602508

HUDSON RIVER PCBs REASSESSMENT RI/FS
RESPONSIVENESS SUMMARY FOR
VOLUME 2E - BASELINE ECOLOGICAL RISK ASSESSMENT

MARCH 2000

UNITED STATES • ENVIRONMENTAL PROTECTION AGENCY •

For

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

Book 1 of 1

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

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

March 24, 2000

To All Interested Parties:

The U.S. Environmental Protection Agency (USEPA) is pleased to release the Responsiveness Summary for the Ecological Risk Assessment (ERA), which is part of Phase 2 of the Reassessment Remedial Investigation/Feasibility Study for the Hudson River PCBs Superfund site. For complete coverage, the ERA and this Responsiveness Summary should be used together.

In the Responsiveness Summary, USEPA has responded to all significant comments received during the public comment period on the ERA. In addition, the Responsiveness Summary contains revised calculations of ecological risk based on the modified future concentrations of PCBs in sediment, water and fish presented in USEPA's January 2000 Revised Baseline Modeling Report. The Responsiveness Summary also contains a comparison of the revised calculations to those reported in the ERA. Importantly, the overall conclusions of the ERA regarding the risks to fish, birds, and mammals due to PCBs in the Hudson River remain unchanged.

The ERA is being peer reviewed by a panel of independent experts. The peer reviewers will discuss their comments on the ERA at a meeting that will be held on June 1 and 2, 2000 at the Holiday Inn in Saratoga Springs, New York. The Human Health Risk Assessment will be peer reviewed by a separate panel on May 30 and 31, 2000 at the same location. Observers are welcome and there will be limited time for observer comment.

If you need additional information regarding the Responsiveness Summary for the ERA, please contact Ann Rychlenski, the Community Relations Coordinator for this site, at (212) 637-3672.

Sincerely yours,

Richard L. Caspe, Director
Emergency and Remedial Response Division

HUDSON RIVER PCBs REASSESSMENT RI/FS
RESPONSIVENESS SUMMARY FOR
VOLUME 2E - BASELINE ECOLOGICAL RISK ASSESSMENT

MARCH 2000

Page

HUDSON RIVER PCBs REASSESSMENT RI/FS
RESPONSIVENESS SUMMARY FOR
VOLUME 2E - BASELINE ECOLOGICAL RISK ASSESSMENT

MARCH 2000

Page

304063

HUDSON RIVER PCBs REASSESSMENT RI/FS
RESPONSIVENESS SUMMARY FOR
VOLUME 2E - BASELINE ECOLOGICAL RISK ASSESSMENT

MARCH 2000

Page

Page

Page

Page

HUDSON RIVER PCBs REASSESSMENT RI/FS
RESPONSIVENESS SUMMARY FOR
VOLUME 2E - BASELINE ECOLOGICAL RISK ASSESSMENT

MARCH 2000

Page

Page

304069

HUDSON RIVER PCBs REASSESSMENT RI/FS
RESPONSIVENESS SUMMARY FOR
VOLUME 2E - BASELINE ECOLOGICAL RISK ASSESSMENT

MARCH 2000

Page

Page

HUDSON RIVER PCBs REASSESSMENT RI/FS
RESPONSIVENESS SUMMARY FOR
VOLUME 2E - BASELINE ECOLOGICAL RISK ASSESSMENT

MARCH 2000

Page

LIST OF TABLES:

Page

Page

Page

Page

Page

LIST OF FIGURES:

ACRONYMS

ATSDR	AGENCY FOR TOXIC SUBSTANCES AND DISEASE REGISTRY
CDI	CHRONIC DAILY INTAKE
CERCLA	COMPREHENSIVE ENVIRONMENTAL RESPONSE, COMPENSATION, AND LIABILITY ACT
CSF	CARCINOGENIC SLOPE FACTOR
EPC	EXPOSURE POINT CONCENTRATION
GE	GENERAL ELECTRIC
HI	HAZARD INDEX
HHRA	HUMAN HEALTH RISK ASSESSMENT
HHRASOW	HUMAN HELATH RISK ASSESSMENT SCOPE OF WORK
HQ	HAZARD QUOTIENT
NCP	NATIONAL OIL AND HAZARDOUS SUBSTANCES POLLUTION CONTINGENCY PLAN
NPL	NATIONAL PRIORITIES LIST
NYSDEC	NEW YORK STATE DEPARTMENT OF ENVIRONMENTAL CONSERVATION
NYSDOH	NEW YORK STATE DEPARTMENT OF HEALTH
PCB	POLYCHLORINATED BIPHENYL
RFD	REFERENCES DOSE
RI	REMEDIAL INVESTIGATION
RI/FS	REMEDIAL INVESTIGATION/FEASIBILITY STUDY
ROD	RECORD OF DECISION
RM	RIVER MILE
RI/FS	REMEDIAL INVESTIGATION/FEASIBILITY STUDY
SARA	SUPERFUND AMENDMENTS AND REAUTHORIZATION ACT OF 1986
TCDD	2,3,7,8-TETRACHLORODIBENZO-P-DIOXIN
TEF	TOXICITY EQUIVALENCY FACTOR
TSCA	TOXIC SUBSTANCES CONTROL ACT
UCL	UPPER CONFIDENCE LIMIT
USEPA	UNITED STATES ENVIRONMENTAL PROTECTION AGENCY

HUDSON RIVER PCBs REASSESSMENT RI/FS
RESPONSIVENESS SUMMARY FOR
VOLUME 2E - BASELINE ECOLOGICAL RISK ASSESSMENT

MARCH 2000

I. INTRODUCTION AND COMMENT DIRECTORY

1.1 INTRODUCTION

The U.S. Environmental Protection Agency (USEPA) has prepared this Responsiveness Summary to address comments received during the public comment period on the Phase 2 Ecological Risk Assessment (ERA) for the Hudson River PCBs Reassessment Remedial Investigation/Feasibility Study (RI/FS), dated August 1999.

For the Hudson River PCBs Reassessment RI/FS, USEPA has established a Community Interaction Program (CIP) to elicit on-going feedback through regular meetings and discussion and to facilitate review of and comment upon work plans and reports prepared during all phases of this Reassessment RI/FS.

Because of the large number of CIP participants and associated costs of reproduction, the ERA is incorporated by reference and is not reproduced herein. No revised ERA will be published. The comment responses and revisions noted herein are considered to amend the ERA. For complete coverage, the ERA and this Responsiveness Summary must be used together.

The first part of this four-part Responsiveness Summary is entitled, "Introduction, Commenting Process, Organization of Comments and Responsiveness Summary and Comment Directory." It describes the ERA review and commenting process, explains the organization and format of comments and responses, and contains a comment directory.

The second part, entitled, "Responses to Comments on the ERA," contains USEPA's responses to all significant comments received on the August 1999 ERA in Section II. Responses are grouped according to the section number of the ERA to which they refer. For example, responses to comments on Section 2.2 of the ERA are found in Section 2.2 of the Responsiveness Summary. Additional information about how to locate responses to comments is contained in the Comment Directory 4.2.

The third section of this part contains the revisions to the risk results based on the revision to the baseline modeling report.

The fourth part, entitled, "Comments on the Phase 2 ERA," contains copies of the comments submitted to USEPA. Not all references provided by the commentors are reproduced in this document. The comments are identified by commentor and comment number, as further explained in the Comment Directory.

2. Commenting Process

This section documents and explains the commenting process and the organization of comments and responses in this document. Readers interested in finding responses to their comments may skip this section and go directly to the tab labeled "Comment Directory."

2.1 Distribution of ERA

The ERA, issued in August 1999, was distributed to federal and state agencies and officials, participants in the CIP and General Electric Company (GE), as shown in Table 1. Distribution was made to approximately 100 agencies, groups, and individuals. Copies of the ERA were also made available for public review in thirteen Information Repositories, as shown in Table 2 and on the USEPA Region 2 internet webpage, entitled "Hudson River PCBs Superfund Site Reassessment," at www.epa.gov/hudson.

2.2 Review Period and Public Availability Meetings

Review of and comment on the August 1999 ERA occurred from August 4, 1999 to September 7, 1999. On August 4 and 5, 1999, USEPA held Joint Liaison Group Meetings open to the public in Albany and Poughkeepsie, NY, respectively. Subsequently, on August 18, 1999, USEPA held public availability sessions at the Holiday Inn Express in Latham, New York. These meetings were conducted in accordance with USEPA's *Community Relations in Superfund: Handbook, Interim Version (1988).* Minutes of the Joint Liaison Group meetings will be available for public review at the Information Repositories listed in Table 2.

As stated in USEPA's letter transmitting the ERA, all citizens were urged to participate in the Reassessment process and to join one of the Liaison Groups formed as part of the CIP.

2.3 Receipt of Comments

Comments on the ERA were received in two ways: letters submitted to USEPA and oral statements made at the August 4 and 5, 1999 Joint Liason Group meetings. USEPA's responses to oral statements made at the Joint Liaison Group meetings are provided in the meeting minutes.

All significant comments received on the ERA are addressed in this Responsiveness Summary. Comments were received from eight commentors. Total comments numbered over 150.

TABLE 1
DISTRIBUTION OF ERA

HUDSON RIVER PCBs OVERSIGHT COMMITTEE MEMBERS

- USEPA ERRD Deputy Division Director (Chair)
- USEPA Project Managers
- USEPA Community Relations Coordinator, Chair of the Steering Committee
- NYSDEC Division of Hazardous Waste Management representative
- NYSDEC Division of Construction Management representative
- National Oceanic and Atmospheric Administration (NOAA) representative
- Agency for Toxic Substances and Disease Registry (ATSDR) representative
- US Army Corps of Engineers representative
- New York State Thruway Authority (Department of Canals) representative
- USDOI (US Fish and Wildlife Service) representative
- NYSDOH representative
- GE representative
- Liaison Group Chairpeople
- Scientific and Technical Committee representative

SCIENTIFIC AND TECHNICAL COMMITTEE MEMBERS

The members of the Science and Technical Committee (STC) are scientists and technical researchers who provide technical input by evaluating the scientific data collected on the Reassessment RI/FS, identifying additional sources of information and on-going research relevant to the Reassessment RI/FS, and commenting on USEPA documents. Members of the STC are familiar with the site, PCBs, modeling, toxicology, and other relevant disciplines.

- Dr. Daniel Abramowicz
- Dr. Donald Aulenbach
- Dr. James Bonner, Texas A&M University
- Dr. Richard Bopp, Rensselaer Polytechnic Institute
- Dr. Brian Bush
- Dr. Lenore Clesceri, Rensselaer Polytechnic Institute
- Mr. Kenneth Darmer
- Mr. John Davis, New York State Dept. of Law
- Dr. Robert Dexter, EVS Consultants, Inc.
- Dr. Kevin Farley, Manhattan College
- Mr. Jay Field, National Oceanic and Atmospheric Administration
- Dr. Ken Pearsall, U.S. Geological Survey
- Dr. John Herbich, Texas A&M University

- Dr. Behrus Jahan-Parwar, SUNY - Albany
- Dr. Nancy Kim, New York State Dept. of Health
- Dr. William Nicholson, Mt. Sinai Medical Center
- Dr. George Putman, SUNY - Albany
- Dr. G-Yull Rhee, New York State Dept. of Health
- Dr. Francis Reilly, Jr., The Reilly Group
- Ms. Anne Secord, U.S. Fish and Wildlife Service
- Dr. Ronald Sloan, New York State Dept. of Environmental Conservation

STEERING COMMITTEE MEMBERS

- USEPA Community Relations Coordinator (Chair)
- Governmental Liaison Group Chair and two Co-chairs
- Citizen Liaison Group Chair and two Co-chairs
- Agricultural Liaison Group Chair and two Co-chairs
- Environmental Liaison Group Chair and two Co-chairs
- USEPA Project Managers
- NYSDEC Technical representative
- NYSDEC Community Affairs representative

FEDERAL AND STATE REPRESENTATIVES

Copies of the ERA were sent to relevant federal and state representatives who have been involved with this project. These include, in part, the following:

- The Hon. Daniel P. Moynihan
- The Hon. Charles E. Schumer
- The Hon. John E. Sweeney
- The Hon. Nita Lowey
- The Hon. Maurice Hinchey
- The Hon. Ronald B. Stafford
- The Hon. Michael McNulty
- The Hon. Sue Kelly
- The Hon. Benjamin Gilman
- The Hon. Richard Brodsky
- The Hon. Bobby D'Andrea

16 INFORMATION REPOSITORIES (*see* Table 2).

TABLE 2
INFORMATION REPOSITORIES

Adriance Memorial Library
93 Market Street
Poughkeepsie, NY 12601

Catskill Public Library
1 Franklin Street
Catskill, NY 12414

^ Cornell Cooperative Extension
Sea Grant Office
74 John Street
Kingston, NY 12401

Crandall Library
City Park
Glens Falls, NY 12801

County Clerk's Office
Washington County Office Building
Upper Broadway
Fort Edward, NY 12828

* ^ Marist College Library
Marist College
290 North Road
Poughkeepsie, NY 12601

* New York State Library
CEC Empire State Plaza
Albany, NY 12230

New York State Department
of Environmental Conservation
Division of Hazardous Waste Remediation
50 Wolf Road, Room 212
Albany, NY 12233

* ^ R. G. Folsom Library
Rensselaer Polytechnic Institute
Troy, NY 12180-3590

Saratoga County EMC
50 West High Street
Ballston Spa, NY 12020

* Saratoga Springs Public Library
49 Henry Street
Saratoga Springs, NY 12866

* ^ SUNY at Albany Library
1400 Washington Avenue
Albany, NY 12222

* ^ Sojourner Truth Library
SUNY at New Paltz
New Paltz, NY 12561

Troy Public Library
100 Second Street
Troy, NY 12180

U. S. Environmental Protection Agency
290 Broadway
New York, NY 10007

White Plains Public Library
100 Martine Avenue
White Plains, NY 12601

* ***Repositories with Database Report CD-ROM (as of 10/98)***

^ ***Repositories without Project Documents Binder (as of 10/98)***

2.4 Distribution of Responsiveness Summary

This Responsiveness Summary will be distributed to the Liaison Chairs and Co-Chairs and interested public officials. This Responsiveness Summary has also been placed in the seventeen Information Repositories and is part of the Administrative Record.

3. Organization of ERA Comments and Responses to Comments

3.1 Identification of Comments

Each submission commenting on the ERA was assigned a letter "E" and one of the following letter codes:

F - Federal agencies and officials;
S - State agencies and officials;
L - Local agencies and officials;
P - Public Interest Groups and Individuals; and
G - GE.

The letter codes were assigned for the convenience of readers and to assist in the organization of this document. Priority or special treatment was neither intended nor given in the responses to comments.

Once a letter code was assigned, each submission was then assigned a number, in the order that it was received and processed such as EF-1, EF-2 and so on. Each different comment within a submission was assigned a separate sub-number. Thus, if a federal agency submission contained three different comments, they are designated as EF-1.1, EF-1.2 and EF-1.3. Comment letters are reprinted in the fourth section of this document.

The alphanumeric code associated with each reprinted written submission is marked at the top right corner of the first page of the comment letter. The sub-numbers designating individual comments are marked in the margin. Comment submissions are reprinted in numerical order by letter code in the following order: EF, ES, EL, EP and EG.

3.2 Location of Responses to Comments

The Comment Directory, following this text, contains a complete listing of all commentors and comments. This directory allows readers to find responses to comments and provides several items of information.

- The first column lists the names of commentors. Comments are grouped first by: EF (Federal), ES (State), EL (Local), EP (Public Interest Group or Individual) or EG (GE).
- The second column identifies the alphanumeric comment code, e.g., EF-1.1, assigned to each comment.

- The third column identifies the location of the response by the ERA Section number. For example, comments raised in Section 2.1 of the ERA can be found in the corresponding Section 2.1 of the Responses, following the third tab of this document.
- The fourth, fifth and sixth columns list key words that describe the subject matter of each comment. Readers will find these key works helpful as a means to identify subjects of interest and related comments.

Responses are grouped and consolidated by section number in order of the ERA so that all responses to related comments appear together for the convenience of the reader interested in responses to related or similar comments.

4.0 COMMENT DIRECTORY

4.1 Guide to Comment Directory

This section contains a diagram illustrating how to find responses to comments. The Comment Directory follows. As stated in the Introduction, this document does not reproduce the ERA. Readers are urged to utilize this Responsiveness Summary in conjunction with the ERA.

4.2 Comment Directory

STEP 1	STEP 2	STEP 3
Find the commentor or the key words words of interest in the Comment Directory.	Obtain the alphanumeric comment codes and the corresponding ERA Section.	Find the responses following the Responses tab. See the Table of Contents to locate the page of the Responsiveness Summary for the ERA Section.
Key to Comment Codes:		
Comment codes are in this format EX-a.b X=Commentor Group (F=Federal, S=State, L=Local, P = Public Interest, G=General Electric) a=Numbered letter containing comments b=Numbered comment		

Example:

COMMENT RESPONSE ASSIGNMENT FOR THE ERA

AGENCY/ Name	COMMENT CODE	REPORT SECTION	KEY WORDS 1	KEY WORDS 2	KEY WORDS 3
NOAA /Rosman	EF-1.1	General	Fate/Transport	Bioaccumulation	BMR

THIS PAGE LEFT BLANK INTENTIONALLY

Comment Directory

304091

4.2 Comment Directory For the ERA

AGENCY/ NAME	COMMENT CODE	REPORT SECTION	KEYWORDS 1	2	3
NOAA/Rosman	EF-1.1	General	Fate and Transport	Bioaccumulation	Baseline Modeling Report
NOAA/Rosman	EF-1.2	General	Fate and Transport	Modeling Parameters	Model Uncertainty
NOAA/Rosman	EF-1.3	General	Food Chain Modeling	Fish Concentrations	Model Prediction
NOAA/Rosman	EF-1.4	3.0	Water Column and Sediment Data	Nearshore Areas	Food Web Pathways
NOAA/Rosman	EF-1.5	4.0	TRV	Fish	Risk
NOAA/Rosman	EF-1.6	4.0	TRV	Study Selection	
NOAA/Rosman	EF-1.7	2.3.2	Floodplains	Risks to Receptors	
NOAA/Rosman	EF-1.8	Exec Sum	Benthic Invertebrates	Risk Characterization	
NOAA/Rosman	EF-1.9	2.5	95th Percentile	PCB	TEQ
NOAA/Rosman	EF-1.10	2.5	Measurement Endpoint	Fish TEQ	Risks to Fish,Birds, and Mammals
NOAA/Rosman	EF-1.11	2.6.5	Peregrine Falcon	Endangered Species List	
NOAA/Rosman	EF-1.12	3.1.2	TEF	Concentration	Factor
NOAA/Rosman	EF-1.13	3.2.1	Data Weighted	Nearshore Areas	Mid Channel
NOAA/Rosman	EF-1.14	3.2.2	Organic Carbon Normalization	Sediment Concentrations	
NOAA/Rosman	EF-1.15	3.2.2	Sediment Data	TIP Variability	Sample Variability
NOAA/Rosman	EF-1.16	3.2.3	Sampling Locations	Deposition	Flow Dynamics
NOAA/Rosman	EF-1.17	3.2.4	Fish Data	Analytical Methods	Systemic Differences
NOAA/Rosman	EF-1.18	3.2.4	TEQ Concentrations	Pumpkinseed	NOAEL/LOAEL
NOAA/Rosman	EF-1.19	3.3.1	Upstream Boundary	Protective	Water Quality Criteria
NOAA/Rosman	EF-1.20	3.4.2	Fish Exposure Pathway	Nearshore Habitats	Seasonal Changes
NOAA/Rosman	EF-1.21	3.4.2	DEC Fish Data	Forage Fish	
NOAA/Rosman	EF-1.22	3.4.3.3	Fish Size	Feeding Strategies	
NOAA/Rosman	EF-1.23	3.4.3.3	Phytoplankton	Bioaccumulation	
NOAA/Rosman	EF-1.24	3.4.3.3	Bald Eagles	Winter Diet	Dietary Exposure
NOAA/Rosman	EF-1.25	3.4.4.3	Fish Size	Feeding Strategies	Uncertainty
NOAA/Rosman	EF-1.26	3.4.3.6	Exposure Data	Mammals	Space Domain
NOAA/Rosman	EF-1.27	3.5.2	Text Edit		
NOAA/Rosman	EF-1.28	3.5.2	Largemouth Bass	Juveniles	Forage Fish
NOAA/Rosman	EF-1.29	3.5.3	Feeding Guild	Piscivores	Perch
NOAA/Rosman	EF-1.30	3.5.3	Fish Body Burdens	Decline	Sediment
NOAA/Rosman	EF-1.31	3.5.3	Water Column	Congener Concentrations	Distance from GE
NOAA/Rosman	EF-1.32	3.5.3	New York Area Discharges	Upriver Transport	PCB Composition
NOAA/Rosman	EF-1.33	3.5.3	Benthic Invertebrates	Lower Hudson	NYC Inputs
NOAA/Rosman	EF-1.34	4.2	PCB Effects	Fish	Reproduction and Development
NOAA/Rosman	EF-1.35	4.1.3	PCB Effects	Congener Concentrations	Underestimate Risk
NOAA/Rosman	EF-1.36	4.2	PCB Effects	Lethality, Growth, Reproduction	Other Effects

4.2 Comment Directory For the ERA

AGENCY/ NAME	COMMENT CODE	REPORT SECTION	KEYWORDS 1	2	3
NOAA/Rosman	EF-1.37	4.2.2.1	Sediment Guidelines	PCB Concentrations	TCDD Guidelines
NOAA/Rosman	EF-1.38	4.2.3	PCB Partitioning	Lipid Phase	
NOAA/Rosman	EF-1.39	4.2.3	TRV	PCB Total Body Burden	Risk Estimate
NOAA/Rosman	EF-1.40	4.2.3	PCB Concentrations	Non-Salamonid	Embryo
NOAA/Rosman	EF-1.41	4.2.3.5	NOAEL	White Perch	Yellow Perch
NOAA/Rosman	EF-1.42	4.2.3.5	White Perch	TRV	
NOAA/Rosman	EF-1.43	5.1.3.1	NYDEC WQC	Wildlife	
NOAA/Rosman	EF-1.44	5.2.1.1	Lower Hudson	Spottail Shinner	Other Species
NOAA/Rosman	EF-1.45	5.2.1	Modeled and Measured Spottail Shinner	Reproductive Effects	TRVs
NOAA/Rosman	EF-1.46	5.2.1	Fish Concentrations	Predicted	Measured vs. Modeled
NOAA/Rosman	EF-1.47	5.2.1.4	Table Edit		
NOAA/Rosman	EF-1.48	5.2.1.5	Brown Bullhead	TEQ	
NOAA/Rosman	EF-1.49	5.2.1.6	Lab vs Field TRVs	White Perch	Yellow Perch
NOAA/Rosman	EF-1.50	5.2.1.7	Text Edit		
NOAA/Rosman	EF-1.51	5.2.3.1	Field Observations	Fishing Bans	Commercial Take
NOAA/Rosman	EF-1.52	5.7.4.1	Text Edit		
NOAA/Rosman	EF-1.53	6.1	Site Related Doses	Calculation of Dietary Doses	
NOAA/Rosman	EF-1.54	6.2	Congener Quantification	PCB 77	Detection Limit
NOAA/Rosman	EF-1.55	6.2	Uncertainty	Comparative Analysis	
NOAA/Rosman	EF-1.56	6.4	Uncertainty	TRVs	Limited Information
NOAA/Rosman	EF-1.57	6.4	TEFs	Non-Standard	TEQs
NOAA/Rosman	EF-1.58	6.4	Fish Size Class	Forage Fish	Piscivorous
NOAA/Rosman	EF-1.59	6.5.1.1	Fish Size Class		
NOAA/Rosman	EF-1.60	6.5.1.1	Prey Ingestion Rates	Gut Contents	BMR
NOAA/Rosman	EF-1.61	6.5.2	Sensitivity Analysis	Fish Toxicity	Exposure Parameters
NOAA/Rosman	EF-1.62	6.5.3	Model Error	Uncertainty	
NOAA/Rosman	EF-1.63	6.5.3.1	Fishrand	Monte Carlo	
NOAA/Rosman	EF-1.64	7.1	Measured and modeled Sediment	Sediment Guidelines	Benthic Community
NOAA/Rosman	EF-1.65	7.2	Forage Fish	Reproductive Effects	
NOAA/Rosman	EF-1.66	7.2	Field Observations	Decreased Fishing Pressure	
USFWS/Stilwell	EF-2.1	General	Avian Receptors	Reproductive Impairment	
USFWS/Stilwell	EF-2.2	2.3.2	Floodplain	Lower PCB Exposure	
NYSDEC/Ports	ES-1.1	2.3.2	River Miles	Fenimore Bridge	Fort Edward Dam
D. Aulenbach	EP-1.1	2.3.2	Model Results	Model Validation	Conclusions

4.2 Comment Directory For the ERA

AGENCY/ NAME	COMMENT CODE	REPORT SECTION	KEYWORDS 1	2	3
D. Aulenbach	EP-1.2	7.0	Model Results	Conclusions	
Chem. Land Hold	EP-2.1	3.0	Exposure Analysis	Habitat Assessment	
Chem. Land Hold	EP-2.2	3.0	Exposure Analysis	Bioavailability	Sediment
Chem. Land Hold	EP-2.3	3.0	Exposure Analysis	Life History	Exposure Sources
Chem. Land Hold	EP-2.4	4.0	Effects Assessment	Deterministic Methods	Probabilistic Methods
Chem. Land Hold	EP-2.5	5.0	Risk Characterization	Field Investigations	Population Level Data
Chem. Land Hold	EP-2.6	5.0	Screening Thresholds	Sediment	Water
Chem. Land Hold	EP-2.7	5.0	Toxicological Effects Thresholds	NOAEL	
Chem. Land Hold	EP-2.8	6.0	Uncertainty Analysis	Uncertainty Factors	
Chem. Land Hold	EP-2.9	6.0	Fish TEQ	Uncertainty Factors	
Chem. Land Hold	EP-2.10	General	Risk Management Decisions	Uncertainty Factors	
SCEMC/Hodgson	EL-1.1	5.2.1	Field Observations	PCB Impacts	
SCEMC/Hodgson	EL-1.2	2.4	Significant Habitats	Assessment Endpoint	
SCEMC/Hodgson	EL-1.3	General	Measured Endpoints	Model Results	
SCEMC/Hodgson	EL-1.4	Exec. Summ.	Uncertainty Factors	Dioxin	
SCEMC/Hodgson	EL-1.5	Exec. Summ.	Tree Swallows		
SCEMC/Hodgson	EL-1.6	2.4	Bald Eagle	Threatened and Endangered	Shortnose Sturgeon
SCEMC/Hodgson	EL-1.7	2.4	Measurement Endpoints	T&E Assessment Endpoints	
SCEMC/Hodgson	EL-1.8	3.1.1	Target and Non-Target Congeners	Data Usability	TEQ
SCEMC/Hodgson	EL-1.9	3.1	Qualified Data		
SCEMC/Hodgson	EL-1.10	3.1.1	Detection Level	BZ#126	Conservatism
SCEMC/Hodgson	EL-1.11	3.1.2	Congener Distribution		
SCEMC/Hodgson	EL-1.12	3.2	Exposure Concentrations	1993 Data	
SCEMC/Hodgson	EL-1.13	3.2.1	Available Water Data	Future vs. Past	
SCEMC/Hodgson	EL-1.14	3.2.3	Benthic Concentrations	GE Data Use	
SCEMC/Hodgson	EL-1.15	3.2.6	PCB Concentrations	Mink	Otter
SCEMC/Hodgson	EL-1.16	3.4.3	Migration factor	Tree Swallow	
SCEMC/Hodgson	EL-1.17	3.4.3.1	Forage Factor		
SCEMC/Hodgson	EL-1.18	3.4.3.3	Avian	Fish Consumption Factor	
SCEMC/Hodgson	EL-1.19	3.4.3.3	Avian Diets	Fish Size	
SCEMC/Hodgson	EL-1.20	3.4.3.3	Belted Kingfisher Diet	Bald Eagle Diet	
SCEMC/Hodgson	EL-1.21	3.4.3.4	Migration Factors	Bald Eagle	Mallard
SCEMC/Hodgson	EL-1.22	3.4.3.6	Migration Factor	Belted Kingfisher	
SCEMC/Hodgson	EL-1.23	3.4.4	Exposure Factors	Mammals	
SCEMC/Hodgson	EL-1.24	3.4.4.1	FE	Raccoon	Mink

4.2 Comment Directory For the ERA

AGENCY/ NAME	COMMENT CODE	REPORT SECTION	KEYWORDS 1	2	3
SCEMC/Hodgson	EL-1.25	3.4.4.3	Diet Source	Raccoon	Mink
SCEMC/Hodgson	EL-1.26	3.4.4.3	Diet Composition	Fish Sizes	
SCEMC/Hodgson	EL-1.27	3.4.4	Diet Composition	Otter	Raccoon
SCEMC/Hodgson	EL-1.28	3.4.4.4	Hibernation	Raccoon	
SCEMC/Hodgson	EL-1.29	3.5.2	Fish Behavior	Data	High Molecular Weight PCBs
SCEMC/Hodgson	EL-1.30	3.5.3	PCBs	New York City	Tidal Action
SCEMC/Hodgson	EL-1.31	4.2.3	PCB Partition	Egg Lipid	Fish Lipid
SCEMC/Hodgson	EL-1.32	4.2.3.1	PCB NOAEL		
SCEMC/Hodgson	EL-1.33	4.2.1	TEQ	Fish Eggs	
SCEMC/Hodgson	EL-1.34	4.2.1	NOAEL	LOAEL	Fish
SCEMC/Hodgson	EL-1.35	4.2.3.4	PCB Body Burden	NOAEL	Striped Bass
SCEMC/Hodgson	EL-1.36	4.2.1	TRVs	Avian Species	
SCEMC/Hodgson	EL-1.37	4.2.1	TRVs	Avian Species	
SCEMC/Hodgson	EL-1.38	4.2.1	TRVs	Avian Eggs	
SCEMC/Hodgson	EL-1.39	4.2.1	TRVs	Avian Eggs	
SCEMC/Hodgson	EL-1.40	4.2.1	NOAEL	LOAEL	Average Value
SCEMC/Hodgson	EL-1.41	5.1.2.1	Sediment	Modeled Concentrations	Overprediction
SCEMC/Hodgson	EL-1.42	5.1.3.1	NYSDEC Water Quality Criteria	PCB Uptake	
SCEMC/Hodgson	EL-1.43	5.5.3.1	Least Bittern	Upland Sand Piper	King Rail
SCEMC/Hodgson	EL-1.44	5.7	Shortnose Sturgeon	Bald Eagle	Threatened and Endangered
SCEMC/Hodgson	EL-1.45	6.5.3	Model Error		
SCEMC/Hodgson	EL-1.46	7.0	Uncertainty Factors	Assessment Endpoint	TQs
GE	EG-1.1	General	Toxicity Quotients	Guidance	Data Collection
GE	EG-1.2	2.4	Assessment Endpoints	Individuals vs. Populations	
GE	EG-1.3	2.4	Assessment Endpoints	Habitats	
GE	EG-1.4	2.5	Measurement Endpoints	Populations	Community
GE	EG-1.5	2.5	Screening Level Models	Empirical Data	Site Specific Data
GE	EG-1.6	2.5	Screening Level Models	Empirical Data	Models
GE	EG-1.7	5.1	Site Specific Data	Benthic Community	
GE	EG-1.8	5.2.1	Fish Population Data	Lower Hudson River	
GE	EG-1.9	5.2.3	Fish Population Data	White Perch	
GE	EG-1.10	5.5	Population Trends	Bald Eagle	Site Specific Data
GE	EG-1.11	5.1	Benthic Community	Tree Swallow	PCB Effects
GE	EG-1.12	7.0	Weight of Evidence	Adverse Reproductive Effects	
GE	EG-1.13	3.0	TQ Approach	Risk Estimates	Screening
GE	EG-1.14	3.1	PCB Detection Limit		
GE	EG-1.15	3.4	Migratory Behavior	Home Range	

4.2 Comment Directory For the ERA

AGENCY/ NAME	COMMENT CODE	REPORT SECTION	KEYWORDS 1	2	3
GE	EG-1.16	3.4.2	Fish Migration Patterns	Anadromous	Semianadromous
GE	EG-1.17	3.0	Exposure Parameters	Probabilistic Analysis	Deterministic Analysis
GE	EG-1.18	4.2.1	TRVs	NOAELs	LOAELs
GE	EG-1.19	4.2.3	TRVs	Fish	
GE	EG-1.20	4.2.1	TRVs	Birds	Mammals
GE	EG-1.21	4.2.1	Field Studies	Laboratory Studies	TRVs
GE	EG-1.22	4.2.4	TRVs	Avian Species	
GE	EG-1.23	4.2.5	TRVs	Bats	Raccoons
GE	EG-1.24	4.2.5	TRVs	Mink	Uncertainty Factor
GE	EG-1.25	4.2.3	Water Quality Criteria	Protection of Aquatic Life	
GE	EG-1.26	4.2.2	Sediment Effect Concentrations	Benthic Community	
GE	EG-1.27	4.1.3	TEQ	Limitations	
GE	EG-1.28	4.1.3	TEQ	Calculations	
GE	EG-1.29	4.2	TEFs	Screening	
GE	EG-1.30	Appendix I	Analytical Data	BZ#126	
GE	EG-1.31	5.1	Site Specific Data	Tree Swallow	Benthic Community
GE	EG-1.32	5.3	Tree Swallow	Reproductive Effects	
GE	EG-1.33	5.1	Benthic Community	PCB Effects	
GE	EG-1.34	7.1	PCB Effects	Benthic Community	
GE	EG-1.35	7.2	PCB Effects	Fish Populations	
GE	EG-1.36	5.5	PCB Effects	Bald Eagle Populations	
GE	EG-1.37	7.4	Bird Populations	Risk Models	Mallard
GE	EG-1.38	General	Assessment Endpoints	Site Specific Data	Weight of Evidence
GE	EG-1.39a	Appendix K	Principal Components	Sources of PCBs	Congener Ratios
GE	EG-1.39b	Appendix K	Principal Components	Sources of PCBs	Congener Ratios
GE	EG-1.39c	Appendix K	Principal Components	Sources of PCBs	Congener Ratios
GE	EG-1.39d	Appendix K	Principal Components	Fingerprinting	Congener Ratios
GE	EG-1.40	5.1.2.1	NOAA SECs		

THIS PAGE INTENTIONALLY LEFT BLANK

Responses

304098

II. RESPONSE TO COMMENTS ON THE ERA FOR THE UPPER HUDSON RIVER

General Comments

Response to EF-1.1 and EL-1.3

The risk calculations presented in the August 1999 Ecological Risk Assessment (ERA) were based on the May 1999 Baseline Modeling Report (USEPA, 1999a) (see, ERA, pp. 44-46). The risk calculations have been revised to reflect changes based on the Revised Baseline Modeling Report (USEPA, 2000a) and comments received on the ERA. The revised calculations are presented in this Responsiveness Summary. The overall conclusions regarding risks have not changed.

Response to EF-1.2

The fate and transport and bioaccumulation models are designed to capture important physical processes in enough detail to be able to predict future sediment, water, and biota concentrations with confidence. The modeling efforts focused on calibrating the models to available data, and did include a validation component. Insofar as the models are able to capture historical data over a 21-year period, and that the data collected over those 21 years reflect a variety of processes, these are captured (if indirectly) in the models.

USEPA agrees that there are aspects of the Hudson River system that are not explicitly modeled by the Agency's fate, transport, and bioaccumulation models. The degree to which the processes mentioned by the commentor may increase resuspension is unknown (i.e., presence of rocks, trees, and root masses in the river, bank erosion, ice scour, daily water level changes due to operation of the hydropower system, and temperature increases in nearshore sediments during summer low flow periods). There are no data to quantify these processes on the Upper Hudson River system and, therefore, no way to constrain them in the model. However, some of the simulations in the sensitivity analysis may indirectly consider the effects of these processes (e.g., the sensitivity analysis for temperature may address the potential for higher temperatures to be found in shallow near-shore areas).

Response to EF-1.3

The FISHRAND model used the "generic" field-validated Gobas growth rate (which has been used in other applications for other species without modification) for the results presented in the May 1999 BMR. In the Revised BMR (USEPA, 2000a), growth rate coefficients for each species were calibrated to fit the observations. This was done by using the growth rate coefficient for each species as a calibration parameter for all locations. This work is presented in Chapter 6 (Book 3) of the Revised BMR.

Response to EF-2.1

USEPA agrees that there is variability among bird species at the same trophic level with respect to their responses to PCBs. This type of variability also exists for the other receptors (i.e., fish and mammals) that were evaluated in the ERA. The correction regarding the years in which the USFWS data were collected, 1994 and 1995 rather than 1993 and 1994, are noted (see, ERA, p. 43).

USEPA acknowledges the comment that USFWS field data of PCBs in a wood duck egg from Griffin Island in the Upper Hudson River and field observations of reduced productivity in wood ducks support the conclusions of ERA regarding the likelihood of reproductive impairment among wood ducks nesting along certain stretches of the Hudson River.

Response to EG-1.1

The analysis presented in the ERA relies on a toxicity quotient approach. This approach was clearly laid out in the September 1998 Scope of Work (SOW). EPA noted that it would use address the uncertainty associated with using reference concentrations derived from the scientific literature, rather than from site-specific toxicological studies, in the ERA (see, Responsiveness Summary for the ERASOW at p. 27 and ERA at pp.157-158). The use of effect doses and burdens are observable, repeatable effects, which are directly related to PCBs and have specific effects which can be predicted from the available data. These are known effects from the scientific literature and insofar as they are real, they meet the USEPA goal of protection of the environment. The toxicity factors and models used to make these predictions have identified parameter ranges that allow one to assess the uncertainty. Instead of adding site-specific toxicological studies to the SOW that would have delayed the release of the ERA, by one year or more (see, Responsiveness Summary for the ERASOW at p. 27), EPA focused its efforts on obtaining site-specific sediment, water, benthic invertebrate, and fish data.

The use of toxicity quotients is part of USEPA's bottom-up approach that gathers data on individuals in order to predict potential effects on local populations and communities that occur or could occur at the site. A recent USEPA directive (OSWER Directive 9285.7-28P, p. 3) states, "Levels that are expected to protect local populations and communities can be estimated by extrapolating from effects on individuals and groups of individuals using a lines-of-evidence approach. The performance of multi-year field studies at Superfund sites to try to quantify or predict long-term changes in local populations is not necessary for appropriate risk management decisions to be made." USEPA used, among other things, observed concentrations of PCBs in benthic invertebrates and fish in the Hudson River and field studies of birds and mammals in and along the Hudson, in a weight-of-evidence approach to characterize risks to ecological receptors (see, ERA Sections 3.2: Observed Exposure Concentrations and 5.0: Risk Characterization).

EPA previously has addressed GE's comment recommending a top-down approach in lieu of the bottom-up approach that was outlined in the ERA SOW (see, Responsiveness Summary for the Ecological Risk Assessment Scope of Work at p. 13). Specifically, EPA noted that the bottom-up approach (calculating Toxicity Quotients, which are ratios of site-specific exposure to toxicity reference values, or TRVs) is consistent with EPA's guidance on conducting ecological risk assessments (see, EPA's 1997 *Ecological Risk Assessment Guidance for Superfund: Process for Designing and Conducting Ecological Risk Assessments* at p. 7-3, "...the quotient method of comparing an estimated exposure concentration to a threshold for a response can be used...").

Further, the ERA provides adequate information for decision makers when considered in conjunction with other parts of the Reassessment RI/FS, such as the Human Health Risk Assessment, the Data Evaluation and Investigation Report, and the results of the modeling. The Hudson River PCBs Superfund Site, stretching for nearly 200 river miles, has been contaminated by PCBs for over 50 years. Consequently, the site is complex on both a spatial and temporal scale and decision-makers will benefit from multiple lines of evidence in reaching site management decisions.

Although the Hudson River and the Clinch River are both large contaminated sites, they are not directly comparable. The Oak Ridge Reservation is owned and administered by the Department of Energy and has fewer outside influences than the Hudson River. Performing top-down studies that start with field population and community information may not accurately represent the effects of PCBs, because other factors (e.g., fishing ban) may mitigate some of the effects of PCBs (see, Responsiveness Summary for the ERASOW at p. 13). Conducting various river specific studies beyond what the New York Department of Environmental Conservation (NYSDEC), US Fish and Wildlife Service (USFWS), and others are already conducting would have provided more elements to the weight of evidence approach, but also would have introduced such broad uncertainties of their own that they are unlikely to have reduced general uncertainty in the assessment. Population numbers, age-class and annual reproductive success vary so widely in nature that only detailed species-specific scientific studies are likely to provide useful data.

Response to EG-1.38

As discussed in the ERA SOW and ERA, the ERA was directed specifically at reassessing the "No Action Decision" related to the PCBs in Hudson River sediments (see, ERA p. 5) and an appropriate level of effort was used for this task. As detailed in responses to other comments (e.g., EG-1.1, 1.2, 1.3, etc.) the methodology and assumptions used in the ERA are supportable and the ERA itself is an important tool for decision makers assessing the Hudson River PCBs Site.

Response to EP-2.10

Consistent with Superfund guidance for ecological risk assessments (see, USEPA 1999d; pp. 3-6), the ERA used a weight-of-evidence approach to evaluate risks to ecological receptors. This weight-of-evidence approach provides sufficient information for the Agency's risk managers to use in making risk management decisions. The degree of uncertainty associated with risks to various receptors in the ERA is considered low to moderate (see, ERA Executive Summary, and Section 7).

EXECUTIVE SUMMARY

Response to EF-1.8

The conclusion that PCBs in the Hudson River pose a risk to benthic invertebrate communities is an important finding of the ERA because these communities are a food source for fish. The conclusions of the ERA that were highlighted in the Executive Summary focused on receptor populations (i.e., fish and wildlife), and so did not include the finding regarding benthic invertebrate communities. However, USEPA agrees that it would be appropriate to include a bullet in the Executive Summary (ERA, p. ES-11) to summarize this finding, as follows:

- Benthic invertebrate communities in the Hudson River are an important food source for many species of fish. PCBs in the Hudson River may adversely affect benthic invertebrate populations.

Response to EL-1.4

For the fish receptors, the selected TEQ-based TRVs were based on concentrations of PCBs in eggs of lake trout. Because lake trout is among the most sensitive of species that have been tested, and the concentration was measured in the egg rather than a dietary dose, the interspecies and subchronic-to-chronic uncertainty factors were not applied to developing the TRVs for the Hudson River fish species. For the avian receptors, the TEQ-based TRV for the tree swallow was based on Hudson River data (USFWS); thus, no uncertainty factors were required. The egg-based TRVs for TEQ congeners for the avian receptors were based on a study of gallinaceous birds, which are among the most sensitive of avian species. For this reason, as with fish, no uncertainty factors were applied. Dietary dose TRVs for the avian receptors incorporated a subchronic-to-chronic uncertainty factor of 10. For the mammalian receptors, an uncertainty factor of 10 was applied in deriving the TEQ-based TRV to account for potential interspecies differences. The derivation of the TRVs for dioxin-like PCBs, including the application of uncertainty factors, if any, is described for each receptor individually in Chapter 4 of the ERA.

Response to EL-1.5

It is unclear to what "serious questions" on the Upper Hudson River tree swallows study the commentor refers. Papers on the study (e.g., McCarty and Secord, 1999a and 1999b) have been published in the peer-reviewed scientific literature. USEPA interpreted the data on behavioral endpoints as No Observed Adverse Effect Levels (NOAELs) rather than Lowest Observed Adverse Effect Levels (LOAELs) because the statistical significance of primarily behavioral endpoints relative to population-level reproductive effects is uncertain.

1.0 INTRODUCTION

No significant comments were received on Section 1.0.

1.1 Purpose of Report

No significant comments were received on Section 1.1.

1.2 Site History

No significant comments were received on Section 1.2.

1.3 Site Investigation and Hudson River Data Sources

1.3.1 EPA Phase 2 Data
1.3.2 NYSDEC/NOAA Data
1.3.3 United States Fish and Wildlife Service (USFWS) Data
1.3.4 GE Data
1.3.5 Other Data Sources

No significant comments were received on Section 1.3.

1.4 Technical Approach and Ecological Assessment in the Superfund Process

No significant comments were received on Section 1.4.

1.5 Report Organization

No significant comments were received on Section 1.5.

2.0 PROBLEM FORMULATION

No significant comments were received on Section 2.0.

2.1 Site Characterization

No significant comments were received on Section 2.1.

2.2 Contaminants of Concern

No significant comments were received on Section 2.2.

2.3 Conceptual Model

No significant comments were received on Section 2.3.

2.3.1 Exposure Pathways in the Hudson River Ecosystem

No significant comments were received on Section 2.3.1.

2.3.1.1 Biological Fate and Transport Processes

No significant comments were received on Section 2.3.1.1.

2.3.2 Ecosystems of the Hudson River

Response to EF-1.7 and EF-2.2

The ERA does not quantify risks to terrestrial receptors on the floodplain. This is because the ERA is limited to ecological risks associated with PCBs in the sediment, water, and biota in the Hudson River, which is consistent with the focus of the Reassessment RI/FS. In addition, there are insufficient data available to characterize the nature and extent of PCBs in floodplain soils (see, ERA, p. 14, see also, Responsiveness Summary for ERA Scope of Work, p. 21). However, to address concerns raised earlier regarding this issue, USEPA qualitatively addressed ecological risks associated with exposure to PCBs in floodplain soils as a source of uncertainty (see, ERA, p. 156). Specifically, terrestrial animals using the Hudson River shoreline may be exposed to elevated concentrations of PCBs and be adversely affected by them. Birds and mammals feeding on prey found in floodplain soils (e.g., earthworms, insects) are likely to have the highest shoreline exposure to PCBs.

Response to ES-1.1

As described in Chapter 1 of the ERA, the Upper Hudson River is defined as the river bed between the Fenimore Bridge in Hudson Falls and the Federal Dam at Troy. River mile designations are approximate; thus, the Upper Hudson River extends from approximately river mile 153 up to river mile 195. All of the data used in the risk assessment was obtained from locations below river mile 194.6.

2.3.3 Aquatic Exposure Pathways

No significant comments were received on Section 2.3.3.

2.3.4 Terrestrial Exposure Pathways

No significant comments were received on Section 2.3.4.

2.4 Assessment Endpoints

Response to EG-1.2

Assessment endpoints are defined as the protection and maintenance (i.e., survival, growth, and reproduction) of local populations (see, ERA, pp. 19-29). Associated measurement endpoints may be at the individual organism, population, or community level. This approach is consistent with USEPA ecological risk assessment guidance (USEPA, 1999d; p. 3).

Response to EL-1.2, EL-1.6, EL-1.7, and EG-1.3

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 (see, ERA, p. 1). The ERA addressed current risks to receptors in the Upper and Lower Hudson River and future risks to receptors in the Upper Hudson River. The December 1999 addendum (USEPA, 1999e) addressed future risks to receptors in the Lower Hudson River (see, ERA, p. 8).

The comment that the bald eagle is no longer a threatened or endangered species and should be removed from the discussion of threatened and endangered species is incorrect. The bald eagle (*Haliaeetus leucocephalus*) is a federal and New York State-listed threatened species. The comment that the shortnose sturgeon (*Acipenser brevirostrum*) and striped bass (*Morone saxatilis)* should be omitted from the discussion because they are not present in the Upper Hudson does not accurately reflect the extent of the study area for the ERA, which includes the Lower Hudson River, where these species are found.

The comment that no definition of "significant habitats" was provided in the ERA is incorrect. Significant habitats along the Hudson River were listed and defined as those areas designated by the NYS Coastal Management Program (see ERA, p. 36 and Table 2-11). All Hudson River significant habitats have been mapped and their biological communities described in a published report (NYSDOS and the Nature Conservancy, 1990), which the ERA cited. All of the significant habitats are located in the Lower Hudson River (i.e., below the Federal Dam at Troy, NY). The ERA evaluated receptor exposure at eight significant habitats and found that current PCB concentrations exceed toxicity reference values for some fish, avian, and mammalian receptors (see ERA, p. 150).

A comparable background station upstream of Fort Edward for the benthic invertebrate sampling could not be located (see, ERA, p. H-2). Detailed information on Hudson River fish, insectivorous birds, piscivorous birds, and wildlife populations prior to the use of PCBs at the General Electric Company capacitor manufacturing plants in the 1940s is not specific enough for comparison to current studies to examine potential effects of PCBs. In addition, USEPA notes that the timeframe of the Reassessment RI/FS is not long enough to collect data to evaluate ecological risk posed by PCBs in the river. The toxicity quotient approach is used specifically because population data alone would not distinguish among changes due to the PCBs in the river and changes due to non-site related factors, such as fisheries management and habitat loss.

2.5 Measurement Endpoints

Response to EF-1.9

The ERA correctly states that median and 95th percentiles were examined for fish, while the mean and 95% UCLM (upper confidence limit on the mean) were evaluated for birds and mammals (see, ERA, pp. 22-27). Predicted fish body burdens are described by distributions to capture the population effects of PCB uptake. Selected fractiles of these distributions are presented (25^{th}, 50^{th}, and 95^{th}) and compared to NOAELs and LOAELs to provide perspective on the range of potential risks.

Mammalian and avian receptors, by contrast, integrate dietary exposures over time and thus the appropriate statistic is the average PCB concentration (in fish, benthic invertebrates, etc.). The 95% UCLM captures the statistical uncertainty in estimating the average concentration. It is not appropriate to characterize mammalian and avian dietary exposures by full distributions of PCB concentrations in fish, as these represent the variability in predicted PCB uptake in the fish population, not the uncertainty in the predicted average fish concentration.

Response to EF-1.10

USEPA agrees that additional measurement endpoints for fish, mammals, and birds that are appropriate for the ERA are a comparison of measured and modeled fish TEQ concentrations reported by USEPA (1993b). For example, low risk to piscivorous fish was associated with fish concentration of 50 pg/g TCDD and high risk at 80 pg/g TCDD. Fish TCDD concentrations of 6 pg/g and 60 pg/g were identified as posing a low to high risk to avian wildlife respectively; where as high risk is defined as causing 50-100% mortality in embryos and young of sensitive species. For mammalian wildlife, fish TCDD concentrations of 0.7 pg/g pose a low risk and 7 pg/g pose a high risk. However, the inclusion of the additional endpoints does not change the overall conclusions of the ERA for the fish, avian, and mammalian receptors.

Response to EG-1.4

Measurements endpoints provide the actual measurements used to evaluate each of the testable hypotheses and to estimate risk (ERA, p. 20). The measurement endpoints selected for use in the ERA are consistent with current USEPA ecological risk assessment guidance (USEPA 1997b, 1999d). A bottom-up approach (calculating Toxicity Quotients, which are ratios of site-specific exposure to toxicity reference values, or TRVs) is consistent with USEPA's guidance on conducting ecological risk assessments (see, EPA's 1997 *Ecological Risk Assessment Guidance for Superfund: Process for Designing and Conducting Ecological Risk Assessments* at p. 7-3, "...the quotient method of comparing an estimated exposure concentration to a threshold for a response can be used..."). A weight-of-evidence approach was used to determine whether concentrations of PCBs in the Hudson River may cause adverse effects in individuals and populations of ecological receptors of concern. USEPA used, among other things, observed concentrations of PCBs in benthic invertebrates and fish in the Hudson River and field studies of birds and mammals in and along the Hudson, in a weight-of-evidence approach to characterize risks to ecological receptors (see, ERA Sections 3.2: Observed Exposure Concentrations and 5.0: Risk Characterization).

Some of the lines of evidence in the ERA include a comparison of measured and modeled concentrations of PCBs in the river to sediment guidelines and water quality criteria. However, the conclusions in the ERA do not rely on generic toxicity benchmarks. Rather, for each assessment endpoint, multiple lines of evidence were examined to evaluate the risk to a receptor posed by PCBs in the Hudson River.

With respect to the TRVs, USEPA disagrees with the comment that the TRVs were derived from the most reasonable single-species toxicity tests available. The TRVs were derived using the methodology outlined in the ERA (see, pp. 79-81) and are comparable with those used at other Superfund sites. For example, for the Sheboygan River, TRVs for mink for total PCBs were the exact same values as those used for the Hudson River. The Fox River ecological risk assessment TRVs for the mink for total PCBs were 0.0021 mg/kg/day for the LOAEL and 0.099 mg/kg/day for the NOAEL; these values are lower than those selected for the Hudson River. The TEQ-based NOAEL developed for bald eagles for the Fox River site was 7 ng/kg egg, while the Hudson River TRV was 10 ng/kg egg. The Fox River ecological risk assessment used 0.75 mg/kg wet weight as a NOAEL for all fish species. Hudson River NOAELs were developed specifically for each individual fish species and range from 0.3 mg/kg wet weight to 5.25 mg/kg wet weight.

Response to EG-1.5

The ERA used site-specific information on receptor species, when available, to provide exposure parameters that were appropriate for the Hudson River (see, ERA, pp. 50-55, 63-64 and Appendices C-F). Field-based observations from a variety of sources provided information on the degree of species diversity and abundance of wildlife along the Hudson River. Most historical and field-based observations do not contain data that is easily comparable to data collected for this study. For example, studies conducted for power plants were collected for a different purpose and are not indicative of biomass estimates in the river generally, but rather were designed to assess the impact of thermal discharge on fish populations. USEPA did not conduct site-specific toxicological studies at different locations along the entire length of the river because it would have required a number toxicological studies conducted over a period of several years, which in turn would have delayed the Superfund process and added substantially to the costs of the ERA (see, Responsiveness Summary for the ERASOW, p. 27).

Response to EG-1.6

The water quality criteria and sediment guidelines were used as measurement endpoints in a weight-of-evidence approach. As noted in the October 7, 1999 letter to GE, USEPA used the NYSDEC ambient water quality criterion for wildlife in the ERA as one of several lines of evidence to evaluate risks to ecological receptors. The NYSDEC criterion is derived from studies of mink (a species known to be sensitive to PCBs) and therefore is particularly appropriate as a measurement endpoint for the mink receptor selected by EPA. However, because the water quality criterion is intended to be protective of all wildlife, it is appropriate for receptors of concern other than mink (see, ERA, Table 2-7). The sediment quality guideline was derived based on the water quality criterion and is also considered appropriate for wildlife receptors of concern.

2.6 Receptors of Concern

No significant comments were received on Section 2.6.

2.6.1 Macroinvertebrate Communities

No significant comments were received on Section 2.6.1.

2.6.2 Fish Receptors

No significant comments were received on Section 2.6.2.

2.6.3 Avian Receptors

No significant comments were received on Section 2.6.3.

2.6.4 Mammalian Receptors

No significant comments were received on Section 2.6.4.

2.6.5 Threatened and Endangered Species

Response to EF-1.11

USEPA agrees that the peregrine falcon (*Falco peregrinus*) was removed from the federal list of threatened and endangered species on August 20, 1999, after the ERA was released. However, the peregrine falcon remains listed as a New York State endangered species, and therefore is still appropriately considered in the discussion of threatened and endangered species of the Hudson River (see, ERA, pp. 34-35).

2.6.6 Significant Habitats

No significant comments were received on Section 2.6.6.

3.0 EXPOSURE ASSESSMENT

Response to EF-1.4 and EP-2.1

To assess risks to ecological receptors, USEPA used measured water column and sediment data from samples that were collected in the river, and modeled concentrations of PCBs in the water column and sediment that were generated using the HUDTOX model. Water column data were averaged to correspond with the ecological sampling locations. Sediment data were averaged in the Thompson Island (TI) Pool across TI Pool sampling locations, but elsewhere only the five samples collected at each particular location were averaged. The HUDTOX model output was averaged as described in the Revised BMR (three locations corresponding to a few miles encompassing the fish sampling locations, i.e., River Miles 189, 168, and 154) (USEPA, 2000a).

Due to size of the site, USEPA did not conduct habitat mapping along the 200-mile length of the Hudson River study area to group data by habitats. However, the majority of receptors modeled feed or live in shallow water habitats, which is where most of the samples were collected, so the data and model output are appropriate for evaluating exposure. Receptors were selected to represent species from different trophic levels, rather than to assess risks to only that specific species. An exposure assessment based on specific habitats and receptors is not appropriate for the Hudson River site because of its large size and the presence of receptors at various points along the river.

Response to EG-1.13

The assumptions used to define parameters for receptors species were based on the best available information. Hudson River specific data were used when available. The TRVs used in this assessment are comparable with those used at other Superfund sites. For example, for the Sheboygan River, TRVs for mink for total PCBs were the exact same values as those used for the Hudson River. The Fox River ecological risk assessment TRVs for the mink for total PCBs were 0.0021 mg/kg/day for the LOAEL and 0.099 mg/kg/day for the NOAEL; these values are lower than those selected for the Hudson River. The TEQ-based NOAEL developed for bald eagles for the Fox River site was 7 ng/kg egg, while the Hudson River TRV was 10 ng/kg egg. The Fox River ecological risk assessment used 0.75 mg/kg wet weight as a NOAEL for all fish species. Hudson River NOAELs were developed specifically for each individual fish species and range from 0.3 mg/kg wet weight to 5.25 mg/kg wet weight.

Response to EG-1.17

In the ERA, USEPA conducted selected Monte Carlo simulations for specific species and years as part of the uncertainty analysis (see, ERA, pp. 164-165). To conduct this analysis for every year, species, and location is not tractable, so a few years were randomly selected for the eagle, kingfisher, mink and otter for the Monte Carlo analysis. The patterns that emerge will be consistent from year to year, thus, meaningful results can be drawn just from random years without having to conduct the analysis for every year from 1993 – 2018. In the Monte Carlo analysis, distributions rather than point estimates were specified for key exposure parameters, including concentrations of PCBs in sediment, water, and prey, ingestion rates, and body weight. The ratios of the predicted 25th percentile to the average (50th percentile) typically range from 0.6 and 0.8 for the avian and mammalian receptors. Because the toxicity quotients are greater than two for all receptors (except the tree swallow), this result suggests that toxicity quotients are greater than one (TQ >1) for most receptors at the 25th percentile.

Response to EP-2.2

The most biologically active zone of sediment (top 5 cm) was sampled in the 1993 ecological sampling program for the Reassessment RI/FS; these data were used to provide field-based exposure concentrations. The bioavailability of PCBs in the sediment is dependent on a number of factors including contaminant concentration and organic carbon content (see, ERA pp. 16-17). Measured and modeled fish concentrations were used, which incorporated bioavailability of PCBs in the sediments to receptors.

Response to EP-2.3

Life history characteristics are considered in the exposure assessment (see, ERA, pp. 54-55). Many of the receptors modeled have both resident and migrant populations in the Hudson River Valley. The resident populations are most at risk and were therefore evaluated in the exposure assessment. Although the tree swallow migrates along the Hudson River, continuous exposure (i.e., 1.0) was used because tree swallows breed along the banks of river and the young are reared and grow to adult size prior to the autumn migration.

3.1 Method to Determine Toxic Equivalencies (TEQ)

Response to EL-1.9

The PCB congener data were evaluated for usability as part of a quality assurance system to monitor the accuracy, precision, representativeness, and sensitivity of the analytical results (see, ERA, Appendix I, p. I-7). As a result of this data quality review, a high percentage (62%) of the PCB congener data were qualified as estimated, primarily due to detection at concentrations below the calibrated quantification limit and/or exceedences in the dual GC column precision criteria (see, ERA, p. I-33). These data were determined to be usable for the ecological risk assessment given the data quality objectives of the sampling program, which were established in the Phase 2B Sampling and Analysis Plan/Quality Assurance Project Plan (USEPA, 1993a). A relatively small percentage of the PCB data (925 of the 59,063 congener measurements, or 1.6%) were rejected due to exceedence of quality control criteria (see, ERA, Tables I-9 to I-12).

Response to EG-1.14

The comment incorrectly describes how USEPA addressed non-detect values in the ERA. All PCB data used in the ERA are contained in the Database for the Hudson River PCBs Reassessment RI/FS, Release 4.1 (USEPA, 1998a) (see, ERA, p. 4). Non-detect values were assumed to be zero if more than 85% of the samples from a given location were below the detection limit. If concentrations above the detection limit were detected in more than 85% of the samples, non-detect samples were assumed to have concentrations at one-half of the non-detect value (see, *Value 2* in USEPA, 1998a). As a result of considering the frequency of detection (i.e., congener presence), USEPA used values that were less conservative than using one-half the detection limit for all non-detect samples.

An exception was made to this method to evaluate the 12 Toxic Equivalency (TEQ) congeners. For the TEQ analysis, BZ#126 was used at the detection limit to compensate for not having quantitated BZ#81 (see, ERA, pp. 38-40). An analysis evaluating the proportion of TEQ congeners in USEPA Phase 2 data and USFWS tree swallow data showed that the proportion of BZ#126 in the Phase 2 data set was roughly equal to the sum of the BZ#126 and BZ#81 in the USFWS data set (see, ERA Appendix J). This approach does not produce an overly conservative estimate of TEQ risks because they are dominated by BZ#126 (and presumably BZ#81) and thus may be overestimated by no more than a factor of 2. This possible overestimation is a relatively small margin of error given that the calculated risk levels exceed acceptable levels by orders of magnitude.

3.1.1 Data Quality Issues for TEQ Congeners

Response to EL-1.8 and EL-1.10

The selection of the 90 target congeners in 1992 is described in the ERA (Appendix I) and in previous Reassessment RI/FS reports (see, USEPA, 1997a; DEIR Chapter 2 and the data usability reports that are included in the DEIR appendices). The selection of the 90 PCB congeners was based on their significance in environmental samples and the commercial availability of calibration standards (see, ERA, p. I-3). Some of the non-target congeners were later calibrated using the calibration curve for target congener BZ#52 and calibration standards that became commercially available in 1993, such that an additional 36 congeners are called calibrated non-target congeners. The target congeners, calibrated non-target congeners, and other non-target congeners are identified by BZ number in the ERA (see, ERA Appendix I, Table I-1). A total of approximately 147 congeners were reported for sediment and biota samples used in the ERA.

The USEPA (1993a) data usability report was revisited in 1999 for the ERA to focus on the 12 TEQ congeners, as opposed to the 12 "principal congeners" that were the focus of the earlier data usability report. The results of the data usability report for the 12 TEQ congeners are presented in the ERA (pp. 38-40).

For the TEQ analysis, BZ#126 was used at the detection limit to compensate for not having quantitated BZ#81 (see, ERA pp. 38-40). An analysis evaluating the proportion of TEQ congeners in USEPA Phase 2 data and USFWS tree swallow data showed that the proportion of BZ#126 in the Phase 2 data set was roughly equal to the sum of the BZ#126 and BZ#81 in the USFWS data set (see, ERA Appendix J). This approach does not produce an overly conservative estimate of TEQ risks because they are dominated by BZ#126 (and presumably BZ#81) and thus may be overestimated by no more than a factor of 2. This possible overestimation is relatively small margin of error given that the calculated risk levels exceed acceptable levels by orders of magnitude.

3.1.2 Estimating Future Baseline TEQ Concentrations

Response to EL-1.11

In the ERA, USEPA assumed that, while absolute concentrations of PCBs in fish generally decrease with time, the congener distributions (patterns) are relatively consistent from year to year (ERA, p. 40). This assumption is reasonable and consistent with the historical data. The congener patterns in fish would be expected to remain relatively consistent as long as the congener patterns to which the fish are exposed remain relatively consistent. Although the biological processes of the fish, such as uptake and depuration, may modify the pattern relative to that of the source (e.g., sediment), these biological processes would continue to be in effect and cause that modification through time. As discussed in the DEIR (USEPA, 1997a), dated sediment core evidence strongly suggests that congener patterns in sediment have been relatively consistent for at least the 17 years prior to the collection of the high resolution cores in 1992. The analytical results from the core samples show the predominance of a single PCB mixture, derived principally from Aroclor 1242, over the period 1975 to 1992, for the entire freshwater Hudson downstream of the GE facilities in Hudson Falls and Fort Edward, New York.

TEQ congeners in fish from the 1993 USEPA/NOAA and the 1995 NOAA/NYSDEC data sets were not specifically compared in terms of their relative proportions, in part because of the large number of non-detect values, which precludes this sort of analysis. However, an analysis of the general congener patterns found in fish was presented in Sections K-3 and K-7 of the ERA. These sections concluded that fish congener patterns were locally consistent across species at a given location as well as over time based on a comparison of the 1993 and 1995 results. The analysis also showed that seasonal variation takes place with preferential enhancement of heavier congeners in fish tissue in spring relative to fall conditions. Because the ERA was based on autumn congener patterns, it is likely that the analysis may slightly underestimate the actual body burdens of the TEQ congeners. Nonetheless, the analyses in Appendix K of the ERA and the DEIR (USEPA, 1997a) directly and indirectly support the assumption of a relatively constant congener pattern in fish over time.

Response to EF-1.12

The "TEF-based factor" (derived for each individual location) was the same within the upper river and the same within the lower river, but different between the two, as Table 3-2 of the ERA shows. Thus, it was appropriate to apply a single factor for the upper vs. lower river (note that single factor is misleading: there is actually a matrix of factors – media as rows; water sed, etc. and receptor as column; avian, mammalian, etc.). Applying upper versus lower versus whole river TEF-based factors does not change the conclusions, given the magnitude of the estimated toxicity quotients.

3.2 Observed Exposure Concentrations

Response to EL-1.12

USEPA Phase 2 (1993) sediment, water, fish and benthic invertebrate data provide a complete synoptic dataset for all of the media. Use of this dataset allows us to minimize the assumptions made in the modeling and to rely to the greatest extent possible on data rather than on modeled concentrations.

3.2.1 Observed Water Concentrations

Response to EF-1.13

Data handling for the TI Pool took several forms depending upon the exposed animal and the period of time examined. For the purposes of examining water quality criteria, the Rogers Island and Thompson Island Dam station data were averaged to create an average for the Thompson Island Pool. Data to enable the comparison of near-shore *vs.* center channel conditions does not exist for the 1993 sampling period, although later data sets obtained by GE do permit such a comparison. These results indicate that for the 1993 period of sampling, the correction would be minimal due to the large, upstream loading conditions that occurred at that time. Additionally, the Rogers Island station differs from the Thompson Island Dam station far more than the suggested difference between near-shore and center-channel conditions measured by GE. Thus the average of these two stations represents a kind of composite representing the range of conditions possible in the Thompson Island Pool.

For the purposes of estimating future fish exposure and risk, the monitoring data were not used directly. Rather, the forecast model FISHRAND was driven using model output from HUDTOX for center channel conditions. These results were subsequently corrected to reflect near-shore conditions based on the concentration and load analysis performed in Appendix Cof the LRC responsiveness summary (USEPA, 1999). Dissolved phase concentrations of the Tri+ congener sum (the water exposure media for fish) were corrected as follows:

Flow (cfs)	Rogers Island Concentration	Correction Factor (unitless)
<4000	<15 ng/L	1.45
<4000	>15 ng/L	1.14
>4000	Not Applicable	No Correction

Note that the Tri+ sum is used because fish do not retain the lighter congeners in their body burdens. These correction factors are simply the inverse of the factors developed to correct TI Dam West results to a center channel result.

For the purposes of mammalian and avian exposures, the current risks were based on the same data set and manipulations as was performed for the water quality criteria review. For future risks, the results from HUDTOX for center channel Tri+ concentrations were used. These results represented whole water concentrations of Tri+ (as opposed to the dissolved phase concentrations used for fish) which were then corrected to represent total PCBs by means of a correction factor (2.31) developed from TI Pool data collected by GE.

Response to EL-1.13

There are additional water samples collected by General Electric, and a few sediment samples from 1991 (predating the start of the ecological risk assessment). USEPA collected the data in 1993 for this assessment and as described in the Scope of Work. The intent was to rely on this USEPA data for the risk assessments. Note that the water concentration data is only used for the ingestion pathway for the avian and mammalian receptors which contributes less than 1% to the overall ingestion of PCBs. Also, these additional samples are from limited locations in the Upper Hudson River while the 1993 dataset provides data from multiple locations in both the Upper and Lower Hudson River.

3.2.2 Observed Sediment Concentrations

Response to EF-1.14

The organic carbon (OC)-normalized sediment data were inadvertently omitted from the ERA. Organic carbon normalized sediment data were compared to NYSDEC Benthic Chronic and Wildlife Guidelines, Persaud SEL, Jones et al. (1997) and Washington State Microtox AET values (see Tables 5-6, 5-8 and 5-9 in the ERA, USEPA, 1999a). Sediments were also compared to dry weight sediment guidelines including: NOAA Hudson River effect concentrations, Persaud et al. (1993) NOAEL and LEL, Long et al. (1995) ERL and ERM, Ingersoll (1996) and Washington State Proposed Freshwater Guidelines (1997). All of these guidelines were used in the evaluation of sediment PCB levels. Based on these comparisons, no impact to the conclusions of the ERA is anticipated.

Response to EF-1.15

The comment correctly describes the surface sediment data used in the ERA (p. 42 and Table 3-4). Within the Thompson Island Pool, sediment (and benthic invertebrate) data were collected from five separate sampling locations. These data were combined, so the average and 95% UCLM concentrations of PCB congeners reflect, to a certain extent, the variability in sediment concentrations within the six-mile Thompson Island Pool. At locations upstream and downstream of the Thompson Island Pool, five replicate samples were generally collected at a single sampling station, so the average and 95% UCLM concentrations of PCB congeners represent only the sample variability at that location.

3.2.3 Observed Benthic Invertebrate Concentrations

Response to EF-1.16

All sampling locations are located within the mainstem of the river, including the NERR stations.

Response to EL-1.14

The two primary objectives of the Phase 2B benthic invertebrate sampling were to examine benthic invertebrate community structure in the Thompson Island Pool and to analyze benthic invertebrate PCB congener body burdens (see, ERA, p. B-4, see also USEPA, 1993a). The observed concentrations of PCBs in benthic invertebrates were accumulated over a period of weeks to months, which equates to the lifetime exposure of many benthic invertebrates to the PCBs in the sediments and water. Data collected by GE were not used because they were not co-located with the USEPA sediment, water, and fish samples.

3.2.4 Observed Fish Concentrations

Response to EF-1.17

USEPA acknowledges that the NOAA (1997) report presented an analysis that suggested a systematic difference between USEPA's congener-specific analytical method and NYSDEC's Aroclor-based method. A similar analysis of lipid fraction in the NOAA (1997) report also showed systematic differences between the USEPA congener data and the NYSDEC Aroclor data. In the ERA, the USEPA and NYSDEC data sets were used in a manner that emphasizes their respective strengths and minimizes uncertainty. For example, the USEPA congener-specific data were used to compare the PCB congener patterns in fish with the patterns observed in sediment and water (see, ERA, Appendix K) as well as to evaluate congener-specific toxicity, such as dioxin toxic equivalency (see, ERA, pp. 38-40). The NYSDEC data set is much more extensive in time and therefore was used to estimate risks attributable to PCB body burden in fish and potential PCB exposure to piscivorous mammals and birds. The only exceptions to this are forage fish and mink: risks for these ecological receptors were based on the USEPA congener-specific data because no NYSDEC data exist for forage fish, which constitute the main fish prey for mink.

The differences between the USEPA and NYSDEC data do not represent a major source of uncertainty for the ERA. Specifically, the USEPA values for fish body burdens were about 31 percent lower than those obtained by NYSDEC for the matching fish samples. Because ecological risks were frequently 10 to 100 and sometimes 1000 times higher than levels of concern, this uncertainty does not affect the overall conclusions of the ERA. Moreover, because the lipid content reported by USEPA was also lower than that reported by NYSDEC by roughly the same percentage, the risks to fish (which are based on lipid-normalized results) would be comparable regardless of the data source.

Response to EF-1.18

USEPA agrees with the comment. Toxic equivalency concentrations were estimated for 15 pumpkinseed samples collected from five locations in 1993 (NYSDEC/NOAA collection) using a fish-cell line bioassay (Tillitt, 1997). TEQs in the pumpkinseed ranged from 1 to 115 pg TCDD-eq/g (wet wt.), with the highest concentrations from fish samples collected in the Thompson Island Pool. These estimated TEQs exceed the NOAEL and LOAEL selected for pumpkinseed (see, ERA, p. 83)

3.2.5 Observed Avian Concentrations

No significant comments were received on Section 3.2.5.

3.2.6 Observed Mammalian Concentrations

Response to EL-1.15

Data on PCB concentrations in mink and otter are currently being collected by NYSDEC scientists. The reports presenting these data have not yet been issued (Mayack, 1999).

3.3 Quantification of PCB Fate and Transport

No significant comments were received on Section 3.3.

3.3.1 Modeled Exposure Concentrations

Response to EF-1.19

The constant upstream boundary assumption in the baseline models of 10 ng/L PCBs is considered protective of aquatic health compared to the 0 ng/L PCBs constant upstream boundary because accounting for this additional input of PCBs reasonably increases risks over what would be calculated assuming zero PCBs in the upstream surface water (see, ERA, p. 44). USEPA agrees that the 10 ng/L upstream boundary condition would reflect an exceedence of surface water quality standards.

3.3.1.1 Modeled Water Concentrations

No significant comments were received on Section 3.3.1.1.

3.3.1.2 Modeled Sediment Concentrations

No significant comments were received on Section 3.3.1.2.

3.3.1.3 Modeled Benthic Invertebrate Concentrations

No significant comments were received on Section 3.3.1.3.

3.3.1.4 Modeled Fish Concentrations

No significant comments were received on Section 3.3.1.4.

3.4 Identification of Exposure Pathways

Response to EG-1.15

Receptor assumptions were selected after reviewing scientific literature and speaking to Hudson River wildlife scientists. Given the size of the Hudson River PCB Superfund site and home ranges of the species modeled, the home range modifying value of 1.0 is considered to be realistic. Because receptors are expected to stay within the Hudson River Valley, the assumption of all prey originating from the Hudson River is also reasonable. The diet of the mink and the raccoon derived from river sources was assumed to be 50.5% and 40%, respectively, which considers that these species forage for prey in areas other than the Hudson River. Habitat availability and utilization for all receptors were evaluated for the ERA (see, ERA Appendices D, E, and F). In addition, exposure durations of receptors are comparable to the duration of the Giesy et al. study (1995) used to derive the biomagnification factor. Similarly, the periods of exposure for fish and wildlife receptors are analogous to the periods of exposure used in the toxicity studies.

3.4.1 Benthic Invertebrate Exposure Pathways

No significant comments were received on Section 3.4.1.

3.4.2 Fish Exposure Pathways

Response to EF-1.20

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

Seasonal accumulation results in a factor of two change in fish body burden over a given year.

Response to EF-1.21

NYSDEC data were used to estimate PCB concentrations in all fish except forage fish (smaller fish < 10 cm) because the data set is extensive both spatially and temporally. USEPA data were used for forage fish because no NYSDEC data are available for them. The USEPA/NOAA Phase 2 data set was used to estimate the PCB concentrations in forage fish (see, ERA, pp. B-5 to B-6). Note that the USEPA/NOAA Phase 2 data for the pumpkinseed, generally considered a small fish, were eliminated because the fish specimens collected were too large to qualify as a small fish.

Response to EG-1.16

For the fish receptors, data from males and females were combined to provide an estimate of exposure for each species. If body burdens in resident striped bass males are higher than in spawning females and migrating striped bass males, then their risks would be higher than the calculated risks based on data from males and females combined. In the ERA, the PCB body burdens for striped bass (male and female combined) were expressed on a standard fillet basis, rather than a whole body basis, and thus are likely to be underestimated for both males and females.

3.4.2.1 Surface Water Sources of PCBs

No significant comments were received on Section 3.4.2.1.

3.4.2.2 Sediment Sources of PCBs

No significant comments were received on Section 3.4.2.2.

3.4.3 Avian Exposure Pathways and Parameters

Response to EL-1.16

Life history characteristics were considered in the exposure assessment (see, ERA, p. 55). Many of the receptors have both resident and migratory populations in the Hudson River Valley. Resident populations are considered to be at greater risk and therefore were evaluated in the exposure assessment. Although the tree swallow is a migratory species, continuous spatial exposure duration (i.e., home range of 1.0) was used because tree swallows breed along the banks of river and the young are reared and grow to adult size along the river prior to the autumn migration.

3.4.3.1 Surface Water Ingestion Pathway

Response to EL-1.17

The size of the Hudson River PCB Superfund site easily covers the range of the bird receptors described, excluding migration. Because the migratory species selected as receptors breed along the Hudson River, they were considered as residents. Therefore, both the foraging effort factor of one (1.0) and the assumption that all prey are derived from the Hudson River are appropriate.

3.4.3.2 Incidental Sediment Ingestion Pathway

No significant comments were received on Section 3.4.3.2.

3.4.3.3 Dietary Exposure Pathway

Response to EF-1.22

To estimate the dietary dose of PCBs for piscivorous birds, the data were divided into smaller (< 10 cm) and larger (> 25 cm) fish, where smaller fish included minnows and sunfish whereas larger fish included catfish and bass (see, ERA, p. 52). This approximation is appropriate for purposes of determining exposure because the exposure is expressed as an average concentration in fish of a given size. For example, the bald eagle consumes large piscivorous fish, which were assumed to be represented by average concentration (measured or modeled) in largemouth bass. Different age classes of fish have different feeding strategies, but within a particular age-class, feeding strategies are similar. For example, largemouth bass above 25 cm in length all feed similarly, but differently from fish smaller than that size range. Largemouth bass feeding references are found in Appendix A of Revised BMR (USEPA, 2000a).

Response to EF-1.23

For the mallard duck, estimated PCB concentrations in phytoplankton (calculated from lipid, K_{ow} and concentrations of PCBs in dissolved water) were used as a surrogate for the vegetative component of the mallard diet. This relationship has been shown to provide reasonable estimates in macrophytes and submergent aquatic plant matter (Gobas, 1993; Swackhamer and Skoglund, 1993; and Lovett-Doust et al., 1997) (see, ERA, p. 53). Additional support for this assumption is provided in supported by Gobas et al. (1991). Linear relationships between the plant-water and fish-water bioconcentration factors and the octanol-water partition coefficient have been demonstrated, indicating that plant-water and fish-water exchanges are largely controlled by the chemical's tendency to partition between the lipids of the plants and water. The Lovett-Doust et al. (1997) work on American wild celery uptake of organochlorines, which suggested that simple bioconcentration is an inadequate description of contaminant dynamics between plants, sediment, and water, did not provide enough quantitative information to evaluate potential uptake from sediment sources.

Predicted fish concentrations from the FISHRAND model are annualized, that is, averaged over the entire year. This is likely to underpredict peak summer concentrations and overpredict lowest winter concentrations, but is considered representative of an average that bald eagles might be exposed to over the course of a year. Observed fish body burden measurements are available for September, 1993, and the 1993 toxicity quotients are based on those observations. Dietary dose is expressed in units of mg/kg/day. Implicit in this quantification is the assumption that the period of exposure is commensurate with the period of exposure in the toxicity study. This is a reasonable assumption, given that most toxicity studies are typically over several months, and all of the ecological receptors are exposed in the field for at least several months in a given year.

Response to EF-1.24

Observed fish body burden measurements are available for September 1993, and 1993 toxicity quotients are based on those observations. Predicted fish concentrations from the FISHRAND model are annualized, that is, averaged over the entire year. This is likely to underpredict peak summer concentrations and overpredict lowest winter concentrations, but is considered representative of an average that eagles might be exposed to over the course of a year. Dietary dose is expressed in units of mg/kg/day. Implicit in this quantification is the assumption that the period of exposure is commensurate with the period of exposure in the toxicity study, which is the case for the bald eagle TRVs.

Response to EL-1.18

The size of the Hudson River PCB Superfund site easily covers the range of the bird receptors described, excluding migration. Therefore, both the foraging effort factor of one (1.0) and the assumption that all prey are derived from the Hudson River are appropriate.

Response to EL-1.19

The percent of large fish and of small fish in diets is specified and justified in detail in Appendicies E and F. This is based on USEPA Wildlife Exposure Factors Handbook (USEPA, 1993c), site specific values from NYSDEC wildlife biologists, and the peer reviewed literature.

Response to EL-1.20

The kingfisher diet of 78% fish was selected based on stomach contents from belted kingfishers in south-central New York State and is considered to be representative of kingfishers living along the Hudson River (see, ERA, p. E-8). The right hand column of Table 3-19 reports the range, rather than the average, for each variable. The range for each parameter represents a variety of conditions and habitats.

The diet of the bald eagle is principally fish (see, ERA, pp. E-13 to E-14). PCBs may also be present in non-fish prey of the bald eagle, such as amphibians and reptiles, birds, and mammals, although PCB concentrations may be greater in these species than in prey fish due to bioaccumulation.

3.4.3.4 Behavioral and Temporal Modifying Factors Relating to Exposure

Response to EL-1.21

The size of the Hudson River PCB Superfund site easily covers the range of the bird receptors described, excluding migration. Therefore, both the foraging effort factor of one (1.0) and the assumption that all prey are derived from the Hudson River are appropriate.

3.4.3.5 Biomagnification Factors for Predicting Egg Concentrations

No significant comments were received on Section 3.4.3.5.

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

Response to EF-1.26

The spatial domain of the average daily dose estimates are approximate. The Hudson River is a large, diverse waterbody with many niche areas. The doses represent expected doses for the particular receptor within a reasonable foraging range centered at the sampling location from which the water, sediment, and prey estimates were derived. PCB concentrations in the exposure media generally follow a gradient from the uppermost sampling locations to the sampling locations closest to the New York - New Jersey Harbor. Between these sampling locations, exposure concentrations generally follow a decreasing trend, but there will be areas of slight increases and slight decreases relative to the general trend. It is approximately true that the calculated doses will represent reasonable estimates of dose between sampling locations. The analysis assumes doses relative to the measured concentrations in the exposure media; the exact spatial domain over which those measured concentrations hold is unknown.

Response to EL-1.22

The right hand column of Table 3-19 reports the range, rather than the average, for each variable. The range for each parameter represents a variety of conditions and habitats. From 245 to 365 days per year represents the range of the annual residence time for the kingfisher and adding the two numbers and then dividing by two does not provide the average residence time. A 365-day per year residence time was selected for the belted kingfisher to represent year-round residents of the Hudson River (see, ERA, p. E-8).

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

No significant comments were received on Section 3.4.3.7.

3.4.4 Mammalian Exposure Pathways and Parameters

Response to EL-1.23 and EL-1.27

The home range of 48 hectares (120 acres) discussed in Appendix F (see, ERA, p. F-5) is based on radio-tracking of raccoons conducted by Urban (1970). Table 3-23 of the ERA is revised to correct the units for the home range of the raccoon from km to hectares (see, ERA, Table 3-23). Due to the size of the Hudson River site, it is reasonable to consider that raccoons have full-time residence near the shoreline and procure all the aquatic components of their diet from the river. Although raccoons in northern US populations (e.g., Minnesota) do hibernate, the climate along the Hudson River is considered appropriate for year-round residency and the ERA assumed that raccoons do not hibernate along the Hudson River.

A 34% fish component for the mink diet was used based on the stomach and intestine contents (winter) of mink from various habitats in New York State (Hamilton, 1959). Fish were found in 34.1% of the minks studied, aquatic invertebrates (i.e., crayfish and molluscs) were found in 16.0% of the minks, and amphibians and insects were found in 21.9% and 6.8% of the minks, respectively. The frequencies of fish and invertebrates occurrence were used as inputs to the percent diet composition as aquatic or semi-aquatic prey. The home range of female mink used in ERA was 1.9 km of shoreline, not square-km (see, ERA Table 3-24). Given that the Hudson River PCBs site is about 322 km in length, an area use factor of one (1.0) is considered appropriate.

The work of Hamilton (1961) on river otter diets (see ERA Appendix F, p. F-12) is revised to read as follows: "Hamilton (1961) examined stomach and intestine contents from 141 trapped individuals from the Adirondacks in NYS. He found that fish occurred in 70.0%, crayfish in 34.7%, frogs in 24.8%, aquatic insects in 13.5%, and mammals in 4.3% of the specimens." Percentages were originally presented as dietary composition, rather than percentage of specimens in which prey were found. Recent field observations by Spinola (1999) suggest that the winter diet of the river otter is composed exclusively of fish. Therefore, use of a 100% fish diet for the river otter is considered appropriate for the ERA. A home range of 10 km of shoreline was used for the river otter in the ERA based on radio tracking of relocated river otters in western New York State (Spinola et al., 1998). The maximum 5,700 hectare home range mentioned in Appendix F was for a population living in a mountain valley in Colorado, which is not relevant for the Hudson River population.

Winter diets were used to estimate dietary composition for most mammals to give full consideration to the aquatic dietary components of mammals living along the Hudson River.

No temporal modifying factor was applied to the little brown bat because all food sources used during the year (i.e., active feeding time plus fat reserves used during hibernation) are assumed to be derived from the Hudson River (see, ERA, pp. F-2 to F-4).

3.4.4.1 Surface Water Ingestion Pathway

Response to EL-1.24

The home range for the raccoon is 48 hectares (120 acres), as discussed in Appendix F (see, ERA, p. F-5), based on radio-tracking of raccoons conducted by Urban (1970). Based on the size of the Hudson River site, it is realistic that raccoons have full-time residence near the shoreline and procure all the aquatic components of their diet from the river.

The home range of female mink used in ERA was 1.9 km (not square-km) of shoreline. Given that the Hudson River PCBs site is about 322 km in length, an area use factor of one (1.0) for the mink is appropriate.

3.4.4.2 Incidental Sediment Ingestion Pathway

No significant comments were received on Section 3.4.4.2.

3.4.4.3 Dietary Exposure Pathway

Response to EF-1.25

To estimate the dietary dose of PCBs for piscivorous birds, the data were divided into smaller (< 10 cm) and larger (> 25 cm) fish, where smaller fish consisted of species such as minnows and sunfish while larger fish were composed of such as catfish and bass (see, ERA, p. 52). This approximation is appropriate for purposes of determining exposure because the exposure is expressed as an average concentration in fish of a given size. For example, the bald eagle consumes large piscivorous fish, which were assumed to be represented by average concentration (measured or modeled) in largemouth bass. Different age classes of fish have different feeding strategies, but within a particular age-class, feeding strategies are similar. For example, largemouth bass above 25 cm in length all feed similarly, but differently from fish smaller than that size range. Largemouth bass feeding references etc. are found in Appendix A of Revised BMR (USEPA, 2000a).

Response to EL-1.25

Based on the size of the Hudson River, it is realistic to assume that raccoons, mink, and otter that reside near the shoreline procure all the aquatic components of their diet from the river.

Response to EL-1.26

Information on size selection of fish in the mink and otter diet can be found on pages F-9 and F-12 of the ERA, respectively. The simplified food web assumes that the otter consumes 100% piscivorous (large) fish and the mink and raccoon consume 100% forage (small) fish of the fish they consume.

3.4.4.4 Behavioral and Temporal Modifying Factors Relating to Exposure

Response to EL-1.28

Although raccoons in northern US populations (e.g., Minnesota) do hibernate, the climate along the Hudson River is considered appropriate for year-round residency and it was assumed that raccoons do not hibernate along the Hudson River.

No temporal modifying factor was applied to the little brown bat because all food sources used during the year (i.e., active feeding time plus fat reserves used during hibernation) are assumed to be derived from the Hudson River (see, ERA p. 64).

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

No significant comments were received on Section 3.4.4.5.

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

No significant comments were received on Section 3.4.4.6.

3.5 Examination of Exposure Pathways Based on Congener Patterns

No significant comments were received on Section 3.5.

3.5.1 Introduction

No significant comments were received on Section 3.5.1.

3.5.2 Identifying Aroclor Patterns for Use in Toxicity Assessment

Response to EF-1.27 and EL-1.29

USEPA acknowledges the comment. The first sentence of the fourth paragraph on p. 69 is revised to change the reference from Figure K-16 to Figure K-14. The fourth sentence of the paragraph is revised to change Figure K-17 to Figure K-15. The first paragraph on p. 70 is revised to cite to the correct figures, as follows:

> Component 1 appears to closely match molecular weight. Note the similarity in the trends of component 1 versus molecular weight in fish and sediments as a function of river mile (see Figure K-14 and the top diagram in Figure K-16). As in Figure K-14, the lines in Figure K-16 represent weighted averages and are used to illustrate general trends. Both component 1 and molecular weight show a gradual rise from the TI Pool to New York City harbor with a plateau in the freshwater Lower Hudson for sediments but not for fish. As shown in the lower diagram in Figure K-16, this rise in molecular weight in fish is paralleled only by a rise in the molecular weight of the water column dissolved-phase PCB fraction. Note the similar slope values as well as the high R^2 values relative to the other matrices plotted.

Similarly, paragraph 2 of p. 71 is revised to read:

> In an examination of seasonal and year-to-year variation in the congener patterns of fish body burdens, a combination of the results of Figures K-44, K-45, K-46 and K-47 suggests a minor shift toward higher molecular weights (*i.e.*, heavier congeners) from Fall 1993 to Fall 1995 and Spring 1995. The shift appears to be much greater for the Fall 1993 to Spring 1995 sampling events than from Fall 1993 to Fall 1995. Based on the last diagram in Figure *K-39*, the Spring 1995 results also appear to have a higher molecular weight than that for Fall 1995. These general trends were also noted in the NOAA report (1997) based on several individual congeners. However, these conclusions must be tempered by the confounding factor of life-stage, which was also shown to coincide with changes in molecular weight. Based on these results plus the direct homologue comparisons provided in Figures K-2 to K-4, it appears likely that seasonal variation in fish body burden does occur, with heavier molecular weights coinciding with the Spring. On the other hand, there does not appear to be a systematic change in the conditions in Fall 1995 relative to Fall 1993. There may be some decline in a few specific congeners, but as shown in Appendix K, some of these congeners may reflect a complexity in their biogeochemistry, which precludes their use as simple markers for recently released PCBs.

The discussion regarding the possible causes of the differences between the congener patterns of the fish and those of the sediments is not intended to be a defining discussion. It is provided as a possible explanation and is not critical to the conclusions of the section. That is, fish body burdens vary toward higher molecular weights beyond that which might be expected from the sediments. The sediments are the ultimate record-keepers of the nature of PCBs present in the river, and thus can be used to rule out the occurrence of any substantial additional sources of PCB to the Hudson between the GE facilities in Hudson Falls and Fort Edward, New York and the saline portion of the river. The fact that the molecular weight of fish PCB body burdens increases despite the absence of additional heavier sources to cause this increase implies that other processes, probably internal to the fish, are at work.

Response to EF-1.28

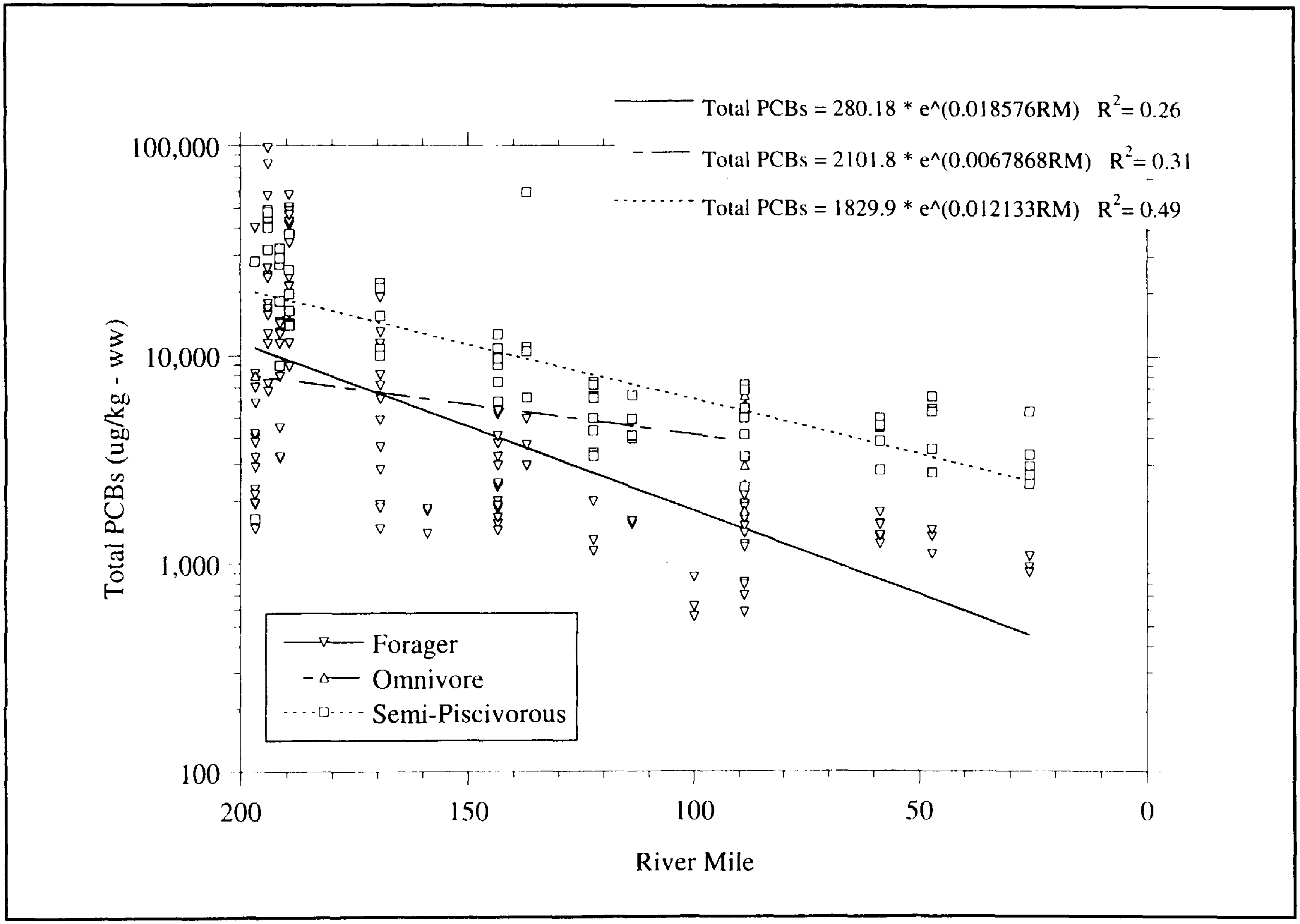
USEPA agrees with the comment. Largemouth bass samples from the Upper Hudson were typically less than 10 cm. Based on a review of all fish data, it became clear that none of the 1993 fish could be correctly classified as piscivores and so this group was removed from the analysis and the samples reassigned based on size. The reclassified results yield the relationships shown in Figures EF-1.28a, EF-1.28b and EF-1.28c. These figures correspond to Figures K-17, K-18 and K-19, respectively. Table NA-28 reflects the reclassified samples replaces Table K-3. Note that the reclassification of several known piscivores in other groups reflects that fact that only juveniles were obtained. Feeding habits of the juveniles would likely be most similar to other fish of comparable size and not to the adult of that species.

The reclassification of the results does not affect the conclusions drawn from these diagrams as they were originally presented in Appendix K. That is, most of the variation in PCB body burden

Table EF-1.28
Feeding Guild Classification for 1993 Hudson River Fish Samples
(Revised Table K-3)

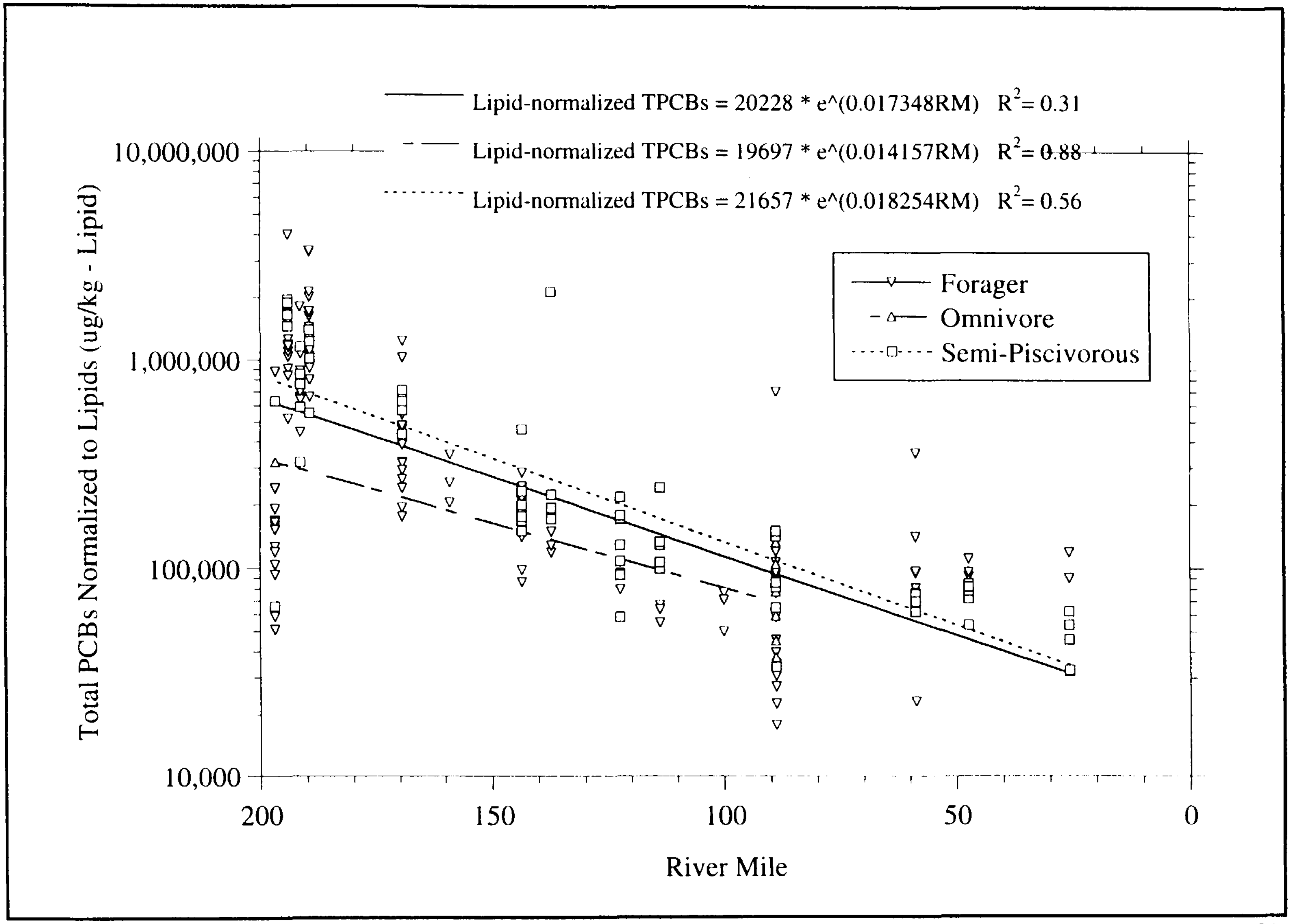
Region	Forager	Semi-Piscivore	Omnivore
Fresh Water Only (RM 196 to 60)	Red Breasted Sunfish Cyprinid Species Tesselated Darter Longnose Dace Sucker Species Largemouth Bass (<= 11 cm) Smallmouth Bass (< 12 cm) Pumpkinseed Spot Tail Shiner Brook Silverside	Rock Bass (15 - 17 cm) Yellow Perch	Brown Bullhead White Catfish
Fresh to Saline (RM 154 to 26)	Atlantic Silverside Striped Bass (< 10 cm)	White Perch	

304128



TAMS/MCA

Figure EF-1.28a (Revised Figure K-17)
Total PCBs Versus River Mile for 1993 Fish Data, Classified by Feeding Guild

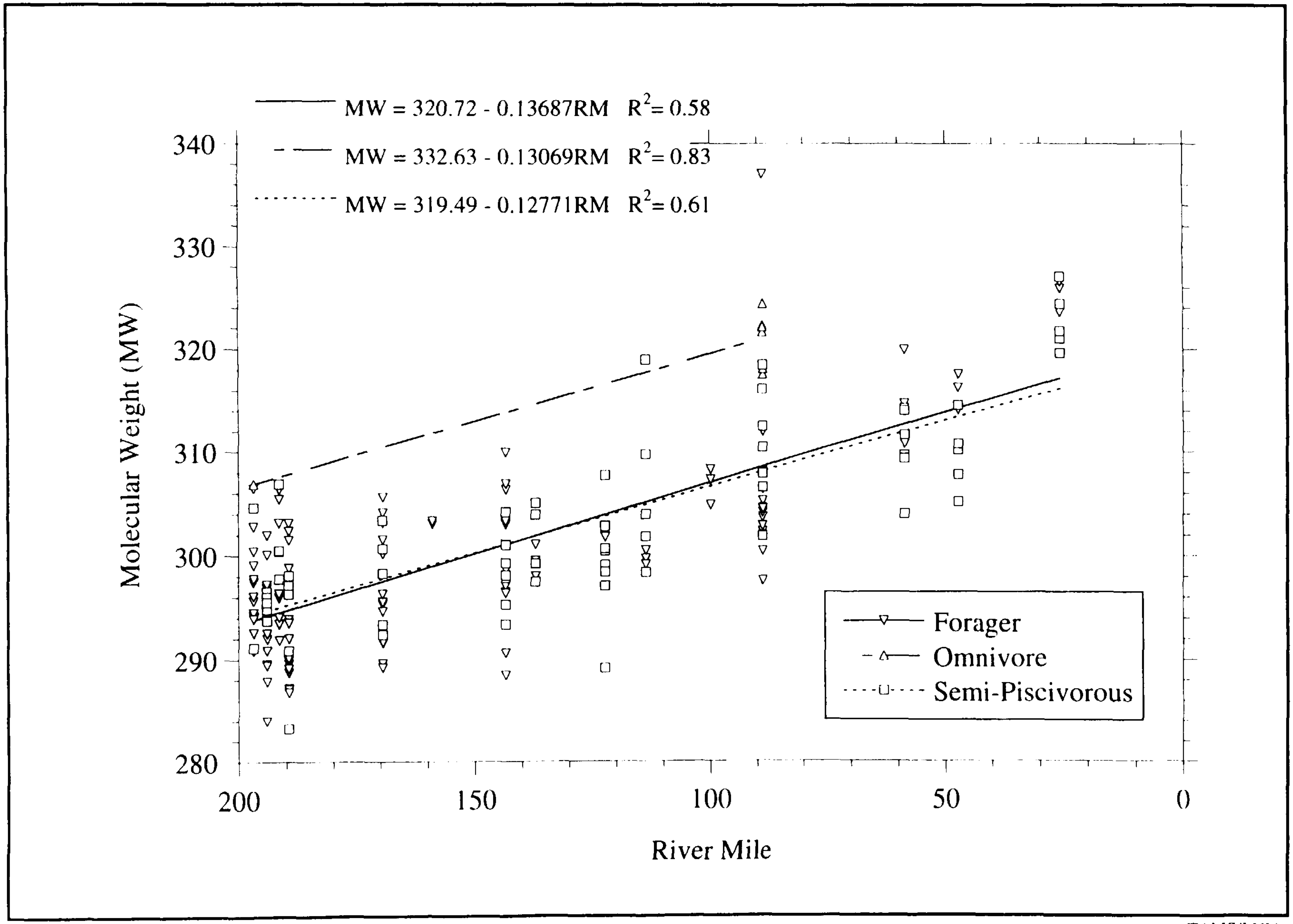


TAMS/MCA

Figure EF-1.28b (Revised Figure K-18)
Normalized Total PCBs Versus River Mile for 1993 Fish Data, Classified by Feeding Guild

304129

304130



TAMS/MCA

Figure EF-1.28c (Revised Figure K-19)
Molecular Weight Versus River Mile for 1993 Fish Data, Classified by Feeding Guild

among species at a given location is attributable to variations in lipid content and not PCB exposure. This assertion is also supported by the consistency in molecular weight in the fish samples for the forage and semi-piscivorous feeding guilds at a given location. The omnivores have a different but parallel relationship in molecular weight *vs.* river mile. This guild is typically associated almost-exclusively with sediment exposure, unlike the other guilds which incorporate both sediment and water pathways.

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

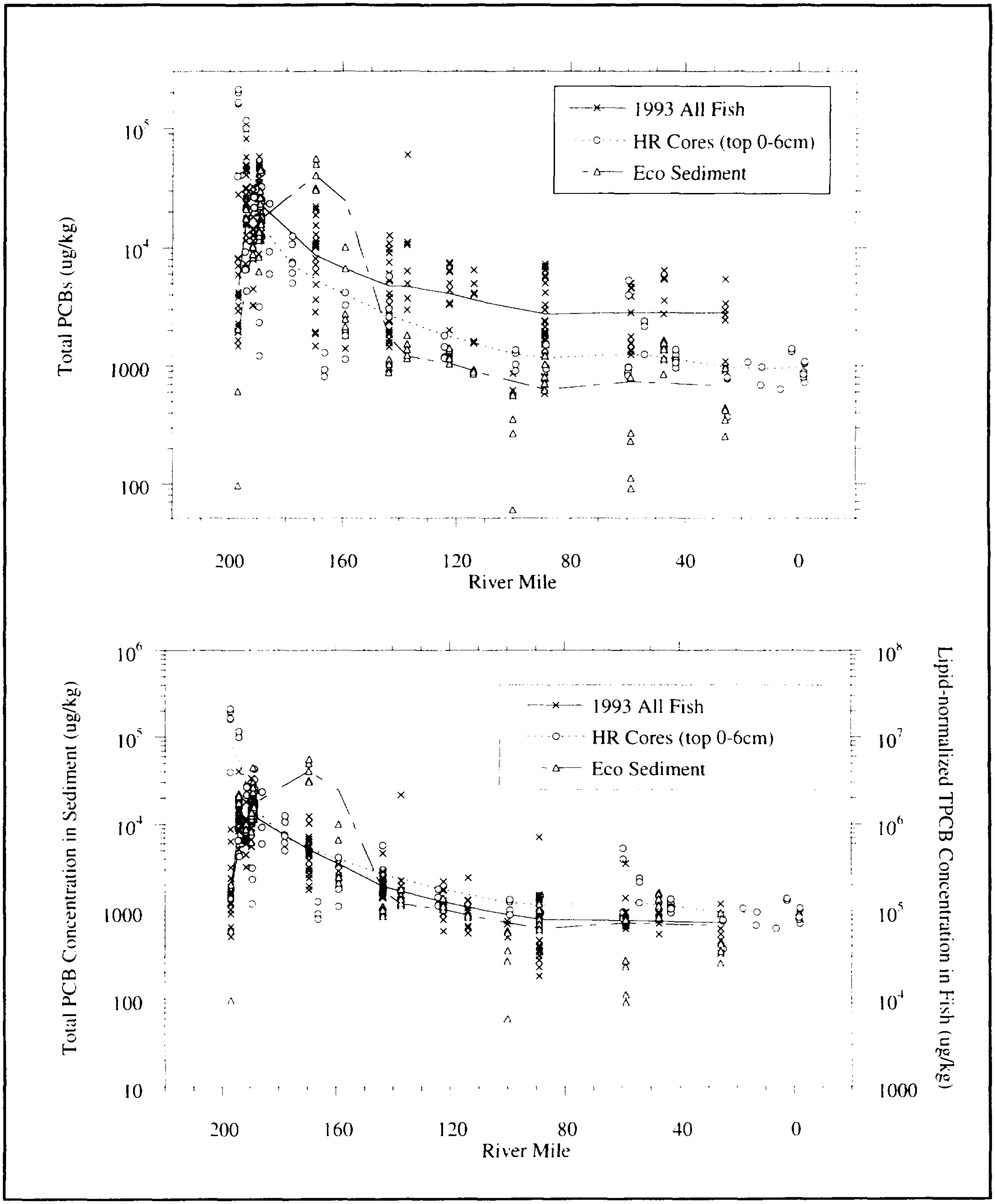
Response to EF-1.29

USEPA agrees with the comment. No piscivorous adults were obtained from the Hudson River as part of the 1993 USEPA sampling. See response to comment EF-1.28 for the effect of this correction on the analysis presented in Appendix K of the ERA.

Response to EF-1.30

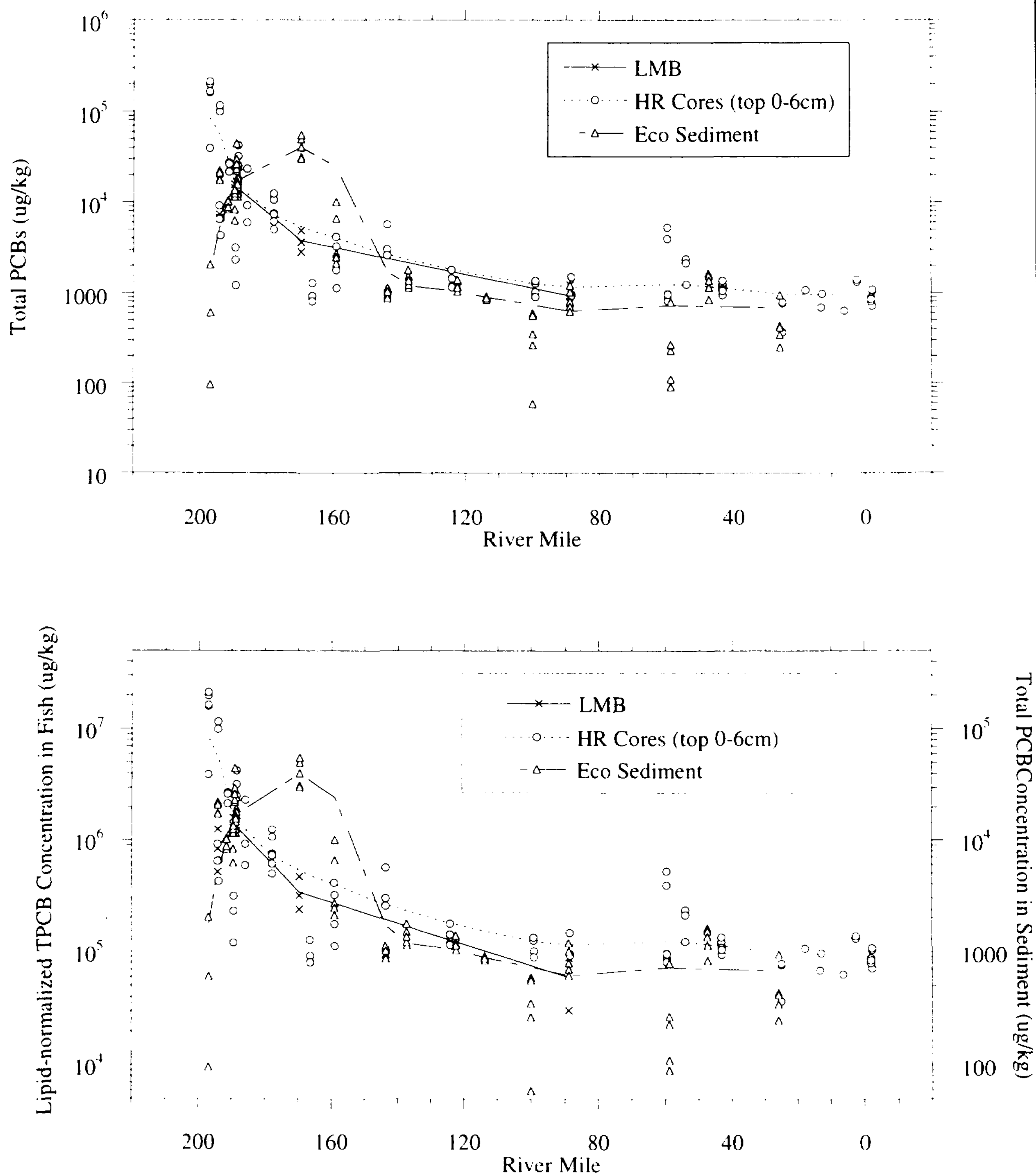
USEPA noted the overall decline in PCBs within sediments and fish throughout the Hudson from RM 200 to 0 (ERA, P. 70). This is illustrated in Figure EF-1.30a. Evident in this figure is the same general level of decline in both sediment and fish concentrations across the region of contamination. Both wet weight and lipid-normalized concentrations are compared to the sediment trend. In each diagram, a weighted-average trend is plotted to show the approximate variation of sediment and fish PCB levels with distance downstream. Note in the lower diagram representing lipid-normalized fish concentrations, the right-hand scale is not the same as the left. This was done to make the results more readily comparable. For both vertical axes, the range is the same when expressed as a function of the lowest scale value (*i.e.*, each scale represents four orders of magnitude). Evident in the diagrams is the consistent degree of decline in both sediments and fish concentrations, nearly 2 orders of magnitude. In most instances, the rates of decline (*i.e.*, the slopes) among sediment and fish are parallel, indicating similar rates of decrease when the river is considered as a whole. It would be inappropriate to consider point-to-point variations in fish and sediment PCB levels within portions of the Hudson, despite apparent trends, because the number of samples at any given location (5 to 7 sediment and 10 to 15 fish) is typically too small to constitue a spatially representative characterization of the location. However, when taken together, these data may be sufficient to characterize the general overall trend. From these diagrams, it is clear that the same level of decline has occurred in both the sediments and fish when considered as a whole.

In Figures EF-1.30b, 1.30c and 1.30d, several individual fish species are examined against river mile. These results show similar levels of decline between wet weight fish concentrations and sediment concentrations. When examined on a lipid-normalized basis (lower diagram in each of the figures), the results still suggest that lipid-normalized concentrations and sediment concentrations decline by approximately the same amount when the entire Hudson is considered.



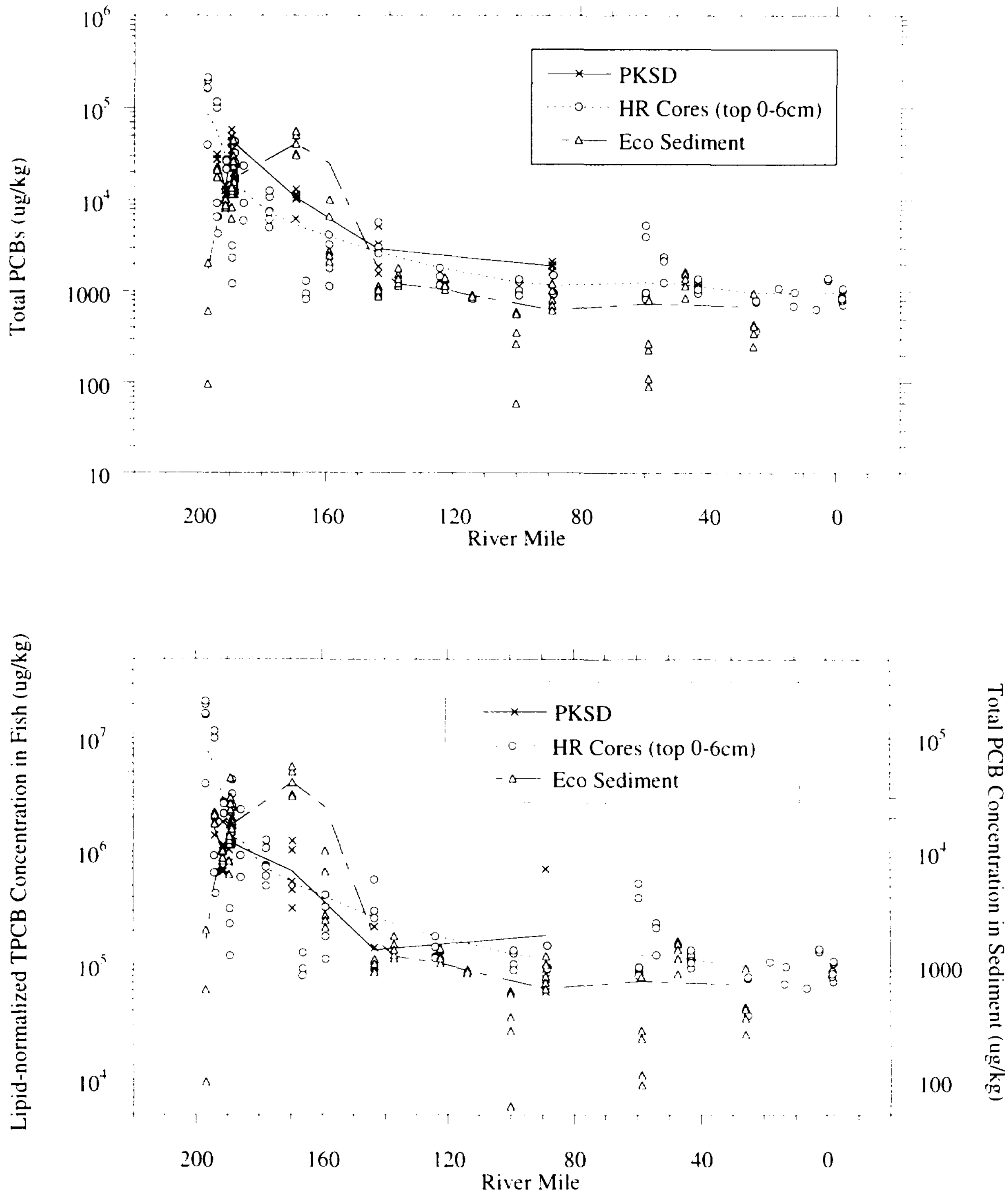
TAMS/MCA

Figure EF-1.30a
Fish Body Burden and Sediment PCB Concentration
Versus River Mile in the Hudson River
(1993 All Fish, Top 0-6 cm of High Resolution Cores and Ecological Sediment)



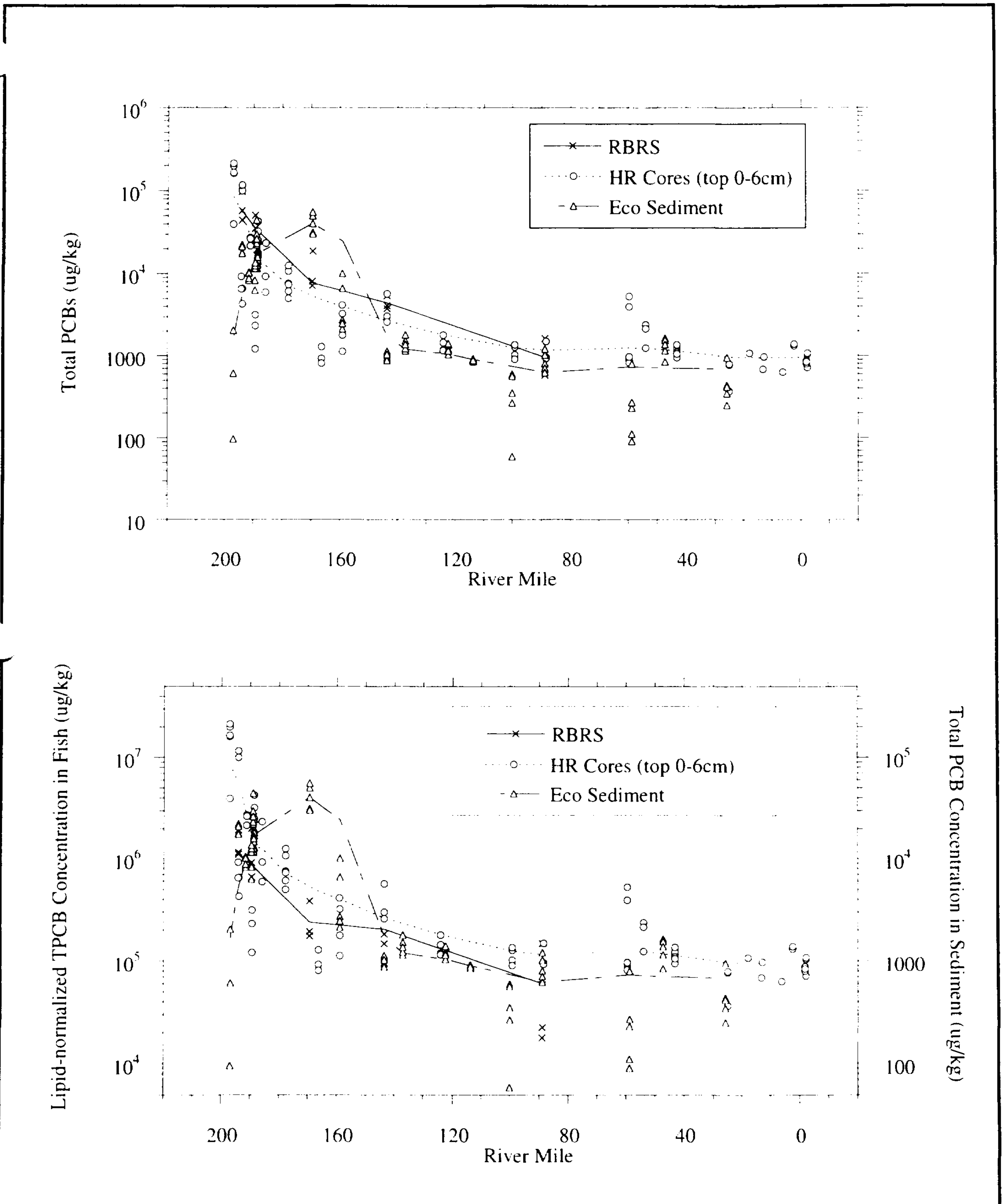
TAMS/MCA

Figure EF-1.30b
Largemouth Bass Body Burden and Sediment PCB Concentration Versus River Mile in the Hudson River (1993 All Fish, Top 0-6 cm of High Resolution Cores and Ecological Sediment)



TAMS/MCA

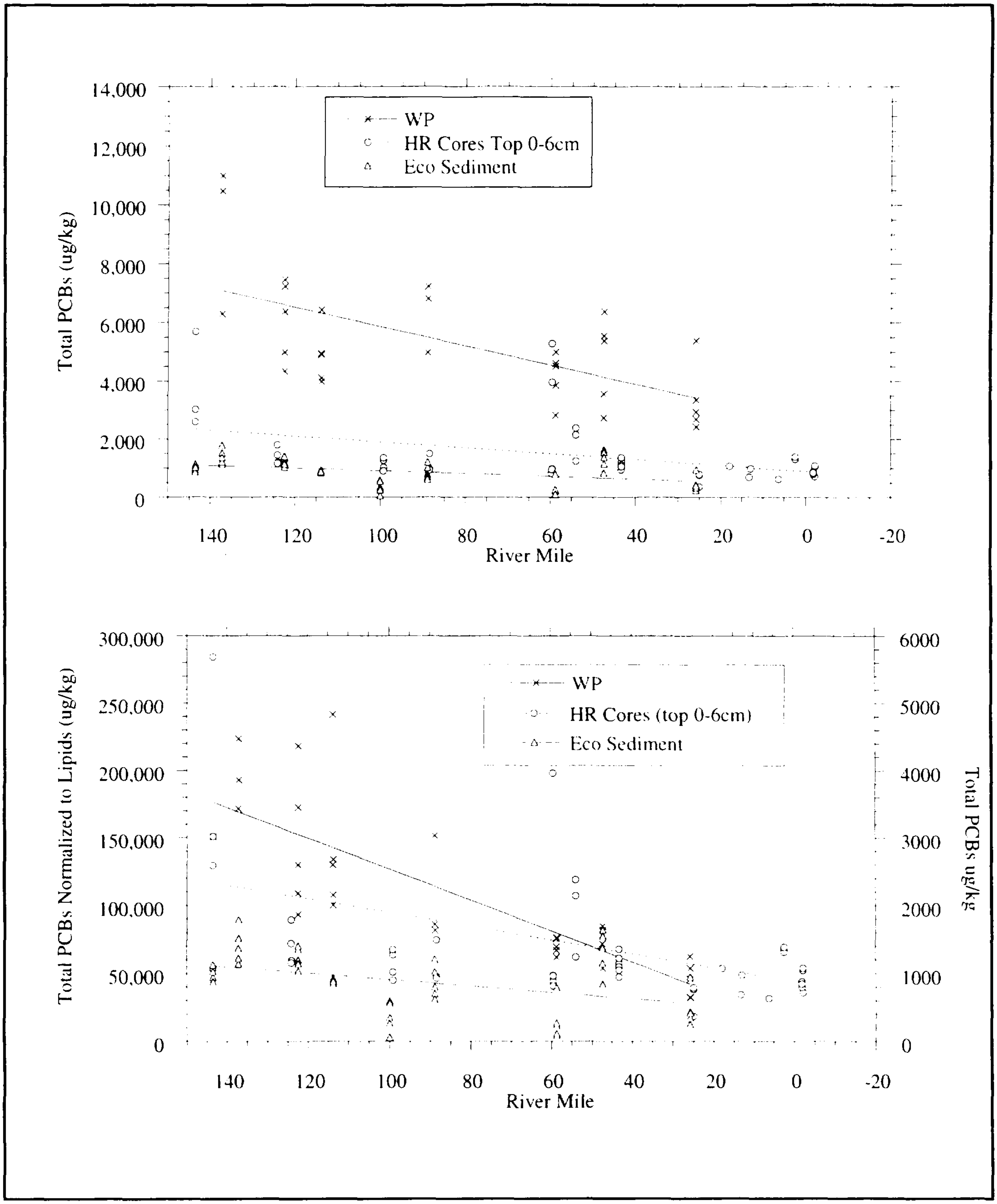
Figure EF-1.30c
Pumpkinseed Body Burden and Sediment PCB Concentration Versus River Mile in the Hudson River
(1993 All Fish, Top 0-6 cm of High Resolution Cores and Ecological Sediment)



TAMS/MCA

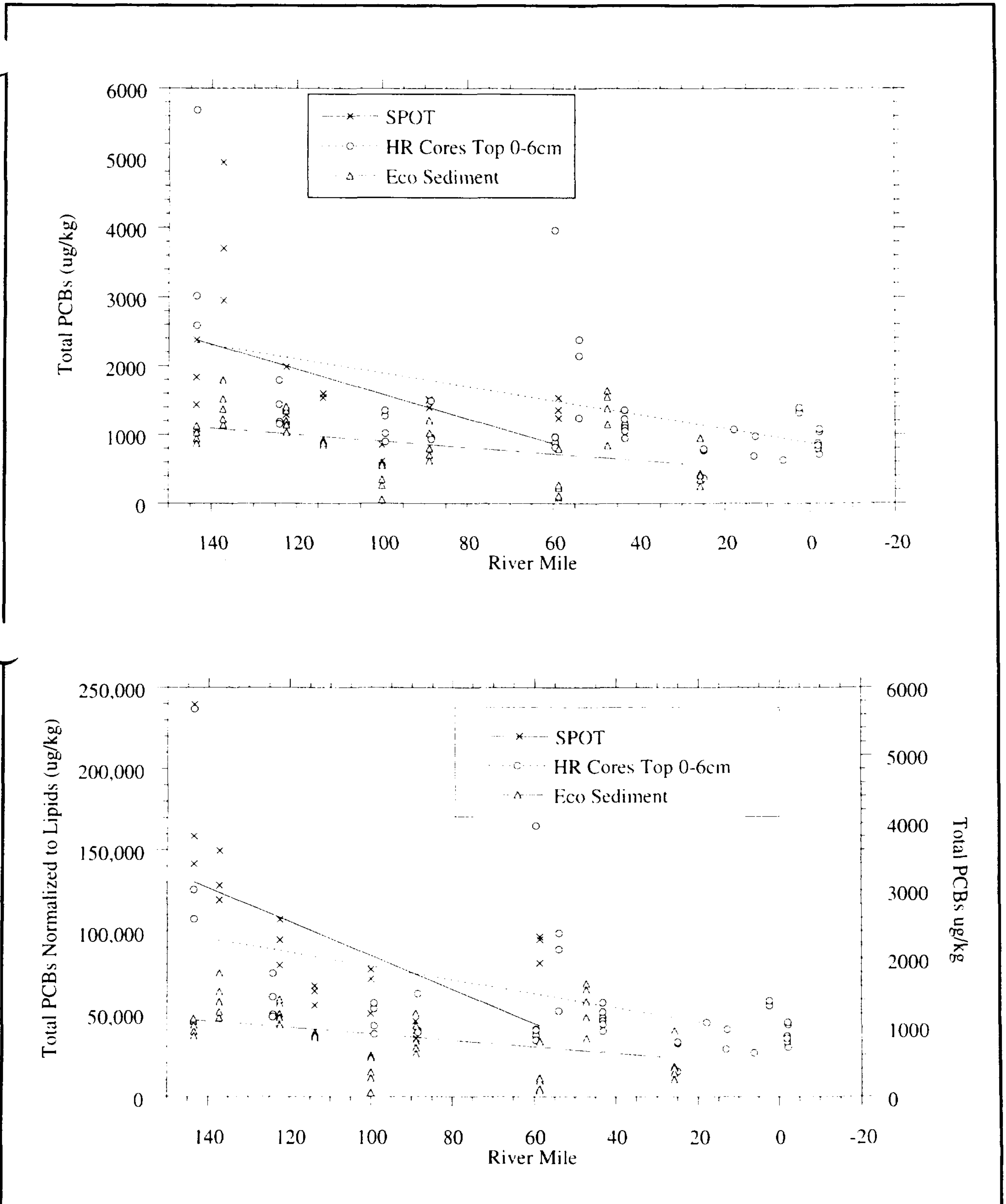
Figure EF-1.30d
Red Breasted Sunfish Body Burden and Sediment PCB Concentration Versus River Mile in the Hudson River
(1993 All Fish, Top 0-6 cm of High Resolution Cores and Ecological Sediment)

304135



TAMS/MCA

Figure EF-1.30e
White Perch Body Burden and Sediment PCB Concentration Versus River Mile in the Lower Hudson River
(1993 All Fish, Top 0-6 cm of High Resolution Cores and Ecological Sediment)



TAMS/MCA

Figure EF-1.30f
Spottail Shiner Body Burden and Sediment PCB Concentration Versus River Mile in the Lower Hudson River (1993 All Fish, Top 0-6 cm of High Resolution Cores and Ecological Sediment)

As compared to the Upper Hudson, the rate of decline in PCBs in the Lower Hudson is less for both sediment and water. In Figures EF-1.30e and 1.30f, the weighted-average trends are presented for both sediment and two species of fish. Note that the scales on these figures are linear and not logarithmic. Both figures show an overall lessening of the slope, meaning a slower rate of decline. The commentor is correct in noting that for this region, fish concentrations decline perhaps two times faster than those of the sediments in the Lower Hudson. This is evident on both a wet weight and a lipid-normalized basis for these species. The decline of PCB levels in fish in this region may result from changes in the relative importance of water and sediment exposure to PCBs. For example, water-based exposure may decline in this region due to losses via processes such as gas exchange.

Response to EF-1.31

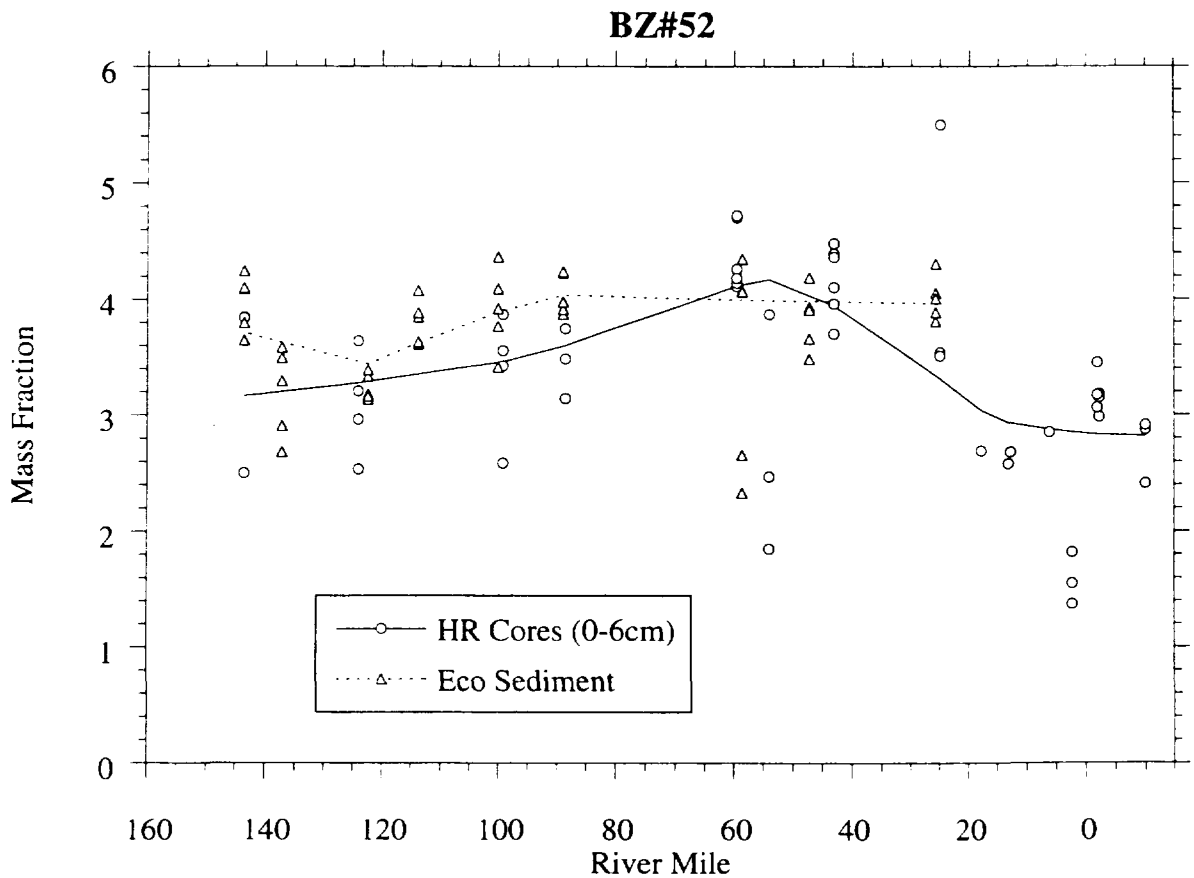
USEPA agrees that some of the change in congener composition and concentration in the water column may be due to the loss of lower chlorinated congeners from the water column. Loss of lower chlorinated congeners may result from gas exchange as well as preferential loss of selected congeners due to *in situ* degradation within the water column (Garvey *et al.*, 1999). This would yield the heavier mixtures seen in the water column and potentially affect fish in the same manner.

Response to EF-1.32

Evidence for the occurrence of heavier PCB mixtures in the NYC harbor area was documented in the DEIR (see, USEPA, 1997a). While there are no data that specifically document the transport of suspended solids upstream in the Hudson, there is much information documenting upstream transport of other NYC harbor-related discharges such as phosphates and nitrates (*e.g.*, Deck, 1980). PCBs are also transported upstream, as shown by the ratios of several heavier congeners relative to BZ#52 in surface sediments below the salt front.

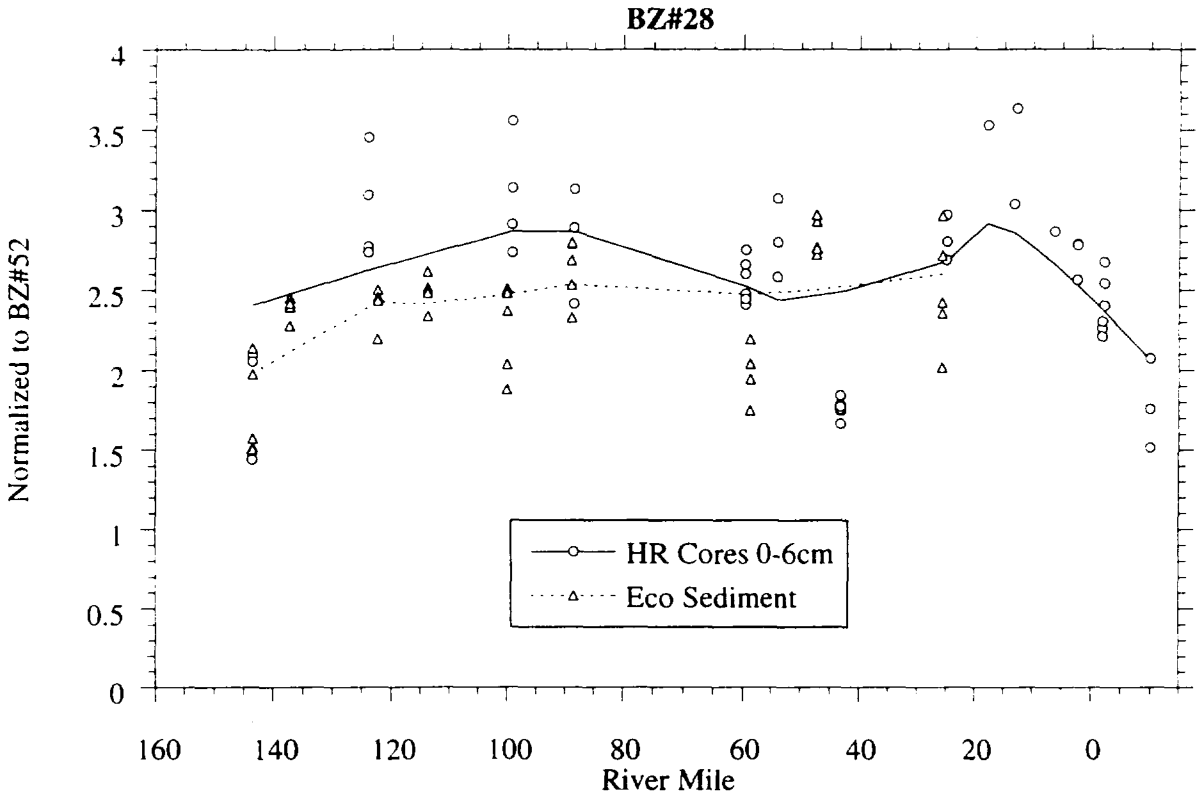
In Figure EF-1.32a, the mass fraction of BZ#52 in the sediments is plotted as a function of river mile. This congener shows a relatively consistent mass fraction throughout the Lower Hudson above RM 60, the approximate location of the salt front. Greater variability is present below this point but there is no trend to a higher or lower value with river mile. This most likely results from several conditions. First, it is a stable congener which is not subject to substantive degradation in the sediments of the Lower Hudson. Second, BZ#52 has comparable mass fractions in both Aroclor 1242 and 1254 (about 4 to 5 percent). Thus, if these Aroclors are the principal forms of PCBs added to the Hudson, then the mass fraction of BZ#52 will tend to remain constant with river mile. It should be noted that additions of Aroclor 1260 will serve to lower the BZ#52 mass fraction in the sediments because this congener is present at less than one percent in the Aroclor.

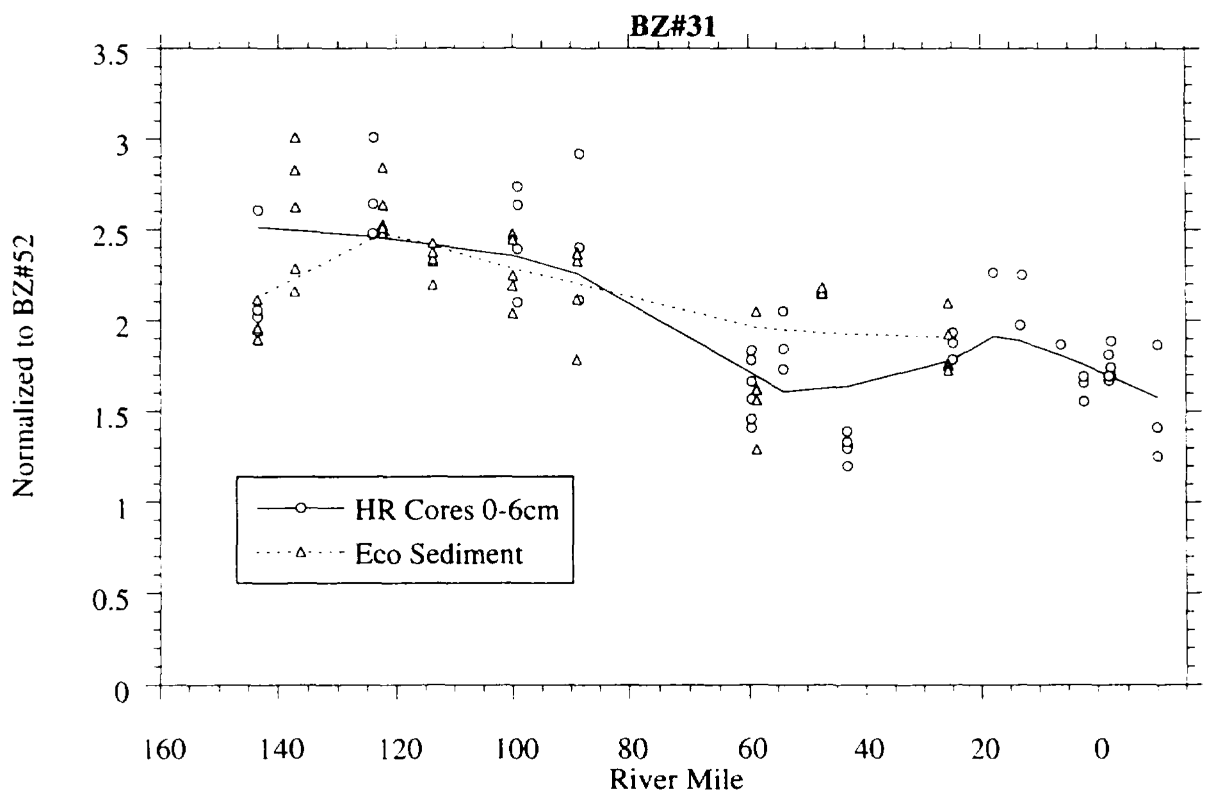
Using BZ#52 as a basis for subsequent comparison, several congeners were compared via their ratio to BZ#52. In Figure EF-32b, the ratios of BZ#28 and BZ#31 relative to BZ#52 in Hudson River sediments are presented. These congeners appear without trend (BZ#28) or with a downward trend with decreasing river mile (BZ#31). A downward trend with respect to BZ#52 would be



TAMS/MCA

Figure EF-1.32a
Mass Fraction of BZ#52 Versus River Mile in the Lower Hudson River





TAMS/MCA

Figure EF-1.32b
BZ#28 and BZ#31 Normalized to BZ#52
Versus River Mile in the Lower Hudson River

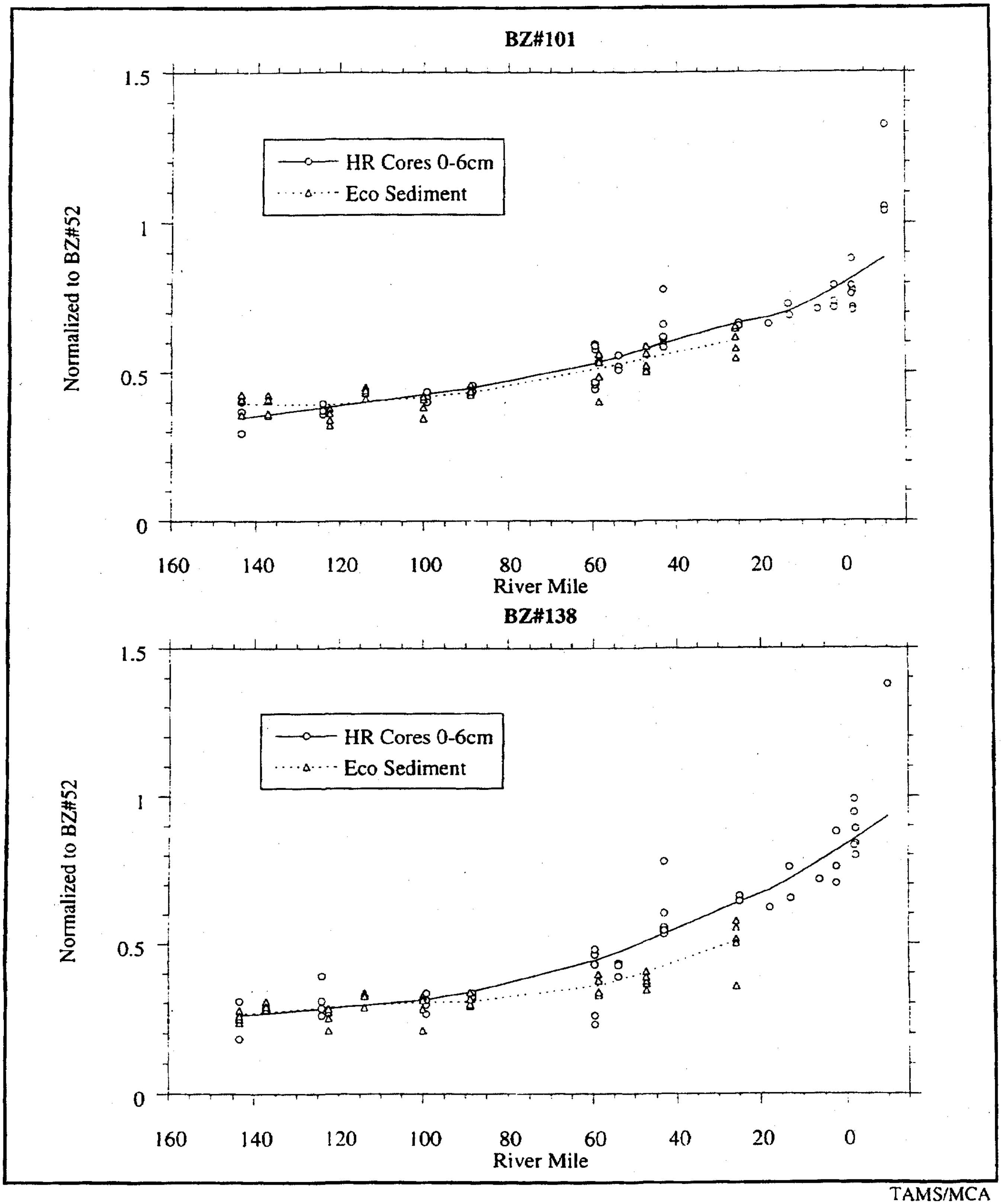
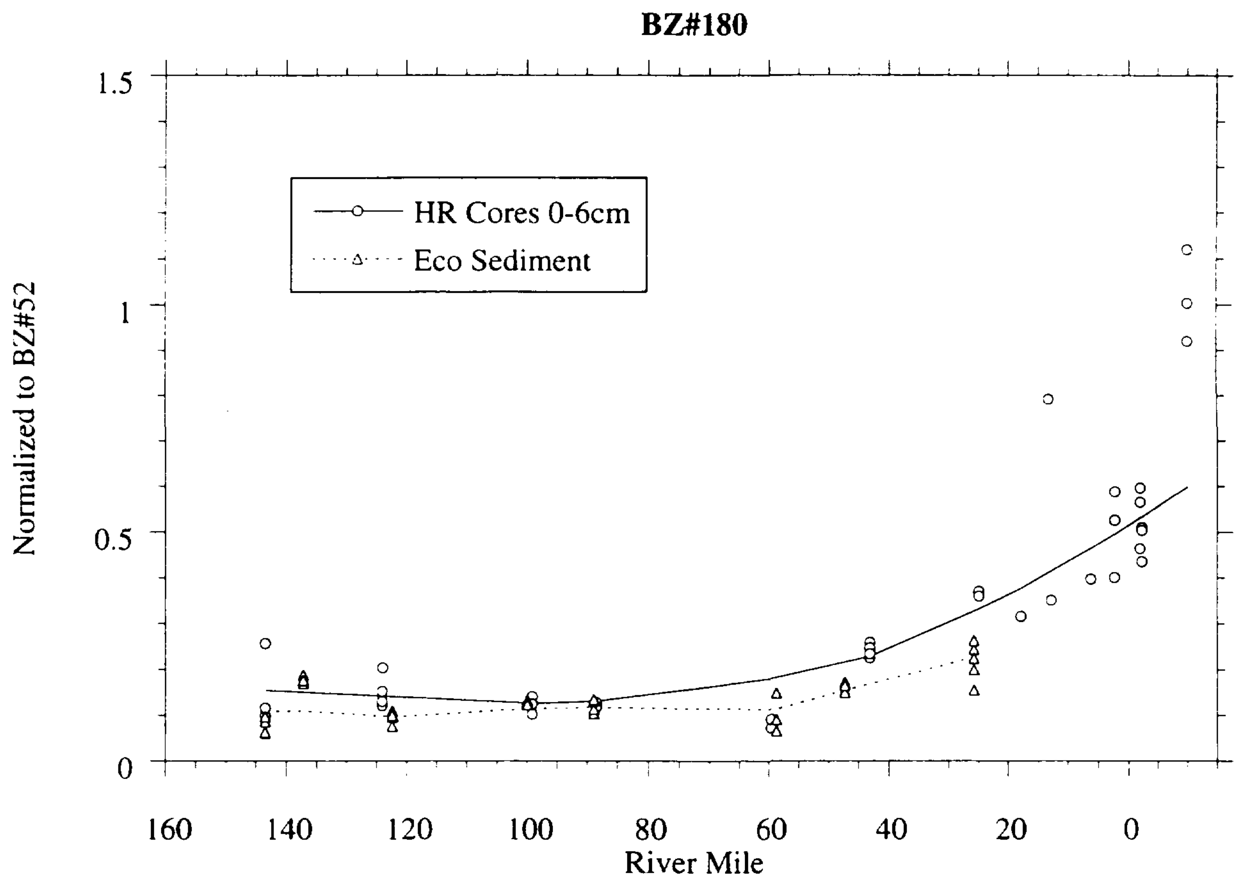


Figure EF-1.32c
BZ#101 and BZ#138 Normalized to BZ#52
Versus River Mile in the Lower Hudson River



TAMS/MCA

Figure EF-1.32d
BZ#180 Normalized to BZ#52
Versus River Mile in the Lower Hudson River

expected if Aroclor 1254 were being added to the river, because this would add BZ#52 without adding the lighter congeners. Note that the downward trend in BZ#31 does not begin until RM 60.

Three congeners heavier than BZ#52 were also examined in Hudson River sediment and are presented in Figures EF-1.32c and EF-1.32d. All three congeners show a rise in ratio with decreasing river mile below RM 60 (i.e., the salt front). A statistical test for trend in the region between RM 150 and RM 60 yielded no significant slope between these two river miles, indicating that these ratios are constant throughout this region. Absence of change in this region is indicative of the absence of additional sources above the salt front besides those attributable to GE.

Conversely, the trends below the salt front (RM 60 and lower) strongly implicate an additional source of heavier congeners in this region. Notably, the trends in BZ#101+90 and BZ#138 appear to rise before that of BZ#180. As discussed in the DEIR, BZ#101+90 and BZ#138 are major components of Aroclor 1254 while BZ#180 is a major component of Aroclor 1260. These differences in trend, combined with the variations seen in BZ#52 below RM 60, may be indicative of two source areas below the salt front. One source, more Aroclor 1254-like, may be present around RM 40 to RM 60 while a second source, consisting of both Aroclor 1254 and Aroclor 1260-like materials, may be present in the lower harbor area. Evidence for the lower harbor source was documented in the DEIR by the core collected at Newtown Creek (USEPA, 1997a).

The fact that the ratios rise fairly steadily with decreasing river mile strongly suggests that the harbor source of heavier congeners is much larger than that at RM 40 to RM 60. To a large degree, the trends in congener ratios result from the upstream transport of the harbor source via estuarine circulation accompanied by mixing with the upstream freshwater load originating from the Upper Hudson. The improved understanding concerning the possibility of a second, smaller source area was obtained from this analysis as well as a principal components analysis of the high resolution core data (Responsiveness Summary for Peer Review 2, USEPA, 2000b, in preparation).

Other evidence presented by Bopp and Simpson (1989) also documents the occurrence of additional loads in the harbor area based on sediment concentrations obtained from several high resolution cores obtained in the 1980s in the Lower Hudson. Similar coring data obtained by USEPA in 1992 would also suggest the occurrence of additional loads in the harbor area based on the failure of surface sediment levels to decline through the harbor.

The USEPA agrees that in the region above the salt front, factors other than additional local sources must come into play in order to cause the change in PCB molecular weight in fish. This is based on the consistency of PCB molecular weight in the sediments as a function of river mile in the freshwater Lower Hudson as well as the demonstrated consistency of the total PCB/cesium-137 ratio discussed in the DEIR (USEPA, 1997a). Both of these conditions would not exist if substantive additional loads were occurring. Further supporting evidence will be provided in a principal component analysis being prepared for the Responsiveness Summary for Peer Review 2 (USEPA, 2000b, in preparation).

Finally, note that although the sediment data clearly show the presence of heavier congener mixtures in the sediment in the saline portion of the Hudson, these heavier mixtures do not represent a significant portion of the PCB mass in the sediment or fish in the upper portion of this region. Indeed, as noted in the DEIR (USEPA, 1997a), the sediment record indicates that not until RM 0 does the PCB contribution from the harbor match or exceed the contribution from upstream (*i.e.*, from GE-related discharges). Additionally, despite the occurence of additonal sources of PCBs in this region, sediment PCB concentrations decline with river mile as would be anticipated from dilution of the GE-related load from upstream. As recorded in the sediments north of Piermont (RM 25), PCBs related to the GE discharges are the main source of fish and sediment contamination in this region of the Hudson.

Response to EF-1.33

The USEPA disagrees that NYC metropolitan area discharges are unlikely to affect the Piermont area (RM 25). Benthic invertebrate samples from the saline Hudson show evidence of minor sources of more chlorinated PCBs to their body burdens in addition to the major source from the GE capacitor manufactuing facilities in the Upper Hudson. There is strong congener ratio evidence which suggests that heavier congener mixtures originate in New York harbor and are transported upstream. Similar evidence of upstream transport was obtained by Chilrud (1996) concerning metal contamination in the harbor. His results showed nearly constant metal concentrations of metals throughout the harbor, indicating rapid mixing and horizontal transport of any particle-bound contamination prior to deposition. These processes would also be expected to apply to PCBs originating in this region as well. Additionally, there is also evidence of other NYC metropolitan area sources to the river. In particular, the USEPA has noted the occurence of a minor source of PCBs in the region between RM 40 and 60 (Responsiveness Summary for Peer Review 2, USEPA, 2000b, in preparation). There is also a documented local source of PCBs in the region near Piermont. Specifically, the Anaconda Wiring (ALCOA) site is located along the river at Harbor at Hastings. PCBs associated with this site, like the other noted NYC metropolitan sources, contain a more chlorinated mixture of PCBs than that associated with the GE discharges.

Response to EL-1.30

Although tidal influences can be measured near the Albany/Troy area (RM 154), tidal mixing does not extend north of the salt water front, which is generally near Newburgh (RM 60). Salinity itself provides a tracer (or signal) for discharges that enter the Hudson at or near the mouth of the river. These discharges can be carried as far north as 60 to 70 miles north of the Battery but no farther; the volume of freshwater flow from the Hudson at this point prevents further upstream movement of salinity and any associated constituents. The large difference in water density between freshwater and saltwater results in a quasi-two layer flow regime in the saline portion of the Hudson River estuary. Transport of salt and New York City-related contaminants to RM 60 is only possible because of the dimensions of the estuary and the quasi-two layer flow. Without this quasi-two layer flow, normal upstream tidal displacement would be expected to be only about 6 to 10 miles.

4.0 EFFECTS ASSESSMENT

Response to EF-1.5

Although Clophen A50 was not used in the United States, the chlorine content of Clophen A50 (50% chlorine) is reasonably similar to the chlorine content of Aroclor 1248 (48% chlorine) and Aroclor 1242 (42% chlorine), which were released into the Hudson River. The chlorine content of Hudson River fish resembles that of Aroclor 1254 (54% chlorine), which is more similar to the chlorine content of Clophen A50, than to that of Aroclor 1248 or 1242 (see, ERA, Appendix K). Therefore, Clophen A50 is a reasonable surrogate of the actual environmental composition of PCBs in Hudson River fish.

The only other reproductive endpoints that Bengtsson et al. (1980) reported to be significantly different in PCB-exposed fish as compared to control fish is the hatching time. Fish in the medium and high exposure groups had significantly reduced hatching times compared with the control group. Although the premature hatching of the exposed fish was thought to result in the premature death of the fry, this result was not tested statistically. Also, differences in the time to spawning and the temperature at which spawning occurred between exposed fish and control fish were not tested statistically. Therefore, hatching time, spawning time, and spawning temperature were not considered as toxicity endpoints in the development of TRVs from this study.

The study by the U.S. Army Corps of Engineers (USACE) was not used for development of laboratory-based TRVs because the sediment used in the study was collected from the field, rather than prepared in the laboratory. Because the field collected sediments contain contaminants other than PCBs, the study is considered to be a field study that could be used to develop NOAEL-based TRVs. A field-based TRV based on the USACE 1988 study was developed and used in the revised risk calculations contained in Section IV of this responsiveness summary.

The study by Hansen et al. (1974) was not identified by the literature search that was conducted for the ERA or the review that was conducted by NOAA (1999b). Laboratory-based TRVs based on the Hansen et al. 1974 study were developed and used in the revised risk calculations contained in Section IV of this responsiveness summary. See response to comment EF-1.39 for further discussion of this study.

Response to EF-1.6

The ERA outlines the criteria that were used to examine the appropriateness of laboratory- and field-based studies (see, ERA, pp. 79-81). The general methodology used to derive toxicity reference values (TRVs) in the ERA was based on approaches that were developed and used by USEPA to derive wildlife criteria (USEPA, 1995) and by Oak Ridge National Laboratory (Sample et al., 1996) to develop toxicological benchmarks. The methodology described the ERA provides a general framework for selection of individual scientific studies for use in calculation of species-specific TRVs. In all cases, best professional judgment was used to select the most appropriate individual scientific studies. The approach used in the ERA develops TRVs on the basis of individual studies that examine appropriately sensitive endpoints, such as reproduction, rather than less sensitive endpoints, such as adult mortality. The ERA approach selects individual studies that examine sensitive endpoints, rather than averaging the endpoints determined for a variety of more or less sensitive endpoints.

Response to EP-2.4

The available toxicological data are insufficient to conduct a probabilistic effects assessment, such as a Monte Carlo analysis, that addresses the range of toxicological effects of PCBs on various organisms within a habitat.

4.1 Polychlorinated Biphenyl Structure and Toxicity

No significant comments were received on Section 4.1.

4.1.1 Structure-Function Relationships of PCBs

No significant comments were received on Section 4.1.1.

4.1.2 Metabolic Activation and Toxicity of PCBs

No significant comments were received on Section 4.1.2.

4.1.3 Estimating the Ecological Effects of PCBs

Response to EF-1.35

The ERA did not examine effects from congeners that have different mechanisms of action from the dioxin-like congeners because insufficient toxicity information is available to determine the effects of non-dioxin-like congeners on ecological receptors. To some extent, these potential effects are addressed through the evaluation of total PCBs. It is not known whether this underestimates true risks.

Response to EG-1.27

The TEQ approach was only one of several approaches used in in the ERA to evaluate potential risks posed by PCBs in the Hudson River. The TEQ approach provides an order-of magnitude estimate of potential risk, which is appropriate for use in the ERA.

Response to EG-1.28

The analysis of the TEQ approach is presented in the text (see, ERA, pp. 38-40). The results are presented in the tables (e.g., Tables 4-7, 4-8, 4-15, 4-16, etc.). Supporting data are provided in Appendix J of the ERA.

4.2 Selection of Measures of Effects

Response to EF-1.34

The NOAA (1999b) report considered reproductive, developmental, and immunotoxic effects on fish. These effects were selected as biological endpoints that are both sensitive to anthropogenic contaminants and ecologically relevant. In the ERA, a more narrow definition of ecologically relevant endpoints was used that included reproductive and developmental effects, but not immunotoxic, sublethal, and other biochemical effects. These other categories of effects were not included because these effects are often less clearly related to ecologically relevant endpoints than are developmental and reproductive effects.

Response to EF-1.36

USEPA agrees that PCBs are known to cause severe adverse effects other than lethality, growth, and reproductive effects, such as immunosuppression. Although immunosuppression can be linked indirectly to population-level effects because organisms with suppressed immune systems are more likely to succumb to disease, this toxicological endpoint is not a direct link to population effects, such as reproductive effects like early life stage lethality.

Response to EG-1.29

The analysis provided in the ERA provides an indication of the potential for risk from dioxin-like congeners. The toxicity factors were based on studies with dioxin, and the TEQ approach has been presented and discussed in the peer-reviewed literature. The statement in the NOAA (1999b) study ("it is currently not possible to evaluate the risk to Hudson River fish larvae from exposure to co-planar PCBs using the TEQ method") referred to the NOAA data set (Monosson, 1999). NOAA concluded that the TEQ analysis may understate the TEQ for the Hudson River fish because not all of the Ah-active congeners were analyzed, in addition to the potential for large interspecies differences.

4.2.1 Methodology Used to Derive TRVs

Response to EG-1.18, EG-1.20, EG-1.21

The lowest appropriate NOAEL and LOAEL TRVs were selected using the methodology described in the ERA (pp. 79-81). The ERA uses the approach that was used in USEPA risk assessment guidance to derive wildlife criteria (USEPA, 1995) and by Oak Ridge National Laboratory (Sample et al., 1996) to develop toxicological benchmarks. Best professional judgment was used to select the most appropriate individual scientific studies. In cases for which no information is available on the sensitivity of a receptor of concern, it is protective, and therefore reasonable, to assume that the receptor could be as sensitive as the most sensitive species tested. The TRVs used in the ERA are comparable with those used at other Superfund sites.

For example, for the Sheboygan River, TRVs for mink for total PCBs were the exact same values as those used for the Hudson River. The Fox River ecological risk assessment TRVs for the mink for total PCBs were 0.0021 mg/kg/day for the LOAEL and 0.099 mg/kg/day for the NOAEL. These values are lower than those selected for the Hudson River. The TEQ-based NOAEL developed for bald eagles for the Fox River site was 7 ng/kg egg, while the Hudson River TRV was 10 ng/kg egg. The Fox River ecological risk assessment used 0.75 mg/kg wet weight as a NOAEL for all fish species. Hudson River NOAELs were developed specifically for each individual fish species and range from 0.3 mg/kg wet weight to 5.25 mg/kg wet weight.

<u>Response to EL-1.33, EL-1.34, EL-1.36, EL-1.37, EL-1.38, EL-1.39, EL-1.40</u>

Averaging values from different studies would provide an average value, but not necessarily a protective one for the receptor species of concern.

The methodology used in applying a subchronic-to-chronic uncertainty factor is described in the ERA (see, pp. 80-81). For the belted kingfisher, a subchronic-to-chronic uncertainty factor was applied to the NOAEL and LOAEL values from the Scott (1977) laboratory study to account for the short-term exposure (4 or 8 weeks); the last sentence in the first paragraph of Section 4.2.4.3 (p. 100) is deleted.

The use of toxic equivalency factors to convert concentrations or doses of PCBs to dioxin equivalents is well established (e.g., Van den Berg et al., 1998) and is used throughout the ERA to estimate risk.

4.2.2 Selection of TRVs for Benthic Invertebrates

<u>Response to EG-1.26</u>

The SECs were used as sediment guidelines, not TRVs, in the ERA to assess sediment quality at the 19 sampling stations in the Upper and Lower Hudson River (see, ERA, pp. 121-122). Their use as guidelines is consistent with USEPA risk assessment protocols and accepted scientific practice.

4.2.2.1 Sediment Guidelines

Response to EF-1.37

USEPA (1993b) sediment guidelines for protection of fish, birds, and mammals based on TCDD concentrations would also be appropriate for inclusion in the discussion of sediment guidelines (see, ERA, pp. 121-122). However, concentrations of TCDD in sediment associated with risk to Great Lakes receptors (e.g. fish, birds, and mammals) were back-calculated using measured biota-to-sediment-accumulation factors (BSAF) that are specific to the Great Lakes, these concentrations are not directly comparable to risk-based concentrations that would be calculated for the Hudson River. These measured BSAF are based on lipid content of fish and organic carbon content of sediment that are specific to the Great Lakes. In addition, these measured BSAFs are specific for the Great Lakes food chain and are likely to be different from those measured for a riverine food chain. Therefore, the risk-based sediment concentrations for the Great Lakes (pg TCDD/g dry weight of sediment) are not directly comparable to the Hudson River because the are based on measured BSAFs that are specific for fish lipid content, sediment organic carbon content, and food chain dynamics of the Great Lakes.

The guidelines selected for use as a measurement endpoint are those considered to be the most relevant for this study area, which are the consensus-based sediment effect concentrations for PCBs in the Hudson River Basin (NOAA, 1999a). Concentrations of PCBs in Hudson River sediments also were compared to NYSDEC (1998) and Washington Department of Ecology (1997) sediment guidelines (see, ERA, p. 122 and Tables 5-8 and 5-9).

4.2.2.2 Body Burden Studies

No significant comments were received on Section 4.2.2.2.

4.2.3 Selection of TRVs for Fish

Response to EF-1.38 and EL-1.31

Much of the recent data on effects of PCBs and dioxin-like compounds on fish have been developed using egg injection studies and are expressed as concentrations of contaminant per wet weight of egg. However, much of the site-specific data for the Hudson River have been collected and expressed as concentrations of PCBs in whole adult fish (wet weight). Because hydrophobic PCBs tend to partition into lipids, and eggs can have significantly different lipid contents than adult tissue, effects concentrations that are presented on a wet weight basis in eggs are not directly comparable to measured or modeled concentrations in adult fish that are expressed on a wet weight basis. However, effects concentrations in eggs expressed on a lipid basis can be compared to concentrations measured or modeled in adult fish tissue that are also expressed on a lipid basis. The underlying assumption is that adult fish with a particular body burden of PCBs per unit of lipid will produce eggs that have a similar burden of PCBs per unit of lipid. Evidence to support this assumption is presented in Nimii (1983).

Therefore, the TRVs presented in the ERA for individual species of fish were presented as the lipid-normalized concentrations in adult fish that are equivalent to the lipid-normalized effects concentrations (NOAEL, LOAEL) in eggs. These lipid-normalized TRVs for adult fish could, of course, also have been back-calculated and expressed as wet weight concentrations. However, it is simpler and easier to clearly present the derivation of these TRVs presented on a lipid-normalized basis.

In summary, the ERA does not assume that lipid-normalized concentrations are more directly related to reproductive effects than the wet weight concentrations in eggs. The ERA does assume, however, that the wet weight effects concentration in eggs is not directly comparable to the wet weight concentration in adults. The ERA assumes that the lipid- normalized effects concentration in eggs is comparable to the lipid-normalized concentration in adult tissue.

Response to EF-1.39

Although Clophen A50 was not used in the United States, the chlorine content of Clophen A50 (50% chlorine) is reasonably similar to the chlorine content of Aroclor 1248 (48% chlorine) and Aroclor 1242 (42% chlorine) that were released into the Hudson River, and to the chlorine content of Hudson River fish which resembles that of Aroclor 1254 (54% chlorine).

The only other reproductive endpoints that Bengtsson et al. (1980) reported to be significantly different in PCB-exposed fish as compared to control fish is the hatching time. Fish in the medium and high exposure groups has significantly reduced hatching times compared with the control group. Although the premature hatching of the exposed fish was thought to result in the premature death of the fry, this result was not tested statistically. Also, differences in the time to spawning and the temperature at which spawning occurred between exposed fish and control fish were not tested statistically. Therefore, hatching time, spawning time, and spawning temperature were not considered as toxicity endpoints in the development of TRVs from this study.

The study by Hansen et al. (1974) was not identified in the literature search that was conducted for the ERA or by the literature search that was conducted by NOAA (1999b). The study did establish a NOAEL of 0.88 mg PCBs/kg egg and a LOAEL of 5.1 mg PCBs/kg egg. The study also established a NOAEL of 1.9 mg PCBs/kg and a LOAEL of 9.3 mg PCBs/kg for adult female fish. The values for adult fish determined in this study are more appropriate for comparison to measured and modeled concentrations in adult Hudson River fish. If this study were used to develop TRVs, uncertainty factors would be applied as described in the methodology. For example, because sheepshead minnow have been shown to be of intermediate sensitivity in a comprehensive, multi-species study on effects of dioxin-like compounds (Elonen et al., 1998), an interspecies uncertainty factor would be applied to develop TRVs for species that are not in the same family as the sheepshead minnow and for which no other toxicity information is available. An uncertainty factor would be applied for development of TRVs for all fish species since they are in different families.

Based on the study by Hansen et al. (1974):

The LOAEL TRV for all fish species is 0.93 mg PCBs/kg tissue
The NOAEL TRV for all fish species is 0.19 mg PCBs/kg tissue

The study by USACE (1988) was inadvertently excluded from Table 4-6 of the ERA. This study is considered to be a field-related study, rather than a laboratory study, because the sediments to which the fish were exposed were field-collected sediments, rather than sediments to which a known contaminant were spiked. Because these field-collected sediments contained contaminants in addition to PCBs, the observed effects could not be solely attributed to the effects of PCBs. As was the case for most field studies considered in the ERA, a NOAEL-based TRV could be developed from these data, because the concentration of PCBs in sediment at which no adverse effects were observed was identified. However, a LOAEL-based TRV could not be developed because of the potential contribution of other contaminants. This study should have was selected for development of a field-based TRV for the spottail shiner, a species in the same family as the fathead minnow that was used in the study, for the revised risk calculations presented in Section IV of this responsiveness summary.

The final NOAEL TRV for the spottail shiner is 5.25 mg PCBs/kg wet wt tissue.

Response to EF-1.40

The ERA (p. 82) is revised to read:

> On the basis of laboratory toxicity studies for non-salmonids:
>
> The LOAEL TRV for the pumpkinseed is 10.3 ug TEQs/kg lipid (Table 4-25).
> The NOAEL TRV for the pumpkinseed is 0.54 ug TEQs/kg lipid (Table 4-25).

Response to EG-1.19

The objective of the NOAA (1999b) report on the effects of PCBs on reproduction and development in fish was not to develop "threshold values," as suggested by the commentor, but rather to summarize the range of values that have been reported to have reproductive or developmental effects (Monosson, 1999). The NOAA (1999b) report demonstrated that a range of PCB concentrations from 5.0 ppm in the whole body of larvae reduces larval survival, and from 25 to 75 ppm in the livers (equivalent to 25 to 75 ppm in the whole body of adults, and half that in fillets) interferes with proper functioning of the reproductive system. The 25 ppm value is not a threshold value; it is the low end of the range of Aroclor 1254 concentrations reported to have reproductive or developmental effects. It is not necessarily the lowest effective concentration, which is an important distinction.

In addition, when comparing the NOAA range values to the TRVs selected in the ERA, it is important to recall that a ten-fold uncertainty factor was used to develop the TRVs in many cases because of interspecific differences. The goal of the NOAA (1999b) report was to summarize the available data, not to derive with threshold numbers or TRVs. Had that been the case, then it is possible that some sort of uncertainty factor may have been applied resulting in lower effective concentrations (Monosson, 1999).

The NOAA (1999b) report considered reproductive, developmental, and immunotoxic effects on fish. These effects were selected as biological endpoints that are both sensitive to anthropogenic contaminants and ecologically relevant. The ERA developed a more narrow definition of ecologically relevant endpoints that included reproductive and developmental effects, but not immunotoxic, sublethal, or biochemical effects. These other types of effects were not included because it was judged that these effects are often less clearly related to ecologically relevant endpoints, than are developmental and reproductive effects.

In addition, NOAA (1999b) reported data that was measured or converted into concentrations in adult liver tissue. The present Hudson River assessment felt that the relationship between the concentration in liver tissue and the concentration in whole fish has not been well studied for most species. Therefore, the Hudson River risk assessment gave preference to studies that measured concentrations in whole fish. For dioxin-like compounds, most studies examined effects on the basis of concentrations in eggs. However, the relationship between concentrations in eggs and whole fish is felt to be better characterized than the relationship between liver concentration and adult tissue concentration. Also, more data is generally available for effects associated with concentrations of PCBs in whole tissue.

Response to EG-1.25

While PCB concentrations above the water quality criterion do not necessarily imply that exposed populations are harmed, they do indicate the potential for some individuals to be affected. The use of the criterion as one line of evidence is consistent with the weight-of-evidence approach used in the ERA.

4.2.3.1 Pumpkinseed *(Lepomis gibbosus)*

Response to EL-1.32

The Nebeker et al. (1974) study states, "Spawning occurred at 1.8 ug/L, but was significantly less than that at lower concentrations." This is the only result for Aroclor 1254 that is reported as being a significant difference. The average tissue concentrations for female fish exposed to 1.8 ug/L was 429 mg/kg. The average tissue concentrations for female fish exposed to the next lower concentration was 105 mg/kg. Therefore, as reported in the ERA, the LOAEL for this study is 429 mg/kg and the NOAEL is 105 mg/kg.

However, results reported by Nebeker et al. (1974) are highly variable and it appears that although spawning at 1.8 ug/L was significantly lower than at lower concentrations, it does not appear to be significantly different from spawning in the control fish. For this reason, this study should not be selected for development of TRVs.

Due to the potential impact of co-occurring contaminants, the ERA uses field studies to develop NOAEL-based TRVs, but not LOAEL-based TRVs. As shown in Table 4-25, the Adams et al. (1989, 1990, 1992) studies are only used to develop NOAEL-based TRVs. The NOAEL in Table 4-6 is listed as 0.5 mg/kg wet wt. However, in the revised risk calculations contained in Section IV of this responsiveness summary a NOAEL of 0.3 is used, based on a different endpoint in the same study.

4.2.3.2 Spottail Shiner *(Notropis hudsonius)*

No significant comments were received on Section 4.2.3.2.

4.2.3.3 Brown Bullhead *(Ictalurs nebulosus)*

No significant comments were received on Section 4.2.3.3.

4.2.3.4 Yellow Perch *(Perca flavescens)*

Response to EL-1.35

The lowest appropriate NOAEL and LOAEL TRVs were selected using the methodology described in the ERA (pp. 79-81). Consistent with this methodology, the striped bass studies (Westin et al., 1985, see, ERA, p. 88) were not used because they are field studies for a species that was not closely related (i.e., in a different taxonomic family). Instead, laboratory studies were used to develop the TRVs for the yellow perch (see, ERA, p. 86). Laboratory studies are controlled, and thus are designed to eliminate the possibility that contaminants or conditions present in the field other than those being studied may cause observed effects.

4.2.3.5 White Perch *(Morone americana)*

Response to EF-1.41

The lowest appropriate NOAEL and LOAEL TRVs were selected using the methodology described in the ERA (pp. 79-81). Consistent with this methodology, the Westin et al. (1985) study of striped bass was used to develop a field-based NOAEL TRV, but not a LOAEL TRV for the white perch, since other contaminants may have been contributing to observed effects. Field studies are not generally extrapolated between families (e.g, Centrarchidae and Moronidae) with uncertainty factors because of the possible effects of other contaminants and stressors.

Response to EF-1.42

USEPA acknowledges the comment on this typographical error. The bottom of p. 88 is revised to read:

The LOAEL TRV for the white perch is 0.6 ug TEQs/kg lipid (Table 4-25)
The NOAEL TRV for the white perch is 0.29 ug TEQs/kg lipid (Table 4-25).

4.2.3.6 Largemouth Bass *(Micropterus salmoides)*

No significant comments were received on Section 4.2.3.6.

4.2.3.7 Striped Bass *(Morone saxatilis)*

No significant comments were received on Section 4.2.3.7.

4.2.3.8 Shortnose Sturgeon *(Acipenser brevirostrum)*

No significant comments were received on Section 4.2.3.8.

4.2.4 Selection of TRVs for Avian Receptors

Response to EG-1.22

Studies on gallinaceous birds were used to develop TRVs only in cases for which no appropriate data are available for a species that is closely related to the receptor of concern. In such cases, it is conservative, but reasonable, to assume that the receptor could be as sensitive as the most sensitive species tested. Because gallinaceous birds are among the most sensitive of birds to the effects of PCBs and dioxin-like compounds, an interspecies uncertainty factor was not applied to account for potential differences in PCB-sensitivity between species.

For the tree swallow, field data were used to derive a field-based TRV, but this does not eliminate the need to examine laboratory-based data. No laboratory-based information is available for the tree swallow and it is assumed that the tree swallow is as sensitive as the most sensitive species tested (see, ERA, p. 94). However, although both laboratory- and field-derived TRVs are listed in Table 4-26, the field-based TRV was selected as the final TRV for the tree swallow.

For the mallard, several studies were identified that examined effects of PCBs on the mallard. The study by Hill et al. (1975) is not selected for development of TRVs because it examined mortality as an endpoint, which is not expected to be as sensitive an endpoint as growth and reproduction. The studies by Riseborough and Anderson (1975), Custer and Heinz (1980), and Heath et al. (1972) found no effects on various reproductive endpoints based on exposure to a single dose (40 ppm, 25 ppm, and 25 ppm in diet, respectively). Haseltine and Prouty (1980) observed no adverse effects on reproductive endpoints after a 12-week exposure to 150 ppm Aroclor 1242 in food, but did observe significantly reduced weight gain in adults. Therefore, the study by Haseltine and Prouty (1980) is selected as the most appropriate study, because it reports a NOAEL value that is bounded by a LOAEL value. Because only a single dose was tested, a LOAEL-to-NOAEL uncertainty factor of ten is applied to estimate a NOAEL from this study. Because the study was conducted over a 12 week period, a sub-chronic to chronic uncertainty factor is not applied.

Based on the results of Haseltine and Prouty (1980) on growth:

The LOAEL TRV for growth effects is: 16 mg/kg/day.
The NOAEL TRV for growth effects is: 1.6 mg/kg/day.

No laboratory data are available on the great blue heron, so the data on domestic chicken (Scott, 1977) were used to develop the TRVs. It is conservatively protective, and therefore reasonable, to assume that the great blue heron could be as sensitive as the most sensitive species tested. A subchronic to chronic uncertainty factor was applied for studies of less than 10 weeks duration (see, ERA, pp. 80-81). The Scott (1977) and other studies on the domestic chicken are all subchronic studies, and so a subchronic-to-chronic uncertainty factor was applied.

For the bald eagle, field studies were selected to develop the final TRVs for the effects of total PCBs PCBs in eggs (see, ERA, pp. 105-106, Table 4-26). The studies by Wiemeyer et al. (1984, 1993) were selected over the study by Eliot et al. (1996) for several reasons. First, Wiemeyer et al. (1993) studied bald eagle production over a much longer time period (i.e., 5- year intervals from 1969 through 1984) than the study by Elliott et al. (1996) (1 year). Second, the studies by Wiemeyer et al. (1993) examined rates of production in the field, while the study by Elliott et al. (1996) examined hatching rates of eggs that were artificially incubated in the laboratory. Third, the studies by Wiemeyer et al. (1993) examined a greater number of eggs than did the study by Elliott et al. (1996). Elliott et al. (1996) found that average percent hatching success was lower (69%, n=16 eggs) for eggs from the contaminated site, but not significantly different from the average percent hatching success for eggs from the reference site (88%, n = 8 eggs).

4.2.4.1 Tree Swallow *(Tachycineta bicolor)*

No significant comments were received on Section 4.2.4.1.

4.2.4.2 Mallard *(Anas platyrhychos)*

No significant comments were received on Section 4.2.4.2.

4.2.4.3 Belted Kingfisher *(Ceryle alcyon)*

No significant comments were received on Section 4.2.4.3.

4.2.4.4 Great Blue Heron *(Ardea herodias)*

No significant comments were received on Section 4.2.4.4.

4.2.4.5 Bald Eagle (*Haliaeetus leucocephalus*)

No significant comments were received on Section 4.2.4.5.

4.2.5 Selection of TRVs for Mammalian Receptors

Response to EG-1.23

The study by Linder et al. (1974) was selected as the most appropriate study for development of TRVs for the bat and the raccoon because it is a multigenerational study. For this reason, it was selected over the shorter-term studies of McCoy et al. (1995) and Linzey (1988). The TRV developed by Sample et al. (1996) from the Linder et al. (1974) study is different than the TRV developed in the ERA for two reasons. First, different food consumption rates were used for the rat to convert the concentration of PCBs in food (5 ppm), which results in different dietary doses (0.4 mg PCBs/kg/day for Sample et al. (1996) and 0.32 mg PCBs/kg/day for the ERA). Second, Sample et al. (1996) used an allometric scaling approach to convert the toxicity results determined for a test species to a TRV for a receptor of concern. The approach in the ERA uses uncertainty factors, rather than allometric scaling, to develop TRVs for the receptors of concern. The allometric scaling method and uncertainty factor method are equally acceptable approaches for the development of TRVs.

Response to EG-1.24

Appropriate field-based and laboratory studies were identified for development of TRVs. The Aulerich and Ringer (1977) laboratory study of mink was used to develop a laboratory-based TRV. The Heaton et al. (1995) field study of mink represents an acceptable, appropriate field-based study and was selected for development of the final TRV for the mink. Both LOAEL- and NOAEL-based TRVs are derived from the Heaton et al. (1995) field study because that study documents that exposure to other co-occurring chlorinated hydrocarbons was minimal.

4.2.5.1 Little Brown Bat (*Myotis lucifugus*)

No significant comments were received on Section 4.2.5.1.

4.2.5.3 Mink (*Mustela vison*)

No significant comments were received on Section 4.2.5.2.

4.2.5.4 River Otter (*Lutra canadensis*)

No significant comments were received on Section 4.2.5.4.

4.3 Summary of Available Literature on Herpetofauna

No significant comments were received on Section 4.3.

4.3.1 Amphibians

No significant comments were received on Section 4.3.1.

4.3.2 Reptiles

No significant comments were received on Section 4.3.2.

4.3.2.1 Snakes

No significant comments were received on Section 4.3.2.1.

4.3.2.2 Turtles

No significant comments were received on Section 4.3.2.2.

5.0 RISK CHARACTERIZATION

Response to EP-2.5

The results of site specific population-level data are considered in the ERA. The tree swallow studies found indications of disturbed reproductive biology, nest abandonment, and low reproductive success (McCarty and Secord, 1999a and 1999b). Population studies of most other receptors have not been performed, although for piscivorous species it is difficult to determine population-level effects due to PCBs because of reduced (human) predation pressures due to the fishing ban and other anthropogenic and natural influences (e.g., point and nonpoint discharges, nutrient enrichment).

Response to EP-2.6

The use of water quality criteria and sediment guidelines as measurement endpoints for the assessment endpoints is an acceptable part of the weight-of-evidence approach.

Response to EP-2.7

All things being equal, it would be appropriate to select the highest NOAEL. However, the studies that were evaluated varied in many aspects, including duration, route of exposure, and type of endpoints examined. Therefore, studies of the longest duration, that examined the appropriate route of exposure and the most sensitive endpoints, were the most appropriate studies. These studies resulted in the most credible and defensible LOAELs and NOAELs.

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

Response to EG-1.7

NYSDEC (1993) and Exponent (1998) benthic invertebrate data were examined in conducting the ERA. The number of taxa/groups collected by Exponent (1998) in the Thompson Island Pool and at Stillwater is similar to the USEPA results (see, ERA, p. 119). NYSDEC (1993) examined 20-year trends in water quality based on macroinvertebrate data and concluded that mayfly/caddisfly species in the Upper Hudson River below Glens Falls increased from 1972 to 1986 following improvements in treatment of municipal and industrial wastes. The NYSDEC (1993) data were not included in the ERA because they were not considered directly relevant to assessing ecological risk from PCB-contaminated sediments in the Hudson River.

Response to EG-1.11 and EG-1.31

The benthic invertebrate study (see, ERA, p. 167) concludes that the reduced macroinvertebrate community indicates a potential for risk from PCBs. This statement is consistent with the lower diversity seen at the two Thompson Island Pool stations with higher PCB concentrations (see, ERA pp. 119-120), although the observed differences may also result from other environmental variables.

In contrast to GE's interpretation of the McCarty and Secord data, the investigators found indications of disturbed reproductive biology, nest abandonment, and low reproductive success (McCarty and Secord, 1999a). The commentor again misinterprets the Secord and McCarty studies (see, Responsiveness Summary for ERASOW, p. 24).

Response to EG-1.33

Although the benthic macroinvertebrate community study (see, ERA, Appendix H) could not separate out the influence of PCBs from other site-related variables, it was one of three lines of evidence used to evaluate benthic community structure as a food source for local fish populations (see, ERA, 118-121). The macroinvertebrate community study, in conjunction with sediment and water PCB concentrations, suggest an adverse effect of PCBs on benthic macroinvertebrates serving as a food source to local fish, which is consistent with the conclusion reached in the ERA for this assessment endpoint (see, ERA, pp. 167-168). Adverse effects are more likely to be seen in the Upper Hudson River than the Lower Hudson River, based on the results of the three measurement endpoints for each section of the river.

5.1.1 Does the Benthic Community Structure Reflect the Influence of PCBs

No significant comments were received on Section 5.1.1.

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

No significant comments were received on Section 5.1.1.1.

5.1.1.2 Measurement Endpoint: Lower Hudson Benthic Invertebrate Community Analysis

No significant comments were received on Section 5.1.1.2.

5.1.2 Do Measured and Modeled Sediment Concentrations Exceed Guidelines?

No significant comments were received on Section 5.1.2.

5.1.2.1 Measurement Endpoint: Comparison of Sediment PCB Concentrations to Guidelines

Response to EL-1.41

The range of mean PCB concentrations in the Upper and Lower Hudson benthic invertebrate sampling stations is provided on the bottom of Table 5-6 (under notes) for comparison to the sediment guidelines.

Modeled sediment concentrations may be higher than observed because the ecological sampling program was biased toward samples containing invertebrates, whereas the HUDTOX model predicts sediment concentrations generally, without particular regard or focus on ecological sediments as opposed to other sediments.

The HUDTOX model was calibrated using available data, and also included a validation component. Agreement with observed sediment concentrations was very good (See Revised BMR, Book 1). Nonetheless, modeled sediment concentrations may be higher than observed because the ecological sampling program was biased toward samples containing invertebrates, whereas the HUDTOX model predicts sediment concentrations generally, without particular regard or focus on ecological sediments as opposed to other sediments. Sediment concentrations are highly variable, and the modeled sediment concentrations are within the uncertainty bounds of the observed ecological program sediment concentrations.

Response to EG-1.40

This comment is answered in the letter transcribed below (MacDonald, 2000). NOAA and EPA concur with this letter:

December 1, 1999

Michael T. Huguenin
President
Industrial Economics, Inc.
2067 Massachusetts Avenue
Cambridge, MA 02140

Dear Michael:

Thank you for the opportunity to review the document entitled, Comments of General Electric Company on Volume 2E-Baseline Ecological Risk Assessment: Hudson River PCBs Reassessment RI/FS (General Electric Company, LWB Environmental Services, Ltd., and Quantitative Environmental Analysis, Inc. 1999; hereafter refered to as GEC et al. 1999). In conducting this review, I have explicitly focused on the information presented in the comment, The Sediment Effects Concentrations (SECs) are not Reasonable Estimates of PCB Toxicity to Benthic Invertebrates Either Individually or as a Population (Section 4.0).

In the subject document, GEC et al. (1999) indicates that the SECs that were developed by NOAA (1999) should not be used as toxicity reference values (TRVs) in the baseline ecological risk assessment because:

- the SEC values have no causal basis; and
- direct relationships between benthic community productivity and the productivity of higher trophic levels cannot be demonstrated.

In addition, GEC et al. (1999) question the applicability of the SECs because "the meaning and utility of the pre-existing SECs is the subject of considerable scientific debate, the authors of several of the methods have warned against their use as risk assessment tools, the no-effects data are not properly considered, the pre-existing SEC values are mostly based on data from sediments for which PCBs have not been shown to be the dominant or only contaminant of concern, and the spiked sediment toxicity study of Swartz et al. (1988) [is improperly used] as a validation of the SEC values." Each of these specific comments are addressed in the following sections.

First GEC et al. (1999) indicate that direct relationships between benthic community productivity and the productivity of higher trophic levels cannot be demonstrated. This statement is counter-intuitive. Countless studies have been published in the scientific literature which indicate that higher trophic levels in the food web are directly dependent on benthic community productivity. While there are a number of other factors that can also influence the productivity of higher trophic levels (e.g., predation, water-borne contaminants, etc.), those species that rely on benthic production for most or all of their energy requirements will necessarily be adversely affected if the food source is removed or reduced in abundance.

Some of the comments included in the GEC et al. (1999) document also indicate that the authors do not have a complete understanding of the sediment quality guidelines (SQGs) that were used to derive the SEC values. For example, GEC et al. (1999) indicate that the TEL/PEL values were promulgated by the Ontario Ministry of the Environment. This statement is incorrect. The TEL/PEL values were promulgated by Enviroment Canada. In addition, GEC et al. (1999) indicate that the TEL and PEL values were not used in an appropriate manner for deriving the consensus-based SECs (i.e., they were used in a manner that is contrary to the guidance provided by the authors). In this respect, GEC et al. (1999) correctly indicated that the Canadian SQGs (i.e., TELs) are intended to define contaminant concentrations below which adverse effects are unlikely to occur. For this reason, the TELs were used to calculate the threshold effect concentrations (TECs). However, GEC et al. (1999) failed to mention that the PELs are intended to identify the concentrations of contaminants above which adverse effects are likely to occur (CCME 1999). Therefore, in contrast to the statements made by GEC et al. (1999) it is appropriate to use the PELs to calculate the mid-range effect concentrations (MECs). Additionally, GEC et al. (1999) indicated that the SEC approach does not properly consider the no-effects data. This statement is also incorrect for several reasons. First, many of the underlying SQGs explicitly consider the distribution of the no-effects data in the derivation of the guideline values (e.g., TEL/PEL values, AETs, NECs, etc.). In addition, both the effects and no-effects data were used to evaluate the predictive ability of the SECs.

GEC et al. (1999) indicate that the authors of several of these methods have warned against the use of SQGs as risk assessment tools. However, none of the reports cited by GEC et al. (1999) provide any such warning regarding the use of SQGs in ecological risk assessments. In contrast, several of these authors have evaluated the SQGs and determined that they provide an accurate basis for predicting the effects of sediment-associated contaminants on sediment-dwelling organisms (Long et al. 1995; Long et al. 1998; Ingersoll et al. 1996; MacDonald et al. 1996; Long and MacDonald 1998; MacDonald and Ingersoll In review). In fact, several of the recently published papers by these authors provide a basis for identifying the probability of observing sediment toxicity using the SQGs (Long and MacDonald 1998; Field et al. 1999). As such, these guidelines are diretly applicable to the ecological risk assessment process. Moreover, these guidelines have been recommended for use in ecological risk assessments by a panel of experts that was assembled by the Society of Environmental Toxicology and Chemistry (Ingersoll et al. 1997) and by several of the authors of these guidelins(Long and MacDonald 1998; Field et al. 1999). Therefore, GEC et al. (1999) seems to be out of step with the most recent guidance on the application of SQGs in ecological risk assessments.

GEC et al. (1999) are correct in their observation that the pre-existing SEC values (they are actually referring to SQGs) are mostly based on data from sediments for which PCBs have not been shown to be the dominant or only contaminant of concern. As a result, it is possible to develop correlations between PCB concentrations and adverse biological effects using the data that have been colloected at most of these sites (i.e., the resultant SQGs are considered to be correlative rather than causually-based). By assembling SQGs that were developed using multiple approaches and unique underlying data sets, it is possible to develop consensus-based SECs that reflect the agreement among the existing SQGs. The fact that the existing SQGs are comparable, in spite of the differences in calculation methods and underlying data sets, increases the level of confidence that they are correctly identifying the concentrations of PCBs below which adverse effects are unlikely to be observed and above which adverse effects are likely to be observed. However, these characteristics, by themselves, are not sufficient to demonstrate that PCBs are causing or substantially contributing to sediment toxicity at concentrations above the two upper SECs (i.e., the MEC and EEC). For this reason, three other evaluaitons of the SECs were conducted, including assessing their predictive ability, assessing their comparability with equilibrium partitioning-based SQGs, and assessing their comparability to the chronic toxicity thresholds that have been estimated from the results of dose-response studies.

The results of theses three additional evaluations indicate that the SECs for PCBs that were developed by NOAA (1999) reflect causal rather than correlative effects. More specifically, the results of the predictive ability evaluation demonstrate that the SECs can be used to accurately classify freshwate, estuarine, and marine sediments as toxic and not toxic. These results can also be used to determine the likelihood that a particular sediment sample will be toxic (i.e., based on PCB concentration alone). This feature is important for conducting ecological risk assessments. The consensus-based SECs were also evaluated to determine if they were comparable to equilibrium partitioning-based SQGs and the results of spiked sediment toxicity test (i.e., dose-response studies); both of these latter assessment tools provide a means of identifying the concentrations of sediment-associated contaminants that are likely to cause sediment toxicity. The results of that analysis indicated that the consensus-based SECs are comparable to the equilibrium partitioning-based SQGs that have been published in the scientific literature and to the chronic toxicity thresholds that have been estimated from the results of spiked sediment toxicity tests. This agreement between the consensus-based SECs, the equilibrium partitioning-based SQGs, and the results of spiked sediment toxicity tests indicates that the SECs are causally-based.

GEC et al. (1999) argued that the results of the Swartz et al. (1988) study were used improperly in the evaluation of the SEC values. This is an interesting argument because Dr. R. Swartz was involved in the development and evaluation of the SEC values and has co-authored a paper on this topic (MacDonald et al. In press). Therefore, it is unlikely that Dr. Swartz would concur with GEC et al. (1999) regarding the use of his spiked sediment toxicity test results in the evaluation of the SECs.

GEC et al. (1999) also used the Swartz et al. (1988) study results to estimate a chronic toxicity threshold of 8 mg/kg DW for PCBs in sediments from the Thompson Island Pool [i.e., which has an average total organic carbon (TOC) of 2%]. This estimated chronic toxicity threshold for this location was then compared to the TEC and the EEC from NOAA (1999). The results of this comparison were then used to suggest that the SECs significantly overstate the toxicity of PCBs. However, this logic is flawed for several reasons. First, GEC et al. (1999) used the nominal concentrations of PCBs to estimate their chronic toxicity thresholds for PCBs; measured concentrations were substantially lower than the nominal concentrations in this study. Second, Swartz et al. (1988) did not report the concentrations of TOC in the test sediment; therefore, the level of TOC used in the GEC et al. (1988) calculations were estimated only, based on other information that was reported in the paper. Third, there is some uncertainty about the application of partitioning model at low levels of TOC. Additionally, the chronic toxicity thresholds that were estimated by GEC et al. (1999), if correct, would only apply to one location on the Hudson River. Such thresholds would not support the type of ecological risk assessment that needed to be conducted on the river. Finally, Swartz et al. (1988) demonstrated that PCB-contaminated sediments tended to be more toxic when they also contain other chemical substances. This fact was not considered by GEC et al. (1999) in the estimation of chronic toxicity thresholds for the Thompson Island Pool. Therefore. The resultant thresholds are unlikely to be relevant for identifying the concentrations of PCBs that are likely to cause or substantially contribute to sediment toxicity in the Hudson River.

Thank you again for the opportunity to review the subject draft. I hope that these review comments provide a helpful perspective on the GEC et al. (1999) document. Cheers and best wishes.

Don MacDonald
President

References

CCME (Canadian Council of Ministers of the Environment), 1999. Canadian Environmental Quality Guidelines. Guidelines and Standards Division. Environment Canada. Ottawa, Canada.

Field L.J., D.D. MacDonald, S.B. Norton, C.G, Severn, and C.G. Ingersoll. 1999. Evaluating sediment chemistry and toxicity data using logistic regression modelling. Environmental Toxicology and Chemistry 18:1311-1322.

General Electric Company, LWB Environmental Services Ltd., and Quantitative Environmental Analysis Inc. 1999. Comments of General Electric Company on Volume 2E - Baseline Ecological Risk Assessment: Hudson River PCBs Reassessment RI/FS. Corporate Environmental Programs. Albany, New York.

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 *Chironomus riparius*. Journal of Great Lakes Research 22:602-623.

Ingersoll, C.G., T. Dllon, and R.G. Biddinger (Eds.). 1997. Methodological uncertainty in sediment ecological risk assessment. *In*: Ecological Risk Assessments of Contaminated Sediment. SETAC Press. Pensacola Florida. 389 pp.

Long, E.R. and D.D. MacDonald. 1998. Recommended uses of the empirically-derived sediment quality guidelines for marine and estuarine ecosystems. Human and Ecological Risk Assessment 47:1019-1039.

Long, E.R., D.D. MacDonald, S.L. Smith and F.D. Calder. 1995. Incidence of adverse biological effects within ranges of chemical concentrations in marine and estuarine sediments. Environmental Management 19:81-97.

MacDonald, D.D., R.S. Carr, F.D. Calder, E.R. Long and C.G. Ingersoll, 1996. Development and evaluation of sediment quality guidelines for Florida coastal waters. Ecotoxicology 5:253-278.

MacDonald, D.D., L.M. DiPinto, J. Field, C.G. Ingersoll, E.R. Long, and R.C. Swartz. In press. Development and evaluation of consensus-based sediment effect concentrations for polychlorinated biphenyls (PCBs). Environmental Toxicology and Chemistry (In press).

MacDonald, D.D. and C.G. Ingersoll. In review. Development and evaluation of consensus-based sediment effect concentrations for freshwater ecosystems. Archives of Environmental Contamination and Toxicology (In review).

NOAA (National Oceanic and Atmospheric Administration). 1999. Development and evaluation of consensus-based sediment effect concentrations for PCBs in the Hudson River. Prepared by MacDonald Environmental Sciences Ltd. Ladysmith, British Columbia..

Swartz, R.C., P.F. Kemp, D.W. Schults and J.O. Lamberson. 1988. Effects of mixtures of sediment contaminants on the marine infaunal amphipod *Rhepoxynius abronius*. Environmental Toxicology and Chemistry 7:1013-1020.

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

No significant comments were received on Section 5.1.3.

5.1.3.1 Measurement Endpoint: Comparison of Water PCB Concentrations to Benchmarks

Response to EF-1.43

As noted by the commentor, the text on page 123 of the ERA contains a typographical error. The NYSDEC surface water standard for protection of wildlife (including bioaccumulation) is 1.2 x 10^{-4} ug/L. This is the value used in all comparisons to data; but the units were stated incorrectly in the table.

Response to EL-1.42

Direct uptake of PCBs from water is a significant pathway of concern for aquatic animals such as invertebrates and fish (see, ERA, p. 16). Although birds and mammals have lower exposure to surface water than aquatic animals, the NYSDEC surface water standard was developed for the protection of wildlife and therefore its use is appropriate in the ERA.

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

No significant comments were received on Section 5.2.

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

Response to EF-1.45

To clarify, the values shown in Table 5-12 for Lower Hudson River spottail shiners are all from spottail shiner samples, and do not include other species of fish (see, ERA, p. 123).

Response to EF-1.46

A comparison of modeled and measured PCB concentrations in fish is provided in the Revised Baseline Modeling Report. Agreement between measured and modeled results is typically within 10 to 50%, and averages around 25%. On an absolute basis, modeled wet weight concentrations are typically within 1 - 2 ppm of measured concentrations.

Response to EG-1.8 and EL-1.1

The available fish population data were evaluated. Population level data are only available for the Lower Hudson River, not the Upper Hudson river, and these were collected specifically to evaluate the impact of power plant discharges on fish population parameters. These data are not directly relevant to the ERA, which assesses ecological risks posed by PCBs in the river, because they do not establish a link between PCB exposure and population abundance. Both the shortnose sturgeon and striped bass data, specifically, were also considered, but again, these data are not helpful for evaluating the potential impact of PCB exposure on changes in fish populations. These data were considered qualitatively, rather than quantitatively, in the ERA.

For obvious reasons it is not possible to retrospectively evaluate population abundance prior to the occurrence of PCBs for each of the receptors of concern. As described in Chapter 5 of the ERA, all available field surveys were included, but these data provide only a qualitative indication of receptor populations.

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

Response to EF-1.44

The comment regarding the typographical error is acknowledged. The NYSDEC surface water standard for protection of wildlife is 1.2×10^{-4} ug/l.

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

No significant comments were received on Section 5.2.1.2.

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

No significant comments were received on Section 5.2.1.3.

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

Response to EF-1.47

The comment regarding the typographical error is acknowledged. On p. 124 of the ERA, the reference to Table 4-16 is revised to read Table 4-25.

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

Response to EF-1.48

The comment regarding the typographical error is acknowledged. The last sentence in Section 5.2.1.5 (p. 125) is revised to read, "On a TEQ basis, the toxicity quotients exceed one for both species at all locations, except brown bullhead at RMs 168 and 189, as shown in Table 5-24."

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

Response to EF-1.49

The final TRVs are shown in Table 4-25 and identified as field-based or laboratory-based. As shown in Table 4-25, a final field-based TRV was selected for the white perch, not the yellow perch, so the text is correct as written (see, ERA, p. 125).

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

Response to EF-1.50

The comment regarding the typographical error is acknowledged. The last sentence in the first paragraph of Section 5.2.1.7 is revised to read, "At river mile 154, just above the Federal Dam, the toxicity quotients begin to fall below one in 1997 on a median basis."

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

No significant comments were received on Section 5.2.1.8.

5.2.1.9 Measurement Endpoint: Comparison of Modeled Tri+ PCB Fish Body Burdens to Toxicity Reference Values for 6 Largemouth Bass for the Period 1993 - 2018

No significant comments were received on Section 5.2.1.9.

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

No significant comments were received on Section 5.2.1.10.

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

No significant comments were received on Section 5.2.1.11.

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

No significant comments were received on Section 5.2.2.

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

No significant comments were received on Section 5.2.2.1.

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

Response to EG-1.9

The Hudson River is a large and complex ecosystem influenced by a variety of factors. Some clear correlations can be seen in the Hudson River ecosystem, such an increase in some fish populations due to the fishing ban or an increase in pollution-intolerant filter feeding macroinvertebrates resulting from improved water quality. The effects expected due to the PCBs include reduced fecundity, decreased hatching success, and similar kinds of reproductive impairment indicators. These effects can be difficult to observe in the field. The gradient of PCB concentrations along the approximately 200 river mile-long study area also increases the difficulty of ascribing particular effects to PCBs. Therefore, the ERA discusses the potential for adverse effects even in apparently healthy receptor populations.

It is difficult to compare data from field observations to assess populations if there are no appropriate reference populations. For example, there are currently at least 4 occupied bald eagle territories along the Hudson River (all below Albany). There has been mixed success in breeding, but because these eagles are the first to breed in approximately one hundred years, there is no appropriate reference population against which to compare these observations.

According to the USFWS, PCB-sensitive piscivorous species may be precluded from nesting in the more heavily contaminated portions of the river (USFWS letter to USEPA, October 4, 1999). Secord and McCarty (1997) concluded, that if a species as PCB-sensitive as the Caspian tern were to nest within portions of the Thompson Island Pool, it would likely experience total reproductive failure.

5.2.3.1 Measurement Endpoint: Evidence from Field Studies

Response to EF-1.51

USEPA agrees with the comment. The effect of restrictions on commercial and recreational fishing is to increase the populations of those species, which would mask adverse impacts to fish populations from PCBs and other anthropogenic sources.

5.3 Evaluation of Assessment Endpoint: Protection and Maintenance (i.e., Survival, Growth, and Reproduction) of Hudson River Insectivorous Bird Species (Tree Swallow)

Response to EG-1.32

USFWS agrees that their study did not demonstrate a dose-response relationship between tree swallow reproduction and PCB concentrations. However, they did observe statistically significant abnormal breeding behavior in 1994 relative to the Ithaca, NY reference colony and relative to data from unimpaired populations documented in the literature. Although USFWS cannot conclude that PCBs impaired reproduction, their 1994 data, in conjunction with the observations of similar abnormal reproductive behavior in other birds exposed to planar halogenated hydrocarbons, suggest that PCBs may have contributed to the observed nest abandonment.

USFWS data more conclusively demonstrated that PCBs likely contributed to or caused abnormal nest construction in tree swallows. Furthermore, it is highly likely that impaired nest quality could have a measurable impact on reproductive success in years of adverse weather conditions or other adverse environmental conditions.

With respect to their reference sites, USFWS did not use any reproductive data from the 1994 Hudson River reference site (the site at which boxes were more closely spaced) because the sample size was too small. Box placement in 1995 was the same at the Hudson River reference site as the Hudson River PCB sites. They determined after the 1995 samples were analyzed that eggs from both Hudson River reference sites and at least adults from the 1995 site contained high concentrations of PCBs, which are believed to have originated from the Hudson River.

USFWS has determined that it is infeasible to find an uncontaminated reference site within the Hudson River watershed due to the widespread PCB contamination in the river and the probable migration habits of the tree swallow along the Hudson River. Therefore, USFWS used reference data from the literature and from another New York state site studied by Dr. John McCarty, formerly with the U.S. Fish and Wildlife Service and currently with the USEPA. Secord and McCarty have published two papers in peer reviewed journals (McCarty and Secord, 1999a and 1999b) in which the Ithaca data were accepted as appropriate reference data. The PCB concentration in a tree swallow egg composite from Ithaca was 103 ng/g.

For the ERA, the most important conclusion of the USFWS tree swallow work may be that the PCB concentrations and dioxin equivalents detected in samples, particularly from the Remnant 4 and SA13 sites, were significantly higher than concentrations known to cause reproductive and developmental impairment in other birds.

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

No significant comments were received on Section 5.3.1.

5.3.1.1 Measurement Endpoint: Modeled Dietary Doses of Total PCBs (i.e., Tri+) to Insectivorous Birds (Tree Swallow) and Predicted Egg Concentrations Using 1993 Data

No significant comments were received on Section 5.3.1.1.

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

No significant comments were received on Section 5.3.1.2.

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

No significant comments were received on Section 5.3.1.3.

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

No significant comments were received on Section 5.3.1.4.

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

No significant comments were received on Section 5.3.1.5.

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

No significant comments were received on Section 5.3.1.6.

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

No significant comments were received on Section 5.4.

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

No significant comments were received on Section 5.4.1.

5.4.1.1 Measurement Endpoint: Modeled Dietary Doses of PCBs and Predicted Egg Concentrations as Total PCBs to Waterfowl (Mallard Ducks) Based on 1993 Data

No significant comments were received on Section 5.4.1.1.

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

No significant comments were received on Section 5.4.1.2.

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

No significant comments were received on Section 5.4.1.3.

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

No significant comments were received on Section 5.4.1.4.

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

No significant comments were received on Section 5.4.1.5.

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

No significant comments were received on Section 5.4.1.6.

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

Response to EG-1.10

NYSDEC wildlife scientists were contacted to obtain data on bird and mammalian receptors and threatened and endangered species (see, ERA, Appendices E, F, and G). NYSDEC is currently studying the bald eagle in the Hudson River area; however the statewide bald eagle data are not directly relevant to any study of the reproductive and developmental vigor of Hudson River bald eagles. Bald eagles have only successfully reproduced along the Hudson River since 1997, with a maximum of three nests in any one year. Very limited chemical data are available thus far. The data from prey taken from a nest (as well as one eagle plasma sample and one eagle fat sample) were analyzed by the USFWS and were provided to General Electric and USEPA in October, 1999. Therefore, these data were not available to be included in the ERA. However, PCB concentrations in these early results are high enough to be of concern (1,329 µg/kg in eagle plasma from a nesting eagle and 85,770 µg/kg in fat sample from an immature eagle found dead) (USFWS letter to USEPA, October 4, 1999). NYSDEC has been collecting eagle serum, prey and unhatched eggs for several years now for PCB analysis, and hopes to develop a solid picture of contaminant loads throughout the eagle's ecosystem over the next few years.

A 100% fish diet was used for bald eagle food chain modeling based on a discussion with Peter Nye of NYSDEC (see, ERA Appendix E p. E-14) and review of other relevant studies. Although bald eagles also may feed on non-fish prey, the inclusion of these prey (e.g., grebes) in the diet may actually increase the calculated PCB dose, because grebes feed on small fish, crustaceans, tadpoles, and insects and thus have higher bioaccumulation potential than some fish species. Fish concentrations used to estimate prey dietary concentrations were based on data collected during the 1993 USEPA Phase 2 ecological field sampling program and 1995 NOAA sampling to provide similar spatial and temporal data for modeling. The Secor (1997) eel data recommended for use by GE are from a different sampling time and location than the USEPA data and would increase the uncertainty of the food chain model.

Bald eagle data collected by NYSDEC indicate a stable population. However, the New State population (consisting of 200-250 individuals) is small enough to be affected by natural or anthropogenic disturbances. The conclusion that current and future concentrations of PCBs are not of a significant magnitude to prevent reproduction of waterfowl (see, ERA, p. ES-7) is in accordance with the secure mallard population found in the Hudson River Estuary. However, exposure to PCBs may reduce or impair the survival, growth, and reproductive capability of waterfowl in the Upper Hudson River. It should be noted that the mallard feeds on plants and aquatic invertebrates and therefore has a relatively low level of exposure to PCBs. The mallard has also been shown to have a relatively high tolerance to PCBs. The limited data that USFWS collected on mallards and wood ducks in the Upper Hudson River indicate that neither species is safe for human consumption, exceeding the USFDA guideline for poultry (3 ug/g lipid weight) by 3 to 31 times. TEFS calculated for the wood duck egg sample collected exceeded the toxicity range at which reduced productivity was observed in wood ducks (USFWS letter to USEPA, October 4, 1999).

Response to EG-1.36

NYSDEC wildlife scientists were contacted to obtain data on bald eagles (see, ERA Appendix E). However, the statewide bald eagle data are not directly relevant to an evaluation of the reproductive and developmental vigor of Hudson River bald eagles. Bald eagles have only successfully reproduced along the Hudson River since 1997, with a maximum of three nests in any one year.

The bald eagle breeding data for the Hudson River are as follows:

Year	#occupied territories	#active (egg laid)	#successful	#yg fledged
1992	1	0	0	0
1993	1	0	0	0
1994	1	1	0	0
1995	1	0	0	0
1996	2	1	0	0
1997	2	1	1	1
1998	3	3	2	4
1999	3	3	3	5

There are probably at least four occupied territories along the river (all below Albany) going into the 2000 breeding season (Nye, 2000). All eagles now breeding in NYS are the result of NYSDEC or other direct release/restoration programs (i.e., either the releases themselves or their progeny).

Very limited chemical data are available thus far. The data from prey taken from a nest (as well as one eagle plasma sample and one eagle fat sample) were analyzed by the USFWS and were provided to General Electric and USEPA in October, 1999. Therefore, these data were not available to be included in the ERA. However, PCB concentrations in these early results are high enough (PCBs) to be of concern (1,329 μg/kg in eagle plasma from a nesting eagle and 85,770 μg/kg in fat sample from an immature eagle found dead) (USFWS letter to USEPA, October 4, 1999). NYSDEC had been collecting eagle serum, prey and unhatched eggs for several years now for PCB analysis, and hopes to develop a solid picture of contaminant loads throughout the eagle's ecosystem over the next few years.

A 100% fish diet was used for bald eagle food chain modeling based on a discussion with Peter Nye of NYSDEC (see, ERA Appendix E, p. E-14) and other relevant studies. Although bald eagles also may feed on non-fish prey, the inclusion of these prey (e.g., grebes) in the diet may actually increase the calculated PCB dose, because grebes feed on small fish, crustaceans, tadpoles, and insect and thus have higher bioaccumulation potential than some fish species.

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

No significant comments were received on Section 5.5.1.

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

No significant comments were received on Section 5.5.1.1.

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

No significant comments were received on Section 5.5.1.2.

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

No significant comments were received on Section 5.5.1.3.

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

No significant comments were received on Section 5.5.1.4.

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

No significant comments were received on Section 5.5.1.5.

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

No significant comments were received on Section 5.5.1.6.

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

No significant comments were received on Section 5.5.2.

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

No significant comments were received on Section 5.5.2.1.

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

No significant comments were received on Section 5.5.3.

5.5.3.1 Measurement Endpoint: Observational Studies

<u>Response to EL-1.43</u>

Observations of the least bittern, upland sandpiper, and king rail are included in the discussion on observational studies of local bird populations because the Northern Hudson River Valley serves as habitat for these New York State species of special concern (<u>see</u>, ERA p. G-7).

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

No significant comments were received on Section 5.6.

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

No significant comments were received on Section 5.6.1.

5.6.1.1 Measurement Endpoint: Modeled Dietary Doses of Tri+ to Insectivorous Mammalian Receptors (Little Brown Bat) Using 1993 Data

No significant comments were received on Section 5.6.1.1.

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

No significant comments were received on Section 5.6.1.2.

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

No significant comments were received on Section 5.6.1.3.

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

No significant comments were received on Section 5.6.1.4.

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

No significant comments were received on Section 5.6.1.5.

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

No significant comments were received on Section 5.6.1.6.

5.6.1.7 Measurement Endpoint: Modeled Dietary Doses on a TEQ Basis to Omnivorous Mammalian Receptors (Raccoon) Using 1993 Data

No significant comments were received on Section 5.6.1.7.

5.6.1.8 Measurement Endpoint: Modeled Dietary Doses on a TEQ Basis to Omnivorous Mammalian Receptors (Raccoon) for the Period 1993 - 2018

No significant comments were received on Section 5.6.1.8.

5.6.1.9 Measurement Endpoint: Measured Total PCB Concentrations in the Liver of Piscivorous Mammalian Receptors (Mink, Otter)

No significant comments were received on Section 5.6.1.9.

5.6.1.10 Measurement Endpoint: Modeled Dietary Doses of Tri+ to Piscivorous Mammalian Receptors (Mink, Otter) Using 1993 Data

No significant comments were received on Section 5.6.1.10.

5.6.1.11 Measurement Endpoint: Modeled Dietary Doses of Tri+ to Piscivorous Mammalian Receptors (Mink, Otter) for the Period 1993 - 2018

No significant comments were received on Section 5.6.1.11.

5.6.1.12 Measurement Endpoint: Modeled Dietary Doses on a TEQ Basis to Piscivorous Mammalian Receptors (Mink, Otter) Using 1993 Data

No significant comments were received on Section 5.6.1.12.

5.6.1.13 Measurement Endpoint: Modeled Dietary Doses on a TEQ Basis to Piscivorous Mammalian Receptors (Mink, Otter) for the Period 1993 - 2018

No significant comments were received on Section 5.6.1.13.

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

No significant comments were received on Section 5.6.2.

5.6.2.1 Measurement Endpoint: Comparison of Measured and Modeled Water Concentrations to Criteria and/or Guidelines for the Protection of Wildlife

No significant comments were received on Section 5.6.2.1.

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

No significant comments were received on Section 5.6.3.

5.6.3.1 Measurement Endpoint: Observational Studies

No significant comments were received on Section 5.6.3.1.

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

No significant comments were received on Section 5.7.

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

No significant comments were received on Section 5.7.1.

5.7.1.1 Measurement Endpoint: Inferences Regarding Shortnose Sturgeon Population

No significant comments were received on Section 5.7.1.1.

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

No significant comments were received on Section 5.7.2.

5.7.2.1 Measurement Endpoint: Inferences Regarding Bald Eagle and Other Raptor Populations

No significant comments were received on Section 5.7.2.1.

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

No significant comments were received on Section 5.7.3.

5.7.3.1 Measurement Endpoint: Comparisons of Measured and Modeled Water Concentrations to Criteria and/or Guidelines for the Protection of Wildlife

No significant comments were received on Section 5.7.3.1.

5.7.4 Do Measured and Modeled Sediment Concentrations Exceed Guidelines for the Protection of Aquatic Health?

No significant comments were received on Section 5.7.4.

5.7.4.1 Measurement Endpoint: Comparisons of Measured and Modeled Sediment Concentrations to Guidelines

Response to EF-1.52

The comment regarding this typographical error is acknowledged. The second sentence of Section 5.7.4.1 is revised to read, "Measured concentrations in the Upper Hudson River exceed the EEC, and all but three locations in the Lower Hudson River exceed the EEC on a 95% UCL basis."

5.8 Evaluation of Assessment Endpoint: Protection of Significant Habitats

No significant comments were received on Section 5.8.

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

No significant comments were received on Section 5.8.1.

5.8.1.1 Measurement Endpoint: Comparison of Measured and Modeled Water Concentrations to Criteria and/or Guidelines for the Protection of Wildlife

No significant comments were received on Section 5.8.1.1.

5.8.1.2 Measurement Endpoint: Comparison of Measured and Modeled Sediment Concentrations to Guidelines for the Protection of Aquatic Health

No significant comments were received on Section 5.8.1.2.

6.0 UNCERTAINTY ANALYSIS

Response to EP-2.8

The rationale for the use of uncertainty factors is documented in the USEPA report, *Great Lakes Water Quality Initiative Technical Support Document for Wildlife Criteria* (USEPA, 1995) (see also, response to comment EG-1.21).

Response to EP-2.9

For estimating the effects to fish on a TEQ basis, the ERA used the study of Walker et al. (1994), but does not apply interspecies or subchronic-to-chronic uncertainty factors. Because salmonids are known to be highly sensitive to the effects of dioxin-like compounds, the ERA also developed TRVs from studies conducted on non-salmonid species.

6.1 Sampling Error and Representativeness

Response to EF-1.53

Individual diets were developed for each receptor species, as detailed in the ERA (see, Appendices D-F). For avian and mammalian receptors, diets were developed for breeding adult females (considered to be sensitive to the effects of PCBs). Prey composition was based on species-specific research conducted in similar habitats. Dietary preferences of fish are incorporated in the bioaccumulation model, and reflect extensive gut content analyses (by MCA, GE, and others) as well as information from the literature.

6.2 Analysis and Quantitation Uncertainties

Response to EF-1.54

Review of the USEPA-funded fish samples indicates that BZ #77 blank contamination was not a significant factor affecting data quality or reporting limits. Blank contamination associated with BZ #77 was detected in only 15% (18 of 120) samples (see details on Table EF-1.54). In only two samples was the blank contamination high enough to result in an increase in the reporting limit for two samples (both were Lower River Freshwater fish samples). BZ #77 was detected in over 85% of the 109 non-background samples (see Table EG-1.30B).

TABLE EF-1.54
BZ #77 and BZ #126 Frequency of Blank Contamination for USEPA-funded Fish Sample Analysis

	BZ #77				BZ#126			
Sample Group	Samples Associated with Blank Contamination	Contamination Greater than PQL	Valid Analyses	Contamination Detection Frequency	Samples Associated with Blank Contamination	Contamination Greater than PQL	Valid Analyses	Contamination Detection Frequency
Background/Tributary	4	0	11	36.4%	7	0	11	63.6%
Upper River	0	0	51	0.0%	13	4	50	26.0%
Lower River - Freshwater	8	2	36	22.2%	5	0	36	13.9%
Lower River - Saline	6	0	22	27.3%	3	0	21	14.3%
EPA-funded Fish Totals	18	2	120	15.0%	28	4	118	23.7%

One Upper River and one Lower River-Saline analysis for BZ #126 were rejected.
Blank contamination was less than the PQL except in samples indicated as "Contamination Greater than PQL."

304181

The TEQ analysis was conducted using data for which non-detects were set at the detection level, as well as using data for which non-detects were set at half the detection level unless the particular congener was detected in less than 15% of the samples in which case the value was set to zero. The results showed at most an order of magnitude difference in the values calculated, and given the magnitude of the estimated TQs, this does not affect the conclusion of adverse risk. This approach provides a valid indication of the potential for risk from dioxin-like congeners.

Response to EF-1.55

To reduce the uncertainty associated with PCB analysis and quantitation, comparative analyses were performed to determine, to the extent possible, a consistent quantitation basis for historical analyses, and to estimate uncertainties present in calculated lipid-normalized PCB body burdens. These analyses resulted in the standardization of PCB analytical results in fish tissue, as described in the Baseline Modeling Report (USEPA, 1999b) and Revised Baseline Modeling Report (USEPA, 2000a). Specifically, three lines of evidence (split sample analyses, interlaboratory comparisons, and theoretical "what if" analysis) were pursued to develop translation procedures for PCB fish tissue concentrations measured between 1977 and 1997. The "what if" analyses provided a reasonable basis for translating Aroclor results to a basis consistent with congener analyses, thereby reducing uncertainty.

6.3 Conceptual Model Uncertainties

No significant comments were received on Section 6.3.

6.4 Toxicological Uncertainties

Response to EF-1.56

As noted by the commentor, the text on page 158 of the ERA contains an error. As documented in Chapter 4 of the ERA, a factor of 10, not 5, was used for the interspecies uncertainty factor.

Response to EF-1.57

Some of the toxicity studies upon which TEQ-based TRVs were based used a different set of TEFs than the Van den Berg WHO/USEPA consensus values used in this assessment. In some cases the specific TEFs the authors used were provided and we were able to take this difference into account, but in some cases they didn't, which introduces uncertainty. The ERA always used the Van den Berg (1998) WHO/USEPA consensus TEFs.

Response to EF-1.58

The toxicity study upon which the TRV is based used a different set of TEFs than the Van den Berg et al. (1998) WHO/USEPA consensus values. In some cases, the specific TEFs used were noted in the studies and it was possible to take this difference into account. In other cases, the studies used to develop the TRVs did not identify the specific TEFs used. The differences were no more than 30% and typically on the order of 13% to 20% (see, ERA, p. 158).

6.5 Exposure and Modeling Uncertainties

No significant comments were received on Section 6.5.

6.5.1 Natural Variation and Parameter Error

No significant comments were received on Section 6.5.1.

6.5.1.1 Food Chain Exposures

Response to EF-1.59

The comment regarding the typographical error is acknowledged. The larger size class on p. 159 is revised to read >25 cm (rather than >20 cm) to reflect the size class used in the ERA.

Response to EF-1.60

All available data on the gut contents of Hudson River fish were used in developing the dietary preferences, for both the ERA and the baseline modeling (USEPA, 1999b, 1999c).

6.5.2 Sensitivity Analysis for Risk Models

Response to EF-1.61

The sensitivity analysis suggested in the comment was conducted. The results are presented in the baseline modeling report (USEPA, 1999b, 2000a). The sensitivity analysis showed that the lipid content of fish is the overwhelming contributor to variability in predicted fish concentrations, followed by K_{ow} and TOC in sediment. All other parameters have been shown to be much less significant contributors to predicted fish body burdens. Long-term trajectories of fish concentrations are changed by more than a factor of two only by adjusting lipid.

6.5.3 Model Error

Response to EF-1.62, EL-1.45

The uncertainty in the ERA associated with model error refers to the conceptual exposure model (see, ERA, p. 164-165). This model error is probably not a significant source of uncertainty because relationships between trophic levels and food web components in the Hudson River are well understood.

6.5.3.1 Uncertainty in FISHRAND Model Predictions

Response to EF-1.63

It is unclear what is meant by "Benthic PCB data was utilized on a wet weight basis because lipid-normalizing the data did not improve the relationships, yet lipid content in prey items was shown to be an important contributor to model uncertainty." The biota sediment accumulation factor is estimated as a lipid normalized benthic invertebrate concentration divided by an average TOC-normalized sediment concentration, and this was how BSAFs were calculated in FISHRAND. Lipid content in prey items (and in the modeled species themselves) is the single most important contributor to variance in predicted fish concentrations.

Because FISHRAND incorporates distributions rather than point estimates for important parameters (e.g., lipid content, weight, etc.), parameters are not averaged. FISHRAND incorporates full distributions for most parameters. Nearshore concentrations were used to the extent possible in the Thompson Island Pool. The limited available data suggest far less of a lateral gradient in downstream areas, and the HUDTOX model does not distinguish between nearshore and center channel below the Thompson Island Pool so as to permit the examination of nearshore effects below the Thompson Island Dam.

The factor of two is based on the relative percent difference between predicted and observed fish body burdens from the Baseline Modeling Report. The Revised Baseline Modeling Report shows typical relative percent differences less than that. However, within year variability in predicted fish concentrations is approximately a factor of two, and that, together with the relative percent differences, suggests that a factor of two is a reasonable upper-bound error limit on predicted fish concentrations.

The sensitivity analysis (presented in the Revised Baseline Modeling Report) shows that changing lipid content can change predicted concentrations by a factor of two. Predicted fish body burdens are influenced most by changes in lipid content.

7.0 CONCLUSIONS

Response to EP-1.1

Because of changes to dynamics of the Hudson River ecosystem (in particular the fishing ban) it is extremely difficult to link current observations to specific conditions in the Hudson River. The results of the models were used to estimate the effects that may be occurring from PCBs in the sediments, regardless of other conditions. See also the responses to EG-1.9, EG-1.54, EF-1.51, and EF-1.66.

Response to EP-1.2

Although the Interim Ecological Risk Assessment (USEPA, 1991b) indicated that PCB concentrations found in the Hudson River may affect the biota found there, the only decision that was made prior to the ERA was that a more detailed quantitative assessment was required. The ERA followed USEPA guidance (1997b) to provide an objective assessment of risks from PCBs in the Hudson River to biota coming into contact with the river.

Response to EG-1.12

As discussed in the response to EG-1.1, the measurement endpoints selected for the weight-of-evidence approach used correspond to the bottom up approach used for the ERA. Techniques to estimate the magnitude and severity of risks can include modeling to predict food-chain transfer and secondary toxicity of bioaccumulative chemical to upper trophic level receptors, measurement of tissue concentrations, species diversity studies, and *in-situ* bioassays (USEPA, 1999d). Most of these techniques were used in the ERA. Although there were no field or lab studies for wildlife species, the ERA looked at the consequences of exposure using various literature data on effects.

Response to EL-1.46

As discussed in the responses to previous comments (see EG-1.9 and EG-1.34, EG-1.38, EP-1.1, and EP-2.10), the presence of healthy populations does not indicate that PCBs have no effect on local fish and wildlife. Improvements in water quality and the fishing ban have undoubtably assisted the recovery and maintenance of many species. The assumptions, methodology, and approach used for the ERA were appropriately used and USEPA intends to use the ERA should be considered along with the other Phase 2 reports in its decision making process.

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

Response to EF-1.64

The following is an addition to the ERA (USEPA, 1999a) which answers the question as to whether measured and modeled sediment concentrations exceed guidelines.

5.1.1.3 Measurement Endpoint: Comparison of Sediment PCB Concentrations to Guidelines

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

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

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

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

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

Response to EG-1.34

The Hudson River is a large and complex ecosystem influenced by a variety of factors. Some clear correlations can be seen in the Hudson River ecosystem, such an increase in some fish populations due to the fishing ban or an increase in pollution-intolerant filter feeding macroinvertebrates resulting from improved water quality. More subtle effects, including those of PCBs, are difficult to discern amid the natural noise of the ecosystem. The gradient of PCB concentrations along the roughly 200 miles of river being examined also increases the difficulty of ascribing particular effects to PCBs. Therefore, the ERA discusses the potential for adverse effects even in apparently healthy receptor populations.

The NYSDEC macroinvertebrate report (1993) states that for the Lower Hudson River below Albany, improvements in species richness is "...attributed to many improvements in municipal and industrial sewage treatment." In the region in the Upper Hudson River below Glens Falls,, mayfly and caddisfly species increased "...following numerous improvements in treatment of municipal and industrial wastes." The report also states, for the Hudson River, "Some problems remain, including PCB deposits downstream of Ft. Edward." (NYSDEC, 1993, p. 132).

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

Response to EF-1.65

The analysis used young-of-year spottail shiner and age 1 pumpkinseed because of the availability of data for these groups. Concentrations of PCBs are likely to be higher in mature adults and therefore risks to local forage fish may be greater than calculated in the ERA.

Response to EF-1.66

The reduction in fishing pressure has provided a respite for Hudson River fish from one of their main predators- man. EPA concurs that this issue should have been discussed in the field-based observations section. See also responses to EG-1.9 and EG-1.54.

7.3 Assessment Endpoint: Protection and Maintenance (i.e., Survival, Growth, and Reproduction) of Local Insectivorous Birds

No significant comments were received on Section 7.3.

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

Response to EG-1.37

The conclusion that current and future concentrations of PCBs are not of a significant magnitude to prevent reproduction of waterfowl (see, ERA, p. ES-7) is in accordance with the secure mallard population found in the Hudson River Estuary. However, exposure to PCBs may reduce or impair the survival, growth, and reproductive capability of waterfowl in the Upper Hudson River. As noted in the response to EG-1.10, the small amount of tissue and egg data that USFWS collected on mallards and wood ducks in the Upper Hudson River indicate that neither species is safe for human consumption.

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

No significant comments were received on Section 7.5.

7.6 Assessment Endpoint: Protection and Maintenance (i.e., Survival, Growth, and Reproduction) of Local Wildlife

No significant comments were received on Section 7.6.

7.7 Assessment Endpoint: Protection of Threatened and Endangered Species

No significant comments were received on Section 7.7.

7.8 Assessment Endpoint: Protection of Significant Habitats

No significant comments were received on Section 7.8.

APPENDICES

APPENDIX A Site Description and Characterization

No significant comments were received on Appendix A.

APPENDIX B Ecological Field Sampling Program

No significant comments were received on Appendix B.

APPENDIX C Life History and Ecology of Dominant Macroinvertebrate Receptors

No significant comments were received on Appendix C.

APPENDIX D Life History and Ecology of Fish Receptors

No significant comments were received on Appendix D.

APPENDIX E Life History and Ecology of Avian Receptors

No significant comments were received on Appendix E.

APPENDIX F Life History and Ecology of Mammalian Receptors

No significant comments were received on Appendix F.

APPENDIX G Threatened, Endangered and Special Concern Species

No significant comments were received on Appendix G.

APPENDIX H Benthic Macroinvertebrate Community Analysis

No significant comments were received on Appendix H.

APPENDIX I Data Usability Report for PCB Congeners Ecological Study

Response to EG-1.30

The analytical data for individual congeners in biota are adequate for calculating TEQs. The base PQL (for these samples, it was expressed as the "calibrated quantitation limit," which is equivalent to the lowest calibration standard analyzed - this is a higher concentration than the actual detection limit determined from MDL studies performed for this project) was about 1.1 ug/kg (wet weight basis). Quantitation factors (i.e., the factor by which the base PQL is multiplied to obtain the sample-specific PQL) ranged from 1.0 to 10 for the 120 vs. EPA-funded fish samples. A breakdown of the quantitation factors by fish sample group is shown on attached table EG-1.30a. Reported quantitation limits were slightly elevated in four fish samples (all Upper River fish) due to blank contamination. (Associated BZ #126 blank contamination in 24 other fish samples was less than the PQL and therefore did not result in an increase in the sample-specific PQL.)

304190

TABLE EG-1.30a
Quantitation Factors for USEPA-funded Fish Sample Analysis

Sample Group	Quantitation Factor (1)								Group Total
	1	2	3	4	5	6	8	10	
Background/Tributary	11	0	0	0	0	0	0	0	11
Upper River	14	14	8	4	2	0	4	5	51
Lower River - Freshwater	10	12	5	4	2	2	1	0	36
Lower River - Saline	11	6	3	2	0	0	0	0	22
EPA-funded Fish Totals	46	32	16	10	4	2	5	5	120

(1) Qunatitation factor back-calculated based on a PQL of 1.0 to 1.2 ug/kg wet weight.

TABLE EG-1.30b
BZ #77 and BZ #126 Frequency of Detection for USEPA-funded Fish Sample Analysis

Sample Group	BZ #77			BZ#126		
	Detections	Valid Analyses	Detection Frequency	Detections	Valid Analyses	Detection Frequency
Background/Tributary	0	11	0.0%	0	11	0.0%
Upper River	51	51	100.0%	15	50	30.0%
Lower River - Freshwater	28	36	77.8%	2	36	5.6%
Lower River - Saline	15	22	68.2%	9	21	42.9%
EPA-funded Fish Totals	94	120	78.3%	26	118	22.0%

One Upper River and one Lower River-Saline analysis for BZ #126 were rejected.

304191

It is true that BZ #126 was not detected in the majority of the samples. The BZ #126 detection frequency ranged from 0% (for background/tributary samples) to 30% (for the Upper River samples). The overall frequency of BZ #126 detection for the non-background samples was about 25% (26 of 107 samples; BZ #126 data were rejected in two samples). The details are shown on Table EG-1.30b.

APPENDIX J Data Supporting TEQ Analysis

No significant comments were received on Appendix J.

APPENDIX K Examination of Exposure Pathways Based on Congener Patterns

Response to EG-1.39a

As noted in Appendix K, three different principal component analyses (PCAs) were completed. The first and foremost of these was performed using the optimized set of 29 congeners to examine all the pertinent matrices in one analysis. The results of this PCA are discussed in Section K-3. Principal component 2 for this analysis is explained on page K-7, second paragraph. Further clarification is provided here. Specifically, the component reflects the relative ratios of three portions of the congener spectrum. The first portion, represented by the positive factors for congeners BZ #4, 10, 19 and 27, reflects the extent of dechlorination products in the mixture since these congeners are all produced as dechlorination proceeds. The second portion, represented by negative factors for congeners BZ #22, 28, 31, 37, 66, 70 and 105, reflects congeners that are associated with Aroclor 1242 and in several cases (*i.e.*, BZ #66 and 70) represents congeners that are transformed to lighter forms by the dechlorination process. The last portion, represented by positive factors for several hexa to octachloro congeners (BZ# 151, 153, 170, 180 and 187), reflects the occurrence of heavier congeners in the mixture. These congeners are generally attributed to the presence of Aroclors 1254 and 1260. In the sediments, values for principal component 2 have the highest values in the Upper Hudson where the extent of dechlorination is highest and Aroclor 1242 is the predominant Aroclor source type. The lowest values occur in the freshwater Lower Hudson where extent of dechlorination is much less. Values for component 2 rise again in the saline Lower Hudson where the presence of heavier Aroclors can be seen in the sediments. These trends are illustrated in the top diagram of Figure EG-1.39a.

Component 2 was also examined in fish and was shown to exhibit quite different behavior (see the second diagram in Figure EG-1.39a). Essentially, component 2 increases with decreasing river mile just as does molecular weight. In this case, the increase in component 2 represents the apparent preferential retention of heavier congeners by fish. This occurs despite the absence of such a gradient in the sediments with river mile. As noted in Appendix K of the ERA, the origin of this trend is not known but may result from changes in the relative importance of sediment and water-based PCB exposure along with preferential loss of lighter congeners from the dissolved fraction of the water column PCB burden.

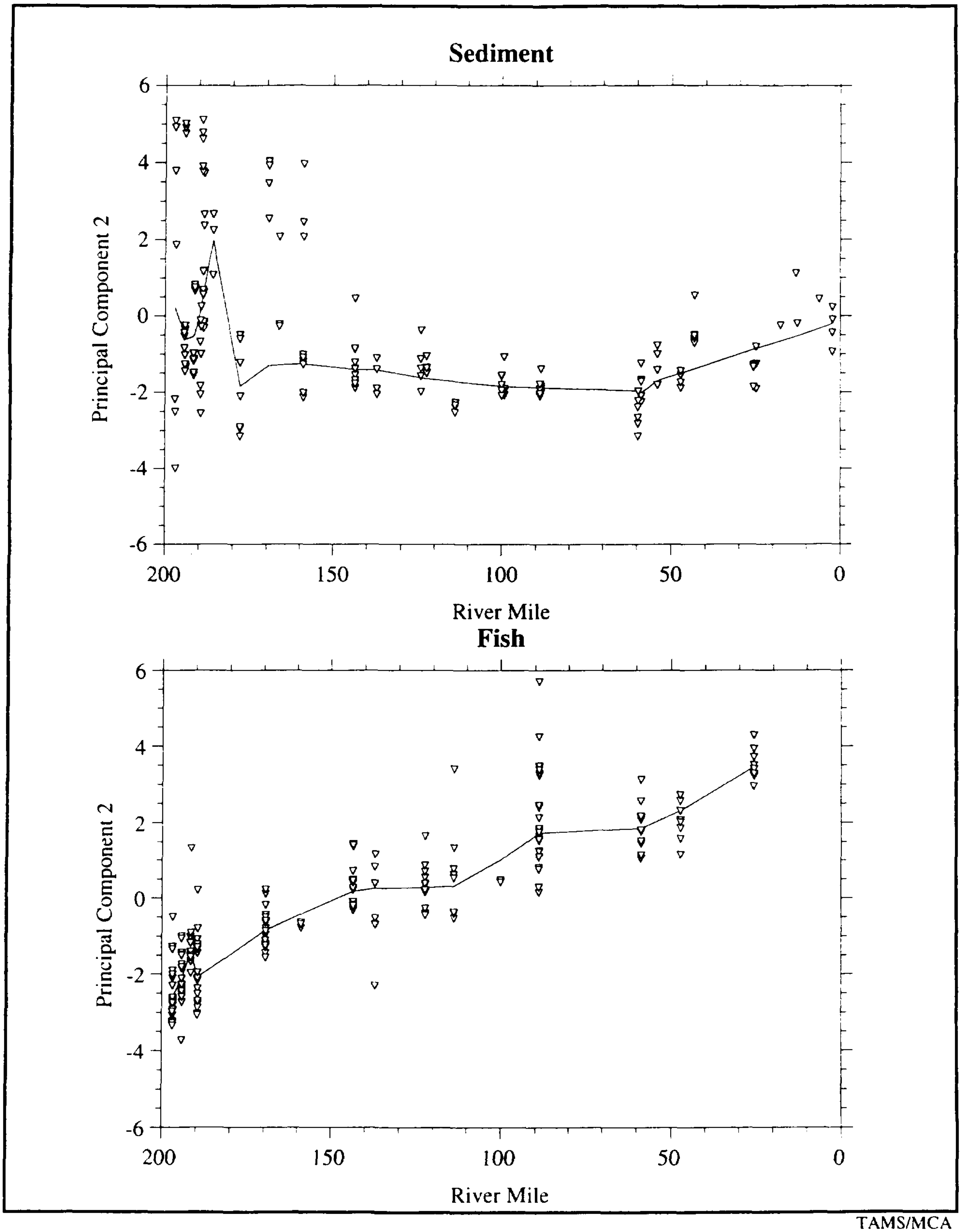


Figure EG-1.39A
Variation of Principal Component 2 with River Mile in Fish and Sediment

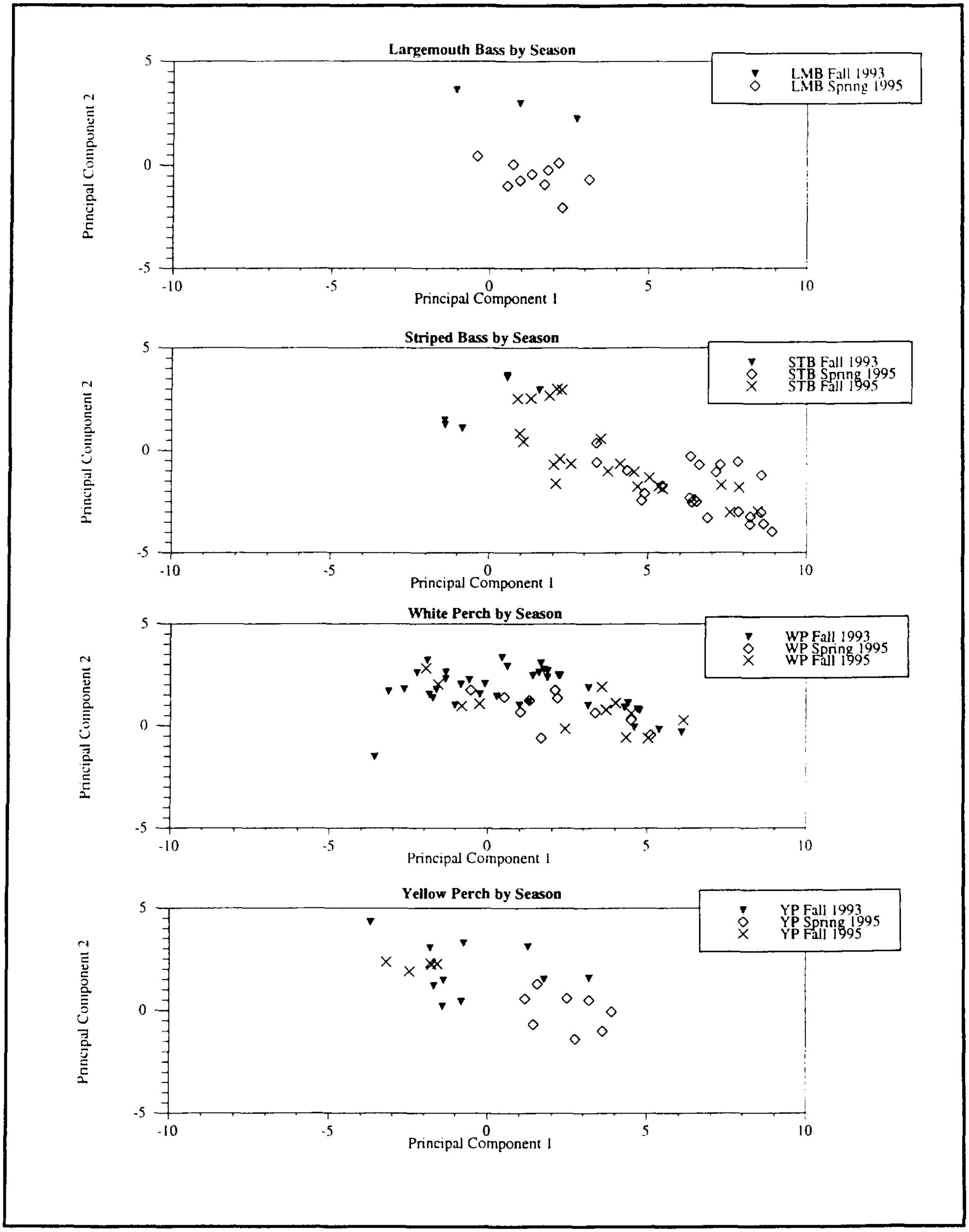
The two additional PCA analyses were used to support the conclusions of the first PCA analysis as well as more closely examine some of the specific relationships among specific sampling events (*i.e.*, Fall 1993 vs. Spring and Fall 1995). In this regard, first the initial 29 congeners were used on 1993 and 1995 fish samples only (no other matrices). Subsequently, the final PCA analysis was performed using NOAA's selection of 46 congeners for comparison purposes. In these analyses, the focus was on attempting to separate 1993 conditions from 1995 based on congener pattern, and not on the meaning of the components themselves. It is not necessary to provide interpretation for the principal components *per se*, because they do not serve to separate the 1993 and 1995 conditions despite trying two separate congener suites. Used in this manner, the statistical analysis merely serves to demonstrate what differences are statistically significant. As demonstrated by the statistical analysis, there is little difference between Fall 1993 and Fall 1995 PCB congener patterns in the Lower Hudson fish. Note that this region of the Hudson is the only place where such a comparison is possible (no data were collected in the Upper Hudson in Fall 1995). Additionally, differences between PCB congener patterns in Fall 1993 and Spring 1995 are quite similar to those between Fall 1995 and Spring 1995 data, indicating that seasonal variability is responsible for the Fall 1993 and Spring 1995 difference and not a long term change in Hudson conditions. More importantly, this analysis indicates that, in the Lower Hudson, seasonal variations in PCB patterns are far greater than the variations attributed to GE's remedial efforts between 1993 and 1995. Hence, USEPA reasonably concluded that remedial efforts by GE did not result in any substantive change in the nature of PCB exposure to Lower Hudson fish. This is further borne out in the lower diagrams of Figures K-52, K-53 and K-54, which show no statistically significant improvement in fish body burdens in the Lower Hudson between 1993 and 1995 (Note the overlap of the uncertainty bounds for each year.).

<u>Response to EG-1.39b</u>

In its analysis in Section K.3, the USEPA presents a statistical analysis of the congener patterns in fish using principal components. As part of this analysis, the USEPA shows that the most of the differences seen between the 1993 USEPA data and the 1995 NOAA data are most likely attributable to differences in life stage and not to actual changes between 1993 and 1995 in terms of the exposures to fish. Of particular importance to this assertion are Figures K-40 and K-41 which examine the principal components analysis in terms of the sampling event and the life stage, respectively. As asserted in the original discussion in Section K.3, it is clear that for largemouth bass and striped bass, all or nearly all of the differences between the two fall events is directly coincident with life stage. For white perch, there is no confounding by life stage since all animals were adults. In this instance, the range and distribution of the data from the two fall events is nearly coincident.

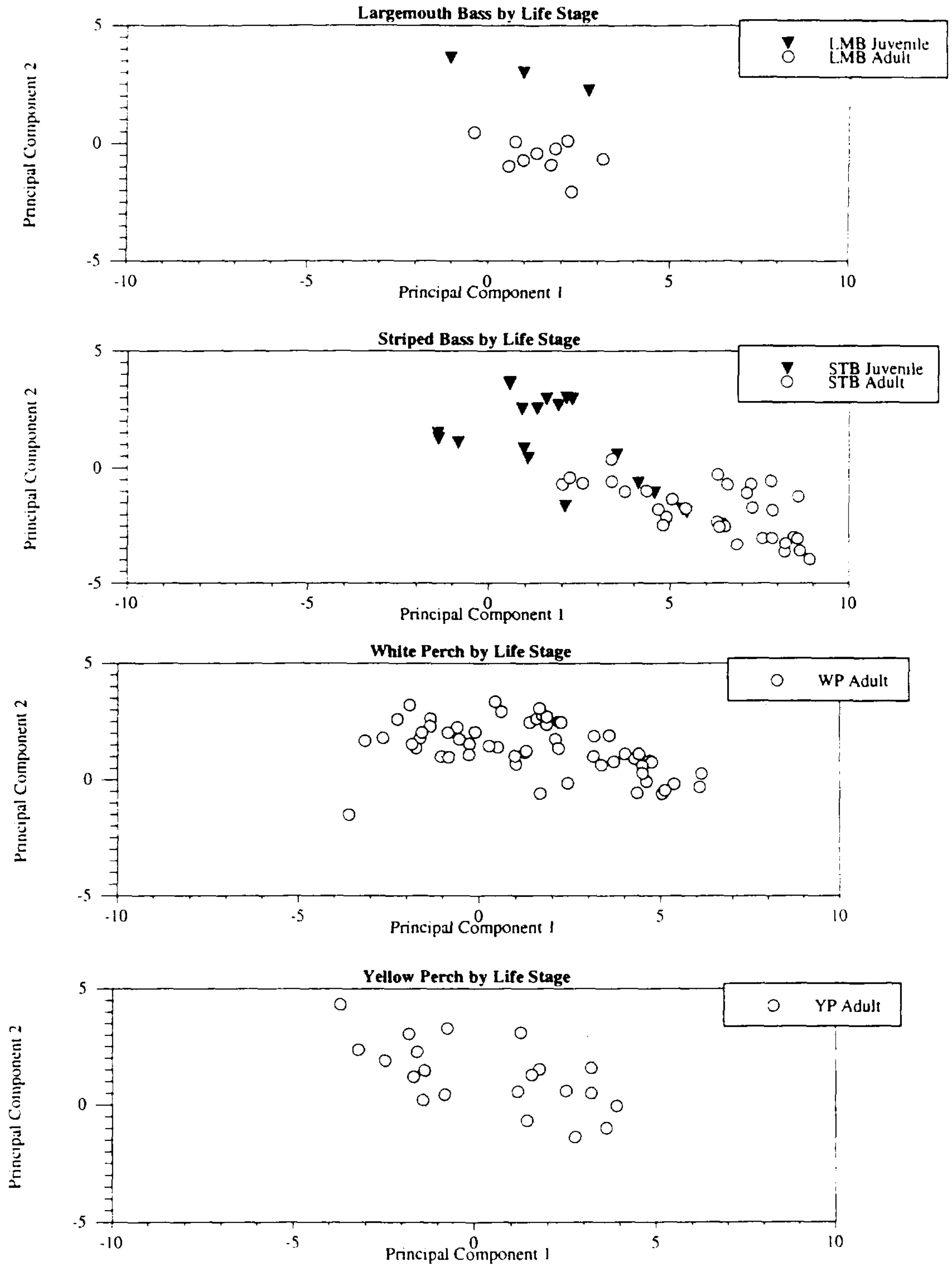
With respect to the yellow perch data, the USEPA rejects this contention and USEPA disagrees with the comment that these data are not as supportive as other fish data, in fact provided further analysis in this regard. The confounding issues of river mile and life stage for the yellow perch are examined and shown to coincide with the remaining differences in several instances (<u>see</u>, Section K.7, Figures K-49 to 51). As further clarification to this issue, Figures EG-1.39B-1 and EG-1.39B-2 were prepared. In these diagrams only Lower Hudson samples were examined. These figures should be compared to the Figures K-40 and K-41 which shows all samples for the various

304194



TAMS/MCA

Figure EG-1.39B-1
Principal Component Results for 1993 and 1995 Fish Samples by Species and Season (Lower River only, Based on 29 Congeners)



TAMS/MCA

Figure EG-1.39B-2
Principal Component Results for 1993 and 1995 Fish Samples by Life Stage (Lower River only, Based on 29 Congeners)

species. Note that there is no change for the striped bass and white perch sample relationships because these animals were only collected from the Lower Hudson. It should be noted as well that the "difference" raised by the commentor in referring to the striped bass data only refers to the small difference between the juveniles in 1993 *vs.* 1995. This difference is small compared to the large difference between Spring and Fall or adults and juveniles. This subtle difference, if real, must be compared to the absence of change apparent in the white perch and yellow perch data.

Examining the yellow perch results after the removal of Upper Hudson samples (collected only in Fall 1993 and Spring 1995), the remaining distribution of Fall 1995 samples is contained within coincides with the range of the Fall 1995 samples. The only substantive difference is that between the two Fall events and that of the Spring 1995 conditions. This evidence directly supports the original assertion in Section K-3 (*i.e.*, no significant difference between the 1993 and 1995 Fall fish congener patterns).

Similar results were seen for white perch, as discussed in Sections K.3 and K.7.

For largemouth bass in the Lower Hudson, there are no data from Fall 1995 and the only apparent difference between the Fall 1993 and Spring 1995 is directly coincident with life stage.

Based on the diagrams presented here as well as those in Appendix K, it is apparent that there was little change in the congener pattern of fish from the Lower Hudson between 1993 and 1995. Any change that may have occurred is very small relative to the seasonal variation apparent in the data. Since the pattern of congeners which constitute the fish body burdens is essentially the same in the same area of the river at the same time of year over a two year period, it is clear that little has changed in the factors which produce these fish body burdens (*i.e.*, the basic routes of exposure – water and sediment pathways and their associated concentrations) in the Lower Hudson. Thus, the remedial efforts at the GE facilities did not result in measurable changes in the patterns of fish body burdens in the Lower Hudson. Although fish body burdens did decline in the Upper Hudson, as noted by the commentor as well as by USEPA in Section K.8 of the ERA, the body burdens in the Lower Hudson were not statistically lower in 1995 relative to 1993. The major change appears to be a decline in the degree of variability in Lower Hudson fish levels. These results suggest the remedial efforts conducted by GE were unimportant to conditions in the Lower Hudson, or at a minimum, it will take a long period of time before the effects of the 1993-1995 remedial efforts will be seen in the Lower Hudson. Notably 1996 fish levels were not different from those in 1993 to 1995 for two of the three species examined in the Lower Hudson.

In regard to the exclusion of Fall 1995 data above the federal dam, it should be noted that no data were collected above the dam during this sampling event.

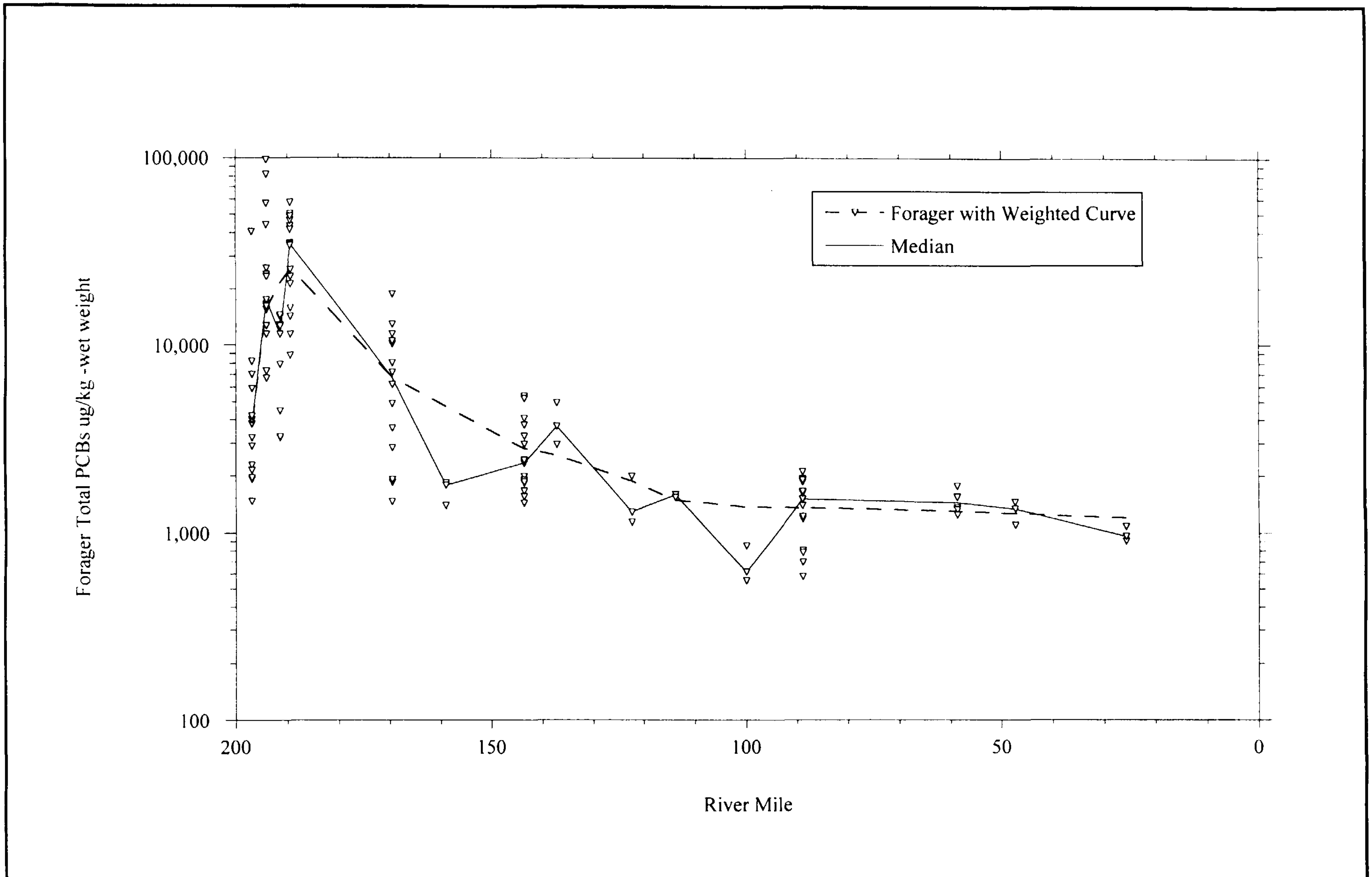
Response to EG-1.39c

The discussion presented in Appendix K of the ERA focused primarily on the resolution of the sources to fish in the context of sediment verses water-derived exposures. As part of the analysis presented, evidence showing a clear deviation in PCB congener pattern in fish relative to sediment ruled out the use of fish PCB patterns for the purposes of tracing PCB sources in the Hudson. The discussion of the occurrence of other external sources to the Hudson was discussed at length in the DEIR (USEPA, 1997a). As discussed in the DEIR, the sediments represent the ultimate recorders of sources to the river, not the biota. This is because the sediment record is governed by the relatively simpler geochemical processes of sediment absorption and desorption. PCB body burdens in fish are subject to various biological processes such as absorption, metabolism and depuration which affect the individual congeners to varying degrees. The combined effect of these processes is not well known but, as documented in Appendix K, section K.9, these processes do not affect all congeners equally. Therefore, the congener patterns within the biota, particularly within fish, cannot serve as "fingerprints" of the various sources to which fish are exposed.

With regard to the assertion that PCB body burdens in forage fish begin rising below RM 120, USEPA strongly disagrees. As shown in Figures EG-1.39C-1 and EG-1.39C-2, PCB levels in forage fish exhibit only a downward trend between river miles 140 and 80. This is evident when viewed on both a wet weight and a lipid-normalized basis. Fish body burdens only begin to rise at RM 60, a location within the saline portion of the Hudson and then only on a lipid basis. There is no evidence of a forage fish body burden rise beginning at RM 120. While the variability at RM 88.9 is quite large, it is important to note that both high and low values are associated with this sampling location. The median lipid-normalized value for this location is the same as that seen in the two upstream locations. The wet weight value is within error of two of the three nearest upstream stations and thus is also without a trend. Thus, the assertion that a substantive source of heavier congeners exists within the freshwater Lower Hudson is not supported by these data.

In the uppermost diagrams of the commentor's Figures 2a and 2b, the commentor fails to present estimates of the uncertainties of the species means. Notably, the range of BZ#138 concentrations for each species at RM 88.9 is quite large. Additionally, three of the six species plotted at RM 88.9 are at or below the value seen at the previous upstream station. Lastly, the diagrams are based on mean values for fish body burdens which are typically log-normal. For these data sets, the mean is a poor measure of central tendency and can be badly distorted by the presence of a single outlier value. A median-based approach would be more appropriate because it is a far more robust estimate of central tendency. Nevertheless, due to the known variability of fish body burdens and the ability of fish to extensively modify PCB patterns relative to their exposures, the fish cannot be used to identify external sources to the Lower Hudson unless body burden changes are large and spatially extensive. This is clearly not the case above RM 60.

The assertion that fish body burdens are not simply related to their PCB exposures is further reinforced by the principal component data analysis presented in Figures K-14 and K-15 of Appendix K of the ERA and Figure EG-1.38A of this responsiveness summary, which show distinctly different trends for fish versus river mile for the two main principal components. That is, principal component values tend to rise much faster for fish than sediment with respect to river mile, indicative of the biological processes related to retention of PCBs by fish and unrelated to source as recorded by the sediments.

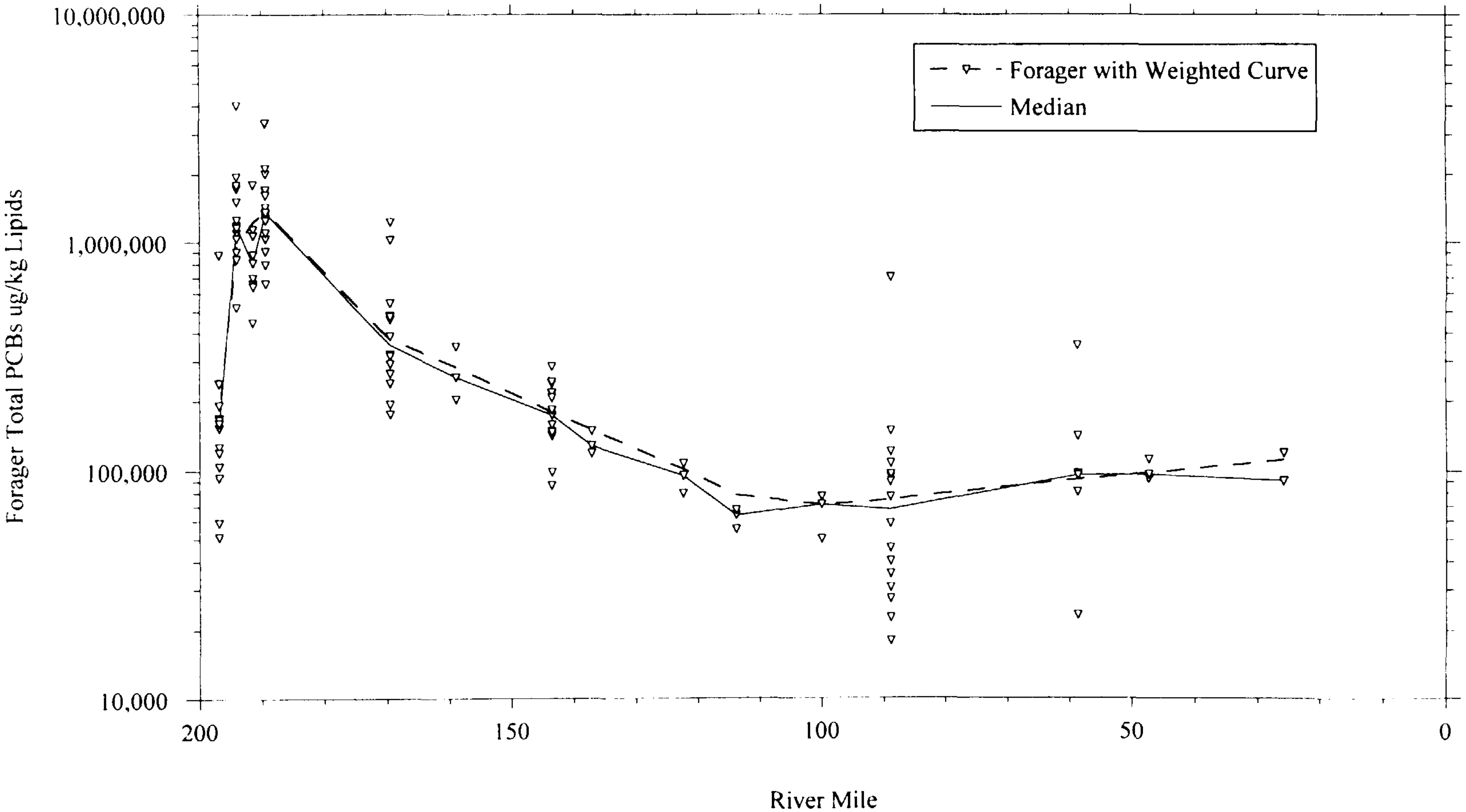


TAMS

Figure EG-1.39C-1
Forager Body Burden Wet Weight versus River Mile

304199

304200



TAMS

Figure EG-1.39C-2
Forager Body Burden Normalized to Lipids versus River Mile

Further evidence of the lack of external sources to the Lower Hudson can be seen in Figures EF-1.32b to EF-1.32e. These figures show the absence of substantive congener ratio variation in the Lower Hudson above RM 60. Below this point, however, there is clear evidence for at least one and possibly two sources. The influence of these sources, as documented by the sediments, is limited to the region below the salt front (see, USEPA, 1997a). In this region, the cumulative effects of these sources appear to be of the same magnitude as the upstream GE-related load.

The last issue raised by the commentor's Figures 2a and 2b pertains to the expected effects of downstream dilution on PCB concentrations in fish. As discussed at length in the DEIR (USEPA, 1997a), Section 3.3.3 as well as in DG-1.17 of the Responsiveness Summary for the DEIR USEPA, 1998c), PCB levels do not decline in the Hudson as would be expected for a simple conservative substance. This is because of differing rates of suspended sediment and flow yields among the various tributaries. These factors can be normalized for sediment PCB concentrations by using cesium-137 as was done in the DEIR. There is no similar normalizing parameter for fish, especially in light of the internal biological processes that serve to modify the PCB burden in fish. Thus, there is no simple model to describe how fish body burdens should decline with river mile. However, as shown in Figures EG-1.30a to 1.30f, fish body burdens do parallel sediment PCB concentrations with river mile, suggesting that the simple model of PCB dilution developed for the DEIR to explain sediment concentrations may have some relation to the fish observations as well.

USEPA disagrees with the commentor's assertion that, below the salt front, PCB concentrations in fish should decline as a conservative substance in the absence of additional sources. PCB concentrations will not act as a simple conservative substance for several reasons. First, significant partitioning behavior for PCBs means that both flow and solids loads must be considered when performing such a calculation. Ocean water, the primary diluant below the salt front, is significantly depleted in suspended solids relative to the freshwater Hudson and thus will not appreciably dilute suspended matter concentrations of PCBs. This concern is of greatest significance to the heavier congeners which also constitute the primary portion of the fish body burden. Thus the lower diagrams are misleading and inappropriate for comparison. More importantly, the USEPA has documented the occurrence of significant PCB sources to the Hudson below the salt front (USEPA, 1997a). Thus conservative behavior is not expected in this region. A rough estimate of the local loading in New York harbor was made in the DEIR. A similar estimate based on fish is not appropriate for all of the reasons discussed previously.

Response to EG-1.39d

USEPA stands by its conclusions regarding the use of congener ratios in fish to ascertain sources. It is clear from the discussion presented in Section K.9 and the Figures K-56 to K-59 that the fish congener ratios for the four congener pairs vary in ways that are not readily explained by the ratios seen in the exposure media. In fact, while in some cases fish congener ratios are bracketed by the range of ratios seen in the exposure media, many times they are not bracketed and clearly lie outside the exposure range, indicating that food web or internal biological processes are affecting the ratio and erasing its relationship to the exposure media. In particular, the USEPA notes that the ratio chosen by the commentor (BZ#56/BZ#49) is just an example, with the fish ratio lying well below the exposure media ratios for nearly the entire Hudson. Notably, this is different from the media to fish relationships seen for the other congener pairs ratios.

While it may be that the BZ#56/BZ#49 ratio is appropriate to apply as the commentor suggests, it is important to note that other congener pairs would lead to different conclusions if applied in the same manner. For congener pairs BZ#66/BZ#49 and BZ#74/BZ#49, ratios seen in the fish of the TI Pool (RM 188.5 to 195) substantively exceed the surface sediment ratio as well as the ratio in Aroclor 1242, perhaps indicating preferential enhancement of the numerator congeners relative to BZ#49 (see, Figures K-58 and K-59, respectively). In the case of BZ#66/BZ#49, no exposure media matches the ratio seen in the fish. In the case of BZ#74/BZ#49, the fish are matched only by the suspended matter. Thus, it is USEPA's assertion that these ratios do not provide a unique interpretation or estimate of exposure. With regard to the conclusion regarding dechlorinated sediments, the USEPA does not accept the assertion that these ratios prove the absence of exposure to these sediments as the report concludes. Because the four congener pairs give four different relationships with respect to the sediments, it is not possible to select the "correct" one without other information. This ambiguity in the congener ratios precludes their usefulness, leading to the following conclusions of the report:

1. Congener patterns are extensively modified by the food web and related biological processes and thus congener ratios are precluded from use as tracer for PCB source (*i.e.*, they cannot be used as fingerprints).

2. Congener patterns in the biota are unique relative to the sediments and water and are produced as the combined result of exposure and biological processes (processes probably internal to the fish).

The commentor is correct in noting that the second sentence of the first paragraph on page K-28 should state that the partition coefficient for BZ#49 about half (not twice) that of the other congeners. This correction does not change the conclusions of the paragraph.

Finally, because of these clearly documented issues with these congener pairs plus the documented differences between fish and the other exposure media based on principal components, the USEPA has concluded that the congener ratios and patterns found in fish are substantively altered and bear only a passing resemblance to their sources, thereby precluding their use as "fingerprints" of the PCB exposure. While it may be that one of the congeners already examined or perhaps one yet to be examined does in fact yield the correct relationship between the fish and the exposure media, there is no *a priori* basis on which to select this congener based on the congener data alone. Thus "fingerprinting" of the fish body burdens through the use of congener patterns in the Hudson is not possible.

References

Adams, S.M., K.L. Shepard, M.S. Greeley Jr., B.D. Jimenez, M.G. Ryon, L.R. Shugart, and J.F. McCarthy. 1989. The use of bioindicators for assessing the effects of pollutant stress on fish. *Marine Environmental Research*. 28:459-464.

Adams, S.M., L.R. Shugart, G.R. Southworth and D.E. Hinton. 1990. Application of bioindicators in assessing the health of fish populations experiencing contaminant stress. In: J.F. McCarthy and L.R. Shugart, eds., Biomarkers of Environmental Contamination. Lewis Publishers, Boca Raton, FL. Pp. 333-353.

Adams, S.M., W.D. Crumby, M.S. Greeley, Jr., M.G. Ryon, and E.M. Schilling. 1992. Relationships between physiological and fish population responses in a contaminated stream. *Environmental Toxicology and Chemistry*. 11:1549-1557.

Aulerich, R.J. and R.K. Ringer. 1977. Current Status of PCB Toxicity, Including Reproduction in Mink. *Arch. Environm. Contam. Toxicol.* 6:279-292.

Bengtsson, B.E. 1980. Long-term effects of PCB (Clophen A50) on growth, reproduction and swimming performance in the minnow, *Phoxinus phoxinus*. *Water Research*, Vol. 1, pp. 681-687.

Bopp, R.F. and H.J. Simpson. 1989. Contamination Of The Hudson River. The Sediment Record. Columbia University.

Chilrud, S. N. 1996. Transport and Fate of Particle Associated Contaminants in the Hudson River Basin. Ph.D. Thesis. Columbia University.

Custer, T.W., and G.H. Heinz. 1980. Reproductive success and nest attentiveness of mallard ducks fed Aroclor 1254. *Environmental Pollution* (Series A). 21:313-318.

Deck, B. L. 1980. Nutrient-Element Distributions in the Hudson Estuary. Ph.D. Thesis. Columbia University.

Elliott, J.E., R.J. Norstrom, A. Lorenzen, L.E. Hart, H. Philibert, S.W. Kennedy, J.J. Stegeman, G.D. Bellward, and K.M. Cheng. 1996. Biological effects of polychlorinated bibenzo-p-dioxins, dibenzofurans, and biphenyls in bald eagle (*Haliaeetus leucocephalus*) chicks. *Environmental Toxicology and Chemistry*. Vol. 1(5):782-793.

Elonen, G.E., R.L. Spehar, G.W. Holcombe, R.D. Johnson, J.D. Fernandex, R.J. Erickson, J.E. Tietge, and P.M. Cook. 1998. Comparative toxicity of 2,3,7,8-tetrachlorodibenzo-p-dioxin to seven freshwater fish species during early life-stage development. *Environmental Toxicology and Chemistry*. 17:3, pp 472-483.

Exponent. 1998. Data report: Macroinvertebrate communities and diets of selected fish species in the Upper Hudson River, Volumes I and II. Prepared for General Electric Company.

Garvey, E. A., C. Hunt, H. Pang, and J.B. Butcher. 1999. Fate and Transport in the Hudson River and Estuary as Documented in the Sediments. SETAC 20th Annual Meeting Abstract Book.

Giesy, J.P., W.W. Bowerman, M.A. Mora, D.A. Verbrugge, R.A. Othoudt, J.L. Newsted, C.L. Summer, R.J. Aulerich, S.J. Bursian, J.P. Ludwig, G.A. Dawson, T.J. Kubiak, D.A. Best, and D.E. Tillitt. 1995. Contaminants in fishes from Great Lakes-influenced sections and above dams of three Michigan rivers: III. Implications for health of bald eagles. *Archives of Environmental Contamination and Toxicology*. 29:309-321.

Gobas, F. A.P.C., E.J. McNeil, L Lovett-Doust and G.D. Haffner. 1991. Bioconcentration of chlorinated aromatic hydrocarbons in aquatic macrophytes. *Environ. Sci. Technol.* 25(5):924-929.

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

Hamilton, W.J. 1959. Foods of mink in New York. N.Y. Fish and Game J. 6: 77-85. In USEPA, 1993. Wildlife Exposure Factors Handbook. Office of Research and Development, Washington, D.C. EPA/600/R-93/187b.

Hamilton, W.J. 1961. Late fall, winter and early spring foods of 141 otters from New York. *N.Y. Fish and Game J.* 8:242-252.

Hansen, D.J., S.C. Schimmel, and E. Matthews. 1974. Avoidance of Aroclor 1254 by shrimp and fishes, *Bull. of Environ. Contam. and Toxicol.* 12(2):253-293.

Haseltine, S.D., and R.M. Prouty. 1980. Aroclor 1242 and reproductive success of adult mallards (*Anas platyrhynchos*). *Environmental Research*. 23:29-34.

Heath, R.G., J.W. Spann, J.F. Kreitzer, and C. Vance. 1972. Effects of polychlorinated biphenyls on birds. pp. 475-485. In: The XVth International Ornithological Congress. K.H. Voous (ed). Leiden, E.J. Brill, The Hague, Netherlands.

Heaton, S.N., S.J. Bursian, J.P. Giesy, D.E. Tillitt, J.A. Render, P.D. Jones, D.A. Verbrugge, T.J. Kubiak, and R.J. Aulerich. 1995. Dietary exposure of mink to carp from Saginaw Bay, Michigan. 1. Effects on reproduction and survival, and the potential risk to wild mink populations. *Archives of Environmental Contamination and Toxicology*. 28:334-343.

Hill, E., R.G Heath, J.W. Spann, and J.D. Williams. 1975. Lethal Dietary Toxicities of Environmental Pollutants to Birds. U.S. Fish and Wildlife Special Scientific Report – Wildlife No. 191, Washington, DC.

Linder, R.E., T.B. Gaines, R.D. Kimbrough. 1974. The effect of polychlorinated biphenyls on rat reproduction. *Food Cosmet. Toxicol.* 12:63-77.

Linzey, A.V. 1988. Effects of chronic polychlorinated biphenyls exposed on growth and reproduction of second generation white-footed mice (*Peromyscus leucopus*). *Arch. Environ. Contam. Toxicol.* 17:39-45.

Lovett-Doust, L., J. Lovett Doust and M. Biernacki. 1997. American wildcelery, Valisneria americana, as a biomonitor of organic contaminants in aquatic ecosystems. *J. Great Lakes Res.* 20(2):333-354.

MacDonald, D. 2000. Letter dated 12/1/99 from Don MacDonald (MacDonald Environmental Sciences Ltd.) to Michael T. Huguenin (Industrial Economics, Inc.). Prepared for NOAA. Personal communication between Lisa Rosmann (NOAA) and Alison Hess (USEPA).

Mayack, D.T. August 30, 1999. NYSDEC Division of Fish, Wildlife and Marine Resources, Conservation Biologist. Hale Creek Field Station, Gloversville, New York. Personal communication by telephone with H. Chernoff, TAMS Consultants, Inc.

McCarty, J.P. and A.L. Secord. 1999a. Reproductive ecology of tree swallows (*Tachycineta bicolor*) with high levels of polychlorinated biphenyl contamination. *Environmental Toxicology and Chemistry.* 18(7):1433-1439.

McCarty, J.P. and A.L. Secord. 1999b. Nest-building behavior in PCB-contaminated tree swallows. *The Auk.* 116(1):55-63.

McCoy, G., M.F. Finlay, A. Rhone, K. James, and G.P. Cobb. 1995. Chronic polychlorinated biphenyls exposure on three generations of oldfield mice (*Peromyscus polionotus*): effects on reproduction, growth, and body residues. *Archives of Environmental Contamination & Toxicolgy.* 28:431-435.

Monosson, E. 1999. Letter to M. Huguenin, President Industrial Economics, Inc. November 5, 1999.

National Oceanographic and Atmospheric Administration (NOAA). 1997. Hudson River Congener-specific Analysis, Data Summary and Analysis Report. Prepared by EVS Environmental Consultants. July, 1997.

National Oceanographic and Atmospheric Administration (NOAA). 1999a. Development and Evaluation of Consensus-Based Sediment Effect Concentrations for PCBs in the Hudson River. Prepared for NOAA Damage Assessment Center, Silver Spring, MD. Prepared through Industrial Economics by MacDonald Environmental Sciences Ltd. March, 1999.

National Oceanographic and Atmospheric Administration (NOAA). 1999b. Reproductive, Developmental and Immunotoxic Effects of PCBs in Fish: a Summary of Laboratory and Field Studies. Prepared for NOAA Damage Assessment Center, Silver Spring, MD. Prepared through Industrial Economics Inc. by E. Monosson. March, 1999.

Nebeker, A.V., F.A. Puglisi, and D.L. Defoe. 1974. Effect of polychlorinated biphenyl compounds the survival and reproduction of the fathead minnow and flagfish. *Trans. Am. Fish. Soc.* 3:562-568.

New York State Department of Environmental Conservation (NYSDEC). 1993. 20 Year Trends in Water Quality of Rivers and Streams in New York State Based on Macroinvertebrate Data 1972-1992. By R.W. Bode, M.A. Novak and L.E. Abele, Stream Biomonitoring Unit, Bureau of Monitoring and Assessment, Division of Water.

New York State Department of Environmental Conservation (NYSDEC). 1998. Technical Guidance for Screening Contaminated Sediments, New York State Department of Environmental Protection, Division of Fish and Wildlife, Division of Marine Resources. March, 1998.

New York State Department of 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.

Nimii, A.J. (1983) Biological and Toxicological Effects of Environmental Contaminants in Fish and Their Eggs. *Can J. Fish Aquat. Sci.* 40: 306-312.

Nye, P. 2000. E-mail from P. Nye, Endangered Species Unit, Division of Fish and Wildlife, New York State Dept. Environmental Conservation to H. Chernoff, TAMS Consultants, Inc. February 25, 2000.

Riseborough, R.W. and D.W. Anderson. 1975. Some effects of DDE and PCB on mallards and their eggs. J. Wild. Manage. 39: 508-513.

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.

Scott, M.L. 1977. Effects of PCBs, DT, and mercury compounds in chickens and Japanese quail. *Federation Proceed.* 36:1888-1893.

Secord, A.L. and J.P. McCarty. 1997. Polychlorinated biphenyl contamination of tree swallows in the Upper Hudson River Valley, New York. US Fish and Wildlife Service, Cortland, NY.

Spinola, R.M., T.L. Serfass, and R.P. Brooks. 1998. Radiotelemetry Study: River Otters Reintroduction at Letchworth State Park, Progress Report.

Spinola, R.M. 1999. Graduate Student, Penn State Cooperative Wetland Center, School of Forest Resources, The Pennsylvania State University, University Park, PA. Personal Communication by telephone with J. Schaffer, TAMS Consultants.

Swackhamer, D.L. and R.S. Skoglund. 1993. Bioaccumulation of PCBs by algae: Kinetics versus equilibrium. *Environ. Toxicol. Chem.* 13:949-962.

Tillett, D.E. 1997. Determination of 2,3,7,8- tetrachloro dibenzo-p-dioxin equivalents (TCDD-eq) with the PLHC-1 bioassay: pumpkinseed collected from the Hudson River, Nov 11, 1997. Final Lab Report Number FY-98-30-01, Report to USFWS and NOAA, 2 pp.

United States Army Corps of Engineers (USACE) Waterways Experiment Station. 1988. Environmental Effects of Dredging Technical Notes. No. EEDP-01-13. Relationship Between PCB Tissue Residues and Reproductive Success of Fathead Minnows. April 1988.

United States Environmental Protection Agency (USEPA). 1991. Phase 1 Report - Reveiw Copy Interim Characterization and Evaluation, Hudson River PCB Reassessment RI/FS. USEPA Work Assignment No. 013-2N84. Prepared by TAMS/Gradient. August, 1991

United States Environmental Protection Agency (USEPA). 1993a. Phase 2B Sampling and Analysis/Quality Assurance Project Plan, Volume 2 Benthic Invertebrate and Sediment Grab Sampling, Hudson River PCB Reassessment RI/FS, February 18, 1993. Prepared for USEPA, Region II, New York. Prepared by TAMS/Gradient. February, 1993.

United States Environmental Protection Agency (USEPA). 1993b. Interim Report on Data and Methods for Assessment of 2,3,7,8-tetrachlorodibenzo-p-dioxin Risks to Aquatic Life and Associated Wildlife. EPA/600/R-93/055. Office of Research and Development. Washington, DC.

United States Environmental Protection Agency (USEPA). 1993c. Wildlife Exposure Factors Handbook. Office of Research and Development, Washington, DC. EPA/600/R-93/187a. December, 1993.

United States Environmental Protection Agency (USEPA). 1995. Great Lakes Water Quality Initiative Criteria Documents for the Protection of Wildlife. Office of Water, Washington, DC. EPA/820/B-95/008. March, 1995.

United States Environmental Protection Agency (USEPA). 1997a. Phase 2 Report, Further Site Characterization and Analysis, Volume 2C- Data Evaluation and Interpretation Report, Hudson River PCBs Reassessment RI/FS. Prepared for USEPA, Region II, New York. Prepared by TAMS *et al.* February, 1997a.

United States Environmental Protection Agency (USEPA). 1997b. Ecological Risk Assessment Guidance for Superfund: Process for Designing and Conducting Ecological Risk Assessments. Interim Final. Environmental Response Team, Edison, NJ. EPA 504/R-97/006.June 5, 1997.

United States Environmental Protection Agency (USEPA). 1998a. Database for the Hudson River PCBs Reassessment RI/FS. Release 4.1 (Compact Disk). Prepared for USEPA, Region II and the US Army Corps of Engineers, Kansas City District. Prepared by TAMS Consultants, Inc. August, 1998.

United States Environmental Protection Agency (USEPA). 1998b. Hudson River PCBs Reassessment RI/FS; Phase 2 Ecological Risk Assessment, Scope of Work. Prepared for USEPA, Region II and the US Army Corps of Engineers, Kansas City District. Prepared by TAMS/MCA. September, 1998.

United States Environmental Protection Agency (USEPA). 1998c. Hudson River PCBs Reassessment RI/FS; Responsiveness Summary for Volume 2A: Database Report; Volume 2B: Preliminary Model Calibration Report; Volume 2C: Data Evaluation and Interpretation Report. Prepared for USEPA, Region II and the US Army Corps of Engineers, Kansas City District. Prepared by TAMS/LTI/MCA/TetraTech. December, 1998.

United States Environmental Protection Agency (USEPA). 1999a. Hudson River PCBs Reassessment RI/FS; Responsiveness Summary for Phase 2- Ecological Risk Assessment, Scope of Work. Prepared for USEPA, Region 2 and the US Army Corps of Engineers, Kansas City District. Prepared by TAMS/MCA. April, 1999.

United States Environmental Protection Agency (USEPA). 1999b. Further Site Characterization and Analysis, Volume 2D- Baseline Modeling Report Hudson River PCBs Reassessment RI/FS. Prepared for USEPA Region 2 and US Army Corps of Engineers, Kansas City District. Prepared by Limno-Tech, Inc. (LTI), Menzie-Cura & Associates, Inc. (MCA), and Tetra-Tech, Inc. May, 1999.

United States Environmental Protection Agency (USEPA). 1999c. Hudson River PCBs Reassessment RI/FS; Volume 2E- Baseline Ecological Risk Assessment. Prepared for USEPA, Region 2 and the US Army Corps of Engineers, Kansas City District. Prepared by TAMS/MCA. August, 1999.

United States Environmental Protection Agency (USEPA). 1999d. Issuance of Final Guidance: Ecological Risk Assessment and Risk Management Principles for Superfund Sites. OSWER Directive 9285.7-28P. October 7, 1999.

United States Environmental Protection Agency (USEPA). 1999e. Hudson River PCBs Reassessment RI/FS; Volume 2E- Baseline Ecological Risk Assessment for Future Risks in the Lower Hudson River. Prepared for USEPA, Region 2 and the US Army Corps of Engineers, Kansas City District. Prepared by TAMS/MCA. December, 1999.

United States Environmental Protection Agency (USEPA). 2000a. Further Site Characterization and Analysis, Revised Baseline Modeling Report Hudson River PCBs Reassessment RI/FS. Prepared for USEPA Region 2 and US Army Corps of Engineers, Kansas City District. Prepared by TAMS Consultants, Inc., Limno-Tech, Inc. (LTI), Menzie-Cura & Associates, Inc. (MCA), and Tetra-Tech, Inc.

United States Environmental Protection Agency (USEPA). 2000b. Responsiveness Summary for the Peer Review 2. Hudson River PCBs Reassessment RI/FS. Prepared for USEPA Region 2 and US Army Corps of Engineers, Kansas City District. Prepared by TAMS Consultants, Inc., Menzie-Cura & Associates, Inc. (MCA), and Tetra-Tech, Inc.

United States Fish and Wildlife Service (USFWS) letter from David A. Stilwell, USFWS to Alison Hess, US EPA Region 2, October 4, 1999.

Van den Berg, M., L. Birnbaum, A.T.C. Bosveld, B. Brunstrom, P. Cook, M. Feeley, J.P. Giesy, A. Hanberg, R. Hasegawa, S.W. Kennedy, T. Kubiak, J. C. Larsen, F.X. Rolaf van Leeuwen, A.K. Jjien Liem, C. Nolt, R.E. Peterson, L. Poellinger, S. Safe, D. Schrenk, D. Tillitt, M. Tyslind, M. Younges, F. Waern, and T. Zacharewski. 1998. Toxic equivalency factors (TEFs) for PCBs, PCDDs, PCDFs for humans and wildlife. *Environmental Health Perspectives*. Vol. 106 (12):775-792.

Walker, M.K., P.M. Cook, A.R. Batterman, B.C. Butterworth, C. Berini, J.J. Libal, L.C. Hufnagle, and R.E. Peterson. 1994. Translocation of 2,3,7,8-tetrachlorodibenzo-p-dioxin from adult female lake trout (*Salvelinus namaycush*) to oocytes: effects on early life stage development and sac fry survival. *Can J. Fish. Aquat. Sci.* **51**: 1410-1419.

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

Westin, D.T., C.E. Olney, and B.A. Rogers. 1985. Effects of parental and dietary organochlorines on survival and body burdens of striped bass larvae. *Transactions of the American Fisheries Society*. 114:125-136.

Wiemeyer, S.N., T.G. Lamont, C.M. Bunck, C.R. Sindelar, F.J. Gramlich, J.D. Fraser, and M.A. Byrd. 1984. Organochlorine pesticide, polychlorobiphenyl, and mercury residues in bald eagle eggs-1969-79- and their relationships to shell thinning and reproduction. Arch.. Environ. Contam. Toxicol. 13: 529-549.

Wiemeyer, S.N., C.M. Bunck, and C.J. Stafford. 1993. Environmental Contaminants in bald eagle eggs – 1980-84- and further interpretations of relationships to productivity and shell thickness. *Archives of Environmental Contamination and Toxicology*. 24:213-227.

304210

Section III. Revised Calculations for the Baseline Ecological Risk Assessment (ERA) August 1999

Overview

This section of the Responsiveness Summary reflects revisions made to the HUDTOX and FISHRAND models, as documented in the Revised Baseline Modeling Report (RBMR)(USEPA, 2000a), and their subsequent effects on the results and implications of the Ecological Risk Assessment. Changes affect the Exposure Assessment (Chapter 3) and the Risk Characterization (Chapter 5) of the August 1999 ERA. In addition, some of the original toxicity reference values (TRVs) selected based on comments received on the ERA which result in changes to the Effects Assessment (Chapter 4). These revisions change some of the risks calculated for the current (1993) Upper and Lower Hudson River conditions and future (1993-2018) Upper Hudson River conditions. However, the revisions do not change the conclusions of the August 1993 ERA for any receptors of concern.

Changes in the Modeled Concentrations of PCBs in Fish, Water and Sediment

The RBMR contains the results of the recalibration of the HUDTOX and FISHRAND models. Because these recalibrations yielded revised values for sediment, water, and fish in the forecast results, it was necessary to revise the ecological risk assessment to reflect the new values. The changes in the HUDTOX and FISHRAND models reflected in the RBMR include the following:

- Use of a revised sediment resuspension model component in HUDTOX;
- Use of the 1998 surface (0-5 cm) sediment data obtained by GE as part of the calibration;
- An extension of the model forecast to a 70 year period (1998 to 2067);
- Use of the 1991 sediment conditions as the initial conditions for the HUDTOX model forecasts (*i.e.,* after calibration, the model was initialized with the 1991 sediment conditions and run to the year 2067);
- Recalibration of the FISHRAND model using Bayesian updating techniques; and
- Incorporation of individual species growth rates in the FISHRAND model.

The revised HUDTOX model results indicate that sediment concentrations increased slightly (10-30%) or remained the same (see Appendix A of the RBMR, USEPA 2000a). The largest difference was in the period 1993 to 1999, for which predicted sediment concentrations are now higher than in the initial modeling results reported in the BMR (USEPA, 1999b). After 1999, predicted sediment concentrations are approximately the same as they were previously. Predicted water concentrations were more or less consistent between the BMR and RBMR. However, in the August, 1999 ERA, Tri+ PCB concentrations were used to predict sediment and water concentrations, while in this Responsiveness Summary, total PCB concentrations were used.

In the RBMR (USEPA, 2000a), the FISHRAND model was formally recalibrated using Bayesian updating. Growth rate coefficients, TOC, lipid content, and Kow distributions were all optimized within the constraints of the data. Comparison of model output to historical data showed significantly better agreement at RMs 168 and 154 than previously.

The FISHRAND model results show that largemouth bass concentrations remained essentially the same at river mile 189 except for the 95th percentile, which decreased. Largemouth bass concentrations decreased at the median and 95th percentile for river miles 168 and 189. Spottail shiner concentrations increased slightly at all locations. (Note: largemouth bass is used as the "large" piscivorous fish consumed by the river otter and bald eagle; spottail shiner is the smaller forage fish species considered as a prey item by the other fish-eating wildlife receptors). Benthic invertebrate concentrations increased in proportion to the increase in sediment concentrations; phytoplankton concentrations remained essentially the same.

Toxicity Reference Value Changes

Fish, mallard, bald eagle, mink, and river otter Toxicity Reference Values (TRVs) were revised based on a reevaluation of toxicity studies, as discussed in the following paragraphs.

Changes in Fish TRVs

The laboratory-based TRVs were revised for all fish receptors (i.e., pumpkinseed, spottail shiner, brown bullhead, yellow perch, white perch, largemouth bass, striped bass, shortnose sturgeon. The study by Hansen et al. (1974) was selected for development of the TRV for PCBs, instead of the study by Bengsston (1980). Hansen et al. established a NOAEL for exposure to Aroclor 1254 of 1.9 mg/kg and a LOAEL of 9.3 mg/kg for adult female fish. The values for adult fish determined in this study are more appropriate for comparison to measured and modeled concentrations in adult Hudson River fish than the study by Bengsston (1980), which examined hatchability in minnows exposed to Clophen A50. Because the sheepshead minnow is not in the same taxonomic family as any Hudson River receptors, an interspecific uncertainty factor of 10 is applied to develop TRVs for all fish.

Therefore, on the basis of laboratory toxicity studies:

- The LOAEL TRV for the pumpkinseed, spottail shiner, brown bullhead, yellow perch, white perch, largemouth bass, spottail shiner, striped bass, and shortnose sturgeon is: 0.93 mg PCBs/kg tissue (Table 4-25).

- The NOAEL TRV for the pumpkinseed, spottail shiner, brown bullhead, yellow perch, white perch, largemouth bass, striped bass, and shortnose sturgeon is: 0.19 mg/kg PCBs/kg tissue (Table 4-25).

The field-based TRVs for the pumpkinseed, spottail shiner, and largemouth bass were revised from the August 1999 ERA. For the pumpkinseed and largemouth mouth, the field studies by Adams et al. (1989, 1990, 1992) on the redbreast sunfish, a species in the same family as the pumpkinseed and largemouth bass, were retained as the studies to establish TRVs. However, the growth endpoint, rather than the reduced fecundity endpoint initially selected, was used to establish a TRV. The NOAEL for growth was reported as being significantly different from a downstream location. Growth is a relevant endpoint, and the NOAEL for growth, 0.3

mg/kg, is used in this assessment. The sunfish (*Lepomis auritus*) in the studies were exposed to PCBs and mercury in the field. However, because other contaminants (e.g., mercury) were measured and reported in these fish and may have been contributing to observed effects, these studies are used to develop a NOAEL TRV, but not a LOAEL TRV, for the pumpkinseed and largemouth bass. An interspecies uncertainty factor is not applied since these three species are all in the same family (Centrachidae). Because the experimental study measured the actual concentration in fish tissue, rather than estimating the dose on the basis of the concentration in external media (e.g., food, water, or sediment, or injected dose), a subchronic-to-chronic uncertainty factor is not applied.

On the basis of the field studies:

- The NOAEL TRV for the pumpkinseed and largemouth bass is: 0.3 mg PCBs/kg tissue (Table 4-25).

The previous NOAEL TRV for the pumpkinseed and largemouth bass was 0.5 mg PCBs/kg tissue based upon the fecundity endpoint in Adam et al. (1992).

In the August 1999 ERA, no field-based TRV was selected for the spottail shiner. However, upon re-examination, the study by USACE (1988) using fathead minnow is considered to be a field-related study, rather than a laboratory study, because the sediments to which the fathead minnow were exposed were field-collected sediments (instead of spiked sediments). This study was selected for development of a field-based TRV for the spottail shiner, a species in the same family as the fathead minnow.

On the basis of the field study:

- The final NOAEL TRV for the spottail shiner is: 5.25 mg PCBs/kg wet wt tissue (Table 4-25).

The field-based TRV was selected for use, rather than the laboratory-based TRVs used in the August 1999 ERA.

Changes in Avian TRVs

The total (Tri+) PCB daily dose TRV in the diet was revised for the mallard duck, as were the total (Tri+) PCB and TEQ concentrations in bald eagle eggs. These changes are discussed below.

<u>Mallard Duck</u>

The development of TRVs for exposure of mallards to PCBs was re-assessed with consideration of two additional studies that were not identified in the literature studies that were conducted for the August 1999 ERA. A total of five studies were identified that examined the effects of PCBs on mallards (Hill et al. 1975, Riseborough and Anderson 1975, Custer and Heinz 1980, Heath et al. 1972, and Haseltine and Prouty 1980).

The study by Hill et al. (1975) is not selected for development of TRVs because it examined mortality as an endpoint, which is not expected to be as sensitive an endpoint as growth and reproduction. The studies by Riseborough and Anderson (1975), Custer and Heinz (1980), and Heath et al. (1972) found no effects on various reproductive endpoints based on exposure to a single dose (40 ppm, 25 ppm, and 25 ppm in diet, respectively). Haseltine and Prouty (1980) observed no adverse effects on reproductive endpoints after a 12-week exposure to 150 ppm Aroclor 1242 in food, but did observe significantly reduced weight gain in adults. Therefore, the study by Haseltine and Prouty (1980) is selected as the most appropriate study, since it is a reports a dose response study that reports a LOAEL on an ecologically relevant endpoint from which a NOAEL can be estimated. Because only a single dose was tested, a LOAEL-to-NOAEL uncertainty factor of ten is applied to estimate a NOAEL from this study. Because the study was conducted over a 12 week period, a sub-chronic to chronic uncertainty factor is not applied.

Based on the results of Haseltine and Prouty (1980) on growth:

- The LOAEL TRV for mallard (growth effects) is: 16 mg/kg/day (Table 4-26).
- The NOAEL TRV for the mallard (growth effects) is: 1.6 mg/kg/day (Table 4-26).

Previously, a LOAEL of 2.6 mg/kg/day and a NOAEL of 0.26 mg/kg/day were used based on Custer and Heinz (1980):

<u>Bald Eagle</u>

Upon reexamination, EPA agree with some commenters that the data collected by Wiemeyer et al. (1993) does not support the development of the previous NOAEL TRV of 3.0 mg/kg for bald eagle egg concentrations. However, USEPA does not agree with GE's assertion that because mean 5-year production was not significantly reduced for the residue *interval* ranging from 5.6 to <13 mg PCBs/kg, that a NOAEL of 13 mg/kg is appropriate. It would be more appropriate to take the average value of the data in the 5.6 to <13 mg/kg interval as a measure of the average concentration for which production was not significantly impacted, as compared to higher concentrations. However, those data are not reported in this paper. As an alternative, the average PCB concentration in eggs from successful eggs (5.5 mg/kg), which was shown to be significantly lower than the concentration measured in unsuccessful nests (8.7 mg/kg) (Wiemeyer et al. 1993, p. 224), is selected as the NOAEL-TRV for bald eagles.

Based on the study by Wiemeyer et al. (1993):

- The NOAEL TRV for PCBs in bald eagle eggs is: 5.5 mg PCBs/kg egg (Table 4-26).

Based on the same study, the previous NOAEL TRV for the bald eagle was 3.0 mg/kg egg.

To determine TEQ-based TRVs PCBs for bald eagle eggs, a study by Elliott et al. (1996) that reports data for TEQ in the yolk sac of the bald eagle egg was used. This study reports a concentration of TEQs of 210 ng/kg wet weight in eggs for the Powell River, a contaminated site with a concentration that is slightly less than another nearby contaminated site, East Vancouver Island. Based on Figure 4 in Elliott et al. (1996) the concentration of TEQs in the East Vancouver Island site is estimated as 13,000 ng TEQs/kg lipid. Using the ratio between weight wet and lipid at the Powell River site, the weight wet concentration at East Vancouver Island wet weight concentration is approximately 217 ng/kg. Since no significant difference was observed between the average hatching rate of the eggs collected from these two contaminated sites and the reference sites, the average concentration in eggs from the contaminated sites (214 ng/kg ww) is selected as the NOAEL for this study.

- The field based NOAEL TRV for TEQs in bald eagle eggs is: 0.214 μg/kg egg (Table 4-26).

Based on Powell et al. (1996), the previous laboratory-based NOAEL and LOAEL TRVs for the bald eagle were 0.02 μg/kg egg and 0.01 μg/kg egg, respectively.

Changes in Mammalian TRVs

USEPA acknowledges that for TEQ-based PCBs in the diet, a LOAEL should not be established from the Tillitt et al. (1996) field study for the mink and river otter. In keeping with accepted scientific practice, only NOAEL TRVs are developed from field studies in the ERA because other contaminants or stressors may be contributing to observed effects. The revised risk estimates remove this comparison (see Table 4-27).

Risk Characterization Changes

In general, conclusions drawn from the results of the August, 1999, Baseline Ecological Risk Assessment Report still hold. In some cases, the year in which predicted toxicity quotients fall from above one to below one will have increased or decreased slightly.

Tables from Chapter 3, 4, and 5 have been revised to reflect the changes due to modeling and TRVs, but these revisions do not affect the general text or the overall conclusions of the August 1999 ERA. Conclusions drawn from the results of the August, 1999, Baseline Ecological Risk Assessment Report have not changed significantly. Specific changes to risk characterization tables are as follows:

Tables 5-1 to 5-8: These tables have not changed.

Table 5-9: Predicted sediment concentrations (Tri+) were adjusted to reflect total PCB concentrations. None of the guidelines changed. Conclusions are unchanged but risks increase at river mile 189, decrease at river miles 168 and 154.

Table 5-10: . The NYSDEC wildlife bioaccumulation criterion comparison was removed, since it is now the same as the EPA criterion (1.2×10^{-4} μg/L).

Table 5-11: Predicted water concentrations (Tri+) were adjusted to reflect total PCB concentrations. The NYSDEC wildlife bioaccumulation criterion comparison was removed, since it is now the same as the EPA criterion (1.2×10^{-4} μg/L). Conclusions mostly unchanged but risks have increased slightly at all locations. Benthic aquatic life: exceedance at all locations for all years except river mile 154 starting in 2012. Previously, river mile 189 exceeded until

2010, river mile 168 until 2016, and river mile 154 until 2007. Based on NYSDEC/USEPA wildlife criterion comparisons, conclusions are unchanged.

Table 5-12: Pumpkinseed field-based NOAEL changed from 0.5 to 0.3 mg/kg (based on Adams et al., 1992; same study but different value). Spottail shiner lab-based NOAEL and LOAEL changed to field-based NOAEL only, 5.25 mg/kg, based on USACE study. Measured concentrations remain unchanged. Conclusions for pumpkinseed remain unchanged, but predicted toxicity quotients increased slightly. Conclusions for spottail shiner are the same for river mile 189, but now predicted toxicity quotients are above one at river mile 168 as well, and above one for the 95% UCL at river mile 137.2. Previously, all locations had predicted toxicity quotients below one at all locations except river mile 189. Comparisons to a laboratory-based LOAEL for the spottail shiner are no longer appropriate as the selected TRV is a field-based NOAEL.

Table 5-13: Pumpkinseed field-based NOAEL changed from 0.5 to 0.3 mg/kg (based on Adams et al., 1992; same study but different value). Predicted pumpkinseed body burdens increased slightly or remained the same at all river miles. Overall conclusions changed slightly: predicted toxicity quotients are exceeded throughout the modeling period at river miles 189 and 168, and at river mile 154 for both the median and 95th percentiles, but only until 2018 for the 25th percentile. Previously, predicted toxicity quotients were exceeded at all locations throughout the modeling period except for the 25th percentile at river mile 154, where toxicity quotients were exceeded until 2005.

Table 5-14: Spottail shiner laboratory-based TRVs changed to a single field-based NOAEL based on the USACE study (previous lab-based NOAEL was 15 mg/kg while field-based NOAEL is 5.25 mg/kg). Predicted spottail shiner body burdens increased slightly or remained the same at all river miles. Conclusions did not change for river miles 168 and 154 (toxicity quotients below one for these locations for all years except for 1993 for the 95th percentile at river mile 168), but changed slightly for river mile 189. Previously, predicted toxicity quotients exceeded one for the 95th percentile for 1993 – 1996, but all other predicted toxicity quotients were below one. The revised risk estimates show that toxicity quotients exceed one for the 25th percentile until 1999, the median until 2000, and the 95th percentile until 2009.

Table 5-15: This table is obsolete, as no LOAEL is derived from the field-based study.

Table 5-16: TRVs have not changed. Predicted pumpkinseed concentrations increased slightly or remained the same at all river miles. Conclusions have changed slightly: Previously at river mile 189, predicted toxicity quotients fell below one for the 25th percentile for the duration of the modeling period, and were above one but below ten until 1998 on a median basis and above one but below ten until 2010 for the 95th percentile. Revised risk estimates show that predicted toxicity quotients are above one but below ten for the 25th percentile until 2000, above one but below ten for the median until 2005, and above one but below ten for the 95th percentile until 2014. At river mile 168, previous predicted toxicity quotients exceeded one but fell below ten until 1999 for the median, and until 2013 for the 95th percentile. Revised risk estimates show that predicted toxicity quotients exceed one but fall below ten for the 25th percentile until 1995, exceed one but fall below ten for the median until 1998, and exceed one but fall below ten for the 95th percentile until 2003. At river mile 154, previous predicted toxicity quotients fell well below one for all years and percentiles. Revised risk estimates show that the 95th percentile exceeds one slightly for 1993 – 1995, but falls below one for all remaining comparisons.

Table 5-17: TRVs have not changed. Predicted pumpkinseed concentrations increased slightly or remained the same at all river miles. Conclusions have changed slightly: Previously at river mile 189, predicted toxicity quotients fell below one for the 25th percentile for the duration of the modeling period, and were above one but below ten for 1993 on a median basis and above one but below ten until 2000 for the 95th percentile. Revised risk estimates show that predicted toxicity quotients are above one but below ten for the 25th percentile until 1995, above one but below ten for the median until 1995, and above one but below ten for the 95th percentile until 2000. At river mile 168, previous predicted toxicity quotients exceeded one but fell below ten until 1995 for the median, and until 2001 for the 95th percentile. Revised risk estimates show that predicted toxicity quotients fall below one for the 25th percentile and median for the duration of the modeling period, and exceed one but fall below ten for the 95th percentile until 1996. At river mile 154, previous predicted toxicity quotients fell well below one for all years and percentiles and the revised risk estimates show the same results.

Tables 5-18 and 5-19: TRVs have not changed. Predicted spottail shiner concentrations increased slightly or remained the same at all river miles. Conclusions are unchanged (predicted toxicity quotients below one for all locations and years).

Table 5-20: Laboratory-based TRVs for brown bullhead have changed: original NOAEL was 1.5 mg/kg based on Bengsston (1980) and revised NOAEL is 0.19 mg/kg based on Hansen et al. (1974). Predicted body burdens remained the same at river mile 189, and decreased slightly at river miles 168 and 154. Overall conclusions have changed slightly: predicted toxicity quotients have increased (by factors of two to three) at all locations resulting in toxicity quotients above ten at river mile 189 and above one at river miles 168 and 154 for all years.

Table 5-21: Laboratory-based TRVs for brown bullhead have changed: The original LOAEL was 17 based on Bengsston (1980) and the revised LOAEL is 0.93 based on Hansen et al. (1974). Predicted body burdens remained the same at river mile 189, and decreased slightly at river miles 168 and 154. Previously, predicted toxicity quotients exceeded one but fell below ten for the median and 95th percentile at river mile 189 until 1998 and 2015, respectively, and until 2003 for the 95th percentile at river mile 168. All other predicted toxicity quotients fell below one. The revised risk estimates show that predicted toxicity quotients exceed one but fall below ten for all years at river miles 189 and 168. At river mile 154, predicted toxicity quotients exceed one but fall below ten for the 25th percentile until 2007, until 2009 for the median, and until 2017 for the 95th percentile.

Tables 5-22 and 5-23: The laboratory-based NOAEL and LOAEL for brown bullhead have not changed. Predicted concentrations for brown bullhead have remained the same or decreased slightly at all locations. Overall conclusions have not changed: predicted toxicity quotients fall below one for all locations and years.

Table 5-24: Observed largemouth bass and brown bullhead concentrations have not changed. However, the field-based NOAEL for largemouth bass has decreased to 0.3 mg/kg from 0.5 mg/kg based on the Adams study. For brown bullhead, the laboratory-derived NOAEL decreased to 0.19 mg/kg based on Hansen et al. (1974) from 1.5 mg/kg based on Bengsston (1980). The laboratory-derived LOAEL decreased to 0.93 mg/kg based on the Hansen study from 17 mg/kg based on the Bengsston study. Overall conclusions remain the same (predicted toxicity quotients exceed ten at all locations for both species based on the NOAEL comparison (and exceed 100 for largemouth bass at river mile 189), and exceed one for all locations based on the LOAEL comparison.

TEQ-based results have not changed.

Table 5-25: Observed white perch and yellow perch concentrations have not changed. The TRV for white perch has not changed. For the yellow perch, the laboratory-derived NOAEL decreased to 0.19 mg/kg based on the Hansen study from 1.5 mg/kg based on the Bengsston study. The laboratory-derived LOAEL decreased to 0.93 mg/kg based on the Hansen study from 17 mg/kg based on the Bengsston study. Overall conclusions have not changed, but predicted toxicity quotients for the yellow perch have increased by almost an order of magnitude. White perch results remain unchanged.

TEQ-based results have not changed.

Table 5-26: The TRV for white perch has not changed. Predicted white perch body burdens have increased slightly at river mile 189 for the 25th percentile, and decreased slightly for the median and 95th percentile at river mile 189 and for all percentiles at river miles 168 and 154. Predicted toxicity quotients at river mile 189 exceed one but fall below ten until 2006 for the 25th percentile (previously until 1998), until 2010 for the median (previously until 2013), and for the duration of the modeling period under both. Conclusions have not changed for river miles 168 and 154.

Table 5-27: The laboratory-derived NOAEL decreased to 0.19 mg/kg based on the Hansen study from 1.5 mg/kg based on the Bengsston study for the yellow perch. Predicted concentrations for yellow perch have increased slightly at all locations and for all percentiles except the 95th percentile at river mile 154, where concentrations decreased slightly. The revised risk estimates show predicted toxicity quotients exceed one but fall below ten at all locations and years except for river mile 154, for which the 25th percentile and median predicted toxicity quotients fall below one in 2010 and 2014, respectively. Previously, predicted toxicity quotients exceeded one but fell below ten at river mile 189 until 2003 for the 25th percentile, until 2017 for the median, and for the entire modeling period for the 95th percentile. At river mile 168, previous predicted toxicity quotients exceeded one but fell below ten until 2003 for the 25th percentile, until 2017 for the median, and for the entire modeling period for the 95th percentile. At river mile 154, predicted toxicity quotients exceeded one but fell below ten until 1996 for the 25th percentile, until 2007 for the median, and until 2017 for the 95th percentile.

Table 5-28: The laboratory-derived LOAEL decreased to 0.93 mg/kg based on the Hansen study from 17 mg/kg based on the Bengsston study for the yellow perch. Predicted concentrations for yellow perch have increased slightly at all locations and for all percentiles except the 95th percentile at river mile 154, where concentrations decreased slightly. Previous risk estimates showed that predicted toxicity quotients fell below one for all locations and years except for 1993 – 1997 for the 95th percentile at river mile 189. Revised risk estimates show that predicted toxicity quotients exceed one for all percentiles and years at river mile 189. At river mile 168, predicted toxicity quotients exceed one until 2002 for the 25th percentile, until 2005 for the median, and for the entire modeling period for the 95th percentile. At river mile 154, predicted toxicity quotients exceed one until 1993 for the 25th percentile, until 1997 for the median, and 2005 for the 95th percentile.

Tables 5-29 and 5-30: TRVs for the white perch have not changed. Predicted concentrations have increased slightly for the 25th percentile and the median at all locations, and decreased slightly for the 95th percentile at all locations. Conclusions have not changed.

Table 5-31: TRVs for the yellow perch have not changed. Predicted concentrations for yellow perch have increased slightly at all locations and for all percentiles except the 95th percentile at river mile 154, where concentrations decreased slightly. Revised risk estimates show that predicted toxicity quotients have increased at all locations and for all years. Predicted toxicity quotients exceed one but fall below ten for all years at river mile 189 (previous predicted toxicity quotients exceeded one but fell below ten until 1993 for the 25th percentile, until 1999 for the median, and until 2016 for the 95th percentile). At river mile 168, predicted toxicity quotients exceed one but fall below ten until 2003 for the 25th percentile, until 2007 for the median, and for the duration of the modeling period for the 95th percentile. Previous predicted toxicity quotients at river mile 168 fell below one for the 25th percentile, fell above one but below ten until 1998 for the median, and until 2016 for the 95th percentile. At river mile 154, predicted toxicity quotients exceed one but fall below ten until 1995 for the 25th percentile, until 2000 for the median, and until 2007 for the 95th percentile. Previous predicted toxicity quotients at river mile 154 fell below one for the 25th percentile, fell above one but below ten until 1996 for the median, and until 2013 for the 95th percentile.

Table 5-32: TRVs for the yellow perch have not changed. Predicted concentrations for yellow perch have increased slightly at all locations and for all percentiles except the 95th percentile at river mile 154, where concentrations decreased slightly. Revised risk estimates show that predicted toxicity quotients have increased at all locations and for all years. Predicted toxicity quotients at river mile 189 exceed one but fall below ten until 2001 for the 25th percentile, until 2006 for the median, and for the duration of the modeling period for the 95th percentile (previous predicted toxicity quotients fell below one for the 25th percentile, exceeded one but fell below ten until 1994 for the median, and until 2005 for the 95th percentile). At river mile 168, predicted toxicity quotients exceed one but fall below ten until 1996 for the 25th percentile, until 2000 for the median, and until 2008 for the 95th percentile. Previous predicted toxicity quotients at river mile 168 fell below one for the 25th percentile, fell above one but below ten until 1994 for the median, and until 2003 for the 95th percentile. At river mile 154, predicted toxicity quotients fall below one for the 25th percentile and the median, and fall above one but below ten until 1999 for the 95th percentile. Previous predicted toxicity quotients at river mile 154 fell below one for the 25th percentile and the median, and fell above one but below ten until 2001 for the 95th percentile.

Table 5-33: The field-based NOAEL for largemouth bass has decreased to 0.3 mg/kg from 0.5 mg/kg based on the Adams study. Predicted largemouth bass concentrations have increased slightly at river mile 189, and decreased slightly at river miles 168 and 154. The conclusions have not changed: predicted toxicity quotients exceed one and sometimes ten for all locations, percentiles, and river miles.

Table 5-34: TRVs on a TEQ basis for largemouth bass have not changed. Predicted lipid-normalized body burdens have increased slightly. Revised risk estimates show that predicted toxicity quotients exceed one at river mile 189 for the duration of the modeling period (previous predicted toxicity quotients exceeded one but fell below ten until 1997 for the 25th percentile, until 2010 for the median, and for the duration of the modeling period for the 95th percentile). At river mile 168, predicted toxicity quotients exceed one but fall below ten until 2010 for the 25th percentile, until 2012 for the median, and for the duration of the modeling period for the 95th percentile. Previous predicted toxicity quotients at river mile 168 fell above one but below ten until 1996 for the 25th percentile, until 2009 for the median, and for the duration of the modeling period for the 95th percentile. At river mile 154, predicted toxicity quotients fall above one but below ten until 2001 for the 25th percentile, until 2006 for the median, and until 2006 for the 95th percentile. Previous predicted toxicity quotients at river mile 154 fell below one for the 25th percentile, and fell above one but below ten until 1997 for the median, and until 2010 for the 95th percentile.

Table 5-35: TRVs on a TEQ basis for largemouth bass have not changed. Predicted lipid-normalized body burdens have increased slightly at all locations. Revised risk estimates show that predicted toxicity quotients exceed one but fall below ten at river mile 189 until 2005 for the 25th percentile, until 2013 for the median, and for the duration of the modeling period for the 95th percentile (previous predicted toxicity quotients fell below one for the 25th percentile, fell above one but below ten until 1998 for the median, and for the duration of the modeling period for the 95th percentile). At river mile 168, predicted toxicity quotients exceed one but fall below ten until 2001 for the 25th percentile, until 2002 for the median, until 2006 for the 95th percentile. Previous predicted toxicity quotients at river mile 168 fell below one for the 25th percentile, fell above one but below ten until 1999 for the median, and for the duration of the modeling period for the 95th percentile. At river mile 154, predicted toxicity quotients fall above one but below ten until 1995 for the 25th percentile, until 1996 for the median, and until 1998 for the 95th percentile. Previous predicted toxicity quotients at river mile 154 fell below one for the 25th percentile and the median, and fell above one but below ten until 1999 for the 95th percentile.

Table 5-36: This table has not changed.

Table 5-37: This table has not changed.

Tables 5-38 and 5-39: TRVs for the tree swallow have not changed. Predicted benthic invertebrate concentrations (the primary dietary item for the tree swallow) have decreased by approximately 10% at all locations. Predicted toxicity quotients have also decreased by approximately 10%, but the conclusions have not changed.

Table 5-40: This table has not changed.

Tables 5-41 and 5-42: TRVs for the tree swallow based on egg concentration have not changed. Predicted benthic invertebrate concentrations (which are used to predict expected egg concentrations) have decreased by approximately 10% at all locations. Predicted toxicity quotients have also decreased by approximately 10%, but the conclusions have not changed.

Table 5-43: The dietary dose TRVs for the mallard have changed. The laboratory-based NOAEL is 1.6 mg/kg/day, and the laboratory based LOAEL is 16 mg/kg/day based on Haseltine and Prouty (1980). The original NOAEL was 0.26 mg/kg/day and LOAEL 2.6 mg/kg/day based on Custer. Predicted toxicity quotients based on dietary dose have decreased slightly but the conclusions have not changed.

Table 5-44: The dietary dose TRVs for the mallard have changed. The laboratory-based NOAEL is 1.6 mg/kg/day, and the laboratory based LOAEL is 16 mg/kg/day based on Haseltine and Prouty. The original NOAEL was 0.26 mg/kg/day and LOAEL 2.6 mg/kg/day based on Custer. Predicted phytoplankton concentrations and benthic invertebrate concentrations have decreased slightly at all locations. Predicted toxicity quotients based on dietary dose have decreased but the conclusions have not changed at river mile 189. At river mil168, NOAEL-based comparisons previously exceeded one but fell below ten for all years. Revised risk estimates show that NOAEL-based comparisons now only exceed one until 1999. At river mile 154, previous predicted toxicity quotients exceeded one but fell below ten until 2001, while the revised risk estimates show predicted toxicity quotients less than one for all years.

Table 5-45: The egg-based TRVs have not changed. Predicted benthic invertebrate concentrations have decreased slightly at all locations. Predicted toxicity quotients have decreased by approximately 5 – 10% but the conclusions have not changed.

Table 5-46: This table has not changed except that the LOAEL-based comparisons for egg concentrations have been removed. The study upon which that TRV was based did not support development of a LOAEL.

Table 5-47: The TRVs have not changed. Predicted phytoplankton and benthic invertebrate concentrations have decreased, thus, predicted toxicity quotients have decreased. Conclusions have not changed except at river mile 154, where LOAEL-based predicted toxicity quotients fall below one in 2008 whereas previously predicted toxicity quotients at river mile 154 exceeded one for all years.

Table 5-48: The TRVs have not changed. Predicted benthic invertebrate concentrations have decreased slightly, but conclusions have not changed.

Tables 5-49 and 5-50: These tables have not changed.

Table 5-51: The field-based NOAEL for egg-based concentrations for the bald eagle has changed from 3.0 mg/kg wet weight to 5.5 mg/kg wet weight. The LOAEL-based comparisons have been removed (it is not appropriate to develop a field-based LOAEL). Conclusions have not changed although predicted toxicity quotients have decreased by less than a factor of two.

Table 5-52: TRVs have not changed. Dietary doses have decreased by almost a factor of three, but the conclusions of risk remain unchanged.

Table 5-53: TRVs have not changed. Dietary doses have decreased by 30 - 50%, but conclusions of risk remain unchanged except for LOAEL based comparisons at river mile 154.

Predicted toxicity quotients previously exceeded one for all years, but the revised risk estimates show that predicted toxicity quotients fall below one in 2002 at this location.

Table 5-54: TRVs have not changed. Dietary doses have decreased by almost a factor of three, but the conclusions of risk remain unchanged, except at river mile 154. Previous LOAEL-based predicted toxicity quotients exceeded one for the entire modeling period, but revised risk estimates show that predicted toxicity quotients fall below one in 2015 at this location.

Table 5-55: TRVs have not changed. Predicted forage fish concentrations have decreased, thus predicted toxicity quotients have decreased slightly but conclusions of risk remain unchanged.

Table 5-56: TRVs have not changed. Predicted forage fish concentrations have decreased, thus predicted toxicity quotients have decreased slightly but conclusions of risk remain unchanged.

Table 5-57: The field-based NOAEL for egg-based concentrations for the bald eagle has changed from 3.0 mg/kg wet weight to 5.5 mg/kg wet weight. The LOAEL-based comparisons have been removed (it is not appropriate to develop a field-based LOAEL). Conclusions have not changed although predicted toxicity quotients have decreased by approximately a factor of two.

Tables 5-58 and 5-59: These tables have not changed.

Table 5-60: The egg-based TRV for bald eagle has changed. The revised field-based NOAEL is 0.000214 mg/kg. It is not appropriate to develop a LOAEL from a field-based study, thus, these comparisons have been removed. Predicted toxicity quotients have decreased over a factor of 10, but conclusions of risk remain unchanged.

Table 5-61: TRVs have not changed. Predicted forage fish concentrations have decreased, thus predicted toxicity quotients have decreased slightly but conclusions of risk remain unchanged.

Table 5-62: TRVs have not changed. Dietary doses have decreased by 30 - 50%, but conclusions of risk remain unchanged except for LOAEL based comparisons at river mile 154. Predicted toxicity quotients previously exceeded one for all years, but the revised risk estimates show that predicted toxicity quotients fall below one in 2004 at this location.

Table 5-63: TRVs have not changed. Dietary doses have decreased by almost a factor of three, but the conclusions of risk remain unchanged at all locations.

Tables 5-64 and 5-65: TRVs have not changed, and predicted forage fish concentrations have decreased slightly. Predicted toxicity quotients have decreased, but conclusions of risk remain unchanged for all locations and years.

Table 5-66: The egg-based TRV for bald eagle has changed. The revised field-based NOAEL is 0.000214 mg/kg. It is not appropriate to develop a LOAEL from a field-based study, thus, these comparisons have been removed. Predicted toxicity quotients have decreased over a factor of 10, but conclusions of risk remain unchanged.

Table 5-67: This table remains unchanged.

Table 5-68: This table remains unchanged.

Table 5-69: TRVs have not changed. Dietary doses have increased slightly at all locations, and predicted toxicity quotients are within 10 – 30% of previous predictions. Conclusions remain unchanged.

Table 5-70: This table remains unchanged.

Table 5-71: TRVs have not changed. Dietary doses have increased slightly at all locations, and predicted toxicity quotients are within 10 – 30% of previous predictions. Conclusions remain unchanged.

Table 5-72: This table remains unchanged.

Table 5-73: TRVs have not changed. Dietary doses have increased slightly at all locations, and predicted toxicity quotients are within 10 – 30% of previous predictions. Conclusions remain unchanged.

Table 5-74: This table remains unchanged.

Table 5-75: TRVs have not changed. Dietary doses have increased slightly at all locations, and predicted toxicity quotients are within 10 – 30% of previous predictions. Conclusions remain unchanged.

Tables 5-76 through 5-78: These tables remain unchanged.

Table 5-79: TRVs have not changed. Forage fish concentrations have increased slightly at all locations, and predicted toxicity quotients are within 10 – 30% of previous predictions. Conclusions remain unchanged.

Table 5-80: TRVs have not changed. Piscivorous (large) fish concentrations have decreased slightly at all locations. Conclusions remain the same except at river mile 154, where LOAEL-based comparisons exceeded one for all years, while revised risk estimates show that these predicted toxicity quotients fall below one in 2001.

Tables 5-81 and 5-82: These tables remain unchanged, except for the removal of the LOAEL-based comparisons. It is not appropriate to develop a LOAEL from a field-based study, thus, only NOAEL-based comparisons are provided.

Table 5-83: TRVs have not changed, except for the removal of the LOAEL-based comparisons. Forage fish concentrations have increased slightly at all locations, and predicted toxicity quotients are within 10 – 30% of previous predictions. Conclusions remain unchanged.

Table 5-84: TRVs have not changed, except for the removal of the LOAEL-based comparisons. Piscivorous (large) fish concentrations have decreased slightly at all locations. Conclusions remain unchanged.

Table 5-85: This table remains unchanged.

Summary

Several revisions were made to the HUDTOX model, FISHRAND model, and toxicity reference values that required recalculation of risks to receptors evaluated in the August 1999 ERA. None of the changes resulted in any significant changes to the conclusions reached in the original risk assessment. Years for which predicted toxicity quotients fall above or below one, have changed slightly (in both directions) based on the recalculated risks.

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

- Benthic invertebrate communities in the Hudson River are an important food source for many species of fish. PCBs in the Hudson River may adversely affect benthic invertebrate populations.

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

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

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

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

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

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

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

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

References

Adams, S.M., W.D. Crumby, M.S. Greeley, Jr., M.G. Ryon, and E.M. Schilling. 1992. Relationships between physiological and fish population responses in a contaminated stream. *Environmental Toxciology and Chemistry*. 11:1549-1557.

Adams, S.M., K.L. Shepard, M.S. Greeley Jr., B.D. Jimenez, M.G. Ryon, L.R. Shugart, and J.F. McCarthy. 1989. The use of bioindicators for assessing the effects of pollutant stress on fish. *Marine Environmental Research*. 28:459-464.

Adams, S.M., L.R. Shugart, G.R. Southworth and D.E. Hinton. 1990. Application of bioindicators in assessing the health of fish populations experiencing contaminant stress. In: J.F. McCarthy and L.R. Shugart, eds., Biomarkers of Environmental Contamination. Lewis Publishers, Boca Raton, FL. Pp. 333-353.

Bengtsson, B.E. 1980. Long-term effects of PCB (Clophen A50) on growth, reproduction and swimming performance in the minnow, *Phoxinus phoxinus*. *Water Research*, Vol. 1, pp. 681-687.

Custer, T.W., and G.H. Heinz. 1980. Reproductive success and nest attentiveness of mallard ducks fed Aroclor 1254. *Environmental Pollution* (Series A). 21:313-318.

Elliott, J.E., R.J. Norstrom, A. Lorenzen, L.E. Hart, H. Philibert, S.W. Kennedy, J.J. Stegeman, G.D. Bellward, and K.M. Cheng. 1996. Biological effects of polychlorinated bibenzo-p-dioxins, dibenzofurans, and biphenyls in bald eagle (*Haliaeetus leucocephalus*) chicks. *Environmental Toxicology and Chemistry*. Vol. 1(5):782-793.

Elonen, G.E., R.L. Spehar, G.W. Holcombe, R.D. Johnson, J.D. Fernandex, R.J. Erickson, J.E. Tietge, and P.M. Cook. 1998. Comparative toxicity of 2,3,7,8-tetrachlorodibenzo-p-dioxin to seven freshwater fish species during early life-stage development. *Environmental Toxicology and Chemistry*. 17:3, pp 472-483.

Hansen, D.J., S.C. Schimmel, and E. Matthews. 1974. Avoidance of Aroclor 1254 by shrimp and fishes, *Bull. of Environ. Contam. and Toxicol.* 12(2):253-293.

Haseltine, S.D., and R.M. Prouty. 1980. Aroclor 1242 and reproductive success of adult mallards (*Anas platyrhynchos*). *Environmental Research*. 23:29-34.

Heath, R.G., J.W. Spann, J.F. Kreitzer, and C. Vance. 1972. Effects of polychlorinated biphenyls on birds. pp. 475-485. In: The XVth International Ornithological Congress. K.H. Voous (ed). Leiden, E.J. Brill, The Hague, Netherlands.

Heinz, G.H., S.D. Haseltine, R.J. Hall, and A.J. Krynitsky. 1980. Organochlorine and mercury residues in snakes from Pilot and Spider Islands, Lake Michigan – 1978. *Bull. Environm. Contam. Toxicol.* 25: 738-743.

Hill, E., R.G Heath, J.W. Spann, and J.D. Williams. 1975. Lethal Dietary Toxicities of Environmental Pollutants to Birds. U.S. Fish and Wildlife Special Scientific Report – Wildlife No. 191, Washington, DC.

Powell, D.C., R.J. Aulerich, J.C. Meadows, D.E. Tillett, J.P. Giesy, K.L. Stromborg, S.J. Bursian. 1996. Effects of 3,3',4,4',5-pentachlorobiphenyl (PCB 126) and 2,3,7,8-tetrachlorodibenzo-p-dioxin (TCDD) injected into the yolks of chicken (*Gallus domesticus*) eggs prior to incubation. *Arch. Environ. Contam. Toxicol.* 31:404-409.

Riseborough, R.W. and D.W. Anderson. 1975. Some effects of DDE and PCB on mallards and their eggs. J. Wild. Manage. 39: 508-513.

Tillett, D.E. 1997. Determination of 2,3,7,8- tetrachloro dibenzo-p-dioxin equivalents (TCDD-eq) with the PLHC-1 bioassay: pumpkinseed collected from the Hudson River, Nov 11, 1997. Final Lab Report Number FY-98-30-01, Report to USFWS and NOAA, 2 pp.

Tillitt, D.E., Gale, R.W., J.C. Meadows, J.L. Zajicek, P.H. Peterman, S.N. Heaton, P.D. Jones, S.J. Bursian, T.J. Kubiak, J.P. Giesy, and R.J. Aulerich. 1996. Dietary exposure of mink to carp from Saginaw Bay. 3. Characterization of dietary exposure to planar halogenated hydrocarbons, dioxin equivalents, and biomagnification. *Environmental Science & Technology.* 30(1):283-291.

United States Army Corps of Engineers (ACE) Waterways Experiment Station. 1988. Environmental Effects of Dredging Technical Notes. No. EEDP-01-13. Relationship Between PCB Tissue Residues and Reproductive Success of Fathead Minnows. April 1988.

United States Environmental Protection Agency (USEPA). 1999a. Further Site Characterization and Analysis, Volume 2D- Baseline Modeling Report Hudson River PCBs Reassessment RI/FS. Prepared for USEPA Region 2 and US Army Corps of Engineers, Kansas City District. Prepared by Limno-Tech, Inc. (LTI), Menzie-Cura & Associates, Inc. (MCA), and Tetra-Tech, Inc. May, 1999.

United States Environmental Protection Agency (USEPA). 1999b. Further Site Characterization and Analysis, Volume 2E – Baseline Ecological Risk Assessment, Hudson River PCBs Reassessment RI/FS. Prepared for USEPA, Region 2 and the US Army Corps of Engineers, Kansas City District. Prepared by TAMS/MCA. August, 1999.

United States Environmental Protection Agency (USEPA). 2000. Further Site Characterization and Analysis, Volume 2D- Revised Baseline Modeling Report Hudson River PCBs Reassessment RI/FS. Prepared for USEPA Region 2 and US Army Corps of Engineers, Kansas City District. Prepared by Limno-Tech, Inc. (LTI), Menzie-Cura & Associates, Inc. (MCA), and Tetra-Tech, Inc. January 2000.

Wiemeyer, S.N., C.M. Bunck, and C.J. Stafford. 1993. Environmental Contaminants in bald eagle eggs – 1980-84- and further interpretations of relationships to productivity and shell thickness. *Archives of Environmental Contamination and Toxicology.* 24:213-227.

TABLE 3-10: SUMMARY OF TRI+ WHOLE WATER CONCENTRATIONS FROM THE HUDTOX MODEL AND TEQ-BASED PREDICTIONS FOR 1993 - 2018
REVISED

	Tri+ Average PCB Results			Tri+ 95% UCL Results			Average Avian TEF			95% Avian TEF			Average Mammalian TEF			95% UCL Mammalian TEF		
Year	189 Whole Water Conc mg/l	168 Whole Water Conc mg/l	154 Whole Water Conc mg/l	189 Whole Water Conc mg/l	168 Whole Water Conc mg/l	154 Whole Water Conc mg/l	189 Whole Water Conc mg/l	168 Whole Water Conc mg/l	154 Whole Water Conc mg/l	189 Whole Water Conc mg/l	168 Whole Water Conc mg/l	154 Whole Water Conc mg/l	189 Whole Water Conc mg/l	168 Whole Water Conc mg/l	154 Whole Water Conc mg/l	189 Whole Water Conc mg/l	168 Whole Water Conc mg/l	154 Whole Water Conc mg/l
1993	1.6E-04	8.9E-05	6.3E-05				1.3E-06	7.2E-07	5.2E-07				1.0E-06	5.6E-07	4.0E-07			
1994	1.4E-04	7.7E-05	5.5E-05				1.1E-06	6.3E-07	4.5E-07				8.7E-07	4.9E-07	3.5E-07			
1995	1.4E-04	7.4E-05	5.3E-05				1.1E-06	6.1E-07	4.3E-07				8.8E-07	4.7E-07	3.3E-07			
1996	7.5E-05	5.2E-05	4.1E-05				6.1E-07	4.2E-07	3.3E-07				4.7E-07	3.3E-07	2.6E-07			
1997	8.4E-05	5.3E-05	4.3E-05				6.8E-07	4.4E-07	3.5E-07				5.3E-07	3.4E-07	2.7E-07			
1998	9.8E-05	4.7E-05	3.6E-05				8.0E-07	3.9E-07	2.9E-07				6.2E-07	3.0E-07	2.3E-07			
1999	8.8E-05	4.5E-05	3.5E-05				7.2E-07	3.7E-07	2.8E-07				5.6E-07	2.8E-07	2.2E-07			
2000	5.6E-05	3.6E-05	2.8E-05				4.5E-07	3.0E-07	2.3E-07				3.5E-07	2.3E-07	1.8E-07			
2001	6.2E-05	3.7E-05	2.9E-05				5.0E-07	3.0E-07	2.3E-07				3.9E-07	2.4E-07	1.8E-07			
2002	6.5E-05	3.1E-05	2.4E-05				5.3E-07	2.5E-07	1.9E-07				4.1E-07	2.0E-07	1.5E-07			
2003	6.1E-05	3.0E-05	2.2E-05				5.0E-07	2.4E-07	1.8E-07				3.8E-07	1.9E-07	1.4E-07			
2004	6.8E-05	3.0E-05	2.2E-05				5.5E-07	2.4E-07	1.8E-07				4.3E-07	1.9E-07	1.4E-07			
2005	5.3E-05	2.7E-05	1.9E-05				4.3E-07	2.2E-07	1.6E-07				3.3E-07	1.7E-07	1.2E-07			
2006	4.8E-05	2.5E-05	1.8E-05				3.9E-07	2.0E-07	1.5E-07				3.0E-07	1.6E-07	1.1E-07			
2007	5.1E-05	2.4E-05	1.7E-05				4.2E-07	2.0E-07	1.4E-07				3.2E-07	1.5E-07	1.1E-07			
2008	5.8E-05	2.2E-05	1.5E-05				4.8E-07	1.8E-07	1.2E-07				3.7E-07	1.4E-07	9.5E-08			
2009	5.4E-05	2.2E-05	1.5E-05				4.4E-07	1.8E-07	1.2E-07				3.4E-07	1.4E-07	9.3E-08			
2010	4.3E-05	2.1E-05	1.5E-05				3.5E-07	1.7E-07	1.2E-07				2.7E-07	1.4E-07	9.6E-08			
2011	4.4E-05	1.9E-05	1.3E-05				3.6E-07	1.6E-07	1.1E-07				2.8E-07	1.2E-07	8.5E-08			
2012	4.3E-05	1.9E-05	1.2E-05				3.5E-07	1.5E-07	1.0E-07				2.7E-07	1.2E-07	7.8E-08			
2013	3.7E-05	1.7E-05	1.1E-05				3.1E-07	1.4E-07	9.4E-08				2.4E-07	1.1E-07	7.3E-08			
2014	3.3E-05	1.6E-05	1.1E-05				2.7E-07	1.3E-07	8.9E-08				2.1E-07	1.0E-07	6.9E-08			
2015	3.2E-05	1.6E-05	1.1E-05				2.6E-07	1.3E-07	8.6E-08				2.0E-07	1.0E-07	6.7E-08			
2016	4.2E-05	1.5E-05	9.9E-06				3.5E-07	1.3E-07	8.0E-08				2.7E-07	9.7E-08	6.2E-08			
2017	4.1E-05	1.6E-05	1.0E-05				3.3E-07	1.3E-07	8.4E-08				2.6E-07	9.9E-08	6.5E-08			
2018	4.0E-05	1.5E-05	9.6E-06				3.2E-07	1.2E-07	7.8E-08				2.5E-07	9.4E-08	6.1E-08			

TABLE 3-11: SUMMARY OF TRI+ SEDIMENT CONCENTRATIONS FROM THE HUDTOX MODEL AND TEQ-BASED PREDICTIONS FOR 1993 - 2018

REVISED

	Tri+ Average PCB Results			Tri+ 95% UCL Results			Average Avian TEF			95% Avian TEF			Average Mammalian TEF			95% UCL Mammalian TEF		
Year	189 Total Sed Conc mg/kg	168 Total Sed Conc mg/kg	154 Total Sed Conc mg/kg	189 Total Sed Conc mg/kg	168 Total Sed Conc mg/kg	154 Total Sed Conc mg/kg	189 Total Sed Conc mg/kg	168 Total Sed Conc mg/kg	154 Total Sed Conc mg/kg	189 Total Sed Conc mg/kg	168 Total Sed Conc mg/kg	154 Total Sed Conc mg/kg	189 Total Sed Conc mg/kg	168 Total Sed Conc mg/kg	154 Total Sed Conc mg/kg	189 Total Sed Conc mg/kg	168 Total Sed Conc mg/kg	154 Total Sed Conc mg/kg
1993	54.12	7.04	2.18				1.5E-01	2.0E-02	6.0E-03				4.2E-02	5.5E-03	1.7E-03			
1994	48.82	6.46	1.99				1.4E-01	1.8E-02	5.5E-03				3.8E-02	5.0E-03	1.5E-03			
1995	46.22	6.09	1.86				1.3E-01	1.7E-02	5.2E-03				3.6E-02	4.7E-03	1.4E-03			
1996	41.48	5.25	1.70				1.2E-01	1.5E-02	4.7E-03				3.2E-02	4.1E-03	1.3E-03			
1997	36.99	4.42	1.52				1.0E-01	1.2E-02	4.2E-03				2.9E-02	3.4E-03	1.2E-03			
1998	22.15	4.78	1.34				6.2E-02	1.3E-02	3.7E-03				1.7E-02	3.7E-03	1.0E-03			
1999	21.09	4.36	1.25				5.9E-02	1.2E-02	3.5E-03				1.6E-02	3.4E-03	9.7E-04			
2000	19.17	3.86	1.14				5.3E-02	1.1E-02	3.2E-03				1.5E-02	3.0E-03	8.8E-04			
2001	17.03	3.21	1.01				4.7E-02	8.9E-03	2.8E-03				1.3E-02	2.5E-03	7.8E-04			
2002	15.75	2.75	0.90				4.4E-02	7.6E-03	2.5E-03				1.2E-02	2.1E-03	7.0E-04			
2003	14.41	2.41	0.82				4.0E-02	6.7E-03	2.3E-03				1.1E-02	1.9E-03	6.4E-04			
2004	13.21	2.31	0.76				3.7E-02	6.4E-03	2.1E-03				1.0E-02	1.8E-03	5.9E-04			
2005	12.31	2.19	0.68				3.4E-02	6.1E-03	1.9E-03				9.6E-03	1.7E-03	5.3E-04			
2006	11.50	1.97	0.61				3.2E-02	5.5E-03	1.7E-03				8.9E-03	1.5E-03	4.7E-04			
2007	10.37	1.79	0.54				2.9E-02	5.0E-03	1.5E-03				8.1E-03	1.4E-03	4.2E-04			
2008	9.43	1.66	0.50				2.6E-02	4.6E-03	1.4E-03				7.3E-03	1.3E-03	3.9E-04			
2009	8.99	1.60	0.47				2.5E-02	4.5E-03	1.3E-03				7.0E-03	1.2E-03	3.6E-04			
2010	8.22	1.48	0.43				2.3E-02	4.1E-03	1.2E-03				6.4E-03	1.1E-03	3.3E-04			
2011	7.36	1.32	0.39				2.0E-02	3.7E-03	1.1E-03				5.7E-03	1.0E-03	3.0E-04			
2012	6.67	1.20	0.36				1.9E-02	3.3E-03	9.9E-04				5.2E-03	9.3E-04	2.8E-04			
2013	6.03	1.09	0.32				1.7E-02	3.0E-03	8.9E-04				4.7E-03	8.5E-04	2.5E-04			
2014	5.49	1.25	0.30				1.5E-02	3.5E-03	8.4E-04				4.3E-03	9.7E-04	2.4E-04			
2015	5.08	1.21	0.29				1.4E-02	3.4E-03	7.9E-04				4.0E-03	9.4E-04	2.2E-04			
2016	4.76	1.19	0.27				1.3E-02	3.3E-03	7.6E-04				3.7E-03	9.3E-04	2.1E-04			
2017	4.49	1.17	0.27				1.2E-02	3.3E-03	7.5E-04				3.5E-03	9.1E-04	2.1E-04			
2018	4.17	1.11	0.26				1.2E-02	3.1E-03	7.2E-04				3.2E-03	8.6E-04	2.0E-04			

304229

TABLE 3-28: SUMMARY OF $ADD_{Expected}$ AND EGG CONCENTRATIONS FOR FEMALE SWALLOW BASED ON TRI+ CONGENERS FOR PERIOD 1993 - 2018 REVISED

Year	Total Average Dietary Dose (mg/Kg/day)			Average Egg Concentration (mg/Kg)		
	189	168	154	189	168	154
1993	1.16E+01	9.54E+00	3.84E+00	2.70E+01	2.23E+01	8.97E+00
1994	1.07E+01	8.88E+00	3.56E+00	2.49E+01	2.07E+01	8.31E+00
1995	9.85E+00	7.99E+00	3.30E+00	2.30E+01	1.87E+01	7.69E+00
1996	8.82E+00	6.85E+00	2.97E+00	2.06E+01	1.60E+01	6.92E+00
1997	7.86E+00	5.86E+00	2.69E+00	1.83E+01	1.37E+01	6.28E+00
1998	4.86E+00	6.44E+00	2.39E+00	1.13E+01	1.50E+01	5.58E+00
1999	4.51E+00	5.81E+00	2.21E+00	1.05E+01	1.35E+01	5.15E+00
2000	4.08E+00	5.02E+00	1.98E+00	9.52E+00	1.17E+01	4.63E+00
2001	3.68E+00	4.23E+00	1.77E+00	8.59E+00	9.88E+00	4.13E+00
2002	3.39E+00	3.65E+00	1.60E+00	7.91E+00	8.51E+00	3.72E+00
2003	3.11E+00	3.34E+00	1.47E+00	7.25E+00	7.80E+00	3.42E+00
2004	2.88E+00	3.17E+00	1.33E+00	6.72E+00	7.40E+00	3.10E+00
2005	2.68E+00	2.94E+00	1.19E+00	6.24E+00	6.85E+00	2.77E+00
2006	2.46E+00	2.65E+00	1.06E+00	5.74E+00	6.19E+00	2.48E+00
2007	2.22E+00	2.44E+00	9.61E-01	5.18E+00	5.69E+00	2.24E+00
2008	2.08E+00	2.28E+00	8.92E-01	4.85E+00	5.33E+00	2.08E+00
2009	1.94E+00	2.18E+00	8.30E-01	4.52E+00	5.08E+00	1.94E+00
2010	1.76E+00	1.98E+00	7.59E-01	4.10E+00	4.61E+00	1.77E+00
2011	1.58E+00	1.78E+00	6.93E-01	3.68E+00	4.15E+00	1.62E+00
2012	1.42E+00	1.62E+00	6.25E-01	3.32E+00	3.77E+00	1.46E+00
2013	1.29E+00	1.65E+00	5.78E-01	3.02E+00	3.84E+00	1.35E+00
2014	1.19E+00	1.74E+00	5.43E-01	2.78E+00	4.06E+00	1.27E+00
2015	1.11E+00	1.70E+00	5.14E-01	2.59E+00	3.97E+00	1.20E+00
2016	1.04E+00	1.66E+00	5.08E-01	2.43E+00	3.87E+00	1.18E+00
2017	9.76E-01	1.62E+00	4.86E-01	2.28E+00	3.77E+00	1.13E+00
2018	9.21E-01	1.55E+00	4.70E-01	2.15E+00	3.62E+00	1.10E+00

TABLE 3-29: SUMMARY OF $ADD_{95\%UCL}$ AND EGG CONCENTRATIONS FOR FEMALE SWALLOW BASED ON TRI+ CONGENERS FOR PERIOD 1993 - 2018 REVISED

Year	Total 95% UCL Dietary Dose (mg/Kg/day)			95% UCL Egg Concentration (mg/Kg)		
	189	168	154	189	168	154
1993	1.22E+01	1.01E+01	4.09E+00	2.85E+01	2.37E+01	9.54E+00
1994	1.13E+01	9.46E+00	3.79E+00	2.63E+01	2.21E+01	8.85E+00
1995	1.04E+01	8.51E+00	3.51E+00	2.43E+01	1.98E+01	8.19E+00
1996	9.32E+00	7.29E+00	3.16E+00	2.17E+01	1.70E+01	7.36E+00
1997	8.30E+00	6.24E+00	2.87E+00	1.94E+01	1.46E+01	6.69E+00
1998	5.13E+00	6.86E+00	2.55E+00	1.20E+01	1.60E+01	5.95E+00
1999	4.77E+00	6.18E+00	2.35E+00	1.11E+01	1.44E+01	5.49E+00
2000	4.31E+00	5.34E+00	2.11E+00	1.01E+01	1.25E+01	4.92E+00
2001	3.89E+00	4.51E+00	1.88E+00	9.08E+00	1.05E+01	4.39E+00
2002	3.59E+00	3.88E+00	1.70E+00	8.37E+00	9.06E+00	3.96E+00
2003	3.29E+00	3.56E+00	1.56E+00	7.67E+00	8.30E+00	3.64E+00
2004	3.04E+00	3.38E+00	1.42E+00	7.10E+00	7.88E+00	3.30E+00
2005	2.83E+00	3.12E+00	1.26E+00	6.60E+00	7.29E+00	2.95E+00
2006	2.60E+00	2.82E+00	1.13E+00	6.07E+00	6.59E+00	2.64E+00
2007	2.35E+00	2.60E+00	1.02E+00	5.48E+00	6.06E+00	2.39E+00
2008	2.20E+00	2.43E+00	9.50E-01	5.12E+00	5.68E+00	2.22E+00
2009	2.05E+00	2.32E+00	8.84E-01	4.77E+00	5.41E+00	2.06E+00
2010	1.86E+00	2.10E+00	8.07E-01	4.33E+00	4.91E+00	1.88E+00
2011	1.67E+00	1.89E+00	7.37E-01	3.89E+00	4.42E+00	1.72E+00
2012	1.51E+00	1.72E+00	6.65E-01	3.51E+00	4.01E+00	1.55E+00
2013	1.37E+00	1.75E+00	6.15E-01	3.19E+00	4.09E+00	1.44E+00
2014	1.26E+00	1.85E+00	5.79E-01	2.93E+00	4.32E+00	1.35E+00
2015	1.17E+00	1.81E+00	5.48E-01	2.74E+00	4.23E+00	1.28E+00
2016	1.10E+00	1.77E+00	5.41E-01	2.57E+00	4.12E+00	1.26E+00
2017	1.03E+00	1.72E+00	5.17E-01	2.41E+00	4.02E+00	1.21E+00
2018	9.74E-01	1.65E+00	5.01E-01	2.27E+00	3.86E+00	1.17E+00

TABLE 3-30: SUMMARY OF $ADD_{Expected}$ AND EGG CONCENTRATIONS FOR FEMALE MALLARD BASED ON 1993 DATA USING SUM OF TRI+ CONGENERS - REVISED

Location	Drinking Water Expected	Macrophyte Expected	Benthic Invertebrate Expected	Sediment Expected	Total Average Daily $Dose_{Expected}$ (mg/Kg/day)	Total Average Concentration in Eggs (mg/Kg)
Upper River						
Thompson Island Pool (189)	4.24E-06	1.49E-01	1.95E+00	1.37E-02	2.11E+00	4.24E+01
Stillwater (168)	7.53E-06	2.41E-01	3.63E+00	3.57E-02	3.91E+00	7.91E+01
Federal Dam (154)	5.26E-06	2.35E-01	8.66E-01	3.21E-03	1.10E+00	1.89E+01
Lower River						
143.5	4.07E-06	1.33E-01	1.21E-01	9.90E-04	2.55E-01	2.63E+00
137.2	4.07E-06	1.33E-01	2.38E-01	1.75E-03	3.73E-01	5.17E+00
122.4	1.86E-06	7.92E-02	1.11E-01	1.11E-03	1.91E-01	2.41E+00
113.8	1.86E-06	7.92E-02	9.52E-02	1.16E-03	1.75E-01	2.07E+00
100	1.86E-06	7.92E-02	5.23E-02	4.59E-04	1.32E-01	1.14E+00
88.9	1.23E-06	5.79E-02	2.63E-02	8.98E-04	8.51E-02	5.72E-01
58.7	1.23E-06	5.79E-02	6.76E-02	2.90E-04	1.26E-01	1.47E+00
47.3	1.23E-06	5.79E-02	9.18E-02	1.77E-03	1.52E-01	2.00E+00
25.8	1.23E-06	5.79E-02	2.72E-02	6.66E-04	8.58E-02	5.92E-01

TABLE 3-31: SUMMARY OF $ADD_{95\%UCL}$ AND EGG CONCENTRATIONS FOR FEMALE MALLARD BASED ON 1993 DATA USING SUM OF TRI+ CONGENERS - REVISED

Location	Drinking Water 95% UCL	Macrophyte 95% UCL	Benthic Invertebrate 95% UCL	Sediment 95% UCL	Total Upper Bound Daily $Dose_{95\%UCL}$ (mg/Kg/day)	Total Concentration in Eggs (95% UCL) (mg/Kg)
Upper River						
Thompson Island Pool (189)	1.34E-05	5.88E-01	3.06E+00	2.00E-02	3.67E+00	6.66E+01
Stillwater (168)	2.39E-05	6.45E-01	6.32E+00	6.23E-02	7.03E+00	1.38E+02
Federal Dam (154)	1.13E-05	4.87E-01	1.51E+00	5.39E-03	2.00E+00	3.28E+01
Lower River						
143.5	4.43E-05	3.99E-01	2.10E-01	1.08E-03	6.10E-01	4.57E+00
137.2	4.43E-05	3.99E-01	4.14E-01	3.53E-03	8.16E-01	9.01E+00
122.4	2.39E-05	1.22E+00	2.78E-01	1.23E-03	1.50E+00	6.06E+00
113.8	2.39E-05	1.22E+00	1.66E-01	1.92E-03	1.39E+00	3.61E+00
100	2.39E-05	1.22E+00	3.58E-01	9.91E-03	1.59E+00	7.79E+00
88.9	5.46E-06	1.13E+00	4.67E-02	2.63E-03	1.17E+00	1.02E+00
58.7	5.46E-06	1.13E+00	1.18E-01	3.22E-03	1.25E+00	2.56E+00
47.3	5.46E-06	1.13E+00	6.74E-01	6.91E-03	1.81E+00	1.47E+01
25.8	5.46E-06	1.13E+00	4.61E-02	1.80E-03	1.17E+00	1.00E+00

TABLE 3-32: SUMMARY OF $ADD_{Expected}$ AND EGG CONCENTRATIONS FOR FEMALE MALLARD BASED ON TRI+ CONGENERS FOR PERIOD 1993 - 2018 REVISED

	Average Dietary Dose (mg/Kg/day)			Average Egg Concentration (mg/Kg)		
Year	189	168	154	189	168	154
1993	1.11E+01	1.91E+00	8.14E-01	4.04E+01	3.34E+01	1.35E+01
1994	9.78E+00	1.79E+00	7.48E-01	3.74E+01	3.11E+01	1.25E+01
1995	7.66E+00	1.55E+00	6.70E-01	3.45E+01	2.80E+01	1.15E+01
1996	5.01E+00	1.31E+00	5.90E-01	3.09E+01	2.40E+01	1.04E+01
1997	4.76E+00	1.15E+00	5.51E-01	2.75E+01	2.05E+01	9.42E+00
1998	5.82E+00	1.26E+00	4.97E-01	1.70E+01	2.26E+01	8.37E+00
1999	5.13E+00	1.10E+00	4.52E-01	1.58E+01	2.03E+01	7.73E+00
2000	3.24E+00	9.53E-01	4.02E-01	1.43E+01	1.76E+01	6.94E+00
2001	3.15E+00	7.93E-01	3.50E-01	1.29E+01	1.48E+01	6.19E+00
2002	3.17E+00	6.91E-01	3.14E-01	1.19E+01	1.28E+01	5.59E+00
2003	3.85E+00	6.68E-01	3.12E-01	1.09E+01	1.17E+01	5.13E+00
2004	3.93E+00	6.25E-01	2.77E-01	1.01E+01	1.11E+01	4.66E+00
2005	3.14E+00	5.86E-01	2.48E-01	9.37E+00	1.03E+01	4.16E+00
2006	3.04E+00	5.15E-01	2.16E-01	8.61E+00	9.29E+00	3.72E+00
2007	2.62E+00	4.75E-01	2.01E-01	7.77E+00	8.54E+00	3.36E+00
2008	3.53E+00	4.61E-01	1.87E-01	7.27E+00	7.99E+00	3.12E+00
2009	3.15E+00	4.46E-01	1.78E-01	6.77E+00	7.62E+00	2.90E+00
2010	2.18E+00	3.86E-01	1.58E-01	6.14E+00	6.92E+00	2.66E+00
2011	2.27E+00	3.54E-01	1.49E-01	5.52E+00	6.23E+00	2.42E+00
2012	2.11E+00	3.25E-01	1.33E-01	4.99E+00	5.66E+00	2.19E+00
2013	1.89E+00	3.26E-01	1.22E-01	4.53E+00	5.77E+00	2.02E+00
2014	1.87E+00	3.50E-01	1.22E-01	4.16E+00	6.09E+00	1.90E+00
2015	1.82E+00	3.42E-01	1.17E-01	3.89E+00	5.96E+00	1.80E+00
2016	2.59E+00	3.34E-01	1.12E-01	3.65E+00	5.80E+00	1.78E+00
2017	2.49E+00	3.26E-01	1.11E-01	3.42E+00	5.66E+00	1.70E+00
2018	2.71E+00	3.37E-01	1.18E-01	3.22E+00	5.44E+00	1.64E+00

TABLE 3-33: SUMMARY OF $ADD_{95\%UCL}$ AND EGG CONCENTRATIONS FOR FEMALE MALLARD BASED ON TRI+ CONGENERS FOR PERIOD 1993 - 2018 REVISED

Year	95% UCL Dietary Dose (mg/Kg/day)			95% UCL Egg Concentration (mg/Kg)		
	189	168	154	189	168	154
1993	1.20E+01	2.05E+00	8.72E-01	4.28E+01	3.55E+01	1.43E+01
1994	1.06E+01	1.91E+00	8.02E-01	3.95E+01	3.31E+01	1.33E+01
1995	8.27E+00	1.66E+00	7.17E-01	3.64E+01	2.98E+01	1.23E+01
1996	5.40E+00	1.40E+00	6.32E-01	3.26E+01	2.55E+01	1.10E+01
1997	5.13E+00	1.23E+00	5.90E-01	2.90E+01	2.18E+01	1.00E+01
1998	6.29E+00	1.35E+00	5.32E-01	1.80E+01	2.40E+01	8.92E+00
1999	5.55E+00	1.18E+00	4.84E-01	1.67E+01	2.16E+01	8.23E+00
2000	3.50E+00	1.02E+00	4.30E-01	1.51E+01	1.87E+01	7.39E+00
2001	3.40E+00	8.47E-01	3.74E-01	1.36E+01	1.58E+01	6.58E+00
2002	3.42E+00	7.39E-01	3.36E-01	1.25E+01	1.36E+01	5.94E+00
2003	4.16E+00	7.15E-01	3.34E-01	1.15E+01	1.25E+01	5.46E+00
2004	4.25E+00	6.69E-01	2.96E-01	1.07E+01	1.18E+01	4.95E+00
2005	3.40E+00	6.27E-01	2.66E-01	9.90E+00	1.09E+01	4.42E+00
2006	3.28E+00	5.51E-01	2.31E-01	9.10E+00	9.88E+00	3.96E+00
2007	2.83E+00	5.08E-01	2.16E-01	8.21E+00	9.09E+00	3.58E+00
2008	3.82E+00	4.93E-01	2.01E-01	7.68E+00	8.52E+00	3.33E+00
2009	3.40E+00	4.78E-01	1.91E-01	7.16E+00	8.11E+00	3.09E+00
2010	2.36E+00	4.12E-01	1.70E-01	6.50E+00	7.36E+00	2.83E+00
2011	2.46E+00	3.79E-01	1.60E-01	5.84E+00	6.63E+00	2.58E+00
2012	2.29E+00	3.48E-01	1.43E-01	5.27E+00	6.02E+00	2.33E+00
2013	2.05E+00	3.49E-01	1.31E-01	4.79E+00	6.14E+00	2.15E+00
2014	2.02E+00	3.75E-01	1.32E-01	4.40E+00	6.48E+00	2.03E+00
2015	1.97E+00	3.66E-01	1.25E-01	4.11E+00	6.34E+00	1.92E+00
2016	2.81E+00	3.57E-01	1.21E-01	3.86E+00	6.18E+00	1.89E+00
2017	2.70E+00	3.49E-01	1.20E-01	3.61E+00	6.03E+00	1.81E+00
2018	2.93E+00	3.61E-01	1.27E-01	3.41E+00	5.79E+00	1.75E+00

TABLE 3-34: SUMMARY OF $ADD_{Expected}$ AND EGG CONCENTRATIONS FOR FEMALE BELTED KINGFISHER BASED ON 1993 DATA USING SUM OF TRI+ CONGENERS - REVISED

Location	Drinking Water Expected	Forage Fish Expected	Benthic Invertebrate Expected	Sediment Expected	Total Average Daily $Dose_{Expected}$ (mg/Kg/day)	Total Average Concentration in Eggs (mg/Kg)
Upper River						
Thompson Island Pool (189)	8.01E-06	6.37E+00	1.14E+00	1.35E-02	7.53E+00	5.71E+02
Stillwater (168)	1.42E-05	2.15E+00	2.00E+00	3.53E-02	4.18E+00	3.15E+02
Federal Dam (154)	9.95E-06	5.05E-01	4.76E-01	3.17E-03	9.84E-01	7.46E+01
Lower River						
143.5	7.70E-06	5.87E-01	6.63E-02	9.78E-04	6.54E-01	4.97E+01
137.2	7.70E-06	1.19E+00	1.31E-01	1.73E-03	1.32E+00	1.00E+02
122.4	3.53E-06	4.53E-01	7.22E-02	1.09E-03	5.27E-01	4.00E+01
113.8	3.53E-06	4.75E-01	7.43E-02	1.15E-03	5.51E-01	4.18E+01
100	3.53E-06	2.06E-01	3.41E-02	4.53E-04	2.41E-01	1.83E+01
88.9	2.32E-06	4.10E-01	1.71E-02	8.87E-04	4.28E-01	3.25E+01
58.7	2.32E-06	4.47E-01	5.27E-02	2.86E-04	5.00E-01	3.80E+01
47.3	2.32E-06	3.97E-01	5.99E-02	1.75E-03	4.59E-01	3.48E+01
25.8	2.32E-06	2.99E-01	1.77E-02	6.57E-04	3.17E-01	2.41E+01

TABLE 3-35: SUMMARY OF $ADD_{95\%UCL}$ AND EGG CONCENTRATIONS FOR FEMALE BELTED KINGFISHER BASED ON 1993 DATA USING SUM OF TRI+ CONGENERS - REVISED

Location	Drinking Water 95% UCL	Fish 95% UCL	Benthic Invertebrate 95% UCL	Sediment 95% UCL	Total Upper Bound Daily $Dose_{95\%UCL}$ (mg/Kg/day)	Total Concentration in Eggs (95 % UCL) (mg/Kg)
Upper River						
Thompson Island Pool (189)	2.54E-05	1.30E+01	2.00E+00	1.98E-02	1.50E+01	1.14E+03
Stillwater (168)	4.52E-05	3.08E+00	9.36E+00	6.16E-02	1.25E+01	9.45E+02
Federal Dam (154)	2.13E-05	7.33E-01	8.09E-01	5.32E-03	1.55E+00	1.17E+02
Lower River						
143.5	8.39E-05	7.05E-01	1.63E-01	1.07E-03	8.69E-01	6.60E+01
137.2	8.39E-05	2.58E+00	5.30E-01	3.49E-03	3.11E+00	2.36E+02
122.4	4.52E-05	7.33E-01	1.82E-01	1.22E-03	9.16E-01	6.96E+01
113.8	4.52E-05	4.93E-01	2.88E-01	1.89E-03	7.83E-01	5.94E+01
100	4.52E-05	3.56E-01	2.33E-01	9.79E-03	5.99E-01	4.48E+01
88.9	1.03E-05	5.62E-01	3.04E-02	2.60E-03	5.95E-01	4.50E+01
58.7	1.03E-05	5.06E-01	4.83E-01	3.17E-03	9.92E-01	7.52E+01
47.3	1.03E-05	5.27E-01	4.39E-01	6.82E-03	9.73E-01	7.35E+01
25.8	1.03E-05	3.59E-01	3.01E-02	1.78E-03	3.91E-01	2.96E+01

TABLE 3-36: SUMMARY OF $ADD_{Expected}$ AND EGG CONCENTRATIONS FOR FEMALE BELTED KINGFISHER BASED ON TRI+ CONGENERS FOR PERIOD 1993 - 2018 REVISED

	Average Dietary Dose (mg/Kg/day)			Average Egg Concentration (mg/Kg)		
Year	189	168	154	189	168	154
1993	6.08E+00	1.76E+00	7.40E-01	4.46E+02	1.32E+02	5.56E+01
1994	6.07E+00	1.65E+00	6.76E-01	4.47E+02	1.24E+02	5.08E+01
1995	5.53E+00	1.46E+00	6.17E-01	4.06E+02	1.09E+02	4.64E+01
1996	3.49E+00	1.23E+00	5.51E-01	2.53E+02	9.19E+01	4.14E+01
1997	2.42E+00	9.05E-01	4.24E-01	1.73E+02	6.75E+01	3.18E+01
1998	2.55E+00	1.14E+00	4.34E-01	1.87E+02	8.56E+01	3.26E+01
1999	2.37E+00	1.03E+00	4.08E-01	1.74E+02	7.71E+01	3.06E+01
2000	2.07E+00	8.90E-01	3.60E-01	1.52E+02	6.65E+01	2.70E+01
2001	1.87E+00	7.50E-01	3.23E-01	1.37E+02	5.61E+01	2.43E+01
2002	1.86E+00	6.56E-01	2.93E-01	1.37E+02	4.91E+01	2.20E+01
2003	1.80E+00	6.02E-01	2.68E-01	1.33E+02	4.51E+01	2.02E+01
2004	1.84E+00	5.70E-01	2.48E-01	1.36E+02	4.27E+01	1.86E+01
2005	1.65E+00	5.32E-01	2.20E-01	1.21E+02	3.98E+01	1.65E+01
2006	1.52E+00	4.78E-01	1.99E-01	1.12E+02	3.58E+01	1.49E+01
2007	1.47E+00	4.35E-01	1.78E-01	1.09E+02	3.26E+01	1.34E+01
2008	1.38E+00	4.20E-01	1.68E-01	1.02E+02	3.14E+01	1.26E+01
2009	1.43E+00	3.98E-01	1.56E-01	1.06E+02	2.98E+01	1.17E+01
2010	1.33E+00	3.58E-01	1.41E-01	9.88E+01	2.67E+01	1.06E+01
2011	1.13E+00	3.21E-01	1.29E-01	8.36E+01	2.40E+01	9.72E+00
2012	1.10E+00	3.02E-01	1.20E-01	8.15E+01	2.26E+01	9.02E+00
2013	9.71E-01	2.94E-01	1.08E-01	7.20E+01	2.21E+01	8.13E+00
2014	9.38E-01	3.15E-01	1.03E-01	6.97E+01	2.36E+01	7.74E+00
2015	8.70E-01	3.08E-01	9.79E-02	6.47E+01	2.30E+01	7.36E+00
2016	9.36E-01	3.05E-01	9.86E-02	6.97E+01	2.29E+01	7.42E+00
2017	9.47E-01	2.94E-01	9.47E-02	7.07E+01	2.20E+01	7.12E+00
2018	1.03E+00	2.94E-01	9.55E-02	7.71E+01	2.20E+01	7.18E+00

TABLE 3-37: SUMMARY OF $ADD_{95\%UCL}$ AND EGG CONCENTRATIONS FOR FEMALE BELTED KINGFISHER BASED ON TRI+ CONGENERS FOR PERIOD 1993 - 2018 REVISED

Year	95% UCL Dietary Dose (mg/Kg/day)			95% UCL Egg Concentration (mg/Kg)		
	189	168	154	189	168	154
1993	6.23E+00	1.86E+00	7.84E-01	4.57E+02	1.40E+02	5.89E+01
1994	6.23E+00	1.75E+00	7.17E-01	4.59E+02	1.31E+02	5.39E+01
1995	5.66E+00	1.55E+00	6.56E-01	4.17E+02	1.16E+02	4.93E+01
1996	3.60E+00	1.30E+00	5.85E-01	2.61E+02	9.75E+01	4.39E+01
1997	2.49E+00	9.65E-01	4.52E-01	1.78E+02	7.20E+01	3.39E+01
1998	2.61E+00	1.21E+00	4.60E-01	1.92E+02	9.08E+01	3.46E+01
1999	2.43E+00	1.09E+00	4.32E-01	1.79E+02	8.18E+01	3.25E+01
2000	2.12E+00	9.43E-01	3.81E-01	1.55E+02	7.05E+01	2.86E+01
2001	1.92E+00	7.95E-01	3.42E-01	1.41E+02	5.95E+01	2.57E+01
2002	1.91E+00	6.95E-01	3.10E-01	1.40E+02	5.20E+01	2.33E+01
2003	1.85E+00	6.38E-01	2.84E-01	1.37E+02	4.78E+01	2.14E+01
2004	1.88E+00	6.05E-01	2.63E-01	1.39E+02	4.53E+01	1.97E+01
2005	1.68E+00	5.64E-01	2.33E-01	1.24E+02	4.22E+01	1.75E+01
2006	1.56E+00	5.07E-01	2.11E-01	1.15E+02	3.79E+01	1.59E+01
2007	1.51E+00	4.61E-01	1.89E-01	1.11E+02	3.46E+01	1.42E+01
2008	1.41E+00	4.46E-01	1.79E-01	1.05E+02	3.34E+01	1.34E+01
2009	1.46E+00	4.21E-01	1.65E-01	1.09E+02	3.16E+01	1.24E+01
2010	1.36E+00	3.79E-01	1.50E-01	1.01E+02	2.84E+01	1.13E+01
2011	1.15E+00	3.40E-01	1.37E-01	8.56E+01	2.54E+01	1.03E+01
2012	1.12E+00	3.20E-01	1.27E-01	8.32E+01	2.40E+01	9.56E+00
2013	9.94E-01	3.12E-01	1.15E-01	7.38E+01	2.34E+01	8.62E+00
2014	9.58E-01	3.33E-01	1.09E-01	7.12E+01	2.50E+01	8.20E+00
2015	8.89E-01	3.26E-01	1.04E-01	6.61E+01	2.44E+01	7.79E+00
2016	9.57E-01	3.24E-01	1.05E-01	7.13E+01	2.43E+01	7.87E+00
2017	9.67E-01	3.11E-01	1.00E-01	7.22E+01	2.33E+01	7.55E+00
2018	1.06E+00	3.11E-01	1.01E-01	7.90E+01	2.33E+01	7.61E+00

TABLE 3-40: SUMMARY OF $ADD_{Expected}$ AND EGG CONCENTRATIONS FOR FEMALE GREAT BLUE HERON BASED ON TRI+ CONGENERS FOR PERIOD 1993 - 2018 REVISED

	Average Dietary Dose (mg/Kg/day)			Average Egg Concentration (mg/Kg)		
Year	189	168	154	189	168	154
1993	2.43E+00	3.96E-01	1.76E-01	4.58E+02	7.20E+01	3.23E+01
1994	2.48E+00	3.77E-01	1.58E-01	4.69E+02	6.86E+01	2.90E+01
1995	2.25E+00	3.26E-01	1.42E-01	4.24E+02	5.92E+01	2.60E+01
1996	1.28E+00	2.67E-01	1.26E-01	2.37E+02	4.83E+01	2.30E+01
1997	7.84E-01	1.53E-01	7.53E-02	1.43E+02	2.68E+01	1.33E+01
1998	1.02E+00	2.45E-01	9.60E-02	1.92E+02	4.43E+01	1.75E+01
1999	9.51E-01	2.21E-01	9.22E-02	1.79E+02	3.99E+01	1.68E+01
2000	8.21E-01	1.90E-01	7.97E-02	1.54E+02	3.44E+01	1.45E+01
2001	7.45E-01	1.60E-01	7.21E-02	1.40E+02	2.89E+01	1.32E+01
2002	7.55E-01	1.43E-01	6.57E-02	1.42E+02	2.59E+01	1.20E+01
2003	7.40E-01	1.32E-01	6.02E-02	1.40E+02	2.39E+01	1.10E+01
2004	7.71E-01	1.24E-01	5.67E-02	1.46E+02	2.25E+01	1.04E+01
2005	6.86E-01	1.17E-01	4.98E-02	1.30E+02	2.12E+01	9.11E+00
2006	6.34E-01	1.04E-01	4.58E-02	1.20E+02	1.89E+01	8.39E+00
2007	6.23E-01	9.39E-02	4.05E-02	1.18E+02	1.70E+01	7.40E+00
2008	5.84E-01	9.41E-02	3.91E-02	1.11E+02	1.71E+01	7.17E+00
2009	6.18E-01	8.84E-02	3.59E-02	1.17E+02	1.61E+01	6.58E+00
2010	5.78E-01	7.85E-02	3.24E-02	1.10E+02	1.42E+01	5.93E+00
2011	4.84E-01	7.02E-02	2.97E-02	9.19E+01	1.27E+01	5.44E+00
2012	4.77E-01	6.91E-02	2.85E-02	9.08E+01	1.26E+01	5.23E+00
2013	4.20E-01	6.37E-02	2.50E-02	7.99E+01	1.16E+01	4.57E+00
2014	4.10E-01	6.91E-02	2.41E-02	7.79E+01	1.25E+01	4.41E+00
2015	3.80E-01	6.75E-02	2.30E-02	7.23E+01	1.23E+01	4.21E+00
2016	4.17E-01	6.86E-02	2.37E-02	7.95E+01	1.25E+01	4.36E+00
2017	4.27E-01	6.48E-02	2.29E-02	8.14E+01	1.18E+01	4.21E+00
2018	4.72E-01	6.85E-02	2.41E-02	9.03E+01	1.25E+01	4.45E+00

TABLE 3-41: SUMMARY OF $ADD_{95\%UCL}$ AND EGG CONCENTRATIONS FOR FEMALE GREAT BLUE HERON BASED ON TRI+ CONGENERS FOR PERIOD 1993 - 2018 REVISED

Year	95% UCL Dietary Dose (mg/Kg/day)			95% UCL Egg Concentration (mg/Kg)		
	189	168	154	189	168	154
1993	2.48E+00	4.17E-01	1.86E-01	4.66E+02	7.60E+01	3.41E+01
1994	2.53E+00	3.98E-01	1.67E-01	4.78E+02	7.25E+01	3.07E+01
1995	2.29E+00	3.46E-01	1.51E-01	4.32E+02	6.28E+01	2.76E+01
1996	1.30E+00	2.82E-01	1.33E-01	2.42E+02	5.10E+01	2.43E+01
1997	7.97E-01	1.64E-01	8.08E-02	1.45E+02	2.89E+01	1.44E+01
1998	1.04E+00	2.59E-01	1.01E-01	1.96E+02	4.68E+01	1.85E+01
1999	9.67E-01	2.33E-01	9.73E-02	1.82E+02	4.22E+01	1.78E+01
2000	8.35E-01	2.00E-01	8.39E-02	1.57E+02	3.62E+01	1.53E+01
2001	7.57E-01	1.69E-01	7.62E-02	1.42E+02	3.06E+01	1.39E+01
2002	7.68E-01	1.51E-01	6.93E-02	1.45E+02	2.73E+01	1.27E+01
2003	7.58E-01	1.39E-01	6.33E-02	1.43E+02	2.52E+01	1.16E+01
2004	7.84E-01	1.31E-01	5.99E-02	1.48E+02	2.38E+01	1.10E+01
2005	6.97E-01	1.23E-01	5.26E-02	1.32E+02	2.24E+01	9.61E+00
2006	6.45E-01	1.10E-01	4.84E-02	1.22E+02	2.00E+01	8.86E+00
2007	6.33E-01	9.90E-02	4.27E-02	1.20E+02	1.79E+01	7.81E+00
2008	5.96E-01	9.95E-02	4.13E-02	1.13E+02	1.81E+01	7.58E+00
2009	6.29E-01	9.32E-02	3.79E-02	1.20E+02	1.69E+01	6.93E+00
2010	5.87E-01	8.27E-02	3.42E-02	1.12E+02	1.50E+01	6.26E+00
2011	4.93E-01	7.39E-02	3.13E-02	9.36E+01	1.34E+01	5.74E+00
2012	4.85E-01	7.29E-02	3.00E-02	9.22E+01	1.33E+01	5.52E+00
2013	4.28E-01	6.72E-02	2.63E-02	8.14E+01	1.22E+01	4.82E+00
2014	4.16E-01	7.29E-02	2.53E-02	7.92E+01	1.32E+01	4.64E+00
2015	3.86E-01	7.12E-02	2.42E-02	7.34E+01	1.29E+01	4.43E+00
2016	4.25E-01	7.24E-02	2.50E-02	8.10E+01	1.32E+01	4.61E+00
2017	4.33E-01	6.83E-02	2.41E-02	8.28E+01	1.24E+01	4.44E+00
2018	4.82E-01	7.20E-02	2.53E-02	9.22E+01	1.32E+01	4.68E+00

TABLE 3-44: SUMMARY OF $ADD_{Expected}$ AND EGG CONCENTRATIONS FOR FEMALE EAGLE BASED ON TRI+ CONGENERS FOR PERIOD 1993 - 2018
REVISED

	Average Dietary Dose (mg/Kg/day)			Average Egg Concentration (mg/Kg)		
Year	189	168	154	189	168	154
1993	1.68E+00	4.86E-01	2.39E-01	3.69E+02	1.07E+02	5.26E+01
1994	1.54E+00	4.39E-01	2.18E-01	3.39E+02	9.65E+01	4.79E+01
1995	1.76E+00	4.38E-01	2.10E-01	3.87E+02	9.61E+01	4.61E+01
1996	1.54E+00	4.01E-01	1.85E-01	3.39E+02	8.81E+01	4.07E+01
1997	1.26E+00	3.58E-01	1.65E-01	2.77E+02	7.86E+01	3.63E+01
1998	1.33E+00	3.36E-01	1.53E-01	2.91E+02	7.38E+01	3.36E+01
1999	1.36E+00	3.16E-01	1.41E-01	2.99E+02	6.95E+01	3.09E+01
2000	1.39E+00	3.10E-01	1.33E-01	3.06E+02	6.81E+01	2.91E+01
2001	1.17E+00	2.78E-01	1.22E-01	2.57E+02	6.10E+01	2.68E+01
2002	1.05E+00	2.48E-01	1.11E-01	2.31E+02	5.44E+01	2.45E+01
2003	1.11E+00	2.31E-01	1.01E-01	2.43E+02	5.07E+01	2.23E+01
2004	9.10E-01	2.16E-01	8.88E-02	2.00E+02	4.74E+01	1.95E+01
2005	8.22E-01	2.23E-01	8.59E-02	1.81E+02	4.90E+01	1.89E+01
2006	7.73E-01	2.22E-01	8.29E-02	1.70E+02	4.88E+01	1.82E+01
2007	8.65E-01	2.21E-01	8.06E-02	1.90E+02	4.86E+01	1.77E+01
2008	9.62E-01	2.18E-01	7.99E-02	2.11E+02	4.79E+01	1.75E+01
2009	9.12E-01	2.10E-01	7.79E-02	2.00E+02	4.62E+01	1.71E+01
2010	9.38E-01	2.04E-01	7.29E-02	2.06E+02	4.48E+01	1.60E+01
2011	8.18E-01	1.96E-01	7.10E-02	1.80E+02	4.31E+01	1.56E+01
2012	7.07E-01	1.86E-01	6.93E-02	1.55E+02	4.08E+01	1.52E+01
2013	6.91E-01	1.73E-01	6.70E-02	1.52E+02	3.80E+01	1.47E+01
2014	7.92E-01	1.77E-01	6.83E-02	1.74E+02	3.89E+01	1.50E+01
2015	7.55E-01	1.70E-01	6.50E-02	1.66E+02	3.74E+01	1.43E+01
2016	6.76E-01	1.51E-01	5.84E-02	1.48E+02	3.32E+01	1.28E+01
2017	6.03E-01	1.44E-01	5.89E-02	1.33E+02	3.17E+01	1.29E+01
2018	7.13E-01	1.50E-01	5.95E-02	1.57E+02	3.30E+01	1.31E+01

TABLE 3-45: SUMMARY OF $ADD_{95\%UCL}$ AND EGG CONCENTRATIONS FOR FEMALE EAGLE BASED ON TRI+ CONGENERS FOR PERIOD 1993 - 2018 REVISED

Year	95% UCL Dietary Dose (mg/Kg/day)			95% UCL Egg Concentration (mg/Kg)		
	189	168	154	189	168	154
1993	1.75E+00	4.93E-01	2.43E-01	3.85E+02	1.08E+02	5.34E+01
1994	1.61E+00	4.46E-01	2.22E-01	3.54E+02	9.79E+01	4.87E+01
1995	1.84E+00	4.44E-01	2.13E-01	4.04E+02	9.75E+01	4.68E+01
1996	1.62E+00	4.07E-01	1.88E-01	3.56E+02	8.93E+01	4.13E+01
1997	1.32E+00	3.63E-01	1.68E-01	2.90E+02	7.97E+01	3.68E+01
1998	1.39E+00	3.41E-01	1.55E-01	3.04E+02	7.48E+01	3.41E+01
1999	1.43E+00	3.21E-01	1.43E-01	3.13E+02	7.05E+01	3.14E+01
2000	1.46E+00	3.15E-01	1.35E-01	3.20E+02	6.91E+01	2.96E+01
2001	1.23E+00	2.82E-01	1.24E-01	2.70E+02	6.19E+01	2.72E+01
2002	1.10E+00	2.51E-01	1.13E-01	2.41E+02	5.52E+01	2.49E+01
2003	1.15E+00	2.34E-01	1.03E-01	2.54E+02	5.14E+01	2.26E+01
2004	9.53E-01	2.19E-01	9.03E-02	2.09E+02	4.81E+01	1.98E+01
2005	8.62E-01	2.26E-01	8.72E-02	1.89E+02	4.97E+01	1.92E+01
2006	8.09E-01	2.25E-01	8.41E-02	1.78E+02	4.95E+01	1.85E+01
2007	9.06E-01	2.24E-01	8.19E-02	1.99E+02	4.93E+01	1.80E+01
2008	1.01E+00	2.21E-01	8.11E-02	2.21E+02	4.86E+01	1.78E+01
2009	9.54E-01	2.13E-01	7.90E-02	2.10E+02	4.68E+01	1.74E+01
2010	9.78E-01	2.07E-01	7.40E-02	2.15E+02	4.54E+01	1.63E+01
2011	8.59E-01	1.99E-01	7.21E-02	1.89E+02	4.37E+01	1.58E+01
2012	7.39E-01	1.88E-01	7.03E-02	1.62E+02	4.14E+01	1.54E+01
2013	7.22E-01	1.76E-01	6.80E-02	1.59E+02	3.86E+01	1.49E+01
2014	8.28E-01	1.80E-01	6.93E-02	1.82E+02	3.94E+01	1.52E+01
2015	7.90E-01	1.73E-01	6.59E-02	1.74E+02	3.80E+01	1.45E+01
2016	7.08E-01	1.53E-01	5.93E-02	1.56E+02	3.37E+01	1.30E+01
2017	6.32E-01	1.46E-01	5.98E-02	1.39E+02	3.22E+01	1.31E+01
2018	7.44E-01	1.52E-01	6.04E-02	1.63E+02	3.35E+01	1.33E+01

TABLE 3-48: SUMMARY OF $ADD_{Expected}$ AND EGG CONCENTRATIONS FOR FEMALE TREE SWALLOW FOR THE PERIOD 1993 - 2018 ON TEQ BASIS REVISED

Year	Total Average Dietary Dose (mg/Kg/day)			Average Egg Concentration (mg/Kg)		
	189	168	154	189	168	154
1993	6.16E-04	5.08E-04	2.05E-04	5.03E-03	4.15E-03	1.67E-03
1994	5.69E-04	4.73E-04	1.90E-04	4.65E-03	3.86E-03	1.55E-03
1995	5.25E-04	4.26E-04	1.76E-04	4.29E-03	3.48E-03	1.43E-03
1996	4.70E-04	3.65E-04	1.58E-04	3.84E-03	2.98E-03	1.29E-03
1997	4.19E-04	3.13E-04	1.43E-04	3.42E-03	2.55E-03	1.17E-03
1998	2.59E-04	3.43E-04	1.28E-04	2.11E-03	2.80E-03	1.04E-03
1999	2.41E-04	3.09E-04	1.18E-04	1.96E-03	2.53E-03	9.61E-04
2000	2.17E-04	2.67E-04	1.06E-04	1.78E-03	2.18E-03	8.63E-04
2001	1.96E-04	2.26E-04	9.43E-05	1.60E-03	1.84E-03	7.69E-04
2002	1.81E-04	1.94E-04	8.51E-05	1.48E-03	1.59E-03	6.94E-04
2003	1.66E-04	1.78E-04	7.81E-05	1.35E-03	1.45E-03	6.37E-04
2004	1.54E-04	1.69E-04	7.09E-05	1.25E-03	1.38E-03	5.79E-04
2005	1.43E-04	1.56E-04	6.33E-05	1.16E-03	1.28E-03	5.17E-04
2006	1.31E-04	1.41E-04	5.66E-05	1.07E-03	1.15E-03	4.62E-04
2007	1.18E-04	1.30E-04	5.12E-05	9.66E-04	1.06E-03	4.18E-04
2008	1.11E-04	1.22E-04	4.76E-05	9.04E-04	9.94E-04	3.88E-04
2009	1.03E-04	1.16E-04	4.43E-05	8.42E-04	9.48E-04	3.61E-04
2010	9.36E-05	1.05E-04	4.05E-05	7.64E-04	8.60E-04	3.30E-04
2011	8.42E-05	9.49E-05	3.69E-05	6.87E-04	7.74E-04	3.01E-04
2012	7.60E-05	8.61E-05	3.33E-05	6.20E-04	7.03E-04	2.72E-04
2013	6.91E-05	8.78E-05	3.08E-05	5.63E-04	7.17E-04	2.51E-04
2014	6.34E-05	9.28E-05	2.90E-05	5.18E-04	7.57E-04	2.36E-04
2015	5.93E-05	9.07E-05	2.74E-05	4.84E-04	7.41E-04	2.24E-04
2016	5.57E-05	8.84E-05	2.71E-05	4.54E-04	7.21E-04	2.21E-04
2017	5.21E-05	8.62E-05	2.59E-05	4.25E-04	7.03E-04	2.11E-04
2018	4.92E-05	8.28E-05	2.51E-05	4.01E-04	6.76E-04	2.04E-04

TABLE 3-49: SUMMARY OF $ADD_{95\%UCL}$ AND EGG CONCENTRATIONS FOR FEMALE TREE SWALLOW FOR THE PERIOD 1993 - 2018 ON TEQ BASIS REVISED

Year	Total 95% UCL Dietary Dose (mg/Kg/day)			95% UCL Egg Concentration (mg/Kg)		
	189	168	154	189	168	154
1993	6.51E-04	5.41E-04	2.18E-04	5.31E-03	4.42E-03	1.78E-03
1994	6.02E-04	5.04E-04	2.02E-04	4.91E-03	4.11E-03	1.65E-03
1995	5.55E-04	4.53E-04	1.87E-04	4.53E-03	3.70E-03	1.53E-03
1996	4.97E-04	3.88E-04	1.68E-04	4.05E-03	3.17E-03	1.37E-03
1997	4.42E-04	3.33E-04	1.53E-04	3.61E-03	2.71E-03	1.25E-03
1998	2.74E-04	3.66E-04	1.36E-04	2.23E-03	2.99E-03	1.11E-03
1999	2.54E-04	3.29E-04	1.25E-04	2.08E-03	2.69E-03	1.02E-03
2000	2.30E-04	2.84E-04	1.13E-04	1.88E-03	2.32E-03	9.18E-04
2001	2.08E-04	2.40E-04	1.00E-04	1.69E-03	1.96E-03	8.19E-04
2002	1.91E-04	2.07E-04	9.05E-05	1.56E-03	1.69E-03	7.39E-04
2003	1.75E-04	1.90E-04	8.31E-05	1.43E-03	1.55E-03	6.78E-04
2004	1.62E-04	1.80E-04	7.55E-05	1.32E-03	1.47E-03	6.16E-04
2005	1.51E-04	1.66E-04	6.74E-05	1.23E-03	1.36E-03	5.50E-04
2006	1.39E-04	1.50E-04	6.03E-05	1.13E-03	1.23E-03	4.92E-04
2007	1.25E-04	1.38E-04	5.45E-05	1.02E-03	1.13E-03	4.45E-04
2008	1.17E-04	1.30E-04	5.07E-05	9.55E-04	1.06E-03	4.13E-04
2009	1.09E-04	1.24E-04	4.71E-05	8.90E-04	1.01E-03	3.84E-04
2010	9.90E-05	1.12E-04	4.30E-05	8.08E-04	9.15E-04	3.51E-04
2011	8.90E-05	1.01E-04	3.93E-05	7.26E-04	8.24E-04	3.21E-04
2012	8.03E-05	9.17E-05	3.55E-05	6.55E-04	7.48E-04	2.89E-04
2013	7.30E-05	9.34E-05	3.28E-05	5.95E-04	7.63E-04	2.68E-04
2014	6.70E-05	9.87E-05	3.09E-05	5.47E-04	8.06E-04	2.52E-04
2015	6.27E-05	9.66E-05	2.92E-05	5.11E-04	7.89E-04	2.38E-04
2016	5.88E-05	9.41E-05	2.88E-05	4.79E-04	7.68E-04	2.35E-04
2017	5.51E-05	9.18E-05	2.76E-05	4.49E-04	7.49E-04	2.25E-04
2018	5.19E-05	8.81E-05	2.67E-05	4.24E-04	7.19E-04	2.18E-04

TABLE 3-50: SUMMARY OF $ADD_{Expected}$ AND EGG CONCENTRATIONS FOR FEMALE MALLARD ON A TEQ BASIS REVISED

Location	Drinking Water Expected	Macrophyte Expected	Benthic Invertebrate Expected	Sediment Expected	Total Average Daily $Dose_{Expected}$ (mg/Kg/day)	Total Average Concentration in Eggs (mg/Kg)
Upper River						
Thompson Island Pool (189)	3.46E-08	1.16E-03	1.04E-04	3.80E-05	1.31E-03	2.11E-02
Stillwater (168)	6.14E-08	1.89E-03	1.94E-04	9.92E-05	2.18E-03	3.93E-02
Federal Dam (154)	4.29E-08	1.84E-03	4.61E-05	8.93E-06	1.89E-03	9.38E-03
Lower River						
143.5	3.32E-08	1.09E-04	1.67E-05	2.75E-06	1.28E-04	3.40E-03
137.2	3.32E-08	1.09E-04	3.29E-05	4.86E-06	1.47E-04	6.69E-03
122.4	1.52E-08	6.46E-05	1.53E-05	3.08E-06	8.30E-05	3.12E-03
113.8	1.52E-08	6.46E-05	1.32E-05	3.23E-06	8.10E-05	2.68E-03
100	1.52E-08	6.46E-05	7.25E-06	1.27E-06	7.31E-05	1.47E-03
88.9	1.00E-08	4.73E-05	3.64E-06	2.50E-06	5.34E-05	7.40E-04
58.7	1.00E-08	4.73E-05	9.36E-06	8.05E-07	5.75E-05	1.90E-03
47.3	1.00E-08	4.73E-05	1.27E-05	4.91E-06	6.49E-05	2.59E-03
25.8	1.00E-08	4.73E-05	3.77E-06	1.85E-06	5.29E-05	7.66E-04

TABLE 3-51: SUMMARY OF $ADD_{95\%UCL}$ AND EGG CONCENTRATIONS FOR FEMALE MALLARD BASED ON 1993 DATA ON A TEQ BASIS

REVISED

Location	Drinking Water 95% UCL	Macrophyte 95% UCL	Benthic Invertebrate 95% UCL	Sediment 95% UCL	Total Upper Bound Daily $Dose_{95\%UCL}$ (mg/Kg/day)	Total Concentration in Eggs (95% UCL) (mg/Kg)
Upper River						
Thompson Island Pool (189)	1.09E-07	4.61E-03	1.63E-04	5.56E-05	4.83E-03	3.31E-02
Stillwater (168)	1.95E-07	5.05E-03	3.37E-04	1.73E-04	5.56E-03	6.85E-02
Federal Dam (154)	9.21E-08	3.82E-03	8.03E-05	1.50E-05	3.91E-03	1.63E-02
Lower River						
143.5	3.62E-07	3.26E-04	2.91E-05	3.01E-06	3.58E-04	5.91E-03
137.2	3.62E-07	3.26E-04	5.73E-05	9.81E-06	3.93E-04	1.16E-02
122.4	1.95E-07	9.95E-04	3.86E-05	3.42E-06	1.04E-03	7.84E-03
113.8	1.95E-07	9.95E-04	2.30E-05	5.33E-06	1.02E-03	4.67E-03
100	1.95E-07	9.95E-04	4.96E-05	2.75E-05	1.07E-03	1.01E-02
88.9	4.45E-08	9.19E-04	6.47E-06	7.30E-06	9.32E-04	1.31E-03
58.7	4.45E-08	9.19E-04	1.63E-05	8.93E-06	9.44E-04	3.31E-03
47.3	4.45E-08	9.19E-04	9.34E-05	1.92E-05	1.03E-03	1.90E-02
25.8	4.45E-08	9.19E-04	6.39E-06	5.00E-06	9.30E-04	1.30E-03

TABLE 3-52: SUMMARY OF $ADD_{Expected}$ AND EGG CONCENTRATIONS FOR FEMALE MALLARD ON A TEQ BASIS FOR PERIOD 1993 - 2018
REVISED

	Average Dietary Dose (mg/Kg/day)			Average Egg Concentration (mg/Kg)		
Year	189	168	154	189	168	154
1993	9.95E-04	1.33E-04	5.51E-05	2.01E-02	1.66E-02	6.69E-03
1994	8.75E-04	1.24E-04	5.05E-05	1.86E-02	1.55E-02	6.20E-03
1995	7.04E-04	1.08E-04	4.50E-05	1.71E-02	1.39E-02	5.74E-03
1996	4.86E-04	9.10E-05	3.96E-05	1.53E-02	1.19E-02	5.16E-03
1997	4.56E-04	8.05E-05	3.70E-05	1.37E-02	1.02E-02	4.68E-03
1998	5.05E-04	8.79E-05	3.35E-05	8.46E-03	1.12E-02	4.16E-03
1999	4.49E-04	7.65E-05	3.04E-05	7.86E-03	1.01E-02	3.84E-03
2000	2.97E-04	6.64E-05	2.71E-05	7.10E-03	8.73E-03	3.45E-03
2001	2.85E-04	5.51E-05	2.34E-05	6.41E-03	7.37E-03	3.08E-03
2002	2.84E-04	4.80E-05	2.10E-05	5.90E-03	6.35E-03	2.78E-03
2003	3.34E-04	4.64E-05	2.11E-05	5.41E-03	5.81E-03	2.55E-03
2004	3.38E-04	4.34E-05	1.87E-05	5.01E-03	5.52E-03	2.32E-03
2005	2.74E-04	4.09E-05	1.68E-05	4.66E-03	5.11E-03	2.07E-03
2006	2.64E-04	3.58E-05	1.45E-05	4.28E-03	4.62E-03	1.85E-03
2007	2.28E-04	3.29E-05	1.36E-05	3.86E-03	4.24E-03	1.67E-03
2008	2.98E-04	3.21E-05	1.26E-05	3.62E-03	3.98E-03	1.55E-03
2009	2.67E-04	3.12E-05	1.21E-05	3.37E-03	3.79E-03	1.44E-03
2010	1.89E-04	2.68E-05	1.07E-05	3.06E-03	3.44E-03	1.32E-03
2011	1.94E-04	2.47E-05	1.01E-05	2.75E-03	3.10E-03	1.21E-03
2012	1.81E-04	2.27E-05	9.05E-06	2.48E-03	2.81E-03	1.09E-03
2013	1.62E-04	2.23E-05	8.22E-06	2.25E-03	2.87E-03	1.01E-03
2014	1.59E-04	2.43E-05	8.35E-06	2.07E-03	3.03E-03	9.46E-04
2015	1.54E-04	2.37E-05	7.96E-06	1.93E-03	2.96E-03	8.95E-04
2016	2.14E-04	2.32E-05	7.62E-06	1.81E-03	2.89E-03	8.84E-04
2017	2.06E-04	2.27E-05	7.61E-06	1.70E-03	2.81E-03	8.45E-04
2018	2.22E-04	2.36E-05	8.17E-06	1.60E-03	2.70E-03	8.18E-04

TABLE 3-53: SUMMARY OF $ADD_{95\%UCL}$ AND EGG CONCENTRATIONS FOR FEMALE MALLARD ON A TEQ BASIS FOR PERIOD 1993 - 2018

REVISED

Year	95% UCL Dietary Dose (mg/Kg/day)			95% UCL Egg Concentration (mg/Kg)		
	189	168	154	189	168	154
1993	1.06E-03	1.41E-04	5.87E-05	2.13E-02	1.77E-02	7.12E-03
1994	9.34E-04	1.32E-04	5.38E-05	1.96E-02	1.65E-02	6.60E-03
1995	7.50E-04	1.15E-04	4.79E-05	1.81E-02	1.48E-02	6.11E-03
1996	5.15E-04	9.63E-05	4.21E-05	1.62E-02	1.27E-02	5.49E-03
1997	4.83E-04	8.53E-05	3.94E-05	1.44E-02	1.09E-02	4.99E-03
1998	5.41E-04	9.31E-05	3.56E-05	8.93E-03	1.19E-02	4.43E-03
1999	4.81E-04	8.09E-05	3.23E-05	8.30E-03	1.08E-02	4.09E-03
2000	3.17E-04	7.03E-05	2.88E-05	7.51E-03	9.29E-03	3.67E-03
2001	3.04E-04	5.82E-05	2.49E-05	6.77E-03	7.84E-03	3.27E-03
2002	3.03E-04	5.08E-05	2.23E-05	6.24E-03	6.76E-03	2.96E-03
2003	3.58E-04	4.92E-05	2.25E-05	5.72E-03	6.19E-03	2.71E-03
2004	3.62E-04	4.60E-05	1.99E-05	5.30E-03	5.88E-03	2.46E-03
2005	2.93E-04	4.34E-05	1.78E-05	4.92E-03	5.44E-03	2.20E-03
2006	2.83E-04	3.79E-05	1.54E-05	4.53E-03	4.91E-03	1.97E-03
2007	2.45E-04	3.49E-05	1.45E-05	4.08E-03	4.52E-03	1.78E-03
2008	3.20E-04	3.40E-05	1.35E-05	3.82E-03	4.23E-03	1.65E-03
2009	2.86E-04	3.31E-05	1.29E-05	3.56E-03	4.03E-03	1.54E-03
2010	2.03E-04	2.84E-05	1.14E-05	3.23E-03	3.66E-03	1.41E-03
2011	2.09E-04	2.62E-05	1.08E-05	2.90E-03	3.30E-03	1.28E-03
2012	1.94E-04	2.41E-05	9.64E-06	2.62E-03	2.99E-03	1.16E-03
2013	1.74E-04	2.37E-05	8.76E-06	2.38E-03	3.05E-03	1.07E-03
2014	1.70E-04	2.58E-05	8.92E-06	2.19E-03	3.22E-03	1.01E-03
2015	1.65E-04	2.52E-05	8.50E-06	2.04E-03	3.15E-03	9.54E-04
2016	2.30E-04	2.46E-05	8.14E-06	1.92E-03	3.07E-03	9.42E-04
2017	2.21E-04	2.41E-05	8.13E-06	1.80E-03	3.00E-03	9.00E-04
2018	2.39E-04	2.51E-05	8.75E-06	1.69E-03	2.88E-03	8.71E-04

TABLE 3-54: SUMMARY OF $ADD_{Expected}$ AND EGG CONCENTRATIONS FOR FEMALE BELTED KINGFISHER BASED ON 1993 DATA ON TEQ BASIS REVISED

Location	Drinking Water Expected	Forage Fish Expected	Benthic Invertebrate Expected	Sediment Expected	Total Average Daily $Dose_{Expected}$ (mg/Kg/day)	Total Concentration in Eggs (mg/Kg)
Upper River						
Thompson Island Pool (189)	6.54E-08	1.64E-03	5.82E-05	3.75E-05	1.73E-03	8.16E-02
Stillwater (168)	1.16E-07	5.52E-04	1.02E-04	9.79E-05	7.52E-04	3.15E-02
Federal Dam (154)	8.12E-08	1.30E-04	2.43E-05	8.82E-06	1.63E-04	7.41E-03
Lower River						
143.5	6.28E-08	1.51E-04	3.38E-06	2.72E-06	1.57E-04	7.42E-03
137.2	6.28E-08	3.05E-04	6.66E-06	4.79E-06	3.16E-04	1.50E-02
122.4	2.88E-08	1.16E-04	3.68E-06	3.04E-06	1.23E-04	5.78E-03
113.8	2.88E-08	1.22E-04	3.79E-06	3.18E-06	1.29E-04	6.06E-03
100	2.88E-08	5.29E-05	1.74E-06	1.26E-06	5.59E-05	2.63E-03
88.9	1.90E-08	1.05E-04	8.73E-07	2.46E-06	1.09E-04	5.11E-03
58.7	1.90E-08	1.15E-04	2.69E-06	7.95E-07	1.18E-04	5.65E-03
47.3	1.90E-08	1.02E-04	3.05E-06	4.85E-06	1.10E-04	5.06E-03
25.8	1.90E-08	7.67E-05	9.03E-07	1.83E-06	7.94E-05	3.74E-03

TABLE 3-55: SUMMARY OF $ADD_{95\%UCL}$ AND EGG CONCENTRATIONS FOR FEMALE BELTED KINGFISHER BASED ON 1993 DATA ON TEQ BASIS REVISED

Location	Drinking Water 95% UCL	Fish 95% UCL	Benthic Invertebrate 95% UCL	Sediment 95% UCL	Total Upper Bound Daily $Dose_{95\%UCL}$ (mg/Kg/day)	Total Concentration in Eggs (95 % UCL) (mg/Kg)
Upper River						
Thompson Island Pool (189)	2.07E-07	3.34E-03	1.02E-04	5.49E-05	3.50E-03	1.66E-01
Stillwater (168)	3.69E-07	7.90E-04	4.77E-04	1.71E-04	1.44E-03	6.10E-02
Federal Dam (154)	1.74E-07	1.88E-04	4.12E-05	1.48E-05	2.44E-04	1.10E-02
Lower River						
143.5	6.84E-07	1.81E-04	8.29E-06	2.97E-06	1.93E-04	9.11E-03
137.2	6.84E-07	6.61E-04	2.70E-05	9.69E-06	6.98E-04	3.31E-02
122.4	3.69E-07	1.88E-04	9.26E-06	3.38E-06	2.01E-04	9.51E-03
113.8	3.69E-07	1.27E-04	1.47E-05	5.26E-06	1.47E-04	6.80E-03
100	3.69E-07	9.12E-05	1.19E-05	2.72E-05	1.31E-04	4.97E-03
88.9	8.42E-08	1.44E-04	1.55E-06	7.21E-06	1.53E-04	7.02E-03
58.7	8.42E-08	1.30E-04	2.46E-05	8.82E-06	1.63E-04	7.44E-03
47.3	8.42E-08	1.35E-04	2.24E-05	1.89E-05	1.77E-04	7.59E-03
25.8	8.42E-08	9.22E-05	1.53E-06	4.93E-06	9.87E-05	4.51E-03

TABLE 3-56: SUMMARY OF $ADD_{Expected}$ AND EGG CONCENTRATIONS FOR FEMALE BELTED KINGFISHER FOR THE PERIOD 1993 - 2018 ON TEQ BASIS REVISED

	Average Dietary Dose (mg/Kg/day)			Average Egg Concentration (mg/Kg)		
Year	189	168	154	189	168	154
1993	1.85E-03	3.16E-04	1.29E-04	6.04E-02	1.15E-02	5.04E-03
1994	1.81E-03	2.97E-04	1.16E-04	6.16E-02	1.09E-02	4.56E-03
1995	1.67E-03	2.64E-04	1.06E-04	5.58E-02	9.49E-03	4.12E-03
1996	1.12E-03	2.20E-04	9.44E-05	3.20E-02	7.82E-03	3.65E-03
1997	8.19E-04	1.50E-04	6.58E-05	1.99E-02	4.87E-03	2.37E-03
1998	7.70E-04	2.02E-04	7.32E-05	2.54E-02	7.21E-03	2.81E-03
1999	7.22E-04	1.83E-04	6.94E-05	2.36E-02	6.50E-03	2.68E-03
2000	6.34E-04	1.59E-04	6.10E-05	2.04E-02	5.61E-03	2.33E-03
2001	5.71E-04	1.33E-04	5.48E-05	1.85E-02	4.72E-03	2.11E-03
2002	5.62E-04	1.17E-04	4.97E-05	1.87E-02	4.19E-03	1.92E-03
2003	5.39E-04	1.06E-04	4.55E-05	1.83E-02	3.85E-03	1.76E-03
2004	5.40E-04	1.01E-04	4.25E-05	1.91E-02	3.64E-03	1.64E-03
2005	4.87E-04	9.51E-05	3.75E-05	1.70E-02	3.42E-03	1.45E-03
2006	4.51E-04	8.51E-05	3.42E-05	1.57E-02	3.06E-03	1.33E-03
2007	4.33E-04	7.69E-05	3.04E-05	1.54E-02	2.76E-03	1.18E-03
2008	4.03E-04	7.50E-05	2.89E-05	1.44E-02	2.74E-03	1.13E-03
2009	4.15E-04	7.11E-05	2.67E-05	1.52E-02	2.58E-03	1.04E-03
2010	3.86E-04	6.38E-05	2.42E-05	1.42E-02	2.29E-03	9.39E-04
2011	3.29E-04	5.72E-05	2.22E-05	1.19E-02	2.05E-03	8.61E-04
2012	3.17E-04	5.46E-05	2.09E-05	1.18E-02	2.00E-03	8.17E-04
2013	2.81E-04	5.09E-05	1.85E-05	1.04E-02	1.87E-03	7.22E-04
2014	2.70E-04	5.57E-05	1.77E-05	1.01E-02	2.02E-03	6.94E-04
2015	2.50E-04	5.44E-05	1.69E-05	9.36E-03	1.98E-03	6.61E-04
2016	2.65E-04	5.45E-05	1.71E-05	1.02E-02	1.99E-03	6.78E-04
2017	2.67E-04	5.22E-05	1.65E-05	1.05E-02	1.89E-03	6.53E-04
2018	2.86E-04	5.31E-05	1.70E-05	1.16E-02	1.97E-03	6.79E-04

TABLE 3-57: SUMMARY OF $ADD_{95\%UCL}$ AND EGG CONCENTRATIONS FOR FEMALE BELTED KINGFISHER FOR THE PERIOD 1993 - 2018 ON TEQ BASIS REVISED

Year	95% UCL Dietary Dose (mg/Kg/day)			95% UCL Egg Concentration (mg/Kg)		
	189	168	154	189	168	154
1993	1.87E-03	3.29E-04	1.35E-04	6.17E-02	1.21E-02	5.33E-03
1994	1.84E-03	3.10E-04	1.22E-04	6.29E-02	1.15E-02	4.83E-03
1995	1.69E-03	2.76E-04	1.11E-04	5.69E-02	1.01E-02	4.37E-03
1996	1.13E-03	2.29E-04	9.90E-05	3.27E-02	8.28E-03	3.87E-03
1997	8.28E-04	1.57E-04	6.94E-05	2.03E-02	5.23E-03	2.54E-03
1998	7.81E-04	2.11E-04	7.65E-05	2.59E-02	7.64E-03	2.97E-03
1999	7.31E-04	1.91E-04	7.26E-05	2.41E-02	6.88E-03	2.84E-03
2000	6.42E-04	1.65E-04	6.37E-05	2.08E-02	5.92E-03	2.46E-03
2001	5.78E-04	1.39E-04	5.73E-05	1.89E-02	4.99E-03	2.23E-03
2002	5.69E-04	1.22E-04	5.20E-05	1.91E-02	4.43E-03	2.03E-03
2003	5.48E-04	1.11E-04	4.75E-05	1.88E-02	4.08E-03	1.85E-03
2004	5.48E-04	1.05E-04	4.45E-05	1.94E-02	3.86E-03	1.74E-03
2005	4.94E-04	9.91E-05	3.93E-05	1.73E-02	3.61E-03	1.53E-03
2006	4.58E-04	8.87E-05	3.58E-05	1.60E-02	3.23E-03	1.40E-03
2007	4.39E-04	8.02E-05	3.18E-05	1.57E-02	2.92E-03	1.24E-03
2008	4.09E-04	7.84E-05	3.03E-05	1.47E-02	2.90E-03	1.20E-03
2009	4.21E-04	7.41E-05	2.79E-05	1.55E-02	2.72E-03	1.10E-03
2010	3.91E-04	6.66E-05	2.54E-05	1.45E-02	2.43E-03	9.93E-04
2011	3.33E-04	5.96E-05	2.32E-05	1.22E-02	2.17E-03	9.10E-04
2012	3.21E-04	5.70E-05	2.19E-05	1.20E-02	2.11E-03	8.63E-04
2013	2.86E-04	5.31E-05	1.94E-05	1.06E-02	1.98E-03	7.63E-04
2014	2.73E-04	5.81E-05	1.85E-05	1.03E-02	2.14E-03	7.31E-04
2015	2.53E-04	5.67E-05	1.76E-05	9.52E-03	2.09E-03	6.97E-04
2016	2.69E-04	5.69E-05	1.79E-05	1.05E-02	2.11E-03	7.17E-04
2017	2.71E-04	5.44E-05	1.73E-05	1.07E-02	2.00E-03	6.90E-04
2018	2.91E-04	5.53E-05	1.77E-05	1.18E-02	2.08E-03	7.17E-04

TABLE 3-60: SUMMARY OF $ADD_{Expected}$ AND EGG CONCENTRATIONS FOR FEMALE GREAT BLUE HERON FOR THE PERIOD 1993 - 2018 ON TEQ BASIS REVISED

	Average Dietary Dose (mg/Kg/day)			Average Egg Concentration (mg/Kg)		
Year	189	168	154	189	168	154
1993	7.41E-04	1.14E-04	4.86E-05	7.44E-02	1.17E-02	5.25E-03
1994	7.43E-04	1.08E-04	4.37E-05	7.62E-02	1.12E-02	4.71E-03
1995	6.77E-04	9.43E-05	3.94E-05	6.90E-02	9.63E-03	4.23E-03
1996	4.17E-04	7.76E-05	3.50E-05	3.85E-02	7.85E-03	3.74E-03
1997	2.81E-04	4.70E-05	2.17E-05	2.32E-02	4.36E-03	2.17E-03
1998	3.10E-04	7.11E-05	2.68E-05	3.12E-02	7.20E-03	2.85E-03
1999	2.89E-04	6.42E-05	2.56E-05	2.91E-02	6.49E-03	2.74E-03
2000	2.52E-04	5.56E-05	2.23E-05	2.51E-02	5.59E-03	2.36E-03
2001	2.28E-04	4.66E-05	2.01E-05	2.27E-02	4.70E-03	2.14E-03
2002	2.28E-04	4.15E-05	1.83E-05	2.31E-02	4.21E-03	1.95E-03
2003	2.21E-04	3.79E-05	1.67E-05	2.27E-02	3.88E-03	1.79E-03
2004	2.26E-04	3.59E-05	1.58E-05	2.37E-02	3.66E-03	1.69E-03
2005	2.02E-04	3.38E-05	1.39E-05	2.11E-02	3.45E-03	1.48E-03
2006	1.87E-04	3.02E-05	1.27E-05	1.95E-02	3.08E-03	1.36E-03
2007	1.82E-04	2.72E-05	1.12E-05	1.92E-02	2.76E-03	1.20E-03
2008	1.70E-04	2.70E-05	1.08E-05	1.80E-02	2.78E-03	1.16E-03
2009	1.78E-04	2.55E-05	9.96E-06	1.91E-02	2.61E-03	1.07E-03
2010	1.66E-04	2.27E-05	9.00E-06	1.79E-02	2.31E-03	9.64E-04
2011	1.40E-04	2.03E-05	8.24E-06	1.49E-02	2.07E-03	8.84E-04
2012	1.37E-04	1.98E-05	7.87E-06	1.48E-02	2.05E-03	8.50E-04
2013	1.21E-04	1.82E-05	6.90E-06	1.30E-02	1.88E-03	7.43E-04
2014	1.17E-04	1.99E-05	6.65E-06	1.27E-02	2.04E-03	7.17E-04
2015	1.08E-04	1.94E-05	6.34E-06	1.17E-02	1.99E-03	6.85E-04
2016	1.17E-04	1.96E-05	6.51E-06	1.29E-02	2.03E-03	7.09E-04
2017	1.19E-04	1.86E-05	6.30E-06	1.32E-02	1.91E-03	6.84E-04
2018	1.30E-04	1.95E-05	6.59E-06	1.47E-02	2.03E-03	7.23E-04

TABLE 3-61: SUMMARY OF $ADD_{95\%UCL}$ AND EGG CONCENTRATIONS FOR FEMALE GREAT BLUE HERON FOR THE PERIOD 1993 - 2018 ON TEQ BASIS REVISED

Year	95% UCL Dietary Dose (mg/Kg/day)			95% UCL Egg Concentration (mg/Kg)		
	189	168	154	189	168	154
1993	7.52E-04	1.19E-04	5.10E-05	7.58E-02	1.23E-02	5.54E-03
1994	7.54E-04	1.13E-04	4.59E-05	7.77E-02	1.18E-02	4.99E-03
1995	6.88E-04	9.91E-05	4.15E-05	7.03E-02	1.02E-02	4.49E-03
1996	4.23E-04	8.12E-05	3.68E-05	3.93E-02	8.29E-03	3.96E-03
1997	2.84E-04	4.97E-05	2.31E-05	2.36E-02	4.69E-03	2.33E-03
1998	3.14E-04	7.44E-05	2.81E-05	3.18E-02	7.60E-03	3.00E-03
1999	2.94E-04	6.72E-05	2.69E-05	2.96E-02	6.85E-03	2.89E-03
2000	2.55E-04	5.80E-05	2.33E-05	2.55E-02	5.89E-03	2.49E-03
2001	2.31E-04	4.88E-05	2.11E-05	2.31E-02	4.97E-03	2.26E-03
2002	2.31E-04	4.34E-05	1.92E-05	2.35E-02	4.44E-03	2.06E-03
2003	2.25E-04	3.97E-05	1.75E-05	2.32E-02	4.09E-03	1.88E-03
2004	2.29E-04	3.76E-05	1.65E-05	2.41E-02	3.87E-03	1.78E-03
2005	2.05E-04	3.54E-05	1.45E-05	2.14E-02	3.64E-03	1.56E-03
2006	1.90E-04	3.16E-05	1.34E-05	1.98E-02	3.25E-03	1.44E-03
2007	1.85E-04	2.84E-05	1.18E-05	1.95E-02	2.92E-03	1.27E-03
2008	1.73E-04	2.83E-05	1.14E-05	1.84E-02	2.94E-03	1.23E-03
2009	1.81E-04	2.66E-05	1.04E-05	1.94E-02	2.75E-03	1.13E-03
2010	1.68E-04	2.37E-05	9.44E-06	1.81E-02	2.44E-03	1.02E-03
2011	1.42E-04	2.12E-05	8.65E-06	1.52E-02	2.18E-03	9.32E-04
2012	1.39E-04	2.07E-05	8.26E-06	1.50E-02	2.16E-03	8.96E-04
2013	1.23E-04	1.90E-05	7.24E-06	1.32E-02	1.98E-03	7.83E-04
2014	1.19E-04	2.08E-05	6.96E-06	1.29E-02	2.15E-03	7.54E-04
2015	1.10E-04	2.03E-05	6.63E-06	1.19E-02	2.10E-03	7.20E-04
2016	1.19E-04	2.06E-05	6.84E-06	1.32E-02	2.14E-03	7.49E-04
2017	1.21E-04	1.95E-05	6.60E-06	1.35E-02	2.02E-03	7.21E-04
2018	1.33E-04	2.03E-05	6.90E-06	1.50E-02	2.14E-03	7.61E-04

TABLE 3-64: SUMMARY OF $ADD_{Expected}$ AND EGG CONCENTRATIONS FOR FEMALE EAGLE FOR THE PERIOD 1993 - 2018 ON TEQ BASIS REVISED

	Average Dietary Dose (mg/Kg/day)			Average Egg Concentration (mg/Kg)		
Year	189	168	154	189	168	154
1993	4.31E-04	1.25E-04	6.15E-05	6.43E-02	1.86E-02	9.16E-03
1994	3.96E-04	1.13E-04	5.60E-05	5.90E-02	1.68E-02	8.35E-03
1995	4.52E-04	1.12E-04	5.39E-05	6.74E-02	1.67E-02	8.03E-03
1996	3.96E-04	1.03E-04	4.76E-05	5.91E-02	1.53E-02	7.09E-03
1997	3.23E-04	9.18E-05	4.24E-05	4.82E-02	1.37E-02	6.32E-03
1998	3.40E-04	8.62E-05	3.93E-05	5.07E-02	1.29E-02	5.85E-03
1999	3.50E-04	8.12E-05	3.62E-05	5.21E-02	1.21E-02	5.39E-03
2000	3.58E-04	7.96E-05	3.40E-05	5.33E-02	1.19E-02	5.07E-03
2001	3.00E-04	7.13E-05	3.13E-05	4.48E-02	1.06E-02	4.66E-03
2002	2.69E-04	6.36E-05	2.86E-05	4.02E-02	9.48E-03	4.26E-03
2003	2.84E-04	5.92E-05	2.60E-05	4.23E-02	8.82E-03	3.88E-03
2004	2.33E-04	5.54E-05	2.28E-05	3.48E-02	8.26E-03	3.40E-03
2005	2.11E-04	5.73E-05	2.20E-05	3.14E-02	8.54E-03	3.29E-03
2006	1.98E-04	5.70E-05	2.13E-05	2.96E-02	8.49E-03	3.17E-03
2007	2.22E-04	5.67E-05	2.07E-05	3.31E-02	8.46E-03	3.09E-03
2008	2.47E-04	5.60E-05	2.05E-05	3.68E-02	8.35E-03	3.06E-03
2009	2.34E-04	5.40E-05	2.00E-05	3.49E-02	8.04E-03	2.98E-03
2010	2.41E-04	5.23E-05	1.87E-05	3.59E-02	7.80E-03	2.79E-03
2011	2.10E-04	5.03E-05	1.82E-05	3.13E-02	7.50E-03	2.72E-03
2012	1.81E-04	4.76E-05	1.78E-05	2.71E-02	7.10E-03	2.65E-03
2013	1.77E-04	4.44E-05	1.72E-05	2.64E-02	6.62E-03	2.56E-03
2014	2.03E-04	4.54E-05	1.75E-05	3.03E-02	6.77E-03	2.61E-03
2015	1.94E-04	4.37E-05	1.67E-05	2.89E-02	6.52E-03	2.49E-03
2016	1.73E-04	3.88E-05	1.50E-05	2.59E-02	5.78E-03	2.23E-03
2017	1.55E-04	3.71E-05	1.51E-05	2.31E-02	5.52E-03	2.25E-03
2018	1.83E-04	3.86E-05	1.53E-05	2.73E-02	5.75E-03	2.28E-03

TABLE 3-65: SUMMARY OF $ADD_{95\%UCL}$ AND EGG CONCENTRATIONS FOR FEMALE EAGLE FOR THE PERIOD 1993 - 2018 ON TEQ BASIS REVISED

Year	95% UCL Dietary Dose (mg/Kg/day)			95% UCL Egg Concentration (mg/Kg)		
	189	168	154	189	168	154
1993	4.50E-04	1.27E-04	6.24E-05	6.71E-02	1.89E-02	9.30E-03
1994	4.14E-04	1.14E-04	5.69E-05	6.17E-02	1.70E-02	8.48E-03
1995	4.71E-04	1.14E-04	5.47E-05	7.03E-02	1.70E-02	8.15E-03
1996	4.16E-04	1.04E-04	4.83E-05	6.20E-02	1.56E-02	7.20E-03
1997	3.39E-04	9.31E-05	4.30E-05	5.05E-02	1.39E-02	6.41E-03
1998	3.56E-04	8.75E-05	3.99E-05	5.30E-02	1.30E-02	5.94E-03
1999	3.66E-04	8.24E-05	3.67E-05	5.45E-02	1.23E-02	5.47E-03
2000	3.74E-04	8.07E-05	3.46E-05	5.58E-02	1.20E-02	5.15E-03
2001	3.15E-04	7.23E-05	3.18E-05	4.70E-02	1.08E-02	4.73E-03
2002	2.82E-04	6.45E-05	2.91E-05	4.20E-02	9.61E-03	4.33E-03
2003	2.96E-04	6.00E-05	2.64E-05	4.42E-02	8.95E-03	3.93E-03
2004	2.45E-04	5.62E-05	2.32E-05	3.65E-02	8.38E-03	3.45E-03
2005	2.21E-04	5.81E-05	2.24E-05	3.30E-02	8.66E-03	3.34E-03
2006	2.08E-04	5.78E-05	2.16E-05	3.09E-02	8.61E-03	3.22E-03
2007	2.33E-04	5.76E-05	2.10E-05	3.47E-02	8.58E-03	3.13E-03
2008	2.58E-04	5.68E-05	2.08E-05	3.85E-02	8.47E-03	3.10E-03
2009	2.45E-04	5.47E-05	2.03E-05	3.65E-02	8.16E-03	3.02E-03
2010	2.51E-04	5.31E-05	1.90E-05	3.74E-02	7.91E-03	2.83E-03
2011	2.20E-04	5.10E-05	1.85E-05	3.29E-02	7.61E-03	2.76E-03
2012	1.90E-04	4.83E-05	1.80E-05	2.83E-02	7.20E-03	2.69E-03
2013	1.85E-04	4.51E-05	1.75E-05	2.76E-02	6.72E-03	2.60E-03
2014	2.12E-04	4.61E-05	1.78E-05	3.17E-02	6.87E-03	2.65E-03
2015	2.03E-04	4.44E-05	1.69E-05	3.02E-02	6.61E-03	2.52E-03
2016	1.82E-04	3.93E-05	1.52E-05	2.71E-02	5.86E-03	2.27E-03
2017	1.62E-04	3.76E-05	1.53E-05	2.42E-02	5.60E-03	2.29E-03
2018	1.91E-04	3.91E-05	1.55E-05	2.85E-02	5.83E-03	2.31E-03

TABLE 3-68: SUMMARY OF $ADD_{Expected}$ FOR FEMALE BAT BASED ON TRI+ PREDICTIONS FOR THE PERIOD 1993 - 2018 REVISED

	Total Average Dietary Dose (mg/Kg/day)		
Year	189	168	154
1993	4.75E+00	3.92E+00	1.58E+00
1994	4.39E+00	3.65E+00	1.46E+00
1995	4.05E+00	3.28E+00	1.35E+00
1996	3.62E+00	2.81E+00	1.22E+00
1997	3.23E+00	2.41E+00	1.11E+00
1998	2.00E+00	2.65E+00	9.83E-01
1999	1.85E+00	2.39E+00	9.07E-01
2000	1.68E+00	2.06E+00	8.15E-01
2001	1.51E+00	1.74E+00	7.26E-01
2002	1.39E+00	1.50E+00	6.56E-01
2003	1.28E+00	1.37E+00	6.02E-01
2004	1.18E+00	1.30E+00	5.46E-01
2005	1.10E+00	1.21E+00	4.88E-01
2006	1.01E+00	1.09E+00	4.36E-01
2007	9.12E-01	1.00E+00	3.95E-01
2008	8.54E-01	9.38E-01	3.67E-01
2009	7.95E-01	8.95E-01	3.41E-01
2010	7.21E-01	8.12E-01	3.12E-01
2011	6.48E-01	7.31E-01	2.85E-01
2012	5.85E-01	6.64E-01	2.57E-01
2013	5.32E-01	6.77E-01	2.37E-01
2014	4.89E-01	7.15E-01	2.23E-01
2015	4.57E-01	6.99E-01	2.11E-01
2016	4.28E-01	6.81E-01	2.09E-01
2017	4.01E-01	6.64E-01	2.00E-01
2018	3.78E-01	6.38E-01	1.93E-01

TABLE 3-69: SUMMARY OF $ADD_{95\%UCL}$ FOR FEMALE BAT BASED ON TRI+ PREDICTIONS FOR THE PERIOD 1993 - 2018 REVISED

Year	Total 95% UCL Dietary Dose (mg/Kg/day)		
	189	168	154
1993	5.02E+00	4.17E+00	1.68E+00
1994	4.64E+00	3.88E+00	1.56E+00
1995	4.28E+00	3.49E+00	1.44E+00
1996	3.83E+00	2.99E+00	1.30E+00
1997	3.41E+00	2.56E+00	1.18E+00
1998	2.11E+00	2.82E+00	1.05E+00
1999	1.96E+00	2.54E+00	9.66E-01
2000	1.77E+00	2.19E+00	8.67E-01
2001	1.60E+00	1.85E+00	7.73E-01
2002	1.47E+00	1.60E+00	6.98E-01
2003	1.35E+00	1.46E+00	6.40E-01
2004	1.25E+00	1.39E+00	5.81E-01
2005	1.16E+00	1.28E+00	5.19E-01
2006	1.07E+00	1.16E+00	4.64E-01
2007	9.64E-01	1.07E+00	4.20E-01
2008	9.02E-01	1.00E+00	3.90E-01
2009	8.40E-01	9.52E-01	3.63E-01
2010	7.62E-01	8.64E-01	3.32E-01
2011	6.85E-01	7.78E-01	3.03E-01
2012	6.19E-01	7.06E-01	2.73E-01
2013	5.62E-01	7.20E-01	2.53E-01
2014	5.16E-01	7.61E-01	2.38E-01
2015	4.83E-01	7.45E-01	2.25E-01
2016	4.52E-01	7.25E-01	2.22E-01
2017	4.24E-01	7.07E-01	2.13E-01
2018	4.00E-01	6.79E-01	2.06E-01

TABLE 3-72: SUMMARY OF $ADD_{Expected}$ FOR FEMALE RACCOON BASED ON TRI+ PREDICTIONS FOR THE PERIOD 1993 - 2018 REVISED

	Total Average Dietary Dose (mg/Kg/day)		
Year	189	168	154
1993	1.09E+00	6.81E-01	2.72E-01
1994	1.01E+00	6.34E-01	2.52E-01
1995	9.38E-01	5.71E-01	2.33E-01
1996	8.18E-01	4.89E-01	2.09E-01
1997	7.19E-01	4.16E-01	1.89E-01
1998	4.57E-01	4.59E-01	1.69E-01
1999	4.27E-01	4.14E-01	1.56E-01
2000	3.85E-01	3.58E-01	1.40E-01
2001	3.47E-01	3.02E-01	1.25E-01
2002	3.22E-01	2.60E-01	1.13E-01
2003	2.96E-01	2.38E-01	1.03E-01
2004	2.76E-01	2.26E-01	9.40E-02
2005	2.56E-01	2.09E-01	8.39E-02
2006	2.36E-01	1.89E-01	7.51E-02
2007	2.15E-01	1.74E-01	6.78E-02
2008	2.00E-01	1.63E-01	6.30E-02
2009	1.89E-01	1.55E-01	5.86E-02
2010	1.72E-01	1.41E-01	5.36E-02
2011	1.54E-01	1.27E-01	4.89E-02
2012	1.40E-01	1.15E-01	4.42E-02
2013	1.27E-01	1.17E-01	4.08E-02
2014	1.17E-01	1.24E-01	3.84E-02
2015	1.09E-01	1.21E-01	3.63E-02
2016	1.04E-01	1.18E-01	3.59E-02
2017	9.86E-02	1.15E-01	3.43E-02
2018	9.48E-02	1.11E-01	3.33E-02

TABLE 3-73: SUMMARY OF $ADD_{95\%UCL}$ FOR FEMALE RACCOON BASED ON TRI+ PREDICTIONS FOR THE PERIOD 1993 - 2018

REVISED

Year	Total 95% UCL Dietary Dose		
	189	168	154
1993	1.14E+00	7.22E-01	2.89E-01
1994	1.05E+00	6.73E-01	2.67E-01
1995	9.77E-01	6.06E-01	2.47E-01
1996	8.52E-01	5.19E-01	2.22E-01
1997	7.48E-01	4.42E-01	2.01E-01
1998	4.76E-01	4.88E-01	1.79E-01
1999	4.45E-01	4.39E-01	1.66E-01
2000	4.01E-01	3.80E-01	1.49E-01
2001	3.61E-01	3.20E-01	1.32E-01
2002	3.35E-01	2.76E-01	1.20E-01
2003	3.08E-01	2.53E-01	1.10E-01
2004	2.87E-01	2.40E-01	9.97E-02
2005	2.66E-01	2.22E-01	8.90E-02
2006	2.46E-01	2.01E-01	7.97E-02
2007	2.23E-01	1.84E-01	7.20E-02
2008	2.08E-01	1.73E-01	6.69E-02
2009	1.97E-01	1.65E-01	6.22E-02
2010	1.79E-01	1.50E-01	5.69E-02
2011	1.60E-01	1.35E-01	5.19E-02
2012	1.46E-01	1.22E-01	4.69E-02
2013	1.32E-01	1.24E-01	4.33E-02
2014	1.22E-01	1.32E-01	4.08E-02
2015	1.13E-01	1.29E-01	3.86E-02
2016	1.08E-01	1.25E-01	3.81E-02
2017	1.03E-01	1.22E-01	3.65E-02
2018	9.86E-02	1.18E-01	3.54E-02

TABLE 3-76: SUMMARY OF $ADD_{Expected}$ FOR FEMALE MINK BASED ON TRI+ PREDICTIONS FOR THE PERIOD 1993 - 2018 REVISED

	Average Dietary Dose (mg/Kg/day)		
Year	189	168	154
1993	1.22E+00	4.27E-01	1.77E-01
1994	1.21E+00	4.00E-01	1.63E-01
1995	1.10E+00	3.56E-01	1.49E-01
1996	7.26E-01	3.00E-01	1.33E-01
1997	5.24E-01	2.31E-01	1.08E-01
1998	5.11E-01	2.80E-01	1.06E-01
1999	4.76E-01	2.53E-01	9.89E-02
2000	4.17E-01	2.18E-01	8.77E-02
2001	3.77E-01	1.84E-01	7.86E-02
2002	3.72E-01	1.60E-01	7.11E-02
2003	3.57E-01	1.47E-01	6.52E-02
2004	3.61E-01	1.39E-01	6.00E-02
2005	3.24E-01	1.30E-01	5.33E-02
2006	2.99E-01	1.17E-01	4.81E-02
2007	2.88E-01	1.07E-01	4.31E-02
2008	2.70E-01	1.02E-01	4.06E-02
2009	2.77E-01	9.68E-02	3.76E-02
2010	2.58E-01	8.72E-02	3.42E-02
2011	2.19E-01	7.84E-02	3.13E-02
2012	2.12E-01	7.30E-02	2.88E-02
2013	1.88E-01	7.20E-02	2.62E-02
2014	1.81E-01	7.68E-02	2.48E-02
2015	1.68E-01	7.51E-02	2.36E-02
2016	1.79E-01	7.41E-02	2.36E-02
2017	1.80E-01	7.15E-02	2.27E-02
2018	1.94E-01	7.09E-02	2.26E-02

TABLE 3-77: SUMMARY OF $ADD_{95\%UCL}$ FOR FEMALE MINK BASED ON TRI+ PREDICTIONS FOR THE PERIOD 1993 - 2018 REVISED

Year	95% UCL Dietary Dose (mg/Kg/day)		
	189	168	154
1993	1.25E+00	4.53E-01	1.88E-01
1994	1.24E+00	4.25E-01	1.73E-01
1995	1.13E+00	3.78E-01	1.59E-01
1996	7.51E-01	3.19E-01	1.42E-01
1997	5.41E-01	2.46E-01	1.15E-01
1998	5.26E-01	2.98E-01	1.12E-01
1999	4.89E-01	2.68E-01	1.05E-01
2000	4.28E-01	2.31E-01	9.30E-02
2001	3.88E-01	1.95E-01	8.34E-02
2002	3.82E-01	1.70E-01	7.55E-02
2003	3.69E-01	1.56E-01	6.92E-02
2004	3.70E-01	1.48E-01	6.36E-02
2005	3.33E-01	1.38E-01	5.65E-02
2006	3.08E-01	1.24E-01	5.10E-02
2007	2.95E-01	1.13E-01	4.58E-02
2008	2.77E-01	1.08E-01	4.31E-02
2009	2.84E-01	1.03E-01	3.99E-02
2010	2.64E-01	9.25E-02	3.63E-02
2011	2.25E-01	8.31E-02	3.32E-02
2012	2.17E-01	7.75E-02	3.06E-02
2013	1.93E-01	7.64E-02	2.77E-02
2014	1.85E-01	8.14E-02	2.63E-02
2015	1.72E-01	7.97E-02	2.50E-02
2016	1.83E-01	7.87E-02	2.51E-02
2017	1.84E-01	7.59E-02	2.40E-02
2018	1.99E-01	7.51E-02	2.40E-02

TABLE 3-80: SUMMARY OF $ADD_{Expected}$ FOR FEMALE OTTER BASED ON TRI+ PREDICTIONS FOR THE PERIOD 1993 - 2018 REVISED

	Total Average Dietary Dose (mg/Kg/day)		
Year	189	168	154
1993	1.65E+00	4.72E-01	2.32E-01
1994	1.51E+00	4.27E-01	2.11E-01
1995	1.72E+00	4.25E-01	2.03E-01
1996	1.51E+00	3.89E-01	1.80E-01
1997	1.23E+00	3.47E-01	1.60E-01
1998	1.29E+00	3.26E-01	1.48E-01
1999	1.32E+00	3.07E-01	1.37E-01
2000	1.35E+00	3.01E-01	1.28E-01
2001	1.14E+00	2.69E-01	1.18E-01
2002	1.02E+00	2.40E-01	1.08E-01
2003	1.07E+00	2.24E-01	9.82E-02
2004	8.84E-01	2.09E-01	8.61E-02
2005	7.99E-01	2.16E-01	8.32E-02
2006	7.51E-01	2.15E-01	8.02E-02
2007	8.40E-01	2.14E-01	7.81E-02
2008	9.33E-01	2.11E-01	7.73E-02
2009	8.84E-01	2.04E-01	7.53E-02
2010	9.08E-01	1.97E-01	7.05E-02
2011	7.92E-01	1.90E-01	6.87E-02
2012	6.85E-01	1.80E-01	6.71E-02
2013	6.69E-01	1.68E-01	6.48E-02
2014	7.66E-01	1.71E-01	6.60E-02
2015	7.30E-01	1.65E-01	6.28E-02
2016	6.54E-01	1.46E-01	5.65E-02
2017	5.84E-01	1.40E-01	5.69E-02
2018	6.90E-01	1.45E-01	5.75E-02

TABLE 3-81: SUMMARY OF $ADD_{95\%UCL}$ FOR FEMALE OTTER BASED ON TRI+ PREDICTIONS FOR THE PERIOD 1993 - 2018 REVISED

Year	Total 95% UCL Dietary Dose (mg/Kg/day)		
	189	168	154
1993	1.72E+00	4.79E-01	2.35E-01
1994	1.58E+00	4.33E-01	2.15E-01
1995	1.79E+00	4.31E-01	2.06E-01
1996	1.58E+00	3.95E-01	1.82E-01
1997	1.29E+00	3.52E-01	1.62E-01
1998	1.35E+00	3.31E-01	1.51E-01
1999	1.39E+00	3.12E-01	1.39E-01
2000	1.42E+00	3.05E-01	1.30E-01
2001	1.19E+00	2.73E-01	1.20E-01
2002	1.07E+00	2.44E-01	1.10E-01
2003	1.12E+00	2.27E-01	9.95E-02
2004	9.26E-01	2.12E-01	8.74E-02
2005	8.38E-01	2.19E-01	8.44E-02
2006	7.86E-01	2.18E-01	8.15E-02
2007	8.79E-01	2.17E-01	7.92E-02
2008	9.74E-01	2.14E-01	7.85E-02
2009	9.25E-01	2.06E-01	7.65E-02
2010	9.47E-01	2.00E-01	7.16E-02
2011	8.32E-01	1.92E-01	6.97E-02
2012	7.16E-01	1.82E-01	6.80E-02
2013	6.99E-01	1.70E-01	6.58E-02
2014	8.01E-01	1.74E-01	6.70E-02
2015	7.64E-01	1.67E-01	6.37E-02
2016	6.85E-01	1.48E-01	5.73E-02
2017	6.12E-01	1.42E-01	5.78E-02
2018	7.20E-01	1.47E-01	5.84E-02

TABLE 3-84: SUMMARY OF $ADD_{Expected}$ FOR FEMALE BAT ON A TEQ BASIS FOR THE PERIOD 1993 - 2018

REVISED

	Total Average Dietary Dose (mg/Kg/day)		
Year	189	168	154
1993	1.41E-04	1.16E-04	4.69E-05
1994	1.30E-04	1.08E-04	4.35E-05
1995	1.20E-04	9.76E-05	4.03E-05
1996	1.08E-04	8.36E-05	3.62E-05
1997	9.60E-05	7.16E-05	3.29E-05
1998	5.94E-05	7.86E-05	2.92E-05
1999	5.52E-05	7.08E-05	2.70E-05
2000	4.98E-05	6.12E-05	2.42E-05
2001	4.50E-05	5.17E-05	2.16E-05
2002	4.14E-05	4.45E-05	1.95E-05
2003	3.80E-05	4.08E-05	1.79E-05
2004	3.52E-05	3.87E-05	1.62E-05
2005	3.27E-05	3.58E-05	1.45E-05
2006	3.00E-05	3.24E-05	1.30E-05
2007	2.71E-05	2.98E-05	1.17E-05
2008	2.54E-05	2.79E-05	1.09E-05
2009	2.37E-05	2.66E-05	1.01E-05
2010	2.15E-05	2.41E-05	9.27E-06
2011	1.93E-05	2.17E-05	8.46E-06
2012	1.74E-05	1.97E-05	7.63E-06
2013	1.58E-05	2.01E-05	7.06E-06
2014	1.45E-05	2.12E-05	6.64E-06
2015	1.36E-05	2.08E-05	6.28E-06
2016	1.28E-05	2.02E-05	6.20E-06
2017	1.20E-05	1.97E-05	5.93E-06
2018	1.13E-05	1.90E-05	5.74E-06

TABLE 3-85: SUMMARY OF $ADD_{95\%UCL}$ FOR FEMALE BAT ON A TEQ BASIS FOR THE PERIOD 1993 - 2018 REVISED

Year	Total 95% UCL Dietary Dose (mg/Kg/day)		
	189	168	154
1993	1.49E-04	1.24E-04	4.99E-05
1994	1.38E-04	1.15E-04	4.63E-05
1995	1.27E-04	1.04E-04	4.29E-05
1996	1.14E-04	8.89E-05	3.85E-05
1997	1.01E-04	7.61E-05	3.50E-05
1998	6.27E-05	8.37E-05	3.11E-05
1999	5.83E-05	7.54E-05	2.87E-05
2000	5.27E-05	6.51E-05	2.58E-05
2001	4.75E-05	5.50E-05	2.30E-05
2002	4.38E-05	4.74E-05	2.07E-05
2003	4.01E-05	4.34E-05	1.90E-05
2004	3.72E-05	4.12E-05	1.73E-05
2005	3.46E-05	3.81E-05	1.54E-05
2006	3.18E-05	3.44E-05	1.38E-05
2007	2.87E-05	3.17E-05	1.25E-05
2008	2.68E-05	2.97E-05	1.16E-05
2009	2.50E-05	2.83E-05	1.08E-05
2010	2.27E-05	2.57E-05	9.86E-06
2011	2.04E-05	2.31E-05	9.00E-06
2012	1.84E-05	2.10E-05	8.12E-06
2013	1.67E-05	2.14E-05	7.51E-06
2014	1.54E-05	2.26E-05	7.07E-06
2015	1.44E-05	2.21E-05	6.69E-06
2016	1.35E-05	2.15E-05	6.61E-06
2017	1.26E-05	2.10E-05	6.32E-06
2018	1.19E-05	2.02E-05	6.11E-06

TABLE 3-88: SUMMARY OF $ADD_{Expected}$ FOR FEMALE RACCOON ON A TEQ BASIS FOR THE PERIOD 1993 - 2018 REVISED

	Total Average Dietary Dose (mg/Kg/day)		
Year	189	168	154
1993	2.21E-04	4.49E-05	1.57E-05
1994	2.01E-04	4.14E-05	1.45E-05
1995	1.89E-04	3.83E-05	1.35E-05
1996	1.69E-04	3.29E-05	1.22E-05
1997	1.50E-04	2.78E-05	1.09E-05
1998	9.09E-05	3.03E-05	9.73E-06
1999	8.63E-05	2.75E-05	9.02E-06
2000	7.84E-05	2.41E-05	8.15E-06
2001	6.98E-05	2.02E-05	7.24E-06
2002	6.46E-05	1.74E-05	6.52E-06
2003	5.91E-05	1.55E-05	5.96E-06
2004	5.44E-05	1.48E-05	5.47E-06
2005	5.07E-05	1.39E-05	4.87E-06
2006	4.72E-05	1.25E-05	4.36E-06
2007	4.27E-05	1.14E-05	3.92E-06
2008	3.90E-05	1.06E-05	3.62E-06
2009	3.71E-05	1.02E-05	3.38E-06
2010	3.39E-05	9.35E-06	3.10E-06
2011	3.03E-05	8.39E-06	2.83E-06
2012	2.75E-05	7.62E-06	2.57E-06
2013	2.49E-05	7.28E-06	2.34E-06
2014	2.27E-05	8.05E-06	2.21E-06
2015	2.10E-05	7.84E-06	2.09E-06
2016	1.98E-05	7.69E-06	2.03E-06
2017	1.87E-05	7.52E-06	1.97E-06
2018	1.75E-05	7.18E-06	1.90E-06

TABLE 3-89: SUMMARY OF $ADD_{95\%UCL}$ FOR FEMALE RACCOON ON A TEQ BASIS FOR THE PERIOD 1993 - 2018

REVISED

Year	Total 95% UCL Dietary Dose (mg/Kg/day)		
	189	168	154
1993	2.23E-04	4.61E-05	1.62E-05
1994	2.02E-04	4.26E-05	1.49E-05
1995	1.90E-04	3.93E-05	1.39E-05
1996	1.70E-04	3.38E-05	1.26E-05
1997	1.51E-04	2.86E-05	1.13E-05
1998	9.15E-05	3.12E-05	1.00E-05
1999	8.69E-05	2.83E-05	9.31E-06
2000	7.88E-05	2.48E-05	8.41E-06
2001	7.02E-05	2.07E-05	7.47E-06
2002	6.50E-05	1.78E-05	6.73E-06
2003	5.95E-05	1.59E-05	6.15E-06
2004	5.47E-05	1.52E-05	5.64E-06
2005	5.10E-05	1.43E-05	5.03E-06
2006	4.75E-05	1.28E-05	4.50E-06
2007	4.29E-05	1.17E-05	4.04E-06
2008	3.92E-05	1.09E-05	3.74E-06
2009	3.73E-05	1.05E-05	3.49E-06
2010	3.41E-05	9.61E-06	3.20E-06
2011	3.05E-05	8.62E-06	2.92E-06
2012	2.77E-05	7.83E-06	2.65E-06
2013	2.50E-05	7.49E-06	2.41E-06
2014	2.29E-05	8.28E-06	2.28E-06
2015	2.12E-05	8.07E-06	2.15E-06
2016	1.99E-05	7.91E-06	2.10E-06
2017	1.88E-05	7.74E-06	2.04E-06
2018	1.76E-05	7.38E-06	1.97E-06

TABLE 3-92: SUMMARY OF $ADD_{Expected}$ FOR FEMALE MINK ON A TEQ BASIS FOR THE PERIOD 1993 - 2018 REVISED

	Average Dietary Dose (mg/Kg/day)		
Year	189	168	154
1993	7.59E-05	1.82E-05	7.25E-06
1994	7.31E-05	1.70E-05	6.62E-06
1995	6.75E-05	1.53E-05	6.08E-06
1996	4.93E-05	1.29E-05	5.44E-06
1997	3.87E-05	9.89E-06	4.35E-06
1998	3.16E-05	1.20E-05	4.30E-06
1999	2.97E-05	1.08E-05	4.03E-06
2000	2.63E-05	9.38E-06	3.58E-06
2001	2.36E-05	7.89E-06	3.20E-06
2002	2.29E-05	6.87E-06	2.90E-06
2003	2.16E-05	6.24E-06	2.65E-06
2004	2.13E-05	5.93E-06	2.45E-06
2005	1.93E-05	5.54E-06	2.17E-06
2006	1.79E-05	4.98E-06	1.96E-06
2007	1.69E-05	4.54E-06	1.76E-06
2008	1.57E-05	4.34E-06	1.65E-06
2009	1.59E-05	4.13E-06	1.53E-06
2010	1.47E-05	3.73E-06	1.40E-06
2011	1.26E-05	3.35E-06	1.28E-06
2012	1.20E-05	3.12E-06	1.18E-06
2013	1.07E-05	3.01E-06	1.06E-06
2014	1.02E-05	3.25E-06	1.01E-06
2015	9.43E-06	3.18E-06	9.59E-07
2016	9.77E-06	3.14E-06	9.58E-07
2017	9.70E-06	3.04E-06	9.25E-07
2018	1.02E-05	3.00E-06	9.22E-07

TABLE 3-93: SUMMARY OF $ADD_{95\%UCL}$ FOR FEMALE MINK ON A TEQ BASIS FOR THE PERIOD 1993 - 2018

REVISED

Year	95% UCL Dietary Dose (mg/Kg/day)		
	189	168	154
1993	7.72E-05	1.91E-05	7.61E-06
1994	7.43E-05	1.78E-05	6.96E-06
1995	6.86E-05	1.60E-05	6.40E-06
1996	5.02E-05	1.35E-05	5.72E-06
1997	3.93E-05	1.04E-05	4.59E-06
1998	3.21E-05	1.25E-05	4.52E-06
1999	3.02E-05	1.13E-05	4.23E-06
2000	2.67E-05	9.81E-06	3.75E-06
2001	2.40E-05	8.26E-06	3.36E-06
2002	2.32E-05	7.19E-06	3.04E-06
2003	2.21E-05	6.54E-06	2.78E-06
2004	2.16E-05	6.21E-06	2.57E-06
2005	1.96E-05	5.80E-06	2.28E-06
2006	1.82E-05	5.22E-06	2.06E-06
2007	1.72E-05	4.75E-06	1.85E-06
2008	1.60E-05	4.55E-06	1.74E-06
2009	1.61E-05	4.32E-06	1.61E-06
2010	1.49E-05	3.90E-06	1.47E-06
2011	1.29E-05	3.50E-06	1.34E-06
2012	1.22E-05	3.27E-06	1.24E-06
2013	1.09E-05	3.15E-06	1.12E-06
2014	1.03E-05	3.41E-06	1.06E-06
2015	9.58E-06	3.33E-06	1.01E-06
2016	9.94E-06	3.30E-06	1.01E-06
2017	9.86E-06	3.18E-06	9.71E-07
2018	1.04E-05	3.14E-06	9.68E-07

TABLE 3-96: SUMMARY OF $ADD_{Expected}$ FOR FEMALE OTTER ON A TEQ BASIS FOR THE PERIOD 1993 - 2018 REVISED

	Total Average Dietary Dose (mg/Kg/day)		
Year	189	168	154
1993	8.99E-05	2.28E-05	1.08E-05
1994	8.22E-05	2.06E-05	9.81E-06
1995	9.03E-05	2.04E-05	9.42E-06
1996	7.95E-05	1.86E-05	8.34E-06
1997	6.61E-05	1.65E-05	7.43E-06
1998	6.33E-05	1.57E-05	6.86E-06
1999	6.44E-05	1.48E-05	6.32E-06
2000	6.49E-05	1.43E-05	5.93E-06
2001	5.48E-05	1.27E-05	5.44E-06
2002	4.94E-05	1.13E-05	4.97E-06
2003	5.12E-05	1.05E-05	4.52E-06
2004	4.27E-05	9.82E-06	3.98E-06
2005	3.87E-05	1.01E-05	3.82E-06
2006	3.63E-05	9.94E-06	3.67E-06
2007	3.97E-05	9.83E-06	3.55E-06
2008	4.34E-05	9.67E-06	3.50E-06
2009	4.11E-05	9.32E-06	3.41E-06
2010	4.19E-05	9.00E-06	3.19E-06
2011	3.66E-05	8.62E-06	3.09E-06
2012	3.18E-05	8.14E-06	3.01E-06
2013	3.09E-05	7.58E-06	2.90E-06
2014	3.48E-05	7.81E-06	2.95E-06
2015	3.32E-05	7.52E-06	2.80E-06
2016	2.98E-05	6.71E-06	2.53E-06
2017	2.67E-05	6.42E-06	2.55E-06
2018	3.11E-05	6.64E-06	2.57E-06

TABLE 3-97: SUMMARY OF $ADD_{95\%UCL}$ FOR FEMALE OTTER ON A TEQ BASIS FOR THE PERIOD 1993 - 2018
REVISED

Year	Total 95% UCL Dietary Dose (mg/Kg/day)		
	189	168	154
1993	9.30E-05	2.31E-05	1.09E-05
1994	8.52E-05	2.09E-05	9.95E-06
1995	9.35E-05	2.07E-05	9.55E-06
1996	8.26E-05	1.88E-05	8.45E-06
1997	6.86E-05	1.67E-05	7.54E-06
1998	6.57E-05	1.59E-05	6.96E-06
1999	6.70E-05	1.49E-05	6.41E-06
2000	6.76E-05	1.45E-05	6.02E-06
2001	5.73E-05	1.29E-05	5.52E-06
2002	5.14E-05	1.15E-05	5.04E-06
2003	5.32E-05	1.06E-05	4.58E-06
2004	4.45E-05	9.96E-06	4.04E-06
2005	4.03E-05	1.02E-05	3.88E-06
2006	3.78E-05	1.01E-05	3.72E-06
2007	4.14E-05	9.97E-06	3.60E-06
2008	4.52E-05	9.80E-06	3.55E-06
2009	4.29E-05	9.44E-06	3.46E-06
2010	4.36E-05	9.12E-06	3.23E-06
2011	3.83E-05	8.74E-06	3.14E-06
2012	3.31E-05	8.25E-06	3.05E-06
2013	3.22E-05	7.69E-06	2.94E-06
2014	3.63E-05	7.91E-06	2.99E-06
2015	3.46E-05	7.62E-06	2.84E-06
2016	3.11E-05	6.80E-06	2.56E-06
2017	2.79E-05	6.51E-06	2.58E-06
2018	3.24E-05	6.73E-06	2.60E-06

304274

TABLE 4-25
TOXICITY REFERENCE VALUES FOR FISH
DIETARY DOSES AND EGG CONCENTRATIONS OF TOTAL PCBs AND DIOXIN TOXIC EQUIVALENTS (TEQs)
REVISED

TRVs		Pumpkinseed (*Lepomis gibbosus*)	Spottail Shiner (*Notropis hudsonius*)	Brown Bullhead (*Ictalurus nebulosus*)	Yellow Perch (*Perca flavescens*)	White Perch (*Morone americana*)	Largemouth Bass (*Micropterus salmoides*)	Striped Bass (*Morone saxatilus*)	Shortnose Sturgeon (*Acipenser brevirostrum*)	References
Tissue Concentration										
Lab-based TRVs for PCBs (mg/kg wet wt.)	LOAEL	0.93	0.93	***0.93***	***0.93***	0.93	0.93	0.93	***0.93***	Hansen et al. (1974)
	NOAEL	0.19	0.19	***0.19***	***0.19***	0.19	0.19	0.19	***0.19***	
Field-based TRVs for PCBs (mg/kg wet wt.)	LOAEL	NA	NA	NA	NA	NA	NA	NA	NA	White perch and striped bass: Westin et al. (1983); spottail shiner: USACE (1988)
	NOAEL	***0.3***	***5.25***	NA	NA	***3.1***	***0.3***	***3.1***	NA	Pumpkinseed and Largemouth bass: Adams et al. (1989, 1990, 1992)
Egg Concentration										
Lab-based TRV for TEQs (ug/kg lipid) from salmonids	LOAEL	***0.6***	Not derived	***18***	***0.6***	***0.6***	***0.6***	***0.6***	***0.6***	Brown Bullhead: Elonen et al. (1998)
	NOAEL	***0.29***	Not derived	***8.0***	***0.29***	***0.29***	***0.29***	***0.29***	***0.29***	All others: Walker et al. (1994)
Lab-based TRV for TEQs (ug/kg lipid) from non salmonids	LOAEL	10.3	***103***	Not derived	10.3	10.3	10.3	10.3	10.3	Oliveri and Cooper (1997)
	NOAEL	0.54	***5.4***	Not derived	0.54	0.54	0.54	0.54	0.54	
Field-based TRVs for TEQs (ug/kg lipid)	LOAEL	NA	NA	NA	NA	NA	NA	NA	NA	
	NOAEL	NA	NA	NA	NA	NA	NA	NA	NA	

Note:

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

Units vary for PCBs and TEQ.

NA = Not available

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

TABLE 4-26
TOXICITY REFERENCE VALUES FOR BIRDS
DIETARY DOSES AND EGG CONCENTRATIONS OF TOTAL PCBs AND DIOXIN TOXIC EQUIVALENTS (TEQs)
REVISED

TRVs		Tree Swallow (*Tachycineta bicolor*)	Mallard Duck (*Anas platyrhychos*)	Belted Kingfisher (*Ceryle alcyon*)	Great Blue Heron (*Ardea herodias*)	Bald Eagle (*Haliaeetus leucocephalus*)	References
Dietary Dose							
Lab-based TRVs for PCBs (mg/kg/day)	LOAEL	0.07	***16***	***0.07***	***0.07***	***0.07***	Mallard: Haseltine and Prouty (1980)
	NOAEL	0.01	***1.6***	***0.01***	***0.01***	***0.01***	All others: Scott (1977)
Field-based TRVs for PCBs (mg/kd/day)	LOAEL	NA	NA	NA	NA	NA	Tree Swallow: US EPA Phase 2 Database (1998)
	NOAEL	***16.1***	NA	NA	NA	NA	
Lab-based TRVs for TEQs (ug/kg/day)	LOAEL	0.014	***0.014***	***0.014***	***0.014***	***0.014***	Nosek et al. (1992)
	NOAEL	0.0014	***0.0014***	***0.0014***	***0.0014***	***0.0014***	
Field-based TRVs for TEQs (ug/kg/day)	LOAEL	NA	NA	NA	NA	NA	US EPA Phase 2 Database (1998)
	NOAEL	***4.9***	NA	NA	NA	NA	
Egg Concentration							
Lab-based TRVs for PCBs (mg/kg egg)	LOAEL	2.21	***2.21***	***2.21***	***2.21***	2.21	Scott (1977)
	NOAEL	0.33	***0.33***	***0.33***	***0.33***	0.33	
Field-based TRVs for PCBs (mg/kg egg)	LOAEL	NA	NA	NA	NA	NA	Bald Eagle: Wiemeyer (1984, 1993)
	NOAEL	***26.7***	NA	NA	NA	***5.5***	Tree Swallow: US EPA Phase 2 Database (1998)
Lab-based TRVs for TEQs (ug/kg egg)	LOAEL	0.02	0.02	***0.02***	NA	0.02	Great Blue Heron: Janz and Bellward (1996)
	NOAEL	0.01	0.01	***0.01***	2	0.01	Others: Powell et al. (1996a)
Field-based TRVs for TEQs (ug/kg egg)	LOAEL	NA	NA	NA	***0.5***	NA	Mallard: White and Segniak (1994); White and Hoffman (1995)
	NOAEL	***13***	***0.005***	NA	***0.3***	***0.214***	Great Blue Heron: Sanderson et al. (1994)
							Eagle: Elliot et al. (1996a)
							Tree Swallow: US EPA Phase 2 Database (1998)

Note: Units vary for PCBs and TEQ.

NA = Not Available

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

TABLE 4-27
TOXICITY REFERENCE VALUES FOR MAMMALS
DIETARY DOSES OF TOTAL PCBs AND DIOXIN TOXIC EQUIVALENTS (TEQs)
REVISED

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

Note: Units vary for PCBs and TEQ.

Note: TRVs for raccoon and bat are based on mulit-generational studies to which interspecies uncertainty factors are applied.

NA = Not Available

Final selected TRVs are ***bolded and italicized*.**

TABLE 5-1
BENTHIC INVERTEBRATES COLLECTED AT TI POOL STATIONS (UNCHANGED)

Taxa in Rank Order	Common Name	Mean % of Total Ind. Collected
Caecidotea racovitzai	Isopod (sowbug)	34.6
Chironomidae[1]	Midges	~30.2
Oligochaeta	Aquatic worms	14.3
Gammarus fasciatus	Amphipod	10.3
Pisidium sp.	Pill Clam	5.0
Canthocamptes sp.	Harpacticoid copepod	1.5
Nematoda	Nematods (worms)	1.1
Phylocentropus sp.	Caddis fly larvae	<1.0
Dubiraphia sp.	Beetle larvae	<1.0
Menetus sp.	Caddis fly larvae	<1.0
Valvata sp.	Snail	<1.0
Sialis sp.	Alderfly larvae	<1.0
Oecetis sp.	Caddisfly larvae	<1.0
Probezzia sp.	Biting midges	<1.0
Enallagma sp.	Damselfly nymph	<1.0
Chydoridae	Water fleas (Cladoceran)	<1.0
Acariformes	Mites	<1.0
Amnicola sp.	Snail	<1.0
Mystacides sp.	Caddisfly larvae	<1.0
Diaphanosoma sp.	Water fleas (Cladoceran)	<1.0
Ceratopogonidae	Biting midges	<1.0
Helobdella fusca	Leech	<1.0
Arthropoda	Arthropods	<1.0
Eukiefferiella sp.	Biting Midges	<1.0

TABLE 5-1
BENTHIC INVERTEBRATES COLLECTED AT TI POOL STATIONS (UNCHANGED)

Taxa in Rank Order	**Common Name**	**Mean % of Total Ind. Collected**
Turbellaria	Flatworms	<1.0
Dugesia tigrina	Flatworm	<1.0
Bithynia tentaculata	Snail	<1.0
Trichoptera	Caddisfly larvae	<1.0
Chydorus sp.	Water fleas (Cladoceran)	<1.0
Caenis sp.	Mayfly nymph	<1.0
Physa sp.	Snail	<1.0
Helobdella sp.	Leech	<1.0
Mesocyclops sp.	Cyclopoid copepods	<1.0
Orthotrichia sp.	Caddis fly larvae	<1.0
Aeschnidae	Dragonfly nymph	<1.0
Hexagenia sp.	Mayfly nymph	<1.0
Hirudinea	Leeches	<1.0
Neureclipsis sp.	Caddisfly larvae	<1.0
Culicoides sp.	Mosquito larvae	<1.0
Corixidae	Water boatman	<1.0
Neoperla sp.	Stonefly nymph	<1.0
Caenidae	Mayfly nymph	<1.0
Donacia sp.	Beetle	<1.0
Hemiptera	True bugs	<1.0
Molanna sp.	Caddisfly larvae	<1.0
Copepoda	Copepods	<1.0
Insecta	Insects	<1.0
Baetidae	Mayfly nymph	<1.0
Macronychus sp.	Riffle beetle	<1.0

TABLE 5-1
BENTHIC INVERTEBRATES COLLECTED AT TI POOL STATIONS (UNCHANGED)

Taxa in Rank Order	**Common Name**	**Mean % of Total Ind. Collected**
Tipulidae	Cranefly larvae	<1.0
Cymatia sp.	Water boatman	<1.0
Notonecta sp.	Water boatman	<1.0
Talitridae	Amphipod	<1.0
Baetis sp.	Mayfly nymph	<1.0
Dromogomphus sp.	Dragonfly nymph	<1.0
Oxyethira sp.	Caddis fly larvae	<1.0
Diptera	Flies and midges	<1.0
Atherix sp.	Snipe fly	<1.0
Tabanidae	Horsefly larvae	<1.0
Elliptio sp.	Eastern elliptio mussel	<1.0

Notes: Taxa are listed in order of absolute abundance.
Mean Percent of individuals is based on the mean of Stations 3 to 7.
[1] Chironomidae were primarily composed of Chironominae, *Procladius* sp., *Tanytarsus* sp., *Dicrotendipes* sp., *Polypedilum* sp., *Clinotanypus* sp., *Tribelos jucundus*, and Tanypodinae.

TABLE 5-2
RELATIVE ABUNDANCE OF FIVE DOMINANT TAXANOMIC GROUPS AT TI POOL STATIONS (UNCHANGED)

Group/Taxa	Station 3 Abundance ind/m²	Station 3 Percent	Station 4 Abundance ind/m²	Station 4 Percent	Station 5 Abundance ind/m²	Station 5 Percent	Station 6 Abundance ind/m²	Station 6 Percent	Station 7 Abundance ind/m²	Station 7 Percent
Total Dominant Isopoda	**653**	**5.6%**	**3245**	**24.6%**	**14256**	**50.9%**	**2347**	**15.2%**	**7286**	**60.9%**
Caecidotea racovitzai										
Total Dominant Chironomids	**3775**	**32.3%**	**3959**	**30.1%**	**7619**	**27.2%**	**3277**	**21.3%**	**1561**	**13.0%**
Unidentified Chironomidae	1398	12.0%	122	0.9%	2232	8.0%	293	1.9%	398	3.3%
Unidentified Chironominae	510	4.4%	1490	11.3%	374	1.3%	1378	8.9%	41	0.3%
Procladius sp.	479	4.1%	204	1.5%	1474	5.3%	128	0.8%	296	2.5%
Tanytarsus sp.	255	2.2%	0	0.0%	1409	5.0%	26	0.2%	0	0.0%
Dicrotendipes sp.	479	4.1%	337	2.6%	560	2.0%	38	0.2%	204	1.7%
Polypedilum sp.	82	0.7%	102	0.8%	396	1.4%	281	1.8%	224	1.9%
Clinotanypus sp.	51	0.4%	133	1.0%	200	0.7%	332	2.2%	194	1.6%
Tribelos jucundus	0	0.0%	867	6.6%	0	0.0%	0	0.0%	0	0.0%
Unidentified Tanypodinae	112	1.0%	571	4.3%	131	0.5%	38	0.2%	0	0.0%
Tribelos sp.	214	1.8%	51	0.4%	194	0.7%	128	0.8%	204	1.7%
Chironomus sp.	41	0.3%	41	0.3%	650	2.3%	0	0.0%	0	0.0%
Cricotopus trifascia	102	0.9%	41	0.3%	0	0.0%	306	2.0%	0	0.0%
Unidentified Orthocladiinae	51	0.4%	0	0.0%	0	0.0%	332	2.2%	0	0.0%
Total Dominant Oligochaeta	**2918**	**25.0%**	**2245**	**17.0%**	**2681**	**9.6%**	**3584**	**23.3%**	**71**	**0.6%**
Unidentified Oligochaeta										
Total Dominant Amphipoda	**1030**	**8.8%**	**1102**	**8.4%**	**682**	**2.4%**	**3176**	**20.6%**	**2296**	**19.2%**
Gammarus fasciatus										
Total Dominant Pelecypoda	**1245**	**10.6%**	**1581**	**12.0%**	**49**	**0.2%**	**1097**	**7.1%**	**0**	**0.0%**
Pisidium sp.										
Subtotals	**9621**	**82.3%**	**12132**	**92.1%**	**25287**	**90.4%**	**13482**	**87.5%**	**11214**	**93.7%**
Total Abundance (all taxa)	**11691**		**13172**		**27983**		**15407**		**11968**	

TABLE 5-3
SUMMARY OF INFAUNA AND TOTAL BENTHOS INDICES – TI POOL (UNCHANGED)

Station	Simpson Diversity D_s		Simpson Dominance I		Eveness Distribution		Species Richness		Abundance No. Ind./Sq M	
	Infauna	Total Benthos	Infauna	Total Benthos	Infauna	Total Benthos	Infauna	Total Benthos	Infauna	Total Benthos
3	0.84	0.87	0.16	0.13	0.88	0.90	25	27	10,008	11,691
4	0.79	0.83	0.21	0.17	0.84	0.87	19	21	8,825	13,172
5	0.81	0.69	0.19	0.31	0.87	0.73	17	19	13,044	27,983
6	0.78	0.84	0.22	0.16	0.82	0.88	22	24	9,884	15,407
7	0.84	0.57	0.16	0.43	0.95	0.61	12	14	2,387	11,968
TI Pool Grand Mean	0.81	0.76	0.19	0.24	0.87	0.80	19	21	8,830	16,044

Notes: Total benthos equals the sum of infaunal and epibenthic macroinvertebrates

TABLE 5-4
RELATIVE PERCENT ABUNDANCE OF MACROINVERTEBRATES -- LOWER HUDSON RIVER (UNCHANGED)

Station 12		Station 14		Station 15		Station 17		Station 18	
Species/Group	**%**	**Species/Group**	**%**	**Species/Group**	**%**	**Species/Group**	**%**	**Species/Group**	**%**
Oligochaeta	42.4%	Chironominae Indet.	36.1%	Oligochaeta	22.0%	*Hobsonia florida*	36.1%	Oligochaeta	18.4%
Chironominae Indet.	12.9%	*Dicrotendipes sp.*	10.5%	Chydoridae	17.3%	Oligochaeta	32.8%	*Cyathura polita*	16.5%
Chironomidae Indet.	10.3%	*Procladius sp.*	10.2%	*Coelotanypus sp.*	14.0%	*Gammarus fasciatus*	11.3%	*Hobsonia florida*	14.2%
Procladius sp.	8.0%	*Polypedilum sp.*	9.0%	Nematoda	7.3%	*Clinotanypus sp.*	6.3%	*Hydrobia minuta*	11.5%
Polypedilum sp.	7.1%	*Clinotanypus sp.*	6.4%	*Clinotanypus sp.*	6.0%	Nemotoda	3.3%	Isopoda	10.8%
Pisidium sp.	2.9%	Oligochaeta	4.1%	*Polypedilum sp.*	5.3%	*Cyathura polita*	2.0%	*Clinotanypus sp.*	10.0%
Tribelos sp.	2.9%	*Gammarus fasciatus*	2.6%	Acariformes	4.0%	*Coelotanypus sp.*	2.0%	*Gammarus fasciatus*	9.7%
Cryptotendipes sp.	2.9%	*Pisidium sp.*	2.3%	*Dicrotendipes sp.*	4.0%	*Procladius sp.*	1.7%	Ostracoda	4.5%
Tanytarsus sp.	2.3%	Chironomidae Indet.	2.3%	*Cladotanytarsus sp.*	3.3%	Pelecypoda	1.3%	*Neanthes succinea*	1.3%
Chironomus sp.	1.9%	*Amnicola limosa*	1.9%	*Amnicola sp.*	3.3%	*Neanthes succinea*	1.0%	Pelecypoda	1.3%
Gammarus fasciatus	1.0%	*Cladotanytarsus sp.*	1.5%	*Synorthocladius sp.*	2.7%	Bryozoa	0.7%	*Procladius sp.*	1.0%
Acariformes	0.6%	*Orthotrichia sp.*	1.1%	*Pisidium sp.*	2.7%	*Balanus improvisus*	0.7%	*Rhithropanopeus harrisii*	0.5%
Tanypodinae Indet.	0.6%	Nematoda	1.1%	*Tribelos sp.*	2.0%	Isopoda	0.3%	*Coelotanypus sp.*	0.3%
Clinotanypus sp.	0.6%	Gastropoda	1.1%	Cyclopoida	1.3%	Orthocladiinae	0.3%		
Coleoptera	0.3%	*Cricotopus bicinctus*	1.1%	*Gammarus fasciatus*	1.3%	*Dicrotendipes sp.*	0.3%		
Bithynia tentaculata	0.3%	*Tanytarsus sp.*	1.1%	Hydroptilidae	1.3%				
Valvata sp.	0.3%	*Triaenodes sp.*	0.8%	*Cyathura polita*	0.7%				
Nematoda	0.3%	Orthocladiinae Indet.	0.8%	*Hydroptila sp.*	0.7%				
Cyathura polita	0.3%	*Chironomus sp.*	0.8%	*Chironomus sp.*	0.7%				
Ostracoda	0.3%	Acariformes	0.4%						
Leptoceridae	0.3%	*Dugesia tigrina*	0.4%						
Ceratopognidae	0.3%	*Diaphanosoma sp.*	0.4%						
Hemiptera	0.3%	*Hydroptila sp.*	0.4%						
Nilothauma sp.	0.3%	*Probezzia sp.*	0.4%						
Cryptochironomus sp.	0.3%	*Bithynia tentaculata*	0.4%						
		Tanypodinae Indet.	0.4%						
		Synorthocladius sp.	0.4%						
		Tribelos sp.	0.4%						
		Djalmabatista sp.	0.4%						
		Labrundinia sp.	0.4%						
		Coelotanypus sp.	0.4%						
		Synorthocladius sp.	0.4%						
		Cryptotendipes sp.	0.4%						

TABLE 5-4
RELATIVE PERCENT ABUNDANCE OF MACROINVERTEBRATES -- LOWER HUDSON RIVER (UNCHANGED)

Station 12		Station 14		Station 15		Station 17		Station 18	
Species/Group	**%**	**Species/Group**	**%**	**Species/Group**	**%**	**Species/Group**	**%**	**Species/Group**	**%**
Oligochaeta	42.4%	Chironominae Indet.	36.1%	Oligochaeta	22.0%	*Hobsonia florida*	36.1%	Oligochaeta	18.4%
Chironominae Indet.	12.9%	*Dicrotendipes sp.*	10.5%	Chydoridae	17.3%	Oligochaeta	32.8%	*Cyathura polita*	16.5%
Chironomidae Indet.	10.3%	*Procladius sp.*	10.2%	*Coelotanypus sp.*	14.0%	*Gammarus fasciatus*	11.3%	*Hobsonia florida*	14.2%
Procladius sp.	8.0%	*Polypedilum sp.*	9.0%	Nematoda	7.3%	*Clinotanypus sp.*	6.3%	*Hydrobia minuta*	11.5%
Polypedilum sp.	7.1%	*Clinotanypus sp.*	6.4%	*Clinotanypus sp.*	6.0%	Nemotoda	3.3%	Isopoda	10.8%
Pisidium sp.	2.9%	Oligochaeta	4.1%	*Polypedilum sp.*	5.3%	*Cyathura polita*	2.0%	*Clinotanypus sp.*	10.0%
Tribelos sp.	2.9%	*Gammarus fasciatus*	2.6%	Acariformes	4.0%	*Coelotanypus sp.*	2.0%	*Gammarus fasciatus*	9.7%
Cryptotendipes sp.	2.9%	*Pisidium sp.*	2.3%	*Dicrotendipes sp.*	4.0%	*Procladius sp.*	1.7%	Ostracoda	4.5%
Tanytarsus sp.	2.3%	Chironomidae Indet.	2.3%	*Cladotanytarsus sp.*	3.3%	Pelecypoda	1.3%	*Neanthes succinea*	1.3%
Chironomus sp.	1.9%	*Amnicola limosa*	1.9%	*Amnicola sp.*	3.3%	*Neanthes succinea*	1.0%	Pelecypoda	1.3%
Gammarus fasciatus	1.0%	*Cladotanytarsus sp.*	1.5%	*Synorthocladius sp.*	2.7%	Bryozoa	0.7%	*Procladius sp.*	1.0%
Acariformes	0.6%	*Orthotrichia sp.*	1.1%	*Pisidium sp.*	2.7%	*Balanus improvisus*	0.7%	*Rhithropanopeus harrisii*	0.5%
Tanypodinae Indet.	0.6%	Nematoda	1.1%	*Tribelos sp.*	2.0%	Isopoda	0.3%	*Coelotanypus sp.*	0.3%
Clinotanypus sp.	0.6%	Gastropoda	1.1%	Cyclopoida	1.3%	Orthocladiinae	0.3%		
Coleoptera	0.3%	*Cricotopus bicinctus*	1.1%	*Gammarus fasciatus*	1.3%	*Dicrotendipes sp.*	0.3%		
Bithynia tentaculata	0.3%	*Tanytarsus sp.*	1.1%	Hydroptilidae	1.3%				
Valvata sp.	0.3%	*Triaenodes sp.*	0.8%	*Cyathura polita*	0.7%				
Nematoda	0.3%	Orthocladiinae Indet.	0.8%	*Hydroptila sp.*	0.7%				
Cyathura polita	0.3%	*Chironomus sp.*	0.8%	*Chironomus sp.*	0.7%				
Ostracoda	0.3%	Acariformes	0.4%						
Leptoceridae	0.3%	*Dugesia tigrina*	0.4%						
Ceratopognidae	0.3%	*Diaphanosoma sp.*	0.4%						
Hemiptera	0.3%	*Hydroptila sp.*	0.4%						
Nilothauma sp.	0.3%	*Probezzia sp.*	0.4%						
Cryptochironomus sp.	0.3%	*Bithynia tentaculata*	0.4%						
		Tanypodinae Indet.	0.4%						
		Synorthocladius sp.	0.4%						
		Tribelos sp.	0.4%						
		Djalmabatista sp.	0.4%						
		Labrundinia sp.	0.4%						
		Coelotanypus sp.	0.4%						
		Synorthocladius sp.	0.4%						
		Cryptotendipes sp.	0.4%						

304283

TABLE 5-5
SUMMARY OF DIVERSITY INDICES AND ABUNDANCE DATA – LOWER HUDSON RIVER (UNCHANGED)

Station	D_s	I	D_{max}	E_s	Species Richness	Abundance ind/m^2	Biomass mg/m^2
Station 12 Stockport Flats	0.70	0.30	0.92	0.76	14	5,289	63
Station 14 Tivoli Bays	0.82	0.18	0.95	0.86	16	4,524	126
Station 15 Esopus Meadows	0.86	0.14	0.93	0.93	11	2,551	65
Station 17 Iona Island	0.71	0.29	0.90	0.79	9	5,136	365
Station 18 Piermont Pier	0.84	0.16	0.90	0.93	9	6,480	291
Grand Mean	0.79	0.21	0.92	0.85	12	4,796	182

TABLE 5-6
SELECTED SEDIMENT SCREENING GUIDELINES: PCBs (UNCHANGED)

	Total PCBs	Aroclor 1254	Aroclor 1248	Aroclor 1016	Aroclor 1260	Aroclor 1242
Sediment Guidelines/Effect Levels						
Hudson River Sediment Effect Concentrations						
(NOAA, 1999) - mg/kg (ppm)						
Threshold Effect Concentration	0.04					
Mid-range Effect Concentration	0.4					
Extreme Effect Concentration	1.7					
NYSDEC (1998) Freshwater (μg/g OC)						
Benthic Aquatic Life Acute Toxicity	2760.8					
Benthic Aquatic Life Chronic Toxicity	19.3					
Wildlife Bioaccumulation	1.4					
Ontario Ministry of the Environment						
Freshwater Guidelines (Persaud et al., 1993)						
No Effect Level (μg/g)	0.01					
Lowest Effect Level (μg/g)	0.07	0.06	0.03	0.007	0.005	
Severe Effect Level (μg/g OC)	530	34	150	53	24	
Long et al. (1995) Marine & Estuaries- ppb						
Effects-Range-Low	22.7					
Effects-Range-Median	180					
Ingersoll et al. (1996) Freshwater Guidelines						
based on *Hyallela azteca* - ppb						
Effects-Range-Low	50					
Effects-Range-Median	730					
Threshold Effect Level	32					
Probable Effect Level	240					
No Effect Concentration	190					
Washington State (1997) Freshwater - ppb						
Probable Apparent Effects Threshold - Microtox	21	7.3	21			
PAET - *Hyalella azteca*	450	240				100
Apparent Effects Threshold - Microtox	21	7.3				
AET - *Hyalella azteca*	820	350				100
Apparent Effects Threshold - Microtox mg/kg OC	2.6	0.73				
AET - *Hyalella azteca* mg/kg OC		18				
Jones et al. (1997) ppb; Eq-P-derived assuming 1% OC						
Recommended TOC adjustment						
Secondary Chronic Values		810	1000		4500000	

Notes: All values are provided in dry weight unless noted
Mean PCB conc.Upper Hudson benthic stations: 9.292 - 29.320 ppm
Mean PCB conc.Lower Hudson benthic stations: 0.367 - 1.313 ppm

TABLE 5-7: FEDERAL AND STATE PCB WATER QUALITY CRITERIA (UNCHANGED)

	Total PCB Water Quality Criteria (μg/L)	Upper Hudson 1993 (μg/L)	
USEPA/NYSDEC - Benthic Aquatic Life		Average	Maximum
Acute Toxicity - Freshwater	2		
Acute Toxicity - Saltwater	10		
Chronic Toxicity - Freshwater	0.014	0.071	0.226
Chronic Toxicity - Saltwater	0.03		
NYSDEC - Wildlife Bioaccumulation			
Freshwater	0.001	0.071	0.226
Saltwater	0.001		
NYSDEC Surface Water Standards			
Wildlife Criterion	0.00012	0.071	0.226

Sources: NYSDEC June, 1998 and March 1998; USEPA, 1991

TABLE 5-8: RATIO OF OBSERVED SEDIMENT CONCENTRATIONS TO GUIDELINES (UNCHANGED)

Location	TEC 0.04 mg/kg dry weight Average Sediment	TEC 0.04 mg/kg dry weight 95% UCL Sediment	MEC 0.4 mg/kg dry weight Average Sediment	MEC 0.4 mg/kg dry weight 95% UCL Sediment	EEC 1.7 mg/kg dry weight Average Sediment	EEC 1.7 mg/kg dry weight 95% UCL Sediment	NYSDEC Benthic Chronic 19.3 mg/kg OC Average Sediment	NYSDEC Benthic Chronic 19.3 mg/kg OC 95% UCL Sediment	NYSDEC Wildlife 1.4 mg/kg OC Average Sediment	NYSDEC Wildlife 1.4 mg/kg OC 95% UCL Sediment
Upper River										
Thompson Island Pool (189)	**297**	**435**	**30**	**43**	**7.0**	**10.2**	**12**	**16**	**169**	**215**
Stillwater (168)	**776**	**1354**	**78**	**135**	**18**	**32**	**43**	**74**	**587**	**1025**
Federal Dam (154)	**70**	**117**	**7.0**	**12**	**1.6**	**2.8**	**9.2**	**15**	**127**	**213**
Lower River										
143.5	**22**	**24**	**2.2**	**2.4**	0.5	0.6	**1.4**	**1.4**	**20**	**20**
137.2	**38**	**77**	**3.8**	**7.7**	0.9	**1.8**	**2.8**	**5.7**	**39**	**79**
122.4	**24**	**27**	**2.4**	**2.7**	0.6	0.6	**2.4**	**2.6**	**33**	**36**
113.8	**25**	**42**	**2.5**	**4.2**	0.6	**1.0**	**1.6**	**2.6**	**22**	**37**
100	**10**	**215**	**1.0**	**22**	0.2	**5.1**	0.9	**17**	**12**	**241**
88.9	**20**	**57**	**2.0**	**5.7**	0.5	**1.3**	1.1	**3.3**	**16**	**45**
58.7	**6.3**	**70**	0.6	**7.0**	0.1	**1.6**	**1.1**	**9.6**	**16**	**133**
47.3	**38**	**150**	**3.8**	**15**	0.9	**3.5**	**2.3**	**8.8**	**32**	**121**
25.8	**14**	**39**	**1.4**	**3.9**	0.3	0.9	**1.5**	**3.9**	**20**	**54**

304288

TABLE 5-8: RATIO OF OBSERVED SEDIMENT CONCENTRATIONS TO GUIDELINES

	Persaud LEL 0.007 mg/kg dry weight		Persaud SEL 53 mg/kg OC		Washington State PAET 1242 100 ppb		Washington State PAET Microtox		Washington State AET Microtox OC	
Location	Average Sediment	95% UCL Sediment	Average Sediment	95% UCL Sediment	Average Sediment	95% UCL Sediment	Average Sediment	95% UCL Sediment	Average Sediment	95% UCL Sediment
Upper River										
Thompson Island Pool (189)	**1697**	**2483**	**4.5**	**5.7**	**119**	**174**	**566**	**828**	**91**	**116**
Stillwater (168)	**4433**	**7739**	**16**	**27**	**310**	**542**	**1478**	**2580**	**316**	**552**
Federal Dam (154)	**399**	**669**	**3.4**	**5.6**	**28**	**47**	**133**	**223**	**69**	**115**
Lower River										
143.5	**123**	**135**	0.5	0.5	**8.6**	**9.4**	**41**	**45**	**11**	**11**
137.2	**217**	**438**	**1.0**	**2.1**	**15**	**31**	**72**	**146**	**21**	**43**
122.4	**138**	**153**	0.9	0.9	**10**	**11**	**46**	**51**	**18**	**19**
113.8	**144**	**238**	0.6	**1.0**	**10**	**17**	**48**	**79**	**12**	**20**
100	**57**	**1230**	0.3	**6.4**	**4.0**	**86**	**19**	**410**	**6.7**	**130**
88.9	**112**	**326**	0.4	**1.2**	**7.8**	**23**	**37**	**109**	**8.5**	**24**
58.7	**36**	**399**	0.4	**3.5**	**2.5**	**28**	**12**	**133**	**8.4**	**71**
47.3	**220**	**857**	0.8	**3.2**	**15**	**60**	**73**	**286**	**17**	**65**
25.8	**83**	**223**	0.5	**1.4**	**5.8**	**16**	**28**	**74**	**11**	**29**

TABLE 5-9: RATIO OF HUDTOX PREDICTED SEDIMENT CONCENTRATIONS TO SEDIMENT GUIDELINES REVISED

Year	Average PCB Results			Average PCB Results			Average PCB Results			Average PCB Results			Average PCB Results		
	189 Total Sed Conc	168 Total Sed Conc	154 Total Sed Conc	189 Total Sed Conc	168 Total Sed Conc	154 Total Sed Conc	189 Total Sed Conc	168 Total Sed Conc	154 Total Sed Conc	189 Total Sed Conc	168 Total Sed Conc	154 Total Sed Conc	189 Total Sed Conc	168 Total Sed Conc	154 Total Sed Conc
	TEC: 0.04 mg/kg dry weight			MEC: 0.4 mg/kg dry weight			EEC: 1.7 mg/kg dry weight			NYSDEC Benthic Chronic 19.3 mg/Kg OC			NYSDEC Wildlife 1.4 mg/Kg OC		
1993	**1353**	**176**	**54**	**135**	**18**	**5.4**	**32**	**4.1**	**1.3**	**141**	**22**	**14**	**1941**	**305**	**194**
1994	**1220**	**161**	**50**	**122**	**16**	**5.0**	**29**	**3.8**	**1.2**	**127**	**20**	**13**	**1750**	**279**	**177**
1995	**1156**	**152**	**46**	**116**	**15**	**4.6**	**27**	**3.6**	**1.1**	**120**	**19**	**12**	**1658**	**264**	**166**
1996	**1037**	**131**	**42**	**104**	**13**	**4.2**	**24**	**3.1**	**1.0**	**108**	**16**	**11**	**1487**	**227**	**152**
1997	**925**	**111**	**38**	**92**	**11**	**3.8**	**22**	**2.6**	0.9	**96**	**14**	**10**	**1326**	**191**	**135**
1998	**554**	**119**	**34**	**55**	**12**	**3.4**	**13**	**2.8**	0.8	**58**	**15**	**9**	**794**	**207**	**120**
1999	**527**	**109**	**31**	**53**	**11**	**3.1**	**12**	**2.6**	0.7	**55**	**14**	**8**	**756**	**189**	**112**
2000	**479**	**96**	**28**	**48**	**10**	**2.8**	**11**	**2.3**	0.7	**50**	**12**	**7.4**	**688**	**167**	**102**
2001	**426**	**80**	**25**	**43**	**8.0**	**2.5**	**10**	**1.9**	0.6	**44**	**10**	**6.5**	**611**	**139**	**90**
2002	**394**	**69**	**23**	**39**	**6.9**	**2.3**	**9.3**	**1.6**	0.5	**41**	**8.6**	**5.9**	**565**	**119**	**81**
2003	**360**	**60**	**21**	**36**	**6.0**	**2.1**	**8.5**	**1.4**	0.5	**37**	**7.6**	**5.3**	**517**	**104**	**73**
2004	**330**	**58**	**19**	**33**	**5.8**	**1.9**	**7.8**	**1.4**	0.4	**34**	**7.3**	**4.9**	**474**	**100**	**68**
2005	**308**	**55**	**17**	**31**	**5.5**	**1.7**	**7.2**	**1.3**	0.4	**32**	**6.9**	**4.4**	**442**	**95**	**61**
2006	**288**	**49**	**15**	**29**	**4.9**	**1.5**	**6.8**	**1.2**	0.4	**30**	**6.2**	**3.9**	**412**	**85**	**54**
2007	**259**	**45**	**14**	**26**	**4.5**	**1.4**	**6.1**	**1.1**	0.3	**27**	**5.6**	**3.5**	**372**	**77**	**48**
2008	**236**	**42**	**12**	**24**	**4.2**	**1.2**	**5.5**	**1.0**	0.3	**25**	**5.2**	**3.2**	**338**	**72**	**44**
2009	**225**	**40**	**12**	**22**	**4.0**	**1.2**	**5.3**	0.9	0.3	**23**	**5.0**	**3.0**	**322**	**69**	**42**
2010	**205**	**37**	**11**	**21**	**3.7**	**1.1**	**4.8**	0.9	0.3	**21**	**4.6**	**2.8**	**295**	**64**	**38**
2011	**184**	**33**	**10**	**18**	**3.3**	**1.0**	**4.3**	0.8	0.2	**19**	**4.1**	**2.5**	**264**	**57**	**35**
2012	**167**	**30**	**8.9**	**17**	**3.0**	0.9	**3.9**	0.7	0.2	**17**	**3.8**	**2.3**	**239**	**52**	**32**
2013	**151**	**27**	**8.0**	**15**	**2.7**	0.8	**3.5**	0.6	0.2	**16**	**3.4**	**2.1**	**216**	**47**	**29**
2014	**137**	**31**	**7.6**	**14**	**3.1**	0.8	**3.2**	0.7	0.2	**14**	**3.9**	**2.0**	**197**	**54**	**27**
2015	**127**	**30**	**7.1**	**13**	**3.0**	0.7	**3.0**	0.7	0.2	**13**	**3.8**	**1.9**	**182**	**53**	**26**
2016	**119**	**30**	**6.9**	**12**	**3.0**	0.7	**2.8**	0.7	0.2	**12**	**3.7**	**1.8**	**171**	**52**	**25**
2017	**112**	**29**	**6.8**	**11**	**2.9**	0.7	**2.6**	0.7	0.2	**12**	**3.7**	**1.8**	**161**	**51**	**24**
2018	**104**	**28**	**6.5**	**10**	**2.8**	0.6	**2.5**	0.7	0.2	**11**	**3.5**	**1.7**	**150**	**48**	**23**

304290

TABLE 5-9: RATIO OF HUDTOX PREDICTED SEDIMENT CONCENTRATIONS TO SEDIMENT GUIDELINES REVISED

Year	Average PCB Results			Average PCB Results			Average PCB Results			Average PCB Results		
	189 Total Sed Conc	168 Total Sed Conc	154 Total Sed Conc	189 Total Sed Conc	168 Total Sed Conc	154 Total Sed Conc	189 Total Sed Conc	168 Total Sed Conc	154 Total Sed Conc	189 Total Sed Conc	168 Total Sed Conc	154 Total Sed Conc
	Persaud LEL 0.007 mg/Kg dry weight			Persaud SEL 53 mg/Kg OC			Washington State PAET 1242 0.1 mg/Kg dry weight			Washington PAET Microtox 0.021 mg/Kg dry weight		
1993	**7731**	**1006**	**311**	**51**	**8.1**	**5.1**	**541**	**70**	**22**	**2577**	**335**	**104**
1994	**6974**	**922**	**284**	**46**	**7.4**	**4.7**	**488**	**65**	**20**	**2325**	**307**	**95**
1995	**6603**	**871**	**266**	**44**	**7.0**	**4.4**	**462**	**61**	**19**	**2201**	**290**	**89**
1996	**5926**	**750**	**243**	**39**	**6.0**	**4.0**	**415**	**52**	**17**	**1975**	**250**	**81**
1997	**5284**	**632**	**217**	**35**	**5.1**	**3.6**	**370**	**44**	**15**	**1761**	**211**	**72**
1998	**3164**	**683**	**192**	**21**	**5.5**	**3.2**	**221**	**48**	**13**	**1055**	**228**	**64**
1999	**3013**	**622**	**178**	**20**	**5.0**	**2.9**	**211**	**44**	**12**	**1004**	**207**	**59**
2000	**2739**	**551**	**162**	**18**	**4.4**	**2.7**	**192**	**39**	**11**	**913**	**184**	**54**
2001	**2433**	**458**	**144**	**16**	**3.7**	**2.4**	**170**	**32**	**10**	**811**	**153**	**48**
2002	**2251**	**393**	**129**	**15**	**3.1**	**2.1**	**158**	**28**	**9.0**	**750**	**131**	**43**
2003	**2058**	**344**	**117**	**14**	**2.8**	**1.9**	**144**	**24**	**8.2**	**686**	**115**	**39**
2004	**1887**	**330**	**109**	**13**	**2.6**	**1.8**	**132**	**23**	**7.6**	**629**	**110**	**36**
2005	**1759**	**313**	**97**	**12**	**2.5**	**1.6**	**123**	**22**	**6.8**	**586**	**104**	**32**
2006	**1643**	**281**	**86**	**11**	**2.2**	**1.4**	**115**	**20**	**6.1**	**548**	**94**	**29**
2007	**1482**	**255**	**77**	**10**	**2.0**	**1.3**	**104**	**18**	**5.4**	**494**	**85**	**26**
2008	**1348**	**237**	**71**	**8.9**	**1.9**	**1.2**	**94**	**17**	**5.0**	**449**	**79**	**24**
2009	**1284**	**229**	**67**	**8.5**	**1.8**	**1.1**	**90**	**16**	**4.7**	**428**	**76**	**22**
2010	**1174**	**211**	**61**	**7.8**	**1.7**	**1.0**	**82**	**15**	**4.3**	**391**	**70**	**20**
2011	**1051**	**189**	**56**	**7.0**	**1.5**	0.9	**74**	**13**	**3.9**	**350**	**63**	**19**
2012	**953**	**171**	**51**	**6.3**	**1.4**	0.8	**67**	**12**	**3.6**	**318**	**57**	**17**
2013	**861**	**155**	**46**	**5.7**	**1.2**	0.8	**60**	**11**	**3.2**	**287**	**52**	**15**
2014	**785**	**178**	**43**	**5.2**	**1.4**	0.7	**55**	**12**	**3.0**	**262**	**59**	**14**
2015	**726**	**173**	**41**	**4.8**	**1.4**	0.7	**51**	**12**	**2.9**	**242**	**58**	**14**
2016	**681**	**171**	**39**	**4.5**	**1.4**	0.6	**48**	**12**	**2.7**	**227**	**57**	**13**
2017	**642**	**167**	**39**	**4.3**	**1.3**	0.6	**45**	**12**	**2.7**	**214**	**56**	**13**
2018	**596**	**159**	**37**	**3.9**	**1.3**	0.6	**42**	**11**	**2.6**	**199**	**53**	**12**

TABLE 5-10: RATIO OF MEASURED WHOLE WATER CONCENTRATIONS TO BENCHMARKS REVISED

Hudson River	USEPA/NYSDEC - Benthic Aquatic Life 0.014 µg/L freshwater and 0.03 saltwater		USEPA/NYSDEC Wildlife Bioaccumulation Criterion 1.2E-04 µg/L	
Location	Average Conc. in Water	95% UCL Conc. In Water	Average Conc. in Water	95% UCL Conc. In Water
Upper River				
Thompson Island Pool (189)	**5.3**	**17**	**613**	**1942**
Stillwater (168)	**9.3**	**30**	**1090**	**3458**
Federal Dam (154)	**6.5**	**14**	**762**	**1634**
Lower River				
143.5	**5.1**	**55**	**589**	**6420**
137.2	**5.1**	**55**	**589**	**6420**
122.4	**2.3**	**30**	**270**	**3460**
113.8	**2.3**	**30**	**270**	**3460**
100	**2.3**	**30**	**270**	**3460**
88.9	**1.5**	**7**	**178**	**790**
58.7	**1.5**	**7**	**178**	**790**
47.3	**1.5**	**7**	**178**	**790**
25.8	**1.5**	**7**	**178**	**790**

Notes:

Source: TAMS/Gradient Database Release 4.1b

304291

TABLE 5-11: RATIO OF HUDTOX PREDICTED WHOLE WATER CONCENTRATIONS TO CRITERIA AND BENCHMARKS - REVISED

	Average Total PCB Results			Average Total PCB Results		
Year	189 Whole Water Conc	168 Whole Water Conc	154 Whole Water Conc	189 Whole Water Conc	168 Whole Water Conc	154 Whole Water Conc
	USEPA/NYSDEC - Benthic Aquatic Life 0.014 ug/L			USEPA/NYSDEC Wildlife Criterion 1.2E-04 ug/l		
1993	**11**	**6.3**	**4.5**	**134**	**74**	**53**
1994	**9.8**	**5.5**	**4.0**	**114**	**64**	**46**
1995	**9.9**	**5.3**	**3.8**	**116**	**62**	**44**
1996	**5.4**	**3.7**	**2.9**	**63**	**43**	**34**
1997	**6.0**	**3.8**	**3.1**	**70**	**44**	**36**
1998	**7.0**	**3.4**	**2.6**	**82**	**39**	**30**
1999	**6.3**	**3.2**	**2.5**	**74**	**38**	**29**
2000	**4.0**	**2.6**	**2.0**	**46**	**30**	**23**
2001	**4.4**	**2.7**	**2.1**	**51**	**31**	**24**
2002	**4.6**	**2.2**	**1.7**	**54**	**26**	**20**
2003	**4.3**	**2.1**	**1.6**	**51**	**25**	**18**
2004	**4.8**	**2.1**	**1.5**	**57**	**25**	**18**
2005	**3.8**	**1.9**	**1.4**	**44**	**22**	**16**
2006	**3.4**	**1.8**	**1.3**	**40**	**21**	**15**
2007	**3.6**	**1.7**	**1.2**	**42**	**20**	**14**
2008	**4.2**	**1.6**	**1.1**	**49**	**19**	**13**
2009	**3.9**	**1.6**	**1.0**	**45**	**18**	**12**
2010	**3.0**	**1.5**	**1.1**	**35**	**18**	**13**
2011	**3.1**	**1.4**	**1.0**	**36**	**16**	**11**
2012	**3.1**	**1.4**	0.9	**36**	**16**	**10**
2013	**2.7**	**1.2**	0.8	**31**	**14**	**9.6**
2014	**2.4**	**1.2**	0.8	**28**	**13**	**9.1**
2015	**2.3**	**1.1**	0.8	**27**	**13**	**8.8**
2016	**3.0**	**1.1**	0.7	**35**	**13**	**8**
2017	**2.9**	**1.1**	0.7	**34**	**13**	**8.6**
2018	**2.8**	1.1	0.7	**33.1**	**12**	**8.0**

TABLE 5-12: RATIO OF MEASURED FORAGE FISH CONCENTRATIONS TO TOXICITY BENCHMARKS
REVISED

	Pumpkinseed field-based NOAEL		Spottail shiner lab-based NOAEL		Spottail shiner lab-based LOAEL	
Location	Average Forage Fish Conc mg/Kg	95% UCL Forage Fish Conc mg/Kg	Average Forage Fish Conc mg/Kg	95% UCL Forage Fish Conc mg/Kg	Average Forage Fish Conc mg/Kg	95% UCL Forage Fish Conc mg/Kg
Upper River						
Thompson Island Pool (189)	**70**	**142**	**110**	**225**	**22**	**46**
Stillwater (168)	**24**	**34**	**37**	**53**	**7.6**	**11**
Federal Dam (154)	**5.5**	**8.0**	**8.7**	**13**	**1.8**	**2.6**
Lower River						
143.5	**6.4**	**7.7**	**10**	**12**	**2.1**	**2.5**
137.2	**13**	**28**	**21**	**44**	**4.2**	**9.1**
122.4	**5.0**	**8.0**	**7.8**	**13**	**1.6**	**2.6**
113.8	**5.2**	**5.4**	**8.2**	**8.5**	**1.7**	**1.7**
100	**2.3**	**3.9**	**3.6**	**6.1**	0.7	**1.3**
88.9	**4.5**	**6.1**	**7.1**	**10**	**1.4**	**2.0**
58.7	**4.9**	**5.5**	**7.7**	**8.7**	**1.6**	**1.8**
47.3	**4.3**	**5.8**	**6.9**	**9.1**	**1.4**	**1.9**
25.8	**3.3**	**3.9**	**5.2**	**6.2**	**1.1**	**1.3**

Source: TAMS/Gradient Database Release 4.1b

304294

TABLE 5-13: RATIO OF PREDICTED PUMPKINSEED CONCENTRATIONS TO FIELD-BASED NOAEL FOR TRI+ PCBS
REVISED

	River Mile 189			River Mile 168			River Mile 154		
Year	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)
1993	**44**	**54**	**82**	**15**	**19**	**33**	**6.5**	**8.4**	**14**
1994	**37**	**47**	**71**	**13**	**17**	**29**	**6.0**	**7.5**	**13**
1995	**37**	**46**	**67**	**12**	**15**	**27**	**5.5**	**6.9**	**12**
1996	**19**	**24**	**42**	**10**	**12**	**22**	**4.6**	**5.7**	**10**
1997	**15**	**17**	**23**	**4.7**	**6.6**	**11**	**2.5**	**3.4**	**5.5**
1998	**16**	**20**	**31**	**8.8**	**11**	**20**	**3.8**	**4.8**	**8.4**
1999	**15**	**19**	**28**	**8.2**	**10**	**18**	**3.6**	**4.4**	**7.7**
2000	**13**	**16**	**23**	**6.9**	**8.7**	**15**	**3.1**	**3.9**	**6.7**
2001	**11**	**14**	**20**	**5.8**	**7.2**	**13**	**2.7**	**3.4**	**5.9**
2002	**11**	**14**	**20**	**5.2**	**6.5**	**11**	**2.5**	**3.1**	**5.4**
2003	**11**	**14**	**23**	**4.9**	**6.2**	**11**	**2.4**	**3.0**	**5.1**
2004	**12**	**15**	**22**	**4.8**	**5.9**	**10**	**2.2**	**2.7**	**4.7**
2005	**11**	**13**	**19**	**4.3**	**5.4**	**9.5**	**2.0**	**2.4**	**4.2**
2006	**9.1**	**11**	**19**	**3.8**	**4.9**	**8.4**	**1.7**	**2.2**	**3.7**
2007	**10**	**12**	**18**	**3.6**	**4.5**	**7.8**	**1.6**	**2.0**	**3.5**
2008	**9.4**	**12**	**17**	**3.4**	**4.3**	**7.5**	**1.4**	**1.8**	**3.1**
2009	**10**	**12**	**19**	**3.4**	**4.2**	**7.2**	**1.4**	**1.8**	**3.0**
2010	**9.3**	**11**	**16**	**2.9**	**3.6**	**6.3**	**1.2**	**1.6**	**2.7**
2011	**7.6**	**9.4**	**15**	**2.7**	**3.4**	**5.8**	**1.2**	**1.5**	**2.5**
2012	**7.7**	**9.4**	**13**	**2.4**	**3.0**	**5.3**	**1.0**	**1.3**	**2.2**
2013	**6.6**	**8.3**	**13**	**2.5**	**3.1**	**5.4**	**1.0**	**1.2**	**2.1**
2014	**6.6**	**8.1**	**12**	**2.6**	**3.2**	**5.6**	**1.0**	**1.2**	**2.1**
2015	**5.8**	**7.2**	**10**	**2.5**	**3.2**	**5.5**	0.9	**1.1**	**2.0**
2016	**6.6**	**8.3**	**13**	**2.5**	**3.1**	**5.3**	0.9	**1.1**	**1.8**
2017	**6.9**	**8.4**	**12**	**2.5**	**3.1**	**5.3**	0.9	**1.1**	**1.9**
2018	**7.2**	**9.1**	**14**	**2.7**	**3.2**	**5.7**	**1.0**	**1.2**	**2.1**

Bold values indicate exceedances

TABLE 5-14: RATIO OF PREDICTED SPOTTAIL SHINER CONCENTRATIONS TO FIELD-DERIVED NOAEL FOR TRI+ PCBS REVISED

	River Mile 189			River Mile 168			River Mile 154		
Year	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)
1993	**2.4**	**2.8**	**4.1**	0.2	0.4	**1.0**	0.1	0.2	0.4
1994	**2.3**	**2.9**	**4.2**	0.2	0.4	0.9	0.1	0.1	0.4
1995	**2.2**	**2.6**	**3.9**	0.2	0.3	0.9	0.1	0.1	0.4
1996	**1.2**	**1.5**	**2.2**	0.2	0.2	0.7	0.1	0.1	0.3
1997	0.8	0.9	**1.2**	0.1	0.1	0.4	0.0	0.1	0.2
1998	**1.0**	**1.2**	**1.8**	0.1	0.2	0.6	0.1	0.1	0.2
1999	**1.0**	**1.1**	**1.6**	0.1	0.2	0.6	0.1	0.1	0.2
2000	0.8	**1.0**	**1.4**	0.1	0.2	0.4	0.0	0.1	0.2
2001	0.7	0.9	**1.2**	0.1	0.1	0.4	0.0	0.1	0.2
2002	0.7	0.9	**1.3**	0.1	0.1	0.3	0.0	0.1	0.2
2003	0.7	0.8	**1.4**	0.1	0.1	0.3	0.0	0.1	0.2
2004	0.8	0.9	**1.3**	0.1	0.1	0.3	0.0	0.1	0.1
2005	0.7	0.8	**1.1**	0.1	0.1	0.3	0.0	0.0	0.1
2006	0.6	0.7	**1.1**	0.1	0.1	0.3	0.0	0.0	0.1
2007	0.6	0.7	**1.0**	0.1	0.1	0.2	0.0	0.0	0.1
2008	0.6	0.7	**1.0**	0.1	0.1	0.2	0.0	0.0	0.1
2009	0.6	0.7	**1.1**	0.1	0.1	0.2	0.0	0.0	0.1
2010	0.6	0.7	0.9	0.0	0.1	0.2	0.0	0.0	0.1
2011	0.5	0.6	0.8	0.0	0.1	0.2	0.0	0.0	0.1
2012	0.5	0.6	0.8	0.0	0.1	0.2	0.0	0.0	0.1
2013	0.4	0.5	0.7	0.0	0.1	0.2	0.0	0.0	0.1
2014	0.4	0.5	0.7	0.0	0.1	0.2	0.0	0.0	0.1
2015	0.4	0.4	0.6	0.0	0.1	0.2	0.0	0.0	0.1
2016	0.4	0.5	0.7	0.0	0.1	0.2	0.0	0.0	0.1
2017	0.4	0.5	0.7	0.0	0.1	0.2	0.0	0.0	0.1
2018	0.5	0.6	0.9	0.0	0.1	0.2	0.0	0.0	0.1

TABLE 5-15: RATIO OF PREDICTED SPOTTAIL SHINER CONCENTRATIONS TO LABORATORY-DERIVED LOAEL FOR TRI+ PCBS - OBSOLETE TABLE

REVISED

Year	River Mile 189 25th (mg/kg wet weight)	River Mile 189 Median (mg/kg wet weight)	River Mile 189 95th Percentile (mg/kg wet weight)	River Mile 168 25th (mg/kg wet weight)	River Mile 168 Median (mg/kg wet weight)	River Mile 168 95th Percentile (mg/kg wet weight)	River Mile 154 25th (mg/kg wet weight)	River Mile 154 Median (mg/kg wet weight)	River Mile 154 95th Percentile (mg/kg wet weight)
1993									
1994									
1995									
1996									
1997									
1998									
1999									
2000									
2001									
2002									
2003									
2004									
2005					**THIS SPECIES IS NOW**				
2006					**COMPARED TO A**				
2007					**FIELD-BASED NOAEL *ONLY***				
2008					**(SEE PAGE XXX)**				
2009									
2010									
2011									
2012									
2013									
2014									
2015									
2016									
2017									
2018									

TABLE 5-16: RATIO OF PREDICTED PUMPKINSEED CONCENTRATIONS TO LABORATORY-DERIVED NOAEL ON A TEQ BASIS REVISED

	River Mile 189			River Mile 168			River Mile 154		
Year	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)
1993	**3.4**	**4.3**	**7.1**	**1.2**	**1.6**	**2.9**	0.6	0.7	**1.3**
1994	**2.9**	**3.8**	**6.1**	**1.1**	**1.4**	**2.6**	0.5	0.6	**1.1**
1995	**2.9**	**3.7**	**5.8**	**1.0**	**1.3**	**2.4**	0.5	0.6	**1.0**
1996	**1.5**	**2.0**	**3.5**	0.8	**1.1**	**2.0**	0.4	0.5	0.9
1997	**1.3**	**1.7**	**2.6**	0.8	**1.0**	**1.8**	0.4	0.4	0.8
1998	**1.3**	**1.6**	**2.6**	0.8	**1.0**	**1.8**	0.3	0.4	0.7
1999	**1.2**	**1.5**	**2.4**	0.7	0.9	**1.6**	0.3	0.4	0.7
2000	**1.0**	**1.3**	**2.0**	0.6	0.7	**1.4**	0.3	0.3	0.6
2001	0.9	**1.1**	**1.7**	0.5	0.6	**1.2**	0.2	0.3	0.5
2002	0.9	**1.1**	**1.7**	0.4	0.6	**1.0**	0.2	0.3	0.5
2003	0.8	**1.1**	**2.0**	0.4	0.5	**1.0**	0.2	0.3	0.5
2004	**1.0**	**1.2**	**1.9**	0.4	0.5	0.9	0.2	0.2	0.4
2005	0.8	**1.1**	**1.6**	0.4	0.5	0.8	0.2	0.2	0.4
2006	0.7	0.9	**1.6**	0.3	0.4	0.8	0.1	0.2	0.3
2007	0.8	0.9	**1.6**	0.3	0.4	0.7	0.1	0.2	0.3
2008	0.7	0.9	**1.5**	0.3	0.4	0.7	0.1	0.2	0.3
2009	0.8	1.0	**1.6**	0.3	0.4	0.6	0.1	0.2	0.3
2010	0.7	0.9	**1.4**	0.2	0.3	0.6	0.1	0.1	0.2
2011	0.6	0.7	**1.3**	0.2	0.3	0.5	0.1	0.1	0.2
2012	0.6	0.7	**1.2**	0.2	0.3	0.5	0.1	0.1	0.2
2013	0.5	0.7	**1.1**	0.2	0.3	0.5	0.1	0.1	0.2
2014	0.5	0.6	**1.0**	0.2	0.3	0.5	0.1	0.1	0.2
2015	0.5	0.6	0.9	0.2	0.3	0.5	0.1	0.1	0.2
2016	0.5	0.7	**1.1**	0.2	0.3	0.5	0.1	0.1	0.2
2017	0.5	0.7	**1.1**	0.2	0.3	0.5	0.1	0.1	0.2
2018	0.6	0.7	**1.2**	0.2	0.3	0.5	0.1	0.1	0.2

Bold values indicate exceedances

304297

304298

TABLE 5-17: RATIO OF PREDICTED PUMPKINSEED CONCENTRATIONS TO LABORATORY-DERIVED LOAEL ON A TEQ BASIS
REVISED

	River Mile 189			River Mile 168			River Mile 154		
Year	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)
1993	**1.6**	**2.1**	**3.4**	0.6	0.8	**1.4**	0.3	0.3	0.6
1994	**1.4**	**1.8**	**3.0**	0.5	0.7	**1.3**	0.2	0.3	0.6
1995	**1.4**	**1.8**	**2.8**	0.5	0.6	**1.2**	0.2	0.3	0.5
1996	0.7	0.9	**1.7**	0.4	0.5	**1.0**	0.2	0.2	0.4
1997	0.6	0.8	**1.3**	0.4	0.5	0.9	0.2	0.2	0.4
1998	0.6	0.8	**1.3**	0.4	0.5	0.9	0.2	0.2	0.4
1999	0.6	0.7	**1.2**	0.3	0.4	0.8	0.1	0.2	0.3
2000	0.5	0.6	**1.0**	0.3	0.4	0.7	0.1	0.2	0.3
2001	0.4	0.5	0.8	0.2	0.3	0.6	0.1	0.1	0.3
2002	0.4	0.5	0.8	0.2	0.3	0.5	0.1	0.1	0.2
2003	0.4	0.5	0.9	0.2	0.3	0.5	0.1	0.1	0.2
2004	0.5	0.6	0.9	0.2	0.2	0.4	0.1	0.1	0.2
2005	0.4	0.5	0.8	0.2	0.2	0.4	0.1	0.1	0.2
2006	0.3	0.4	0.8	0.2	0.2	0.4	0.1	0.1	0.2
2007	0.4	0.5	0.8	0.1	0.2	0.3	0.1	0.1	0.1
2008	0.4	0.5	0.7	0.1	0.2	0.3	0.1	0.1	0.1
2009	0.4	0.5	0.8	0.1	0.2	0.3	0.1	0.1	0.1
2010	0.4	0.4	0.7	0.1	0.2	0.3	0.1	0.1	0.1
2011	0.3	0.4	0.6	0.1	0.1	0.3	0.0	0.1	0.1
2012	0.3	0.4	0.6	0.1	0.1	0.2	0.0	0.1	0.1
2013	0.2	0.3	0.5	0.1	0.1	0.2	0.0	0.0	0.1
2014	0.3	0.3	0.5	0.1	0.1	0.2	0.0	0.0	0.1
2015	0.2	0.3	0.4	0.1	0.1	0.2	0.0	0.0	0.1
2016	0.3	0.3	0.5	0.1	0.1	0.2	0.0	0.0	0.1
2017	0.3	0.3	0.5	0.1	0.1	0.2	0.0	0.0	0.1
2018	0.3	0.4	0.6	0.1	0.1	0.2	0.0	0.1	0.1

Bold values indicate exceedances

TABLE 5-18: RATIO OF PREDICTED SPOTTAIL SHINER CONCENTRATIONS TO LABORATORY-DERIVED NOAEL ON A TEQ BASIS REVISED

	River Mile 189			River Mile 168			River Mile 154		
Year	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)
1993	0.37	0.49	0.97	0.08	0.12	0.36	0.04	0.06	0.15
1994	0.36	0.47	0.86	0.07	0.12	0.33	0.03	0.05	0.14
1995	0.33	0.42	0.76	0.06	0.10	0.29	0.03	0.04	0.13
1996	0.20	0.26	0.49	0.05	0.08	0.24	0.03	0.04	0.11
1997	0.15	0.19	0.36	0.05	0.07	0.20	0.02	0.03	0.10
1998	0.15	0.19	0.35	0.05	0.07	0.22	0.02	0.03	0.09
1999	0.14	0.18	0.31	0.04	0.07	0.20	0.02	0.03	0.08
2000	0.12	0.15	0.27	0.04	0.06	0.16	0.02	0.03	0.07
2001	0.11	0.14	0.24	0.03	0.05	0.14	0.01	0.02	0.06
2002	0.11	0.14	0.25	0.03	0.04	0.13	0.01	0.02	0.06
2003	0.10	0.13	0.27	0.03	0.04	0.12	0.01	0.02	0.05
2004	0.11	0.14	0.26	0.03	0.04	0.11	0.01	0.02	0.05
2005	0.10	0.13	0.22	0.02	0.04	0.10	0.01	0.02	0.04
2006	0.09	0.12	0.21	0.02	0.03	0.09	0.01	0.01	0.04
2007	0.09	0.12	0.20	0.02	0.03	0.08	0.01	0.01	0.04
2008	0.09	0.11	0.20	0.02	0.03	0.08	0.01	0.01	0.03
2009	0.09	0.11	0.21	0.02	0.03	0.08	0.01	0.01	0.03
2010	0.08	0.11	0.19	0.02	0.02	0.07	0.01	0.01	0.03
2011	0.07	0.09	0.16	0.01	0.02	0.06	0.01	0.01	0.03
2012	0.07	0.09	0.15	0.01	0.02	0.06	0.01	0.01	0.02
2013	0.06	0.08	0.14	0.01	0.02	0.06	0.01	0.01	0.02
2014	0.06	0.08	0.13	0.01	0.02	0.06	0.01	0.01	0.02
2015	0.06	0.07	0.12	0.01	0.02	0.06	0.005	0.01	0.02
2016	0.06	0.08	0.15	0.01	0.02	0.06	0.005	0.01	0.02
2017	0.06	0.08	0.14	0.01	0.02	0.06	0.005	0.01	0.02
2018	0.07	0.09	0.17	0.01	0.02	0.06	0.005	0.01	0.02

Bold values indicate exceedances

304300

TABLE 5-19: RATIO OF PREDICTED SPOTTAIL SHINER CONCENTRATIONS TO LABORATORY-DERIVED LOAEL ON A TEQ BASIS
REVISED

	River Mile 189			River Mile 168			River Mile 154		
Year	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)
1993	0.019	0.025	0.051	0.004	0.006	0.019	0.002	0.003	0.008
1994	0.019	0.025	0.045	0.004	0.006	0.017	0.002	0.003	0.007
1995	0.017	0.022	0.040	0.003	0.005	0.015	0.001	0.002	0.007
1996	0.010	0.014	0.026	0.003	0.004	0.013	0.001	0.002	0.006
1997	0.008	0.010	0.019	0.003	0.004	0.011	0.001	0.002	0.005
1998	0.008	0.010	0.019	0.003	0.004	0.011	0.001	0.002	0.005
1999	0.007	0.009	0.016	0.002	0.004	0.010	0.001	0.002	0.004
2000	0.006	0.008	0.014	0.002	0.003	0.008	0.001	0.001	0.004
2001	0.006	0.007	0.013	0.002	0.003	0.007	0.001	0.001	0.003
2002	0.006	0.007	0.013	0.001	0.002	0.007	0.001	0.001	0.003
2003	0.005	0.007	0.014	0.001	0.002	0.006	0.001	0.001	0.003
2004	0.006	0.007	0.013	0.001	0.002	0.006	0.001	0.001	0.003
2005	0.005	0.007	0.012	0.001	0.002	0.005	0.001	0.001	0.002
2006	0.005	0.006	0.011	0.001	0.002	0.005	0.0005	0.001	0.002
2007	0.005	0.006	0.011	0.001	0.002	0.004	0.0004	0.001	0.002
2008	0.004	0.006	0.011	0.001	0.002	0.004	0.0004	0.001	0.002
2009	0.005	0.006	0.011	0.001	0.001	0.004	0.0004	0.001	0.002
2010	0.004	0.006	0.010	0.001	0.001	0.004	0.0003	0.001	0.001
2011	0.004	0.005	0.009	0.001	0.001	0.003	0.0003	0.0005	0.001
2012	0.004	0.005	0.008	0.001	0.001	0.003	0.0003	0.0005	0.001
2013	0.003	0.004	0.008	0.001	0.001	0.003	0.0003	0.0004	0.001
2014	0.003	0.004	0.007	0.001	0.001	0.003	0.0003	0.0004	0.001
2015	0.003	0.004	0.006	0.001	0.001	0.003	0.0003	0.0004	0.001
2016	0.003	0.004	0.008	0.001	0.001	0.003	0.0003	0.0004	0.001
2017	0.003	0.004	0.007	0.001	0.001	0.003	0.0003	0.0004	0.001
2018	0.004	0.005	0.009	0.001	0.001	0.003	0.0003	0.0004	0.001

Bold values indicate exceedances

TABLE 5-20: RATIO OF PREDICTED BROWN BULLHEAD CONCENTRATIONS TO LABORATORY-DERIVED NOAEL FOR TRI+ PCBS
REVISED

	River Mile 189			River Mile 168			River Mile 154		
Year	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)
1993	**105**	**134**	**209**	**48**	**56**	**90**	**20**	**23**	**37**
1994	**93**	**120**	**192**	**44**	**52**	**84**	**18**	**22**	**35**
1995	**89**	**115**	**178**	**40**	**46**	**75**	**17**	**20**	**32**
1996	**73**	**94**	**148**	**33**	**39**	**63**	**15**	**18**	**29**
1997	**64**	**82**	**118**	**42**	**50**	**70**	**19**	**23**	**32**
1998	**48**	**61**	**97**	**30**	**35**	**58**	**12**	**13**	**22**
1999	**42**	**54**	**85**	**27**	**32**	**54**	**11**	**13**	**22**
2000	**37**	**48**	**77**	**23**	**27**	**46**	**10**	**11**	**19**
2001	**33**	**43**	**67**	**19**	**23**	**39**	**8.6**	**10**	**17**
2002	**31**	**40**	**62**	**17**	**20**	**34**	**7.8**	**9.2**	**16**
2003	**28**	**36**	**57**	**16**	**19**	**31**	**7.2**	**8.5**	**14**
2004	**28**	**35**	**55**	**15**	**18**	**30**	**6.5**	**7.7**	**13**
2005	**25**	**32**	**50**	**14**	**16**	**27**	**5.8**	**6.8**	**12**
2006	**22**	**28**	**46**	**12**	**15**	**25**	**5.2**	**6.1**	**10**
2007	**21**	**27**	**43**	**11**	**13**	**23**	**4.8**	**5.6**	**9.4**
2008	**20**	**26**	**40**	**11**	**13**	**22**	**4.4**	**5.2**	**8.7**
2009	**19**	**25**	**38**	**10**	**12**	**20**	**4.1**	**4.8**	**8.1**
2010	**17**	**22**	**35**	**9.2**	**11**	**18**	**3.8**	**4.4**	**7.4**
2011	**16**	**20**	**31**	**8.4**	**10**	**17**	**3.4**	**4.0**	**6.8**
2012	**15**	**18**	**29**	**7.6**	**8.9**	**15**	**3.1**	**3.6**	**6.1**
2013	**13**	**16**	**26**	**8.1**	**9.4**	**16**	**2.9**	**3.4**	**5.6**
2014	**12**	**15**	**23**	**8.2**	**10**	**16**	**2.7**	**3.2**	**5.3**
2015	**11**	**14**	**22**	**8.1**	**9.5**	**16**	**2.6**	**3.0**	**5.1**
2016	**11**	**14**	**21**	**7.9**	**9.3**	**16**	**2.5**	**3.0**	**4.9**
2017	**11**	**13**	**20**	**7.6**	**9.0**	**15**	**2.5**	**2.9**	**4.8**
2018	**10**	**12**	**19**	**7.0**	**8.4**	**15**	**2.3**	**2.7**	**4.6**

304302

TABLE 5-21: RATIO OF PREDICTED BROWN BULLHEAD CONCENTRATIONS TO LABORATORY-DERIVED LOAEL FOR TRI+ PCBS
REVISED

Year	River Mile 189 25th (mg/kg wet weight)	River Mile 189 Median (mg/kg wet weight)	River Mile 189 95th Percentile (mg/kg wet weight)	River Mile 168 25th (mg/kg wet weight)	River Mile 168 Median (mg/kg wet weight)	River Mile 168 95th Percentile (mg/kg wet weight)	River Mile 154 25th (mg/kg wet weight)	River Mile 154 Median (mg/kg wet weight)	River Mile 154 95th Percentile (mg/kg wet weight)
1993	**13**	**17**	**26**	**6.0**	**7.1**	**11**	**2.5**	**2.9**	**4.7**
1994	**12**	**15**	**24**	**5.6**	**6.6**	**11**	**2.3**	**2.7**	**4.4**
1995	**11**	**15**	**23**	**5.0**	**5.9**	**9.5**	**2.1**	**2.5**	**4.0**
1996	**9.3**	**12**	**19**	**4.2**	**4.9**	**8.0**	**1.9**	**2.2**	**3.6**
1997	**8.1**	**10**	**15**	**5.4**	**6.4**	**8.9**	**2.4**	**2.9**	**4.0**
1998	**6.0**	**7.7**	**12**	**3.8**	**4.4**	**7.4**	**1.5**	**1.7**	**2.8**
1999	**5.4**	**6.9**	**11**	**3.4**	**4.0**	**6.9**	**1.4**	**1.6**	**2.7**
2000	**4.6**	**6.0**	**9.7**	**2.9**	**3.4**	**5.9**	**1.2**	**1.4**	**2.4**
2001	**4.2**	**5.4**	**8.5**	**2.5**	**2.9**	**5.0**	**1.1**	**1.3**	**2.2**
2002	**4.0**	**5.1**	**7.9**	**2.1**	**2.5**	**4.3**	**1.0**	**1.2**	**2.0**
2003	**3.6**	**4.6**	**7.2**	**2.0**	**2.4**	**4.0**	0.9	**1.1**	**1.8**
2004	**3.5**	**4.5**	**6.9**	**1.9**	**2.2**	**3.8**	0.8	**1.0**	**1.6**
2005	**3.2**	**4.1**	**6.4**	**1.7**	**2.0**	**3.5**	0.7	0.9	**1.5**
2006	**2.8**	**3.6**	**5.8**	**1.6**	**1.8**	**3.1**	0.7	0.8	**1.3**
2007	**2.7**	**3.4**	**5.4**	**1.5**	**1.7**	**2.9**	0.6	0.7	**1.2**
2008	**2.6**	**3.3**	**5.0**	**1.4**	**1.6**	**2.7**	0.6	0.7	**1.1**
2009	**2.5**	**3.1**	**4.8**	**1.3**	**1.5**	**2.6**	0.5	0.6	**1.0**
2010	**2.2**	**2.7**	**4.4**	**1.2**	**1.4**	**2.3**	0.5	0.6	0.9
2011	**2.0**	**2.5**	**3.9**	**1.1**	**1.2**	**2.1**	0.4	0.5	0.9
2012	**1.8**	**2.3**	**3.6**	**1.0**	**1.1**	**1.9**	0.4	0.5	0.8
2013	**1.6**	**2.1**	**3.2**	**1.0**	**1.2**	**2.0**	0.4	0.4	0.7
2014	**1.5**	**1.9**	**3.0**	**1.0**	**1.2**	**2.1**	0.3	0.4	0.7
2015	**1.4**	**1.8**	**2.7**	**1.0**	**1.2**	**2.0**	0.3	0.4	0.6
2016	**1.4**	**1.7**	**2.6**	**1.0**	**1.2**	**2.0**	0.3	0.4	0.6
2017	**1.3**	**1.7**	**2.6**	**1.0**	**1.1**	**1.9**	0.3	0.4	0.6
2018	**1.2**	**1.6**	**2.4**	0.9	**1.1**	**1.8**	0.3	0.3	0.6

TABLE 5-22: RATIO OF PREDICTED BROWN BULLHEAD CONCENTRATIONS TO LABORATORY-DERIVED NOAEL ON A TEQ BASIS
REVISED

	River Mile 189			River Mile 168			River Mile 154		
Year	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)
1993	0.16	0.21	0.38	0.07	0.09	0.16	0.03	0.04	0.07
1994	0.14	0.18	0.34	0.07	0.08	0.15	0.03	0.03	0.06
1995	0.13	0.18	0.32	0.06	0.07	0.13	0.03	0.03	0.06
1996	0.11	0.15	0.26	0.05	0.06	0.11	0.02	0.03	0.05
1997	0.09	0.12	0.21	0.04	0.05	0.10	0.02	0.02	0.04
1998	0.07	0.09	0.16	0.05	0.06	0.10	0.02	0.02	0.04
1999	0.06	0.08	0.15	0.04	0.05	0.09	0.02	0.02	0.04
2000	0.05	0.07	0.13	0.04	0.04	0.08	0.01	0.02	0.03
2001	0.05	0.07	0.12	0.03	0.04	0.07	0.01	0.02	0.03
2002	0.05	0.06	0.11	0.03	0.03	0.06	0.01	0.01	0.03
2003	0.04	0.06	0.10	0.02	0.03	0.05	0.01	0.01	0.03
2004	0.04	0.05	0.10	0.02	0.03	0.05	0.01	0.01	0.02
2005	0.04	0.05	0.09	0.02	0.03	0.05	0.01	0.01	0.02
2006	0.03	0.04	0.08	0.02	0.02	0.04	0.01	0.01	0.02
2007	0.03	0.04	0.08	0.02	0.02	0.04	0.01	0.01	0.02
2008	0.03	0.04	0.07	0.02	0.02	0.04	0.01	0.01	0.02
2009	0.03	0.04	0.07	0.02	0.02	0.04	0.01	0.01	0.01
2010	0.03	0.03	0.06	0.01	0.02	0.03	0.01	0.01	0.01
2011	0.02	0.03	0.05	0.01	0.02	0.03	0.01	0.01	0.01
2012	0.02	0.03	0.05	0.01	0.01	0.03	0.005	0.01	0.01
2013	0.02	0.02	0.04	0.01	0.01	0.03	0.004	0.01	0.01
2014	0.02	0.02	0.04	0.01	0.02	0.03	0.004	0.01	0.01
2015	0.02	0.02	0.04	0.01	0.02	0.03	0.004	0.005	0.01
2016	0.02	0.02	0.04	0.01	0.01	0.03	0.004	0.005	0.01
2017	0.02	0.02	0.04	0.01	0.01	0.03	0.004	0.005	0.01
2018	0.02	0.02	0.03	0.01	0.01	0.03	0.004	0.004	0.01

Bold values indicate exceedances

TABLE 5-23: RATIO OF PREDICTED BROWN BULLHEAD CONCENTRATIONS TO LABORATORY-DERIVED LOAEL ON A TEQ BASIS
REVISED

	River Mile 189			River Mile 168			River Mile 154		
Year	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)
1993	0.07	0.09	0.17	0.03	0.04	0.07	0.01	0.02	0.03
1994	0.06	0.08	0.15	0.03	0.04	0.07	0.01	0.02	0.03
1995	0.06	0.08	0.14	0.03	0.03	0.06	0.01	0.01	0.03
1996	0.05	0.06	0.12	0.02	0.03	0.05	0.01	0.01	0.02
1997	0.04	0.05	0.10	0.02	0.02	0.05	0.01	0.01	0.02
1998	0.03	0.04	0.07	0.02	0.02	0.05	0.01	0.01	0.02
1999	0.03	0.04	0.07	0.02	0.02	0.04	0.01	0.01	0.02
2000	0.02	0.03	0.06	0.02	0.02	0.03	0.01	0.01	0.01
2001	0.02	0.03	0.05	0.01	0.02	0.03	0.01	0.01	0.01
2002	0.02	0.03	0.05	0.01	0.01	0.03	0.01	0.01	0.01
2003	0.02	0.02	0.04	0.01	0.01	0.02	0.005	0.01	0.01
2004	0.02	0.02	0.04	0.01	0.01	0.02	0.005	0.01	0.01
2005	0.02	0.02	0.04	0.01	0.01	0.02	0.004	0.005	0.01
2006	0.01	0.02	0.04	0.01	0.01	0.02	0.004	0.004	0.01
2007	0.01	0.02	0.03	0.01	0.01	0.02	0.003	0.004	0.01
2008	0.01	0.02	0.03	0.01	0.01	0.02	0.003	0.004	0.01
2009	0.01	0.02	0.03	0.01	0.01	0.02	0.003	0.003	0.01
2010	0.01	0.01	0.03	0.01	0.01	0.01	0.003	0.003	0.01
2011	0.01	0.01	0.02	0.01	0.01	0.01	0.002	0.003	0.01
2012	0.01	0.01	0.02	0.01	0.01	0.01	0.002	0.003	0.005
2013	0.01	0.01	0.02	0.01	0.01	0.01	0.002	0.002	0.004
2014	0.01	0.01	0.02	0.01	0.01	0.01	0.002	0.002	0.004
2015	0.01	0.01	0.02	0.01	0.01	0.01	0.002	0.002	0.004
2016	0.01	0.01	0.02	0.01	0.01	0.01	0.002	0.002	0.004
2017	0.01	0.01	0.02	0.01	0.01	0.01	0.002	0.002	0.004
2018	0.01	0.01	0.02	0.005	0.01	0.01	0.002	0.002	0.004

Bold values indicate exceedances

TABLE 5-24: RATIO OF OBSERVED LARGEMOUTH BASS AND BROWN BULLHEAD CONCENTRATIONS TO TOXICITY BENCHMARKS USING NYSDEC DATASET
REVISED

<<<<<<<<<<<<<<< ---------------------------------- RATIO OF WET WEIGHT CONCENTRATION TO NOAEL ---------------------------------->>>>>>>>>>>

	Largemouth Bass 113			Largemouth Bass 168			Brown Bullhead 168			Largemouth Bass 189			Brown Bullhead 189		
	Average	95% UCL	Maximum	Average	95% UCL	Maximum	Average	95% UCL	Maximum	Average	95% UCL	Maximum	Average	95% UCL	Maximum
1993	37	*	115	56	72	127	67	105	137	315	610	1153	134	262	223
1994	52	120	174	46	91	106	45	105	143	150	227	321	138	181	546
1995	27	36	99	44	60	96	47	61	101	187	312	427	104	137	142
1996	31	54	89							93	124	190	85	*	99

<<<<<<<<<<<<<<< ---------------------------------- RATIO OF WET WEIGHT CONCENTRATION TO LOAEL ---------------------------------->>>>>>>>>>>

	Largemouth Bass 113			Largemouth Bass 168			Brown Bullhead 168			Largemouth Bass 189			Brown Bullhead 189		
	Average	95% UCL	Maximum	Average	95% UCL	Maximum	Average	95% UCL	Maximum	Average	95% UCL	Maximum	Average	95% UCL	Maximum
1993	NA	*	NA	NA	NA	NA	14	22	28	NA	NA	NA	27	53	45
1994	NA	NA	NA	NA	NA	NA	9	22	29	NA	NA	NA	28	37	112
1995	NA	NA	NA	NA	NA	NA	10	12	21	NA	NA	NA	21	28	29
1996	NA	NA	NA							NA	NA	NA	17	*	20

<<<<<<<<<<<< ----------------------------RATIO OF LIPID NORMALIZED CONCENTRATIONS: TEQ BASIS TO NOAEL ---------------------------->>>>>>>>

	Largemouth Bass 113			Largemouth Bass 168			Brown Bullhead 168			Largemouth Bass 189			Brown Bullhead 189		
	Average	95% UCL	Maximum	Average	95% UCL	Maximum	Average	95% UCL	Maximum	Average	95% UCL	Maximum	Average	95% UCL	Maximum
1993	2.4	5.1	4.2	9.5	11	15	0.2	*	0.3	42	64	94	0.6	1.1	1.2
1994	2.8	4.2	8.2	9.0	17	21	0.1	0.2	0.5	23	29	48	0.5	0.7	2.1
1995	2.7	3.3	4.3	10.5	12	18	0.1	0.1	0.3	20	31	34	0.2	0.3	0.4
1996	2.2	3.0	4.5							15	19	27	0.2	*	0.3

<<<<<<<< ---------------------------------- RATIO OF LIPID NORMALIZED CONCENTRATIONS: TEQ BASIS TO LOAEL---------------------------------->>>>>

	Largemouth Bass 113			Largemouth Bass 168			Brown Bullhead 168			Largemouth Bass 189			Brown Bullhead 189		
	Average	95% UCL	Maximum	Average	95% UCL	Maximum	Average	95% UCL	Maximum	Average	95% UCL	Maximum	Average	95% UCL	Maximum
1993	1.2	2.5	2.0	4.6	5.4	7.2	0.1	*	0.1	20	31	45	0.3	0.5	0.5
1994	1.4	2.1	4.0	4.4	8.1	10.1	0.1	0.1	0.2	11	14	23	0.2	0.3	0.9
1995	1.3	1.6	2.1	5.1	6.0	8.9	0.0	0.1	0.1	9.8	15	17	0.1	0.1	0.2
1996	1.1	1.5	2.2							7.1	9.0	13	0.1	*	0.1

304305

304306

TABLE 5-25: RATIO OF OBSERVED WHITE PERCH AND YELLOW PERCH CONCENTRATIONS TO TOXICITY BENCHMARKS USING NYSDEC DATASET
REVISED

<<<<<<<<<<<<<<< ---------- RATIO OF WET WEIGHT CONCENTRATION TO NOAEL ---------- >>>>>>>>>>>

	White Perch 113			White Perch 152			Yellow Perch 113			Yellow Perch 168			Yellow Perch 189		
	Average	95% UCL	Maximum	Average	95% UCL	Maximum	Average	95% UCL	Maximum	Average	95% UCL	Maximum	Average	95% UCL	Maximum
1993	0.8	*	**1.8**	0.8	*	**1.2**	**5.6**	**6.6**	**17**	**55**	*	**112**	**191**	**349**	**691**
1994	0.3	*	0.7	**1.6**	*	**2.8**	**2.7**	**3.5**	**4.2**						
1995															
1996	**1.6**	*	**2.9**	0.9	*	**2.6**									

<<<<<<<<<<<<<<< ---------- RATIO OF WET WEIGHT CONCENTRATION TO LOAEL ---------- >>>>>>>>>>>

	White Perch 113			White Perch 152			Yellow Perch 113			Yellow Perch 168			Yellow Perch 189		
	Average	95% UCL	Maximum	Average	95% UCL	Maximum	Average	95% UCL	Maximum	Average	95% UCL	Maximum	Average	95% UCL	Maximum
1993	NA	NA	NA	NA	NA	NA	**1.1**	**1.4**	**3.5**	**11**	*	**23**	**39**	**71**	**141**
1994	NA	NA	NA	NA	NA	NA	0.6	0.7	0.9						
1995															
1996	NA	NA	NA	NA	NA										

<<<<<<<<<<<<<<< ----------RATIO OF LIPID NORMALIZED CONCENTRATIONS: TEQ BASIS TO NOAEL ---------- >>>>>>>>>>>

	White Perch 113			White Perch 152			Yellow Perch 113			Yellow Perch 168			Yellow Perch 189		
	Average	95% UCL	Maximum	Average	95% UCL	Maximum	Average	95% UCL	Maximum	Average	95% UCL	Maximum	Average	95% UCL	Maximum
1993	**2.3**	**3.3**	**4.1**	**2.0**	**2.3**	**3.0**	**2.0**	**2.5**	**8.0**	**9.0**	*	**13**	**51**	**75**	**255**
1994	**2.1**	**3.0**	**3.6**	**5.8**	**7.1**	**12**	**1.0**	**1.3**	**1.4**						
1995															
1996	**2.8**	**7.3**	**6.5**												

<<<<<<<<<<<<<< ---------- RATIO OF LIPID NORMALIZED CONCENTRATIONS: TEQ BASIS TO LOAEL---------- >>>>>>>>>

	White Perch 113			White Perch 152			Yellow Perch 113			Yellow Perch 168			Yellow Perch 189		
	Average	95% UCL	Maximum	Average	95% UCL	Maximum	Average	95% UCL	Maximum	Average	95% UCL	Maximum	Average	95% UCL	Maximum
1993	**1.1**	**1.6**	**2.0**	**1.0**	**1.1**	**1.5**	**1.0**	**1.2**	**3.9**	**4.4**	*	**6.5**	**25**	**36**	**123**
1994	**1.0**	**1.5**	**1.8**	**2.8**	**3.4**	**5.9**	0.5	0.6	0.7						
1995															
1996	**1.4**	**3.5**	**3.2**												

TABLE 5-26: RATIO OF PREDICTED WHITE PERCH CONCENTRATIONS TO FIELD-BASED NOAEL FOR TRI+ PCBS
REVISED

	River Mile 189			River Mile 168			River Mile 154		
Year	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)
1993	**4.7**	**5.9**	**10.4**	**1.8**	**2.3**	**5.1**	0.8	**1.0**	**2.1**
1994	**3.8**	**4.9**	**8.6**	**1.7**	**2.1**	**4.8**	0.7	0.9	**2.0**
1995	**3.8**	**4.8**	**8.3**	**1.5**	**1.9**	**4.3**	0.7	0.8	**1.8**
1996	**2.8**	**3.6**	**6.4**	**1.3**	**1.6**	**3.6**	0.6	0.7	**1.6**
1997	**2.3**	**3.0**	**5.3**	**1.1**	**1.5**	**3.4**	0.7	0.9	**1.3**
1998	**2.0**	**2.5**	**4.2**	**1.1**	**1.5**	**3.4**	0.4	0.6	**1.3**
1999	**1.8**	**2.3**	**3.8**	**1.0**	**1.4**	**3.1**	0.4	0.6	**1.2**
2000	**1.5**	**1.9**	**3.3**	0.9	**1.1**	**2.6**	0.4	0.5	**1.1**
2001	**1.4**	**1.7**	**2.9**	0.7	**1.0**	**2.2**	0.3	0.4	0.9
2002	**1.3**	**1.7**	**2.8**	0.7	0.9	**1.9**	0.3	0.4	0.9
2003	**1.2**	**1.5**	**2.5**	0.6	0.8	**1.8**	0.3	0.4	0.8
2004	**1.2**	**1.5**	**2.5**	0.6	0.8	**1.7**	0.3	0.3	0.7
2005	**1.1**	**1.4**	**2.3**	0.5	0.7	**1.6**	0.2	0.3	0.6
2006	**1.0**	**1.2**	**2.1**	0.5	0.6	**1.4**	0.2	0.3	0.6
2007	0.9	**1.2**	**1.9**	0.4	0.6	**1.3**	0.2	0.2	0.5
2008	0.9	**1.1**	**1.9**	0.4	0.5	**1.2**	0.2	0.2	0.5
2009	0.9	**1.1**	**1.8**	0.4	0.5	**1.2**	0.2	0.2	0.5
2010	0.8	**1.0**	**1.7**	0.4	0.5	**1.1**	0.1	0.2	0.4
2011	0.7	0.9	**1.5**	0.3	0.4	**1.0**	0.1	0.2	0.4
2012	0.7	0.8	**1.4**	0.3	0.4	0.9	0.1	0.2	0.3
2013	0.6	0.7	**1.2**	0.3	0.4	0.9	0.1	0.1	0.3
2014	0.5	0.7	**1.2**	0.3	0.4	0.9	0.1	0.1	0.3
2015	0.5	0.6	**1.1**	0.3	0.4	0.9	0.1	0.1	0.3
2016	0.5	0.6	**1.1**	0.3	0.4	0.9	0.1	0.1	0.3
2017	0.5	0.6	**1.1**	0.3	0.4	0.9	0.1	0.1	0.3
2018	0.5	0.6	**1.1**	0.3	0.4	0.8	0.1	0.1	0.3

Bold values indicate exceedances

304307

304308

TABLE 5-27: RATIO OF PREDICTED YELLOW PERCH CONCENTRATIONS TO LABORATORY-DERIVED NOAEL FOR TRI+ PCBS
REVISED

	River Mile 189			River Mile 168			River Mile 154		
Year	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)
1993	**82**	**105**	**174**	**11**	**16**	**33**	**4.7**	**6.6**	**14**
1994	**61**	**78**	**132**	**11**	**14**	**32**	**4.5**	**6.5**	**13**
1995	**63**	**80**	**130**	**9.2**	**13**	**28**	**4.1**	**5.8**	**12**
1996	**36**	**46**	**80**	**7.3**	**10**	**22**	**3.4**	**4.8**	**10**
1997	**31**	**39**	**67**	**3.5**	**5.9**	**15**	**1.8**	**2.9**	**6.8**
1998	**30**	**37**	**61**	**7.3**	**10**	**22**	**2.8**	**4.0**	**9.1**
1999	**28**	**35**	**57**	**6.6**	**8.9**	**20**	**2.6**	**3.8**	**8.2**
2000	**24**	**31**	**48**	**5.4**	**7.1**	**16**	**2.4**	**3.2**	**7.1**
2001	**19**	**25**	**41**	**4.7**	**6.3**	**14**	**2.0**	**2.9**	**6.4**
2002	**19**	**24**	**40**	**4.2**	**5.6**	**13**	**1.9**	**2.7**	**5.9**
2003	**18**	**23**	**41**	**3.6**	**5.2**	**11**	**1.7**	**2.4**	**5.2**
2004	**20**	**25**	**41**	**3.7**	**5.1**	**11**	**1.6**	**2.3**	**5.0**
2005	**19**	**23**	**37**	**3.4**	**4.5**	**10**	**1.4**	**2.0**	**4.4**
2006	**16**	**20**	**33**	**3.0**	**4.1**	**9.0**	**1.3**	**1.8**	**4.0**
2007	**16**	**20**	**32**	**2.8**	**3.7**	**8.3**	**1.2**	**1.7**	**3.6**
2008	**16**	**21**	**34**	**2.7**	**3.7**	**8.0**	**1.1**	**1.6**	**3.3**
2009	**16**	**20**	**34**	**2.6**	**3.5**	**7.6**	**1.0**	**1.5**	**3.1**
2010	**15**	**19**	**29**	**2.3**	**3.0**	**6.8**	0.9	**1.3**	**2.8**
2011	**13**	**16**	**25**	**2.0**	**2.8**	**6.1**	0.8	**1.2**	**2.6**
2012	**12**	**15**	**25**	**1.9**	**2.6**	**5.8**	0.8	**1.1**	**2.4**
2013	**11**	**14**	**22**	**1.8**	**2.5**	**5.4**	0.7	**1.0**	**2.1**
2014	**11**	**13**	**21**	**1.9**	**2.6**	**5.7**	0.7	0.9	**2.0**
2015	**9.3**	**12**	**19**	**1.9**	**2.6**	**5.6**	0.6	0.9	**1.9**
2016	**11**	**14**	**23**	**1.9**	**2.6**	**5.7**	0.6	0.9	**1.9**
2017	**11**	**13**	**22**	**1.9**	**2.5**	**5.5**	0.6	0.9	**1.8**
2018	**11**	**13**	**22**	**1.9**	**2.6**	**5.6**	0.6	0.9	**1.9**

Bold values indicate exceedances

TABLE 5-28: RATIO OF PREDICTED YELLOW PERCH CONCENTRATIONS TO LABORATORY-DERIVED LOAEL FOR TRI+ PCBS REVISED

	River Mile 189			River Mile 168			River Mile 154		
Year	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)
1993	**17**	**21**	**36**	**2.2**	**3.2**	**6.8**	**1.0**	**1.3**	**2.8**
1994	**12**	**16**	**27**	**2.2**	**2.9**	**6.5**	0.9	**1.3**	**2.7**
1995	**13**	**16**	**27**	**1.9**	**2.7**	**5.6**	0.8	**1.2**	**2.4**
1996	**7.4**	**9.5**	**16**	**1.5**	**2.1**	**4.6**	0.7	**1.0**	**2.1**
1997	**6.3**	**8.0**	**14**	0.7	**1.2**	**3.0**	0.4	0.6	**1.4**
1998	**6.0**	**7.6**	**13**	**1.5**	**2.0**	**4.4**	0.6	0.8	**1.9**
1999	**5.7**	**7.2**	**12**	**1.3**	**1.8**	**4.1**	0.5	0.8	**1.7**
2000	**4.9**	**6.2**	**10**	**1.1**	**1.5**	**3.3**	0.5	0.7	**1.5**
2001	**4.0**	**5.0**	**8.4**	**1.0**	**1.3**	**2.9**	0.4	0.6	**1.3**
2002	**4.0**	**5.0**	**8.1**	0.8	**1.1**	**2.6**	0.4	0.6	**1.2**
2003	**3.7**	**4.8**	**8.3**	0.7	**1.1**	**2.2**	0.4	0.5	**1.1**
2004	**4.1**	**5.2**	**8.4**	0.8	**1.0**	**2.3**	0.3	0.5	**1.0**
2005	**3.9**	**4.8**	**7.6**	0.7	0.9	**2.0**	0.3	0.4	0.9
2006	**3.2**	**4.0**	**6.8**	0.6	0.8	**1.8**	0.3	0.4	0.8
2007	**3.2**	**4.1**	**6.6**	0.6	0.8	**1.7**	0.2	0.3	0.7
2008	**3.3**	**4.2**	**6.9**	0.5	0.7	**1.6**	0.2	0.3	0.7
2009	**3.2**	**4.1**	**6.9**	0.5	0.7	**1.6**	0.2	0.3	0.6
2010	**3.1**	**3.8**	**5.9**	0.5	0.6	**1.4**	0.2	0.3	0.6
2011	**2.6**	**3.2**	**5.2**	0.4	0.6	**1.3**	0.2	0.2	0.5
2012	**2.5**	**3.2**	**5.0**	0.4	0.5	**1.2**	0.2	0.2	0.5
2013	**2.2**	**2.8**	**4.5**	0.4	0.5	**1.1**	0.1	0.2	0.4
2014	**2.2**	**2.7**	**4.2**	0.4	0.5	**1.2**	0.1	0.2	0.4
2015	**1.9**	**2.4**	**3.8**	0.4	0.5	**1.1**	0.1	0.2	0.4
2016	**2.2**	**2.8**	**4.6**	0.4	0.5	**1.2**	0.1	0.2	0.4
2017	**2.2**	**2.7**	**4.4**	0.4	0.5	**1.1**	0.1	0.2	0.4
2018	**2.2**	**2.7**	**4.5**	0.4	0.5	**1.1**	0.1	0.2	0.4

Bold values indicate exceedances

304309

304310

TABLE 5-29: RATIO OF PREDICTED WHITE PERCH CONCENTRATIONS TO LABORATORY-DERIVED NOAEL ON A TEQ BASIS
REVISED

	River Mile 189			River Mile 168			River Mile 154		
Year	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)
1993	**4.0**	**5.4**	**10.3**	**1.6**	**2.0**	**4.8**	0.6	0.9	**2.3**
1994	**3.2**	**4.5**	**8.6**	**1.5**	**1.9**	**4.5**	0.6	0.8	**2.1**
1995	**3.2**	**4.4**	**8.4**	**1.3**	**1.7**	**4.0**	0.5	0.7	**1.9**
1996	**2.4**	**3.4**	**6.4**	**1.1**	**1.4**	**3.4**	0.5	0.6	**1.7**
1997	**2.0**	**2.8**	**5.3**	**1.0**	**1.3**	**3.1**	0.4	0.6	**1.5**
1998	**1.7**	**2.4**	**4.3**	**2.6**	**3.5**	**6.8**	0.4	0.5	**1.3**
1999	**1.5**	**2.1**	**3.9**	**2.4**	**3.2**	**6.3**	0.3	0.5	**1.2**
2000	**1.3**	**1.8**	**3.4**	**1.7**	**2.3**	**4.5**	0.3	0.4	**1.1**
2001	**1.1**	**1.6**	**3.0**	**1.8**	**2.3**	**4.7**	0.3	0.4	**1.0**
2002	**1.1**	**1.5**	**2.8**	**1.6**	**2.1**	**4.1**	0.3	0.3	0.9
2003	**1.0**	**1.4**	**2.5**	**1.8**	**2.4**	**4.8**	0.2	0.3	0.8
2004	**1.0**	**1.4**	**2.5**	**1.8**	**2.3**	**4.6**	0.2	0.3	0.8
2005	0.9	**1.3**	**2.4**	**1.4**	**1.9**	**3.8**	0.2	0.3	0.7
2006	0.8	**1.1**	**2.1**	**1.4**	**1.9**	**3.8**	0.2	0.2	0.6
2007	0.8	**1.1**	**2.0**	**1.4**	**1.8**	**3.6**	0.2	0.2	0.5
2008	0.7	**1.0**	**1.9**	**1.5**	**1.9**	**3.8**	0.1	0.2	0.5
2009	0.7	**1.0**	**1.8**	**1.4**	**1.9**	**3.7**	0.1	0.2	0.5
2010	0.7	0.9	**1.7**	**1.1**	**1.5**	**2.9**	0.1	0.2	0.4
2011	0.6	0.8	**1.5**	**1.1**	**1.5**	**2.9**	0.1	0.2	0.4
2012	0.6	0.8	**1.4**	**1.0**	**1.4**	**2.6**	0.1	0.1	0.4
2013	0.5	0.7	**1.3**	**1.0**	**1.3**	**2.5**	0.1	0.1	0.3
2014	0.5	0.6	**1.2**	0.8	**1.1**	**2.2**	0.1	0.1	0.3
2015	0.4	0.6	**1.1**	0.8	**1.1**	**2.1**	0.1	0.1	0.3
2016	0.4	0.6	**1.1**	**1.0**	**1.3**	**2.5**	0.1	0.1	0.3
2017	0.4	0.6	**1.1**	0.9	**1.3**	**2.4**	0.1	0.1	0.3
2018	0.4	0.6	**1.0**	**1.0**	**1.3**	**2.5**	0.1	0.1	0.3

Bold values indicate exceedances

TABLE 5-30: RATIO OF PREDICTED WHITE PERCH CONCENTRATIONS TO LABORATORY-DERIVED LOAEL ON A TEQ BASIS
REVISED

	River Mile 189			River Mile 168			River Mile 154		
Year	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)
1993	**1.9**	**2.6**	**5.0**	0.8	**1.0**	**2.3**	0.3	0.4	**1.1**
1994	**1.6**	**2.2**	**4.1**	0.7	0.9	**2.2**	0.3	0.4	**1.0**
1995	**1.5**	**2.1**	**4.0**	0.6	0.8	**1.9**	0.3	0.4	0.9
1996	**1.2**	**1.6**	**3.1**	0.5	0.7	**1.7**	0.2	0.3	0.8
1997	0.9	**1.3**	**2.6**	0.5	0.6	**1.5**	0.2	0.3	0.7
1998	0.8	**1.1**	**2.1**	**1.3**	**1.7**	**3.3**	0.2	0.2	0.6
1999	0.7	**1.0**	**1.9**	**1.2**	**1.5**	**3.0**	0.2	0.2	0.6
2000	0.6	0.9	**1.6**	0.8	**1.1**	**2.2**	0.1	0.2	0.5
2001	0.6	0.8	**1.4**	0.9	**1.1**	**2.3**	0.1	0.2	0.5
2002	0.5	0.7	**1.4**	0.8	**1.0**	**2.0**	0.1	0.2	0.4
2003	0.5	0.7	**1.2**	0.9	**1.2**	**2.3**	0.1	0.2	0.4
2004	0.5	0.7	**1.2**	0.9	**1.1**	**2.2**	0.1	0.1	0.4
2005	0.4	0.6	**1.1**	0.7	0.9	**1.8**	0.1	0.1	0.3
2006	0.4	0.5	**1.0**	0.7	0.9	**1.8**	0.1	0.1	0.3
2007	0.4	0.5	**1.0**	0.7	0.9	**1.7**	0.1	0.1	0.3
2008	0.4	0.5	0.9	0.7	0.9	**1.8**	0.1	0.1	0.2
2009	0.4	0.5	0.9	0.7	0.9	**1.8**	0.06	0.09	0.2
2010	0.3	0.4	0.8	0.5	0.7	**1.4**	0.06	0.08	0.2
2011	0.3	0.4	0.7	0.5	0.7	**1.4**	0.05	0.07	0.2
2012	0.3	0.4	0.7	0.5	0.7	**1.3**	0.05	0.07	0.2
2013	0.2	0.3	0.6	0.5	0.6	**1.2**	0.04	0.06	0.2
2014	0.2	0.3	0.6	0.4	0.5	**1.0**	0.04	0.06	0.1
2015	0.2	0.3	0.5	0.4	0.5	**1.0**	0.04	0.06	0.1
2016	0.2	0.3	0.5	0.5	0.6	**1.2**	0.04	0.05	0.1
2017	0.2	0.3	0.5	0.5	0.6	**1.2**	0.04	0.05	0.1
2018	0.2	0.3	0.5	0.5	0.6	**1.2**	0.04	0.05	0.1

Bold values indicate exceedances

304311

304312

TABLE 5-31: RATIO OF PREDICTED YELLOW PERCH CONCENTRATIONS TO LABORATORY-DERIVED NOAEL ON A TEQ BASIS
REVISED

	River Mile 189			River Mile 168			River Mile 154		
Year	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)
1993	**7.4**	**9.9**	**19.4**	**2.8**	**3.9**	**8.4**	**1.2**	**1.6**	**3.5**
1994	**5.5**	**7.3**	**14.8**	**2.7**	**3.6**	**8.1**	**1.2**	**1.6**	**3.4**
1995	**5.7**	**7.5**	**14.8**	**2.3**	**3.3**	**7.0**	**1.0**	**1.4**	**3.0**
1996	**3.3**	**4.4**	**8.9**	**1.9**	**2.6**	**5.8**	0.9	**1.2**	**2.6**
1997	**2.8**	**3.7**	**7.7**	**1.8**	**2.7**	**5.9**	0.8	**1.2**	**2.7**
1998	**2.7**	**3.6**	**7.1**	**1.9**	**2.5**	**5.6**	0.7	**1.0**	**2.3**
1999	**2.5**	**3.3**	**6.4**	**1.7**	**2.3**	**5.1**	0.7	**1.0**	**2.1**
2000	**2.1**	**2.9**	**5.3**	**1.3**	**1.8**	**4.3**	0.6	0.8	**1.8**
2001	**1.8**	**2.3**	**4.6**	**1.2**	**1.6**	**3.7**	0.5	0.7	**1.6**
2002	**1.7**	**2.3**	**4.5**	**1.1**	**1.4**	**3.2**	0.5	0.7	**1.5**
2003	**1.7**	**2.2**	**4.6**	0.9	**1.3**	**2.8**	0.4	0.6	**1.3**
2004	**1.8**	**2.4**	**4.7**	0.9	**1.3**	**2.8**	0.4	0.6	**1.3**
2005	**1.7**	**2.2**	**4.2**	0.8	**1.1**	**2.6**	0.4	0.5	**1.1**
2006	**1.4**	**1.9**	**3.7**	0.7	**1.0**	**2.3**	0.3	0.5	**1.0**
2007	**1.4**	**1.9**	**3.7**	0.7	0.9	**2.1**	0.3	0.4	0.9
2008	**1.5**	**1.9**	**3.8**	0.7	0.9	**2.1**	0.3	0.4	0.8
2009	**1.4**	**1.9**	**3.7**	0.6	0.9	**2.0**	0.3	0.4	0.8
2010	**1.4**	**1.8**	**3.3**	0.6	0.8	**1.8**	0.2	0.3	0.7
2011	**1.1**	**1.5**	**2.9**	0.5	0.7	**1.6**	0.2	0.3	0.7
2012	**1.1**	**1.5**	**2.8**	0.5	0.7	**1.5**	0.2	0.3	0.6
2013	**1.0**	**1.3**	**2.5**	0.5	0.6	**1.4**	0.2	0.2	0.5
2014	0.9	**1.2**	**2.3**	0.5	0.7	**1.5**	0.2	0.2	0.5
2015	0.8	**1.1**	**2.1**	0.5	0.7	**1.4**	0.2	0.2	0.5
2016	**1.0**	**1.3**	**2.5**	0.5	0.7	**1.5**	0.2	0.2	0.5
2017	**1.0**	**1.3**	**2.4**	0.5	0.6	**1.4**	0.2	0.2	0.5
2018	**1.0**	**1.3**	**2.5**	0.5	0.6	**1.4**	0.2	0.2	0.5

Bold values indicate exceedances

TABLE 5-32: RATIO OF PREDICTED YELLOW PERCH CONCENTRATIONS TO LABORATORY-DERIVED LOAEL ON A TEQ BASIS
REVISED

	River Mile 189			River Mile 168			River Mile 154		
Year	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)
1993	**3.6**	**4.8**	**9.4**	**1.4**	**1.9**	**4.1**	0.6	0.8	**1.7**
1994	**2.7**	**3.5**	**7.2**	**1.3**	**1.8**	**3.9**	0.6	0.8	**1.6**
1995	**2.7**	**3.6**	**7.1**	**1.1**	**1.6**	**3.4**	0.5	0.7	**1.5**
1996	**1.6**	**2.1**	**4.3**	0.9	**1.3**	**2.8**	0.4	0.6	**1.3**
1997	**1.4**	**1.8**	**3.7**	0.9	**1.3**	**2.9**	0.4	0.6	**1.3**
1998	**1.3**	**1.7**	**3.4**	0.9	**1.2**	**2.7**	0.4	0.5	**1.1**
1999	**1.2**	**1.6**	**3.1**	0.8	**1.1**	**2.5**	0.3	0.5	**1.0**
2000	**1.0**	**1.4**	**2.6**	0.6	0.9	**2.1**	0.3	0.4	0.9
2001	0.8	**1.1**	**2.2**	0.6	0.8	**1.8**	0.3	0.4	0.8
2002	0.8	**1.1**	**2.2**	0.5	0.7	**1.6**	0.2	0.3	0.7
2003	0.8	**1.1**	**2.2**	0.4	0.6	**1.4**	0.2	0.3	0.6
2004	0.9	**1.2**	**2.3**	0.5	0.6	**1.4**	0.2	0.3	0.6
2005	0.8	**1.1**	**2.0**	0.4	0.5	**1.3**	0.2	0.2	0.5
2006	0.7	0.9	**1.8**	0.4	0.5	**1.1**	0.2	0.2	0.5
2007	0.7	0.9	**1.8**	0.3	0.5	**1.0**	0.1	0.2	0.4
2008	0.7	0.9	**1.8**	0.3	0.4	**1.0**	0.1	0.2	0.4
2009	0.7	0.9	**1.8**	0.3	0.4	0.9	0.1	0.2	0.4
2010	0.7	0.9	**1.6**	0.3	0.4	0.9	0.1	0.2	0.3
2011	0.5	0.7	**1.4**	0.2	0.3	0.8	0.1	0.1	0.3
2012	0.5	0.7	**1.3**	0.2	0.3	0.7	0.1	0.1	0.3
2013	0.5	0.6	**1.2**	0.2	0.3	0.7	0.1	0.1	0.3
2014	0.5	0.6	**1.1**	0.2	0.3	0.7	0.1	0.1	0.2
2015	0.4	0.5	**1.0**	0.2	0.3	0.7	0.1	0.1	0.2
2016	0.5	0.6	**1.2**	0.2	0.3	0.7	0.1	0.1	0.2
2017	0.5	0.6	**1.2**	0.2	0.3	0.7	0.1	0.1	0.2
2018	0.5	0.6	**1.2**	0.2	0.3	0.7	0.1	0.1	0.2

Bold values indicate exceedances

304314

TABLE 5-33: RATIO OF PREDICTED LARGEMOUTH BASS CONCENTRATIONS TO FIELD-BASED NOAEL FOR TRI+ PCBS
REVISED

	River Mile 189			River Mile 168			River Mile 154		
Year	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)
1993	**86**	**141**	**403**	**30**	**35**	**46**	**14**	**16**	**21**
1994	**65**	**103**	**252**	**28**	**33**	**42**	**13**	**15**	**20**
1995	**78**	**124**	**260**	**26**	**31**	**40**	**12**	**14**	**18**
1996	**39**	**62**	**167**	**20**	**24**	**31**	**10**	**12**	**15**
1997	**35**	**55**	**133**	**19**	**23**	**32**	**4.2**	**7**	**11**
1998	**37**	**55**	**131**	**18**	**22**	**29**	**7.9**	**10**	**13**
1999	**38**	**57**	**133**	**17**	**20**	**26**	**7.0**	**8.7**	**11**
2000	**26**	**41**	**105**	**15**	**19**	**24**	**6.5**	**8.1**	**11**
2001	**24**	**37**	**87**	**12**	**14**	**19**	**5.5**	**6.8**	**8.9**
2002	**25**	**38**	**87**	**11**	**12**	**16**	**5.1**	**6.1**	**8.1**
2003	**23**	**35**	**82**	**9.4**	**11**	**15**	**4.5**	**5.6**	**7.3**
2004	**27**	**40**	**89**	**9.5**	**11**	**15**	**4.4**	**5.4**	**7.0**
2005	**22**	**34**	**83**	**8.7**	**10**	**14**	**3.9**	**4.8**	**6.2**
2006	**18**	**28**	**68**	**7.8**	**9.2**	**12**	**3.5**	**4.2**	**5.6**
2007	**19**	**30**	**67**	**7.3**	**8.6**	**11**	**3.2**	**3.9**	**5.1**
2008	**20**	**31**	**72**	**6.8**	**8.1**	**11**	**3.0**	**3.6**	**4.7**
2009	**20**	**31**	**73**	**6.6**	**8.0**	**10**	**2.7**	**3.4**	**4.4**
2010	**16**	**26**	**64**	**6.0**	**7.1**	**9.4**	**2.5**	**3.1**	**4.1**
2011	**16**	**24**	**55**	**5.4**	**6.4**	**8.4**	**2.3**	**2.9**	**3.7**
2012	**16**	**25**	**55**	**5.0**	**5.9**	**7.8**	**2.2**	**2.6**	**3.4**
2013	**13**	**20**	**49**	**4.5**	**5.5**	**7.3**	**1.8**	**2.3**	**3.0**
2014	**12**	**18**	**44**	**4.8**	**5.7**	**7.5**	**1.8**	**2.2**	**2.9**
2015	**11**	**17**	**40**	**4.9**	**5.7**	**7.5**	**1.7**	**2.1**	**2.8**
2016	**13**	**19**	**47**	**4.8**	**5.7**	**7.5**	**1.7**	**2.1**	**2.7**
2017	**14**	**22**	**49**	**4.7**	**5.6**	**7.3**	**1.7**	**2.0**	**2.7**
2018	**13**	**21**	**47**	**4.6**	**5.4**	**7.1**	**1.6**	**2.0**	**2.6**

TABLE 5-34: RATIO OF PREDICTED LARGEMOUTH BASS CONCENTRATIONS TO LABORATORY-DERIVED NOAEL ON A TEQ BASIS
REVISED

	River Mile 189			River Mile 168			River Mile 154		
Year	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)
1993	**7.2**	**12.2**	**38.1**	**4.8**	**5.6**	**7.3**	**2.2**	**2.6**	**3.4**
1994	**5.5**	**8.8**	**23.8**	**4.4**	**5.1**	**6.7**	**2.0**	**2.4**	**3.1**
1995	**6.5**	**10.4**	**23.5**	**4.1**	**4.8**	**6.3**	**1.9**	**2.2**	**2.9**
1996	**3.3**	**5.3**	**16.3**	**3.2**	**3.7**	**4.9**	**1.6**	**1.8**	**2.4**
1997	**2.9**	**4.8**	**12.3**	**3.0**	**3.7**	**4.8**	**1.4**	**1.8**	**2.3**
1998	**3.1**	**4.8**	**13.0**	**2.9**	**3.2**	**4.4**	**1.2**	**1.5**	**1.9**
1999	**3.1**	**4.9**	**12.4**	**2.6**	**3.1**	**4.1**	**1.1**	**1.4**	**1.8**
2000	**2.1**	**3.4**	**9.7**	**2.2**	**2.7**	**3.5**	**1.0**	**1.2**	**1.6**
2001	**2.1**	**3.1**	**8.2**	**1.9**	**2.2**	**3.0**	0.9	**1.1**	**1.4**
2002	**2.1**	**3.2**	**8.1**	**1.7**	**1.9**	**2.6**	0.8	**1.0**	**1.3**
2003	**1.9**	**2.9**	**7.7**	**1.5**	**1.8**	**2.3**	0.7	0.9	**1.2**
2004	**2.2**	**3.3**	**8.2**	**1.5**	**1.8**	**2.3**	0.7	0.8	**1.1**
2005	**1.8**	**3.0**	**7.7**	**1.4**	**1.6**	**2.1**	0.6	0.7	**1.0**
2006	**1.5**	**2.4**	**6.3**	**1.2**	**1.4**	**1.9**	0.6	0.7	0.9
2007	**1.6**	**2.5**	**6.0**	**1.1**	**1.3**	**1.8**	0.5	0.6	0.8
2008	**1.7**	**2.6**	**6.6**	**1.1**	**1.3**	**1.7**	0.5	0.6	0.7
2009	**1.7**	**2.7**	**6.6**	**1.0**	**1.2**	**1.6**	0.4	0.5	0.7
2010	**1.3**	**2.2**	**6.0**	0.9	**1.1**	**1.5**	0.4	0.5	0.6
2011	**1.3**	**2.0**	**5.1**	0.8	**1.0**	**1.3**	0.4	0.4	0.6
2012	**1.3**	**2.1**	**5.1**	0.8	0.9	**1.2**	0.3	0.4	0.5
2013	**1.1**	**1.7**	**4.6**	0.7	0.9	**1.1**	0.3	0.4	0.5
2014	**1.0**	**1.6**	**4.1**	0.8	0.9	**1.2**	0.3	0.3	0.5
2015	0.9	**1.5**	**3.7**	0.8	0.9	**1.2**	0.3	0.3	0.4
2016	**1.1**	**1.6**	**4.2**	0.8	0.9	**1.2**	0.3	0.3	0.4
2017	**1.2**	**1.9**	**4.5**	0.7	0.9	**1.2**	0.3	0.3	0.4
2018	**1.1**	**1.8**	**4.4**	0.7	0.9	**1.2**	0.3	0.3	0.4

Bold values indicate exceedances

304315

304316

TABLE 5-35: RATIO OF PREDICTED LARGEMOUTH BASS CONCENTRATIONS TO LABORATORY-DERIVED LOAEL ON A TEQ BASIS
REVISED

	River Mile 189			River Mile 168			River Mile 154		
Year	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)
1993	**3.5**	**5.9**	**18.4**	**2.3**	**2.7**	**3.5**	**1.1**	**1.2**	**1.6**
1994	**2.7**	**4.2**	**11.5**	**2.1**	**2.5**	**3.2**	**1.0**	**1.2**	**1.5**
1995	**3.1**	**5.0**	**11.4**	**2.0**	**2.3**	**3.0**	0.9	**1.1**	**1.4**
1996	**1.6**	**2.6**	**7.9**	**1.5**	**1.8**	**2.4**	0.8	0.9	**1.2**
1997	**1.4**	**2.3**	**5.9**	**1.4**	**1.8**	**2.3**	0.7	0.9	**1.1**
1998	**1.5**	**2.3**	**6.3**	**1.4**	**1.6**	**2.1**	0.6	0.7	0.9
1999	**1.5**	**2.3**	**6.0**	**1.3**	**1.5**	**2.0**	0.5	0.7	0.9
2000	**1.0**	**1.6**	**4.7**	**1.1**	**1.3**	**1.7**	0.5	0.6	0.8
2001	**1.0**	**1.5**	**4.0**	0.9	**1.1**	**1.4**	0.4	0.5	0.7
2002	**1.0**	**1.5**	**3.9**	0.8	0.9	**1.3**	0.4	0.5	0.6
2003	0.9	**1.4**	**3.7**	0.7	0.9	**1.1**	0.3	0.4	0.6
2004	**1.1**	**1.6**	**4.0**	0.7	0.8	**1.1**	0.3	0.4	0.5
2005	0.9	**1.4**	**3.7**	0.7	0.8	**1.0**	0.3	0.4	0.5
2006	0.7	**1.2**	**3.1**	0.6	0.7	0.9	0.3	0.3	0.4
2007	0.8	**1.2**	**2.9**	0.6	0.7	0.9	0.2	0.3	0.4
2008	0.8	**1.3**	**3.2**	0.5	0.6	0.8	0.2	0.3	0.4
2009	0.8	**1.3**	**3.2**	0.5	0.6	0.8	0.2	0.3	0.3
2010	0.7	**1.1**	**2.9**	0.5	0.5	0.7	0.2	0.2	0.3
2011	0.6	**1.0**	**2.5**	0.4	0.5	0.6	0.2	0.2	0.3
2012	0.6	**1.0**	**2.4**	0.4	0.4	0.6	0.2	0.2	0.3
2013	0.5	0.8	**2.2**	0.3	0.4	0.6	0.1	0.2	0.2
2014	0.5	0.8	**2.0**	0.4	0.4	0.6	0.1	0.2	0.2
2015	0.4	0.7	**1.8**	0.4	0.4	0.6	0.1	0.2	0.2
2016	0.5	0.8	**2.0**	0.4	0.4	0.6	0.1	0.2	0.2
2017	0.6	0.9	**2.2**	0.4	0.4	0.6	0.1	0.2	0.2
2018	0.5	0.8	**2.1**	0.4	0.4	0.6	0.1	0.2	0.2

Bold values indicate exceedances

TABLE 5-36
COMPARISON OF MEASURED STRIPED BASS CONCENTRATIONS TO TOXICITY REFERENCE VALUES (UNCHANGED)

		<<--- Tri+ in Tissue -->>		<< -- TEQ in Eggs: Lipid Normalized -->>			
		Field-Based NOAEL		LOAEL		NOAEL	
River Mile	Year	Average	95%UCL	Average	95% UCL	Average	95% UCL
12	1993	0.4	0.7	0.2	0.4	0.5	0.7
27	1993	0.8	**1.3**	0.4	0.6	0.9	**1.3**
33	1993	**1.4**	**2.7**	0.5	0.9	**1.0**	**1.8**
40	1993	0.5	0.7	0.3	0.4	0.6	0.8
74	1993	**1.0**	**1.6**	0.7	**1.2**	**1.4**	**2.5**
112	1993	**1.2**	**1.8**	**1.1**	**1.9**	**2.3**	**3.9**
152	1993	**4.0**	**5.7**	**2.8**	**4.9**	**5.8**	**10**
26	1994	0.5	0.8	0.3	0.5	0.7	**1.0**
37	1994	0.6	**1.0**	0.4	0.8	0.9	**1.6**
40	1994	0.6	0.9	0.4	0.5	0.8	**1.1**
74	1994	0.8	**1.4**	0.5	0.7	0.9	**1.4**
112	1994	**1.0**	**2.7**	**1.2**	**4.3**	**2.5**	**9.0**
152	1994	**2.1**	**3.2**	**2.3**	**3.1**	**4.8**	**6.4**
27	1995	0.6	0.9	0.5	0.8	**1.0**	**1.7**
36	1995	0.4	0.5	0.2	0.3	0.5	0.7
59	1995	0.7	0.9	**1.5**	**2.3**	**3.1**	**4.7**
76	1995	0.6	0.7	0.3	0.3	0.6	0.7
113	1995	0.5	0.9	0.3	0.5	0.7	**1.1**
152	1995	**2.1**	**2.7**	**1.6**	**2.1**	**3.3**	**4.4**
12	1996	0.4	0.6	0.3	0.5	0.7	**1.0**
29	1996	0.6	0.9	0.4	0.5	0.8	**1.1**
40	1996	0.5	0.7	0.4	0.7	0.9	**1.3**
74	1996	0.6	0.8	0.4	0.6	0.9	**1.3**
112	1996	0.6	**1.0**	0.8	**1.7**	**1.6**	**3.5**
152	1996	**1.6**	**3.6**	**2.0**	**4.8**	**4.1**	**10**

Bold values indicate exceedances

TABLE 5-37: RATIO OF MODELED DIETARY DOSE AND EGG CONCENTRATIONS BASED ON 1993 DATA FOR FEMALE TREE SWALLOW FOR TRI+ CONGENERS (UNCHANGED)

	<<<< ----- Dietary Dose ---- >>>>				<<<< ---- Egg Concentration ---- >>>>			
	LOAEL	LOAEL	NOAEL	NOAEL	LOAEL	LOAEL	NOAEL	NOAEL
Location	vs. Average ADD Hazard Quotient	vs. 95% UCL ADD Hazard Quotient	vs. Average ADD Hazard Quotient	vs. 95% UCL ADD Hazard Quotient	vs. Average Conc. Hazard Quotient	vs. 95% UCL Conc Hazard Quotient	vs. Average Conc. Hazard Quotient	vs. 95% UCL Conc. Hazard Quotient
Upper River								
Thompson Island Pool (189)	NA	NA	0.7	**1.2**	NA	NA	**1.0**	**1.7**
Stillwater (168)	NA	NA	**1.2**	**5.5**	NA	NA	**1.7**	**7.8**
Federal Dam (154)	NA	NA	0.3	0.5	NA	NA	0.4	0.7
Lower River								
143.5	NA	NA	0.04	0.1	NA	NA	0.1	0.1
137.2	NA	NA	0.08	0.3	NA	NA	0.1	0.4
122.4	NA	NA	0.04	0.1	NA	NA	0.1	0.2
113.8	NA	NA	0.04	0.2	NA	NA	0.1	0.2
100	NA	NA	0.02	0.1	NA	NA	0.03	0.2
88.9	NA	NA	0.01	0.02	NA	NA	0.01	0.03
58.7	NA	NA	0.03	0.3	NA	NA	0.04	0.4
47.3	NA	NA	0.04	0.3	NA	NA	0.05	0.4
25.8	NA	NA	0.01	0.0	NA	NA	0.01	0.03

Bold value indicates exceedances

TABLE 5-38: RATIO OF MODELED DIETARY DOSE BASED ON FISHRAND FOR FEMALE TREE SWALLOWS BASED ON THE SUM OF TRI+ CONGENERS FOR THE PERIOD 1993 - 2018
REVISED

Year	LOAEL 189 Average	LOAEL 189 95% UCL	NOAEL 189 Average	NOAEL 189 95% UCL	LOAEL 168 Average	LOAEL 168 95% UCL	NOAEL 168 Average	NOAEL 168 95% UCL	LOAEL 154 Average	LOAEL 154 95% UCL	NOAEL 154 Average	NOAEL 154 95% UCL
1993	NA	NA	0.7	0.8	NA	NA	0.6	0.6	NA	NA	0.2	0.3
1994	NA	NA	0.7	0.7	NA	NA	0.6	0.6	NA	NA	0.2	0.2
1995	NA	NA	0.6	0.6	NA	NA	0.5	0.5	NA	NA	0.2	0.2
1996	NA	NA	0.5	0.6	NA	NA	0.4	0.5	NA	NA	0.2	0.2
1997	NA	NA	0.5	0.5	NA	NA	0.4	0.4	NA	NA	0.2	0.2
1998	NA	NA	0.3	0.3	NA	NA	0.4	0.4	NA	NA	0.15	0.2
1999	NA	NA	0.3	0.3	NA	NA	0.4	0.4	NA	NA	0.14	0.15
2000	NA	NA	0.3	0.3	NA	NA	0.3	0.3	NA	NA	0.12	0.13
2001	NA	NA	0.2	0.2	NA	NA	0.3	0.3	NA	NA	0.11	0.12
2002	NA	NA	0.2	0.2	NA	NA	0.2	0.2	NA	NA	0.10	0.11
2003	NA	NA	0.2	0.2	NA	NA	0.2	0.2	NA	NA	0.09	0.10
2004	NA	NA	0.2	0.2	NA	NA	0.2	0.2	NA	NA	0.08	0.09
2005	NA	NA	0.2	0.2	NA	NA	0.2	0.2	NA	NA	0.07	0.08
2006	NA	NA	0.2	0.2	NA	NA	0.2	0.2	NA	NA	0.07	0.07
2007	NA	NA	0.14	0.15	NA	NA	0.2	0.2	NA	NA	0.06	0.06
2008	NA	NA	0.13	0.14	NA	NA	0.14	0.2	NA	NA	0.06	0.06
2009	NA	NA	0.12	0.13	NA	NA	0.14	0.14	NA	NA	0.05	0.05
2010	NA	NA	0.11	0.12	NA	NA	0.12	0.13	NA	NA	0.05	0.05
2011	NA	NA	0.10	0.10	NA	NA	0.11	0.12	NA	NA	0.04	0.05
2012	NA	NA	0.09	0.09	NA	NA	0.10	0.11	NA	NA	0.04	0.04
2013	NA	NA	0.08	0.08	NA	NA	0.10	0.11	NA	NA	0.04	0.04
2014	NA	NA	0.07	0.08	NA	NA	0.11	0.12	NA	NA	0.03	0.04
2015	NA	NA	0.07	0.07	NA	NA	0.11	0.11	NA	NA	0.03	0.03
2016	NA	NA	0.06	0.07	NA	NA	0.10	0.11	NA	NA	0.03	0.03
2017	NA	NA	0.06	0.06	NA	NA	0.10	0.11	NA	NA	0.03	0.03
2018	NA	NA	0.06	0.06	NA	NA	0.10	0.10	NA	NA	0.03	0.03

Bold value indicates exceedances

304319

TABLE 5-39: RATIO OF MODELED EGG CONCENTRATIONS TO BENCHMARKS FOR FEMALE TREE SWALLOWS BASED ON THE SUM OF TRI+ CONGENERS FOR THE PERIOD 1993 - 2018
REVISED

Year	LOAEL 189 Average	LOAEL 189 95% UCL	NOAEL 189 Average	NOAEL 189 95% UCL	LOAEL 168 Average	LOAEL 168 95% UCL	NOAEL 168 Average	NOAEL 168 95% UCL	LOAEL 154 Average	LOAEL 154 95% UCL	NOAEL 154 Average	NOAEL 154 95% UCL
1993	NA	NA	**1.0**	**1.1**	NA	NA	0.8	0.9	NA	NA	0.3	0.4
1994	NA	NA	0.9	**1.0**	NA	NA	0.8	0.8	NA	NA	0.3	0.3
1995	NA	NA	0.9	0.9	NA	NA	0.7	0.7	NA	NA	0.3	0.3
1996	NA	NA	0.8	0.8	NA	NA	0.6	0.6	NA	NA	0.3	0.3
1997	NA	NA	0.7	0.7	NA	NA	0.5	0.5	NA	NA	0.2	0.3
1998	NA	NA	0.4	0.4	NA	NA	0.6	0.6	NA	NA	0.2	0.2
1999	NA	NA	0.4	0.4	NA	NA	0.5	0.5	NA	NA	0.2	0.2
2000	NA	NA	0.4	0.4	NA	NA	0.4	0.5	NA	NA	0.2	0.2
2001	NA	NA	0.3	0.3	NA	NA	0.4	0.4	NA	NA	0.2	0.2
2002	NA	NA	0.3	0.3	NA	NA	0.3	0.3	NA	NA	0.14	0.15
2003	NA	NA	0.3	0.3	NA	NA	0.3	0.3	NA	NA	0.13	0.14
2004	NA	NA	0.3	0.3	NA	NA	0.3	0.3	NA	NA	0.12	0.12
2005	NA	NA	0.2	0.2	NA	NA	0.3	0.3	NA	NA	0.10	0.11
2006	NA	NA	0.2	0.2	NA	NA	0.2	0.2	NA	NA	0.09	0.10
2007	NA	NA	0.2	0.2	NA	NA	0.2	0.2	NA	NA	0.08	0.09
2008	NA	NA	0.2	0.2	NA	NA	0.2	0.2	NA	NA	0.08	0.08
2009	NA	NA	0.2	0.2	NA	NA	0.2	0.2	NA	NA	0.07	0.08
2010	NA	NA	0.2	0.2	NA	NA	0.2	0.2	NA	NA	0.07	0.07
2011	NA	NA	0.14	0.15	NA	NA	0.2	0.2	NA	NA	0.06	0.06
2012	NA	NA	0.12	0.13	NA	NA	0.14	0.2	NA	NA	0.05	0.06
2013	NA	NA	0.11	0.12	NA	NA	0.14	0.2	NA	NA	0.05	0.05
2014	NA	NA	0.10	0.11	NA	NA	0.2	0.2	NA	NA	0.05	0.05
2015	NA	NA	0.10	0.10	NA	NA	0.15	0.2	NA	NA	0.04	0.05
2016	NA	NA	0.09	0.10	NA	NA	0.14	0.2	NA	NA	0.04	0.05
2017	NA	NA	0.09	0.09	NA	NA	0.14	0.2	NA	NA	0.04	0.05
2018	NA	NA	0.08	0.09	NA	NA	0.14	0.14	NA	NA	0.04	0.04

Bold value indicates exceedances

TABLE 5-40: RATIO OF MODELED DIETARY DOSE AND EGG CONCENTRATIONS TO BENCHMARKS BASED ON 1993 DATA FOR FEMALE TREE SWALLOW ON TEQ BASIS (UNCHANGED)

	<<<< ----- Dietary Dose ---- >>>>				<<<< ---- Egg Concentration ---- >>>>			
	LOAEL	LOAEL	NOAEL	NOAEL	LOAEL	LOAEL	NOAEL	NOAEL
Location	vs. Average ADD Hazard Quotient	vs. 95% UCL ADD Hazard Quotient	vs. Average ADD Hazard Quotient	vs. 95% UCL ADD Hazard Quotient	vs. Average Conc. Hazard Quotient	vs. 95% UCL Conc Hazard Quotient	vs. Average Conc. Hazard Quotient	vs. 95% UCL Conc. Hazard Quotient
Upper River								
Thompson Island Pool (189)	NA	NA	0.12	0.21	NA	NA	0.36	0.64
Stillwater (168)	NA	NA	0.21	0.97	NA	NA	0.64	**2.99**
Federal Dam (154)	NA	NA	0.05	0.08	NA	NA	0.15	0.26
Lower River								
143.5	NA	NA	0.01	0.02	NA	NA	0.02	0.05
137.2	NA	NA	0.01	0.06	NA	NA	0.04	0.17
122.4	NA	NA	0.01	0.02	NA	NA	0.02	0.06
113.8	NA	NA	0.01	0.03	NA	NA	0.02	0.09
100	NA	NA	0.00	0.02	NA	NA	0.01	0.07
88.9	NA	NA	0.00	0.00	NA	NA	0.01	0.01
58.7	NA	NA	0.01	0.05	NA	NA	0.02	0.15
47.3	NA	NA	0.01	0.05	NA	NA	0.02	0.14
25.8	NA	NA	0.00	0.00	NA	NA	0.01	0.01

304321

304322

TABLE 5-41: RATIO OF MODELED DIETARY DOSE BASED ON FISHRAND FOR FEMALE TREE SWALLOW USING TEQ FOR THE PERIOD 1993 - 2018
REVISED

Year	LOAEL 189 Average	LOAEL 189 95% UCL	NOAEL 189 Average	NOAEL 189 95% UCL	LOAEL 168 Average	LOAEL 168 95% UCL	NOAEL 168 Average	NOAEL 168 95% UCL	LOAEL 154 Average	LOAEL 154 95% UCL	NOAEL 154 Average	NOAEL 154 95% UCL
1993	NA	NA	0.13	0.13	NA	NA	0.10	0.11	NA	NA	0.04	0.04
1994	NA	NA	0.12	0.12	NA	NA	0.10	0.10	NA	NA	0.04	0.04
1995	NA	NA	0.11	0.11	NA	NA	0.09	0.09	NA	NA	0.04	0.04
1996	NA	NA	0.10	0.10	NA	NA	0.07	0.08	NA	NA	0.03	0.03
1997	NA	NA	0.09	0.09	NA	NA	0.06	0.07	NA	NA	0.03	0.03
1998	NA	NA	0.05	0.06	NA	NA	0.07	0.07	NA	NA	0.03	0.03
1999	NA	NA	0.05	0.05	NA	NA	0.06	0.07	NA	NA	0.02	0.03
2000	NA	NA	0.04	0.05	NA	NA	0.05	0.06	NA	NA	0.02	0.02
2001	NA	NA	0.04	0.04	NA	NA	0.05	0.05	NA	NA	0.02	0.02
2002	NA	NA	0.04	0.04	NA	NA	0.04	0.04	NA	NA	0.02	0.02
2003	NA	NA	0.03	0.04	NA	NA	0.04	0.04	NA	NA	0.02	0.02
2004	NA	NA	0.03	0.03	NA	NA	0.03	0.04	NA	NA	0.01	0.02
2005	NA	NA	0.03	0.03	NA	NA	0.03	0.03	NA	NA	0.01	0.01
2006	NA	NA	0.03	0.03	NA	NA	0.03	0.03	NA	NA	0.01	0.01
2007	NA	NA	0.02	0.03	NA	NA	0.03	0.03	NA	NA	0.01	0.01
2008	NA	NA	0.02	0.02	NA	NA	0.02	0.03	NA	NA	0.01	0.01
2009	NA	NA	0.02	0.02	NA	NA	0.02	0.03	NA	NA	0.01	0.01
2010	NA	NA	0.02	0.02	NA	NA	0.02	0.02	NA	NA	0.01	0.01
2011	NA	NA	0.02	0.02	NA	NA	0.02	0.02	NA	NA	0.01	0.01
2012	NA	NA	0.02	0.02	NA	NA	0.02	0.02	NA	NA	0.01	0.01
2013	NA	NA	0.01	0.01	NA	NA	0.02	0.02	NA	NA	0.01	0.01
2014	NA	NA	0.01	0.01	NA	NA	0.02	0.02	NA	NA	0.01	0.01
2015	NA	NA	0.01	0.01	NA	NA	0.02	0.02	NA	NA	0.01	0.01
2016	NA	NA	0.01	0.01	NA	NA	0.02	0.02	NA	NA	0.01	0.01
2017	NA	NA	0.01	0.01	NA	NA	0.02	0.02	NA	NA	0.01	0.01
2018	NA	NA	0.01	0.01	NA	NA	0.02	0.02	NA	NA	0.01	0.01

TABLE 5-42: RATIO OF MODELED EGG CONCENTRATIONS BASED ON FISHRAND FOR FEMALE TREE SWALLOW USING TEQ FOR THE PERIOD 1993 - 2018

REVISED

Year	LOAEL 189 Average	LOAEL 189 95% UCL	NOAEL 189 Average	NOAEL 189 95% UCL	LOAEL 168 Average	LOAEL 168 95% UCL	NOAEL 168 Average	NOAEL 168 95% UCL	LOAEL 154 Average	LOAEL 154 95% UCL	NOAEL 154 Average	NOAEL 154 95% UCL
1993	NA	NA	0.39	0.41	NA	NA	0.32	0.34	NA	NA	0.13	0.14
1994	NA	NA	0.36	0.38	NA	NA	0.30	0.32	NA	NA	0.12	0.13
1995	NA	NA	0.33	0.35	NA	NA	0.27	0.28	NA	NA	0.11	0.12
1996	NA	NA	0.30	0.31	NA	NA	0.23	0.24	NA	NA	0.10	0.11
1997	NA	NA	0.26	0.28	NA	NA	0.20	0.21	NA	NA	0.09	0.10
1998	NA	NA	0.16	0.17	NA	NA	0.22	0.23	NA	NA	0.08	0.09
1999	NA	NA	0.15	0.16	NA	NA	0.19	0.21	NA	NA	0.07	0.08
2000	NA	NA	0.14	0.14	NA	NA	0.17	0.18	NA	NA	0.07	0.07
2001	NA	NA	0.12	0.13	NA	NA	0.14	0.15	NA	NA	0.06	0.06
2002	NA	NA	0.11	0.12	NA	NA	0.12	0.13	NA	NA	0.05	0.06
2003	NA	NA	0.10	0.11	NA	NA	0.11	0.12	NA	NA	0.05	0.05
2004	NA	NA	0.10	0.10	NA	NA	0.11	0.11	NA	NA	0.04	0.05
2005	NA	NA	0.09	0.09	NA	NA	0.10	0.10	NA	NA	0.04	0.04
2006	NA	NA	0.08	0.09	NA	NA	0.09	0.09	NA	NA	0.04	0.04
2007	NA	NA	0.07	0.08	NA	NA	0.08	0.09	NA	NA	0.03	0.03
2008	NA	NA	0.07	0.07	NA	NA	0.08	0.08	NA	NA	0.03	0.03
2009	NA	NA	0.06	0.07	NA	NA	0.07	0.08	NA	NA	0.03	0.03
2010	NA	NA	0.06	0.06	NA	NA	0.07	0.07	NA	NA	0.03	0.03
2011	NA	NA	0.05	0.06	NA	NA	0.06	0.06	NA	NA	0.02	0.02
2012	NA	NA	0.05	0.05	NA	NA	0.05	0.06	NA	NA	0.02	0.02
2013	NA	NA	0.04	0.05	NA	NA	0.06	0.06	NA	NA	0.02	0.02
2014	NA	NA	0.04	0.04	NA	NA	0.06	0.06	NA	NA	0.02	0.02
2015	NA	NA	0.04	0.04	NA	NA	0.06	0.06	NA	NA	0.02	0.02
2016	NA	NA	0.03	0.04	NA	NA	0.06	0.06	NA	NA	0.02	0.02
2017	NA	NA	0.03	0.03	NA	NA	0.05	0.06	NA	NA	0.02	0.02
2018	NA	NA	0.03	0.03	NA	NA	0.05	0.06	NA	NA	0.02	0.02

304324

TABLE 5-43: RATIO OF MODELED DIETARY DOSE AND EGG CONCENTRATIONS BASED ON 1993 DATA FOR FEMALE MALLARD FOR TRI+ CONGENERS
REVISED

Location	LOAEL vs. Average ADD Hazard Quotient	LOAEL vs. 95% UCL ADD Hazard Quotient	NOAEL vs. Average ADD Hazard Quotient	NOAEL vs. 95% UCL ADD Hazard Quotient	LOAEL vs. Average Conc. Hazard Quotient	LOAEL vs. 95% UCL Conc Hazard Quotient	NOAEL vs. Average Conc. Hazard Quotient	NOAEL vs. 95% UCL Conc. Hazard Quotient
	<<<< ----- Dietary Dose ---- >>>>				<<<< ---- Egg Concentration ---- >>>>			
Upper River								
Thompson Island Pool (189)	0.13	0.2	**1.3**	**2.3**	**19**	**30**	**129**	**202**
Stillwater (168)	0.2	0.4	**2.4**	**4.4**	**36**	**62**	**240**	**417**
Federal Dam (154)	0.07	0.1	0.7	**1.2**	**8.5**	**15**	**57**	**99**
Lower River								
143.5	0.02	0.04	0.2	0.4	**1.2**	**2.1**	**8**	**14**
137.2	0.02	0.05	0.2	0.5	**2.3**	**4.1**	**16**	**27**
122.4	0.01	0.09	0.1	0.9	**1.1**	**2.7**	**7.3**	**18**
113.8	0.01	0.09	0.1	0.9	0.9	**1.6**	**6.3**	**11**
100	0.01	0.10	0.1	**1.0**	0.5	**3.5**	**3.5**	**24**
88.9	0.01	0.07	0.1	0.7	0.3	0.5	**1.7**	**3.1**
58.7	0.01	0.08	0.1	0.8	0.7	**1.2**	**4.5**	**7.8**
47.3	0.01	0.11	0.1	**1.1**	0.9	**6.6**	**6.1**	**44**
25.8	0.01	0.07	0.1	0.7	0.3	0.5	**1.8**	**3.0**

Bold values indicate exceedances

TABLE 5-44: RATIO OF MODELED DIETARY DOSE FOR FEMALE MALLARD BASED ON FISHRAND RESULTS FOR THE TRI+ CONGENERS
REVISED

Year	LOAEL 189 Average	LOAEL 189 95% UCL	NOAEL 189 Average	NOAEL 189 95% UCL	LOAEL 168 Average	LOAEL 168 95% UCL	NOAEL 168 Average	NOAEL 168 95% UCL	LOAEL 154 Average	LOAEL 154 95% UCL	NOAEL 154 Average	NOAEL 154 95% UCL
1993	0.7	0.8	**7.0**	**7.5**	0.12	0.13	**1.2**	**1.3**	0.05	0.05	0.5	0.5
1994	0.6	0.7	**6.1**	**6.6**	0.11	0.12	**1.1**	**1.2**	0.05	0.05	0.5	0.5
1995	0.5	0.5	**4.8**	**5.2**	0.10	0.10	**1.0**	**1.0**	0.04	0.04	0.4	0.4
1996	0.3	0.3	**3.1**	**3.4**	0.08	0.09	0.8	0.9	0.04	0.04	0.4	0.4
1997	0.3	0.3	**3.0**	**3.2**	0.07	0.08	0.7	0.8	0.03	0.04	0.3	0.4
1998	0.4	0.4	**3.6**	**3.9**	0.08	0.08	0.8	0.8	0.03	0.03	0.3	0.3
1999	0.3	0.3	**3.2**	**3.5**	0.07	0.07	0.7	0.7	0.03	0.03	0.3	0.3
2000	0.2	0.2	**2.0**	**2.2**	0.06	0.06	0.6	0.6	0.03	0.03	0.3	0.3
2001	0.2	0.2	**2.0**	**2.1**	0.05	0.05	0.5	0.5	0.02	0.02	0.2	0.2
2002	0.2	0.2	**2.0**	**2.1**	0.04	0.05	0.4	0.5	0.02	0.02	0.2	0.2
2003	0.2	0.3	**2.4**	**2.6**	0.04	0.04	0.4	0.4	0.02	0.02	0.2	0.2
2004	0.2	0.3	**2.5**	**2.7**	0.04	0.04	0.4	0.4	0.02	0.02	0.2	0.2
2005	0.2	0.2	**2.0**	**2.1**	0.04	0.04	0.4	0.4	0.02	0.02	0.2	0.2
2006	0.2	0.2	**1.9**	**2.1**	0.03	0.03	0.3	0.3	0.01	0.01	0.13	0.14
2007	0.2	0.2	**1.6**	**1.8**	0.03	0.03	0.3	0.3	0.01	0.01	0.13	0.13
2008	0.2	0.2	**2.2**	**2.4**	0.03	0.03	0.3	0.3	0.01	0.01	0.12	0.13
2009	0.2	0.2	**2.0**	**2.1**	0.03	0.03	0.3	0.3	0.01	0.01	0.11	0.12
2010	0.1	0.1	**1.4**	**1.5**	0.02	0.03	0.2	0.3	0.01	0.01	0.10	0.11
2011	0.1	0.2	**1.4**	**1.5**	0.02	0.02	0.2	0.2	0.01	0.01	0.09	0.10
2012	0.1	0.1	**1.3**	**1.4**	0.02	0.02	0.2	0.2	0.01	0.01	0.08	0.09
2013	0.1	0.1	**1.2**	**1.3**	0.02	0.02	0.2	0.2	0.01	0.01	0.08	0.08
2014	0.1	0.1	**1.2**	**1.3**	0.02	0.02	0.2	0.2	0.01	0.01	0.08	0.08
2015	0.1	0.1	**1.1**	**1.2**	0.02	0.02	0.2	0.2	0.01	0.01	0.07	0.08
2016	0.2	0.2	**1.6**	**1.8**	0.02	0.02	0.2	0.2	0.01	0.01	0.07	0.08
2017	0.2	0.2	**1.6**	**1.7**	0.02	0.02	0.2	0.2	0.01	0.01	0.07	0.07
2018	0.2	0.2	**1.7**	**1.8**	0.02	0.02	0.2	0.2	0.01	0.01	0.07	0.08

Bold values indicate exceedances

304325

304326

TABLE 5-45: RATIO OF EGG CONCENTRATIONS FOR FEMALE MALLARD BASED ON FISHRAND RESULTS FOR THE TRI+ CONGENERS
REVISED

Year	LOAEL 189 Average	LOAEL 189 95% UCL	NOAEL 189 Average	NOAEL 189 95% UCL	LOAEL 168 Average	LOAEL 168 95% UCL	NOAEL 168 Average	NOAEL 168 95% UCL	LOAEL 154 Average	LOAEL 154 95% UCL	NOAEL 154 Average	NOAEL 154 95% UCL
1993	**18**	**19**	**123**	**130**	**15**	**16**	**101**	**108**	**6.1**	**6.5**	**41**	**43**
1994	**17**	**18**	**113**	**120**	**14**	**15**	**94**	**100**	**5.6**	**6.0**	**38**	**40**
1995	**16**	**16**	**104**	**110**	**13**	**13**	**85**	**90**	**5.2**	**5.6**	**35**	**37**
1996	**14**	**15**	**93**	**99**	**11**	**12**	**73**	**77**	**4.7**	**5.0**	**31**	**33**
1997	**12**	**13**	**83**	**88**	**9.3**	**10**	**62**	**66**	**4.3**	**4.5**	**29**	**30**
1998	**7.7**	**8.1**	**52**	**54**	**10**	**11**	**68**	**73**	**3.8**	**4.0**	**25**	**27**
1999	**7.1**	**7.6**	**48**	**51**	**9.2**	**10**	**62**	**66**	**3.5**	**3.7**	**23**	**25**
2000	**6.5**	**6.8**	**43**	**46**	**7.9**	**8.5**	**53**	**57**	**3.1**	**3.3**	**21**	**22**
2001	**5.8**	**6.2**	**39**	**41**	**6.7**	**7.1**	**45**	**48**	**2.8**	**3.0**	**19**	**20**
2002	**5.4**	**5.7**	**36**	**38**	**5.8**	**6.1**	**39**	**41**	**2.5**	**2.7**	**17**	**18**
2003	**4.9**	**5.2**	**33**	**35**	**5.3**	**5.6**	**35**	**38**	**2.3**	**2.5**	**16**	**17**
2004	**4.6**	**4.8**	**31**	**32**	**5.0**	**5.3**	**34**	**36**	**2.1**	**2.2**	**14**	**15**
2005	**4.2**	**4.5**	**28**	**30**	**4.6**	**4.9**	**31**	**33**	**1.9**	**2.0**	**13**	**13**
2006	**3.9**	**4.1**	**26**	**28**	**4.2**	**4.5**	**28**	**30**	**1.7**	**1.8**	**11**	**12**
2007	**3.5**	**3.7**	**24**	**25**	**3.9**	**4.1**	**26**	**28**	**1.5**	**1.6**	**10**	**11**
2008	**3.3**	**3.5**	**22**	**23**	**3.6**	**3.9**	**24**	**26**	**1.4**	**1.5**	**9.5**	**10**
2009	**3.1**	**3.2**	**21**	**22**	**3.4**	**3.7**	**23**	**25**	**1.3**	**1.4**	**8.8**	**9.4**
2010	**2.8**	**2.9**	**19**	**20**	**3.1**	**3.3**	**21**	**22**	**1.2**	**1.3**	**8.0**	**8.6**
2011	**2.5**	**2.6**	**17**	**18**	**2.8**	**3.0**	**19**	**20**	**1.1**	**1.2**	**7.3**	**7.8**
2012	**2.3**	**2.4**	**15**	**16**	**2.6**	**2.7**	**17**	**18**	**1.0**	**1.1**	**6.6**	**7.1**
2013	**2.1**	**2.2**	**14**	**15**	**2.6**	**2.8**	**17**	**19**	0.9	**1.0**	**6.1**	**6.5**
2014	**1.9**	**2.0**	**13**	**13**	**2.8**	**2.9**	**18**	**20**	0.9	0.9	**5.8**	**6.1**
2015	**1.8**	**1.9**	**12**	**12**	**2.7**	**2.9**	**18**	**19**	0.8	0.9	**5.5**	**5.8**
2016	**1.7**	**1.7**	**11**	**12**	**2.6**	**2.8**	**18**	**19**	0.8	0.9	**5.4**	**5.7**
2017	**1.5**	**1.6**	**10**	**11**	**2.6**	**2.7**	**17**	**18**	0.8	0.8	**5.2**	**5.5**
2018	**1.5**	**1.5**	**10**	**10**	**2.5**	**2.6**	**16**	**18**	0.7	0.8	**5.0**	**5.3**

Bold values indicate exceedances

TABLE 5-46: RATIO OF MODELED DIETARY DOSE TO BENCHMARKS FOR FEMALE MALLARD BASED ON 1993 DATA ON A TEQ BASIS REVISED

Location	<<< ---- Dietary Dose ---->>>>>>> LOAEL vs. Average ADD Hazard Quotient	LOAEL vs. 95% UCL ADD Hazard Quotient	NOAEL vs. Average ADD Hazard Quotient	NOAEL vs. 95% UCL ADD Hazard Quotient	<<<< ---- Egg Concentration ---- >>>> LOAEL vs. Average Conc. Hazard Quotient	LOAEL vs. 95% UCL Conc Hazard Quotient	NOAEL vs. Average Conc. Hazard Quotient	NOAEL vs. 95% UCL Conc. Hazard Quotient
Upper River								
Thompson Island Pool (189)	**93**	**345**	**933**	**3450**	NA	NA	**4218**	**6627**
Stillwater (168)	**156**	**397**	**1556**	**3973**	NA	NA	**7870**	**13698**
Federal Dam (154)	**135**	**279**	**1353**	**2795**	NA	NA	**1875**	**3264**
Lower River								
143.5	**9.2**	**26**	**92**	**256**	NA	NA	**680**	**1183**
137.2	**10**	**28**	**105**	**281**	NA	NA	**1339**	**2330**
122.4	**5.9**	**74**	**59**	**741**	NA	NA	**624**	**1569**
113.8	**5.8**	**73**	**58**	**731**	NA	NA	**536**	**933**
100	**5.2**	**77**	**52**	**766**	NA	NA	**295**	**2016**
88.9	**3.8**	**67**	**38**	**666**	NA	NA	**148**	**263**
58.7	**4.1**	**67**	**41**	**674**	NA	NA	**381**	**663**
47.3	**4.6**	**74**	**46**	**737**	NA	NA	**517**	**3796**
25.8	**3.8**	**66**	**38**	**664**	NA	NA	**153**	**260**

Bold values indicate exceedances

304327

304328

TABLE 5-47: RATIO OF MODELED DIETARY DOSE TO BENCHMARKS FOR FEMALE MALLARD FOR PERIOD 1993 - 2018 ON A TEQ BASIS
REVISED

Year	LOAEL 189 Average	LOAEL 189 95% UCL	NOAEL 189 Average	NOAEL 189 95% UCL	LOAEL 168 Average	LOAEL 168 95% UCL	NOAEL 168 Average	NOAEL 168 95% UCL	LOAEL 154 Average	LOAEL 154 95% UCL	NOAEL 154 Average	NOAEL 154 95% UCL
1993	**71**	**76**	**710**	**759**	**9.5**	**10**	**95**	**101**	**3.9**	**4.2**	**39**	**42**
1994	**62**	**67**	**625**	**667**	**8.9**	**9.4**	**89**	**94**	**3.6**	**3.8**	**36**	**38**
1995	**50**	**54**	**503**	**536**	**7.7**	**8.2**	**77**	**82**	**3.2**	**3.4**	**32**	**34**
1996	**35**	**37**	**347**	**368**	**6.5**	**6.9**	**65**	**69**	**2.8**	**3.0**	**28**	**30**
1997	**33**	**35**	**326**	**345**	**5.7**	**6.1**	**57**	**61**	**2.6**	**2.8**	**26**	**28**
1998	**36**	**39**	**361**	**387**	**6.3**	**6.7**	**63**	**67**	**2.4**	**2.5**	**24**	**25**
1999	**32**	**34**	**321**	**344**	**5.5**	**5.8**	**55**	**58**	**2.2**	**2.3**	**22**	**23**
2000	**21**	**23**	**212**	**226**	**4.7**	**5.0**	**47**	**50**	**1.9**	**2.1**	**19**	**21**
2001	**20**	**22**	**204**	**217**	**3.9**	**4.2**	**39**	**42**	**1.7**	**1.8**	**17**	**18**
2002	**20**	**22**	**203**	**216**	**3.4**	**3.6**	**34**	**36**	**1.5**	**1.6**	**15**	**16**
2003	**24**	**26**	**239**	**255**	**3.3**	**3.5**	**33**	**35**	**1.5**	**1.6**	**15**	**16**
2004	**24**	**26**	**241**	**259**	**3.1**	**3.3**	**31**	**33**	**1.3**	**1.4**	**13**	**14**
2005	**20**	**21**	**196**	**210**	**2.9**	**3.1**	**29**	**31**	**1.2**	**1.3**	**12**	**13**
2006	**19**	**20**	**188**	**202**	**2.6**	**2.7**	**26**	**27**	**1.0**	**1.1**	**10**	**11**
2007	**16**	**17**	**163**	**175**	**2.4**	**2.5**	**24**	**25**	**1.0**	**1.0**	**9.7**	**10**
2008	**21**	**23**	**213**	**229**	**2.3**	**2.4**	**23**	**24**	0.9	**1.0**	**9.0**	**9.6**
2009	**19**	**20**	**191**	**205**	**2.2**	**2.4**	**22**	**24**	0.9	0.9	**8.6**	**9.2**
2010	**14**	**14**	**135**	**145**	**1.9**	**2.0**	**19**	**20**	0.8	0.8	**7.6**	**8.1**
2011	**14**	**15**	**139**	**149**	**1.8**	**1.9**	**18**	**19**	0.7	0.8	**7.2**	**7.7**
2012	**13**	**14**	**129**	**138**	**1.6**	**1.7**	**16**	**17**	0.6	0.7	**6.5**	**6.9**
2013	**12**	**12**	**116**	**124**	**1.6**	**1.7**	**16**	**17**	0.6	0.6	**5.9**	**6.3**
2014	**11**	**12**	**113**	**122**	**1.7**	**1.8**	**17**	**18**	0.6	0.6	**6.0**	**6.4**
2015	**11**	**12**	**110**	**118**	**1.7**	**1.8**	**17**	**18**	0.6	0.6	**5.7**	**6.1**
2016	**15**	**16**	**153**	**165**	**1.7**	**1.8**	**17**	**18**	0.5	0.6	**5.4**	**5.8**
2017	**15**	**16**	**147**	**158**	**1.6**	**1.7**	**16**	**17**	0.5	0.6	**5.4**	**5.8**
2018	**16**	**17**	**158**	**171**	**1.7**	**1.8**	**17**	**18**	0.6	0.6	**5.8**	**6.2**

Bold values indicate exceedances

TABLE 5-48: RATIO OF MODELED EGG CONCENTRATION TO BENCHMARKS FOR FEMALE MALLARD FOR PERIOD 1993 - 2018 ON A TEQ BASIS
REVISED

Year	LOAEL 189 Average	LOAEL 189 95% UCL	NOAEL 189 Average	NOAEL 189 95% UCL	LOAEL 168 Average	LOAEL 168 95% UCL	NOAEL 168 Average	NOAEL 168 95% UCL	LOAEL 154 Average	LOAEL 154 95% UCL	NOAEL 154 Average	NOAEL 154 95% UCL
1993	NA	NA	**4022**	**4252**	NA	NA	**3320**	**3533**	NA	NA	**1338**	**1424**
1994	NA	NA	**3718**	**3929**	NA	NA	**3091**	**3292**	NA	NA	**1240**	**1320**
1995	NA	NA	**3428**	**3624**	NA	NA	**2783**	**2961**	NA	NA	**1147**	**1222**
1996	NA	NA	**3068**	**3244**	NA	NA	**2384**	**2537**	NA	NA	**1032**	**1098**
1997	NA	NA	**2736**	**2888**	NA	NA	**2041**	**2172**	NA	NA	**937**	**997**
1998	NA	NA	**1691**	**1786**	NA	NA	**2243**	**2388**	NA	NA	**833**	**887**
1999	NA	NA	**1571**	**1660**	NA	NA	**2021**	**2151**	NA	NA	**769**	**818**
2000	NA	NA	**1420**	**1501**	NA	NA	**1746**	**1858**	NA	NA	**690**	**735**
2001	NA	NA	**1282**	**1355**	NA	NA	**1474**	**1568**	NA	NA	**615**	**655**
2002	NA	NA	**1181**	**1248**	NA	NA	**1270**	**1351**	NA	NA	**556**	**591**
2003	NA	NA	**1082**	**1144**	NA	NA	**1163**	**1238**	NA	NA	**510**	**543**
2004	NA	NA	**1002**	**1059**	NA	NA	**1103**	**1175**	NA	NA	**463**	**493**
2005	NA	NA	**932**	**985**	NA	NA	**1022**	**1087**	NA	NA	**413**	**440**
2006	NA	NA	**856**	**905**	NA	NA	**924**	**983**	NA	NA	**370**	**394**
2007	NA	NA	**773**	**817**	NA	NA	**849**	**904**	NA	NA	**334**	**356**
2008	NA	NA	**723**	**764**	NA	NA	**795**	**847**	NA	NA	**311**	**331**
2009	NA	NA	**674**	**712**	NA	NA	**758**	**807**	NA	NA	**289**	**308**
2010	NA	NA	**611**	**646**	NA	NA	**688**	**732**	NA	NA	**264**	**281**
2011	NA	NA	**549**	**581**	NA	NA	**620**	**659**	NA	NA	**241**	**257**
2012	NA	NA	**496**	**524**	NA	NA	**563**	**599**	NA	NA	**218**	**232**
2013	NA	NA	**451**	**476**	NA	NA	**573**	**610**	NA	NA	**201**	**214**
2014	NA	NA	**414**	**438**	NA	NA	**606**	**645**	NA	NA	**189**	**201**
2015	NA	NA	**387**	**409**	NA	NA	**593**	**631**	NA	NA	**179**	**191**
2016	NA	NA	**363**	**383**	NA	NA	**577**	**615**	NA	NA	**177**	**188**
2017	NA	NA	**340**	**359**	NA	NA	**563**	**599**	NA	NA	**169**	**180**
2018	NA	NA	**321**	**339**	NA	NA	**541**	**575**	NA	NA	**164**	**174**

Bold values indicate exceedances

304329

304330

TABLE 5-49: RATIO OF MODELED DIETARY DOSE AND EGG CONCENTRATIONS BASED ON 1993 DATA FOR FEMALE BELTED KINGFISHER FOR TRI+ CONGENERS

REVISED

Location	<<<< ----- Dietary Dose ---- >>>>				<<<< ---- Egg Concentration ---- >>>>			
	LOAEL vs. Average ADD Hazard Quotient	LOAEL vs. 95% UCL ADD Hazard Quotient	NOAEL vs. Average ADD Hazard Quotient	NOAEL vs. 95% UCL ADD Hazard Quotient	LOAEL vs. Average Conc. Hazard Quotient	LOAEL vs. 95% UCL Conc Hazard Quotient	NOAEL vs. Average Conc. Hazard Quotient	NOAEL vs. 95% UCL Conc. Hazard Quotient
Upper River								
Thompson Island Pool (189)	**108**	**215**	**753**	**1503**	**259**	**516**	**1732**	**3459**
Stillwater (168)	**60**	**179**	**418**	**1250**	**143**	**428**	**956**	**2865**
Federal Dam (154)	**14**	**22**	**98**	**155**	**34**	**53**	**226**	**355**
Lower River								
143.5	**9.3**	**12**	**65**	**87**	**22**	**30**	**151**	**200**
137.2	**19**	**44**	**132**	**311**	**45**	**107**	**304**	**716**
122.4	**7.5**	**13**	**53**	**92**	**18**	**31**	**121**	**211**
113.8	**7.9**	**11**	**55**	**78**	**19**	**27**	**127**	**180**
100	**3.4**	**8.6**	**24**	**60**	**8.3**	**20**	**55**	**136**
88.9	**6.1**	**8.5**	**43**	**60**	**15**	**20**	**98**	**137**
58.7	**7.1**	**14**	**50**	**99**	**17**	**34**	**115**	**228**
47.3	**6.6**	**14**	**46**	**97**	**16**	**33**	**105**	**223**
25.8	**4.5**	**5.6**	**32**	**39**	**11**	**13**	**73**	**90**

Bold values indicate exceedances

TABLE 5-50: RATIO OF MODELED DIETARY DOSE AND EGG CONCENTRATIONS BASED ON 1993 DATA FOR FEMALE GREAT BLUE HERON FOR TRI+ CONGENERS (UNCHANGED)

	<<<< ---- Dietary Dose ---- >>>>				<<<< ---- Egg Concentration ---- >>>>			
Location	LOAEL vs. Average ADD Hazard Quotient	LOAEL vs. 95% UCL ADD Hazard Quotient	NOAEL vs. Average ADD Hazard Quotient	NOAEL vs. 95% UCL ADD Hazard Quotient	LOAEL vs. Average Conc. Hazard Quotient	LOAEL vs. 95% UCL Conc Hazard Quotient	NOAEL vs. Average Conc. Hazard Quotient	NOAEL vs. 95% UCL Conc. Hazard Quotient
Upper River								
Thompson Island Pool (189)	**47**	**95**	**327**	**667**	**284**	**580**	**1902**	**3883**
Stillwater (168)	**17**	**25**	**116**	**178**	**96**	**137**	**642**	**918**
Federal Dam (154)	**3.8**	**5.6**	**27**	**39**	**22**	**33**	**151**	**219**
Lower River								
143.5	**4.3**	**5.2**	**30**	**36**	**26**	**31**	**175**	**210**
137.2	**8.7**	**19**	**61**	**132**	**53**	**115**	**354**	**768**
122.4	**3.3**	**5.4**	**23**	**38**	**20**	**33**	**135**	**219**
113.8	**3.5**	**3.7**	**24**	**26**	**21**	**22**	**142**	**147**
100	**1.5**	**2.8**	**11**	**19**	**9**	**16**	**61**	**106**
88.9	**3.0**	**4.1**	**21**	**29**	**18**	**25**	**122**	**168**
58.7	**3.3**	**3.8**	**23**	**27**	**20**	**23**	**133**	**151**
47.3	**2.9**	**4.0**	**20**	**28**	**18**	**23**	**119**	**157**
25.8	**2.2**	**2.6**	**15**	**18**	**13**	**16**	**89**	**107**

Bold values indicate exceedances

304331

TABLE 5-51: RATIO OF MODELED DIETARY DOSE AND EGG CONCENTRATIONS BASED ON 1993 DATA FOR FEMALE EAGLE FOR TRI+ CONGENERS

REVISED

	<<<< ----- Dietary Dose ---- >>>>				<<<< ---- Egg Concentration ---- >>>>			
Location	LOAEL vs. Average ADD Hazard Quotient	LOAEL vs. 95% UCL ADD Hazard Quotient	NOAEL vs. Average ADD Hazard Quotient	NOAEL vs. 95% UCL ADD Hazard Quotient	LOAEL vs. Average Conc. Hazard Quotient	LOAEL vs. 95% UCL Conc Hazard Quotient	NOAEL vs. Average Conc. Hazard Quotient	NOAEL vs. 95% UCL Conc. Hazard Quotient
Upper River								
Thompson Island Pool (189)	**172**	**333**	**1204**	**2331**	NA	NA	**481**	**931**
Stillwater (168)	**31**	**39**	**214**	**276**	NA	NA	**85**	**110**
Federal Dam (154)	**22**	**40**	**155**	**279**	NA	NA	**62**	**111**
Lower River								
143.5	**22**	**40**	**155**	**279**	NA	NA	**62**	**111**
137.2	**84**	**199**	**586**	**1394**	NA	NA	**234**	**557**
122.4	**19**	**27**	**136**	**188**	NA	NA	**55**	**75**
113.8	**18**	**25**	**123**	**173**	NA	NA	**49**	**69**
100	**20**	**63**	**142**	**438**	NA	NA	**57**	**175**
88.9	**13**	**25**	**91**	**174**	NA	NA	**36**	**69**
58.7	**15**	**22**	**106**	**157**	NA	NA	**42**	**63**
47.3	**17**	**47**	**121**	**327**	NA	NA	**48**	**131**
25.8	**12**	**24**	**86**	**170**	NA	NA	**34**	**68**

Bold values indicate exceedances

TABLE 5-52: RATIO OF MODELED DIETARY DOSE BASED ON FISHRAND FOR FEMALE KINGFISHER BASED ON THE SUM OF TRI+ CONGENERS FOR THE PERIOD 1993 - 2018
REVISED

Year	LOAEL 189 Average	LOAEL 189 95% UCL	NOAEL 189 Average	NOAEL 189 95% UCL	LOAEL 168 Average	LOAEL 168 95% UCL	NOAEL 168 Average	NOAEL 168 95% UCL	LOAEL 154 Average	LOAEL 154 95% UCL	NOAEL 154 Average	NOAEL 154 95% UCL
1993	87	89	608	623	25	27	176	186	11	11	74	78
1994	87	89	607	623	24	25	165	175	10	10	68	72
1995	79	81	553	566	21	22	146	155	8.8	9.4	62	66
1996	50	51	349	360	18	19	123	130	7.9	8.4	55	58
1997	35	36	242	249	13	14	90	96	6.1	6.5	42	45
1998	36	37	255	261	16	17	114	121	6.2	6.6	43	46
1999	34	35	237	243	15	16	103	109	5.8	6.2	41	43
2000	30	30	207	212	13	13	89	94	5.1	5.4	36	38
2001	27	27	187	192	11	11	75	80	4.6	4.9	32	34
2002	27	27	186	191	9.4	10	66	70	4.2	4.4	29	31
2003	26	26	180	185	8.6	9.1	60	64	3.8	4.1	27	28
2004	26	27	184	188	8.1	8.6	57	60	3.5	3.8	25	26
2005	24	24	165	168	7.6	8.1	53	56	3.1	3.3	22	23
2006	22	22	152	156	6.8	7.2	48	51	2.8	3.0	20	21
2007	21	22	147	151	6.2	6.6	44	46	2.5	2.7	18	19
2008	20	20	138	141	6.0	6.4	42	45	2.4	2.6	17	18
2009	20	21	143	146	5.7	6.0	40	42	2.2	2.4	16	16
2010	19	19	133	136	5.1	5.4	36	38	2.0	2.1	14	15
2011	16	16	113	115	4.6	4.9	32	34	1.8	2.0	13	14
2012	16	16	110	112	4.3	4.6	30	32	1.7	1.8	12	13
2013	14	14	97	99	4.2	4.5	29	31	1.5	1.6	11	11
2014	13	14	94	96	4.5	4.8	31	33	1.5	1.6	10	11
2015	12	13	87	89	4.4	4.7	31	33	1.4	1.5	10	10
2016	13	14	94	96	4.4	4.6	31	32	1.4	1.5	10	10
2017	14	14	95	97	4.2	4.4	29	31	1.4	1.4	9.5	10
2018	15	15	103	106	4.2	4.4	29	31	1.4	1.4	10	10

Bold values indicate exceedances

TABLE 5-53: RATIO OF MODELED DIETARY DOSE BASED ON FISHRAND FOR FEMALE BLUE HERON BASED ON THE SUM OF TRI+ CONGENERS FOR THE PERIOD 1993 - 2018
REVISED

Year	LOAEL 189 Average	LOAEL 189 95% UCL	NOAEL 189 Average	NOAEL 189 95% UCL	LOAEL 168 Average	LOAEL 168 95% UCL	NOAEL 168 Average	NOAEL 168 95% UCL	LOAEL 154 Average	LOAEL 154 95% UCL	NOAEL 154 Average	NOAEL 154 95% UCL
1993	**35**	**35**	**243**	**248**	**5.7**	**6.0**	**40**	**42**	**2.5**	**2.7**	**18**	**19**
1994	**35**	**36**	**248**	**253**	**5.4**	**5.7**	**38**	**40**	**2.3**	**2.4**	**16**	**17**
1995	**32**	**33**	**225**	**229**	**4.7**	**4.9**	**33**	**35**	**2.0**	**2.2**	**14**	**15**
1996	**18**	**19**	**128**	**130**	**3.8**	**4.0**	**27**	**28**	**1.8**	**1.9**	**13**	**13**
1997	**11**	**11**	**78**	**80**	**2.2**	**2.3**	**15**	**16**	**1.1**	**1.2**	**7.5**	**8.1**
1998	**15**	**15**	**102**	**104**	**3.5**	**3.7**	**24**	**26**	**1.4**	**1.4**	**10**	**10**
1999	**14**	**14**	**95**	**97**	**3.2**	**3.3**	**22**	**23**	**1.3**	**1.4**	**9.2**	**10**
2000	**12**	**12**	**82**	**84**	**2.7**	**2.9**	**19**	**20**	**1.1**	**1.2**	**8.0**	**8.4**
2001	**11**	**11**	**74**	**76**	**2.3**	**2.4**	**16**	**17**	**1.0**	**1.1**	**7.2**	**7.6**
2002	**11**	**11**	**76**	**77**	**2.0**	**2.2**	**14**	**15**	0.9	**1.0**	**6.6**	**6.9**
2003	**11**	**11**	**74**	**76**	**1.9**	**2.0**	**13**	**14**	0.9	0.9	**6.0**	**6.3**
2004	**11**	**11**	**77**	**78**	**1.8**	**1.9**	**12**	**13**	0.8	0.9	**5.7**	**6.0**
2005	**9.8**	**10**	**69**	**70**	**1.7**	**1.8**	**12**	**12**	0.7	0.8	**5.0**	**5.3**
2006	**9.1**	**9.2**	**63**	**65**	**1.5**	**1.6**	**10**	**11**	0.7	0.7	**4.6**	**4.8**
2007	**8.9**	**9.0**	**62**	**63**	**1.3**	**1.4**	**9.4**	**10**	0.6	0.6	**4.0**	**4.3**
2008	**8.3**	**8.5**	**58**	**60**	**1.3**	**1.4**	**9.4**	**10**	0.6	0.6	**3.9**	**4.1**
2009	**8.8**	**9.0**	**62**	**63**	**1.3**	**1.3**	**8.8**	**9.3**	0.5	0.5	**3.6**	**3.8**
2010	**8.3**	**8.4**	**58**	**59**	**1.1**	**1.2**	**7.8**	**8.3**	0.5	0.5	**3.2**	**3.4**
2011	**6.9**	**7.0**	**48**	**49**	**1.0**	**1.1**	**7.0**	**7.4**	0.4	0.4	**3.0**	**3.1**
2012	**6.8**	**6.9**	**48**	**48**	**1.0**	**1.0**	**6.9**	**7.3**	0.4	0.4	**2.8**	**3.0**
2013	**6.0**	**6.1**	**42**	**43**	0.9	**1.0**	**6.4**	**6.7**	0.4	0.4	**2.5**	**2.6**
2014	**5.9**	**5.9**	**41**	**42**	**1.0**	**1.0**	**6.9**	**7.3**	0.3	0.4	**2.4**	**2.5**
2015	**5.4**	**5.5**	**38**	**39**	**1.0**	**1.0**	**6.8**	**7.1**	0.3	0.3	**2.3**	**2.4**
2016	**6.0**	**6.1**	**42**	**42**	**1.0**	**1.0**	**6.9**	**7.2**	0.3	0.4	**2.4**	**2.5**
2017	**6.1**	**6.2**	**43**	**43**	0.9	**1.0**	**6.5**	**6.8**	0.3	0.3	**2.3**	**2.4**
2018	**6.7**	**6.9**	**47**	**48**	**1.0**	**1.0**	**6.9**	**7.2**	0.3	0.4	**2.4**	**2.5**

Bold values indicate exceedances

TABLE 5-54: RATIO OF MODELED DIETARY DOSE BASED ON FISHRAND FOR FEMALE BALD EAGLE BASED ON THE SUM OF TRI+ CONGENERS FOR THE PERIOD 1993 - 2018

REVISED

Year	LOAEL 189 Average	LOAEL 189 95% UCL	NOAEL 189 Average	NOAEL 189 95% UCL	LOAEL 168 Average	LOAEL 168 95% UCL	NOAEL 168 Average	NOAEL 168 95% UCL	LOAEL 154 Average	LOAEL 154 95% UCL	NOAEL 154 Average	NOAEL 154 95% UCL
1993	**24**	**25**	**168**	**175**	**6.9**	**7.0**	**49**	**49**	**3.4**	**3.5**	**24**	**24**
1994	**22**	**23**	**154**	**161**	**6.3**	**6.4**	**44**	**45**	**3.1**	**3.2**	**22**	**22**
1995	**25**	**26**	**176**	**184**	**6.3**	**6.3**	**44**	**44**	**3.0**	**3.0**	**21**	**21**
1996	**22**	**23**	**154**	**162**	**5.7**	**5.8**	**40**	**41**	**2.6**	**2.7**	**19**	**19**
1997	**18**	**19**	**126**	**132**	**5.1**	**5.2**	**36**	**36**	**2.4**	**2.4**	**17**	**17**
1998	**19**	**20**	**133**	**139**	**4.8**	**4.9**	**34**	**34**	**2.2**	**2.2**	**15**	**16**
1999	**19**	**20**	**136**	**143**	**4.5**	**4.6**	**32**	**32**	**2.0**	**2.0**	**14**	**14**
2000	**20**	**21**	**139**	**146**	**4.4**	**4.5**	**31**	**31**	**1.9**	**1.9**	**13**	**13**
2001	**17**	**18**	**117**	**123**	**4.0**	**4.0**	**28**	**28**	**1.7**	**1.8**	**12**	**12**
2002	**15**	**16**	**105**	**110**	**3.5**	**3.6**	**25**	**25**	**1.6**	**1.6**	**11**	**11**
2003	**16**	**16**	**111**	**115**	**3.3**	**3.3**	**23**	**23**	**1.4**	**1.5**	**10**	**10**
2004	**13**	**14**	**91**	**95**	**3.1**	**3.1**	**22**	**22**	**1.3**	**1.3**	**8.9**	**9.0**
2005	**12**	**12**	**82**	**86**	**3.2**	**3.2**	**22**	**23**	**1.2**	**1.2**	**8.6**	**8.7**
2006	**11**	**12**	**77**	**81**	**3.2**	**3.2**	**22**	**23**	**1.2**	**1.2**	**8.3**	**8.4**
2007	**12**	**13**	**87**	**91**	**3.2**	**3.2**	**22**	**22**	**1.2**	**1.2**	**8.1**	**8.2**
2008	**14**	**14**	**96**	**101**	**3.1**	**3.2**	**22**	**22**	**1.1**	**1.2**	**8.0**	**8.1**
2009	**13**	**14**	**91**	**95**	**3.0**	**3.0**	**21**	**21**	**1.1**	**1.1**	**7.8**	**7.9**
2010	**13**	**14**	**94**	**98**	**2.9**	**3.0**	**20**	**21**	**1.0**	**1.1**	**7.3**	**7.4**
2011	**12**	**12**	**82**	**86**	**2.8**	**2.8**	**20**	**20**	**1.0**	**1.0**	**7.1**	**7.2**
2012	**10**	**11**	**71**	**74**	**2.7**	**2.7**	**19**	**19**	**1.0**	**1.0**	**6.9**	**7.0**
2013	**10**	**10**	**69**	**72**	**2.5**	**2.5**	**17**	**18**	**1.0**	**1.0**	**6.7**	**6.8**
2014	**11**	**12**	**79**	**83**	**2.5**	**2.6**	**18**	**18**	**1.0**	**1.0**	**6.8**	**6.9**
2015	**11**	**11**	**75**	**79**	**2.4**	**2.5**	**17**	**17**	0.9	0.9	**6.5**	**6.6**
2016	**10**	**10**	**68**	**71**	**2.2**	**2.2**	**15**	**15**	0.8	0.8	**5.8**	**5.9**
2017	**8.6**	**9.0**	**60**	**63**	**2.1**	**2.1**	**14**	**15**	0.8	0.9	**5.9**	**6.0**
2018	**10**	**11**	**71**	**74**	**2.1**	**2.2**	**15**	**15**	0.8	0.9	**5.9**	**6.0**

Bold values indicate exceedances

304335

304336

TABLE 5-55: RATIO OF MODELED EGG CONCENTRATIONS TO BENCHMARKS FOR FEMALE KINGFISHER BASED ON THE SUM OF TRI+ CONGENERS FOR THE PERIOD 1993 - 2018
REVISED

Year	LOAEL 189 Average	LOAEL 189 95% UCL	NOAEL 189 Average	NOAEL 189 95% UCL	LOAEL 168 Average	LOAEL 168 95% UCL	NOAEL 168 Average	NOAEL 168 95% UCL	LOAEL 154 Average	LOAEL 154 95% UCL	NOAEL 154 Average	NOAEL 154 95% UCL
1993	202	207	1351	1386	60	63	399	423	25	27	168	179
1994	202	208	1355	1391	56	59	375	398	23	24	154	163
1995	184	189	1231	1263	50	53	332	352	21	22	141	149
1996	115	118	767	791	42	44	278	295	19	20	125	133
1997	78	81	524	540	31	33	204	218	14	15	96	103
1998	85	87	567	582	39	41	259	275	15	16	99	105
1999	79	81	528	541	35	37	234	248	14	15	93	98
2000	69	70	460	471	30	32	202	214	12	13	82	87
2001	62	64	416	427	25	27	170	180	11	12	73	78
2002	62	64	415	425	22	24	149	158	10	11	67	71
2003	60	62	402	414	20	22	137	145	9.1	10	61	65
2004	61	63	411	421	19	20	129	137	8.4	8.9	56	60
2005	55	56	368	377	18	19	121	128	7.5	7.9	50	53
2006	51	52	340	348	16	17	108	115	6.8	7.2	45	48
2007	49	50	330	337	15	16	99	105	6.1	6.4	41	43
2008	46	47	309	317	14	15	95	101	5.7	6.1	38	41
2009	48	49	322	329	13	14	90	96	5.3	5.6	35	38
2010	45	46	300	306	12	13	81	86	4.8	5.1	32	34
2011	38	39	253	259	11	12	73	77	4.4	4.7	29	31
2012	37	38	247	252	10	11	69	73	4.1	4.3	27	29
2013	33	33	218	224	10	11	67	71	3.7	3.9	25	26
2014	32	32	211	216	11	11	71	76	3.5	3.7	23	25
2015	29	30	196	200	10	11	70	74	3.3	3.5	22	24
2016	32	32	211	216	10	11	69	74	3.4	3.6	22	24
2017	32	33	214	219	10	11	67	71	3.2	3.4	22	23
2018	35	36	234	239	10	11	67	71	3.2	3.4	22	23

Bold values indicate exceedances

TABLE 5-56: RATIO OF MODELED EGG CONCENTRATIONS TO BENCHMARKS FOR FEMALE BLUE HERON BASED ON THE SUM OF TRI+ CONGENERS FOR THE PERIOD 1993 - 2018

REVISED

Year	LOAEL 189 Average	LOAEL 189 95% UCL	NOAEL 189 Average	NOAEL 189 95% UCL	LOAEL 168 Average	LOAEL 168 95% UCL	NOAEL 168 Average	NOAEL 168 95% UCL	LOAEL 154 Average	LOAEL 154 95% UCL	NOAEL 154 Average	NOAEL 154 95% UCL
1993	207	211	1388	1413	33	34	218	230	15	15	98	103
1994	212	216	1421	1448	31	33	208	220	13	14	88	93
1995	192	196	1286	1310	27	28	180	190	12	12	79	84
1996	107	109	718	733	22	23	146	155	10	11	70	74
1997	65	66	432	439	12	13	81	87	6.0	6.5	40	44
1998	87	89	582	593	20	21	134	142	7.9	8.4	53	56
1999	81	82	542	551	18	19	121	128	7.6	8.0	51	54
2000	70	71	468	475	16	16	104	110	6.6	6.9	44	46
2001	63	64	424	431	13	14	88	93	6.0	6.3	40	42
2002	64	66	431	439	12	12	78	83	5.4	5.7	36	38
2003	63	65	423	433	11	11	72	76	5.0	5.2	33	35
2004	66	67	442	450	10	11	68	72	4.7	5.0	31	33
2005	59	60	393	399	10	10	64	68	4.1	4.3	28	29
2006	54	55	363	370	8.6	9.0	57	61	3.8	4.0	25	27
2007	53	54	358	364	7.7	8.1	52	54	3.4	3.5	22	24
2008	50	51	336	342	7.7	8.2	52	55	3.2	3.4	22	23
2009	53	54	356	362	7.3	7.7	49	51	3.0	3.1	20	21
2010	50	51	333	338	6.4	6.8	43	46	2.7	2.8	18	19
2011	42	42	279	284	5.8	6.1	39	41	2.5	2.6	16	17
2012	41	42	275	279	5.7	6.0	38	40	2.4	2.5	16	17
2013	36	37	242	247	5.2	5.5	35	37	2.1	2.2	14	15
2014	35	36	236	240	5.7	6.0	38	40	2.0	2.1	13	14
2015	33	33	219	222	5.5	5.9	37	39	1.9	2.0	13	13
2016	36	37	241	245	5.6	6.0	38	40	2.0	2.1	13	14
2017	37	37	247	251	5.3	5.6	36	38	1.9	2.0	13	13
2018	41	42	274	279	5.7	6.0	38	40	2.0	2.1	13	14

Bold values indicate exceedances

304338

TABLE 5-57: RATIO OF MODELED EGG CONCENTRATIONS TO BENCHMARKS FOR FEMALE BALD EAGLES BASED ON THE SUM OF TRI+ CONGENERS FOR THE PERIOD 1993 - 2018

REVISED

Year	LOAEL 189 Average	LOAEL 189 95% UCL	NOAEL 189 Average	NOAEL 189 95% UCL	LOAEL 168 Average	LOAEL 168 95% UCL	NOAEL 168 Average	NOAEL 168 95% UCL	LOAEL 154 Average	LOAEL 154 95% UCL	NOAEL 154 Average	NOAEL 154 95% UCL
1993	NA	NA	**67**	**70**	NA	NA	**19**	**20**	NA	NA	**10**	**10**
1994	NA	NA	**62**	**64**	NA	NA	**18**	**18**	NA	NA	**8.7**	**8.9**
1995	NA	NA	**70**	**73**	NA	NA	**17**	**18**	NA	NA	**8.4**	**8.5**
1996	NA	NA	**62**	**65**	NA	NA	**16**	**16**	NA	NA	**7.4**	**7.5**
1997	NA	NA	**50**	**53**	NA	NA	**14**	**14**	NA	NA	**6.6**	**6.7**
1998	NA	NA	**53**	**55**	NA	NA	**13**	**14**	NA	NA	**6.1**	**6.2**
1999	NA	NA	**54**	**57**	NA	NA	**13**	**13**	NA	NA	**5.6**	**5.7**
2000	NA	NA	**56**	**58**	NA	NA	**12**	**13**	NA	NA	**5.3**	**5.4**
2001	NA	NA	**47**	**49**	NA	NA	**11**	**11**	NA	NA	**4.9**	**4.9**
2002	NA	NA	**42**	**44**	NA	NA	**10**	**10**	NA	NA	**4.5**	**4.5**
2003	NA	NA	**44**	**46**	NA	NA	**9.2**	**9.3**	NA	NA	**4.0**	**4.1**
2004	NA	NA	**36**	**38**	NA	NA	**8.6**	**8.8**	NA	NA	**3.5**	**3.6**
2005	NA	NA	**33**	**34**	NA	NA	**8.9**	**9.0**	NA	NA	**3.4**	**3.5**
2006	NA	NA	**31**	**32**	NA	NA	**8.9**	**9.0**	NA	NA	**3.3**	**3.4**
2007	NA	NA	**35**	**36**	NA	NA	**8.8**	**9.0**	NA	NA	**3.2**	**3.3**
2008	NA	NA	**38**	**40**	NA	NA	**8.7**	**8.8**	NA	NA	**3.2**	**3.2**
2009	NA	NA	**36**	**38**	NA	NA	**8.4**	**8.5**	NA	NA	**3.1**	**3.2**
2010	NA	NA	**37**	**39**	NA	NA	**8.1**	**8.3**	NA	NA	**2.9**	**3.0**
2011	NA	NA	**33**	**34**	NA	NA	**7.8**	**7.9**	NA	NA	**2.8**	**2.9**
2012	NA	NA	**28**	**30**	NA	NA	**7.4**	**7.5**	NA	NA	**2.8**	**2.8**
2013	NA	NA	**28**	**29**	NA	NA	**6.9**	**7.0**	NA	NA	**2.7**	**2.7**
2014	NA	NA	**32**	**33**	NA	NA	**7.1**	**7.2**	NA	NA	**2.7**	**2.8**
2015	NA	NA	**30**	**32**	NA	NA	**6.8**	**6.9**	NA	NA	**2.6**	**2.6**
2016	NA	NA	**27**	**28**	NA	NA	**6.0**	**6.1**	NA	NA	**2.3**	**2.4**
2017	NA	NA	**24**	**25**	NA	NA	**5.8**	**5.8**	NA	NA	**2.4**	**2.4**
2018	NA	NA	**28**	**30**	NA	NA	**6.0**	**6.1**	NA	NA	**2.4**	**2.4**

Bold values indicate exceedances

TABLE 5-58: RATIO OF MODELED DIETARY DOSE AND EGG CONCENTRATIONS TO BENCHMARKS BASED ON 1993 DATA FOR FEMALE BELTED KINGFISHER ON TEQ BASIS (UNCHANGED)

	<<<< ----- Dietary Dose ---- >>>>				<<<< ---- Egg Concentration ---- >>>>			
Location	LOAEL vs. Average ADD Hazard Quotient	LOAEL vs. 95% UCL ADD Hazard Quotient	NOAEL vs. Average ADD Hazard Quotient	NOAEL vs. 95% UCL ADD Hazard Quotient	LOAEL vs. Average Conc. Hazard Quotient	LOAEL vs. 95% UCL Conc Hazard Quotient	NOAEL vs. Average Conc. Hazard Quotient	NOAEL vs. 95% UCL Conc. Hazard Quotient
Upper River								
Thompson Island Pool (189)	**124**	**250**	**1237**	**2498**	**4078**	**8287**	**8157**	**16574**
Stillwater (168)	**54**	**103**	**537**	**1027**	**1575**	**3049**	**3149**	**6099**
Federal Dam (154)	**12**	**17**	**116**	**174**	**370**	**552**	**741**	**1104**
Lower River								
143.5	**11**	**14**	**112**	**138**	**371**	**456**	**742**	**911**
137.2	**23**	**50**	**226**	**499**	**750**	**1657**	**1500**	**3313**
122.4	**8.8**	**14**	**88**	**144**	**289**	**475**	**578**	**951**
113.8	**9.2**	**10**	**92**	**105**	**303**	**340**	**606**	**680**
100	**4.0**	**9.3**	**40**	**93**	**131**	**248**	**263**	**497**
88.9	**7.8**	**11**	**78**	**109**	**255**	**351**	**511**	**702**
58.7	**8.4**	**12**	**84**	**117**	**282**	**372**	**565**	**744**
47.3	**7.8**	**13**	**78**	**126**	**253**	**379**	**506**	**759**
25.8	**5.7**	**7.1**	**57**	**71**	**187**	**226**	**374**	**451**

Bold values indicate exceedances

TABLE 5-59: RATIO OF MODELED DIETARY DOSE AND EGG CONCENTRATIONS TO BENCHMARKS BASED ON 1993 DATA FOR FEMALE GREAT BLUE HERON ON TEQ BASIS (UNCHANGED)

	<<<< ---- Dietary Dose ---- >>>>				<<<< ---- Egg Concentration ---- >>>>			
Location	LOAEL vs. Average ADD Hazard Quotient	LOAEL vs. 95% UCL ADD Hazard Quotient	NOAEL vs. Average ADD Hazard Quotient	NOAEL vs. 95% UCL Conc. Hazard Quotient	LOAEL vs. Average Conc. Hazard Quotient	LOAEL vs. 95% UCL Conc Hazard Quotient	NOAEL vs. Average Conc. Hazard Quotient	NOAEL vs. 95% UCL Conc. Hazard Quotient
Upper River								
Thompson Island Pool (189)	**62**	**125**	**616**	**1245**	**204.0**	**416.6**	**340**	**694**
Stillwater (168)	**26**	**39**	**256**	**388**	**68.9**	**98.5**	**115**	**164**
Federal Dam (154)	**5.2**	**7.7**	**52**	**77**	**16.2**	**23.5**	**27**	**39**
Lower River								
143.5	**5.6**	**6.8**	**56**	**68**	**18.8**	**22.6**	**31**	**38**
137.2	**11**	**25**	**114**	**246**	**38.0**	**82.4**	**63**	**137**
122.4	**4.4**	**7.0**	**44**	**70**	**14.5**	**23.5**	**24**	**39**
113.8	**4.6**	**4.9**	**46**	**49**	**15.2**	**15.8**	**25**	**26**
100	**2.0**	**4.9**	**20**	**49**	**6.6**	**11.4**	**11**	**19**
88.9	**4.0**	**5.6**	**40**	**56**	**13.1**	**18.0**	**22**	**30**
58.7	**4**	**5.2**	**42**	**52**	**14.3**	**16.2**	**24**	**27**
47.3	**4**	**6.0**	**40**	**60**	**12.7**	**16.9**	**21**	**28**
25.8	**3**	**3.6**	**29**	**36**	**9.6**	**11.5**	**16**	**19**

Bold values indicate exceedances

TABLE 5-60: RATIO OF MODELED DIETARY DOSE AND EGG CONCENTRATIONS TO BENCHMARKS BASED ON 1993 DATA FOR FEMALE BALD EAGLE ON TEQ BASIS
REVISED

Location	<<<< ----- Dietary Dose ---- >>>>				<<<< ---- Egg Concentration ---- >>>>			
	LOAEL vs. Average ADD Hazard Quotient	LOAEL vs. 95% UCL ADD Hazard Quotient	NOAEL vs. Average ADD Hazard Quotient	NOAEL vs. 95% UCL ADD Hazard Quotient	LOAEL vs. Average Conc. Hazard Quotient	LOAEL vs. 95% UCL Conc Hazard Quotient	NOAEL vs. Average Conc. Hazard Quotient	NOAEL vs. 95% UCL Conc. Hazard Quotient
Upper River								
Thompson Island Pool (189)	**221**	**427**	**2208**	**4272**	NA	NA	**2153**	**4167**
Stillwater (168)	**39**	**51**	**392**	**505**	NA	NA	**383**	**493**
Federal Dam (154)	**28**	**51**	**285**	**511**	NA	NA	**278**	**499**
Lower River								
143.5	**28**	**51**	**285**	**512**	NA	NA	**278**	**499**
137.2	**107**	**256**	**1074**	**2555**	NA	NA	**1047**	**2492**
122.4	**25**	**34**	**250**	**344**	NA	NA	**244**	**335**
113.8	**23**	**32**	**226**	**317**	NA	NA	**221**	**309**
100	**26**	**80**	**260**	**803**	NA	NA	**253**	**783**
88.9	**17**	**32**	**167**	**318**	NA	NA	**163**	**310**
58.7	**19**	**29**	**194**	**288**	NA	NA	**190**	**281**
47.3	**22**	**60**	**223**	**599**	NA	NA	**217**	**585**
25.8	**16**	**31**	**157**	**312**	NA	NA	**153**	**305**

Bold values indicate exceedances

304341

304342

TABLE 5-61: RATIO OF MODELED DIETARY DOSE BASED ON FISHRAND FOR FEMALE BELTED KINGFISHER USING TEQ FOR THE PERIOD 1993 - 2018
REVISED

Year	LOAEL 189 Average	LOAEL 189 95% UCL	NOAEL 189 Average	NOAEL 189 95% UCL	LOAEL 168 Average	LOAEL 168 95% UCL	NOAEL 168 Average	NOAEL 168 95% UCL	LOAEL 154 Average	LOAEL 154 95% UCL	NOAEL 154 Average	NOAEL 154 95% UCL
1993	**132**	**134**	**1320**	**1338**	**23**	**24**	**226**	**235**	**9.2**	**10**	**92**	**96**
1994	**130**	**132**	**1296**	**1315**	**21**	**22**	**212**	**222**	**8.3**	**8.7**	**83**	**87**
1995	**119**	**121**	**1190**	**1206**	**19**	**20**	**189**	**197**	**7.6**	**7.9**	**76**	**79**
1996	**80**	**81**	**800**	**811**	**16**	**16**	**157**	**164**	**6.7**	**7.1**	**67**	**71**
1997	**59**	**59**	**585**	**591**	**11**	**11**	**107**	**112**	**4.7**	**5.0**	**47**	**50**
1998	**55**	**56**	**550**	**558**	**14**	**15**	**144**	**151**	**5.2**	**5.5**	**52**	**55**
1999	**52**	**52**	**516**	**522**	**13**	**14**	**131**	**136**	**5.0**	**5.2**	**50**	**52**
2000	**45**	**46**	**453**	**459**	**11**	**12**	**113**	**118**	**4.4**	**4.5**	**44**	**45**
2001	**41**	**41**	**408**	**413**	**10**	**10**	**95**	**99**	**3.9**	**4.1**	**39**	**41**
2002	**40**	**41**	**401**	**407**	**8.4**	**8.7**	**84**	**87**	**3.6**	**3.7**	**36**	**37**
2003	**38**	**39**	**385**	**392**	**7.6**	**7.9**	**76**	**79**	**3.3**	**3.4**	**33**	**34**
2004	**39**	**39**	**386**	**391**	**7.2**	**7.5**	**72**	**75**	**3.0**	**3.2**	**30**	**32**
2005	**35**	**35**	**348**	**353**	**6.8**	**7.1**	**68**	**71**	**2.7**	**2.8**	**27**	**28**
2006	**32**	**33**	**322**	**327**	**6.1**	**6.3**	**61**	**63**	**2.4**	**2.6**	**24**	**26**
2007	**31**	**31**	**310**	**314**	**5.5**	**5.7**	**55**	**57**	**2.2**	**2.3**	**22**	**23**
2008	**29**	**29**	**288**	**292**	**5.4**	**5.6**	**54**	**56**	**2.1**	**2.2**	**21**	**22**
2009	**30**	**30**	**296**	**301**	**5.1**	**5.3**	**51**	**53**	**1.9**	**2.0**	**19**	**20**
2010	**28**	**28**	**276**	**279**	**4.6**	**4.8**	**46**	**48**	**1.7**	**1.8**	**17**	**18**
2011	**23**	**24**	**235**	**238**	**4.1**	**4.3**	**41**	**43**	**1.6**	**1.7**	**16**	**17**
2012	**23**	**23**	**227**	**230**	**3.9**	**4.1**	**39**	**41**	**1.5**	**1.6**	**15**	**16**
2013	**20**	**20**	**201**	**204**	**3.6**	**3.8**	**36**	**38**	**1.3**	**1.4**	**13**	**14**
2014	**19**	**20**	**193**	**195**	**4.0**	**4.1**	**40**	**41**	**1.3**	**1.3**	**13**	**13**
2015	**18**	**18**	**179**	**181**	**3.9**	**4.1**	**39**	**41**	**1.2**	**1.3**	**12**	**13**
2016	**19**	**19**	**189**	**192**	**3.9**	**4.1**	**39**	**41**	**1.2**	**1.3**	**12**	**13**
2017	**19**	**19**	**191**	**193**	**3.7**	**3.9**	**37**	**39**	**1.2**	**1.2**	**12**	**12**
2018	**20**	**21**	**204**	**208**	**3.8**	**3.9**	**38**	**39**	**1.2**	**1.3**	**12**	**13**

Bold values indicate exceedances

TABLE 5-62: RATIO OF MODELED DIETARY DOSE BASED ON FISHRAND FOR FEMALE GREAT BLUE HERON USING TEQ FOR THE PERIOD 1993 - 2018 REVISED

Year	LOAEL 189 Average	LOAEL 189 95% UCL	NOAEL 189 Average	NOAEL 189 95% UCL	LOAEL 168 Average	LOAEL 168 95% UCL	NOAEL 168 Average	NOAEL 168 95% UCL	LOAEL 154 Average	LOAEL 154 95% UCL	NOAEL 154 Average	NOAEL 154 95% UCL
1993	**53**	**54**	**529**	**537**	**8.1**	**8.5**	**81**	**85**	**3.5**	**3.6**	**35**	**36**
1994	**53**	**54**	**530**	**539**	**7.7**	**8.1**	**77**	**81**	**3.1**	**3.3**	**31**	**33**
1995	**48**	**49**	**484**	**491**	**6.7**	**7.1**	**67**	**71**	**2.8**	**3.0**	**28**	**30**
1996	**30**	**30**	**298**	**302**	**5.5**	**5.8**	**55**	**58**	**2.5**	**2.6**	**25**	**26**
1997	**20**	**20**	**201**	**203**	**3.4**	**3.6**	**34**	**36**	**1.6**	**1.6**	**16**	**16.5**
1998	**22**	**22**	**221**	**225**	**5.1**	**5.3**	**51**	**53**	**1.9**	**2.0**	**19**	**20**
1999	**21**	**21**	**207**	**210**	**4.6**	**4.8**	**46**	**48**	**1.8**	**1.9**	**18**	**19**
2000	**18**	**18**	**180**	**182**	**4.0**	**4.1**	**40**	**41**	**1.6**	**1.7**	**16**	**16.6**
2001	**16**	**16**	**163**	**165**	**3.3**	**3.5**	**33**	**35**	**1.4**	**1.5**	**14**	**15.1**
2002	**16**	**16**	**163**	**165**	**3.0**	**3.1**	**30**	**31**	**1.3**	**1.4**	**13**	**13.7**
2003	**16**	**16**	**158**	**161**	**2.7**	**2.8**	**27**	**28**	**1.2**	**1.3**	**12**	**12.5**
2004	**16**	**16**	**162**	**164**	**2.6**	**2.7**	**26**	**27**	**1.1**	**1.2**	**11**	**11.8**
2005	**14**	**15**	**145**	**147**	**2.4**	**2.5**	**24**	**25**	**1.0**	**1.0**	**9.9**	**10.4**
2006	**13**	**14**	**134**	**136**	**2.2**	**2.3**	**22**	**23**	0.9	**1.0**	**9.1**	**9.5**
2007	**13**	**13**	**130**	**132**	**1.9**	**2.0**	**19**	**20**	0.8	0.8	**8.0**	**8.4**
2008	**12**	**12**	**122**	**124**	**1.9**	**2.0**	**19**	**20**	0.8	0.8	**7.7**	**8.1**
2009	**13**	**13**	**127**	**129**	**1.8**	**1.9**	**18**	**19**	0.7	0.7	**7.1**	**7.4**
2010	**12**	**12**	**119**	**120**	**1.6**	**1.7**	**16**	**17**	0.6	0.7	**6.4**	**6.7**
2011	**10**	**10**	**100**	**102**	**1.4**	**1.5**	**14**	**15**	0.6	0.6	**5.9**	**6.2**
2012	**9.8**	**10**	**98**	**99**	**1.4**	**1.5**	**14**	**15**	0.6	0.6	**5.6**	**5.9**
2013	**8.6**	**8.8**	**86**	**88**	**1.3**	**1.4**	**13**	**14**	0.5	0.5	**4.9**	**5.2**
2014	**8.4**	**8.5**	**84**	**85**	**1.4**	**1.5**	**14**	**15**	0.5	0.5	**4.8**	**5.0**
2015	**7.7**	**7.9**	**77**	**79**	**1.4**	**1.5**	**14**	**15**	0.5	0.5	**4.5**	**4.7**
2016	**8.4**	**8.5**	**84**	**85**	**1.4**	**1.5**	**14**	**15**	0.5	0.5	**4.7**	**4.9**
2017	**8.5**	**8.6**	**85**	**86**	**1.3**	**1.4**	**13**	**14**	0.4	0.5	**4.5**	**4.7**
2018	**9.3**	**9.5**	**93**	**95**	**1.4**	**1.5**	**14**	**15**	0.5	0.5	**4.7**	**4.9**

Bold values indicate exceedances

304343

304344

TABLE 5-63: RATIO OF MODELED DIETARY DOSE BASED ON FISHRAND FOR FEMALE BALD EAGLE USING TEQ FOR THE PERIOD 1993 - 2018
REVISED

Year	LOAEL 189 Average	LOAEL 189 95% UCL	NOAEL 189 Average	NOAEL 189 95% UCL	LOAEL 168 Average	LOAEL 168 95% UCL	NOAEL 168 Average	NOAEL 168 95% UCL	LOAEL 154 Average	LOAEL 154 95% UCL	NOAEL 154 Average	NOAEL 154 95% UCL
1993	**31**	**32**	**308**	**322**	**8.9**	**9.0**	**89**	**90**	**4.4**	**4.5**	**44**	**45**
1994	**28**	**30**	**283**	**296**	**8.1**	**8.2**	**81**	**82**	**4.0**	**4.1**	**40**	**41**
1995	**32**	**34**	**323**	**337**	**8.0**	**8.1**	**80**	**81**	**3.8**	**3.9**	**38**	**39**
1996	**28**	**30**	**283**	**297**	**7.3**	**7.5**	**73**	**75**	**3.4**	**3.4**	**34**	**34**
1997	**23**	**24**	**231**	**242**	**6.6**	**6.7**	**66**	**67**	**3.0**	**3.1**	**30**	**31**
1998	**24**	**25**	**243**	**254**	**6.2**	**6.2**	**62**	**62**	**2.8**	**2.8**	**28**	**28**
1999	**25**	**26**	**250**	**261**	**5.8**	**5.9**	**58**	**59**	**2.6**	**2.6**	**26**	**26**
2000	**26**	**27**	**256**	**267**	**5.7**	**5.8**	**57**	**58**	**2.4**	**2.5**	**24**	**25**
2001	**21**	**23**	**214**	**225**	**5.1**	**5.2**	**51**	**52**	**2.2**	**2.3**	**22**	**23**
2002	**19**	**20**	**192**	**201**	**4.5**	**4.6**	**45**	**46**	**2.0**	**2.1**	**20**	**21**
2003	**20**	**21**	**203**	**212**	**4.2**	**4.3**	**42**	**43**	**1.9**	**1.9**	**19**	**19**
2004	**17**	**17**	**167**	**175**	**4.0**	**4.0**	**40**	**40**	**1.6**	**1.7**	**16**	**17**
2005	**15**	**16**	**151**	**158**	**4.1**	**4.1**	**41**	**41**	**1.6**	**1.6**	**16**	**16**
2006	**14**	**15**	**142**	**148**	**4.1**	**4.1**	**41**	**41**	**1.5**	**1.5**	**15**	**15**
2007	**16**	**17**	**159**	**166**	**4.1**	**4.1**	**41**	**41**	**1.5**	**1.5**	**15**	**15**
2008	**18**	**18**	**176**	**184**	**4.0**	**4.1**	**40**	**41**	**1.5**	**1.5**	**15**	**15**
2009	**17**	**17**	**167**	**175**	**3.9**	**3.9**	**39**	**39**	**1.4**	**1.4**	**14**	**14**
2010	**17**	**18**	**172**	**179**	**3.7**	**3.8**	**37**	**38**	**1.3**	**1.4**	**13**	**14**
2011	**15**	**16**	**150**	**157**	**3.6**	**3.6**	**36**	**36**	**1.3**	**1.3**	**13**	**13**
2012	**13**	**14**	**130**	**136**	**3.4**	**3.5**	**34**	**35**	**1.3**	**1.3**	**13**	**13**
2013	**13**	**13**	**127**	**132**	**3.2**	**3.2**	**32**	**32**	**1.2**	**1.2**	**12**	**12**
2014	**15**	**15**	**145**	**152**	**3.2**	**3.3**	**32**	**33**	**1.3**	**1.3**	**13**	**13**
2015	**14**	**14**	**138**	**145**	**3.1**	**3.2**	**31**	**32**	**1.2**	**1.2**	**12**	**12**
2016	**12**	**13**	**124**	**130**	**2.8**	**2.8**	**28**	**28**	**1.1**	**1.1**	**11**	**11**
2017	**11**	**12**	**111**	**116**	**2.6**	**2.7**	**26**	**27**	**1.1**	**1.1**	**11**	**11**
2018	**13**	**14**	**131**	**136**	**2.8**	**2.8**	**28**	**28**	**1.1**	**1.1**	**11**	**11**

Bold values indicate exceedances

TABLE 5-64: RATIO OF MODELED EGG CONCENTRATIONS BASED ON FISHRAND FOR FEMALE BELTED KINGFISHER USING TEQ FOR THE PERIOD 1993 - 2018 REVISED

Year	LOAEL 189 Average	LOAEL 189 95% UCL	NOAEL 189 Average	NOAEL 189 95% UCL	LOAEL 168 Average	LOAEL 168 95% UCL	NOAEL 168 Average	NOAEL 168 95% UCL	LOAEL 154 Average	LOAEL 154 95% UCL	NOAEL 154 Average	NOAEL 154 95% UCL
1993	**3022**	**3083**	**6045**	**6165**	**574**	**607**	**1149**	**1214**	**252**	**266**	**504**	**533**
1994	**3080**	**3144**	**6160**	**6289**	**545**	**577**	**1090**	**1153**	**228**	**241**	**456**	**483**
1995	**2790**	**2846**	**5580**	**5693**	**475**	**504**	**949**	**1007**	**206**	**218**	**412**	**437**
1996	**1600**	**1637**	**3201**	**3275**	**391**	**414**	**782**	**828**	**182**	**193**	**365**	**387**
1997	**997**	**1016**	**1993**	**2032**	**244**	**261**	**487**	**523**	**118**	**127**	**237**	**254**
1998	**1268**	**1295**	**2537**	**2589**	**361**	**382**	**721**	**764**	**141**	**149**	**281**	**297**
1999	**1181**	**1203**	**2362**	**2407**	**325**	**344**	**650**	**688**	**134**	**142**	**268**	**284**
2000	**1021**	**1040**	**2041**	**2079**	**280**	**296**	**561**	**592**	**117**	**123**	**233**	**246**
2001	**925**	**943**	**1851**	**1886**	**236**	**250**	**472**	**499**	**105**	**111**	**211**	**223**
2002	**937**	**955**	**1874**	**1910**	**209**	**221**	**419**	**443**	**96**	**101**	**192**	**203**
2003	**917**	**940**	**1834**	**1879**	**193**	**204**	**385**	**408**	**88**	**93**	**176**	**185**
2004	**953**	**970**	**1905**	**1940**	**182**	**193**	**364**	**386**	**82**	**87**	**164**	**174**
2005	**848**	**863**	**1697**	**1726**	**171**	**181**	**342**	**361**	**72**	**77**	**145**	**153**
2006	**783**	**799**	**1567**	**1598**	**153**	**162**	**306**	**323**	**66**	**70**	**133**	**140**
2007	**770**	**783**	**1539**	**1566**	**138**	**146**	**276**	**292**	**59**	**62**	**118**	**124**
2008	**722**	**737**	**1444**	**1474**	**137**	**145**	**274**	**290**	**56**	**60**	**113**	**120**
2009	**762**	**777**	**1524**	**1553**	**129**	**136**	**258**	**272**	**52**	**55**	**104**	**110**
2010	**712**	**724**	**1424**	**1449**	**115**	**121**	**229**	**243**	**47**	**50**	**94**	**99**
2011	**597**	**609**	**1194**	**1217**	**103**	**108**	**205**	**217**	**43**	**45**	**86**	**91**
2012	**588**	**598**	**1176**	**1195**	**100**	**106**	**200**	**211**	**41**	**43**	**82**	**86**
2013	**518**	**529**	**1036**	**1057**	**94**	**99**	**187**	**198**	**36**	**38**	**72**	**76**
2014	**504**	**513**	**1009**	**1026**	**101**	**107**	**202**	**214**	**35**	**37**	**69**	**73**
2015	**468**	**476**	**936**	**952**	**99**	**104**	**198**	**209**	**33**	**35**	**66**	**70**
2016	**512**	**523**	**1025**	**1045**	**100**	**105**	**199**	**211**	**34**	**36**	**68**	**72**
2017	**524**	**533**	**1047**	**1066**	**95**	**100**	**189**	**200**	**33**	**34**	**65**	**69**
2018	**579**	**591**	**1157**	**1182**	**99**	**104**	**197**	**208**	**34**	**36**	**68**	**72**

Bold values indicate exceedances

304346

TABLE 5-65: RATIO OF MODELED EGG CONCENTRATIONS BASED ON FISHRAND FOR FEMALE GREAT BLUE HERON USING TEQ FOR THE PERIOD 1993 - 2018 REVISED

Year	LOAEL 189 Average	LOAEL 189 95% UCL	NOAEL 189 Average	NOAEL 189 95% UCL	LOAEL 168 Average	LOAEL 168 95% UCL	NOAEL 168 Average	NOAEL 168 95% UCL	LOAEL 154 Average	LOAEL 154 95% UCL	NOAEL 154 Average	NOAEL 154 95% UCL
1993	149	152	248	253	23	25	39	41	10	11	17	18
1994	152	155	254	259	22	24	37	39	9.4	10	16	17
1995	138	141	230	234	19	20	32	34	8.5	9.0	14	15
1996	77	79	128	131	16	17	26	28	7.5	7.9	12	13
1997	46	47	77	79	8.7	9.4	15	16	4.3	4.7	7.2	7.8
1998	62	64	104	106	14	15	24	25	5.7	6.0	9.5	10
1999	58	59	97	99	13	14	22	23	5.5	5.8	9.1	10
2000	50	51	84	85	11	12	19	20	4.7	5.0	7.9	8.3
2001	45	46	76	77	9.4	9.9	16	17	4.3	4.5	7.1	7.5
2002	46	47	77	78	8.4	8.9	14	15	3.9	4.1	6.5	6.9
2003	45	46	76	77	7.8	8.2	13	14	3.6	3.8	6.0	6.3
2004	47	48	79	80	7.3	7.7	12	13	3.4	3.6	5.6	5.9
2005	42	43	70	71	6.9	7.3	12	12	3.0	3.1	4.9	5.2
2006	39	40	65	66	6.2	6.5	10	11	2.7	2.9	4.5	4.8
2007	38	39	64	65	5.5	5.8	9.2	10	2.4	2.5	4.0	4.2
2008	36	37	60	61	5.6	5.9	9.3	10	2.3	2.5	3.9	4.1
2009	38	39	64	65	5.2	5.5	8.7	9.2	2.1	2.3	3.6	3.8
2010	36	36	60	60	4.6	4.9	7.7	8.1	1.9	2.0	3.2	3.4
2011	30	30	50	51	4.1	4.4	6.9	7.3	1.8	1.9	2.9	3.1
2012	30	30	49	50	4.1	4.3	6.8	7.2	1.7	1.8	2.8	3.0
2013	26	26	43	44	3.8	4.0	6.3	6.6	1.5	1.6	2.5	2.6
2014	25	26	42	43	4.1	4.3	6.8	7.2	1.4	1.5	2.4	2.5
2015	23	24	39	40	4.0	4.2	6.6	7.0	1.4	1.4	2.3	2.4
2016	26	26	43	44	4.1	4.3	6.8	7.1	1.4	1.5	2.4	2.5
2017	26	27	44	45	3.8	4.0	6.4	6.7	1.4	1.4	2.3	2.4
2018	29	30	49	50	4.1	4.3	6.8	7.1	1.4	1.5	2.4	2.5

Bold values indicate exceedances

TABLE 5-66: RATIO OF MODELED EGG CONCENTRATIONS BASED ON FISHRAND FOR FEMALE BALD EAGLE USING TEQ FOR THE PERIOD 1993 - 2018

REVISED

Year	LOAEL 189 Average	LOAEL 189 95% UCL	NOAEL 189 Average	NOAEL 189 95% UCL	LOAEL 168 Average	LOAEL 168 95% UCL	NOAEL 168 Average	NOAEL 168 95% UCL	LOAEL 154 Average	LOAEL 154 95% UCL	NOAEL 154 Average	NOAEL 154 95% UCL
1993	NA	NA	**300**	**314**	NA	NA	**87**	**88**	NA	NA	**43**	**43**
1994	NA	NA	**276**	**288**	NA	NA	**78**	**80**	NA	NA	**39**	**40**
1995	NA	NA	**315**	**328**	NA	NA	**78**	**79**	NA	NA	**38**	**38**
1996	NA	NA	**276**	**290**	NA	NA	**72**	**73**	NA	NA	**33**	**34**
1997	NA	NA	**225**	**236**	NA	NA	**64**	**65**	NA	NA	**30**	**30**
1998	NA	NA	**237**	**248**	NA	NA	**60**	**61**	NA	NA	**27**	**28**
1999	NA	NA	**244**	**255**	NA	NA	**57**	**57**	NA	NA	**25**	**26**
2000	NA	NA	**249**	**261**	NA	NA	**55**	**56**	NA	NA	**24**	**24**
2001	NA	NA	**209**	**220**	NA	NA	**50**	**50**	NA	NA	**22**	**22**
2002	NA	NA	**188**	**196**	NA	NA	**44**	**45**	NA	NA	**20**	**20**
2003	NA	NA	**198**	**206**	NA	NA	**41**	**42**	NA	NA	**18**	**18**
2004	NA	NA	**163**	**170**	NA	NA	**39**	**39**	NA	NA	**16**	**16**
2005	NA	NA	**147**	**154**	NA	NA	**40**	**40**	NA	NA	**15**	**16**
2006	NA	NA	**138**	**145**	NA	NA	**40**	**40**	NA	NA	**15**	**15**
2007	NA	NA	**155**	**162**	NA	NA	**40**	**40**	NA	NA	**14**	**15**
2008	NA	NA	**172**	**180**	NA	NA	**39**	**40**	NA	NA	**14**	**14**
2009	NA	NA	**163**	**171**	NA	NA	**38**	**38**	NA	NA	**14**	**14**
2010	NA	NA	**168**	**175**	NA	NA	**36**	**37**	NA	NA	**13**	**13**
2011	NA	NA	**146**	**154**	NA	NA	**35**	**36**	NA	NA	**13**	**13**
2012	NA	NA	**126**	**132**	NA	NA	**33**	**34**	NA	NA	**12**	**13**
2013	NA	NA	**123**	**129**	NA	NA	**31**	**31**	NA	NA	**12**	**12**
2014	NA	NA	**142**	**148**	NA	NA	**32**	**32**	NA	NA	**12**	**12**
2015	NA	NA	**135**	**141**	NA	NA	**30**	**31**	NA	NA	**12**	**12**
2016	NA	NA	**121**	**127**	NA	NA	**27**	**27**	NA	NA	**10**	**11**
2017	NA	NA	**108**	**113**	NA	NA	**26**	**26**	NA	NA	**11**	**11**
2018	NA	NA	**128**	**133**	NA	NA	**27**	**27**	NA	NA	**11**	**11**

Bold values indicate exceedances

304347

TABLE 5-67: WILDLIFE SURVEY RESULTS - Birds (UNCHANGED)
Hudson River
New York

Information Source	Date	Contact	Response	Contact Information	Data Available	Information/Findings
Birds						
Hudsonia	2-Jun-99	Call/Fax	YES; Spoke with on 6/2/1999	Eric Kiviat, Executive Director; (914) 758-7273 (7274) OR (914) 758-7053; FAX: (914) 758-7033; EMAIL: kiviat@bard.edu	He has no direct knowledge of the upper Hudson but provided names	WATERFOWL/MALLARD: Steve Brown - Delmar NYSDEC KINGFISHER: Breeding bird atlas - DEC now computerized on web page; Bob Anderle/Janet Carroll - NYSDEC NYSDEC - Natural Resource Damage Assessment
NYS Department of Environmental Conservation Endangered Species Unit	3-Jun-99	Call	No	Peter Nye (518) 439-7635x9 (Eagle Specialist); www.dec.state.ny.us	Left Message - Will call back	
Manomet Center for Conservation Sciences	2-Jun-99	Email	No	John M. Hagen, Division Director (Conservation Forestry Staff); jmhagan@ime.net; www.manomet.org;	Left Message - Will call back	
Saratoga National Historic Park, Stillwater, NY	4-Jun-99	Call	No	Chris (wildlife manager) (518) 664-9821x5; also can contact Richard Beresford	Left Message - Will call back	
Federation of New York State Bird Clubs	3-Jun-99	Email	No	Valeria Freer, President (vfreer@sullivan.suny.edu); http://www.birds.cornell.edu/fnysbc		
Union College Professor Emeritus	2-Jun-1999 7-Jun-1999	Call Call	No Yes	Carl George (518) 388-6330; Bird Expert; (John Waldman - Hudson River Foundation Recommended I call)	He did not have any specific data, but recommended a number of different sources	He recommended that I contact: Bob Daniels (mammals) - NY State Museum; Walter Sabin (Hudson-Mohawk Bird Club, they do an intensive waterbird survey and publish results in the *Kingbird Journal* (518) 439-7344; Also Union College has survey information for a lake in Scotia near the Hudson for Collins Lake in Scotia (across river from Schenectady) - http://tardis.union.edu/~birds, presents 10 years of bird information - 15 air miles from Hudson; also recommended contacting Robert Yunick for regional baseline information from Audubon Christmas count and the mid-May Big Day
Manomet Center for Conservation Sciences	7-Jun-99	Email	No	Dr. Treavor Lloyd-Evans (tlloyd-evans@manomet.org) - avian expert	Avian Conservationist	

TABLE 5-67: WILDLIFE SURVEY RESULTS - Birds (UNCHANGED)

Hudson River

New York

Information Source	Date	Contact	Response	Contact Information	Data Available	Information/Findings
American Birding Association - Online	7-Jun-99	WWW	No	www.americanbirding.org	Good links - possibility for some bird information on Hudson	
Breeding Bird Survey - OnLine	7-Jun-99	WWW	No	www.mbr.nbs.gov/bbs/bbs.html	Regional trend analysis by species - region=NY State, some additional details may be available	
Hudson-Mohawk Bird Club	7-Jun-99	Call	No	Walter Sabin Home: (518) 439-7344	Intensive waterbird survey every year - publish results in Kingbird Journal	
Ornithologist	7-Jun-99	Need Number	No	Robert Yunick	regional baseline data from Audubon Christmas count and mid-May Big Day	
NYS Department of Environmental Conservation Endangered Species Unit	8-Jun-99	WWW	No	www.dec.state.ny.us/website/dfwmr/wildlife/endspec/enspbird.html	Brief summaries, listed by species, for NY State.	*Ixobrychus exilis* (Least Bittern): Populations along Hudson River Valley, uncommon and rare breeder, declines due to loss of marsh habitat due to drainage, vegetational changes, pollution, insecticides. *Rallus elegans* (King Rail): Nesting was reported in northern Hudson Valley, however there are no confirmed nests in NY state currently, decline due to degradation of wetlands. *Bartramia longicauda* (Upland Sand Piper): once common around NY state including Hudson, less than 250 breeding sites to date in NY, decline due to loss of grassland habitat. All considered threatened species.
Andrle, R. F. and Carroll, J. R. (ed.) 1988. The Atlas of Breeding Birds in New York State. Cornell University Press, Ithica.	8-Jun-99				Regional trend analysis by species - region=NY State, some additional details may be available	*Tachycineta bicolor* (Tree Swallow): Common breeder throughout entire state. *Ceryle alcyon* (Belted Kingfisher): Common summer resident throughout entire state. *Ardea herodias* (Great Blue Heron): Observed in Northern Hudson Valley, possibility of breeding there. *Anas platyrhynchos* (Mallard): Common breeder in wetlands. In the 1900's, rarely if ever seen as a breeder; creation/improvement of wetlands in mid-1900's and release of captive-bred adults and ducklings in the 1950's caused populations to increase. Birds not found in Northern Hudson Valley: Eagles and Osprey.

304349

TABLE 5-67: WILDLIFE SURVEY RESULTS - Birds (UNCHANGED)
Hudson River
New York

Information Source	Date	Contact	Response	Contact Information	Data Available	Information/Findings
NYSDEC	16-Jun-99	Call	Yes	Mark Brown (518) 623-3671	Familiar with the area regarding mammals, birds, and herps. Good source. See General Info page.	This area is rich in birds including water fowl. Bald Eagle is only a winter resident, migrates in the summer. Lots of Canada geese and mallard. Has not seen any Osprey nests. They only feed here and spend most of their time around the near-by lakes. Has also seen tree swallow, kingfisher, and great blue heron. Most of the water fowl and larger birds use the area for feeding but do not breed here. He hasn't seen many nests except those built by species which live in the more wooded areas. Here's a list of the other species he has seen in the area: Common Mergenser (Diving Duck), red tailed hawk, sparrow hawk, rough grouse, wild turkey, killdare, wood cock, morning dove, barn owl, bard owl, sawhat owl (occupying nest boxes built for ducks), swallows, ravens, crows, wrens, eastern blue bird, starlings.
Ndakinna Wilderness Project	6/3/1999 6/16/99	Email Call Call	No No Yes	Jim Brushek (518) 583-9980x3, 23 Middle Grove Road, Greenfield Center, NY 12833; Received address from Saratoga County Information - Annamaria Dalton (annamaria@spa.net)	Professional Tracker	Saw some bald eagles 3 or 4 weeks ago. Hasn't seen any osprey. Great Blue Heron and kingfisher in large numbers. Hasn't seen any tree swallow. Lots of mallards and Canada geese. Could not recall seeing any nests.

TABLE 5-68: RATIO OF MODELED DIETARY DOSES TO BENCHMARKS FOR FEMALE BATS BASED ON 1993 DATA FOR THE TRI+ CONGENERS (UNCHANGED)

Location	LOAEL vs. Average ADD Hazard Quotient	LOAEL vs. 95% UCL ADD Hazard Quotient	NOAEL vs. Average ADD Hazard Quotient	NOAEL vs. 95% UCL ADD Hazard Quotient
Upper River				
Thompson Island Pool (189)	**30**	**52**	**140**	**244**
Stillwater (168)	**52**	**244**	**245**	**1146**
Federal Dam (154)	**12**	**21**	**58**	**99**
Lower River				
143.5	**1.7**	**4.3**	**8.1**	**20**
137.2	**3.4**	**14**	**16**	**65**
122.4	**1.9**	**4.7**	**8.8**	**22**
113.8	**1.9**	**7.5**	**9.1**	**35**
100	0.9	**6.1**	**4.2**	**29**
88.9	0.4	0.8	**2.1**	**3.7**
58.7	**1.4**	**13**	**6.5**	**59**
47.3	**1.6**	**11**	**7.3**	**54**
25.8	0.5	0.8	**2.2**	**3.7**

Bold values indicate exceedances

TABLE 5-69: RATIO OF MODELED DIETARY DOSES TO TOXICITY BENCHMARKS FOR FEMALE BAT FOR TRI+ CONGENERS FOR THE PERIOD 1993 - 2018 REVISED

Year	LOAEL 189 Average	LOAEL 189 95% UCL	NOAEL 189 Average	NOAEL 189 95% UCL	LOAEL 168 Average	LOAEL 168 95% UCL	NOAEL 168 Average	NOAEL 168 95% UCL	LOAEL 154 Average	LOAEL 154 95% UCL	NOAEL 154 Average	NOAEL 154 95% UCL
1993	32	33	148	157	26	28	122	130	11	11	49	53
1994	29	31	137	145	24	26	114	121	9.8	10	46	49
1995	27	29	126	134	22	23	103	109	9.0	9.6	42	45
1996	24	26	113	120	19	20	88	94	8.1	8.6	38	41
1997	22	23	101	106	16	17	75	80	7.4	7.8	35	37
1998	13	14	62	66	18	19	83	88	6.6	7.0	31	33
1999	12	13	58	61	16	17	75	79	6.0	6.4	28	30
2000	11	12	52	55	14	15	64	69	5.4	5.8	25	27
2001	10	11	47	50	12	12	54	58	4.8	5.2	23	24
2002	9.3	10	44	46	10	11	47	50	4.4	4.7	20	22
2003	8.5	9.0	40	42	9.2	9.7	43	46	4.0	4.3	19	20
2004	7.9	8.3	37	39	8.7	9.2	41	43	3.6	3.9	17	18
2005	7.3	7.7	34	36	8.0	8.6	38	40	3.3	3.5	15	16
2006	6.7	7.1	32	33	7.3	7.7	34	36	2.9	3.1	14	15
2007	6.1	6.4	29	30	6.7	7.1	31	33	2.6	2.8	12	13
2008	5.7	6.0	27	28	6.3	6.7	29	31	2.4	2.6	11	12
2009	5.3	5.6	25	26	6.0	6.3	28	30	2.3	2.4	11	11
2010	4.8	5.1	23	24	5.4	5.8	25	27	2.1	2.2	9.7	10
2011	4.3	4.6	20	21	4.9	5.2	23	24	1.9	2.0	8.9	9.5
2012	3.9	4.1	18	19	4.4	4.7	21	22	1.7	1.8	8.0	8.5
2013	3.5	3.7	17	18	4.5	4.8	21	23	1.6	1.7	7.4	7.9
2014	3.3	3.4	15	16	4.8	5.1	22	24	1.5	1.6	7.0	7.4
2015	3.0	3.2	14	15	4.7	5.0	22	23	1.4	1.5	6.6	7.0
2016	2.9	3.0	13	14	4.5	4.8	21	23	1.4	1.5	6.5	6.9
2017	2.7	2.8	13	13	4.4	4.7	21	22	1.3	1.4	6.2	6.6
2018	2.5	2.7	12	12	4.3	4.5	20	21	1.3	1.4	6.0	6.4

Bold values indicate exceedances

TABLE 5-70: RATIO OF MODELED DIETARY DOSES TO BENCHMARKS FOR FEMALE BAT BASED ON 1993 DATA ON A TEQ BASIS (UNCHANGED)

Location	LOAEL vs. Average ADD Hazard Quotient	LOAEL vs. 95% UCL ADD Hazard Quotient	NOAEL vs. Average ADD Hazard Quotient	NOAEL vs. 95% UCL ADD Hazard Quotient
Upper River				
Thompson Island Pool (189)	**133**	**232**	**1328**	**2323**
Stillwater (168)	**232**	**1089**	**2324**	**10885**
Federal Dam (154)	**55**	**94**	**554**	**943**
Lower River				
143.5	**8**	**20**	**78**	**197**
137.2	**15**	**62**	**153**	**624**
122.4	**8.4**	**22**	**84**	**215**
113.8	**8.7**	**34**	**87**	**339**
100	**4.0**	**28**	**40**	**276**
88.9	**2.0**	**3.6**	**20**	**36**
58.7	**6.2**	**56**	**62**	**562**
47.3	**7.0**	**51**	**70**	**512**
25.8	**2.1**	**3.6**	**21**	**36**

Bold values indicate exceedances

TABLE 5-71: RATIO OF MODELED DIETARY DOSES TO TOXICITY BENCHMARKS FOR FEMALE BAT ON A TEQ BASIS FOR THE PERIOD 1993 - 2018

REVISED

Year	LOAEL 189 Average	LOAEL 189 95% UCL	NOAEL 189 Average	NOAEL 189 95% UCL	LOAEL 168 Average	LOAEL 168 95% UCL	NOAEL 168 Average	NOAEL 168 95% UCL	LOAEL 154 Average	LOAEL 154 95% UCL	NOAEL 154 Average	NOAEL 154 95% UCL
1993	141	149	1410	1491	116	124	1164	1238	47	50	469	499
1994	130	138	1304	1378	108	115	1084	1154	43	46	435	463
1995	120	127	1203	1271	98	104	976	1038	40	43	403	429
1996	108	114	1076	1137	84	89	836	889	36	39	362	385
1997	96	101	960	1012	72	76	716	761	33	35	329	350
1998	59	63	594	627	79	84	786	837	29	31	292	311
1999	55	58	552	583	71	75	708	754	27	29	270	287
2000	50	53	498	527	61	65	612	651	24	26	242	258
2001	45	48	450	475	52	55	517	550	22	23	216	230
2002	41	44	414	438	45	47	445	474	19	21	195	207
2003	38	40	380	401	41	43	408	434	18	19	179	190
2004	35	37	352	372	39	41	387	412	16	17	162	173
2005	33	35	327	346	36	38	358	381	15	15	145	154
2006	30	32	300	318	32	34	324	344	13	14	130	138
2007	27	29	271	287	30	32	298	317	12	12	117	125
2008	25	27	254	268	28	30	279	297	11	12	109	116
2009	24	25	237	250	27	28	266	283	10	11	101	108
2010	21	23	215	227	24	26	241	257	9.3	9.9	93	99
2011	19	20	193	204	22	23	217	231	8.5	9.0	85	90
2012	17	18	174	184	20	21	197	210	7.6	8.1	76	81
2013	16	17	158	167	20	21	201	214	7.1	7.5	71	75
2014	15	15	145	154	21	23	212	226	6.6	7.1	66	71
2015	14	14	136	144	21	22	208	221	6.3	6.7	63	67
2016	13	13	128	135	20	22	202	215	6.2	6.6	62	66
2017	12	13	120	126	20	21	197	210	5.9	6.3	59	63
2018	11	12	113	119	19	20	190	202	5.7	6.1	57	61

Bold values indicate exceedances

TABLE 5-72: RATIO OF MODELED DIETARY DOSES TO BENCHMARKS FOR FEMALE RACCOON BASED ON 1993 DATA FOR THE TRI+ CONGENERS REVISED

Location	LOAEL vs. Average ADD Hazard Quotient	LOAEL vs. 95% UCL ADD Hazard Quotient	NOAEL vs. Average ADD Hazard Quotient	NOAEL vs. 95% UCL ADD Hazard Quotient
Upper River				
Thompson Island Pool (189)	**5.8**	**10**	**27**	**47**
Stillwater (168)	**9.5**	**42**	**45**	**195**
Federal Dam (154)	**2.2**	**3.7**	**10**	**17**
Lower River				
143.5	0.4	0.8	**1.7**	**3.7**
137.2	0.7	**2.6**	**3.4**	**12**
122.4	0.4	0.9	**1.8**	**4.1**
113.8	0.4	**1.3**	**1.9**	**6.2**
100	0.2	**1.3**	0.8	**6.1**
88.9	0.1	0.3	0.6	**1.2**
58.7	0.3	**2.2**	**1.3**	**10**
47.3	0.3	**2.1**	**1.6**	**9.9**
25.8	0.1	0.2	0.6	**1.0**

Bold values indicate exceedances

304356

TABLE 5-73: RATIO OF MODELED DIETARY DOSES TO TOXICITY BENCHMARKS FOR FEMALE RACCOON FOR TRI+ CONGENERS FOR THE PERIOD 1993 - 2018 REVISED

Year	LOAEL 189 Average	LOAEL 189 95% UCL	NOAEL 189 Average	NOAEL 189 95% UCL	LOAEL 168 Average	LOAEL 168 95% UCL	NOAEL 168 Average	NOAEL 168 95% UCL	LOAEL 154 Average	LOAEL 154 95% UCL	NOAEL 154 Average	NOAEL 154 95% UCL
1993	**7.3**	**7.6**	**34**	**36**	**4.5**	**4.8**	**21**	**23**	**1.8**	**1.9**	**8.5**	**9.0**
1994	**6.7**	**7.0**	**32**	**33**	**4.2**	**4.5**	**20**	**21**	**1.7**	**1.8**	**7.9**	**8.4**
1995	**6.3**	**6.5**	**29**	**31**	**3.8**	**4.0**	**18**	**19**	**1.6**	**1.6**	**7.3**	**7.7**
1996	**5.5**	**5.7**	**26**	**27**	**3.3**	**3.5**	**15**	**16**	**1.4**	**1.5**	**6.5**	**6.9**
1997	**4.8**	**5.0**	**22**	**23**	**2.8**	**2.9**	**13**	**14**	**1.3**	**1.3**	**5.9**	**6.3**
1998	**3.0**	**3.2**	**14**	**15**	**3.1**	**3.3**	**14**	**15**	**1.1**	**1.2**	**5.3**	**5.6**
1999	**2.8**	**3.0**	**13**	**14**	**2.8**	**2.9**	**13**	**14**	**1.0**	**1.1**	**4.9**	**5.2**
2000	**2.6**	**2.7**	**12**	**13**	**2.4**	**2.5**	**11**	**12**	0.9	**1.0**	**4.4**	**4.6**
2001	**2.3**	**2.4**	**11**	**11**	**2.0**	**2.1**	**9.4**	**10**	0.8	0.9	**3.9**	**4.1**
2002	**2.1**	**2.2**	**10**	**10**	**1.7**	**1.8**	**8.1**	**8.6**	0.8	0.8	**3.5**	**3.7**
2003	**2.0**	**2.1**	**9.3**	**9.6**	**1.6**	**1.7**	**7.4**	**7.9**	0.7	0.7	**3.2**	**3.4**
2004	**1.8**	**1.9**	**8.6**	**9.0**	**1.5**	**1.6**	**7.1**	**7.5**	0.6	0.7	**2.9**	**3.1**
2005	**1.7**	**1.8**	**8.0**	**8.3**	**1.4**	**1.5**	**6.5**	**6.9**	0.6	0.6	**2.6**	**2.8**
2006	**1.6**	**1.6**	**7.4**	**7.7**	**1.3**	**1.3**	**5.9**	**6.3**	0.5	0.5	**2.3**	**2.5**
2007	**1.4**	**1.5**	**6.7**	**7.0**	**1.2**	**1.2**	**5.4**	**5.8**	0.5	0.5	**2.1**	**2.3**
2008	**1.3**	**1.4**	**6.2**	**6.5**	**1.1**	**1.2**	**5.1**	**5.4**	0.4	0.4	**2.0**	**2.1**
2009	**1.3**	**1.3**	**5.9**	**6.1**	**1.0**	**1.1**	**4.9**	**5.2**	0.4	0.4	**1.8**	**1.9**
2010	**1.1**	**1.2**	**5.4**	**5.6**	0.9	**1.0**	**4.4**	**4.7**	0.4	0.4	**1.7**	**1.8**
2011	**1.0**	**1.1**	**4.8**	**5.0**	0.8	0.9	**4.0**	**4.2**	0.3	0.3	**1.5**	**1.6**
2012	0.9	**1.0**	**4.4**	**4.6**	0.8	0.8	**3.6**	**3.8**	0.3	0.3	**1.4**	**1.5**
2013	0.8	0.9	**4.0**	**4.1**	0.8	0.8	**3.7**	**3.9**	0.3	0.3	**1.3**	**1.4**
2014	0.8	0.8	**3.7**	**3.8**	0.8	0.9	**3.9**	**4.1**	0.3	0.3	**1.2**	**1.3**
2015	0.7	0.8	**3.4**	**3.5**	0.8	0.9	**3.8**	**4.0**	0.2	0.3	**1.1**	**1.2**
2016	0.7	0.7	**3.3**	**3.4**	0.8	0.8	**3.7**	**3.9**	0.2	0.3	**1.1**	**1.2**
2017	0.7	0.7	**3.1**	**3.2**	0.8	0.8	**3.6**	**3.8**	0.2	0.2	**1.1**	**1.1**
2018	0.6	0.7	**3.0**	**3.1**	0.7	0.8	**3.5**	**3.7**	0.2	0.2	**1.0**	**1.1**

Bold values indicate exceedances

TABLE 5-74: RATIO OF MODELED DIETARY DOSES TO BENCHMARKS FOR FEMALE RACCOON BASED ON 1993 DATA ON A TEQ BASIS (UNCHANGED)

Location	LOAEL vs. Average ADD Hazard Quotient	LOAEL vs. 95% UCL ADD Hazard Quotient	NOAEL vs. Average ADD Hazard Quotient	NOAEL vs. 95% UCL ADD Hazard Quotient
Upper River				
Thompson Island Pool (189)	**69**	**107**	**685**	**1067**
Stillwater (168)	**150**	**374**	**1504**	**3736**
Federal Dam (154)	**19**	**33**	**195**	**328**
Lower River				
143.5	**4.8**	**7.3**	**48**	**73**
137.2	**8.8**	**23**	**88**	**231**
122.4	**5.1**	**8.0**	**51**	**80**
113.8	**5.4**	**12**	**54**	**120**
100	**2.2**	**36**	**22**	**359**
88.9	**3.4**	**9.2**	**34**	**92**
58.7	**2.2**	**20**	**22**	**196**
47.3	**7.0**	**30**	**70**	**304**
25.8	**2.6**	**6.5**	**26**	**65**

Bold values indicate exceedances

304358

TABLE 5-75: RATIO OF MODELED DIETARY DOSES TO TOXICITY BENCHMARKS FOR FEMALE RACCOON ON A TEQ BASIS FOR THE PERIOD 1993 - 2018
REVISED

Year	LOAEL 189 Average	LOAEL 189 95% UCL	NOAEL 189 Average	NOAEL 189 95% UCL	LOAEL 168 Average	LOAEL 168 95% UCL	NOAEL 168 Average	NOAEL 168 95% UCL	LOAEL 154 Average	LOAEL 154 95% UCL	NOAEL 154 Average	NOAEL 154 95% UCL
1993	221	223	2214	2228	45	46	449	461	16	16	157	162
1994	201	202	2006	2018	41	43	414	426	14	15	145	149
1995	189	190	1893	1905	38	39	383	393	13	14	135	139
1996	169	170	1688	1699	33	34	329	338	12	13	122	126
1997	150	151	1501	1510	28	29	278	286	11	11	109	113
1998	91	91	909	915	30	31	303	312	10	10	97	100
1999	86	87	863	869	28	28	275	283	9.0	9.3	90	93
2000	78	79	784	788	24	25	241	248	8.2	8.4	82	84
2001	70	70	698	702	20	21	202	207	7.2	7.5	72	75
2002	65	65	646	650	17	18	174	178	6.5	6.7	65	67
2003	59	60	591	595	15	16	155	159	6.0	6.1	60	61
2004	54	55	544	547	15	15	148	152	5.5	5.6	55	56
2005	51	51	507	510	14	14	139	143	4.9	5.0	49	50
2006	47	48	472	475	12	13	125	128	4.4	4.5	44	45
2007	43	43	427	429	11	12	114	117	3.9	4.0	39	40
2008	39	39	390	392	11	11	106	109	3.6	3.7	36	37
2009	37	37	371	373	10	11	102	105	3.4	3.5	34	35
2010	34	34	339	341	9.4	10	94	96	3.1	3.2	31	32
2011	30	31	303	305	8.4	8.6	84	86	2.8	2.9	28	29
2012	28	28	275	277	7.6	7.8	76	78	2.6	2.7	26	27
2013	25	25	249	250	7.3	7.5	73	75	2.3	2.4	23	24
2014	23	23	227	229	8.1	8.3	81	83	2.2	2.3	22	23
2015	21	21	210	212	7.8	8.1	78	81	2.1	2.2	21	22
2016	20	20	198	199	7.7	7.9	77	79	2.0	2.1	20	21
2017	19	19	187	188	7.5	7.7	75	77	2.0	2.0	20	20
2018	18	18	175	176	7.2	7.4	72	74	1.9	2.0	19	20

Bold values indicate exceedances

TABLE 5-76
CONCENTRATIONS TO BENCHMARKS (UNCHANGED)

Comparison to Low Range LOAEL				
Species and Statistic	North Hudson Valley	South Hudson Valley	Hudson Valley	Other NY State
Mink liver - average	0.5	0.6		
Mink liver - minimum	0.1	0.1		
Mink liver - maximum	**1.4**	**2.8**		
Otter liver - average			**1.9**	
Otter liver - minimum			0.6	
Otter liver - maximum			**5.9**	

Comparison to Upper Range LOAEL				
Species and Statistic	North Hudson Valley	South Hudson Valley	Hudson Valley	Other NY State
Mink liver - average	0.2	0.2		
Mink liver - minimum	0.0	0.0		
Mink liver - maximum	0.5	**1.1**		
Otter liver - average			0.7	
Otter liver - minimum			0.2	
Otter liver - maximum			**2.4**	

TABLE 5-77: RATIO OF MODELED DIETARY DOSES TO BENCHMARKS FOR FEMALE MINK BASED ON 1993 DATA FOR THE TRI+ CONGENERS (UNCHANGED)

Location	LOAEL vs. Average ADD Hazard Quotient	LOAEL vs. 95% UCL ADD Hazard Quotient	NOAEL vs. Average ADD Hazard Quotient	NOAEL vs. 95% UCL ADD Hazard Quotient
Upper River				
Thompson Island Pool (189)	**11**	**17**	**359**	**566**
Stillwater (168)	**5.8**	**23**	**188**	**760**
Federal Dam (154)	**1.8**	**2.8**	**58**	**92**
Lower River				
143.5	**1.0**	**1.3**	**31**	**43**
137.2	**1.9**	**4.6**	**62**	**150**
122.4	0.8	**1.4**	**25**	**45**
113.8	0.8	**1.3**	**27**	**43**
100	0.4	**1.0**	**12**	**34**
88.9	0.6	0.8	**19**	**26**
58.7	0.7	**1.8**	**24**	**58**
47.3	0.7	**1.7**	**22**	**56**
25.8	0.5	0.6	**15**	**18**

Bold values indicate exceedances

TABLE 5-78: RATIO OF MODELED DIETARY DOSES TO BENCHMARKS
FOR FEMALE OTTER BASED ON 1993 DATA FOR THE TRI+ CONGENERS (UNCHANGED)

Location	LOAEL vs. Average ADD Hazard Quotient	LOAEL vs. 95% UCL ADD Hazard Quotient	NOAEL vs. Average ADD Hazard Quotient	NOAEL vs. 95% UCL ADD Hazard Quotient
Upper River				
Thompson Island Pool (189)	**89**	**173**	**2906**	**5623**
Stillwater (168)	**16**	**21**	**520**	**671**
Federal Dam (154)	**12**	**21**	**375**	**673**
Lower River				
143.5	**12**	**21**	**375**	**673**
137.2	**43**	**103**	**1413**	**3362**
122.4	**10**	**14**	**329**	**453**
113.8	**9.2**	**13**	**298**	**417**
100	**11**	**33**	**342**	**1057**
88.9	**6.8**	**13**	**220**	**419**
58.7	**7.9**	**12**	**256**	**379**
47.3	**9.0**	**24**	**293**	**789**
25.8	**6.4**	**13**	**207**	**411**

Bold values indicate exceedances

TABLE 5-79: RATIO OF MODELED DIETARY DOSES TO TOXICITY BENCHMARKS FOR FEMALE MINK FOR TRI+ CONGENERS FOR THE PERIOD 1993 - 2018 REVISED

Year	LOAEL 189 Average	LOAEL 189 95% UCL	NOAEL 189 Average	NOAEL 189 95% UCL	LOAEL 168 Average	LOAEL 168 95% UCL	NOAEL 168 Average	NOAEL 168 95% UCL	LOAEL 154 Average	LOAEL 154 95% UCL	NOAEL 154 Average	NOAEL 154 95% UCL
1993	**9.4**	**9.6**	**304**	**313**	**3.3**	**3.5**	**107**	**113**	**1.4**	**1.4**	**44**	**47**
1994	**9.3**	**9.6**	**302**	**310**	**3.1**	**3.3**	**100**	**106**	**1.3**	**1.3**	**41**	**43**
1995	**8.5**	**8.7**	**275**	**283**	**2.7**	**2.9**	**89**	**94**	**1.1**	**1.2**	**37**	**40**
1996	**5.6**	**5.8**	**182**	**188**	**2.3**	**2.5**	**75**	**80**	**1.0**	**1.1**	**33**	**35**
1997	**4.0**	**4.2**	**131**	**135**	**1.8**	**1.9**	**58**	**62**	0.8	0.9	**27**	**29**
1998	**3.9**	**4.0**	**128**	**131**	**2.2**	**2.3**	**70**	**74**	0.8	0.9	**26**	**28**
1999	**3.7**	**3.8**	**119**	**122**	**1.9**	**2.1**	**63**	**67**	0.8	0.8	**25**	**26**
2000	**3.2**	**3.3**	**104**	**107**	**1.7**	**1.8**	**55**	**58**	0.7	0.7	**22**	**23**
2001	**2.9**	**3.0**	**94**	**97**	**1.4**	**1.5**	**46**	**49**	0.6	0.6	**20**	**21**
2002	**2.9**	**2.9**	**93**	**95**	**1.2**	**1.3**	**40**	**43**	0.5	0.6	**18**	**19**
2003	**2.7**	**2.8**	**89**	**92**	**1.1**	**1.2**	**37**	**39**	0.5	0.5	**16**	**17**
2004	**2.8**	**2.8**	**90**	**92**	**1.1**	**1.1**	**35**	**37**	0.5	0.5	**15**	**16**
2005	**2.5**	**2.6**	**81**	**83**	**1.0**	**1.1**	**32**	**34**	0.4	0.4	**13**	**14**
2006	**2.3**	**2.4**	**75**	**77**	0.9	**1.0**	**29**	**31**	0.4	0.4	**12**	**13**
2007	**2.2**	**2.3**	**72**	**74**	0.8	0.9	**27**	**28**	0.3	0.4	**11**	**11**
2008	**2.1**	**2.1**	**67**	**69**	0.8	0.8	**25**	**27**	0.3	0.3	**10**	**11**
2009	**2.1**	**2.2**	**69**	**71**	0.7	0.8	**24**	**26**	0.3	0.3	**9.4**	**10**
2010	**2.0**	**2.0**	**64**	**66**	0.7	0.7	**22**	**23**	0.3	0.3	**8.6**	**9.1**
2011	**1.7**	**1.7**	**55**	**56**	0.6	0.6	**20**	**21**	0.2	0.3	**7.8**	**8.3**
2012	**1.6**	**1.7**	**53**	**54**	0.6	0.6	**18**	**19**	0.2	0.2	**7.2**	**7.6**
2013	**1.4**	**1.5**	**47**	**48**	0.6	0.6	**18**	**19**	0.2	0.2	**6.5**	**6.9**
2014	**1.4**	**1.4**	**45**	**46**	0.6	0.6	**19**	**20**	0.2	0.2	**6.2**	**6.6**
2015	**1.3**	**1.3**	**42**	**43**	0.6	0.6	**19**	**20**	0.2	0.2	**5.9**	**6.2**
2016	**1.4**	**1.4**	**45**	**46**	0.6	0.6	**19**	**20**	0.2	0.2	**5.9**	**6.3**
2017	**1.4**	**1.4**	**45**	**46**	0.6	0.6	**18**	**19**	0.2	0.2	**5.7**	**6.0**
2018	**1.5**	**1.5**	**48**	**50**	0.5	0.6	**18**	**19**	0.2	0.2	**5.6**	**6.0**

Bold values indicate exceedances

TABLE 5-80: RATIO OF MODELED DIETARY DOSE TO TOXICITY BENCHMARKS FOR FEMALE OTTER FOR TRI+ CONGENERS FOR THE PERIOD 1993 - 2018

REVISED

Year	LOAEL 189 Average	LOAEL 189 95% UCL	NOAEL 189 Average	NOAEL 189 95% UCL	LOAEL 168 Average	LOAEL 168 95% UCL	NOAEL 168 Average	NOAEL 168 95% UCL	LOAEL 154 Average	LOAEL 154 95% UCL	NOAEL 154 Average	NOAEL 154 95% UCL
1993	**13**	**13**	**412**	**430**	**3.6**	**3.7**	**118**	**120**	**1.8**	**1.8**	**58**	**59**
1994	**12**	**12**	**378**	**395**	**3.3**	**3.3**	**107**	**108**	**1.6**	**1.7**	**53**	**54**
1995	**13**	**14**	**430**	**449**	**3.3**	**3.3**	**106**	**108**	**1.6**	**1.6**	**51**	**52**
1996	**12**	**12**	**377**	**396**	**3.0**	**3.0**	**97**	**99**	**1.4**	**1.4**	**45**	**46**
1997	**9.5**	**10**	**308**	**323**	**2.7**	**2.7**	**87**	**88**	**1.2**	**1.2**	**40**	**41**
1998	**10**	**10**	**322**	**337**	**2.5**	**2.5**	**82**	**83**	**1.1**	**1.2**	**37**	**38**
1999	**10**	**11**	**331**	**346**	**2.4**	**2.4**	**77**	**78**	**1.1**	**1.1**	**34**	**35**
2000	**10**	**11**	**338**	**354**	**2.3**	**2.3**	**75**	**76**	**1.0**	**1.0**	**32**	**33**
2001	**8.7**	**9.2**	**284**	**298**	**2.1**	**2.1**	**67**	**68**	0.9	0.9	**30**	**30**
2002	**7.9**	**8.2**	**255**	**267**	**1.8**	**1.9**	**60**	**61**	0.8	0.8	**27**	**27**
2003	**8.3**	**8.6**	**268**	**280**	**1.7**	**1.7**	**56**	**57**	0.8	0.8	**25**	**25**
2004	**6.8**	**7.1**	**221**	**232**	**1.6**	**1.6**	**52**	**53**	0.7	0.7	**22**	**22**
2005	**6.1**	**6.4**	**200**	**209**	**1.7**	**1.7**	**54**	**55**	0.6	0.6	**21**	**21**
2006	**5.8**	**6.0**	**188**	**196**	**1.7**	**1.7**	**54**	**55**	0.6	0.6	**20**	**20**
2007	**6.5**	**6.8**	**210**	**220**	**1.6**	**1.7**	**54**	**54**	0.6	0.6	**20**	**20**
2008	**7.2**	**7.5**	**233**	**244**	**1.6**	**1.6**	**53**	**54**	0.6	0.6	**19**	**20**
2009	**6.8**	**7.1**	**221**	**231**	**1.6**	**1.6**	**51**	**52**	0.6	0.6	**19**	**19**
2010	**7.0**	**7.3**	**227**	**237**	**1.5**	**1.5**	**49**	**50**	0.5	0.6	**18**	**18**
2011	**6.1**	**6.4**	**198**	**208**	**1.5**	**1.5**	**47**	**48**	0.5	0.5	**17**	**17**
2012	**5.3**	**5.5**	**171**	**179**	**1.4**	**1.4**	**45**	**46**	0.5	0.5	**17**	**17**
2013	**5.1**	**5.4**	**167**	**175**	**1.3**	**1.3**	**42**	**42**	0.5	0.5	**16**	**16**
2014	**5.9**	**6.2**	**192**	**200**	**1.3**	**1.3**	**43**	**43**	0.5	0.5	**17**	**17**
2015	**5.6**	**5.9**	**183**	**191**	**1.3**	**1.3**	**41**	**42**	0.5	0.5	**16**	**16**
2016	**5.0**	**5.3**	**164**	**171**	**1.1**	**1.1**	**37**	**37**	0.4	0.4	**14**	**14**
2017	**4.5**	**4.7**	**146**	**153**	**1.1**	**1.1**	**35**	**35**	0.4	0.4	**14**	**14**
2018	**5.3**	**5.5**	**173**	**180**	**1.1**	**1.1**	**36**	**37**	0.4	0.4	**14**	**15**

Bold values indicate exceedances

304363

TABLE 5-81: RATIO OF MODELED DIETARY DOSES TO BENCHMARKS FOR FEMALE MINK BASED ON 1993 DATA ON A TEQ BASIS REVISED

Location	LOAEL vs. Average ADD Hazard Quotient	LOAEL vs. 95% UCL ADD Hazard Quotient	NOAEL vs. Average ADD Hazard Quotient	NOAEL vs. 95% UCL ADD Hazard Quotient
Upper River				
Thompson Island Pool (189)	NA	NA	**792**	**1233**
Stillwater (168)	NA	NA	**510**	**1536**
Federal Dam (154)	NA	NA	**120**	**191**
Lower River				
143.5	NA	NA	**69**	**96**
137.2	NA	NA	**137**	**322**
122.4	NA	NA	**57**	**97**
113.8	NA	NA	**61**	**93**
100	NA	NA	**26**	**121**
88.9	NA	NA	**46**	**71**
58.7	NA	NA	**50**	**120**
47.3	NA	NA	**55**	**139**
25.8	NA	NA	**34**	**49**

Bold values indicate exceedances

TABLE 5-82: RATIO OF MODELED DIETARY DOSES TO BENCHMARKS FOR FEMALE OTTER BASED ON 1993 DATA ON A TEQ BASIS REVISED

Location	LOAEL vs. Average ADD Hazard Quotient	LOAEL vs. 95% UCL ADD Hazard Quotient	NOAEL vs. Average ADD Hazard Quotient	NOAEL vs. 95% UCL ADD Hazard Quotient
Upper River				
Thompson Island Pool (189)	NA	NA	**6286**	**12140**
Stillwater (168)	NA	NA	**1254**	**1683**
Federal Dam (154)	NA	NA	**817**	**1467**
Lower River				
143.5	NA	NA	**808**	**1453**
137.2	NA	NA	**3038**	**7230**
122.4	NA	NA	**711**	**978**
113.8	NA	NA	**644**	**904**
100	NA	NA	**735**	**2309**
88.9	NA	NA	**476**	**910**
58.7	NA	NA	**550**	**827**
47.3	NA	NA	**635**	**1720**
25.8	NA	NA	**447**	**890**

Bold values indicate exceedances

TABLE 5-83: RATIO OF MODELED DIETARY DOSES TO TOXICITY BENCHMARKS FOR FEMALE MINK ON A TEQ BASIS FOR THE PERIOD 1993 - 2018
REVISED

Year	LOAEL 189 Average	LOAEL 189 95% UCL	NOAEL 189 Average	NOAEL 189 95% UCL	LOAEL 168 Average	LOAEL 168 95% UCL	NOAEL 168 Average	NOAEL 168 95% UCL	LOAEL 154 Average	LOAEL 154 95% UCL	NOAEL 154 Average	NOAEL 154 95% UCL
1993	NA	NA	**949**	**965**	NA	NA	**227**	**238**	NA	NA	**91**	**95**
1994	NA	NA	**913**	**929**	NA	NA	**213**	**223**	NA	NA	**83**	**87**
1995	NA	NA	**843**	**857**	NA	NA	**191**	**200**	NA	NA	**76**	**80**
1996	NA	NA	**617**	**627**	NA	NA	**161**	**169**	NA	NA	**68**	**72**
1997	NA	NA	**484**	**491**	NA	NA	**124**	**130**	NA	NA	**54**	**57**
1998	NA	NA	**395**	**402**	NA	NA	**150**	**157**	NA	NA	**54**	**56**
1999	NA	NA	**371**	**377**	NA	NA	**135**	**142**	NA	NA	**50**	**53**
2000	NA	NA	**329**	**334**	NA	NA	**117**	**123**	NA	NA	**45**	**47**
2001	NA	NA	**295**	**300**	NA	NA	**99**	**103**	NA	NA	**40**	**42**
2002	NA	NA	**286**	**290**	NA	NA	**86**	**90**	NA	NA	**36**	**38**
2003	NA	NA	**271**	**276**	NA	NA	**78**	**82**	NA	NA	**33**	**35**
2004	NA	NA	**266**	**270**	NA	NA	**74**	**78**	NA	NA	**31**	**32**
2005	NA	NA	**241**	**245**	NA	NA	**69**	**73**	NA	NA	**27**	**29**
2006	NA	NA	**224**	**227**	NA	NA	**62**	**65**	NA	NA	**25**	**26**
2007	NA	NA	**212**	**215**	NA	NA	**57**	**59**	NA	NA	**22**	**23**
2008	NA	NA	**197**	**200**	NA	NA	**54**	**57**	NA	NA	**21**	**22**
2009	NA	NA	**198**	**201**	NA	NA	**52**	**54**	NA	NA	**19**	**20**
2010	NA	NA	**183**	**186**	NA	NA	**47**	**49**	NA	NA	**17**	**18**
2011	NA	NA	**158**	**161**	NA	NA	**42**	**44**	NA	NA	**16**	**17**
2012	NA	NA	**150**	**153**	NA	NA	**39**	**41**	NA	NA	**15**	**15**
2013	NA	NA	**134**	**136**	NA	NA	**38**	**39**	NA	NA	**13**	**14**
2014	NA	NA	**127**	**129**	NA	NA	**41**	**43**	NA	NA	**13**	**13**
2015	NA	NA	**118**	**120**	NA	NA	**40**	**42**	NA	NA	**12**	**13**
2016	NA	NA	**122**	**124**	NA	NA	**39**	**41**	NA	NA	**12**	**13**
2017	NA	NA	**121**	**123**	NA	NA	**38**	**40**	NA	NA	**12**	**12**
2018	NA	NA	**127**	**129**	NA	NA	**38**	**39**	NA	NA	**12**	**12**

Bold values indicate exceedances

TABLE 5-84: RATIO OF MODELED DIETARY DOSES TO TOXICITY BENCHMARKS FOR FEMALE OTTER ON A TEQ BASIS FOR THE PERIOD 1993 - 2018
REVISED

Year	LOAEL 189 Average	LOAEL 189 95% UCL	NOAEL 189 Average	NOAEL 189 95% UCL	LOAEL 168 Average	LOAEL 168 95% UCL	NOAEL 168 Average	NOAEL 168 95% UCL	LOAEL 154 Average	LOAEL 154 95% UCL	NOAEL 154 Average	NOAEL 154 95% UCL
1993	NA	NA	**1124**	**1163**	NA	NA	**285**	**289**	NA	NA	**135**	**136**
1994	NA	NA	**1027**	**1065**	NA	NA	**258**	**261**	NA	NA	**123**	**124**
1995	NA	NA	**1129**	**1168**	NA	NA	**256**	**259**	NA	NA	**118**	**119**
1996	NA	NA	**994**	**1033**	NA	NA	**232**	**235**	NA	NA	**104**	**106**
1997	NA	NA	**826**	**857**	NA	NA	**206**	**209**	NA	NA	**93**	**94**
1998	NA	NA	**791**	**821**	NA	NA	**197**	**199**	NA	NA	**86**	**87**
1999	NA	NA	**805**	**837**	NA	NA	**184**	**187**	NA	NA	**79**	**80**
2000	NA	NA	**811**	**845**	NA	NA	**179**	**181**	NA	NA	**74**	**75**
2001	NA	NA	**686**	**716**	NA	NA	**159**	**161**	NA	NA	**68**	**69**
2002	NA	NA	**618**	**643**	NA	NA	**141**	**143**	NA	NA	**62**	**63**
2003	NA	NA	**640**	**665**	NA	NA	**131**	**133**	NA	NA	**56**	**57**
2004	NA	NA	**533**	**556**	NA	NA	**123**	**124**	NA	NA	**50**	**50**
2005	NA	NA	**483**	**504**	NA	NA	**126**	**128**	NA	NA	**48**	**48**
2006	NA	NA	**454**	**473**	NA	NA	**124**	**126**	NA	NA	**46**	**47**
2007	NA	NA	**497**	**518**	NA	NA	**123**	**125**	NA	NA	**44**	**45**
2008	NA	NA	**543**	**565**	NA	NA	**121**	**122**	NA	NA	**44**	**44**
2009	NA	NA	**514**	**536**	NA	NA	**116**	**118**	NA	NA	**43**	**43**
2010	NA	NA	**524**	**545**	NA	NA	**113**	**114**	NA	NA	**40**	**40**
2011	NA	NA	**458**	**479**	NA	NA	**108**	**109**	NA	NA	**39**	**39**
2012	NA	NA	**397**	**414**	NA	NA	**102**	**103**	NA	NA	**38**	**38**
2013	NA	NA	**386**	**402**	NA	NA	**95**	**96**	NA	NA	**36**	**37**
2014	NA	NA	**436**	**454**	NA	NA	**98**	**99**	NA	NA	**37**	**37**
2015	NA	NA	**414**	**433**	NA	NA	**94**	**95**	NA	NA	**35**	**36**
2016	NA	NA	**372**	**389**	NA	NA	**84**	**85**	NA	NA	**32**	**32**
2017	NA	NA	**334**	**348**	NA	NA	**80**	**81**	NA	NA	**32**	**32**
2018	NA	NA	**389**	**405**	NA	NA	**83**	**84**	NA	NA	**32**	**33**

Bold values indicate exceedances

304367

304368

TABLE 5-85: WILDLIFE SURVEY RESULTS Mammals (UNCHANGED)
Hudson River
New York

Information Source	Date	Contact	Response	Contact Information	Data Available	Information/Findings
Mammals						
Hudsonia	2-Jun-99	Call/Fax	YES; spoke with on 6/2/999	Eric Kiviat, Executive Director; (914) 758-7273 (7274) OR (914) 758-7053; FAX: (914) 758-7033; EMAIL: kiviat@bard.edu; inside.bard.edu/specialprog/arch/hudsonia.html	He has no direct knowledge of the upper Hudson but provided names	RIVER OTTER: very rare; he has only seen one on the Hudson in 30 years RACCOON: Fur bearer unit - NYSDEC; trapper prices currently very low so may not have information LITTLE BROWN BAT: Endangered species Unit - Allen Hicks (Delmar NYSDEC Endangered Species) NYSDEC - Natural Resource Damage Assessment
NYS Department of Environmental Conservation - Endangered Species Unit	3-Jun-99	Call	Yes	Al Hicks (Mammal Biologist) (518) 478-3056; www.dec.state.ny.us	Left Message - Will call back	
The New York River Otter Project	2-Jun-99	Email	No	Dennis Money, Dennis_Money@rge.com; www.nyotter.org	Left Message - Will call back	
Professional Trapper	4-Jun-99	Call	No	Jim Comstock	Left Message - Will call back	
New York State Trappers Association	4-Jun-99	Email	Yes	Jerry Leggieir (montcalm@earthlink.net)	Asked me to give him a call at night; also suggested that I call Everett Nack (518) 851-2901 - a commercial fisherman on the river	
Professional Fisherman on the Hudson				Everett Nack (518) 851-2901	Recommended by Jerry Leggieir	
NYSDEC	16-Jun-99	Call	Yes	Mark Brown (518) 623-3671	Familiar with the area regarding mammals, birds, and herps. Good source. See General Info page.	Otter, Mink, Musk Rat present. PCB contamination reduced their numbers severly but in the past 10 years, they have rebounded after clean-up work. Has also seen raccoon, short and long tail weasels, big and little brown bat, skunk, oppossum. The red fox, grey fox, and coyote especially common in the northern Hudson, and plenty of white tail deer suggesting no bears.
Ndakinna Wilderness Project	6/3/1999 6/16/99	Email Call Call	No No Yes	Jim Brushek (518) 583-9980x3, 23 Middle Grove Road, Greenfield Center, NY 12833; Received address from Saratoga County Information - Annamaria Dalton (annamaria@spa.net)	Professional Tracker	Quite a few otter. Mink numbers are large and increasing. Tons of raccoons ("road-kill count is staggering"). Some musk rat. Lots of beavers. Very recent reports of moose in the center of Saratoga, about 5 miles from Hudson. He expects moose to inhabit the Hudson very soon but he thinks they are already there. Sees fisher cats cruising the water occassionally. Frequently sees red fox, grey fox , and deer visiting the water. The coyote population is very large. Coyotes and foxes will feed on the smaller aquatic mammals. Sees the occassional black bear.

Comments

Federal

U.S. DEPARTMENT OF COMMERCE
National Oceanic and Atmospheric Administration
National Ocean Service
Office of Response and Restoration
Coastal Protection and Restoration Division290 Broadway, Rm 1831
New York, New York 10007

EF-1

September 7, 1999

Alison Hess
U.S. EPA
Sediment Projects/Caribbean Team
290 Broadway, 19th Floor
New York, NY 10007

Dear Alison:

Thank you for the opportunity to review the August 1999 Phase 2 Report - Review Copy, Further Site Characterization and Analysis, Volume 2E - Baseline Ecological Risk Assessment, Hudson River PCBs Reassessment RI/FS. The following comments are submitted by the National Oceanic and Atmospheric Administration (NOAA).

Background

The primary objectives of the baseline ecological risk assessment (ERA) are to quantify risks to selected biological receptors and communities exposed to releases of PCBs in the Hudson River. The Hudson River is divided into the Upper, freshwater non-tidal portion of the river between Federal Dam and Hudson Falls; and the Lower, tidal estuarine portion of the river between Federal Dam and the Battery. The Upper Hudson is further subdivided into three sections: RM 189- Thompson Island Pool (TIP), RM 168 - Stillwater, and RM 154 -Waterford.

Current risk evaluations are derived primarily from Phase 2 ecological and other investigations, NYSDEC/NOAA 1993 and 1995 PCB congener-specific data fish data, NYSDEC fish monitoring data, NYSDOH water column invertebrate studies, USFWS tree swallow data, and numerous GE studies.

Summary

The Hudson River Superfund Site encompasses the 200 miles of the Hudson from the Verrazano to Hudson Falls, encompassing freshwater, brackish and estuarine habitats. The ERA focuses on three distinct sections of the Upper Hudson River: the Thompson Island Pool (TIP), Stillwater and Federal Dam and on the Lower Hudson as a whole. PCBs were examined as total PCBs (expressed as tri+ PCBs) and toxic equivalents (TEQs).

Eight species of fish comprised of foragers, omnivores, semi- piscivores and piscivores were evaluated. Measured PCB tissue contaminant levels were utilized for all species except the federally endangered shortnose sturgeon for which body burdens were modeled. Five species each of birds and four species of mammals were evaluated to represent various trophic positions.

Macroinvertebrates were sampled for PCB-congener body burdens and for benthic community ecological metrics. Structure was measured by abundance and diversity. Survival, growth and reproduction was assessed using sediment and surface water guidance or criteria and PCB tissue residues.

Toxic reference values (TRVs) based on body burden or dietary dose were selected for fish, birds and mammals. For aquatic invertebrates, the selected No Observed Adverse Effect Levels (NOAELs) ranged between 5.4 and 127 mg/kg wet wt PCBs and the Lowest Observed Adverse Effect Levels (LOAELs) ranged between 27 and 1570 mg/kg wet wt. No TRVs for benthic invertebrates were established due to limited data available in the literature.

Selected fish NOAELs ranged from 0.5 to 15 mg/kg wet weight PCBs. Fish LOAELs ranged from 17 to 170 mg/kg wet weight PCBs. Fish egg NOAELs, which were reported as lipid-normalized TEQ concentrations, ranged from 0.29 to 8 ug/kg lipid and LOAELs ranged from 0.6 to 103 ug/kg lipid. The lowest whole body TRVs were calculated for the pumpkinseed and largemouth bass. Based on TEQ in egg, the lowest TRVs were calculated for pumpkinseed, yellow perch, white perch, largemouth bass, striped bass and shortnose sturgeon. Spottail shiner had the highest TRVs (egg or whole body).

TRVs were developed for each bird species based on dietary dose and egg concentration for total PCBs and TEQs from available field and laboratory studies. In general, TRVs from laboratory studies were lower than those derived from field studies. Based on lab studies, NOAELs for total dietary PCBs ranged from 0.01 to 0.26 mg/kg/day and the LOAELs ranged from 0.07 to 2.6 mg/kg/day. The NOAEL and LOAEL for total PCBs in eggs was 0.33 and 2.21 mg PCBs/kg egg, respectively. TRVs derived from dietary TEQs were lower than those from egg TEQs. The highest TRVs were associated with the mallard for lab diet studies, the blue heron for TEQs in eggs (lab study) and the bald eagle for total PCBs in eggs (field data).

Mink and otter were more sensitive to dietary intake (lab studies) of PCBs (NOAEL=0.01 mg PCBs/kg/day; LOAEL 0.07 mg PCBs/kg/day) than raccoon or little brown bat (NOAEL=0.032 mg/kg/day PCBs; LOAEL 0.15 mg/kg/day PCBs). The NOAEL and LOAEL across species based on laboratory dietary doses of TEQ was the same (NOAEL= 0.0001 ug TEQ/kg/day; LOAEL=0.001 ug TEQ/kg/day). The total PCB and TEQ NOAELs for mink and otter developed from field studies was up to an order of magnitude lower than those based on laboratory toxicity studies.

Benthic invertebrates showed a trend of decreasing diversity, species/taxa richness, and abundance associated with increasing PCB concentrations.

Congener patterns in fish and benthic invertebrates were examined to identify Aroclor patterns; determine relative importance of water, sediment and food exposure pathways; evaluate importance of upstream vs downstream sources of PCBs; evaluate recent or current vs historical releases of PCBs, and evaluate exposure through marker compounds and congener ratios.

A weight of evidence approach was followed to assess risks of adverse effects to receptors of concern exposed to Hudson River PCBs. Assessment endpoints were evaluated against the various lines of evidence available. The following conclusions were drawn about exposure to Hudson River PCBs:

- risks are greatest in the Upper Hudson River (especially the TIP) and decrease downstream with concomitant decreasing PCB concentrations,

- many species are expected to be at risk at least through 2018, the upper bound of the forecasting exercise,

- current and future exposures to PCBs may adversely effect the benthic community serving as a food source to local fish and wildlife,

- current and future concentrations of PCBs are of sufficient magnitude to adversely affect particular habitats in the Hudson River ability to support healthy and sustainable aquatic biota populations,

- current and future sediment and water concentrations generally exceed existing guidelines and criteria for the protection of aquatic health,
- current and future exposures to PCBs may reduce or impair survival, growth, and reproductive capability of resident fish in the Upper Hudson River,
- current exposures to PCBs may reduce or impair survival, growth, and reproductive capability of upper trophic level fish in the Lower Hudson River,
- current and future exposures to PCBs may reduce or impair survival, growth, and reproductive capability of waterfowl, insectivorous and piscivorous birds in the Upper Hudson River, and may have a similar effects in the Lower Hudson River to varying degrees,
- current and future exposures to PCBs may reduce or impair survival, growth, and reproductive capability of insectivorous, omnivorous and piscivorous birds in the Upper Hudson River, and may have a similar effects in the Lower Hudson River to varying degrees, and
- current and future concentrations of PCBs are of sufficient magnitude to adversely affect the reproductive capability of threatened and endangered species.

Comments

The Baseline Ecological Risk Assessment (ERA) closely follows EPA ERA guidance. The report describes the problem formulation including assessment and measurement endpoints, the exposure assessment including modeled exposure concentrations and exposure pathways, the effects assessment including development and selection of TRVs, risk characterization including an evaluation of the assessment endpoints and an uncertainty analysis. Overall, the document is well-organized and clearly written.

The fate and transport and bioaccumulation modeling presented in the Baseline Modeling Report provides the primary exposure information for the ERA. While the ERA describes the quantification of PCB fate and transport and discusses the modeled exposure concentrations (sediment, water, benthos, fish), there is no substantial discussion of the limitations of the models. Moreover, future work is planned on the fate and transport and bioaccumulation models that could significantly affect the predictions in the ERA. How will the results of these supplemental analyses be incorporated into the models and the prediction of risk? **EF-1.1**

There are a number of aspects of the Hudson River system that the fate and transport and bioaccumulation models are not addressing. For example, sediment resuspension may result from debris in the river (including large rocks, trees and root masses), bank erosion, or ice scour during high flow events. The daily changes in water level associated with hydropower generation act as a regular tidal action in shallow water, nearshore sediments, which may increase the release of PCBs from these sediments. Temperature in the shallow nearshore areas, during the summer low flow period may be higher than the mid-channel, which would affect temperature-dependent partitioning. All of these factors may result in significant underestimation of resuspension of sediments and/or PCB loading to the river. This represents major uncertainty in the exposure assessment for the risk assessment, since the future sediment, water and fish tissue PCB concentrations forecasted by these models are used to predict future risk. The implications of the uncertainty resulting from the model inputs to risk assessment should be addressed. **EF-1.2**

The food chain modeling used a generic growth rate for lake trout as an input parameter for all 8 species of fish modeled rather than attempting to capture the difference in their growth. Sensitivity analysis was not conducted on growth rates, but GE issued corrections to QEA model predictions **EF-1.3** due to initial errors in growth input, which indicates the importance of this parameter in forecasting fish concentrations.

Water column and sediment data used in the exposure assessment are averaged without regard for habitat occupied by the receptors of concern. As NOAA and others have previously pointed out, nearshore areas represent important habitat for most of the food web pathways evaluated in the ERA, yet water column and sediment data apparently were averaged across area groupings, in spite of the much-discussed differences between along-shore and mid-channel PCB concentrations. **EF-1.4**

Selection of the toxicity reference values (TRV) for fish total PCB body burden relied to a great extent on a single study (Bengtsson 1980) that used a commercial mixture (Clophen A50) which was not available in the United States and is different from mixtures used in the Hudson River. In addition, Bengtsson reported lower NOAEL and LOAEL values (based on for a more sensitive reproductive endpoint) than used as TRVs in the ERA. Other studies on closely related species, which also reported lower effect-values, were excluded with no rationale provided (e.g., Hansen et al 1974; USACE 1988). Therefore, the TRVs developed for Hudson River total PCB body burdens in fish may underestimate risk. **EF-1.5**

The risk assessment did not provide clear criteria for selection of laboratory studies that are used to define TRVs for fish species other than giving preference to studies on closely-related species. Because of the importance of the TRV in the determination of risk and all of the uncertainty associated with the selection of appropriate TRVs, relying on one or two laboratory studies to determine the TRVs for effects in fish should be evaluated in the context of other studies, particularly considering the limited number of studies available. **EF-1.6**

The ERA acknowledges that floodplains serve as an important habitat area, and thus could serve as a significant source of PCBs for animals utilizing the floodplain for food and habitat (p. 14) but risk associated with PCB-contaminated floodplain soils is not considered within the ERA. PCB-contaminated floodplain soils can also act as a significant source of PCBs to the Hudson River, thereby contributing to the recontamination of remediated areas. Hence, their relative importance needs to be considered within the RI/FS. The report also states that "plants and animals utilizing the Hudson River shoreline are likely exposed to levels lower than aquatic-based exposures" but the "degree and spatial extent of PCB contamination in floodplain soils has not been investigated". Risks to receptors utilizing the floodplain should be characterized even though they may be exposed to lower levels than receptors utilizing the river. According to a risk assessment conducted by EPA on the Sheboygan River (Chapman 1999), robins nesting within 100 ft of floodplain soil averaging 4 ppm PCBs are "at risk of reproductive impairment." **EF-1.7**

Specific Comments

Executive Summary

Page ES-11: Risk to benthic invertebrates should have been included in the major findings. **EF-1.8**

Chapter 2

Page 22-27: Median and 95th percentile total PCBs and TEQ PCBs apparently were examined for fish while mean and 95th percentile total PCBs and TEQ PCBs were evaluated for birds and mammals. Is this correct? **EF-1.9**

Pages 22, 24-28: Additional measurement endpoints should have included a comparison of measured and modeled fish TEQ concentrations reported by EPA (1993) to pose a risk to fish, avian and mammalian receptors. For example, low risk to piscivorous fish was associated with fish concentration of 50 pg/g TCDD and high risk at 80 pg/g TCDD. Fish TCDD concentrations of 6 pg/g and 60 pg/g were identified as posing a low to high risk to avian wildlife respectively; where high risk is defined as causing 50-100% mortality in embryos and young of sensitive **EF-1.10** species. For mammalian wildlife, fish TCDD concentrations of 0.7 pg/g pose a low risk and 7 pg/g pose a high risk.

Page 35, Para 3: The peregrine falcon was recently removed from the endangered species list. **EF-1.11**

Chapter 3

Page 40, Section 3.1.2, Item 2: It is not appropriate to average "TEF-based concentrations" or "TEF-based factors" over the entire lower river or the entire river without first demonstrating that this does not bias the results. **EF-1.12**

Page 41-42, Section 3.2.1: In the TIP, were data weighted equally for nearshore and mid-channel locations? Would it be appropriate to use a "correction factor" for water column samples that were not collected in the nearshore habitat, where the fish and sediment samples were collected? **EF-1.13**

Page 42, Section 3.2.2: Why were OC-normalized sediment concentrations not included? **EF-1.14**

It should be pointed out that the statistics for the sediment data (and for the benthic data) from TIP are different from other locations, due to the fact that sample data from 5 separate station locations within the TIP were apparently averaged, while most other stations represent a single station with 5 replicates collected from a very limited areas. Thus, the mean and UCL for TIP reflect (to a limited extent) the variability within the TIP, while for the other locations only represent the sample variability. **EF-1.15**

Page 42, Section 3.2.3: Are all the sampling locations within the mainstem of the river or are some (i.e. NERRs stations) separated from the river by the embayment created when the railroad was constructed. If some stations are within these embayments, then there should be a discussion of the different flow dynamics and deposition rates compared to PCB exposure in the mainstem. **EF-1.16**

Page 43, Section 3.2.4: The fish data from EPA/NOAA Phase 2 used a different analytical method than the NYSDEC analysis: are total PCBs (and percent lipid) determined by these two approaches directly comparable? An empirical analysis conducted by NOAA (1997) indicates that there may be systematic differences. **EF-1.17**

Toxic equivalency concentrations were estimated for 15 pumpkinseed samples collected from 5 locations in 1993 (NYSDEC/NOAA collection) using a fish-cell line bioassay (Tillitt 1997). TEQs in the pumpkinseed ranged from 1 to 115 pg TCDD-eq/g (wet wt.), with the highest concentrations from fish samples collected in TIP. These estimated TEQs exceed the NOAEL and LOAEL selected for pumpkinseed on page 83. **EF-1.18**

Page 44, Section 3.3.1: The last sentence states that the constant upstream boundary assumption is considered protective of aquatic health. According to the BMR (Book 1, page 86), the assumed constant upstream boundary condition is 9.9 ng/l, which exceeds the NYSDEC Wildlife Criterion and EPA Great Lakes Initiative water quality criteria. **EF-1.19**

The upstream boundary conditions used in the BMR assumes that there will no flow-related change (increase) in loading during high flow events. The BMR model does not address potential impact of high flow events on the Interim Cap on the Remnant Deposits or other areas of high concentrations of PCBs that may remain between the plant sites and Rogers Island. Data from the January 1999 high flow event suggest that this is not true and that setting the upstream boundary at 9.9 ng/l could underestimate the loading. This uncertainty should be addressed.

Page 47, Section 3.4.2:. The description of fish exposure pathways is minimal at best. There should be some discussion of the importance of nearshore habitats for key fish species and how that will be addressed in the exposure assessment. In addition, although seasonal differences in accumulation are discussed later, the importance of processes affecting seasonal changes in exposure need to be addressed. **EF-1.20**

Explain why NYSDEC data was used for all fish except forage fish while USEPA/NOAA Phase 2 data was limited to forage fish only. EF-1..

Page 52 Para 1: Fish were divided into small (< 10 cm) and larger (> 10 cm), where small fish were identified as minnows and sunfish and larger fish as catfish and bass. This ignores different feeding strategies for different age classes. EF-1.22

Page 53 Para 2: Estimated phytoplankton PCB concentrations (calculated from lipid, Kow and dissolved water PCBs) are used as a surrogate for the vegetative component of the mallard diet. This calculation excludes the importance of sediments in the bioaccumulation of PCBs in plants. Research conducted by Lovett-Doust et al. (1994) on American wild celery uptake of organochlorines demonstrated that "simple bioconcentration is an inadequate description of contaminant dynamics between plants, sediment, and water." EF-1.23

Page 54: The winter diet of bald eagles seems to consist mainly of fish. Were the concentrations modeled for dietary exposure based on summer and fall data or only the fall data? EF-1.24

Page 62: The approach presented for segregating fish data into two size classes fails to account for an alternative feeding strategy by juveniles which as adults occupy a higher trophic level. The model should be adjusted if data is available; otherwise, the uncertainty associated with this should be discussed. EF-1.25

Page 55-58, 64-65: It would be helpful in Summary of ADD_{exp} and $ADD_{95\% UCL}$ sections to state specifically the space domains of the dose/exposure data. For example, is it correct to assume that the doses to mammals are intended to represent all animals within the river reach centered on the river mile for which the data are presented and extending half way to the next reported concentration? EF-1.26

Pages 69, Para 4 and 71, Para 2: Narrative and figures do not match. EF-1.27

Page 69, Para 3: "...typical largemouth bass sample..." It should be noted that these typical largemouth bass collected in 1993 study were juveniles and appropriately classified as forage fish (i.e., less than 10 cm). EF-1.28

Page 70, Last Para: Discussion of feeding guild analysis from Appendix K, which identifies some fish as piscivores. No piscivores were collected in the 1993 samples. Adult white perch and yellow perch could be classified as semi-piscivores. EF-1.29

Page 70, Section 3.5.3: "Fish body burdens were shown to decline with river mile to about the same degree as changes in the sediment PCB concentration." This statement is incorrect—fish concentrations decline more rapidly in fish, especially in the lower river. EF-1.30

Page 70, Section 3.5.3: The change in water column congener composition and concentration with increasing distance from the GE plants may also be due to loss of the lower chlorinated congeners. EF-1.31

Page 70, Section 3.5.3: "Lastly, metropolitan New York discharges present higher molecular weight mixtures for fish exposure in the saline portion of the Lower Hudson River." This assertion is made without any supporting justification. Are there data showing upriver transport of suspended particulates for 25 - 50 miles in the river? It is also misleading, since the change in composition reflects a decrease in concentration. A more likely explanation for the shift in composition is the increased importance of the local sediment as the water column load is diluted by tributary input. The lipid-normalized concentrations of individual key tri- (BZ# 28), tetra- (BZ#52 and BZ#66), penta- (BZ#101 and BZ#110), and hexa-chlorobiphenyl congeners (BZ#138) in white perch collected from eight locations between the Federal Dam and Piermont (rivermile 24) all show a consistent decline (Field and Sloan 1996). EF-1.32

Page 71: " Benthic invertebrates in the saline Lower Hudson distinctly show the impact of the New York City metropolitan area inputs." The furthest downstream benthic station was located at EF-1...

304376

Piermont, about 25 miles upriver from NYC. It is highly unlikely that NYC inputs are the source of the PCB signal at Piermont. Likewise, PCB concentrations were relatively low in this sample.

Chapter 4

Page 76: The NOAA (1999) report on the effects of PCBs on fish reproduction and development demonstrates that Hudson River fish contain PCBs at concentrations above levels shown to cause reproductive and developmental effects and that these effect levels are, in some cases, below the TRVs presented later in this Chapter. **EF-1.34**

Page 76: While the ERA attempts to address effects associated with congeners eliciting dioxin-like behavior, it does not attempt to examine effects from congeners that have different mechanisms of action. This potentially further underestimates risk to receptors associated with releases from the GE facilities and should be addressed within the uncertainty section. **EF-1.35**

Page 77: The ERA focuses on the effects of PCBs on lethality, growth and reproduction, although PCBs are known to cause other severe adverse effects (e.g., immunosuppression). **EF-1.36**

Page 81, section 4.2.2.1 and Table 4-3: The sediment guidelines section provides PCB guidelines and standards. It should also include sediment guidelines developed by EPA (1993) for protection of fish, birds and mammals based on TCDD sediment concentrations since PCBs are also evaluated as TEQs. It should also be noted that measured and modeled PCB concentrations in sediments were also compared to NYSDEC and Washington Department of Ecology guidelines. **EF-1.37**

Page 81-82, Section 4.2.3: The authors assume that PCBs partition equally into the lipid phase of eggs and into the lipid phase of adult fish "tissue". There is good justification for this assumption, but it does not necessarily follow that it is appropriate to establish TRVs based on lipid-normalized concentrations. Is there evidence to indicate that lipid-normalized concentrations are more directly related to the reproductive effects than the wet weight concentrations in eggs? **EF-1.38**

Page 81-93, Section 4.2.3: Toxicity reference value (TRV) development includes separate analyses of field and laboratory studies. The selection of a total PCB body burden NOAEL and LOAEL for the eight receptor fish species, primarily relied on one study by Bengtsson (1980) on the minnow, *Phoxinus phoxinus*, exposed to Clophen A50, a PCB product not available in the United States. Bengtsson not only observed a decrease in ova hatchability but also a delay in time to spawning and an associated higher temperature required to trigger spawning. These effects on time of spawning were documented at lower concentrations than those affecting hatchability of minnow ova. No effects on initiation of spawning were observed at 1.6 ppm PCBs (compared to an NOAEL of 15 ppm based on hatching success) but spawning was delayed one week at 15 ppm and three weeks at 170 ppm when temperatures were around 16°C. In addition, other studies on closely related species were not considered. For example, a study by Hansen et al. (1974), showed a LOAEL of 5.1 mg/kg for sheepshead minnow (family Cypriniformes) based on decreased fry survival after hatch. Another study, by the USACE (1988) examined effects of PCBs on fathead minnows where the NOAEL and LOAEL for reduced fecundity and frequency of reproduction were 5.6 and 13.7 mg/kg, respectively. These results suggest that some of the TRVs developed for Hudson River total PCB body burdens in fish may underestimate risk. **EF-1.39**
Page 83: Pumpkinseed TRVs (TEQs in eggs) were selected from salmonid and non-salmonid lab studies. The derivation of TRVs based on non-salmonids are described but not presented. The report should have stated that the NOAEL and LOAEL were 0.54 and 10.3 ug TEQs/kg lipid, respectively.

Pages 83, 84, 87, 89, 90, 92 and 93: According to Table 4-7, the non-salmonid concentrations were based on the concentration in an embryo, not an egg, as stated in the second to last paragraph. **EF-1.40**

Page 88 Middle: An NOAEL from field data was developed for white perch but not for yellow perch since "no field studies were identified that examined effects of PCBs on yellow perch or on a fish in the same family as the yellow perch or on a species in the same family as the yellow perch." It is not clear why the values for white perch or other species could not be used with the application of an uncertainty factor, as done for laboratory studies. **EF-1.41**

Page 88 Bottom: There is a typographical error for the white perch TRVs from lab studies on TEQs in salmonid eggs. The NOAEL and LOAEL are transposed. The NOAEL should read 0.29 ug TEQs/kg lipid, the LOAEL 0.6 ug TEQs/kg lipid. **EF-1.42**

Chapter 5

Page 123 Top: The NYSDEC surface water standard for protection of wildlife is 1.2×10^{-4} ug/l. The report incorrectly identifies the units as g/l. **EF-1.43**

Page 123, Section 5.2.1.1, First Para.: "In the Lower Hudson R., the shown values are for spottail shiners (the only fish less than 10 cm in length)..." Eighteen of the 39 fish samples less than 10 cm in length collected from the lower river represented other species, including striped bass and largemouth bass juveniles, and brook and Atlantic silversides. **EF-1.44**

Page 123-124: A revised discussion of modeled and measured spottail shiner (and other fish) should account for the other reproductive effects observed by Bengtsson (1980) and USACE (1988) at concentrations lower than the selected TRVs. **EF-1.45**

Page 123-127: Fish concentrations were predicted for years 1993-2018. This includes years where there is overlap with measured concentrations. How do the measured and modeled values compare? **EF-1.46**

Page 124: Table 4-16 should read Table 4-25. **EF-1.47**

Page 125 Para 1: Append " except brown bullhead at RM 168 and 189.to last sentence beginning "On a TEQ basis, ...". **EF-1.48**

Page 125, Section 5.2.1.6: Table 5-25 does not distinguish between lab and field derived TRVs. Also a field TRV was developed for yellow perch but not white perch as stated in first paragraph of this section. **EF-1.49**

Page 126: Replace "1993-1999" with "1997". **EF-1.50**

Page 128-129: This section provides evidence from field observations on populations of receptor fish species. An important factor omitted from discussion (except for sturgeon) is the potential affect recreational and commercial fishing bans and fish consumption advisories have had on populations in the Lower and Upper Hudson. For example, the ban on fishing in the Upper Hudson was only recently lifted and a catch and release policy instated. Commercial takes of striped bass have been prohibited since 1976 and commercial restrictions have varied for other lower river species. Reductions in fishing pressure may mask cumulative adverse impacts to fish populations from multiple anthropogenic activities. **EF-1.51**

Page 149, Section 5.7.4.1: "Several locations" refers to all but 3 stations in the Lower Hudson exceeding the EEC on a 95% UCL basis. **EF-1.52**

Chapter 6

Page 133 and elsewhere: Modeled and measured PCBs concentrations in fish, invertebrates, sediment and water were used to derive site-related doses. For a given receptor (i.e. largemouth bass, mink, heron), was the diet developed from relevant species and size class data? For example, on page 64, otter obtain 100 % of their diet from fish. How were dietary PCBs calculated i.e., from a given size class of fish, limited to certain fish species; employing 25%, **EF-1.53** median or 95% UCL concentrations across fish species? An explanation is warranted for each receptor species.

Page 154: The quantification of the congeners used in the TEQ analysis should have included a discussion of PCB 77 where concentrations were frequently classified as below detection due to blank contamination (although reported concentrations were consistent with the composition of other samples). In addition, the problem of using water column data, where the congeners of primary importance (in weight and toxicologically) were mostly below detection, should be discussed. **EF-1.54**

p. 156, Para 3: "To reduce the uncertainty..., comparative analyses were performed...Results of the analyses were employed so as to enhance study comparability while reducing inherent uncertainty." It is not clear what analyses were performed or how they were used to reduce uncertainty. **EF-1.55**

Page 158: Uncertainty associated with the development of TRVs is discussed here. According to the first bullet, "a factor of 5 for extrapolations between families and a factor of 10..." were used; these are not consistent with the discussion of uncertainty factors on page 80. The major uncertainty here is the limited available information on the differences in fish sensitivity to PCBs and to different PCB mixtures. **EF-1.56**

Page 158, Last Full Para: "...use of different TEF when evaluating TEQ concentrations." Does this mean that non-standardized TEFs were used in calculating TEQs, even when data were available to use the same TEFs? Where is this analysis presented? **EF-1.57**

Page 159, Para 4: "uncertainty associated with fish that fall into the smaller size class as juveniles and the larger size class as adults." Does this mean that species were classified as forage or piscivorous, regardless of size? If not, then what is the uncertainty? **EF-1.58**

Page 159: Fish were categorized into two size classes. The larger size class presented here (> 20 cm) is not consistent with the discussion on page 52 or elsewhere in the main body of the report. **EF-1.59**

Page 161: Under prey ingestion rates, a discussion could have been provided comparing Hudson River gut contents of fish relative to the diets structured into the BMR model. **EF-1.60**

Page 164, Para 2: The report only discusses sensitivity analyses for avian and mammalian receptors. Such an analysis should be conducted for fish toxicity values taking into account exposure parameters (i.e., growth rates, lipid), TRVs (i.e., assuming 1:1 egg:tissue for TEQs, fillet to whole body ratios) and exposure media concentrations (i.e., nearshore vs channel sediment and water column concentrations). **EF-1.61**

Pages 165, top: "model error is probably not a significant source of uncertainty." If this refers to the entire exposure model, justification for this assertion should be provided. **EF-1.62**

Pages 165, Section 6.5.3.1: FISHRAND modeled parameters were averaged temporally, spatially and across species. Monte Carlo analysis was used to analyze uncertainties in model predictions. Benthic PCB data was utilized on a wet weight basis because lipid-normalizing the data did not improve the relationships, yet lipid content in prey items was shown to be an important contributor to model uncertainty. This should be addressed. Moreover, the discussion on FISHRAND ignores the averaging of parameters and the importance of nearshore habitats to receptor species. The basis for the "factor of two" statements should be presented. **EF-1.63**

Chapter 7

Page 167 Second question: Measured and modeled sediment concentrations were also compared to other sediment guidelines (exceedance of NOAA, Washington Department of Ecology and Ontario Ministry of the Environment guidelines) and discussed elsewhere in the report including page 187. These should have been included in the concluding remarks under benthic community structure. **EF-1.64**

Page 168 Forage fish reproductive effects: The analysis assumes that measurements of young-of-year spottail shiner and age 1 pumpkinseed are equivalent to concentrations in mature adults. EF-1.

Page 173, Second question: The discussion on available field-based observations ignores the impact of decreased fishing pressure on local populations. EF-1.66

Thank you for your continual efforts in keeping NOAA apprised of the progress at this site. Please contact me at (212) 637-3259 or Jay Field at 206-526-6404 should you have any questions or would like further assistance.

Sincerely,

Lisa Rosman

Lisa Rosman

NOAA Coastal Resource Coordinator

References

Field, L.J. , R. Sloan, L. Read, C. Severn and R. Dexter 1996. PCBs in Hudson River Fish: Comparisons of Congener Patters Over a Geographic Gradient, Poster Presentation at the 17th Annual SETAC meeting, Washington DC, November 1996.

Hansen, D.J., S.C, Schimmel, and J. Forester. 1974. Aroclor 1254 in eggs of sheepshead minnows: effect on fertilization success and survival of embryos and fry. Proc. Southeastern Assoc. Game Fish. Comm. pp. 805-812.
NOAA 1999. Reproductive, Developmental and Immunotoxic Effects of PCBs in Fish: a Summary of Laboratory and Field Studies. Prepared by Emily Monosson for National Oceanic Atmospheric Administration, March 1999.

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

Tillett, D.E. 1997. Determination of 2,3,7,8- tetrachloro dibenzo-p-dioxin equivalents (TCDD-eq) with the PLHC-1 bioassay: pumpkinseed collected from the Hudson River, Nov 11, 1997. Final Lab Report Number FY-98-30-01, Report to USFWS and NOAA, 2 pp.

USACE 1998. Relationship between PCB tissue residues and reproductive success of fathead minnows. Environmental effects of dredging. Technical notes. EEDP-01-13. U.S. Army Corps of Engineers, Engineer Waterways Experiment Station, Environmental Laboratory. Vicksburg, MS

USACE 1994. Biological Investigation for the Remedial Investigation/Feasibility Study, Sangamo Weston, Inc./Twelve Mile Creek/ Lake Hartwell PCB Contamination Superfund Site Operable Unit Two. U.S. Army Corps of Engineers, Savannah District, Savannah, Georgia.

cc: Mindy Pensak, DESA/HWSB
Gina Ferreira, ERRD/SPB
Robert Hargrove, DEPP/SPMM
Charles Merckel, USFWS
Anne Secord, USFWS
William Ports, NYSDEC
Ron Sloan, NYSDEC
Anton P. Giedt, NOAA

State

N~w York State Department of Environmental Conservation
L_,sion of Environmental Remediation
Bureau of Central Remedial Action, Room 228
_50 Wolf Road, Albany, New York 12233-7010
Phone: (518) 457-1741 • **FAX:** (518) 457-7925
Website: www.dec.state.ny.us

John P. Cahill
Commissioner

ES-1

September 7, 1999

Allison A. Hess
Project Manager
U.S. Environmental Protection Agency
Region 2
290 Broadway, 19th Floor
New York, New York 10007-1866

Dear Ms. Hess:

RE: Hudson River PCB Reassessment RI/FS
Site No. 5-46-031

The New York State Department of Environmental Conservation has completed its review of the Phase 2 Report - Further Site Characterization and Analysis, Volume 2E - Baseline Ecological Risk Assessment (BERA), Hudson River PCBs Reassessment RI/FS, dated August 1999. Our comments on the BERA are provided below.

Page 13 Section 2.3.2 The first paragraph contains an inconsistency with other portions of the report. In this paragraph, the Upper Hudson River is described as being between river mile 153 and river mile 195. As you know, river mile 195 is within the Remnant Area and located two miles downstream of Fenimore Bridge. This description conflicts with the text found on page 1 in Chapter 1: "For purposes of the Reassessment, the area of the Upper Hudson is defined as the river bed between the Fenimore Bridge in Hudson Falls (just south of Glens Falls) and the Federal Dam at Troy." The latter description is more accurate for consideration in the ecological risk assessment boundaries. **ES-1.1**

In general, we agree with EPA's conclusion that receptors in close contact with the Hudson River are at an increased ecological risk as a result of exposure to PCBs in sediments, water, and/or prey.

If you have any questions regarding the comments please contact this office at 518-457-5637.

Sincerely,

William T. Ports

William T. Ports P.E.
Project Manager
Remedial Section A
Bureau of Central Remedial Action
Division of Environmental Remediation

Local

SARATOGA COUNTY
ENVIRONMENTAL MANAGEMENT COUNCIL
PETER BALET
CHAIRMAN
GEORGE HODGSON
DIRECTOR

EL-1

September 2, 1999

Alison A. Hess, C.P.G.
USEPA Region 2
290 Broadway, 19th Floor
New York, New York 10007-1866

Attn: Hudson River HHRA
ERA Comments

Dear Ms. Hess:

Enclosed you will find the Saratoga County Environmental Management Council's (SCEMC's) comments prepared by member David Adams on the Hudson River PCB's Reassessment Phase 2 Human Health Risk and Ecological Risk Assessment Reports.

The Council, although sensitive to the need to provide conservative estimates when assessing human health and ecological risks related to the Hudson River PCB Reassessment, finds both the HHR and ER Assessments to reflect an unrealistic degree of "scientific" over-conservatism often based upon inaccuracies and what we believe to be fallacious scientific assumptions.

Sincerely,

George Hodgson

George Hodgson, Jr.
Director

Enc.
cc: Doug Tomchuk, USEPA, Region 2
SCEMC Members
Darryl Decker, Chr., Government Liaison Committee, CIP
The Honorable John Sweeney

50 WEST HIGH STREET BALLSTON SPA, N.Y. 12020 (518) 884-4778

SARATOGA COUNTY
ENVIRONMENTAL MANAGEMENT COUNCIL
PETER BALET
CHAIRMAN
GEORGE HODGSON
DIRECTOR

COMMENTS ON BASELINE ECOLOGICAL RISK ASSESSMENT REPORT
VOLUME 2E; AUGUST 1999
HUDSON RIVER PCBs REASSESSMENT RI/FS

Prepared by: David D. Adams, Member, Saratoga County EMC and Government Liaison Committee
August 30, 1999

1. **General:** This report is characterized by a general lack of specific information about the species of concern in the Upper Hudson River area. The true measure of whether the potential effects predicted by TQs is greater than one are actually occurring should be what is happening to the species of interest. This would best be evaluated by having information on what the populations were before PCBs, determining what population would be expected today without any adverse effects from PCBs, and finally comparing this expected population without any effects from PCBs to today's population. P. 29 points out that people have higher confidence in actual data than projections or characterizations that require assumptions. The usefulness of long-term trend data is also pointed out on P. 29. While not very long-term, it is unfortunate that EPA did not see fit at the beginning of this reassessment to study the species populations of concern in the Upper Hudson. If this had been done, we would now be approaching 9 years worth of data instead of having to rely on conjecture about what might happen. This failure to collect data is another example of the poor planning which went into EPA's data acquisition program. As will be commented on later, the calculated TQs are in several cases so high that effects on species would be expected to be apparent (otherwise it is hard to see any merit to the TQs), yet visible effects are not seen in the limited information available on species populations in the Upper Hudson River. This discrepancy between the high calculated TQs and the lack of observed effects on species makes this whole report of questionable value – a report based on conjecture and speculation, raising alarms about things which have not been observed. EL-1.1

2. **Executive Summary, PP. ES-3 and ES-4, P. ES-3:** This lists protection of significant habitats as an assessment endpoint and P. ES-4 lists striped bass and shortnose sturgeon as receptors of concern. Table 2-11, which lists areas of significant habitat, does not appear to include any areas in the Upper Hudson River. Also, striped bass and shortnose sturgeon are not species present in the Upper Hudson River. If the scope of this report is the Upper Hudson River as it has been understood is the subject of EPA's reassessment, then these items should be deleted from this report. EPA is requested to clarify this matter. EL-1.2

3. **Executive Summary, P. ES-3:** The "measured endpoints" include modeled PCB body burdens. Are the model results in this report based on the models as presented in EPA's published Baseline Model Report or are they based on the model revisions EPA has previously indicated to be required? If the former, this report should be withdrawn until the results are updated. If the latter, EPA should promptly make the revised models available for review. In either case, it is not proper that EPA has requested a review of a report for which all the backup information is not available to the public. EL-1.3

50 WEST HIGH STREET BALLSTON SPA, N.Y. 12020 (518) 884-4778

4. **Executive Summary, P. ES-7:** What is the reason why no uncertainty factors were needed for dioxin-like PCBs? EL-1.4

5. **Executive Summary, P. ES-7:** EPA should mention that there have been serious questions raised about the tree swallow study conducted in the Upper Hudson River. EL-1.5

6. **Executive Summary, PP. ES-10 & 11:** The bald eagle is no longer a threatened or endangered species and so should be removed from this discussion as should the sturgeon as it is not in the Upper Hudson River. EL-1.6

7. **Section 2.4, PP. 21, 22,23,25,26,27, & 28:** To properly evaluate Measurement Endpoint 1 for the benthic community requires data on which to base expected values of benthic community abundance and composition in the absence of PCBs. Do these data exist and if so, has EPA used the data? The same comment applies to Measurement Endpoints 5 for fish, 6 for insectivorous birds, 6 for waterfowl, 6 for piscivorous birds, and 5 for wildlife. The Assessment Endpoints should be deleted for threatened and endangered species and for significant habitats as there are none of interest identified for the Upper Hudson River. EL-1.7

8. **Section 3.1.1, P. 38:** Please explain what is meant by "target" and "non-target" congeners. There should be discussion of the fact that Gradient Corporation did data usability for any of the congeners important to the TEQ in fish and other species except for BZ #118 which is a small contributor (11%) to the total TEQ. How can the validity of the congener data used be judged without such a discussion? EL-1.8

9. **Section 3.1.1, P. 39:** Does the term "qualified" data mean "questionable" data? Also, why was it considered acceptable to use the 62% of the ecological data qualified as estimated? EL-1.9

10. **Section 3.1.1, PP. 39 & 40:** The use of the detection level for BZ #126 when it was quantitated at the detection level is troublesome since BZ #126 has one of the highest TEFs and is calculated to account for 52% on average of the TEQ in the Upper Hudson River. When BZ #126 is measured at the detection level, the actual value could be anywhere between zero and the detection level. Using the detection level for such a significant congener as B2 #126 imparts an unnecessary degree of conservatism to the calculations. EPA should instead use the average of zero and the detection level. The possible error of a factor of 10 (stated on P. 40) produced by using the BZ #126 detection limit is very significant. Also, EPA is requested to explain why the fish-based data in Table J-2 were used rather than the USFWS data in Table J-3 for tree swallows. The data in Table J-3 show a very small contribution from BZ #126 compared to the data in Table J-2. EL-1.10

11. **Section 3.1.2, P. 40:** Why should the congener distribution stay constant? What do the past congener data show? EL-1.11

12. **Section 3.2, P. 40:** What is the purpose and use of the exposure concentrations developed from the 1993 dataset? Isn't the purpose of the report to look at the future and not past history? EL-1.12

13. **Section 3.2.1, P. 41:** Aren't there a lot more water data that could be used than the data mentioned here? EL-1.13

14. **Section 3.2.3, P. 42:** What was the time period for the Benthic concentrations? Is the GE data also being used? EL-1.14

15. **Section 3.2.6, P. 44:** Why aren't data being taken on PCB concentrations in mink and otter today to compare the 1983-86 data? EL-1.15

16. **Section 3.4.3, P. 48:** A mitigation factor of 0.5 instead of 1.0 should be used for the tree swallow (see Table 3-17) since Appendix E gives the residence time in the Upper Hudson River of 122 to 242 days/year or an average of 182 days/year which is a migration factor of 0.5. Using 1.0 introduces unnecessary conservatism into the analysis. EL-1.16

17. **Section 3.4.3.1, P. 49:** Given the presence of many other water bodies in the ranges given in the Appendix E, the assumption of 1.0 for the areal foraging effort factor (FE) is not reasonable without presenting analysis and justification. Just assuming 1.0, is another unjustified conservatism. EL-1.17

18. **Section 3.4.3.3, P. 52:** Considering the range of many of the bird species given in Appendix E, the assumption that all fish and benthic macroinvertibrate prey come from the Hudson River is unjustified and introduces unnecessary conservatism. EPA should do an analysis to determine factors less than 1 for birds which can range away from the river. EL-1.18

19. **Section 3.4.3.3, P. 52:** Where are the percent of large fish and of small fish in diets specified and justified? EL-1.19

20. **Section 3.4.3.3, P. 54:** Why is 78% fish used for the belted kingfisher diet when the average in Table 3-19 is 73%? Also, why is no consideration given to invertebrate and non-river related food as part of the bald eagle diet when Table 3-21 shows up to 18% invertebrate and up to 4.3% non-river food in the eagle diet? EL-1.20

21. **Section 3.4.3.4, P. 54:** Are comments 15, 16, and 17 for previous comments on range and migration factors? The use of 1.0 for a range factor seems especially inappropriate for the ranges given for the bald eagle (3-7km, Table 3-21 and Mallard Duck 540-620 km^2 or range of 40 – 1440 Ha, Table 3-18). EL-1.21

22. **Section 3.4.3.6, P. 56:** The use of a migration factor of 1.0 for the belted kingfisher is not justified based on the statements in Appendix E that the residence in the Upper Hudson River ranges from 245 to 365 days/year giving an average of 304 days/year. The factor .84 based on 304 days/year should be used to avoid unnecessary conservatism. EL-1.22

23. **Section 3.4.4., P. 58:** The exposure parameters given in Table 3-23, 3-24 and 3-25 are in some cases unnecessarily conservative and should be modified. For the raccoon (see Table 4-23), the dietary percentages for food from the Hudson River (fish and aquatic invertebrates) are taken at the high end of the range given in the references cited in Table 4-23. Average values should be used and the non-river percentage increased accordingly. Also, the use of a factor of 1.0 for range and hibernation should be reduced based on the raccoons home range of 48 km^2 and the statements in Appendix F that the raccoon hibernates up to 4 months/year. Referring to Table 3-24 for mink, why is the percent fish in the mink's diet shown as 34% when the percent fish given in the references cited in Table 3-29 are 18.8% and 27.3% (see Appendix F, P. F-9)? EPA should use the average of 23% fish. Also, the habitat factor for mink should be reduced below 1.0 based on home range of 1.9 to 3.4 km^2 given in Table 3-24. For the river other (see Table 3-25), the use of a 100% fish diet is at odds with Appendix F, P. F-12 which gives a 70% fish diet. (Note: While 70% seems the upper limit for fish, the value is confined somewhat by the fact that the diet percentages on P. F-12 add up to more than 100%). EL-1.23

24. **Section 3.4.4.1, P. 59:** The use of a FE of 1.0 is unnecessarily conservative for the raccoon and mink based on their home ranges given in Tables 3-23 and 3-24 and the FE should be reduced accordingly. EL-1.24

25. **Section 3.4.4.3, P. 61:** Again, the assumption that only the Hudson River is the source for fish and macroinvertebrates in the diets of the raccoon, mink is unnecessarily conservative given the ranges of these mammals. Based upon home range information (approximately 50 mi^2) and the nomadic food seeking characteristics of the river otter, it is highly unlikely that this species is solely dependent upon the Hudson River for its year round food sources. Some factor less than 1.0 should be used. EL-1.25

26. **Section 3.4.4.3, P. 62:** Where is information on the percent of small fish and percent of large fish in the mammalian diets given? EL-1.26

27. **Section 3.4.4.3, PP. 63 & 64:** The use year round of a winter diet composition for raccoons and otter is not right. The diet compositions should be adjusted as stated in Comment 23. Also, the diet percents for mink should be adjusted or stated in Comment 23. EL-1.27

28. **Section 3.4.4.4, P. 64:** The need to account for hibernation of the raccoon has already been commented on (the statement that the only mammal species that hibernates is the little brown bat contradicts Appendix F). Regarding the bat, shouldn't the duration of feeding during the year be less than 365days to account for hibernation? EL-1.28

29. **Section 3.5.2, PP. 69 & 70:** Are the figure references in the last paragraph correct? It seems Figure K-16 should be Fig. K-14 and Fig. K-17 should be Fig. K-15. The speculation in this paragraph about the fish behaving differently in one region of the river from the rest of the river is unfounded unless EPA can put forth some plausible theory as to why this should occur. EPA should search deeper to see if it can find a plausible explanation for the difference in behavior (perhaps the data is faulty?). Similarly, speculation about the effect being due to the loss of lighter weight congeners should be backed up by analysis showing this effect. Again, in the first full paragraph on P. 70, it appears that the reference to Fig. K-16 should be Fig. K-14 and the reference to Fig. K-18 should be Fig. K-16. The speculation in the last paragraph of this section should again be backed up by analysis to show the postulated effects have a basis. The higher molecular weight in this portion of the river could also be coming from ingestion of higher molecular weight PCBs coming up from the New York City area. EL-1.29

30. **Section 3.5.3, P. 71:** Couldn't PCBs from the New York City area be pushed further upstream than the saline portion of the river by the tidal action which reaches Albany? EL-1.30

31. **Section 4.2.3, P. 80:** Are there any data to support the assumption that TEQs partition equally into the lipid phase of the egg and into the lipids in the tissue of adult fish? If so, the data should be cited, and if not, EPA should present some theoretical basis to justify this assumption. EL-1.31

32. **Sections 4.2.3.1 (P. 82), 4.2.3.2 (P. 84), 4.2.3.3 (P. 85), 4.2.3.4 (P. 86), 4.2.3.6 (P. 89) and 4.2.3.8 (P. 92) Total PCB Body Burdens:** All of these sections rely on Bengtsson's (1980) study to estimate total PCB body burdens for the fish species given in each of the sections. This study exposed the fish to Clophen A50 (Chlorine content of 50%) which would seem to be very different from PCBs. The Nebeker study of 1974 used arochlor 1254 and gives a NOAEL of 429 mg PCBs/kg tissue or 42.9 when the interspecies factor of 10 is applies vs the 1.5 from the Bengtsson study. The Nebeker study seems much more applicable than the Bengtsson study and should be used to remove unnecessary conservatism. Regarding the pumpkinseed (Section 4.2.3.1), the Nebeker study also seems preferable to the Adams et. al. (1989, 1990, 1992) field studies which are confused by not only the presence of mercury but also PAHs and Chlorine. EPA is asked to say why a value of .5 for NOAEL was selected from the Adams studies when Table 4-6 shows a value of .95. EL-1.32

33. **Sections 4.2.3.1 (p. 83), 4.2.3.4 (P. 86), 4.2.3.5 (P. 88), 4.2.3.6 (P. 90), 4.2.3.7 (P. 91) and 4.2.3.8 (P 93) Total Dioxin Equivalents (TEQs) in Eggs:** All of these sections rely on Walker's (1994) study of Lake Trout to estimate the egg TEQs for the fish species given in each of the sections. The Walker study selected is the one with the lowest LOAEL and NOAEL values. This puts unnecessary conservatism into the analysis. EPA should use the average of all the lake trout data and also average in the brook trout data since brook trout share the same genus level classification. EL-1.3

34. **Section 4.2.3.2, P. 84:** Instead of using only the lowest NOAEL and LOAEL values from applicable studies, EPA should remove unnecessary conservatism by averaging all applicable studies, including the white sucker and flathead minnow study by Elonen. EL-1.3

35. **Section 4.2.3.4, P. 86:** Instead of using either the Bengtsson study or Nebeker study for the total PCB body burden, why is the striped bass field study not used as was done for white perch? Even though the yellow perch and striped bass are in different families, they are of the same classification order so the interspecies uncertainty factor could be eliminated. This would give a NOAEL of 3.1 mg PCBs/kg tissue instead of the 1.5 value given in Section 4.2.3.4. EL-1.35

36. **Section 4.2.4.2, P. 98:** To remove excess conservatism, the average of the three studies for Mallard Ducks should be used for total PCBs rather than the lowest study. Similarly the average of the chicken and pheasant data should be averaged for the TEQs as the chicken and pheasants are in the same family. EL-1.3

37. **Section 4.2.4.3, PP. 101, 102 & 103:** There are many studies on chickens and pheasants (same family). Why aren't all or at least some of these studies averaged to get more realistic and less conservative values of LOAEL and NOAEL for total PCBs and TEQs? Also, the first paragraph in 4.2.4.3 contradicts itself. It first says a subchronic-to-chronic uncertainty factor of 10 is needed and then says it is not needed. Based on the information on P. 101 for total PCBs in eggs, non-use of this factor seems the way to go and the total PCBs LOAEL and NOAEL should be increased by a factor of 10. The average of the Scott, Britton, and Huston data for total PCBs in eggs should be used to reduce conservatism. For TEQs in eggs, the data of Powell which is based directly on dioxin seems more appropriate to me. This gives a LOAEL of .16 and a NOAEL of .08. EL-1.3

38. **Sections 4.2.4.4, PP. 102, 103 & 104:** The comments in Comment 36 pertaining to the use of averages for total PCBs, total TEQs, and total PCBs in eggs apply here also. EL-1.3

39. **Section 4.2.4.5, PP. 104, 105, & 106:** The comments in Comment 36 pertaining to the use of averages for total PCBs and total TEQs as well as the comment on TEQs in eggs apply here also. EL-1.3

40. **Section 4.2.5.1 and 4.2.5.2, PP. 106-109:** Despite the Linder, et. al. (1974) study being more robust, there is such a wide range of LOAEL and NOAEL values in Table 4-17, that some average value should be used to remove excess conservatism for total PCBs LOAEL and NOAEL values. Similarly, an average should be used rather than simply the Murray, et. al. (1979) study for total TEQs. EL-1.4

41. **Section 5.1.2.1, PP. 121 & 122:** Table 5-6 gives only the Guideline values and no comparison. Should a different table be referenced? Also, comparison of the ratios given in Tables 5-8 and 5-9 shows that the modeled sediment concentrations greatly exceed the observed sediment concentrations. This is very disturbing in that it indicates the EPA model may overpredict future PCB concentrations, not only in sediment but also in fish (and so throughout the food chain, including humans). EPA should take action to try to reduce this conservatism in their model. EL-1.4

42. **Section 5.1.3.1, PP. 122 & 123:** The exceedance of the NYSDEC Water Quality Criteria (here and in succeeding sections) seems of little significance because the water has been shown to be a minor player at best in the PCB uptake of the species of concern. EL-1.42

43. **Section 5.5.3.1, P. 138:** The inclusion of the paragraph about the least bittern, upland sand piper, and king rail is not understood. None of these species are evaluated in the report and their decline is attributed to factors beyond PCBs. Inclusion seems only for the purpose of alarming the reader. This paragraph should be deleted. EL-1.43

44. **Section 5.7, PP. 148-150:** The discussion in this section is irrelevant and should be deleted. Fist, the short-nosed sturgeon is not present in the Upper Hudson River. Second, the bald eagle is no longer on the endangered and/or threatened list. EL-1.44

45. **Section 6.5.3, P. 165:** The statement here that model error is not a significant source of uncertainty is unsubstantiated. The uncertainty between observed and model results was previously noted in Comment 41. Also, EPA's model has not been peer reviewed and EPA has not yet responded to the many significant comments submitted on their model. A statement such as that on P. 165 is premature and misleading. EL-1.45

46. **Section 7.0, PP. 167-188:** The conclusions in this section obviously follow from the information in Section 5, Risk Characterization. Rather than comment on each conclusion, this comment will focus on an overall evaluation. The total conservatism in calculating the risks is very great. Uncertainty of factors of 10 to 100 (and possibly more) are applied to LOAEL and NOAEL data. In most cases EPA has chosen the lowest LOAEL and NOAEL values from the data available. Dietary data is mostly lacking so standard formulas have been used instead and winter diets are used in some cases for year-round exposure. The effects of range, migration, and hibernation on dietary exposure have been ignored. Also, there are indications that EPA's model may be overpredicting the PCB concentrations in the food chain. EL-1.46

    Given all of the above, it is not surprising that EPA finds that none of its Assessment Endpoints are satisfied and that all of the species studied are at risk. However, how real is that risk? All indications of wildlife in the Upper Hudson River are that the wildlife is thriving. Reference the reports on field studies in Section 5. Observers in the area of the river report the area to be rich in waterfowl, populations of avian species increasing, otter numbers rebounding, "tons of raccoons" present, large number of mink and mink populations increasing. Fish populations are reported as robust and fish species historically present remain. There have not been stories in the local press (at least in the last 10 years and probably longer) that I have seen sending out alarms about dying or mal-formed wildlife in the Upper Hudson River area.

    Despite the lack of evidence of a problem, this report's TQ's (see Table 5-12 through 5-14) exceed 1.0 for many of the species and for periods up to 2018. Of particular note, are TQ's for Mallard Ducks which consistently exceed 100, with values even over 1000, and for Mallard egg TQ's which consistently range over 1000 and even over 5000. Values for other fish eating birds are also high ranging into the 100's or higher for eggs (sometimes in the 1000's with peak values over 25,000). Numbers are in the 100's and 1000's for the bald eagle which has triumphantly returned to the Hudson. Similar TQs are calculated for mammals. The otter has TQ values mainly in the hundreds with some values well over 1,000 and one over 10,000. The mink has TQs consistently in the 100's and many values over 1,000.

Even though the TQs only show a "potential" risk, I submit that the TQ values are so high in many cases that either the effects of PCBs on species should be evident in the area of the Hudson River or that the TQ analysis is overly conservative and not useful for predicting risks. Based on the lack of observed problems (rather, healthy populations are being seen), the conservatism in the report discussed above coupled with the lack of real data specific to the Hudson River, the only conclusion possible is that the report so overestimates risks as to be not useful.

304392

DONALD B. AULENBACH, PHD

24 VALENCIA LANE
CLIFTON PARK, NY 12065-5800
(518) 371-7572

EP-1

To comply with the standard 911 numbering system, effective October 1, 1999 our house number will be: 28 Valencia Lane

2 September 1999

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

Dear Alison Hess:

I have reviewed the PHASE 2 REPORT - REVIEW COPY, FURTHER SITE CHARACTERIZATION AND ANALYSIS, VOLUME 2E - BASELINE ECOLOGICAL RISK ASSESSMENT, HUDSON RIVER PCBs REASSESSMENT RI/FS, dated August 1999. Herein are my comments.

Basically I find the report inconclusive and misleading. As with VOLUME 2F - HUMAN HEALTH RISK ASSESSMENT, there are so many "weasel" words (words that one can easily slip out of) such as *may, can, could, might, probably, perhaps, estimate, etc.* The one difference compared with the HUMAN HEALTH report is that in this report the word *but* is used to express the report's conclusion despite the evidence. I refrain from citing the report section and page numbers for all these words, since they would fill this response. They appear in nearly every CONCLUSION (Section 7.0).

A grammatical comment, the author of the report should be aware that a singular subject (*a total, a combination, etc.*) takes a singular verb and is not changed when followed by a phrase starting with *of.* Thus: *A total of 10,000* still requires the verb *is*. It would be better to spell out the number, or use another subject noun such as *Altogether*. This is a common error of present day writing that should not be continued in this report. Again, this occurs throughout the report, and I shall not cite specific section and page numbers.

The report goes into great detail on how the models were derived and how the values inserted into the model were obtained. This is commendable, and leaves little question to the reviewer.

However, models are all supposed to be subject to validation. This is done by comparing measured results with the results obtained using the model. Instead of pointing out the lack of validation in the development of the conclusions, I shall refer only to the CONCLUSIONS (Section 7.0). In all cases where field observations were made (Section 7.2 for fish, 7.3 for insectivorous birds, 7.4 for waterfowl, 7.5 for piscivorous birds, and 7.6 for wildlife) a modification of the following statement is made: "Collectively, these lines of evidence indicate that current and future concentrations of PCBs are not of sufficient magnitude to prevent [concern and animal], <u>but</u>" [emphasis mine], and then it goes on to say that the model concentrations "typically exceed benchmarks". In the two cases (Section 7.7 for threatened or endangered species and 7.8 for significant habitats) where field tests were not made or included, the results of the model, only. EP-1.1

were used to conclude positively (the only instances where positive conclusions were made) that the PCB levels would have an adverse effect.

Simply put: Regardless of observed facts, we have designed a model that confirms our previously made decision that PCBs in the Upper Hudson River are harmful to the ecology there. Thus the conclusions are based on this model. EP-1.2

Since this is the overall summary of this report, I shall not go over the specific items point by point. That would be redundant and merely take up more space and time.

Thank you for this opportunity to comment on this report.

Sincerely,

Donald B Aulenbach

Donald B. Aulenbach, PhD, P.E., DEE

1015 Belleville Turnpike
Kearny, New Jersey 07032

CLH
CHEMICAL LAND HOLDINGS, INC.

HAND DELIVERED

September 7, 1999

EP-2

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

Dear Ms. Hess:

Chemical Land Holdings, Inc. (CLH)is pleased to submit the following technical memorandum entitled "Comments on the U.S. Environmental Protection Agency's Hudson River Ecological and Human Health Risk Assessments." The comments provided in this memorandum represent CLH's position on the technical approaches that were used by U.S. Environmental Protection Agency (USEPA) to assess risks to humans and ecological receptors from polychlorinated biphenyls (PCBs) in the Hudson River.

We view the Hudson River risk assessments as an example of how USEPA is going to evaluate ecological and human health risks due to organochlorines and other persistent chemicals in large river systems. We submit these comments to help ensure that USEPA assesses these risks in a technically sound manner, in keeping with applicable regulations and guidance, and in a fashion that is useful to facilitate effective risk management and decision making.

In USEPA's August 4, 1999 memorandum regarding the release of the Hudson River risk assessments, USEPA stated that "comments...should include the report section and page number for each comment." To the extent possible we have tried to provide specific section and page numbers for each of our comments. However, it was not CLH's desire to provide comments on the site-specific details of the Hudson River risk assessments. Rather, the comments contained in this memorandum are focused on the "big picture" technical approaches used by USEPA to assess chemical risks in a large riverine system, and that will likely become the basis for other riverine risk assessments conducted by USEPA in the future. For this reason, the comments are not all specifically targeted towards a page and/or paragraph of the risk assessments. Rather, several comments deal with a more general technical approach that is contained within an entire section of the assessment. We have tried to be as specific as possible in referencing either the page or section number that a comments is targeted towards.

We hope that USEPA will strongly consider these comments and re-think several of the technical approaches used to conduct the Hudson River risk assessments.

Sincerely,

Alex Pittignano

Alex Pittignano
Senior Project Engineer

Technical Memorandum

Comments on the U.S. Environmental Protection Agency's Hudson River Ecological and Human Health Risk Assessments

Prepared for

Chemical Land Holdings, Inc.
1015 Belleville Turnpike
Kearny, New Jersey 07032

Prepared by

Exponent
8201 Corporate Drive, Suite 680
Landover, Maryland 20785

September 1999

Doc. No.: 8601068.001 1301 TIF1

Comments on the U.S. Environmental Protection Agency's Hudson River Ecological and Human Health Risk Assessments

Chemical Land Holdings, Inc., (CLH) is pleased to submit these comments to the August 1999 *Hudson River PCBs Reassessment RI/FS Baseline Ecological Risk Assessment* and *Human Health Risk Assessment* report . We view this assessment as an example of how the U.S. Environmental Protection Agency (USEPA) is going to evaluate ecological and human health risks due to organochlorines and other persistent chemicals in large river systems. We submit these comments to help ensure that USEPA assesses these risks in a technically sound manner, in keeping with applicable regulations and guidance, and in a fashion useful to facilitate effective risk management decision making.

Baseline Ecological Risk Assessment (ERA) (Volume 2E)

Exposure Assessment (Section 3 of Volume 2E)

Comment 1

The exposure analysis in the ERA is conducted by simply averaging data from water, sediment, benthic invertebrate, and forage fish samples taken in various locations EP-2.1 representing relatively long reaches of the river (Volume 2E Sections 2.3.2 and 3.2). The ERA states (Volume 2E, Section 2.3.2, page 15) that the river segments represented in this scheme are "large enough to encompass the foraging areas of local populations of fish and wildlife, and provide information at an appropriate scale...[to] capture changes in spatial concentrations of PCBs."

This approach to ecological exposure analysis is inadequate for assessing chemical risks in large river systems. Risk Assessment Forum Guidelines for Ecological Risk Assessment ("the Guidelines," USEPA 1998, Section 4.2.1) clearly state that "Exposure is contact or co-occurrence between a stressor [chemical] and a receptor. The objective is to describe exposure in terms of intensity, space, and time units that can be combined with the effects assessment...A complete picture of how, when, and where exposure occurs or has occurred is developed by evaluating sources and releases, the distribution of the stressor in the environment, and the extent and pattern of contact or co-occurrence." River systems are highly heterogeneous, and heterogeneity is not captured by simply treating vertical river reaches as if they were uniform exposure habitat (which is what the ERA does). There is substantial and important horizontal structure in river systems (NRC 1992). For example, deep mid-channel environments have quite different levels and kinds of biological activity from shallow, near-shore sediments. Riffles differ from pools. Shoreline characteristics, aquatic vegetation types and substrate features typically determine the relative value of near-shore habitats for a variety of aquatic organisms.

The distribution of receptors in a river is largely a function of these habitat differences. For example, many fish-eating birds feed on small forage fish in very shallow waters. These fish are exposed to sediments and food only in limited areas of the river. Consequently, the bird exposure derives from those sediments, and not from others. Thus, the approach taken in the ERA (simply lumping habitats within river reaches as if they were equivalent from an exposure standpoint) is inadequate and does not reflect the guidance.

In general, key parameters are habitat type (e.g., foraging, breeding, loafing, and migrating), distribution, and quality. If there is no habitat for particular receptors in a particular watershed or river system, or river reach, there can be no exposure for those receptors. Because organochlorine compounds do not impact habitat *per se*, habitat conditions are the exposure baseline. If some habitat areas are present, but of relatively poor quality for particular receptors, exposure will be less in those poor quality areas. The more urbanized and degraded a watershed or river reach is, the less important it is as an exposure area. In the ERA, exposure area was by river reach with no consideration of habitat. Quantitative consideration of habitat is important for the technical and regulatory credibility of the assessment, and should be incorporated to reflect the ecological reality of exposure in this large river system. EP-2.1 (continued)

Therefore, we recommend that USEPA conduct a habitat assessment of the River, and then conduct a realistic evaluation of exposure for each receptor of interest based on their relative use of specific areas of the River. This type of analysis can be done using tools such as geographic information system (GIS) to map and quantify habitat types, and then evaluate the likely and relative use of each habitat or habitat type by the receptors of interest. This type of analysis is key to conducting a realistic assessment of exposure in aquatic systems.

Comment 2 EP-2.2

The exposure analysis in the ERA implicitly assumes that all polychlorinated biphenyl (PCB) molecules within a river reach have an equal likelihood of contacting ecological receptors. This is simply not true for PCBs or any other chemical contained in river sediments.

As described in Comment 1, exposure is properly quantified by overlaying the spatial and temporal distributions of chemicals and different types of habitats for representative receptors. In other words, not all organochlorine molecules in a river system are equal—some are more important in the exposure pool than others.

A substantial portion of the PCB in the sediments are bound and have no or limited bioavailability. Others are buried beneath the biologically active surface zone of the sediments, or are in habitats or microhabitats (Resh et al. 1996) that limit or eliminate bioaccessibility. In a particular river system, a relatively large proportion of organochlorine molecules may be in sediments that are not bioavailable or bioaccessible, and thus cannot drive ecological risks. USEPA should evaluate and document the particular areas in the River that contain PCBs at levels that may pose ecological risk, based on a realistic exposure assessment as described above.

Comment 3

The exposure analysis in the ERA fails to account adequately for life history characteristics of particular receptors. Among the receptors identified for ecological risk assessment, there is a wide range of life history parameters that affect exposure in important ways, and should, therefore, be accounted for in the analysis. Some species (including anadromous fish like striped bass and shortnose sturgeon and migratory birds including tree swallow, mallard, belted kingfisher, great blue heron, and bald eagle) may acquire substantial doses and/or body burdens of PCBs in areas remote from the Hudson. For example, the birds migrate to the southern United States and/or to Central and South America, where they feed actively in preparation for the return migration in the spring (Welty 1982). Striped bass and shortnose sturgeon leave the Hudson and migrate along the coast to deeper and/or more southern waters. EP-2.3

In both cases, there is substantial likelihood that these species acquire PCBs from sources unrelated to the Hudson. Source is an important exposure parameter (USEPA 1998, Section 4.2.1.1). Yet the ERA treats all PCBs as if the source of exposure was the Hudson River system. Relatively simple tools are available to evaluate the ecology of fish and bird movement, and many readily available sources (including published information on bird and fish migration routes and wintering ground populations) track the time spent in summer vs. winter habitats. In addition, if resident subpopulations of some species (such as the striped bass) are present, ecological risks should be quantified separately for this subpopulation because the exposure sources will differ. The potential for exposure in other areas (e.g., waterfowl and tree swallows migrate to Central and South America) should be addressed and, to the extent possible, quantified in the Hudson River risk assessment.

Effects Assessment (Section 4 of Volume 2E)

Comment 4

The ERA relies on a deterministic method for identifying toxic effects thresholds for the ecological receptors that is, a yes/no description of the likelihood of response. This is a common and widely accepted approach to conducting ecological risk assessments. However, for assessments as complex as those involving organochlorine compounds in large river systems, probabilistic analysis of toxicity may be as important for credible risk assessment as is probabilistic analysis (on a habitat basis) of exposure. This is particularly critical for risk assessments involving organochlorine compounds for which susceptibility of organisms is known to differ enormously (by several orders of magnitude across major taxa, by more than an order of magnitude within a single class such as fishes). EP-2.4

By not employing a probabilistic analysis of toxicity, the risk assessment necessitates the application of arbitrary and unjustified "uncertainty factors" (see separate comments below) that hinder utility of the entire risk assessment. The Guidelines (USEPA 1998, Section 4.3.1.1) state that "Point estimates may be adequate for simple assessments or comparative studies of risk...," neither of which is the case for the Hudson River ecological risk assessment. Furthermore, when point estimates are used for ecological risk assessment, they should be derived based on the slope of the dose-response curve (Chapman et al. 1998), and the ERA fails to provide any information whatsoever on quantitative aspects of the dose-response relationship for PCBs. The ERA should consider probabilistic alternatives to the deterministic toxicity thresholds.

Risk Characterization (Section 5 of Volume 2E)

Comment 5

The ERA identifies a number of site-specific field investigations of population-level parameters for some receptors, but then dismisses these studies or gives them little or no weight in the "weight of evidence" analysis. This is a serious shortcoming. EP-2.5

Site-specific population-level data (such as field studies of reproductive impairment and population parameters) are the most relevant and useful data for risk assessment. The Guidelines (USEPA 1998, Section 4.3.1.3.2) clearly state: "Risks to organisms in field situations are best estimated from studies at the site of interest. However, such data are not always available." For the Hudson River, such data are available, and should therefore be used and given appropriate weight in the risk assessment. It is not appropriate for the ERA to discard such data, particularly when the results (such as the findings of tree swallow field studies) are consistent and credible.

Comment 6

EP-2.6

The ERA includes screening thresholds for water and sediment quality explicitly as a component of the definitive risk characterization. This is inappropriate from both a scientific and regulatory viewpoint.

Screening thresholds are applied only to guide quantitative risk characterization. Such thresholds are "...based on generic assessment endpoints (e.g., protection of aquatic communities from changes in structure or function) and are assumed to be widely applicable to sites around the United States" (USEPA 1997). Such generic thresholds include water quality criteria and sediment effects thresholds, both of which are designed to identify chemical concentrations below which adverse effects are unlikely. These thresholds are not intended to and cannot be used to quantify risk in an remedial investigation and feasibility study (RI/FS) context for supporting risk management decision making. USEPA (1997) clearly states that "...requiring cleanup based solely on [the information developed during risk screening assessment] would not be technically defensible." The ERA should be modified to eliminate screening thresholds from the definitive risk characterization.

Comment 7

The ERA consistently misapplies toxicological effect thresholds. In calculating hazard quotients, it is appropriate to use highest no-observed-adverse-effect-level (NOAEL) when a range of choices is available. USEPA (1997) states: "For those contaminants with documented adverse effects, one should also identify the highest exposure level that is a NOAEL." Yet the ERA, without explanation, uses the lowest NOAEL. This fundamental toxicological error should be corrected in a revised version of the ERA. EP-2.7

Uncertainty Analysis (Section 6 of Volume 2E)

Comment 8

In the uncertainty analysis of the risk assessment USEPA uses the "uncertainty factor approach" to estimate safe concentrations of PCBs. Unfortunately, there is no foundation in the technical literature for applying "uncertainty factors" of 10 to toxicity thresholds. In fact, comparative toxicological studies have clearly established that the "analogy with human health risk assessment" on this is outdated and indefensible (Chapman et al. 1998). Indeed, this was reflected in USEPA's decision to not apply uncertainty factors when applying toxicity data developed for gallinaceous birds to fish eating birds in deriving the Great Lakes water quality criteria (USEPA 1995). EP-2.8

We suggest that it is important for USEPA to revise this document to properly address this issue. When inappropriate factors-of-ten uncertainty factors are applied, it is difficult or impossible to tell whether risk management decisions are being made to reduce real potential risks or analytical uncertainty. Unless uncertainty bounds can be quantified so that risk management decisions can be understood in the context of risk and uncertainty (which is impossible with the "factors of 10" approach employed in the Hudson risk assessment), effective risk management decisions cannot be made. The ERA should be revised to provide uncertainty bounds or technically defensible "uncertainty factors," and not rely on simplistic, outdated "factors of 10".

Comment 9

For estimating effects levels on a toxicity equivalence (TEQ) basis to fish, the ERA repeatedly applies Walker et al. (1994) studies on Lake Trout, generally including "factors of 10" uncertainty divisors. This approach is simplistic, not credible, and scientifically indefensible. Salmonids like Lake Trout are highly sensitive to organochlorine compounds, and the application of salmonid studies, particularly with an uncertainty factor of 10 to non-salmonid fishes is inappropriate. For example, in determining effluent quality under the Clean Water Act, USEPA guidance provides a "resident species recalculation" procedure for water bodies lacking certain receptors (such as sensitive salmonids) on which generic standards may be based. The intent of this procedure is to assure that risk management decisions are not made to inappropriately stringent standards. The same procedure should be followed in risk assessments. EP-2.9

For watersheds, water bodies, or river reaches where only warm water fish communities exist, salmonids toxicity thresholds should not be applied. The ERA should be modified to identify areas of the Hudson supporting only warm-water fish communities, and apply a separate and appropriate toxicity threshold for these areas. A salmonid-based threshold should be applied only to areas supporting a cold-water fish community, and then without "factors of 10" uncertainty divisors.

Conclusions

Comment

The Hudson risk assessment cannot be used to support effective risk management decision making for the Hudson River. This is not in keeping with applicable USEPA requirements under Comprehensive Environmental Response, Compensation and Liability Act of 1980 (CERCLA); the National Contingency Plan (NCP); and implementing guidance such as that for conducting RI/FSs. It is clear that risk assessments are one of the most critical decision tools to be applied to risk management at Superfund sites (e.g., USEPA 1988, NCP at 300.430(d) and (e)). For the Hudson, the result of not using appropriate exposure and toxicological analyses to develop an accurate characterization of risk, renders this document nothing more than a broad-brush and generic risk assessment. EP-2.10

Sediment parcels that might be associated with higher levels of exposure or toxicity cannot be identified or prioritized for risk management. Given the gross importance of "uncertainty factors of 10" in the technical conclusions, risk managers cannot even know if they would be managing real risks or simply analytical uncertainty if actions were to be taken. Given that the job of risk assessment is to support sound risk management decision making, a risk assessment that concludes, on a generic basis, that risks are "everywhere and all the time" is useless and unacceptable. The ERA should be revised to reflect the realities of exposure and toxicology in such a way that clear, credible, and defensible risk management decisions can be made. Otherwise, the entire exercise is a waste of time and effort.

Literature Cited

Chapman, P., A. Fairbrother, and D. Brown. 1998. A critical evaluation of safety (uncertainty) factors for ecological risk assessment. Env. Tox. Chem. 17:99–108.

NRC (National Research Council). 1992. Restoration of aquatic ecosystems. National Academy Press, Washington, DC.

Resh, V.H., M.J. Meyers, and M.J. Hannaford. 1996. Macroinvertebrates as biotic indicators of environmental quality. pp 647–667. In: Methods in stream ecology. Hauer, F.R., and G.A. Lambert (eds.), Academic Press, NY.

USEPA. 1988. Guidance for conducting remedial investigations and feasibility studies under CERCLA. EPA 540 G-89-004. U.S. Environmental Protection Agency, Office of Emergency and Remedial Response, Washington, DC.

USEPA. 1995. Great Lakes water quality initiative criteria documents for the protection of wildlife: DDT; mercury; 2,3,7,8 TCDD; PCBs. EPA/820/B-95/008. U.S. Environmental Protection Agency, Office of Water, Washington, DC.

USEPA. 1997. Ecological risk assessment guidance for Superfund: Process for designing and conducting ecological risk assessments. Interim Final. EPA 540-R-97-OCS, OSWER 9285.7-25, PB97-963211. U.S. Environmental Protection Agency, Office of Solid Waste and Emergency Response, Washington, DC.

USEPA. 1998. Guidelines for ecological risk assessment. EPA/630/R-95/002F. U.S. Environmental Protection Agency, Risk Assessment Forum, Washington, DC.

Walker, M.K., P.M. Cook, A.R. Batterman, B.C. Buttrworth, C. Berini, J.J. Libal, L.C. Hufnagle, and R.E. Peterson. 1994. Translocation of 2,3,7,8-tetrachlorodibenzo-*p*-dioxin from adult female lake trout (*Salvelinus namaycush*) to oocytes: effects on early life stage development and sac fry survival. Can. J. Fish. Aquat. Sci. 51:1410–1419.

Welty, J.C. 1982. The life of birds. Saunders College Publishing, NY.

General Electric

John G. Haggard, Manager
Hudson River Program

General Electric Company
320 Great Oaks Office Park, Ste. 323
Albany, NY 12203
Fax: (518) 862-2731
Telephone: (518) 862-2739
Dial Comm: 8* 232-2739
E-Mail:John.Haggard@corporate.ge.com
Pager: 518-484-3177

EG-1

Hand Delivered

September 7, 1999

Alison A. Hess, C.P.G.
U.S. Environmental Protection Agency
290 Broadway, 19th Floor
New York, NY 10007-1866

RE: HUDSON RIVER ECOLOGICAL RISK ASSESSMENT – COMMENTS

Dear Ms. Hess:

Enclosed are the comments of the General Electric Company (GE) on the U.S. Environmental Protection Agency's (EPA) "Volume 2E – Baseline Ecological Risk Assessment Hudson River PCBs Reassessment RI/FS" (BERA).

Given the scale of the Upper Hudson River site, EPA should strive to bring the best science to bear to understand the risks to ecological receptors. This, however, is not reflected in the recently released report. The ecological risk assessment is best described as a "screening" analysis that one would perform to determine if a site-specific assessment was needed. In addition to other problems, the report relies on overly conservative assumptions concerning toxicity and exposure; fails to consider a significant amount of field data; misrepresents important conclusions for two field studies; and fails to use the weight-of-evidence method in a useful way.

Without significant revisions, the ecological risk assessment findings are too unreliable to guide development of remedial objectives or to predict what impact a remedy will have on the river ecology.

Please place a copy of this letter and associated comments in the site administrative record.

If you have any questions on these comments, please let me know.

Yours truly;

John G. Haggard

Encl:

304408

cc: Richard Caspe, U.S. EPA
William McCabe, U.S. EPA
Douglas Fischer, U.S. EPA (ORC)
Erin Crotty, NYDEC
Walter Demick, NYDEC
William Ports, NYDEC
Ron Sloan, NYDEC
Bob Montione, NYDOH
Jay Field, NOAA
Anthony Geidt, NOAA
Lisa Rosman, NOAA (New York)
Tom Brosman, NOAA

Comments of General Electric Company on Volume 2E – Baseline Ecological Risk Assessment Hudson River PCBs Reassessment RI/FS

September 7, 1999

General Electric Company
Corporate Environmental Programs
320 Great Oaks Office Park, Suite 323
Albany, NY 12203

LWB Environmental Services, Inc.
105 Wesley Lane
Oak Ridge, TN 37830

Quantitative Environmental Analysis, Inc.
305 West Grand Avenue
Montvale, NJ 07645

TABLE OF CONTENTS

Appendix A: Critique of Analyses Presented in Appendix K of EPA's Ecological Risk Assessment

Appendix B: Critique of the Evaluation of the Predictive Capability of the NOAA (1999) SEC Values

1.0 Introduction And Executive Summary

General Electric Company (GE) submits these comments on the Phase 2 Report – Review Copy, Further Site Characterization and Analysis, Volume 2E – Baseline Ecological Risk Assessment Hudson River PCBs Reassessment RI/FS (BERA) which was released by the U.S. Environmental Protection Agency (EPA) on August 4, 1999.

GE's comments are premised on the understanding that the objective of the BERA is to support remedial decisionmaking for the Upper Hudson River.[1] To achieve this objective, the assessment must provide:

- a sound and reliable description of the effects of current PCB exposures on biota in the Hudson River Valley;
- a foundation for projecting the responses of those biota to alternative remedies; and
- a sound technical underpinning for comparing the ecological benefits gained through remediation to the ecological costs of implementing remedial actions.

The scale of the sociological, ecological and economic impacts of a remedy for a large, complex ecosystem such as the Hudson River dictate that the best science be employed to reduce uncertainty in decision making. The assessment should reflect best scientific practice in ecological risk assessment as described in EPA's Guidelines for Ecological Risk Assessment (EPA 1998a) and exemplified in ecological risk assessments that have been published in the peer-reviewed scientific literature. EPA's assessment does not reflect best scientific practice. It is excessively conservative because it relies on screening level benchmarks. It is deficient both because of

[1] The Upper Hudson River is the 40 mile stretch between Hudson Falls and the Federal Dam at Troy. For reasons explained previously to the Agency, GE maintains its position that the Hudson River PCBs Superfund Site encompasses only these 40 miles and does not extend to the Lower Hudson River.

EPA's failure to collect relevant data over the last ten years and because of its failure to examine and utilize existing site-specific data. It is grossly insufficient for use in determining the need for a remedy or selecting a remedy.

<u>The Approach Used bv EPA is Inconsistent with Best Scientific Practice</u>.

Best scientific practice in ecological risk assessment differs in two fundamental respects from best practice in human health risk assessment. First, although there are limited circumstances, such as protection of endangered species, in which adverse effects on individual organisms are sufficient to warrant management action, effects on populations and communities are the prime ecological focus and should be the basis for analysis (EPA 1998b). Second, whereas most human health risk assessments must be based on predictions from models, ecological risk assessments can be based on observed exposures and effects measured in well-designed, site-specific studies. EPA's work fails to meet either of these basic benchmarks. The hallmark of this assessment is its repeated use of literature-based screening values to project effects on individual organisms.

<u>EPA's Assessment Focuses on Individuals, Not Populations or Communities</u>.

With the exception of the analysis of benthic invertebrates, EPA's assessment endpoints address risks to individual organisms, not populations. No data or methods are presented that either evaluate effects on populations or communities directly or provide a basis to extrapolate from individual level effects to population effects.

<u>EPA Failed to Collect Ecological Information on the Hudson and Its Assessment Ignores or Dismisses Substantial and Valuable Site-Specific Data</u>.

Despite spending ten years on this Reassessment, EPA has failed to examine the wild populations of the River and its Valley with the exception of benthic invertebrates. This is -indefensible. Moreover, EPA's assessment repeatedly ignores or dismisses the substantial and valuable data that have been collected about the biota of the Hudson over the last thirty years and which provide both probative evidence as to the health of the wildlife populations and a mechanism to test the results derived from generic or hypothetical analyses. To give two examples, the New York State Department of Environmental Conservation (NYSDEC) has examined macroinvertebrate populations in the Hudson and could not identify any adverse effects from the exposure to PCBs. Extensive data on the fish populations of the Lower Hudson are available but were not analyzed.

<u>**EPA Misstates the Results of the Site-Specific Studies**</u>.

EPA relies on two Hudson-specific studies: EPA's own study of benthic invertebrates and the Fish and Wildlife Service examination of tree swallows. EPA's Risk Characterization, in the body of its report, correctly concluded that the benthic invertebrate community analyses could not distinguish any clear effects from PCBs. The BERA's conclusions misstated the result of this study by claiming that the analysis showed a reduced macroinvertebrate community with potential risk due to the site. The tree swallow study was unable to show a dose-response relationship between tree swallow reproduction and PCB exposures. The behavioral responses that were identified are not correlated with reproductive success. The Assessment inaccurately claims that the study showed decreased reproductive success related to PCB exposures.

<u>**Available Population and Community Data Conflict with EPA's Conclusions**</u>.

In addition to the two site-specific studies that were misstated by EPA, other available data on the status of Hudson River biological resources conflict with EPA's conclusions. The New York State Department of Environmental Conservation (NYSDEC) has examined macroinvertebrate populations in the Hudson and could not identify any adverse effects from exposure to PCBs. National Marine Fisheries Service (NMFS) and the Fish and Wildlife Service analyzed the effects of PCBs on the striped bass population under the Atlantic Striped Bass Conservation Act and concluded that PCBs were not the cause of declines in the Hudson or coastal striped bass populations. Data available to EPA demonstrate that populations of striped bass, shortnose sturgeon, and other Hudson River fish species have increased in recent years. NYSDEC has examined the growing number of eagles in the Hudson Valley and throughout the state. Breeding Bird Survey data document healthy populations of many other bird species in the Hudson Valley. These data and other similar data reflect the actual health of the wild animal populations of the Hudson. These are the facts that count.

EPA's Assessment Fails to Use the Weight-of-Evidence Approach In a Sound and Meaningful Manner.

Because any single study can produce ambiguous results, multiple lines of evidence should be developed using different types of data. Each line of evidence should be evaluated, and all the lines together should be used to draw conclusions concerning the existence, causes, and magnitudes of risks. Evaluation criteria are discussed in the Guidelines for Ecological Risk Assessment (EPA 1998a), and in the refereed scientific literature (Suter and Loar 1992, Suter 1993, Menzie et al. 1996, Suter et al. 1999) This is not a simple process of counting up a number of studies and keeping score of the findings. The results of 10 bad studies cannot be compared on an equal footing with the results of one high quality, relevant study. The quality and relevance of each study must be closely evaluated.

While the BERA claims to have developed and analyzed several lines of evidence for a wide variety of species, in fact, the BERA reflects a basic misunderstanding of how to perform a weight-of-evidence assessment. Most of the assessment endpoints are addressed using only one line of evidence: comparison of measured or modeled exposure concentrations to generic, non-specific toxicity benchmarks, particularly Toxicity Reference Values (TRVs) and Sediment Effect Concentrations (SECs). EPA failed to collect the information required to implement the weight-of-evidence approach properly. EPA began its reassessment almost ten years ago; its failure to collect site-specific data and to examine existing data closely is indefensible. Beginning a "field survey" two months before releasing the BERA by making phone calls to collect anecdotal information is no substitute for the comprehensive data collection demanded by a site of this size and complexity. EPA's cavalier attitude toward factual evidence cannot be reconciled with a true weight-of-the-evidence analysis.

EPA's Assessment is Excessively Conservative.

The approaches used for exposure assessment and effects assessment result in overestimates of actual exposures and risk. In the exposure assessment, unnecessarily conservative assumptions are made concerning (1) treatment of samples in which a target chemical was not detected yet was still assumed to be present, (2) diet composition and food consumption rates, and (3) habitat utilization. The effects assessment relies on screening-level criteria and Toxicity Reference Values (TRVs). These values are intended to identify the lowest doses that could potentially affect organisms, not the values at which a population will exhibit adverse effects at relevant ecological endpoints. The Toxicity Quotients (TQs) developed from these exposure and effects estimates greatly inflate the ecological risks of PCBs present at the Hudson River site.

Since the ultimate question for the risk manager is what effect an array of possible remedial actions will have on wildlife populations, the assessment should reflect prudent realism rather than conservatism. Use of excessively conservative assessment calculations will lead to a misrepresentation of site conditions and result in the prediction of excessively beneficial results from various remedial actions, which will not be borne out in fact. EPA policy on this point was articulated by Administrator Browner in her cover letter on EPA's Guidance for Risk Characterization: "while I believe that the American public expects us to err on the side of protection in the face of scientific uncertainty, I do not want our assessments to be unrealistically conservative. We cannot lead the fight for environmental protection into the next century unless we use common sense in all we do." EPA's Assessments does not follow this basic Agency policy.

The SECs are not Reasonable Estimates of Effects of PCBs on Benthic Invertebrates.

For benthic invertebrates, EPA relies on SECs which operate as TRVs. These also are screening-level criteria that do not reflect effects on benthic invertebrates demonstrably caused by PCBs. The SECs incorporate the assumption that the supposed effects on benthic invertebrates will be reflected in adverse effects on fish populations, threatened and endangered species, and the ability of particular habitats to support sustainable, healthy animal populations. There is no demonstration that an exceedance of SEC values for PCBs has any identifiable adverse effect on other biotic populations.

EPA's Assessment Should Not Rely on the TEQ Approach Because It Is Not Sufficiently Developed and Has Been Applied Improperly in the BERA.

The Toxicity Equivalence (TEQ) approach, used by EPA as one method for assessing the risks of PCBs to exposed fish and wildlife, converts concentrations of "dioxin-like" organic chemicals to equivalent concentrations of dioxin. The Toxicity Equivalency Factors (TEFs) used by EPA are, according to their developers, order-of-magnitude approximations (Van den Berg et al. 1998). The analytical methods used by EPA to measure concentrations of individual congeners are not sensitive enough to distinguish biologically significant and insignificant concentrations in fish tissue. EPA's treatment of data below the minimum quantification level for these congeners produces highly inflated estimates of both exposures and effects. Moreover, according to an expert review performed for NOAA, information on the relative sensitivities of different fish species to dioxin-like compounds is insufficient to support application of the TEQ approach to Hudson River fish species. This technique is not developed to the point where it is an effective tool for realistic risk assessment.

EPA's Use of the Upper Hudson Food-Chain Bioaccumulation Model is Premature.

GE has previously noted significant deficiencies in the Upper Hudson River model used by EPA to quantify the bioaccumulation of PCBs in fish tissue. These deficiencies have not yet been addressed by EPA, and the model still predicts higher tissue concentrations than are actually observed. The Agency has properly elected not to use the Thomann-Farley model for a risk assessment of the Lower Hudson until it has been fully vetted and reviewed. The same logic applies to EPA's Upper Hudson River model: it should not be used for risk assessment until it is validated against field data, the public comments have been addressed, and it has undergone a rigorous peer review.

* * *

Each of these major limitations and deficiencies is described more fully in the text which follows. Another way to summarize the deficiencies in the BERA is to compare it against the standards for admissibility of expert scientific evidence established by the Supreme Court in Daubert v. Merrell Dow Pharmaceuticals, Inc., 509 U.S. 579 (1993). When such a comparison is made, it is clear that the BERA falls far short of these standards for sound science and would not be admitted for consideration by a jury deciding a scientific question to which it was allegedly relevant. By the same token, it should not be used for decision-making in this Reassessment.

2.0 The Ecological Risk Assessment Does Not Conform To Best Scientific Practice

EG-1.1

EPA's ecological risk assessment is based principally on "Toxicity Quotients" (TQs), i.e., comparisons between measured or modeled exposure concentrations and concentrations believed to be potentially harmful to organisms. Conservative, "screening-level" data and assumptions are used to define both the exposures and the effects. Screening-level data and models, such as those used by EPA, are deliberately designed to be conservative, i.e., to minimize the possibility that any potential adverse effects will be missed. They overstate the actual effects of most chemicals at most sites. The Ecological Risk Assessment Guidance for Superfund (EPA 1997) explicitly states that decisions to require remedial action based solely on screening-level data "would not be technically defensible." A scientifically defensible ecological risk assessment should be based on the methods described below, not on TQs.

A wide variety of techniques for measuring and characterizing ecological risks at contaminated sites are described in EPA's Guidelines for Ecological Risk Assessment (EPA 1998) and Ecological Risk Assessment Guidance for Superfund (EPA 1997). These include:

- Measurements of the abundance, diversity, and other characteristics of exposed invertebrate, fish, and wildlife communities.
- Measurements of reproductive success in fish, birds, and mammals.
- *In-situ*, whole-media, and dietary toxicity tests using selected receptors or appropriate surrogate species.

Each type of measurement typically requires knowledge of and data relevant to the population dynamics of the species for appropriate use in assessing risks to wild populations. Measures of effects on individual organisms must be interpreted in the context of the distribution, abundance, and temporal dynamics of the exposed populations.

These methods are described in available EPA guidance documents and in the refereed scientific literature. Experience and practice at the other comparable sites demonstrates the inadequacy of EPA's BERA for the Hudson River.

The assessment performed for the Clinch River Study Area in Tennessee is a particularly appropriate example of an approach consistent with best scientific practice because: 1) it involved a study area similar in scale to the Hudson River, and 2) sediment-derived PCBs were a major concern. The Clinch River ecological risk assessment was documented recently in a series of peer-reviewed articles in *Environmental Toxicology and Chemistry* (Volume 18, no.4, 1999, pp.579–654). Table 1 compares the assessment endpoints, data types, and assessment methodologies used in these two assessments.

Both assessments address risks of sediment-derived PCBs to benthic macroinvertebrates, fish, birds, and mammals. However, far more information was used in the Clinch River assessment. Whereas the Hudson River BERA primarily relies on TQs, the Clinch River assessment employed a wide variety of site-specific data. In addition to TQs, the Clinch River assessment used site-specific toxicity tests, histopathological studies, avian reproduction studies, a mink dietary toxicity test, and local/regional fish and benthic macroinvertebrate surveys. In contrast with the deterministic TQs used in the Hudson River assessment, Monte Carlo analyses and other probabilistic approaches were used in the Clinch River risk assessment to characterize the likelihood that adverse effects might occur as a result of exposure to PCBs and other chemicals.

Data collection to support the Clinch River assessment began in 1989, the same year EPA initiated its reassessment of PCBs in the Hudson River. The draft assessment for the Clinch River was completed in 1995 and the final assessment was issued in 1996. EPA had ample time to perform similar studies for the Hudson River. The Hudson River BERA repeatedly cites lack of data on population trends or parameters but never offers an explanation for why such data were not collected. EPA's attempt to patch this glaring omission by making phone calls to collect anecdotal information beginning two months before releasing the BERA (Tables 5-67 and 5-85) falls far short of the mark.

2.1 The Assessment Endpoints for Fish and Wildlife Receptors Pertain to Effects on

Individuals, not Populations

EG-1.2

EPA's assessment endpoints for fish and wildlife refer to protection and maintenance of "survival, growth, and reproduction" of individual organisms rather than to the sustainability of populations.

The *Guidelines for Ecological Risk Assessment* (EPA 1998A) permit assessment endpoints to be defined at any level of biological organization including the individual organism. This latitude is necessary because the guidelines are intended to be applicable to all of EPA's regulatory activities. Many of these activities (e.g., development of water-quality criteria and registration of new chemicals) do not employ site-specific data and cannot directly address effects of chemicals on populations and communities. In making decisions concerning remedial action needs for the Hudson River, however, decisionmakers must determine (1) whether the sustainability of exposed biological populations and communities is being threatened by the presence of PCBs in Hudson River sediment, and (2) whether the positive effects of a particular remedy will be greater than any negative ecological effects of carrying out the remedy.

A focus on populations rather than individuals is necessary because compensatory mechanisms that operate in all biological populations permit these populations to sustain themselves in spite of impacts to some individual organisms. Even if there were statistically significant reductions in survival, growth and reproduction of individuals, such data alone cannot be used directly as surrogates for evaluating adverse effects to populations, communities, or ecosystems. Survival, growth, and reproduction rates are interrelated in complex ways, and the contribution of each factor to eventual population indices depends on the life history of the organism and compensatory mechanisms at the population and community levels. EPA's draft Risk Management Guidance clearly states that populations are the appropriate level of ecological organization for assessment. (EPA 1998b)

Large numbers of the fish and wildlife species are routinely harvested for recreation or human consumption without threat to stock abundance. EPA's focus on the individual organism is inappropriate since it does not rest on a showing that effects on individuals will be reflected in effects on the relevant populations. Consequently, it cannot support a reasoned remedial action decision for the Hudson River.

An appropriate example of an assessment endpoint for fish or wildlife is provided in the *Ecological Risk Assessment Guidance for Superfund* (1997, Highlight I-2, p. I-6; emphasis added): "[s]ufficient rates of survival, growth, and reproduction *to sustain populations of carnivores typical for the area.*"

2.3 The Assessment Endpoint for Habitat As Presently Stated is Meaningless

EG-1.3

EPA lists "Protection of significant habitats" as an assessment endpoint. No definition of "habitat" is provided, and generic water and sediment quality criteria are the only measures employed by EPA to evaluate effects on habitat. Because some of the remedial actions (i.e., dredging of sediment) being considered by EPA would destroy the contaminated habitat, it is important to define and examine this endpoint realistically so that the adverse ecological effect of such remedies can be weighed and taken into account in considering remedial alternatives.

It is also important to note that no one has alleged that levels of PCBs found in these "significant habitats" are causing damages to the habitat. The question should be directed at whether the organisms supported by the habitat are adversely impacted by PCBs. This illustrates another basic flaw with the BERA; EPA has made no attempt to map habitats and determine how different species utilize different habitats.

2.4 EPA's Measurement Endpoints Are Not Predictive of Population or Community Effects

EG-1.4

With the exception of the benthic invertebrate community survey, all of the measurement endpoints used by EPA are generic toxicity benchmarks: sediment-quality criteria, water-quality criteria, and TRVs derived from the most conservative single-species toxicity tests available. These benchmarks cannot be validly used to infer the existence of adverse effects on populations or communities.

To support a remedial action decision for the Hudson River, the measurement endpoints used in the assessment must either (1) directly address the abundance and distributions of populations and communities, or (2) provide an appropriate line of evidence regarding effects on populations and communities. Quantitative biological surveys can provide information to assess effects on populations and communities directly. Supporting lines of evidence can be developed from data such as site-specific toxicity tests, histopathological and biochemical studies of exposed populations, and dose-response data from all relevant laboratory tests. Models are available that can potentially be used to quantify effects of chemicals on exposed populations (Barnthouse 1993). Even if data on the dynamics of exposed populations are insufficient to support quantitative modeling, the above measurement endpoints can be used to estimate the fraction of a population that could potentially be impaired by exposure to PCBs.

2.5 EPA's Assessment is Based on Screening-Level Models and Ignores, Discounts, or Misinterprets Empirical Data

EG-1.5

EPA's Assessment does not examine and incorporate site-specific data such as biological surveys, whole-media toxicity tests, or reproductive effects studies. In fact, with the exception of the TRVs for tree swallows that were based on field studies, none of the benchmarks are based on site-specific data. According to Suter (1999), site-specific ecotoxicological studies "can provide a firm basis for decision making, often resulting in savings in remedial costs far beyond the cost of performing the studies." As documented in Table 1, a wide variety of site-specific data were collected for the Clinch River BERA. Generic criteria and TRVs provided only one of many lines of evidence used in the assessment.

EPA's use of water and sediment-quality criteria as measurement endpoints for the BERA is inappropriate and redundant with earlier uses of the same criteria in the screening assessment performed for the Phase 1 investigation (EPA 1991). The SOW for the BERA correctly notes that comparisons of exposure concentrations to ambient water quality criteria and sediment quality guidelines merely indicates that there is a "potential for risk" to aquatic organisms. Notwithstanding this admission, EPA uses these criteria as separate lines of evidence for *every assessment endpoint* addressed in the assessment, and using these criteria concludes for each that actual risks are present. This is not a substitute for a site-specific ecological risk assessment. **EG-1.6**

Rather than measuring exposures of birds and mammals to PCBs, EPA calculates exposure using biota-sediment accumulation factors (BSAFs) and the FISHRAND model, which simulates the bioaccumulation of sediment-derived PCBs in aquatic food chains. Where data is available it should be used. For instance, EPA should have directly estimated avian exposures using measured concentrations in eggs. PCB concentrations in fish for the period 1993–1996 were computed by the model; data is available for this period and should have been used.

EPA ignores or discounts other existing site-specific data. In addition to the benthic community data collected by EPA and used in the assessment, data on benthic community structure are available from NYSDEC (1993) and Exponent (1998 a,b). These data were not used. **EG-1.7**

EPA ignores the large quantity of fish population data available for the Lower Hudson River from surveys conducted by NYSDEC and the Hudson River utility companies. Abundance trends for striped bass are reported by NOAA and are used in stock assessments performed by the Atlantic States Marine Fisheries Commission (NMFS 1998a). Estimates of the abundance of shortnose sturgeon in the Hudson River are available for the 1970s (Dovel et al. 1992) and the 1990s (Bain et al. 1995); these studies are summarized in the Final Recovery Plan for shortnose sturgeon (NMFS 1998b). **EG-1.8**

EPA acknowledges (p. 129) the growth in the white perch population in the lower Hudson, but discounts the significance of this growth on the grounds that "it is possible that PCBs could influence rates of reproduction and recruitment to a degree that is not manifested in recent populations trends." A similar argument is made in discussing the continued presence of apparently healthy fish populations in the upper Hudson, and of apparently healthy bird and mammal populations in both the upper and lower Hudson valleys. It appears that EPA's position is that a decline in abundance would indicate an adverse effect due to PCBs; an increase indicates only that the adverse effects (which are purportedly demonstrated by the TQs) are being masked by other factors. This argument is obviously contrary to established principles of scientific inference. **EG-1.9**

Data on population trends and reproductive success in bald eagles and other bird species are available but were not considered by EPA. NYSDEC has been monitoring the winter use and breeding activity, tissue contaminant concentrations, and reproduction of bald eagles in New York State and in the Hudson River area for many years. In addition to these reproductive data, the collection of prey from eagle nests by NYSDEC provides empirical, site-specific information on the diets of bald eagles that should have been used by EPA to improve the realism of its exposure model. Peter Nye of NYSDEC found remains of grebes, eels, pickerel, bullhead, herring and carp in eagle nests. In addition, NYSDEC collected unhatched eggs which could be analyzed. Information on the PCB concentrations in some of these prey, including bullhead and eels, are available (Secor 1997). Other data for bird populations are available from the U.S. Fish and Wildlife Service (USFWS 1997), NYSDEC (1997), and the North American Breeding Bird Survey (Sauer et al. 1997). EPA used none of these data. **EG-1.10**

Finally, as discussed in Section VI of these comments, the benthic community study and tree swallow reproduction study performed to support EPA's assessment were misinterpreted as supporting EPA's conclusions, even though adverse effects that could be validly attributed to PCBs were not detected in either study. **EG-1.11**

2.6 EPA Improperly Applies the Weight of Evidence Approach

EPA claims (p. 167) to have used a "weight of evidence" approach to "assess the potential for adverse reproductive effects in the receptors of concern as a result of exposure to PCBs in the Hudson River." EPA's assessment, however, presents virtually no lines of evidence other than screening-level TQs and fails to present a framework for resolving conflicting lines of evidence.[2] Many of the so-called "lines of evidence" are based on the same or similar data and are not truly **EG-1.12**

[2] EPA's failure to present a framework for resolving conflicting lines of evidence is remarkable in light of its statement in the Responsivenss Summary for the Ecological Risk Assessment Scope of Work: "The quality of each measurement endpoint will be evaluated according to the attributes identifed by Menzie et al. (1996) and will be discussed in ERA. USEPA notes that Dr. Menzie will be directly involved for the Hudson River PCBs Reassessment ERA" (Responsiveness Summary at 19).

independent. For example, TQs for fish and wildlife are presented using TRVs based alternatively on total PCBs and TEQs. Both approaches to TRV-development are based on the same types of data; they simply have different theoretical foundations and use different exposure estimates. Similarly, the same water and sediment-quality-based TQs are cited as evidence for risks to every receptor group. In reality, most of the assessment endpoints are addressed using only one line of evidence: comparison of measured or modeled exposure concentrations to generic toxicity benchmarks. In contrast, appropriately defined multiple lines of evidence would include completely independent study designs, such as: (1) benchmark comparisons; (2) field evaluations of community structure or reproduction; and (3) toxicity bioassays. (Suter et al. 1999, Jones et al. 1999)

EPA simply failed to collect the information required to implement the weight-of-evidence approach properly. For at least a decade, the "sediment quality triad" approach (Chapman et al. 1997) has been recognized in assessments of effects of chemicals on benthic invertebrate communities. The triad approach has been used in other large-scale ecological risk assessments, including both the Clinch River assessment and the assessment performed for the Clark Fork River, Montana (Canfield et al. 1994). EPA did not collect the data needed for such an examination. Similar concepts should have been used to evaluate all receptors of interest on the Hudson River. For example, rather than limiting evaluation of birds to literature-based TQ comparisons, multiple lines of evidence for effects on bird populations can be generated through quantitative field studies of reproductive success, density and diversity. Likewise, site-specific, field based community structure and reproductive studies on small mammals are relatively straightforward to execute and would support a true evaluation of multiple lines of evidence.

Both EPA's Hudson River Assessment and the Clinch River assessment addressed risks of sediment-derived PCBs to benthic macroinvertebrates, fish, birds, and mammals. However, far more information was used in the Clinch River ERA (Table 1). Five independent lines of evidence were developed for fish, three were developed for benthic invertebrates, and two were developed for piscivorous birds and wildlife. Ample time was available for EPA to perform similar studies, however, virtually no ecological data beyond those available for the Phase I assessment were collected.[3]

3.0 The Ecological Risk Assessment is Excessively Conservative

EG-1.13

Even if, arguendo, the TQ approach can, in principle, provide information that is useful in a baseline ecological risk assessment, EPA's application of the TQ approach provides highly inflated risk estimates that are not useful in remedial decisionmaking. Both the exposure assessment and the effects assessment employ data, models, and assumptions that are, at best, appropriate for screening.

3.1 The Exposure Assumptions Employed by EPA Result in Overestimates of Actual Exposures

In contrast to assumptions used in the companion human health risk assessment, the BERA assumes that samples with non-detect values contained PCBs at levels equal to the detection limit. No explanation is provided to support this assumption. The use of detection limits as estimates of concentrations actually present is an acceptable practice in screening assessments, but is not acceptable for use in a baseline assessment. The guidance prepared by EPA Region 3 (available at http://www.epa.gov/reg3hwmd/risk/guide3.htm) states that the approach used in the BERA "always produces a mean concentration which is biased high, and is not consistent with Region 3's policy of using best science in risk assessments." Less conservative and more acceptable approaches either (1) assume that nondetects are present at one-half the detection limit, or (2) if EG-1.14

[3] Clearly, use of the best available science involves development of a report that is free of mathematical errors. While time constraints prevented completion of a detailed mathematical review of all calculations, Tables 3-26 through 3-65 appear to contain an important miscalculation that leads to erroneous predictions of egg concentrations and predicted risks to avian embryos. Tables 3-26 through 3-65 calculate the predicted egg concentrations (in mg/Kg) for each of the bird species by multiplying the total average daily dose (in mg/Kg/day) by the biomagnification factor (BMF) (apparently unitless per page 55 of the BERA). Clearly the units in this equation do not cancel out. Either the units of the BMF were inadvertently not reported in the text (and should be mg/kg/day) or it is necessary to convert the total average daily dose of a concentration in food (in mg/Kg) prior to applying the BMF. Because BMFs usually reflect the ratio of the concentration of a chemical in the diet to the concentration in tissue, the latter error is the more likely of the two. In that case, the reported egg concentrations in all of these tables are erroneous, and the resultant predicted risks to avian embryos are also reported in error.

the data set contains a high proportion of positive detects (typically, greater than 50%), use a statistical estimation procedure to estimate the distribution of concentrations below the detection limit. At low detection levels one must also recognize the contribution of "background" PCBs which do not originate from the site and will not be addressed by any conceivable site remedy.

The BERA fails to consider realistically the influence of migratory behavior, home range, and landscape pattern on the distribution of exposures within fish and wildlife populations. For example, all avian receptors are assumed to have a home range (modifying value of 1.0) consisting solely of the Hudson River. For species that can exploit wetlands or other nonriverine habitats, this assumption is excessively conservative when applied to entire populations rather than to maximally exposed individuals. Piscivorous birds, such as blue heron, would be expected to forage in ponds and tributary streams as well as in the Hudson River itself and to obtain a significant fraction of their diets from sources other than fish (Henning et al. 1999). Similarly, insectivorous birds, such as tree swallows, can be expected to obtain part of their diets from terrestrial insects and to exploit insect emergences from ponds and tributary streams. Biomagnification factors (BMFs) reported by Giesy et al (1995) and employed by EPA to predict egg concentrations of piscivorous birds are 4 to 15 times greater than site-specific BMFs for tree swallows. Mink live primarily in wetland areas, and raccoons are abundant in hardwood swamps, flood plain forests, fresh and salt marshes, mesic hardwood stands, cultivated and abandoned farmlands, and suburban residential areas (Kaufman 1982). EPA could and should have studied habitat availability and utilization by avian and mammalian receptor species. **EG-1.15**

Anadromous and semianadromous fish species, such as striped bass and white perch undergo complex seasonal migrations that limit their exposures to PCBs. Individuals of both species range widely throughout the lower Hudson, and, in the case of striped bass, along the Atlantic coast from North Carolina to Maine. NYSDEC's data have consistently shown that the adult striped bass with the highest PCB tissue concentrations are collected in the vicinity of the Federal dam at Troy. Secor and Baker (1999) have shown that these fish are predominantly males that remain in freshwater for most or all of their lifetimes. Concentrations of PCBs in these fish are not representative of concentrations found in spawning females (or most Hudson striped bass) which **EG-1.16** are migratory and have much lower exposures to PCBs.

Differences in professional judgment regarding specific exposure parameter values can often be resolved through the use of probabilistic analyses, such as Monte Carlo analysis. By using distributions to represent the full range of values for exposure, both the most extreme and the most likely values are incorporated into the assessment, to a degree commensurate with the actual distribution in the population. Such an approach would have been far more scientifically defensible than the use of only the most conservative exposure assumptions in a deterministic analysis. EG-1.17

3.2 The Effects Assessment Relies on Excessively Conservative TRVs and Criterion Values

EPA's approach to developing TRVs (pp. 79-80) is unnecessarily conservative and, in many cases, results in the use of TRVs that are many times lower than the lowest concentration or dose ever observed to affect exposed organisms. The approach develops a single value rather than a range of values for each receptor species. In all cases where studies are not available of the taxonomic family or order of interest, the lowest applicable No Observed Adverse Effect Level (NOAEL) is used to define the TRV. NOAELs are appropriate for screening because they define a dose or exposure concentration below which no effects should occur; they are inappropriate for baseline assessments because they do not define a concentration or dose above which effects are likely. The approach used in the Clinch River assessment (Sample and Suter 1999) would be more appropriate for a baseline assessment. In the Clinch River assessment, NOAELs and Lowest Observed Adverse Effect Levels (LOAELs) for each receptor species were used to define ranges of exposures associated with negligible (dose lower than NOAEL), possible (dose exceeds NOAEL), and probable (dose exceeds LOAEL) effects on individual organisms. EG-1.18

The TRVs used by EPA to address risks to fish are whole body concentrations ranging from 0.5 mg/Kg to 15 mg/Kg. These values are inconsistent with values developed in two recent reviews of the literature on toxicity of PCBs to fish. NOAA (1999b) performed a review of the literature on reproductive, developmental, and immunotoxic effects of PCBs in fish. This review was published in March 1999 and is cited in the BERA (NOAA 1999a). According to NOAA's EG-1.19

evaluation of the toxicity of Aroclor 1254, the threshold for occurrence of physiological and biochemical changes related to reproduction in adult fish is a liver concentration of approximately 25 ppm (equivalent to a whole-body concentration of approximately 12.5 ppm). This review implies that a valid screening benchmark for Aroclor 1254 and related PCB mixtures in fish tissue (whole body) would be no lower than 12.5 ppm. Actual reductions in egg production or viability may require even higher exposure levels. Using actual reductions in survival or reproduction rather than physiological and biochemical endpoints, Niimi (1996) concluded that the weight of evidence from numerous species indicated that adverse reproductive effects are typically observed at whole body concentrations >100 mg/kg wet weight. Similarly, adverse effects on growth and survival of the progeny have generally been observed at whole body concentrations > 50 mg/kg wet weight. Only one of the TRV's used by EPA to calculate TQs is greater than NOAA's screening benchmark (spottail shiner, 15 mg/kg).

Similar reviews of the PCB toxicity literature are unavailable for birds and mammals. However, EPA's use of the lowest measured NOAEL, rather than the full range of available NOAELs and LOAELs, is an excessively conservative approach to assess the effects of PCBs on exposed species and does not provide a realistic description of risk. In many cases, laboratory studies provide the basis for the only TRV derived despite the many limitations in the ability of laboratory studies to simulate actual field conditions. Laboratory studies generally overestimate potential adverse effects. In the wild, organisms are exposed to widely fluctuating dose rates, temperatures, environmental stresses, competition, and food availability. EG-1.20

Regardless of the relative merits of field and laboratory study designs, we disagree with EPA's selection of studies on which to base TRVs (always the most conservative study, unless a study is available on a species of the same taxonomic family or order), as well as its interpretation of the studies selected. When sufficient data are available from both laboratory and field-based studies to generate TRVs, the BERA provides no information as to which TRV (laboratory or field-based) is actually used to predict risks. These multiple sources of conservatism are further compounded by the use of several ten-fold uncertainty factors to account for interspecies differences and subchronic-to-chronic exposure durations. EG-1.21

A notable problem with the EPA's Assessment relates to the use of gallinaceous birds (e.g., chickens) to evaluate the effects of PCBs on all avian receptors in the Hudson River region. This assumption is overly conservative; other data sources should be considered. For example, the site-specific tree swallow studies eliminate the need to predict PCB effects from the extrapolation of laboratory data. As a second example, in the analysis of mallards, three studies were examined for PCB toxicity data, and the TRV was based on the study with the lowest NOAEL; when there is more than one equally valid NOAEL, the highest value should be selected to provide the most realistic estimate of the effects threshold. In a third example, the analysis of great blue herons included the addition of an uncertainty factor due to the relatively short length of the study (Scott 1977). Longer term studies than the one chosen are available and would eliminate the need for an uncertainty factor. Fourth, despite the availability of field data on bald eagles and related predatory birds (Elliot et al. 1996), the TRV was developed using data from chicken studies. A far more representative laboratory study was conducted on screech owls (McLane and Hughes 1980), which are similar in feeding guild and taxonomy although they are not in the same family or order as bald eagles. **EG-1.22**

For bats and raccoons, EPA based TRVs on a laboratory study of rats conducted by Linder et al. (1974), despite the many limitations associated with the use of a laboratory species to evaluate wild species. EPA should have based TRVs for bats and raccoons on studies of wild species, such as Linzey (1987) and McCoy et al. (1995), which would not be subject to such extreme extrapolations. Even if there were a defensible basis for using Linder et al. (1974) instead of Linzey (1987) or McCoy et al. (1995), the uncertainty factor used by EPA to derive a TRV is overly conservative. For example, Sample et al. (1996) used Linder et al. (1974) to derive a NOAEL of 0.4 mg/kg-d, a value more than ten-fold higher than EPA's TRV of 0.032 mg/kg-d for bats. **EG-1.23**

For mink, the most scientifically defensible basis for a TRV is provided by Auerlich and Ringer (1977), rather than Heaton et al. 1995), which was used by EPA. Auerlich and Ringer (1977) fed mink Aroclor 1254 at multiple dose groups over a 4.5 month period. This period included critical **EG-1.24** life stages, so that no subchronic-to-chronic uncertainty factor should be necessary. Heaton et al.'s (1995) field study was confounded by the concurrent exposures of mink to other chemicals, and was of shorter duration than Auerlich and Ringer (1977) (4 months vs. 4.5 months). The most defensible TRV for mink would be based on Auerlich and Ringer (1977), without the factor-of-ten adjustment.

EPA's use of water quality criteria is similarly over conservative. As described by EPA (EPA 1986), water-quality criteria for the protection of aquatic life are intended to protect 99% of the individuals in 95% of the species exposed to a toxic chemical. Chemicals present at concentrations lower than the criterion clearly should not harm any exposed population. Concentrations above the criterion, however, do not necessarily imply that any of the exposed populations at a site are being adversely affected. **EG-1.25**

4.0 The Sediment Effect Concentrations (SECs) are not Reasonable Estimates of PCB Toxicity to Benthic Invertebrates Either Individually or As a Population

Because the BERA claims to address benthic invertebrates at the population level, we address the SECs used as TRVs for benthic invertebrates in greater detail. The deficiencies discussed in this section are illustrative of problems found in many of the other TRVs used in EPA's Assessment. EG-1.26

EPA relies on SECs developed by NOAA (1999b) as TRVs for benthic invertebrates. These SECs are used in assessments of the likelihood that PCBs are impacting benthic invertebrate populations, fish populations, threatened and endangered species and the ability of particular habitats to support sustainable, healthy populations of biota. The presumption is that exceedance of SEC values is evidence that some unspecified toxic effect is occurring to benthic invertebrates and that this direct effect results in secondary effects to fish, threatened and endangered species and other organisms.

This presumption lacks scientific merit for two reasons:

- The SEC values have no causal basis.

- Direct relationships between benthic community productivity and the productivity of higher trophic level populations cannot be demonstrated.

The SECs developed by NOAA (1999b) and termed "Consensus-Based" SECs are the geometric means of pre-existing SECs developed from correlating measurements of sediment chemical concentrations and the results of sediment bioassay tests. The meaning and utility of the pre-existing SECs is the subject of considerable scientific debate (for example, see the discussion by O'Connor in the January 1999 issue of SETAC News). The principal arguments center on the lack of consideration of cause and effect. Absent an understanding of the agents responsible for observed toxicity and in the presence of the typical co-variation among sediment contaminants, it is inappropriate to use simple bivariate correlations to ascribe threshold concentrations for individual chemicals. This difficulty is compounded by the aggregation of data from sites that differ physically and in the suite of chemicals present. As indicated by Swartz and DiToro (1997) "correlation of chemical concentration and biological response only establishes potential exposure. The effects assessment must be based on independent evaluation of causality." This limitation makes SEC values appropriate for use only in the problem formulation stage of an ecological risk assessment (Chapman and Mann, 1999).

Authors of several of these methods have warned against their use as risk assessment tools. Long and Morgan (1991) and Long et al. (1995), who developed the current effects-range approach, have clearly stated the limits of these Sediment Quality Values (SQVs) in their primary publications. Long et al (1995): "The numerical guidelines should be used as informal screening tools in environmental assessments. They are not intended to preclude the use of toxicity tests or other measures of environmental effects." Like these scientists, Cubbage et al. (1997) inform the reader that their SQV (the PAET used by NOAA 1999b) have not been peer reviewed. These authors warn managers that the freshwater SQV "delineate a level below which biological effects are unlikely to occur...stations above these levels could be tested with bioassays to substantiate implied deleterious effects." The authors of the TEL/PEL values promulgated by the Ontario Ministry of the Environment (Smith et al. 1996) repeat this warning for their users: "[T]he guidelines are intended to be used in Canada as an indication that no adverse effects on aquatic organisms are expected if the measured concentrations of substances in sediments are equal to or lower than the recommended sediment quality guidelines. In contrast, measured concentrations of substances in sediments that are higher than the recommended sediment quality guidelines indicate only that there is the potential for adverse biological effects to occur." The use of these values to derive SECs and the subsequent use of SECs to *predict biological effects* in a baseline risk assessment is completely inconsistent with the intent of the basic SQVs as stated by their authors.

A key problem in the SEC approach is that no-effects data are not properly considered. The distribution of no-effects data is important because it is only in no-effects samples that *all* chemicals must be present below toxic levels, including the chemical of interest. Therefore the no-effects distribution defines a concentration range over which the chemical of interest assuredly has no toxic effect. The converse, however, is not true of the distribution of effects data; the effects distribution does not define a concentration range over which the chemical of interest assuredly produces a toxic effect because virtually all environmental samples are mixtures of chemicals. Where an effect is observed in a chemical mixture, a variety of measured and unmeasured chemicals could be responsible for the observed toxic effect: the effect cannot be positively attributed to any single chemical. Measured toxic effects have a high probability of being attributed to the wrong chemical(s).

The authors of the NOAA (1999b) SECs argue that the central tendency of the various pre-existing SECs for PCBs "reflect(s) causal rather than correlative effects ... and account for the effects of contaminant mixtures." There is no logical basis underlying the idea that causation exists in the central tendency of numbers that do not reflect causation. If ten researchers independently demonstrate and quantify correlations between the frequency of skin cancer and annual average air temperature, the central tendency of those studies does not provide evidence that skin cancer is caused by exposure to high air temperature. Correlation does not define causation, and multiple studies of correlation cannot overcome this fact. (See Appendix B).

The pre-existing SEC values are mostly based on data from sediments for which PCBs have not been shown to be the dominant or only contaminant of concern. For example, of the nine sites used to develop the Ingersoll et al. (1996) SEC values, only one (Waukegan Harbor) or possibly two (Saginaw River) could even be considered as primarily dominated by PCBs. The others contain substantial quantities of metals, PAHs, and/or petroleum hydrocarbons. As a result, the data set used to assign SEC values includes observations of toxicity at relatively low PCB concentrations. However, it is incorrect to infer that PCBs cause toxicity if these low concentrations are exceeded. Because the BERA does not present an evaluation of the studies on which each SEC is based, it fails to demonstrate that the SEC values have any meaning with regard to PCB toxicity. In fact, a recent analysis of field data (Anid and Connolly 1998), has shown that such evaluations can significantly alter the interpretation of SEC values. This analysis suggests that existing SEC values significantly over-estimate PCB toxicity because of the co-variation of PCBs and other chemicals.

The significant overstatement of PCB toxicity by the SEC values is illustrated by the spiked sediment toxicity study of Swartz et al. (1988) that NOAA (1999a) improperly uses as a validation of the SEC values. This study used sediment with a TOC content of about 0.25 percent. Most sediments contain a TOC content in the range of 1 to 4 percent. The fine sediment of the Thompson Island Pool has an average TOC of about 2 percent. Based on the relationship between the bioavailability of organic chemicals in sediments and sediment TOC that forms the basis of EPA's Sediment Quality Criteria (USEPA, 1993), the Swartz et al. toxicity results would have to be adjusted by about a factor of 8 to be applicable to the Hudson River. Thus, the applicable LC_{50} and LC_{10} values for comparison to the SEC values are 86 and 54 mg/kg DW. For argument sake, accepting the acute-to-chronic ratio of 11 cited in NOAA (1999b), PCBs would not *begin* to cause chronic toxicity to amphipods until concentrations exceeded about 8 mg/kg DW. In comparison, the NOAA (1999b) SEC values indicate the threshold is 0.04 mg/kg and that extreme effects are expected if the sediment concentration exceeds 1.7 mg/kg.

5.0 EPA's Assessment Should Not Rely on the TEQ Approach Because It Is Not Sufficiently Developed and Has Not Been Applied Properly

The Toxicity Equivalence (TEQ) approach converts concentrations of "dioxin-like" organic chemicals to equivalent concentrations of dioxin. EPA has used the TEQ approach as a method of assessing the risks of PCB exposure to fish and wildlife, in spite of its substantial limitations. The TEQ approach provides only order of magnitude estimates of toxicity. Moreover, the analytical methods used by EPA cannot accurately measure PCB congeners in fish or animal tissue. EPA has inappropriately handled non-detect readings of these congeners. **EG-1.27**

In addition, the TEQ calculations in the BERA are not well-documented. The procedure for estimating individual congener concentrations and TEQs is unclear and poorly justified. The report does not provide enough information to permit one to recheck the calculations. For these reasons, EPA's presentation of its analysis is markedly below the standard of "best practices." **EG-1.28**

5.1 The TEFs are Improperly Applied

EPA seems to consider the use of total PCB and PCB TEQ as equally valid means of assessing risks, regardless of the species and endpoint being evaluated. Given its current state of development, the use of the TEQ approach should be considered as a screening level filter rather than as a primary assessment approach. This reflects the cautions issued by the scientists who have contributed to the development of the TEQ approach for PCBs (Van den Berg et al. 1998; Tillit et al. 1991; Safe 1990, 1994). The TEFs used to convert coplanar congener concentrations to dioxin-equivalents are, at best, order-of-magnitude approximations useful primarily for screening purposes. **EG-1.29**

The stringent data requirements and the lack of a comprehensive toxicological database currently preclude the routine application of the approach to all receptor species. With the possible exception of mink, insufficient information is available concerning the species addressed in the BERA for TEQs to provide defensible risk estimates. For example, results of field studies for fish indicate that expression of PCB exposure in TEQs does not improve correlations between exposure and adverse effects (Giesy et al. 1994). In reviewing the applicability of the TEQ approach to Hudson River fish species, NOAA (1999a) concluded that "it is currently not possible to evaluate the risk to Hudson River fish larvae from exposure to coplanar PCBs using the TEQ method."

As a second example, the calculations of some TEFs are based on enzyme induction studies, notably BZ#81, one of the two most potent TEQ congeners to avian species (Van den Berg, et al. 1998). However, Yorks, et al. (1998) clearly demonstrated the lack of induction in tree swallows when dosed with PCBs and likewise observed a lack of metabolic activity in field studies, thus negating the TEQ approach for this species.

5.2 The Analytical Data are Inadequate

The analytical data for individual congeners in biota are inadequate for calculating TEQs. EG-1.30
In particular, the practical quantitation limit (PQL) for BZ#126 was too high to permit reliable measures of its concentration in biological samples. These TEQ values are based on non-detect concentrations. EPA assigned the PQL to concentrations of BZ#126 below the quantitation limit and then used those values in the risk assessment. This deficiency is critical to the assessment because, based on EPA's calculations, BZ#126 comprises from 52 to 85 percent of the PCB TEQ in fishes from the Hudson River (BERA Table 3-1). Furthermore, the BERA implies that this overestimate of BZ#126 is compensated for by the fact that BZ#81 was not measured. The BERA provides no justification for its unusual assumptions but states that the magnitude of error associated with the omission of BZ#81 and the use of the detection limit for BZ#126 is within an order of magnitude at most. There is no basis for this conclusion. The end result of this assumption is that the TEQ-based risk assessments are driven by non-quantified concentrations of BZ#126.

6.0 EPA Has Misstated the Results of Field Studies

EPA considered two site specific field studies as part of the BERA: the USFWS tree swallow reproduction study (Secord and McCarty 1997, McCarty and Secord 1999a, 1999b), and EPA's study of the benthic macroinvertebrate community in Thompson Island Pool (Appendix H of the BERA). EPA concluded that both studies support a finding of significant risks related to PCB exposures. Neither study supports this conclusion. EG-1.31

6.1 The Tree Swallow Study Did Not Demonstrate PCB-Related Reproductive Effects

EG-1.32

According to EPA's assessment, McCarty and Secord (1999a) observed "decreased reproductive success relative to reference areas and the occurrence of unusual parental and/or nesting behavior relative to reference areas" (BERA at p. 175). EPA states that "the behavioral endpoints have been shown to be statistically related to PCB exposures." EPA's inference from these results is that "PCB exposures may have significant effects on tree swallow nesting behavior. Alterations in behavior may also be reflected in changes in reproductive success of this species over time."

These statements are misleading. McCarty and Secord have been unable to demonstrate a dose-response relationship between tree swallow reproduction and PCB concentrations. The differences in reproductive parameters between the Ithaca and Hudson River tree swallow population are very likely due to the natural and temporal variation of these parameters between populations. The behavioral responses, although statistically related to PCB doses, are not correlated with reproductive success.

The theory of a relationship between PCB contamination and reproductive effects in tree swallows is not supported by the 1995 data set (McCarty and Secord 1999a). No significant differences in reproductive success of tree swallows nesting on the Hudson River in 1995 were found when comparing to the Ithaca reference data. Reproductive success was not related to PCB dose in either data set. The behavioral endpoints mentioned in the Hudson River BERA and measured by McCarty and Secord (1999b) are nest quality metrics; these metrics were not correlated with reproductive success.

Problems with reference site selection severely compromise all of McCarty and Secord's results. Both reference sites chosen for comparison with the Hudson River sites are inadequate. The original reference site was located on Champlain Canal. However, the tree swallow eggs at the Champlain reference site were determined to contain high concentrations of PCBs (Secord and McCarty 1997). Moreover, space limitations at this site forced the researchers to place tree swallow nest boxes much closer together (10-15 meters apart) than at all other Hudson River sites (30 meters apart) (Secord and McCarty 1997). Robertson and Rendell (1990), Muldal et al. (1985) and others have found that tree swallows prefer distantly spaced nests; adverse effects of close spacing confound effects of PCBs on tree swallows nesting at this site.

Data collected at a site in Ithaca during McCarty's thesis studies at Cornell University (McCarty 1995), were chosen as an alternative reference data set. However, the Ithaca study was conducted prior to 1994, and very limited information is provided regarding the site. Although the field methods used at Ithaca are reported to be the same as those used at the Hudson River sites, other factors that affect reproductive success, such as weather conditions, habitat characteristics, and tissue residue levels, have not been documented. The dissimilarities (i.e., the years sampled and habitats represented) between the Ithaca and Hudson River sites greatly weaken the already ambiguous conclusions that can be drawn from these studies.

6.2 The Benthic Macroinvertebrate Study Did Not Demonstrate PCB-Related Effects

EG-1.33

In presenting conclusions from the benthic macroinvertebrate study documented in Appendix H to the BERA, EPA states that "[t]he analysis shows a reduced macroinvertebrate community, indicating the potential for risk above regional conditions due to site-related influences" (BERA at p. 167). This statement contradicts the statement in the Risk Characterization section (p. 121) that the benthic invertebrate community analyses could not distinguish any clear effects from PCBs in the Upper or Lower Hudson River (BERA at 121).

In fact, EPA's benthic macroinvertebrate study did not employ a design capable of separating effects of PCBs from effects of environmental variables such as site depth, grain size, total organic carbon (TOC), and other potentially toxic chemicals. The results presented in Appendix H, Table H-6 show that concentrations of PCBs, TOC, cadmium, chromium, lead, and mercury all co-vary at the five stations studied. Hence, although benthic community metrics differ between Stations 5 and 7 (higher PCB concentrations) vs. Stations 3, 4, and 6 (lower PCB concentrations), it is not possible to infer that PCBs are responsible for the differences in macroinvertebrate community metrics between these two groups of sites.

These results flatly contradict claims made by EPA, (pp. ES-6, 167) that PCBs are adversely affecting benthic macroinvertebrate populations in the Upper Hudson River.

7.0 Available Population and Community Data Conflict With EPA's Conclusions

Many studies of the biological resources of the Hudson River and Valley have been carried out over the last 25 years. As a result, many sources of field data on the status of benthic communities and of fish and wildlife populations are available. EPA failed to use these data. They generally demonstrate the presence of healthy populations and communities in the upper and lower Hudson in spite of exposures to PCBs.

7.1 Benthic Macroinvertebrates

No effects of PCBs have been seen in Hudson River benthic macroinvertebrate communities as evidenced by the increase in abundance of pollution-intolerant filter feeders (NYSDEC 1993) over a 25 year period. **EG-1.34**

7.2 Fish

Fish population data are available for the Lower Hudson River from surveys conducted by NYSDEC and the Hudson River utility companies. Abundance trends for striped bass are reported to NOAA and are used in stock assessments performed by the Atlantic States Marine Fisheries Commission. These data, reported in the 1998 striped bass stock assessment (NMFS 1998a) clearly show (Figure 1) that the abundance of young-of-the-year striped bass has remained stable since 1980 and that the abundance of the spawning stock in the Hudson River has increased over the same period. There is no evidence of any adverse effects due to PCB exposure. Following the decline of the coastal striped bass stock in the mid-1970s, the National Marine Fisheries Service and the U.S. Fish and Wildlife Service investigated the possible causes of the decline. Those agencies did not find that PCBs posed a threat to the striped bass population and they concluded, given the restrictions on striped bass fishing in the Hudson, that the "Hudson River striped bass stock is likely to increase to near the maximum level supportable by that ecosystem." (Atlantic States Marine Fisheries Commission, 1990). **EG-1.35**

EPA also neglected the documented positive trends in the populations of shortnose sturgeon in the Hudson River. Two studies have generated estimates of the adult shortnose sturgeon population using mark-recapture methods. Dovel et al. (1992) used recapture counts from 1975 to 1980 to estimate the shortnose sturgeon population size in the Hudson River at 13,000 adult fish. In 1995, Bain et al. used comparable methods to estimate the shortnose sturgeon population at 38,024 (standard error = 7,199). According to Bain et al. (1995), both studies probably underestimate the sturgeon population because the samples did not cover the full range of habitat used by sturgeon in the Hudson River. Nonetheless, these studies suggest that the numbers of shortnose sturgeon are increasing in the Hudson River.

7.3 Wildlife

NYSDEC has been monitoring the winter use and breeding activity, tissue contaminant concentrations, and reproduction of bald eagles in New York State and the Hudson River area for many years. Statewide, there are approximately 45 breeding pairs, and the recent wintering population includes 200–250 individuals (Nye 1999, pers. comm.). Since 1990, bald eagle productivity in the state has ranged from 0.55 to 1.3 fledglings per occupied territory (NYSDEC 1999). Production has been greater than 0.7 (the minimum for a stable population [Sprunt et al. 1973]) in eight out of nine years, and greater than 1.0 in six out of nine years (the rate assumed for a healthy population by USFWS (1997). In 1996, 37 young (including one introduced chick) were fledged in the state; 43 eaglets (including 3 introduced chicks) fledged in 1997; and 40 eaglets (including 1 introduced chick) fledged in 1998. **EG-1.36**

Nesting attempts in three bald eagle territories on the Hudson resumed in 1992, and fledglings were successfully produced in 1997 (Nye 1999, pers. comm.). Four eaglets were fledged from Hudson River nests in 1998.

In addition to these reproductive data, the collection of prey from eagle nests by NYSDEC provides empirical, site-specific information on the diets of bald eagles that should have been used by EPA to improve the realism of its exposure model. Peter Nye of NYSDEC found remains of grebes, eels, pickerel, bullhead, herring and carp in eagle nests. Information on the PCB concentrations in some of these prey, including bullhead and eels, are available (Secor 1997). Remaining samples are being stored with the plasma and egg samples for analysis of organochlorines. EPA should have worked with NYSDEC to measure the PCB concentrations in these samples. Measured PCBs in eggs could have been used to calibrate the rough estimate provided by the biomagnification factor (BMF) approach in EPA's risk assessment, improving the reliability of the BCF model, and could have been considered as an indicator of exposure on their own. Measured PCBs in prey items could have been used to calibrate the food web exposure model.

Other data for bird populations that relate directly to EPA's risk models, are available and should have been included in EPA's analysis. For example, data show that mallards are "demonstrably secure" throughout the New York Bight watershed and are "widespread, abundant and secure in the state of New York" (USFWS 1997). NYSDEC (1997) reports that, on the basis of breeding surveys, the mallard population using the Hudson River estuary is "stable to increasing." Mid-winter counts of waterfowl show generally increasing numbers of mallards and other species with a peak in 1995 of more than 16,000 birds (NYSDEC 1997). North American Breeding Bird Survey data (analyzed in Sauer et al. 1997) indicate that populations of mallard ducks have significantly increased at a rate of 5.7 percent per year within the region that includes the Hudson River (i.e., the Ridge and Valley Province) since 1966. EG-1.37

8.0 Conclusion

The BERA is significantly flawed. It does not reflect best scientific practice, is excessively conservative, and is grossly insufficient in determining the need for or selecting a remedy. These weaknesses make the assessment of little use for the remedial decision maker. EG-1.38

- The BERA is deficient because its assessment endpoints inappropriately focus on risks to individuals, not populations or communities, and it is insufficiently based on observed exposures and effects measured in well-designed, site-specific studies. EPA had ample time – nearly 10 years – to collect the necessary data to perform a defensible and valid ecological risk assessment.

- The BERA ignores a wealth of valuable site-specific data and misrepresents the two site-specific studies on which it relies. Available population and community data, in fact, contradict the BERA's conclusions.

- The BERA fails to use the weight-of-evidence approach in a sound manner. Although EPA claims to have examined several lines of evidence, most of its assessment endpoints are addressed using only one line of evidence: comparison of measured or modeled exposure concentrations to generic, non-specific toxicity benchmarks. Other, more probative lines of evidence are ignored.

- The BERA contains a number of assumptions and approaches that are excessively conservative. These include, employing generic screening values such as TRVs and SEC as predictors of site-specific risk; treating non-detects as if they show the presence of a chemical; and mistaken and unrealistic assumptions about diet composition, food consumption rates and habitat utilization. The BERA should not have relied on the TEQ approach, which is insufficiently developed and was misapplied.

- The BERA should not have used EPA's food-chain bioaccumulation model, which is flawed, undergoing changes and has not yet been peer-reviewed.

EPA should ignore this assessment when making a remedial decision for the Site.

9.0 REFERENCES

Anid P.J., Connolly J.P. 1998. Limitations and uncertainties of sediment effects concentrations (SECs) in the evaluation of sediment toxicity and chemistry. Presentation at the SETAC 19th Annual Meeting, Charlotte NC. November 19, 1998.

Atlantic States Marine Fisheries Commission (ASMFC). 1990. Source Document for the Supplement to the Striped Bass FMP, Amendment #4

Aulerich, R.J., and R.K. Ringer. 1977. Current status of PCB toxicity to mink, and effect on their reproduction. Arch. Environ. Contam. Toxicol. 6:279–292.

Bain, M.B., S. Nack, and J.G. Knight. 1995. Population status of shortnose sturgeon in the Hudson River. Phase 1 Project Report to the U.S. Army Corps of Engineers. North Atlantic Division, New York, New York.

Barnthouse, L.W. 1993. Population level effects. Ch. 8 in Suter, G.W. II (ed.) Ecological Risk Assessment Lewis Publishers, Chelsea, Mirhison

Canfield, T.J., N.E. Kemble, W.G. Brumbaugh, F.J. Dwyer, C.G. Ingersoll, and J.F. Fairchild. 1994. Use of benthic invertebrate community structure and the Sediment Quality Triad to evaluate metal-contaminated sediment in the upper Clark Fork River, Montana. Environmental Toxicology and Chemistry 13(12); 1999-2012

Carr RS, Long ER, Windom HL, Chapman DC, Thursby, G, Sloane GM, Wolfe DA. 1996. Sediment quality assessment studies of Tampa Bay, Florida. Environ. Tox. Chem. 15:1218-1231.

Chapman PM, Mann GS. 1999. Sediment quality values (SQVs) and ecological risk assessment. *Marine Pollution Bulletin.* 38:339-344.

Chapman, P.M., B. Anderson, S. Carr, et al, 1997. General Guidelines for Using the Sediment Quality Triad. Mar. Pollut. Bull. 34:368-372.

Cubbage J, Batts, D. Briedenbach S. 1997. Creation and analysis of freshwater sediment quality values in Washington State. Environmental Investigations and Laboratory Services Program. Washington Department of Ecology. Olympia WA.
Carlson 1986

Dovel, W.L., A.W. Pekovitch, and T.J. Berggren. 1992. Biology of the shortnose sturgeon (*Acipenser brevirostrum* Lesueur, 1818) in the Hudson River estuary, New York. pp. 187-216. In: Estuarine Research in the 1980s. Hudson River Environmental Society 7th Symposium on Hudson River Ecology. C.L. Smith (ed). State University of New York Press, Albany, NY.

Elliot, J., Norstrom, F., Lorenzen, A., Hart, L., Philibert, H., Kennedy, S., Stegman, J., Bellward, G., and K. Cheng. (1996). Biological effects of polychlorinated dibenzo-p-dioxins, dibenzofurans, and biphenyls in bald eagels, *Haliaeetus leucocephalus,* chicks. *Environ. Toxicol. and Chem.*

15(5): 782-793

EPA 1986. Quality criteria for water. EPA/440/5-86/001. Office of Water, U.S. Environmental Protection Agency, Washington, D.C.

EPA. 1991. Phase 1 Report - Review Copy Interim Characterization and Evaluation, Hudson River PCBs Reassessment RI/FS. Volume 1. EPA No. 013-2N84. U.S. Environmental Protection Agency, Washington, DC.

EPA. 1992a. Final Phase 2 Work Plan and Sampling Plan, Hudson River PCBs Reassessment RI/FS. EPA No. 013-2N84. U.S. Environmental Protection Agency, Washington, DC.

EPA. 1993. Wildlife Exposure Factors Handbook. U.S. Environmental Protection Agency, Office of Research and Development, Washington, DC. EPA/600/R-93/187a,b. December.

EPA. 1997. Ecological Risk Assessment Guidance for Superfund: Process for Designing and Conducting Ecological Risk Assessments. EPA 540-R-97-0CS.

EPA. 1998a. Ecological Risk Management Principles for Superfund Sites. Draft. OSWER Directive 9285-7-28P.

EPA. 1998b. Guidelines for Ecological Risk Assessment. EPA/630/R-95/002F.

Exponent. 1998. Macroinvertebrate community and fish stomach content analysis for sampling conducted in 1997. Data report. Exponent, Bellevue, WA.

Exponent. 1998b. Data report, macroinvertebrate communities and diets of selected fish species in the Upper Hudson River. Draft. Volumes I and II. Prepared for General Electric Company, Albany, NY. Exponent, Bellevue, WA.

Gale RW, Huckins JN, Petty JD, Peterman PH, Williams LL, Morse D, Schwartz TR, Tillet DE. 1997. Comparison of the uptake of dioxin-like compounds by caged channel catfish and semipermeable membrane devices in the Saginaw River, Michigan. Environ. Sci. Technol. 31:178-187.

Giesy, J.P., J.P. Ludwig, and D.E. Tillitt. 1994. Chapter 9, dioxins, dibenzofurans, PCBs and colonial, fish-eating water birds.

Giesy, J.P., W.W. Bowerman, M.A. Mora, D.A. Verbrugg, R.A. Othoudt, J.L. Newsted, C.L. Summer, R.J. Aulerich, S.J. Bursian, J.P. Ludwig, G.A. Dawson, Kubiak T.J., D.A. Best and D.E. Tillitt. (1995). "Contaminants in fishes from Great Lakes-influenced section and above dams of three Michigan rivers: III. Implications for Health of Bald Eagles." Archives of Environmental Contamination and Toxicology. 29:209-271

Heaton, S.N., S.J. Bursian, J.P. Giesy, D.E. Tillitt, J.A. Render, P.D.

304449

Jones, D.A. Verbrugge, T.J. Kubiak and R.J. Aulerich. (1995). "Dietary exposure of mink to carp from Saginaw Bay, Michigan: I. Effects on reproduction and survival, and the potential risk to wild mink populations." Archives of Environmental Contamination and Toxicology 28:334-343.

Henning, M. H., N.M. Shear Weinberg, N.D. Wilson, and T. J. Jamuzzi. 1999. Distrubtions for key exposure factors controlling the uptake of xenobiotic chemicals by Great Blue Herons (Ardea herodius) through ingestion of fish. *Human and Ecological Risk Assessment* 5(1):125-144.

Hoke RA, Giesy JP, Zabik M, Unger M. 1993. Toxicity of sediments and sediment pore waters from the Grand Calumet River-Indiana Harbor, Indiana area of concern. Ecotox. Environ. Safety. 26:86-112.

Ingersoll CG, Haverland PS, Brunson EL, Canfield TJ, Dwyer FJ, Henke CE, Kemble NE, Mount DR, Fox RG. 1995. Calculation and evaluation of sediment effect concentrations for the amphipod *Hyalella azteca* and the midge *Chironomus riparius*. J. Great Lakes Res. 22:602-623.

Jones, D.S., L. W. Barnthouse, G.W. Suter II, R. A. Effroymson, J.M. Field, and J.J. Beachamp. 1999. Ecological risk assessment in large river-reservoirs. 3-Benthic Invertebrates. Environmental Toxicology and Chemistry 18(4): 599-609

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

Long, ER, Morgan LG. 1991. The potential for biological effects of sediment-sorbed contaminants tested in the National Status and Trends Program. NOAA Technical Memorandum NOS OMA 52, Seattle WA.

Linder, R.E., T.B. Gaines, R.D. Kimbrough. (1974). "The effect of polychlorinated biphenyls on rat reproduction." Food Cosmet. Toxicol. 12: 63-77.

Long ER, MacDonald DD, Smith SL, Calder FD. 1995. Incidence of adverse biological effects within ranges of chemical concentrations in marine and estuarine sediments. Environ. Management. 19:81-97.

McCarty, J.P., 1995. Effects of Short-Term Changes in Environmental Conditions on the Foraging Ecology and Reproductive Success of Tree Swallows, Tachycineta bicolor. Doctoral Thesis. Cornell University

McCarty, J.P., Secord, A.L., 1999a. Reproductive ecology of tree swallows (Tachycineta bicolor) with high levels of polychlorinated biphenyl contamination. Environ. Toxicol. Chem. 18(7):1433-1439

McCarty, J.P., Secord, A.L., 1999b. Nest-building behavior in PCB-contaminated tree swallows. Auk 116(1):55-63

McLane, M.A., and D.L. Hughes. 1980. Reproductive success of screech owls fed Aroclor® 1248. Arch. Environ. Contam. Toxicol. 9:661–665.

Menzie, C., M.H. Henning, J. Cura, K. Finkelstein, J. Gentile, J. Maughan, D. Mitchell, S. Petron, B. Potocki, S. Svirsky, and P. Tyler. 1996. Special report of the Massachusetts weight-of-evidence workgroup: a weight-of-evidence approach for evaluating ecological risks. Human Ecol. Risk Assess. 2(2): 277–304.

Moll RA, Jude D, Rossmann R, Kantak GV, Barres J, DeBoe S, Giesy J, Tuchman M. 1995. Movement and loading of inorganic contaminants through the lower Saginaw River. J. Great Lakes Res. 21:17-34.

Muldal, A., H.L. Gibbs and R.J. Robertson. 1985. Preferred nest spacing of an obligate cavity-nesting bird, the tree swallow. *The Cooper Ornithological Society* 87:356-363. 14424

National Oceanographic and Atmospheric Administration (NOAA). 1999a. Reproductive, Developmental and Immunotoxic Effects of PCBs in Fish: a Summary of Laboratory and Field Studies. Prepared for NOAA Damage Assessment Center, Silver Spring, MD. Prepared through Industrial Economics Inc. by E. Monosson. March, 1999.

National Oceanic and Atmospheric Administration. 1999. Development and evaluation of consensus-based sediment effect concentrations for PCBs in the Hudson River. Prepared by MacDonald Environmental Services Ltd. Ladysmith BC. March 1999.

Niimi, A.J. 1996. PCBs in aquatic organisms. In: Environmental Contaminants in Wildlife—Interpreting Tissue Concentrations. W.N. Beyer, G.H. Heinz, and A.W. Redmon-Norwood (eds). SETAC Special Publications Series, CRC Press, Lewis Publishers, Boca Raton, FL.

NMFS 1998a. 26th Northeast Regional Stock Assessment WorkshopL Stock Assessment Review Committee (SARC) Consensus Summary of Assessments. Northeast Fisheries Science Center Reference Document 98-03.

NMFS 1998b. Final recovery plan for the shortnose sturgeon (*Acipenser brevirostrum*). Prepared by the Shortnose Sturgeon Recovery Team for the National Marine Fisheries Service, Silver Spring, MD. 104 pages..

Nye, P. 1999. Personal communication (conversations with Johanna Salatas, Exponent, Boulder, CO, on April 21 and 29). New York Department of Environmental Conservation.

New York State Department of Environmental Conservation. 1993. Technical guidance for screening contaminated sediments. Albany NY. November 1993.

NYSDEC. 1993. 20 year tends in water quality of rivers and streams in New York state based on macroinvertebrate data, 1972-1992. New York State Department of Environmental Conservation, Albany, NY.

NYSDEC. 1997. HREMP Annual Report and State of the Hudson Report for Period 4/1/97-3/31/98. Albany, NY. 69 pp.

NYSDEC. 1999. New York State Bald Eagle Breeding Summary 1990–1998. New York State Department of Environmental Conservation, Division of Fish and Wildlife, Endangered Species Unit, Delmar, NY. Faxed pages received April 29, 1999.

Quantitative Environmental Analysis, LLC. 1999. PCBs in the Upper Hudson River. Report to the General Electric Company Corporate Environmental Programs, Albany NY.

QEA 1999. PCBs in the Upper Hudson River. Volume 2. A Model of PCB Fate, Transport and Bioaccumulation. Prepared for General Electric Co., Albany, New York.
Safe, S. 1990. Polychlorinated biphenyls (PCBs), dibenzo-p-dioxins (PCDDs), dibenzofurans (PCDFs), and related compounds: Environmental and mechanistic considerations which support the development of toxic equivalency factors (TEFs). Crit. Rev. Toxicol. 21:51-88.

Rendell, W.B. and R.J. Robertson. 1990. Influence of Forest Edge on Nest-Site Selection by Tree Swallows. *Wilson Bulletin* 102:634-644.14113

Safe, S.H. 1994. Polychlorinated biphenyls (PCBs): Environmental impact, biochemical and toxic responses, and implications for risk assessment. Crit. Rev. Toxicol. 24:87–149.

Sample, B.E., D.M. Opresko, and G.W. Suter, II. 1996. Toxicological benchmarks for wildlife: 1996 revision. ES/ER/TM-86/RS. Prepared for the U.S. Department of Energy, Office of Environmental Management. Oak Ridge National Laboratory, Risk Assessment Program, Health Sciences Research Division, Oak Ridge, TN.

Sample, B.E. and G.W. Suter II. 1999. Ecological risk assessment in a large river-reservoir. 4. Piscivorous birds and wildlife. Environmental Toxicology and Chemistry 18(4). 610-620.

Sauer, J.R., J.E. Hines, G. Gough, I. Thomas, and B.G. Peterjohn. 1997. The North American breeding bird survey results and analysis. Version 96.4. Patuxent Wildlife Research Center, Laurel, MD. (From website http://www.mbr.gov/bbs/bbs.html)

Scott, M.L. 1977. Effects of PCBs, DTT, and mercury compounds in chickens and Japanese quail. Federation Proceed. 36:1888-1893.

Secor, D. 1997. Personal communication (telephone conversation with J. Sampson, Exponent, Bellevue, WA 1998 regarding PCB concentrations in eels). Chesapeake Biological Laboratory, U. of Maryland for Environmental Science, Solomons, MD.

Secor, D. H., and J. E. Baker. 1999. Effects of migration on polychlorinated biphenyl concentrations in Hudson River striped bass. Final report to the Hudson River Foundation, 40 West 20th Street, Ninth Floor, New York, N.Y. 10011

Secord, A.L., and McCarty, J.P., 1997. Polychlorinated Biphenyl Contamination of Tree Swallows in the Upper Hudson River Valley, New York. U.S. Fish & Wildlife Service, New York

Field Office

Smith SL, MacDonald DD, Keenleyside CG, Ingersoll CG, Field J. 1996. A preliminary evaluation of sediment quality assessment values for freshwater ecosystems. J. Great Lakes Res. 22:624-638.

Sprunt IV, A., W.B. Robertson, Jr., S. Postupalsky, R.J. Hensel, C.E. Knoder, and F.J. Ligas. 1973. Comparative productivity of six bald eagle populations. Thirty-eighth North American Wildlife Conference. pp. 96–106.

Suter, G.W. II 1999. Lessons for small sites from assessments of large sites. Environmental Toxicology and Chemistry 18(4): 579-580

Suter, G.W. II, L.W. Barnthouse, R.A. Effroymson and H. Jager, 1999. Ecological risk assessment in a large river reservoir: 2 fish community. Environmental Toxicology and Chemistry 18 (4): 589-598

Suter, G. W. II (ed.) 1993. Ecological Risk Assessment. Lewis Publishers, Chelsea, MI

Suter, G.W. II, and J. M. Loar, 1992. Weighing the ecological risks of hazardous waste sites: the Oak Ridge case. Environmental Toxicology and Chemistry 26:432-438.

Swartz RC, Kemp PF, Schults DW, Lamberson JO. 1988. Effects of mixtures of sediment contaminants on the marine infaunal amphipod, *Repoxynius abronius*. Envrion. Tox. Chem. 7:1013-1020.

Swartz R.C., Di Toro DM. 1997. Sediments as complex mixtures: an overview of methods to assess ecotoxicological significance. In: Ingersoll CG, Dillon T, Biddinger GR, editors. Ecological risk assessment of contaminated sediments. SETAC Press, FL. p 255-269.

Tillitt, D.E., R.W. Gale, J.C. Meadows, J.L. Zajicek, P.H. Peterman, S.N. Heaton, P.D. Jones, S.J. Bursian, T.J. Kubiak, J.P. Giesy, and R.J. Aulerich. 1996. Dietary exposure of mink to carp from Saginaw Bay. 3. Characterization of dietary exposure to planar halogenated hydrocarbons, dioxin equivalents, and biomagnification. Environ. Sci. Technol. 30:283–291.

U.S. Environmental Protection Agency. 1993. Technical basis for deriving sediment quality criteria (SQC) for nonionic organic contaminants for the protection of benthic organisms by using equilibrium partitioning. Washington DC: USEPA. EPA/822/R-93/011.

USFWS. 1997. Significant Habitats and Habitat Complexes of the New York Bight Watershed. Southern New England - New York Bight Coastal Ecosystems Program. Charlestown, Rhode Island. 1200 pp. (on CD)

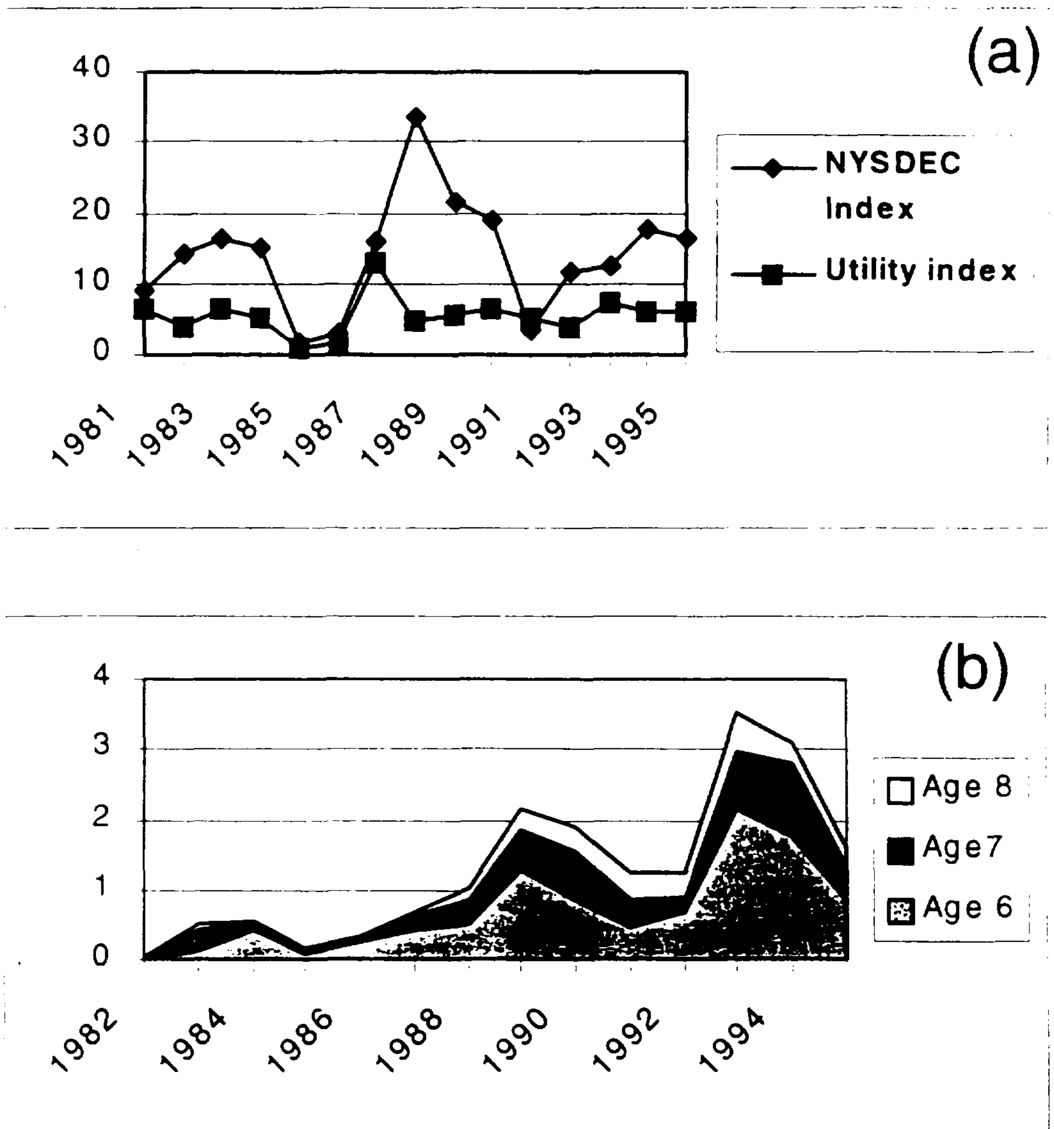
Van den Berg, M., L. Birnbaum, A.T.C. Bosveld, B. Brunström, P. Cook, M. Feeley, J.P. Giesy, A. Hanberg, R. Hasegawa, S.W. Kennedy, T. Kubiak, J.C. Larsen, F.X.R. van Leeuwen, A.K.D. Liem, C. Nolt, R.E. Peterson, L. Poellinger, S. Safe, D. Schrenk, D. Tillitt, M. Tysklind, M.

304453

Younes, F. Wærn, and T. Zacharewski. 1998. Toxic Equivalency Factors (TEFs) for PCBs, PCDDs, PCDFs for humans and wildlife. Environ. Health Perspect. 106(12):775–792.

Yorks, A.L., M.J. Melancon and C.S. Hulse. 1998. Cytochrome PA50 Monooxygenase Activities as a Biomarker for PCB exposure and Effect in Field Collected and Manually Dosed Tree Swallow (Titachycineta bicolor) Nestlings Platform Presentation #257 Society of Environmental Toxicology and Chemistry, 19th Annual Meeting. Charlotte, NC.

LWB
Environmental Services, Inc.



Indices of striped bass abundance in the Hudson River. (a) Young-of-the-year indices from beach seine surveys conduced by the Hudson River utilities and the NYSDEC. (b) CPUE for age 6 through 8 striped bass caught as bycatch in the gillnet fishery for American shad. Data from SARC (1998).

Table 1: Comparison of Hudson River ERA and Clinch River ERA

Hudson River ERA	Clinch River ERA
Problem Formulation	
Assessment endpoints: Maintenance of benthic community structure; protection and maintenance of local fish, insectivorous birds, waterfowl, piscivorous birds, and wildlife; protection of threatened and endangered species; protection of significant habitats.	**Assessment endpoints:** Reductions in benthic community richness or abundance; reductions in fish species richness or abundance; increased frequency of gross pathologies in fish communities; reduced abundance or production of piscivorous and insectivorous wildlife
Measurement endpoints: Near-field benthic community study, water and sediment-quality criteria, Chronic TRVs (reproduction endpoint) for fish, birds, and mammals	**Measurement endpoints:** Near-field and far-field biological survey data (fish and benthic invertebrates), whole-sediment toxicity tests; whole-water toxicity tests, fish histopathology, water and sediment-quality criteria; chronic TRVs for fish, birds, and mammals, blue heron reproductive success, mink dietary toxicity studies
Exposure Assessment	
Measured concentrations of aroclors and PCB congeners in fish (whole body), water, and sediment Modeled oral doses (aroclors and TEQs) to avian and mammalian receptors using conservative exposure assumptions; modeled egg concentrations in birds	Measured concentrations of aroclors in fish (whole body), water, and sediment Measured concentrations of aroclors in great blue heron eggs and chicks Modeled oral doses to avian and mammalian receptors (by subarea), using (1) conservative exposure assumptions, and (2) Monte Carlo analysis of all exposure parameters

Effects Assessment	
Hudson River ERA	**Clinch River ERA**
TRVs for PCB and TEQ concentrations in fish tissue	TRV for PCB concentrations in fish tissue (whole body, adult)
Field-derived (tree swallow) or literature-derived (other species) TRVs for fish, birds, mammals	Literature-derived TRVs for birds and mammals
Analysis of local-scale benthic community diversity	Site-specific assessment of fish histopathology and reproductive condition
	Whole-sediment toxicity tests
	Whole-water toxicity tests
	Analysis of fish and benthic community composition at local and regional scales
	Site-specific mink dietary toxicity study
	Site-specific study of great blue heron reproductive success

Risk Characterization	
Hudson River ERA	**Clinch River ERA**
All assessment endpoints: Comparison of water and sediment concentrations to water and sediment-quality criteria **Benthic Invertebrates**: Correlation of local-scale benthic community diversity with PCB concentrations in sediment **Fish**: Comparison of aroclor and TEQ concentrations in fish tissue to literature-derived NOAEL TRVs Overview of population trends for selected species **Birds**: Comparison of modeled oral doses and egg concentrations (aroclors and TEQs) to field-derived (tree swallow) or literature-derived (other species) TRVs Qualitative overview of occurrence data for various species **Mammals**: Comparison of modeled doses (aroclors and TEQs) to literature-derived TRVs	**Benthic Invertebrates**: Comparison of maximum sediment concentration to sediment-quality criteria; comparison of empirical distribution functions for sediment toxicity to cumulative distribution of measured sediment concentrations Whole-sediment toxicity tests **Fish:** Comparison of observed concentration in fish tissue to TRVs Whole-water toxicity test results Comparison of frequencies of histopathological and reproductive condition indicators in study area to observed values in unexposed upstream reservoir Canonical discriminant analysis of fish community composition (reservoir scale); analysis of species richness (reservoir scale and local scale) **Birds:** Comparisons of modeled dose distributions (cumulative frequencies from Monte Carlo analysis) to TRVs Comparison of blue heron reproductive success in on-site and off-site rookeries; comparison of osprey reductive success in nests adjacent to site to observed range of North American values **Mammals**: Comparisons of modeled dose distributions (cumulative frequencies from Monte Carlo analysis) to TRVs Comparison of toxicity observed in mink dietary study to toxicity predicted from exposure model and literature-derived TRVs

304458

Appendix A

Critique of Analyses Presented in Appendix K of EPA's Ecological Risk Assessment

Appendix K to EPA's Baseline Ecological Risk Assessment (BERA) is not relevant to the risk assessment itself. In addition, there are significant problems associated with several of the conclusions. In particular, the usefulness of the "Principal Components Analysis" (PCA) set out in the Appendix rests on the analyst's ability to understand the biological or physical meaning behind the components. EPA presents little analysis of the interpretation of component 2 and therefore leaves open the question of the meaning of differences or similarities among samples. Appendix K should not be included in the BERA, and its conclusion should be disregarded. Below, we comment specifically on several points brought up in Appendix K. **EG-1.39a**

Impact of Upstream Remediation Between 1993 and 1995

Appendix K suggests that remediation of upstream PCB sources will not lead to significant reductions in fish PCB levels. This conclusion is unwarranted, is contradicted by other lines of evidence, and is not relevant to the ecological risk assessment. **EG-1.39b**

In Section K.6 of Appendix K, EPA compares the 1993 and 1995 fish congener data collected by NOAA and EPA using PCA. From this analysis, EPA concludes that "little difference was evident between the two fall sampling events, suggesting that little had occurred (such as GE remediation of the Hudson Falls releases) to affect the congener patterns, and, by inference, the basic routes of exposure in fish." BERA (p. 68 of Book 1).

Although Section K.6 presents several comparisons between fish collected in fall 1993 and fall 1995, EPA states that most of the comparisons cannot be used because of confounding variables. In fact, the differences seen in the fall 1993 and 1995 samples are due in part to differences between juvenile and adult fish (Figure K-41). When age class is not considered, EPA states that for striped bass "there may actually be a discernable difference between the Fall 1993 results and the Fall 1995 results since there is a portion of both data sets based on the same life-stage which appear to be different" (p. K-19). For white perch and yellow perch, life-stage does not appear to influence the comparison, and based upon Figure K-40, EPA concludes that there is no difference between fall 1993 and fall 1995. In fact, these comparison provide little support for EPA's conclusion of no difference: for yellow perch, there are only five values presented for fall 1995. These are tightly clustered and lie within the range of the Fall 1993 values. However, all but four of the values for fall 1993 lie below and to the right of the Fall 1995 values (Figure K-40), suggesting that the populations as a whole may in fact be different.

This leaves only the white perch data (Figure K-40) to support the conclusion of no effect. The 1993 and 1995 values data do indeed overlap. However, the white perch data must be balanced against the results seen in the striped bass and yellow perch. In short, EPA's conclusion that there are no differences in the fall 1993 and fall 1995 fish data is very weak.

In addition, no data for the fall 1995 collection above RM 152 were used (page K-18), so any conclusions drawn must relate only to the Lower Hudson River.

Finally, it is unclear what EPA means by effects on the "basic routes of exposure." The most likely meaning is that the PCB composition of the source did not change. However, this does not mean that remediation activities upstream of Thompson Island Pool had no impact on PCB levels. It simply means that either there was no change in the composition of the source to the fish during this period or that the tool used was too imprecise to tell potentially different sources apart.

The most direct way to assess the impact of upstream remediation on fish PCB levels is to analyze fish PCB levels. For example, lipid-based levels in largemouth bass, pumpkinseed and brown bullhead in Thompson Island Pool declined from 1993 to 1995 (Figure 1), consistent with a decrease in exposure level.

In conclusion, the analysis presented in Section K.6 does not support EPA's assertion that site-based remediation had no effect on fish levels. The limited number of data points, the scatter observed in the PCA, the fact that only a qualitative appraisal of the results was presented, the differences among species in apparent trends from 1993 to 1995, and the limitation of the data to the Lower Hudson River all undermine EPA's conclusion that remediation of the Hudson Falls releases did not affect the basic routes of exposure in fish. This conclusion also is contradicted, at least in Thompson Island Pool, by the more direct analysis of PCB levels in fish.

Sources of PCBs to the Fish in the Freshwater Portion of the Lower Hudson River

In discussing spatial patterns in the congener composition in fish, EPA, referring to the freshwater portion of the Lower Hudson River, states that "additional, substantive, higher molecular weight PCB load to this region is not in evidence" (p. K-10). This conclusion is based upon analyses presented in the DEIR, but is contradicted by the data presented by EPA. **EG-1.39c**

EPA states that PCB concentrations decrease from the Thompson Island Pool to the Lower Hudson River, but that there is a "trend to a nearly constant average value for each feeding guild in the Lower Hudson" (p. K-11), presumably referring to the fact that the decline in lipid-based total PCB concentrations in fish observed in the Upper Hudson River and in the upper portion of the Lower Hudson River stops, and concentrations in the lower portion of the Lower Hudson River do not decline towards the mouth (Figure K-18). In fact, total PCB concentrations in foragers in Figure K-18 actually increase towards the mouth downstream of approximately mile 120. Over this region, concentrations would be expected to decline due to dilution. This is shown in the bottom panels of Figures 2a and 2b, each of which contains a line indicating the expected degree of dilution by freshwater inflow (mile 153 to approximately mile 60), and by tidal mixing (mile 60 to the mouth). On each plot, the value on the Y-axis represents relative concentration and is set to 1.0 at mile 153. For example, PCBs in the water at Troy are expected to be diluted by 80% at the mouth of the river due to freshwater inflow and tidal mixing. Comparison of calculated dilution with the observed lack of gradient in forage fish total PCB levels downstream of approximately mile 120 (Figure K-18) suggests that lower river sources may be affecting PCB levels even above the salt wedge, perhaps as far north as mile 120.

This conclusion is brought out more clearly by studying spatial patterns in the concentrations of individual congeners in fish collected by EPA and NOAA in 1993 (Figures 2a-2b). Above approximately mile 100, concentrations of each congener decline more rapidly than expected by dilution. Below the region between mile 90 and 110, concentrations of all PCBs do not decline as fast as expected by dilution, and concentrations of higher chlorinated PCBs actually increase. Note that actual concentrations (mg/kg lipid) are plotted here, not weight proportion. This provides evidence for a higher molecular weight PCB load to the region of the freshwater Lower Hudson River downstream of the region between mile 90 and 110. Thus, EPA's conclusion is contradicted by the available data it presents.

Use of Congener Ratios to Explore PCB Sources to the Fish

EPA concludes that the use of congener ratios provides "little clue as to the nature of the source" of PCBs to the fish (p. K-30-31). This is based upon variation in spatial trends among media (fish, sediment and water) as well as variation among fish. EPA's analysis is focused on assessing the relative importance of upper and lower river sources to the fish. Interpretation of these ratios for the purposes of Appendix K is difficult, but EPA's conclusion is too broad. These ratios can provide useful information concerning PCB sources to fish in other contexts, as demonstrated in QEA (1999). **EG-1.39d**

QEA used these ratios to answer a different question: are the fish in the Upper Hudson River exposed to dechlorinated PCBs or relatively undechlorinated PCBs? The observation that the average ratio was more similar to surface sediments than buried dechlorinated sediments was very clear (Figure 5-15 of QEA 1999).

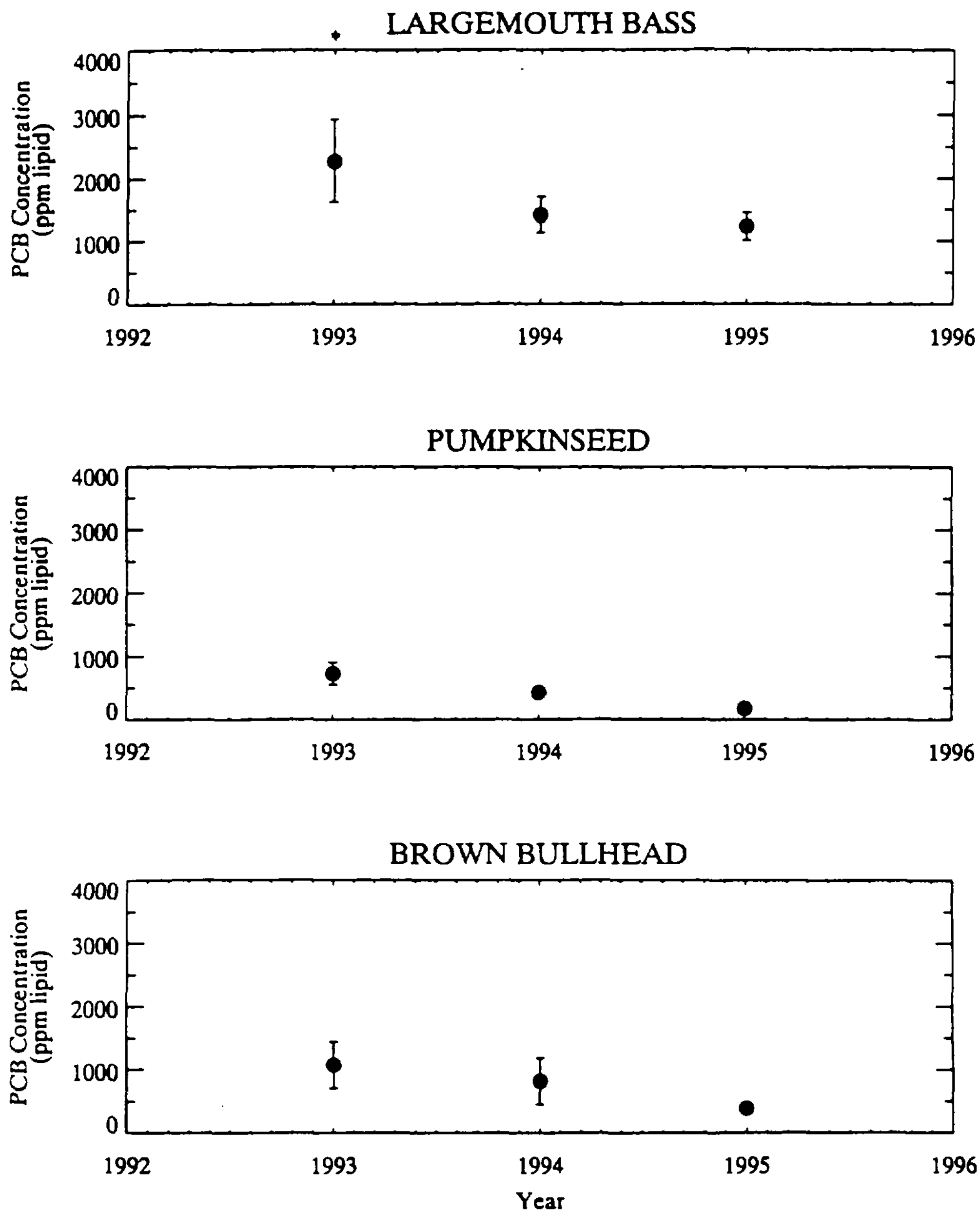
On a related point, EPA concludes that because the value of the ratios is greater in the particulate phase, the partition coefficient for BZ#49 is greater than the congeners used in the numerators (p. K-28). The opposite is true, and indeed the K_{OW} values of the congeners used in the numerators are approximately a factor of two greater than BZ#49 (Hawker and Connell 1988).

In conclusion, the statement that congener ratios provide little clue as to the nature of PCB sources is too broad and is contradicted by data. A more correct statement would be that the analysis presented in Appendix K does not lead to definitive conclusions concerning variation in PCB sources with river mile.

References

Hawker DW, Connell DW. 1988. Octanol-water partition coefficients of polychlorinated biphenyl congeners. Environ. Sci. Technol. 22:382-385.

[QEA] Quantitative Environmental Analysis, LLC. 1999. PCBs in the Upper Hudson River. Report to the General Electric Company Corporate Environmental Programs, Albany NY.



Data are arithmetic means +/- 2 standard errors.
NYSDEC Data

Figure 1. Lipid-normalized PCB concentrations in largemouth bass, pumpkinseed and brown bullhead collected from Thompson Island Pool.

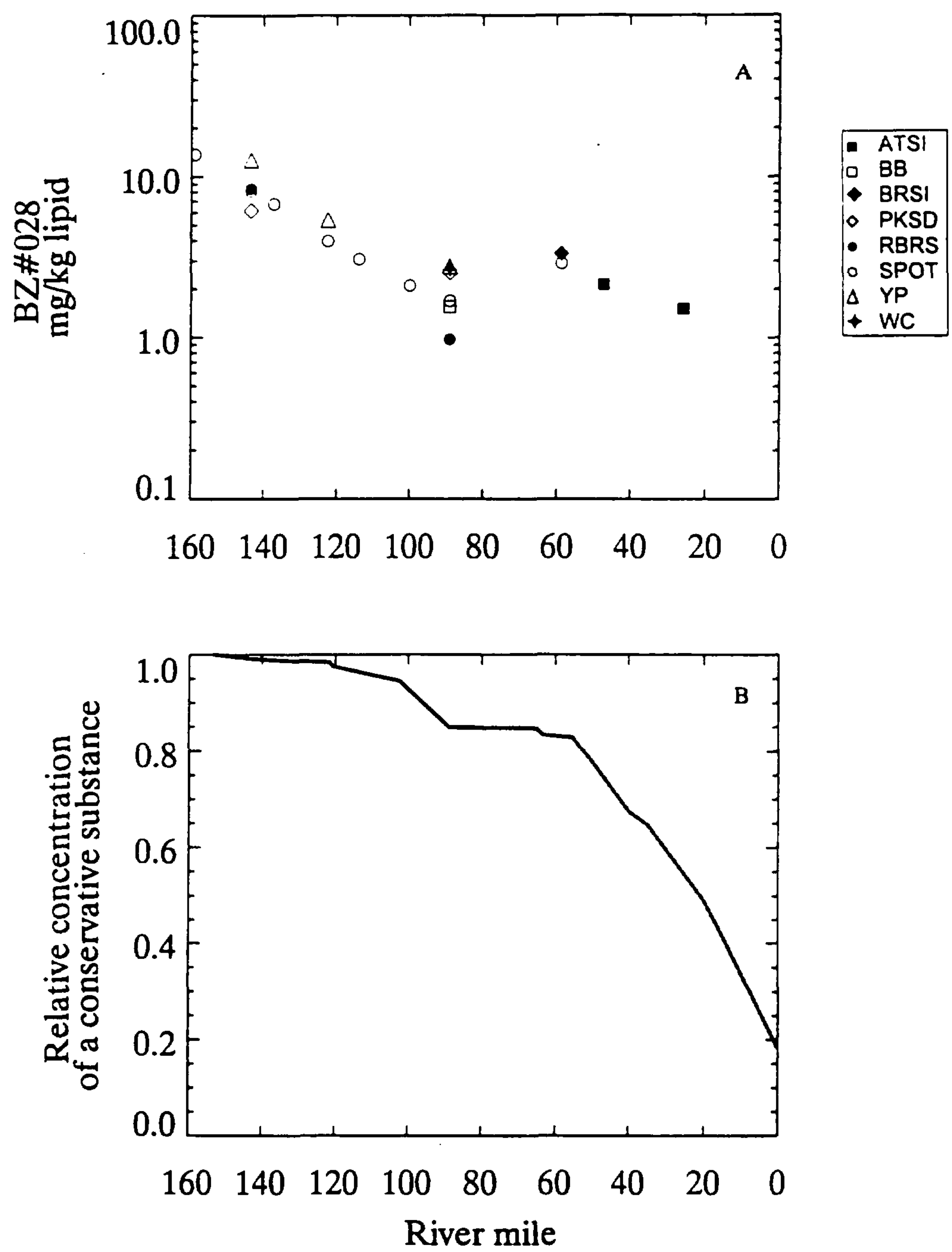


Figure 2a. Spatial Gradients in Concentration of PCB Congener BZ#028 in Fish from the Hudson River Compared With Gradients Expected Due to Dilution of PCBs Originating Upstream.

A: Species Specific Average Concentration of BZ#028
B: Concentration of a Conservative Substance Relative to MP153.
Dilution estimated based upon tributary inflow above MP60 and tidal mixing below MP60.
Data: EPA Phase II and NOAA, 1993

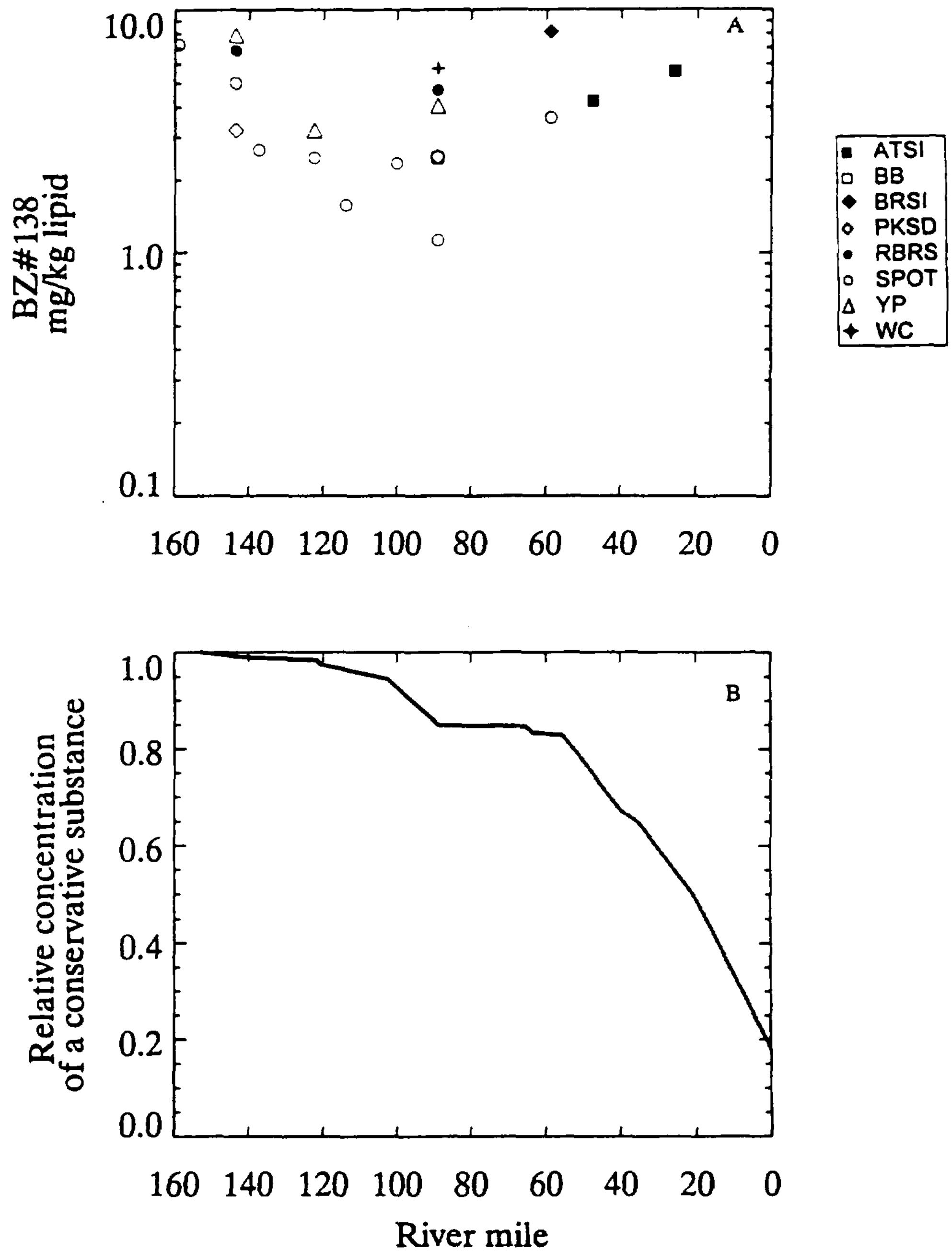


Figure 2b. Spatial Gradients in Concentration of PCB Congener BZ#138 in Fish from the Hudson River Compared With Gradients Expected Due to Dilution of PCBs Originating Upstream.

A: Species Specific Average Concentration of BZ#138

B: Concentration of a Conservative Substance Relative to MP153.

Dilution estimated based upon tributary inflow above MP60 and tidal mixing below MP60.

Data: EPA Phase II and NOAA, 1993

Appendix B

Critique of the Evaluation of the Predictive Capability of the NOAA (1999) SEC Values

NOAA (1999) used three approaches to evaluate the predictive capability of their Sediment Effect Concentration (SEC) values:

- Comparison to field data from numerous freshwater and estuarine/marine sites
- Comparison to the results of a laboratory spiked-sediment bioassay
- Comparison to screening level sediment quality criteria developed by New York State Department of Environmental Conservation (NYSDEC) using equilibrium partitioning

All of these approaches are flawed and do not validate the use of the SEC values as reasonable predictors of PCB toxicity.

1.0 Comparison to Field Data

This evaluation of SEC predictive capability was conducted using a data set that included the results of studies from eight freshwater bodies, eleven estuarine or marine sites and the Environmental Monitoring and Assessment Program (EMAP) Virginia Province. These data were used to test the frequency at which the SEC values correctly predicted the presence or absence of toxicity. This testing contained the following flaws:

- The data set was not independent of the data used to develop the SEC values
- It is not likely that PCBs were the toxic agent at most of the sites
- Comparison over multiple sites does not test the predictive capability at individual sites

1.1 Lack of Independence of the Validation Data Set

The pre-existing SEC values used by NOAA are not independent of the SEC validation data to develop the "Consensus-based" SEC values. For example, the Ingersoll et al. (1996) SECs were developed using data from 5 of the 8 freshwater sites that form the testing data set (i.e., Indiana Harbor; Saginaw River; Trinity River; Upper Mississippi River; Waukegan Harbor).

1.2 Lack of Relevance to PCB Toxicity

A validation of the relevance of the SEC values to PCB toxicity in the Hudson River would be best achieved by comparison to data from sites in which PCBs are the probable toxic agent. No attempt was made to use such a criterion in site selection. Five of the freshwater sites in the validation data set clearly fail to meet this criterion. Two of the water bodies (Trinity River and Upper Mississippi River) had little or no PCB present, as indicated by the overwhelming frequency of non-detect concentrations. Three of the water bodies contain chemicals other than PCBs that could account for all of the observed toxicity (Indiana Harbor; Grand Calumet River; Potomac River). In fact, the reference cited for the Grand Calumet River data (Hoke et al. 1993) states that "... ammonia, polycyclic aromatic hydrocarbons, metals, petroleum hydrocarbons, and bicarbonate ion appear to be the major contaminants of environmental significance to benthic invertebrates within the study area." A sixth site (Saginaw River) is also problematic because it contains significant concentrations of several heavy metals (Moll et al., 1995) and significant TCDD TEQ values that are attributable mostly to dioxin and furan congeners (Gale et al., 1997).

All of the estuarine or marine sites in the validation data set fail the PCB dominance criterion. Most of the sites contain relatively high concentrations of PAHs. Some contain significant amounts of pesticides or metals. For example, toxicity in Tampa Bay was significantly correlated with trace metals, pesticides, PAHs and ammonia in addition to

PCBs (Carr et al., 1996). Similarly, as shown in Figure B-1, amphipod mortality in the Hudson-Raritan Estuary and LA Harbor exhibited no evident dose-response relationship to PCBs, but did show such a relationship for PAHs (Anid and Connolly, 1998).

1.3 Lack of Predictive Ability at Individual Sites

The two freshwater studies in which PCBs are perceived to be a primary contaminant demonstrate that the SEC values lack predictive ability. The Lower Fox River and Green Bay data exhibit 86 percent false positives for the Extreme Effect Concentration (EEC) and 84 percent false positives for the Medium Effect Concentration (MEC). Although all but one of the Waukegan Harbor samples are classified as toxic, neither the mortality or growth endpoints exhibit a dose-response relationship with PCBs. Further, the one non-toxic sample had the third highest PCB concentration in the data set (7.4 mg/kg DW).

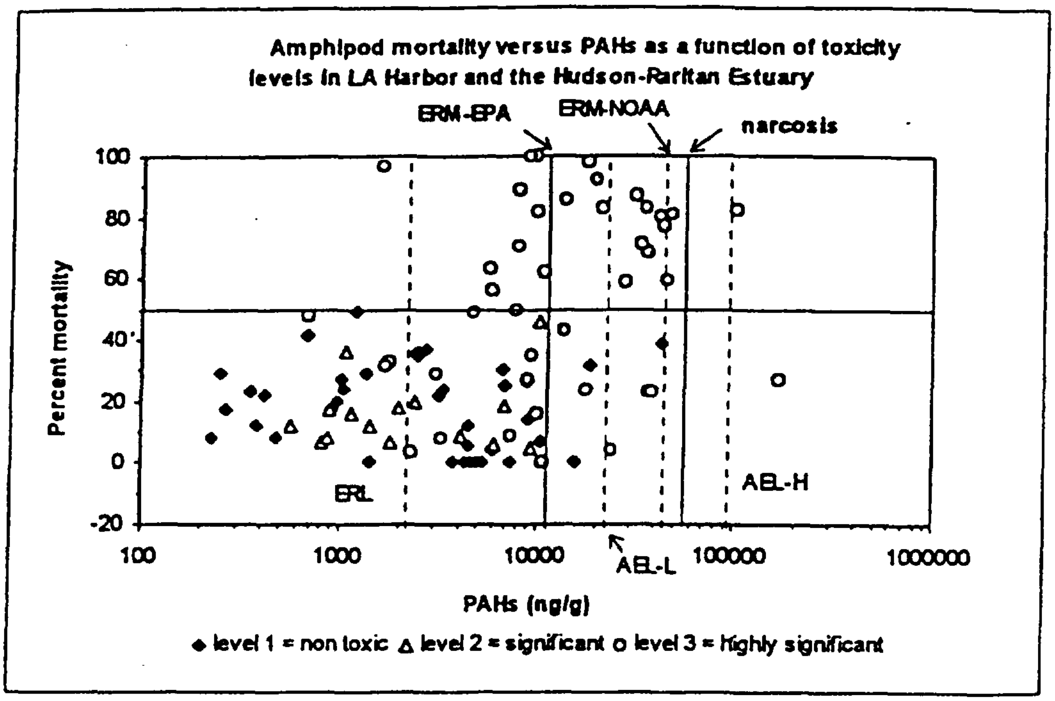
2.0 Comparison to the Laboratory-Spiked Sediment Bioassay

The Swartz et al. (1988) spiked sediment toxicity study cannot be compared to the SEC values. This study used sediment with a TOC content of about 0.25 percent. Most sediments contain a TOC content in the range of 1 to 4 percent. The fine sediment of the Thompson Island Pool has an average TOC of about 2 percent. Based on the relationship between the bioavailability of organic chemicals in sediments and sediment TOC that forms the basis of EPAs Sediment Quality Criteria (EPA, 1993), the Swartz et al, toxicity results would have to be adjusted by about a factor of 8 to be applicable to the Hudson River. Thus, the applicable LC_{50} and LC_{10} values for comparison to the SEC values are 86 and 54 mg/kg DW. For arguments sake, accepting the acute-to-chronic ratio of 11 cited by NOAA (1999) and assuming the validity of this study, one would at most conclude that PCBs would not even *begin* to cause chronic toxicity to amphipods until concentrations exceeded about 8 mg/kg DW.

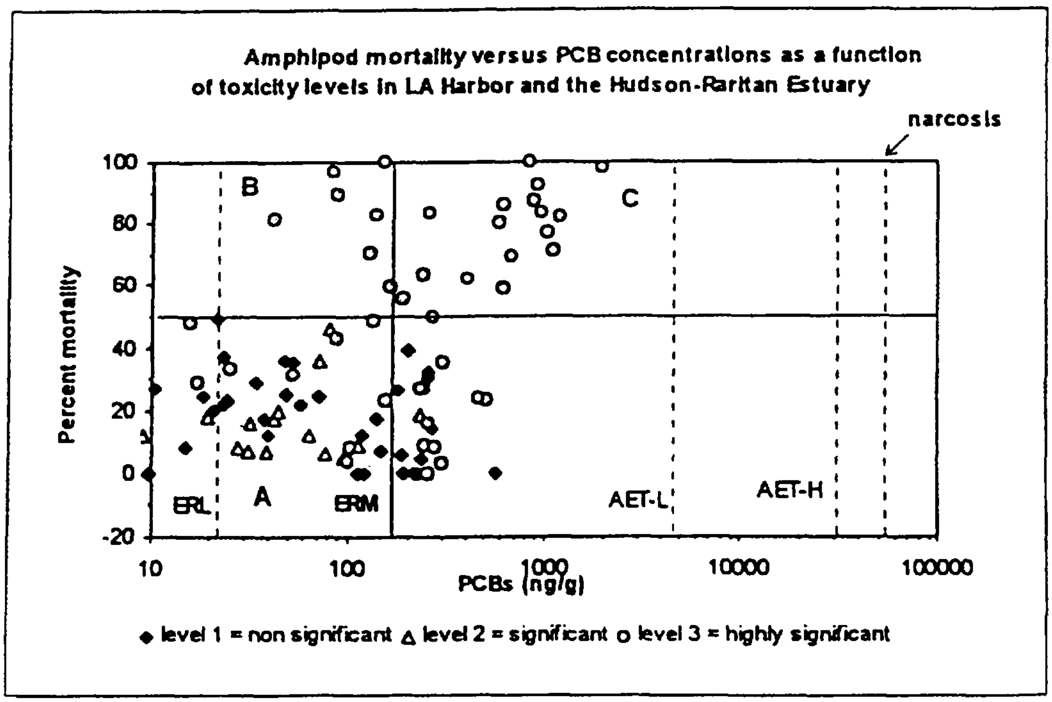
3.0 Comparison to Screening Level Sediment Quality Criteria Developed using Equilibrium Partitioning (EqP)

EqP sediment quality criteria are by definition sediment concentrations at which no effect is expected. They are derived using two conservative assumptions. The first is that sediment chemical is fully bioavailable. It is well accepted that some fraction of the chemical in the sediments is not bioavailable. The second is that the water quality criterion is the maximum allowable pore water concentration. The water quality criterion is determined using procedures that ensure that it is protective, i.e., it is not a toxicity threshold and may be significantly below a toxicity threshold. It is for these reasons that sediment quality criteria are best used as screening values. Sediment concentrations lower than the EqP are presumed safe. Concentrations greater than the EqP may, or may not, indicate toxicity.

The NYSDEC (1993) sediment quality guidelines were calculated using the EPA 1991 water quality criterion and an organic carbon normalized partition coefficient (K_{oc}) of $10^{6.14}$. The chosen partition coefficient is generic and not necessarily applicable to the Hudson River. In fact, Hudson River field data indicate a K_{oc} about a factor of 3 to 4 lower than the generic value (QEA 1999). Thus, the NYSDEC numbers provide no validation of the SEC values.



Amphipod mortality versus PAHs as a function of toxicity levels in LA Harbor and the Hudson-Raritan Estuary
ERM-EPA
ERM-NOAA
narcosis
Percent mortality
ERL
AEL-L
AEL-H
PAHs (ng/g)
◆ level 1 = non toxic △ level 2 = significant o level 3 = highly significant



Amphipod mortality versus PCB concentrations as a function of toxicity levels in LA Harbor and the Hudson-Raritan Estuary
narcosis
Percent mortality
B
C
A
ERL
ERM
AET-L
AET-H
PCBs (ng/g)
◆ level 1 = non significant △ level 2 = significant o level 3 = highly significant

This Page Was Intentionally Left Blank For Pagination Purposes.

References

Anid PJ, Connolly JP. 1998. Limitations and uncertainties of sediment effects concentrations (SECs) in the evaluation of sediment toxicity and chemistry. Presentation at the SETAC 19th Annual Meeting, Charlotte NC. November 19, 1998.

Carr RS, Long ER, Windom HL, Chapman DC, Thursby, G, Sloane GM, Wolfe DA. 1996. Sediment quality assessment studies of Tampa Bay, Florida. Environ. Tox. Chem. 15:1218-1231.

Gale RW, Huckins JN, Petty JD, Peterman PH, Williams LL, Morse D, Schwartz TR, Tillet DE. 1997. Comparison of the uptake of dioxin-like compounds by caged channel catfish and semipermeable membrane devices in the Saginaw River, Michigan. Environ. Sci. Technol. 31:178-187.

Hoke RA, Giesy JP, Zabik M, Unger M. 1993. Toxicity of sediments and sediment pore waters from the Grand Calumet River-Indiana Harbor, Indiana area of concern. Ecotox. Environ. Safety. 26:86-112.

Ingersoll CG, Haverland PS, Brunson EL, Canfield TJ, Dwyer FJ, Henke CE, Kemble NE, Mount DR, Fox RG. 1995. Calculation and evaluation of sediment effect concentrations for the amphipod *Hyalella azteca* and the midge *Chironomus riparius*. J. Great Lakes Res. 22:602-623.

Moll RA, Jude D, Rossmann R, Kantak GV, Barres J, DeBoe S, Giesy J, Tuchman M. 1995. Movement and loading of inorganic contaminants through the lower Saginaw River. J. Great Lakes Res. 21:17-34.

[NOAA] National Oceanic and Atmospheric Administration. 1999. Development and evaluation of consensus-based sediment effect concentrations for PCBs in the Hudson River. Prepared by MacDonald Environmental Services Ltd. Ladysmith BC. March 1999.

[NYSDEC] New York State Department of Environmental Conservation. 1993. Technical guidance for screening contaminated sediments. Albany NY. November 1993.

Swartz RC, Kemp PF, Schults DW, Lamberson JO. 1988. Effects of mixtures of sediment contaminants on the marine infaunal amphipod, *Repoxynius abronius*. Envrion. Tox. Chem. 7:1013-1020.

[USEPA] U.S. Environmental Protection Agency. 1993. Technical basis for deriving sediment quality criteria (SQC) for nonionic organic contaminants for the protection of benthic organisms by using equilibrium partitioning. Washington DC: USEPA. EPA/822/R-93/011.