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PHASE 2 REPORT- REVIEW COPY
FURTHER SITE CHARACTERIZATION AND ANALYSIS

VOLUME 2E - BASELINE ECOLOGICAL RISK ASSESSMENT

HUDSON RIVER PCBs REASSESSMENT RI/FS

AUGUST 1999

For

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

Book 1 of 3
Text

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

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

August 4, 1999

To All Interested Parties:

The U.S. Environmental Protection Agency (USEPA) is pleased to release the baseline Ecological Risk Assessment for the Hudson River (ERA), which is part of Phase 2 of the Reassessment Remedial Investigation/Feasibility Study (Reassessment RI/FS) for the Hudson River PCBs Superfund site. The ERA evaluates current and future risk to the environment in the Upper Hudson River (Hudson Falls, New York to Federal Dam at Troy, New York) and current risk to the environment in the Lower Hudson River (Federal Dam to the Battery in New York City) in the absence of remediation. The 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 for the Hudson River PCBs site.

As stated in the April 1999 Responsiveness Summary for Phase 2 - Ecological Risk Assessment Scope of Work, USEPA will complete the assessment of future ecological risk in the Lower Hudson River following review of the revised Thomann-Farley model developed for the Hudson River Foundation.

USEPA will accept comments on the ERA until September 7, 1999. 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 Comments

USEPA will hold two Joint Liaison Group meetings to discuss the findings of the ERA. The first meeting will be on the date of release of the report, August 4, 1999, and will be held at 7:30 p.m. at the Marriott Hotel, 189 Wolf Road, Albany, New York. The second meeting will be on August 5, 1999 at 7:30 p.m. at the Sheraton Hotel, 40 Civic Center Plaza, Poughkeepsie, New York. Both meetings are open to the general public. Notification of these meetings was sent to Liaison Group members, interested parties, and the press several weeks prior to the meetings.

During the public comment period, USEPA will hold availability sessions to answer questions from the public regarding the ERA. The availability sessions will be held from 2:30 to 4:30 p.m. and from

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6:30 to 8:30 p.m. on August 18, 1999 at the Holiday Inn Express, 946 New Loudon Road, Latham, New York.

If you need additional information regarding the ERA, the availability sessions, 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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Ecological Risk Assessment
Executive Summary
August 1999

This document presents the baseline Ecological Risk Assessment (ERA) for the Hudson River, which is part of Phase 2 of the Reassessment Remedial Investigation/Feasibility Study (Reassessment RI/FS) for the Hudson River PCBs site in New York. The ERA quantitatively evaluates the current and future risks to the environment in the Upper Hudson River (Hudson Falls, New York to Federal Dam at Troy, New York) and the current risks to the environment in the Lower Hudson River (Federal Dam to the Battery in New York City) posed by polychlorinated biphenyls (PCBs) in the absence of remediation.[1] This report uses current U.S. Environmental Protection Agency (USEPA) policy and guidance as well as additional site data and analyses to follow up 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 indicates that PCBs in the Hudson River generally exceed levels that have been shown to cause adverse ecological effects, and that those levels will continue to be exceeded in the Upper Hudson through 2018 (the entire forecast period). The results of the ERA 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

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[1] A separate ecological risk assessment is being conducted to evaluate the future risks to the environment in the Lower Hudson River.

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 to the Battery in New York City. As defined in the ERA, the Upper Hudson River is the 40 mile (64 km) stretch from Hudson Falls to the Federal Dam at Troy. The Lower Hudson River extends approximately 160 miles (258 km) from the Federal Dam to the Battery.

The Hudson River is home to a wide variety of ecosystems. These ecosystems differ between the Upper Hudson River and the Lower Hudson River. The Upper Hudson River is non-tidal, consists of a series of pools separated by dams, and is entirely freshwater. In contrast, 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. Both the Upper and Lower Hudson have deep water environments as well as shallow nearshore areas with aquatic vegetation.

PCBs were released from two General Electric Company 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 were selected to include direct exposure to PCBs in 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 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 (forage, omnivorous, and piscivorous)
- Protection and maintenance (survival, growth, and reproduction) of local insectivorous birds
- Protection and maintenance (survival, growth, and reproduction) of local waterfowl
- Protection and maintenance (survival, growth, and reproduction) of local piscivorous birds
- Protection and maintenance (survival, growth, and reproduction) of local wildlife
- 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 generally include measured or 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 are:

1) Benthic community indices, such as richness, abundance, diversity and biomass;

2) Concentrations of PCBs in fish and invertebrates to evaluate food-chain exposure;

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

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

5) Exceedence of criteria for concentrations of PCBs in river water that are protective of fish

and wildlife;

6) Exceedence of guidelines for concentrations of PCBs in sediments that are protective of aquatic health; and

7) Field observations.

Receptors of Concern

The 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 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:

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)* P/S
- Shortnose sturgeon (*Acipenser brevirostrum)* O

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 provide current and future (*i.e.,*measured and modeled) concentrations of PCBs in fish, sediments and river water, and form the basis of the site data collection and analyses that were used in conducting the ERA. Exposure parameters were obtained from USEPA references, the scientific literature, and directly from researchers.

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, TRVs were selected 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 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 (measured or 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. Based on a comparison of modeled concentrations to measured values, the PCB concentrations in fish presented in the May 1999 Baseline Modeling Report are expected to be within an order of magnitude, but likely closer to a factor of two, of future measured values.

To integrate the various components of the ERA, 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 originating in the 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

Benthic community structure as a food source for local fish populations was assessed using three lines of evidence. All three suggest an adverse effect of PCBs on benthic invertebrate populations serving as a food source to local fish in the Upper Hudson River. Two lines of evidence 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 evaluated using seven lines of evidence. Collectively, they indicate that current (1993) and future PCB exposures are not expected to be of a sufficient magnitude to prevent reproduction or recruitment of common fish species in the Hudson River. However, current and future exposures to the PCBs may reduce or impair the survival, growth, and reproductive capability of resident fish in the Upper Hudson River, and current exposure to PCBs may have similar adverse effects on upper trophic level fish (such as largemouth bass and white perch) in the Lower Hudson River.

Current fish body burdens exceed most TRVs (*i.e.*, TQ $\geq$ 1) in the Upper Hudson River for all species, and body burdens consistently exceed TRVs for upper trophic level fish in both the Upper and Lower Hudson River. Future body burdens in fish are expected to exceed TRVs through 2018 (the entire forecast period) in the Upper Hudson River for several of the upper trophic level fish species. 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 dioxin-like PCBs, no uncertainty factors were needed). Given the magnitude of the TQs, they would have to decrease by an order of magnitude or more to fall below 1 for most fish species in the Upper Hudson River and for upper trophic level fish in the Lower Hudson River.

Measured and modeled concentrations of PCBs in river water in the Upper Hudson River and sediment show exceedences of their respective criteria and guidelines for protection of fish through 2018 (the entire forecast period). Measured concentrations of PCBs in river water and sediment in the Lower Hudson River typically exceed all but the least stringent criteria and guidelines for protection of fish at most locations.

Insectivorous Birds

Risks to insectivorous birds were evaluated using six lines of evidence. Collectively, they indicate that current and future concentrations of PCBs are not of a sufficient magnitude to prevent reproduction of insectivorous birds. However, anomalous nesting behavior has been observed in tree swallows in the Upper Hudson River and these behaviors may adversely affect reproductive capability at the population level. There is a moderate degree of uncertainty in the calculated doses of PCBs in tree swallow diet and the concentrations of PCBs in eggs. There is a low degree of uncertainty associated with the tree swallow TRVs, which were derived from field studies of Hudson River tree swallows.

Measured and modeled concentrations of PCBs in Upper Hudson River water exceed criteria developed for the protection of wildlife through 2018 (the entire forecast period). Measured concentrations of PCBs in Lower Hudson River water exceed criteria developed for the protection of wildlife at most locations.

Waterfowl

Risks to waterfowl were evaluated using six lines of evidence. Collectively, they indicate that current and future concentrations of PCBs are not of a sufficient magnitude to prevent reproduction of the waterfowl. However, current and future exposures to the PCBs may reduce or impair the survival, growth, and reproductive capability of waterfowl in the Upper Hudson River. To a lesser degree, current exposures may have similar adverse effects on waterfowl in the Lower Hudson River.

Calculated dietary doses of PCBs and concentrations of PCBs in eggs under current and future conditions typically exceed their respective TRVs. TQs for the dioxin-like PCBs consistently show greater exceedences than TQs for total PCBs. Exceedences of TRVs are expected to occur through 2018 (the entire forecast period). 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 1 for the mallard duck in the Upper Hudson River.

Measured and modeled concentrations of PCBs in Upper Hudson River water exceed criteria developed for the protection of wildlife through 2018 (the entire forecast period). Measured concentrations of PCBs in Lower Hudson River water exceed criteria developed for the protection of wildlife at most locations.

Piscivorous Birds

Risks to piscivorous birds were evaluated using six lines of evidence. Collectively, they indicate that current and future concentrations of PCBs are not of a sufficient magnitude to prevent reproduction of these piscivorous species, which have been observed along the Hudson River. However, current and future exposures to the PCBs may reduce or impair the survival, growth, and reproductive capability of piscivorous birds in the Upper Hudson River, and current exposure to PCBs may have similar adverse effects on piscivorous birds in the Lower Hudson River. Calculated dietary doses of PCBs and concentrations of PCBs in eggs exceed all TRVs for the Upper Hudson River through 2018 (the entire forecast period) and current exposures exceed all TRVs in the Lower Hudson River. There is a moderate degree of uncertainty in the calculated dietary doses and concentrations in eggs. Given the magnitude of the majority of the TQs, they would have to decrease by an order of magnitude or more to fall below 1 for piscivorous birds in the Upper Hudson River.

Measured and modeled concentrations of PCBs in Upper Hudson River water exceed criteria developed for the protection of wildlife through 2018 (the entire forecast period). Measured concentrations of PCBs in Lower Hudson River water exceed criteria developed for the protection of wildlife at most locations.

Insectivorous Mammals

Risks to insectivorous mammals were evaluated using four lines of evidence. Collectively, they indicate that current and future concentrations of PCBs are not of a sufficient magnitude to prevent reproduction of insectivorous mammals. However, current and future exposures to the PCBs may reduce or impair the survival, growth, and reproductive capability of mammals in the Upper Hudson River. To a lesser degree, current exposures may have similar adverse effects on insectivorous mammals in the Lower Hudson River. Modeled dietary doses for the little brown bat exceed TRVs under current and future conditions in the Upper Hudson River. Given the magnitude of the majority of the TQs, they would have to decrease by an order of magnitude or more to fall below 1. There is a moderate degree of uncertainty in the calculated dietary doses.

Measured and modeled concentrations of PCBs in Upper Hudson River water exceed criteria developed for the protection of wildlife through 2018 (the entire forecast period). Measured concentrations of PCBs in Lower Hudson River water exceed criteria developed for the protection of wildlife at most locations.

Omnivorous Mammals

Risks to omnivorous mammals were evaluated using four lines of evidence. Collectively, they indicate that current and future concentrations of PCBs are not of a sufficient magnitude to prevent reproduction of omnivorous mammals. However, current and future exposures to the PCBs may reduce or impair the survival, growth, and reproductive capability of mammals in the Upper Hudson River. To a lesser degree, current exposures may have similar adverse effects on omnivorous mammals in the Lower Hudson River. Modeled dietary doses for the raccoon exceed TRVs under current and future conditions in the Upper Hudson River. Given the magnitude of the majority of the TQs, they would have to decrease by an order of magnitude or more to fall below 1. There is a moderate degree of uncertainty in the calculated dietary doses.

Measured and modeled concentrations of PCBs in Upper Hudson River water exceed criteria developed for the protection of wildlife through 2018 (the entire forecast period). Measured concentrations of PCBs in Lower Hudson River water exceed criteria developed for the protection of wildlife at most locations.

Piscivorous Mammals

Risks to piscivorous mammals were evaluated using four lines of evidence. Collectively, they indicate that current and future concentrations of PCBs are not of a sufficient magnitude to prevent reproduction of piscivorous mammals. However, current exposures to the PCBs may reduce or impair the survival, growth, and reproductive capability of mammals in the Upper and Lower Hudson River.. Future exposures may have adverse effects on piscivorous mammals in the Upper Hudson River. Modeled dietary doses for the mink and river otter exceed TRVs under current and future conditions in the Upper Hudson River. Given the magnitude of the majority of the TQs, they would have to decrease by an order of magnitude or more to fall below 1. There is a moderate degree of uncertainty in the calculated dietary doses.

Measured and modeled concentrations of PCBs in Upper Hudson River water exceed criteria developed for the protection of wildlife through 2018 (the entire forecast period). Measured concentrations of PCBs in Lower Hudson River water exceed criteria developed for the protection of wildlife at most locations.

Threatened and Endangered Species

Risks to threatened and endangered species were evaluated using four lines of evidence. Collectively, they indicate that current and future concentrations of PCBs are of a sufficient magnitude to adversely affect the reproductive capability of these fragile populations.

TQs for the bald eagle exceed 1 through 2018 (the entire forecast period) in the Upper Hudson River and exceed 1 for all locations in the Lower Hudson River. TQs for the shortnose sturgeon exceed 1 through 2018 (the entire forecast period) for most TRVs developed on the basis of reproductive effects at all locations in the Upper Hudson River. TQs for the shortnose sturgeon exceed 1 for all locations in the Lower Hudson River. There is a moderate degree of uncertainty in the modeled fish body burdens and calculated dietary doses and egg concentrations of PCBs. Given the magnitude of the TQs for the bald eagle, they would have to decrease by two orders of magnitude or more to fall below 1 in the Upper Hudson River.

Measured and modeled concentrations of PCBs in Upper Hudson River water exceed criteria developed for the protection of wildlife through 2018 (the entire forecast period). Measured concentrations of PCBs in Lower Hudson River water exceed criteria developed for the protection of wildlife at most locations. Measured and modeled concentrations of PCBs in sediment in the Upper Hudson River exceed all but the least stringent guidelines at all locations through 2018 (the entire forecast period). Measured concentrations of PCBs in sediment in the Lower Hudson River exceed all but the least stringent guidelines at all locations.

Significant Habitats

Risks to significant habitats were evaluated using two lines of evidence. Together, they indicate that current and future concentrations of PCBs are of a sufficient magnitude to adversely affect the ability of particular habitats in the Hudson River to support sustainable, healthy wildlife populations.

Measured and modeled concentrations of PCBs in Upper Hudson River water exceed criteria developed for the protection of wildlife through 2018 (the entire forecast period). Measured concentrations of PCBs in Lower Hudson River water exceed criteria developed for the protection of wildlife at most locations. Measured and modeled concentrations of PCBs in sediment in the Upper Hudson River exceed all but the least stringent guidelines at all locations through 2018 (the entire forecast period). Measured concentrations of PCBs in sediment in the Lower Hudson River exceed all but the least stringent guidelines at all locations.

Conclusions

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

- Fish in the Hudson River are at risk from 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.

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

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

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

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

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

- The risks to fish and wildlife are greatest in the Upper Hudson River (in particular the Thompson Island Pool) and decrease in relation to PCB concentrations down river. Based on modeled future PCB concentrations, many species are expected to be at considerable risk through 2018 (the entire forecast period).

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Chapter 1

1.0 INTRODUCTION

1.1 Purpose of Report

This report is part of the Phase 2 investigation of Hudson River sediment polychlorinated biphenyl (PCB) contamination. This investigation is being conducted under the direction of the United States Environmental Protection Agency (USEPA) and is part of a three-phase Reassessment Remedial Investigation/Feasibility Study (Reassessment RI/FS) to reassess USEPA's 1984 No Action decision with respect to the PCB-contaminated sediments in the Upper Hudson River. For purposes of the Reassessment, the area of the Upper Hudson is defined as the river bed between the Fenimore Bridge at Hudson Falls (just south of Glens Falls) and the Federal Dam at Troy. However, the Hudson River PCB Superfund site encompasses the Hudson River from Hudson Falls to the Battery in New York Harbor, a stretch of nearly 200 river miles (322 km). Figure 1-1 presents a map of the general site location and the Hudson River drainage basin.

In December 1990, USEPA issued a Scope of Work (SOW) for reassessing the No Action decision for the Hudson River PCB site. The scope of work indicated that the Reassessment RI/FS would be conducted in three phases:

- Phase 1 - Interim Characterization and Evaluation;
- Phase 2 - Further Site Characterization and Analysis; and
- Phase 3 - Feasibility Study.

In August 1991, USEPA issued a Phase 1 Report describing the results of Phase 1 studies (USEPA, 1991b). The Phase 1 Report contains a compendium of background material, discussion of findings, and preliminary assessment of risks. The Phase 2 work began in December 1991 (upon approval of the earlier Phase 2A Sampling Plan) and is still ongoing. Six reports have been or will be released from this phase of the investigation, specifically:

(1) Phase 2 Report, Volume 2A: Database Report - October 1995;
(2) Phase 2 Report, Volume 2B: Preliminary Model Calibration Report - October 1996;
(3) Phase 2 Report, Volume 2C: Data Evaluation and Interpretation Report (DEIR) - February 1997;
(3A) Phase 2 Report, Volume 2C-A: Low Resolution Sediment Coring Report - July 1998;
(4) Phase 2 Report, Volume 2D: Baseline Modeling Report (BMR) - May 1999;
(5) Phase 2 Report, Volume 2E: Baseline Ecological Risk Assessment (ERA) - August 1999; and
(6) Phase 2 Report, Volume 2F: Human Health Risk Assessment (HHRA) - August 1999.

The Responsiveness Summary for the first three volumes of the Phase 2 Report (Volumes 2A to 2C) was released in December 1998 and the Responsiveness Summary for the Low

Resolution Sediment Coring Report (Volume 2C-A) was released in February 1999. The Database for the Hudson River PCBs Reassessment RI/FS was most recently updated in August 1998 (USEPA, 1998c).

In September 1998, USEPA issued its Phase 2 Ecological Risk Assessment Scope of Work (ERASOW). USEPA solicited public comment on the ERASOW from September 23 to November 2, 1998. On April 30, 1999, USEPA released its Responsiveness Summary for the ERASOW, in which USEPA responded to all significant written comments received on the ERASOW.

The Comprehensive Environmental Response, Compensation, and Liability Act (CERCLA), as amended by the Superfund Amendments and Reauthorization Act of 1986 (SARA), authorizes USEPA to protect public health and welfare and the environment from releases or potential releases of hazardous substances. The National Oil and Hazardous Substances Pollution Contingency Plan (NCP) calls for a baseline risk assessment to determine whether contaminants identified at a site pose a current or future risk to human health and the environment in the absence of any remediation. The results of the baseline risk assessment will be considered in developing remedial alternatives in the FS.

Consistent with USEPA guidance (USEPA, 1997b), the Hudson River PCBs Reassessment baseline ERA calculates the risk to individual receptor species of concern. The receptor species for the site were selected to represent various trophic levels, a variety of feeding types, and a diversity of habitats associated with the Hudson River. The ERA does not characterize injury to, impact on, or threat to every species of plant or animal at the site; such a characterization is beyond the scope of the ERA.

1.2 Site History

During an approximately 30-year period ending in 1977, two General Electric (GE) facilities, one in Fort Edward, NY and the other in Hudson Falls, NY, used PCBs in the manufacture of electrical capacitors. Estimates of the total quantity of PCBs discharged from the two plants to the Hudson River from the 1940s to 1977 range from 209,000 to 1,330,000 pounds (95,000 to 603,000 kg) (USEPA, 1991b). In 1977, manufacture processing and distribution commerce of PCBs within the US was restricted under provisions of the Toxic Substances and Control Act (TSCA).

Many of the PCBs discharged to the river adhered to sediments and accumulated downstream with the sediments as they settled in the impounded pool behind the former Fort Edward Dam (River Mile [RM] 195), as well as in other impoundments farther downstream. Because of its deteriorating condition, the Fort Edward dam was removed in 1973. During subsequent spring floods, PCB-contaminated sediments were scoured and transported downstream. A substantial portion of these sediments were stored in relatively quiescent areas of the river. These areas, which were surveyed by New York State Department of Environmental Conservation (NYSDEC) in 1976 to 1978 and 1984, have been described as PCB *hot spots*. Exposed sediments from the former pool behind the dam, called the "remnant deposits," have been capped by GE under a consent decree with USEPA.

Although commercial uses of PCBs was restricted in 1977, loading of PCBs derived from the GE plants to the Hudson River continued, due primarily to erosion of contaminated remnant deposits, discharges of PCBs via bedrock fractures from the GE Hudson Falls plant, and erosion from contaminated deposits above the water line near the GE Fort Edward plant outfall. Capping of the remnant deposits (in the area of RM195 to RM196) was completed in 1991. In September 1991, high PCB concentrations were again detected in Hudson River water. GE attributed the higher levels to the collapse of a wooden gate structure within the abandoned Allen Mill located adjacent to the GE Hudson Falls capacitor plant (RM ~197) (O'Brien and Gere, 1993). As reported by GE, the gate had kept water from flowing through a tunnel cut into bedrock below the mill, which contained oil-phase PCBs that migrated there via subsurface bedrock fractures. During 1993 to 1995, extensive PCB contamination was detected in water conduits within the mill and approximately 45 tons of PCBs and 3,340 tons of sediment were eventually removed (O'Brien and Gere, 1995). In 1994, GE documented the presence of PCB dense non-aqueous phase liquid (DNAPL) seeps in a dewatered portion of the river bottom at Bakers Falls adjacent to the Hudson Falls plant site. GE instituted a number of mitigation efforts that have resulted in a decline, but not total cessation, of these seeps (O'Brien and Gere, 1995). A more in-depth discussion of external PCB sources, including the GE facilities, the remnant deposits, and other sources in both the Upper and Lower Hudson River, is contained in the Phase 2 Data Evaluation and Interpretation Report (USEPA, 1997a).

USEPA issued a Record of Decision (ROD) for the site in 1984. The ROD selected:

- An interim No Action decision concerning river sediments;
- In-place capping, containment, and monitoring of remnant deposit sediments; and
- A treatability study (at the Waterford Water Works) to evaluate the effectiveness of removing PCBs from the Hudson River for domestic water supply.

In December 1989, USEPA Region II began a reassessment of the No Action decision for the Hudson River sediments based on, among other things, the CERCLA's five-year reevaluation requirement for remedies that leave contamination on site; the specification of future evaluations of the No Action decision contained in the 1984 ROD; and the request from the NYSDEC that USEPA reassess the 1984 No-Action decision.

The 1984 ROD does not address PCB DNAPL seeps near the GE Hudson Falls plant, which were unknown at the time. GE is conducting remedial activities at the GE Hudson Falls Plant Site under an Order on Consent between the NYSDEC and GE.

1.3 Site Investigation and Hudson River Data Sources

Many studies have been conducted to define the nature and extent of PCB contamination in the Hudson River. Studies have included identifying areas of the river with large PCB deposits, examining PCB concentrations in fish, investigating the historical deposition of PCBs, and evaluating various remedial options to address the PCBs. The ERA uses USEPA data collected during the Phase 2 ecological sampling and data collected by other sources. These data are identified and discussed below. All PCB data used in this report are contained in the Database for the Hudson River PCBs Reassessment RI/FS Release 4.1 (USEPA, 1998c) and are described in more detail in Chapter 3.

1.3.1 USEPA Phase 2 Data

An ecological sampling program was conducted in August 1993 to obtain data for this assessment. The ecological field sampling effort collected collocated surficial sediment (0 to 5 cm), benthic invertebrates, and fish for PCB congener-specific analysis at 19 locations in the Upper Hudson River and the Lower Hudson River. Benthic invertebrates were identified and counted to provide data for a community-level analysis. The fish analyzed for the risk assessment were collected by NYSDEC and the National Oceanic and Atmospheric Administration (NOAA). A detailed description of the sampling stations and the ecological field sampling effort are provided in Appendices A and B, respectively.

Data from other USEPA Phase 2 investigations conducted as part of the Hudson River PCBs Reassessment are also used in this report. Water column flow-averaged and transect samples collected from 14 stations between April 1993 to September 1993 (USEPA, 1997a) were used to calculate water column concentrations for 1993. High resolution sediment samples (USEPA, 1997a) provided additional information on PCB congener concentrations in Hudson River sediments.

The USEPA Phase 2 dataset was used to characterize water and sediment exposures to the receptors for 1993. It was also used to determine benthic invertebrate and forage fish body burdens. Projected future concentrations of PCBs in the Hudson River were calculated from the results of the Baseline Modeling Report (USEPA, 1999c). The models presented in that report were used to estimate water, sediment, benthic invertebrate, and fish concentrations for the period 1993 - 2018. The water, sediment, benthic invertebrate, and fish concentrations were used in exposure models to estimate dietary doses for the mammalian and avian receptors.

1.3.2 NYSDEC/NOAA Data

NYSDEC and NOAA collected resident fish at 16 of the ecological sampling stations in 1993 for PCB congener-specific analysis. Between three and ten fish were sampled at each location in this program. In 1995, NOAA conducted an additional study to build on the congener data and the historical database for resident fish established in the 1993 study (NOAA, 1997a). These additional studies collected between three and five fish per sampling location. Congener concentrations were measured in resident fish and juvenile striped bass collected in spring and fall at the same locations along the river. Data from the 1993 and 1995 NYSDEC/ NOAA fish collections are used in this evaluation.

NYSDEC has conducted historical monitoring of total PCB concentrations in Hudson River fish since the 1970s. Fish were collected on an annual basis from 1975 to 1988. In 1988, fish sampling frequency shifted to biannual collections. NYSDEC historical data provide Aroclor 1016, 1254, and in some years Aroclor 1221 and 1242 concentrations. Typically, approximately 20 fish samples are available from four or five locations in the river (RM189, RM168, RM152, RM113 and sometimes RM155 or RM67). In addition to PCB data, NYSDEC also provided data on the distribution of biological receptors covered in this report.

1.3.3 United States Fish and Wildlife Service (USFWS) Data

The USFWS has studied PCB congener concentrations in tree swallows breeding in the Upper Hudson River (McCarty and Secord, 1999; USFWS, 1997). USFWS tree swallow data and other information on biological receptors are included in this report. Database Release 4.1 also presents a sample for a mallard (adult hen and egg) collected from river mile 173 (Saratoga National State Park) during 1995.

1.3.4 GE Data

GE has conducted many studies on PCB contamination in the Hudson River. In 1998 GE commissioned a report on macroinvertebrate communities and diets of selected fish species in the Upper Hudson River (Exponent, 1998). This study was used in conjunction with other research to characterize dietary preferences for the fish receptors examined in this report.

1.3.5 Other Data Sources

The FISHRAND bioaccumulation food chain model used data on water column invertebrates from NYSDOH studies done as part of the Hudson River PCB Reclamation Demonstration Project (Simpson et al., 1986). NYSDOH samples were analyzed for Aroclors 1016 and 1254. Total PCB values were obtained by summing the individual values for these two Aroclors.

1.4 Technical Approach and Ecological Assessment in the Superfund Process

The Ecological Risk Assessment is part of a focused evaluation directed specifically at reassessing the "No Action Decision" related to the presence of PCBs in Hudson River sediments. This reassessment is required under the CERCLA five-year re-evaluation for no action alternatives. The reassessment which was initiated in 1989 consists of three phases: interim characterization and evaluation (phase 1); further site characterization and analysis (phase 2); and a Feasibility Study (phase 3). This Ecological Risk Assessment is being completed as part of Phase 2.

As part of reassessing the "No Action Decision", two major key technical questions were identified pertinent to ecological issues:

- What are the ecological risks associated with PCBs in sediments under the current "no action" baseline conditions?

- How will these baseline risks change in the future if "no action" is taken with respect to deposits of PCBs in the sediments?

The primary objective of this ERA is to answer these questions in order to support the needs of the reassessment. Because of the focused nature of the reassessment, a number of technical decisions have been made which serve to structure and focus the ERA. Most of these decisions were reached after discussion among technical team members and with input from technical and managerial personnel from USEPA, NOAA, NYSDEC, USFWS, and the New York State Department of Health (NYSDOH). These technical decisions were also discussed with representatives of General Electric. This section of the report discusses a number of key technical decisions that were made in order to support the goals and objectives of the reassessment.

1. **The baseline ERA considers current and future exposures and risks.** It is well recognized that exposure to PCBs in aquatic systems can be temporally variable. In the case of the Hudson, considerable effort has been focused on how exposure may change over a period of years. Because of the importance of time as a factor in characterizing exposure and risk, this ERA incorporates reasonably foreseeable future conditions under a "no-action" alternative into the assessment. These reasonably foreseeable conditions are represented by models that capture many of the factors influencing the fate and transport of PCBs and thus, future exposure of ecological receptors to PCBs. "Current" exposures are characterized by existing data found in Database Release 4.1 (USEPA, 1998c). "Future" exposures are characterized by the HUDTOX model for water and sediment, and FISHRAND for invertebrates and fish (USEPA, 1999c).

2. **The ERA considers spatial and temporal dimensions at scales that are appropriate for the assessment of local populations and for decision making.** Spatial and temporal dimensions for analyses were selected based on several criteria including: 1) ecological considerations concerning the areas that may be used by local populations of fish and wildlife; and 2) the level of detail that can be resolved practically with available fate and transport models and that can be reasonably supported by the underlying data. In planning the ecological risk assessment, numerous technical discussions were held concerning the degree of resolution needed for sediments (within river segments) as well as the extent of individual river segments (and associated shorelines). Detail was balanced against the extent to which actual exposure conditions were known as well as the degree of resolution that can be practically achieved through modeling future conditions.

3. **The assessment focuses on particular categories of PCBs that can be supported by the available data and are amenable to modeling**. Selection of PCB 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 selected congeners. The "tri and higher" group is expected to include the PCB compounds that are most toxic to fish and wildlife and is therefore considered to reflect a category that captures most of the toxicity associated with PCB compounds. Historical quantitation of PCBs in biota

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was done on an Aroclor basis; an analysis of these data show that the sum of particular Aroclors is equivalent to the Tri+ and higher congeners (USEPA, 1999c, Chapter 4) and that the Tri+ congeners represent total PCBs in biota. Information on selected congeners (i.e., those used as part of the toxic equivalency methodology) is also used to evaluate risk to fish and wildlife.

1.5 Report Organization

This ERA follows *Ecological Risk Assessment Guidance for Superfund, Process for Designing and Conducting Ecological Risk Assessments* (ERAGS) (USEPA, 1997b). The ERAGS guidance is composed of eight steps, as shown in Figure 1-2. The first two steps consisting of screening-level problem formulation, ecological effects evaluation, exposure estimate, and preliminary risk calculations were completed in the Phase 1 Report (USEPA, 1991b). Steps 3 and 4 encompassing further problem formulation, study design and the data quality objectives (DQO) process were addressed in the Final Phase 2 Work Plan and Sampling Plan (USEPA, 1992b) and Step 5, verification of the field sampling design, was completed in the Phase 2B Sampling and Analysis/Quality Assurance Project Plan (USEPA, 1993a). The ecological field sampling program was completed in August 1993.

A revised Scope of Work was issued in 1998 (USEPA, 1998d) to bring the previously released documents up to date with the 1997 ERAGS guidance. In April, 1999, a responsiveness summary was issued to address comments submitted on the Scope of Work (USEPA, 1999b). This ERA covers Steps 6 and 7 of the ERAGS process, analysis of ecological exposures and effects and risk characterization, including an uncertainty analysis. 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.

Due to the nature of the available data, the ecological risk assessment for the Hudson River will follow a deterministic evaluation.

In keeping with ERAGS, the format of this baseline ERA is as follows:

- Chapter 1, the introduction, provides background on the purpose of the report, Hudson River PCBs site history, site investigation and available data, and ecological risk assessment in the Superfund process.

- Chapter 2, problem formulation, presents the contaminants of concern (COCs), the conceptual model, assessment and measurement endpoints, and brief profiles of the receptors of concern.

- Chapter 3, the exposure assessment, discusses observed and modeled PCB concentrations (based on the results of the BMR), identifies exposure pathways for receptors, and selects exposure parameters for each of the avian and mammalian receptors used for food chain modeling.

- Chapter 4, the effects assessment, is divided into two parts. The first part provides an overview of PCB structure and toxicity. In the second half of the chapter, toxicity reference values (TRVs) are selected for each receptor based on laboratory and field studies.

- 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, discusses various uncertainties associated with the assessment and presents a sensitivity analysis on the exposure and risk models.

- 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.

The ERA is presented in three books. Books 1 through 3 address potential current and future risks in the Upper Hudson River (defined as the TI Pool down to Federal Dam) and current risks in the Lower Hudson River. Book 1 contains the report text, Book 2 presents the tables and figures, and Book 3 provides the report appendices. Detailed descriptions of the site, sampling program, and receptor species are contained in the appendices. An addendum will be issued to Book 1 addressing the potential future risks in the Lower Hudson River when the results of the Thomann/Farley model become available. This is anticipated to occur in the Fall of 1999.

Chapter 2

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2.0 PROBLEM FORMULATION

Problem Formulation for a Baseline Risk Assessment is reflected as "Step 3" of the USEPA's Ecological Risk Assessment Process for Superfund (1997b) as shown in Figure 1-2. Problem Formulation establishes the goals, breadth, and focus of the assessment. Receptors are identified and Assessment Endpoints are established. Through Problem Formulation, the questions and issues that will be addressed are defined based on identifiable potentially 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.

For the Hudson River Reassessment RI/FS, Problem Formulation has been an ongoing process that was initiated in the early 1990s (formal meetings began in 1993) and culminated in the ERA Scope of Work that was reviewed by agencies, General Electric, and other interested parties. While not formally referred to as Problem Formulation, most of the issues that are considered during this part of an assessment were discussed with various agency personnel, other agencies (e.g., NOAA, USFWS, NYSDEC), and General Electric. These discussions occurred during a number of technical and public meetings. Much of the discussion involved various aspects of the conceptual model(s) for exposure pathways, the methods by which exposures would be determined, and the selection of receptors.

2.1 Site Characterization

The Hudson River PCBs Reassessment Site is defined as the 200 miles (322 km) of river from Hudson Falls to the Battery in New York Harbor. The Upper Hudson River, in the context of this baseline ERA, is the 40-mile (64-km) stretch from Hudson Falls to Federal Dam (Figure 2-1). Detailed sections of the Upper Hudson River are shown in Figures 2-3A to 2-3H. The Lower Hudson River runs from Federal Dam to the Battery (Figure 2-2), and is distinguished from the Upper Hudson River by different physical and hydrologic regimes.

The Upper Hudson is an entirely freshwater reach of the river that supports a variety of aquatic and terrestrial wildlife. Large quantities of relatively high concentrations of PCBs have been found in the sediments of the Thompson island Pool (TI Pool) (188.5-194) stretch of the Upper Hudson River (e.g., USEPA, 1998b; 1997a) REF FIGURES. Several tributaries, including Snook Kill and Moses Kill, enter the Hudson River at the TI Pool. Griffin Island, on the western side of the river, is also located within the TI Pool.

The Lower Hudson River is tidal, in contrast to the Upper Hudson River, and includes freshwater, brackish, and estuarine habitats. Most of the unique ecological areas in the river (i.e., significant habitats) and several endangered or threatened species are found in the Lower Hudson River (see subchapters 2.6.5 and 2.6.6, respectively).

2.2 Contaminants of Concern

To focus on the charge of reassessing the 1984 No Action decision of the USEPA concerning PCB-contaminated sediments in the Upper Hudson River, the contaminants of concern (COC) are limited to PCBs. This decision is consistent with the overall purpose of the reassessment. 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 this reassessment of that decision. Consistent with that focus, the evaluation examines risks posed by the presence of in-place PCBs in sediments. PCBs are described as individual congeners, Aroclors, and total PCBs in this ERA. Total PCBs are represented by the trichlorinated and higher congeners (designated Tri+) for the purposes of modeling (USEPA, 1999c). Analyses conducted as part of the Baseline Modeling Report show that the trichlorinated and higher PCB congeners approximate total PCBs in biota.

PCBs consist of a group of 209 distinct chemical compounds, known as congeners, that contain one to ten chlorine atoms attached to a biphenyl molecule. Homologue groups are named according to the number of chlorine atoms present (e.g., monochlorobiphenyls have one chlorine atom, dichlorobiphenyls have two chlorine atoms). Most PCBs manufactured were made up of complex mixtures of congeners.

PCBs were used in a variety of products including: dielectric fluids in capacitors and transformers, printing inks, plasticizer in paints, carbonless paper, coolants, lubricants, adhesives, and dusting agents. Their chemical and physical stability and electrical insulating properties accounted for their widespread usage, but these same characteristics make them persistent in the environment. Monsanto Corporation produced more than 95% of the PCBs used in the US and marketed them under the trade name of Aroclor. Aroclor products were sold from 1930 to 1977, when the TSCA ban on PCB sales became effective.

The most widely marketed mixtures included Aroclors 1016, 1221, 1242, 1248, 1254, and 1260. Chlorination levels of PCB formulations differed markedly (Eisler, 1986). Among the Aroclor formulations, the second half of the number indicated the percent of chlorine by weight in the mixture. For example, Aroclor 1242 is 42% chlorine by weight. The exception to this nomenclature is Aroclor 1016 which is 41% chlorine by mass, not 16%. The difference between 1242 and 1016 reflects differences in homologue composition rather than percent chlorine. Pentachlorinated (i.e., five chlorine atoms) and higher homologues comprise approximately 6.5% of Aroclor 1242 in contrast to Aroclor 1016, which has a composition of only about 0.5% pentachlorinated or higher PCBs. It should be noted that the more highly chlorinated PCB congeners are in evidence throughout the river, especially in fish.

Water solubility decreases as chlorination increases, for example at 75 °F (24 °C), the water solubility of Aroclor 1242 is less than 1 mg/L while that of Aroclor 1260 is less than 0.1 mg/L (Mackay et al., 1992). As water solubility decreases, the tendency to accumulate in lipids increases, with the exception of the most highly chlorinated PCBs. Therefore, more highly chlorinated congeners are frequently found in biota. PCBs have been detected as contaminants in a variety of environmental and biological media including air, water, soils, sediments, plants, domestic animals, wildlife, and human adipose tissue, milk, and serum.

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 this ERA (Figure 2-4). In this model, the initial sources of PCBs are releases from the two GE facilities located in Hudson Falls and Fort Edward. PCB releases into the Hudson River began in the 1940s and have continued to date. Releases were reduced after the halt in PCB production in 1977, and over the last few years with the remedial measures taken by GE around the old Allen Mill.

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. High flow events may also increase the bioavailability of contaminants to organisms in the water column (Petty et al., 1993). Organisms moving between the river and shore may also provide a pathway for PCB transfer to the terrestrial ecosystem.

Animals and plants living in or near the 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. The exposure pathways by which these species could be exposed to PCBs are discussed in the following section.

2.3.1 Exposure Pathways in the 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 covered extensively in previously released Phase 2 reports, such as the Baseline Modeling Report (USEPA, 1999c), Data Evaluation and Interpretation Report (USEPA, 1997a), and Low Resolution Sediment Coring Report (USEPA, 1998b). Therefore, the contaminant fate and transport discussion in this report is limited to the role of biological processes and the ecosystems present in the Hudson River.

2.3.1.1 Biological Fate and Transport Processes

Biological fate and transport processes occur when an organism is exposed to a contaminant. Bioaccumulation is the net result when uptake of a chemical by a biological organism exceeds the depuration of the chemical from the organism (NOAA, 1997a). Uptake may occur directly from the water, sediment, soil, and air, or indirectly through the ingestion of food containing the chemical. Bioconcentration is the process by which a chemical is taken up (by absorption only) from water and is accumulated to levels greater than those found in surrounding water. Biomagnification is the increase in tissue concentrations of a bioaccumulated chemical as the chemical passes up through two or more trophic levels (NOAA, 1997a).

Physical characteristics, such as the octanol-water-partition coefficient (K_{ow}), influence the fate of the PCB molecule once it enters an organism. K_{ow} is a measure of the tendency of a substance to partition from the water into the less polar organic solvent octanol (representative of lipid). The higher the K_{ow}, the greater the tendency to partition to lipophilic substances and the greater the bioconcentration, as shown by a higher bioconcentration factor (BCF). The BCF is the ratio of the concentration in the biological tissue to the dissolved water concentration. The less-chlorinated homologue groups are more readily metabolized and/or excreted than the more highly chlorinated congeners because PCBs with few chlorine molecules fit more readily into binding sites of metabolic enzymes. The log K_{ow} increases with percent chlorine, for example the K_{ow} of Aroclor 1242 (42% chlorine) is 5.6 and the K_{ow} of Aroclor 1260 (60% chlorine) is 6.8 (Mackay et al., 1992).

Bioconcentration of Aroclor 1254 in selected species of freshwater and marine organisms varied from 60 to 340,000 times the concentration of the surrounding water (Eisler, 1986). The lowest bioconcentration factor (BCF) was seen in the protozoan *Tetrahymena pyriformis* exposed to Aroclor 1254 for seven days (Cooley et al., 1972 cited in USEPA, 1980), while the highest BCF was seen in the lipid of the rotifer *Brachionus plicatilis* exposed for 45 days (USEPA, 1980). Aroclor 1260 BCFs in the fathead minnow (*Pimephales promelas*) exposed for 250 days ranged from 160,000 to 270,000 (Defoe et al., 1978). Oysters (*Crassostrea virginica*) held for 65 days in seawater solutions containing 0.0055 to 0.06 µg/L of di-, tri-, tetra-, penta-, or hexachlorobiphenyls had BCFs of 1,200 to 4,800 (Ernst, 1984). Uptake was lowest for dichlorophenyls and became progressively higher with increasing chlorination of PCB congeners. Califano et al. (1982) found that Aroclor 1254 bioaccumulation rates in early life history stages of Hudson River striped bass (*Morone saxatilis*) ranged from 34,700 to 114,000. Eisler (1986) reviewed PCB studies and concluded that for all PCBs BCFs were generally higher with increasing exposure period, with increasing PCB concentration, and with increasing chlorination of PCB congeners.

In aquatic species, PCBs taken up through the water column via the gills are absorbed into the circulatory system, which distributes PCBs throughout the body. Unlike terrestrial species that generally are exposed to PCBs via ingestion, aquatic species living in contaminated surface water are exposed continuously to ambient concentrations. In this way, species exposed to low level water concentrations can accumulate large amounts of PCBs (e.g., Barron,1990; Ankley et al., 1992).

Ankley et al. (1992) demonstrated bioselectivity due to specific lipophilic characteristics of different homologue groups in a study of measured concentrations of PCB homologues and total PCBs in field-collected and laboratory fish and oligochaetes. Although the sediment contained mainly trichlorobiphenyls, both the field and lab oligochaetes and fish were tetrachlorobiphenyl dominant, indicating that the less chlorinated homologue groups are readily metabolized and/or excreted. Concentrations of the more highly chlorinated PCB homologues in field oligochaetes were greater or equal to concentrations found in the sediment, while concentrations of less chlorinated congeners were lower apparently due to metabolization or excretion.

Assimilation and depuration of PCB congeners is related to their chlorine content. Generally, as the number of chlorine atoms increases, the maximum uptake also increases. Although the equilibrium uptake of the less chlorinated congeners is reached quickly (within hours in mammals), they are significantly metabolized and/or excreted. More chlorinated congeners, such as hexachlorobiphenyls, can take days to months to reach their maximum storage in fat tissue (USEPA, 1980). Aroclor 1254, can bioconcentrate significantly in a relatively short period of time, as its congeners are poorly excreted. Elimination of PCBs can be influenced by growth, biotransformation, and maternal transfer (Sijm et al., 1992). In a long-term study on the elimination of PCBs in eels (*Anguilla anguilla*) under natural conditions, the half-lives of particular congeners were on the order of years and no elimination was observed for the more chlorinated congeners, mainly hexachlorobiphenyls through octachlorobiphenyls, during the eight-year study (De Boer et al., 1994).

Jones et al. (1989) found that caged fathead minnows in the Upper Hudson selectively accumulated tetrachlorinated and pentachlorinated biphenyls, suggesting a congener-specific bioaccumulation preference. Further information on PCB bioaccumulation can be found in the Baseline Modeling Report (USEPA, 1999c) and PCB toxicity is discussed in detail in Chapter 4 of this report.

2.3.2 Ecosystems of the Hudson River

The Hudson River is home to a wide variety of ecosystems. The ecosystems present in the river differ between the Upper Hudson River and the Lower Hudson River, divided by the Federal Dam at Troy (RM153). The Upper Hudson River is entirely freshwater and non-tidal and, in the context of this baseline ERA, extends from RM153 to RM195. This area includes deeper water environments as well as littoral zones characterized by aquatic vegetation. In contrast, the Lower Hudson River (running from RM153 down to the New York Harbor) is tidal and becomes increasingly saline towards the mouth of the River. The salt front of the Hudson River shows an annual pattern linked to seasonal weather conditions and resulting freshwater runoff (Stanne et al., 1996). In a year with typical precipitation, the salt front reaches Newburgh Bay, 60 miles north of the Battery during summer or early fall. Reduced freshwater runoff allows the salt front to push further north. Spring runoffs and major storms can push the salt front well below the Tappan Zee Bridge, and sometimes south to Manhattan.

The following attributes are used to classify particular reaches of the Hudson River:

- Non-tidal freshwater (RM154 to RM195 above Federal Dam) – this area includes deeper water environments as well as littoral zones characterized by aquatic vegetation.

- Tidal freshwater (from RM153 to ~RM60 from Federal Dam to Newburgh and below) - this area includes deeper water environments as well as littoral zones that are characterized by aquatic vegetation

- Estuarine (RM60 to RM0 from Newburgh to Manahattan) – this zone reflects a steady gradient of salinity from about 0.5 $^{o}/_{oo}$ to 25 $^{o}/_{oo}$. Note that freshwater often occurs as far south as Haverstraw Bay in a typical year. This area also includes deeper water environments as well as littoral zones that are characterized by aquatic vegetation

- Marine – Hudson River discharge to New York harbor and beyond.

- Marsh environments: these include non-tidal freshwater, tidal freshwater, and estuarine to salt marshes.

- Riparian habitats along the border of the entire length of the river.

The Hudson River provides diverse habitats for all trophic levels of the river's ecosystem. Plants, plankton, aquatic invertebrates, fish, amphibians, reptiles, birds, and mammals use the Hudson River for feeding, reproduction, and shelter. In addition to the aquatic communities associated with the Hudson River, riparian, wetland, floodplain, and upland communities are also dependent on the river. These transition zones between aquatic and terrestrial habitat provide pathways for PCB transfer via the food chain or floodplain sediments. PCBs may be intermittently deposited in nearshore areas that are irregularly flooded during high flow events. Areas that are regularly flooded, such as intertidal areas in the Lower Hudson River, are exposed to contaminants in the river on a continual basis.

Animals found in these transition habitats include a diverse assemblage of mammals (e.g., shrews and meadow voles), birds (e.g., passerines, raptors), reptiles, amphibians, and soil invertebrates (e.g., earthworms, burrowing insect larvae). Many river bank and floodplain species depend on prey, such as insects with aquatic larval stages, that use the river and are exposed to river PCBs during part of their life cycle. Animals with a partial aquatic life history have been shown to transport PCBs into terrestrial environments (Larsson, 1984). Upper trophic level avian and mammalian species living in habitats near the river are also exposed to PCBs originating in the Hudson River.

The degree and spatial extent of PCB contamination in floodplain soils has not been investigated. However, over the last 50 years some PCBs have likely been deposited along the Hudson River shoreline. Consistent with the primary focus of the reassessment, the ERA does not quantitatively estimate PCB exposure from floodplain soils, but plants and animals utilizing the Hudson River shoreline are likely exposed to levels lower than aquatic-based exposures.

Vertebrates potentially found in or along the Hudson River are listed in Table 2-1 and Tables 2-3 to 2-6. Table 2-1 provides a list of the fishes found in the Hudson River system and their predominant habitat (i.e., freshwater vs. saltwater). Fish aggregations observed in the Upper Hudson River Estuary (NYSDEC, 1989) are given in Table 2-2. Potential amphibians and reptiles found near the Hudson River are listed in Tables 2-3 and 2-4, respectively.

Many birds use the Hudson River for feeding and breeding. A list of breeding birds of the Hudson River based on Andrle and Carroll (1988) is provided in Table 2-5. Because of the reproductive effects associated with PCBs, breeding individuals are considered to be more vulnerable to the effects of PCBs than non-breeding individuals. Mammals that may potentially be found along the Hudson River are listed in Table 2-6.

This assessment evaluates exposures for selected river segments that are individually large enough to encompass the foraging areas of local populations of fish and wildlife, and provide information at an appropriate scale for different parts of the Hudson that capture changes in spatial concentrations of PCBs. Assessment locations are referred to by a single river mile to correspond with the Baseline Modeling Report (USEPA, 1999c) but actually reflect a wider area. Accordingly, the Upper Hudson River was divided into the following three reaches for the purposes of this assessment:

- RM189- TI Pool;
- RM168 - Stillwater; and
- RM154- Federal Dam.

These reaches were chosen to represent three distinct locations in terms of spatial concentrations of PCBs, and they correspond to the modeling locations (USEPA, 1999c). Receptor species are assumed to utilize a foraging area encompassing several miles on either side of the specific river miles shown above.

Estimates of future sediment and water PCB concentrations at these reaches were made using the HUDTOX model (USEPA, 1999c). Future fish PCB body burdens at these reaches were predicted using the FISHRAND food chain model (USEPA, 1999c).

The Lower Hudson River is an estuary, a semi-enclosed coastal body of water freely connected to the sea. The Hudson River Estuary is a valuable state and local resource, as well as an integral part of the North Atlantic coast environment (NYSDEC, 1998b). Freshwater, brackish, and marine conditions are found in the estuary. 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 (see subchapter 2.6.6). The NYS Natural Heritage Program has identified many areas along the estuary where rare plants, animals, or natural communities are found. The estuary also serves as an important resting and feeding area for migratory birds, such as eagles, osprey, songbirds, and waterfowl (NYSDEC, 1998b).

2.3.3 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/or,
- Uptake via food (including plants).

The early life stages of many species (e.g., frogs, salamanders, and fish) are entirely aquatic allowing for contaminant exposure at sensitive developmental stages through the contact with egg membranes, gills, and skin. PCBs tend to accumulate in the fatty tissue of organisms due to their hydrophobicity. PCBs also accumulate in predators at higher concentrations than would be predicted by surrounding environmental media, such as water and sediment, in a process known as biomagnification. Depending on the position of an organism in the food chain, exposure may be intensified through food sources as organisms consume other organisms that have accumulated PCBs in the lipid portion of their tissues.

The length of time that an organism is exposed to contaminants contributes to the body burden of an organism. An attached organism (e.g., mussels) or an animal with a small home-range, that is permanently in an area of low PCB contamination may have the same body burden as a mobile organism, or an animal with a larger home-range, that spends only a short period of time in a highly contaminated area.

Direct uptake from water. Many aquatic animals breathe through a series of flexible blood- filled filaments known as a gill. A gill passes water over the blood-filled filaments that extract oxygen from it. While the oxygen is being taken in the gills are simultaneously releasing carbon dioxide and wastes back into the water. During this exchange, PCBs may be taken up from the water column into an organism.

For fish, direct uptake of PCBs from the water occurs primarily across the gills, although uptake can also occur across the linings of the mouth and gastrointestinal tract and the sensory organs. No significant evidence exists for absorption from the water column through the epidermis (Shaw and Connell, 1984).

Organisms exposed to PCBs primarily via the water column include lower trophic level pelagic or planktonic species that live suspended or swimming in the water column. Water column uptake could account for PCB concentrations observed in zooplankton, if PCB concentrations were normalized for lipid content of the organism (Clayton et al., 1977).

Uptake from sediment. PCBs tend to bind to fine grained sediments, probably due to the larger surface area (Phillips, 1986) and/or fraction of organic carbon in sediment particles. Bioavailability of contaminants is dependent on a number of factors including contaminant and organic carbon concentrations. Habitat selection of aquatic organisms plays a role in the potential exposure to PCBs in sediments. Organisms that prefer fine-grained sediments may be exposed to higher concentrations of PCBs, particularly in areas with *hot spots*, such as the TI Pool.

Direct contact with and ingestion of contaminated sediment and associated pore water are the primary routes of exposure for benthic infauna that live in close association with or are buried in the sediment. Deposit-feeding organisms that feed by ingesting sediment, such as oligochaetes, also ingest contaminants that are bound to the sediment. Epifaunal organisms living on the surface of the sediment receive exposure from both the sediment and the overlying water.

Equilibrium partitioning (EQP) has been suggested to be the major factor controlling bioaccumulation in sediment-based benthic communities (Bierman, 1990). EQP assumes that chemicals in interstitial water are the major source of toxicity to sediment dwelling organisms and tries to predict chemical concentrations in water from bulk sediment concentrations. EQP is estimated by multiplying the K_{ow} by the percent carbon in the sediment to derive an organic carbon normalized partition coefficient (K_{oc}) which is used in turn to derive a quantifying partition coefficient (K_p). PCBs are continually being released from the sediment into the interstitial or pore water, from which uptake by benthic organisms occurs. EQP does not consider body-wall absorption and ingestion effects.

Bierman (1990) predicted bioaccumulation factors based upon equilibrium partitioning to account for concentrations of hydrophobic organic contaminants in animals at the lower and middle parts of the food chain from the Great Lakes. Animals modeled on field data included oligochaetes, chironomids, amphipods, sculpin, small smelt, and large smelt. Ankley et al. (1992) compared field and laboratory data and confirmed that for oligochaetes, concentrations of PCBs in sediments could be used to predict concentrations of PCBs in organisms, but that for other species, food or ingestion of contaminated food or possibly ingestion of contaminated particles could affect concentrations.

Uptake via food (including plants). Both field studies and modeling efforts indicate that biomagnification through the food chain is an important component of bioaccumulation. For example, the presence of higher chlorinated Aroclor mixtures congeners in fish of the Lower Hudson River suggests a food chain bioaccumulation component (Sloan et al., 1985). In food chain models using existing field data, almost all of the existing PCB body burden in top predators, such as Hudson River striped bass, could be attributed to a food source (Thomann, 1989; Thomann, 1981). In a modeling study of factors influencing PCB accumulation in Lake Michigan trout, transfer through the food chain accounted for up to 99% of the PCB body burden (Thomann and Connolly, 1984). Ingestion of contaminated food was shown as an important factor in the accumulation of PCBs throughout the food web in a freshwater lake (Van der Oost et al., 1988).

Many aquatic receptors consume macrophytes, including submerged aquatic vegetation, and phytoplankton. In addition, many fish species use the areas in and around submerged aquatic vegetation as habitat areas. Exponent (1998a) documented the occurrence and relationship of submerged aquatic vegetation and fish communities during a survey of the Upper Hudson River in 1998.

Macrophytes and submerged aquatic plants can accumulate PCBs through a direct relationship with dissolved concentrations in the water (Gobas et al., 1991; Lovett-Doust et al., 1997a; Swackhamer and Skoglund, 1993) or through root uptake via sediment sources (Richard et al., 1997; Lovett-Doust et al., 1997b). Submerged aquatic vegetation can alter the oxygen content and pH of the water, and has been shown to affect nutrient cycling, sediment deposition, and sequestration of contaminants (Stewart et al., 1992).

2.3.4 Terrestrial Exposure Pathways

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

- Food uptake (including plants);
- Surface water ingestion;
- Incidental sediment ingestion;
- Contact with floodplain sediments/soils; and/or,
- Inhalation of air.

PCBs enter the terrestrial food chain primarily via food uptake of contaminated prey. Surface water ingestion and incidental sediment ingestion may also contribute to the dietary ingestion of PCBs. Terrestrial animals, such as piscivorous birds, mink, otter, raccoon, and little brown bat, may come into contact with contaminated floodplain soils and/or river sediments while burrowing or foraging. All terrestrial animals may inhale volatilized PCBs. As mentioned previously, floodplain soils are not evaluated in this report because this Reassessment RI/FS focuses on contaminated sediments in the Upper Hudson River. The inhalation exposure pathway is not considered any further for the same reason.

Uptake via Food. Uptake via food constitutes the primary PCB exposure pathway for terrestrial animals living in the Hudson River watershed. PCB-contaminated prey include:

- Animals that spend their entire life in the Hudson River, such as fish (e.g., largemouth bass, pumpkinseed) and some aquatic invertebrates (e.g., oligochaetes, amphipods, mollusks);

- Animals that spend a portion of their life cycle in the Hudson River and the remainder on land, such as aquatic insects (e.g., chironomids, odonates, tricoptera);

- Animals that are entirely aquatic but migrate in and out of the Hudson River, such as striped bass and eels; and

- Animals that are entirely terrestrial, but consume contaminated prey (e.g., reptiles, small birds and mammals) that have been exposed to PCBs originating in the Hudson River.

- Macrophytes and terrestrial plant matter in floodplain areas that may have been exposed to PCB-contaminated sediment or water.

Because of the number of ways that PCBs can be transferred from aquatic organisms to terrestrial organisms, there is the potential for dispersal of PCBs to neighboring ecosystems.

Uptake via Surface Water. Terrestrial animals, including vertebrates and invertebrates, may use the Hudson River as a regular or intermittent drinking water source. PCBs present in the surface water are ingested into the organism where they have the potential to accumulate.

Uptake via Sediment Ingestion. Animals that feed near the river, mainly birds and mammals, may ingest sediment during prey capture and ingestion. The quantity of sediment ingested varies according to feeding method and prey selectivity. Once ingested, contaminants in the sediment may be absorbed or retained by an organism or may remain adsorbed to the sediment and be excreted with body wastes. Sediment may also be incidentally ingested during non-feeding related behaviors such as grooming and cleaning.

The metabolic rate of an animal may also affect exposure to PCBs. Species with higher metabolic rates may accumulate higher concentrations of PCBs than species with lower rates. Animals that generate heat to maintain their body temperature, known as endotherms, generally have higher metabolic rates than animals that regulate their body temperature largely by exchanging heat with their surroundings, known as ectotherms. Therefore, endothermic birds and mammals living in terrestrial communities along the river may be even more exposed to PCBs originating in river sediments than ectothermic aquatic organisms, such as fish and amphibians. Similarly, small passerine insectivorous birds or mammals may accumulate PCBs at higher rates than larger piscivorous birds or mammals, so that their PCB body burdens approach those of higher trophic level species.

Contact with Floodplain Sediments/Soils. Consistent with the primary focus of the Reassessment RI/FS, the analysis focuses specifically on the exposure and risk associated with in-place sediments. It is anticipated that these PCBs are most likely to pose greatest future risk to aquatic receptors or terrestrial receptors such as birds and mammals that rely on aquatic receptors for food. While floodplains may be influenced by PCBs in the aquatic environment, the extent to which this regime may be modified in the future by processes involving in-place sediments is expected to be less than exposure within the river itself. A detailed examination of this issue is beyond the scope of the reassessment.

2.4 Assessment Endpoints

Assessment endpoints are explicit expressions of actual environmental values (e.g., ecological resources) that are to be protected (USEPA, 1992a; 1997b). 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). A population is a group of organisms of the same species, generally occupying a contiguous area and which are capable of interbreeding (USEPA, 1989a). A community is composed of an association of species in the same area. Communities interact continuously with the nonliving components of the environment in an ecosystem. Energy and matter flow through ecosystems by means of complex systems known as food chains and food webs. Food chains are hierarchically arranged into trophic levels that generally consist of primary producers (plants), primary consumers (herbivores), secondary consumers (carnivores), and tertiary consumers (top carnivores) (USEPA, 1989a).

In addition to protection of ecological values, assessment endpoints may also encompass a function or quality that is to be maintained or protected. The selection of assessment endpoints is based on:

- Mechanisms of toxicity of PCBs to different groups of organisms;
- Ecologically relevant receptor groups that are potentially sensitive or highly exposed to PCBs; and
- Potentially complete exposure pathways.

The assessment endpoints for this ERA were selected to include direct exposure 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 indirect exposure endpoints at various levels of the food chain to address PCB-related risks at higher trophic levels. The assessment endpoints selected are:

- Benthic community structure as a food source for local fish and wildlife.

- Protection and maintenance (i.e., survival, growth, and reproduction) of:
    - local forage fish populations;
    - local omnivorous fish populations; and
    - local piscivorous fish populations.

- Protection and maintenance (i.e., survival, growth, and reproduction) of local wildlife including:
    - semi-piscivorous/piscivorous, and insectivorous birds;
    - waterfowl; and
    - semi-piscivorous/piscivorous, insectivorous, and omnivorous mammals.

- Protection of threatened and endangered species.

- Protection of significant habitats.

The selected assessment endpoints along with respective measurement endpoints are listed in Table 2-7. These endpoints reflect a combination of values that have been identified by USEPA, NYSDEC, USFWS, and NOAA as being important, as well as habitats or species that have been identified as ecologically valuable. Each assessment endpoint is framed by testable hypotheses or "risk questions", which provide the measurement endpoint(s) that will be used to evaluate each assessment endpoint.

2.5 Measurement Endpoints

Measurement endpoints provide the actual measurements used to evaluate each of the testable hypotheses and to estimate risk. Measurement endpoints are selected to represent mechanisms of toxicity and exposure pathways, and consider:

- The strength of association between the measurement endpoint and assessment endpoint;

- Data quality; and
- Study design and execution.

Strength of association refers to how well a measurement endpoint represents an assessment endpoint. The greater the strength of association between the measurement and assessment endpoint, the greater the weight given to that measurement endpoint in the risk analysis. Measurement endpoints include measured or modeled PCB concentrations in water, sediment, fish, birds, and/or mammals, laboratory toxicity studies, and field observations.

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 factors discussed above. Therefore, it is common practice to use more than one measurement endpoint to evaluate each assessment endpoint, when possible. Measurement endpoints considered in this analysis relative to the assessment endpoints include:

Assessment Endpoint: Benthic community structure as a food source for local fish and wildlife

Does the benthic community structure reflect the influence of PCBs?

Measurement Endpoint 1: Field observations of benthic community abundance and composition in relation to measured PCB concentrations and habitat characteristics.

Do measured and modeled sediment PCB concentrations exceed guidelines for the protection of aquatic health?

Measurement Endpoint 3: Measured and modeled average and 95% upper confidence limit PCB concentrations in sediment compared to sediment benchmarks such as NOAA Sediment Effect Concentrations for PCBs in the Hudson River (NOAA, 1999a), NYSDEC Technical Guidance for Screening Contaminated Sediments (March 1998a), Persaud et al. (1993), Ingersoll et al. (1996), Washington Department of Ecology (1997), and Jones et al. (1997), for protection of aquatic life.

Do measured and modeled PCB water concentrations exceed criteria and/or guidelines for the protection of wildlife?

Measurement Endpoint 2: Measured and modeled average and 95% upper confidence limit PCB concentrations in water (freshwater and saline) compared to NYS Ambient Water Quality Criteria (AWQC) for the protection of benthic aquatic life (NYSDEC, 1998c) or wildlife (NYSDEC, 1998c).

Assessment Endpoint: Protection and maintenance (i.e., survival, growth, and reproduction) of local fish populations (forage, omnivorous, piscivorous)

Do measured and/or modeled total PCB body burdens in local fish exceed benchmarks for adverse effects on fish reproduction?

Measurement Endpoint 1: Measured and modeled median and 95th percentile PCB body burdens in fish for each river segment over 25 years to determine exceedance of effect-level thresholds based on toxicity reference values (TRV) derived in Chapter 4.

Do measured and/or modeled PCB body burdens expressed on a TEQ basis in local fish exceed benchmarks for adverse effects on fish reproduction?

Measurement Endpoint 2: Measured and modeled TEQ-based median and 95th percentile PCB body burdens in fish for each river segment over 25 years to determine exceedance of effect-level thresholds based on toxicity reference values (TRV) derived in Chapter 4.

Do measured and modeled PCB water concentrations exceed criteria and/or guidelines for the protection of wildlife?

Measurement Endpoint 3: Measured and modeled median and 95th percentile PCB concentrations in water (freshwater and saline) compared to NYS Ambient Water Quality Criteria (AWQC) for the protection of benthic aquatic life (NYSDEC, 1998c).

Do measured and modeled sediment PCB concentrations exceed guidelines for the protection of aquatic health?

Measurement Endpoint 4: Measured and modeled average and 95% upper confidence limit PCB concentrations in sediment compared to sediment benchmarks such as NOAA Sediment Effect Concentrations for PCBs in the Hudson River (NOAA, 1999a), NYSDEC Technical Guidance for Screening Contaminated Sediments (March 1998a), Persaud et al. (1993), Ingersoll et al. (1996), Washington Department of Ecology (1997), and Jones et al. (1997), for protection of aquatic life.

What do the 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 Hudson River for each river segment as an indication of the ability of the species to maintain populations.

Assessment Endpoint: Protection and maintenance (i.e., survival, growth, and reproduction) of local insectivorous birds

Do modeled total PCB dietary doses to insectivorous exceed benchmarks for adverse effects on reproduction?

Measurement Endpoint 1: Modeled total average and 95% upper confidence limit PCB dietary doses to the tree swallow to determine exceedance of effect-level thresholds based on toxicity reference values (TRV) derived in Chapter 4.

Do modeled TEQ-based dietary doses of PCBs to insectivorous birds exceed benchmarks for adverse effects on reproduction?

Measurement Endpoint 2: Modeled TEQ-based average and 95% upper confidence limit PCB dietary doses to the tree swallow for each river segment over 25 years to determine exceedance of effect-level thresholds based on toxicity reference values (TRV) derived in Chapter 4.

Do modeled total PCB concentrations in insectivorous bird eggs exceed benchmarks for adverse effects on reproduction?

Measurement Endpoint 3: Modeled total average and 95% upper confidence limit PCB concentrations in tree swallow eggs to determine exceedance of effect-level thresholds based on toxicity reference values (TRV) derived in Chapter 4.

Do modeled TEQ-based PCB concentrations in insectivorous bird eggs exceed benchmarks for adverse effects on reproduction?

Measurement Endpoint 4: Modeled TEQ-based average and 95% upper confidence limit PCB concentrations in tree swallow eggs for each river segment over 25 years to determine exceedance of effect-level thresholds based on toxicity reference values (TRV) derived in Chapter 4.

Do measured and modeled whole water concentrations exceed criteria and/or guidelines for the protection of wildlife?

Measurement Endpoint 5: Modeled and measured average and 95% upper confidence limit PCB concentrations in water (freshwater and saline) compared to NYS Ambient Water Quality Criteria (AWQC) for the protection of wildlife (NYSDEC, 1998c).

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 Hudson River for each river segment as an indication of the ability of the species to maintain populations.

Assessment Endpoint: Protection and maintenance (i.e., survival, growth, and reproduction) of local waterfowl

Do modeled total PCB dietary doses to waterfowl exceed benchmarks for adverse effects on reproduction?

Measurement Endpoint 1: Modeled total average and 95% upper confidence limit PCB dietary doses to the mallard duck to determine exceedance of effect-level thresholds based on toxicity reference values (TRV) derived in Chapter 4.

Do modeled TEQ-based dietary doses of PCBs to waterfowl exceed benchmarks for adverse effects on reproduction?

Measurement Endpoint 2: Modeled TEQ-based average and 95% upper confidence limit PCB dietary doses to the mallard duck for each river segment over 25 years to determine exceedance of effect-level thresholds based on toxicity reference values (TRV) derived in Chapter 4.

Do modeled total PCB concentrations in waterfowl eggs exceed benchmarks for adverse effects on reproduction?

Measurement Endpoint 3: Modeled total average and 95% upper confidence limit PCB concentrations in mallard duck eggs to determine exceedance of effect-level thresholds based on toxicity reference values (TRV) derived in Chapter 4.

Do modeled TEQ-based PCB concentrations in waterfowl eggs exceed benchmarks for adverse effects on reproduction?

Measurement Endpoint 4: Modeled TEQ-based average and 95% upper confidence limit PCB concentrations in mallard duck eggs for each river segment over 25 years to determine exceedance of effect-level thresholds based on toxicity reference values (TRV) derived in Chapter 4.

Do measured and modeled whole water concentrations exceed criteria and/or guidelines for the protection of wildlife?

Measurement Endpoint 5: Measured and modeled average and 95% upper confidence limit PCB concentrations in whole water (freshwater and saline) compared to NYS Ambient Water Quality Criteria (AWQC) for the protection of wildlife (NYSDEC, 1998c).

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 duck species within the Hudson River for each river segment as an indication of the ability of the species to maintain populations.

Assessment Endpoint: Protection and maintenance (i.e., survival, growth, and reproduction) of Hudson River piscivorous bird species

Do modeled total PCB dietary doses to piscivorous birds exceed benchmarks for adverse effects on reproduction?

Measurement Endpoint 1: Modeled total average and 95% upper confidence limit PCB dietary doses to the belted kingfisher, great blue heron, and bald eagle to determine exceedance of effect-level thresholds based on toxicity reference values (TRV) derived in Chapter 4.

Do modeled TEQ-based dietary doses of PCBs to piscivorous birds exceed benchmarks for adverse effects on reproduction?

Measurement Endpoint 2: Modeled TEQ-based average and 95% upper confidence limit PCB dietary doses to the belted kingfisher, great blue heron, and bald eagle for each river segment over 25 years to determine exceedance of effect-level thresholds based on toxicity reference values (TRV) derived in Chapter 4.

Do modeled total PCB concentrations in piscivorous bird eggs exceed benchmarks for adverse effects on reproduction?

Measurement Endpoint 3: Modeled total average and 95% upper confidence limit PCB concentrations in the eggs of the belted kingfisher, great blue heron, and bald eagle to determine exceedance of effect-level thresholds based on toxicity reference values (TRV) derived in Chapter 4.

Do modeled TEQ-based PCB concentrations in piscivorous bird eggs exceed benchmarks for adverse effects on reproduction?

Measurement Endpoint 4: Modeled TEQ-based average and 95% upper confidence limit PCB concentrations in the eggs of the belted kingfisher, great blue heron, and bald eagle for each river segment over 25 years to determine exceedance of effect-level thresholds based on toxicity reference values (TRV) derived in Chapter 4.

Do measured and modeled whole water concentrations exceed criteria and/or guidelines for the protection of wildlife?

Measurement Endpoint 5: Measured and modeled average and 95% upper confidence limit PCB concentrations in water (freshwater and saline) compared to NYS Ambient Water Quality Criteria (AWQC) for the protection of wildlife (NYSDEC, 1998c).

What do the available field-based observations suggest about the health of local piscivorous bird populations?

Measurement Endpoint 6: Available field observations on the presence and relative abundance of piscivorous avian species within the Hudson River for each river segment as an indication of the ability of the species to maintain populations.

Assessment Endpoint: Protection and maintenance (i.e., survival, growth, and reproduction) of local wildlife

Do modeled total PCB dietary doses to local wildlife species exceed benchmarks for adverse effects on reproduction?

Measurement Endpoint 1: Modeled total average and 95% upper confidence limit PCB dietary doses to the little brown bat, raccoon, mink, and otter to determine exceedance of effect-levels based on toxicity reference values (TRV) derived in Chapter 4.

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 average and 95% upper confidence limit PCB dietary doses to the little brown bat, raccoon, mink, and otter for each river segment over 25 years to determine exceedance of effect-level thresholds based on toxicity reference values (TRV) determined in Chapter 4.

Do measured total PCB concentrations in local wildlife species exceed benchmarks for adverse effects on reproduction?

Measurement Endpoint 3: Measured total PCB concentrations in the liver of mink and otter.

Do measured and modeled whole water concentrations exceed criteria and/or guidelines for the protection of wildlife?

Measurement Endpoint 4: Measured and modeled PCB concentrations in water (freshwater and saline) compared to NYS Ambient Water Quality Criteria (AWQC) for the protection of wildlife (NYSDEC, 1998c).

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 the wildlife species within the Hudson River for each river segment 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 fish species exceed benchmarks for adverse effects on fish reproduction?

Measurement Endpoint 1: Modeled total median and 95th percentile PCB body burdens in shortnose sturgeon (fish) using surrogate upper trophic level fish species to determine exceedance of effect-level thresholds based on toxicity reference values (TRV) derived in Chapter 4.

Do modeled TEQ-based PCB body burdens in local threatened or endangered fish species exceed benchmarks for adverse effects on fish reproduction?

Measurement Endpoint 2: Modeled TEQ-based median and 95th percentile PCB body burdens in shortnose sturgeon (fish) using surrogate upper trophic level fish species to determine exceedance of effect-level thresholds based on toxicity reference values (TRV) derived in Chapter 4.

Do modeled total PCB dietary doses to local threatened or endangered avian species exceed benchmarks for adverse effects on reproduction?

Measurement Endpoint 3: Modeled total PCB dietary doses to the bald eagle to determine exceedance of effect-level thresholds based on toxicity reference values (TRV) derived in Chapter 4.

Do modeled TEQ-based PCB dietary doses to local threatened or endangered avian species exceed benchmarks for adverse effects on reproduction?

Measurement Endpoint 4: Modeled TEQ-based PCB dietary doses to the bald eagle to determine exceedance of effect-level thresholds based on toxicity reference values (TRV) derived in Chapter 4.

Do measured and modeled whole water concentrations exceed criteria and/or guidelines for the protection of wildlife?

Measurement Endpoint 5: Measured and modeled PCB concentrations in water (freshwater and saline) compared to NYS Ambient Water Quality Criteria (AWQC) for the protection of wildlife (NYSDEC, 1998c).

Do measured and modeled sediment PCB concentrations exceed guidelines for the protection of aquatic health?

Measurement Endpoint 6: Measured and modeled PCB concentrations in sediment compared to applicable sediment benchmarks such as NOAA Sediment Effect Concentrations for PCBs in the Hudson River (NOAA, 1999a), NYSDEC Technical Guidance for Screening Contaminated Sediments (1998a), Persaud et al. (1993), Ingersoll et al. (1996), Washington Department of Ecology (1997), and Jones et al. (1997), for protection of aquatic life.

Assessment Endpoint: Protection of Significant Habitats

Do measured and modeled whole water concentrations exceed criteria and/or guidelines for the protection of wildlife?

Measurement Endpoint 1: Measured and modeled PCB concentrations in water (freshwater and saline) compared to NYS Ambient Water Quality Criteria (AWQC) for the protection of benthic aquatic life (NYSDEC, 1998c) or wildlife (NYSDEC, 1998c).

Do measured and modeled sediment PCB concentrations exceed guidelines for the protection of aquatic health?

Measurement Endpoint 2: Measured and modeled PCB concentrations in sediment compared to applicable sediment benchmarks such as NOAA Sediment Effect Concentrations for PCBs in the Hudson River (NOAA, 1999a), NYSDEC Technical Guidance for Screening Contaminated Sediments (1998a), Persaud et al. (1993), Ingersoll et al. (1996), Washington Department of Ecology (1997), and Jones et al. (1997), for protection of aquatic life.

Effect level-concentrations are measured by toxicity reference values (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 dietary dose, egg concentration, and/or body burden is described in Chapter 3 and selection of the toxicity reference values is addressed in Chapter 4.

Population-level effects are determined for each receptor species by evaluating the species life-history and the order of magnitude of the toxicity quotients. The results of all measurement endpoints are evaluated in a weight-of-evidence approach. For receptors with small populations, individual-level effects may place the population at risk. Individual-level effects are considered to occur when the toxicity quotient is greater to or equal to one. Populations are based on species ranges along the Hudson River and are not considered as completely isolated groups.

Direct observations of fish and wildlife populations can also provide insight into the risks or impacts associated with historical releases of chemicals to the environment. Such information can be used for retrospective assessments where contamination is expected to either remain the same or decrease.

As with any measurement endpoint, direct observations offer certain strengths and limitations for risk assessment purposes. Direct observations begin with the ecological receptor (population, community, or system) and attempt to determine if that receptor is exhibiting effects (i.e., reduced reproduction). This is often accomplished by comparing the potentially affected receptor to reference or control populations or systems. Temporal information (e.g., trends in abundance or reproductive status) may also be used to evaluate the relationship between the timing of observed effects and the occurrence and timing of the stress. Possible cause and effect relationships are judged using a series of criteria (e.g., Hill's criteria). Observational studies are typically epidemiological-type assessments.

The major strength of observational studies is that the receptor is examined directly and the results have a "real world" feel. People often have higher confidence in information that reflects actual conditions as compared to projections or characterizations that incorporate assumptions.

The major weakness of observational studies is that they may not be a sensitive detector of potentially important environmental effects. This is because natural systems are variable and effects may occur over time scales larger than those captured by the observations. It is also because the receptor may be affected by a variety of factors unrelated to the stressor of interest. Further, in the case of chemical exposures, there may be a number of sources of exposure that cannot be easily appropriately attributed from a direct examination of the receptor alone. Long-term trend data can be helpful in reducing weaknesses associated with using observational approaches.

Because it is often difficult to quantify cause and effect relationships using observational approaches alone, these approaches may not yield data that can be used for predictive purposes. While trend information may be useful for identifying the direction of change, such historical information may not be indicative of future rates of change.

Historical studies of Hudson River fish populations have focused primarily on the Lower Hudson River, primarily in support of power plant impact studies (Klauda, et al. 1988; Beebe and Savidge, 1988). These quantitative studies have been conducted since the late 1960s and were extensive during the 1970s. In contrast, studies in the upper river (above the Federal Dam) are relatively limited but include collection of fish for examination of PCB body burden analysis.

Population-level information on wildlife bordering the Hudson is also limited. There are many observations related to the presence of species in different areas of the Hudson but relatively little data on population trends or population parameters. What information is available is summarized in the risk characterization chapter.

2.6 Receptors of Concern

Potential adverse effects are evaluated for selected receptor species that represent various trophic levels living in or near the Hudson River. These receptors are used to establish assessment endpoints for evaluation of risk. Species higher in the food chain (e.g., piscivores, carnivores) generally demonstrate higher concentrations of PCBs than species at lower levels (e.g., herbivores, detritivores) (Figure 2-4). In addition, birds and mammals that live and feed along the river would be expected to have higher exposures than upland species.

Receptors were selected to represent different trophic levels, a variety of feeding types, and a diversity of habitats (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 Hudson River. Species were selected based on:

- Species sensitivity to PCBs;
- Societal relevance of selected species;
- Discussions with agency representatives; and,
- Comments received on the ERA SOW.

A summary of the species selected to serve as receptors and other groups/species that inhabit similar habitats along the Hudson River but were not selected as receptor species for this ERA is provided in Table 2-8. Receptor species were chosen to represent the various groups of animals (described in the following subchapters) feeding at different trophic levels.

While various fish species are identified and evaluated, the assessment is applicable to a broad range of animals. The selected species serve primarily as recognizable surrogates for the hundreds of different species that may be exposed to PCBs in Hudson River sediments. An assessment of exposure to each and every species would not be practical. However, if the range of exposures can be captured by a subset of species that fill various ecological niches, then confidence can be achieved that lesser known or recognizable species have been adequately considered.

2.6.1 Macroinvertebrate Communities

Benthic macroinvertebrate communities that provide a source of food for fish and wildlife are evaluated. The Upper Hudson River benthic macroinvertebrate community is composed of freshwater invertebrates, dominated by groups such as isopods, oligochaetes, and chironomids. The Lower Hudson River invertebrate community has a greater diversity of organisms because of the range of salinities found there. The upper reaches of the Lower Hudson above RM50 are dominated by freshwater arthropods and oligochaetes. The middle reaches from RM25 to RM50 have a mixture of freshwater and marine forms and the lower reaches below RM25 support a typical marine assemblage including marine oligochaetes, polychaetes, and crustaceans. Profiles of the dominant macroinvertebrate species/groups found in Hudson River the are provided in Appendix C.

2.6.2 Fish Receptors

The Hudson River is home to over 200 species of fish (Stanne et al., 1996). Eight fish species, representing a range of trophic levels, are evaluated in the ERA (Tables 2-8 and 2-9; see Appendix D for profiles). These species feed on a variety of prey and are divided into forage fish, piscivorous/semi-piscivorous fish, and omnivorous fish. These fish species are selected as surrogates to 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. The fish species selected as receptor species include:

- Spottail shiner *(Notropis hudsonius)*;
- Pumpkinseed (*Lepomis gibbosus)*;
- Brown bullhead *(Ictalurus nebulosus)*;
- White perch (*Morone americana)*;
- Yellow perch (*Perca flavescens)*;
- Largemouth bass (*Micropterus salmoides)*;
- Striped bass (*Morone saxatilis)*; and,
- Shortnose sturgeon (*Acipenser brevirostrum).*

Fish species were selected for modeling based on a consideration of ecological risk as well as a consideration of human health risk. This list of fish species was reviewed by personnel from various state and federal agencies and discussed with representatives from GE. Several criteria were applied for selecting fish species and these were discussed with personnel from the various agencies.

Lower trophic level forage fish, such as the spottail shiner *(Notropis hudsonius);* and pumpkinseed (*Lepomis gibbosus*), feed primarily on invertebrates, plants, and detritus. Omnivorous fish, such as the brown bullhead *(Ictalurus nebulosus*) and shortnose sturgeon (*Acipenser brevirostrum*), feed indiscriminately upon benthic organisms, emergent vegetation, and, in some cases, small amounts of other fish. Yellow perch (*Perca flavescens*), and white perch (*Morone americana*), are considered semi-piscivorous in that they consume primarily invertebrates but will consume small amounts of other fish.

Fish that generally feed primarily on other fish (piscivorous), include the largemouth bass (*Micropterus salmoides*), and striped bass (*Morone saxatilis*). These fish generally feed at higher trophic levels than forage fish. Detailed profiles of the fish species are found in Appendix D.

2.6.3 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 (Tables 2-5, 2-8, 2-9). Detailed life history profiles of the species listed below are provided in Appendix E.

- Tree swallow (*Tachycineta bicolor*);
- Mallard (*Anas platyrhychos*);

- Belted kingfisher (*Ceryle alcyon*);
- Great blue heron (*Ardea herodias*); and,
- Bald eagle (*Haliaeetus leucocephalus*).

The tree swallow is a migratory bird that breeds along the Hudson River. It has steely blue-green-black coloration above and is clear white below. It averages 5 to 6 in (13 to 15 cm) in size. USFWS studied the uptake of PCBs and their affect on nesting colonies along the Upper Hudson River (Secord and MCarty, 1997). As an aerial insectivore, the Hudson River tree swallow feeds primarily on flying insects during the breeding season.

The mallard is a surface-feeding duck that feeds by dabbling and filtering through sediments for food. The male is recognized by his glossy-green head and white neck ring, while the female is a mottled brown. They average 20 to 28 in (50 to 70 cm). Mallards feed primarily on aquatic vegetation, seeds, and aquatic invertebrates. In spring, females shift from a largely herbivorous diet to a diet of mainly invertebrates to obtain protein for their prebasic molt and then for egg production. The animal diet continues throughout the summer as many females lay clutches to replace destroyed nests. Ducklings also consume mainly aquatic invertebrates, particularly during the period of rapid growth. The mallard is a year-round resident of the Hudson River (Stanne et al., 1996).

The belted kingfisher is a medium-sized bird, measuring about 13 in (33 cm) (Peterson, 1980). It is blue-gray with a ragged bushy crest and broad gray breastband. It generally feeds on fish that swim near the surface or in shallow water (USEPA, 1993b). The kingfisher may also feed on crayfish, and in times of food shortages it can feed on a variety of invertebrates and vertebrates. Kingfishers nest in burrows that they excavate in embankments. Only a small number of kingfishers spend the winter near the Hudson.

The great blue heron is the largest wading bird found along the Hudson River. It can stand over 4 ft high (ave. 42 to 52 in) with a wing span of 6 to 7 ft. It has a blue-gray color and adults are white about the head. Their long legs, necks, and bills are adapted for wading in the shallow water and stabbing prey. Fish are the preferred prey of great blue herons, but they also eat amphibians, reptiles, crustaceans, insects, birds, and mammals (USEPA, 1993b). There are currently two breeding colonies along the Hudson River, one in the Upper Hudson River and one in the Lower Hudson River. Other great blue herons may feed along the Hudson River, either during migration or as part of feeding forays from other breeding colonies.

The adult bald eagle is a distinctive bird with a white head and white tail. In 1997 the status of the bald eagle was changed from a federally-listed endangered species to a federally-listed threatened species. The bald eagle ranges in size from 30 to 43 in (75 to 108 cm), with females being larger in size than the males. Bald eagles are opportunistic feeders; taking advantage of whatever food source is most abundant and easy to scavenge or capture (USEPA, 1993b). They feed on a variety of prey including small birds, mammals, fish, and carrion. Bald eagles build large stick nests near the water. There is substantial use of the Hudson River by overwintering bald eagles. NYSDEC has used satellite tracking to follow bald eagles along their migration routes (Nye, 1999). In conjunction with the eagle tracking, NYSDEC is trying to define essential habitat areas and chemical contaminant loads in both eagles and prey along the Hudson River (Nye, 1999).

Avian species were selected based on discussions with representatives from the state, NOAA and USFWS. Inclusion of species was also guided by comments received on the ERA Scope of Work. Selection was also based on experience at other sites with respect to what is known concerning the sensitivity of species to PCBs. The societal relevance of selected species was also considered.

2.6.4 Mammalian Receptors

The potential mammalian receptors found along the Hudson River also represent various trophic levels and habitats (Tables 2-6, 2-8 and 2-9). The four mammals selected to serve as representative receptors in this assessment are:

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

Detailed profiles of these species are provided in Appendix F.

Bats in New York State feed entirely on insects (NYSDOH, 1997). Some of their prey, such as aquatic invertebrates, spend the first part of their lives in water bodies, such as the Hudson River, where they would be exposed to PCB contamination via sediments and the water column. Little brown bats are nocturnal and feed in open forest canopies, open shorelines, and basins of rivers, lakes, streams, and wetlands.

The raccoon is a medium-sized opportunistic omnivore commonly found throughout North America. Raccoons exploit seasonally abundant food including aquatic invertebrates, fish, berries, fruit, or refuse. Although smaller prey items are preferred, raccoons can catch and feed upon larger prey, such as waterfowl and small mammals, and are significant waterfowl egg predators (Doutt et al., 1977).

The mink is a small carnivore that is widely distributed throughout North America. Generally, mink are opportunistic in their feeding habits and prey varies according to seasonal abundance of prey and habitat. They feed on a variety of prey including fish, aquatic invertebrates, and small mammals.

The river otter is a medium-sized carnivore that has historically lived in or near water bodies throughout North America. Otters feed primarily on fish and supplement their diet with aquatic invertebrates (particularly crayfish), birds, mammals, and turtles. Prey depends on availability and ease of capture. River otters are primarily nocturnal, but may be active in the early morning and late afternoon in remote areas. They are active all winter except during the most severe periods, when they take shelter for a few days. Otters belong to the same family as the mink (Mustelidae) and based on a study of the bioaccumulation of PCBs in closely related mustelid species are probably at least as sensitive to PCBs as mink (Leonards et al., 1998).

Wildlife species were selected based on discussions with representatives from the state, NOAA and U.S. Fish and Wildlife Service. Inclusion of species was also guided by comments received on the ERA Scope of Work. Selection was also based on experience at other sites with respect to what is known concerning the sensitivity of species to PCBs. The societal relevance of selected species was also considered. Experience has shown that this last factor is extremely important for providing a basis for decision making that is ultimately acceptable to the public.

The selected mammalian species serve primarily as recognizable surrogates for the many different species that may be exposed to PCBs in Hudson River sediments. An assessment of exposure to each and every species would not be practical. However, if the range of exposures can be captured by a subset of species that fill various ecological niches, then confidence can be achieved that lesser known or recognizable species have been adequately considered.

2.6.5 Threatened and Endangered Species

The federal Endangered Species Act (16 USC Section 1531, et seq.) divides animals and plants in danger of extinction into two categories, "threatened" and "endangered." Endangered species are faced with imminent extinction. Threatened species are in less danger, but require special protection to maintain their populations. There is also a category of species of special concern. These species have no legal protection but are listed because the stability of their populations is unknown. The USFWS encourages government agencies and appropriate parties to consider these species during evaluations.

New York State also maintains separate lists of animals and plants that are considered endangered, threatened, or of special concern at the state level.

All threatened, endangered and special concern species mentioned in this section are found in and along the Lower Hudson River, while some of them (e.g., bald eagle and peregrine falcon) also inhabit the Upper Hudson River Valley. Profiles of threatened and endangered species found in and along the Hudson River are provided in Appendix G.

The habitats of the Hudson River support a number of rare plant species. NYS-listed threatened plant species found along the Hudson River include estuary beggar-ticks (*Bidens bidentoides*), golden seal (*Hydrastis canadensis*), heartleaf plantain (*Plantago cordata*), southern yellow flax (*Linum medium var. texanum*), and swamp cottonwood (*Populus heterophylla*). NYS-listed endangered plant species found in its vicinity are American waterwort (*Elantine americana*), saltmarsh bulrush (*Scirpus novae-angliae*), and water pigmyweed (*Crassula aquatica)*. NYS rare plant species of special concern found include Bicknell's sedge (*Carex bicknelli*), clustered sedge (*Carex cumulata*), Davis' sedge (*Carex davisii*), false hop sedge (*Carex lupiformes*), glaucous sedge (*Carex Flaccosperma var. glaucodea*), Illinois pinweed (*Lechea racemulosa*), marsh straw sedge(*Carex hormathodes*), mock-pennyroyal (*Hedeoma hispidum*), Schweinitz's flatsedge(*Cyperus schweinitizii*), slender crabgrass (*Digitaria filiformis),* smooth bur-marigold (*Bidens laevis)*, spongy arrowhead (*Sagittaria calycina var. spongiosa),* swamp lousewort (*Pedicularis lanceolata*), violet lespedeza (*Lespedeza violacea),* and weak stellate sedge (*Carex seorsa).* Furthermore, two federal species of special concern, handsome sedge (*Carex formosa)* and micrantherum (*Micranthemum micranthemoides*) are found in the vicinity of the Hudson River.

The Karner blue butterfly (*Lycaeides melissa samuelis*), a federal and NYS-listed endangered species, is the only protected invertebrate found along the Hudson River.

The shortnose sturgeon (*Acipenser brevirostrum*) is a federal and NYS-listed endangered species found in the Hudson River. No threatened fish species or fish species of special concern are found in the Hudson River.

NYS-listed threatened reptiles found along the Hudson River include Blanding's turtle (*Emydoidea blandingii*) and the timber rattlesnake (*Crotalus horridus*). The bog turtle is also a federal-listed threatened species. NYS-listed endangered species of herpetofauna found along the Hudson River are the northern cricket frog (*Acris crepitans*) and bog turtle (*Clemmys muhlenbergii*). NYS reptile species of special concern found in and near the Hudson River are spotted turtle (*Clemmys guttata*), wood turtle (*Clemmys insculpta*), diamondback terrapin (*Malaclemys terrapin*), and fence lizard (*Sceloporous undulatus*).

The Hudson River Valley is home to many bird species, including a number of threatened and endangered species and species of special concern. The bald eagle (*Haliaeetus leucocephalus*) is a federal-listed threatened species and a NYS-listed endangered species. The osprey (*Pandion haliaetus*), northern harrier (*Circus cyaneus*), and red-shouldered hawk (*Buteo lineatus)* are NYS-listed threatened species found in the Hudson River Valley. The peregrine falcon (*Falco peregrinus*) is listed as endangered by both the federal and NYS governments. NYS species of special concern found in the vicinity of the Hudson River are the least bittern (*Ixobrychus exilis*), Cooper's hawk (*Accipiter cooperii*), upland sandpiper (*Bartramia longicauda*), short-eared owl (*Asio flammeus*), barn owl (*Tyto alba*), king rail (*Rallus elegans*), common nighthawk (*Chordeiles minor*), eastern bluebird (*Sialia sialis*), grasshopper sparrow (*Ammodramus savannarum*), and vesper sparrow (*Pooecetes gramineus).*

The only federal-listed mammal known to occur along the Hudson River or within one mile of it is the endangered Indiana bat (*Myotis sodalis*) (USFWS, 1999). The eastern woodrat (*Neotoma magister*) is a NYS-listed endangered mammal that has been sighted along the Hudson River. There are no federal or State-listed threatened mammals or mammals of special concern found along or in the vicinity of the Hudson River.

This ERA evaluates risks to threatened and endangered species based on threatened and endangered receptors that are being evaluated for other assessment endpoints. The sustainability of Hudson River piscivorous birds, including the bald eagle (*Haliaeetus leucocephalus*), is being evaluated as an assessment endpoint (see subchapters 2.4 and 2.5). The sustainability of Hudson River omnivorous fish, including the shortnose sturgeon (*Acipenser brevirostrum*), is being evaluated as another assessment endpoint. Therefore, these two species were used to represent risks to threatened and endangered species found along the Hudson River.

In many ERAs conducted at Superfund Sites, it is assumed that the assessment of risks to threatened and endangered species will be preformed by Natural Resource Trustee agencies. While this is also true for the Hudson River, various agencies (including trustee agencies) requested that these species be considered within the ERA.

2.6.6 Significant Habitats

All portions of the Hudson River have value for plants and animals. However, 34 specific sites in the tidal portion of the 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, listed in Table 2-11 (NYSDOS, 1990). These areas are unique, unusual, or necessary for continued propagation of key species. Four of these areas comprise the Hudson River National Estuarine Research Reserve (NERR), administered by NYS in partnership with NOAA. The following NYSDOS-designated significant habitats were sampled during the 1993 Ecological Sampling Program:

- Stockport Flats (NERR) - RM123;
- Tivoli Bays (NERR) - RM100;
- Iona Island (NERR) - RM40;
- Piermont Marsh (NERR) - RM24;
- Shad Island- RM135;
- Roger's Island - RM118;
- Esopus Meadows- RM88; and
- Moodna Creek - RM58.

The significant habitats found along the Hudson River are home to many species, including rare and endangered species. Rare ecological communities and areas of concern (Table 2-11) often form part or all of the areas considered to be significant habitats.

Chapter 3

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 expressed as total PCBs (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.

- Observed USEPA and NOAA fish, sediment, water and benthic invertebrate data from the 1993 sampling program, and NYSDEC fish data from 1993 - 1996 were used to estimate exposures from each of these media to receptors of concern on a total PCB basis (expressed as Tri+). These data were used to estimate fish body burdens and dietary doses to the avian and mammalian receptors.
- Individual observed mink and otter liver concentrations from the 1982 NYSDEC sampling program as described in Foley et al. (1988) are used to compare to toxicity reference values.
- Observed tree swallow and mallard data from the 1993 and 1995 USFWS sampling program are used to derive field-based toxicity reference values, and ratios of egg to emergent aquatic insects are used to determine egg/benthos biomagnification factors. These egg/benthos biomagnification factors are applied to measured or modeled benthic invertebrate concentrations to obtain PCB concentrations in the eggs of the avian receptors.
- Modeled sediment and water concentrations from the HUDTOX model and fish body burdens from the FISHRAND model (USEPA, 1999c) were used to estimate dietary doses to the avian and mammalian receptors for the period 1993-2018. The modeled fish body burdens were used 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 the following methods:

- 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;

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- Using dioxin-like toxic equivalents (TEQs) exposure concentrations based on toxic equivalency factors (TEFs) in a series of body burden, dietary dose and/or egg concentration models; and,

- Using a principal components analysis.

The first two approaches involve constructing exposure models to estimate body burdens, dietary doses, and/or egg concentrations to compare to the toxicity reference values derived in Chapter 4. The final approach provides perspective on the factors contributing to observed variability in fish concentrations, and is used to highlight pertinent issues of spatial and temporal scale relative to exposure.

Subchapter 4.1.3 describes the toxicological basis and the specific PCB congeners used in the TEQ approach. TEQ-based concentrations for the 1993 dataset are estimated by multiplying the individual TEQ congener concentrations by the appropriate TEF and summing the individual congener TEQs. Future TEQ concentrations are estimated according to the procedure described in the next section.

3.1 Method to Determine Toxic Equivalencies (TEQ)

3.1.1 Data Quality Issues for TEQ Congeners

The TEQ congeners (found in Table 4-2) include: BZ#77, BZ#126, BZ#169, BZ#105, BZ#114, BZ#118, BZ#123, BZ#156, BZ#157, BZ#167, and BZ#189. Of these congeners, BZ#118 was explicitly evaluated in the detailed data usability conducted for the ecological program (Appendix I). The data usability report (Appendix I) for the ecological sampling program (sediments, fish, and invertebrates) prepared by Gradient Corporation focused on the 12 "principal" congeners; i.e., BZ#1, BZ#4, BZ#8, BZ#10, BZ#18, BZ#19, BZ#28, BZ#52, BZ#101, BZ#118, BZ#138, and BZ#180.

Of the 11 other TEQ congeners, one - BZ#81 - was not analyzed or reported by Aquatec. Of the remaining 10 TEQ congeners, two (BZ #169 and 114) were "non-target" congeners, one (BZ#156) is an "additional calibrated congener", and the remaining seven (BZ#77, BZ#126, BZ#105, BZ#123, BZ#157, BZ#167, and BZ#189), as well as BZ#118, are target congeners. Quantitation of the two non-target congeners is therefore estimated in all samples (but is appropriate for comparison of concentrations of either of those congeners with data for that congener in other samples analyzed by Aquatec), since no calibration standards were analyzed for these two congeners.

Four of the TEQ congeners (BZ#77, BZ#105, BZ#118, and BZ#126) were part of the suite of matrix spike compounds. No issues specific to any of these congeners were noted; although it was noted that recoveries were uniformly high in one of the invertebrate sample groups.

BZ#77 was one of the congeners for which more than 10% of the sediment data were rejected due to dual column imprecision (13% of the sediment BZ#77 data were rejected). Twelve percent of the BZ#189 data were rejected in the invertebrate samples for the same reason. No other TEQ congeners were rejected in any of the three ecological media at frequencies of 10% or more.

Results for BZ#118 were qualified in a small percentage (less than 2%) of the fish samples (both the USEPA and NOAA fish) due to blank contamination, and also in two of the invertebrate samples. No other TEQ congeners were qualified in any of the other samples for blank contamination.

Other than noted above, there were no issues associated with TEQ congener data quality evident from the data usability report. It is noted that, overall, a high percentage of the ecological data (62%) were qualified as estimated; however, these data were considered usable for the purposes of the ecological risk assessment. Rejected data, which are not usable, amounted to about 1.6% of the total congener data generated for the ecological program.

There are two important issues in estimating TEQ-based PCB concentrations from the Phase 2 dataset:

1. BZ#81 was not quantitated; and,
2. BZ#126 is typically present at the detection level in fish tissue samples, and because the samples required dilution, detected values are often less than the reported detection level.

As mentioned above, BZ#81 was not evaluated in the analytical program. Because BZ#81 was not quantitated, this congener is excluded from TEQ-based estimates of PCB concentrations. This clearly underrepresents the potential influence of BZ#81 in the overall analysis. This is most significant for the avian receptors, as the TEF for BZ#81 is equal to the TEF for BZ#126 (0.1, the highest TEF for any congener). For fish, the TEF for BZ#81 is an order of magnitude less than the highest TEF (which is also for BZ#126). For mammals, the TEF for BZ#81 is three orders of magnitude lower than the highest TEF (BZ#126) and equal to the TEF for BZ#77.

In addition, BZ#126 is often quantitated at the detection level. For the purpose of this analysis, the reported detection level of BZ#126 was used.

To evaluate the impact of using BZ#126 at the detection level and using BZ#126 as a surrogate for BZ#81, the following analysis was conducted. First, all the TEQ-based fish concentrations were compiled and the individual fish-based TEF applied (setting all non-detects equal to the detection level). These values were then summed and each individual congener expressed as a proportion of the TEQ sum for that sample. The results for each individual sample are presented in Appendix J. Since the USFWS tree swallow dataset quantitated BZ#81, this same procedure was again followed using this dataset (only 1995 was used because the 1994 dataset did not quantitate as many congeners) and again applying the fish-based TEF. Table J-2 in Appendix J presents the results obtained by applying the fish-based TEF to the tree swallow

TEQ congener concentrations and expressing the results as proportions of the total TEQ for each individual sample.

Table 3-1 shows the comparison of the TEQ-proportion for each individual congener on an average basis from the fish-based analysis using the Phase 2 dataset (USEPA and NOAA fish data) and the USFWS data. The results presented in this table demonstrate that on a TEQ basis, BZ#77, BZ#81, BZ#105, BZ#118 and BZ#126 comprise nearly 97% of the total TEQ concentration. For the fish-based results, the proportion of BZ#126 (even at the detection level) is much higher than the USFWS-based results, and in fact roughly equal to the sum of BZ#126 and BZ#81 from the USFWS dataset. This analysis shows that it is a reasonable assumption to use the Phase 2 dataset in evaluating TEQ-based exposures. The exact magnitude of the error introduced by the omission of BZ#81 and setting BZ#126 equal to the detection level is not known, but is likely within an order of magnitude at most.

3.1.2 Estimating Future Baseline TEQ Concentrations

The following method was used to estimate future TEQ concentrations using the results from the FISHRAND bioaccumulation model:

1) Divide individual congener concentrations (i.e., BZ#77, BZ#126, BZ#169, BZ#105, BZ#114, BZ#118, BZ#123, BZ#156, BZ#157, BZ#167, and BZ#189) by the Tri+ total PCB concentration for each sample (whole water, dissolved water, sediment, benthic invertebrate, and fish). Non-detects in samples were set equal to the detection level based on the rationale described previously;

2) Next, multiply these fractions by the TEF for each individual congener and biota category (fish, avian, and mammal) and average across the Upper Hudson River, Lower Hudson River, and entire river; and,

3) Finally, sum across the congeners to obtain the TEF weighting factor to apply to future predicted concentrations.

This process provides the fraction of the Tri+ concentration for each medium that is represented by TEQs (Table 3-2). A different fraction is obtained depending on the receptor category (fish, avian, mammalian) and for each of the media (water, sediment, benthic invertebrate, fish, avian, mammalian). This weighted TEF fraction is applied to future Tri+ predictions under the assumption that while absolute concentrations change, the congener distribution is relatively consistent from year to year.

3.2 Observed Exposure Concentrations

This chapter summarizes the data used to develop exposure concentrations for the receptor species of concern. The 1993 USEPA/NOAA Phase 2 dataset was used at each of the sampling locations to obtain measurements for water, sediment, benthic invertebrates, and forage fish. The NYSDEC dataset was used for piscivorous fish. Total PCB concentrations based on observed data in sediment, whole water, benthic invertebrates, and fish are described as averages and 95% UCLs on the mean. Total PCB concentrations are expressed in terms of the Tri+ and higher PCB congeners. TEQ exposure concentrations are estimated by multiplying individual congener concentrations by the appropriate weighted TEF (see Table 4-2) and summing them.

Observed PCB concentrations are best described by lognormal distributions (USEPA, 1999c). Lognormality was determined by log-transforming observed concentrations and running standard normality tests. The formula to estimate 95% upper confidence limits for lognormal distributions is given by Gilbert (1987):

$$UL_{0.95} = Exp\left[\overline{X} + 0.50s^2 + \frac{H_{1-\alpha}S}{\sqrt{n-1}}\right] \qquad \text{Equation 3-1}$$

where:

$\overline{X}$ = log-transformed arithmetic average from the data or model,

s^2 = variance of the natural log-transformed data,

s = sample standard deviation of the natural log-transformed data,

$H_{1-\alpha}$ = $H_{\alpha\text{-}2}$ is a function of the standard deviation of the log-transformed data and the number of samples in the data set. $H_{1-\alpha}$ was taken from a standard table of calculated values (Gilbert, 1987) or linearly interpolated between values given in the table where necessary; and

n = the number of samples in the data set.

3.2.1 Observed Water Concentrations

Water column data were collected at 14 locations in the Hudson River over the course of one year (USEPA, 1998c; 1997a). These locations are not the same as the ecological program sampling locations. Spatially, data were averaged over water column sampling stations to represent a water concentration for a particular reach encompassed by an ecological sampling station, and temporally, this assessment uses summer-averaged water column concentrations of PCBs as the basis for modeling exposure to aquatic organisms and for comparison to water quality benchmarks. For example, water samples collected between April and September at three locations in the TI Pool were used to obtain a TI Pool average water concentration. Stillwater average water concentrations were estimated from water samples collected at RM181.3 and 168.3 during April, June, and August. The area just above the Federal Dam (RM154) was characterized by water samples collected from RM156.5 in April, May, June, July, August, and September. Samples collected in April and August from RM151.7 and 125 were used to obtain average water concentrations for ecological stations at RM143.5 and 137.2. RM122.4, RM113.8, and RM100 were characterized by average water column concentrations over RM125 and RM77 from April and September. The final four ecological stations were characterized by average water column concentrations at RM77 in April and September. Water concentrations are expressed on a whole water basis (particulate plus dissolved) and are shown in Table 3-3. All water samples were above the detection limit. RM77 is just above the saltfront so these concentrations may not adequately reflect concentrations in the more saline waters leading to the mouth of the harbor.

Table 3-3 also provides whole water concentrations of PCBs described as TEQs. Different TEF are applied to the water concentrations depending on whether the receptor is mammalian or avian. The TEF used from Table 3-2 is a weighted TEF from the analysis presented previously. Consequently, separate columns are provided for avian- and mammalian-based TEQ water column concentrations.

3.2.2 Observed Sediment Concentrations

Sediment data were collected at 19 locations in the Hudson River during the 1993 USEPA ecological sampling program (see Appendix B). Sediment samples were taken in the most biologically active zone of 0 to 5 cm (0 to 2 inches). Five samples from each location were analyzed on a PCB congener basis, from which Aroclor, homologue totals, and total PCB concentrations were obtained. Table 3-4 provides average and 95% UCL sediment concentrations for three Upper Hudson River locations and nine Lower Hudson River locations. (Note that data from stations within the TI Pool were combined.)

Table 3-4 also provides observed sediment concentrations described as TEQ. Different TEF are applied to the sediment concentrations depending on whether the receptor is mammalian or avian. The TEF used from Table 3-2 is a weighted TEF from the analysis presented previously. Consequently, separate columns are provided for avian- and mammalian-based TEQ sediment concentrations.

3.2.3 Observed Benthic Invertebrate Concentrations

Data on benthic invertebrate communities and PCB body burdens were collected at the ecological monitoring stations. PCB concentrations were analyzed in benthic invertebrate communities, and for identifiable taxa when sufficient mass was available. Total PCB concentrations are averaged using all samples to obtain exposure point concentrations for fish, birds, and mammals that may be consuming invertebrates as prey items. Statistical tests (t-tests) showed no significant difference in PCB concentrations between benthic invertebrate species; thus, it was appropriate to consider overall benthic invertebrate concentrations as representative of any particular species. The congener analysis presented in Appendix K also supports this assumption. Table 3-5 provides average and 95% UCL benthic invertebrate concentrations used in this analysis.

Table 3-5 also provides observed benthic invertebrate concentrations described on a TEQ basis. Different TEF are applied to the benthic invertebrate concentrations depending on whether the receptor is mammalian or avian. The TEF used from Table 3-2 is a weighted TEF from the analysis presented previously. Consequently, separate columns are provided for avian- and mammalian-based TEQ benthic invertebrate concentrations.

3.2.4 Observed Fish Concentrations

Fish were collected at 16 of the ecological sampling locations along the Hudson River. Only three sampling locations in the TI Pool, selected specifically for the benthic invertebrate community study, were not sampled for fish. Sample sizes are too small to estimate average and 95% UCL PCB concentrations for each species based on the USEPA Phase 2 dataset. Thus, we consider a composite forage fish (less than 10 cm in size) using the USEPA Phase 2 dataset and provide individual species PCB concentrations from the NYSDEC dataset. These PCB concentrations for the composite forage fish are provided in Table 3-6 with avian- and mammalian-based TEQs. Table 3-7 provides wet weight and lipid-normalized concentrations for largemouth bass, brown bullhead, and white and yellow perch for river miles 113, 168, and 189 for the years 1993 through 1996. Figures 3-1 and 3-2 provide wet weight and lipid-normalized average PCB concentrations in several species across several river miles based on the NYSDEC data.

Table 3-8 provides observed striped bass concentrations for several river miles from the NYSDEC sampling program. Striped bass are not typically observed in the Upper Hudson River, although individual fish may be capable of crossing into the Upper Hudson River at Federal Dam.

The observed fish concentrations for all species except pumpkinseed and spottail shiner in both the USEPA Phase 2 and NYSDEC sampling programs are given as standard fillets. Since ecological receptors do not distinguish between standard fillets and whole fish, and toxicity reference values 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, 1997d). For largemouth bass, this ratio is 2.5 and for brown bullhead, the factor is 1.5. These values were discussed with NYSDEC and thought to be comparable to values for Hudson River fish. 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 in this assessment. Note that this is likely to underestimate wet weight concentrations in the whole body but has no effect on lipid-normalized concentrations.

3.2.5 Observed Avian Concentrations

USFWS conducted PCB monitoring in tree swallow eggs and nestlings during 1993 and 1994 (USFWS, 1997). A summary of their results is provided in Table 3-9. One mallard sample from river mile 173 was presented in the USFWS database. There are no other recent direct measurements of PCBs in kingfisher, great blue herons, mallard, or bald eagles. One observation of PCBs in a bald eagle was obtained during 1997 (NY Times, Sept. 17, 1997), but this does not provide enough data with which to assess potential exposures and effects from Hudson River sources. During the early 1980's, NYSDEC conducted some limited monitoring throughout New York State of PCBs in peregrine falcons, heron, a few mallard ducks, and several other species. However, these data are not suitable for assessing potential risks to these species.

NYSDEC is currently studying contaminant loads in bald eagles and their prey along the Hudson River.

3.2.6 Observed Mammalian Concentrations

The New York State Toxic Substances Monitoring Program (1982) and Foley et al. (1988) provide limited data on PCB concentrations in mink and otter livers from three locations within the Hudson River watershed area. These concentrations are summarized in Table 3-9.

3.3 Quantification of PCB Fate and Transport

Models have been developed to describe the fate, transport, and bioaccumulation potential of PCBs in the Upper Hudson River. The HUDTOX model provides sediment and water concentrations (USEPA, 1999c), and the FISHRAND model provides benthic invertebrate, water column invertebrate, macrophyte, and selected fish species concentrations (USEPA, 1999c). 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 95th percentile, 95% of the population is expected to experience the predicted concentration or less than the predicted concentration.

3.3.1 Modeled Exposure Concentrations

Concentrations of PCBs in the Hudson River ecosystem were estimated for the period 1993 to 2018 based on models for sediment and water and aquatic biota (USEPA, 1999c). The models were run using a constant upstream boundary assumption. Under this assumption, nominal amounts of PCB are continually released into the river via the water column from the boundary (above Fort Edward). Model results show that the zero upstream boundary and the constant upstream boundary assumptions yield very similar results (USEPA, 1999c). The constant upstream boundary assumption is considered protective of aquatic health.

3.3.1.1 Modeled Water Concentrations

The HUDTOX model was used to predict whole water and dissolved water concentrations of PCBs for the period 1993 to 2018. Details of specific model assumptions and parameters can be found in the Baseline Modeling Report (USEPA, 1999c). Table 3-10 provides the predicted average and 95% UCL whole water concentrations on a Tri+ total PCB basis.

Table 3-10 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 3-10 by the toxic equivalency weighting factors developed in subchapter 3.2 to describe the proportion of the Tri+ total expressed as a TEQ.

3.3.1.2 Modeled Sediment Concentrations

The HUDTOX model was used to predict sediment concentrations of PCBs for the period 1993 to 2018. Details of specific model assumptions and parameters can be found in the Baseline Modeling Report (USEPA, 1999c). Table 3-11 provides the predicted average and 95% UCL sediment concentrations on a Tri+ total PCB basis.

Table 3-11 also provides the predicted average and 95% UCL sediment concentrations expressed on a TEQ basis. These values were obtained by multiplying the Tri+ predictions in the first six columns of Table 3-11 by the toxic equivalency weighting factors developed in subchapter 3.2 to describe the proportion of the Tri+ total expressed as a TEQ.

3.3.1.3 Modeled Benthic Invertebrate Concentrations

Benthic invertebrate concentrations of PCBs for the period 1993 to 2018 were predicted using a biota sediment accumulation factor (BSAF), given as:

$$BSAF = \frac{C_{Invert}}{C_{Sediment}} \qquad \text{Equation 3-2}$$

where:

BSAF = biota-sediment accumulation factor (µg PCB per g lipid/µg PCB per g organic carbon);

$Conc_{Invert}$ = the concentration of PCB in an organism (µg/g lipid); and

$Conc_{Sediment}$ = the concentration of PCB in sediments (µg/g organic carbon).

Table 3-12 provides the predicted average and 95% UCL benthic invertebrate concentrations expressed on a total PCB basis. Table 3-12 also provides the predicted average and 95% UCL benthic invertebrate concentrations expressed as TEQs. These values were obtained by multiplying the predicted benthic invertebrate concentration by the appropriate TEF for that receptor species from the analysis presented in subchapter 3.2.

3.3.1.4 Modeled Fish Concentrations

Fish concentrations of PCBs for the period 1993 to 2018 were predicted using the FISHRAND model (USEPA, 1999c). Tables 3-13 through 3-16 provide the 25th and 95th

percentile values as well as the median of the predicted distribution for each of the receptor fish species (largemouth bass, brown bullhead, white perch, and yellow perch) expressed on a wet weight basis for Tri+ total PCBs at three locations (TI Pool, RM168, and RM154).

Predictions are not provided for the striped bass or the shortnose sturgeon. Specific bioaccumulation models have not been developed for these species. The striped bass will be modeled in the Lower Hudson River using the results from the Thomann/Farley bioaccumulation model. For this analysis, assuming that individual fish may cross into the Upper Hudson River, largemouth bass and brown bullhead results will serve as order-of-magnitude surrogate fish species to assess potential risks to sturgeon and striped bass. Observed data will be used to compare to toxicity reference values for striped bass for 1993 – 1996.

As described above, the model also predicts standard fillet concentrations in fish. Thus, the wet weight model results were adjusted by a factor of 1.5 for the brown bullhead and 2.5 for the largemouth bass. No factors were available for the white and yellow perch, and no factors were required for the pumpkinseed and spottail shiner as these were modeled on a whole body basis.

To obtain an expected value (mean) and standard deviation from the FISHRAND probabilistic model to use in the estimate of the 95% UCL, the following procedure was used:

1. The model predicts 25^{th}, 50^{th}, and 95^{th} percentiles;
2. These three percentiles provide three points that when plotted against the inverse of the normal cumulative distribution yield a straight line; and,
3. The slope of this straight line is the inverse of the geometric standard deviation.

3.4 Identification of Exposure Pathways

Potential exposure pathways to PCBs for aquatic and terrestrial receptors were identified in subchapters 2.3.3 and 2.3.4, respectively. The exposure pathways considered in the quantitative exposure calculations in this assessment are discussed below.

3.4.1 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 (Thomann et al., 1992). Benthic invertebrates also provide an important food source for demersal (bottom-feeding) fish, such as the brown bullhead, and represent a portion of the diet of other fish species, including largemouth bass and white perch.

Benthic invertebrate concentrations for 1993 are estimated using the USEPA Phase 2 dataset. Predicted benthic invertebrate concentrations for the period 1993 to 2018 are estimated using a BSAF multiplied by the predicted sediment concentration described as:

$$Conc_{Invert} = BSAF \times Conc_{Sediment}$$ Equation 3-3

where: $Conc_{Invert}$ = the concentration of PCBs in an organism (µg/g lipid);
$BSAF$ = biota-sediment accumulation factor (µg PCB per g lipid/µg PCB per g organic carbon); and
$Conc_{Sediment}$ = sediment concentration (µg/g).

3.4.2 Fish Exposure Pathways

Fish are exposed to PCBs in water and sediments both directly as well as indirectly through the food chain. Fish exposure to PCBs is described by a wet weight tissue concentration. NYSDEC data for 1993 – 1996 are used for largemouth bass, brown bullhead, white perch, yellow perch, and striped bass, while the USEPA/NOAA Phase 2 dataset was used to estimate the expected concentration in forage fish (smaller fish). The FISHRAND model was used to estimate wet weight tissue concentrations for all the fish receptor species for the period 1993 – 2018.

3.4.2.1 Surface Water Sources of PCBs

Fish are exposed to PCBs in the water column through respiration, direct dermal contact, and through the food chain. Typically, PCBs are found at relatively low concentrations in the dissolved phase in the water column, due to their low solubility and preferential lipophilic partitioning to suspended matter and sediment. The dissolved phase is believed to control uptake kinetics, with PCBs sorbed to particulate matter or complexed to dissolved organic carbon. Significant levels of PCBs can be detected in tissue of biota living in contaminated areas, particularly in organs that contain high concentrations of lipids (e.g., reproductive and digestive organs). Biota have been shown to bioaccumulate concentrations of PCBs greater than concentrations present in the water-column or sediment (e.g., Ankley et al., 1992; Eisler, 1986), a process likely attributable to a slower depuration rate relative to the uptake rate.

In aquatic species, PCBs taken up through the water column via the gills are absorbed into the systemic circulation system and, depending on the specific congener, typically preferentially sequestered in lipid tissue. Unlike terrestrial species, which are generally exposed to PCBs via ingestion, aquatic species living in contaminated surface water are continuously exposed to low-level ambient concentrations. In this way, species exposed to low-level water concentrations of PCBs can accumulate large amounts of PCBs (e.g., Barron,1990; Ankley et al., 1992).

3.4.2.2 Sediment Sources of PCBs

Bioaccumulation of PCBs from contaminated sediments can occur via several mechanisms, including uptake from the interstitial or overlying water via respiration, direct dermal absorption, ingestion of sediment, or indirectly through the food web. Although dermal absorption is theoretically a pathway of potential concern, this pathway will not be quantitatively assessed due to a lack of appropriate data and models. PCBs in sediments adsorbed to particles and in interstitial water represent the primary source of exposure for benthic invertebrates. In addition, epibenthic species may derive a larger portion of their exposure from overlying water. Sediments also represent an important exposure source for demersal (bottom-feeding) fish, such as the brown bullhead. Fish may also experience indirect exposure to sediments by consuming benthic invertebrates and emergent aquatic insects that have traveled into the water column. Terrestrial receptors are also indirectly exposed to PCBs in sediments by consuming as prey items organisms that experience sediments as their primary route of exposure.

3.4.3 Avian Exposure Pathways and Parameters

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 subchapter 2.3.4). Intake is calculated as an average daily dosage (ADD) value, expressed as mg PCB/kg/day. The ADD from each of the three calculated exposure pathways are summed to develop the total ADD of PCBs from riverine sources. The equation is provided as:

$$ADD_{River} = ADD_{Diet} + ADD_{Water} + Add_{Sediment} \qquad \text{Equation 3-4}$$

where:

ADD_{River} = Potential average daily dosage of PCBs to receptor from Hudson River sources (mg/kg/day);
ADD_{Diet} = Average daily dosage of PCBs via dietary sources of fish and invertebrates (mg/kg/day);
ADD_{Water} = Average daily dosage of PCBs via drinking water (mg/kg/day); and
$ADD_{Sediment}$ = Average daily dosage of PCBs via incidental ingestion of sediments (mg/kg/day).

The direct ingestion of surface water for drinking and the incidental ingestion of sediments are generic exposure pathways that were developed based upon allometric relationships and guidance described in USEPA (1993b) and Nagy (1987). Ingestion rates are derived based upon body weight, free living metabolic rate, and diet composition. Dietary exposure differs between receptors since the percentage of diet derived from the Hudson River, type of prey consumed (e.g., fish or invertebrates), and size selectivity of prey species varies with the receptor. The equations used to calculate intakes for each of exposure pathways are provided below. Parameters used for the tree swallow, mallard, belted kingfisher, great blue heron and bald eagle, are summarized in Tables 3-17 to 3-21.

3.4.3.1 Surface Water Ingestion Pathway

The receptor-specific average daily dosage rate ADD_{Water} (mg/kg/day) is derived as the quotient of the mass of PCBs ingested on a daily basis and the body mass of the species being evaluated:

$$ADD_{Water} = \frac{(PCB_{Surfacewater} \times NWI_{Bird})}{BW_{Bird}} \times (FE) \qquad \text{Equation 3-5}$$

where:

ADD_{Water}	=	Daily dose of PCBs from consuming Hudson River surface water (mg/kg/day);
$PCB_{Surfacewater}$	=	Mean/95% UCL PCB exposure concentration (mg/L) in surface water;
NWI_{Bird}	=	Normalized water ingestion rate (L/day) for avian receptor;
FE	=	Areal forage effort (unitless) as fraction of home or forage range; and,
BW_{Bird}	=	Body weight (kg) of receptor.

Given the size of the Hudson River site, exposure to Hudson River-derived PCB sources (water, sediment, biota) was considered continuous, and the areal foraging effort factor (FE) for all receptors was set to a value of 1.0.

The normalized water ingestion rate (NWI) (L/day) was estimated from the following equation (USEPA, 1993b):

$$NWI_{(Birds)} = (0.0582 * BW^{0.67}) \qquad \text{Equation 3-6}$$

where:

$NWI_{(Birds)}$	=	Receptor specific normalized water ingestion rate (L/day); and
BW	=	Body weight of avian receptor (kg).

PCBs ingested on a daily basis are calculated for both the mean and 95% UCL concentration of PCBs in surface water (mg/L) and the normalized water ingestion rate (L/day).

3.4.3.2 Incidental Sediment Ingestion Pathway

Incidental ingestion of Hudson River sediments by avian receptors may occur through feeding and non-feeding activities, such as cleaning and preening of the feathers. The equation for this pathway is considered on a dry weight basis for evaluation and accounts for only the fraction of the total diet represented by abiotic material (USEPA, 1993b). The incidental ingestion is calculated as:

$$ADD_{Sediment} = \frac{(PCB_{Sediment} \times FS_{Media} \times NIR_{Total})}{BW_{Bird}} \times FE \qquad \text{Equation 3-7}$$

where:

$ADD_{Sediment}$	=	Average/95% UCL daily dose of PCB via incidental ingestion of sediments (mg/kg/day dry wt.);
$PCB_{Sediment}$	=	Mean/95% UCL PCB concentration (mg/kg dry weight) in sediment;
FS_{media}	=	Fraction of abiotic media in diet (%);
NIR_{Total}	=	Total food ingestion (FI) rate (kg/day dry wt); estimated using FI (kg/day) = $0.0582(BW)^{0.651}$ (USEPA, 1993b);
FE	=	Areal foraging effort (1.0); and
BW_{Bird}	=	Body weight (kg) of receptor.

The fraction of incidental sediment ingestion in the diet is specific to each of the avian endpoint receptors. Most incidental ingestion occurs during feeding (Beyer et al., 1994) and the greatest potential for this exposure pathway occurs while feeding on aquatic benthic invertebrates in the river. Therefore, receptors having a diet including an important benthic invertebrate component, such as the mallard, are likely to have higher incidental exposures than species preferring to feed on fish, such as the belted kingfisher, great blue heron, and bald eagle. Incidental sediment ingestion for the mallard, an omnivore that consumes a large percentage of aquatic invertebrates (50%), has been estimated to be 2.0 % (Beyer et al., 1994).

Quantitative estimates of percent composition of sediments in the diet of the tree swallow, belted kingfisher, great blue heron and bald eagle were not available. Therefore, incidental sediment ingestion for these species was estimated based on their feeding patterns. The diet of the tree swallow consists entirely of flying insects captured in flight. Since the swallows have no direct contact with submerged sediments, the incidental sediment ingestion pathway is considered incomplete and a value of 0% diet composition of sediment is applied. The bald eagle and belted kingfisher feed mainly on fish they catch swimming near the surface or in shallow water (USEPA, 1993b). The belted kingfisher generally nests in banks near a body of water, while the bald eagle usually nests in trees, but may also nest on cliffs (Andrle and Carroll, 1988). The incidental ingestion of sediments was considered negligible for the bald eagle and a value of 0% (on a dry weight basis) was applied. Since the belted kingfisher contacts bank sediment during nesting and grooming, a value of 1% (on a dry weight basis) was applied.

Great blue herons fish in shallow waters (up to 0.5 m) with a firm substrate (USEPA, 1993b). They capture fish by thrusting the beak into the fish's side or back (Eckert and Karalus, 1983). Based on the great blue heron's fishing technique, a value of 2% (on a dry weight basis) was applied based on incidental ingestion during feeding and grooming.

3.4.3.3 Dietary Exposure Pathway

Hudson River avian receptors are exposed to PCBs in their diet primarily through consumption of fish and aquatic invertebrates. In the absence of information on feeding habits and dietary composition of Hudson River receptor populations, available literature and discussions with NYSDEC wildlife specialists were used to develop dietary profiles for Hudson River populations. Given the tendency of PCBs to be biomagnified within aquatic food webs, exposure point concentrations for fish were divided into forage fish species and larger piscivorous fish species. Benthic macroinvertebrates are considered as a single dietary source, inclusive of all taxa.

To evaluate the dietary sources of PCBs to avian receptors, a total daily dietary normalized ingestion rate (kg/day on a wet weight basis) for each receptor was referenced from the available literature or developed using the normalized field metabolic rate (NFMR) (kcal/g-day) and the average metabolizable energy (ME_{Ave}) content (kcal/kg) of fish and invertebrates based on USEPA guidance (USEPA, 1993b). Total daily dietary ingestion rates for all the avian receptors were calculated using the normalized field metabolic rate, the typical diet composition for the Hudson River populations, and the average metabolizable energy content of the diet.

A normalized field metabolic rate was estimated for avian receptors based upon the allometric relationship developed by Nagy (1987) and USEPA (December 1993):

$$NFMR = \frac{2.601(BW_a)^{0.640}}{BW_b} \qquad \text{Equation 3-8}$$

where:

NFMR	=	Normalized field metabolic rate (kcal/g-day);
BW_a	=	Body weight of avian receptor (gm); and
BW_b	=	Body weight of avian receptor (kg).

The metabolizable energy content for fish and benthic macroinvertebrates is calculated as the product of the gross energy content (kcal/g) and percent assimilative efficiency of the dietary item by avian consumers (USEPA, 1993b):

$$ME = GE \times AE \qquad \text{Equation 3-9}$$

where:

ME = Metabolizable energy content of dietary component (kcal/gm wet wt);
GE = Gross energy content of dietary component (kcal/gm wet wt); and
AE = Assimilation efficiency value for diet component (unitless).

Gross energy contents of 1.2 kcal/gm for fish, 1.1 kcal/gm for benthic invertebrates (based on isopods and amphipods), and 1.5 kcal/gm for flying insects (based on beetles) were used (USEPA, 1993b). Assimilation efficiencies of 79% and 77% were used for fish and invertebrate prey, respectively (USEPA, 1993b).

The dietary ingestion rate for each of the avian receptors is calculated as the quotient of the receptor-specific NFMR and ME_{Ave} for the specific diet:

$$NIR_{Total} = \frac{NFMR_{Bird}}{ME_{Ave} \times BW_{Bird}} \times 0.001 \qquad \text{Equation 3-10}$$

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where:

NIR_{Total}	=	Species-specific total normalized ingestion rate for avian receptor (kg/day);
$NFMR_{Bird}$	=	Species-specific normalized field metabolic rate (kcal/g-day) for avian receptor;
ME_{Ave}	=	Average metabolizable energy content of dietary component (kcal/g wet wt);
BW_{Bird}	=	Body weight of avian receptor (gm); and
0.001	=	Conversion term from grams to kilograms (kg/gm).

This analysis assumes that all fish and benthic macroinvertebrate prey are obtained from the Hudson River.

The modeled fish component of the avian receptor diet considers two distinct fish trophic levels defined by size, based upon the tendency for PCBs to bioaccumulate to a greater degree in longer-lived, higher-trophic level species. Small fish (< 10 cm) include planktivorous/insectivorous forage fish, such as minnows and sunfish, and large fish (> 25 cm) include benthic/piscivorous fish, such as catfish and bass. Ingestion rates of forage fish and benthic/piscivorous fish are based upon size selectiveness observed in the diet (see Appendix E). The average daily dosage of PCBs to the avian receptor from the fish-derived portion of the diet is expressed as:

$$ADD_{Fish} = \frac{(PCB_{Fish} \times NIR_{Total} \times PD_{Fish})}{BW_{Bird}} \times FE \qquad \text{Equation 3-11}$$

where:

ADD_{Fish}	=	Average/95% UCL daily dietary dose of PCBs from ingestion of fish (mg/kg/day wet wt);
PCB_{Fish}	=	Mean/95% UCL concentration of PCBs observed in fish tissue (mg/kg wet wt);
NIR_{Total}	=	Total normalized ingestion rate for avian receptor (kg/day, wet wt);
PD_{Fish}	=	Fraction of total diet of avian receptor represented by forage and/or large fish (unitless);
FE	=	Areal forage effort as fraction of home range of the endpoint (unitless); and
BW_{Bird}	=	Body weight (kg) of avian receptor.

The modeled benthic invertebrate portion of the avian receptor diet follows an approach similar to that outlined for fish, with the exception that all invertebrate body burdens are deemed comparable and do not consider feeding group-specific bioaccumulative effects. The average daily dosage for the invertebrate portion of the avian receptor diet is expressed as:

$$ADD_{Invert} = \frac{(PCB_{Invert} \times NIR_{Total} \times PD_{Invert})}{BW_{Bird}} \times FE \qquad \text{Equation 3-12}$$

where:

ADD_{Invert}	=	Average/95% UCL daily dietary dose of PCBs from ingestion of benthic invertebrates (mg/kg/day wet wt);
PCB_{Invert}	=	Mean/95% UCL of PCB concentration in invertebrate tissue (mg/kg wet wt);
NIR_{Total}	=	Total normalized ingestion rate for avian receptor (kg/day);
PD_{Invert}	=	Fraction of total diet of avian receptor represented by benthic invertebrates (unitless);
FE	=	Areal forage effort as fraction of home or forage range (unitless); and,
BW_{Bird}	=	Body weight of avian receptor (kg).

The mallard duck feeds on both aquatic invertebrates and plants (USEPA, 1993b). This analysis assumes a macrophyte compartment as a surrogate for the vegetative portion of the diet. Macrophyte concentrations are estimated by:

$$Conc_{macro} = (K_{ow}\ x\ Conc_{diss}\ x\ Lipid_{macro}) \qquad \text{Equation 3-13}$$

where:

$Conc_{macro}$	=	Concentration of PCBs in phytoplankton (mg/kg);
K_{ow}	=	Octanol-water partition coefficient;
$Conc_{diss}$	=	Concentration of PCBs in dissolved water (mg/L); and
$Lipid_{macro}$	=	Organic fraction of macrophytes expressed as lipid (assumed at 1%).

This relationship has been shown to provide reasonable estimates of concentrations in macrophytes and submerged aquatic plant matter (Gobas et al., 1991; Swackhamer and Skoglund, 1993; Lovett-Doust et al., 1997a). Uptake of PCBs from sediment sources may also be significant but there is less quantitative information available to characterize this relationship. Equation 3-13 is likely to provide protective estimates of bioconcentration.

Scientific literature and NYSDEC and USFWS staff were consulted to determine the dietary composition of avian receptors in the Hudson River Valley. Geographical preference for diet-related information followed the order: Upper and Lower Hudson River Valley, other regions of New York State (NYS), populations from the northeastern US, and populations from other regions of the contiguous United States. Wherever possible, multiple data sources were used to define the diet composition.

Tree swallow diets for the Hudson River Valley were based upon bolus sampling conducted by Secord and McCarty (USFWS, 1997) on the Hudson River near Saratoga Springs, NY. Secondary sources for diet composition included Robertson et al. (1992) and McCarty (1999). A diet of 100% flying insects with partial aquatic life histories was used in the exposure assessment.

Mallard diet information for Hudson River or NYS populations in regional proximity

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were not available. Diet studies provided in USEPA (1993b) were reviewed and evaluated for seasonal or habitat specific trends. Mallards feed approximately equally on invertebrates and vegetation during the spring and summer. The invertebrate component of their diet decreases during the fall and winter. No fish were documented in the diets summarized in USEPA (1993b), and therefore fish are not considered in the mallard exposure assessment. Based upon spring and summer feeding patterns, a 50% aquatic invertebrate component, and a 50% vegetation component were used in the exposure assessment.

The primary sources used for the belted kingfisher diet are south-central NYS populations (Gould, unpublished data provided in Salyer and Lagler, 1946) and Davis (1982). Secondary sources include Bull (1998) and Brooks and Davis (1987). The belted kingfisher diet is considered to consist exclusively of forage fish species and aquatic invertebrates. Dietary percentages of 78% fish (as forage fish) and 22% aquatic invertebrates were applied in the exposure assessment.

Data on great blue heron diet information in Hudson River or NYS were not available. The primary sources of diet information for the great blue heron include Alexander (1977) for Michigan populations and Hoffman (1978) for southwestern Lake Erie populations. Diets are derived exclusively from aquatic sources for both studies. Secondary sources for dietary information include Eckert and Karalus (1983) and Krebs (1974). The heron diet was assume to consist of 98% fish (composed primarily of forage fish and small numbers of piscivorous fish), 1% aquatic invertebrates, and 1% non-river related diet sources.

Bald eagle dietary information for Hudson River resident populations was primarily based upon Nye (1999) and Bull (1998). Secondary sources for information included Nye and Suring (1978) and diet studies provided in USEPA (1993b). Diet composition can be highly variable; however, winter diets in the Hudson River resident populations appear to be dominated by fish. Fish species captured tend to be larger species and the diet is restricted to larger fish. Therefore, a diet of 100% fish (as piscivorous fish) derived from the river is applied in the exposure assessment.

3.4.3.4 Behavioral and Temporal Modifying Factors Relating to Exposure

Potential behavioral and temporal modifying characteristics relating to PCB exposure to avian receptors were considered when calculating exposure. The values account for either a species-specific behavioral (e.g., home range) or temporal relationship (e.g., migration, hibernation) resulting in discontinuous exposure duration. Modifying factors typically range in value from 0 to 1.0, with 1.0 representing a continuous exposure duration.

Home range considers the size of the habitat associated with the territorial characteristics of the receptors. The size of the Hudson River site, combined with a preference for riverine environments, resulted in a value of 1.0 (i.e., continuous spatial exposure duration) for all receptors. The river segments selected for evaluation are large enough to encompass the foraging areas of local populations of avian species. These species will integrate exposure over temporal and spatial scales as approximated by the modeling.

Migration considers both spatial and temporal displacement of a receptor in regard to changing seasonal factors, such as dwindling food supplies or severity of weather. The mallard, belted kingfisher, bald eagle and great blue heron have both resident (i.e., year-round) and migrant populations in the Hudson River Valley. It is assumed that the resident populations of these receptors are most at risk and therefore remain continually exposed on a temporal basis. The tree swallow migrates along the Hudson River and is temporally exposed only during spring and summer residency. However, tree swallows breed along the banks of the Hudson River and the young are reared and grow to near adult size prior to the autumn migration.

3.4.3.5 Biomagnification Factors for Predicting Egg Concentrations

Biomagnification factors from the literature were used to predict the concentration of total PCBs and TEQ in the eggs of piscivorous birds from the mean and 95% UCL concentration in fish. A biomagnification factor of 30 has been used to predict these concentrations (USEPA, 1998a). This report uses the results presented in Geisy et al., (1995) which shows a biomagnification factor of 28 for piscivorous birds for total PCBs and 19 on a TEQ basis. A biomagnification factor of 2 was used to estimate diet to egg concentrations for the tree swallow for total PCBs based on the USFWS data, and a factor of 7 on a TEQ basis. The USFWS data provided one mallard sample and two wood duck samples. From these, biomagnification factors of 3 on a total PCB basis and 28 on a TEQ basis were used for the mallard duck analysis.

3.4.3.6 Summary of $ADD_{Expected}$, $ADD_{95\%UCL}$, and Egg Concentrations for Avian Receptors on a Total (Tri+) PCB Basis

Tree Swallow

Tables 3-26 and 3-27 provide the expected average daily dose and 95% UCL daily dose on a total PCB basis for the tree swallow from water and dietary sources based on 1993 data for water and benthic invertebrate concentrations. These tables also show the predicted egg concentrations based on a biomagnification factor derived from the Phase 2 dataset. Tables 3-28 and 3-29 present the expected average daily dose and 95% UCL for the modeling period 1993 – 2018 based on the results from HUDTOX for water and FISHRAND for benthic invertebrates. These tables also show the predicted egg concentrations based on a biomagnification factor derived from the Phase 2 dataset.

Mallard Duck

Tables 3-30 and 3-31 provide the expected average daily dose and 95% UCL daily dose on a total PCB basis for the mallard duck from water, sediment and dietary sources based on 1993 data for water, sediment, and benthic invertebrate concentrations. These tables also show the predicted egg concentrations based on a biomagnification factor derived from the Phase 2 dataset. Tables 3-32 and 3-33 present the expected average daily dose and 95% UCL for the modeling period 1993 – 2018 based on the results from HUDTOX for water and sediment and FISHRAND for benthic invertebrates and macrophytes. These tables also show the predicted egg concentrations based on a biomagnification factor derived from the Phase 2 dataset.

Belted Kingfisher

Tables 3-34 and 3-35 provide the expected average daily dose and 95% UCL daily dose on a total PCB basis for the belted kingfisher from water, sediment, and dietary sources based on 1993 data for water, sediment, benthic invertebrate and forage fish concentrations. These tables also show the predicted egg concentrations based on a biomagnification factor obtained from the literature (Giesy et al., 1995). Tables 3-36 and 3-37 present the expected average daily dose and 95% UCL for the modeling period 1993 – 2018 based on the results from HUDTOX for water and sediment and FISHRAND for benthic invertebrates and forage fish. These tables also show the predicted egg concentrations based on a biomagnification factor derived from the Phase 2 dataset.

Great Blue Heron

Tables 3-38 and 3-39 provide the expected average daily dose and 95% UCL daily dose on a total PCB basis for the great blue heron from water, sediment, and dietary sources based on 1993 data for water, sediment, benthic invertebrate and forage fish concentrations. These tables also show the predicted egg concentrations based on a biomagnification factor obtained from the literature (Giesy et al., 1995). Tables 3-40 and 3-41 present the expected average daily dose and 95% UCL for the modeling period 1993 – 2018 based on the results from HUDTOX for water and sediment and FISHRAND for benthic invertebrates and forage fish. These tables also show the predicted egg concentrations based on a biomagnification factor derived from the Phase 2 dataset.

Bald Eagle

Tables 3-42 and 3-43 provide the expected average daily dose and 95% UCL daily dose on a total PCB basis for the bald eagle from water, sediment, and dietary sources based on 1993 data for water, sediment, benthic invertebrate and piscivorous fish concentrations. These tables also show the predicted egg concentrations based on a biomagnification factor obtained from the literature (Giesy et al., 1995). Tables 3-44 and 3-45 present the expected average daily dose and 95% UCL for the modeling period 1993 – 2018 based on the results from HUDTOX for water and sediment and FISHRAND for benthic invertebrates and piscivorous fish. These tables also show the predicted egg concentrations based on a biomagnification factor derived from the Phase 2 dataset.

3.4.3.7 Summary of $ADD_{Expected}$, $ADD_{95\%UCL}$, and Egg Concentrations for Avian Receptors on a TEQ Basis

Tree Swallow

Tables 3-46 and 3-47 provide the expected average daily dose and 95% UCL daily dose on a TEQ basis for the tree swallow from water and dietary sources based on 1993 data for water and benthic invertebrate concentrations. These tables also show the predicted egg concentrations based on a biomagnification factor derived from the Phase 2 dataset. Tables 3-48 and 3-49 present the expected average daily dose and 95% UCL for the modeling period 1993 – 2018 based on the results from HUDTOX for water and FISHRAND for benthic invertebrates. These tables also show the predicted egg concentrations based on a biomagnification factor derived from the Phase 2 dataset.

Mallard Duck

Tables 3-50 and 3-51 provide the expected average daily dose and 95% UCL daily dose on a TEQ PCB basis for the mallard duck from water, sediment and dietary sources based on 1993 data for water, sediment, and benthic invertebrate concentrations. These tables also show the predicted egg concentrations based on a biomagnification factor derived from the Phase 2 dataset. Tables 3-52 and 3-53 present the expected average daily dose and 95% UCL for the modeling period 1993 – 2018 based on the results from HUDTOX for water and sediment and FISHRAND for benthic invertebrates and macrophytes. These tables also show the predicted egg concentrations based on a biomagnification factor derived from the Phase 2 dataset.

Belted Kingfisher

Tables 3-54 and 3-55 provide the expected average daily dose and 95% UCL daily dose on a TEQ PCB basis for the belted kingfisher from water, sediment, and dietary sources based on 1993 data for water, sediment, benthic invertebrate, and forage fish concentrations. These tables also show the predicted egg concentrations based on a biomagnification factor obtained from the literature (Giesy et al., 1995). Tables 3-56 and 3-57 present the expected average daily dose and 95% UCL for the modeling period 1993 – 2018 based on the results from HUDTOX for water and sediment and FISHRAND for benthic invertebrates and forage fish. These tables also show the predicted egg concentrations based on a biomagnification factor obtained from the literature (Giesy et al., 1995).

Great Blue Heron

Tables 3-58 and 3-59 provide the expected average daily dose and 95% UCL daily dose on a TEQ PCB basis for the great blue heron from water, sediment, and dietary sources based on 1993 data for water, sediment, benthic invertebrate, and forage fish concentrations. These tables also show the predicted egg concentrations based on a biomagnification factor obtained from the literature (Giesy et al., 1995). Tables 3-60 and 3-61 present the expected average daily dose and 95% UCL for the modeling period 1993 – 2018 based on the results from HUDTOX for water and sediment and FISHRAND for benthic invertebrates and forage fish. These tables also show the predicted egg concentrations based on a biomagnification factor obtained from the literature (Giesy et al., 1995).

Bald Eagle

Tables 3-62 and 3-63 provide the expected average daily dose and 95% UCL daily dose on a TEQ PCB basis for the bald eagle from water, sediment, and dietary sources based on 1993 data for water, sediment, benthic invertebrate, and piscivorous fish concentrations. These tables also show the predicted egg concentrations based on a biomagnification factor obtained from the literature (Giesy et al., 1995). Tables 3-64 and 3-65 present the expected average daily dose and

95% UCL for the modeling period 1993 – 2018 based on the results from HUDTOX for water and sediment and FISHRAND for benthic invertebrates and piscivorous fish. These tables also show the predicted egg concentrations based on a biomagnification factor obtained from the literature (Giesy et al., 1995).

3.4.4 Mammalian Exposure Pathways and Parameters

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 subchapter 2.3.4). Intake is calculated as an average daily dosage (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 equation is provided as:

$$ADD_{River} = ADD_{Diet} + ADD_{Water} + ADD_{Sediment} \qquad \text{Equation 3-14}$$

where:

ADD_{River}	=	Potential average daily dosage of PCBs to receptor from Hudson River sources (mg/kg/day);
ADD_{Diet}	=	Average daily dosage of PCBs via dietary sources (mg/kg/day);
ADD_{Water}	=	Average daily dosage of PCBs via drinking water (mg/kg/day); and,
$ADD_{Sediment}$	=	Average daily dosage of PCBs via incidental ingestion of sediments (mg/kg/day).

The direct ingestion of surface water for drinking and the incidental ingestion of sediments are generic exposure pathways that were developed based upon allometric relationships and guidance described in USEPA (1993b) and Nagy (1987). Ingestion rates are derived based upon a single variable, body weight, which is expressed as kilograms or grams wet weight. Dietary exposure is the most variable pathway, since the percentage of mammalian receptor diet derived from the Hudson River, type of prey consumed (i.e., fish or invertebrates), and size selectivity of prey species differs between receptors. The equations used to calculate intakes for each exposure pathway is provided below. Parameters used for the little brown bat, raccoon, mink, and river otter are summarized in Tables 3-22 to 3-25.

3.4.4.1 Surface Water Ingestion Pathway

For mammalian receptors, the normalized water ingestion rate (NWI) (L/day) was estimated from the following equation (USEPA, 1993b):

$$NWI_{(Mammal)} = (0.099 * BW^{0.90}) \qquad \text{Equation 3-15}$$

where:

$NWI_{(Mammal)}$	=	Receptor-specific normalized water ingestion rate (L/day); and
BW	=	Body weight of receptor (kg).

PCBs ingested on a daily basis are calculated as the product of the concentration of PCBs in surface water (mg/L) and the normalized water ingestion rate (L/day). The receptor-specific average daily dosage rate ADD_{Water} (mg/kg/day) is derived as the quotient of the mass of PCBs ingested on a daily basis and the body mass of the species being evaluated:

$$ADD_{Water} = \frac{(PCB_{Surfacewater} \times NWI_{Mammal})}{BW_{Mammal}} \times (FE) \qquad \text{Equation 3-16}$$

where:

ADD_{Water}	=	Average/95% UCL daily dose of PCB via drinking of Hudson River surface water (mg/kg/day);
$PCB_{Surfacewater}$	=	Mean/95% UCLPCB exposure concentration (mg/L) in surface water;
NWI_{Mammal}	=	Normalized water ingestion rate (L/day) for mammalian receptor;
FE	=	Areal forage effort (unitless) as fraction of home or forage range; and,
BW_{Mammal}	=	Body weight (kg) of mammalian receptor.

Given the size of the Hudson River site, exposure to Hudson River derived PCBs sources (water, sediment, biota) was considered continuous and the areal foraging effort factor (FE) for all receptors was set at a value of 1.0 for all parameters.

3.4.4.2 Incidental Sediment Ingestion Pathway

Incidental ingestion of Hudson River sediments by mammalian receptors may occur through feeding and non-feeding activities, such as cleaning and grooming of the fur. The equation for this pathway is considered on a dry weight basis for evaluation and accounts for only the fraction of the total diet represented by abiotic material (USEPA, 1993a). The incidental ingestion is calculated as:

$$ADD_{Sediment} = \frac{(PCB_{Sediment} \times FS_{Media} \times NIR_{Total})}{BW_{Mammal}} \times FE \qquad \text{Equation 3-17}$$

where:

$ADD_{Sediment}$	=	Average/95% UCL daily dose of PCB via incidental ingestion of sediments (mg/kg/day);
$PCB_{Sediment}$	=	Mean/95% UCL PCB concentration (mg/kg) in sediment;
FS_{Media}	=	Fraction of abiotic media in diet (%);
NIR_{Total}	=	Total normalized food ingestion (FI) rate (kg/day dry wt); estimated using FI (kg/day) = $0.0687(BW)^{0.822}$ (USEPA, 1993a);
FE	=	Areal foraging effort (1.0)(unitless);
BW_{Mammal}	=	Body weight (kg) of mammalian receptor.

The fraction of incidental sediment ingestion in the mammalian diet is specific to each of the endpoint receptors. Most incidental ingestion occurs during feeding (Beyer et al., 1994) and the greatest potential for this exposure pathway occurs while feeding on aquatic benthic invertebrates. Therefore, mammalian receptors, such as the raccoon, that have diets inclusive of an important benthic invertebrate component are likely to have higher incidental exposures to PCBs via sediment ingestion than largely piscivorous species.

Incidental sediment ingestion for the raccoon, an omnivore that consumes a large percentage of aquatic invertebrates, has been estimated to be 9.4% (Beyer et al., 1994). Quantitative estimates of percent composition of sediments in the diet of the little brown bat, mink, and river otter were not available. Therefore, incidental sediment ingestion for these receptors was estimated based on their feeding patterns. The diet of the little brown bat consists entirely of flying insects captured in flight. Since bats have no contact with submerged sediments, they are not considered to ingest any sediment in their diet. The mink and the otter are largely piscivorous and incidental ingestion of sediments during feeding is considered to be limited. An incidental sediment ingestion value of 1% (on a dry weight basis) was used to cover incidental sediment ingestion during feeding and grooming for both receptors.

3.4.4.3 Dietary Exposure Pathway

Hudson River mammalian receptors are exposed to PCBs in their diet primarily through the consumption of fish and aquatic invertebrates. In the absence of information on feeding habits and dietary composition of Hudson River receptor populations, available literature and discussions with NYSDEC wildlife specialists were used to develop dietary profiles. Given the tendency of PCBs to be biomagnified within aquatic food webs, modeled exposure point concentrations for fish were divided into forage fish species and larger piscivorous fish species. Benthic macroinvertebrates are considered as a single dietary source, inclusive of all taxa.

To evaluate the dietary sources of PCBs to mammalian receptors, a total daily dietary normalized ingestion rate (kg/day on a wet weight basis) for each receptor was referenced from the available literature or developed using the normalized field metabolic rate (NFMR) (kcal/gm-day) and the average metabolizable energy (ME_{Ave}) content (kcal/kg) of fish and invertebrates based on USEPA guidance (USEPA, 1993a). Total daily dietary ingestion rates for the little brown bat (Fenton and Barclay, 1980), mink (Bleavins and Aulerich, 1981) and river otter (Harris, 1968; USEPA, 1993a) were based on published literature (see Appendix F). An estimated daily dietary ingestion rate was developed for the raccoon using the normalized field metabolic rate, the typical NYS diet composition of the raccoon, and the average metabolizable energy content of the diet.

A normalized field metabolic rate was estimated for the raccoon (a non-herbivore) based upon the allometric relationship developed by Nagy (1987):

$$NFMR = \frac{0.6167 \times (BW_a)^{0.862}}{BW_b} \qquad \text{Equation 3-18}$$

where:

NFMR	=	Normalized field metabolic rate (kcal/g-day);
BW_a	=	Body weight of mammalian receptor (gms); and

BW_b	=	Body weight of mammalian receptor (kg).

The metabolizable energy content for fish and benthic macroinvertebrates is calculated as the product of the gross energy content (kcal/g) and percent assimilative efficiency of the dietary item by mammalian consumers (USEPA, 1993a):

$$ME = GE \times AE \qquad \text{Equation 3-19}$$

where:

ME = Metabolizable energy content of dietary component (kcal/gm wet wt);
GE = Gross energy content of dietary component (kcal/gm wet wt); and
AE = Assimilation efficiency value for diet component (unitless).

Gross energy contents of 1.2 kcal/gm for fish, 1.1 kcal/gm for benthic invertebrates (based on isopods and amphipods), and 1.5 kcal/gm for flying insects (based on beetles) were used (USEPA, 1993a). Assimilation efficiencies of 91% and 87% were used for fish and invertebrate prey, respectively (USEPA, 1993a).

The dietary ingestion rate for each of the mammalian receptors is calculated as the quotient of the receptor-specific NFMR and ME_{Ave} for the specific diet:

$$NIR_{Total} = \frac{NFMR_{Mammal}}{ME_{Ave}} \times BW_{Mammal} \times 0.001 \qquad \text{Equation 3-20}$$

where:

NIR_{Total}	=	Species-specific total normalized ingestion rate for mammalian receptor (kg/day);
$NFMR_{Mammal}$	=	Species-specific normalized field metabolic rate (kcal/gm-day) for mammalian receptor;
ME_{Ave}	=	Average metabolizable energy content of dietary component (kcal/gm wet wt);
BW_{Mammal}	=	Body weight of mammalian receptor (gm); and
0.001	=	Conversion term from grams to kilograms (kg/gm).

This analysis assumes that fish and benthic macroinvertebrates consumed by mammalian receptors are obtained only from the Hudson River. Therefore, the average daily dosage (ADD) of diet derived sources of PCBs to mammalian receptors is expressed as:

$$ADD_{Diet} = ADD_{Fish} + ADD_{Invert} \qquad \text{Equation 3-21}$$

where:

ADD_{Diet}	=	Cumulative average/95% UCL daily dose of PCBs from diet (mg/kg/day);
ADD_{Fish}	=	Average/95% UCL daily dietary dose of PCBs from ingestion of fish (mg/kg/day); and,
ADD_{Invert}	=	Average/95% UCL daily dietary dose of PCBs from ingestion of invertebrates (mg/kg/day).

The fish component of the modeled mammalian receptor diet considers two distinct fish trophic levels, defined by size, based upon the tendency for PCBs to bioaccumulate to a greater degree in longer-lived, higher trophic-level species. Small fish (< 10 cm) include planktivorous/insectivorous forage fish, such as minnows and sunfish, and large fish (> 10 cm) include benthic/piscivorous fish, such as catfish and bass. Mammalian receptor ingestion rates of forage fish and benthic/piscivorous fish are based upon size selectiveness observed in the diet (see Appendix F). The average daily dosage of PCBs to receptors from the fish-derived portion of the diet is expressed as:

$$ADD_{Fish} = \frac{(PCB_{Fish} \times NIR_{Total} \times PD_{Fish})}{BW_{Mammal}} \times FE \qquad \text{Equation 3-22}$$

where:

ADD_{Fish}	=	Average/95% UCL daily dietary dose of PCBs from ingestion of fish (mg/kg/day wet wt);
PCB_{Fish}	=	Mean/95% UCL concentration of PCBs in fish tissue (mg/kg wet wt);
NIR_{Total}	=	Total normalized ingestion rate for mammalian receptor (kg/day, wet wt);
PD_{Fish}	=	Fraction of total diet of mammalian receptor represented by forage and/or large fish (unitless);
FE	=	Areal forage effort as fraction of home range of the receptor (unitless); and,
BW_{Mammal}	=	Body weight (kg) of mammalian receptor.

The benthic invertebrate portion of the modeled mammalian diet follows an approach similar to that outlined for fish, with the exception that all invertebrate body burdens are deemed comparable and do not consider feeding group-specific bioaccumulative effects. The average daily dosage for the modeled mammalian invertebrate portion of the diet is expressed as:

$$ADD_{Invert} = \frac{(PCB_{Invert} \times NIR_{Total} \times PD_{Invert})}{BW_{Mammal}} \times FE \qquad \text{Equation 3-23}$$

where:

ADD_{Invert}	=	Average/95% UCL daily dietary dose of PCBs from ingestion of benthic invertebrates (mg/kg/day wet wt.);

PCB_{Invert}	=	Mean/95% UCL of PCB concentration in invertebrate tissue (mg/kg wet wt.);
NIR_{Total}	=	Total normalized ingestion rate for mammalian receptor (kg/day);
PD_{Invert}	=	Fraction of total diet of mammalian receptor represented by benthic invertebrates (unitless);
FE	=	Areal forage effort as fraction of home or forage range (unitless); and,
BW_{Mammal}	=	Body weight of mammalian receptor (kg).

The scientific literature and NYSDEC and USFWS wildlife specialists were consulted for identifying the dietary composition of mammalian receptors in the Hudson River study area. As with avian receptors, the geographical preference for diet-related information for receptor populations followed the order: Upper and Lower Hudson River Valley, other regions of NYS, populations from the northeastern US, and populations from other regions of the contiguous US. Wherever possible, collaborative information from secondary sources was used to better define the diet composition which the receptor populations would be expected to consume. Prey consumption rates were selected to reflect the general regional feeding preferences found in the literature, which did not always represent the maximum reported consumption of river-related prey for that receptor.

The little brown bat diet studies of Buchler (1976) and Belwood and Fenton (1976) in NYS were used as the primary sources for diet composition. Secondary information was drawn from Anthony and Kunz (1977) for Nova Scotia populations. The little brown bat diet may consist of 87% to 100% insects with partial aquatic life histories. Based upon these data, a diet composition of 100% aquatic invertebrates (as insects with partial aquatic life histories) was applied to Hudson River Valley little brown bat populations.

The raccoon diet studies of Tabatabai and Kennedy (1988) on Tennessee populations, Llewellyn and Ulher (1952) on Maryland populations, and Hamilton (1940) on NYS populations of raccoon were utilized as primary sources for raccoon diet composition of fish and aquatic invertebrates. Review of the literature revealed a marked difference in raccoon diet composition based on habitats and seasons. Raccoons from forested bottom land and riverine environments (like those of the Hudson River) had a larger aquatic component in the diet than populations from marshes or more agricultural land uses, with winter diets for both groups accounting for the largest percentage of aquatic sources (i.e., fish and aquatic invertebrates). Based upon this review, a winter diet composition of 3.0% fish and 37.0% aquatic invertebrates (Llewellyn and Ulher, 1952) was applied for Hudson River Valley raccoon populations.

Diet studies by Hamilton (1959, 1940, 1936) for NYS populations of mink were utilized as the primary sources for mink diet composition of fish and aquatic invertebrates. Secondary information was drawn from staff of the NYSDEC Furbearer Units in Bath, NY (Mayack, 1999) and Delmar, NY (Batcheller, 1999). Review of the literature revealed a marked seasonality in mink diet components from aquatic sources, with winter diets accounting for the largest percentage of aquatic sources (i.e., fish and aquatic invertebrates). Based upon this review, a winter diet of 34.0% fish and 16.5% aquatic invertebrates (as cited in Hamilton, 1959) was applied to Hudson River Valley mink populations.

The diet studies by Hamilton (1961) on NYS river otter populations, Sheldon and Toll (1964) on Massachusetts populations, and personal communications with Penrod (1999) and Spinola (1999) of the NYSDEC River Otter Project were used as primary sources for the diet of Hudson River otter populations. These and other studies (Newell et al. 1987; Knudsen and Hale, 1968; Geer, 1955) showed that although there was seasonality in diet components, fish are the preferred prey of river otters. Recent field observations by Spinola (1999) suggest that the winter diet of the river otter is composed exclusively of fish. Hence, a diet composition of 100% fish was applied to Hudson River otter populations.

3.4.4.4 Behavioral and Temporal Modifying Factors Relating to Exposure

Potential modifying characteristics related to PCB exposure were considered when calculating exposure to mammalian receptors. The values account for a species-specific behavioral (e.g., home range) or temporal relationship (e.g., migration, hibernation) resulting in discontinuous exposure duration. Modifying factors typically range in value from 0 to 1.0, with 1.0 being a continuous exposure duration.

Home range considers the size of the habitat associated with the territorial characteristics of the receptors. The size of the Hudson River site, combined with the receptors' preference for riverine environments, resulted in a value of 1.0 (i.e., continuous spatial exposure duration) for all receptors. The river segments selected for evaluation are large enough to encompass the foraging areas of local populations of mammalian species. These species will integrate exposure over temporal and spatial scales as approximated by the modeling.

Migration considers the spatial and temporal displacement of a receptor in regard to changing seasonal factors, such as dwindling food supplies or severity of weather. All four of the mammalian receptors are considered year-round residents of the Hudson River. The little brown bat is the only receptor that hibernates. Although the little brown bat hibernates part of the year, all food sources used during the year (i.e., active feeding time plus fat reserves used during hibernation) are assumed to be derived from the Hudson river. Therefore, no temporal modifying factor was applied to the little brown bat.

3.4.4.5 Summary of $ADD_{Expected}$ and $ADD_{95\%UCL}$ for Mammalian Receptors Based on Total (Tri+) PCBs

Little Brown Bat

Tables 3-66 and 3-67 provide the expected average daily dose and 95% UCL daily dose on a total PCB basis for the little brown bat from water and dietary sources based on 1993 data for water and benthic invertebrate concentrations. Tables 3-68 and 3-69 present the expected average daily dose and 95% UCL for the modeling period 1993 – 2018 based on the results from HUDTOX for water and FISHRAND for benthic invertebrates.

Raccoon

Tables 3-70 and 3-71 provide the expected average daily dose and 95% UCL daily dose on a total PCB basis for the raccoon from water and dietary sources based on 1993 data for water, sediment, benthic invertebrate, and forage fish concentrations. Tables 3-72 and 3-73 present the expected average daily dose and 95% UCL for the modeling period 1993 – 2018 based on the results from HUDTOX for water and sediment and FISHRAND for benthic invertebrates and forage fish.

Mink

Tables 3-74 and 3-75 provide the expected average daily dose and 95% UCL daily dose on a total PCB basis for the mink from water and dietary sources based on 1993 data for water, sediment, benthic invertebrate, and forage fish concentrations. Tables 3-76 and 3-77 present the expected average daily dose and 95% UCL for the modeling period 1993 – 2018 based on the results from HUDTOX for water and sediment and FISHRAND for benthic invertebrates and forage fish.

Otter

Tables 3-78 and 3-79 provide the expected average daily dose and 95% UCL daily dose on a total PCB basis for the otter from water and dietary sources based on 1993 data for water, sediment, and piscivorous fish concentrations. Tables 3-80 and 3-81 present the expected average daily dose and 95% UCL for the modeling period 1993 – 2018 based on the results from HUDTOX for water and sediment and FISHRAND for piscivorous fish.

3.4.4.6 Summary of $ADD_{Expected}$ and $ADD_{95\%UCL}$ for Mammalian Receptors on a TEQ Basis

Little Brown Bat

Tables 3-82 and 3-83 provide the expected average daily dose and 95% UCL daily dose on a TEQ PCB basis for the little brown bat from water and dietary sources based on 1993 data for water and benthic invertebrate concentrations. Tables 3-84 and 3-85 present the expected average daily dose and 95% UCL for the modeling period 1993 – 2018 based on the results from HUDTOX for water and FISHRAND for benthic invertebrates.

Raccoon

Tables 3-86 and 3-87 provide the expected average daily dose and 95% UCL daily dose on a TEQ PCB basis for the raccoon from water and dietary sources based on 1993 data for water, sediment, benthic invertebrate, and forage fish concentrations. Tables 3-88 and 3-89 present the expected average daily dose and 95% UCL for the modeling period 1993 – 2018 based on the results from HUDTOX for water and sediment and FISHRAND for benthic invertebrates and forage fish.

Mink

Tables 3-90 and 3-91 provide the expected average daily dose and 95% UCL daily dose on a TEQ PCB basis for the mink from water and dietary sources based on 1993 data for water, sediment, benthic invertebrate, and forage fish concentrations. Tables 3-92 and 3-93 present the expected average daily dose and 95% UCL for the modeling period 1993 – 2018 based on the results from HUDTOX for water and sediment and FISHRAND for benthic invertebrates and forage fish.

Otter

Tables 3-94 and 3-95 provide the expected average daily dose and 95% UCL daily dose on a TEQ PCB basis for the otter from water and dietary sources based on 1993 data for water, sediment, and piscivorous fish concentrations. Tables 3-96 and 3-97 present the expected average daily dose and 95% UCL for the modeling period 1993 – 2018 based on the results from HUDTOX for water and sediment and FISHRAND for piscivorous fish.

3.5 Examination of Exposure Pathways Based on Congener Patterns

3.5.1 Introduction

As part of the ecological risk assessment, an examination of the fish exposures to PCBs in the environment was completed. Because fish exposures to PCBs from sediment, water and through the food chain all contribute to the fish body burden, it is useful to examine fish body burdens relative to these sources. Specifically, the congener pattern of a fish's body burden reflects, to varying degrees, the nature and history of its exposure. Thus an examination of the congener patterns in fish and other matrices may provide useful clues in designating the main PCB sources to the fish. If the congener "fingerprint" remains unaltered from source to the fish, this analysis can directly link the source(s) to the fish body burden. Information linking fish body burdens to their sources is clearly useful in selecting effective remedial actions. However, as will be shown, the links between fish body burden and source are not straightforward.

Patterns of PCB contamination in fish and benthic invertebrates were examined using the congener-specific PCB data from the 1993 USEPA Phase 2 ecological investigation, the 1993 NOAA fish analyses, and the 1995 NOAA fish analyses. Additionally, the long-term monitoring records for fish obtained by NYSDEC were examined along with USGS water column data to establish current trends between PCB body burden and water column concentrations for several fish species. This analysis represents the biological extension of the geochemical analysis presented in the Data Evaluation and Interpretation Report (USEPA, 1997a) and the Low Resolution Sediment Coring Report (LRC) (USEPA, 1998b), examining the correlations among fish and invertebrate body burdens, sediment, and water column conditions. Details of the analysis are found in Appendix K.

Also in Appendix K, the congener patterns contained in fish are examined from the context of classifying the mixture for the purposes of assigning risk-based criteria (i.e., toxicity benchmarks). This examination addresses, to a limited extent, the "best" basis for quantifying current fish body burdens in terms of Aroclor-based analyses and standards. This issue arises from the historical analytical protocols that characterized fish body burdens in terms of Aroclors 1248 and 1254, despite the documented presence of a predominantly Aroclor 1242-based source throughout the freshwater Hudson River (USEPA, 1997a).

The objectives in conducting this analysis include:

- Identifying Aroclor patterns for use in toxicity assessment;
- Determining the relative importance of water, sediment, and food exposures;
- Evaluating the importance of upstream *versus* downstream sources of PCBs through spatial and temporal patterns;
- Importance of ongoing or recent releases in comparison to historical releases; and,
- Use of marker compounds and ratios to understand exposure.

Conclusions from the analysis presented in Appendix K are summarized as follows:

- The PCB mixture contained in the fish of the Hudson River can be best characterized as a Aroclor 1248-type mixture in the Upper Hudson with a trend toward a heavier mixture (*i.e.*, Aroclor 1254) in fish from the freshwater Lower Hudson and the harbor. These congener mixtures do not imply the increased presence of these Aroclors in the freshwater Lower Hudson but rather are indicative of the enhanced bioaccumulation of the heavier congeners contained in the mixture released by GE. For the purposes of toxicity assessment, Upper Hudson fish are best classified as containing Aroclor 1248, based on the molecular weight and homologue patterns contained in the fish. Similarly, Lower Hudson fish are best classified as containing a mixture of Aroclors 1248 and 1254.

- The PCB body burden of benthic invertebrates represents an intermediate stage between the sediments and fish body burdens based on congener pattern. These benthic invertebrates are still most similar to Aroclor 1248 although less so than the fish. The principal components analysis showed a slightly closer association of the sediments and benthic invertebrate congener pattern. Similarly, the magnitude of the benthic body burdens is seen to vary with the sediment concentrations, with lower body burdens associated with lower sediment concentrations.

- Examination of fish congener patterns using principal components analysis showed that the fish are distinct from their exposure media, in that a readily discernable molecular weight and congener pattern shift occurs with the accumulation of PCBs. This shift increased with decreasing river mile despite the overall decrease in fish body burden. Specifically, an enhancement of the proportion of heavier congeners (penta- and hexachlorohomologues) occurs at the same time that the fish body burdens decline. This occurs despite a much smaller change in the congener composition of the sediments. Changes in water column concentrations may be partially responsible for the enhanced PCB molecular weight in fish, largely attributed to the loss of the lighter congeners from the water column during transport from the Upper River, and not to the introduction of additional heavier Aroclor mixture to the freshwater Lower Hudson. The principal component analysis also shows that benthic invertebrates results typically lie part way between the fish and sediment domains, as might be expected based on trophic level.

- Fish body burdens decrease downstream of the GE facilities, regardless of species. However, the congener properties do not remain constant and the fraction of higher molecular weight congeners increases with decreasing river mile.

- The ratios of BZ#56, 60, 66 and BZ#70 to 49 were examined for several different matrices with the intent of using these ratios as tracers or "fingerprints" of the PCB sources to the fish. These ratios exhibited a large degree of variation in fish that was not shown to occur in any other media. Additionally, comparison of dissolved and suspended matter ratios suggested that the geochemistries of these congeners are not identical and may be different enough to preclude their usefulness as tracers. Overall, these ratios showed a general decline in fish with distance downstream although the ratios themselves were only somewhat similar to those seen in the dissolved phase water column and were distinctly lower than downriver sediments. These poorly understood variations in the ratios preclude their use as tracers. Essentially, the environmental modifications, particularly those produced by fish, serve to erase the "fingerprint" of the original PCB source material. Ultimately, the ratios found in fish (and benthic invertebrates) were unique to the biota, and provided little clue as to the nature of the source.

- Using two different sets of congeners, principal components analysis was used to compare the 1993 and the 1995 fish congener patterns. Using the larger of the two congener sets (46 congeners), the analysis largely confirmed the prior analyses performed by NOAA (1997) as well as in previous subchapters of this report. In particular, spring conditions in 1995 were distinctly different (higher molecular weight in spring) from those of the two fall sampling events. Little difference was evident between the two fall sampling events, suggesting that little had occurred (such as GE remediation of the Hudson Falls releases) to affect the congener patterns and, by inference, the basic routes of exposure in fish. Alternatively, the lack of difference in fall conditions may be partially the result of the bioaccumulation processes which simply serve to create the same general congener pattern in the fall, so long as exposure routes and congener concentrations are approximately the same.

The first two objectives are summarized in greater detail next as they relate directly to the evaluation of ecological risk.

3.5.2 Identifying Aroclor Patterns for Use in Toxicity Assessment

The analysis prepared by NOAA, as well as those of the DEIR and LRC, demonstrated the complexities of the PCB congener patterns in the Hudson River among the various matrices (i.e., sediments, water, fish and benthic invertebrates). In order to capture and reflect these complexities in the data analysis, a principal components analysis (PCA) was undertaken. Effectively, PCA reduces the data set and its associated variables into a minimum number of variables which can then be used to examine the data. This PCA analysis provides a means of showing the appropriateness of using toxicity reference values (TRVs) based on Aroclor 1254 and will explore the ability to trace the source of PCBs in fish.

The first principal component is constructed as a linear combination of the original variables so as to encompass (or "explain") the greatest amount of the variance for the original data set. Subsequent principal components encompass the largest amount of the remaining variance of the data set while being uncorrelated (orthogonal) to all previously constructed principal components. Detailed information on the selection of the congener variables selected for the analysis and the analysis itself can be found in Appendix K. This subchapter summarizes the results of the PCA.

The PCA suggests a strong similarity between the fish body burdens and Aroclor 1248. This is largely due to the bioaccumulation of the tetrachlorocongeners which are most prevalent in this Aroclor. As is suggested by the loading to components 1 and 2, this PCA strongly reflects the molecular weight of the congener mixture and emphasizes its importance in examining the congener data.

The agreement between Aroclor 1248 and the body burden for Upper Hudson River fish is demonstrated by comparing Upper Hudson River fish samples to Aroclor standards on a mass fraction basis. Figure K-12 presents several regressions between a typical Upper River 1993 largemouth bass sample from RM190 versus several Aroclors standards on a mass fraction basis. The regressions represent double hit pairs only, that is congeners which were detected in both sediment and the Aroclor. Although agreement is best for Aroclor 1248, the result is not a true line and several congener proportions fall well away from line. This analysis was repeated using a typical Lower Hudson River white perch sample from RM26 and is shown in Figure K-13. Based on the previous principal components analysis, fish in the Lower Hudson appeared to approach Aroclor 1260. However, when all congeners are considered via regressions such as those in Figure K-13, the best regressions are obtained against Aroclor 1254. For the Lower Hudosn River fish sample shown in the figure, the best fit is achieved against Aroclor 1254 with a regression coefficient of 0.65 that is relatively close to the regression coefficient of 0.7 for the Upper Hudson River fish sample against Aroclor 1248. The fact that the regression coefficients are highest for two different Aroclors is simply indicative of the shift in molecular weight of the fish PCB body burden while moving downstream.

Component 1 itself was examined as a function of river mile for both sediment and fish (see Figure K-16). Though the variance observed is nontrivial, trends in the data are evident. The more pronounced rise in the value of component 1 for the fish data relative to the sediment data is clearly in evidence. In the figure, the lines represent a weighted average of the data. While the fish data appear to rise relatively steadily, the sediment results show several distinct features, including a marked drop in the Upper Hudson River, a near-plateau level in the freshwater Lower Hudson River and finally a sharp rise near the salt front at RM60. The plateau value of the freshwater Lower Hudson River is directly contrasted against the rising fish component 1 levels in Figure K-17. The consistency of the component 1 value in the sediments versus the rising values in the fish may indicate a change in the absorption and retention of PCBs in fish in this region of the river because an additional, substantive, higher molecular weight PCB load to this region is not in evidence (USEPA, 1997a). Alternatively, this may be attributable to a change in the PCB exposures to the fish resulting from the loss of the lighter congeners from the water column during transport downstream. This would yield fish body burdens which had higher molecular weight but lower total PCB mass.

Component 1 appears to closely match molecular weight. Note the similarity in the trends of component 1 versus molecular weight in fish and sediments as function of river mile (see Figure K-16 and the top diagram in Figure K-18). As in Figure K-16, the lines in Figure K-18 represent weighted averages and are used to simply illustrate general trends. Both component 1 and molecular weight show a gradual rise from the TI Pool to New York City harbor with a plateau in the freshwater Lower Hudson for sediments but not for fish. As shown in the lower diagram in Figure K-18, this rise in molecular weight in fish is paralleled only by a rise in the molecular weight of the water-column dissolved-phase PCB fraction. Note the similar slope values as well as the high R^2 values relative to the other two matrices plotted.

The reason for the parallel trends in the fish and water column dissolved phase matrices in this region is unclear because, in general, the dissolved phase contains a higher proportion of less chlorinated congeners due to partitioning while the congeners in fish are more chlorinated. Most likely, the molecular weight increase in the dissolved phase is due to gas exchange plus degradative losses of the lighter dissolved congeners as well as the possible partial replenishment via the resuspension of less dechlorinated, higher molecular weight PCBs from the sediments of the Lower Hudson River, similar to interpretation of the water column data of the Upper Hudson River presented in Appendix C of the Responsiveness Summary for the LRC (USEPA, 1999a). To the extent that water column exposure to fish is important, the increase in the molecular weight of the dissolved phase combined with its absolute decline in concentration may produce the observed trends in fish body burden. Alternatively, the simple decline in water column concentrations alone with river mile would serve to decrease the overall fish exposure (resulting in lower body burdens) while raising the mean molecular weight of the mixture to which the fish are exposed (resulting in higher molecular weights).

3.5.3 Determining the Relative Importance of Water, Sediment, and Dietary Exposures

Fish body burdens were shown to decline with river mile to about the same degree as the changes in the sediment PCB concentration. Similarly, molecular weight in fish samples increased with distance from the Upper Hudson River source areas. Differences in total PCB concentration among species was shown to be significant based on feeding guild (i.e., food source). However, when normalized to lipid content, the interspecies differences disappeared and the largest changes in PCB concentration coincided with river mile. Similarly, the molecular weight of the PCB body burdens in fish was not found to vary by feeding guild but simply by river mile. These results indicate that PCB uptake and biomagnification of individual congeners in fish is largely related to distance downstream of the GE facilities and not to trophic level. In addition, the reason for the increase in molecular weight with distance downstream was not known but may be attributed to one or more several causes including decreasing importance of water column exposure for fish due to declining water column concentrations, particularly for lighter congeners. Alternatively, water column concentrations may simply become higher in molecular weight due to replenishment from less-dechlorinated, Lower Hudson sediments, yielding a higher molecular weight for water-based exposure. Lastly, metropolitan New York discharges present higher molecular weight mixtures for fish exposure in the saline portion of the Lower Hudson River.

Benthic invertebrate data were examined and shown to be similar to the results for fish for much of the Hudson River. Benthic invertebrates in the freshwater Hudson River typically have lower molecular weights than the fish from the same location, but have higher molecular weights than the sediments in which they live. Benthic invertebrate body burdens decline with river mile. Benthic invertebrates in the saline Lower Hudson distinctly show the impact of the New York City metropolitan area inputs. These invertebrates have a substantially higher molecular weight than that of the Upper Hudson River. Epibenthic invertebrates appeared to have lower body burdens but similar molecular weights relative to other benthic invertebrates collected from the same station. This suggests that the bioaccumulation process may be dependent on PCB congener type or perhaps molecular weight.

Combining the results of Figures K-41, K-42 and K-43, there appears to be a minor shift toward higher molecular weights (i.e., heavier congeners) from Fall 1993 to Fall 1995 and Spring 1995. The shift appears to be much greater for the Fall 1993 to Spring 1995 sampling than from Fall 1993 to Fall 1995. Based on the last diagram in Figure K-41, the Spring 1995 results also appear to have a higher molecular weight than that for Fall 1995. These general trends were also noted in the NOAA report (1997) based on several individual congeners. However, these conclusions must be tempered by the confounding factor of life-stage which was also shown to coincide with changes in molecular weight. Based on these results plus the direct homologue comparisons provided in Figures K-4 to K-6, it appears likely that seasonal variation in fish body burden does occur, with heavier molecular weights coinciding with the spring. On the other hand, there does not appear to be a systematic change in the fall conditions in 1995 relative to Fall 1993. There may be some decline in a few specific congeners, but as shown later, some of these congeners may reflect a complexity in their biogeochemistry which precludes their use as simple markers for recently released PCBs.

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Chapter 4

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4.0 EFFECTS ASSESSMENT

This chapter 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.

4.1 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 4-1 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 4-1), 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.

4.1.1 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 4-1), 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 4-2 lists the coplanar non-*ortho* and mono-*ortho* congeners.

4.1.2 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).

4.1.3 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 4-2.

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.

4.2 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, 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 used to select TRVs for the present risk assessment. Long-term studies are also preferred since 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).

4.2.1 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). Since 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, 1993b).

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

4.2.2 Selection of TRVs for Benthic Invertebrates

4.2.2.1 Sediment Guidelines

Various guidelines and standards exist for concentrations of PCBs in sediment (Table 4-3). Measured and 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).

4.2.2.2 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 4-4). 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.

4.2.3 Selection of TRVs for Fish

In this Chapter, 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 4-5. 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 4-7) 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 concenctrations 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.

4.2.3.1 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 4-5, Figure 4-2). 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 4-23). 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 4-5).

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 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. 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. 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 17 mg PCBs/kg tissue (Table 4-25).
The NOAEL TRV for the pumpkinseed is 1.5 mg PCBs/kg tissue (Table 4-25).

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 4-6 and 4-23). 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 4-25).

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 (Table 4-7, Figure 4-3). Therefore, concentrations of TEQs in the pumpkinseed will be compared to the lowest appropriate NOAEL and LOAEL from the selected studies (Table 4-7). 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 4-7). 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 4-7), 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 4-25).
The NOAEL TRV for the pumpkinseed is 0.29 μg TEQs/kg lipid (Table 4-25).

Because salmonids are known to be highly sensitive to effects of dioxin-like compounds (Table 4-7), 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 will be 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 4-7) 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 4-25).

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 4-8).

4.2.3.2. 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 4-5). The study by Bengtsson (1980) on the minnow is selected as the lowest appropriate NOAEL (170 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 spottail shiner is 170 mg PCBs/kg tissue (Table 4-25).
The NOAEL TRV for spottail shiner is 15 mg PCBs/kg tissue (Table 4-25).

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 4-6 and 4-23).

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 (Table 4-7, Figure 4-3). The study by Olivieri and Cooper (1997) on the fathead minnow provides the lowest appropriate LOAEL and NOAEL from the selected applicable studies (Table 4-7). 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. 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 4-25).
The NOAEL TRV for the spottail shiner is 5.4 μg TEQs/kg lipid (Table 4-25).

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 4-8).

4.2.3.3 Brown Bullhead *(Ictalurus 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 4-5, Figure 4-2). Therefore, concentrations of PCBs in the brown bullhead will be compared to the lowest appropriate LOAEL and NOAEL from the selected applicable studies (Table 4-5). 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. Hatchability was significantly reduced in fish with an average total PCB concentration of 170 mg PCBs/kg, but not in fish with an average concentration of 15 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 4-5, 4-7), 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 17 mg PCBs/kg tissue (Table 4-25).
The NOAEL TRV for the brown bullhead is 1.5 mg PCBs/kg tissue is selected as (Table 4-25).

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 4-6).

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 4-7). The study by Elonen et al. (1998) on the channel catfish (Table 4-7) 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 4-23). 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 4-25).
The NOAEL TRV for the brown bullhead is 8.0 μg TEQs/kg lipid (Table 4-25).

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 4-8).

4.2.3.4 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 4-5, Figure 4-2). 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 yellow 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 4-5). 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, hatchability was significantly reduced in fish with an average total PCB concentration of 170 mg/kg, but not in fish with an average concentration of 15 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 4-5, 4-7), an interspecies uncertainty factor of 10 is applied.

On the basis of laboratory toxicity studies:

The LOAEL TRV for the yellow perch is 17 mg PCBs/kg tissue (Table 4-25).
The NOAEL TRV for the yellow perch is 1.5 mg PCBs/kg tissue (Table 4-25).

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 4-6 and 4-23).

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 (Table 4-7, Figure 4-3). 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 4-7). 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 4-7), 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 4-25).
The NOAEL TRV for the yellow perch is 0.29 μg TEQs/kg lipid (Table 4-25).

Because salmonids are known to be highly sensitive to effects of dioxin-like compounds (Table 4-7), 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) for a non-salmonid species (Table 4-7), 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 4-25).

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 4-8).

4.2.3.5 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 4-5, Figure 4-2). 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 4-5). 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, hatchability was significantly reduced in fish with an average total PCB concentration of 170 mg/kg, but not in fish with an average concentration of 15 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 4-5, 4-7), an interspecies uncertainty factor of 10 is applied.

On the basis of laboratory toxicity studies:

The LOAEL TRV for the white perch is 17 mg PCBs/kg tissue (Table 4-25).
The NOAEL TRV for the white perch is 1.5 mg PCBs/kg tissue (Table 4-25).

Two field studies were identified that examined the effects of PCBs on striped bass (Table 4-6). 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 4-23). 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 4-25).

On the basis of the field study:

The NOAEL TRV for the white perch is 3.1 mg PCBs/kg tissue (Table 4-25).

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 (Table 4-7, Figure 4-3). Therefore, concentrations of TEQs in the white perch will be compared to the lowest appropriate LOAEL and NOAEL from the selected studies (Table 4-7). 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 4-7). 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 4-7), 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 4-25).
The NOAEL TRV for the white perch is 0.6 μg TEQs/kg lipid (Table 4-25).

Because salmonids are known to be highly sensitive to effects of dioxin-like compounds (Table 4-7), 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 TEQs/kg lipid) and LOAEL (103 μg TEQs/kg lipid) for a non-salmonid species (Table 4-7), 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 4-25).

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 4-8).

4.2.3.6 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 4-5, Figure 4-2). 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 4-5). 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. Hatchability was significantly reduced in fish with an average total PCB concentration of 170 mg/kg, but not in fish with an average concentration of 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. 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 4-5, 4-7), 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 17 mg PCBs/kg tissue (Table 4-25).
The NOAEL TRV for the largemouth bass is 1.5 mg PCBs/kg tissue (Table 4-25).

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 4-6 and 4-23). 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 4-25).

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 4-7, Figure 4-3). Therefore, concentrations of TEQs in the largemouth bass will be compared to the lowest appropriate NOAEL and corresponding LOAEL from the selected studies (Table 4-7). 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 4-7). 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 4-7), 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 4-25).
The NOAEL TRV for the largemouth bass is 0.29 µg TEQs/kg lipid (Table 4-25).

Because salmonids are known to be highly sensitive to effects of dioxin-like compounds (Table 4-7), 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) 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 4-25).

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 4-8).

4.2.3.7 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 4-5, Figure 4-2). 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 4-5). 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, hatchability of eggs from adult fish with an average total PCB concentration of 170 mg PCBs/kg was significantly reduced in comparison to control fish. Hatchability was not reduced in eggs from adult fish with an average concentration of 15 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 4-5, 4-7), an interspecies uncertainty factor of 10 is applied.

On the basis of laboratory toxicity studies:

The LOAEL TRV for the striped bass is 17 mg PCBs/kg tissue (Table 4-25).
The NOAEL TRV for the striped bass is 1.5 mg PCBs/kg tissue (Table 4-25).

Two field studies were identified that examined the effects of PCBs on striped bass (Table 4-6). 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 4-25).

On the basis of the field study:

The NOAEL TRV for the striped bass is 3.1 mg PCBs/kg tissue (Table 4-25).

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 4-7, Figure 4-3). 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 4-7). 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 4-7). 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 4-7), 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 4-25).
The NOAEL TRV for the striped bass is 0.29 µg TEQs/kg lipid (Table 4-25).

Because salmonids are known to be highly sensitive to effects of dioxin-like compounds (Table 4-7), 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 4-7) 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 4-25).

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 4-8).

4.2.3.8 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 4-5, Figure 4-2). Therefore, concentrations of PCBs in the shortnose sturgeon will be compared to the lowest appropriate LOAEL and NOAEL from the selected applicable studies (Table 4-5). 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, hatchability of eggs from adult fish with an average total PCB concentration of 170 mg PCBs/kg was significantly reduced. No effects were seen for fish with an average concentration of 15 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 4-5, 4-7), an interspecies uncertainty factor of 10 is applied.

On the basis of laboratory toxicity studies:

The LOAEL TRV for the shortnose sturgeon is 17 mg PCBs/kg tissue (Table 4-25).
The NOAEL TRV for the shortnose sturgeon is 1.5 mg PCBs/kg tissue (Table 4-25).

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 4-6).

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 4-7, Figure 4-3). Therefore, the lowest NOAEL and corresponding LOAEL from the selected applicable studies (Table 4-7) 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 4-25).
The NOAEL TRV for the shortnose sturgeon is 0.29 μg TEQs/kg lipid (Table 4-25).

Because salmonids are known to be highly sensitive to effects of dioxin-like compounds (Table 4-7), 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) 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 4-25).

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 4-8).

4.2.4 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.

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4.2.4.1 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 4-9, Figure 4-4). 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 4-26).
The NOAEL TRV for the tree swallow is 0.01 mg PCBs/kg/day (Table 4-26).

Two field studies were identified that examined concentrations of PCBs in food of tree swallows in comparison to measures of reproductive effects (Table 4-10). 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 (USEP, 1998c). 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 4-26).

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 4-11 and Figure 4-5). Therefore, the lowest values from the selected applicable studies (Table 4-11), 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 4-11), 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 4-26).
The NOAEL TRV for the tree swallow is 0.0014 μg TEQs/kg/day (Table 4-26).

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, 1993a).

Two field studies were identified that examined the effects of dioxin-like compounds in the diets of tree swallows (Table 4-12). 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, 1998c). 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 4-26).

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 4-13 and Figure 4-6). Therefore, the lowest appropriate NOAEL and corresponding LOAEL from the selected applicable studies (Table 4-13) 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

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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 4-26).
The NOAEL TRV for the tree swallow egg is 0.33 mg PCBs/kg egg (Table 4-26).

Several field studies were identified that examined effects of PCBs on eggs of the tree swallow (Table 4-14). 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, in press). 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 4-26).

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 4-15 and Figure 4-7). 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 4-26).
The NOAEL TRV for the tree swallow is 0.01 μg TEQs/kg egg (Table 4-26).

Two field studies were identified that examined effects of dioxin-like compounds on tree swallows (Table 4-16). 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, 1998c). 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 4-26).

4.2.4.2 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 4-9, Figure 4-4). 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 4-26).
The NOAEL TRV for the mallard is 0.26 mg PCBs/kg/day (Table 4-26).

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 4-10).

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 4-11 and Figure 4-5). 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 4-11) (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, 1995a). 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 4-11), 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 4-26).
The NOAEL TRV for the mallard is 0.0014 μg TEQs/kg/day (Table 4-26).

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 4-12).

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 4-13 and Figure 4-6). Therefore, the lowest appropriate LOAEL and NOAEL from the selected applicable studies (Table 4-13) 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.

The LOAEL TRV for the mallard egg is 2.21 mg PCBs/kg egg (Table 4-26).
The NOAEL TRV for the mallard egg is 0.33 mg PCBs/kg egg (Table 4-26).

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 4-14).

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 4-15 and Figure 4-7). 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 4-15), 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.

The LOAEL TRV for the mallard egg is 0.02 µg TEQs/kg egg (Table 4-26).
The NOAEL TRV for the mallard egg is 0.01 µg TEQs/kg egg (Table 4-26).

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 4-16 and 4-23). 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 (1989b) 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 4-26).
The NOAEL TRV for the mallard egg is 0.005 μ TEgQs/kg egg (Table 4-26).

4.2.4.3 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 4-9, Figure 4-4). 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 4-9), 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 4-26).
The NOAEL TRV for the belted kingfisher is 0.01 mg PCBs/kg/day (Table 4-26).

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 taxomomic family as the kingfisher (Table 4-10).

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 4-11 and Figure 4-5). Therefore, the lowest appropriate values from the selected applicable studies (Table 4-11), 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 4-11), 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 4-26).
The NOAEL TRV for the belted kingfisher is 0.0014 μg TEQs/kg/day (Table 4-26).

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 4-12).

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 4-13 and Figure 4-6). Therefore, the lowest appropriate NOAEL and LOAEL from the selected applicable studies (Table 4-13) 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 4-26).
The NOAEL TRV for the belted kingfisher is 0.33 mg PCBs/kg egg (Table 4-26).

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 4-14).

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 4-15 and Figure 4-7). 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 4-15), 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 4-26).
The NOAEL TRV for the belted kingfisher egg is 0.01 μg TEQs/kg egg (Table 4-26).

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 4-16).

4.2.4.4 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 4-9, Figure 4-4). 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 4-26).
The NOAEL TRV for the great blue heron is 0.01 mg PCBs/kg/day (Table 4-26).

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 4-10).

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 4-11 and Figure 4-5). Therefore, the lowest appropriate values from the selected applicable studies (Table 4-11), 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 4-11), 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 4-26).
The NOAEL TRV for the great blue heron is 0.0014 μg TEQs/kg/day (Table 4-26).

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 4-12).

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 4-13 and Figure 4-6). Therefore, the lowest appropriate NOAEL and LOAEL (Scott, 1977) from the selected applicable studies (Table 4-13) 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 4-13), 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 4-26).
The NOAEL TRV for great blue heron eggs is 0.33 mg PCBs/kg egg (Table 4-26).

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 4-14).

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 4-15). 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 4-26).

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 4-16). 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 4-26).
The NOAEL TRV for the great blue heron is 0.3 μg TEQs/kg egg (Table 4-26).

4.2.4.5 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 4-9, Figure 4-4). 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 4-26).
The NOAEL TRV for the bald eagle is 0.01 mg PCBs/kg/day (Table 4-26).

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 4-10).

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 4-11 and Figure 4-5). Therefore, the lowest values from the selected applicable studies (Table 4-11), 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 4-11), 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 4-26).
The NOAEL TRV for the bald eagle is 0.0014 μg TEQs/kg/day (Table 4-26).

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 4-12).

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 4-13 and Figure 4-6). Therefore, the lowest appropriate NOAEL and corresponding LOAEL from the selected applicable studies (Table 4-13) 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 4-13), 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 4-26).
The NOAEL TRV for the bald eagle is 0.33 mg PCBs/kg egg (Table 4-26).

Several field studies were identified that examined the effects of PCBs in eggs of bald eagles (Table 4-14). 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

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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 4-26).

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 4-15 and Figure 4-7). 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 4-15) 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 4-15), 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 4-26).
The NOAEL TRV for the bald eagle is 0.01 μg TEQs/kg egg (Table 4-26).

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.

4.2.5 Selection of TRVs for Mammalian Receptors

4.2.5.1 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 4-17 and Figure 4-9). Therefore, the lowest appropriate NOAEL (0.32 mg/kg/day) and corresponding LOAEL (1.5 mg/kg/day) from the applicable studies (Table 4-17) 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 4-27). 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 4-27).
The NOAEL TRV for the little brown bat is 0.032 mg PCBs/kg/day (Table 4-27).

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 4-18 and Figure 4-10). 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 4-27).
The NOAEL TRV for the little brown bat is 0.0001 μg TEQs/kg/day (Table 4-27).

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, 1993a).

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.

4.2.5.2 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 4-17 and Figure 4-9). 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 4-17) 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 4-17) 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 (2 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 raccoons (Table 4-27).
The NOAEL TRV for the raccoon is 0.032 mg PCBs/kg/day (Table 4-27).

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 4-18). 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 4-18). 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 4-27).
The NOAEL TRV for the raccoon is 0.0001 µg TEQs/kg/day (Table 4-27).

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.

4.2.5.3 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 4-19 and Figure 4-8). The lowest effective dose in the selected applicable studies (Table 4-19) (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 4-27).
The NOAEL TRV for the mink is 0.01 mg PCBs/kg/day (Table 4-27).

Two field studies were identified that examined effects of PCBs in the diet of the mink (Table 4-20). 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 4-27).
The NOAEL TRV for the mink is 0.004 mg PCBs/kg/day (Table 4-27).

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 4-18). 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 4-18). 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 4-18, 4-21) 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 4-27).
The NOAEL TRV is for the mink is 0.0001 µg TEQs/kg/day (Table 4-27).

Two field studies were identified which examined effects of dioxin-like compounds on reproduction and survival in mink (Table 4-22). 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 4-27).
The NOAEL for the mink is 0.00008 μg TEQs/kg/day (Table 4-27).

4.2.5.4 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 4-17 and Figure 4-9). Because river otter and mink are in the same phylogenetic family (Table 4-23), 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 4-27).
The NOAEL TRV for the river otter is 0.01 mg PCBs/kg/day (Table 4-27).

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 4-20). 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 (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 4-27).
The NOAEL TRV for the river otter is 0.004 mg PCBs/kg/day (Table 4-27).

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 4-18 and Figure 4-10). 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 4-27).
The NOAEL TRV for the river otter is 0.0001 μg TEQs/kg/day (Table 4-27).

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 4-22). 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 4-27).
The NOAEL TRV for the river otter is 0.00008 μg TEQs/kg/day (Table 4-27).

4.3 Summary of Available Literature on Herpetofauna

This subchapter summarizes a review of the available literature on amphibian and reptile exposure and responses to PCBs (and dioxins). Typically, the literature documents concentrations in wild species but contains very few references to actual effect levels. No TRVs were developed for these species due to the lack of available data and because these species were not selected as the receptor species.

4.3.1 Amphibians

This subchapter presents available information on the PCBs (and dioxins) in frogs. No literature was available on the effects of PCBs on salamanders.

Most of the available literature on PCBs in frogs relates to body burdens without reference to effects. Rico et al. (1987) presented data on PCBs in frogs (*Rana perezi*) in a national park in Spain. They presented values ranging from 0.05 to 1.08 mg/kg wet weight for total PCBs. Dowd et al. (1985) reported 0.017 mg/kg PCBs as Aroclor 1260 in a frog leg sample from Lake Verret in Louisiana. They did not detect PCBs in legs or whole bodies of frogs from two other locations (at a detection limit of 0.1 mg/kg). They did not report whether their results were wet or dry weight. Niethammer et al. (1984) did not detect PCBs (as Aroclor 1254) in bullfrogs (*Rana catesbeiana*), bronze frogs (*Rana c. clamitans*), or southern leopard frogs (*Rana sphenocephala*) from Louisiana oxbow lakes at a detection limit of 0.2 mg/kg wet weight.

Similarly, most of the literature on dioxin in frogs reports body burdens but does not discuss effects. Korfmacher et al. (1986a and 1986b) measured 2,3,7,8-TCDD concentrations in bullfrog organs from a contaminated creek in Arkansas. Concentrations in livers ranged from 1.2 to 48 µg/kg wet weight. Concentrations in ovaries were similar to livers at 2.38 to 10.4 µg/kg wet weight while concentrations in muscle and skin were lower. They did not present information related to effects of these body burdens. Fanelli et al. (1980) reported 0.2 µg/kg TCDD (wet or dry weight not recorded) in a whole toad captured from an area contaminated with TCDD.

Few papers were found in the literature that discussed effects of PCBs on amphibians. Huang et al. (1998) studied the induction of cytochrome P450-associated enzymes in the northern leopard frog (*Rana pipens*). They injected BZ#126 dissolved in corn oil intraperitoneally into frogs at doses ranging from 0.2 to 7.8 mg/kg of body weight. They found increased enzyme activity at concentrations greater than 2.3 mg/kg body weight, and no effects at doses less than 0.7 mg/kg. They concluded that because they were unable to induce changes in enzyme activity at low PCB doses, these enzyme activities are unlikely to be a sensitive biomarker for coplanar PCB exposure in frogs. They also found no effect on body weight, liver weight, gross kidney appearance, behavior, skin or organ condition or mortality at the highest doses tested. It is not possible to convert these values into TRVs due to the method of dosing the frogs (injection) and the lack of clear ecological significance of the effects measured.

Reader et al. (1998) found that sites with higher concentrations of PCBs and PCDFs had cricket frogs (*Acris crepitans*) with higher sex-ratio reversals than areas with lower concentrations. They did not present concentrations in water or sediment that caused these effects.

Beatty et al. (1976) injected TCDD dissolved in olive oil intraperitoneally into tadpoles and adult bullfrogs at doses ranging from 0.25 to 1000 µg/kg of body weight. There was no effect on survival or metamorphosis of tadpoles at the highest concentrations. There was no effect on survival of adult bullfrogs. Adult frogs receiving a dose of 500 µg/kg of body weight exhibited decreased food intake, but frogs receiving the highest dose exhibited similar food intake to those receiving the lowest dose.

Jung and Walker (1997) exposed frog and toad eggs and tadpoles to 2,3,7,8-TCDD in water. The species they studied were American toad, leopard frog, and green frog. They found that American toad eggs and tadpoles exposed to 0.03 to 30 µg/l and green frog eggs and tadpoles exposed to 0.3 to 100 μg/l did not exhibit mortality, but that leopard frogs exposed to 3 μg/l exhibited significant mortality. They reported no difference in numbers of deformities for any treatment. Leopard and green frog tadpoles were lighter in pigmentation after exposure to 3 μg/l TCDD. Leopard frog tadpoles exposed to 3 μg/l TCDD had shorter lengths than control tadpoles whether or not the dosed tadpoles were on high or low fat diets or unfed. There was no effect on swimming speed in green frogs tadpoles or American toad tadpoles at any exposure level. American toads exposed as eggs seemed to metamorphose earlier, but there was no statistically significant difference between treatments.

Jung and Walker (1997) measured BCFs in amphibian eggs and tadpoles (1 to 4 for American toad eggs; 4 to 7 for leopard frog eggs; 1 to 3 for green frog eggs; 17 to 20 for unfed American toad tadpoles; and 8 to 10 for fed American toad tadpoles).

4.3.2 Reptiles

This section discusses the available literature PCBs and reptiles. Literature on alligators was not reviewed given their absence in the Hudson River.

4.3.2.1 Snakes

The literature is sparse with regard to effects of PCBs on snakes, although some information is available on concentrations in snake tissue. Sabourin et al. (1984) measured concentrations in two species of water snake from Southern Louisiana, *Nerodia rhombifera* and *Nerodia cyclopion*. They measured PCBs in fat bodies, liver, and muscle of *Nerodia rhombifera* from areas of varying soil contamination and found highest concentrations in fat bodies (up to 13.65 mg/kg wet weight total PCBs measured as Aroclors) and lowest in muscle (up to 0.02 mg/kg wet weight total PCBs measured as Aroclors). Livers had intermediate PCB concentrations (up to 0.66 mg/kg wet weight). In comparison, average whole snake (*N. rhombifera* and *N. Cyclopion*) concentrations ranged from 0.25 mg/kg to 0.58 mg/kg wet weight. Snake embryos contained 0.8 to 1.3 mg/kg Aroclor 1260 wet weight. Their results indicated that water snakes from areas contaminated with higher levels of PCBs had higher concentrations in their tissues. They did not document effects of these body burdens on the snakes or their behavior.

Heinz et al. (1980) measured PCBs in one northern water snake (*Nerodia sipedon*) and six common garter snakes (*Thamnophis sirtalis*) from islands in Lake Michigan. They also measured PCB concentrations in the stomach contents of the snakes. Total PCB concentrations (as Aroclor 1260) in the whole snakes ranged from 1.3 to 5.8 mg/kg wet weight while concentrations in stomach contents ranged up to 1.3 mg/kg wet weight. They presented no data on the effects of these body burdens in snakes. Dowd et al. (1985) did not detect PCBs as Aroclor 1260 in water snake tissue from two sites in Louisiana (detection limit of 0.1 mg/kg – wet or dry weight not recorded). Niethammer et al. (1984) reported less than 0.2 to 1.61 mg/kg wet weight PCBs (measured as Aroclor 1254) in cottonmouth snakes (*Agkistrodon piscivorus*) and less than 0.2 to 1.62 mg/kg wet weight PCBs in water snakes (*N. cyclopion* and *N. rhombifera*) in Louisiana oxbow lakes.

Dioxin levels have also been measured in snake tissue, with no mention of effects. Fanelli et al. (1980) detected 2.7 µg/kg TCDD in snake liver and 16 µg/kg in snake adipose tissue from a snake caught in a contaminated area. They did not report whether these results were wet or dry weight.

Due to the lack of effects concentrations, doses, or body burdens in the literature, no TRVs could be developed for snakes.

4.3.2.2 Turtles

The greatest available body of information is available for PCBs in snapping turtles (*Cheldrya serpentina*). Helwig and Hora (1983) reported a range of less than 0.025 to 0.086 mg/kg PCBs in leg meat and a range of less than 0.2 to 60.5 mg/kg PCBs in turtle fat in Minnesota (wet or dry weight not recorded). Bryan et al. (1987a and 1987b) measured PCB congeners in a turtle from an area with low contamination and one from an area with high contamination in New York. They reported congener concentrations in fat and total PCB concentrations in fat and various organs. The turtle from the contaminated area had much higher PCB concentrations in its organs (a range from 13 mg/kg in lungs to 1600 mg/kg in fat – wet or dry weight not reported). They did not report effects data. Olafsson et al. (1983) measured PCB congeners and homologies in two turtles, one from Lake Ontario and one from the Hudson River near Hudson Falls. They measured 3,608 mg/kg wet weight total PCBs in fat from the Hudson River turtle and 633 mg/kg wet weight PCBs in the Lake Ontario turtle. They did not report effects, but commented that their study indicated that snapping turtles could store high concentrations of PCBs in their fat without apparent effect. Hebrew et al. (1993) measured 28 PCB congeners and Aroclors 1254 and 1260 in snapping turtle adults and eggs from Southern Ontario. Detected concentrations of total PCBs in muscle ranged from less than 0.2 µg/kg to 655.28 µg/kg wet weight. They reported that the larger, heavier, older turtles had the highest PCB concentrations. They cited an Environment Canada (1991) report that found that turtles from Lake Ontario with the lowest contaminant levels also had the greatest degree of reproductive success, but did not provide the Environment Canada data.

Stone et al. (1980) measured PCBs in snapping turtle tissue from water bodies in upstate New York including the Hudson River. They found that turtles from the Hudson River had higher levels of PCBs in their tissues than turtles from other water bodies. They reported 306 to 7990 mg/kg PCBs in fat, 0.54 to 683 mg/kg PCBs (wet weight) in liver, and 0.19 to 27.62 mg/kg PCBs (wet weight) in muscle from Hudson River turtles. They also reported 10.4 to 42.9 mg/kg PCBs wet weight in Hudson River snapping turtle eggs. They noted that despite these elevated body burdens, snapping turtles appeared to be abundant in the Hudson River.

Other data are available for PCBs in snapping turtle eggs. Struger et al. (1993) compared PCB concentrations in snapping turtle eggs from the Great Lakes – St. Lawrence River basin to eggs from a background location. Turtle eggs from the Great Lakes averaged around 1 mg/kg wet weight total PCBs, while those from the background location had an average of 0.187 mg/kg wet weight PCBs. In a comparison between Lake Ontario snapping turtle eggs and eggs from a background area, Bishop et al. (1991) found a correlation between PCB concentrations in eggs and hatching success of eggs and deformities in hatchlings. The lowest hatching success and highest level of deformities were recorded where egg concentrations ranged from 257 to 3322 µg/kg wet weight (measured as the sum of six congeners). In comparison, healthy hatch success rates and low numbers of deformities were measured where egg PCB concentrations were 28 to 76 µg/kg. Bishop et al. (1994) measured seven PCB congeners in snapping turtle eggs from Lake Ontario and found no correlation between PCB concentrations in eggs and maternal body mass, carapace length, carapace width, and plastron length.

Ryan et al. (1986) measured dioxins in snapping turtles from the upper St. Lawrence River. They reported less than 2 to 107 pg/g 2,3,7,8-TCDD in liver samples and 232 to 474 pg/g TCDD in fat samples (wet or dry weight not reported). They noted that the turtles appeared to be normal in appearance and behavior.

A few researchers reported effects information as well as body burden data for PCBs in snapping turtles. Albers et al. (1986) measured PCBs in Maryland and New Jersey snapping turtles and looked for effects on growth and blood and blood plasma characteristics. They reported up to 291 mg/kg lipid PCBs in visceral fat in the New Jersey turtles. They did not detect significant differences in growth or blood chemistry in any turtles examined.

Effects of PCBs on turtle eggs have also been reported. PCBs has been demonstrated to reverse gonadal sex in a turtle species that exhibits temperature-dependent sex determination. Bergeron et al. (1994) and Crews et al. (1995) found that two PCB derivatives (2',4',6'-trichloro-4-biphenylol and 2',3',4',5'-tetrachloro-4-biphenylol) significantly reversed the sex of red-eared slider turtles (*Trachemys scripta*) when applied to the eggs at doses of 10 and 100 µg.

TRVs could not be developed for the herps species due to the lack of quantitative data on effects, concentrations, and doses.

Chapter 5

5.0 RISK CHARACTERIZATION

Risk Characterization evaluates the likelihood of adverse effects occurring as a result of exposure to a stressor, such as PCB contamination, and discusses the qualitative and quantitative assessment of risks to ecological receptors with regard to toxic effects. Risk characterization is made up of two steps, risk estimation and risk description (USEPA, 1992a; 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 next.

In the toxicity quotient approach, potential risks to ecological receptors are assessed by comparing measured or modeled concentrations (Chapter 3) to toxicity benchmarks developed in Chapter 4. The quantitative assessment relies on a toxicity quotient approach in which measured or modeled concentrations are compared to appropriate benchmarks derived for the receptors. PCBs are described as total PCBs (Tri+) as well as TEQ.

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

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

Toxicity quotients exceeding one are typically considered to indicate potential risk to ecological receptors. The toxicity quotient 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 (5.1.1);
- Health and maintenance of local fish populations (5.1.2) 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., su rvival and reproduction) of local wildlife including:
  - semi-piscivorous/piscivorous and insectivorous birds and waterfowl (5.1.3);
  - semi-piscivorous/piscivorous, insectivorous, and omnivorous mammals (5.1.4).
- Protection of threatened and endangered species (5.1.5).

- Protection of significant habitats (5.1.6).

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

5.1.1 Does the Benthic Community Structure Reflect the Influence of PCBs?

The benthic macroinvertebrate community is closely associated with sediment and pore water, relying on these media for habitat, food, and exchange of gases. Therefore, the characteristics of the benthic invertebrate community are strongly affected by, and reflect, the quality of the sediment that the organisms inhabit. Benthic macroinvertebrate communities in the TI Pool (Upper Hudson River) and at selected significant habitats in the Lower Hudson River were sampled and their ecological metrics were analyzed for each sampling station. The overall health and structure of the benthic community can affect organisms, such as fish, that depend upon the benthic community for food. An impoverished or unhealthy benthic community can affect not only the animals feeding directly on benthic invertebrates, but also upper trophic level receptors.

Benthic invertebrate communities and sediment PCB concentrations were sampled at ten locations in the Hudson River (five in the Upper Hudson River and five in the Lower Hudson River) to determine if PCB concentrations have affected benthic community structure. These analyses are used as measurement endpoints for evaluating benthic community structure as food source for local fish and wildlife community assessment endpoint. A detailed report of the benthic invertebrate study is provided in Appendix H. Upper and Lower Hudson River sediment and water concentrations are compared to New York State and federal guidelines (Subchapter 5.1.3) as another measurement endpoint in a weight-of evidence approach.

5.1.1.1 Measurement Endpoint: TI Pool (Upper Hudson River) Benthic Invertebrate Community Analysis

The TI Pool Benthic Invertebrate Community PCB Study investigated macroinvertebrate communities in areas of varying PCB concentrations within the lower reach of the TI Pool (RM 188.5 to 191.5). The objectives of the TI Pool study were to create a general profile of community characteristics and determine whether ecologically-based effects of PCBs could be inferred. Sampling focused on the overall community characteristics and sediment properties at five selected stations (Stations 3 to 7; see Figure 2-3B) that were selected based on prior PCB screening results (see Appendix B for details). Replicate macroinvertebrate samples were characterized by examining species/taxa richness (number of taxa), abundance (number of individuals), species diversity (a combination of richness and equitability), biomass (grams), and community similarity.

The selection of the five ecological stations was, by necessity, a compromise between habitats and PCB concentrations. Differences in community characteristics between stations were analyzed in relation to physical and chemical properties that may contribute to the observed variations, such as differences in grain size, total organic carbon, total PCBs, and metals.

A total of 86 taxa were collected from the 30 Ekman grabs taken at the five TI Pool stations. Table 5-1 lists the taxa in rank order, excluding the colonial bryozoans, which could not be individually counted. Approximately 90% of the total taxa collected were members of five major taxonomic groups: isopods (sow bugs); chironomids (midge larvae); oligochaetes (aquatic worms); amphipods (scuds/sideswimmers); and pelecypods (mussels and clams). The numerical abundance (individuals/m^2) and the percent abundance data for each of the five major groups are presented in Table 5-2. Profiles of dominant invertebrate receptor groups can be found in Appendix C.

The number of taxa/groups collected is similar to results found by a General Electric study, wherein a total of 100 macroinvertebrate species were collected in 86 cores (3-in diameter) taken during September 1997 (Exponent, 1998). These cores were taken from below Roger's Island and Griffin Island in the TI Pool and at Stillwater.

Species richness, abundance, diversity, evenness and dominance at each of the five stations are summarized in Table 5-3. Species richness (i.e., number of taxa), abundance, and biomass are direct measurements. The Simpson Index (D_s) was used to calculate species diversity. The Simpson Index is more sensitive to the relative abundance of species and to dominance as opposed to evenness of species abundance (Magurran, 1988). Given the shifts in relative abundance in the TI Pool (Table 5-2), the Simpson Index was considered the most appropriate choice. Evenness (E_s) is a measure of the distribution of individuals among the component taxa; the higher the E_s, the more even the distribution. Formulas to calculate species diversity, evenness, and community similarity are provided in Appendix H.

When all species/taxa collected (i.e., benthic and epibenthic) were considered, species diversity, evenness, and taxa richness were higher at Stations 3, 4, and 6 than at Stations 5 and 7. Dominance was higher at Stations 5 and 7, indicating lower equitability at these stations (Table 5-3).

To quantitatively assess overall community similarity in a more robust fashion, the Morisita Index (I_m) was used to compare all species, rather than just the dominant taxa. The Morisita Index of community similarity is based on Simpson's index of dominance (l) and ranges from zero (no similarity) to 1.0 (identical). Stations with similar indices are considered to be more similar in community structure than stations with large differences in their indices.

A dendrogram was drawn based on the Morisita Indices calculated for the five TI Pool Stations to provide a visual representation of community similarity (Figure 5-1). The benthic invertebrate communities were divided into two distinct clusters. The first cluster (Cluster 1), comprised of Stations 5 and 7, exhibited lower species richness, species diversity, dominance, and diversity than Cluster 2, which was made up of Stations 3, 4, and 6. Differences in community characteristics may be a function, at least in part, of sediment characteristics. The stations in Cluster 1 have a higher proportion of fine-grained silty material than the stations in Cluster 2 (Figure 5-2). With the exception of the more even grain size distribution at Station 3, Cluster 2 can be characterized as a predominately fine sand habitat with a lower percentage of silt. Total organic carbon (TOC), often associated with fine-grained sediments, was greatest at

Cluster 1 stations (Figure 5-3). These sediment characteristics may contribute to the differences seen between clusters.

When total PCB concentrations at the five TI Pool stations were examined (Figure 5-4), concentrations could be divided into the same two general groups that were seen in the Morisita Index analysis (Figure 5-1). Using an analysis of variance (ANOVA), the total PCB concentration at Station 5 (29.32 mg/kg) was significantly greater ($p<0.05$) than at Stations 3 (9.29 mg/kg), 4 (10.49 mg/kg), and 6 (14.33 mg/kg). The total PCB concentration at Station 7 (18.51 mg/kg) was significantly greater ($p<0.05$) than at Stations 3 and 4 (see Appendix H). There were no significant differences in total PCBs between Stations 5 and 7 and among Stations 3, 4, and 6. Stations 3, 4, and 6 constitute a lower total PCB concentration cluster and Stations 5 and 7 comprise a higher total PCB concentration cluster. When PCB concentrations were normalized to TOC, there were no significant differences between stations. TOC-normalized concentrations may be a more accurate representation of available PCBs; however the available data were not sufficient for an evaluation of PCB-bioavailability in the TI Pool.

Examination of the data showed that benthic invertebrate populations at Stations 5 and 7 are dominated by the isopod *Caecidotea racovitzai* (Table 5-2). Isopods are crustaceans that are often numerous in sediments having high organic content and low oxygen levels (NYSDEC, 1993). Data were reanalyzed, excluding epibenthic invertebrates, to examine community structure without the isopod *Caecidotea racovitzai.* The infaunal analysis indicates that species diversity, dominance, and evenness are similar between all five stations when only infauna are considered (Table 5-3). The numerical abundance of infauna at Station 7 (one of the two Cluster 1 stations) is low compared to the other four stations. However, Station 7 had the highest biomass of the TI Pool stations due to the presence of the eastern elliptio mussel (*Elliptio* sp.). In contrast, Station 5, with the highest overall number of individuals, had the lowest total biomass of any of the TI Pool stations (Figure 5-4), because of the many juvenile *Caecidotea racovitzai* found at this station.

5.1.1.2 Measurement Endpoint: Lower Hudson Benthic Invertebrate Community Analysis

Benthic macroinvertebrate communities were characterized at five significant habitat locations in the Lower Hudson River, including the four sites comprising the NERR (see Chapter 2). Macroinvertebrates in the Lower Hudson River represent a heterogeneous group of organisms with a wide range of life history strategies and environmental tolerances (Table 5-4). Studies and reviews of invertebrates found in the Lower Hudson River indicate that they are distributed in distinct patterns, corresponding to their distance to the mouth of the Hudson, where the saltwater of the ocean salinizes the water (e.g., Ristich et al., 1977; Weinstein, 1977; Gladden et al., 1988; and Moran and Limburg, 1986). The lower reaches, below RM25, support a typical marine assemblage of benthic invertebrates, including marine oligochaetes, polychaetes, and crustaceans. The middle reaches, from RM25 to RM50, have a mixture of freshwater and marine benthic invertebrates, and the upper reaches, above RM50, are dominated by freshwater arthropods and oligochaetes.

The benthic macroinvertebrate communities collected in the Lower Hudson River reflect the variety of habitats and conditions found along the river (Table 5-4). Because of the habitat diversity and salinity gradient found in the Lower Hudson River, it is difficult to make direct comparisons between any of the stations. A summary of indices and abundance data for Lower Hudson River benthic communities is provided in Table 5-5. Stations 14, 17, and 18 had a higher proportion of fine-grained sediments than Stations 12 and 15 (Figure 5-6). However, grain size was not closely correlated with TOC (Figure 5-7) or total PCB concentrations (Figure 5-8). Stations 14 (Tivoli Bay), 15 (Esopus Meadows), and 18 (Piermont Pier), with mean total PCB concentrations of 0.37, 0.87, and 0.48 mg/kg respectively, had higher species diversity indices than Stations 17 (Iona Island) and 12 (Stockport Flats), which had average total PCB concentrations of 1.31 and 1.23 mg/kg, respectively (Table 5-5).

The invasion of the zebra mussel (*Dreissena polymorpha*), first detected in the Hudson River in May 1991 (Strayer et al., 1996), may increase the bioavailability of PCBs to benthic and pelagic organisms found in the river. The Hudson River zebra mussel population reached 550 billion animals (4,000/m^2) by the end of 1992, constituting more than 70% of the zoobenthic biomass and filtering a volume equivalent to the entire water column in one day (Strayer et al., 1996). Morrison et al. (1998) estimated the effects of zebra mussels on the trophodynamics of PCB congeners in western Lake Erie. The reduction concentration of particulate organic carbon (POC), attributed to the prodigious filter feeding of large zebra mussel populations, was theorized to have caused increases, ranging from 2.9% to 9.3%, in the freely dissolved concentrations of PCB congeners. The effect of the zebra mussel on Lower Hudson River PCB concentrations is not known at this time.

Community-level measurements may be confounded by the influence of abiotic factors (e.g., grain size) and the difficulty of distinguishing between directional (e.g., response to a trend or gradient) and nondirectional (e.g., seasonal or annual) variability (Ingersoll et al., 1998). Although the benthic invertebrate community analyses could not distinguish any clear effects from PCBs in the Upper or Lower Hudson River, they are part of a weight-of-evidence approach in evaluating sediment quality. Another component of determining sediment quality is the comparison of PCB concentrations to applicable guidelines, which is discussed in the following section.

5.1.2 Do Measured and Modeled Sediment Concentrations Exceed Guidelines?

5.1.2.1 Measurement Endpoint: Comparison of Sediment PCB Concentrations to Guidelines

Mean concentrations of PCBs at each station were compared to sediment guidelines for PCBs (Table 5-6). Consensus-based sediment effect concentrations (SECs) for PCBs in the Hudson River Basin were developed to support an assessment to sediment-dwelling organisms (NOAA, 1999a). The consensus-based SECs:

- Provide a unifying synthesis of existing sediment quality guidance (SQG);
- Reflect causal rather than correlative effects; and,
- Account for the effects of PCB mixtures.

The SEC for PCBs refers to all of the polychlorinated biphenyls found in the Hudson River, plus the degradation products and metabolites of these chemicals. The SECs do not consider the potential for: 1) bioaccumulation in aquatic species; or 2) potential effects that could occur throughout the food web as a result of PCB bioaccumulation. The Hudson River SECs, as well as the NYSDEC Technical Guidance for Screening Contaminated Sediments (NYSDEC, 1998c), were used as the primary sediment guidelines for comparison in this baseline ERA.

The Hudson River SECs provide threshold effect concentration (TEC), mid-range effect concentration (MEC), and extreme effect concentration (EEC). 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.

The Hudson River TEC (0.04 mg/kg), MEC (0.4 mg/kg), and EEC (1.7 mg/kg) were exceeded at all TI Pool stations, which had mean concentrations ranging from 9.29 to 29.32 mg/kg (Table 5-6). In the Lower Hudson River, all stations had mean total PCB concentrations (range of 0.367 mg/kg to 1.313 mg/kg) above the TEC and MEC values, except Stations 14 (Tivoli Bays) and Station 18 (Piermont Pier), which were slightly below the MEC. All Lower Hudson River stations had mean total PCB concentrations below the EEC (1.7 mg/kg).

Although all TI Pool stations had viable benthic macroinvertebrate communities that could support local fish populations, the PCB concentrations measured at these stations indicate that some benthic species may be adversely affected by the levels of PCBs present in the sediment.

Tables 5-8 and 5-9 provide the ratios of observed and modeled sediment concentrations to a number of sediment guidelines established for the protection of benthic life (NOAA, 1999a; NYSDEC, 1998; Washington, 1997).

The threshold effect concentration (TEC), mid-range effect concentration (MEC), and extreme effect concentration (EEC) were exceeded at the Upper Hudson River locations for both the average and upper 95% UCL. All Lower Hudson River locations exceeded the TEC and the majority of locations also exceeded the MEC.

5.1.3 Do Measured and Modeled PCB Water Concentrations Exceed Appropriate Criteria and/or Guidelines for the Protection of Wildlife?

5.1.3.1 Measurement Endpoint: Comparison of Water PCB Concentrations to Benchmarks

Benthic macroinvertebrates are also exposed to PCBs in the water column. Water column samples taken from January through September 1993, from RM194.6 to RM156.5, were compared to NYSDEC Water Quality Criteria (WQC) (1998c; based upon USEPA, 1991a)

(Table 5-7). The average Upper Hudson River water column PCB concentration of 0.071 µg/L exceeded the chronic freshwater WQC (0.014 µg/L). The NYSDEC surface water standard for protection of wildlife (including bioaccumulation) of 1.2 x 10^{-4} g/L (NYSDEC, 1998a) was exceeded by all water column samples collected. Again, the PCB concentrations measured at these stations indicate that some aquatic species and wildlife may be adversely affected by the levels of PCBs present in the water.

Comparison of total PCB concentrations in the sediment and water column to guidelines indicates that the level of PCBs present in the Hudson River, particularly in the Upper River, may cause adverse effects to aquatic life. Fish and wildlife may be more severely affected than aquatic invertebrates by the concentrations of PCBs present in the Hudson River, owing to bioaccumulation and biomagnification of PCBs, as discussed in the following sections.

Table 5-10 presents the results of the comparison between measured whole water concentrations and appropriate benchmarks, while Table 5-11 presents the results for the comparison between HUDTOX modeled whole water concentrations and appropriate benchmarks.

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

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

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

Table 5-12 shows the results of the comparison between measured forage fish body burdens and toxicity reference values developed for pumpkinseed and spottail shiners under current conditions. The measured forage fish concentrations are obtained from the USEPA/NOAA dataset and consist of all fish less than 10 cm in length. In the Lower Hudson River, the shown values are for spottail shiners (the only fish less than 10 cm in length) and typically spottail shiners comprise the majority of the small fish collected at any given station.

This table shows that measured forage fish concentrations exceed the field-based NOAEL derived for pumpkinseed, but only exceed the laboratory-based NOAEL derived for spottail shiners in the TI Pool.

Tables 5-13 through 5-15 present the results of the comparison between predicted percentiles of pumpkinseed and spottail shiner to selected toxicity reference values on a total PCB basis (expressed as Tri+) under future conditions. These two species show very different results. While pumpkinseed consistently exceed a toxicity quotient of one on a NOAEL basis for the TI Pool and Stillwater for the 25th percentile, median, and 95th percentile, the spottail shiner only exceeds one until 1996 for the 95th percentile. On a 95th percentile basis, pumpkinseed exceed one at all three modeling locations until the end of the modeling period (2018) for the

95th percentile. This is interpreted to mean that 95% of pumpkinseed will experience the shown toxicity quotient or less for that year.

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

Tables 5-16 through 5-19 present the results of the comparison between predicted percentiles of pumpkinseed and spottail shiner to selected toxicity reference values on a TEQ basis under future conditions. Tables 5-16 and 5-17 present the results for pumpkinseed based on comparisons to a NOAEL and LOAEL, respectively. On a NOAEL basis, the toxicity quotients exceed one on a 95th percentile basis until approximately 2010 for both the TI Pool and Stillwater. On a LOAEL basis, the toxicity quotients exceed one until approximately 1999.

Tables 5-18 and 5-19 presents the results for the spottail shiner. The TRVs derived for the spottail shiner differ from the pumpkinseed by several orders of magnitude (0.6 µg/kg on a LOAEL basis for pumpkinseed versus 103 µg/kg on a LOAEL basis for spottail shiners). Consequently, spottail shiner toxicity quotients are quite different from pumpkinseed. Toxicity quotients for spottail shiners do not exceed one at any time 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 5-20 and 5-21 present the results of the comparison between predicted percentiles of brown bullhead concentrations to selected toxicity reference values under future conditions. On a NOAEL basis, brown bullhead toxicity quotients consistently exceed one for both the median and 95th percentiles at all three locations. Within the TIP, toxicity quotients exceed one for most of the modeling period for all three percentiles, suggesting the potential for significant risk to this omnivorous demersal fish.

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

Tables 5-22 and 5-23 present the results of the comparison between predicted percentiles of brown bullhead concentrations to selected toxicity reference values under future conditions. The toxicity reference values derived for the brown bullhead are significantly higher than for the other fish species (see Table 4-16). Brown bullhead results show that the estimated toxicity quotients do not exceed one for any percentiles or locations on the basis of the selected toxicity reference values.

5.2.1.5 Measurement Endpoint: Comparison of Observed Total PCB and TEQ Basis Fish Body Burdens to Toxicity Reference Values for Largemouth Bass and Brown Bullhead

Table 5-24 shows the results of the comparison between observed average and 95% UCL concentrations for largemouth bass and brown bullhead from the NYSDEC data set to selected toxicity reference values under current conditions. This table shows that toxicity quotients on both a total PCB and TEQ basis exceed one for all locations for the NOAEL. Both the average and 95% UCL (or maximum, as appropriate) toxicity quotients exceed one. This suggests the potential for significant risk to these fish species, including river mile 113 in the Lower Hudson River.

For the LOAEL-based comparison, the results in Table 5-24 suggest that on a total PCB body burden basis, toxicity quotients exceed one at all locations. On a TEQ basis, the toxicity quotients exceed one for both species at all locations.

5.2.1.6 Measurement Endpoint: Comparison of Measured Total and TEQ-Based PCB Fish Body Burdens to Toxicity Reference Values for White and Yellow Perch Based on NYSDEC Data

Table 5-25 presents the results of the comparison between measured body burdens of white perch and yellow perch Tri+ PCB concentrations to selected toxicity reference values under current conditions. The measured white perch body burdens exceed the field-based NOAEL derived in Chapter 4 at RM113 for 1996 and at RM152 during 1994 and on a maximum basis at RM152 during 1996. The 1996 concentrations reflect an increase in concentrations relative to prior years, suggesting that although a TRV may not be exceeded in a particular year, variability in body burdens can cause exceedances in subsequent years.

The comparisons for yellow perch show that both the laboratory-based NOAEL and LOAEL are exceeded at the TI Pool for the average, 95% UCL, and maximum. For the NOAEL, predicted toxicity quotients exceed ten. The NOAEL is exceeded at Stillwater, and the LOAEL on a maximum basis. Note that measured body burdens are expressed on a wet weight basis for the standard fillet, while toxicity reference values have been derived on a whole body basis. Thus, an adjustment is required to express the measured body burden on a whole body basis. Unfortunately, there were no data available with which to make this conversion, consequently, the toxicity quotients were calculated on a fillet basis. This is likely to underestimate true body burdens.

On a lipid-normalized basis, all NOAEL-based comparisons are exceeded for both species at all locations. On a LOAEL basis, white perch exceed at all locations, and yellow perch at all locations except RM113 during 1994. As body burdens are expressed on a lipid-normalized basis, there is no need to convert from fillet to whole body for this comparison.

5.2.1.7 Measurement Endpoint: Comparison of Modeled Total PCB Fish Body Burdens to Toxicity Reference Values for White and Yellow Perch for the Period 1993 - 2018

Tables 5-26 through 5-28 present the results of the comparison between predicted percentiles of white perch and yellow perch Tri+ PCB concentrations to selected toxicity reference values under future conditions. For white perch (Table 5-24), this table shows that toxicity quotients on a NOAEL basis consistently exceed one for the 95th percentile in the TI Pool and at Stillwater. On a median basis, Table 5-26 shows that the estimated toxicity quotients exceed one until 2013 and 2008 for the TI Pool and Stillwater, respectively. At river mile 154,

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just above the Federal Dam, the toxicity quotients begin to fall below one in 1993 – 1999 on a median basis.

Tables 5-27 and 5-28 present the results for the yellow perch. On a NOAEL basis (Table 5-25), the estimated toxicity quotients exceed one until 2017 for all three locations for the 95th percentile. The median estimated toxicity quotients exceed one for virtually the entire modeling period in the TI Pool and at Stillwater, and exceed one until 2006 at RM154, just above the Federal Dam.

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, since the presented results are based on predicted standard fillet concentrations, true risks are likely underestimated for these two species.

5.2.1.8 Measurement Endpoint: Comparison of Modeled TEQ Basis Body Burdens to Toxicity Reference Values for White and Yellow Perch for the Period 1993 - 2018

Tables 5-29 through 5-32 present the results of the comparison between predicted percentiles of white perch and yellow perch TEQ–based PCB concentrations to selected toxicity reference values under future conditions. Tables 5-29 and 5-30 compare modeled white perch concentrations to NOAEL and LOAEL based toxicity reference values, respectively. On a NOAEL basis, the 95th percentile exceeds one for the entire modeling period in the TI Pool, until 2016 at Stillwater, and until 1998 at RM154. The median-based toxicity quotients exceed one until 2003 in the TI Pool and 1997 at Stillwater. The median-based toxicity quotient at RM154 does not exceed one. On a LOAEL basis, the 95th percentile exceeds one until 2013 in the TI Pool and 2004 at Stillwater. This is interpreted to mean that 95% of the white perch population will experience toxicity quotients at the values shown in the appropriate column or less.

Results for yellow perch are shown in Tables 5-31 and 5-32. These tables show similar results to white perch, but yellow perch predicted toxicity quotients fall below one a few years before white perch.

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

5.2.1.9 Measurement Endpoint: Comparison of Modeled Tri+ PCB Fish Body Burdens to Toxicity Reference Values for Largemouth Bass for the Period 1993 - 2018

Table 5-33 presents the results of the comparison between modeled largemouth bass concentrations and the field-based TRV derived in Chapter 4 under future conditions. This table shows that predicted toxicity quotients exceed one for all percentiles and locations for the duration of the modeling period.

5.2.1.10 Measurement Endpoint: Comparison of Modeled TEQ Based Fish Body Burdens to Toxicity Reference Values for Largemouth Bass for the Period 1993 - 2018

Tables 5-34 and 5-35 present the results of the comparison between modeled largemouth bass concentrations and the laboratory-based TRVs derived in Chapter 4 for TEQ under future conditions. On a NOAEL basis, predicted largemouth bass toxicity quotients exceed one for the 95^{th} percentile in the TI Pool and at Stillwater.

5.2.1.11 Measurement Endpoint: Comparison of Observed Striped Bass Concentrations to Toxicity Reference Values on a Total (Tri+) and TEQ PCB Basis

Tables 5-36 presents the results of the comparison between observed striped bass concentrations at various river miles to toxicity reference values developed in Chapter 4 under current conditions. This table shows that at river mile 152, just below the Federal Dam, predicted toxicity quotients exceed one for both tissue and egg-based concentrations. As mentioned previously, striped bass wet weight body burdens are expressed on a standard fillet basis. Although an adjustment to a whole body basis is required, there was not enough data to make this adjustment. Thus, true risks are likely underestimated.

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

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

Tables 5-10 and 5-11 present the results of the comparison between observed and modeled whole water PCB concentrations and appropriate criteria and guidelines. These tables show that observed and predicted water concentrations exceed appropriate and relevant criteria. 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 and guidelines are based on total PCBs (the sum of all congeners).

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

5.2.3.1 Measurement Endpoint: Evidence from Field Studies

Extensive observational data for Hudson River fish are available for the Lower Hudson River (e.g., see Klauda et al. 1988) and limited data are available for the Upper Hudson River (above Federal Dam). The strengths and limitations of observational data have been previously described. Based on the available data, the following observations provide insights into the current and 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 epidemiological studies (other than for tree swallows) that have directly addressed impacts associated with the presence of PCBs to Hudson River fish and wildlife.

1. Collections made by NYSDEC over the past few decades indicate that populations of the receptor species - largemouth bass, brown bullhead, spottail shiner, yellow perch, pumpkinseed – have continued to be present in the upper area in reaches where exposure to PCBs is occurring. The continued presence of these and other species over this period of time indicates that exposure levels of PCBs are not high enough ***to prevent*** reproduction of these species or recruitment of new individuals to these areas. 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 Chapter 5.2 that compares measured and predicted body burdens to TRV values is required to judge the possible magnitude of these risks.

2. 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 stickelback, and sharksucker) have been collected within the last 20 years." To our knowledge, there are no obvious losses of species that have occurred over the past few decades during which PCB exposures have been greatest. As noted above, while this true, it is also not possible to say from these data that reproductive or recruitment rates have not been influenced by PCB exposure. Such influences may not be discernable within the timeframe of the monitoring studies and has not been specifically examined in relation to PCB exposure. The analysis of potential effects presented in Chapter 5.2 must be relied upon in order to determine the magnitude of potential risks that PCBs have on reproduction and recruitment rates.

3. Studies of the abundance of short nose sturgeon indicate that this species is reproducing in the Lower Hudson River (below the Federal Dam) and that the population numbers are increasing. The shortnose sturgeon has been listed as a federally endangered species since 1967. 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). The species matures late and spawns infrequently (maturity at 7-10 years). While available data indicate that the population growth of shortnose sturgeon in the Hudson is positive (i.e., numbers are increasing rather than decreasing), it is not possible to quantify from these data the extent to which PCB exposures might impair or reduce these population growth rates. The analysis provided in Chapter 5.2 provides information that can be used to evaluate this possibility.

4. Population data indicate that white perch, a semi-anadromous fish in the Lower Hudson River, have 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. 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. studies 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. Chapter 5 provides 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 Hudson River Insectivorous Bird Species (Tree Swallow)

5.3.1 Do Measured and 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 of Total PCBs (i.e., Tri+) to Insectivorous Birds (Tree Swallow) and Predicted Egg Concentrations Using 1993 Data

Table 5-37 provides the results of the comparison between modeled dietary doses and egg concentrations of total PCBs and toxicity reference values for the tree swallow using the 1993 data under current conditions. Dietary doses to adult tree swallows and egg concentrations are estimated by applying biomagnification factors to observed benthic invertebrate PCB concentrations on a Tri+ basis from the USEPA Phase 2 dataset.

The NOAEL-based 95% UCL comparisons exceed one using the Phase 2 1993 dataset (Table 5-35), and are below 10 for the TI Pool and at Stillwater. On an average basis for the NOAEL, Stillwater also exceeds one. A LOAEL was not derived for this species (see Chapter 4).

5.3.1.2 Measurement Endpoint: Modeled Dietary Doses on a Tri+ PCB Basis to Insectivorous Birds (Tree Swallow) for the Period 1993 – 2018

Table 5-38 compares modeled dietary doses for the period 1993 – 2018 for the tree swallow to the field-based TRV derived in Chapter 4 under future conditions. This TRV was derived from the USFWS data from the Hudson River. Only the modeled result for 1993 exceeds one. For the remainder of the modeling period, the toxicity quotients for this insectivorous bird are below one.

5.3.1.3 Measurement Endpoint: Predicted Egg Concentrations on a Tri+ PCB Basis to Insectivorous Birds (Tree Swallow) for the Period 1993 – 2018

Table 5-39 compares predicted egg concentrations for the period 1993 – 2018 for the tree swallow to the field-based TRV derived in Chapter 4 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 95% UCL-based comparison exceeds one at the TI Pool until 1998, and falls below one thereafter.

5.3.1.4 Measurement Endpoint: Modeled Dietary Doses of PCBs and Predicted Egg Concentrations Expressed as TEQ to Insectivorous Birds (Tree Swallow) Based on 1993 Data

Table 5-40 compares the estimated TEQ-based dietary dose and predicted egg concentration to the piscivorous birds to the toxicity benchmarks presented in Table 4-26. The NOAEL-based comparison for the 95% UCL in eggs exceeds one at Stillwater. This is the only exceedance. The remainder of the predicted toxicity quotients fall below one.

5.3.1.5 Measurement Endpoint: Modeled Dietary Doses of PCBs Expressed as TEQ to Insectivorous Birds (Tree Swallow) for the Period 1993 - 2018

Table 5-41 compares the estimated TEQ-based dietary dose and predicted egg concentration to the piscivorous birds to the toxicity benchmarks presented in Table 4-26. None of the predicted dietary doses expressed as TEQ exceed the field-based TRV.

5.3.1.6 Measurement Endpoint: Predicted Egg Concentrations Expressed as TEQ to Insectivorous Birds (Tree Swallow) for the Period 1993 - 2018

Table 5-42 compares the estimated TEQ-based predicted egg concentrations for insectivorous birds to the toxicity benchmarks presented in Table 4-26 under future conditions. None of the toxicity quotients for the predicted egg concentrations exceed one.

5.4 Evaluation of Assessment Endpoint: Protection and Maintenance (i.e., Survival, Growth and Reproduction) of Local Waterfowl

5.4.1 Do Measured and 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 PCBs and Predicted Egg Concentrations as Total PCBs to Waterfowl (Mallard Ducks) Based on 1993 Data

Table 5-43 provides the results of comparisons between modeled dietary doses and predicted egg concentrations of total PCBs to toxicity reference values for the mallard duck based on 1993 data under current conditions. The NOAEL-based comparison for the 95% UCL on a dietary dose basis exceeds one for all river locations. On an average basis for the NOAEL, the Upper Hudson River locations exceed one, but Lower Hudson River locations fall below one. On a LOAEL basis, the 95% UCL exceeds one for the TI Pool and Stillwater, and only at Stillwater on an average basis.

For the predicted egg concentrations, the NOAEL-based comparisons exceed one for both the average and 95% UCL along the entire river. In the TI Pool and at Stillwater, the predicted toxicity quotients exceed 100. On a LOAEL basis, the 95% UCL comparisons exceed ten in the Upper Hudson River and exceed one at most Lower Hudson River locations. On an average basis, the predicted toxicity quotients exceed ten for the TI Pool and at Stillwater, and exceed one until RM122.4. Thereafter the predicted toxicity quotients fall below one.

5.4.1.2 Measurement Endpoint: Modeled Dietary Doses of Tri+ PCBs to Waterfowl (Mallard Ducks) for the Period 1993 - 2018

Table 5-44 provides the results of the comparison between predicted dietary doses based on predictions for the modeling period 1993 to 2018 to the toxicity reference values developed in Chapter 4 under future conditions. On a NOAEL basis, the predicted toxicity quotients exceed one for the entire modeling period on both an average and 95% UCL period in the TI Pool and at Stillwater. At RM154, the 95% UCL exceeds one until 2001, and the average until 2000.

On a LOAEL basis, only the 95% UCL for the TI Pool exceeds one until 1996. Thereafter for the TI Pool and at all other locations, predicted toxicity quotients do not exceed one.

5.4.1.3 Measurement Endpoint: Predicted Egg Concentrations of Tri+ PCBs to Waterfowl (Mallard Ducks) for the Period 1993 – 2018

Table 5-45 provides the results of the comparison between predicted egg concentrations and toxicity reference values based on model results for the period 1993 to 2018 under future conditions. These results show that predicted toxicity quotients exceed one for the duration of the modeling period on both a NOAEL and LOAEL basis, for both the average and 95% UCL. NOAEL-based comparisons for both the average and 95% UCL exceed ten and some exceed 100 in the TI Pool and at Stillwater.

5.4.1.4 Measurement Endpoint: Modeled Dietary Doses and Predicted Egg Concentrations of TEQ-Based PCBs to Waterfowl (Mallard Ducks) Using 1993 Data

Table 5-46 provides the results of the comparison between predicted dietary doses and egg concentrations on a TEQ basis to toxicity reference values developed in Chapter 4 using the 1993 data under current conditions. The results presented in this table show that the predicted toxicity quotients exceed one for all locations, for both the average and the 95% UCL, and for both the NOAEL and LOAEL-based comparisons.

5.4.1.5 Measurement Endpoint: Modeled Dietary Doses of TEQ-Based PCBs to Waterfowl (Mallard Ducks) for the Period 1993 – 2018

Table 5-47 provides the results of the comparison between predicted dietary doses and appropriate toxicity reference values for the period 1993 – 2018 under future conditions. These results show that predicted toxicity quotients exceed one for all locations and all comparisons. Predicted toxicity quotients exceed 100 on a NOAEL basis for all three modeling locations for the duration of the modeling period.

5.4.1.6 Measurement Endpoint: Predicted Egg Concentrations of TEQ-Based PCBs to Waterfowl (Mallard Ducks) for the Period 1993 – 2018

Table 5-48 provides the results of the comparison between predicted egg concentrations and appropriate toxicity reference values for the period 1993 – 2018 under future conditions. These results show that predicted toxicity quotients exceed 100 for all locations and all comparisons. This suggests the potential for adverse reproductive effects to waterfowl species.

5.5 Evaluation of Assessment Endpoint: Protection and Maintenance (i.e., Survival, Growth, and Reproduction) of Hudson River Piscivorous Bird Species

5.5.1 Do Measured and 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 PCBs and Predicted Egg Concentrations for Total PCBs for Piscivorous Birds (Belted Kingfisher, Great Blue Heron, Bald Eagle) Using 1993 Data

Tables 5-49 through 5-51 compare the estimated total PCB (i.e., Tri+) dietary dose of the female bald eagle, great blue heron, and kingfisher to the toxicity benchmarks presented in Table 4-26. The site-related doses are based on:

- Measured concentrations in forage fish, piscivorous fish, benthic invertebrates, whole water, and sediment using the 1993 USEPA Phase 2 datasets in species-specific exposure models under current conditions.

Female belted kingfisher modeled dietary doses on a total PCB (i.e., Tri+) basis are compared to toxicity reference values in Table 5-49. These values exceed one for all comparisons. In the Upper Hudson River, predicted toxicity quotients exceed 1000 in the TI Pool and at Stillwater on a NOAEL basis for the 95% UCL, and exceed 500 for the average in the TI Pool. Comparisons on the basis of predicted egg concentrations show higher toxicity quotients than dietary dose-based comparisons.

Table 5-50 provides comparisons of modeled dietary doses to great blue heron to toxicity reference values developed in Chapter 4. Again, all comparisons exceed one and in many cases exceed ten.

Table 5-51 presents the results for the bald eagle based on observed data. Again, all comparisons exceed one. For the TI Pool, the $ADD_{expected}$ compared to the LOAEL is 172, suggesting that site related concentrations of PCBs have the ability to adversely affect the reproductive capability of eagles. The field-based TRV derived for egg concentrations (Table 4-26) is exceeded by several orders of magnitude in the Upper Hudson River. Predicted toxicity quotients in the Lower Hudson River exceed 50 on an average basis and 100 on a 95% UCL basis.

Reproductive effects toxicity quotients for great blue heron, belted kingfisher, and bald eagle using average and upper confidence limits all exceed one on a dietary dose and predicted egg concentration basis using the 1993 data. This indicates that PCBs from the Hudson River in the diet and water present a significant risk of reproductive effects to these species on the basis of modeled total PCB dietary doses as compared to appropriate toxicity reference values. These results suggest it is reasonable to expect population-level effects, given the consistent exceedance of a reproductive-based endpoint.

5.5.1.2 Measurement Endpoint: Modeled Dietary Doses of Total PCBs for Piscivorous Birds (Belted Kingfisher, Great Blue Heron, Bald Eagle) Using 1993 Data

Tables 5-52 through 5-54 compare the estimated total PCB (i.e., Tri+) dietary dose of the female belted kingfisher, great blue heron, and bald eagle to the toxicity benchmarks presented in Table 4-26. 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 bioaccumulation model (USEPA, 1999c) and used in exposure models under future conditions.

Table 5-52 shows the comparison of modeled dietary doses to toxicity reference values for the period 1993 to 2018 for the kingfisher. All comparisons exceed one for the entire modeling period.

Table 5-53 presents the results for the great blue heron. This table shows that estimated toxicity quotients exceed one for all locations and years on both a NOAEL and LOAEL basis.

These results suggest the potential for adverse reproductive effects to the great blue heron.

Table 5-54 presents the results for the bald eagle. Again, all comparisons exceed one for the duration of the modeling period at all locations.

Reproductive effects toxicity quotients for great blue heron, belted kingfisher, and bald eagle using average and upper confidence limits all exceed one, in many cases by several orders of magnitude. This indicates that PCBs from the Hudson River in the diet and water present a significant 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 toxicity quotients are based on reproductive effects, and consistently exceed one (by a significant margin) over the course of the modeling period.

5.5.1.3 Measurement Endpoint: Predicted Egg Concentrations Expressed as Tri+ to Piscivorous Birds (Eagle, Great Blue Heron, Kingfisher) for the Period 1993 – 2018

Tables 5-55 through 5-57 compare the estimated total PCB (i.e., Tri+) predicted egg concentrations for the kingfisher, great blue heron, and bald eagle to the toxicity benchmarks presented in Table 4-26 under future conditions. Egg concentrations are estimated using biomagnification factors from the literature (Giesy et al., 1995) based on a fish concentration. Predicted fish concentrations were obtained using the FISHRAND model (USEPA, 1999c).

Table 5-55 presents the results for the kingfisher. 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. Early in the modeling period, predicted toxicity quotients exceed 1000 in the TI Pool and at Stillwater on a NOAEL basis.

Table 5-56 presents the results for the great blue heron. 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. Early in the modeling period, predicted toxicity quotients exceed 1000 in the TI Pool and at Stillwater on a NOAEL basis.

Table 5-57 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. As described in Chapter 4, only a field-derived NOAEL was obtained for the bald eagle. The predicted toxicity quotients using this NOAEL all exceed one, and several are greater than 100.

All of the predicted toxicity quotients greatly 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 toxicity quotient 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.

5.5.1.4 Measurement Endpoint: Modeled Dietary Doses and Predicted Egg Concentrations of PCBs on a TEQ Basis to Piscivorous Birds (Belted Kingfisher, Great Blue Heron, Bald Eagle) Using 1993 Data

Tables 5-58 through 5-60 compare the estimated predicted TEQ-based dietary dose and egg concentrations for the piscivorous bird species to the toxicity benchmarks presented in Table 4-26. The site-related dietary doses are based on:

- Measured concentrations in forage fish, piscivorous fish, benthic invertebrates, whole water, and sediment using the 1993 USEPA Phase 2 datasets in species-specific exposure models under current conditions.

Egg concentrations are predicted from biogmagnification factors presented in Chapter 3. Biomagnification factors are applied to observed forage or piscivorous fish concentrations to obtain predicted egg concentrations.

Table 5-58 presents the results for the belted kingfisher. This table shows that all comparisons exceed one by a considerable margin. The NOAEL based comparison for the 95% UCL for the predicted egg concentration in the TI Pool approaches 17,000. All toxicity quotients estimated on the basis of egg concentrations exceed 100 and exceed 1000 in the TI Pool and at Stillwater for both the LOAEL and NOAEL on both an average and 95% UCL basis.

Table 5-59 presents the results for the great blue heron. All comparisons exceed one, but predicted toxicity quotients for the great blue heron are comparably less than for the belted kingfisher. Nonetheless, the NOAEL-based dietary dose comparison exceeds 1000 in the TI Pool. Note that the toxicity reference values derived for the great blue heron are an order of magnitude higher than for the belted kingfisher.

Table 5-60 presents the results for the bald eagle. Predicted toxicity quotients exceed ten for all comparisons at all locations, and several are above 1000. For the predicted egg concentrations, the 95% UCL NOAEL based comparisons exceed 1000 for all locations, and in some cases exceed 10,000.

The results for modeled dietary doses and egg concentrations as compared to appropriate toxicity reference values based on 1993 data on a TEQ basis suggest that exposure to PCBs by piscivorous bird species is likely to result in adverse reproductive effects. All toxicity quotients exceed one, in many cases by many orders of magnitude, for all locations in the river. The consistency of these results suggests the potential for adverse reproductive effects on a population level.

5.5.1.5 Measurement Endpoint: Modeled Dietary Doses of PCBs Expressed as TEQs to Piscivorous Birds (Belted Kingfisher, Great Blue Heron, Bald Eagle) for the Period 1993 - 2018

Tables 5-61 through 5-63 present the results of the comparison between modeled dietary doses expressed as TEQ to the piscivorous bird species for the 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 HUDTOX and FISHRAND models (USEPA, 1999c) and used in exposure models under future conditions. Model results were multiplied by the weighted TEF derived in Chapter 3.

All locations and all comparisons exceed one for all three bird species on a dietary dose basis.

Reproductive effects toxicity quotients for 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 or even 1000. 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 it is reasonable to expect adverse population-level effects, given the consistent exceedance of a reproductive-based endpoint.

5.5.1.6 Measurement Endpoint: Modeled Dietary Doses of PCBs Expressed as TEQs to Piscivorous Birds (Belted Kingfisher, Great Blue Heron, Bald Eagle) for the Period 1993 - 2018

Tables 5-64 through 5-66 present the results of the comparison between egg concentrations expressed as TEQ to the piscivorous bird species for the period 1993 – 2018. Egg concentrations were estimated using:

- Modeled concentrations in forage fish and piscivorous fish from the FISHRAND bioaccumulation model (USEPA, 1999c) under future conditions. Model results were multiplied by the weighted TEF derived in Chapter 3 and then multiplied by a biomagnification factor of 19 (Giesy et al., 1995).

All locations and all comparisons exceed one for all three bird species on an egg concentration basis.

Toxicity quotients based on reproductive effects for 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 or even 1000, and several of the bald eagle toxicity quotients exceed 10000. 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 toxicity reference values. These results suggest it is reasonable to expect adverse population-level effects, given the consistent exceedance of a reproductive-based endpoint.

5.5.2 Do Measured and Modeled Water Concentrations Exceed Criteria and/or Guidelines for the Protection of Wildlife?

5.5.2.1 Measurement Endpoint: Comparison of Measured and Modeled Water Concentrations to Criteria and/or Guidelines

Measured average and 95% UCL total PCB concentrations exceed the NYSDEC criteria developed for the protection of wildlife under current conditions. All locations in the river exceed 100 for all comparisons, and some exceed 1000. Modeled water concentrations from HUDTOX also exceed the NYSDEC criteria for the duration of the modeling period under future conditions.

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

5.5.3.1 Measurement Endpoint: Observational 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.

In this subchapter, the results of a survey undertaken to examine wildlife activity throughout the assessment area are presented. The survey information provided in the following sections is based on the insights of a variety of observers living and working along the Upper Hudson River between the TI Pool to Federal Dam. Before initiating the survey, it was clear that the level of assessment along this stretch of the river was limited in comparison to the region below Federal Dam.

The survey was not formally structured because of the diversity of different experts included in the interview process. The primary focus of the interviews was to determine which species are utilizing the Upper Hudson River, and to understand and identify patterns of changes along this stretch of the Hudson River. Determining a direct causative linkage between contamination and population changes is not realistic, however many of the observers have been working on the Hudson for forty years and can provide anecdotal information about population patterns over time. The wildlife survey confirms that the selection of representative species for the ecological risk assessment is appropriate and provides first-hand, qualitative information about the status of wildlife in the Upper Hudson River.

Quantitative data were collected whenever possible, however the majority of the data were qualitative in nature. Few studies focusing specifically on wildlife in the Upper Hudson River through quantitative population assessments are available.

An information matrix (Table 5-67) was developed to guide the survey. The matrix includes contact information, dates, sources, a compilation of available data and relevant information. In many cases the first person contacted could not provide information on the Upper

Hudson River, but recommended additional contacts. The additional contacts are provided where appropriate.

Contacts included scientists in non-profit organizations (Hudsonia, Hudson River Foundation, Institute for Ecological Studies, The Nature Conservancy, Manomet Center for Conservation Sciences, New York State Trappers Association, Saratoga National Historic Park, Ndakinna Wilderness Project), state and local government agencies (Department of Environmental Conservation, New York River Otter Project, Endangered Species Units, New York State Museum, NY Natural Heritage Program, Amphibian and Reptile Atlas Project, Federation of New York State Bird Clubs, Breeding Bird Survey Online, Hudson-Mohawk Bird Club), and academic departments (Union College, SUNY Stonybrook, Cornell University).

In general, information on wildlife usage of the Upper Hudson River is sparse and unconnected. The survey was very broad in terms of the contact list, however studies in this area are limited. The most valuable and detailed information sources were a fisher and hunter who cover the Upper Hudson River routinely and have spent the last forty years observing Hudson wildlife. They provided invaluable insights into the patterns and populations occupying this stretch of the Hudson.

Avian wildlife are well studied along the Upper Hudson River. Both species-specific studies and general, annual bird surveys provide extensive databases on the species that live and migrate through the region.

The New York State Department of Environmental Conservation – Endangered Species Unit reports three threatened species that live in the Northern Hudson Valley. The least bittern (*Ixobrychus exilis*) and upland sand piper (*Bartramia longicauda*), once common along the Northern Hudson, have declined due to loss of habitat, pollution and insecticides. The king rail (*Rallus elegans*) was reported to be nesting in the northern Hudson, but nests have not been confirmed.

Tree swallows (*Tachycineta bicolor*) are very common along the Hudson River. John McCarty, of Cornell University and Anne Secord, of the US Fish and Wildlife Service, studied the tree swallow's reproductive ecology and behavior in relation to PCB contamination. The swallows are common along the Upper Hudson River during the spring when they are feeding in preparation for breeding (McCarty and Secord, 1999). Though details about PCB toxicity are discussed in other portions of the risk assessment, McCarty and Secord provide insight into the importance of identifying additional population effects in order to understand wildlife distribution along this stretch of the Hudson. A large proportion of the information presented in this section is based on years of observations along the Hudson. However, issues such as low reproductive success and the occurrence of unusual parental behavior may have a hidden impact on the populations. Over time additional trends may be better understood. McCarty and Secord do confirm that tree swallows are able to use this stretch of the Upper Hudson River.

Everett Nack identified a number of bird species in the Upper Hudson River. He recently observed a family of mallards (6-8 chicks) in the TI Pool and has observed 15-20 great blue heron nests around Schuylerville. Canada geese are common along the entire Hudson. He observes small numbers of osprey following the herring runs.

Both the New York State Endangered Species Unit and The Atlas of Breeding Birds in New York (1988) provide general information regarding the bird species using the Upper Hudson River. Birds that are not rare in this area include the least bittern (*Ixobrychus exilis*), the king rail (*Rallus elegans*) and the upland sandpiper (*Bartramia longicauda*). The breeding bird survey concludes that the tree swallow (*Tachycineta bicolor*) is a common breeder in the area, the belted kingfisher (*Ceryle alcyon*) is a common summer resident along the Hudson River, the great blue heron (*Ardea herodias*) is observed along the Upper Hudson River, while the mallard (*Anas platyrhynchos*) is a frequent breeder in the wetland along the Upper Hudson River. Both eagles and osprey are not common in the Upper Hudson River Valley based on the breeding bird survey results.

Mark Brown of the New York State Department of Environmental Conservation reports that the area is rich in waterfowl. The bald eagle is a winter resident, but migrates in the summer. Mallards and Canada geese are common resident waterfowl in the Upper Hudson. In addition, he has seen tree swallows, kingfishers and great blue herons. Osprey do use the Northern Hudson River for feeding, but tend to breed along the shores of nearby lakes. In general, the larger birds feed along the Hudson, but breed and nest elsewhere. He has also observed: common merganser (diving duck), red-tailed hawk, sparrow hawk, rough grouse, wild turkey, killdeer, woodcock, morning dove, barn owl, bard owl, sawhat owl, swallows, ravens, crows, wrens, Eastern blue bird and starlings.

Jim Brushek, a professional tracker, has seen bald eagles, great blue heron and large numbers of kingfishers, mallards and Canada geese. However, he has not seen osprey.

The "Collins Lake Waterbird Study" in Scotia, New York provides a well-developed daily log of bird visits to a lake that is 15 miles west of the Hudson River. The study is valuable because it records trends over the past ten years of observation at Collins Lake. The study found all of the avian species of interest using the lake except the osprey, with population numbers generally increasing from 1988 to 1998.

5.6 Evaluation of Assessment Endpoint: Protection (i.e., Survival and Reproduction) of Local Wildlife

5.6.1 Do Measured and Modeled Total and TEQ-Based PCB Dietary Doses to 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) using 1993 Data

Modeled total PCB dietary dose comparisons to toxicity reference values are presented for the little brown bat in Table 5-68 using 1993 data under current conditions. These results show that LOAEL-based toxicity quotients exceed ten in the Upper Hudson River and exceed 100 at Stillwater, and exceed one at most locations in the Lower Hudson River for the 1993 USEPA dataset. All NOAEL based comparisons exceed one in the Lower Hudson River and exceed 50 in the Upper Hudson River.

These results suggest the potential for adverse reproductive effects to insectivorous mammalian species at all locations in the river based on using 1993 data in the exposure models. Given the consistency of the results and the magnitude of the exceedances, these results suggest the potential for population-level adverse reproductive effects.

5.6.1.2 Measurement Endpoint: Modeled Dietary Doses of Tri+ to Insectivorous Mammalian Receptors (Little Brown Bat) for the Period 1993 - 2018

Modeled total PCB dietary dose comparisons to toxicity reference values are presented for the little brown bat in Table 5-69 for the period 1993 – 2018 under future conditions. These results show that all comparisons exceed one for all locations except the LOAEL based average and 95% UCL at RM154 (just above Federal Dam) starting in 2013 and 2016, respectively.

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.

5.6.1.3 Measurement Endpoint: Modeled Dietary Doses on a TEQ Basis to Insectivorous Mammalian Receptors (Little Brown Bat) using 1993 Data

Modeled total PCB dietary dose comparisons to toxicity reference values are presented for the little brown bat in Table 5-70 based on using 1993 data in the exposure models under current conditions. Modeled dietary doses are obtained by using 1993 observed benthic invertebrate and water concentrations in the exposure models. Observed conentrations are adjusted by the weighted TEF presented in Chapter 3.

These results show that all comparisons exceed one for all locations. Predicted toxicity quotients are higher than those predicted on a Tri+ PCB basis. Predicted toxicity quotients in the TI Pool and at Stillwater exceed 100 across all comparisons.

These results suggest the potential for adverse reproductive effects to insectivorous mammalian species at all locations in the river on a TEQ basis using 1993 data in the exposure models. Given the consistency of the results and the magnitude of the exceedances, these results suggest the potential for population-level adverse reproductive effects.

5.6.1.4 Measurement Endpoint: Modeled Dietary Doses of Tri+ to Insectivorous Mammalian Receptors (Little Brown Bat) for the Period 1993 - 2018

Modeled total PCB dietary dose comparisons to toxicity reference values are presented for the little brown bat in Table 5-71 for the period 1993 – 2018 under future conditions. These results show that all comparisons exceed ten for all locations except the LOAEL based average and 95% UCL at RM154 (just above Federal Dam) starting in 2001 and 2005, respectively. All comparisons exceed one for all locations during the entire modeling period.

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.

5.6.1.5 Measurement Endpoint: Modeled Dietary Doses of Tri+ to Omnivorous Mammalian Receptors (Raccoon) using 1993 Data

Modeled total PCB dietary dose comparisons to toxicity reference values are presented for the raccoon in Table 5-72 using 1993 data under current conditions. In the Upper Hudson River, all comparisons exceed one. The NOAEL based comparison for the 95% UCL exceeds one at all locations in both the Upper and Lower Hudson River. In the Lower Hudson River, the LOAEL based comparison for the average does not exceed one at any location, but does exceed one for RM137.2, RM113.8, RM100, RM58.7 and RM47.3 for the 95% UCL.

These results suggest the potential for adverse reproductive effects to omnivorous mammalian species in the Upper Hudson River based on using 1993 data in the exposure models for dietary dose. Given the consistency of the results and the magnitude of the exceedances, these results suggest the potential for population-level adverse reproductive effects in the Upper Hudson River.

5.6.1.6 Measurement Endpoint: Modeled Dietary Doses of Tri+ to Omnivorous Mammalian Receptors (Raccoon) for the Period 1993 - 2018

Modeled total PCB dietary dose comparisons to toxicity reference values are presented for the raccoon in Table 5-73 for the period 1993 – 2018 under future conditions. Dietary doses are estimated by using predicted water concentrations from the HUDTOX model and predicted forage fish and benthic invertebrate concentrations from the FISHRAND model.

Predicted toxicity quotients for the TI Pool exceed one on both a NOAEL and LOAEL basis for both the average and 95% UCL. Predicted toxicity quotients exceed one at Stillwater on a LOAEL basis until 2005 and 2007 for the average and 95% UCL, and until 1995 at RM154 (just above Federal Dam). On a NOAEL basis, both the average and 95% UCL comparisons exceed one for the duration of the modeling period at Stillwater, and until 2013 at RM154.

These results suggest the potential for adverse reproductive effects to omnivorous mammalian species in the Upper Hudson River based on using FISHRAND and HUDTOX 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 in the Upper Hudson River.

5.6.1.7 Measurement Endpoint: Modeled Dietary Doses on a TEQ Basis to Omnivorous Mammalian Receptors (Raccoon) using 1993 Data

Modeled total PCB dietary dose comparisons to toxicity reference values are presented for the raccoon in Table 5-74 using 1993 data under current conditions. Dietary doses are estimated by using 1993 sediment, water, benthic invertebrate and forage fish concentrations in the exposure models. All comparisons consistently exceed one, and in some cases exceed 100.

These results suggest the potential for adverse reproductive effects to omnivorous mammalian species in the Hudson River based on using 1993 data in the exposure models for dietary dose. Given the consistency of the results and the magnitude of the exceedances, these results suggest the potential for population-level adverse reproductive effects in the Upper Hudson River.

5.6.1.8 Measurement Endpoint: Modeled Dietary Doses on a TEQ Basis to Omnivorous Mammalian Receptors (Raccoon) for the Period 1993 - 2018

Modeled total PCB dietary dose comparisons to toxicity reference values are presented for the raccoon in Table 5-75 for the period 1993 – 2018 under future conditions. Dietary doses are estimated by using predicted water concentrations from the HUDTOX model and predicted forage fish and benthic invertebrate concentrations from the FISHRAND model.

All comparisons exceed one at all locations for the duration of the modeling period. Predicted toxicity quotients for the TI Pool exceed ten on a LOAEL basis and exceed 100 on a NOAEL basis for both the average and 95% UCL. Predicted toxicity quotients exceed ten at Stillwater on a LOAEL basis until 2017 for both the average and 95% UCL, and until 1999 at RM154 (just above Federal Dam). On a NOAEL basis, both the average and 95% UCL comparisons exceed 100 until 2016 and 2017 for the average and 95% UCL, respectively. At RM154, the NOAEL based comaprisons exceed ten for the duration of the modeling period.

These results suggest the potential for adverse reproductive effects to omnivorous mammalian species in the Upper Hudson River based on using FISHRAND and HUDTOX 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 in the Upper River.

5.6.1.9 Measurement Endpoint: Measured Total PCB Concentrations in the Liver of Piscivorous Mammalian Receptors (Mink, Otter)

Table 5-76 presents the results of a comparison between PCB concentrations in mink and otter liver tissue (NYS Toxic Substances Monitoring Program, 1987) to the toxicity reference values presented in Chapter 4. The data were collected in the mid-1980's, and provide some field evidence that mink and otter tissue concentrations have been observed at levels known to cause reproductive effects. At the upper range LOAEL (the bottom half of Table 5-76), total reproductive failure was observed in mink experiencing liver concentrations of 3.2 mg/kg of PCBs.

5.6.1.10 Measurement Endpoint: Modeled Dietary Doses of Tri+ to Piscivorous Mammalian Receptors (Mink, Otter) using 1993 Data

Tables 5-77 and 5-78 present the results of the comparison between modeled dietary doses to mink and otter on the basis of 1993 data for total (Tri+) PCBs under current conditions. Modeled dietary doses are estimated from observed water, sediment, forage fish and piscivorous fish concentrations using the 1993 dataset.

Table 5-77 shows the results for the mink. On a dietary dose basis for total (Tri+) PCBs, predicted toxicity quotients exceed ten on a NOAEL basis for both the average and 95% UCL at all locations in the river. For the LOAEL based comparison, predicted toxicity quotients exceed one in the Upper Hudson River and exceed one for the 95% UCL in the Lower Hudson River at all locations except at 88.9 and 25.8. For the average, the predicted toxicity quotients on a LOAEL basis exceed one in the Upper Hudson River and in the Lower Hudson River down to 137.2. Thereafter, the LOAEL based comparison for the average falls below one.

Table 5-78 shows the results for the otter. On a dietary dose basis for total (Tri+) PCBs, predicted toxicity quotients exceed one at all locations across all comparisons. On a NOAEL basis, the predicted toxicity quotients for the otter for both the average and 95% UCL exceed 100 at all locations. The otter consumes a larger size range of fish and is likely to obtain fish from deeper in the river. Thus, the exposure of the 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 1993 data in the exposure models for dietary dose. Given the consistency of the results and the magnitude of the exceedances, these results suggest the potential for population-level adverse reproductive effects for mink and otter consuming fish from the Hudson River.

5.6.1.11 Measurement Endpoint: Modeled Dietary Doses of Tri+ to Piscivorous Mammalian Receptors (Mink, Otter) for the Period 1993 - 2018

Tables 5-79 and 5-80 present the results of the comparison between modeled dietary doses to mink and otter under future conditions for the period 1993 - 2018 for total (Tri+) PCBs under future conditions. Modeled dietary doses are estimated by using HUDTOX model results for water and sediment, and FISHRAND results for forage fish and piscivorous fish concentrations.

Table 5-79 shows the results for the mink. On a dietary dose basis for total (Tri+) PCBs, predicted toxicity quotients exceed one for the duration of the modeling period for both the LOAEL and NOAEL based comparisons in the TI Pool. NOAEL based results exceed one for all locations for the duration of the modeling period. At Stillwater, the LOAEL based comparison exceeds one for the average and 95% UCL until 2004 and 2006, respectively. At RM154, the LOAEL based comparison does not exceed one on an average basis and is exactly one for the 95% UCL until 1999.

Table 5-80 shows the results for the otter. On a dietary dose basis for total (Tri+) PCBs, predicted toxicity quotients exceed one at all locations across all comparisons. On a NOAEL basis, the predicted toxicity quotients for the otter for both the average and 95% UCL exceed 100 in the TI Pool and at Stillwater for the duration of the modeling period. The otter consumes a larger size range of fish and is likely to obtain fish from deeper in the river. Thus, the exposure of the 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 HUDTOX 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 otter consuming fish from the Hudson River.

Reproductive effects toxicity quotients for the mink and otter using average and upper confidence limits all exceed one. This indicates that PCBs from the Hudson River in the diet and water are likely to present a significant risk of reproductive effects to the mink and otter on the basis of modeled total PCB dietary doses as compared to appropriate toxicity reference values. Limited field validation is provided by the comparison between observed liver concentrations and a toxicity reference value for complete reproductive failure. Results for the bat and raccoon suggest that levels of PCBs in the Upper Hudson River pose a greater risk than concentrations of PCBs in the Lower Hudson River for these receptors.

5.6.1.12 Measurement Endpoint: Modeled Dietary Doses on a TEQ Basis to Piscivorous Mammalian Receptors (Mink, Otter) using 1993 Data

Tables 5-81 and 5-82 present the results of the comparison between modeled dietary doses to mink and otter under current conditions on the basis of 1993 data for PCBs expressed as TEQ. Modeled dietary doses are estimated from observed water, sediment, forage fish and piscivorous fish concentrations using the 1993 dataset.

Table 5-81 shows the results for the mink. On a dietary dose basis for total (Tri+) PCBs, predicted toxicity quotients exceed one for all locations across all comparisons. LOAEL based comparisons exceed ten in the TI Pool and at Stillwater for both the average and 95% UCL, and exceed 100 for both these locations on a NOAEL basis. In the Lower Hudson River, the LOAEL based comparisons exceed one but fall below ten for the average and 95% UCL, and exceed ten and in some cases 100 for the NOAEL based comparison.

Table 5-82 shows the results for the otter. On a dietary dose basis for total (Tri+) PCBs, predicted toxicity quotients exceed ten at all locations across all comparisons. On a NOAEL basis, the predicted toxicity quotients for the otter for both the average and 95% UCL exceed 100 at all locations. The otter consumes a larger size range of fish and is likely to obtain fish from deeper in the river. Thus, the exposure of the 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 1993 data in the exposure models for dietary dose. Given the consistency of the results and the magnitude of the exceedances, these results suggest the potential for population-level adverse reproductive effects for mink and otter consuming fish from the Hudson River.

5.6.1.13 Measurement Endpoint: Modeled Dietary Doses on a TEQ Basis to Piscivorous Mammalian Receptors (Mink, Otter) for the Period 1993 - 2018

Tables 5-82 and 5-83 present the results of the comparison between modeled dietary doses to mink and otter under future conditions for the period 1993 - 2018 for total (Tri+) PCBs. Modeled dietary doses are estimated by using HUDTOX model results for water and sediment, and FISHRAND results for forage fish and piscivorous fish concentrations.

Table 5-82 shows the results for the mink. On a dietary dose basis for total (Tri+) PCBs, predicted toxicity quotients exceed one for the duration of the modeling period for both the LOAEL and NOAEL based comparisons in the TI Pool and at Stillwater. NOAEL based results exceed one for all locations for the duration of the modeling period. At RM154, the LOAEL based comparison exceeds one on an average basis until 2005 and exceeds one for the 95% UCL until 2005.

Table 5-83 shows the results for the otter. On a dietary dose basis for total (Tri+) PCBs, predicted toxicity quotients exceed ten at all locations across all comparisons except on a LOAEL basis at RM154. At this location, predicted toxicity quotients exceed ten for all comparisons except the LOAEL based comparison from 2007 onward for the average and 2010 for the 95% UCL. The otter, which consumes larger fish and obtains 100% of its diet from fish, demonstrates higher toxicity quotients than the mink, as shown by comparing Tables 5-80 and 5-81.

These results suggest the potential for adverse reproductive effects to piscivorous mammalian species in the Hudson River based on using HUDTOX 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 otter consuming fish from the Hudson River.

TEQ based toxicity quotients are generally higher than the total PCB dietary dose results. Reproductive effects toxicity quotients for the mink and otter using average and upper confidence limits all exceed one. This indicates that PCBs from the Hudson River in the diet and water are likely to present a significant risk of reproductive effects to the mink and otter on the basis of modeled total PCB dietary doses as compared to appropriate toxicity reference values. Limited field validation is provided by the comparison between observed liver concentrations and a toxicity reference value for complete reproductive failure.

5.6.2 Do Measured and Modeled Water Concentrations Exceed Criteria and/or Guidelines for the Protection of Wildlife?

5.6.2.1 Measurement Endpoint: Comparison of Measured and Modeled Water Concentrations to Criteria and/or Guidelines for the Protection of Wildlife

Measured average and 95% UCL total PCB concentrations exceed the NYSDEC criteria developed for the protection of wildlife under current conditions. All locations in the river exceed 100 for all comparisons, and some exceed 1000. Modeled water concentrations from HUDTOX also exceed the NYSDEC criteria for the duration of the modeling period under future conditions.

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

5.6.3.1 Measurement Endpoint: Observational 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.

In this subchapter, the results of a survey undertaken to examine wildlife activity throughout the assessment area are presented. The survey information provided in the following sections is based on the insights of a variety of observers living and working along the Upper Hudson River between the TI Pool to Federal Dam. Before initiating the survey, it was clear that the level of assessment along this stretch of the river was limited in comparison to the region below Federal Dam.

The survey was not formally structured because of the diversity of different experts included in the interview process. The primary focus of the interviews was to determine which species are utilizing the Upper Hudson, River and to understand and identify patterns of changes along this stretch of the Hudson River. Determining a direct causative linkage between contamination and population changes is not realistic, however many of the observers have been working on the Hudson for forty years and can provide anecdotal information about population patterns over time. The wildlife survey confirms that the selection of representative species for the ecological risk assessment is appropriate and provides first-hand, qualitative information about the status of wildlife in the Upper Hudson River.

Quantitative data were collected whenever possible, however the majority of the data were qualitative in nature. Few studies focusing specifically on wildlife in the Upper Hudson River through quantitative population assessments are available.

An information matrix (Table 5-84) was developed to guide the survey. The matrix includes contact information, dates, sources, a compilation of available data and relevant information. In many cases the first person contacted could not provide information on the Upper Hudson River, but recommended additional contacts. The additional contacts are provided where appropriate.

Contacts included scientists in non-profit organizations (Hudsonia, Hudson River Foundation, Institute for Ecological Studies, The Nature Conservancy, Manomet Center for Conservation Sciences, New York State Trappers Association, Saratoga National Historic Park, Ndakinna Wilderness Project), state and local government agencies (Department of Environmental Conservation, New York River Otter Project, Endangered Species Units, New York State Museum, NY Natural Heritage Program, Amphibian and Reptile Atlas Project, Federation of New York State Bird Clubs, Breeding Bird Survey Online, Hudson-Mohawk Bird Club), and academic departments (Union College, SUNY Stonybrook, Cornell University). The complete survey matrix is provided upon request.

In general, information on wildlife usage of the Upper Hudson River is sparse and unconnected. The survey was very broad in terms of the contact list, however studies in this area are limited. The most valuable and detailed information sources were a fisher and hunter who cover the Upper Hudson River routinely and have spent the last forty years observing Hudson wildlife. They provided invaluable insights into the patterns and populations occupying this stretch of the Hudson.

Mammals:

In general, the Upper Hudson River is not studied as thoroughly as the Lower Hudson River. In most cases the interviewed person worked and studied the Lower Hudson River and could only provide rough estimates of wildlife use in the Upper Hudson River.

Mark Brown, of the New York Department of Environmental Conservation, is responsible for a number of counting and banding projects with birds and is undertaking a herpotological study along the Hudson. He has a good working knowledge of the Hudson. The river otter (*Lutra* spp.), mink (*Mustela vison*) and muskrat (*Ondatra zibethica*) are all present along this stretch of the river. He noted a severe decline in population numbers, but they have rebounded in the past ten years coincident with cleanup work. Mark Brown has also identified raccoon (*Procyon lotor*), short and long-tailed weasels (*Mustela* spp.), both big and little brown bats (order: *Chiroptera*), skunk (*Mephitis mephitis*) and oppossum (*Didelphis marsupialis*). Along the northern portion of the Hudson, red fox (*Vulpes vulpes*), grey fox (*Vulpes cinereoargenteus*) and coyote (*Canis latrans*) are very common and he noted that the growing white tail deer (*Odocoileus virginianus*) population is a good indicator that the bear (*Ursus americanus*) population is minimal along this stretch of the river.

Jim Brushek, a professional tracker who works with the Ndakinna Wilderness Project, confirmed the information provided by Mark Brown of NYSDEC. He described the fauna as rich along the Hudson River. He sees many otter and beaver (*Castor* spp.) and the mink population is large and increasing. Based on a 'road-kill count' the raccoon population is staggering. Though not at levels consistent with the otter and mink, there are a few muskrat. Moose (*Alces alces*), if not already using the Upper Hudson River, are expected to use the

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Hudson soon based on reports of moose sightings in Saratoga. He has seen fisher cats (*Martes pennanti*) cruising the shoreline on occasion. Consistent with Mark Brown's observations, he sees many fox (red and grey) and deer. The coyote population is also very large and they generally feed on the smaller aquatic mammals. He has seen a few black bear though their populations appear to be patchy.

Professional fishers are an excellent resource because they work along the river day and night, and often grew up on the river. Everett Nack has been working along the river as a professional fisherperson for over 60 years. He rarely sees otter north of the dam, though he has seen them around Albany. Mr. Nack observes a few mink and many beaver in and around the TI Pool.

In general, mammals are using the Upper Hudson River. The only species that appear to be rare are muskrat, though there is some question about mink populations as well.

5.7 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 these wildlife species, and the shortnose sturgeon a representative surrogate for fish.

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

5.7.1.1 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, we compare observed and modeled largemouth bass total and TEQ based PCB concentrations to toxicity reference values.

The derived values (Table 4-26) 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 largemouth bass and brown bullhead, because the toxicity reference values are the same.

Under the assumption that shortnose sturgeon experience body burdens and effect levels similar to either largemouth bass or brown bullhead, PCB body burdens resulting from exposure to Hudson River sources are likely to result in adverse reproductive effects to fish.

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

5.7.2.1 Measurement Endpoint: Inferences Regarding Bald Eagle and Other Raptor Populations

The modeled results for the bald eagle were presented in Chapter 5.1.3. All comparisons across all locations and PCB types exceeded one, and in some instances by several 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 *(Buteo lineatus)* are likely to exhibit similar exposures.

5.7.3 Do Measured and Modeled Water Concentrations Exceed Criteria and/or Guidelines for the Protection of Wildlife?

5.7.3.1 Measurement Endpoint: Comparisons of Measured and Modeled Water Concentrations to Criteria and/or Guidelines for the Protection of Wildlife

Measured average and 95% UCL total PCB concentrations exceed the NYSDEC criteria developed for the protection of wildlife. All locations in the river exceed 100 for all comparisons, and some exceed 1000. Modeled water concentrations from HUDTOX also exceed the NYSDEC criteria for the duration of the modeling period.

5.7.4 Do Measured and Modeled Sediment Concentrations Exceed Guidelines for the Protection of Aquatic Health?

5.7.4.1 Measurement Endpoint: Comparisons of Measured and Modeled Sediment Concentrations to Guidelines

Measured sediment concentrations exceed the NOAA TEC and MEC consensus values, both on an average and 95% UCL basis, at all locations except RM58.7 for the average as compared to the MEC under current conditions. Measured concentrations in the Upper Hudson River exceed the EEC, and several locations in the Lower Hudson River exceed the EEC on a 95% UCL basis. For future conditions, modeled average and 95% UCL sediment concentrations predicted by HUDTOX exceed the TEC, MEC, and EEC for the TI Pool and Stillwater for the duration of the modeling period. Modeled average and 95% UCL sediment concentrations at RM154 exceed the TEC and MEC for the duration of the modeling period, and exceed the EEC until 2001.

Measured sediment concentrations based on the 1993 USEPA dataset exceed the NYSDEC chronic criteria for the protection of benthic aquatic life on both an average and 95%

UCL basis for all locations in the river except RM100 (average only). Average and 95% UCL measured sediment concentrations at all locations exceed the NYSDEC wildlife criterion. Expressed as a ratio, these values are above ten for all locations and above 1000 at Stillwater for the 95% UCL comparison. For future conditions, modeled average and 95% UCL sediment concentrations predicted by HUDTOX exceed the benthic chronic criterion for all locations for the duration of the modeling period.

Measured sediment concentrations exceed the SELs from Ontario, and all Washington State criteria. The same is true for future conditions.

Many of the ratios of measured and modeled sediment concentrations to appropriate guidelines exceed 10 or even 100. Thus, even in the unlikely event that predicted sediment concentrations were to decrease by an order of magnitude or more, comparisons to sediment guidelines would show exceedances.

5.8 Evaluation of Assessment Endpoint: Protection of Significant Habitats

The significant habitats found along the Hudson River (Table 2-11) 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.

Based on the comparisons of observed and modeled body burdens to toxicity reference values presented in the previous subchapters (Chapters 5.2.2 to 5.2.5), current PCB concentrations found in the Lower Hudson River (i.e., RM137.2, RM122.4, RM113.8, RM100, RM88.9, RM58.7, RM47.3, and RM25.8) exceed toxicity reference values for avian and mammalian receptors (fish receptors will be evaluated in a subsequent report). 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.

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. The exceedance of sediment and surface water criteria is another measurement endpoint which indicates that populations using significant habitats along the Hudson River may affected by PCBs.

5.8.1 Do Measured and Modeled Water Column Concentrations Exceed Criteria and/or Guidelines for the Protection of Aquatic Wildlife?

5.8.1.1 Measurement Endpoint: Comparison of Measured and Modeled Water Concentrations to Criteria and/or Guidelines for the Protection of Wildlife

Measured average and 95% UCL total PCB concentrations exceed the NYSDEC criteria developed for the protection of wildlife under current conditions. All locations in the river exceed 100 for all comparisons, and some exceed 1000. Modeled water concentrations from HUDTOX also exceed the NYSDEC criteria for the duration of the modeling period under future conditions.

5.8.1.2 Measurement Endpoint: Comparison of Measured and Modeled Sediment Concentrations to Guidelines for the Protection of Aquatic Health

Under current conditions, measured sediment concentrations exceed the NOAA TEC and MEC consensus values, both on an average and 95% UCL basis, at all locations except RM58.7 for the average as compared to the MEC under current conditions. Measured concentrations in the Upper Hudson River exceed the EEC, and several locations in the Lower Hudson River exceed the EEC on a 95% UCL basis. For future conditions, modeled average and 95% UCL sediment concentrations predicted by HUDTOX exceed the TEC, MEC, and EEC for the TI Pool and Stillwater for the duration of the modeling period. Modeled average and 95% UCL sediment concentrations at RM154 exceed the TEC and MEC for the duration of the modeling period, and exceed the EEC until 2001.

Measured sediment concentrations based on the 1993 USEPA dataset exceed the NYSDEC chronic criteria for the protection of benthic aquatic life on both an average and 95% UCL basis for all locations in the river except RM100 (average only). Average and 95% UCL measured sediment concentrations at all locations exceed the NYSDEC wildlife criterion. Expressed as a ratio, these values are above ten for all locations and above 1000 at Stillwater for the 95% UCL comparison. For future conditions, modeled average and 95% UCL sediment concentrations predicted by HUDTOX exceed the benthic chronic criterion for all locations for the duration of the modeling period.

Measured sediment concentrations exceed the SELs from Ontario, and all Washington State criteria. The same is true for future conditions.

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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 include:

- Sampling error and representativeness;
- Analysis and quantitation uncertainties;
- Conceptual model uncertainties;
- Toxicological study uncertainties;
- Natural variation and parameter error; and
- Model error.

Each of these potential sources of uncertainty is discussed next.

6.1 Sampling Error and Representativeness

Errors may occur during sampling activities. Examples of errors include use of contaminated sampling equipment or effectiveness of the sampling device in the collection of a discrete and representative sample. To minimize any uncertainties associated with the above sources of error and uncertainty, guidance set forth and described in the Work Plan and Sampling Plan (USEPA, 1992b) and Sampling and Analysis/Quality Assurance Project Plan (QAPP) (USEPA, 1993a) were followed. A field quality assurance audit was conducted on August 11, 1993 to ensure that the Field Sampling Plan (FSP) and QAPP were adhered to. The uncertainty associated with sampling error is considered to be low. The procedures set forth in the above plans were developed to minimize uncertainties associated with sampling error.

Representativeness accounts for the effective assessment of the nature and extent of contamination based upon sampling of a defined population. Uncertainty may be introduced into an assessment if the samples are not representative of "true" concentrations over appropriate spatial and temporal scales. PCB concentrations in Hudson River water and sediments are highly variable in space and time, resulting in sample uncertainty for representation of actual conditions in each reach of the river. The Hudson River is contaminated with PCBs originating from the General Electric Hudson Falls and Ft. Edward plants for almost 200 miles (322 km). Consequently, all potentially contaminated locations along the river could not be sampled. To focus on locations that were considered most appropriate for the ecological risk assessment a literature search, field reconnaissance, and PCB field screening were performed prior to site selection. USEPA coordinated with NYSDEC and NOAA so that all ecological samples (i.e., sediment, benthic invertebrate, fish, and historical fish samples) were collected from the same sampling locations.

Pilot samples taken during the field reconnaissance indicated that most biological activity was found in the upper 5 cm (2 inches) of sediment. After discussion with other agencies, a surficial sampling depth of 5 cm was selected. The 5 cm sampling depth is less than the depth at which bioturbation can occur, but since the ERA focuses on the exposure of the ecological community to PCBs, the 5 cm sampling depth was used. USEPA coordinated with NYSDEC and NOAA so that all ecological samples (i.e., sediment, benthic invertebrate, fish, and historical fish samples) were collected from approximately the same sampling locations.

Fine-grained, depositional areas where PCBs often accumulate were selected as sampling stations. However, microvariation in grain size, TOC, etc. was present at all sampling stations. To reduce the uncertainty associated with this ubiquitous microvariation, two samples were composited for each of the five replicates taken at each station.

PCB congener patterns and concentrations in Hudson River fish vary both spatially and seasonally. Both striped bass and resident fish collected in the spring (May) showed consistently higher PCB concentrations than fish collected in the fall (August) (NOAA, 1997a). Although the differences were not large, potential risks to fish and upper-trophic level receptors may be underestimated because the Phase 2 ecological samples were collected in the fall.

6.2 Analysis and Quantitation Uncertainties

The analysis and quantitation of PCBs and other parameters was minimized by developing and adhering to strict quality assurance (QA)/quality control (QC) protocols. Procedures were developed specifically for this project by Inchcape Laboratories, Inc. to detect low concentrations of many PCB-congeners. USEPA reviewed all the procedures developed and reviewed new protocols. Although the accuracy of the laboratory analyses is considered to be quite high, there is always some level of uncertainty associated with all laboratory analyses due to matrix interference, handling, and analytical equipment limitations.

There is some uncertainty associated with the quantification of the congeners used in the TEQ analysis (see Table 4-2). Eleven of the 12 TEQ congeners were quantified (BZ#81 was not). Concentrations of BZ#126 were often below the detection limit, so concentrations of BZ#126 were expressed as the detection level. Chapter 3 provides a rationale for the approach taken to estimate TEQ congeners. As a check on the method, predicted TEQ-based egg concentrations using a biomagnification factor on the FISHRAND output were compared to measured egg concentrations.

Data validation provided an additional check on laboratory procedures and quantitation. Data that did not meet USEPA standards were rejected or qualified as estimated. There is no systematic bias in the laboratory results used in this report, and therefore associated uncertainty is low.

Based on the results of NOAA's mussel method detection limit (MDL) study (see USEPA, 1993a for details), the percent lipid determination for benthic invertebrates was considered to be estimated. Therefore, the percent lipid of benthic invertebrates was determined as the mean of all invertebrate taxa analyzed in the Phase 2 study. The variability seen in the percent lipid composition was probably associated with the small sample mass (1 gram wet weight) available on a sample by sample basis. The confidence of percent lipids was higher for fish samples, which had more material available for analysis.

The analytical chemistry program implemented by USEPA for the Hudson River ecological sampling was extremely sophisticated, requiring the use of state-of-the-art GC methodology. A total of 93 sediment, 120 USEPA funded fish, 115 NOAA funded fish, and 83 invertebrate samples were analyzed for 108 target and up to 38 non-target congeners. Considering the complexity of the program, the outcome of the analytical chemistry program was successful.

A total of 59,063 congener measurements were recorded, of which 925 values (1.6%) were rejected. A 98.4% completeness rate was achieved for the overall program, which successfully exceeded the 95% completeness requirement. The only principal congener which did not meet the completeness requirement was BZ #4 (93.5% completeness), however, this did not impair the overall integrity of the program.

A majority of all congener results (both detects and nondetects) were qualified as estimated or estimated and presumptively present (62%). The main reason for most of the qualifications was detection at concentrations below the calibrated quantitation limit and/or exceedances in the dual GC column precision criteria. Numerous congeners for nearly all sample delivery groups had calculated concentrations on each GC column which differed by more than 25%, but less than 50%, which warranted qualification as estimated values. With the level of background organic material present in Hudson sediments and in tissue samples, resultant interferences, particularly for congeners with low concentrations, likely caused these differences between the GC columns. Data users were recommended (see Appendix I) to consider all detect and nondetect results, which were estimated to be usable relative to the data quality objectives of the program.

The water-column sample analysis generally met data quality requirements (USEPA, 1997a). A small number congeners were rejected for dual gas chromatograph (GC) column imprecision. However, the completeness ratio (i.e., [number of total data - number of rejected data]/number of total data * 100) for the water-column monitoring study was 98.2% (USEPA, 1997a).

Data from a number of sources, each of which has used a slightly different standard in quantitating PCBs (i.e., Aroclors versus congeners, laboratory methods, etc.), were employed and the Phase 2 results were combined with these earlier data sources. Within the historical NYSDEC database of fish PCB concentrations, significant differences in reported total PCB body burden results can occur as a result of analytical method changes in 1975, 1977, and 1982 (Butcher et al., 1997). In addition, several additional changes in analytical methodology occurred in the 1990's. The earlier analytical packed-column methods are likely to significantly under-report the total concentration of mono- and dichlorobiphenyls than would be obtained using a congener-based capillary column methodology, as was done for the Phase 2 analyses.

Additional uncertainty in the interpretation of historical results is attributable to differences in laboratory determination of lipid content of fish tissue. PCBs are lipophilic, stored mainly in fatty tissue, and it is generally agreed that lipid normalization (i.e., expressing PCB body burden on a lipid basis) provides a more consistent basis for evaluating bioaccumulation.

Lipid-normalized PCB body burden is calculated as the reported wet-weight PCB concentration divided by the corresponding lipid concentration for the tissue sample. Although extraction and determination of lipid content is subject to uncertainty, it does not present a major problem in the modeling. Laboratory analyses for PCBs are based on a lipid extract ; thus the lipid normalized concentration should be consistent as long as the extraction procedure used for PCB and lipid analysis are consistent (USEPA, 1999c).

Inter-laboratory comparisons conducted by NYSDEC in September 1992 showed an average variability between laboratories of 10% percent in determining lipid content of biological specimens. NOAA (1997b) also evaluated lipid concentrations reported by Hazleton and Inchcape Laboratories. They found Hazleton values were consistently higher, attributed the interlaboratory differences to use of different extraction solvents, and based their lipid-normalized analyses only on Inchcape data. Both the NYSDEC and USEPA Phase 2 datasets were used in the FISHRAND bioaccumulation model. A consistent quantitation basis, or translation procedure, was used to standardize the Hazleton and Hale Creek results in the NYSDEC database (see USEPA, 1999c subchapter 4.1 for details).

To reduce the uncertainty associated with these issues, comparative analyses were performed to determine, to the extent possible, a consistent quantitation basis for historical analyses, and to estimate uncertainties present in calculated lipid-normalized PCB body burdens. Results of the analyses were employed so as to enhance study comparability while reducing inherent uncertainty.

6.3 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, since 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 subchapter 6.5.

The conceptual model used in this assessment is limited to animals exposed to 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 present assessment. Inclusion of these pathways in this assessment would increase the risks to the mink and raccoon, whose risks were calculated assuming 49.5% and 60% non-river related diet sources, respectively. In addition, risks for terrestrial species , such as shrews and moles (see tables in Chapter 2 for a complete listing), exposed to PCBs originating in the Hudson River may be above acceptable levels.

6.4 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.

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. 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.

Specific uncertainty factors used in this assessment include:

- NOAEL for phylogenetically similar species: Factor of 5 for extrapolations between families and factor of 10 for extrapolations between orders.

- LOAEL to NOAEL conversion: Factor of 10.

- Chronic LOAEL from subchronic data: Factor of 10.

Not all uncertainty factors were used in each derivation. Typically, no more than a factor of 10 was applied.

There is also uncertainty in the manner in which TEQ concentrations are characterized. Some toxicity studies used slightly different TEF 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%.

6.5 Exposure and Modeling Uncertainties

6.5.1 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, etc.) as well as variability (e.g., ingestion rate, body weight, etc.). Some parameters can be both uncertain and variable. It is important to distinguish

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uncertainty from variability. Variability represents known variations in parameters based on observed heterogeneity in the characteristics of a particular endpoint species. Variability can often be reduced with additional data collection, whereas uncertainty can be reduced directly through the confirmation of applied assumptions or inferences through direct measurement.

6.5.1.1 Food Chain Exposures

The exposure assessment evaluates two principal dietary pathways of PCB exposure to the avian and mammalian receptors: fish and invertebrate consumption. These are considered the only diet derived exposure sources in the assessment for evaluation of PCBs solely from riverine sources.

The invertebrate component of the diet was assessed from samples of the infaunal community collectively as a total component with no preference or dominance being prescribed to specific taxa or trophic level groups. This collective assessment approach introduces some degree of uncertainty with the heterogeneity of benthic community composition varying by location, taxa composition and sample mass. However, statistical analyses (t-tests) based on the USEPA Phase 2 1993 dataset found no significant difference between PCB concentrations among taxa at a given sampling location, although in many cases sample sizes were small. The principal components analysis presented in Appendix K found similar results.

The fish component was divided into two distinct, exposure groups based upon fish size and observed tendencies of PCBs to biomagnify at higher trophic levels in aquatic food chains. Fish were categorized into: forage fish species (<10 centimeters total length and diets representative of primary or secondary consumers) and piscivorous fish species (>20 cm total length with diets of secondary and/or tertiary consumers). There is some uncertainty associated with fish that fall into the smaller size class as juveniles and the larger size class as adults.

Body mass plays a significant role in development of exposure dosages to contaminants. Body mass plays a quantitative role in the water, dietary and incidental sediment ingestion pathways as part of the average daily dosage term from each pathway on a per kilogram body weight basis. As the ERA considers risks to receptors from the Hudson River, body masses from endemic Hudson River populations are associated with only a small degree of uncertainty.

Representative mammalian body masses were available for individuals/populations from the Hudson River Valley region from historical specimens curated in the New York State Museum (NYSM) (Bopp, 1999) which were compared to ranges provided in USEPA (1993b). This comparison showed similar body masses for endemic populations to Hudson Valley or NYS to other North American populations suggesting that the Hudson River populations are typical in meristic measurements. Therefore, body masses employed in the exposure pathway modeling for mammalian receptors are considered reliable and representative of Hudson River populations.

Body mass was not recorded for most historical avian specimens collected from the Hudson River Valley at the time of collection (Bopp, 1999). The tree swallow was the only avian receptor with Hudson River specific body masses available (Secord and McCarty, 1997). A limited number of body mass data was available for the bald eagle (N=3). However, all specimens were considered juveniles or described as emaciated when collected and were therefore excluded from consideration. Body masses for adults of the belted kingfisher, great blue heron, mallard and bald eagle were based on mean or median body masses provided in Dunning (1993) or (USEPA, 1993b). The use of a centralized value introduces some uncertainty when used to represent a meristic characteristic for a specific population. On a cumulative dosage basis, a higher body mass estimate would reduce the estimated daily cumulative daily dosage fraction of PCBs on a per kg body weight basis. Likewise, a lower body mass estimate would result in a higher average daily dosage estimate. Since it is not known if typical body masses for Hudson River populations are indicative of either extreme in the range of body masses, no systematic bias is associated with these estimates.

Dietary ingestion rates may also contribute to uncertainty. A dietary ingestion rate for little brown bat populations from the Hudson River Valley (Millbrook, NY) is the only ingestion rate estimate available for wild populations for the mammalian receptors considered. Daily ingestion rates for the mink, river otter and raccoon were based upon captive studies or were calculated from allometric relationships derived from USEPA (1993b) and Nagy (1987). The ingestion rate applied to mink (Bleavins and Aulerich, 1981) and river otter (Harris, 1968) are based upon captive populations. Captive based studies consider metabolic and physiological needs of animals under controlled environments and nutritional demands. Such studies are performed for purposes of identifying optimal nutritional quality and metabolic demands specifically for captive populations (Bleavins and Aulerich, 1981). These studies remain confined to a select age groups and/or sexes and remain limited in duration. Diets typically consist of processed feeds (Bleavins and Aulerich, 1981) or whole food (Harris, 1968). Timing and availability of food in such studies can be provided on a consistent temporal or on demand basis.

Diets of wild populations can vary based upon age, sex, physical state of the individual, prey availability, and nutritional quality of the diet. Consequently, application of ingestion rates from captive population studies requires the assumption of similar metabolic demands and nutritional needs in wild populations. Such an assumption will likely underestimate the ingestion rates in wild populations of mink and otter as increased physiological (i.e., the pursuit and capture of prey) and behavioral (i.e., territorial) demands would likely increase their corresponding metabolic rates and their corresponding ingestion needs (Nagy, 1987). Without confirmatory studies in free living populations, the application of ingestion rates from captive studies to wild populations may result in the potential underestimation of dietary derived exposure to PCBs in the mink and river otter assessments.

For the raccoon and all avian receptors, food ingestion rates are estimated based upon guidance in USEPA (1993b) which recommends an allometric estimation methodology using diet, normalized metabolic rate and metabolizable energy content of specific foods consumed (USEPA, 1993b). Use of this methodology does incorporate some degree of uncertainty in the absence of field verification. Ingestion rates are calculated as the quotient of the species specific normalized metabolic rate and the average metabolizable energy content of the diet. Estimation of the average gross energy content in wildlife foods remains limited to select broad phylogenic groups and is rarely available for species level evaluations for prey included in the diet. Reliance upon the gross energy estimates for representative taxa groups introduces some uncertainty in derivation of the ingestion rates, as it is assumed that the gross energy content and assimilative efficiency of select groups of invertebrates and fish taxa are equivalent to other freshwater benthic invertebrates taxa. This assumption in the energy content of the diet can influence the ingestion rate estimate if under- or overestimated. An overestimate of the average metabolizable energy in the diet will decrease the ingestion rate (i.e, actual metabolic average is lower than estimated). An underestimate of the metabolic average results in an overestimate of the ingestion rate. To be consistent in application and minimize error across receptors, gross energy contents for aquatic invertebrates and bony fish were held constant. Additionally, assimilative efficiencies for mammalian and avian receptors were applied on a consistent basis for all receptors.

Water Ingestion Rates

Water ingestion rates for avian and mammalian receptors were estimated based upon allometric relationships developed for mammals and birds outlined in USEPA (1993b). For this pathway, it is assumed that avian and mammalian receptors utilize the Hudson River as their exclusive source for drinking water. This assumption excludes the use of non-contaminated sources in close proximity to the Hudson River. The dosage estimate for water ingestion does not account for metabolic or dietary derived sources of water for the individual receptors. Consequently, the allometric methods assume that hydration demands in the receptors are solely accounted for by direct ingestion of surface water. This assumption may result in an overestimate of surface water derived PCBs exposures through the drinking water pathway through exclusion of metabolic and dietary sources.

Prey Ingestion Rates

The most prevalent exposure pathway for which endpoint receptors are exposed to PCBs is via dietary ingestion of contaminated prey items. Dietary composition estimates the fraction of total intake represented by each food type (USEPA, 1993b). For this assessment, the basis for the exposure to PCBs from Hudson River sources is limited to ingestion of aquatic invertebrates and fish solely from the Hudson River. The dietary characteristics of the mammalian and avian receptors being evaluated span a diverse range of dietary percentages represented by fish and/or aquatic invertebrates.

Composition of fish and aquatic invertebrates in diets can be affected by a variety of elements which can contribute to the uncertainty in dietary exposure estimates. These factors include environmental factors (e.g., seasons, geographic region, prey susceptibility and abundance) and receptor specific factors (e.g., age, sex and reproductive state).

The little brown bat was the only receptor for which Hudson River diet data for endemic populations were available. The mink, river otter, and raccoon diets relied upon results for other NYS populations. Neither specific habitat types nor location for samples of mink or river otter diets were specified in the available studies. This primary information is supported by secondary information from wildlife specialists with NYSDEC. Most of the studies on stomach content or scat analyses did not discriminate by sex or age for the mink, river otter or raccoon. Consequently, age or sex related effects related to diet composition could not be inferred for mink, river otter or raccoon diets for NYS populations. For the little brown bat, stomach contents are differentiated by sex and age (> 1 year old). Therefore, age or sex could be not be assessed for effects on diet composition in the mink, river otter, or raccoon for the exposure assessment.

A distinct preference for dominance of aquatic based food items (i.e., fish and invertebrates) in winter was observed for mink and river otter based upon multiple samples from NYS populations and preliminary results of the NYSDEC River Otter Reintroduction Program. Therefore a winter maximum diet composition for aquatic prey items was applied to assess risks to these two receptors.

Raccoon diet data for NYS and Minnesota populations were limited to marsh type habitats in summer. The median aquatic invertebrate component was applied and a minor fish component added based upon potential opportunistic exposures documented in other studies. The greatest uncertainty in diet composition appears related to the raccoon diet being based in part on professional judgement for inclusion of a minor fish component in the exposure assessment.

Tree swallow diets for the Hudson River Valley were based upon bolus sampling conducted by Secord and McCarty (1997) on the Hudson River near Saratoga Springs, NY. Secondary sources for diet composition included Robertson et al (1992) and McCarty (1999). A diet of 100% flying insects with partial aquatic life histories was applied in the exposure assessment for dietary pathway for this receptor.

Mallard diet information for Hudson River or NYS populations in regional proximity were not available. Diet studies provided in USEPA (1993b) were reviewed and evaluated for seasonal or habitat specific trends. The invertebrate component of the mallard diet increases during the spring and summer to a near equal percentage as systemic vegetation and seeds form a lower percentage of the diet in fall and winter. No fish were documented in the diets summarized in USEPA (1993b) and are not considered as in the mallard exposure assessment. Based upon the trend towards a higher percentage of the invertebrate component in spring and summer, a 70% aquatic invertebrate component was applied in the exposure assessment.

The belted kingfisher diet was based primarily on south-central NYS populations (Gould unpublished data cited in Salyer and Lagler, 1946) and Ohio populations (Davis, 1982). Bull (1998) and Brooks and Davis (1987) were used as secondary sources. The diet is considered to consist exclusively of forage fish species and aquatic invertebrates. Dietary percentages of 78% fish (as forage fish) and 22% aquatic invertebrates were used in the exposure assessment.

Great blue heron diet information for Hudson River or NYS populations in regional proximity was not available. The primary sources for diet information for the great blue heron is Alexander (1977) for Michigan populations and Hoffman (1978) for southwestern Lake Erie populations. The diets are derived exclusively from aquatic sources for both studies. Secondary sources for dietary information include Eckert and Karalus (1988) and Krebs (1974). The fish fraction observed was 95% to 98% (composed primarily of forage fish) and 1% aquatic invertebrates. These values were applied in the exposure assessment.

Primary sources of bald eagle dietary information for Hudson River resident populations included discussions with NYSDEC wildlife specialist (Nye, 1999) and Bull (1998). Secondary sources for information included Nye and Suring (1978) and diet studies provided in USEPA (1993b). Since NYS-specific information was available, the diet composition of 100% fish used can be considered reliable and applicable to Hudson River populations.

Incidental Sediment Ingestion

Of the receptors evaluated, only the mallard and raccoon have published estimates for ingestion of soil/sediment. The values of 2.0% for the mallard and 9.4% for the raccoon are quantified estimates based upon Beyer et al. (1994) and are considered reliable for application to Hudson River populations.

Estimates of incidental sediment ingestion were made based upon: feeding behavior used to capture prey, prey consumption, and nesting/resting habitats of each species. Both the tree swallow and little brown bat feed primarily on flying insects that are captured and consumed in flight. The tree swallow nests in trees, while the little brown bat roosts in sheltered locations, such as caves and abandoned buildings. These feeding and roosting preferences result in incomplete pathways for incidental sediment ingestion. Therefore, a 0% percent incidental sediment ingestion rate was applied to both receptors.

The great blue heron, belted kingfisher, and bald eagle were characterized as primarily piscivorous in diet. All three receptor species visually follow their prey and seize the specific prey item using their bill (i.e., the great blue heron and belted kingfisher) or talons (i.e., the bald eagle). The great blue heron may ingest some incidental sediment during prey capture, prey consumption, and grooming. Therefore, an incidental sediment ingestion rate of 2%was used for the great blue heron. A sediment ingestion rate of 1% was applied to the belted kingfisher, which has little contact with sediments during feeding, but may ingest some sediments during grooming because it nests in river banks. A sediment ingestion rate of 0% was applied to the bald eagle based on its feeding and nesting habits.

These rates do not consider sediment contained in the digestive system of fish prey. A study evaluating the stomach contents of bluegills reported that an average of 9.6% of the diet consisted of detritus and sediment (Kolehmainen, 1974). Since many of the fish analyzed for this study were filets, rather than whole fish, the incidental sediment ingestion rate of piscivorous receptors may be underestimated.

Stomach content and scat analyses on mink and feeding behavior of otter described by Liers (1951) suggest either the presence of soil/sediment component in the diet or the potential for exposure to occur. Stomach content and scat analyses of mink from NYS revealed trace quantities of sand present. The term "trace" was assumed to account for less than or equal to 1% of the diet based upon the frequency distribution of other items. Based upon these reports and the potential for the mink to also ingest sediments during grooming, a 1% incidental ingestion composition in the diet of the mink was applied. No quantitative dietary information regarding the occurrence of soils/sediments in the diet of the river otter was available. Liers (1951) observed that sediments may be ingested when river otters feed on bivalves. Although the river otter is considered to feed exclusively on fish, the potential for river otter to ingest sediments during feeding and grooming exists. A 1% ingestion rate, used for the mink, was also applied to the river otter. These values may underestimate the actual diet composition if the invertebrate component of the mink and otter is under represented.

6.5.2 Sensitivity Analysis for Risk Models

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 Chapter 3. 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, body weights, etc.). 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.. 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).

6.5.3 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 this assessment, model error is probably not a significant source of uncertainty. Relationships between trophic levels and food web components in the Hudson River are well understood.

6.5.3.1 Uncertainty in FISHRAND Model Predictions

The literature review and experimental data collected for the Hudson River has shown that:

- River ecosystem characteristics vary significantly from one location to another depending on flow rate, depth, sediment structure, etc.; and,

- Available experimental data for the model parameters are somewhat unreliable due to the scarcity of measurements.

Moreover, most of the measurements are not easily related to the FISHRAND generic input parameters because experimental measurements are taken at a specific time and space. The FISHRAND model parameters are, in contrast, values corresponding to averages over time, space and species.

The effect of variation of all input parameters on all model outputs were evaluated in a sensitivity analysis using a Monte-Carlo methodology. This is a powerful tool to analyze uncertainties in model predictions. 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. The combination of all possible outputs generated in this manner is used to construct the distribution of model outputs, which reflect the influence of the undetermined parameters on the output values.

The partial rank regression technique (Morgan and Henrion, 1990) is used as a formal method to determine the most important parameters for model performance. If the partial rank regression coefficient (PRRC) is close to 1 or -1 for a specific input model parameter, this parameter significantly influences model output. The results of this analysis (presented in greater detail in USEPA, 1999c) show that the percent lipid in fish, percent lipid in prey items, and K_{ow} contribute most to the uncertainty in the model output. Based on comparisons between predicted and observed concentrations (USEPA, 1999c), approximately a factor of two uncertainty is contained in the lipid-normalized fish body burden estimates, and the wet weight body burden estimates are within an order of magnitude, but likely also closer to a factor of two.

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Chapter 7

7.0 CONCLUSIONS

This chapter summarizes the results of the risk assessment. Each assessment endpoint and associated measurement endpoint is presented, along with a summary of the results. The results of the risk characterization are evaluated in the context of the uncertainty analysis 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 originating in the Hudson River. The weight-of-evidence approach considers both the results of the toxicity quotient analysis and insights from the field-based observational data.

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

Does the benthic community structure reflect the influence of PCBs?

The analysis shows a reduced macroinvertebrate community, indicating the potential for risk above regional conditions due to site-related influences.

Do measured and modeled sediment PCB concentrations exceed appropriate guidelines?

Measured sediment concentrations based on the 1993 USEPA dataset exceed the NYSDEC chronic criteria for the protection of benthic aquatic life on both an average and 95% UCL basis for all locations in the river except RM100 (average only).

Do measured and modeled PCB water concentrations exceed appropriate criteria and/or guidelines for the protection of benthic invertebrates?

Based on observed data from the USEPA 1993 sampling program, both the average and 95% UCL total PCB water concentrations exceed the USEPA/NYSDEC chronic criteria for the protection of benthic aquatic life in the Upper Hudson River. In the Lower Hudson River, chronic criteria on an average basis are exceeded down to RM88.9, and are exceeded at all locations for the 95% UCL.

Modeled sediment concentrations from the HUDTOX model exceed the NYSDEC chronic criteria for the protection of benthic aquatic life for the duration of the modeling period across all three locations.

Summary: Benthic community structure as a food source for local fish populations was assessed using three lines of evidence. The first was to evaluate community structure and abundance relative to regional conditions. The second was to compare measured and modeled water column concentrations to water quality criteria developed specifically for the protection of aquatic life. The third was to compare measured and modeled sediment concentrations to sediment guidelines developed specifically for the protection of benthic invertebrates. All three lines of evidence suggest an adverse effect of PCBs on benthic invertebrate populations serving as a food source to local fish. Uncertainty in this analysis is considered low.

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

Do measured and/or modeled total PCB body burdens in local forage fish exceed benchmarks for adverse effects on forage fish reproduction?

Measured PCB body burdens for forage fish exceed the field-based NOAEL developed in Chapter 4 for pumpkinseed for all locations in the river on both an average and 95% UCL basis, but only exceed the laboratory-based NOAEL developed for spottail shiners in the TI Pool. Modeled PCB body burdens for pumpkinseed exceed the field-based NOAEL developed in Chapter 4 on a median and 95th percentile basis for the duration of the modeling period at both the TI Pool and Stillwater. At RM154, modeled body burdens exceed the field-based NOAEL until 2017 for the median and for the duration of the modeling period for the 95th percentile. Modeled spottail shiner body burdens exceed the laboratory-based NOAEL on a 95th percentile basis until 1995 and 1996 in the TI Pool and at Stillwater, respectively, but show no other exceedances on either a LOAEL or NOAEL basis.

These results suggest the potential for adverse effects on forage fish reproduction based on exceedances of measured and modeled body burdens to TRVs developed in Chapter 4. Body burdens were measured and modeled on a whole body basis, thus, there was no factor applied to convert from a standard fillet to a whole body concentration. There were no interspecies or subchronic to chronic uncertainty factors applied in the development of the field-based NOAEL selected for the pumpkinseed. All of the uncertainties in the HUDTOX exposure concentrations and in FISHRAND suggest approximately a factor of two uncertainty in the predicted body burdens. If the model results were to decrease by a factor of two, pumpkinseed would still exceed the field-based NOAEL on a median and 95th percentile basis for both the TI Pool and Stillwater. The 95th percentile predicted body burden for pumpkinseed at RM154 would still exceed for the duration of the modeling period, and would exceed the NOAEL until approximately 2004 on a median basis.

Do modeled PCB body burdens expressed on a TEQ basis in local forage fish exceed benchmarks for adverse effects on forage fish reproduction?

Modeled PCB body burdens in pumpkinseed exceed the laboratory-based NOAEL on a median basis until 1998 for the median in the TI Pool and at Stillwater. On a 95th percentile basis, modeled pumpkinseed body burdens exceed the laboratory-based NOAEL until 2010 in the TI Pool and at Stillwater. Modeled pumpkinseed body burdens on a TEQ basis do not exceed the laboratory-based NOAEL at RM154. On a LOAEL basis, the modeled 95th percentile body burden exceeds one until 1999 in the TI Pool and at Stillwater. All other percentiles do not exceed the laboratory-based LOAEL for pumpkinseed. None of the predicted spottail shiner body burdens exceed the TEQ-based NOAEL or LOAEL.

These results suggest the potential for adverse effects on forage fish reproduction based on exceedances of measured and modeled body burdens to TRVs developed in Chapter 4. Body burdens were expressed on a lipid-normalized basis (and measured for whole body fish), thus, there was no factor applied to convert from a standard fillet to a whole body concentration. There is uncertainty in the specific TEF used in the derivation of the NOAEL and LOAEL, but no interspecies or subchronic to chronic uncertainty factors were applied. Consequently, uncertainty in the derived TRV is low.

Do measured and/or modeled total PCB body burdens in local omnivorous fish exceed benchmarks for adverse effects on omnivorous fish reproduction?

Measured PCB body burdens for brown bullhead exceed the laboratory-based NOAEL developed in Chapter 4 for all locations in the river on both an average and 95% UCL basis, but only exceed the laboratory-based LOAEL on a 95% UCL and maximum basis at Stillwater. In the TI Pool, modeled brown bullhead concentrations exceed the laboratory-based LOAEL on an average basis until 1995, but exceed on a 95% UCL and maximum basis for all years. Modeled brown bullhead body burdens exceed the laboratory-derived NOAEL across all percentiles for the TI Pool and Stillwater. At RM154, the median predicted brown bullhead body burden exceeds the laboratory-based NOAEL until 2009, and the 95th percentile exceeds for the duration of the modeling period. The modeled brown bullhead body burden exceeds the laboratory-based LOAEL until 1998 on a median basis in the TI Pool, and until 2015 on a 95th percentile basis. At Stillwater, the 95th percentile exceeds the LOAEL until 2002, and the modeled body burdens do not exceed the laboratory-based LOAEL at RM154.

These results suggest the potential for adverse effects on omnivorous fish reproduction based on exceedances of measured and modeled body burdens to TRVs developed in Chapter 4. Body burdens were measured and modeled on a fillet basis, thus, a factor of 1.5 was applied to convert from a standard fillet to a whole body concentration. An interspecies factor of 10 was applied in the derivation of the TRVs. All of the uncertainties in the HUDTOX exposure concentrations and in FISHRAND suggest approximately a factor of two uncertainty in the predicted body burdens. Uncertainty in the predicted toxicity quotients for the brown bullhead is therefore roughly within a factor of 30. On this basis, the predicted 95th percentile body burden would still exceed the laboratory-based NOAEL in the TI Pool until 2005. However, none of the other comparisons would show exceedances. Note that uncertainty may overestimate or underestimate the predicted results.

Do modeled PCB body burdens expressed on a TEQ basis in local omnivorous fish exceed benchmarks for adverse effects on omnivorous fish reproduction?

Measured average brown bullhead lipid-normalized body burdens fall below the laboratory-based NOAEL and laboratory-based LOAEL expressed on a TEQ basis at both Stillwater and in the TI Pool for all years. Only the 95% UCL and maximum exceed the NOAEL in the TI Pool for one and two years, respectively. Modeled body burdens also did not exceed any TRVs.

These results suggest low potential for adverse effects on omnivorous fish reproduction based on exceedances of measured and modeled body burdens to TRVs developed in Chapter 4. Body burdens were measured and modeled on lipid-normalized basis, thus, there was no necessity to convert from a standard fillet to a whole body concentration. There is uncertainty in the specific TEF used in the derivation of the NOAEL and LOAEL, but no interspecies or subchronic to chronic uncertainty factors were applied. Consequently, uncertainty in the derived TRV is low.

Do measured and/or modeled total PCB body burdens in local piscivorous and semi-piscivorous fish exceed benchmarks for adverse effects on fish reproduction?

White Perch: Measured PCB body burdens for white perch exceed the field-based NOAEL developed in Chapter 4 for RM113 and RM152 on an average and 95% UCL basis. The maximum measured concentration in white perch exceeds the field-based TRV. The modeled white perch concentrations exceed the field-based NOAEL until 2013 and 2008 on a median basis at the TI Pool and Stillwater, respectively. The modeled 95th percentiles exceed the field-based NOAEL for the duration of the modeling period at the TI Pool and at Stillwater, and until 2004 at RM154.

Yellow Perch: The measured yellow perch concentrations exceed the laboratory-based NOAEL and LOAEL in the TI Pool for the average, 95% UCL, and maximum. At Stillwater, average measured body burdens at Stillwater exceed the NOAEL but not the LOAEL. Modeled median yellow perch body burdens exceed the laboratory-based NOAEL until 2016, 2017, and 2006 at the TI Pool, Stillwater, and RM154, respectively. The modeled 95th percentiles exceed the NOAEL for all three locations for the duration of the modeling period. However, when the modeled body burdens are compared to the laboratory-based LOAEL, the only exceedances are for the 95th percentile for several years at the TI Pool and at Stillwater.

Largemouth Bass: Measured largemouth bass concentrations exceed the field-based NOAEL for the average, 95% UCL, and maximum at RM113, Stillwater, and the TI Pool. A LOAEL was not derived for this species (see Chapter 4). Modeled largemouth bass concentrations exceed the field-based NOAEL for all locations and percentiles for the duration of the modeling period.

Striped Bass: Measured striped bass concentrations exceed the field-based NOAEL derived in Chapter 4 at RM152 for the average and 95% UCL for the range of observations. At RM112, the average and 95% UCL body burdens exceed the NOAEL in 1993 and 1994. In 1996, the observed 95% UCL compared to the NOAEL was exactly one. In 1995 and 1996, the only exceedances between measured concentrations and observations occurred at RM152. However, in 1993, the average and 95%UCL at RM33, RM74, RM112, and RM152 all exceed the NOAEL.

These results suggest the potential for adverse effects on piscivorous and semi-piscivorous fish reproduction based on exceedances of measured and modeled body burdens to TRVs developed in Chapter 4. Body burdens were measured and modeled on a fillet basis for all species. For largemouth bass, a factor of 2.5 from the literature was available to convert concentrations from a standard fillet to a whole body concentration. In the absence of data, none of the other fish body burdens were adjusted (that is, measured and modeled fillet concentrations were compared to TRVs). This is likely to underestimate body burdens for white and yellow perch, and striped bass. An interspecies factor of 10 was applied in the derivation of the TRVs for all species. All of the uncertainties in the HUDTOX exposure concentrations and in FISHRAND suggest approximately a factor of two uncertainty in the predicted body burdens. Uncertainty in the predicted toxicity quotients for the largemouth bass is therefore roughly within a factor of 50. On this basis, measured body burdens would still show toxicity quotients greatly in excess of one at Stillwater and the TI Pool on a 95% UCL basis. Predicted median and 95th percentile body burdens would still exceed the field-based NOAEL for several years in the TI Pool and at Stillwater. Since the magnitude of the difference between fillet and whole body concentrations is unknown for the other species, the magnitude of uncertainty cannot be addressed. Note that uncertainty may overestimate or underestimate predicted results.

Do modeled PCB body burdens expressed on a TEQ basis in local piscivorous and semi-piscivorous fish exceed benchmarks for adverse effects on omnivorous fish reproduction?

White Perch: Measured PCB body burdens for white perch exceed the laboratory-based NOAEL developed in Chapter 4 for RM113 and RM152 on an average and 95% UCL basis. The maximum measured concentration in white perch exceeds the field-based TRV. The modeled white perch concentrations exceed the field-based NOAEL until 2013 and 2008 on a median basis at the TI Pool and Stillwater, respectively. The modeled 95th percentiles exceed the field-based NOAEL for the duration of the modeling period at the TI Pool and at Stillwater, and until 2004 at RM154.

Yellow Perch: The measured yellow perch concentrations exceed the laboratory-based NOAEL and LOAEL in the TI Pool for the average, 95% UCL, and maximum. At Stillwater, average measured body burdens at Stillwater exceed the NOAEL but not the LOAEL. Modeled median yellow perch body burdens exceed the laboratory-based NOAEL until 2016, 2017, and 2006 at the TI Pool, Stillwater, and RM154, respectively. The modeled 95th percentiles exceed the NOAEL for all three locations for the duration of the modeling period. However, when the modeled body burdens are compared to the laboratory-based LOAEL, the only exceedances are for the 95th percentile for several years at the TI Pool and at Stillwater.

Largemouth Bass: Measured largemouth bass concentrations exceed the laboratory-based NOAEL and LOAEL for the average, 95% UCL, and maximum at RM113, Stillwater, and the TI Pool. In the TI Pool, the measured average, 95% UCL and maximum concentrations exceed ten for all years except for the LOAEL in 1996.

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Modeled 95th percentile largemouth bass body burdens exceed the laboratory-based NOAEL for the duration of the modeling period in the TI Pool and at Stillwater, and exceed until 2007 at RM154. Median modeled body burdens exceed the NOAEL until 2009 in the TI Pool, until 2007 at Stillwater, and until 1996 at RM154. Modeled 95th percentile largemouth bass body burdens exceed the laboratory-based LOAEL for the duration of the modeling period in the TI Pool and at Stillwater, and exceed until 1998 at RM154. Median modeled body burdens exceed the LOAEL until 1997 in the TI Pool and at Stillwater, and not at all at RM154.

Striped Bass: Measured 95% UCL striped bass concentrations exceed the laboratory-based NOAEL derived in Chapter 4 at all river miles during 1996. At RM152, the average and 95% UCL body burdens exceed the NOAEL and LOAEL for all years. Modeled results are interpreted as for largemouth bass.

These results suggest the potential for adverse effects on piscivorous and semi-piscivorous fish reproduction based on exceedances of measured and modeled lipid-normalized body burdens to egg-based TRVs developed in Chapter 4. Since body burdens were lipid-normalized, the potential adjustment factor for the conversion of standard fillet to whole body concentrations was not required. No interspecies or subchronic-to-chronic uncertainty factors were applied in the derivation of the TRVs for these species. All of the uncertainties in the HUDTOX exposure concentrations and in FISHRAND suggest approximately a factor of two uncertainty in the predicted body burdens. In general, the lipid-normalized model results show better agreement with observed results than wet weight concentrations, suggesting a higher degree of confidence in these results. Uncertainty in the predicted toxicity quotients for these species is therefore considered low. Note that uncertainty may overestimate or underestimate the predicted results.

Do measured and modeled PCB water concentrations exceed appropriate criteria and/or guidelines for the protection of wildlife?

Comparison of measured average and 95% UCL total PCB concentrations exceed the NYSDEC criteria developed for the protection of wildlife on the basis of bioaccumulation potential. All locations in the river exceed ten for all comparisons. On a 95% UCL basis, all locations down to RM100 exceed 100, and at Stillwater on an average basis. Modeled water concentrations from HUDTOX also exceed the NYSDEC criteria for the duration of the modeling period.

These results suggest the potential for risk based on the potential of PCBs to bioaccumulate from water to fish and wildlife. A preliminary analysis of sources of uncertainty in the HUDTOX model suggest a factor of two uncertainty. Even if the model results were to decrease by a factor of two, the NYSDEC wildlife bioaccumulation criterion would be exceeded at all three modeling locations for the duration of the modeling period.

Do measured and modeled sediment PCB concentrations exceed appropriate guidelines?

Measured sediment concentrations exceed the NOAA guidelines on a TEC and MEC basis for all locations. The EEC is exceeded in the Upper Hudson River on an average and 95% UCL basis. Modeled sediment concentrations from HUDTOX exceed all three guideline values on both an average and 95% UCL basis for the three Upper Hudson River locations. Both the Persaud LEL guideline and the NYSDEC wildlife criteria are exceeded for all measured locations on both an average and 95% UCL basis.

What do the available field-based observations suggest about the health of local fish populations?

The continued presence of the receptor species over time indicates that exposure levels of PCBs are not high enough ***to prevent*** reproduction of these species or recruitment of new individuals to these areas. The qualitative observational data presented in Chapter 5.1 can not be used to provide insight into the possibility that PCBs have ***reduced or impaired*** reproduction or rates of recruitment. Risks to these receptors likely exists even if the fish species are able to maintain themselves in these areas.

The toxicity quotient approach comparing measured and predicted body burdens to TRV values is required to judge the possible magnitude of potential risks to fish species. Based on the analysis presented in Chapter 5.1, the potential for adverse effects resulting from exposure to PCBs is likely to occur for the piscivorous and semi-piscivorous fish species.

Summary: Risks to local fish populations were evaluated using seven lines of evidence. Four of these are based on comparing measured and modeled body burdens of PCBs to a number of toxicity reference values derived from the literature; one is based on comparing measured and modeled water column concentrations to water quality criteria developed for the protection of aquatic life; one is based on comparing measured and modeled sediment concentrations to guidelines developed for the protection of aquatic life; and the fourth is based on qualitative field observations. Collectively these lines of evidence indicate that current and future PCB exposures not of a sufficient magnitude to prevent reproduction or recruitment of common fish species in all areas investigated. However, body burdens are above TRVs in the Upper Hudson River for all species, and body burdens consistently exceed TRVs for the upper trophic level fish such as largemouth bass. Model results show that body burdens are expected to remain above these levels for the duration of the modeling period for several of the upper trophic level fish species. There is a moderate degree of uncertainty in the modeled body burdens used to evaluate exposure, and is at most an order of magnitude uncertainty in the various TRVs (although for the TEQ-based assessment, no uncertainty factors were applied). Despite this potential uncertainty, body burdens still exceed benchmarks for a number of the species. Measured and modeled water and sediment concentrations typically exceed all but the least stringent guidelines at most locations in the river. All concentrations in the Upper Hudson river show exceedances.

7.3 Assessment Endpoint: Protection and Maintenance (i.e., Survival, Growth, and Reproduction) of Local Insectivorous Birds

Do modeled total PCB dietary doses to insectivorous birds exceed benchmarks for adverse effects on reproduction?

Modeled total average PCB dietary doses to the tree swallow exceed the field-based NOAEL derived in Chapter 4 at Stillwater using 1993 data. The modeled 95% UCL dietary doses exceed the field-based NOAEL in the TI Pool and at Stillwater using 1993 data. There were no other exceedances on the basis of 1993 data. Modeled 95% UCL dietary doses to tree swallows under future conditions exceed the field-based NOAEL until 1995. There were no other exceedances. This field-based NOAEL was derived on the basis of Hudson River data.

Do modeled TEQ-based dietary doses of PCBs to insectivorouos birds exceed benchmarks for adverse effects on reproduction?

The predicted 95% UCL dietary dose to tree swallows exceed the field-based NOAEL at Stillwater on the basis of 1993 data. There were no other exceedances. Modeled dietary doses did not exceed the field-based NOAEL.

Do modeled total PCB concentrations in insectivorous bird eggs exceed benchmarks for adverse effects on reproduction?

Modeled total average PCB concentrations in the eggs of tree swallows exceed the field-based NOAEL derived in Chapter 4 at Stillwater using 1993 data. The modeled 95% UCL egg concentrations exceed the field-based NOAEL in the TI Pool and at Stillwater using 1993 data. There were no other exceedances on the basis of 1993 data. Modeled 95% UCL egg concentrations to tree swallows under future conditions exceed the field-based NOAEL until 1995. There were no other exceedances. This field-based NOAEL was derived on the basis of Hudson River data.

Do modeled TEQ-based PCB concentrations in insectivorous bird eggs exceed benchmarks for adverse effects on reproduction?

The predicted 95% UCL PCB concentrations in the eggs of tree swallows exceed the field-based NOAEL at Stillwater on the basis of 1993 data. There were no other exceedances. Modeled egg concentrations did not exceed the field-based NOAEL.

Do measured and modeled whole water concentrations exceed appropriate criteria and/or guidelines for the protection of wildlife?

Comparison of measured average and 95% UCL total PCB concentrations exceed the NYSDEC criteria developed for the protection of wildlife. All locations in the river exceed 100 for all comparisons, and some exceed 1000. Modeled water concentrations from HUDTOX also exceed the NYSDEC criteria for the duration of the modeling period.

These results suggest the potential for risk based on the criteria developed for the protection of wildlife. A preliminary analysis of sources of uncertainty in the HUDTOX model suggest a factor of two uncertainty. Even if the model results were to decrease by a factor of two, the NYSDEC wildlife criterion would be exceeded at all three modeling locations for the duration of the modeling period.

What do the available field-based observations suggest about the health of local insectivorous bird populations?

Tree swallows (*Tachycineta bicolor*) commonly observed along the Upper Hudson River during the spring when they are feeding in preparation for breeding (McCarty and Secord, 1999). These researchers have observed decreased reproductive success relative to reference areas and the occurrence of unusual parental and/or nesting behavior relative to reference areas. Although tree swallows are able to use this stretch of the Upper Hudson River, behavioral endpoints have shown to differ significantly from reference areas. The behavioral endpoints have been shown to be statistically related to PCB exposures.

Although the modeled dietary doses and egg concentrations indicate only very few exceedances of field-based NOAELs, the observational studies suggest that PCB exposures may have significant effects on tree swallow nesting behavior. Alterations in behavior may also be reflected in changes in reproductive success of this species over time.

Summary: Risks to insectivorous bird species were evaluated using six lines of evidence. Four of these are based on comparing modeled dietary doses and egg concentrations to various toxicity reference values developed from Hudson River field data; one is based on comparing measured and modeled water column concentrations to water quality criteria developed for the protection of aquatic life; and one is based on qualitative field observations. Collectively, these lines of evidence indicate that current and future concentrations of PCBs are not of a sufficient magnitude to prevent reproduction of insectivorous bird species, but that anomalous behavior has been observed at these levels that may be reflected at the population level. There is a moderate degree of uncertainty in the dietary dose and egg concentration estimates, but all of the data used for the tree swallow was obtained from Hudson River information. Measured and modeled water column concentrations exceed criteria developed for the protection of wildlife at all locations under current conditions, and for the duration of the modeling period for future conditions.

7.4 Assessment Endpoint: Protection and Maintenance (i.e., Survival, Growth, and Reproduction) of Local Waterfowl

Do modeled total PCB dietary doses to waterfowl exceed benchmarks for adverse effects on reproduction?

Average and 95% UCL modeled PCB dietary doses to the mallard duck exceed the laboratory-based NOAEL in the Upper Hudson River on the basis of 1993 data. In the Lower Hudson River, the modeled 95% UCL dietary doses exceed the NOAEL, and the average doses exceed at RM143.5 and RM137.2. Modeled average and 95% UCL dietary doses to the mallard duck did not exceed the LOAEL in the Lower Hudson River, and the average exceeds only at Stillwater in the Upper Hudson River. The modeled 95% UCL dietary doses to the mallard duck exceed the LOAEL in the TI Pool and at Stillwater on the basis of 1993 data. Future average and 95% UCL modeled results exceed the NOAEL at both the TI Pool and at Stillwater, but only until approximately 1999 at RM154. Modeled results only exceed the laboratory-based LOAEL in the TI Pool until 1996.

These results suggest the potential for adverse reproductive effects as a result of PCB exposure to waterfowl. An uncertainty factor of 10 for subchronic to chronic exposure was applied to the TRV for this species. On the basis of dietary doses modeled using 1993 data, an order of magnitude reduction in the average and 95% UCL would still result in an exceedance of the NOAEL at Stillwater and at the TI Pool. However, all other toxicity quotients would fall below one. Modeled dietary doses for future conditions would fall below the NOAEL and LOAEL if the factor of 10 were removed.

Do modeled TEQ-based dietary doses of PCBs to waterfowl exceed benchmarks for adverse effects on reproduction?

The modeled average and 95% UCL PCB dietary doses to the mallard duck exceed the NOAEL and LOAEL on the basis of 1993 data. The predicted toxicity quotients exceed ten for all comparisons, and exceed 100 for the predicted 95% UCL compared to the LOAEL, and average and 95% UCLs compared to the NOAEL.

These results suggest the potential for adverse reproductive effects as a result of PCB exposure via dietary dose expressed as TEQ to waterfowl. An uncertainty factor of 10 for subchronic to chronic exposure was applied to the TRV for this species. On the basis of dietary doses modeled using 1993 data, an order of magnitude reduction in the average and 95% UCL would still result in exceedances of the NOAEL and the LOAEL at all locations. Modeled dietary doses for future conditions would still exceed both the LOAEL and the NOAEL for all locations.

Do modeled total PCB concentrations in waterfowl eggs exceed benchmarks for adverse effects on reproduction?

Modeled average and 95% UCL total PCB concentrations in eggs of mallard ducks exceed the laboratory-based NOAEL at all locations in the river by at least a factor of 50 on the basis of 1993 data. Average and 95% UCL modeled egg concentrations exceed the laboratory-based LOAEL in the Upper Hudson River, and down to RM122.4 in the Lower Hudson River. The average modeled egg concentrations fall below the laboratory-based LOAEL in the Lower Hudson River, but the 95% UCL exceeds except at RM100 and at RM25.8 on the basis of 1993 data. Modeled average and 95% UCL egg concentrations under future conditions exceed the laboratory-based LOAEL and NOAEL at both the TI Pool and at Stillwater. The average and 95% UCL modeled concentrations at RM154 exceed the LOAEL until 2010, and the NOAEL for the duration of the modeling period.

These results suggest the potential for adverse reproductive effects as a result of predicted total PCB concentrations in the eggs of waterfowl. No uncertainty factors were applied to the selected TRVs for this species. Since the toxicity quotients are so high, a reduction of even 100 times the predicted toxicity quotient would still result in exceedances of modeled egg concentrations as compared to the laboratory-based LOAEL and NOAEL.

Do modeled TEQ-based PCB concentrations in waterfowl eggs exceed benchmarks for adverse effects on reproduction?

Modeled average and 95% UCL PCB concentrations in mallard duck eggs exceed the NOAEL and LOAEL by at least a factor of ten for all locations in the river based on 1993 data. The predicted toxicity quotients in the TI Pool and at Stillwater exceed 1000 for all comparisons based on 1993 data. Modeled results for future conditions show that the toxicity quotients exceed ten for the duration of the modeling period at all three locations. In the TI Pool and at Stillwater, predicted toxicity quotients exceed 100 for the entire modeling period.

These results suggest the potential for adverse reproductive effects as a result of predicted TEQ-based PCB concentrations in the eggs of waterfowl. No uncertainty factors were applied to the selected TRVs for this species. Since the toxicity quotients are so high, a reduction of even 100 times the predicted toxicity quotient would still result in exceedances of modeled egg concentrations as compared to the laboratory-based LOAEL and NOAEL.

Do measured and modeled whole water concentrations exceed appropriate criteria and/or guidelines for the protection of wildlife?

Comparison of measured average and 95% UCL total PCB concentrations exceed the NYSDEC criteria developed for the protection of wildlife. All locations in the river exceed 100 for all comparisons, and some exceed 1000. Modeled water concentrations from HUDTOX also exceed the NYSDEC criteria for the duration of the modeling period.

These results suggest the potential for risk based on the criteria developed for the protection of wildlife. A preliminary analysis of sources of uncertainty in the HUDTOX model suggest a factor of two uncertainty. Even if the model results were to decrease by a factor of two, the NYSDEC wildlife criterion would be exceeded at all three modeling locations for the duration of the modeling period.

What do the available field-based observations suggest about the health of local waterfowl populations?

In general, anecdotal evidence suggests the continued presence of mallard ducks and other similar waterfowl utilizing the Upper Hudson River and Lower Hudson River as breeding grounds and habitat. Nonetheless, the continued presence of these species does not imply a lack of reproductive effects. The results of the risk characterization suggest that mallards are likely to experience adverse reproductive effects as a result of exposure to PCBs.

Summary: Risks to waterfowl species were evaluated using six lines of evidence. Four of these are based on comparing modeled dietary doses and egg concentrations to various toxicity reference values; one is based on comparing measured and modeled water column concentrations to water quality criteria developed for the protection of aquatic life; and one is based on qualitative field observations. Collectively, these lines of evidence indicate that current and future concentrations of PCBs are not of a sufficient magnitude to prevent reproduction of the mallard duck, but that modeled dietary doses and egg concentrations under current and future conditions typically exceed benchmarks. TEQ-based concentrations show greater exceedances than total PCB-based concentrations. Exceedances are expected to occur for the duration of the modeling period. There is a moderate degree of uncertainty in the dietary dose and egg concentration estimates, but even assuming an order of magnitude uncertainty or more, measured and modeled dietary doses and egg concentrations still exceed benchmarks. Measured and modeled water column concentrations exceed criteria developed for the protection of wildlife at all locations under current conditions, and for the duration of the modeling period for future conditions.

7.5 Assessment Endpoint: Protection and Maintenance (i.e., Survival, Growth, and Reproduction) of Hudson River Piscivorous Bird Species

Do modeled total PCB dietary doses to piscivorous birds exceed benchmarks for adverse effects on reproduction?

Belted kingfisher: Modeled average and 95% UCL PCB dietary doses to the belted kingfisher exceed both the LOAEL and NOAEL for all locations in the river under current conditions (1993 data) and for future conditions.

Great blue heron: Modeled average and 95% UCL PCB dietary doses to the great blue heron exceed both the LOAEL and NOAEL for all locations in the river under current conditions (1993 data) and for future conditions.

Bald eagle: Modeled average and 95% UCL PCB dietary doses to the bald eagle exceed both the LOAEL and NOAEL for all locations in the river under current conditions (1993 data) and for future conditions.

These results suggest the potential for adverse reproductive effects to piscivorous bird species as a result of exposure to total PCBs via dietary and water sources. A subchronic-to-chronic uncertainty factor of ten was applied to the laboratory-based LOAEL and NOAEL. For current conditions, measured concentrations were used for sediment, water, and fish. Uncertainty in the predicted fish concentrations (which comprise the bulk of the diet for these species) for future conditions is estimated at roughly a factor of two. Many of the predicted toxicity quotients exceed 100 or even 1000 (which is likely much greater than the range of uncertainty), so even if dietary doses decreased by that amount, modeled doses would typically still exceed TRVs for observed data and well into the future.

Do modeled TEQ-based dietary doses of PCBs to piscivorous birds exceed benchmarks for adverse effects on reproduction?

Belted kingfisher: Modeled average and 95% UCL TEQ-based PCB dietary doses to the belted kingfisher exceed both the LOAEL and NOAEL for all locations in the river under current conditions (1993 data) and for future conditions.

Great blue heron: Modeled average and 95% UCL TEQ-based PCB dietary doses to the great blue heron exceed both the LOAEL and NOAEL for all locations in the river under current conditions (1993 data) and for future conditions.

Bald eagle: Modeled average and 95% UCL TEQ-based PCB dietary doses to the bald eagle exceed both the LOAEL and NOAEL for all locations in the river under current conditions (1993 data) and for future conditions.

These results suggest the potential for adverse reproductive effects to piscivorous bird species as a result of exposure to total PCBs via dietary and water sources. No uncertainty factors were applied in development of the TRVs. For current conditions, measured concentrations were used for sediment, water, and fish. Uncertainty in the predicted fish concentrations (which comprise the bulk of the diet for these species) for future conditions is estimated at roughly a factor of two. Many of the predicted toxicity quotients exceed 100 or even 1000, so even if dietary doses decreased by that amount, modeled doses would typically still exceed TRVs for observed data and well into the future.

Do modeled total PCB concentrations in piscivorous bird eggs exceed benchmarks for adverse effects on reproduction?

Belted kingfisher: Modeled average and 95% UCL total PCB concentrations in the eggs of the belted kingfisher exceed both the LOAEL and NOAEL for all locations in the river under current conditions (1993 data) and for future conditions.

Great blue heron: Modeled average and 95% UCL total PCB concentrations in the eggs of the great blue heron exceed both the LOAEL and NOAEL for all locations in the river under current conditions (1993 data) and for future conditions.

Bald eagle: Modeled average and 95% UCL total PCB concentrations in the eggs of the bald eagle exceed both the LOAEL and NOAEL for all locations in the river under current conditions (1993 data) and for future conditions.

These results suggest the potential for adverse reproductive effects to piscivorous bird species as a result of exceedances of predicted egg concentrations to toxicity benchmarks. No uncertainty factors were applied in the development of the TRVs. For current conditions, measured concentrations were used for sediment, water, and fish. Uncertainty in the predicted fish concentrations (which comprise the bulk of the diet for these species) for future conditions is estimated at roughly a factor of two. Many of the predicted toxicity quotients exceed 100 or even 1000, so even if the egg concentrations decreased by that amount, modeled concentrations would typically still exceed TRVs for observed (1993) data and well into the future.

Do modeled TEQ-based PCB concentrations in piscivorous bird eggs exceed benchmarks for adverse effects on reproduction?

Belted kingfisher: Modeled average and 95% UCL TEQ-based PCB concentrations in the eggs of the belted kingfisher exceed both the LOAEL and NOAEL for all locations in the river under current conditions (1993 data) and for future conditions.

Great blue heron: Modeled average and 95% UCL TEQ-based PCB concentrations in the eggs of the great blue heron exceed both the LOAEL and NOAEL for all locations in the river under current conditions (1993 data) and for future conditions.

Bald eagle: Modeled average and 95% UCL TEQ-based PCB concentrations in the eggs of the bald eagle both the LOAEL and NOAEL for all locations in the river under current conditions (1993 data) and for future conditions.

These results suggest the potential for adverse reproductive effects to piscivorous bird species as a result of predicted TEQ-based PCB concentrations in eggs. No uncertainty factors were applied in the derivation of the TRVs. For current conditions, measured concentrations were used for sediment, water, and fish. Uncertainty in the predicted fish concentrations (which comprise the bulk of the diet for these species) for future conditions is estimated at roughly a factor of two. Many of the predicted toxicity quotients exceed 100 or even 1000, so even if dietary doses decreased by that amount, modeled doses would typically still exceed TRVs for observed data and well into the future.

Do measured and modeled whole water concentrations exceed appropriate criteria and/or guidelines for the protection of wildlife?

Measured average and 95% UCL total PCB concentrations exceed the NYSDEC criteria developed for the protection of wildlife. All locations in the river exceed 100 for all comparisons, and some exceed 1000. Modeled water concentrations from HUDTOX also exceed the NYSDEC criteria for the duration of the modeling period.

These results suggest the potential for risk based on the criteria developed for the protection of wildlife. A preliminary analysis of sources of uncertainty in the HUDTOX model suggest a factor of two uncertainty. Even if the model results were to decrease by a factor of two, the NYSDEC wildlife criterion would be exceeded at all three modeling locations for the duration of the modeling period.

What do the available field-based observations suggest about the health of local piscivorous bird populations?

Measurement Endpoint 6: Available field observations on the presence and relative abundance of piscivorous avian species within the Hudson River for each river segment as an indication of the ability of the species to maintain populations.

Summary: Risks to piscivorous bird species were evaluated using six lines of evidence. Four of these are based on comparing modeled dietary doses and egg concentrations to various toxicity reference values; one is based on comparing measured and modeled water column concentrations to water quality criteria developed for the protection of aquatic life; and one is based on qualitative field observations. Collectively, these lines of evidence indicate that current and future concentrations of PCBs are not of a sufficient magnitude to prevent reproduction of these piscivorous species, which have all been observed along the Hudson River, but that modeled dietary doses and egg concentrations under current and future conditions exceed all benchmarks. Modeled dietary doses and egg concentrations exceed benchmarks developed on the basis of reproductive effects at all locations and for the duration of the modeling period, indicating the potential for risk to piscivorous bird species. There is a moderate degree of uncertainty in the dietary dose and egg concentration estimates, but even assuming an order of magnitude uncertainty or more, measured and modeled dietary doses and egg concentrations still exceed benchmarks. Measured and modeled water column concentrations exceed criteria developed for the protection of wildlife at all locations under current conditions, and for the duration of the modeling period for future conditions.

7.6 Assessment Endpoint: Protection and Maintenance (i.e., Survival, Growth, and Reproduction) of Local Wildlife

Do modeled total PCB dietary doses to local wildlife species exceed benchmarks for adverse effects on reproduction?

Little brown bat: Modeled average and 95% UCL total PCB dietary doses to the little brown bat exceed both the LOAEL and the NOAEL for most locations in the river under current conditions (1993 data) and for future conditions. Modeled average dietary doses do not exceed the LOAEL at RM100, RM88.9 and RM25.8, and the modeled 95% UCL does not exceed at RM88.9 and RM25.8.

Raccoon: Modeled average and 95% UCL total PCB dietary doses to the raccoon exceed both the LOAEL and the NOAEL for all locations in the Upper Hudson River under current conditions (1993 data) and in the TI Pool under future conditions. Under current conditions, the modeled 95% UCL dietary doses exceed the NOAEL in the Lower Hudson River, and the average modeled dietary doses exceed at all locations except RM100, RM88.9 and RM25.8. On a LOAEL-basis, the modeled 95% UCL dietary doses show exceedances at all Lower Hudson River locations except RM143.5, RM122.4, RM88.9, and RM25.8. The average modeled dietary dose does not exceed the LOAEL at any Lower River locations. Under future conditions, modeled average and 95% UCL dietary doses exceed both the LOAEL and the NOAEL for the duration of the modeling period in the TI Pool. At Stillwater, the NOAEL-based comparison shows exceedances for the duration of the modeling period, but the LOAEL-based comparison only exceeds until 2005 for the average and 2007 for the 95% UCL. At RM154, the LOAEL-based comparison shows exceedances only until 1995, and the NOAEL-based comparison shows exceedances until 2013.

Mink: Modeled average and 95% UCL total PCB dietary doses to the mink exceed both the LOAEL and NOAEL for all locations in the Upper Hudson River under current conditions (1993 data). In the Lower Hudson River, modeled average and 95% UCL dietary doses exceed the NOAEL for all locations. Modeled 95% UCL dietary doses exceed the LOAEL in the Lower Hudson River for all locations except RM88.9 andRM25.8. Average modeled dietary doses only exceed the LOAEL at RM143.5 and RM137.2. Under future conditions, modeled average and 95% UCL dietary doses exceed both the LOAEL and the NOAEL in the TI Pool for the duration of the modeling period. At Stillwater, the NOAEL-based comparison shows exceedances for the duration of the modeling period, but the LOAEL-based comparison only exceeds until 2004 for the average and 2006 for the 95% UCL. At RM154, the LOAEL-based comparison shows exceedances only until 1995 on a 95% UCL basis, but the NOAEL-based comparison shows exceedances for the duration of the modeling period.

Otter: Modeled average and 95% UCL total PCB dietary doses to the otter exceed both the LOAEL and NOAEL for all locations in the river under current conditions (1993 data) and for future conditions.

Do modeled TEQ-based PCB dietary doses to local wildlife species exceed benchmarks for adverse effects on reproduction?

Little brown bat: Modeled average and 95% UCL total PCB dietary doses to the little brown bat exceed both the LOAEL and NOAEL for all locations in the river under current conditions (1993 data) and for future conditions.

Raccoon: Modeled average and 95% UCL total PCB dietary doses to the raccoon exceed both the LOAEL and NOAEL for all locations in the river under current conditions (1993 data) and for future conditions.

Mink: Modeled average and 95% UCL total PCB dietary doses to the mink exceed both the LOAEL and NOAEL for all locations in the river under current conditions (1993 data) and for future conditions.

Otter: Modeled average and 95% UCL total PCB dietary doses to the otter exceed both the LOAEL and NOAEL for all locations in the river under current conditions (1993 data) and for future conditions.

Do measured total PCB concentrations in local wildlife species exceed benchmarks for adverse effects on reproduction?

Maximum measured PCB concentrations in the liver of mink and otter exceed both the low-range LOAEL and the upper-range LOAEL at all locations except the North Hudson Valley for mink. The average measured otter concentration also exceeds the low-range LOAEL.

Do measured and modeled whole water concentrations exceed appropriate criteria and/or guidelines for the protection of wildlife?

Measured average and 95% UCL total PCB concentrations exceed the NYSDEC criteria developed for the protection of wildlife. All locations in the river exceed 100 for all comparisons, and some exceed 1000. Modeled water concentrations from HUDTOX also exceed the NYSDEC criteria for the duration of the modeling period.

These results suggest the potential for risk based on the criteria developed for the protection of wildlife. A preliminary analysis of sources of uncertainty in the HUDTOX model suggest a factor of two uncertainty. Even if the model results were to decrease by a factor of two, the NYSDEC wildlife criterion would be exceeded at all three modeling locations for the duration of the modeling period.

What do the available field-based observations suggest about the health of local wildlife populations?

Anecdotal evidence suggests that little brown bat and raccoon are common along most locations in the Upper Hudson River. Further anecdotal evidence suggests that the mink population declined for many years but currently appears to be increasing. There were very few reports of otter in and around the TI Pool but they have been observed in the Albany area and below.

Summary: Risks to the mammalian species were evaluated using four lines of evidence. Two of these are based on comparing modeled dietary doses to various toxicity reference values; one is based on comparing measured and modeled water column concentrations to water quality criteria developed for the protection of aquatic life; and one is based on qualitative field observations. Collectively, these lines of evidence indicate that current and future concentrations of PCBs are not of a sufficient magnitude to prevent reproduction of these mammalian species, which have all been observed along the Hudson River, but that modeled dietary doses under current and future conditions exceed all benchmarks. Modeled dietary doses exceed benchmarks developed on the basis of reproductive effects at all locations and for the duration of the modeling period, indicating the potential for risk to these mammalian species. There is a moderate degree of uncertainty in the dietary dose estimates, but even assuming an order of magnitude uncertainty or more, measured and modeled dietary doses and egg concentrations still exceed benchmarks. Measured and modeled water column concentrations exceed criteria developed for the protection of wildlife at all locations under current conditions, and for the duration of the modeling period for future conditions.

7.7 Assessment Endpoint: Protection of Threatened and Endangered Species

Do modeled total PCB body burdens in local, threatened, or endangered fish species exceed benchmarks for adverse effects on fish reproduction?

The endangered fish species selected for evaluation in the risk assessment is the shortnose sturgeon. As there were no measurements available, and none of the modeling was specifically designed for the sturgeon, the results of the bioaccumulation modeling for other species are used to make inferences about the potential for PCB effects in the shortnose sturgeon. Results for upper trophic level fish, such as the largemouth bass, or omnivorous fish, such as brown bullhead, suggest there is the potential for adverse reproductive effects to the shortnose sturgeon if body burdens in the sturgeon approximate those of the brown bullhead or largemouth bass. As sturgeon can live 30 years or more, it is likely they will accumulate more PCBs than trophic level alone would suggest.

Do modeled TEQ-based PCB body burdens in local threatened or endangered fish species exceed benchmarks for adverse effects on fish reproduction?

The endangered fish species selected for evaluation in the risk assessment is the shortnose sturgeon. As there were no measurements available, and none of the modeling was specifically designed for the sturgeon, the results of the bioaccumulation modeling for other species are used to make inferences about the potential for PCB effects in the shortnose sturgeon. Results for upper trophic level fish, such as the largemouth bass, or omnivorous fish, such as brown bullhead, suggest there is the potential for adverse reproductive effects to the shortnose sturgeon if body burdens in the sturgeon approximate those of the brown bullhead or largemouth bass. As sturgeon can live 30 years or more, it is likely they will accumulate more PCBs than trophic level alone would suggest.

Do modeled PCB dietary doses to local threatened or endangered avian species exceed benchmarks for adverse effects on reproduction?

The threatened avian species selected for evaluation in this risk assessment is the bald eagle. Results for this species were presented in the avian section. All modeled dietary doses to the bald eagle exceed the field-based NOAEL derived in Chapter 4.

Do modeled PCB concentrations in the eggs of local threatened or endangered avian species exceed benchmarks for adverse effects on reproduction?

The threatened avian species selected for evaluation in this risk assessment is the bald eagle. Results for this species were presented in the avian section. All modeled PCB concentrations in the eggs of the bald eagle exceed the field-based NOAEL derived in Chapter 4.

Do measured and modeled whole water concentrations exceed appropriate criteria and/or guidelines for the protection of wildlife?

Measured average and 95% UCL total PCB concentrations exceed the NYSDEC criteria developed for the protection of wildlife. All locations in the river exceed 100 for all comparisons, and some exceed 1000. Modeled water concentrations from HUDTOX also exceed the NYSDEC criteria for the duration of the modeling period.

These results suggest the potential for risk based on the criteria developed for the protection of wildlife. A preliminary analysis of sources of uncertainty in the HUDTOX model suggest a factor of two uncertainty. Even if the model results were to decrease by a factor of two, the NYSDEC wildlife criterion would be exceeded at all three modeling locations for the duration of the modeling period.

Do measured and modeled sediment PCB concentrations exceed appropriate guidelines?

Measured concentrations also exceed the NOAA TEC and MEC consensus values, both on an average and 95% UCL basis, at all locations except RM58.7 for the average as compared to the MEC. Measured concentrations in the Upper Hudson River exceed the EEC, and several locations in the Lower Hudson River exceed the EEC on a 95% UCL basis. For future conditions, modeled average and 95% UCL sediment concentrations predicted by HUDTOX exceed the TEC, MEC, and EEC for the TI Pool and Stillwater for the duration of the modeling period. Modeled average and 95% UCL sediment concentrations at RM154 exceed the TEC and MEC for the duration of the modeling period, and exceed the EEC until 2001.

Measured sediment concentrations based on the 1993 USEPA dataset exceed the NYSDEC chronic criteria for the protection of benthic aquatic life on both an average and 95% UCL basis for all locations in the river except RM100 (average only). Average and 95% UCL measured sediment concentrations at all locations exceed the NYSDEC wildlife criterion. Expressed as a ratio, these values are above ten for all locations and above 1000 at Stillwater for the 95% UCL comparison. For future conditions, modeled average and 95% UCL sediment concentrations predicted by HUDTOX exceed the benthic chronic criterion for all locations for the duration of the modeling period.

Measured sediment concentrations exceed the SELs from Ontario, and all Washington State criteria. The same is true for future conditions.

Many of the ratios of measured and modeled sediment concentrations to appropriate guidelines exceed 10 or even 100. Thus, even in the unlikely event that predicted sediment concentrations were to decrease by an order of magnitude or more, comparisons to sediment guidelines would show exceedances.

Summary: Risks to threatened and endangered species were evaluated using four lines of evidence. Two of these are based on comparing modeled body burdens in fish and dietary doses and egg concentrations in piscivorous birds to various toxicity reference values; one is based on comparing measured and modeled water column concentrations to water quality criteria developed for the protection of aquatic life; and one is based on comparisons of measured and modeled sediment concentrations to guidelines developed specifically for the protection of aquatic life. Collectively, these lines of evidence indicate that current and future concentrations of PCBs are of a sufficient magnitude to adversely affect the reproductive capability of these fragile populations. Modeled fish body burdens and avian dietary doses and egg concentrations exceed benchmarks developed on the basis of reproductive effects at all locations and for the duration of the modeling period, indicating the potential for risk to these species. There is a moderate degree of uncertainty in these estimates, but exceedances would still persist for the bald eagle even if the predicted dietary doses and/or egg concentrations decreased by an order of magnitude, and in some cases several orders of magnitude. Measured and modeled water column concentrations exceed criteria developed for the protection of wildlife at all locations under current conditions, and for the duration of the modeling period for future conditions. Measured and modeled sediment concentrations exceed all but the least stringent guidelines at all locations under current conditions and at all locations for the duration of the modeling period under future conditions.

7.8 Assessment Endpoint: Protection of Significant Habitats

Do measured and modeled whole water concentrations exceed appropriate criteria and/or guidelines for the protection of wildlife?

Measured average and 95% UCL total PCB concentrations exceed the NYSDEC criteria developed for the protection of wildlife. All locations in the river exceed 100 for all comparisons, and some exceed 1000. Modeled water concentrations from HUDTOX also exceed the NYSDEC criteria for the duration of the modeling period.

These results suggest the potential for risk based on the criteria developed for the protection of wildlife. A preliminary analysis of sources of uncertainty in the HUDTOX model suggest a factor of two uncertainty. Even if the model results were to decrease by a factor of two, the NYSDEC wildlife criterion would be exceeded at all three modeling locations for the duration of the modeling period.

Do measured and modeled sediment PCB concentrations exceed appropriate guidelines?

Measured sediment concentrations exceed the NOAA TEC and MEC consensus values, both on an average and 95% UCL basis, at all locations except RM58.7 for the average as compared to the MEC. Measured concentrations in the Upper Hudson River exceed the EEC, and several locations in the Lower Hudson River exceed the EEC on a 95% UCL basis. For future conditions, modeled average and 95% UCL sediment concentrations predicted by HUDTOX exceed the TEC, MEC, and EEC for the TI Pool and Stillwater for the duration of the modeling period. Modeled average and 95% UCL sediment concentrations at RM154 exceed the TEC and MEC for the duration of the modeling period, and exceed the EEC until 2001.

Measured sediment concentrations based on the 1993 USEPA dataset exceed the NYSDEC chronic criteria for the protection of benthic aquatic life on both an average and 95% UCL basis for all locations in the river except RM100 (average only). Average and 95% UCL measured sediment concentrations at all locations exceed the NYSDEC wildlife criterion. Expressed as a ratio, these values are above ten for all locations and above 1000 at Stillwater for the 95% UCL comparison. For future conditions, modeled average and 95% UCL sediment concentrations predicted by HUDTOX exceed the benthic chronic criterion for all locations for the duration of the modeling period.

Measured sediment concentrations exceed the SELs from Ontario, and all Washington State criteria. The same is true for future conditions.

Many of the ratios of measured and modeled sediment concentrations to appropriate guidelines exceed 10 or even 100. Thus, even in the unlikely event that predicted sediment concentrations were to decrease by an order of magnitude or more, comparisons to sediment guidelines would show exceedances.

Summary: Risks to significant habitats were evaluated using two lines of evidence. One is based on comparing measured and modeled water column concentrations to water quality criteria developed for the protection of aquatic life; and one is based on comparisons of measured and modeled sediment concentrations to guidelines developed specifically for the protection of aquatic life. Collectively, these lines of evidence indicate that current and future concentrations of PCBs are of a sufficient magnitude to adversely affect the ability of particular habitats in the Hudson River to support sustainable, healthy aquatic biota populations. Measured and modeled water column concentrations exceed criteria developed for the protection of wildlife at all locations under current conditions, and for the duration of the modeling period for future conditions. Measured and modeled sediment concentrations exceed all but the least stringent guidelines at all locations under current conditions and at all locations for the duration of the modeling period under future conditions.

REFERENCES

Adams, S.M., W.D. Crumby, M.S. Greeley, Jr., M.G. Ryon, and E.M. Schilling. 1992. Relationships between physiological and fish population responses in a contaminated stream. *Environmental Toxciology and Chemistry.* 11:1549-1557.

Adams, S.M., K.L. Shepard, M.S. Greeley Jr., B.D. Jimenez, M.G. Ryon, L.R. Shugart, and J.F. McCarthy. 1989. The use of bioindicators for assessing the effects of pollutant stress on fish. Marine Environmental Research. 28:459-464.

Adams, S.M., L.R. Shugart, G.R. Southworth and D.E. Hinton. 1990. Application of bioindicators in assessing the health of fish populations experiencing contaminant stress. In: J.F. McCarthy and L.R. Shugart, eds., Biomarkers of Environmental Contamination. Lewis Publishers, Boca Raton, FL. Pp. 333-353.

Albers, P.H., S. Sileo, and B.M Hulhern. 1986. Effects of environmental contaminants on snapping turtles of a tidal wetland. *Arch. Environ. Contam. Toxicol.* 15, 39-49.

Alexander, G.R. 1977. Food of vertebrate predators on trout waters in north central lower Michigan. Michigan Academician. 10: 181-195. In. USEPA, 1993. Wildlife Exposure Factors Handbook. Volume II of II. Office of Research and Development, Washington, D.C. EPA/600/R-93/187b.

Allen, J.R., Barsotti, D.A., Carstens, L.A. 1980. Residual effects of polychlorinated biphenyls on adult nonhuman primates and their offspring. *J. Toxicol. Environ. Health.* 6:55-66.

Allen, J.R., D.A. Barsotti, L.K. Lambrecht, and J.P. Van Miller. 1979. Reproductive effects of halogenated aromatic hydrocarbons on nonhuman primates. *Ann. NY Acad. Sci.* 320:419-425.

Allen, J.R., and D.A. Barsotti. 1976. The effects of transplacental and mammary movement of PCBs on infant rhesus monkeys. *Toxicology.* 6:331-340.

Andrle, R.F. and J.R. Carroll, eds. 1988. The Atlas of Breeding Birds in New York State. Cornell University Press, Ithaca, NY 551 pp.

Ankley, G.T., K. Lodge, D.J. Call, M.D. Balcer, L.T. Brooke, P.M. Cook, and R.G. Kreis, Jr. 1992. Integrated assessment of contaminated sediments in the Lower Fox River and Green Bay, Wisconsin. *Ecotoxicol. Environ. Safety* 23:46-63.

Ankley, G.T., R.D. Johnson, G.Toth, L.C. Folmar, N.E. Detenbeck, and S.P. Bradbury. 1997. Development of a research strategy for assessing the ecological risk of endocrine disruptors. Rev. Toxicol. Ser. B: *Environ. Toxicol.*

Ankley, G.T., D.E. Tillitt, J.P. Giesy, P.D. Jones, and D.A. Verbrugge. 1991. Bioassay-derived 2,3,7,8-tetrachlorodibenzo-p-dioxin equivalents in PCB-containing extracts from the flesh and eggs of Lake Michigan chinook salmon (*Oncorhynchus tshawytscha*) and possible implications for reproduction. *Canadian Journal of Aquatic Sciences*, 48:1685-1690.

Anthony, E.L.P. and T.H. Kunz. 1977. Feeding strategies of the little brown bat, *Myotis lucifugus*, in southern New Hampshire. *Ecology* 58: 775-786.

Arnold, D.L., J. Mes, D. Bryce, et al. 1990. A pilot study on the effects of Aroclor 1254 ingestion by rhesus and cynomolgus monkeys as a model for human ingestion of PCBs. *Food. Chem. Toxicol.* 28:847-857.

Agency for Toxic Substances and Disease Registry (ATSDR). 1996. PCBs In: Toxicological Profiles for Group I Chemicals. Agency for Toxic Substances and Disease Registry, US Public Health Service.

Aulerich, R.J., S.J. Bursian, W.J. Breslin, B.A. Olson, and R.K. Ringer. 1985. Toxicological manifestations of 2,4,5,2',4',5'-,2,3,6,2',3',6'-,and 3,4,5,3',4',5'- hexachlorobiphenyl and Aroclor 1254 in Mink. *J.Toxicol. Environ. Health.* 15:63-79.

Aulerich, R.J., S.J. Bursian, and A.C. Napolitano. 1988. Biological effects of epidermal growth factor and 2,3,7,8-tetrachlorodibenzo-p-dioxin on developmental parameters of neonatal mink. *Arch. Environ. Contam. Toxicol.* 17:27-31.

Aulerich, R.J. and R.K. Ringer. 1977. Current Status of PCB Toxicity, Including Reproduction in Mink. *Arch. Environm. Contam. Toxicol.* 6:279-292.

Aulerich, R.J., R.K. Ringer, and J. Safronoff. 1986. Assessment of primary vs. secondary toxicity of aroclor 1254 to mink. *Arch. Environ. Contam. Toxicol.* 15:393-399.

Bain, M.B. 1999. Cornell Univeristy. Personal communication by telephone.

Barron, M.G. 1990. Bioconcentration - Will water-borne organic chemicals accumulate in aquatic animals? *Environ. Sci. Technol.*, Vol. 24, 11:1612-1618.

Barsotti, D.A., R.J. Marlar, and J.R. Allen. 1976. Reproductive dysfunction in rhesus monkeys exposed to low levels of polychlorinated biphenyls (Aroclor 1248). *Food Cosmet. Toxicol.* 14:99-103.

Barsotti, D.A., and J.P. Van Miller. 1984. Accumulation of a commercial polychlorinated biphenyl mixture (Aroclor 1016) in adult rhesus monkeys and their nursing infants. *Toxicology.* 30:31-40.

Batcheller, G., 1999. NYSDEC Division of Fish, Wildlife and Marine Resources, Furbearer Unit. Delamar, New York. Personal communication by telephone.

Beatty, P.W., M. A. Holscher, and R.A. Neal. 1976. Toxicity of 2,3,7,8-tetracholordobenzo-p-dioxin in larval and adult forms of Rana catesbeiana. *Bulletin of Environmental Contamination and Toxicology*, Vol. 16, No. 5, pp. 578-581.

Becker, P.H. S. Schuhmann, and C. Koepff. 1993. Hatching failure in Common *Terns (Sterna hirundo*) in relation to environmental chemicals. *Environmental Pollution*. Vol. 79:207-213.
Becker, G.M., W.P. McNulty, and M. Bell. 1979. Polychlorinated biphenyls-induced morphologic changes in the gastric mucosa of the rhesus monkey. *Invest.* 40:373.

Beebe, C.A. and A.R. Savidge. 1988. Historical perspective on fish species composition and distribution in the Hudson River Estuary. *Amer. Fish. Soc. Monogr.* 4:25-36.

Belwood, J.J. and M.B. Fenton. 1976. Variation in the diet of *Myotis lucifugus*. (Chiroptera: Vespertilionidae). *Can. J. Zool.* 54: 1674-1678.

Bengtsson, B.E. 1980. Long-term effects of PCB (Clophen A50) on growth, reproduction and swimming performance in the minnow, *Phoxinus phoxinus*. *Water Research*, Vol. 1, pp. 681-687.

Bergeron, J.M., D. Crews, and J.A. McLachlan. 1994. PCBs as environmental estrogens: Turtle sex determination as a biomarker of environmental contamination. *Environmental Health Perspectives*, Vol. 102, No. 9, pp. 780-781.

Beyer, W.N., E. Conner, and S. Gerould. 1994. Estimates of soil ingestion by wildlife. *J. Wildl. Manage.* 58: 375-382.

Bierman, V.J., Jr. 1990. Equilibrium Partitioning and Biomagnification of Organic Chemicals in Benthic Animals. *Environmental Science and Technology*, Vol. 24, No. 9, pp. 1407-1412

Bishop, C.A., Brooks, R.J., J.H. Carey, P. Ng, R.J. Nostrom, and D.R.S. Lean. 1991. The case for a cause-effect linkage between environmental contamination and development in eggs of the common snapping turtle (*Chelydra S. Serpentina*) from Ontario, Canada. *Journal of Toxicology and Environmental Health*, 33: 521-547.

Bishop, C.A., G.P. Brown, R.J. Brooks, D.R.S. Lean, and J.H. Carey. 1994. Organochlorine contaminant concentrations in eggs and their relationship to body size, and clutch characteristics of the female common snapping turtle *(Chelydra serpentina serpentina*) in Lake Ontario, Canada. *Arch. Environ. Contam. Toxicol.* 27, 82-87.

Black, D.E., Gutzahr-Gobell, R., Pruell, R.J., Bergen, B., and McElroy, A.E. 1998a. Effects of a mixture of non-ortho- and mono-ortho-polychlorinated biphenyls on reproduction *in fundulus heteroclitus* (linnaeus). *Environmental Toxicology and Chemistry*, Vol. 17, No. 7, pp. 1396-1404.

Black, D.E., R. Gutjahr-Gobell, R.J. Pruell, B. Bergen, L. Mills, and A.E. McElroy. 1998b. Reproduction and polychlorinated biphenyls in *Fundulus heteroclitus* (Linnaeus) from New

Bedford Harbor, Massachusetts, USA. *Environmental Toxicology and Chemistry*, 17(7):1405-1414.

Black, D.E., D.K. Phelps, and R.L. Lapan. 1988. The effect of inherited contamination on egg and larval winter flounder, *Pseudopleuronectes americanus. Marine Environmental Research.* 25:45-62.

Bleavins, M R. and R. J. Aulerich. 1981. Feed consumption and food passage time in mink (*Mustela vison*) and European ferrets (*Mustela putorius furo). Lab. Anim. Sci.* 31: 268-269.

Bleavins, M.R., R.J. Aulerich, and R.K. Ringer. 1980. Polychlorinated biphenyls (Aroclors 1016 and 1242): Effects on survival and reproduction in mink and ferrets. *Archives of Environmental Contamination & Toxicology.* 9:627-635.

Bopp, J. 1999. Collections Manager, New York State Museum. Albany, New York. Personal communication by telephone with J. Schaffer, TAMS Consultants, Inc. January 22, 1999.

Borgmann, U., W.P. Norwood, and K.M. Ralph. 1990. Chronic toxicity and bioaccumulation of 2,5,2',5'- and 3,4,3',4'-tetrachlorobiphenyl and Aroclor 1242 in the amphipod Hyalella azteca. *Arch. Environ. Contam. Toxicol.* 19:558-564.

Bowerman, W.W., J.P. Giesy, D.A. Best, and V.J. Kramer. 1995. A review of factors affecting productivity of bald eagles in the Great Lakes region: implications for recovery. *Environmental Health Perspectives.* 103(4):51-59.

Bowman, R.E., S.L. Schantz, N.C.A. Weerasinghe, M.L. Gross, and D.A. Barsotti. 1989. Chronic dietary intake of 2,3,7,8-tetrachlorodibenzo-p-dioxin (TCDD) at 5 and 25 parts per trillion in the monkey: TCDD kinetics and dose-effect estimate of reproductive toxicity. *Chemosphere.* 18:243-252.

Brezner, E., J. Terkel, A.S. Perry. 1984. The effect of Aroclor 1254 (PCB) on the physiology of reproduction in the female rat – I. *Comp. Biochem. Physiol.* 77:65-70.

Britton, W.M. and T.M. Huston. 1973. Influence of polychlorinated biphenyls in the laying hen. *Poultry Science* 52:1620-1624.

Brooks, R.P., and W.J. Davis. 1987. Habitat selection by breeding Belted Kingfishers (*Ceryle alcyon*). *Amer. Midl. Nat.* 117:6370.

Brown, M.P., M.B. Werner, C.R. Carusone and M. Klein. 1988. Distribution of PCBs in the Thompson Island Pool of the Hudson River. Final Report of the Hudson River PCB Reclamation Demonstration Project Sediment Survey. NYSDEC, Albany, N.Y.

Broyles, R.H., and M.I. Noveck. 1979. Uptake and distribution of 2,4,5,2',4',5'-hexachlorobiphenyl in fry of lake trout and chinook salmon and its effects on viability. *Toxicology and Applied Pharmacology.* 50: 299-308.

Bryan, A.M., P.G. Olafsson, and W.B. Stone. 1987a. Disposition of low and high environmental concentrations of PCBs in snapping turtle tissues. *Bull. Environ. Contam. Toxicol.* 38: 1000-1005.

Bryan, A.M., P.G. Olafsson, and W.B. Stone. 1987b. Disposition of toxic PCB congeners in snapping turtle eggs: Expressed as toxic equivaalents of TCDD. *Bull. Environ. Contam. Toxicol.* 39: 791-796.

Buchler, E.R. 1976. Prey selection by *Myotis lucifugus* (Chiroptera: Vespertilionidae). *Am. Natur.* 110(974): 619-628.

Bull, J. L. 1998. Bull's birds of New York State. Comstock Publishing Associates, Cornell University Press. Ithaca, New York. 622 pp.

Butcher, J.B., T.D. Gauthier and E.A. Garvey. 1997. Use of historical PCB Aroclor measurements: Hudson River fish data. *Environ. Toxicol. Chem.* 16 (8): 1618-1623.

California Environmental Protection Agency, Department of Toxic Substances Control, Human and Ecological Risk Division. July 4, 1996. Guidance for Ecological Risk Assessment at Hazardous Waste Sites and Permitted Facilities. Part A. Overview.

Caraco, N.F., J.J. Cole, P.A. Raymond, D.L. Strayer, M.L. Pace, S.E.G. Findlay, and D.T. Fischer. 1997. Zebra mussel invasion in a large turbid river: phytoplankton response to increased grazing. *Ecology* 78(2): 588-602.

Carlson, R.W., and R.T. Duby. 1973. Embryotoxic effects of three PCBs in the chicken. *Bulletin of Environmental Contamination & Toxicology.* 9:5, 261-266.

Casillas, E., D. Misitano, L.L. Johnson, L.D. Rhodes, T.K. Collier, J.E. Stein, B.B. McCain and U. Varanasi. 1991. Inducibility of spawning and reproductive success of female English sole (*Parophrys vetulus*) from urban and nonurban areas of Puget Sound, Washington. *Marine Environmental Research.* 31:99-122.

Cecil, H.C., J. Bitman, R.J. Lillie, and G.F. Fries. 1974. Embryotoxic and teratogenic effects in unhatched fertile eggs from hens fed polychlorinated biphenyls (PCBs). *Bulletin of Environmental Contamination & Toxicology.* 11:6, pp 489-495.

Chang, E.S., and E.R.L. Stokstad. 1975. Effects of chlorinated hudrocarbons on shell gland carbonic anhydrase and eggshell thickness in Japanese quail. *Poult. Sci.* 54:3-10.

Clark, K.E., L.J. Niles, and W. Stansley. 1998. Environmental contaminants associated with reproductive failure in bald eagles (*Haliaeetus leucocephalus*) eggs in New Jersey. *Bull. Environ. Contamin. Toxicol.* 61:247-254.

Clark, D.R., Jr. 1978. Uptake of dietary PCB by pregnant big brown bats (*Eptesicus fuscus*) and their fetuses. *Bull. Environ. Contam. Toxicol.* 19:707-714.

Clark, D.R., Jr., and A. Krynitsky. 1978. Organochlorine residues and reproduction in the little brown bat, Laurel, Maryland – June 1976. *Pestic. Monit.* J. 12:113-116.

Clark, D.R., Jr., and T.G. Lamont. 1976. Organochlorine residues and reproduction in the big brown bat. *J. Wildl. Manage.* 40:249-254.

Clayton, J.R., S.P. Pavlou, and N.F. Brietner. 1977. Polychlorinated biphenyls in coastal marine zooplankton: bioaccumulation by equilibrium partitioning. *Env. Sci. Tech.* 19:57-62.

Collins, W.T., C.C. Capen. 1980. Fine structural lesions and hormonal alterations in thyroid glands of prinatal rats exposed in utero and by milk to polychlorinated biphenyls. *Am. J. Pathol.* 99:125-142.

Crews, D., J.M. Bergeron, and J.A. McLachlan. 1995. The role of estrogen in turtle sex determination and the effect of PCBs. Environmental Health Perspectives. Vol. 103, Supplement 7, pp. 73-77.

Custer, T.W., and G.H. Heinz. 1980. Reproductive success and nest attentiveness of mallard ducks fed Aroclor 1254. *Environmental Pollution* (Series A). 21:313-318.

Custer, C.M., T.W. Custer, P.D. Allen, K.L. Strombord, and M.J. Melancons. 1998. Reproduction and environmental contamination in tree swallows nesting in the Fox River drainage and Green Bay, Wisconsin, USA. *Environmental Toxicology and Chemistry.* 17(9):1786-1798.

Dahlgren, R.B., R.L. Linder, and C.W. Carlson. 1972. Polychlorinated biphenyls: Their effects on penned pheasants. *Environmental Health Perspectives.* 1:89-101.

Davis, W.J. 1982. Territory size in *Megaceryle alcyon* along a stream habitat. *Auk* 99:353-362.

De Boer, J., F. Van der Valk, M. Kerkhoff, and P. Hagel . 1994. 8-Year study on the elimination of PCBs and other organochlorine compounds from eel (*Anguilla anguilla*) under natural conditions. *Environ. Sci. and Tech.* 28 (13): 2242-2248.

Defoe,D.L., G.D.Veith, and R.W.Carlson. 1978. Effects of Aroclor 1248 and 1260 on the Fathead Minnow (*Pimephales promelas*). *J. Fish. Res. Board Can.* 35(7):997-1002.

Doutt, J.K., C.A. Heppenstall and J.E. Guilday. 1977. Mammals of Pennsylvania. The Pennsylvania Game Commission. Harrisburg, PA. 285 p.

Dowd, P.F., G.U. Mayfield, D.P. Coulon, J.B. Graves, and J.D. Newsom. 1985. Organochlorine residues in animals from three Louisiana watersheds in 1978 and 1979. *Bull. Environ. Contam. Toxicol.* 34: 832-841.

Dunning, J.B. Jr. 1993. CRC Handbook of Avian Body Masses. CRC Press. Ann Arbor Michigan. 371 pp.

Eckert, A. W. and K.E. Karalus. 1983. The Wading Birds of North America: North of Mexico. Weathervane Books, New York, New York. 226 pp.

Eisler, R. 1986. Polychlorinated Biphenyl Hazards to Fish, Wildlife, and Invertebrates: A Synoptic Review. Fish and Wildlife Service, U.S. Dept. of the Interior. Biological Report 85(1.7). 72 pp.

Elliott, J.E., R.W. Butler, R.J. Norstrom, and P.E. Whitehead. 1989. Environmental contaminants and reproductive success of Great Blue Herons (*Ardea herodias*) in British Columbia, 1986-87. *Environmental Pollution*. Vol. 59:91-114.

Elliott, J.E., R.J. Norstrom, A. Lorenzen, L.E. Hart, H. Philibert, S.W. Kennedy, J.J. Stegeman, G.D. Bellward, and K.M. Cheng. 1996. Biological effects of polychlorinated bibenzo-p-dioxins, dibenzofurans, and biphenyls in bald eagle (*Haliaeetus leucocephalus*) chicks. *Environmental Toxicology and Chemistry*. Vol. 1(5):782-793.

Elsie, R. and A.A. Belisle. 1996. Planar PCB Hazards to Fish, Wildlife, and Invertebrates: A Synoptic Review. Contaminants Hazard Review, Report 31.U.S. Department of the Interior, National Biological Service. Washington, DC.

Elonen, G.E., R.L. Spehar, G.W. Holcombe, R.D. Johnson, J.D. Fernandex, R.J. Erickson, J.E. Tietge, and P.M. Cook. 1998. Comparative toxicity of 2,3,7,8-tetrachlorodibenzo-p-dioxin to seven freshwater fish species during early life-stage development. *Environmental Toxicology and Chemistry*. 17:3, pp 472-483.

Environment Canada. 1991 (cited in Hebert et al., 1993). Toxic chemicals in the Great Lakes and associated effects. Vols. 1-2. Ministry of Supply and Services, Ottawa, Ontario, Canada

Ernst, W. 1984. Pesticides and technical organic chemicals. In Marine Ecology, Vol. V, Part 4 (Ed: O. Kinne). John Wiley, New York. pp. 1617-1709.

Exponent. 1998a. Data documentation and interpretation report: Submerged aquatic vegetation and fish community analysis. Prepared for General Electric Company.

Exponent. 1998b. Data report: Macroinvertebrate communities and diets of selected fish species in the Upper Hudson River, Volumes I and II. Prepared for General Electric Company.

Fanelli, R., M.G. Castelli, G.P. Martelli, A. Noseda, and S. Garattini. 1980. Presence of 2,3,7,8-tetrachlorodibenzo-p-dioxin in wildlife living near Seveso, Italy: A preliminary study. *Bull Environm. Contam. Toxicol.* 24, 460-462.

Fenton, M.B. and G.P. Barclay. 1980. Ecolocation and feeding behavior in four species of *Myotis* (Chiroptera). *Can. J. Zool.* 57: 1271-1277.

Field, L.J. and R. Sloan. 1996. PCBs in Hudson River Fish: Comparisons of Congener Patterns Over a Geographic Gradient. In Abstract Book of SETAC 17th Annunal Meeting *Partnerships For The Environment: Science, Education, and Policy*, Washington, DC, 17 - 21 November 1996.

Fisher, S.W., S.W. Chordas III, P.F. Landrum. 1999. Lethal and sublethal body residues for PCB intoxication in the oligochaete, *Lumbriculus variegatus*. *Aquatic Toxicology*. 45:115-126.

Foley, R.E., S.J. Jackling, R.J. Sloan and M.K. Brown. 1988. Organochlorine and mercury residues in wild mink and otter: comparison with fish. *Environ. Toxicol. Chem.* 7: 363-374.

Fossi, C., C. Leonzio, S. Focardi, and D.B. Peakall. 1990. Avian mixed function oxidase induction as a monitoring device: the influence of normal physiological functions. In: Biomarkers of Environmental Contamination. Pp. 877-121. J.F. McCarthy and L.R. Shugart, Editors. Lewis Publishers. Boca Raton, FL.

Freeman, H.C., and D.R. Idler. 1974. The effect of polychlorinated biphenyl on steroidogenesis and reproduction in the brook trout (Salvelinus fontinalis). Canadian Journal of Biochemistry. 75:666-670.

Garcia, E.F., R.J. McPherson, T.H. Martin, R.A. Poth, and M.S. Greeley, Jr. 1997. Liver cell estrogen receptor binding in prespawning female largemouth bass, *Micropterus salmoides*, environmentally exposed to polychlorinated biphenyls. *Archives of Environmental Contamination and Toxicology*. 32:309-315.

Garthoff, L.H., F.E. Cerra, E.M. Marks. 1981. Blood chemistry alteration in rats after single and multiple gavage administration of polychlorinated biphenyls. *Toxicol. Appl. Pharmacol.* 60:33-44.

Geer, K.R. 1955. Yearly food habits of the river otter in the Thompson Lakes region, Northwestern Montana, as indicated by scat analysis. *Am. Mid. Nat.* 54: 299-313.

Giesy, J.P., W.W. Bowerman, M.A. Mora, D.A. Verbrugge, R.A. Othoudt, J.L. Newsted, C.L. Summer, R.J. Aulerich, S.J. Bursian, J.P. Ludwig, G.A. Dawson, T.J. Kubiak, D.A. Best, and D.E. Tillitt. 1995. Contaminants in fishes from Great Lakes-influenced sections and above dams of three Michigan rivers: III. Implications for health of bald eagles. *Archives of Environmental Contamination and Toxicology*. 29:309-321.

Giesy, J.P., J. Newsted, and D.L. Garling. 1986. Relationships between chlorinated hydrocarbon concentrations and rearing mortality of chinook salmon (*Oncorhynchus tshawytscha*) eggs from Lake Michigan. *Journal of Great Lakes Research*. 12(1):82-98.

Gilbert, R.O. 1987. Statistical Methods for Environmental Pollution Monitoring. John Wiley and Sons, NY

Gilbert, F.F. and E.G. Nancekivell. 1982. Food habits of mink (*Mustela vison*) and otter (*Lutra canadensis*) in northeastern Alberta. *Can. J. Zool.* 60: 1282-1288.

Gladden, J., F.R. Cantelmo, J.M. Croom, and R. Shapot. 1988. Evaluation of the Hudson River ecosystem in relation to the dynamics of fish populations. *Amer. Fish. Soc. Mono.* 4:37-52.

Gobas, F. A.P.C., E.J. McNeil, L Lovett-Doust and G.D. Haffner. 1991. Bioconcentration of chlorinated aromatic hydrocarbons in aquatic macrophytes. *Environ. Sci. Technol.* 25(5):924-929.

Gobas, F.A.P.C. 1993. A model for predicting the bioaccumulation of hydrophobic organic chemicals in aquatic food-webs: application to Lake Ontario. *Ecol. Modelling* 69:1-17.

Gould, J.C., K.R. Cooper, and C.G. Scanes. 1997. Effects of polychlorinated biphenyl mixtures and three specific congeners on growth and circulating growth-related hormones. *General and Comparative Endocrinology*. 106:221-230.

Greig, J.B., G. Jones, W.H. Butler, and J.M. Barnes. 1973. Toxic effects of 2,3,7,8-tetrachloro-dibenzo-p-dioxin. *Fd. Cosmet. Toxcicol.* 11:585-595.

Gulney, P.D., P.M. Cook, J.M. Casselman, J.D. Fitzsimmons, H.A. Simonin, E.W. Zabel, and R.E. Peterson. 1996. Assessment of 2,3,7,8-tetrachlorodibenzo-p-dioxin induced sac fry mortality in lake trout (*Salvelinus namaycush*) from different regions of the Great Lakes. *Canadian Journal of Aquatic Sciences*. 53:2080-2092.

Hansen, P.D. 1985. Chlorinated hydrocarbons and hatching success in Baltic Herring spring spawners. *Marine Environmental Research*. 15:59-76.

Hansen, D.J., S.C. Schimmel, and E. Matthews. 1974. Avoidance of Aroclor 1254 by shrimp and fishes, *Bull. of Environ. Contam. and Toxicol.* 12(2):253-293.

Hamilton, W. J. 1936. Food habits of the mink in New York. *Journ. Wild. Manag.* 17(2): 169.

Hamilton, W.J. 1940. The summer food of minks and raccoons on the Montezuma Marsh, New York. *Journ. Wild. Manag.* 4(1): 80-84.

Hamilton, W.J. 1943. The food and breeding habits of the raccoon. *Ohio Journ. Sci.* 36: 131-140.

Hamilton, W.J. 1959. Foods of mink in New York. N.Y. Fish and Game J. 6: 77-85. In USEPA, 1993a. Wildlife Exposure Factors Handbook. Volume II of II. Office of Research and Development, Washington, D.C. EPA/600/R-93/187b.

Hamilton, W.J. 1961. Late fall, winter and early spring foods of 141 otters from New York. *N.Y: Fish and Game J.* 8:242-252.

Hansen, L.G., D.W. Wilson, and C.S. Byerly. 1976. Effects on growing swine and sheep of two polychlorinated biphenyls. *Am. J. Vet. Res.* 37:1021-1024.

Hansen, L.G., C.S. Byerly, R.L. Metcalf, R.F. Bevill. 1975. Effect of a polychlorinated biphenyl mixture on swine reproduction and tissue residues. *Am. J. Vet. Res.* 36(1):23-26.

Hansen, D.J., Parrish, P.R., and Forester, J. 1974. Aroclor 1016: toxicity to and uptake by estuarine animals. *Environmental Research*, Vol. 7, pp 363-373.

Hansen, D.J., Parrish, P.R., Lowe, J.I., Wilson, A.J., Jr., and P.D. Wilson. 1971. Chronic toxicity, uptake, and retention of aroclor 1254 in two estuarine fishes. *Bulletin of Environmental Contamination & Toxicology*, Vol. 6, No. 2. pp. 113-119.

Harris, C.J. 1968. Otters: a study of the recent Lutrinae. London, U.K.: Weidenfield and Nicolson. In USEPA, 1993. Wildlife Exposure Factors Handbook. Volume I of II. Office of Research and Development, Washington, D.C. EPA/600/R-93/187a.

Harris, M., T. Zacharewski, and S. Safe. 1993. Comparative potencies of Aroclors 1232, 1242, 1248, 1254, and 1260 in male wistar rats – assessment of the toxic equivalency factor (TEF) approach for polychlorinated biphenyls (PCBs). *Fundamental and Applied Toxicology.* 20:456-463.

Hart, L.E., K.M. Cheng, P.E. Whitehead, R.M. Shah, R.J. Lewis, S.R. Ruschkowski, R.W. Blair, D.C. Bennett, S.M. Bandiera, R.J.Norstrom, and G.D. Bellward. 1991. Dioxin contamination and growth and development in great blue heron embryos. *Journal of Toxicology and Environmental Health.* 32:331-344.

Haseltine, S.D., and R.M. Prouty. 1980. Aroclor 1242 and reproductive success of adult mallards (*Anas platyrhynchos*). *Environmental Research.* 23:29-34.

Heaton, S.N., S.J. Bursian, J.P. Giesy, D.E. Tillitt, J.A. Render, P.D. Jones, D.A. Verbrugge, T.J. Kubiak, and R.J. Aulerich. 1995. Dietary exposure of mink to carp from Saginaw Bay, Michigan. 1. Effects on reproduction and survival, and the potential risk to wild mink populations. *Archives of Environmental Contamination and Toxicology.* 28:334-343.

Hebert, C.E., V. Glooschenko, G.D. Haffner, and R. Lazar. 1993. Organic contaminants in snapping turtle (*Chelydra serpentina*) populations from Southern Ontario, Canada. *Arch. Environ. Contam. Toxicol.* 24, 35-43.

Heinz, G.H., S.D. Haseltine, R.J. Hall, and A.J. Krynitsky. 1980. Organochlorine and mercury residues in snakes from Pilot and Spider Islands, Lake Michigan – 1978. *Bull. Environm. Contam. Toxicol.* 25: 738-743.

Helwig, D.D and M.E. Hora. 1983. Polychlorinated biphenyl, mercury, and cadmium concentrations in Minnesota snapping turtles. *Bull. Environm. Contam. Toxicol.* 30, 186-190.

Henry, T.R., J.M. Spitsbergen, M.W. Hornung, C.C. Abnet, and R.E. Peterson. 1997. Early life stage toxicity of 2,3,7,8-tetrachlorodibenzo-p-dioxin in zebrafish *(Danio rerio). Toxicology and Applied Pharmacology.* 142:56-68. Article No. TO968024.

Hill, E., R.G Heath, J.W. Spann, and J.D. Williams. 1975. Lethal Dietary Toxicities of Environmental Pollutants to Birds. U.S. Fish and Wildlife Special Scientific Report – Wildlife No. 191, Washington, DC.

Hochstein, J.R., R.J. Aulerich, and S.L. Bursian. 1988. Acute toxicity of 2,3,7,8-tetrachloro-dibenzo-p-dioxin in mink. *Arch. Environ. Contam. Toxicol.* 17:33-37.

Hoff, T.B., R.J. Klauda, and J.R. Young. 1988. Contribution to the biology of shortnose sturgeon in the Hudson River Estuary. Chapter 5 in C.L. Lavett, ed. *Fisheries Research in the Hudson River.* State University of New York Press. Albany, NY. pp. 171-189.

Hoffman, R.D. 1978. The Diets of Herons and Egrets in Southwestern Lake Erie. In: Sprunt, A. Ogden, J. Winckler, S., Editors: Wading Birds. Natl. Audobon Soc. Res. Report 7: 365-369.

Hoffman, D.J., M.J. Melancon, P.N. Klein, J.D. Eismann, and J.W. Spann. 1998. Comparative developmental toxicity of planar polychlorinated biphenyl congeners in chickens, American kestrels, and common terns. *Environmental Toxicology and Chemistry.* 17:4, pp 747-757.

Hoffman, D.J., G.J. Smith, and B.A. Rattner. 1993. Biomarkers of contaminant exposure in Common Terns and Black-Crowned Night Herons in the Great Lakes. *Environmental Toxicology and Chemistry.* Vol. 12:1095-1103.

Hornshaw, T.C. 1984. Development of dietary LC_{50} and reproduction test protocols using mink and ferrets as representative mammalian carnivores. PhD Thesis, Michigan State University, E. Lansing, MI. 212 pp.

Hornshaw, T.C., J. Safronoff, R.K. Ringer, and R.J. Aulerich. 1986. LC50 results in polychlorinated biphenyl-fed mink: age, season, and diet comparisons. *Arch. Environ. Contam. Toxicol.* Vol. 15, pp 717-723.

Huang, Y., M.J. Melancon, R.E. Jung, and W.H. Karasov. 1998. Induction of cytochrome P450-associated monooxygenases in northern leopard frogs, *Rana pipiens*, by 3,3',4,4',5-pentachlorobiphenyl. *Environmental Toxicology and Chemistry*, Vol. 17, No. 8, pp 1564-1569.

Hudson, R.H., R.K. Tucker, and M.A. Haegele. 1984. Handbook of toxicity of pesticides to wildlife. Second edition. US Fish and Wildlife Serv. Resour. Publ. 153. 90 pp.

Ingersoll, C.G., G.T. Ankley, R. Baudo, G.A. Burton, W. Lick, S.N. Luoma, D.D. MacDonald,

T. B. Reynoldson, K. R. Solomon, R. C. Swatrz, and W.J. Warren-Hicks. 1998. Workgroup summary report on uncertainty evaluation of measurement endpoints used in sediment ecological risk assessment, Chapter 18. In: Ecological Risk Assessment of Contaminated Sediments. SETAC press.

Ingersoll C.G., P.S. Haverland, E.L. Brunson, T.J. Canfield, F.J. Dwyer, C.E. Henke, N.E. Kemble, D.R. Mount and R.G. Fox. 1996. Calculation and evaluation of sediment effect concentrations for the amphipod *Hyalella azteca* and the midge *Chironomous riparius*. *J. Great Lakes Res.* 22(3):602-623.

Isensee, A.R. 1978. Bioaccumulation of 2,3,6,7-tetrachlorodibenzo-para-dioxin. *Ecol. Bull.* (Stockholm). 27:255-262.

Isensee, A.R., and G.E. Jones. 1975. Distribution of 2,3,7,8-tetrachlorodibenzo-p-dioxin (TCDD) in aquatic model ecosystem. *Environmental Science and Technology*. 9:668-672.

Jensen, S., J.E. Kihlstrom, M. Olsson, C. Lundberg, and J. Orberg. 1977. Effects of PCBs and DDT on mink (*Mustela vision*) during the reproductive season. *Ambio*. 6:239.

Johnson, L.L., S.Y. Sol, D.P. Lomax, G.M. Nelson, C.A. Sloan, and E. Casillas. 1997. Fecundity and egg weight in English sole, *Pleuronectes vetulus*, from Puget Sound, Washington: influence of nutritional status and chemical contaminants. *Fishery Bulletin*. 95:231-249.

Jones, P.A., R.J. Sloan, and M.P. Brown. 1989. PCB congeners to monitor with caged juvenile fish in the Upper Hudson River. *Environ. Toxicol. Chem.* 8:793-803.

Jones, D. S., G. W. Suter II and R.N. Hall. 1997. Toxicological Benchmarks for Screening Contaminants of Potential Concern for Effects on Sediment-Associated Biota: 1997 Revision. Lockheed Martin Systems, Inc. ES/ER/TM-95/R4.

Jung, R.E. and M.K. Walker. 1997. Effects of 2,3,7,8-tetrachlorodibenzo-p-dioxin (TCDD) on development of anuran amphibians. *Environmental Toxicology and Chemistry*, Vol. 16, No. 2., pp. 230-240.

Kaufmann. J.H. 1982. Raccoon and allies. In: Chapman, J.S.; Feldhamer, G.A., eds. Wild Mammals of North America. Johns Hopkins University Press, Baltimore, MD. pp. 567-585.

Khera, K.S., and J.A. Ruddick. 1973. Polychlorodibenzo-p-dioxins: Perinatal effects and the dominant lethal test in Wistar rats. In: E.H. Blair, ed., Chlorodioxins – Origin and Fate. Advances in Chemistry Series 120. Amer. Chem. Soc., Washington, DC.

Kihlstrom, J.E., M. Olsson, S. Jensen,A. Johansson, J. Ahlbom, and A. Bergman. 1992. Effects of PCB and different fractions of PCB on the reproduction of the mink (Mustela vison). *Ambio*. 21:8, pp 563-569.

301522

Kimbrough, R.D., R.E. Linder, T.B. Gaines. 1972. Morphological changes in livers of rats fed polychlorinated biphenyls. *Arch. Environ. Health.* 25:354-364.

Klauda, R.J., P.H. Muessig, and J.A. Matousek. 1988. Fisheries data sets compiled by utility-sponsored research in the Hudson River Estuary. In: Fisheries Research in the Hudson River. Edited by C. Lavett Smith. State University of New York Press, Albany, NY.

Kling, D., J. Kunkle, A.S. Roller, et al. 1978. Polychlorinated biphenyls. In vivo and in vitro modifications of cholesterol and fatty acid biosynthesis. *J. Environ. Pathol. Toxicol.* 1:813-828.

Knight, G.C. and C.H. Walker. 1982. A study of hepatic microsomal epoxide hydrolase in sea birds. *Comp. Biochem. Physiol.* 73C:211-221.

Knudsen, G.J. and J.B. Hale. 1968. Food habits of otters in the Great Lakes region. *J. of Wildlife Management* 32(1):89-93.

Kociba, R.J., D.G. Keyes, J.E. Beyer, R.M. Carreon, C.E. Wade, D.A. Dittenber, R.P. Kalnins, L.E. Frauson, C.N. Park, S.D. Barnard, R.A. Hummel, and C.G. Humiston. 1978. Results of a two-year chronic toxicity and oncogenicity study of 2,3,7,8-tetrachlorodibenzo-p-dioxin in rats. *Toxicol. Appl. Pharmacol.* 46:279-303.

Kociba, R.J., and B.A. Schwetz. 1982. Toxicity of 2,3,7,8-tetrachlorobenzo-p-dioxin (TCDD). *Drug Metab. Rev.* 13:387-406.8

Kolehmainen, S.E. 1974. Daily feeding rates of bluegill (Lepomis machrochirus) determined by a refined radioisotope method. *J. Fish. Res. Bd. Can.* 31:67-74.

Korfmacher, W.A., E.G. Hansen, Jr., and K.L. Rowland. 1986a. Tissue distribution of 2,3,7,8-TCDD in bullfrogs obtained from a 2,3,7,8-TCDD-contaminated area. *Chemosphere.* Vol. 15, No. 2, 121-126.

Korfmacher, W.A., E.B. Hansen, Jr., and K.L. Rowland. 1986b. Use of bullfrogs *(Rana catesbeiana*) as biological markers for 2,3,7,8-tetrachlorodibenzo-p-dioxin contamination in the environment. *The Science of the Total Environment*, Vol. 57, No. 257-262.

Krebs, J. R. 1974. Colonial Nesting and Social Feeding as Strategies for Exploiting Food Resources in the Great Blue Heron (*Ardea herodias*). *Behaviour* LI 1-2: 99-134.

Kubiak, T.J., H.J. Harris, L.M. Smith, T.R. Schwartz. D.L. Stalling, J.A. Trick. L. Sileo, D.E. Docherty, and T.C. Erdman. 1989. Microcontaminants and reproductive impairment of the Forster's Tern on Green Bay, Lake Michigan – 1983. *Archives of Environmental Contamination and Toxciology.* Vol. 18:706-727.

Larsson, P. 1984. Transport of PCBs from aquatic to terrestrial environments by emerging chironomids. *Environmental Pollution (Series A).* 34: 283-289.

Leonards P.E.G., S. Broekhuizen, P. De Voogt, N.M. Van Straalen, U. Brinkman, W.P. Cofino, and B. Van Hattum. 1998. Studies of bioaccumulation and biotransformation of PCBs in mustelids based on concentration and congener patterns in predators and preys. *Archives of Environ.mental Contamination and Toxicology;* 35 (4). 1998. 654-665.

Levin, E.D., S.L. Schantz, and R.E. Bowman. 1988. Delayed spatial alternation deficits resulting from perinatal PCB exposure in monkeys. *Arch. Toxicol.* 62:267-273.

Liers, E.E. 1951. Notes on the river otter (*Lutra canadensis*). *J. Mammal.* 32:1-9 In: USEPA. 1993a. Wildlife Exposure Factors Handbook. Volume I of II. Office of Research and Development. Washington, D.C. EPA/600/R-93/187a.

Lillie, R.J., H.C. Cecil, J. Bitman, G.G. Fries, and J. Verrett. 1974. Differences in response of caged white leghorn layers to various polychlorinated biphenyls (PCBs) in the diet. *Poult. Sci.* 53:726-732.

Lillie, R.J., H.C. Cecil, J. Bitman, G.F. Fries, and J. Verrett. 1975. Toxicity of certain polychlorinated and polybrominated biphenyls on reproductive efficiency of caged chickens. *Poult. Sci.* 54:1550-1555.

Linder, R.E., T.B. Gaines, R.D. Kimbrough. 1974. The effect of polychlorinated biphenyls on rat reproduction. *Food Cosmet. Toxicol.* 12:63-77.

Linzey, A.V. 1988. Effects of chronic polychlorinated biphenyls exposed on growth and reproduction of second generation white-footed mice (*Peromyscus leucopus*). *Arch. Environ. Contam. Toxicol.* 17:39-45.

Llewellyn,. L. M. and F.M. Uhler. 1952. The foods of fur animals of the Patuxent Research Refuge, Maryland. *Am. Midl. Nat.* 48: 193-203.

Long, E.R., and M.F. Buchman. 1990. A comparative evaluation of selected measures of biological effects of exposure of marine organisms to toxic chemicals. In: Biomarkers of Environmental Contamination. Pp 355-394. J.F. McCarthy, and L.R. Shugart, Editros. Lewis Publishers. Boca Raton, FL.

Long, E.R., D.D. MacDonald, S.L. Smith, F.D. Calder 1995. Incidence of Adverse Biological Effects Within Ranges of Chemical Concentrations in Marine and Estuarine Sediments. *Environmental Management.* 19(1):8197.

Lovett-Doust, L., J. Lovett Doust and M. Biernacki. 1997a. American wildcelery, Valisneria americana, as a biomonitor of organic contaminants in aquatic ecosystems. *J. Great Lakes Res.* 20(2):333-354.

Lovett-Doust, J., L Lovett-Doust, M. Biernacki, T. K. Mal and R. Lazar. 1997b. Organic contaminants in submersed macrophytes drifting in the Detroit River. *Can. Journal of Aquatic Sciences*, 54(10):2417-2327.

Lundkvist, U. 1990. Clinical and reproductive effects of Clophen-A50 (PCB) administered during gestation on pregnant guinea pigs and their offspring. *Toxicology*. 6:249-257.

Mac, M.J., T.R. Schwartz, C.C. Edsall, and A.M. Frank. 1993. Polychlorinated biphenyls in Great Lakes lake trout and their eggs: relations to survival and congener composition 1979-1988. *Journal of Great Lakes Research*. 19(4):752-765.

Mac, M.J., and J.G. Seelye. 1981. Patterns of PCB accumulation by Fry of Lake Trout. *Bulletin of Environmental Contamination & Toxicology*. Vol. 27, pp 368-375.

Mackay, D., W.Y. Shiu, and K.C. Ma, and K. Ching. 1992. Illustrated Handbook of Physical-Chemical Properties and Environmental Fate for Organic Chemicals. Vol. 1. Monoaromatic Hydrocarbons, Chlorobenzenes, and PCBs. Boca Raton, Florida. Lewis Publishers.

Magurran, A.E. 1988. Ecological Diversity and it Measurement. Princeton University Press, Princeton, NJ.

Mauck, W.L., P.M. Mehrle, and F.L. Mayer. 1978. Effects of the polychlorinated biphenyl Aroclor 1254 on growth, survival, and bone development in brook trout (*Salvelinus fontinalis*). *J. Fish Res. Board Can.* 35:1084-1088.

Mayak, D.T. 1999. NYSDEC Division of Fish, Wildlife and Marine Resources, Conservation Biologist. Hale Creek Field Station, Gloversville, New York. Personal communication by telephone

McCarty, JP and AL Secord. 1999. Reproductive ecology of tree swallows (Tachycineta bicolor) with high levels of polychlorinated biphenyl contamination. *Environmental Toxicology and Chemistry*. 18(7):1433-1439.

McCoy, G., M.F. Finlay, A. Rhone, K. James, and G.P. Cobb. 1995. Chronic polychlorinated biphenyls exposure on three generations of oldfield mice (*Peromyscus polionotus*): effects on reproduction, growth, and body residues. *Archives of Environmental Contamination & Toxicolgy*. 28:431-435.

McFarland, V.A. and J.U. Clarke. 1989. Environmental occurrence, abundance and potential toxicity of polychlorinated biphenyl congeners: considerations for a congener-specific analysis. *Environ. Health Perspectives* 81:225-239.

McLane, M.A. and D.L. Hughes. 1980. Reproductive success of screech owls fed Aroclor 1248. *Arch. Environ. Contam. Toxicol.* 9:661-665.

Menzie-Cura & Associates, Inc. (MCA). 1997. Guidance for Ecological Risk Assessment at Petroleum Release Sites. Prepared for American Petroleum Institute, Washington, DC.

Monod, G. 1985. Egg mortality of Lake Geneva charr (*Salvelinus alpinus* L.) contaminated by PCB and DDT derivatives. *Bulletin of Environmental Contamination and Toxicology*. 35:531-536.

Montz, W.E., W.C. Card, and R.L. Kirkpatrick. 1982. Effects of polychlorinated biphenyls and nutritional restriction on barbituate-induced sleeping times and selected blood characteristics in raccoons (*Procyon lotor*). *Bull. Environ. Contam. Toxicol.* 28:578-583.

Moore, R.W., and R.E. Peterson. 1996. Reproductive and developmental toxicity of polychlorinated biphenyls: to what extent are the effects aryl hydrocarbon receptor-independent. *Comments Toxicology*, Vol. 5(4-5)347-365.

Moran, M.A., and K.E. Limburg. 1986. The Hudson River Ecosystem. Springer-Verlag, New York, pp. 6-39.

Morgan, M.G., Henrion, M. 1990. Uncertainty, A guide to dealing with uncertainty in quantitative risk and policy analysis. Cambridge University Press, New York.

Morrison, H.A, Gobas F.A.P.C., Lazar R., Whittle D.M., and G.D. Haffner. 1998. Projected changes to the trophodynamics of PCBs in the Western Lake Erie ecosystem attributed to the presence of zebra mussels (*Dreissena polymorpha*). *Environmental Science & Technology*; 32 (24):3862-3867.

Murray, F.J., F.A. Smith, K.D. Nitschke, C.G. Huniston, R.J. Kociba, and B.A. Schwetz. 1979. Three-generation reproduction study of rats given 2,3,7,8-tetrachlorodobenzo-p-dioxin (TCDD) in the diet. *Toxicol. Appl. Pharmacol.* 50:241-252.

Nagy, K. A. 1987. Field metabolic rarte and food requirement scaling in mammals and birds. *Ecologicl. Monographs.* 57(2): 111-128.

National Cancer Institute (NCI). 1978. Bioassay of Aroclor 1254 for possible carcinogenicity. NCI-GC-TR-38. Bethesda, MD. NTIS PB279624.

National Council on Radiation Protection (NCRP). 1996. A guide for uncertainty analysis in dose and risk assessments related to environmental contamination. NCRP Commentary No. 14. National Council on Radiation Protection and Measurements, Bethesda.

Niethammer, K.R., D.H.White, T.S. Baskett, and M.W. Sayre. 1984. Presence and biomagnification of organochlorine chemical residues in oxbow lakes of Northeastern Louisiana. *Arch. Environ. Contam. Toxicol.* 13, 63-74.

Niimi, A.J. 1983. Biological and toxicological effects of environmental contaminants in fish and their eggs. *Can. J. Fish. Aquat. Sci.* 40:306-312.

National Oceanographic and Atmospheric Administration (NOAA). 1990. The Potential for Biological Effects of Sediment-Sorbed Contaminants Tested in the National Status and Trends Program. NOAA Technical Memorandum NOS OMA 52. Seattle, Washington. March, 1990.

National Oceanographic and Atmospheric Administration (NOAA). 1994. Aquatic Ecological Risk Assessment for Metal Bank of America, Cottman Avenue. NPL Site. Final Draft prepared for USEPA Region 3, Philadelphia, Pennsylvania. March 1994.

National Oceanographic and Atmospheric Administration (NOAA). 1997a. Hudson River Congener-specific Analysis, Data Summary and Analysis Report. Prepared by EVS Environmental Consultants. July, 1997.

National Oceanographic and Atmospheric Administration (NOAA). 1997b. The Coastal Resources Coordinator's Bioassessment Manual. Report No. HAZMAT 93-1 (revised). Seattle, WA. Prepared by MacDonald, D.A., M.B. Matta, L.J. Field, C. Cairncross, and M.D. Munn. 147 pp + appendices. August, 1997.

National Oceanographic and Atmospheric Administration (NOAA). 1999a. Development and Evaluation of Consensus-Based Sediment Effect Concentrations for PCBs in the Hudson River. Prepared for NOAA Damage Assessment Center, Silver Spring, MD. Prepared through Industrial Economics by MacDonald Environmental Sciences Ltd. March, 1999.

National Oceanographic and Atmospheric Administration (NOAA). 1999b. Reproductive, Developmental and Immunotoxic Effects of PCBs in Fish: a Summary of Laboratory and Field Studies. Prepared for NOAA Damage Assessment Center, Silver Spring, MD. Prepared through Industrial Economics Inc. by E. Monosson. March, 1999.

Nebeker, A.V., and F.A. Puglisi. 1974a. Effect of polychlorinated biphenyls (PCBs) on survival and reproduction of daphnia, gammarus, and tanytarsus, *Trans. Am. Fish. Soc.* 103(4):722-728.

Nebeker, A.V., F.A. Puglisi, and D.L. Defoe. 1974b. Effect of polychlorinated biphenyl compounds the survival and reproduction of the fathead minnow and flagfish. *Trans. Am. Fish. Soc.* 3:562-568.

New York State Department of Environmental Conservation (NYSDEC). 1987. Toxic substances in fish and wildlife. Analyses since May 1, 1982. Volume 6. September 1987. Bureau of Environmental Protection, Division of Fish and Wildlife, Albany, New York. September, 1987.

New York State Department of Environmental Conservation (NYSDEC). 1989. Fish and Their Habitats in the Upper Hudson Estuary. Prepared by D.M. Carlson, Division of Fish and Wildlife. April, 1989.

New York State Department of Environmental Conservation (NYSDEC). 1993. 20 Year Trends in Water Quality of Rivers and Streams in New York State Based on Macroinvertebrate Data

1972- 1992. By R.W. Bode, M.A. Novak, and L.E. Abele, Stream Biomonitoring Unit, Bureau of Monitoring and Assessment Division of Water.

New York State Department of Environmental Conservation (NYSDEC). 1998a. Technical Guidance for Screening Contaminated Sediments, New York State Department of Environmental Protection, Division of Fish and Wildlife, Division of Marine Resources. March, 1998.

New York State Department of Environmental Conservation (NYSDEC). 1998b. Draft Hudson River Estuary Management Action Plan. The Hudson River Estuary Management Program. April, 1998.

New York State Department of Environmental Conservation (NYSDEC). 1998c. Ambient Water Quality Standards and Guidance Values and Groundwater Effluent Limitations. Division of Water Technical and Operational Guidance Series (1.1.1). June, 1998.

New York State Department of Environmental Conservation (NYSDEC). 1999a. New York State Amphibian and Reptile Atlas 1990-1998, Interim Report. www.dec.state.ny.us/website/dfwmr/wildlife/herp. February, 1999.

New York State Department of Environmental Conservation (NYSDEC). 1999b. Natural Heritage Program Report on Rare Species and Ecological Communities. Prepared by NY Natural Hertiage Program, NYSDEC, Latham, New York upon the request of TAMS Consultants, Inc. May 5, 1999.

New York State Department of Health (NYSDOH). October 1974. Macroinvertebrate Survey of the Upper Hudson River New York - 1972. Environmental Health Report No. 2. By K. Simpson, T.B. Lyons, and S.P. Allen.

New York State Department of Health (NYSDOH). 1997. Bat Control. http://pmep.cce.cornell.edu/ recommends/vertebrates/vertbats.html.

New York State Department of State (NYSDOS). 1990. Division of Coastal Resources and Waterfront Revitalization and The Nature Conservancy. Hudson River Significant Tidal Habitats: A guide to the functions, values, and protection of the river's natural resources. 184 pp.

New York Times. 1997. Newspaper Article. September 17, 1997.

Newell, A.J., D.W. Johnson, and L.K. Allen. 1987. Niagara River Biota Contamination Project: Fish Flesh Criteria for Piscivorous Wildlife. NYSDEC. Division of Fish and Wildlife. Technical Rpt. 87-3. 182 pp.

Niimi, A.J. 1996. PCBs in Aquatic Organisms. In Environmental Contaminants in Wildlife: Interpreting Tissue Concentrations. W. Beyer, G.H. Heinz, and A.W. Redmon-Norwood (eds.). Lewis Publishers. Boca Raton, FL.

Nimmo, D.R., J. Forester, P.T. Heitmuller, and G.H. Cook. 1974. Accumulation of Aroclor 1254 in grass shrimp (*Palaemonetes pugio*) in laboratory and field exposures. *Bull. Environ. Contam. Toxicology.* 11:303-308.

Nosek, J.A., J.R. Sullivan, T.E. Amundson, S.R. Craven, L.M. Miller, A.G. Fitzpatrick, M.E. Cook, and R.E. Peterson. 1993. Embryotoxicity of 2,3,7,8-tetrachlorodibenzo-p-dioxin in the ring-necked pheasants. *Environ. Toxicol. Chem.* 12:1215-1222.

Nosek, J.A., J.R. Sullivan, S.S. Hurley, S.R. Craven, and R.E. Peterson. 1992. Toxicity and reproductive effects of 2,3,7,8-tetrachlorodibenzo-p-dioxin toxicity in ring-necked pheasant hens. *J.Toxicol. Environ. Health.* 35:187-198.

Novak, M.A., A.A. Reilly, and S.J. Jackling. 1988. Long-term monitoring of polychlorinated biphenyls in the Hudson River (New York) using caddisfly larvae and other macroinvertebrates. *Arch. Environ. Contam. Toxicol.* 17:669-710.

Nye, P. 1999a. Eastern Bald Eagle Migration Update. http://www.learner.org /jnorth/spri...9/species/eagle/EUpdate021099.html. February 10, 1999.

Nye, P. 1999b. New York State Division of Fish, Wildlife and Marine Resources. Endangered Species Unit. Delmar, NY. Personal Communication by telephone. March, 1999.

Nye, P. and L.H. Suring. 1978. Observations concerning a wintering population of Bald Eagles on an area in southeastern New York. N.Y. Fish and Game Journal. 25(2): 91-107.

O'Brien & Gere Engineers, Inc. 1993. Hudson River Project Sampling and Analysis Program, 1991 Sediment Sampling and Analysis Program. May 1993. General Electric Co., Albany, New York.

O'Brien & Gere Engineers, Inc. 1995. Fort Edward Dam PCB Remnant Containment 1994 Post-Construction Monitoring Program. Prepared for General Electric Company, Albany, NY.

Olafsson, P.G., A.M. Bryan, B.Bush, and W. Stone. 1983. Snapping turtles – A biological screen for PCB's. Chemosphere, Vol. 12, No. 11/12, pp. 1525-1532.

Olivieri, C.E., and K.R. Cooper. 1997. Toxicity of 2,3,7,8-tetrachlorodibenzo-p-dioxin (TCDD) in embryos and larvae of the fathead minnow (*Pimephales promelas*). *Chemosphere* 34(5-7):1139-1150.

Overmann, S.R., J. Kostas, L.R. Wilson, et al. 1987. Neurobehavioral and somatic effects of perinatal PCB exposure in rats. *Environ. Res.* 44:56-70.

Peakall, D.B., J.L. Lincer, and S.E. Bloom. 1972. Embryonic mortality and chromosomal alterations caused by Aroclor 1254 in ring doves. *Environmental Health Pespectives.* Vol. 1:103-104.

301529

Peakall, D.B., and M.L. Peakall. 1973. Effects of a polychlorinated biphenyl on the reproduction of artificially and naturally incubated dove eggs. *J. Appl. Ecol.* 10:863-868.

Penrod, B. 1999. NYSDEC Division of Fish, Wildlife and Marine Resources, Furbearer Unit. Warrensburg, New York. Personal communication by telephone.

Persaud, D., R. Jaagumagi and A. Hayton. August 1993. Guidelines for the protection and management of aquatic sediment quality in Ontario. Ontario Ministry of the Environment and Energy.

Peterson, R. 1980. A Field Guide to the Birds East of the Rockies, 4th ed. Houghton Miffin Co., Boston, MA. 384 pp.

Petty, J.D., B.C. Poulton, C.S. Charbonneau, J.N. Huckins, S.B. Jones, J.T. Cameron, and H. Prest. 1998. Determination of bioavailable contaminants in the Lower Missouri River following the flood of 1993. *Environ. Sci. and Tech.* 32(7):837-842.

Phillips, D.J.H. 1986. Use of organisms to quantify PCBs in marine and estuarine environments. In: J.S. Waid (ed). PCBs and the Environment. Boca Raton, FL: CRC Press, Inc. pp. 127-182.

Phillips, W.E.J., G. Hatina, D.C. Villeneuve, et al. 1972. Effect of parathion administration in rats following long term feeding with PCBs. *Environ. Physiol. Biochem.* 2:351-354.

Platonow, N. and C. Karstad. 1973. Dietary effects of polychlorinated biphenyls on mink. *Can. J. Comp. Med.* 37:391-400.

Platanow, N.S., and B.S. Reinhart. 1973. The effects of polychlorinated biphenyls (Aroclor 1254) on chicken egg production, fertility and hatchability. *Can. J. Comp. Med.* 37:341-346.

Powell, D.C., R.J. Aulerich, J.C. Meadows, D.E. Tillett, J.P. Giesy, K.L. Stromborg, S.J. Bursian. 1996a. Effects of 3,3',4,4',5-pentachlorobiphenyl (PCB 126) and 2,3,7,8-tetrachlorodibenzo-p-dioxin (TCDD) injected into the yolks of chicken (*Gallus domesticus*) eggs prior to incubation. *Arch. Environ. Contam. Toxicol.* 31:404-409.

Powell, D.C., R.J. Aulerich, K.L. Stromberg, S.J. Bursian. 1996b. Effects of 3,3',4,4',-tetrachlorobiphenyl, 2,3,3',4,4',-pentachlorobiphenyl, and 3,3',4,4',5-pentachlorobiphenyl on the developing chicken embryo when injected prior to incubation. *Journal of Toxicology and Environmental Health.* 49:319-338.

Powell, D.C., R.J. Aulerich, J.C. Meadows, D.E. Tillitt, J.F. Powell, J.C. Restum, K.L. Stromborg, J.P. Giesy, and S.J. Bursian. 1997. Effects of 3,3',4,4',5-pentachlorobiphenyl (PCB 126), 2,3,7,8-tetrachlorodibenzo-p-dioxin (TCDD), or an extract derived from field-collected cormorant eggs injected into double-crested cormorant (*Phalacrocoras auritus*) eggs. *Environmental Toxicology and Chemistry.* 16:7, pp 145-1455.

Reader, A.L., G.L. Foley, D.K. Nichols, L.G. Hansen, B. Wikoff, S. Faeh, J. Eisold, M.B. Wheeler, R. Warner, J.E. Murphy, and V.R. Beasley. 1998. Forms and prevalence of intersexuality in cricket frogs (*Acris crepitans*). *Environmental Health Perspectives*, Vol. 106, No. 5, pp. 261-266.

Restum, J.C., S.J. Bursian, J.P. Giesy, J.A. Render, W.G. Helferich, E.B. Shipp, and D.A. Verbrugge. 1998. Multigenerational study of the effects of consumption of PCB-contaminated carp from Saginaw Bay, Lake Huron, on mink. 1. Effects on mink reproduction, kit growth and survival, and selected biological parameters. *Journal of Toxicology and Environmental Health*, Part A, 54:343-375.

Richard, M.R., M.E. Fox and F.R. Pick. 1997. PCB concentrations and congener composition in macrophytes and sediments in the St. Lawrence River near Cornwall, Ontario. *J. Great Lakes Res.* 23(3):297-306.

Rico, M.C., L.M. Hernandez, M.J. Gonzale, M.A. Fernandez, and M.C. Montero. 1987. Organochlorine and metal pollution in the Donana National Park during the period 1983-1988. *Bull. Environ. Contam. Toxicol.* 39: 1076-1083.

Ringer, R.K., R.J. Aulerich, and M.R. Bleavins. 1981. Biological effects of PCBs and PBBs on mink and ferrets: a review. In: Khan MAQ, ed. Pergamon Press, Elmsford, NY. 329-343. Halogenated Hydrocarbons: Health and Ecological Effects.

Ristich, S.S., M. Crandall, and J. Fortier. 1977. Benthic and epibenthic macroinvertebrates of the Hudson River. I. Distribution, natural history and community structure. *Est. Coastal Mar. Sci.* 5:255-266.

Roberts, J.R., D.W. Rodgers, J.r. Bailey, and M.A. Rorke. 1978. Polychlorinated biphenyls: biological criteria for an assessment of their effects on environmental quality. National Research Council of Canada. Report No. 16077, 172 pp.

Robertson, R.J., B.J. Stutchbury and R.R. Cohen. 1992. Tree Swallow, *Tachycineta bicolor*. In. The Birds of North America. No. 11 (A.Poole, P. Stettenheim and F. Gill Eds.). Philadelphia: The Academy of Natural Sciences; Washington, DC. The American Ornithologists Union.

Ryan, J.J., P.F.Y. Lau, J.A. Hardy, W. Stone, P. O'Keefe, and J.F. Gierthy. 1986. 2,3,7,8-tetrachlorodibenzo-p-dioxin and related dioxins and furans in snapping turtles *(Chelydra serpentina)* tissues from the Upper St. Lawrence River. Chemosphere, Vol. 15, No. 5, pp. 537-548.

Sabourin, T.D., W.B. Stickle, T.C.Michot, C.E. Villars, D.W. Garton, and H.R. Mushinsky. 1984. Organochlorine residue levels in Mississippi River water snakes in Southern Louisiana. *Bull. Environ. Contam. Toxicol.* 32: 460-468.

Safe, S. 1990. Polychlorinated biphenyls (PCBs), dibenzo-p-dioxins (PCDDs), dibenzofurans (PCDFs) and related compounds: environmental and mechanistic considerations which support the development of Toxic Equivalency Factors (TEFs). *Critical Reviews in Toxicol.* 21:51-88.

Safe, S., S. Bandiera, T. Sawyer, L. Robertson, L. Safe, A. Parkinson, P.E. Thomas, D.E. Ryan, L.M. Reik, W. Levin, M.A. Denomme, and T. Fujita. 1985a. PCBs: structure – function relationships and mechanisms of action. *Environmental Health Perspectives* 60: 47-56.

Safe, S., S. Bandiera, T. Sawyer, B. Zmudzka, G. Mason, M. Romkes, M.A. Denomme, J. sparling, A.B. Okey, and T. Fujita. 1985b. Effects of structure on binding to the 2,3,7,8-TCDD receptor protein and AHH induction – halogenated biphenyls. *Environmental Health Perspectives* 61: 21-33.

Sager, D.B., and D.M. Girard. 1994. Long-term effects on reproductive parameters in female rats after translactational exposure to PCBs. *Environmental Research.* 66:52-76.

Salyer, J.C. and Lagler, K.F. 1946. The eastern belted kingfisher, Megaceryle alcyon alcyon (Linnaeus), in relation to fish management. Trans. Am. Fish. Soc. 76: 97-117.

Sample, B.E., DM. Opresko, and G.W. Suter II. June 1996. Toxicological Benchmarks for Wildlife: 1996 Revision. Lockheed Martin Energy Systems, Inc. ES/ER/TM-96/R3.

Sanderson, J.T., J.E. Elliott, R.J. Norstrom, P.E. Whitehead, L.E. Hart, K.M. Cheng, and G.D. Bellward. 1994. Monitoring biological effects of polychlorinated bibenxo-p-dioxins, dibenzofurans, and biphenyls in great blue heron chicks (*Ardea herodias*) in British Columbia. *Journal of Toxicology and Environmental Health.* 41:435-450.

Schwetz, A.B., J.M. Norris, G.L. Sparschu, V.K. Rowe, P.J. Gehring, J.L. Emerson, and C.G. Gerbig. 1973. Toxicology of chlorinated dobenzo-p-dioxins. *Environ. Health Perspect.* 5:87-99.

Scott, M.L. 1977. Effects of PCBs, DT, and mercury compounds in chickens and Japanese quail. *Federation Proceed.* 36:1888-1893.

Secord, A.L. and J.P. McCarty. 1997. Polychlorinated biphenyl contamination of tree swallows in the Upper Hudson River Valley, New York. US Fish and Wildlife Service, Cortland, NY.

Shaw, G.R. and D.W. Connell. 1984. Factors controlling bioaccumulation of PCBs. In *PCBs and the Environment Volume 1.* National Technical Information Service. pp. 501-516.

Sijm, D.T.H.M., W. Selnen, A. Opperhulzen. 1992. Life-cycle biomagnification study in fish. *Environ. Sci. Technol.* MISSING PART OF REF

Simpson, K.W. 1986. "Final Report Biomonitoring of PCBs in the Hudson River I. Results of Long-Term Monitoring using Caddisfly (Insecta: *Trichoptera: Hydropsychidae*) Larvae and Multiple Residues II. Development of Field Protocol for Monitoring PCB uptake by caged live

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Chironomus tentans (Insecta: *Diptera: Chironomidae*) larvae during dredging operations." Report prepared by M.A. Novak and A.A. Reilly, NYS Department of Health 99pp. plus appendices.

Sloan, R., M. Brownm R. Brandt, and C. Barnes. 1985. Hudson River PCB relationships between resident fish, water and sediment. *Northeastern Environmental Science* 3:137-151.

Spies, R.B. and D.W. Rice, Jr. 1988. Effects of organic contaminants on reproduction of the starry flounder *Platichthys stellatus* in San Francisco Bay. II. Reproductive success of fish captured in San Francisco Bay and spawned in the laboratory. *Marine Biology*. 98:191-200.

Spies, R.B., J.J. Stegeman, D.W. Rice, B. Woodlin, P. Thomas, J.E. Hose, J.N. Cross, and M. Prieto. 1990. Sublethal responses of *Platichthys stellatus* to organic contamination in San Francisco Bay with emphasis on reproduction in biomarkers of environmental contamination. pp. 87-121. J.F. McCarthy and L.R. Shugart, Editors. Lewis Publishers. Boca Raton, FL.

Spinola, R.M. 1999. Graduate Student, Penn State Cooperative Wetland Center, School of Forest Resources, The Pennsylvania State University, University Park, PA. Personal Communication by telephone.

Stanne, S.P., R.G. Panetta, and B.E. Forist. 1996. The Hudson: An Illustrated Guide to the Living River. Rutgers University Press. New Brunswick, NJ.

Stewart, A.J., G.J. Haynes, and M.I. Martinez. 1992. Fate and biological effects of contaminated vegetation in a Tennessee stream. *Environ. Toxicol. Chem.* 11:653-664.

Stickel, W.H., L.F. Stickel, R.A. Dyrland, and D.L. Hughes. 1984. Aroclor 1254 residues in birds: lethal levels and loss rates. *Arch. Environ. Contam. Toxicol.* 13:7-13.

Stone, W.B., Kiviat, E., and Butkas, S.A. 1980. Toxicants in snapping turtles. New York Fish and Game Journal. Vol. 27, No. 1, pp. 39-50.

Strayer, D.L., J. Powell, P. Ambrose, L.C. Smith, M.L. Pace, and D.T. Fischer. 1996. Arrival, spread, and early dynamics of a zebra mussel (*Dreisssena polymorpha*) population in the Hudson River estuary. *Can. J. Fish. Aquat. Sci.* 53:1143-1149.

Strayer, D.L. and L.C. Smith. 1996. Relationships between zebra mussels (*Dreisssena polymorpha*) and unionid clams during the early stages of the zebra mussel invasion of the Hudson River. *Freshwater Biology* 36:771-779.

Struger, J., J.E. Elliott, C.A. Bishop, M.E. Obbard, R.J. Norstrom, D.V. Weseloh, M. Simon, and P. Ng. 1993. Environmental contaminants in eggs of the common snapping turtle (Chelydra serpentina) from the Great Lakes – St. Lawrence River Basin of Ontario, Canada (1981, 1984). J. Great Lakes Res. 19 (4): 681-694

Suter, G.W. 1993. Ecological Risk Assessment. Boca Raton, FL: Lewis Publishers.

Suter II, G.W. 1996. Risk Characterization for Ecological Risk Assessment of Contaminated Sites. Lockheed Martin Energy Systems, Inc. ES/ER/TM-200.

Swackhamer, D.L. and R.S. Skoglund. 1993. Bioaccumulation of PCBs by algae: Kinetics versus equilibrium. *Environ. Toxicol. Chem.* 13:949-962.

Tabatabai, F.R. and M.L. Kennedy. 1988. Food habits of the racccoon (*Procyon lotor*) in Tennessee. J. *Tenn. Acad. Sci.* 63: 89-94.

Thomann, R.V. 1989. Bioaccumulation model of organic chemical distribution in aquatic food chains. *Environ. Sci. Technol.* 23:699-707.

Thomann, R.V. 1981. Equilibrium model of fate of microcontaminants in diverse aquatic food chains. *Can. J. Fish Aquat. Sci.* 38:280-296.

Thomann, R.V. and J.P. Connolly. 1984. Model of PCB in Lake Michigan lake trout food chain. *Environ. Sci. Technol.* 18:65-71.

Thomann, R.V., J.P. Connolly, and T.F. Parkerton. 1992. An Equilibrium Model of Organic Chemical Accumulation in Aquatic Food Webs with Sediment Interaction. *Environ. Toxicol. and Chem.*, Vol. 11, pp. 615-629.

Tillitt, D.E., G.T. Ankley, J.P. Giesy, J.P. Ludwig, H.Kurita-Matsuba, L. Sileo, K.L. Stromborg, J. Larson, T.J. Kubiak. 1992. Polychlorinated biphenyl residues and egg mortality in double-crested cormorants from the Great Lakes. *Environmental Toxicology and Chemistry.* Vol. 11:1282-1288.

Tillitt, D.E., T.J. Kubiak, G.T. Ankley, and J.P. Giesy. 1993. Dioxin-like toxic potency in Forster's Tern eggs from Green Bay, Lake Michigan, North America. *Chemosphere.* Vol. 26(11):2079-2084.

Tillitt, D.E., Gale, R.W., J.C. Meadows, J.L. Zajicek, P.H. Peterman, S.N. Heaton, P.D. Jones, S.J. Bursian, T.J. Kubiak, J.P. Giesy, and R.J. Aulerich. 1995. Dietary exposure of mink to carp from Saginaw Bay. 3. Characterization of dietary exposure to planar halogenated hydrocarbons, dioxin equivalents, and biomagnification. *Environmental Science & Technology.* 30(1):283-291.

Truelove, J., D. Grant, J. Mes, H. Tryphonas, L. Trypohonas, and Z. Zawidzka. 1982. Polychlorinated biphenyl toxicity in the pregnant cynomolgus monkey: A pilot study. *Arch. Environ. Contam. Toxicol.* 11:583-588.

Tumasonis, C.F., B. Bush, and F.D. Baker. 1973. PCB levels in egg yolks associated with embryonic mortality and deformity of hatched chicks. *Arch. Environ. Contam. Toxicol.* 1(4):312-324.

US Army Corps of Engineers (ACE) Waterways Experiment Station. 1988. Environmental Effects of Dredging Technical Notes. No. EEDP-01-13. Relationship Between PCB Tissue Residues and Reproductive Success of Fathead Minnows. April 1988.

United States Environmental Protection Agency (USEPA). 1980. *Ambient Water Quality Criteria Document for Polychlorinated Biphenyls.* Office of Health and Environmental Assessment, Cincinnati, OH. EPA 440/5-80-068.

United States Environmental Protection Agency (USEPA). 1989a. Risk Assessment Guidance for Superfund: Volume 2 – Environmental Evaluation Manual, Interim Final. Office of Solid Waste and Emergency Response. EPA/540/1-89/001A.

United States Environmental Protection Agency (USEPA). 1989b. Interim Procedures for estimating risks associated with exposure to mixtures of chlorinated dibenzo-p-dioxins and –dibenzofurans (CDDs and CDFs). EPA/625/3-89/016. Risk Assessment Forum, Washington, DC.

United States Environmental Protection Agency (USEPA). 1991a. Proposed Technical Basis for Establishing Sediment Quality Criteria for Nonionic Organic Chemicals Using Equilibrium Partitioning. Office of Science and Technology, Health and Ecological Criteria Division, Washington, DC.

United States Environmental Protection Agency (USEPA). 1991b. Phase 1 Report - Reveiw Copy Interim Characterization and Evaluation, Hudson River PCB Reassessment RI/FS. USEPA Work Assignment No. 013-2N84. Prepared by TAMS/Gradient. August, 1991

United States Environmental Protection Agency (USEPA). 1991c. Ecological assessment of Superfund sites: an overview. ECO Update 1(2). Office of Solid Waste and Emergency Response, USEPA: Washington, DC. Publication 9345.0-05I. December, 1991.

United States Environmental Protection Agency (USEPA). 1992a. Framework for Ecological Risk Assessment. Washington, DC. Risk Assessment Forum. EPA/630/R-92/001. February, 1992.

United States Environmental Protection Agency (USEPA). 1992b. Final Phase 2 Work Plan and Sampling Plan, Hudson River PCB Reassessment RI/FS. USEPA Region II, New York. Prepared by TAMS/Gradient. September, 1992.

United States Environmental Protection Agency (USEPA). 1992c. Phase 2B Sampling and Analysis Plan/Quality Assurance Project Plan, Volume 1: Flow-Averaged Water-Column Sampling, Hudson River PCB Reassessment RI/FS, December 30, 1992. USEPA, Region II, New York. Prepared by TAMS/Gradient. December, 1992.

United States Environmental Protection Agency (USEPA). 1993a. Phase 2B Sampling and Analysis/Quality Assurance Project Plan, Volume 2 Benthic Invertebrate and Sediment Grab Sampling, Hudson River PCB Reassessment RI/FS, February 18, 1993. USEPA, Region II, New

York. Prepared by TAMS/Gradient. February, 1993.

United States Environmental Protection Agency (USEPA). 1993b. Wildlife Exposure Factors Handbook. Office of Research and Development, Washington, DC. EPA/600/R-93/187a. December, 1993.

United States Environmental Protection Agency (USEPA). 1995a. Great Lakes Water Quality Initiative Criteria Documents for the Protection of Wildlife. Office of Water, Washington, DC. EPA/820/B-95/008. March, 1995.

United States Environmental Protection Agency (USEPA). 1995b. Phase 2 Report- Review Copy. Further Site Characterization and Analysis, Database Report, Hudson River PCBs Reasssessment RI/FS. Prepared for USEPA, Region II, New York. Prepared by TAMS/Gradient. October, 1995.

United States Environmental Protection Agency (USEPA). 1996. Supplemental Risk Assessment Guidance for Superfund. Office of Environmental Assessment Risk Evaluation Unit with the Assistance of: ICF Kaiser under ESAT TID 10-9510-718.

United States Environmental Protection Agency (USEPA). 1997a. Phase 2 Report, Further Site Characterization and Analysis, Volume 2C- Data Evaluation and Interpretation Report, Hudson River PCBs Reasssessment RI/FS. Prepared for USEPA, Region II, New York. Prepared by TAMS *et al.* February, 1997c.

United States Environmental Protection Agency (USEPA). 1997b. Ecological Risk Assessment Guidance for Superfund: Process for Designing and Conducting Ecological Risk Assessments. Interim Final. Environmental Response Team, Edison, NJ. EPA 504/R-97/006. June 5, 1997.

United States Environmental Protection Agency (USEPA). 1997c. Revision to Rapid Bioassessment Protocols For Use in Streams and Rivers: Periphyton, Benthic, Macroinvertebrates, and Fish. EPA 841-D-97-002. Office of Water. July 28, 1997.

United States Environmental Protection Agency (USEPA). 1997d. The incidence and severity of sediment contamination in surface waters of the United States. EPA 823/R-97-006. Office of Science and Technology, Washington, DC.

United States Environmental Protection Agency (USEPA). 1998a. Draft Report from the Workshop on the Application of 2,3,7,8-TCDD To Fish and Wildlife. Risk Assessment Forum, US Environmental Protection Agency, Washington, D.C. February, 1998.

United States Environmental Protection Agency (USEPA). 1998b. Phase 2 Report- Further Site Characterization and Analysis. Volume 2C-A Low Resolution Sediment Coring Report. Addendum to the Data Evaluation and Interpretation Report, Hudson River PCBs Reassessment RI/FS. USEPA, Region II, New York. Prepared by TAMS/Gradient/Tetra Tech. July, 1998.

United States Environmental Protection Agency (USEPA). 1998c. Database for the Hudson River PCBs Reasssessment RI/FS. Release 4.1 (Compact Disk). Prepared for USEPA, Region II and the US Army Corps of Engineers, Kansas City District. Prepared by TAMS Consultants, Inc. August, 1998.

United States Environmental Protection Agency (USEPA). 1998d. Hudson River PCBs Reasssessment RI/FS; Phase 2 Ecological Risk Assessment, Scope of Work. Prepared for USEPA, Region II and the US Army Corps of Engineers, Kansas City District. Prepared by TAMS/MCA. September, 1998.

United States Environmental Protection Agency (USEPA). 1999a. Responsiveness Summary for the Low Resolution Sediment Coring Report. USEPA, Region 2, New York. Prepared by TAMS/Gradient/Tetra Tech. February, 1999.

United States Environmental Protection Agency (USEPA). 1999b. Hudson River PCBs Reasssessment RI/FS; Responsiveness Summary for Phase 2- Ecological Risk Assessment, Scope of Work. Prepared for USEPA, Region 2 and the US Army Corps of Engineers, Kansas City District. Prepared by TAMS/MCA. April, 1999.

United States Environmental Protection Agency (USEPA). 1999c. Further Site Characterization and Analysis, Volume 2D- Baseline Modeling Report Hudson River PCBs Reassessment RI/FS. Prepared for USEPA Region 2 and US Army Corps of Engineers, Kansas City District. Prepared by Limno-Tech, Inc. (LTI), Menzie-Cura & Associates, Inc. (MCA), and Tetra-Tech, Inc. May, 1999.

US Fish and Wildlife Service (USFWS). 1997. Polychlorinated Biphenyl Contamination of Tree Swallows in the Upper Hudson River Valley, New York. Effects on Breeding Biology and Implications for Other Bird Species. Preparers: A.L. Secord and J.P. McCarty.

US Fish and Wildlife Service (USFWS). April 14, 1999. Letter from M. W. Clough acting for D. A. Stilwell, Acting Field Supervisor to H. Chernoff, TAMS.

Van den Berg, M., L. Birnbaum, A.T.C. Bosveld, B. Brunstrom, P. Cook, M. Feeley, J.P. Giesy, A. Hanberg, R. Hasegawa, S.W. Kennedy, T. Kubiak, J. C. Larsen, F.X. Rolaf van Leeuwen, A.K. Jjien Liem, C. Nolt, R.E. Peterson, L. Poellinger, S. Safe, D. Schrenk, D. Tillitt, M. Tyslind, M. Younges, F. Waern, and T. Zacharewski. 1998. Toxic equivalency factors (TEFs) for PCBs, PCDDs, PCDFs for humans and wildlife. *Environmental Health Perspectives*. Vol. 106 (12):775-792.

Van der Oost, R., H. Heida, and A. Oppenhuizen. 1988. Polychlorinated biphenyl congeners in sediments, plankton, molluscs, crustaceans, and eel in a freshwater lake: Implications of using reference chemicals and indicator organisms in bioaccumulation studies. *Arch. Environ. Contam. Toxicol.* 17:721-729.

Villeneuve, D.C., D.L. Grant, K. Khera, et al. 1971. The fetotoxicity of a polychlorinated biphenyl mixture (Aroclor 1254) in the rabbit and in the rat. *Environ. Physiol.* 1:67-71.

von Westernhagen, H., H. Rosenthal, V. Dethlefsen, W. Ernst, U. Harms, and P.D. Hansen. 1981. Bioaccumulating substances and reproductive success in Baltic flounder *Platichthys flesus*. *Aquatic Toxicology*, 1:85-99.

Vos, J.G., J.J.T.W.A. Strik, C.W.M. van Holsteyn, and J.H. Pennings. 1971. Polychlorinated biphenyls as inducers of hepatic porphyria in Japanese quail, with special reference to (-aminolevulinic acid synthetase activity, fluorescence, and residues in the liver. *Toxicology and Applied Pharmacology*. 20:232-240.

Walker, M.K., P.M. Cook, A.R. Batterman, B.C. Buttrworth, C.Berini, J.J. Libal, L.C. Hufnagle, and R.E. Peterson. 1994. Translocation of 2,3,7,8-tetrachlorodibenzo-p-dioxin from adult female lake trout (*Salvelinus namaycush*) to oocytes: effects on early life stage development and sac fry survival. *Can. J. Fish. Aquat. Sci.* 51:1410-1419.

Walker, C.H., C.G. Knight, J.K. Chipman, and J.J. Ronsis. 1984. Hepatic microsomal monoxygenase of sea birds. *Mar. Env. Research.* 14:416-419.

Walker, C.H. and R.E. Peterson. 1991. Potencies of polychlorinated dibenzo-p-dioxin, dibenzofuran, and biphenyl congeners, relative to 2,2,7,8-tetrachlorodibenzo-p-dioxin, for producing early life stage mortality in rainbow trout (*Oncorhynchus mykiss*). *Aquat. Toxicol.* 21: 219-238.

Walker, M.K., L.C. Hufnagle, Jr., M.K. Clayton, and R.E. Peterson. 1992. An egg injection method for assessing early life stage mortality of polychlorinated dibenzo-p-dioxins, dibenzofurans, and biphenyls in rainbow trout, (*Oncorhynchus mykiss*). *Aquatic Toxicology*. 22:15-38.

Walker, M.K., and R.E. Peterson. 1994. Toxicity of 2,3,7,8-tetrachlorodibenzo-p-dioxin to brook trout (*Salvelinus fontinalis*) during early development. *Environmental Toxicology and Chemistry*. 15(5):817-820.

Washington State Department of Ecology. July 1997. Creation and Analysis of Freshwater Sediment Quality Values in Washington State, Publication No. 97-323a.

Weinstein, L.H. (ed.). 1977. *An Atlas of the Biologic Resources of the Hudson River Estuary.* Boyce Thompson Institute of Plant Research. Yonkers, N.Y. 104 pp.

Wells, A.W., J.A. Matousek, and J.B. Hutchinson. Abundance Trends in Hudson River White perch. In: *Fishery Biology.*

Weseloh, D.V., and S.M. Teeple. 1983. Double-crested Cormorants of the Great Lakes: egg-laying parameters, reproductive failure, and contaminant residues in eggs, Lake Huron 1972-1973. *Can. J. Zool.* Vol. 61:427-436.

Westin, D.T., C.E. Olney, and B.A. Rogers. 1983. Effects of parental and dietary PCBs and survival, growth, and body burdens of larval striped bass. *Bull. Environ. Contam. Toxicol.* 30:50-57.

Westin, D.T., C.E. Olney, and B.A. Rogers. 1985. Effects of parental and dietary organochlorines on survival and body burdens of striped bass larvae. *Transactions of the American Fisheries Society.* 114:125-136.

White, D.H., and D.J. Hoffman. 1995. Effects of polychlorinated dibenzo-p-dioxins and *dibenzofurans on nesting wood ducks (Aix Sponsa) at Bayou Meto, Arkansas. Environmental Health Perspectives.* 103(4):37-39.

White, D.H., and J.T. Seginak. 1994. Dioxins and furans linked to reproductive impairment in wood ducks. *J. Wild. Manage.* 58(1):100-106.

Wiemeyer, S.N., C.M. Bunck, and C.J. Stafford. 1993. Environmental Contaminants in bald eagle eggs – 1980-84- and further interpretations of relationships to productivity and shell thickness. *Archives of Environmental Contamination and Toxicology.* 24:213-227.

Wiemeyer, S.N., T.G. Lamont, C.M. Bunck, C.R. Sindelar, F.J. Gramlich, J.D. Fraser, and M.A. Byrd. Organochlorine pesticide, polychlorobiphenyl, and mercury residues in bald eagle eggs-1969-79-and their relations to shell thinning and reproduction. *Archives of Environmental Contamination and Toxicology.* 13:529-549.

Woodford, J.E., W.H. Karasov, M.W. Meyer, and L. Chambers. 1998. Impact of 2,3,7,8-TCDD exposure on survival growth, and behavior of Ospreys breeding in Wisconsin, USA. *Environmental Toxicology and Chemistry.* Vol. 17(7):1323-1331.

Wren, C.D. 1991. Cause and effect linkages between chemicals and populations of mink (*Mustela vison*) and otter (*Lutra canadensis*) in the Great Lakes Basis. *J. Toxicol. Environ. Health.* 33: 549-585.

Wren, C.J., D.B. Hunter, J.F. Leatherland, and P.M. Stokes. 1987. The effects of polychlorinated biphenyls and methylmercury, singly and in combination on mink. II: Reproduction and kit development. *Arch. Environ. Contam. Toxicol.* Vol. 16, pp 449-454.

Yuan, Z,X., S. Kumar and H. C. Sikka. 1997. Comparative metabolism of benzo(a)pyrene by liver microsomes of channel catfish and brown bullhead. *Environ. Toxicol. Chem.* 16(4): 835-836.

Wright, P.J., and D.E. Tillitt. 1999. Embryotoxicity of Great Lakes lake trout extracts to developing rainbow trout. *Aquatic Toxicology.* 000:1-16.

Yamashita, N., S. Tanabe, J.P. Ludwig, H. Kurita, M.E. Ludwig, and R. Tatsukawa. 1993. Embryonic abnormalities and organochlorine contamination in double-crested cormorants

(*Phalacrocorax auritus*) and Caspan Terns (*Hydroprogne caspia*) from the upper Great Lakes in 1988. *Environ. Pollution.* 79:163-173.

Zabel, E.W., P.M. Cook, and R.E. Peterson. 1995. Potency of 3,3',4,4',5-pentachlorobiphenyl (PCB 126), alone and in combination with 2,3,7,8-tetrachlorodibenzo-p-dioxin (TCDD), to produce lake trout early life-stage mortality. *Environmental Toxicolgy and Chemistry.* 14(12):2175-2179.

Zabel, E.W., R.E. Peterson. 1996. TCDD-like activity of 2,3,6,7-tetrachloroxanthene in rainbow trout early life stages and in a rainbow trout gonadal cell line (RTG-2). *Environmental Toxicology and Chemistry.* 15:12, pp 2305-2309.