PHASE 2 REPORT
FURTHER SITE CHARACTERIZATION AND ANALYSIS
VOLUME 2E - REVISED BASELINE ECOLOGICAL RISK ASSESSMENT
HUDSON RIVER PCBs REASSESSMENT RI/FS

NOVEMBER 2000

For

U.S. Environmental Protection Agency
Region 2
and
U.S. Army Corps of Engineers
Kansas City District

Book 1 of 2
Text

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

305873

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

November 29, 2000

To All Interested Parties:

The U.S. Environmental Protection Agency (USEPA) is pleased to release the Revised Baseline Ecological Risk Assessment (Revised 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 Revised ERA evaluates current and future risks to ecological receptors posed by PCBs in the Hudson River in the absence of remediation of PCBs in sediments of the Upper Hudson River. The Revised ERA shows that risks to fish-eating birds and mammals are above USEPA's levels of concern.

On June 1-2, 2000, USEPA, through its contractor, Eastern Research Group (ERG), convened a panel of independent scientific experts to conduct a peer review of the August 1999 Baseline Ecological Risk Assessment for the Upper Hudson River and the March 2000 Responsiveness Summary for that report. In conjunction with this Revised ERA, USEPA is issuing a Response to Peer Review Comments on the Ecological Risk Assessment. The November 2000 Response to Peer Review Comments describes how USEPA incorporated the peer review comments or provides the technical rationale for not incorporating a comment.

The Revised ERA combines into a single report the August 1999 ERA, the March 2000 Responsiveness Summary, and the November 2000 Response to Peer Review Comments. The Revised ERA also includes revisions to the December 1999 ERA for Future Risks in the Lower Hudson River and August 2000 Responsiveness Summary for that report. USEPA is using the results of the Revised ERA to establish acceptable PCB exposure levels for ecological receptors, which will in turn be used to develop remedial alternatives for the PCBs in the sediments of the Upper Hudson River.

If you need additional information regarding the Revised ERA or the Reassessment RI/FS, please contact Ann Rychlenski at 212-637-3672.

Sincerely yours,

William M. [illegible]

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

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Revised Ecological Risk Assessment
Hudson River PCBs Reassessment
Executive Summary
November 2000

This document presents the Revised Baseline Ecological Risk Assessment (Revised 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 Revised 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 Lower Hudson River (Federal Dam to the Battery in New York City) posed by polychlorinated biphenyls (PCBs) in the absence of remediation. 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 interim baseline risk assessment.

A baseline ERA for current and future risks in the Upper Hudson River and future risks in the Lower Hudson River was issued by USEPA in August 1999 (USEPA, 1999c) and an associated Responsiveness Summary was issued in March 2000 (USEPA, 2000b). An ERA for Future Risks in the Lower Hudson was issued in December 1999 (USEPA, 1999e) and a Responsiveness Summary followed in August 2000 (USEPA, 2000c). On June 1-2, 2000, USEPA, through its contractor, Eastern Research Group (ERG), convened a panel of independent scientific experts to conduct a peer review of the baseline Ecological Assessment (ERA) for the Hudson River PCBs Site, consistent with the Agency's Peer Review Handbook (USEPA, 1998a). Based on comments received during the Peer Review, USEPA is issuing this Revised ERA in conjunction with a Response to the Peer Review Comments. The Revised ERA, in addition to incorporating to Peer Review comments, has been modified to incorporate all previous ERA reports into one report and update data, as appropriate.

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) the exposure assessment, which describes concentrations or dietary doses of contaminants of concern to which the selected receptors are or may be exposed; 6) 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 7) 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 Revised ERA indicates that PCBs in the Hudson River generally exceed levels that have been shown to cause adverse ecological effects in piscivorous birds and mammals, and that those levels will continue to be exceeded in the Upper Hudson through 2018[1] (the entire forecast period). Piscivorous birds and mammals are also at risk, to a lesser extent, in the Lower Hudson River. The results of the Revised 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 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 nearly 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

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[1]The 25-year forecast period is appropriate for the ERA based on receptor life spans.

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 the actual environmental values that are to be protected, operationally defined by an ecological entity and its attributes. 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:

- Sustainability of a benthic community structure, which is a food source for local fish and wildlife
- Sustainability (*i.e.*, survival, growth, and reproduction) of local fish (forage, omnivorous, and piscivorous) populations
- Sustainability (*i.e.*, survival, growth, and reproduction) of local insectivorous bird populations
- Sustainability (*i.e.*, survival, growth, and reproduction) of local waterfowl populations
- Sustainability (*i.e.*, survival, growth, and reproduction) of local piscivorous bird populations
- Sustainability (*i.e.*, survival, growth, and reproduction) of local insectivorous mammal populations
- Sustainability (*i.e.*, survival, growth, and reproduction) of local omnivorous mammal populations, and
- Sustainability (*i.e.*, survival, growth, and reproduction) of local piscivorous and semi-piscivorous mammal populations.

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 Revised 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) including a probabilistic dose-response analysis for selected receptors;

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.

Representative Receptors

The risks to the environment were evaluated for receptors that were selected to be representative of various feeding preferences, predatory levels, and habitats (aquatic, wetland, shoreline). Individual assessment endpoints are evaluated with at a minimum of one "model" (receptor) species. The following receptors were selected for the Revised ERA:

<u>Aquatic Invertebrates</u>

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

<u>Fish</u>

- 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*)

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 assessment endpoint 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, January 2000 Revised Baseline Modeling Report, and associated responsiveness summaries). 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. 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.

To integrate the various components of the ERA, the results of the risk characterization and associated uncertainties were evaluated to assess the risk of adverse effects in the receptors of concern as a result of exposure to PCBs originating in the Hudson River. This approach considers both the results of the TQ analysis and field observations for each assessment endpoint.

Sustainability of a Benthic Community Structure, Which Serves as a Food Source For Local Fish and Wildlife

Benthic community structure as a food source for local fish populations was assessed using three lines of evidence. Overall, there was no strong evidence of adverse effects due to PCBs at the community level.

Sustainability (*i.e.*, Survival, Growth, and Reproduction) of Local Fish (Forage, Omnivorous, and Piscivorous) Populations

Risks to local fish populations were evaluated using seven lines of evidence. Collectively, they indicate that current (1993) and future PCB exposures may reduce or impair the survival, growth, and reproductive capability of resident omnivorous (*e.g.*, brown bullhead) and piscivorous fish (*e.g.*, largemouth bass) in the Upper Hudson River and piscivorous fish (*e.g.*, largemouth bass, striped bass) 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. Fish in the Lower Hudson River showed limited exceedance at current levels. Future body burdens in fish on total PCB (Tri+) basis 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. Concentrations on a lipid-normalized TEQ basis showed fewer exceedances. There is a moderate degree of uncertainty in the modeled body burdens used to evaluate exposure. The lower river modeling results are considered to have a greater degree of uncertainty than the upper river results.

Measured and modeled concentrations of PCBs in river water and sediment in the Upper Hudson River and 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 some criteria and guidelines for protection of fish; however, fewer sediment guidelines or water criteria/guidelines are exceeded in the lower river than the upper river during the modeling period (1993 - 2018).

Sustainability (*i.e.*, Survival, Growth, and Reproduction) of Local Insectivorous Bird Populations

Risks to insectivorous birds, using the tree swallow as a model, were evaluated using six lines of evidence. Collectively, they indicate that current and future concentrations of PCBs are not of a sufficient magnitude to impair 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. PCB concentrations detected in tree swallow samples were significantly higher than concentrations known to cause reproductive and developmental impairment in other birds. 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 and Lower Hudson River water exceed criteria and guidelines developed for the protection of wildlife through 2018 (the entire forecast period).

Sustainability (*i.e.*, Survival, Growth, and Reproduction) of Local Waterfowl Populations

Risks to waterfowl, using the mallard duck as a model, were evaluated using six lines of evidence. Collectively, they indicate that current and future concentrations of PCBs are not of a sufficient magnitude to impair reproduction of waterfowl, but modeled dietary doses and egg concentrations under current and future conditions exceed some benchmarks.

Calculated dietary doses of PCBs and concentrations of PCBs in eggs based on 1993 data typically did not exceed their respective TRVs, except at Stillwater (RM 168). TQs for the dioxin-like PCBs are consistently higher than TQs for total PCBs and exceed one at most locations for both the body burden and egg concentrations. There is a moderate degree of uncertainty in the dietary dose and egg concentration estimates.

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

The large number of mallards observed along the Hudson River indicate that mallard populations are stable along the river.

Sustainability (*i.e.*, Survival, Growth, and Reproduction) of Local Piscivorous Bird Populations

Risks to piscivorous birds, using the belted kingfisher, great blue heron, and bald eagle as models, were evaluated using six lines of evidence. Collectively, they indicate that current and future concentrations of PCBs may reduce or impair the survival, growth, and reproductive capability of piscivorous birds in the Upper and Lower Hudson River. Calculated concentrations of total PCBs in eggs exceed most TRVs for the Upper and Lower Hudson River through 2018 (the entire forecast period). On a TEQ basis all calculated body burden and egg concentrations of the bald eagle exceeded TRVs for the duration of the modeling period, as did the majority of the belted kingfisher and great blue heron exposures. 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.

The probabilistic dose response analysis showed female eagles at RM 189 show approximately a 45% probability of experiencing at least a 50% reduction in fecundity in 1993 going down to about a 10% reduction in fecundity in 2015. Female eagles at RM 168 in 1993 show approximately a 30% probability of experiencing a 20% reduction in fecundity, which decrease to low probabilities (<10%) of experiencing small reductions (<5%) in fecundity by 2015. Female kingfishers showed similar results.

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

The bald eagle is on both federal and NY State lists of threatened and endangered species. Therefore, individual (rather than population) level effects could adversely affect the Hudson River populations. Based on the results in this report, Hudson River bald eagles are considered to be at risk.

<u>Sustainability (*i.e.*, Survival, Growth, and Reproduction) of Local Insectivorous Mammal Populations</u>

Risks to insectivorous mammals, using the little brown bat as a model, were evaluated using four lines of evidence. Collectively, they indicate that current and future concentrations of PCBs may reduce or impair the survival, growth, and reproductive capability of insectivorous mammals in the Upper Hudson River. To a lesser degree, current and future 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 and lower river, particularly from the Thompson Island Pool to Stillwater. TRVs are exceeded for almost all comparisons for the duration of the modeling period (1993-2018) at all locations on a TEQ basis. There is a moderate degree of uncertainty in the calculated dietary doses.

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

<u>Sustainability (*i.e.*, Survival, Growth, and Reproduction) of Local Omnivorous Mammal Populations</u>

Risks to omnivorous mammals, using the raccoon as a model, were evaluated using four lines of evidence. Collectively, they indicate that current and future concentrations of PCBs may reduce or impair the survival, growth, and reproductive capability of individuals who feed extensively near the Upper Hudson River. To a lesser degree, current and future exposures may have similar adverse effects on omnivorous mammals in the Lower Hudson River. Modeled dietary doses for the raccoon exceed TRVs on a TEQ basis under current and future conditions in the Upper Hudson River, but only limited exceedances are seen (in the upper river) on a total PCB basis. There is a moderate degree of uncertainty in the calculated dietary doses.

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

Sustainability (*i.e.*, Survival, Growth, and Reproduction) of Local Piscivorous Mammal Populations

Risks to piscivorous mammals were evaluated using four lines of evidence. Collectively, they indicate that current and future concentrations of PCBs are of a sufficient magnitude to prevent the reproduction of piscivorous mammals in the Thompson Island Pool area and reduce or impair the survival, growth, and reproductive capability of mammals in the Upper and Lower Hudson River. Modeled dietary doses for the mink and river otter exceed all TRVs under current conditions on a total PCB and TEQ basis at all stations in the Upper and Lower Hudson River, with one exception. Measured PCBs in mink and otter liver also exceeded TRVs. Toxicity quotients were up to three orders of magnitude above one. Future modeled dietary doses of PCBs in mink exceeded all TRVs, with the exception of some of the LOAELs from RM 154 to RM 50 on a total PCB basis and the LOAEL at RM 154 after 2006 on a TEQ basis. Future modeled dietary doses of PCBs (total and TEQ basis) for the river otter exceeded all NOAEL and LOAEL comparisons (1993-2018) at all locations in the upper and lower river by up to three orders of magnitude. Given the magnitude of the majority of the TQs, they would have to decrease by an order of magnitude or more to fall below one. There is a moderate degree of uncertainty in the calculated dietary doses.

The probabilistic dose response analysis indicates that in 1993, female mink at RM 189 and 168 show a high probability (90 to 100%) of experiencing a severe reduction (>80%) in fecundity, and females at RM 154 still show a high probability (>95%) of experiencing at least a 50% reduction in fecundity. In 2015, mink at RM 189 still show a high probability (>95%) of experiencing substantially reduced (>50%) fecundity. River otters show even more severe effects. In 1993, female river otters at RM 189, 168 and 154 show high probabilities (80 to 100%) of experiencing severe decreases (>90%) in fecundity, in comparison to otters that are not exposed to PCBs. In the year 2015, female otters at RM 189 still show high probabilities (>70%) of experiencing severely reduced (100%) fecundity. River otters at RM 168 still show high probabilities (>80%) of experiencing a substantial decrease (>80%) in 2015, while otters at RM 154 show a 30% probability of experiencing at least a 50% reduction in fecundity.

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

The results of the point estimate toxicity quotients and probabilistic dose response analysis combined with field observations suggesting reduced mink and river otter populations in the upper river indicate that these animals are experiencing adverse effects at the population level, and that these effects are likely to persist into the future.

Uncertainty

At each step of the risk assessment process there are sources of uncertainty. Uncertainty exists because of lack of knowledge (*e.g.*, TRVs) and variability (*e.g.*, fish tissue concentrations). Quantifiable sources of uncertainty were included to the extent possible in sensitivity and uncertainty analyses. The results showed that even at the 5th percentile, predicted toxicity quotients for the bald eagle egg, belted kingfisher egg, mink and river otter did not fall below one for any location or year, except for mink at RM 154 in 2015.

Conclusions

The results of the risk assessment indicate that upper trophic level 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, avian and mammalian piscivorous receptors are at risk. In summary, the major findings of the report are:

- Piscivorous fish (*e.g.*, largemouth bass and striped bass) and omnivorous fish (*e.g.*, brown bullhead) in the Hudson River may be adversely affected (*i.e.*, reduced survival, growth, and/or reproduction) from exposure to PCBs. Forage fish are unlikely to be affected outside of the Thompson Island Pool.

- 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, may be adversely affected (*i.e.*, reduced survival, growth, and/or reproduction), particularly insectivorous mammals living in the Thompson Island Pool area.

- Waterfowl feeding on animals and plants in the Hudson River are unlikely to be adversely affected (*i.e.*, reduced survival, growth, and/or reproduction) from exposure to PCBs.

- Omnivorous animals, such as the raccoon, that derive a large portion of their food from the Hudson River may be adversely affected (*i.e.*, reduced survival, growth, and/or reproduction) from exposure to PCBs.

- 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 at the population level. PCBs may adversely affect the survival, growth, and reproduction of these species. Piscivorous mammals are at the greatest risk due to their feeding patterns.

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

- PCB concentrations in water and sediments in the Upper and Lower Hudson River generally exceed standards and criteria and guidelines established to be protective of the environment.

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

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Chapter I

1.0 INTRODUCTION

1.1 Purpose of Report

This report is part of the Phase 2 investigation of Hudson River 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 interim No Action decision with respect to the PCB-contaminated sediments in the Upper Hudson River. For purposes of the Reassessment RI/FS, 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 PCBs 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, and the Upper and Lower Hudson are shown in Figures 1-2 and 1-3, respectively.

In December 1990, USEPA issued a Scope of Work (SOW) for reassessing the interim No Action decision for the Hudson River PCBs 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 scheduled to be completed by the end of 2000. Six major reports have been released from the Phase 2 investigation, specifically:

(1) Volume 2A: Database Report - October 1995;

(2) Volume 2B: Preliminary Model Calibration Report - October 1996;

(3) Volume 2C: Data Evaluation and Interpretation Report (DEIR) - February 1997;

(3A) Volume 2C-A: Low Resolution Sediment Coring Report - July 1998;

(4) Volume 2D: Baseline Modeling Report (BMR) - May 1999 and Revised BMR January 2000;

(5) Volume 2E: Baseline Ecological Risk Assessment (ERA) - August 1999 and ERA Addendum December 1999; and

(6) Volume 2F: Upper Hudson River Human Health Risk Assessment (HHRA) - August 1999 and Mid-Hudson River Human Health Risk Assessment - December 1999.

The Responsiveness Summaries were released as follows: the first three volumes of the Phase 2 Report (Volumes 2A to 2C) - December 1998, Low Resolution Sediment Coring Report (Volume 2C-A) - February 1999, Baseline Modeling Report - February 2000 (Volume 2D), and ERA and HHRA (Volumes 2E and 2F) - March 2000, ERA and HHRA Addendums (Volumes 2E-A and 2F-A) - August 2000. The Database for the Hudson River PCBs Reassessment RI/FS was most recently updated in October 2000 (USEPA, 2000e).

The Comprehensive Environmental Response, Compensation, and Liability Act (CERCLA) 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) (USEPA, 1990) 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.

This Revised ERA is a revision of the Baseline ERA and ERA Addendum, based upon public comments and comments from the peer reviewers received during the ERA Peer Review conducted on June 1 and 2, 2000 (USEPA, 2000d). The baseline ERA and ERA Addendum followed USEPA guidance (USEPA, 1997b and USEPA, 1998b) and the Phase 2 Ecological Risk Assessment Scope of Work (ERASOW) (USEPA, 1998e) and Responsiveness Summary for the ERASOW (USEPA, 1999a), in which USEPA responded to all significant written comments received on the ERASOW. This Revised ERA incorporates all Hudson River Reassessment RI/FS documents related to the ERA (*i.e.*, USEPA, 1998e, 1999a, 1999c, 1999e, 2000b, 2000d, and 2000e) to provide an integrated and updated document. This Revised ERA also follows USEPA guidance (USEPA, 1997b and USEPA, 1998b).

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 were restricted under provisions of the Toxic Substances and Control Act (TSCA).

In addition to direct discharges from the two capacitor production plants, GE may have indirectly contributed additional PCBs to the watershed and ultimately to the river as a result of its practice of disposing manufacturing wastes in nearby landfills and possibly wastewater collection systems (sewers and municipal wastewater treatment plants). More recently, additional discharge of PCBs into the Hudson River continues to occur as a consequence of migration of PCBs from the overburden or bedrock at GE's Hudson Falls and Fort Edward plants and adjoining areas.

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 was stored in relatively quiescent areas of the river. These areas, which were surveyed by the New York State Department of Environmental Conservation (NYSDEC) in 1976-1978 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 were 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, higher PCB concentrations were detected in Hudson River water. The higher levels have been attributed 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).

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 detailed evaluation of the Waterford Water Works to see if an upgrade or alterations to the facilities were needed to the public water supply.

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

1.2.1 Summary of PCB Sources to the Upper and Lower Hudson River

External PCB sources, including the GE plants, the remnant deposits, and other sources in both the Upper and Lower Hudson River, are discussed in the Phase 2 Data Evaluation and Interpretation Report (DEIR) (USEPA, 1997a).

Other sources of PCBs have also existed within the Upper Hudson River valley. These include electric utilities and manufacturers who may have purchased equipment containing PCBs, paper mills (from paper production as well as from electrical equipment), other industries, transportation sources, and electrical component scavengers. In addition to these more-or-less direct inputs of PCBs, the Upper Hudson is also being affected by redistribution of earlier discharges; landfilling of dredged material or contaminated soil is an example of a modified PCB source derived from historical releases. Also, PCBs were historically introduced throughout New York State by paper mills recycling carbonless copy paper (also known as NCR paper) which contained Aroclor 1242. The total discharge of PCBs during 1977 and 1978 from all recycle mills in New York State was estimated at a maximum of 20 kg/year (45 lb/year), with less than 2.3 kg/year (5 lb/year) to the Hudson River from Bakers Falls to Troy (NYSDEC, 1978). This is, however, an insignificant amount compared to GE's estimated 14 kg/day (30 lb/day) or 5,000 kg/year (11,000 lb/year) discharges at Fort Edward and Hudson Falls during the early 1970s (Tofflemire and Quinn, 1979).

The DEIR (USEPA, 1997a) identified PCB-contaminated sites near the Upper Hudson River, including riverbank sediments (remnant deposits), dredge spoil areas, industrial sites, dump sites, and municipal landfills. These include the following sites on NYSDEC's Registry of Inactive Hazardous Waste Disposal Sites:

- Niagara Mohawk Power Corporation – Queensbury
- General Electric Company - Hudson Falls Plant and Vicinity
- General Electric Company - Fort Edward Plant and Vicinity
- Moreau Landfill
- Kingsbury and Fort Edward Municipal Landfills

Other "dump sites" include South Glens Falls Dragstrip, GE Moreau (formerly Caputo Dump), West Glens Falls Containment Site, and Old Fort Edward Landfill. These sites have either been remediated or are currently under remediation and do not represent potential loadings of PCBs to the Hudson River, or insufficient data currently exist to estimate impacts to the Hudson River.

Other sources of PCBs include GE's remnant deposits, New York State Department of Transportation's (NYSDOT) dredge spoil sites, tributaries to the Hudson River, point sources to the NY/NJ Harbor in the lower river (*e.g.*, sewage treatment plant influent and effluent), and sources that are not directly measured (*e.g.*, stormwater, atmospheric deposition, and leachate). However, the area of the site upstream of the Thompson Island Dam (*i.e.*, the source considered in this assessment) represents the primary source of PCBs to the freshwater Hudson, as described in the next section.

1.2.2 Summary of Phase 2 Geochemical Analyses

As a result of its geochemical analyses, USEPA concluded that the sediments of the Thompson Island Pool strongly impact the water column, generating a significant PCB load to the water column whose congener pattern can often be seen throughout the Upper Hudson. Using side-scan sonar geophysical techniques, USEPA also found a number of areas of cohesive sediment at locations that closely resemble the *hot spot* areas defined previously by NYSDEC. These *hot spot*-related sediments appear to be intact despite the time between the NYSDEC studies and USEPA's Phase 2 investigation. Given the strong link between PCBs in sediment and water, the large inventory of PCBs in the Upper Hudson, and the apparent lack of substantial reduction in PCB concentrations via *in situ* degradation, it is unlikely that the PCB levels in the water column downstream of the Thompson Island Dam will substantially decline beyond current levels until the active sediments are depleted of their PCB inventory or remediated.

The decrease in PCB inventories in the more contaminated sediments of the Thompson Island Pool and from several of the studied *hot spots* below the Thompson Island Dam, along with the indication of an inventory gain in the coarse sediments of the Thompson Island Pool, indicate that PCBs are being redistributed within the Hudson River system. These results show that the stability of the sediment deposits cannot be assured.

Burial of contaminated sediment by cleaner material is not occurring universally. Burial of more PCB-contaminated sediment by less contaminated sediment has occurred at limited locations, while significant portions of the PCB inventories at other *hot spots* have been re-released to the environment. It is likely that PCBs will continue to be released from Upper Hudson River sediments.

Patterns of contamination found throughout the Hudson all contain the "finger print" of GE-related contamination. In the freshwater Hudson, GE-related contamination represents 80 to 100 percent of the in-place and water-borne contamination. In the Upper Hudson, this percentage is quite close to 100 percent. In the saline Hudson, GE-related contamination represents perhaps 50 percent of the in-place and recently deposited PCB inventory.

1.2.3 Extent of Contamination in the Upper Hudson River

This section summarizes the current conditions of the Upper Hudson River with respect to PCB contamination of the sediment, water, and fish. Sixteen years after USEPA's interim No Action decision, PCB concentrations remain elevated in the Hudson River in all three environmental media. Concentrations generally decrease with distance down river, away from the original source areas of the GE Hudson Falls and Fort Edward plants. While some changes have occurred during this period, in general, conditions have not improved substantially from about 1995 to the present.

1.2.3.1 PCBs in Sediment

Areas of elevated sediment concentrations, *i.e.*, *hot spots*, are found in depositional areas throughout the Upper Hudson River. This section discusses the extent of PCBs in sediments as characterized by the NYSDEC 1976-1978 Sediment Survey, the NYSDEC 1984 Sediment Survey, the General Electric 1991 Sediment Composite Survey, the USEPA 1992 High Resolution Sediment Coring Program, the USEPA 1994 Low Resolution Sediment

Coring Program, the General Electric 1998 Sediment Composite Survey, and the General Electric 1998-1999 Sediment Coring Program.

The Thompson Island (TI) Pool (RM194.6-188.5) contains 20 of the 40 *hot spots* identified by NYSDEC in 1977 and 1984 (Brown *et al.*, 1988) and Malcolm Pirnie (MPI, 1992). The sediments exhibit a high degree of heterogeneity with respect to the distribution of PCBs. Historically, the highest sediment concentrations have been observed within the cohesive sediments of the TI Pool and are generally lower within the non-cohesive sediments. The maximum concentration of PCBs measured was approximately 2000 mg/kg. The average concentration of PCBs in surficial sediments (0-25 cm) in 1991 for the area between RM 186 and RM 194 was 42 mg/kg.

Thompson Island Dam to Northumberland Dam near Lock 5 (RM 188.5-183.4) contains 15 of the 40 NYSDEC-defined *hot spots*. The maximum concentration of PCBs found in the Hudson River, approximately 4,000 mg/kg, was measured in a thin section of core from Hot Spot 28 in this section of the river. The average concentration of PCBs in surficial sediments (0-25 cm) in 1991 for this section of the river was approximately 26 mg/kg.

Northumberland Dam to Federal Dam at Troy (RM 183.4-153.9) contains 5 of the 40 NYSDEC-defined *hot spots*. The average concentration of PCBs in surficial sediments (0-25 cm) in 1991 for this section of the river was approximately 9 mg/kg.

1.2.3.2 PCBs in the Water Column

The dominant sources of PCB load to the water column of the Upper Hudson River may be separated into two groups: (1) PCBs-contaminated oil in bedrock seeps from the GE Hudson Falls plant and other discharges upstream of Rogers Island; and (2) PCB-contaminated sediments that accumulated behind the former Fort Edward Dam and were remobilized and transported downstream. The sediments of the TI Pool are the major source of PCBs to the water column during low flow conditions from May to October, which includes the period of greatest biological activity.

USGS monitoring of PCBs in the water of the Upper Hudson River began in 1977. In the Thompson Island Pool, the data of PCB concentrations in water indicate significant gains in PCB load. The concentrations may be converted to load estimates by integration with the flow series, using a ratio estimator. The PCB load from the TI Dam-West sampling station above TI Dam for the period of January 1998 to March 2000 is estimated to be approximately 1.03 kg/day. Estimating load gain across the TI Pool as the difference in loads at Rogers Island and TI Dam-West yields an estimate for this time period of a gain of 0.86 kg/day. During this same period, approximately 0.07 kg/day total PCB load derived from upstream of Bakers Falls, and about 0.10 kg/day from the Bakers Falls area. The recent rate of apparent load gain across the TI Pool is higher than the estimated load gain over the period of record from April 1991 to March 2000 of 0.81 kg/day, indicating that PCB load continues to be generated from the TI Pool at an approximately constant rate.

Samples collected at the TID-West station above the TI Dam are believed to be higher than the PCB concentrations that are actually transported across the Dam in the center channel due to reduced lateral mixing. PCB concentrations in the channel appear to be on the order of 50 to 80 percent of the TID-West concentrations. After adjusting for this potential bias, the load generated from the TI Pool still is on the order of 0.5 to 0.7 kg/day, and represents the main source of PCB load present at the TI Dam.

During the summer of 1998 (June-September), the average concentration at TI Dam West station was 134 ng/L. Concentrations from January 1996 through March 2000 averaged 90 ng/L. Five observations in excess of 300 ng/L were noted during the winter of 1999-2000.

In recent years, GE has also resumed monitoring at the Route 29 bridge in Schuylerville. The average concentration at Schuylerville during the summer of 1998 (June-September) was 80.4 ng/L. From August 1997 to May 2000, the concentrations averaged 75.6 ng/L. PCB concentrations in the water column below Schuylerville tend to reflect the same loads present at Schuylerville, with a reduction in concentration associated with tributary dilution.

Evaluation of these data (USEPA, 1997a) indicates that annual PCB loads at Stillwater (reflecting all upstream sources) were approximately 3,000 kg/yr in 1977-79, and 1000 kg/yr in 1980-84, then declined to about 200 kg/yr by 1991. From 1980 to 1991, the upstream loads at Rogers Island appear to have declined from about 500 kg/yr to less than 200 kg/yr. The declining trend in loads at Stillwater primarily reflects the washout of readily erodible PCB-contaminated sediments left by the removal of the Fort Edward Dam and shows a gradual increase in the relative importance of sources upstream of Rogers Island.

1.2.3.3 PCBs in Fish

PCB concentrations observed in fish are a result of exposure to PCBs in water and surface sediment, through either an aquatic food chain or a benthic food chain, respectively. Because biota integrate exposures over time, they provide a time-averaged indicator of trends in exposure concentrations.

NYSDEC continues to collect and analyze fish tissue data from many locations in the Upper Hudson River (Table 1-1). Converted to a Tri+ PCB basis (trichlorinated and higher congeners represent total PCBs in biota, discussed in Section 1.4), the concentrations in the TI Pool in 1998 averaged about 28.6 mg/kg (wet weight) in carp, and about 16.1 mg/kg (wet weight) in largemouth bass. The maximum PCB concentrations measured were 83.2 mg/kg (wet weight) in carp, and 40.4 mg/kg (wet weight) in largemouth bass. Concentrations at Stillwater averaged about 41.3 mg/kg (wet weight) in carp and 6.9 mg/kg (wet weight) in largemouth bass and the maximum concentrations measured were 105.9 mg/kg (wet weight) in carp and 32.3 mg/kg (wet weight) in largemouth bass.

Because PCBs tend to accumulate in fatty tissues, it is also useful to examine concentrations on a lipid basis, as shown in Table 1-2. The lipid-based Tri+ concentrations for 1998 are generally similar to those observed from 1995 to 1997 in both the TI Pool and the Stillwater/Coveville reach, with little evidence for a consistent decline. In particular, the largemouth bass results appear to have been nearly stable throughout the 1990s.

The PCB principal components analysis contained in the Baseline ERA has shown that fish body burdens decline with river mile to about the same degree as the changes in the PCB concentration in sediment (USEPA, 1999c, Appendix K). Similarly, the average molecular weight of the PCB body burden 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 and not to trophic level.

Table 1-3 summarizes half-life data for the three species discussed above, plus yellow perch. For example, based on consistent Tri+ PCB data for 1995 through 1998 or 1999, the half-life for brown bullhead in the TI Pool is 50 years, and the half-lives for largemouth bass and pumpkinseed are increasing. In the Stillwater reach the half-life of brown bullhead is increasing, the half-life of largemouth bass is about 42 years, and the half-life of pumpkinseed is about 2.8 years.

The consistent Tri+ PCB data include both Aroclor-based data reported by NYSDEC and direct estimates of Tri+ from homologue-based analyses from NEA Laboratories that are included in the NYSDEC database. In addition to the consistent PCB Tri+ data, Table 1-2 also includes the trends from NYSDEC-reported lipid-based total PCBs (NYSDEC-collected data only) and Aroclor 1254 concentrations without correction to a consistent Tri+ basis. These data are included for comparison; however, it is believed that analytical changes in 1990 and 1992 may distort the interpretation of trends.

1.3 Data Sources

PCB contamination in the Hudson River has been examined in many studies over the last couple of decades (*e.g.*, Normandeau Associates, 1997; Malcolm Pirnie, 1978; O'Brien and Gere, 1993; and Exponent, 1998a and 1998b). These studies have identified areas of the river with large PCB deposits, examined PCB concentrations in fish and invertebrates, investigated the historical deposition of PCBs, and evaluated various remedial options to address the PCBs. The data that were selected for use in the Revised ERA (Figure 1-4) and the rationale behind their selection are described below. PCB data used in this report are contained in the Database for the Hudson River PCBs Reassessment RI/FS Release 5.0 (USEPA, 2000e).

The Revised ERA relies primarily upon USEPA data collected during the Phase 2 sampling, which was conducted specifically to obtain data to be used in the Reassessment reports. Although many other studies have been performed, data from the Phase 2 program are used as the preferred data set because:

- The Phase 2 data is exhaustive, providing information on both the Upper and Lower Hudson River;
- Samples in all matrices (*i.e.*, sediment, water, fish, and invertebrates) were analyzed for PCBs;
- PCBs were analyzed at the congener-specific level by the same laboratory for all matrices (*i.e.*, sediment, water, fish, and invertebrates);
- Samples were collocated (to the maximum practical extent) to provide an overall picture of PCB distribution; and

- Data were validated under protocols developed specifically for this project (*e.g.*, see Appendix I of USEPA 1999c).

The Phase 2 ecological sampling program was conducted in August 1993 to obtain data for this assessment. This 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 (see Figures 1-2 and 1-3). Benthic invertebrates were identified and counted to provide data for a community-level analysis. Fish analyzed for the risk assessment were collected by NYSDEC and the National Oceanic and Atmospheric Administration (NOAA), who also provided 1995 fish data that are used in this report. Detailed descriptions of the sampling stations and the ecological field sampling effort are provided in Appendices A and B of USEPA 1999c, respectively.

Data from the DEIR (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) are 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.

When there was not enough information available from the Phase 2 study to characterize a medium, other data sources, such as the NYSDEC fish database were used, as described below.

- **NYSDEC/NOAA Data** - NYSDEC and NOAA collected resident fish at 16 of the ecological sampling stations (3-10 fish per location) in 1993 for PCB congener-specific analysis. In 1995, NOAA conducted an additional study (3-5 fish per location) to build on the congener data and the historical database for resident fish established in the 1993 study (NOAA, 1997). Data from both 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.

- **USFWS Data** - The United States Fish and Wildlife Service (USFWS) has conducted a detailed study on PCB congener concentrations in tree swallows breeding in the Upper Hudson River (McCarty and Secord, 1999a; McCarty and Secord, 1999b; and USFWS, 1997). USFWS tree swallow, mallard, wood duck, and eagle data (some collected in conjunction with NYSDEC) are included in this report.

- **NYSDOH Data** - The FISHRAND bioaccumulation food chain model used data on water column invertebrates from New York State Department of Health (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.

- **General Electric 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, 1998b). This study was used in conjunction with other research to characterize dietary preferences for the fish receptors examined in this report. Vegetation mapping in Thompson Island Pool performed by GE (Exponent, 1998a) was considered in the habitat characterization of that area.

1.4 Technical Approach and Ecological Assessment in the Superfund Process

The Revised ERA is part of a focused evaluation directed specifically at reassessing the interim No Action decision related to the presence of PCBs in Hudson River sediments. This reassessment is required under the CERCLA provision for five-year reviews for remedial actions at sites where hazardous substances, pollutants, or contaminants remain on-site above levels that allow for unlimited use and unrestricted exposure. The Reassessment RI/FS was initiated in 1989, prior to the issuance of Agency guidance on ecological risk assessment. The Ecological Risk Assessment Guidance for Superfund was published in 1997 (USEPA, 1997b) and Guidelines for Ecological Risk Assessment was published in 1998 (USEPA, 1998b). The conceptual approach used in the Revised ERA is consistent with available guidance and an effort was made to incorporate guidance as it became available. The approach relied on the input of a number of affected and interested parties to help define the problem, consistent with what is currently referred to as Problem Formulation, and to develop a scope of study.

As part of reassessing the interim 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 PCBs in the sediments?

The primary objective of this Revised ERA is to answer these questions in order to support the needs of the Reassessment RI/FS. Because of the focused nature of the Reassessment RI/FS, a number of technical decisions were made which serve to structure and focus the Revised 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 NYSDOH. Technical issues were also discussed with representatives of GE. 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 vary temporally. 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" (*i.e.* 1993) exposures are characterized by existing data found in Database Release 5.0 (USEPA, 2000e). "Future" (*i.e.*, 1993-2018) exposures are characterized by the HUDTOX model for water and sediment, and FISHRAND for invertebrates and fish (USEPA, 2000a).

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 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, 2000a) 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 Revised 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 1-5. 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; however, the

Revised ERA includes data available through the first part of 2000 collected by other agencies (*i.e.*, NYSDEC, USFWS).

A revised Scope of Work was issued in 1998 (USEPA, 1998e) 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, 1999a). This Revised ERA focuses on 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.

Due to the nature of the available data, the ecological risk assessment for the Hudson River follows a deterministic risk evaluation with a probabilistic evaluation used to evaluate the sensitivity of key parameters.

In keeping with ERAGS, the format of this Revised 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 site characterization, contaminants of concern (COCs), conceptual model, assessment and measurement endpoints, and representative receptors.

- Chapter 3, the Exposure Assessment, discusses observed and modeled PCB concentrations (based on the results of the RBMR), 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.

This report is presented in two books that address potential current and future ecological risks in the Upper and Lower Hudson River. Book 1 contains the report text and Book 2 presents the tables and figures. Appendices to these books were released as part of the August 1999 ERA and December 1999 ERA Addendum. Since the few changes made to the appendices are incorporated into the first two books of this report, the appendices contained in USEPA 1999c and 1999e are not being reissued with this report. A list of the appendices in those documents, which are considered part of this Revised ERA follows:

August 1999 ERA:

APPENDIX A	Site Description and Characterization
APPENDIX B	Ecological Field Sampling Program
APPENDIX C	Life History and Ecology of Dominant Macroinvertebrate Receptors
APPENDIX D	Life History and Ecology of Fish Receptors
APPENDIX E	Life History and Ecology of Avian Receptors
APPENDIX F	Life History and Ecology of Mammalian Receptors
APPENDIX G	Threatened, Endangered and Special Concern Species
APPENDIX H	Benthic Macroinvertebrate Community Analysis
APPENDIX I	Data Usability Report for PCB Congeners Ecological Study
APPENDIX J	Data Supporting TEQ Analysis
APPENDIX K	Examination of Exposure Pathways Based on Congener Patterns

December 1999 ERA Addendum:

APPENDIX A - Conversion from Tri+ PCB Loads to Dichloro through Hexachloro Homologue Loads at the Federal Dam

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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 1-5. 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 PCBs 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 1998 ERA SOW that was reviewed by agencies, GE, and other interested parties. While not formally referred to as Problem Formulation, most of the issues that are considered Step 3 of an assessment were discussed with various agency personnel, other agencies (*e.g.*, NOAA, USFWS, NYSDEC), and GE. 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 Site is defined as the nearly 200 miles (322 km) of river from Hudson Falls to the Battery in New York Harbor. The Upper Hudson River, in the context of the Reassessment RI/FS and this baseline ERA, is the 40-mile (64-km) stretch from Hudson Falls to Federal Dam (Figure 1-2). The Lower Hudson River extends from Federal Dam to the Battery (Figure 1-3) 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) (about RM188.5-195) stretch of the Upper Hudson River (*e.g.*, USEPA, 1998c; 1997a). Several tributaries, including Snook Kill and Moses Kill, enter the Hudson River at 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 (*e.g.*, significant habitats) and threatened and endangered species found in the Hudson River are found in the lower river. The following sections describe the habitats, fauna, threatened and endangered species, and significant habitats in the Hudson River. Plate 1 provides detailed wetland habitat maps and bathymetry of the Hudson River. Information shown on these maps is taken from National Wetland Inventory (NWI), NYSDEC wetlands (based on 6 NYCRR Part 664), and NOAA nautical charts. Wetland descriptions of the NWI and NYSDEC classifications can be found in Cowardin *et al.* (1997) and NYSDEC (1980), respectively. Tables 2-1 to 2-6 contain species lists (common and scientific names) of fish, amphibians, reptiles, breeding birds, and mammals of the Hudson River.

2.1.1 Habitat Descriptions

The following habitat descriptions are based primarily on *Ecological Communities of New York State* (Reschke, 1990). Primary groups of organisms and environmental characteristics are used as an index to habitat conditions (Reschke, 1990). For estuarine, palustrine, and terrestrial systems vegetation is used as the primary group of organisms, while for riverine systems fish are used as the primary group of organisms. Plate 1 shows National Wetland Inventory (NWI) and NYSDEC wetlands along the river along with the bathymetry. The upper river can be found on sheets 1-4 of Plate 1 and the lower river is on sheets 4-16 of Plate 1.

2.1.1.1 Upper Hudson River Habitats

Main Channel Stream – is the aquatic community of a large, quiet, base level sections of streams where there are no distinct riffles. Main channel streams usually have clearly distinguished meanders. They are characterized by considerable deposition, with a relatively minor amount of erosion. Although the middle of the main channel is too deep for aquatic macrophytes to occur, the shallow shores and backwaters typically have rooted macrophytes. Mosses in the genus *Fontinalis* are characteristic of shallow areas. Two exotic weeds, Eurasian milfoil (*Myriophyllum spicatum)* and water chestnut *(Trapa natans*) are common along shores and backwaters. Characteristic fishes are deep-bodied fishes such as suckers (Catostomids) and shad and warmwater fishes such as pickerel, northern pike, smallmouth bass, and largemouth bass.

Riverine Cultural Subsystems- this grouping includes communities that are either created and maintained by human activities, or are modified by human influence to such a degree that the stream flow, morphology, water chemistry, or the biological composition of the resident community is substantially different from the character of the substrate or community as it existed prior to human influence. Cultural riverine communities found in the Upper Hudson River include acidified streams and canals.

Palustrine System – the palustrine system consists of non-tidal perennial wetlands characterized by emergent vegetation. Palustrine subsystems found along the upper river include deep emergent marsh and shallow emergent marsh, as described below.

Deep Emergent Marsh – a marsh community that occurs on mineral soils or fine-grained organic soils; the substrate is flooded by waters that are not subject to violent wave action. Water depths can range from 6 in to 6.6 ft (15 cm to 2 m); water levels may fluctuate seasonally but the substrate is rarely dry, and there is usually standing water in the fall. Characteristic vegetation include emergent aquatics such as yellow pond-lily (*Nuphar luteum*), white water-lily (*Nymphaea odorata),* cattails (*Typha latifolia, T. angustifolia*), soft-stem bulrush (*Scirpus tabernaemontanii*), hard-stem bulrush (*S. acutus*), bur-reed (*Sparganium eurycarpum*), arrow arum (*Peltandra virginica*), and wild rice (*Zizania aquatica*). Marshes that have been disturbed are frequently dominated by aggressive weedy species such as purple loosestrife (*Lythrum salicaria*) and common reed (*Phragmites australis*).

Characteristic animals include American bittern, least bittern, red-winged blackbird, marsh wren, Virginia rail, pied-billed grebe, bullfrog, and painted turtle.

Shallow Emergent Marsh – a marsh community that occurs on mineral soils or fine-grained organic soils. This marsh is better drained than a deep emergent marsh; water depths may range from 6 in to 3.3 ft (15 cm to 1 m) during flood stages, but the water level usually drops by mid to late summer and the substrate is exposed. Characteristic vegetation include bluejoint grass (*Calamagrostis canadensis*), reed canary grass (*Phalaris arundinacea*), rice cutgrass (*Leersia oryzoides*), mannagrass (*Glyceria canadensis*), sedges (*Carex stricta, C. lacustris*), three-way sedge (*Dulichium arundina*), bulrushs (*Scirpus cyperinus, S. atrovirens*), sweetflag (*Acrosus americanus),* wild iris (*Iris versicolor*), water smartweed (*Polygonum amphibium*), marsh bellflower (*Campanula aparinoides*), and tufted loosestrife (*Lythrum thrysiflora).*

Palustrine Cultural Subsystems - this group includes communities that are either created and maintained by human activities, or are modified by human influence to such a degree that the physical conformation of the substrate, the hydrology, or the biological composition of the resident community is substantially different from the character of the substrate, hydrology, or community as it existed prior to human influence. Cultural palustrine communities found in the Upper Hudson River include farmed land that may have been partially drained or impounded.

Forests- Hardwood forests are found along the Hudson River. These include ash-maple floodplain forests (*e.g.*, green and black ash, red maple, and slippery elm) and black locust forests found along the banks of the river near Saratoga National Historic Park (SNHP, 1981).

2.1.1.2 Lower Hudson River Habitats

A number of distinct ecological communities are found in the Lower Hudson River, including tidal river, brackish subtidal aquatic bed, brackish tidal marsh, brackish intertidal shore, brackish intertidal mudflats, freshwater swamp, freshwater subtidal aquatic bed, freshwater tidal marsh, freshwater intertidal shore, and freshwater intertidal mudflats. Brief descriptions of these communities based on Reschke (1990) are provided below.

Tidal River- refers to the aquatic community of continuously flooded substrates that support no emergent vegetation. These habitats are found along the Lower Hudson River, from Troy to New York City. Within the Lower Hudson River there are two zones; the deepwater zone includes sections of the lower river with water depths greater than six feet at low tide and the shallows zone includes submerged areas less than six feet at low tide that lack rooted aquatic vegetation. Hence, vegetation is limited to phytoplankton in the upper layers of the water column. In the river there is vertical salinity gradient, with a surface layer of freshwater (salinity less than 0.5 parts per thousand [ppt]) floating over a deeper layer of brackish water (salinity 0.5 and 18.0 ppt). Plate 1 shows the general salinity distribution based on NWI classifications along the lower river (Plate 1: Sheets 4 –16). Salinities at any one place in the river may fluctuate as the tides flow in and out because the "salt wedge" of the brackish water alternatively rises and falls with the tides.

The tidal river community is composed of abundant animal life supported by organic material originating in the watershed. Characteristic fishes include year-round residents as well as seasonal migrants or anadromous species that enter the river as adults to spawn and return to the ocean afterwards. The progeny of these anadromous fish occupy the river as nursery area for the remainder of the year or longer. Fish found in the deepwater community include Atlantic tomcod (*Microgadus tomcod*), hogchoker (*Trinectes maculatus*), rainbow smelt (*Osmerus mordax*), Atlantic sturgeon (*Acipenser oxyrhynchus*) and shortnose sturgeon (*Acipenser brevirostrum*). Characteristic fish of the shallows include striped bass (*Morone saxatilis*), American shad (*Alosa sapidissima*), banded killifish (*Fundulus diaphanous*), spottail shiner (*Notropis hudsonius*), tessellated darter (*Etheostoma olmsteadi*), and pumpkinseed (*Lepomis gibbosus*). Fishes that occur in both deepwater and shallows include bay anchovy (*Anchoa mitchilli*), blueback herring (*Alosa aestivalis*), white perch (*Morone americana*), and alewife (*Alosa pseudoharengus*) (Gladden *et al.*, 1988). Fish predators can capture fish near the water's surface (*e.g.*, bald eagles, osprey) or below the surface of the water (*e.g.*, cormorants, loons, and diving ducks).

Brackish Subtidal Aquatic Bed- is the aquatic community of continuously flooded substrates with rooted aquatic vegetation. The water is brackish (salinity between 0.5 and 18.0 ppt) and is usually less than six feet deep at low tide. Characteristic plant species are waterweed (*Elodea nuttallii*), coontail (*Ceratophyllum demersum*), naiad (*Najas guadalupensis*), sago pondweed (*Potamogeton pectinatus*), horned pondweed (*Zannichellia palustris*), and widgeon grass (*Ruppia maritime*). A common weedy exotic is Eurasian milfoil (*Myriophyllum spicatum).* This community is found along the Hudson River from Newburgh to New York City (*e.g.*, Piermont Marsh).

Freshwater Subtidal Aquatic Bed- is the freshwater (salinity less than 0.5 ppt) aquatic community of continuously flooded substrates with rooted aquatic vegetation. The water is usually less than six feet deep at low tide. Characteristic plant species are waterweed (*Elodea nuttallii*), water celery (*Vallisneria americana*), naiads (*Najas guadalupensis* and *N. minor*), and pondweed (*Potamogeton perfoliatus*). Two exotic weeds, Eurasian milfoil (*Myriophyllum spicatum)* and water chestnut *(Trapa natans*) are common in the Hudson River aquatic beds. This community is found along the Hudson River from Troy to Newburgh.

Brackish Tidal Marsh- this includes sections of the Hudson River where salinities range from 0.5 to 18.0 ppt, and the water is less than six feet deep at high tide. The plant community consists of a mixture of salt marsh and freshwater marsh tidal marsh species, with no species attaining dominance over extensive areas, although some species are locally abundant in patches. The vegetation in a brackish tidal marsh is dense and dominated by tall grasses. Characteristic plants are narrow-leaved cattail (*Typha angustifolia*), arrow arum (*Peltandra virginica*), pickerelweed (*Pontederia cordata*), water smartweed (*Polygonum punctanum*), common reed (*Phragmites australis*), marsh fern (*Thelypteris palustris*), wild rice (*Zizania aquatica*), soft-stem bulrush (*Scirpus tabernaemontanii*), river bulrush (*S. fluviatilis*), dwarf spikerush (*Eleocharis parvula*), arrowhead (*Sagittaria latifolia*), lilaeopsis (*Lilaeopsis chinensis*), rose-mallow (*Hibiscus mosheutos*), yellow iris (*Iris pseudocorus*), and saltmarsh fleabane (*Pluchea ordorata*). Purple loosestrife (*Lythrum salicaria*) is a common weed in brackish marshes.

Tidal marshes provide important feeding and breeding areas for many resident and transient aquatic and terrestrial animals. Fish (*e.g.*, killifish, darters, mummichogs, sunfish, and carp) come into marshes at high tide to feed on invertebrates such as cladocerans, copepods, ostracods, and chironomids. A variety of amphibians, reptiles, birds, and mammals feed on the fish and invertebrates found in marshes. Hudson River tidal marshes support many bird species and large populations of nesting birds, which includes a high density of breeding marsh birds. Characteristic birds include red-winged blackbird, swamp swallow, marsh wren, yellow warbler, common yellowthroat, song sparrow, Virginia rail, American goldfinch, and eastern kingbird.

Brackish tidal marshes are found along the Hudson River from Newburgh to New York City. The downstream limit of brackish marsh communities begins where cordgrass (*Spartina alterniflora*) no longer dominates tidal creek or river banks and the upstream limits extend to where the green seaweed *Entermorpha intestinalis* can no longer be found. Brackish tidal marshes can be distinguished from freshwater tidal marshes by the lack of species restricted to freshwater, such as spatterdock (*Nuphar luteum*), sweetflag (*Acorus americanus*), and blue flag (*Iris versicolor*), and a decrease in the cover of sedges (*Carex* spp. and *Cyperus* spp.).

Brackish Intertidal Mudflats- is a sparsely vegetated community, characterized by low-growing, rosette-leaved aquatics. The community occurs on exposed intertidal mudflats where water salinity ranges from 0.5 to 18.0 ppt. This community is best developed where the mudflats are nearly level so that broad expanses are exposed at low tide. The rosette-leaved aquatics are completely submerged at high tide, and they are usually coated with mud. Characteristic species are spongy arrowhead (*Sagittaria calycina*), strap-leaf arrowhead (*Sagittaria subulata*), mudwort (*Limosella australis*), three-square bulrush (*Scirpus americanus*), and water celery (*Vallisneria americana*). Brackish intertidal mudflats are found along the Hudson River from Newburgh to New York City.

Brackish Intertidal Shore – is a community of the intertidal gravelly or rocky shores of brackish tidal rivers and creeks where water salinity ranges from 0.5 to 18.0 ppt. Brackish intertidal shore is found along the Hudson River from Newburgh to New York City.

Freshwater Tidal Swamp- is a forested or shrub-dominated tidal wetland that occurs in lowlands along large river systems characterized by gentle slope gradients coupled with tidal influence over considerable distances. The swamp substrate is always wet and is subject to semidiurnal flooding by fresh tidal water.

The characteristic trees are green ash (*Fraxinus pennsylvanica*), black ash (*F. nigra*), red maple (*Acer rubrum*), slippery elm (*Ulmus rubra*), American hornbeam (*Carpinus caroliniana),* and sometimes northern white cedar (*Thuja occidentalis*). Common shrubs and vines are alders (*Alnus serrulata, A. rugosa*), spicebush (*Lindera benzoin*), arrowwood (*Viburnum recognitum*), silky dogwood (*Cornus amomum*), gray dogwood (*C. Foemina*), red-osier dogwood (*C. sericea*), Virginia creeper *(Parthenocissus quinquefolius*), and poison ivy (*Toxicodendron radicans*). Characteristic groundlayer species are rice cutgrass (*Leersia oryzoides*), sensitive fern (*Onoclea sensibilis*), clearweed (*Pilea pumila*), spotted jewelweed (*Impatiens capensis*), common monkeyflower (*Mimulus ringens*), knotweeds (*Polygonum* spp.), skunk cabbage (*Symplocarpus foetidus*), hog peanut (*Amphicarpae bracteata*), groundnut (*Apios americana*), wild yam

(*Dioscorea villosa*), sedge (*Carex grayi*), Jack-in-the-pulpit (*Arisaema triphyllum*), and swamp milkweed (*Asclepias incarnata*). This community is found along the Hudson River from Troy to Newburgh.

Freshwater Tidal Marsh- is a marsh community that occurs in shallow bays, shoals, and at the mouth of tributaries of large tidal river systems, where the water is usually fresh (salinity less than 0.5 ppt) and less than six feet deep at high tide. The vegetation is dominated by aquatics that are emergent at high tide. Typically there are two zones in a freshwater tidal marsh: a low-elevation area dominated by short, broad-leaf emergents bordering mudflats or open water, and a slightly higher elevation area dominated by tall grasses.

Characteristic plants of the low elevation broad-leaf emergent zone include spatterdock, pickerelweed, arrowleaf, and fowl mannagrass (*Glyceria striata*). Under the canopy of emergents there may be a sparse understory of rosette-leaved aquatics, such as narrow-leaved arrowheads and mud plantain (*Heteranthera reniformis*). Characteristic plants of the higher zone include narrowleaf cattail, river bulrush, burreed (*Sparganium eurcarpum*), wild rice, and blue flag.

Other characteristic plants that occur in both zones include arrowhead, rice cutgrass, water-hemp, spotted jewelweed (*Impatiens capensis*), estuary beggar-ticks (*Bidens bidentoides*), sweetflag (*Acorus americanus*), softstem bulrush, sedges, and cyperus (*Cyperus* spp.). Purple loosestrife and common reed are common exotics in this community.

Characteristic birds include marsh wren, red-winged blackbird, swamp sparrow, Virginia rail, song sparrow, yellow warbler, least bittern, American goldfinch, willow flycatcher, and common yellowthroat. Freshwater tidal marshes are found along the Hudson River from Troy to Newburgh.

Freshwater Intertidal Mudflats - is a sparsely vegetated community, characterized by low-growing, rosette-leaved aquatics. The community occurs on exposed intertidal mudflats where the water is fresh (salinity less than 0.5 ppt). This community is best developed where the mudflats are nearly level so that broad expanses are exposed at low tide. The plants are completely submerged in three to four feet of water at high tide, and they are usually coated in mud. Characteristic species are strap-leaf arrowhead, mud- plantain (*Heteranthera renifromis*), grass-leaf arrowhead, stiff arrowhead *(Sagittaria rigida)*, three-square bulrush (*Scirpus americanus*), golden club (*Orontium aquaticum*), and wild rice. Freshwater intertidal mudflats are found along the Hudson River from Troy to Newburgh.

Freshwater Intertidal Shore – is a community of the intertidal gravelly or rocky shores of freshwater tidal rivers and creeks. Vegetation may be very sparse. Characteristic species include heartleaf plantain, estuary beggar-ticks, water-hemp, smartweed, cardinal flower (*Lobelia cardinalis*), Pennsylvania bittercress (*Cardamine pennsylvanica*), mud-hyssop (*Gratiola neglecta*), golden club, and an exotic black mustard (*Brassica niger*). Freshwater intertidal shore is found along the Hudson River from Troy to Newburgh.

Estuarine Cultural Subsystems- this grouping includes communities that are either created and maintained by human activities, or are modified by human influence to such a degree that the physical conformation of the substrate, or the biological composition of the resident community is substantially different from the character of the substrate or community as it existed prior to human influence. Cultural communities found in the Lower Hudson River include estuarine channel/artificial impoundment; estuarine impoundment marsh; estuarine dredge spoil shore; and estuarine riprap/artificial shore.

2.1.2 Hudson River Natural History

The Hudson River is home to a wide variety of fish and wildlife species. Many of these animals feed on the abundant vegetation and invertebrates found in the Hudson River. Much of the primary production of organic matter by photosynthesis is accomplished by the phytoplankton. In the Lower Hudson River there is a gradient with respect to species composition and abundance that corresponds to salinity (Boyce Thompson Institute, 1997). Common phytoplankton include diatoms (*e.g., Asterionella* spp., *Coscinodiscus* spp., *Cyclotella* spp., *Melosira* spp., *Skeletonema* spp.), green algae (*e.g., Pediastrum* spp., *Scenedesmus* spp., *Ankistrodesmus* spp.), dinoflagellates (*Ceratium* spp., *Prorocentrum* spp.), and blue-green algae (*Anacystis* spp., *Anabaena* spp.). The maximum gross primary productivity is highest in the brackish/saltwater region just north of New York City (Boyce Thompson Institute, 1997).

Invertebrates

The zooplankton community of the Hudson River is diverse and includes copepods (*e.g., Acartoa tonsa, Eurytemona affinis*), young snails (*e.g., Valvata sincera*), water fleas (*e.g., Bosmina longirostris, Daphanosoma* sp., *Moina* sp.), and immature barnacles (*e.g., Balanus* spp.).

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 RM 50 are dominated by freshwater arthropods and oligochaetes. The middle reaches from RM 25 to RM 50 have a mixture of freshwater and marine forms and the lower reaches below RM 25 support a typical marine assemblage including marine oligochaetes, polychaetes, and crustaceans. In the lower river decapods, such as the penaeid shrimp (*Penaeus* spp.) and blue crab (*Callinectes sapidus*) are found. Profiles of the dominant macroinvertebrate species/groups found in the Hudson River are provided in Appendix C of USEPA 1999c.

An unwelcome invertebrate in the Hudson River is the zebra mussel (*Dreissena polymorpha*). It was first detected near Catskill in the Lower Hudson River in May 1991 (Strayer *et al.*, 1996) and its distribution is strongly controlled by the distribution of suitable substrata. The highest densities (average 17,000/m^2) are found on rocks in deep (> 5 m) water. Even though such deep-water rocky areas cover only 7% of the estuary, they support 95% of the zebra mussel population (Strayer *et al.*, 1996). Zebra mussels are not a major problem in the Upper Hudson River, probably due to lack of suitable substrata.

Fish

The Hudson River supports a diverse assemblage of fish. Fish in the river can be classified according to predominant habitat or the habitat in which they reproduce. Table 2-1 contains a list of Hudson River fishes along with their predominant habitat (*i.e.*, freshwater, freshwater/brackish, saltwater, anadromous, and catadromous). 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. The nutrient-rich brackish water portion of the estuary provides food composed primarily of immature and mature invertebrates, such as shrimp, polychaetes, copepods, crabs, barnacles, oysters, and clams. Estuarine spawners include the bay anchovy, hogchoker, and mummichogs. Freshwater and anadromous species include the Atlantic tomcod, Atlantic sturgeon, shortnose sturgeon, American shad, alewife, blueback herring, white perch, and striped bass. Marine spawners include the American eel, Atlantic menhaden, bluefish, weakfish, longhorn sculpin, and winter flounder.

NYSDEC (1989) collected data on the distribution of fish and their use of habitats in the Hudson River Estuary from Troy to Saugerties, NY. The data were combined with existing information to describe generalized fish distribution, aggregations, and patterns of habitat use (Table 2-2). Fish distribution in the Hudson Estuary is habitat dependent for most resident species. The habitat with the greatest diversity of fish was vegetated backwaters, while that with the highest fish abundance (in spring) was the tailwater behind Federal Dam at Troy. Fish abundance changed seasonally between habitats, and shallow water habitats had higher fish abundance in summer and fall than spring. Offshore fish aggregations were most diverse around rock piles and least diverse in the main channel.

Herpetofauna

Amphibians hatch from eggs laid in water and live for a time as aquatic larvae before metamorphosing into air-breathing terrestrial animals. Even as adults, amphibians require moist conditions for survival. Though most have lungs, they also exchange oxygen through their skin. For the transfer to happen efficiently, their skin must be moist. Amphibians found along the Hudson River based on the *New York State Amphibian and Reptile Atlas 1990-1998* (NYSDEC, 1999b) are listed in Table 2-3 and include salamanders, toads, and frogs. Amphibians that have been sighted in Saratoga National Historic Park, Saratoga County, NY (SNHP, 2000) are marked with an asterisk.

The amphibian populations of the lower river are fairly low (Stanne *et al.*, 1996). This may be due to: 1) cycles of exposure and flooding in the tidal zone may pose difficulties for adult amphibians and certainly are problematic for their eggs, which must stay wet; 2) although sometimes found in slightly brackish water, amphibians avoid salt water and are absent from the lower portions of the river; 3) intertidal waters are subject to high temperatures in summer and ice scour in the winter, conditions that threaten any animal that can not leave or burrow in the mud; 4) many predators (*e.g.*, large fish, herons, and snapping turtles) prowl these areas and may limit populations; and 5) some amphibians are very sensitive to pollutants.

Reptiles found along the Hudson River include turtles, snakes, and lizards (NYSDEC, 1999b), as listed in Table 2-4. As is the case with amphibians, many reptiles common in wetland habitats elsewhere along the Hudson River are much less common in the river's tidal wetlands (Stanne *et al.*, 1996). Representative species of turtles include the snapping turtle and painted turtle. Snakes found along the river include the water snake and garter snake.

Birds

Among the Hudson River's vertebrates, birds are second only to fish in overall numbers and they rival fish in their diversity (Stanne *et al.*, 1996). The highly productive estuary produces large supplies of food for birds to feed on. The Hudson River also serves as a flyway, a route that birds follow as they migrate north in spring and south in the fall. Birds exploit all the habitats available in the Hudson River ecosystem, and are active in all seasons.

A list of breeding birds found along the Hudson River based on Andrle and Carroll (1988) is provided in Table 2-5. Birds sighted in Saratoga NHP (SNHP, 2000) are marked with an asterisk. There are many other birds that may also be sighted along the river, but have not been confirmed to breed along the river. Examples of birds found along the river, grouped as swimming, wading, perching birds of wetland habitats, and wide-ranging river birds (Stanne *et al.*, 1996) are summarized below.

Swimming birds include many commonly seen waterfowl, such as ducks, geese, and swans, and also gulls, cormorants, and coots. Surface-feeding ducks are found in shallow wetlands and feed on underwater vegetation and invertebrates by tipping up or dabbling (scooping up water and food and allowing the water to drain out the sides of their bills). These ducks are numerous in the spring and fall during migration, but only the mallard and black ducks winter in any significant numbers. Diving ducks (*e.g.*, greater scaup, bufflehead, and common merganser) are adapted for swimming underwater and feed on a variety of aquatic organisms including plants, clams, mussels, crabs and other crustaceans, and fish. Large numbers of diving ducks are seen during spring and fall migrations, and many winter on the Lower Hudson, staying south of the ice cover or using openings in the ice for underwater feeding.

Wading birds stay along the river's edge following the tide in and out over the shoreline, tidal flats, and marshes. These birds include shorebirds, herons, egrets, bitterns, and rails. Shorebirds (*e.g.*, killdeer, spotted sandpiper, greater yellowlegs) feed mainly on invertebrates, such as worms, crustaceans, insects, and mollusks. Herons (*e.g.*, great blue heron, green heron), egrets (*e.g.*, snowy egret), and bitterns (*e.g.*, least bittern) have long, dagger like bills adapted for catching fish and other small animals. Many of these species nest along the river. Although many wading birds nest and raise their young along the Hudson River, few overwinter on the river since much of their preferred habitat is covered in ice.

Perching birds of wetland habitats include thrushes, blackbirds, wrens, finches, sparrows, flycatchers, swallows, and jays, which all belong to the largest order of birds, the passerines, or perching birds. Most feed on insects and other small invertebrates, but some also eat seeds. Most are migratory, but many species breed along the river. Common summer birds include the marsh wren, red-winged blackbird, and swamp sparrow.

Wide-ranging river birds include raptors, kingfishers, and gulls. Most typically hunt over the open water. Of the raptors (*i.e.*, eagles, hawks, falcons, and owls) found along the river the bald eagle and osprey are most dependent on the river because fish is their preferred food. The bald eagle, a federal-listed threatened species and a NYS-listed endangered species, is monitored by NYSDEC. The belted kingfisher is the only kingfisher found along the Hudson River. They catch small fish using their strong, dagger like bills. Gulls (*e.g.*, herring gull, great black-backed gull) are opportunistic feeders, feeding upon fish, mollusks, crustaceans, human food scraps, and even other birds and small mammals. Gulls can be seen in all seasons along the river.

Mammals

Many mammals are found close to the Hudson River and take advantage of the resources provided by it. However, only a few mammalian species live and reproduce by the river. A list of mammals potentially found along the Hudson River is provided in Table 2-6. Mammals sighted in Saratoga NHP (SNHP, 2000) are marked with an asterisk. Herbivores, such as whitetail deer and small rodents (*e.g.*, meadow vole, white-footed mouse) can be found feeding along marshes. The muskrat is a rodent that commonly inhabits both the freshwater and brackish marshes of the Hudson River. They build houses of plant stems and mud, but may also live in burrows excavated along the shoreline. Muskrats feed on plants, favoring cattails. Mammals in families of squirrels (*e.g.*, chipmunk, red squirrel, woodchuck), hares and rabbits (e.g., cottontail, snowshoe hare), and moles (eastern mole, star-nosed mole) may also feed near the river. The most common omnivore found along the river is the raccoon.

A variety of insectivores, such as shrews and bats, are found along the river. Shrews (*e.g.*, short-tailed shrew, masked shrew) feed mainly on insects and other invertebrates living in the ground. Bats (*e.g.*, little brown bat, eastern pipistrelle) feed on insects, many of which are emergent insects with an aquatic life stage.

Large predatory mammals found along the Hudson include canids (*e.g.*, coyote, gray fox, red fox), and members of the weasel family, including both species commonly found in wetland habitats (mink and river otter) and species more commonly found in other habitats (*e.g.*, striped skunk, longtail weasel). The mink and river otter stay close to the water, denning along the shoreline, and feeding largely on fish. These animals and other large carnivores, often active at night, are elusive and seldom seen (Stanne *et al.*, 1996).

Marine mammals, such as whales, dolphins, and seals are found at the mouth of the Hudson River and are rarely seen in the river.

2.1.3 Threatened and Endangered Species

The federal Endangered Species Act (16 USC Sections 1531-1544) 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. The USFWS was contacted to obtain a list of listed or proposed endangered or threatened species potentially found in or near the Hudson River (USFWS, 2000).

New York State also maintains its own separate list of animals and plants that are considered endangered, threatened, or of special concern at the state level. The New York State Natural Heritage Program was contacted to obtain a current listing of rare or state-listed animals and plants, significant natural communities, and other significant habitats found within a one-mile corridor on either side of the Hudson River (NYSNHP, 2000). All threatened, endangered and special concern species listed in Table 2-7 have been sighted in and along the Lower Hudson River, and some of them (*e.g.*, bald eagle and short-eared owl) are also found in the Upper Hudson River Valley, as noted in the table. Profiles of threatened and endangered species found in and along the Hudson River are provided in Appendix G of USEPA 1999c.

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*), blunt-lobe grape fern (*Botrychium oneidense*), 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 (*Micrantherum 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, although several rare dragonflies and the tawny emperor butterfly are also found near the river.

The shortnose sturgeon (*Acipenser brevirostrum*) is a federal and NYS-listed endangered species found in the Lower Hudson River. No threatened fish species or fish species of special concern are found in the Hudson River, although the rare bluespotted sunfish may occur along the 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 potentially 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

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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 is the endangered Indiana bat (*Myotis sodalis*) (USFWS, 2000). 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.

The Revised ERA does not evaluate specific risks to most threatened and endangered species, but risks to organisms that have similar habitats and feeding strategies can be used to infer risks to threatened and endangered species at the individual, rather than population, level. The Hudson River Natural Resources Damage Assessment will evaluate injuries to bird species, particularly federal or State listed threatened and endangered species (*e.g.* bald eagle, peregrine falcon), species that have been shown to be sensitive to PCBs or other hazardous substances of concern (*e.g.* black-crowned night heron, wood duck), and species that are consumed by humans (*e.g.*, waterfowl) (Hudson River Natural Resources Trustee Council, 1998).

2.1.4 Significant Habitats

New York State's Coastal Management Program (CMP) has a policy aimed at the protection of fish and wildlife resources of statewide significance. The specific policy statement is as follows: "Significant coastal fish and wildlife habitats will be protected, preserved, and where practical, restored so as to maintain their viability as habitats." NYSDEC evaluates the significance of coastal fish and wildlife habitats and following a recommendation from NYSDEC, the Department of State (DOS) designates and maps specific areas under the authority of the Coastal Management Program's enabling legislation, the Waterfront Revitalization and Coastal Resources Act (Executive Law of New York, Article 42). These designations are subsequently incorporated into the Coastal Management Program under authority provided by the Federal Coastal Management Act.

Thirty-four (34) sites in the tidal portion of the Hudson River have been designated as Significant Coastal Fish and Wildlife Habitats under the NYS Coastal Management Program (NYSDOS, 1987). 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 2-8 (NYSDOS and the Nature Conservancy, 1990). These areas are unique, unusual, or necessary for continued propagation of key species. Habitats (and their associated communities) present in significant habitats include freshwater and brackish water shallows, mudflats, marshes, swamp forest, deepwater, and creeks. Many areas provide spawning areas for fish and are used as resting and feeding areas for migratory birds. A summary of important resources at each Hudson River significant habitat, including community types, rare species, and resource value is provided in Table 2-9.

Four of the significant habitats comprise the Hudson River National Estuarine Research Reserve (NERR), administered by NYS in partnership with NOAA. These areas and several other NYSDOS-designated significant habitats were sampled during the ecological risk assessment field sampling effort (see Plate 1 for locations in Stockport Flats (NERR) RM 123; Tivoli Bays (NERR) RM 100; Iona Island (NERR) – RM 40; Piermont Marsh (NERR) RM 24; Shad Island RM 135; Roger's Island RM 118; Esopus Meadows RM 88; and Moodna Creek RM 58).

2.1.5 Human Use of the River

The Hudson River is an important source of energy, natural resources, and transportation to populations along the river, much as it has been to prehistoric and historic populations of the region. During the thousands of years following the final northerly retreat of the Wisconsin Glacier approximately 14,000 years ago, the river and its drainages gradually transformed the landscape, providing a rich habitat and supporting a substantial prehistoric population. During the formative years of America's historic settlement, the Hudson River often proved to be of vital logistical importance and was the site of numerous military engagements. During the 17th through 19th centuries, this region was gradually settled by European immigrants who cleared the land, established towns, and built a variety of industries along the river. Efforts to maximize the industrial use of the river led to the construction of locks, dams, gates, channels, and related structures.

In 1609, the Englishman Henry Hudson was looking for a quick passage to China as he sailed along America's North Atlantic coast. Hudson thought he found what he was looking for when he entered New York Bay and what is now the river named for him. He and his crew, sailing a ship called the Half Moon, traveled about 150 miles up the river near what is now Albany before realizing it would not lead them to their destination of choice. Hudson had been hired for the journey by a Dutch trading company, the Dutch East India Company, and his explorations led to the area first being settled by the Dutch.

The 60-mile (96.5 km) Champlain Canal was completed in 1825. This canal linked the Hudson River at Troy, New York with the southern end of Lake Champlain at Whitehall, New York. During the heyday of the Champlain Canal, between 1823 and the early 20th century, thousands of canal boats passed between Lake Champlain and the Hudson River, transporting raw materials and finished products, linking the farmers and merchants of the Hudson Valley with the rest of the world. Canal boats were by far the most common type of working craft; these boxy vessels efficiently and inexpensively transported heavy cargoes, and at the same time served as home for canal boatmen and their families.

The first American school of landscape painting is known as the Hudson River School. In 1825, a young artist named Thomas Cole arrived in the Hudson Valley along the Lower Hudson. He was captivated by the scenery and began a sketching trip through the Hudson River Valley. His subsequent paintings celebrating nature inspired other artists to do the same. Their style of dramatic and uniquely American landscapes became known as the Hudson River School of Painting, which flourished from 1825 to 1870.

Recently, the Hudson River has been designated an American Heritage River, which is an initiative designed to more effectively use the federal government's many resources. Through this program, environmental, economic, and social concerns will be addressed in a plan that is designed and driven by the local community. The American Heritage Rivers initiative is intended to help communities revitalize their rivers and the banks along them--the streets, the historic buildings, the natural habitats, the parks--to help celebrate their history and their heritage.

A site file search of the records of the New York Office of Parks, Recreation, and Historic Preservation (OPRHP), the New York State Museum, and the National Register of Historic Places was conducted in 1990 in the Towns of Moreau and Fort Edward. The search resulted in the documentation of 20 cultural resources (Collamer & Associates, Inc., 1990), including three prehistoric sites (one of which was a stratified, multi-component seasonal campsite); one site dating to the French and Indian War; one multi-component prehistoric site also containing French and Indian War and Revolutionary War encampments; the Fort Edward Blockade; the Satterlee Lane Historic Deposits; eight historic houses or former houses; the historic Ferry Landing; a mid- to late-19th century mill (Allen Mill); the site of a ferry house and blockhouse; and the location of the Royal Blockhouse. It is likely that similar types of cultural resources are present along other portions of the Hudson River. In addition, the Saratoga National Historic Park lies on the western bank of the Hudson River in the Town of Stillwater.

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 (COCs) are limited to PCBs. This decision is consistent with the overall purpose of the reassessment, as PCBs are the chemicals that are the basis for the Reassessment RI/FS, and allow the risk assessment to focus on the contaminants of greatest concern, as supported by NYSDEC fish analyses summarized below.

In 1997, NYSDEC analyzed total DDT, total chlordane, total endrin, total endosulfan, dieldrin, aldrin, mirex, total heptachlor, total hexachlorobenzene, toxaphene, meoxychl, individual polycyclic aromatic hydrocarbons (PAHs), cadmium, mercury, dioxins and dibenzofurans in fish collected from above Federal Dam (RM 201) to the George Washington

Bridge (RM 12). Fish species analyzed were brown bullhead, largemouth bass, pumpkinseed, smallmouth bass, yellow perch, white perch, striped bass, white catfish, and American eel. For the most part, concentrations of these contaminants were relatively low or below detection limits (Sloan, 1999). Mercury is present in most locations and in all species with some largemouth bass, smallmouth bass, and striped bass individuals above 1 part per million (ppm), the federal action level. Total DDT is about 0.5 ppm in some species. The action level for DDT is 5 ppm and the guideline for protection of piscivorous wildlife is 0.1 ppm. With few exceptions, PAHs were below detection limits at the locations sampled (*i.e.*, RMs 189, 175, 147, 112, 27, and 12). Phenanthrene was found at 5 parts per billion (ppb) in two samples, a white catfish and a largemouth bass at Catskill (RM 112) and fluorine was detected at 10 ppb at the Tappan Zee Bridge (RM 27). Although concentrations of dioxins and dibenzofurans are relatively low, they are of some concern to fish-eating wildlife, but not to the same degree as PCBs (Sloan, 1999). Overall, NYSDEC concluded that levels of these contaminants are not as problematic as the concern caused by PCBs.

Consistent with the focus of the Reassessment RI/FS, this 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 fate, transport and bioaccumulation modeling. Analyses conducted as part of the RBMR (USEPA, 2000a) show that the trichlorinated and higher PCB congeners approximate total PCBs in biota.

PCBs are 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.

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. The more highly chlorinated PCB congeners are in evidence throughout the Hudson River, especially in fish. 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. Processes that govern PCB distributions in the environment and biological fate and transport processes are discussed in Section 2.3.1.

According to scientists at GE, at least 80 percent of the total PCBs discharged during the production of electrical capacitors are believed to have been Aroclor 1242, with lesser amounts of Aroclors 1254, 1221, and 1016. However, the Aroclors that were discharged varied over time, with Aroclor 1254 being 75 percent or more of the total until about 1955; Aroclor 1242 being at least 95 percent of the discharges from 1955 through 1971; and Aroclor 1016 being close to 100 percent of the discharge from 1971 through 1977 (Brown *et al.*, 1984).

Since the cessation of manufacturing discharges, extensive evidence has been found to document continued leakage of PCBs into the Hudson River beginning in 1983. The largest known leakage event occurred during 1991 to 1993, apparently initiated by a partial failure of a gate structure in 1991 at the abandoned Allen Mill at Bakers Falls, which is located on the river adjacent to the GE Hudson Falls plant (see Section 1.2). Congener patterns in PCB loads at Rogers Island indicate the presence of freshly released Aroclor 1242, consistent with the observed leakage of non-aqueous phase PCB-bearing oils from the bedrock beneath the GE Hudson Falls plant site.

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). It serves as a communication tool that illustrates the major pathways by which ecological receptors might be exposed to PCBs associated with releases from the GE facility. Preliminary conceptual models were presented to various agencies and to GE during the early 1990s to identify the exposure pathways that would be included in modeling and ultimately in the risk assessments. Comments on these early conceptual models were incorporated. For example, the exposure pathways linking the river with selected wildlife species were added to the conceptual model as a result of comments received during this initial review process.

The exposure models initiated during the modeling efforts were eventually developed into an integrated site conceptual model for the ERA (Figure 2-1). 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 continue 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. PCBs present in the water and sediment can be accumulated by plants, invertebrates, and fish and transferred through aquatic food webs. Some wildlife species rely partially or fully on aquatic plants and animals for food and, therefore, PCBs present in aquatic biota can also be transferred to these wildlife species and the species that prey on them. PCBs may also enter the terrestrial food chain through sediments deposited on the floodplain during high flow events. Such high flow events may also increase the availability 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 species models to evaluate assessment endpoints. Species selected as models are intended to be representative of other species at the same trophic level that share similar ecological characteristics. These groups of species are often referred to as guilds. By evaluating a representative member of a guild and by accounting for the predominant guilds, the uncertainty associated with missing an important species group or pathway is reduced. Input received from interested parties indicated that the ERA should be comprehensive and should consider the major guilds of species that rely on the river for habitat or food. Emphasis was given to fish and wildlife that are higher in the food chain; risk to plants or microorganisms are not considered in this assessment. This reflects experience with the types of effects associated with exposure to PCBs as well as the fact that the chemicals are biomagnified from one trophic level to the next. Fish and wildlife that are higher in the food chain are more likely to be exposed to higher concentrations of PCBs than are animals lower in the food chain. The 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 (*i.e.*, fish and wildlife) 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).

Chemical fate and transport and the magnitude and extent of contamination have been covered extensively in previously released Phase 2 reports, such as the Revised Baseline Modeling Report (USEPA, 2000a), Data Evaluation and Interpretation Report (USEPA, 1997a), and Low Resolution Sediment Coring Report (USEPA, 1998c). Therefore, the chemical fate and transport discussion in this report is limited to the processes that govern PCB distributions in the environment and biological fate and transport.

2.3.1.1 Processes That Govern PCB Distributions in the Environment

A challenge to developing a modeling framework for PCB bioaccumulation is that PCBs consist of 209 individual congeners, each of which exhibits varying degrees of bioaccumulation

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potential, depending on the degree and substitution of chlorination. In general, the higher chlorinated congeners tend to preferentially accumulate in biota.

Studies that have measured PCBs as individual congeners have provided insights into the bioaccumulation processes for water column- and sediment-based communities. Several researchers have noted that whether or not total PCB levels increase with position in the food chain, chlorine content of PCB body burdens tends to increase (Smith *et al.*, 1985; Oliver and Niimi, 1988; Van der Oost *et al.*, 1988; MacDonald *et al.*, 1993). Congener patterns of caged fathead minnows and feral brown bullhead from the area around Thompson Island Pool in the Hudson River were generally similar, sharing 60 percent of their 20 most abundant peaks, but the bullhead had higher concentrations of hexa- and heptachlorobiphenyls (Jones *et al.*, 1989). The fish contained 17 peaks that were not detectable in water samples. It has been noted that when young bluefish enter the Hudson River from offshore, heavier, more chlorinated congeners were accumulated to a greater level than lighter, less chlorinated congeners (LeBlanc and Brownawell, 1994).

A variety of factors control accumulation of PCB congeners (Shaw and Connell, 1984; Jones *et al.*, 1989; Kadlec and Bush, 1994; Ankley *et al.*, 1992; LeBlanc and Brownawell, 1994; Bright *et al.*, 1995; Willman *et al.*, 1998). Accumulation of PCB congeners is influenced both by the tendency of the congener to adsorb onto a surface as well as the tendency to partition into organic-rich matrices (*e.g.*, organic carbon in sediment, particulate organic carbon in the water column and lipid in biota). These factors include:

1. Individual PCB congener characteristics, including solubility and partition coefficients, degree of chlorination, and stereochemistry. Shaw and Connell (1984) found that more planar molecules are more strongly absorbed that those with more regular shapes, that is, the stereochemistry of the molecule has the greatest influence on adsorption. Degree of chlorination, by contrast, has a greater influence on partitioning into organic-rich matrices, up to a K_{ow} of approximately 7 and decreasing thereafter.

2. Characteristics of the fish, including lipid content of gills, blood, and tissue; cardiac output; ventilation volume; gill surface area; epithelium layer of gill; aqueous stagnant layer of gill; ability to biotransform PCBs; and, excretion rates.

3. Environmental factors, including temperature, pH, light, current, suspended particles, and dissolved organic compounds.

2.3.1.2 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, 1997b). 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, 1997b).

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).

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 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).

PCBs have also been found to accumulate in predatory fish tissues at higher concentrations than the concentrations in the surrounding water would predict (Thomann and Connolly, 1984), a process known as biomagnification. Depending upon the position of an aquatic organism within the aquatic food web, exposure may be intensified through food sources as organisms consume other organisms that have bioaccumulated PCBs in the lipid portion of their tissues. Because of the important role of food as an exposure pathway, the feeding ecology of a fish species is a key aspect in distinguishing between the relative contribution of the water column and sediments to body burdens of PCBs.

Direct Uptake from Water

For fish, direct uptake of PCBs from water occurs primarily across the gills. No significant evidence exists for absorption through the epidermis (Shaw and Connell, 1984).

The significance of direct uptake from water of PCBs has been debated. Based upon laboratory studies, Shaw and Connell (1984) argued that uptake via the gills is the major route for accumulation of PCBs. Some field studies have indicated that water column uptake could account for PCB concentrations observed in biota, if PCB concentrations were normalized for lipid content of the organism (*e.g.*, Clayton *et al.*, 1977).

Other researchers have continued to examine the potential for bioconcentration through the gills to account for PCB concentrations. Caged rainbow trout that were fed clean, commercial food appeared to accumulate PCBs directly from contaminated waters of the St. Lawrence River (Kadlec, 1994; Kadlec and Bush, 1994). Barron (1990) noted that simple evaluations of uptake directly from the water column have assumed that bioconcentration is controlled by the hydrophobicity of the compound, as measured by its octanol-water partition coefficient and argued that bioconcentration appears to be independent of octanol-water partition coefficients when the coefficient is small or when the molecule to be accumulated is large. Other factors that affect bioconcentration include: molecular shape, degree to which the compound is bound to dissolved organic matter, lipid content of the gills, size of the organism, blood flow, variations in enzyme content and activity, and exposure temperature and ionic content.

Uptake from Sediments

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.

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. Equilibrium partitioning has been suggested to be the major factor controlling bioaccumulation in sediment-based benthic communities.

A steady-state food chain model with a benthic invertebrate component was developed to account for both water column and sediment sources of contaminants (Thomann *et al.*, 1992). This model considered four exposure routes for ingestion of particulate contaminants: sediment organic carbon, overlying plankton, interstitial water, and overlying water. Applying the model to an amphipod-sculpin food web in Lake Ontario (Oliver and Niimi, 1988), Thomann and his colleagues (1992) found that accumulation was based primarily upon a benthic food web rather than upon direct uptake from the water column. They noted however, that including the overlying water and phytoplankton as a food source was necessary to explain the field data. Considering only interstitial water and sediment particles as contaminant sources did not explain the observed concentrations.

Uptake via Food

Field studies and modeling efforts have indicated that biomagnification through the food chain is an important component for bioaccumulation. Sloan *et al.*, (1985), for example, suggested that the presence of higher chlorinated Aroclor mixtures in fish of the Lower Hudson River might reflect a food chain component to bioaccumulation. Using existing field data, Thomann (1981, 1989) derived steady-state food chain models, considering uptake of contaminants from both water and food sources through several trophic levels. The models indicated that food assimilation, excretion, and net weight gain were important characteristics that determined bioaccumulation levels. They also demonstrated that for top predators, such as Hudson River striped bass, almost all the observed PCB body burden could be attributed to a food source. In Lake Michigan lake trout, only 2 to 3 percent of the PCB accumulation could be predicted from water column concentrations using an age-dependent model (Thomann and Connolly, 1984), while transfer through the food chain accounted for up to 99 percent of the body burden of PCBs in Lake Michigan lake trout.

Many researchers have tested, refined, or elaborated upon Thomann's food chain models. One test of the approach examined PCB accumulation in young-of-the-year bluefish which enter the Hudson River Estuary from relatively uncontaminated offshore waters and grow quickly (LeBlanc and Brownawell, 1994). Connolly *et al.*, (1985) considered growth rates, respiration rates, food assimilation efficiency, predator-prey relationships, PCB assimilation efficiency, and bioconcentration factors for PCBs when they applied a model to existing data from the Hudson River system. They predicted PCB levels in Hudson River striped bass, assuming various reductions in concentrations of PCBs in the water column. They also began efforts to incorporate lipid and non-lipid components of the striped bass into the model. Pizza and O'Connor (1983) conducted laboratory experiments to determine rates of PCB accumulation from the gut and elimination from the body in young-of-the-year striped bass from the Hudson River. An EPA model, Food and Gill Exchange of Toxic Substances, or FGETS, has been used to predict average concentrations of contaminants in the food web over time (*e.g.*, Woolfolk *et al.*, 1994). This model incorporates bioconcentration of contaminants from the water column and biomagnification in the food chain.

Gobas and his colleagues (Gobas, 1993; and Gobas *et al.*, 1995 and 1999) examined the roles of food digestion, food absorption, and rates of gill elimination and metabolic transformation upon bioaccumulation. The model has recently been updated to include exposure from both water and sediment sources, and a pharmacokinetic module. The mechanistic model relied on in this ERA (FISHRAND) is based on the aforementioned studies (Gobas, 1993; and Gobas *et al.*, 1995 and 1999).

2.3.1.3 Spatial and Temporal Issues in Congener-specific Uptake

A number of competing factors govern the environmental fate of individual congeners. In general, the physical-chemical properties of the congeners together with the physiological characteristics and metabolic pathways of biota dictate uptake profiles. Metabolic modification in either the organism or its prey items could strongly influence congener signatures and lead to temporal changes in congener profiles. That is, congener signatures of PCBs are generally a reflection of differential rates of enzyme-mediated hydroxylation and excretion (Bright *et al.*, 1995). For example, there is some evidence that BZ #77, which is readily accumulated in fish and birds, has been shown not to bioaccumulate in otter in a field study (Leonards *et al.*, 1998). By contrast, these authors found that BZ#126 contributed 30 to 50% to total TEQ concentrations in fish, while the same congener contributed 60 to 80% in otter, suggesting enrichment. BZ#169 was also enriched.

In general, congeners with less *ortho*-substitution (or higher *meta*- and *para*- chlorine substitution) typically have higher K_{ow} values (Willman *et al.*, 1997; Fisk *et al.*, 1998). These are the congeners that theoretically should demonstrate the lowest elimination and greatest bioaccumulation potential. A laboratory study involving juvenile rainbow trout evaluated sixteen *meta*- and *para*-substituted congeners (BZ#18, BZ#28, BZ#44, BZ#52, BZ#66, BZ#101, BZ#105, BZ#118, BZ#128, BZ#138, BZ#153, BZ#187, BZ#189, BZ#195, BZ#206, and BZ#209) to determine whether there were any observable and significant relationships between bioaccumulation parameters and the K_{ow} of these congeners (Fisk *et al.*, 1998). The authors found that all of these congeners (except BZ#118) biomagnified in fish, and that the estimated half-lives showed significant curvilinear relationship with K_{ow} (increasing up to a K_{ow} of approximately 7 and thereafter decreasing), consistent with the observations of other authors. The assimilation efficiency did not relate as well to the K_{ow} as did the half-lives and biomagnification factors. However, K_{ow}s for the congeners are obtained from other sources and were determined under specific conditions, and may not be representative of the apparent K_{ow} in this study that might differ from literature K_{ow}s.

A field study in Cambridge Bay, Northwest Territories (Bright *et al.*, 1995) found that the percent composition of total PCB levels in four-horn sculpin livers contributed by BZ#77 and BZ#126 was relatively constant regardless of total PCB concentration, suggesting that for a given species increased exposure to PCBs does not lead to increased relative concentrations of these congeners. Limited data also found that these congeners were diminished rather than enriched relative to the total PCB concentration in this particular food web. They found no significant difference in the congener distribution of the whole body as compared to the liver, and they found that the seven dominant congeners (68 to 92% of total PCBs in sculpin species) contained chlorines on the *para* positions and that none showed adjacent unsubstituted *meta* and *para* sites. There was no evidence for metabolism of *ortho*- and *meta*-unsubstituted congeners, although this has been shown to be the case in some marine mammals (Bright *et al.*, 1995; Boon *et al.*, 1994).

A field study in the Canadian arctic (Norstrom *et al.*, 1988) concluded that polar bears appear able to metabolize PCB congeners in which there are nonchlorinated *para* positions, adjacent nonchlorinated *ortho-meta* positions, or both *ortho* positions are chlorinated in one ring.

Willman *et al.* (1997) found that mono-*ortho* and non-*ortho* congeners were not systematically enriched within a sediment-plankton-fish foodweb in a freshwater estuary. In fact, many coplanar congeners, including BZ #77, were depleted with increasing trophic level.

Note that there are numerous uncertainties associated with the kinds of field studies described here. Organisms identified as prey may not be representative of the organisms actually consumed, and the exposure zones may be different for the top level predator and its prey items. In general, the uncertainties are of a sufficient magnitude that this analysis considers congener profiles in the overall Tri+ mixture to be relatively consistent over time. The field data show that the congener most likely to be preferentially retained over time is BZ#126, which is the congener that is typically at concentrations below the detection level in the Hudson River. Thus, the assumption of BZ#126 at the detection level would appear to be protective and an overestimate rather than an underestimate of the contribution this congener makes to the total PCB mixture.

2.3.2 Ecosystems of the Hudson River

As discussed in Section 2.1, the Hudson River is home to a wide variety of ecosystems. Broad categories of ecosystems present in the river include non-tidal freshwater (RM 154 to RM 195 above Federal Dam); tidal freshwater (from RM 153 to ~RM 60 from Federal Dam to Newburgh and below); and estuarine (RM 60 to RM 0 from Newburgh to the Battery). These aquatic ecosystems are considered to be the primary ecosystems at risk and are therefore the focus of this ecological risk assessment. However, in addition to the aquatic communities associated with the Hudson River, many species found in 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 have not been extensively investigated. Nevertheless, over the last 50 years, some PCBs have likely been deposited along the Hudson River shoreline; however there are insufficient data available to characterize the nature and extent of PCBs in floodplain soils. Consistent with the primary focus of the Reassessment RI/FS, the Revised ERA does not quantitatively estimate PCB exposure from floodplain soils, but does discuss exposure to PCB soils as a source of uncertainty in Chapter 6.

2.3.3 Aquatic Exposure Pathways

Aquatic and semi-aquatic organisms, such as fish, invertebrates, amphibians, and reptiles are exposed to PCBs through direct uptake from water; uptake from sediment; and/or uptake via food (including plants), as described in Section 2.3.1.2. Exposure is dependent on timing (*e.g.*, life-stage), feeding preferences, and length of time of exposure.

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.

Uptake from sediment 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. Epifaunal organisms living on the surface of the sediment receive exposure from both the sediment and the overlying water.

Uptake via food 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 Hudson River. The inhalation exposure pathway is not considered any further for the same reason.

Uptake via food constitutes the primary PCB exposure pathway for terrestrial animals living in the Hudson River watershed. PCB-contaminated prey include: 1) 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); 2) 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, odonata, tricoptera); 3) animals that are entirely aquatic but migrate in and out of the Hudson River (*e.g.*, striped bass and eels); 4) 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; and 5) 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.

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.

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. In addition, metabolic rate affects the ingestion rate such that, in order to sustain a high metabolic rate, endotherms need to eat more food or food with a higher caloric value (*e.g.*, high in fat).

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 were developed from the conceptual model, by considering input received from interested parties, and from experience at other sites contaminated with PCBs. Assessment endpoints are explicit expressions of the actual environmental values that are to be protected, operationally defined by an ecological entity and its attributes (USEPA, 1998b). They are expressed in terms of the ecological receptor (*e.g.*, a species, community of organisms, or other ecosystem component) and an attribute (*e.g.*, survival or reproduction). Most of the assessment endpoints developed for this ERA evaluate risks to local populations of fish and wildlife species. Therefore, the assessment endpoints are expressed in terms of particular species (representative of larger guilds) and population attributes such as survival, growth, and reproduction. 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) as shown in the conceptual model (Figure 2-1).

The assessment endpoints selected are:

- Sustainability of a benthic invertebrate community that can serve as a food source for local fish and wildlife.

- Sustainability (*i.e.*, survival, growth, and reproduction) of:

  - local forage fish populations;
  - local omnivorous fish populations; and
  - local piscivorous fish populations.

- Sustainability (*i.e.*, survival, growth, and reproduction) of local wildlife including:

  - insectivorous birds;
  - waterfowl;
  - semi-piscivorous/piscivorous birds;
  - insectivorous mammals;
  - omnivorous mammals; and
  - semi-piscivorous/piscivorous mammals.

The selected assessment endpoints along with respective measurement endpoints are listed in Table 2-10.

2.5 Measurement Endpoints

Measurement endpoints (also known as measures of effect and measures of exposure) are the actual measurements or estimates used to evaluate each of the assessment endpoints and are the basis for evaluating risk. The ERA relies primarily on evaluating exposure to fish and wildlife using either measured (for current conditions) or modeled (for current and future conditions) concentrations of PCBs. The emphasis placed on measures of exposure reflects the need to evaluate the current conditions and the degree and rate of change in these conditions under the no-action alternative. Predictions of future exposure levels – while uncertain – are more amenable to modeling than are responses in local populations or ecosystems, especially when these populations are influenced by many other factors. The emphasis on evaluating current and future exposure also reflects the fact that historical management decisions concerning the river have focused on PCB body burdens in fish. As a result, the assessment has focused on documenting exposure concentrations in water, sediment, invertebrates, and fish and on modeling the temporal change in these concentrations. Because the assessment relies strongly on future predictions of exposure, the effects assessment relies primarily on literature that report on the types of effects that may occur at various exposure levels.

The Revised ERA is not an impact statement. It does not attempt to document the actual degree to which reductions in reproduction, growth or survival have or are occurring. Instead it focuses primarily on the question of whether PCB exposures are at levels that could impair (*i.e.*, pose a risk to) one or more assessment endpoints. The sustainability of fish and wildlife populations depends on many factors that interact with one another over time. The Revised ERA does not attempt to predict how these factors may change in the future or how PCBs may interact with them in influencing survival, growth, or reproduction. It does consider whether PCBs might reduce the fitness of the population thereby making it more susceptible to population decline either due to PCBs alone or in combination with other factors that may negatively affect the population in the future. Although the Revised ERA is not an impact assessment, it makes use of observations on biological conditions in the river and for the wildlife species that rely on the river for habitat and food. Such data (although limited) serve as a useful reality check on the assessment and also provide information on exposure levels that could be harmful to biota.

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*: Sustainability of a benthic invertebrate community, which is 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 2: Measured average and 95% upper confidence limit and modeled average 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 (1999a), Persaud *et al.* (1993), Ingersoll *et al.* (1996), and Washington Department of Ecology (1997) for protection of aquatic life.

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

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

Measurement Endpoint 4: Available field observations on the presence and relative abundance of fish species within the Hudson River as an indication of the ability of the species to maintain populations.

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

Do measured and modeled total PCB dietary doses to insectivorous birds exceed toxicity reference values for adverse effects on reproduction?

Measurement Endpoint 1: Measured average and 95% upper confidence limit and modeled total average 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 measured and modeled TEQ-based dietary doses of PCBs to insectivorous birds exceed toxicity reference values for adverse effects on reproduction?

Measurement Endpoint 2: Measured average and 95% upper confidence limit and modeled TEQ-based 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 toxicity reference values 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 toxicity reference values 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: Measured average and 95th percentile and modeled average PCB concentrations in water (freshwater and saline) compared to chronic 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 along the Hudson River as an indication of the ability of the species to maintain populations.

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

Do modeled total PCB dietary doses to waterfowl exceed toxicity reference values 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 toxicity reference values 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 toxicity reference values 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 toxicity reference values 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 chronic 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 waterfowl along the Hudson River as an indication of the ability of the species to maintain populations.

Assessment Endpoint: Sustainability (i.e., survival, growth, and reproduction) of Hudson River semi-piscivorous/piscivorous bird species

Do modeled total PCB dietary doses to semi-piscivorous/piscivorous birds exceed toxicity reference values 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 semi-piscivorous/piscivorous birds exceed toxicity reference values 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 semi-piscivorous/piscivorous bird eggs exceed toxicity reference values 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 toxicity reference values 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 chronic 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 semi-piscivorous/piscivorous bird populations?

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

Assessment Endpoint: Sustainability (i.e., survival, growth, and reproduction) of local insectivorous mammals

Do modeled total PCB dietary doses to local insectivorous mammals exceed toxicity reference values for adverse effects on reproduction?

Measurement Endpoint 1: Modeled total average and 95% upper confidence limit PCB dietary doses to the little brown bat 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 insectivorous mammals exceed toxicity reference values 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 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 and modeled whole water concentrations exceed criteria and/or guidelines for the protection of wildlife?

Measurement Endpoint 3: Measured and modeled PCB concentrations in water (freshwater and saline) compared to chronic 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 4: Available field observations on the presence and relative abundance of insectivorous mammals along the Hudson River as an indication of the ability of the species to maintain populations.

Assessment Endpoint: Sustainability (i.e., survival, growth, and reproduction) of local omnivorous mammals

Do modeled total PCB dietary doses to local omnivorous mammals species exceed toxicity reference values for adverse effects on reproduction?

Measurement Endpoint 1: Modeled total average and 95% upper confidence limit PCB dietary doses to the raccoon 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 omnivorous mammals exceed toxicity reference values for adverse effects on reproduction?

Measurement Endpoint 2: Measured and modeled TEQ-based average and 95% upper confidence limit PCB dietary doses to the raccoon 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 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 chronic 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 omnivorous wildlife species along the Hudson River as an indication of the ability of the species to maintain populations.

Assessment Endpoint: Sustainability (i.e., survival, growth, and reproduction) of local *semi-piscivorous/****piscivorous mammals***

Do modeled total PCB dietary doses to local semi-piscivorous/piscivorous mammals exceed toxicity reference values for adverse effects on reproduction?

Measurement Endpoint 1: Modeled total average and 95% upper confidence limit PCB dietary doses to the mink and river 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 semi-piscivorous/piscivorous mammals exceed toxicity reference values for adverse effects on reproduction?

Measurement Endpoint 2: Measured and modeled TEQ-based average and 95% upper confidence limit PCB dietary doses to the mink and river 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 semi-piscivorous/piscivorous mammals exceed toxicity reference values for adverse effects on reproduction?

Measurement Endpoint 3: Measured total PCB concentrations in the liver of mink and river otter as compared to concentrations at which impaired reproduction and growth have been observed.

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 chronic 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 semi-piscivorous/piscivorous mammals along the Hudson River as an indication of the ability of the species to maintain populations.

Effect level concentrations are represented by toxicity reference values (TRVs). Toxicity quotients are exceeded when the modeled dose or concentration is greater than the toxicity reference value expressed either as a dose or concentration (*i.e.*, toxicity quotient [TQ] exceeds 1). Toxicity quotients are calculated on a total PCB (trichlorinated and higher) and dioxin-like toxic equivalency (TEQ) basis. The total PCB and TEQ-based toxicity quotients are used as separate measurement endpoints. The total PCB toxicity quotients carry slightly more weight, since data were not available for all dioxin-like congeners, as described in Section 3.1.2.

Exceedance of a TRV is considered indicative of a risk to a population function (*e.g.*, survival, growth, or reproduction) which could reduce the fitness of the local population to sustain itself. Reduced fitness could render the population more susceptible to other natural and/or man-made stresses or could slow a population's recovery following a decline. PCBs themselves are among the stresses imposed upon the local population and may act alone or in concert with other stresses the population may encounter in the future. 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 discussed in Chapter 4.

The potential risks to population functions are evaluated for each representative receptor by considering the species' life-history and the degree to which exposures exceed the toxicity reference values. While the magnitude of an exceedance does not translate directly into the degree to which a population function may be impaired (*e.g.*, a percent reduction in reproduction), it does influence the confidence that can be placed in the conclusions. Where exposure levels greatly exceed toxicity reference values (*e.g.*, orders of magnitude) there is greater confidence that there is a risk to population functions than where toxicity reference values are slightly exceeded. The spatial extent of local populations is based on species ranges along the Hudson River.

As described earlier, 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, 1965). 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 and they do not reflect the degree to which the population is vulnerable to future stresses or the ability of the species to recover from a population decline. 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 (*e.g.*, fishing ban, prey availability). 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.

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; Central Hudson and Gas *et al.*, 1999). 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 Representative Receptors

Wildlife species were selected by USEPA based on discussions with representatives of New York 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.

Receptors were selected to represent animals from different trophic levels, a variety of feeding types, and a diversity of habitats that may be exposed to PCBs from the Hudson River. Specific fish, avian, and mammalian species were selected for evaluation as surrogate species for modeling the range of species likely to be exposed to PCBs in the Hudson River. During the development of the ERA, USEPA invited and incorporated input from stakeholders and the general public on valued species in the Hudson River, which resulted in adding the river otter as a receptor species (see USEPA, 2000b).

While various fish and wildlife species are identified and evaluated, this assessment is applicable to a broad range of animals, as shown in Table 2-11. The selected receptor species serve primarily as recognizable surrogate models for the hundreds of different species that may be exposed to PCBs in Hudson River sediments and emphasizes species that are likely to receive the highest doses of PCBs (*e.g.*, piscivores). 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 exposed species have been adequately considered. The following subsections describe the receptors selected to represent the ecosystem at risk by class (*i.e.*, fish, birds, mammals), with the exception of benthic invertebrate communities.

2.6.1 Macroinvertebrate Communities

Benthic macroinvertebrate communities were selected for evaluation because they inhabit the sediments where PCBs eventually accumulate. As a result, they can experience longer-term exposures. Because these animals are relatively sessile (they are not carried great distances in the water column and they do not migrate to a large degree), they are useful for indicating possible effects of chemicals that may be accumulated in sediments. The invertebrate animals that comprise these communities, such as various insect larvae, crustaceans, other arthropods, mollusks, and worms, provide an important source of food for fish and wildlife. Hence, the benthic macroinvertebrate community is important to the functioning of the entire aquatic community (*e.g.*, fish). Different benthic macroinvertebrates provide the food base for the different size fish (*e.g.*, chironomids are important food for developing bass, but are too small once the fish reach a certain size). Profiles of the dominant macroinvertebrate species/groups found in Hudson River the are provided in Appendix C of USEPA 1999c.

Species found exclusively in the Lower Hudson River, such as blue crab and zebra mussels, may have a large effect on resources in that portion of the river. However, in light of the purpose of the Reassessment RI/FS, which is to evaluate the need to address PCB-contaminated sediments in the Upper Hudson River, USEPA determined that the invertebrate community as a food source for local fish and wildlife was a more relevant assessment endpoint than the health of crayfish, blue crab and zebra mussels as individual species.

2.6.2 Fish Receptors

The Hudson River is home to more than 200 species of fish (Stanne *et al.*, 1996). Eight fish species, representing a range of trophic levels, are evaluated in the ERA (Tables 2-1 and 2-2; see Appendix D in USEPA, 1999c 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 surrogate models 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*); and, striped bass (*Morone saxatilis*).

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 and pumpkinseed, feed primarily on invertebrates, plants, and detritus. Omnivorous fish, such as the brown bullhead, feed indiscriminately upon benthic organisms, emergent vegetation, and, in some cases, small amounts of other fish. Yellow perch and white perch 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 and striped bass. These fish generally feed at higher trophic levels than forage fish. Detailed profiles of the fish species are found in Appendix D of USEPA, 1999c.

2.6.3 Avian Receptors

Five avian receptors (tree swallow (*Tachycineta bicolor*); mallard (*Anas platyrhychos*); belted kingfisher (*Ceryle alcyon*); great blue heron (*Ardea herodias*); and bald eagle (*Haliaeetus leucocephalus*) 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). Detailed life history profiles of the species listed below are provided in Appendix E of USEPA, 1999c.

The tree swallow is a migratory bird that breeds along the Hudson River. As an aerial insectivore, the Hudson River tree swallow feeds primarily on flying insects during the breeding season. USFWS studied the uptake of PCBs and their affect on nesting colonies along the Upper Hudson River (USFWS, 1997) and is currently analyzing data on PCB concentrations in tree swallow adults, nestlings, and eggs (Stilwell, 2000).

The mallard is a surface-feeding duck that feeds by dabbling and filtering through sediments for food. 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 that 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 River.

The great blue heron is the largest wading bird found along the Hudson River. Its long legs, neck, and bill 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 fish, small birds, mammals, 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 and USFWS are measuring chemical contaminant loads in both eagles and prey along the Hudson River (Nye, 2000; Secord, 2000; Stilwell, 2000).

The bald eagle was selected, rather than the osprey, to serve as a receptor species for piscivorous birds feeding on large fish (*e.g.*, largemouth bass). A short profile of the osprey was provided in Appendix G (Threatened, Endangered, and Special Concern Species) of the ERA (USEPA, 1999c). A careful reading of this profile provides some of the reasons why the osprey was not selected as a receptor species, and additional reasons why the osprey was not selected are as follows: 1) There are no known osprey breeding sites along the Hudson, in comparison to bald eagles who have started to breed along the Hudson River in recent years; 2) Ospreys are extremely sensitive to organochlorine pesticide residues, which could confound PCB effects; 3) There are documented occurrences of the bald eagle along the Hudson River in the NY Natural Heritage database (NYSDEC, 1999, 2000), while there are no listings of the osprey for the same area; 4) NYSDEC and USFWS have been collecting Hudson River bald eagle blood, egg, and prey samples for PCB tissue analysis (Stilwell, 2000), while osprey samples are not being analyzed; 5) As noted in the ERA Responsiveness Summary (USEPA, 2000b), the bald eagle was selected rather than other birds of prey because it is on both the federal (threatened) and New York State (endangered) threatened and endangered species lists and there have been recent sightings of it along the Hudson River. In any event, even if the osprey had been selected, the risks are expected to be similar to those calculated for the bald eagle receptor model because of similar exposure parameters and toxicity.

2.6.4 Mammalian Receptors

The potential mammalian receptors found along the Hudson River also represent various trophic levels and habitats (Table 2-9). The four mammals selected to serve as representative receptors in this assessment are the 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 of USEPA 1999c.

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.

The selected wildlife species serve primarily as recognizable surrogates for the many different species that may be exposed to PCBs in the Hudson River. An assessment of exposure to each and every species would not be practical. However, a subset of species that fill various ecological niches was selected to capture a range of exposures, to provide confidence that PCB exposures to lesser known or recognizable species have been adequately considered.

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3.0 EXPOSURE ASSESSMENT

Risks of PCBs within the river to invertebrates, fish, and wildlife depends on the magnitude, extent, and duration of exposure. These characteristics of exposure are examined and quantified within the Exposure Assessment. The Revised ERA evaluates exposure to fish and wildlife using either measured (for current conditions) or modeled (for current and future conditions) concentrations of PCBs. Emphasis is placed in the Revised ERA on measures of exposure in order to evaluate current conditions as well as the degree and rate of change in these conditions under the no-action alternative.

Modeling is required to predict these future exposure levels and combinations of fate and transport as well as bioaccumulation and food chain models were applied for that purpose. Exposures are characterized as either media concentrations, dietary doses, body burdens, and/or egg concentrations depending on the representative receptor. Exposure concentrations are based on either measurements or estimates of the PCB concentrations modeled under site-specific assumptions and expressed as either total PCBs (Tri+) or dioxin-like toxic equivalencies (TEQs).

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. This dataset represents the most complete, recent synoptic dataset for each of the media for total PCBs and the individual congeners. In addition, the NYSDEC dataset was used for piscivorous fish. There are only limited measurements available for the avian and mammalian receptors consisting of a few samples for mink, river otter, tree swallows, and mallard duck. There are no measurements for great blue heron, belted kingfisher, bald eagle, little brown bat, and raccoon. However, USFWS is currently collecting additional data on great blue heron (nestlings and prey), bald eagle (blood, eggs, and prey), and tree swallow (adults, nestlings, and egg), some of which are discussed in Section 3.6 and others are expected to be available in early 2001. NYSDEC is currently collecting and mink and river otter data, which are anticipated to be available in 2001.

Exposures were evaluated for various segments of the river. These segments differ in PCB exposure concentrations. They also vary somewhat in habitat type and ecological receptors. The biggest differences are between the Upper and Lower Hudson River delineated by Federal Dam at Albany. The Upper Hudson River is characterized by a series of pools divided by dams and locks (Figure 1-2). Thompson Island Pool (TI Pool) is one of these pools and is the location where the highest concentrations of PCBs occur in sediments. It is also a primary focus for the Reassessment. Therefore, extensive data have been gathered for this segment of the river. Exposure concentrations of PCBs generally decrease in the Upper Hudson pools below the TI Pool. The river below the Federal Dam is tidal and supports freshwater, estuarine, and anadromous marine species. The distribution of these species within the Lower Hudson depends on salinity and the behavior of the species. For example, the striped bass is an anadromous fish and uses the Lower Hudson as a spawning and nursery ground. Adults swim from the saltwater to the estuarine and freshwater segments of the river to spawn and larvae and juvenile fish eventually migrate back down river. Current conditions in the TI Pool, other pools in the Upper Hudson and in the Lower Hudson were evaluated using information in the vicinity of the ecological sampling stations (Figure 1-2 and 1-3 and Plate 1).

A subset of Upper Hudson pools and Lower Hudson locations were selected to model exposures for predicting future conditions. The selection process took into account gradients in PCB concentrations, spatial factors for individuals and/or local populations of ecological receptors, and technical/practical constraints associated with physical fate and transport modeling. Future exposures were evaluated for three pools in the Upper Hudson: 1) the TI Pool which extends approximately from Fort Edward at River Mile (RM) 195 down to the Thompson Island Dam (RM 188.5); 2) the Stillwater reach which extends from the Northumberland Dam (RM 183.4) to the Stillwater Dam (RM 168.2); and the Waterford reach which extends from the Lock 1 Dam (RM 159.4) down to the Federal Dam (RM 153.9). These three pools or reaches cover a 41 mile stretch of river (RM 195 to RM 153.9). Within each of these pools, exposures were predicted and averaged at a spatial scale of five miles. This scale reflects the balance of the concentration, ecological, and modeling factors described above. The pools delineate to some degree local populations of fish although there could be movement of individuals among pools and recruitment from upper pools to lower pools. The selected pools are also large enough and distributed along the river for a sufficient length (41 miles) that they are appropriate for evaluating exposures to wildlife bordering the river. Certain wildlife forage over larger spatial scales while others use smaller scales (receptor Tables 3-21 to 3-25 and 3-67 to 3-70). However, the spatial dimensions selected to represent exposure are judged to provide a good basis for considering exposures along various reaches of the Upper Hudson for both fish and wildlife.

The Lower Hudson River is not segmented into discrete pools (*i.e.*, there are no dams or locks), but is tidal throughout and exhibits a gradient in salinity from freshwater to estuarine and eventually near-marine salinities where the river flows into the New York Harbor. Predictions of exposure in the Lower Hudson are based on the Farley model, described later in this chapter. Four five-mile segments of the Lower Hudson were used for the Exposure Assessment. These were located at RM 152 (encompassing RM 153.5 - 123.5); RM 113 (encompassing RM 123.5 - 93.5); RM 90 (encompassing RM 93.5 - 63.5); and RM 50 (encompassing RM 63.5 - 33.5). This covers the Hudson from below Albany to Ossining, a river length of about 120 miles.

The Exposure Assessment is organized as follows:

- Quantifying PCB mixtures and TEQs;
- Estimating current and future exposures;
- Exposure concentrations in water and sediments;
- Exposure to benthic invertebrates;
- Exposure to fish;
- Exposure to avian wildlife;
- Exposure to mammalian wildlife;
- Uncertainty and sensitivity in exposure; and
- Examination of exposure pathways based on congener patterns.

3.1 Quantifying PCB Mixtures and TEQs

As discussed in Chapter 2, PCBs are mixtures of compounds that vary in physicochemical properties and toxicity. In order to estimate current and potential future exposures to invertebrates, fish, and wildlife and relate such exposures to available information on toxic effects, decisions were made about how to represent the PCB mixtures. These decisions took into account differences among the various analytical methods used to characterize PCBs in the river.

Total PCB concentrations based on observed data in sediment, whole water, benthic invertebrates, and fish are described as arithmetic averages. The exposure point concentrations (EPC) in this ERA are based on the time- and space-dependency of the PCB concentrations in fish, invertebrates, sediment, and water. The EPC for PCBs in each of these media is based upon modeled projections of future concentrations in each medium (although the models are based upon a large monitoring record). As a result, the typical approach adopted in Superfund risk assessments of calculating an upper confidence limit on a mean concentration (*i.e.*, 95% UCLM), in some instances no longer strictly applies. One reason for its inapplicability is that the 95% UCLM calculation is based upon the notion that the estimate of the mean exposure point concentration from a finite sample set is uncertain and is a function of the number of samples available to estimate the true mean. However, when a model is used to predict the EPC there is no corollary to sample size; with a model an almost unlimited number of model-predicted values can be calculated. As the number of model-projected concentration estimates increases (in time or space), the model mean and model 95% UCLM converge to the same value because the 95% UCLM reflects statistical uncertainty rather than uncertainty in the modeling estimates themselves. Only if model inputs are varied to reflect environmental variability of the model input parameters, and repeated model estimates of the mean are obtained over the range of parameters, can an average and 95% upper confidence limit on the modeled means be calculated.

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:

$\overline{X}$ = arithmetic average of the individual natural log-transformed concentrations from the data;

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

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

$H_{1-\alpha}$ = $H_{1-\alpha}$ is a function of the standard deviation of the log-transformed data or model results and the number of samples in the data set or number of simulations in the modeling. $H_{1-\alpha}$ was taken from a standard table of

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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.

3.1.1 Quantifying PCB mixtures as Tri+ PCBs

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 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 and that the Tri+ congeners represent total PCBs in biota (see, RBMR USEPA, 2000a, Book 3, Chapter 4). The fate and transport of PCBs in the river and future environmental concentrations were predicted as Tri+ using the HUDTOX model described later in this chapter. Tri+ was used as a common metric for representing exposure levels in water, sediments, and biota. It is acknowledged that the composition of PCBs within the Tri+ group can vary due to differences in fate and transport as well as accumulation into biota. The implications of such variations is discussed later in this chapter. However, use of a PCB Tri+ metric for exposure is consistent with much of the available toxicological literature for PCB effects expressed as total PCBs or Aroclors.

3.1.2 Quantifying Toxic Equivalencies (TEQ)

An objective of the Exposure Assessment is to estimate exposure concentrations or doses that can be related to the toxicity of the compounds. As discussed in the next chapter (Chapter 4), PCBs may elicit a variety of effects including those that are mechanistically similar to dioxin, although the potency is much less. There are twelve individual PCB congeners that are thought to elicit toxicity via a dioxin-like mechanism. 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 to the aryl hydrocarbon receptor (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). The toxicological aspects of the method are further described in Chapter 4.

The 1993 EPA dataset provides monitoring data for eleven of the individual congeners (with appropriate caveats as described later in the chapter.) However, the models used in this assessment (HUDTOX and FISHRAND for the Upper Hudson River and the Farley model and FISHRAND for the Lower Hudson River) are not designed to predict future concentrations for each of the individual dioxin-like congeners. There are not enough data available to calibrate and constrain the models for each of these congeners. Approaches used to predict future concentrations on a TEQ basis are described later in this Exposure Assessment.

There are a number of data quality issues that needed to be addressed in order to make use of the available congener information in risk assessment. The TEQ congeners (listed in Tables 3-1 and 4-2) include: BZ#77, BZ#81, 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 of USEPA, 1999c). The data usability report (Appendix I of USEPA, 1999c) for the ecological sampling program (sediments, fish, and invertebrates) 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 Laboratories), 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, primarily due to detection at concentrations below the calibrated quantification limit and/or exceedances in the dual column precision criteria (see USEPA, 1999c Appendix I p. I-33). These data were determined to be usable for the ecological risk assessment given the data quality objectives of the sampling program, which were established in the Phase 2B Sampling and Analysis Plan/Quality Assurance Project Plan (USEPA, 1993a). A relatively small percentage of the PCB data (925 of the 59,063 congener measurements, or 1.6%) were rejected due to exceedence of quality control criteria (see, USEPA, 1999c Tables I-9 to I-12).

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 under represents 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. This contrasts with how non-detect values were addressed in the rest of the ERA. In all other analyses, non-detects were assumed to be zero if more than 85% of the samples from a given location were below the detection limit. If concentrations above the detection limit were detected in more than 85% of the samples, non-detect samples were assumed to have concentrations at one-half of the non-detect value (see, *Value 2* in USEPA, 2000e). As a result of considering the frequency of detection (*i.e.*, congener presence), USEPA used values that were less conservative than using one-half the detection limit for all non-detect samples. The effect of using half the detection limit, or setting BZ#126 equal to zero, is discussed in the uncertainty analysis.

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 of USEPA 1999c. Because 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 (USEPA 1999c) 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 this analysis suggests it is on an order of magnitude basis. The fraction of the Tri+ concentration for each medium that is represented by TEQs is provided in Table 3-2. The methodology used to calculate these fractions is discussed in Section 3.3.3.

3.2 Estimating Current and Future Exposures

The ERA examines risks associated with current conditions as well as how these risks will change over time. Current exposure conditions rely on a combination of measurements and models, which will be discussed for each of the major groups of representative receptors. Fate and transport models are used to predict future exposure levels on water and sediments. Bioaccumulation models are used to translate these future exposure levels into body burdens of PCBs for invertebrates and fish. These estimates of body burdens are also used to predict future exposures to wildlife that use invertebrates and fish as food sources.

Predictive models play an important role in the exposure assessment and much of the effort in the Reassessment has been focused on developing, calibrating, and applying these models. The models that are used to describe the fate and transport and bioaccumulation of PCBs in the river are described below.

3.2.1 Upper Hudson River Models

Future risks in the Upper Hudson River are characterized using the HUDTOX and FISHRAND models, as described in the Revised Baseline Modeling Report (RBMR) (USEPA, 2000a). A large body of information from site-specific field measurements (documented in Hudson River Database Release 5.0), laboratory experiments and the scientific literature was synthesized within the models to develop the PCB transport and fate and the PCB bioaccumulation models. Data for these models were taken from numerous sources including USEPA, NYSDEC, NOAA, US Geological Survey (USGS), and General Electric. The proposed modeling approach and preliminary demonstrations of model outputs were made available for public review in the Preliminary Model Calibration Report, which was issued in October 1996.

The modeling framework was revised based on a peer review held in September 1998 and public comment, as well as the incorporation of additional data. The baseline modeling effort and results were documented in the Baseline Modeling Report (BMR) issued in May 1999 (USEPA, 1999b). USEPA decided to revise the BMR to reflect changes to the models based on public comment and additional analyses that were conducted. The Revised BMR (USEPA, 2000a) included model refinements, additional years of data, longer model forecasts, validation to an independent dataset, and additional model sensitivity analyses.

The Upper Hudson River Toxic Chemical Model (HUDTOX) was developed to simulate PCB transport and fate for 40 miles of the Upper Hudson River from Fort Edward to Troy, New York. HUDTOX is a transport and fate model, which is based on the principle of conservation of mass. The fate and transport model simulates PCBs in the water column and sediment bed, but not in fish. It balances inputs, outputs and internal sources and sinks for the Upper Hudson River. Mass balances are constructed first for water, then solids and bottom sediment, and finally PCBs. External inputs of water, solids loads and PCB loads, plus values for many internal model coefficients, were specified from field observations. Once inputs are specified, the remaining internal model parameters are calibrated so that concentrations computed by the model agree with field observations. The forecast baseline conditions used in this revised risk assessment were revised to a constant load condition at Rogers Island of 16 kg/day. This load was based on the 1996-1999 GE monitoring data obtained at Rogers Island and nominally corresponds to a concentration of 13 ng/L at this location. The original ERA (USEPA, 1999c) assumed a constant concentration at Rogers Island of 10 ng/L. Model calculations of forecasted PCB concentrations in water and sediment from HUDTOX are used as inputs for the forecasts of the FISHRAND bioaccumulation model.

The FISHRAND model is based on the peer-reviewed uptake model developed by Gobas (1993) and Gobas *et al.* (1995) and provides a mechanistic, process-based, time-varying representation of PCB bioaccumulation. This is the same form of the model that was used to develop criteria under the Great Lakes Initiative (USEPA, 1995a). The FISHRAND model incorporates distributions instead of point estimates for input parameters, and calculates distributions of fish body burdens from which particular point estimates can be obtained, for example, the 25^{th} percentile, median, or 95th percentile. FISHRAND was used to predict all future fish PCB body burdens, with the exception of striped bass, used in this assessment. The Revised Baseline Modeling Report was the subject of an external peer review during 2000 and found to be generally acceptable with some revisions.

3.2.2 Lower Hudson River Models

Four separate models are used to calculate the exposure point concentrations in the lower river. The HUDTOX fate and transport model for the upper river provides the flux of PCBs over the Federal Dam into the Lower Hudson River. These results represent an external input to the Lower Hudson River fate and transport model (*i.e.*, the Farley model). The Farley fate-and-transport model (Farley *et al.*, 1999) developed at Manhattan College specifically for the Lower Hudson River is used to generate the water and sediment concentrations for the Lower Hudson River risk assessments. The Farley bioaccumulation model (updated per Cooney, 1999) is then applied to yield PCB concentrations in striped bass. The water and sediment concentrations from the Farley fate-and-transport model are also input for the FISHRAND bioaccumulation model to generate the lower river concentrations for pumpkinseed, spottail shiner, yellow perch, brown bullhead, largemouth bass and white perch (*i.e.*, all other fish species examined in this report).

3.2.2.1 Use of the Farley Models

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

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

In this application, the flux over the Federal Dam for each homologue is derived from the flux of Tri+ PCBs given by the HUDTOX model. The HUDTOX model was developed for the Upper Hudson River and is described in the Revised Baseline Modeling Report (USEPA, 2000a). The HUDTOX model results used to estimate external loads for the Lower Hudson were obtained from LTI (LTI, 1999a and 1999b). In order to use the Tri+ flux given by the HUDTOX model, a basis for conversion of the Tri+ load to individual homologue loads was required. This was accomplished through the use of Tri+ to homologue conversion factors for each homologue group. These factors were determined by analyzing the available USEPA and General Electric (GE) water column data. Table 3-3 provides the means of conversion for each homologue during both the calibration and forecast periods. The complete analysis can be found in Appendix A of the ERA Addendum (USEPA, 1999e).

The Farley models were originally designed to run for a 15 year period, 1987-2002. Because a 70-year forecast of concentrations is required for the human health risk assessments, the models are run in 15 year increments with the final conditions in each model segment and each modeled species becoming the initial conditions for the next 15 years. For this assessment, only the model output from the period 1993 to 2018 was required.

The major external PCB load to the Lower Hudson, *i.e.*, the load from the Upper Hudson, was originally estimated using the 70-year forecast from the HUDTOX model in the BMR (USEPA, 1999b), assuming a constant concentration at Rogers Island of 10 ng/L (LTI, 1999a and 1999b). The concentrations at the Federal Dam were obtained from HUDTOX under this condition and the annual Tri+ PCB load to the Lower Hudson River calculated. However, based on recent data, a constant load condition at Rogers Island of 16 kg/day, corresponding to a concentration of 13 ng/L, was used in this assessment. The ratios of the annual Tri+ PCB loads from this model run to the annual load estimates used in the ERA Addendum (USEPA,1999e) provide a means of comparing the estimates and adjusting the Lower Hudson River model predictions without rerunning the fate, transport and bioaccumulation models. These annual ratios are multiplied by the original Lower Hudson River water, sediment and fish modeled concentrations (USEPA, 1999e) to generate the current estimated concentrations. Although it is unlikely that the models would exhibit a purely linear response to the magnitude of the Upper Hudson River PCB load, this method provides a first order approximation of the actual response. The prior and current predicted annual loads and their ratio are presented in Table 3-3a. For the entire period ecological modeling period (1993-2018) the ratio of the predictions remains close to one with the ratio ranging from 0.98 to 1.18, with the exception of 1998 which has a ratio of 2.3. This ratio is unreasonably high and results from high flow events that occurred in 1998. The concentrations from 1994 are substituted for this year, because using the ratio results in higher concentrations than previous years where the loadings were even greater than in 1998.

In addition to examining the forecast from the Farley models, an examination of the Farley model results was also performed for the calibration period 1987 to 1997. In this examination, the original calibration curve developed by Farley *et al.* was compared with model results produced using the HUDTOX model loads to the Lower Hudson. In this fashion, the effects of any differences in Upper Hudson load assumptions could be examined.

Differences from the application of the FISHRAND model to the upper river to the lower river are:

- Water and sediment concentrations estimated from the Farley fate-and-transport model are used;
- The percent lipid distribution (based on data) is significantly different for the lower river largemouth bass with an average lipid content of 2.5% in the lower river versus 1.3% in the upper river;
- The total organic content value for sediment segments used in the Farley fate-and-transport model is used; and,
- The K_{OW} values specified in USEPA (2000a) for the Upper Hudson River below the Thompson Island Dam are applied to the lower river.

These adjustments are required to make the FISHRAND model specific to the Lower Hudson River.

3.2.2.2 Estimation of Striped Bass Body Burdens in the Lower Hudson

The Farley bioaccumulation model was used to estimate PCB levels for striped bass which migrate up to food web region 2. The model does not provide striped bass concentrations in region 1. In order to estimate striped bass body burdens for the human health and ecological risk assessments in region 1, the largemouth bass body burdens estimated from the FISHRAND model are multiplied by the ratio of striped bass to largemouth bass body burdens. Observed striped bass and largemouth bass concentrations from NYSDEC data are used to construct the ratio at river miles 152 and 113. The averaged concentrations for each year and species are shown in Table 3-4. White perch concentrations are also presented in the table for comparison.

A comprehensive discussion of the models used to create a 70-year forecast for the Lower Hudson River is contained in the ERA Addendum (USEPA, 1999e). Appendix A: Conversion from Tri+ PCB loads to dichloro through hexachloro homologue loads at Federal Dam of the ERA Addendum is incorporated by reference into this Revised ERA. In general, fish body burdens estimated by the models tended to fall below actual measurements by about 16 percent. The model results were able to capture the general trend of decreasing PCB concentration with time and distance down river, but not year-to-year variability. The agreement is considered sufficient for use in this ERA and the Revised HHRA.

3.3 Exposure Concentrations in Water and Sediments

Invertebrates, fish, and wildlife can be exposed to PCBs present in water and sediments. Current conditions for exposure are based on available measurements from the USEPA Phase 2 dataset, which represents the most complete synoptic measurements of any dataset. Future conditions were evaluated using the models described below. Modeled concentrations of water and sediments are used later in the Exposure Assessment to derive body burdens for PCBs in invertebrates and fish.

3.3.1 Measured Concentrations in Water and Sediments

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 (Plate 2). 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 RM 181.3 and 168.3 during April, June, and August. The area just above the Federal Dam (RM 154) was characterized by water samples collected from RM 156.5 in April, May, June, July, August, and September. Samples collected in April and August from RM 151.7 and 125 were used to obtain average water concentrations for ecological stations at RM 143.5 and 137.2. RM 122.4, RM 113.8, and RM 100 were characterized by average water column concentrations over RM 125 and RM 77 from April and September. The final four ecological stations were characterized by average water column concentrations at RM 77 in April and September. Water concentrations are expressed on a whole water basis (particulate plus dissolved) and are shown in Table 3-5. All water samples were above the detection limit. RM 77 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 3-5 also provides whole water concentrations of PCBs described as TEQs. Different TEFs are applied to the water concentrations depending on whether the receptor is mammalian or avian. The TEF used is a weighted TEF from the analysis contained in Appendix J of USEPA 1999c. Consequently, separate columns are provided for avian- and mammalian-based TEQ water column concentrations.

Sediment data were collected at 19 locations in the Hudson River during the 1993 USEPA ecological sampling program (see Appendix B of USEPA 1999c). 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 3-6 provides average 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 3-6 also provides observed sediment concentrations described as TEQ. Different TEFs 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.

3.3.2 Modeled Concentrations in Water and Sediments

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 Revised Baseline Modeling Report (USEPA, 2000a). Table 3-7 provides the predicted average whole water concentrations on a Tri+ total PCB basis.

Table 3-7 also provides the predicted average whole water concentrations expressed on a TEQ basis. These values were obtained by multiplying the Tri+ predictions in Table 3-7 by the toxic equivalency weighting factors in Table 3-2 to describe the proportion of the Tri+ total expressed as a TEQ.

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 Revised Baseline Modeling Report (USEPA, 2000a). Table 3-8 provides the predicted average sediment concentrations on a Tri+ total PCB basis and Table 3-9 provides organic carbon normalized sediment concentrations.

3.3.3 Estimating Future Baseline TEQ Concentrations

Table 3-8 provides the predicted average sediment concentrations expressed on a TEQ basis for birds and mammals. These values were obtained by multiplying the Tri+ predictions in the first seven columns of Table 3-8 by the toxic equivalency weighting factors to describe the proportion of the Tri+ total expressed as a TEQ. As discussed previously, the HUDTOX and FISHRAND models do not predict individual PCB congener concentrations in environmental media. 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) in the 1993 USEPA dataset. 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). The "TEF-based factor" (derived for each individual location) was the same within the upper river and the same within the lower river, but different between the two sections. 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.

3.4 Exposure to Benthic Invertebrates

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 shortnose sturgeon, and represent a portion of the diet of other fish species, including largemouth bass and white perch.

Benthic invertebrate concentrations for 1993 are obtained from the measurements in the USEPA Phase 2 dataset. Predicted distributions of benthic invertebrate concentrations for the period 1993 to 2018 are estimated in the FISHRAND model assuming steady-state conditions between the lipid content of invertebrates and the organic carbon of sediment (see Equation 3-2).

3.4.1 Observed Benthic Invertebrate Concentrations

Data on benthic invertebrate communities and PCB body burdens were collected at the ecological monitoring stations, all located in the main stem of the river (Figures 1-2 and 1-3). 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 of USEPA 1999c also supports this assumption. Table 3-10 provides average benthic invertebrate concentrations used in this analysis.

Table 3-10 also provides observed benthic invertebrate concentrations described on a TEQ basis. Different TEFs 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.

3.4.2 Modeled Benthic Invertebrate Concentrations

Benthic invertebrate concentrations of PCBs for the period 1993 to 2018 were predicted assuming equilibrium partitioning between organic carbon in sediment and lipid in benthic invertebrates. Distributions were assigned for organic carbon and lipid in benthic invertebrates:

$$C_{Invert} = \frac{C_{Sed}}{TOC} * Lip_{Invert} \qquad \text{Equation 3-2}$$

where:

C_{Invert} = the concentration of PCB in an organism (µg/g wet weight);

C_{Sed} = the concentration of PCB in sediment (µg/g wet weight);

TOC = Total organic carbon in sediment (fraction); and

Lip_{Invert} = Percent lipid in invertebrates (fraction)

Table 3-11 provides the predicted average benthic invertebrate concentrations expressed on a total PCB basis. Table 3-11 also provides the predicted average 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 Section 3.1.2.

3.5 Exposure to Fish

Fish are exposed to PCBs in water and sediments both directly as well as indirectly through the food chain. PCB concentrations in fish are described as wet weight or lipid normalized tissue concentrations. Data from males and females were combined to provide an estimate of exposure for each species.

To address the importance of nearshore habitats for fish species (such as pumpkinseed, spottail shiner, brown bullhead, white perch, and yellow perch) using the available data, water column concentrations in the Thompson Island Pool were weighted toward nearshore areas. However, water column concentrations for locations downstream of Thompson Island Pool were averaged across the river. Lateral gradients are of greater importance in the lower Thompson Island Pool and of less importance downstream of Thompson Island Pool because (1) downstream dams have generally smaller, narrower pools plus higher flows, so lateral mixing would be increased; (2) the lateral gradient in the Thompson Island Pool is strong when flows are low because upstream water is relatively clean; because the lower reaches receive the relatively contaminated Thompson Island Pool water as their upstream water, the lateral gradients are not as strong; (3) the density of hot spots and surface sediment concentrations are generally lower downstream, thus the lateral gradient should be less; and (4) lateral gradients are likely enhanced by the numerous shallow macrophyte beds in the Thompson Island Pool.

Body burdens in fish may change seasonally as lipid pools in the fish increase or decrease and as the activity of the fish changes with changes in water temperature. There may also be seasonal differences in exposure concentrations that reflect temperature as well as the activity of invertebrates used as food items or which mix sediments. This Exposure Assessment focuses on warmer water periods (late spring to early fall) when fish are expected to be most active and when spawning occurs for most of the fish species considered in this assessment. Thus, the estimates of PCB body burdens reflect this time of the year.

3.5.1 Observed Fish Concentrations

Fish have been collected and analyzed for PCB concentrations on a number of occasions. To represent "current exposures" data are used for body burdens in fish 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 3-12 with avian- and mammalian-based TEQs. Table 3-13 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. Tables 3-13a and 13b provide 1998 NYSDEC upper river sampling data for wet weight and lipid-based concentrations, respectively. Figures 3-2 and 3-3 provide wet weight and lipid-normalized average PCB concentrations in several species across several river miles based on the NYSDEC data.

Average largemouth bass concentrations at RM 189 range from 94 to 28 ppm wet weight from 1993 to 1996, respectively. The corresponding maximum concentrations range from 346 to 57 ppm over that same time period. Average brown bullhead concentrations range from 26 to 16 ppm wet weight from 1993 to 1996, respectively, with the corresponding maximum concentrations ranging from 104 to 19 ppm wet weight. Average wet weight concentrations at RM 168, near Stillwater, are 17 to 13 ppm for largemouth bass and 13 to 9 ppm for brown bullhead. Maximum concentrations are 38 to 29 ppm for the largemouth bass and 27 to 19 ppm for brown bullhead. In the Lower Hudson River, average and maximum largemouth bass concentrations at RM 113 range from 11 to 9 ppm wet weight and 34 to 27 ppm wet weight, respectively, from 1993 to 1996.

Table 3-14 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. Wet weight concentrations in the Lower Hudson River during 1996 range from 1.3 ppm wet weight at RM 12 to 4.9 ppm wet weight at RM 152, just below 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.

For the lower Hudson River observed data are used to compare to toxicity reference values for striped bass for 1993 – 1996.

3.5.2 Modeled Fish Concentrations

Fish concentrations of PCBs for the period 1993 to 2018 were predicted using the FISHRAND model (USEPA, 2000a), with the exception of the striped bass which was predicted using the Farley *et al.* (1999) model. Phase II fish data were of limited use in development of the FISHRAND model because all the largemouth bass were too small to be piscivorous and in fact, were smaller than their prey (pumpkinseed). Therefore, primarily NYSDEC data were used in FISHRAND. Tables 3-15 through 3-19 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, yellow perch, and striped bass) expressed on a wet weight basis for Tri+ total PCBs at the upper and lower river modeling locations.

As described above, the model is designed to predict PCB concentrations in the standard fillet of piscivorous fish. 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. 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, the following procedure was used:

1. Take the model-predicted 25th, 50th, and 95th percentiles;

2. Log-transform the model output for the 25th, 50th and 95th percentiles and plot the results against the inverse of the normal cumulative distribution, yielding a straight line;

3. Obtain the parameters of the regression to estimate a μ and geometric standard deviation (GSD) where μ equals the intercept * GSD and GSD equals 1/slope; and,

4. Obtain the mean (expected value, or E[x], of the distribution) as $E[x] = e^{\ln(x)+\sigma^2/2}$ where σ equals the GSD.

3.6 Exposure to Avian Wildlife

3.6.1 Measured Concentrations in Birds

USFWS conducted PCB monitoring in tree swallow eggs and nestlings during 1993 and 1994 (USFWS, 1997) and is currently analyzing sample from tree swallows, great blue herons, bald eagles, and bald eagle prey collected from 1997-1999. A summary of some of their results is provided in Table 3-20a. One mallard sample from river mile 173 was presented in the USFWS database. USFWS, NYSDEC, and the United States Geological Survey (USGS) are currently analyzing bald eagle, great blue heron, and tree swallow tissue, egg and prey samples. Several preliminary reports are available (USGS, 2000a; 2000b; 2000c; and 2000d) that summarize the results of the data collected thus far. The reports provide results for congener specific PCBs in bald eagle blood (USGS, 2000a), dioxins and furans in bald eagle fat, plasma, and prey items (USGS, 2000b), total PCBs and selected congeners in great blue heron nestling brains, tree swallow nestlings, and tree swallow adults (USGS, 2000c), and dioxins, furans, non-*ortho* PCBs in bald eagle blood (USGS, 2000d). These results are not directly comparable to the toxicity reference values derived in Chapter 4 which are expressed as dietary doses in mg/kg-day based on the way in which data are typically presented in the studies. However, these data (Table 3-20a) show that total PCBs in the brains of great blue heron nestlings obtained from Castleton Island in the Lower Hudson River range from 35 to 560 ng/g wet weight (USGS, 2000c). One great blue heron sample, obtained from Saratoga National Historic Park, measured 1,000 ng/g total PCBs wet weight. Total PCB concentrations in tree swallow nestlings obtained from various locations in the Upper Hudson River valley during 1998 and 1999 ranged from

1,800 to 12,000 ng/g total PCBs wet weight, with the highest concentrations seen at the Special Area 13 and Remnant 4 Sites. . Lower Hudson River concentrations were lower, ranging from 170 to 890 ng/g total PCBs wet weight. Adult tree swallows from the Upper Hudson River ranged from 2,300 to 16,000 ng/g wet weight (USGS, 2000c).

The bald eagles were obtained primarily from locations in the Lower Hudson River (roughly RM 80 to 138) and total PCB concentrations ranged from 471 to 14,240 ng/g in serum and 214 to 755 ng/g in whole blood. One sample, taken near Lock 1 in the Upper Hudson River (approximately RM 159.4), showed total PCB concentrations of 1,009 ng/g in serum.

During the early 1980's, NYSDEC conducted some limited monitoring throughout New York State of PCBs in the tissues of peregrine falcons, great blue heron, mallard ducks, and several other species. However, these data are not adequate to assess potential exposures and effects from Hudson River sources. The toxicity reference values derived in this assessment are expressed as dietary doses in mg/kg-day, which are not directly comparable to specific tissue concentrations.

3.6.2 Avian Exposure Models

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

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

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 3-21 to 3-25.

3.6.2.1 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:

$$ADD_{Water} = \frac{(PCB_{Surfacewater} \times WI_{Receptor})}{BW_{Bird}} \times (FE) \qquad \text{Equation 3-4}$$

where:

ADD_{Water}	=	Daily dose of PCBs from consuming Hudson River surface water (mg/kg/day);
$PCB_{Surfacewater}$	=	Mean PCB exposure concentration (mg/L) in surface water;
$WI_{Receptor}$	=	Water ingestion rate (L/day) for avian receptor;
FE	=	Areal forage effort (unitless) as fraction of home or forage range; and,
$BW_{Receptor}$	=	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. Many of the avian receptors have both resident and migratory populations in the Hudson River Valley. Resident populations are considered to be at greater risk (due to breeding and growth) and therefore are evaluated in the exposure assessment.

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

$$WI_{(Birds)} = (0.0582 * BW^{0.67}) \qquad \text{Equation 3-5}$$

where:

$WI_{(Bird)}$	=	Bird specific 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).

3.6.2.2 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:

$$ADD_{Sediment} = \frac{(PCB_{Sediment} \times FS_{Media} \times IR_{Total})}{BW_{Bird}} \times FE \qquad \text{Equation 3-6}$$

where:

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

3.6.2.3 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 ingestion rate (kg/day on a wet weight basis) for each receptor was referenced from the available literature or developed using the field metabolic rate (FMR) (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 field metabolic rate, the typical diet composition for the Hudson River populations, and the average metabolizable energy content of the diet.

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

$$FMR = 2.601(BW)^{0.640} \qquad \text{Equation 3-7}$$

where:

FMR	=	Field metabolic rate (kcal/day);
BW	=	Body weight of avian receptor (gm); and

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):

$$ME = GE \times AE \qquad \text{Equation 3-8}$$

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 FMR and ME_{Ave} for the specific diet:

$$IR_{Total} = \frac{FMR_{Receptor}}{ME_{Ave} \times BW_{Receptor}} \times 0.001 \qquad \text{Equation 3-9}$$

where:

IR_{Total}	=	Species-specific total ingestion rate for avian receptor (kg/day);
$FMR_{Receptor}$	=	Species-specific field metabolic rate (kcal/day) for avian receptor;
ME_{Ave}	=	Average metabolizable energy content of dietary component (kcal/gm wet wt);
$BW_{Receptor}$	=	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. This approximation is appropriate for purposes of determining exposure because the expose is expressed as an average concentration in fish of a given size. Different age classes of fish have different feeding strategies, but within a particular age-class, feeding strategies are similar. For example, largemouth bass above 25 cm in length all feed similarly, but differently from fish smaller than that size range (see Appendix A of USEPA, 2000a).

Ingestion rates of forage fish and benthic/piscivorous fish are based upon size selectiveness observed in the diet (see Appendix E of USEPA, 1999c). The average daily dosage of PCBs to the avian receptor from the fish-derived portion of the diet is expressed as:

$$ADD_{Fish} = \frac{(PCB_{Fish} \times IR_{Total} \times PD_{Fish})}{BW_{Receptor}} \times FE \qquad \text{Equation 3-10}$$

where:

ADD_{Fish}	=	Average daily dietary dose of PCBs from ingestion of fish (mg/kg/day wet wt);
PCB_{Fish}	=	Average concentration of PCBs observed in fish tissue (mg/kg wet wt);
IR_{Total}	=	Total 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

$BW_{Receptor}$ = 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:

$$ADD_{Invert} = \frac{(PCB_{Invert} \times IR_{Total} \times PD_{Invert})}{BW_{Receptor}} \times FE \qquad \text{Equation 3-11}$$

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);
IR_{Total}	=	Total 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,
$BW_{Receptor}$	=	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:

$$Conc_{macro} = (K_{ow} \ x \ Conc_{diss} \ x \ Lipid_{macro}) \qquad \text{Equation 3-12}$$

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). Linear relationships between the plant-water and fish-water bioconcentration factors and the octanol-water partition coefficient have been demonstrated, indicating that plant-water and fish-water exchanges are largely controlled by the chemical's tendency to partition between the lipids of the plants and water. Uptake of PCBs from sediment sources may also be significant but there is less quantitative information available to characterize this relationship. Equation 3-12 is likely to provide protective estimates of bioconcentration.

Scientific literature and wildlife biologists 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 United States, and populations from other regions of the contiguous United States. Wherever possible, multiple data sources are 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 is used in the exposure assessment.

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. 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 are 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 are 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 is 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 (1999b) 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 Lower Hudson River populations appear to be dominated by fish. Fish species captured tend to be larger species and the diet is restricted to larger fish. Eagles overwintering north of Federal Dam may feed extensively on waterfowl (Nye, 2000). In the absence of data on PCB concentrations in waterfowl, a diet of 100% fish (as piscivorous fish) derived from the river is applied to both upper and lower river eagles in the exposure assessment.

3.6.2.4 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 and therefore a temporal exposure factor of one is applied.

3.6.2.5 Biomagnification Factors for Predicting Egg Concentrations

Biomagnification factors (BMFs) from the literature are used to predict the concentration of total PCBs and TEQ in the eggs of piscivorous birds from the mean concentration in fish. Biomagnification factors are typically based on field studies in which measured egg concentrations are compared to synoptic measured sediment, benthic invertebrate, or fish concentrations. The same uncertainties that apply to the field studies of bioaccumulation are also applicable here: it is not known whether the denominator of the calculated BMF ratios represent the true exposure concentrations that led to the accumulation of PCBs in the eggs of avian receptors.

Total PCB Biomagnification Factors

Table 3-26 provides a summary of biomagnification factors from the literature. A biomagnification factor of 30 has been used to predict total PCB concentrations in piscivorous bird eggs from concentrations in prey fish (USEPA, 1998a). The only value obtained from field data specifically for bald eagles is 28, as presented in Giesy *et al.*, (1995). This value (28) was used for the belted kingfisher and bald eagle in this assessment. No information was provided on uncertainty or variability in this estimate. A mean factor of 8 was obtained for the great blue heron (Halbrook *et al.*, 1999) with a range from 0.3 to 44, and is used for that species in this report. Biomagnification factors across all species for total PCBs are relatively similar, ranging from 8 to 53 on a mean basis, suggesting a less than order of magnitude range in BMFs.

A biomagnification factor of 3 is used to estimate benthic invertebrate to egg concentrations for the tree swallow for total PCBs based on the USFWS data (Table 3-26a). The range from the observed data was 0.4 to 11.7 based on 11 samples. The USFWS data provided one mallard sample from which to estimate a biomagnification factor. From this, a biomagnification factor of approximately 3 on a total PCB basis was obtained.

TEQ Biomagnification Factors

Table 3-26 also provides a summary of TEQ biomagnification factors from the literature. A biomagnification factor of 19 has been used to predict total PCB concentrations in piscivorous bird eggs from concentrations in prey fish (Kubiak and Best, 1993). This value is for TCDD-equivalents from northern pike to bald eagle. No further information was provided, and nothing was stated about uncertainty in this estimate. Giesy *et al.* (1995), referring to the BMF of 19, states that "the uncertainty in the BMF accumulation of TCDD-EQ from fish to the eggs of bald eagles is not as great as that for estimates of the NOAEC," and they estimate the uncertainty in the NOAEC as being two orders of magnitude. Braune and Norstrom (1989) provide a TCDD BMF of 21 with an apparent standard deviation of 5 for alewife to herring gull eggs. USEPA (1994) provides a value of 200, based on Dr. Giesy's comments on the report, based on a maximum observed bald eagle egg concentration as compared to an average forage fish concentration. This information suggests the range of BMFs across all species is less than one to 200. Based on these data, a TEQ biomagnification factor of 19 was used for the belted kingfisher, great blue heron, and bald eagle.

A biomagnification factor of 7 was used to estimate benthic invertebrate to egg concentrations for the tree swallow on a TEQ basis using USFWS data (Table 3-26a). The range from the observed data was 2.2 to 20.2 based on 7 samples. The USFWS data provided one mallard sample from which to estimate a biomagnification factor. From this, a biomagnification factor of 7 on a TEQ basis was obtained. These biomagnification factors are based upon a diet of mixed insects. Birds that preferentially feed on odonata (*i.e.*, dragonflies and damselflies) are likely to accumulate higher levels of PCBs, as the BMFs for this group range up to an order of magnitude greater than for total insects.

3.6.3 Exposure Estimates for Avian Wildlife on a Total (Tri+) PCB Basis

Tree Swallow

Tables 3-27 and 3-28 provide the expected average daily dose and 95% UCL daily dose, respectively, on a total PCB basis for the tree swallow from water and dietary sources based on 1993 data for water and benthic invertebrate concentrations. Table 3-29 and 3-30 present the expected average daily dose in the upper and lower river, respectively, for the modeling period 1993 – 2018. These tables also all show the predicted egg concentrations based on a total PCB biomagnification factor derived from the Phase 2 dataset.

Mallard Duck

Tables 3-31 and 3-32 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. Table 3-33 and 3-34 present the expected average daily dose in the upper river and lower river, respectively, for the modeling period 1993 – 2018. These tables also show the predicted egg concentrations based on a total PCB biomagnification factor derived from the Phase 2 dataset. These biomagnification factors are based upon a diet of mixed insects. Birds that preferentially feed on odonata are likely to accumulate higher levels of PCBs, as the BMFs for this group range up to 27 percent greater than for total insects.

Belted Kingfisher

Tables 3-35 and 3-36 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. Tables 3-37 and 3-38 present the expected average daily dose in the upper river and lower river, respectively, for the modeling period 1993 – 2018. These tables also show the predicted egg concentrations based on a total PCB biomagnification factor obtained from the literature (Giesy *et al.*, 1995).

Great Blue Heron

Tables 3-39 and 3-40 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. Tables 3-41 and 3-42 present the expected average daily dose for the modeling period 1993 – 2018 for the upper and lower river, respectively. These tables also show the predicted egg concentrations based on a total PCB biomagnification factor obtained from the literature (Halbrook *et al.*, 1999).

Bald Eagle

Tables 3-43 and 3-44 provide the expected average daily dose and 95% UCL daily dose on a total PCB basis for the bald eagle from water and dietary sources based on 1993 data for water and piscivorous fish concentrations. Tables 3-45 and 3-46 present the expected average daily dose for the modeling period 1993 – 2018 for the upper and lower river, respectively. These tables also show the predicted egg concentrations based on a biomagnification factor obtained from the literature (Giesy *et al.*, 1995).

3.6.4 Exposure Estimates for Avian Wildlife on a TEQ Basis

Tree Swallow

Tables 3-47 and 3-48 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. Tables 3-49 and 3-50 present the expected average daily dose for the modeling period 1993 – 2018 for the upper and lower river, respectively. These tables also all show the predicted egg concentrations based on a TEQ-based biomagnification factor derived from the Phase 2 dataset.

Mallard Duck

Tables 3-51 and 3-52 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. Tables 3-53 and 3-54 present the expected average daily dose for the modeling period 1993 – 2018 for the upper and lower river, respectively. These tables also all show the predicted egg concentrations based on a TEQ-based biomagnification factor derived from the Phase 2 dataset.

Belted Kingfisher

Tables 3-55 and 3-56 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. Tables 3-57 and 3-58 present the expected average daily dose for the modeling period 1993 – 2018 for the upper and lower river, respectively. These tables also all show the predicted egg concentrations based on a TEQ-based biomagnification factor obtained from the literature (Giesy *et al.*, 1995).

Great Blue Heron

Tables 3-59 and 3-60 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. Tables 3-61 and 3-62 present the expected average daily dose for the modeling period 1993 – 2018 for the upper and lower river, respectively. These tables also all show the predicted egg concentrations based on a TEQ-based biomagnification factor obtained from the literature (Giesy *et al.*, 1995).

Bald Eagle

Tables 3-63 and 3-64 provide the expected average daily dose and 95% UCL daily dose on a TEQ PCB basis for the bald eagle from water and dietary sources based on 1993 data for water, and piscivorous fish concentrations. Tables 3-65 and 3-66 present the expected average daily dose for the modeling period 1993 – 2018 for the upper and lower river, respectively. These tables also all show the predicted egg concentrations based on a TEQ-based biomagnification factor obtained from the literature (Giesy *et al.*, 1995).

3.7 Exposure to Mammalian Wildlife

3.7.1 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 3-20b. NYSDEC is currently collecting mink, otter, and muskrat data, including concentrations of PCBs in blood and liver and population estimates (Mayack, 2000a). Preliminary results suggest that highest concentrations are found for animals with home ranges directly on the Hudson River (Loukmas, 2000).

3.7.2 Mammalian Wildlife Exposure Models

The exposure equation to calculate mammalian daily dose is the same as that used for birds (Equation 3-3), and is equal to the sum of diet, water, and sediment exposure. 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 are discussed below. Parameters used for the little brown bat, raccoon, mink, and river otter are summarized in Tables 3-67 to 3-70.

3.7.2.1 Surface Water Ingestion Pathway

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

$$WI_{(Mammal)} = (0.099 * BW^{0.90}) \qquad \text{Equation 3-13}$$

where:

$WI_{(Mammal)}$ = Receptor-specific water ingestion rate (L/day); and
BW = Body weight of mammalian 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 water ingestion rate (L/day). The receptor-specific average daily dosage rate ADD_{Water} (mg/kg/day) is calculated using Equation 3-4.

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 mammalian receptors was set at a value of 1.0 for all parameters.

3.7.2.2 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 using Equation 3-6.

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.

3.7.2.3 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 field metabolic rate (FMR) (kcal/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 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 in USEPA, 1999c). An estimated daily dietary ingestion rate was developed for the raccoon using the field metabolic rate, the typical NYS diet composition of the raccoon, and the average metabolizable energy content of the diet.

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

$$FMR = 0.6167 \times (BW_a)^{0.862} \qquad \text{Equation 3-14}$$

where:

FMR = Field metabolic rate (kcal/g-day); and
BW = Body weight of mammalian receptor (gms).

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 as provided in Equation 3-8.

The dietary ingestion rate for each of the mammalian receptors is calculated as the quotient of the receptor-specific FMR and ME_{Ave} for the specific diet according to Equation 3-9.

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:

$$ADD_{Diet} = ADD_{Fish} + ADD_{Invert} \qquad \text{Equation 3-15}$$

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 (> 25 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 in USEPA, 1999c). The average daily dosage of PCBs to receptors from the fish-derived portion of the diet is provided in Equation 3-10.

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 given in Equation 3-11.

The scientific literature and 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 United States, and populations from other regions of the contiguous United States. 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 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, 1999b) 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 a*l. 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. Based on their preference for fish and the winter diet, a diet composition of 100% fish was applied to Hudson River otter populations.

3.7.2.4 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. In addition, reproduction and growth (the most sensitive time periods) occur when the little brown bat is active along the Hudson River. Therefore, no temporal modifying factor was applied to the little brown bat.

3.7.3 Exposure Estimates for Mammalian Wildlife on a Total (Tri+) PCB Basis

Little Brown Bat

Tables 3-71 and 3-72 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 3-73 and 3-74 present the expected average daily dose for the modeling period 1993 – 2018 for the upper and lower river , respectively.

Raccoon

Tables 3-75 and 3-76 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 3-77 and 3-78 present the expected average daily dose for the modeling period 1993 – 2018 for the upper and lower river, respectively.

Mink

Tables 3-79 and 3-80 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 3-81 and 3-82 present the expected average daily dose for the modeling period 1993 – 2018 for the upper and lower river, respectively.

River Otter

Tables 3-83 and 3-84 provide the expected average daily dose and 95% UCL daily dose on a total PCB basis for the river otter from water and dietary sources based on 1993 data for water, sediment, and piscivorous fish concentrations. Tables 3-85 and 3-86 present the expected average daily dose for the modeling period 1993 – 2018 for the upper and lower river, respectively.

3.7.4 Exposure Estimates for Mammalian Wildlife on a TEQ Basis

Little Brown Bat

Tables 3-87 and 3-88 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 3-89 and 3-90 present the expected average daily dose and for the modeling period 1993 – 2018 for the upper and lower river, respectively.

Raccoon

Tables 3-91 and 3-92 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 3-93 and 3-94 present the expected average daily dose for the modeling period 1993 – 2018 for the upper and lower river, respectively.

Mink

Tables 3-95 and 3-96 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 3-97 and 3-98 present the expected average daily dose for the modeling period 1993 – 2018 for the upper and lower river, respectively.

River Otter

Tables 3-99 and 3-100 provide the expected average daily dose and 95% UCL daily dose on a TEQ PCB basis for the river otter from water and dietary sources based on 1993 data for water, sediment, and piscivorous fish concentrations. Tables 3-101 and 3-102 present the expected average daily dose for the modeling period 1993 – 2018 for the upper and lower river, respectively.

3.8 Uncertainty and Sensitivity in Exposure

This assessment evaluates uncertainty in the predicted exposure doses through the use of a Monte Carlo analysis. The Monte Carlo analysis assigns distributions for key input parameters in the exposure models to predict distributions, or cumulative frequencies, of exposure (dose in mg/kg-day). The output distributions are then used in several different ways: to evaluate the percent of the population expected to experience a particular dose, to compare to dose-response curves to obtain population-level responses, and to evaluate the sensitivity of predicted exposure (dose) to the input parameters.

Table 3-103 summarizes the distributions and distribution parameters used in the exposure analyses. Sediment and water concentrations were set at point estimates as they have been shown to be minor contributors to the exposure concentrations. Based on a comparison of predicted fish concentrations to observations over the historical period (1977 – 1997), the uncertainty in the mean estimate of predicted fish concentrations is approximately a factor of two (USEPA, 2000a). In addition, the results of the FISHRAND model show that within-year variability in predicted fish body burdens (attributable to seasonality, lipid content, etc.) is also approximately a factor of two. Mammalian and avian receptors integrate exposure to PCBs in fish over spatial and temporal scales. Because of this, the appropriate statistic to evaluate exposure is the mean and the uncertainty about the mean. This uncertainty was characterized by a normal distribution (under the assumption that errors are normally distributed about a mean value), and also as a lognormal distribution (under the assumption that environmental data are lognormally distributed and that the error in predicted estimates is likely to be biased toward the right tail). The analysis was also run assuming a correlation between body weight and ingestion rate (based on allometric equations relating body weight, ingestion rate, and metabolic rate.

The remaining exposure parameters were described by triangular distributions in the absence of data with which to better constrain these distributions. All of the exposure parameters were developed for Hudson River receptors and use information specific to the Hudson River where possible. The Monte Carlo analysis was carried out in Excel™ using the Crystal Ball™ add-in. Each model was run for 10,000 iterations. Sensitivity of the predicted exposure concentrations to input parameters was estimated using percent contribution to variance as well as rank correlation. These results are provided in Chapter 6.

The results for all of the runs were within 5% of each other, so the results that are presented are for the lognormally distributed fish concentrations. Figures 3-4 and 3-5 provide the results of the Monte Carlo analysis for the belted kingfisher, bald eagle, mink, and river otter. These graphs show the cumulative distribution on the y-axis and the predicted exposure dose in mg/kg-day on the x-axis. These predicted concentrations are compared to the dose-response curves presented in Chapter 4 and the results of those comparisons are provided in Chapter 5.

3.9 Examination of Exposure Pathways Based on Congener Patterns

Decisions related to controlling exposures to PCBs depend, in part, on understanding how PCBs are behaving in the river and the degree to which water and sediments are contributing to body burdens within the TI Pool as well as at downriver pools and the Lower Hudson River.

Chapter 1 of this report describes the current understanding of sources based on geochemical and historical considerations. Another approach for understanding sources is to examine the PCB congener patterns in fish in comparison to water and sediment at different locations in the river.

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 in USEPA 1999c.

The congener patterns contained in fish are also examined in Appendix K in USEPA 1999c 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: 1) identifying Aroclor patterns for use in toxicity assessment; 2) determining the relative importance of water, sediment, and food exposures; 3) evaluating the importance of upstream *versus* downstream sources of PCBs through spatial and temporal patterns; 4) importance of ongoing or recent releases in comparison to historical releases; and, 5) use of marker compounds and ratios to understand exposure.

Conclusions from the analysis presented in Appendix K in USEPA 1999c 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 components 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 sections 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.

3.9.1 Identifying Aroclor Patterns for Use in the Toxicity Assessment

The analysis prepared by NOAA (NOAA, 1997a), as well as those of the DEIR (USEPA, 1997a) and LRC (USEPA, 1998b), 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 in USEPA 1999c. This section 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 3-6 presents several regressions between a typical upper river 1993 largemouth bass sample from RM 190 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 RM 26 and is shown in Figure 3-7. 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 3-7, the best regressions are obtained against Aroclor 1254. For the Lower Hudson 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 3-8). 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 3-9. 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 and molecular weight in fish and sediments as function of river mile (see Figure 3-8 and Figure 3-10). As in Figure 3-8, the lines in upper figure in Figure 3-10 represent weighted averages and are used to simply illustrate general trends while the lines in the lower figure are linear regressions. 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 3-8, 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. 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).

3.9.2 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 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.

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 3-11, 3-12 and 3-13, 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 3-11, 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 (Appendix K of USEPA. 1999c), 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

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.

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

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. Most PCB toxicity research has concentrated on fish, birds, and mammals and therefore individual toxicity values are developed for species in these groups (Section 4.3, USEPA, 1999c). In contrast, few detailed studies have been performed on amphibians and reptiles and hence toxicity values would not have been able to be developed for species in these groups, had they been selected as receptors.

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

A toxicity reference value (TRV) is a contaminant dose or body burden that is compared to site-specific doses or body burdens to assess the potential risk to an ecological receptor. A TRV can be based on results from laboratory or field studies. 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, a comprehensive literature search was conducted on the toxicity of PCBs to animals. A variety of databases were searched for references containing toxicity information. These included the National Library of Medicine (NLM) MEDLINE and TOXLINE databases and the Aquatic Information Retrieval Database (AQUIRE). Secondary sources that were used to identify studies that may have been overlooked in the database searches included U.S. Fish and Wildlife Service Contaminant Hazard Reviews, the Agency for Toxic Substances Disease Registry (ATSDR) documents, and the U.S. EPA Great Lakes Water Quality Initiative documents.

A number of criteria were considered in order to evaluate the appropriateness of a particular study for inclusion in the database used for this assessment. First of all, doses should be quantified and reported. An appropriate study design, including the use of adequate sample size and an appropriate negative control group, should be included in the design. Appropriate statistical analyses should be conducted and the statistical significance of the results reported. The remainder of this chapter describes the rationale that was used to select TRVs for the representative receptors.

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 survival of the population. 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. Therefore studies of chronic exposure are used to select TRVs for the present risk assessment. Long-term studies are also preferred because reproductive effects of PCBs are typically studied after long-term exposure.

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

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

- LOAEL (Lowest-Observed-Adverse-Effect-Level) is the lowest exposure level shown to produce adverse effect in organisms exposed to a range of doses. LOAELs may also be expressed as dietary doses (*e.g.*, mg PCBs consumed/kg body weight/d), as concentrations in external media (*e.g.*, mg PCBs/kg food), or as concentrations in tissue of the effected organisms (*e.g.*, mg chemical/kg egg).

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

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

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

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

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

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

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

Most USEPA risk assessments typically estimate risk by comparing the exposure of receptors of concern to TRVs that are based on NOAELs. TRVs for the present baseline risk assessments are developed on the basis of both NOAELs and LOAELs to provide perspective on the range of potential effects relative to measured or modeled exposures.

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

- TRVs for benthic invertebrates are expressed as concentrations in external media (*e.g.*, mg/kg sediment). Critical body burdens (*e.g.*, mg/kg body weight) for benthic invertebrates are presented, but a TRV is not selected due to limited data.

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

- 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).

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.

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. The studies that were reviewed for this risk assessment are presented in Tables 4-5 through 4-22. 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. This not a great disadvantage to the risk assessment, since the assessment endpoints evaluate feeding groups, as represented by individual receptor models. 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.

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 or field studies on less closely related species 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 the receptors from appropriate studies.

When appropriate chronic-exposure toxicity studies on the effects of PCBs or dioxin-like compounds on lethality, growth, or reproduction are not available for the species examined for a particular assessment endpoint, studies on other species are used to develop 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 LOAEL is unavailable for a phylogenetically similar species (*e.g.* within the same taxonomic family), the assessment uses a study conducted on another species, preferably one that is closely related to the receptor of concern. The most appropriate LOAEL is used whenever several studies are available. Professional judgement was used in some cases to select the most appropriate study. Interspecies uncertainty factors, which account for potential differences in sensitivity between a test species and a receptor, are not used in the development of the final TRVs for the risk assessment. However, for illustrative purposes, a secondary set of TRVs are developed using interspecies uncertainty factors where appropriate. If the surrogate test species is known to be highly sensitive to the effects of PCBs or dioxin (*e.g.* salmonids, mink), an interspecies uncertainty factor is not applied to the secondary TRV.

- In the absence of an appropriate NOAEL, an appropriate LOAEL may be divided by a conversion factor of 10 to estimate a NOAEL. 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 (Dourson and Stara, 1983).

- 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 a conversion factor of 10 to estimate chronic TRVs. The use of a conversion factor of 10 is consistent with the methodology used to derive human health RfDs (Dourson and Stara, 1983). These factors are applied to account for uncertainty in using an external dose (mg/kg/d in diet) as a surrogate for the dose at the site of toxic action (*e.g.* mg/kg in tissue). Because organisms may attain a toxic dose at the site of toxic action (*e.g.* in tissues or organs) via a large dose administered over a short period, or via a smaller dose administered over a longer period, conversion 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 generally 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), a sub-chronic to chronic conversion factor is not 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 TRVs 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).

In order to illustrate the range of uncertainty involved in deriving TRVs, TRVs are developed from both laboratory and field studies, with and without the use of interspecies uncertainty factors. However, the no interspecies uncertainty factors were used to develop the final TRVs that are used in the risk assessment, which are presented in bold type in Tables 4-25 through 4-27. The sensitivity of the risk estimates to the use of these various approaches is examined in the uncertainty chapter (Chapter 6.0) of this report.

4.2.2 Selection of TRVs for Benthic Invertebrates

Various guidelines exist for concentrations of PCBs in sediment (Table 4-3). Concentrations of PCBs in sediments of the Hudson River will be compared to the Sediment Effects Concentrations (SEC) developed for this site (NOAA, 1999a), NYSDEC sediment guidelines (NYSDEC, 1999a), Ontario guideline (Persaud *et al.,* 1993), and Washington State sediment guidelines (1997), which are considered to be the guidelines most relevant to this study.

A measurement endpoint of measured and modeled benthic invertebrate body burdens to TRVs was not included because relatively few studies were identified that examined the effects of PCBs or dioxin-like compounds on the basis of body burdens in aquatic invertebrates (Table 4-4). Therefore, a body burden-based TRV is not developed for benthic invertebrates.

4.2.3 Selection of TRVs for Fish

In this section, TRVs are developed for the forage fish receptors (pumpkinseed and spottail shiner) and for fish receptors that feed at higher trophic levels (brown bullhead, yellow perch, white perch, largemouth bass, and striped bass).

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. Field studies on total PCBs or Aroclors are presented in Table 4-6. Laboratory studies and field studies on the effects of dioxin-like compounds (TEQs) on fish (Tables 4-7 and 4-8, respectively) typically report concentrations of TEQs in fish eggs, rather than in whole body, since eggs represent a more sensitive life stage. Comparison of effect levels (*e.g.*, NOAELs or LOAELs) reported as wet weight concentrations in eggs to whole body tissue concentrations in adult Hudson River fish is complicated by the fact that eggs and whole body adult fish tend to have different lipid contents and concentrations of lipophilic contaminants, such as TEQs. However, if we assume that TEQs partition equally into the lipid phase of the egg and into the lipids in the tissue of adult fish (Niimi, 1983), then lipid-normalized concentrations in fish eggs that are associated with adverse effects (μg

TEQs/kg lipid in egg) can be compared to lipid-normalized tissue concentrations of TEQs in adult Hudson River fish (ug TEQs/kg lipid in whole body adult). 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

Of the laboratory studies examined (Table 4-5), no 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 4-23). Two studies (Hansen *et al.*, 1971 and Hansen *et al.*, 1974a) were identified that examined toxicity of PCBs to species in the same order as the pumpkinseed (Table 4-23). However, these studies by Hansen *et al.* (1971, 1974a) are not selected for the development of TRVs because the 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 LOAEL and corresponding NOAEL 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) is not selected because is based on exposure to Clophen A50. Although the chlorine content of Clophen A50 (50%) is similar to that of the chlorine content of Hudson River fish, this mixture was never used in the United States.

The study by Hansen *et al.* (1974b) is selected as the most appropriate study. This study established a NOAEL of 1.9 mg PCBs/kg and a LOAEL of 9.3 mg PCBs/kg for the sheepshead minnow. This study was based on a flow-through bioassay of Aroclor 1254 on adult female fish. Fish were exposed for 28 days, and then egg production was induced. The eggs were fertilized and placed in PCB-free flowing seawater and observed for mortality. Survival of fry to one week of age was 77% for eggs from adults from the 0.32 ug/L treatment (average 9.3 mg/kg in tissue of females), as compared to 95% survival of fry from control adults and 97% survival of fry from adults from the NOAEL treatment (0.1 ug/L; average 1.9 mg/kg in tissue of females). The TRVs resulting from this study are comparable to the TRVs from Bengtsson (1980).

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 conversion factor is not applied.

On the basis of laboratory toxicity studies:

The LOAEL TRV for the pumpkinseed is 9.3 mg PCBs/kg tissue (Table 4-25a).
The NOAEL TRV for the pumpkinseed is 1.9 mg PCBs/kg tissue (Table 4-25a).

These are selected as the final TRVs for use in the risk assessment.

Because the test species and the pumpkinseed are not from the same taxonomic family, an interspecies uncertainty factor of ten could be applied. For comparative purposes, Table 4-25b presents TRVs that would be derived using interspecies uncertainty factors. However, interspecies uncertainty factors are not used in the derivation of final TRVs for this risk assessment (Table 4-25a).

Of all of the field studies examined (Table 4-6), several studies were identified that examined the effect of PCBs on the redbreast sunfish, a species in the same family as the pumpkinseed (Table 4-23). Field studies by Adams *et al.* (1989, 1990, 1992) reported reduced fecundity and growth in redbreast sunfish (*Lepomis auritus*) that were exposed to PCBs and mercury in the field. Growth was the more sensitive endpoint; length and weight of sunfish at three sites (C1, C2, and C3) along a gradient moving downstream from a pollutant source were significantly reduced at each site in comparison to a fourth site (C4) further downstream. The average concentration of PCBs in tissue of fish from site C3 was 0.4 mg/kg, which is selected as the LOAEL TRV for pumpkinseed. Mean length of fish at this site was 11% lower than site C4 and mean weight was 29% lower than site C4. Average tissue PCB concentration of fish from site C4 was 0.3 mg/kg; this value is selected as the NOAEL TRV for pumpkinseed. 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 conversion factor is not applied.

On the basis of the field studies:

The LOAEL TRV for the pumpkinseed is 0.4 mg PCBs/kg tissue (Table 4-25a).
The NOAEL TRV for the pumpkinseed is 0.3 mg PCBs/kg tissue (Table 4-25a).

Because of the presence of substantial amounts of co-occurring contaminants, especially mercury, at this field site, this study is not selected for derivation of final TRVs for the risk assessment.

An interspecies uncertainty factor would not be applied because the redbreast sunfish and the pumpkinseed are in the same family (Table 4-25b).

Total Dioxin Equivalents (TEQs) in Eggs of the Pumpkinseed

Of all of the studies examined (Table 4-7), 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 4-23). Studies of salmonids are not used to develop the primary TRVs because salmonids are among the most sensitive species tested (Table 4-7). Therefore, the lowest appropriate non-salmonid LOAEL and NOAEL from the selected applicable studies are used to derive TRVs for the pumpkinseed. Elonen *et al.* (1998) reported a

NOAEL of 8 μg TEQs/kg lipid and a LOAEL of 18 μg TEQs/kg lipid for eggs of the channel catfish, based on early life stage mortality. Survival of juveniles from eggs containing 18 μg TEQs/kg lipid was 18%, compared to 100% survival in both the control and the solvent control. Because the experimental study is based on the concentration in the egg, rather than an estimated dose, a subchronic-to-chronic conversion factor is not applied.

On the basis of laboratory studies:

The LOAEL TRV for the pumpkinseed is 18 μg TEQs/kg lipid (Table 4-25a).
The NOAEL TRV for the pumpkinseed is 8 μg TEQs/kg lipid (Table 4-25a).

These are selected as the final TRVs for use in the risk assessment.

An interspecies uncertainty factor of ten could be applied because the pumpkinseed and the channel catfish are not in the same taxonomic family (Table 4-25b). For comparative purposes, Table 4-25b presents TRVs that would be derived using interspecies uncertainty factors. However, interspecies uncertainty factors are not used in the derivation of final TRVs for this risk assessment (Table 4-25a).

Alternative TRVs developed from laboratory studies conducted on salmonids are presented for comparison. The lowest salmonid LOAEL (0.6 μg TEQs/kg lipid) and corresponding NOAEL (0.29 μg TEQs/kg lipid) from the selected applicable studies are used to derive alternative TRVs for the pumpkinseed (Table 4-25a). In a study by Walker *et al.* (1994), 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 TEQs/kg lipid. Because the experimental study is based on the concentration in the egg, rather than an estimated dose, a subchronic-to-chronic conversion factor is not applied. Because salmonids are among the most sensitive species tested, an interspecies uncertainty factor of ten would not be applied (Table 4-25b).

Of all the field studies examined (Table 4-8), 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. Therefore, the most appropriate NOAEL from the selected applicable studies is used to derive a TRV for the pumpkinseed. Guiney *et al.* (1996) found no effect on early life stage mortality in eggs from lake trout from Lake Ontario compared to eggs from a hatchery. Eggs from Lake Ontario contained an average of 0.1 μg TEQs/kg lipid, while hatchery eggs contained only trace levels of TEQs. The value of 0.1 μg TEQs/kg lipid is selected as the basis for the NOAEL TRV for the pumpkinseed. Because the experimental study is based on the concentration in the egg, rather than an estimated dose, a subchronic-to-chronic conversion factor is not applied.

On the basis of field studies:

The NOAEL TRV for the pumpkinseed is 0.1 μg TEQs/kg lipid (Table 4-25a).

Because this study was conducted on a highly sensitive species, it is not selected for development of the final TRV for this risk assessment.

Because salmonids are among the most sensitive species tested, an interspecies uncertainty factor of ten would not be applied (Table 4-25b).

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 LOAEL and corresponding NOAEL from the selected applicable studies (Table 4-5). The study by Hansen *et al.* (1974b) on the sheepshead minnow is selected as the lowest appropriate LOAEL (9.3 mg/kg) and corresponding NOAEL (1.9 mg/kg) for development of TRVs for 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 conversion factor is not applied.

On the basis of laboratory toxicity studies:

The LOAEL TRV for the spottail shiner is 9.3 mg PCBs/kg tissue (Table 4-25a).
The NOAEL TRV for the spottail shiner is 1.9 mg PCBs/kg tissue (Table 4-25a).

An interspecies uncertainty factor of ten could be applied because the spottail shiner and the sheepshead minnow are not in the same taxonomic family (Table 4-25b). For comparative purposes, Table 4-25b presents TRVs that would be derived using interspecies uncertainty factors. However, interspecies uncertainty factors are not used in the derivation of final TRVs for this risk assessment (Table 4-25a).

Of all the field studies examined (Table 4-6), one study was identified that examined the effects of PCBs on the fathead minnow, a species in the same taxonomic family (Cyprinidae) as the spottail shiner. The USACE (1988) exposed adult fathead minnows for 16 weeks to field-collected sediment contaminated with varying levels of PCBs. Fecundity and frequency of reproduction were significantly impaired in the "medium" and "high" level treatments, but not the "low" level treatment, in comparison to the control. In the medium-level treatment, fecundity was 75% lower than the control and frequency of reproduction was 84% lower than the control. Fish sacrificed after 7 weeks had tissue concentrations of 5.25 mg PCBs/kg in the low-level treatment and 13.7 mg PCBs/kg in the medium-level treatment. These values are selected for development of the NOAEL and LOAEL TRVs for 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 conversion factor is not applied.

On the basis of the field study:

The LOAEL TRV for the spottail shiner is 13.7 mg PCBs/kg tissue (Table 4-25a).
The NOAEL TRV for the spottail shiner is 5.25 mg PCBs/kg tissue (Table 4-25a).

These are selected as the final TRVs for use in the risk assessment because the test species is in the same family as the spottail shiner.

Because the spottail shiner and the fathead minnow are in the same family, an interspecies uncertainty factor would not be applied (Table 4-25b).

Total Dioxin Equivalents (TEQs) in Eggs of Spottail Shiner

Of all of the laboratory studies examined (Table 4-7), several studies were identified that examined toxicity of dioxin-like compounds on fish in the same family as the spottail shiner. The study by Elonen *et al.* (1998) 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 18 μg TEQs/kg lipid. This effect was not observed at a concentration of 9.8 μg TEQs/kg lipid. Survival of juveniles from eggs containing 18 μg TEQs/kg lipid was 73%, compared to 100% survival in both the control and the solvent control. Because the experimental study is based on the concentration in the egg, rather than an estimated dose, a subchronic-to-chronic conversion factor is not applied.

On the basis of laboratory toxicity studies:

The LOAEL TRV for the spottail shiner is 18 μg TEQs/kg lipid (Table 4-25a).
The NOAEL TRV for the spottail shiner is 9.8 μg TEQs/kg lipid (Table 4-25a).

These are selected as the final TRVs for use in the risk assessment.

Because the fathead minnow and the spottail shiner are in the same taxonomic family, an interspecies uncertainty factor would not be applied (Table 4-25b).

Alternative TRVs for dioxin-like compounds based on studies of salmonids are not developed for the spottail shiner because the laboratory-based TRVs for the spottail shiner are based on data for a species from the same taxonomic family as the spottail shiner.

Of all the field studies examined (Table 4-8), no 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). Therefore, the most appropriate NOAEL from the selected applicable studies, the value of 0.1 µg TEQs/kg lipid identified by Guiney *et al.* (1996) for the lake trout, is used to derive a TRV for the spottail shiner. Because the experimental study is based on the concentration in the egg, rather than an estimated dose, a subchronic-to-chronic conversion factor is not applied.

On the basis of field studies:

The NOAEL TRV for the spottail shiner is 0.1 µg TEQs/kg lipid (Table 4-25a).

Because this study was conducted on a highly sensitive species, it is not selected for development of the final TRV for this risk assessment.

Because salmonids such as the lake trout are among the most sensitive species tested, an interspecies uncertainty factor of ten would not be applied (Table 4-25b).

4.2.3.3 Brown Bullhead (*Ictalurus nebulosus*)

Total PCB Body Burden in the Brown Bullhead

Of all the laboratory studies examined (Table 4-5), 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. Therefore, concentrations of PCBs in the brown bullhead will be compared to the lowest appropriate LOAEL and corresponding 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 Hansen *et al.* (1974b) on the sheepshead minnow is selected for development of the TRV. 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 conversion factor is not applied

On the basis of laboratory toxicity studies:

The LOAEL TRV for the brown bullhead is 9.3 mg PCBs/kg tissue (Table 4-25a).
The NOAEL TRV for the brown bullhead is 1.9 mg PCBs/kg tissue (Table 4-25a).

These are selected as the final TRVs for use in the risk assessment rather than the TRVs developed based on field studies.

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 ten could be applied to develop TRVs for the brown bullhead (Table 4-25b).For comparative purposes, Table 4-25b presents TRVs that would be derived using interspecies uncertainty factors. However, interspecies uncertainty factors are not used in the derivation of final TRVs for this risk assessment (Table 4-25a).

Of all field studies examined (Table 4-6), 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. Therefore, the lowest appropriate LOAEL and corresponding NOAEL from the applicable studies are used to develop TRVs for the brown bullhead. The study by USACE (1988) on the fathead minnow is selected for development of the TRV. 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 conversion factor is not applied.

On the basis of the field study:

The LOAEL TRV for the brown bullhead is 13.7 mg PCBs/kg tissue (Table 4-25a).
The NOAEL TRV for the brown bullhead is 5.25 mg PCBs/kg tissue (Table 4-25a).

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 ten could be applied (Table 4-25b). For comparative purposes, Table 4-25b presents TRVs that would be derived using interspecies uncertainty factors. However, interspecies uncertainty factors are not used in the derivation of final TRVs for this risk assessment (Table 4-25a).

The laboratory study (Hansen et al., 1974) and field-based study (USACE, 1988) are comparable in terms of exposure duration, sample number, and sensitivity of endpoint examined. The laboratory study is chosen as a conservative TRV since effects were observed at slightly lower concentrations.

Total Dioxin Equivalents (TEQs) in Eggs of the Brown Bullhead

Of all laboratory studies examined (Table 4-7), no studies were identified that examined toxicity of dioxin-like compounds on the brown bullhead. 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 (72% compared to none in the control) 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 conversion factor is not applied.

On the basis of laboratory toxicity studies:

The LOAEL TRV for the brown bullhead is 18 μg TEQs/kg lipid (Table 4-25a).
The NOAEL TRV for the brown bullhead is 8.0 μg TEQs/kg lipid (Table 4-25a).

These are selected as the final TRVs for use in the risk assessment.

An interspecies uncertainty factor would not be applied because channel catfish and brown bullhead are in the same taxonomic family (Table 4-25b).

Alternative TRVs for dioxin-like compounds based on studies of salmonids are not developed for the brown bullhead because the laboratory-based TRVs for the brown bullhead are based on data for a species from the same taxonomic family as the brown bullhead.

Of all the field studies examined (Table 4-8), no field 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. Therefore, the most appropriate NOAEL from the selected applicable studies, the value of 0.1 μg TEQs/kg lipid identified by Guiney *et al.* (1996) for the lake trout, is used to derive a TRV for the brown bullhead. Because the experimental study is based on the concentration in the egg, rather than an estimated dose, a subchronic-to-chronic conversion factor is not applied.

On the basis of field studies:

The NOAEL TRV for the brown bullhead is 0.1 μg TEQs/kg lipid (Table 4-25a).

This study is not selected as the final TRV because it was conducted on a highly sensitive species.

Because salmonids such as the lake trout are among the most sensitive species tested, an interspecies uncertainty factor of ten would not be applied (Table 4-25b).

4.2.3.4 Yellow Perch (*Perca flavescens*)

Total PCB Body Burden in the Yellow Perch

Of all the laboratory studies examined (Table 4-5), no laboratory studies were identified that examined toxicity of PCBs to the yellow perch. Two studies (Hansen *et al.*, 1974a 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 Hansen *et al.* (1974b) on the sheepshead minnow is selected as the lowest appropriate NOAEL for development of the TRV. 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 conversion factor is not applied.

On the basis of laboratory toxicity studies:

The LOAEL TRV for the yellow perch is 9.3 mg PCBs/kg tissue (Table 4-25a).
The NOAEL TRV for the yellow perch is 1.9 mg PCBs/kg tissue (Table 4-25a).

These are selected as the final TRVs for use in the risk assessment rather than the TRVs based on field studies (see below).

Because results of studies of dioxin-like compounds and PCBs on fish eggs have shown minnows to be of intermediate sensitivity compared to all other fish species tested (Tables 4-5, 4-7), an interspecies uncertainty factor of ten could be applied (Table 4-25b). For comparative purposes, Table 4-25b presents TRVs that would be derived using interspecies uncertainty factors. However, interspecies uncertainty factors are not used in the derivation of final TRVs for this risk assessment (Table 4-25a).

Of all the field studies examined (Table 4-6), no 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. Therefore, the lowest appropriate LOAEL and corresponding NOAEL from the applicable studies are used to develop TRVs for the yellow perch. The study by USACE (1988) on the fathead minnow is selected for development of the TRV. 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 conversion factor is not applied.

On the basis of the field study:

The LOAEL TRV for the yellow perch is 13.7 mg PCBs/kg tissue (Table 4-25a).
The NOAEL TRV for the yellow perch is 5.25 mg PCBs/kg tissue (Table 4-25a).

Because the yellow perch and the fathead minnow are not in the same taxonomic family, an interspecies uncertainty factor of ten could be applied (Table 4-25b). For comparative purposes,

Table 4-25b presents TRVs that would be derived using interspecies uncertainty factors. However, interspecies uncertainty factors are not used in the derivation of final TRVs for this risk assessment (Table 4-25a).

The laboratory study (Hansen *et al.*, 1974) and field-based study (USACE, 1988) are comparable in terms of exposure duration, sample number, and sensitivity of endpoint examined. The laboratory study is chosen as a conservative TRV since effects were observed at slightly lower concentrations.

Total Dioxin Equivalents (TEQs) in Eggs of the Yellow Perch

Of all of the laboratory studies examined (Table 4-7), no 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. Studies of salmonids are not used to develop the primary TRVs because salmonids are among the most sensitive species tested (Table 4-7). Therefore, the lowest appropriate non-salmonid LOAEL and NOAEL from the selected applicable studies are used to derive TRVs for the yellow perch. The study by Elonen *et al.* (1998) on the channel catfish (Table 4-7) is selected for development of TRVs for the yellow perch. In that study, significant early life stage mortality (72% compared to none in the control) 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 conversion factor is not applied.

On the basis of laboratory toxicity studies:

The LOAEL TRV for the yellow perch is 18 μg TEQs/kg lipid (Table 4-25a).
The NOAEL TRV for the yellow perch is 8.0 μg TEQs/kg lipid (Table 4-25a).

These are selected as the final TRVs for use in the risk assessment.

Because the yellow perch and the channel catfish are not in the same taxonomic family, an interspecies uncertainty factor of ten could be applied (Table 4-25b). For comparative purposes, Table 4-25b presents TRVs that would be derived using interspecies uncertainty factors. However, interspecies uncertainty factors are not used in the derivation of final TRVs for this risk assessment (Table 4-25a).

Alternative TRVs developed from laboratory studies conducted on salmonids are presented for comparison. The lowest salmonid LOAEL (0.6 μg TEQs/kg lipid) and corresponding NOAEL (0.29 μg TEQs/kg lipid) from the selected applicable studies are used to derive alternative TRVs for the yellow perch (Table 4-25a). In a study by Walker *et al.* (1994), 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 TEQs/kg lipid. Because the experimental study is based on the concentration in the egg, rather than an estimated dose, a subchronic-to-chronic conversion factor is not applied. Because salmonids are among the most sensitive species tested, an interspecies uncertainty factor of ten would not be applied (Table 4-25b).

Of all of the field studies examined (Table 4-8), no 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. Therefore, the most appropriate NOAEL from the selected applicable studies, the value of 0.1 μg TEQs/kg lipid identified by Guiney *et al.* (1996) for the lake trout, is used to derive a TRV for the yellow perch. Because the experimental study is based on the concentration in the egg, rather than an estimated dose, a subchronic-to-chronic conversion factor is not applied.

On the basis of field studies:

The NOAEL TRV for the yellow perch is 0.1 μg TEQs/kg lipid (Table 4-25a).

This study is not selected for development of final TRVs because it was conducted on a highly sensitive species.

Because salmonids such as the lake trout are among the most sensitive species tested, an interspecies uncertainty factor of ten would not be applied (Table 4-25b).

4.2.3.5 White Perch (*Morone americana*)

Total PCB Body Burden in the White Perch

Of all the laboratory studies examined (Table 4-5), no studies were identified that examined toxicity of PCBs to the white perch. 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 LOAEL and corresponding 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 Hansen *et al.* (1974b) on the sheepshead minnow is selected as the lowest appropriate LOAEL and corresponding NOAEL for development of the TRV. 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 conversion factor is not applied.

On the basis of laboratory toxicity studies:

The LOAEL TRV for the white perch is 9.3 mg PCBs/kg tissue (Table 4-25a).
The NOAEL TRV for the white perch is 1.9 mg PCBs/kg tissue (Table 4-25a).

This LOAEL is selected as the final LOAEL TRV. The NOAEL from this study is not used, because a NOAEL for a more a closely related species (described below) is selected for development of the final NOAEL.

Because results of studies of dioxin-like compounds and PCBs on fish eggs have shown minnows to be of intermediate sensitivity compared to all other fish species tested (Tables 4-5, 4-7), an interspecies uncertainty factor of ten could be applied (Table 4-25b). For comparative purposes, Table 4-25b presents TRVs that would be derived using interspecies uncertainty factors. However, interspecies uncertainty factors are not used in the derivation of final TRVs for this risk assessment (Table 4-25a).

Of all the field studies examined (Table 4-6), two studies were identified that examined the effects of PCBs on striped bass, a species in the same family as the white perch (Table 4-23). 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. Because the study measured the concentration in the larval tissue, rather than estimating a dose, a subchronic-to-chronic conversion factor is not applied.

On the basis of the field study:

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

This study is selected for development of the final NOAEL TRV because it was conducted on a closely related species that was collected from the Hudson River.

An interspecies uncertainty factor would not be applied because white perch and striped bass are in the same taxonomic family (Table 4-25b).

Total Dioxin Equivalents (TEQs) in Eggs of the White Perch

Of all the laboratory studies examined (Table 4-7), no 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. Studies of salmonids are not used to develop the primary TRVs because salmonids are among the most sensitive species tested (Table 4-7). Therefore, the lowest appropriate non-salmonid LOAEL and NOAEL from the selected applicable studies are used to derive TRVs for the white perch. The study by Elonen *et al.* (1998) on the channel catfish (Table 4-7) is selected for development of TRVs for the white perch. In that study, significant early life stage mortality (72% compared to none in the control) 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 conversion factor is not applied.

On the basis of laboratory toxicity studies:

The LOAEL TRV for the white perch is 18 µg TEQs/kg lipid (Table 4-25a).
The NOAEL TRV for the white perch is 8.0 µg TEQs/kg lipid (Table 4-25a).

These are selected as the final TRVs for use in the risk assessment.

Because the white perch and the channel catfish are not in the same taxonomic family, an interspecies uncertainty factor of ten could be applied (Table 4-25b). For comparative purposes, Table 4-25b presents TRVs that would be derived using interspecies uncertainty factors. However, interspecies uncertainty factors are not used in the derivation of final TRVs for this risk assessment (Table 4-25a).

Alternative TRVs developed from laboratory studies conducted on salmonids are presented for comparison. The lowest salmonid LOAEL (0.6 µg TEQs/kg lipid) and corresponding NOAEL (0.29 µg TEQs/kg lipid) from the selected applicable studies are used to derive alternative TRVs for the white perch (Table 4-25a). In a study by Walker *et al.* (1994), 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 TEQs/kg lipid. Because the experimental study is based on the concentration in the egg, rather than an estimated dose, a subchronic-to-chronic conversion factor is not applied. Because salmonids are among the most sensitive species tested, an interspecies uncertainty factor of ten would not be applied (Table 4-25b).

Of all the field studies examined (Table 4-8), no 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). Therefore, the most appropriate NOAEL from the selected applicable studies, the value of 0.1 µg TEQs/kg lipid identified by Guiney *et al.* (1996) for the lake trout, is used to derive a TRV for the white perch. Because the experimental study is based on the concentration in the egg, rather than an estimated dose, a subchronic-to-chronic conversion factor is not applied.

On the basis of field studies:

The NOAEL TRV for the white perch is 0.1 µg TEQs/kg lipid (Table 4-25a).

This study is not selected for development of final TRVs because it was conducted on a highly sensitive species.

Because salmonids such as the lake trout are among the most sensitive species tested, an interspecies uncertainty factor of ten would not be applied (Table 4-25b).

4.2.3.6 Largemouth Bass (*Micropterus salmoides*)

Total PCB Body Burden in the Largemouth Bass

Of all the laboratory studies examined (Table 4-5), no studies were identified that examined toxicity of PCBs to the largemouth bass. Two studies (Hansen *et al.*, 1974a and Hansen *et al.*, 1971) were identified that examined toxicity of PCBs to species of the same order as the largemouth bass. However, these studies are not selected for the development of TRVs because they 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 LOAEL and corresponding 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 Hansen *et al.* (1974b) on the sheepshead minnow is selected as the lowest appropriate LOAEL and corresponding NOAEL for development of the TRV. 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 conversion factor is not applied.

On the basis of laboratory toxicity studies:

The LOAEL TRV for the largemouth bass is 9.3 mg PCBs/kg tissue (Table 4-25a).
The NOAEL TRV for the largemouth bass is 1.9 mg PCBs/kg tissue (Table 4-25a).

These are selected as the final TRVs for use in the risk assessment.

Because results of studies of dioxin-like compounds and PCBs on fish eggs have shown minnows to be of intermediate sensitivity compared to all other fish species tested (Tables 4-5, 4-7), an interspecies uncertainty factor of ten could be applied (Table 4-25b). For comparative purposes, Table 4-25b presents TRVs that would be derived using interspecies uncertainty factors. However, interspecies uncertainty factors are not used in the derivation of final TRVs for this risk assessment (Table 4-25a).

Of all of the field studies examined (Table 4-6), several studies were identified that examined effect of PCBs on the redbreast sunfish, a species in the same family as the largemouth bass (Table 4-23). Field studies by Adams *et al.* (1989, 1990, 1992) reported reduced fecundity and growth in redbreast sunfish (*Lepomis auritus*) that were exposed to PCBs and mercury in the field. Growth was the more sensitive endpoint, with a NOAEL of 0.3 mg PCBs/kg tissue and a LOAEL of 0.4 mg PCBs/kg tissue. 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 conversion factor is not applied.

On the basis of the field studies:

The LOAEL TRV for largemouth bass is 0.4 mg PCBs/kg tissue (Table 4-25a).
The NOAEL TRV for largemouth bass is 0.3 mg PCBs/kg tissue (Table 4-25a).

An interspecies uncertainty factor would not be applied because the largemouth bass and the redbreast sunfish are in the same family (Table 4-25b).

Because of the presence of substantial amounts of co-occurring contaminants, especially mercury, at this field site, this study is not selected for derivation of final TRVs for the risk assessment.

Total Dioxin Equivalents (TEQs) in Eggs of the Largemouth Bass

Of all the laboratory studies examined (Table 4-7), no 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. Studies of salmonids are not used to develop the primary TRVs because salmonids are among the most sensitive species tested (Table 4-7). Therefore, the lowest appropriate non-salmonid LOAEL and NOAEL from the selected applicable studies are used to derive TRVs for the largemouth bass. The study by Elonen *et al.* (1998) on the channel catfish (Table 4-7) is selected for development of TRVs for the largemouth bass. In that study, significant early life stage mortality (72% compared to none in the control) 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 conversion factor is not applied.

On the basis of laboratory toxicity studies:

The LOAEL TRV for the largemouth bass is 18 μg TEQs/kg lipid (Table 4-25a).
The NOAEL TRV for the largemouth bass is 8.0 μg TEQs/kg lipid (Table 4-25a).

These are selected as the final TRVs for use in the risk assessment.

Because the largemouth bass and the channel catfish are not in the same taxonomic family, an interspecies uncertainty factor of ten could be applied (Table 4-25b). For comparative purposes, Table 4-25b presents TRVs that would be derived using interspecies uncertainty factors. However, interspecies uncertainty factors are not used in the derivation of final TRVs for this risk assessment (Table 4-25a).

Alternative TRVs developed from laboratory studies conducted on salmonids are presented for comparison. The lowest salmonid LOAEL (0.6 μg TEQs/kg lipid) and corresponding NOAEL (0.29 μg TEQs/kg lipid) from the selected applicable studies are used to derive alternative TRVs for the largemouth bass (Table 4-25a). In a study by Walker *et al.* (1994), 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 TEQs/kg lipid. Because the experimental study is based on the concentration in the egg, rather than an estimated dose, a subchronic-to-chronic conversion factor is not applied. Because salmonids are among the most sensitive species tested, an interspecies uncertainty factor of ten would not be applied (Table 4-25b).

Of all the field studies examined (Table 4-8), no 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. Therefore, the most appropriate NOAEL from the selected applicable studies, the value of 0.1 μg TEQs/kg lipid identified by Guiney *et al.* (1996) for the lake trout, is used to derive a TRV for the largemouth bass. Because the experimental study is based on the concentration in the egg, rather than an estimated dose, a subchronic-to-chronic conversion factor is not applied.

On the basis of field studies:

The NOAEL TRV for the largemouth bass is 0.1 μg TEQs/kg lipid (Table 4-25a).

This study is not selected for development of final TRVs because it was conducted on a highly sensitive species.

Because salmonids such as the lake trout are among the most sensitive species tested, an interspecies uncertainty factor of ten would not be applied (Table 4-25b).

4.2.3.7 Striped Bass (*Morone saxatilis*)

PCB Body Burdens in the Striped Bass

Of all the laboratory studies examined (Table 4-5), no studies were identified that examined toxicity of PCBs to the striped bass. 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, 1974a). 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 LOAEL and corresponding 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 Hansen *et al.* (1974b) on the sheepshead minnow is selected for development of the TRV. 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 conversion factor is not applied.

On the basis of laboratory toxicity studies:

The LOAEL TRV for the striped bass is 9.3 mg PCBs/kg tissue (Table 4-25a).
The NOAEL TRV for the striped bass is 1.9 mg PCBs/kg tissue (Table 4-25a).

This LOAEL is selected as the final LOAEL TRV. The NOAEL from this study is not used, because a NOAEL for a more a closely related species (described below) is selected for development of the final NOAEL.

Because results of studies of dioxin-like compounds and PCBs on fish eggs have shown minnows to be of intermediate sensitivity compared to all other fish species tested (Tables 4-5, 4-7), an interspecies uncertainty factor of ten could be applied (Table 4-25b). For comparative purposes, Table 4-25b presents TRVs that would be derived using interspecies uncertainty factors. However, interspecies uncertainty factors are not used in the derivation of final TRVs for this risk assessment (Table 4-25a).

Of all of the field studies examined (Table 4-6), two studies were identified that examined the effects of PCBs on striped bass. 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 striped bass. Because this study measured the concentration in the larval tissue, rather than estimating a dose, a subchronic-to-chronic conversion factor is not applied.

On the basis of the field study:

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

This NOAEL is selected as the final NOAEL TRV for the assessment because it was conducted on striped bass collected from the Hudson River.

Because the study was conducted on striped bass, an interspecies uncertainty factor would not be applied (Table 4-25b).

Total Dioxin Equivalents (TEQs) in Eggs of Striped Bass

Of all the laboratory studies examined (Table 4-7), no 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. Studies of salmonids are not used to develop the primary TRVs because salmonids are among the most sensitive species tested (Table 4-7). Therefore, the lowest appropriate non-salmonid LOAEL and NOAEL from the selected applicable studies are used to derive TRVs for the striped bass. The study by Elonen *et al.* (1998) on the channel catfish (Table 4-7) is selected for development of TRVs for the striped bass. In that study, significant early life stage mortality (72% compared to none in the control) 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 conversion factor is not applied.

On the basis of laboratory toxicity studies:

The LOAEL TRV for the striped bass is 18 μg TEQs/kg lipid (Table 4-25a).
The NOAEL TRV for the striped bass is 8.0 μg TEQs/kg lipid (Table 4-25a).

These are selected as the final TRVs for use in the risk assessment.

Because the striped bass and the channel catfish are not in the same taxonomic family, an interspecies uncertainty factor of ten could be applied (Table 4-25b). For comparative purposes, Table 4-25b presents TRVs that would be derived using interspecies uncertainty factors. However, interspecies uncertainty factors are not used in the derivation of final TRVs for this risk assessment (Table 4-25a).

Alternative TRVs developed from laboratory studies conducted on salmonids are presented for comparison. The lowest salmonid LOAEL (0.6 μg TEQs/kg lipid) and corresponding NOAEL (0.29 μg TEQs/kg lipid) from the selected applicable studies are used to derive alternative TRVs for the striped bass (Table 4-25a). In a study by Walker *et al.* (1994), 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 TEQs/kg lipid. Because the experimental study is based on the concentration in the egg, rather than an estimated dose, a subchronic-to-chronic conversion factor is not applied. Because salmonids are among the most sensitive species tested, an interspecies uncertainty factor of ten would not be applied (Table 4-25b).

Of all field studies examined (Table 4-8), no 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. Therefore, the most appropriate NOAEL from the selected applicable studies, the value of 0.1 μg TEQs/kg lipid identified by Guiney *et al.* (1996) for the lake trout, is used to derive a TRV for the striped bass. Because the experimental study is based on the concentration in the egg, rather than an estimated dose, a subchronic-to-chronic conversion factor is not applied.

On the basis of field studies:

The NOAEL TRV for the striped bass is 0.1 µg TEQs/kg lipid (Table 4-25a).

This study is not selected for development of final TRVs because it was conducted on a highly sensitive species.

Because salmonids such as the lake trout are among the most sensitive species tested, an interspecies uncertainty factor of ten would not be applied (Table 4-25b).

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.

4.2.4.1 Tree Swallow (*Tachycineta bicolor*)

Total PCBs in the Diet of the Tree Swallow

Of all laboratory studies examined (Table 4-9), no 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. Therefore, the lowest appropriate LOAEL and NOAEL from the selected studies, the LOAEL (7.1 mg/kg/d) and NOAEL (1.8 mg/kg/d) for the ring-necked pheasant (Dahlgren *et al.*, 1972), are used to develop TRVs for the tree swallow. Dahlgren *et al.* (1972) found significantly reduced ($p<0.01$) egg production by hens that had been fed PCBs for a period of 16 weeks. Egg production by hens fed PCBs at the LOAEL was 32-97% that of control hens. Because the study was conducted over a 16-week period, a subchronic to chronic conversion factor is not applied.

On the basis of laboratory toxicity studies:

The LOAEL TRV for the tree swallow is 7.1 mg PCBs/kg/day (Table 4-26a).
The NOAEL TRV for the tree swallow is 1.8 mg PCBs/kg/day (Table 4-26a).

Because gallinaceous birds, such as the ring-necked pheasant, are among the most sensitive of avian species to the effects of PCBs, an interspecies uncertainty factor would not be applied (Table 4-26b).

Of all the field studies examined (Table 4-10), two studies were identified that examined concentrations of PCBs in food of tree swallows in comparison to measures of reproductive effects. Custer *et al.* (1998) reported that measures of reproductive success (*e.g.*, clutch and egg success) were not significantly different for birds from a PCB-contaminated site in comparison to birds from a reference site. In that study, dietary doses of PCBs, estimated on the basis of average measured food concentrations at the site (2 samples) and a food ingestion rate of 0.9 kg food/kg body wt/day for the tree swallow, ranged from 0.38 to 0.55 mg PCBs/kg/day.

Dietary doses of PCBs to tree swallows can also be estimated on the basis of composite samples of food taken from feeding tree swallows on the Hudson River in 1995 (USEPA, 1998). Dietary doses (estimated using the aforementioned food ingestion rate) for the tree swallow at three locations on the Hudson River are 0.08, 6.0, and 16.1 mg PCBs/kg/day. The final TRV is based on the highest concentration shown to be without adverse effects in both field studies, a value of 16.1 mg PCBs/kg/day.

On the basis of field studies:

The NOAEL TRV for the tree swallow is 16.1 mg PCBs/kg/day (Table 4-26a).

This study is selected as the final NOAEL TRV for the risk assessment. A LOAEL TRV is not developed from the laboratory study, because tree swallows appear to be much less sensitive than other species, and the field-based NOAEL TRV for tree swallows is higher than the laboratory-based LOAEL for most other species.

Because tree swallows were the subject of both these studies, an interspecies uncertainty factor would not be applied (Table 4-26b).

Total Dioxin Equivalents (TEQs) in the Diet of the Tree Swallow

Of all the laboratory studies examined (Tale 4-11), no 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. Therefore, the lowest appropriate LOAEL (0.14 μg TEQs/kg/day) and corresponding 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 tree swallow. Nosek *et al.* (1992) observed reduced fertility (64% lower than control) and increased embryo mortality (100% lower than the control) in ring-necked pheasants that received weekly intraperitoneal injections of 2,3,7,8-TCDD over the course of 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 of the short-term nature of the exposure (10 weeks), a subchronic-to-chronic conversion 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 4-26a).
The NOAEL TRV for the tree swallow is 0.0014 μg TEQs/kg/day (Table 4-26a).

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 would not be applied (Table 4-26b). Note that the study by Nosek *et* al. (1992) was also selected by the USEPA as the basis for development of concentrations of 2,3,7,8-TCDD associated with risk to avian receptors (USEPA, 1993).

Of al the field studies examined (Table 4-12), two studies were identified that examined the effects of dioxin-like compounds in the diets of tree swallows. Custer *et* al. (1998) reported that measures of reproductive success (*e.g.*, clutch and egg success) were not significantly different for birds from a PCB-contaminated site in comparison to birds from a reference site. In that study, dietary doses of dioxin-like compounds were as high as 0.08 μg TEQs/kg/day.

Dietary doses of dioxin-like compounds to the tree swallow can also be estimated on the basis of composite samples of food taken from feeding tree swallows on the Hudson River in 1995 (USEPA, 1998). Dietary doses (estimated using the aforementioned food ingestion rate) for the tree swallow at three locations on the Hudson River are: 0.12, 1.8, and 4.9 μg TEQs/kg/day. The final TRV is based on the highest concentration shown to be without adverse effects in the 1995 field study, a value of 4.9 μg TEQs/kg/day.

On the basis of the field studies:

The NOAEL TRV for the tree swallow is 4.9 μg TEQs/kg/day (Table 4-26a).

This study is selected as the final NOAEL TRV for the risk assessment. A LOAEL TRV is not developed from the laboratory study, because tree swallows appear to be much less sensitive than other species, and the field-based NOAEL TRV for tree swallows is higher than the LOAELs for most other species.

Because this TRV is derived from studies of tree swallows, an interspecies uncertainty factor of 10 would not be applied (Table 4-26b).

Total PCBs in Eggs of the Tree Swallow

Of all the laboratory studies examined (Table 4-13), no 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. Therefore, the lowest appropriate LOAEL from the selected applicable studies (Table 4-13) is used to develop TRVs for the tree swallow. The study by Peakall and Peakall (1973) on ring doves is selected for development of TRVs. Peakall and Peakall (1973) found significantly reduced hatching (23-64% less than control) and fledgling success (70-79% less than control) in eggs of ring doves that had been fed PCBs over two generations. Because only a single dose was tested, a LOAEL to NOAEL conversion factor of ten is applied to estimate a NOAEL from this study. Because the experimental study measured actual concentrations in the egg, rather than reporting a surrogate dose, a subchronic-to-chronic conversion factor is not applied.

On the basis of laboratory toxicity studies:

The LOAEL TRV for the tree swallow egg is 16 mg PCBs/kg egg (Table 4-26a).
The NOAEL TRV for the tree swallow egg is 1.6 mg PCBs/kg egg (Table 4-26a).

Because the ring dove and the tree swallow are not in the same taxonomic family, an uncertainty factor of ten could be applied (Table 4-26b). For comparative purposes, Table 4-26b presents TRVs that would be derived using interspecies uncertainty factors. However, interspecies uncertainty factors are not used in the derivation of final TRVs for this risk assessment (Table 4-26a).

Of all the field studies examined (Table 4-14), several studies were identified that examined effects of PCBs on eggs of the tree swallow. 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 (USFWS, 1997, McCarty and Secord, 1999a, 1999b). 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 (USFWS, 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 swallow egg is 26.7 mg PCBs/kg egg (Table 4-26a).

This study is selected as the final NOAEL TRV for the risk assessment. A LOAEL TRV is not developed from the laboratory study, because tree swallows appear to be much less sensitive than other species, and the field-based NOAEL TRV for tree swallows is higher than the laboratory-based LOAEL for most other species.

Because this TRV is developed from studies of tree swallows, an interspecies uncertainty factor of ten would not be applied (Table 4-26b).

Total Dioxin Equivalents (TEQs) in Eggs of the Tree Swallow

Of all the laboratory studies examined (Table 4-15), no 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. Therefore, the lowest appropriate LOAEL (4.0 μg TEQs/kg egg) and corresponding NOAEL (1.0 μg TEQs/kg egg) from the applicable studies are used to develop TRVs for the tree swallow. Powell *et al.* (1997) found significantly increased embryo mortality (almost twice that of the control) in eggs of double-crested cormorants that were injected with 4.0 μg 2,3,7,8-TCDD/kg egg. This effect was not observed in eggs injected with 1.0 or less μg 2,3,7,8-TCDD/kg egg. Because the study measured the actual dose to the eggs, rather than reporting a surrogate dose, a subchronic-to-chronic conversion factor is not applied.

On the basis of laboratory toxicity studies:

The LOAEL TRV for the tree swallow egg is 4.0 μg TEQs/kg egg (Table 4-26a).
The NOAEL TRV for the tree swallow egg is 1.0 μg TEQs/kg egg (Table 4-26a).

Because the double-crested cormorant and the tree swallow are in different taxonomic families, an interspecies uncertainty factor could be applied (Table 4-26b). For comparative purposes, Table 4-26b presents TRVs that would be derived using interspecies uncertainty factors. However, interspecies uncertainty factors are not used in the derivation of final TRVs for this risk assessment (Table 4-26a).

Of all the field studies examined (Table 4-16), two studies were identified that examined effects of dioxin-like compounds on tree swallows. 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. 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 swallow egg is 13 μg TEQs/kg egg (Table 4-26a).

This study is selected as the final NOAEL TRV for the risk assessment. A LOAEL TRV is not developed from laboratory studies, because tree swallows appear to be much less sensitive than other species, and the field-based NOAEL TRV for tree swallows is higher than the laboratory-based LOAEL for most other species.

Because this study evaluated tree swallows, an interspecies uncertainty factor would not be applied (Table 4-26b).

4.2.4.2 Mallard (*Anas platyrhychos*)

Total PCBs in Diet of the Mallard

Of all of the laboratory studies examined (Table 4-9), five studies were identified which examined effects of PCBs in the diet on mallards. Of two studies that identified a LOAEL, the study by Haseltine and Prouty (1980) examined a more sensitive endpoint (growth rather than mortality). Haseltine and Prouty (1980) observed significantly reduced weight gain in adult mallards (11% less than the control) after a 12-week exposure to 150 ppm Aroclor-1242 in food. Because only a single dose was tested, a LOAEL to NOAEL conversion factor of ten is applied to estimate a NOAEL from this study. Because the study was conducted over a 12-week period, a subchronic to chronic conversion factor is not applied (Table 4-26b).

On the basis of laboratory toxicity studies:

The LOAEL TRV for the mallard is 41 mg PCBs/kg/day (Table 4-26a).
The NOAEL TRV for the mallard is 4.1 mg PCBs/kg/day (Table 4-26).

These are selected as the final TRVs for use in the risk assessment.

Because the study was conducted using mallards, an interspecies uncertainty factor of ten would not be applied (Table 4-26b).

Of all the field studies examined (Table 4-10), no 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. The studies on the tree swallow are not used because the tree swallow appears to be highly insensitive to the effects of PCBs. The study on the tern is not used because the dose is estimated, not measured.

Total Dioxin Equivalents (TEQs) in Diet of the Mallard

Of all the laboratory studies examined (Table 4-11), no 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. Therefore, the lowest appropriate LOAEL (0.14 μg TEQs/kg/day) and corresponding 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. Because of the short-term nature of the exposure in this study (10 weeks), a subchronic-to-chronic conversion 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 4-26a).
The NOAEL TRV for the mallard is 0.0014 μg TEQs/kg/day (Table 4-26a).

These are selected as the final TRVs for use in the risk assessment.

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 would not be applied (Table 4-26b).

Of all the field studies examined (Table 4-12), no appropriate 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. The study on the tree swallow is not used because the tree swallow appears to be highly insensitive to the effects of PCBs.

Total PCBs in Eggs of the Mallard

Of all the laboratory studies examined (Table 4-13), one study was identified that examined the toxicity of PCBs in eggs of the mallard. Haseltine and Prouty (1980) found no significant effect on reproductive endpoints such as hatching success, survival, or weight gain in the young of mallards fed 150 ppm Aroclor-1242 in food for 12 weeks. The mean PCB concentration measured in eggs in this study was 105 ppm. Because the study measured actual concentrations in the egg, rather than reporting a surrogate dose, a subchronic-to-chronic conversion factor is not applied.

On the basis of laboratory toxicity studies:

The NOAEL TRV for the mallard egg is 105 mg PCBs/kg egg (Table 4-26a).

This study is selected as the final NOAEL TRV for the risk assessment. A LOAEL TRV is not developed from the laboratory study, because the mallard appears to be less sensitive than other species, and the field-based NOAEL TRV for mallard is higher than the laboratory-based LOAELs for most other species.

Because this study was conducted on mallards, an interspecies uncertainty factor would not be applied (Table 4-26b).

Of all the field studies examined (Table 4-14), one study was identified that examined effects of PCBs in eggs of the red-breasted merganser, *Mergus serrator*, a species in the same taxonomic family as the mallard (Tables 4-23). Heinz *et al.* (1983) found that levels of PCBs they measured did not have a significant effect on red-breasted merganser reproductive success followed to the point of departure of ducklings from the nest. The geometric mean of PCBs measured in randomly selected eggs was 17.58 mg PCBs/kg egg, while in unhatched eggs it was slightly higher (although not significantly different) at 19.3 mg PCBs/kg egg. Thus, 19.3 mg PCBs/kg egg was the NOAEL for this study. Because the NOAEL is based on measured concentrations, a subchronic-to-chronic conversion factor is not applied.

On the basis of field studies:

The NOAEL TRV for the mallard egg is 19.3 mg PCBs/kg egg (Table 4-26a).

This study is not selected as the final TRV because the laboratory study described above is conducted on mallards, and the field study is conducted on a less closely related species.

Because the mallard and the red-breasted merganser are in the same taxonomic family, an interspecies uncertainty factor would not be applied (Table 4-26b).

Total Dioxin Equivalents (TEQs) in Eggs of the Mallard

Of all laboratory examined (Table 4-15), one study was identified that examined the toxicity of dioxin-like compounds in the eggs of the mallard. Brunstrom and Reutergardh found no effect on embryo mortality in mallard eggs injected with 100 µg BZ#77/kg egg. The effective concentrations of BZ#77 are multiplied by the avian TEF for BZ#77 (0.05) to estimate TRVs on a dioxin basis. Because the study is based on an actual measured dose to the egg, rather than on a surrogate dose, a subchronic-to-chronic conversion factor is not applied.

On the basis of laboratory toxicity studies:

The NOAEL TRV for the mallard egg is 5 µg TEQs/kg egg (Table 4-26a).

This study is selected as the final NOAEL TRV for the risk assessment. A LOAEL TRV is not developed from laboratory study, because mallards appear to be less sensitive than other species, and the field-based NOAEL TRV for the mallard is higher than the laboratory-based LOAELs for most other species.

Because this study was conducted on mallards, an interspecies uncertainty factor would not be applied (Table 4-26b).

Of all the field studies examined (Table 4-16), two studies were identified that examined effects of dioxin-like compounds in eggs of the wood duck, *Aix sponsa*, a species in the same family as the mallard. 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 (30-40%) reduced nest success, hatching success , and duckling production, at concentrations of 0.020 µg TEQs/kg egg. In general, these effects were not observed at concentrations of 0.005 µg TEQs/kg egg; hatching success was slightly reduced at that concentration, but the number of live ducklings produced was not significantly reduced. Measured concentrations of organochlorine pesticides and PCBs were low and were not believed to be biologically significant. Because of the relevance of this study to the mallard, the LOAEL (0.02 µg TEQs/kg egg) and NOAEL (0.005 µg TEQs/kg egg) from these studies are selected for development of a field-based TRV for the mallard. Note that this study used TEFs provided by USEPA (1989) to calculate TEQs, which may differ slightly from TEFs used in this report (Van den Berg *et* al., 1998). Potential differences in effect concentrations that are based on use of differing TEFs are estimated at 12 to 30% (See sections on great blue herons and mink). Because the LOAEL and NOAEL are based on measured concentrations, a subchronic-to-chronic conversion 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 4-26a).
The NOAEL TRV for the mallard egg is 0.005 μg TEQs/kg egg (Table 4-26a).

This study is not selected as the final TRV because the laboratory study described above is conducted on mallards and the field study is conducted on a less closely related species. Because the mallard and the wood duck are in the same family, an interspecies uncertainty factor would not be applied (Table 4-26b).

4.2.4.3 Belted Kingfisher (*Ceryle alcyon*)

Total PCBs in the Diet of the Belted Kingfisher

Of al the laboratory studies examined (Table 4-9), no 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. Therefore, the lowest appropriate LOAEL and NOAEL from the selected studies, the LOAEL (7.1 mg/kg/d) and NOAEL (1.8 mg/kg/d) for the ring-necked pheasant (Dahlgren *et al.*, 1972), are used to develop TRVs for the belted kingfisher. Dahlgren *et al.* (1972) found significantly reduced ($p<0.01$) egg production by hens that had been fed PCBs for a period of 16 weeks. Egg production by hens fed PCBs at the LOAEL was 32-97% that of control hens. Because the study was conducted over a 16-week period, a subchronic-to-chronic conversion factor is not applied.

On the basis of laboratory toxicity studies:

The LOAEL TRV for the belted kingfisher is 7.1 mg PCBs/kg/day (Table 4-26a).
The NOAEL TRV for the belted kingfisher is 1.8 mg PCBs/kg/day (Table 4-26a).

These are selected as the final TRVs for use in the risk assessment.

Because gallinaceous birds, such as the ring-necked pheasant, are among the most sensitive of avian species to the effects of PCBs, an interspecies uncertainty factor would not be applied (Table 4-26b).

Of all the field studies examined (Table 4-10), no studies were identified that examined effects of dietary exposure to PCBs on growth, reproduction, or mortality of the belted kingfisher or to a species in the same taxonomic family as the kingfisher. The tree swallow studies are not used, because tree swallows appear to be much less sensitive than other species to the effects of PCBs. The tern study is not used because an estimated, rather than a measured, dose is reported.

Total Dioxin Equivalents (TEQs) in the Diet of the Belted Kingfisher

Of all the laboratory studies examined (Table 4-11), no 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. 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 of the short-term nature of the exposure (10 weeks), a subchronic-to-chronic conversion 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 4-26a).
The NOAEL TRV for the belted kingfisher is 0.0014 μg TEQs/kg/day (Table 4-26a).

These are selected as the final TRVs for use in the risk assessment.

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 would not be applied (Table 4-26b).

Of all the field studies examined (Table 4-12), no 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. The tree swallow studies are not used because the tree swallows appear to be much less sensitive than other species to the effects of PCBs.

Total PCBs in Eggs of the Belted Kingfisher

Of all the laboratory studies examined (Table 4-13), no 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. Therefore, the lowest appropriate LOAEL from the selected applicable studies (Table 4-13) is used to develop TRVs for the belted kingfisher. The study by Peakall and Peakall (1973) on ring doves is selected for development of TRVs. Peakall and Peakall (1973) found significantly reduced hatching (23-64% less than control) and fledgling (70-79% less than control) success in eggs of ring doves that had been fed PCBs over two generations. Because only a single dose was tested, a LOAEL to NOAEL conversion factor of ten is applied to estimate a NOAEL from this study. Because the experimental study measured actual concentrations in the egg, rather than reporting a surrogate dose, a subchronic-to-chronic conversion factor is not applied.

On the basis of laboratory toxicity studies:

The LOAEL TRV for the belted kingfisher egg is 16 mg PCBs/kg egg (Table 4-26a).
The NOAEL TRV for the belted kingfisher egg is 1.6 mg PCBs/kg egg (Table 4-26a).

This study is not selected for development of final TRVs because only one dose was tested, and the NOAEL must be estimated. The field study described below is the preferred study.

Because the ring dove and the belted kingfisher are not in the same taxonomic family, an uncertainty factor of ten could be applied (Table 4-26b). For comparative purposes, Table 4-26b presents TRVs that would be derived using interspecies uncertainty factors. However, interspecies uncertainty factors are not used in the derivation of final TRVs for this risk assessment (Table 4-26a).

Of all the field studies examined (Table 4-14), no studies were identified that examined effects of PCBs in eggs of the belted kingfisher or in eggs of a species in the same taxonomic family as the kingfisher. Therefore, the lowest appropriate LOAEL (7.6 mg PCBs/kg egg) and corresponding NOAEL (4.7 mg PCBs/kg egg) from the applicable studies are used to develop TRVs for belted kingfisher eggs. Hoffman *et al.* (1993) found that hatching success in common tern eggs from PCB-contaminated industrial areas was significantly reduced compared to non-industrialized control areas. Hatching success in non-industrialized areas ranged from 73-85% while hatching success in industrialized areas ranged from 24-71%. The lowest mean egg PCB concentration from the industrial areas was considered to be the LOAEL while the highest mean egg PCB concentration from the non-industrialized areas was considered to be the NOAEL. Because the study is based on an actual measured concentration in the egg, rather than a surrogate dose, a subchronic-to-chronic conversion factor is not applied.

On the basis of field studies:

The LOAEL TRV for the belted kingfisher egg is 7.6 mg PCBs/kg egg (Table 4-26a).
The NOAEL TRV for the belted kingfisher egg is 4.7 mg PCBs/kg egg (Table 4-26a).

These are selected as the final TRVs for use in the risk assessment because the study presents a measured NOAEL, rather than an estimated NOAEL, as is presented in the laboratory study.

Because the common tern and the kingfisher are not in the same taxonomic family, an uncertainty factor of ten could be applied (Table 4-26b). For comparative purposes, Table 4-26b presents TRVs that would be derived using interspecies uncertainty factors. However, interspecies uncertainty factors are not used in the derivation of final TRVs for this risk assessment (Table 4-26a).

Total Dioxin Equivalents (TEQs) in Eggs of the Belted Kingfisher

Of all the laboratory studies examined (Table 4-15), no 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. Therefore, the lowest appropriate LOAEL (4.0 μg TEQs/kg egg) and corresponding NOAEL (1.0 μg TEQs/kg egg) (Powell *et al.*, 1997) from the applicable studies are used to develop TRVs for the belted kingfisher. Powell *et al.* (1997) found significantly increased embryo mortality (almost twice that of the control) in eggs of double-crested cormorants that were injected with 4.0 μg 2,3,7,8-TCDD/kg egg. This effect was not observed in eggs injected with 1.0 or less μg 2,3,7,8-TCDD/kg egg. Because the study measured the actual dose to the eggs, rather than reporting a surrogate dose, a subchronic-to-chronic conversion factor is not applied.

On the basis of laboratory toxicity studies:

The LOAEL TRV for the belted kingfisher egg is 4.0 μg TEQs/kg egg (Table 4-26a).
The NOAEL TRV for the belted kingfisher egg is 1.0 μg TEQs/kg egg (Table 4-26a).

These are selected as the final TRVs for use in the risk assessment.

Because the double-crested cormorant and the belted kingfisher are in different taxonomic families, an interspecies uncertainty factor could be applied (Table 4-26b). For comparative purposes, Table 4-26b presents TRVs that would be derived using interspecies uncertainty factors. However, interspecies uncertainty factors are not used in the derivation of final TRVs for this risk assessment (Table 4-26a).

Of all the field studies examined (Table 4-16), no 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. Therefore, the lowest appropriate LOAEL (1.06 μg TEQs/kg egg) and corresponding NOAEL (0.61 μg TEQs/kg) from the selected studies are used to develop TRVs for the belted kingfisher. Harris *et al.* (1993) compared hatching success in Forster's tern eggs from Green Bay, Michigan in 1983 to eggs from Green Bay in 1988, after a period of low flows and associated reduced PCB loading into Green Bay. Hatching success in the 1988 eggs was significantly higher (27%) than in 1983; median TEQs were 1.06 μg TEQs/kg egg in 1983 and 0.61 μg TEQs/kg egg in 1988. Hatching success in 1988 eggs was not significantly different from eggs at a clean site studied in 1983. Because the study measured the actual dose to the eggs, rather than reporting a surrogate dose, a subchronic-to-chronic conversion factor is not applied.

On the basis of field studies:

The LOAEL TRV for the belted kingfisher egg is 1.06 μg TEQs/kg egg (Table 4-26a).
The NOAEL TRV for the belted kingfisher egg is 0.61 μg TEQs/kg egg (Table 4-26a).

This study is not selected for development of final TRVs because of the small sample size (n=6 eggs) used in the study in comparison to the laboratory study described above.

Because the Forster's tern and the belted kingfisher are in different taxonomic families, an interspecies uncertainty factor could be applied (Table 4-26b). For comparative purposes, Table 4-26b presents TRVs that would be derived using interspecies uncertainty factors. However, interspecies uncertainty factors are not used in the derivation of final TRVs for this risk assessment (Table 4-26a).

4.2.4.4 Great Blue Heron (*Ardea herodias*)

Total PCBs in the Diet of the Great Blue Heron

Of all the laboratory studies examined (Table 4-9), no 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. Therefore, the lowest appropriate LOAEL and NOAEL from the selected studies, the LOAEL (7.1 mg/kg/d) and NOAEL (1.8 mg/kg/d) for the ring-necked pheasant (Dahlgren *et al.*, 1972), are used to develop TRVs for the great blue heron. Dahlgren *et al.* (1972) found significantly reduced ($p<0.01$) egg production by hens that had been fed PCBs for a period of 16 weeks. Egg production by hens fed PCBs at the LOAEL was 32-97% that of control hens. Because the study was conducted over a 16-week period, a subchronic to chronic conversion factor is not applied.

On the basis of laboratory toxicity studies:

The LOAEL TRV for the great blue heron is 7.1 mg PCBs/kg/day (Table 4-26a).
The NOAEL TRV for the great blue heron is 1.8 mg PCBs/kg/day (Table 4-26a).

These are selected as the final TRVs for use in the risk assessment.

Because gallinaceous birds, such as the ring-necked pheasant, are among the most sensitive of avian species to the effects of PCBs, an interspecies uncertainty factor would not be applied (Table 4-26b).

Of all the field studies examined (Table 4-10), no 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. The studies on tree swallows are not used because tree swallows appear to be much less sensitive than other species to the effects of PCBs. The study on terns is not used because it reports an estimated dose, rather than a measured dose.

Total Dioxin Equivalents (TEQs) in the Diet of the Great Blue Heron

Of all the laboratory studies examined (Table 4-11), no 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. 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 of the short-term nature of the exposure of the experimental study (10 weeks), a subchronic-to-chronic conversion 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 4-26a).
The NOAEL TRV for the great blue heron is 0.0014 μg TEQs/kg/day (Table 4-26a).

These are selected as the final TRVs for use in the risk assessment.

Because gallinaceous birds, such as the pheasant, are among the most sensitive birds to the effects of 2,3,7,8-TCDD (Table 4-11), an interspecies uncertainty factor would not be applied (Table 4-26b).

Of all the field studies examined (Table 4-12), no 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. The tree swallow studies are not used because the tree swallows appear to be much less sensitive than other species to the effects of PCBs.

Total PCBs in Eggs of the Great Blue Heron

Of all the laboratory studies examined (Table 4-13), no 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. Therefore, the lowest appropriate LOAEL from the selected applicable studies (Table 4-13) is used to develop TRVs for the great blue heron. The study by Peakall and Peakall (1973) on ring doves is selected for development of TRVs. Peakall and Peakall (1973) found significantly reduced hatching (23-64% less than control) and fledgling (70-79% less than control) success in eggs of ring doves that had been fed PCBs over two generations. Because only a single dose was tested, a LOAEL to NOAEL conversion factor of ten is applied to estimate a NOAEL from this study. Because the experimental study measured actual concentrations in the egg, rather than reporting a surrogate dose, a subchronic-to-chronic conversion factor is not applied.

On the basis of laboratory toxicity studies:

The LOAEL TRV for the great blue heron egg is 16 mg PCBs/kg egg (Table 4-26a).
The NOAEL TRV for the great blue heron egg is 1.6 mg PCBs/kg egg (Table 4-26a).

Because the ring dove and the great blue heron are not in the same taxonomic family, an uncertainty factor of ten could be applied (Table 4-26b). For comparative purposes, Table 4-26b presents TRVs that would be derived using interspecies uncertainty factors. However, interspecies uncertainty factors are not used in the derivation of final TRVs for this risk assessment (Table 4-26a).

Of all the field studies examined (Table 4-14), one study was identified that examined effects of PCBs in eggs of the great blue heron, and one study that examined effects in black-crowned night herons. The study by Halbrook *et al.* (1999), which found no difference in reproductive success (mean chicks fledged per nest) among great blue herons from PCB-contaminated areas and non-contaminated areas, is selected because it examined the great blue heron. The highest concentration of Aroclor 1260 in eggs from contaminated areas (2.01 mg PCBs/kg egg) is used to develop the NOAEL TRV for great blue heron eggs.

Because the aforementioned study only identified a NOAEL, a LOAEL from another study is used to develop a LOAEL TRV for great blue heron eggs. Hoffman *et al.* (1993) found that hatching success in common tern eggs (7.6 mg PCBs/kg egg) from PCB-contaminated industrial areas was significantly reduced compared to non-industrialized control areas. Hatching success in non-industrialized areas ranged from 73-85% while hatching success in industrialized areas ranged from 24-71%. The lowest mean egg PCB concentration from the industrial areas was considered to be the LOAEL. Because the study is based on an actual measured concentration in the egg, rather than a surrogate dose, a subchronic-to-chronic conversion factor is not applied.

On the basis of field studies:

The LOAEL TRV for the great blue heron egg is 7.6 mg PCBs/kg egg (Table 4-26a).
The NOAEL TRV for the great blue heron egg is 2.01 mg PCBs/kg egg (Table 4-26a).

These are selected as the final TRVs for use in the risk assessment.

Because the Halbrook *et al.* study evaluated great blue herons, an interspecies uncertainty factor would not be applied to the NOAEL TRV (Table 4-26b).

Total Dioxin Equivalents (TEQs) in Eggs of the Great Blue Heron

Of all the laboratory studies examined (Table 4-15), one study was identified that examined effects of dioxin-like compounds on eggs of the great blue heron. Janz and Bellward (1996) found no substantial adverse effect on 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 conversion factor is applied.

On the basis of the laboratory toxicity study:

The NOAEL TRV for the great blue heron egg is 2.0 μg TEQs/kg egg (Table 4-26a).

This study is not selected for development of final TRVs because of the small sample size (n=6) in comparison to sample size reported in the field studies described below (n=11), and because the field study provides both a LOAEL and a NOAEL.

Because the study was conducted on the great blue heron, an interspecies uncertainty factor would not be applied (Table 4-26b).

Of all the field studies examined (Table 4-16), three 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. 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). These studies are selected for development of TRVs for the great blue heron because the study reported concentrations of PCBs, in addition to concentrations of dioxins and furans. These studies found no significant difference in hatchability of eggs, but a significant reduction in body weight (9% lower than controls, Hart *et al.*, 1991) 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 conversion factor is not applied. Because the study was conducted on the great blue heron, an interspecies uncertainty factor would not be applied (Table 4-26b).

On the basis of field studies:

The LOAEL TRV for the great blue heron egg is 0.5 μg TEQs/kg egg (Table 4-26a).
The NOAEL TRV for the great blue heron egg is 0.3 μg TEQs/kg egg (Table 4-26a).

These are selected as the final TRVs for use in the risk assessment.

Because the study was conducted on the great blue heron, an interspecies uncertainty factor would not be applied (Table 4-26b).

4.2.4.5 Bald Eagle (*Haliaeetus leucocephalus*)

Total PCBs in the Diet of the Bald Eagle

Of all the laboratory studies examined (Table 4-9), no 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. Therefore, the lowest appropriate LOAEL and NOAEL from the selected studies, the LOAEL (7.1 mg/kg/d) and NOAEL (1.8 mg/kg/d) for the ring-necked pheasant (Dahlgren *et al.*, 1972), are used to develop TRVs for the bald eagle. Dahlgren *et al.* (1972) found significantly reduced ($p<0.01$) egg production by hens that had been fed PCBs for a period of 16 weeks. Egg production by hens fed PCBs at the LOAEL was 32-97% that of control hens. Because the study was conducted over a 16-week period, a subchronic to chronic conversion factor is not applied.

On the basis of laboratory toxicity studies:

The LOAEL TRV for the bald eagle is 7.1 mg PCBs/kg/day (Table 4-26a).
The NOAEL TRV for the bald eagle is 1.8 mg PCBs/kg/day (Table 4-26a).

These are selected as the final TRVs for use in the risk assessment.

Because gallinaceous birds, such as the ring-necked pheasant, are among the most sensitive of avian species to the effects of PCBs, an interspecies uncertainty factor would not be applied (Table 4-26b).

Of all the field studies examined (Table 4-10), no 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. The studies on tree swallows are not used because tree swallows appear to be much less sensitive than other species to the effects of PCBs. The tern study is not used because it reports an estimated dose, rather than a measured dose.

Total Dioxin Equivalents (TEQs) in the Diet of the Bald Eagle

Of all the laboratory studies examined (Table 4-11), no 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. 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 of the short-term nature of the exposure (10 weeks), a subchronic-to-chronic conversion 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 4-26a).
The NOAEL TRV for the bald eagle is 0.0014 μg TEQs/kg/day (Table 4-26a).

These are selected as the final TRVs for use in the risk assessment.

Because gallinaceous birds, such as the pheasant, are among the most sensitive birds to the effects of 2,3,7,8-TCDD (Table 4-11), an interspecies uncertainty factor would not be applied (Table 4-26b).

Of all the field studies examined (Table 4-12), no 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. The studies on tree swallows are not used because tree swallows appear to be much less sensitive than other species to the effects of PCBs.

Total PCBs in Eggs of the Bald Eagle

Of all the laboratory studies examined (Table 4-13), no 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. Therefore, the lowest appropriate LOAEL from the selected applicable studies (Table 4-13) is used to develop TRVs for the bald eagle. The study by Peakall and Peakall (1973) on ring doves is selected for development of TRVs. Peakall and Peakall (1973) found significantly reduced hatching (23-64% less than control) and fledgling (70-79% less than control) success in eggs of ring doves that had been fed PCBs over two generations. Because only a single dose was tested, a LOAEL to NOAEL conversion factor of ten is applied to estimate a NOAEL from this study. Because the experimental study measured actual concentrations in the egg, rather than reporting a surrogate dose, a subchronic-to-chronic conversion factor is not applied.

On the basis of laboratory toxicity studies:

The LOAEL TRV for the bald eagle egg is 16 mg PCBs/kg egg (Table 4-26a).
The NOAEL TRV for the bald eagle egg is 1.6 mg PCBs/kg egg (Table 4-26a).

Because the ring dove and the bald eagle are not in the same taxonomic family, an uncertainty factor of ten could be applied (Table 4-26b). For comparative purposes, Table 4-26b presents TRVs that would be derived using interspecies uncertainty factors. However, interspecies uncertainty factors are not used in the derivation of final TRVs for this risk assessment (Table 4-26a).

Of all the field studies examined (Table 4-14), several field studies were identified that examined the effects of PCBs in eggs of bald eagles. 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) examined relationships between bald eagle reproductive endpoints and contaminant concentrations in a number sites around the United States. Wiemeyer *et al.* (1993) reported significantly lower concentrations of PCBs in eggs in successful bald eagle nests (5.5 mg PCBs/kg egg) as compared to unsuccessful nests (8.7 mg PCBs/kg egg).

On the basis of field toxicity studies:

The LOAEL TRV for the bald eagle egg is 8.7 mg PCBs/kg egg (Table 4-26a).
The NOAEL TRV for the bald eagle egg is 5.5 mg PCBs/kg egg (Table 4-26a).

These are selected as the final TRVs for use in the risk assessment because the study was conducted on bald eagles.

Because this study was conducted on bald eagles, an interspecies uncertainty factor would not be applied (Table 4-26b).

Total Dioxin Equivalents (TEQs) in Eggs of the Bald Eagle

Of all the laboratory studies examined (Table 4-15), no 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. Therefore, the lowest appropriate LOAEL (5 μg TEQs/kg egg) and corresponding NOAEL (2 μg TEQs/kg egg) from the applicable studies (Table 4-15) are used to develop TRVs for the bald eagle. Hoffman *et al.* (1998) found significantly increased embryo mortality (33% as compared to none in the control) in American kestrel eggs that were injected with 100 μg BZ#77/kg egg (5 μg TEQs/kg egg). This effect was not observed in eggs injected with 23 μg BZ#126/kg egg (2 μg TEQs/kg egg). The effective concentrations of BZ#126 and BZ#77 are multiplied by the avian TEFs for BZ#126 (0.1) and BZ#77 (0.05) to estimate TRVs on a dioxin basis. Because the study reports a measured dose to the egg rather than a surrogate dose, no subchronic-to-chronic conversion factor is applied.

On the basis of laboratory toxicity studies:

The LOAEL TRV for the bald eagle egg is 5 μg TEQs/kg egg (Table 4-26a).
The NOAEL TRV for the bald eagle egg is 2 μg TEQs/kg egg (Table 4-26a).

This study is selected as the final LOAEL TRV for the risk assessment. The associated NOAEL TRV is not selected because a field-derived NOAEL for the bald eagle is described below.

Because the American kestrel and the bald eagle are not in the same taxonomic family, an interspecies uncertainty factor could be applied (Table 4-26b). For comparative purposes, Table 4-26b presents TRVs that would be derived using interspecies uncertainty factors. However, interspecies uncertainty factors are not used in the derivation of final TRVs for this risk assessment (Table 4-26a).

Of all the field studies examined (Table 4-16), two studies were identified that examined concentrations of PCBs in eggs of bald eagles in comparison to measures of reproductive effects. 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 a NOAEL TRV. A field study by Elliott *et al.* (1996) reported data for TEQ in the yolk sac of the bald eagle egg. This study reports a concentration of TEQs of 210 ng/kg wet weight in eggs for the Powell River, a contaminated site with a concentration that is slightly less than another nearby contaminated site, East Vancouver Island. Based on Figure 4 in Elliott *et al.* (1996), the concentration of TEQs in the East Vancouver Island site is estimated as 13,000 ng TEQs/kg lipid. Using the ratio between wet weight and lipid at the Powell River site, the wet weight concentration at East Vancouver Island is approximately 217 ng/kg. Since no significant difference was observed between the average hatching rate of the eggs collected from these two contaminated sites and the reference sites, the average concentration in eggs from the contaminated sites (214 ng/kg wet weight) is selected as the NOAEL for this study. Because the study reports a measured dose to the egg rather than a surrogate dose, no subchronic-to-chronic conversion factor is applied.

On the basis of the field studies:

The NOAEL TRV for the bald eagle egg is 0.21 μg TEQs/kg egg (Table 4-26a).

This study is selected as the final NOAEL TRV for the risk assessment.

Because the study evaluated bald eagles, an interspecies uncertainty factor would not be applied (Table 4-26b).

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

Of all the laboratory studies that were examined (Table 4-17), no studies were identified that examined the effects of PCBs on bats or on a species in the same taxonomic family or order as the bat were identified. Therefore, the lowest appropriate LOAEL (1.5 mg/kg/day) and corresponding NOAEL (0.32 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 (reduction of 15-24%) in comparison to controls. This effect was not observed at a dose of 0.32 mg/kg/day. Because of the extended duration of the experimental study (2 generations) a subchronic-to-chronic conversion factor is not applied.

On the basis of laboratory toxicity studies:

The LOAEL TRV for the little brown bat is 1.5 mg PCBs/kg/day (Table 4-27a).
The NOAEL TRV for the little brown bat is 0.32 mg PCBs/kg/day (Table 4-27a).

These are selected as the final TRVs for use in the risk assessment.

Because the rat and the little brown bat are not in the same taxonomic family, an interspecies uncertainty factor of ten could be applied (Table 4-27b). For comparative purposes, Table 4-27b presents TRVs that would be derived using interspecies uncertainty factors. However, interspecies uncertainty factors are not used in the derivation of final TRVs for this risk assessment (Table 4-27a).

Several studies were identified that examined the effects of PCBs on bats (*i.e.*, 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

Of all the laboratory studies examined (Table 4-18), no studies were identified that examined effects of dioxin-like compounds on bats or on a species in the same taxonomic family or order as the bat. 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. Fertility (number of females delivering a litter divided by the number of females placed with a male) was 57% in the 0.01 μg/kg/day F1 generation and 55% in the 0.01 μg/kg/day F2 generation, compared with 85% and 88% in the respective control groups. Because the experimental study examined exposure over three generations, a subchronic-to-chronic conversion factor is not applied.

On the basis of laboratory toxicity studies:

The LOAEL TRV for the little brown bat is 0.01 μg TEQs/kg/day (Table 4-27a).
The NOAEL TRV for the little brown bat is 0.001 μg TEQs/kg/day (Table 4-27a).

These are selected as the final TRVs for use in the risk assessment.

Because the rat and the little brown bat are not in the same taxonomic family, an interspecies uncertainty factor of ten could be applied (Table 4-27b). For comparative purposes, Table 4-27b presents TRVs that would be derived using interspecies uncertainty factors. However, interspecies uncertainty factors are not used in the derivation of final TRVs for this risk assessment (Table 4-27a).

Note that the study by Murray *et al.* (1979) was also selected by the USEPA as the basis for development of concentrations of 2,3,7,8-TCDD associated with risk to mammalian receptors (USEPA, 1993).

No field studies were identified that examined effects of dietary exposure to dioxin-like compounds on growth, reproduction, or mortality of the little brown bat or on a species in the same taxonomic family as the little brown bat.

4.2.5.2 Raccoon (*Procyon lotor*)

Total PCBs in the Diet of the Raccoon

Of all of the laboratory studies examined (Table 4-17), 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). Therefore, the lowest appropriate LOAEL (1.5 mg/kg/day) and corresponding NOAEL (0.32 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 (reduction of 15-24%) in comparison to controls. This effect was not observed at a dose of 0.32 mg/kg/day. Because of the extended duration of the experimental study (two generations), a subchronic-to-chronic conversion factor is not applied.

On the basis of laboratory toxicity studies:

The LOAEL TRV for the raccoon is 1.5 mg PCBs/kg/day (Table 4-27a).
The NOAEL TRV for the raccoon is 0.32 mg PCBs/kg/day (Table 4-27a).

These are selected as the final TRVs for use in the risk assessment.

Because the rat and the raccoon are not in the same taxonomic family, an interspecies uncertainty factor of ten could be applied (Table 4-27b). For comparative purposes, Table 4-27b presents TRVs that would be derived using interspecies uncertainty factors. However, interspecies uncertainty factors are not used in the derivation of final TRVs for this risk assessment (Table 4-27a).

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

Of all of the laboratory studies examined (Table 4-18), no studies were identified that examined effects of dioxin-like compounds on raccoons or a species in the same taxonomic family as the raccoon. 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. Because the experimental study examined exposure over three generations, a subchronic-to-chronic conversion factor is not applied.

On the basis of laboratory toxicity studies:

The LOAEL TRV for the raccoon is 0.01 μg TEQs/kg/day (Table 4-27a).
The NOAEL TRV for the raccoon is 0.001 μg TEQs/kg/day (Table 4-27a).

These are selected as the final TRVs for use in the risk assessment.

Because the rat and the raccoon are not in the same taxonomic family, an interspecies uncertainty factor of ten could be applied (Table 4-27b). For comparative purposes, Table 4-27b presents TRVs that would be derived using interspecies uncertainty factors. However, interspecies uncertainty factors are not used in the derivation of final TRVs for this risk assessment (Table 4-27a).

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). 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 0.3 mg/kg/day for a period of 9 months (from before reproduction until kits born to the females were 4 weeks of age). A total of one live kit was born to females in the dosed group while a total of 28 live kits were born to the females in the control group. Because this study evaluated a single dose only, a LOAEL-to-NOAEL conversion factor of ten was used to estimate a NOAEL. Because the study was conducted for a relatively long period over a sensitive life stage (reproduction), a subchronic-to-chronic conversion factor of ten is not applied in developing the TRVs for mink.

On the basis of laboratory toxicity studies:

The LOAEL TRV for the mink is 0.3 mg PCBs/kg/day (Table 4-27a).
The NOAEL TRV for the mink is 0.03 mg PCBs/kg/day (Table 4-27a).

This study is not selected for development of final TRVs. A field-based study (described below) is selected because it represents a longer, multigenerational study.

Because the study was conducted on mink, an interspecies uncertainty factor would not be applied (Table 4-27b).

Two field studies were identified that examined effects of PCBs in the diet of the mink (Table 4-20). The study that reported reproductive effects at the lowest dose is used to develop TRVs for the mink. Seven-month-old mink were fed diets containing various amounts of PCB-contaminated carp from Saginaw Bay, Lake Huron (Restum *et al.*, 1998); the study was continued over two generations. 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 for reduced growth rate of kits in the F1 generation was 0.04 mg PCBs/kg/day. Mean weight of F1 kits of mothers in the 0.04 mg PCBs/kg/day group was 15% lower than controls at 6 weeks of age.

Because this was the lowest concentration of PCBs tested, a LOAEL-to-NOAEL conversion factor of ten is used to estimate a NOAEL. Because the study was conducted for a relatively long period (6 months until weaning of F1 generation) over a sensitive life stage (reproduction), a subchronic-to-chronic conversion factor is not applied.

On the basis of field toxicity studies:

The LOAEL TRV for the mink is 0.04 mg PCBs/kg/day (Table 4-27a).
The NOAEL TRV for the mink is 0.004 mg PCBs/kg/day (Table 4-27a).

Because this study is multigenerational, these are selected as the final TRVs for use in the risk assessment.

Because the study was conducted on mink, an interspecies uncertainty factor would not be applied (Table 4-27b).

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 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 reproductive effects in rats. Because the experimental study examined exposure to 2,3,7,8-TCDD over three generations, a subchronic-to-chronic conversion factor is not applied.

On the basis of laboratory toxicity studies:

The LOAEL TRV for the mink is 0.01 µg TEQs/kg/day (Table 4-27a).
The NOAEL TRV is for the mink is 0.001 µg TEQs/kg/day (Table 4-27a).

These are not selected as final TRVs, because the field-based study is conducted on the mink (described below), rather than the rat.

Information on the short-term toxicity (LD50) of 2,3,7,8-TCDD to the rat and the mink (Tables 4-18, 4-21) indicates that the mink is much more sensitive than the rat, so an interspecies uncertainty factor of ten could be applied (Table 4-27b). For comparative purposes, Table 4-27b presents TRVs that would be derived using interspecies uncertainty factors. However, interspecies uncertainty factors are not used in the derivation of final TRVs for this risk assessment (Table 4-27a).

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 were 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 was significantly reduced (20% lower) 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 4-27a).
The NOAEL for the mink is 0.00008 µg TEQs/kg/day (Table 4-27a).

These are selected as the final TRVs for use in the risk assessment.

Because this study was conducted on mink, an interspecies uncertainty factor of ten would not be applied (Table 4-27b).

4.2.5.4 River Otter (*Lutra canadensis*)

Total PCBs in the Diet of the River Otter

Of all the laboratory studies examined (Table 4-17 and 4-19), no studies were identified that examined the toxic effects of PCBs on otters. Because river otter and mink are in the same

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phylogenetic family (Table 4-23), the LOAEL (0.3 mg Aroclor 1254/kg/day) for the mink from the study by Aulerich and Ringer (1977) is used to develop TRVs for the otter. Because this study evaluated a single dose only, a LOAEL-to-NOAEL conversion factor of ten is used to estimate a NOAEL. Because the study was conducted for a relatively long period over a sensitive life stage (reproduction), a subchronic-to-chronic conversion factor of ten is not applied in developing the TRVs.

On the basis of laboratory toxicity studies:

The LOAEL TRV for the river otter is 0.3 mg PCBs/kg/day (Table 4-27).
The NOAEL TRV for the river otter is 0.03 mg PCBs/kg/day (Table 4-27a).

This study is not selected for development of final TRVs. A field-based study (described below) is selected because it represents a longer, multigenerational study.

Since mink are generally considered to be among the most sensitive of mammalian species and otter are not expected to be more sensitive, and the otter and the mink are in the same taxonomic family, an interspecies uncertainty factor would not be applied (Table 4-27b).

Because river otters are closely related to mink, the LOAEL and NOAEL selected from field studies of dietary exposure to PCBs to mink are used to develop TRVs for the river otter. Restum *et al.* (1998) identified a LOAEL for reproductive effects of 0.04 mg PCBs/kg/day. Because this was the lowest concentration of PCBs tested, a LOAEL-to-NOAEL conversion factor of ten is used to estimate a NOAEL. Because the study was conducted for a relatively long period over a sensitive life stage (reproduction), a subchronic-to-chronic conversion factor is not applied.

On the basis of field studies:

The LOAEL TRV for the river otter is 0.04 mg PCBs/kg/day (Table 4-27a).
The NOAEL TRV for the river otter is 0.004 mg PCBs/kg/day (Table 4-27a).

Because this study is multigenerational, these are selected as the final TRVs for use in the risk assessment.

Since mink are generally considered to be among the most sensitive of mammalian species and otter are not expected to be more sensitive, and the otter and the mink are in the same taxonomic family, an interspecies uncertainty factor would not be applied (Table 4-27b).

Total Dioxin Equivalents (TEQs) in the Diet of the River Otter

Of all the laboratory studies examined (Table 4-18 and 4-21), 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. 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 the experimental study examined exposure over three generations, a subchronic-to-chronic conversion factor is not applied.

On the basis of laboratory toxicity studies:

The LOAEL TRV for the river otter is 0.01 μg TEQs/kg/day (Table 4-27a).
The NOAEL TRV for the river otter is 0.001 μg TEQs/kg/day (Table 4-27a).

These are not selected as final TRVs, because a field-based study conducted on the more closely-related mink (described below), rather than the rat, is available.

Because of the lack of any acute or chronic toxicity data for effects of dioxin-like compounds on the river otter, an interspecies uncertainty factor of 10 could be applied to account for potential differences in sensitivity to dioxin-like compounds between the rat and the river otter (Table 4-27b). For comparative purposes, Table 4-27b presents TRVs that would be derived using interspecies uncertainty factors. However, interspecies uncertainty factors are not used in the derivation of final TRVs for this risk assessment (Table 4-27a).

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 (20% lower) 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 of the extended exposure period of the study (182 days) a subchronic-to-chronic conversion 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).

These are selected as final TRVs, because the field-based study was conducted on the mink, rather than the rat.

Because mink and river otter are in the same taxonomic family, an interspecies uncertainty factor would not be applied (Table 4-27b).

4.3 Dose-Response Functions from the Literature

This assessment evaluates the potential for population level risks by comparing predicted cumulative frequencies of exposure to published dose-response curves from the literature (Moore *et al.*, 1999). This approach provides a method to estimate the probability of exceeding effects of varying magnitudes. These dose-response curves were only available for mink and pheasant, and were applied to predicted exposure (dose) frequencies for river otter, mink, bald eagle, and belted kingfisher. The otter and the eagle both consume larger fish, while the mink and kingfisher consume primarily forage fish. Thus, these receptors represent a range of exposure to PCB concentrations in fish.

The dose-response curves are based on data for the pheasant and mink. The pheasant data are the same data that were used to develop TRVs for great blue heron, belted kingfisher and bald eagle in this assessment (Dahlgren *et al.*, 1972) and have been applied in this way to kingfisher in the literature (Moore *et al.*, 1999). The mink data are the same data that were used to develop laboratory-based TRVs for mink and otter (however, the field-based TRVs were used in the toxicity quotient calculations). The laboratory-based TRVs for mink and otter are based on Aulerich and Ringer (1977) and are higher than the Restum *et al.* (1998) field study used to derive the point estimate TRVs. The toxicity data were used to estimate dose-response relationships using generalized linear modeling (Moore *et al.*, 1999) and the details of those estimates can be found in that publication. The lead author of that report provided the dose-response estimates for this assessment. For both of these dose-response relationships, mean responses were used in the analyses, as the original data from the toxicity studies were not available (Moore *et al.*, 1999). Risk functions for the belted kingfisher, bald eagle, mink, and river otter based on the dose-response curves are discussed for each of these receptors in the following chapter, risk characterization.

Chapter 5

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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 in this section.

In the toxicity quotient approach used in this assessment, potential risks to ecological receptors are assessed by comparing measured and modeled concentrations (Chapter 3) to toxicity benchmarks (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+) and toxic equivalency (TEQ) relative to the toxicity of the potent dioxin 2,3,7,8-TCDD.

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:

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

The following example is provided to demonstrate the calculation of the TQ. The TQ is calculated as an exposure dose estimated in Chapter 3 (in mg/kg-day) divided by a TRV derived in Chapter 4 (in mg/kg-day). The result is a unitless quotient. To calculate the NOAEL-based TQ for the mink based on modeled fish for 2001 at RM 168, take the average daily dose from Table 3-81 (1.96E-01), divided by the TRV from Table 4-27a (0.004) to yield the predicted TQ found in Table 5-73:

$$1.96E\text{-}01/0.004 = 49$$

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. This bottom-up approach is consistent with USEPA's Risk Management Guidance (USEPA, 1999d) and is used specifically because population data alone would not distinguish among changes due to the PCBs in the river and changes due to non-site related factors, such as fisheries management and habitat loss.

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

- Sustainability of a benthic invertebrate community, which is as a food source for local fish and wildlife.

- Sustainability (*i.e.*, survival, growth, and reproduction) of:

  - local forage fish populations;
  - local omnivorous fish populations; and

- local piscivorous fish populations.

- Sustainability (*i.e.*, survival, growth, and reproduction) of local wildlife including:

  - insectivorous birds;
  - waterfowl;
  - semi-piscivorous/piscivorous birds;
  - insectivorous mammals;
  - omnivorous mammals; and
  - semi-piscivorous/piscivorous omnivorous mammals.

These assessment endpoints and their associated measurement endpoints are discussed below.

5.1 Evaluation of Assessment Endpoint: Sustainability of a Benthic Invertebrate Community That Can Serve 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. The characteristics of the benthic invertebrate community are strongly affected by, and reflect, the quality of the sediment that the organisms inhabit. 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 benthic community can affect not only the animals feeding directly on benthic invertebrates, but also upper trophic level receptors.

Benthic macroinvertebrate communities in the lower TI Pool (RM 188.5 to 191.5) and at selected significant habitat locations in the Lower Hudson River (RMs 122.4, 100, 88.9, 47.3, and 25.8) were sampled and their ecological metrics were analyzed for each sampling station to determine if there is an association between PCB concentrations and benthic community structure as a measurement endpoint. A detailed analysis of the benthic invertebrate study is provided in Appendix H of the ERA (USEPA, 1999c). Sediment and water concentrations are compared to New York State and federal guidelines as other measurement endpoints.

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. 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 1-2 and Plate 1) that were selected based on PCB screening results (see Appendix B of USEPA 1999c 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 of USEPA 1999c.

The number of taxa/groups collected is similar to results found by a General Electric study, where a total of 100 macroinvertebrate species were collected in 86 cores (3-in diameter) taken during September 1997 (Exponent, 1998b). These cores were taken from below Rogers 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 of USEPA 1999c.

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 are 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 of USEPA 1999c). 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. However, 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; but data are insufficient 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 5-5) 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 Hudson River National Estuarine Research Reserve (see Chapter 2). Macroinvertebrates in the Lower Hudson River represent a heterogeneous group of organisms with a wide range of life history strategies, habitat preferences 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 RM 25, support a typical marine assemblage of benthic invertebrates, including marine oligochaetes, polychaetes, and crustaceans. The middle reaches, from RM 25 to RM 60, have a mixture of freshwater and marine benthic invertebrates, and the upper reaches, above RM 60, 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 (RM 100), 17 (RM 47.3), and 18 (RM 25.8) had a higher proportion of fine-grained sediments than Stations 12 (RM 122.4) and 15 (RM 88.9) (Figure 5-6). However, grain size was not closely correlated with TOC (Figure 5-7) or total PCB concentrations (Figure 5-8). Stations 14, 15, and 18, with mean total PCB concentrations of 0.37, 0.87, and 0.48 mg/kg respectively, had higher species diversity indices than Stations 17 and 12, which had average total PCB concentrations of 1.31 and 1.23 mg/kg, respectively (Table 5-5).

Influence of Zebra Mussels in the Lower Hudson River

The invasion of the zebra mussel (*Dreissena polymorpha*), was first detected near Catskill in the Lower Hudson River in May 1991 (Strayer *et al.*, 1996). Their distribution is strongly controlled by the distribution of suitable substrata. The highest densities (average 17,000/m^2) are found on rocks in deep (> 5 m) water. Even though such deep-water rocky areas cover only 7% of the estuary, they support 95% of the zebra mussel population (Strayer *et al.*, 1996). Zebra mussels are not a major problem in the upper river (probably due to lack of suitable substrata) and therefore research has focused on the lower river (*e.g.*, Caraco *et al.*, 2000; Strayer *et al*, 1999; 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 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). Although zebra mussels may potentially increase the bioavailability of PCBs to benthic and pelagic organisms found in the river, the effect of the zebra mussel on Lower Hudson River PCB concentrations is not known at this time. Effects of the zebra mussel, such as decline of edible particles in the water column (*e.g.*, phytoplankton and small zooplankton) and dissolved oxygen declines in the Lower Hudson River, have much more serious implication on ecosystem health than potentially increasing PCB bioavailability. Zebra mussels are not considered to influence PCB concentrations in the upper river

Summary

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.*, 1996). The benthic invertebrate community analyses could not distinguish any clear adverse effects associated with increasing PCB concentrations in the Upper or Lower Hudson River, and is considered in association with other measurement endpoints, discussed in the following sections.

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 (see Table 4-3). 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 effect, and account for the effects of PCB mixtures. The Hudson River threshold effect concentration (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 mid-range effect concentration (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 extreme effect concentration (EEC).

The SECs can be used to accurately classify freshwater, estuarine, and marine sediment as toxic and not toxic (MacDonald, 1999). They can also be used to determine the likelihood that a particular sediment sample will be toxic based on PCB concentration alone. The SECs are comparable to equilibrium partitioning-based sediment quality guidelines and to chronic toxicity thresholds that have been estimated from the results of spike toxicity tests. The SECs for PCBs refer to total PCBs found in the Hudson River, plus the degradation products and metabolites of these chemicals. The SECs do not consider the potential for bioaccumulation in aquatic species or potential effects that could occur throughout the food web as a result of PCB bioaccumulation.

The Hudson River SECs and the NYSDEC Technical Guidance for Screening Contaminated Sediments (NYSDEC, 1999a), were used as the primary sediment guidelines for comparison in this report. The NYSDEC freshwater benthic aquatic life chronic toxicity sediment criterion of 19.3 μg/gOC for freshwater was used for all stations, except stations with brackish or salt water (*i.e.*, RMs 58.7, 47.3, and 25.8) for which the saltwater criterion of 41.4 μg/gOC was used.

Table 5-6 provides ratios of observed sediment concentrations to guidelines. Ratios greater than one indicate that a concentration exceeds the guideline. The Hudson River TEC (0.04 mg/kg), MEC (0.4 mg/kg), and EEC (1.7 mg/kg) are exceeded at all upper river locations (Table 5-6). Mean PCB concentrations in the TI Pool ranged from 9.29 to 29.32 mg/kg in 1993. In the Lower Hudson, the TEC and MEC are exceeded by the average and 95% UCL sediment concentrations at all stations, with the exception of the MEC for the average PCB concentration at RM 58.7 (Table 5-6). Mean total PCB concentrations in 1993 at lower river locations ranged from 0.367 mg/kg to 1.313 mg/kg, below the EEC (1.7 mg/kg). The New York State benthic aquatic life chronic toxicity value (NYSDEC, 1999a) is exceeded at all locations (Table 5-6), except for the average concentrations at RMs 100, 58.7, and 25.8.

Persaud *et al.* (1993) lowest effect level (LEL) and Washington State 1997 *Hyallela azteca* and Microtox probable apparent effects thresholds (PAETs) are also exceeded at all upper and lower river locations. The Persaud *et al.* (1993) severe effect level of 530 mg/kg OC is exceeded at the Stillwater (RM 168) location.

Table 5-7 provides the ratios of predicted 1993-2018 sediment concentrations to sediment guidelines, using the HUDTOX model for the upper river and the Farley model for the lower river. In the Thompson Island Pool (RM 189), predicted sediment concentrations exceed the NOAA TEC, MEC, EEC, NYSDEC benthic chronic toxicity value, wildlife bioaccumulation value, Persaud *et al.* LEL, and Washington State guidelines for the entire modeling period. Results are similar for RM 168, with the exception that the EEC is only exceeded for a portion of the modeling period (until 2010). At RM 154 the TEC, LEL, NYSDEC benthic and wildlife values, and Washington State guidelines are exceeded for the duration of the modeling period. The MEC and EEC are exceeded for a portion of the modeling period.

Predicted sediment concentrations in the lower river exceed the TEC, NYSDEC wildlife value, LEL, and Washington State guidelines for the duration of the modeling period (Table 5-7). The MEC and NYSDEC benthic chronic value are exceeded for a portion of the modeling period.

5.2 Evaluation of Assessment Endpoint: Sustainability (*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 5-10 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). 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 laboratory-based NOAELs and LOAELs derived for both pumpkinseed and spottail shiner in the TI Pool (RM 189). At Stillwater, measured forage fish body burdens exceed NOAELs, but only the pumpkinseed 95% UCL exceeds the LOAEL. In the Lower Hudson River, measured forage fish concentrations exceed the pumpkinseed NOAEL at several locations (*i.e.*, RM 143.5, 137.2 for the average and the 95% UCL; 122.3 and 88.9 for the 95% UCL), but none of the concentrations exceed the LOAEL. For measured forage fish concentrations compared to the spottail shiner TRVs, none exceed one except for the 95% UCL at RM 137.2.

Tables 5-11a through 5-14b 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. The pumpkinseed predicted 25th, median, and 95th percentiles exceed one in the TI Pool on a NOAEL-basis (Table 5-11a), and fall below one after 1998 on a LOAEL basis (Table5-12a). At Stillwater, the pumpkinseed exceeds one for the NOAEL until 2002 for the 25th percentile, 2006 for the median, and for the duration of the modeling period for the 95th percentile, and exceeds one for a few years on a LOAEL basis for the 95th percentile. At RM 154, the predicted 95th percentile pumpkinseed concentration exceeds one until 2002 on a NOAEL basis. This is interpreted to mean that 95% of the population will experience the TQ that is shown or less, so by 2002, 95% of the population will experience a TQ of one or less. The pumpkinseed shows few exceedences in the Lower Hudson River on a NOAEL basis and none on a LOAEL basis (Tables 5-11b, 5-12b).

The spottail shiner exceeds one for a few years at the TI Pool and Stillwater on a NOAEL basis (Table 5-13a), and LOAEL basis (Table 5-14b). The spottail shiner shows no exceedences in the Lower Hudson River on a NOAEL or LOAEL basis (Tables 5-13b, 5-14b).

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

Tables 5-15a through 5-16b 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 5-15a and 5-15b present the results for pumpkinseed based on comparisons to a NOAEL, for the upper and lower river, respectively. None of the predicted percentiles exceed the NOAEL. Tables 5-16a and 5-16b present the results for the spottail shiner for the upper and lower river, respectively. Predicted toxicity quotients for spottail shiners do not exceed one at any time during the modeling period on a NOAEL basis.

LOAEL comparison tables are not presented because the NOAEL does not exceed the predicted concentration for any comparison (*i.e.*, if the PCB concentration is less than the NOAEL, it is also less than the LOAEL).

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

Tables 5-17a to 5-18b 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 exceed one for both the median and 95th percentiles at the TI Pool and Stillwater for the duration of the modeling period. At RM 154, the predicted median exceeds the NOAEL until 2003 and the predicted 95th percentile until 2009 (Table 5-17a). In the lower river, the predicted 95th percentiles for RM 152 and RM 113 both exceed one for the duration of the modeling period on a NOAEL basis (Table 5-17b). Brown bullhead in the 95th percentile exceed the LOAEL until 2006 at the TI Pool and until 2001 at Stillwater (Table 5-18a). They typically do not exceed the LOAEL in the Lower Hudson River (Table 5-18b).

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

Tables 5-19a and 5-19b present the results of the comparison between predicted percentiles of brown bullhead concentrations to the laboratory-derived NOAEL on a TEQ basis under future conditions for the upper and lower river, respectively. The predicted brown bullhead body burdens do not exceed the NOAEL (or the higher LOAEL) in either the Upper or Lower Hudson River for the duration of the modeling period.

LOAEL comparison tables are not presented because the NOAEL does not exceed the predicted concentration for any comparison (*i.e.*, if the PCB concentration is less than the NOAEL, it is also less than the LOAEL).

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 5-20 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 a total PCB basis exceed one for all locations for the NOAEL and LOAEL. Both the average and 95% UCL (or maximum, as appropriate) toxicity quotients exceed one. This suggests the potential for risk to the largemouth bass and brown bullhead in the Upper Hudson River down to RM 113 in the Lower Hudson River.

For the TEQ-based comparison (Table 5-20), toxicity quotients exceed one for largemouth bass and brown bullhead in the TI Pool.

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 5-21 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 average measured white perch body burdens exceed the field-based NOAEL at RM113 for 1996 and at RM152 in 1994 and on a maximum basis at RM152 during 1993, 1994 and 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 exceedences in subsequent years.

The comparisons for yellow perch show that both the laboratory-based NOAEL and LOAEL are exceeded at the TI Pool and Stillwater for the average, 95% UCL, and maximum concentrations. For the NOAEL, predicted toxicity quotients exceed ten at the TI Pool. 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, since there were no data available with which to make this conversion, the toxicity quotients were calculated on a fillet basis, which is likely to underestimate true body burdens.

On a lipid-normalized basis, all comparisons do not exceed one except for yellow perch in the TI Pool. 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 5-22 through 5-23 present the results of the comparison between predicted percentiles of white perch Tri+ PCB concentrations to the NOAEL and LOAEL, respectively. For white perch NOAEL comparison toxicity quotients at river mile 154, just above the Federal Dam, fall below one in 1995 on a median basis (Table 5-22). Predicted concentrations are not provided above the Federal Dam as white perch are typically found mainly in the Lower Hudson River. Below the Federal Dam, predicted toxicity quotients for median values fall below one on a NOAEL basis at all locations and years except during 1998. On a LOAEL basis, all predicted TQs fall below one at all locations for the entire modeling period (Table 5-23).

Tables 5-24a to 5-25b present the results for the yellow perch. On a NOAEL basis (Tables 5-24a and 5-24b), the estimated toxicity quotients exceed one for all percentiles for the duration of the modeling period in the TI Pool. At Stillwater and RM 154, predicted TQs for the 95th percentile exceed one until 2007 and 1998, respectively, and fall below one in the Lower Hudson River, with the exception of the 95th percentile of RM 152 in 1998. On a LOAEL basis (Tables 5-25a and 5-25b), the median estimated toxicity quotients at the TI Pool exceed one until 1996.

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 5-26 through 5-27b 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. All of the predicted TQs fall below one in both the Upper and Lower Hudson River for both species on a NOAEL and LOAEL basis.

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

Tables 5-28a and 5-28b present the results of the comparison between modeled largemouth bass concentrations and the laboratory-based TRVs under future conditions for the upper and lower river, respectively. The predicted toxicity quotients exceed one for all percentiles in the TI Pool (RM 189), Stillwater (RM 168), RM 152, and RM 113 for the duration of the modeling period. RM 154 toxicity quotients for median concentrations exceed one through 2004. RMs 90 and 50 show exceedences for a few years until 2000. The differences between RMs 154 and 152 are due to the use of the HUDTOX model for sediment and water concentrations in the upper river and the use of the Farley model for these variables in the lower river.

Tables 5-29a and 5-29b show that median predicted TQs in the TI Pool exceed one on a LOAEL basis until 2010, at Stillwater until 1996, and at RMs 152 and 113 until 1999. Toxicity quotients are below one at Federal Dam (RM 154) and RMs 90 and 50.

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 5-30a and 5-30b present the results of the comparison between modeled largemouth bass concentrations and the laboratory-based TRVs for TEQ under future conditions. All of the predicted TQs fall below one on a median basis in both the Upper and Lower Hudson River. Only toxicity quotients for the 95th percentile at the TI Pool until 1998 are greater than one.

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

Table 5-31 presents the results of the comparison between observed striped bass concentrations at various river miles to toxicity reference values under current conditions. This table shows that there are several exceedences on a total PCB wet weight body burden basis in 1993, 1994, and 1996. All TEQ-based egg comparisons fall below one. 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.

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

Table 5-31a presents the results of the comparison between modeled striped bass concentrations at river miles 152 and 113 to the field-based NOAEL and laboratory-derived LOAEL. At RM 152 all NOAEL comparisons and the 95th percentile LOAEL comparison exceed one for the duration of the modeling period. At RM 113 the NOAEL comparisons exceed one for the 95th percentile until 2005, and until 1999 for the median and 25th percentile. All TEQ-based comparisons fall below one at both river miles (Table 5-31b). 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

Observed whole water concentrations exceed water quality benchmarks at all locations, with the exception of the average water concentration as compared to the USEPA/NYSDEC chronic benthic aquatic life saltwater value of 0.03 μg/L at RMs 58.7, 47.3 and 25.8 (Table 5-8). Modeled water column concentrations predict that the chronic benthic aquatic life value of 0.014 μg/L (freshwater) is exceeded at RMs 189 and 168 for the entire modeling period, at RM 154 until 2006, and at lower river locations, with the exception of RM 50, for a portion of the modeling period (Table 5-9). The USEPA/NYSDEC wildlife bioaccumulation value is exceeded at all upper and lower river locations for the duration of the modeling period.

Comparison of total PCB concentrations in the water column to guidelines indicates that the level of PCBs present in the Upper and Lower Hudson River may cause adverse effects to aquatic life. Comparisons may underestimate 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 Do Measured and Modeled Sediment Concentrations Exceed Appropriate Criteria and/or Guidelines for the Protection of Aquatic Life and Wildlife?

5.2.3.1 Measurement Endpoint: Comparison of Sediment PCB Concentrations to Guidelines

Mean concentrations of PCBs at each station are compared to sediment guidelines for PCBs (see Table 4-3). The Hudson River SECs and the NYSDEC Technical Guidance for Screening Contaminated Sediments (NYSDEC, 1999a), are used as the primary sediment guidelines for comparison in this report. The NYSDEC freshwater benthic aquatic life chronic toxicity sediment criterion of 1.4 μg/gOC for freshwater and 0.03 μg/gOC for saltwater criterion are used. The NYSDEC benthic chronic value and wildlife bioaccumulation value (NYSDEC, 1999a) are exceeded at all locations in the upper and lower river on a 95% UCL basis, and on an average basis with the exception of benthic value at RMs 100, 58.7, and 25.8 (Table 5-6).

The Hudson River TEC (0.04 mg/kg), MEC (0.4 mg/kg), and EEC (1.7 mg/kg) are exceeded at all upper river locations (Table 5-6). Mean PCB concentrations in the TI Pool ranged from 9.29

to 29.32 mg/kg in 1993. In the Lower Hudson, the TEC and MEC are exceeded by the average and 95% UCL sediment concentrations at all stations, with the exception of the MEC for the average PCB concentration at RM 58.7 (Table 5-6). Mean total PCB concentrations in 1993 at lower river locations ranged from 0.367 mg/kg to 1.313 mg/kg, below the EEC (1.7 mg/kg).

Table 5-7 provides the ratios of predicted 1993-2018 sediment concentrations to sediment guidelines, using the HUDTOX model for the upper river and the Farley model for the lower river. In the Thompson Island Pool (RM 189), predicted sediment concentrations exceed the NOAA TEC, MEC, EEC, NYSDEC benthic chronic toxicity value, wildlife bioaccumulation value, Persaud *et al.* LEL, and Washington State guidelines for the entire modeling period. Results are similar for RM 168, with the exception that the EEC is only exceeded for a portion of the modeling period (until 2010). At RM 154 the TEC, LEL, NYSDEC benthic and wildlife values, and Washington State guidelines are exceeded for the duration of the modeling period. The MEC and EEC are exceeded for a portion of the modeling period.

Predicted sediment concentrations in the lower river exceed the TEC, NYSDEC wildlife value, LEL, and Washington State guidelines for the duration of the modeling period (Table 5-7). The MEC and NYSDEC benthic chronic value are exceeded for a portion of the modeling period.

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

5.2.4.1 Measurement Endpoint: Evidence from Field Studies

Extensive observational data for Hudson River fish are available for the Lower Hudson River (*e.g.*, see Klauda *et al.* 1988; Barnthouse, 2000) 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 cannot 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 stickleback, 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 is 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 have 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 federal and NY State-listed endangered shortnose sturgeon indicate that this species is reproducing in the Lower Hudson River (below the Federal Dam) and that the population numbers are increasing (*e.g.*, Bain *et al.*, 1995). 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 may be 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' populations is likely to be slow as a result of its biology. 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, it is not possible to quantify from these data the extent to which PCB exposures might impair or reduce these population growth rates. Concentrations of PCBs ranging from 22.1 to 997 ppm have been measured in sturgeon flesh (Dovel, 1981).

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.* (1992) have reported on studies of the white perch during the 1970s and 1980s. This species is a permanent resident in the Hudson and, together with the shortnose sturgeon, is one of two Hudson River species that are representative primarily of the Lower Hudson River. Wells *et al.* studied several sources of Hudson River data for the period 1975 through 1987 and concluded that the population of white perch had increased over this period. This positive population growth has occurred during a period when PCB exposure has 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.

5. Barnthouse (2000) reviewed the Draft Environmental Impact Statement (DEIS) prepared by Hudson River Utility companies (Central Hudson *et al.*, 1999) as part of the requirement for a power plant permit along the Hudson River. The fish population data are important in regard to power plant construction because cooling water intake pipes and the discharge of heated water into the river can kill fish by the millions.

Among other data, the DEIS summarized data from the striped bass mark-recapture program, initiated in 1984 and NYSDEC's striped bass sampling program, established in 1976. Adult striped bass have shown to be increasing over the last twenty years, which is not surprising considering that there has been a fishing ban in place much of that time. General improvement in water quality and density-dependence may also affect striped bass populations.

A permit was granted to the Pacific Gas & Electric company on June 2, 2000 (NY Times, 2000) to build a power plant at Athens, 30 miles south of Albany. However, the water permit allows an intake of just 180,000 gallons daily, instead of the expected 4 million, indicating that there is still concern about maintaining the Hudson River fish population and species that feed on them.

Summary

The toxicity quotients calculated for forage fish (pumpkinseed and spottail shiner), white perch, indicate that these fish are unlikely to experience adverse reproductive effects at the current time (*i.e.*, 2000). Toxicity quotients for the omnivorous brown bullhead and piscivorous yellow perch and largemouth bass show that these species may experience adverse reproductive effects, particularly in the upper river. The striped bass may also be affected in the upper reaches of the lower river.

Comparisons of measured and predicted sediment and water column concentrations also indicate the potential for risk. Field observations, conducted primarily in the lower river, show that fish populations are maintaining their numbers and even increasing. The effects of influences other than PCBs, such on the fishing ban, on fish populations have not been quantitated.

5.3 Evaluation of Assessment Endpoint: Sustainability (*i.e.*, Survival, Growth, and Reproduction) of Hudson River Insectivorous Birds (as Represented by the 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 5-32 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.

For both dietary dose and egg concentration, the NOAEL-based comparisons for the 95% UCL exceed one using the Phase 2 1993 dataset (Table 5-32) at the TI Pool and Stillwater, but are below one at Federal Dam. The NOAEL-based comparisons for the average egg concentration also exceed one at the TI Pool and Stillwater. All comparisons for the Lower Hudson River are below 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 5-33 compares total PCB modeled dietary doses for the period 1993 – 2018 for the tree swallow to the field-based NOAEL under future conditions. This TRV was derived from the USFWS data from the Hudson River. All of the modeled results for this insectivorous bird for the entire modeling period are below one. A LOAEL was not derived for this species (see Chapter 4).

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 5-34 compares predicted total PCB egg concentrations for the period 1993 – 2018 for the tree swallow to the field-based TRV. 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. Only the NOAEL-based comparison at Stillwater in 1993 exceeds one.

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 5-35 compares the estimated TEQ-based dietary dose and predicted egg concentration, respectively, to the toxicity benchmarks presented in Table 4-26a. The NOAEL-based comparison for the 95% UCL dietary dose and egg concentration exceeds one at Stillwater. The remainder of the predicted toxicity quotients for the Upper and Lower Hudson River 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 5-36 compares the estimated TEQ-based dietary dose and predicted egg concentration to insectivorous birds to field-based TRVs. None of the predicted dietary doses expressed as TEQ exceed the field-based NOAEL. A LOAEL was not used to calculate toxicity quotients, as the laboratory-derived value was greater than the NOAEL.

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

Table 5-37 compares the estimated TEQ-based predicted egg concentrations for insectivorous birds to field-based TRVs under future conditions. None of the toxicity quotients for the predicted egg concentrations exceed one. A LOAEL was not used to calculate toxicity quotients, as the laboratory-derived value was greater than the NOAEL.

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

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

All observed whole water concentrations exceeded the USEPA/NYSDEC wildlife bioaccumulation value at all locations (Table 5-8). This value is exceeded at all upper and lower river locations for the duration of the modeling period (Table 5-9).

Comparison of total PCB concentrations in the water column to guidelines indicates that the level of PCBs present in the Upper and Lower Hudson River may cause adverse effects to wildlife feeding on aquatic life. Comparisons may underestimate 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).

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

5.2.3.1 Measurement Endpoint: Observational Studies

Tree swallows (*Tachycineta bicolor*) are common along the Upper Hudson River during the spring when they are feeding in preparation for breeding (McCarty and Secord, 1999a). Anne Secord, of the US Fish and Wildlife Service, and John McCarty, of Cornell University studied the tree swallow's reproductive ecology and behavior in relation to PCB contamination.

The USFWS study (USFWS, 1997) did not demonstrate a dose-response relationship between tree swallow reproduction and PCB concentrations, but the investigators did observe statistically significant abnormal breeding behavior in 1994 relative to the Ithaca, NY reference colony and relative to data from unimpaired populations documented in the literature. Although USFWS cannot conclude that PCBs impaired reproduction, their 1994 data, in conjunction with the observations of similar abnormal reproductive behavior in other birds exposed to planar halogenated hydrocarbons, suggest that PCBs may have contributed to the observed nest abandonment. Their data more conclusively demonstrated that PCBs likely contributed to or caused abnormal nest construction in tree swallows. Impaired nest quality could have a measurable impact on reproductive success in years of adverse weather conditions or other adverse environmental conditions.

The most important conclusion developed from the USFWS tree swallow work may be that the PCB concentrations and dioxin equivalents detected in samples, particularly from the Remnant 4 and SA13 sites, were significantly higher than concentrations known to cause reproductive and developmental impairment in other birds (*e.g.*, Caspian Tern).

Summary

Toxicity quotients based on measured and modeled concentrations of PCBs in tree swallows and their eggs are generally below one. Field studies indicate that overall reproductive success is not impaired, but more subtle effects, such as abnormal nest construction, could have measurable effects if combined with other adverse environmental conditions. Insectivorous species that are more sensitive to PCBs than the modeled receptor (*i.e.*, tree swallow) may also be affected.

5.4 Evaluation of Assessment Endpoint: Sustainability (*i.e.*, Survival, Growth and Reproduction) of Local Waterfowl (as represented by Mallards)

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 5-38 provides the results of comparisons between modeled total PCB 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 and average concentration on a dietary dose basis exceeds one only at Stillwater. The rest of the Upper and Lower Hudson River locations fall below one. On a LOAEL basis, both the average and 95% UCL are below one for all river mile locations.

For the predicted egg concentrations, the NOAEL-based comparisons exceed one for the 95% UCL at Stillwater, but nowhere else along the entire river. A LOAEL was not available for egg concentrations.

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

Table 5-39 provides the results of the comparison between predicted total PCB dietary doses based on predictions for the modeling period 1993 to 2018 to the toxicity reference values. On a NOAEL basis the predicted toxicity quotients exceed one sporadically over the entire modeling period for the average dietary dose in the TI Pool. NOAEL comparisons do not exceed one at other river miles. No LOAEL based benchmarks are exceeded by modeled doses at any river mile for the entire modeling period.

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

Table 5-40 provides the results of the comparison between predicted egg concentrations and toxicity reference values based on total PCB model results for the period 1993 to 2018. These results show that predicted toxicity quotients do not exceed one for the duration of the modeling period on a NOAEL basis. A LOAEL was not available for egg concentrations.

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 5-41 provides the results of the comparison between predicted dietary doses and egg concentrations on a TEQ basis to toxicity reference values using the 1993 data. 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 dietary dose, and for both the NOAEL and LOAEL-based comparisons. For the Upper Hudson River locations, the NOAEL- and LOAEL-based comparisons for the average and 95% UCL exceed 100. For the Lower River locations, the NOAEL- and LOAEL-based comparisons for both the average and 95% UCL exceed 10, except for the LOAEL-based comparisons for average dietary dose.

For predicted egg concentrations, all NOAEL-based comparisons for the average and 95% UCL exceed one in the upper river. In the lower river, comparisons for RMs 143.5, 137.2, 122.4, 100, and 47.3 exceed one using the 95% UCL and at RM 137.2 using the average concentration.

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

Table 5-42 provides the results of the comparison between predicted TEQ-based 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, except for LOAEL-based comparisons for the Federal Dam (RM 154) after 2009. Predicted toxicity quotients exceed 100 on a NOAEL basis for the TI Pool 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 5-43 provides the results of the comparison between predicted TEQ-based egg concentrations and appropriate toxicity reference values for the period 1993 – 2018. These results show that predicted NOAEL-based toxicity quotients are above 1 for at all locations for at least a portion of the modeling period (1998 at RMs 90 and 50). All toxicity quotients fall below one by 2009. An appropriate LOAEL benchmark based on Tri+ PCBs for mallard egg concentrations was not available.

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

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

All observed whole water concentrations exceeded the USEPA/NYSDEC wildlife bioaccumulation value at all locations (Table 5-8). This value is exceeded at all upper and lower river locations for the duration of the modeling period (Table 5-9).

Comparison of total PCB concentrations in the water column to guidelines indicates that the level of PCBs present in the Upper and Lower Hudson River may cause adverse effects to wildlife feeding on aquatic life. Comparisons may underestimate 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).

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

5.4.3.1 Measurement Endpoint: Observational Studies

No PCB studies have been performed on the mallard or other waterfowl along the Hudson River. There are numerous resident and migratory waterfowl utilizing the resources of the Hudson River, which is to be expected given the high habitat quality of many areas of the Hudson River (see Section 2.1.1). Although there are no obvious PCB-related effects that have been observed in waterfowl, PCBs may be passed up to the next trophic level when waterfowl are consumed by predators, such as eagles.

Summary

Toxicity quotients for the mallard and mallard egg are generally below one on a total (Tri+) PCB basis and above one on a TEQ-basis. Water quality values indicate that some wildlife may be adversely affected by the concentrations of PCB detected in the Hudson River. Based on field observations of large mallard populations, there is no discernable population impact on the mallard; however, waterfowl with greater sensitivity to PCBs may be affected based on the TEQ results.

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

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 5-44 through 5-46 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 4-26a. 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 5-44. These values exceed one for all comparisons at the TI Pool. The modeled average and 95% UCL dietary dose exceeds the NOAEL, but only the 95% UCL exceeds the LOAEL at Stillwater. There are no dietary does TQs above one at RM154. For the lower river, the 95% UCL dietary dose exceeds the NOAEL at River Mile 137.2. Comparisons on the basis of predicted egg concentrations all exceed one (to a greater extent than the dietary dose-based comparisons).

Table 5-45 provides comparisons of great blue heron modeled dietary doses to toxicity reference values. The dietary dose-based comparisons for both the average and 95% UCL exceed the NOAEL at the TI Pool. At Stillwater, the NOAEL-based comparison for the 95% UCL dietary dose is one. Most egg concentration-based comparisons exceed one, and in many cases exceed ten at the TI Pool and Stillwater.

Table 5-46 presents the results for the bald eagle based on observed data. Both the average and 95% UCL dietary doses exceed the NOAEL and LOAEL benchmarks at the TI Pool. At Stillwater, the NOAEL is exceeded by the average and 95% UCL dietary dose. The 95% UCL dietary dose exceeds the NOAEL by one at Federal Dam and several Lower River locations. The 95% UCL dietary dose exceeds the LOAEL at River Mile 137.2. The field-based TRV derived for egg concentrations is exceeded by one to two orders of magnitude in the Upper and Lower Hudson River 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 on predicted egg concentration basis using the 1993 data for all river miles. However, reproductive effects toxicity quotients for these receptors in the Lower Hudson River using average and upper confidence limits rarely exceed one based on dietary dose outside of the TI Pool/Stillwater reach, except for the eagle. The risk to heron on a dietary dose basis is lower than of the kingfisher and eagle. This indicates that PCBs from the Hudson River in the diet and water may present a significant risk of reproductive effects at sensitive life stages 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 exceedence of a reproductive-based endpoint.

5.5.1.2 Measurement Endpoint: Modeled Dietary Doses of Total (Tri+) PCBs to Piscivorous Birds (Belted Kingfisher, Great Blue Heron, Bald Eagle) for the Period 1993-2018.

Tables 5-47 through 5-49 compare the estimated total PCB dietary dose of the female belted kingfisher, great blue heron, and bald eagle to the toxicity benchmarks. 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, 2000a) and used in exposure models under future conditions.

Table 5-47 shows the comparison of modeled dietary doses to toxicity reference values for the period 1993 to 2018 for the kingfisher. The average modeled dietary doses exceeds the NOAEL by one at the TI Pool and at Stillwater until 2010 and 1996, respectively. All other comparisons do not exceed one for the entire modeling period.

Table 5-48 presents the results for the great blue heron. This table shows that estimated toxicity quotients exceed one in the TI Pool until 1996 on a NOAEL basis. All other comparisons do not exceed one for the entire modeling period.

Table 5-49 presents the results for the bald eagle. Average modeled dietary doses exceed the NOAEL by one at the TI Pool until 2000. In the lower river, the NOAEL is exceeded at RM 152 in 1993 and 1998 and at RM 113 in 1998. All other comparisons do not exceed one for the entire modeling period.

Reproductive effects toxicity quotients for great blue heron, belted kingfisher, and bald eagle all exceed one at the TI Pool during the early to mid phase of the modeling period. However, dietary dose exceedences are not by orders of magnitude

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

Tables 5-50 through 5-52 compare the estimated total PCB (*i.e.*, Tri+) predicted egg concentrations for the belted kingfisher, great blue heron, and bald eagle to the toxicity benchmarks presented in Table 4-26a 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, 2000a).

Table 5-50 presents the results for the belted kingfisher. All comparisons at all upper and lower river locations exceed one for the duration of the modeling period. Early in the modeling period, predicted toxicity quotients exceed 10 in the TI Pool and at Stillwater on a NOAEL basis, but decrease with time and distance from the TI Pool.

Table 5-51 presents the results for the great blue heron. All NOAEL-based comparisons in the upper and lower river exceed one for the duration of the modeling period, with the exception of from 2011 on at RM 154. Many LOAEL-based comparisons also exceeded one.

Table 5-52 presents the results for the bald eagle. All NOAEL and LOAEL comparisons at all locations in the upper and lower river exceed one for the duration of the modeling period. The highest toxicity quotients are seen in the Thompson Island Pool.

All of the predicted toxicity quotients for the eggs of the belted kingfisher and eagle exceed one on the basis of estimated egg concentrations. This is also the case for most of the great blue heron NOAEL comparisons. These results suggest that exposure of piscivorous birds to PCBs from the Hudson River may result in adverse reproductive effects, particularly in areas around the Thompson Island Pool. 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 5-53 through 5-55 compare the estimated predicted TEQ-based dietary dose and egg concentrations for the piscivorous bird species to the toxicity benchmarks presented in Table 4-26a. 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 biomagnification factors presented in Chapter 3. Biomagnification factors are applied to observed forage or piscivorous fish concentrations to obtain predicted egg concentrations.

Table 5-53 presents the results for the belted kingfisher. This table shows that all comparisons exceed one, except for the LOAEL on an average egg concentration-basis for RMs 100 and 25.8. The NOAEL-based comparison for the 95% UCL dietary dose exceeds one by a considerable margin for all river miles. The NOAEL based comparison for the 95% UCL for the predicted dietary dose in the TI Pool approaches 2500. All toxicity quotients estimated on the basis of egg concentrations exceed 10 in the TI Pool on both an average and 95% UCL basis.

Table 5-54 presents the results for the great blue heron. All comparisons exceed one, with most toxicity quotients two orders of magnitude greater than one in the TI Pool.

Table 5-55 presents the results for the bald eagle. Predicted toxicity quotients exceed one for all comparisons at all locations, and many are above 1000. For the TI Pool, all of the predicted toxicity quotients are above 100, and some are above 1000. For the predicted egg concentrations, the 95% UCL NOAEL based comparisons exceed 100 for all locations, and in some cases exceed 1000.

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 may result in adverse reproductive effects. All toxicity quotients exceed one, in many cases by several 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 5-56 through 5-58 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. Model results were multiplied by the weighted TEF derived in Chapter 3.

All upper and lower river locations and all comparisons exceed one for the belted kingfisher and bald eagle on a dietary dose basis for the entire modeling period (Tables 5-56 and 5-58). For the great blue heron, all comparisons exceed one on a NOAEL-basis for the entire modeling period at all locations (Table 5-57). On a LOAEL-basis, some values fall below one at Federal Dam and locations in the lower river for a portion of the modeling period (Table 5-57).

Reproductive effects toxicity quotients for great blue heron, belted kingfisher, and bald eagle on a TEQ basis all exceed one, and in many cases exceed 100. The predicted risk is slightly less for the heron, than for the kingfisher and eagle. These results indicate that PCBs from the Hudson River in the diet and water may 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 and suggest it is reasonable to expect adverse population-level effects, given the consistent exceedence of a reproductive-based endpoint.

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

Tables 5-59 through 5-61 present the results of the comparison between egg concentrations expressed on a TEQ basis to 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, 2000a) 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).

Table 5-59 presents the results for the belted kingfisher. This table shows that all comparisons exceed one at the TI Pool. Comparisons also exceed one on a NOAEL for the entire modeling period at Stillwater (RM 168), RM 152, and RM 113. The remainder of the comparisons exceed one during a portion of the modeling period.

Table 5-60 presents the results for the great blue heron. All comparisons exceed one for all river miles over the entire modeling period, with the exception of the LOAEL at RM 50 in 2015 and 2018.

Table 5-61 presents the results for the bald eagle. All comparisons exceed one for all river miles over the entire modeling period, except for the LOAEL-based comparisons after 2013 at RM 154, after 2000 at RM 90, and after 1999 at RM 50.

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 in many cases. In the TI Pool, predicted toxicity quotients exceed 10 or even 100, and some of the bald eagle toxicity quotients exceed 1000. This indicates that PCBs from the Hudson River in fish as they translate to egg concentrations are likely to result in adverse reproductive effects to these species on the basis of modeled TEQ-based PCB egg concentrations when compared to appropriate toxicity reference values. These results suggest it is reasonable to expect adverse population-level effects, given the consistent exceedence 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

All observed whole water concentrations exceeded the USEPA/NYSDEC wildlife bioaccumulation value at all locations (Table 5-8). This value is exceeded at all upper and lower river locations for the duration of the modeling period (Table 5-9).

Comparison of total PCB concentrations in the water column to guidelines indicates that the level of PCBs present in the Upper and Lower Hudson River may cause adverse effects to wildlife feeding on aquatic life. Comparisons may underestimate 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).

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

5.5.3.1 Measurement Endpoint: Observational Studies

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. However, determining a direct causative linkage between contamination and population changes is not realistic, given the many changes that have occurred along the Hudson River in the last 50 years and other populations pressures (*e.g.*, prey and habitat availability)

Both the New York State Endangered Species Unit and The Atlas of Breeding Birds in New York (Andrle and Carroll, 1988) provide general information regarding the bird species using the Hudson River. Some belted kingfishers may be remain in the Hudson River area year-round, as evidenced by sightings in the Christmas bird count (Cornell University, 1999; 2000). The belted kingfisher (*Ceryle alcyon*) appears to breed along the Hudson River north of Westchester County in areas such as Oscawana and George's Island Parks (Bickford, 1999).

The great blue heron (*Ardea herodias*) is found along the Hudson River and has been observed in most count circles during the Christmas bird count (Cornell University, 1999; 2000). There is one breeding colony of herons in the upper river and another in the freshwater portion of the Lower Hudson River (Rensselaer County). The presence of two breeding colonies of great blue herons does not indicate that they are breeding throughout the Hudson River area. NYSDEC and USFWS are currently analyzing concentrations of PCBs and other organochlorines in great blue heron nestlings, eggs, and prey. Initial results (Table 3-20) show the presence of PCBs in the brains of nestlings. The concentration of the individual (1 ppm) collected from Saratoga National Historic Park in the upper river had PCB concentrations five times greater than the average of the individuals collected from Castleton Island in the lower river (0.19 ppm).

Peter Nye (NYSDEC) studies the bald eagle in New York State. The bald eagle is a winter resident of both the upper and lower river. Up to 40 eagles have wintered in the 30 miles between Danskammer Point (Orange County) and Croton Point (Westchester County) in the last few years (USGS, 1999) and eagles have recently started breeding at several locations in the lower river. After five years of unsuccessful attempts, a bald eagle was fledged in the Hudson Valley in 1997 for the first time in about 100 years. The succeeding three years have produced four, five, and this year ten new eagles from four separate nests (NYSDEC, 2000b). All eagles now breeding in NYS are the result of NYSDEC or other direct release/restoration programs (Nye, 2000b). Bald eagle data collected by NYSDEC indicate a stable population.

However, the New York State population (consisting of 200-250 individuals) is small enough to be affected by natural or manmade disturbances. It is encouraging to see some successful nesting, but it is too early to call the Hudson River population re-established. In addition, all breeding individuals nest in the lower river. The reproductive effects on bald eagles that winter along the upper river is unknown.

NYSDEC and USFWS have been collecting eagle serum, prey and unhatched eggs for several years to evaluate contaminant loads throughout the eagles ecosystem (Nye, 2000b). Preliminary PCB results from two samples indicated levels that are high enough to be of concern (1,329 μg/kg in eagle plasma from a nestling eagle and 85,770 μg/kg in fat sample from an immature eagle found dead). Recent data on PCB concentrations in bald eagle blood (Table 3-20a) show PCBs in the blood of all bald eagles samples, with concentrations as high as 14, 240 ng/g. These eagles were all sampled in the lower river, bald eagles wintering in the upper river may have even higher concentrations of PCBs. More data from USFWS and NYSDEC and are expected to be available in late 2000/early 2001 (Secord, 2000).

It should be noted that the bald eagle is on the federal and NY State list of threatened and endangered species. Therefore, individual level effects could adversely affect the Hudson River populations.

Summary

Toxicity quotients of the three piscivorous birds modeled (belted kingfisher, great blue heron, and bald eagle) are generally well above one for the duration of the modeling period in the Upper and Lower Hudson River. Measured and modeled water concentrations of PCBs are high enough to be of concern to piscivorous wildlife for the duration of the modeling period (through 2018) at all locations in the upper and lower river. The three receptors modeled for piscivorous birds have all been observed along the river, and bald eagles have begun to breed successfully in the lower river over the last few years. Nonetheless, the concentrations of PCBs detected in piscivorous birds from the Hudson River are high enough to be of concern. These data, in combination with the food chain and water concentration modeling, indicate that piscivorous birds along the Hudson River may be experience adverse reproductive effects from exposure to PCBs.

5.6 Evaluation of Assessment Endpoint: Sustainability (*i.e.*, Survival and Reproduction) of Local Insectivorous Mammals (as Represented by the Little Brown Bat)

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

5.6.1.1 Measurement Endpoint: Modeled Dietary Doses of Tri+ to Insectivorous Mammalian Receptors (Little Brown Bat) using 1993 Data

Modeled total PCB dietary dose comparisons to toxicity reference values are presented for the little brown bat in Table 5-62 using 1993 data under current conditions. These results show that LOAEL- and NOAEL-based toxicity quotients exceed one in the Upper Hudson River. The LOAEL toxicity quotients for the average dietary dose are below one in the Lower Hudson River for the 1993 USEPA dataset, although LOAEL toxicity quotients for the 95% UCL exceeds one at River Miles 137.2, 58.7 and 47.3. All NOAEL toxicity quotients for the 95% UCL exceed one in the Lower Hudson River, except for RMs 88.9 and 25.8. For the average dietary does in the lower river, the NOAEL toxicity quotients exceed one at River Mile 137.2.

These results suggest the potential for adverse reproductive effects to insectivorous mammalian species at Upper Hudson River locations based on using 1993 data in the exposure models. However, the risk of impaired reproduction is lower at lower river locations. Given the consistency of the results and the magnitude of the exceedences for the upper river locations, these results suggest the potential for population-level adverse reproductive effects in those areas.

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 5-63 for the period 1993 – 2018 under future conditions. NOAEL-based comparisons exceed or equal one for the TI Pool and Stillwater during the entire modeling period. LOAEL-based comparisons exceed one for the TI Pool and Stillwater until 2002 and 2004, respectively. The LOAEL-based comparisons for Federal Dam exceed one until 1994, whereas the NOAEL-based comparisons for this location exceed one until 2012.

In the lower river the NOAEL is exceeded for a portion of the modeling period at all locations. LOAEL-based comparisons do not exceed one at any location in the lower river.

These results suggest a low potential for adverse reproductive effects to insectivorous mammalian species along the upper and lower river, with populations at the TI Pool and Stillwater experiencing the greatest risk.

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

Modeled TEQ-based dietary dose comparisons to toxicity reference values are presented for the little brown bat in Table 5-64 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 concentrations are adjusted by the weighted TEF presented in Chapter 3.

These results are similar to those predicted on a Tri+ PCB basis, but predicted toxicity quotients based on TEF are higher than those predicted on a Tri+ PCB basis. Predicted toxicity quotients in the TI Pool and at Stillwater exceed 10 across all LOAEL comparisons. For the lower river, the 95% UCL but not the average based LOAEL comparisons exceed one consistently, except for River Miles 88.9 and 25.8.

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 exceedences for the Hudson River, these results suggest the potential for population-level adverse reproductive effects in 1993. The predicted TQs for insectivorous mammals in the Lower Hudson River are lower than those of the upper river.

5.6.1.4 Measurement Endpoint: Modeled Dietary Doses on a TEQ Basis to Insectivorous Mammalian Receptors (Little Brown Bat) for the Period 1993 - 2018

Modeled TEQ-based dietary dose comparisons to toxicity reference values are presented for the little brown bat in Table 5-65 for the period 1993 – 2018 under future conditions. These results show that all comparisons exceed one for all upper and lower locations except the LOAEL based comparisons at RM 154 starting in 2011. All NOAEL-based comparisons exceed one for all locations during the entire modeling period.

These results suggest the potential for adverse reproductive effects to insectivorous mammalian species feeding on emergent insects from the Hudson River, particularly to populations found near the TI Pool or Stillwater. However, based on the differences seen between the total PCB and TEQ-based results, the magnitude of population-level adverse reproductive effects is uncertain.

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

All observed whole water concentrations exceeded the USEPA/NYSDEC wildlife bioaccumulation value at all locations (Table 5-8). This value is exceeded at all upper and lower river locations for the duration of the modeling period (Table 5-9).

Comparison of total PCB concentrations in the water column to guidelines indicates that the level of PCBs present in the Upper and Lower Hudson River may cause adverse effects to wildlife feeding on aquatic life. Comparisons may underestimate 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).

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

5.6.3.1 Measurement Endpoint: Observational Studies

A limited amount of data is available on little brown bat populations in the Upper and Lower Hudson River, and only a small subset of that data is within a time frame relevant to this study. Little brown bats populations are found along river, but the limited number of field-based observations available do not provide enough information to evaluate this measurement endpoint.

Summary

Based on the results of the food chain and water column modeling, little brown bats may be experience adverse reproductive effects from exposure to PCBs, particularly populations in the Thompson Island Pool/Stillwater area.

5.7 Evaluation of Assessment Endpoint: Sustainability (*i.e.*, Survival and Reproduction) of Local Omnivorous Mammals (as Represented by the Raccoon)

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

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

Modeled total PCB dietary dose comparisons to toxicity reference values are presented for the raccoon in Table 5-66 using 1993 data under current conditions. In the Upper Hudson River, all NOAEL-based comparisons exceed or equal one. On LOAEL-basis, both the 95% UCL and average comparisons exceed or equal one at Stillwater. At the TI Pool, the LOAEL-based comparison for the 95% UCL, but not the average, exceeds one. On a LOAEL-basis, neither the 95% UCL or the average toxicity quotient exceeds one at Federal Dam. For the LOAEL-based comparisons, both the 95% UCL and average do not exceed one at any of the Lower Hudson River location. At Lower River Miles 137.2, 58.7, and 47.3, the NOAEL-based comparison for the 95% UCL equals or exceeds one.

These results suggest a low 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. The potential for adverse reproductive effects is greater in the upper reaches of the river.

5.7.1.2 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 5-67 for the period 1993 – 2018 under future conditions. Dietary doses are estimated by using predicted water concentrations from the HUDTOX (upper river) and Farley (lower river) models and predicted forage fish and benthic invertebrate concentrations from the FISHRAND model.

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For the TI Pool and Stillwater, the NOAEL toxicity quotients equal or exceed one until 2003 and 2002, respectively. Predicted LOAEL toxicity quotients for all upper river locations are below one. The NOAEL toxicity quotients for the Federal Dam location and all lower river locations are below one.

5.7.1.3 Measurement Endpoint: Modeled Dietary Doses on a TEQ Basis to Omnivorous Mammalian Receptors (Raccoon) using 1993 Data

Modeled TEQ-based dietary dose comparisons to toxicity reference values are presented for the raccoon in Table 5-68 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 for the Upper Hudson River locations consistently exceed one, and in some cases exceed 10. For the Lower Hudson River, all NOAEL toxicity quotients exceed one. The LOAEL toxicity quotients for the 95% UCL for River Miles 137.2, 113.8, 100, and 58.7, and 47.3 exceed one. No other comparisons exceed one for the lower river locations.

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.

5.7.1.4 Measurement Endpoint: Modeled Dietary Doses on a TEQ Basis to Omnivorous Mammalian Receptors (Raccoon) for the Period 1993 - 2018

Modeled TEQ-based comparisons to toxicity reference values are presented for the raccoon in Table 5-69 for the period 1993 – 2018 under future conditions. Predicted NOAEL toxicity quotients at all upper and lower river locations exceed one for the duration of the modeling period. All LOAEL comparisons exceed one at the TI Pool location for the duration of the modeling period. On a LOAEL basis, predicted toxicity quotients at Stillwater and Federal Dam exceed one until 2011 and 1999, respectively. There are some LOAEL exceedances in the lower river in the early modeling years.

These results suggest the potential for adverse reproductive effects to omnivorous mammalian species in the upper river, particularly in the TI Pool. The potential for risk to omnivorous mammals from Federal Dam downriver is much lower than Upper River locations.

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

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

All observed whole water concentrations exceeded the USEPA/NYSDEC wildlife bioaccumulation value at all locations (Table 5-8). This value is exceeded at all upper and lower river locations for the duration of the modeling period (Table 5-9).

Comparison of total PCB concentrations in the water column to guidelines indicates that the level of PCBs present in the Upper and Lower Hudson River may cause adverse effects to wildlife feeding on aquatic life. Comparisons may underestimate 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).

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

5.7.3.1 Measurement Endpoint: Observational Studies

A survey of wildlife biologists and trappers working along the Hudson River (see USEPA, 1999c), indicated that Hudson River raccoon populations appear to be thriving. Raccoons are commonly seen by residents in the upper and lower river. As discussed in Appendix F in USEPA 1999c, the raccoon is an opportunistic feeder. Their tendency to eat easily available food, including garbage, may make them less dependent on Hudson River food sources than other species.

Summary

Based on the results of the food chain and water column modeling, raccoons may be experience adverse reproductive effects from exposure to PCBs, particularly populations in the Thompson Island Pool/Stillwater area. However, field observations indicate that populations of raccoons are abundant along the Hudson River, suggesting that only a small number of individuals living close to the river are consuming the estimated river-related diet sources (40% fish and invertebrates).

5.8 Evaluation of Assessment Endpoint: Sustainability (*i.e.*, Survival and Reproduction) of Local Piscivorous Mammals (as Represented by the Mink and River Otter)

5.8.1 Measurement Endpoint: Measured Total PCB Concentrations in the Liver of Piscivorous Mammalian Receptors (Mink, River Otter)

Table 5-70 presents the results of a comparison between total PCB concentrations in mink and otter liver tissue (NYS Toxic Substances Monitoring Program, 1987) to concentrations at which impaired reproduction (Platnow and Karsted, 1973) and growth (Wren *et al.*, 1987) had been observed. The mink and otter 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 adverse effects. Mink concentrating upper range concentrations of PCBs and river otter concentrating average to upper range concentrations are likely to experience adverse reproductive or growth effects.

5.8.1.1 Measurement Endpoint: Modeled Dietary Doses of Tri+ to Piscivorous Mammalian Receptors (Mink, River Otter) using 1993 Data

Tables 5-71 and 5-72 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 5-71 shows the results for the mink. On a dietary dose basis for total (Tri+) PCBs, predicted toxicity quotients exceed one on a NOAEL and LOAEL basis for both the average and 95% UCL at all locations in the upper and lower river. For the LOAEL based comparisons at the TI Pool and Stillwater, predicted toxicity quotients exceed ten. On a NOAEL-basis, toxicity quotients for these areas exceed 100 and exceed 10 for the rest of the river.

Table 5-72 shows the results for the river 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 with several greater than 1,000. The otter's diet is composed almost entirely of fish. It consumes a larger size range of fish than the mink and is likely to obtain fish from deeper in the river. Thus, the exposure of the 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 exceedences, these results suggest the potential for adverse reproductive effects on mink and river otter consuming fish from the Hudson River.

5.8.1.2 Measurement Endpoint: Modeled Dietary Doses of Tri+ to Piscivorous Mammalian Receptors (Mink, River Otter) for the Period 1993 - 2018

Tables 5-73 and 5-74 present the results of the comparison between modeled dietary doses to mink and river otter under future conditions for the period 1993 - 2018 for total (Tri+) PCBs. NOAEL-based results exceed one for all upper and lower locations for the duration of the modeling period. On a dietary dose basis for total (Tri+) PCBs, predicted toxicity quotients for the mink exceed one for the duration of the modeling period for the LOAEL based comparisons in the TI Pool, Stillwater, and RM 152 (Table 5-73). At Federal Dam, the LOAEL based comparison equal or exceeds one until 2011 and for a portion of the modeling period at the remaining lower river stations. The disparity between the toxicity quotients at Federal Dam (RM154) and RM 152 is due to the use of the HUDTOX model for sediment and water concentrations in the upper river and the use of the Farley models for these concentrations in the lower river.

Table 5-74 shows the results for the river otter. On a dietary dose basis for total (Tri+) PCBs, predicted toxicity quotients exceed one at all upper and lower locations across all comparisons. On a NOAEL basis, the predicted toxicity quotients for the river otter exceed 200 at the TI Pool for the duration of the modeling period.

These results suggest the potential for adverse reproductive effects to piscivorous mammalian species in the Hudson River, in particular to the otter. Given the consistency of the results, the magnitude of the exceedences, and the duration of the exceedences, these results suggest the potential for adverse reproductive effects on mink and otter consuming fish from the Hudson River.

Reproductive effects toxicity quotients for the mink and otter using averages 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.

5.8.1.3 Measurement Endpoint: Modeled Dietary Doses on a TEQ Basis to Piscivorous Mammalian Receptors (Mink, River Otter) using 1993 Data

Tables 5-75 and 5-76 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 5-75 shows the results for the mink. On a TEQ basis, predicted toxicity quotients exceed one for all upper and lower river locations across all comparisons, with the exception of the LOAEL at RM 100. 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 with two exceedances greater than 1,000.

Predicted toxicity quotients for the river otter exceed ten at all upper and lower locations across all comparisons (Table 5-76). On a NOAEL basis, the predicted toxicity quotients for the otter for both the average and 95% UCL exceed 100 at all locations with many above 1,000 and one above 10,000. The river otter consumes a greater percentage of fish in the diet and larger size range of fish than the mink and therefore its exposure 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 exceedences, these results suggest the potential for adverse reproductive effects on mink and otter consuming fish from the Hudson River.

5.8.1.4 Measurement Endpoint: Modeled Dietary Doses on a TEQ Basis to Piscivorous Mammalian Receptors (Mink, River Otter) for the Period 1993 - 2018

Tables 5-77 and 5-78 present the results of the comparison between modeled TEQ-based dietary doses to mink and otter under future conditions for the period 1993 - 2018 for total (Tri+) PCBs.

Table 5-77 shows the results for the mink. On a TEQ dietary dose basis, predicted toxicity quotients exceed one at all upper and lower river locations for the duration of the modeling period for both the LOAEL and NOAEL based comparisons, with the exception of the LOAEL based comparison at Federal Dam after 2006. The disparity between the toxicity quotients at Federal Dam (RM154) and RM 152 is due to the use of the HUDTOX model for sediment and water concentrations in the upper river and the use of the Farley models for these concentrations in the lower river.

The predicted toxicity quotients exceed one at all locations across all comparisons for the river otter (Table 5-78). In early modeling years, NOAEL toxicity exceedences are greater than 100, and in some cases greater than 1,000.

These results suggest the potential for adverse reproductive effects to piscivorous mammalian species in the Hudson River based on using HUDTOX (upper river), Farley (lower river), and FISHRAND model results in the exposure models. Given the consistency of the total PCB and TEQ-based toxicity quotient results, the magnitude of the exceedences, and the duration of the exceedences, these results suggest the potential for population-level adverse reproductive effects for mink and otter consuming fish from the Hudson River.

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

5.8.2.1 Measurement Endpoint: Comparison of Measured and Modeled Water Concentrations to Criteria and/or Guidelines for the Protection of Wildlife

All observed whole water concentrations exceeded the USEPA/NYSDEC wildlife bioaccumulation value at all locations (Table 5-8). This value is exceeded at all upper and lower river locations for the duration of the modeling period (Table 5-9).

Comparison of total PCB concentrations in the water column to guidelines indicates that the level of PCBs present in the Upper and Lower Hudson River may cause adverse effects to wildlife feeding on aquatic life. Comparisons may underestimate 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).

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

5.8.3.1 Measurement Endpoint: Observational Studies

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

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

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

NYSDEC has also been interviewing trappers to pinpoint the take (abundance), habitat, and habitat quality of where the piscivorous mammals are found (Mayack, 2000a). Many of the trappers interviewed have been working along the Hudson River for 30 to 40 years. Over the last season (September to October), a trapper has been employed by NYSDEC to trap furbearing mammals. The trapper worked from Corinth to Stillwater and found nine mink within 0.25 miles of the river or right on the river (Mayack, 2000b). There were several otter at Fish Creek. The average number of animals trap per trap night on the river was three, versus 26 per trap night off the river. Preliminary results indicate that the animals are relatively PCB free away from the river (less than six km) (Mayack, 2000b).

NYSDEC is continuing to compile tissue from mink, otter, and muskrat for total PCBs, some PCB congener-specific analyses, Aroclors, and pesticides (not all analyses are being performed on every sample) from the upper river (Mayack, 2000a). These data are anticipated to be available in 2001.

5.9 Results of the Probabilistic Dose-Response Analysis

The potential for population-level effects is evaluated by comparing dose-response curves from the literature (Moore *et al.*, 1999 as presented in Section 4.3) and cumulative distributions of exposure developed in Section 3.8. Analyses are presented for belted kingfisher, bald eagle, mink and river otter. Results are presented for 1993 and 2015, representing a range of exposures (1993 represents the maximum exposure while 2015 represents exposures representative towards the end of the modeling period).

To compare the cumulative distributions developed in Chapter 3 with the dose response curves from the literature, the following procedure was used. First, the Monte Carlo exposure models were used to generate the cumulative frequency of predicted dietary doses for each receptor. Output concentrations were log-transformed, and the associated cumulative frequencies, expressed as fractions, were transformed by the inverse of the normal cumulative distribution. The log-transformed Monte Carlo concentrations and the transformed cumulative frequencies yield straight lines when plotted against each other. The parameters of those regressions (one for each river mile-year-species combination) were used to obtain the cumulative frequency for the specified doses in the dose-response curves from the literature. The resulting curves can then be compared directly by plotting the probability of exceedence on the y-axis (see Figures 3-4 and 3-5) and the percent reduction in fecundity on the x-axis (see Figure 4-2).

5.9.1 Belted Kingfisher

Figure 5-9 shows that in 1993, female kingfishers at RM 189 show approximately a 65% probability of experiencing at least a 50% reduction in fecundity. Females at RM 168 show approximately a 45% probability of experiencing at least a 20% reduction in fecundity, while females at RM 154 show approximately a 30% probability of experiencing at least a 10% reduction in fecundity.

In 2015, female kingfishers at RM 189 show approximately an 80% probability of experiencing at least a 10% decrease in fecundity (Figure 5-9). Females at RM 168 and 154 have low probabilities (<10%) of experiencing small reductions (<5%) in fecundity.

5.9.2 Bald Eagle

In 1993, female eagles at RM 189 show approximately a 45% probability of experiencing at least a 50% reduction in fecundity (Fig 5-9). Female eagles at RM 168 show approximately a 30% probability of experiencing a 20% reduction in fecundity, while at RM 154, females show approximately a 15% probability of experiencing at least a 10% reduction in fecundity.

In the year 2015, female eagles at RM 189 show approximately a 45% probability of experiencing at least a 10% reduction in fecundity (Fig 5-9). Females at RM 168 and 154 have low probabilities (<10%) of experiencing small reductions (<5%) in fecundity.

5.9.3 Mink

In 1993, female mink at RM 189 and 168 show a high probability (90 to 100%) of experiencing a severe reduction (>80 %) in fecundity (Fig 5-10), and females at RM 154 still show a high probability (>95%) of experiencing at least a 50% reduction in fecundity.

In the year 2015, mink at RM 189 still show a high probability (>95%) of experiencing substantially reduced (>50%) fecundity (Fig 5-10). However, mink at RM 168 show a lower probability (35%) of experiencing at least a 40% reduction, while mink at RM 154 show only a 10% probability of experiencing at least a 20% reduction in fecundity.

5.9.4 River Otter

In 1993, female river otters at RM 189, 168 and 154 show high probabilities (80 to 100%) of experiencing severe decreases (>90%) in fecundity (Fig 5-10), in comparison to otters that are not exposed to PCBs.

In the year 2015, female otters at RM 189 still show high probabilities (>70%) of experiencing severely reduced (100%) fecundity (Fig 5-10). Otters at RM 168 still show high probabilities (>80%) of experiencing a substantial decrease (>80%) in 2015, while otters at RM 154 show a 30% probability of experiencing at least a 50% reduction in fecundity.

Summary

The probabilistic dose-response analysis shows that piscivorous birds have a high likelihood of experiencing a moderate reduction in fecundity around the Thompson Island Pool (RM 189) and a low likelihood of a small decrease in fecundity near Federal Dam (RM 154). Piscivorous mammals have a high likelihood of experiencing a significant reduction in fecundity around the Thompson Island Pool (RM 189) and a moderate likelihood of a moderate decrease in fecundity near Federal Dam (RM 154).

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 Revised 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 in the following sections.

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 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 it was used since the ERA focuses on the exposure of the ecological community to PCBs.

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 small scale variation, 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 late summer/early fall (August) (NOAA, 1997a). Although the differences were not large, potential risks to fish and upper-trophic level receptors may be slightly underestimated because the Phase 2 ecological samples were collected in August 1993.

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.

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 gas chromatograph 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 of USEPA, 1999c) 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 of congeners were rejected for dual 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, 2000a).

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 (1997a) 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, 2000a Section 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.

6.2.1 TEQ Quantitation

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. To evaluate the effect of using BZ#126 at the detection level, the ratio of the TEF with BZ#126 at the detection level was compared to the value of the TEF with BZ#126 set to 0 (following the rule that any individual congener detected in less than 15% of samples was set to 0, otherwise, set to 1/2 the detection limit). The values of the TEF are found in Table 3-2. For fish concentrations, the ratio across fish, mammals and avian receptors was never more than a factor of five, and typically less than a factor of 2. In other words, the difference in the predicted TEQ toxicity quotient for the avian and mammalian receptors that feed primarily on fish is approximately a factor of two based on using BZ#126 at the detection level.

There is evidence in the literature that BZ#126 becomes enriched in biological tissues over time. Thus, it is not inappropriate to use BZ#126 at the detection level rather than set to zero or half the detection limit.

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, surface waters, and prey. However, since it is a generalized model, it is not intended to represent specific individuals currently living along the Hudson River. The actual linkages between the biotic levels often depend on seasonal availability of various prey and food items. However, the results of the risk characterization show that the majority of risk is due to exposure to contaminated prey, which is consistent with other studies. Specific uncertainties in the exposure and food web modeling are discussed in Section 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 may exist for terrestrial species , such as shrews and moles (see Table 2-6), exposed to PCBs originating in the Hudson River.

6.4 Toxicological Uncertainties

Hundreds of studies on the toxic effects of PCBs on animals were reviewed and summarized for the effects assessment chapter. The results of these studies were summarized, organized and presented in Tables 4-4 through 4-22. These studies represent a wide variety of laboratory and field-based studies that examine a variety of test species, doses, exposures, instruments, and analytical methods. For each Hudson River representative receptor, individual studies were chosen to represent the expected toxic effect of PCBs on that receptor. Each of the selected studies has limitations and varying degrees of uncertainties associated with the design and results of the study and it's applicability to the Hudson River receptors. The goal of this section is to discuss the uncertainties generally associated with the effects assessment, and the uncertainties associated with the development of individual toxicity reference values (TRVs).

One of the most important sources of uncertainty in the effects assessment is the difference among species in sensitivity to PCBs. Certain taxonomic groups of animals, such as salmonid fish, gallinaceous birds, and mink have been shown to be highly sensitive to the reproductive effects of PCBs (Beyer *et al.*, 1996). Other groups, such as the tree swallow, have been shown to be less sensitive (McCarty and Secord, 1999a). To minimize this source of uncertainty in the ERA, the effects assessment selects appropriate studies that were conducted on species that are the same as, or closely related to (within the same taxonomic family), the Hudson River representative receptors.

If an appropriate study on a closely related species is not available, the final TRV is developed from a study on a less closely-related species. In these cases, the final TRV could over- or underestimate the sensitivity of a Hudson River receptor. Some information is available on the potential magnitude of the uncertainty associated with these interspecies extrapolations. For example, studies on mammals have shown that the ratio of sensitivity of the least to the most sensitive species, on the basis of dietary dose of toxicant, ranges from 1.9 to 100 (Hayes, 1967). An interspecies uncertainty factor of ten has been proposed to account for this interspecies variability in toxicity (Dourson and Stara, 1983). A similar study on interspecies variability in birds found that the most sensitive individuals are within a factor of four of the median sensitivity for 75% of the chemicals tested, and 95% are within a factor of 10 (Hill *et al.*, 1975 and USEPA 1995a). A similar comparative study is not available for fish, however the range of lowest to highest LOAELs for effects of total PCBs and Aroclors on fry mortality is about 100 (Table 4-5), and the range of lowest to highest LOAELs for effects of dioxin-like compounds on early life stage mortality is 125 (Table 4-7). Uncertainty associated with the development of the

TRV for the effect of total PCBs on fish may be greater than for other taxonomic groups since fewer studies are available for fish (Tables 4-5 through 4-8) than for birds (Tables 4-9 through 4-16) and mammals (Tables 4-18 and 4-19).

To examine the uncertainty associated with interspecies variability in sensitivity to PCBs, an alternative set of TRVs was developed using an interspecies uncertainty factor of 10 for extrapolation between test species and Hudson River receptors that are not in the same taxonomic family. If the test species is known to be particularly sensitive to PCBs (*e.g.*, lake trout), an uncertainty factor is not applied since the likelihood that the final TRV underestimates the sensitivity of the Hudson River receptor is small. If the test species is known to be of intermediate sensitivity, then the alternative TRV is developed using an uncertainty factor of ten, reflecting the range of reported interspecies variability. These alternative TRVs are not selected as the final TRVs, however, since the Hudson River receptor could also be less sensitive than the test species. The alternative TRVs are, however, used to examine uncertainty in the final TRVs, and provide a conservative estimate of the potential that the final TRVs underestimate the sensitivity of Hudson River receptors.

Another important area of uncertainty in the effects assessment is the potential for differences between effects observed in laboratory studies or field studies and those experienced by Hudson River receptors. Both laboratory and field studies have advantages and disadvantages for use in the development of TRVs. Laboratory experiments offer the advantage of being able to control exposure conditions, while field studies may more closely represent actual exposure conditions. For example, the concentrations of PCB congeners in environmental media, especially tissue, are strongly influenced by differential rates of transport, uptake, metabolism, and elimination for these congeners. Congeners that are resistant to metabolism are more persistent and tend to be present at higher concentrations in environmental media than in a commercial mixture, such as Aroclor 1254. Therefore, mixtures of PCB congeners in environmental media (*e.g.*, fish tissue or bird eggs) may be is more toxic than the commercial mixture, and TRVs based on dietary dose of an Aroclor may underestimate the toxicity of the dietary dose received by a receptor in the field. This uncertainty is reduced in the effects assessment by developing TRVs based on total equivalents of dioxin-like PCBs (TEQs). TRVs for TEQs are based on the toxicity of individual congeners in comparison to the toxicity of 2,3,7,8-TCDD. Assessing toxicity based on the toxicity of the individual congeners is less uncertain than assessing toxicity in comparison to the toxicity of an Aroclor mixture.

There is also uncertainty in the manner in which TEQ concentrations are characterized. Some toxicity studies used slightly different TEFs when evaluating TEQ concentrations. When possible, the results that were obtained using TEFs reported in the original study are compared to results that would be calculated using more recent TEFs. This difference was no more than 30% and typically on the order of 13% - 20%.

A controlled laboratory study can be designed to test the effect of a single PCB mixture or congener on a test species in the absence 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. In field studies, organisms are typically exposed to other co-occurring contaminants. The presence of co-occurring contaminants may be a disadvantage to the use of field studies for development of TRVs, since observed effects may not be solely attributable to exposure to PCBs. Some field studies that were identified in the effects assessment were not selected for development of final TRVs because co-occurring contaminants were present at the study site in high concentrations. For example, field studies conducted by Adams *et al.* (1989, 1990, 1992) were not selected as final TRVs because the study site is contaminated with large quantities of mercury. In other cases, field studies are selected for development of final TRVs because observed effects are estimated to be attributable to the effects of PCBs, rather than co-occurring contaminants (Restum *et al.*, 1998).

Additional areas of uncertainty are encountered when the best available study for the development of a final TRV uses a sub-chronic, rather than a chronic, exposure. A conversion factor of 10 is used to estimate a chronic TRV from a sub-chronic TRV. A conversion factor differs from an uncertainty factor in that the direction of the uncertainty is known. For example, the chronic TRV is expected to be lower than the sub-chronic TRV. An interspecies uncertainty factor estimates differences in sensitivity, but the test species could be either more or less sensitive than the receptor of concern. Use of a subchronic-to-chronic conversion factor of ten is supported by the results of a study that compared sub-chronic to chronic NOAELs and LOAELs (Weil and McCollister, 1963; Dourson and Stara, 1983). For more than half of the chemicals studied, the ratio of sub-chronic to chronic endpoints was 2.0 or less, and for 96% of the chemicals the ratios were below 10. A subchronic-to-chronic conversion factor is applied in developing the following final TRVs: dietary doses of TEQs to the mallard, belted kingfisher, great blue heron, and bald eagle (Nosek *et al.*, 1992). However, uncertainty associated with use of a conversion factor for this study (Nosek *et al.*, 1992) is low since the authors independently estimated the time required to reach steady state in comparison to the exposure duration of the experiment. Because TRVs for fish are based on actual body burdens, rather than on dietary doses, subchronic-to-chronic conversion factors are not applied.

Uncertainty also exists when conversion factors are used to estimate NOAELs from LOAELs. Data on the ratio of LOAEL to NOAEL indicates that all chemicals examined have a LOAEL to NOAEL ratio of 10 or less and 96% have a ratio of 5 or less (Weil and McCollister, 1963, and Dourson and Stara, 1983). As previously noted, the uncertainty associated with the use of a LOAEL to NOAEL conversion factor is less than for an interspecies uncertainty factor since the direction of uncertainty is known, and NOAELs are always expected to be lower than LOAELs. LOAEL to NOAEL conversion factors were used only to develop the final TRV for dietary doses of PCBs to the mallard (Haseltine and Prouty, 1980). LOAEL-to-NOAEL conversion factors were not used in the development of any final TRVs for mammals or fish.

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 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. Distributions of important parameters (*e.g.*, PCB concentrations, lipid content, weight, etc.) are incorporated in the FISHRAND model (see Section 6.5.3.1) reducing the uncertainty associated with these parameters. 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 of USEPA (1999c) 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 (>25 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, 1999a) 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, 1999a). The tree swallow was the only avian receptor with Hudson River specific body masses available (USFWS, 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 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). Data for penned mink might not represent ingestion rates for wild mink, which typically expend more energy foraging for food and defending territories. 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 increase their corresponding metabolic rates and their corresponding ingestion needs (Nagy, 1987). Therefore, the application of ingestion rates from penned mink 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

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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). For avian and mammalian receptors diets were developed for the breeding adult females, considered to be most sensitive to reproductive PCB effects.

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 judgment 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 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 applied in the exposure assessment for dietary pathway for this receptor.

Mallard diet information for Hudson River or NYS populations in regional proximity was 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 50% 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 bald eagle specialist Peter Nye (Nye, 1999b; 2000) 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, although some individuals may have lower fish consumption rates.

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 fillets, rather than whole fish, the incidental sediment ingestion rate of piscivorous receptors may be underestimated.

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Stomach content and scat analyses on mink and feeding behavior of otter described by Liers (1951) suggest either the presence of a 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.

Exposure Duration

The 25-year modeling duration period used for ecological modeling covers the average lifespan of receptor species. A few individuals of some species (*e.g.*, striped bass, bald eagle may live longer than 25years, but these individuals are not considered typical of the population. Life span information is presented below.

Fish: largemouth bass - up to 15 years (Smith, 1985); pumpkinseed sunfish – 8 to 10 years (in Canadian populations) (Scott and Crossman, 1975); brown bullhead - 6 to 7 years (Smith, 1985); yellow perch - 9 years (Smith, 1985); white perch - 5 to 7 years; some live 14 to 17 years (Smith, 1985); spottail shiner - 4 years (Pflieger, 1997); striped bass - Smith (1988) reports the oldest fish studied at 14 to 18 years and Cooper (1983) reports a single female estimated to be 30 years.

Birds: tree swallow –maximum 6 years (Long Point Bird Observatory, 2000); average mallard – 1-2 years (Palmer, 1976); belted kingfisher – maximum 16 years (note: no species-specific information was available so the oldest nonpasserine land bird was used [red-cockaded woodpecker]; Klimkiewicz, 1997); great blue heron – maximum 23 years (Klimkiewicz, 1997); and bald eagle – average 15-20 years; may live up to 30 years (Nebraska Games and Parks Commission, 2000).

Mammals: little brown bat - 20 to 30 years (University of New Hampshire, 2000); raccoon - 5 to 6 years (Georgia Wildlife Web, 2000); mink - up to 10 years (Walker, 1997); and river otter - 10 to 15 years (Ohio Division of Wildlife, 2000).

Based on these lifespans, the modeling period used is consider to cover lifetime exposure of receptors.

6.5.2 Sensitivity and Uncertainty Analysis for Risk Models

Because of the quantity of receptors, modeling locations, and number of years modeled, the quantitative uncertainty and sensitivity analyses were only run for four species for two years at Upper Hudson River locations (*i.e.,* RMs 189, 168, and 154). The four species analyzed are the belted kingfisher, bald eagle, mink, and river otter. 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.

The toxicity quotients of kingfisher and kingfisher egg, eagle and eagle egg, mink, and otter were evaluated. All were evaluated on a LOAEL basis and the kingfisher egg NOAEL was also evaluated. Distributions were described as triangular and were based on the ranges for exposure parameters including piscivorous fish concentration, forage fish concentration, benthic invertebrate concentration, body weight, prey ingestion rate, percent of the diet consisting of fish, percent of the diet consisting of invertebrates, water intake rate, and biomagnification factors for eggs (see Chapter 3; Table 3-103). Environmental concentrations were described as lognormal by a mean and standard deviation. Only the toxicity reference value for the kingfisher egg contained an uncertainty factor (factor of ten; see discussion above) which was described as uniform. There were no uncertainty factors recommended for any of the other TRVs (see chapter 4), thus, all other TRVs were set at the calculated point estimate.

Output distributions for the belted kingfisher and egg (Figure 6-1), bald eagle and egg (Figure 6-2), and mink and otter (Figure 6-3) represent the cumulative distribution of predicted toxicity quotients (on a LOAEL basis except for the kingfisher egg, which also includes a NOAEL comparison). These figures show that even at the 5th percentile, predicted toxicity quotients for the eagle egg, kingfisher egg, mink and otter do not fall below one for any location or year, except for mink at RM 154 in 2015. The toxicity quotient results are also shown for selected percentiles in Table 6-1.

Tables 6-2 and 6-3 show the results of the sensitivity analyses. The results are expressed as percent contribution to the variance (Table 6-2) as well as rank correlation (Table 6-3). Results indicate that the fish concentration generally shows the highest contribution to the predicted dietary dose variance and/or shows the highest positive correlation (R^2 typically 0.52 to 0.98). Body weight is negatively correlated with predicted dietary dose (and TQ) but the R^2 is not very high. The belted kingfisher egg toxicity quotient is strongly influenced by the TRV, as this is the only TQ which uses a TRV with an order of magnitude uncertainty factor.

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: 1) river ecosystem characteristics vary significantly from one location to another depending on flow rate, depth, sediment structure, etc.; and 2) certain parameters in the model (such as feeding preferences) are only imprecisely known. Moreover, most of the measurements are not easily related to the FISHRAND generic input parameters because, by their own nature, experimental measurements are taken at specific temporal and spatial scales while the FISHRAND model parameters are, in contrast, values corresponding to averages over time, space and species.

The effect of variation in all input parameters on model output was evaluated in a sensitivity analysis using Monte Carlo methodology and is presented in the Revised Baseline Modeling Report (USEPA, 2000a). 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 and Spearman rank regression techniques (Morgan and Henrion, 1990) are used as a formal method to find the most important parameters for the model performance. If the Spearman or partial rank regression coefficient (PRRC or SRRC) is close to 1 or -1 for a specific input model parameter, this parameter significantly influences model output. The estimated correlation coefficients for the percent lipid in water column invertebrates are above 0.5 for most species and location for the lipid normalized results. The percent lipid in fish is strongly negatively correlated with PCB body burden expressed on a lipid-normalized basis. This is because increases in lipid increase the PCB storage capacity of the fish, reducing the apparent concentration. As expected, the percent lipid in fish is positively associated for the wet weight results, but less so. This confirms that particularly on a lipid-normalized basis, the percent lipid distribution is very important. K_{ow} and benthic percent lipid are also important for some species on a wet weight basis. Feeding preferences are only weakly correlated with body burdens in terms of sensitivity to this parameter.

As described in detail in the Revised Baseline Modeling Report (USEPA, 2000a), sensitivity to model constants was evaluated by approximating an analytical solution to the model and then taking partial derivatives of all the model constants with respect to fish concentration. Derivatives of the model constants were evaluated across the full range of all parameters to determine the sign and magnitude of each of the derivatives. Both the derivatives and the rate constants were plotted over time. The assimilation efficiency and growth rate were determined to be the most important parameters in terms of effect on predicted fish concentration. This procedure was described in the approach to calibration in Chapters 3 and 6 of that report.

The greatest source of uncertainty in the FISHRAND model is in the specification of key assumptions to the HUDTOX model which in turn generates the water and sediment exposure concentrations. These assumptions include upstream boundary conditions, hydrology, and sediment loads and behavior. Also, year-to-year variability in the lipid content of fish affects both the central tendency and the distribution of predicted fish concentrations, and there is no defensible way to predict lipid content.

The RBMR shows that the relative percent difference between FISHRAND predictions and observed data is typically within 25-40%, and significantly less than that for many individual years, species, and locations. This suggests roughly a factor of two, or less, uncertainty in the mean estimate of fish concentration and was the basis for the estimated distributions in the exposure models.

6.5.3.2 Uncertainty in the Farley Model

Uncertainty in the application of the Farley *et al.* (1999) model for the purposes of characterizing risks to ecological receptors in the Lower Hudson River arises from several sources. These sources of uncertainty can be classified as one of two types: uncertainties which originate from the parameterization of the model, and uncertainties concerning the assumptions of future conditions in the Hudson.

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

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

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

Use of the Federal Dam load ratios to adjust the Farley Model and FISHRAND results adds additional uncertainty to the Lower Hudson River model concentrations. It is unlikely that the model response would be directly proportional to the change in the Upper Hudson River PCB loads in any given year. However, in the freshwater Lower Hudson River, where the Upper Hudson River load is the dominant source and given the small adjustment (ranging from 0.98 to 1.18, with an average ratio of for the 25 year period) this approximation is reasonable. Because of other loads to the system to the Lower Hudson River sediment, tributaries and New York City sources, the change in the system would most likely not be directly proportional to the change in the Upper Hudson River load, but again, this is a reasonable approximation.

The Lower Hudson River concentrations for the various media are at times higher in concentration than the Upper Hudson River concentrations which implies that the Upper Hudson River recovers more quickly than the Lower Hudson, particularly in the later years. This results from using substantially different models for the Upper and Lower Hudson River. For instance, the Upper Hudson model uses cohesive sediment concentrations as input to the FISHRAND model, whereas the Farley Model generates a segment concentration without regard to sediment type. Further analysis would be required to integrate the models in order to yield consistent results.

6.6 Summary

This chapter summarizes sources of uncertainty in the fate and transport and bioaccumulation models, toxicity reference values, and exposure calculations. Quantifiable sources of uncertainty were included to the extent possible in the sensitivity and uncertainty analyses described in Section 6.5.2. The results of that exercise showed that even at the 5^{th} percentile, predicted toxicity quotients for the bald eagle egg, belted kingfisher egg, mink and river otter did not fall below one for any location or year, except for mink at RM 154 in 2015 (Table 6-1). These results support the toxicity quotients calculated in Chapter 5.

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Chapter 7

7.0 CONCLUSIONS

This chapter summarizes the results of the risk assessment. Each assessment endpoint and associated measurement endpoints are presented along with a summary of the results. The results of the risk characterization are evaluated in the context of the uncertainty analysis 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 results of the toxicity quotient analysis, probabilistic dose-response analysis, field-based observational data, and uncertainties are considered in the conclusions.

7.1 Assessment Endpoint: Sustainability of a Benthic Invertebrate Community, Which Serves as a Food Source for Local Fish and Wildlife

Does the benthic community structure reflect the influence of PCBs?

The analysis was inconclusive since the differences seen between TI Pool locations could be due to factors such as TOC and grain size, rather than PCB concentrations.

Do measured and modeled sediment PCB concentrations exceed appropriate guidelines?

Measured sediment concentrations in the upper river based on the 1993 USEPA dataset exceed all Hudson River sediment effect concentrations (NOAA, 1999a) and NYSDEC's chronic guideline for the protection of benthic aquatic life on both an average and 95% UCL.

In the lower river, the threshold effect concentration (TEC) and mid-range effect (MEC) concentrations were exceeded at all locations on an average and 95% UCL basis. NYSDEC's chronic benthic protection guideline was exceeded at all locations in the river except RMs100. 58-7, and 25.8 on an average basis.

Predicted concentrations (1993-2018) were exceeded throughout the upper river. The TEC and NYSDEC chronic benthic protection guideline were exceeded at all upper river locations (RMs 189, 168, and 154) for the duration of the modeling period. The TEC was also exceeded at all lower river stations. The MEC was exceeded at RMs 189 and 168 for the duration of the modeling period and at other locations for a portion of the modeling period. The EEC was exceeded at RM 189 for the duration of the modeling timeframe, and at other upper river locations for a portion of the modeling period.

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 guideline for the protection of benthic aquatic life in the Upper and Lower Hudson River, with the exception of the average PCB concentrations at RMs 58.7, 47.3, and 25.8.

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Modeled surface water concentrations from the HUDTOX model (upper river) and Farley model (lower river) exceed the NYSDEC chronic guideline for the protection of benthic aquatic life for the duration of the modeling period at RMs 189 and 168 and until 2006 at RM 154. Predicted concentrations in the lower river exceed the guideline for a portion of the modeling period at all locations except RM 50.

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 sediment concentrations to sediment guidelines developed specifically for the protection of benthic invertebrates. The third was to compare measured and modeled water column concentrations to water quality criteria developed specifically for the protection of aquatic life. The benthic invertebrate community study was inconclusive, but exceedance of sediment and water guidelines, particularly in the upper river, indicate that some sensitive benthic aquatic life may be affected by concentrations of PCBs in the sediments and water.

7.2 Assessment Endpoint: Sustainability (*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 laboratory-derived NOAEL for pumpkinseed and the field-based NOAEL developed for spottail shiners in the TI Pool. Modeled PCB body burdens for pumpkinseed exceed the NOAEL on a 95th percentile basis for the duration of the modeling period at both the TI Pool and Stillwater. The LOAEL was only exceeded at the beginning of the modeling period. Modeled spottail shiner body burdens exceed the laboratory-based NOAEL for a portion of the modeling period at the TI Pool and Stillwater, and show a few exceedances on a NOAEL basis at the TI Pool. There were only a few exceedances predicted in the lower river.

These results suggest that there is limited potential for adverse effects on forage fish reproduction in the upper river (Thompson Island Pool to Stillwater) based on the calculated toxicity quotients. Body burdens were measured and modeled on a whole body basis; thus, there were no factors applied to convert from standard fillet to a whole body concentration. There were no uncertainty factors applied in the development of TRVs selected. The uncertainties in the FISHRAND model suggest approximately a factor of two uncertainty in the predicted body burdens.

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 and spottail shiner did not exceed the NOAEL at any location in the Hudson River. These results suggest that forage fish populations are unlikely to experience adverse effects. 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 are no uncertainty factors applied in the specific TEQ used in the derivation of the NOAEL and LOAEL.

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 and LOAEL in the upper river using the NYSDEC dataset for the years 1993 to 1998.

Modeled brown bullhead body burdens exceed the laboratory-derived NOAEL across most percentiles for the TI Pool and Stillwater for the duration of the modeling period. At RM 154, the median predicted brown bullhead body burden exceeds the laboratory-based NOAEL until 2003, and the 95th percentile exceeds the NOAEL until 2009. The NOAEL is also exceeded in the lower river using the 95th percentile for the duration of modeling period at RMs 152 and 113. Brown bullhead exceed the LOAEL until 2006 for the 95th percentile in the TI Pool, and typically do not exceed the LOAEL in the Lower Hudson River.

These results suggest the potential for adverse effects on omnivorous fish reproduction based on exceedances of measured and modeled body burdens compared to TRVs, particularly in the upper river. 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. No interspecies factor was applied in the derivation of the TRVs. The FISHRAND suggest approximately a factor of two uncertainty in the predicted body burdens. 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?

The predicted brown bullhead body burdens do not exceed the NOAEL (or the higher LOAEL) in either the Upper or Lower Hudson River for the duration of the modeling period. These results suggest low potential for adverse effects on omnivorous fish reproduction based on exceedances of modeled body burdens to TRVs. There are no uncertainty factors used in the derivation of the TEQ NOAEL and LOAEL.

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: The white perch is found mainly in the lower river. Measured PCB body burdens for white perch exceed the field-based NOAEL for RM 152 and RM 113 on an average and 95% UCL basis. The median modeled white perch concentrations exceed the field-based NOAEL until 1995 at RM 154 and the LOAEL was not exceeded at any river mile.

Yellow Perch: The measured yellow perch concentrations exceed the laboratory-based NOAEL and LOAEL in the TI Pool and at Stillwater for the average, 95% UCL, and maximum concentrations. Modeled median yellow perch body burdens exceed the laboratory-based NOAEL for the duration of the modeling period at the TI Pool and for a portion of the modeling period at Stillwater and RM 154. When the modeled body burdens are compared to the laboratory-based LOAEL, exceedances are seen at the TI Pool for a portion of the modeling period.

Largemouth Bass: Measured largemouth bass concentrations exceed the laboratory-derived NOAEL and LOAEL for the average, 95% UCL, and maximum at RMs 189, 168 and 113. Modeled largemouth bass concentrations exceed the laboratory-based NOAEL at RMs 189, 168, 152 and 113 for the duration of the modeling period. Using the LOAEL, toxicity quotients exceed one for the entire modeling period for 95% UCL at the TI Pool. At Stillwater, the 95% UCL exceeds toxicity quotients exceed one for a portion of the modeling period.

Striped Bass: Measured striped bass concentrations show several exceedences of the field-based NOAEL and lab-based LOAEL on a total PCB wet weight body burden basis in 1993, 1994, and 1996. Modeled concentrations of striped bass (using the Farley Model) exceeded the NOAEL for the duration of the modeling period at RM 152 and the LOAEL for all years and percentiles except for the 25th percentile starting at year 2009. The field-based NOAEL was exceeded for a portion of the sampling period at RM 113. 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 may be underestimated.

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, particularly in the reach of the river from the Thompson Island Pool to Stillwater. 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. No uncertainty factors were applied in the derivation of the TRVs for all species.

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: The measured and predicted white perch body burdens do not exceed the NOAEL (or the higher LOAEL) in either the Upper or Lower Hudson. These results suggest low potential for adverse effects on piscivorous fish reproduction based on exceedances of measured and modeled body burdens as compared to TEQ-based TRVs. There are no uncertainty factors used in the derivation of the TEQ NOAEL and LOAEL.

Yellow Perch: The measured yellow perch concentrations exceed the laboratory-based TEQ NOAEL and LOAEL in the TI Pool for the 95% UCL and maximum concentration. The predicted yellow perch body burdens do not exceed the NOAEL (or the higher LOAEL) in either the Upper or Lower Hudson River for the duration of the modeling period. These results suggest low potential for adverse effects on piscivorous fish reproduction based on exceedances of measured and modeled body burdens to TRVs. There are no uncertainty factors used in the derivation of the TEQ NOAEL and LOAEL.

Largemouth Bass: Measured largemouth bass concentrations exceed the laboratory-based TEQ NOAEL in the TI Pool for 1993 through 1995 and for the LOAEL in 1993. Modeled largemouth bass concentrations TEQ-based toxicity quotients are all less than one, with the exception of the 95th percentile at RM 189 through 1997.

Striped Bass: All TEQ-based striped bass TQs fall below one for the entire modeling period for all concentrations at RMs 152 and RM 113 for both NOAEL AND LOAEL TRVs.

These results using TEQ-based toxicity quotients suggest the potential for adverse effects on piscivorous and semi-piscivorous fish reproduction is limited. 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.

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

All observed whole water concentrations exceeded water quality benchmarks at all upper and lower river locations, with the exception of the average PCB concentration was lower than USEPA/NYSDEC chronic guideline for the protection of benthic aquatic at RMs 58.7, 47.3, and 25.8.

Modeled surface water concentrations from the HUDTOX model (upper river) and Farley model (lower river) exceed the NYSDEC chronic guideline for the protection of benthic aquatic life for the duration of the modeling period at RMs 189 and 168 and until 2006 at RM 154. Predicted concentrations in the lower river exceed the aquatic life guideline for a portion of the modeling period at all locations except RM 50. The USEPA/NYSDEC wildlife bioaccumulation value is exceeded at all upper and lower river locations for the duration of the modeling period.

Comparison of total PCB concentrations in the water column to guidelines indicates that the level of PCBs present in the Upper and Lower Hudson River may cause adverse effects to aquatic life. Comparisons may underestimate 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).

Do measured and modeled sediment PCB concentrations exceed appropriate guidelines?

The Hudson River SECs and the NYSDEC Technical Guidance for Screening Contaminated Sediments (NYSDEC, 1999a), were used as the primary sediment guidelines for comparison in this report. The NYSDEC benthic aquatic life chronic toxicity sediment criteria was exceeded at all locations in the upper and lower river on 95% UCL basis and on an average basis for all locations except of RMs 100, 58.7, and 25.8. The NYSDEC wildlife bioaccumulation value of 1.4 μg/gOC was exceeded in all comparisons.

The Hudson River TEC (0.04 mg/kg), MEC (0.4 mg/kg), and EEC (1.7 mg/kg) are exceeded at all upper river locations. In the Lower Hudson, the TEC and MEC are exceeded by the average and 95% UCL sediment concentrations at all stations, with the exception of the MEC for the average PCB concentration at RM 58.7.

In the TI Pool (RM 189), predicted sediment concentrations exceed the NOAA TEC, MEC, and EEC, NYSDEC benthic chronic and wildlife bioaccumulation values, Persaud *et al.* LEL, and Washington State guidelines for the entire modeling period. Results are similar for RM 168, with the exception that the EEC is only exceeded until 2010. At RM 154 the TEC, NYSDEC values, LEL, and Washington State guidelines are exceeded for the duration of the modeling period. The MEC and EEC are exceeded for a portion of the modeling period. Predicted sediment concentrations in the lower river exceed sediment guidelines, except for the EEC and SEC, for a portion of the modeling period.

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

The qualitative observational data presented in Section 5.2.4 can not be used to provide insight into the possibility that PCBs have reduced or impaired reproduction or rates of recruitment. Risks to some receptors may exist 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 Section 5.2.1, the potential for adverse effects resulting from exposure to PCBs may occur for the omnivorous, piscivorous and semi-piscivorous fish species in the upper river.

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 last is based on qualitative field observations. Collectively these lines of evidence indicate that current and future PCB exposures are 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 Hudson River for some species, with the most exceedances seen in upper trophic level fish, such as largemouth bass and striped bass. Toxicity quotients are greatest in the reach between the TI Pool and Stillwater. Model results show that body burdens on a total PCB (Tri+) basis are expected to remain above these levels for the duration of the modeling period for a few of the upper trophic level fish species. However, the results of the TEQ modeling indicate little potential for risk. There is a moderate degree of uncertainty in the modeled body burdens used to evaluate exposure and no uncertainty factors were applied to the TRVs. Measured water and sediment concentrations exceed guidelines at most locations in the river. Modeled water and sediment concentrations consistently exceed guidelines in the upper and lower river.

7.3 Assessment Endpoint: Sustainability (*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 average PCB dietary doses to the tree swallow exceed the field-based NOAEL at Stillwater using 1993 data, and the modeled 95% UCL dietary doses exceed the field-based NOAEL at the TI Pool and Stillwater. There were no other exceedances on the basis of 1993 data. Modeled average dietary doses to tree swallows under future conditions do not exceed the field-based NOAEL. The NOAEL was derived on the basis of Hudson River data.

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

The predicted 95% UCL dietary dose exceeds 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?

NOAEL-based comparisons at TI Pool and Stillwater exceed one for the average and 95% UCL in 1993. Modeled egg concentrations to tree swallows under future conditions do not exceed the field-based NOAEL for any year and RM except for RM 168 in 1993.

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 exceedances of the NOAEL for the 1993-2018 modeling period.

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

All observed whole water concentrations exceeded the wildlife water quality benchmark at all upper and lower river locations. Modeled water column concentrations also exceed the USEPA/NYSDEC wildlife bioaccumulation value at all upper and lower river locations for the duration of the modeling period.

Comparison of total PCB concentrations in the water column to guidelines indicates that the level of PCBs present in the Upper and Lower Hudson River may cause adverse effects to wildlife. Comparisons may underestimate 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).

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

Tree swallows (*Tachycineta bicolor*) are commonly observed along the Upper Hudson River during the spring when they are feeding in preparation for breeding (McCarty and Secord, 1999a). These researchers have observed 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 in the upper river. 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 using the tree swallow as a model. 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 wildlife; 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 impact reproduction of insectivorous bird species, but that anomalous behavior has been observed at these levels. 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 and future conditions.

7.4 Assessment Endpoint: Sustainability (*i.e.*, Survival, Growth, and Reproduction) of Local Waterfowl

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

The NOAEL-based comparison for the 95% UCL and average concentration on a dietary dose basis exceeds one at Stillwater on the basis of 1993 data. On a NOAEL basis, the predicted toxicity quotients exceed one sporadically over the entire modeling period for the average dietary dose in the TI Pool. NOAEL comparisons do not exceed one at other river miles.

These results suggest a limited potential for adverse reproductive effects as a result of PCB exposure to waterfowl in the TI Pool area.

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

The modeled average and 95% UCL TEQ-based dietary doses to the mallard duck exceed the NOAEL and LOAEL on the basis of 1993 data at all locations along the river.

These results suggest the potential for adverse reproductive effects as a result of PCB exposure via dietary dose expressed as TEQ to waterfowl.

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

For the predicted egg concentrations based on the 1993 data, the NOAEL-based comparisons exceed one for the 95% UCL at Stillwater, but nowhere else along the entire river. Modeled egg concentrations do not exceed the NOAEL at any location during the sampling period (1993-2018). A LOAEL was not available for egg concentrations.

These results suggest limited potential for adverse reproductive effects as a result of predicted total PCB concentrations in the eggs of waterfowl.

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

All NOAEL-based comparisons for the average and 95% UCL exceed one in the upper river. In the lower river, comparisons for RMs 143.5, 137.2, 122.4, 100 and 47.3 exceed one using the 95% UCL and at RM 137.2 using the average concentration. Future modeled egg concentrations (1993-2018) show that predicted NOAEL-based toxicity quotients are above 1 at all locations for at least a portion of the modeling period (only 1998 at RMs 90 and 50). All toxicity quotients fall below one by 2007.

These results suggest a potential for adverse reproductive effects as a result of predicted TEQ-based PCB concentrations in the eggs of waterfowl in the upper river.

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

All observed whole water concentrations exceeded the wildlife water quality benchmark at all upper and lower river locations. Modeled water column concentrations also exceed the USEPA/NYSDEC wildlife bioaccumulation value at all upper and lower river locations for the duration of the modeling period

Comparison of total PCB concentrations in the water column to guidelines indicates that the level of PCBs present in the Upper and Lower Hudson River may cause adverse effects to wildlife. Comparisons may underestimate 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).

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 to individual birds. The results of the TEQ-based risk calculations suggest that mallards may experience adverse reproductive effects as a result of exposure to PCBs; however, the calculations based on total PCBs indicate limited risk.

Summary: Risks to waterfowl species were evaluated using six lines of evidence using the mallard as a model. 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 impair reproduction of the mallard duck, but that modeled dietary doses and egg concentrations under current and future conditions exceed some 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. Measured and modeled water column concentrations exceed criteria developed for the protection of wildlife at all locations under current and future conditions.

7.5 Assessment Endpoint: Sustainability (*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 PCB dietary doses to the belted kingfisher exceed both the LOAEL (except for the average at Stillwater) and NOAEL at the TI Pool and Stillwater under current conditions (1993 data). For future conditions, the NOAEL was exceeded at the TI Pool and Stillwater for a portion of the modeling period.

Using a probabilistic dose-response analysis, in 1993 female kingfishers at RM 168 show approximately a 45% probability of experiencing at least a 20% reduction in fecundity, while females at RM 154 show approximately a 30% probability of experiencing at least a 10% reduction in fecundity. In 2015, female kingfishers at RM 189 show approximately an 80% probability of experiencing at least a 10% decrease in fecundity, while females at RM 168 and 154 have low probabilities (<10%) of experiencing small reductions (<5%) in fecundity.

Great blue heron: Modeled 95% UCL PCB dietary doses to the great blue heron exceed the NOAEL at the TI Pool and Stillwater under current conditions (1993 data). For future conditions, the NOAEL was exceeded at the TI Pool until 1996.

Bald eagle: Modeled 95% UCL PCB dietary doses to the bald eagle exceed the NOAEL for almost all locations in the river under current conditions (1993 data). All TRVs are exceeded at the TI Pool. For future conditions, the NOAEL was exceeded for a portion of the modeling period at the TI Pool.

Using a probabilistic dose-response analysis, in 1993, female eagles at RM 189 show approximately a 45% probability of experiencing at least a 50% reduction in fecundity. Female eagles at RM 168 show approximately a 30% probability of experiencing a 20% reduction in fecundity, while at RM 154, females show approximately a 15% probability of experiencing at least a 10% reduction in fecundity. In the year 2015, female eagles at RM 189 show approximately a 45% probability of experiencing at least a 10% reduction in fecundity. Females at RM 168 and 154 have low probabilities (<10%) of experiencing small reductions (<5%) in fecundity.

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. The birds most likely to experience adverse effects are those living in the TI Pool area. No uncertainty factors were applied to the TRVs. For current conditions, measured concentrations were used for sediment, water, and fish. Uncertainty in the predicted fish

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concentrations (which comprise the bulk of the diet for these species) for future conditions is estimated at roughly a factor of two.

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). For future conditions, the NOAEL is exceeded at all locations in the river and the LOAEL at several locations for a portion of the modeling period.

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 (1993 data) and 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. 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.

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 (1993 data) and future conditions for the duration of the modeling period.

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), with the exception of the average concentration compared to the LOAEL at RM 100. For future conditions NOAEL comparisons were greater than one at all locations for most of the modeling time frame. At the TI Pool, the LOAEL was also exceeded for the duration of the modeling period.

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 in the upper river exceed 100.

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 almost all locations in the river under current conditions (1993 data). For future conditions, both the NOAEL and LOAEL are exceeded at the TI Pool for the duration of the modeling and the NOAEL is exceeded at all upper and lower locations for at least a portion of the modeling.

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 (except LOAEL for RM 50 in 2015 and 2018).

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). For future conditions, both the LOAEL and NOAEL are exceeded at the TI Pool and Stillwater for the duration of the modeling period and the NOAEL is exceeded for the duration of the sampling period at all locations.

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, 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?

All observed whole water concentrations exceeded the wildlife water quality benchmark at all upper and lower river locations. Modeled water column concentrations also exceed the USEPA/NYSDEC wildlife bioaccumulation value at all upper and lower river locations for the duration of the modeling period.

Comparison of total PCB concentrations in the water column to guidelines indicates that the level of PCBs present in the Upper and Lower Hudson River may cause adverse effects to wildlife. Comparisons may underestimate 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).

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

Available field observations on the presence and relative abundance of piscivorous avian species along the Hudson River are an indication of the ability of these species to maintain populations.

The Hudson River supports many piscivorous birds. Many birds feed along the river, but a smaller number breed along the river. The bald eagle has begun breeding successfully in the lower river over the last four years, but it is too early to determine whether it has been successfully reestablished. Preliminary PCB results from USFWS eagle samples are high enough to be of concern.

Summary: Risks to piscivorous bird species, using the kingfisher, heron, and eagle as models, 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 may impair reproduction of these piscivorous species. 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, many 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 (1993) and future conditions.

It should be noted that the bald eagle is on the federal and NY State list of threatened and endangered species. Therefore, individual (rather than population) level effects could adversely affect the Hudson River populations. Based on the results in this report, Hudson River bald eagles are considered to be at risk.

7.6 Assessment Endpoint: Sustainability (*i.e.*, Survival, Growth, and Reproduction) of Local Insectivorous Mammals

Do modeled total PCB dietary doses to local insectivorous mammals 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 all upper river locations in the

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river under current conditions (1993 data). In the lower river most locations exceed the NOAEL using the 95% UCL. Future modeled average dietary doses exceed the NOAEL at the TI Pool, Stillwater and RM 152 for the duration of the modeling period (1993-2018) and at other upper and lower river stations for a portion of the modeling. The LOAEL was also exceeded at the TI Pool and Stillwater for the first third of the modeling period.

Do modeled TEQ-based PCB dietary doses to local insectivorous mammals 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 upper river under current conditions (1993 data). Lower river locations exceeded one for NOAEL comparisons and the 95% UCL LOAEL comparison, with the exception of RMs 88.9 and 25.8. Predicted toxicity quotients exceed the NOAEL and LOAEL at all locations for the duration of the modeling period, with the exception of the LOAEL at RM 154 from 2011 on. Toxicity quotients in the TI Pool and at Stillwater exceed 10 across all NOAEL comparisons.

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

All observed whole water concentrations exceeded the wildlife water quality benchmark at all upper and lower river locations. Modeled water column concentrations also exceed the USEPA/NYSDEC wildlife bioaccumulation value at all upper and lower river locations for the duration of the modeling period.

Comparison of total PCB concentrations in the water column to guidelines indicates that the level of PCBs present in the Upper and Lower Hudson River may cause adverse effects to wildlife. Comparisons may underestimate 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).

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

Anecdotal evidence suggests that little brown bat is common along most locations along the Hudson River.

Summary: Risks to the insectivorous mammalian species using the little brown bat as a model 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 may impair reproduction of insectivorous mammals, particularly in the upper river.

There is a moderate degree of uncertainty in the dietary dose estimates, but even assuming an order of magnitude uncertainty or more, modeled TEQ dietary doses still exceed benchmarks. Measured and modeled water column concentrations exceed criteria developed for the protection of wildlife at all locations under current and future conditions and for the duration of the modeling period in the upper and lower river.

7.7 Assessment Endpoint: Sustainability (*i.e.*, Survival, Growth, and Reproduction) of Local Omnivorous Mammals

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

Raccoon: Modeled average and 95% UCL total PCB dietary doses to the raccoon exceed the NOAEL for all locations in the Upper Hudson River under current conditions (1993 data) and the LOAEL at the TI Pool (95% UCL dietary dose only) and Stillwater. There are few exceedances in the lower river using the current data. Under future conditions, modeled average dietary doses exceed the NOAEL at the TI Pool and Stillwater through 2003 and 2002, respectively. No exceedances are seen in the lower river.

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

Raccoon: Modeled average and 95% UCL total PCB dietary doses to the raccoon exceed the NOAEL for all locations in the upper and lower river under current conditions (1993 data). All LOAEL comparisons exceed one in the upper river, as do about half of the 95% UCL comparisons in the lower river. Predicted future NOAEL toxicity quotients at all upper and lower river locations exceed one for the duration of the modeling period. All LOAEL comparisons exceed one at the TI Pool location for the duration of the modeling period. On a LOAEL basis, predicted toxicity quotients at Stillwater and Federal Dam exceed one until 2011 and 1999, respectively. There are some LOAEL exceedances in the lower river in the early modeling years.

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

All observed whole water concentrations exceeded the wildlife water quality benchmark at all upper and lower river locations. Modeled water column concentrations also exceed the USEPA/NYSDEC wildlife bioaccumulation value at all upper and lower river locations for the duration of the modeling period.

Comparison of total PCB concentrations in the water column to guidelines indicates that the level of PCBs present in the Upper and Lower Hudson River may cause adverse effects to wildlife. Comparisons may underestimate 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).

What do the available field-based observations suggest about the health of local omnivorous wildlife populations?

Anecdotal evidence suggests that the raccoon is common along the Hudson River.

Summary: Risks to omnivorous mammals using the raccoon as a model 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 may impair reproduction of omnivorous mammals. Omnivores deriving a greater portion of their diet from non-river sources are at lower risk than those obtaining a greater proportion of their diet from the river. Modeled TEQ-based dietary doses exceed benchmarks developed on the basis of reproductive effects at all locations and for the duration of the modeling period, however there is uncertainty associated with the effects, as there is limited risk calculated for future exposures using total PCB toxicity values. There is a moderate degree of uncertainty in the dietary dose estimates. Measured and modeled water column concentrations exceed criteria developed for the protection of wildlife at all locations under current and future conditions and for the duration of the modeling period in the upper and lower river.

7.8 Assessment Endpoint: Sustainability (*i.e.*, Survival, Growth, and Reproduction) of Local Piscivorous Mammals

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

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 and Lower Hudson River under current conditions (1993 data). Under future conditions, modeled average dietary doses exceed the NOAEL at all locations in the upper and lower river. The LOAEL-based comparison is exceeded for the duration of the modeling period at the TI Pool, Stillwater, and RM 152 and for a portion of the modeling period at all other locations

The probabilistic dose response analysis indicates that in 1993, female mink at RMs 189 and 168 show a high probability (90 to 100%) of experiencing a severe reduction (>80 %) in fecundity, and females at RM 154 still show a high probability (>95%) of experiencing at least a 50% reduction in fecundity. In 2015, mink at RM 189 still show a high probability (>95%) of experiencing substantially reduced (>50%) fecundity. However, mink at RM 168 show a lower probability (35%) of experiencing at least a 40% reduction, while mink at RM 154 show only a 10% probability of experiencing at least a 20% reduction in fecundity.

River 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. The NOAEL is consistently exceeded by two orders of magnitude in the TI Pool for the duration of the modeling period.

The probabilistic dose response analysis indicates that in 1993, female river otters at RMs 189, 168 and 154 show high probabilities (80 to 100%) of experiencing severe decreases (>90%) in fecundity, in comparison to otters that are not exposed to PCBs. In the year 2015, female otters at RM 189 still show high probabilities (>70%) of experiencing severely reduced (100%) fecundity (Fig 5-10). River otters at RM 168 still show high probabilities (>80%) of experiencing a substantial decrease (>80%) in 2015, while otters at RM 154 show a 30% probability of experiencing at least a 50% reduction in fecundity.

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

Mink: Modeled average and 95% UCL TEQ-based PCB dietary doses to the mink exceed both the LOAEL and NOAEL for all locations in the river under current conditions (1993 data), with the exception of the average LOAEL comparison at RM 100. For future conditions, the average dietary dose exceeds the NOAEL and LOAEL at all locations for the duration of the modeling period, with the exception of the LOAEL at RM 154 from 2007 on. The NOAEL is consistently exceeded by two orders of magnitude in the TI Pool for the duration of the modeling period.

River 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. The NOAEL is consistently exceeded by two orders of magnitude at almost all upper and lower river locations for the duration of the modeling period.

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 river 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?

All observed whole water concentrations exceeded the wildlife water quality benchmark at all upper and lower river locations. Modeled water column concentrations also exceed the USEPA/NYSDEC wildlife bioaccumulation value at all upper and lower river locations for the duration of the modeling period.

Comparison of total PCB concentrations in the water column to guidelines indicates that the level of PCBs present in the Upper and Lower Hudson River may cause adverse effects to wildlife. Comparisons may underestimate 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).

All observed whole water concentrations exceeded water quality benchmarks at all upper and lower river locations. Modeled water column concentrations exceed the USEPA/NYSDEC wildlife bioaccumulation value at all upper river locations for the duration of the modeling period. There are no exceedances of this value in the lower river, other than in 1993 and 1995 at RM 152.

Comparison of total PCB concentrations in the water column to guidelines indicates that the level of PCBs present in the Upper Hudson River may cause adverse effects to wildlife. Predicted surface water concentrations in the Lower Hudson indicate that levels are likely to be below those of concern. Comparisons may underestimate 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).

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

The intensive trapping effort by NYSDEC in the Upper Hudson River resulted in fewer mink and river otter being captured than would be expected. The average catch of three animals per trap night on the river versus 26 animals per trap night off the river strongly suggests that animals are avoiding the river or have low reproduction/high mortality rates there.

Summary: Risks to the piscivorous mammalian species using the mink and otter as models 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 may impair current and future reproduction, particularly in the upper river. 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 still exceed benchmarks. Measured and modeled water column concentrations exceed criteria developed for the protection of wildlife at all upper and lower river locations under current and future conditions.

7.9 Summary

Measured and modeled PCB concentrations were examined in receptors covering a range of feeding preferences and trophic levels in the Upper and Lower Hudson River. These analyses used the calibrated HUDTOX model to calculate upper river sediment and water concentrations, the Farley model to calculate lower river sediment and water concentrations and striped bass concentrations, and the FISHRAND model to calculate upper and lower river fish concentrations. Avian and mammalian body burdens were calculated using food web models described in Chapter 3. Receptor body burdens and egg concentrations were then compared to toxicity reference values derived in Chapter 4. No uncertainty factors were applied to TRVs derived for each species. Although most receptors exceeded some toxicity quotients for current (1993) and/or future (1993-2018) conditions, piscivorous birds and mammals showed consistent exceedances by large orders of magnitude. Effects on bald eagle populations may be occurring at the individual level, as it is a threatened and endangered species.

Piscivorous mammals, such as the mink and river otter, are at the greatest risk from PCB exposure. These animals are not being found at expected numbers in the Upper Hudson River during intensive field trapping efforts conducted by NYSDEC. Both the point estimate toxicity quotients and the probabilistic dose response analysis indicate that many individuals are likely to show reproductive failure at current and future PCB concentrations.

In 1993, female mink at RMs 189 and 168 show a high probability (90 to 100%) of experiencing a severe reduction (>80 %) in fecundity and female river otters at RMs 189, 168 and 154 show high probabilities (80 to 100%) of experiencing severe decreases (>90%) in fecundity, in comparison to otters that are not exposed to PCBs. These risks continue through to 2015 when mink at RM 189 still show a high probability (>95%) of experiencing substantially reduced (>50%) fecundity female otters at RM 189 still show high probabilities (>70%) of experiencing severely reduced (100%) fecundity.

Comparison of total PCB concentrations in the water column to guidelines also indicate that the concentrations of PCBs present in the Upper and Lower Hudson River may cause adverse effects to wildlife feeding on aquatic life.

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