HUDSON RIVER PCBs REASSESSMENT RI/FS
PHASE 2 ECOLOGICAL RISK ASSESSMENT
SCOPE OF WORK

SEPTEMBER 1998

Prepared for:

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

Prepared by:

TAMS Consultants, Inc.
New York, NY
and
Menzie-Cura & Associates, Inc.
Chelmsford, MA

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

SEP 23 1998

To All Interested Parties:

The U.S. Environmental Protection Agency (EPA) is pleased to release the Ecological Risk Assessment Scope of Work for the Hudson River PCBs Superfund site Reassessment. This document describes the approach to be taken by EPA to develop the ecological risk assessment for the Hudson River PCBs site. The ecological risk assessment will evaluate potential risk to several species of organisms exposed to PCBs in both the Upper and Lower Hudson River.

Please note that EPA has completed numerous tasks relating to the ecological risk assessment. Nevertheless, this Scope of Work is being provided so that the public is fully aware of the process that EPA is using to conduct the ecological risk assessment. The Ecological Risk Assessment Report is scheduled to be released in August 1999, after modeling work essential to the report is completed.

EPA will accept comments on the Ecological Risk Assessment Scope of Work until **Monday, November 2, 1998.** Comments should be marked with the name of the document and should include the document section and page number for each comment. Comments should be sent to:

Douglas Tomchuk
USEPA - Region 2
290 Broadway - 20th Floor
New York, NY 10007-1866

Attn: ERA SOW Comments

Similar to the release of previous Reassessment reports, EPA will make presentations on the Ecological Risk Assessment Scope of Work, as well as the Feasibility Study Scope of Work, at a Joint Liaison Group meeting on the day of release. EPA will follow-up with an availability session to answer the public's questions regarding these documents. The availability session will be held on Tuesday, October 20, 1998 at the Marriott Hotel, 189 Wolf Road, Albany, New York from 2:30 to 4:30 p.m. and from 6:30 to 8:30 p.m.

If you need additional information regarding this Scope of Work, or with respect to the Reassessment in general, please contact Ann Rychlenski, the Community Relations Coordinator for this site, at (212) 637-3672.

Sincerely yours,

William McCabe

William McCabe, Deputy Director
Emergency and Remedial Response Division

HUDSON RIVER PCBs REASSESSMENT RI/FS
PHASE 2 ECOLOGICAL RISK ASSESSMENT
SCOPE OF WORK

SEPTEMBER 1998

Prepared for:

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

Prepared by:

TAMS Consultants, Inc.
New York, NY
and
Menzie-Cura & Associates, Inc.
Chelmsford, MA

PHASE 2 - ECOLOGICAL RISK ASSESSMENT

SCOPE OF WORK

HUDSON RIVER PCBs REASSESSMENT RI/FS

TABLE OF CONTENTS

PHASE 2 - ECOLOGICAL RISK ASSESSMENT

SCOPE OF WORK

HUDSON RIVER PCBS REASSESSMENT RI/FS

TABLE OF CONTENTS

PHASE 2 - ECOLOGICAL RISK ASSESSMENT

SCOPE OF WORK

HUDSON RIVER PCBS REASSESSMENT RI/FS

TABLE OF CONTENTS

PHASE 2 - ECOLOGICAL RISK ASSESSMENT

SCOPE OF WORK

HUDSON RIVER PCBs REASSESSMENT RI/FS

TABLE OF CONTENTS

List of Tables
(tables provided at end of report)

List of Figures
(figures provided at end of report)

APPENDICES

ACRONYMS

Ah	Aryl Hydrocarbon
ARAR	Applicable or Relevant and Appropriate Requirement
AWQC	Ambient Water Quality Criteria
BAF	Bioaccumulation Factor
BSAF	Biota:Sediment Accumulation Factors
CBR	Critical Body Residue
CERCLA	Comprehensive Environmental Response, Compensation, and Liability Act
COE	Corps of Engineers
DEIR	Data Evaluation and Interpretation Report
DNAPL	Dense Non-Aqueous Phase Liquid
DQO	Data Quality Objectives
ERA	Ecological Risk Assessment
ERL	Effects Range-Low
ERM	Effects Range-Median
FDA	Food and Drug Administration
FFBAF	Foraging Fish Bioaccumulation Factor
FS	Feasibility Study
GE	General Electric
GM	Geometric Mean
GSD	Geometric Standard Deviation
HROC	Hudson River PCBs Oversight Committee
JLG	Joint Liaison Group
LOAEL	Lowest-Observed-Adverse-Effect-Level
NCP	National Oil and Hazardous Substances Pollution Contingency Plan
NPL	National Priorities List
NOAA	National Oceanic and Atmospheric Administration
NOAEL	No-Observed-Adverse-Effect-Level
NYSDEC	New York State Department of Environmental Conservation
NYSDOH	New York State Department of Health
NYSDOS	New York State Department of Sanitation
ORNL	Oak Ridge National Laboratories
PBAF	Pelagic Invertebrate Bioaccumulation Factor
PCB	Polychlorinated Biphenyl
PEL	Probable Effect Level
PFBAF	Piscivorous Fish Bioaccumulation Factor

ACRONYMS

RI	Remedial Investigation
.RI/FS	Remedial Investigation/Feasibility Study
ROD	Record of Decision
RM	River Mile
RPI	Rensselaer Polytechnic Institute
RRI/FS	Reassessment Remedial Investigation/Feasibility Study
SARA	Superfund Amendments and Reauthorization Act of 1986
SMDP	Scientific/Management Decision Point
SOW	Scope of Work
STC	Science and Technical Committee
TAGM	Technical and Administrative Guidance Memorandum
TCDD	2,3,7,8-Tetrachlorodibenzo-p-dioxin
TEF	Toxicity Equivalency Factor
TIP	Thompson Island Pool
TRV	Toxicity Reference Value
TSCA	Toxic Substances Control Act
USEPA	United States Environmental Protection Agency
USFWS	US Fish and Wildlife Service
WHO	World Health Organization
WQC	Water Quality Criteria

1. INTRODUCTION

This Scope of Work (SOW) outlines the procedures that the United States Environmental Protection Agency (USEPA) will use to develop the baseline Ecological Risk Assessment (ERA) for the Hudson River, as required under the National Oil and Hazardous Substances Pollution Contingency Plan (more commonly called the National Contingency Plan [NCP]). The assessment will quantify risks to selected biological species and communities exposed to polychlorinated biphenyls (PCBs) in the Hudson River and follow appropriate ecological risk assessment policies and guidance. The ERA will evaluate current and future risks based on the assumption of no remediation or institutional controls (USEPA, 1990).

Figure 1 is an organization chart of the individuals contributing to the ERA and their roles in the assessment.

1.1 Site History

The Hudson River PCB Superfund site encompasses the Hudson River from Hudson Falls to the Battery in New York Harbor, a stretch of nearly 200 river miles (322 km). 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 river from the 1940s to 1977 range from 209,000 to 1,330,000 pounds (95,000 to 603,000 kg) (TAMS/Gradient, 1991). In 1977, manufacture and sale of PCBs within the U.S. was stopped under provisions of the Toxic Substances and Control Act (TSCA).

PCBs discharged from the GE facilities were distributed downstream of Hudson Falls. 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. Because of its deteriorating condition, the dam was removed in 1973. Subsequent spring floods scoured PCB-contaminated sediments from the area behind the former dam and they were released downstream. The sediments released from the former Fort Edward Dam are a continuing source of PCBs. The exposed sediments from the former pool behind the dam, called the "remnant deposits," have been the subject of several remedial efforts. Capping of the remnant deposits was completed in 1991.

Although commercial uses of PCBs ceased in 1977, loading of PCBs derived from the GE plants to the Hudson River has continued, from contaminated sediments and leakage of dense non-aqueous phase liquid (DNAPL) PCBs from bedrock fractures. In September 1991 high PCB concentrations were detected in Hudson River water and traced to the collapse of a wooden gate structure within the abandoned Allen Mill adjacent to the GE Hudson Falls capacitor plant. The gate 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 PCB-bearing oils and sediments were eventually removed (O'Brien and Gere, 1995). In 1994, GE documented the presence of PCB DNAPL seeps below Hudson Falls in a dewatered portion of the river bottom on Bakers Falls. GE instituted a number of mitigation efforts that have resulted in a decline but not total cessation of these seeps.

In 1984, USEPA issued a Record of Decision (ROD) for the site. The ROD selected: 1) an interim No Action decision concerning river sediments; 2) in-place capping, containment, and monitoring of remnant deposit sediments; and 3) a treatability study (at the Waterford Water Works) to evaluate the effectiveness of removing PCBs from the Hudson River for domestic water supply.

In December 1989, USEPA Region II began a reassessment of the No Action decision for the Hudson River sediments based on the Comprehensive Environmental Response, Compensation, and Liability Act (CERCLA) five-year reevaluation requirement for remedies that leave contamination on site; the reopener in the 1984 ROD; and the request from the New York State

Department of Environmental Conservation (NYSDEC) to conduct the reassessment. The ongoing reassessment consists of three phases: Phase 1 - Interim Characterization and Evaluation: Phase 2 - Further Site Characterization and Analysis; and Phase 3 - Feasibility Study (FS). This document represents the scope of work for the Phase 2 ERA that will be developed for the Reassessment.

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. The changing upstream loading from the Hudson Falls site must be accounted for in any evaluation of PCB bioaccumulation within the Hudson River. In addition, the GE Fort Edward Plant outfall area is likely a continuing source of PCBs to the Hudson River.

1.2 Ecological Risk Assessment in the Superfund Process

This ERA will address ecological concerns of CERCLA, as amended by the Superfund Amendments and Reauthorization Act of 1986 (SARA), which authorizes USEPA to protect public health and welfare and the environment with respect to releases or potential releases of contaminants from hazardous waste sites. The NCP calls for identification and mitigation of the environmental impacts (such as toxicity, bioaccumulation, death, reproductive impairment, growth impairment, and loss of critical habitat) at hazardous waste sites, and for the selection of remedial actions to protect the environment (USEPA, 1997). In addition, numerous other federal and state laws and regulations concerning environmental protection are potentially "applicable or relevant and appropriate requirements" (ARARs). Compliance with these laws and regulations may require evaluation of site-related ecological effects and the measures needed to mitigate those effects.

Ecological risk assessment specifically for the Superfund process (USEPA, 1997) refers to a qualitative and/or quantitative appraisal of the actual or potential impacts of contaminants from a hazardous waste site on plants and animals other than humans and domesticated species.

The ERA will evaluate current and future risks. The assessment of current risk will rely primarily on the PCB congener-specific data collected during the 1993 Phase 2 ecological sampling program (Figures 2 and 3), including data collected in 1993 and 1995 by NYSDEC and the National Oceanographic and Atmospheric Administration (NOAA). The ERA will also include data collected for other Phase 2 studies, such as the Data Evaluation and Interpretation Report (DEIR) (TAMS *et al.*, 1997) and Low Resolution Coring Report (TAMS, 1998). The assessment of future risk will be based on the Baseline Modeling Report (to be released in 1999). Other data that will be evaluated during the ERA may include:

- Data collected during the late 1970s and early 1980s that was used for the 1984 FS;
- All relevant fish tissue PCB data, including data collected annually by NYSDEC since 1971, when NYSDEC added PCBs to its statewide analyses of pesticide residues in fish and GE fish data;
- New York State Department of Health (NYSDOH) benthic invertebrate surveys conducted in 1972 using multiplate samplers; and
- Hudson River avian PCB data collected by US Fish and Wildlife Service (USFWS) and NYSDEC.

1.3 Results of Phase 1 Ecological Risk Assessment

In 1991, USEPA issued the Phase 1 Report - Interim Characterization and Evaluation for the Hudson River PCB Reassessment Remedial Investigation/Feasibility Study (RRI/FS), including an interim ERA. The interim ecological risk assessment determined that:

- Data were insufficient to conduct a quantitative ecological risk assessment and recommended that additional studies be conducted;

- The interim assessment showed that PCB levels exceeded freshwater Ambient Water Quality Criteria (AWQC) for the protection of aquatic life by two- to five-fold;

- Concentrations of PCBs in sediment lower than one ppm may impact biota based on the probable effect level (PEL) of 0.277 ppm for freshwater sediments (Smith *et al.*, 1996) and effects range-median (ERM) of 0.18 ppm for saltwater sediments (Long *et al.*, 1995);

- Levels of PCBs in the Upper Hudson fish exceeded the USFWS guidelines for trout (Eisler, 1986) by a factor of ten; and

- Estimated PCB concentrations in the diets of fish eating birds and mammals at the site appear to be similar or somewhat higher than dietary concentrations recommended by USFWS or NYSDEC (TAMS/Gradient, 1991).

1.4 Changes in EPA Risk Assessment Guidance Since the Phase 1 Assessment

Since the Phase 1 risk assessment, the USEPA has issued new risk assessment policies and guidance documents. A brief summary of the new documents and their impact on the risk assessment follows.

- "Ecological Risk Assessment Guidance for Superfund: Process for Designing and Conducting Ecological Risk Assessments" was released in 1997 (USEPA, June 1997). The new guidance has eight steps and several scientific/management decision points (SMDPs). The Hudson River ERA will incorporate the new guidance into the process, as discussed in Section 1.5.

- "Guiding Principles for Monte Carlo Analysis" (USEPA, March 1997). These guidelines set forth basic approaches for developing a probabilistic risk assessment and determining when a probabilistic assessment is appropriate. The ERA will develop probabilistic estimates of exposure expressed as distributions of concentrations in media, doses, or tissue levels, and then combine them with both point estimates of effects and probabilistic estimates of effects (*e.g.*, by considering the uncertainty associated with effects within a class of animals).

- "USEPA Workshop on the Application of 2,3,7,8,-TCDD Toxicity Equivalency Factors to Fish and Wildlife" on January 20-22, 1998. A draft report of the meeting was released in February 1998 (ERG, 1998). The ERA will consider recommendations on the application of Toxicity Equivalency Factors (TEFs) to PCBs.
- "Priorities for Ecological Protection: An Initial List and Discussion Document for EPA" (USEPA, January 1997) was distributed for discussion. The purpose of this document was to stimulate discussion on ecological entities that should be considered priorities for protection and to propose a process by which decision makers can set specific ecological objectives to guide assessment and action.

- The "Great Lakes Water Quality Initiative Criteria Documents for the Protection of Wildlife; DDT, Mercury, 2,3,7,8-TCDD, and PCBs" provides the methodology to develop site-specific water quality criteria (USEPA, 1995).

1.5 Additional Toxicological Benchmarks Developed by ORNL Since the Phase 1 Assessment

In addition to the new USEPA guidance documents, Oak Ridge National Laboratories (ORNL) has released several reports pertinent to ecological risk assessment. These reports provide bioaccumulation models and toxicological benchmarks that may be used in ecological risk assessments. A subset of these publications includes:

- "Development and Validation of Bioaccumulation Models for Small Mammals" (Sample *et al.*, 1998);

- "Toxicological Benchmarks for Wildlife: 1996 Revision" (Sample *et al.*, 1996);

- "Toxicological Benchmarks for Screening Potential Contaminants of Concern for Effects on Aquatic Biota: 1996 Revision" (Suter and Tsao, 1996);

- "Risk Characterization for Ecological Risk Assessment of Contaminated Sites" (Suter, 1996); and
- "Toxicological Benchmarks for Screening Contaminants of Potential Concern for Effects on Sediment Associated Biota: 1997 Revision" (Jones *et al.*, 1997).

1.6 Organization of the Phase 2 ERA Based on USEPA 1997 Guidance

The eight steps of the ERA process, as outlined in USEPA's 1997 guidance (Figure 4), described below, provide an outline for this SOW.

Step 1 is the Screening-Level Problem Formulation and Ecological Effects Evaluation and Step 2 is the Screening-Level Preliminary Exposure Estimate and Risk Calculation. These two steps are screening-level activities that include the development of a conceptual model, selection of conservative toxicity values, and conservative estimates of exposure. Field studies of the health and condition of ecological receptors are usually not performed. These activities have already been completed as part of the Phase 1 Report (TAMS/Gradient, 1991). The Phase 1 Report concluded that PCB concentrations in the surface water, sediments, and fish exceeded federal (*i.e.*, USEPA and USFWS) and state (*i.e.*, NYSDEC) guidelines. The decision at this point was that the potential for adverse impacts exists and a more thorough assessment was warranted (Section 1.3).

Step 3 is the Baseline Risk Assessment Problem Formulation. Problem formulation at Step 3 involves further characterization of effects, refining information on fate and transport of contaminants from the source area(s), selecting assessment endpoints, and developing a conceptual model with hypotheses or questions.

Step 4 is the Study Design and Data Quality Objectives. Step 4 establishes the measurement endpoints and how the data that has been or will be generated will be used in the ERA. This step completes the conceptual model begun during Step 3. The decisions on what data will be collected, and how it will be used in the evaluation of risks to the assessment endpoints are made during this step. Data Quality Objectives (DQO) based on statistical considerations are developed in this step that will be used to analyze the data. including data reduction techniques. data interpretation methods. and statistical analyses. The product of this step is the work plan and the sampling and analysis plan. Considerations for selecting the measurement endpoints include:

- Species/community/habitat considerations - measurement endpoints should be selected to be inclusive of risks to all of the species, populations, or groups included in the assessment endpoints that are not directly measured (USEPA, 1997). In selecting a measurement endpoint, the species and life stage, population, or community chosen should be the one(s) most susceptible to the contaminant for the assessment endpoint in question.

- Relationship of the measurement endpoints to the contaminant of concern - properties such as physiology, behavioral characteristics, or life history make a particular species useful in evaluating specific contaminants. For example, minks have been shown to be among the most sensitive of mammalian test species to toxic effects of PCBs (USEPA, 1995)

- Mechanisms of ecotoxicity- toxicity issues are reviewed to ensure that the measurement endpoint will appropriately measure the assessment endpoint's toxic response of concern (USEPA, 1997).

Step 5 is the Field Verification of Sampling Design. This involves a check on the scope to determine whether it is appropriate and can be implemented. During this step all previously obtained data should be checked and the feasibility of sampling will need to be verified. Reference areas also need to be finalized at this point.

Step 6 is the Site Investigation and Analysis of Exposure and Effects. Information that was collected during the site investigation is used to characterize exposures and ecological effects. These steps follow the outline put forth in Steps 3 and 4. The exposure characterization relies heavily on data from the site investigation and can involve fate-and-transport modeling. Results from the various modeling tasks outside of the ERA will be placed into the framework developed for the ERA to calculate fish and invertebrate body burdens using a variety of models (*i.e.*, bivariate statistical model, Upper Hudson Probabilistic model, Gobas steady state model, Lower Hudson food web model, and GE bioenergetic model). The information for characterizing potential ecological effects gathered from the literature review will be combined with results from the site investigation to calculate exposures.

Step 7 is the Risk Characterization. In the risk characterization step, data on exposure and effects are integrated into a statement about risk using risk estimation and risk description. Risk estimation consists of integrating the exposure profiles with the exposure effects information and summarizing the associated uncertainties (USEPA, 1997). The risk description provides information for interpreting the risk results and identifies a threshold for adverse effects on the assessment endpoints.

Step 8 is Risk Management, which occurs after the assessment is completed. This step is the responsibility of the USEPA site risk manager, who must balance risk reductions associated with cleanup of contaminants with potential impacts of the remedial actions themselves.

These eight steps outlined above are discussed in the following sections.

This Page Was Intentionally Left Blank For Pagination Purposes.

2. PROBLEM FORMULATION

Problem formulation provides the foundation for proceeding with the subsequent portions of the ecological risk assessment. It describes the specific objectives, the scope of the ecological assessment, and the rationale for the study site. It identifies potential exposure pathways, endpoints of concern, known ecological effects, and ecological receptors.

2.1 Site Characterization

The Hudson River PCBs National Priorities List (NPL) Site is defined as the 200 miles of river from Hudson Falls to the Battery in New York Harbor. The Upper Hudson 40-mile (64-km) stretch (Hudson Falls to Federal Dam [Figure 2]) is distinguished from the Lower Hudson stretch (Federal Dam to the Battery [Figure 3]), by different physical and hydrologic regimes. The ERA will discuss the Hudson River in three sections: the Upper Hudson, Thompson Island Pool (a section of the Upper Hudson), and the Lower Hudson. Each of these sections is described below.

2.1.1 Upper Hudson River

The Upper Hudson River in the context of this ERA covers the area between Hudson Falls and the Federal Dam in Troy, New York to Fort Edward, a length of approximately 40 river miles (RM), with the exception of the Thompson Island Pool (TIP). The Upper Hudson is an entirely freshwater reach of the river. It supports a variety of aquatic and terrestrial wildlife, but no endangered or threatened species or habitats have been recorded in the Upper Hudson River (NYSDEC Natural Heritage Program Search, March 1994). However, the potential does exist for the presence of the small whorled pogonia (*Isotria medeoloides*) in Washington County.

2.1.2 Thompson Island Pool

The TIP (RM 188.5-194) is a 5.5-mile (8.9-km) stretch of the river below Hudson Falls in the upper portion of the Hudson River. The Thompson Island Pool is discussed separately from the Upper Hudson due to the large quantities and high concentrations of PCBs deposited there (TAMS *et al.*, 1997).

2.1.3 Lower Hudson River

The Lower Hudson River risk assessment will cover selected areas from Albany, New York to the Battery, a length of approximately 160 RM. The Lower Hudson River includes freshwater, brackish, and estuarine habitats. Selected sites in the Lower Hudson River are being evaluated owing to the presence of significant fish and wildlife habitats in these regions. Although PCB concentrations generally decrease along the Hudson, there are several unique natural areas of ecological importance in the Lower Hudson that are considered sensitive areas.

2.2 Contaminants of Concern

This ERA is being prepared as part of the three-phase RI/FS to reassess the 1984 No Action decision of the USEPA concerning sediments contaminated with PCBs in the Upper Hudson River. To focus on this charge, the contaminants of concern in this ERA are limited to PCBs. PCBs will be examined as:

- Congener-specific PCBs;
- Total PCBs; or
- Aroclors.

A literature search identifying No-Observed-Adverse-Effect-Levels (NOAELs), Lowest-Observed-Adverse-Effect-Levels (LOAELs), and exposure-response functions for congeners and total PCBs will be performed for the ERA. The mechanisms of toxic responses as it pertains to the various PCB groups will be discussed.

2.3 Assessment Endpoints

Risks will be evaluated with respect to the assessment endpoints. Assessment endpoints focus the risk assessment on particular components of the ecosystem that could be adversely affected by contaminants from the site (USEPA, 1997). These endpoints are expressed in terms of a group of species or populations, or habitats and ecosystem with some common characteristics (*e.g.*, feeding preferences). Assessment endpoints may also encompass a function or quality that is to be maintained or protected. The selection of assessment endpoints (USEPA, 1997) depends on:

- The contaminants present and their concentrations;
- Mechanisms of toxicity of the contamination to different groups of organisms;
- Ecologically relevant receptor groups that are potentially sensitive or highly exposed to the contaminants and attributes of their natural history; and
- Potentially complete exposure pathways.

The assessment endpoints for the Hudson River PCBs Reassessment ERA were selected to include direct exposure to contaminated media through sediment ingestion and indirect exposure to the original contaminated media via the food chain. Because PCBs are known to bioaccumulate, an emphasis was placed on indirect exposure endpoints.

The assessment endpoints selected for the Hudson River PCB Reassessment ERA are:

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

- Survival, growth, and reproduction of:
  - localized benthic macroinvertebrate community;
  - local forage fish populations; and
  - local piscivorous fish populations;

- Protection (*i.e.*, survival and reproduction) of local wildlife including:
  - piscivorous and insectivorous birds; and
  - piscivorous, insectivorous, and omnivorous mammals; and

- Protection of significant habitats.

The selected endpoints reflect a combination of values that have been identified by USEPA, NYSDEC, USFWS, and NOAA as being important, as well as ecological characteristics or species that have been identified as valuable to protect. The selected assessment endpoints along with respective measurement endpoints are listed in Table 1. It should be understood that other factors such as metals could effect ecological receptor populations and communities associated with the Hudson River.

2.4 Site Conceptual Model

The site conceptual model identifies the source, media, pathway, and route of exposure that will be evaluated in the ecological risk assessment, and the relationship of the measurement endpoints to the assessment endpoints (USEPA, 1997).

Based on the information obtained from the Phase 1 and 2 activities completed to date and the assessment endpoints, an integrated conceptual model was developed (Figure 5). In the Hudson

River PCBs Reassessment conceptual model, the initial sources of PCBs are releases from the two GE facilities located in Hudson Falls and Fort Edward. The PCBs entered the Hudson River and adhered to sediments or were redistributed into the water column. Releases into the Hudson River have continued but have been reduced in recent times. Aquatic organisms, such as macroinvertebrates and fish, are exposed to the PCBs from both contaminated sediments and water. Receptors are grouped into general trophic levels based on their diets. Trophic level is estimated by evaluating the overall diet, rather than basing it on a small proportion of the diet.

Potential exposure pathways (*i.e.*, links between the sources of contamination and the receptors exposed) will be identified by considering the source locations, the media through which contaminants may be transported, the potential for bioaccumulation, and characteristics of the receptors. The approach that will be used to evaluate exposure is discussed in Section 3.

2.5 Measurement Endpoints

Measurement endpoints provide the actual measurements used to estimate risk. They direct data collection needs. In the ERA, each of the measurement endpoints is weighed qualitatively by considering:

- Strength of association between the measurement endpoint and assessment endpoint;
- Data quality; and
- Study design and execution.

Strength of association refers to how well a measurement endpoint represents an assessment endpoint. The greater the strength of association between the measurement and assessment endpoint, the greater the weight given to that measurement endpoint in the risk analysis. Measures include:

- PCB concentrations:
- Laboratory toxicity studies:
- Field observations: and
- Food-web models.

Because ecological systems are complex and exhibit high natural variability, there is considerable uncertainty associated with estimating risks. Measurement endpoints typically have specific strengths and weaknesses related to the factors discussed above. Because of this, it is common practice to use more than one measurement endpoint to evaluate each assessment endpoint, when possible.

Measurement endpoints that may be considered include:

- Benthic community indices in relation to transfer of PCBs through the food chain (*e.g.*, richness, abundance, diversity, biomass);

- PCB body burdens in fish for use in evaluating exposure via the food chains;

- PCB body burdens in fish and wildlife populations along the Hudson River to determine exceedance of effect-level thresholds;

- PCB concentrations in water (freshwater and saline) compared to NYS Ambient Water Quality Criteria (AWQC) for the protection of wildlife (NYSDEC, 1998): and

- PCB concentrations in sediment compared to applicable sediment benchmarks such as NYSDEC Technical and Administrative Guidance Memorandums (TAGMs) (1993), Persaud *et al.*, 1993, Ingersoll *et al.* (1996), Smith *et al.* (1996), Washington Department of Ecology (1997), and Jones *et al.*, 1997 for protection of aquatic life.

In addition, observations made on disease and deformities during sampling will be noted as an indication of general organism health. PCBs can cause effects such as deformities or hormonal aberrations that can impact the ability of organisms to move, feed, or survive a normal lifespan.

2.6 Receptors of Concern

Analysis of the potential for adverse effects will be based upon selecting representative species (*i.e.*, assessment endpoint models) to represent the various trophic levels living in or near the Hudson River. Although species are categorized here by trophic level, species often feed on varied diets that do not lend themselves easily to strict categorizations.

Receptors of concern will be characterized using information on feeding habits, life histories, habitat preferences, trophic status, migratory habits, reproductive strategies, and other attributes that could influence their exposure or sensitivity to contaminants. USEPA guidance indicates that the ecological risk assessment must focus on a limited number of receptors in order to develop a "reasonable and practical evaluation" (USEPA, 1991). Due to the size and complexity of the Hudson River NPL Site, an effort was made to include species or groups that represent different trophic levels, a variety of feeding types, and several habitats (aquatic, wetland, shoreline). Not every receptor of concern will be evaluated throughout the entire Hudson River. The list of potential ecological receptors was developed with consideration for "species of concern."

Specific species were selected for evaluation within each vertebrate class examined (*i.e.*, fish, birds, and mammals) to represent a variety of trophic levels and functions in the Hudson River ecosystem. Amphibians (such as turtles) and reptiles are also found along the Hudson River, but there are currently limited tissue data available on concentrations of PCBs in herpetological fauna. Benthic invertebrates were also selected as receptors; however, they are discussed at the community.

rather than species, level. Characterization of receptors will be derived from guidance documents (*e.g.*, USEPA, 1993; Sample *et al.*, 1996) and scientific literature.

2.6.1 Macroinvertebrate Communities

The Hudson River ecological field sampling program included measuring the species richness (number of taxa), abundance (number of individuals), and biomass at a subset of the sampling stations. Benthic macroinvertebrate communities, rather than individuals or populations, will be examined based upon the assessment endpoints selected. Benthic community structure, measured by diversity (D_s), evenness (E_s), and dominance (l), will be evaluated as a food source for local fish and wildlife. PCB concentrations detected in the water column will be compared to freshwater AWQC, while PCB sediment concentrations will be compared to guidelines such as the NYSDEC TAGMs (1993), Smith *et al.* (1996), Ingersoll *et al.* (1996), Washington Department of Ecology (1997), Persaud *et al.* (1993) and Long *et al.* (1995) to determine if PCB concentrations exceed probable effect levels.

The Upper Hudson River benthic macroinvertebrate community is composed of freshwater species, while the Lower Hudson River community is comprised of a heterogeneous group of organisms adapted to various salinities. The lower reaches below RM 25 support a typical marine assemblage including marine oligochaetes, polychaetes, and crustaceans. The middle reaches from RM 25 to 50 have a mixture of freshwater and marine forms and the upper reaches above RM 50 are dominated by freshwater arthropods and oligochaetes.

2.6.2 Fish Receptors

Eight fish species, representing a range of trophic levels, will be evaluated in the ERA. These species are divided into forage fish, piscivorous fish, and omnivorous fish. Forage fish feed primarily on invertebrates, plants, and detritus. Piscivorous fish may feed on other fish in addition to the forage fish prey, and omnivorous fish feed indiscriminately upon benthic organisms, emergent vegetation, and fishes. The species that will be considered are listed below.

- Forage Fish
  - Spottail shiner (*Notropis hudsonius*); and
  - Pumpkinseed (*Lepomis gibbosus*).

- Piscivorous and Semi-piscivorous Fish
  - Yellow perch (*Perca flavescens*);
  - White perch (*Morone americana*);
  - Largemouth bass (*Micropterus salmoides*); and
  - Striped bass (*Morone saxatilis*).

- Omnivorous Fish
  - Brown bullhead (*Ameiurus nebulosus*); and
  - Shortnose sturgeon (*Acipenser brevirostrum*).

Historical databases, as well as recent field sampling efforts, will be used to develop contaminant profiles for these species. Several of the species are distributed throughout the Hudson River (*e.g.*, white perch, spottail shiner, pumpkinseed), while others are found primarily in the Upper Hudson (*e.g.*, largemouth bass, yellow perch) or Lower Hudson (*e.g.*, shortnose sturgeon, striped bass, brown bullhead).

2.6.3 Avian Receptors

Avian receptors selected will represent various trophic levels. Potential species to be evaluated include the tree swallow (*Iridoprocne bicolor*), mallard (*Anas platyrhychos*), belted kingfisher (*Ceryle alcyon*), great blue heron (*Ardea herodias*), and bald eagle (*Haliaeetus leucocephalus*).

The tree swallow is an insectivorous bird that resides along the shore of the Hudson River. The mallard is a benthivorous feeder that feeds primarily on vegetation and aquatic invertebrates. The belted kingfisher and the great blue heron are medium and large (respectively) piscivorous (fish-eating) birds found along the Hudson River. The bald eagle feeds on a variety of prey including small birds, mammals, and live and dead fish.

2.6.4 Mammalian Receptors

Mammalian receptors will also represent various trophic levels. Representatives of different feeding strategies include the insectivorous little brown bat (*Myotis* spp.), piscivorous mink (*Mustela vison*), and omnivorous raccoon (*Procyon lotor*).

Bats in New York State feed entirely on insects (NYSDOH, 1997). Some of their prey (*e.g.*, dragonflies, midges) 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.

The mink is the most abundant and widespread carnivorous mammal in North America (USEPA, 1993). Mink feed on a variety of prey including fish, aquatic invertebrates, and small mammals. Mink are particularly sensitive to PCBs and have been found to accumulate PCBs in subcutaneous fat (Hornshaw *et al.* 1983; as cited in USEPA, 1993).

The raccoon is the most abundant and widespread medium-sized omnivore in North America (USEPA, 1993). The raccoon is an omnivorous and opportunistic feeder. They feed primarily on fleshy fruits, nuts, acorn, and corn (Kaufmann, 1982; as cited in USEPA 1993), but also eat grains, insects, frogs, crayfish, eggs, and virtually any animal or vegetable matter (Palmer and Fowler, 1975; as cited in USEPA, 1993).

2.6.5 Threatened and Endangered Species

Federal and New York state-listed threatened and endangered species, including the shortnose sturgeon, peregrine falcon, and bald eagle, are found along the Hudson River. Adult shortnose sturgeon feed indiscriminately upon bottom organisms and off emergent vegetation consuming polychaete worms, molluscs, crustaceans, aquatic insects, and small bottom-dwelling fishes (Gilbert, 1989). The bald eagle and peregrine falcon are upper-trophic level birds feeding on a variety of prey.

State-listed threatened species that are found along the Hudson include the osprey and northern harrier (NYS Department of State [NYSDOS], 1990). State-recognized species of special concern include the least bittern, spotted turtle, and wood turtle.

2.6.6 Significant Habitats

Areas considered by NYSDEC, USFWS, and NOAA to be "unique, unusual, or necessary for continued propagation of key species" (USEPA, 1989) will be discussed in the ERA. The evaluation of sensitive habitats will focus on the four NOAA Hudson River Estuarine Sanctuaries, all of which are located in the Lower Hudson River. Other areas designated by the NYSDOS (1990) to be significant habitats will also be considered.

2.7 Risk Questions

Risk questions are used to determine relationships among assessment endpoints and their predicted responses when exposed to contaminants. Risk questions are based on assessment endpoints and are used during the study design to evaluate the results of the site investigation in the analysis phase and during risk characterization. The basic question at the Hudson River PCBs site is whether PCBs are causing, or have the potential to cause, adverse effects on the assessment endpoints. Formal hypotheses may be used to define explicit error rates and magnitudes of effect (USEPA, 1997). However, many of the measurement endpoints used to evaluate assessment endpoints can not be measured by formal hypothesis testing. A weight of evidence approach, using various measurement endpoints (Table 1), will provide the basis for determining whether, and to what extent, PCBs are impacting the biological resources of the Hudson River.

3. EXPOSURE ASSESSMENT

This section describes the proposed approach to characterizing the exposure of ecological receptors, including aquatic and terrestrial biota, to PCBs from the Hudson River. Categories of important Hudson River ecological receptors will be defined, and representative species from each category will be selected. This section will also present potential exposure pathways to these various ecological receptors. The exposure assessment will present exposure concentrations of PCBs in sediment, water, and/or food, to which ecological receptors may be exposed. Exposure concentrations will include measured concentrations from monitoring data as well as modeled concentrations.

Exposure of a receptor is influenced by the life histories of the species of concern. For example, an ecological receptor integrates PCB concentrations over a typical foraging and habitat area. The exposure assessment assumes that each species forages randomly over a spatial scale that is typical for that species. As described in Section 2.1, the Hudson River is divided into three areas: the Thompson Island Pool in the Upper Hudson, the remainder of the Upper Hudson to the Federal Dam at Troy, and the Lower Hudson. The exposure assessment assumes that the selected species of concern are exposed over appropriate spatial and temporal scales within each of these areas.

Exposure of a receptor is influenced by the temporal and spatial characteristics of the exposure concentrations. Exposure data will be expressed in two ways: 1) as deterministic exposure point values, defined as the concentrations experienced by the receptor and generally expressed as the 95th percent upper confidence limits on the arithmetic means of the concentrations; and 2) as probability distributions representing the temporal and spatial variability of the concentrations.

The site conceptual model (Figure 5) illustrates the potential exposure pathways. A complete exposure pathway occurs whenever there is a source of contamination, a fate and transport mechanism that delivers the contaminant to the receptor, and exposure pathways that result in uptake of the contaminant by the receptor. Existing exposure will be examined along with potential future exposure, which will be modeled. Table 2 presents potential exposure pathways for each of the endpoint species of concern, including typical foraging preferences. Existing data will be used to develop exposure concentrations under current conditions. Models will be developed to evaluate future exposure concentrations. Since there are no data available for the avian (except for the tree swallow) and mammalian receptors, all current and future exposures for these receptors will be modeled.

3.1 Exposure Pathways

3.1.1 PCBs in Sediments

Bioaccumulation of PCBs from contaminated sediments can occur via several mechanisms, including uptake from the interstitial or overlying water via respiration, direct dermal absorption, ingestion of sediment, or indirectly through the food web. PCBs in sediments adsorbed to particles and in interstitial water represent the primary sources of exposure for benthic invertebrates. In addition, epibenthic species may derive a larger portion of their exposure from overlying water. Sediments also represent an important exposure source for demersal fish such as the brown bullhead. Fish may experience indirect exposure to sediments by consuming benthic invertebrates and emergent aquatic insects that have traveled into the water column. Terrestrial receptors are also indirectly exposed to PCBs in sediments by consuming organisms as prey items that experience sediments as their primary route of exposure.

3.1.2 PCBs in Water

Aquatic organisms are exposed to PCBs in the water column through respiration, direct dermal contact, and ingestion via the food chain. Terrestrial receptors are exposed to PCBs in the water column via direct ingestion of water, direct dermal absorption, and consumption of fish and invertebrates.

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

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

3.1.3 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 represent a portion of the diet for other fish species including largemouth bass and white perch.

3.1.4 Fish Receptors

Fish accumulate PCBs from direct uptake of dissolved-phase PCBs in the water column, direct contact with water and sediments, and dietary exposures. Adult fish can be categorized as either forage fish (spottail shiner and pumpkinseed), omnivorous and primarily demersal fish (brown bullhead), piscivorous fish (largemouth bass and striped bass), omnivorous (shortnose sturgeon), or semi-piscivorous (white perch and yellow perch). Forage fish primarily consume pelagic and/or benthic invertebrates, zooplankton, and phytoplankton, while piscivorous fish primarily consume forage fish. Semi-piscivorous fish, such as yellow perch and white perch, consume a combination of smaller forage fish as well as invertebrates and plankton. Omnivorous fish are opportunistic feeders, consuming a variety of pelagic and benthic invertebrates. Omnivorous demersal fish primarily consume benthic invertebrates. Several categories of fish receptors are being considered in this analysis. These categories along with examples of representative species are discussed below:

- **Forage fish**: spottail shiner and pumpkinseed. These fish represent intermediate trophic level fish because they primarily consume plankton and macroinvertebrates. The pumpkinseed is primarily a pelagic invertebrate feeder, while the spottail generally consumes approximately equal proportions of pelagic and benthic invertebrates.

- **Semi-piscivorous**: white perch and yellow perch. Much less than 50 percent of the diet of semi-piscivorous fish is comprised of other fish. Semi-piscivorous fish primarily consume pelagic and benthic invertebrates and small amounts of forage fish. The white perch is a semi-anadromous species, spending most of its time in the Hudson River but integrating exposure over a larger area. White perch appear to feed predominantly on benthic invertebrates. The yellow perch is resident in the Hudson River year round and feeds on benthic organisms and in the water column.

- **Piscivorous fish**: largemouth bass and striped bass. These fish primarily consume other fish. The largemouth bass is a resident fish species and derives all its exposure from Hudson River sources. It feeds on both fish and larger benthic invertebrates, such as crayfish. The striped bass, a migratory species, is only resident in the Hudson River for a portion of the year but represents an important commercial fish.

- **Omnivorous fish**: brown bullhead and shortnose sturgeon (also an endangered species). The brown bullhead is primarily a bottom-feeding fish, feeding opportunistically on invertebrates, some forage fish, and other organic material that falls to the river bottom. They will, however, opportunistically consume pelagic invertebrates as well. The sturgeon is an endangered species and therefore of particular interest. This fish can live 30 years or more; thus, there is greater potential for accumulation of PCBs. Shortnose sturgeon typically feed on chironomids, isopods, amphipods, crustacea, and molluscs (Bain, 1997).

3.1.5 Avian Receptors

Tree swallows are exposed to PCBs via contact with water and through dietary exposure. All avian and mammalian receptors' PCB body burdens may be partly attributable to other pathways of PCB exposure not considered in this assessment, such as airborne exposure and exposure from sources other than the Hudson River.

Kingfisher, great blue heron, and the bald eagle (a threatened species) are exposed to PCBs via direct contact with water, ingestion of invertebrates and fish, and, in the case of the bald eagle, ingestion of small mammals that may themselves have been exposed to PCBs via contact with water, ingestion of prey, and ingestion of sediments. The mallard is exposed to PCBs via direct contact with the water, ingestion of aquatic plants and invertebrates, and dabbling and filtering through sediments.

3.1.6 Mammalian Receptors

Mink and raccoon are exposed to PCBs primarily via the food chain from ingestion of invertebrates and fish. They are also exposed to PCBs through direct contact with water and ingestion of water. Mammalian receptors may experience exposure via a terrestrial pathway through the floodplain. However, as the focus of this reassessment is on addressing PCB-contaminated sediment within the river, floodplains will not be a primary concern in the ERA.

Little brown bats are exposed to PCBs via direct contact with water as well as ingestion of emergent aquatic insects. Much of the little brown bat food source is attributable to insects that emerge from sediments and travel up the water column; thus, they are exposed to both water and sediment sources of PCBs (Kovats and Ciborowski, 1989). Exposure to water column sources of PCBs can occur through feeding activities, diurnal behavior, or during emergence. The life history of an adult emergent insect is very short (*i.e.*, on the order of days). Bats represent an important receptor in the overall food chain due to their particular feeding strategy.

3.2 Quantification of PCB Fate and Transport

Fate and transport and food chain models are used to assess conditions beyond the time of data collection and to interpolate between point-in-time measurements for areas other than those at which data were collected and for species for which data were not obtained. Fate and transport models are being developed to describe the distribution of PCBs in the Hudson River. These include mass balance models that will be used to predict summer-averaged water and sediment concentrations for future years. These modeled sediment and water concentrations will provide initial concentrations for the food chain models. Several bioaccumulation models are being developed for six fish species (Appendix A). Results from these models will be extrapolated for the fish species that are not explicitly being modeled for the reassessment (*i.e.*, striped bass and shortnose sturgeon). Models will also be developed for the terrestrial species for which there are no direct observations, including the bald eagle, great blue heron, kingfisher, mallard, mink, bat, and raccoon. There are some data available for the tree swallow (USFWS, 1997).

The Hudson River PCBs Reassessment RI/FS DEIR (TAMS/Cadmus/Gradient, 1997) provides a detailed discussion of the fate and transport of PCBs in the Hudson River. More information on the fate and transport models, including water and sediment mass balance models as well as bioaccumulation models, is included in the Hudson River PCBs Reassessment RI/FS Preliminary Modeling Calibration Report (Limno-Tech *et al.*, 1996).

To assess future conditions assuming baseline conditions in the absence of any remediation, the analysis will rely on modeled exposure concentrations evaluated at future times relative to the natural life history, migration habits, and time of sexual maturity of the specific receptor. Typically, the initial concentration of a five- or ten-year interval will conservatively be assumed to hold, even though models show a time-varying decrease over time. For example, risk might be evaluated at five-year intervals for a receptor with a life span of ten years (See Section 5 for further details).

3.3 Observed Exposure Concentrations

To assess PCB exposure to aquatic receptors, sediment, benthic, and fish samples were collected by TAMS, NYSDEC, and NOAA in August 1993, in both the Upper and Lower Hudson River (Figures 2 and 3). These samples were analyzed for congener-specific PCBs as part of the Phase 2 ecological sampling program. Water samples collected during this period in both the upper and lower river were analyzed for congener-specific PCBs as part of the Phase 2 water-column transect and flow-averaged sampling programs. Aroclor and PCB totals were estimated from these data. For further information and specific details on the field sampling program see the Phase 2 Sampling and Analysis Plan/Quality Assurance Project Plan, Vol. 1 & 2, TAMS/Gradient 1992 and 1993. NYSDEC has been collecting fish tissue data on an Aroclor basis since 1971. NYSDEC and NOAA collected congener-specific PCB data during 1995. Data were collected for tree swallows by the USFWS for some locations (USFWS, 1997). GE has also performed a number of studies examining PCB concentrations in various media in the Hudson River. Each of these types of data will be discussed in the following sections.

Appropriate statistics will be used from the observed data to characterize exposures and body burdens (also known as critical body residue). These include arithmetic averages and 95 percent upper confidence limits on these averages. In some cases, data are sufficient to allow exposures, dietary doses, and/or body burdens to be characterized as distributions, typically lognormal in shape, and described by a mean and standard deviation of the underlying normal distribution.

3.3.1 Sediment Concentrations

Sediment data were collected at 20 locations in the Hudson River during the 1993 USEPA field program. Sediment samples were taken in the most biologically active zone of 0 to five cm (0 to two inches). Five samples were obtained for each location and analyzed on a congener basis from which Aroclor, homologue totals, and total PCBs were obtained. Sediment data taken for the high-resolution and low-resolution sediment sampling programs will be used as needed in the ERA.

3.3.2 Water Column Concentrations

Water column data were collected at 14 locations in the Hudson River over the course of one year. The ERA will use the summer-averaged water column concentrations as the basis for exposure to aquatic organisms and for comparison to water quality benchmarks. Data collected by GE will be used to supplement EPA data.

3.3.3 Benthic Invertebrate Concentrations

Data on benthic invertebrate communities and PCB body burdens were collected at the ecological monitoring stations. PCB concentrations were analyzed in benthic invertebrate communities and for individual species, when sufficient mass was available. PCB concentrations will be averaged using all samples to obtain exposure point concentrations for fish that may be consuming invertebrates as prey items.

3.3.4 Fish Concentrations

Fish were collected at 16 of the ecological sampling locations along the Hudson River. Only three sampling locations in the Thompson Island Pool, selected specifically for the benthic invertebrate community study, were not sampled for fish. The fish species and number of samples collected at each monitoring location varied according to the fish caught. Not every species was collected at every location. Data for individual species will be compiled by location. Observed body burdens are assumed to reflect integrated exposure over appropriate spatial and temporal scales.

3.3.5 Avian Concentrations

Data collected by the USFWS (1997) will be used to evaluate tree swallow body burdens for those locations at which data are available. There are no measurements of PCBs in kingfisher, great blue herons, mallard, or bald eagles; thus, these will be modeled. One bald eagle was analyzed for PCB in 1997 (*NY Times*, Sept. 17, 1997), but this does not provide enough data with which to assess potential exposures and effects from Hudson River sources.

3.3.6 Mammalian Concentrations

All mammalian exposures will be modeled as there are no observations available.

3.4 Modeled Exposure Concentrations

As discussed above, fate and transport models will be used to obtain exposure concentrations for future years, for areas at which data were not collected and/or for species for which data are unavailable. This section provides the framework for developing modeling approaches for each of the identified receptors.

Bioaccumulation models are being developed for the aquatic receptors (Appendix A). Exposure of mammalian and avian receptors will be described as doses averaged over appropriate temporal and spatial scales. Typically, primary exposure of organochlorines in aquatic systems expected results from ingestion pathways (McCarty and Mackay, 1993). Inhalation, direct dermal contact, and incidental ingestion of sediments pathways are not anticipated to contribute significant amounts of risk, based on the physical-chemical properties of PCBs and the life histories of most of the endpoint species. Screening-level calculations will be used to determine the relative contributions of these pathways. There are only limited measurements in air; consequently, a simple model based on liquid/air diffusion principles will be used to calculate expected air concentrations. Parameters to quantify incidental ingestion of sediments on a species-specific basis are rarely available or highly uncertain (USEPA, 1993).

The general form of the model for direct ingestion of water is as follows:

$$Dose = \frac{C_{water} \bullet IR \bullet FR}{BW}$$

where:

Dose = average dose from water averaged over appropriate temporal scale or per day
C_{water} = average concentration of PCBs in water (µg/L)
FR = fraction of total water ingestion from Hudson River
IR = ingestion rate (L/day)
BW = body weight (kg).

The general form of the equation for dietary doses from ingestion of prey items is as follows:

$$Dose = \sum_{i=1}^{n} \left[\frac{IR_i \bullet C_i \bullet TUF}{BW} \right]$$

where:

Dose = expected dose from prey items (µg PCB/kg body weight/day, wet weight)
IR_i = ingestion rate of i^{th} food item (kg wet weight)
C_i = concentration of PCBs in ith food item (µg PCB/kg body weight)
TUF = adjustment factor to account for foraging range and migration factor (unitless)
BW = body weight of endpoint species (kg).

Additional terms, such as assimilation and metabolic efficiencies, are required to express exposed doses as absorbed doses (*i.e.*, critical body residues), and depend on physiological factors of each species. Whether a dose is expressed as a critical body residue or as an average daily exposed dose depends on the toxicity reference value being used. In some cases, a biomagnification factor can be applied to exposure concentrations to determine the critical body residue of the receptor species. Biomagnification factors are also available to predict concentrations of PCBs in the eggs of piscivorous birds.

The general form of this model will be modified for the particular endpoint species of concern. The models will be parameterized by obtaining values from the literature, including USEPA (USEPA, 1993). Specific parameters will depend on the spatial and temporal characteristics of PCB contamination as well as the natural life history of the endpoint species. The mass balance models will provide initial sediment and water concentrations for future years from which expected body burdens will be estimated.

3.4.1 Benthic Invertebrate Receptors

Concentrations of PCBs in invertebrates are estimated through a distribution of site specific biota: sediment accumulation factors (BSAF) derived from the USEPA Phase 2 data. The BSAF does not distinguish between sediment porewater and PCBs adsorbed to particles, but reflects the general relationship between sediment concentrations and observed body burdens.

3.4.2 Fish Receptors

Several models, including species-specific probabilistic bioaccumulation models, are being developed for six fish species based on feeding preferences (Appendix A) and relationships between trophic levels. These models will be extrapolated to striped bass and sturgeon based on the natural life history of these fish species.

3.4.3 Avian Receptors

The avian receptors considered in this ecological risk assessment include the tree swallow, bald eagle (an endangered species about to be downgraded to threatened), great blue heron, and kingfisher. Models will be developed for these species based on exposure factors obtained from the "Wildlife Exposure Factors Handbook" (USEPA, 1993) and from other literature sources. The models will take into account spatial and temporal characteristics of PCB contamination in water and fish as well as feeding preferences of the avian receptors.

Expected concentrations will be expressed as dietary doses. In addition, a biomagnification model will be used to predict concentrations in the eggs of piscivorous birds (USEPA, 1994).

3.4.4 Mammalian Receptors

The mammalian receptors considered in this ecological risk assessment include the little brown bat, raccoon, and mink. The little brown bat feeds primarily on aquatic insects while the raccoon and mink feed on a combination of forage fish and other small mammals. Models will be developed for these mammalian receptors based on the concentrations of PCBs in prey items and water in conjunction with feeding preferences and habitat ranges obtained from the literature.

This Page Was Intentionally Left Blank For Pagination Purposes.

4. EFFECTS ASSESSMENT

PCBs are considered to be the main stressor of concern in this ERA. The exposure assessment (Section 3) described various exposure pathways for the receptors of concern. This chapter discusses PCB toxicity and the measures of effect for the receptors of concern.

4.1 Estimating the Toxicity of PCBs

Toxicity measurement endpoints will be established for each receptor species or group using published toxicity studies. Toxicological endpoints that will be evaluated in the ERA are:

- Survival;
- Growth; and
- Reproduction.

Each endpoint will not be evaluated for every receptor.

Studies will be evaluated and applied to the ERA, rather than conducting site-specific toxicity studies, due to the size of the Hudson River site and the associated level of effort required to obtain site- and species-specific data. PCB toxicity has most commonly been assessed on a total PCB or Aroclor mixture basis. This approach is subject to considerable inaccuracy, because it ignores the fact that toxicity is due to specific PCB congeners and the environmental distribution of congeners is typically very different from those found in pure Aroclors. As more data have become available, other approaches including the TCDD-Toxicity Equivalent Factors (TEF) and congener-specific toxicity can be used, as discussed in the following sections.

4.1.1 Total PCBs and Aroclor Toxicities

The toxicity of PCB mixtures to aquatic and terrestrial organisms varies according to composition of the PCB mixture. Differences in factors such as percent chlorine, solubility, congener structure, organism sensitivity, and species-specific sensitivity contribute to the overall complexity in evaluating PCB toxicity. Toxic effects of PCBs are generally chronic, rather than acute. The threshold for estimated adverse ecological effects will generally use NOAEL toxicity values, with safety factors depending on factors such as test species, length of study, and life stage.

Toxicity values will be examined for total PCBs and Aroclor mixtures. Total PCBs will be calculated by summing the congeners on non-overlapping Aroclors. Issues of Aroclor quantitation are more complex, as various Aroclor quantitation measures have been used to measure PCBs in the Hudson River and associated biota over the last 20 years. Appendix B discusses approaches that will be used to evaluate Aroclor data.

4.1.2 Congener-specific Toxicity and the Toxic Equivalency Factors (TEF) Approach

Individual PCB congeners have been shown to induce mortality and produce reproductive, developmental, and neurological effects. Study of structure-function relationships for PCB congeners have identified two major structural classes of PCBs that elicit "2,3,7,8-Tetrachlorodibenzo-p-dioxin (TCDD)-like" responses. These are the coplanar PCBs, also referred to as non-ortho-chloro-substituted congeners, and the mono-ortho-chloro-substituted congeners that have one chlorine in the ortho position. Both of these classes of congeners bind to the aryl hydrocarbon (Ah) receptor, as does 2,3,7,8-TCDD. Binding to the Ah receptor is used as an index of dioxin reactivity and toxicity.

Based on PCBs' mechanistic similarity to TCDD and the fact that they often exist as complex mixtures in the environment, efforts have been made to derive toxic equivalency factors (TEFs) to express the toxicity of individual PCB congeners relative to the toxicity of TCDD. The ERA will follow the recommendations of the USEPA Workshop on the Application of 2,3,7,8-TCDD Toxicity Equivalency Factors to Fish and Wildlife (ERG, 1998), and will use values that have been developed by USEPA and USFWS. PCB congeners that may play a significant role in the effects assessment, based on toxicity, distribution, persistence, and concentration, will be evaluated individually for toxicity effects and TEFs, when warranted and when adequate data is available. Of the most toxic (coplanar) congeners, the Phase 2 database includes usable data for BZ#77 only. If the data for BZ#77 is determined to be adequate, TEFs will be used to compare measured and modeled congener-specific PCB tissue concentrations in receptor species to concentrations that may result in adverse ecological effects.

4.2 Measures of Effect

Toxicological measurement endpoints will be examined for all receptors. The threshold for effects will be determined for each receptor population, as described below. Sources of toxicological data will include refereed scientific literature, the USEPA AQUIRE database, and government publications.

4.2.1 Benthic Invertebrate Communities

The effects measures that will be used for benthic invertebrates include community analyses, concentrations of PCBs in sediment, and measurement of body burdens. A benthic invertebrate community assessment was conducted in the 1993 sampling program to examine community diversity, species abundance, and potential effects of PCBs on the benthic community. The endpoint for this measure is the correlation of community indices (*e.g.*, diversity [D_s], evenness [E_s], and dominance [*l*]) with PCB concentrations taken at the sampling areas. Areas with low PCB concentrations are considered to be more representative of reference areas than areas with elevated PCB concentrations. This approach is taken due to the variability associated with areas along the river and because community indices are difficult to compare owing to the large number of parameters that affect community structure (*e.g.*, grain size, oxygen levels).

The toxicological measurement endpoints, to be referred to as Toxicity Reference Values (TRVs), will represent PCB concentrations that have been shown to cause adverse effects in test species. TRVs will be taken from published studies, generally derived from the most sensitive individual (based on species and age class). Body burdens measured in the 1993 field sampling effort will be used to estimate body burdens of individual species and of the entire benthic invertebrate community.

4.2.2 Fish Receptors

Fish effects will be measured using measured and modeled PCB body burdens. This approach is known as the Critical Body Residue (CBR) approach. TRVs will be based on published studies, as for the invertebrate studies. Measurement effects will be based on data available for both the test species most similar to the receptor and on the most sensitive age class, since all age classes are assumed to be exposed. Current body burden concentrations, with the exception of the shortnose sturgeon, will be based on measured body burdens. Future exposure concentrations and shortnose sturgeon concentrations will be based on body burden models. Exposure models are primarily designed for adults, which are generally the longest-lived age class. Body burdens will be directly compared to literature-based TRVs, rather than introducing an additional adjustment or modeling step.

4.2.3 Avian Receptors

Measurement endpoints are measured (for tree swallows) and modeled PCB body burden concentrations. Body burden models for birds will extend to upper trophic level receptors (*i.e.*, the bald eagle), providing the highest level of bioaccumulation presented in the ERA. PCB body burdens will be compared to appropriate literature-based TRVs.

4.2.4 Mammalian Receptors

The measurement endpoints for mammalian species will be based on modeled PCB body burden concentrations that will be compared to literature-based TRVs.

4.2.5 Threatened and Endangered Species

For the protection of threatened and endangered species, measurement endpoints will be based on the most sensitive species or age class. Modeled PCB body burden concentrations will be compared to literature-based TRVs. In addition, PCB concentrations will be compared to federal and New York State AWQC for the protection of aquatic species and wildlife. The WQC are directed to the most sensitive species in general, which are considered to be protective of aquatic species and wildlife, including threatened and endangered species.

4.2.6 Significant Habitats

The measurement endpoint used for significant habitats will be federal and New York State AWQC for the protection of aquatic species and wildlife.

This Page Was Intentionally Left Blank For Pagination Purposes.

5. RISK CHARACTERIZATION

Risk characterization integrates the results of the exposure assessment and effects assessment to obtain an estimate of the level of effects that will result from exposure of the endpoint species to concentrations of PCBs in the Hudson River. Consistent with USEPA guidance (USEPA, 1997), a weight-of-evidence approach is utilized to evaluate potential risks. This approach includes field observations (obtained from the literature specific to the Hudson River), comparison of measured and modeled exposure to biota to TRVs, and qualitative assessments on community structure and abundance. This information is integrated to provide a perspective on the potential for impacts to biota that use the Hudson River as a habitat or as a foraging and drinking water source.

The primary information available consists of analytical chemistry data for sediment, surface water, benthic invertebrates, fish tissues (seven species), and tree swallows (limited locations). Thus, the risk characterization will rely primarily on a toxicity quotient approach in which exposure concentrations are compared to toxicity reference values resulting in a ratio. Generally, if exposure exceeds appropriate benchmarks (typically expressed as a threshold effect level), the potential for risk is considered to exist. Such values do not necessarily indicate that an effect will occur but only that a lower threshold has been exceeded. The toxicity quotient method provides some insight into general effects upon individual animals in the local population. If effects are judged to be insignificant at the average individual level, they are not likely to be significant at the population level. However, if risks are present at the individual level, they may or may not be important at the population level.

Exposure can be described as either a single point value or a distribution, and the same is true for the toxicity reference values. A probabilistic approach will be taken for the fish species, but the avian and mammalian analyses will consist of point estimates with an appropriate uncertainty analysis to determine the effect of changes in specific model assumptions. The probabilistic approach estimates community risk by estimating risk as a percent of individuals affected by acute or chronic toxicity on a critical body residue basis. This is described in Subchapter 5.2 below.

5.1 Surface Water Concentrations

Water column concentrations will be compared to ambient water quality criteria and applicable state criteria. Ambient water quality criteria have been developed for the protection of aquatic health and to insure a healthy, diverse community. Exceedances provide an indication that there is the potential for risk to aquatic organisms, including invertebrates, fish, and terrestrial receptors using the Hudson River as a food source. Exceedances of ambient water quality criteria may also indicate a potential risk to wetland community structure and may reduce the habitat value of a particular area. Appropriate criteria will be used depending on the salinity gradient of the specific portion of the river being assessed.

5.2 Sediment Concentrations

NOAA (Long *et al.*, 1995) assembled data on concentrations of PCBs in sediments and measures of effects from many sources. From this information, NOAA estimated effects-based sediment criteria and published ER-L (effects range - low), and ER-M (effects range - median). These sediment concentrations are not criteria or standards, but provide some perspective on the PCB levels in the Hudson River. Sediment concentrations in the Lower Hudson will be compared to ER-L and ER-M to qualitatively assess the potential for risk to sediment-based organisms. For the Upper Hudson, Persaud *et al.* (1993), NYSDEC TAGMs (1993), Smith *et al.* (1996), Jones *et al.* (1997), Ingersoll *et al.* (1996), and Washington Department of Ecology (1997) guidelines will be considered. All benchmarks used will be appropriate for the sediment-type/salinity being considered.

5.3 Benthic Invertebrates

The survival, growth, and successful reproductive capability of benthic invertebrates represent assessment endpoints both in terms of the invertebrates themselves and as prey items for fish species. The corresponding measurement effects are based on concentrations of PCBs in sediments leading to body burdens in benthic invertebrates as well as measured PCB body burdens.

Qualitative analyses of community structure and abundance at each of the sampling locations will be used as another line of evidence. Because of the difficulty in attributing specific results or differences between stations to PCBs alone, this information will be used qualitatively to provide another perspective on PCB contamination in the Hudson River.

Effects ranges of PCB concentrations in sediments as reported by NOAA will be used in the marine/estuarine reaches of the Hudson River where Persaud *et al.* (1993), NYSDEC TAGMS (1993), Smith *et al.* (1996), Jones *et al.* (1997), Ingersoll *et al.* (1996), and Washington Department of Ecology (1997) guidelines will be considered as an additional method to assess effects and to provide perspective on the potential for risk to aquatic biota. All benchmarks used will be appropriate for the sediment-type/salinity being considered.

5.4 Fish Receptors

The survival, growth, and reproductive capability of local fish populations represent an assessment endpoint. The corresponding measurement endpoints are fish body burdens, and surface water and sediment concentrations leading to body burdens in fish at which effects have been observed.

A number of proposed approaches exist to characterizing population-level risks (Suter, 1993). Generally, the approach involves the following steps:

- Define the effects assessment as an extrapolation of series or statistical extrapolations;
- Develop a statistical model for each extrapolation;
- Calculate point estimates and cumulative variances to generate probability; distributions for the test endpoint that serves as the surrogate for the assessment endpoint; and
- Calculate the probability of exceeding the endpoint (or distribution of endpoints) given a distribution of exposure concentrations.

A risk model that relates PCB body burdens to the percent of affected individuals will be used to estimate population-level risks. The expected body burden of fish can be expressed as a distribution of the expected variability in body burdens resulting from differences in species-specific exposures. This function, *f*(BB), is a lognormal distribution with parameters $\mu_{\ln(x)}$ and $\sigma_{\ln(x)}$. Depending on the type and availability of toxicological data, a cumulative distribution function can be constructed based on critical body residue effects data to represent expected effects, given as g(TRV). If these distributions can be described by a mean and standard deviation, then the probability that *f*(BB) > *g*(TRV) is given by:

$$p = \phi_z\left(\frac{\mu_{BB} - \mu_{TRV}}{\sigma_{BB} + \sigma_{TRV}}\right)$$

where:

p = probability of *f*(BB) > *g*(TRV)
Φ_z = cumulative distribution function of a standard normal random variable
μ_{BB} = expected value (mean) of natural log of *f*(BB)
μ_{RBB} = expected value (mean) of natural log of *g*(TRV)
σ_{BB} = variance of natural log of *f*(BB)
σ_{RBB} = variance of natural log of *g*(TRV).

If only a point value is available for g(TRV), then the natural log of that value is used in place of the mean, and only the standard deviation for f(BB) included in the denominator.

The probability of y individuals out of a total n experiencing a f(BB) > g(TRV) is estimated using a cumulative binomial probability function defined as:

$$R = \left(\frac{n}{y} \right) p^{y} (1 - p)^{n-y}$$

where:

R = probability of y individuals out of a total of n experiencing f(BB) > effect level
n = total number of individuals (based on biomass estimates, field observations, judgment)
p = probability of f(BB) > effect level
y = number of individuals experiencing f(BB) > effect level.

The exact form of the model will depend primarily on the availability of toxicological data and the form in which exposure data are expressed (*i.e.*, critical body residue or dietary dose). The model shown here is an example of how a population level risk characterization might proceed. Other alternatives include logit or probit functions to describe dose-effect and a logistic model to express the probability that the receptor of concern will exceed a particular effect level. The model that is ultimately developed will provide perspective on the expected exposure concentration relative to the TRV in a population context which can be interpreted relative to risk management goals.

5.5 Avian Receptors

The survival and reproductive capability of piscivorous birds (bald eagle, great blue heron, kingfisher) represent assessment endpoints. The corresponding measurement endpoints are observed and modeled body burdens and/or dietary doses in avian receptors, as compared to appropriate toxicity benchmarks. Modeled body burdens and/or dietary doses will be based upon point estimates, but the models will be evaluated in an uncertainty analysis to determine the impact of specific assumptions on the results. The uncertainty analysis is described in Section 6. An appropriate biomagnification factor to estimate the expected concentration in eggs will also be used (USEPA, 1994).

The survival and reproductive capability of tree swallows and mallards is another assessment endpoint. These species are not piscivorous, but are considered to be representative receptors due to their size, position in the trophic food web and foraging strategies.

5.6 Mammalian Receptors

The survival and reproductive capability of piscivorous mammals (mink and raccoon) represent assessment endpoints. The corresponding measurement endpoints are modeled body burdens and/or dietary doses in mammalian receptors, as compared to appropriate toxicity benchmarks. Mink have been shown to be particularly sensitive to the effects of PCBs. Modeled body burdens will be based upon point estimates, but the models will undergo an uncertainty analysis to evaluate the impact of specific assumptions on the results. This is described in Section 6. The survival and reproductive capability of brown bats also represent assessment endpoints. Bats consume emergent aquatic insects and occupy a unique position in the food web of the Hudson River.

5.7 Threatened and Endangered Species

The survival and reproductive capability of threatened and endangered species (bald eagle and shortnose sturgeon) represent assessment endpoints. The corresponding measurement endpoints are modeled body burdens in fish and birds, as compared to appropriate TRVs as discussed in Sections 5.4 and 5.5.

5.8 Significant Habitats

Wetland community structure and habitat value represents assessment endpoints. The corresponding measurement endpoints include fish and invertebrate body burdens (modeled and/or observed), as compared to TRVs, and surface water and sediment PCB levels (modeled and/or observed) as compared to federal and New York state benchmarks. The surface water and sediment levels provide perspective on the ability of the habitat area to support a diverse array of receptors.

This Page Was Intentionally Left Blank For Pagination Purposes.

6. UNCERTAINTY ANALYSIS

There are many potential sources of uncertainty in assessing ecological risks. These can be roughly grouped as follows:

- **Measurement and/or sampling error**: Potential error or bias can result from sampling design and collection in terms of whether particular samples are indicative of true concentrations over an appropriate spatial and temporal scale. Further, PCB concentrations in Hudson River water and sediments are highly variable in space and time, resulting in sample uncertainty for representation of "true" population parameters. There is also uncertainty in quantitation techniques. In particular, this ecological risk assessment will rely on 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.). USEPA is assessing the differences between data sets in order to apply correction factors as appropriate (Appendix B).

- **Conceptual model uncertainties**: The conceptual model links PCB sources, likely exposure pathways, and potential ecological receptors. There are uncertainties in the specification of these linkages. This source of uncertainty will be discussed qualitatively.

- **Natural variation and parameter error**: Parameter error includes both uncertainty in estimating specific values of parameters and forcing functions in the exposure models (*i.e.*, sediment and water concentrations, etc.) as well as variability (*i.e.*, 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 a particular endpoint species. Variability generally cannot be further reduced with additional data collection, whereas uncertainty can be reduced by collecting additional data. Uncertainty is truly unknown but could be known better if more data were available. Both uncertainty and variability can be represented by distributions, but it is important to separate them analytically in order to be able to distinguish true population heterogeneity from that which is poorly known and would benefit from additional data. When variability and uncertainty are operationally indistinguishable, this represents the risk to an individual selected at random from the particular population.

- **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 and this analysis proposes to evaluate model error qualitatively.

Another significant source of uncertainty in the ecological risk assessment lies in the effects assessment. Generally, toxicological data will not be available for the specific species of concern, so there is uncertainty in species-to-species extrapolation. Toxicity data are frequently based on acute tests but may be used in the analysis to predict chronic effects. Congener profiles of PCBs in test mixtures are often very different from environmental mixtures of PCBs; thus, there is uncertainty extrapolating effects observed from exposure to one type of commercial PCB mixture versus that actually experienced in the field.

6.1 Approaches to Assessing Uncertainty

Exposure models will be developed for each of the endpoint species of concern. One means of assessing uncertainty is to conduct a sensitivity analysis in which individual model parameters are varied while holding all other parameters constant. For example, one analysis could use the upper bound on the plausible range for the particular variable. We will run the avian and mammalian models using the upper bound on the ingestion rate of a prey item, holding all other variables constant to evaluate the effect on the outcome.

Another approach to characterizing uncertainty specifies distributions for each of the uncertain parameters in a Monte Carlo analysis. The distributions represent true uncertainty, not variability. Variable parameters are held at the expected value or mean to be able to determine the quantitative impact of changes in how the uncertain parameters are specified. To combine both of these steps, a second-order Monte Carlo analysis can be conducted (USEPA, 1996b). In this analysis, uncertainty and variability are explicitly identified by nesting the variability simulations within an uncertainty loop.

The approach to assessing uncertainty in the Hudson River ERA will begin with a sensitivity analysis to evaluate the relative importance of each of the parameters in the exposure models. Each of the parameters will be identified and appropriate statistics provided to the extent the data allow. For example, the typical body-weight normalized feeding rate of a piscivorous bird will be identified along with a quantitative or qualitative indication of the relative confidence in the estimate. This will be done for each of the parameters of interest.

In the case of fish, uncertainty and variability are explicitly included in the risk characterization models. The probabilistic models are designed to provide an estimate of the expected distribution of body burdens in a particular fish species, given a set of starting sediment and water concentrations. This is done with a Monte Carlo analysis. which consists of the following steps:

1. Define input parameter distributions
2. Randomly sample from these distributions
3. Perform repeated model simulations
4. Analyze the output.

REFERENCES

Ankley, G.T., K. Lodge, D.J. Call, M.D. Balcer, L.T. Brooke, P.M. Cook, and R.G. Kreis, Jr. 1992. Integrated assessment of contaminated sediments in the Lower Fox River and Green Bay, Wisconsin. *Ecotoxicol. Environ. Safety* 23:46-63.

Barron, M.G. 1990. Bioconcentration - Will water-borne organic chemicals accumulate in aquatic animals? *Environ. Sci. Technol.*, Vol. 24, 11:1612-1618.

Eastern Research Group, Inc. (ERG). 1998. Report from the Workshop on the Application of 2,3,7,8-TCDD Toxicity Equivalency Factors to Fish and Wildlife. First Draft, February 19, 1998.

Eisler, R. 1986. "Polychlorinated Biphenyl Hazards to Fish, Wildlife, and Invertebrates: A Synoptic Review." Fish and Wildlife Service, U.S. Dept. of the Interior. Biological Report 85(1.7). 72 pp.

Gobas, F.A.P.C. 1993. A model for predicting the bioaccumulation of hydrophobic organic chemicals in aquatic food-webs: application to Lake Ontario. *Ecol. Modelling* 69:1-17.

Hornshaw, T.C., R.J. Auerlich, and H.E. Johnson. 1983. Feeding Great Lakes fish to mink: effects on mink and accumulation and elimination of PCBs by mink. *J. Toxicol. Environ. Health.* 11: 933-946.

Ingersoll C.G., P.S. Haverland, E.L. Brunson, T.J. Canfield, F.J. Dwyer, C.E. Henke, N.E. Kemble, D.R. Mount and R.G. Fox. 1996. Calculation and evaluation of sediment effect concentrations for the amphipod *Hyalella azteca* and the midge *Chironomous riparius*. *J. Great Lakes Res.* 22(3):602-623.

Jones, D. S., G. W. Suter II and R.N. Hall. 1997. Toxicological Benchmarks for Screening Contaminants of Potential Concern for Effects on Sediment-Associated Biota: 1997 Revision. Lockheed Martin Systems, Inc. ES/ER/TM-95/R4.

Kaufmann, J.H. 1982. Raccoon and allies. In: Chapman, J.S.; Feldhamer, G.A., eds. Wild Mammals of North America. Johns Hopkins University Press, Baltimore, MD. pp. 567-585.

Kovats, Z.E. and J.J.H. Ciborowski. 1989. Aquatic Insect Adults as Indicators of Organochlorine Contamination. J. Great Lakes Res. 15(4):623-634.

Limno-Tech, Inc., Menzie-Cura & Associates, Inc., and the CADMUS Group, Inc. October 1996. Further Site Characterization and Analysis, Volume 2B- Preliminary Model Calibration Report Hudson River PCBs Reassessment RI/FS. Prepared for USEPA Region II.

Long, E.R., D.D. MacDonald. S.L. Smith, F.D. Calder 1995. Incidence of Adverse Biological Effects Within Ranges of Chemical Concentrations in Marine and Estuarine Sediments. *Environmental Management* 19(1):81-97.

McCarty. L.S. and D. Mackay. 1993. Enhancing Ecotoxicological Modeling and Assessment. *Environmental Science and Technology* 27(9):1719-1728.

Palmer, E.L. and H.S. Fowler. 1975. Fieldbook of Natural History. McGraw-Hill. New York, NY.

New York State Department of Environmental Conservation (NYSDEC). June 1998 (reissue of October 22, 1998). Division of Water Technical and Operational Guidance Series (1.1.1), Ambient Water Quality Standards and Guidance Values and Groundwater Effluent Limitations.

NYSDEC.March 1994. Natural Heritage Program Search performed for TAMS Consultants, Inc.

NYSDEC. November 1993. Technical Guidance for Screening Contaminated Sediments, New York State Department of Environmental Protection, Division of Fish and Wildlife, Division of Marine Resources.

New York State Department of Health (NYSDOH). 1997. Bat Control. http://pmep.cce.cornell.edu/recommends/vertebrates/vertbats.html.

New York State Department of State (NYSDOS), Division of Coastal Resources and Waterfront Revitalization and The Nature Conservancy. 1990. Hudson River Significant Tidal Habitats: A guide to the functions, values, and protection of the river's natural resources. 184 pp.

Persaud, D., R. Jaagumagi and A. Hayton. August 1993. Guidelines for the protection and management of aquatic sediment quality in Ontario. Ontario Ministry of the Environment and Energy.

Phillips, D.J.H. 1986. Use of organisms ro quantify PCBs in marine and estuarine environments. In: J.S. Waid (ed). PCBs and the Environment. Boca Raton, FL: CRC Press, Inc. pp. 127-182.

Sample, B.E., J.J. Beauchamp, R.A. Efroymson, G.W. Suter II, and T.L. Ashwood. February 1998. Development and Validation of Bioaccumulation Models for Small Mammals. Lockheed Martin Energy Systems, Inc. ES/ER/TM-219.

Sample, B.E., DM. Opresko, and G.W. Suter II. June 1996. Toxicological Benchmarks for Wildlife: 1996 Revision. Lockheed Martin Energy Systems, Inc. ES/ER/TM-96/R3.

Sawhney, B.L. 1986. Chemistry and properties of PCBs in relation to environmental effects. In: J.S. Waid (ed). PCBs and the Environment. Boca Raton, FL: CRC Press, Inc. pp. 147-65.

Smith S.L., D.D. MacDonald, K.A. Keenleyside, C.G. Ingersoll, and L.J. Field 1996. A preliminary evaluation of sediment quality assessment values for freshwater ecosystems. *J. Great Lakes Res.* 22(3):624-638.

Suter II, G.W. 1996. Risk Characterization for Ecological Risk Assessment of Contaminated Sites. Lockheed Martin Energy Systems, Inc. ES/ER/TM-200.

Suter, G.W. 1993. Ecological risk assessment. Boca Raton, FL: Lewis Publishers.
USEPA. Febraury 1998. Draft Report from the Workshop on the Application of 2,3,7,8-TCDD To Fish and Wildlife. Risk Assessment Forum, US Environmental Protection Agency, Washington, DC.

Suter II, G.W. and C.L.Tsao. 1996. Toxicological Benchmarks for Screening Potential Contaminants of Concern for Effects on Aquatic Biota: 1996 Revision. Lockheed Martin Energy Systems, Inc. ES/ER/TM-96/R2.

TAMS/Gradient/Tetra Tech. 1998. Phase 2 Report-Further Site Characterization and Analysis. Volume 2C-A Low Resolution Sediment Coring Report. Addendum to the Data Evaluation and Interpretation Report. Hudson River PCBs Reassessment RI/FS. July 1998. USEPA, Region II, New York.

TAMS/Gradient, 1993. Phase 2B Sampling and Analysis/Quality Assurance Project Plan, Volume 2 Benthic Invertebrate and Sediment Grab Sampling, Hudson River PCB Reassessment RI/FS, February 18, 1993. USEPA, Region II, New York.

TAMS/Gradient, 1992. Phase 2B Sampling and Analysis Plan/Quality Assurance Project Plan, Volume 1: Flow-Averaged Water-Column Sampling, Hudson River PCB Reassessment RI/FS. December 30, 1992. USEPA, Region II, New York.

TAMS/Gradient. August 1991. Phase 1 Report - Reveiw Copy Interim Characterization and Evaluation, Hudson River PCB Reassessment RI/FS. EPA Work Assignment No. 013-2N84.

Thomann, R.V., J.P. Connolly, and T.F. Parkerton. 1992. An Equilibrium Model of Organic Chemical Accumulation in Aquatic Food Webs with Sediment Interaction. *Environmental Toxicology and Chemistry*, Vol. 11, pp. 615-629.

USEPA. June 1997. Ecological Risk Assessment Guidance for Superfund: Process for Designing and Conducting Ecological Risk Assessments. USEPA, Environmental Response Team, Edison, NJ. EPA540-R-97-006. June 2, 1997.

USEPA. March 1997. Guiding Principles for Monte Carlo Analysis. EPA/630/R-97/001.

USEPA. February 1997. Special Report on Environmental Endocrine Disruption: An Effects Assessment and Analysis. EPA/630/R-96/012.

USEPA. January 1997. Priorities for Ecological Protection: An Initial List and Discussion Document for EPA. Discussion Document. EPA/600/S-97/002.

USEPA. September 1996a. PCBs: Cancer Dose-Response Assessment and Application to Environmental Mixtures. National Center for Environmental Assessment. EPA/600/P-96/001F.

USEPA. September 1996b. Summary Report for the Workshop on Monte Carlo Analysis. Washington, D.C. EPA/630/R-96-010.

USEPA. 1995. Great Lakes Water Quality Initiative Criteria Documents for the Protection of Wildlife. Office of Water, Washington, DC. EPA/820/B-95/008.

USEPA. May, 1994. Workshop on the Use of Available Data and Methods for Assessing Ecological Risks of 2,3,7,8-Tetrachloro-p-dibenzo-dioxin to Aquatic Life and Associated Wildlife. Washington, D.C. EPA/630/R-94-002.

USEPA. 1993. Wildlife Exposure Factors Handbook. Office of Research and Development, Washington, DC. EPA/600/R-93/187a.

USEPA. 1991. Ecological assessment of Superfund sites: an overview. ECO Update 1(2). Office of Solid Waste and Emergency Response, USEPA: Washington, DC.

US Fish and Wildlife Service (USFWS). March 1997. Polychlorinated Biphenyl Contamination of Tree Swallows in the Upper Hudson River Valley, New York. Effects on Breeding Biology and Implications for Other Bird Species. Preparers: A.L. Secord and J.P. McCarty.

Washington State Department of Ecology. 1997. Creation and Analysis of Freshwater Sediment Quality Values in Washington State, Publication No. 97-323a, July 1997.

TABLES

TABLE 1

ASSESSMENT AND MEASUREMENT ENDPOINTS

Assessment Endpoint	Specific Ecological Receptor ("Endpoint Species")	Measures	
		Exposure	Effect
Benthic community structure as food source for local fish and wildlife.	• Benthic macroinvertebrate community	• Ecological community indices (diversity, evenness, dominance) • PCB levels in sediments and surface water	• Estimated exceedance of Toxicity Reference Values (TRVs) • Differences in benthic community indices
Survival, growth and reproduction of benthic invertebrates.	• Benthic macroinvertebrate community	• PCB body burdens in individual species and community composites • PCB concentrations in sediments (biologically active zone) and water column	• Estimated exceedance of Toxicity Reference Values (TRVs) • Exceedance of water quality criteria (WQC)
Survival, growth, and reproduction of local forage fish populations.	• Spottail shiner • Pumpkinseed • Shortnose sturgeon	• Food chain modeling • PCB concentrations in food items (plankton, invertebrates) • Measured PCB body burdens • PCB concentrations in sediments and water column	• Estimated exceedance of TRVs • Estimated exceedance of population-level effect thresholds • Exceedance of WQC
Survival, growth, and reproduction of local piscivorous fish populations.	• Yellow perch • White perch • Largemouth bass • Smallmouth bass • Striped bass • Brown bullhead	• Food chain modeling • Measured PCB concentrations in food items (forage fish) • Measured PCB body burdens • PCB concentrations in sediments and water column	• Estimated exceedance of TRVs • Estimated exceedance of population-level effect thresholds • Exceedance of WQC

TABLE 1

ASSESSMENT AND MEASUREMENT ENDPOINTS

Assessment Endpoint	Specific Ecological Receptor ("Endpoint Species")	Measures	
		Exposure	Effect
Protection (i.e., survival and reproduction) of insectivorous, birds and mammals.	• Tree swallow • Little brown bat	• Food chain modeling • Measured PCB concentrations in food items (aquatic insects) • Measured PCB body burdens in tree swallows • Reproductive success and nestling growth and survival of tree swallows • PCB concentrations in sediments and water column	• Reduced reproductive and hatchling success • Estimated exceedance of TRVs • Estimated exceedance of population-level effect thresholds • Exceedance of WQC
Protection (i.e., survival and reproduction) of benthivorous birds.	• Mallard	• Food chain modeling • Measured PCB concentrations in food items (benthic invertebrates) • PCB concentrations in sediments and water column	• Estimated exceedance of TRVs • Estimated exceedance of population-level effect thresholds • Exceedance of WQC
Protection of piscivorous birds and mammals.	• Belted kingfisher • Great blue heron • Mink	• Food chain modeling • Measured PCB concentrations in food items (forage fish) • PCB concentrations in sediments and water column	• Estimated exceedance of TRVs • Estimated exceedance of population-level effect thresholds • Exceedance of WQC
Protection of omnivorous mammals	• Raccoon	• Food chain modeling • Measured PCB concentrations in food items (fish, invertebrates, etc.) • PCB concentrations in sediments and water column	• Estimated exceedance of TRVs • Estimated exceedance of population-level effect thresholds • Exceedance of WQC

TABLE 1

ASSESSMENT AND MEASUREMENT ENDPOINTS

Assessment Endpoint	Specific Ecological Receptor ("Endpoint Species")	Measures	
		Exposure	Effect
Protection of upper trophic level wildlife	• Bald eagle	• Food chain modeling (small mammals) • Measured PCB concentrations in food items (fish) • PCB concentrations in sediments and water column	• Estimated exceedance of TRVs • Estimated exceedance of population-level effect thresholds • Exceedance of WQC
Protection of endangered and threatened species	• Bald eagle • Shortnose sturgeon	• Food chain modeling • Measured PCB concentrations in food items (fish) • PCB concentrations in sediments and water column	• Estimated exceedance of TRVs • Estimated exceedance of population-lvel effect thresholds • Exceedance of WQC
Protection of significant habitats	• NOAA Estuarine Sanctuaries • Selected NYS Dept. of State significant habitats	• PCB concentrations in sediments and water column	• Exceedance of federal and state WQC

Notes: The best available TRVs based on protection of sensitive species or age-classes will be used to estimate effects levels.
Populations are defined based on ranges along the Hudson River and are not considered to be completely isolated groups.

TABLE 2

TROPHIC LEVELS, EXPOSURE PATHWAYS, AND FOOD SOURCES

Endpoint Species	Level	Exposure Pathways	Food Sources (based on preliminary estimates)
		Fish	
Benthic Invertebrates	1	• Direct contact with sediments • Direct contact with interstitial water • Direct contact with water (epibenthic and filter feeders)	Interstitial water; invertebrates can be scavengers, filter feeders, burrowers, clingers
Spottail Shiner	2	• Direct contact with water (respiration, dermal) • Food chain exposures (both water and sediment-based) • Direct contact with sediments	50% benthic invertebrates, 50% pelagic invertebrates
Pumpkinseed	2	• Direct contact with water (respiration, dermal) • Food chain exposures (both water and sediment-based) • Direct contact with sediments	80% pelagic invertebrates, 20% benthic invertebrates
White Perch	2-3	• Direct contact with water (respiration, dermal) • Food chain exposures (both water and sediment-based) • Direct contact with sediments	10-20% forage fish, 30-40% benthic invertebrates, 50-60% pelagic invertebrates
Yellow Perch	2-3	• Direct contact with water (respiration, dermal) • Food chain exposures (both water and sediment-based) • Direct contact with sediments	<10% forage fish, 20-30% benthic invertebrates, 60-80% pelagic invertebrates
Largemouth Bass	3	• Direct contact with water (respiration, dermal) • Food chain exposures (both water and sediment-based) • Direct contact with sediments	90% forage fish, 10% benthic invertebrates
Brown Bullhead	2	• Direct contact with water (respiration, dermal) • Food chain exposures (primarily sediment-based) • Direct contact with sediment	90% benthic invertebrates, <10% pelagic invertebrates or forage fish

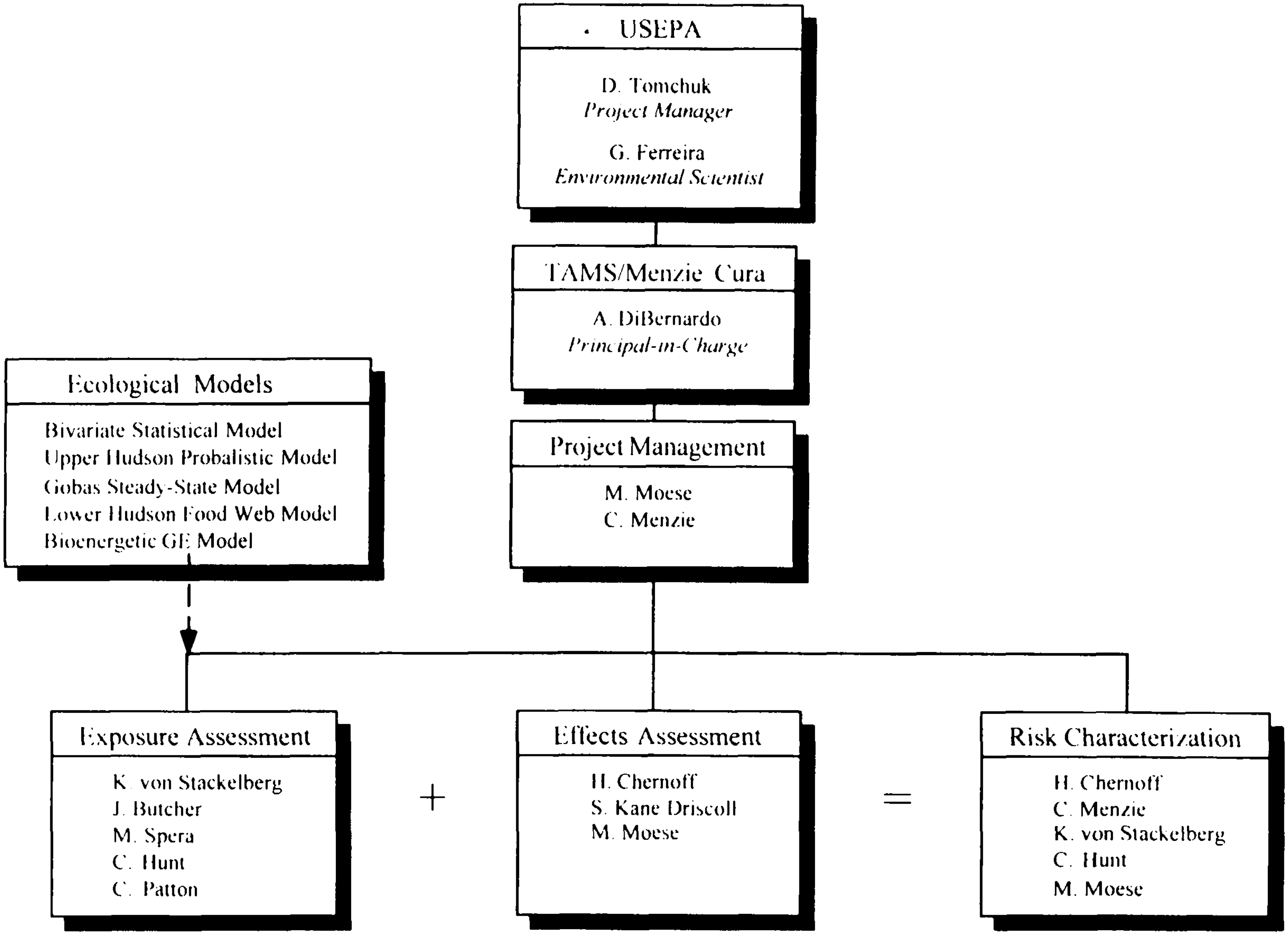
TABLE 2

TROPHIC LEVELS, EXPOSURE PATHWAYS, AND FOOD SOURCES

Endpoint Species	Level	Exposure Pathways	Food Sources (based on preliminary estimates)
Striped Bass	3	• Direct contact with water (respiration, dermal) • Food chain exposures (both water and sediment-based) • Direct contact with sediments	Predominantly forage fish
Shortnose Sturgeon	3	• Direct contact with water (respiration, dermal) • Food chain exposures (both water and sediment-based) • Direct contact with sediments	Predominantly forage fish
Birds			
Tree swallow	2	• Water ingestion • Food chain exposures	Insects, worms; note that exposure to PCBs may be from unmeasured sources
Mallard	2	• Water ingestion • Food chain exposures	vegetation, benthic invertebrates
Kingfisher	3	• Water ingestion • Food chain exposures	Invertebrates, forage fish
Great Blue Heron	3	• Water ingestion • Food chain exposures • Direct contact with sediments	Invertebrates, forage fish
Bald Eagle	4	• Ingestion of water • Food chain exposures	Forage fish, small mammals
Mammals			
Brown Bat	2	• Ingestion of water • Food chain: ingestion of emergent aquatic insects	Food source may originate in sediments and travel up the water column
Raccoon	3	• Ingestion of water • Food chain exposures • Direct contact with sediments	Forage fish, insects, invertebrates
Mink	4-5	• Ingestion of water • Food chain exposures • Direct contact with sediments	Forage fish, insects, invertebrates

FIGURES

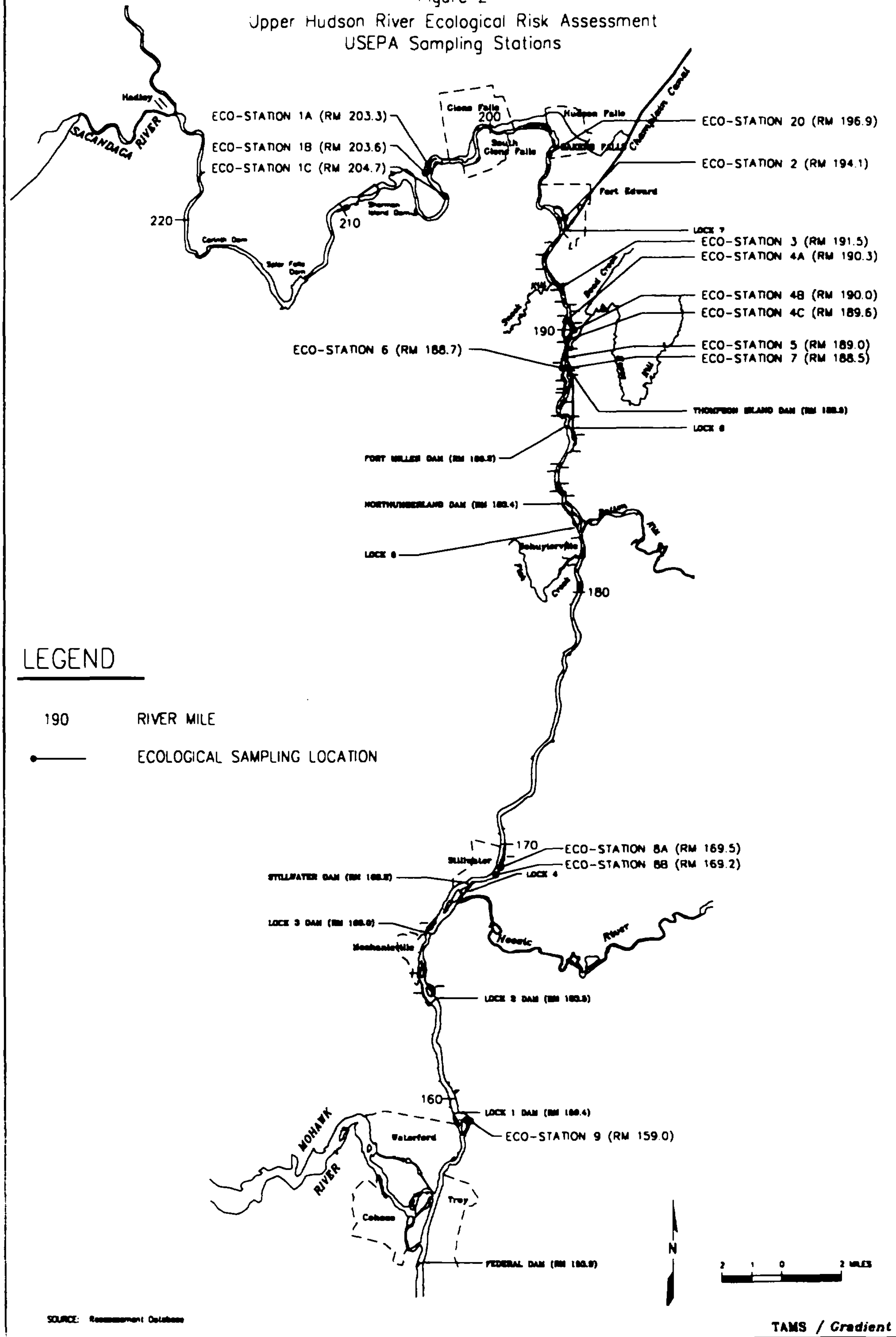
Figure 1
Organization Chart
Hudson River PCB Reassessment
Ecological Risk Assessment



Note:
1. TAMS Menzie-Cura Internal Reviewers: E. Garvey, J. Butcher

Filename: U:\PROJECT 33\HUD-CADD\ECOSOW\FIG2.dwg / Date: 4-27-98 / User: CH

Figure 2
Upper Hudson River Ecological Risk Assessment
USEPA Sampling Stations



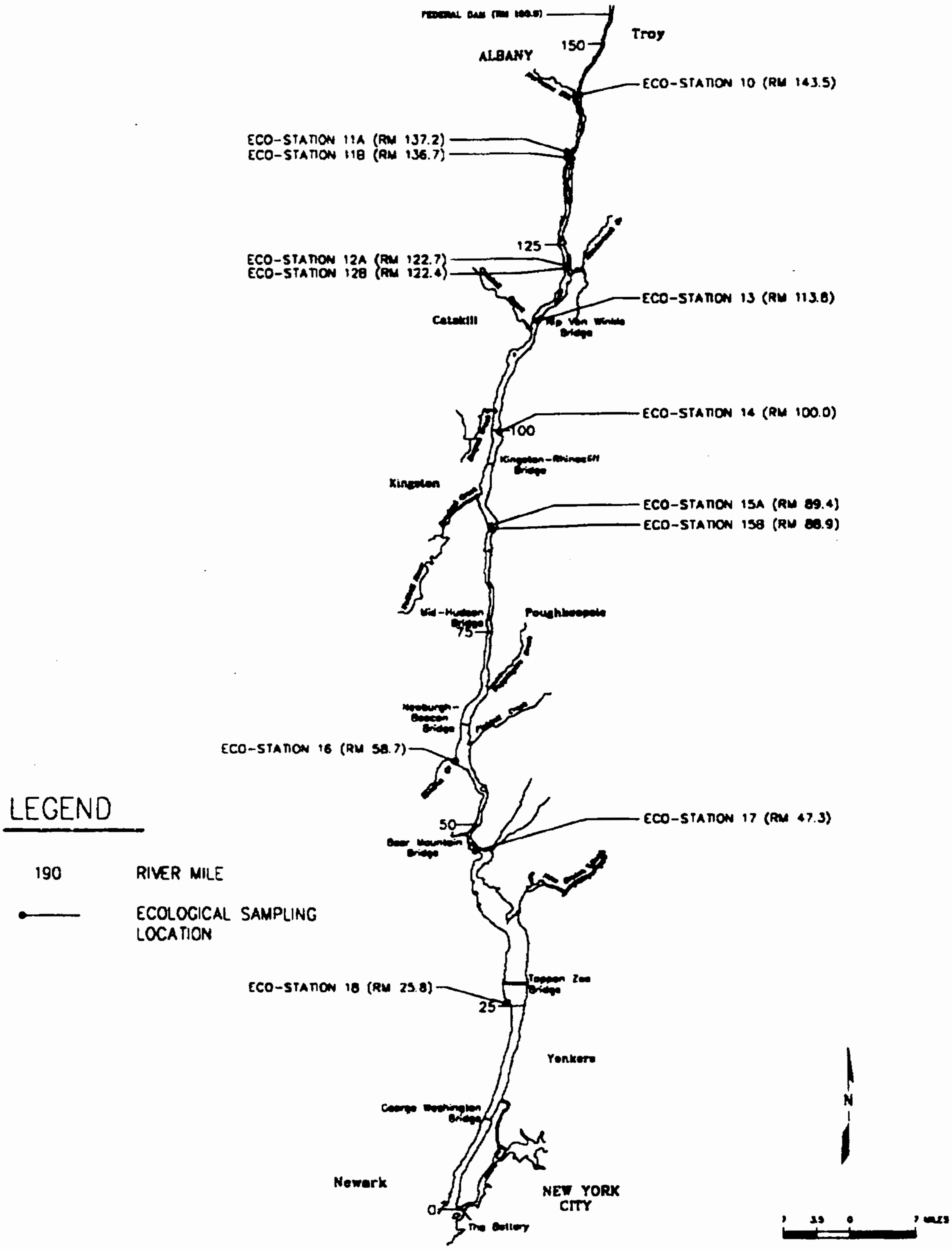
SOURCE: Reassessment Database

TAMS / Gradient

300629

Filename: U:\PROJECT.33\HUD-CADD\ECOSOW\FIG3.dwg / Date: 3/20/98 / User: CH

Figure 3
Lower Hudson River Ecological Risk Assessment
USEPA Sampling Stations



SOURCE: Reassessment Database

TAMS / *Gradient*

Figure 4
Eight-Step Ecological Risk Assessment Process for Superfund
Hudson River PCB Reassessment
Ecological Risk Assessment

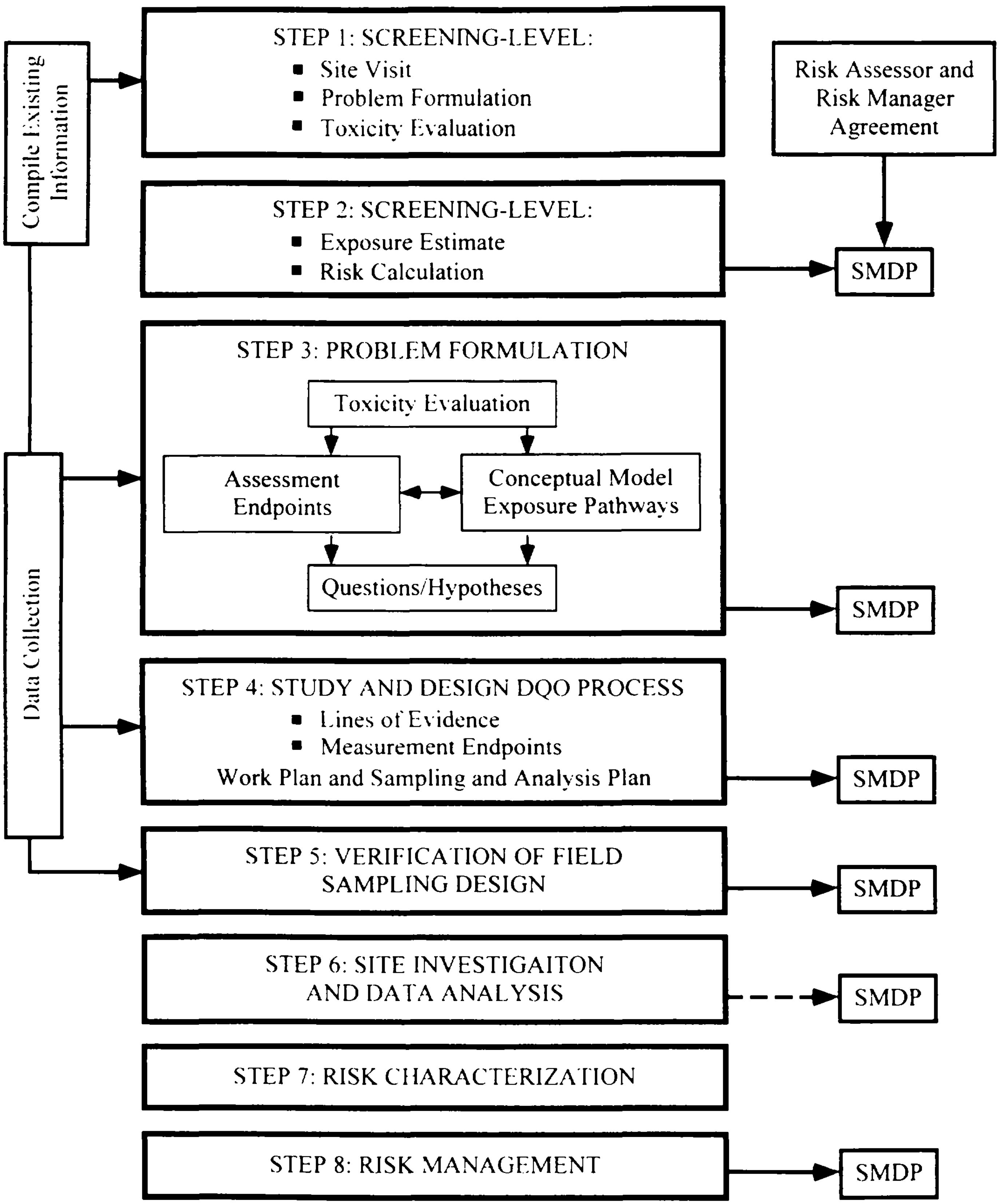
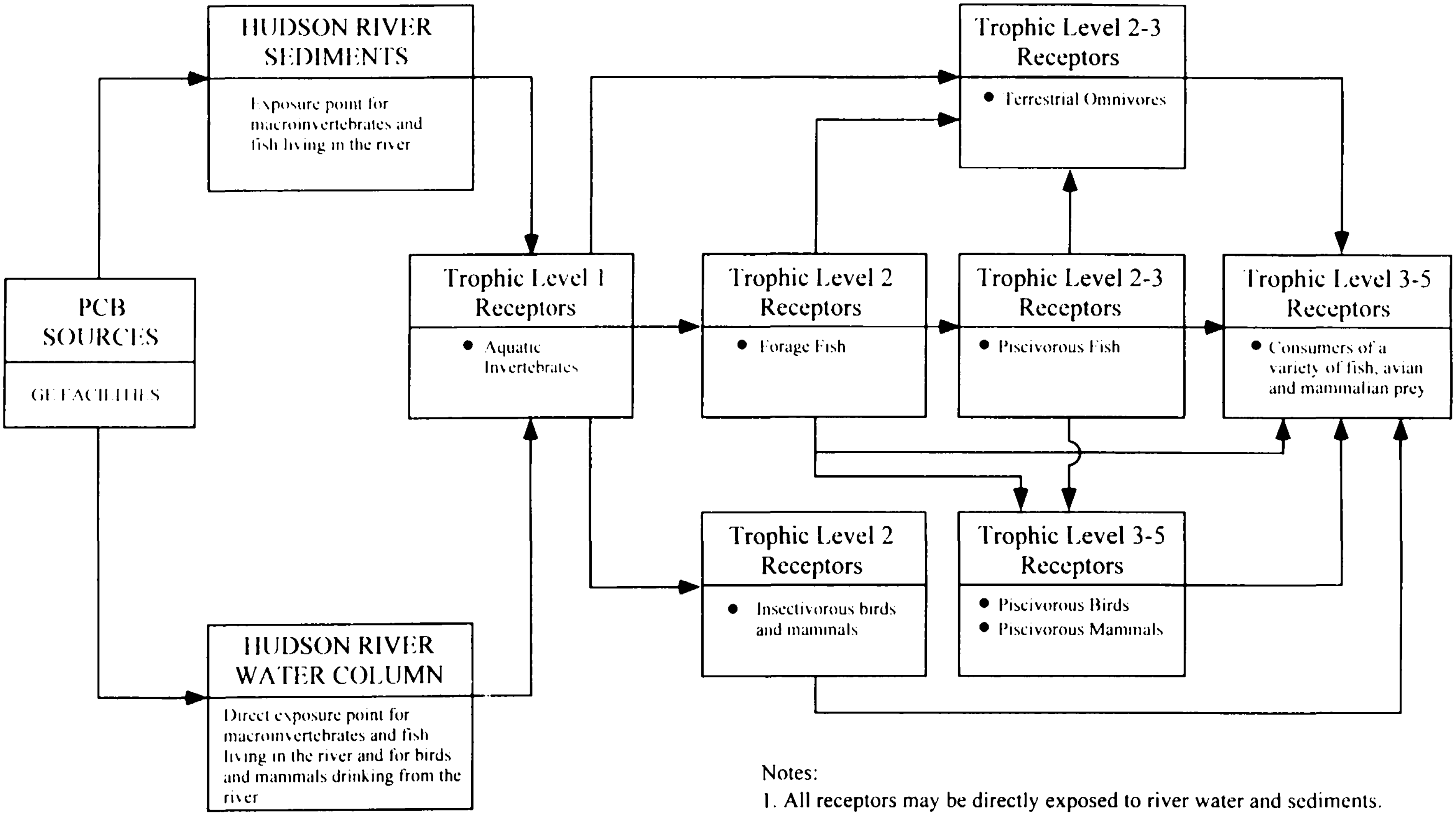


Figure 5
Hudson River PCB Reassessment
Conceptual Model Diagram



Notes:
1. All receptors may be directly exposed to river water and sediments.
2. Trophic levels are provided as a general guide to bioaccumulation potential, but vary according to species and food availability.

APPENDIX A

APPENDIX A

Modeling Approaches

The goal of the modeling effort is to develop a framework for relating body burdens of PCBs in fish to exposure concentrations in Hudson River water and sediments. This framework is used to understand historical and current relationships as well as to predict fish body burdens for future conditions. Estimates of PCB body burdens in fish are intended to be used for the ecological risk assessment and aid in decision making regarding options for addressing PCB-contaminated sediments in the upper Hudson.

The objectives of the body burden modeling effort are based on discussions with the investigators responsible for the ecological risk assessment and with the fate and transport modeling team. Because PCB analytical protocols have varied over time, the framework needs to account for historical as well as current data to the extent possible. Accordingly, the framework is structured to meet the following objectives:

- Relate historical body burden data (as PCB Aroclors and Aroclor totals) to exposure concentrations in water and sediments;
- Relate current and future body burdens (as PCB Aroclors, totals, and individual congeners)to exposure concentrations in water and sediments; and
- Provide estimates in a form that can be used for ecological risk assessment.

To meet these objectives, two modeling approaches have been developed to relate body burdens to water and sediment concentrations. One – used with the historical PCB Aroclor data base – is referred to as the Bivariate Statistical Model. The other – derived using historical and Phase 2 USEPA data – is referred to as the Probabilistic Bioaccumulation Food Chain Model. In each case, the model relates PCB exposure concentrations in water and sediments to body burdens. The major difference between the two approaches is that the Bivariate Statistical Model uses available time series data to develop statistical relationships between concentrations in water and sediments and those in fish, while the Probabilistic Bioaccumulation Food Chain Model relies upon feeding relationships to link body burdens to water and/or sediments.

The two approaches complement one another. Each utilizes derived Bioaccumulation Factors (BAFs). The agreement between these and the resultant estimates of body burdens provide a check on the two approaches. It is anticipated that there will be some modeling applications for which the Bivariate Statistical Model is the better tool and other applications where the Probabilistic Bioaccumulation Model will provide the desired information. The probabilistic bioaccumulation model explicitly incorporates feeding preference data and uncertainty and variability information.

A third modeling approach will be used to provide independent verification for the results of the bivariate and probabilistic models. This will be done using a steady state bioaccumulation model called the Gobas Model (Gobas, 1993). This model combines the toxicokinetics of chemical uptake, elimination, and bioaccumulation in individual organisms and the trophodynamics of food-webs to estimate chemical concentrations at different trophic levels under steady state conditions. The model incorporates multiple feeding interactions, including benthic and pelagic food chains, and has been used for PCBs in Lake Ontario and western Lake Erie.

The Bivariate Statistical, Probabilistic Bioaccumulation, and the Gobas Models share a common theoretical basis, including:

- PCB body burdens in fish are related ultimately to exposure concentrations in water and/or sediments;
- PCBs in the water column and sediments are not necessarily in equilibrium with each other;
- Within the water and sediment compartments, an equilibrium or *quasi*-steady state condition exists at temporal scales on the order of a year, and spatial scales on the order of a river segment; and

- Fish body burdens are in *quasi*-steady state with the water and/or sediment at time scales on the order of one or more years.

PCB concentrations measured in biota are assumed to be in steady state with PCBs in the environment for the development of BAFs, and thus can be related by linear coefficients or bioaccumulation factors similar to partitioning coefficients.

A steady state condition is usually considered to hold within a given year; thus, the BAF approach represents temporal changes only annually. The simplest approach considers that biota and *all* environmental compartments are in equilibrium with one another, in which case the concentration in any medium can be predicted from the concentration in any other medium. The BAF method is readily modified to address situations in which disequilibrium exists at steady state between different environmental compartments.

The three modeling approaches each use existing data differently. Agreement between the expected values from each of the models will provide a degree of validation for each of them. The Probabilistic Bioaccumulation Model is specifically designed to provide probability distributions on a body burden basis for fish, while the Bivariate and the Gobas Models provide central tendency estimates of bioaccumulation under specific exposure conditions.

General Form of the Probabilistic Bioaccumulation Model

The first step in developing the probabilistic bioaccumulation model is to characterize the observed body burden data in each of the fish species using a geometric mean (GM) and geometric standard deviation (GSD). The Hudson River database is used to estimate the GM and GSD for benthic invertebrates. The NYSDOH multiplate data provides the basis for constructing mathematical relationships between whole water and water column invertebrates. This relationship is used to predict a GM and GSD for the pelagic species.

An extensive literature search as well as qualitative data from the Hudson River provides estimates of species-specific dietary composition in the form of feeding preferences expressed as average fraction of prey species consumed (*i.e.*, pelagic invertebrates, benthic invertebrates, and forage fish). The feeding preference information combined with the concentration distributions results in a GM and GSD for the diet of each fish species. The dietary BAF is then defined as the observed PCB body burden distribution divided by the expected distribution in the diet of that species.

The biota:sediment bioaccumulation factor (BSAF) predicts expected benthic invertebrate concentrations using the Hudson River Phase 2 dataset. This relationship is based on a dynamic equilibrium (steady state) assumption between benthic invertebrates and the sediment in which they reside. An individual lipid-normalized benthic invertebrate concentration is divided by an average (consisting of five samples) TOC-normalized sediment concentration to derive a distribution of BSAF for the Hudson River. Statistical analyses evaluate patterns by species and/or location, particularly for those species which may experience additional exposure via overlying water. The final distribution is characterized by a GM and GSD. To predict benthic invertebrate concentrations, a new TOC-normalized sediment concentration is used together with the BSAF distribution.

The pelagic invertebrate bioaccumulation factor (PBAF) predicts expected pelagic invertebrate concentrations using mathematical relationships derived from the NYSDOH multiplate data from the early 1980s (Novak, 198x). These data showed that typical water column invertebrates quickly achieve steady state (within 96 hours) with the surrounding water. An individual measured lipid-normalized multiplate concentration is divided by an appropriate total water concentration (obtained from concurrent United States Geological Survey data) to derive a distribution of PBAF. This distribution is characterized by a GM and GSD. A new total water concentration together with the PBAF distribution predicts expected concentrations in pelagic invertebrates.

The forage fish diet consists of benthic and pelagic invertebrates. The exact proportion depends on the individual species (spottail shiner and pumpkinseed). The dietary forage fish bioaccumulation factor (FFBAF) is calculated by dividing the observed forage fish concentration distribution by the concentration distribution in the diet (*i.e.*, proportion of benthic invertebrates from the Phase 2 dataset and predicted pelagic invertebrate concentration from the NYSDOH dataset). Piscivorous fish bioaccumulation factors (PFBAF) are derived in the same way, except that a portion of the piscivorous fish diet will also consist of forage fish.

This Page Was Intentionally Left Blank For Pagination Purposes.

APPENDIX B

APPENDIX B

Quantitation of PCBs and Lipid Content

This Appendix will address issues related to quantitation of PCBs and lipid content of biological specimens. Analysis of biological specimens involves many complexities, associated with both sample extraction and analytical procedures. When data collected and analyzed under a variety of protocols are combined, it is essential to ensure a common basis for analysis. Systematic differences between samples may result in a spurious attribution of trends or responses.

A consistent quantitation basis is most important when data obtained under highly different sampling and analytical protocols are combined. It is not, for instance, an issue for analysis of the Phase 2 data, which were collected using consistent methods and subjected to a single, documented analytical protocol. However, combining Phase 2 results and earlier data, as is required for the bivariate statistical analysis of bioaccumulation, can present a significant problem. For instance, the Phase 2 congener-based analytical methods can be expected to provide consistently different results than historic analyses using packed-column gas chromatography against Aroclor standards.

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.*, 1996). Several additional changes in analytical methodology occurred in the 199's: 1990–1992 results were analyzed by NYSDOH using a packed-column Webb and McCall method, while post–1992 results were analyzed by a contract lab using a limited number of quantitation peaks and a shift from an Aroclor 1016 standard (as used in the pre-1990 results) to Aroclor 1242 and Aroclor 1248 standards. All these 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. The results of these changes in PCB quantitation methodology in the NYSDEC 1990 results have not yet been analyzed.

Additional uncertainty in the interpretation of 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 that wet-weight PCB concentrations. Lipid-normalized PCB body burden is calculated as the reported wet-weight PCB concentration divided by lipid concentration. Unfortunately, any imprecision in the determination of lipid concentration will also result in imprecision in the calculation of lipid-normalized PCB body burden. Further, the propagation of uncertainty will be non-linear, as the lipid-normalized concentration involves division by the lipid content. Therefore, estimation of the uncertainty in lipid-based PCB concentrations must also include an analysis of the uncertainty in determination of lipid concentration. Interlaboratory comparisons conducted by NYSDEC in September 1992 showed an average variability between laboratories of ten percent in determining lipid content of biological specimens, with results from some pairs of laboratories showing a consistent relative bias.

The work proposed for this Appendix consists of comparative analyses to (1) determine, to the extent possible, a consistent quantitation basis for historical analyses, and (2) estimate uncertainties present in calculated lipid-normalized PCB body burdens. To complete this effort, the following activities are proposed:

1. PCB Quantitation
   - i. Document analytical methods used during the 1990's by Hale Creek and NYSDEC's contract laboratory.
   - ii. Obtain additional interlaboratory comparison results from NYSDEC, if available.
   - iii. Obtain peak-area results for PCB quantitation from each laboratory, to the extent available.
   - iv. Using the Phase 2 PCB congener analytical results as a basis. compare "what if" results for each PCB analytical method used in historical analyses.
   - v. Estimate analytical uncertainty in historical PCB quantitation methods.

vi. Propose correction factors to convert historical PCB analyses in biota to a consistent basis for comparison to Phase 2 results.

2. Lipid Quantitation

i. Document and compare extraction and analysis methodologies for determining percent lipid in historical and Phase 2 samples.

ii. Obtain additional interlaboratory comparison results, if available.

iii. Estimate uncertainty in reported lipid concentration results.

iv. Analyze propagated uncertainty in lipid-normalized PCB concentrations.

v. If analysis indicates consistent bias between lipid determination methods, propose appropriate correction factors.

This Page Was Intentionally Left Blank For Pagination Purposes.