HUDSON RIVER PCBs REASSESSMENT RI/FS
RESPONSIVENESS SUMMARY FOR VOLUME 2E-A BASELINE ECOLOGICAL RISK ASSESSMENT FOR FUTURE RISKS IN THE LOWER HUDSON RIVER

AUGUST 2000

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

August 29, 2000

To All Interested Parties:

The U.S. Environmental Protection Agency (USEPA) is pleased to release the Responsiveness Summary for the baseline Ecological Risk Assessment for Future Risks in the Lower Hudson River (ERA Addendum), 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 Addendum and this Responsiveness Summary should be used together.

In the Responsiveness Summary, USEPA has responded to all significant written comments received during the public comment period on the ERA Addendum. In addition, the Responsiveness Summary contains revised calculations of ecological risks based on the January 2000 Revised Baseline Modeling Report and comments received on the Ecological Risk Assessment and the ERA Addendum. Importantly, the overall conclusions regarding the future risks to ecological receptors due to PCBs in the Lower Hudson River remain unchanged.

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

Sincerely yours,

William McCabe

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

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Acronyms

BSAF	Biota-Sediment Accumulation Factor
CIP	Community Interaction Program
DEIR	Data Interpretation and Evaluation Report
ERA	Ecological Risk Assessment
ERASOW	Ecological Risk Assessment Scope of Work
GE	General Electric
HHRA	Human Health Risk Assessment
LOAEL	Lowest-Observed-Adverse-Effect-Level
NOAA	National Oceanic and Atmospheric Administration
NOAEL	No-Observed-Adverse-Effect-Level
NYSDEC	New York State Department of Environmental Conservation
PCB	Polychlorinated Biphenyl
RBMR	Revised Baseline Modeling Report
RI/FS	Remedial Investigation/Feasibility Study
RM	River Mile
RI/FS	Remedial Investigation/Feasibility Study
STC	Science and Technical Committee
SWEM	System-Wide Eutrophication Model
TCDD	2,3,7,8-Tetrachlorodibenzo-p-dioxin
TEQ	Toxicity Equivalency
TI	Thompson Island
TOC	Total Organic Carbon
TRV	Toxicity Reference Value
TQ	Toxicity Quotient
UCL	Upper Confidence Limit
USEPA	United States Environmental Protection Agency
USACE	United States Army Corps of Engineers
USFWS	United States Fish and Wildlife Service
USGS	United States Geological Survey

Introduction

HUDSON RIVER PCBs REASSESSMENT RI/FS
RESPONSIVENESS SUMMARY FOR
VOLUME 2E-A BASELINE ECOLOGICAL RISK ASSESSMENT
FOR FUTURE RISKS IN THE LOWER HUDSON RIVER

AUGUST 2000

I. INTRODUCTION AND COMMENT DIRECTORY

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 for Future Risks in the Lower Hudson River (ERA Addendum) for the Hudson River PCBs Reassessment Remedial Investigation/Feasibility Study (RI/FS), dated December 1999.

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

The ERA Addendum is incorporated by reference and is not reproduced herein. The comment responses and revisions noted herein are considered to amend the ERA Addendum. For complete coverage, the ERA Addendum and this Responsiveness Summary must be used together.

The first part of this Responsiveness Summary is entitled, "Introduction and Comment Directory." It describes the ERA Addendum 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 for Future Risks in the Lower Hudson River," contains USEPA's responses t o all significant comments received on the ERA Addendum. Responses are grouped according to the section number of the ERA Addendum to which they refer. For example, responses to comments on Section 2.2 of the ERA Addendum 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.

The third part, entitled "Risk Assessment Revisions," presents the revised results for the ERA Addendum, incorporating the modified forecast concentrations of PCBs in fish, sediments, and river water from the Revised Baseline Modeling Report (USEPA, 2000a) and other revisions based on comments received on the ERA Addendum. To facilitate comparison to the December 1999 ERA Addendum, all table and figure numbers have retained their original designations.

The fourth part, entitled "Comments on the ERA Addendum," contains copies of the comments on the ERA Addendum submitted to USEPA. Not all references provided by the commenters are reproduced in this document. The comments are identified by commenter 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 Addendum, issued in December 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 Addendum were also made available for public review in 16 Information Repositories, as shown in Table 2 and on the USEPA Region 2 Internet web page, entitled "Hudson River PCBs Superfund Site Reassessment," at www.epa.gov/hudson.

2.2 Review Period and Public Availability Meetings

USEPA held a formal comment period on the ERA Addendum from December 29, 1999 to January 28, 2000. USEPA held a Joint Liaison Group meeting on January 11, 2000 in Poughkeepsie, New York that was open to the public to present the ERA Addendum. Subsequently, USEPA sponsored an availability session to answer questions on January 18, 2000 in Poughkeepsie, New York. These meetings were conducted in accordance with USEPA's "Community Relations in Superfund: Handbook, Interim Version" (1998a). Minutes of the Joint Liaison Group meeting are available for public review at the Information Repositories listed in Table 2.

As stated in USEPA's letter transmitting the ERA Addendum, 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 January 11, 2000 Joint Liaison Group meeting. USEPA's responses to oral statements made at the Joint Liaison Group meetings are provided in the meeting minutes. Written comments were received from four commenters; total comments number 100. All significant written comments received on the ERA Addendum are addressed in this Responsiveness Summary.

2.4 Distribution of Responsiveness Summary

This Responsiveness Summary is being distributed to, among others, the Liaison Chairs and Co-Chairs and interested public officials. This Responsiveness Summary is also being placed in the 16 Information Repositories and is part of the Administrative Record.

TABLE 1
DISTRIBUTION OF ERA ADDENDUM

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 Chairpersons
- 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

- Dr. 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 Addendum 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

Copies of the ERA Addendum were placed in 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 1222

*^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)***

3. Organization of ERA Addendum Comments and Responses to Comments

3.1 Identification of Comments

Each submission commenting on the ERA Addendum 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; 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, and EG.

3.2 Location of Responses to Comments

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

- The first column lists the names of commenters. Comments are grouped first by: EF (Federal), ES (State), EL (Local) 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 Addendum Section number. For example, comments raised in Section 2.1 of the ERA Addendum 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 Addendum so that all responses to related comments appear together for the convenience of the reader interested in responses to related or similar comments.

4. 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 Addendum. Readers are urged to utilize this Responsiveness Summary in conjunction with the ERA Addendum.

4.2 Comment Directory

STEP 1	STEP 2	STEP 3
Find the commenter or the key words of interest in the Comment Directory.	Obtain the alphanumeric comment codes and the corresponding ERA Addendum Section.	Find the responses following the Responses tab. See the Table of Contents to locate the page of the Responsiveness Summary for the ERA Addendum Section.
Key to Comment Codes:		
Comment codes are in this format EX-a.b X=Commenter Group (F=Federal, S=State, L=Local, 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	2	3
NOAA /Rosman	EF-1.1	General	Fate/Transport	Bioaccumulation	BMR

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Comment Directory

4.2 Comment Directory - Lower Hudson River Future Risks					
AGENCY/ NAME	COMMENT CODE	REPORT SECTION	KEYWORDS 1	2	3
NOAA/Rosman	EF-1.1	General	Fate and Transport	Bioaccumulation	Baseline Modeling Report Revisions
NOAA/Rosman	EF-1 2	Appendix A	Farley Model	HUDTOX	Model Uncertainty
NOAA/Rosman	EF-1.3	General	Water Column and Sediment Data	Nearshore Areas	Food Web Pathways
NOAA/Rosman	EF-1.4	General	Food Chain Modeling	Fish Concentrations	Model Prediction
NOAA/Rosman	EF-1.5	General	Water and Sediment Values	Model Output	TOC - %Lipid
NOAA/Rosman	EF-1.6	General	TRV's	Study Selection	
NOAA/Rosman	EF-1 7	General	Field Duplicatre Data	NOAELs and LOAELs	Uncertainty Factors
NOAA/Rosman	EF-1 8	General	TRV's	Underestimate Risk	Modeling Results
NOAA/Rosman	EF-1.9	Exec Sum	Bald Eagles	Future Risks	Breeding Success
NOAA/Rosman	EF-1.10	3.1 1 1	Upstream Boundry	BMR High Flow Conditions	Underestimate Risk
NOAA/Rosman	EF-1 11	3.1 1 1	Farley Model	FISHRAND	Output Parameters
NOAA/Rosman	EF-1.12	3.0	Striped Bass	Body Burdens	Normalization
NOAA/Rosman	EF-1 13	3 1 1.2	Farley Model	Overestimates Water Column Loss	Effects on Fish Uptake of PCBs
NOAA/Rosman	EF-1.14	3.1 1.4	Figure 3-7	Two Locations Outside Boundaries	
NOAA/Rosman	EF-1.15	3 1 1 6	Modeled Fish Concentrations	NYSDEC 1998 Data	
NOAA/Rosman	EF-1.16	3 2.2	TOC	Sediment Guidelines	
NOAA/Rosman	EF-1 17	3.2.2	Table 3-7	EPA 1993 Guidelines Based on TCDD	PCB TEQs
NOAA/Rosman	EF-1.18	3.2.3	Predicted Benthic PCB Concentrations	Empirical Data	Reasonable Estimate
NOAA/Rosman	EF-1.19	3 2 3	Striped Bass	Food Web Region 1	Striped Bass to Largemouth Bass Ratio
NOAA/Rosman	EF-1.20	3 3 2	NOAA 1999	Fish PCB Concentrations	Reproduction and Development
NOAA/Rosman	EF-1 21	4.1.2	TRV Laboratory and Field Studies	Hansen et al, and Bengtsson	Other Effects than Reproduction and Development
NOAA/Rosman	EF-1 22	4 1.2	TEFs and TEQs	Data Quality Issues	PCBs BZ# 81 and 126
NOAA/Rosman	EF-1 23	5 1 1.1	Table Reference	Underestimate Sediment PCBs	Farley Model TOC
NOAA/Rosman	EF-1.24	5.1.1.1	Forecasted Sediment Concentrations	NYSDEC Benthic Criterion	
NOAA/Rosman	EF-1.25	5 1 1 1	Table 5-1	Wrong SEL	
NOAA/Rosman	EF-1 26	5 1 2 1	NYSDEC Surface Water Criterion		
NOAA/Rosman	EF-1 27	5 2 1	Forage Fish	Reproductive Effects	TRV
NOAA/Rosman	EF-1 28	5 2 1 2	PCB Partitioning	Eggs vs Fish Tissue	Lipid Normalized vs Wet Weight
NOAA/Rosman	EF-1 29	5 2 1 3	TRV	Whole Body Concentration Studies	

4.2 Comment Directory - Lower Hudson River Future Risks					
AGENCY/ NAME	COMMENT CODE	REPORT SECTION	KEYWORDS 1	2	3
NOAA/Rosman	EF-1 30	5 2 1 3	Measurement Endpoints	TEQ Concentrations	EPA 1993
NOAA/Rosman	EF-1 31	5 2 1 5	Field NOAEL	White Perch - Yellow Perch	Application of Uncertainty Factor
NOAA/Rosman	EF-1 32	5 4	Current Trend in Bird Usage	Historical Bird Usage	GE's Use of PCBs
NOAA/Rosman	EF-1 33	6 2	Dioxin Like Effects Based on Congeners	Other Effects of Congeners	Underestimates Risk
NOAA/Rosman	EF-1 34	6 3 2 1	Model - General Trends	Year to Year Changes in Fish Concentrations	Underestimate Risk
NOAA/Rosman	EF-1 35	6 3 2 1	Uncertainty	Upstream Boundary	Releases From Remnant Deposits
NOAA/Rosman	EF-1 36	6 3 2 2	Sensitivity Analysis	BMR and FISHRAND	Other Factors
NOAA/Rosman	EF-1 37	7 1	Uncertainty	Sediment and Water Forcasts	
NOAA/Rosman	EF-1 38	7 3	Sensitivity of Tree Swallows	Compare to other insectivorous Birds	Underestimate Risk
NOAA/Rosman	EF-1 39	Appendix A	Text Edit	Hexachloro Homologue Ratio	
NOAA/Rosman	EF-1 40	Appendix B	See Comment Section 5 2 1 2		
NOAA/Rosman	EF-1 41	Appendix B	TRVs	Hansen et al, 1971 vs 1974	
NOAA/Rosman	EF-1 42	Appendix B	Bengtsson et al	TRVs	Other Effects than Reproduction and Development
NOAA/Rosman	EF-1 43	Appendix B	NOAEL and LOAEL Transposed	Text Edit	
NOAA/Rosman	EF-1 44	Appendix B	Tables B-5 and B-6	Include USACE 1988	
NOAA/Rosman	EF-1 45	Appendix B	El-No Effect and EL Effect	NOAELs and LOAELs	
NOAA/Rosman	EF-1 46	Appendix B	Table B-7	Lipid Values Reported by Study or Estimated	Estimation Procedure
NOAA/Rosman	EF-1 47	Appendix B	Figure B-2	Referenced NOAEL	Figure Focus - Laboratory Studies
NOAA/Rosman	EF-1 48	Appendix B	Figure B-3	Toxicity Endpoints for Egg Dioxin Equivalencies	Figure Focus - Laboratory Studies
NYSDEC/Ports	ES-1 1	5 1 2 1	NYSDEC Wildlife Criterion	Revised Value	
SCEMC/Hodgson	EL-1 1	General	FISHRAND	Farley Model	
SCEMC/Hodgson	EL-1 2a-h	General	Farley Model	EPA Review	
SCEMC/Hodgson	EL-1 3	General	PCB Source	Upper Hudson Load	Risks Based on Various Loads
SCEMC/Hodgson	EL-1 4	General	Exposure Assessment	Toxicity Assessment	Previous Comments on August Upper River
SCEMC/Hodgson	EL-1 5	Appendix A	Tree Swallows		
SCEMC/Hodgson	EL-1 6	3 1 1 1	Farley Model	15 Year Increments	Future Predictions
SCEMC/Hodgson	EL-1 7a-e	3 1 1 3	Striped Bass	HHRA	
SCEMC/Hodgson	EL-1 8	3 1 1 3	Table 3-3	HUDTOX	

4.2 Comment Directory - Lower Hudson River Future Risks

AGENCY/ NAME	COMMENT CODE	REPORT SECTION	KEYWORDS 1	2	3
SCEMC/Hodgson	EL-1 9	3 1 1 4	Striped Bass	Model vs Data	Food Web Region 1
SCEMC/Hodgson	EL-1 10	3 1 1 5	Figure 3-10	Farley vs FISHRAND	
SCEMC/Hodgson	EL-1 11	3 1 1 6	Figure 3-12	Striped Bass	Data Comparisons
SCEMC/Hodgson	EL-1 12	3 1 2 2	Figures 3-16 and 3-17	Explanation	Figure 3-17 Region 2 only
SCEMC/Hodgson	EL-1 13	3 1 2 3	Figures 3-16 to 3-19	Internal consistency	Use of Farley or FISHRAND
SCEMC/Hodgson	EL-1 14	3 2	Selection of River Mile Range	Fish Body Burdens	Selected Mile vs Area Average
SCEMC/Hodgson	EL-1 15	3 2 4	Brown Bullhead	Shortnosed Sturgeon	Uncertainty
SCEMC/Hodgson	EL-1 16	3 3	See Comments on August ERA		
SCEMC/Hodgson	EL-1 17	4 0	See Comments on August ERA	Bengtsson	Chlophen
SCEMC/Hodgson	EL-1 18	5 0	See Comments on August ERA		
SCEMC/Hodgson	EL-1 19	5 2 1 9	Measurement Endpoints	Striped Bass	River Miles 152 and 113
SCEMC/Hodgson	EL-1 20	5 2 4 1	Body Burden	Shortnosed Sturgeon	Uncertainty
SCEMC/Hodgson	EL-1 21	5 4 3	Trends in Christmas Bird Counts	Health of Bird Populations	
SCEMC/Hodgson	EL-1 22	5 7 3 1	Raccoon	Potential Risk	
SCEMC/Hodgson	EL-1 23	Appendix A	Duplicate Sample	GE Samples	EPA Samples
SCEMC/Hodgson	EL-1 24	Appendix A	Figures A-1 to A-5	Geochemical Trends	Factor 1 and Factor 2
SCEMC/Hodgson	EL-1 25	Appendix A	TID Values	GE Data	EPA Data
SCEMC/Hodgson	EL-1 26	Appendix A	Citation for Data	Decline in PCB Loads	
SCEMC/Hodgson	EL-1 27	Appendix A	Figures A-1 to A-5	Validity of Factors in Table A-2	Factors Constant for 40 years
SCEMC/Hodgson	EL-1 28	Appendix A	Bakers Falls Releases	Post 1990	Post 1990 Releases not of Concern
GE	EG-1 1	General	PCB Levels in the Hudson River	Environmental Health of River	Ecological Risks
GE	EG-1 2	General	Historical Databases	Ecological Risk Assessment	Fish and Wildlife Populations
GE	EG-1 3	General	Future Ecological Risk	Remedial Action	
GE	EG-1 4	2 0	Population vs Individual Risks	GE's Previous Comments	
GE	EG-1 5	2 5	Site Specific Data	Biological Surveys	Toxicity Tests
GE	EG-1 6	3 0	Conservative Assumptions	TQ Approach	Clich River Similarities
GE	EG-1 7	4 0	NOAA SECs	Inherent Limitations	Primary Measure of Risks
GE	EG-1 8	4 0	TEQ Approach	Screening Approach	NOAA 1999 Comments on TEQ Approach
GE	EG-1 9	4 0	NOAA Review of PCB Effects on Fish	Effects Exceeding 5 ppm	
GE	EG-1 10	4 0	TQ Approach	Not Scientifically Defensible	Other Similar Assessments
GE	EG-1 11	3 0	Model Deficiencies	HUDTOX	FISHRAND

4.2 Comment Directory - Lower Hudson River Future Risks					
AGENCY/ NAME	**COMMENT CODE**	**REPORT SECTION**	**KEYWORDS 1**	**2**	**3**
GE	EG-1.12a	3.0	HUDTOX Use in Lower River	PCB Homologs	Application to historical period and Future
GE	EG-1.12b	Appendix A	Farley Model	Model Uncertainty	TI Dam Station
GE	EG-1.13	3.0	Farley Model	Prediction of Lower River Water and Sediment	Critcle Review of Model
GE	EG-1.14	3.0	Lower River Fish Concentrations	FISHRAND vs Farley	Food Web Structure
GE	EG-1.15a-c	3.0	Farley Model	Temporal Trends	Striped Bass
GE	EG-1.16a&b	3.0	Fish modeled not Representative of Wildlife Consumption	Growth Rates not Site-Specific	
GE	EG-1.17	5.0	Condition of Ecological Resources	Benthic Community	Fish and Wildlife Populations
GE	EG-1.18a&b	4.0	EPAs Approach Conservative	Relies on Small Subset of Data	Impropper Interpretation of Data
GE	EG-1.19	4.0	Fish TRVs	Values Developed by Monosson	
GE	EG-1.20	4.0	TRVs	Birds	Uncertainty Factor
GE	EG-1.21	4.0	TRVs	Mammals	
GE	EG-1.22	4.0	Limitations of TRVs and TQ Approach	Selective Treatment of Literature	Uncertainty Factor
GE	EG-1.23	7.0	Reliance on Models	Ignore site-specific Data	Scientific Foundation of Lower River BERA

Responses

II. RESPONSIVENESS SUMMARY FOR THE ERA ADDENDUM FOR FUTURE RISKS IN THE LOWER HUDSON RIVER

General Comments

Response to EF-1.1

Exposure modeling for the ERA Addendum combines the output of HUDTOX with the Farley model and FISHRAND. Uncertainties and potential limitations in HUDTOX and FISHRAND forecasts are discussed in the Revised Baseline Modeling Report (RBMR) (USEPA, 2000a). Discussions of potential limitations of the Farley model approach are provided in Farley et al. (1999). Limitations inherent in USEPA's use of the Farley model approach, and general modeling uncertainties associated with the Lower Hudson ERA, are discussed in the ERA Addendum (see, USEPA, 1999c, pp. 70-73).

The ERA Addendum was completed in December 1999, prior to the revisions in the HUDTOX modeling of the Upper Hudson, which are presented in the RBMR (USEPA, 2000a). The revisions result in some relatively small changes in the forecasts of Tri+ PCB load across Federal Dam. The Farley model for the Lower River was subsequently re-run using the output from the revised HUDTOX model. The resulting forecasts for the Lower Hudson River are presented in Section III of this Responsiveness Summary. The revised risk results do not change the overall conclusions of the ERA Addendum.

Response to EF-1.3

The USEPA agrees that the effect of daily water level changes on PCB exchange and non-scour related movement of PCB-contaminated sediments may add important additional loads that are not specified by any HUDTOX model mechanism. As a result, these mechanisms will not be represented directly in the estimation of PCB loads to the Lower Hudson. However, the HUDTOX model is not strictly mechanistic and incorporates several empirical and semi-empirical components. Specifically, the process or processes responsible for the non-scour PCB loads identified in the RBMR (USEPA, 2000a) have been empirically represented in the model based on the observed data. To the extent that daily water level changes and non-scour related movement are important, much of their effect is captured by this empirical representation. Similarly, the mechanistic components of transport represented in the HUDTOX model rely on long-term records for calibration. To the extent that either process suggested by the commenter is important to the mechanistic components of transport, its effect will be reflected in the adjustments to the model parameters incorporated in the mechanistic expression. Given that both the mechanistic and empirical components of the HUDTOX model are based on long-term records, it is unlikely that any major PCB load has not been represented in the model calibration.

USEPA recognizes that although the model will effectively capture all major loads by the empirical and semi-empirical components, the issue of the exact sources of these loads is less certain.

This uncertainty affects the reliability of the model forecasts to the extent that the release process may change relative to the representation derived from the model. Near-shore/shoreline regions and the possibility of non-scour-related sediment movement can be considered independently of the model results when decision making occurs.

Response to EF-1.4

In the RMBR (USEPA, 2000a), USEPA did not use a generic fish growth rate for lake trout in the FISHRAND model, but rather used species-specific growth rates (see Section III of this Responsiveness Summary). Growth rate is not the most sensitive parameter in the FISHRAND model, but was considered sensitive enough to focus on for calibration. The spottail shiner growth rate was not a sensitive parameter, and there are virtually no data available for this species. Thus, the spottail shiner growth rate is the "generic" growth rate used in the Gobas model. Note that the calibrated growth rates for the other species modeled (e.g., pumpkinseed, brown bullhead, largemouth bass) are very close to the "generic" growth rate used in the Gobas model for all species.

Response to EF-1.5

The fate and transport portion of the Farley et al. (1999) model represents the Lower Hudson River on a finite segment basis, with each one-dimensional segment being 10 miles in length and occupying the whole width of the river. Given this model segmentation and the use of seasonal average flows, the Farley model is appropriate for the calculation of long-term, segment-wide average environmental concentrations, and not appropriate for the fine spatial scale estimates of concentrations within specific habitat types identified by the commenter. Given the scale of the Farley model, it is appropriate to use averaged exposure concentrations within a segment to assess the accumulation of PCBs in biota. The fit between model predictions and observation data could be improved by use of a more detailed spatial and temporal representation of exposure concentrations and individual species feeding patterns, but this is not possible within the context of the Farley et al. (1999) model.

Sediment TOC was assumed to be constant throughout the Lower Hudson. TOC actually varies; however, use of a constant average value is consistent with the segment-averaged nature of the Farley model. Lipid content of fish was also set to a constant value by Farley et al. (1999). It should be noted, however, that in the version of the Farley model used by USEPA, the lipid content of fish was modified from that given in Farley et al. (1999) to reflect the lipid content in fish sampled by NYSDEC in the 1990s (Cooney, 1999).

These simplifying assumptions reflect the fact that the Farley model is designed for prediction of long-term trends in average fish tissue concentrations. USEPA does not consider the model to be appropriate for predicting short-term variability in fish concentrations or response to transient events. Variability in these factors will affect predictions of long-term trends and associated ecological risks only to the extent that the selected average values are biased.

Lipid content in the FISHRAND model is described by a distribution. TOC was set to the single value that is used by the Farley model in order to be consistent with the assumptions of that model. TOC is a relatively sensitive parameter, and shows an inverse relationship with predicted body burdens (i.e., increases in TOC lead to decreases in predicted fish body burden).

Response to EF-1.6

Based on the comments received on the ERA(USEPA, 1999c) and the ERA Addendum (USEPA, 1999c), the studies that were used to derive TRVs were reexamined. This reexamination is detailed in Section III of this document.

Based on the reexamination, 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 sheepshead minnow study by Hansen et al. (1974) was selected for development of the laboratory TRV, instead of Bengsston (1980). Hansen et al. (1974) established a NOAEL for exposure to Aroclor 1254 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 Hansen et al. (1974) are more appropriate for comparison to measured and modeled concentrations in adult Hudson River fish than the Bengsston (1980), which examined hatchability in minnows exposed to Clophen A50. Because the sheepshead minnow is not in the same taxonomic family as any of the Hudson River fish receptors, an interspecies 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, striped bass, and shortnose sturgeon is: 0.93 mg PCBs/kg tissue.
- 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.

The field-based TRVs of the pumpkinseed, spottail shiner, and largemouth bass were also revised. For the pumpkinseed and largemouth bass, the field studies by Adams et al. (1989, 1990, 1992) on the redbreast sunfish, a species in the same family as the pumpkinseed, were retained as the studies to establish TRVs. However, the growth endpoint, rather than the higher fecundity endpoint initially selected, was used to establish a TRV. The NOAEL for growth was reported as being significantly different from one downstream location, but no comparison to the reference sites was provided. Growth is a relevant endpoint, and the NOAEL for growth, 0.3 mg/kg, was selected. 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 because 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.

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

In the ERA Addendum, 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.

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

The associated future risks to fish in the Lower Hudson River using these revised TRVs are presented in Section III.

Response to EF-1.7

The LOAEL and NOAEL values was used together to bracket risk.

As outlined in Section B.2.1 of the ERA Addendum (USEPA, 1999c), both laboratory and field studies have advantages and disadvantages for the purpose of deriving TRVs. For example, a controlled laboratory study can test the effect of a single formulation or congener on the test species in the absence of the effects of other co-occurring contaminants or confounding field conditions. Therefore, greater confidence can be placed in the conclusion that observed effects are related to exposure to the test compound and both NOAEL- and LOAEL-based TRVs can be developed Field studies have the advantage that organisms are exposed to a more realistic mixture of PCB congeners (with different toxic potencies) than laboratory studies. Both types of studies may have the disadvantage that they are conducted on species that are not closely related to the receptors of concern at the Hudson River. Because each approach has both advantages and disadvantages, the ERA Addendum developed TRVs and evaluated risk based on both laboratory and field studies.

If an appropriate field study is available for a species in the same taxonomic family as the receptor of concern, that field study is used to derive a NOAEL TRV. However, in many cases, appropriate field studies were not available for any species that were closely related (e.g., in the same taxonomic family) as the receptor of concern. In such cases, the advantages of a controlled laboratory study, in particular the ability to derive both LOAEL- and NOAEL-based TRVs, were felt to outweigh those of field studies conducted on less closely related species, and a field-based TRV was not developed.

Response to EF-1.8

The TRV selection process specifically focused on the toxicity endpoints of greatest population relevance, i.e, survival, growth, and reproductive capacity. Although immune suppression may have implications on a population level (e.g., reduced capacity to recover from stress), the link to population level effects is far less direct than reproductive and growth effects, such as survival and reproductive capacity.

The FISHRAND model does not consistently underestimate risks; typically predicted results are within the error bars of the data. If there is an underestimate in concentration, it is generally within a factor of two, which would not change the overall conclusions of the ERA Addendum.

Response to EL-1.1

The RBMR (USEPA, 2000a) was not available when the ERA Addendum was released. USEPA provided the commenter a copy of Farley et al. (1999) report and added a copy to each of the 16 information repositories. The final models are documented in the RBMR(USEPA, 2000a). The only documentation for the Farley model is in the March 1999 report (Farley et al., 1999). An update regarding lipid content was made to the Farley et al. (1999) model prior to its use in the ERA Addendum; this update (Cooney, 1999) is noted in the ERA Addendum, the Responsiveness Summary for the ERA, and this Responsiveness Summary for the ERA Addendum. The change to the Farley et al. (1999) model used in the ERA Addendum is minor and does not affect the overall conclusions of the ERA Addendum.

Response to EL-1.2

USEPA reviewed the Farley model to ensure that it was an appropriate tool for use in predicting future risks in the Lower Hudson River. USEPA is not arranging a peer review of the Farley model, although it is USEPA's understanding that the authors of the Farley et al. (1999) report intend to submit their work in a paper to be published in a peer reviewed scientific journal.

Response to EL-1.2a

The data presented in Figure 1-1 of the Farley et al. (1999) report represent results from individual cores. Two results are shown for RM 159 because two different cores are available.

Throughout the freshwater portion of the Hudson there is a significant amount of variability in core profiles from nearby locations, due to local heterogeneity in depositional patterns and different histories of dechlorination. The first core shown in Figure 1-1 appears to reflect a more contaminated location in which a significant amount of dechlorination has occurred. The second core, with lower total PCB concentration in the surface layer, does not have a strong dechlorination signature.

Response to EL-1.2b

USEPA agrees that the Farley model contains a number of simplifying assumptions, such as the use of a constant seasonal pattern of flows based on a typical flow year. These simplifying assumptions are appropriate for the purpose for which the model was built, which was to examine the effects of PCB loading rates on the long-term trajectory of PCB concentrations in fish. Year-to-year variability in flow and sedimentation will cause short-term fluctuations in PCB concentrations, but will have a lesser effect on long term trends. This type of approach is particularly appropriate for the Lower Hudson, which is far removed from the major upstream source of PCBs, resulting in a smoothing out of temporal variations in loads. Further, mixing and sedimentation in much of the Lower Hudson are strongly affected by tidal influences, which are relatively constant from year to year. Regarding model-assigned sediment thicknesses, it should be noted that these are designed to be representative of the vertical profile that potentially interacts with the water column, and not actual sediment thickness. The assumptions are considered to be appropriate, and changing these values by small amounts would have little effect on model results. It is incorrect to say that sedimentation rates and sediment loads are assigned by Farley "with little or no justification." The sources of data are cited on pp. 26-27 of Farley et al. (1999), along with a discussion of the rationale for extrapolating or interpolating results. The resulting parameters are reasonable based on the best available data; however, it is true that detailed data were not available to fully constrain specific components, such as annual sediment loads from individual tributaries. It is also incorrect to imply there is "little or no justification" for the model representation of organic carbon, as this was taken from the detailed System-Wide Eutrophication Model (SWEM) effort completed by HydroQual. Finally, USEPA agrees that the PCB loads from the New Jersey tributaries are subject to high degree of uncertainty. These loads, however, appear to be of minor significance relative to loads from other sources, particularly for Food Web Regions 1 and 2 of the Farley model, which are upstream of the New Jersey tributaries.

Response to EL-1.2c

It is misleading to say that these components were specified "rather than" modeled. Page 18 of Farley et al. (1999) report states that these components were not directly simulated within the Farley model itself. Instead, they were "specified based on field observations, other modeling work, or simple mass balance calculations...". In fact, hydrodynamics and organic carbon were represented in the Farley model based on the detailed SWEM modeling effort. The sediment transport component combines SWEM hydrodynamics with a solids mass balance

Response to EL-1.2d and EL-1.2e

It is a common feature of essentially all modeling efforts that more data are desired than are available. The Farley model lacks field data for comparison at certain locations, for certain conditions, and times. However, available data were used to evaluate calibration of the model. The lack of additional data for calibration increases uncertainty in model results, but does not invalidate the approach.

Response to EL-1.2f

The Farley model is based on the earlier modeling effort of Thomann et al. (1989). For the recalibration of the bioaccumulation model, Farley et al. (1999) adjusted only three parameters in the model: the volatilization rate coefficient, the gill transfer efficiency, and the phytoplankton uptake rate/growth rate ratio. (Several other parameters were adjusted relative to the original Thomann Model, c.f. Table 2-2, but based directly on new data rather than calibration.) USEPA disagrees with the comment that this constitutes a "large number of parameter adjustments;" rather it represents a parsimonious set of parameters.

Response to EL-1.2g

Results presented by Farley et al. (1999) show some discrepancies in homologue distribution between model results and high-resolution sediment core data collected by USEPA in 1992 (Figure 3-5), although total surface sediment concentrations are well replicated (Figure 3-4). In part, the small error in replicating concentrations in individual cores is due to the expected spatial variability among sediment samples. Despite this spatial variability, the Farley et al. (1999) results appear to underestimate the dichlorobiphenyl and trichlorobiphenyl sediment concentrations on a fairly consistent basis (e.g., Figure 3-5). These discrepancies are believed to be largely due to Farley's assumptions for upstream loads over Federal Dam, which were replaced with HUDTOX generated loads by USEPA (USEPA, 2000a). As shown in Table 3-3 and Figure 3-2 of the ERA Addendum (USEPA, 1999c), the HUDTOX model output produces higher loads of dichlorobiphenyl and trichlorobiphenyl than the estimates used by Farley et al. (1999) for the period prior to 1992.

Response to EL-1.2h

Figure 3-14 in Farley et al. (1999) shows a fairly good fit between model predictions and observations in white perch, with most model predictions lying within confidence bounds on sample data. For Food Web Region 1, it appears that the model overpredicts more than underpredicts NYSDEC white perch data, while closely replicating the USEPA/NOAA white perch data. This is in large part due to the fact that Farley et al. (1999) did not account for the effects of different analytical methods between the USEPA/NOAA and NYSDEC results. In any case, the results reported by Farley et al. have been superseded by results using the revised HUDTOX load estimates across Federal Dam. Farley model predictions using the HUDTOX-generated load are compared to observed body burdens in white perch in Figure 3-8 of the ERA Addendum.

Response to EL-1.3

The USEPA fate, transport and bioaccumulation models are designed to capture the general trends in the concentrations of PCBs in river water, sediment, and fish. This is demonstrated by comparing the model output to calibration data. There is good agreement between the model hindcast and observed data. These models do not examine cases such as the nearshore environment or PCB load variations caused by underestimating resuspension. This is an added uncertainty to the risk assessments, not a modeling uncertainty.

Additionally, the USEPA has already recognized the importance of Lower Hudson River contributions of PCBs (USEPA, 1997). However, as also noted by Farley et al. (1999), lower river contributions are restricted to the region below the salt front. Thus, USEPA's statements that the Upper Hudson River is the only significant source of PCBs to the freshwater Lower Hudson are correct.

Response to EL-1.4

USEPA agrees that the framework and methodology used for the Upper Hudson and Lower Hudson exposure and toxicity assessments are consistent. USEPA's response to comments on the ERA regarding items such as exposure and the toxicity assessment are addressed in the Responsiveness Summary for the ERA (USEPA, 2000b).

Response to EG-1.1

Although historic data for some species exist from the last 25 years, changes in populations and communities can not viewed from a strictly PCB-oriented perspective. The fishing ban and overall improvement in water quality have contributed to population increases in some fish species; however, this does not indicate that there are no adverse effects from PCBs.

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 variability of the ecosystem. The kinds of effects expected from PCBs include reduced fecundity, decreased hatching success, and similar kinds of reproductive impairment indicators, which are often difficult to detect. The gradient of PCB concentrations along the roughly 200 miles of river being examined in the Reassessment RI/FS also increases the difficulty of ascribing particular effects to PCBs. Therefore, the ERA Addendum discusses the potential for adverse effects even in apparently healthy receptor populations.

Part of the difficulty of assessing receptor populations is that there are no data against which to measure abundance. Limited breeding success in bald eagle nests along the Hudson does not establish that the bald eagle is re-established there. Since these eagles are the first to breed in approximately one hundred years, there is no appropriate reference population for comparison. In addition, all eagles now breeding in NYS are the result of NYSDEC or other direct release/restoration programs (Nye, 2000).

As noted in the Responsiveness Summary for the ERA (see, USEPA, 2000b response to EL-1.1), population-specific information prior to PCBs, the fishing ban, and other anthropogenic influences would be valuable in assessing the effects of PCBs on today's population. However, the time frame necessary for these data is far longer than the nine years since the initiation of the ERA.

Response to EG-1.2

This comment on USEPA's ecological risk assessment approach has been addressed in the Responsiveness Summary for the ERA (USEPA, 2000b). Responses to general points provided in the Responsiveness Summary for the ERA also apply to the ERA Addendum (see responses to the following comments: EG-1.38 (general response), EG-1.2 and EG-1.4 (inadequate consideration of population versus individual-level effects), EG-1.12 (improper use of weight of evidence approach), EG-1.5, EG-1.8, EG- 1.9, and EG-1.11, EG-1.-31, EG-1.32, and EG-1.33 (ignoring or dismissing site-specific data), EG-1.6 (use of sediment guidelines and criteria as a measurement endpoint), EG-1.1, EG-1.13, EG-1.15 and EG-1.18 (conservative exposure and effects assumptions), EG-1.27 and EG-1.29 (TEQ Approach), EG-1.19 (NOAA's expert review of PCB effects on fish)).

Response to EG-1.3

Consistent with USEPA guidance, the purpose of the ERA Addendum is to assess ecological risk posed by site-related contaminants, and does not include decision-making or risk management. USEPA will evaluate risk reduction in the Feasibility Study and Proposed Plan for the Reassessment.

EXECUTIVE SUMMARY

Response to EF-1.9

For clarity, p. ES-9 is corrected to read "limited breeding success" (of bald eagles) rather than "lack of breeding success" (fourth paragraph, last sentence). The first and second sentences of the fifth paragraph on p. ES-11 is corrected to read, "Collectively the evidence indicates that future PCB exposures (predicted from 1993 to 2018) may impair reproduction or recruitment of threatened or endangered species. Using the TEQ-based..."

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 Report Organization

No significant comments were received on Section 1.2.

2.0 PROBLEM FORMULATION

Response to EG-1.4

USEPA's bottom-up approach uses data on individuals in order to predict potential effects on local populations and communities that occur or could occur at the site. USEPA's Risk Management Guidance (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 River, in a weight-of-evidence approach to characterize risks to ecological receptors (see, ERA Addendum, Section 5.0: Risk Characterization. Also see response to EG-1.1 in USEPA, 2000b).

The life span of each receptor examined in the ERA Addendum is less than 25 years (with the exception of the shortnose sturgeon), indicating that the 25-year modeling duration covers the life span of most receptors. Life span information is presented below.

Fish: largemouth bass - up to 15 years (Smith, 1985); pumpkinseed sunfish - 8 to 10 years (in Canadian populations) (Scott and Crossman, 1975); brown bullhead - 6 to 7 years (Smith, 1985); yellow perch - 9 years (Smith, 1985); white perch - 5 to 7 years; some live 14 to 17 years (Smith, 1985); spottail shiner - 4 years (Pflieger, 1997); striped bass - Smith (1988) reports the oldest fish studied at 14 to 18 years and Cooper (1983) reports a single female estimated to be 30 years; shortnose sturgeon - 25 to 30 years, sometimes more (Dovel, 1981).

Birds (maximum longevity): tree swallow - 10 years; mallard - 26 years; belted kingfisher - 16 years (note: no species-specific information was available so the oldest nonpasserine land bird was used [red-cockaded woodpecker]); great blue heron - 23 years; and bald eagle - 22 years (Klimkiewicz, 1997).

Mammals: little brown bat - 6 to 7 years; raccoon less than 5 years; mink - up to 10 years; and river otter - up to 23 years (Walker, 1997).

The toxicity quotient (TQ) exceeds one (on a Tri+ and/or TEQ basis) in the Lower Hudson River for the life span of the pumpkinseed, brown bullhead, yellow perch, white perch, largemouth bass, striped bass, mallard, belted kingfisher, great blue heron, bald eagle, little brown bat, raccoon, mink, and otter. These exceedances indicate that population level effects are possible in these species.

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 Lower Hudson River Ecosystem

No significant comments were received on Section 2.3.1.

2.3.2 Ecosystems of the Lower Hudson River

No significant comments were received on Section 2.3.2.

2.3.3 Exposure Pathways

No significant comments were received on Section 2.3.3.

2.3.3.1 Aquatic Exposure Pathways

No significant comments were received on Section 2.3.3.1.

2.3.3.2 Terrestrial Exposure Pathways

No significant comments were received on Section 2.3.3.2.

2.4 Assessment Endpoints

No significant comments were received on Section 2.4.

2.5 Measurement Endpoints (Measures of Effect)

Response to EG-1.5

The comparison between the Hudson River ERA and the Clinch River ERA was addressed in the response to comment EG-1.1 in the Responsiveness Summary for the ERA (USEPA, 2000b). Although the Hudson River and the Clinch River are both large contaminated sites, they are not directly comparable. The Oak Ridge Reservation (ORR) is owned and administered by the Department of Energy (DOE) 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, since others factors (e.g., fishing ban) may mitigate some of the effects of PCBs (see, Responsiveness Summary for the ERASOW [USEPA, 1999b] at p. 13). Due to concerns that a top-down approach would not be protective of biological resources of the Hudson River, more weight was placed on use of toxicity quotients.

USEPA's bottom-up approach uses data on individuals in order to predict potential effects on local populations and communities that occur or could occur at the site. USEPA's Risk Management Guidance (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 River, in a weight-of-evidence approach to characterize risks to ecological receptors (see, ERA Addendum, Section 5.0: Risk Characterization. Also see response to EG-1.1 in USEPA, 2000b).

Conducting various studies on the Lower Hudson River beyond what NYSDEC, US Fish and Wildlife Service, 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, as most of the data used were collected in 1993. The decision not to conduct new site-specific toxicity studies was described in the Responsiveness Summary for the ERA Scope of Work (see, USEPA 1999a, p. 27).

2.6 Receptors of Concern

No significant comments were received on Section 2.6

2.6.1 Fish Receptors

No significant comments were received on Section 2.6.1.

2.6.2 Avian Receptors

No significant comments were received on Section 2.6.2.

2.6.3 Mammalian Receptors

No significant comments were received on Section 2.6.3.

2.6.4 Threatened and Endangered Species

No significant comments were received on Section 2.6.4.

2.6.5 Significant Habitats

No significant comments were received on Section 2.6.5.

3.0 EXPOSURE ASSESSMENT

<u>Response to EF-1.12</u>

Estimation of striped bass body burdens was based on wet weight.

<u>Response to EG-1.6</u>

The TQ approach used in the ERA Addendum was described in the September 1998 Scope of Work for the ERA (USEPA, 1998b). USEPA noted that in the ERA it would address the uncertainty associated with using reference concentrations derived from the scientific literature, rather than from site-specific toxicological studies (<u>see</u>, Responsiveness Summary for the ERA Scope of Work [USEPA, 1999a] at p. 27). This issue is addressed in the ERA (USEPA, 1999b) at pp. 157-158.

The use of TQs is part of USEPA's bottom-up approach that uses 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."

Response to EG-1.11

USEPA disagrees that the models used to predict future PCB concentrations are deficient. The revised HUDTOX and FISHRAND models (USEPA, 2000a) were peer reviewed by a panel of independent experts and found to be acceptable with revisions. USEPA is currently evaluating the recommendations of the peer review panel and will issue a written response to the reviewers' recommendations.

Response to EG-1.12a

The commenter raises several concerns regarding the use of the HUDTOX model to estimate PCB loads delivered to the Lower Hudson via the Troy dam. These comments were addressed in the Responsiveness Summary for the BMR (USEPA, 2000c). Of relevance are the responses to GE's comments on PCB fate and transport issues (BG-17 through BG-20); description of model development (BG-21 through 28); and prediction of water column levels (BG-1.32). Also, the commenter questions whether the HUDTOX model is consistent in its use of equations and coefficients, whether USEPA converted Tri+ loads to individual homologue groups on an adequate data set, and whether some of the data used may be limited in usefulness. With regard to the first concern, the HUDTOX model consistently applies a set of mathematical formulations to the entire Upper Hudson. Different equations are not used in different river regions. However, the model does apply different coefficients among the various reaches to account for the observed differences in PCB loading and other conditions that vary as a function of river reach. These adjustments were derived during the calibration of the HUDTOX model to the data (see, Sections 6 and 7 of the RBMR (USEPA, 2000a) for discussions of the data used and the model calibration, respectively).

Response to EG-1.13

The commenter asserts that the Farley model is largely untested and therefore the uncertainty associated with the model's veracity undermines its application in the ERA Addendum. However, USEPA reviewed the Farley model specifically for use in the ERA Addendum. The USEPA acknowledges that the data set available to calibrate a PCB fate and transport model in the Lower Hudson is limited. However other data and analyses are available (USEPA, 1997; USEPA 1999c) that independently confirm the conclusions drawn from the modeling analysis. For example, the conclusion that the principal source of PCBs to the Lower Hudson is the Upper Hudson is directly supported by the high-resolution core analysis presented in the DEIR (USEPA, 1997). Similarly, the gradual decline in concentration of PCBs in surface sediment as shown in Figure 3-7 of the ERA Addendum is confirmed by the analysis of the high-resolution cores presented in USEPA (1997). Additionally, although this version of the model has not yet been subjected to peer review, earlier versions of the model developed by Thomann were peer reviewed and published (see, Thomann et al., 1989 and Thomann, 1989). It is USEPA's understanding that the authors of Farley et al. (1999) will submit their work in a paper for publication in a peer reviewed scientific journal.

The comment that the model is biased toward the lower chlorinated congeners ignores the application of the model in the ERA Addendum. The original model presentation is not "biased" but simply based on the measured loads at the TI Dam as reported by GE. As documented in the ERA Addendum (USEPA, 1999c) and the RBMR (USEPA, 2000a), the load measured at the TI Dam is not the same as that delivered at Waterford. A significant change takes place during transit, largely resulting from the loss of lighter congeners from the water column. To account for this, in the ERA Addendum USEPA utilized loads derived from the HUDTOX model runs, which addressed these losses and more correctly estimated the homologue patterns at Waterford. The effect of this correction can be seen in Figure 3-2 of the ERA Addendum, which compares the cumulative loads as estimated from HUDTOX and by Farley et al. (1999). As a result of the correction, the proportion of dichloro homologue in the Waterford load is greatly decreased. Figure 3-6 shows that while the model estimates of water column concentration are low, the proportions of the various homologues as predicted by the model are very similar to those measured in the water column. Thus the model is not "biased toward lower chlorinated congeners."

The USEPA acknowledges that the period of data available for the fate and transport model is largely limited to one year. However, an integrative test of the model for the purposes of risk assessment is provided by the long-term record of fish body burdens obtained by NYSDEC. This data set suggests that the combined fate, transport, and bioaccumulation models are able to predict fish body burdens within an acceptable range of accuracy for the ERA Addendum, given that risk is largely resolved on an order-of-magnitude basis.

Lastly, USEPA disagrees that high resolution cores should be incorporated in Figure 3-7 of the ERA Addendum. The high resolution cores represent unusual depositional environments, whereas the purpose of Figure 3-7 is to depict more spatially representative sediment samples. For this reason, the sediment samples collected for the ERA and ERA Addendum are included in Figure 3-7. Nonetheless, even these data set are not truly representative of all sediment environments. While these data show general agreement with modeled concentrations of PCBs in sediment, the data are not intended as a calibration point for the model.

Response to EG-1.14

USEPA does not agree that differences between the Farley model and FISHRAND must be reconciled before reliable forecasts can be made. Any model of PCB bioaccumulation involves approximations (i.e., empirical assumptions and coefficients) regardless of the degree of mechanistic representation. As demonstrated in Appendix K of the ERA (USEPA, 1999b), fish body burdens resulting from Lower Hudson River exposures cannot be exclusively linked to either their sediment or water exposures. As noted in the DEIR (USEPA, 1997), sediment and water column concentrations are not independent because resuspension and sediment release yields PCBs to overlying water, which can then serve to redeposit these PCBs in quiescent regions downstream.

Given that these matrices are linked, it is possible that the use of the semi-empirically determined coefficients would yield similar results from two different mechanistic representations, especially when considered over the long term. The only reliable test as to the veracity of the model forecasts is the degree to which the model replicates existing data to within an acceptable level of error. Both model forms (FISHRAND and Farley) have been shown to do this.

The commenter also suggests that the striped bass body burdens as estimated from the largemouth bass underestimate contributions from the saline portion of the Lower Hudson and consequently over predict site-related risks. As noted by the commenter, the FISHRAND model was not used to characterize striped bass body burdens directly. Rather, an order-of-magnitude approximation was used based on an empirical, data-based approach (i.e., ratios of PCB concentrations in striped bass and largemouth bass from fish data for these two species sampled at the same location and same time frame). This empirical ratio approach captures the current relationship between the freshwater and saline regions without making an explicit assumptions regarding the striped bass migratory patterns with a level of precision that is appropriate for use in the ERA Addendum. In the freshwater region of the Lower Hudson, striped bass body burdens are clearly dominated by PCBs from the Upper Hudson, as shown by an increase in body burden with increasing river mile towards the GE plants, which are the source of the PCBs in the Upper Hudson River sediments (see Table EG-1.14 below). Given that the Upper Hudson source dominates in the Farley Food Web Region 1, the relative proportion of PCBs from the freshwater and saline Hudson is unimportant in this region.

Table EG-1.14
Comparison of Mean Striped Bass Body Burdens at Two Long-Term Monitoring Locations (Data from NYSDEC)

Year	RM113	RM152
1990	4.64	9.02
1991	NA	NA
1992	2.94	15.32
1993	3.27	10.92
1994	2.30	5.61
1995	1.11	NA
1996	1.66	4.28

Response to EG-1.15a

As shown in Figure 3-2 of the ERA Addendum (USEPA, 1999c), the Farley model results derived using HUDTOX loads to the Lower Hudson tend to yield slightly lower values before 1992 and slightly higher values after 1992 relative to the available data. Given that the Farley model was calibrated using the PCBs loads estimated by Farley et al. (1999) rather than HUDTOX, a difference in values would be expected. The slight lack of agreement in PCB loads has a minimal effect on the estimate of risks to striped bass because the uncertainty introduced is minimal and does not change the overall conclusions regarding risk to striped bass. Moreover, the model output for striped bass aged 6-16 years in Food Web Region 2 is not the only metric used to measure risks to striped bass in this region (body burden estimates for striped bass aged 2-6 years also were used). Thus, USEPA's decision to use the Farley model results with the HUDTOX PCB loads was appropriate in the ERA Addendum.

Response to EG-1.15b

The commenter suggests that the fact that the FISHRAND model does not yield fish body burdens reflecting an impact of the 1991-1992 Allen Mills event, which in turn indicates that exposure point concentrations and food web structure may be inaccurate in the model. However, the data referred to by the commenter do not present a consistent picture of response to an increased load from the Allen Mills event. Specifically, body burdens (wet-weight basis) for white perch and brown bullhead peak in 1992, largemouth bass peak in 1993 and yellow perch steadily increase from 1991 to 1996. Young-of-the-year pumpkinseed at RM 142-152 (Figure 3-12c) show only a gradual decline in both wet-weight and lipid-normalized PCBs from 1988 to 1996, with 1993 being the lowest value recorded. Lipid-normalized results reflect similar disagreement among the species: largemouth bass data from 1992 are not different from 1990 data; white perch, brown bullhead and yellow perch peak in 1992 but are similar to the values seen in prior and subsequent years. These data suggest that the Allen Mills pulse is of relatively minor importance over the long-term.

Predicted concentrations from the USEPA's FISHRAND model represent annualized concentrations, which tend to average out varying concentrations throughout the year. Thus, the response to the pulse loads may be partially smoothed by the approach. (The fish data represent conditions during a few brief weeks of sample collection.) Additionally, the Farley model is not designed to represent short-term behavior of PCB fate and transport. Rather, it is focused on long-term trends. Nonetheless, USEPA is satisfied that the model is reasonably able to capture the long-term trend and it therefore appropriate for use in the ERA Addendum. USEPA does not plan to use the Farley model to quantitatively assess the effects of remedial activities in the Upper Hudson on the Lower Hudson.

Response to EG-1.15c

In the ERA Addendum, FISHRAND model comparisons to data on a wet-weight basis were emphasized over lipid-normalized predictions (Figures 3-12a,b, c, and d). This is because risk estimates for all avian and mammalian receptors attributable to ingestion of PCB-contaminated fish are based on predicted wet-weight concentrations. (Lipid-normalized results are used to evaluate TEQ-based risks to fish). Additionally, a single version of the model for largemouth bass was applied to all of the freshwater Lower Hudson. In this manner, the model was calibrated to meet available data at both RMs 152 and 113. The model provides acceptable output on a wet-weight basis at both locations and tends to underestimate lipid-normalized results, particularly at RM 152. The lipid contents used in the model were obtained from NYSDEC and were based on monitoring data. The potential underprediction in comparisons to data suggests that true TEQ-based risks to fish may be underestimated at RM 152.

Response to EG-1.16a

The largest piscivorous ecological receptors, represented by bald eagle and river otter, will consume any fish within a particular size range and favor large fish. 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 while larger fish included catfish and bass (see, USEPA 1999c, p. 52). The FISHRAND model predicts population distributions of PCB concentrations in several larger fish species, including largemouth bass, brown bullhead, white perch and yellow perch. Because there were no data available to suggest preferential fish selection by these two species (i.e., no data to suggest that otter or eagle preferentially consume white perch, for example), the largemouth bass was used as a surrogate species to represent larger piscivorous species likely to be consumed by otter and eagle. Largemouth bass are the most abundant species in the river and are more likely to inhabit the nearshore areas that serve as foraging areas for the otter and the eagle.

The remaining ecological receptors have been shown to consume a much smaller size of fish. For these species, the predicted population distributions of PCB concentrations in the forage fish were used, as modeled for pumpkinseed and spottail shiner. Again, as there was no information available to suggest preferential selection of these fish by any of the ecological receptors, the spottail shiner was chosen as the representative surrogate forage fish. Most of the data in the Lower Hudson River were available for the spottail shiner, and the spottail shiner has been shown to be one of the most abundant forage fish species. Note that predicted PCB concentrations in spottail shiner are very similar to those predicted for the pumpkinseed, suggesting it is appropriate to use one species as representative of typical expected PCB concentrations in forage fish generally.

Response to EG-1.16b

The FISHRAND model was calibrated in the Upper Hudson River using growth rate coefficients for each species as a calibration parameter. Consequently, the growth rates used in the FISHRAND model were not the generic growth rates of the Gobas Model (except for spottail shiner, which is not sensitive to this parameter and for which no data are available, see Response to EF-1.4).

3.1 Quantification of PCB Fate and Transport: Modeling Exposure Concentrations

No significant comments were received on Section 3.1.

3.1.1. Modeling Approach

No significant comments were received on Section 3.1.1.

3.1.1.1 Use of the Farley Model

Response to EF-1.10

The models presented in the RBMR (USEPA, 2000a) address the potential impact of high flow events on the Remnant Deposits and other areas of relatively high concentrations of PCBs upstream from Rogers Island by modeling 0 ng/L and 30 ng/L upstream boundary conditions in addition to 10 ng/L. The largemouth bass body burdens for the 10 ng/L and 30 ng/L conditions are given in Tables 7-10 and 7-13 of the RBMR (USEPA, 2000a). These figures show approximately a threefold increase in concentration in the long term "steady-state" value (asymptote) between upstream boundary conditions of 10 ng/L and 30 ng/L. Risks vary in direct proportion to concentration, resulting in a threefold increase in risk at 30 ng/L. These increases in concentration and risk are also seen for the brown bullhead, Tables 7-11 and 7-14 of the RBMR (USEPA, 2000a). In the Lower Hudson, the system response is expected to be less than this one-to-one response estimated for the Upper Hudson. That is, body burdens will not respond in a strict proportional fashion to changes in the Upper Hudson load, because the local sediment inventory resulting from previous GE related contamination is expected to continue to effect fish levels for the entire forecast period.

Response to EF-1.11

The Farley model output is sediment, water column, and fish body burdens. The modeled species are white perch in Food Web Regions 1 and 2 and striped bass in Food Web Region 2 only. The FISHRAND model output is for pumpkinseed, spottail shiner, yellow perch, white perch, brown bullhead and largemouth bass. Striped bass concentrations in Food Web Region 1 were estimated from FISHRAND forecasts for PCBs in largemouth bass, based on the ratio between the two species in sample data for the same location and time period (see, Section 3.1.1.2 of the ERA Addendum). PCB concentrations in white perch from the Farley model were used in the ERA Addendum in preference to the FISHRAND output, because the Farley model has been designed specifically for migratory species.

Response to EL-1.6

As discussed in Section 3.1.1.1 of the ERA Addendum, the Farley model was used with few adjustments to predict future concentrations in Lower Hudson River sediment, water, and fish. The

PCB loads from HUDTOX were converted from Tri+ PCBs to di through hexa homologues. These homologue values replaced the Upper Hudson River PCB loads that were used in Farley et al. (1999) and provide consistency between the Upper and Lower Hudson for purposes of the ERA and ERA Addendum. Because the Farley model was designed to run for a period of up to 15 years, the models were run four times to generate a 40-year forecast. The Upper Hudson River PCB loads from the HUDTOX model for the appropriate time period were put into each of the four sets of input files. The initial conditions for each model segment for the fate and transport models and species for the bioaccumulation model were updated with data from the previous model run. The final concentration in a segment (or species) becomes the initial concentration in the same segment for the next model run. Each initial condition is determined by the previous model run, which is no different than running the model on a continuous basis for the 40-year simulation period. This 15-year step approach is simply a requirement of the original model coding and does not affect the model results.

3.1.1.2 Use of FISHRAND

Response to EF-1.13

USEPA's comparison of the Farley model output to August and September 1993 water column data suggests that the model underpredicts late summer water column concentrations (but not spring concentrations). A possible reason for this discrepancy may be an overestimation of water column losses to volatilization. However, this inference is based on only three observations (i.e., data points), which are not sufficient to clearly diagnose the existence of a bias, given the temporal variability typically observed in water column concentrations. Different PCB loads at the Federal Dam upper boundary loads is also a possible explanation. It should also be noted that the Farley model is not strictly mechanistic, does not use fine-scale dynamic simulation, and is calibrated to the available fish data, which integrate over time. Thus, while the Farley model may not represent seasonal changes in water column concentrations, it does appear to do a reasonable job in replicating average concentrations in fish.

In addition, overestimating loss of PCBs from the water column is likely to reduce water column concentrations, but this loss is likely to occur for the lighter chlorinated and less toxic congeners, which do not tend to accumulate in tissue. Thus, although the loss may be overestimated (which would underestimate risk), if the loss occurs primarily for these lighter chlorinated and less toxic congeners, as would be expected, then there would be little or no effect on estimated risks.

3.1.1.3 Comparison to the March 1999 Farley Model (1987-1997)

Response to EL-1.7a

All fish body burdens, except for striped bass and white perch, are modeled using the FISHRAND model. The FISHRAND model provides species-specific results for several different species, while the Farley model only provides two to three estimates, depending on river region.

Specifically, the Farley model provides estimates for white perch in all regions, a generic forage fish in all regions, and striped bass in Food Web Regions 2 to 5. As noted by the commenter, the Farley model does not provide striped bass estimates in Food Web Region 1, which is between RMs 152 and 73.5. Yet, because PCB concentrations in striped bass were needed in this region, it was necessary to apply the FISHRAND model in this region (using a ratio approach based on largemouth bass). The fact that Farley et al. (1999) did not develop a striped bass estimate for Food Web Region 1 is likely a result of the model's focus on the saline portion of the Lower Hudson and New York Harbor. It is not for lack of data above RM 73.5, as NYSDEC collected over 200 adult striped bass samples from 1977 to 1997 at RM 152 and RM 113. As to its use in the ERA Addendum, the FISHRAND model provides a finer spatial scale than does the Farley model, and to be consistent with how body burdens are presented for the other species, the ERA Addendum uses the FISHRAND results. Although the Farley model provides sufficient detail for sediment and water column estimates, it does not provide sufficient fish body burden data to support the requirements of the ERA Addendum (and the Mid-Hudson Human Health Risk Assessment).

Response to EL-1.7b

As discussed above and in response EL-1.14, the locations at RM 152 and 113 represent the primary fish sampling locations for all species in the freshwater Lower Hudson and thus form the main calibration and forecast locations in this region of the river.

Response to EL-1.7c

The bald eagle and river otter, representing the largest top-level piscivores, are assumed to eat the fish portion of their diet as largemouth bass, as these are the fish most likely to be caught by these receptors. All other ecological receptors that consume fish (i.e., mink, raccoon, great blue heron, belted kingfisher) consume a smaller fish, which was assumed to be the pumpkinseed. Thus, adult striped bass are not included in subsequent higher trophic level avian and mammalian exposures.

Response to EL-1.7d

The ratio approach was designed to approximate PCB concentrations in striped bass because the Farley model did not provide striped bass results for Region 1 despite the documented occurrence of striped bass at this location (i.e., RM 152, RM 113) and because the FISHRAND model was not parameterized for striped bass. The ratio approach was based on the observed relationship between measured PCB concentrations in striped bass and in largemouth bass from the historical NYSDEC data. There are not sufficient data available for striped bass and largemouth bass to compare concentrations at RMs 90 and 50. The Farley model provides predicted concentrations of PCBs in striped bass for the region between RMs 73 and 33, and thus the ratio approach was not required there.

With regard to the use of the largemouth bass, Figure K-17 of the ERA (USEPA, 1999b) shows that the largemouth bass (piscivores) has the most gradual decline of any feeding guild. Thus, while the use of the ratio approach adds uncertainty, it is unlikely that this approach is unduly conservative.

In constructing the ratios, only adult fish for each species were used. The difference between the ratios for RMs 152 and 113 may be attributable to the migratory behavior of the striped bass relative to the resident behavior of largemouth bass. However, given that this approach is purely empirical, it is of little importance as long as the body burdens are correlated (see also, response to EL-1.14).

Response to EL-1.7e

The FISHRAND model also addresses the age of the animal by approximating the distribution of fish sizes and ages in the population. This is described in detail in the RBMR (USEPA, 2000a).

Response to EL-1.8

Revised Table 3-3 and Figure 3-2 are given in Table EL-1.8 and Figure EL-1.8, respectively, which show consistent units for the Upper Hudson River PCB loads presented in the RBMR (USEPA, 2000a). Revised text for Section 3.1.1.3, paragraph 4 of the ERA Addendum is as follows:

Comparison of HUDTOX and Farley et al. (1999) PCB Load Estimates at the Federal Dam

The revision of the flux of PCBs over the Federal Dam at Troy is the only modification made to the March 1999 Farley fate and transport model for the ERA Addendum and Mid-Hudson HHRA. The difference in magnitude between Farley's original flux estimate and that derived from the HUDTOX model can be seen in Table 3-3. This table shows the two estimates of the PCB homologue loads. The cumulative tri-through-hexa-load estimates over the Federal Dam from the Farley model compare favorably with the estimates from HUDTOX for the period 1987-1997. The largest difference among the tri to hexa homologues is 278 kg for the tetra homologue, representing a cumulative difference of about 13 percent relative to the estimate by Farley et al. (1999) (see, Table 3-3). Conversely, the estimates for the di homologue differ by a greater amount, 428 kg (36 percent relative to Farley et al. 1999). The Farley et al. (1999) model used the General Electric Company water column samples at TI Dam to estimate all homologue loads during the calibration period. As described in Appendix A and presented in Table A-2, the di homologue fraction based on HUDTOX was calculated from the Tri+ PCBs by applying a ratio developed from the USEPA Phase 2 water column data. Notably, the largest differences are for the di homologue, which matters least to Lower Hudson fish body

Table EL-1.8
Cumulative Loads Over the Troy Dam (kg)

Homologue	Farley Model	HUDTOX Converted According to Appendix A	Difference
Di	1182	1610	428
Tri	2320	2097	-224
Tetra	1664	1386	-278
Penta	715	617	-98
Hexa	270	220	-50
Total 1987-1997	6151	5930	-222

Homologue	Farley Model	HUDTOX Converted According to Appendix A	TI Dam Estimate from the DEIR[1]
Di	857	566	638
Tri	1645	856	1072
Tetra	1081	593	672
Penta	406	249	214
Hexa	145	83	65
Total 4/91-2/96	4134	2348	2662

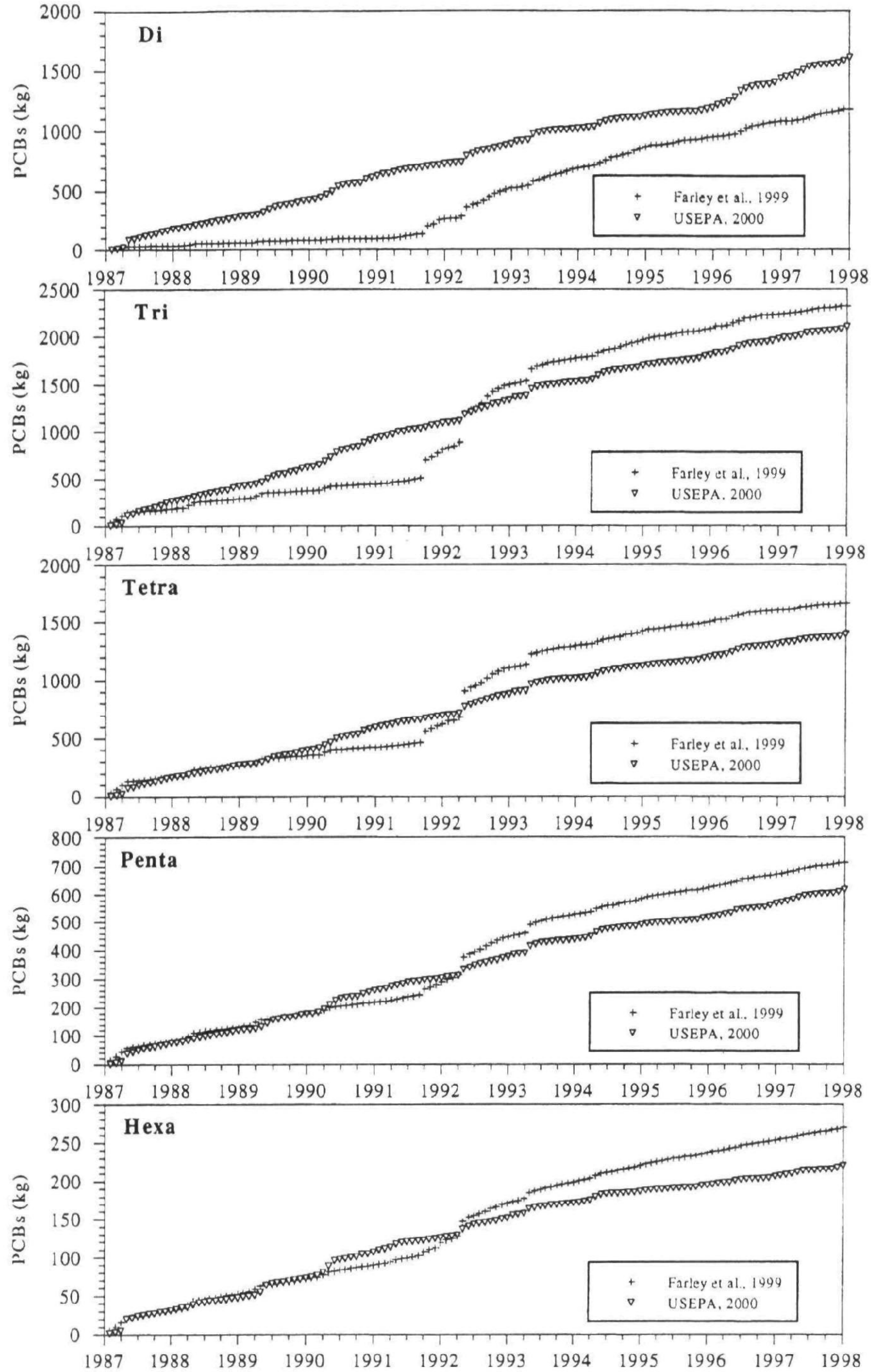
Note.
1. Homologue loads were recalculated using the averaging estimator formula described in the DEIR (USEPA, 1997) as originally given in Dolan et al (1981):

$$\boxed{L_m = \sum_{j=1}^{N_m} q_j \left[\sum_{i=1}^{n_m} \frac{c_i}{n_m} \right]}$$

where.

L_m is the load estimate for month m;
N_m is the number of days in month m,
q_j is the daily mean flow on day j,
n_m is the number of days on which PCB observations were made during month m, and
c_i is a measured concentration within the month

Reanalyzed GE water column data are used (GE, 2000) In addition the TI Dam bias is accounted for as described in Appendix C of the Responsiveness to the Low Resolution Sediment Coring Report (USEPA, 1998)



Sources: Farley et al., 1999 and USEPA, 2000

Figure EL-1.8 (Revised Figure 3-2)
Comparison of Cumulative PCB Loads at Waterford from Farley et al., 1999 and USEPA, 2000

burdens. It is noteworthy as well that the cumulative HUDTOX loads are closer to the load estimates made on a strictly statistical basis, as presented in the DEIR (USEPA, 1997).

The second part of Table 3-3 compares three estimates of the PCB loads from April 1991 through February 1996 all of which are derived from data obtained by GE's federally mandated Post-Construction Remnant Deposit Monitoring Program. The estimates are from Farley et al. (1999), a conversion of HUDTOX model output (ERA Addendum) and the DEIR (USEPA, 1997). The DEIR estimate in Table EL-1.8 has been updated to account for the TI Dam bias issue (USEPA, 1999d). The Farley Model and DEIR estimates are of loads over the TI Dam assuming no gain or loss of PCBs between the TI Dam and the Troy Dam while the HUDTOX estimate uses the modeled concentrations over the Troy Dam. The HUDTOX estimate is the lowest estimate for di through hexa homologues at 2,348 kg versus 4,134 kg for the Farley Model estimate and 2,662 kg for the DEIR estimate. For the di homologue, the converted HUDTOX value is lower (72 kg) than the DEIR estimate and substantially less than the Farley estimate (291 kg). For tri, tetra and penta homologues, the converted HUDTOX estimates are considerably less than the Farley Model estimate, but similar to the DEIR estimate. The higher Farley Model load estimate is based on the original GE data which was not corrected for the TI Dam bias. The corrected data (QEA, 1998) were used for the HUDTOX and revised DEIR estimates.

3.1.1.4 Comparison Between Model Output and Sample Data

Response to EF-1.14

The text on p. 20, paragraph 3 of the ERA Addendum is corrected to read:

Modeled surface sediment concentrations from 0-2.5 cm and 2.5-5 cm are plotted against the USEPA Phase 2 ecological samples (approximately 5 cm in depth). The modeled data fall within the range of the sampled concentrations for all RMs except for RM 59, which falls above and below the modeled data and for RM 47, which falls above. At RM 47, the modeled values are about 0.1 ppm below the lowest sampled value. These results suggest that the model is able to represent the general level of sediment contamination in the river as a function of distance downstream.

Response to EL-1.9

Concentrations of PCBs in striped bass are not explicitly modeled for Food Web Region 1 because the striped bass has not been parameterized within the FISHRAND model (that is, it was not one of the six fish species targeted for modeling). Consequently, the approach to estimating concentrations of PCBs in striped bass was to develop a ratio based on observed concentrations of PCBs in largemouth bass (wet weight basis), which is a target species for FISHRAND, with observed concentrations of PCBs in striped bass at the same location and within the same time period.

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

Response to EL-1.10

The concentrations of PCBs in white perch estimated by FISHRAND at each RM are presented in Figure 3-10 of the ERA Addendum. While the average of the FISHRAND data from each station would provide a more concise comparison between the FISHRAND and Farley models, the values shown in the figure were presented because they were the values used to calculate risk in the ERA Addendum.

3.1.1.6 Comparison Between FISHRAND Output and Sample Data

Response to EF-1.15

The NYSDEC 1998 data were not available in the database at the time the ERA Addendum was issued.

Response to EL-1.11

Because PCB concentrations in striped bass were not directly modeled, it is not possible to compare modeled results to data, as was done for the other fish species.

3.1.2. Model Results

No significant comments were received on Section 3.1.2.

3.1.2.1 Farley Model Forecast Water Column and Sediment Concentrations

No significant comments were received on Section 3.1.2.1.

3.1.2.2 Farley Model Forecast Fish Body Burdens

Response to EL-1.12

Figures 3-16 and 3-17 show the fish body burden bimonthly concentrations of Tri+ PCBs modeled by the Farley bioaccumulation model (represented by "x") versus monthly load of Tri+ PCBs from the Upper Hudson River as modeled by HUDTOX. The range of values show the seasonal variation. Figure EL-1.12 shows the revised Figure 3-17 with the corrected title (i.e., Striped Bass Body Burdens in Food Web Region 2).

3.1.2.3 FISHRAND Forecast Fish Body Burdens

Response to EL-1.13

The FISHRAND model compares favorably to data and provides concentrations of PCBs in at finer spatial scales than does the Farley model. Consequently, the risk estimates presented in the ERA Addendum rely primarily on the results from the FISHRAND model. Because striped bass were not directly modeled, it is not possible to compare modeled results to data, as was done for other fish species.

3.1.3 Modeling Summary

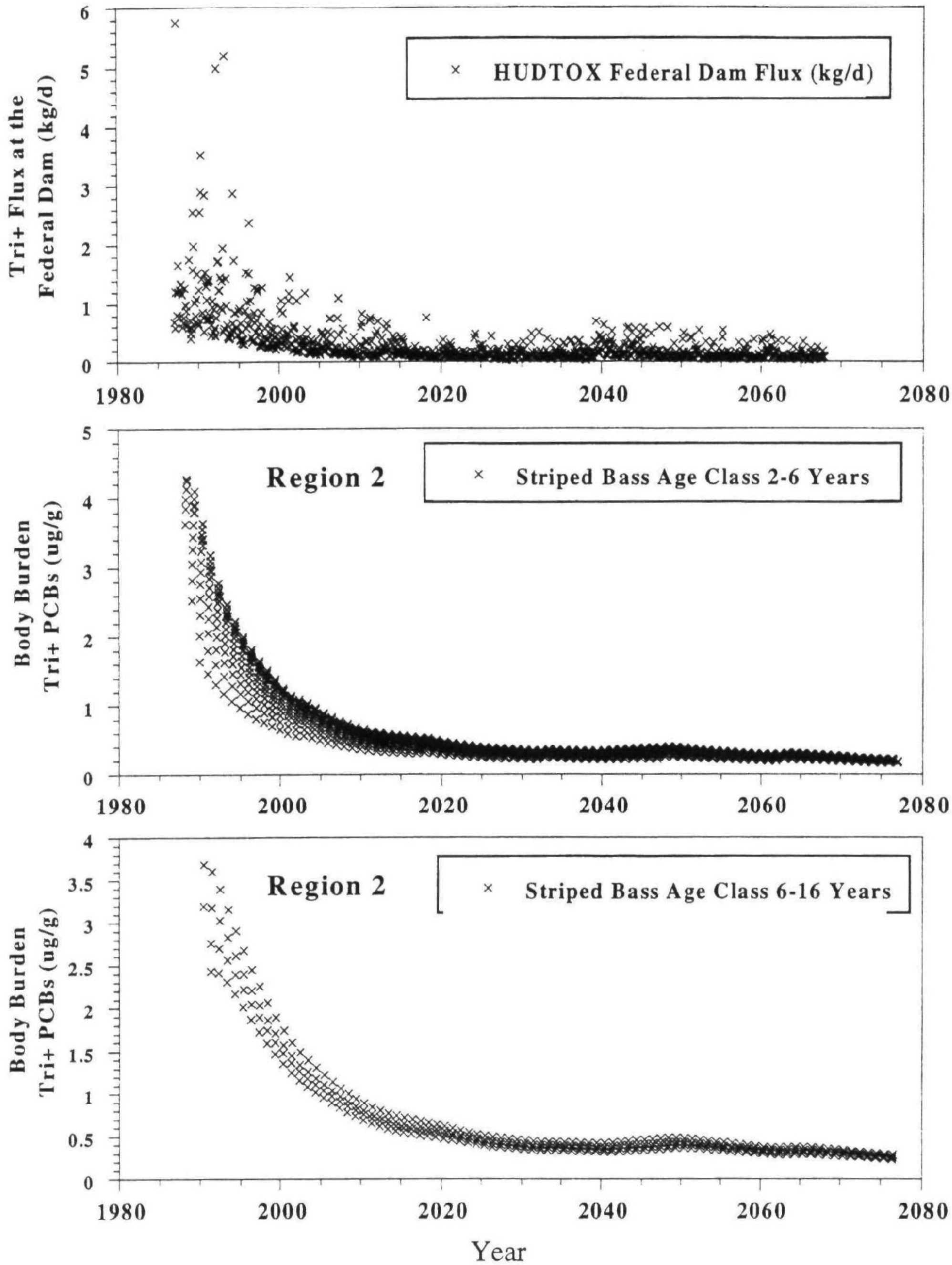
No significant comments were received on Section 3.1.3.

3.2 Exposure Point Concentrations

Response to EL-1.14

The selection of the specific river miles for the purposes of the FISHRAND simulations in the Lower Hudson was based partially on the available data for the Lower Hudson. The RMs 152 and 113 correspond to the locations used by NYSDEC in its long-term (20 years or so) fish monitoring. Therefore, these locations represent the best locations for the calibration of the FISHRAND model. Because the model has been calibrated at these locations, they are also the best locations for forecasting fish body burdens, and thus were used in the ERA Addendum. The other locations, RMs 90 and 50, were added to achieve approximately equal spacing among the simulated locations. For each of these locations, FISHRAND is used with the sediment and water results from the Farley model to estimate fish body burdens in the resident fish species (i.e., all species except striped bass). In the ERA Addendum, striped bass are only estimated for RMs 152 and 113 because these are the only locations with sufficient striped bass and largemouth bass data to establish a ratio between the two species. Thus striped bass are calculated from the FISHRAND output for largemouth bass by the application of the ratios described in Section 3.2. (For the Mid-Hudson Human Health Risk Assessment, striped bass body burdens also were estimated at RM 90 by applying the ratio from RM 113 to the forecast concentrations of PCBs in largemouth bass at RM 90.)

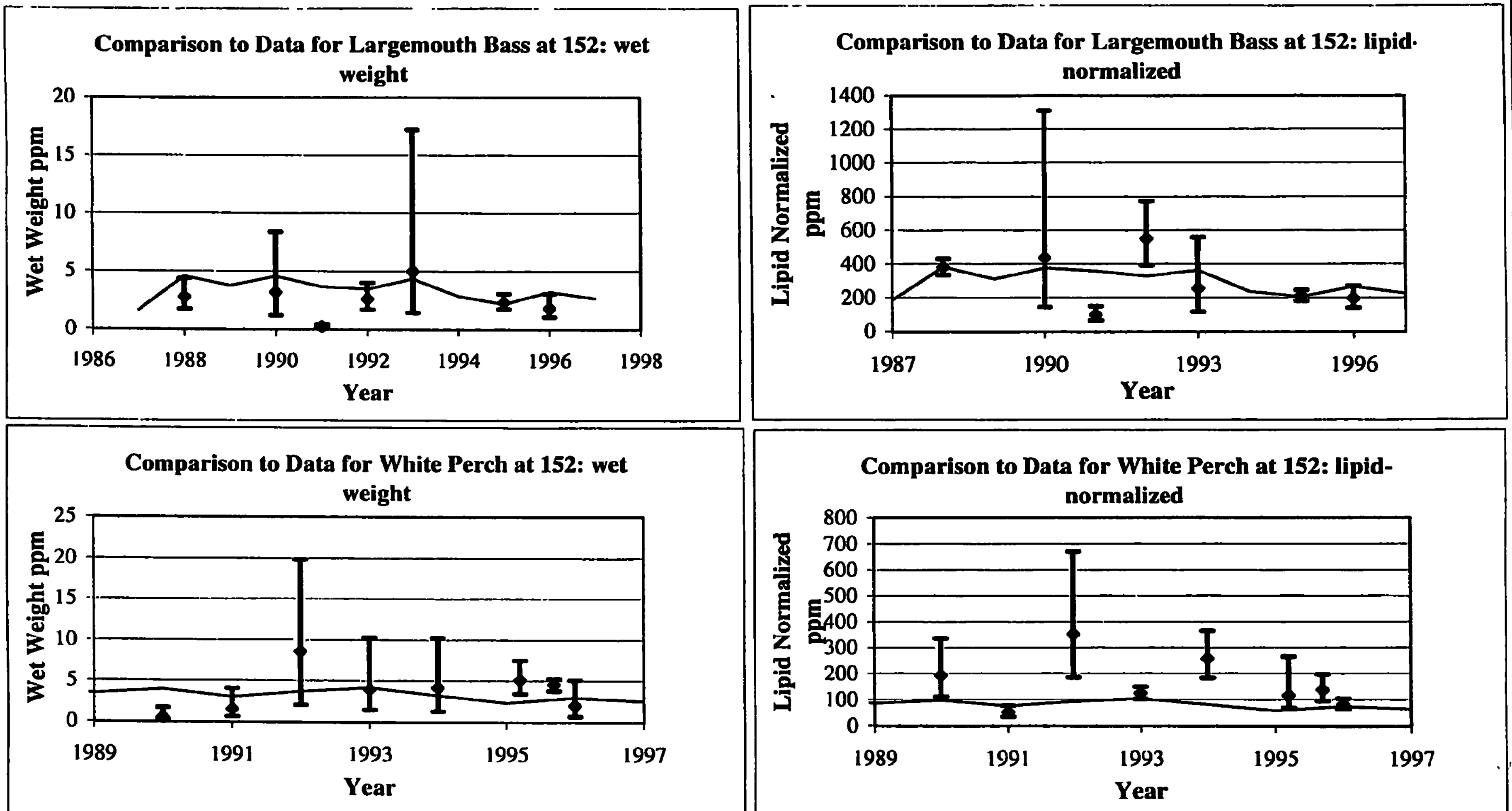
Overall, the range of FISHRAND stations spans from RM 152 to 50, which is comparable to the sediment and water ranges simulated by the model, RM 153.5 to 33.5. The differences in historical fish body burdens between adjacent monitoring stations are generally less than the uncertainty associated with the median value (compare single species for individual years among Figures EL-14a, b and c [revised Figures 3-12a to 3-12c]; see also Figure 12 in the report



Souces: Farley et al., 1999, Hudson River Database Release 4.1 and USEPA, 2000

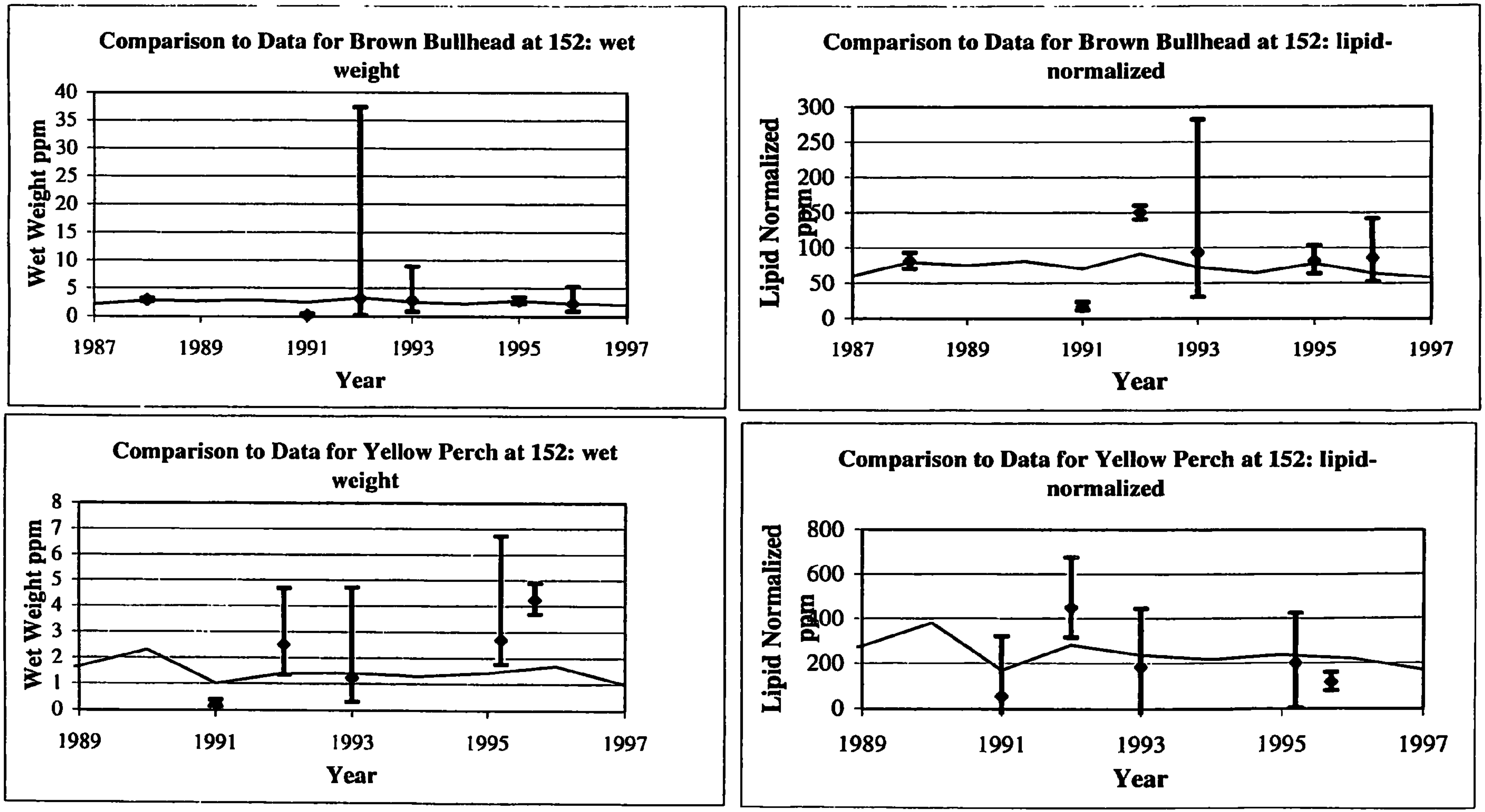
Figure EL-1.12 (Revised Figure 3-17)
Comparison Among the HUDTOX Upper River Load and Farley Model Estimates
Striped Bass Body Burdens in Food Web Region 2
(1987-2067)

FIGURE EL-1.14a (Revised Figure 12a): Comparison Between FISHRAND Results and Measurements at RM 152



MCA/TAMS

FIGURE EL-1.14a (Revised Figure 12a): Comparison Between FISHRAND Results and Measurements at RM 152



MCA/TAMS

FIGURE EL-1.14b (Revised Figure 12b): Comparison Between FISHRAND Results and Measurements at RM 113

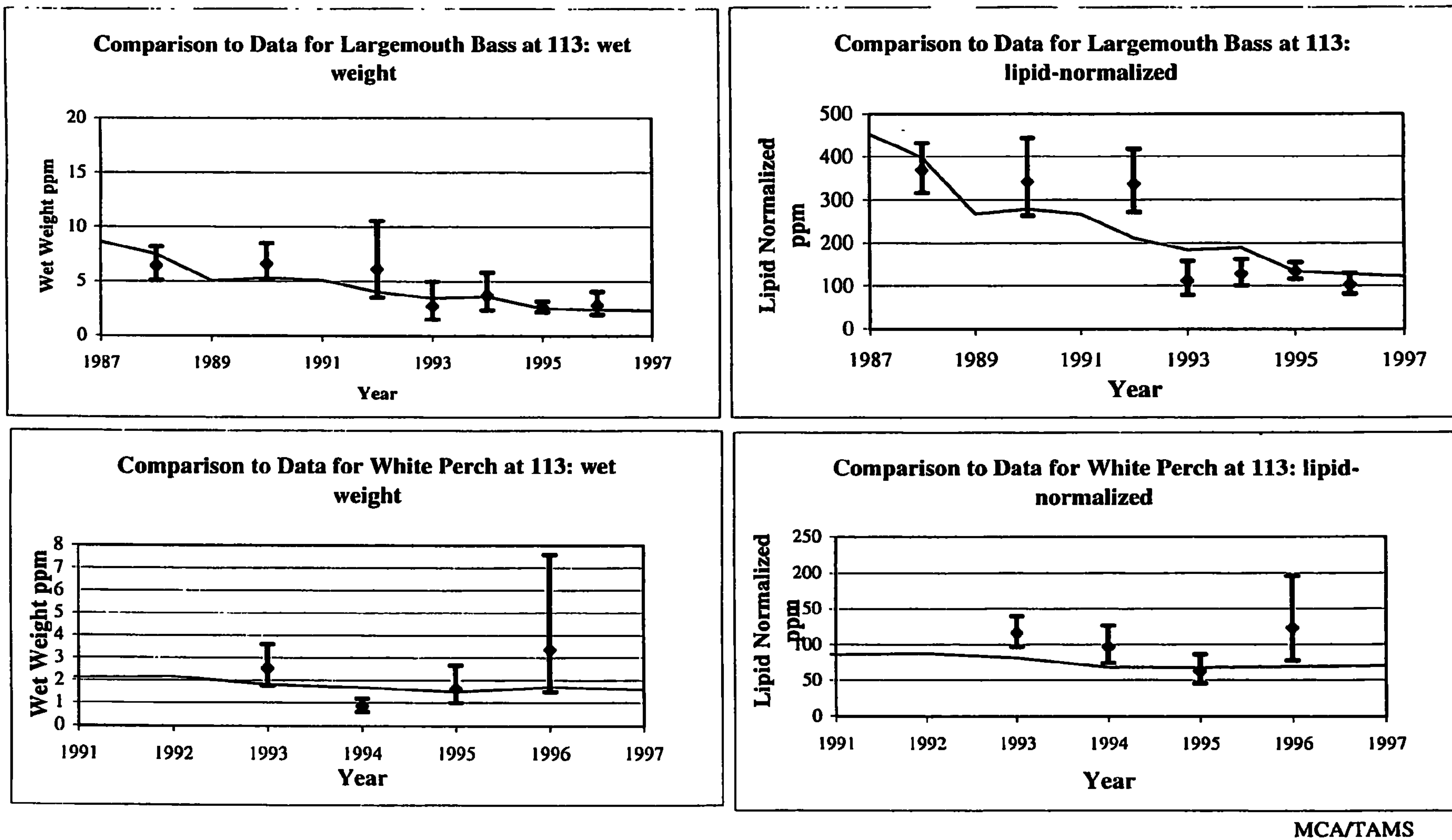
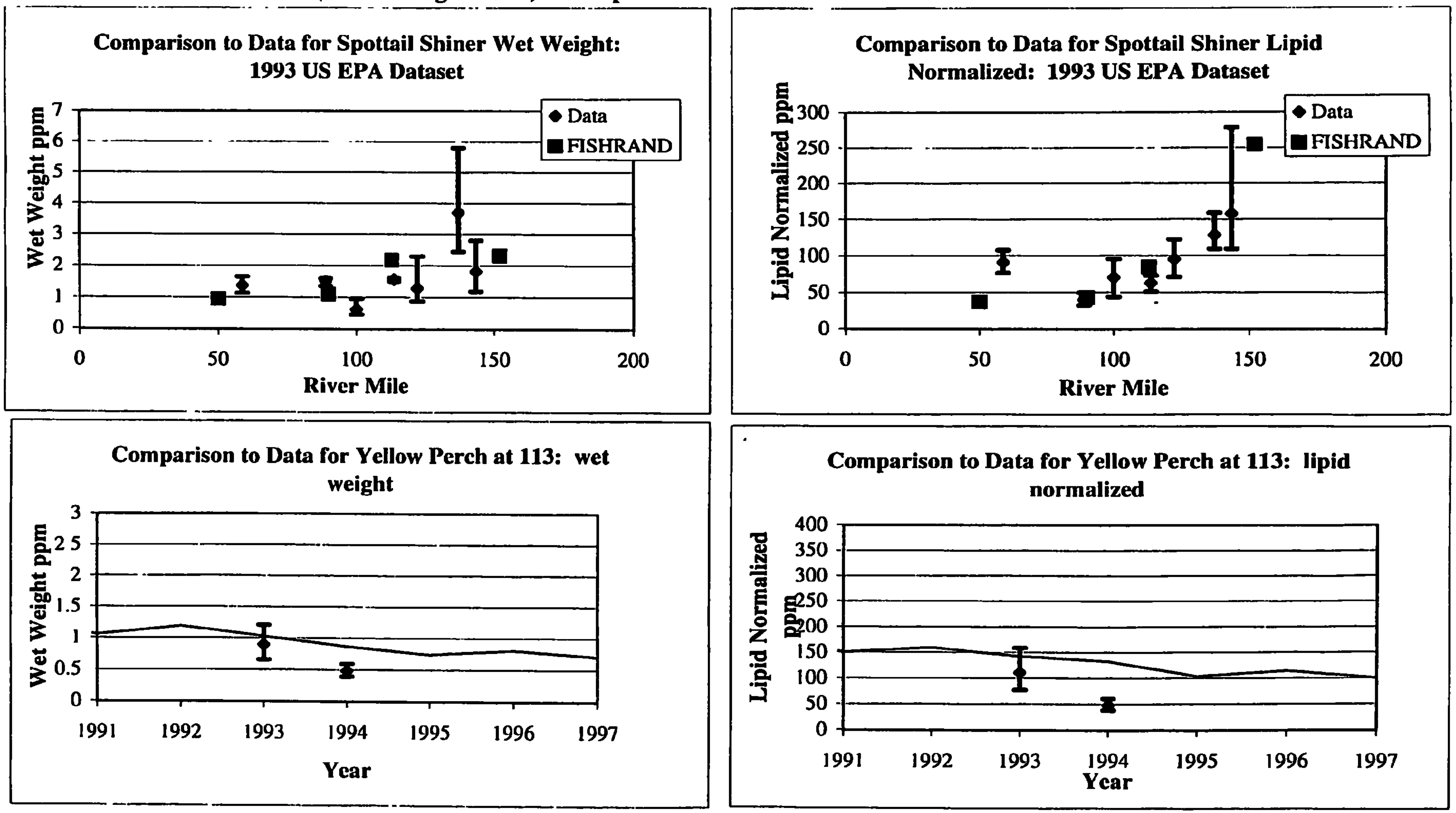
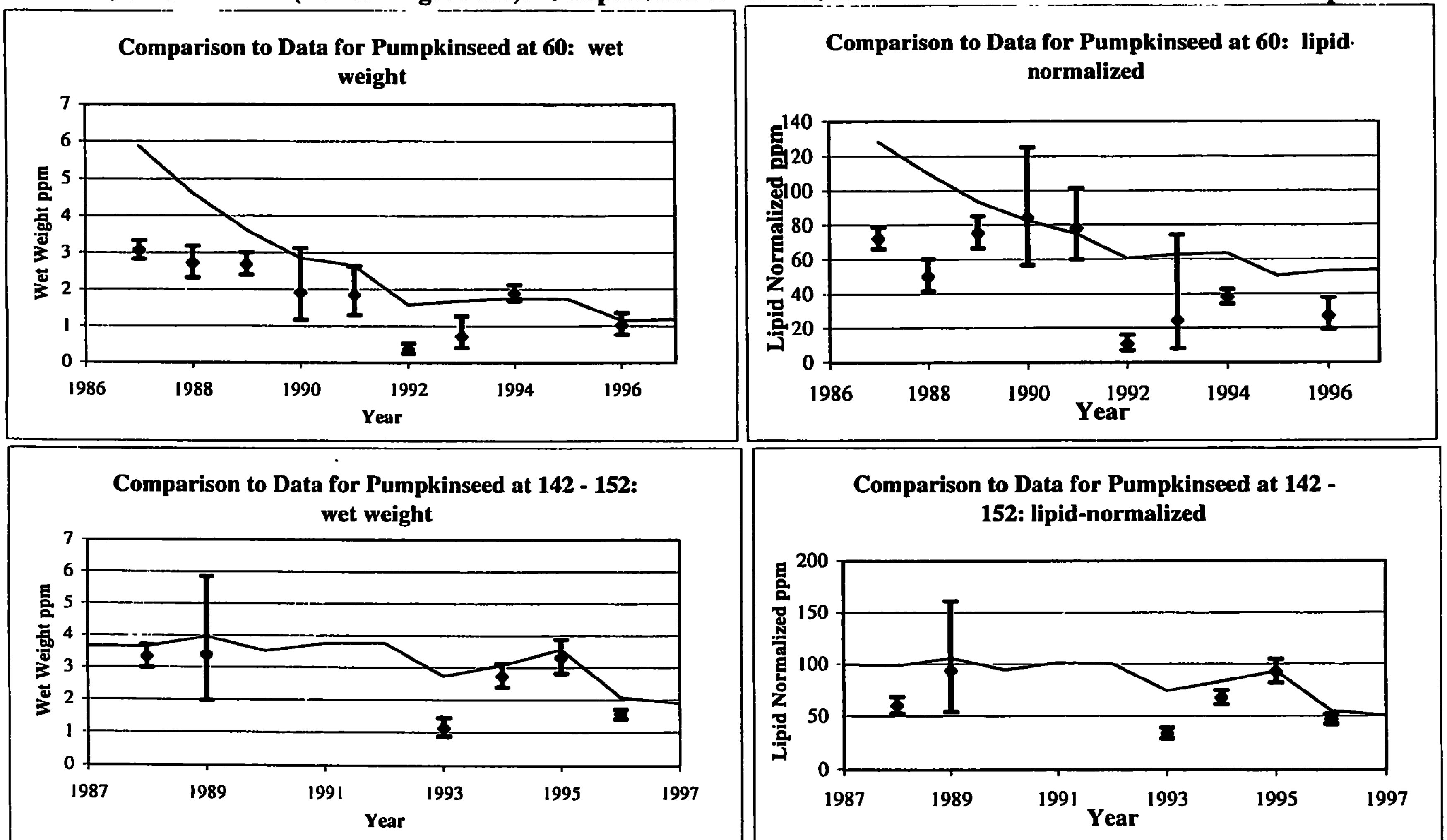


FIGURE EL-1.14b (Revised Figure 12b): Comparison Between FISHRAND Results and Measurements at RM 113



MCA/TAMS

FIGURE EL-1.14c (Revised Figure 12c): Comparison Between FISHRAND Results and Measurements of Pumpkinseed



MCA/TAMS

by Farley et al., 1999). Thus, the selection of these RM stations does not introduce excessive conservatism.

3.2.1 Modeled Water Concentrations

No significant comments were received on Section 3.2.1.

3.2.2 Modeled Sediment Concentrations

Response to EF-1.16

The Farley model predicts sediment concentrations on a dry weight basis. Organic carbon-normalized concentrations must then be estimated from the dry weight concentration and an assumed TOC. In fact, lipophilic compounds such as PCBs are strongly sorbed to particulate organic carbon, and, among sediments of the same depositional age, the concentration normalized to organic carbon is often a more consistent and stable measure than dry weight concentration. Indeed, much of the variability among dry weight concentration measurements is due to variability in TOC content. Therefore, to back-calculate an average organic carbon-normalized PCB sediment concentration, it is important to use the same estimate of TOC as was used in the Farley et al. (1999) report, which is 2.5%. Given that the average sediment PCB concentration at a given location and time is really an estimate of the expected (dry weight) concentration for an average TOC, it would not be appropriate to compute organic carbon-normalized concentrations using extreme values from the observed range of TOC concentrations in the Lower Hudson. In general, sediment areas with very low TOC concentrations (e.g., sands) will have correspondingly low dry weight PCB concentrations, although organic carbon-normalized concentrations may be similar to other locations.

Response to EF-1.17

The TCDD sediment guidelines developed by USEPA (1993) for the protection of fish, birds, and mammals are:

- Sediment concentrations associated with low risk: 60 pg/g dry weight (dw) for fish, 21 pg/g dw for birds, and 2.5 pg/g dw for mammals; and
- Sediment concentrations associated with high risk to sensitive species: 100 pg/g dw for fish, 210 pg/g dw for birds, and 25 pg/g dw for mammals.

However, because these TCDD sediment guidelines associated with risk to Great Lakes receptors (i.e., fish, birds, and mammals) were back-calculated using measured biota-to-sediment-accumulation factors (BSAF) 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 BSAFs are based on lipid content of fish and organic carbon content of sediment that are measured in the Great Lakes. In addition, these BSAFs are specific for the Great Lakes food chain and are likely to be different from those measured for a riverine food chain. Based on these differences, the risk-based sediment concentrations for the Great Lakes (pg TCDD/g dry weight of sediment) are not directly comparable to the Hudson River.

3.2.3 Modeled Benthic Invertebrate Concentrations

Response to EF-1.18

Benthic invertebrate data from 1993 are available for the following RMs: 122.4, 100, 88.9, 47.3, and 25.8. No comparisons are available for river mile 152. River miles 122.4 and 100 were averaged to obtain a comparable estimate for RM 113. River miles 88.9 and 100 were averaged to obtain a comparable estimate for RM 90, and RM 47.3 was used to compare for RM 50. Concentrations are shown as wet weight in mg/kg PCBs. These comparisons, which are provided below, show that the concentrations of PCBs in benthic invertebrates are reasonably estimated.

	RM 113		**RM 90**		**RM 50**	
	Data	Model	Data	Model	Data	Model
Mean	0.82	0.98	0.40	0.80	0.60	0.58
Standard Error	0.14	0.10	0.21	0.20	0.20	0.11

Response to EF-1.19

The FISHRAND model was not designed to capture bioaccumulation in highly migratory fish species such as the striped bass, nor was it parameterized for this species (e.g., growth rate). The Farley model explicitly models striped bass, but concentrations were only available for locations below RM 60. Because striped bass are known to occur throughout the Lower Hudson River, and direct modeled concentrations were not available the Lower Hudson River from the FISHRAND model or from the Farley model above RM 60, the ratio approach was used for RMs 152 and 113. The Farley model provides predictions of PCB concentrations in striped bass at greater spatial scales than the FISHRAND model results for all the other species.

3.2.4 Modeled Fish Concentrations

Response to EL-1.15

There are no PCB body burden data available for shortnose sturgeon (an endangered species). Thus, there are no data to calibrate a model. The approach taken in the ERA Addendum approximates concentrations of PCBs in shortnose sturgeon based on the similarity to the brown bullhead (feeding preferences, etc.).

Development of the FISHRAND model (and all of the bioaccumulation models) is constrained by data availability. All of the monitoring data from 1977 to 1997 for largemouth bass, yellow perch, white perch, brown bullhead are expressed on a fillet basis. To calibrate the model and demonstrate model functionality, predictions from the FISHRAND model are also expressed as a concentration of PCBs in fillet.

3.3 Identification of Exposure Pathways

Response to EL-1.16

The responses to comments on the on exposure pathways of the August 1999 ERA, which were not reiterated in the comment, have been provided in the Responsiveness Summary for the ERA (USEPA, 2000b, see responses in Section 3.4, Identification of Exposure Pathways).

3.3.1 Benthic Invertebrate Exposure Pathways

No significant comments were received on Section 3.3.1.

3.3.2 Fish Exposure Pathways

Response to EF-1.20

The NOAA (1999) 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 (USEPA, 1999b) developed a more narrow definition of ecologically relevant endpoints, which included reproductive and developmental effects but not immunotoxic effects. Immunotoxic effects were not included because such effects are often less clearly related to the assessment endpoints than are developmental and reproductive effects.

In addition, NOAA (1999) reported data that were measured or converted into concentrations in adult liver tissue. The relationship between the concentration in liver tissue and the concentration in whole fish has not been well studied for most species. Therefore, the ERA Addendum gives 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 better characterized than the relationship between liver concentration and adult tissue concentration. In addition, more data are generally available for effects associated with concentrations of PCBs in whole tissue.

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

No significant comments were received on Section 3.3.3.

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

No significant comments were received on Section 3.3.3.1.

3.3.4 Mammalian Exposure Pathways, Parameters, and Daily Doses

No significant comments were received on Section 3.3.4.

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

No significant comments were received on Section 3.3.4.1.

4.0 EFFECTS ASSESSMENT

Response to EL-1.17

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 General Electric Company 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, Appendix K of USEPA, 1999b). Therefore, it is believed that Clophen A50 is a reasonable surrogate for the composition of PCBs in Hudson River fish.

The revisions to the TRVs based on the Bengsston study followed a review of the comments received on the ERA (USEPA, 2000b). The Bengsston study presented results for three dose groups and a control group. Originally, the values of 170 mg/kg and 15 mg/kg were selected based on a hatchability endpoint. For this endpoint, the high dose group was significantly different from the control group, but not the low and medium dose groups. However, for the hatching time endpoint, the medium and high dose groups were significantly different from the control group. Hatching time is a less relevant endpoint than hatchability, however, Bengsston (1980) noted that premature hatching "resulted in premature death of the fry" and that "very few survived for more than one week after hatching." Because there were no formal statistics conducted in association with this statement, the TRVs selected for the ERA were based on hatchability. However, USEPA received numerous comments that, given the observed premature death of the fry, hatching time was a relevant endpoint. As a result, USEPA revised the TRVs to reflect the concentrations derived from the hatching time endpoint in the Bengsston (1980) study.

Based on additional comments received on the ERA Addendum, USEPA has selected the Hansen et al. (1974) study rather than the Bengsston (1980) study, based on the rationale explained in the response to comment EF-1.6 and Section III of this Responsiveness Summary.

Response to EG-1.7

The NOAA Sediment Effects Concentrations (SECs) were used as sediment guidelines, not TRVs, in the ERA Addendum (see, USEPA, 1999c, p. 38). Their use as guidelines is consistent with accepted scientific practice. A detailed response on the development and use of the SECs by the author of the document is contained in response to EG-1.40 of the Responsiveness Summary for the ERA (see, USEPA, 2000b, pp. 75-80).

Response to EG-1.8

For the TEQ analysis, BZ#126 was used at the detection limit to compensate for not having quantitated BZ#81, as described in the ERA (see, USEPA, 1999b, 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 dataset was approximately equal to the sum of the BZ#126 and BZ#81 in the USFWS dataset (see, USEPA, 1999b, 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 too high by at most a factor of two. This is a relatively small margin of error, given that the calculated risk levels exceed USEPA's levels of concern by orders of magnitude.

The NOAA report (1999) applies the TEQ approach for estimating risk to Hudson River fish, but identifies two areas of uncertainty. First, the NOAA (1999) report notes that the available data indicate large interspecies differences in early life stage toxicity. Second, the report notes the risk estimates are based on data for measured concentrations of only two dioxin-like compounds of PCBs in Hudson River fish, and that these results may underestimate the concentrations and effects of total dioxin equivalents in these fish. The ERA Addendum addresses the first source of uncertainty by presenting interspecies differences in sensitivity of early life stages to dioxin-like compounds in Table B-7 and acknowledging that salmonid species are the most sensitive group thus far tested. The ERA Addendum accounts for this uncertainty by using two sets of TRVs, one based on data for salmonids and one based on data for non-salmonids, to bracket the potential range of sensitivity of Hudson River fish. The ERA Addendum addresses the second source of uncertainty by acknowledging that concentrations of some dioxin-like compounds may be underestimated in measurements of Hudson River fish and using a modeling approach to approximate the concentration of total dioxin equivalents in these fish.

Response to EG-1.9

The review conducted by Dr. Emily Monosson for NOAA is cited in the ERA (and thereby for the ERA Addendum) as NOAA (1999b). The NOAA report was reviewed during the process of selection of TRVs for the ERA and ERA Addendum. However, results presented in the NOAA report and in USEPA's ERA and ERA Addendum are based on different approaches and assertions about the type of studies that are most appropriate for assessment of risk to Hudson River fish. The study by NOAA reported measured or estimated concentrations in eggs, larvae, or in liver of adult fish that result in adverse reproductive, developmental, or immunotoxic effects. In contrast, USEPA developed TRVs for total PCBs on the basis of studies that report measured concentrations of total PCBs in tissue of larval and adult fish which result in adverse effects on survival growth or reproduction, but not on sublethal immunotoxic or biochemical effects. USEPA believes that effect levels reported as measured whole-body concentrations are most appropriate for comparison to concentrations of PCBs in Hudson River fish.

As noted by the commenter, the NOAA (1999) report finds that concentrations of PCBs as low as 5 ppm (whole body, wet wt.) in larvae can impair survival. This finding is based on studies by Hansen et al. (1974) and Shimmel et al. (1974). NOAA (1999) reports a measured concentration from the Hansen et al. study and an estimated concentration in larvae from the Shimmel et al. (1974) study. Because the factor used by NOAA to estimate the concentration in larvae is highly uncertain (e.g. NOAA reports ratios ranging from 0.1 to 545 in other fish), the study by Shimmel et al. (1974) was not used by USEPA to develop TRVs.

The study by Hansen et al. (1974) was selected by USEPA for development of a TRV. USEPA notes, however, that the concentration measured by Hansen et al. (1974) and reported by NOAA as 5.1 mg/kg in larvae, was actually measured in eggs. USEPA did not use this concentration for development of a TRV because no other studies were identified that examined concentrations of total PCBs in eggs. Rather, USEPA developed a TRV from the effective concentration in adult tissue that was reported in the same paper (Hansen et al. 1974). Effect concentrations determined as concentrations in tissue of adult fish are believed to be more directly comparable to PCB concentrations in adult Hudson River fish than are effect concentrations determined in eggs. Therefore, USEPA used the NOAEL (1.9 mg/kg) and LOAEL (9.3 mg/kg) effect levels reported in Hansen et al. (1974) for development of TRVs.

The comment states that adverse effects on adult fish might occur at concentrations exceeding 12.5 ppm (whole body, wet weight). Actually, NOAA (1999) reports that effects may occur at greater than 25 ppm in adult *liver*, a concentration that is expected to be equivalent to a concentration of 12.5 ppm in *fillet*, not whole body as stated by the commenter. The NOAA report compiled concentrations that were measured or estimated in liver of adult fish. Studies that reported concentrations in liver were not used by USEPA for development of TRVs because uncertainty in the ratio of concentration in liver to concentration in whole body or fillet was believed to be too great. As noted in the NOAA report (1999), the liver/muscle ratio varies with a number of factors, and can range from <1 to 77. USEPA selected studies that reported actual measured concentrations of PCBs in whole body fish tissue.

<u>Response to EG-1.10</u>

The issues of the TQ approach and comparison to the Clinch River Assessment are addressed in the responses to EG-1.1, EG-1.5, and EG-1.6 of this Responsiveness Summary.

Response to EG-1.18a

The study by Bengsston (1980) was replaced by a study by Hansen et al. (1974) that was not identified for the ERA. As described in the response to comment EF-1.6, the Hansen et al. (1974) study was selected because it examined reproductive effects of Aroclor 1254, rather than Clophen, a mixture of PCBs that is similar in chlorine content but which was not used in the United States.

USEPA conducted an extensive review of the available literature on the effects of PCBs on wildlife species. All of the studies presented in Tables B-4 through B-22 in the ERA Addendum (USEPA, 1999c) were considered in the development of the TRVs. Only after consideration of all of the studies were the most appropriate individual studies selected for development of the TRVs.

In deriving the TRVs, in cases for which there is no appropriate information available on the sensitivity of a receptor of concern, it is conservatively protective to assume that the receptor could be as sensitive as the most sensitive species tested. However, based on comments received from the peer reviewers, the sensitivity of wild birds is expected to be less than that of gallinaceous birds, such as the chicken, which are often used as test species. USEPA is evaluating how best to revise its selection of TRVs on the basis of the peer reviewers' comments.

Response to EG-1.18b

In regard to benthic invertebrate community endpoints, the ERA Addendum (USEPA, 1999c) examined future risk to the benthic invertebrate community of the Lower Hudson River and therefore used sediment guidelines and water quality criteria as measurement endpoints. The ERA (USEPA, 1999b) used 1993 data on macroinvertebrate communities as a measurement endpoint to evaluate current risk to the Lower Hudson River benthic invertebrate community.

Response to EG-1.19

The studies by Adams et al. (1989, 1990, 1992) are mistakenly listed in Table B-6 as EL-effect and EL-no effect, meaning that they examined a single effect level rather than a range of doses. In fact, a range of doses was examined and the NOAEL is not unbounded (Adams et al., 1992). Adverse effects that could be attributed to PCBs (or other co-occurring contaminants) were observed.

USEPA acknowledges the uncertainty associated with using the unbounded field study by Westin et al. (1983). However, the study is used to develop TRVs because it was conducted on striped bass from the Hudson River.

As stated in response to comment EG-1.9, the review conducted by NOAA is cited in the ERA as NOAA (1999b). The NOAA report was reviewed during the process of selection of TRVs for the ERA and ERA Addendum. However, results presented NOAA report and in the ERA and ERA Addendum are based on different approaches and assertions about the type of studies that are most appropriate for assessment of risk to Hudson River fish. For example, the study by NOAA reported measured or estimated concentrations in eggs, larvae, or in liver of adult fish that result in adverse reproductive, developmental, or immunotoxic effects. In contrast, USEPA developed TRVs for total PCBs on the basis of studies that report measured concentrations of total PCBs in tissue of larval and adult fish that result in adverse effects on survival, growth, or reproduction but not on sublethal immunotoxic or biochemical effects. In addition, the NOAA report compiled concentrations that were measured or estimated in liver of adult fish. Studies that reported concentrations in liver were not used by USEPA for development of TRVs because uncertainty in the ratio of concentration in liver to concentration in whole body or fillet was believed to be too great. As noted in the NOAA report (1999), the liver/muscle ratio varies with a number of factors, and can range from <1 to 77. USEPA selected studies that reported actual measured concentrations of PCBs in whole body fish tissue.

The NOAA report (1999) applies the TEQ approach for estimating risk to Hudson River fish, but identifies two areas of uncertainty. First, the NOAA (1999) report notes that the available data indicate large interspecies differences in early life stage toxicity. Second, the reports notes the risk estimates are based on data for measured concentrations of only two dioxin-like compounds of PCBs in Hudson River fish, and that these results may underestimate the concentrations and effects of total dioxin equivalents in these fish. The ERA Addendum addresses the first source of uncertainty by presenting interspecies differences in sensitivity of early life stages to dioxin-like compounds in Table B-7 and acknowledging that salmonid species are the most sensitive group thus far tested. The ERA Addendum accounts for this uncertainty by using two sets of TRVs, one based on data for salmonids and one based on data for non-salmonids, to bracket the potential range of sensitivity of Hudson River fish. The ERA Addendum addresses the second source of uncertainty by acknowledging that concentrations of some dioxin-like compounds may be underestimated in measurements of Hudson River fish and using a modeling approach to approximate the concentration of total dioxin equivalents in these fish.

Niimi (1996) noted in his review, "These estimates represent the threshold concentrations that were derived from a limited information base that may not be representative of the more sensitive species and should be interpreted accordingly." In fact, the Niimi (1996) review does not include or consider the studies that were found to be most relevant to the assessment of risk to Hudson River fish. For example, it does not consider the results of the studies by Bengtsson (1980) or Hansen et al. (1974) (<u>see</u>, response to comment EF-1.6). Therefore, the review by Niimi (1996) cannot be considered to be a comprehensive overview of the relevant studies that can be used to assess risk or develop TRVs.

<u>Response to EG-1.20</u>

As the commenter notes, all things being equal among several studies, selection of the *highest* NOAEL for the development of TRVs would be appropriate. However, many different types of studies varying in aspects such as exposure time, exposure route, toxicological endpoint examined and species examined were reviewed for the ERA and ERA Addendum. Therefore, studies were evaluated based on many criteria in order to select the most appropriate study. The selected study was often lower than other available endpoints, but was not selected solely on that basis. For example, a chronic study on a sensitive reproductive endpoint may report a lower NOAEL/LOAEL than a subchronic study that examined adult mortality. In this example, it is appropriate to select the lower value because it is a more appropriate study.

The ERA Addendum did not use uncertainty factors in an overly conservative fashion. For example, if a TRV was developed on the basis of a study that was conducted on a very sensitive species, an additional interspecies uncertainty factor was not applied because it is unlikely that the receptor of concern would be more sensitive than the highly sensitive species. If, however, a TRV was developed on the basis of a study that was conducted on a species that was known to be of intermediate sensitivity, an uncertainty factor of 10 was applied in case the receptor of concern is more sensitive than the test species.

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). This report summarizes several studies that analyzed the variability in acute sensitivity of birds and mammals to a variety of chemicals. For the effect of an individual chemical on birds, the ratio of the LC_{50} for the least sensitive species to that of the most sensitive species was usually less than 10. For mammals, the ratio for the least sensitive species to the most sensitive species was usually less than 100. These analyses of variability in the acute sensitivity provided support for the use of a recommended range in interspecies uncertainty factors of 1 to 100. In addition, a smaller set of data on chronic exposures indicated that an interspecies sensitivity ratio of 100 encompasses 84% of the chronic data. Therefore, in cases for which no appropriate information is available on the chronic sensitivity of a receptor of concern, it is conservatively protective to assume that the receptor could be 10 times more sensitive than the species used to establish the TRV. However, as previously noted, when the test species that was used to establish the TRV is known to be a highly sensitive species, the ERA Addendum did not apply an interspecies uncertainty factor in order to estimate the TRV for the receptor of concern.

Similarly, USEPA (1995) provides support for the conceptual basis for use of a subchronic-to-chronic uncertainty factor. The report summarizes the results of three studies that examined the ratio of subchronic-to-chronic toxicity endpoints. The first study reported that 97% of the ratios were 9 or below. The second study reported that 98% of the ratios were less than 4 and all of the ratios were less than 7. The third study reported that 90% of the ratios were within a factor of 5. Therefore, use of an uncertainty factor of 10 in the ERA and ERA Addendum to estimate chronic toxicity endpoints from sub-chronic studies is a conservatively protective approach.

Tree Swallow

Although tree swallows inhabiting areas along the Lower Hudson River are likely to be less exposed to PCBs than those along the Upper Hudson River, the USFWS findings (summarized in Secord and McCarty, 1997 and McCarty and Secord, 1999a,1999b) are relevant for the Lower

Hudson River because they can be used to estimate a NOAEL for field exposure of tree swallows to PCBs.

As stated in the ERA Addendum, USEPA agrees that the data by McCarty and Secord do not demonstrate a consistent relationship between exposure to PCBs and adverse reproductive effects in the tree swallow. Although significant adverse effects on reproduction were observed in the first year of the study, significant adverse effects on reproduction were not observed in the second year of the study. Reproductive success in the first year may have been influenced by the large number of young females that typically inhabit nest boxes in the first year that the boxes are placed in the field. These data were therefore used to establish a NOAEL, but not LOAEL TRV (see, USEPA, 1999c, Appendix B, p. 26).

Mallard

Five studies were identified that examined effects of PCBs on the mallard. The study by Hill et al. (1975) was not selected for development of TRVs for exposure of mallards to PCBs 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) was selected as the most appropriate study because it 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 10 was applied to estimate a NOAEL from this study. The study was conducted over a 12 week period, so a sub-chronic to chronic uncertainty factor was not applied.

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

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

Great Blue Heron

The study by Speich et al. (1992) was designed to examine eggshell thinning rather than the more sensitive endpoint of egg mortality. The author reports, "In this study, we found no evidence of current reproductive failure, high incidence of eggshell breakage, eggshell flaking, or low hatching success (S.M. Speich, unpubl. field notes)." However, the study provides no data on egg mortality, other than this reference to unpublished field notes. Unpublished field results that have not tested statistically are not considered an appropriate basis for the development of TRVs.

The description of the development of the TRVs for TEQs in eggs of the great blue heron is revised to better explain how the TRV was developed. The description is revised to note that the TRV was developed on the basis of both the study by Sanderson et al. (1994) and the study by Hart et al. (1991). As noted, Sanderson et al. (1994) do not present data on reduced growth rate. Sanderson et al. (1994) presents data on the concentration of TEQs in eggs that were collected from a highly contaminated site (Crofton) and a less contaminated site (Vancouver) in 1988. Hart et al. (1991) report that the yolk-free body weights of chicks collected in 1998 from Crofton were significantly different from a reference site, but that the weights of chicks from Vancouver were not different from the reference site. Therefore, the data from both Hart et al. (1991) and Sanderson et al. (1994) were used in development of the TRVs for the great blue heron:

The LOAEL TRV for the great blue heron is 0.5 ug TEQs/kg egg.
The NOAEL TRV for the great blue heron is 0.3 ug TEQs/kg egg.

Belted Kingfisher

Taxonomic similarity is considered to be a better predictor of sensitivity to PCBs and dioxin-like compounds than is similarity in feeding habits. Because no information is available on the sensitivity of the belted kingfisher or for a species in the same family as the belted kingfisher, the assessment conservatively assumes that the kingfisher could be as sensitive as the most sensitive species tested.

Bald Eagle

USEPA agrees that the data by Wiemeyer et al. (1993) do not support the development of a NOAEL TRV of 3.0 mg/kg for the bald eagle. However, USEPA does not agree with the commenter's assertion that because mean five-year production was not significantly reduced for the residue *interval* ranging from 5.6–<13 mg PCBs/kg, 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-<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 by Wiemeyer et al. (1993). As an alternative, USEPA is using the average PCB concentration in eggs from successful nests (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), as the NOAEL TRV for bald eagles.

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

The NOAEL TRV for the bald eagle is 5.5 mg/kg egg.

As noted in the ERA Addendum, USEPA agrees that because of the presence of co-occurring contaminants, adverse effects observed in field studies cannot be attributed solely to the presence of PCBs. Therefore, USEPA did not use any field studies to establish LOAELs, the concentrations or doses at which adverse effects are expected to occur. USEPA did, however, use field studies to establish NOAELs, the concentration or doses below which adverse effects are not expected to occur. USEPA acknowledges that because of the confounding influence of co-occurring contaminants, that actual NOAEL TRVs could be higher than those observed in field-based studies.

The study by Elliott et al. (1996) was included in USEPA's review (see, ERA Addendum, Table B-14). USEPA notes, however, that the study by Wiemeyer et al. (1993) was determined to be a more appropriate study for the development of a TRV for the bald eagle because the Wiemeyer et al. (1993) study examined numerous eggs in 15 states over a period of many years, whereas the Elliott et al. (1996) study examined only 16 eggs from a contaminated area and eight eggs from reference areas.

USEPA inadvertently excluded the study by Elliott et al. (1996) from Table B-16, the compilation of field studies on the effects of dioxin-like contaminants on bird eggs. The study by Elliott et al. (1996) reports data for TEQ in the yolk sac of the bald eagle egg. The authors do report 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 the other contaminated site (East Vancouver Island). If the concentration of TEQs at the East Vancouver Island site is estimated as 13,000 ng TEQs/kg lipid, the estimated wet weight concentration would be approximately 217 ng/kg ww for East Vancouver eggs. Because no significant difference was observed between the average hatching rate of the eggs collected from the pulp mill sites (East Vancouver and Powell River) and the non-pulp mill sites, a NOAEL based on the average egg concentration of these two sites could be developed. Based on the results of Elliott et al. (1996), an average field based NOAEL of 214 ng TEQs/kg ww would be established for the bald eagle. This is lower than the value of 400 ng/kg ww that was suggested by the commenter, the derivation of which is unclear.

The field based NOAEL for the bald eagle eggs would be 0.214 ug/kg egg.

The study by Donaldson et al. (1999) was not available when the literature search was conducted for the ERA Addendum. If the study were available, USEPA would still have selected the study by Wiemeyer et al. (1993) for development of TRVs because this study examined many more eggs. Donaldson et al. (1999) examined concentrations of PCBs in 6 eggs from a contaminated site, whereas Wiemeyer et al. (1993) examined numerous eggs in 15 states over a period of many years.

Response to EG-1.21

Mammals

USEPA acknowledges that limited data are available to assess the potential for risk to the little brown bat or the raccoon and that laboratory studies conducted on rats must be used to make conservative estimates risk to these organisms. However, as noted in Table B-27 of the ERA Addendum, field studies on the mink, rather than laboratory studies, are used to develop final TRVs for the mink and the otter.

Mink

USEPA agrees that a LOAEL should not be established from the Tillitt et al. (1996) field study. The revised risk estimates remove this comparison.

USEPA does not concur with the assertion of Sample et al. (1996) that an exposure over a longer period would not result in a lower effective dose. As described in Section B.2.1, USEPA's approach follows the approach used by the Great Lakes Water Quality Initiative (USEPA, 1995). This approach uses uncertainty factors to account for the well-recognized observation that subchronic toxicity studies may be of insufficient length to measure adverse effects that would be observed in chronic tests of longer duration.

USEPA examined the available data on body burdens of PCBs and dioxin-like compounds in mink of the Hudson River area and found that insufficient data were available to assess risk on this basis. Therefore, the study by Leonards et al. (1995) was not used to develop TRVs for the assessment.

River Otter

The paper by Harding et al. (1999) was not available when the literature search was done for the ERA and ERA Addendum. However, the study would not be selected to develop a TRV because it reports exposure on the basis of concentration in liver and reports effects on baculum length in juvenile males.

Response to EG-1.22

The TQ approach isolates the effects of PCBs versus other confounding influences from field-based studies. The TQ approach suggests the potential for risk attributable to PCBs alone. As explained in the response to EG-20, the ERA Addendum did not rely on overly conservative application of uncertainty factors in deriving TRVs and evaluated available literature in deriving TRVs.

4.1 Selection of Measures of Effects

No significant comments were received on Section 4.1.

4.1.1 Methodology Used to Derive TRVs

No significant comments were received on Section 4.1.1.

4.1.2 Selection of TRVs

Response to EF-1.21

The study by USACE (1988), which examined field-collected sediments, was inadvertently excluded from Appendix B. Results from the USACE (1988) study, the lab studies by Hansen et al. (1974) and Bengtsson (1980), and field studies by Adams et al. (1989, 1990, 1992) yield similar toxicity values (see, USEPA, 1999c, Table B-5), thereby providing further weight of evidence to support the selection of the TRVs.

The field-based NOAEL (0.5 mg/kg) reported in the ERA Addendum for pumpkinseed and largemouth bass was based on a reproductive endpoint (Adams et al. 1989, 1990, 1992). USEPA is revising the ERA Addendum to use the NOAEL of 0.3 mg/kg based on a growth endpoint from Adams et al. (1989, 1990, 1992). It should be noted that Adams et al. (1989, 1990, 1992) examined other endpoints that occurred at concentrations below 0.5 mg/kg. Adverse effects were associated with DNA integrity, detoxification enzymes, lipid metabolism, community structure and histological indices.

Response to EF-1.22

For the TEQ analysis, BZ#126 was used at the detection limit to compensate for not having quantitated BZ#81, as described in the ERA (see, USEPA, 1999b, 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 dataset was approximately equal to the sum of the BZ#126 and BZ#81 in the USFWS dataset (see, USEPA, 1999b, 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 too high by at most a factor of two. This is a relatively small margin of error considering that the calculated risk levels exceed USEPA's levels of concern by orders of magnitude.

Direct water column exposures represent a tiny fraction of the overall daily dose for all receptors, thus, the issue of detection limits is far less important for this medium.

5.0 RISK CHARACTERIZATION

Response to EL-1.18

The responses to comments on the risk characterization presented in the ERA can be found in the Responsiveness Summary for the ERA (see, USEPA, 2000b, Section 5.0 Risk Characterization).

Response to EG-1.17

The condition of the ecological resources of the Lower Hudson River in the relation to PCBs cannot be evaluated simply by examining trends over the last 30 years (see, response to EG-1.1). Although macroinvertebrate communities in the Hudson River have improved since the 1970s, much of the change can be attributed to improvements in treatment of municipal and industrial wastes rather than to a direct response to lower PCB concentrations (see, ERA Responsiveness Summary [USEPA, 2000b] responses to EG-1.7 and EG-1.34).

Fish population trends have also been addressed in the Responsiveness Summary for the ERA (see, responses to comments EG-1.9 and EG1-34 in USEPA, 2000b). The kinds of effects due to PCBs expected in the field include reduced fecundity, decreased hatching success, and similar kinds of reproductive impairment indicators, which are often difficult to discern, particularly against the background of the fishing ban.

Tree swallows are present throughout the Lower Hudson River Valley (no adverse effects were predicted for the tree swallow). Waterfowl are abundant, which is expected given the high habitat quality of many areas of the Hudson River (see, Section 2.6.5 of ERA Addendum). The presence of one breeding colony of great blue herons does not indicate that they are breeding throughout the Lower Hudson River Valley. Similarly, the mixed success of bald eagle nests along the Hudson River in the last several years does not indicate that the bald eagle is re-established along the Hudson River. Certainly, it is encouraging to see some successful nesting, but it is too early to call the Hudson River population re-established. NYSDEC has been collecting eagle serum, prey and unhatched eggs for several years to evaluate contaminant loads throughout the eagles ecosystem (Nye, 2000). Preliminary PCB results from only two samples are high enough to be of concern, and more data on PCB concentrations in birds along the Hudson River are expected to be available in late 2000/early 2001 (Secord, 2000).

The abundance of raccoons along the Hudson River is addressed in the response to EL-1.22. Although mink and river otter are present along the Hudson River, their numbers are generally low. Preliminary results from a NYSDEC study (Mayack, 1999) indicate that PCBs may adversely affect litter size and possibly kit survival of river otter in the Hudson River.

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

No significant comments were received on Section 5.1.

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

No significant comments were received on Section 5.1.1.

5.1.1.1 Measurement Endpoint: Comparisons of Modeled Sediment Concentrations to Guidelines

Response to EF-1.23

Tables 3-2 and 3-3 are revised to read, "Tables 3-6 and 3-7." Although the predicted concentrations of PCBs in sediment consistently underestimate the mean concentrations measured at RMs 152, 113, 90 and 50, they do fall within the range of the sampled concentrations for all RMs except for RM 47 (see, Figure 3-7). In addition, the predicted mean sediment concentrations are based on dicholoro to hexachloro homologues and therefore are expected to be slightly lower than total PCB concentrations. Although average TOC values were generally greater than the TOC of 2.5% used by Farley et al. (1999), a TOC of 2.5% was used to provide consistency in the model (see response to EF-1.16).

Response to EF-1.24

The first complete sentence is revised to read, "Forecast sediment concentrations exceed the NYSDEC benthic aquatic life chronic toxicity criterion at RMs 152 and 113 for the duration of the modeling period based on the 95% UCL."

Response to EF-1.25

The correction of the organic carbon-normalized SEL (Persaud et al., 1993) from 1.3 mg/kg to 13 mg/kg in Table 5-1 and the associated text is noted. Ratios in Table 5-1 were calculated using the correct organic carbon-normalized SEL of 13 mg/kg.

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

No significant comments were received on Section 5.1.2.

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

Response to ES-1.1 and EF-1.26

The change in the NYSDEC surface water standard for the protection of wildlife from 0.001 μg/L total PCBs to 1.2×10^{-4} μg/L in 1998 (6 NYCRR Part 703) is noted. Table 5-2 (see, Section III) is revised accordingly. Use of the earlier standard underestimated the ratio of predicted whole water concentrations to the wildlife standard by an order of magnitude.

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

Response to EF-1.27

The assumption that measurements of young-of-year spottail shiner and age 1 pumpkinseed are equivalent to concentrations in mature adults may underestimate concentrations of PCBs in those species and animals that feed on them. The TRV used in the ERA Addendum for the spottail shiner on a NOAEL basis is 1.6 mg/kg, not 15 mg/kg. Therefore, if comparisons are made between field and laboratory based NOAELs, the difference is reduced to three-fold, as stated by the commenter.

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

No significant comments were received on Section 5.2.1.1.

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

Response to EF-1.28

The FISHRAND model generates lipid-normalized (and wet weight) fillet concentrations. The model does not explicitly model an egg concentration, which was developed based on the correlation between lipid normalized egg and whole body PCB concentrations (Niimi, 1983).

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

Response to EF-1.29

RM 133 is revised to read 113. The first sentence of Section 5.2.1.3 is revised to read, "As literature-derived TRVs were based on whole body concentration studies, the fish fillets were converted to whole body for direct comparison."

Response to EF-1.30

The comparison of the concentrations of TCDD in fish tissue associated with low and high levels of risk to Great Lakes receptors (EPA, 1993) is provided below. The study used by USEPA (1993) to establish concentrations of TCDD associated with low risk to piscivorous fish, Walker et al. 1992, is presented in Table B-7. This study found that for waterborne exposures, a residue of 0.034 ug TCDD/kg ww in lake trout eggs (estimated by USEPA to be about 0.050 ug TCDD/kg ww in adult fish) did not exhibit significant effects relative to controls. The study used by USEPA to establish the concentration of TCDD associated with high risk to piscivorous fish is also presented in Table B-7. Walker et al. (1992) reported effects on fry survival at 0.055 ug TCDD/kg ww in trout eggs (estimated by USEPA to be about 0.075 ug/kg ww in parent fish). The ERA Addendum reports these egg concentrations in Table B-7 as both wet weight and as lipid normalized concentrations (0.43 and 0.7 ug TEQ/kg lipid, respectively). However, in the Responsiveness Summary for the ERA, USEPA selected a more recent study for the development of TRVs, Walker et al. (1994). This study is selected for development of TRVs for salmonids because it reports a NOAEL that was measured using a different and more realistic exposure route, maternal transfer of TCDD to eggs. This study reported a NOAEL of 0.023 ug TCDD/kg ww egg (0.29 ug /kg lipid) and a LOAEL of 0.05 ug TCDD/kg ww (0.6 ug/kg lipid). Thus, the NOAEL TRV developed for the ERA Addendum is slightly lower than that developed in USEPA (1993) and the LOAEL TRV is similar.

For mammals and birds, the ERA Addendum estimated risk on the basis of dietary dose (mg TCDD/kg body weight/day), rather than as concentration in diet (mg TCDD/ kg fish). However, the studies used in the USEPA (1993) report to develop risk-based concentrations in prey were the same studies that were used to develop TRVs for some receptors in the present risk assessment. These are Murray et al. (1979) and Nosek et al (1992), which are included in Tables B-11 and B-18. The USEPA (1993) approach assumed that avian and mammalian receptors consume 100% fish, whereas the ERA Addendum assumes that most receptors consume a variety of prey types, which include fish in most cases. Therefore, although the same studies were used in some cases, different assumptions were used about the types of prey consumed by Hudson River receptors in comparison to Great Lakes receptors, and conclusions about protective levels in fish prey items are not directly comparable.

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

No significant comments were received on Section 5.2.1.4.

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

Response to EF-1.31

The rationale for not applying interspecies uncertainty factors to field studies is provided in the response to comment EF-1.7. Because white perch and yellow perch are not in the same taxonomic family, if the NOAEL TRV for the white perch were used to develop a NOAEL TRV for the yellow perch, an interspecies uncertainty factor of 10 would be applied. In that hypothetical case, the NOAEL TRV for the yellow perch would be 0.31 mg PCBs/kg tissue, rather than the 0.16 mg/kg laboratory-based NOAEL-based TRV, and the NOAEL-based toxicity quotients would be approximately half of what was reported in the ERA Addendum.

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

No significant comments were received on Section 5.2.1.6.

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

No significant comments were received on Section 5.2.1.7.

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

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 Striped Bass

Response to EL-1.19

Striped bass are known to occur throughout the upper portion of the Lower Hudson River (NOAA, 1985). Concentrations of PCBs in striped bass were related to concentrations in largemouth bass in the absence of explicitly modeled results from either the Farley or FISHRAND models. There are no monitoring data available for largemouth bass at RMs 90 and 50; thus, ratios could not be estimated for these locations. Although the Farley model provides results for Food Web Region 2, this area is a much larger area than that used for the remaining fish species. Note also that the ERA provides risk estimates based on observed concentrations in striped bass, and that these results suggest risk to the striped bass at some locations in the Lower Hudson River (see, Table 5-36 [unchanged] in ERA and ERA Responsiveness Summary [USEPA, 1999b and 2000b]).

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

No significant comments were received on Section 5.2.1.10.

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

No significant comments were received on Section 5.2.2.

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

No significant comments were received on Section 5.2.2.1.

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

No significant comments were received on Section 5.2.3.

5.2.3.1 Measurement Endpoint: Comparisons of Modeled Sediment Concentrations to Guidelines

No significant comments were received on Section 5.2.3.1.

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

No significant comments were received on Section 5.2.4.

5.2.4.1 Measurement Endpoint: Evidence from Field Studies

Response to EL-1.20

As discussed in the Responsiveness Summary for the ERA (see, USEPA, 2000b, responses to EG-1.9 and EG-1.34, EG-1.38, EP-1.1, EP-2.10, and EL-1.46), the presence of healthy populations does not indicate that PCBs have no adverse 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 shortnose sturgeon in particular has benefitted from being listed as an endangered species and the fishing ban in the Hudson River. These factors have allowed the population of Hudson River shortnose sturgeon to increase despite any potential adverse effects from PCB exposure.

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

No significant comments were received on Section 5.3.

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

No significant comments were received on Section 5.3.1.

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

No significant comments were received on Section 5.3.1.1.

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

No significant comments were received on Section 5.3.1.2.

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

No significant comments were received on Section 5.3.1.3.

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

No significant comments were received on Section 5.3.1.4.

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

No significant comments were received on Section 5.3.2.

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

No significant comments were received on Section 5.3.2.1.

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

No significant comments were received on Section 5.3.3.

5.3.3.1 Measurement Endpoint: Evidence from Field Studies

No significant comments were received on Section 5.3.3.1.

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

Response to EF-1.32

The comparison of current trends in bird usage to historical usage (e.g., prior to GE's use of PCBs at its two Hudson River facilities) would be of limited use in assessing ecological risk due to PCBs due to the changes in habitat use that have occurred along the Hudson River (and, for migratory species, other areas as well) over the last 50 years. The complexity of the ecosystem and number of variables affecting bird usage does not allow direct effects to be determined based on PCB concentrations.

5.4.1 Do 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 Tri+ PCBs to Waterfowl (Mallard)

No significant comments were received on Section 5.4.1.1.

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

No significant comments were received on Section 5.4.1.2.

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

No significant comments were received on Section 5.4.1.3.

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

No significant comments were received on Section 5.4.1.4.

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

No significant comments were received on Section 5.4.2.

5.4.2.1 Measurement Endpoint: Comparison of Modeled Water Concentrations to Criteria

No significant comments were received on Section 5.4.2.1.

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

Response to EL-1.21

The Christmas bird count species records are known to contain errors (Cornell University, 1999). Therefore, the database cannot be used for scientific studies until it has been reviewed and corrected. In addition, count efforts (e.g., number of participants, skill level) are not consistent between years or count circles. Based on these factors, it is difficult to discern any meaningful trends in the data without intensive data analyses. In general, the greatest number of species was observed near the mouth of the Hudson River, which is consistent with the locations of various habitats.

5.4.3.1 Measurement Endpoint: Observational Studies

No significant comments were received on Section 5.4.3.1.

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

No significant comments were received on Section 5.5.

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

No significant comments were received on Section 5.5.1.

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

No significant comments were received on Section 5.5.1.1.

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

No significant comments were received on Section 5.5.1.2.

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

No significant comments were received on Section 5.5.1.3.

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

No significant comments were received on Section 5.5.1.4.

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

No significant comments were received on Section 5.5.2.

5.5.2.1 Measurement Endpoint: Comparison of Modeled Water Concentrations to Criteria

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 Piscivorous Bird Populations?

No significant comments were received on Section 5.5.3.

5.5.3.1 Measurement Endpoint: Observational Studies

No significant comments were received on Section 5.5.3.1.

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

No significant comments were received on Section 5.6

5.6.1 Do Modeled Total and TEQ-Based PCB Dietary Doses to Insectivorous 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)

No significant comments were received on Section 5.6.1.1.

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

No significant comments were received on Section 5.6.1.2.

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

No significant comments were received on Section 5.6.2.

5.6.2.1 Measurement Endpoint: Comparison of Modeled Water Concentrations to Criteria 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 Insectivorous 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 (i.e., Survival and Reproduction) of Local Omnivorous Mammal Populations (as represented by the Raccoon)

No significant comments were received on Section 5.7.

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

No significant comments were received on Section 5.7.1.

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

No significant comments were received on Section 5.7.1.1.

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

No significant comments were received on Section 5.7.1.2.

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

No significant comments were received on Section 5.7.2.

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

No significant comments were received on Section 5.7.2.1.

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

No significant comments were received on Section 5.7.3.

5.7.3.1 Measurement Endpoint: Observational Studies

Response to EL-1.22

The ERA Addendum focuses on fish and wildlife found along the Hudson River. The raccoon was selected to represent omnivorous mammal populations living near the Hudson River. Although a large proportion of the raccoon population in the Lower Hudson River obtains food from sources other than the Hudson River, those individuals using the Hudson River as their primary food source may experience adverse effects.

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

No significant comments were received on Section 5.8.

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

No significant comments were received on Section 5.8.1.

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

No significant comments were received on Section 5.8.1.1.

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

No significant comments were received on Section 5.8.1.2.

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

No significant comments were received on Section 5.8.2.

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

No significant comments were received on Section 5.8.2.1.

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

No significant comments were received on Section 5.8.3.

5.8.3.1 Measurement Endpoint: Observational Studies

No significant comments were received on Section 5.8.3.1.

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

No significant comments were received on Section 5.9.

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

No significant comments were received on Section 5.9.1.

5.9.1.1 Measurement Endpoint: Inferences Regarding Shortnose Sturgeon Population

No significant comments were received on Section 5.9.1.1.

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

No significant comments were received on Section 5.9.2.

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

No significant comments were received on Section 5.9.2.1.

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

No significant comments were received on Section 5.9.3.

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

No significant comments were received on Section 5.9.3.1.

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

No significant comments were received on Section 5.9.4.

5.9.4.1 Measurement Endpoint: Comparisons of Modeled Sediment Concentrations to Guidelines

No significant comments were received on Section 5.9.4.1.

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

No significant comments were received on Section 5.9.5.

5.9.5.1 Measurement Endpoint: Observational Studies

No significant comments were received on Section 5.9.5.1.

5.10 Evaluation of Assessment Endpoint: Protection of Significant Habitats

No significant comments were received on Section 5.10.

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

No significant comments were received on Section 5.10.1.

5.10.1.1 Measurement Endpoint: Inferences Regarding Receptor Populations

No significant comments were received on Section 5.10.1.1.

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

No significant comments were received on Section 5.10.2.

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

No significant comments were received on Section 5.10.2.1.

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

No significant comments were received on Section 5.10.3.

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

No significant comments were received on Section 5.10.3.1.

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

No significant comments were received on Section 5.10.4.

5.10.4.1 Measurement Endpoint: Observational Studies

No significant comments were received on Section 5.10.4.1.

6.0 UNCERTAINTY ANALYSIS

No significant comments were received on Section 6.0.

6.1 Conceptual Model Uncertainties

No significant comments were received on Section 6.1.

6.2 Toxicological Uncertainties

Response to EF-1.33

The ERA Addendum does not attempt to examine effects from congeners that have different mechanisms of action from the dioxin-like congeners because much less data are available on the non-dioxin effects. The effect of those congeners on risk is unknown.

6.3 Exposure and Modeling Uncertainties

No significant comments were received on Section 6.3.

6.3.1 Natural Variation and Parameter Error

No significant comments were received on Section 6.3.1.

6.3.2 Model Error

No significant comments were received on Section 6.3.2.

6.3.2.1 Uncertainty in the Farley Model

Response to EF-1.34

The goal of the fate, transport and bioaccumulation models is to capture the general, long-term trend of PCBs in water, sediment, and fish. Capturing the year-to-year variability is not a goal of the ERA Addendum or the Farley modeling effort. The Phase 2 sediment data shown in Figure 3-7 of the ERA Addendum is for the 0-5 cm layer. The majority of the data falls between the modeled results for the 0-2.5 cm and 2.5-5 cm layers, capturing the trend in the data with the exception of RM 47, which falls above the modeled data. The average of the sediment sample data would fall near the average of the modeled results from the 0-2.5 cm and 2.5-5 cm layers. This is good agreement, particularly because the sediment data are not spatially representative of the Lower Hudson River sediments and show the heterogeneity of the sediments. This heterogeneity results from the significant variability in deposition history for sections of the river. The Phase 2 high resolution cores show a fourfold decline between the Albany Turning basin and Lents Cove (see Figure 3-60 of the DEIR, USEPA, 1997). The cesium-normalized PCB concentrations show a three-fold decline in the Lower Hudson River in Figure 3-64 of the DEIR (USEPA, 1997).

Response to EF-1.35

USEPA agrees that there is uncertainty associated with changes in the upstream boundary conditions and potential releases from the remnant deposits. This is addressed in the response to comment EF-1.10.

6.3.2.2 Uncertainty in FISHRAND Model Predictions

Response to EF-1.36

The sensitivity analyses presented in the RBMR (USEPA, 2000a) addressed parameters in the FISHRAND model (e.g., growth rate, lipid). These parameters were adjusted in the FISHRAND model to optimize the fit between predicted body burdens and observed body burdens for the period of the hindcast (i.e, calibration). In terms of TRVs, lipid normalization, which is how the egg versus tissue TRVs were developed, represents a standardization of the TRV results. Because the normalization is based on observed lipid in the egg and tissue, respectively, there is no sensitivity to evaluate. The fillet to whole body ratios, derived on the basis of large datasets of observed ratios between percent lipid in the fillet and the whole body, which is estimated at 1.5 for brown bullhead and 2.5 for largemouth bass, would reduce estimated risks by these factors if the "true" ratio were 1:1. If the "true" ratio were higher, risks would accordingly increase, but the increase is unlikely to be even a factor of 2 (i.e., from 2.5 to 5). Only the river otter and bald eagle risk estimates rely on predicted body burdens using the 2.5 ratio (otter and eagle are assumed to consume largemouth bass), and given the magnitude of risks for these receptors, a decrease in risk by a factor of 2.5 would not change the overall conclusions of the ERA Addendum.

Regarding exposure media concentrations, the FISHRAND model incorporates annual average sediment and monthly average water concentrations as inputs. Sediment concentrations are stable and show little temporal variability. That is, monthly average sediment concentrations are not significantly different from annual average concentrations. Water concentrations are much more variable, and consequently, the FISHRAND model explicitly incorporates this variability by characterizing water concentrations on a monthly basis. For the avian and mammalian receptors, which are assumed to be exposed to summer average water concentrations, this period of exposure coincides with the typical length of the toxicity study. This is also the period of greatest feeding, particularly for migratory or hibernating species, and the period for which concentrations of PCBs in water are highest. Thus, an annual average (which is longer than the period of exposure in the toxicity study) would decrease risks, but given the magnitude of the predicted TQ, again, the overall conclusions of the ERA Addendum would not change.

6.3.3 Sensitivity Analysis for Risk Models for Avian and Mammalian Receptors

No significant comments were received on Section 6.3.3.

7.0 CONCLUSIONS

Response to EG-1.23

The ERA and ERA Addendum are based on USEPA policy and guidance and standard ecological risk assessment practices. To the extent the ERA and ERA Addendum are used in decision-making, along with the Human Health Risk Assessment, the Data Evaluation and Investigation Report, and the results of the modeling, USEPA will document that use in the FS, the Proposed Plan, and the Record of Decision (see also, responses to EG-1.1, EG-1.3, and EG-1.4). Moreover, USEPA disagrees with the commenter's statement that the conclusions of the ERA Addendum are "unambiguously contradicted" by data (see, response to EG-1.20).

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

Response to EF-1.37

The uncertainty in sediment and water forecasts is estimated to be on the order of a factor of two. This value is based on professional judgment. The parameterized model shows agreement between the model predictions and the calibration data of a factor of two or better (see, Figures 3-5 and 3-7 of the ERA Addendum). As stated in the ERA Addendum (p. 71), "the fact that the model is able to reproduce the general trends of the existing sediment, water and fish data suggests that the model uncertainty from parameterization is similar to the scale of the differences between the model calibration and the data themselves."

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

No significant comments were received on Section 7.2.

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

Response to EF-1.38

The sensitivity of tree swallows to PCBs as compared to other insectivorous birds is not well documented. Other insectivorous bird species may be more sensitive to PCBs; however, even with an uncertainty factor of ten to account for interspecies variation, most TQs would still fall below one. The tree swallows and other insectivores in the Upper Hudson River may experience reproductive impairment due to PCB exposure.

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

No significant comments were received on Section 7.4.

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

No significant comments were received on Section 7.5.

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

No significant comments were received on Section 7.6.

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

No significant comments were received on Section 7.7.

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

No significant comments were received on Section 7.8.

7.9 Assessment Endpoint: Protection of Threatened and Endangered Species

No significant comments were received on Section 7.9.

7.10 Assessment Endpoint: Protection of Significant Habitats

No significant comments were received on Section 7.10.

7.11 Summary

No significant comments were received on Section 7.11.

APPENDICES

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

<u>Response to EF-1.2, EL-1.5 and EG-1.12b</u>

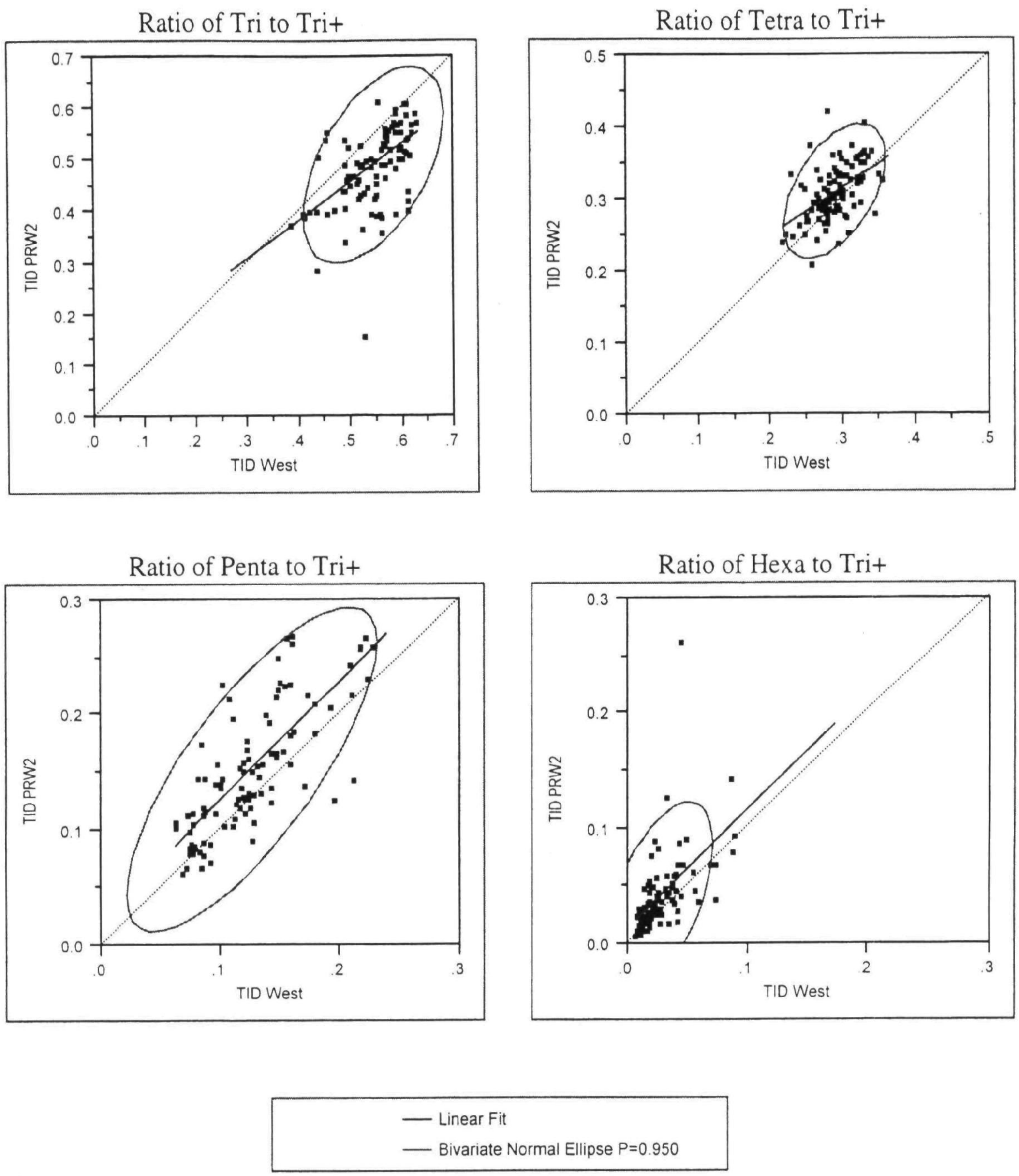
The state variable modeled by HUDTOX is tri and higher PCBs (Tri+ PCBs). This variable was chosen for the Hudson River because historic data exist for this form of PCBs, primarily from the 1977 and 1984 NYSDEC sediment surveys and the USGS water column monitoring program (1977 to the present). There is little historical homologue data on which to base a model calibration. The Farley models are based on di through hexa homologues, requiring a conversion from Tri+ PCBs to homologues.

USEPA's conversion uses HUDTOX load estimates at Federal Dam (not Thompson Island Dam) to predict the upstream boundary loads for the Farley model during the forecast period (only). The processes used to convert HUDTOX Tri+ output to estimates of dichloro through hexachloro homologues are explained in detail in Appendix A of the ERA Addendum (USEPA, 1999c), and are based on observed data stratified by season. As with any forecast, there is uncertainty in these estimates; however, uncertainty in the ratios between trichloro through hexachloro homologues is expected to be much less than the uncertainty in forecasting total Tri+ PCB load, due to the uncertainty in future contributions from the GE Bakers Falls source. Uncertainty is greatest for the dichloro homologue, but this is the homologue with least significance to bioaccumulation in fish. Variability in the homologue composition of future loads would have little effect on total PCB concentrations in water and sediment, but would have an impact on bioaccumulation, as the more chlorinated homologues generally have greater apparent bioaccumulation factors.

Thompson Island Dam and Waterford data were used to estimate the change in homologue ratios relative to Tri+. This approach is reasonable for the following two reasons. As documented in Figures A-24 to A-27 of Appendix A of the ERA Addendum (USEPA, 1999c), the ratios of the homologue groups trichloro to hexachlorobiphenyl to the Tri+ sum have not varied greatly over time. Indeed, most of the variation seen is due to seasonal changes that were addressed in the Appendix A. Additionally, the results show very consistent trends over the period 1996-1998. This period was utilized for the ratio estimates delivered at Thompson Island Dam. These results suggest that the variations in these ratios have been well characterized and can be extrapolated without introducing large amounts of uncertainty. Notably, the dichloro homologue has a much greater degree of variability compared to the other four groups. However, its importance to downstream exposures is much less, given that the dichloro homologue group does not tend to bioaccumulate and thus constitutes a negligible portion of fish body burdens (see, Appendix K of the ERA, USEPA, 1999b). As a result, human and ecological exposures to this homologue group are minimal. Thus, the greater uncertainty in the dichloro homologue loads does not limit the usefulness of the loading calculation.

As to the examination of the changes in load between TI Dam and Waterford and the effect on the homologue ratios, the presence or absence of a large upstream load above Rogers Island does not affect the nature of the transport processes downstream. Specifically, the processes of sediment-to-water exchange, gas exchange, and similar geochemical processes will occur in any event. Thus the 1993 data are not inappropriate for examining the effects of transport between TI Dam and Waterford. Recognizing that the geochemical processes will vary temporally, the results have been grouped according to season. Additionally, the 1993 USEPA data have the distinction of either tracking or integrating PCB loads in such a fashion so as to closely document the changes in homologue ratios between these stations. Specifically, the transect data tracks and monitors a single water parcel through the Upper Hudson during each sampling event. The flow-averaged samples integrate PCB loads on the basis of flow over a 15-day period. Thus, a large number of randomly collected data points is not necessary to establish the degree of change between the stations.

Regarding the representativeness of the USEPA Thompson Island Dam monitoring station, USEPA has previously acknowledged that its Thompson Island Dam monitoring location as well as the west wing station occupied by GE do not match the PCB load estimates obtained from a center channel monitoring location (USEPA, 1999b). However, the USEPA does not agree that the center channel is the true measure of the load at the Thompson Island Dam but rather, that the center channel load is probably closer to the true value, which lies between the loads derived from the center channel and west-wing-wall locations. Nonetheless, the use of the long-term wing wall station in Appendix A does not examine load but simply the ratios among the congeners. Ratios at this station are similar but not identical to those of the center channel. Figure EG-1.12 illustrates this with data from GE, showing the ratio of each homologue group to Tri+ at the Thompson Island Dam west wall and center channel stations. These results show the west wing wall station to have a higher proportion of trichloro homologues and lower proportions of tetra through hexachloro homologues relative to the center channel station. Thus, the use of the west wing wall station may slightly underestimate the Upper Hudson contribution of the heaviest congeners. More importantly, these figures show that each of the homologue groups represents a fairly consistent proportion of the



Notes:

TID West = GE's Thompson Island Dam West-wing-wall station
TID PRW2 = GE's central channel station near the Thompson Island Dam
Dash line represents the 1:1 unity line.

Figure EG-1.12
Relationship Between the TI Dam West and Central Channel Stations for Homologue to Tri+ Ratios
GE Data (1997 – 1999)

Tri+ sum regardless of the choice of monitoring location. Additionally, the difference between the stations for each homologue group is less than or equal to the variability of the homologue-to-Tri+ ratio. This is illustrated in each instance by the 95th percentile ellipse that is elongated and close to the line of perfect agreement. Lack of correlation between the stations would tend to yield a more circular ellipse, indicating lack of correlation.

Ultimately, it must be noted that the ratios developed from the Thompson Island Dam west station were only used to determine the mean proportion of each homologue in the Tri+ sum, corrected on the basis of flow or season. The correlations behind these ratios have uncertainties associated with them but these uncertainties should be relatively small and unlikely to affect the long-term forecast results. Specifically, the differences in the ratios seen at the center channel and west wing wall stations represent likely bounding values. Given that these differences were smaller than the actual variations in the ratios as a function of flow or season, the uncertainty derived from the use of the Thompson Island Dam west wing wall monitoring station does not introduce an important additional degree of uncertainty.

Response to EF-1.39

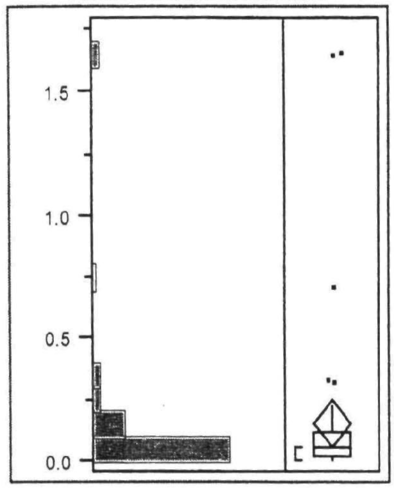
The text on p. A-6 paragraph 1 of the ERA Addendum is revised to read:

The TI Dam data from 1996-1998 are grouped by season for each homologue of concern in Figures A-28 through A-31. The data are grouped by flow in Figures A-32 through A-35. The best separation (greatest distance between the Tukey-Kramer circles) of the means is given by grouping on season. The ratio variations among these groups are relatively small, typically only a few percent of the total Tri+ mixture. The importance of these variations increases as the fraction of the homologue decreases, as would be expected. Thus, the summer to spring variation of 8 percent (54 - 46 percent) in the trichloro homologue percentage represents about 15 percent of the total trichloro mass. However, the 2.4 percent summer to fall-winter change in the hexachloro homologue ratio represents nearly a 50 percent decline in the ratio from fall-winter to summer. These results should be compared to the dichloro homologue results, which show large changes on both absolute and relative scales.

Response to EL-1.23

A total of 49 field duplicate samples at the Thompson Island Dam and Waterford stations are given in the GE database (QEA, 1999). These samples are from the end of 1995 through 1999. Averages of the values were not used because the concentration and homologue pattern of the samples are similar. Using the sample as opposed to the average of the sample pair would not change the conclusions of the analysis. This is seen in Figure EL-23a which shows the relative percent difference (RPD) for the concentration of tri through hexa homologues. The differences are small for the majority of samples with a median RPD of only 6%. Because the homologue patterns were used in the conversion analysis, a comparison of the homologue distributions is more relevant.

Relative Percent Difference



Quantiles

maximum	100.0%	167
	99.5%	167
	97.5%	167
	90.0%	33
quartile	75.0%	12
median	50.0%	5.5
quartile	25.0%	2.5
	10.0%	1.1
	2.5%	0.18
	0.5%	0.08
minimum	0.0%	0.08

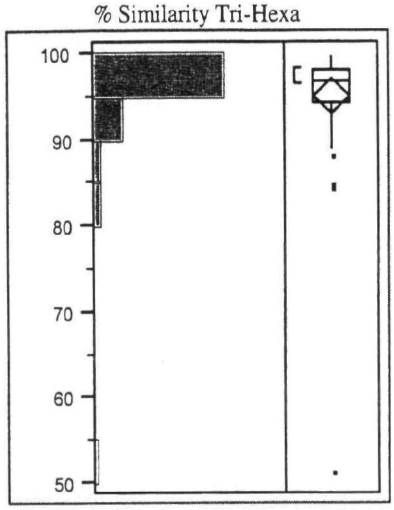
Moments

Mean	15
Std Dev	34
Std Error Mean	4.8
Upper 95% Mean	25
Lower 95% Mean	58
N	49
Sum Weights	49

Sources: GE, 1999

$$\text{Relative Percent Difference } (RPD) = \frac{|Measurement1 - Measurement2|}{\frac{(Measurement1 + Measurement2)}{2}} \times 100\%$$

Figure EL-1.23a
Relative Percent Difference for GE Water Column Sample Duplicates at the TI Dam



Quantiles		
maximum	100.0%	99.7
	99.5%	99.7
	97.5%	99.7
	90.0%	99.1
quartile	75.0%	98.0
median	50.0%	96.8
quartile	25.0%	94.4
	10.0%	89.0
	2.5%	59.6
	0.5%	51.3
minimum	0.0%	51.3

Moments	
Mean	95.0
Std Dev	7.23
Std Error Mean	1.03
Upper 95% Mean	97.2
Lower 95% Mean	93.0
N	49
Sum Weights	49

Sources: GE, 1999

$$\%Similarity = \sum_{i=3}^{6} \left(Minimum\,Value\,of\,[Homologue]_1, [Homologue]_2 \right)$$

For example:

	Sample 1	Sample 2	Lesser	
Tri (%)	45	48	45	
Tetra	22	20	20	
Penta	18	19	18	
Hexa	15	13	13	
	100%	100%	96%	= 96% Similarity

Figure EL-1.23b
Percent Similarity of GE Water Column Sample Duplicates for the Tri through Hexa Homologues at the TI Dam

Figure EL-1.23b shows the percent similarity for the tri through hexa homologues. Percent similarity is a means of comparing distributions. The lower value for each of the homologue percent of Tri+ PCBs is summed. The closer the sum is to 100%, the more similar are the distributions. The agreement between the homologue distributions also is satisfactory with a median value of 96.8% and a mean value of 95.1%.

Response to EL-1.24

The conversion from Tri+ PCBs to homologues assumes that the factors can be applied to a 40-year forecast because the geochemical processes creating the ratios among the homologues are unlikely to change without remediation. The homologue patterns in the water column are generated from the sediment inventories. Without altering the sediment inventory, the patterns in the water column should remain relatively constant. Note the small variation in the mean mass percent of Tri+ PCBs using Thompson Island Dam data for tri through penta given by +/- two standard errors. More variation is evident for di and hexa, but this is of less concern because Tri+ PCBs are modeled for the fish body burdens and hexa is a small fraction of Tri+ PCBs. The mean mass percent ratio for Waterford/TID is less well constrained. However, the di homologue is less important to the ERA Addendum, which is primarily based on exposure to Tri+ PCBs through the food chain (see, EG-1.12 for further discussion). The commenter is correct in noting that the discussion presents the estimation of the Thompson Island Dam to Waterford correction first and the Thompson Island Dam ratio estimates second.

Response to EL-1.25

USEPA Phase 2 water column samples were used to determine a correction to the PCB load between the Thompson Island Dam and Waterford. 12 samples from the Thompson Island Dam and 12 samples from the Waterford station were used in the calculation. There are a total of 53 water column samples taken at Waterford by GE in 1991 and 1992. Although the GE data set provides more than four times the number of USEPA samples at the Waterford station, the collection method that generates the data is inappropriate for this analyses. This is discussed in the ERA Addendum, Appendix A, p. A-3, as follows:

> The data set to establish the TI Dam to Waterford ratio is limited. In particular, the 1991 GE samples at TI Dam and Waterford were not timed to capture the same parcel of water as it traveled from the TI Dam to Waterford. Thus, these samples do not directly track the changes to the water column loads originating from the geochemical processes which occur en route. Given the relatively low number of samples collected at the two stations that year, there are not enough samples to develop an average ratio to accurately represent the effects of the geochemical processes as a function of flow and season. Table A-1 lists the calculated time for each flow rate at Fort Edward for water to travel from TI Dam to Waterford and the hours between sampling at these stations. None of the travel times are similar to the

sampling times, indicating that the sampling were not timed to capture the same parcel of data. Because of this aspect of the GE sampling method, only the USEPA Phase 2 samples, which were purposely timed to capture the same parcel of water, will be used to compare TI Dam to Waterford. As discussed below, all of the GE and Phase 2 samples at TI Dam will be used to examine the temporal changes in homologue percentages.

Response to EL-1.26

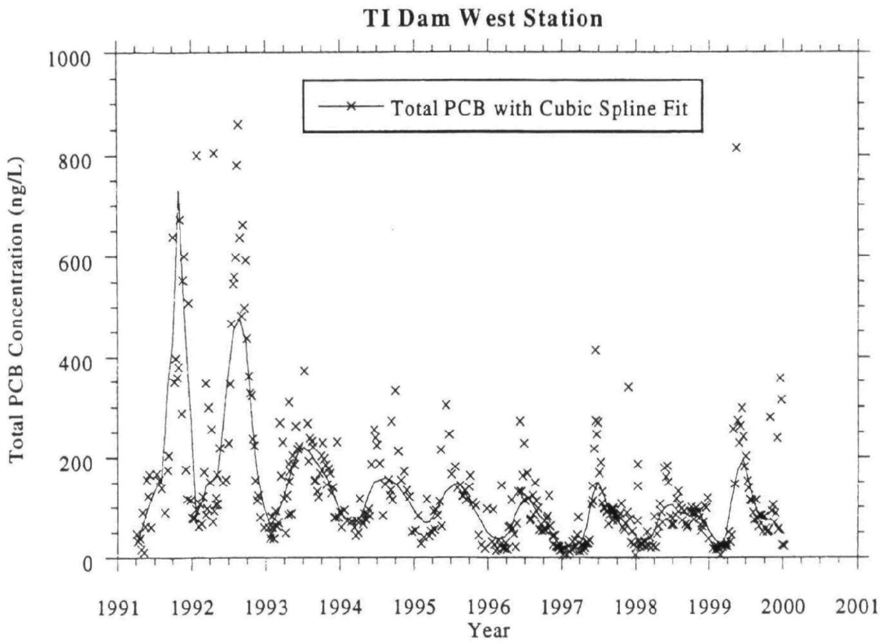
Figure EL-1.26a shows the most recent GE water column data at the Thompson Island Dam (QEA, 2000). A decline in PCB concentrations at the Thompson Island Dam is not evident in the data presented in this figure. This is further discussed in the Responsiveness Summary for the LRC (USEPA, 1999d). While a decline in summer loads has been noted, this can be largely ascribed to a decline in flow (see, Figure EL-1.26b). The fact that water column concentrations have not declined with time after 1996 suggests a mechanism that releases PCBs from the sediment and establishes a constant water column condition regardless of flow. Note how loads correlate with flow in Figure EL-1.26c.

Response to EL-1.27

It is likely that these factors will remain constant for decades to come because the homologue patterns found in the water column are a reflection of the patterns found in the source of the contamination. The primary source of contamination is the river sediments. Without remediation, PCBs will continue to be released from the sediment in the appropriate proportions found there. Additionally, there are few data to establish an *a priori* basis for estimating a change in these ratios. Thus, these values are assumed constant. See, response to comment EG-1.12 and EL-1.24.

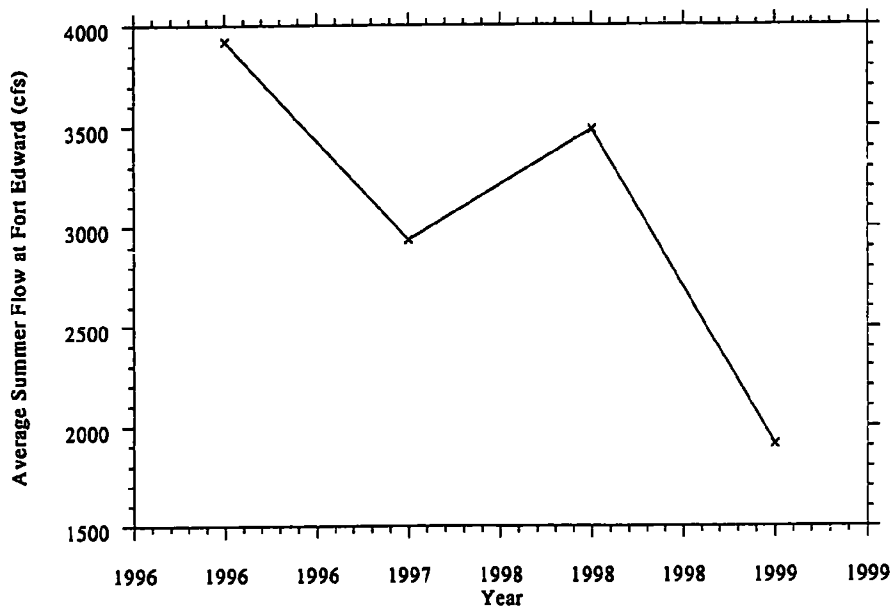
Response to EL-1.28

Between 1987 and 1990, there are no homologue data available because the GE monitoring program had not begun. The 1991 GE sample data was used to represent this time period. Releases from the Bakers Falls area, which occurred in 1991, might have yielded a different homologue pattern than was actually present in 1987 to 1990. Thus, the 1991 pattern might not be representative of the prior three years. However, the model output between 1987 and 1990 is not used in the ERA Addendum. Thus, the lack of data between 1987 and 1990 does not effect the results of the ERA Addendum.



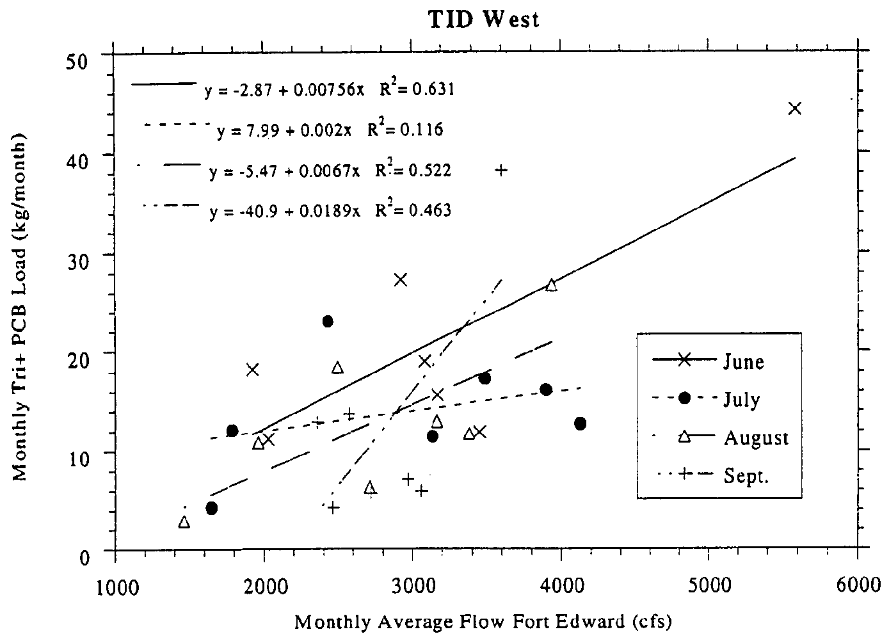
Source: GE, 2000

Figure EL-1.26a
Total PCB Concentrations at the Thompson Island Dam (1991-2000)



Source: Hudson River Database Release 4.2

Figure EL-1.26b
Fort Edward Summer Average Flows



Sources: Hudson River Database Release 4.2 and GE, 2000

Figure EL-1.26c
Tri+ Loads at the TI Dam Compared to Flow at Fort Edward

APPENDIX B - Effects Assessment

Response to EF-1.40

As discussed in the response to EF-1.6, Hansen et al. (1974) and USACE (1988) were reexamined and selected to develop TRVs.

Response to EF-1.41

As discussed in the response to EF-1.6, Hansen et al. (1974), Adams et al. (1989, 1990, 1992), and USACE (1988) were reexamined and selected to develop TRVs.

Endpoints of greatest relevance to the population and the ability of the population to successfully reproduce were considered the most significant endpoints in developing TRVs. Although other significant effects may have been observed at lower concentrations than those of the selected TRVs, these endpoints have less direct relevance to population-level endpoints.

Response to EF-1.42

Comment acknowledged. See response to EF-1.6.

Response to EF-1.43

Comment acknowledged. The revised sentences read:

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

Response to EF-1.44

Comment acknowledged. Tables B-5 and B-6 are revised to include USACE (1988).

Response to EF-1.45

The studies should be listed as dose-response studies, which estimate a NOAEL/LOAEL rather than an EL-no effect or EL-effect.

Response to EF-1.46

The table states in a footnote that the lipid values were reported by a referenced study for that species.

Response to EF-1.47

These toxicity endpoints are from the USACE (1988) study and were inadvertently included in Figure B-2. The sediments for this study were collected from the field and this study is not considered a laboratory study. The purpose of the figure was to visually illustrate the range of endpoints that were identified from the literature. Therefore, not all studies were included, but a few representative studies were selected to show this range.

Response to EF-1.48

The purpose of Figure B-3 was to visually illustrate the range of toxicity endpoints that were identified in the literature. Therefore, all studies were included in the evaluation, but very few studies were selected to visually represent this range.

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Risk Assessment
Revision

III. RISK ASSESSMENT REVISIONS

1. Summary

This section of the Responsiveness Summary presents the revised results of the baseline Ecological Risk Assessment for Future Risks in the Lower Hudson River (ERA Addendum). The revisions are based on modified forecast concentrations of PCBs in fish, sediment, and water, which in turn result from the revised upstream PCB boundary load into the Lower Hudson that was presented in the Revised Baseline Modeling Report (RBMR)(USEPA, 2000a), and its subsequent effects on output of the modeling for the Lower Hudson River. The revisions in this section also incorporate changes to the toxicity reference values (TRVs), based on comments received on the ERA and ERA Addendum. This section also compares the revised ecological risk results and associated conclusions with those of the ERA Addendum.

The overall conclusions from the ERA Addendum (USEPA, 1999c) remain unchanged. The revised calculations for the ERA Addendum show that there are ecological risks to receptors of concern (except the tree swallow) above USEPA levels of concern. In addition, site-related risks due to PCBs in Lower Hudson River are greatest for the top-level piscivorous receptors, such as the river otter and the bald eagle.

2. Introduction

Part III of this Responsiveness Summary summarizes the modifications made and presents the results of the revised risk calculations for the ERA Addendum. Those tables and figures that were modified are labeled "Revised." To facilitate in the ease of comparing revised results with the ERA Addendum results (USEPA, 1999c), all tables and figures have retained their number designations.

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

The RBMR (USEPA, 2000a) 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 ERA Addendum to reflect these new values. The changes in the HUDTOX and FISHRAND models reflected in the RBMR (USEPA, 2000a) 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, 1999a). 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 ERA Addendum, Tri+ PCB concentrations were used to predict sediment and water concentrations, while in this Responsiveness Summary, total PCB concentrations were used.

The models for the Lower Hudson River have been rerun due to changes made to the HUDTOX forecast discussed above. The current forecast of PCB loads from the Upper Hudson River given by HUDTOX are used. The revised Lower River fate and transport model is also used (Cooney, 1999). The minor changes made to the Farley models since the March 1999 Report (Farley et al., 1999) are discussed below.

2.1.1 Changes to the Farley Models between December 1999 and August 2000

Revised Farley fate, transport and bioaccumulation models (Cooney, 1999) were used for this Responsiveness Summary for the ERA Addendum. The changes to the fate and transport model are as follows (Cooney, 2000):

- The A_{DOC} for sediment was changed from 1 to 0.1; and
- A slight change to the solids balance that affects other flow parameters such as the settling and resuspension rates.

Changes to the bioaccumulation model are:

- The chemical and food assimilation for zooplankton have been set to 0.3;
- Striped bass and white perch lipid content are the average lipid content given by NYSDEC fish samples taken in the 1990s; and
- A minor correction to the prey pattern has been made. Striped bass in one compartment of the model fed upon white perch that were a year younger.

<u>Comparison of Forecasted Water, Sediment and Fish Data from the ERA Addendum and this Responsiveness Summary</u>

Comparisons were made between the Tri+ PCBs in the dissolved phase of the water column between the revised model output and the data presented in the ERA Addendum. As seen in Figure III-1, the R^2 ranged from 0.996 at RM 152 to 1.0 at RMs 90 and 50, indicating that there is virtually no difference between the original and revised results.

Comparisons were also made between the total PCBs in the water column (whole water) between the revised model output and the data presented in the ERA Addendum for future risks in the Lower Hudson River. As seen in Figure III-2, the R^2 ranged from 0.994 at RM 152 to 1.0 at RMs 90 and 50, indicating that there was again virtually no difference between the original and revised results.

The final comparisons were also made between the total PCBs in the sediment (0-2.5 cm) between the revised model output and the data presented in the ERA Addendum. As seen in Figure III-3, the R^2 ranged from 0.998 at RM 152 to 1.0 at RMs 113, 90, and 50, indicating that there is virtually no difference between the original and revised results of sediment concentrations.

These three comparisons show that although the Farley et al. (1999) was revised slightly, changes were minimal and did not significantly change calculated water and sediment concentrations.

2.1.2 Changes to FISHRAND between December 1999 and August 2000

In the RBMR (USEPA, 2000a), the FISHRAND model was formally recalibrated using Bayesian updating. Growth rate coefficients, TOC, lipid content, and K_{ow} distributions were all optimized within the constraints of the data.

Table 3-8 (Revised) shows the revised Tri+ PCB average and 95% UCL concentrations for benthic invertebrates for the duration of the modeling period. Overall, concentrations in benthic invertebrates are similar to those uses in the ERA Addendum. From 1993 to 2010-2011, concentrations are 0.8 to 1.0 times the prediction used in the ERA Addendum. In the later portion of the modeling period, concentrations are calculated to be up to 1.2 times higher than predicted earlier. This pattern was fairly consistent between locations.

Revised predictions for the two forage fish modeled, the spottail shiner and the pumpkinseed, are provided in Tables 3-9 and 3-10 (Revised), respectively. Predicted concentrations for the spottail shiner are about 4.4 to 9.0 times higher than those used in the ERA Addendum. The spottail shiner is the smaller forage fish species considered as a prey item by the other fish-eating wildlife receptors. Revised concentrations for the pumpkinseed are 2.0 to 3.1 times higher than those used in the ERA Addendum. Revised pumpkinseed concentrations were fairly consistent between locations and over the duration of the sampling period.

Revised predictions for the yellow perch and white perch were 0.8 to 2.9 times the values used in the ERA Addendum (Tables 3-11 and 3-12 Revised, respectively). There were fewer changes in the values predicted in the 25th percentile (0.8 to 1.4 and 0.7 to 1.0 times the original prediction for the yellow and white perch, respectively) than in the 95th percentile (1.4 to 2.9 and 1.6 to 2.2 times the original prediction for the yellow and white perch, respectively). The values for the median were between the two percentiles (0.9 to 1.6 and 0.9 to 1.3 times the original prediction for the yellow and white perch, respectively).

Revised concentrations for brown bullhead (Table 3-13 Revised) were higher than those used in the ERA Addendum. Concentrations were 1.1 to 1.9 times greater than earlier predictions. Concentrations were higher at all locations for the duration of the modeling period. The greatest increases were seen in the 25th percentile.

Revised concentrations for largemouth bass ranged from 0.45 to 3.17 times the values used in the ERA Addendum(Table 3-14 Revised). Revised largemouth bass concentrations were lower than initial predictions at RMs 152 and 113, with the exception of the 25th percentile at RM 152 after 1998. Revised concentrations at RMs 90 and 50 were 2.5 to 3.2 times higher than the initial predictions. The largemouth bass is used as the "large" piscivorous fish consumed by the river otter and bald eagle. Striped bass concentrations are calculated by applying a factor to the largemouth bass concentrations (see Table 3-2), thus the changes in concentration for the striped bass are proportional to the changes in concentration for the largemouth bass.

Revised exposure concentrations tables for avian and mammalian receptors based on the revised forecasts are provided in Tables 3-25 to 3-60 (Revised).

2.2 Changes in Toxicity Reference Values

Toxicity Reference Values (TRVs) for fish, mallard, bald eagle, mink, and river otter were revised from the ERA (USEPA, 1999b) and ERA Addendum (1999c) based on a reevaluation of toxicity studies, as discussed in the following paragraphs.

2.2.1 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 was 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 4-1).

• 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 4-1).

The field-based TRVs for the pumpkinseed, spottail shiner, and largemouth bass were revised from the ERA Addendum. For the pumpkinseed and largemouth bass, 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, because 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 was 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 4-1).

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

In the ERA Addendum, 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 4-1).

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

2.2.2 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.

Mallard Duck

The development of TRVs for exposure of mallards to PCBs was re-examined with consideration of two additional studies that were not identified in the literature studies that were conducted for the ERA Addendum. 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) was 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) was selected as the most appropriate study, given that it is 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 was 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 4-2).
- The NOAEL TRV for the mallard (growth effects) is: 1.6 mg/kg/day (Table 4-2).

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):

Bald Eagle

Upon reexamination, USEPA agrees that the data collected by Wiemeyer et al. (1993) does not support the development of the previous NOAEL total PCB TRV of 3.0 mg/kg for bald eagle egg concentrations. However, USEPA does not agree that because mean five-year production was not significantly reduced for the residue *interval* ranging from 5.6 to <13 mg PCBs/kg, 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 nests (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 4-2).

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 wet weight and lipid at the Powell River site, the weight wet concentration at East Vancouver Island is approximately 217 ng/kg. Because no significant difference was observed between the average hatching rate of the eggs collected from these two contaminated sites and the reference sites, the average concentration in eggs from the contaminated sites (214 ng/kg wet weight) was 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 4-2).

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.

2.2.3 Changes in Mammalian TRVs

USEPA acknowledges that for TEQ-based PCBs in the diet, a LOAEL should not be established from the Tillett 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 4-3).

3. Results

The overall conclusions drawn from the results of the ERA Addendum do not change as a result of the revised risk calculations. Tables from Chapter 3, 4, and 5 have been revised to reflect the changes due to modeling and TRVs. In some cases, toxicity quotients (TQs) have increased or decreased slightly, but these revisions do not affect the general text or the overall conclusions of the ERA Addendum. Specific changes to risk characterization tables are as follows:

Table 5-1: Predicted sediment concentrations (Tri+) were adjusted to reflect total PCB concentrations. None of the guidelines changed. Only average, rather than average and 95% UCL results are calculated. Conclusions are unchanged but ratios increase slightly at all river miles (i.e., RM 152, 113, 90, and 50).

Table 5-2: 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 USEPA criterion (1.2×10^{-4} μg/L). Conclusions are unchanged but risks have decreased slightly at all locations.

Table 5-3: Pumpkinseed field-based NOAEL changed from 0.5 to 0.3 mg/kg (based on Adams et al., 1992; same study but different value). Toxicity quotients at all locations increased to 1.0 or higher for the duration of the modeling period (1993 to 2018). Conclusions for pumpkinseed remain unchanged, but predicted toxicity quotients increased slightly. Previously, all locations had predicted toxicity quotients below one for a portion of the modeling period.

Table 5-4: 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, as all toxicity quotients remained below one, except for the 95th percentile at RM 152 in 1993.

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

Table 5-6: TEQ-based TRVs have not changed. Predicted pumpkinseed concentrations increased slightly or remained the same at all river miles. Conclusions have changed slightly, because revised risk estimates show that predicted toxicity quotients exceed one for a greater proportion of the modeling period at all river miles.

Table 5-7: TEQ-based TRVs have not changed. Predicted pumpkinseed concentrations increased slightly or remained the same at all river miles. Conclusions have changed slightly, previously all predicted toxicity quotients fell below one at all locations. Revised risk estimates show that predicted toxicity quotients are above one at RM 152 for the median in 1993, and above one but below ten for the 95th percentile until 2003. At RM 113 revised risk estimates exceed one for the 95th percentile until 1998. At RMs 90 and 50, revised risk estimates exceed one for the 95th percentile until 1996.

Tables 5-8 and 5-9: TEQ-based TRVs have not changed. Predicted spottail shiner concentrations increased slightly at all river miles. Conclusions are unchanged (predicted toxicity quotients below one for all locations and years).

Table 5-10: 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).

Overall conclusions have changed slightly: predicted toxicity quotients have increased at all locations.

Table 5-11: Laboratory-based TRVs for brown bullhead have changed: The original LOAEL was 1.5 based on Bengsston (1980) and the revised LOAEL is 0.93 based on Hansen et al. (1974). Overall conclusions have changed slightly, as predicted toxicity quotients have increased and exceed one at all locations for the duration of the modeling period.

Tables 5-12 and 5-13: The TEQ-based laboratory-based NOAEL and LOAEL for brown bullhead have not changed. Predicted concentrations for brown bullhead have remained the same or changed slightly at all locations. Overall conclusions have not changed: predicted toxicity quotients fall below one for all locations and years.

Table 5-14: The field-based TRV for white perch has not changed. Conclusions have changed slightly, previously only the 95th percentile toxicity quotient at RM 152 in 1993 was greater than one. Revised numbers predict the 95th percentile to exceed one at RM 152 until 2015, at RM 113 until 1999, and at RM 90 until 1995. The median TQ at RM 152 also was greater than one in 1993.

Table 5-15: The laboratory-derived NOAEL for the yellow perch increased slightly to 0.19 mg/kg based on the Hansen study from 0.16 mg/kg based on the Bengsston study. Overall conclusions have not changed, but predicted toxicity quotients for the yellow perch have decreased for the median and 25th percentile values for the later part of the modelling period.

Table 5-16: The laboratory-derived LOAEL for the yellow perch decreased to 0.93 mg/kg based on the Hansen study from 1.5 mg/kg based on the Bengsston study. Conclusions have changed slightly, previously all predicted toxicity quotients fell below one at all locations for the duration of the modeling period. Revised risk estimates show that predicted toxicity quotients are above one at RM 152 for the 25th percentile until 1997, the median until 1998, and generally above one for the 95th percentile until 2015. At RM 113 revised risk estimates exceed one for the median until 1995 and the 95th percentile until 2004. At RMs 90 and 50, revised risk estimates exceed one for the 95th percentile until 2000 and 1999, respectively.

Table 5-17: The TEQ-based NOAEL for the white perch has not changed. Predicted concentrations have increased slightly for the 95th percentile at RMs 113, 90 and 50. Conclusions have not changed.

Table 5-18: The TEQ-based LOAEL for the white perch has not changed. Predicted toxicity quotients exceed one for the 95th percentile at all river miles for a greater proportion of the modeling duration than predicted previously. Conclusions have not changed.

Table 5-19: The TEQ-based NOAEL for the yellow perch has not changed. Predicted concentrations for yellow perch have increased slightly at all locations and for all percentiles. Predicted toxicity quotients for the 95th percentile exceed one for the duration of the sampling period at all locations.

Predicted toxicity quotients for the median and 25th percentiles also exceed one for a portion of the modeling period at all locations.

Table 5-20: The TEQ-based LOAEL for the yellow perch has not changed. Predicted concentrations for yellow perch have increased slightly at all locations and for all percentiles. Predicted toxicity quotients for the 95th percentile exceed one for the duration of the sampling period at RM 152 and for a portion of the modeling period at other locations. Predicted toxicity quotients for the median and 25th percentiles also exceed one for a portion of the modeling period at all locations.

Table 5-21: The field-based total PCB 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 decreased slightly at RM 152 and increased slightly at RMs 113, 90, and 50. All toxicity quotients exceed one (and sometimes ten) at all river miles for the duration of the modeling period at the 25th percentile, median concentration, and 95th percentile. Overall conclusions have not changed.

Tables 5-22 and 5-23: TRVs on a TEQ basis for largemouth bass have not changed. Revised risk estimates show that predicted toxicity quotients exceed one on a NOAEL basis at all river miles for the duration of the modeling period using the 95th percentile concentration. On a LOAEL basis, toxicity quotients also slightly increased at all locations. Toxicity quotients exceed one for a greater proportion of the modeling time frame than in the ERA Addendum.

Table 5-24: This table has not changed.

Table 5-25: The total PCB dietary dose TRV for the tree swallow has not changed. Overall, there are slight decreases in predicted toxicity quotients, and all toxicity quotients remain below one for the duration of the modeling period at all river miles. Conclusions have not changed.

Table 5-26: Total PCB egg concentration TRV for the tree swallow have not changed. Overall, there are very slight decreases in predicted toxicity quotients, and all toxicity quotients remain below one for the duration of the modeling period at all river miles.

Tables 5-27 and 5-28: TEQ-based TRVs for the tree swallow have not changed. The toxicity quotients in these tables have decreased slightly.

Table 5-29: The dietary dose TRVs for the mallard have changed. The laboratory-based body burden total PCB 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 and Heinz (1980). Predicted toxicity quotients based on dietary dose have decreased slightly and do not exceed one for any location, concentration, or time period. Previously, calculated toxicity quotients exceeded one for a portion of the modeling period at all locations.

Table 5-30: The mallard egg-based TRVs have not changed. Predicted toxicity quotients have decreased slightly during the first part of the modeling period, but the conclusions have not changed. Revised TQs exceed one on a NOAEL and LOAEL basis at RMs 152 and 113 for the duration of the modeling period (1993-2018). NOAELs are exceeded at all RMs for the duration of the modeling period.

Tables 5-31 and 5-32: Mallard TEQ-based TRVs have not changed. Toxicity quotients have decreased slightly during the first part of the modeling period and increased slightly during the later period. All TQs still exceed one at all locations for the duration of the modeling period.

Tables 5-33 and 5-34: Dietary dose TRVs did not change for the belted kingfisher and great blue heron. Predicted concentrations of prey (spottail shiner) did increase at all locations, resulting in an increase of the calculated toxicity quotients (generally less than a factor of two).

Table 5-35: Dietary dose TRVs did not change for the bald eagle. Predicted concentrations of prey (largemouth bass) generally increased at RMs 152 and 113 and decreased at RM 90 and 50. Revised toxicity quotients reflect these changes, decreasing slightly up river and increasing slightly down river.

Tables 5-36 and 5-37: Egg concentration TRVs did not change for the belted kingfisher and great blue heron. Predicted concentrations of prey (spottail shiner) did increase at all locations, resulting in an increase of the calculated toxicity quotients (generally less than a factor of two). Calculated TQs remain well above one.

Table 5-38: 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. Conclusions have not changed although predicted toxicity quotients have decreased by about a factor of two at RMs 152 and 113 and increased by less than a factor of two at RMs 90 and 50 because of changes in prey concentration.

Tables 5-39: TEQ-based TRVs for the kingfisher have not changed. Dietary doses have increased by roughly a factor of two, but the conclusions of risk remain unchanged.

Tables 5-40: TEQ-based TRVs for the great blue heron have not changed. Dietary doses have increased by roughly a factor of three. All LOAEL-based toxicity quotients exceed one (with the exception of the LOAEL at RM 50 in 2017 and 2018), in contrast to the earlier numbers where the LOAEL-based TQs only exceeded one for a portion of the modeling period.

Table 5-41: TEQ-based TRVs for the bald eagle have not changed. Conclusions have not changed although predicted toxicity quotients have decreased at RMs 152 and 113 and increased by more than a factor of two at RMs 90 and 50 because of changes in prey concentration. All NOAEL and LOAEL-based toxicity quotients exceed one for the duration of the modeling period.

Tables 5-42 and 5-43: TEQ-based egg concentration TRVs for the belted kingfisher and great blue heron have not changed. Dietary doses have increased (within a factor of two) resulting in increases in TQs. All NOAEL and LOAEL-based toxicity quotients exceed one by up to three orders of magnitude. Conclusions do not change from the ERA Addendum.

Table 5-44: TEQ-based egg concentration TRVs for the bald eagle have 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. TQs have decreased at RMs 152 and 113 and increased at RMs 90 and 50 owing to changes in prey concentration. Conclusions have not changed and TQs exceed one by up to four orders of magnitude at all locations.

Tables 5-45 and 5-46: TRVs for the little brown bat have not changed. TQs reflect changes in benthic invertebrate concentrations, and are slightly lower during the first portion of the modeling period and slightly higher at the end of the modeling period. Conclusions are unchanged.

Tables 5-47 and 5-48: TRVs for the raccoon have not changed. TQs have decreased slightly during the first portion of the modeling period and increased slightly in later years. Conclusions remain unchanged.

Table 5-49: TRVs for the mink have not changed. TQs have increased throughout the modeling period because of increases in prey (forage fish) concentrations. Conclusions remain unchanged.

Table 5-50: TRVs for the river otter have not changed. TQs have decreased at RMs 152 and 113 and increased at RMs 90 and 50 throughout the modeling period because of increases in prey (piscivorous fish) concentrations. TQs now exceed one for both the NOAEL and LOAEL at all locations for the duration of the modeling period.

Tables 5-51: The LOAEL-based comparisons for the mink have been removed since it is not appropriate to develop a LOAEL from a field-based study. Consequently, only NOAEL-based comparisons are provided on a TEQ basis. TQs for all NOAEL comparisons have increased because of increases in prey (forage fish).

Table 5-52: The LOAEL comparisons on a TEQ basis for the river otter have been removed since it is not appropriate to develop a LOAEL from a field-based study. Consequently, only NOAEL-based comparisons are provided. TQs have decreased at RMs 152 and 113 and increased at RMs 90 and 50 throughout the modeling period because of increases in prey (piscivorous fish) concentrations. TQs now exceed one by up to four orders of magnitude at all locations.

3.1 Comparison/Discussion

Several revisions were made to the HUDTOX model (input into the Farley model), FISHRAND model, Farley model, and toxicity reference values that required recalculation of risks to receptors evaluated in the ERA Addendum. None of the changes resulted in any significant changes to the conclusions reached in the ERA Addendum for future risks in the Lower Hudson River. Years for which predicted toxicity quotients fall above or below one, have changed slightly (in both directions) based on the recalculated risks, as have the toxicity quotients.

The major findings of the ERA Addendum 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:

- Fish in the Lower Hudson River are at risk from future exposure to PCBs. Omnivorous and piscivorous fish (*i.e.*, which are higher on the food chain) are at greater risk than forage fish. PCBs may adversely affect fish survival, growth, and reproduction.

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

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

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

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

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

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

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

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

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TABLE 3-5: SUMMARY OF TRI+ WHOLE WATER CONCENTRATIONS FROM THE FARLEY 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	152 Whole Water Conc mg/l	113 Whole Water Conc mg/l	90 Whole Water Conc mg/l	50 Whole Water Conc mg/l	152 Whole Water Conc mg/l	113 Whole Water Conc mg/l	90 Whole Water Conc mg/l	50 Whole Water Conc mg/l	152 Whole Water Conc mg/l	113 Whole Water Conc mg/l	90 Whole Water Conc mg/l	50 Whole Water Conc mg/l	152 Whole Water Conc mg/l	113 Whole Water Conc mg/l	90 Whole Water Conc mg/l	50 Whole Water Conc mg/l	152 Whole Water Conc mg/l	113 Whole Water Conc mg/l	90 Whole Water Conc mg/l	50 Whole Water Conc mg/l	152 Whole Water Conc mg/l	113 Whole Water Conc mg/l	90 Whole Water Conc mg/l	50 Whole Water Conc mg/l
1993	3.4E-05	2.4E-05	1.9E-05	1.5E-05					2.9E-08	2.0E-08	1.6E-08	1.3E-08					2.2E-08	1.6E-08	1.2E-08	9.8E-09				
1994	3.7E-05	2.2E-05	1.6E-05	1.3E-05					3.2E-08	1.9E-08	1.4E-08	1.1E-08					2.4E-08	1.4E-08	1.1E-08	8.6E-09				
1995	1.6E-05	1.4E-05	1.3E-05	1.1E-05					1.4E-08	1.2E-08	1.1E-08	9.4E-09					1.1E-08	9.2E-09	8.4E-09	7.2E-09				
1996	4.9E-05	2.4E-05	1.5E-05	1.1E-05					4.2E-08	2.1E-08	1.3E-08	9.2E-09					3.2E-08	1.6E-08	9.6E-09	7.1E-09				
1997	3.0E-05	1.8E-05	1.3E-05	9.9E-06					2.5E-08	1.6E-08	1.1E-08	8.4E-09					1.9E-08	1.2E-08	8.5E-09	6.5E-09				
1998	1.9E-05	1.4E-05	1.1E-05	8.6E-06					1.6E-08	1.1E-08	9.2E-09	7.3E-09					1.2E-08	8.8E-09	7.1E-09	5.6E-09				
1999	1.6E-05	1.1E-05	9.2E-06	7.5E-06					1.4E-08	9.5E-09	7.8E-09	6.4E-09					1.1E-08	7.3E-09	6.0E-09	4.9E-09				
2000	2.6E-05	1.4E-05	9.3E-06	7.1E-06					2.2E-08	1.2E-08	7.9E-09	6.0E-09					1.7E-08	9.2E-09	6.1E-09	4.6E-09				
2001	3.0E-05	1.6E-05	9.5E-06	6.8E-06					2.5E-08	1.3E-08	8.1E-09	5.8E-09					1.9E-08	1.0E-08	6.2E-09	4.5E-09				
2002	1.5E-05	1.0E-05	8.0E-06	6.1E-06					1.3E-08	8.9E-09	6.8E-09	5.2E-09					9.8E-09	6.8E-09	5.2E-09	4.0E-09				
2003	1.7E-05	1.1E-05	7.7E-06	5.8E-06					1.4E-08	9.0E-09	6.5E-09	4.9E-09					1.1E-08	6.9E-09	5.0E-09	3.8E-09				
2004	1.0E-05	7.3E-06	6.1E-06	4.9E-06					8.5E-09	6.2E-09	5.2E-09	4.2E-09					6.5E-09	4.8E-09	4.0E-09	3.2E-09				
2005	1.5E-05	8.0E-06	5.7E-06	4.5E-06					1.3E-08	6.8E-09	4.8E-09	3.8E-09					9.7E-09	5.2E-09	3.7E-09	2.9E-09				
2006	1.9E-05	9.4E-06	5.9E-06	4.3E-06					1.6E-08	8.0E-09	5.0E-09	3.7E-09					1.2E-08	6.2E-09	3.9E-09	2.8E-09				
2007	1.8E-05	9.4E-06	5.8E-06	4.1E-06					1.5E-08	8.0E-09	4.9E-09	3.5E-09					1.2E-08	6.2E-09	3.8E-09	2.7E-09				
2008	7.5E-06	5.7E-06	4.6E-06	3.6E-06					6.4E-09	4.8E-09	3.9E-09	3.1E-09					4.9E-09	3.7E-09	3.0E-09	2.3E-09				
2009	7.8E-06	5.2E-06	4.1E-06	3.3E-06					6.6E-09	4.5E-09	3.5E-09	2.8E-09					5.1E-09	3.4E-09	2.7E-09	2.1E-09				
2010	1.5E-05	7.6E-06	4.6E-06	3.3E-06					1.3E-08	6.5E-09	3.9E-09	2.8E-09					9.9E-09	5.0E-09	3.0E-09	2.2E-09				
2011	1.4E-05	7.6E-06	4.7E-06	3.3E-06					1.2E-08	6.4E-09	4.0E-09	2.8E-09					8.9E-09	4.9E-09	3.1E-09	2.1E-09				
2012	9.0E-06	6.0E-06	4.3E-06	3.1E-06					7.6E-09	5.1E-09	3.7E-09	2.7E-09					5.9E-09	3.9E-09	2.8E-09	2.0E-09				
2013	1.3E-05	7.4E-06	4.6E-06	3.2E-06					1.1E-08	6.3E-09	3.9E-09	2.7E-09					8.7E-09	4.8E-09	3.0E-09	2.1E-09				
2014	1.0E-05	6.2E-06	4.2E-06	3.0E-06					8.7E-09	5.3E-09	3.6E-09	2.6E-09					6.7E-09	4.1E-09	2.8E-09	2.0E-09				
2015	9.7E-06	5.8E-06	4.0E-06	2.9E-06					8.3E-09	4.9E-09	3.4E-09	2.4E-09					6.3E-09	3.8E-09	2.6E-09	1.9E-09				
2016	4.7E-06	3.8E-06	3.2E-06	2.6E-06					4.0E-09	3.3E-09	2.8E-09	2.2E-09					3.1E-09	2.5E-09	2.1E-09	1.7E-09				
2017	4.6E-06	3.3E-06	2.9E-06	2.3E-06					3.9E-09	2.8E-09	2.4E-09	2.0E-09					3.0E-09	2.2E-09	1.9E-09	1.5E-09				
2018	5.2E-06	3.7E-06	2.9E-06	2.3E-06					4.4E-09	3.1E-09	2.5E-09	2.0E-09					3.4E-09	2.4E-09	1.9E-09	1.5E-09				

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

REVISED

Year	Tri+ Average PCB Results				Tri+ 95% UCL Results			
	152 Total Sed Conc mg/kg	113 Total Sed Conc mg/kg	90 Total Sed Conc mg/kg	50 Total Sed Conc mg/kg	152 Total Sed Conc mg/kg	113 Total Sed Conc mg/kg	90 Total Sed Conc mg/kg	50 Total Sed Conc mg/kg
1993	1.106	0.843	0.664	0.484				
1994	1.015	0.805	0.634	0.461				
1995	0.929	0.758	0.603	0.440				
1996	0.957	0.740	0.580	0.422				
1997	0.942	0.726	0.563	0.408				
1998	0.875	0.695	0.542	0.394				
1999	0.820	0.661	0.520	0.380				
2000	0.817	0.643	0.502	0.367				
2001	0.838	0.640	0.490	0.356				
2002	0.806	0.630	0.482	0.349				
2003	0.771	0.611	0.469	0.340				
2004	0.725	0.585	0.454	0.331				
2005	0.705	0.564	0.439	0.321				
2006	0.715	0.557	0.428	0.313				
2007	0.706	0.549	0.419	0.306				
2008	0.676	0.536	0.410	0.299				
2009	0.646	0.518	0.400	0.292				
2010	0.654	0.512	0.392	0.286				
2011	0.657	0.509	0.386	0.281				
2012	0.643	0.503	0.381	0.276				
2013	0.638	0.497	0.376	0.272				
2014	0.621	0.488	0.370	0.267				
2015	0.603	0.477	0.363	0.263				
2016	0.578	0.463	0.355	0.258				
2017	0.560	0.451	0.347	0.254				
2018	0.556	0.443	0.340	0.248				

Year	Average Avian TEF				95% Avian TEF			
	152 Total Sed Conc mg/kg	113 Total Sed Conc mg/kg	90 Total Sed Conc mg/kg	50 Total Sed Conc mg/kg	152 Total Sed Conc mg/kg	113 Total Sed Conc mg/kg	90 Total Sed Conc mg/kg	50 Total Sed Conc mg/kg
1993	9.4E-04	7.2E-04	5.6E-04	4.1E-04				
1994	8.6E-04	6.8E-04	5.4E-04	3.9E-04				
1995	7.9E-04	6.4E-04	5.1E-04	3.7E-04				
1996	8.1E-04	6.3E-04	4.9E-04	3.6E-04				
1997	8.0E-04	6.2E-04	4.8E-04	3.5E-04				
1998	7.4E-04	5.9E-04	4.6E-04	3.3E-04				
1999	7.0E-04	5.6E-04	4.4E-04	3.2E-04				
2000	6.9E-04	5.5E-04	4.3E-04	3.1E-04				
2001	7.1E-04	5.4E-04	4.2E-04	3.0E-04				
2002	6.9E-04	5.4E-04	4.1E-04	3.0E-04				
2003	6.6E-04	5.2E-04	4.0E-04	2.9E-04				
2004	6.2E-04	5.0E-04	3.9E-04	2.8E-04				
2005	6.0E-04	4.8E-04	3.7E-04	2.7E-04				
2006	6.1E-04	4.7E-04	3.6E-04	2.7E-04				
2007	6.0E-04	4.7E-04	3.6E-04	2.6E-04				
2008	5.7E-04	4.6E-04	3.5E-04	2.5E-04				
2009	5.5E-04	4.4E-04	3.4E-04	2.5E-04				
2010	5.6E-04	4.4E-04	3.3E-04	2.4E-04				
2011	5.6E-04	4.3E-04	3.3E-04	2.4E-04				
2012	5.5E-04	4.3E-04	3.2E-04	2.3E-04				
2013	5.4E-04	4.2E-04	3.2E-04	2.3E-04				
2014	5.3E-04	4.1E-04	3.1E-04	2.3E-04				
2015	5.1E-04	4.1E-04	3.1E-04	2.2E-04				
2016	4.9E-04	3.9E-04	3.0E-04	2.2E-04				
2017	4.8E-04	3.8E-04	3.0E-04	2.2E-04				
2018	4.7E-04	3.8E-04	2.9E-04	2.1E-04				

Year	Average Mammalian TEF				95% UCL Mammalian TEF			
	152 Total Sed Conc mg/kg	113 Total Sed Conc mg/kg	90 Total Sed Conc mg/kg	50 Total Sed Conc mg/kg	152 Total Sed Conc mg/kg	113 Total Sed Conc mg/kg	90 Total Sed Conc mg/kg	50 Total Sed Conc mg/kg
1993	7.2E-04	5.5E-04	4.3E-04	3.2E-04				
1994	6.6E-04	5.3E-04	4.1E-04	3.0E-04				
1995	6.1E-04	4.9E-04	3.9E-04	2.9E-04				
1996	6.2E-04	4.8E-04	3.8E-04	2.8E-04				
1997	6.2E-04	4.7E-04	3.7E-04	2.7E-04				
1998	5.7E-04	4.5E-04	3.5E-04	2.6E-04				
1999	5.4E-04	4.3E-04	3.4E-04	2.5E-04				
2000	5.3E-04	4.2E-04	3.3E-04	2.4E-04				
2001	5.5E-04	4.2E-04	3.2E-04	2.3E-04				
2002	5.3E-04	4.1E-04	3.1E-04	2.3E-04				
2003	5.0E-04	4.0E-04	3.1E-04	2.2E-04				
2004	4.7E-04	3.8E-04	3.0E-04	2.2E-04				
2005	4.6E-04	3.7E-04	2.9E-04	2.1E-04				
2006	4.7E-04	3.6E-04	2.8E-04	2.0E-04				
2007	4.6E-04	3.6E-04	2.7E-04	2.0E-04				
2008	4.4E-04	3.5E-04	2.7E-04	2.0E-04				
2009	4.2E-04	3.4E-04	2.6E-04	1.9E-04				
2010	4.3E-04	3.3E-04	2.6E-04	1.9E-04				
2011	4.3E-04	3.3E-04	2.5E-04	1.8E-04				
2012	4.2E-04	3.3E-04	2.5E-04	1.8E-04				
2013	4.2E-04	3.2E-04	2.5E-04	1.8E-04				
2014	4.1E-04	3.2E-04	2.4E-04	1.7E-04				
2015	3.9E-04	3.1E-04	2.4E-04	1.7E-04				
2016	3.8E-04	3.0E-04	2.3E-04	1.7E-04				
2017	3.7E-04	2.9E-04	2.3E-04	1.7E-04				
2018	3.6E-04	2.9E-04	2.2E-04	1.6E-04				

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

	Tri+ Average PCB Results			
Year	152 Total Sed Conc mg/kg	113 Total Sed Conc mg/kg	90 Total Sed Conc mg/kg	50 Total Sed Conc mg/kg
1993	44.25	33.73	26.54	19.34
1994	40.62	32.20	25.35	18.42
1995	37.17	30.31	24.13	17.60
1996	38.27	29.59	23.19	16.90
1997	37.70	29.06	22.51	16.31
1998	34.99	27.81	21.70	15.75
1999	32.79	26.45	20.80	15.19
2000	32.66	25.72	20.07	14.68
2001	33.52	25.60	19.61	14.26
2002	32.25	25.19	19.28	13.96
2003	30.86	24.42	18.77	13.60
2004	29.02	23.40	18.17	13.22
2005	28.22	22.56	17.56	12.85
2006	28.59	22.27	17.13	12.52
2007	28.25	21.97	16.78	12.23
2008	27.03	21.42	16.41	11.96
2009	25.85	20.73	15.99	11.69
2010	26.16	20.47	15.67	11.44
2011	26.29	20.37	15.44	11.23
2012	25.72	20.12	15.23	11.04
2013	25.51	19.87	15.02	10.86
2014	24.82	19.51	14.79	10.69
2015	24.12	19.09	14.53	10.51
2016	23.11	18.52	14.21	10.32
2017	22.41	18.03	13.89	10.14
2018	22.24	17.71	13.59	9.94

average TOC from Farley model 2.5%

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

	Tri+ Average PCB Results				Tri+ 95% UCL Results				Average Avian TEF				95% Avian TEF				Average Mammalian TEF				95% UCL Mammalian TEF			
Year	152 Total Benthic Conc mg/kg	113 Total Benthic Conc mg/kg	90 Total Benthic Conc mg/kg	50 Total Benthic Conc mg/kg	152 Total Benthic Conc mg/kg	113 Total Benthic Conc mg/kg	90 Total Benthic Conc mg/kg	50 Total Benthic Conc mg/kg	152 Total Benthic Conc mg/kg	113 Total Benthic Conc mg/kg	90 Total Benthic Conc mg/kg	50 Total Benthic Conc mg/kg	152 Total Benthic Conc mg/kg	113 Total Benthic Conc mg/kg	90 Total Benthic Conc mg/kg	50 Total Benthic Conc mg/kg	152 Total Benthic Conc mg/kg	113 Total Benthic Conc mg/kg	90 Total Benthic Conc mg/kg	50 Total Benthic Conc mg/kg	152 Total Benthic Conc mg/kg	113 Total Benthic Conc mg/kg	90 Total Benthic Conc mg/kg	50 Total Benthic Conc mg/kg
1993	1 530	1 209	0 967	0 713	1 611	1 275	1 018	0 751	2.1E-04	1 7E-04	1.3E-04	9.9E-05	2 2E-04	1 8E-04	1.4E-04	1 0E-04	1 7E-04	1 3E-04	1 0E-04	7.7E-05	1 7E-04	1 4E-04	1 1E-04	8 1E-05
1994	1 410	1 146	0 923	0 686	1.488	1.209	0 973	0 723	2.0E-04	1 6E-04	1.3E-04	9 5E-05	2 1E-04	1 7E-04	1 3E-04	1 0E-04	1 5E-04	1.2E-04	1 0E-04	7.4E-05	1 6E-04	1 3E-04	1 0E-04	7 8E-05
1995	1 354	1 094	0 883	0 654	1 430	1 154	0 932	0.690	1 9E-04	1 5E-04	1 2E-04	9 1E-05	2 0E-04	1.6E-04	1 3E-04	9 6E-05	1 5E-04	1.2E-04	9.5E-05	7 1E-05	1 5E-04	1.2E-04	1 0E-04	7 4E-05
1996	1 357	1 056	0 847	0 630	1 432	1 116	0 894	0 665	1 9E-04	1.5E-04	1 2E-04	8.7E-05	2 0E-04	1 5E-04	1 2E-04	9.2E-05	1 5E-04	1 1E-04	9 1E-05	6 8E-05	1.5E-04	1 2E-04	9 6E-05	7 2E-05
1997	1 289	1 028	0 816	0.600	1 363	1 087	0 862	0 634	1 8E-04	1.4E-04	1 1E-04	8.3E-05	1 9E-04	1 5E-04	1 2E-04	8.8E-05	1 4E-04	1 1E-04	8 8E-05	6 5E-05	1.5E-04	1 2E-04	9 3E-05	6 8E-05
1998	1 217	0 982	0 786	0.582	1 292	1 040	0 832	0.616	1 7E-04	1.4E-04	1 1E-04	8.1E-05	1 8E-04	1 4E-04	1 2E-04	8 5E-05	1.3E-04	1.1E-04	8 5E-05	6 3E-05	1.4E-04	1 1E-04	9 0E-05	6 6E-05
1999	1 165	0 949	0 757	0 566	1 238	1.008	0 802	0 600	1 6E-04	1.3E-04	1 0E-04	7 8E-05	1 7E-04	1 4E-04	1 1E-04	8.3E-05	1 3E-04	1 0E-04	8 2E-05	6 1E-05	1 3E-04	1 1E-04	8 6E-05	6 5E-05
2000	1 182	0 926	0 731	0 549	1 254	0 983	0 775	0 582	1 6E-04	1.3E-04	1 0E-04	7.6E-05	1 7E-04	1.4E-04	1.1E-04	8.1E-05	1 3E-04	1 0E-04	7 9E-05	5.9E-05	1 4E-04	1.1E-04	8.4E-05	6 3E-05
2001	1 167	0 918	0 723	0 534	1 236	0 975	0 767	0 566	1.6E-04	1 3E-04	1 0E-04	7.4E-05	1.7E-04	1.4E-04	1 1E-04	7.9E-05	1 3E-04	9 9E-05	7.8E-05	5 8E-05	1 3E-04	1 1E-04	8.3E-05	6 1E-05
2002	1 123	0 889	0.702	0 516	1.193	0.946	0 745	0 549	1.6E-04	1 2E-04	9 7E-05	7 1E-05	1 7E-04	1.3E-04	1.0E-04	7 6E-05	1 2E-04	9 6E-05	7 6E-05	5 6E-05	1 3E-04	1.0E-04	8 0E-05	5 9E-05
2003	1 066	0 858	0.677	0 505	1 138	0 914	0 720	0 537	1.5E-04	1.2E-04	9 4E-05	7 0E-05	1 6E-04	1.3E-04	1 0E-04	7.4E-05	1.2E-04	9 3E-05	7.3E-05	5 4E-05	1 2E-04	9 9E-05	7.8E-05	5 8E-05
2004	1 058	0 855	0 656	0.486	1 133	0 914	0 699	0 518	1 5E-04	1.2E-04	9 1E-05	6 7E-05	1 6E-04	1 3E-04	9 7E-05	7.2E-05	1 1E-04	9 2E-05	7 1E-05	5 2E-05	1 2E-04	9 9E-05	7.5E-05	5 6E-05
2005	1 046	0 846	0 652	0.488	1 120	0 905	0 697	0.522	1 4E-04	1 2E-04	9.0E-05	6 8E-05	1 6E-04	1 3E-04	9.7E-05	7.2E-05	1.1E-04	9 1E-05	7.0E-05	5.3E-05	1 2E-04	9 8E-05	7 5E-05	5 6E-05
2006	1 038	0 829	0 639	0.478	1 110	0.887	0.684	0.512	1 4E-04	1.1E-04	8 9E-05	6 6E-05	1 5E-04	1 2E-04	9 5E-05	7.1E-05	1 1E-04	8 9E-05	6 9E-05	5 2E-05	1 2E-04	9 6E-05	7 4E-05	5 5E-05
2007	1 021	0 818	0 631	0 470	1 093	0 876	0 676	0.504	1.4E-04	1.1E-04	8 8E-05	6 5E-05	1 5E-04	1 2E-04	9.4E-05	7 0E-05	1 1E-04	8 8E-05	6 8E-05	5 1E-05	1.2E-04	9 5E-05	7 3E-05	5 4E-05
2008	1 007	0 809	0 620	0 465	1 081	0 868	0.664	0 498	1 4E-04	1.1E-04	8 6E-05	6 4E-05	1 5E-04	1 2E-04	9 2E-05	6.9E-05	1 1E-04	8.7E-05	6.7E-05	5 0E-05	1 2E-04	9 4E-05	7.2E-05	5 4E-05
2009	0 991	0 795	0 613	0 458	1 064	0 854	0 657	0 491	1 4E-04	1.1E-04	8 5E-05	6 3E-05	1 5E-04	1 2E-04	9 1E-05	6 8E-05	1 1E-04	8 6E-05	6 6E-05	4 9E-05	1 1E-04	9 2E-05	7 1E-05	5 3E-05
2010	0 978	0 787	0 607	0 454	1 048	0 845	0 651	0 487	1 4E-04	1.1E-04	8 4E-05	6 3E-05	1 5E-04	1 2E-04	9 0E-05	6 7E-05	1 1E-04	8 5E-05	6 6E-05	4.9E-05	1.1E-04	9 1E-05	7 0E-05	5.3E-05
2011	0 969	0 775	0 597	0 445	1 037	0 832	0.640	0 477	1 3E-04	1.1E-04	8 3E-05	6 2E-05	1 4E-04	1.2E-04	8 9E-05	6 6E-05	1 0E-04	8 4E-05	6 4E-05	4 8E-05	1 1E-04	9 0E-05	6 9E-05	5 1E-05
2012	0 954	0 765	0 590	0 439	1 022	0 820	0 633	0 471	1 3E-04	1 1E-04	8 2E-05	6 1E-05	1 4E-04	1.1E-04	8 8E-05	6.5E-05	1 0E-04	8 3E-05	6 4E-05	4 7E-05	1.1E-04	8 9E-05	6 8E-05	5 1E-05
2013	0 938	0 754	0 581	0 433	1 004	0 809	0 623	0.465	1 3E-04	1.0E-04	8 0E-05	6.0E-05	1 4E-04	1.1E-04	8.6E-05	6 4E-05	1 0E-04	8.1E-05	6 3E-05	4 7E-05	1 1E-04	8 7E-05	6 7E-05	5 0E-05
2014	0 923	0 744	0 574	0 427	0.990	0 799	0 615	0.458	1 3E-04	1 0E-04	8 0E-05	5 9E-05	1 4E-04	1 1E-04	8.5E-05	6 4E-05	1 0E-04	8.0E-05	6.2E-05	4.6E-05	1 1E-04	8 6E-05	6.6E-05	4 9E-05
2015	0 912	0 732	0 564	0 421	0 979	0 786	0 605	0.452	1 3E-04	1 0E-04	7 8E-05	5.8E-05	1 4E-04	1 1E-04	8 4E-05	6 3E-05	9 8E-05	7 9E-05	6.1E-05	4 5E-05	1 1E-04	8 5E-05	6.5E-05	4 9E-05
2016	0 917	0 721	0 554	0 415	0 991	0 775	0 595	0.445	1 3E-04	1 0E-04	7 7E-05	5 7E-05	1 4E-04	1.1E-04	8 2E-05	6.2E-05	9 9E-05	7 8E-05	6.0E-05	4 5E-05	1 1E-04	8 4E-05	6.4E-05	4 8E-05
2017	0 917	0 720	0.548	0 409	0 993	0 777	0 588	0 439	1 3E-04	1.0E-04	7.6E-05	5 7E-05	1 4E-04	1 1E-04	8 2E-05	6 1E-05	9 9E-05	7 8E-05	5.9E-05	4 4E-05	1 1E-04	8 4E-05	6.3E-05	4 7E-05
2018	0 920	0 728	0 537	0 402	0 998	0 788	0 577	0 433	1 3E-04	1 0E-04	7 4E-05	5 6E-05	1 4E-04	1 1E-04	8 0E-05	6.0E-05	9.9E-05	7 9E-05	5 8E-05	4 3E-05	1 1E-04	8 5E-05	6 2E-05	4 7E-05

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

	River Mile 152			River Mile 113			River Mile 90			River Mile 50		
Year	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)
1993	1.66	2.34	5.25	1.67	2.15	4.40	1.33	1.70	3.35	1.25	1.63	3.31
1994	1.20	1.73	3.54	1.54	1.96	3.70	1.21	1.54	3.00	1.11	1.44	2.89
1995	1.11	1.54	3.22	1.21	1.56	3.19	1.05	1.34	2.70	0.98	1.28	2.55
1996	1.35	1.99	4.17	1.34	1.70	3.32	0.98	1.25	2.42	0.90	1.15	2.26
1997	1.10	1.61	3.45	1.18	1.52	3.11	0.91	1.16	2.24	0.82	1.05	2.04
1998	0.91	1.24	2.63	1.01	1.31	2.62	0.83	1.07	2.04	0.75	0.96	1.87
1999	0.81	1.17	2.37	0.90	1.16	2.18	0.74	0.95	1.78	0.68	0.87	1.68
2000	0.77	1.06	2.15	0.90	1.15	2.05	0.69	0.87	1.59	0.62	0.79	1.49
2001	0.89	1.23	2.47	0.89	1.15	2.16	0.66	0.85	1.50	0.58	0.74	1.36
2002	0.82	1.09	2.29	0.84	1.08	2.05	0.64	0.81	1.50	0.55	0.71	1.30
2003	0.64	0.95	1.94	0.80	1.02	1.82	0.61	0.77	1.40	0.52	0.67	1.23
2004	0.54	0.76	1.49	0.68	0.87	1.59	0.55	0.70	1.29	0.49	0.62	1.16
2005	0.57	0.77	1.48	0.65	0.83	1.45	0.51	0.65	1.18	0.45	0.58	1.06
2006	0.65	0.88	1.72	0.66	0.85	1.50	0.49	0.63	1.11	0.43	0.55	0.98
2007	0.53	0.76	1.47	0.64	0.82	1.44	0.48	0.62	1.06	0.41	0.52	0.93
2008	0.48	0.69	1.36	0.58	0.76	1.40	0.45	0.58	1.05	0.39	0.50	0.89
2009	0.41	0.58	1.17	0.52	0.68	1.20	0.42	0.54	0.97	0.36	0.47	0.83
2010	0.54	0.72	1.40	0.54	0.70	1.24	0.42	0.54	0.91	0.35	0.46	0.80
2011	0.50	0.72	1.46	0.58	0.74	1.30	0.42	0.54	0.93	0.35	0.45	0.78
2012	0.49	0.71	1.40	0.56	0.73	1.30	0.41	0.54	0.92	0.34	0.45	0.77
2013	0.53	0.76	1.51	0.57	0.73	1.30	0.41	0.53	0.92	0.34	0.44	0.76
2014	0.45	0.67	1.36	0.55	0.71	1.27	0.40	0.52	0.91	0.33	0.43	0.75
2015	0.42	0.60	1.18	0.52	0.67	1.17	0.39	0.51	0.88	0.32	0.42	0.73
2016	0.38	0.52	1.05	0.47	0.61	1.08	0.36	0.48	0.85	0.31	0.40	0.71
2017	0.37	0.52	1.05	0.44	0.57	1.02	0.34	0.45	0.80	0.30	0.39	0.68
2018	0.38	0.54	1.05	0.43	0.57	1.00	0.33	0.44	0.76	0.29	0.37	0.65

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

	River Mile 152			River Mile 113			River Mile 90			River Mile 50		
Year	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)
1993	3.24	4.09	7.73	2.04	2.58	5.08	1.58	1.99	3.74	1.56	2.02	3.79
1994	2.47	3.10	5.16	1.79	2.25	4.22	1.41	1.77	3.31	1.36	1.76	3.29
1995	2.07	2.67	4.58	1.38	1.72	3.59	1.21	1.50	2.93	1.20	1.53	2.86
1996	2.80	3.54	5.91	1.55	1.96	3.59	1.12	1.41	2.64	1.07	1.36	2.54
1997	2.18	2.73	5.11	1.37	1.69	3.36	1.04	1.30	2.41	0.97	1.23	2.27
1998	1.49	1.89	3.80	1.13	1.42	2.90	0.92	1.17	2.21	0.88	1.11	2.05
1999	1.29	1.64	3.14	0.96	1.22	2.39	0.81	1.03	1.91	0.79	1.00	1.83
2000	1.48	1.84	2.99	0.99	1.24	2.08	0.75	0.94	1.64	0.72	0.89	1.63
2001	1.57	1.99	3.41	1.01	1.27	2.33	0.72	0.90	1.58	0.66	0.83	1.50
2002	1.23	1.57	3.13	0.90	1.14	2.22	0.69	0.87	1.51	0.63	0.79	1.41
2003	1.27	1.61	2.66	0.85	1.08	1.91	0.65	0.81	1.41	0.59	0.74	1.31
2004	0.88	1.10	1.92	0.70	0.88	1.60	0.59	0.73	1.30	0.55	0.68	1.20
2005	0.98	1.19	1.93	0.67	0.84	1.41	0.54	0.66	1.15	0.50	0.62	1.08
2006	1.18	1.46	2.31	0.71	0.87	1.53	0.51	0.64	1.07	0.46	0.57	1.00
2007	0.94	1.15	1.81	0.69	0.85	1.44	0.49	0.62	1.03	0.44	0.54	0.94
2008	0.75	0.94	1.73	0.60	0.75	1.34	0.47	0.58	0.98	0.42	0.52	0.89
2009	0.75	0.95	1.46	0.52	0.65	1.11	0.42	0.53	0.90	0.39	0.48	0.81
2010	0.86	1.06	1.81	0.58	0.71	1.22	0.43	0.53	0.85	0.37	0.47	0.77
2011	1.02	1.25	1.92	0.60	0.75	1.27	0.43	0.53	0.86	0.37	0.46	0.75
2012	0.87	1.08	1.77	0.57	0.72	1.23	0.42	0.52	0.85	0.36	0.45	0.73
2013	0.97	1.23	1.96	0.59	0.73	1.28	0.42	0.53	0.86	0.36	0.45	0.73
2014	0.83	1.05	1.74	0.56	0.71	1.21	0.41	0.51	0.84	0.35	0.44	0.72
2015	0.78	0.96	1.48	0.53	0.66	1.12	0.40	0.49	0.81	0.34	0.43	0.70
2016	0.55	0.69	1.23	0.46	0.57	0.97	0.37	0.46	0.78	0.33	0.40	0.67
2017	0.53	0.67	1.17	0.42	0.53	0.90	0.35	0.42	0.72	0.31	0.38	0.64
2018	0.57	0.69	1.18	0.42	0.52	0.86	0.33	0.41	0.67	0.30	0.37	0.61

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

	River Mile 152			River Mile 113			River Mile 90			River Mile 50		
Year	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)
1993	0.98	1.35	3.21	0.72	1.00	2.32	0.57	0.79	1.80	0.54	0.75	1.78
1994	0.94	1.30	2.64	0.67	0.93	2.06	0.52	0.72	1.61	0.48	0.67	1.56
1995	0.70	0.98	2.41	0.52	0.72	1.62	0.45	0.62	1.39	0.42	0.59	1.36
1996	0.99	1.36	2.92	0.58	0.80	1.75	0.43	0.59	1.28	0.39	0.54	1.22
1997	0.72	0.99	2.33	0.51	0.70	1.58	0.39	0.54	1.16	0.36	0.49	1.10
1998	0.57	0.78	1.72	0.44	0.61	1.31	0.36	0.49	1.04	0.32	0.45	0.99
1999	0.50	0.68	1.47	0.40	0.53	1.13	0.32	0.44	0.91	0.29	0.40	0.87
2000	0.62	0.83	1.56	0.40	0.54	1.09	0.31	0.41	0.85	0.27	0.37	0.79
2001	0.61	0.82	1.67	0.40	0.54	1.11	0.30	0.39	0.82	0.26	0.35	0.74
2002	0.48	0.65	1.40	0.37	0.50	1.04	0.29	0.39	0.78	0.25	0.33	0.70
2003	0.48	0.66	1.35	0.35	0.48	0.95	0.27	0.36	0.73	0.23	0.32	0.65
2004	0.37	0.49	1.00	0.30	0.40	0.79	0.25	0.33	0.65	0.22	0.29	0.60
2005	0.41	0.54	1.06	0.29	0.39	0.76	0.23	0.31	0.60	0.20	0.27	0.55
2006	0.46	0.62	1.30	0.30	0.40	0.79	0.22	0.29	0.58	0.19	0.26	0.52
2007	0.41	0.54	1.02	0.29	0.38	0.76	0.22	0.28	0.57	0.18	0.24	0.49
2008	0.32	0.43	0.88	0.25	0.35	0.71	0.20	0.27	0.53	0.18	0.23	0.47
2009	0.30	0.41	0.86	0.23	0.31	0.62	0.18	0.25	0.49	0.16	0.21	0.43
2010	0.35	0.46	0.97	0.24	0.33	0.64	0.19	0.25	0.50	0.16	0.21	0.43
2011	0.38	0.52	1.07	0.25	0.34	0.68	0.19	0.25	0.50	0.16	0.21	0.42
2012	0.34	0.47	1.01	0.25	0.33	0.67	0.19	0.25	0.49	0.15	0.20	0.41
2013	0.37	0.50	0.99	0.25	0.34	0.66	0.19	0.25	0.49	0.15	0.20	0.41
2014	0.33	0.44	0.94	0.24	0.33	0.65	0.18	0.24	0.48	0.15	0.20	0.40
2015	0.31	0.42	0.87	0.23	0.31	0.62	0.18	0.23	0.47	0.15	0.19	0.39
2016	0.25	0.34	0.68	0.20	0.28	0.55	0.16	0.22	0.43	0.14	0.19	0.38
2017	0.24	0.33	0.65	0.19	0.26	0.51	0.16	0.21	0.41	0.13	0.18	0.36
2018	0.24	0.33	0.66	0.19	0.26	0.51	0.15	0.20	0.40	0.13	0.17	0.35

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

	River Mile 152			River Mile 113			River Mile 90			River Mile 50		
Year	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)
1993	2.19	3.20	7.03	1.40	1.96	4.34	1.09	1.55	3.41	0.92	1.30	2.94
1994	2.00	2.84	5.88	1.29	1.83	3.96	1.02	1.44	3.16	0.84	1.20	2.70
1995	1.67	2.40	5.24	1.18	1.64	3.64	0.95	1.33	2.94	0.78	1.10	2.45
1996	1.93	2.83	5.50	1.17	1.63	3.44	0.90	1.26	2.75	0.73	1.03	2.27
1997	1.74	2.48	5.21	1.12	1.54	3.36	0.86	1.20	2.60	0.69	0.97	2.13
1998	1.53	2.17	4.60	1.04	1.45	3.13	0.81	1.14	2.45	0.65	0.91	2.00
1999	1.41	2.01	4.23	0.96	1.34	2.87	0.76	1.07	2.31	0.61	0.86	1.87
2000	1.48	2.09	4.17	0.94	1.33	2.71	0.73	1.03	2.17	0.58	0.82	1.76
2001	1.47	2.10	4.42	0.95	1.32	2.72	0.71	1.01	2.09	0.56	0.79	1.67
2002	1.36	1.94	4.11	0.92	1.27	2.71	0.70	0.98	2.05	0.55	0.77	1.62
2003	1.29	1.84	3.83	0.87	1.22	2.56	0.67	0.95	1.98	0.53	0.74	1.58
2004	1.18	1.70	3.52	0.82	1.15	2.43	0.64	0.90	1.90	0.51	0.72	1.52
2005	1.20	1.71	3.44	0.79	1.12	2.32	0.62	0.87	1.82	0.49	0.69	1.45
2006	1.25	1.76	3.47	0.79	1.13	2.29	0.60	0.85	1.77	0.48	0.67	1.39
2007	1.19	1.70	3.42	0.78	1.10	2.26	0.59	0.84	1.73	0.46	0.65	1.35
2008	1.11	1.58	3.27	0.76	1.06	2.21	0.58	0.81	1.70	0.45	0.64	1.32
2009	1.06	1.53	3.12	0.72	1.03	2.13	0.56	0.79	1.65	0.44	0.62	1.29
2010	1.10	1.57	3.18	0.73	1.02	2.09	0.55	0.79	1.62	0.43	0.61	1.26
2011	1.11	1.57	3.22	0.73	1.02	2.11	0.55	0.78	1.60	0.42	0.60	1.24
2012	1.06	1.50	3.09	0.71	1.00	2.08	0.54	0.77	1.58	0.42	0.59	1.22
2013	1.07	1.54	3.13	0.71	1.00	2.06	0.53	0.76	1.56	0.41	0.58	1.20
2014	1.03	1.48	3.02	0.69	0.98	2.02	0.53	0.75	1.54	0.41	0.57	1.19
2015	1.01	1.45	2.94	0.68	0.96	1.98	0.52	0.73	1.51	0.40	0.56	1.17
2016	0.96	1.37	2.87	0.66	0.93	1.93	0.50	0.72	1.48	0.39	0.55	1.15
2017	0.94	1.34	2.77	0.64	0.91	1.89	0.49	0.70	1.45	0.38	0.54	1.12
2018	0.93	1.35	2.72	0.64	0.91	1.86	0.49	0.69	1.42	0.38	0.53	1.10

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

	River Mile 152			River Mile 113			River Mile 90			River Mile 50		
Year	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)
1993	3.11	3.83	6.47	2.39	2.94	4.90	1.90	2.33	3.88	1.49	1.82	3.06
1994	2.84	3.51	5.69	2.25	2.76	4.60	1.80	2.21	3.66	1.40	1.72	2.86
1995	2.58	3.19	5.43	2.09	2.57	4.32	1.70	2.08	3.46	1.32	1.62	2.69
1996	2.73	3.43	5.57	2.06	2.55	4.18	1.62	2.00	3.30	1.25	1.54	2.55
1997	2.57	3.16	5.34	1.98	2.44	4.06	1.56	1.92	3.18	1.20	1.47	2.43
1998	2.38	2.94	4.99	1.88	2.32	3.90	1.49	1.84	3.04	1.15	1.41	2.33
1999	2.23	2.78	4.62	1.79	2.23	3.67	1.43	1.77	2.90	1.10	1.35	2.24
2000	2.26	2.82	4.51	1.76	2.20	3.57	1.38	1.72	2.80	1.06	1.31	2.15
2001	2.29	2.84	4.60	1.75	2.18	3.53	1.35	1.68	2.74	1.03	1.28	2.09
2002	2.18	2.68	4.50	1.70	2.12	3.49	1.32	1.65	2.69	1.00	1.25	2.04
2003	2.04	2.56	4.24	1.65	2.05	3.38	1.28	1.60	2.62	0.98	1.21	1.99
2004	1.95	2.47	4.07	1.58	1.99	3.27	1.24	1.55	2.56	0.95	1.18	1.94
2005	1.95	2.46	3.97	1.55	1.95	3.19	1.21	1.51	2.48	0.92	1.15	1.89
2006	1.97	2.46	3.99	1.54	1.93	3.15	1.18	1.48	2.43	0.90	1.12	1.85
2007	1.93	2.40	3.90	1.52	1.91	3.10	1.16	1.46	2.38	0.88	1.10	1.82
2008	1.84	2.33	3.85	1.48	1.87	3.07	1.14	1.43	2.35	0.87	1.09	1.79
2009	1.80	2.28	3.73	1.44	1.83	2.99	1.12	1.41	2.31	0.85	1.07	1.76
2010	1.84	2.27	3.67	1.44	1.81	2.94	1.10	1.39	2.26	0.84	1.06	1.72
2011	1.81	2.24	3.68	1.42	1.79	2.92	1.09	1.37	2.24	0.82	1.04	1.70
2012	1.76	2.20	3.61	1.40	1.76	2.88	1.07	1.35	2.21	0.81	1.03	1.67
2013	1.76	2.18	3.59	1.38	1.74	2.85	1.06	1.34	2.18	0.80	1.01	1.65
2014	1.70	2.15	3.52	1.36	1.72	2.81	1.04	1.32	2.15	0.79	1.00	1.63
2015	1.67	2.13	3.45	1.34	1.70	2.75	1.03	1.30	2.11	0.78	0.98	1.60
2016	1.63	2.09	3.41	1.31	1.67	2.72	1.01	1.28	2.08	0.76	0.97	1.58
2017	1.62	2.06	3.38	1.29	1.65	2.68	0.99	1.26	2.05	0.75	0.95	1.55
2018	1.60	2.05	3.31	1.28	1.64	2.65	0.98	1.24	2.01	0.74	0.94	1.53

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

	River Mile 152			River Mile 113			River Mile 90			River Mile 50		
Year	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)
1993	9.23	10.61	15.61	7.26	8.25	12.02	5.59	6.34	9.11	5.45	6.20	9.07
1994	6.71	7.54	10.49	6.23	7.09	10.38	5.04	5.72	8.28	4.81	5.46	7.96
1995	5.49	6.26	9.30	5.12	5.86	8.86	4.42	5.04	7.38	4.24	4.82	7.03
1996	7.39	8.22	11.08	5.36	6.04	8.56	4.00	4.52	6.53	3.78	4.30	6.21
1997	6.08	6.97	10.25	4.94	5.61	8.22	3.74	4.23	6.09	3.44	3.90	5.64
1998	4.94	4.89	5.60	4.24	4.84	7.10	3.40	3.85	5.60	3.13	3.55	5.12
1999	5.63	4.16	4.73	3.69	4.20	6.13	3.03	3.43	4.98	2.82	3.20	4.62
2000	5.98	4.07	4.58	3.41	3.88	5.64	2.72	3.07	4.46	2.55	2.89	4.19
2001	6.20	4.54	5.11	3.58	4.04	5.84	2.57	2.93	4.23	2.36	2.66	3.84
2002	6.51	4.10	4.67	3.43	3.91	5.64	2.53	2.85	4.11	2.25	2.52	3.62
2003	4.72	3.68	4.13	3.12	3.52	5.12	2.39	2.69	3.89	2.12	2.39	3.43
2004	4.23	2.76	3.14	2.72	3.08	4.54	2.21	2.49	3.63	1.97	2.24	3.21
2005	4.63	2.77	3.15	2.44	2.78	4.06	2.00	2.25	3.28	1.82	2.06	2.95
2006	5.06	3.23	3.58	2.54	2.87	4.13	1.87	2.12	3.10	1.70	1.91	2.75
2007	4.84	2.78	3.14	2.48	2.80	4.06	1.81	2.06	3.00	1.61	1.82	2.61
2008	3.99	2.51	2.87	2.34	2.64	3.86	1.76	1.99	2.88	1.54	1.73	2.50
2009	3.23	2.28	2.58	2.06	2.33	3.42	1.65	1.86	2.70	1.43	1.62	2.36
2010	3.87	2.44	2.72	2.05	2.33	3.33	1.56	1.77	2.59	1.37	1.55	2.25
2011	4.33	2.85	3.20	2.22	2.52	3.61	1.57	1.78	2.61	1.34	1.52	2.21
2012	4.64	2.44	2.74	2.13	2.40	3.52	1.56	1.77	2.59	1.32	1.49	2.17
2013	4.86	2.74	3.14	2.18	2.48	3.57	1.55	1.76	2.57	1.31	1.48	2.15
2014	5.17	2.48	2.76	2.09	2.35	3.44	1.53	1.72	2.54	1.29	1.45	2.11
2015	3.76	2.22	2.53	1.96	2.22	3.27	1.49	1.67	2.46	1.26	1.43	2.06
2016	3.47	1.99	2.27	1.85	2.09	3.07	1.43	1.61	2.34	1.22	1.38	2.00
2017	4.04	1.82	2.05	1.69	1.91	2.78	1.34	1.51	2.20	1.17	1.32	1.91
2018	4.41	1.81	2.02	1.62	1.84	2.69	1.27	1.43	2.11	1.11	1.26	1.82

TABLE 4-1
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 saxatilis*)	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 4-2
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 4-3
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: RATIO OF FARLEY PREDICTED SEDIMENT CONCENTRATIONS TO SEDIMENT GUIDELINES
REVISED

Year	Average PCB Results — TEC: 0.04 mg/kg dry weight				Average PCB Results — MEC: 0.4 mg/kg dry weight				Average PCB Results — EEC: 1.7 mg/kg dry weight				Average PCB Results — NYSDEC Ben. Chr. 19.3 mg/Kg OC				Average PCB Results — NYSDEC Wildlife 1.4 mg/Kg OC			
	152 Total Sed Conc	113 Total Sed Conc	90 Total Sed Conc	50 Total Sed Conc	152 Total Sed Conc	113 Total Sed Conc	90 Total Sed Conc	50 Total Sed Conc	152 Total Sed Conc	113 Total Sed Conc	90 Total Sed Conc	50 Total Sed Conc	152 Total Sed Conc	113 Total Sed Conc	90 Total Sed Conc	50 Total Sed Conc	152 Total Sed Conc	113 Total Sed Conc	90 Total Sed Conc	50 Total Sed Conc
1993	**28**	**21**	**17**	**12**	**2.8**	**2.1**	**1.7**	**1.2**	0.7	0.5	0.4	0.3	**2.3**	**1.7**	**1.4**	**1.0**	**32**	**24**	**19**	**14**
1994	**25**	**20**	**16**	**12**	**2.5**	**2.0**	**1.6**	**1.2**	0.6	0.5	0.4	0.3	**2.1**	**1.7**	**1.3**	**1.0**	**29**	**23**	**18**	**13**
1995	**23**	**19**	**15**	**11**	**2.3**	**1.9**	**1.5**	**1.1**	0.5	0.4	0.4	0.3	**1.9**	**1.6**	**1.3**	0.9	**27**	**22**	**17**	**13**
1996	**24**	**18**	**14**	**11**	**2.4**	**1.8**	**1.4**	**1.1**	0.6	0.4	0.3	0.2	**2.0**	**1.5**	**1.2**	0.9	**27**	**21**	**17**	**12**
1997	**24**	**18**	**14**	**10**	**2.4**	**1.8**	**1.4**	**1.0**	0.6	0.4	0.3	0.2	**2.0**	**1.5**	**1.2**	0.8	**27**	**21**	**16**	**12**
1998	**22**	**17**	**14**	**9.8**	**2.2**	**1.7**	**1.4**	**1.0**	0.5	0.4	0.3	0.2	**1.8**	**1.4**	**1.1**	0.8	**25**	**20**	**15**	**11**
1999	**20**	**17**	**13**	**9.5**	**2.0**	**1.7**	**1.3**	0.9	0.5	0.4	0.3	0.2	**1.7**	**1.4**	**1.1**	0.8	**23**	**19**	**15**	**10.8**
2000	**20**	**16**	**13**	**9.2**	**2.0**	**1.6**	**1.3**	0.9	0.5	0.4	0.3	0.2	**1.7**	**1.3**	**1.0**	0.8	**23**	**18**	**14**	**10.5**
2001	**21**	**16**	**12**	**8.9**	**2.1**	**1.6**	**1.2**	0.9	0.5	0.4	0.3	0.2	**1.7**	**1.3**	**1.0**	0.7	**24**	**18**	**14**	**10.2**
2002	**20**	**16**	**12**	**8.7**	**2.0**	**1.6**	**1.2**	0.9	0.5	0.4	0.3	0.2	**1.7**	**1.3**	**1.0**	0.7	**23**	**18**	**14**	**10.0**
2003	**19**	**15**	**12**	**8.5**	**1.9**	**1.5**	**1.2**	0.8	0.5	0.4	0.3	0.2	**1.6**	**1.3**	**1.0**	0.7	**22**	**17**	**13**	**9.7**
2004	**18**	**15**	**11**	**8.3**	**1.8**	**1.5**	**1.1**	0.8	0.4	0.3	0.3	0.2	**1.5**	**1.2**	0.9	0.7	**21**	**17**	**13**	**9.4**
2005	**18**	**14**	**11**	**8.0**	**1.8**	**1.4**	**1.1**	0.8	0.4	0.3	0.3	0.2	**1.5**	**1.2**	0.9	0.7	**20**	**16**	**13**	**9.2**
2006	**18**	**14**	**11**	**7.8**	**1.8**	**1.4**	**1.1**	0.8	0.4	0.3	0.3	0.2	**1.5**	**1.2**	0.9	0.6	**20**	**16**	**12**	**8.9**
2007	**18**	**14**	**10**	**7.6**	**1.8**	**1.4**	**1.0**	0.8	0.4	0.3	0.2	0.2	**1.5**	**1.1**	0.9	0.6	**20**	**16**	**12**	**8.7**
2008	**17**	**13**	**10**	**7.5**	**1.7**	**1.3**	**1.0**	0.7	0.4	0.3	0.2	0.2	**1.4**	**1.1**	0.9	0.6	**19**	**15**	**11.7**	**8.5**
2009	**16**	**13**	**10**	**7.3**	**1.6**	**1.3**	**1.0**	0.7	0.4	0.3	0.2	0.2	**1.3**	**1.1**	0.8	0.6	**18**	**15**	**11.4**	**8.3**
2010	**16**	**13**	**9.8**	**7.2**	**1.6**	**1.3**	**1.0**	0.7	0.4	0.3	0.2	0.2	**1.4**	**1.1**	0.8	0.6	**19**	**15**	**11.2**	**8.2**
2011	**16**	**13**	**9.6**	**7.0**	**1.6**	**1.3**	**1.0**	0.7	0.4	0.3	0.2	0.2	**1.4**	**1.1**	0.8	0.6	**19**	**15**	**11.0**	**8.0**
2012	**16**	**13**	**9.5**	**6.9**	**1.6**	**1.3**	**1.0**	0.7	0.4	0.3	0.2	0.2	**1.3**	**1.0**	0.8	0.6	**18**	**14**	**10.9**	**7.9**
2013	**16**	**12**	**9.4**	**6.8**	**1.6**	**1.2**	0.9	0.7	0.4	0.3	0.2	0.2	**1.3**	**1.0**	0.8	0.6	**18**	**14**	**10.7**	**7.8**
2014	**16**	**12**	**9.2**	**6.7**	**1.6**	**1.2**	0.9	0.7	0.4	0.3	0.2	0.2	**1.3**	**1.0**	0.8	0.6	**18**	**14**	**10.6**	**7.6**
2015	**15**	**12**	**9.1**	**6.6**	**1.5**	**1.2**	0.9	0.7	0.4	0.3	0.2	0.2	**1.2**	**1.0**	0.8	0.5	**17**	**14**	**10.4**	**7.5**
2016	**14**	**12**	**8.9**	**6.4**	**1.4**	**1.2**	0.9	0.6	0.3	0.3	0.2	0.2	**1.2**	**1.0**	0.7	0.5	**17**	**13**	**10.1**	**7.4**
2017	**14**	**11**	**8.7**	**6.3**	**1.4**	**1.1**	0.9	0.6	0.3	0.3	0.2	0.1	**1.2**	0.9	0.7	0.5	**16**	**12.9**	**9.9**	**7.2**
2018	**14**	**11**	**8.5**	**6.2**	**1.4**	**1.1**	0.8	0.6	0.3	0.3	0.2	0.1	**1.2**	0.9	0.7	0.5	**16**	**12.6**	**9.7**	**7.1**

exceedances are bolded

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

Year	Average PCB Results				Average PCB Results				Average PCB Results				Average PCB Results			
	152 Total Sed Conc	113 Total Sed Conc	90 Total Sed Conc	50 Total Sed Conc	152 Total Sed Conc	113 Total Sed Conc	90 Total Sed Conc	50 Total Sed Conc	152 Total Sed Conc	113 Total Sed Conc	90 Total Sed Conc	50 Total Sed Conc	152 Total Sed Conc	113 Total Sed Conc	90 Total Sed Conc	50 Total Sed Conc
	Persaud LEL 0.07 mg/Kg dw				Persaud SEL 530 mg/Kg OC				WA PAET 1242 0.1 mg/Kg dw				WA PAET Microtox 0.021 mg/Kg			
1993	16	12	9	7	0.08	0.06	0.05	0.04	11	8.4	6.6	4.8	53	40	32	23
1994	15	11	9	7	0.08	0.06	0.05	0.03	10	8.0	6.3	4.6	48	38	30	22
1995	13	11	9	6	0.07	0.06	0.05	0.03	9.3	7.6	6.0	4.4	44	36	29	21
1996	14	11	8	6	0.07	0.06	0.04	0.03	9.6	7.4	5.8	4.2	46	35	28	20
1997	13	10	8	6	0.07	0.05	0.04	0.03	9.4	7.3	5.6	4.1	45	35	27	19
1998	12	10	8	6	0.07	0.05	0.04	0.03	8.7	7.0	5.4	3.9	42	33	26	19
1999	12	9	7	5	0.06	0.05	0.04	0.03	8.2	6.6	5.2	3.8	39	31	25	18
2000	12	9	7	5	0.06	0.05	0.04	0.03	8.2	6.4	5.0	3.7	39	31	24	17
2001	12	9	7	5	0.06	0.05	0.04	0.03	8.4	6.4	4.9	3.6	40	30	23	17
2002	12	9	7	5	0.06	0.05	0.04	0.03	8.1	6.3	4.8	3.5	38	30	23	17
2003	11	9	7	5	0.06	0.05	0.04	0.03	7.7	6.1	4.7	3.4	37	29	22	16
2004	10	8	6	5	0.05	0.04	0.03	0.02	7.3	5.9	4.5	3.3	35	28	22	16
2005	10	8	6	5	0.05	0.04	0.03	0.02	7.1	5.6	4.4	3.2	34	27	21	15
2006	10	8	6	4	0.05	0.04	0.03	0.02	7.1	5.6	4.3	3.1	34	27	20	15
2007	10	8	6	4	0.05	0.04	0.03	0.02	7.1	5.5	4.2	3.1	34	26	20	15
2008	10	8	6	4	0.05	0.04	0.03	0.02	6.8	5.4	4.1	3.0	32	26	20	14
2009	9	7	6	4	0.05	0.04	0.03	0.02	6.5	5.2	4.0	2.9	31	25	19	14
2010	9	7	6	4	0.05	0.04	0.03	0.02	6.5	5.1	3.9	2.9	31	24	19	14
2011	9	7	6	4	0.05	0.04	0.03	0.02	6.6	5.1	3.9	2.8	31	24	18	13
2012	9	7	5	4	0.05	0.04	0.03	0.02	6.4	5.0	3.8	2.8	31	24	18	13
2013	9	7	5	4	0.05	0.04	0.03	0.02	6.4	5.0	3.8	2.7	30	24	18	13
2014	9	7	5	4	0.05	0.04	0.03	0.02	6.2	4.9	3.7	2.7	30	23	18	13
2015	9	7	5	4	0.05	0.04	0.03	0.02	6.0	4.8	3.6	2.6	29	23	17	13
2016	8	7	5	4	0.04	0.03	0.03	0.02	5.8	4.6	3.6	2.6	28	22	17	12
2017	8	6	5	4	0.04	0.03	0.03	0.02	5.6	4.5	3.5	2.5	27	21	17	12
2018	8	6	5	4	0.04	0.03	0.03	0.02	5.6	4.4	3.4	2.5	26	21	16	12

TABLE 5-2: RATIO OF FARLEY PREDICTED WHOLE WATER CONCENTRATIONS TO BENCHMARKS - REVISED

Year	Tri+ Average PCB Results				Tri+ Average PCB Results			
	152 Whole Water Conc	113 Whole Water Conc	90 Whole Water Conc	50 Whole Water Conc	152 Whole Water Conc	113 Whole Water Conc	90 Whole Water Conc	50 Whole Water Conc
	USEPA/NYSDEC - Ben. Aqu. 0.014 ug/L				USEPA/NYSDEC - W. Bio. 1.2 E-04 ug/L			
1993	**2.4**	**1.7**	**1.3**	**1.1**	**286**	**199**	**155**	**125**
1994	**2.7**	**1.6**	**1.2**	0.9	**310**	**184**	**136**	**109**
1995	**1.2**	**1.0**	0.9	0.8	**136**	**117**	**108**	**92**
1996	**3.5**	**1.7**	**1.1**	0.8	**412**	**203**	**123**	**91**
1997	**2.1**	**1.3**	0.9	0.7	**249**	**154**	**108**	**82**
1998	**1.3**	**1.0**	0.8	0.6	**157**	**113**	**90**	**72**
1999	**1.2**	0.8	0.7	0.5	**135**	**93**	**77**	**63**
2000	**1.9**	**1.0**	0.7	0.5	**220**	**117**	**77**	**59**
2001	**2.1**	**1.1**	0.7	0.5	**247**	**130**	**80**	**57**
2002	**1.1**	0.7	0.6	0.4	**125**	**87**	**67**	**51**
2003	**1.2**	0.8	0.5	0.4	**138**	**88**	**64**	**48**
2004	0.7	0.5	0.4	0.4	**84**	**61**	**51**	**41**
2005	**1.1**	0.6	0.4	0.3	**123**	**67**	**47**	**37**
2006	**1.3**	0.7	0.4	0.3	**155**	**79**	**49**	**36**
2007	**1.3**	0.7	0.4	0.3	**152**	**79**	**48**	**34**
2008	0.5	0.4	0.3	0.3	**63**	**48**	**38**	**30**
2009	0.6	0.4	0.3	0.2	**65**	**44**	**34**	**27**
2010	**1.1**	0.5	0.3	0.2	**126**	**63**	**39**	**28**
2011	**1.0**	0.5	0.3	0.2	**114**	**63**	**39**	**27**
2012	0.6	0.4	0.3	0.2	**75**	**50**	**36**	**26**
2013	**1.0**	0.5	0.3	0.2	**111**	**61**	**38**	**26**
2014	0.7	0.4	0.3	0.2	**85**	**52**	**35**	**25**
2015	0.7	0.4	0.3	0.2	**81**	**48**	**33**	**24**
2016	0.3	0.3	0.2	0.2	**39**	**32**	**27**	**21**
2017	0.3	0.2	0.2	0.2	**38**	**28**	**24**	**19**
2018	0.4	0.3	0.2	0.2	**44**	**31**	**24**	**19**

exceedances are bolded

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

	River Mile 152			River Mile 113			River Mile 90			River Mile 50		
Year	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)
1993	10.8	13.6	25.8	6.8	8.6	16.9	5.3	6.6	12.5	5.2	6.7	12.6
1994	8.2	10.3	17.2	6.0	7.5	14.1	4.7	5.9	11.0	4.5	5.9	11.0
1995	6.9	8.9	15.3	4.6	5.7	12.0	4.0	5.0	9.8	4.0	5.1	9.5
1996	9.3	11.8	19.7	5.2	6.5	12.0	3.7	4.7	8.8	3.6	4.5	8.5
1997	7.3	9.1	17.0	4.6	5.6	11.2	3.5	4.3	8.0	3.2	4.1	7.6
1998	5.0	6.3	12.7	3.8	4.7	9.7	3.1	3.9	7.4	2.9	3.7	6.8
1999	4.3	5.5	10.5	3.2	4.1	8.0	2.7	3.4	6.4	2.6	3.3	6.1
2000	4.9	6.1	10.0	3.3	4.1	6.9	2.5	3.1	5.5	2.4	3.0	5.4
2001	5.2	6.6	11.4	3.4	4.2	7.8	2.4	3.0	5.3	2.2	2.8	5.0
2002	4.1	5.2	10.4	3.0	3.8	7.4	2.3	2.9	5.0	2.1	2.6	4.7
2003	4.2	5.4	8.9	2.8	3.6	6.4	2.2	2.7	4.7	2.0	2.5	4.4
2004	2.9	3.7	6.4	2.3	2.9	5.3	2.0	2.4	4.3	1.8	2.3	4.0
2005	3.3	4.0	6.4	2.2	2.8	4.7	1.8	2.2	3.8	1.7	2.1	3.6
2006	3.9	4.9	7.7	2.4	2.9	5.1	1.7	2.1	3.6	1.5	1.9	3.3
2007	3.1	3.8	6.0	2.3	2.8	4.8	1.6	2.1	3.4	1.5	1.8	3.1
2008	2.5	3.1	5.8	2.0	2.5	4.5	1.6	1.9	3.3	1.4	1.7	3.0
2009	2.5	3.2	4.9	1.7	2.2	3.7	1.4	1.8	3.0	1.3	1.6	2.7
2010	2.9	3.5	6.0	1.9	2.4	4.1	1.4	1.8	2.8	1.2	1.6	2.6
2011	3.4	4.2	6.4	2.0	2.5	4.2	1.4	1.8	2.9	1.2	1.5	2.5
2012	2.9	3.6	5.9	1.9	2.4	4.1	1.4	1.7	2.8	1.2	1.5	2.4
2013	3.2	4.1	6.5	2.0	2.4	4.3	1.4	1.8	2.9	1.2	1.5	2.4
2014	2.8	3.5	5.8	1.9	2.4	4.0	1.4	1.7	2.8	1.2	1.5	2.4
2015	2.6	3.2	4.9	1.8	2.2	3.7	1.3	1.6	2.7	1.1	1.4	2.3
2016	1.8	2.3	4.1	1.5	1.9	3.2	1.2	1.5	2.6	1.1	1.3	2.2
2017	1.8	2.2	3.9	1.4	1.8	3.0	1.2	1.4	2.4	1.0	1.3	2.1
2018	1.9	2.3	3.9	1.4	1.7	2.9	1.1	1.4	2.2	1.0	1.2	2.0

Bold values indicate exceedances

TABLE 5-4: RATIO OF PREDICTED SPOTTAIL SHINER CONCENTRATIONS TO FIELD-BASED NOAEL FOR TRI+ PCBS REVISED

	River Mile 152			River Mile 113			River Mile 90			River Mile 50		
Year	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)
1993	0.11	0.45	1.00	0.32	0.41	0.84	0.25	0.32	0.64	0.24	0.31	0.63
1994	0.08	0.33	0.68	0.29	0.37	0.71	0.23	0.29	0.57	0.21	0.28	0.55
1995	0.074	0.29	0.61	0.23	0.30	0.61	0.20	0.26	0.51	0.19	0.24	0.49
1996	0.09	0.38	0.80	0.26	0.32	0.63	0.19	0.24	0.46	0.17	0.22	0.43
1997	0.07	0.31	0.66	0.22	0.29	0.59	0.17	0.22	0.43	0.16	0.20	0.39
1998	0.060	0.24	0.50	0.19	0.25	0.50	0.16	0.20	0.39	0.14	0.18	0.36
1999	0.054	0.22	0.45	0.17	0.22	0.41	0.14	0.18	0.34	0.13	0.17	0.32
2000	0.051	0.20	0.41	0.17	0.22	0.39	0.13	0.17	0.30	0.12	0.15	0.28
2001	0.059	0.23	0.47	0.17	0.22	0.41	0.13	0.16	0.29	0.11	0.14	0.26
2002	0.055	0.21	0.44	0.16	0.21	0.39	0.12	0.16	0.29	0.10	0.13	0.25
2003	0.042	0.18	0.37	0.15	0.19	0.35	0.12	0.15	0.27	0.10	0.13	0.24
2004	0.036	0.15	0.28	0.13	0.17	0.30	0.11	0.13	0.25	0.09	0.12	0.22
2005	0.038	0.15	0.28	0.12	0.16	0.28	0.10	0.12	0.22	0.09	0.11	0.20
2006	0.043	0.17	0.33	0.13	0.16	0.29	0.09	0.12	0.21	0.08	0.10	0.19
2007	0.036	0.14	0.28	0.12	0.16	0.27	0.09	0.12	0.20	0.08	0.10	0.18
2008	0.032	0.13	0.26	0.11	0.15	0.27	0.09	0.11	0.20	0.07	0.10	0.17
2009	0.027	0.11	0.22	0.10	0.13	0.23	0.08	0.10	0.18	0.07	0.09	0.16
2010	0.036	0.14	0.27	0.10	0.13	0.24	0.08	0.10	0.17	0.07	0.09	0.15
2011	0.033	0.14	0.28	0.11	0.14	0.25	0.08	0.10	0.18	0.07	0.09	0.15
2012	0.033	0.13	0.27	0.11	0.14	0.25	0.08	0.10	0.18	0.07	0.08	0.15
2013	0.035	0.15	0.29	0.11	0.14	0.25	0.08	0.10	0.18	0.06	0.08	0.14
2014	0.030	0.13	0.26	0.11	0.14	0.24	0.08	0.10	0.17	0.06	0.08	0.14
2015	0.028	0.11	0.22	0.10	0.13	0.22	0.07	0.10	0.17	0.06	0.08	0.14
2016	0.025	0.10	0.20	0.09	0.12	0.21	0.07	0.09	0.16	0.06	0.08	0.14
2017	0.025	0.10	0.20	0.08	0.11	0.19	0.07	0.09	0.15	0.06	0.07	0.13
2018	0.025	0.10	0.20	0.08	0.11	0.19	0.06	0.08	0.14	0.05	0.07	0.12

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

	River Mile 152			River Mile 113			River Mile 90			River Mile 50		
Year	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)
1993												
1994												
1995												
1996												
1997												
1998												
1999												
2000												
2001												
2002				THIS SPECIES IS NOW COMPARED TO A FIELD-BASED NOAEL ONLY								
2003												
2004												
2005												
2006												
2007												
2008												
2009												
2010												
2011												
2012												
2013												
2014												
2015												
2016												
2017												
2018												

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

	River Mile 152			River Mile 113			River Mile 90			River Mile 50		
Year	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)
1993	**1.4**	**2.1**	**5.4**	0.9	**1.3**	**3.5**	0.7	**1.0**	**2.7**	0.7	**1.0**	**2.7**
1994	**1.1**	**1.5**	**3.6**	0.8	**1.1**	**2.9**	0.6	0.9	**2.4**	0.6	0.9	**2.3**
1995	0.9	**1.3**	**3.3**	0.6	0.9	**2.4**	0.5	0.8	**2.1**	0.5	0.8	**2.1**
1996	**1.2**	**1.7**	**4.2**	0.7	**1.0**	**2.6**	0.5	0.7	**1.8**	0.5	0.7	**1.8**
1997	0.9	**1.4**	**3.5**	0.6	0.9	**2.4**	0.4	0.6	**1.7**	0.4	0.6	**1.6**
1998	0.6	**1.0**	**2.6**	0.5	0.7	**2.0**	0.4	0.6	**1.6**	0.4	0.6	**1.5**
1999	0.6	0.8	**2.2**	0.4	0.6	**1.6**	0.3	0.5	**1.4**	0.3	0.5	**1.3**
2000	0.6	0.9	**2.1**	0.4	0.6	**1.5**	0.3	0.5	**1.2**	0.3	0.4	**1.2**
2001	0.7	**1.0**	**2.4**	0.4	0.6	**1.6**	0.3	0.5	**1.1**	0.3	0.4	**1.0**
2002	0.5	0.8	**2.1**	0.4	0.6	**1.5**	0.3	0.4	**1.1**	0.3	0.4	**1.0**
2003	0.5	0.8	**1.9**	0.4	0.5	**1.3**	0.3	0.4	**1.0**	0.2	0.4	0.9
2004	0.4	0.5	**1.4**	0.3	0.4	**1.1**	0.2	0.4	0.9	0.2	0.3	0.9
2005	0.4	0.6	**1.4**	0.3	0.4	**1.0**	0.2	0.3	0.8	0.21	0.3	0.8
2006	0.5	0.7	**1.6**	0.3	0.4	**1.1**	0.2	0.3	0.8	0.19	0.3	0.7
2007	0.4	0.6	**1.3**	0.3	0.4	**1.0**	0.21	0.3	0.7	0.19	0.3	0.7
2008	0.3	0.5	**1.2**	0.2	0.4	0.9	0.20	0.3	0.7	0.18	0.3	0.6
2009	0.3	0.5	**1.0**	0.2	0.3	0.8	0.18	0.3	0.6	0.16	0.2	0.6
2010	0.4	0.5	**1.2**	0.2	0.4	0.8	0.18	0.3	0.6	0.16	0.2	0.5
2011	0.4	0.6	**1.4**	0.3	0.4	0.9	0.18	0.3	0.6	0.15	0.2	0.5
2012	0.4	0.5	**1.3**	0.2	0.4	0.9	0.18	0.3	0.6	0.15	0.2	0.5
2013	0.4	0.6	**1.4**	0.2	0.4	0.9	0.18	0.3	0.6	0.15	0.2	0.5
2014	0.4	0.5	**1.2**	0.2	0.3	0.9	0.17	0.3	0.6	0.15	0.2	0.5
2015	0.3	0.5	**1.1**	0.2	0.3	0.8	0.17	0.2	0.6	0.14	0.21	0.5
2016	0.2	0.3	0.8	0.19	0.3	0.7	0.16	0.2	0.5	0.14	0.20	0.5
2017	0.22	0.3	0.8	0.18	0.3	0.6	0.14	0.21	0.5	0.13	0.19	0.5
2018	0.24	0.3	0.8	0.18	0.3	0.6	0.14	0.20	0.5	0.12	0.18	0.4

Bold values indicate exceedances

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

	River Mile 152			River Mile 113			River Mile 90			River Mile 50		
Year	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)
1993	0.7	**1.0**	**2.6**	0.4	0.6	**1.7**	0.3	0.5	**1.3**	0.3	0.5	**1.3**
1994	0.5	0.7	**1.8**	0.4	0.6	**1.4**	0.3	0.4	**1.1**	0.3	0.4	**1.1**
1995	0.4	0.6	**1.6**	0.3	0.4	**1.2**	0.2	0.4	**1.0**	0.3	0.4	**1.0**
1996	0.6	0.8	**2.0**	0.3	0.5	**1.2**	0.2	0.3	0.9	0.22	0.3	0.9
1997	0.5	0.7	**1.7**	0.3	0.4	**1.1**	0.21	0.3	0.8	0.20	0.3	0.8
1998	0.3	0.5	**1.3**	0.2	0.3	0.9	0.19	0.3	0.8	0.18	0.3	0.7
1999	0.3	0.4	**1.0**	0.20	0.3	0.8	0.17	0.2	0.7	0.16	0.2	0.6
2000	0.3	0.4	**1.0**	0.20	0.3	0.7	0.15	0.2	0.6	0.15	0.21	0.6
2001	0.3	0.5	**1.2**	0.21	0.3	0.8	0.15	0.22	0.5	0.13	0.20	0.5
2002	0.3	0.4	**1.0**	0.18	0.3	0.7	0.14	0.21	0.5	0.13	0.19	0.5
2003	0.3	0.4	0.9	0.17	0.3	0.7	0.13	0.20	0.5	0.12	0.18	0.4
2004	0.18	0.3	0.7	0.14	0.21	0.5	0.12	0.18	0.4	0.11	0.16	0.4
2005	0.20	0.3	0.7	0.14	0.20	0.5	0.11	0.16	0.4	0.10	0.15	0.4
2006	0.24	0.3	0.8	0.14	0.21	0.5	0.10	0.15	0.4	0.09	0.14	0.3
2007	0.19	0.3	0.6	0.14	0.21	0.5	0.10	0.15	0.3	0.09	0.13	0.3
2008	0.15	0.23	0.6	0.12	0.18	0.4	0.09	0.14	0.3	0.09	0.12	0.3
2009	0.15	0.23	0.5	0.11	0.16	0.4	0.09	0.13	0.3	0.08	0.11	0.28
2010	0.18	0.3	0.6	0.12	0.17	0.4	0.09	0.13	0.3	0.08	0.11	0.27
2011	0.21	0.3	0.7	0.12	0.18	0.4	0.09	0.13	0.3	0.07	0.11	0.26
2012	0.18	0.3	0.6	0.12	0.17	0.4	0.09	0.13	0.3	0.07	0.11	0.25
2013	0.20	0.3	0.7	0.12	0.18	0.4	0.09	0.13	0.3	0.07	0.11	0.25
2014	0.17	0.3	0.6	0.11	0.17	0.4	0.08	0.12	0.3	0.07	0.10	0.25
2015	0.16	0.23	0.5	0.11	0.16	0.4	0.08	0.12	0.28	0.07	0.10	0.24
2016	0.11	0.17	0.4	0.09	0.14	0.3	0.08	0.11	0.26	0.07	0.10	0.23
2017	0.11	0.16	0.4	0.09	0.13	0.3	0.07	0.10	0.24	0.06	0.09	0.22
2018	0.11	0.17	0.4	0.08	0.12	0.3	0.07	0.10	0.23	0.06	0.09	0.21

Bold values indicate exceedances

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

	River Mile 152			River Mile 113			River Mile 90			River Mile 50		
Year	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)
1993	0.18	0.26	0.64	0.06	0.09	0.20	0.05	0.07	0.16	0.04	0.06	0.16
1994	0.13	0.18	0.41	0.05	0.08	0.17	0.04	0.06	0.14	0.04	0.06	0.14
1995	0.12	0.17	0.40	0.04	0.06	0.15	0.04	0.05	0.13	0.04	0.05	0.12
1996	0.15	0.21	0.50	0.05	0.07	0.16	0.04	0.05	0.11	0.03	0.05	0.11
1997	0.12	0.18	0.41	0.04	0.06	0.14	0.03	0.05	0.11	0.03	0.04	0.10
1998	0.09	0.14	0.32	0.04	0.05	0.12	0.03	0.04	0.10	0.03	0.04	0.09
1999	0.09	0.13	0.29	0.03	0.05	0.10	0.03	0.04	0.08	0.02	0.03	0.08
2000	0.08	0.11	0.25	0.03	0.04	0.09	0.02	0.03	0.07	0.02	0.03	0.07
2001	0.09	0.13	0.29	0.03	0.04	0.10	0.02	0.03	0.07	0.021	0.03	0.06
2002	0.08	0.12	0.28	0.03	0.04	0.10	0.02	0.03	0.07	0.020	0.03	0.06
2003	0.07	0.10	0.22	0.03	0.04	0.09	0.022	0.03	0.06	0.019	0.03	0.06
2004	0.06	0.08	0.17	0.024	0.03	0.08	0.020	0.03	0.06	0.017	0.02	0.05
2005	0.06	0.08	0.18	0.023	0.03	0.07	0.018	0.03	0.05	0.016	0.023	0.05
2006	0.07	0.10	0.21	0.024	0.03	0.07	0.018	0.02	0.05	0.015	0.021	0.04
2007	0.06	0.08	0.17	0.023	0.03	0.07	0.017	0.024	0.05	0.015	0.020	0.04
2008	0.051	0.07	0.16	0.021	0.03	0.06	0.016	0.023	0.05	0.014	0.020	0.04
2009	0.044	0.06	0.13	0.019	0.03	0.06	0.015	0.021	0.04	0.013	0.018	0.04
2010	0.06	0.08	0.17	0.020	0.03	0.06	0.015	0.021	0.04	0.013	0.018	0.04
2011	0.05	0.08	0.17	0.021	0.03	0.06	0.015	0.021	0.04	0.012	0.018	0.04
2012	0.052	0.08	0.16	0.020	0.03	0.06	0.015	0.021	0.04	0.012	0.017	0.03
2013	0.06	0.08	0.18	0.020	0.03	0.06	0.015	0.021	0.04	0.012	0.017	0.03
2014	0.049	0.07	0.15	0.020	0.03	0.06	0.014	0.020	0.04	0.012	0.017	0.03
2015	0.045	0.06	0.14	0.019	0.03	0.05	0.014	0.020	0.04	0.012	0.016	0.03
2016	0.039	0.056	0.12	0.017	0.024	0.05	0.013	0.019	0.04	0.011	0.016	0.03
2017	0.039	0.056	0.12	0.016	0.022	0.05	0.012	0.018	0.04	0.011	0.015	0.03
2018	0.041	0.057	0.12	0.015	0.022	0.04	0.012	0.017	0.03	0.010	0.015	0.03

Bold values indicate exceedances

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

Year	River Mile 152 25th (mg/kg wet weight)	River Mile 152 Median (mg/kg wet weight)	River Mile 152 95th Percentile (mg/kg wet weight)	River Mile 113 25th (mg/kg wet weight)	River Mile 113 Median (mg/kg wet weight)	River Mile 113 95th Percentile (mg/kg wet weight)	River Mile 90 25th (mg/kg wet weight)	River Mile 90 Median (mg/kg wet weight)	River Mile 90 95th Percentile (mg/kg wet weight)	River Mile 50 25th (mg/kg wet weight)	River Mile 50 Median (mg/kg wet weight)	River Mile 50 95th Percentile (mg/kg wet weight)
1993	0.009	0.014	0.033	0.003	0.004	0.011	0.002	0.004	0.008	0.002	0.003	0.008
1994	0.007	0.010	0.022	0.003	0.004	0.009	0.002	0.003	0.007	0.002	0.003	0.007
1995	0.006	0.009	0.021	0.002	0.003	0.008	0.002	0.003	0.007	0.002	0.003	0.006
1996	0.008	0.011	0.026	0.002	0.003	0.008	0.002	0.003	0.006	0.002	0.002	0.006
1997	0.006	0.009	0.021	0.002	0.003	0.008	0.002	0.002	0.006	0.002	0.002	0.005
1998	0.005	0.007	0.017	0.002	0.003	0.006	0.002	0.002	0.005	0.001	0.002	0.005
1999	0.005	0.007	0.015	0.002	0.002	0.005	0.001	0.002	0.004	0.001	0.002	0.004
2000	0.004	0.006	0.013	0.002	0.002	0.005	0.001	0.002	0.004	0.001	0.002	0.004
2001	0.005	0.007	0.015	0.002	0.002	0.005	0.001	0.002	0.004	0.001	0.002	0.003
2002	0.004	0.006	0.014	0.002	0.002	0.005	0.001	0.002	0.004	0.001	0.001	0.003
2003	0.004	0.005	0.011	0.001	0.002	0.004	0.001	0.002	0.003	0.001	0.001	0.003
2004	0.003	0.004	0.009	0.001	0.002	0.004	0.001	0.001	0.003	0.001	0.001	0.003
2005	0.003	0.004	0.009	0.001	0.002	0.003	0.001	0.001	0.003	0.001	0.001	0.003
2006	0.004	0.005	0.011	0.001	0.002	0.004	0.001	0.001	0.003	0.001	0.001	0.002
2007	0.003	0.004	0.009	0.001	0.002	0.003	0.001	0.001	0.003	0.001	0.001	0.002
2008	0.003	0.004	0.009	0.001	0.002	0.003	0.001	0.001	0.002	0.001	0.001	0.002
2009	0.002	0.003	0.007	0.001	0.001	0.003	0.001	0.001	0.002	0.001	0.001	0.002
2010	0.003	0.004	0.009	0.001	0.001	0.003	0.001	0.001	0.002	0.001	0.001	0.002
2011	0.003	0.004	0.009	0.001	0.002	0.003	0.001	0.001	0.002	0.001	0.001	0.002
2012	0.003	0.004	0.009	0.001	0.001	0.003	0.001	0.001	0.002	0.001	0.001	0.002
2013	0.003	0.004	0.009	0.001	0.001	0.003	0.001	0.001	0.002	0.001	0.001	0.002
2014	0.003	0.004	0.008	0.001	0.001	0.003	0.001	0.001	0.002	0.001	0.001	0.002
2015	0.002	0.003	0.007	0.001	0.001	0.003	0.001	0.001	0.002	0.001	0.001	0.002
2016	0.002	0.003	0.006	0.001	0.001	0.003	0.001	0.001	0.002	0.001	0.001	0.002
2017	0.002	0.003	0.006	0.001	0.001	0.002	0.001	0.001	0.002	0.001	0.001	0.002
2018	0.002	0.003	0.006	0.001	0.001	0.002	0.001	0.001	0.002	0.001	0.001	0.002

Bold values indicate exceedances

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

Year	River Mile 152			River Mile 113			River Mile 90			River Mile 50		
	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)
1993	16	20	34	13	15	26	10	12	20	8	10	16
1994	15	18	30	12	15	24	9	12	19	7	9	15
1995	14	17	29	11	14	23	9	11	18	7	9	14
1996	14	18	29	11	13	22	9	11	17	7	8	13
1997	14	17	28	10	13	21	8	10	17	6.3	8	13
1998	13	15	26	9.9	12	21	7.8	9.7	16	6.0	7.4	12
1999	12	15	24	9.4	12	19	7.5	9.3	15	5.8	7.1	12
2000	12	15	24	9.3	12	19	7.3	9.0	15	5.6	6.9	11
2001	12	15	24	9.2	11	19	7.1	8.9	14	5.4	6.7	11
2002	11	14	24	8.9	11	18	7.0	8.7	14	5.3	6.6	11
2003	11	13	22	8.7	11	18	6.8	8.4	14	5.1	6.4	10
2004	10	13	21	8.3	10	17	6.5	8.2	13	5.0	6.2	10
2005	10	13	21	8.2	10	17	6.3	8.0	13	4.8	6.0	10
2006	10	13	21	8.1	10	17	6.2	7.8	13	4.7	5.9	9.7
2007	10	13	21	8.0	10	16	6.1	7.7	13	4.6	5.8	9.6
2008	9.7	12	20	7.8	9.8	16	6.0	7.6	12	4.6	5.7	9.4
2009	9.4	12	20	7.6	9.7	16	5.9	7.4	12	4.5	5.6	9.2
2010	9.7	12	19	7.6	9.5	15	5.8	7.3	12	4.4	5.6	9.1
2011	9.5	12	19	7.5	9.4	15	5.7	7.2	12	4.3	5.5	8.9
2012	9.2	12	19	7.4	9.3	15	5.6	7.1	12	4.3	5.4	8.8
2013	9.3	11	19	7.3	9.2	15	5.6	7.0	11	4.2	5.3	8.7
2014	8.9	11	19	7.1	9.0	15	5.5	6.9	11	4.2	5.3	8.6
2015	8.8	11	18	7.0	8.9	14	5.4	6.8	11	4.1	5.2	8.4
2016	8.6	11	18	6.9	8.8	14	5.3	6.7	11	4.0	5.1	8.3
2017	8.5	11	18	6.8	8.7	14	5.2	6.6	11	4.0	5.0	8.2
2018	8.4	11	17	6.7	8.6	14	5.2	6.5	11	3.9	5.0	8.0

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

	River Mile 152			River Mile 113			River Mile 90			River Mile 50		
Year	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)
1993	3.3	4.1	7.0	2.6	3.2	5.3	2.0	2.5	4.2	1.6	2.0	3.3
1994	3.0	3.8	6.1	2.4	3.0	4.9	1.9	2.4	3.9	1.5	1.9	3.1
1995	2.8	3.4	5.8	2.2	2.8	4.6	1.8	2.2	3.7	1.4	1.7	2.9
1996	2.9	3.7	6.0	2.2	2.7	4.5	1.7	2.1	3.6	1.3	1.7	2.7
1997	2.8	3.4	5.7	2.1	2.6	4.4	1.7	2.1	3.4	1.3	1.6	2.6
1998	2.6	3.2	5.4	2.0	2.5	4.2	1.6	2.0	3.3	1.2	1.5	2.5
1999	2.4	3.0	5.0	1.9	2.4	3.9	1.5	1.9	3.1	1.2	1.5	2.4
2000	2.4	3.0	4.9	1.9	2.4	3.8	1.5	1.8	3.0	1.1	1.4	2.3
2001	2.5	3.1	4.9	1.9	2.3	3.8	1.5	1.8	2.9	1.1	1.4	2.2
2002	2.3	2.9	4.8	1.8	2.3	3.8	1.4	1.8	2.9	1.1	1.3	2.2
2003	2.2	2.8	4.6	1.8	2.2	3.6	1.4	1.7	2.8	1.0	1.3	2.1
2004	2.1	2.7	4.4	1.7	2.1	3.5	1.3	1.7	2.7	1.0	1.3	2.1
2005	2.1	2.6	4.3	1.7	2.1	3.4	1.3	1.6	2.7	1.0	1.2	2.0
2006	2.1	2.6	4.3	1.7	2.1	3.4	1.3	1.6	2.6	1.0	1.2	2.0
2007	2.1	2.6	4.2	1.6	2.0	3.3	1.2	1.6	2.6	0.9	1.2	2.0
2008	2.0	2.5	4.1	1.6	2.0	3.3	1.2	1.5	2.5	0.9	1.2	1.9
2009	1.9	2.5	4.0	1.6	2.0	3.2	1.2	1.5	2.5	0.9	1.1	1.9
2010	2.0	2.4	3.9	1.5	1.9	3.2	1.2	1.5	2.4	0.9	1.1	1.8
2011	1.9	2.4	4.0	1.5	1.9	3.1	1.2	1.5	2.4	0.9	1.1	1.8
2012	1.9	2.4	3.9	1.5	1.9	3.1	1.2	1.5	2.4	0.9	1.1	1.8
2013	1.9	2.3	3.9	1.5	1.9	3.1	1.1	1.4	2.3	0.9	1.1	1.8
2014	1.8	2.3	3.8	1.5	1.8	3.0	1.1	1.4	2.3	0.8	1.1	1.7
2015	1.8	2.3	3.7	1.4	1.8	3.0	1.1	1.4	2.3	0.8	1.1	1.7
2016	1.8	2.2	3.7	1.4	1.8	2.9	1.1	1.4	2.2	0.8	1.0	1.7
2017	1.7	2.2	3.6	1.4	1.8	2.9	1.1	1.4	2.2	0.8	1.0	1.7
2018	1.7	2.2	3.6	1.4	1.8	2.8	1.1	1.3	2.2	0.8	1.0	1.6

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

	River Mile 152			River Mile 113			River Mile 90			River Mile 50		
Year	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)
1993	0.04	0.05	0.09	0.03	0.04	0.07	0.02	0.03	0.05	0.02	0.02	0.04
1994	0.04	0.04	0.08	0.03	0.03	0.06	0.02	0.03	0.05	0.02	0.02	0.04
1995	0.03	0.04	0.07	0.03	0.03	0.06	0.02	0.03	0.05	0.016	0.02	0.04
1996	0.03	0.04	0.08	0.03	0.03	0.06	0.02	0.03	0.04	0.016	0.02	0.03
1997	0.03	0.04	0.07	0.02	0.03	0.05	0.02	0.02	0.04	0.015	0.02	0.03
1998	0.03	0.04	0.07	0.02	0.03	0.05	0.02	0.02	0.04	0.014	0.02	0.03
1999	0.03	0.03	0.06	0.02	0.03	0.05	0.02	0.02	0.04	0.014	0.02	0.03
2000	0.03	0.04	0.06	0.02	0.03	0.05	0.017	0.02	0.04	0.013	0.02	0.03
2001	0.03	0.04	0.06	0.02	0.03	0.05	0.017	0.02	0.04	0.013	0.02	0.03
2002	0.03	0.03	0.06	0.02	0.03	0.05	0.016	0.02	0.04	0.012	0.02	0.03
2003	0.03	0.03	0.06	0.02	0.03	0.05	0.016	0.02	0.04	0.012	0.02	0.03
2004	0.02	0.03	0.05	0.02	0.02	0.04	0.015	0.02	0.03	0.012	0.015	0.03
2005	0.02	0.03	0.05	0.02	0.02	0.04	0.015	0.02	0.03	0.011	0.014	0.03
2006	0.02	0.03	0.05	0.019	0.02	0.04	0.015	0.02	0.03	0.011	0.014	0.02
2007	0.02	0.03	0.05	0.019	0.02	0.04	0.014	0.02	0.03	0.011	0.014	0.02
2008	0.02	0.03	0.05	0.018	0.02	0.04	0.014	0.02	0.03	0.011	0.014	0.02
2009	0.02	0.03	0.05	0.018	0.02	0.04	0.014	0.02	0.03	0.011	0.013	0.02
2010	0.02	0.03	0.05	0.018	0.02	0.04	0.014	0.02	0.03	0.010	0.013	0.02
2011	0.02	0.03	0.05	0.018	0.02	0.04	0.014	0.02	0.03	0.010	0.013	0.02
2012	0.02	0.03	0.05	0.017	0.02	0.04	0.013	0.02	0.03	0.010	0.013	0.02
2013	0.02	0.03	0.05	0.017	0.02	0.04	0.013	0.017	0.03	0.010	0.013	0.02
2014	0.021	0.03	0.05	0.017	0.02	0.04	0.013	0.017	0.03	0.010	0.013	0.02
2015	0.021	0.03	0.05	0.017	0.02	0.04	0.013	0.016	0.03	0.010	0.012	0.02
2016	0.020	0.03	0.05	0.016	0.02	0.04	0.013	0.016	0.03	0.010	0.012	0.02
2017	0.020	0.03	0.05	0.016	0.02	0.04	0.012	0.016	0.03	0.009	0.012	0.02
2018	0.020	0.03	0.04	0.016	0.02	0.04	0.012	0.02	0.03	0.009	0.012	0.02

Bold values indicate exceedances

TABLE 5-13: RATIO OF PREDICTED BROWN BULLHEAD CONCENTRATIONS TO LABORATORY-DERIVED LOAEL ON A TEQ BASIS
REVISED

	River Mile 152			River Mile 113			River Mile 90			River Mile 50		
Year	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)
1993	0.02	0.02	0.04	0.013	0.02	0.03	0.011	0.013	0.02	0.008	0.010	0.02
1994	0.02	0.02	0.03	0.013	0.02	0.03	0.010	0.012	0.02	0.008	0.010	0.02
1995	0.01	0.02	0.03	0.012	0.01	0.03	0.009	0.012	0.02	0.007	0.009	0.02
1996	0.02	0.02	0.03	0.011	0.01	0.02	0.009	0.011	0.02	0.007	0.009	0.02
1997	0.014	0.02	0.03	0.011	0.014	0.02	0.009	0.011	0.02	0.007	0.008	0.01
1998	0.013	0.02	0.03	0.010	0.013	0.02	0.008	0.010	0.02	0.006	0.008	0.014
1999	0.012	0.02	0.03	0.010	0.012	0.02	0.008	0.010	0.02	0.006	0.008	0.013
2000	0.013	0.02	0.03	0.010	0.012	0.02	0.008	0.010	0.02	0.006	0.007	0.013
2001	0.013	0.02	0.03	0.010	0.012	0.02	0.007	0.009	0.02	0.006	0.007	0.012
2002	0.012	0.015	0.03	0.009	0.012	0.02	0.007	0.009	0.02	0.006	0.007	0.012
2003	0.011	0.014	0.03	0.009	0.011	0.02	0.007	0.009	0.02	0.005	0.007	0.012
2004	0.011	0.014	0.02	0.009	0.011	0.02	0.007	0.009	0.015	0.005	0.007	0.012
2005	0.011	0.014	0.02	0.009	0.011	0.02	0.007	0.008	0.015	0.005	0.006	0.011
2006	0.011	0.014	0.02	0.009	0.011	0.02	0.007	0.008	0.015	0.005	0.006	0.011
2007	0.011	0.013	0.02	0.008	0.011	0.02	0.006	0.008	0.014	0.005	0.006	0.011
2008	0.010	0.013	0.02	0.008	0.010	0.02	0.006	0.008	0.014	0.005	0.006	0.011
2009	0.010	0.013	0.02	0.008	0.010	0.02	0.006	0.008	0.014	0.005	0.006	0.010
2010	0.010	0.013	0.02	0.008	0.010	0.02	0.006	0.008	0.014	0.005	0.006	0.010
2011	0.010	0.013	0.02	0.008	0.010	0.02	0.006	0.008	0.013	0.005	0.006	0.010
2012	0.010	0.012	0.02	0.008	0.010	0.02	0.006	0.008	0.013	0.005	0.006	0.010
2013	0.010	0.012	0.02	0.008	0.010	0.017	0.006	0.007	0.013	0.004	0.006	0.010
2014	0.009	0.012	0.02	0.008	0.010	0.017	0.006	0.007	0.013	0.004	0.006	0.010
2015	0.009	0.012	0.02	0.007	0.009	0.017	0.006	0.007	0.013	0.004	0.005	0.010
2016	0.009	0.012	0.02	0.007	0.009	0.016	0.006	0.007	0.012	0.004	0.005	0.009
2017	0.009	0.011	0.02	0.007	0.009	0.016	0.006	0.007	0.012	0.004	0.005	0.009
2018	0.009	0.011	0.02	0.007	0.009	0.016	0.005	0.007	0.012	0.004	0.005	0.009

Bold values indicate exceedances

TABLE 5-14: RATIO OF PREDICTED WHITE PERCH CONCENTRATIONS TO FIELD-BASED NOAEL FOR TRI+ PCBS REVISED

	River Mile 152			River Mile 113			River Mile 90			River Mile 50		
Year	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)
1993	0.7	**1.0**	**2.3**	0.5	0.6	**1.4**	0.4	0.5	**1.1**	0.3	0.4	0.9
1994	0.6	0.9	**1.9**	0.4	0.6	**1.3**	0.3	0.5	**1.0**	0.3	0.4	0.9
1995	0.5	0.8	**1.7**	0.4	0.5	**1.2**	0.3	0.4	0.9	0.3	0.4	0.8
1996	0.6	0.9	**1.8**	0.4	0.5	**1.1**	0.3	0.4	0.9	0.2	0.3	0.7
1997	0.6	0.8	**1.7**	0.4	0.5	**1.1**	0.3	0.4	0.8	0.2	0.3	0.7
1998	0.5	0.7	**1.5**	0.3	0.5	**1.0**	0.3	0.4	0.8	0.2	0.3	0.6
1999	0.5	0.6	**1.4**	0.3	0.4	0.9	0.2	0.3	0.7	0.2	0.3	0.6
2000	0.5	0.7	**1.3**	0.3	0.4	0.9	0.2	0.3	0.7	0.2	0.3	0.6
2001	0.5	0.7	**1.4**	0.3	0.4	0.9	0.2	0.3	0.7	0.2	0.3	0.5
2002	0.4	0.6	**1.3**	0.3	0.4	0.9	0.2	0.3	0.7	0.2	0.2	0.5
2003	0.4	0.6	**1.2**	0.3	0.4	0.8	0.2	0.3	0.6	0.2	0.2	0.5
2004	0.4	0.5	**1.1**	0.3	0.4	0.8	0.2	0.3	0.6	0.2	0.2	0.5
2005	0.4	0.6	**1.1**	0.3	0.4	0.7	0.2	0.3	0.6	0.2	0.2	0.5
2006	0.4	0.6	**1.1**	0.3	0.4	0.7	0.2	0.3	0.6	0.2	0.2	0.4
2007	0.4	0.5	**1.1**	0.3	0.4	0.7	0.2	0.3	0.6	0.1	0.2	0.4
2008	0.4	0.5	**1.1**	0.2	0.3	0.7	0.2	0.3	0.5	0.1	0.2	0.4
2009	0.3	0.5	**1.0**	0.2	0.3	0.7	0.2	0.3	0.5	0.1	0.2	0.4
2010	0.4	0.5	**1.0**	0.2	0.3	0.7	0.2	0.3	0.5	0.1	0.2	0.4
2011	0.4	0.5	**1.0**	0.2	0.3	0.7	0.2	0.3	0.5	0.1	0.2	0.4
2012	0.3	0.5	**1.0**	0.2	0.3	0.7	0.2	0.2	0.5	0.1	0.2	0.4
2013	0.3	0.5	**1.0**	0.2	0.3	0.7	0.2	0.2	0.5	0.1	0.2	0.4
2014	0.3	0.5	**1.0**	0.2	0.3	0.7	0.2	0.2	0.5	0.1	0.2	0.4
2015	0.3	0.5	0.9	0.2	0.3	0.6	0.2	0.2	0.5	0.1	0.2	0.4
2016	0.3	0.4	0.9	0.2	0.3	0.6	0.2	0.2	0.5	0.1	0.2	0.4
2017	0.3	0.4	0.9	0.2	0.3	0.6	0.2	0.2	0.5	0.1	0.2	0.4
2018	0.3	0.4	0.9	0.2	0.3	0.6	0.2	0.2	0.5	0.1	0.2	0.4

Bold values indicate exceedances

TABLE 5-15: RATIO OF PREDICTED YELLOW PERCH CONCENTRATIONS TO LABORATORY-DERIVED NOAEL FOR TRI+ PCBS
REVISED

	River Mile 152			River Mile 113			River Mile 90			River Mile 50		
Year	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)
1993	**5.2**	**7.1**	**16.9**	**3.8**	**5.3**	**12.2**	**3.0**	**4.2**	**9.5**	**2.8**	**3.9**	**9.4**
1994	**4.9**	**6.8**	**13.9**	**3.5**	**4.9**	**10.8**	**2.7**	**3.8**	**8.5**	**2.5**	**3.5**	**8.2**
1995	**3.7**	**5.2**	**12.7**	**2.8**	**3.8**	**8.5**	**2.4**	**3.3**	**7.3**	**2.2**	**3.1**	**7.2**
1996	**5.2**	**7.2**	**15.4**	**3.0**	**4.2**	**9.2**	**2.3**	**3.1**	**6.7**	**2.0**	**2.8**	**6.4**
1997	**3.8**	**5.2**	**12.3**	**2.7**	**3.7**	**8.3**	**2.1**	**2.8**	**6.1**	**1.9**	**2.6**	**5.8**
1998	**3.0**	**4.1**	**9.0**	**2.3**	**3.2**	**6.9**	**1.9**	**2.6**	**5.5**	**1.7**	**2.4**	**5.2**
1999	**2.6**	**3.6**	**7.8**	**2.1**	**2.8**	**5.9**	**1.7**	**2.3**	**4.8**	**1.5**	**2.1**	**4.6**
2000	**3.2**	**4.4**	**8.2**	**2.1**	**2.8**	**5.7**	**1.6**	**2.2**	**4.5**	**1.4**	**2.0**	**4.2**
2001	**3.2**	**4.3**	**8.8**	**2.1**	**2.8**	**5.8**	**1.6**	**2.1**	**4.3**	**1.3**	**1.8**	**3.9**
2002	**2.5**	**3.4**	**7.4**	**1.9**	**2.6**	**5.5**	**1.5**	**2.0**	**4.1**	**1.3**	**1.8**	**3.7**
2003	**2.5**	**3.5**	**7.1**	**1.9**	**2.5**	**5.0**	**1.4**	**1.9**	**3.9**	**1.2**	**1.7**	**3.4**
2004	**1.9**	**2.6**	**5.3**	**1.6**	**2.1**	**4.2**	**1.3**	**1.7**	**3.4**	**1.1**	**1.5**	**3.2**
2005	**2.1**	**2.9**	**5.6**	**1.5**	**2.0**	**4.0**	**1.2**	**1.6**	**3.2**	**1.1**	**1.4**	**2.9**
2006	**2.4**	**3.3**	**6.8**	**1.6**	**2.1**	**4.2**	**1.2**	**1.5**	**3.1**	**1.0**	**1.3**	**2.7**
2007	**2.2**	**2.8**	**5.4**	**1.5**	**2.0**	**4.0**	**1.1**	**1.5**	**3.0**	**1.0**	**1.3**	**2.6**
2008	**1.7**	**2.3**	**4.6**	**1.3**	**1.8**	**3.7**	**1.1**	**1.4**	**2.8**	0.9	**1.2**	**2.5**
2009	**1.6**	**2.1**	**4.5**	**1.2**	**1.6**	**3.3**	**1.0**	**1.3**	**2.6**	0.8	**1.1**	**2.2**
2010	**1.8**	**2.4**	**5.1**	**1.3**	**1.7**	**3.4**	**1.0**	**1.3**	**2.6**	0.8	**1.1**	**2.3**
2011	**2.0**	**2.8**	**5.6**	**1.3**	**1.8**	**3.6**	**1.0**	**1.3**	**2.6**	0.8	**1.1**	**2.2**
2012	**1.8**	**2.5**	**5.3**	**1.3**	**1.8**	**3.5**	**1.0**	**1.3**	**2.6**	0.8	**1.1**	**2.2**
2013	**1.9**	**2.6**	**5.2**	**1.3**	**1.8**	**3.5**	**1.0**	**1.3**	**2.6**	0.8	**1.1**	**2.2**
2014	**1.7**	**2.3**	**4.9**	**1.3**	**1.7**	**3.4**	**1.0**	**1.3**	**2.5**	0.8	**1.0**	**2.1**
2015	**1.7**	**2.2**	**4.6**	**1.2**	**1.6**	**3.3**	0.9	**1.2**	**2.5**	0.8	**1.0**	**2.1**
2016	**1.3**	**1.8**	**3.6**	**1.1**	**1.5**	**2.9**	0.9	**1.2**	**2.3**	0.7	**1.0**	**2.0**
2017	**1.2**	**1.7**	**3.4**	**1.0**	**1.4**	**2.7**	0.8	**1.1**	**2.2**	0.7	0.9	**1.9**
2018	**1.3**	**1.7**	**3.5**	**1.0**	**1.4**	**2.7**	0.8	**1.1**	**2.1**	0.7	0.9	**1.8**

Bold values indicate exceedances

TABLE 5-16: RATIO OF PREDICTED YELLOW PERCH CONCENTRATIONS TO LABORATORY-DERIVED LOAEL FOR TRI+ PCBS
REVISED

	River Mile 152			River Mile 113			River Mile 90			River Mile 50		
Year	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)
1993	**1.1**	**1.4**	**3.5**	0.8	**1.1**	**2.5**	0.6	0.9	**1.9**	0.6	0.8	**1.9**
1994	**1.0**	**1.4**	**2.8**	0.7	**1.0**	**2.2**	0.6	0.8	**1.7**	0.5	0.7	**1.7**
1995	0.8	**1.1**	**2.6**	0.6	0.8	**1.7**	0.5	0.7	**1.5**	0.5	0.6	**1.5**
1996	**1.1**	**1.5**	**3.1**	0.6	0.9	**1.9**	0.5	0.6	**1.4**	0.4	0.6	**1.3**
1997	0.8	**1.1**	**2.5**	0.5	0.7	**1.7**	0.4	0.6	**1.3**	0.4	0.5	**1.2**
1998	0.6	0.8	**1.8**	0.5	0.7	**1.4**	0.4	0.5	**1.1**	0.3	0.5	**1.1**
1999	0.5	0.7	**1.6**	0.4	0.6	**1.2**	0.3	0.5	**1.0**	0.3	0.4	0.9
2000	0.7	0.9	**1.7**	0.4	0.6	**1.2**	0.3	0.4	0.9	0.3	0.4	0.8
2001	0.7	0.9	**1.8**	0.4	0.6	**1.2**	0.3	0.4	0.9	0.3	0.4	0.8
2002	0.5	0.7	**1.5**	0.4	0.5	**1.1**	0.3	0.4	0.8	0.3	0.4	0.8
2003	0.5	0.7	**1.4**	0.4	0.5	**1.0**	0.3	0.4	0.8	0.2	0.3	0.7
2004	0.4	0.5	**1.1**	0.3	0.4	0.8	0.3	0.4	0.7	0.2	0.3	0.6
2005	0.4	0.6	**1.1**	0.3	0.4	0.8	0.2	0.3	0.6	0.2	0.3	0.6
2006	0.5	0.7	**1.4**	0.3	0.4	0.8	0.2	0.3	0.6	0.2	0.3	0.6
2007	0.4	0.6	**1.1**	0.3	0.4	0.8	0.2	0.3	0.6	0.2	0.3	0.5
2008	0.3	0.5	0.9	0.3	0.4	0.8	0.2	0.3	0.6	0.2	0.3	0.5
2009	0.3	0.4	0.9	0.2	0.3	0.7	0.2	0.3	0.5	0.2	0.2	0.5
2010	0.4	0.5	**1.0**	0.3	0.3	0.7	0.2	0.3	0.5	0.2	0.2	0.5
2011	0.4	0.6	**1.2**	0.3	0.4	0.7	0.2	0.3	0.5	0.2	0.2	0.5
2012	0.4	0.5	**1.1**	0.3	0.4	0.7	0.2	0.3	0.5	0.2	0.2	0.4
2013	0.4	0.5	**1.1**	0.3	0.4	0.7	0.2	0.3	0.5	0.2	0.2	0.4
2014	0.3	0.5	**1.0**	0.3	0.4	0.7	0.2	0.3	0.5	0.2	0.2	0.4
2015	0.3	0.5	0.9	0.2	0.3	0.7	0.2	0.3	0.5	0.2	0.2	0.4
2016	0.3	0.4	0.7	0.2	0.3	0.6	0.2	0.2	0.5	0.2	0.2	0.4
2017	0.3	0.4	0.7	0.2	0.3	0.6	0.2	0.2	0.4	0.1	0.2	0.4
2018	0.3	0.4	0.7	0.2	0.3	0.6	0.2	0.2	0.4	0.1	0.2	0.4

Bold values indicate exceedances

TABLE 5-17: RATIO OF PREDICTED WHITE PERCH CONCENTRATIONS TO LABORATORY-DERIVED NOAEL ON A TEQ BASIS
REVISED

	River Mile 152			River Mile 113			River Mile 90			River Mile 50		
Year	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)
1993	**1.0**	**1.5**	**3.5**	**1.0**	**1.5**	**3.5**	0.8	**1.2**	**2.7**	0.7	**1.0**	**2.3**
1994	0.9	**1.3**	**2.9**	0.9	**1.4**	**3.1**	0.7	**1.1**	**2.5**	0.6	0.9	**2.1**
1995	0.8	**1.1**	**2.6**	0.9	**1.2**	**2.9**	0.7	**1.0**	**2.3**	0.6	0.8	**1.9**
1996	0.9	**1.3**	**2.7**	0.9	**1.3**	**2.7**	0.7	**1.0**	**2.2**	0.5	0.8	**1.8**
1997	0.8	**1.2**	**2.6**	0.8	**1.2**	**2.7**	0.6	0.9	**2.0**	0.5	0.7	**1.7**
1998	0.7	**1.0**	**2.3**	0.8	**1.1**	**2.5**	0.6	0.9	**1.9**	0.5	0.7	**1.6**
1999	0.6	0.9	**2.1**	0.7	**1.0**	**2.3**	0.6	0.8	**1.8**	0.4	0.7	**1.5**
2000	0.7	**1.0**	**2.0**	0.7	**1.0**	**2.2**	0.5	0.8	**1.7**	0.4	0.6	**1.4**
2001	0.7	**1.0**	**2.2**	0.7	**1.0**	**2.1**	0.5	0.8	**1.7**	0.4	0.6	**1.3**
2002	0.6	0.9	**2.0**	0.7	**1.0**	**2.2**	0.5	0.7	**1.6**	0.4	0.6	**1.3**
2003	0.6	0.9	**1.9**	0.6	0.9	**2.0**	0.5	0.7	**1.6**	0.4	0.6	**1.2**
2004	0.5	0.8	**1.7**	0.6	0.9	**1.9**	0.5	0.7	**1.5**	0.4	0.5	**1.2**
2005	0.5	0.8	**1.7**	0.6	0.9	**1.9**	0.4	0.7	**1.5**	0.4	0.5	**1.2**
2006	0.6	0.8	**1.7**	0.6	0.9	**1.8**	0.4	0.7	**1.4**	0.3	0.5	**1.1**
2007	0.5	0.8	**1.7**	0.6	0.8	**1.8**	0.4	0.6	**1.4**	0.3	0.5	**1.1**
2008	0.5	0.7	**1.6**	0.5	0.8	**1.8**	0.4	0.6	**1.4**	0.3	0.5	**1.1**
2009	0.5	0.7	**1.6**	0.5	0.8	**1.7**	0.4	0.6	**1.3**	0.3	0.5	**1.0**
2010	0.5	0.7	**1.6**	0.5	0.8	**1.7**	0.4	0.6	**1.3**	0.3	0.5	**1.0**
2011	0.5	0.7	**1.6**	0.5	0.8	**1.7**	0.4	0.6	**1.3**	0.3	0.5	**1.0**
2012	0.5	0.7	**1.5**	0.5	0.8	**1.7**	0.4	0.6	**1.3**	0.3	0.5	**1.0**
2013	0.5	0.7	**1.6**	0.5	0.8	**1.7**	0.4	0.6	**1.2**	0.3	0.4	**1.0**
2014	0.5	0.7	**1.5**	0.5	0.7	**1.6**	0.4	0.6	**1.2**	0.3	0.4	0.9
2015	0.5	0.7	**1.5**	0.5	0.7	**1.6**	0.4	0.6	**1.2**	0.3	0.4	0.9
2016	0.4	0.6	**1.4**	0.5	0.7	**1.6**	0.4	0.5	**1.2**	0.3	0.4	0.9
2017	0.4	0.6	**1.4**	0.5	0.7	**1.5**	0.4	0.5	**1.2**	0.3	0.4	0.9
2018	0.4	0.6	**1.4**	0.5	0.7	**1.5**	0.4	0.5	**1.1**	0.3	0.4	0.9

Bold values indicate exceedances

TABLE 5-18: RATIO OF PREDICTED WHITE PERCH CONCENTRATIONS TO LABORATORY-DERIVED LOAEL ON A TEQ BASIS REVISED

	River Mile 152			River Mile 113			River Mile 90			River Mile 50		
Year	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)
1993	0.5	0.7	**1.7**	0.5	0.7	**1.7**	0.4	0.6	**1.3**	0.3	0.5	**1.1**
1994	0.4	0.6	**1.4**	0.5	0.7	**1.5**	0.4	0.5	**1.2**	0.3	0.4	**1.0**
1995	0.4	0.5	**1.3**	0.4	0.6	**1.4**	0.3	0.5	**1.1**	0.3	0.4	0.9
1996	0.4	0.6	**1.3**	0.4	0.6	**1.3**	0.3	0.5	**1.0**	0.3	0.4	0.9
1997	0.4	0.6	**1.3**	0.4	0.6	**1.3**	0.3	0.4	**1.0**	0.2	0.4	0.8
1998	0.3	0.5	**1.1**	0.4	0.5	**1.2**	0.3	0.4	0.9	0.2	0.3	0.8
1999	0.3	0.5	**1.0**	0.3	0.5	**1.1**	0.3	0.4	0.9	0.2	0.3	0.7
2000	0.3	0.5	**1.0**	0.3	0.5	**1.0**	0.3	0.4	0.8	0.2	0.3	0.7
2001	0.3	0.5	**1.1**	0.3	0.5	**1.0**	0.2	0.4	0.8	0.2	0.3	0.6
2002	0.3	0.4	**1.0**	0.3	0.5	**1.0**	0.2	0.4	0.8	0.2	0.3	0.6
2003	0.3	0.4	0.9	0.3	0.4	**1.0**	0.2	0.3	0.8	0.2	0.3	0.6
2004	0.3	0.4	0.8	0.3	0.4	0.9	0.2	0.3	0.7	0.2	0.3	0.6
2005	0.3	0.4	0.8	0.3	0.4	0.9	0.2	0.3	0.7	0.2	0.3	0.6
2006	0.3	0.4	0.8	0.3	0.4	0.9	0.2	0.3	0.7	0.2	0.2	0.5
2007	0.3	0.4	0.8	0.3	0.4	0.9	0.2	0.3	0.7	0.2	0.2	0.5
2008	0.2	0.4	0.8	0.3	0.4	0.9	0.2	0.3	0.7	0.2	0.2	0.5
2009	0.2	0.3	0.7	0.3	0.4	0.8	0.2	0.3	0.6	0.2	0.2	0.5
2010	0.2	0.4	0.8	0.3	0.4	0.8	0.2	0.3	0.6	0.2	0.2	0.5
2011	0.2	0.4	0.8	0.3	0.4	0.8	0.2	0.3	0.6	0.15	0.2	0.5
2012	0.2	0.3	0.7	0.2	0.4	0.8	0.2	0.3	0.6	0.15	0.2	0.5
2013	0.2	0.4	0.8	0.2	0.4	0.8	0.2	0.3	0.6	0.14	0.2	0.5
2014	0.2	0.3	0.7	0.2	0.4	0.8	0.2	0.3	0.6	0.14	0.2	0.5
2015	0.2	0.3	0.7	0.2	0.4	0.8	0.2	0.3	0.6	0.14	0.2	0.4
2016	0.2	0.3	0.7	0.2	0.3	0.8	0.2	0.3	0.6	0.14	0.2	0.4
2017	0.2	0.3	0.7	0.2	0.3	0.7	0.2	0.3	0.6	0.13	0.2	0.4
2018	0.2	0.3	0.7	0.2	0.3	0.7	0.2	0.3	0.5	0.13	0.2	0.4

Bold values indicate exceedances

TABLE 5-19: RATIO OF PREDICTED YELLOW PERCH CONCENTRATIONS TO LABORATORY-DERIVED NOAEL ON A TEQ BASIS
REVISED

	River Mile 152			River Mile 113			River Mile 90			River Mile 50		
Year	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)
1993	**3.1**	**4.3**	**10.2**	**1.9**	**2.7**	**6.3**	**1.5**	**2.1**	**4.9**	**1.4**	**2.0**	**4.9**
1994	**2.9**	**4.1**	**8.6**	**1.8**	**2.5**	**5.6**	**1.4**	**1.9**	**4.4**	**1.3**	**1.8**	**4.3**
1995	**2.2**	**3.1**	**7.8**	**1.4**	**2.0**	**4.5**	**1.2**	**1.7**	**3.8**	**1.1**	**1.6**	**3.7**
1996	**3.1**	**4.3**	**9.5**	**1.6**	**2.2**	**4.8**	**1.1**	**1.6**	**3.6**	**1.0**	**1.4**	**3.3**
1997	**2.2**	**3.1**	**7.4**	**1.4**	**1.9**	**4.3**	**1.1**	**1.5**	**3.2**	**1.0**	**1.3**	**3.0**
1998	**1.8**	**2.5**	**5.5**	**1.2**	**1.6**	**3.7**	**1.0**	**1.3**	**2.9**	0.9	**1.2**	**2.7**
1999	**1.6**	**2.2**	**4.8**	**1.0**	**1.4**	**3.1**	0.9	**1.2**	**2.5**	0.8	**1.1**	**2.4**
2000	**1.9**	**2.6**	**5.0**	**1.1**	**1.5**	**3.0**	0.8	**1.1**	**2.3**	0.7	**1.0**	**2.2**
2001	**1.9**	**2.6**	**5.3**	**1.1**	**1.5**	**3.1**	0.8	**1.1**	**2.3**	0.7	0.9	**2.0**
2002	**1.5**	**2.1**	**4.6**	**1.0**	**1.4**	**2.9**	0.8	**1.0**	**2.2**	0.7	0.9	**1.9**
2003	**1.5**	**2.1**	**4.3**	0.9	**1.3**	**2.6**	0.7	**1.0**	**2.0**	0.6	0.9	**1.8**
2004	**1.1**	**1.5**	**3.2**	0.8	**1.1**	**2.2**	0.7	0.9	**1.8**	0.6	0.8	**1.6**
2005	**1.3**	**1.7**	**3.4**	0.8	**1.1**	**2.1**	0.6	0.8	**1.7**	0.5	0.7	**1.5**
2006	**1.4**	**2.0**	**4.1**	0.8	**1.1**	**2.2**	0.6	0.8	**1.6**	0.5	0.7	**1.4**
2007	**1.3**	**1.7**	**3.3**	0.8	**1.0**	**2.1**	0.6	0.8	**1.6**	0.5	0.7	**1.4**
2008	**1.0**	**1.4**	**2.8**	0.7	0.9	**2.0**	0.5	0.7	**1.5**	0.5	0.6	**1.3**
2009	0.9	**1.3**	**2.7**	0.6	0.8	**1.7**	0.5	0.7	**1.3**	0.4	0.6	**1.2**
2010	**1.1**	**1.5**	**3.1**	0.6	0.9	**1.8**	0.5	0.7	**1.4**	0.4	0.6	**1.2**
2011	**1.2**	**1.7**	**3.4**	0.7	0.9	**1.9**	0.5	0.7	**1.4**	0.4	0.6	**1.2**
2012	**1.1**	**1.5**	**3.2**	0.7	0.9	**1.8**	0.5	0.7	**1.4**	0.4	0.5	**1.1**
2013	**1.1**	**1.6**	**3.2**	0.7	0.9	**1.8**	0.5	0.7	**1.4**	0.4	0.5	**1.1**
2014	**1.0**	**1.4**	**3.0**	0.6	0.9	**1.8**	0.5	0.7	**1.3**	0.4	0.5	**1.1**
2015	**1.0**	**1.3**	**2.8**	0.6	0.8	**1.7**	0.5	0.6	**1.3**	0.4	0.5	**1.1**
2016	0.8	**1.1**	**2.2**	0.5	0.8	**1.5**	0.4	0.6	**1.2**	0.4	0.5	**1.0**
2017	0.7	**1.0**	**2.1**	0.5	0.7	**1.4**	0.4	0.6	**1.1**	0.4	0.5	**1.0**
2018	0.7	**1.1**	**2.1**	0.5	0.7	**1.4**	0.4	0.5	**1.1**	0.3	0.5	**1.0**

Bold values indicate exceedances

TABLE 5-20: RATIO OF PREDICTED YELLOW PERCH CONCENTRATIONS TO LABORATORY-DERIVED LOAEL ON A TEQ BASIS
REVISED

	River Mile 152			River Mile 113			River Mile 90			River Mile 50		
Year	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)
1993	**1.5**	**2.1**	**4.9**	0.9	**1.3**	**3.1**	0.7	**1.0**	**2.4**	0.7	**1.0**	**2.4**
1994	**1.4**	**2.0**	**4.1**	0.9	**1.2**	**2.7**	0.7	0.9	**2.1**	0.6	0.9	**2.1**
1995	**1.1**	**1.5**	**3.8**	0.7	0.9	**2.2**	0.6	0.8	**1.8**	0.6	0.8	**1.8**
1996	**1.5**	**2.1**	**4.6**	0.7	**1.0**	**2.3**	0.6	0.8	**1.7**	0.5	0.7	**1.6**
1997	**1.1**	**1.5**	**3.6**	0.7	0.9	**2.1**	0.5	0.7	**1.6**	0.5	0.6	**1.5**
1998	0.9	**1.2**	**2.7**	0.6	0.8	**1.8**	0.5	0.6	**1.4**	0.4	0.6	**1.3**
1999	0.8	**1.1**	**2.3**	0.5	0.7	**1.5**	0.4	0.6	**1.2**	0.4	0.5	**1.2**
2000	0.9	**1.3**	**2.4**	0.5	0.7	**1.5**	0.4	0.5	**1.1**	0.4	0.5	**1.1**
2001	0.9	**1.2**	**2.6**	0.5	0.7	**1.5**	0.4	0.5	**1.1**	0.3	0.5	**1.0**
2002	0.7	**1.0**	**2.2**	0.5	0.7	**1.4**	0.4	0.5	**1.0**	0.3	0.4	0.9
2003	0.7	**1.0**	**2.1**	0.5	0.6	**1.3**	0.4	0.5	**1.0**	0.3	0.4	0.9
2004	0.6	0.7	**1.6**	0.4	0.5	**1.1**	0.3	0.4	0.9	0.3	0.4	0.8
2005	0.6	0.8	**1.6**	0.4	0.5	**1.0**	0.3	0.4	0.8	0.3	0.4	0.7
2006	0.7	0.9	**2.0**	0.4	0.5	**1.0**	0.3	0.4	0.8	0.2	0.3	0.7
2007	0.6	0.8	**1.6**	0.4	0.5	**1.0**	0.3	0.4	0.8	0.2	0.3	0.7
2008	0.5	0.7	**1.4**	0.3	0.5	0.9	0.3	0.4	0.7	0.2	0.3	0.6
2009	0.5	0.6	**1.3**	0.3	0.4	0.8	0.2	0.3	0.7	0.2	0.3	0.6
2010	0.5	0.7	**1.5**	0.3	0.4	0.9	0.2	0.3	0.7	0.2	0.3	0.6
2011	0.6	0.8	**1.7**	0.3	0.5	0.9	0.2	0.3	0.7	0.2	0.3	0.6
2012	0.5	0.7	**1.6**	0.3	0.4	0.9	0.2	0.3	0.7	0.20	0.3	0.6
2013	0.5	0.8	**1.5**	0.3	0.4	0.9	0.2	0.3	0.7	0.20	0.3	0.5
2014	0.5	0.7	**1.5**	0.3	0.4	0.9	0.2	0.3	0.6	0.19	0.3	0.5
2015	0.5	0.6	**1.3**	0.3	0.4	0.8	0.2	0.3	0.6	0.19	0.3	0.5
2016	0.4	0.5	**1.1**	0.3	0.4	0.7	0.2	0.3	0.6	0.18	0.2	0.5
2017	0.4	0.5	**1.0**	0.2	0.3	0.7	0.2	0.3	0.5	0.17	0.2	0.5
2018	0.4	0.5	**1.0**	0.2	0.3	0.7	0.2	0.3	0.5	0.17	0.2	0.5

Bold values indicate exceedances

TABLE 5-21: RATIO OF PREDICTED LARGEMOUTH BASS CONCENTRATIONS TO FIELD-BASED NOAEL FOR TRI+ PCBS REVISED

Year	River Mile 152 25th (mg/kg wet weight)	River Mile 152 Median (mg/kg wet weight)	River Mile 152 95th Percentile (mg/kg wet weight)	River Mile 113 25th (mg/kg wet weight)	River Mile 113 Median (mg/kg wet weight)	River Mile 113 95th Percentile (mg/kg wet weight)	River Mile 90 25th (mg/kg wet weight)	River Mile 90 Median (mg/kg wet weight)	River Mile 90 95th Percentile (mg/kg wet weight)	River Mile 50 25th (mg/kg wet weight)	River Mile 50 Median (mg/kg wet weight)	River Mile 50 95th Percentile (mg/kg wet weight)
1993	31	35	52	24	27	40	19	21	30	18	21	30
1994	22	25	35	21	24	35	17	19	28	16	18	27
1995	18	21	31	17	20	30	15	17	25	14	16	23
1996	25	27	37	18	20	29	13	15	22	13	14	21
1997	20	23	34	16	19	27	12	14	20	11	13	19
1998	16	19	28	14	16	24	11	13	19	10	12	17
1999	14	16	23	12	14	20	10	11	17	9.4	11	15
2000	14	15	21	11	13	19	9.1	10	15	8.5	9.6	14
2001	15	17	24	12	13	19	8.6	9.8	14	7.9	8.9	13
2002	14	16	23	11	13	19	8.4	9.5	14	7.5	8.4	12
2003	12	14	19	10	12	17	8.0	9.0	13	7.1	8.0	11
2004	9.2	10	15	9.1	10	15	7.4	8.3	12	6.6	7.5	11
2005	9.2	10	14	8.1	9.3	14	6.7	7.5	11	6.1	6.9	9.8
2006	11	12	17	8.5	9.6	14	6.2	7.1	10	5.7	6.4	9.2
2007	9.3	10	15	8.3	9.3	14	6.0	6.9	10	5.4	6.1	8.7
2008	8.4	9.6	14	7.8	8.8	13	5.9	6.6	9.6	5.1	5.8	8.3
2009	7.6	8.6	12	6.9	7.8	11	5.5	6.2	9.0	4.8	5.4	7.9
2010	8.1	9.1	13	6.8	7.8	11	5.2	5.9	8.6	4.6	5.2	7.5
2011	9.5	11	15	7.4	8.4	12	5.2	5.9	8.7	4.5	5.1	7.4
2012	8.1	9.1	13	7.1	8.0	12	5.2	5.9	8.6	4.4	5.0	7.2
2013	9.1	10	14	7.3	8.3	12	5.2	5.9	8.6	4.4	4.9	7.2
2014	8.3	9.2	13	7.0	7.8	11	5.1	5.7	8.5	4.3	4.8	7.0
2015	7.4	8.4	12	6.5	7.4	11	5.0	5.6	8.2	4.2	4.8	6.9
2016	6.6	7.6	11	6.2	7.0	10	4.8	5.4	7.8	4.1	4.6	6.7
2017	6.1	6.8	9.9	5.6	6.4	9.3	4.5	5.0	7.3	3.9	4.4	6.4
2018	6.0	6.7	9.4	5.4	6.1	9.0	4.2	4.8	7.0	3.7	4.2	6.1

TABLE 5-22: RATIO OF PREDICTED LARGEMOUTH BASS CONCENTRATIONS TO LABORATORY-DERIVED NOAEL ON A TEQ BASIS REVISED

	River Mile 152			River Mile 113			River Mile 90			River Mile 50		
Year	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)
1993	**5.0**	**6.7**	**13.3**	**2.4**	**3.3**	**6.7**	**1.9**	**2.5**	**5.1**	**1.8**	**2.5**	**5.0**
1994	**3.6**	**4.7**	**9.1**	**2.1**	**2.8**	**5.8**	**1.7**	**2.3**	**4.6**	**1.6**	**2.2**	**4.4**
1995	**2.9**	**4.0**	**8.1**	**1.7**	**2.3**	**4.9**	**1.5**	**2.0**	**4.1**	**1.4**	**1.9**	**3.9**
1996	**3.9**	**5.1**	**9.6**	**1.8**	**2.4**	**4.7**	**1.3**	**1.8**	**3.7**	**1.3**	**1.7**	**3.4**
1997	**3.3**	**4.4**	**8.7**	**1.7**	**2.2**	**4.6**	**1.3**	**1.7**	**3.4**	**1.1**	**1.6**	**3.1**
1998	**2.6**	**3.5**	**7.2**	**1.4**	**1.9**	**4.0**	**1.1**	**1.5**	**3.1**	**1.1**	**1.4**	**2.9**
1999	**2.2**	**3.0**	**5.9**	**1.2**	**1.7**	**3.4**	**1.0**	**1.4**	**2.7**	0.9	**1.3**	**2.6**
2000	**2.2**	**2.9**	**5.5**	**1.1**	**1.5**	**3.1**	0.9	**1.2**	**2.5**	0.9	**1.2**	**2.3**
2001	**2.4**	**3.2**	**6.3**	**1.2**	**1.6**	**3.2**	0.9	**1.2**	**2.4**	0.8	**1.1**	**2.1**
2002	**2.2**	**3.0**	**6.0**	**1.1**	**1.6**	**3.1**	0.8	**1.1**	**2.3**	0.8	**1.0**	**2.0**
2003	**2.0**	**2.6**	**5.0**	**1.1**	**1.4**	**2.9**	0.8	**1.1**	**2.2**	0.7	**1.0**	**1.9**
2004	**1.5**	**2.0**	**4.0**	0.9	**1.2**	**2.5**	0.7	**1.0**	**2.0**	0.7	0.9	**1.8**
2005	**1.5**	**2.0**	**3.8**	0.8	**1.1**	**2.3**	0.7	0.9	**1.8**	0.6	0.8	**1.7**
2006	**1.7**	**2.2**	**4.4**	0.9	**1.1**	**2.3**	0.6	0.8	**1.7**	0.6	0.8	**1.5**
2007	**1.5**	**2.0**	**3.9**	0.8	**1.1**	**2.2**	0.6	0.8	**1.7**	0.5	0.7	**1.5**
2008	**1.4**	**1.8**	**3.6**	0.8	**1.1**	**2.1**	0.6	0.8	**1.6**	0.5	0.7	**1.4**
2009	**1.2**	**1.6**	**3.1**	0.7	0.9	**1.9**	0.6	0.7	**1.5**	0.5	0.6	**1.3**
2010	**1.3**	**1.7**	**3.4**	0.7	0.9	**1.9**	0.5	0.7	**1.4**	0.5	0.6	**1.3**
2011	**1.5**	**2.0**	**3.8**	0.7	**1.0**	**2.0**	0.5	0.7	**1.5**	0.5	0.6	**1.2**
2012	**1.3**	**1.7**	**3.4**	0.7	**1.0**	**2.0**	0.5	0.7	**1.4**	0.4	0.6	**1.2**
2013	**1.5**	**2.0**	**3.8**	0.7	**1.0**	**2.0**	0.5	0.7	**1.4**	0.4	0.6	**1.2**
2014	**1.3**	**1.7**	**3.4**	0.7	0.9	**1.9**	0.5	0.7	**1.4**	0.4	0.6	**1.2**
2015	**1.2**	**1.6**	**3.1**	0.7	0.9	**1.8**	0.5	0.7	**1.4**	0.4	0.6	**1.2**
2016	**1.1**	**1.4**	**2.9**	0.6	0.8	**1.7**	0.5	0.6	**1.3**	0.4	0.5	**1.1**
2017	**1.0**	**1.3**	**2.6**	0.6	0.8	**1.5**	0.5	0.6	**1.2**	0.4	0.5	**1.1**
2018	**1.0**	**1.3**	**2.5**	0.5	0.7	**1.5**	0.4	0.6	**1.2**	0.4	0.5	**1.0**

Bold values indicate exceedances

TABLE 5-23: RATIO OF PREDICTED LARGEMOUTH BASS CONCENTRATIONS TO LABORATORY-DERIVED LOAEL ON A TEQ BASIS REVISED

	River Mile 152			River Mile 113			River Mile 90			River Mile 50		
Year	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)	25th (mg/kg wet weight)	Median (mg/kg wet weight)	95th Percentile (mg/kg wet weight)
1993	**2.4**	**3.2**	**6.4**	**1.2**	**1.6**	**3.3**	0.9	**1.2**	**2.5**	0.9	**1.2**	**2.4**
1994	**1.7**	**2.3**	**4.4**	**1.0**	**1.4**	**2.8**	0.8	**1.1**	**2.2**	0.8	**1.1**	**2.1**
1995	**1.4**	**1.9**	**3.9**	0.8	**1.1**	**2.4**	0.7	**1.0**	**2.0**	0.7	0.9	**1.9**
1996	**1.9**	**2.5**	**4.6**	0.9	**1.2**	**2.3**	0.6	0.9	**1.8**	0.6	0.8	**1.7**
1997	**1.6**	**2.1**	**4.2**	0.8	**1.1**	**2.2**	0.6	0.8	**1.6**	0.6	0.8	**1.5**
1998	**1.3**	**1.7**	**3.5**	0.7	0.9	**1.9**	0.6	0.7	**1.5**	0.5	0.7	**1.4**
1999	**1.1**	**1.4**	**2.9**	0.6	0.8	**1.6**	0.5	0.7	**1.3**	0.5	0.6	**1.2**
2000	**1.1**	**1.4**	**2.6**	0.6	0.7	**1.5**	0.4	0.6	**1.2**	0.4	0.6	**1.1**
2001	**1.2**	**1.5**	**3.0**	0.6	0.8	**1.6**	0.4	0.6	**1.1**	0.4	0.5	**1.0**
2002	**1.1**	**1.4**	**2.9**	0.6	0.8	**1.5**	0.4	0.5	**1.1**	0.4	0.5	**1.0**
2003	**1.0**	**1.3**	**2.4**	0.5	0.7	**1.4**	0.4	0.5	**1.1**	0.3	0.5	0.9
2004	0.7	**1.0**	**1.9**	0.4	0.6	**1.2**	0.4	0.5	**1.0**	0.3	0.4	0.9
2005	0.7	**1.0**	**1.8**	0.4	0.5	**1.1**	0.3	0.4	0.9	0.3	0.4	0.8
2006	0.8	**1.1**	**2.1**	0.4	0.6	**1.1**	0.3	0.4	0.8	0.3	0.4	0.7
2007	0.7	**1.0**	**1.9**	0.4	0.5	**1.1**	0.3	0.4	0.8	0.3	0.3	0.7
2008	0.7	0.9	**1.8**	0.4	0.5	**1.0**	0.3	0.4	0.8	0.3	0.3	0.7
2009	0.6	0.8	**1.5**	0.3	0.5	0.9	0.3	0.4	0.7	0.2	0.3	0.6
2010	0.6	0.8	**1.6**	0.3	0.4	0.9	0.3	0.3	0.7	0.2	0.3	0.6
2011	0.7	**1.0**	**1.9**	0.4	0.5	**1.0**	0.3	0.3	0.7	0.2	0.3	0.6
2012	0.6	0.8	**1.6**	0.3	0.5	0.9	0.3	0.3	0.7	0.2	0.3	0.6
2013	0.7	**1.0**	**1.8**	0.4	0.5	**1.0**	0.3	0.3	0.7	0.2	0.3	0.6
2014	0.6	0.8	**1.7**	0.3	0.5	0.9	0.2	0.3	0.7	0.2	0.3	0.6
2015	0.6	0.8	**1.5**	0.3	0.4	0.9	0.2	0.3	0.7	0.2	0.3	0.6
2016	0.5	0.7	**1.4**	0.3	0.4	0.8	0.2	0.3	0.6	0.2	0.3	0.5
2017	0.5	0.6	**1.3**	0.3	0.4	0.7	0.2	0.3	0.6	0.2	0.3	0.5
2018	0.5	0.6	**1.2**	0.3	0.4	0.7	0.2	0.3	0.6	0.2	0.2	0.5

Bold values indicate exceedances

TABLE 5-25: RATIO OF MODELED DIETARY DOSE BASED ON FISHRAND FOR FEMALE TREE SWALLOWS BASED ON THE SUM OF TRI+ CONGENERS FOR THE PERIOD 1993 - 2018

REVISED

Year	LOAEL 152 Average	LOAEL 152 95% UCL	NOAEL 152 Average	NOAEL 152 95% UCL	LOAEL 113 Average	LOAEL 113 95% UCL	NOAEL 113 Average	NOAEL 113 95% UCL	LOAEL 90 Average	LOAEL 90 95% UCL	NOAEL 90 Average	NOAEL 90 95% UCL	LOAEL 50 Average	LOAEL 50 95% UCL	NOAEL 50 Average	NOAEL 50 95% UCL
1993	NA	NA	0.08	0.09	NA	NA	0.06	0.07	NA	NA	0.05	0.05	NA	NA	0.04	0.04
1994	NA	NA	0.08	0.08	NA	NA	0.06	0.06	NA	NA	0.05	0.05	NA	NA	0.04	0.04
1995	NA	NA	0.07	0.08	NA	NA	0.06	0.06	NA	NA	0.05	0.05	NA	NA	0.03	0.04
1996	NA	NA	0.07	0.08	NA	NA	0.06	0.06	NA	NA	0.05	0.05	NA	NA	0.03	0.04
1997	NA	NA	0.07	0.07	NA	NA	0.05	0.06	NA	NA	0.04	0.05	NA	NA	0.03	0.03
1998	NA	NA	0.06	0.07	NA	NA	0.05	0.06	NA	NA	0.04	0.04	NA	NA	0.03	0.03
1999	NA	NA	0.06	0.07	NA	NA	0.05	0.05	NA	NA	0.04	0.04	NA	NA	0.03	0.03
2000	NA	NA	0.06	0.07	NA	NA	0.05	0.05	NA	NA	0.04	0.04	NA	NA	0.03	0.03
2001	NA	NA	0.06	0.07	NA	NA	0.05	0.05	NA	NA	0.04	0.04	NA	NA	0.03	0.03
2002	NA	NA	0.06	0.06	NA	NA	0.05	0.05	NA	NA	0.04	0.04	NA	NA	0.03	0.03
2003	NA	NA	0.06	0.06	NA	NA	0.05	0.05	NA	NA	0.04	0.04	NA	NA	0.03	0.03
2004	NA	NA	0.06	0.06	NA	NA	0.05	0.05	NA	NA	0.03	0.04	NA	NA	0.03	0.03
2005	NA	NA	0.06	0.06	NA	NA	0.05	0.05	NA	NA	0.03	0.04	NA	NA	0.03	0.03
2006	NA	NA	0.06	0.06	NA	NA	0.04	0.05	NA	NA	0.03	0.04	NA	NA	0.03	0.03
2007	NA	NA	0.05	0.06	NA	NA	0.04	0.05	NA	NA	0.03	0.04	NA	NA	0.03	0.03
2008	NA	NA	0.05	0.06	NA	NA	0.04	0.05	NA	NA	0.03	0.04	NA	NA	0.02	0.03
2009	NA	NA	0.05	0.06	NA	NA	0.04	0.05	NA	NA	0.03	0.03	NA	NA	0.02	0.03
2010	NA	NA	0.05	0.06	NA	NA	0.04	0.04	NA	NA	0.03	0.03	NA	NA	0.02	0.03
2011	NA	NA	0.05	0.06	NA	NA	0.04	0.04	NA	NA	0.03	0.03	NA	NA	0.02	0.03
2012	NA	NA	0.05	0.05	NA	NA	0.04	0.04	NA	NA	0.03	0.03	NA	NA	0.02	0.03
2013	NA	NA	0.05	0.05	NA	NA	0.04	0.04	NA	NA	0.03	0.03	NA	NA	0.02	0.02
2014	NA	NA	0.05	0.05	NA	NA	0.04	0.04	NA	NA	0.03	0.03	NA	NA	0.02	0.02
2015	NA	NA	0.05	0.05	NA	NA	0.04	0.04	NA	NA	0.03	0.03	NA	NA	0.02	0.02
2016	NA	NA	0.05	0.05	NA	NA	0.04	0.04	NA	NA	0.03	0.03	NA	NA	0.02	0.02
2017	NA	NA	0 05	0.05	NA	NA	0.04	0.04	NA	NA	0.03	0.03	NA	NA	0.02	0.02
2018	NA	NA	0.05	0.05	NA	NA	0.04	0.04	NA	NA	0.03	0.03	NA	NA	0.02	0.02

Bold value indicates exceedances

TABLE 5-26 : RATIO OF MODELED EGG CONCENTRATIONS TO BENCHMARKS FOR FEMALE TREE SWALLOWS BASED ON THE SUM OF TRI+ CONGENERS FOR THE PERIOD 1993 - 2018

REVISED

Year	LOAEL 152 Average	LOAEL 152 95% UCL	NOAEL 152 Average	NOAEL 152 95% UCL	LOAEL 113 Average	LOAEL 113 95% UCL	NOAEL 113 Average	NOAEL 113 95% UCL	LOAEL 90 Average	LOAEL 90 95% UCL	NOAEL 90 Average	NOAEL 90 95% UCL	LOAEL 50 Average	LOAEL 50 95% UCL	NOAEL 50 Average	NOAEL 50 95% UCL
1993	NA	NA	0.1	0.1	NA	NA	0.09	0.1	NA	NA	0.07	0.08	NA	NA	0.05	0.06
1994	NA	NA	0.1	0.1	NA	NA	0.09	0.09	NA	NA	0.07	0.07	NA	NA	0.05	0.05
1995	NA	NA	0.1	0.1	NA	NA	0.08	0.09	NA	NA	0.07	0.07	NA	NA	0.05	0.05
1996	NA	NA	0.1	0.1	NA	NA	0.08	0.08	NA	NA	0.06	0.07	NA	NA	0.05	0.05
1997	NA	NA	0.1	0.1	NA	NA	0.08	0.08	NA	NA	0.06	0.06	NA	NA	0.04	0.05
1998	NA	NA	0.09	0.1	NA	NA	0.07	0.08	NA	NA	0.06	0.06	NA	NA	0.04	0.05
1999	NA	NA	0.09	0.09	NA	NA	0.07	0.08	NA	NA	0.06	0.06	NA	NA	0.04	0.04
2000	NA	NA	0.09	0.09	NA	NA	0.07	0.07	NA	NA	0.05	0.06	NA	NA	0.04	0.04
2001	NA	NA	0.09	0.09	NA	NA	0.07	0.07	NA	NA	0.05	0.06	NA	NA	0.04	0.04
2002	NA	NA	0.08	0.09	NA	NA	0.07	0.07	NA	NA	0.05	0.06	NA	NA	0.04	0.04
2003	NA	NA	0.08	0.09	NA	NA	0.06	0.07	NA	NA	0.05	0.05	NA	NA	0.04	0.04
2004	NA	NA	0.08	0.08	NA	NA	0.06	0.07	NA	NA	0.05	0.05	NA	NA	0.04	0.04
2005	NA	NA	0.08	0.08	NA	NA	0.06	0.07	NA	NA	0.05	0.05	NA	NA	0.04	0.04
2006	NA	NA	0.08	0.08	NA	NA	0.06	0.07	NA	NA	0.05	0.05	NA	NA	0.04	0.04
2007	NA	NA	0.08	0.08	NA	NA	0.06	0.07	NA	NA	0.05	0.05	NA	NA	0.04	0.04
2008	NA	NA	0 08	0.08	NA	NA	0.06	0.06	NA	NA	0.05	0.05	NA	NA	0.03	0.04
2009	NA	NA	0.07	0.08	NA	NA	0.06	0.06	NA	NA	0.05	0.05	NA	NA	0.03	0.04
2010	NA	NA	0.07	0.08	NA	NA	0.06	0.06	NA	NA	0.05	0.05	NA	NA	0.03	0.04
2011	NA	NA	0.07	0.08	NA	NA	0.06	0.06	NA	NA	0.04	0.05	NA	NA	0.03	0.04
2012	NA	NA	0.07	0.08	NA	NA	0.06	0.06	NA	NA	0.04	0.05	NA	NA	0.03	0.04
2013	NA	NA	0.07	0.08	NA	NA	0.06	0.06	NA	NA	0.04	0.05	NA	NA	0.03	0.03
2014	NA	NA	0.07	0.07	NA	NA	0.06	0.06	NA	NA	0.04	0.05	NA	NA	0.03	0.03
2015	NA	NA	0.07	0.07	NA	NA	0.05	0.06	NA	NA	0.04	0.05	NA	NA	0.03	0.03
2016	NA	NA	0.07	0.07	NA	NA	0.05	0.06	NA	NA	0.04	0.04	NA	NA	0.03	0.03
2017	NA	NA	0.07	0.07	NA	NA	0.05	0.06	NA	NA	0.04	0.04	NA	NA	0.03	0.03
2018	NA	NA	0.07	0.07	NA	NA	0.05	0.06	NA	NA	0.04	0.04	NA	NA	0.03	0.03

Bold value indicates exceedances

TABLE 5-27: RATIO OF MODELED DIETARY DOSE BASED ON FISHRAND FOR FEMALE TREE SWALLOW USING TEQ FOR THE PERIOD 1993 - 2018
REVISED

Year	LOAEL 152 Average	LOAEL 152 95% UCL	NOAEL 152 Average	NOAEL 152 95% UCL	LOAEL 113 Average	LOAEL 113 95% UCL	NOAEL 113 Average	NOAEL 113 95% UCL	LOAEL 90 Average	LOAEL 90 95% UCL	NOAEL 90 Average	NOAEL 90 95% UCL	LOAEL 50 Average	LOAEL 50 95% UCL	NOAEL 50 Average	NOAEL 50 95% UCL
1993	NA	NA	0.04	0.04	NA	NA	0.03	0.03	NA	NA	0.02	0.02	NA	NA	0.02	0.02
1994	NA	NA	0.03	0.04	NA	NA	0.03	0.03	NA	NA	0.02	0.02	NA	NA	0.02	0.02
1995	NA	NA	0.03	0.03	NA	NA	0.03	0.03	NA	NA	0.02	0.02	NA	NA	0.02	0.02
1996	NA	NA	0.03	0.03	NA	NA	0.03	0.03	NA	NA	0.02	0.02	NA	NA	0.02	0.02
1997	NA	NA	0.03	0.03	NA	NA	0.02	0.03	NA	NA	0.02	0.02	NA	NA	0.01	0.02
1998	NA	NA	0.03	0.03	NA	NA	0.02	0.03	NA	NA	0.02	0.02	NA	NA	0.01	0.01
1999	NA	NA	0.03	0.03	NA	NA	0.02	0.02	NA	NA	0.02	0.02	NA	NA	0.01	0.01
2000	NA	NA	0.03	0.03	NA	NA	0.02	0.02	NA	NA	0.02	0.02	NA	NA	0.01	0.01
2001	NA	NA	0.03	0.03	NA	NA	0.02	0.02	NA	NA	0.02	0.02	NA	NA	0.01	0.01
2002	NA	NA	0.03	0.03	NA	NA	0.02	0.02	NA	NA	0.02	0.02	NA	NA	0.01	0.01
2003	NA	NA	0.03	0.03	NA	NA	0.02	0.02	NA	NA	0.02	0.02	NA	NA	0.01	0.01
2004	NA	NA	0.03	0.03	NA	NA	0.02	0.02	NA	NA	0.02	0.02	NA	NA	0.01	0.01
2005	NA	NA	0.03	0.03	NA	NA	0.02	0.02	NA	NA	0.02	0.02	NA	NA	0.01	0.01
2006	NA	NA	0.03	0.03	NA	NA	0.02	0.02	NA	NA	0.02	0.02	NA	NA	0.01	0.01
2007	NA	NA	0.02	0.03	NA	NA	0.02	0.02	NA	NA	0.02	0.02	NA	NA	0.01	0.01
2008	NA	NA	0.02	0.03	NA	NA	0.02	0.02	NA	NA	0.02	0.02	NA	NA	0.01	0.01
2009	NA	NA	0.02	0.03	NA	NA	0.02	0.02	NA	NA	0.01	0.02	NA	NA	0.01	0.01
2010	NA	NA	0.02	0.03	NA	NA	0.02	0.02	NA	NA	0.01	0.02	NA	NA	0.01	0.01
2011	NA	NA	0.02	0.03	NA	NA	0.02	0.02	NA	NA	0.01	0.02	NA	NA	0.01	0.01
2012	NA	NA	0.02	0.02	NA	NA	0.02	0.02	NA	NA	0.01	0.02	NA	NA	0.01	0.01
2013	NA	NA	0.02	0.02	NA	NA	0.02	0.02	NA	NA	0.01	0.02	NA	NA	0.01	0.01
2014	NA	NA	0.02	0.02	NA	NA	0.02	0.02	NA	NA	0.01	0.01	NA	NA	0.01	0.01
2015	NA	NA	0.02	0.02	NA	NA	0.02	0.02	NA	NA	0.01	0.01	NA	NA	0.01	0.01
2016	NA	NA	0.02	0.02	NA	NA	0.02	0.02	NA	NA	0.01	0.01	NA	NA	0.01	0.01
2017	NA	NA	0.02	0.02	NA	NA	0.02	0.02	NA	NA	0.01	0.01	NA	NA	0.01	0.01
2018	NA	NA	0.02	0.02	NA	NA	0.02	0.02	NA	NA	0.01	0.01	NA	NA	0.01	0.01

TABLE 5-28: RATIO OF MODELED EGG CONCENTRATIONS BASED ON FISHRAND FOR FEMALE TREE SWALLOW USING TEQ FOR THE PERIOD 1993 - 2018 REVISED

Year	LOAEL 152 Average	LOAEL 152 95% UCL	NOAEL 152 Average	NOAEL 152 95% UCL	LOAEL 113 Average	LOAEL 113 95% UCL	NOAEL 113 Average	NOAEL 113 95% UCL	LOAEL 90 Average	LOAEL 90 95% UCL	NOAEL 90 Average	NOAEL 90 95% UCL	LOAEL 50 Average	LOAEL 50 95% UCL	NOAEL 50 Average	NOAEL 50 95% UCL
1993	NA	NA	0.1	0.1	NA	NA	0.1	0.1	NA	NA	0.07	0.08	NA	NA	0.05	0.06
1994	NA	NA	0.1	0.1	NA	NA	0.1	0.1	NA	NA	0.07	0.07	NA	NA	0.05	0.05
1995	NA	NA	0.1	0.1	NA	NA	0.08	0.1	NA	NA	0.07	0.07	NA	NA	0.05	0.05
1996	NA	NA	0.1	0.1	NA	NA	0.08	0.1	NA	NA	0.06	0.07	NA	NA	0.05	0.05
1997	NA	NA	0.1	0.1	NA	NA	0.08	0.08	NA	NA	0.06	0.06	NA	NA	0.04	0.05
1998	NA	NA	0.1	0.1	NA	NA	0.07	0.08	NA	NA	0.06	0.06	NA	NA	0.04	0.05
1999	NA	NA	0.1	0.1	NA	NA	0.07	0.08	NA	NA	0.06	0.06	NA	NA	0.04	0.04
2000	NA	NA	0.1	0.1	NA	NA	0.07	0.07	NA	NA	0.05	0.06	NA	NA	0.04	0.04
2001	NA	NA	0.09	0.1	NA	NA	0.07	0.07	NA	NA	0.05	0.06	NA	NA	0.04	0.04
2002	NA	NA	0.08	0.1	NA	NA	0.07	0.07	NA	NA	0.05	0.06	NA	NA	0.04	0.04
2003	NA	NA	0.08	0.08	NA	NA	0.06	0.07	NA	NA	0.05	0.05	NA	NA	0.04	0.04
2004	NA	NA	0.08	0.08	NA	NA	0.06	0.07	NA	NA	0.05	0.05	NA	NA	0.04	0.04
2005	NA	NA	0.08	0.08	NA	NA	0.06	0.07	NA	NA	0.05	0.05	NA	NA	0.04	0.04
2006	NA	NA	0.08	0.08	NA	NA	0.06	0.07	NA	NA	0.05	0.05	NA	NA	0.04	0.04
2007	NA	NA	0.08	0.08	NA	NA	0.06	0.07	NA	NA	0.05	0.05	NA	NA	0.04	0.04
2008	NA	NA	0.08	0.08	NA	NA	0.06	0.06	NA	NA	0.05	0.05	NA	NA	0.03	0.04
2009	NA	NA	0.07	0.08	NA	NA	0.06	0.06	NA	NA	0.05	0.05	NA	NA	0.03	0.04
2010	NA	NA	0.07	0.08	NA	NA	0.06	0.06	NA	NA	0.05	0.05	NA	NA	0.03	0.04
2011	NA	NA	0.07	0.08	NA	NA	0.06	0.06	NA	NA	0.04	0.05	NA	NA	0.03	0.04
2012	NA	NA	0.07	0.08	NA	NA	0.06	0.06	NA	NA	0.04	0.05	NA	NA	0.03	0.04
2013	NA	NA	0.07	0.07	NA	NA	0.06	0.06	NA	NA	0.04	0.05	NA	NA	0.03	0.03
2014	NA	NA	0.07	0.07	NA	NA	0.06	0.06	NA	NA	0.04	0.05	NA	NA	0.03	0.03
2015	NA	NA	0.07	0.07	NA	NA	0.05	0.06	NA	NA	0.04	0.05	NA	NA	0.03	0.03
2016	NA	NA	0.07	0.07	NA	NA	0.05	0.06	NA	NA	0.04	0.04	NA	NA	0.03	0.03
2017	NA	NA	0.07	0.07	NA	NA	0.05	0.06	NA	NA	0.04	0.04	NA	NA	0.03	0.03
2018	NA	NA	0.07	0.07	NA	NA	0.05	0.06	NA	NA	0.04	0.04	NA	NA	0.03	0.03

TABLE 5-29: RATIO OF MODELED DIETARY DOSE FOR FEMALE MALLARD BASED ON FISHRAND RESULTS FOR THE TRI+ CONGENERS
REVISED

Year	LOAEL 152 Average	LOAEL 152 95% UCL	NOAEL 152 Average	NOAEL 152 95% UCL	LOAEL 113 Average	LOAEL 113 95% UCL	NOAEL 113 Average	NOAEL 113 95% UCL	LOAEL 90 Average	LOAEL 90 95% UCL	NOAEL 90 Average	NOAEL 90 95% UCL	LOAEL 50 Average	LOAEL 50 95% UCL	NOAEL 50 Average	NOAEL 50 95% UCL
1993	0.03	0.03	0.3	0.3	0.02	0.03	0.2	0.3	0.02	0.02	0.2	0.2	0.02	0.02	0.2	0.2
1994	0.03	0.03	0.3	0.3	0.02	0.02	0.2	0.2	0.02	0.02	0.2	0.2	0.02	0.02	0.2	0.2
1995	0.02	0.02	0.2	0.2	0.02	0.02	0.2	0.2	0.02	0.02	0.2	0.2	0.01	0.01	0.1	0.1
1996	0.03	0.03	0.3	0.3	0.02	0.02	0.2	0.2	0.01	0.02	0.1	0.2	0.01	0.01	0.1	0.1
1997	0.02	0.03	0.2	0.3	0.02	0.02	0.2	0.2	0.01	0.01	0.1	0.1	0.01	0.01	0.1	0.1
1998	0.02	0.02	0.2	0.2	0.02	0.02	0.2	0.2	0.01	0.01	0.1	0.1	0.01	0.01	0.1	0.1
1999	0.02	0.02	0.2	0.2	0.01	0.01	0.1	0.1	0.01	0.01	0.1	0.1	0.01	0.01	0.1	0.1
2000	0.02	0.02	0.2	0.2	0.01	0.01	0.1	0.1	0.01	0.01	0.1	0.1	0.009	0.01	0.09	0.1
2001	0.02	0.02	0.2	0.2	0.01	0.01	0.1	0.1	0.01	0.01	0.1	0.1	0.009	0.009	0.09	0.09
2002	0.02	0.02	0.2	0.2	0.01	0.01	0.1	0.1	0.01	0.01	0.1	0.1	0.008	0.009	0.08	0.09
2003	0.02	0.02	0.2	0.2	0.01	0.01	0.1	0.1	0.009	0.01	0.09	0.1	0.008	0.008	0.08	0.08
2004	0.01	0.01	0.1	0.1	0.01	0.01	0.1	0.1	0.009	0.009	0.09	0.09	0.007	0.008	0.07	0.08
2005	0.01	0.01	0.1	0.1	0.01	0.01	0.1	0.1	0.008	0.009	0.08	0.09	0.007	0.008	0.07	0.08
2006	0.01	0.02	0.1	0.2	0.01	0.01	0.1	0.1	0.008	0.009	0.08	0.09	0.007	0.007	0.07	0.07
2007	0.01	0.01	0.1	0.1	0.01	0.01	0.1	0.1	0.008	0.009	0.08	0.09	0.006	0.007	0.06	0.07
2008	0.01	0.01	0.1	0.1	0.01	0.01	0.1	0.1	0.008	0.008	0.08	0.08	0.006	0.007	0.06	0.07
2009	0.01	0.01	0.1	0.1	0.009	0.01	0.1	0.1	0.007	0.008	0.07	0.08	0.006	0.007	0.06	0.07
2010	0.01	0.01	0.1	0.1	0.01	0.01	0.1	0.1	0.007	0.008	0.07	0.08	0.006	0.006	0.06	0.06
2011	0.01	0.01	0.1	0.1	0.009	0.01	0.1	0.1	0.007	0.008	0.07	0.08	0.006	0.006	0.06	0.06
2012	0.01	0.01	0.1	0.1	0.009	0.01	0.1	0.1	0.007	0.008	0.07	0.08	0.006	0.006	0.06	0.06
2013	0.01	0.01	0.1	0.1	0.01	0.01	0.1	0.1	0.007	0.008	0.07	0.08	0.006	0.006	0.06	0.06
2014	0.01	0.01	0.1	0.1	0.009	0.01	0.09	0.1	0.007	0.007	0.07	0.07	0.005	0.006	0.05	0.06
2015	0.01	0.01	0.1	0.1	0.009	0.01	0.09	0.1	0.007	0.007	0.07	0.07	0.005	0.006	0.05	0.06
2016	0.01	0.01	0.1	0.1	0.008	0.009	0.08	0.09	0.006	0.007	0.06	0.07	0.005	0.006	0.05	0.06
2017	0.01	0.01	0.1	0.1	0.008	0.009	0.08	0.09	0.006	0.007	0.06	0.07	0.005	0.005	0.05	0.05
2018	0.01	0.01	0.1	0.1	0.008	0.009	0.08	0.09	0.006	0.007	0.06	0.07	0.005	0.005	0.05	0.05

Bold values indicate exceedances

TABLE 5-30: RATIO OF EGG CONCENTRATIONS FOR FEMALE MALLARD BASED ON FISHRAND RESULTS FOR THE TRI+ CONGENERS
REVISED

Year	LOAEL 152 Average	LOAEL 152 95% UCL	NOAEL 152 Average	NOAEL 152 95% UCL	LOAEL 113 Average	LOAEL 113 95% UCL	NOAEL 113 Average	NOAEL 113 95% UCL	LOAEL 90 Average	LOAEL 90 95% UCL	NOAEL 90 Average	NOAEL 90 95% UCL	LOAEL 50 Average	LOAEL 50 95% UCL	NOAEL 50 Average	NOAEL 50 95% UCL
1993	2.1	2.2	14	15	1.6	1.7	11	12	1.3	1.4	8.8	9.3	1.0	1.0	6.5	6.8
1994	1.9	2.0	13	14	1.6	1.6	10	11	1.3	1.3	8.4	8.8	0.9	1.0	6.2	6.6
1995	1.8	1.9	12	13	1.5	1.6	9.9	10	1.2	1.3	8.0	8.5	0.9	0.9	5.9	6.3
1996	1.8	1.9	12	13	1.4	1.5	9.6	10	1.1	1.2	7.7	8.1	0.9	0.9	5.7	6.0
1997	1.7	1.9	12	12	1.4	1.5	9.3	9.9	1.1	1.2	7.4	7.8	0.8	0.9	5.5	5.8
1998	1.7	1.8	11	12	1.3	1.4	8.9	9.5	1.1	1.1	7.1	7.6	0.8	0.8	5.3	5.6
1999	1.6	1.7	11	11	1.3	1.4	8.6	9.2	1.0	1.1	6.9	7.3	0.8	0.8	5.1	5.5
2000	1.6	1.7	11	11	1.3	1.3	8.4	8.9	1.0	1.1	6.6	7.0	0.7	0.8	5.0	5.3
2001	1.6	1.7	11	11	1.2	1.3	8.3	8.9	1.0	1.0	6.6	7.0	0.7	0.8	4.9	5.1
2002	1.5	1.6	10	11	1.2	1.3	8.1	8.6	1.0	1.0	6.4	6.8	0.7	0.7	4.7	5.0
2003	1.4	1.5	9.7	10	1.2	1.2	7.8	8.3	0.9	1.0	6.2	6.5	0.7	0.7	4.6	4.9
2004	1.4	1.5	9.6	10	1.2	1.2	7.8	8.3	0.9	0.9	6.0	6.4	0.7	0.7	4.4	4.7
2005	1.4	1.5	9.5	10	1.1	1.2	7.7	8.2	0.9	0.9	5.9	6.3	0.7	0.7	4.4	4.7
2006	1.4	1.5	9.4	10	1.1	1.2	7.5	8.1	0.9	0.9	5.8	6.2	0.6	0.7	4.3	4.7
2007	1.4	1.5	9.3	9.9	1.1	1.2	7.4	8.0	0.9	0.9	5.7	6.1	0.6	0.7	4.3	4.6
2008	1.4	1.5	9.2	9.8	1.1	1.2	7.4	7.9	0.8	0.9	5.6	6.0	0.6	0.7	4.2	4.5
2009	1.3	1.4	9.0	9.7	1.1	1.2	7.2	7.8	0.8	0.9	5.6	6.0	0.6	0.7	4.2	4.5
2010	1.3	1.4	8.9	9.5	1.1	1.1	7.2	7.7	0.8	0.9	5.5	5.9	0.6	0.7	4.1	4.4
2011	1.3	1.4	8.8	9.4	1.1	1.1	7.0	7.6	0.8	0.9	5.4	5.8	0.6	0.6	4.0	4.3
2012	1.3	1.4	8.7	9.3	1.0	1.1	7.0	7.5	0.8	0.9	5.4	5.8	0.6	0.6	4.0	4.3
2013	1.3	1.4	8.5	9.1	1.0	1.1	6.9	7.4	0.8	0.8	5.3	5.7	0.6	0.6	3.9	4.2
2014	1.3	1.3	8.4	9.0	1.0	1.1	6.8	7.3	0.8	0.8	5.2	5.6	0.6	0.6	3.9	4.2
2015	1.2	1.3	8.3	8.9	1.0	1.1	6.7	7.1	0.8	0.8	5.1	5.5	0.6	0.6	3.8	4.1
2016	1.2	1.3	8.3	9.0	1.0	1.1	6.6	7.0	0.8	0.8	5.0	5.4	0.6	0.6	3.8	4.0
2017	1.2	1.3	8.3	9.0	1.0	1.1	6.5	7.1	0.7	0.8	5.0	5.3	0.6	0.6	3.7	4.0
2018	1.2	1.4	8.4	9.1	1.0	1.1	6.6	7.2	0.7	0.8	4.9	5.2	0.5	0.6	3.7	3.9

Bold values indicate exceedances

TABLE 5-31: RATIO OF MODELED DIETARY DOSE TO BENCHMARKS FOR FEMALE MALLARD FOR PERIOD 1993 - 2018 ON A TEQ BASIS REVISED

Year	LOAEL 152 Average	LOAEL 152 95% UCL	NOAEL 152 Average	NOAEL 152 95% UCL	LOAEL 113 Average	LOAEL 113 95% UCL	NOAEL 113 Average	NOAEL 113 95% UCL	LOAEL 90 Average	LOAEL 90 95% UCL	NOAEL 90 Average	NOAEL 90 95%UCL	LOAEL 50 Average	LOAEL 50 95% UCL	NOAEL 50 Average	NOAEL 50 95% UCL
1993	14	15	137	150	11	12	107	117	8.8	9.4	88	94	14	9.2	137	92
1994	12	13	121	133	9.6	10	96	104	7.7	8.4	77	84	12	8.1	121	81
1995	9.3	10.2	93	102	7.5	8.2	75	82	6.8	7.0	68	70	9.1	7.1	91	71
1996	13	14	128	140	8.1	8.8	81	88	6.1	6.8	61	68	13	6.3	128	63
1997	10	11	103	113	7.4	8.1	74	81	5.5	6.1	55	61	10.2	5.7	102	57
1998	7.7	8.4	77	84	6.0	6.6	60	66	5.0	5.4	50	54	7.4	5.1	74	51
1999	6.7	7.3	67	73	5.1	5.5	51	55	4.5	4.7	45	47	6.3	4.6	63	46
2000	7.3	8.0	73	80	5.0	5.4	50	54	4.1	4.3	41	43	6.9	4.1	69	41
2001	8.2	9.0	82	90	5.3	5.8	53	58	3.8	4.2	38	42	7.9	3.8	79	38
2002	6.7	7.3	67	73	4.8	5.2	48	52	3.6	4.0	36	40	6.3	3.6	63	36
2003	5.8	6.3	58	63	4.5	4.9	45	49	3.4	3.8	34	38	5.4	3.4	54	34
2004	4.7	5.1	47	51	3.6	4.0	36	40	3.1	3.3	31	33	4.1	3.1	41	31
2005	4.6	5.0	46	50	3.4	3.7	34	37	2.9	3.1	29	31	4.1	2.9	41	29
2006	4.9	5.3	49	53	3.5	3.8	35	38	2.7	2.9	27	29	4.4	2.7	44	27
2007	4.5	4.9	45	49	3.4	3.7	34	37	2.5	2.8	25	28	4.0	2.5	40	25
2008	4.1	4.4	41	44	3.1	3.3	31	33	2.4	2.6	24	26	3.5	2.4	35	24
2009	3.3	3.6	33	36	2.8	3.0	28	30	2.3	2.5	23	25	2.7	2.2	27	22
2010	4.2	4.6	42	46	3.0	3.2	30	32	2.2	2.4	22	24	3.7	2.1	37	21
2011	3.9	4.2	39	42	3.0	3.3	30	33	2.1	2.4	21	24	3.3	2.1	33	21
2012	4.0	4.4	40	44	3.0	3.3	30	33	2.1	2.4	21	24	3.5	2.0	35	20
2013	4.6	5.0	46	50	3.1	3.4	31	34	2.1	2.4	21	24	4.2	2.0	42	20
2014	3.9	4.2	39	42	2.9	3.2	29	32	2.0	2.3	20	23	3.4	2.0	34	20
2015	3.7	4.0	37	40	2.7	3.0	27	30	2.0	2.3	20	23	3.2	1.9	32	19
2016	3.0	3.2	30	32	2.3	2.5	23	25	1.9	2.1	19	21	2.4	1.9	24	19
2017	2.9	3.1	29	31	2.2	2.3	22	23	1.8	1.9	18	19	2.3	1.8	23	18
2018	2.9	3.2	29	32	2.2	2.4	22	24	1.7	1.9	17	19	2.3	1.7	23	17

Bold values indicate exceedances

TABLE 5-32: RATIO OF MODELED EGG CONCENTRATION TO BENCHMARKS FOR FEMALE MALLARD FOR PERIOD 1993 - 2018 ON A TEQ BASIS REVISED

Year	LOAEL 152 Average	LOAEL 152 95% UCL	NOAEL 152 Average	NOAEL 152 95% UCL	LOAEL 113 Average	LOAEL 113 95% UCL	NOAEL 113 Average	NOAEL 113 95% UCL	LOAEL 90 Average	LOAEL 90 95% UCL	NOAEL 90 Average	NOAEL 90 95%UCL	LOAEL 50 Average	LOAEL 50 95% UCL	NOAEL 50 Average	NOAEL 50 95% UCL
1993	297	313	1187	1250	235	247	938	990	188	197	750	790	138	146	553	583
1994	274	289	1094	1154	222	235	889	938	179	189	716	755	133	140	532	561
1995	263	277	1051	1110	212	224	849	896	171	181	685	724	127	134	508	535
1996	263	278	1053	1111	205	217	819	866	164	173	657	694	122	129	489	516
1997	250	264	1000	1058	199	211	798	844	158	167	634	669	116	123	465	492
1998	236	251	945	1002	191	202	762	807	153	161	610	645	113	120	452	478
1999	226	240	904	961	184	196	737	782	147	156	587	622	110	116	439	465
2000	229	243	917	973	180	191	718	763	142	150	567	601	106	113	426	452
2001	226	240	905	959	178	189	713	757	140	149	561	595	104	110	414	440
2002	218	232	872	926	172	183	690	734	136	145	545	578	100	106	400	426
2003	207	221	828	883	167	177	666	710	131	140	525	559	98	104	392	417
2004	205	220	821	880	166	177	663	710	127	136	509	542	94	100	377	402
2005	203	217	812	869	164	176	656	703	126	135	506	541	95	101	379	405
2006	201	215	805	862	161	172	643	689	124	133	496	531	93	99	371	397
2007	198	212	792	848	159	170	635	680	123	131	490	524	91	98	365	391
2008	195	210	782	839	157	168	628	673	120	129	481	515	90	97	361	386
2009	192	206	769	826	154	166	617	662	119	128	476	510	89	95	356	381
2010	190	203	759	813	153	164	611	656	118	126	471	505	88	94	352	378
2011	188	201	752	805	150	161	602	645	116	124	463	497	86	93	345	370
2012	185	198	740	793	148	159	594	637	115	123	458	491	85	91	341	366
2013	182	195	728	779	146	157	585	628	113	121	451	483	84	90	336	361
2014	179	192	716	768	144	155	578	620	111	119	445	478	83	89	331	356
2015	177	190	708	760	142	152	568	610	109	117	438	470	82	88	327	351
2016	178	192	712	769	140	150	559	602	108	115	430	462	80	86	322	346
2017	178	193	712	770	140	151	559	603	106	114	425	456	79	85	317	341
2018	179	194	714	774	141	153	565	612	104	112	417	448	78	84	312	336

Bold values indicate exceedances

TABLE 5-33: 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 152 Average	LOAEL 152 95% UCL	NOAEL 152 Average	NOAEL 152 95% UCL	LOAEL 113 Average	LOAEL 113 95% UCL	NOAEL 113 Average	NOAEL 113 95% UCL	LOAEL 90 Average	LOAEL 90 95% UCL	NOAEL 90 Average	NOAEL 90 95% UCL	LOAEL 50 Average	LOAEL 50 95% UCL	NOAEL 50 Average	NOAEL 50 95% UCL
1993	14	14	95	99	12	12	83	86	9.3	9.6	65	67	8.7	9.0	61	63
1994	10	11	72	75	11	11	75	77	8.4	8.7	59	61	7.8	8.0	54	56
1995	9.2	9.6	65	67	8.9	9.2	62	64	7.5	7.8	53	55	6.9	7.2	48	50
1996	12	12	81	84	9.4	9.7	66	68	7.0	7.2	49	51	6.3	6.5	44	45
1997	9.6	10	67	70	8.6	8.9	60	63	6.5	6.8	46	47	5.7	5.9	40	41
1998	7.6	7.9	53	56	7.5	7.8	53	55	6.0	6.3	42	44	5.3	5.5	37	38
1999	7.2	7.5	51	53	6.7	6.9	47	49	5.5	5.7	38	40	4.8	5.0	34	35
2000	6.6	6.9	46	48	6.5	6.8	46	47	5.0	5.2	35	36	4.4	4.6	31	32
2001	7.4	7.7	52	54	6.6	6.8	46	48	4.9	5.0	34	35	4.2	4.3	29	30
2002	6.7	7.0	47	49	6.2	6.5	44	45	4.7	4.9	33	34	4.0	4.1	28	29
2003	6.1	6.3	42	44	5.8	6.1	41	42	4.5	4.6	31	33	3.8	3.9	27	27
2004	5.0	5.3	35	37	5.2	5.4	36	38	4.1	4.3	29	30	3.5	3.7	25	26
2005	5.0	5.3	35	37	5.0	5.1	35	36	3.9	4.0	27	28	3.3	3.5	23	24
2006	5.6	5.8	39	41	5.0	5.2	35	36	3.8	3.9	26	27	3.2	3.3	22	23
2007	5.0	5.2	35	36	4.9	5.0	34	35	3.7	3.8	26	27	3.1	3.2	21	22
2008	4.7	4.9	33	34	4.6	4.8	32	34	3.5	3.7	25	26	2.9	3.1	21	21
2009	4.1	4.3	29	30	4.2	4.4	29	31	3.3	3.4	23	24	2.8	2.9	19	20
2010	4.7	4.9	33	34	4.3	4.4	30	31	3.3	3.4	23	24	2.7	2.8	19	20
2011	4.8	5.0	33	35	4.5	4.6	31	32	3.3	3.4	23	24	2.7	2.8	19	19
2012	4.7	4.9	33	34	4.4	4.6	31	32	3.3	3.4	23	24	2.6	2.7	18	19
2013	4.9	5.1	34	36	4.4	4.5	31	32	3.2	3.4	23	24	2.6	2.7	18	19
2014	4.5	4.7	31	33	4.3	4.5	30	31	3.2	3.3	22	23	2.6	2.7	18	19
2015	4.1	4.2	28	30	4.1	4.2	29	30	3.1	3.2	22	22	2.5	2.6	18	18
2016	3.7	3.9	26	27	3.8	3.9	26	27	2.9	3.1	21	21	2.4	2.5	17	18
2017	3.7	3.9	26	27	3.6	3.8	25	26	2.8	2.9	20	21	2.3	2.4	16	17
2018	3.8	4.0	27	28	3.6	3.7	25	26	2.7	2.9	19	20	2.3	2.3	16	16

Bold values indicate exceedances

TABLE 5-34: 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 152 Average	LOAEL 152 95% UCL	NOAEL 152 Average	NOAEL 152 95% UCL	LOAEL 113 Average	LOAEL 113 95% UCL	NOAEL 113 Average	NOAEL 113 95% UCL	LOAEL 90 Average	LOAEL 90 95% UCL	NOAEL 90 Average	NOAEL 90 95% UCL	LOAEL 50 Average	LOAEL 50 95% UCL	NOAEL 50 Average	NOAEL 50 95% UCL
1993	**5.9**	**6.2**	**42**	**43**	**5.3**	**5.4**	**37**	**38**	**4.1**	**4.3**	**29**	**30**	**4.0**	**4.1**	**28**	**29**
1994	**4.3**	**4.5**	**30**	**31**	**4.7**	**4.8**	**33**	**34**	**3.7**	**3.8**	**26**	**27**	**3.5**	**3.6**	**25**	**25**
1995	**3.8**	**4.0**	**27**	**28**	**3.8**	**3.9**	**27**	**28**	**3.3**	**3.4**	**23**	**24**	**3.1**	**3.2**	**22**	**22**
1996	**5.0**	**5.2**	**35**	**36**	**4.1**	**4.2**	**29**	**30**	**3.0**	**3.1**	**21**	**22**	**2.8**	**2.9**	**20**	**20**
1997	**4.1**	**4.2**	**28**	**30**	**3.7**	**3.8**	**26**	**27**	**2.8**	**2.9**	**20**	**20**	**2.5**	**2.6**	**18**	**18**
1998	**3.1**	**3.2**	**22**	**22**	**3.2**	**3.3**	**22**	**23**	**2.6**	**2.7**	**18**	**19**	**2.3**	**2.4**	**16**	**17**
1999	**2.9**	**3.0**	**20**	**21**	**2.8**	**2.9**	**20**	**20**	**2.3**	**2.4**	**16**	**17**	**2.1**	**2.2**	**15**	**15**
2000	**2.6**	**2.7**	**18**	**19**	**2.7**	**2.8**	**19**	**20**	**2.1**	**2.1**	**15**	**15**	**1.9**	**2.0**	**13**	**14**
2001	**3.0**	**3.1**	**21**	**22**	**2.8**	**2.8**	**19**	**20**	**2.0**	**2.1**	**14**	**15**	**1.8**	**1.8**	**12**	**13**
2002	**2.7**	**2.8**	**19**	**20**	**2.6**	**2.7**	**18**	**19**	**2.0**	**2.0**	**14**	**14**	**1.7**	**1.7**	**12**	**12**
2003	**2.4**	**2.5**	**17**	**17**	**2.4**	**2.5**	**17**	**17**	**1.9**	**1.9**	**13**	**13**	**1.6**	**1.7**	**11**	**12**
2004	**1.9**	**2.0**	**13**	**14**	**2.1**	**2.1**	**15**	**15**	**1.7**	**1.7**	**12**	**12**	**1.5**	**1.5**	**10**	**11**
2005	**1.9**	**2.0**	**13**	**14**	**2.0**	**2.0**	**14**	**14**	**1.6**	**1.6**	**11**	**11**	**1.4**	**1.4**	**9.7**	**10**
2006	**2.2**	**2.2**	**15**	**16**	**2.0**	**2.1**	**14**	**15**	**1.5**	**1.6**	**11**	**11**	**1.3**	**1.3**	**9.2**	**9.4**
2007	**1.9**	**1.9**	**13**	**14**	**1.9**	**2.0**	**14**	**14**	**1.5**	**1.5**	**10**	**11**	**1.3**	**1.3**	**8.8**	**9.0**
2008	**1.7**	**1.8**	**12**	**13**	**1.8**	**1.9**	**13**	**13**	**1.4**	**1.4**	**9.8**	**10**	**1.2**	**1.2**	**8.4**	**8.6**
2009	**1.4**	**1.5**	**10**	**11**	**1.6**	**1.7**	**11**	**12**	**1.3**	**1.3**	**9.1**	**9.3**	**1.1**	**1.1**	**7.8**	**8.0**
2010	**1.8**	**1.8**	**12**	**13**	**1.7**	**1.7**	**12**	**12**	**1.3**	**1.3**	**9.0**	**9.2**	**1.1**	**1.1**	**7.7**	**7.9**
2011	**1.8**	**1.9**	**13**	**13**	**1.8**	**1.8**	**12**	**13**	**1.3**	**1.3**	**9.1**	**9.3**	**1.1**	**1.1**	**7.5**	**7.8**
2012	**1.8**	**1.8**	**12**	**13**	**1.7**	**1.8**	**12**	**13**	**1.3**	**1.3**	**8.9**	**9.2**	**1.1**	**1.1**	**7.4**	**7.6**
2013	**1.9**	**2.0**	**13**	**14**	**1.7**	**1.8**	**12**	**13**	**1.3**	**1.3**	**8.9**	**9.2**	**1.1**	**1.1**	**7.4**	**7.5**
2014	**1.7**	**1.8**	**12**	**12**	**1.7**	**1.8**	**12**	**12**	**1.2**	**1.3**	**8.7**	**9.0**	**1.0**	**1.1**	**7.2**	**7.4**
2015	**1.5**	**1.5**	**10**	**11**	**1.6**	**1.7**	**11**	**12**	**1.2**	**1.2**	**8.5**	**8.7**	**1.0**	**1.0**	**7.1**	**7.3**
2016	**1.3**	**1.3**	**9.0**	**9.4**	**1.5**	**1.5**	**10**	**10**	**1.1**	**1.2**	**8.0**	**8.2**	**1.0**	**1.0**	**6.8**	**7.0**
2017	**1.3**	**1.3**	**9.0**	**9.4**	**1.4**	**1.4**	**9.6**	**9.9**	**1.1**	**1.1**	**7.6**	**7.8**	0.9	**1.0**	**6.5**	**6.7**
2018	**1.3**	**1.4**	**9.4**	**9.8**	**1.4**	**1.4**	**9.5**	**9.8**	**1.1**	**1.1**	**7.4**	**7.6**	0.9	0.9	**6.2**	**6.4**

Bold values indicate exceedances

TABLE 5-35: 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 152 Average	LOAEL 152 95% UCL	NOAEL 152 Average	NOAEL 152 95% UCL	LOAEL 113 Average	LOAEL 113 95% UCL	NOAEL 113 Average	NOAEL 113 95% UCL	LOAEL 90 Average	LOAEL 90 95% UCL	NOAEL 90 Average	NOAEL 90 95% UCL	LOAEL 50 Average	LOAEL 50 95% UCL	NOAEL 50 Average	NOAEL 50 95% UCL
1993	20	20	139	141	15	16	108	110	12	12	83	84	12	12	81	82
1994	14	14	98	99	13	13	93	94	11	11	75	76	10	10	71	72
1995	12	12	82	83	11	11	77	78	9.4	10	66	67	9.0	9.1	63	64
1996	15	15	106	108	11	11	79	80	8.4	8.6	59	60	8.0	8.2	56	57
1997	13	13	91	93	10	11	73	75	7.9	8.0	55	56	7.3	7.4	51	52
1998	8.9	9.0	62	63	9.0	9.2	63	64	7.2	7.3	50	51	6.6	6.7	46	47
1999	8.1	7.8	56	55	7.8	8.0	55	56	6.4	6.5	45	46	6.0	6.1	42	42
2000	7.9	7.7	55	54	7.2	7.4	51	52	5.7	5.8	40	41	5.4	5.5	38	38
2001	8.7	8.5	61	59	7.5	7.7	53	54	5.5	5.5	38	39	5.0	5.0	35	35
2002	8.1	7.8	56	55	7.3	7.4	51	52	5.3	5.4	37	38	4.7	4.8	33	33
2003	7.1	6.9	49	48	6.6	6.7	46	47	5.0	5.1	35	36	4.4	4.5	31	32
2004	5.4	5.2	38	37	5.8	5.9	40	41	4.6	4.7	33	33	4.2	4.2	29	30
2005	5.5	5.3	39	37	5.2	5.3	36	37	4.2	4.3	29	30	3.8	3.9	27	27
2006	6.3	6.1	44	43	5.3	5.4	37	38	3.9	4.0	28	28	3.6	3.6	25	25
2007	5.6	5.4	39	38	5.2	5.3	37	37	3.8	3.9	27	27	3.4	3.4	24	24
2008	4.9	4.8	35	34	4.9	5.0	35	35	3.7	3.8	26	26	3.2	3.3	23	23
2009	4.4	4.3	31	30	4.4	4.4	30	31	3.5	3.5	24	25	3.0	3.1	21	21
2010	4.8	4.6	33	32	4.3	4.4	30	31	3.3	3.4	23	24	2.9	2.9	20	21
2011	5.5	5.4	39	38	4.7	4.8	33	33	3.3	3.4	23	24	2.8	2.9	20	20
2012	5.0	4.8	35	34	4.5	4.5	31	32	3.3	3.4	23	23	2.8	2.8	19	20
2013	5.5	5.3	39	37	4.6	4.7	32	33	3.3	3.3	23	23	2.8	2.8	19	20
2014	5.2	5.0	36	35	4.4	4.5	31	31	3.2	3.3	23	23	2.7	2.8	19	19
2015	4.4	4.3	31	30	4.1	4.2	29	30	3.1	3.2	22	22	2.7	2.7	19	19
2016	4.0	3.9	28	27	3.9	4.0	27	28	3.0	3.0	21	21	2.6	2.6	18	18
2017	3.9	3.7	27	26	3.6	3.6	25	25	2.8	2.9	20	20	2.5	2.5	17	17
2018	4.0	3.8	28	27	3.4	3.5	24	24	2.7	2.7	19	19	2.3	2.4	16	17

Bold values indicate exceedances

TABLE 5-36: RATIO OF MODELED EGG CONCENTRATIONS TO BENCHMARKS FOR FEMALE KINGFISHER BASED ON THE SUM OF TRI+ CONGENERS FOR THE PERIOD 1993 - 2018
REVISED

Year	LOAEL 152 Average	LOAEL 152 95% UCL	NOAEL 152 Average	NOAEL 152 95% UCL	LOAEL 113 Average	LOAEL 113 95% UCL	NOAEL 113 Average	NOAEL 113 95% UCL	LOAEL 90 Average	LOAEL 90 95% UCL	NOAEL 90 Average	NOAEL 90 95% UCL	LOAEL 50 Average	LOAEL 50 95% UCL	NOAEL 50 Average	NOAEL 50 95% UCL
1993	32	34	218	227	28	29	190	197	22	23	150	155	21	22	140	145
1994	25	26	164	171	26	27	172	178	20	21	136	140	19	19	125	129
1995	22	23	148	154	21	22	142	148	18	19	121	125	17	17	111	115
1996	28	29	185	193	23	23	151	156	17	17	112	116	15	16	101	104
1997	23	24	154	161	21	21	138	143	16	16	105	109	14	14	92	95
1998	18	19	122	127	18	19	121	125	14	15	97	100	13	13	85	88
1999	17	18	116	121	16	17	107	111	13	14	88	91	12	12	78	80
2000	16	17	106	111	16	16	105	109	12	12	80	83	11	11	71	74
2001	18	19	119	124	16	16	105	109	12	12	78	81	10	10	67	69
2002	16	17	108	112	15	15	100	104	11	12	76	78	9.6	9.9	64	66
2003	14	15	97	101	14	14	94	97	11	11	72	74	9.1	9.4	61	63
2004	12	13	81	84	12	13	83	86	9.9	10	66	69	8.5	8.8	57	59
2005	12	13	81	84	12	12	79	82	9.3	9.7	62	65	8.0	8.3	53	55
2006	13	14	89	93	12	12	80	83	9.0	9.4	61	63	7.6	7.9	51	53
2007	12	12	80	83	12	12	78	81	8.8	9.1	59	61	7.3	7.6	49	51
2008	11	12	75	78	11	12	74	77	8.4	8.8	57	59	7.0	7.3	47	49
2009	10	10	66	69	10	10	67	70	7.9	8.2	53	55	6.6	6.9	44	46
2010	11	12	75	79	10	11	68	71	7.9	8.2	53	55	6.5	6.8	44	45
2011	11	12	76	80	11	11	72	74	7.9	8.2	53	55	6.4	6.7	43	45
2012	11	12	75	78	11	11	70	73	7.8	8.1	52	54	6.3	6.5	42	44
2013	12	12	79	82	10	11	70	73	7.7	8.0	52	54	6.2	6.5	42	43
2014	11	11	72	75	10	11	69	71	7.6	7.9	51	53	6.1	6.4	41	43
2015	9.7	10	65	68	9.8	10	65	68	7.4	7.7	50	52	6.0	6.2	40	42
2016	8.8	9.3	59	62	9.0	9.4	60	63	7.0	7.3	47	49	5.8	6.0	39	40
2017	8.8	9.3	59	62	8.6	9.0	58	60	6.7	7.0	45	47	5.6	5.8	37	39
2018	9.1	9.6	61	64	8.6	9.0	58	60	6.6	6.8	44	46	5.4	5.6	36	38

Bold values indicate exceedances

TABLE 5-37: RATIO OF MODELED EGG CONCENTRATIONS TO BENCHMARKS FOR FEMALE BLUE HERON BASED ON THE SUM OF TRI+ CONGENERS FOR THE PERIOD 1993 - 2018

REVISED

Year	LOAEL 152 Average	LOAEL 152 95% UCL	NOAEL 152 Average	NOAEL 152 95% UCL	LOAEL 113 Average	LOAEL 113 95% UCL	NOAEL 113 Average	NOAEL 113 95% UCL	LOAEL 90 Average	LOAEL 90 95% UCL	NOAEL 90 Average	NOAEL 90 95% UCL	LOAEL 50 Average	LOAEL 50 95% UCL	NOAEL 50 Average	NOAEL 50 95% UCL
1993	36	37	241	251	32	33	214	221	25	26	168	173	24	25	162	168
1994	26	27	175	182	29	29	192	197	23	23	151	156	21	22	143	148
1995	23	24	155	161	23	24	155	160	20	21	133	137	19	20	127	131
1996	30	32	203	211	25	26	167	172	18	19	123	127	17	18	114	117
1997	25	26	165	172	23	23	151	157	17	18	114	118	15	16	103	106
1998	19	19	125	130	19	20	130	134	16	16	105	108	14	15	94	97
1999	18	18	119	123	17	17	114	117	14	14	93	96	13	13	85	88
2000	16	16	106	109	17	17	111	114	13	13	85	87	12	12	77	80
2001	18	19	123	127	17	17	112	115	12	13	82	84	11	11	72	74
2002	16	17	110	114	16	16	106	109	12	12	79	81	10	11	69	71
2003	14	15	97	101	15	15	98	101	11	12	75	77	9.7	10	65	67
2004	11	12	76	79	13	13	84	87	10	11	68	70	9.0	9.3	60	62
2005	11	12	76	79	12	12	80	82	9.4	9.7	63	65	8.4	8.6	56	58
2006	13	14	88	91	12	13	82	84	9.2	9.4	61	63	7.9	8.2	53	55
2007	11	12	76	78	12	12	79	81	8.9	9.1	60	61	7.6	7.8	51	52
2008	10	11	70	72	11	11	75	77	8.5	8.7	57	58	7.3	7.5	49	50
2009	8.7	9.0	58	61	10	10	66	68	7.8	8.0	52	54	6.7	6.9	45	46
2010	11	11	71	74	10	10	68	69	7.8	8.0	52	53	6.6	6.8	45	46
2011	11	11	73	75	11	11	72	74	7.8	8.0	52	54	6.5	6.7	44	45
2012	11	11	71	74	11	11	71	73	7.7	7.9	52	53	6.4	6.6	43	44
2013	11	12	77	80	11	11	70	72	7.7	7.9	52	53	6.4	6.5	43	44
2014	10	11	68	71	10	11	69	71	7.6	7.8	51	52	6.2	6.4	42	43
2015	8.9	9.2	60	62	9.7	10	65	67	7.3	7.5	49	50	6.1	6.3	41	42
2016	7.8	8.1	52	54	8.8	9.0	59	61	6.9	7.1	46	48	5.9	6.0	39	40
2017	7.8	8.1	52	54	8.3	8.5	56	57	6.5	6.7	44	45	5.6	5.8	38	39
2018	8.1	8.4	54	56	8.2	8.5	55	57	6.4	6.5	43	44	5.4	5.5	36	37

Bold values indicate exceedances

TABLE 5-38: RATIO OF MODELED EGG CONCENTRATIONS TO BENCHMARKS FOR FEMALE BALD EAGLES BASED ON THE SUM OF TRI+ CONGENERS FOR THE PERIOD 1993 - 2018
REVISED

Year	LOAEL 152 Average	LOAEL 152 95% UCL	NOAEL 152 Average	NOAEL 152 95% UCL	LOAEL 113 Average	LOAEL 113 95% UCL	NOAEL 113 Average	NOAEL 113 95% UCL	LOAEL 90 Average	LOAEL 90 95% UCL	NOAEL 90 Average	NOAEL 90 95% UCL	LOAEL 50 Average	LOAEL 50 95% UCL	NOAEL 50 Average	NOAEL 50 95% UCL
1993	NA	NA	55	56	NA	NA	43	44	NA	NA	32	33	NA	NA	32	33
1994	NA	NA	39	40	NA	NA	37	38	NA	NA	28	29	NA	NA	28	29
1995	NA	NA	33	33	NA	NA	31	31	NA	NA	25	26	NA	NA	25	26
1996	NA	NA	43	43	NA	NA	31	32	NA	NA	22	23	NA	NA	22	23
1997	NA	NA	36	37	NA	NA	29	30	NA	NA	20	21	NA	NA	20	21
1998	NA	NA	29	30	NA	NA	25	26	NA	NA	18	19	NA	NA	18	19
1999	NA	NA	25	25	NA	NA	22	22	NA	NA	17	17	NA	NA	17	17
2000	NA	NA	24	24	NA	NA	20	21	NA	NA	15	15	NA	NA	15	15
2001	NA	NA	27	27	NA	NA	21	21	NA	NA	14	14	NA	NA	14	14
2002	NA	NA	24	25	NA	NA	20	21	NA	NA	13	13	NA	NA	13	13
2003	NA	NA	21	22	NA	NA	18	19	NA	NA	12	13	NA	NA	12	13
2004	NA	NA	16	17	NA	NA	16	16	NA	NA	12	12	NA	NA	12	12
2005	NA	NA	16	17	NA	NA	14	15	NA	NA	11	11	NA	NA	11	11
2006	NA	NA	19	19	NA	NA	15	15	NA	NA	10	10	NA	NA	10	10
2007	NA	NA	16	17	NA	NA	15	15	NA	NA	9.5	9.6	NA	NA	9.5	9.6
2008	NA	NA	15	15	NA	NA	14	14	NA	NA	9.0	9.2	NA	NA	9.0	9.2
2009	NA	NA	13	14	NA	NA	12	12	NA	NA	8.4	8.6	NA	NA	8.4	8.6
2010	NA	NA	14	14	NA	NA	12	12	NA	NA	8.1	8.2	NA	NA	8.1	8.2
2011	NA	NA	17	17	NA	NA	13	13	NA	NA	7.9	8.1	NA	NA	7.9	8.1
2012	NA	NA	14	14	NA	NA	12	13	NA	NA	7.8	7.9	NA	NA	7.8	7.9
2013	NA	NA	16	17	NA	NA	13	13	NA	NA	7.7	7.8	NA	NA	7.7	7.8
2014	NA	NA	14	15	NA	NA	12	12	NA	NA	7.6	7.7	NA	NA	7.6	7.7
2015	NA	NA	13	13	NA	NA	12	12	NA	NA	7.4	7.6	NA	NA	7.4	7.6
2016	NA	NA	12	12	NA	NA	11	11	NA	NA	7.2	7.3	NA	NA	7.2	7.3
2017	NA	NA	11	11	NA	NA	9.9	10	NA	NA	6.9	7.0	NA	NA	6.9	7.0
2018	NA	NA	10	11	NA	NA	9.6	9.8	NA	NA	6.5	6.7	NA	NA	6.5	6.7

Bold values indicate exceedances

TABLE 5-39: RATIO OF MODELED DIETARY DOSE BASED ON FISHRAND FOR FEMALE BELTED KINGFISHER USING TEQ FOR THE PERIOD 1993 - 2018
REVISED

Year	LOAEL 152 Average	LOAEL 152 95% UCL	NOAEL 152 Average	NOAEL 152 95% UCL	LOAEL 113 Average	LOAEL 113 95% UCL	NOAEL 113 Average	NOAEL 113 95% UCL	LOAEL 90 Average	LOAEL 90 95% UCL	NOAEL 90 Average	NOAEL 90 95% UCL	LOAEL 50 Average	LOAEL 50 95% UCL	NOAEL 50 Average	NOAEL 50 95% UCL
1993	13	13	129	134	11	12	113	117	8.9	28	89	279	8.3	23	83	229
1994	9.8	10	98	101	10	10	102	105	8.0	26	80	256	7.4	21	74	213
1995	8.8	9.1	88	91	8.4	8.7	84	87	7.2	23	72	232	6.6	20	66	197
1996	11	11	110	114	8.9	9.2	89	92	6.7	23	67	231	6.0	19	60	187
1997	9.2	9.5	92	95	8.2	8.5	82	85	6.2	22	62	224	5.4	18	54	180
1998	7.3	7.5	73	75	7.2	7.4	72	74	5.8	21	58	208	5.0	17	50	170
1999	6.9	7.1	69	71	6.4	6.6	64	66	5.2	19	52	193	4.6	16	46	160
2000	6.3	6.5	63	65	6.2	6.4	62	64	4.8	19	48	188	4.2	15	42	153
2001	7.1	7.3	71	73	6.3	6.5	63	65	4.6	19	46	190	4.0	15	40	150
2002	6.4	6.7	64	67	5.9	6.1	59	61	4.5	18	45	183	3.8	15	38	146
2003	5.8	6.0	58	60	5.6	5.7	56	57	4.3	18	43	175	3.6	14	36	141
2004	4.8	5.0	48	50	4.9	5.1	49	51	3.9	16	39	164	3.4	13	34	134
2005	4.8	5.0	48	50	4.7	4.9	47	49	3.7	16	37	158	3.2	13	32	129
2006	5.3	5.5	53	55	4.8	4.9	48	49	3.6	16	36	159	3.0	13	30	126
2007	4.7	4.9	47	49	4.6	4.8	46	48	3.5	16	35	156	2.9	12	29	123
2008	4.4	4.6	44	46	4.4	4.6	44	46	3.4	15	34	149	2.8	12	28	120
2009	3.9	4.1	39	41	4.0	4.1	40	41	3.1	14	31	142	2.6	12	26	115
2010	4.5	4.6	45	46	4.1	4.2	41	42	3.1	14	31	143	2.6	11	26	114
2011	4.5	4.7	45	47	4.2	4.4	42	44	3.1	14	31	144	2.5	11	25	113
2012	4.4	4.6	44	46	4.2	4.3	42	43	3.1	14	31	141	2.5	11	25	111
2013	4.7	4.9	47	49	4.1	4.3	41	43	3.1	14	31	140	2.5	11	25	110
2014	4.3	4.4	43	44	4.1	4.2	41	42	3.0	14	30	137	2.4	11	24	108
2015	3.9	4.0	39	40	3.9	4.0	39	40	2.9	13	29	133	2.4	11	24	106
2016	3.5	3.7	35	37	3.6	3.7	36	37	2.8	13	28	127	2.3	10	23	102
2017	3.5	3.6	35	36	3.4	3.5	34	35	2.7	12	27	123	2.2	9.9	22	99
2018	3.6	3.7	36	37	3.4	3.5	34	35	2.6	12	26	121	2.1	9.7	21	97

Bold values indicate exceedances

TABLE 5-40: RATIO OF MODELED DIETARY DOSE BASED ON FISHRAND FOR FEMALE GREAT BLUE HERON USING TEQ FOR THE PERIOD 1993 - 2018
REVISED

Year	LOAEL 152 Average	LOAEL 152 95% UCL	NOAEL 152 Average	NOAEL 152 95% UCL	LOAEL 113 Average	LOAEL 113 95% UCL	NOAEL 113 Average	NOAEL 113 95% UCL	LOAEL 90 Average	LOAEL 90 95% UCL	NOAEL 90 Average	NOAEL 90 95% UCL	LOAEL 50 Average	LOAEL 50 95% UCL	NOAEL 50 Average	NOAEL 50 95% UCL
1993	**5.8**	**6.0**	**58**	**60**	**5.1**	**5.2**	**51**	**52**	**4.0**	**4.1**	**40**	**41**	**3.8**	**3.9**	**38**	**39**
1994	**4.2**	**4.4**	**42**	**44**	**4.6**	**4.7**	**46**	**47**	**3.6**	**3.7**	**36**	**37**	**3.4**	**3.5**	**34**	**35**
1995	**3.7**	**3.9**	**37**	**39**	**3.7**	**3.8**	**37**	**38**	**3.2**	**3.3**	**32**	**33**	**3.0**	**3.1**	**30**	**31**
1996	**4.9**	**5.0**	**49**	**50**	**4.0**	**4.1**	**40**	**41**	**2.9**	**3.0**	**29**	**30**	**2.7**	**2.8**	**27**	**28**
1997	**4.0**	**4.1**	**40**	**41**	**3.6**	**3.7**	**36**	**37**	**2.7**	**2.8**	**27**	**28**	**2.4**	**2.5**	**24**	**25**
1998	**3.0**	**3.1**	**30**	**31**	**3.1**	**3.2**	**31**	**32**	**2.5**	**2.6**	**25**	**26**	**2.2**	**2.3**	**22**	**23**
1999	**2.9**	**3.0**	**29**	**30**	**2.7**	**2.8**	**27**	**28**	**2.2**	**2.3**	**22**	**23**	**2.0**	**2.1**	**20**	**21**
2000	**2.6**	**2.7**	**26**	**27**	**2.7**	**2.7**	**27**	**27**	**2.0**	**2.1**	**20**	**21**	**1.9**	**1.9**	**19**	**19**
2001	**3.0**	**3.1**	**30**	**31**	**2.7**	**2.8**	**27**	**28**	**2.0**	**2.0**	**20**	**20**	**1.7**	**1.8**	**17**	**18**
2002	**2.7**	**2.8**	**27**	**28**	**2.5**	**2.6**	**25**	**26**	**1.9**	**2.0**	**19**	**20**	**1.7**	**1.7**	**17**	**17**
2003	**2.4**	**2.4**	**24**	**24**	**2.4**	**2.4**	**24**	**24**	**1.8**	**1.9**	**18**	**19**	**1.6**	**1.6**	**16**	**16**
2004	**1.9**	**1.9**	**19**	**19**	**2.0**	**2.1**	**20**	**21**	**1.7**	**1.7**	**17**	**17**	**1.5**	**1.5**	**15**	**15**
2005	**1.9**	**1.9**	**19**	**19**	**1.9**	**2.0**	**19**	**20**	**1.5**	**1.6**	**15**	**16**	**1.4**	**1.4**	**14**	**14**
2006	**2.1**	**2.2**	**21**	**22**	**2.0**	**2.0**	**20**	**20**	**1.5**	**1.5**	**15**	**15**	**1.3**	**1.3**	**13**	**13**
2007	**1.9**	**1.9**	**19**	**19**	**1.9**	**2.0**	**19**	**20**	**1.4**	**1.5**	**14**	**15**	**1.2**	**1.3**	**12**	**13**
2008	**1.7**	**1.8**	**17**	**18**	**1.8**	**1.9**	**18**	**19**	**1.4**	**1.4**	**14**	**14**	**1.2**	**1.2**	**12**	**12**
2009	**1.4**	**1.5**	**14**	**15**	**1.6**	**1.7**	**16**	**17**	**1.3**	**1.3**	**13**	**13**	**1.1**	**1.1**	**11**	**11**
2010	**1.8**	**1.8**	**18**	**18**	**1.6**	**1.7**	**16**	**17**	**1.3**	**1.3**	**13**	**13**	**1.1**	**1.1**	**11**	**11**
2011	**1.8**	**1.8**	**18**	**18**	**1.7**	**1.8**	**17**	**18**	**1.3**	**1.3**	**13**	**13**	**1.1**	**1.1**	**11**	**11**
2012	**1.7**	**1.8**	**17**	**18**	**1.7**	**1.8**	**17**	**18**	**1.3**	**1.3**	**13**	**13**	**1.0**	**1.1**	**10**	**11**
2013	**1.9**	**1.9**	**19**	**19**	**1.7**	**1.8**	**17**	**18**	**1.3**	**1.3**	**13**	**13**	**1.0**	**1.1**	**10**	**11**
2014	**1.7**	**1.7**	**17**	**17**	**1.7**	**1.7**	**17**	**17**	**1.2**	**1.3**	**12**	**13**	**1.0**	**1.0**	**10**	**10**
2015	**1.5**	**1.5**	**15**	**15**	**1.6**	**1.6**	**16**	**16**	**1.2**	**1.2**	**12**	**12**	**1.0**	**1.0**	**9.9**	**10**
2016	**1.3**	**1.3**	**13**	**13**	**1.4**	**1.5**	**14**	**15**	**1.1**	**1.2**	**11**	**12**	**1.0**	**1.0**	**9.5**	**9.7**
2017	**1.3**	**1.3**	**13**	**13**	**1.4**	**1.4**	**14**	**14**	**1.1**	**1.1**	**11**	**11**	0.9	0.9	**9.1**	**9.3**
2018	**1.3**	**1.4**	**13**	**14**	**1.3**	**1.4**	**13**	**14**	**1.0**	**1.1**	**10**	**11**	0.9	0.9	**8.8**	**9.0**

Bold values indicate exceedances

TABLE 5-41: RATIO OF MODELED DIETARY DOSE BASED ON FISHRAND FOR FEMALE BALD EAGLE USING TEQ FOR THE PERIOD 1993 - 2018
REVISED

Year	LOAEL 152 Average	LOAEL 152 95% UCL	NOAEL 152 Average	NOAEL 152 95% UCL	LOAEL 113 Average	LOAEL 113 95% UCL	NOAEL 113 Average	NOAEL 113 95% UCL	LOAEL 90 Average	LOAEL 90 95% UCL	NOAEL 90 Average	NOAEL 90 95% UCL	LOAEL 50 Average	LOAEL 50 95% UCL	NOAEL 50 Average	NOAEL 50 95% UCL
1993	19	19	188	191	15	15	146	148	11	11	112	114	11	11	110	112
1994	13	13	133	135	13	13	126	128	10	10	101	103	9.7	9.8	97	98
1995	11	11	111	113	10	11	104	106	8.9	9.1	89	91	8.5	8.7	85	87
1996	14	15	144	146	11	11	107	108	8.0	8.1	80	81	7.6	7.7	76	77
1997	12	13	124	126	9.9	10	99	101	7.5	7.6	75	76	6.9	7.0	69	70
1998	9.9	10	99	101	8.6	8.7	86	87	6.8	6.9	68	69	6.3	6.4	63	64
1999	8.4	8.5	84	85	7.4	7.6	74	76	6.1	6.2	61	62	5.7	5.7	57	57
2000	8.1	8.2	81	82	6.9	7.0	69	70	5.4	5.5	54	55	5.1	5.2	51	52
2001	9.0	9.1	90	91	7.1	7.3	71	73	5.2	5.3	52	53	4.7	4.8	47	48
2002	8.3	8.4	83	84	6.9	7.0	69	70	5.0	5.1	50	51	4.5	4.5	45	45
2003	7.3	7.4	73	74	6.2	6.3	62	63	4.8	4.8	48	48	4.2	4.3	42	43
2004	5.6	5.7	56	57	5.5	5.6	55	56	4.4	4.5	44	45	4.0	4.0	40	40
2005	5.5	5.6	55	56	4.9	5.0	49	50	4.0	4.1	40	41	3.6	3.7	36	37
2006	6.3	6.4	63	64	5.1	5.2	51	52	3.7	3.8	37	38	3.4	3.4	34	34
2007	5.5	5.6	55	56	5.0	5.0	50	50	3.6	3.7	36	37	3.2	3.3	32	33
2008	5.1	5.2	51	52	4.7	4.8	47	48	3.5	3.6	35	36	3.1	3.1	31	31
2009	4.5	4.6	45	46	4.1	4.2	41	42	3.3	3.3	33	33	2.9	2.9	29	29
2010	4.8	4.9	48	49	4.1	4.2	41	42	3.1	3.2	31	32	2.7	2.8	27	28
2011	5.6	5.7	56	57	4.5	4.5	45	45	3.2	3.2	32	32	2.7	2.7	27	27
2012	4.8	4.9	48	49	4.2	4.3	42	43	3.1	3.2	31	32	2.6	2.7	26	27
2013	5.5	5.6	55	56	4.4	4.5	44	45	3.1	3.2	31	32	2.6	2.7	26	27
2014	4.9	4.9	49	49	4.2	4.2	42	42	3.1	3.1	31	31	2.6	2.6	26	26
2015	4.5	4.5	45	45	3.9	4.0	39	40	3.0	3.0	30	30	2.5	2.6	25	26
2016	4.0	4.1	40	41	3.7	3.8	37	38	2.8	2.9	28	29	2.4	2.5	24	25
2017	3.6	3.7	36	37	3.4	3.4	34	34	2.7	2.7	27	27	2.3	2.4	23	24
2018	3.5	3.6	35	36	3.3	3.3	33	33	2.5	2.6	25	26	2.2	2.3	22	23

Bold values indicate exceedances

TABLE 5-42: RATIO OF MODELED EGG CONCENTRATIONS BASED ON FISHRAND FOR FEMALE BELTED KINGFISHER USING TEQ FOR THE PERIOD 1993 - 2018
REVISED

Year	LOAEL 152 Average	LOAEL 152 95% UCL	NOAEL 152 Average	NOAEL 152 95% UCL	LOAEL 113 Average	LOAEL 113 95% UCL	NOAEL 113 Average	NOAEL 113 95% UCL	LOAEL 90 Average	LOAEL 90 95% UCL	NOAEL 90 Average	NOAEL 90 95% UCL	LOAEL 50 Average	LOAEL 50 95% UCL	NOAEL 50 Average	NOAEL 50 95% UCL
1993	**412**	**430**	**825**	**860**	**362**	**375**	**724**	**751**	**268**	**278**	**537**	**556**	**268**	**278**	**537**	**556**
1994	**308**	**321**	**616**	**641**	**326**	**337**	**652**	**674**	**239**	**247**	**477**	**494**	**239**	**247**	**477**	**494**
1995	**277**	**288**	**553**	**575**	**269**	**279**	**538**	**558**	**213**	**220**	**425**	**440**	**213**	**220**	**425**	**440**
1996	**350**	**365**	**700**	**729**	**286**	**296**	**572**	**592**	**192**	**199**	**384**	**397**	**192**	**199**	**384**	**397**
1997	**289**	**302**	**579**	**604**	**261**	**271**	**522**	**542**	**175**	**181**	**350**	**362**	**175**	**181**	**350**	**362**
1998	**227**	**236**	**453**	**472**	**227**	**236**	**455**	**471**	**161**	**166**	**322**	**333**	**161**	**166**	**322**	**333**
1999	**215**	**224**	**430**	**448**	**201**	**208**	**402**	**416**	**147**	**152**	**294**	**304**	**147**	**152**	**294**	**304**
2000	**196**	**204**	**391**	**407**	**197**	**203**	**393**	**406**	**134**	**139**	**269**	**278**	**134**	**139**	**269**	**278**
2001	**222**	**230**	**443**	**461**	**198**	**205**	**396**	**409**	**126**	**130**	**252**	**260**	**126**	**130**	**252**	**260**
2002	**200**	**208**	**400**	**416**	**187**	**194**	**375**	**388**	**120**	**124**	**241**	**249**	**120**	**124**	**241**	**249**
2003	**179**	**187**	**358**	**373**	**175**	**181**	**350**	**362**	**114**	**118**	**229**	**236**	**114**	**118**	**229**	**236**
2004	**147**	**153**	**294**	**307**	**154**	**159**	**308**	**319**	**106**	**110**	**213**	**220**	**106**	**110**	**213**	**220**
2005	**147**	**153**	**294**	**306**	**147**	**152**	**294**	**304**	**100**	**103**	**200**	**207**	**100**	**103**	**200**	**207**
2006	**164**	**171**	**328**	**341**	**149**	**154**	**298**	**308**	**95**	**98**	**190**	**197**	**95**	**98**	**190**	**197**
2007	**145**	**151**	**290**	**303**	**144**	**149**	**288**	**298**	**91**	**94**	**182**	**189**	**91**	**94**	**182**	**189**
2008	**136**	**142**	**271**	**283**	**137**	**142**	**275**	**285**	**88**	**91**	**176**	**182**	**88**	**91**	**176**	**182**
2009	**118**	**123**	**236**	**246**	**124**	**129**	**248**	**257**	**82**	**85**	**164**	**170**	**82**	**85**	**164**	**170**
2010	**137**	**143**	**275**	**286**	**126**	**131**	**252**	**261**	**81**	**84**	**162**	**168**	**81**	**84**	**162**	**168**
2011	**139**	**145**	**278**	**290**	**132**	**137**	**265**	**274**	**80**	**82**	**159**	**165**	**80**	**82**	**159**	**165**
2012	**136**	**142**	**272**	**284**	**130**	**135**	**260**	**270**	**78**	**81**	**157**	**162**	**78**	**81**	**157**	**162**
2013	**144**	**151**	**289**	**301**	**129**	**134**	**259**	**268**	**78**	**80**	**155**	**160**	**78**	**80**	**155**	**160**
2014	**131**	**137**	**262**	**274**	**127**	**132**	**254**	**264**	**76**	**79**	**153**	**158**	**76**	**79**	**153**	**158**
2015	**117**	**123**	**235**	**245**	**121**	**125**	**241**	**250**	**75**	**77**	**149**	**155**	**75**	**77**	**149**	**155**
2016	**106**	**111**	**212**	**222**	**111**	**115**	**221**	**230**	**72**	**75**	**144**	**149**	**72**	**75**	**144**	**149**
2017	**106**	**111**	**212**	**222**	**106**	**110**	**211**	**219**	**69**	**72**	**139**	**144**	**69**	**72**	**139**	**144**
2018	**109**	**115**	**219**	**229**	**105**	**109**	**210**	**219**	**67**	**69**	**134**	**139**	**67**	**69**	**134**	**139**

Bold values indicate exceedances

TABLE 5-43: RATIO OF MODELED EGG CONCENTRATIONS BASED ON FISHRAND FOR FEMALE GREAT BLUE HERON USING TEQ FOR THE PERIOD 1993 - 2018
REVISED

Year	LOAEL 152 Average	LOAEL 152 95% UCL	NOAEL 152 Average	NOAEL 152 95% UCL	LOAEL 113 Average	LOAEL 113 95% UCL	NOAEL 113 Average	NOAEL 113 95% UCL	LOAEL 90 Average	LOAEL 90 95% UCL	NOAEL 90 Average	NOAEL 90 95% UCL	LOAEL 50 Average	LOAEL 50 95% UCL	NOAEL 50 Average	NOAEL 50 95% UCL
1993	19	20	32	33	17	18	28	29	13	14	22	23	13	13	21	22
1994	14	14	23	24	15	16	25	26	12	12	20	21	11	12	19	20
1995	12	13	21	21	12	13	21	21	11	11	18	18	10	10	17	17
1996	16	17	27	28	13	14	22	23	10	10	16	17	9.0	9.3	15	15
1997	13	14	22	23	12	12	20	21	9	9.3	15	16	8.2	8.4	14	14
1998	9.9	10	17	17	10	11	17	18	8	8.5	14	14	7.5	7.7	12	13
1999	9.4	9.8	16	16	9.0	9.3	15	15	7	7.6	12	13	6.8	7.0	11	12
2000	8.4	8.7	14	14	8.8	9.1	15	15	6.7	6.9	11	12	6.1	6.3	10	11
2001	9.7	10	16	17	8.9	9.2	15	15	6.5	6.7	11	11	5.7	5.9	9.5	9.8
2002	8.7	9.0	14	15	8.4	8.6	14	14	6.3	6.5	10	11	5.5	5.6	9.1	9.4
2003	7.7	8.0	13	13	7.8	8.0	13	13	5.9	6.1	9.9	10	5.2	5.3	8.6	8.9
2004	6.0	6.3	10	10	6.7	6.9	11	11	5.4	5.6	9.0	9.3	4.8	4.9	8.0	8.2
2005	6.1	6.3	10	10	6.4	6.5	11	11	5.0	5.2	8.4	8.6	4.5	4.6	7.4	7.6
2006	7.0	7.2	.12	12	6.5	6.7	11	11	4.9	5.0	8.1	8.3	4.2	4.3	7.0	7.2
2007	6.0	6.2	10	10	6.3	6.4	10	11	4.7	4.9	7.9	8.1	4.0	4.1	6.7	6.9
2008	5.5	5.7	9.2	9.5	5.9	6.1	9.9	10	4.5	4.6	7.5	7.7	3.9	4.0	6.4	6.6
2009	4.6	4.8	7.7	8.0	5.2	5.4	8.7	9.0	4.2	4.3	6.9	7.1	3.6	3.7	6.0	6.1
2010	5.7	5.9	9.4	9.8	5.4	5.5	8.9	9.2	4.1	4.2	6.9	7.1	3.5	3.6	5.9	6.0
2011	5.8	6.0	9.6	10	5.7	5.9	9.5	9.8	4.2	4.3	6.9	7.1	3.5	3.6	5.8	5.9
2012	5.6	5.8	9.4	9.7	5.6	5.8	9.3	9.6	4.1	4.2	6.8	7.0	3.4	3.5	5.7	5.8
2013	6.1	6.3	10	11	5.6	5.7	9.3	9.6	4.1	4.2	6.8	7.0	3.4	3.5	5.6	5.8
2014	5.4	5.6	9.0	9.4	5.5	5.6	9.1	9.4	4.0	4.1	6.7	6.9	3.3	3.4	5.5	5.7
2015	4.7	4.9	7.9	8.2	5.2	5.3	8.6	8.8	3.9	4.0	6.5	6.7	3.2	3.3	5.4	5.6
2016	4.1	4.3	6.9	7.1	4.7	4.8	7.8	8.0	3.7	3.8	6.1	6.3	3.1	3.2	5.2	5.3
2017	4.1	4.3	6.9	7.1	4.4	4.5	7.3	7.5	3.5	3.6	5.8	6.0	3.0	3.1	5.0	5.1
2018	4.3	4.5	7.2	7.4	4.4	4.5	7.3	7.5	3.4	3.5	5.6	5.8	2.9	2.9	4.8	4.9

Bold values indicate exceedances

TABLE 5-44: RATIO OF MODELED EGG CONCENTRATIONS BASED ON FISHRAND FOR FEMALE BALD EAGLE USING TEQ FOR THE PERIOD 1993 - 2018
REVISED

Year	LOAEL 152 Average	LOAEL 152 95% UCL	NOAEL 152 Average	NOAEL 152 95% UCL	LOAEL 113 Average	LOAEL 113 95% UCL	NOAEL 113 Average	NOAEL 113 95% UCL	LOAEL 90 Average	LOAEL 90 95% UCL	NOAEL 90 Average	NOAEL 90 95% UCL	LOAEL 50 Average	LOAEL 50 95% UCL	NOAEL 50 Average	NOAEL 50 95% UCL
1993	NA	NA	3924	3993	NA	NA	3046	3098	NA	NA	2338	2376	NA	NA	2291	2330
1994	NA	NA	2770	2812	NA	NA	2619	2664	NA	NA	2110	2145	NA	NA	2016	2050
1995	NA	NA	2319	2360	NA	NA	2176	2216	NA	NA	1862	1894	NA	NA	1781	1811
1996	NA	NA	3010	3050	NA	NA	2223	2258	NA	NA	1666	1694	NA	NA	1588	1614
1997	NA	NA	2578	2623	NA	NA	2072	2108	NA	NA	1560	1585	NA	NA	1439	1463
1998	NA	NA	1767	1776	NA	NA	1791	1822	NA	NA	1420	1444	NA	NA	1308	1330
1999	NA	NA	1595	1553	NA	NA	1553	1580	NA	NA	1266	1287	NA	NA	1180	1199
2000	NA	NA	1555	1516	NA	NA	1432	1457	NA	NA	1133	1152	NA	NA	1067	1085
2001	NA	NA	1722	1681	NA	NA	1491	1515	NA	NA	1081	1099	NA	NA	982	998
2002	NA	NA	1596	1549	NA	NA	1443	1467	NA	NA	1050	1067	NA	NA	929	944
2003	NA	NA	1396	1362	NA	NA	1298	1320	NA	NA	992	1009	NA	NA	881	895
2004	NA	NA	1066	1036	NA	NA	1140	1159	NA	NA	920	935	NA	NA	825	839
2005	NA	NA	1089	1055	NA	NA	1026	1043	NA	NA	832	846	NA	NA	760	773
2006	NA	NA	1240	1207	NA	NA	1058	1075	NA	NA	782	795	NA	NA	705	716
2007	NA	NA	1100	1064	NA	NA	1034	1051	NA	NA	760	773	NA	NA	669	680
2008	NA	NA	980	950	NA	NA	976	993	NA	NA	733	745	NA	NA	639	649
2009	NA	NA	870	848	NA	NA	862	877	NA	NA	687	699	NA	NA	598	608
2010	NA	NA	941	916	NA	NA	858	872	NA	NA	655	666	NA	NA	572	581
2011	NA	NA	1093	1064	NA	NA	930	945	NA	NA	659	671	NA	NA	562	572
2012	NA	NA	987	951	NA	NA	885	900	NA	NA	653	664	NA	NA	551	561
2013	NA	NA	1097	1059	NA	NA	914	929	NA	NA	651	663	NA	NA	546	555
2014	NA	NA	1031	988	NA	NA	867	882	NA	NA	637	648	NA	NA	537	546
2015	NA	NA	876	848	NA	NA	821	835	NA	NA	618	629	NA	NA	526	535
2016	NA	NA	792	765	NA	NA	774	787	NA	NA	594	604	NA	NA	508	517
2017	NA	NA	777	742	NA	NA	704	716	NA	NA	557	566	NA	NA	485	493
2018	NA	NA	797	757	NA	NA	680	692	NA	NA	530	539	NA	NA	464	471

Bold values indicate exceedances

TABLE 5-45: RATIO OF MODELED DIETARY DOSES TO TOXICITY BENCHMARKS FOR FEMALE BAT FOR TRI+ CONGENERS FOR THE PERIOD 1993 - 2018

REVISED

Year	LOAEL 152 Average	LOAEL 152 95% UCL	NOAEL 152 Average	NOAEL 152 95% UCL	LOAEL 113 Average	LOAEL 113 95% UCL	NOAEL 113 Average	NOAEL 113 95% UCL	LOAEL 90 Average	LOAEL 90 95% UCL	NOAEL 90 Average	NOAEL 90 95%UCL	LOAEL 50 Average	LOAEL 50 95% UCL	NOAEL 50 Average	NOAEL 50 95% UCL
1993	3.6	3.8	17	18	2.8	3.0	13	14	2.3	2.4	11	11	17	18	7.8	8.3
1994	3.3	3.5	16	16	2.7	2.8	13	13	2.2	2.3	10	11	16	17	7.5	8.0
1995	3.2	3.4	15	16	2.6	2.7	12	13	2.1	2.2	9.7	10	15	16	7.2	7.6
1996	3.2	3.4	15	16	2.5	2.6	12	12	2.0	2.1	9.3	9.8	15	16	6.9	7.3
1997	3.0	3.2	14	15	2.4	2.6	11	12	1.9	2.0	9.0	9.5	14	15	6.6	7.0
1998	2.9	3.0	13	14	2.3	2.4	11	11	1.8	2.0	8.7	9.2	14	14	6.4	6.8
1999	2.7	2.9	13	14	2.2	2.4	10	11	1.8	1.9	8.3	8.8	13	14	6.2	6.6
2000	2.8	2.9	13	14	2.2	2.3	10	11	1.7	1.8	8.0	8.5	13	14	6.0	6.4
2001	2.7	2.9	13	14	2.2	2.3	10	11	1.7	1.8	8.0	8.4	13	13	5.9	6.2
2002	2.6	2.8	12	13	2.1	2.2	9.8	10	1.6	1.7	7.7	8.2	12	13	5.7	6.0
2003	2.5	2.7	12	13	2.0	2.1	9.4	10	1.6	1.7	7.4	7.9	12	13	5.6	5.9
2004	2.5	2.7	12	12	2.0	2.1	9.4	10	1.5	1.6	7.2	7.7	11	12	5.3	5.7
2005	2.5	2.6	12	12	2.0	2.1	9.3	10	1.5	1.6	7.2	7.7	11	12	5.4	5.7
2006	2.4	2.6	11	12	1.9	2.1	9.1	9.8	1.5	1.6	7.0	7.5	11	12	5.3	5.6
2007	2.4	2.6	11	12	1.9	2.1	9.0	9.6	1.5	1.6	6.9	7.4	11	12	5.2	5.5
2008	2.4	2.5	11	12	1.9	2.0	8.9	9.5	1.5	1.6	6.8	7.3	11	12	5.1	5.5
2009	2.3	2.5	11	12	1.9	2.0	8.8	9.4	1.4	1.5	6.7	7.2	11	12	5.0	5.4
2010	2.3	2.5	11	12	1.8	2.0	8.7	9.3	1.4	1.5	6.7	7.2	11	11	5.0	5.4
2011	2.3	2.4	11	11	1.8	2.0	8.5	9.2	1.4	1.5	6.6	7.0	10	11	4.9	5.3
2012	2.2	2.4	10	11	1.8	1.9	8.4	9.0	1.4	1.5	6.5	7.0	10	11	4.8	5.2
2013	2.2	2.4	10	11	1.8	1.9	8.3	8.9	1.4	1.5	6.4	6.9	10	11	4.8	5.1
2014	2.2	2.3	10	11	1.7	1.9	8.2	8.8	1.3	1.4	6.3	6.8	10	11	4.7	5.0
2015	2.1	2.3	10	11	1.7	1.8	8.1	8.6	1.3	1.4	6.2	6.7	9.9	11	4.6	5.0
2016	2.2	2.3	10	11	1.7	1.8	7.9	8.5	1.3	1.4	6.1	6.5	9.7	10	4.6	4.9
2017	2.2	2.3	10	11	1.7	1.8	7.9	8.5	1.3	1.4	6.0	6.5	9.6	10	4.5	4.8
2018	2.2	2.3	10	11	1.7	1.9	8.0	8.7	1.3	1.4	5.9	6.3	9.4	10	4.4	4.8

Bold values indicate exceedances

TABLE 5-46: RATIO OF MODELED DIETARY DOSES TO TOXICITY BENCHMARKS FOR FEMALE BAT ON A TEQ BASIS FOR THE PERIOD 1993 - 2018
REVISED

Year	LOAEL 152 Average	LOAEL 152 95% UCL	NOAEL 152 Average	NOAEL 152 95% UCL	LOAEL 113 Average	LOAEL 113 95% UCL	NOAEL 113 Average	NOAEL 113 95% UCL	LOAEL 90 Average	LOAEL 90 95% UCL	NOAEL 90 Average	NOAEL 90 95%UCL	LOAEL 50 Average	LOAEL 50 95% UCL	NOAEL 50 Average	NOAEL 50 95% UCL
1993	58	61	581	612	46	48	459	484	37	39	367	387	27	29	271	285
1994	54	57	536	565	44	46	435	459	35	37	351	370	26	27	260	275
1995	51	54	514	543	42	44	416	439	34	35	336	354	25	26	248	262
1996	52	54	515	544	40	42	401	424	32	34	322	340	24	25	240	253
1997	49	52	490	518	39	41	391	413	31	33	310	328	23	24	228	241
1998	46	49	463	491	37	40	373	395	30	32	299	316	22	23	221	234
1999	44	47	442	470	36	38	361	383	29	30	288	305	22	23	215	228
2000	45	48	449	476	35	37	352	374	28	29	278	294	21	22	209	221
2001	44	47	443	470	35	37	349	370	27	29	275	291	20	22	203	215
2002	43	45	427	453	34	36	338	359	27	28	267	283	20	21	196	208
2003	41	43	405	432	33	35	326	347	26	27	257	274	19	20	192	204
2004	40	43	402	431	32	35	325	347	25	27	249	265	18	20	185	197
2005	40	43	397	426	32	34	321	344	25	26	248	265	19	20	186	198
2006	39	42	394	422	31	34	315	337	24	26	243	260	18	19	182	194
2007	39	42	388	415	31	33	311	333	24	26	240	257	18	19	179	191
2008	38	41	383	411	31	33	307	330	24	25	236	252	18	19	176	189
2009	38	40	377	404	30	32	302	324	23	25	233	250	17	19	174	187
2010	37	40	372	398	30	32	299	321	23	25	231	247	17	19	172	185
2011	37	39	368	394	29	32	294	316	23	24	227	243	17	18	169	181
2012	36	39	362	388	29	31	291	312	22	24	224	240	17	18	167	179
2013	36	38	356	382	29	31	287	307	22	24	221	237	16	18	164	177
2014	35	38	351	376	28	30	283	303	22	23	218	234	16	17	162	174
2015	35	37	346	372	28	30	278	299	21	23	214	230	16	17	160	172
2016	35	38	348	376	27	29	274	295	21	23	211	226	16	17	157	169
2017	35	38	348	377	27	30	274	295	21	22	208	223	16	17	155	167
2018	35	38	350	379	28	30	277	299	20	22	204	219	15	16	153	165

Bold values indicate exceedances

TABLE 5-47: RATIO OF MODELED DIETARY DOSES TO TOXICITY BENCHMARKS FOR FEMALE RACCOON FOR TRI+ CONGENERS FOR THE PERIOD 1993 - 2018
REVISED

Year	LOAEL 152 Average	LOAEL 152 95% UCL	NOAEL 152 Average	NOAEL 152 95% UCL	LOAEL 113 Average	LOAEL 113 95% UCL	NOAEL 113 Average	NOAEL 113 95% UCL	LOAEL 90 Average	LOAEL 90 95% UCL	NOAEL 90 Average	NOAEL 90 95% UCL	LOAEL 50 Average	LOAEL 50 95% UCL	NOAEL 50 Average	NOAEL 50 95% UCL
1993	0.7	0.7	**3.3**	**3.4**	0.6	0.6	**2.6**	**2.6**	0.4	0.5	**2.1**	**2.2**	0.3	0.4	**1.6**	**1.7**
1994	0.6	0.7	**2.9**	**3.1**	0.5	0.5	**2.5**	**2.5**	0.4	0.4	**2.0**	**2.1**	0.3	0.3	**1.5**	**1.6**
1995	0.6	0.6	**2.8**	**2.9**	0.5	0.5	**2.3**	**2.3**	0.4	0.4	**1.9**	**2.0**	0.3	0.3	**1.4**	**1.5**
1996	0.6	0.6	**2.9**	**3.0**	0.5	0.5	**2.3**	**2.3**	0.4	0.4	**1.8**	**1.9**	0.3	0.3	**1.4**	**1.4**
1997	0.6	0.6	**2.7**	**2.8**	0.5	0.5	**2.2**	**2.2**	0.4	0.4	**1.7**	**1.8**	0.3	0.3	**1.3**	**1.4**
1998	0.5	0.6	**2.5**	**2.6**	0.4	0.5	**2.1**	**2.2**	0.4	0.4	**1.7**	**1.7**	0.3	0.3	**1.2**	**1.3**
1999	0.5	0.5	**2.4**	**2.5**	0.4	0.4	**2.0**	**2.1**	0.3	0.4	**1.6**	**1.7**	0.3	0.3	**1.2**	**1.3**
2000	0.5	0.5	**2.4**	**2.5**	0.4	0.4	**1.9**	**2.0**	0.3	0.3	**1.5**	**1.6**	0.2	0.3	**1.2**	**1.2**
2001	0.5	0.5	**2.4**	**2.5**	0.4	0.4	**1.9**	**2.0**	0.3	0.3	**1.5**	**1.6**	0.2	0.3	**1.1**	**1.2**
2002	0.5	0.5	**2.3**	**2.4**	0.4	0.4	**1.8**	**2.0**	0.3	0.3	**1.5**	**1.5**	0.2	0.2	**1.1**	**1.1**
2003	0.5	0.5	**2.2**	**2.3**	0.4	0.4	**1.8**	**1.9**	0.3	0.3	**1.4**	**1.5**	0.2	0.2	**1.1**	**1.1**
2004	0.5	0.5	**2.1**	**2.3**	0.4	0.4	**1.7**	**1.9**	0.3	0.3	**1.3**	**1.4**	0.2	0.2	**1.0**	**1.1**
2005	0.4	0.5	**2.1**	**2.2**	0.4	0.4	**1.7**	**1.8**	0.3	0.3	**1.3**	**1.4**	0.2	0.2	**1.0**	**1.1**
2006	0.4	0.5	**2.1**	**2.2**	0.4	0.4	**1.7**	**1.8**	0.3	0.3	**1.3**	**1.4**	0.2	0.2	**1.0**	**1.0**
2007	0.4	0.5	**2.0**	**2.2**	0.4	0.4	**1.7**	**1.8**	0.3	0.3	**1.3**	**1.4**	0.2	0.2	**1.0**	**1.0**
2008	0.4	0.5	**2.0**	**2.1**	0.4	0.4	**1.6**	**1.8**	0.3	0.3	**1.3**	**1.3**	0.2	0.2	**1.0**	**1.0**
2009	0.4	0.4	**2.0**	**2.1**	0.3	0.4	**1.6**	**1.7**	0.3	0.3	**1.2**	**1.3**	0.2	0.2	0.9	**1.0**
2010	0.4	0.4	**2.0**	**2.1**	0.3	0.4	**1.6**	**1.7**	0.3	0.3	**1.2**	**1.3**	0.2	0.2	0.9	**1.0**
2011	0.4	0.4	**1.9**	**2.1**	0.3	0.4	**1.6**	**1.7**	0.3	0.3	**1.2**	**1.3**	0.2	0.2	0.9	**1.0**
2012	0.4	0.4	**1.9**	**2.0**	0.3	0.4	**1.6**	**1.7**	0.3	0.3	**1.2**	**1.3**	0.2	0.2	0.9	**1.0**
2013	0.4	0.4	**1.9**	**2.0**	0.3	0.3	**1.5**	**1.6**	0.3	0.3	**1.2**	**1.3**	0.2	0.2	0.9	0.9
2014	0.4	0.4	**1.9**	**2.0**	0.3	0.3	**1.5**	**1.6**	0.2	0.3	**1.2**	**1.2**	0.2	0.2	0.9	0.9
2015	0.4	0.4	**1.8**	**1.9**	0.3	0.3	**1.5**	**1.6**	0.2	0.3	**1.1**	**1.2**	0.2	0.2	0.9	0.9
2016	0.4	0.4	**1.8**	**1.9**	0.3	0.3	**1.5**	**1.6**	0.2	0.3	**1.1**	**1.2**	0.2	0.2	0.8	0.9
2017	0.4	0.4	**1.8**	**1.9**	0.3	0.3	**1.4**	**1.5**	0.2	0.3	**1.1**	**1.2**	0.2	0.2	0.8	0.9
2018	0.4	0.4	**1.8**	**2.0**	0.3	0.3	**1.5**	**1.6**	0.2	0.2	**1.1**	**1.2**	0.2	0.2	0.8	0.9

Bold values indicate exceedances

TABLE 5-48: RATIO OF MODELED DIETARY DOSES TO TOXICITY BENCHMARKS FOR FEMALE RACCOON ON A TEQ BASIS FOR THE PERIOD 1993 - 2018

REVISED

Year	LOAEL 152 Average	LOAEL 152 95% UCL	NOAEL 152 Average	NOAEL 152 95% UCL	LOAEL 113 Average	LOAEL 113 95% UCL	NOAEL 113 Average	NOAEL 113 95% UCL	LOAEL 90 Average	LOAEL 90 95% UCL	NOAEL 90 Average	NOAEL 90 95% UCL	LOAEL 50 Average	LOAEL 50 95% UCL	NOAEL 50 Average	NOAEL 50 95% UCL
1993	**12**	**13**	**121**	**127**	**9.7**	**9.7**	**97**	**97**	**7.7**	**8.1**	**77**	**81**	**5.9**	**6.2**	**59**	**62**
1994	**11**	**11**	**109**	**114**	**9.1**	**9.2**	**91**	**92**	**7.3**	**7.7**	**73**	**77**	**5.6**	**5.9**	**56**	**59**
1995	**10**	**11**	**103**	**109**	**8.5**	**8.6**	**85**	**86**	**6.9**	**7.3**	**69**	**73**	**5.3**	**5.5**	**53**	**55**
1996	**11**	**11**	**107**	**112**	**8.4**	**8.4**	**84**	**84**	**6.6**	**6.9**	**66**	**69**	**5.1**	**5.3**	**51**	**53**
1997	**10**	**11**	**100**	**105**	**8.1**	**8.1**	**81**	**81**	**6.4**	**6.7**	**64**	**67**	**4.8**	**5.0**	**48**	**50**
1998	**9.3**	**9.8**	**93**	**98**	**7.6**	**8.0**	**76**	**80**	**6.1**	**6.4**	**61**	**64**	**4.6**	**4.8**	**46**	**48**
1999	**8.8**	**9.3**	**88**	**93**	**7.3**	**7.7**	**73**	**77**	**5.8**	**6.1**	**58**	**61**	**4.4**	**4.7**	**44**	**47**
2000	**8.9**	**9.3**	**89**	**93**	**7.1**	**7.5**	**71**	**75**	**5.6**	**5.9**	**56**	**59**	**4.3**	**4.5**	**43**	**45**
2001	**8.9**	**9.4**	**89**	**94**	**7.1**	**7.4**	**71**	**74**	**5.5**	**5.8**	**55**	**58**	**4.1**	**4.3**	**41**	**43**
2002	**8.5**	**9.0**	**85**	**90**	**6.8**	**7.2**	**68**	**72**	**5.4**	**5.6**	**54**	**56**	**4.0**	**4.2**	**40**	**42**
2003	**8.0**	**8.5**	**80**	**85**	**6.6**	**6.9**	**66**	**69**	**5.2**	**5.4**	**52**	**54**	**3.9**	**4.1**	**39**	**41**
2004	**7.8**	**8.3**	**78**	**83**	**6.5**	**6.8**	**65**	**68**	**5.0**	**5.3**	**50**	**53**	**3.7**	**3.9**	**37**	**39**
2005	**7.7**	**8.2**	**77**	**82**	**6.3**	**6.7**	**63**	**67**	**4.9**	**5.2**	**49**	**52**	**3.7**	**3.9**	**37**	**39**
2006	**7.7**	**8.2**	**77**	**82**	**6.2**	**6.6**	**62**	**66**	**4.8**	**5.1**	**48**	**51**	**3.6**	**3.8**	**36**	**38**
2007	**7.6**	**8.0**	**76**	**80**	**6.1**	**6.5**	**61**	**65**	**4.7**	**5.0**	**47**	**50**	**3.6**	**3.8**	**36**	**38**
2008	**7.4**	**7.9**	**74**	**79**	**6.1**	**6.4**	**61**	**64**	**4.6**	**4.9**	**46**	**49**	**3.5**	**3.7**	**35**	**37**
2009	**7.2**	**7.7**	**72**	**77**	**5.9**	**6.3**	**59**	**63**	**4.6**	**4.8**	**46**	**48**	**3.4**	**3.6**	**34**	**36**
2010	**7.2**	**7.7**	**72**	**77**	**5.9**	**6.2**	**59**	**62**	**4.5**	**4.8**	**45**	**48**	**3.4**	**3.6**	**34**	**36**
2011	**7.2**	**7.6**	**72**	**76**	**5.8**	**6.2**	**58**	**62**	**4.4**	**4.7**	**44**	**47**	**3.3**	**3.5**	**33**	**35**
2012	**7.1**	**7.5**	**71**	**75**	**5.7**	**6.1**	**57**	**61**	**4.4**	**4.7**	**44**	**47**	**3.3**	**3.5**	**33**	**35**
2013	**7.0**	**7.4**	**70**	**74**	**5.6**	**6.0**	**56**	**60**	**4.3**	**4.6**	**43**	**46**	**3.2**	**3.4**	**32**	**34**
2014	**6.8**	**7.2**	**68**	**72**	**5.6**	**5.9**	**56**	**59**	**4.3**	**4.5**	**43**	**45**	**3.2**	**3.4**	**32**	**34**
2015	**6.7**	**7.1**	**67**	**71**	**5.5**	**5.8**	**55**	**58**	**4.2**	**4.5**	**42**	**45**	**3.1**	**3.3**	**31**	**33**
2016	**6.6**	**7.1**	**66**	**71**	**5.3**	**5.7**	**53**	**57**	**4.1**	**4.4**	**41**	**44**	**3.1**	**3.3**	**31**	**33**
2017	**6.6**	**7.1**	**66**	**71**	**5.3**	**5.7**	**53**	**57**	**4.0**	**4.3**	**40**	**43**	**3.0**	**3.2**	**30**	**32**
2018	**6.6**	**7.1**	**66**	**71**	**5.3**	**5.7**	**53**	**57**	**4.0**	**4.2**	**40**	**42**	**3.0**	**3.2**	**30**	**32**

Bold values indicate exceedances

TABLE 5-49: RATIO OF MODELED DIETARY DOSES TO TOXICITY BENCHMARKS FOR FEMALE MINK FOR TRI+ CONGENERS FOR THE PERIOD 1993 - 2018
REVISED

Year	LOAEL 152 Average	LOAEL 152 95% UCL	NOAEL 152 Average	NOAEL 152 95% UCL	LOAEL 113 Average	LOAEL 113 95% UCL	NOAEL 113 Average	NOAEL 113 95% UCL	LOAEL 90 Average	LOAEL 90 95% UCL	NOAEL 90 Average	NOAEL 90 95% UCL	LOAEL 50 Average	LOAEL 50 95% UCL	NOAEL 50 Average	NOAEL 50 95% UCL
1993	**1.4**	**1.5**	**46**	**48**	**1.2**	**1.3**	**40**	**41**	**1.0**	**1.0**	**31**	**33**	0.9	0.9	**29**	**30**
1994	**1.1**	**1.1**	**35**	**37**	**1.1**	**1.2**	**36**	**37**	0.9	0.9	**29**	**30**	0.8	0.8	**26**	**27**
1995	**1.0**	**1.0**	**32**	**34**	0.9	**1.0**	**30**	**32**	0.8	0.8	**26**	**27**	0.7	0.7	**23**	**24**
1996	**1.2**	**1.3**	**39**	**41**	**1.0**	**1.0**	**32**	**33**	0.7	0.8	**24**	**25**	0.6	0.7	**21**	**22**
1997	**1.0**	**1.1**	**33**	**35**	0.9	0.9	**29**	**31**	0.7	0.7	**22**	**23**	0.6	0.6	**19**	**20**
1998	0.8	0.9	**27**	**28**	0.8	0.8	**26**	**27**	0.6	0.7	**21**	**22**	0.6	0.6	**18**	**19**
1999	0.8	0.8	**25**	**27**	0.7	0.7	**23**	**24**	0.6	0.6	**19**	**20**	0.5	0.5	**16**	**17**
2000	0.7	0.8	**24**	**25**	0.7	0.7	**23**	**24**	0.5	0.6	**17**	**18**	0.5	0.5	**15**	**16**
2001	0.8	0.8	**26**	**27**	0.7	0.7	**23**	**24**	0.5	0.5	**17**	**18**	0.4	0.5	**14**	**15**
2002	0.7	0.8	**24**	**25**	0.7	0.7	**22**	**23**	0.5	0.5	**16**	**17**	0.4	0.4	**14**	**14**
2003	0.7	0.7	**22**	**23**	0.6	0.7	**20**	**21**	0.5	0.5	**16**	**16**	0.4	0.4	**13**	**14**
2004	0.6	0.6	**18**	**19**	0.6	0.6	**18**	**19**	0.4	0.5	**15**	**15**	0.4	0.4	**12**	**13**
2005	0.6	0.6	**18**	**19**	0.5	0.6	**18**	**18**	0.4	0.4	**14**	**14**	0.4	0.4	**12**	**12**
2006	0.6	0.6	**20**	**21**	0.5	0.6	**18**	**18**	0.4	0.4	**13**	**14**	0.3	0.4	**11**	**12**
2007	0.6	0.6	**18**	**19**	0.5	0.6	**17**	**18**	0.4	0.4	**13**	**14**	0.3	0.3	**11**	**11**
2008	0.5	0.6	**17**	**18**	0.5	0.5	**16**	**17**	0.4	0.4	**13**	**13**	0.3	0.3	**10**	**11**
2009	0.5	0.5	**15**	**16**	0.5	0.5	**15**	**16**	0.4	0.4	**12**	**12**	0.3	0.3	**9.8**	**10**
2010	0.5	0.6	**17**	**18**	0.5	0.5	**15**	**16**	0.4	0.4	**12**	**12**	0.3	0.3	**9.6**	**10**
2011	0.5	0.6	**17**	**18**	0.5	0.5	**16**	**17**	0.4	0.4	**12**	**12**	0.3	0.3	**9.5**	**9.9**
2012	0.5	0.5	**17**	**18**	0.5	0.5	**16**	**16**	0.4	0.4	**12**	**12**	0.3	0.3	**9.3**	**9.7**
2013	0.5	0.6	**18**	**19**	0.5	0.5	**16**	**16**	0.4	0.4	**12**	**12**	0.3	0.3	**9.2**	**9.6**
2014	0.5	0.5	**16**	**17**	0.5	0.5	**15**	**16**	0.3	0.4	**11**	**12**	0.3	0.3	**9.1**	**9.5**
2015	0.5	0.5	**15**	**16**	0.4	0.5	**15**	**15**	0.3	0.4	**11**	**12**	0.3	0.3	**8.9**	**9.3**
2016	0.4	0.5	**14**	**15**	0.4	0.4	**14**	**14**	0.3	0.3	**11**	**11**	0.3	0.3	**8.6**	**9.0**
2017	0.4	0.5	**14**	**15**	0.4	0.4	**13**	**14**	0.3	0.3	**10**	**11**	0.3	0.3	**8.3**	**8.7**
2018	0.4	0.5	**14**	**15**	0.4	0.4	**13**	**14**	0.3	0.3	**9.9**	**10**	0.2	0.3	**8.1**	**8.4**

Bold values indicate exceedances

TABLE 5-50: RATIO OF MODELED DIETARY DOSE TO TOXICITY BENCHMARKS FOR FEMALE OTTER FOR TRI+ CONGENERS FOR THE PERIOD 1993 - 2018

REVISED

Year	LOAEL 152 Average	LOAEL 152 95% UCL	NOAEL 152 Average	NOAEL 152 95% UCL	LOAEL 113 Average	LOAEL 113 95% UCL	NOAEL 113 Average	NOAEL 113 95% UCL	LOAEL 90 Average	LOAEL 90 95% UCL	NOAEL 90 Average	NOAEL 90 95% UCL	LOAEL 50 Average	LOAEL 50 95% UCL	NOAEL 50 Average	NOAEL 50 95% UCL
1993	10	10	335	341	8.0	8.1	260	264	6.1	6.2	199	203	6.0	6.1	195	199
1994	7.3	7.4	236	240	6.9	7.0	223	227	5.5	5.6	180	183	5.3	5.4	172	175
1995	6.1	6.2	198	201	5.7	5.8	186	189	4.9	5.0	159	162	4.7	4.8	152	155
1996	7.9	8.0	257	260	5.8	5.9	190	193	4.4	4.4	142	145	4.2	4.2	135	138
1997	6.8	6.9	220	224	5.4	5.5	177	180	4.1	4.2	133	135	3.8	3.8	123	125
1998	5.4	5.5	177	180	4.7	4.8	153	155	3.7	3.8	121	123	3.4	3.5	112	113
1999	4.6	4.7	149	151	4.1	4.1	133	135	3.3	3.4	108	110	3.1	3.1	101	102
2000	4.4	4.5	144	146	3.8	3.8	122	124	3.0	3.0	97	98	2.8	2.8	91	93
2001	4.9	5.0	160	163	3.9	4.0	127	129	2.8	2.9	92	94	2.6	2.6	84	85
2002	4.5	4.6	147	150	3.8	3.9	123	125	2.8	2.8	90	91	2.4	2.5	79	81
2003	4.0	4.0	129	131	3.4	3.5	111	113	2.6	2.6	85	86	2.3	2.3	75	76
2004	3.0	3.1	99	101	3.0	3.0	97	99	2.4	2.5	79	80	2.2	2.2	70	72
2005	3.0	3.1	99	100	2.7	2.7	88	89	2.2	2.2	71	72	2.0	2.0	65	66
2006	3.5	3.5	112	114	2.8	2.8	90	92	2.1	2.1	67	68	1.9	1.9	60	61
2007	3.0	3.1	99	100	2.7	2.8	88	90	2.0	2.0	65	66	1.8	1.8	57	58
2008	2.8	2.8	91	92	2.6	2.6	83	85	1.9	2.0	63	64	1.7	1.7	55	55
2009	2.5	2.5	81	82	2.3	2.3	74	75	1.8	1.8	59	60	1.6	1.6	51	52
2010	2.6	2.7	86	87	2.3	2.3	73	74	1.7	1.7	56	57	1.5	1.5	49	50
2011	3.1	3.1	100	102	2.4	2.5	79	81	1.7	1.8	56	57	1.5	1.5	48	49
2012	2.6	2.7	86	87	2.3	2.4	76	77	1.7	1.7	56	57	1.4	1.5	47	48
2013	3.0	3.1	98	100	2.4	2.4	78	79	1.7	1.7	56	57	1.4	1.5	47	47
2014	2.7	2.7	87	88	2.3	2.3	74	75	1.7	1.7	54	55	1.4	1.4	46	47
2015	2.4	2.5	79	81	2.2	2.2	70	71	1.6	1.7	53	54	1.4	1.4	45	46
2016	2.2	2.2	72	73	2.0	2.1	66	67	1.6	1.6	51	52	1.3	1.4	43	44
2017	2.0	2.0	65	66	1.8	1.9	60	61	1.5	1.5	48	48	1.3	1.3	41	42
2018	1.9	2.0	63	64	1.8	1.8	58	59	1.4	1.4	45	46	1.2	1.2	40	40

Bold values indicate exceedances

TABLE 5-51: RATIO OF MODELED DIETARY DOSES TO TOXICITY BENCHMARKS FOR FEMALE MINK ON A TEQ BASIS FOR THE PERIOD 1993 - 2018
REVISED

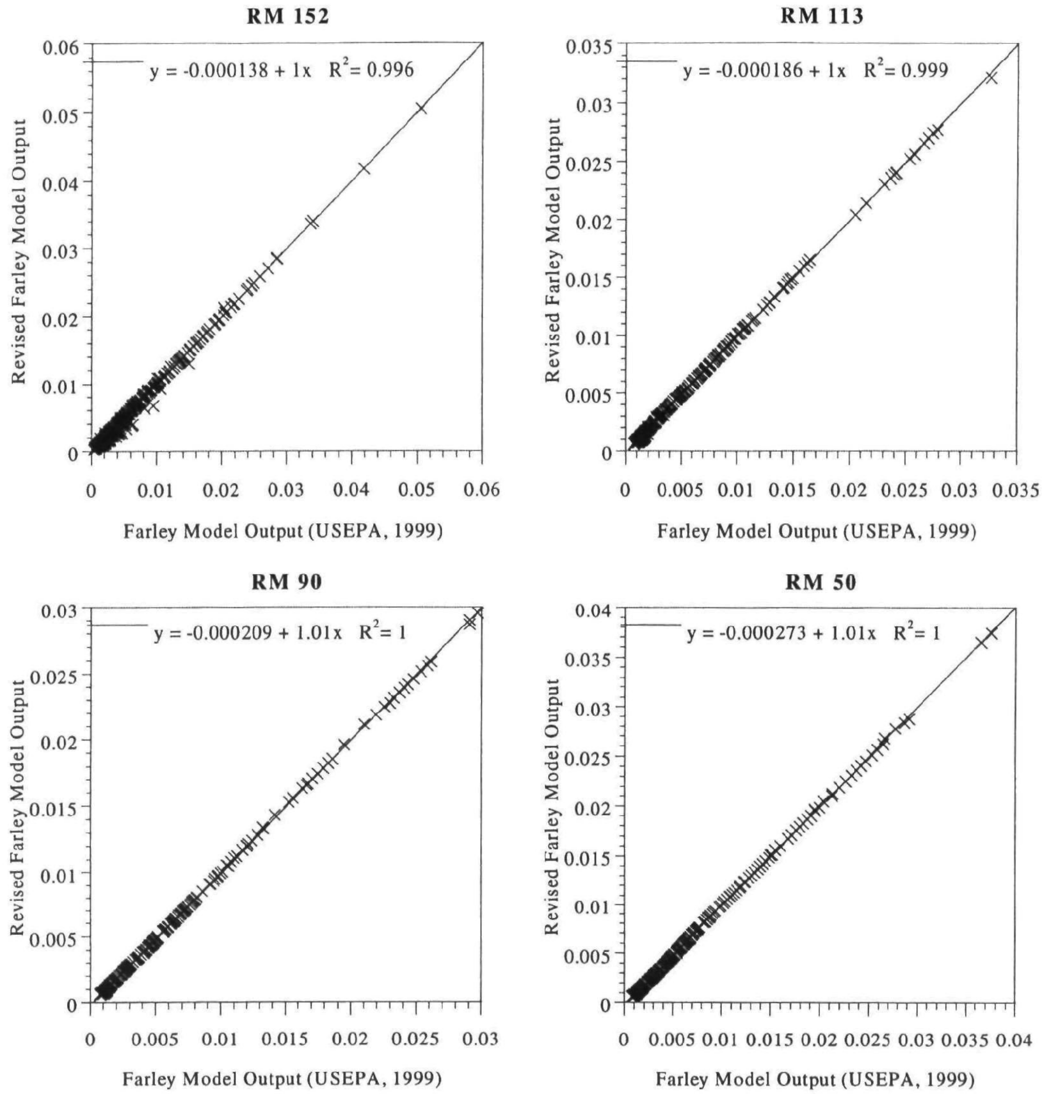
Year	LOAEL 152 Average	LOAEL 152 95% UCL	NOAEL 152 Average	NOAEL 152 95% UCL	LOAEL 113 Average	LOAEL 113 95% UCL	NOAEL 113 Average	NOAEL 113 95% UCL	LOAEL 90 Average	LOAEL 90 95% UCL	NOAEL 90 Average	NOAEL 90 95% UCL	LOAEL 50 Average	LOAEL 50 95% UCL	NOAEL 50 Average	NOAEL 50 95% UCL
1993	NA	NA	**265**	**277**	NA	NA	**230**	**239**	NA	NA	**181**	**188**	NA	NA	**167**	**173**
1994	NA	NA	**204**	**212**	NA	NA	**209**	**216**	NA	NA	**165**	**171**	NA	NA	**149**	**155**
1995	NA	NA	**185**	**192**	NA	NA	**175**	**182**	NA	NA	**148**	**154**	NA	NA	**134**	**139**
1996	NA	NA	**226**	**236**	NA	NA	**184**	**191**	NA	NA	**138**	**143**	NA	NA	**122**	**126**
1997	NA	NA	**191**	**199**	NA	NA	**169**	**176**	NA	NA	**129**	**134**	NA	NA	**111**	**116**
1998	NA	NA	**154**	**160**	NA	NA	**149**	**155**	NA	NA	**120**	**124**	NA	NA	**103**	**107**
1999	NA	NA	**146**	**152**	NA	NA	**134**	**139**	NA	NA	**109**	**113**	NA	NA	**95**	**98**
2000	NA	NA	**135**	**141**	NA	NA	**131**	**135**	NA	NA	**100**	**104**	NA	NA	**87**	**91**
2001	NA	NA	**150**	**156**	NA	NA	**131**	**136**	NA	NA	**98**	**101**	NA	NA	**82**	**85**
2002	NA	NA	**137**	**143**	NA	NA	**125**	**129**	NA	NA	**95**	**98**	NA	NA	**79**	**82**
2003	NA	NA	**123**	**129**	NA	NA	**117**	**121**	NA	NA	**90**	**93**	NA	NA	**75**	**78**
2004	NA	NA	**105**	**110**	NA	NA	**105**	**109**	NA	NA	**83**	**87**	NA	NA	**70**	**73**
2005	NA	NA	**105**	**110**	NA	NA	**101**	**105**	NA	NA	**79**	**82**	NA	NA	**67**	**69**
2006	NA	NA	**114**	**120**	NA	NA	**102**	**106**	NA	NA	**77**	**80**	NA	NA	**64**	**66**
2007	NA	NA	**103**	**108**	NA	NA	**99**	**103**	NA	NA	**75**	**78**	NA	NA	**61**	**64**
2008	NA	NA	**98**	**102**	NA	NA	**94**	**98**	NA	NA	**72**	**75**	NA	NA	**59**	**62**
2009	NA	NA	**87**	**92**	NA	NA	**87**	**90**	NA	NA	**68**	**71**	NA	NA	**56**	**58**
2010	NA	NA	**98**	**103**	NA	NA	**88**	**91**	NA	NA	**67**	**70**	NA	NA	**55**	**58**
2011	NA	NA	**99**	**104**	NA	NA	**91**	**95**	NA	NA	**68**	**70**	NA	NA	**54**	**56**
2012	NA	NA	**97**	**101**	NA	NA	**90**	**93**	NA	NA	**67**	**69**	NA	NA	**53**	**56**
2013	NA	NA	**101**	**106**	NA	NA	**89**	**93**	NA	NA	**66**	**69**	NA	NA	**53**	**55**
2014	NA	NA	**93**	**98**	NA	NA	**87**	**91**	NA	NA	**65**	**68**	NA	NA	**52**	**54**
2015	NA	NA	**85**	**90**	NA	NA	**83**	**87**	NA	NA	**63**	**66**	NA	NA	**51**	**53**
2016	NA	NA	**79**	**83**	NA	NA	**78**	**81**	NA	NA	**60**	**63**	NA	NA	**49**	**51**
2017	NA	NA	**79**	**83**	NA	NA	**75**	**78**	NA	NA	**58**	**61**	NA	NA	**48**	**50**
2018	NA	NA	**81**	**85**	NA	NA	**75**	**78**	NA	NA	**57**	**59**	NA	NA	**46**	**48**

Bold values indicate exceedances

TABLE 5-52: RATIO OF MODELED DIETARY DOSES TO TOXICITY BENCHMARKS FOR FEMALE OTTER ON A TEQ BASIS FOR THE PERIOD 1993 - 2018 REVISED

Year	LOAEL 152 Average	LOAEL 152 95% UCL	NOAEL 152 Average	NOAEL 152 95% UCL	LOAEL 113 Average	LOAEL 113 95% UCL	NOAEL 113 Average	NOAEL 113 95% UCL	LOAEL 90 Average	LOAEL 90 95% UCL	NOAEL 90 Average	NOAEL 90 95% UCL	LOAEL 50 Average	LOAEL 50 95% UCL	NOAEL 50 Average	NOAEL 50 95% UCL
1993	NA	NA	1954	1988	NA	NA	1517	1542	NA	NA	1164	1183	NA	NA	1141	1160
1994	NA	NA	1380	1400	NA	NA	1304	1327	NA	NA	1051	1068	NA	NA	1004	1021
1995	NA	NA	1155	1175	NA	NA	1084	1104	NA	NA	927	943	NA	NA	887	902
1996	NA	NA	1499	1519	NA	NA	1107	1124	NA	NA	830	844	NA	NA	791	804
1997	NA	NA	1284	1306	NA	NA	1032	1050	NA	NA	777	789	NA	NA	716	728
1998	NA	NA	881	885	NA	NA	892	908	NA	NA	707	719	NA	NA	651	662
1999	NA	NA	795	774	NA	NA	774	787	NA	NA	631	641	NA	NA	588	597
2000	NA	NA	775	756	NA	NA	714	726	NA	NA	565	574	NA	NA	532	540
2001	NA	NA	858	838	NA	NA	743	755	NA	NA	538	547	NA	NA	489	497
2002	NA	NA	795	772	NA	NA	719	731	NA	NA	523	532	NA	NA	463	470
2003	NA	NA	696	679	NA	NA	647	658	NA	NA	494	502	NA	NA	439	446
2004	NA	NA	532	517	NA	NA	568	578	NA	NA	458	466	NA	NA	411	418
2005	NA	NA	543	526	NA	NA	511	520	NA	NA	415	421	NA	NA	379	385
2006	NA	NA	618	602	NA	NA	527	536	NA	NA	389	396	NA	NA	351	357
2007	NA	NA	549	531	NA	NA	515	524	NA	NA	379	385	NA	NA	333	339
2008	NA	NA	488	474	NA	NA	486	495	NA	NA	365	371	NA	NA	318	324
2009	NA	NA	434	423	NA	NA	430	437	NA	NA	342	348	NA	NA	298	303
2010	NA	NA	469	457	NA	NA	428	435	NA	NA	326	332	NA	NA	285	290
2011	NA	NA	545	530	NA	NA	463	471	NA	NA	329	334	NA	NA	280	285
2012	NA	NA	492	474	NA	NA	441	448	NA	NA	325	331	NA	NA	275	279
2013	NA	NA	547	528	NA	NA	456	463	NA	NA	325	330	NA	NA	272	277
2014	NA	NA	514	493	NA	NA	432	439	NA	NA	317	323	NA	NA	267	272
2015	NA	NA	437	423	NA	NA	409	416	NA	NA	308	313	NA	NA	262	266
2016	NA	NA	395	381	NA	NA	386	392	NA	NA	296	301	NA	NA	253	257
2017	NA	NA	388	370	NA	NA	351	357	NA	NA	277	282	NA	NA	242	246
2018	NA	NA	398	377	NA	NA	339	345	NA	NA	264	269	NA	NA	231	235

Bold values indicate exceedances

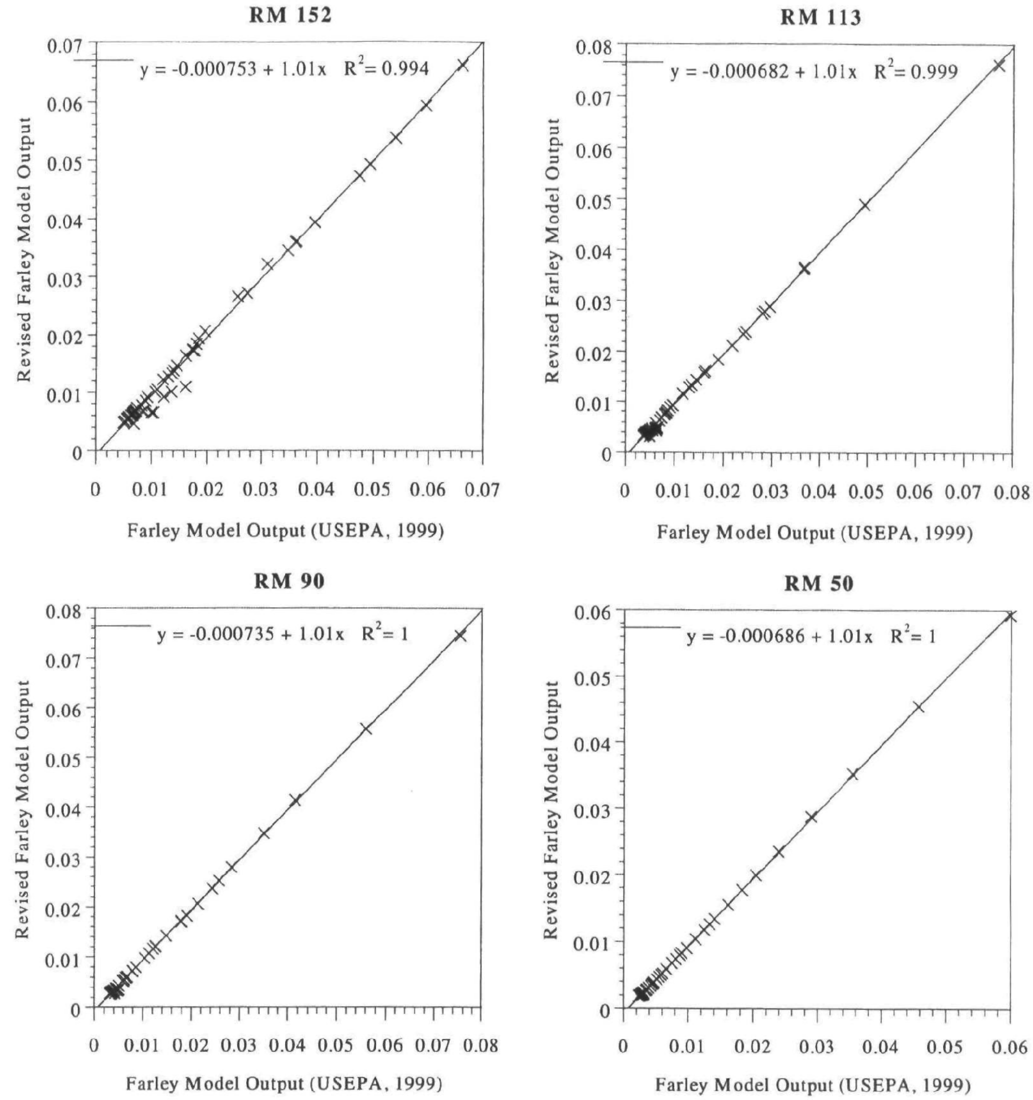


Sources: Farley et al., 1999, Hudson River Database Release 4.1 and USEPA, 2000a

Notes:

1. The x-axis is the model data used in the Addendum to the Baseline Ecological Risk Assessment for Future Risks in the Lower Hudson River (USEPA, 1999c).
2. The y-axis is the revised Farley fate and transport model with the Upper River Loads from the RBMR (USEPA, 2000a).
3. Monthly arithmetic averages are shown.

Figure III-1
Comparison for Tri+ PCBs in the Dissolved Phase of the Water Column Between the Revised Model Output versus the Data Presented in the Lower River Ecological Risk Assessment

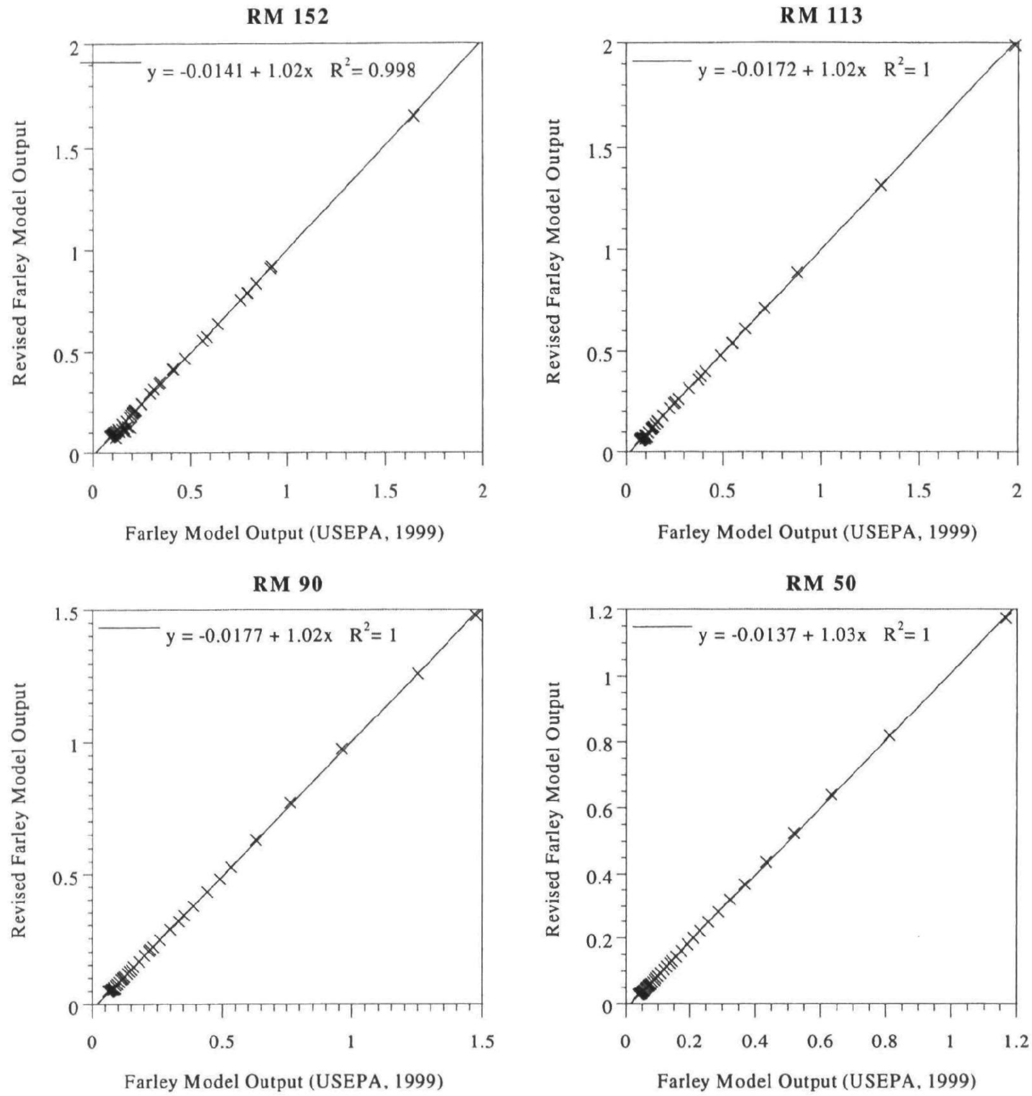


Sources: Farley et al., 1999, Hudson River Database Release 4.1 and USEPA, 2000

Notes:

1. The x-axis is the model data used in the Addendum to the Baseline Ecological Risk Assessment for Future Risks in the Lower Hudson River (USEPA, 1999c).
2. The y-axis is the revised Farley fate and transport model with the Upper River Loads from the RBMR (USEPA, 2000a).
3. Annual arithmetic averages are compared.

Figure III-2
Comparison for Total PCBs in the Water Column (Whole Water) Between the Revised Model Output versus the Data Presented in the Lower River Ecological Risk Assessment



Sources: Farley et al., 1999, Hudson River Database Release 4.1 and USEPA, 2000a

Notes:

1. The x-axis is the model data used in the Addendum to the Baseline Ecological Risk Assessment for Future Risks in the Lower Hudson River (USEPA, 1999c).
4. The y-axis is the revised Farley fate and transport model with the Upper River Loads from the RBMR (USEPA, 2000a).
5. Annual arithmetic averages are compared.

Figure III-3

Comparison for Total PCBs in the Sediment (0-2.5 cm) Between the Revised Model Output versus the Data Presented in the Lower River Ecological Risk Assessment

Comments

Federal

U.S. DEPARTMENT OF COMMERCE
National Oceanic and Atmospheric Administration
National Ocean Service
Office of Response and Restoration
Coastal Protection and Restoration Division, 290 Broadway, Rm 1831
New York, New York 10007

EF-1

January 28, 2000

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 December 1999 Phase 2 Report - Review Copy, Further Site Characterization and Analysis, Volume 2E - Baseline Ecological Risk Assessment for Future Risks in the Lower Hudson River, 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 addendum to the baseline ecological risk assessment (ERA) are to quantify future risks to selected biological receptors and communities exposed to releases of PCBs in the lower tidal estuarine portion of the river between Federal Dam and the Battery in the absence of remediation. The upper freshwater non-tidal portion of the river between Federal Dam and Hudson Falls Hudson River was the primary focus of a August 1999 ERA although the Lower Hudson was also evaluated.

Modeled concentrations of PCBs in sediment, water column, striped bass and white perch were obtained from a model developed by Farley et al. 1999. Future concentrations of fish were also derived from a modified FISHRAND model (EPA 1999, 2000). White perch was the only species whose concentration was estimated from both models. Risk evaluations were based on future exposure from sediment, water and fish.

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 Addendum focuses on Lower Hudson River: 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.

Risks to benthic invertebrate communities were determined by comparing modeled concentrations of sediment and water to existing guidelines, standards and criteria. Toxic reference values

(TRVs) based on body burden or dietary dose were selected for survival, growth and reproductive endpoints of fish, birds and mammals.

Selected fish NOAELs ranged from 0.16 to 3.1 mg/kg wet weight PCBs. Fish LOAELs ranged from 1.5 to 15 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. Based on whole body concentrations, the lowest TRVs were calculated for pumpkinseed, yellow perch, white perch, largemouth bass, striped bass, brown bullhead and shortnose sturgeon. Spottail shiner had the highest whole body. Brown bullhead had the highest TEQ TRV based on NOAELs while spottail shiners had the highest based on LOAELs. All other fish species had the lowest TRV for TEQs.

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 tree swallow for field conducted diet and egg concentration studies.

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 PCBs/kg/day; LOAEL 0.15 mg PCBs /kg/day). 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.

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 future exposure to Lower Hudson River PCBs:

- risks to fish and wildlife are greatest in the upper reaches of the Lower Hudson River and decrease downstream with concomitant decreasing PCB concentrations,
- many species are expected to be at risk at least through the year 2018 - the upper bound of the forecasting exercise,
- modeled sediment and water concentrations generally exceed existing guidelines, standards and criteria for the protection of aquatic health;
- animals using areas designated as significant habitats may be adversely affected by PCBs,
- PCBs may adversely affect survival, growth, and reproduction of fish, especially the higher trophic levels;
- PCBs may adversely affect survival, growth, and reproduction of waterfowl, omnivorous, and piscivorous birds while no risk is expected for insectivorous birds;
- PCBs may adversely affect survival, growth, and reproduction of insectivorous, omnivorous, and piscivorous mammals; and
- threatened and endangered species are particularly susceptible.

Comments

The Baseline Ecological Risk Assessment Addendum (ERA) is a companion document to the August Hudson River ERA. The report determines the future risks in the Lower Hudson River. 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 are described. Overall the document is well-organized and clearly written.

The fate and transport and bioaccumulation modeling presented in the Baseline Modeling Report plus Farley's model for the Lower Hudson provides the primary exposure information for the ERA Addendum. While the ERA Addendum 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, revisions to the BMR will be released in a report at the end of January. Have these modifications been accounted for in the predictions contained within the ERA Addendum? If not, how will these changes impact predictions of risk? **EF-1.1**

The Farley model relies on HUDTOX load estimates at the Thompson Island Dam (TIP) to predict sediment, water and fish (white perch, striped bass) PCBs in the Lower Hudson. This required the conversion of tri+ PCB loads to homologue loads at the Federal Dam because dichloro through hexachloro homologues are state variables in the Farley model of the Lower Hudson River. The conversion process is not clearly explained. What are the uncertainties associated with this conversion and the potential implications for the predicted sediment, water and fish tissue concentrations. **EF-1.2**

There are a number of aspects of the Hudson River system that the fate and transport and bioaccumulation models are not addressing. For example, as identified in our comments to the May 1999 Baseline Modeling Report (7/1/99) and the August 1999 Ecological Risk Assessment (9/7/99), potential effects of daily changes in water level on nearshore shallow-water PCB deposits and non-scour related movement of PCB-contaminated sediment 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 employ HUDTOX loads at Federal Dam to predict future risk in the Lower Hudson River. The implications of the uncertainty resulting from the model inputs to risk assessment should be addressed. **EF-1.3**

The food chain modeling (FISHRAND) used a generic growth rate for lake trout as an input parameter for the species of fish modeled rather than attempting to capture the difference in their growth. Did Farley et al. use species-specific growth rates for white perch and striped bass? How sensitive are the two models to growth rates and what implications does this have for predictions of future PCBs in fish and risks to receptors? **EF-1.4**

Water column and sediment data used in the exposure assessment are averaged without regard for habitat occupied by the receptors of concern or changes in physico-chemical conditions of the river. In addition, TOC and lipid content were set to a single value for the entire Lower Hudson. How sensitive is the model output to these assumptions and what implications does this have for the derived risks? **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. For example, the selection of the **EF-1.6** 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. Other studies (Hansen et al. 1974, USACE 1988) generated similar NOAELs and LOAELs and should have been presented to support the laboratory- and field-based TRVs.

Field- and laboratory-based fish results were handled differently (Section B.2.1). When a laboratory study could not be identified for a particular fish species or one in the same taxonomic family, an interspecies uncertainty factor was applied to derive a TRV. For field-based studies, NOAELs and LOAELs were only developed for the species studied or one within the same family. An interspecies uncertainty factor was not applied if the receptor of interest belonged to a different family. An explanation for the different data treatments should be provided. **EF-1.7**

Toxicity quotients were calculated using the field TRV as the denominator when both field and laboratory TRVs were developed for the same fish species. An alternative approach would have been to develop toxicity quotients from field and lab TRVs to bracket risk.

The TRVs developed for Hudson River total PCB body burdens in fish focus solely on growth, survival and reproductive capacity. They may underestimate risk because they do not consider other adverse effects such as immune suppression reported in the literature. The uncertainty section should address these other adverse effects since they are not accounted for in the TRV selection process. Additionally, modeling results tend to underestimate fish concentrations which would result in lower toxicity quotients. This could also contribute to an underestimation of risk. **EF-1.8**

Specific Comments

Executive Summary

Pages ES-9 and ES-11: Statements about bald eagles appear to be inconsistent. Page ES-11 states that "future PCB exposures (predicted from 1003 to 2018) are not expected to be of a sufficient magnitude to prevent reproduction or recruitment" appears to contradict page ES-9's reference to the lack of breeding success. **EF-1.9**

Chapter 3

Page 15 Para 3: The upstream boundary conditions used in the BMR assumes that flow-related changes (increases) in loading during high flow events will not occur. 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 setting the upstream boundary at 10 ng/l could underestimate the loading. This uncertainty should be addressed. **EF-1.10**

Pages 13, 14, 16, 17 , 24 and 26: Farley et al. (1999) and FISHRAND model output parameters are stated but the descriptions are inconsistent. For example, the Farley et al. (1999) model generated sediment, water and fish (white perch) PCB concentrations (Section 3.0, Para 5), and sediment, water and fish (striped bass, white perch) PCB concentrations (Section 3.1.1; Section 3.1.1.3, Para 2). In the case of FISHRAND model, PCB body burdens were calculated for all fish (Section 3.0, Para 2; Section 3.1.1), and all fish receptors except striped bass (Section 3.1.1.2, Para 1; Section 3.1.2.3; Section 3.2.4). **EF-1.11**

Page 16, Estimation of striped bass body burdens: It is not clear whether this estimation was based on wet weight or lipid-normalized values. **EF-1.12**

Page 20 Para 2: It is suggested that the Farley et al. model overestimates loss of PCBs from the water column during the summer. The importance of this finding relative to uptake of PCBs by fish should be discussed. **EF-1.13**

Page 20 Para 3: According to Figure 3-7, empirical data at two locations fell outside the boundaries of the modeled data. The text indicates only RM 47. **EF-1.14**

Pages 22-23: Modeled fish concentrations are compared against empirical data. Does the model capture the PCB concentrations reported by NYSDEC for 1998? **EF-1.15**

Page 25, Para 5: A TOC of 2.5% was used to estimate organic carbon-normalized PCB sediment concentrations. What effect does selection of a 2.5% TOC have on the outcome of the comparisons as TOC concentrations in the Lower Hudson ranged from 0.35% to 4.9% for individual samples? The sediment guidelines section provides PCB guidelines and standards. **EF-1.16**

Table 3-7 should 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. **EF-1.17**

Page 25, Para 7: Predicted benthic invertebrate PCB concentrations for the year 1993 could have been compared to empirical data. Are predictions a reasonable estimate of actual measurements? **EF-1.18**

Page 27 Para 4: Elsewhere the report indicates that PCB concentrations in striped bass in food web region 1 are predicted using a striped bass to largemouth bass ratio. The last sentence implies the ratio was used for all of the Lower Hudson striped bass estimates. **EF-1.19**

Chapter 4

Table 4-1: 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 in Table 4-1. **EF-1.20**

Table 4-1: Other laboratory (Hansen et al. 1974) and field studies (Adams et al. 1989, 1990, 1992, USACE 1988) document fish effect levels similar to Bengtsson (1980) thereby providing further weight of evidence to support selection of these TRVs. Hansen et al. (1974) is described in Appendix B but USACE (1988) was not. The NOAEL from the USACE (1988) study on fathead minnows are 5.6 mg/kg. Dividing by a factor of 10 for all species not in the Cyprinidae family would result in an NOAEL of 0.56 mg/kg. Hence Hansen et al. (1974), Bengtsson (1980) and USACE (1988) yield similar TRVs. The field NOAEL developed for pumpkinseed and largemouth bass from USACE (1988) is also comparable to the field NOAEL in Adams et al. (1989, 1990, 1992). The field-based NOAEL (0.5 mg/kg PCBs) reported in the ERA Addendum was based on reproduction (Adams et al. 1989, 1990, 1992). A lower value (0.3 mg/kg PCBs) should have been selected based on growth. It should be noted that Adams et al. (1989, 1990, 1992) assessed endpoints besides those related to survival, growth and reproduction and these effects were also observed at concentrations below 0.5 mg/kg PCBs. Adverse effects were associated with DNA integrity, detoxification enzymes, lipid metabolism, community structure and histological indices. **EF-1.21**

Page 36: The discussion on the use of TEFs to derive TEQs should have included an explanation of data quality issues and the impacts on the TEQ calculations. For example, PCB 126 was frequently classified as below detection and PCB 81 was not measured. In addition, the congeners of primary importance (by weight and toxicologically) for the water column were mostly below detection. **EF-1.22**

Chapter 5

Page 38, Section 5.1.1.1: Tables 3-2 and 3-3 are actually Tables 3-6 and 3-7. In 1993, mean PCBs were 1.213 ppm at RM 152, 0.828 ppm at RM 113, 0.872 ppm at RM 90 and 0.806 ppm at RM 50. Predicted sediment PCBs (Table 3-6) underestimate empirical means in 1993 by 0.07 to 0.42 ppm. For organic carbon-normalized sediments, estimates also underpredict observed concentrations at RM 152, RM 113 and RM 50 with the greatest difference again observed for RM 50. A TOC of 2.5% was used from Farley's model while average TOC for each of these RM segments ranged from 2.5% to 3.6% and individual samples ranged from 0.35% to 5.3%. **EF-1.23**

Page 39 Top: Forecasted sediment concentrations exceeded NYSDEC benthic aquatic life chronic toxicity criterion at RM 152 and RM 113 for the duration of the modeling period at the 95% UCL. RM 90 only exceeded criterion until 2011. **EF-1.24**

Page 39 Para 2 and Table 5-1: The organic carbon-normalized SEL (Persaud et al. 1993) should read 13 mg/kg instead of 1.3 mg/kg. **EF-1.25**

Page 40, Section 5.1.2.1; Page 44, Section 5.2.2.1; Page 47, Section 5.3.2.1; Page 49, Section 5.4.2.1; Page 53, Section 5.5.2.1; Page 55, Section 5.6.2.1; Page 56 Section 5.7.2.1;, Page 59, Section 5.8.2.1; Page 61, Section 5.9.3.1; Page 63, Section 5.10.2.1: The NYSDEC surface water standard for protection of wildlife is 1.2×10^{-4} ug/l total PCBs (NYSDEC 1998). It replaced the wildlife criterion of 0.001 ug/l in 1998 (Stoner 2000). **EF-1.26**

Page 40, Last Para: The analysis on forage fish reproductive effects assumes that measurements of young-of-year spottail shiner and age 1 pumpkinseed are equivalent to concentrations in mature adults. According to Table 4-1, the TRV for spottail shiner on an NOAEL basis is 1.6 mg/kg not 15 mg/kg as stated in this paragraph. The NOAEL derived from laboratory studies, therefore, resulted in a TRV for pumpkinseed that is an order of magnitude lower than for spottail shiner. If comparisons are made between laboratory and field studies then the difference is reduced to 3 fold. **EF-1.27**

Page 41, Section 5.2.1.2: 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. What is the evidence to indicate that lipid-normalized concentrations are more directly related to the reproductive effects than the wet weight concentrations in eggs? **EF-1.28**

Page 41, Section 5.2.1.3: RM 133 should read RM 113. This discussion should also note that since the literature-derived TRVs were based on whole body concentration studies, the fish fillets were converted to whole body for direct comparison. **EF-1.29**

Pages 41-43, 52-53, 58, 61: 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 species. For mammalian wildlife, fish TCDD concentrations of 0.7 pg/g pose a low risk and 7 pg/g pose a high risk. **EF-1.30**

Page 42, Section 5.2.1.5: 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 species in the same genera or family. 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. If this was done, how would the TRVs change and what impact would it have on the toxicity quotients? **EF-1.31**

Pages 48, 50, and 54: Current trends in bird usage should be compared to historical usage, especially prior to GE's use of PCBs at their two Hudson River facilities. **EF-1.32**

Chapter 6

Page 69: 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-1.33**

Page 71 Top: While the model may be able to reproduce general trends, it was incapable of picking up year to year changes in fish concentrations and generally underestimated average surface sediment concentrations in 1993, the only year data are available for the Lower Hudson. **EF-1.34**

Page 71 Para 1: There is also uncertainty associated with changes in upstream boundary conditions and potential releases from the remnant deposits. **EF-1.35**

Page 71-72, Section 6.3.2.2: The report discusses sensitivity analyses for avian and mammalian receptors and refers to the BMR for a more detailed analysis of uncertainty and sensitivity in the FISHRAND model. A sensitivity 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., annually averaged-sediment and summer averaged-water column concentrations). **EF-1.36**

Chapter 7

Page 75, Para 4: The report states that the uncertainty in sediment and water forecasts is approximately a factor of two. The basis for this statement should be explained. **EF-1.37**

Page 76: The sensitivity of tree swallows to PCBs should be compared to other insectivorous avian species since the lack of predicted risk may underestimate threats (impairment of survival, growth, reproduction) to others in the same feeding guild. **EF-1.38**

Appendix A

Page A-6 Para 1: "However, the 2.4 percent summer-to-spring change in hexachloro homologue ratio". The 2.4% change was between fall-winter and spring. Between summer and spring the change was smaller (1.4%). **EF-1.39**

Appendix B

Page B-10 Para 1: See comment above on Page 41, Section 5.2.1.2. **EF-1.40**

Pages B-10 to B-11, Section B.2.3.1, Para 1: Neither Hansen et al. (1971) nor Hansen et al. (1974) focused on adult mortality. Hansen et al. (1971) evaluated toxicity of Aroclor 1254 to juvenile spot and Hansen et al. (1974) examined the effect of Aroclor 1254 on the eggs of sheepshead minnow. Their 1974 study represents a more sensitive endpoint than the (1971) study and should have been considered in the development of TRVs. **EF-1.41**

Pages B-10 to B-11, Section B.2.3.1, Para 2, and Sections 2.3.3-2.3.8: Hansen et al (1974) established an NOAEL of 1.9 mg/kg PCBs and a LOAEL of 9.3 mg/kg PCBs based on early life stage survival, where TRVs were based on adult female sheepshead minnow concentrations that were directly associated with effects on their respective eggs.

The field study by USACE (1988) with fathead minnows should also be described since the toxicity endpoint was reproductive success and the NOAEL 5.6 mg/kg. For less closely related species (non-Cyprinidae), the NOAEL be 0.56 mg/kg, upon employing an uncertainty factor of 10.

Pages B-10 to B-11, Section B.2.3.1, Para 3, and Sections 2.3.3-2.3.8: NOAA does not support the rationale for selecting Bengtsson (1980)[a] over Hansen et al. (1974)[b]. The Hansen et al. (1974) laboratory study should have been selected along with Bengtsson (1980) and the USACE (1988) field study along with Adams et al. (1989, 1990, 1992). For spottail shiner, the resultant laboratory NOAELs and LOAELs would be 1.6[a] and 1.9[b] mg/kg PCBs and 9.3[b] and 15[a] mg/kg PCBs, respectively, based on these two studies. All of the other seven fish receptors would have NOAELs and LOAELs an order of magnitude lower (NOAEL: 0.16[a] and 0.19[b] mg/kg PCBs; LOAEL: 0.93[b]and 1.5[a] mg/kg PCBs). The field-based NOAELs (mg/kg PCBs) were 3.1 for white perch and striped bass (Westin et al. 1983), 0.3 for pumpkinseed and largemouth bass (Adams et al. 1989, 1990, 1992), and 5.6 for spottail shiner (USACE 1988). Values are summarized in the table below. **EF-1.42**

TRVs for PCBs (mg/kg PCBs wet wt.)		White Perch, Striped Bass	Largemouth Bass, Pumpkinseed	Spottail Shiner	Yellow Perch, Brown Bullhead, Shortnose Sturgeon	References
Tissue Concentration						
Lab-based	NOAEL	0.16	0.16	1.6	0.16	Bengtsson 1980
Lab-based	NOAEL	0.19	0.19	1.9	0.19	Hansen et al. 1974
Field-based	NOAEL	3.1	0.31	0.31	0.31	Westin et al. 1983
Field-based	NOAEL	0.03	0.3	0.03	0.03	Adams et al. 1989, 1990, 1992
Field-based	NOAEL	0.56	0.56	5.6	0.56	USACE 1988
Lab-based	LOAEL	0.93	0.93	9.3	0.93	Hansen et al. 01974
Lab-based	LOAEL	1.5	1.5	15	1.5	Bengtsson 1980

Pages B-11, Section B.2.3.1, Para 3, and Sections 2.3.3, 2.3.4, 2.3.6, 2.3.8: Adams et al. (1989, 1990, 1992) also documented reductions in length, weight and growth potential (RNA/DNA ratio). The NOAEL for growth was 0.3 mg/kg PCBs while the NOAEL of 0.5 ppm PCB was based on fecundity (clutch size). In addition, other adverse effects were reported at concentrations lower than the NOAEL selected for lethality, growth and reproduction. These included significant differences between reference and contaminated site fish for the following parameters: DNA integrity (strand breaks), detoxification enzymes (P450, CB5, NADPH, EROD), histological indices (liver and spleen parasites, macrophage aggregates in the liver, necrotic liver parenchyma) and lipid metabolism (serum triglycerides, body triglycerides, phospholipids). Still, other effects (i.e., species richness, total body lipid, liver-somatic index) were observed but results were either not significantly different from the reference or statistical analyses were not presented.

Page B-17. The NOAEL and LOAEL are transposed. The LOAEL TRV for white perch should read 0.6 ug TEQs/kg lipid. The NOAEL TRV for white perch should read 0.29 ug TEQs/kg lipid. **EF-1.43**

Tables B-5 and B-6 should include USACE (1988). **EF-1.44**

Table B-6: This table lists Adams et al. (1989, 1990, 1992) as providing EL-no effect and EL-effect while the text in Appendix B indicates that an NOAEL and LOAEL was derived from these studies. The Adams et al. (1989, 1990, 1992) field investigations evaluated redbreast sunfish from four sites along the East Fork Poplar Creek. Contaminant concentrations and observed **EF-1.45** effects encompass a range of values and should be considered representative of NOAEL/LOAELs rather than EL-no effect or EL-effect.

Table B-7: It is not clear from the table whether lipid values were reported by the referenced study or estimated by the authors of the report. If estimated, the estimation procedure and the potential implications should be discussed. **EF-1.46**

Figure B-2: Selected toxicity endpoints are shown for selected aroclors. Two of them, an NOAEL of 11.6 mg/kg PCB and an LOAEL of 36 mg/kg PCB in fathead minnow cannot be found by cross-referencing Table B-5. What is the reference? Why were these endpoints selected over others presented in Table B-5? Why was the focus of the figure limited to laboratory-based studies? **EF-1.47**

Figure B-3: Selected toxicity endpoints are shown for selected fish egg dioxin equivalencies. Why were these endpoints selected over others presented in Table B-7? Why was the focus of the figure limited to laboratory-based studies? **EF-1.48**

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

NOAA Coastal Resource Coordinator

References

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.

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USEPA 1993. 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., March 1993.

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ES-1

John P. Cahill
Commissioner

February 4, 2000

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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) for Future Risks in the Lower Hudson, Hudson River PCBs Reassessment RI/FS, dated December 1999. Our comment on the Lower Hudson BERA is provided below.

On page 40, the reference to the "NYSDEC wildlife bioaccumulation criterion of 0.001 µg/L" for PCB is an older number which has changed. The current number for PCB is 1.2x10-4 µg/L (ppb) for protection of wildlife which is a promulgated New York State ambient water quality standard (see 6 NYCRR Part 703). 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.
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Local

SARATOGA COUNTY
ENVIRONMENTAL MANAGEMENT COUNCIL
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EL-1

January 26, 2000

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New York, N.Y. 10007-1866

Dear Ms. Hess:

Enclosed you will find the Saratoga County Environmental Management Council's (SCEMC's) comments on the **Baseline Ecological Risk Assessment For Future Risks in the Lower Hudson River** and the **Human Health Risk Assessment for the Mid-Hudson River** prepared by the Council's chief technical advisor, David Adams.

Many of the SCEMC's previous comments on the Hudson River Reassessment's Phase 2 Human Health Risk and Ecological Risk Assessment Reports transmitted to you on September 2, 1999 apply to these reports as well. The Council believes these latest Ecological and Human Health Assessments also reflect an unrealistic and excessive degree of "scientific" over-conservatism in calculating the human health and ecological risks.

In the enclosed comments, David Adams makes a number of appropriate and what we feel are valid observations relating to the unavailability and inconsistencies of important modeling information not being provided to the public for its review prior to its being used by EPA in these reports. The unavailability of EPA's revised baseline modeling information and EPA's lack of agency/peer review of the Farley model are important areas of methodological concern as these tools are crucial in determining the magnitude of the Reassessment's risk assessments. The SCEMC requests, at this time, a copy of EPA's revised modeling information for our review and comment. This information should also be provided to all Reassessment public information repositories.

Once again, it becomes apparent that EPA **has not developed an adequate overall methodological framework for the Reassessment** when it relies on a model (Farley's) to assess mid and lower river risks which requires PCB monitoring information on a homolog basis rather than a congener basis which was the type of data collected during the Reassessment monitoring period. This lack of adequate pre-project planning now requires the need for data conversion which introduces yet "another undefined level of uncertainty into the calculated risks". The Council also feels it is inappropriate to utilize a limited number of striped bass samples to draw what we believe to be erroneous conclusions in regarding PCB concentrations found in largemouth bass populations. Again, the need for additional PCB Homolog sampling for representative fish species found in the mid and lower Hudson River should have been anticipated and is indicative of the poor methodological planning inherent throughout EPA's Hudson River PCB Reassessment process.

Sincerely,

Peter M. Balet
Chairman

Enc.
cc: Doug Tomchuk, USEPA, Region 2
SCEMC Members
Darryl Decker, Chr., Government Liaison Committee, CIP
The Honorable John Sweeney
John Wanska, USGAO
Dr. George Putman, Scientific & Technical Committee, CIP
William Ports, NYSDEC
Ned Sullivan, Scenic Hudson

SARATOGA COUNTY

ENVIRONMENTAL MANAGEMENT COUNCIL

PETER BÁLET
CHAIRMAN

GEORGE HODGSON
DIRECTOR

COMMENTS ON PHASE 2 - VOLUME 2E
A BASELINE ECOLOGICAL RISK ASSESSMENT FOR FUTURE RISKS
IN THE LOWER HUDSON RIVER
AND ON VOLUME 2F
A HUMAN HEALTH RISK ASSESSMENT FOR THE MID-HUDSON RIVER
HUDSON RIVER PCB'S REASSESSMENT RI/FS
DECEMBER, 1999

Prepared By: David D. Adams, Member, Saratoga County EMC and Government Liaison Committee, January 2, 2000

<u>General Comments</u>

1. Both of these risk assessments and the revised EPA FISHRAND Model for the Upper Hudson River are based on the revised EPA PCB Fate and Transport Model and the Farley, et. al. Model for the Lower Hudson River. Reports describing these models and the model results were not made available by EPA with the risk assessment reports. It is improper for EPA to present reports to the EL-1.1 public for review and comments when information vital to the review is not available to the general public. Before presenting these reports, EPA should have made the revised EPA model reports and the Farley, et. al. Model report available in the designated PCB Reassessment repositories for review along with the risk assessment reports. I was able to obtain a copy of the Farley, et. al. Model report through the courtesy of Alison Hess of EPA. Results of my review of the Farley Model are presented as appropriate in the comments on the Risk Assessment Reports. My review was constrained, however, by not having the model revisions made after March, 1999. EPA is requested to forward information on these revisions. I still await the revised EPA model reports which have not yet been issued.

2. In EPA's public presentation of the Risk Assessment Reports, EPA stated that EPA does not plan to review the Farley Model. The reason given was that the Reassessment and subsequent remediation decision being done by EPA is for the Upper Hudson only. The logic of this position is difficult to EL-1.2 understand. If the risk assessments of the Mid and Lower Hudson are of no significance to EPA's study of the Upper Hudson, then why were the risk assessments done? If the results of the risk assessments may have bearing on EPA's decision about remedial action in the Upper Hudson, then EPA owes the public the assurance that the risk assessments have been done on a sound basis. This assurance requires EPA's review of the Farley Model and also review by an appropriate independent review panel. EPA is requested to respond as to the use of these risk assessments and based on that response, as to whether the Farley Model will be reviewed. While overall the Farley Model appears

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to be a good and credible model, the following are some of my questions/concerns that arose from my review of the report by Farley, et. al. which illustrate why review of the Farley Model is needed:

a. The very sharp concentration gradient shown in Fig. 1-1 for di PCB's between RM159 and RM144 is suspect as it is not clear what could cause such a gradient. Also, there is no explanation for the second bar graph at RM159. If this bar graph is selected, the sharp gradient for di disappears. Is it possible there is something wrong with the data presented in the first bar graph? EL-1.2a

b. In many places, values of parameters are stated or assumed with little or no justification. Examples are the sediment thicknesses assigned to each model segment (p. 19); the use of the 1989 Mohawk River and Upper Hudson River flows as a constant yearly flow repeated annually throughout the PCB simulations (P. 24); sedimentation rates, suspended solids concentrations, settling velocity, suspended sediment loads from the Upper Hudson and Mohawk River during high and low flow periods, sediment loads from the Lower Hudson Watershed and their distribution in the model segments (P. 26); production rate of solids by phytoplankton, the stoichiometric conversion factor, the decomposition percentage for phytoplankton, and average-annual sedimentation rates (P. 27); fraction of organic carbon in sediments (P. 30); the values for a_{DOC} (P. 56); use of Mohawk River PCB concentrations for Passaic, Hackensack, and Puritan Rivers (P. 40). EL-1.2b

c. The specification rather than modeling of hydrodynamic, organic carbon, and sediment transport (P. 18). EL-1.2c

d. The lack of data to support model calculated values (see P. 28 & Fig. 2-5 where data are lacking above RM25 for low flow and RM12 for high flow and P. 55 & Fig. 3-1 where data are lacking below RM80). EL-1.2d

e. The assignment of PCB initial conditions for sediments for model segments missing sediment cores. Based on the distribution of cores, it appears only 6 or 7 segments out of 26 segments in the model have core data (PP. 41 & 45). EL-1.2e

f. There seems to be a very large number of parameter adjustments required to calibrate the bio-accumulation model (P. 54). EL-1.2f

g. The rather poor fit in several instances of the data to the model calculations for PCB homologue concentrations in surface sediments (P. 59 & Fig. 3-5). EL-1.2g

h. The apparent over prediction of total PCB's in perch (P. 75 & Fig. 3-14). EL-1.2h

3. EPA also stated in its public presentation that the only PCB source considered to the Lower Hudson was the PCB's coming over the Troy Dam. While I could not find an explicit statement in the model discussion in the Ecological Risk Assessment Report to this effect, the presentation in the Report appears to be based on the Upper Hudson as the only source to the Lower Hudson. Farley, et. al. state on P. 41 of their report that while the Upper Hudson dominated the loading to the Lower Hudson in the early 1990's, the Upper Hudson loads continued to decrease in the 1990's and by 1997 are estimated to be slightly less than one-half of the total PCB load to the Lower Hudson. EPA is requested to justify assuming all the PCB loading comes from the Upper Hudson in view of the position stated by Farley, et. al. As a minimum, EPA should provide values for the risks assuming that the Upper Hudson load is eliminated and 50% of the PCB load to the Lower Hudson remains into the future as no action to remove these loads appear to be underway. These risk values would put into proper perspective the possible contribution of PCB loads from the Upper Hudson to risks in the Lower Hudson. EL-1.3

4. Much of the information in the December, 1999 reports regarding such items as exposure and toxicity assessment is a copy of similar information in the August, 1999 Risk Assessment Reports for the Upper Hudson. Comments were previously submitted on these sections for the Upper Hudson in the Saratoga County EMC's letter to EPA of September 2, 1999 as corrected by the EMC letter of October 1, 1999. Therefore, the earlier comments will not be repeated here but will be referenced as appropriate. EL-1.4

5. The need to convert EPA model Upper Hudson PCB inputs to the Farley Model from tri+congeners of the EPA model to the homologue distribution of the Farley Model, as discussed in App. A of the Ecological Risk Assessment, is another example of the lack of planning which has plagued EPA's investigation since the beginning. The need for evaluation of the Lower Hudson should have been seen at the start of the study and plans made to obtain data and a model which would fit together without the manipulations of App.A which introduce another undefined level of uncertainty into the calculated risks. Comments on the procedure EPA used to make the extrapolation are given later in comments on Appendix A. EL-1.5

Vol. 2E Baseline Ecological Risk Assessment Comments

Section 3.1.1.1; P.15: Please identify the "few changes" needed to make the Farley Model usable by EPA. Also, EPA is requested to provide an evaluation of the potential effects of starting the model over after each 15-year increment with possibly imprecise initial conditions. Is there the possibility of increasing error in the future predictions? EL-1.6

Section 3.1.1.2; P.16&17: The treatment of PCB body burdens for striped bass throughout this report and the comparison Human Health Risk Assessment Report is puzzling and a major source of concern. The discussion starting on P. 16 focuses on predicting striped bass body burdens in Region 1 because the Farley Model only predicted striped bass body burdens as far as Region 2. This focus on striped bass in Region 1 continues throughout both Reports as calculated striped bass body burdens are only reported for RM152 and RM113 whereas calculations are made for other fish species at RM90 and RM50 also. This focus by EPA solely on Region 1 for striped bass is puzzling because apparently Farley, et. al. did not consider striped bass to be significant in Region 1. The Farley report discusses the migratory behavior of striped bass on P. 78 and following pages of the report. This discussion only mentions striped bass as going as far north as Region 2 which ends at RM73.5, implying Farley, et. al. felt no need to consider Region 1. It must be that some striped bass appear in Region 1 as EPA on P. 16 discusses data at RM152 and RM113. However nowhere in the EPA reports are the data for striped bass shown. EL-1.7
Comparisons of model results to data for other fish species are shown in Fig. 3-12 but not for striped bass. Therefore, there is no way of evaluating the significance of the data on striped bass for Region 1. EPA is requested to provide an explanation of the basis for considering body burdens in striped bass at RM152 and RM113 while excluding striped bass at RM90 and RM50. The Farley, et. al. report would indicate just the opposite. EPA is also requested to furnish information on the number and age of fish samples of each species sampled at the RM's152, 113, 90 and 50 used in the risk analysis so the size of the data base on which the model is based can be evaluated. EPA should also show a comparison of the model results to the data for striped bass as was done for other species of fish. EL-1.7a

The EPA focus on RM152 and RM113 for striped bass is a major concern because of the significance of striped bass to the risk assessments. In the Human Health Risk Assessment Report, Tables 2-6 and 2-7 show that striped bass are the second largest species eaten by anglers. The concentration of PCB's in EL-1.7b striped bass are the highest of any of the fish species ranging up to twice the PCB concentration in brown bullheads which represent the major fraction of fish consumed (52% per Table 2-7 of Vol. 2F). Thus, the product of the percent species in the diet times the PCB concentration makes striped bass as significant as brown bullhead in contributing to the human health risk from eating fish. EL-1.7c

The situation for avian and mammal populations is less clear. While many include fish in their diet, in most cases, but not all, the fish seem to be smaller than striped bass. Because EPA does not provide definitive information, either in the August, 1999 or December, 1999 reports, it is not possible to determine the fraction of the avian and mammal receptors diet that is assumed to come from striped bass but it is likely striped bass contribute in EPA's analysis to at least some of the avian and mammal receptors.

Because of the major significance of striped bass to the risk assessments, it is very important that proper selection be made of the modeled PCB concentrations in striped bass to be used in the risk assessments. The trend for PCB concentration with decreasing river mile shows declining concentrations with decreasing river mile until New York City is reached. Review of Figure 3-18 for largemouth bass from Vol. 2E (the species EPA uses to estimate striped bass PCB concentrations at RM150 and RM113) indicates this decline is not linear but rather decreases from RM113 to RM90, and finally has a much more gradual decline from RM90 to RM50. This trend is important because of how EPA calculates the future yearly PCB concentrations in each fish species used in the human health risk assessment. While not stated, (see comments on Sect. 2.3.1, P. 9 of Vol. 2F) it appears this average is calculated assuming a linear variation with distance. This assumption would overestimate the PCB concentration in largemouth bass and therefore striped bass. Use of a technique such as graphical integration would seem to be a more appropriate way to calculate the average concentration for these species. It is also of note that EPA provides curves vs. time for all fish species at each river mile except for striped bass. EPA is requested to provide the curve for striped bass. But of more consequence is the fact that EPA has chosen to use striped bass concentrations only at RM152 & 113 in both the ecological and human health risk assessments, while using concentrations at RM152, RM113, RM90 and RM50 for all other species in the ecological risk assessment and RM152, RM113, and RM90 in the human health risk assessment. This is done, despite the fact that Farley, et. al. do not even consider striped bass in this region (Region 1) and the likely sharp drop-off in PCB concentration in striped bass from RM152 to RM90.

The approach EPA has taken for striped bass is certainly overly conservative and likely incorrect in calculating the contribution of striped bass to the risk assessments. EPA should recalculate the risks using a more accurate approach. It is recommended that EPA use striped bass concentrations at RM90 in the human health risk assessment, and that the ecological risk to striped bass be evaluated at RM90 and RM50 as was done for other fish species. Whether the lack of striped bass PCB concentrations for these river miles affects the ecological risk to other species at these locations is unclear because EPA has not identified the amount of striped bass in the diets of receptors. In recalculating the PCB concentrations in striped bass, EPA should also define and account for any size restrictions New York imposes on catching and retaining striped bass. Size is related to age and is important because PCB concentration in striped bass decreases with age due to the migratory nature of striped bass as discussed in the Farley, et. al. report on P. 78 and shown by Figs. 3-16 through 3-19 of the report. It is my understanding that NYS limits keeping striped bass to fish 18" or greater. Fish of this size would be expected to be older than 0-2 yr. Age class which exhibits peak PCB concentrations. The excess conservatism in the EPA calculation of PCB concentration in striped bass is illustrated by comparing Table 3-18 of EPA's Vol. 2E with Fig.

3-16 of the Farley report. Table 3-18 shows median values for the years from 1993 to 1997 of 36 to 24 at RM152 and 5 to 3.5 for RM1.13. For fish born in 1987, Fig. 3-16 gives a mean of about 3 for Food Region 2. Fig. 3-19 shows data points ranging from 1 to 2 (one year about 5) over this time period for fish 6 to 17-years-old.

The use of largemouth bass, which are a non-migratory fish as a surrogate for striped bass, a migratory fish, is in itself questionable. More uncertainty in the calculation for striped bass arises from the large difference between the ratios of striped bass to largemouth bass PCB concentrations at RM152 (2.5) and RM113 (.52) (see P.17). EPA is requested to provide an explanation for this difference as there is no apparent reason for it. What are the ratios for RM90 and RM50? It is also of interest that the ratios (and also those for White Perch) have dropped considerably in recent years. Shouldn't any ratio, if used to calculate striped bass concentrations, be based on the more recent data for future predictions? EL-1.7d

Going back to P. 16, EPA is requested to explain why the FISHRAND Model was used for all fish species except striped bass as again the reasons are not apparent. Would using FISHRAND for striped bass eliminate or reduce some of the concerns discussed above? Also, Farley, et. al. make a distinction between ages of striped bass (2-6 yrs. and 6-16 yrs.). Does EPA modeling do this? If not, why not? EL-1.7e

Section 3.1.1.3; PP.17&18: Why is there no discussion of the second part of Table 3-3, the period from 4/91 to 2/96? Table 3-3 does not seem to agree with Fig. 3-2. Table 3-3 shows more penta coming from HUDTOX but Fig. 3-2 shows the opposite. Also, Table 3-3 shows a delta of −18 kg for hexa but Fig. 3-2 shows a delta of about −52 kg. Please explain these differences. It would be helpful if EPA would stick to one set of units as less arithmetic would be required. EL-1.8

Section 3.1.1.4;P.20: The comparison of measured striped bass body burdens to modeled values in Fig. 3-9 is for Region 2 only, whereas EPA uses only modeled values in Region 1 in its health risk assessment. EPA is requested to show a plot of the EPA model results vs. data for Region 1 (RM152 & RM113) so the proper comparison can be made. EL-1.9

Section 3.1.1.5;P.21: Referring to Fig. 3-10, would it make more sense to plot the average of FISHRAND values in Region 1 to compare to the Farley Model as it uses averages for Region 1? EL-1.10

Section 3.1.1.6;P.21: EPA is requested to supply a comparison similar to Fig. 3-12 for striped bass. Why are striped bass often omitted from data comparisons? EL-1.11

Section 3.1.2.2;P.23: Please explain what all the "x's represent on Figs. 3-16 & 3-17. It is also noted Fig. 3-17 shows results only for Region 2 despite the title on the figure. EL-1.12

Section 3.1.2.3;P.24: Comparing Fig. 3-16 to Fig. 3-19, it appears the average value for Region 1 from Fig. 3-19 is about 50% higher for the year 2020 than the value from Fig. 3-16, but for Region 2 it appears Fig. 3-16 gives a somewhat higher value. Please explain why this changeover should occur. Would using the Farley Model throughout give more internally consistent results and thus be preferred over FISHRAND? Again, why is there no forecast for striped bass? EL-1.13

Section 3.2, P.25: The selection of a river mile towards the upper end of each range to represent the range is another example of the excessive conservatism in the EPA assessments. Given the known drop EL-1.14

off of PCB body burden with decreasing river mile, using the body burden at the selected river miles instead of an appropriate average over the river mile segment introduces unnecessary extra conservatism.

Section 3.2.4;P.26: The use of brown bullhead results to represent short-nosed sturgeon makes the risk assessment for the sturgeon very uncertain and of dubious value because of the unknown uncertainty. Also the need to extrapolate the fish PCB concentration data from standard fillets basis to whole body wet weight basis produces more uncertainty of unknown magnitude into the risk assessment, again decreasing the value of the calculated risks. EL-1.15

Section 3.3;PP.27-30: These sections are very similar to those in the August, 1999 Risk Assessment Reports. The comments previously submitted on these items apply to this report as well and will not be repeated here. EL-1.16

Section 4; PP.31-36: These sections are very similar to those in the August, 1999 Risk Assessment Reports. The comments previously submitted on these items apply to this report as well and will not be repeated here. Additional comments come from PP. B-10 & B-11 of Appendix B. The presentation in Section B.2.3.1 on P. B-10 answers the question asked in the EMC's comments to the August, 1999 Risk Assessment Reports as to the amount of chlorine in chlophen compared to PCB's. However, no information is given to justify that the behavior in fish of the chlorine in chlophen duplicates that of PCB's. Page B-11 says "Hatchability was significantly reduced in fish with an average total PCB concentration of 170 mg/kg...." I thought Bengtsson's testing was done with chlophen A50 and not PCB's. This sentence should be corrected to state what was actually tested. The discussion here introduces another factor of about 10 conservatism in the results by not using the 170 mg/kg and 15mg/kg data from Bentgsson study but rather the 15 mg/kg and 1.6 mg/kg data. This further adds to the total excessive conservatism in the EPA risk assessments (also applies to other fish species in Section B.2.3 of Appendix B). Does this new conservatism mean that EPA now considers the ecological risk evaluation of these fish species in the August, 1999 risk assessment to be wrong? EL-1.17

Section 5.;P.37-55: Comments previously made on the August 1999 ERA regarding the over conservatism in EPA's risk characterization apply to the report as well and will not be repeated here. EL-1.18

Section 5.2.1.9;P.43: As previously questioned, EPA is requested to explain why EPA reports Measurement Endpoints for striped bass only for RM152 and 113 and why these river miles should be considered at all for striped bass. EL-1.19

Section 5.2.4.1;P.45&46: In view of the unquantified uncertainty in the calculation of body burdens in the shortnosed sturgeon and the positive statements about the health of the shortnosed sturgeon in the last paragraph on this page, why does EPA insist on putting forth a negative risk evaluation for the shortnosed sturgeon? This question also applies to white perch as the discussion on P. 46 again indicates a healthy situation and the discussion at the end of the paragraph represents speculation based on only extremely conservative calculations and is inconsistent with the facts shown by the field studies. EL-1.20

Section 5.4.3;P.50,Section 5.5.3.1;PP.53&54,Section 5.3.3.1;PP.47&48A: EPA is requested to provide information on what trends were seen in the Christmas bird counts. This information would be helpful in assessing what is happening to the health of birds in the region. EL-1.21

Section 5.7.3.1;P.57: The discussion in this paragraph leads to the conclusion that not enough raccoons would be affected by the PCB's in the Hudson to have an impact on the raccoon population so why is EPA insisting on singling out the potential risk to those few raccoons that might be affected? EL-1.22

Section A.2;P.A-2: It is not clear what is meant by the phrase "duplicate samples are equivalent." Does this mean the PCB data from the duplicate samples are exactly equal? If not the case, why weren't the duplicate GE samples averaged as were the EPA duplicates? EL-1.23

Section A.3;P.A-3: EPA is requested to provide some discussion of what factors could effect the geochemical processes and why these factors are not expected to change to justify the assumption made here. The discussion of the steps taken is confusing in that it appears the first step described applies to Factor 2 and the second step to Factor 1. Is this correct? EL-1.24

Section A.3;P.A-3 and Figs.A-1toA-5: The EPA mean values shown on these figures for the TID (presumably from years prior to 1996) agree more with GE means (see Fig. A-9) for post 1996 data and not at all with GE means for prior 1996 data. Since the GE data set for the TID is much larger (225 samples prior to 1996 and 293 samples after 1996) than the EPA data set of 4 to 12 samples, the use of the EPA data at the TID to calculate the ratio for homologues at Waterford (or the Troy Dam) is very questionable. Shouldn't the GE data be used to calculate the factors in Table A-2? EPA is requested to address this issue regarding the calculation EPA used to get input to the Farley Model. EL-1.25

Section A.3;P.A-4: EPA is requested to provide the citation of the data used as the basis for the statement that there is little evidence of decline in PCB loads at the TID post-1995. Is this still true based on 1999 data? EL-1.26

Section A.3;P.A-4: See comment above on A-3 and Fig. A-1 - A-5 questioning validity of factors given in Table A-2. Also, why should these factors stay constant for 40 years? EL-1.27

Section A.5,P.A-7: The basis for the statement at the top of the page about releases from Baker Falls is unclear. Weren't the major releases from Baker Falls post 1990? If so, EPA is requested to clarify why the post 1990 releases are not of concern. EL-1.28

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Section 2;PP.5-21: Comments previously submitted on Section 2 of the August, 1999 Risk Assessment apply to this report as well and will not be repeated here.

Section 3;PP.23&24: Comments previously submitted on the August 1999 risk assessment regarding non-cancer toxicity values and cancer toxicity apply to this report and will not be repeated here.

Section 2.3.1;P.8: Comments given above on the Ecological Risk Assessment regarding the EPA approach to calculating PCB concentrations in striped bass apply here also.

Section 2.3.1;P.9: The comment on Section 3.2, P. 25 of the Ecological Risk Assessment applies here also to the selection of river miles to represent sections of the river as do comments about selecting a more appropriate way to average values than straight linear averages.

EG-1

COMMENTS OF GENERAL ELECTRIC COMPANY ON

Hudson River PCBs Reassessment RI/FS Phase 2 Baseline Ecological Risk Assessment for Future Risks in the Lower Hudson River

February 4, 2000

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1.0 Executive Summary and Introduction

General Electric Company (GE) submits these comments on the *Hudson River PCBs Reassessment RI/FS Phase 2 Baseline Ecological Risk Assessment for Future Risks in the Lower Hudson River (Future Risk ERA)*, issued by the U.S. Environmental Protection Agency (EPA) on December 29, 1999. EG-1.1

PCBs have been present in the Hudson River environment for 50 years, and at significantly higher levels than are found today. For the last 25 years, PCB concentrations in fish and wildlife in the Hudson River have been declining. During this period, other pollutants in this river have generally declined and the management of wild populations, particularly fish, has materially improved. EPA has studied and analyzed Hudson River PCBs for the last 10 years and, even before this reassessment began, the Agency was fully familiar with the river's aquatic resources through its involvement in the issuance of the first water discharge permits to power plants on the Lower Hudson River in the 1970s.

As a result of public, scientific and regulatory interest in the environmental health of the Hudson River, volumes of data on fish, wildlife, sediment and water quality have been collected over the last 25 years. The data documenting conditions in the Lower Hudson for this period are particularly rich for fish.

When it began its ecological risk assessment for the Lower Hudson, EPA had at its disposal the entire record of a living river laboratory, a quarter century in length. These data, collected at a time when PCB levels were higher, provided an unusual opportunity to explore relationships between PCB levels and the sustainability of populations of fish, birds, and mammals. For a number of animal populations, there was sufficient data for EPA to examine the potential for impacts due to PCBs and to determine whether at lower future levels it is reasonable to suggest that animal populations would be affected.

EPA could have built on this extensive historical record to produce a first-class ecological risk assessment. Unfortunately, the Agency did nothing to collect data on wildlife or biotic populations in the Lower Hudson over the past 10 years and disregarded the mine of data which it examined in the 1970s power plant cases and which has grown larger with new data in each year since. EPA likewise ignored the extensive work of the U.S. Fish and Wildlife Service and the National Marine Fisheries Service in addressing the most obvious, large-scale, Hudson-related biological emergency of the last 25 years – the late '70s-early '80s crash of the coastal striped bass population, to which the Hudson stock contributes, an event for which PCBs were considered, but rejected, as a cause, before the real cause, overfishing, was established (Atlantic States Marine Fisheries Commission [ASMFC], 1990). EG-1.2

What EPA produced is superficial, theoretical speculation that implies future risks to wildlife populations without providing evidence of past effects and while ignoring clear evidence that key wildlife populations are, in general, healthy and the communities diverse. For many of the fish and wildlife species evaluated by EPA, the facts clearly contradict EPA's conclusions. For example, the facts demonstrate that:

- The white perch population of the Lower Hudson River is relatively stable and that the striped bass and shortnose sturgeon populations have increased dramatically since the 1970s. The upward trend in striped bass is especially important because EPA has concluded that risks to this species are especially high.

- Although EPA predicts that PCB levels in kingfishers range from 4 to 280 times the level EPA says may pose a risk, a kingfisher population is documented by EPA as successfully reproducing in the Lower Hudson.

According to reports from various sources, including the New York State Department of Environmental Conservation (NYSDEC), the U.S. Fish and Wildlife Service (USFWS), the Audubon Society and others, the populations of other species are present and growing, including bald eagles, which have returned to the Hudson after an absence of more than 100 years and, contrary to EPA statements, are successfully reproducing in the Lower Hudson River; mallard ducks, whose population is characterized as "demonstrably secure," great blue herons, and raccoons. In some cases, EPA's report does not even acknowledge these facts, and where it does, it discounts the data for no legitimate reason.

EPA's approach, including selective use of data, discounting information in a manner that is inconsistent with the Agency's guidance and scientifically defensible practices, and uncorroborated speculation about risks for which no site-specific evidence exists, is highly misleading to the public and fails to provide regulators with a risk assessment that is useful for choosing the most appropriate, scientifically defensible management options for the Upper Hudson River. There is no sound basis to accept EPA's analytical approach as plausible when it at dramatic variance with the facts.

The objective of the risk assessment should be to provide data and analysis on which to base remedial decisionmaking for the Upper Hudson River. To the extent that an examination of risks in the lower river is appropriate, the assessment must be useful to the remedial manager as:

- A sound and reliable description of the effects of current and future PCB exposures emanating solely from the Upper Hudson on biota in the Hudson River Valley.
- A foundation for projecting the responses of those biota to alternative remedies taking into account the effects of chemicals other than PCBs and PCBs whose source is not the Upper Hudson River.
- A sound technical underpinning for comparing the ecological benefits gained through remediation to the ecological costs of implementing remedial actions.

Like EPA's Baseline Ecological Risk Assessment (BERA), the Future Risk ERA is simply a screening-level assessment. As such, it does not reflect acceptable scientific practice, is excessively conservative, and is insufficient for use in determining the effect of a remedy or selecting an appropriate remedy.

The Future Risk ERA repeats critical flaws identified by GE and others in the BERA including:

- Inadequate consideration of population vs. individual-level effects.
- Ignoring or dismissing site-specific data.
- Failure to use a weight-of-evidence approach correctly.
- Use of excessively conservative assumptions concerning exposures and effects.
- Interpretation of exceedances of Sediment Effects Concentrations and other sediment quality guidelines as measures of actual effects.
- Inappropriate use of the TEQ approach.
- Failure to evaluate the usefulness of or even cite the expert review of PCB effects on fish prepared for NOAA.
- Mathematical errors.

Rather than altering the assessment procedures to minimize or eliminate the identified flaws, EPA used exactly the same approach in the Future Risk ERA. Consequently, this assessment suffers from the same flaws as the BERA.

In the following sections, GE provides comments on EPA's Future Risk ERA, specifically addressing:

- The Future Risk ERA does not provide the information necessary to support remedial action decisions.
- EPA has repeated critical flaws identified in previous reviews of the BERA.
- The Future Risk ERA does not conform to best scientific practice.

- The models used to project future PCB concentrations in media have been inadequately reviewed and are seriously deficient.
- Available data on ecological resources of the Lower Hudson River were not used and directly contradict EPA's conclusions.
- EPA's approach to effects assessment for fish and wildlife is excessively conservative, relies on a small subset of the available data, and ignores or improperly interprets key studies.

By concluding that PCBs may or may not pose risks to wildlife populations and offering no evidence of past effects from PCBs, EPA failed to abide by the most fundamental tenet of its own internal guidance -- it did not quantify impacts on wildlife populations. The Agency failed to use realistic exposure scenarios, failed to consider effects that might be attributable to contaminants other than PCBs, and failed to distinguish PCBs from the Upper Hudson and those originating in the mid-Hudson or elsewhere. This final point is most important. EPA is preparing to make a remedial decision for the Upper Hudson River. If it intends to assert that its decision would benefit lower parts of the river as well as the Upper Hudson, it must be able to show that it has the ability to distinguish between one PCB source and another. There is no indication in this report or any report that the agency has thus far produced for this project, that EPA can do that with any scientific certainty.

Therefore, this report should be given no weight in the Agency's deliberations over the appropriate remedial strategy for the Upper Hudson River.

2.0 The Future Risk ERA does not provide the information necessary to support remedial action decisions

As we have previously explained, it is inappropriate for EPA to base a remedial decision for sediments in the Upper Hudson on risk reduction to biota in the Lower Hudson.[1] Should EPA nevertheless persist in examining risks in the Lower Hudson, it is clear that, like the BERA, the Future Risk ERA in its present form will not provide useful information for the risk manager. EG-1.3

To support remedial action decisions for the Upper Hudson River, the Future Risk ERA must be based on an objective evaluation of all available information concerning the risks to ecological resources posed by present and future exposures to PCBs. As described in the following sections of GE's comments, this information should include:

- Site-specific data concerning PCB and other chemical exposures and effects on populations and communities based on a variety of independent lines of evidence.

- Estimates of concentrations of PCBs in sediment, water, and biota based on properly calibrated and verified models.

- A thorough review of all available data.

The Future Risk ERA fails to include any of the above information. It is based on inadequately verified models, excessively conservative Toxicity Quotients (TQs) based on a limited evaluation of literature-derived test data, a focus on individual organisms, and a failure to consider important and relevant site-specific data. Therefore, the Future Risk ERA cannot support scientifically sound decisions about remedial actions on the Hudson River.

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[1] See Nov. 6, 1997 letter from Angus Macbeth to Richard Caspe; May 5, 1998 letter from Angus Macbeth to Douglas Fischer

3.0 EPA has repeated critical flaws identified in GE's and others' review of the Baseline ERA

GE's and other's comments on the BERA identified a number of critical flaws, which render the document inadequate for supporting remedial decisionmaking for the Hudson River. In the Future Risk ERA, EPA has not addressed *any* of these flaws. EG-1.4

3.1 Inadequate consideration of population vs. individual-level effects

As noted in GE's comments on the BERA, decisions concerning remedial action needs for the Hudson River must consider:

(1) Whether the sustainability of exposed biological populations and communities is being threatened by the presence of PCBs in Upper Hudson River sediment.

(2) Whether the positive effects of a particular remedy will be greater than any negative ecological effects of carrying out the remedy. EPA's Risk Management Guidance clearly states that populations are the appropriate level of ecological organization for assessment. (EPA 1999a, Ecological Risk Assessment and Risk Management Principles for Superfund Sites. USEPA Office of Solid Waste and Emergency Response, Washington, D.C., Directive 9285.7-28P).

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 the death or impairment of some individuals that occurs due to natural and anthropogenic stressors. Even if statistically significant reductions in survival, growth and reproduction of some individuals are observed, such data alone cannot be used directly to estimate adverse effects to populations, communities, or ecosystems (Forbes and Calow, 1999). Survival, growth, and reproductive rates are interrelated in complex ways, and apparent adverse changes in one of these factors (e.g., a reduction in fecundity) are often offset by compensatory changes in others (e.g., increased growth and survival of young).

In the Future Risk ERA, EPA indicates that it considers population-level effects by comparing the magnitudes of TQs over the 25-year modeling period to the life spans of the receptor species (p. 9). EPA asserts that population-level effects are more likely if the TQ exceeds 1 for the life span of a species. This approach does not consider compensatory processes and is not supported by any published studies. In fact, EPA did not even implement the approach described on page 9. The risk characterization in Section 5 does not even discuss the life spans of the various receptor species, much less compare them to the duration of the modeling period.

3.2 Ignoring or dismissing site-specific data

GE's comments on the BERA noted that EPA had not examined or incorporated site-specific data such as biological surveys, whole-media toxicity tests, or reproductive effects studies. 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." This is particularly true where, as in the Lower Hudson, PCB concentrations in biota have been declining over a long period of time. GE's previous comments included a comparison between the data used by EPA and the data collected by the Department of Energy for the Clinch River ecological assessment. Table 1 presents a similar comparison between the Future Risk ERA and the Clinch River ERA. Whereas the BERA included limited site-specific data concerning the effects of PCBs on Hudson River biota, the Future Risk ERA includes *no* data specific to the Lower Hudson River. EG-1.5

Like the BERA, the Future Risk ERA ignores or discounts existing site-specific data. For the Lower Hudson, extensive data on the condition of ecological resources are available, especially for fish. As in the BERA, EPA explicitly discounts these data for risk assessment, arguing on page 45 that reproduction and recruitment of fish might be impaired by exposure to PCBs, even though populations are increasing. The implication is that only comparisons between measured or modeled exposures and Toxicity Reference Values (TRVs) are relevant. This conflicts with established principles of ecological risk assessment (e.g., Suter, 1993) and with EPA's own Superfund guidance (EPA, 1997a).

3.3 Use of excessively conservative assumptions concerning exposures and effects

EG-1.6

In its comments on the BERA, GE noted that, even accepting the proposition that the TQ approach provides useful information for an assessment, EPA's application of TQs in the BERA provides highly inflated risk estimates that are not useful in remedial decisionmaking. Both the exposure assessment and the effects assessment used by EPA employed data, models, and assumptions that are inappropriate for site-specific assessments.

Like the BERA, the Future Risk ERA employs water and sediment-quality guidelines designed to be protective such that exposure concentrations *below* the criteria can be confidently presumed to be safe. Site-specific studies of the type EPA chose *not* to perform (such as those used in the Clinch River ERA) are required to determine whether exposures that exceed the guidelines are actually causing any adverse effects. Similarly, in selecting TRVs for use in assessing effects on fish and wildlife, EPA consistently chose the lowest value from the range of available test results, and often adjusted those values even lower with 10x uncertainty factors. The resulting TRVs are generally lower than any exposure concentrations at which effects have been observed in any test system. We may be confident that exposures that are lower than the TRVs will have no adverse effects, but additional information – again, information that EPA chose not to collect – is required to determine whether adverse effects will occur at the exposure levels actually seen in the lower Hudson.

3.4 Interpretation of exceedences of Sediment Effects Concentrations and other sediment quality guidelines as actual measures of effects

GE's comments on the BERA included an extensive discussion of the lack of validity of NOAA's Sediment Effects Concentrations (SECs) as measures of actual effects on benthic invertebrate communities. GE provided a thorough review of the inherent limitations of the SECs and other generic sediment quality guidelines, including statements from the *developers of the guidelines themselves* that these values are intended as screening values, not as measures of effects. In the Future Risk ERA, EPA continues to use generic sediment-quality criteria as the primary measure of risks to benthic invertebrates. EG-1.7

3.5 Inappropriate use of the TEQ approach

GE previously noted that the toxicity equivalency (TEQ) approach, in its current state of development, is a screening approach rather than a primary assessment approach. The developers of the approach themselves have expressed caution concerning improper use of the TEQs. EPA has inappropriately handled non-detect readings of PCB congeners by using full detection limits for non-detect values, even though standard risk assessment practice typically involves using one-half of the detection limit for non-detects and in the human health risk assessment a value of 0 was used for non-detect. As noted by GE in comments on the BERA, EPA has assumed that nondetects of BZ#126 are present at the detection limit. This results in the TEQ-based risk assessments being driven by a chemical not even detected (non-quantified concentrations of BZ#126). EG-1.8

In the case of fish, the review performed for NOAA of the TEQ approach concluded that, because of insufficient understanding of inter-species variations in sensitivity to dioxin-like compounds, the approach should not be applied to Hudson River fish species (NOAA, 1999).

In these circumstances, the Future Risk ERA should not employ the TEQ approach.

3.6 Failure to cite the expert review of PCB effects on fish prepared for NOAA

EG-1.9

In its previous comments, GE noted that NOAA commissioned a review by Dr. Emily Monosson of effects of PCBs on fish, with specific reference to Hudson River fish populations (NOAA, 1999). The review concluded that adverse effects on early life stages of Hudson River fish species might occur at tissue concentrations exceeding 5 ppm (whole body, wet weight), and that physiological effects on adult fish might occur at tissue concentrations exceeding 12.5 ppm (whole body, wet weight). One might question these values in light of the site-specific data, but in any event, they are far higher than the TRVs used by EPA in both the BERA and the Future Risk ERA.

This review was published by the same NOAA office that published the report on Sediment Effects Concentrations that EPA used in its assessment of risks to benthic invertebrates. Both reports were issued in March, 1999. There is no indication that EPA evaluated the applicability of the Monosson study. EPA's failure to examine the Monosson review violates common sense and the Agency's own guidelines, which require the EPA to consider all relevant evidence when performing its risk assessments. Will EPA choose the results that give the lowest possible acceptable PCB levels regardless of the quality of the data? This is scientifically indefensible.

4.0 The ERA for Future Risks does not conform to best scientific practice

EG-1.10

Like the BERA, the Future Risk ERA relies almost exclusively on "Toxicity Quotients" (TQs), i.e., comparisons between measured or modeled exposure concentrations and concentrations believed to be potentially harmful to organisms. Such screening-level data and models, as applied by EPA, are deliberately designed to be conservative, i.e., to minimize the possibility that any potential adverse effects will be missed. They necessarily 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 the screening-level calculations performed by EPA "would not be technically defensible." As noted by GE in comments on the BERA, a scientifically defensible ecological risk assessment should use a variety of independent techniques for measuring and characterizing ecological risks, e.g.:

- 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.

These techniques are described in EPA's Guidelines for Ecological Risk Assessment (EPA, 1998) and Ecological Risk Assessment Guidance for Superfund (EPA, 1997). 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.

As noted in GE's comments on the BERA, these techniques have been successfully applied at other large Superfund sites such as the Clark Fork River (Canfield et al., 1994) and the Clinch River Study Area, Tennessee (Cook et al., 1999). Table 1 contrasts the assessment performed for the Clinch River Study Area to the EPA's Future Risk ERA. In addition to the TQ approach used by EPA, 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. EPA had ample time to perform similar studies for the Hudson River, but chose not to do so.

EPA's approach to evaluating the small amount of field data that were discussed in the Future Risk ERA also fails to meet accepted standards of scientific inference. In the Clinch River assessment, all of the lines of evidence were considered together in making determinations concerning the existence and magnitude of risks. Lack of concordance between different types of evidence relevant to a given endpoint was taken to indicate that the risk assessment was inconclusive. In the Future Risk ERA, EPA discounted all lines of evidence other than TQs, arguing that the failure of field data to support the TQs simply showed that other factors were masking the adverse effects caused by exposure to PCBs. Such an approach is scientifically indefensible.

5.0 The models used to project future PCB concentrations in water, sediment, and biota have been inadequately reviewed and are seriously deficient

EG-1.11

All three of the models used by EPA in the exposure assessment component of the Future Risk ERA have deficiencies that compromise their value for projecting future PCB concentrations in sediment, water, and biota. Two of these models – EPA's HUDTOX and FISHRAND models – were recently revised, and it is the modified models that were used in the risk assessments. Our comments are based on oral presentations of the modified models to the peer reviewers of EPA's Baseline Modeling Report (BMR), and we reserve our right to supplement these comments after further review of the revised BMR, which EPA just released in late January 2000.

5.1 EPA Upper Hudson River model (HUDTOX) used to predict PCB loads to the Lower Hudson River

EG-1.12a

The use of the EPA Upper Hudson River model (HUDTOX) to predict PCB load passing Troy to the Lower Hudson River relies on the presumption that this model accurately predicts the time trends of PCB concentrations at Troy. As detailed in GE's Comments on the BMR (GE, 1999), GE has concerns that HUDTOX has not been properly and fully developed and is inadequate for predicting future PCB concentrations. One of the most significant of these concerns relates to the model's ability to describe PCB fate downstream of the Thompson Island Dam (TID). The equations and coefficients describing sediment transport in the 34 miles between the TID and Troy are inconsistent with the equations and coefficients used in the Thompson Island Pool and inaccurately represent the processes critical to PCB fate in the river (GE, 1999).

The inaccuracy of the HUDTOX-predicted PCB load to the Lower Hudson River is exacerbated by the necessity to convert the HUDTOX PCB metric (PCBs with 3 or more chlorine atoms; tri+) to the homolog characterization of PCBs used in the Farley et al. (1999) Lower Hudson River model. This conversion was made using factors that may EG-1.12b not be generally applicable because they were developed from 1993 TID and Waterford data that were influenced by the 1991-1993 elevated upstream source.

The ratio of each PCB homolog to tri+ was calculated in two steps. The first step was to calculate the seasonal averages of these ratios for all of the measurements made at the GE TID West sampling station between 1991 and 1998. The second step was to convert these ratios to equivalent ratios at Waterford. This step was accomplished using the differences in PCB composition between the TID and Waterford observed in the 1993 EPA Phase 2 sampling program. This assumes that the differences observed in 1993 apply over all times, a presumption that was never tested. There are several reasons why the presumption may be invalid. First, the 1993 EPA Phase 2 TID station was located along the west shoreline 200 feet upstream of the GE TID West station. Both stations provide poor representations of the overall PCB flux passing TID and they are not replicate locations. Second, the 1993 EPA Phase 2 data reflect a period in which PCB load from the vicinity of Hudson Falls was a significant component of the PCBs passing the TID. This condition is not representative of the entire 1991 to 1998 period; a period over which conditions have transitioned from one in which the Hudson Falls source dominates to one in which sediment sources dominate. Thus, a ratio developed from a snapshot in time may not be applicable to the full historical period or to the future.

5.2 Farley et al. Lower Hudson River model used to predict Lower Hudson River water and sediment PCB concentrations EG-1.13

EPA has used the Farley et al. (1999) Lower Hudson River model without having conducted a critical review to determine its validity and accuracy. EPA has not developed an understanding of the veracity of the predicted water and sediment PCB concentrations and the relationship of those concentrations to the various PCB sources. Because the predictions are the basis for the risk calculations, the lack of understanding of model veracity undermines the utility of the risk assessment.

Concerns about model veracity are pertinent in view of apparent deviations between the model and site data. These deviations raise questions about the ability of the model to accurately describe the relative contributions of external and sediment PCB sources and to accurately predict time trends.

The model is biased toward lower chlorinated PCBs relative to the observed PCB composition. For example, data indicate that dichlorobiphenyl constitutes about 20 percent of the sum of di- through pentachlorobiphenyl present at river mile 125, whereas the model computes that it constitutes about 40 percent. (See Figure 3-2 of Farley et al. 1999). Dichlorobiphenyl is a reasonable tracer of the Upper Hudson River source and the upward bias of the model may indicate underestimation of the rate at which the Upper Hudson River source declines as water moves downstream.

The water column and sediment model-data comparisons were limited to a single year (1993), an inadequate duration to test the model's ability to predict time trends accurately. Water column data for comparison to the model were available for only 3 locations over the more than 150 miles of river. The model predicts PCB levels that compare poorly with these data. The model's predictions are significantly lower than the summer data and do not predict the extent of concentration decline from Troy to the mid-river in April (Figure 3-5 of the Future Risk ERA report). These differences suggest that the model underestimates sources within the lower river (probably local sediments) and under estimates the loss rate of Upper Hudson River PCBs. The comparison of model and surface sediment data (Figure 3-7 of the Future Risk ERA report) excludes important data (i.e., the USEPA Phase 2 high resolution cores) that indicate that the model under predicts 1993 surface sediment PCB levels.

5.3 Models used to predict PCB concentrations in Lower Hudson River fish (FISHRAND and Farley et al.)

EG-1.14

PCB concentrations in fish in the Lower Hudson River were computed using two models, FISHRAND (EPA, 1999b) and Farley et al., (1999). Each model was used to predict

PCB concentrations in selected species in the Lower Hudson River (Table 2). These models are similar, in that they are mechanistic bioenergetic-based simulation models of bioaccumulation in aquatic organisms. However, they differ in some of the formulations used to describe the key processes, and the impacts of these differences have not been evaluated. In addition, as mentioned above, EPA has used the Farley, et al., (1999) Lower Hudson River model without having conducted a critical review to determine its validity and accuracy. Thus, the validity of the predicted fish PCB concentrations has not been fully evaluated, undermining the utility of the risk assessment.

A preliminary review of Farley et al. (1999) and FISHRAND (EPA, 1999b) has revealed several weaknesses in parameterization and calibration of the models. These are divided into three categories: food web structure, calibration, and other issues associated with model development.

5.3.1 Food web structure

FISHRAND and Farley are inconsistent in their characterization of the food web.

Fish can accumulate PCBs from both the surface sediments and the water column. PCB concentrations in the sediments and water column may exhibit different rates of natural recovery and different responses to remedial activities. Thus, the realism of the projected fish concentrations is affected by the accuracy of the presumed food web. The two bioaccumulation models of the Lower Hudson River are inconsistent in their descriptions of contaminant sources to the food web. FISHRAND includes both sediment- and water column-associated food webs for the resident fish and the striped bass, based on the fact that the striped bass concentrations are computed from the largemouth bass concentrations, and the statement that the parameterization of FISHRAND is the same as in the Upper Hudson River. In contrast, Farley includes only a water column source to the food web of the striped bass. To develop reliable projections, this inconsistency must be reconciled, and the final food web structure must be considered in light of the available information.

Striped bass migration patterns are described inaccurately.

Largemouth bass is a resident fish, while striped bass is migratory. Because predicted largemouth bass PCB concentrations are used to estimate striped bass concentrations, the contribution to the striped bass of PCBs originating south of Region 1, that is, in the estuary, is underestimated in the ERA. Projected concentrations in the striped bass are determined by the changes in the loads from the various PCB sources in the Lower Hudson River. Migratory striped bass migrate between the coastal ocean, the river and the Harbor and are therefore exposed to PCBs from many sources. Inaccurate description of the relative contributions of each source can therefore lead to inaccurate projections.

5.3.2 Calibration

Farley does not compute realistic temporal trends in striped bass PCB levels.

Computed total PCB concentrations in striped bass ages 6-16 years are consistently lower than the data prior to 1992 and generally greater than the data after 1992 (Figure 3-9 of the Future Risk ERA report). This is important because it indicates that the rate of natural recovery is not being accurately modeled. It may be due to inaccuracies in the food web structure, in particular the contribution of sediment and water column PCBs, or to inaccurate temporal trends in water column PCBs computed by the fate model. EG-1.15a

Response of model fish at RM 152 to the events of 1991 is unrealistic.

At river mile (RM) 152, lipid-based PCB concentrations in largemouth bass, white perch, brown bullhead and yellow perch increased in 1992 following the Allen Mill event and decreased thereafter (Figure 3-12a of the Future Risk ERA report). In contrast, model calculations for these fish exhibit no response to these events. This suggests that exposure concentrations and food web structure may be inaccurate. EG-1.15b

FISHRAND computations on a wet weight and lipid basis are inconsistent.

For largemouth bass and white perch at RM 152, wet weight-based concentrations computed by the model run through the error bars and exhibit limited bias with respect to the data. In contrast, lipid-based levels are generally lower than the data (Figure 3-12a of the Future Risk ERA report). This suggests that the lipid contents are not representative of the fish for which PCB data are available. EG-1.15c

5.4 Other model development issues

Size of fish modeled may not reflect consumption patterns by ecological receptors.

To develop a relationship between largemouth bass and striped bass concentrations, EPA compared concentrations in fish greater than 25 centimeters (cm) in length, because those are consumed by anglers. It is unclear what size classes are used in the model calculations. Size classes consumed by wildlife should be used. EG-1.16a

Fish growth rates are not site-specific.

Fish growth rates can control the computed PCB concentrations. For example, if growth rates are unrealistically high, then the predicted degree of bioaccumulation is likely to be unrealistically low. To calibrate a model with less bioaccumulation, the exposure concentrations must be increased. This is done, for example, by increasing the contribution to the food web from more contaminated sources. Thus, realistic growth rates are needed to characterize the contaminant sources to the food web as accurately as possible. It is our understanding that FISHRAND employed generic growth rates; site-specific data should be used when available. EG-1.16b

6.0 Available data on ecological resources of the Lower Hudson directly contradict EPA's conclusions EG-1.17

Substantial data are available concerning the condition of the ecological resources of the Lower Hudson River. Information concerning long-term trends in the abundance of various fish species, including three of the receptor species considered in the Future Risk ERA, are especially complete. This information directly contradicts EPA's conclusions concerning the risks posed by future exposures to PCBs.

6.1 Benthic macroinvertebrates

Based on the comparison of modeled Lower Hudson River PCB surface water and sediment concentrations with screening criteria and guidelines, EPA contends that there is the potential for adverse effects on benthic organisms. As noted in GE's comments on the BERA, NYSDEC (1993) found that the abundance of pollution-intolerant filter-feeding macroinvertebrates has increased throughout the Hudson River as a result of improved water quality since 1972. Hudson River macroinvertebrate communities are comparable in structure to those in other New York rivers, and currently considered slightly impacted based on the type of species present in the river (Plafkin et al., 1989; NYSDEC, 1993).

In addition to improvements at several sites in the Upper Hudson River, NYSDEC (1993) noted improvements in macroinvertebrate populations in the Lower Hudson River over the last two decades. The number of pollution-sensitive species increased below Troy Dam at Castleton and Saugerties between 1973 and 1983. Numbers declined from 1983 to 1991, but 1991 values were still higher than those of the early 1970s. These data demonstrate that: (1) the benthic community improved even in the presence of PCB concentrations greater than levels currently exhibited; and (2) changes in species composition appear to occur independent of changes in PCB levels.

There is more evidence that the improvements in macroinvertebrate communities of the Hudson River noted by NYSDEC (1993) are likely independent of any changes in PCB concentrations. Exponent (1998a,b) found that the macroinvertebrate communities of the Upper Hudson River had abundant populations and high species richness (i.e., total number of taxa), in areas with higher PCB concentrations. These results together with the results of macroinvertebrate surveys conducted by EPA (as reported in the BERA) suggest that PCBs currently have no major impact on macroinvertebrate communities of the Hudson River. Because it is highly unlikely that PCB concentrations in the Lower Hudson River reach the high concentrations in study area sediments sampled by Exponent (1998a,b), it can be concluded that there is no apparent risk, present or future, from GE-associated PCBs to macroinvertebrates of the Lower Hudson River.

6.2 Fish

The Hudson River utility companies recently completed a comprehensive assessment of the impacts of power plants on the biological resources of the Hudson River (Central Hudson Gas & Electric Corporation et al., 1999) as part of a Draft Environmental Impact Statement (DEIS). The assessment summarizes 25 years of data on the distribution and abundance of the major fish populations inhabiting the Lower Hudson. Trends in the abundance of 16 fish species were evaluated, including striped bass, white perch, and shortnose sturgeon. The major conclusions from the DEIS are summarized below.

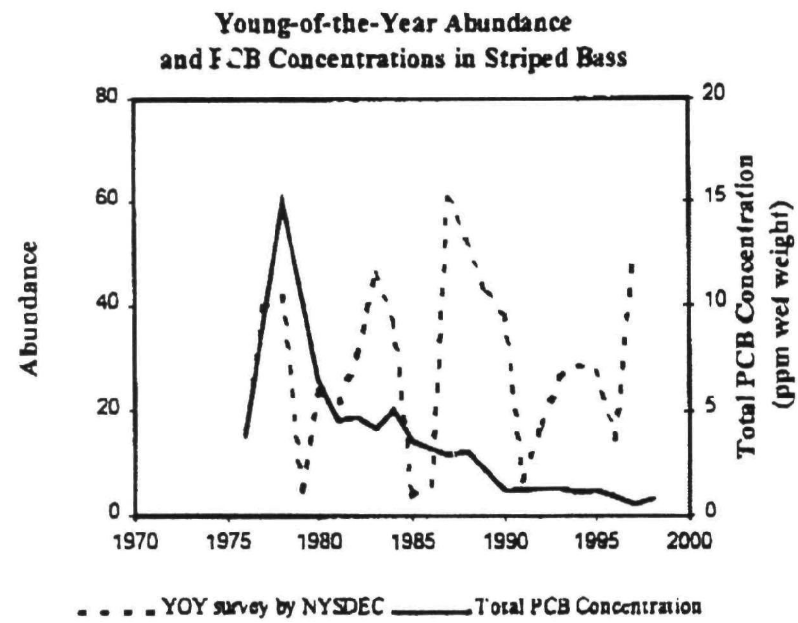
6.2.1 Striped bass

Information on the abundance of striped bass life stages in the Lower Hudson is available from sampling programs conducted both by the utility companies and by NYSDEC. These data include a river wide ichthyoplankton sampling program, two beach seine surveys, a trawl survey, and a mark-recapture program. NYSDEC also samples striped bass in 7 bays around western Long Island Sound, conducts a haul seine survey to obtain information on the length, age, sex distribution, and mortality rates for the adult population, and monitors the striped bass bycatch in the American shad fishery. The data derived from these programs represent one of the most extensive data sets available for any estuarine fish species.

As documented in the DEIS, large year classes of striped bass, as measured by the utility and NYSDEC beach seine surveys, were produced in 1977, 1978, 1983, and 1984. When these fish reached reproductive age in the mid and late 1980s, numbers of striped bass larvae collected in the utilities' river wide ichthyoplankton survey increased dramatically. Correspondingly strong year classes, as measured in the beach seine surveys, were produced in four consecutive years, from 1987 through 1990. The abundance of adult striped bass increased steadily from 1980 through the mid-1990s. According to the DEIS, the Hudson River striped bass population may now have reached its carrying capacity. Striped bass are, according to the DEIS, now a dominant predator in the estuary, controlling the abundance of many other fish species.

In addition to the utility-sponsored studies, research on the migratory behavior of striped bass has shown that adult striped bass collected immediately below Troy Dam (RM 152) appear to be a cohort of nonmigratory male fish that have resided in fresh water for their entire lifetimes (Secor, 1999). These fish, which frequently have higher PCB body burdens, are unrepresentative of the population as a whole. Fish that migrate annually between marine and fresh water, and probably dominate the spawning stock, have much lower body burdens. The adult females sampled by NYSDEC in April and May, in the mid and lower estuary, provide the most relevant data concerning PCB concentrations in spawning female striped bass and are the only data that should be used for risk assessment.

Figure 1 compares time trends in PCB concentrations in adult female striped bass, collected during the spawning season in the mid and lower Hudson, to trends in the NYSDEC striped bass juvenile index. This index, which is a measure of the density of juvenile striped bass present in the Hudson River estuary during the late summer and early fall, has been accepted by the Atlantic States Marine Fisheries Commission (ASMFC) as a valid indicator of year-class production in the Hudson River striped bass population and is used in the ASMFC's annual striped bass stock assessments. From 1976 through 1997, the annual production of young striped bass from the Hudson has fluctuated without trend; PCB concentrations in the spawning females that produced these fish have declined steadily over the same period. The ASMFC concluded that "[g]iven the very healthy status of the Hudson River stock, which is well documented to have relatively high tissue concentrations of PCBs, it would appear that such levels ... may not pose a threat to striped bass from a population biology perspective" (ASMFC, 1990). Clearly, there is no evidence that high maternal PCB concentrations in the late 1970s adversely affected striped bass recruitment. The obvious implication of this result is that future, lower maternal concentrations will similarly have no effect on striped bass recruitment.



Total PCB Concentration: Average +/- 2SE for female striped bass (>1000g, April/May fillet)
Source: hudorg.dbf, NYSDEC database
Abundance: Geometric mean number per 200' seine haul for 6 week sampling period +/- 2 SE
Source: Draft Environmental Impact Statement, December 1999

Figure 1. Total PCB Concentration and Young-of-the-Year Production for Striped Bass in the Lower Hudson River

6.2.2 White perch

White perch are sampled in many of the same programs that sample striped bass. The abundance of white perch larvae and juveniles increased rapidly in the late 1970s, but has fluctuated and generally declined since the mid-1980s. A variety of factors may have contributed to the decline; however, the DEIS concluded that competition with young striped bass and predation by older striped bass are the most likely cause (Central Hudson Gas & Electric Corporation et al., 1999). In addition, the re-growth of large beds of water chestnut in the upper estuary following cessation of herbicide treatments in 1976 is believed to have reduced the quality of the habitat for juvenile fish and may also have contributed to the recent decline (Central Hudson Gas & Electric Corporation et al., 1999).

6.2.3 Shortnose sturgeon

Published mark-recapture studies discussed in GE's comments on the BERA show a large increase in the abundance of shortnose sturgeon in the Lower Hudson between the 1970s and the 1990s. These studies indicate that the size of the spawning stock of shortnose sturgeon in the Hudson has increased fourfold, from approximately 14,000 fish to 60,000 fish during that interval. These studies are supported by data on the abundance of yearling shortnose from the utilities' monitoring program. The utilities' data show a substantial increase in abundance of young sturgeon since 1990. In light of these data, NMFS has recommended that the status of the population be changed from "endangered" to "threatened."

6.2.4 Atlantic Tomcod

The Atlantic tomcod is relevant to the Future Risk ERA because studies performed in the in the 1970s found liver tumors in 80% of the adult tomcod examined (Klauda et al., 1981). Exposure to PCBs was suggested as a possible cause; however elevated levels of PAH-sensitive biomarkers in Hudson River tomcod suggest increased exposure to polycyclic aromatic hydrocarbons (PAHs), consistent with previous studies (Wirgin et al., 1994). Thermal stress to tomcod during warmer months and the potential occurrence of a genetically distinct population of tomcod in the Hudson River that is predisposed to neoplasia may also contribute to the prevalence of tumors (El-Zahr, et al., 1993; Schultz et al., 1993; Wirgin et al., 1991). Despite the tumors, population trends in this species have been relatively stable, with abundance increasing somewhat from 1983-1989 and decreasing somewhat from 1989 through 1997. The DEIS concludes that improved sewage treatment in the lower estuary, resulting in reduced food availability and increased competition, may be responsible for the recent decline. Data collected during the 1995-1996 spawning season indicate that the incidence of liver tumors has dropped to less than 2%.

6.2.5 Summary of Risks to Fish Community of the Lower Hudson River

Changes in the fish community as a whole, measured by the number of species present, appear to have been determined by three factors based on analyses performed by experts in fisheries biology (Central Hudson Gas & Electric Corporation et al., 1999):

(1) Improved water quality in the Lower Hudson, which increased the number of marine species entering the lower estuary.

(2) Increased abundance of striped bass, which reduced the abundance of many species throughout the lower estuary.

(3) Increased abundance of water chestnut, which has reduced the availability of habitat for freshwater fish in the upper estuary.

PCB exposures, which have declined steadily over the entire period covered in the DEIS, do not explain any of the observed changes. The observation of increasing, i.e., recovering, populations of fish occurring in previous periods of relative high PCB concentrations suggests that PCBs are unlikely to have a significant impact on population dynamics in the future when PCB levels are expected to decline.

6.3 Birds and Mammals

As noted by GE in comments on the BERA, data demonstrating the health of bird and mammal populations throughout the Hudson Valley are available from a variety of sources. 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.

The Future Risk ERA itself acknowledges that Audubon Society Christmas bird counts and other sources of local information on the bird species present in the Lower Hudson Valley show that:

(1) Tree swallows are present throughout the Lower Hudson Valley.

(2) Waterfowl are extremely abundant.

(3) Belted kingfishers and great blue herons are breeding throughout the Lower Hudson.

(4) Bald eagles are returning.

EPA's statement that the eagles have not successfully reproduced is incorrect. In fact, the Hudson River bald eagle population has become reestablished in recent years. The first bald eagle nesting attempt on the Hudson River in over 100 years occurred in 1992 along the Lower Hudson River, but no fledglings were successfully produced at this nest until 1997 (Nye 1999, pers. comm.). Since then, three bald eagle territories have been active on the Lower Hudson River. Four eaglets were fledged from these territories in 1998, including three from a single nest in Columbia County. Four eaglets were also fledged in 1999, including three from a single nest in Green County.

The Future Risk ERA also acknowledges that raccoons are abundant throughout the Lower Hudson Valley, and that mink and river otter are present. EPA discounts the significance of the occurrence of raccoon populations on the grounds that raccoons likely obtain food from sources other than the Hudson River. In the 1960s, the Hudson River Valley Commission (HRVC, 1966) reported that the raccoon, cottontail rabbit, gray squirrel, muskrat, skunk, and beaver were plentiful along the Hudson River. Numerous localized studies of biota in wetland and riparian areas along the Lower Hudson River reported the presence of mammalian species that are common throughout the eastern U.S., including raccoon, muskrat, beaver, and white-tailed deer (Kiviat, 1986, 1997; Kiviat and Tashiro, 1987; Kiviat and Stapleton, 1987).

7.0 EPA's approach to effects assessment for fish and wildlife is excessively conservative, relies on a small subset of the available data, and ignores or improperly interprets key studies.

All of GE's comments on the TRVs used in the BERA apply equally to the Future Risk ERA, because in almost all cases the same TRVs are used in both documents. The only exception is the study of Bengsston (1980), for which EPA apparently *lowered* the NOAEL and LOAEL in response to comments from NOAA on the BERA. In addition to its previous comments, GE believes it is important to emphasize that the effects assessment component of the Future Risk ERA is based on a mere handful of studies that are treated in an excessively conservative manner. Therefore, not only does EPA make inappropriate use of an overly conservative screening-level approach, its approach is further compromised by a biased treatment of the available literature-derived toxicological data. EG-1.8a

7.1 Benthic Community Structure

EPA states that the assessment endpoint to be used for evaluation of risks to the benthic community is benthic community structure,[2] but the measurement endpoints selected were (1) comparison of modeled water column chemical concentrations to water quality criteria and (2) comparison of modeled sediment chemical concentrations to guideline values. Neither of these endpoints that were actually used is directly representative of benthic community structure. These methods are suitable only for screening assessments. The Future Risk ERA should rely on direct measurement of the abundance, diversity, and other characteristics of invertebrate communities. Data on benthic community structure are available from EPA (1993) (reported as part of the BERA), Exponent (1998a,b), and NYSDEC (1993). EG-1.8b

[2] The text of the Future Risk ERA uses the ambiguous phrase "benthic community structure as a food source"—whether this is intended to mean community structure or biomass is unclear, but in either case, the measurement endpoints used are inappropriate.

Water and sediment quality criteria (or guideline values) are inappropriate measurement endpoints for assessment of benthic community structure. Criteria values are derived from toxicity tests on individuals, and do not represent community-level effects.

EG-1.19

7.2 Fish

The following studies provided *all* of the TRVs for the eight fish species evaluated:

- Bengtsson (1980), effects of exposure to Clophen A50 on the minnow *Phoxinus phoxinus*.
- Walker et al. (1994), effects of dioxin on lake trout eggs and fry.
- Adams et al. (1989, 1990, 1992) study of redbreast sunfish (*Lepomis auritus*) exposed to multiple chemicals in the field.
- Olivieri and Cooper (1997), study of effects of dioxin on the fathead minnow (*Pimephales promelas*).
- Elonen et al. (1998), study of the effects of dioxin on channel catfish (*Ictalurus punctatus*).
- Westin et al. (1993), study of effects of PCBs on larval striped bass (*Morone saxatilis*).

The study by Bengtsson (1980) was the source of laboratory-derived TRVs for 7 of the 8 fish species. The TRVs for 6 of these species were derived by applying 10x uncertainty factors to the NOAEL and LOAEL calculated in the paper. The study by Walker et al. (1994) was the source of TEQ-based TRVs for 6 of the 8 species. No uncertainty factors were applied to results from this study; however, because salmonids appear to be uniquely sensitive to dioxin compared to other tested taxonomic groups, the relevance of the study to Hudson River fish species is questionable. The NOAELs derived from the two field studies used by EPA (Adams et al., Westin et al.) are *unbounded* NOAELs, meaning that no effects on survival, growth, or reproduction attributable to PCBs were actually observed.

The review performed by Monosson (NOAA, 1999) for NOAA, which evaluated all available literature on the toxicity of Aroclor 1254 to fish, concluded that adverse effects could be expected at exposure concentrations of approximately 25 ppm in the livers of adult fish (equivalent to approximately 12.5 ppm in fillets of Hudson River fish) or approximately 5 ppm (whole body) in larvae. The NOAA value for adult fish is nearly an order of magnitude higher than the LOAEL TRVs EPA used for pumpkinseed, brown bullhead, yellow perch, white perch, largemouth bass, striped bass, and shortnose sturgeon. As noted in Section 2 of these comments, EPA ignored the report's conclusion that the TEQ approach should not be applied to Hudson River fish species.

As noted in GE's comments on the BERA, the values developed in the Monosson report are still conservative: a review by Niimi (1996) concluded that even higher exposures may be required before actual reductions in survival or reproduction are observed in typical fish species. Thus, EPA's approach to evaluating the toxicity of PCBs to fish is highly selective and superficial and the effects predicted by EPA's TQs have not been observed in the exposed populations themselves.

EG-1.20

7.3 Birds

For birds, the following laboratory studies on gallinaceous birds (*e.g.*, chickens and pheasants) provided a large fraction of the TRVs used by EPA:

- Scott (1977), effects of PCBs on the chicken.
- Nosek et al. (1992), effects of dioxin on the pheasant.
- Powell et al. (1996), effects of PCB congeners on the chicken.

EPA acknowledges that gallinaceous birds, such as chickens and pheasants, are extremely sensitive to PCBs. The use of TRVs derived from these studies is therefore expected to significantly overstate the actual risks of PCBs to wild birds. Alternative data sources more relevant to avian receptors at the Hudson River which avoid this overprediction are discussed in the following sections.

As GE explained in its comments on the BERA, EPA's use of the lowest available NOAEL when multiple studies were available is inappropriate. Because a NOAEL can be considerably lower than an effects threshold, selection of the *highest* NOAEL for the species of interest or a surrogate will minimize the gap between the NOAEL and the actual threshold for observable effects.

The derivation of TRVs in the Future Risk ERA also follows an outdated "margin-of-safety" method in applying uncertainty factors which introduces unnecessary conservatism into the risk assessment. Rather than using default uncertainty factors of 10, human health risk assessors (Dourson et al., 1996) use a method that considers values from 1 to 10 where appropriate, depending on the availability of data for the chemical in question. Ecological risk assessors seem to be following suit, particularly with regard to interspecies extrapolations (e.g., EPA Region 10, 1997 [EPA, 1997b]; Hoff and Henningsen, 1998). EPA's ERA guidelines (EPA, 1998) note that "uncertainty factors can be misused, especially when used in an overly conservative fashion, as when chains of factors are multiplied together without sufficient justification."

In several instances, EPA considers a 10-week exposure period to be subchronic, and a subchronic-to-chronic uncertainty factor of 10 is applied to the NOAEL. This is the case for the tree swallow, mallard, great blue heron, bald eagle, and belted kingfisher's dietary TEQ-based TRV. However, according to Sample et al. (1996), 10 weeks is considered the transition point from a subchronic to a chronic exposure duration for avian species, rendering such a large uncertainty factor unnecessary.

7.3.1 Tree Swallow

The field studies conducted by the U.S. Fish and Wildlife Service which addressed effects of PCBs at concentrations higher than likely to be found in the Lower Hudson make it irrelevant to predict PCB-related effects on the basis of extrapolations of data from laboratory studies. Ample field data have been collected from areas adjacent to the

Hudson River (Secord and McCarty, 1997; McCarty and Secord, 1999 a,b). These data indicate that the reproductive success of tree swallows is not being affected by PCBs in the Hudson River. EPA's statements regarding these studies are misleading. McCarty and Secord have been unable to illustrate a dose-response relationship between tree swallow reproduction and PCB contamination. The differences in reproductive parameters between the Ithaca and Hudson River tree swallow populations fall within the natural variation observed elsewhere in tree swallow populations. Likewise, the behavioral data referred to by EPA do not correlate with reproductive parameters.

7.3.2 Mallard

Out of the three studies that have examined PCB toxicity in mallards, EPA selected the study with the *lowest* NOAEL for TRV development. As shown above, this approach is erroneous. The NOAEL found by Risebrough and Anderson (1975), based on a dietary Aroclor 1254 dose of 40 ppm, is recommended as the TRV. Risebrough and Anderson (1975) did not measure PCB concentrations in eggs associated with this level of exposure. However, Heath et al. (1972) established a NOAEL for Aroclor 1254 at a slightly lower dose (25 ppm), and measured a corresponding egg concentration of 45 ppm. Additionally, because these two studies used exposure durations of 150 and 511 days (Risebrough and Anderson, 1975; Heath et al., 1972, respectively), should not apply a subchronic-to-chronic uncertainty factor as it did for the Custer and Heinz (1980) study.

7.3.3 Great Blue Heron

The studies selected by EPA for TRV development for the great blue heron were less appropriate than other available studies and were incorrectly interpreted. Speich et al. (1992) examined potential effects of environmental concentrations of PCBs, from both pristine and industrialized areas, on great blue heron reproduction in western Washington State. The authors noted that they were unable to detect any PCB-related effects on egg mortality that would have been predicted on the basis of chicken studies. Therefore, the egg concentration of 16 ppm (wet weight), representing the highest reported mean egg concentration in a reproductively healthy colony, could be considered an unbounded NOAEL. This concentration is 48-fold higher than the TRV (0.33 mg/kg egg) derived by EPA on the basis of effects in chickens.

Field data in Sanderson et al. (1994) are used to derive a TEQ-based TRV in great blue heron eggs. However, the authors reported an improvement in the reproductive success of the colony with the highest measured TEQ concentrations. Though EPA used an egg concentration of 0.5 ug TEQ/kg egg as a LOAEL based on a reduction in body weight, Sanderson et al. (1994) did not find reduced body weights in the birds.

7.3.4 Belted Kingfisher

Species-specific studies are not available for the kingfisher; however, the studies selected by EPA for TRV development were less appropriate than other available studies for species similar to the kingfisher. As indicated above, there are available studies for species with similar feeding habits to those of the kingfisher (e.g., great blue heron) which would provide more representative TRVs than those derived using gallinaceous bird studies.

7.3.5 Bald Eagle

The TRV for total PCB concentrations in bald eagle eggs - 3.0 mg/kg - is based on a field study of population productivity and egg contaminant concentrations for a large number of sites (Wiemeyer et al., 1993). This value is inappropriate for two reasons:

(1) Wiemeyer et al. (1993) report that productivity was not statistically different in eggs in three concentration ranges: <3.0, 3.0 - <5.6, 5.6-<13 (Wiemeyer et al., 1993 Table 10). Productivity was significantly reduced for PCB concentrations >13 mg/kg. Thus, based upon these data, a NOAEL of 13 mg/kg is more appropriate.

(2) Wiemeyer et al. (1993) could not demonstrate impacts of PCBs on productivity because of the strong correlation between PCB and DDE levels. Thus, a LOAEL cannot be determined, and the degree of conservatism in the NOAEL of 13 mg/kg is unknown.

DDE concentrations in fish collected recently near Catskill, New York average approximately 0.27 ppm whole body (NYSDEC database: HUDORG.dbf). Using an egg/fish DDE ratio of 22 (Giesy et al., 1995), an egg level of approximately 6 mg/kg is estimated. This is greater than the NOAEL of 3.6 mg/kg estimated by Wiemeyer et al. (1993) for DDE in bald eagles. This suggests that DDE may be having an impact on bald eagle productivity in the Lower Hudson River.

EPA also ignored or discounted two other field studies on potential effects of PCBs on bald eagles. Elliot et al. (1996) evaluated hatching success and morphological, physiological, and histological parameters in bald eagle eggs collected near pulp mills in British Columbia. Laboratory hatching success did not differ between eggs from pulp mill sites and from reference locations, though Elliot et al. (1996) did find positive associations between PCB exposure and biochemical and morphological responses. The unbounded NOAEL for hatching success based on this data is >400 pg/g TEQ (wet weight) in eggs. Additionally, Donaldson et al. (1999) studied reproductive success of breeding bald eagles along Lake Erie in Canada from 1980 to 1996. The author concluded that the reproductive success of the colony was not impaired, and found an unbounded NOAEL of >26.4 mg/kg total PCBs (wet weight) in eggs based on nest reproductive success. Both of these NOAELs are significantly higher than those selected by EPA.

EG-1.21

7.4 Mammals

As noted in GE's comments on the BERA, the TRVs for little brown bat and raccoon are based on laboratory studies of rats (Murray et al. 1979; Linder et al. 1974). The study by Murray et al. (1979) was also used to derive TEQ-based dietary TRVs for mink and river otter. EPA calculated TRVs by applying 10x uncertainty factor to the LOAELs and NOAELs from these studies.

The very limited available data concerning effects of PCBs on mammalian species other than rodents and mink indicate that EPA should be very cautious about basing remedial decisions on TQs calculated for these species. Data sources and approaches that EPA could use to more appropriately assess potential effects of PCBs on mink and river otter are described below.

7.4.1 Mink

EPA used a field study by Tillett et al. (1996) to derive both a NOAEL and a LOAEL for TEQs in the diet of mink at Lake Michigan. However, the method used to administer PCBs to the test animals did not exclude other environmental toxicants known to be present in Great Lakes fish (Giesy, et al. 1994), the study is inappropriate for use in deriving a LOAEL. On page 34 of the Future Risk ERA, EPA states that "because of the potential contribution of other contaminants (e.g., metals, pesticides, etc.) to observed effects in field studies, [this] ERA and ERA Addendum use field studies to establish NOAEL TRVs, but not LOAEL TRVs." According to EPA's own selection criteria, this study should not have been used to derive a LOAEL TRV.

Mink laboratory studies that investigate the reproductive effects of Aroclor 1254 resulting from chronic dietary exposure are typically considered relevant and scientifically sound for the development of protective mink NOAEL and LOAEL values for PCBs. EPA's choice of the study by Aulerich and Ringer (1977) is consistent with Sample et al. (1996); however, it should be used similarly to derive a TRV. While EPA applies a subchronic-to-chronic uncertainty factor of 10 to the NOAEL and LOAEL, Sample et al. (1996) states that because the treatment period extended before and throughout the reproductive stage, the study should be considered chronic in duration. As a result, the NOAEL and LOAEL should not be conservatively adjusted to account for the exposure duration.

An alternative approach to TRV development based on dietary levels of PCBs is the determination of critical body residues of PCBs developed from dose-response relationships. A study by Leonards et al. (1995) evaluated dose-response relationships for PCB body burdens and mink reproductive parameters from nine feeding studies. Leonards et al. (1995) proposed critical body residues of 1.2 ug/g total PCBs (wet weight) and 160 pg/g TEQ (wet weight) based on effects on mink litter size. Because PCB whole-body concentrations in mink were more closely correlated with reproductive effects than PCB concentrations in food, these critical whole-body residue levels should serve as PCB TRVs. EPA should use the results of ongoing residue studies for furbearers by NYSDEC in conjunction with these TRVs.

7.4.2 River Otter

EPA selected TRVs for the river otter using NOAEL and LOAEL TRVs for mink, based on the assumption that because the two species are in the same phylogenetic family, they must be similarly sensitive to PCBs. Recent data examining reproductive health in mustelids found that river otters were not as susceptible to PCB-induced effects as mink (Harding et al., 1999). The Agency should take account of this information.

7.5 General Limitations of TRVs and the TQ Approach

EG-1.22

As previously indicated, the TQ approach, which incorporates the TRVs, is a highly conservative screening-level approach that is inappropriate for use in an ecological risk assessment of the scale of the Hudson River assessments. Since this approach focuses on potential risks to individuals, it is not sufficient to demonstrate a significant risk at the population, community, or ecosystem level. EPA's selective treatment of the available scientific literature and overly conservative application of uncertainty factors in deriving TRVs further negates any use this approach has on decisions regarding remedial actions.

8.0 Conclusions

In its *Baseline Ecological Risk Assessment for Future Risks in the Lower Hudson River,* EPA relied exclusively on models and ignored site-specific data demonstrating that PCBs have not adversely affected ecological resources of the Lower Hudson River in the past, and will not do so in the future. The models used by EPA to predict future concentrations of PCBs in water, sediment, and fish tissue contain many deficiencies and have been inadequately reviewed to date. The Toxicity Reference Values used by EPA to estimate risks to fish and wildlife are conservative, screening-level values selectively derived from the scientific literature. EPA's conclusions, which are that important fish and wildlife species in the lower Hudson are presently at risk and will in the future continue to be at risk, are unambiguously contradicted by a wealth of data on the past and present status of those species. Data that were available to EPA show that: EG-1.23

- The reproductive success of the Hudson River striped bass population, as measured by the number of juvenile fish produced each year, was as high in the 1970s, when PCB concentrations in adult female striped bass were at their highest measured levels, as in recent years, when concentrations are much lower. The abundance of adult striped bass has increased dramatically over that same period, as has the abundance of shortnose sturgeon.

- The Lower Hudson River Valley supports healthy, reproducing populations of the wildlife populations addressed by EPA. These include piscivorous birds such as the kingfisher, for which EPA predicted that reproductive effects would occur as a result of PCB exposures.

- Bald eagles are now successfully reproducing in the Lower Hudson River Valley, for the first time in 100 years.

EPA's failure to properly consider these facts in the Future Risk ERA is inconsistent with best scientific practice in ecological risk assessment and with the agency's own guidelines.

This assessment *does not* provide a sound and reliable description of the effects of current and risks of future PCB exposures on biota in the Hudson River Valley. It does not provide a scientifically valid foundation for either estimating the responses of the biota of the Lower Hudson River to alternative remedies that would reduce inputs of PCBs from the upper Hudson or for comparing the ecological benefits gained through remedial actions to the ecological costs of implementing remedial actions.

The report should not be used by EPA in making decisions regarding remedial actions in the upper Hudson River.

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Table 1. Comparison of Lower Hudson River Future Risk 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 Measurement endpoints: Water and sediment-quality criteria, Chronic TRVs (reproduction endpoint) for fish, birds, and mammals	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 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	
Modeled concentrations of PCBs (tri+) and TEQs in fish Modeled oral doses (tr+ 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 sub-area), using (1) conservative exposure assumptions, and (2) Monte Carlo analysis of all exposure parameters

<table>
<tr><th colspan="2">Effects Assessment</th></tr>
<tr><th>Hudson River ERA</th><th>Clinch River ERA</th></tr>
<tr><td>TRVs for PCB and TEQ concentrations in fish tissue

Field-derived (tree swallow and bald eagle) or literature-derived (other species) TRVs for fish, birds, mammals</td><td>TRV for PCB concentrations in fish tissue (whole body, adult)

Literature-derived TRVs for birds and mammals

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</td></tr>
</table>

<table>
<tr><th colspan="2">Risk Characterization</th></tr>
<tr><th>Hudson River ERA</th><th>Clinch River ERA</th></tr>
<tr><td>All assessment endpoints: Comparison of water and sediment concentrations to water and sediment-quality criteria

Fish: Comparison of tri+ and TEQ concentrations in fish tissue to literature-derived TRVs

Overview of population trends for selected species

Birds: Comparison of modeled oral doses and egg concentrations (tri+ and TEQs) to field-derived (tree swallow and bald eagle) or literature-derived (other species) TRVs .

Qualitative overview of occurrence data for various species

Mammals: Comparison of modeled doses (tri+ and TEQs) to literature-derived TRVs</td><td>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</td></tr>
</table>

Table 2. Computation of PCB Levels in Fish - Future Risk ERA

Species	Location (River Mile)	Method of estimation	Sediment exposure	Water column exposure
Largemouth bass, White perch, brown bullhead, pumpkinseed, yellow perch, spottail shiner	60-152	FISHRAND	✓	✓
White perch	113,152 Region 1 (60-152)	FISHRAND FARLEY	?	? ✓
White perch	Region 2 (12-60)	FARLEY		✓
Striped bass	113	FARLEY		✓
Striped bass	152	Largemouth bass from FISHRAND multiplied by a data-based STB/LMB ratio	✓	✓