressed as concentrations in external media (*e.g.*, water) or as accumulated concentrations in the tissue of the exposed organism (also called a "body burden"). In some studies, doses are administered via gavage, intraperitoneal injection into an adult, or injection into a fish or bird egg. If appropriate studies are available, TRVs for the present baseline risk assessment are selected on the basis of the most likely route of exposure, as described below:

- TRVs for benthic invertebrates are expressed as concentrations in external media (*e.g.*, mg/kg sediment). Critical body burdens (*e.g.*, mg/kg body weight) for benthic invertebrates are presented, but a TRV is not selected due to limited data.

- TRVs for fish are expressed as critical body residues (CBR) (*e.g.*, mg/kg whole body weight and mg/kg lipid in eggs).

- TRVs for terrestrial receptors (*e.g.*, birds and mammals) are expressed as daily dietary doses (*e.g.*, mg/kg whole body wt/d).

- TRVs for birds are also expressed as concentrations in eggs (*e.g.* mg/kg wet wt egg).

B.2.1 Methodology Used to Derive TRVs

The literature on toxic effects of PCBs to animals includes studies conducted solely in the laboratory, as well as studies including a field component. Each type of study has advantages and disadvantages for the purpose of deriving TRVs for a risk assessment. For example, a controlled laboratory study can be designed to test the effect of a single formulation or congener (*e.g.* Aroclor 1254 or PCB 126) on the test species in the absence of the effects of other co-occurring contaminants. This is an advantage since greater confidence can be placed in the conclusion that observed effects are related to exposure to the test compound. However, laboratory studies are often conducted on species that are easily maintained in the laboratory, rather than on wildlife species. Therefore, laboratory studies may have the disadvantage of being conducted on species that are less closely related to a particular receptor of concern. Field studies have the advantage that organisms are exposed to a more realistic mixture of PCB congeners (with differences in toxic potencies), than, for example, laboratory tests that expose organisms to a commercial mixture, such as Aroclor 1254. Field studies have the disadvantage that organisms are usually exposed to other contaminants and observed effects may not be attributable solely to exposure to PCBs. Field studies can be used most successfully, however, to establish concentrations of PCBs or TEQs at which adverse effects are not observed (*e.g.*, a NOAEL). Because of the potential contribution of other contaminants (*e.g.* metals, pesticides, etc.) to observed effects in field studies, the present risk assessment uses field studies to establish NOAEL TRVs, but not LOAEL TRVs.

If appropriate field studies are available for species in the same taxonomic family as the receptor of concern, those field studies will be used to derive NOAEL TRVs for receptors of concern. Appropriateness of a field study will be based on the following considerations:

• whether the study examines sensitive endpoints, such as reproductive effects, in a species that is closely related (*e.g.* within the same taxonomic family) to the receptor of concern;

• whether measured exposure concentrations of PCBs or dioxin-like compounds are reported for dietary doses, whole organisms, or eggs;

• whether the study establishes a dose-response relationship between exposure concentrations of PCBs or dioxin-like contaminants and observed effects; and

• whether contributions of co-occurring contaminants are reported and considered to be negligible in comparison to contribution of PCBs or dioxin-like compounds.

If appropriate field studies are not available for a test species in the same taxonomic family as the receptor species of concern, laboratory studies will be used to establish TRVs for the receptor species. The general methodology described in the following paragraphs will be used to derive TRVs for receptors of concern from appropriate studies.

When appropriate chronic-exposure toxicity studies on the effects of PCBs on lethality, growth, or reproduction are not available for a species of concern to the risk assessment, extrapolations from other studies are made in order to estimate appropriate TRVs. For example, if toxicity data is unavailable for a particular species of bird, toxicity data for a related species of bird is used if appropriate information was available. Several methodologies have been developed for deriving TRVs for wildlife species (*e.g.*, Sample *et al.* , 1996; California EPA, 1996; USEPA, 1996; Menzie-Cura & Associates, 1997). The general methodology that is used to develop LOAEL and NOAEL toxicity reference values (TRVs) for the present study is described below.

• If an appropriate NOAEL is unavailable for a phylogenetically similar species (*e.g.* within the same taxonomic family), the assessment adjusts NOAEL values for other species (as closely related as possible) by dividing by an uncertainty factor of 10 to account for extrapolations between species. The lowest appropriate NOAEL is used whenever several studies are available. However, if the surrogate test species is known to be the most sensitive of all species tested in that taxonomic group (*e.g.* fish, birds, mammals), then an interspecies uncertainty factor is not applied

• In the absence of an appropriate NOAEL, if a LOAEL is available for a phylogenetically similar species, these may be divided by an uncertainty factor of 10 to account for a LOAEL to NOAEL conversion. The LOAEL to NOAEL conversion is similar to USEPA's derivation of human health RfD (Reference Dose) values, where LOAEL studies are adjusted by a factor of 10 to estimate NOAEL values.

• When calculating chronic dietary dose-based TRVs (*e.g.* mg/kg/d) from data for sub-chronic tests, the sub-chronic LOAEL or NOAEL values are divided by an additional uncertainty factor of 10 to estimate chronic TRVs. The use of an uncertainty factor of 10 is consistent with the methodology used to derive human health RfDs. These factors are applied to account for uncertainty in using an external dose (mg/kg/d in diet) as a surrogate for the dose at the site of toxic action (*e.g.* mg/kg in tissue). Because organisms may attain a toxic dose at the site of toxic action (*e.g.* in tissues or organs) via a large dose administered over a short period, or via a smaller dose administered over a longer period, uncertainty factors are used to estimate the smallest dose that, if administered chronically, would result in a toxic dose at the site of action. USEPA has not established a definitive line between sub-chronic and chronic exposures for ecological receptors. The present risk assessment follows recently developed guidance (Sample *et al.*, 1996) which considers 10 weeks to be the minimum time for chronic exposure of birds and 1 year for chronic exposure of mammals.

• For studies that actually measure the internal toxic dose (*e.g.* mg PCBs/kg tissue), no sub-chronic to chronic uncertainty factor is applied. This is appropriate since effects are being compared to measured internal doses, rather than to external dietary doses that are used as surrogates for the internal dose.

• In cases where NOAELs are available as a dietary concentration (*e.g.*, mg contaminant per kg food), a daily dose for birds or mammals is calculated on the basis of standard estimates of food intake rates and body weights (*e.g.*, USEPA, 1993).

The sensitivity of the risk estimates to the use of these various uncertainty factors is examined in the uncertainty chapter (Chapter 6.0) of the ERA Addendum.

B.2.2 Selection of TRVs for Benthic Invertebrates

B.2.2.1 Sediment Guidelines

Various guidelines exist for concentrations of PCBs in sediment (Table B-3). Modeled concentrations of PCBs in sediments of the Hudson River will be compared to the Sediment Effects Concentrations (SEC) developed for this site (NOAA, 1999a).

B.2.2.2 Body Burden Studies

Relatively few studies were identified that examined the effects of PCBs or dioxin-like compounds on the basis of body burdens in aquatic invertebrates. Concentrations of PCBs that are without adverse effects range from 5.4 to 127 mg/kg wet wt (Table B-4). Lowest-observed-adverse-effect-levels range from 27 to 1570 mg/kg wet wt. A body burden-based TRV is not developed because of the limited amount of data that is available for benthic invertebrates.

B.2.3 Selection of TRVs for Fish

In this section, TRVs are developed for the forage fish receptors (pumpkinseed and spottail shiner), as well as for fish that feed at higher trophic levels, such as the brown bullhead, yellow perch, white perch, largemouth bass, striped bass, and shortnose sturgeon.

Laboratory studies that examine the effects of total PCBs or Aroclors on fish are summarized in Table B-5. Most of these studies report measured concentrations of PCBs in whole body fish tissue, although one study (Black *et al.*, 1998a) reported a nominal injected dose. Laboratory studies on the effects of dioxin-like compounds (TEQs) on fish (Table B-7) typically report concentrations of TEQs in fish eggs, rather than in whole body, since eggs represent a more sensitive life stage. Comparison of effect levels (*e.g.* NOAELs or LOAELs) reported as wet weight concentrations in eggs to whole body tissue concentrations in adult Hudson River fish is complicated by the fact that eggs and whole body adult fish tend to have different lipid contents and concentrations of lipophilic contaminants, such as TEQs. However, if we assume that TEQs partition equally into the lipid phase of the egg and into the lipids in the tissue of adult fish, then lipid-normalized concentrations in fish eggs that are associated with adverse effects (µg TEQs/kg lipid) can be compared to lipid-normalized tissue concentrations of TEQs in adult Hudson River fish. Therefore, this assessment establishes TRVs for TEQs in fish on a lipid-normalized basis so that measured or predicted whole body concentrations of TEQs in Hudson River fish can be compared to TRVs established from studies on fish eggs.

B.2.3.1 Pumpkinseed (*Lepomis gibbosus*)

Total PCB Body Burden in Pumpkinseed

No laboratory studies were identified that examined toxicity of PCBs to the pumpkinseed forage fish receptor, or to a fish species in the same family as the pumpkinseed (Table B-5, Figure B-2). Two studies (Hansen *et al.*, 1971 and Hansen *et al.*, 1974) were identified that examined toxicity of PCBs to species in the same order as the pumpkinseed (Table B-23). However, the studies by Hansen *et al.* (1971, 1974) are not selected for the development of TRVs because these studies examined adult mortality, which is not expected to be a sensitive endpoint. Therefore, concentrations of PCBs in the pumpkinseed will be compared to the lowest appropriate NOAEL and corresponding LOAEL from the available appropriate studies (Table B-5).

Hansen *et al.*, (1974) established a NOAEL of 1.9 mg PCBs/kg and a LOAEL of 9.3 mg PCBs/kg for adult female fish. This study was based on a flow-through bioassay of Aroclor 1254 on sheepshead minnow. Fish were exposed for 28 days, and then egg production was induced. The eggs were fertilized and placed in PCB-free flowing seawater and observed for mortality. The TRVs resulting from this study are comparable to the TRVs for the study that was selected (Bengtsson, 1980).

The study by Black *et al.* (1998a) is not selected because it reports a nominal dose, rather than a measured whole body concentration. The Hansen *et al.* (1974) study was not selected because the Bengtsson study was more recent and of longer duration. The study by Bengtsson (1980) on the minnow is selected as the lowest appropriate NOAEL for development of the TRV for pumpkinseed. In this study, fish were exposed to Clophen A50 (a commercial mixture with a chlorine content of 50%) in food for 40 days. Although Clophen A50 was not used in the United States, the chlorine content of Clophen A50 (50% chlorine) is reasonably similar to the chlorine content of Aroclor 1248 (48% chlorine) and Aroclor 1242 (42% chlorine) that were released into the Hudson River. The chlorine content of Hudson River fish resembles that of Aroclor 1254 (54% chlorine), which is more similar to the chlorine content of Clophen A50, than to that of Aroclor 1248 or 1242 (Appendix K USEPA, 1999). Therefore, it is believed that Clophen A50 is a reasonable surrogate of the actual environmental composition of PCBs in Hudson River fish.

Hatchability was significantly reduced in fish with an average total PCB concentration of 170 mg/kg (measured on day 171 of the experiment), but not in fish with an average concentration of 15 mg/kg or 1.6 mg/kg. The only other reproductive endpoints that Bengsston *et al.* (1980) reported to be significantly different in PCB-exposed fish as compared to control fish is the hatching time. Fish in the medium and high exposure groups had significantly reduced hatching times compared with the control group. Exposed fish that hatched prematurely all died within a week of hatching, however, this result was not tested statistically. Nonetheless, because the prematurely hatched fry all died, the low dose group is considered a NOAEL (1.6 mg/kg), and the medium dose group a LOAEL (15 mg/kg).

Because the experimental study measured the actual concentration in fish tissue, rather than estimating the dose on the basis of the concentration in external media (*e.g.*, food, water, or sediment, or injected dose), a subchronic-to-chronic uncertainty factor is not applied. An interspecies uncertainty factor of 10 is applied to develop TRVs for the pumpkinseed.

On the basis of laboratory toxicity studies:

The LOAEL TRV for the pumpkinseed is 1.5 mg PCBs/kg tissue (Table B-25).
The NOAEL TRV for the pumpkinseed is 0.16 mg PCBs/kg tissue (Table B-25).

Several field studies were identified that examined the effect of PCBs on the redbreast sunfish, a species in the same family as the pumpkinseed (Tables B-6 and B-23). Field studies by Adams *et al.* (1989, 1990, 1992) reported reduced fecundity, clutch size and growth in redbreast sunfish (*Lepomis auritus*) that were exposed to PCBs and mercury in the field. However, since other contaminants (*e.g.* mercury) were measured and reported in these fish and may have been contributing to observed effects, these studies are used to develop a NOAEL TRVs, but not a LOAEL TRV, for the pumpkinseed. An interspecies uncertainty factor is not applied since these species are in the same family. Because the experimental study measured the actual concentration in fish tissue, rather than estimating the dose on the basis of the concentration in external media (*e.g.*, food, water, or sediment, or injected dose), a subchronic-to-chronic uncertainty factor is not applied.

On the basis of the field studies:

The NOAEL TRV for the pumpkinseed is 0.5 mg PCBs/kg tissue (Table B-25).

As described previously, a LOAEL is not derived from the field studies because of the potential for interactive effects of other contaminants in addition to PCBs.

Total Dioxin Equivalents (TEQs) in Eggs of the Pumpkinseed

No laboratory studies were identified that examined toxicity of dioxin-like compounds to the pumpkinseed or to a species in the same taxonomic family or order as the pumpkinseed (Tables B-7, Figure B-3). Therefore, concentrations of TEQs in the pumpkinseed will be compared to the lowest appropriate NOAEL and LOAEL from the selected studies (Table B-7). The study by Walker *et al.* (1994) for the lake trout is selected as the lowest appropriate LOAEL and NOAEL from the selected applicable studies (Table B-7). In that study, significant early life stage mortality was observed in lake trout eggs with a concentration of 0.6 μg TEQs/kg lipid. This effect was not observed at a concentration of 0.29 μg/kg lipid. Because the experimental study is based on the concentration in the egg, rather than an estimated dose, a subchronic-to-chronic uncertainty factor is not applied. Because salmonids, such as the lake trout, are among the most sensitive species tested (Table B-7), an interspecies uncertainty factor is not applied.

On the basis of laboratory toxicity studies for salmonids:
The LOAEL TRV for the pumpkinseed is 0.6 μg TEQs/kg lipid (Table B-25).
The NOAEL TRV for the pumpkinseed is 0.29 μg TEQs/kg lipid (Table B-25).

Because salmonids are known to be highly sensitive to effects of dioxin-like compounds (Table B-7), alternative TRVs, developed from laboratory studies conducted on non-salmonid species, are presented for comparison. (Uncertainty associated with comparison of Hudson River fish to these TRVs is discussed in the uncertainty chapter). The lowest non-salmonid NOAEL (5.4 μg TEQ/kg lipid) and LOAEL (103 μg TEQs/kg lipid) from the selected applicable studies (Table B-7) for the fathead minnow, are used to derive alternative TRVs for the pumpkinseed. Because the experimental study is based on the concentration in the egg, rather than an estimated dose, a subchronic-to-chronic uncertainty factor is not applied. An interspecies uncertainty factor of 10 is applied to account for potential differences between fathead minnow and pumpkinseed (Table B-25).

No field studies were identified that examined effects of dioxin-like compounds on reproduction, growth or mortality of the pumpkinseed or on a fish in the same taxonomic family as the pumpkinseed (Table B-8).

B.2.3.2 Spottail Shiner (*Notropis hudsonius*)

Total PCB Body Burden in Spottail Shiner

Concentrations of PCBs in spottail shiner will be compared to the lowest appropriate NOAEL and corresponding LOAEL from the selected applicable studies (Table B-5). The study by Bengtsson (1980) on the minnow is selected as the lowest appropriate NOAEL (1.6 mg/kg) and corresponding LOAEL (15 mg/kg) for development of the TRV for the spottail shiner because the minnow is in the same family as the spottail shiner. Because the experimental study measured the actual concentration in fish tissue, rather than estimating the dose on the basis of the concentration in external media (*e.g.*, food, water, or sediment, or injected dose), a subchronic-to-chronic uncertainty factor is not applied. Because the spottail shiner and the minnow are in the same family, an interspecies uncertainty factor is not applied.

On the basis of laboratory toxicity studies:

The LOAEL TRV for the spottail shiner is 15 mg PCBs/kg tissue (Table B-25).
The NOAEL TRV for the spottail shiner is 1.6 mg PCBs/kg tissue (Table B-25).

No field studies were identified that examined the effects of PCBs on the spottail shiner or on a species in the same taxonomic family as the spottail shiner (Tables B-6 and B-23).

Total Dioxin Equivalents (TEQs) in Eggs of Spottail Shiner

Several laboratory studies were identified that examined toxicity of dioxin-like compounds on fish in the same family as the spottail shiner (Tables B-7, Figure B-3). The study by Olivieri and Cooper (1997) on the fathead minnow provides the lowest appropriate LOAEL and NOAEL from the selected applicable studies (Table B-7). In that study, significant early life stage mortality was observed in fathead minnow eggs with a concentration of 103 μg TEQs/kg lipid. This effect was not observed at a concentration of 5.4 μg TEQs/kg lipid. The study did not report a lipid content for fathead minnow eggs, so the 2.4% reported in Elonen *et al.* (1998) was used to obtain lipid normalized results based on Olivieri and Cooper (1997). Because the experimental study is based on the concentration in the egg, rather than an estimated dose, a subchronic-to-chronic uncertainty factor is not applied. Because fathead minnow and spottail shiner are in the same taxonomic family, an interspecies uncertainty factor is not applied.

Alternative TRVs for dioxin-like compounds are not developed for the spottail shiner since the laboratory-based TRVs for the spottail shiner are not based on data for highly sensitive salmonids.

On the basis of laboratory toxicity studies:
The LOAEL TRV for the spottail shiner is 103 μg TEQs/kg lipid (Table B-25).
The NOAEL TRV for the spottail shiner is 5.4 μg TEQs/kg lipid (Table B-25).

No field studies were identified that examined the effects of dioxin-like compounds on reproduction, growth or mortality of the spottail shiner or on a species in the same taxonomic family as the spottail shiner (Table B-8).

B.2.3.3 Brown bullhead (*Ameiurus nebulosus*)

Total PCB Body Burden in the Brown Bullhead

No laboratory studies were identified that examined toxicity of PCBs to the brown bullhead or to a species in the same taxonomic family or order as the brown bullhead (Table B-5, Figure B-2). Therefore, concentrations of PCBs in the brown bullhead will be compared to the lowest appropriate LOAEL and NOAEL from the selected applicable studies (Table B-5). The study by Black *et al.*

(1998a) is not selected because it reports a nominal dose, rather than a measured whole body concentration. The study by Bengtsson (1980) on the minnow is selected for development of the TRV. Hatching time was significantly reduced in fish with an average total PCB concentration of 15 mg PCBs/kg, but not in fish with an average concentration of 1.6 mg PCBs/kg. Because the experimental study measured the actual concentration in fish tissue, rather than estimating the dose on the basis of the concentration in external media (*e.g.*, food, water, or sediment, or injected dose), a subchronic-to-chronic uncertainty factor is not applied. Because results of studies of PCBs and dioxin-like compounds on fish eggs have shown that minnows are of intermediate sensitivity in comparison to other fish (Tables B-5, B-7), an interspecies uncertainty factor of 10 is applied to develop TRVs for the brown bullhead.

On the basis of laboratory toxicity studies:

The LOAEL TRV for the brown bullhead is 1.5 mg PCBs/kg tissue (Table B-25).
The NOAEL TRV for the brown bullhead is 0.16 mg PCBs/kg tissue (Table B-25).

No field studies were identified that examined effects of PCBs on reproduction, growth or mortality of the brown bullhead or on a species in the same taxonomic family as the brown bullhead (Table B-6).

Total Dioxin Equivalents (TEQs) in Eggs of the Brown Bullhead

No laboratory studies were identified that examined toxicity of dioxin-like compounds on the brown bullhead (Table B-7). The study by Elonen *et al.* (1998) on the channel catfish (Table B-7) is selected for development of TRVs for the brown bullhead because the channel catfish and the brown bullhead are in the same taxonomic family (Table B-23). In that study, significant early life stage mortality was observed in catfish eggs having a concentration of 18 μg TEQs/kg lipid. This effect was not observed at a concentration of 8.0 μg TEQs/kg lipid. Because the experimental study is based on the concentration in the egg, rather than an estimated dose, a subchronic-to-chronic uncertainty factor is not applied. An interspecies uncertainty factor is not applied because channel catfish and brown bullhead are in the same taxonomic family.

On the basis of laboratory toxicity studies:

The LOAEL TRV for the brown bullhead is 18 μg TEQs/kg lipid (Table B-25).
The NOAEL TRV for the brown bullhead is 8.0 μg TEQs/kg lipid (Table B-25).

Because TRVs for effects of dioxin-like compounds on the brown bullhead were not based on data for sensitive salmonid species, alternative TRVs are not derived.

No field studies were identified that examined effects of dioxin-like compounds on reproduction, growth or mortality of brown bullhead or a fish in the same taxonomic family as brown bullhead (Table B-8).

B.2.3.4 Yellow Perch (*Perca flavescens*)

Total PCB Body Burden in the Yellow Perch

No laboratory studies were identified that examined toxicity of PCBs to the yellow perch (Table B-5, Figure B-2). Two studies (Hansen *et al.*, 1974 and Hansen *et al.*, 1971) were identified that examined toxicity of PCBs to species of the same order as the yellcw perch. However, the studies by Hansen *et al.* are not selected for the development of TRVs because these studies examined adult mortality, which is not expected to be a sensitive endpoint. Therefore, concentrations of PCBs in the yellow perch will be compared to the lowest appropriate NOAEL and corresponding LOAEL from the selected applicable studies (Table B-5). The study by Black *et al.* (1998a) is not selected because it reports a nominal dose, rather than a measured whole body concentration. The study by Bengtsson (1980) on the minnow is selected as the lowest appropriate NOAEL for development of the TRV. In this study, hatching time was significantly reduced in fish with an average total PCB concentration of 15 mg/kg, but not in fish with an average concentration of 1.6 mg PCBs/kg. Because the experimental study measured the actual concentration in fish tissue, rather than estimating the dose on the basis of the concentration in external media (*e.g.*, food, water, or sediment, or injected dose), a subchronic-to-chronic uncertainty factor is not applied. Because results of studies of dioxin-like compounds and PCBs on fish eggs have shown another species of minnow to be of intermediate sensitivity compared to all other fish species tested (Tables B-5, B-7), an interspecies uncertainty factor of 10 is applied.

On the basis of laboratory toxicity studies:

The LOAEL TRV for the yellow perch is 1.5 mg PCBs/kg tissue (Table B-25).
The NOAEL TRV for the yellow perch is 0.16 mg PCBs/kg tissue (Table B-25).

No field studies were identified that examined effects of PCBs on yellow perch or on a fish in the same family as the yellow perch or on a species in the same family as the yellow perch (Tables B-6 and B-23).

Total Dioxin Equivalents (TEQs) in Eggs of the Yellow Perch

No laboratory studies were identified that examined toxicity of dioxin-like compounds to the yellow perch or to a species in the same taxonomic family or order as the yellow perch (Tables B-7, Figure B-3). Therefore, concentrations of TEQs in the yellow perch will be compared to the lowest appropriate NOAEL and corresponding LOAEL from the selected laboratory studies (Table B-7). The study by Walker *et al.* (1994) reported significant early life stage mortality in lake trout eggs with a concentration of 0.6 TEQs/kg lipid. This effect was not observed at a concentration of 0.29 µg/kg lipid. Because the experimental study is based on the concentration in the egg, rather than an estimated dose, a subchronic-to-chronic uncertainty factor is not applied. Because lake trout are among the most sensitive species tested (Table B-7), an interspecies uncertainty factor is not applied.

On the basis of laboratory toxicity studies for salmonids:

The LOAEL TRV for the yellow perch is 0.6 μg TEQs/kg lipid (Table B-25).
The NOAEL TRV for the yellow perch is 0.29 μg TEQs/kg lipid (Table B-25).

Because salmonids are known to be highly sensitive to effects of dioxin-like compounds (Table B-7), alternative TRVs, developed from studies conducted on non-salmonid species, are presented for comparison. (Uncertainty associated with comparison of Hudson River fish to these TRVs is discussed in Chapter 6 of the ERA Addendum.) The lowest NOAEL (5.4 μg TEQ/kg lipid) and corresponding LOAEL (103 μg TEQs/kg lipid) for a non-salmonid species (Table B-7), the fathead minnow, are presented as alternative TRVs for the yellow perch. An interspecies uncertainty factor of 10 is applied to account for potential differences between the fathead minnow and the yellow perch. Because the experimental study measured the concentration in the egg, rather than estimating a dose, a subchronic-to-chronic uncertainty factor is not applied (Table B-25).

No field studies were identified that examined effects of dioxin-like compounds on reproduction, growth or mortality of the yellow perch or on a species in the same taxonomic family as the yellow perch (Table B-8).

B.2.3.5 White Perch (*Morone americana*)

Total PCB Body Burden in the White Perch

No laboratory studies were identified that examined toxicity of PCBs to the white perch (Table B-5, Figure B-2). Two studies (Hansen *et al.*, 1974 and Hansen *et al.*, 1971) were identified that examined toxicity of PCBs to species of the same order as the white perch. However, the studies by Hansen *et al.* are not selected for the development of TRVs because these studies examined adult mortality, which is not expected to be a sensitive endpoint. Therefore, concentrations of PCBs in the white perch will be compared to the lowest appropriate NOAEL and corresponding LOAEL from the selected applicable studies (Table B-5). The study by Black *et al.* (1998a) is not selected because it reports a nominal dose, rather than a measured whole body concentration. The study by Bengtsson (1980) on the minnow is selected as the lowest appropriate NOAEL and corresponding LOAEL for development of the TRV. In that study, hatching time was significantly reduced in fish with an average total PCB concentration of 15 mg/kg, but not in fish with an average concentration of 1.6 mg PCBs/kg. Because the experimental study measured the actual concentration in fish tissue, rather than estimating the dose on the basis of the concentration in external media (*e.g.*, food, water, or sediment, or injected dose), a subchronic-to-chronic uncertainty factor is not applied. Because results of studies of dioxin-like compounds and PCBs on fish eggs have shown another species of minnow to be of intermediate sensitivity compared to all other fish species tested (Tables B-5, B-7), an interspecies uncertainty factor of 10 is applied.

On the basis of laboratory toxicity studies:

The LOAEL TRV for the white perch is 1.5 mg PCBs/kg tissue (Table B-25).
The NOAEL TRV for the white perch is 0.16 mg PCBs/kg tissue (Table B-25).

Two field studies were identified that examined the effects of PCBs on striped bass (Table B-6). In one study, larval mortality was observed at concentrations of 0.1 to 10 mg PCBs/kg eggs, but a NOAEL was not reported (Westin *et al.* , 1985). Another study found no adverse effect on survival of striped bass larvae with average concentrations of 3.1 mg PCBs/kg larval tissue (Westin *et al.* , 1983). This study is selected for development of a NOAEL-based TRV for the white perch. An interspecies uncertainty factor is not applied because white perch and striped bass are in the same taxonomic family (Table B-23). Because the study measured the concentration in the larval tissue, rather than estimating a dose, a subchronic-to-chronic uncertainty factor is not applied (Table B-25).

On the basis of the field study:

The NOAEL TRV for the white perch is 3.1 mg PCBs/kg tissue (Table B-25).

Total Dioxin Equivalents (TEQs) in Eggs of the White Perch

No laboratory studies were identified that examined the toxicity of dioxin-like compounds to the white perch or to a species in the same taxonomic family or order as the white perch (Tables B-7, Figure B-3). Therefore, concentrations of TEQs in the white perch will be compared to the lowest appropriate LOAEL and NOAEL from the selected studies (Table B-7). The study by Walker *et al.* (1994) for the lake trout is selected as the lowest appropriate LOAEL and NOAEL from the selected applicable studies (Table B-7). In that study, significant early life stage mortality was observed in lake trout eggs with a concentration of 0.6 μg TEQs/kg lipid. This effect was not observed at a concentration of 0.29 μg/kg lipid. Because the experimental study is based on the concentration in the egg, rather than an estimated dose, a subchronic-to-chronic uncertainty factor is not applied. Because lake trout are among the most sensitive species tested (Table B-7), an interspecies uncertainty factor is not applied.

On the basis of laboratory toxicity for salmonid studies:

The LOAEL TRV for the white perch is 0.29 μg TEQs/kg lipid (Table B-25).
The NOAEL TRV for the white perch is 0.6 μg TEQs/kg lipid (Table B-25).

Because salmonids are known to be highly sensitive to effects of dioxin-like compounds (Table B-7), alternative TRVs, developed from studies conducted on non-salmonid species, are presented for comparison. (Uncertainty associated with comparison of Hudson River fish to these TRVs is discussed in Chapter 6 of the ERA Addendum.) The lowest NOAEL (5.4 μg TEQs/kg lipid) and LOAEL (103 μg TEQs/kg lipid) for a non-salmonid species (Table B-7), the fathead minnow, are used to develop alternative TRVs for the white perch (Olivieri and Cooper, 1997). An uncertainty factor of 10 is applied to account for potential differences between the fathead minnow and the white perch. Because the experimental study is based on the concentration in the egg, rather than an estimated dose, a subchronic-to-chronic uncertainty factor is not applied (Table B-25).

No field studies were identified that examined effects of dioxin-like compounds on reproduction, growth or mortality of the white perch or on a species in the same taxonomic family as the white perch (Table B-8).

B.2.3.6 Largemouth bass (*Micropterus salmoides*)

Total PCB Body Burden in the Largemouth Bass

No laboratory studies were identified that examined toxicity of PCBs to the largemouth bass (Table B-5, Figure B-2). Two studies (Hansen *et al.*, 1974 and Hansen *et al.*, 1971) were identified that examined toxicity of PCBs to species of the same order as the largemouth bass. However, the studies by Hansen *et al.* are not selected for the development of TRVs because these studies examined adult mortality, which is not expected to be a sensitive endpoint. Therefore, concentrations of PCBs in the largemouth bass will be compared to the lowest appropriate NOAEL and corresponding LOAEL from the selected applicable studies (Table B-5). The study by Black *et al.* (1998a) is not selected because it reports a nominal dose, rather than a measured whole body concentration. The study by Bengtsson (1980) on the minnow is selected as the lowest appropriate NOAEL and corresponding LOAEL for development of the TRV. Hatching time was significantly reduced in fish with an average total PCB concentration of 15 mg/kg, but not in fish with an average concentration of 1.6 mg/kg. Because the experimental study measured the actual concentration in fish tissue, rather than estimating the dose on the basis of the concentration in external media (*e.g.*, food, water, or sediment, or injected dose), a subchronic-to-chronic uncertainty factor is not applied. Because results of studies of dioxin-like compounds and PCBs on fish eggs have shown another species of minnow to be of intermediate sensitivity compared to all other fish species tested (Tables B-5, B-7), an interspecies uncertainty factor of 10 is applied to the LOAEL (170 mg/kg) and NOAEL (15 mg/kg) from this study to develop TRVs for the largemouth bass.

On the basis of laboratory toxicity studies:

The LOAEL TRV for the largemouth bass is 1.5 mg PCBs/kg tissue (Table B-25).
The NOAEL TRV for the largemouth bass is 0.16 mg PCBs/kg tissue (Table B-25).

Several field studies were identified that examined effect of PCBs on the redbreast sunfish, a species in the same family as the largemouth bass (Table B-6 and B-23). Field studies by Adams *et al.* (1989, 1990, 1992) reported reduced fecundity, clutch size and growth in redbreast sunfish (*Lepomis auritus*) that were exposed to PCBs and mercury in the field. However, since other contaminants (*e.g.*, mercury) were measured and reported in these fish and may have been contributing to observed effects, these studies are used to develop a NOAEL TRVs, but not a LOAEL TRV, for the largemouth bass. An interspecies uncertainty factor is not applied since these species are in the same family. Because the experimental study measured the actual concentration in fish tissue, rather than estimating the dose on the basis of the concentration in external media (*e.g.*, food, water, or sediment, or injected dose), a subchronic-to-chronic uncertainty factor is not applied.

On the basis of the field studies:

The NOAEL TRV for largemouth bass is 0.5 mg PCBs/kg tissue (Table B-25).

Total Dioxin Equivalents (TEQs) in Eggs of the Largemouth Bass

No laboratory studies were identified that examined toxicity of dioxin-like compounds to the largemouth bass or to a species in the same taxonomic family or order as the largemouth bass (Table B-7, Figure B-3). Therefore, concentrations of TEQs in the largemouth bass will be compared to the lowest appropriate NOAEL and corresponding LOAEL from the selected studies (Table B-7). The study by Walker *et al.* (1994) for the lake trout is selected as the lowest appropriate LOAEL and NOAEL from the selected applicable studies (Table B-7). In that study, significant early life stage mortality was observed in lake trout eggs with a concentration of 0.6 TEQs/kg lipid. This effect was not observed at a concentration of 0.29 μg/kg lipid. Because the study is based on the concentration in the egg, rather than an estimated dose, a subchronic-to-chronic uncertainty factor is not applied. Because lake trout are among the most sensitive species tested (Table B-7), an interspecies uncertainty factor is not applied.

On the basis of laboratory toxicity for salmonid studies:

The LOAEL TRV for the largemouth bass is 0.6 μg TEQs/kg lipid (Table B-25).
The NOAEL TRV for the largemouth bass is 0.29 μg TEQs/kg lipid (Table B-25).

Because salmonids are known to be highly sensitive to effects of dioxin-like compounds (Table B-7), alternative TRVs, developed from studies conducted on non-salmonid species, are presented for comparison. (Uncertainty associated with comparison of Hudson River fish to these TRVs is discussed in Chapter 6 of the ERA Addendum.) The lowest NOAEL (5.4 μg TEQ/kg lipid) and corresponding LOAEL (103 μg TEQs/kg lipid) for a non-salmonid species, the fathead minnow, are presented as alternative TRVs for the largemouth bass. An uncertainty factor of 10 is applied to account for potential differences between the fathead minnow and the largemouth bass. Because the experimental study is based on the concentration in the egg, rather than an estimated dose, a subchronic-to-chronic uncertainty factor is not applied (Table B-25).

No field studies were identified that examined effects of dioxin-like compounds on reproduction, growth or mortality of the largemouth bass or on a species in the same taxonomic family as the largemouth bass (Table B-8).

B.2.3.7 Striped bass (*Morone saxatilis*)

PCB Body Burdens in the Striped Bass

No laboratory studies were identified that examined toxicity of PCBs to the striped bass (Table B-5, Figure B-2). Two studies were identified that examined toxicity of PCBs to species that are in the same taxonomic order as the striped bass (Hansen *et al.*, 1971, 1974). However, these studies are not selected for the development of TRVs because they examined adult mortality, which is not considered a sensitive endpoint. Therefore, concentrations of PCBs in the striped bass will be compared to the lowest appropriate NOAEL and corresponding LOAEL from the selected applicable studies (Table B-5). The study by Black *et al.* (1998a) is not selected because it reports a nominal dose, rather than a measured whole body concentration. The study by Bengtsson (1980) on the minnow is selected for development of the TRV. In this study, hatching time of eggs from adult fish with an average total PCB concentration of 15 mg PCBs/kg was significantly reduced in comparison to control fish. Hatching time was not reduced in eggs from adult fish with an average concentration of 1.6 mg PCBs/kg. Because the study measured the actual concentration in fish tissue, rather than estimating the dose on the basis of the concentration in external media (*e.g.*, food, water, or sediment, or injected dose), a subchronic-to-chronic uncertainty factor is not applied. Because results of studies of dioxin-like compounds and PCBs on fish eggs have shown another species of minnow to be of intermediate sensitivity compared to all other fish species tested (Table B-5, B-7), an interspecies uncertainty factor of 10 is applied.

On the basis of laboratory toxicity studies:

The LOAEL TRV for the striped bass is 1.5 mg PCBs/kg tissue (Table B-25).
The NOAEL TRV for the striped bass is 0.16 mg PCBs/kg tissue (Table B-25).

Two field studies were identified that examined the effects of PCBs on striped bass (Table B-6). In one study, larval mortality was observed at concentrations of 0.1 to 10 mg PCBs/kg eggs, but a NOAEL was not reported (Westin *et al.* , 1985). Another study found no adverse effect on survival of striped bass larvae with average concentrations of 3.1 mg PCBs/kg larval tissue (Westin *et al.* , 1983). This study is selected for development of a TRV for the striped bass. Because this study measured the concentration in the larval tissue, rather than estimating a dose, a subchronic-to-chronic uncertainty factor is not applied. An interspecies uncertainty factor is not applied (Table B-25).

On the basis of the field study:

The NOAEL TRV for the striped bass is 3.1 mg PCBs/kg tissue (Table B-25).

Total Dioxin Equivalents (TEQs) in Eggs of Striped Bass

No laboratory studies were identified that examined toxicity of dioxin-like compounds to the striped bass or to a species in the same taxonomic family or order as the striped bass (Table B-7, Figure B-3). Therefore, concentrations of PCBs in the striped bass will be compared to the lowest appropriate NOAEL and corresponding LOAEL from the selected applicable studies (Table B-7). The study by Walker *et al.* (1994) for the lake trout is selected as having the lowest appropriate LOAEL and NOAEL from the selected applicable studies (Table B-7). In that study, significant early life stage mortality was observed in lake trout eggs with a concentration of 0.6 TEQs/kg lipid. This effect was not observed at a concentration of 0.29 µg/kg lipid. Because the experimental study is based on the concentration in the egg, rather than an estimated dose, a subchronic-to-chronic uncertainty factor is not applied. Because lake trout are among the most sensitive species tested (Table B-7), an interspecies uncertainty factor is not applied.

On the basis of laboratory toxicity studies:

The LOAEL TRV for the striped bass is 0.6 µg TEQs/kg lipid (Table B-25).
The NOAEL TRV for the striped bass is 0.29 µg TEQs/kg lipid (Table B-25).

Because salmonids are known to be highly sensitive to effects of dioxin-like compounds (Table B-7), alternative TRVs, developed from studies conducted on non-salmonid species, are presented for comparison. (Uncertainty associated with comparison of Hudson River fish to these TRVs will be discussed in the uncertainty chapter.) The lowest NOAEL (5.4 µg TEQ/kg lipid) and corresponding LOAEL (103 µg TEQs/kg lipid) from the selected applicable studies (Table B-7) for a non-salmonid species, the fathead minnow, are presented as alternative TRVs for the striped bass. An uncertainty factor of 10 is applied to account for potential differences between the fathead minnow and the striped bass. Because the study is based on the concentration in the egg, rather than estimating a dose, a subchronic-to-chronic uncertainty factor is not applied (Table B-25).

No field studies were identified that examined effects of dioxin-like compounds on reproduction, growth or mortality of the striped bass or on a species in the same taxonomic family as the striped bass (Table B-8).

B.2.3.8 Shortnose sturgeon (*Acipenser brevirostrum*)

Total PCB Body Burden in the Shortnose Sturgeon

No laboratory studies were identified that examined toxicity of PCBs to the shortnose sturgeon or to a species in the same taxonomic family or order as the shortnose sturgeon (Table B-5, Figure B-2). Therefore, concentrations of PCBs in the shortnose sturgeon will be compared to the lowest appropriate LOAEL and NOAEL from the selected applicable studies (Table B-5). The study by Black *et al.* (1998a) is not selected because it reports a nominal dose, rather than a measured whole body concentration. The study by Bengtsson (1980) on the minnow is selected for development of the TRV. In this study, hatching time of eggs from adult fish with an average total PCB concentration of 15 mg PCBs/kg was significantly reduced. No effects were seen for fish with an average concentration of 1.6 mg PCBs/kg. Because the experimental study measured the actual concentration in fish tissue, a subchronic-to-chronic uncertainty factor is not applied. Because results of studies of dioxin-like compounds and PCBs on fish eggs have shown another species of minnow to be of intermediate sensitivity compared to all other fish species tested (Table B-5, B-7), an interspecies uncertainty factor of 10 is applied.

On the basis of laboratory toxicity studies:

The LOAEL TRV for the shortnose sturgeon is 1.5 mg PCBs/kg tissue (Table B-25).
The NOAEL TRV for the shortnose sturgeon is 0.16 mg PCBs/kg tissue (Table B-25).

No field studies were identified that examined effects of PCBs on reproduction, growth or mortality of the shortnose sturgeon or on a species in the same taxonomic family as the sturgeon (Table B-6).

Total Dioxin Equivalents (TEQs) in Eggs of the Shortnose Sturgeon

No laboratory studies were identified that examined toxicity of dioxin-like compounds to the shortnose sturgeon or to a species in the same taxonomic family or order as the shortnose sturgeon (Table B-7, Figure B-3). Therefore, the lowest NOAEL and corresponding LOAEL from the selected applicable studies (Table B-7) are selected for development of TRVs. Walker *et al.* (1994) observed significant early life stage mortality in lake trout eggs with a concentration of 0.6 μg TEQs/kg lipid. This effect was not observed at a body burden of 0.29 mg/kg lipid. Because the study is based on the concentration in the egg, rather than estimating a dose, a subchronic-to-chronic uncertainty factor is not applied.

On the basis of laboratory toxicity studies:

The LOAEL TRV for the shortnose sturgeon is 0.6 μg TEQs/kg lipid (Table B-25).
The NOAEL TRV for the shortnose sturgeon is 0.29 μg TEQs/kg lipid (Table B-25).

Because salmonids are known to be highly sensitive to effects of dioxin-like compounds (Table B-7), alternative TRVs, developed from studies conducted on non-salmonid species, are presented for comparison. (Uncertainty associated with comparison of Hudson River fish to these TRVs is discussed in Chapter 6 of the ERA Addendum.) The lowest NOAEL (5.4 μg TEQ/kg lipid) and corresponding LOAEL (103 μg TEQs/kg lipid) for a non-salmonid species, the fathead minnow, are used to develop alternative TRVs for the shortnose sturgeon. An uncertainty factor of 10 is applied to account for differences between the fathead minnow and the shortnose sturgeon. Because the study is based on the concentration in the egg, rather than estimating a dose, a subchronic-to-chronic uncertainty factor is not applied (Table B-25).

No field studies were identified that examined effects of dioxin-like compounds on reproduction, growth or mortality of the shortnose sturgeon or on a species in the same taxonomic family as the sturgeon (Table B-8).

B.2.4 Selection of TRVs for Avian Receptors

Toxicity studies for birds are typically based on dietary doses fed to the birds or on concentrations of chemicals in eggs. Concentrations in eggs may be expressed as actual measured concentrations, as is typical of field studies, or as nominal doses that are injected into the egg. TRVs are developed for birds according to the methodology described previously.

B.2.4.1 Tree swallow (*Tachycineta bicolor*)

Total PCBs in the Diet of the Tree Swallow

No laboratory studies were identified that examined the toxicity of PCBs in the diet of the tree swallow or a bird in the same taxonomic family or order as the tree swallow (Table B-9, Figure B-4). Therefore, the lowest appropriate LOAEL and NOAEL from the selected studies, the LOAEL (0.7 mg/kg/d) and NOAEL (0.1 mg/kg/d) for the domestic chicken (Scott, 1977), are used to develop TRVs for the tree swallow. This study is selected for calculating TRVs for the tree swallow because it shows a clear dose-response relationship with a meaningful endpoint. Scott (1977) found significantly reduced hatchability in the eggs of hens that had been fed PCBs for a period of 4 or 8 weeks. A subchronic-to-chronic uncertainty factor of 10 is applied to the reported value to account for the short-term exposure. Because gallinaceous birds, such as the chicken, are among the most sensitive of avian species to the effects of PCBs, an interspecies uncertainty factor is not applied.

On the basis of laboratory toxicity studies:

The LOAEL TRV for the tree swallow is 0.07 mg PCBs/kg/day (Table B-26).
The NOAEL TRV for the tree swallow is 0.01 mg PCBs/kg/day (Table B-26).

Two field studies were identified that examined concentrations of PCBs in food of tree swallows in comparison to measures of reproductive effects (Table B-10). Custer *et al.* (1998) reported that measures of reproductive success (*e.g.*, clutch and egg success) were not significantly different for birds from a PCB-contaminated site in comparison to birds from a reference site. In that study, dietary doses of PCBs, estimated on the basis of average measured food concentrations at the site (2 samples) and a food ingestion rate of 0.9 kg food/kg body wt/day for the tree swallow, ranged from 0.38 to 0.55 mg PCBs/kg/day.

Dietary doses of PCBs to tree swallows can also be estimated on the basis of composite samples of food taken from feeding tree swallows on the Hudson River in 1995 (USEPA, 1998). Dietary doses (estimated using the aforementioned food ingestion rate) for the tree swallow at three locations on the Hudson River are 0.08, 6.0, and 16.1 mg PCBs/kg/day. The final TRV is based on the highest concentration shown to be without adverse effects in both field studies, a value of 16.1 mg PCBs/kg/day.

On the basis of field studies:

The NOAEL TRV for the tree swallow is 16.1 mg PCBs/kg/day (Table B-26).

Total Dioxin Equivalents (TEQs) in the Diet of the Tree Swallow

No laboratory studies were identified that examined the toxicity of dioxin-like compounds in the diet of the tree swallow or for a bird in the same taxonomic family or order as the tree swallow

(Tables B-11 and Figure B-5). Therefore, the lowest values from the selected applicable studies (Table B-11), the NOAEL (0.014 μg TEQs/kg/day) and corresponding LOAEL (0.0014 μg TEQs/kg/day) for the pheasant (Nosek *et al.*, 1992) are used to develop TRVs for the tree swallow. Because gallinaceous birds, such as the pheasant, are among the most sensitive to 2,3,7,8-TCDD (Table B-11), an interspecies uncertainty factor is not applied. Because of the short-term nature of the exposure (10 weeks), a subchronic-to-chronic uncertainty factor of 10 is applied.

On the basis of laboratory toxicity studies:
The LOAEL TRV for the tree swallow is 0.014 μg TEQs/kg/day (Table B-26).
The NOAEL TRV for the tree swallow is 0.0014 μg TEQs/kg/day (Table B-26).

Note that the study by Nosek *et al.* (1992) was also selected by the USEPA as the basis for development of concentrations of 2,3,7,8-TCDD associated with risk to avian receptors (USEPA, 1993).

Two field studies were identified that examined the effects of dioxin-like compounds in the diets of tree swallows (Table B-12). Custer *et al.* (1998) reported that measures of reproductive success (*e.g.*, clutch and egg success) were not significantly different for birds from a PCB-contaminated site in comparison to birds from a reference site. In that study, dietary doses of dioxin-like compounds were as high as 0.08 μg TEQs/kg/day.

Dietary doses of dioxin-like compounds to the tree swallow can also be estimated on the basis of composite samples of food taken from feeding tree swallows on the Hudson River in 1995 (USEPA, 1998). Dietary doses (estimated using the aforementioned food ingestion rate) for the tree swallow at three locations on the Hudson River are: 0.12, 1.8, and 4.9 μg TEQs/kg/day. The final TRV is based on the highest concentration shown to be without adverse effects in the 1995 field study, a value of 4.9 μg TEQs/kg/day.

On the basis of the field studies:

The NOAEL TRV for the tree swallow is 4.9 μg TEQs/kg/day (Table B-26).

Total PCBs in Eggs of the Tree Swallow

No laboratory studies were identified that examined the toxicity of PCBs in eggs of the tree swallow or for a bird in the same taxonomic family or order as the tree swallow (Table B-13 and Figure B-6). Therefore, the lowest appropriate NOAEL and corresponding LOAEL from the selected applicable studies (Table B-13) are used to develop TRVs for the tree swallow. The study by Scott (1977) on chickens is selected for development of TRVs. This study is selected for calculating TRVs for the tree swallow because it shows a clear dose-response with a meaningful endpoint. Scott (1977) found significantly reduced hatchability in the eggs of hens that had been fed PCBs for a period of 4 or 8 weeks. Because gallinaceous birds, such as the chicken, are among the most sensitive of avian species to the effects of PCBs, an interspecies uncertainty factor is not applied. Because the experimental study measured actual concentrations in the egg, rather than reporting a surrogate dose, a subchronic-to-chronic uncertainty factor is not applied.

On the basis of laboratory toxicity studies:

The LOAEL TRV for the tree swallow egg is 2.21 mg PCBs/kg egg (Table B-26).
The NOAEL TRV for the tree swallow egg is 0.33 mg PCBs/kg egg (Table B-26).

Several field studies were identified that examined effects of PCBs on eggs of the tree swallow (Table B-14). Custer *et al.* (1998) found that clutch success (the probability of a clutch hatching at least one young) and egg success (the probability of an egg hatching in a successful nest) were not significantly lower at two contaminated sites in comparison to reference sites. Average concentrations of total PCBs in eggs and pippers (newly hatched young) near a PCB contaminated site ranged from 0.95 to 3.85 mg PCBs/kg and were significantly higher than concentrations from the reference site, which ranged from 0.05 to 0.77 mg PCBs/kg.

The United States Fish and Wildlife Service (USFWS) studied the effects of PCB contamination on tree swallows in the Upper Hudson River Valley in 1994 and 1995 (Secord and McCarty, 1997, McCarty and Secord, 1999). Concentrations of PCBs were measured in tree swallow eggs and nestlings from three sites on the Hudson River, one reference site on the Champlain Canal, and one reference site in Ithaca, NY. Because concentrations of PCBs are not usually measured in whole birds, concentrations of PCBs measured in whole bodies of Hudson River tree swallows are not considered in this risk assessment.

In 1994, the mean mass of nestlings on the day of hatching from all of the Hudson River sites combined was significantly less than the mean mass of nestlings from the Ithaca site. Reproductive success at the Hudson sites was significantly impaired relative to other sites in New York due to reduced hatchability and increased levels of nest abandonment during incubation, but clutch size, nestling survival, and nestling growth and development were all normal. Average concentrations of total PCBs in swallow eggs measured in 1994 were 11.7, 12.4, and 42.1 mg/kg wet wt for three Hudson River sites, and 6.28 mg/kg wet wt for the Champlain Canal reference site (Secord and McCarty, 1997).

In 1995 reproductive output of swallows at the Hudson sites was normal, but higher than expected rates of abandonment and supernormal clutch size persisted. Growth and development of nestlings was not significantly impaired. Average concentrations of PCBs in swallow eggs reported in this subsequent study were 5.3, 24.1, and 26.7 mg/kg wet wt at the three Hudson sites, 5.9 mg/kg at the Champlain Canal reference site, 1.85 mg/kg wet wt at an inland reference site, and 0.209 mg/kg wet wt at the Ithaca reference site.

Reproductive success in 1994 may have been influenced by the large number of young females that typically inhabit nest boxes the first year that they are placed in the field (Secord and McCarty, 1997). Because of the lack of a consistent pattern of reproductive success between the two years of the study, these results are not used to establish a LOAEL TRV for the swallow. These results do suggest, however, that tree swallows are more resistant to the effects of PCBs than are many other species studied, and results can be used to derive a NOAEL TRV. Because of the obvious relevance of the Hudson River study to the present assessment, the data from Secord and McCarty are selected for development of a field-based TRV for the tree swallow. The highest concentration from the year without significant effects is used to establish this field-based NOAEL TRV for tree swallows.

On the basis of field toxicity studies:

The NOAEL TRV for tree swallows is 26.7 mg PCBs/kg egg (Table B-26).

Total Dioxin Equivalents (TEQs) in Eggs of the Tree Swallow

No laboratory studies were identified that examined the toxicity of dioxin-like compounds in the eggs of the tree swallow or for a bird in the same taxonomic family as the tree swallow (Table B-15 and Figure B-7). Therefore, the lowest appropriate NOAEL (0.01 µg TEQs/kg egg) and LOAEL (0.02 µg TEQs/kg egg) from the applicable studies are used to develop TRVs for the tree swallow. Powell *et al.* (1996a) found significantly reduced hatchability in eggs of domestic chickens that were injected with 0.2 µg PCB 126/kg egg. This effect was not observed in eggs injected with 0.1 µg PCB 126/kg egg. The effective concentrations of BZ#126 are multiplied by the TEF (0.1) for BZ#126 to estimate TRVs. Because gallinaceous birds, such as the chicken, are among the most sensitive of avian species to the effects of dioxin-like compounds, an interspecies uncertainty factor is not applied. Because by nature, a hatching period is a short-term event, a subchronic-to-chronic uncertainty factor is not applied.

On the basis of laboratory toxicity studies:

The LOAEL TRV for the tree swallow is 0.02 µg TEQs/kg egg (Table B-26).
The NOAEL TRV for the tree swallow is 0.01 µg TEQs/kg egg (Table B-26).

Two field studies were identified that examined effects of dioxin-like compounds on tree swallows (Table B-16). Field studies conducted in 1994 and 1995 reported elevated concentrations of dioxin-like compounds in tree swallow eggs at contaminated Hudson River sites in comparison to reference sites (USEPA, 1998). As noted in the discussion above regarding PCBs in tree swallow eggs, reproductive success was significantly reduced in 1994, but not in 1995. Because of the lack of a consistent pattern of reproductive success between the two years of the study, these results are not used to establish a LOAEL TRV for the swallow. The results do suggest, however, that tree swallows are more resistant to the effects of PCBs than are many other species studied, and the results can be used to derive a NOAEL TRV. The highest average concentration from the year without significant adverse effects on reproduction, growth, or mortality (13 µg TEQs/kg egg at the Remnant Site in 1995) is used to establish this field-based NOAEL TRV for tree swallows.

On the basis of field toxicity studies:

The NOAEL TRV for the tree swallows is 13 µg TEQs/kg egg (Table B-26).

B.2.4.2 Mallard (*Anas platyrhychos*)

Total PCBs in Diet of the Mallard

Three laboratory studies were identified which examined effects of PCBs in the diet on mallards (Table B-9, Figure B-4). The study that reported the lowest NOAEL is selected for development of TRVs for the mallard. Custer and Heinz (1980) observed no adverse effects on reproduction after approximately 1 month on a dosage of 2.6 mg Aroclor 1254/kg/day. Because of the short-term exposure period of the experimental study (1 month), a subchronic-to-chronic uncertainty factor of 10 is applied to the reported NOAEL. A LOAEL was not provided in this study, so the LOAEL is assumed to be 10 times the estimated NOAEL for the mallard.

On the basis of laboratory toxicity studies:

The LOAEL TRV for the mallard is 2.6 mg PCBs/kg/day (Table B-26).
The NOAEL TRV for the mallard is 0.26 mg PCBs/kg/day (Table B-26).

No field studies were identified that examined effects of dietary exposure to PCBs on reproduction, growth or mortality of the mallard or on a species in the same taxonomic family as the mallard (Table B-10).

Total Dioxin Equivalents (TEQs) in Diet of the Mallard

No laboratory studies were identified that examined the toxicity of dioxin-like compounds in the diet of the mallard or for a bird in the same taxonomic family or order as the mallard (Tables B-11 and Figure B-5). Therefore, the lowest appropriate LOAEL (0.14 µg TEQs/kg/day) and NOAEL (0.014 µg TEQs/kg/day) from the selected applicable studies (Table B-11) (Nosek *et al.*, 1992) are used to develop TRVs for the mallard. Nosek *et al.* (1992) observed reduced fertility and increased embryo mortality in ring-necked pheasants that received weekly intraperitoneal injections of 2,3,7,8-TCDD over the course 10 weeks. It is generally acknowledged that intraperitoneal injection and oral routes of exposure are similar because in both instances the chemical is absorbed by the liver, thereby permitting first-pass metabolism (USEPA, 1995). Because data indicate that the mallard (LD_{50} > 108 mg/kg/day for a single dose) is less sensitive than the pheasant (LD_{75} = 25 mg/kg/day for a single dose) to the acute effects of 2,3,7,8-TCDD (Table B-11), an interspecies uncertainty factor is not applied. Because of the short-term nature of the exposure in this study (10 weeks), a subchronic-to-chronic uncertainty factor of 10 is applied.

On the basis of laboratory toxicity studies:

The LOAEL TRV for the mallard is 0.014 µg TEQs/kg/day (Table B-26).
The NOAEL TRV for the mallard is 0.0014 µg TEQs/kg/day (Table B-26).

No field studies were identified that examined effects of dietary exposure to dioxin-like compounds on reproduction, growth or mortality of the mallard or on a species in the same taxonomic family as the mallard (Table B-12).

Total PCBs in Eggs of the Mallard

No laboratory studies were identified that examined the toxicity of PCBs in eggs of the mallard or for a bird in the same taxonomic family or order as the mallard (Table B-13 and Figure B-6). Therefore, the lowest appropriate LOAEL and NOAEL from the selected applicable studies (Table B-13) are used to develop TRVs for the mallard. The study by Scott (1977) on chickens is selected for development of TRVs. This study is selected for calculating TRVs for the mallard because it shows a clear dose-response with a meaningful endpoint. Scott (1977) found significantly reduced hatchability in the eggs of hens that had been fed PCBs for a period of either 4 or 8 weeks. Because gallinaceous birds, such as the chicken, are among the most sensitive of avian species to the effects of PCBs, an interspecies uncertainty factor is not applied. Because the study measured actual concentrations in the egg, rather than reporting a surrogate dose, a subchronic-to-chronic uncertainty factor is not applied.

On the basis of laboratory toxicity studies:

The LOAEL TRV for the mallard egg is 2.21 mg PCBs/kg egg (Table B-26).
The NOAEL TRV for the mallard egg is 0.33 mg PCBs/kg egg (Table B-26).

No field studies were identified that examined effects of PCBs in eggs of the mallard or in eggs of a species in the same taxonomic family as the mallard (Table B-14).

Total Dioxin Equivalents (TEQs) in Eggs of the Mallard

No laboratory studies were identified that examined the toxicity of dioxin-like compounds in the eggs of the mallard or for a bird in the same taxonomic family as the mallard (Table B-15 and Figure B-7). Therefore, the lowest appropriate NOAEL (0.01 μg TEQs/kg egg) and corresponding LOAEL (0.02 μg TEQs/kg egg) from the applicable studies are used to develop TRVs for the mallard. Powell *et al.* (1996a) found significantly reduced hatchability in domestic chicken eggs that were injected with 0.2 μg BZ#126/kg egg. This effect was not observed in eggs injected with 0.1 μg BZ#126/kg egg. The effective concentrations of BZ#126 are multiplied by the avian TEF for BZ#126 (0.1) to estimate TRVs on a dioxin basis. Because gallinaceous birds, such as the chicken, are among the most sensitive of avian species to the effects of dioxin-like compounds (Table B-15), an interspecies uncertainty factor is not applied. Because the experimental study is based on an actual measured dose to the egg, rather than on a surrogate dose, a subchronic-to-chronic uncertainty factor is not applied.

On the basis of laboratory toxicity studies:

The LOAEL TRV for the mallard egg is 0.02 μg TEQs/kg egg (Table B-26).

The NOAEL TRV for the mallard egg is 0.01 μg TEQs/kg egg (Table B-26).

Two field studies were identified that examined effects dioxin-like compounds in eggs of the wood duck, *Aix sponsa*, a species in the same family as the mallard (Tables B-16 and B-23). These studies reported significant negative correlations between measures of reproductive effects and concentrations of TEQs in eggs of wood ducks (White and Segniak, 1994 White and Hoffman, 1995). These studies reported substantially reduced nest success, hatching success, and duckling production, at concentrations of 0.020 μg TEQs/kg egg. These effects were not observed at concentrations of 0.005 μg TEQs/kg egg. Measured concentrations of organochlorine pesticides and PCBs were low and were not believed to be biologically significant. Because of the relevance of this study to the mallard, the LOAEL (0.02 μg TEQs/kg egg) and NOAEL (0.005 μg TEQs/kg egg) from these studies are selected for development of a field-based TRV for the mallard. Note that this study used TEFs provided by USEPA (1989) to calculate TEQs, which may differ slightly from TEFs used in this report (Van den Berg *et al.* , 1998). Potential differences in effect concentrations that are based on use of differing TEFs are estimated at 12 to 30% (See sections on great blue herons and mink). Because the mallard and the wood duck are in the same family, an interspecies uncertainty factor is not applied. Because the LOAEL and NOAEL are based on measured concentrations, a subchronic-to-chronic uncertainty factor is not applied.

On the basis of field studies:

The LOAEL TRV for the mallard egg is 0.02 μg TEQs/kg egg (Table B-26).
The NOAEL TRV for the mallard egg is 0.005 μ TEgQs/kg egg (Table B-26).

B.2.4.3 Belted kingfisher (*Ceryle alcyon*)

Total PCBs in the Diet of the Belted Kingfisher

No laboratory studies were identified that examined the toxicity of PCBs in the diet of the belted kingfisher or for a bird in the same taxonomic family or order as the kingfisher (Table B-9, Figure B-4). Therefore, the lowest appropriate NOAEL (0.1 mg/kg/d) and corresponding LOAEL (0.7 mg/kg/d) for the domestic chicken (Scott, 1977) are used to develop TRVs for the belted kingfisher. This study is selected for calculating TRVs because it shows a clear dose-response relationship with a meaningful endpoint. Scott (1977) found significantly reduced hatchability in the eggs of hens that had been fed PCBs for a period of 4 or 8 weeks. A subchronic-to-chronic uncertainty factor of 10 is applied to the reported value to account for the short-term exposure. Because gallinaceous birds, such as the chicken, are among the most sensitive of avian species to the effects of PCBs (Table B-9), an interspecies uncertainty factor is not applied. Because by nature a hatching period is a short-term event, a subchronic-to-chronic uncertainty factor is not applied.

On the basis of laboratory toxicity studies:

The LOAEL TRV for the belted kingfisher is 0.07 mg PCBs/kg/day (Table B-26).
The NOAEL TRV for the belted kingfisher is 0.01 mg PCBs/kg/day (Table B-26).

No field studies were identified that examined effects of dietary exposure to PCBs on growth, reproduction, or mortality of the belted kingfisher or to a species in the same taxonomic family as the kingfisher (Table B-10).

Total Dioxin Equivalents (TEQs) in the Diet of the Belted Kingfisher

No laboratory studies were identified that examined the toxicity of dioxin-like compounds in the diet of the belted kingfisher or for a bird in the same taxonomic family or order as the kingfisher (Tables B-11 and Figure B-5). Therefore, the lowest appropriate values from the selected applicable studies (Table B-11), the NOAEL (0.014 µg TEQs/kg/day) and LOAEL (0.14 µg TEQs/kg/day) for the pheasant (Nosek *et al.* , 1992), are used to develop TRVs for the kingfisher. Because gallinaceous birds, such as the pheasant, are among the most sensitive birds to the effects of dioxin-like compounds (Table B-11), an interspecies uncertainty factor is not applied. Because of the short-term nature of the exposure (10 weeks), a subchronic-to-chronic uncertainty factor of 10 is applied.

On the basis of laboratory toxicity studies:

The LOAEL TRV for the belted kingfisher is 0.014 µg TEQs/kg/day (Table B-26).
The NOAEL TRV for the belted kingfisher is 0.0014 µg TEQs/kg/day (Table B-26).

No field studies were identified that examined effects of dietary exposure to dioxin-like compounds on growth, reproduction, or mortality of the belted kingfisher or a species in the same family as the kingfisher (Table B-12).

Total PCBs in Eggs of the Belted Kingfisher

No laboratory studies were identified that examined the toxicity of PCBs in eggs of the belted kingfisher or in eggs of a bird in the same order as the kingfisher (Tables B-13 and Figure B-6). Therefore, the lowest appropriate NOAEL and LOAEL from the selected applicable studies (Table B-13) are used to develop TRVs for the belted kingfisher. The study by Scott (1977) is selected for development of TRVs since this study reports the lowest effect levels and provides both a NOAEL and a LOAEL. Because gallinaceous birds, such as the chicken, are among the most sensitive of avian species to the effects of PCBs, an interspecies uncertainty factor is not applied. Because by nature, a hatching period is a short-term event, a subchronic-to-chronic uncertainty factor is not applied.

On the basis of laboratory toxicity studies:

The LOAEL TRV for the belted kingfisher is 2.21 mg PCBs/kg egg (Table B-26).
The NOAEL TRV for the belted kingfisher is 0.33 mg PCBs/kg egg (Table B-26).

No field studies were identified that examined effects of PCBs in eggs of the belted kingfisher or on a species in the same taxonomic family as the kingfisher (Table B-14).

Total Dioxin Equivalents (TEQs) in Eggs of the Belted Kingfisher

No laboratory studies were identified that examined the toxicity of dioxin-like compounds in the eggs of the belted kingfisher or for a bird in the same taxonomic family as the kingfisher (Tables B-15 and Figure B-7). Therefore, the lowest appropriate NOAEL (0.01 µg TEQs/kg egg) and LOAEL (0.02 µg TEQs/kg egg) from the applicable studies are used to develop TRVs for the belted kingfisher. Powell *et al.* (1996a) found significantly reduced hatchability in domestic chicken eggs that were injected with 0.2 µg PCB 126/kg egg. This effect was not observed in eggs injected with 0.1 µg BZ#126/kg egg. The effective concentrations of BZ#126 are multiplied by the avian TEF for BZ#126 (0.1) to estimate TRVs on a dioxin basis. Because gallinaceous birds, such as the chicken, are among the most sensitive of avian species to the effects of dioxin-like compounds (Table B-15), an interspecies uncertainty factor is not applied. Because by nature a hatching period is a short-term event, a subchronic-to-chronic uncertainty factor is not applied.

On the basis of laboratory toxicity studies:

The LOAEL TRV for the belted kingfisher egg is 0.02 µg TEQs/kg egg (Table B-26).
The NOAEL TRV for the belted kingfisher egg is 0.01 µg TEQs/kg egg (Table B-26).

No field studies were identified that examined effects of dioxin-like compounds on eggs of the belted kingfisher or on a bird in the same taxonomic family as the kingfisher (Table B-16).

B.2.4.4 Great Blue Heron (*Ardea herodias*)

Total PCBs in the Diet of the Great Blue Heron

No laboratory studies were identified that examined the toxicity of PCBs in the diet of the great blue heron or a bird in the same taxonomic family or order as the heron (Table B-9, Figure B-4). Therefore, the lowest appropriate LOAEL and NOAEL from the applicable studies, the LOAEL (0.7 mg/kg/d) and NOAEL (0.1 mg/kg/d) for the domestic chicken (Scott, 1977), are used to develop TRVs for the great blue heron. Scott (1977) found significantly reduced hatchability in the eggs of hens that had been fed PCBs for a period of 4 or 8 weeks. A subchronic-to-chronic uncertainty factor of 10 is applied to the reported value to account for the short-term exposure. Because gallinaceous birds, such as the chicken, are among the most sensitive of avian species to the effects of PCBs, an interspecies uncertainty factor is not applied.

On the basis of laboratory toxicity studies:

The LOAEL TRV for the great blue heron is 0.07 mg PCBs/kg/day (Table B-26).
The NOAEL TRV for the great blue heron is 0.01 mg PCBs/kg/day (Table B-26).

No field studies were identified that examined effects of dietary exposure to PCB compounds on growth, reproduction, or mortality of the great blue heron or on a species in the same taxonomic family as the great blue heron (Table B-10).

Total Dioxin Equivalents (TEQs) in the Diet of the Great Blue Heron

No laboratory studies were identified that examined the toxicity of dioxin-like compounds in the diet of the great blue heron or for a bird in the same taxonomic family or order as the heron (Tables B-11 and Figure B-5). Therefore, the lowest appropriate values from the selected applicable studies (Table B-11), the NOAEL (0.014 µg TEQs/kg/day) and LOAEL (0.14 µg TEQs/kg/day) for the pheasant (Nosek *et al.* , 1992), are used to develop TRVs for the great blue heron. Because gallinaceous birds, such as the pheasant, are among the most sensitive birds to the effect 2,3,7,8-TCDD (Table B-11), an interspecies uncertainty factor is not applied. Because of the short-term nature of the exposure of the experimental study (10 weeks), a subchronic-to-chronic uncertainty factor of 10 is applied.

On the basis of laboratory toxicity studies:

The LOAEL TRV for the great blue heron is 0.014 µg TEQs/kg/day (Table B-26).
The NOAEL TRV for the great blue heron is 0.0014 µg TEQs/kg/day (Table B-26).

No field studies were identified that examined effects of dietary exposure to dioxin-like compounds on growth, reproduction, or mortality of the great blue heron or on a species in the same taxonomic family as the great blue heron (Table B-12).

Total PCBs in Eggs of the Great Blue Heron

No laboratory studies were identified that examined the toxicity of PCBs in eggs of the great blue heron or for a bird in the same taxonomic family or order as the heron (Tables B-13 and Figure B-6). Therefore, the lowest appropriate NOAEL and LOAEL (Scott, 1977) from the selected applicable studies (Table B-13) are used to develop TRVs for the great blue heron. Because gallinaceous birds, such as the chicken, are among the most sensitive of avian species to the effects of PCBs (Table B-13), an interspecies uncertainty factor is not applied. Because by nature, a hatching period is a short-term event, a subchronic-to-chronic uncertainty factor is not applied.

On the basis of laboratory toxicity studies:

The LOAEL TRV for great blue heron eggs is 2.21 mg PCBs/kg egg (Table B-26).
The NOAEL TRV for great blue heron eggs is 0.33 mg PCBs/kg egg (Table B-26).

No field studies were identified that examined effects of PCBs to eggs of the great blue heron or for eggs of a species in the same taxonomic family as the great blue heron (Table B-14).

Total Dioxin Equivalents (TEQs) in Eggs of the Great Blue Heron

One laboratory study was identified that examined effects of dioxin-like compounds on eggs of the great blue heron (Table B-15). Janz and Bellward (1996) found no substantial adverse effect on hatchability or growth rate of chicks from great blue heron eggs that were injected with 2 µg

2,3,7,8-TCDD/kg egg. Because the study reports a measured dose to the egg rather than a surrogate dose, no subchronic-to-chronic uncertainty factor is applied. Because the study was conducted on the great blue heron, no interspecies uncertainty factor is applied.

On the basis of the laboratory toxicity study:
The NOAEL TRV for the great blue heron is 2.0 µg TEQs/kg egg (Table B-26).

Three field studies were identified that examined the effects of dioxins, furans, and PCBs in field-collected eggs of the great blue heron at a site in British Columbia (Table B-16). One of the studies documented complete reproductive failure in a colony of great blue herons with average egg concentrations of 0.23 µg TEQs/kg egg in the 1986-1987 season (Elliott *et al.* , 1989). Average concentrations of TEQs in great blue heron eggs from the same failed colony in 1988 were greater than 0.5 µg TEQs/kg egg (Hart *et al.* , 1991, Sanderson *et al.* , 1994). The study by Sanderson *et al.* (1994) is selected for development of TRVs for the great blue heron because this study reported concentrations of PCBs, in addition to concentrations of dioxins and furans. Sanderson *et al.* (1994) reported no significant difference in hatchability of eggs, but a significant reduction in body weight associated with egg concentrations greater than 0.5 µg TEQs/kg egg (Sanderson *et al.* , 1994). This effect was not observed at egg concentrations of approximately 0.3 µg TEQs/kg egg (Sanderson *et al.* , 1994). TEQs calculated by Sanderson *et al.* (1994) at the same site using the TEF values of Safe *et al.* (1990) are estimated to be 30% lower than the concentration of TEQs that would be calculated using the TEFs of Van den Berg *et al.* (1998) that are used in the present report. The LOAEL (0.5 µg/kg egg) and NOAEL (0.3 µg TEQs/kg egg) from this study (Sanderson *et al.* , 1994) are selected for development of a field-based TRV for the great blue heron. Because the LOAEL and NOAEL endpoints are based on measured concentrations, a subchronic-to-chronic uncertainty factor is not applied.

On the basis of field toxicity studies:

The LOAEL TRV for the great blue heron is 0.5 µg TEQs/kg egg (Table B-26).
The NOAEL TRV for the great blue heron is 0.3 µg TEQs/kg egg (Table B-26).

B.2.4.5 Bald eagle (*Haliaeetus leucocephalus*)

Total PCBs in the Diet of the Bald Eagle

No laboratory studies were identified that examined the toxicity of PCBs in the diet of the bald eagle or a bird in the same taxonomic family or order as the bald eagle (Table B-9, Figure B-4). Therefore, the lowest appropriate the NOAEL (0.1 mg/kg/d) and corresponding LOAEL (0.7 mg/kg/d) for the domestic chicken (Scott, 1977), are used to develop TRVs for the great blue heron. Scott (1977) found significantly reduced hatchability in the eggs of hens that had been fed PCBs for a period of 4 or 8 weeks. A subchronic-to-chronic uncertainty factor of 10 is applied to the reported value to account for the short exposure period of the experimental study (up to 8 weeks). Because gallinaceous birds, such as the chicken, are among the most sensitive of avian species to the effects of PCBs, an interspecies uncertainty factor is not applied.

On the basis of laboratory toxicity studies:

The LOAEL TRV for the bald eagle is 0.07 mg PCBs/kg/day (Table B-26).
The NOAEL TRV for the bald eagle is 0.01 mg PCBs/kg/day (Table B-26).

No field studies were identified that examined effects of dietary exposure to PCBs on growth, reproduction, or mortality of the bald eagle or on a species in the same taxonomic family as the bald eagle (Table B-10).

Total Dioxin Equivalents (TEQs) in the Diet of the Bald Eagle

No laboratory studies were identified that examined the toxicity of dioxin-like compounds in the diet of the bald eagle or for a bird in the same taxonomic family or order as the bald eagle (Tables B-11 and Figure B-5). Therefore, the lowest values from the selected applicable studies (Table B-11), the NOAEL (0.014 μg TEQs/kg/day) and LOAEL (0.14 μg TEQs/kg/day) for the pheasant (Nosek *et al.* , 1992) are used to develop TRVs for the bald eagle. Because gallinaceous birds, such as the pheasant, are among the most sensitive birds to the effects 2,3,7,8-TCDD (Table B-11), an interspecies uncertainty factor is not applied. Because of the short-term nature of the exposure (10 weeks), a subchronic-to-chronic uncertainty factor of 10 is applied. These TRVs are expected to be protective of the bald eagle.

On the basis of laboratory toxicity studies:

The LOAEL TRV for the bald eagle is 0.014 μg TEQs/kg/day (Table B-26).
The NOAEL TRV for the bald eagle is 0.0014 μg TEQs/kg/day (Table B-26).

No field studies were identified that examined effects of dietary exposure to dioxin-like compounds on growth, reproduction, or mortality of the bald eagle or on a species in the same taxonomic family as the bald eagle (Table B-12).

Total PCBs in Eggs of the Bald Eagle

No laboratory studies were identified that examined the toxicity of PCBs in eggs of the bald eagle or for a bird in the same taxonomic family or order as the bald eagle (Table B-13 and Figure B-6). Therefore, the lowest appropriate NOAEL and corresponding LOAEL from the selected applicable studies (Table B-13) are used to develop TRVs for the bald eagle. The study by Scott (1977) is selected for development of TRVs since this study reports a NOAEL and a LOAEL for a meaningful reproductive endpoint. Because gallinaceous birds, such as the chicken, are among the most sensitive of avian species to the effects of PCBs (Table B-13), an interspecies uncertainty factor is not applied. Because by nature, a hatching period is a short-term event, a subchronic-to-chronic uncertainty factor is not applied. These TRVs are expected to be protective of the bald eagle.

On the basis of laboratory toxicity studies:

The LOAEL TRV for the bald eagle is 2.21 mg PCBs/kg egg (Table B-26).
The NOAEL TRV for the bald eagle is 0.33 mg PCBs/kg egg (Table B-26).

Several field studies were identified that examined the effects of PCBs in eggs of bald eagles (Table B-14). Clark *et al.* (1998) presented information on concentrations of total PCBs (range = 20 to 54 mg/kg egg) and TEQs in eggs from two sites in New Jersey where reproductive failures have occurred, but the data could not be used to establish NOAEL or LOAELs. Studies by Wiemeyer *et al.* (1984, 1993) reported adverse effects on mean 5-year production in bald eagle with egg concentrations greater than 3.0 mg PCBs/kg egg. Because significant intercorrelation of many contaminants made it difficult to determine which contaminants had cause the adverse effects (Wiemeyer, 1993), these studies can not be used to establish a field-based LOAEL for the effects of PCBs. However, a field-based NOAEL of 3.0 mg PCBs/kg egg can be established on the basis of this study for the bald eagle (Wiemeyer *et al.*, 1993). This NOAEL is expected to be protective of the bald eagle.

On the basis of field toxicity studies:

The NOAEL TRV for the bald eagle is 3.0 mg PCBs/kg egg (Table B-26).

Total Dioxin Equivalents (TEQs) in Eggs of the Bald Eagle

No laboratory studies were identified that examined the toxicity of dioxin-like compounds in the eggs of the bald eagle or for eggs of a bird in the same taxonomic family as the bald eagle (Table B-15 and Figure B-7). Therefore, the lowest appropriate NOAEL (0.01 μg TEQs/kg egg) and corresponding LOAEL (0.02 μg TEQs/kg egg) from the applicable studies (Table B-15) are used to develop TRVs for the bald eagle. Powell *et al.* (1996a) found significantly reduced hatchability in domestic chicken eggs that were injected with 0.2 μg BZ#126/kg egg. This effect was not observed in eggs injected with 0.1 μg BZ#126/kg egg. The effective concentrations of BZ#126 are multiplied by the avian TEF for BZ#126 (0.1) to estimate TRVs on a dioxin basis. Because gallinaceous birds, such as the chicken, are among the most sensitive of avian species to the effects of dioxin-like compounds (Table B-15), an interspecies uncertainty factor is not applied. Because by nature, a hatching period is a short-term event, a subchronic-to-chronic uncertainty factor is not applied.

On the basis of laboratory toxicity studies:

The LOAEL TRV for the bald eagle is 0.02 μg TEQs/kg egg (Table B-26).
The NOAEL TRV for the bald eagle is 0.01 μg TEQs/kg egg (Table B-26).

A field study by Clark *et al.* (1998) presented information regarding concentrations of TEQs (range = 0.513 to 1.159 μg/kg) in bald eagle eggs from two sites in New Jersey where reproductive failures have occurred. However, these data were not detailed enough to establish NOAEL TRV.

B.2.5 Selection of TRVs for Mammalian Receptors

B.2.5.1 Little brown bat (*Myotis lucifugus*)

Total PCBs in the Diet of the Little Brown Bat

No laboratory studies that examined the effects of PCBs on bats or on a species in the same taxonomic family or order as the bat were identified (Table B-17 and Figure B-9). Therefore, the lowest appropriate NOAEL (0.32 mg/kg/day) and corresponding LOAEL (1.5 mg/kg/day) from the applicable studies (Table B-17) are selected for the development of TRVs for the little brown bat. The study by Linder *et al.* (1974) is selected over other studies because it is a multigenerational study, and thus more robust. In this study, mating pairs of rats and their offspring were fed PCBs in the diet. Offspring of rats fed Aroclor 1254 at a dose of 1.5 mg/kg/day exhibited decreased litter size in comparison to controls. This effect was not observed at a dose of 0.32 mg/kg/day. An uncertainty factor of 10 is applied to account for potential differences in sensitivity to PCBs between the rat and the little brown bat (Table B-27). Because of the extended duration of the experimental study (2 generations) a subchronic-to-chronic uncertainty factor is not applied.

On the basis of laboratory toxicity studies:

The LOAEL TRV for the little brown bat is 0.15 mg PCBs/kg/day (Table B-27).
The NOAEL TRV for the little brown bat is 0.032 mg PCBs/kg/day (Table B-27).

Several field studies were identified that examined the effects of PCBs on bats (Clark, 1978, Clark and Krynitsky, 1978; Clark and Lamont, 1976). However, these studies are not used to select TRVs because effect endpoints in these studies are reported on the basis of concentrations of PCBs in bat tissue, rather than as dietary doses. No field studies were identified that examined effects of dietary exposure to PCBs on growth, reproduction, or mortality of the little brown bat or on a species in the same family as the little brown bat. These studies are not presented in a table due to their overall lack of relevance to the development of TRVs for mammals.

Total Dioxin Equivalents (TEQs) in the Diet of the Little Brown Bat

No laboratory studies were identified that examined effects of dioxin-like compounds on bats bats or on a species in the same taxonomic family or order as the bat were identified (Tables B-18 and Figure B-10). Therefore, the multigenerational study by Murray *et al.* (1979) is selected to derive the TRV for the little brown bat. The study by Murray *et al.* (1979) was selected over the study of Bowman *et al.* , (1989b) on rhesus monkeys because the length of exposure was significantly longer than that used in the rhesus monkey study. Murray *et al.* (1979) reported a LOAEL of 0.01 μg/kg/day and a NOAEL of 0.001 μg/kg/day for adverse reproductive effects in the rat. An uncertainty factor of 10 is applied to account for potential differences between the rat and the little brown bat in sensitivity to dioxin-like compounds. Because the experimental study examined over three generations, a sub-chronic-to-chronic uncertainty factor is not applied.

On the basis of laboratory toxicity studies:

The LOAEL TRV for the little brown bat is 0.001 µg TEQs/kg/day (Table B-27).
The NOAEL TRV for the little brown bat is 0.0001 µg TEQs/kg/day (Table B-27).

Note that the study by Murray *et al.* (1979) was also selected by the USEPA as the basis for development of concentrations of 2,3,7,8-TCDD associated with risk to mammalian receptors (USEPA, 1993).

No field studies were identified that examined effects of dietary exposure to dioxin-like compounds on growth, reproduction, or mortality of the little brown bat or on a species in the same taxonomic family as the little brown bat.

B.2.5.2 Raccoon (*Procyon lotor*)

Total PCBs in the Diet of the Raccoon

One study was identified that examined acute effects (8-day exposure) of PCBs on the growth of raccoons (Montz *et al.* , 1982). Because of the difficulty in estimating chronic LOAELs and NOAELs from acute studies, this study is not used to estimate TRVs for the raccoon.

No appropriate experiments that examined the effects of PCBs on raccoons or on species in the same taxonomic family or order were identified (Table B-17 and Figure B-9). Therefore, the lowest appropriate NOAEL (0.32 mg/kg/day) and corresponding LOAEL (1.5 mg/kg/day) from the selected applicable mammalian studies (Table B-17) are selected for the development of TRVs for the raccoon. The study by Linder *et al.* (1974) is selected over other studies because it is a robust multigenerational study, in which mating pairs of rats and their offspring were fed PCBs in their diets. Offspring of rats fed Aroclor 1254 at a dose of 1.5 mg/kg/day exhibited decreased litter size in comparison to controls. This effect was not observed at a dose of 0.32 mg/kg/day.

Because acute effects of PCBs on raccoons (Montz *et al.* 1982, Table B-17) are not directly comparable to sub-chronic or chronic effects of PCBs on the rat, the sensitivities of the two species to PCBs cannot be compared. Therefore, an uncertainty factor of 10 is applied to account for potential differences in sensitivity to PCBs between the rat and the raccoon. Because of the extended duration of the experimental study (two generations), a subchronic-to-chronic uncertainty factor is not applied.

On the basis of laboratory toxicity studies:

The LOAEL TRV for the raccoon is 0.15 mg PCBs/kg/day (Table B-27).
The NOAEL TRV for the raccoon is 0.032 mg PCBs/kg/day (Table B-27).

No field studies were identified that examined effects of dietary exposure to PCBs on growth, reproduction, or mortality of the raccoon or on a species in the same taxonomic family as the raccoon.

Total Dioxin Equivalents (TEQs) in the Diet of the Raccoon

No studies were identified that examined effects of dioxin-like compounds on raccoons or a species in the same taxonomic family as the racoon (Table B-18). Therefore, the multigenerational study by Murray *et al.* (1979) is selected to derive the TRV for raccoons. Murray *et al.* (1979) observed reduced reproductive capacity in two generations of offspring of the rats that were exposed to 2,3,7,8-TCDD in the diet (Table B-18). Murray *et al.* (1979) reported a LOAEL of 0.01 µg/kg/day and a NOAEL of 0.001 µg/kg/day for these reproductive effects. An uncertainty factor of 10 is applied to account for potential differences between the rat and the raccoon in sensitivity to dioxin-like compounds. Because the experimental study examined exposure over three generations, a subchronic-to-chronic uncertainty factor is not applied.

On the basis of laboratory toxicity studies:

The LOAEL TRV for the raccoon is 0.001 µg TEQs/kg/day (Table B-27).
The NOAEL TRV for the raccoon is 0.0001 µg TEQs/kg/day (Table B-27).

No field studies were identified that examined effects of dietary exposure to dioxin-like compounds on growth, reproduction, or mortality of the raccoon or on a species in the same taxonomic family as the raccoon.

B.2.5.3 Mink (*Mustela vison*)

Total PCBs in the Diet of the Mink

Numerous studies have evaluated the effects of total PCBs on mortality, growth and reproduction in mink (Table B-19 and Figure B-8). The lowest effective dose in the selected applicable studies (Table B-19) (Platanow and Karstad, 1973) is not selected for development of TRVs because that study compared growth and reproduction of PCB-treated mink to the performance of an institutional herd of mink, rather than to a true experimental control group. Instead, the study of Aulerich and Ringer (1977) is selected for calculating TRVs for the mink. In this study, reproduction was markedly reduced when female mink were fed Aroclor 1254 at a dose of 0.7 mg/kg/day for a period of 4 months. These effects were not observed at a dose of 0.1 mg/kg/day. A subchronic-to-chronic uncertainty factor of 10 is applied to the reported LOAEL and NOAEL to account for the short exposure duration of the study.

On the basis of laboratory toxicity studies:

The LOAEL TRV for the mink is 0.07 mg PCBs/kg/day (Table B-27).
The NOAEL TRV for the mink is 0.01 mg PCBs/kg/day (Table B-27).

Two field studies were identified that examined effects of PCBs in the diet of the mink (Table B-20). The study that reported a lack of adverse reproductive effects at the lowest dose is used to develop TRVs for the mink. Adult ranch mink were fed diets containing various amounts of PCB-contaminated carp from Lake Michigan (Heaton *et al.* , 1995). Mink fed the contaminated diet before and during reproduction had reduced reproduction and/or growth and survival of offspring. Concentrations of other contaminants were measured and were substantially lower than concentrations of PCBs. The dietary LOAEL was 0.13 mg PCBs/kg/day. The dietary NOAEL was 0.004 mg PCBs/kg/day. Because of the extended period of exposure (128 days) a subchronic-to-chronic uncertainty factor is not applied.

On the basis of field toxicity studies:

The LOAEL TRV for the mink is 0.13 mg PCBs/kg/day (Table B-27).
The NOAEL TRV for the mink is 0.004 mg PCBs/kg/day (Table B-27).

This field study was accepted as appropriate for use in developing TRVs for the mink, and these TRVs are accepted as final TRVs for the mink, rather than the laboratory-based TRVs.

Total PCBs in the Liver of the Mink

Two studies were identified that related concentrations of PCBs in the liver of mink to adverse reproductive effects. Platanow and Karstad (1973) reported that a liver concentration of 1.23 mg/kg (weathered Aroclor 1254) corresponded to impaired reproductive success (as reported in Wren, 1991). It should be noted, however, that reproductive success in the control group of that study was also very poor in relation to that of control groups in other experiments. Reduced growth of mink kits was observed in female mink with 3.1 mg Aroclor 1254/gm liver (Wren *et al.* , 1987).

Total Dioxin Equivalents (TEQs) in the Diet of the Mink

Two studies were identified that examined acute effects (12- and 28-day exposures) of dioxin-like compounds on mink (Hochstein *et al.* , 1988, Aulerich *et al.* , 1988) (Table B-18). Because of the difficulty in estimating chronic LOAELs and NOAELs from acutely lethal doses, these studies are not used to derive TRVs for the effects of dioxin-like compounds on the mink. Instead, the study by Murray *et al.* (1979) is selected to derive TRVs for mink (Table B-18). Murray *et al.* (1979) observed reduced reproductive capacity in two generations of the offspring of rats that were exposed to 2,3,7,8-TCDD in the diet. This study was selected over the study of Bowman *et al.* , (1989b) on rhesus monkeys because: (1) the length of exposure was significantly longer than that used in the rhesus monkey study, and (2) information on the short-term toxicity (LD50) of 2,3,7,8-TCDD to the rat and the mink (Tables B-18, B-21) helps indicate the sensitivity of these two animals relative to one another. This data indicates that the mink is much more sensitive than the rat, so an inter-order uncertainty factor should be applied. Murray *et al.* (1979) reported a LOAEL of 0.01 μg/kg/day and a NOAEL of 0.001 μg/kg/day for reproductive effects in rats. An uncertainty factor of 10 is used to account for the extreme sensitivity of the mink in comparison to the rat. Because the experimental studies examined exposure to 2,3,7,8-TCDD over three generations, a subchronic-to-chronic uncertainty factor is not applied.

On the basis of laboratory toxicity studies:

The LOAEL TRV for the mink is 0.001 μg TEQs/kg/day (Table B-27).
The NOAEL TRV is for the mink is 0.0001 μg TEQs/kg/day (Table B-27).

Two field studies were identified which examined effects of dioxin-like compounds on reproduction and survival in mink (Table B-22). The study that reports adverse reproductive effects at the lowest dose is used to develop TRVs for the mink. In this study, mink were fed diets containing contaminated carp from Lake Michigan (Tillitt *et al.*, 1996). Concentrations of TEQs in the food was quantified by two methods: standard analytical chemistry and with a bioassay conducted on an extract of the food. The growth rate of kits born to the adults that were fed the carp diet were significantly reduced in comparison to controls. This effect was observed at a dose of 0.00224 μg/kg/day, but not at a dose of 0.00008 μg/kg/day. TEQs calculated by Tillitt *et al.* (1996) are estimated to be 12% higher than the concentration of TEQs that would be calculated using the TEFs of van den Berg *et al.* (1998) that are used in the present report.

On the basis of field toxicity studies:
The LOAEL for the mink is 0.00224 μg TEQs/kg/day (Table B-27).
The NOAEL for the mink is 0.00008 μg TEQs/kg/day (Table B-27).

B.2.5.4 River Otter (*Lutra canadensis*)

Total PCBs in the Diet of the River Otter

No studies were identified that examined the toxic effects of PCBs on otters (Table B-17 and Figure B-9). Because river otter and mink are in the same phylogenetic family (Table B-23), the LOAEL TRV (0.07 mg Aroclor 1254/kg/day) and NOAEL TRV (0.01 mg Aroclor 1254/kg/day) for the mink are used to develop TRVs for the otter. Since mink are generally considered to be among the most sensitive of mammalian species and otter are not expected to be more sensitive, the interspecies uncertainty factor is set to 1.

On the basis of laboratory toxicity studies:

The LOAEL TRV for the river otter is 0.07 mg PCBs/kg/day (Table B-27).
The NOAEL TRV for the river otter is 0.01 mg PCBs/kg/day (Table B-27).

Because river otters are closely related to mink, the field studies that examined effects of dietary exposure to PCBs to mink are used to develop TRVs for the river otter. Two field studies were identified that examined effects of PCBs in the diet of the mink (Table B-20). The study that reported adverse reproductive effects at the lowest dose is used to develop TRVs for the mink and the otter. Adult ranch mink were fed diets containing various amounts of PCB-contaminated carp

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(Heaton *et al.* , 1995). Mink fed the contaminated diet before and during reproduction had reduced reproduction and/or growth and survival of offspring. Concentrations of other contaminants were measured and were substantially lower than concentrations of PCBs. The dietary LOAEL was 0.13 mg PCBs/kg/day. The dietary NOAEL was 0.004 mg PCBs/kg/day.

On the basis of field studies:

The LOAEL TRV for the river otter is 0.13 mg PCBs/kg/day (Table B-27).
The NOAEL TRV for the river otter is 0.004 mg PCBs/kg/day (Table B-27).

Total Dioxin Equivalents (TEQs) in the Diet of the River Otter

No studies were identified that examined effects of dioxin-like compounds to otters or on a species in the same taxonomic family as the otter (Table B-18 and Figure B-10). The multi-generational study by Murray *et al.* (1979), which was selected as appropriate for the mink, is selected to derive TRVs for the closely related river otter. The study of Murray *et al.*, (1979) was selected over the study of Bowman *et al.* (1989b) on rhesus monkeys because the length of exposure was significantly longer than that used in the rhesus monkey study. Murray *et al.* (1979) reported a LOAEL of 0.01 μg/kg/day and a NOAEL of 0.001 μg/kg/day for adverse reproductive effects in the rat. Because of the lack of any acute or chronic toxicity data for effects of dioxin-like compounds on the river otter, an uncertainty factor of 10 is applied to account for potential differences in sensitivity to dioxin-like compounds between the rat and the river otter. Because the experimental study examined exposure over three generations, a subchronic-to-chronic uncertainty factor is not applied.

On the basis of laboratory toxicity studies:

The LOAEL TRV for the river otter is 0.001 μg TEQs/kg/day (Table B-27).
The NOAEL TRV for the river otter is 0.0001 μg TEQs/kg/day (Table B-27).

Because otters are closely related to mink, the field studies that examined effects of dietary exposure to dioxin-like compounds to mink are used to develop TRVs for the otter. Two field studies were identified that examined effects of dioxin-like compounds on reproduction and survival in mink (Table B-22). The study that reports adverse reproductive effects at the lowest dose is used to develop TRVs for the otter. In this study, mink were fed diets containing contaminated carp from Lake Michigan (Tillitt *et al.*, 1996). Concentrations of TEQs in the food was quantified by two methods: standard analytical chemistry and with a bioassay conducted on the extract of the food. The growth rate of kits born to the adults that were fed the carp diet were significantly reduced in comparison to controls. This effect was observed at a dose of 0.00224 μg/kg/day, but not at a dose of 0.00008 μg/kg/day. TEQs calculated by Tillitt *et al.* (1996) are estimated to be 12% higher than the concentration of TEQs that would be calculated using the TEFs of van den Berg *et al.* (1998) that are used in the present report. Because mink and river otter are in the same taxonomic family, an interspecies uncertainty factor is not applied. Because of the extended exposure period of the study (182 days) a subchronic-to-chronic uncertainty factor is not applied.

On the basis of field toxicity studies:

The LOAEL TRV for the river otter is 0.00224 µg TEQs/kg/day (Table B-27).
The NOAEL TRV for the river otter is 0.00008 µg TEQs/kg/day (Table B-27).

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TABLE B-1
COMMON EFFECTS OF PCB EXPOSURE IN ANIMALS

Hepatotoxicity
Hepatomegaly; bile duct hyperplasia, proliferation of smooth ER
Focal necrosis; fatty degeneration
Induction of microsomal enzymes; implications for hormone imbalances, pancreas and reproductive effects
Depletion of fat soluble vitamins (predominantly vitamin A)
Porphyria
Immunotoxicity
Atrophy of lymphoid tissues
Reduction in circulating leukocytes and lymphocytes
Suppressed antibody responses
Enhanced susceptibility to viruses
Suppression of natural killer cells
Neurotoxicity
Impaired behavioral responses
Alterations in catecholamine levels
Depressed spontaneous motor activity
Developmental deficits
Numbness in extremities
Reproduction
Increased abortion; low birth weights
Decreased survival and mating success
Increased length of estrus
Embryo and fetal mortality
Gross teratogenic effects
Biochemical, neurological, and functional changes following *in utero* exposure (mammals)
Decreased libido, decreased sperm numbers and motility
Gastrointestinal
Gastric hyperplasia
Ulceration and necrosis
Respiratory
Chronic bronchitis
Decreased vital capacity
Dermal Toxicity
Chloracne
Hyperplasia and hyperkeratosis of epithelium
Edema
Mutagenic Effects
Commercial mixtures are weakly mutagenic
Carcinogenic Effects
Preneoplastic changes
Neoplastic changes
Promotion considered main contribution
Attenuation of other carcinogens under certain conditions
Source: Hansen, L. G.. 1987. Environmental Toxicology of Polychlorinated Biphenyls in Environmental Toxin Series 1. eds. Safe, S. and Hutzinger, O., p. 32.

TABLE B-2
WORLD-HEALTH ORGANIZATION FOR TOXIC EQUIVALENCY FACTORS (TEFs) FOR HUMANS, MAMMALS, FISH, AND BIRDS

Congener	Toxic Equivalency Factor		
	Humans/Mammals	**Fish**	**Birds**
Non-*ortho* PCBs			
3,4,4',5-TetraCB (81)	0.0001	0.0005	0.1
3,3',4,4'-TetraCB (77)	0.0001	0.0001	0.05
3,3',4,4',5-PentaCB (126)	0.1	0.005	0.1
3,3',4,4',5,5'-HexaCB (169)	0.01	0.00005	0.001
Mono-*ortho* PCBs			
2,3,3',4,4'-PentaCB (105)	0.0001	<0.000005	0.0001
2,3,4,4',5-PentaCB (114)	0.0005	<0.000005	0.0001
2,3',4,4',5-PentaCB (118)	0.0001	<0.000005	0.00001
2',3,4,4',5-PentaCB (123)	0.0001	<0.000005	0.00001
2,3,3',4,4',5-HexaCB (156)	0.0005	<0.000005	0.0001
2,3,3',4,4',5'-HexaCB (157)	0.0005	<0.000005	0.0001
2,3',4,4',5,5'-HexaCB (167)	0.00001	<0.000005	0.00001
2,3,3',4,4',5,5'-HeptaCB (18[illegible]	0.0001	<0.000005	0.00001

Notes: CB = chlorinated biphenyls
Reference: van den Berg, et al. (1998). Toxic Equivalency Factors (TEFs) for PCBs PCDDs, PCDFs for Humans and Wildlife. Environmental Health Perspectives, 106:12, 775-791.

TABLE B-3
SELECTED SEDIMENT SCREENING GUIDELINES: PCBs

	Total PCBs	Aroclor 1254	Aroclor 1248	Aroclor 1016	Aroclor 1260
Hudson River Sediment Effect Concentrations ***(mg/kg, or ppm)***					
(MacDonald Env. Sci., 1999)					
(Estuarine, freshwater, and saltwater)					
Threshold Effect Concentration	0.04				
Mid-range Effect Concentration	0.4				
Extreme Effect Concentration	1.7				
NYSDEC (1998) (Freshwater) ***(mg/kg organic carbon)***					
Benthic Aquatic Life Acute Toxicity	2760.8				
Benthic Aquatic Life Chronic Toxicity	19.3				
Wildife Bioaccumulation	1.4				
NYSDEC (1998) (Saltwater) ***(mg/kg organic carbon)***					
Benthic Aquatic Life Acute Toxicity	13803.3				
Benthic Aquatic Life Chronic Toxicity	41.4				
Wildlife Bioaccumulation	1.4				
Ontario Ministry of the Environment Sediment Guidelines (Freshwater)					
(Persaud et al., 1993)					
No Effect Level ***(mg/kg)***	0.01				
Lowest Effect Level ***(mg/kg)***	0.07	0.06	0.03	0.007	0.005
Severe Effect Level ***(mg/kg organic carbon)***	530	34	150	53	24
Long et al. (1995) Sediment Guidelines ***(ug/kg)***					
(Marine and Estuarine)					
Effects-Range-Low	22.7				
Effects-Range-Median	180				
Ingersoll et al. (1996) Sediment Guidelines ***(ug/kg, or ppb)***					
(Freshwater)					
(Derived from 28-day Hyalella azteca *data)*					
Effects-Range-Low	50				
Effects-Range-Median	730				
Threshold Effect Level	32				
Probable Effect Level	240				
No Effect Concentration	190				
				Aroclor 1242	
Washington State Dep't of Ecology 1997 Sediment Guidelines					
(Freshwater) ***(ug/kg, or ppb)***[1]					
Apparent Effects Threshold (Microtox)	21	7.3			
Apparent Effects Threshold (*Hyalella azteca*)	820	350		100	
Probable Apparent Effects Threshold (Microtox)	21	7.3	21		
Probable Apparent Effects Threshold (*Hyalella azteca*)	450	240		100	
Lowest Apparent Effects Threshold	21	7.3	21		
(between Microtox and *H. azteca*)					
Florida Department of Environmental Protection ***(ug/kg, or ppb)***					
(MacDonald, D.D., et al., 1996) (Marine and Estuarine)					
Threshold Effect Level	21.6				
Probable Effect Level	189				
Jones et al. (1997) ***(ug/kg, or ppb)***					
EqP-derived; recommended TOC adjustment					
Secondary Chronic Value		810	1000		450000
Smith et al. (1996) ***(ug/kg, or ppb)***					
Threshold Effect Level	34.1				
Probable Effect Level	277				

Note: All values are dry weight unless noted.
Please note that for Washington state values, the Aroclor 1016 column becomes Aroclor 1242. This applies only to this one set of values.
[1] Some values also available in mg/kg organic carbon

TABLE B-4
TOXICITY ENDPOINTS FOR BENTHIC INVERTEBRATES
EFFECTIVE CONCENTRATIONS OF TOTAL PCBs, AROCLORS, AND DIOXIN TOXIC EQUIVALENTS (TEQs)

SPECIES	EXPOSURE MEDIA	PCB TYPE	EXPOSURE DURATION	EFFECT LEVEL	EFFECT CONC, WHOLE BODY CONC. (mg/kg wet wt)	EFFECT ENDPOINT	REFERENCE
Amphipod (*Gammarus pseudolimnaeus*)	Water	Aroclor 1248	2 months	LD_{50}	552	Mortality	Nebeker and Puglisi (1974)
Amphipod (*Hyalella azteca*)	Water	PCB 52	> or = 10 weeks	LD_{100}	180	Mortality	Borgmann et al. (1990)
Amphipod (*Hyalella azteca*)	Water	Aroclor 1242	> or = 10 weeks	LD_{100}	100	Mortality	Borgmann et al. (1990)
Amphipod (*Gammarus pseudolimnaeus*)	Water	Aroclor 1242	2 months	LD_{50}	316	Mortality	Nebeker and Puglisi (1974)
Cladoceran (*Daphnia magna*)	Model ecosystem	2,3,7,8-TCDD	33 days	EL (no effect)	1570	Mortality	Isensee and Jones (1975)
Amphipod (*Gammarus pseudolimnaeus*)	Water	Aroclor 1248	2 months	LOAEL	552	Reproduction reduced by at least 50%	Nebeker and Puglisi (1974)
Snail (*Physa* spp.)	Water	2,3,7,8-TCDD	33 days	EL (no effect)	502	Mortality	Isensee and Jones (1975) Isensee (1978)
Amphipod (*Gammarus pseudolimnaeus*)	Water	Aroclor 1242	2 months	EL (effect)	316	No reproduction	Nebeker and Puglisi (1974)
Oligochaete (*Lumbriculus variegatus*)	Algae (Food)	PCB 153	35 days	LOAEL	126	Mortality	Fisher et al. (1998)
Oligochaete (*Lumbriculus variegatus*)	Algae (Food)	PCB 153	35 days	LOAEL	126	Weight loss	Fisher et al. (1998)
Oligochaete (*Lumbriculus variegatus*)	Algae (Food)	PCB 15	35 days	LOAEL	119	Mortality	Fisher et al. (1998)
Oligochaete (*Lumbriculus variegatus*)	Algae (Food)	PCB 15	35 days	LOAEL	119	Weight loss	Fisher et al. (1998)
Oligochaete (*Lumbriculus variegatus*)	Algae (Food)	PCB 47	35 days	LOAEL	113	Mortality	Fisher et al. (1998)
Oligochaete (*Lumbriculus variegatus*)	Algae (Food)	PCB 47	35 days	LOAEL	113	Weight loss	Fisher et al. (1998)
Grass shrimp (*Palaemonetes pugio*)	Water	Aroclor 1254	7 days	LOAEL	65	Mortality (60%)	Nimmo et al. (1974)
Oligochaete (*Lumbriculus variegatus*)	Algae (Food)	PCB 1	35 days	LOAEL	64	Mortality	Fisher et al. (1998)
Oligochaete (*Lumbriculus variegatus*)	Algae (Food)	PCB 1	35 days	LOAEL	64	Weight loss	Fisher et al. (1998)
Grass shrimp (*Palaemonetes pugio*)	Water	Aroclor 1254	16 days	LOAEL	27	Mortality (45%)	Nimmo et al. (1974)
Amphipod (*Gammarus pseudolimnaeus*)	Water	Aroclor 1248	2 months	NOAEL	127	Reproduction	Nebeker and Puglisi (1974)
Amphipod (*Gammarus pseudolimnaeus*)	Water	Aroclor 1242	2 months	NOAEL	76	Reproduction	Nebeker and Puglisi (1974)
Oligochaete (*Lumbriculus variegatus*)	Algae (Food)	PCB 153	35 days	NOAEL	65	Mortality	Fisher et al. (1998)
Oligochaete (*Lumbriculus variegatus*)	Algae (Food)	PCB 153	35 days	NOAEL	65	Weight loss	Fisher et al. (1998)
Oligochaete (*Lumbriculus variegatus*)	Algae (Food)	PCB 15	35 days	NOAEL	63.1	Mortality	Fisher et al. (1998)

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TABLE B-5
TOXICITY ENDPOINTS FOR FISH - LABORATORY STUDIES
EFFECTIVE CONCENTRATIONS OF TOTAL PCBs AND AROCLORS

SPECIES	EXPOSURE MEDIA	PCB TYPE	EXPOSURE DURATION	EFFECT LEVEL	EFFECT CONCENTRATION WHOLE BODY CONCENTRATION mg/kg wet wt.	EFFECT ENDPOINT	REFERENCE
Laboratory studies							
Lake trout (*Salvelinus namaycush*)	Water	PCB-153	15 days	LD100	7.6	Fry mortality	Broyles and Noveck, 1979
Chinook salmon (Oncorhnchus tshawytscha)	Water	PCB-153	15 days	LD100	3.6	Fry mortality	Broyles and Noveck, 1979
Adult Fathead Minnow (*Pimephales promelas*)	Water	Aroclor 1254	9 months	LOAEL	999	Adult mortality	Nebeker et al., 1974
Adult Fathead Minnow (*Pimephales promelas*)	Water	Aroclor 1254	9 months	LOAEL	429	Spawning	Nebeker et al., 1974
Brook trout fry (*Salvelinus fontinalis*)	Water	Aroclor 1254	118 days	LOAEL	125	Fry mortality	Mauck et al., 1978
2	Water	Aroclor 1254	21 days	EL-effect	32.8 in muscle	Egg hatchabilty	Freeman and Idler, 1974
Brook trout fry (*Salvelinus fontinalis*)	Water	Aroclor 1254	21 days	EL-effect	77.9 in eggs	Egg hatchability	Freeman and Idler, 1974
Junvenile Spot (*Leiostomus xanthurus*)	Water	Aroclor 1254	20 days	LOAEL	46	Adult mortality	Hansen et al., 1971
Adult pinfish (*Lagodon rhomboides*)	Water	Aroclor 1016	42 days	LOAEL	42	Adult mortality	Hansen et al., 1974
Adult Minnow (*Phoxinus phoxinus*)	Diet	Clophen A50	40 days; studied for 300 days	LOAEL	15	Hatching time; fry survival	Bengtsson, B., 1980
Killifish (*Fundulus heteroclitus*)	Single intraperitoneal injection into adults	PCB mixture	Single injection, 40 d of observation	LOAEL	19 (nominal dose)	Adult female mortality	Black et al., 1998a
Sheepshead minnow (*Cyprinodon variegatus*)	Water	Aroclor 1254	28 days	LOAEL	9.3	Fry mortality	Hansen et al., 1974
Lake trout fry (*Salmo gairdneri*)	Water	Aroclor 1254	48 days	EL-effect	4.5	Fry mortality	Mac and Seelye, 1981
Killifish (*Fundulus heteroclitus*)	Single intraperitoneal injection into adults	PCB mixture	Single injection, 40 days of observation	LOAEL	3.8 (nominal dose)	Egg production and food consumption	Black et al., 1998a
Adult Fathead Minnow (*Pimephales promelas*)	Water	Aroclor 1242	9 months	NOAEL	436	Adult mortality	Nebeker et al., 1974
Adult Fathead Minnow (*Pimephales promelas*)	Water	Aroclor 1254	9 months	NOAEL	429	Egg hatchability	Nebeker et al., 1974
Adult pinfish (*Lagodon rhomboides*)	Water	Aroclor 1016	42 days	NOAEL	170	Adult mortality	Hansen et al., 1974
Adult Fathead Minnow (*Pimephales promelas*)	Water	Aroclor 1254	9 months	NOAEL	105	Spawning	Nebeker et al., 1974
Brook trout fry (*Salvelinus fontinalis*)	Water	Aroclor 1254	118 days	NOAEL	71	Fry mortality	Mauck et al., 1978
Juvenile Spot (*Leiostomus xanthurus*)	Water	Aroclor 1254	Lab Stu	NOAEL	27	Adult mortality	Hansen et al., 1971

TAE
TOXICITY ENDPOINTS FOR ...THIC INVERTEBRATES
EFFECTIVE CONCENTRATIONS OF TOTAL PCBs, AROCLORS, AND DIOXIN TOXIC EQUIVALENTS (TEQs)

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SPECIES	EXPOSURE MEDIA	PCB TYPE	EXPOSURE DURATION	EFFECT LEVEL	EFFECT CONC, WHOLE BODY CONC. (mg/kg wet wt)	EFFECT ENDPOINT	REFERENCE
Oligochaete (*Lumbriculus variegatus*)	Algae (Food)	PCB 15	35 days	NOAEL	63.1	Weight loss	Fisher et al. (1998)
Amphipod (*Hyalella azteca*)	Water	PCB 52	> or = 10 weeks	NOAEL	54	Mortality	Borgmann et al. (1990)
Oligochaete (*Lumbriculus variegatus*)	Algae (Food)	PCB 47	35 days	NOAEL	49.3	Mortality	Fisher et al. (1998)
Oligochaete (*Lumbriculus variegatus*)	Algae (Food)	PCB 47	35 days	NOAEL	49.3	Weight loss	Fisher et al. (1998)
Oligochaete (*Lumbriculus variegatus*)	Algae (Food)	PCB 1	35 days	NOAEL	33.2	Mortality	Fisher et al. (1998)
Oligochaete (*Lumbriculus variegatus*)	Algae (Food)	PCB 1	35 days	NOAEL	33.2	Weight loss	Fisher et al. (1998)
Amphipod (*Hyalella azteca*)	Water	Aroclor 1242	> or = 10 weeks	NOAEL	30	Mortality	Borgmann et al. (1990)
Grass shrimp (*Palaemonetes pugio*)	Water	Aroclor 1254	16 days	NOAEL	18	Mortality	Nimmo et al. (1974)
Grass shrimp (*Palaemonetes pugio*)	Water	Aroclor 1255	7 days	NOAEL	5.4	Mortality	Nimmo et al. (1974)

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TABLE B-6
TOXICITY ENDPOINTS FOR FISH - FIELD STUDIES
EFFECTIVE CONCENTRATIONS OF TOTAL PCBs AND AROCLORS

SPECIES	FIELD COMPONENT	CONTAMINANT TYPE	EFFECT LEVEL	EFFECT CONCENTRATION mg/kg wet wt (or as noted below)	EFFECT ENDPOINT	REFERENCE
Field studies						
Arctic charr (*Salvelinus alpinus*)	Adult fish and eggs collected from Lake Geneva	PCBs DDT	EL-effect	10 to 78 mg/kg lipid in eggs	Embryomortality	Monod, 1985
Winter flounder (*Pseudopleuronectes americanus*)	Adult and eggs collected from New Bedford Harbor	PCBs	EL-effect	39.6 mg/kg dry wt in eggs	Growth rate of larvae	Black et al., 1988b
Killifish (*Fundulus heteroclitus*)	Fish collected from New Bedford Harbor	PCBs	LOAEL	29.2 mg/kg dry wt in liver	Embryo and larval survival	Black et al., 1998b
Killifish (*Fundulus heteroclitus*)	Fish collected from New Bedford Harbor	PCBs	LOAEL	20.8 mg/kg dry wt in liver	Adult female mortality	Black et al., 1998b
English sole (*Parophrys vetulus*)	Fish collected from Puget Sound	PCBs, PAHs	EL-effect	Approx. 10 mg/kg in liver	Increased fecundity	Johnson et al., 1997
Striped bass (*Morone saxatilis*)	Eggs from hatcheries. Larvae fed naturally contaminated food.	PCBs, HCB, pesticides	EL-effect	0.1 to 10 in eggs	Larval mortality	Westin et al., 1985
Chinook salmon (*Oncorhynchus tshawytscha*)	Adult fish and eggs collected from Lake Michigan	PCBs, pesticides	EL-effect	2.8 to 9.9 A-1254 in eggs	Hathcing success	Giesy et al., 1986
Chinook salmon (*Oncorhynchus tshawytscha*)	Adult fish and eggs collected from Lake Michigan	PCBs	El-effect	2.75 to 5.75 in eggs	Hatching success	Ankley et al., 1981
Rainbow trout (*Salmo gairdneri*)	Adult fish and eggs hatchery	PCBs, DDT	EL-effect	2.7 in eggs	Embryomortality	Hogan and Braun, 1975
English sole (*Parophrys vetulus*)	Adults and eggs collected from Puget Sound	PCBs	LOAEL	2.56 in liver	Prodcution of normal larvie	Casillas et al., 1991
Lake trout (*Salvelinus namaycush*)	Adult fish and eggs collected from Great Lakes	PCBs	EL-effect	0.25 to 7.77 in eggs	Egg mortality and percent of normal fry hatching	Mac et al., 1993
Chinook salmon (*Oncorhynchus tshawytscha*)	Adult fish and eggs collected from Lake Michigan	PCBs, pesticides	EL-effect	0.322 to 2.6 A-1260 in eggs	Hathcing success	Giesy et al., 1986
Starry flounder (*Platichthys stellatus*)	Adult fish and eggs collected from area of San Francisco Bay	PCBs, HCB, Pthalates	EL-effect	about 50 t0 200 in eggs	Hathcing success	Spies and Rice, 1988
Redbreast sunfish (*Lepomis auritus*)	Adult fish collected from East Tennessee stream	PCBs, PAHs, metals, chlorine	EL-effect	0.95	Fecundity, clutch size, growth	Adams et al., 1989, 1990, 1992
Baltic herring (*Clupea harengus*)	Adult fish and eggs collected from Baltic Sea	PCBs, pesticides	EL-effect	> 0.120 in ovaries	Hathcing success	Hansen et el., 1985
Baltic flounder (*Platichthys flesus*)	Adult fish and eggs collected from Baltic Sea	PCBs, pesticides, metals	EL-effect	> 0.120 in ovaries	Hathcing success	von Westernhagen et al., 1981
Killifish (*Fundulus heteroclitus*)	Fish collected from New Bedford Harbor	PCBs	NOAEL	9.5 mg/kg dry wt in liver	Embryo and larval mortality	Black et al., 1998b
Striped bass (*Morone saxatilis*)	Eggs from Hudson River fish. Larvae fed naturally contaminated food	PCBs	EL-no effect	3.1 in post yolk sac larvae	Larval mortality	Westin et al., 1983
Winter flounder (*Pseudopleuronectes americanus*)	Adult and eggs collected from New Bedford Harbor	PCBs	EL-no effect	1.08 mg/kg dry wt in eggs	Growth rate of larvae	Black et al., 1988b
English sole (*Parophrys vetulus*)	Adults and eggs collected from Puget Sound	PCBs	NOAEL	0.09 in liver	Prodcution of normal larvie	Casillas et al., 1991

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TABLE B-5
TOXICITY ENDPOINTS FOR FISH - LABORATORY STUDIES
EFFECTIVE CONCENTRATIONS OF TOTAL PCBs AND AROCLORS

SPECIES	EXPOSURE MEDIA	PCB TYPE	EXPOSURE DURATION	EFFECT LEVEL	EFFECT CONCENTRATION WHOLE BODY CONCENTRATION mg/kg wet wt.	EFFECT ENDPOINT	REFERENCE
Killifish (*Fundulus heteroclitus*)	Single intraperitoneal injection into adults	PCB mixture	Single injection, 40 days of observation	NOAEL	3.8 (nominal dose)	Adult female mortality	Black et al., 1998a
Sheepshead minnow (*Cyprinodon variegatus*)	Water	Aroclor 1254	28 days	NOAEL	1.9	Fry mortality	Hansen et al., 1974
Adult Minnow (*Phoxinus phoxinus*)	Diet	Clophen A50	40 days; studied for 300 days	NOAEL	1.6	Hatching time; fry survival	Bengtsson, B., 1980
Killifish (*Fundulus heteroclitus*)	Single intraperitoneal injection into adults	PCB mixture	Single injection, 40 days of observation	NOAEL	0.76 (nominal dose)	Egg production and food consumption	Black et al., 1998a

TABLE B-6
TOXICITY ENDPOINTS FOR FISH - FIELD STUDIES
EFFECTIVE CONCENTRATIONS OF TOTAL PCBs AND AROCLORS

Redbreast sunfish (*Lepomis auritus*)	Fish from an East Tennessee stream	PCBs, PAHs, metals, chlorine	EL-no effect	0.5	Fecundity, clutch size, growth	Adams et al., 1989, 1990, 1992
Killifish (*Fundulus heteroclitus*)	Fish collected from New Bedford Harbor	PCBs	NOAEL	0.461 mg/kg dry wt in liver	Adult female mortality	Black et al., 1998b
Arctic charr (*Salvelinus alpinus*)	Adult fish and eggs collected from Lake Geneva	PCBs DDT	EL- no effect	0.1 to 0.31 in eggs	Embryomortality	Monod, 1985

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TABLE B-7
TOXICITY ENDPOINTS FOR FISH - LABORATORY STUDIES
EFFECTIVE CONCENTRATIONS OF DIOXIN TOXIC EQUIVALENTS (TEQs)

SPECIES	EXPOSURE MEDIA	EFFECT LEVEL	TISSUE	CONTAMINANT TYPE	EFFECT CONC. (ug/kg ww)	LIPID CONTENT OF EGG (g lipid/gww egg)	TEF	EFFECT CONC. DIOXIN EQUIVALENTS (ug TEQ/kg lipid)	EFFECT ENDPOINT	REFERENCE
Laboratory studies[a]										
Fathead minnow (*Pimephales promelas*)	Water	LD50	Embryo	2,3,7,8-TCDD	25.7	0.024	1	1071	Early life stage mortality	Olivieri and Cooper, 1997[b]
Zebrafish (*Danio danio*)	Water	LD50	Egg	2,3,7,8-TCDD	2.61	0.017	1	154	Early life stage mortality	Elonen et al., 1998
Zebrafish (Danio danio)	Water	LD50	Egg	2,3,7,8-TCDD	2.5	0.017	1	147	Early life stage mortality	Henry et al., 1997
White sucker (*Catastomus commersoni*)	Water	LD50	Egg	2,3,7,8-TCDD	1.89	0.025	1	76	Early life stage mortality	Elonen et al., 1998
Northern Pike (*Esox lucius*)	Water	LD50	Egg	2,3,7,8-TCDD	2.46	0.042	1	59	Early life stage mortality	Elonen et al., 1998
Medaka (*Oryzias latipes*)	Water	LD50	Egg	2,3,7,8-TCDD	1.11	0.029	1	38	Early life stage mortality	Elonen et al., 1998
Fathead minnow (*Pimephales promelas*)	Water	LD50	Egg	2,3,7,8-TCDD	0.539	0.024	1	22	Early life stage mortality	Elonen et al., 1998
Lake herring (*Coregonus artedii*)	Water	LD50	Egg	2,3,7,8-TCDD	0.902	0.066	1	14	Early life stage mortality	Elonen et al., 1998
Channel catfish (*Ictalurus punctatus*)	Water	LD50	Egg	2,3,7,8-TCDD	0.644	0.048	1	13	Early life stage mortality	Elonen et al., 1998
Rainbow Trout (*Salmo gairderi*) - Erwin strain	Water	LD50	Egg	2,3,7,8-TCDD	0.439	0.087	1	5.0	Early life stage mortality	Walker et al., 1992
Rainbow Trout (*Salmo gairderi*) - Erwin strain	Injection	LD50	Egg	2,3,7,8-TCDD	0.421	0.087	1	4.8	Early life stage mortality	Walker et al., 1992
Brook Trout (Salvenius fontinalis)	Water	LD100	Egg	2,3,7,8-TCDD	0.324	0.068	1	4.8	Early life stage mortality	Walker and Peterson, 1994
Rainbow Trout (*Salmo gairderi*) - Erwin strain	Egg injection	LD50	Egg	2,3,7,8-TCDD	0.409	0.087	1	4.7	Early life stage mortality	Zabel & Peterson, 1996
Rainbow Trout (*Salmo gairderi*)	Egg injection	LD50	Egg	2,3,7,8-TCDD	0.374	0.087	1	4.3	Early life stage mortality	Walker and Peterson, 1991
Rainbow Trout (*Salmo gairderi*)	Egg injection	LD50	Egg	PCB 126	74	0.087	0.005	4.3	Early life stage mortality	Walker and Peterson, 1991
Brook Trout (*Salvenius fontinalis*)	Water	LD50	Egg	2,3,7,8-TCDD	0.200	0.068	1	2.9	Early life stage mortality	Walker and Peterson, 1994
(*Salmo gairdneri*) Erwin strain	Egg injection	LD50	Egg	2,3,7,8-TCDD	0.242	0.087	1	2.8	Early life stage mortality	Zabel & Peterson, 1996
Lake trout (*Salvenius namaycush*)	Water	LD50	Egg	PCB 126	29	0.08	0.005	1.8	Early life stage mortality	Zabel et al., 1995
Fathead minnow (Pimephales promelas)	Water	LD50	Embryo	2,3,7,8-TCDD	0.026	0.024	1	1.1	Early life stage mortality	Olivieri and Cooper, 1997
Lake trout (*Salvenius namaycush*)	Water	LD50	Egg	2,3,7,8-TCDD	0.085	0.08	1	1.1	Early life stage mortality	Zabel et al., 1995
Lake trout (*Salvenius namaycush*)	Water	LD50	Egg	2,3,7,8-TCDD	0.065	0.08	1	0.8	Early life stage mortality	Walker et al., 1992
Lake trout (*Salvenius namaycush*)	Injection	LD50	Egg	2,3,7,8-TCDD	0.047	0.08	1	0.6	Early life stage mortality	Walker et al., 1992
Fathead minnow (Pimephales promelas)	Water	LD100	Larvae	2,3,7,8-TCDD	163	Not reported for larvae	1		Early life stage mortality	Olivieri and Cooper, 1997

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TABLE B-7
TOXICITY ENDPOINTS FOR FISH - LABORATORY STUDIES
EFFECTIVE CONCENTRATIONS OF DIOXIN TOXIC EQUIVALENTS (TEQs)

SPECIES	EXPOSURE MEDIA	EFFECT LEVEL	TISSUE	CONTAMINANT TYPE	EFFECT CONC. (ug/kg ww)	LIPID CONTENT OF EGG (g lipid/gww egg)	TEF	EFFECT CONC. DIOXIN EQUIVALENTS (ug TEQ/kg lipid)	EFFECT ENDPOINT	REFERENCE
Fathead minnow (Pimephales promelas)	Water	LD50	Larvae	2,3,7,8-TCDD	70.9	Not reported for larvae	1		Early life stage mortality	Olivieri and Cooper, 1997

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TABLE B-7
TOXICITY ENDPOINTS FOR FISH - LABORATORY STUDIES
EFFECTIVE CONCENTRATIONS OF DIOXIN TOXIC EQUIVALENTS (TEQs)

SPECIES	EXPOSURE MEDIA	EFFECT LEVEL	TISSUE	CONTAMINANT TYPE	EFFECT CONC. (ug/kg ww)	LIPID CONTENT OF EGG (g lipid/gww egg)	TEF	EFFECT CONC. DIOXIN EQUIVALENTS (ug TEQ/kg lipid)	EFFECT ENDPOINT	REFERENCE
Zebrafish (*Danio danio*)	Water	LOAEL	Egg	2,3,7,8-TCDD	2	0.017	1	118	Early life stage mortality	Elonen et al., 1998
Fathead minnow (*Pimephales promelas*)	Water	LOAEL	Embryo	2,3,7,8-TCDD	2.46	0.024	1	103	Early life stage mortality	Olivieri and Cooper, 1997
White sucker (*Catastomus commersoni*)	Water	LOAEL	Egg	2,3,7,8-TCDD	1.22	0.025	1	49	Early life stage mortality	Elonen et al., 1998
Northern Pike (*Esox lucius*)	Water	LOAEL	Egg	2,3,7,8-TCDD	1.8	0.042	1	43	Early life stage mortality	Elonen et al., 1998
Medaka (*Oryzias latipes*)	Water	LOAEL	Egg	2,3,7,8-TCDD	0.949	0.029	1	33	Early life stage mortality	Elonen et al., 1998
Fathead minnow (*Pimephales promelas*)	Water	LOAEL	Egg	2,3,7,8-TCDD	0.435	0.024	1	18	Early life stage mortality	Elonen et al., 1998
Channel catfish (*Ictalurus punctatus*)	Water	LOAEL	Egg	2,3,7,8-TCDD	0.855	0.048	1	18	Early life stage mortality	Elonen et al., 1998
Lake herring (*Coregonus artedii*)	Water	LOAEL	Egg	2,3,7,8-TCDD	0.27	0.066	1	4.1	Early life stage mortality	Elonen et al., 1998
Rainbow Trout (*Salmo gairderi*)	Injection	LOAEL	Egg	2,3,7,8-TCDD	0.291	0.087	1	3.3	Early life stage mortality	Walker et al., 1992
Rainbow Trout (*Salmo gairderi*)	Water	LOAEL	Egg	2,3,7,8-TCDD	0.279	0.087	1	3.2	Early life stage mortality	Walker et al., 1992
Brook Trout (*Salvenius fontinalis*)	Water	LOAEL	Egg	2,3,7,8-TCDD	0.185	0.068	1	2.7	Early life stage mortality	Walker and Peterson, 1994
Lake trout (*Salvelinus namaycush*)	Injection	LOAEL	Egg	2,3,7,8-TCDD	0.058	0.08	1	0.7	Early life stage mortality	Walker et al., 1992
Lake trout (*Salvelinus namaycush*)	Injection	LOAEL	Egg	2,3,7,8-TCDD	0.055	0.08	1	0.7		Walker et al., 1994
Lake trout (*Salvelinus namaycush*)	Water	LOAEL	Egg	2,3,7,8-TCDD	0.055	0.08	1	0.7	Early life stage mortality	Walker et al., 1992
Lake trout (*Salvelinus namaycush*)	Maternal transfer	LOAEL	Egg	2,3,7,8-TCDD	0.05	0.08	1	0.6		Walker et al., 1994
Lake trout (*Salvelinus namaycush*)	Water	LOAEL	Egg	2,3,7,8-TCDD	0.04	0.08	1	0.5		Walker et al., 1994
Fathead minnow (*Pimephales promelas*)	Water	LOAEL	Larvae	2,3,7,8-TCDD	20	Not reported for larvae	1		Early life stage mortality	Olivieri and Cooper, 1997
White sucker (*Catastomus commersoni*)	Water	NOAEL	Egg	2,3,7,8-TCDD	0.848	0.025	1	34	Early life stage mortality	Elonen et al., 1998
Northern Pike (*Esox lucius*)	Water	NOAEL	Egg	2,3,7,8-TCDD	1.19	0.042	1	28	Early life stage mortality	Elonen et al., 1998
Zebrafish (*Danio danio*)	Water	NOAEL	Egg	2,3,7,8-TCDD	0.424	0.017	1	25	Early life stage mortality	Elonen et al., 1998
Medaka (*Oryzias latipes*)	Water	NOAEL	Egg	2,3,7,8-TCDD	0.455	0.029	1	16	Early life stage mortality	Elonen et al., 1998
Fathead minnow (*Pimephales promelas*)	Water	NOAEL	Egg	2,3,7,8-TCDD	0.235	0.024	1	9.8	Early life stage mortality	Elonen et al., 1998
Channel catfish (*Ictalurus punctatus*)	Water	NOAEL	Egg	2,3,7,8-TCDD	0.385	0.048	1	8.0	Early life stage mortality	Elonen et al., 1998
Fathead minnow (*Pimephales promelas*)	Water	NOAEL	Embryo	2,3,7,8-TCDD	0.13	0.024	1	5.4	Early life stage mortality	Olivieri and Cooper, 1997

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TABLE B-7
TOXICITY ENDPOINTS FOR FISH - LABORATORY STUDIES
EFFECTIVE CONCENTRATIONS OF DIOXIN TOXIC EQUIVALENTS (TEQs)

SPECIES	EXPOSURE MEDIA	EFFECT LEVEL	TISSUE	CONTAMINANT TYPE	EFFECT CONC. (ug/kg ww)	LIPID CONTENT OF EGG (g lipid/gww egg)	TEF	EFFECT CONC. DIOXIN EQUIVALENTS (ug TEQ/kg lipid)	EFFECT ENDPOINT	REFERENCE
Lake herring (*Coregonus artedii*)	Water	NOAEL	Egg	2,3,7,8-TCDD	0.175	0.066	1	2.7	Early life stage mortality	Elonen et al., 1998
Rainbow Trout (*Salmo gairderi*)	Injection	NOAEL	Egg	2,3,7,8-TCDD	0.291	0.087	1	3.3	Early life stage mortality	Walker et al., 1992
Brook Trout (*Salvenius fontinalis*)	Water	NOAEL	Egg	2,3,7,8-TCDD	0.135	0.068	1	2.0	Early life stage mortality	Walker and Peterson, 1994

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TABLE B-7
TOXICITY ENDPOINTS FOR FISH - LABORATORY STUDIES
EFFECTIVE CONCENTRATIONS OF DIOXIN TOXIC EQUIVALENTS (TEQs)

SPECIES	EXPOSURE MEDIA	EFFECT LEVEL	TISSUE	CONTAMINANT TYPE	EFFECT CONC. (ug/kg ww)	LIPID CONTENT OF EGG (g lipid/gww egg)	TEF	EFFECT CONC. DIOXIN EQUIVALENTS (ug TEQ/kg lipid)	EFFECT ENDPOINT	REFERENCE
Lake trout (*Salvelinus namaycush*)	Injection	NOAEL	Egg	2,3,7,8-TCDD	0.044	0.08	1	0.55	Early life stage mortality	Walker et al., 1992
Lake trout (*Salvelinus namaycush*)	Injection	NOAEL	Egg	2,3,7,8-TCDD	0.044	0.08	1	0.55		Walker et all., 1994
Lake trout (*Salvelinus namaycush*)	Water	NOAEL	Egg	2,3,7,8-TCDD	0.034	0.08	1	0.43	Early life stage mortality	Walker et al., 1992
Lake trout (*Salvelinus namaycush*)	Water	NOAEL	Egg	2,3,7,8-TCDD	0.034	0.08	1	0.43		Walker et all., 1994
Lake trout (*Salvelinus namaycush*)	Maternal transfer	NOAEL	Egg	2,3,7,8-TCDD	0.023	0.08	1	0.29		Walker et all., 1994
Fathead minnow (*Pimephales promelas*)	Water	NOAEL	Larvae	2,3,7,8-TCDD	3.59	Not reported for larvae	1		Early life stage mortality	Olivieri and Cooper, 1997

Notes:

[a] No relevant field studies were found.

[b] Fathead minnow embryo is assumed to have same lipid content as reported for eggs (Elonen et al., 1998)

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TABL

TOXICITY ENDPOINTS FOR FISH - FIELD STUDIES

EFFECTIVE CONCENTRATIONS OF DIOXIN TOXIC EQUIVALENTS (TEQs)

SPECIES	EXPOSURE MEDIA	EFFECT LEVEL	TISSUE	CONTAMINANT TYPE	EFFECT CONC. (ug/kg ww, unless noted differently below)	LIPID CONTENT OF EGG (g lipid/gww egg)	EFFECT CONC. (ug/kg lipid)	TEF	EFFECT CONC. DIOXIN EQUIVALENTS (ug TEQ/kg lipid)	EFFECT ENDPOINT	REFERENCE
Rainbow Trout - Arlee strain (*Salmo gairdneri*)	Egg injection of extract from field-collected fish	LD50	Eggs	TEQs	0.514	0.087	5.9	1	5.9	Embryomortality	Wright and Tillitt, 1999
Rainbow Trout - Erwin strain (*Salmo gairdneri*)	Egg injection of extract from field-collected fish	LD50	Eggs	TEQs	0.206	0.087	2.4	1	2.4	Embryomortality	Wright and Tillitt, 1999
Rainbow Trout - Lake Superior (*Salmo gairdneri*)	Egg injection of extract from field-collected fish	LD50	Eggs	TEQs	1.43	0.087	16.4	1	16.4	Embryomortality	Wright and Tillitt, 1999
Killifish (*Fundulus heteroclitus*)	Fish collected from New Bedford Harbor	LOAEL	Liver	TEQs	1.56 ug/kg dry et	Not available	Not available	1	Not available	Embryo and larval survival	Black et al., 1998
Killifish (*Fundulus heteroclitus*)	Fish collected from New Bedford Harbor	LOAEL	Liver	TEQs	0.543 ug/kg dry wt	Not available	Not available	1	Not available	Adult female mortality	Black et al., 1998
Killifish (*Fundulus heteroclitus*)	Fish collected from New Bedford Harbor	NOAEL	Liver	TEQs	0.132 ug/kg dry wt	Not available	Not available	1	Not available	Embryo and larval survival	Black et al., 1998
Lake trout (*Salvelinus namaycush*)	Fish collected from Lake Ontario	EL-no effect	Eggs	TEQs	0.011	0.08	0.1	1	0.1	Early life stage mortality	Guiney et al., 1996
Killifish (*Fundulus heteroclitus*)	Fish collected from New Bedford Harbor	NOAEL	Liver	TEQs	0.00572 ug/kg dry wt	Not available	Not available	1	Not available	Adult female mortality	Black et al., 1998

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TABLE B-9
TOXICITY ENDPOINTS FOR AVIANS - LABORATORY STUDIES
EFFECTIVE DIETARY DOSES OF TOTAL PCBs AND AROCLORS

SPECIES	EXPOSURE MEDIA	EXPOSURE DURATION	EFFECT LEVEL	PCB TYPE	EFFECTIVE DOSE (mg/kg/day)	EFFECTIVE FOOD CONC. (mg/kg)	EFFECT ENDPOINT	REFERENCE
Laboratory studies								
Mallard Duck (*Anas platrhynchos*)		5 day	LD50	Aroclor 1254	853	8122	Mortality	Hill et al., 1975
Japanese Quail (*Coturnix coturnix*)		5 day	LD50	Aroclor 1254	759	6737	Mortality	Hill et al., 1975
Bobwhite Quail (*Colinus virginianus*)		5 day	LD50	Aroclor 1254	141	1516	Mortality	Hill et al., 1975
Brown-headed Cowbird (*Molothrus ater*)	Diet	7 days	EL-effect	Aroclor 1254	333	1500	Mortality	Stickel et al., 1984
Red-winged Blackbird (*Agelaius phoeniceus*)	Diet	6 days	EL-effect	Aroclor 1254	321	1500	Mortality	Stickel et al., 1984
Japanese Quail (*Coturnix coturnix*)	Oral by syringe	7 days	LOAEL	Aroclor 1260	100	888	Weight loss	Vos et al., 1971
Mallard Duck (*Anas platrhynchos*)	Diet	12 weeks	EL-effect	Aroclor1242	16	150	Decreased weight gain in hens, eggshell thinning	Haseltine and Prouty, 1980
Domestic Chicken (*Gallus domesticus*)	Drinking water	6 weeks	EL-effect	Aroclor 1254	3.5	50	Hatching success	Tumasonis et al., 1973
Ring-Necked Pheasant (*Phasianus colchicus*)	Diet, in gelatin capsules	Once per week for 17 weeks	LOAEL	Aroclor 1254	2.9	50	Egg production	Dahlgren et al., 1972
Ring-Necked Pheasant (*Phasianus colchicus*)	Diet	Not available	LOAEL	Aroclor 1254	2.9	50	Female fertility	Roberts et al., 1978
Domestic Chicken (*Gallus domesticus*)	Diet	9 weeks	LOAEL	Aroclor 1242	1.4	20	Egg production, hatching success, chick growth	Lillie et al., 1974
Domestic Chicken (*Gallus domesticus*)	Diet	9 weeks	LOAEL	Aroclor 1248	1.4	20	Egg production, hatching success, chick growth	Lillie et al., 1974
Domestic Chicken (*Gallus domesticus*)	Diet	9 weeks	LOAEL	Aroclor 1254	1.4	20	Egg production, hatching success, chick growth	Lillie et al., 1974
Domestic Chicken (*Gallus domesticus*)	Diet	9 weeks	LOAEL	Aroclor1242	1.4	20	Hatching success	Cecil et al., 1974
Domestic Chicken (*Gallus domesticus*)	Diet	9 weeks	LOAEL	Aroclor 1254	1.4	20	Hatching success	Cecil et al., 1974
Domestic Chicken (*Gallus domesticus*)	Diet	9 weeks	LOAEL	Aroclor1248	1.4	20	Hatching success	Cecil et al., 1974
Ringed Turtle Dove (*Streptopelia risoria*)	Diet	3 months	EL-effect	Aroclor 1254	1.1	10	Hatching success	Peakall et al, 1972
Ringed Turtle Dove (*Streptopelia risoria*)	Diet		LOAEL	Aroclor 1254	1.1	10	Hatching success	Peakall and Peakall, 1973
Domestic Chicken (*Gallus domesticus*)	Diet	6 weeks	LOAEL	Aroclor 1242	0.7	10	Hatching success	Britton and Huston, 1973
Domestic Chicken (*Gallus domesticus*)	Diet	8 weeks	LOAEL	Aroclor 1242	0.7	10	Hatching success	Lillie et al., 1975
Domestic Chicken (*Gallus domesticus*)	Diet	8 weeks	LOAEL	Aroclor 1248	0.7	10	Hatching success	Lillie et al., 1975
Domestic Chicken (*Gallus domesticus*)	Diet	8 weeks	LOAEL	Aroclor 1248	0.7	10	Hatching success	Scott, 1977
Domestic Chicken (*Gallus domesticus*)	Diet		LOAEL	Aroclor 1254	0.3	5	Fertility and egg production	Platonow and Reinhart, 1973

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TOXICITY ENDPOINTS FOR AVIANS - LABORATORY STUDIES
EFFECTIVE DIETARY DOSES OF TOTAL PCBs AND AROCLORS

SPECIES	EXPOSURE MEDIA	EXPOSURE DURATION	EFFECT LEVEL	PCB TYPE	EFFECTIVE DOSE (mg/kg/day)	EFFECTIVE FOOD CONC. (mg/kg)	EFFECT ENDPOINT	REFERENCE
Laboratory studies								
European Starling (*Sternus vulgaris*)	Diet	4 days	EL-effect	Aroclor 1254	Not available	1,500	Mortality	Stickel et al., 1984
Common Grackle (*Quiscalus quiscula*)	Diet	8 days	EL-effect	Aroclor 1254	Not available	1,500	Mortality	Stickel et al., 1984
Mallard Duck (*Anas platrhynchos*)	Diet	12 weeks	EL-no effect	Aroclor1242	16	150	Reproduction success, hatching success, survival and growth of chicks	Haseltine and Prouty, 1980
Japanese Quail (*Coturnix coturnix*)	Diet	14 weeks	EL-no effect	Aroclor 1254	5.6	50	Mortality and growth rates of adults	Chang and Stokstad, 1975
Mallard Duck (*Anas platyrhynchos*)	Diet	Approx. 1 month	EL-no effect	Aroclor 1254	2.6	25	Reproduction success	Custer and Heinz, 1980
Japanese Quail (*Coturnix coturnix*)	Diet	Not reported	NOAEL	Aroclor1248	2.3	20	Hatching success	Scott, 1977
Domestic Chicken (*Gallus domesticus*)	Diet	8 weeks	NOAEL	Aroclor 1016	1.4	20	Egg production	Lillie et al., 1975
Domestic Chicken (*Gallus domesticus*)	Diet	8 weeks	NOAEL	Aroclor 1254	1.4	20	Egg production	Lillie et al., 1975
Domestic Chicken (*Gallus domesticus*)	Diet	9 weeks	EL-no effect	Aroclor1221	1.4	20	Hatching success	Cecil et al., 1974
Domestic Chicken (*Gallus domesticus*)	Diet	9 weeks	EL-no effect	Aroclor1232	1.4	20	Hatching success	Cecil et al., 1974
Domestic Chicken (*Gallus domesticus*)	Diet	9 weeks	EL-no effect	Aroclor1268	1.4	20	Hatching success	Cecil et al., 1974
Domestic Chicken (*Gallus domesticus*)	Diet	9 weeks	EL-no effect	Aroclor 5442	1.4	20	Hatching success	Cecil et al., 1974
Ring-Necked Pheasant (*Phasianus colchicus*)	Diet, in gelatin capsules	Once per week for 17 weeks	NOAEL	Aroclor 1254	0.7	12.5	Egg production	Dahlgren et al., 1972
Screech Owl (*Otus asio*)	Diet	> 8 weeks	EL-no effect	Aroclor1248	0.4	3	Egg prdoduction, hatching success, fledging success	McLane and Hughes, 1980
Domestic Chicken (*Gallus domesticus*)	Diet	6 weeks	NOAEL	Aroclor1242	0.3	5	Hatching success	Britton and Huston, 1973
Domestic Chicken (*Gallus domesticus*)	Diet	8 weeks	NOAEL	Aroclor 1242	0.3	5	Hatching success	Lillie et al., 1975
Domestic Chicken (*Gallus domesticus*)	Diet	8 weeks	NOAEL	Aroclor1248	0.3	5	Hatching success	Lillie et al., 1975
Domestic Chicken (*Gallus domesticus*)	Diet	9 weeks	NOAEL	Aroclor 1242	0.1	2	Egg production, hatching success, chick growth	Lillie et al., 1974
Domestic Chicken (*Gallus domesticus*)	Diet	9 weeks	NOAEL	Aroclor 1248	0.1	2	Egg production, hatching success, chick growth	Lillie et al., 1974
Domestic Chicken (*Gallus domesticus*)	Diet	9 weeks	NOAEL	Aroclor 1254	0.1	2	Egg production, hatching success, chick growth	Lillie et al., 1974
Domestic Chicken (*Gallus domesticus*)	Diet	9 weeks	NOAEL	Aroclor1242	0.1	2	Hatching success	Cecil et al., 1974
Domestic Chicken (*Gallus domesticus*)	Diet	9 weeks	NOAEL	Aroclor1248	0.1	2	Hatching success	Cecil et al., 1974
Domestic Chicken (*Gallus domesticus*)	Diet	9 weeks	NOAEL	Aroclor 1254	0.1	2	Hatching success	Cecil et al., 1974

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TABLE B-9
TOXICITY ENDPOINTS FOR AVIANS - LABORATORY STUDIES
EFFECTIVE DIETARY DOSES OF TOTAL PCBs AND AROCLORS

SPECIES	EXPOSURE MEDIA	EXPOSURE DURATION	EFFECT LEVEL	PCB TYPE	EFFECTIVE DOSE (mg/kg/day)	EFFECTIVE FOOD CONC. (mg/kg)	EFFECT ENDPOINT	REFERENCE
Laboratory studies								
Domestic Chicken (*Gallus domesticus*)	Diet	8 weeks	NOAEL	Aroclor1248	0.1	1	Hatching success	Scott, 1977

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TAB... -10
TOXICITY ENDPOINTS FOR AVIANS - FIELD STUDIES
EFFECTIVE DIETARY DOSES OF TOTAL PCBs AND AROCLORS

SPECIES	FIELD COMPONENT	EFFECT LEVEL	CONTAMINANT TYPE	EFFECTIVE DOSE (mg/kg/day)	EFFECTIVE FOOD CONC. (mg/kg)	EFFECT ENDPOINT	REFERENCE
Field studies							
Tree Swallow (*Tachycineta bicolor*)	Populations in Fox River and Green Bay, Lake Michigan, studied	NOAEL	PCBs, DDE	0.55	up to 0.61	Clutch and egg success	Custer et al., 1998
Tree Swallow (*Tachycineta bicolor*)	Populations along Hudson River studied	NOAEL	PCBs	16.1	up to 17.9	Growth, mortality, reproduction	US EPA Phase 2 Database (1998)

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TABLE B-11
TOXICITY ENDPOINTS FOR AVIANS - LABORATORY STUDIES
EFFECTIVE DIETARY DOSES OF DIOXIN TOXIC EQUIVALENTS (TEQs)

SPECIES	EXPOSURE MEDIA	EXPOSURE DURATION	EFFECT LEVEL	CONTAMINANT TYPE	EFFECTIVE DOSE DIOXIN EQUIVALENTS (ug/kg/day)	EFFECT ENDPOINT	REFERENCE
Laboratory studies[a]							
Ringed turtle dove (*Streptopelia risoria*)	Oral	Singe dose	LD_{50}	2,3,7,8-TCDD	> 810	Mortality	Hudson et al., 1984
Mallard (*Anas platrhyncos*)	Oral	Single dose		2,3,7,8-TCDD	> 108	Mortality	Hudson et al., 1984
Chicken (*Gallus domesticus*)	Oral	21 days	LD_{100}	2,3,7,8-TCDD	25 - 50	Mortality	Greig et al., 1973
Ring-necked pheasant (*Phasianus colchicus*)	Intraperitoneal	Single dose	LD_{75}	2,3,7,8-TCDD	25	Mortality	Nosek et al., 1992
Northern bobwhite quail (*Colinus virginianus*)	Oral	Single dose	LD_{50}	2,3,7,8-TCDD	15	Mortality	Hudson et al., 1984
Chicken (*Gallus domesticus*)	Oral	21 days	LOAEL	2,3,7,8-TCDD	1.0	Mortality	Schwetz et al., 1973
Ring-necked pheasant (*Phasianus colchicus*)	Intraperitoneal	10 weeks	LOAEL	2,3,7,8-TCDD	0.14	Fertility, embryo mortality	Nosek et al., 1992
Chicken (*Gallus domesticus*)	Oral	21 days	NOAEL	2,3,7,8-TCDD	0.1	Mortality	Schwetz et al., 1973
Ring-necked pheasant (*Phasianus colchicus*)	Intraperitoneal	10 weeks	NOAEL	2,3,7,8-TCDD	0.014	Fertility, embryo mortality	Nosek et al., 1992

Notes:

[a] No relevant field studies were found.

Note units of ug/kg/day.

TABLE B-12
TOXICITY ENDPOINTS FOR AVIANS - FIELD STUDIES
EFFECTIVE DIETARY DOSES OF DIOXIN TOXIC EQUIVALENTS (TEQs)

SPECIES	FIELD COMPONENT	EFFECT LEVEL	CONTAMINANT TYPE	EFFECTIVE DOSE DIOXIN EQUIVALENTS (ug/kg/day)	EFFECTIVE FOOD CONC. (ug/kg)	EFFECT ENDPOINT	REFERENCE
Field studies							
Tree Swallow (*Tachycineta bicolor*)	Populations along Hudson River studied	EL-no effect	TEQs	4.9	up to 5.41	Growth, mortality, reproduction	US EPA Phase 2 Database, 1998
Tree Swallow (*Tachycineta bicolor*)	Populations in Fox River and Green Bay, Lake Michigan,	EL-no effect	TEQs, DDE	0.08	up to 0.091	Clutch and egg success	Custer et al., 1998

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TABLE B-13
TOXICITY ENDPOINTS FOR AVIAN EGGS - LABORATORY STUDIES
EFFECTIVE CONCENTRATIONS OF TOTAL PCBs AND AROCLORS

SPECIES	EXPOSURE MEDIA	EXPOSURE DURATION	EFFECT LEVEL	PCB TYPE	EFFECTIVE EGG CONC. (mg/kg egg)	EFFECT ENDPOINT	REFERENCE
Laboratory studies							
Chicken (*Gallus domesticus*)	Drinking water	6 weeks	EL-effect	Aroclor 1254	> 10-15 ppm in yolk	Deformities	Tumasonis et al., 1973
Chicken (*Gallus domesticus*)	Egg injection		LOAEL	Aroclor 1260	10	Growth rate of chicks	Carlson and Duby, 1973
Chicken (*Gallus domesticus*)	Egg injection		LOAEL	Aroclor 1254	6.7	Growth and mortality of embryos	Gould et al., 1997
Chicken (*Gallus domesticus*)	Egg injection		LOAEL	Aroclor 1242	5	Hatching success	Carlson and Duby, 1973
Chicken (*Gallus domesticus*)	Egg injection		LOAEL	Aroclor 1254	5	Hatching success	Carlson and Duby, 1973
Chicken (*Gallus domesticus*)	Egg injection		LOAEL	Aroclor 1242	5	Growth rate of chicks	Carlson and Duby, 1973
Chicken (*Gallus domesticus*)			LOAEL		5	Egg production and hatching success	Platanow and Reinhart, 1973
Chicken (*Gallus domesticus*)	Diet	6 weeks	LOAEL	Aroclor 1242	3.7	Hatching success	Britton and Huston, 1973
Chicken (*Gallus domesticus*)	Diet	4 weeks	LOAEL	Aroclor 1248	2.21	Hatching success	Scott, 1977
Chicken (*Gallus domesticus*)	Egg injection		NOAEL	Aroclor 1260	10	Hatching success	Carlson and Duby, 1973
Screech owl (*Otus asio*)	Diet of hens	> 8 weeks	NOAEL	Aroclor 1248	7.1	Egg production, hatching success, and fledging success	McLane and Hughes, 1980
Chicken (*Gallus domesticus*)	Egg injection		NOAEL	Aroclor 1260	5	Growth rate of chicks	Carlson and Duby, 1973
Chicken (*Gallus domesticus*)	Egg injection		NOAEL	Aroclor 1242	2.5	Hatching success	Carlson and Duby, 1973
Chicken (*Gallus domesticus*)	Egg injection		NOAEL	Aroclor 1254	2.5	Hatching success	Carlson and Duby, 1973
Chicken (*Gallus domesticus*)	Egg injection		NOAEL	Aroclor 1242	2.5	Growth rate of chicks	Carlson and Duby, 1973
Chicken (*Gallus domesticus*)	Diet	6 weeks	NOAEL	Aroclor 1242	1.7	Hatching success	Britton and Huston, 1973
Chicken (*Gallus domesticus*)	Egg injection		NOAEL	Aroclor 1254	0.67	Growth and mortality of embryos	Gould et al., 1997
Chicken (*Gallus domesticus*)	Diet	4 weeks	NOAEL	Aroclor 1248	0.33	Hatching success	Scott, 1977

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TABLE B-14
TOXICITY ENDPOINTS FOR AVIAN EGGS - FIELD STUDIES
EFFECTIVE CONCENTRATIONS OF TOTAL PCBs AND AROCLORS

SPECIES	EFFECT LEVEL	CONTAMINANT TYPE	EFFECTIVE EGG CONC. (mg/kg egg)	EFFECT ENDPOINT	REFERENCE
Field studies					
Bald eagle (*Haliaeetus leucocephalus*)	EL--Effect level	PCBs, Pesticides	20-54	Reproductive success	Clark et al., 1988
Double-crested cormorant (*Phalacrocorax auritus*)	EL-Effect level	PCBs, Pesticides, Hg	23.8	Hatching success and fledging success	Weseloh et al., 1983
Caspian tern (*Hydropogne caspia*)	EL-Effect level	PCBs, Pesticides	4.2 - 18	Increased rate of embryo deformities	Yamashita et al., 1993
Forster's tern (*Sterna forsteri*)	LOAEL	PCBs, Pesticides, Dioxins, Furans	22.2	Hatching success	Kubiak et al., 1989
Common tern (*Sterna hirundo*)	LOAEL	PCBs, Pesticides, Hg	7	Hatching success	Becker et al., 1993
Common tern (*Sterna hirundo*)	LOAEL	PCBs, Pesticides, Hg	9.8	Hatching success	Hoffman et al., 1993
Bald eagle (*Haliaeetus leucocephalus*)	LOAEL	PCBs, Pesticides, Hg	3 - 5.6	10 % reduction in reproductive success	Wiemeyer et al., 1984, 1993
Bald eagle (*Haliaeetus leucocephalus*)	EL- No Effect	PCBs, TEQs, Pesticides	33.2 - 64 in yolk sac	Hatching success	Elliott et al., 1996
Tree swallow (*Tachycineta bicolor*)	NOAEL	PCBs	26.7	Reproductive output	Secord and McCarty, 1997, McCarty and Secord, 1999, U.S. EPA Phase 2 Database Release 4.1b, 1998
Common tern (*Sterna hirundo*)	NOAEL	PCBs, Pesticides, Hg	6.7	Hatching success	Hoffman et al., 1993
Common tern (*Sterna hirundo*)	NOAEL	PCBs, Pesticides, Hg	5.2	Hatching success	Becker et al., 1993
Forster's tern (Sterna forsteri)	NOAEL	PCBs, Pesticides, Dioxins, Furans	4.5	Hatching success	Kubiak et al., 1989
Tree swallow (*Tachycineta bicolor*)	NOAEL	PCBs, DDE	3.24 in eggs and pippers	Clutch success, egg success	Custer et al., 1998
Bald eagle (*Haliaeetus leucocephalus*)	NOAEL	PCBs, Pesticides, Hg	< 3	Reproductive success	Wiemeyer et al., 1984, 1993

TABLE B-15
TOXICITY ENDPOINTS FOR AVIAN EGGS - LABORATORY STUDIES
EFFECTIVE CONCENTRATIONS OF DIOXIN TOXIC EQUIVALENTS (TEQs)

SPECIES	EXPOSURE MEDIA	EXPOSURE DURATION	EFFECT LEVEL	CONTAMINANT TYPE	EFFECTIVE EGG CONC. (ug/kg egg)	TEF	EFFECTIVE EGG CONC. DIOXIN EQUIVALENTS (ug TEQ/kg egg)	EFFECT ENDPOINT	REFERENCE
Laboratory studies									
American kestrel (*Falco sparverius*)	Egg injection	18 days	LD50	PCB 77	316	0.05	16	Embryo mortality	Hoffman et al., 1998
Double-crested cormorant (*Phalacrocorax auritus*)	Egg injection	21 days	LD50	PCB 126	158	0.1	16	Embryo mortality	Powell et al., 1997
Common tern (*Sterna hirundo*)	Egg injection	18 days	LD50	PCB 126	104	0.1	10	Embryo mortality	Hoffman et al., 1998
American kestrel (*Falco sparverius*)	Egg injection	20 days	LD50	PCB 126	65	0.1	7	Embryo mortality	Hoffman et al., 1998
Ring-necked pheasant (*Phasianus colchicus*)	Egg injection	28 days	LD50	2,3,7,8-TCDD	1.35	1	1	Embryo mortality	Nosek et al., 1993
Chicken (*Gallus domesticus*)	Egg injection	18 days	LD50	PCB 105	5592	0.0001	1	Embryo mortality	Powell et al., 1996b
Chicken (*Gallus domesticus*)	Egg injection	18 days	LD50	PCB 77	8.8	0.05	0.4	Embryo mortality	Powell et al., 1996b
Chicken (*Gallus domesticus*)	Egg injection	24 days	LD50	PCB 126	2.3	0.1	0.2	Embryo mortality	Powell et al., 1996a
Chicken (*Gallus domesticus*)	Egg injection	24 days	LD50	2,3,7,8-TCDD	0.15	1	0.2	Embryo mortality	Powell et al., 1996a
Chicken (*Gallus gallus*)	Egg injection	20 days	LD50	PCB 77	2.6	0.05	0.1	Embryo mortality	Hoffman et al., 1998
Chicken (*Gallus gallus*)	Egg injection	18 days	LD50	PCB 126	0.4	0.1	0.04	Embryo mortality	Hoffman et al., 1998
Chicken (*Gallus domesticus*)	Egg injection	18 days	LD50	PCB 126	0.6	0.1	0.1	Embryo mortality	Powell et al., 1996b
Double-crested cormorant (*Phalacrocorax auritus*)	Egg injection	21 days	LOAEL	PCB 126	800	0.1	80	Embryo mortality	Powell et al., 1997
American kestrel (*Falco sparverius*)	Egg injection	20 days	LOAEL	PCB 126	233	0.1	23	Embryo mortality	Hoffman et al., 1998
American kestrel (*Falco sparverius*)	Egg injection	20 days	LOAEL	PCB 77	100	0.05	5	Embryo mortality	Hoffman et al., 1998
Common tern (*Sterna hirundo*)	Egg injection	18 days	LOAEL	PCB 126	44	0.1	4	Embryo mortality	Hoffman et al., 1998
Double-crested cormorant (*Phalacrocorax auritus*)	Egg injection	21 days	LOAEL	2,3,7,8-TCDD	4	1	4	Embryo mortality	Powell et al., 1997
Ring-necked pheasant (*Phasianus colchicus*)	Egg injection	21 days	LOAEL	2,3,7,8-TCDD	1	1	1.0	Embryo mortality	Nosek et al., 1993
Chicken (*Gallus domesticus*)	Egg injection	18 days	LOAEL	PCB 105	8100	0.0001	1	Embryo mortality	Powell et al., 1996b
Chicken (*Gallus domesticus*)	Egg injection	18 days	LOAEL	PCB 77	9	0.05	0.5	Embryo mortality	Powell et al., 1996b
Chicken (*Gallus gallus*)	Egg injection	18 days	LOAEL	PCB 77	6	0.05	0.3	Embryo mortality	Hoffman et al., 1998
Chicken (*Gallus domesticus*)	Egg injection	24 days	LOAEL	2,3,7,8-TCDD	0.16	1	0.2	Embryo mortality	Powell et al., 1996a
Pidgeon (*Columba livia*)	Egg injection	Embryonic Day 3 through hatch	EL-Effect	2,3,7,8-TCDD	1	1	1.0	Hatchability	Janz and Bellward, 1996
Chicken (*Gallus domesticus*)	Egg injection	18 days	LOAEL	PCB 126	0.9	0.1	0.09	Embryo mortality	Powell et al., 1996b
Chicken (*Gallus gallus*)	Egg injection	18 days	LOAEL	PCB 126	0.5	0.1	0.05	Embryo mortality	Hoffman et al., 1998
Chicken (*Gallus domesticus*)	Egg injection	24 days	LOAEL	PCB 126	0.2	0.1	0.02	Embryo mortality	Powell et al., 1996a
Double-crested cormorant (*Phalacrocorax auritus*)	Egg injection	21 days	NOAEL	PCB 126	400	0.1	40	Embryo mortality	Powell et al., 1997

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TAB... 15

TOXICITY ENDPOINTS FOR AVIAN EGGS - LABORATORY STUDIES

EFFECTIVE CONCENTRATIONS OF DIOXIN TOXIC EQUIVALENTS (TEQs)

SPECIES	EXPOSURE MEDIA	EXPOSURE DURATION	EFFECT LEVEL	CONTAMINANT TYPE	EFFECTIVE EGG CONC. (ug/kg egg)	TEF	EFFECTIVE EGG CONC. DIOXIN EQUIVALENTS (ug TEQ/kg egg)	EFFECT ENDPOINT	REFERENCE
Great Blue Heron (*Ardea herodias*)	Egg injection	Embryonic Day 9 through hatch	EL-No effect	2,3,7,8-TCDD	2	1	2	Hatchability	Janz and Bellward, 1996
American kestrel (*Falco sparverius*)	Egg injection	20 days	NOAEL	PCB 126	23	0.1	2	Embryo mortality	Hoffman et al., 1998
Double-crested cormorant (*Phalacrocorax auritus*)	Egg injection	21 days	NOAEL	2,3,7,8-TCDD	1	1	1	Embryo mortality	Powell et al., 1997

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TABLE B-15
TOXICITY ENDPOINTS FOR AVIAN EGGS - LABORATORY STUDIES
EFFECTIVE CONCENTRATIONS OF DIOXIN TOXIC EQUIVALENTS (TEQs)

SPECIES	EXPOSURE MEDIA	EXPOSURE DURATION	EFFECT LEVEL	CONTAMINANT TYPE	EFFECTIVE EGG CONC. (ug/kg egg)	TEF	EFFECTIVE EGG CONC. DIOXIN EQUIVALENTS (ug TEQ/kg egg)	EFFECT ENDPOINT	REFERENCE
Chicken (*Gallus domesticus*)	Egg injection	18 days	NOAEL	PCB 105	2700	0.0001	0.3	Embryo mortality	Powell et al., 1996b
Chicken (*Gallus domesticus*)	Egg injection	18 days	NOAEL	PCB 77	3	0.05	0.2	Embryo mortality	Powell et al., 1996b
Ring-necked pheasant (*Phasianus colchicus*)	Egg injection	28 days	NOAEL	2,3,7,8-TCDD	0.1	1	0.1	Embryo mortality	Nosek et al., 1993
Chicken (*Gallus domesticus*)	Egg injection	24 days	NOAEL	2,3,7,8-TCDD	0.08	1	0.1	Embryo mortality	Powell et al., 1996a
Chicken (*Gallus gallus*)	Egg injection	18 days	NOAEL	PCB 77	1.2	0.05	0.1	Embryo mortality	Hoffman et al., 1998
Chicken (*Gallus gallus*)	Egg injection	Embryonic Day 4 through hatch	EL-No effect	2,3,7,8-TCDD	0.1	1	0.1	Hatchability	Janz and Bellward, 1996
Chicken (*Gallus gallus*)	Egg injection	18 days	NOAEL	PCB 126	0.3	0.1	0.03	Embryo mortality	Hoffman et al., 1998
Chicken (*Gallus domesticus*)	Egg injection	18 days	NOAEL	PCB 126	0.3	0.1	0.03	Embryo mortality	Powell et al., 1996b
Chicken (*Gallus domesticus*)	Egg injection	24 days	NOAEL	PCB 126	0.1	0.1	0.01	Embryo mortality	Powell et al., 1996a

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TABLE B-16
TOXICITY ENDPOINTS FOR AVIAN EGGS - FIELD STUDIES
EFFECTIVE CONCENTRATIONS OF DIOXIN TOXIC EQUIVALENTS (TEQs)

SPECIES	EFFECT LEVEL	CONTAMINANT TYPE	EFFECTIVE EGG CONC. DIOXIN EQUIVALENTS (ug TEQ/kg egg)	EFFECT ENDPOINT	REFERENCE
Field studies					
Osprey (*Pandion haliaetus*)	EL-Effect level	TCDD	29 - 162	Growth rate of chicks	Woodford, et al., 1998
Bald eagle (*Haliaeetus leucocephalus*)	EL-Effect level	TEQs, DDE	0.51-1.2	Reproductive success	Clark et al., 1998
Great blue heron (*Ardea herodias*)	LOAEL	TEQs	0.5	Growth rate	Sanderson et al., 1994
Great blue heron (*Ardea herodias*)	EL-Effect level	TEQs, pesticides	0.5	Growth rate	Hart et al., 1991
Cormorant (*Phalacrocorax auritus*)	EL-effect level	TEQ	0.035 - 0.344	Egg mortality	Tillitt et al., 1992
Great blue heron (*Ardea herodias*)	EL-Effect level	TEQs, pesticides	0.23	Reproductive success	Elliott et al., 1989
Forster's tern (*Sterna forsteri*)	EL-Effect	TEQs, pesticides	2.20	Hatching success, growth rate of chicks	Kubiak et al., 1989
Forster's tern (*Sterna forsteri*)	EL-Effect level	TEQ	0.21	Hatching success	Tillitt et al., 1993
Wood duck (*Aix sponsa*)	LOAEL	TEQs, pesticides	0.02	Nest success, hatching success, duckling production	White and Seginak, 1994; White and Hoffman, 1995
Tree swallow (*Tachycineta bicolor*)	NOAEL	TEQs	13	Reproductive success	US EPA Phase 2 Database (1998)
Tree swallow (*Tachycineta bicolor*)	EL-No effect	TEQs	0.589 in pippers	Reproductive success	Custer et al., 1998
Great blue heron (*Ardea herodias*)	NOAEL	TEQs	0.3	Reduced body weight	Sanderson et al., 1994
Great blue heron (*Ardea herodias*)	NOAEL	TEQs	0.24	Growth rate	Hart et al., 1991
Forster's tern (*Sterna forsteri*)	EL-no effect	TEQs, pesticides	0.2	Hatchabilty, growth rate of chicks	Kubiak et al., 1989
Great blue heron (*Ardea herodias*)	EL-No effect	TEQs, pesticides	0.079	Reproductive success	Elliott et al., 1989
Osprey (*Pandion haliaetus*)	EL-no effect	TCDD, TEQs	ND - 23.8	Growth rate of chicks	Woodford et al., 1998
Osprey (*Pandion haliaetus*)	EL-no effect	TEQs	0.136	Embryo survival	Woodford et al., 1998
Foster's tern (*Sterna forsteri*)	EL-no effect	TEQs	0.023	Hatching success	Tillitt et al., 1993
Wood duck (*Aix sponsa*)	NOAEL	TEQs, pesticides	0.005	Nest success, hatching success, duckling production	White and Seginak, 1994; White and Hoffman, 1995

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TABLE B-17
TOXICITY ENDPOINTS FOR OTHER MAMMALS - LABORATORY STUDIES
EFFECTIVE DIETARY DOSES OF TOTAL PCBs AND AROCLORS

SPECIES	EXPOSURE MEDIA	EXPOSURE DURATION	EFFECT LEVEL	PCB TYPE	EFFECTIVE DOSE (mg/kg/day)	FOOD INGESTION RATE (kg/kg/day)	EFFECTIVE FOOD CONC. (mg/kg)	EFFECT ENDPOINT	REFERENCE
Laboratory studies[a]									
Osborne-Mendel Rat	Oral-gavage	2.5 wk, 2 d per week	LD_{50}	Aroclor 1254	1530	0.099		Mortality	Garthoff et al., 1981 (ATSDR)
Osborne-Mendel Rat	Oral-gavage	2.5 wk, 2 d per week	LD_{50}	Aroclor 1254	1530	0.099		Mortality	Garthoff et al., 1981 (ATSDR)
Wistar Rat	Diet	From mating to weaning of pups	LD_{50}	Aroclor 1254	22	0	269	2 day postnatal mortality of offspring	Overmann et al., 1987
Juvenile Male Rat	Single intraperitoneal injection	Observed after 14 days	LOAEL	Aroclor 1248	2000			Growth rate of juveniles	Harris et al., 1993
Juvenile Male Rat	Single intraperitoneal injection	Observed after 14 days	LOAEL	Aroclor 1232	2000			Growth rate of juveniles	Harris et al., 1993
Sherman Rat	Diet	8 months	LOAEL	Aroclor 1260	72.4	0.08		Mortality	Kimbrough et al., 1972 (ATSDR)
Raccoon (*Procyon lotor*)	Diet	8 days	EL-effect	Arochor 1254	50			Decreased weight gain	Montz et al., 1982
Osborne-Mendel Rat	Diet	During pregnancy and lactation	LOAEL	Not reported	49.471	0.080	500	Reduced litter size	Collins & Capen, 1980
Balb/c Mouse	Oral	6 months	LOAEL	Aroclor 1254	48.75	0.18		Mortality	Koller et al., 1977 (ATSDR)
Adult Female Rat	Oral	Day 1,3,5,7 and 9 of lactation	LOAEL	Aroclor 1254	32	0.08		Reduced growth rate of offspring	Sager & Girard, 1994
Wistar Rat	Oral-gavage	1 month	LOAEL	Aroclor 1254	30	0.08		Decreased litter size, survival of weanlings	Brezner et al., 1984 (ATSDR)
White-footed Mouse (*Peromyscus leucopus*)	Diet	12 weeks	EL-effect	Aroclor 1254	17		10	Reduced growth rate reproduction in second generation	Linzey, 1988 (Golub)
Wistar Rat	Diet	42 days	LOAEL	Aroclor 1254	13.5	0.08		Neonatal death	Overmann, 1987 (ATSDR)
Mouse	Diet	108 days	LOAEL	Aroclor 1254	12.5	0.18		Decreased conception	Welsch, 1975 (ATSDR)
Rabbit	Oral-gavage	28 days	LOAEL	Aroclor 1254	12.5	0.034		Fetal death	Villeneuve et al., 1971 (ATSDR)
Pig	Diet	91 days	LOAEL	Aroclor 1242	9.2			Decreased weight gain	Hansen et al., 1976 (ATSDR)
New Zealand White Rabbit	Diet	> 4 weeks	LOAEL	Aroclor 1248	8.9	0.0	250	Reduced growth rate in offspring	Thomas and Hinsdill, 1980 (Golub)
Osborne-Mendel Rat	Diet	During pregnancy and lactation	LOAEL	Not reported	4.947	0.080	50	Reduced growth rate of offspring	Collins & Capen, 1980
Rhesus Monkey (*Macaca mulatta*)	Diet	2 months	LOAEL	Aroclor 1248	4.3	0.2		Decreased conception	Allen et al., 1974a (ATSDR)
Rhesus Monkey (*Macaca mulatta*)	Diet	2 months	LOAEL	Aroclor 1248	4.3	0.2		Abortion	Allen et al., 1974a (ATSDR)
Fischer Rat	Diet	105 weeks	LOAEL	Aroclor 1254	2.5	0.08		Decreased survival	NCI, 1978 (ATSDR)
Guinea Pig	Oral-gavage	Gestational day 18-60	LOAEL	Clophen A50	2.5			Fetal death	Lundkvist, 1990 (ATSDR)
Sherman Rat	Diet	Multigenerational	LOAEL	Aroclor 1254	1.5	0.08	20	Decreased litter size	Linder et al., 1974
Wistar Rat	Diet	52 weeks	LOAEL	Aroclor 1254	1	0.08		Decreased growth rate	Phillips et al., 1972 (ATSDR)
Oldfield Mouse (*Peromyscus polionotus*)	Diet	12 months	EL-effect	Aroclor 1254	0.68	0.01	5	Decreased offspring born per mated pair, birth weight, % survival of offspring to weaning	McCoy et al., 1995
Rhesus Monkey (*Macaca mulatta*)	Diet	38 weeks	LOAEL	Aroclor 1254	0.2	0.2		No conception, abortion	Arnold et al., 1990 (ATSDR)

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TABLE B-18
TOXICITY ENDPOINTS FOR OTHER MAMMALS - LABORATORY STUDIES
EFFECTIVE DIETARY DOSES OF DIOXIN TOXIC EQUIVALENTS (TEQs)

SPECIES	EXPOSURE MEDIA	EXPOSURE DURATION	EFFECT LEVEL	CONTAMINANT TYPE	EFFECTIVE DOSE DIOXIN EQUIVALENTS (ug TEQ/kg/day)*	EFFECT ENDPOINT	REFERENCE
Laboratory studies							
Hamster	Oral	Single dose	LD_{50}	2,3,7,8-TCDD	1,160 - 5,050	Mortality	Kociba and Schwetz, 1982
Mouse	Oral	Single dose	LD_{50}	2,3,7,8-TCDD	114 -284	Mortality	Kociba and Schwetz, 1982
Dog	Oral	Single dose	LD_{50}	2,3,7,8-TCDD	about 100 - 200	Mortality	Kociba and Schwetz, 1982
Rabbit	Oral	Single dose	LD_{50}	2,3,7,8-TCDD	115	Mortality	Schwetz et al., 1973
Rhesus monkey (*Macaca mulatta*)	Oral	Single dose	LD_{50}	2,3,7,8-TCDD	approx. 70	Mortality	Kociba and Schwetz, 1982
Rat	Oral	Single dose	LD_{50}	2,3,7,8-TCDD	22 - 45	Mortality	Schwetz et al., 1973
Guinea pig	Oral	Single dose	LD_{52}	2,3,7,8-TCDD	0.6 - 2.1	Mortality	Schwetz et al., 1973
Rat		Gestation days 6 to 15	LOAEL	2,3,7,8-TCDD	0.25	Litter size, pup weight	Khera and Ruddick, 1973
Rat		2 years	LOAEL	2,3,7,8-TCDD	0.1	Female mortality	Kociba et al., 1978
Rat		3 generations	LOAEL	2,3,7,8-TCDD	0.01	Reproductive capacity	Murray et al.,1979
Rhesus monkey (*Macaca mulatta*)		7 months	LOAEL	2,3,7,8-TCDD	0.0021	Number of births	Allen et al., 1979
Rhesus monkey (*Macaca mulatta*)		7 - 48 months, maternal	LOAEL	2,3,7,8-TCDD	0.00059	Reproductive	Bowman et al., 1989b
Rat		Gestation days 6 to 15	NOAEL	2,3,7,8-TCDD	0.125	Litter size, pup weight	Khera and Ruddick, 1973
Rat		2 years	NOAEL	2,3,7,8-TCDD	0.01	Female mortality	Kociba et al., 1978
Rat		3 generations	NOAEL	2,3,7,8-TCDD	0.001	Reproductive capacity	Murray et al. 1979
Rhesus monkey (*Macaca mulatta*)		7 to 48 months, maternal	NOAEL	2,3,7,8-TCDD	0.00012	Reproductive	Bowman et al., 1989

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TOXICITY ENDPOINTS FOR OTHER MAMMALS - LABORATORY STUDIES
EFFECTIVE DIETARY DOSES OF TOTAL PCBs AND AROCLORS

SPECIES	EXPOSURE MEDIA	EXPOSURE DURATION	EFFECT LEVEL	PCB TYPE	EFFECTIVE DOSE (mg/kg/day)	FOOD INGESTION RATE (kg/kg/day)	EFFECTIVE FOOD CONC. (mg/kg)	EFFECT ENDPOINT	REFERENCE
Rhesus Monkey (*Macaca mulatta*)	Diet	7 months	LOAEL	Aroclor 1248	0.2	0.2		Decreased conception	Barsotti et al., 1976 (ATSDR)
Wistar Rat	Diet	From mating to weaning of pups	LOAEL	Aroclor 1254	0.2	0.08	2.5	Reduced growth rate in offspring	Overmann et al., 1987
Rhesus Monkey (*Macaca mulatta*)	Diet	2 months	LOAEL	Aroclor 1242	0.12	0.2		No weight gain	Becker et al., 1979 (ATSDR)
Rhesus Monkey (*Macaca mulatta*)	Diet	1.5 years	LOAEL	Aroclor 1248	0.12	0.2	5	Reduced birth weight	Allen and Barsotti, 1976 (Golub)
Rhesus Monkey (*Macaca mulatta*)	Diet	18 months	LOAEL	Aroclor 1248	0.1	0.2		Infant mortality	Allen et al., 1980 (ATSDR)
Cynomolgus Monkey	Diet	238 days	LOAEL	Aroclor 1254	0.1			100% fetal death	Truelove et al., 1982 (ATSDR)
Rhesus Monkey (*Macaca mulatta*)	Diet	18.2	LOAEL	Aroclor 1248	0.08	0.2		Decreased birth weight	Levin et al., 1988 (ATSDR)
Rhesus Monkey (*Macaca mulatta*)	Diet	> 8 months	LOAEL	Aroclor 1016	0.04	0.2	1	Reduced birth weight	Barsotti and Van Miller, 1984 (Golub)
Swine	Diet	Throughout gestation	EL-effect	Aroclor 1242	Not available		20	Decreased litter size	Hansen et al., 1975 (Golub)
Juvenile Male Rat	Single intraperitoneal injection	Observed after 14 days	NOAEL	Aroclor 1248	480			Growth rate of juveniles	Harris et al., 1993
Juvenile Male Rat	Single intraperitoneal injection	Observed after 14 days	NOAEL	Aroclor 1232	480			Growth rate of juveniles	Harris et al., 1993
Wistar Rat	Diet	52 weeks	NOAEL	Aroclor 1254	10	0.08		Decreased growth rate	Phillips et al., 1972 (ATSDR)
Rabbit	Oral-gavage	28 days	NOAEL	Aroclor 1254	10	0.034		Fetal death	Villeneuve et al., 1971 (ATSDR)
Adult Female Rat	Oral	Day 1,3,5,7 and 9 of lactation	NOAEL	Aroclor 1254	8	0.099		Growth rate of offspring	Sager & Girard, 1994
New Zealand White Rabbit	Diet	> 4 weeks	NOAEL	Aroclor 1248	3.6	0.034	100	Reduced growth rate in offspring	Thomas and Hinsdill, 1980 (Golub)
Sherman Rat	Diet	Multigenerational	NOAEL	Aroclor 1254	0.32	0.08	5	Decreased litter size	Linder et al., 1974
Osborne-Mendel Rat	Diet	During pregnancy and lactation	NOAEL	Aroclor 1254	0.059	0.08	50	Reduced litter size	Collins & Capen, 1980
Rhesus Monkey (*Macaca mulatta*)	Diet	> 8 months	NOAEL	Aroclor 1016	0.01	0.2	0.25	Reduced birth weight	Barsotti and Van Miller, 1984 (Golub)
Wistar Rat	Diet	From mating to weaning of pups	NOAEL	Aroclor 1254	0.0016	0.08	0.02	Reduced growth rate in offspring	Overmann et al., 1987

Notes:

*No relevant field studies were found.

Dose to rhesus monkey calculated using food ingestion rate of 0.2 kg/day and body weight of 5 kg (Sample et al., 1996)

TAB. 19
TOXICITY ENDPOINTS FOR MINK - LABORATORY STUDIES
EFFECTIVE DIETARY DOSES OF TOTAL PCBs AND AROCLORS

SPECIES	EXPOSURE MEDIA	EXPOSURE DURATION	EFFECT LEVEL	PCB TYPE	EFFECTIVE DOSE (mg/kg/day)	EFFECTIVE FOOD CONC. (mg/kg)	EFFECT ENDPOINT	REFERENCE
Laboratory studies								
Mink (*Mustela vision*)	Diet	4 weeks	LD50	Aroclor 1254	11.5	84	Adult mortality	Hornshaw (1984), as cited in Aulerich et al. (1986)
Mink (*Mustela vision*)	Diet	4 weeks	LD50	Aroclor 1254	10.8	79	Adult mortality	Aulerich et al. (1986)
Mink (*Mustela vision*)	Diet	4 weeks	LD50	Aroclor 1254	6.4	47	Adult mortality	Hornshaw et al. (1986)
Mink (*Mustela vision*)	Diet	4 weeks	LD50	Aroclor 1254 (weathered)	6.4	47	Adult mortality	Aulerich et al. (1986)
Mink (*Mustela vision*)	Diet	9 months	LD50	Aroclor 1254	0.9	6.6	Mortality	Ringer et al. (1981)
Mink (*Mustela vision*)	Diet	8 months	EL-effect	Aroclor 1016	2.7	20	Reduced birth weight and growth rate of kits	Bleavins et al., 1980
Mink (*Mustela vision*)	Diet	8 months	EL-effect	Aroclor 1016	2.7	20	Adult mortality	Bleavins et al., 1980
Mink (*Mustela vision*)	Diet	4 weeks	LOAEL	Aroclor 1254	1.4	10	Reduced weight gain in juveniles	Hornshaw et al. (1986)
Mink (*Mustela vision*)	Diet	8 months	LOAEL	Aroclor 1242	1.4	10	Adult mortality	Bleavins et al., 1980
Mink (*Mustela vision*)	Diet	3 months	EL-effect	Clophen A-50	2	Not reported	Decreased number of kits born alive	Kihlstom et al., 1992
Mink (*Mustela vision*)	Diet	3 months	EL-effect	Aroclor 1254	2	Not reported	Decreased number of kits born alive	Kihlstom et al., 1992
Mink (*Mustela vision*)	Diet	8 months	LOAEL	Aroclor 1242	0.7	5	Reduced reproduction	Bleavins et al., 1980
Mink (*Mustela vision*)	Diet	4 months	LOAEL	Aroclor 1254	0.7	5	Decreased number of kits born alive	Aulerich and Ringer (1977)
Mink (*Mustela vision*)	Diet	105 days	LOAEL	Aroclor 1254 (weathered)	0.5	3.57	Adult mortality	Platonow & Karstad (1973)
Mink (*Mustela vision*)	Diet	66 days	LOAEL	Not reported	0.5	3.3	Decreased number of kits born alive	Jensen et al. (1977)
Mink (*Mustela vision*)	Diet	4 months	EL-effect	Aroclor 1254	0.3	2.5	Decreased number of kits born alive	Aulerich et al. (1985)
Mink (*Mustela vision*)	Diet	6 months	EL-effect	Aroclor 1254	0.1	1	Reduced growth rates of kits	Wren et al., 1987
Mink (*Mustela vision*)	Diet	160 days	LOAEL	Aroclor 1254 (weathered)	0.09	0.64	Reduced number of kits born alive	Platanow & Karstad (1973)
Mink (*Mustela vision*)	Diet	8 months	NOAEL	Aroclor 1242	0.9	5	Adult mortality	Bleavins et al., 1980
Mink (*Mustela vision*)	Diet	4 months	NOAEL	Aroclor 1254	0.1	1	Decreased number of kits born alive	Aulerich & Ringer (1977)

TABLE B-20
TOXICITY ENDPOINTS FOR MINK - FIELD STUDIES
EFFECTIVE DIETARY DOSES OF TOTAL PCBs AND AROCLORS

SPECIES	FIELD COMPONENT	STUDY DURATION	EFFECT LEVEL	CONTAMINANT TYPE	EFFECTIVE DOSE (mg/kg/day)	EFFECTIVE FOOD CONC. (mg/kg)	EFFECT ENDPOINT	REFERENCE
Field studies								
Mink (*Mustela vision*)	Fed contaminated carp from Saginaw Bay, MI	Mink were fed prior to and throughout the reproductive period	LOAEL	PCBs, TEQs, others	0.13	N/A	Reproductive success, growth/survival of offspring	Heaton et al. (1995)
Mink (*Mustela vision*)	Fed contaminated carp from Saginaw Bay, MI	Mink fed prior to breeding and over two generations	LOAEL	PCBs, pesticides	0.08	0.5	Kit survival	Restum et al., 1998
Mink (*Mustela vision*)	Fed contaminated carp from Saginaw Bay, MI	Mink fed prior to breeding and over two generations	LOAEL	PCBs, pesticides	0.04	0.25	Reduced growth rate of kits	Restum et al., 1998
Mink (*Mustela vision*)	Fed contaminated carp from Saginaw Bay, MI	Mink fed prior to breeding and over two generations	LOAEL	PCBs, pesticides	0.04	0.25	Kit survival	Restum et al., 1998
Mink (*Mustela vision*)	Fed contaminated carp from Saginaw Bay, MI	Mink were fed prior to and throughout the reproductive period	NOAEL	PCBs, TEQs, others	0.004	N/A	Reproductive success, growth/survival of offspring	Heaton et al. (1995)

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TOXICITY ENDPOINTS FOR MINK - LABORATORY STUDIES
EFFECTIVE DIETARY DOSES OF DIOXIN TOXIC EQUIVALENTS (TEQs)

SPECIES	FIELD COMPONENT	STUDY DURATION	EFFECT LEVEL	CONTAMINANT TYPE	EFFECTIVE DOSE (mg/kg/day)	EFFECTIVE DOSE DIOXIN EQUIVALENTS (ug TEQ/kg/day)	EFFECT ENDPOINT	REFERENCE
Laboratory studies								
Mink kits (*Mustela vison*)	Intraperitoneal	12 days	LD_{54}	2,3,7,8-TCDD	< 0.01	< 0.01	Mortality	Aulerich et al., 1988
Mink males (*Mustela vison*)	Oral	Single dose	LD_{51}	2,3,7,8-TCDD	4.2	4.2	Mortality	Hochstein et al., 1988

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TABLE B-22
TOXICITY ENDPOINTS FOR MINK - FIELD STUDIES
EFFECTIVE DIETARY DOSES OF DIOXIN TOXIC EQUIVALENTS (TEQs)

SPECIES	FIELD COMPONENT	STUDY DURATION	EFFECT LEVEL	CONTAMINANT TYPE	EFFECTIVE DOSE DIOXIN EQUIVALENTS (ug TEQ/kg/day)	EFFECT ENDPOINT	REFERENCE
Field studies							
Mink (*Mustela vision*)	Fed contaminated carp from Saginaw Bay, MI	Fed prior to and throughout breeding period	LOAEL	TEQs, pesticides	0.0036	Growth rate of kits	Heaton et al. (1995)
Mink (*Mustela vision*)	Fed contaminated carp from Saginaw Bay, MI	Fed prior to and throughout breeding period	LOAEL	TEQs (chemically derived)	0.00224	Growth and survival rate of kits	Tillitt et al., 1996
Mink (*Mustela vision*)	Fed contaminated carp from Saginaw Bay, MI	Fed prior to and throughout breeding period	LOAEL	TEQs (bioassay derived)	0.00027	Growth and survival rate of kits	Tillitt et al., 1996
Mink (*Mustela vision*)	Fed contaminated carp from Saginaw Bay, MI	Fed prior to and throughout breeding period	NOAEL	TEQs (bioassay derived)	0.00344	Growth and survival rate of kits	Tillitt et al., 1996
Mink (*Mustela vision*)	Fed contaminated carp from Saginaw Bay, MI	Fed prior to and throughout breeding period	NOAEL	TEQs, pesticides	0.00025	Growth rate of kits	Heaton et al. (1995)
Mink (*Mustela vision*)	Fed contaminated carp from Saginaw Bay, MI	Fed prior to and throughout breeding period	NOAEL	TEQs (chemically derived)	0.00008	Growth and survival rate of kits	Tillitt et al., 1996

TABLE B-23
TAXONOMY OF STUDIED ORGANISMS

Phylum	*Class*	*Subclass*	*Order*	*Family*	*Genus*	*Species*	*Common name*
Chordata	Mammalia		Carnivora	Mustelidae	*Lutra*	*canadensis*	River Otter
Chordata	Mammalia		Carnivora	Mustelidae	*Mustela*	*vision*	Mink
Chordata	Mammalia		Carnivora	Procyonidae	*Procyon*	*lotor*	Raccoon
Chordata	Mammalia		Chiroptera	Vespertilionidae	*Myotis*	*lucifugus*	Little Brown Bat
Chordata	Mammalia		Lagomorphus	Leporidae	*[Sylvilagus]*	*[transitionalis]*	Rabbit [Eastern Cottontail]
Chordata	Mammalia		Rodentia	Muridae	*[Peromyscus]*	*[polionotus]*	Mouse [Oldfield Mouse]
Chordata	Mammalia		Rodentia	Muridae	*[Rattus]*	*[rattus]*	Rat
							Birds
Chordata	Aves		Anseriformes	Anatidae	*Aix*	*sponsa*	Wood Duck
Chordata	Aves		Anseriformes	Anatidae	*Anas*	*platyrhynchos*	Mallard Duck
Chordata	Aves		Charadriiformes	Laridae	*Hydropogne*	*caspia*	Caspian tern
Chordata	Aves		Charadriiformes	Laridae	*Sterna*	*hirundo*	Common tern
Chordata	Aves		Charadriiformes	Laridae	*Sterna*	*forsteri*	Forster's tern
Chordata	Aves		Ciconiiformes	Ardeidae	*Ardea*	*herodias*	Great Blue Heron
Chordata	Aves		Coraciiformes	Alcedinidae	*Ceryle*	*alcyon*	Kingfisher
Chordata	Aves		Falconiiformes	Accipitridae	*Haliaeetus*	*leucocephalus*	Bald Eagle
Chordata	Aves		Falconiiformes	Falconidae	*Falco*	*sparvenius*	American Kestrel
Chordata	Aves		Falconiiformes	Pandionidae	*Pandion*	*haliaeetus*	Osprey
Chordata	Aves		Galliformes	Phasianidae	*Colinus*	*virginianus*	Northern Bobwhite
Chordata	Aves		Galliformes	Phasianidae	*Coturnix*	*coturnix*	Japanese Quail
Chordata	Aves		Galliformes	Phasianidae	*Gallus*	*domesticus*	Domestic Chicken
Chordata	Aves		Galliformes	Phasianidae	*Phasianus*	*colchicus*	Ring-Necked Pheasant
Chordata	Aves		Passeriformes	Hirundinidae	*Tachycineta*	*bicolor*	Tree Swallow
Chordata	Aves		Passeriformes	Icteridae	*Agelaius*	*phoeniceus*	Red-Winged Blackbird
Chordata	Aves		Passeriformes	Icteridae	*Molothrus*	*ater*	Brown-Headed Cowbird
Chordata	Aves		Passeriformes	Icteridae	*Quiscalus*	*quiscula*	Common Grackle
Chordata	Aves		Passeriformes	Sturnidae	*Sturnus*	*vulgaris*	European Starling
Chordata	Aves		Pelecaniformes	Phalacrocoracidae	*Phalacrocorax*	*auritus*	Double-Crested Cormorant
Chordata	Aves		Strigiformes	Strigidae	*Otus*	*asio*	Screech Owl
							Fish
Chordata	Pisces	Actinopterygii	Acipenseriformes	Acipenseridae	*Acipenser*	*brevirostrum*	Shortnose Sturgeon
Chordata	Pisces	Actinopterygii	Beloniformes	Adrianichthydiae	*Oryzias*	*latipes*	Medaka
Chordata	Pisces	Actinopterygii	Clupeiformes	Clupeidae	*Clupea*	*harengus*	Baltic Herring
Chordata	Pisces	Actinopterygii	Cypriniformes	Catostomidae	*Catastomus*	*commersoni*	White sucker
Chordata	Pisces	Actinopterygii	Cypriniformes	Cyprinidae	*Danio*	*danio*	Zebrafish
Chordata	Pisces	Actinopterygii	Cypriniformes	Cyprinidae	*Notropis*	*hudsonius*	Spottail Shiner
Chordata	Pisces	Actinopterygii	Cypriniformes	Cyprinidae	*Phoxinus*	*phoxinus*	Minnow
Chordata	Pisces	Actinopterygii	Cypriniformes	Cyprinidae	*Pimephalus*	*promelas*	Fathead Minnow
Chordata	Pisces	Actinopterygii	Cypriniformes	Cyprinodontidae	*Fundulus*	*heteroclitus*	Killifish
Chordata	Pisces	Actinopterygii	Perciformes	Centrarchidae	*Lepomis*	*gibbosus*	Pumpkinseed
Chordata	Pisces	Actinopterygii	Perciformes	Centrarchidae	*Lepomis*	*auritus*	Redbreast Sunfish
Chordata	Pisces	Actinopterygii	Perciformes	Centrarchidae	*Micropterus*	*salmoides*	Largemouth Bass
Chordata	Pisces	Actinopterygii	Perciformes	Moronidae	*Morone*	*americana*	White Perch
Chordata	Pisces	Actinopterygii	Perciformes	Moronidae	*Morone*	*saxatilis*	Striped Bass
Chordata	Pisces	Actinopterygii	Perciformes	Percidae	*Perca*	*flavescens*	Yellow Perch
Chordata	Pisces	Actinopterygii	Perciformes	Sciaenidae	*Leiostomus*	*xanthurus*	Spot

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TABLE B-23
TAXONOMY OF STUDIED ORGANISMS

Phylum	*Class*	*Subclass*	*Order*	*Family*	*Genus*	*Species*	*Common name*
Chordata	Pisces	Actinopterygii	Perciformes	Sparidae	*Lagodon*	*rhomboides*	Pinfish
Chordata	Pisces	Actinopterygii	Pleuronectiformes	Pleuronectidae	*Parophrys*	*vetulus*	English Sole
Chordata	Pisces	Actinopterygii	Pleuronectiformes	Pleuronectidae	*Platichthys*	*flesus*	Baltic Flounder
Chordata	Pisces	Actinopterygii	Pleuronectiformes	Pleuronectidae	*Platichthys*	*stellatus*	Starry Flounder
Chordata	Pisces	Actinopterygii	Pleuronectiformes	Pleuronectidae	*Pseudopleuronectes*	*americanus*	Winter Flounder
Chordata	Pisces	Actinopterygii	Salmoniformes	Esocidae	*Esox*	*lucius*	Northern Pike
Chordata	Pisces	Actinopterygii	Salmoniformes	Salmonidae	*Coregonus*	*artedii*	Lake Herring
Chordata	Pisces	Actinopterygii	Salmoniformes	Salmonidae	*Oncorhynchus*	*tshawytscha*	Chinook Salmon
Chordata	Pisces	Actinopterygii	Salmoniformes	Salmonidae	*Salmo*	*gairdneri*	Rainbow Trout
Chordata	Pisces	Actinopterygii	Salmoniformes	Salmonidae	*Salvelinus*	*alpinus*	Arctic Charr
Chordata	Pisces	Actinopterygii	Salmoniformes	Salmonidae	*Salvelinus*	*fontinalis*	Brook Trout
Chordata	Pisces	Actinopterygii	Salmoniformes	Salmonidae	*Salvelinus*	*namaycush*	Lake Trout
Chordata	Pisces	Actinopterygii	Siluriformes	Ictaluridae	*Ictalurus*	*nebulosus*	Brown Bullhead
Chordata	Pisces	Actinopterygii	Siluriformes	Ictaluridae	*Ictalurus*	*punctatus*	Channel Catfish

TABLE B-24
STANDARD ANIMAL BODY WEIGHTS AND FOOD INTAKE RATES

Animal	*Body Weight (kg)*	*Food Ing. Rate (g/d)*	*Food Ingestion Rate (kg/d)*	*Food factor (kg/kg body wt/d)*
MAMMALS				
Mink	1		0.137	0.137
Mouse	0.03		0.0055	0.180
	0.028			
Mean Mouse	0.029			
Mouse, Oldfield	0.014	1.9	0.0019	
Rabbit	3.8		0.135	0.034
Rhesus Monkey	5		0.2	0.040
Rat	0.35		0.028	0.080
	0.435			
	0.303			
	0.273		0.0375	0.137
	0.365			
	0.26			
Mean Rat	0.331		0.03275	0.099
BIRDS				
Blackbird, Red-Winged	0.064		0.0137	0.214
Chicken, Domestic--adult	1.6		0.11	0.069
	1.5		0.106	0.071
Mean Chicken, Domestic--adult	1.55		0.108	0.070
Chickens, Domestic--chick	0.121		0.0126	0.104
	0.534		0.044	0.082
Mean Chicken, Domestic--chick	0.3275		0.0283	0.086
Cowbird, Brown-headed	0.049		0.01087	0.222
Dove, Ringed	0.155		0.017	0.110
Duck, Mallard--adult	1		0.1	0.100
	1.153		0.11	0.095
	1.15	115	0.115	0.100
	1		0.128	0.128
	1.17		0.121	0.103
Mean Duck, Mallard--adult	1.0946		0.1148	0.105
Duck, Mallard--duckling	0.782	78.2	0.0782	0.100
Kestrel, American	0.13		0.01	0.077
Owl, Screech	0.181	25	0.025	0.138
Pheasant, Ring-necked	1		0.0582	0.058
Quail, Japanese	0.15		0.0169	0.113
Quail, Japanese--3 months	0.072			

Note: All values are from Toxicological Benchmarks for Wildlife:1996 Revision (USEPA, 1996) unless oth

TABLE B-25
TOXICITY REFERENCE VALUES FOR FISH
DIETARY DOSES AND EGG CONCENTRATIONS OF TOTAL PCBs AND DIOXIN TOXIC EQUIVALENTS (TEQs)

TRVs		Pumpkinseed (*Lepomis gibbosus*)	Spottail Shiner (*Notropis hudsonius*)	Brown Bullhead (*Ictalurus nebulosus*)	Yellow Perch (*Perca flavescens*)	White Perch (*Morone americana*)	Largemouth Bass (*Micropterus salmoides*)	Striped Bass (*Morone saxatilus*)	Shortnose Sturgeon (*Acipenser brevirostrum*)	References
Tissue Concentration										
Lab-based TRVs for PCBs (mg/kg wet wt.)	LOAEL	1.5	***15***	***1.5***	***1.5***	1.5	1.5	1.5	***1.5***	Bengtsson (1980)
	NOAEL	0.16	***1.6***	***0.16***	***0.16***	0.16	0.16	0.16	***0.16***	
Field-based TRVs for PCBs (mg/kg wet wt.)	LOAEL	NA	NA	NA	NA	NA	NA	NA	NA	White perch and striped bass: Westin et al. (1983)
	NOAEL	***0.5***	NA	NA	NA	***3.1***	***0.5***	***3.1***	NA	Pumpkinseed and Largemouth bass: Adams et al. (1989, 1990, 1992)
Egg Concentration										
Lab-based TRV for TEQs (ug/kg lipid) from salmonids	LOAEL	***0.6***	Not derived	***18***	***0.6***	***0.6***	***0.6***	***0.6***	***0.6***	Brown Bullhead: Elonen et al. (1998)
	NOAEL	***0.29***	Not derived	***8.0***	***0.29***	***0.29***	***0.29***	***0.29***	***0.29***	All others: Walker et al. (1994)
Lab-based TRV for TEQs (ug/kg lipid) from non-salmonids	LOAEL	10.3	***103***	Not derived	10.3	10.3	10.3	10.3	10.3	Oliveri and Cooper (1997)
	NOAEL	0.54	***5.4***	Not derived	0.54	0.54	0.54	0.54	0.54	
Field-based TRVs for TEQs (ug/kg lipid)	LOAEL	NA	NA	NA	NA	NA	NA	NA	NA	
	NOAEL	NA	NA	NA	NA	NA	NA	NA	NA	

Note:
[a] Pumpkinseed (*Lepomis gibbosus*) and spottail shiner (*Notropis hudsonius*)
Units vary for PCBs and TEQ.
NA = Not available
Selected TRVs are ***bolded and italicized***.

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TABLE B-26
TOXICITY REFERENCE VALUES FOR BIRDS
DIETARY DOSES AND EGG CONCENTRATIONS OF TOTAL PCBs AND DIOXIN TOXIC EQUIVALENTS (TEQs)

TRVs		Tree Swallow (*Tachycineta bicolor*)	Mallard Duck (*Anas platyrhychos*)	Belted Kingfisher (*Ceryle alcyon*)	Great Blue Heron (*Ardea herodias*)	Bald Eagle (*Haliaeetus leucocephalus*)	References
Dietary Dose							
Lab-based TRVs for PCBs (mg/kg/day)	LOAEL	0.07	***2.6***	***0.07***	***0.07***	***0.07***	Mallard: Custer and Heinz (1980)
	NOAEL	0.01	***0.26***	***0.01***	***0.01***	***0.01***	All others: Scott (1977)
Field-based TRVs for PCBs (mg/kd/day)	LOAEL	NA	NA	NA	NA	NA	Tree Swallow: US EPA Phase 2 Database (1998)
	NOAEL	***16.1***	NA	NA	NA	NA	
Lab-based TRVs for TEQs (ug/kg/day)	LOAEL	0.014	***0.014***	***0.014***	***0.014***	***0.014***	Nosek et al. (1992)
	NOAEL	0.0014	***0.0014***	***0.0014***	***0.0014***	***0.0014***	
Field-based TRVs for TEQs (ug/kg/day)	LOAEL	NA	NA	NA	NA	NA	US EPA Phase 2 Database (1998)
	NOAEL	***4.9***	NA	NA	NA	NA	
Egg Concentration							
Lab-based TRVs for PCBs (mg/kg egg)	LOAEL	2.21	***2.21***	***2.21***	***2.21***	2.21	Scott (1977)
	NOAEL	0.33	***0.33***	***0.33***	***0.33***	0.33	
Field-based TRVs for PCBs (mg/kg egg)	LOAEL	NA	NA	NA	NA	NA	Bald Eagle: Wiemeyer (1984, 1993)
	NOAEL	***26.7***	NA	NA	NA	***3.0***	Tree Swallow: US EPA Phase 2 Database (1998)
Lab-based TRVs for TEQs (ug/kg egg)	LOAEL	0.02	0.02	***0.02***	NA	***0.02***	Great Blue Heron: Janz and Bellward (1996)
	NOAEL	0.01	0.01	***0.01***	2	***0.01***	Others: Powell et al. (1996a)
Field-based TRVs for TEQs (ug/kg egg)	LOAEL	NA	***0.02***	NA	***0.5***	NA	Mallard: White and Segniak (1994); White and Hoffman (1995)
	NOAEL	***13***	***0.005***	NA	***0.3***	NA	Great Blue Heron: Sanderson et al. (1994)
							Tree Swallow: US EPA Phase 2 Database (1998)

Note: Units vary for PCBs and TEQ.
NA = Not Available
Selected TRVs are ***bolded and italicized***.

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TABLE B-27
TOXICITY REFERENCE VALUES FOR MAMMALS
DIETARY DOSES OF TOTAL PCBs AND DIOXIN TOXIC EQUIVALENTS (TEQs)

TRVs		**Little Brown Bat (*Myotis lucifugus*)**	**Raccoon (*Procyon lotor*)**	**Mink (*Mustela vison*)**	**Otter (*Lutra canadensis*)**	**References**
Lab-based TRVs for PCBs (mg/kg/day)	LOAEL	***0.15***	***0.15***	0.07	0.07	Mink and otter: Aulerich and Ringer (1977)
	NOAEL	***0.032***	***0.032***	0.01	0.01	Raccoon and bat: Linder et al. (1984)
Field-based TRVs for PCBs (mg/kg/day)	LOAEL	NA	NA	***0.13***	***0.13***	Heaton et al. (1995)
	NOAEL	NA	NA	***0.004***	***0.004***	
Lab-based TRVs for TEQs (ug/kg/day)	LOAEL	***0.001***	***0.001***	0.001	0.001	Murray et al. (1979)
	NOAEL	***0.0001***	***0.0001***	0.0001	0.0001	
Field-based TRVs for TEQs (ug/kg/day)	LOAEL	NA	NA	***0.00224***	***0.00224***	Tillitt et al. (1996)
	NOAEL	NA	NA	***0.00008***	***0.00008***	

Note: Units vary for PCBs and TEQ.
Note: TRVs for raccoon and bat are based on mulit-generational studies to which interspecies uncertainty factors are applied.
NA = Not Available
Final selected TRVs are ***bolded and italicized*.**

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TABLE B-28: WILDLIFE SURVEY RESULTS Amphibians
Hudson River
New York

Information Source	Date	Contact	Response	Contact Information	Data Available	Information/Findings
				Amphibians		
Amphibian Expert	1-Jun-99	Email	Yes	Thomas Palmer, frog consultant for Wellesley Project; Ophis@world.std.com	He doesn't know anything about PCB effects on frogs; posted message on amphibian web page	Recommended the following website: http://cciw.ca/green-lane/herptox/
NYSDEC - Amphibian and Reptile Atlas Project	3-Jun-99	Email	No	herps@gw.dec.state.ny.us; http://www.dec.state.ny.us/website/dfwmr/wildlife/herp/index.html		
NYS Department of Environmental Conservation - Endangered Species Unit	8-Jun-99	WWW	No	www.dec.state.ny.us/website/dfwmr/wildlife/endspec/enspamphib.html	Brief summaries, listed by species, for NY state.	*Eurycea longicauda* (Longtail Salamander): nocturnal salamander which occupies shallow rocky streams and moist forested areas. Found in Cattaraugus County and mid Hudson Valley. Very few in NY. Status: Special Concern.
NYS Department of Environmental Conservation	8-Jun-99	WWW	No	www.dec.state.ny.us/website/dfwmr/wildlife/herp/atproj.html	10 year survey documenting geographic distribution of herpetofauna in NY state.	Common frogs and toads abundant, snapping turtles abundant, some box turtles present.
NYSDEC	16-Jun-99	Call	Yes	Mark Brown (518) 623-3671	Familiar with the area regarding mammals, birds, and herps. Good source. See General Info page.	Reports snapping and painted turtles, red back and two-line salamanders. Frogs: bull, spring peepers, gray tree, northern leopard, and pickeral. American toad. Garter and water snakes (none are poisonous). Currently working on a herp survey.
Ndakinna Wilderness Project	6/3/1999 6/16/99	Email Call Call	No No Yes	Jim Brushek (518) 583-9980x3, 23 Middle Grove Road, Greenfield Center, NY 12833; Received address from Saratoga County Information - Annamaria Dalton (annamaria@spa.net)	Professional Tracker	Common amphibians present in strong numbers. Box, snapper, and painted turtles. Some snakes which he could not identify.

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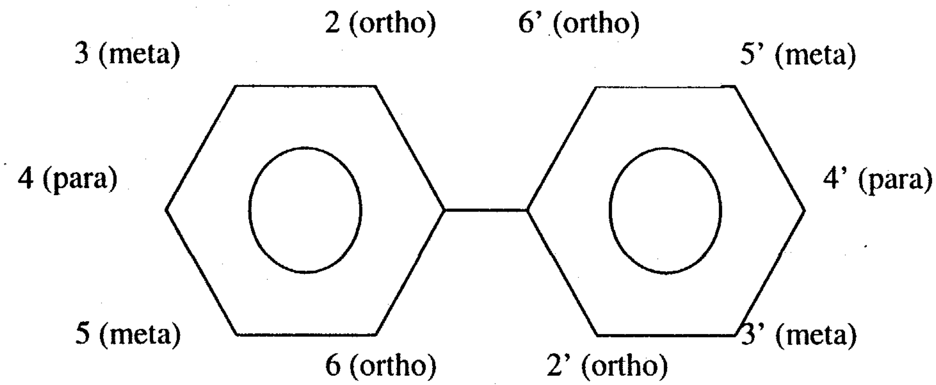

Figure B-1: Shape of Biphenyl and Substitution Sites

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Figure B-2
Selected Fish Aroclor and Total PCB Toxicity Endpoints

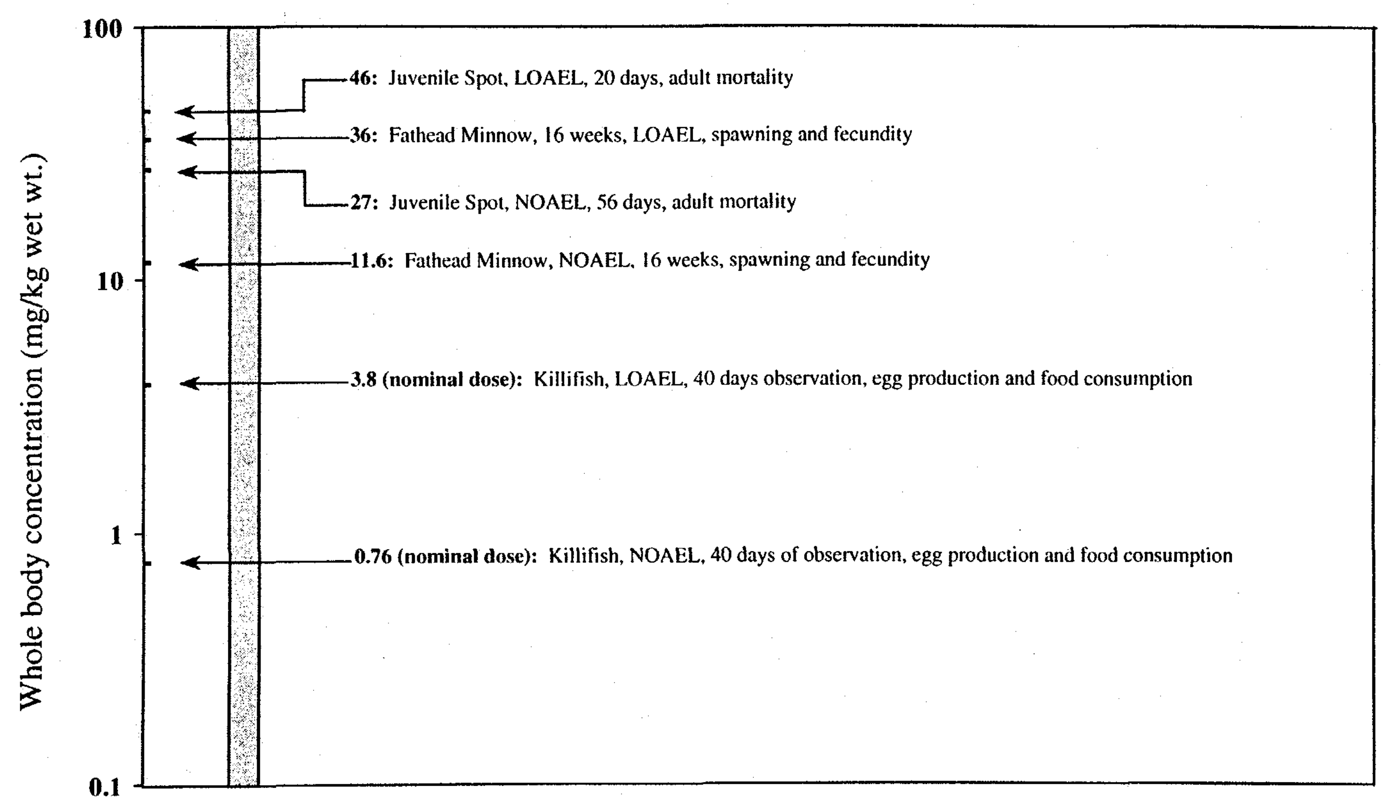

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Figure B-3
Selected Fish Egg Dioxin Equivalent Toxicity Endpoints

Endpoint: Early Life Stage Mortality

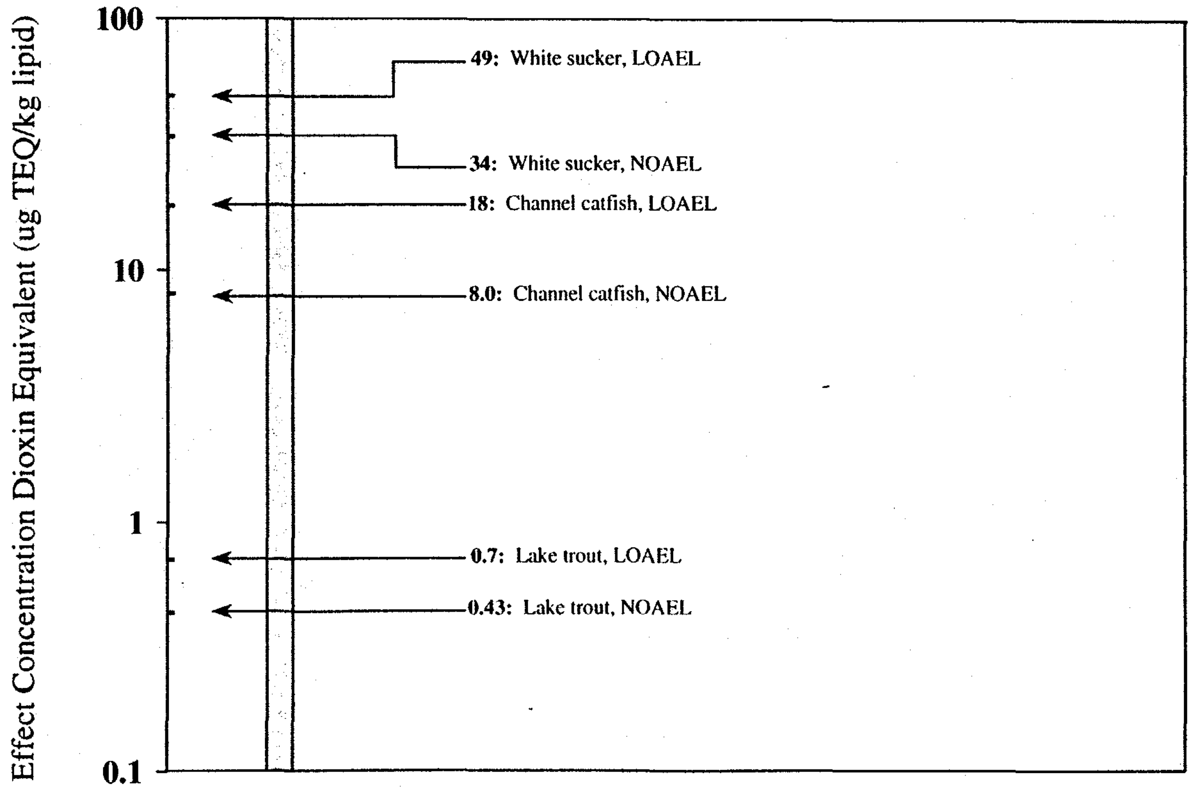

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Figure B-4
Selected Bird Diet Aroclor and Total PCB Toxicity Endpoints

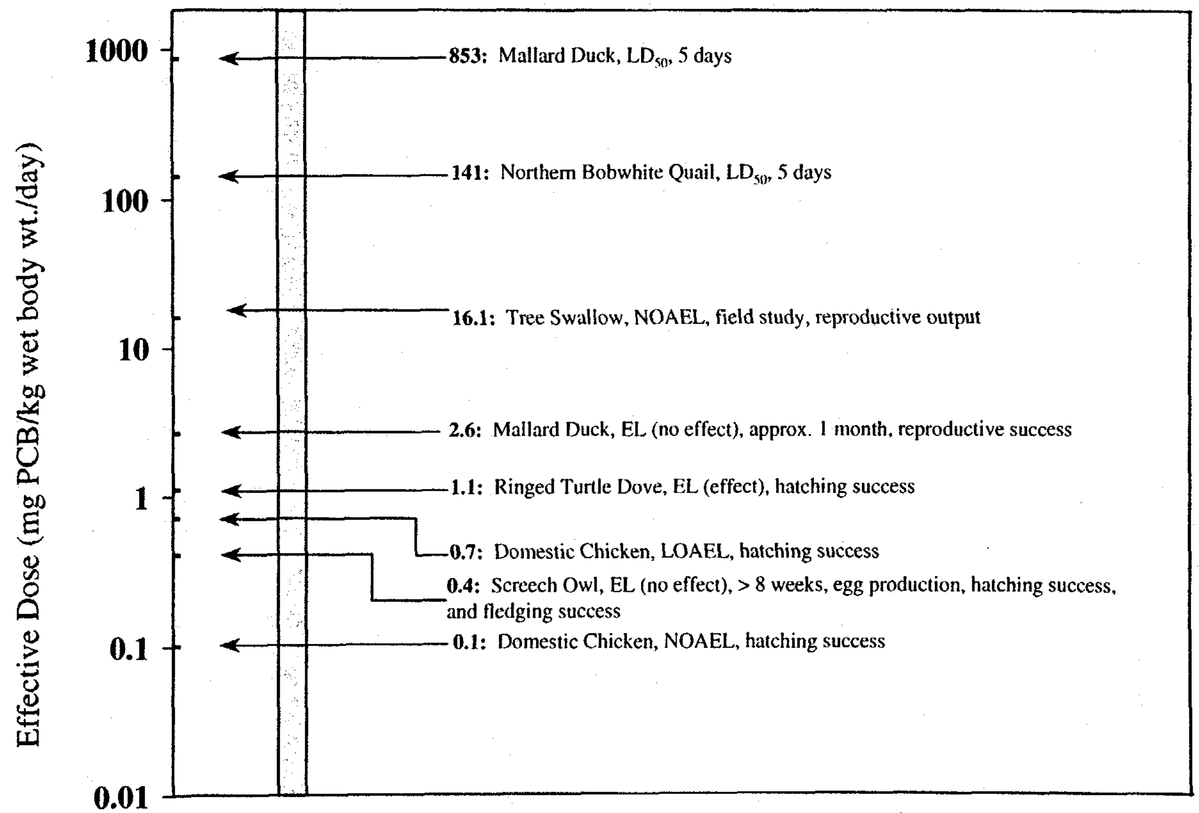

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Figure B-5
Selected Bird Diet Dioxin Equivalent Toxicity Endpoints

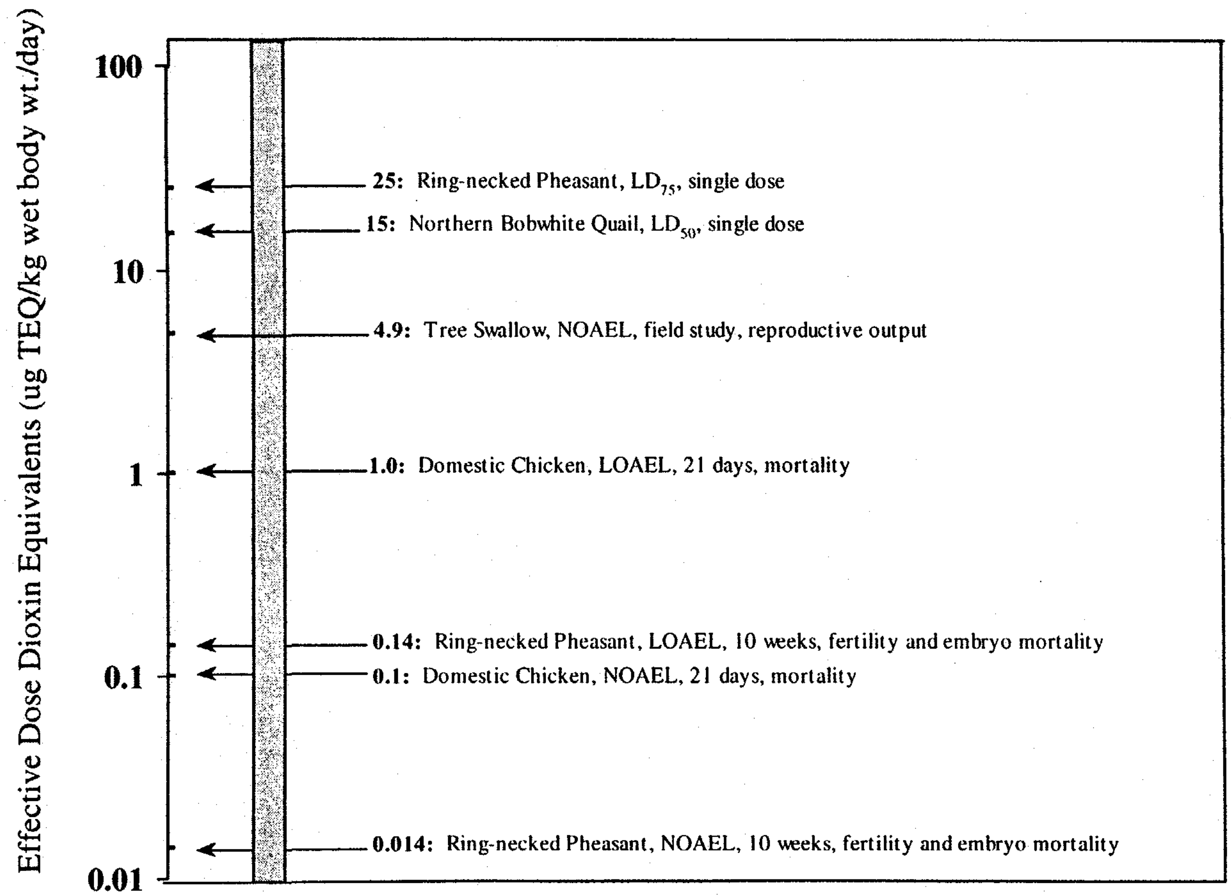

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Figure B-6
Selected Bird Egg Aroclor and Total PCB Toxicity Endpoints

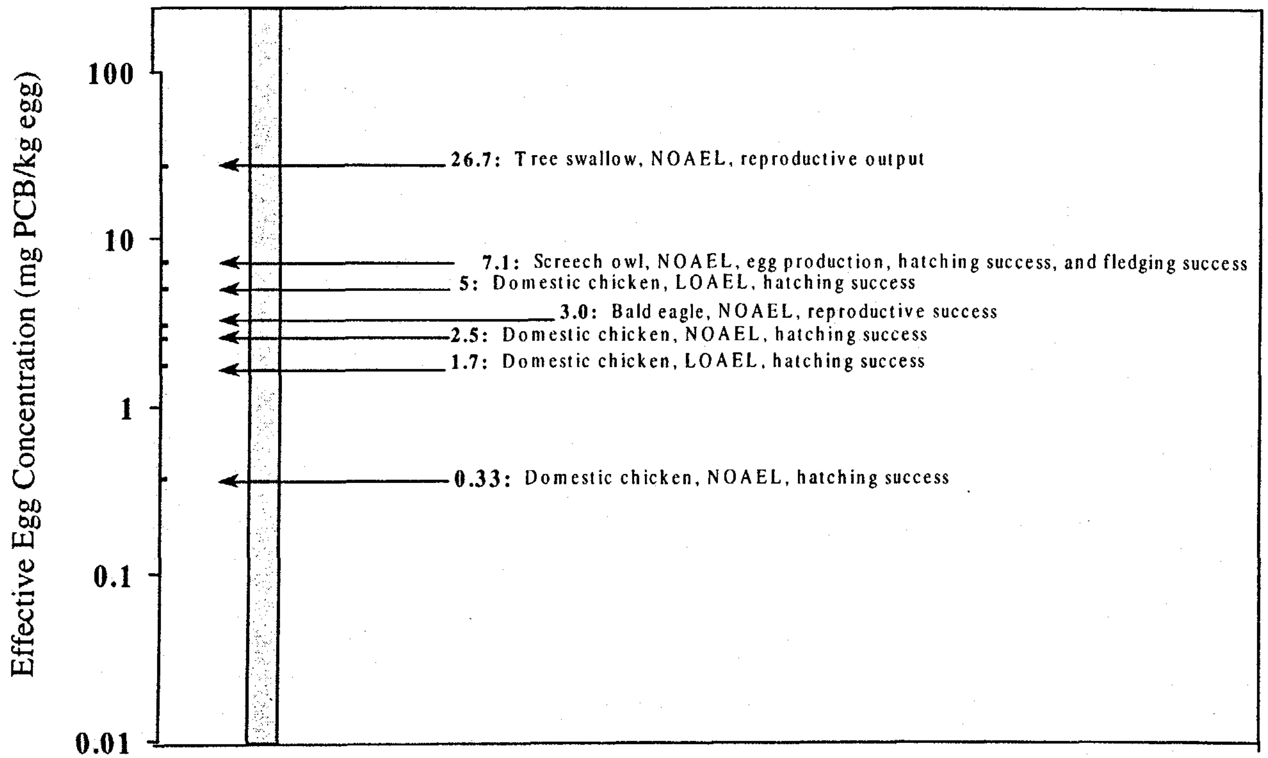

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Figure B-7

Selected Bird Egg Dioxin Equivalent Toxicity Endpoints

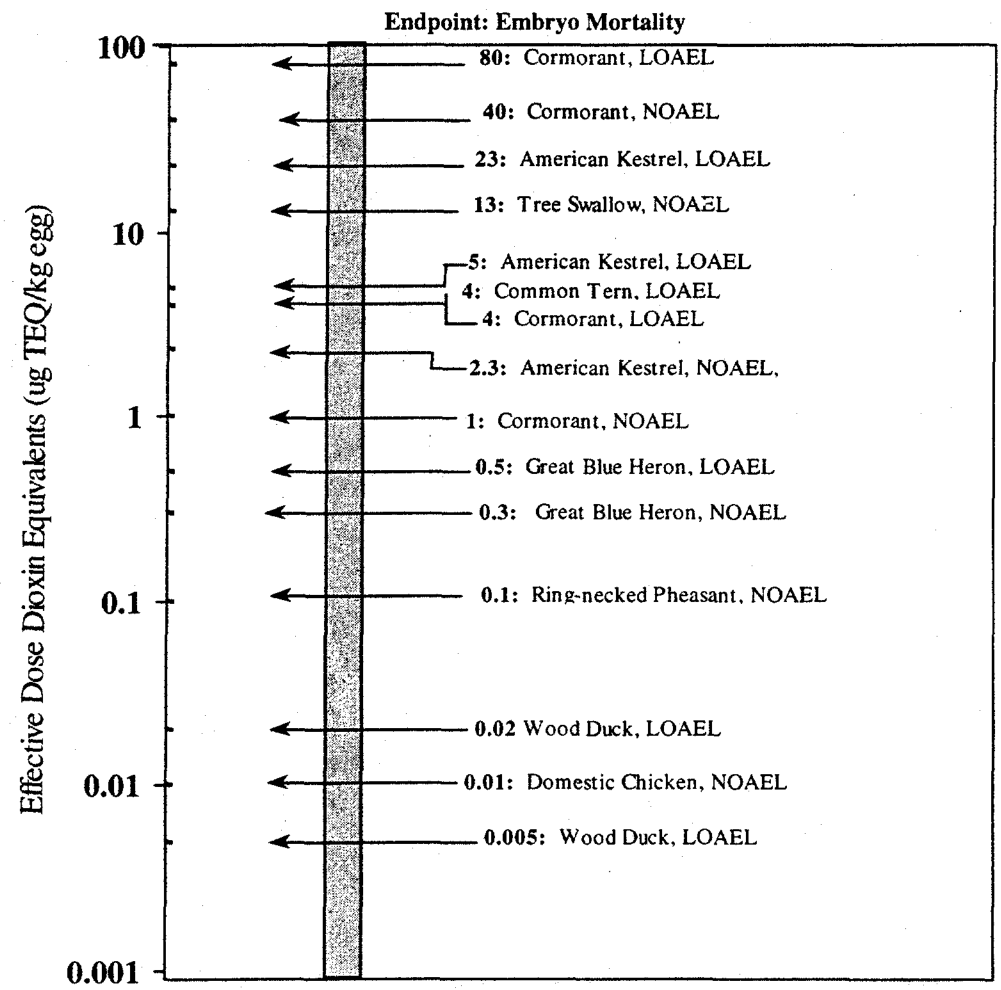

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Figure B-8
Selected Mink Aroclor and Total PCB Toxicity Endpoints

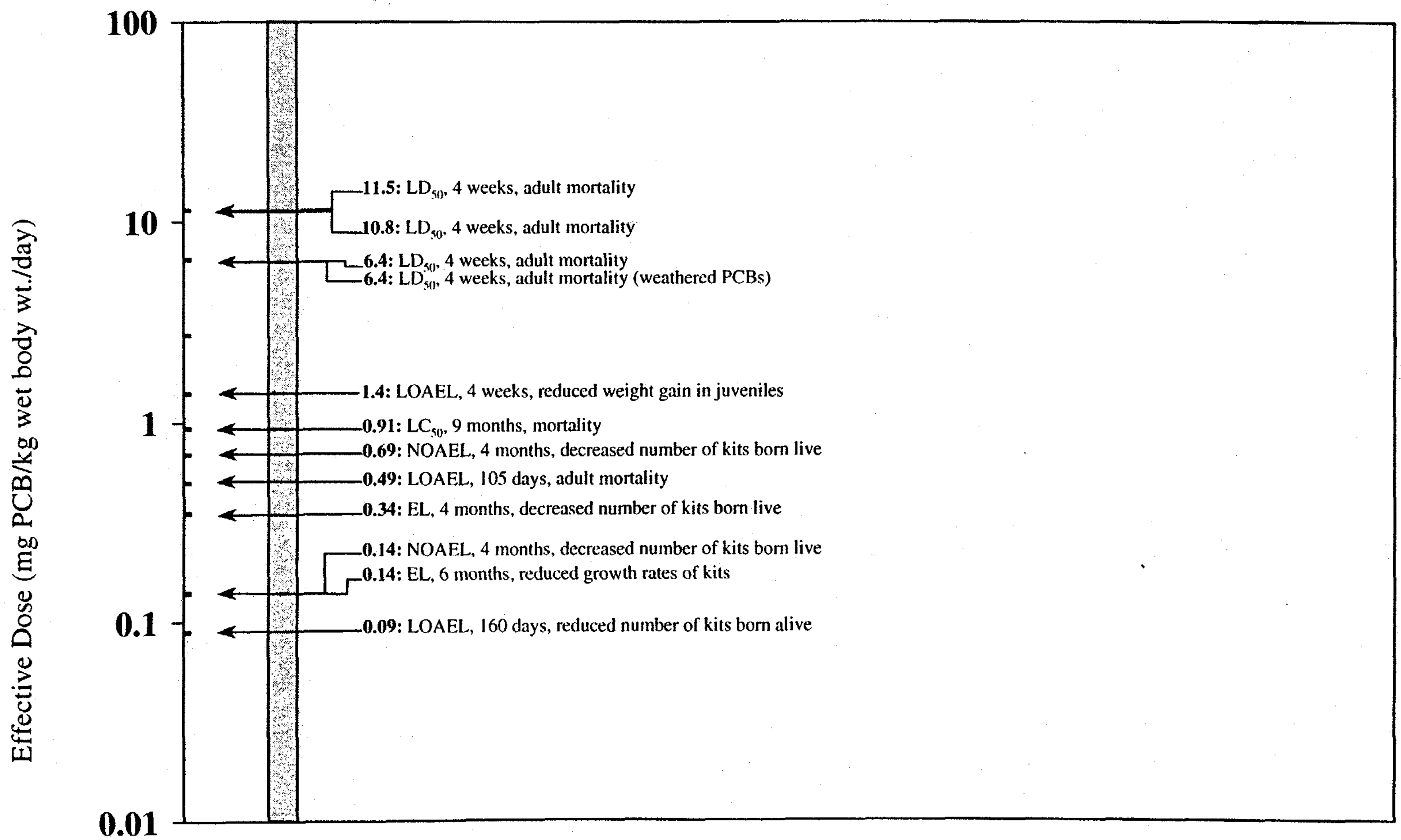

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Figure B-9
Selected Mammal Aroclor and Total PCB Toxicity Endpoints

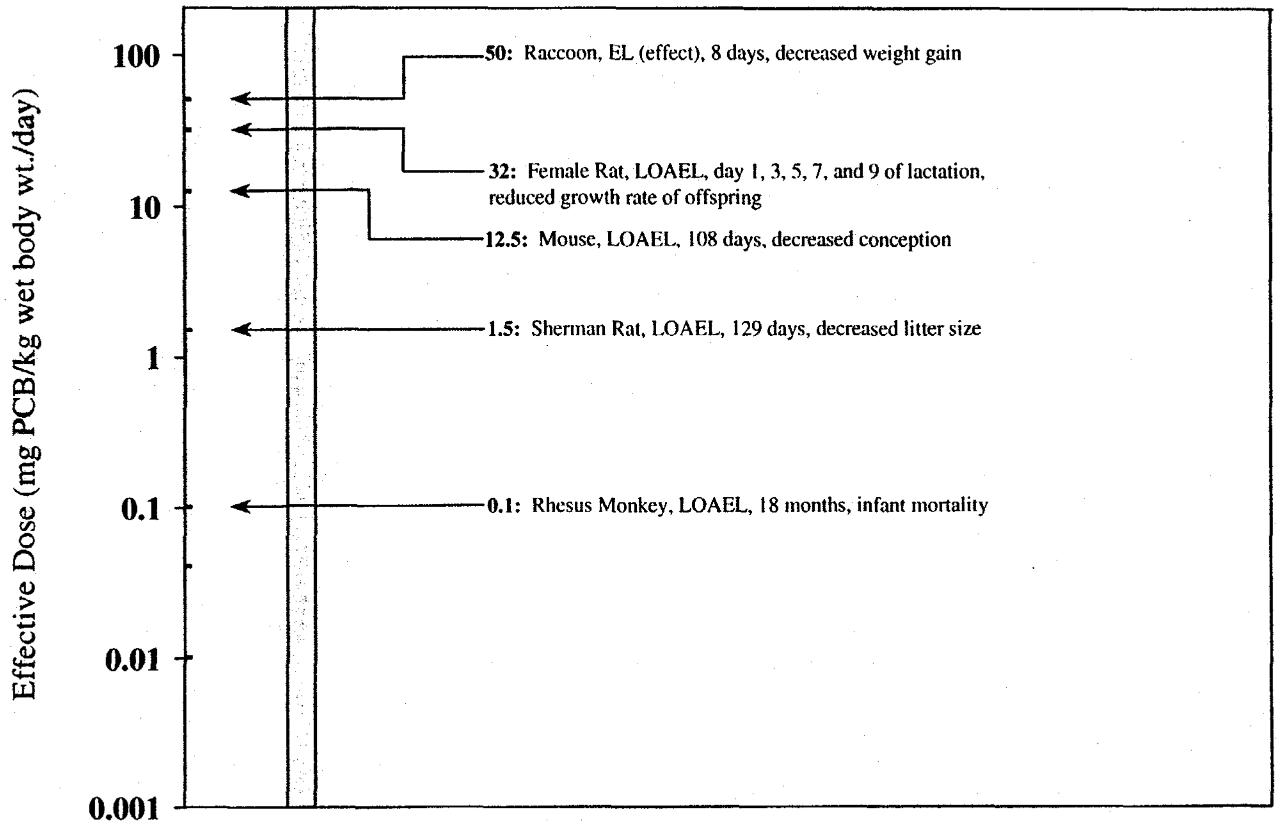
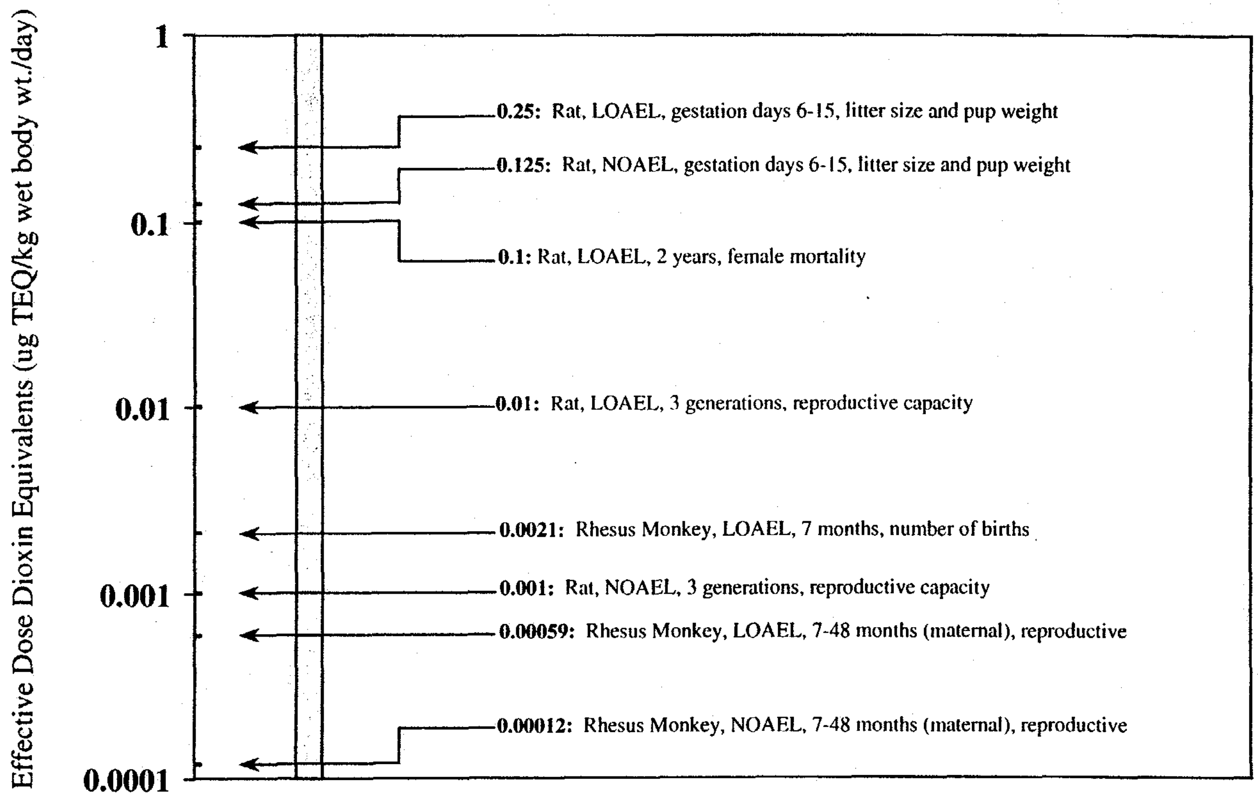


Figure B-10
Selected Mammal Dioxin Equivalent Toxicity Endpoints



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