TABLE OF CONTENTS FOR MASTER CD
ECOLOGICAL RISK ASSESSMENT
FOR GENERAL ELECTRIC (GE)/HOUSATONIC RIVER SITE, REST OF RIVER

1. ERA MAIN DOCUMENT AND APPENDICES

U.S. Army
Corps of Engineers

New England District
Concord, Massachusetts

U.S. Environmental
Protection Agency

New England Region
Boston, Massachusetts

ECOLOGICAL RISK ASSESSMENT FOR GENERAL ELECTRIC (GE)/HOUSATONIC RIVER SITE, REST OF RIVER

Volumes 1 and 2
Sections 1-12

DCN: GE-100504-ACJS

November 12, 2004

Environmental Remediation Contract
GE/Housatonic River Project
Pittsfield, Massachusetts

Contract No. DACW33-00-D-0006

Task Order 0003

04P-1447-vo1&2

ECOLOGICAL RISK ASSESSMENT FOR
GENERAL ELECTRIC (GE)/HOUSATONIC RIVER SITE
REST OF RIVER

VOLUME 1
SECTIONS 1-5

ENVIRONMENTAL REMEDIATION CONTRACT
GENERAL ELECTRIC (GE)/HOUSATONIC RIVER PROJECT
PITTSFIELD, MASSACHUSETTS

Contract No. DACW33-00-D-0006
Task Order No. 0003

DCN: GE-100504-ACJS

Prepared for

U.S. ARMY CORPS OF ENGINEERS
New England District
Concord, Massachusetts

and

U.S. ENVIRONMENTAL PROTECTION AGENCY
New England Region
Boston, Massachusetts

Prepared by

WESTON SOLUTIONS, INC.
West Chester, Pennsylvania

November 12, 2004

Work Order No. 20123.001.096.0735

ECOLOGICAL RISK ASSESSMENT FOR GENERAL ELECTRIC (GE)/HOUSATONIC RIVER SITE REST OF RIVER

VOLUME 2
SECTIONS 6-12

ENVIRONMENTAL REMEDIATION CONTRACT
GENERAL ELECTRIC (GE)/HOUSATONIC RIVER PROJECT
PITTSFIELD, MASSACHUSETTS

Contract No. DACW33-00-D-0006
Task Order No. 0003

DCN: GE-100504-ACJS

Prepared for

U.S. ARMY CORPS OF ENGINEERS
New England District
Concord, Massachusetts

and

U.S. ENVIRONMENTAL PROTECTION AGENCY
New England Region
Boston, Massachusetts

Prepared by

WESTON SOLUTIONS, INC.
West Chester, Pennsylvania

November 12, 2004

Work Order No. 20123.001.096.0735

AUTHORS/CONTRIBUTORS

Tod Delong
Avatar Environmental, LLC
West Chester, Pennsylvania

Scott Ferson, Troy Tucker
Applied Biomathematics, Inc.
Setauket, New York

Dwayne Moore, Roger Breton, Scott Teed, Ryan Thompson
CANTOX Environmental, Inc.
Ottawa, Ontario

Gary Lawrence
EVS Environment Consultants
Vancouver, British Columbia

Richard McGrath, Richard DiNitto
Sleeman Hanley & DiNitto, Inc.
Boston, Massachusetts

Francesca Langford
District of Squamish
Squamish, British Columbia

Susan Svirsky
U.S. Environmental Protection Agency
Boston, Massachusetts

Scott Campbell
Weston Solutions, Inc.
West Chester, Pennsylvania; Pittsfield, Massachusetts

John Lortie, Bob Roy, Michael Thompson
Woodlot Alternatives, Inc.
Topsham, Maine

TABLE OF CONTENTS

Section **Page**

VOLUME 1

TABLE OF CONTENTS
(Continued)

Section **Page**

TABLE OF CONTENTS
(Continued)

Section **Page**

TABLE OF CONTENTS
(Continued)

Section **Page**

TABLE OF CONTENTS
(Continued)

Section **Page**

TABLE OF CONTENTS
(Continued)

Section **Page**

VOLUME 2

TABLE OF CONTENTS
(Continued)

Section **Page**

TABLE OF CONTENTS
(Continued)

Section **Page**

TABLE OF CONTENTS
(Continued)

Section **Page**

TABLE OF CONTENTS
(Continued)

Section **Page**

TABLE OF CONTENTS
(Continued)

Section **Page**

TABLE OF CONTENTS
(Continued)

Section **Page**

LIST OF TABLES

Title **Page**

LIST OF TABLES
(Continued)

Title **Page**

LIST OF TABLES
(Continued)

Title **Page**

LIST OF TABLES
(Continued)

Title **Page**

LIST OF TABLES
(Continued)

Title **Page**

LIST OF TABLES
(Continued)

Title **Page**

LIST OF TABLES
(Continued)

Title **Page**

LIST OF TABLES
(Continued)

Title **Page**

LIST OF FIGURES

Title **Page**

LIST OF FIGURES
(Continued)

Title **Page**

LIST OF FIGURES
(Continued)

Title **Page**

LIST OF FIGURES
(Continued)

Title **Page**

LIST OF FIGURES
(Continued)

Title **Page**

LIST OF FIGURES (Continued)

Title **Page**

LIST OF FIGURES
(Continued)

Title **Page**

LIST OF FIGURES
(Continued)

Title **Page**

LIST OF FIGURES
(Continued)

Title **Page**

LIST OF FIGURES
(Continued)

Title **Page**

LIST OF FIGURES
(Continued)

Title **Page**

LIST OF FIGURES
(Continued)

Title **Page**

LIST OF FIGURES
(Continued)

Title **Page**

LIST OF FIGURES
(Continued)

Title **Page**

LIST OF FIGURES
(Continued)

Title **Page**

LIST OF FIGURES
(Continued)

Title **Page**

LIST OF FIGURES
(Continued)

Title **Page**

LIST OF FIGURES
(Continued)

Title **Page**

LIST OF FIGURES
(Continued)

Title **Page**

LIST OF FIGURES
(Continued)

Title **Page**

LIST OF FIGURES
(Continued)

Title **Page**

LIST OF FIGURES
(Continued)

Title **Page**

LIST OF FIGURES
(Continued)

Title **Page**

TABLE OF CONTENTS FOR APPENDICES

Title **Page**

APPENDICES

VOLUME 3

APPENDIX A ECOLOGICAL CHARACTERIZATION (Presented on CD)

VOLUME 4

APPENDIX B PRE-ERA

TABLE OF CONTENTS FOR APPENDICES (Continued)

Title **Page**

ATTACHMENTS

TABLE OF CONTENTS FOR APPENDICES
(Continued)

Title **Page**

APPENDIX C SUPPORTING TECHNICAL INFORMATION

TABLE OF CONTENTS FOR APPENDICES (Continued)

Title **Page**

TABLE OF CONTENTS FOR APPENDICES
(Continued)

Title **Page**

TABLE OF CONTENTS FOR APPENDICES
(Continued)

Title **Page**

APPENDIX D ASSESSMENT ENDPOINT—COMMUNITY STRUCTURE, SURVIVAL, GROWTH, AND REPRODUCTION OF BENTHIC INVERTEBRATES

TABLE OF CONTENTS FOR APPENDICES (Continued)

Title **Page**

TABLE OF CONTENTS FOR APPENDICES
(Continued)

Title **Page**

LIST OF ATTACHMENTS

TABLE OF CONTENTS FOR APPENDICES (Continued)

Title **Page**

VOLUME 5

APPENDIX E ASSESSMENT ENDPOINT—COMMUNITY CONDITION, SURVIVAL, REPRODUCTION, DEVELOPMENT, AND MATURATION OF AMPHIBIANS

TABLE OF CONTENTS FOR APPENDICES
(Continued)

Title **Page**

TABLE OF CONTENTS FOR APPENDICES (Continued)

Title **Page**

TABLE OF CONTENTS FOR APPENDICES
(Continued)

Title **Page**

TABLE OF CONTENTS FOR APPENDICES
(Continued)

Title **Page**

APPENDIX F ASSESSMENT ENDPOINT—SURVIVAL, GROWTH, AND REPRODUCTION OF FISH

TABLE OF CONTENTS FOR APPENDICES (Continued)

Title **Page**

TABLE OF CONTENTS FOR APPENDICES
(Continued)

Title **Page**

TABLES (Pages 1 to 49)

FIGURES (Pages 1 to 44)

ATTACHMENTS

TABLE OF CONTENTS FOR APPENDICES (Continued)

Title **Page**

APPENDIX G ASSESSMENT ENDPOINT— SURVIVAL, GROWTH, AND REPRODUCTION OF INSECTIVOROUS BIRDS

TABLE OF CONTENTS FOR APPENDICES
(Continued)

Title **Page**

TABLES (Pages 1 to 84)

FIGURES (Pages 1 to 25)

ATTACHMENTS

TABLE OF CONTENTS FOR APPENDICES
(Continued)

Title **Page**

VOLUME 6

APPENDIX H ASSESSMENT ENDPOINT— SURVIVAL, GROWTH, AND REPRODUCTION OF PISCIVOROUS BIRDS

H. ASSESSMENT ENDPOINT—SURVIVAL, GROWTH, AND REPRODUCTION OF PISCIVOROUS BIRDS

TABLE OF CONTENTS FOR APPENDICES
(Continued)

Title **Page**

TABLE OF CONTENTS FOR APPENDICES
(Continued)

Title **Page**

APPENDIX I ASSESSMENT ENDPOINT— SURVIVAL, GROWTH, AND REPRODUCTION OF PISCIVOROUS MAMMALS

TABLE OF CONTENTS FOR APPENDICES
(Continued)

Title **Page**

TABLE OF CONTENTS FOR APPENDICES (Continued)

Title **Page**

APPENDIX J ASSESSMENT ENDPOINT—SURVIVAL, GROWTH, AND REPRODUCTION OF OMNIVOROUS AND CARNIVOROUS MAMMALS

TABLE OF CONTENTS FOR APPENDICES
(Continued)

Title **Page**

TABLE OF CONTENTS FOR APPENDICES (Continued)

Title **Page**

APPENDIX K ASSESSMENT ENDPOINT—SURVIVAL, GROWTH, AND REPRODUCTION OF THREATENED AND ENDANGERED SPECIES

TABLE OF CONTENTS FOR APPENDICES
(Continued)

Title **Page**

TABLE OF CONTENTS FOR APPENDICES (Continued)

Title **Page**

TABLE OF CONTENTS FOR APPENDICES
(Continued)

Title **Page**

LIST OF ACRONYMS

ACO	Administrative Consent Order
Ah	aryl hydrocarbon
AHH	aryl hydrocarbon hydroxylase
BW	body weight
C	chemistry
C/C	coarse-grained contaminated
C/R	Coarser Reference
CAE	chemical absorption efficiency
CAWQC	chronic ambient water quality criterion
CBS	Carolina Biological Supply
CDD:CDF	chlorinated dibenzo-p-dioxin and dibenzofuran
CERC	Columbia Environmental Research Center
Cl/BP	chlorines per biphenyl
COC	contaminant of concern
COPC	contaminant of potential concern
DL	detection limit
DQO	data quality objective
EPA	U.S. Environmental Protection Agency
EPC	exposure point concentration
ERA	ecological risk assessment
EROD	ethoxyresorufin-O-deethylase
F/C	Finer Contaminated
F/R	Finer Reference
FIR	food intake rate
FMR	free metabolic rate
FOMC	First Order Monte Carlo
FS	Field Study
FT	Foraging Time
GE	General Electric Company
GLiM	Generalized Linear Model
HQ	hazard quotient
IDW	inverse distance weighted
LCL	lower confidence limit
LMB	largemouth bass
MATC	Maximum Acceptable Threshold Concentration

LIST OF ACRONYMS
(Continued)

MCP	Massachusetts Contingency Plan
MDEP	Massachusetts Department of Environmental Protection
MDPH	Massachusetts Department of Public Health
MDS	multidimensional scaling
MFD	Modeling Framework Design
MFO	mixed function oxidase
MHBI	Modified Hilsenhoff Biotic Index
MNHESP	Massachusetts Natural Heritage and Endangered Species Program
MSU	Michigan State University
ND	non-detect
ng/L	nanogram per liter
NPL	National Priorities List
OC	organic carbon
PAH	polycyclic aromatic hydrocarbon
PCA	principal components analysis
PCB	polychlorinated biphenyl
PCDD	polychlorinated dibenzo-p-dioxin
PCDF	polychlorinated dibenzo-furan
PCH	planar chlorinated hydrocarbons
pg/L	picogram per liter
PSA	Primary Study Area
QA	quality assurance
QC	quality control
RAGS	Risk Assessment Guidance for Superfund
RCRA	Resource Conservation and Recovery Act
RFI	RCRA Facility Investigation
RPP	relative performance proportion
SCOX	Side Channels and Oxbows
SIWP	Supplemental Investigation Work Plan
SMDP	scientific/management decision point
SOP	standard operating procedure
SPE	solid phase extraction
SQG	sediment quality guideline
SQL	sample quantitation limit
SQT	Sediment Quality Triad

LIST OF ACRONYMS
(Continued)

SQV	sediment quality value
SSD	species sensitivity distribution
SVOC	semivolatile organic compound
TCDD	2,3,7,8-tetrachlorodibenzo-p-dioxin
TDI	Total Daily Intake
TEC	threshold effect concentration
TEF	toxic equivalency factor
TEQ	toxic equivalence
TIE	Toxicity Identification Evaluation
TOC	total organic carbon
tPCB	total PCB
TSK	Trimmed Spearman Karber
UCL	upper confidence limit
USACE	U.S. Army Corps of Engineers
VP	vernal pool
WHO	World Health Organization’s
WOE	weight-of-evidence
WSU	Wright State University
ww	wet weight
WWTP	Wastewater Treatment Plant

ES. ECOLOGICAL RISK ASSESSMENT EXECUTIVE SUMMARY

Highlights of the ERA

- Total PCBs (tPCBs) and other contaminants of concern (COCs) in the Primary Study Area (PSA) of the Housatonic River pose intermediate or high risks to some assessment endpoints, and low risk to others.
- Risk is high for amphibians and piscivorous mammals. Confidence in this conclusion is high because (1) multiple lines of evidence with concordant results were available; (2) models used to estimate risk were not conservative; and (3) consideration of uncertainty indicates a high probability of effects.
- Risk is high for some insectivorous birds (wood duck).
- Risk is high for selected threatened and endangered birds (bald eagle and American bittern) and intermediate for a T&E mammal species (small-footed myotis). There is uncertainty regarding these conclusions because corroborating lines of evidence were generally not available.
- Risk is intermediate to high for benthic invertebrates, but risk varies across reaches and at smaller spatial scales. Confidence in this conclusion is high because multiple lines of evidence with concordant results were available.
- Risk is intermediate to high for piscivorous birds (osprey and belted kingfisher).
- Risk is intermediate for omnivorous and carnivorous mammals (red fox and short-tailed shrew).
- Risk is low to intermediate for warmwater fish in the PSA, and confidence in this conclusion is moderate. Two of the three major lines of evidence (site-specific toxicity, fish tissue chemistry relative to effects thresholds) suggest intermediate risk to fish. However, the field surveys suggest that PCBs and/or other COCs are not causing obvious effects to fish populations.
- Risk is low for some insectivorous birds (e.g., tree swallow and American robin), but confidence in this conclusion is not high because of conflicting results between the available lines of evidence.
- Other species not included in the quantitative risk assessments may also be at risk in the PSA.
- Assessment of risks to the most susceptible endpoints downstream of the PSA indicates that benthic invertebrates, amphibians, coldwater fish, mink, river otter, and bald eagles may be at risk.

ES.1 OVERVIEW

The purpose of this ecological risk assessment (ERA) is to characterize and quantify the current and potential risks to biota exposed to contaminants of potential concern (COPCs) in the

Housatonic River below the confluence of the East and West Branches (known as the "Rest of River"), focusing on polychlorinated biphenyls (PCBs) and other hazardous substances originating from the General Electric Company (GE) facility in Pittsfield, MA.

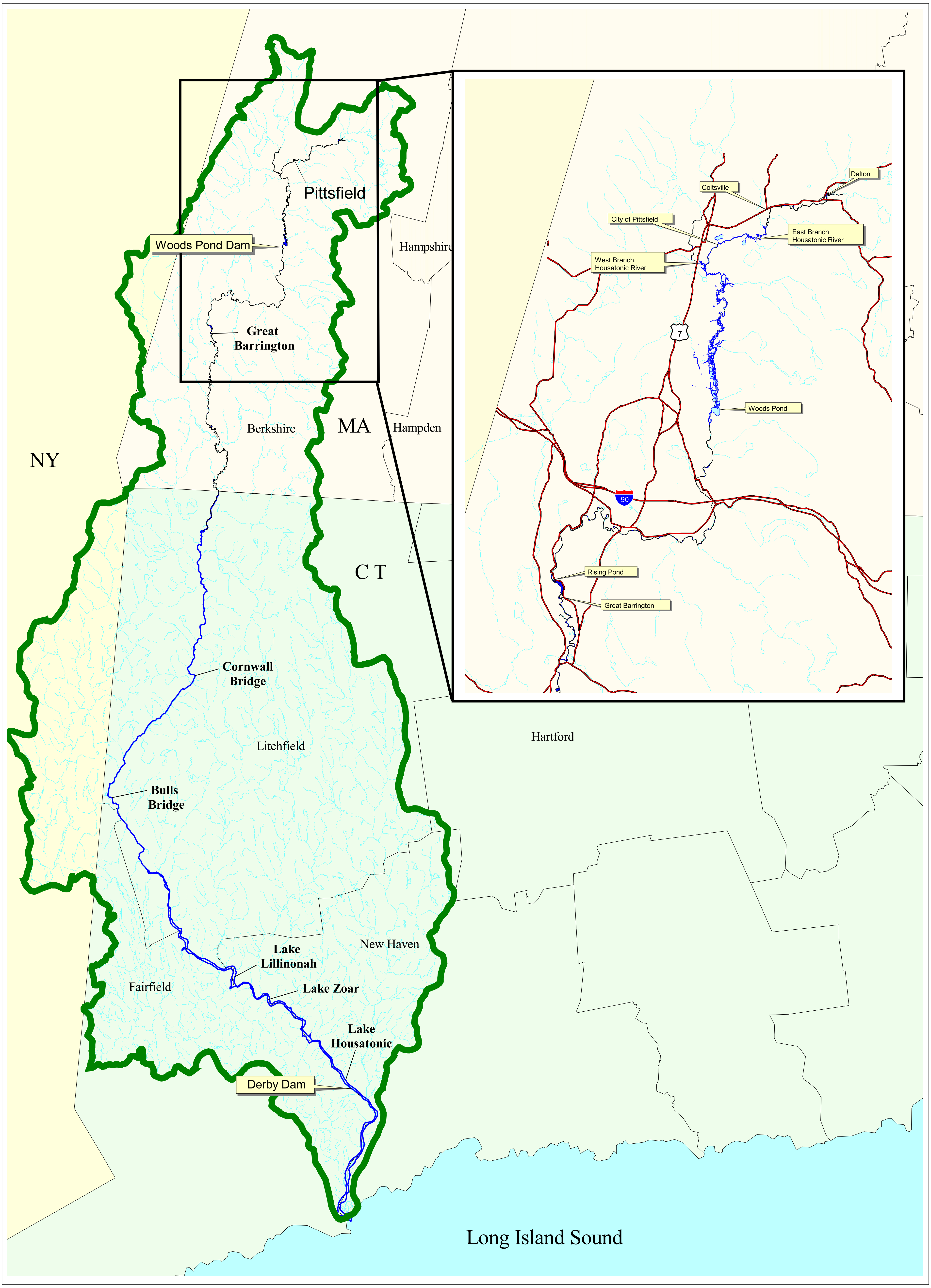
This information is synthesized, through a weight-of-evidence (WOE) approach, into a discussion of the nature and magnitude of the risks for the assessment endpoints, and the uncertainties associated with the characterization of these risks. Multiple lines of evidence for each assessment endpoint are evaluated, including where applicable or available:

- Field surveys and studies.
- Site-specific toxicity tests.
- Comparison of effects thresholds to site-specific COC concentrations or doses (measured or modeled).

ES.2 SITE DESCRIPTION AND HISTORY

The Housatonic River flows from east of Pittsfield, MA, to Long Island Sound and drains an area of approximately 1,950 square miles in Massachusetts, New York, and Connecticut (Figure ES-1). The river is located in a predominantly rural area of western Massachusetts and Connecticut, where farming was the main occupation from colonial settlement through the late 1800s. The entire site, known as the General Electric/Housatonic River Site, consists of the 254-acre (103-hectare [ha]) GE manufacturing facility; the Housatonic River and associated riverbanks and floodplain from Pittsfield, MA, to Long Island Sound; as well as other properties or areas that have become contaminated as a result of GE facility operations.

Widespread contamination of the river downstream of the GE facility has resulted from the transport of PCB-contaminated river sediment and floodplain soil by river flow, sediment transport, and overbank flooding. Numerous studies conducted since 1988 have documented PCB contamination of soil within the floodplain of the Housatonic River downstream of the GE facility. PCBs have been detected in river sediment in Massachusetts as far downstream as the border with Connecticut (BBL 1995), and in Connecticut as far as the Derby Dam and beyond into Long Island Sound (other sources have been identified downstream of this dam). The PCBs detected in Housatonic River floodplain soil and sediment consist of predominantly Aroclor 1260, with a minor contribution of Aroclor 1254.



LEGEND:

Housatonic River

Housatonic River Basin

Primary Road

N W E S

4 0 4 8 Miles

5 0 5 10 15 Kilometers

Ecological Risk Assessment
GE/Housatonic River Site
Rest of River

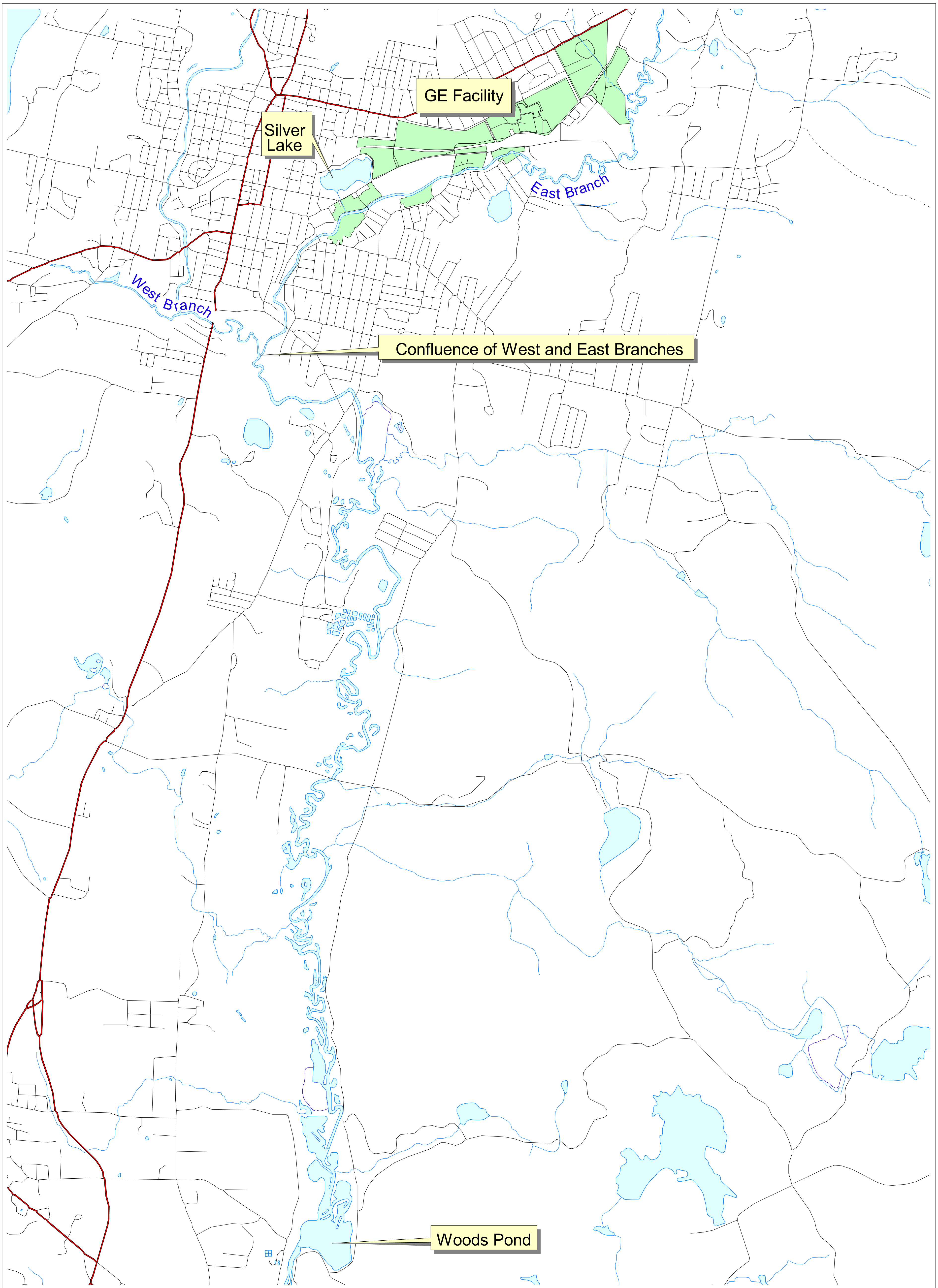
FIGURE ES-1
HOUSATONIC RIVER

The GE facility in Pittsfield is the only known source of PCBs found in the Housatonic River sediment and floodplain soil in Massachusetts. GE began operations at its present location in 1903. Three manufacturing divisions have operated at the GE facility (Transformer, Ordnance, and Plastics). Although GE performed many functions at the Pittsfield facility throughout the years, the activities of the Transformer Division, including the construction and repair of electrical transformers using dielectric fluids, some of which contained PCBs (primarily Aroclor 1260 and, to a lesser extent, 1254), were one likely significant source of PCB contamination.

Because of its size and complexity, the GE/Housatonic River Site has been divided into several areas for investigation and cleanup. The Rest of River is the portion of the river from the confluence of the East and West Branches of the Housatonic River (the confluence) to the Massachusetts border with Connecticut, a distance of approximately 54 miles (87 km), and beyond into Connecticut to Long Island Sound. In addition to the river itself, the Rest of River includes the associated riverbank and floodplain. The lateral extent of the area under investigation includes the floodplain extending to the 1-ppm tPCB isopleth, which is approximately equivalent to the 10-year floodplain. The Rest of River portion of the Housatonic River flows through one of the most biologically diverse regions of Massachusetts and Connecticut. Dams play an integral role in the downstream migration of PCBs and other COPCs from the GE facility.

The ERA focuses on the portion of the river from the confluence, 2 miles (3 km) below the GE facility, to Woods Pond Dam, a distance of 10.7 river miles (17.3 km). This area is referred to as the PSA (Figure ES-2), and is where much of the PCB contamination was found in previous studies. The ERA also includes an evaluation of the river and floodplain downstream of the PSA to the Derby Dam in Connecticut, approximately 14 miles upstream from Long Island Sound.

The first 10.1 miles (16.3 km) from the confluence to the headwaters of Woods Pond is referred to as Reach 5. Next to the initial 0.5-mile (0.8-km) reach bordering the GE facility, Reach 5 has the highest concentrations and highest frequency of detections of PCBs in sediment. Reach 5 is subdivided further into four segments: Reach 5A, from the confluence to just above the Pittsfield Wastewater Treatment Plant (WWTP); Reach 5B, from the WWTP to Roaring Brook; Reach 5C,



LEGEND:

Roads

Housatonic River

GE Facility

N W E S

0.1 0 0.1 0.2 Miles

0.2 0 0.2 Kilometers

Ecological Risk Assessment
GE/Housatonic River Site
Rest of River

FIGURE ES-2
PRIMARY STUDY AREA (PSA)

from Roaring Brook to the headwaters of Woods Pond; and Reach 5D, the backwaters above Woods Pond. Reach 6, which begins approximately 10 miles (16 km) downstream of the confluence at Woods Pond, is 0.57 mile (0.9 km) in length. The pond has an area of 54 acres (22 ha). This reach contains the first impoundment downstream from the GE facility and is a depositional environment. In the PSA, the river channel ranges from 40 to 125 ft (12 to 38 m) in width, is bordered by extensive floodplain (up to 3,600 ft [1,100 m] wide), and has a meandering pattern with numerous oxbows and backwaters.

The remaining reaches downstream of the PSA are designated as follows:

- ***Reach 7* –** From downstream of Woods Pond Dam to the upstream extent of Rising Pond (18.5 miles).
- ***Reach 8* –** Rising Pond (45 acres).
- ***Reach 9*–** From downstream of Rising Pond Dam to the Massachusetts/Connecticut border (23.9 miles).
- ***Reach 10* –** From below the Massachusetts/Connecticut border to Great Falls Dam (7.4 miles).
- ***Reach 11* –** From downstream of Great Falls Dam to Cornwall Bridge (11.5 miles).
- ***Reach 12* –** From downstream of Cornwall Bridge to Bulls Bridge Dams (13.1 miles).
- ***Reach 13* –** From downstream of Bulls Bridge Dams to Bleachery Dam (New Milford, CT) (10.9 miles).
- ***Reach 14* –** From downstream of Bleachery Dam to Shepaug Dam (Lake Lillinonah) (12.5 miles).
- ***Reach 15* –** From downstream of Shepaug Dam to Stevenson Dam (Lake Zoar) (10.2 miles).
- ***Reach 16* –** From downstream of Stevenson Dam to Derby Dam (Lake Housatonic) (6.0 miles).
- ***Reach 17* –** From downstream of the Derby Dam to Long Island Sound (tidal, and not part of GE/Housatonic River site due to other sources of PCBs) (13.7 miles).

The land uses of the floodplain properties in Massachusetts include residential, commercial/industrial, agricultural, recreational (such as canoeing, fishing, and hunting), wildlife management, and parks and a golf course. The Housatonic River floodplain is an attractive area for recreation, including fishing and waterfowl hunting.

The State of Connecticut posted a fish consumption advisory for most of the Connecticut section of the river in 1977 as a result of the PCB contamination in the river sediment and fish tissue. In 1982, the Massachusetts Department of Public Health (MDPH) issued a consumption advisory for fish, frogs, and turtles for the Housatonic River. In addition, in 1999, MDPH issued a waterfowl consumption advisory from Pittsfield to Great Barrington due to PCB concentrations in wood ducks and mallards collected from the river by the U.S. Environmental Protection Agency (EPA).

ES.3 REGULATORY BACKGROUND

The GE/Housatonic River site has been subject to regulatory investigations dating back to the early 1980s. In 1991, EPA issued a RCRA Corrective Action Permit to the General Electric Company (GE) Pittsfield facility. Following appeals by GE and others and subsequent modification, the permit became effective in 1994. The permit included the 254-acre facility, some filled former oxbows, Silver Lake, the Housatonic River and its floodplains and adjacent wetlands, and all sediment contaminated by PCBs migrating from the GE facility.

EPA proposed the site to the Superfund National Priorities List (NPL) in September 1997. Several federal and state government agencies and GE entered into negotiations late in 1997 with the goal of reaching a comprehensive settlement, which included remediation, redevelopment, and restoration components.

In September 1998, representatives of the federal and state government agencies, GE, the City of Pittsfield, and the Pittsfield Economic Development Authority reached an agreement in principle. This agreement was translated into a Consent Decree that was entered by the court on 27 October 2000. The agreement provides for, in general, the cleanup of the GE plant facility and surrounding areas that have become contaminated as a result of facility operations, including the Housatonic River.

The GE/Housatonic River site is made up of several separate response actions (as described in the Consent Decree), including three actions in the river:

- Upper ½-Mile Reach Housatonic River Removal Action (½-Mile Reach)
- 1 ½-Mile Housatonic River Removal Action (1 ½-Mile Reach)
- Rest of River

The primary COPCs are PCBs; other COPCs include volatile organics, dioxins/furans, polycyclic aromatic hydrocarbons (PAHs), semivolatiles, and metals. Certain PCB congeners (known as coplanar or dioxin-like congeners) exhibit a mechanism of toxicity similar to that of the most toxic dioxin congener (2,3,7,8-TCDD). The combined toxicity of these coplanar congeners can be expressed and evaluated as the equivalent toxicity of 2,3,7,8-TCDD using the concept of toxic equivalence (TEQ).

EPA completed an investigation of the Rest of River below the 1½-Mile Reach into Connecticut, which focused on collecting information for and preparing the human health and ecological risk assessments, and modeling PCB fate and transport in the river. The ecological risk assessment, together with the human health risk assessment (HHRA) and the model of PCB fate, transport, and bioaccumulation, will inform EPA's decision on what additional remedial actions, if any, may be required in the future.

ES.4 OVERVIEW OF TECHNICAL APPROACH

This ERA follows the eight-step technical approach and guidelines detailed in EPA's Ecological Risk Assessment Guidance for Superfund: Process for Designing and Conducting Ecological Risk Assessments. The first two steps of the ERA process (Screening-Level Problem Formulation and Screening-Level Exposure Estimate and Risk Calculation) were first addressed in the *Upper Reach-Housatonic River Ecological Risk Assessment* and subsequently refined in Appendix B of this document. Steps 3, 4, and 5 (Baseline Problem Formulation, Study Design and DQO Process, and Verification of Field Data Analysis) are iterative components of the eight-step ERA process. These three steps were initially presented in the *Supplemental Investigation Work Plan for the Lower Housatonic River* and were modified as necessary during the data collection phase of the project. Steps 6 and 7 (Site Investigation and Data Analysis and Risk Characterization) are presented in detail in this Ecological Risk Assessment report. Step 8

(Risk Management) will be addressed upon completion of the ERA and HHRA. A roadmap for the ERA is provided in Figure ES-3. A brief overview of each of the eight steps, particularly as they relate to this document, is presented below.

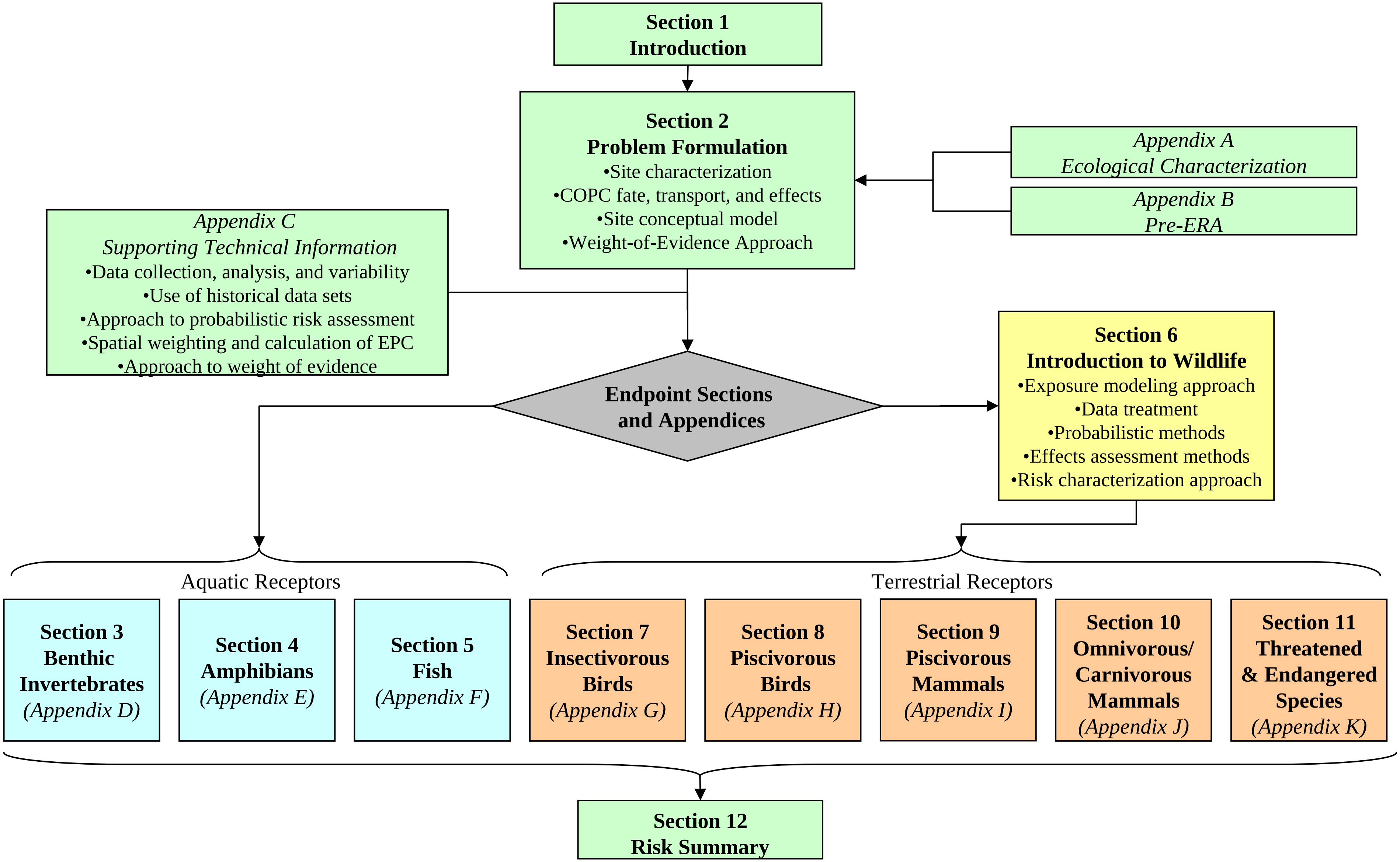
Problem formulation is an important component of the ERA process that establishes the goals, objectives, and scope for the ERA. The problem formulation portion of the ERA is discussed in Section 2 and was outlined in the *Supplemental Investigation Work Plan for the Lower Housatonic River*.

An extensive physical and ecological characterization of the Housatonic River is presented in Section 2.2 and Appendix A (Ecological Characterization) of this document. These sections detail the physical setting, habitats, and biotic communities of the river in both the aquatic and terrestrial environments.

Investigations of the nature and extent of contaminants in the Housatonic River watershed have previously been conducted by GE, EPA, and others. PCBs have been identified as the main COPC, and other contaminants such as dioxins/furans and PAHs have also been identified at the GE facility. In Section 2.3, a summary of the sources, amounts, and patterns of contaminant releases and receiving bodies are presented. Further discussion of the nature and extent of contamination is provided in the Resource Conservation and Recovery Act (RCRA) Facility Investigation Report (BBL and QEA 2003).

The purpose of the Pre-ERA was to identify contaminants that warranted more detailed analyses in the ERA, and those that could be removed from further consideration because they pose minimal risk. A summary of the Pre-ERA is provided in Section 2.4. The complete Pre-ERA is included as Appendix B to this document.

An overview of the environmental behavior of PCBs and other COPCs is presented in Section 2.5. This section includes discussions of the transport of the contaminants from their point(s) of release, partitioning behavior in different media, and biotic and abiotic degradation in these media.



Relevant to all endpoints

+ Relevant to specific assessment endpoints

Wildlife assessment approach

MK01|O:\20123001.096\ERA_FNL\ERA_FNL_ES_Fig ES-3.ppt

Ecological Risk Assessment
GE/Housatonic River Site
Rest of River

Figure ES-3
Ecological Risk Assessment Roadmap

The effects and mechanisms of toxicity of the contaminants identified as COPCs within the Housatonic River and floodplain are discussed, with an emphasis on PCBs, in Section 2.6, with additional detail in the effects assessment portion of each assessment endpoint section and corresponding appendix. Assessment and measurement endpoints are defined and described in Section 2.8. An assessment endpoint is defined as the "explicit expression of the environmental value that is to be protected." A measurement endpoint is defined as "a measurable ecological characteristic that is related to the valued characteristic chosen as the assessment endpoint," and is a measure of biological effects (e.g., mortality, reproduction, growth).

Assessment Endpoints Selected for the Rest of River ERA

- Community structure, survival, growth, and reproduction of benthic invertebrates.
- Community condition, survival, reproduction, development, and maturation of amphibians.
- Survival, growth, and reproduction of fish.
- Survival, growth, and reproduction of insectivorous birds.
- Survival, growth, and reproduction of piscivorous birds.
- Survival, growth, and reproduction of piscivorous mammals.
- Survival, growth, and reproduction of omnivorous and carnivorous mammals.
- Survival, growth, and reproduction of Special Status Species (Endangered, Threatened).

The conceptual model outlined in Section 2.7 describes the relationship between the COPCs and the assessment endpoints. Section 2.9 describes the analytical approach used to estimate risks and the weight-of-evidence approach used to develop the conclusions.

Sections 3 through 11 (and their corresponding appendices) provide an overview of the exposure assessment, the effects assessment, and the risk characterization for each representative species or representative group of species. The exposure assessment sections include a description of the data collection activities and the studies conducted to determine concentrations of contaminants of potential concern (COPCs) in water, soil, sediment, and biota samples. The list of COPCs was further narrowed with additional screening for the specific endpoint, resulting in the list of contaminants of concern (COCs) retained for that endpoint risk assessment.

The effects assessment sections begin with an overview of the toxicity of PCBs and the other COCs. The effects literature was reviewed for each major representative species group and COC. The goal of this review was to identify studies that could be used to develop effects metrics for use in risk characterization. The effects metrics developed ranged from concentration- or dose-response curves to benchmarks, depending on the quality and relevance of the data available.

The risk characterization sections first provide an overview of the site-specific studies, analyses of the results, and conclusions, and then consider the three major lines of evidence (where available) using a weight-of-evidence approach. There is also a discussion of the uncertainties associated with the assessment for that endpoint, an evaluation of potential risks to receptors other than those chosen as the representative species, and a discussion of potential risks downstream of the PSA for receptors of concern.

Section 12 summarizes the conclusions of the ERA, and presents a discussion of these conclusions in the context of the uncertainties and other factors that could not be expressly quantified in the evaluation of the assessment endpoints.

ES.5 OVERVIEW OF THE ASSESSMENT ENDPOINT CONCLUSIONS

ES.5.1 Risks in the Primary Study Area

The assessments in the ERA were conducted using various lines of evidence including, in many cases, different measurement endpoints and effects metrics. These lines of evidence—defined as information derived from different sources that can be used to describe and interpret risk—were integrated into a graphical representation of risk using the weight-of-evidence (WOE) approach. The WOE provides an objective process by which measurement endpoints are related to an assessment endpoint to evaluate whether significant risk is posed to the environment. A formal WOE can range from a simple qualitative assessment to a highly quantitative evaluation; however, no matter what form the WOE assessment takes, it should provide documentation of the thought process used when assessing potential ecological risk.

In the first step of the WOE approach, weights are assigned to measurement endpoints based on 10 attributes related to: (1) strength of association between assessment and measurement endpoints; (2) data and study quality; and (3) study design and execution. The second step of the approach is to evaluate the magnitude of response in the measurement endpoint, considering whether the measurement endpoint indicates the presence or absence of harm, and if the magnitude of response is low, intermediate, or high. The WOE process concludes by plotting the output of the two preceding steps in a matrix for all measurement endpoints associated with a given assessment endpoint. The matrix allows easy visual examination of agreements or divergences among measurement endpoints, facilitating interpretation with respect to the assessment endpoint.

The results of the WOE process and the final WOE matrix are summarized below for each of the eight assessment endpoints considered in this ERA. Following that discussion, to facilitate comparison of risks among aquatic life and wildlife receptors and to give an overview of the findings of the risk assessment, assessment results are converted to probabilistic hazard quotients (HQs). An HQ is the expected environmental concentration or dose of a contaminant divided by its estimated low or no toxic effect concentration or dose. Although risk is not necessarily linearly related to the magnitude of HQ, in general, higher quotients indicate greater risk. The methods used to calculate the probabilistic HQs and the results of these analyses for each endpoint are summarized in this section.

ES.5.1.1 Aquatic Assessment Endpoints

Benthic Invertebrates—The WOE results for the benthic invertebrate assessment endpoint are shown in Table ES-1. In this table, the measurement endpoints for the three lines of evidence: water, sediment, and tissue chemistry (C); toxicity tests (T); and benthic community measures (B) are listed, as are the weight of the measurement endpoint, evidence of harm, and magnitude of response. This table indicates that the majority of endpoints suggest some risk for benthic communities in both coarse- and fine-grained sediment. The conclusion is that there is intermediate to high risk to much of the benthic community.

Amphibians—The results of the WOE assessment for amphibians are presented in Table ES-2. In the amphibian WOE matrix, the measurement endpoints for the three lines of evidence:

Table ES-1

Evidence of Harm and Magnitude of Effects for Measurement Endpoints Related to Maintenance of a Healthy Benthic Community

Measurement Endpoints	Weighting Value	Evidence of Harm	Magnitude of Harm (High, Intermediate, or Low)	
			Coarse-Grained	**Fine-Grained**
C. Chemistry Endpoints				
C-1. Concentration of PCB in water in relation to concentrations reported (in literature) to be harmful to benthos.	Low/Moderate	Yes	Intermediate	Intermediate
C-2. Concentration of PCB in sediment in relation to concentrations reported (in literature) to be harmful to benthos.	Moderate	Yes	High	High
C-3. Concentration of PCB in invertebrate tissues in relation to concentrations reported (in literature) to be harmful to benthos.	Moderate	Yes	High	High
T. Toxicity Endpoints				
T-1. Sediment toxicity to multiple invertebrate species, as measured in the laboratory toxicity tests (*Hyalella*, *Chironomus*).	Moderate/High	Yes	High	High
T-2. Sediment toxicity to multiple invertebrate species, as measured in the in situ toxicity tests (*Hyalella*, *Chironomus*, *Daphnia*).	Moderate/High	Yes	High	High
T-3. Indications of PCB as toxicity driver in TIE investigations.	Moderate	Yes	–	Intermediate
B. Benthic Community Endpoints				
B-1. Abundance, richness, biomass, and diversity of assemblages, relative to reference stations of comparable substrate and habitat.	Moderate	Yes	Intermediate	Low
B-2. Benthic community structure, as assessed using multivariate analysis of benthic metrics (rank analysis, multidimensional scaling) and multiple regression analyses.	Moderate	Yes	Intermediate	Low
B-3. Water quality assessment using modified Hilsenhoff Biotic Index (MHBI)	Moderate	No	–	–
B-4. Evaluation of individual taxa using species sensitivity distribution (SSD) analysis of abundance endpoint.	Moderate/High	Yes	High	Intermediate

Table ES-2

Evidence of Harm and Magnitude of Effects for Measurement Endpoints Related to Maintenance of Amphibian Populations in the Housatonic River PSA

Measurement Endpoints	Weighting Value (High, Moderate, Low)	Evidence of Harm (Yes, No, Undetermined)	Magnitude (High, Intermediate, Low)
C. Chemical Measures			
C. Concentration of PCB in frog tissues in relation to concentrations reported to be harmful to amphibians	Moderate	Yes	Low
W. Wood Frog Toxicological Measures			
W-1. Sediment toxicity to hatchling/late embryo life stages	Mod/High	No	--
W-2. Sediment toxicity to larval life stages	Mod/High	Yes	Intermediate
W-3. Sediment toxicity to late larval/metamorph life stage	Mod/High	Yes	High
W-4. GE Study (juvenile wood frogs)	Low	Undetermined	--
L. Leopard Frog Toxicological Measures			
L-1. Sediment toxicity to hatchling/late embryo life stages	Mod/High	Yes	Low
L-2. Sediment toxicity to larval life stages	Mod/High	Yes	High
L-3. Sediment toxicity to late larval/metamorph life stage	Mod/High	Yes	High
L-4. Sediment toxicity to adult leopard frogs (reproductive health)	Mod/High	Yes	High
B. Biology			
B-1. Vernal pool community study	Mod/High	Yes	Low
B-2. GE leopard frog egg mass survey	Low/Mod	Undetermined	--
B-3. Anecdotal observations during collections for reproductive study	Moderate	Yes	Low

tissue chemistry (C); wood frog toxicity tests (W) and leopard frog toxicity tests (L); and field surveys (B) are listed. As shown in the table, many of the endpoints indicated some degree of risk. Evidence of harm to amphibians based on the two GE studies was undetermined due to limitations in the study designs. The only endpoint that did not indicate potential risk was the earliest life stage wood frog toxicity endpoint, for which there is a mechanistic explanation for the lack of response. The plots also indicate that four endpoints exhibited a high degree of risk combined with a moderate to high confidence rating.

In addition, a population model was constructed for wood frogs to determine whether effects from PCBs on individual wood frogs influence the populations within the PSA. A 10-year simulation, both with and without the effects of PCBs, was conducted. The model demonstrated that effects observed in the toxicity studies would result in population-level impacts.

The conclusion is that there is a significant risk to amphibians as indicated by the preponderance of the evidence, the relative weights of the measurement endpoints, and the population modeling. The "no harm" value of measurement endpoint W-1 does not diminish the overall conclusion, because the study demonstrated that embryo/early larval life stages are fairly insensitive to the effects of maternally transferred PCBs relative to later juvenile life stages exposed to contaminated media.

Fish—The weight-of-evidence results for fish in the PSA are shown in Table ES-3. In the fish WOE matrix, the measurement endpoints for the three lines of evidence: site-specific toxicity tests (T); fish tissue chemistry (C); and field surveys (F) are listed. This table illustrates that for two of the three lines of evidence (site-specific toxicity and fish tissue concentrations compared to MATCs), intermediate risks to fish in the Housatonic River are predicted. However, the field surveys suggest that PCBs and/or other COCs are not causing obvious effects to fish populations. Therefore, the overall risk conclusion for fish is low/intermediate risk. The lines of evidence collectively indicate that although widespread and obvious effects (e.g., acute mortality to adults or total reproductive failure) are not currently occurring, there is evidence of biological

Table ES-3

Evidence of Harm and Magnitude of Effects for Measurement Endpoints Related to Maintenance of a Healthy Fish Community

Measurement Endpoints	Weighting Value (High, Moderate, Low)	Evidence of Harm (Yes, No, Undetermined)	Magnitude of Harm (High, Intermediate, or Low)
C. Chemistry Endpoints			
C-1. Concentration of PCB in tissues of representative species in relation to threshold concentrations reported (in literature) to be harmful to fish.	Mod	Yes	Intermediate
C-2. Concentration of PCB in tissues of representative species in relation to threshold concentrations reported (in Phase I study; Tillitt et al. 2003a) to be harmful to fish	Mod/High	Yes	Intermediate
C-3. Concentration of PCB in tissues of representative species in relation to threshold concentrations reported (in Phase II study; Tillitt et al. 2003b) to be harmful to fish.	Mod/High	Yes	Intermediate
T. Toxicity Endpoints			
T-1. Indications of reproductive and developmental impairment of largemouth bass spawned from Housatonic River adults in Phase I study (Tillitt et al. 2003a).	Mod/High	Yes	Intermediate
T-2. Indications of reproductive and developmental impairment of bass, medaka, and rainbow trout fry reared from eggs injected with Housatonic River extracts in Phase II study (Tillitt et al. 2003b).	High	Yes	Intermediate
F. Field Study Endpoints			
F-1. Abundance and biomass of fish species observed in EPA fish collections, and GE fish community assessment.	Mod	No	--
F-2. Reproduction and nest condition metrics from GE reproduction study (R2 Resource Consultants 2002) including evidence of largemouth bass reproduction, nest conditions, YOY abundance and growth, and adult growth rate.	Mod	No	--
F-3. Largemouth bass population demographics including age-structure analysis and adult condition.	Low/Mod	No	--

impairment during sensitive life stages. Biological indicators such as histopathology data, disease and deformity data, and induction of reproductive/endocrine biomarkers all support the conclusion that the health of individual fish has been impaired. The effect of this impairment on local fish population size, recruitment, and/or resilience to natural or anthropogenic stressors is not known.

ES.5.1.2 Wildlife Assessment Endpoints

Insectivorous Birds—The weight-of-evidence results for exposure of insectivorous birds to tPCBs are presented in Table ES-4, and for exposure to TEQ in Table ES-5. Two lines of evidence are presented in the table: field studies, and modeled exposure and effects. The results from the modeled exposure and effects line of evidence suggest that tPCBs and TEQ pose intermediate to high risks to tree swallows living in the PSA. However, the field study line of evidence suggests that, although effects are occurring, they are minor. The uncertainty concerning the field-based threshold range for tPCBs likely means that risks of this COC are overestimated for the PSA. Even the upper end of the tPCB threshold range is associated with equivocal evidence for adverse effects to tree swallows. For TEQ, the threshold range is quite broad. The available evidence from field studies indicates that tree swallows are tolerant to exposure to persistent organochlorines such as tPCBs and TEQ. If the tree swallow threshold is near the upper end of the threshold range, then the current modeled exposure and effects line of evidence is overestimating risks of TEQ to tree swallows. Thus, the WOE assessment supports a finding of low risk for tree swallows exposed to tPCBs and TEQ in the PSA. This conclusion, however, is uncertain because of the conflicting results in the WOE assessment.

The results from the modeled exposure and effects lines of evidence suggest that tPCBs and TEQ pose an intermediate to high risk to American robins inhabiting the PSA. The American robin field study, however, suggests that reproductive success is not being impaired by the tPCBs and TEQ in the PSA. The uncertainty in the modeled exposure and effects line of evidence, outlined below, likely means the approach overestimates the risks of tPCBs and TEQ to American robins in the PSA. The WOE assessment, therefore, supports a conclusion of low risk to American robins exposed to tPCBs and TEQ in the PSA. This conclusion, however, is uncertain because of the conflicting results in the WOE assessment.

Table ES-4

Evidence of Harm and Magnitude of Effects for Insectivorous Birds Exposed to tPCBs in the Housatonic River PSA

Measurement Endpoints	Weighting Value (High, Moderate, Low)	Evidence of Harm (Yes, No, Undetermined)	Magnitude (High, Intermediate, Low)
Field Study	High (Tree swallow) Moderate/High (American robin)	Yes (Tree swallow) No (American robin)	Low (Tree swallow) -- (American robin)
Modeled Exposure and Effects	Moderate (Tree swallow) Moderate (American robin) Moderate (Wood duck)	Yes (Tree swallow) Yes (American robin) Yes (Wood duck)	High (Tree swallow) High (American robin) High (Wood duck)

Table ES-5

Evidence of Harm and Magnitude of Effects for Insectivorous Birds Exposed to TEQ in the Housatonic River PSA

Measurement Endpoints	Weighting Value (High, Moderate, Low)	Evidence of Harm (Yes, No, Undetermined)	Magnitude (High, Intermediate, Low)
Field Study	High (Tree swallow) Moderate/High (American robin)	Yes (Tree swallow) No (American robin)	Low (Tree swallow) -- (American robin)
Modeled Exposure and Effects	Moderate (Tree swallow) Moderate (American robin) Moderate/High (Wood duck)	Yes (Tree swallow) Yes (American robin) Yes (Wood duck)	Intermediate (Tree swallow) Intermediate (American robin) High (Wood duck)

The results from the modeled exposure and effects lines of evidence suggest that tPCBs and TEQ pose high risk to wood ducks inhabiting the Housatonic River PSA. For TEQ, the threshold range is quite narrow. The available evidence from literature field studies indicates that wood ducks are sensitive to exposure to TEQ congeners. The uncertainty in the modeled exposure and effects line of evidence likely means the modeling approach overestimates the risks of tPCBs to wood ducks in the PSA. However, the modeled egg TEQ concentrations were similar to TEQ concentrations in wood duck egg samples from the PSA, and well within the probability bounds.

Piscivorous Birds—The WOE analysis indicates that exposure of piscivorous birds, such as the belted kingfisher and osprey (Tables ES-6 and ES-7), to tPCBs and TEQ in the PSA, could lead to adverse reproductive effects in some species. The two lines of evidence used to support this conclusion were the field study of belted kingfisher productivity and the comparison of modeled exposure with effects to piscivorous birds (for belted kingfisher and osprey).

Table ES-6

Evidence of Harm and Magnitude of Effects for Piscivorous Birds Exposed to tPCBs in the Housatonic River PSA

Measurement Endpoints	Weighting Value (High, Moderate, Low)	Evidence of Harm (Yes, No, Undetermined)	Magnitude (High, Intermediate, Low)
Modeled Exposure and Effects	Moderate	Yes (Belted kingfisher) Yes (Osprey)	High (Belted kingfisher) High (Osprey)
Belted Kingfisher Field Study (Henning 2002)	Moderate	No (Belted kingfisher)	--

Table ES-7

Evidence of Harm and Magnitude of Effects for Piscivorous Birds Exposed to TEQ in the Housatonic River PSA

Measurement Endpoints	Weighting Value (High, Moderate, Low)	Evidence of Harm (Yes, No, Undetermined)	Magnitude (High, Intermediate, Low)
Modeled Exposure and Effects	Moderate	Yes (Belted kingfisher) Yes (Osprey)	Intermediate (Belted kingfisher) Intermediate (Osprey)
Belted Kingfisher Field Study (Henning 2002)	Moderate	No (Belted kingfisher)	--

For the assessment of risks to belted kingfishers, both lines of evidence were employed. The modeled exposure and effects line of evidence indicated that belted kingfishers in the PSA are likely to receive a tPCB dose greater than what the most tolerant species known from the literature can be exposed to without effects. For TEQ, the risk picture is less clear because the threshold range for this COC is very wide and the exposure estimates for belted kingfishers fell within this range. Thus, without effects data specific to belted kingfishers, it is difficult to make definitive conclusions about the risks of TEQ to this species. The field study of belted kingfisher productivity, however, indicated that these birds are able to reproduce in the PSA. This line of evidence was given weighting equal to the exposure and effects modeling line of evidence, despite concerns about the design and conduct of the field study. Therefore, belted kingfishers are considered to be at intermediate risk in the PSA as a result of exposure to tPCBs and TEQ. The conclusion of intermediate risk to belted kingfishers is uncertain because the two lines of evidence did not give concordant results.

For ospreys, only the modeled exposure and effects line of evidence was available to assess risk to these birds. As with belted kingfishers, this line of evidence indicated that ospreys in the PSA are likely to receive a tPCB dose that is greater than what the most tolerant species known from the literature can be exposed to without effects. The risks due to exposure to TEQ are unclear, as the estimates for exposure also fell within toxicity threshold range. However, a site-specific study that investigated the effects of COCs on osprey in the PSA was not conducted. The PSA contains suitable habitat for ospreys, with abundant prey. If ospreys were to nest in the PSA, they will be at risk as a result of exposure to tPCBs and TEQ.

Piscivorous Mammals—The results of the WOE assessment for piscivorous mammals are presented for tPCBs and TEQ, respectively, in Tables ES-8 and ES-9. All three lines of evidence: field studies (EPA), mink feeding study, and modeled exposure and effects indicated that the elevated concentrations of tPCBs and TEQ in the PSA of the Housatonic River are causing adverse effects of high magnitude to mink and river otter. The field surveys indicated that mink and river otter are rarely present in the PSA, except during winter, and likely have not established home territories close to the main channel despite suitable mink and otter habitat. The MSU feeding study indicated that feeding adult female mink with a diet containing approximately 0.9% fish from the PSA caused a 20% reduction in kit survival from 0 to 6 weeks of age. Because mink in the wild typically consume 20% or more fish in their diet, the associated risk is correspondingly higher. In addition, other components of the mink diet in the PSA (e.g., crayfish) have high concentrations of tPCBs and TEQ. Further, the jaw lesion study indicated that erosion of the jaw occurs at even lower doses and exhibits a dose-response relationship. The occurrence of jaw lesions coincides with the induction of Ah-receptor-regulated enzymes (ECOD and EROD), also in a dose-response manner.

Table ES-8

Evidence of Harm and Magnitude of Effects for Piscivorous Mammals Exposed to tPCBs in the Housatonic River PSA

Measurement Endpoints		Weighting Value (High, Moderate, Low)	Evidence of Harm (Yes, No, Undetermined)	Magnitude (High, Intermediate, Low)
Field Surveys	EPA	Moderate/High	Yes (Mink) Yes (River otter)	High (Mink) High (River otter)
	GE	Moderate	No (Mink) No (River otter)	– (Mink) – (River otter)
Feeding Study		High	Yes (Mink)	High (Mink)
Modeled Exposure and Effects		Moderate/High	Yes (Mink) Yes (River otter)	High (Mink) High (River otter)

Table ES-9

Evidence of Harm and Magnitude of Effects for Piscivorous Mammals Exposed to TEQ in the Housatonic River PSA

Measurement Endpoints		Weighting Value (High, Moderate, Low)	Evidence of Harm (Yes, No, Undetermined)	Magnitude (High, Intermediate, Low)
Field Surveys	EPA	Moderate/High	Yes (Mink) Yes (River otter)	High (Mink) High (River otter)
	GE	Moderate	No (Mink) No (River otter)	– (Mink) – (River otter)
Feeding Study		High	Yes (Mink)	High (Mink)
Modeled Exposure and Effects		Moderate/High	Yes (Mink) Yes (River otter)	High (Mink) High (River otter)

The high risks evident from the feeding study are further supported by the modeled exposure and effects line of evidence. The estimated potential for exposure is so high that even individual mink and otter that forage in the PSA for only short periods of time (10% of foraging time or less) are at an intermediate or higher risk from tPCBs and TEQ.

Omnivorous and Carnivorous Mammals—The WOE results for omnivorous and carnivorous mammals are shown in Table ES-10 for tPCB and Table ES-11 for TEQ. Data from three lines of evidence were available: field surveys, a population demography study of short-tailed shrew, and exposure and effects modeling.

The weight-of-evidence analysis indicates, based on modeling results, high risk for short-tailed shrews exposed to tPCBs and no appreciable risk to short-tailed shrew exposed to TEQs in the PSA. This conclusion, however, is uncertain because of the lack of species-specific measures of effects in the literature used for the exposure and effects modeling. The conclusions in the short-tailed shrew demographic study at the site conducted for GE are not in agreement with the results from the modeling of exposure and effects line of evidence. However, the results of the supplemental analyses conducted by EPA of the data from the GE demographic study of short-tailed shrews are in agreement with the modeling results, suggesting that there are intermediate effects from exposure to COCs in the contaminated areas of the PSA. Risk to short-tailed shrew based on field surveys was undetermined.

The WOE also suggests, based on the modeling results for red fox, an intermediate risk to fox exposed to tPCBs and TEQ in the PSA. This assessment is uncertain because measures of effects of tPCBs and TEQ to fox were not available in the literature and there are no other lines of evidence, because the field study was not designed to distinguish the potential impacts from exposure to contaminants.

Table ES-10

Evidence of Harm and Magnitude of Effects for Omnivorous and Carnivorous Mammals Exposed to tPCBs in the Housatonic River PSA

Measurement Endpoints	Weighting Value (High, Moderate, Low)	Evidence of Harm (Yes, No, Undetermined)	Magnitude (High, Intermediate, Low)
Field Surveys	Moderate	Undetermined (Shrew) Undetermined (Red fox)	— (Shrew) — (Red fox)
Population Demography Field Study	Moderate/High	Yes* (Shrew)	Intermediate (Shrew)
Modeled Exposure and Effects	Moderate/High	Yes (Shrew) Yes(Red fox)	High (Shrew) Intermediate (Red fox)

* Based on EPA's re-analysis of the demographic study data.

Table ES-11

Evidence of Harm and Magnitude of Effects for Omnivorous and Carnivorous Mammals Exposed to TEQ in the Housatonic River PSA

Measurement Endpoints	Weighting Value (High, Moderate, Low)	Evidence of Harm (Yes, No, Undetermined)	Magnitude (High, Intermediate, Low)
Field Surveys	Moderate	Undetermined (Shrew) Undetermined (Red fox)	— (Shrew) — (Red fox)
Modeled Exposure and Effects	Moderate/High	No (Shrew) Yes (Red fox)	— (Shrew) Intermediate (Red fox)

Threatened and Endangered Species—The results of the WOE evaluation for threatened and endangered (T&E) species using a single line of evidence, modeled exposures and effects, are shown in Table ES-12 and Table ES-13 for tPCBs and TEQ, respectively. The results of the risk characterization showed that the highest risk to T&E species is to bald eagles and American bittern from exposure to tPCBs and TEQ.

The WOE analysis indicates that risks to bald eagles and American bittern exposed to tPCBs are high. Risks to small-footed myotis exposed to tPCBs and TEQ are intermediate.

ES.5.1.3 Hazard Quotients

To facilitate comparison of risks among aquatic life and wildlife receptors and to give an overview of the findings of the risk assessment, assessment results were converted to probabilistic hazard quotients (HQs). An HQ is the expected environmental concentration or dose of a contaminant divided by its estimated low or no toxic effect concentration or dose, similar to a hazard index that is used to describe noncancer risks to people. Higher quotients generally indicate greater risk, although the magnitude of risk is not necessarily linearly proportional to the HQ. Figures ES-4 through ES-7 summarize the ranges of HQs for exposure to tPCBs and TEQ determined for each the assessment endpoints in the PSA. The methods used to calculate the probabilistic HQs and the results of these analyses for each endpoint are summarized below.

The thresholds used in HQ calculations represent concentrations at, or close to, those demonstrated to cause adverse responses to organisms in site-specific studies. Thus, HQ exceedances of 1 are cause for concern.

Benthic Invertebrates—Hazard quotients were calculated by dividing concentrations of COCs in site sediment by 3 mg/kg, the site-specific effects benchmark for benthic invertebrates exposed to tPCBs in sediment. These results indicate that significant risk was observed in all reaches of the PSA. Predicted risks were greatest in the upstream (Reach 5A) and Woods Pond (Reach 6) sediment. Because of small-scale variation in sediment tPCB concentrations, although the majority of benthic invertebrates in the PSA are at risk (i.e., HQ > 1), some individuals in less-contaminated areas are not.

Table ES-12

Evidence of Harm and Magnitude of Effects of T&E Species Exposed to tPCBs in the Housatonic River PSA

Measurement Endpoints	Weighting Value (High, Moderate, Low)	Evidence of Harm (Yes, No, Undetermined)	Magnitude (High, Intermediate, Low)
Modeled exposure and effects, Bald Eagle	Moderate/High	Yes	High
Modeled exposure and effects, American Bittern	Moderate	Yes	High
Modeled exposure and effects, Small-Footed Myotis	Moderate	Yes	Intermediate

Table ES-13

Evidence of Harm and Magnitude of Effects for T&E Species Exposed to TEQ in the Housatonic River PSA

Measurement Endpoints	Weighting Value (High, Moderate, Low)	Evidence of Harm (Yes, No, Undetermined)	Magnitude (High, Intermediate, Low)
Modeled exposure and effects, Bald Eagle	Moderate/High	Yes	High
Modeled exposure and effects, American Bittern	Moderate	Yes	High
Modeled exposure and effects, Small-Footed Myotis	Moderate	Yes	Intermediate

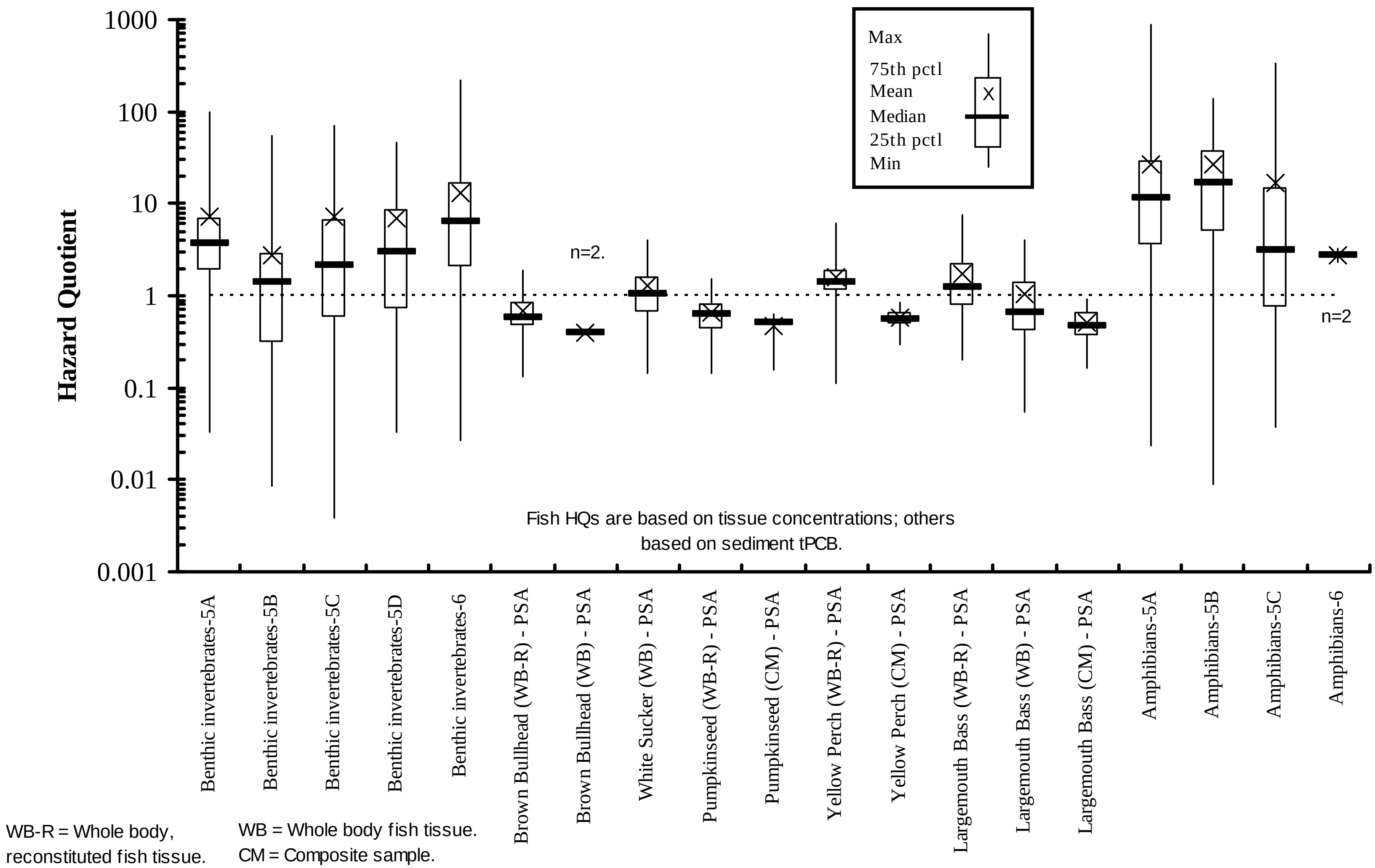


Figure ES-4 Hazard Quotients for Aquatic Biota Exposed to tPCBs in the Housatonic River PSA

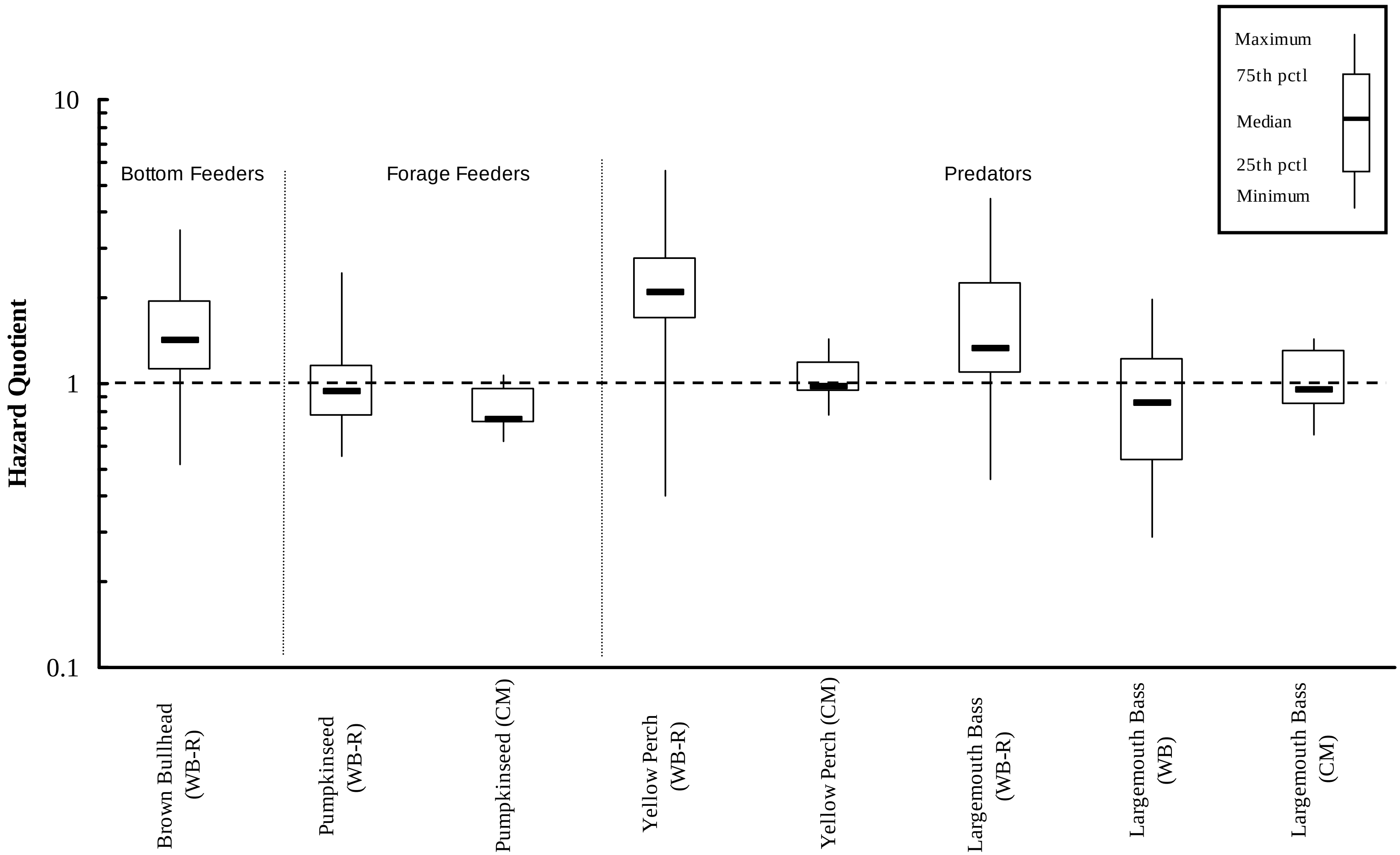


Figure ES-5 Hazard Quotients for Fish Exposed to 2,3,7,8-TCDD TEQ in the Housatonic River PSA

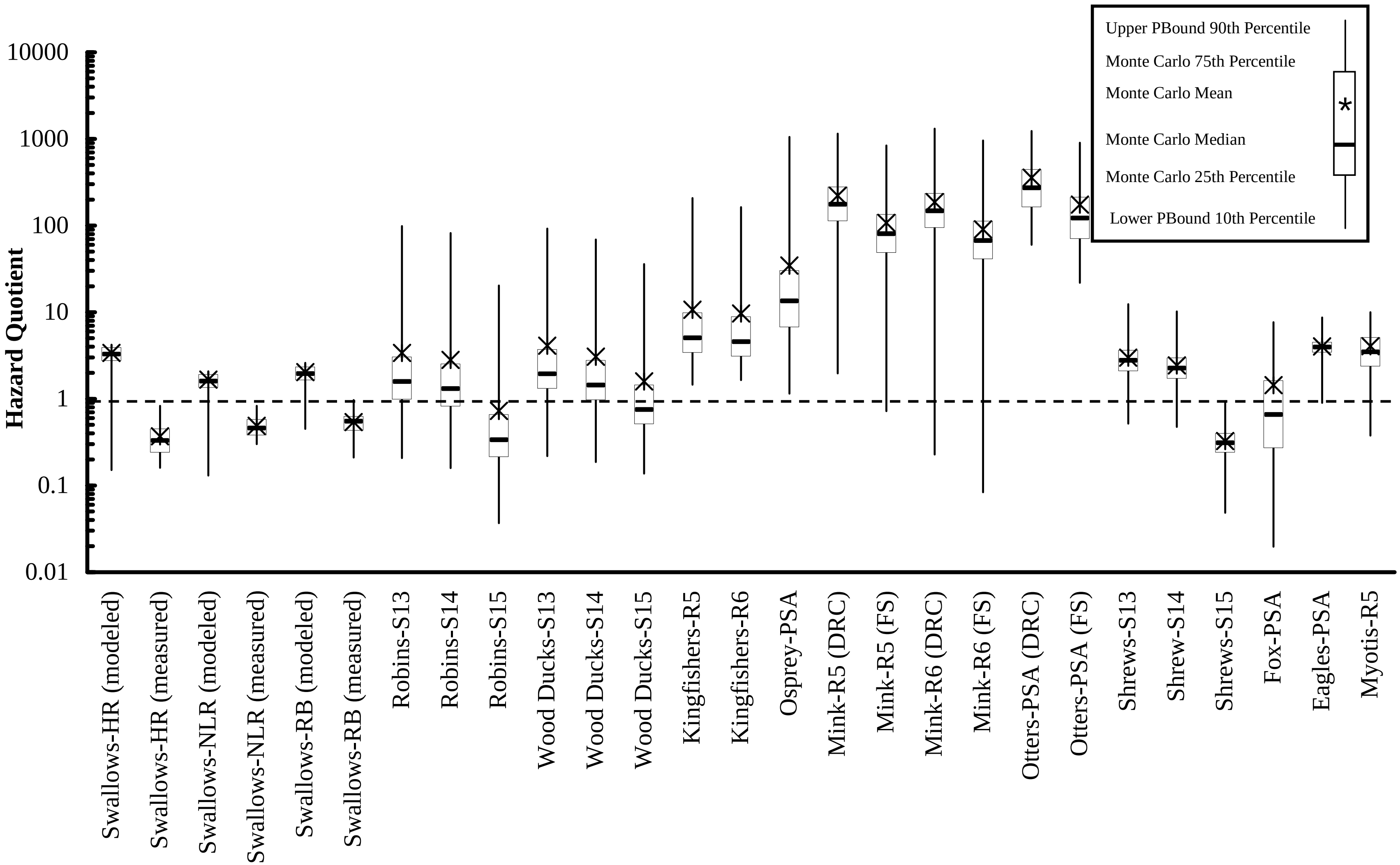


Figure ES-6 Hazard Quotients for Wildlife Exposed to tPCBs in the Housatonic River PSA

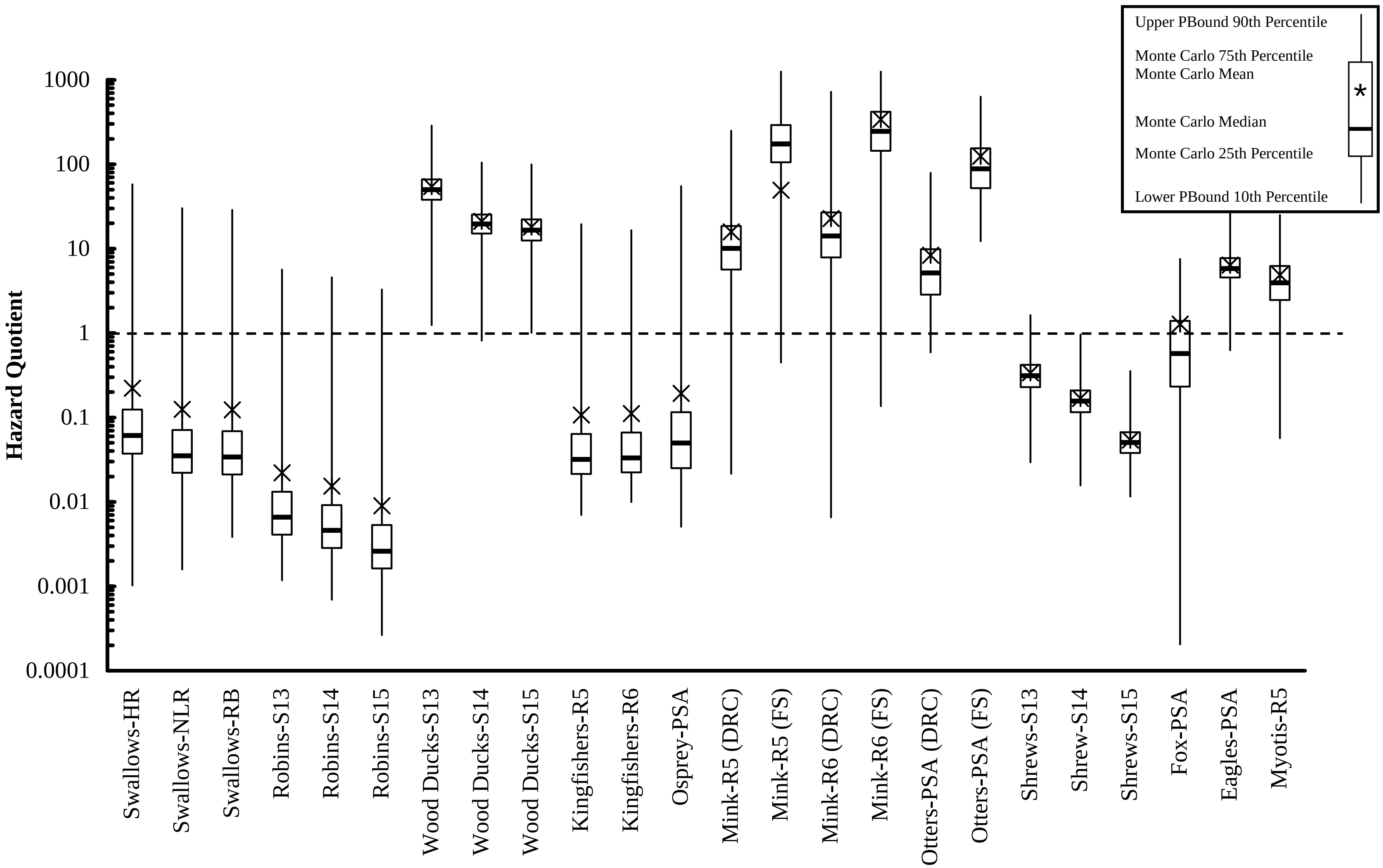


Figure ES-7 Hazard Quotients for Wildlife Exposed to 2,3,7,8-TCDD TEQ in the Housatonic River PSA

Amphibians—For amphibians, HQs were calculated by dividing summary statistics for vernal pool sediment concentrations by the site-specific effects benchmark for amphibians exposed to tPCBs in sediment (3.27 mg/kg tPCBs). These results indicate significant risk in all reaches of the PSA, with HQs above 1. Predicted risks were greatest in the upstream (Reach 5A) vernal pool habitats. These results indicate that the majority of amphibians are at risk (i.e., HQ > 1), with higher levels of risk (i.e., HQ > 5) in a large percentage of vernal pools (about 50% of pools in Reaches 5A and 5B).

Fish—For fish, HQs were calculated separately for the five representative warmwater species by dividing summary statistics for exposure by the site-specific tissue effects benchmark protective of all species of warmwater PSA fish (55 mg/kg ww tPCB). These results indicate that risk occurs in all reaches of the PSA.

Predicted risks were greatest for predator fish at the top of the food web (e.g., largemouth bass) and when fish reach their maximum adult tPCB concentrations. The ERA indicates that these HQs are indicative of sublethal (e.g., reproductive and developmental) responses to offspring; the pathway for the manifestation of effects is through the maternal transfer of tPCBs to eggs. Acute mortality to adults is not expected for most fish.

In addition to tPCBs, fish HQs were derived for TEQ using the average of the effects thresholds relevant to warmwater fish species obtained from site-specific toxicity studies (44 ng/kg TEQ). The magnitudes and probabilities of risk for TEQ are generally similar to tPCB risks.

Wildlife Endpoints—For wildlife, the distributions from Monte Carlo analyses for total daily intake of COCs by representative species were each divided by the corresponding effects metrics used to estimate risks. In the case of a dose-response curve effects metric (e.g., mink exposed to tPCBs), the effects metric was specified as a uniform distribution of dose ranging from 10% to 20% effect. A similar approach was used for field-based effects metrics (e.g., tree swallows exposed to tPCBs), and threshold ranges (e.g., belted kingfishers exposed to TEQ).

In addition to plots developed for mink and otter exposed to tPCBs and TEQ using the results of the literature-based dose-response curve, plots were also developed using the results of the mink feeding study conducted in support of this ERA. In this case, the denominators were the 10%

and 20% effects doses estimated from the mink feeding study for survival of mink kits from 0 to 6 weeks of age.

Unlike traditional HQs, the probabilistic HQs for wildlife do not include safety factors, i.e., are not conservative. No safety factors were used to estimate the effects metrics (except in the case of the bald eagle), and uncertainties regarding exposure model inputs were explicitly propagated through the probability bounds and Monte Carlo analyses. Thus, HQ exceedances of 1 are cause for concern.

Wildlife risks from tPCBs and TEQ are highest for mink and river otter, with HQs between 100 and 500 for tPCBs, and between 5 and 10 for TEQ, using the results of the literature-based dose-response curve. The HQs for tPCBs were similar when the results of the site-specific mink feeding study were used to derive the effects threshold. Wildlife risks from tPCBs are generally higher than risks from TEQ by one to several orders of magnitude, with the exception of wood duck, which was more sensitive to TEQ.

The risks from tPCBs and TEQ to many of the wildlife species are uncertain to the extent that the range of HQs spans 1. The highest and lowest values depicted for the wildlife HQs are the extreme representations of uncertainty because they are tail outputs from the probability bounds analyses, a technique designed to propagate all forms of uncertainty (e.g., inability to precisely specify distribution type or parameter values for a distribution). Thus, the range of HQs shown in the boxes should be interpreted as representing a reasonable range within which the HQ estimate occurs for the receptor of interest, and the lines should be interpreted as representing the extremes within which the HQ could occur.

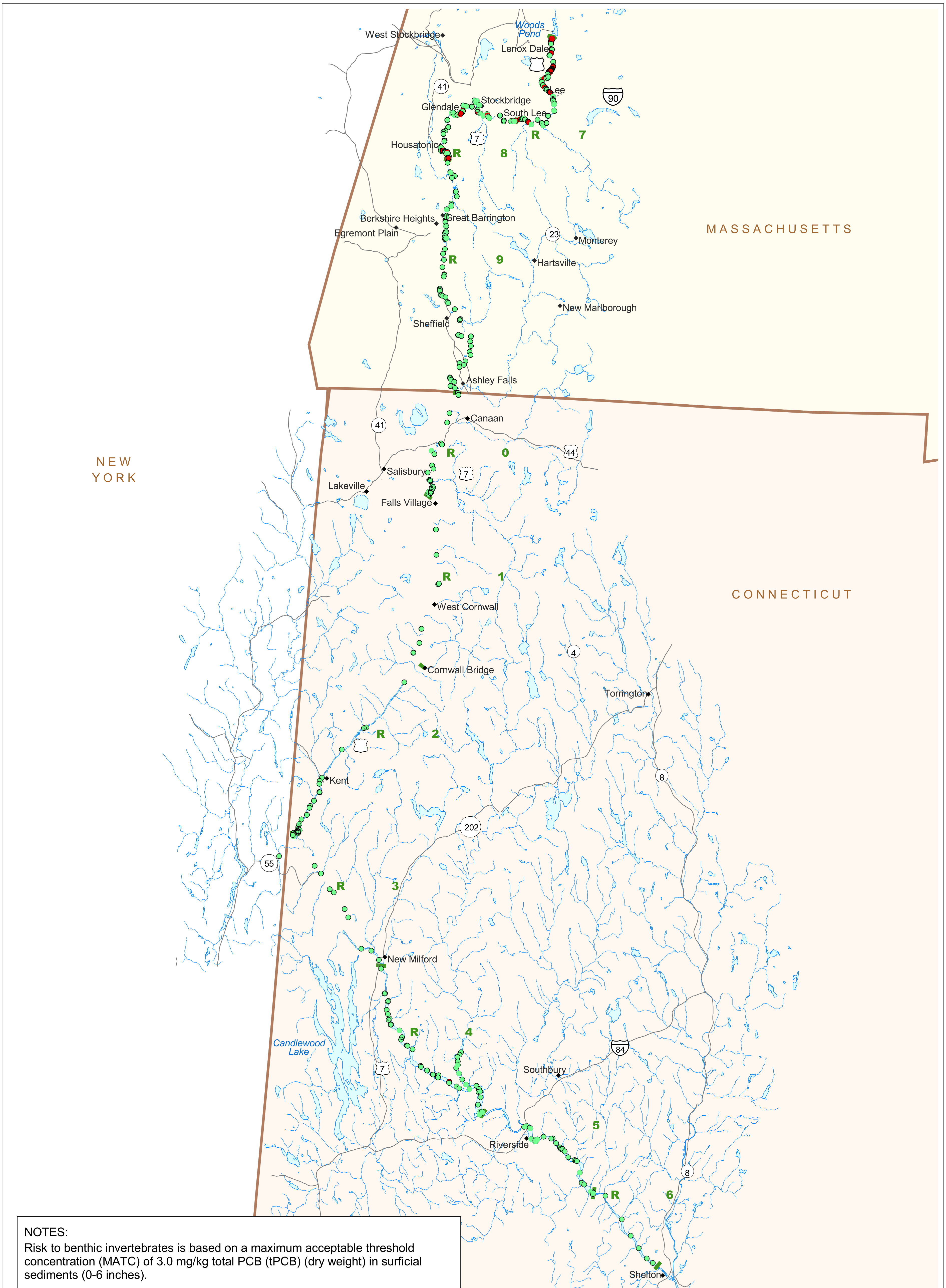
ES.5.2 Risks Downstream of the Primary Study Area

Because of the reduced contaminant concentrations downstream of the Primary Study Area (PSA) and shifts in aquatic habitat types associated both with river gradient and location of dams, a different approach than that applied in the PSA was followed to assess potential ecological risks of tPCBs to biota in areas downstream of the PSA. The assessment of potential ecological risks was conducted using mapping (GIS) techniques and threshold concentrations that indicate potential risk for seven taxonomic groups or species selected based on the outcome of the evaluations performed in the PSA and the habitat characteristics found downstream. These groups or species are benthic invertebrates, amphibians, warmwater fish, trout, mink, otter, and bald eagles. A screening assessment for the downstream floodplain was also conducted for short-tailed shrews exposed to tPCBs in soil. Prey tissue concentration data downstream of the PSA are lacking for shrews, making a more refined assessment not feasible.

For each of these groups, a maximum acceptable threshold concentration (MATC) for tPCBs was developed based primarily on the detailed risk assessment performed for the PSA. Each MATC was then compared to medium-specific data for areas downstream of the PSA to Long Island Sound. Areas with MATC exceedances, indicating potential risk, were plotted on maps of the river. The methods used for each of the seven representative groups and the results of the analyses are discussed in the following sections.

Benthic Invertebrates—For benthic invertebrates, the medium of interest was river sediment. A sediment MATC of 3 mg/kg tPCBs was used as a measure of the potential for adverse affects to benthic invertebrates downstream of the PSA. The MATC of 3 mg/kg tPCBs was compared to recent surficial sediment data downstream of the PSA, and the results were plotted to indicate samples above and below the MATC (see Figure ES-8).

In general, potential risks to benthic invertebrates occur in limited areas downstream of Woods Pond to Rising Pond. These areas are depositional and tend to have higher concentrations of tPCBs. Below Rising Pond, sediment does not contain concentrations of tPCBs that represent a potential risk to benthic invertebrates. The latter conclusion is supported by comparison of field-collected invertebrate tissue residue data (West Cornwall, CT) to literature-derived PCB tissue thresholds.



NOTES:
Risk to benthic invertebrates is based on a maximum acceptable threshold concentration (MATC) of 3.0 mg/kg total PCB (tPCB) (dry weight) in surficial sediments (0-6 inches).

LEGEND:

BENTHIC INVERTEBRATE RISK
- < 3 mg/kg
- >= 3 mg/kg

- Town
- Reach Break
- Roads
- Housatonic River Basin Hydrology
- State Boundary

2 0 2 4 6 8 Miles

Ecological Risk Assessment
GE/Housatonic River Site
Rest of River

FIGURE ES-8
ASSESSMENT OF RISK TO BENTHIC INVERTEBRATES EXPOSED TO tPCBs DOWNSTREAM OF THE PSA

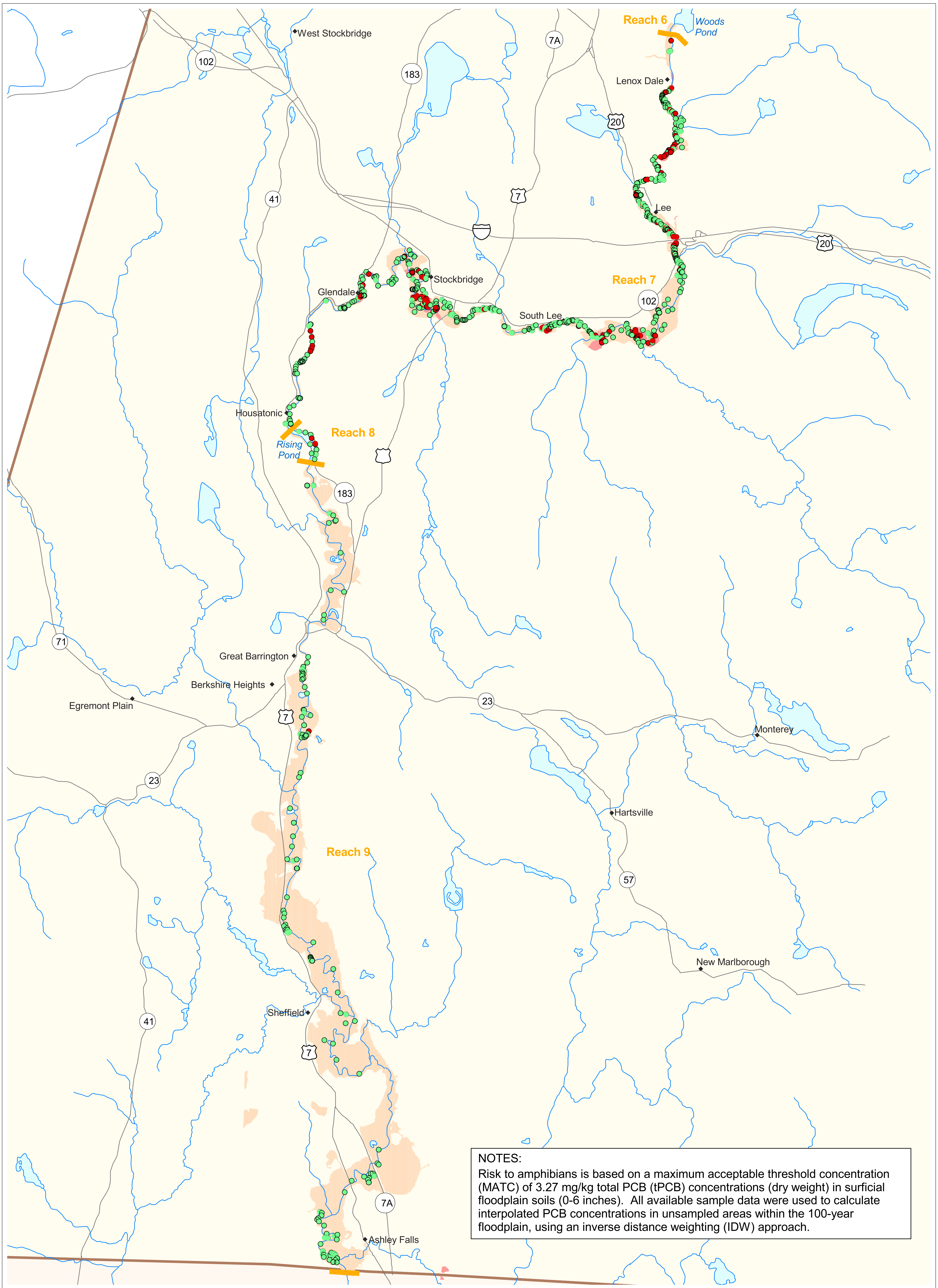
Amphibians—A floodplain soil MATC of 3.27 mg/kg tPCB (dry weight) was derived from the amphibian risk assessment conducted for the PSA. The MATC of 3.27 mg/kg tPCBs was compared to recent surficial sediment data downstream of the PSA, and the results were plotted (Figure ES-9) to indicate samples above and below the MATC.

Several large areas of the floodplain may pose risk to amphibians between Woods Pond and Rising Pond, with only small isolated areas of potential risk downstream of Rising Pond (Figure ES-9).

Downstream of the Massachusetts/Connecticut state line, the risk mapping for amphibians was conducted for sediment only because the concentrations in floodplain soil had decreased below the MATC throughout Reach 9 upstream of the border, and the extent of the floodplain is limited in Connecticut (Figure ES-10).

Warmwater Fish—As was done for the PSA, risk to fish downstream was evaluated based on concentrations of tPCBs in fish tissue. An MATC of 55 mg/kg tPCB in tissue (whole body, wet weight) developed for the PSA based on site-specific effects to warmwater fish was applied to areas downstream of the PSA using the available (e.g., bass, perch, sunfish) tissue data for warmwater species. Each downstream reach (Reaches 7 through 16) was evaluated as a unit, and the mean adult fish tissue concentration in each reach was compared with the MATC to determine potential risk (Figure ES-11). No risks were indicated in any of the reaches below the PSA.

Trout—Trout were evaluated separately from warmwater fish species because of differences in habitat requirements and differences in the sensitivity of some trout species to tPCBs documented in the literature. Trout also tend to have higher tPCB concentrations because of their higher lipid content. The strain of rainbow trout (Fish Lake strain) used in the site-specific toxicity tests is less sensitive than other strains used widely in toxicity testing. Furthermore, there are other trout species found downstream of the PSA (e.g., brown trout) for which sensitivity has not been assessed. Given that some trout species have been documented to have greater sensitivity of PCBs and dioxins, relative to the warmwater species considered in the development of the 55 mg/kg tPCB warmwater MATC, a factor of 4 was applied in recognition



NOTES:
Risk to amphibians is based on a maximum acceptable threshold concentration (MATC) of 3.27 mg/kg total PCB (tPCB) concentrations (dry weight) in surficial floodplain soils (0-6 inches). All available sample data were used to calculate interpolated PCB concentrations in unsampled areas within the 100-year floodplain, using an inverse distance weighting (IDW) approach.

LEGEND:

AMPHIBIAN RISK

SEDIMENT SAMPLES
- < 3.27 mg/kg
- >= 3.27 mg/kg

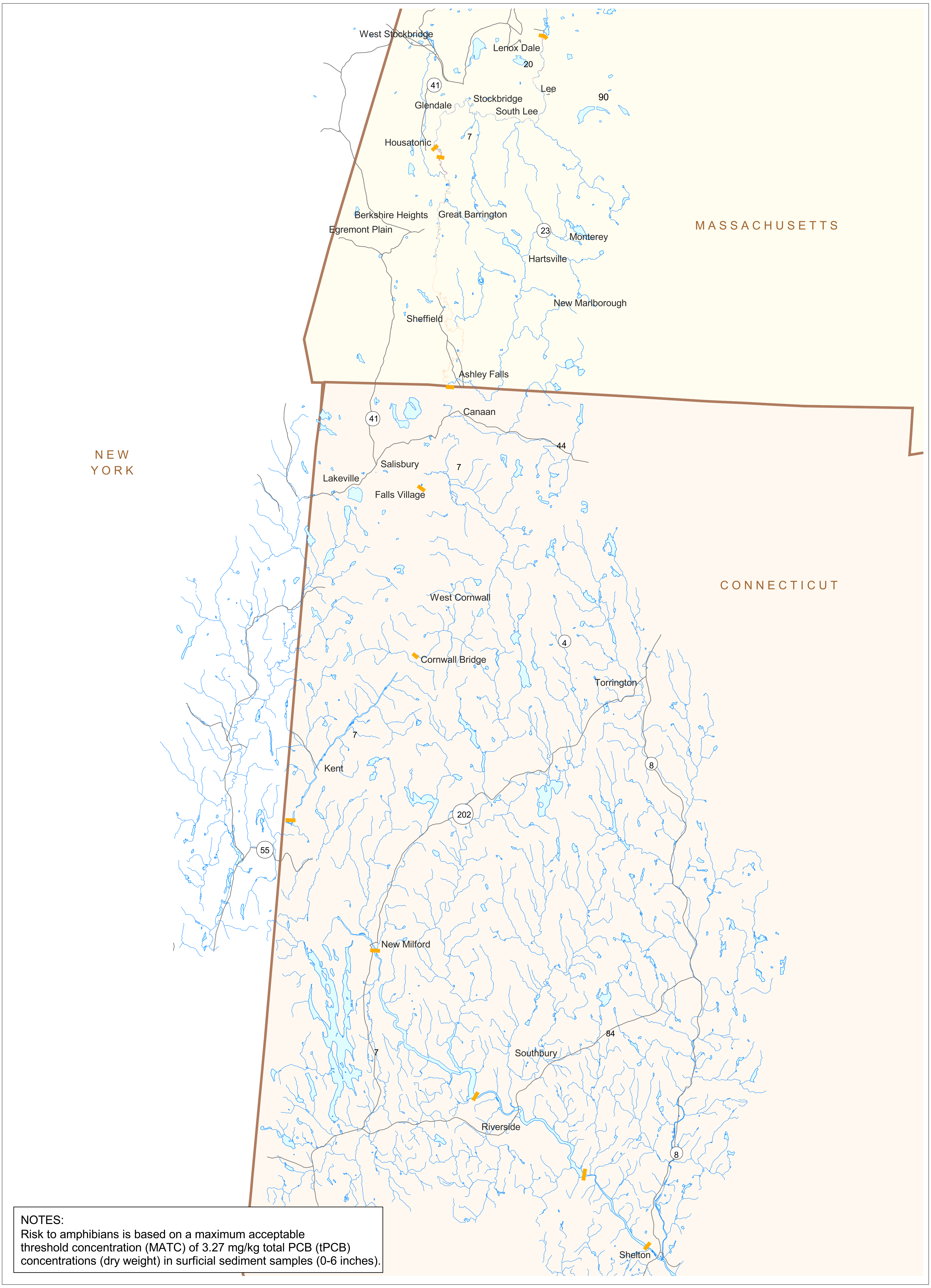
FLOODPLAIN SOIL
- < 3.27 mg/kg
- >= 3.27 mg/kg

- Town
- Reach Breaks
- Roads
- Housatonic River Basin Hydrology
- State Boundary

0.5 0 0.5 1 1.5 2 Miles

Ecological Risk Assessment
GE/Housatonic River Site
Rest of River

FIGURE ES-9
ASSESSMENT OF RISK TO AMPHIBIANS EXPOSED TO tPCBs DOWNSTREAM OF THE PSA IN MASSACHUSETTS



NOTES:
Risk to amphibians is based on a maximum acceptable threshold concentration (MATC) of 3.27 mg/kg total PCB (tPCB) concentrations (dry weight) in surficial sediment samples (0-6 inches).

LEGEND:

Town

Reach Breaks

AMPHIBIAN RISK

< 3.27 mg/kg

>= 3.27 mg/kg

Roads

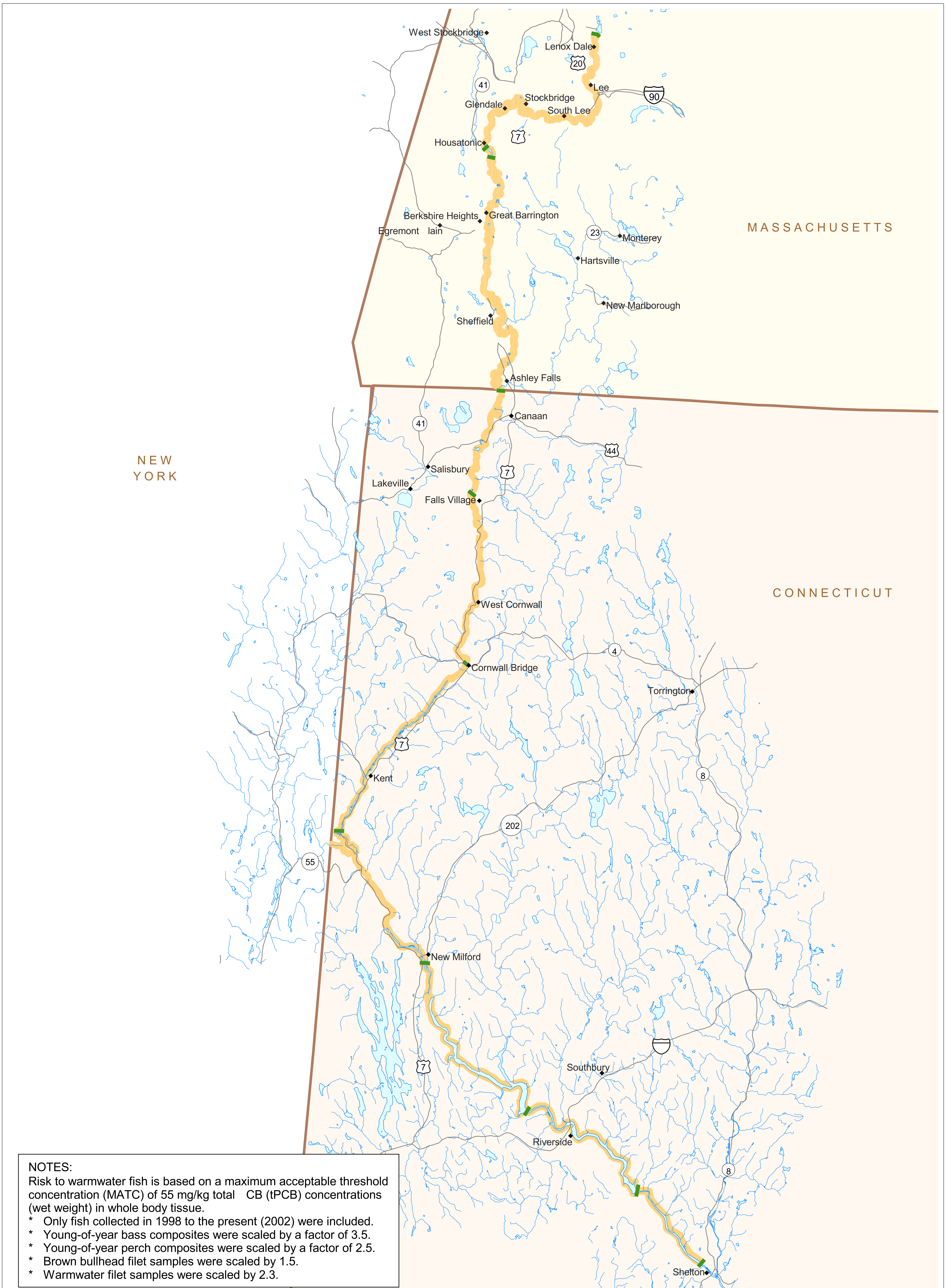
Housatonic River Basin Hydrology

State Boundary

2 0 2 4 6 8 Miles

Ecological Risk Assessment
GE/Housatonic River Site
Rest of River

FIGURE ES-10
ASSESSMENT OF RISK TO AMPHIBIANS IN SEDIMENT EXPOSED TO tPCBs DOWNSTREAM OF THE PSA IN CONNECTICUT



NOTES:
Risk to warmwater fish is based on a maximum acceptable threshold concentration (MATC) of 55 mg/kg total CB (tPCB) concentrations (wet weight) in whole body tissue.
* Only fish collected in 1998 to the present (2002) were included.
* Young-of-year bass composites were scaled by a factor of 3.5.
* Young-of-year perch composites were scaled by a factor of 2.5.
* Brown bullhead filet samples were scaled by 1.5.
* Warmwater filet samples were scaled by 2.3.

LEGEND:

WARMWATER FISH RISK
- < 55 mg/kg
- >= 55 mg/kg

- Town
- Reach Break
- Roads
- Housatonic River Basin Hydrology
- State Boundary

2 0 2 4 6 8 Miles

Ecological Risk Assessment
GE/Housatonic River Site
Rest of River

FIGURE ES-11
ASSESSMENT OF RISK TO WARM WATER FISH EXPOSED TO tPCBs DOWNSTREAM OF THE PSA

of these potential interspecies differences. Therefore, a tissue MATC of 14 mg/kg tPCBs (whole body, wet weight) was derived for trout.

The results of this evaluation indicate that trout are potentially at risk in Reach 7, but not in reaches with suitable habitat farther downstream (Figure ES-12). This assessment has high uncertainty due to the number of extrapolations required and the low magnitude of exceedance of the MATC value. Potential risk to trout was not evaluated downstream of Reach 12 due to lack of suitable trout habitat.

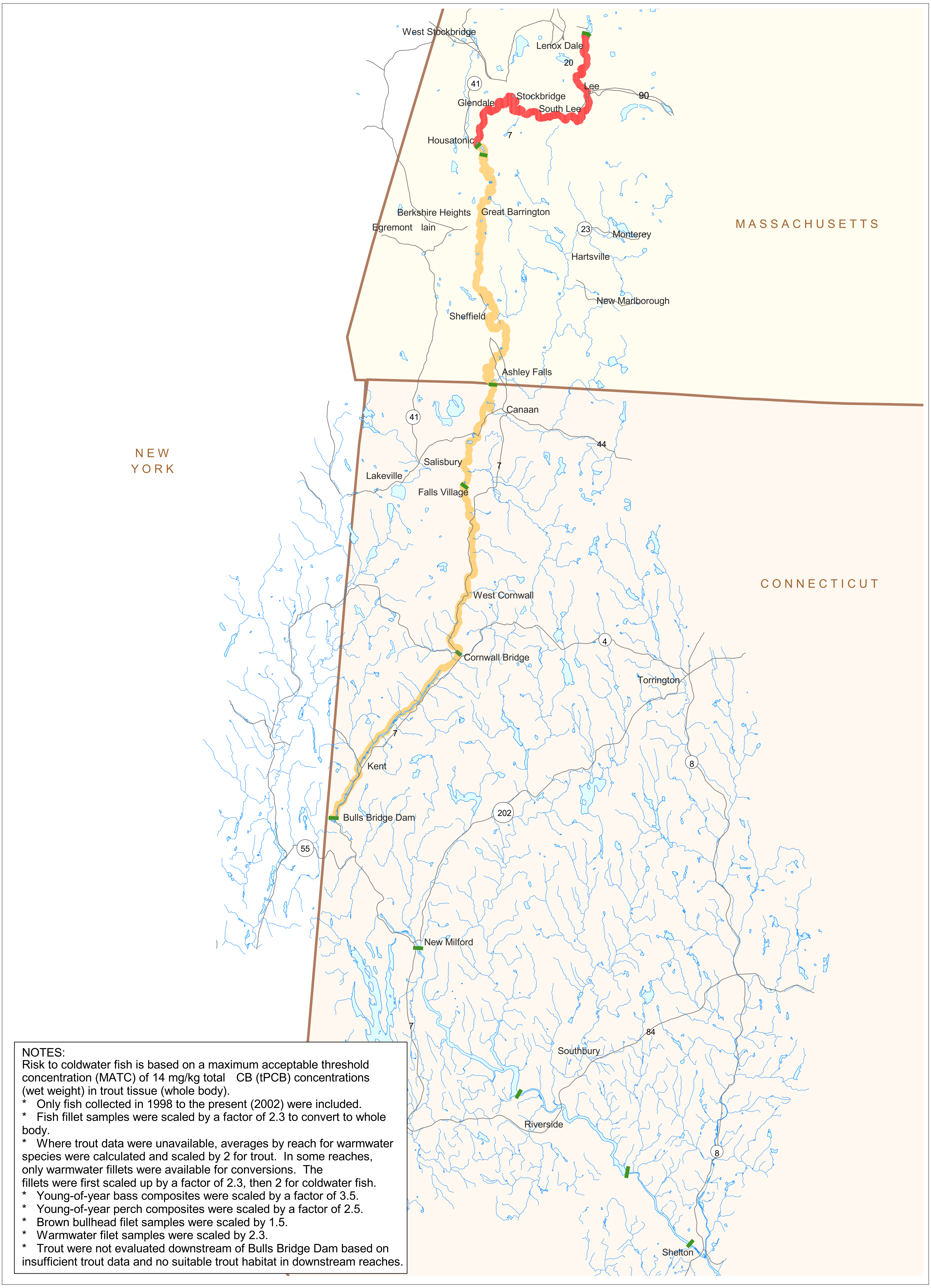
Mink—An MATC for mink of 0.984 mg/kg tPCBs in fish (whole body, wet weight) is equal to the LC_{20} for survival of mink kits from 0 to 6 weeks developed in a site-specific study of the toxicity of a diet containing Housatonic River fish to mink. Mean fish concentrations were calculated for each river reach downstream of the PSA using available whole body fish tissue data from samples of fish with an overall body length between 7 and 20 cm, corresponding to the size commonly preyed on by mink. Potential risk to mink due to consumption of contaminated fish occurs from the Woods Pond Dam downstream to the end of Reach 15 (Figure ES-13).

River Otter—The mink MATC of 0.984 mg/kg tPCB in fish (whole body, wet weight) was also used for river otter. Mean fish concentrations were calculated for river reaches downstream of the PSA using available whole body fish tissue data from fish with an overall body length between 5 and 50 cm, corresponding to the size commonly preyed on by otter.

Potential risk to otter due to consumption of contaminated fish occurs from the Woods Pond Dam downstream to Reach 15 (Figure ES-14).

Bald Eagle—An MATC of 30.4 mg/kg tPCBs (whole body fish tissue, wet weight) was developed for wintering bald eagles, assuming that the eagle diet was composed of 78% fish, and that the remainder of the diet included non-aquatic species that, for the purpose of this analysis, were assumed to be uncontaminated. Potential risk to nesting bald eagles was evaluated using methods similar to those discussed above for mink (Figure ES-15).

A more in-depth analysis was performed for Reaches 14 and 15 where bald eagles have nested. On average, bald eagles consume a summer diet consisting of 78.2% fish, 16.3% birds, and 5% mammals.



NOTES:
Risk to coldwater fish is based on a maximum acceptable threshold concentration (MATC) of 14 mg/kg total CB (tPCB) concentrations (wet weight) in trout tissue (whole body).

* Only fish collected in 1998 to the present (2002) were included.
* Fish fillet samples were scaled by a factor of 2.3 to convert to whole body.
* Where trout data were unavailable, averages by reach for warmwater species were calculated and scaled by 2 for trout. In some reaches, only warmwater fillets were available for conversions. The fillets were first scaled up by a factor of 2.3, then 2 for coldwater fish.
* Young-of-year bass composites were scaled by a factor of 3.5.
* Young-of-year perch composites were scaled by a factor of 2.5.
* Brown bullhead filet samples were scaled by 1.5.
* Warmwater filet samples were scaled by 2.3.
* Trout were not evaluated downstream of Bulls Bridge Dam based on insufficient trout data and no suitable trout habitat in downstream reaches.

LEGEND:

COLD WATER FISH RISK
< 14 mg/kg
>= 14 mg/kg

Town
Reach Break
Roads
Housatonic River Basin Hydrology
State Boundary

2 0 2 4 6 8 Miles

Ecological Risk Assessment
GE/Housatonic River Site
Rest of River

FIGURE ES-12
ASSESSMENT OF RISK TO TROUT EXPOSED TO tPCBs DOWNSTREAM OF THE PSA

NOTES:
Risk to mink is based on a maximum acceptable threshold concentration (MATC) of 0.984 mg/kg total CB (tPCB) concentration (wet weight) in whole body fish tissue 7-20 cm in length.

* Only fish collected in 1998 to the present (2000) were included.
* Young-of-year bass composites were scaled by a factor of 3.5.
* Young-of-year perch composites were scaled by a factor of 2.5.
* Trout filet samples were scaled by 1.47.
* Brown bullhead filet samples were scaled by 1.5.
* Warmwater filet samples were scaled by 2.3.

LEGEND:

MINK RISK
- < 0.984 mg/kg
- >= 0.984 mg/kg

- Town
- Reach Break
- Roads
- Housatonic River Basin Hydrology
- State Boundary

2 0 2 4 6 8 Miles

Ecological Risk Assessment
GE/Housatonic River Site
Rest of River

FIGURE ES-13
ASSESSMENT OF RISK TO MINK EXPOSED TO tPCBs DOWNSTREAM OF THE PSA

NOTES:
Risk to otter is based on a maximum acceptable threshold concentration (MATC) of 0.984 mg/kg total CB (tPCB) concentration (wet weight) in whole body fish tissue 5-50 cm in length.

* Only fish collected in 1998 to the present (2000) were included.
* Young-of-year bass composites were scaled by a factor of 3.5.
* Young-of-year perch composites were scaled by a factor of 2.5.
* Trout filet samples were scaled by 1.47.
* Brown bullhead filet samples were scaled by 1.5.
* Warmwater filet samples were scaled by 2.3.

LEGEND:

OTTER RISK
- < 0.984 mg/kg
- >= 0.984 mg/kg

- Town
- Reach Break
- Roads
- Housatonic River Basin Hydrology
- State Boundary

2 0 2 4 6 8 Miles

Ecological Risk Assessment
GE/Housatonic River Site
Rest of River

FIGURE ES-14
ASSESSMENT OF RISK TO OTTER EXPOSED TO tPCBs DOWNSTREAM OF THE PSA

NOTES:
Risk to eagles is based on a maximum acceptable threshold concentration (MATC) of 30.4 mg/kg total CB (tPCB) concentration (wet weight) in whole body fish tissue greater than or equal to 12 cm.
* Only fish collected in 1998 to the present (2000) were included.
* Young-of-year bass composites were scaled by a factor of 3.5.
* Young-of-year perch composites were scaled by a factor of 2.5.

LEGEND:

EAGLE RISK
- < 30.4 mg/kg
- >= 30.4 mg/kg

- Town
- Reach Break
- Roads
- Housatonic River Basin Hydrology
- State Boundary

2 0 2 4 6 8 Miles

Ecological Risk Assessment
GE/Housatonic River Site
Rest of River

FIGURE ES-15
ASSESSMENT OF RISK TO BALD EAGLE EXPOSED TO tPCBs DOWNSTREAM OF THE PSA

The results of the evaluation indicate that potential risks to bald eagles from consuming contaminated fish in areas downstream of the PSA are restricted to Reach 8, corresponding to Rising Pond. However, Rising Pond is smaller than the typical eagle foraging area. It is likely that eagles nesting at Rising Pond would be exposed through a foraging area including but greater than Rising Pond. The concentrations of COCs in fish tissue in the adjacent subreaches of the river are not known, but are expected to be elevated.

ES.6 BROADER IMPLICATIONS

The WOE assessments indicate that COCs in the PSA of the Housatonic River, particularly tPCBs, are causing risks to many of the species selected to represent the assessment endpoints. Risks from COCs, however, may potentially extend beyond adverse effects to survival, growth, and reproduction of representative species. The Housatonic River ERA also explores the broader implications of the risks of COCs to representative species, including extension of the ecological risk assessment to species that occur in the Housatonic River watershed, but that had not been considered explicitly in the quantitative ecological risk assessments, and additional ecological implications.

ES.6.1 Implications for Other Species in the Primary Study Area

The major factors that influence exposure to tPCBs and TEQ and that were considered in the analysis include:

- Dietary composition.
- Foraging behavior and home range.
- Size, metabolism, and life history characteristics.
- Sensitivity to COCs.

The ERA compares these factors between the representative species and other species in their foraging groups. The comparison highlights similarities and differences, and their potential to influence exposure and hence risks from tPCBs and TEQ.

Benthic Invertebrates—The benthic invertebrate ERA included the entire benthic community; benthic community composition analysis was a measurement endpoint considered in the WOE assessment. Both the status of sensitive taxa and community composition are considered indicators of overall health and productivity of the benthic community. Thus, there is no need to extrapolate the findings of the benthic invertebrate assessment described previously to other benthic invertebrate species in the PSA.

Amphibians—Certain amphibian species that were not studied may be more susceptible to the effects of tPCBs because of their life history characteristics. For example, blue-spotted (*Ambystoma laterale*) and spotted salamanders (*Ambystoma maculatum*) have a lifestage as aquatic carnivorous, bottom-dwelling larvae. Thus, they could potentially bioaccumulate PCBs more quickly than herbivorous amphibians. Salamanders appeared in lower numbers in vernal pools with high sediment tPCB concentrations. Several salamander species occur in contaminated habitat in the PSA, including the spotted salamander, the Jefferson salamander (*Ambystoma jeffersonianum*, formerly considered a variety of blue-spotted salamander), and the four-toed salamander (*Hemidactylium scutatum*), the latter two of which are Species of Special Concern.

Fish—There is evidence in the literature that salmonid species may have a higher sensitivity to the effects of PCBs and other dioxin-like COPCs. The use of rainbow trout in the site-specific toxicity testing program, combined with effects data from the literature, provides a high degree of confidence that the ERA included an evaluation of fish species with equal or greater sensitivities than the representative species listed above. However, the procedure used to establish MATCs for fish in the PSA placed a low weight on studies conducted with fish species known to be highly sensitive (e.g., lake trout), to avoid an overly conservative assessment. Risks to coldwater fisheries (e.g., trout) downstream of the PSA were explicitly evaluated using benchmarks developed for salmonids; the uncertainty in these downstream risk estimates is high due to the number of extrapolations required. The risk of COCs to the occasional salmonid occurring within the PSA is considered to be intermediate. The PSA, however, is considered to be a warmwater fishery, and thus salmonid abundance is expected to be low in this portion of the river, even in the absence of chemical stressors.

Insectivorous Birds—The WOE assessment indicated that exposure of insectivorous birds, such as tree swallows and American robins, to tPCBs and TEQ is high but unlikely to lead to adverse reproductive effects. Confidence in this conclusion, however, is not high because the two available lines of evidence for both species did not produce concordant results. There are a number of insectivorous birds with similar feeding habits as tree swallows in the PSA, and these are generally believed to be at the same low to moderate risk as tree swallows, although some species with higher food intake could be at higher risk. Modeling of exposure and effects indicated that wood ducks are at high risk from exposure to tPCBs and TEQ. Several other insectivorous waterfowl species may also be at high risk in the PSA (e.g., hooded merganser, American black duck).

Compared to American robins, eastern bluebirds and eastern towhees are expected to experience lower to similar levels of risk from exposure to tPCBs and TEQ. The level of risk for the hermit thrush, northern mockingbird, veery, and wood thrush is expected to be similar to the level of risk for American robins. With the exception of earthworms in the robin diet, the dietary preferences of these birds are similar to the American robin. The absence of earthworms, a major dietary source of contaminants, will decrease their exposure to tPCBs and TEQ. However, their smaller body sizes result in higher food intake rates and, hence, greater exposure to tPCBs and TEQ through diet compared to American robins.

Piscivorous Birds—The WOE assessment indicates that risks of tPCBs and TEQ to belted kingfisher are low; however, risks of these COCs to osprey are high and could lead to adverse reproductive effects.

The belted kingfisher and osprey were chosen to represent piscivorous birds inhabiting the Housatonic River area. Belted kingfisher and osprey are common piscivorous birds in the PSA. Great blue herons are also found in the PSA, and are discussed in Appendix K with other piscivorous birds (e.g., American bittern).

Piscivorous Mammals—Mink and river otter, the representative species for piscivorous mammals, are the only piscivorous mammals commonly found in the watershed of the Housatonic River.

Omnivorous and Carnivorous Mammals—The WOE assessment indicates that omnivorous and carnivorous mammals, such as red fox and short-tailed shrew, are at risk in the PSA as a result of exposure to tPCBs and TEQ. Masked shrews are expected to experience a level of risk similar to northern short-tailed shrews and smoky shrews are expected to be at higher risk, based on their metabolic rates relative to short-tailed shrews. All three have similar foraging behaviors and ranges.

Coyotes have a larger body size and foraging range which decreases their exposure to tPCBs and TEQ. Considering these characteristics, coyotes are expected to experience lower risks from exposure to tPCBs and TEQ than red fox. Gray and red foxes are expected to experience similar risks from exposure to tPCBs and TEQ. Gray fox have a larger foraging range than red fox and that may decrease their exposure to tPCBs and TEQ. Gray fox, however, have a greater reliance on animal matter and therefore greater exposure to tPCBs and TEQ.

Fishers, long-tailed weasels, and short-tailed weasels are expected to experience similar to higher levels of risk from exposure to tPCBs and TEQ compared to the red fox due to greater consumption of animal matter and/or higher metabolic rate.

Threatened and Endangered Species—The bald eagle, American bittern, and small-footed myotis were chosen to represent T&E species that are likely to be highly exposed to COCs in the Housatonic River PSA. Other T&E species that occur in the area include one mussel (triangle floater); three dragonflies (riffle snaketail, zebra clubtail, and arrow clubtail); a turtle (wood turtle); three salamanders (Jefferson salamander, four-toed salamander, and northern spring salamander); three hawks (northern harrier, sharp-shinned hawk, and Cooper's hawk); two warblers (northern parula and blackpoll warbler); a wading bird (common moorhen); and a shrew (northern water shrew). Some of these species were qualitatively assessed in other appendices and compared to other, more appropriate, assessment endpoints (e.g., amphibians for salamanders).

The level of risk for soras[1] is expected to be lower than for American bitterns because of greater consumption of vegetable matter. Great blue herons and king rails are expected to experience a

[1] Several of the species included in this section (i.e., sora, great blue heron, green heron, Virginia rail, northern myotis, little brown bat) are not threatened and endangered species either federally or in Massachusetts and Connecticut (Appendix A). They are included in the discussion of T&E species because they are taxonomically and ecologically similar to either American bittern or to small-footed myotis.

similar level of risk as American bitterns because of a combination of size differences and some differences in dietary preferences. The least bittern, green heron, Virginia rail, and pied-billed grebe are expected to experience higher levels of risk compared to the American bittern. The foraging and life history characteristics of these birds are similar to the American bittern. However, these birds are much smaller than the American bittern. Their smaller body sizes result in a higher metabolism and greater exposure to tPCBs and TEQ.

The Indiana bat, northern myotis, and little brown bat are expected to have similar levels of risk as the small-footed myotis. These species belong to the same genus (*Myotis*) and have similar foraging behaviors and life histories.

ES.7 SOURCES OF UNCERTAINTY

The assessment of risks of COCs to aquatic and wildlife species in the Housatonic River contains uncertainties. Each source of uncertainty can influence the estimates of risk; therefore, it is important to describe and, when possible, specify the magnitude and direction of such uncertainties. In this section, the most significant sources of uncertainty commonly encountered throughout the ERA are described. The sources of uncertainty are grouped by phase of the ERA (i.e., problem formulation, exposure assessment, effects assessment, risk assessment).

The problem formulation is intended to define the linkages between stressors, potential exposure, and predicted effects on ecological receptors. As such, the conceptual model provides the scientific basis for selecting assessment and measurement endpoints to support the risk assessment process. Potential uncertainties arise from lack of knowledge regarding ecosystem functions, failure to adequately address spatial and temporal variability in the evaluations of sources, fate and effects, omission of stressors, and overlooking secondary effects (EPA 1998). The types of uncertainties associated with the conceptual model that links contaminant sources to effects include those associated with the identification of COCs, environmental fate and transport of COCs, exposure pathways, receptors at risk, and ecological effects. Of these, the identification of exposure pathways probably represents the primary source of uncertainty in the conceptual model. The detailed ecological characterization performed at this site has greatly reduced the uncertainties associated with problem formulation.

The exposure assessment is intended to describe the actual or potential co-occurrence of stressors with receptors. As such, the exposure assessment identifies the exposure pathways and the intensity and extent of contact with stressors for each receptor or group of receptors at risk. The exposure models for wildlife were energetics-based models requiring information on body weight, free living metabolic rate, proportions of food items in the diets, and the concentrations of COCs in these food items. Each of these variables has associated uncertainties, most of which were propagated through the exposure models. The effects assessment is intended to describe the effects caused by stressors, link them to the assessment endpoints, and evaluate how effects change with fluctuations in the levels (i.e., concentrations or doses) of the various stressors. In this assessment, the effects of tPCBs and other COCs to representative species were assessed. There are several sources of uncertainty in the assessment of effects, including extrapolation errors and a limited number of toxicity studies conducted with the representative species.

For benthic invertebrates and amphibians, the effects benchmarks derived from the literature had a high degree of uncertainty because of the need to extrapolate across sites and species. The site-specific fish toxicity studies indicated variations in the dose-response relationships observed across species, reaches, and treatments and introduced uncertainty into the development of effects thresholds. The methodology used in site-specific fish studies was developed recently, and there are potential uncertainties inherent to extrapolating these laboratory-based results to Housatonic River fish. Similarly, the extrapolation of concentrations of tPCBs in egg to whole body concentrations has a degree of associated uncertainty.

The greatest potential source of uncertainty for the fish and wildlife effects assessments, however, was associated with the lack of toxicity studies involving the representative species.

A WOE procedure was used to assess risks of tPCBs and TEQ to the assessment endpoints in the Housatonic River PSA. The analysis follows the methodology proposed by the Massachusetts Weight-of-Evidence Workgroup (Menzie et al. 1996; see Section 2.9 for details).

In general, the WOE approach is an inclusive process whereby multiple lines of evidence are considered prior to determining risk. For the wildlife risk assessments, these lines of evidence included the exposure and effects modeling results and, in some cases, field survey results, and/or in situ or whole media toxicity test results. For the fish and benthic invertebrate risk assessments, available lines of evidence included field survey results (e.g., community evaluation for benthos), site-specific toxicity tests, and comparison of tissue and sediment concentrations to benchmarks (both from the literature and site-specific benchmarks). The largest source of uncertainty in the WOE process was the development of conclusions based upon only one or two lines of evidence.

ES.8 SUMMARY AND CONCLUSIONS

The weight-of-evidence assessments indicated that aquatic life and wildlife in the Housatonic River PSA are experiencing intermediate or high risks as a result of exposure to tPCBs and other COCs. Confidence in this conclusion is high for benthic invertebrates, amphibians, and piscivorous mammals because multiple lines of evidence gave concordant results.

The risks of tPCBs and other COCs likely extend beyond the representative species considered in the quantitative risk assessments described herein. Qualitative risk assessments indicated that many other species in the PSA are potentially at risk. Further, there are likely indirect effects (e.g., changes in predator-prey relationships, changes in metapopulation dynamics) occurring inside and outside the PSA as a result of the direct impacts caused by tPCBs and other COCs.

An assessment of risk downstream of the PSA indicated that tPCBs could potentially be causing adverse effects to benthic organisms in depositional areas as far as Reach 8, amphibians in floodplain areas as far as Reach 9, trout in Reach 7, mink as far as Reach 15, and river otter as far as Reach 15, and bald eagle in Reach 8. However, the magnitude of risks in these areas is lower than in the PSA.

1. INTRODUCTION

> ***Purposes of ERA***
>
> This ecological risk assessment (ERA) characterizes and quantifies the current and potential risks to biota exposed to contaminants in the Housatonic River downstream of the confluence of the East and West Branches. Specific purposes of the ERA include:
>
> 1. To consider the fate and transport of PCBs and other contaminants of potential concern (COPCs) to ecological receptors in the river and associated floodplain.
> 2. To identify the potential routes of exposure and toxicological effects of the COPCs for these receptors.
> 3. To identify assessment endpoints representative of species potentially at risk.
> 4. To determine the risk to the assessment endpoints selected.

1.1 OVERVIEW

The purpose of this ecological risk assessment (ERA) is to characterize and quantify the current and potential risks to biota exposed to contaminants of potential concern (COPCs) in the Housatonic River downstream of the confluence of the East and West Branches, focusing on polychlorinated biphenyls (PCBs) and other COPCs originating from the General Electric Company (GE) facility in Pittsfield, MA. This ERA considers the fate and transport of PCBs and other COPCs to ecological receptors in the river and associated floodplain, identifies assessment endpoints that are representative of species potentially at risk, and identifies the potential routes of exposure and toxicological effects of the COPCs for these receptors.

This information is synthesized, through a weight-of-evidence approach, into a discussion of the nature and magnitude of the risks for the assessment endpoints, and the uncertainties associated with the characterization of these risks.

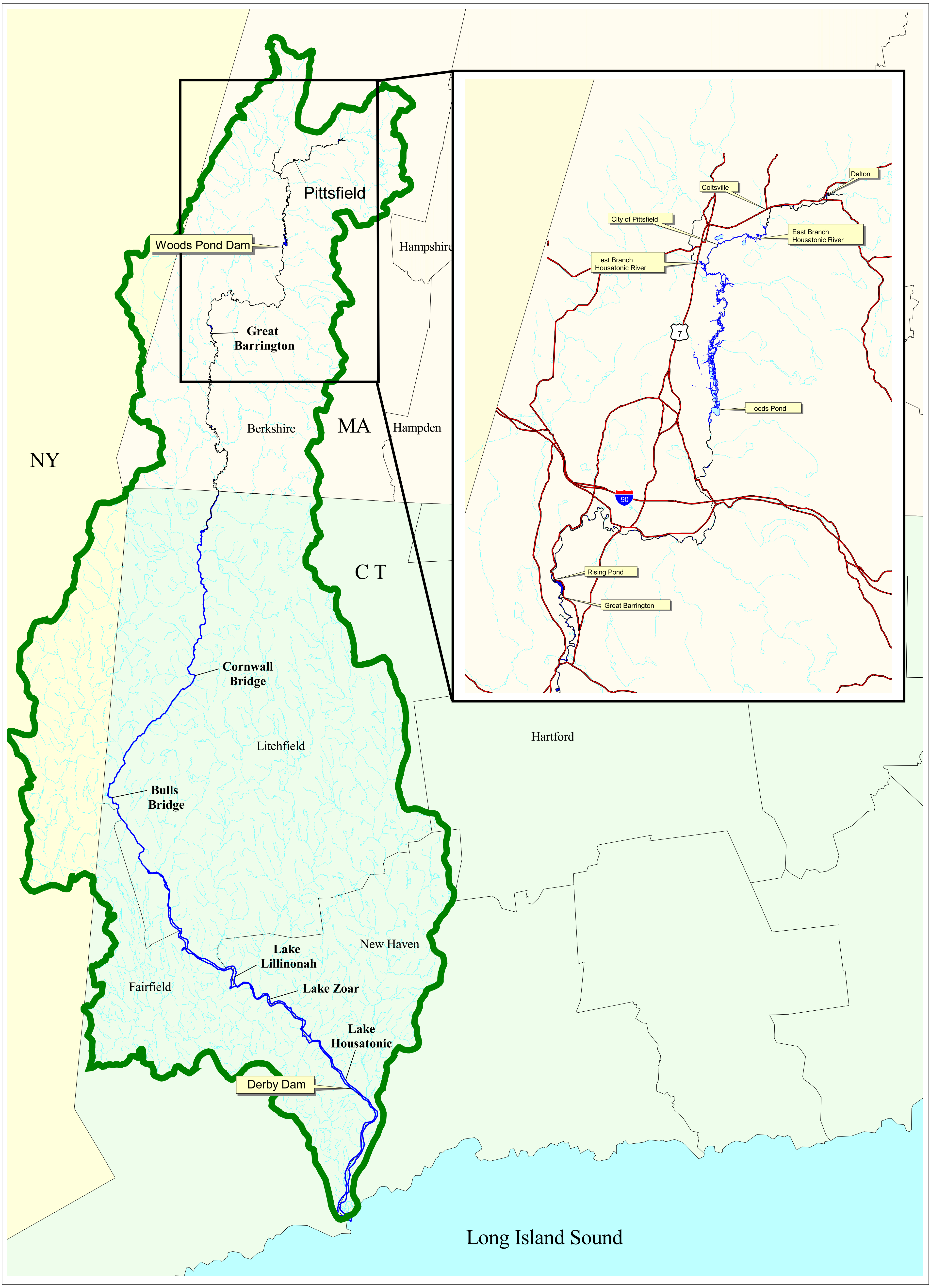
Multiple lines of evidence for each assessment endpoint are evaluated, including where applicable or available:

- Field surveys/studies.
- Toxicity tests.
- Comparison of effects in the literature to a site-specific exposure model.

The Housatonic River flows from east of Pittsfield, MA, to Long Island Sound and drains an area of approximately 1,950 square miles (500,000 hectares) in Massachusetts, New York, and Connecticut (Figure 1.1-1). The Housatonic River, its sediment, and associated floodplain have been contaminated with polychlorinated biphenyls (PCBs) and other hazardous substances released from the General Electric Company (GE) facility located in Pittsfield, MA. The entire site, known as the General Electric/Housatonic River Site, consists of the 254-acre (103-hectare) GE manufacturing facility; the Housatonic River and associated riverbanks and floodplain from Pittsfield, MA, to Long Island Sound; former river oxbows that have been filled; neighboring commercial properties; Allendale School; Silver Lake; and other properties or areas that have become contaminated as a result of GE's facility operations.

Because of its size and complexity, the GE/Housatonic River Site has been divided into several areas for investigation and cleanup. The "Rest of River" is the portion of the river from the confluence of the East and West Branches of the Housatonic River (the confluence) to the Massachusetts border with Connecticut, a distance of approximately 54 miles (87 km), and beyond into Connecticut to Long Island Sound. The total distance from the confluence to Long Island Sound is approximately 139 miles (224 km). In addition to the river itself, the Rest of River includes the associated riverbank and floodplain. The Rest of River is further defined in the Consent Decree entered with the U.S. District Court, Western Region, Massachusetts, in October 2000. The Rest of River includes areas of the Housatonic River and its sediment and floodplain (except Actual/Potential Lawns), in which contaminants originating from the GE facility are located. The lateral extent of the area under investigation includes the floodplain extending to the 1-ppm total PCB (tPCB) isopleth, which is approximately equivalent to the 10-year floodplain.

The ERA focuses on the portion of the river from the confluence of the East and West Branches 2 miles (3 km) downstream of the GE facility, to Woods Pond Dam, a distance of 10.7 river miles (17.3 km). This area is referred to as the Primary Study Area (PSA) in the Supplemental Investigation (Figure 1.1-2), and is where much of the PCB contamination was found in previous studies. The river (which includes free-flowing and impounded sections) and the floodplain downstream of the PSA to the Derby Dam in Connecticut are also considered in the ERA.



LEGEND:

Housatonic River

Housatonic River Basin

rimary Road

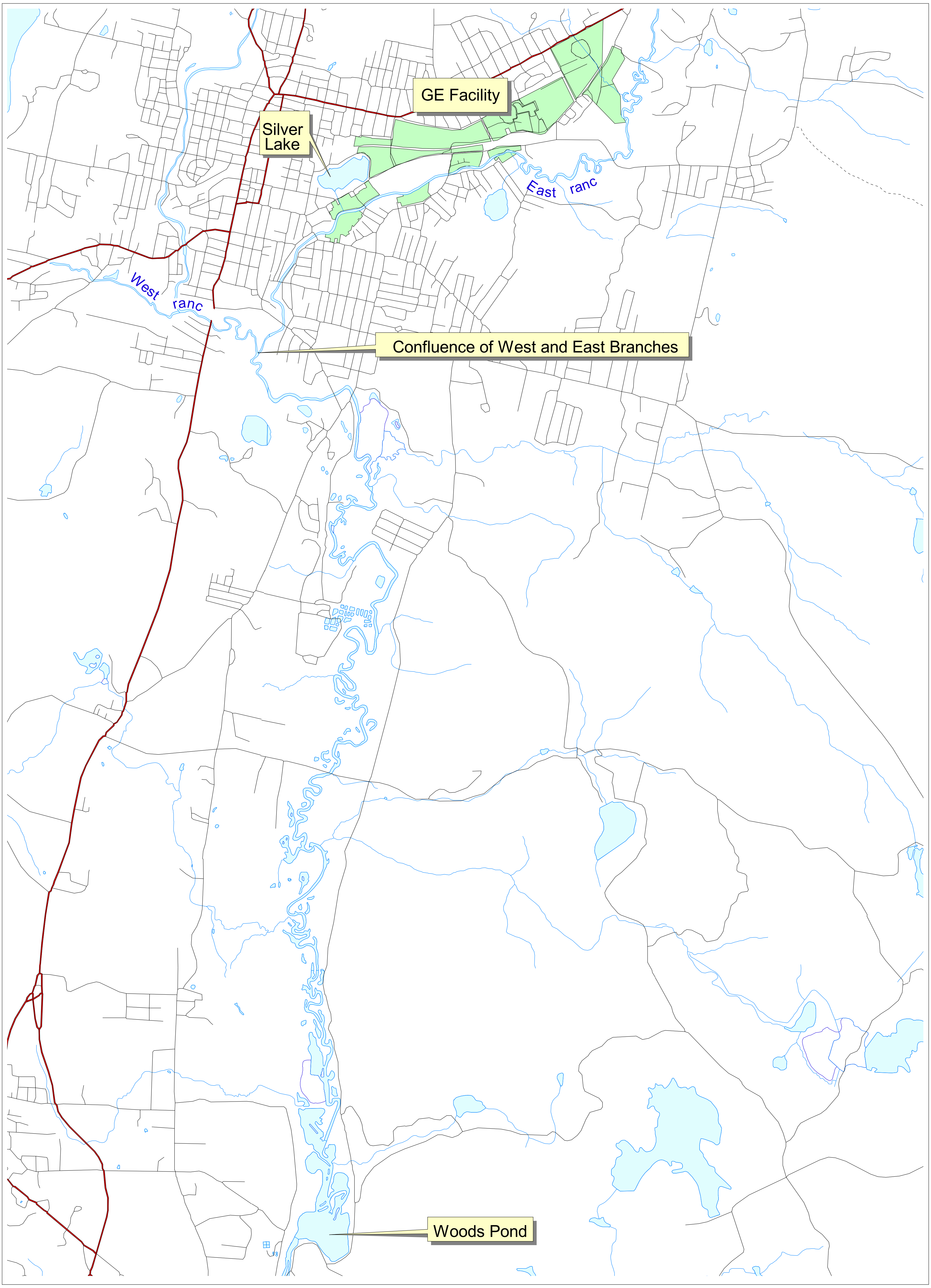
N W E S

4 0 4 8 Miles

5 0 5 10 15 Kilometers

Ecological Risk Assessment
GE/Housatonic River Site
Rest of River

FIGURE 1.1-1
HOUSATONIC RIVER



LEGEND:

- Roads
- Housatonic River
- GE Facility

N W E S

0.1 0 0.1 0.2 Miles

0.2 0 0.2 Kilometers

Ecological Risk Assessment
GE/Housatonic River Site
Rest of River

FIGURE 1.1-2
PRIMARY STUDY AREA (PSA)

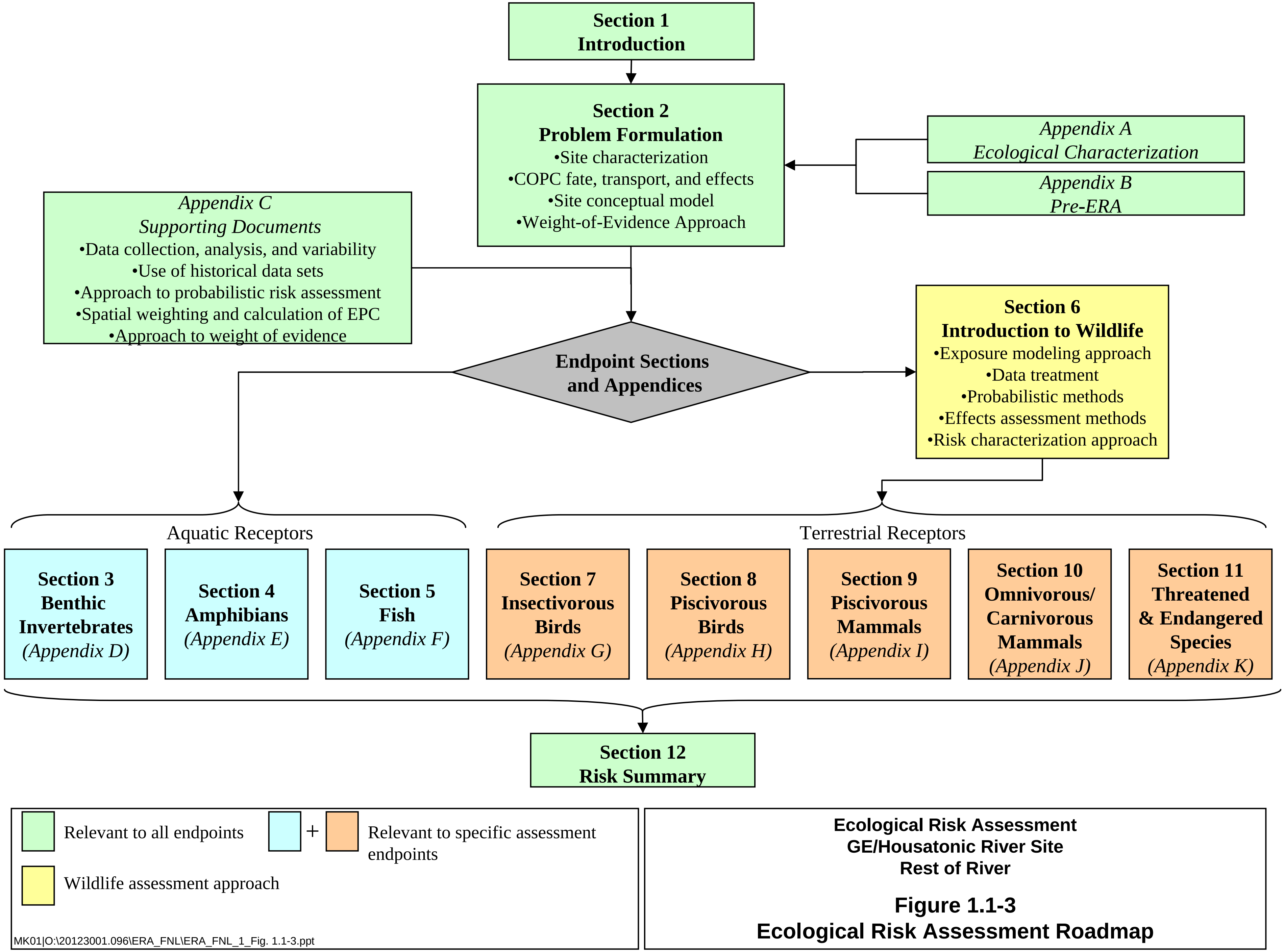
Beyond this dam, the river is subject to tidal influence, as well as COPCs (including PCBs) from other hazardous waste sites.

This ERA is structured as an integrated report summarizing information included in the supporting appendices and providing background common to all the assessment endpoints. The potential risk for each assessment endpoint is discussed in detail in Appendices D through K and is summarized in Sections 3 through 11. Other appendices provide additional information such as a comprehensive Ecological Characterization, the identification of COPCs and the extent of contamination for further consideration in the ERA, and other supporting information. Figure 1.1-3 provides a roadmap for the ERA and supporting appendices.

1.2 SITE HISTORY

The Housatonic River is located in a predominantly rural area of western Massachusetts and Connecticut, where farming was the main occupation from colonial settlement through the late 1800s. As with most rivers, the onset of the industrial revolution in the late 1800s brought manufacturing to the banks of the Housatonic River in Pittsfield, MA. GE began operations in its present location in 1903. Three manufacturing divisions have operated at the GE facility (Transformer, Ordnance, and Plastics).

The 254-acre (103-ha) GE facility in Pittsfield (Figure 1.2-1) has historically been the major handler of PCBs in western Massachusetts, and is the only known source of PCBs found in the Housatonic River sediment and floodplain soil in Massachusetts. Although GE performed many functions at the Pittsfield facility throughout the years, the activities of the Transformer Division, including the construction and repair of electrical transformers using dielectric fluids, some of which contained PCBs (primarily Aroclor 1260 and, to a lesser extent, 1254), were one likely significant source of PCB contamination. According to GE reports, from 1932 through 1977, releases of PCBs reached the wastewater and stormwater systems associated with the facility and were subsequently conveyed to the East Branch of the Housatonic River and to Silver Lake, a 25-acre (10-ha) lake adjacent to the GE facility.



Ecological Risk Assessment
GE/Housatonic River Site
Rest of River

Figure 1.1-3
Ecological Risk Assessment Roadmap

Ecological Risk Assessment
GE/Housatonic River Site
Rest of River

FIGURE 1.2-1
GE PLANT AREA: REMOVAL ACTION AREAS

| O:\gepitt\aprs\raas_locs.apr | layout - fig 1.2-1 | o:\gepitt\epsfiles\plots\in\fig1-2-1.eps | 10:31 AM, 7/10/2003 |

During the 1940s, efforts to straighten the Pittsfield reach of the Housatonic River by the City of Pittsfield and the U.S. Army Corps of Engineers (USACE) resulted in 11 former oxbows being isolated from the river channel. The oxbows were filled with material, much of which was later discovered to contain PCBs and other hazardous substances.

The State of Connecticut posted a fish consumption advisory for most of the Connecticut section of the river in 1977 as a result of the PCB contamination in the river sediment and fish tissue. In 1982, the Massachusetts Department of Public Health (MDPH) issued a consumption advisory for fish, frogs, and turtles for the Housatonic River. In addition, in 1999, MDPH issued a waterfowl consumption advisory from Pittsfield to Great Barrington due to PCB concentrations in wood ducks and mallards collected from the river by the U.S. Environmental Protection Agency (EPA).

Although much of the first 2 miles (3 km) downstream from the facility have been channelized, the remainder of the river's course is relatively unaffected (with the exception of the numerous dams downstream) in areas south of Pittsfield. The river, from the confluence of the East and West Branches of the Housatonic to Woods Pond Dam in Lenox, is 10.7 miles (17.3 km) long. The channel ranges from 40 to 125 ft (12 to 38 m) in width, is bordered by an extensive floodplain (up to 3,600 feet [1,100 m] wide), and has a meandering pattern with numerous oxbows and backwaters. Woods Pond, the first impoundment downstream of the GE facility, is a shallow 54-acre (22-ha) impoundment that was formed by the construction of a dam in 1864 (Harza 2001, as cited in BBL and QEA 2003).

The land uses of the floodplain properties in Massachusetts include residential, commercial/industrial, agricultural, recreational (such as canoeing, fishing, and hunting), wildlife management, and parks and a golf course. The Housatonic River floodplain is an attractive area for recreation, including fishing and waterfowl hunting.

Numerous studies conducted since 1988 have documented PCB contamination of soil within the floodplain of the Housatonic River downstream of the GE facility. PCBs have been detected in river sediment in Massachusetts as far downstream as the border with Connecticut (BBL 1995), and in Connecticut as far as the Derby Dam and beyond into Long Island Sound (other sources have been identified downstream of this dam). The PCBs detected in Housatonic River floodplain soil and sediment consist of predominantly Aroclor 1260, with a minor contribution of Aroclor 1254.

The highest concentrations of Aroclors 1254 and 1260 have been detected in the vicinity of the plant and downstream of Building 68 (WESTON 2000; BBL 1994, 1995; O'Brien & Gere Engineers, Inc. 1995). Widespread contamination of the river downstream of the GE facility has resulted from the transport of PCB-contaminated river sediment and floodplain soil by river flow, sediment transport, and overbank flooding (WESTON 2000). Total PCBs have been detected at concentrations of greater than 1 ppm in floodplain soil as far downstream as Bartholomew's Cobble in Massachusetts, close to the Massachusetts-Connecticut state line.

Numerous residential properties have been the focus of efforts by the Massachusetts Department of Environmental Protection (MDEP) to coordinate cleanup of residential soil contaminated with PCBs that was brought to the properties as fill from GE. GE has cleaned up approximately 170 properties to date under this program.

Other properties or areas in Pittsfield and the surrounding communities have been discovered over the years to have received waste from the GE facility and/or are contaminated with PCBs, including the Pittsfield Landfill, Rose Disposal Site (National Priorities List [NPL] site), and Dorothy Amos Park located on the West Branch of the Housatonic River. Actions to address these properties have been taken or investigation is underway.

1.3 REGULATORY BACKGROUND

The GE/Housatonic River site has been subject to regulatory investigations dating back to the early 1980s. For several years, these investigations were consolidated under the following regulatory mechanisms: two Administrative Consent Orders (ACOs) with MDEP and a Corrective Action Permit with EPA under the Hazardous and Solid Waste Amendments to the Resource Conservation and Recovery Act (RCRA).

In 1991, EPA issued a RCRA Corrective Action Permit to the GE Pittsfield facility. Following appeals by GE and others, and subsequent modification, the permit became effective in 1994. The permit included the 254-acre facility, some filled former oxbows, Silver Lake, the

Housatonic River and its floodplains and adjacent wetlands, and all sediment contaminated by PCBs migrating from the GE facility.

In addition to the permit, the two ACOs between GE and MDEP became effective in 1990 and included those areas defined in the permit, as well as additional filled former oxbows and Allendale Elementary School. Under the ACOs, GE has performed several investigations and short-term cleanups.

EPA proposed the site to the Superfund National Priorities List (NPL) in September 1997. Several federal and state government agencies and GE entered into negotiations late in 1997 with the goal of reaching a comprehensive settlement, which included remediation, redevelopment, and restoration components.

In September 1998, representatives of the federal and state government agencies, GE, the City of Pittsfield, and the Pittsfield Economic Development Authority reached an agreement in principle relating to GE's Pittsfield facility, other contaminated areas in Pittsfield, and the Housatonic River. This agreement was translated into a Consent Decree, lodged with the federal court on 7 October 1999, and entered by the court on 27 October 2000. The agreement provides for, among other things, the cleanup of the GE plant facility, cleanup and restoration of the former oxbows, cleanup and restoration of Silver Lake, cleanup of Allendale School, environmental restoration projects related to the Housatonic River and floodplains, monetary compensation for natural resource damages, and government recovery of past and future response costs. Entry of the agreement also made possible the start of large-scale redevelopment of the GE facility.

The GE/Housatonic River site is made up of several separate response actions (as described in the Consent Decree), including three actions in the river:

- Upper ½-Mile Reach Housatonic River Removal Action (½-Mile Reach).
- 1 ½-Mile Housatonic River Removal Action (1 ½-Mile Reach).
- Rest of River.

and actions outside the river, including:

- GE plant site soil remediation.
- Unkamet Brook and floodplain remediation.
- Hill 78/Building 71 consolidation areas.

- Groundwater remediation.
- Former oxbow areas.
- Allendale School.
- Floodplain current residential and nonresidential properties.
- Silver Lake.

The primary COPCs are PCBs, specifically Aroclor 1260 and, to a lesser extent, 1254. Other COPCs include volatile organics, dioxins/furans, polycyclic aromatic hydrocarbons (PAHs), semivolatiles, and metals. These contaminants vary in their distribution in different areas.

EPA completed an investigation of the Rest of River downstream of the 1½-Mile Reach into Connecticut, which focused on collecting information for and preparing the ecological and human health risk assessments, and modeling PCB fate and transport in the river. Under the terms of the Consent Decree, both of the risk assessments and three aspects of the modeling effort are to undergo formal external Peer Review. The Peer Review of the Modeling Framework Design occurred in April 2001; of the Human Health Risk Assessment, in November 2003; and of the Ecological Risk Assessment, in January 2004. Two additional Peer Reviews of the Model Calibration and Validation Reports still remain. The ecological risk assessment, together with the human health risk assessment and the model of PCB fate, transport, and bioaccumulation, will inform EPA's decision on what additional remedial actions, if any, may be required in the river and floodplain downstream of the confluence.

Following the investigations, as required in the Revised RCRA Permit, GE has prepared a Supplemental RCRA Facility Investigation Report (BBL and QEA 2003), will propose cleanup levels (Interim Media Protection Goals), and will analyze cleanup alternatives (Corrective Measures Study) for review and approval by EPA. EPA will propose the draft Statement of Basis (cleanup plan, scheduled for 2007) for the Corrective Measure(s) for the Rest of River and, after public comment, will finalize the Statement of Basis. GE and other members of the public may then appeal EPA's decision to the EPA Environmental Appeals Board (EAB) and the First Circuit District Court. GE is then required, under the Consent Decree, to implement and pay for the remedy selected after resolution of any appeals. The Rest of River response action, if any, will be implemented through a modification to the Revised RCRA Permit and an amendment to the CERCLA Consent Decree, and is estimated to begin in 2008.

1.4 SITE DESCRIPTION

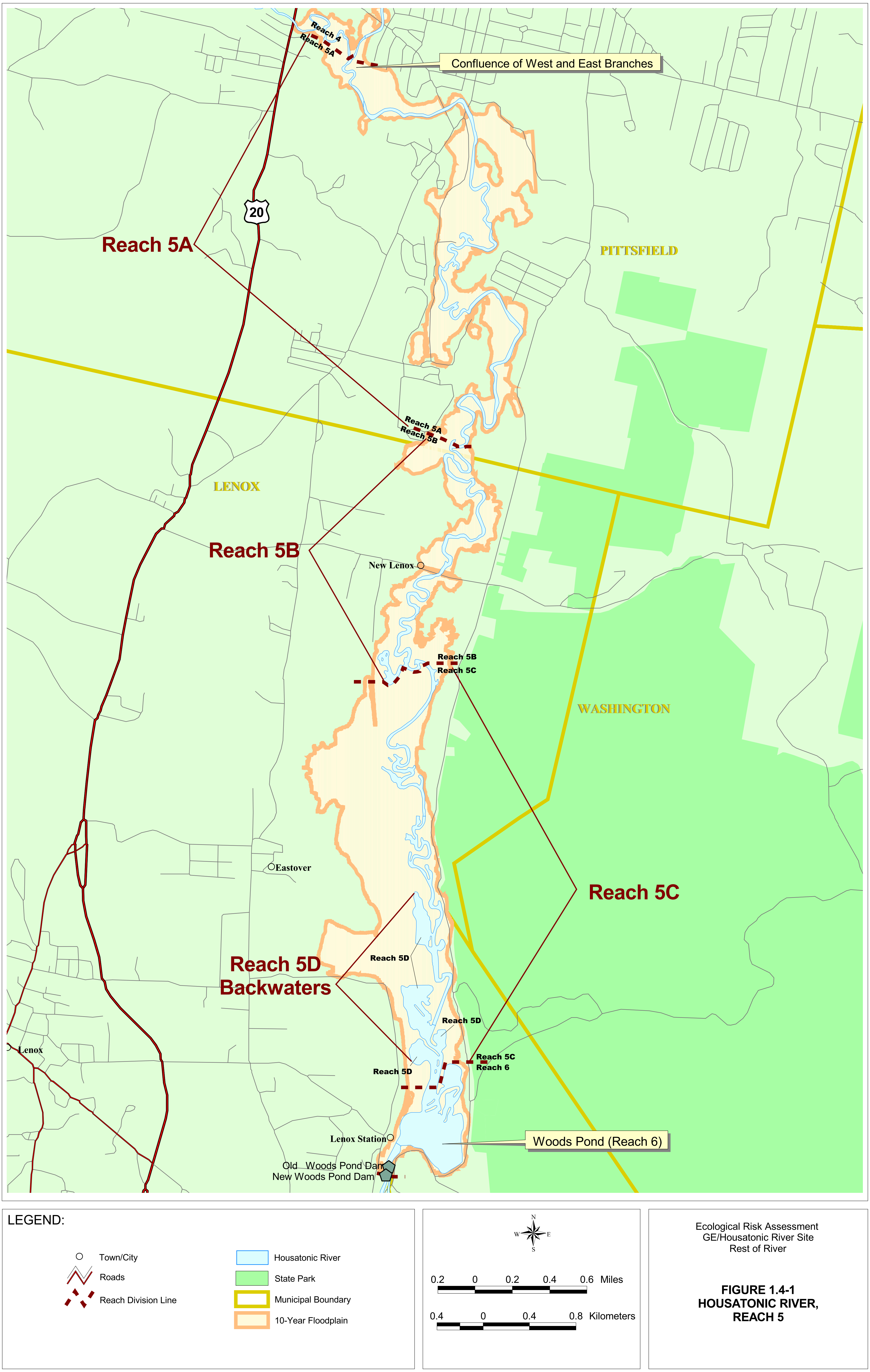
The Rest of River portion of the Housatonic River flows through one of the most biologically diverse regions of Massachusetts (Barbour et al. 1998) and Connecticut. Dams play an integral role in the downstream migration of PCBs and other COPCs from the GE facility.

The first 10.1 miles (16.3 km) from the confluence to the headwaters of Woods Pond is referred to as Reach 5 (Figure 1.4-1). Other than the pre-remediation concentrations in the initial 0.5-mile (0.8-km) reach bordering the GE facility, Reach 5 has the highest concentrations and highest frequency of detections of PCBs in sediment. Reach 5 is subdivided further into four parts: Reach 5A, from the confluence to just above the Pittsfield Wastewater Treatment Plant (WWTP); Reach 5B, from the WWTP to the mouth of the Roaring Brook tributary; Reach 5C, from Roaring Brook to the headwaters of Woods Pond; and Reach 5D, the backwaters above Woods Pond (Woods Pond is Reach 6) (Figures 1.4-2 and 1.4-3).

The Housatonic River meanders through Reach 5A, with widths between 40 and 120 ft (12 and 37 m) and depths up to 11 ft (3.4 m).

Aquatic habitat includes snags (large woody debris), undercut banks, and rocks. Land use in this section is predominantly forested and cleared, with some residential areas. Reach 5B is similar to Reach 5A from the WWTP to New Lenox Road. The land near New Lenox Road is predominantly agricultural and forested. Below New Lenox Road, the river widens and becomes shallower.

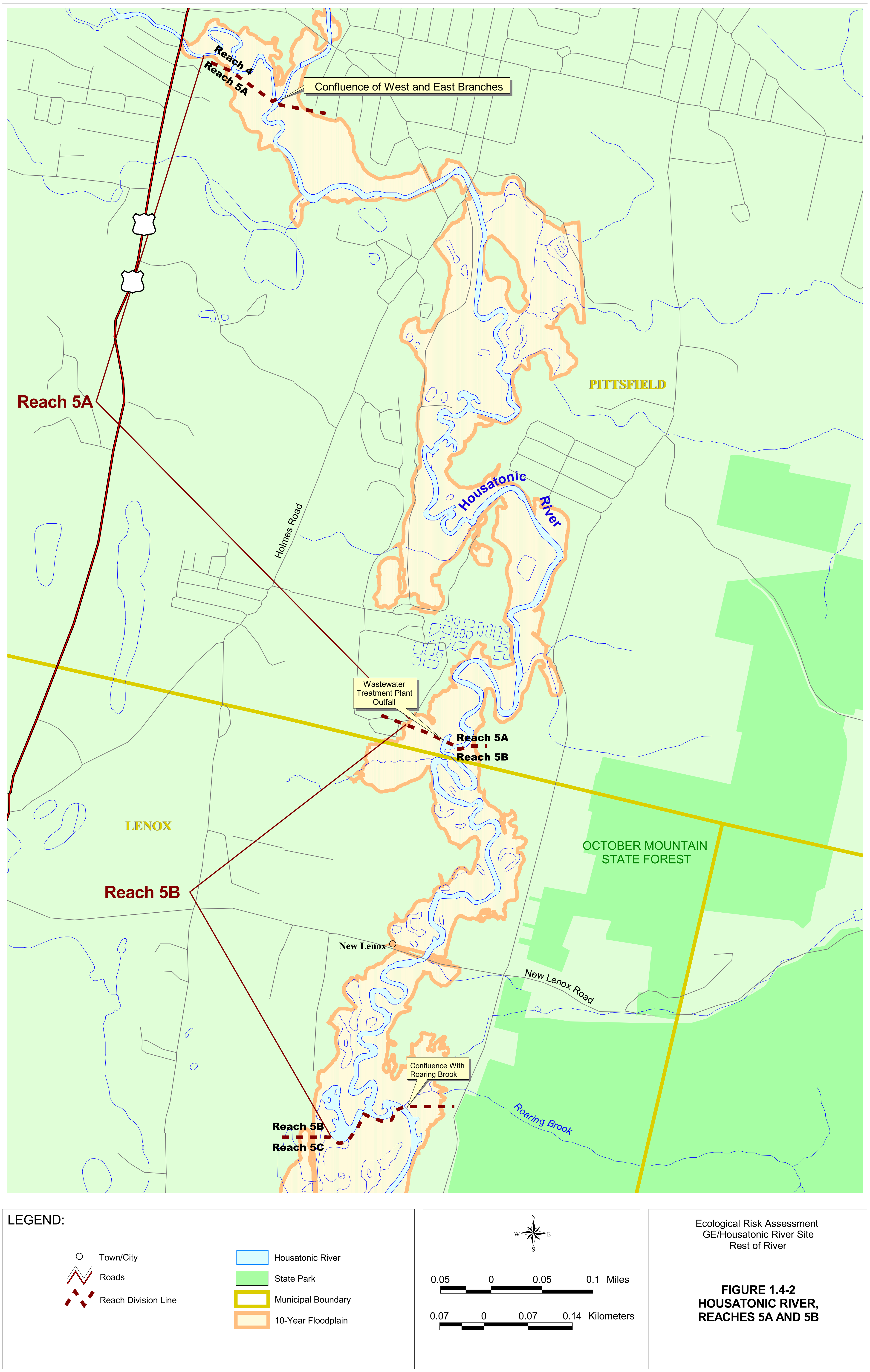
This portion of Reach 5B is dominated by a broad wetland floodplain, ranging up to 2,200 ft (671 m) wide (see Appendix A). Reach 5C is similar to Reach 5B, although as the Housatonic River approaches Woods Pond, the velocity decreases and deep pools occur (up to and exceeding 7 ft [2 m]), created by large snags that divert water flow, and the effect of Woods Pond Dam becomes apparent. Dense vegetation lines the banks of the river in the upper portion of this section, while extensive backwaters border the lower section. Reach 5D consists of several upstream backwater areas associated with Woods Pond and covers more than 120 acres (49 ha).



Ecological Risk Assessment
GE/Housatonic River Site
Rest of River

FIGURE 1.4-1
HOUSATONIC RIVER,
REACH 5

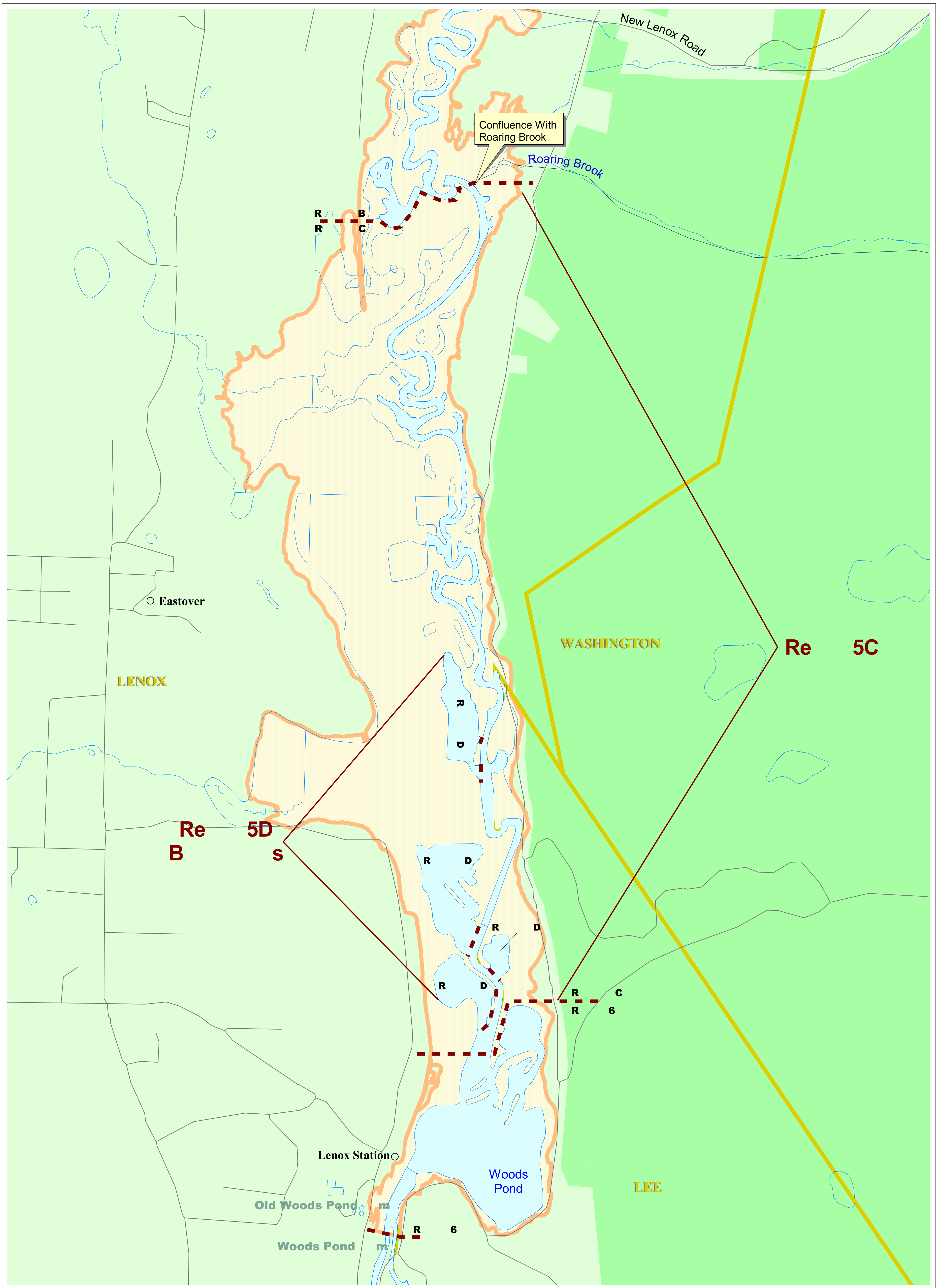
| O:\gepitt\aprs\era_figures2.apr | layout - fig1.4-1 | o:\gepitt\epsfiles\plots\in\fig1-4-1.eps | 3:14 PM, 10/11/2004 |



Ecological Risk Assessment
GE/Housatonic River Site
Rest of River

FIGURE 1.4-2
HOUSATONIC RIVER, REACHES 5A AND 5B

| O:\gepitt\aprs\era_figures2.apr | layout - fig1.4-2 | o:\gepitt\epsfiles\plots\in\fig1-4-2.eps | 10:16 AM, 3/7/2003 |



LEGEND:

- Town/City
- Dam
- Roads
- Reach Division Line
- Housatonic River
- State Park
- Municipal Boundary
- 10-Year Floodplain

N W E S

0.04 0 0.04 0.08 Miles

0.06 0 0.06 0.12 Kilometers

Ecological Risk Assessment
GE/Housatonic River Site
Rest of River

FIGURE 1.4-3
HOUSATONIC RIVER,
REACHES 5C AND 5D

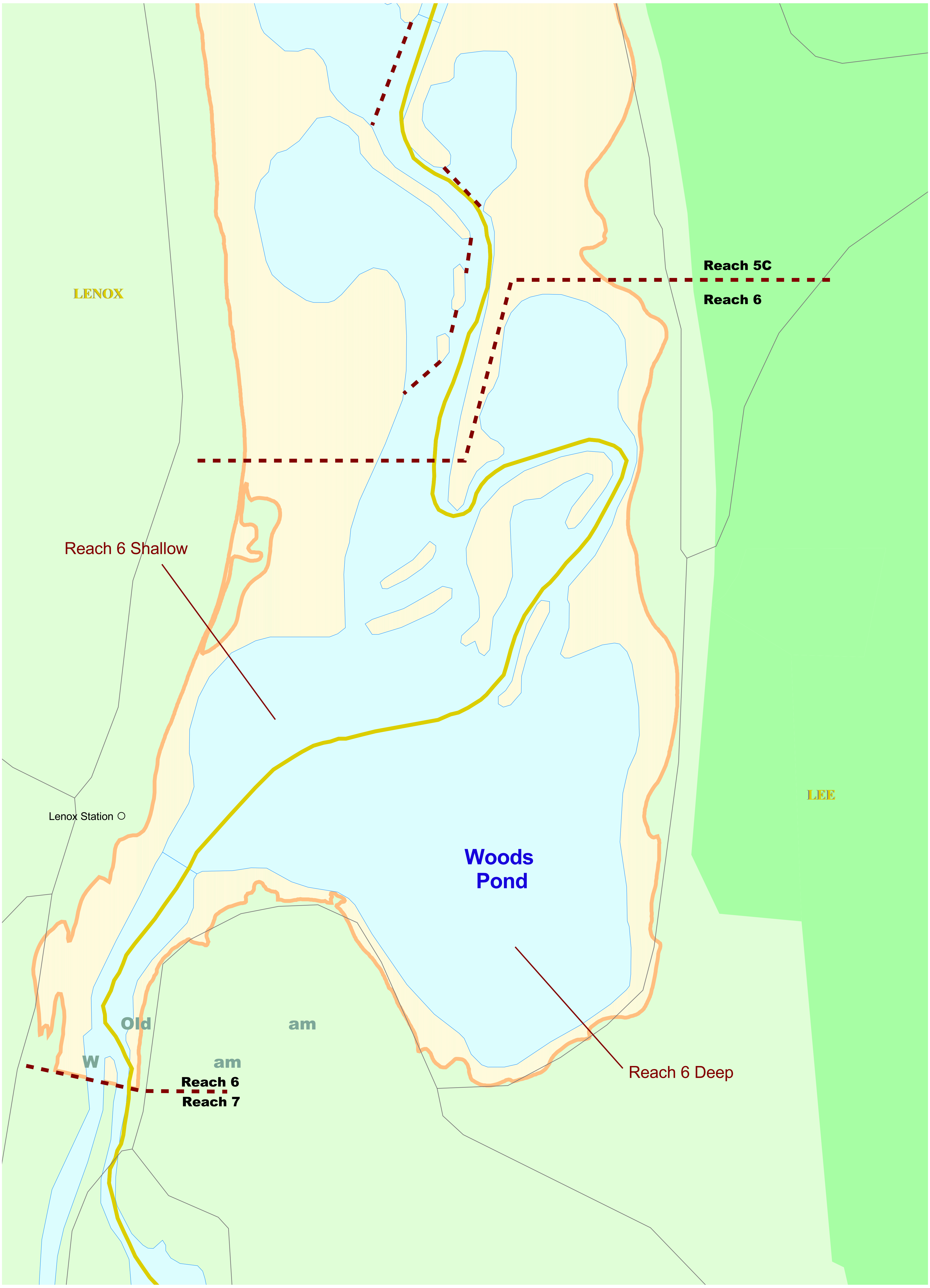
| O:\gepitt\aprs\era_figures2.apr | layout - fig1.4-3 | o:\gepitt\epsfiles\plots\in\fig1-4-3.eps | 10:35 AM, 3/7/2003 |

Reach 6 begins approximately 10.1 miles (16.3 km) downstream of the confluence at a point 700 ft (213 m) upstream of Woods Pond, and is approximately 0.57 mile (0.9 km) in length. This reach contains the first impoundment downstream from the GE facility and is a depositional environment (HEC 1996). Woods Pond, the dominant feature in Reach 6, is approximately 0.2 mile (0.3 km) in length and has an area of 54 acres (22 ha) (Figure 1.4-4).

The water in Woods Pond is relatively slow-moving and contains aquatic habitat characteristic of a standing-water environment. The maximum depth is 16 ft (5 m), but most of the pond is 1 to 3 ft (0.3 to 0.9 m) deep (HEC 1996; Stewart Laboratories, Inc. 1982; CR Environmental 1998). The banks of the pond provide extensive cover, such as overhanging vegetation, woody debris, rock piles, and submerged macrophytes. The Woods Pond Dam was built in 1864. In 1989, GE replaced the original dam with a concrete weir dam located 150 ft (46 m) downstream of the original dam site.

Reach 7 extends 18.5 miles (29.8 km) from Woods Pond to the upstream end of Rising Pond in Great Barrington (Figure 1.4-5). There are five dams in this reach, and the river has an average gradient of 14.5 ft (4.42 m) per mile (Stewart Laboratories, Inc. 1982). The average depth of the river is between 3 and 5 ft (0.9 to 1.5 m) in the faster flowing sections of the river channel and upwards of 20 feet (6 m) just upstream of the dams. Agricultural activity becomes more common in this area than in the upstream reaches.

Reach 7 ends above Rising Pond, which is Reach 8 (Figure 1.4-5). This 45-acre (18-ha) pond was created by the construction of a dam at the Rising Paper Company (WESTON 2000). Rising Pond has depositional characteristics similar to Woods Pond.



LEGEND:

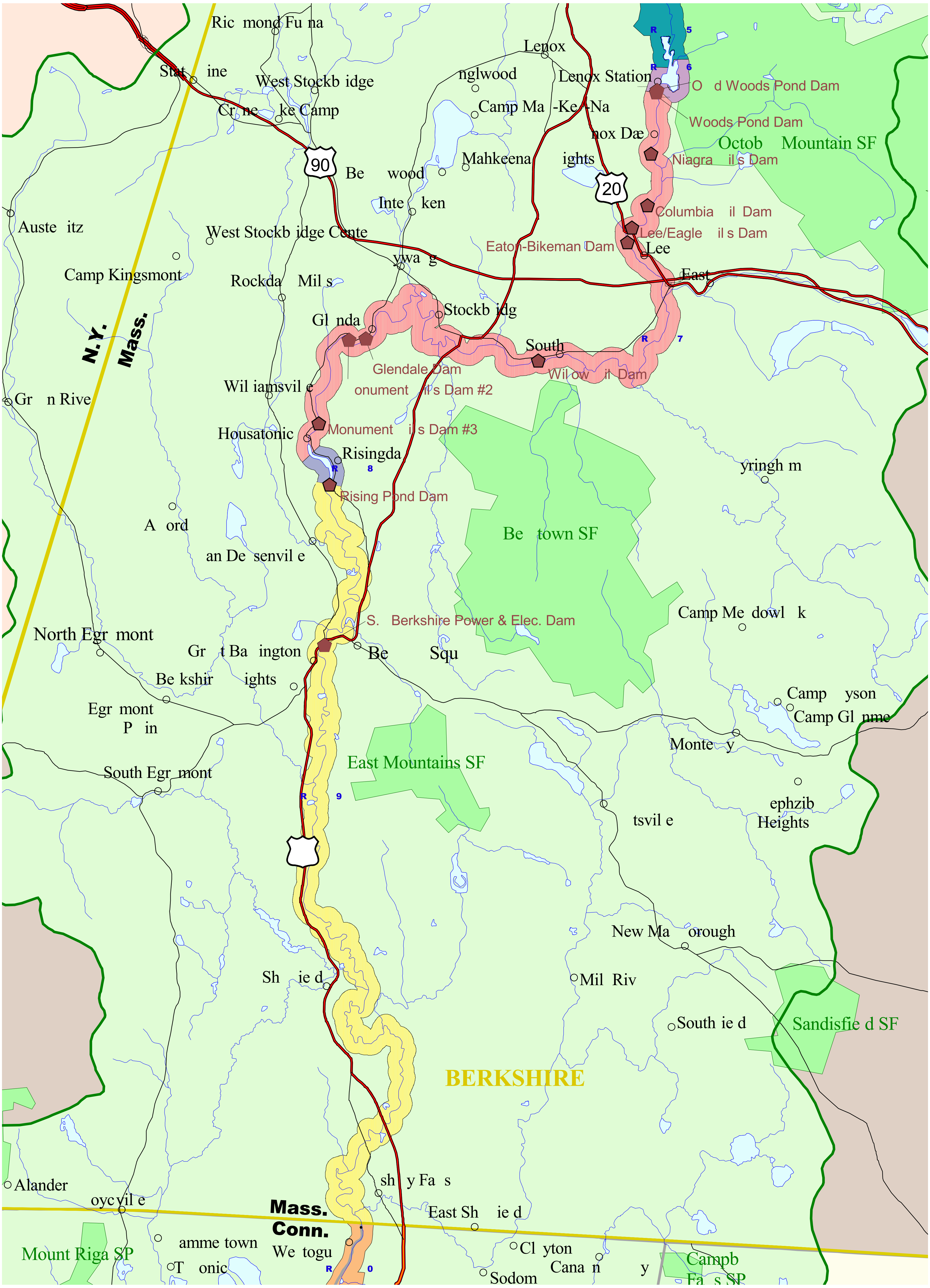
- Town/City
- Dam
- Roads
- Reach Division Line
- Housatonic River
- State Park
- Municipal Boundary
- 10-Year Floodplain

0.02 0 0.02 Miles

0.03 0 0.03 Kilometers

Ecological Risk Assessment
GE/Housatonic River Site
Rest of River

FIGURE 1.4-4
HOUSATONIC RIVER, REACH 6



LEGEND:

- Town/City
- Dam
- Roads
- Housatonic River
- Housatonic Watershed
- State Park
- County Boundary
- Reach 5
- Reach 6
- Reach 7
- Reach 8
- Reach 9
- Reach 10

N, W, E, S

1 0 1 2 Miles

1.5 0 1.5 3 Kilometers

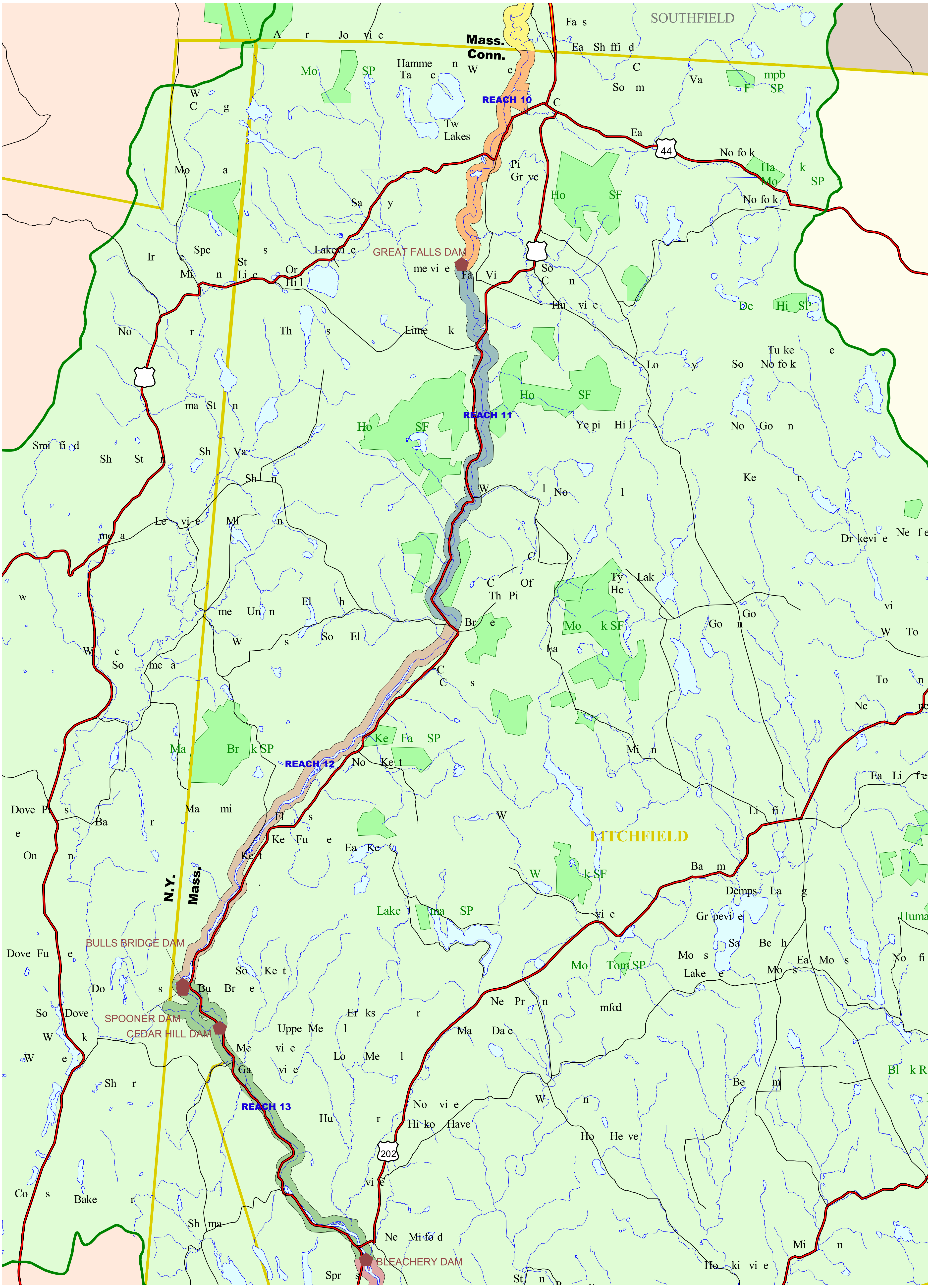
Ecological Risk Assessment
GE/Housatonic River Site
Rest of River

FIGURE 1.4-5
HOUSATONIC RIVER, REACHES 7 TO 9

Reach 9 begins downstream of Rising Pond and extends for 23.9 miles (38.5 km) to the Massachusetts/Connecticut state line (Figure 1.4-5). It contains low-gradient slow-moving sections, as well as moderate gradient sections with riffle habitat. This reach is wide with flat floodplains and several oxbows, and agriculture is a predominant land use.

Reach 10 begins at the Massachusetts/Connecticut border and extends 7.4 miles (12 km) to the dam at Great Falls Village (Figure 1.4-6). The river characteristics are similar to those of Reach 9, with a meandering river course. Reach 11 begins downstream of the dam at Great Falls and ends at Cornwall Bridge, where Route 7 crosses the river (Figure 1.4-6). This reach is 11.5 miles (18.5 km) long. Reach 11 is mostly shallow and fast flowing, and much of the reach is designated as a Trout Management Area. Reach 12 extends from Cornwall Bridge to the dams at Bulls Bridge (Figure 1.4-6), a length of 13.1 miles (21.1 km). The river is relatively straight through this reach and flows quickly for most of the reach. Near the town of Kent, the river slows and deepens as it enters the backwaters from the dams at Bulls Bridge. Reach 13 starts on the downstream side of the dams at Bulls Bridge and runs 10.9 miles (17.5 km) to the Bleachery Dam at New Milford, CT (Figure 1.4-6). The Bleachery Dam is virtually submerged as a result of the backwater created by the Shepaug Dam farther downstream. The river meanders more than in the previous reach and, as in Reaches 11 and 12, is relatively fast-flowing.

Reach 14, from the Bleachery Dam to Shepaug Dam, is known as Lake Lillinonah (Figure 1.4-7). The reach is 12.5 miles (20.1 km) long. The Shepaug Dam is approximately 100 ft (30 m) high. The backwater effect from the Shepaug Dam extends all the way to the upstream Bleachery Dam. The Shepaug Dam is used for power generation and this may affect water levels during the year. Water movement is slower through this reach and the river is deep. Reach 15 encompasses Lake Zoar, from Shepaug Dam to Stevenson Dam (Figure 1.4-7). This predominantly slow-moving reach is 10.2 miles (16.4 km) long. The backwater effect of the Stevenson Dam extends upstream for almost the entire reach. The Stevenson Dam is approximately 100 ft (30 m) high and supports power generation. Some homes and boat launches are located on Lake Zoar.



LEGEND:

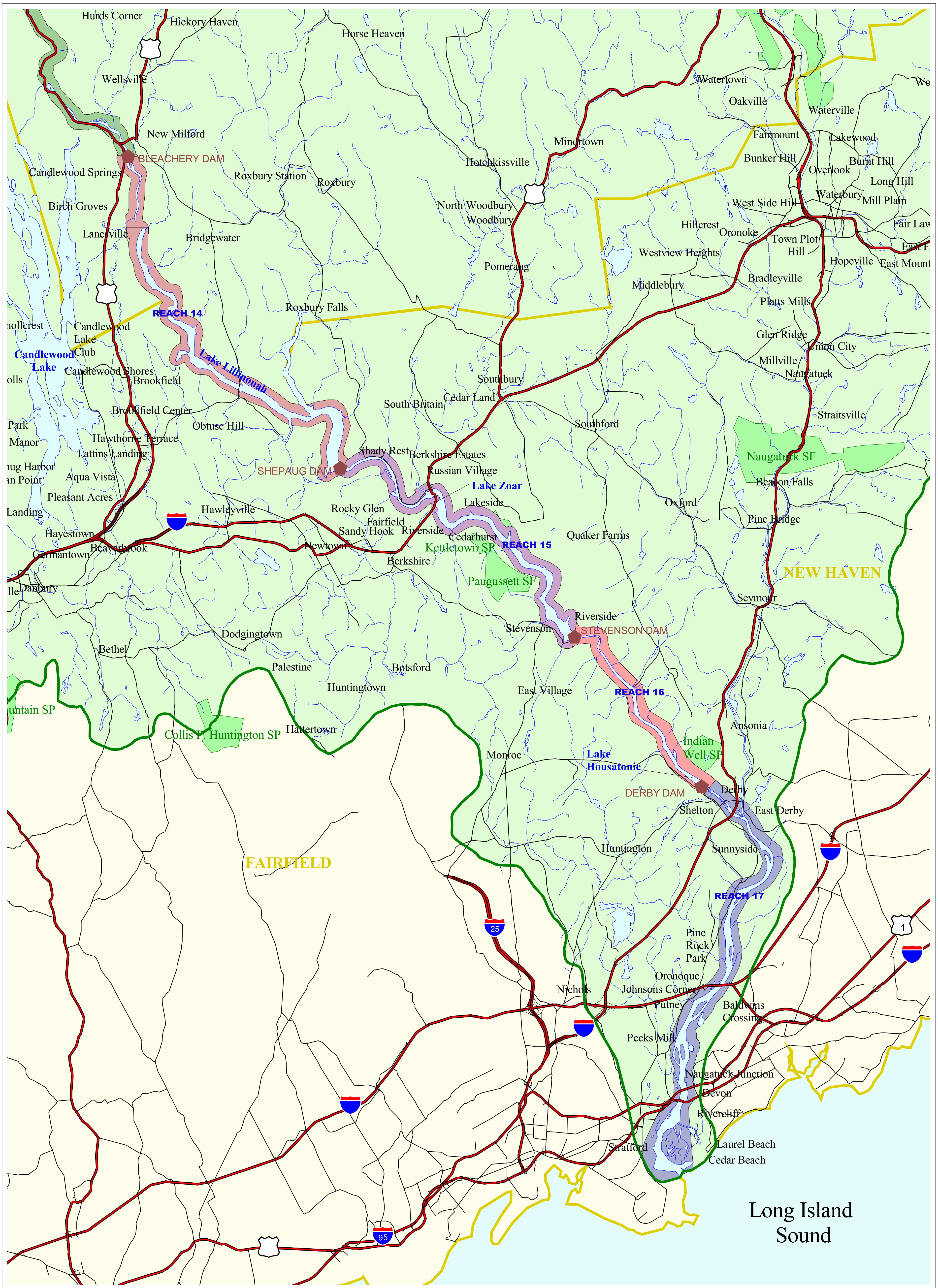
- Town/City
- Dam
- Roads
- Housatonic River
- State Park
- County Boundary
- Housatonic Watershed
- Reach 9
- Reach 10
- Reach 11
- Reach 12
- Reach 13
- Reach 14

1 0 1 2 3 4 Miles

2 0 2 4 Kilometers

Ecological Risk Assessment
GE/Housatonic River Site
Rest of River

FIGURE 1.4-6
HOUSATONIC RIVER,
REACHES 10 TO 13



LEGEND:

Town/City	Reach 13
Dam	Reach 14
Roads	Reach 15
Housatonic River	Reach 16
State Park	Reach 17
County Boundary	
Housatonic Watershed	

1.5 0 1.5 3 Miles

2 0 2 4 Kilometers

Ecological Risk Assessment
GE/Housatonic River Site
Rest of River

FIGURE 1.4-7
HOUSATONIC RIVER, REACHES 14 TO 17

| O:\gepitt\aprs\era_figures2.apr | layout - fig1.4-7 | o:\gepitt\epsfiles\plots\in\fig1-4-7.eps | 12:33 PM, 10/27/2004 |

Reach 16 is Lake Housatonic and is bounded by the Stevenson Dam and the Derby Dam (Figure 1.4-7). The reach is 6.0 miles (9.7 km) long and, like the previous two upstream reaches, is slow moving. The Derby Dam (approximately 25 ft [7.6 m] high) is smaller than either the Shepaug or Stevenson Dams. More homes and boat launches occur along this reach. Reach 17, from Derby Dam to Long Island Sound, is 13.7 miles (22.0 km) long (Figure 1.4-7). This reach is entirely tidally influenced. It is shallow in its upstream portions and deepens downstream. The Naugatuck River enters the Housatonic River approximately 1 mile (1.6 km) from the upstream end of this reach.

1.5 OVERVIEW OF TECHNICAL APPROACH

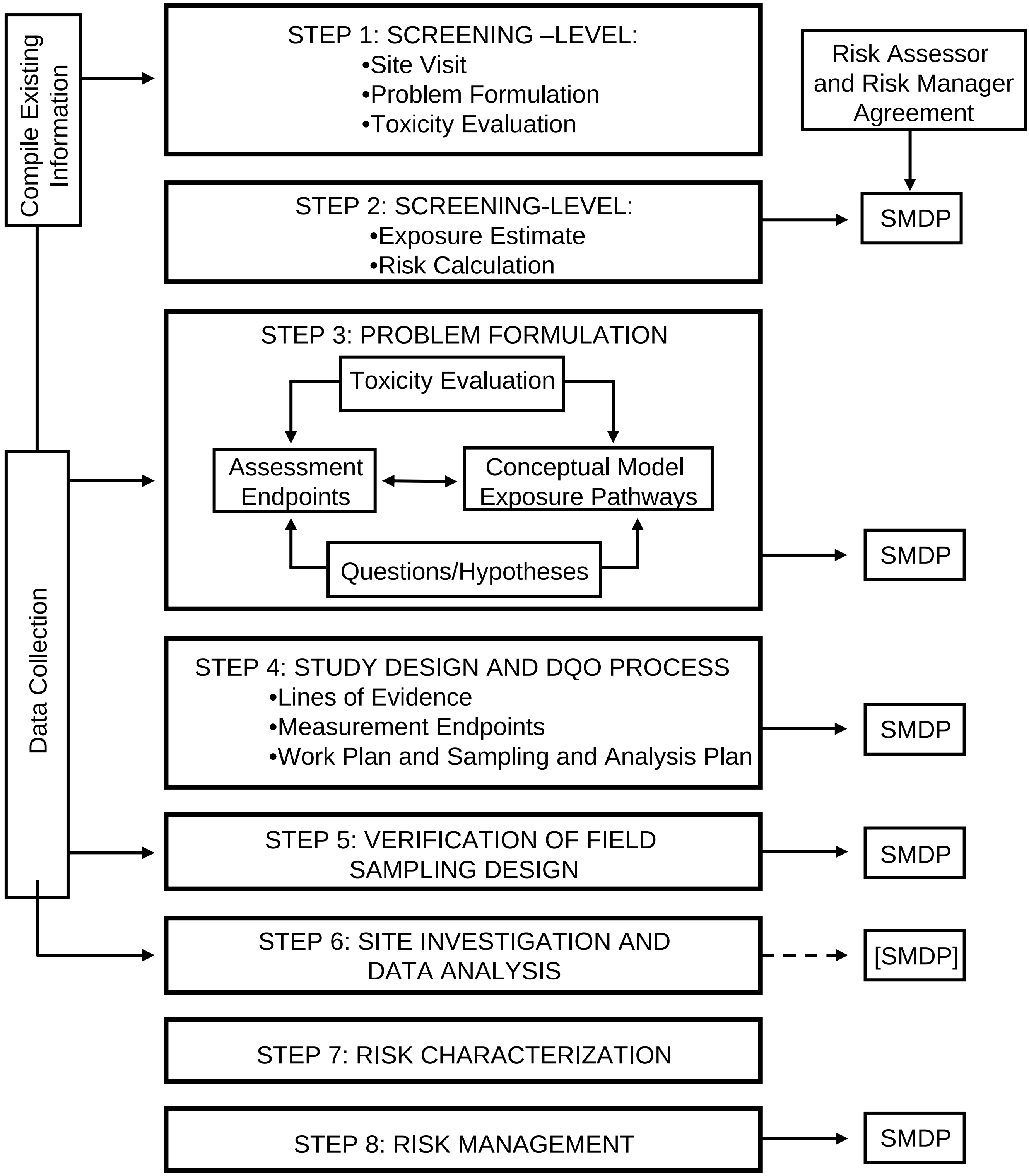
This ERA follows the technical approach and guidelines detailed in EPA's *Ecological Risk Assessment Guidance for Superfund: Process for Designing and Conducting Ecological Risk Assessments* (EPA, 1997a). Additional documents were also consulted, including, but not limited to, the following:

- *Framework for Ecological Risk Assessment* (EPA/630/R-92/001, 1992a).
- *Guidelines for Ecological Risk Assessment* (EPA/630/R-95-002F, April 1998).
- *Ecological Risk Assessment Issue Papers* (EPA/630/R-94/009, November 1994, 1994e).
- *Wildlife Exposure Factors Handbook*, Volumes I and II (EPA 600/R-93/187a and 187b, December 1993, 1993a).
- *Guidance for Disposal Site Risk Characterization: Method 3 Environmental Risk Characterization* (MDEP 1996).
- *The Role of BTAGs in Ecological Assessment*, *ECO Update*, Volume 1, Number 1 (EPA 1991a).
- *Ecological Assessment of Superfund Sites: An Overview*, *ECO Update*, Volume 1, Number 2 (EPA 1991b).
- *The Role of Natural Resource Trustees in the Superfund Process, ECO Update*, Volume 1, Number 3 (EPA 1992b).
- *Using Toxicity Tests in Ecological Risk Assessment*, *ECO Update*, Volume 2, Number 1 (EPA 1994a).

- *Catalogue of Standard Toxicity Tests for Ecological Risk Assessment, ECO Update*, Volume 2, Number 2 (EPA 1994b).
- *Field Studies for Ecological Risk Assessment, ECO Update*, Volume 2, Number 3 (EPA 1994c).
- *Selecting and Using Reference Information in Superfund Ecological Risk Assessments, ECO Update*, Volume 2, Number 4 (EPA 1994d).
- *Ecotox Thresholds, ECO Update*, Volume 3, Number 2 (EPA 1996).
- *RAGS*, Volume 3, *Part A: Process for Conducting Probabilistic Risk Assessment* (EPA 540-R-02-002, December 2001).
- *Guiding Principles for Monte Carlo Analysis* (EPA/63C/R-97/001, 1997b).
- *Ecological Risk Assessment and Risk Management Principles for Superfund Sites* (EPA 1999).

The *Ecological Risk Assessment Guidance for Superfund* (EPA 1997a) details an eight-step process for conducting an ERA (Figure 1.5-1). This document provides the user with a basic framework for the ERA process and ensures a consistent approach to conducting ERAs. In addition to these steps, there are several scientific/management decision points (SMDPs). These are opportunities for the risk manager and the risk assessment team to communicate ideas concerning the scope, focus, and direction of the ERA. The first two steps of the ERA process (Screening-Level Problem Formulation and Screening-Level Exposure Estimate and Risk Calculation) were first addressed in the *Upper Reach-Housatonic River Ecological Risk Assessment* (WESTON 1998) and subsequently refined in Appendix B of this document. Steps 3, 4, and 5 (Baseline Problem Formulation, Study Design and DQO Process, and Verification of Field Data Analysis) are iterative components of the eight-step ERA process. Steps 3 through 5 were initially presented in the *Supplemental Investigation Work Plan for the Lower Housatonic River* (SIWP) (WESTON 2000) and were modified as necessary during the data collection phase of the project. Steps 6 and 7 (Site Investigation and Data Analysis and Risk Characterization) are presented in detail in the following sections and appendices. Step 8 (Risk Management) will be addressed after the ERA has been completed.

Ecological Risk Assessment for the Housatonic River



Legend:
SMDP - Scientific/management decision point
[SMDP] - only if change to the sampling and analysis plan is necessary

Source: EPA (U.S. Environmental Protection Agency), Environmental Response Team. 1997. *Ecological Risk Assessment Guidance for Superfund: Process for Designing and Conducting Ecological Risk Assessments.* Interim Final. EPA 540-R-97-006.

Housatonic River Project
Pittsfield, Massachusetts

Figure 1.5-1

EIGHT-STEP ECOLOGICAL RISK ASSESSMENT PROCESS FOR SUPERFUND

MK01|O:\20123001.096\ERA_FNL\ERA_FNL_1_Fig. 1.5-1.ppt

1.5.1 Problem Formulation

Problem formulation is an important component of the ERA process that establishes the goals, objectives, and scope for the ERA. Products of problem formulation include the identification of assessment endpoints, illustration of exposure pathways (relating fate and transport to ecological effects), a conceptual model depicting the relationships between COPCs and the assessment endpoints, and risk hypotheses and questions that can be drawn from evident or suspected effects. The problem formulation portion of the ERA is discussed in Section 2 and was outlined in the *Supplemental Investigation Work Plan for the Lower Housatonic River* (WESTON 2000).

1.5.1.1 Physical and Ecological Characterization

An extensive physical and ecological characterization of the Housatonic River is presented in Section 2.2 and Appendix A (Ecological Characterization) of this document. These sections detail the physical setting, habitats, and biotic communities of the river in both the aquatic and terrestrial environments. The physical and ecological characterizations aid in identifying representative species and exposure pathways, provide information for the exposure analyses, and can inform risk managers on the potential impacts of future remedial actions.

1.5.1.2 Stressors and Their Sources

Investigations of the nature and extent of contaminants in the Housatonic River watershed have previously been conducted by GE, EPA, and others. PCBs have been identified as the main COPC, and other contaminants such as dioxins and furans and PAHs have also been identified at the GE facility. In addition, other COPCs, such as pesticides, were detected in the PSA, although at much lower concentrations and frequencies of detection. In Section 2.3, the sources, amounts, and patterns of contaminant releases and receiving bodies are presented.

1.5.1.3 Pre-Ecological Risk Assessment

> ***COPC vs COC***
>
> In the ERA, contaminants of potential concern (COPC) refer to contaminants considered before, during, and immediately after the Pre-ERA process. A contaminant is considered a contaminant of concern (COC) if it has passed through all screening-level processes and is included as part of the exposure and effects assessment conducted for a specific assessment endpoint.

The purpose of the Pre-ERA (Appendix B) was to identify contaminants that warranted more detailed analyses in the ERA, and those that could be removed from further consideration because they pose minimal risk. For those contaminants that screened through to the ERA, the primary media of concern as well as the sections of the study area that are potentially impacted are identified. A summary of the Pre-ERA is provided in Section 2.4. The complete Pre-ERA is included as Appendix B to this document.

1.5.1.4 Fate and Transport of Contaminant Stressors

An overview of the environmental behavior of PCBs and other COPCs is presented in Section 2.5. This section includes discussions of the transport of the contaminants from their point(s) of release, partitioning behavior in different media, and biotic and abiotic degradation in these media.

1.5.1.5 Effects on Representative Species

The effects and mechanisms of toxicity to biota of the contaminants identified as COPCs within the Housatonic River and floodplain are discussed, with an emphasis on PCBs, in Section 2.6, and in further detail in the effects assessment portion of each assessment endpoint section and corresponding appendix.

1.5.1.6 Conceptual Model, Selection of Assessment and Measurement Endpoints, and Analysis Plan

The conceptual model outlined in Section 2.7 describes the relationship between the COPCs and the biota at the site. Development of a conceptual model includes review of sources of contamination, evaluation of the spatial scale for the assessment, description of the exposure pathways, and formulation of risk questions to be addressed.

Assessment and measurement endpoints are defined and described in Section 2.8. An assessment endpoint is defined as the "explicit expression of the environmental value that is to be protected" (EPA 1997a). Because it is often unrealistic to perform an assessment for all species present at a contaminated site, species or populations are often grouped based on their similarities (e.g., exposure pathway, contaminant sensitivity) or societal importance (e.g., threatened and endangered species), and assessment endpoints are established for these groups of similar species. A measurement endpoint is defined as "a measurable ecological characteristic that is related to the valued characteristic chosen as the assessment endpoint," and is a measure of biological effects (e.g., mortality, reproduction, growth) (EPA 1997a). Measurement endpoints are frequently numerical expressions of observations (e.g., toxicity test results, community diversity measures) that can be compared statistically to a reference site to detect adverse responses to a site contaminant (EPA 1997a).

Section 2.9 describes the analytical approach used to estimate risks and the weight-of-evidence approach used to develop the conclusions.

1.5.2 Assessment of Representative Species

Sections 3 through 11 (and their corresponding appendices) provide an overview of the exposure assessment, the effects assessment, and the risk characterization for each representative species or representative group of species.

1.5.2.1 Exposure Assessment

The exposure assessment sections include a description of the data collection activities and the studies conducted to determine concentrations of COPCs in water, soil, sediment, and biota samples. Previous sampling and monitoring studies are also described in this section. Variation in PCB and COPC concentrations over space and time in each environmental medium is briefly characterized. For each assessment endpoint, one or more representative species were selected and an appropriate exposure model identified.

1.5.2.2 Effects Assessment

The effects assessment sections begin with an overview of the toxicity of PCBs and other COPCs. For each major representative species group and COPC, the effects literature was reviewed. The goal of this review was to identify important studies that could be used to develop effects metrics for use in risk characterization. The effects metrics developed ranged from concentration- or dose-response curves to threshold concentrations depending on the quality and relevance of the data available.

1.5.2.3 Risk Characterization

The risk characterization sections for each assessment endpoint consider three major lines of evidence (where available): (1) comparison of estimated exposures to laboratory-based effects metrics, (2) results of in situ or whole media toxicity tests, and (3) results of field surveys. Probabilistic methods were used to integrate COPC exposure distributions in the study area with laboratory-derived benchmarks and effects curves. The format of the discussion includes an overview of the study, statistical analyses of the results and conclusions stating the observed effects, and likely causal agents. The risk characterization concludes with a weight-of-evidence assessment for each assessment endpoint. Primary sources of uncertainty are also identified.

1.6 DATA SOURCES

The Housatonic River ERA generally relies on data from studies and research specifically designed for this assessment. Field surveys were conducted to support the ecological characterization and ecological risk assessments for benthic invertebrates, amphibians, fish, birds, and mammals in the Housatonic River floodplain. Prior to the surveys, literature reviews were conducted to establish historic populations and habitats for species within the study area. Surveys were also conducted at several reference sites (i.e., areas of relatively low contamination within the Housatonic River watershed).

A variety of site-specific studies were conducted, including the following:

- Survival, growth, and reproduction of benthic organisms as part of the Sediment Quality Triad (SQT) approach.
- Reproductive success of amphibians in the Housatonic River floodplain and the effects of exposure to PCBs and other COPCs on these species.
- Studies with largemouth bass (*Micropterus salmoides*) to determine if exposure of adults to PCBs and COPCs in river water and sediment adversely affect the survival and development of offspring.
- Investigation of tree swallows (*Tachycineta bicolor*) to determine the extent to which PCBs and other COPCs are impairing their reproduction.
- A reproductive toxicology study with farm-raised mink (*Mustela vison*) exposed to PCBs and other COPCs in their diet from fish collected from the Housatonic River.

These and other studies are described in more detail in the *Supplemental Investigation Work Plan for the Lower Housatonic River* (WESTON 2000) and its appendices. Information on study design, methodology, and quality assurance (QA)/quality control (QC) procedures can also be found in the *Supplemental Investigation Work Plan for the Lower Housatonic River*.

In addition, GE, without review by or consultation with EPA, conducted the following studies in the PSA (unless otherwise noted):

- Productivity of robins.
- Productivity and density of belted kingfishers *(Ceryle alcyon)*.
- Analysis of context-dependent effects on early life stages on wood frogs *(Rana sylvatica)*.
- Spatial and demographic effects on tree swallows (performed in Canada).
- Demographics of short-tailed shrews *(Blarina brevicauda)*.
- Field observations of presence/absence of mink.
- Evaluation of largemouth bass habitat, population structure, and reproduction.

EPA project data are managed using a relational structure in Microsoft Access. The database contains information on PCBs and other COPCs in soil, sediment, and tissue samples, and other field study data collected by EPA and other parties, constituting more than 2 million records.

ArcView (geographic information system [GIS]) was used to illustrate spatial patterns. Data originating from previous or concurrent studies conducted by GE and other sources were used if data quality was acceptable. The procedure followed for evaluating data quality of historical studies is described in Appendix C.

1.7 QA/QC

QA and QC procedures and techniques are established to guide data collection, analysis, modeling, administration, and auditing. Three documents, the *Quality Assurance Project Plan* (WESTON 2001a), the *Supplemental Investigation Work Plan for the Lower Housatonic River* (WESTON 2000), and the *Field Sampling Plan* (WESTON 2001b), outline the QA/QC procedures and techniques used in the studies conducted by EPA in support of this ecological risk assessment. These documents also provide details on the methods used in the collection and analyses of data.

1.8 REFERENCES

Barbour, H., T. Simmons, P. Swain, and H. Woolsey. 1998. *Our Irreplaceable Heritage, Protecting Biodiversity in Massachusetts*. Massachusetts Heritage and Endangered Species Program and The Massachusetts Chapter of the Nature Conservancy. 83 pp.

BBL (Blasland, Bouck & Lee, Inc.). 1994. *MCP Interim Phase II Report and Current Assessment Summary for East Street Area 2/U.S. EPA Area 4*. Volumes I, II, III, IV, V, VI, VII, IX, X, and XII.

BBL (Blasland, Bouck & Lee, Inc.). 1995. *MCP Supplemental Phase II Scope of Work and Proposal for RCRA Facility Investigation of Unkamet Brook Area/U.S. EPA Area 1*.

BBL (Blasland, Bouck & Lee, Inc.) and QEA (Quantitative Environmental Analysis, LLC). 2003. *Housatonic River- Rest of River RCRA Facility Investigation Report.* . Prepared for General Electric Company..

CR Environmental. 1998. *Housatonic River Supplemental Investigation Sub-bottom Profiling, Woods and Rising Ponds*. Prepared for Roy F. Weston, Inc. December 1998.

EPA (U.S. Environmental Protection Agency). 1991a. *The Role of BTAGs in Ecological Assessment*. Office of Solid Waste and Emergency Response, Office of Emergency and Remedial Response, Hazardous Site Evaluation Division (OS-230). *ECO Update*, Intermittent Bulletin, Volume 1, Number 1. Publication 9345.0-05I. September 1991.

EPA (U.S. Environmental Protection Agency). 1991b. *Ecological Assessment of Superfund Sites: An Overview*. Office of Solid Waste and Emergency Response, Office of Emergency and Remedial Response, Hazardous Site Evaluation Division (OS-230). *ECO Updat*e, Intermittent Bulletin, Volume 1, Number 2. Publication 9345.0-05I. December 1991.

EPA (U.S. Environmental Protection Agency). 1992a. *Framework for Ecological Risk Assessment*. Risk Assessment Forum, Washington, DC. EPA/630R-92/001.

EPA (U.S. Environmental Protection Agency). 1992b. *The Role of Natural Resource Trustees in the Superfund Process. ECO Update,* Volume 1, Number 3.

EPA (U.S. Environmental Protection Agency). 1993a. *Wildlife Exposure Factors Handbook*. Volumes I and II. EPA/600/R-93/187a, EPA/600/R-93/187b. U.S. Environmental Protection Agency, Office of Research and Development.

EPA (U.S. Environmental Protection Agency). 1994a. *Using Toxicity Tests in Ecological Risk Assessment*. Office of Solid Waste and Emergency Response, Office of Emergency and Remedial Response, Hazardous Site Evaluation Division (OS-230). *ECO Update,* Intermittent Bulletin, Volume 2, Number 1. Publication 9345.0-05I, EPA 540-F-94-012, PB94-963303. September 1994.

EPA (U.S. Environmental Protection Agency). 1994b. *Catalogue of Standard Toxicity Tests for Ecological Risk Assessment*. Office of Solid Waste and Emergency Response, Office of Emergency and Remedial Response, Hazardous Site Evaluation Division (5204G). *ECO Update,* Intermittent Bulletin, Volume 2, Number 2. Publication 9345.0-05I, EPA 540-F-94-013, PB94-963304. September 1994.

EPA (U.S. Environmental Protection Agency). 1994c. *Field Studies for Ecological Risk Assessment*. Office of Solid Waste and Emergency Response, Office of Emergency and Remedial Response, Hazardous Site Evaluation Division (5204G). *ECO Update,* Intermittent Bulletin, Volume 2, Number 3. Publication 9345.0-05I. EPA 540-F-94-014, PB94-963305. September 1994.

EPA (U.S. Environmental Protection Agency). 1994d. *Selecting and Using Reference Information in Superfund Ecological Risk Assessments*. Office of Solid Waste and Emergency Response, Office of Emergency and Remedial Response, Hazardous Site Evaluation Division (5204G). *ECO Update,* Intermittent Bulletin, Volume 2, Number 4. Publication 9345.0-10I, EPA 540-F-94-050, PB94-963319. September 1994.

EPA (U.S. Environmental Protection Agency). 1994e. *Ecological Risk Assessment Issue Papers.* EPA/630/R-94/009. November 1994.

EPA (U.S. Environmental Protection Agency). 1996. *Ecotox Thresholds*. Office of Solid Waste and Emergency Response, Office of Emergency and Remedial Response. *ECO Update,* Intermittent Bulletin, Volume 3, Number 2. Publication 9345.0-12FSI, EPA 540/F-95/038, PN95-963324. January 1996.

EPA (U.S. Environmental Protection Agency). 1997a. *Ecological Risk Assessment Guidance for Superfund: Process for Designing and Conducting Ecological Risk Assessments*. Interim Final. EPA 540-R-97-006. U.S. Environmental Protection Agency, Environmental Response Team.

EPA (U.S. Environmental Protection Agency). 1997b. *Guiding Principles for Monte Carlo Analysis*. EPA/63C/R-97/001. March 1997.

EPA (U.S. Environmental Protection Agency). 1998. *Guidelines for Ecological Risk Assessment*. Risk Assessment Forum, Washington, DC. EPA/630/R-95/002F.

EPA (U.S. Environmental Protection Agency). 1999. *Ecological Risk Assessment and Risk Management Principles for Superfund Sites.*

EPA (U.S. Environmental Protection Agency). 2001. *Risk Assessment Guidance for Superfund (RAGS)*, Volume 3, *Part A – Process for Conducting Probabilistic Risk Assessment.* EPA 540-R-02-002. December 2001.

Harza. 2001. *Woods Pond Dam: Structural Integrity Assessment*. Prepared for General Electric Company, Pittsfield, MA.

HEC (Harrington Engineering and Construction, Inc.). 1996. *Report on the Preliminary Investigation of Corrective Measures for Housatonic River and Silver Lake Sediment.* Prepared for General Electric Company.

MDEP (Massachusetts Department of Environmental Protection). 1996. *Guidance for Disposal Site Characterization in Support of the Massachusetts Contingency Plan*. Chapter 9, Method 3, "Environmental Risk Characterization," Interim Final Policy. MDEP, Bureau of Waste Site Cleanup, and Office of Research and Standards. April 1996.

O'Brien & Gere Engineers, Inc. 1995. *Phase I Report (MCP)/Current Assessment Summary, Hill 78 Area/Area 2*.

Stewart Laboratories, Inc. 1982. *Housatonic River Study 1980 and 1982.* Volumes I and II.

TechLaw (TechLaw, Inc.). 1998. *Preliminary Report: Wetland Characterization and Function-Value Assessment, Housatonic River from Newell Street to Woods Pond.* 4 May 1998.

United States of America, State of Connecticut, and Commonwealth of Massachusetts, Plaintiffs vs. General Electric Company, Defendant. 1999. *Consent Decree-Main Document and Appendices A through W.* October 1999.

WESTON (Roy F. Weston, Inc.). 1998. *Upper Reach-Housatonic River Ecological Risk Assessment.* Prepared for U.S. Environmental Protection Agency.

WESTON (Roy F. Weston, Inc.). 2000. *Supplemental Investigation Work Plan for the Lower Housatonic River*. Prepared for U.S. Army Corps of Engineers and U.S. Environmental Protection Agency. 22 February 2000. DCN GEP2-020900-AAME.

WESTON (Roy F. Weston, Inc.). 2001a. *Final Quality Assurance Project Plan*, Vol. I – Text, Vol. II – Appendix A, Vol. IIa – Appendix A cont'd., Vol. IV – Appendix E & F. Prepared for U.S Army Corps of Engineers and U.S. Environmental Protection Agency. DCN GE-021601-AAHM.

WESTON (Roy F. Weston, Inc.). 2001b. *Final Field Sampling Plan*. Prepared for U.S. Army Corps of Engineers and U.S. Environmental Protection Agency. DCN GE-053001-AAMA.

WESTON (Weston Solutions, Inc.). 2002. *Rest of River Site Investigation Data Report*. Prepared for U.S. Army Corps of Engineers and U.S. Environmental Protection Agency. DCN GE-080202-ABDK.

2. PROBLEM FORMULATION

Problem Formulation Highlights

The problem formulation establishes the goals, scope, and focus of the baseline ecological risk assessment (ERA). It is a process for generating and evaluating preliminary hypotheses about why ecological effects have occurred, or may occur, as a result of human activity. The problem formulation includes discussions of the following topics:

- Identification and sources of stressors.
- Determination of contaminants of potential concern (COPCs).
- Fate and transport of contaminant stressors.
- Contaminant effects on receptors.
- Site conceptual model.
- Assessment and measurement endpoints.
- Weight-of-evidence (WOE) approach.

2.1 OVERVIEW

Problem formulation, the planning phase of the ecological ERA, plays an important role in the development and direction of the risk assessment. It builds on and refines the screening-level problem formulation and, with input from stakeholders and other parties, shapes the analysis of ecological issues of concern at a site (EPA 1997). Problem formulation results in three products:

- Conceptual model(s).
- Assessment and measurement endpoints.
- Analysis plan.

This section describes the process that was followed in developing and refining the problem formulation phase of this ERA.

The screening-level problem formulation provides initial guidance to the risk assessors and managers by providing a preliminary look at potential issues of concern (see Figure 1.5-1). The screening-level problem formulation describes: (1) the environmental setting and contaminants known or suspected at the site; (2) contaminant fate and transport mechanisms; (3) ecotoxicity mechanisms and receptor categories of concern; (4) exposure pathways from contaminant sources to receptors; and (5) the results of a screening of conservative ecotoxicity values. Subsequently, the problem formulation is expanded and refined as data collection and analysis proceed. The ERA performed for the Upper Reach of the Housatonic River (WESTON 1998), with the ERA Work Plan developed by GE during the previous RCRA process (ChemRisk 1997), together fulfill the functional requirements of the screening problem formulation.

A detailed ecological characterization of the site was conducted (and subsequently refined) to expand on the information used in the screening-level problem formulation and to refine the initial conceptual model for the site. The final ecological characterization is summarized in Section 2.2 and presented in its entirety as Appendix A. The objective of the ecological characterization was to characterize the ecosystems within the Housatonic River watershed, including both plant and animal communities, with a focus on the Primary Study Area (PSA). Table 2.1-1 summarizes the specific ecological characterization surveys performed to characterize the ecosystems potentially at risk, as well as the specific survey objective(s), and references to the SI Work Plan appendix containing the detailed standard operating procedure (SOP) for each survey. This information was then used as input to the problem formulation.

In Section 2.3, the sources, concentrations, and distribution of contaminants in the study area are discussed. Contaminants of potential concern (COPCs) identified in the screening-level problem formulation were re-examined to determine whether they should be retained in the Pre-ERA screen for the ERA (Section 2.4). The availability of new data, information, or changes in assumptions can alter the results of the preliminary screening. Lack of data was not a reason alone to eliminate a potential contaminant, rather best professional judgment was used, and discussion regarding the uncertainty surrounding the decision is presented.

Table 2.1-1

Surveys Conducted for Ecosystem Characterization and Their Specific Objective(s)

Survey	Specific Objective(s)	SI Work Plan Appendix
Rare Plants and Natural Communities	Determine the potential rare, threatened, or endangered plants or animals occurring within the study area. Determine the presence and areal extent of habitats capable of supporting special status species potentially occurring within the study area. Determine the presence and areal extent of exemplary natural communities within the study area.	A.6
Dragonfly	Determine species of dragonflies present in the study area, with particular attention to rare species.	A.7
Mussel	Determine historical and current distribution within and upstream of the study area. Identify potential mussel hosts. Identify wildlife species that prey upon mussels.	A.8
Reptile and Amphibian Use	Estimate amphibian and reptile species richness in the study area by habitat type. Sample larval amphibians in breeding habitats over a range of PCB concentrations. Determine chemical concentrations in herptiles incidentally succumbing during trapping. Note: The latter two objectives were intended for use in ERA exposure and effects characterization (see Section 7.3 of the SI Work Plan).	A.9
Raptors and Waterfowl	Identify raptors and waterfowl breeding in study area.	A.10
Forest Bird and Marsh and Wading Bird	Identify birds using the study area floodplain forests and scrub-shrub habitats. Identify birds using the study area wetland and aquatic habitats.	A.11
River Otter, Mink, and Bat	Determine if mink and otter are present in the study area and reference areas. Identify bat species present in the study area. Determine habitats bats use for feeding.	A.12

In the problem formulation, the fate and transport of contaminants in the ecosystem potentially at risk and the description of exposure pathways were expanded beyond those in the screening-level problem formulation. This was generally accomplished through the collection of data or other information (e.g., field studies, modeling results, observations) on the fate and transport of contaminants, the ecological setting and flora and fauna of the site, and the extent of contamination (Section 2.5). In addition, the potential effects and impacts associated with COPCs are described in the context of site-specific environmental conditions (Section 2.6).

The next step in the problem formulation was to establish the assessment endpoints for the study. Assessment endpoints are an "explicit expression of the environmental value that is to be protected" (EPA 1998). Specific assessment endpoints focus the ERA on the issues that are important at the site and identify the appropriate measurement endpoints required to address these endpoints. Potential adverse effects on local populations and communities, such as reproduction, growth and survival, or changes in community structure or function, respectively, were identified and described using measurement endpoints to quantify effects for the assessment endpoints (Section 2.8). The identification of assessment and measurement endpoints and the exposure pathway analysis were used to refine the conceptual model for the site (Section 2.7). The intent of the conceptual model was, through the iterative process described above, to develop a thorough understanding and description of the site in a systematic and representative manner, and to identify important data or information gaps.

The problem formulation culminates in a scientific/management decision point (SMDP). A SMDP is an agreement between the risk manager and risk assessor on the assessment endpoints selected, exposure pathways, and questions presented in the conceptual model (EPA 1997).

The initial problem formulation for this ERA focused on the area within the 10-year floodplain of the Housatonic River extending from the confluence of the East and West Branches of the river to and including Woods Pond (Figures 1.4-2 and 1.4-3). This area, referred to as the Primary Study Area (PSA), is downstream from the source of COPCs from the GE facility, as well as the area where cleanup activity (including river sediment, bank soil, upland soil, and groundwater) is currently in progress, and includes the river sediment and floodplain soil where a majority of the PCBs are located, as indicated by the historical data and the evaluation of the recent EPA data, and summarized in the RCRA Facility Investigation Report (RFI) (BBL and QEA 2003). The RFI states that most of the PCB mass in the Housatonic River and floodplain downstream of the GE facility is in the PSA.

In addition, risks are also evaluated for the portion of the river downstream of Woods Pond, MA, to the point of tidal influence downstream of the Derby Dam, approximately 13 miles (21 km) from Long Island Sound and 128 miles downstream from the PSA, but using a less quantitative approach than that used for the PSA and for a focused set of endpoints (see Section 2.4, and Appendix A).

2.2 PHYSICAL AND ECOLOGICAL CHARACTERIZATION OF THE HOUSATONIC RIVER

2.2.1 Physical Characteristics of the Housatonic River Basin

The Housatonic River is located in Berkshire County, MA, and western Litchfield, eastern Fairfield, and western New Haven Counties, CT. The river flows approximately 166 miles (240 km) from the headwaters above Dalton, MA, to Long Island Sound, and drains an area of approximately 1,950 square miles in Massachusetts, New York, and Connecticut (BBL and QEA 2003).

For much of its path through Berkshire County, MA, the river lies in a wide alluvial plain called the Central Valley (Weatherbee 1996). The Central Valley is bounded to the east by the Berkshire Plateau, a southern extension of Vermont’s Green Mountains, and to the west by the Taconic Range, extending from Vermont to New York. In Connecticut, this same alluvial valley is called the Marble Valley. East of the valley, the Berkshire Plateau from Massachusetts continues southward and is called the Litchfield Hills Plateau.

In general, the plateaus and mountains bounding the river valley are typified by rounded hills and mountains draped with glacial deposits, and relatively narrow, steep-sided valleys cut into the hills by streams and rivers. The principal bedrock underlying much of the river basin is marble formed during the Devonian period, approximately 350 to 400 million years ago. Because of the prevalence of marble, the Housatonic River basin exhibits characteristics different from most other river systems in the northeastern United States. In particular, soil and water pH in the valley are high (7.9 to 8.3) and the groundwater contains high concentrations of calcium and magnesium (Harris 1997; Olcott 1995).

The area has a continental climate, similar to the rest of interior New England, characterized by cold winters and hot summers. In Stockbridge, MA, near the northern end of the study area, average annual temperature was 8 ºC, and average daily July and January temperatures were 20 and -6 ºC, respectively, for the period between 1951 and 1974. At Cornwall, CT, at approximately the midpoint of the watershed, average annual temperature was 9 ºC, and average daily July and January temperatures were 21 and -4 ºC, respectively. At Danbury, CT, nearer the southern end of the study area, the average annual temperature was 10 ºC, and average daily July and January temperatures were 22 and –3 ºC (SCS 1970, 1981, 1988). The number of frost-free days (growing season) at those locations ranges from 103 to 183 days. Moisture supply usually exceeds evaporation, except during periods of drought. Average total rainfall is 43 inches (110 cm) in Berkshire County, increases slightly southward to 45 inches (114 cm) in Litchfield County, and 47 inches (119 cm) in Fairfield and New Haven Counties (SCS 1970, 1979, 1981, 1988), and is evenly distributed throughout the year. Conversely, average total snowfall for these counties decreases markedly north to south and is 71, 61, 39, and 32 inches (180, 155, 99, and 81 cm), respectively (SCS 1970, 1979, 1981, 1988).

2.2.2 Ecological Characterization of the Study Area

For the purposes of the EPA Supplemental Investigation, the Housatonic River was divided into 17 reaches from the headwaters to Long Island Sound, with some reaches further subdivided. Reaches 1 to 17 were the focus of earlier ecological characterization studies (Chadwick & Associates, Inc. 1994). Reaches 5 and 6, comprising the PSA, were further investigated in detail by EPA from 1998 to 2000 (see Figure 1.1-2). As a result of that work, an ecological characterization of the PSA was prepared (see Appendix A.1, Ecological Characterization of the Housatonic River). Reaches 7 to 17 were also characterized, using aerial photograph interpretation and data provided by regional references and state natural resource agencies (see Appendix A.2, Ecological Characterization of the Housatonic River Downstream of Woods Pond).

2.2.2.1 Primary Study Area (PSA) Characteristics

Much of the PSA (approximately 770 acres) consists of state lands. Portions of the Housatonic River Valley State Wildlife Management Area, which totals 818 acres including land ranging from the confluence of the East and West Branches of the Housatonic River to Woods Pond (MassWildlife 2002), fall within the PSA. Approximately two-thirds of the State Wildlife Management Area is a continuous parcel from just north of New Lenox Road south to Woods Pond. Additional large parcels occur near the confluence (approximately 80 acres) and north of the Pittsfield wastewater treatment plant (WWTP) (approximately 120 acres). This area includes most of the forested habitat within the PSA. October Mountain State Forest, which comprises approximately 15,940 acres, occurs immediately adjacent to the eastern side of the lower PSA. This large area consists mainly of mature hardwood, softwood, and mixed forests. The City of Pittsfield owns a 45-acre parcel of land associated with the WWTP. Much of the land associated with the WWTP has been developed and includes buildings, paved parking areas, access roads, and maintained lawns. The remaining WWTP land near the river consists of transitional forests, shrub swamps, and shallow emergent marsh. Canoe Meadows, a Massachusetts Audubon Society property, is located just downstream of Holmes Road. This area contains forests and fields, as well as a large wetland complex.

A total of 18 natural communities occur within the PSA: 1 lacustrine community; 10 palustrine communities primarily associated with the Housatonic River floodplain and shoreline; 3 riverine communities either within the channel itself or draining into it; and 4 upland communities included within the 10-year floodplain[1] (Appendix A).

Portions of the PSA have been cleared for various purposes, primarily agriculture, residences, and various rights-of-way (e.g., roads, railroads, power lines). Agricultural development was the primary source of forest clearing within the floodplain. Several large wet meadows can be found in the PSA in which the species composition is influenced by past farming practices. Shrub swamps are common along pool and river channel borders, but are especially frequent as an

[1] Natural communities have been identified and classified according to Swain and Kearsley (2000). Weatherbee (1996) and Weatherbee and Crow (1992) were used to classify some river and lake systems.

intermediate successional stage in areas where pasture is reverting to forested floodplain. Even within some transitional floodplain forests, it is clear from the subcanopy species present that these areas were farmed in the past century. For example, dotted hawthorn routinely colonizes regenerating pastureland, and survives in the subcanopy of floodplain forests for some time after the tree stratum has returned to the site.

Significant portions of the PSA are open wetlands and riverine systems dominated by submersed, floating-leaved, and emergent herbaceous vegetation. Riverine point bars and beaches occur occasionally along the river, primarily near bends in the river channel. Mud flats of limited size begin to appear later in the season as the water level declines and exposes previously inundated sediment. Deep emergent marshes, which are usually inundated through the season and vegetated by robust herbs, are frequent along the river channel and backwater edges (Figure 2.2-1). These areas become much more abundant south of New Lenox Road, where backwater sloughs, old oxbows, and cut-off channels are common due to the influence of Woods Pond. Shallow emergent marshes, which are areas with saturated soil or shallow water and lower herbs, are less common in the study area and most frequently observed within more permanent vernal pools.

2.2.2.2 Housatonic River Downstream of the PSA

The Housatonic River downstream of the PSA extends to Long Island Sound in Connecticut, encompassing Reaches 7 through 17 (see Figures 1.4-5 to 1.4-7). Reach 8 comprises the next significant impoundment downstream of Woods Pond, formed by Rising Pond Dam. Reach 17 is the tidal portion of the river downstream of the Derby Dam and was not included in the Rest of River investigations due to its tidal nature and a number of other sources of COPCs. The river valley in Connecticut becomes narrower with steep uplands flanking both sides, and the free-flowing reaches of the river flow over a harder, coarser substrate of limestone, quartz, and granite (HVA 2001). Because of the constricting valley walls, the floodplain becomes narrower than in much of Massachusetts. However, some localized areas of broader floodplain exist. In these areas, agricultural activities are the dominant land use.

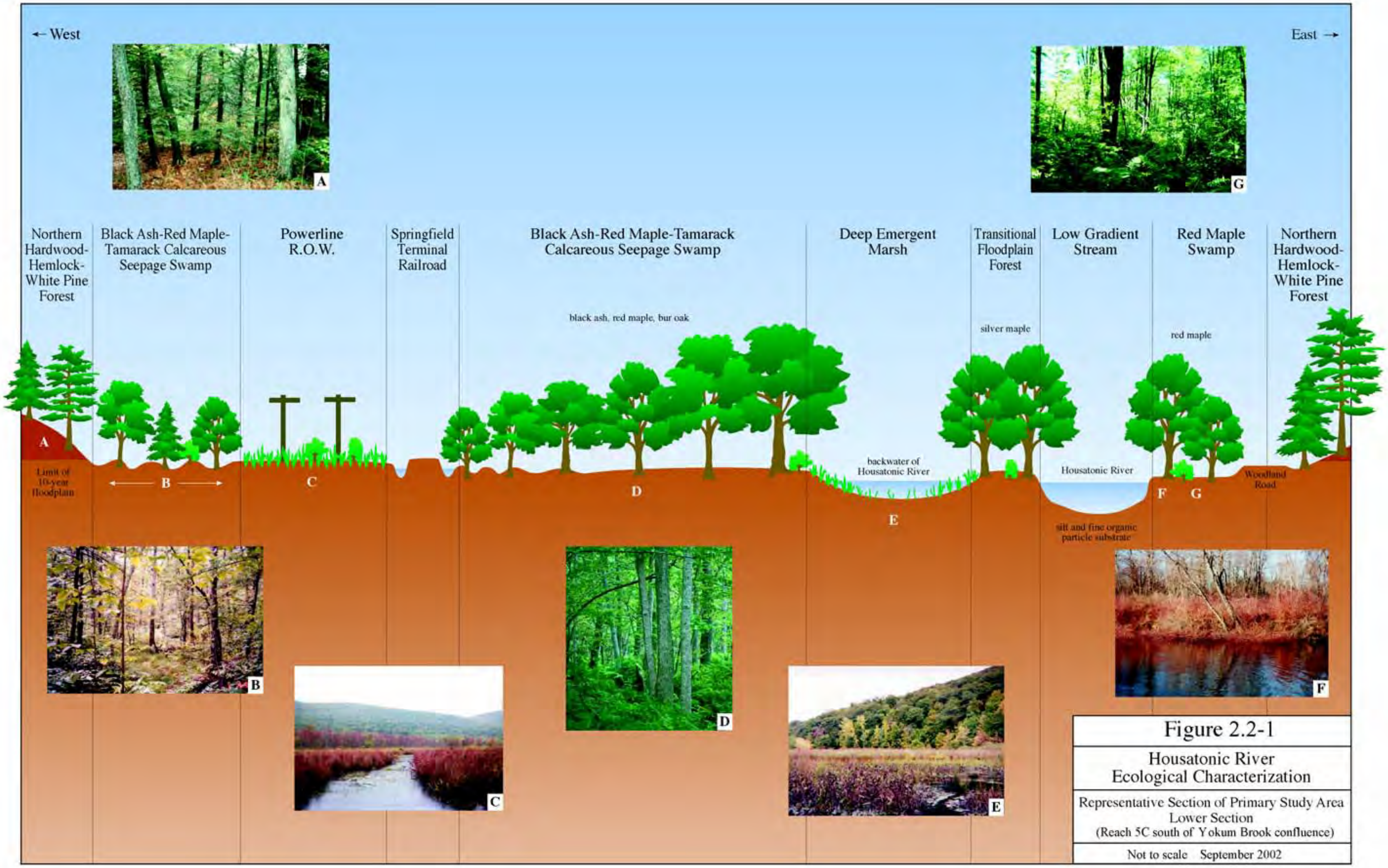


Figure 2.2-1 Housatonic River Ecological Characterization

The Housatonic River in Connecticut is affected by six dams, all of which create impoundments; five of the dams are used for electric power generation and all the impoundments are used for recreational purposes. These impoundments are medium to large, deep reservoirs of lacustrine habitat. Between these reservoirs, the free-flowing river is characterized as a medium-gradient stream with moderate to fast currents and pool, riffle, and run habitats.

A total of 28 natural communities occur in the Lower Housatonic River study area. Aquatic communities include moderately alkaline lakes and ponds in impounded reaches and low-, medium-, and high-gradient stream communities in free-flowing riverine areas. Palustrine communities include deep emergent marshes, shallow emergent marshes, wet meadows, mud flats, riverside seeps, calcareous sloping fens, shrub swamps, red maple swamps, black ash-red maple-tamarack calcareous seepage swamps, transitional floodplain forests, and high-terrace floodplain forests. Within the terrestrial systems, there are riverine point bars and beaches, high-energy riverbanks, riverside rock outcrops, calcareous rock cliff communities, northern hardwoods-hemlock-white pine forests, red oak–sugar maple transition forests, spruce-fir-northern hardwood forests, successional northern hardwoods, rich mesic forests, and cultural grasslands. Developed land uses include agricultural, residential, commercial, and public development, along with transportation.

2.3 IDENTIFICATION AND SOURCES OF STRESSORS

2.3.1 Contaminant Stressors

In this section, the sources, concentrations, and distribution of contaminants in the study area are identified. More detailed discussions of these topics, including information regarding the amounts, form, and conditions of release, are presented in the *Modeling Framework Design* (MFD) (WESTON 2004), the *Supplemental Investigation Work Plan for the Lower Housatonic River* (WESTON 2000), and the GE *RCRA Facility Investigation Report* (BBL and QEA 2003).

The GE facility in Pittsfield was the major handler of PCBs in western Massachusetts, and is the only known point source of PCBs in the PSA and downstream to the Derby Dam (approximately 13 miles from Long Island Sound). According to previous GE reports, from 1932 through 1977, releases of PCBs reached the wastewater and storm systems associated with the facility and were subsequently conveyed to the East Branch of the Housatonic River and to Silver Lake, or were released directly to these waters. In addition to the Housatonic River and Silver Lake, areas of the 254-acre GE facility, filled former river oxbows, neighboring commercial properties, Allendale School, and other properties or areas have become contaminated as a result of the GE operations.

Based on historical data and facility operations, the following contaminants have been found in the source areas and may have migrated to the Housatonic River:

- PCBs.
- Dioxins/furans.
- Semivolatile organics (e.g., bis(2-ethylhexyl)phthalate, methylphenol, phenol, and polycyclic aromatic hydrocarbon [PAHs]).
- Volatile organics (e.g., acetone, benzene, chlorobenzene, tetrachloroethene, trichloroethene, toluene, xylene, and other chlorinated hydrocarbons).
- Inorganics (e.g., lead and zinc).

According to the Source Area Characterization Report, there are five general categories of contaminant sources potentially impacting the river (WESTON 1998):

- Nonaqueous phase liquid (NAPL) discharge.
- Contaminated groundwater discharge.
- Riverbank soil/river sediment transport.
- Desorption/adsorption of residual riverbank and sediment contaminants.
- Direct stormwater discharge and surface runoff to the river.

The major areas of contamination designated in the Consent Decree for purposes of investigation and response are shown in Figure 1.2-1.

2.3.2 Physical and Biological Stressors

In addition to contaminant stressors (e.g., PCBs), physical and biological stressors can alter processes within ecosystems, affect habitat types, and ultimately influence natural communities by changing the diversity and abundance of species within habitat types. Physical stressors include structures and events such as dams, ice scour, floods, and droughts; biological stressors consist of changes in the biological components of a community, such as invasive plants out-competing native plants within riparian areas. Examples of physical and biological stressors that occur within the PSA and their subsequent effects on the natural communities are presented in the following paragraphs.

As mentioned in previous sections, the Housatonic River is a low-gradient river with a complex and diverse array of aquatic, riparian, and terrestrial habitats. These habitat types are largely determined by local conditions such as geology, climate, and soil, but are also influenced by a broad network of watershed processes such as hydrology and sediment transport. These processes can alter habitats by changing river morphology (e.g., eroding banks and creating pools) or by resetting high floodplain forest succession (e.g., uprooting overstory trees during a windstorm). Thus, the variety of habitat types within the PSA varies both spatially and temporally.

One of the natural watershed processes of the Housatonic River is for the river to meander laterally within its valley by eroding riverbanks on the outside of bends and depositing sediment on the inside of bends to create point bars. This typically occurs during or after high-flow events and can affect various animals within localized natural communities. For example, high flows undermine riverbanks and cause bank collapse, which may result in nest failures of bank-nesting birds such as the belted kingfisher (*Ceryle alcyon*). High flows also can flood out animals that den in riverbanks, such as muskrats (*Ondatra zibethicus*), causing mortality or forced relocation of burrows, or flood nests in the floodplain for species such as waterfowl.

Floods also increase river velocities and shear stresses that can cause the riverbed to scour, transport sediment, and then, as high flows subside, deposit sediment downstream and onto the floodplain. Movement of bed sediment can affect aquatic organisms including benthic macroinvertebrates and macrophytes that depend on specific substrate types. In addition, changes in macroinvertebrate communities can stress localized populations of other aquatic organisms that depend on these animals as food.

High winds, such as those that occur during hurricanes or nor'easters, are another physical stressor that occurs along the Housatonic River. These winds can cause blowdown of localized areas of the overstory floodplain vegetation (e.g., silver maple [*Acer saccharinum*] and black willow [*Salix nigra*]), which then resets forest succession to pioneer herbaceous species such as goldenrod (*Solidago*) that can tolerate increases in light and decreases in soil moisture.

Blown-down trees and those undermined by bank failure fall into the river and create complex habitat (large woody debris) for a host of aquatic organisms. Large woody debris alters localized channel processes and influences the development of natural communities. When a tree falls into a riffle, it can cause local bed scour and pool formation that then provides areas of refuge during high flows, hiding cover, rearing areas, and food sources for specific aquatic organisms. For example, pools typically have slower velocities and deeper water depths that are used by fish species such as northern pike (*Esox lucius),* largemouth bass (*Micropterus salmoides),* and common carp (*Cyprinus carpio*). Riffles and runs provide relatively faster moving water for different fish species (e.g., longnose dace [*Rhinichthys cataractae*]).

Biological stressors within the PSA influence natural communities by changing the distribution of species, which can then affect other components of the food chain. Such stressors include insect infestations, diseases and pathogens, and exotic or invasive species. Invasive plant species are common in the PSA and include species such as Asiatic bittersweet (*Celastrus orbiculata*), garlic mustard (*Alliaria petiolata*), common buckthorn (*Rhamnus cathartica*), Japanese knotweed (*Polygonum cuspidatum*), and purple loosestrife (*Lythrum salicaria*). These plants can invade natural communities and out-compete native plants, create localized monocultures, provide prolific seed source areas, and reduce species diversity. Such invasions can stress species that depend on specific native plants (e.g., some butterflies require specific plants for food).

Both physical and biological stressors within the PSA are influenced by anthropomorphic changes that have occurred within the watershed. These changes include channelization, riverbank armoring, dams, urban runoff, riparian area management, introduction of non-native and exotic species, invasive plants, business and residential development, watershed restoration projects, stormflow routing, bridge and railroad construction, wastewater transport and treatment facilities, agricultural clearing and ditching, and power lines.

2.4 OVERVIEW OF PRE-ERA

2.4.1 Introduction

The purpose of the Pre-ERA was to narrow the scope of the ERA by refining the list of contaminants to only those that pose potential risks to biota. The primary objectives of the Pre-ERA are as follows:

1. Identify COPCs other than PCBs for the PSA. (Downstream of the PSA, numerous potential sources of COPCs, other than the GE facility, exist along the river.)

2. Determine the downstream boundary beyond which PCBs from the GE facility pose a negligible risk to aquatic biota and wildlife.

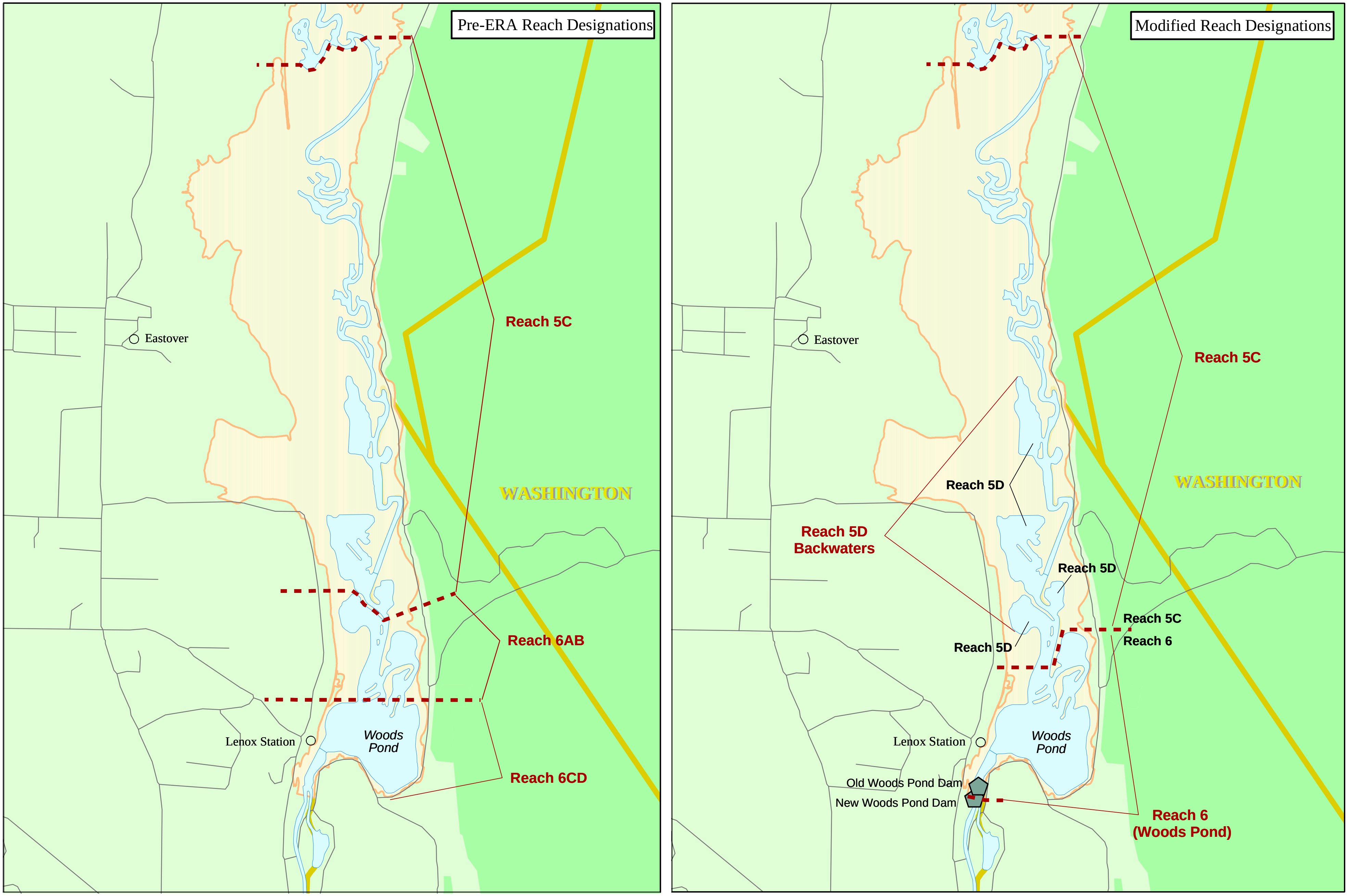
The following discussion provides a brief overview of the steps taken to identify COPCs for the PSA and to determine the downstream extent of the ERA. The COPCs from the Pre-ERA are further refined in the discussion of each individual assessment endpoint, resulting in endpoint-specific COCs, as appropriate. A more detailed presentation of the Pre-ERA approach and results is provided in Appendix B.

2.4.2 Data

Data sets were developed for the primary media of concern for the PSA, background areas, and for the area downstream of Woods Pond.

2.4.2.1 Primary Study Area

Data in the PSA were grouped by media (i.e., sediment, surface water, soil, and fish tissue), subreach, and geomorphological type. The reaches used for Pre-ERA evaluation are hydraulically similar sections of the Housatonic River; however, subsequent evaluation of the river in the PSA, as part of the Modeling Framework Design (WESTON 2004), resulted in further refinement of the Reach 5 and 6 designations to more accurately characterize existing flow regimes and the hydraulic influences of Woods Pond (Figure 2.4-1). Reaches 5A, 5B, and 5C did not change since the Pre-ERA was completed; however, the original Woods Pond backwater area (Reaches 6A and 6B) changed substantially and was renamed Reach 5D. Woods Pond proper (formerly Reaches 6C and 6D) was modified along its northern boundary and is



LEGEND:

- Town/City
- Roads
- Reach Division Line
- Housatonic River
- State Park
- Municipal Boundary
- 10-Year Floodplain

N W E S

0 0.1 0.2 0.4 0.6 0.8 Miles

0 0.15 0.3 0.6 0.9 1.2 Kilometers

Ecological Risk Assessment
GE/Housatonic River Site
Rest of River

FIGURE 2.4-1
COMPARISON OF PRE-ERA REACH DESIGNATIONS TO MODIFIED REACH DESIGNATIONS

currently designated as Reach 6. A detailed discussion of the current reach designation can be found in Section 2 of the Modeling Framework Design (WESTON 2004).

Geomorphological terrain descriptions (geomorph codes) were assigned to sediment, soil, and surface water samples collected by EPA. Each geomorph code represents a depositional or erosional feature or a terrain type that was formed by a specific geologic process (e.g., main channel, vernal pools, and side channels). The sediment and water data categories used for the Pre-ERA are shown in Table 2.4-1.

Floodplain and riverbank soil were evaluated separately for Reaches 5A, 5B, 5C, 6A, and 6B. Soil adjacent to Woods Pond (referred to as Reaches 6C and 6D in the Pre-ERA) was also evaluated. A more detailed description of the reach designations is provided in Section 1.4.

Fish tissue samples were grouped based on reach (5A, 5B, 5C, 6C, and 6D) and size class. Three size classes were evaluated: small (< 3 inches [7.6 cm]), medium (≥ 3 inches [7.6 cm] but < 12 inches [30.5 cm]), and large (≥ 12 inches [30.5 cm]).

2.4.2.2 Background Data

Background data are media-specific (i.e., sediment, surface water, soil, and fish tissue) chemistry data collected within the Housatonic River watershed that were not believed to be influenced by contamination directly originating from the GE Pittsfield facility. The objective of the determination of background concentrations was to identify what the media-specific chemical concentrations would be in the absence of releases from the GE facility, and to use this information in evaluating COPCs for the ERA.

Table 2.4-1

Sediment and Surface Water Data Categories

Medium/Reach	Geomorphological Terrain Type			
	Main Channel/ Aggrading Bars	Side Channels and Oxbows (SCOX)	Vernal Pools	Pond
Sediment				
5A	√	√	√	
5B	√	√	√	
5C	√	√	√	
6A, 6B	√	√	√	
6C, 6D				√
Surface Water				
5A	√	√	√	
5B	√		√	
5C	√	√	√	
6C, 6D				√

2.4.3 Primary Study Area (PSA) Evaluation and Results

The procedures used to screen potential COPCs were applied to the data groupings summarized above. Three progressive evaluation tiers were used to determine COPCs for the PSA.

- Tier I – A three-step process was used to establish the initial COPC list evaluating:
  - Frequency of detection.
  - Exceedance of benchmarks.
  - Comparisons to background concentrations.
- Tier II – A more detailed evaluation of frequency of exceedance of benchmarks was performed for contaminants that were not eliminated from further consideration in the Tier I evaluation.
- Tier III – The spatial extent of contamination, magnitude of benchmark exceedance, presence in other media, and mechanism of toxicity were evaluated subjectively for contaminants not removed in Tier I or Tier II.

Using the three-tier approach, a final list of COPCs was developed for each medium within a reach/terrain. The final generic COPC lists are presented in Tables 2.4-2 through 2.4-5. A detailed description of this approach is provided in Appendix B (Pre-ERA). Although several pesticides were retained as COPCs from the Pre-ERA process, a subsequent review of pesticide concentrations indicated, in general, relatively few detects and low concentrations. Therefore, it is believed that pesticides are generally not a site-related COPC. The COPCs were then further evaluated for each assessment endpoint; this is discussed in detail in the specific assessment endpoint appendices.

2.4.4 PCB Screening Evaluation Downstream of the PSA and Results

To determine the downstream limit for the ERA and potential areas of concern, the PCB concentrations measured in sediment at locations downstream of the PSA to Derby Dam were compared with available benchmarks. PCB concentrations in sediment (rather than another medium) were selected as an indicator of the spatial extent of potential ecological risk, because:

- Sediment serves as a reservoir of PCBs released from the GE facility.
- Sediment concentrations generally reflect the relative concentrations that could be expected in the floodplain.
- Exposure to PCBs in sediment is a major route of exposure for lower trophic levels (and subsequently higher trophic levels) in the aquatic food chain.
- Relatively extensive data on PCB concentrations in sediment are available.

The conservative benchmark used for this analysis was a threshold effect concentration (TEC) of 0.0598 mg PCB/kg sediment (MacDonald et al. 2000).

Hazard quotients (HQs) were developed using detected PCB concentrations or sample quantitation limits (SQLs), and the MacDonald TEC benchmark (0.0598 mg PCB/kg). After evaluation of the magnitude by which the benchmark was exceeded and the consistency and frequency of exceedances, the reaches from Woods Pond Dam to Derby Dam were retained for quantitative evaluation of risk from exposure to PCBs in the ERA.

Table 2.4-2

COPCs for Sediment Based on Tier III Evaluation

Chemical	5A			5B			5C			6AB			6CD
	MC & AB	SCOX	VP	MC & AB	SCOX	VP	MC & AB	SCOX	VP	MC & AB	SCOX	VP	Pond
Semivolatiles													
Dibenzofuran	X	---	---	---	---	---	---	---	---	---	---	---	---
PAHs													
Acenaphthene	X	---	---	---	---	---	X	---	---	---	---	---	---
Acenaphthylene	X	---	---	---	---	---	X	---	---	---	---	---	---
Anthracene	X	---	---	---	---	---	X	---	---	---	---	---	---
Benzo(a)anthracene	X	---	X	X	X	X	X	---	X	---	---	---	---
Benzo(b)fluoranthene	X	X	X	X	X	X	X	---	X	X	---	---	X
Benzo(k)fluoranthene	X	---	X	---	---	---	X	---	X	---	---	---	---
Benzo(g,h,I)perylene	X	X	X	X	X	X	X	---	X	X	---	---	X
Benzo(a)pyrene	X	---	X	X	X	X	X	---	X	---	---	---	---
Chrysene	X	---	X	---	X	X	X	---	X	---	---	---	---
Dibenzo(a,h)anthracene	X	X	X	---	---	X	X	---	X	---	---	---	---
Fluoranthene	X	---	---	---	---	---	X	---	X	---	---	---	---
Fluorene	X	---	---	---	---	---	X	---	---	---	---	---	---
Indeno(1,2,3-CD)pyrene	X	X	X	X	X	X	X	---	X	X	---	---	X
Naphthalene	X	---	---	---	---	---	X	---	---	---	---	---	---
Phenanthrene	X	---	---	X	X	---	X	---	X	---	---	---	---
Pyrene	X	---	---	X	X	X	X	---	X	---	---	---	---

Table 2.4-2

COPCs for Sediment Based on Tier III Evaluation (Continued)

Chemical	5A			5B			5C			6AB			6CD
	MC & AB	SCOX	VP	MC & AB	SCOX	VP	MC & AB	SCOX	VP	MC & AB	SCOX	VP	Pond
Dioxins/Furans	X	X	X	X	X	X	X	X	X	X	---	---	X
PCBs	X	X	X	X	X	X	X	X	X	X	X	X	X
Metals													
Antimony	---	---	---	---	---	---	X	---	X	---	---	---	---
Barium	---	---	---	---	---	---	---	---	X	X	---	---	---
Beryllium	---	---	---	---	---	---	---	X	X	---	---	---	---
Cadmium	---	---	---	---	---	---	X	---	X	X	---	---	---
Chromium	---	---	---	---	---	---	X	---	X	X	---	---	X
Copper	---	---	X	---	---	X	X	---	X	X	---	---	X
Lead	---	---	X	---	---	X	X	---	X	---	---	---	X
Mercury	---	---	X	---	X	X	X	X	X	X	---	---	X
Selenium	---	---	---	---	---	---	---	---	X	---	---	---	X
Silver	---	---	X	---	X	X	X	---	X	X	---	---	X
Tin	---	---	---	---	---	X	X	---	X	X	---	---	X

MC – main channel
AB – aggrading bars
SCOX – side channels and oxbows
VP – vernal pools
Pond – Woods Pond

Note: Reach designations reflect previous reach boundaries; it is assumed that the revised reach designations do not impact reach-specific COPCs determined in Appendix B.

Table 2.4-3

COPCs for Surface Water Based on Tier III Evaluation

Chemical	Reach/Geomorphological Type						
	5A		5B		5C		6CD
	MC & AB	VP	MC & AB	VP	MC & AB	VP	Pond
Dioxins/Furans	X	X	X	X	X	X	X
PCBs	X	X	X	X	X	X	X

MC – main channel
AB – aggrading bars
VP – vernal pools
Pond – Woods Pond

Note: Reach designations reflect previous reach boundaries; it is assumed that the revised reach designations do not impact reach-specific COPCs determined in Appendix B.

Table 2.4-4

COPCs for Soil Based on Tier III Evaluation

Chemical	Reach/Geomorphological Type						
	5A		5B		5C		6CD
	Floodplain	Riverbank	Floodplain	Riverbank	Floodplain	Riverbank	Floodplain
Semivolatiles							
Dibenzofuran	X	X	X	X	---	---	---
PAHs							
Benzo(a)pyrene	---	X	---	---	---	---	---
Pyrene	---	X	---	---	---	---	---
Dioxins/Furans	X	X	X	X	X	X	X
PCBs	X	X	X	X	X	X	X
Metals							
Chromium	X	X	X	X	X	X	X
Lead	---	---	---	---	X	---	X
Mercury	---	X	X	X	X	X	X
Selenium	---	---	---	---	X	---	X

Note: Reach designations reflect previous reach boundaries; it is assumed that the revised reach designations do not impact reach-specific COPCs determined in Appendix B.

Table 2.4-5

COPCs for Fish Based on Tier III Evaluation

Chemical	Reach/Fish Size								
	5A			5BC			6CD		
	Small	Medium	Large	Small	Medium	Large	Small	Medium	Large
Pesticides									
4,4’-DDE	---	---	---	---	---	X	---	X	X
O,p’-DDT	X	X	X	X	X	X	X	X	X
4,4’-DDT	---	X	---	---	X	X	---	---	X
Heptachlor epoxide	X	X	X	---	---	X	X	X	X
Cis-Nonachlor	X	X	X	X	X	X	X	X	X
Trans-nonachlor	X	X	X	X	X	X	X	X	X
Oxychlordane	X	X	X	X	X	X	---	X	X
Dioxins/Furans	X	X	X	X	X	X	X	X	X
PCBs	X	X	X	X	X	X	X	X	X

Note: Reach designations reflect previous reach boundaries; it is assumed that the revised reach designations do not impact reach-specific COPCs determined in Appendix B.

2.5 FATE AND TRANSPORT OF CONTAMINANTS

Understanding the fate and transport characteristics of COPCs is a major component of the problem formulation phase of an ERA. Although other COPCs are present in the study area, the focus of the Rest of River evaluation is on PCBs, as well as dioxins/furans; therefore, the objectives of this discussion are to:

- Provide a general description of PCB fate and transport mechanisms (Section 2.5.1).
- Present a summary of PCB distribution within the Housatonic River floodplain (Section 2.5.2).
- Identify exposure pathways (Section 2.5.3).
- Discuss how PCB congener patterns change in environmental media (Section 2.5.4).
- Present a general overview of the fate and transport mechanisms of dioxins/furans (Section 2.5.5).

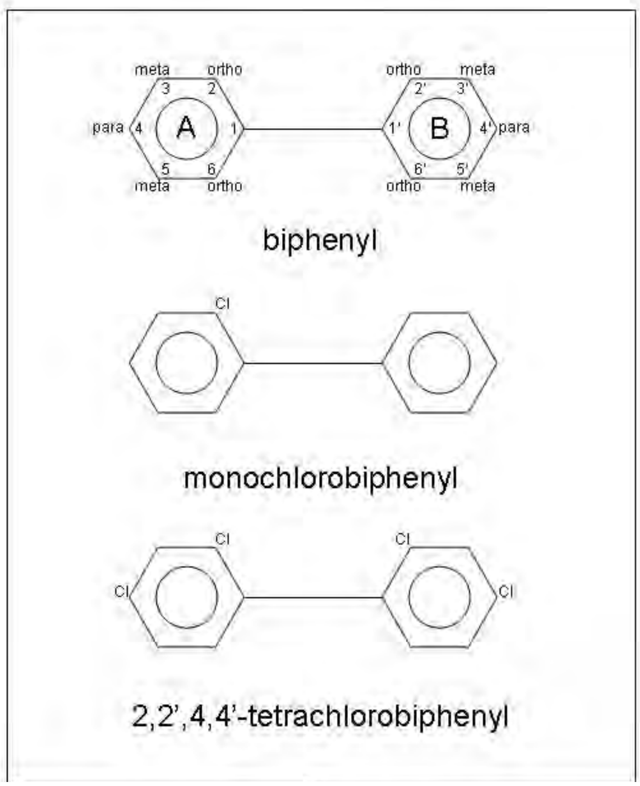
2.5.1 Fate and Transport of PCBs

Polychlorinated biphenyls (PCBs) are formed when hydrogen atoms on a biphenyl molecule are replaced by 1 to 10 chlorine atoms. First manufactured approximately 75 years ago, PCBs are extremely persistent contaminants that are now ubiquitous in the global ecosystem (Eisler 1986). There are 209 possible configurations of PCB molecules, based on the number and position of chlorine substitutions on the biphenyl ring; these individual PCB configurations are known as congeners. Although all possible congeners have been synthesized, only approximately 175 of the 209 congeners were included in the various commercial formulations. Groups of PCB congeners with similar numbers of substituted chlorine atoms are referred to as homologs, including: mono-, di-, tri-, tetra-, penta-, hexa-, hepta-, octa-, nona-, and decachlorobiphenyl (EPA 1996). Aroclors (Aroclor is a trade name of the Monsanto Company) are commercial mixtures of PCB congeners that were formulated to have specific physical properties, which are based, in general, on the overall amount of chlorine substitution (Figure 2.5-1).

The level of chlorination affects various physicochemical properties of the PCB molecule, such as the octanol/water partition coefficient (K_{ow}), solubility, vapor pressure, and Henry's Law constant. These properties affect processes such as volatilization, and partitioning to water, sediment, and floodplain soil. Similarly, the level of chlorination also controls (in part) biologically mediated processes such as biotransformation, uptake, and bioaccumulation (WESTON 2004). In general, more chlorinated PCBs have greater stability and environmental persistence (EPA 1996).

PCBs in the environment occur as mixtures of congeners that differ in composition from commercial mixtures because of partitioning, contaminant transformation, and preferential bioaccumulation over time (Aulerich et al. 1986; Hornshaw et al. 1983; EPA 1980). Some congeners are retained in sediment, soil, and biological tissue. Bioaccumulated PCBs appear to be more toxic than commercial PCBs (Aulerich et al. 1986; Hornshaw et al. 1983).

More detailed discussions on the fate and transport of PCBs can be found in the *Modeling Framework Design* (WESTON 2004).



Source: Adapted from Eisler 1986.

Figure 2.5-1 Biphenyl and Representative PCB Congeners

2.5.2 PCB Distribution by Media

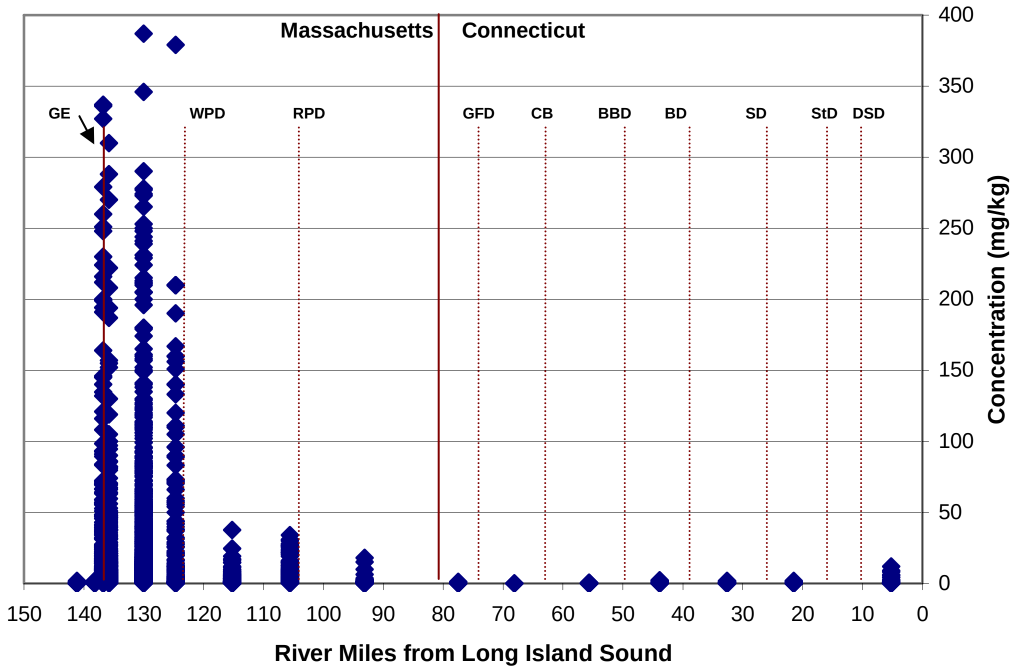
This section provides an overview of the distribution of PCBs in sediment, soil, surface water, and biota of the PSA. This section also presents a discussion of the sediment grain size analysis and the concentrations of organic carbon in the sediment, soil, and water samples from the PSA and their relationship with PCBs. Sediment and soil data used for this analysis included all samples collected by any organization between 1998 and 2002, a span of 5 years. Earlier data were not included to ensure that any potential temporal trends or the influence of different analytical methods in the data would not potentially mask current spatial patterns. The analysis of surface water included samples collected between 1996 and 2002. This slightly longer span of time was used because of the more robust data set that was available. A full presentation of the spatial and temporal trends is presented in the MFD (WESTON 2004) and the RFI report (BBL and QEA 2003).

The term *sediment* is defined for this study as solid material typically inundated under normal hydrologic conditions. Soil samples are defined as those samples collected from areas not typically inundated under normal hydrologic conditions. Sediment and soil samples were collected from across the PSA and classified by the geomorphic terrains (i.e., main river channel, floodplain, riverbanks, etc.) from which they were originally collected. The distribution of PCBs in the PSA and in Reaches 7, 8, and 9 between the PSA and the Massachusetts/Connecticut state line is illustrated in the stack bar figures (see Attachment 2.1).

2.5.2.1 Sediment

PCBs have been detected in sediment samples collected from all reaches of the Housatonic River from just upstream of the GE facility through the PSA and downstream in Massachusetts and in Connecticut. Figure 2.5-2 presents sediment PCB data for the entire Housatonic River (from the vicinity of the GE facility to the point where the river empties into Long Island Sound). Historically, over 7,500 sediment samples have been collected from the main channel of the river in Massachusetts and Connecticut; almost 5,000 samples since 1998 alone. The highest concentrations of PCBs have been detected in sediment adjacent to the GE facility (river mile 137; 9,411 mg/kg in a surficial sample) and continuing downstream to Woods Pond Dam (at river mile 124.37). Within the PSA, the highest PCB concentrations detected by EPA were 614 mg/kg in Reach 5A, 165 mg/kg in Reach 5B, 213 mg/kg in Reach 5C, and 668 mg/kg in Reach 6 (Woods Pond). Sediment samples collected prior to 1998 detected PCBs as high as 2,270 mg/kg in Reach 5A. PCBs have also been detected as deep as 6 to 8 ft below the riverbed surface throughout Reaches 5 and 6 (BBL and QEA 2003; WESTON 2004).

PCB concentrations in sediment decrease downstream of Woods Pond Dam in Reaches 7, 8, and 9, and decrease further in concentration through most of Connecticut (BBL and QEA 2003). An increase in PCBs was detected in the most downstream reach (Reach 17 – from the Derby Dam to Long Island Sound), attributable to other Superfund or designated hazardous waste sites located within that portion of the river.



Notes:

1. All data are plotted at the approximate mid-point of each reach, and represent samples collected from within the top 3 feet of the riverbed.
2. Total PCB concentrations above 400 mg/kg were not plotted.
3. Symbols represent significant features/names of reach boundaries: GE = General Electric facility; WPD = Woods Pond Dam; RPD = Rising Pond Dam; GFD = Great Falls Dam; CB = Cornwall Bridge; BBD = Bulls Bridge Dam; BD = Bleachery Dam; SD = Shepaug Dam; STD = Stevenson Dam; DSD = Derby Dam.

Figure 2.5-2 Distribution of tPCB Concentrations Detected in Sediment Samples from the GE Facility to Long Island Sound

The mean tPCB concentrations in sediment samples are plotted by reach in Figure 2.5-3. The data have been presented on a log scale to capture the mean tPCB concentration of 393 mg/kg in Reach 3 (adjacent to the GE facility). For the purposes of calculating statistics for this exercise, values for non-detects were treated as half the reported detection limit. Likewise, the most commonly reported detection limit of 0.5 mg/kg is shown on the figure for comparison.

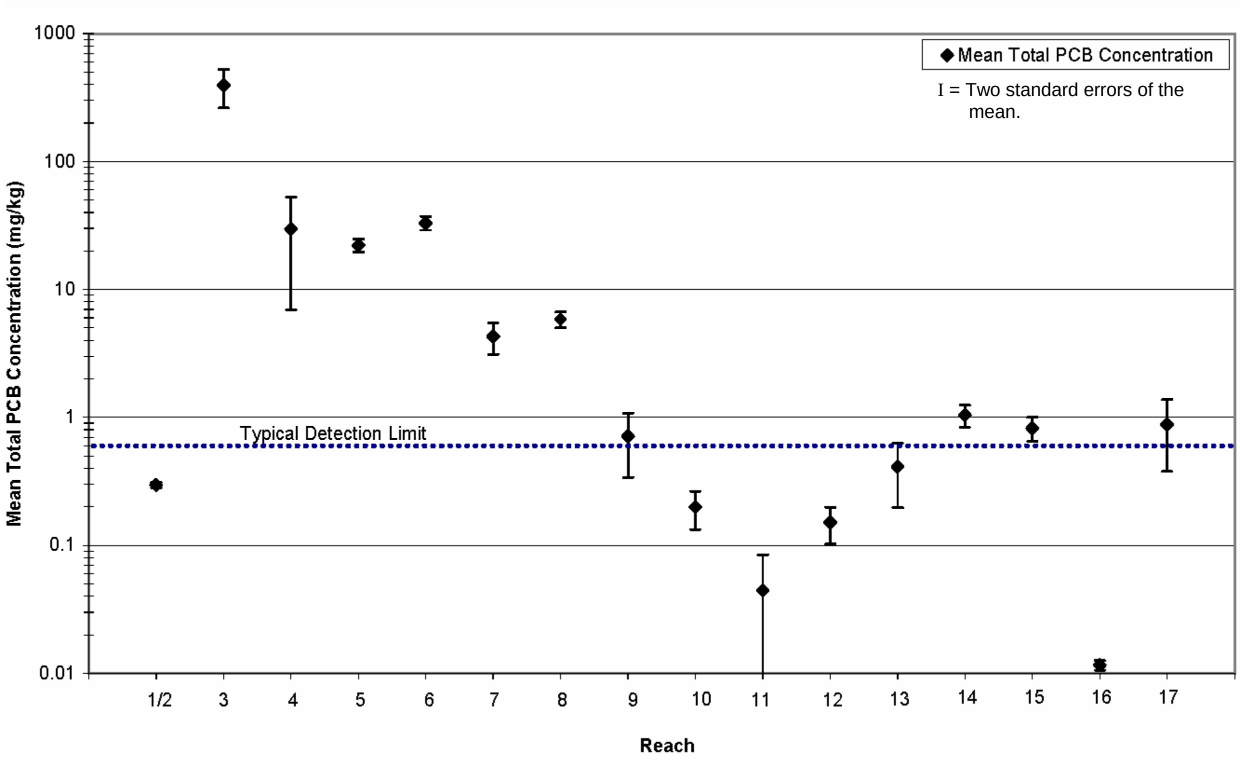


Figure 2.5-3 Mean Total Sediment PCB Concentrations by Reach

As shown in Figure 2.5-2, the mean tPCB concentrations are highest adjacent to the GE facility and on through the PSA to Woods Pond Dam (Reaches 3 to 6) and then generally decrease through the remaining reaches in Massachusetts and Connecticut. Many samples from Connecticut were non-detect, resulting in low (<0.5 mg/kg) mean concentrations of PCBs. Reach 11, and approximately half of the length of Reach 10, is shallow and fast-flowing with mostly a gravel to cobble stream bed where PCB-containing solids are not likely to be deposited, resulting in very few samples having PCBs. Reach 16 represents the last impoundment along the Housatonic River, and approximately 99% of the samples collected there were non-detect for PCBs.

2.5.2.2 Soil

Soil samples were collected from the floodplain and riverbanks along the Housatonic River within Massachusetts) (approximately 4,300 samples were collected by EPA, and 3,300 samples were collected by GE) in the reaches upstream of Woods Pond Dam. Additional samples (approximately 1,600 collected by EPA, and 200 collected by GE) were collected downstream of the PSA, and PCBs have been detected in floodplain soil in all reaches of the Housatonic River from the GE facility to the Massachusetts/Connecticut state line. The highest concentration of tPCBs detected in floodplain soil was 907 mg/kg from soil in Reach 5C above Woods Pond. Conversely, the highest tPCBs detected in riverbank soil were adjacent to the GE facility and just downstream, in Reaches 2 through 4 (between river miles 138 and 135).

Figure 2.5-4 presents the spatial distribution of the mean and median tPCB concentrations for all surficial (0 to 6 inches [0 to15 cm]) floodplain soil by reach for the portion of the river upstream of the Massachusetts/Connecticut state line, at which point the average PCB concentration in floodplain soil is less than 1 mg/kg. In addition, little floodplain exists within the Connecticut portion of the river; therefore, no samples were collected from those reaches. Mean tPCB concentrations are broadly similar within Reaches 4, 5, and 6, averaging slightly more than 15 mg/kg, and then on average decreasing by an order of magnitude in Reaches 7, 8, and 9. However, localized areas of higher concentrations are found in Reach 7.

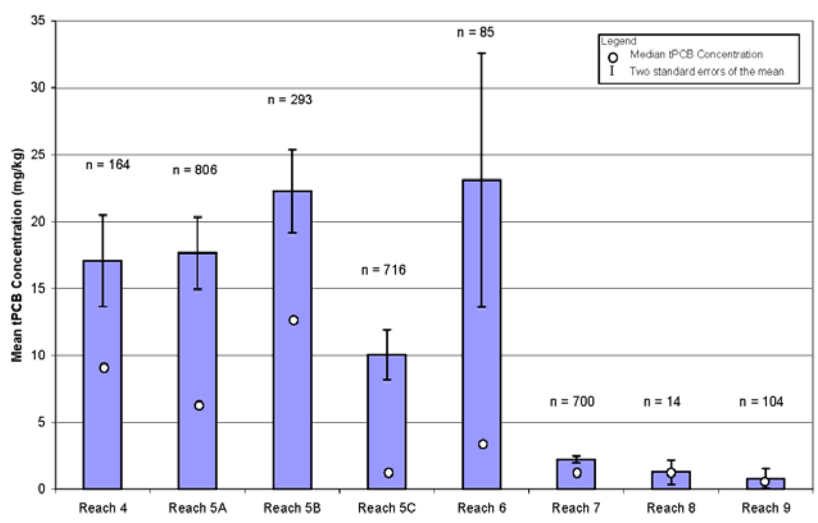


Figure 2.5-4 Mean Total Surficial Soil PCB Concentrations at Floodplain Locations by Reach

Figures 2.5-5 through 2.5-11 display the spatially weighted floodplain tPCB concentrations in the PSA using the inverse distance weighting procedures described in Appendix C.3.

Riverbank soil PCB concentrations are broadly similar in concentration ranges to the floodplain soil and river sediment samples, being highest adjacent to the GE facility in Reach 3 and immediately downstream in Reach 4, and decreasing in concentration within Reaches 6 and 7.

2.5.2.3 Surface Water

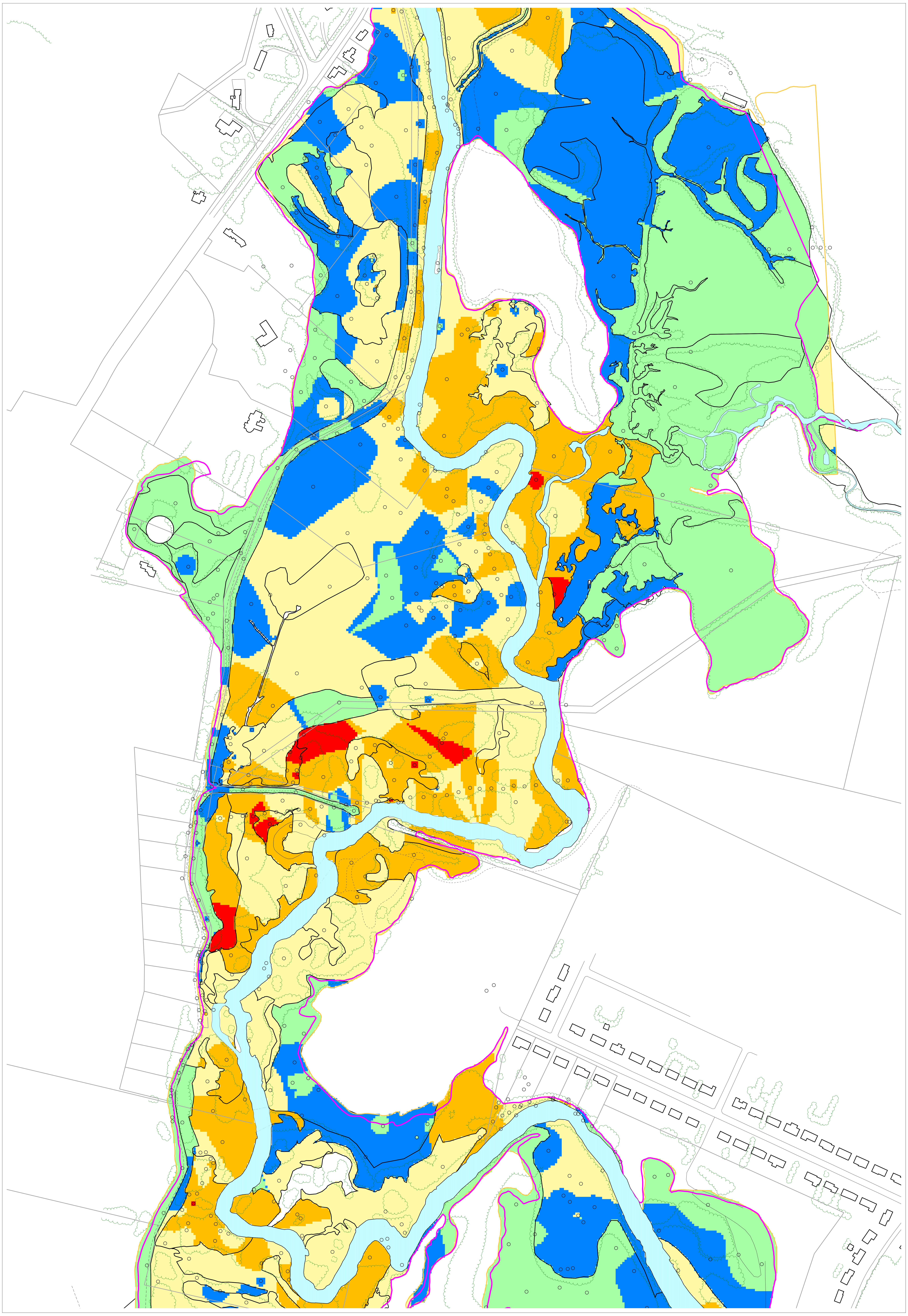
Sampling for PCBs in surface water was conducted during both low flow conditions and during higher or storm flow conditions. During lower flow conditions, PCB-contaminated sediment act as the primary source of PCBs in the water column through the processes of diffusion and groundwater advection through the sediment and associated porewater. During higher flows, the principal source of PCBs in the water column is from resuspended sediment, from both upstream and within the PSA.

Results for all of the surface water samples collected and analyzed for tPCBs since 1980 are presented in Figure 2.5-12 by river mile. In addition, Figures 2.5-12 and 2.5-13 identify the locations of major impoundment structures found along the Housatonic River from the GE facility to Long Island Sound.

- GE = General Electric facility.
- WPD = Woods Pond Dam.
- RPD = Rising Pond Dam.
- GF = Great Falls Dam.
- CB = Cornwall Bridge.
- BBD = Bulls Bridge Dam.
- RDD = Bleachery Dam.
- SD (River Mile 25) = Shepaug Dam.
- SD (River Mile 15) = Stevenson Dam.
- DSD = Derby Dam.

While the analysis of spatial patterns discussed below only used data from 1996 to 2002, all historical data were plotted to show the full set of results, because most of the data downstream of the PSA, especially in Connecticut, were collected prior to 1996. Figure 2.5-13 presents only the surface water data collected since 1996. As indicated in this figure, tPCB concentrations increase at the GE facility and then decrease downstream through to Rising Pond Dam. Concentrations of tPCBs continue to decrease by an order of magnitude downstream of Rising Pond Dam and into the Connecticut portion of the river. Most (approximately 80%) of the samples collected in Connecticut, both before and since 1996, were non-detect. Within Reaches 5 and 6, PCBs were detected at all surface water sampling locations and were fairly constant in concentrations across the study area. More than half of the samples collected from Reaches 5 and 6 contained tPCBs above the chronic ambient water quality criterion (cAWQC) for protection of aquatic life of 0.014 μg/L.

| d:\housatonic\ecora\swa_d1_hdl_2_7tiles_era.apr | layout - TILE 1/7 | d:\housatonic\ecora\exports\in\idw_era_2_5-5.eps | 11:26 AM, 4/9/2003 |



LEGEND:

	Super Habitat Boundary
	I ppm PCB Isopleth (Historic)
	10-Year Floodplain

Inverse Distance Weighting PCB Concentration (mg/kg)	Total PCB (tPCB) (mg/kg) (0-6" depth)
< 1	< 1
1 - 5	1 - 5
5 - 25	5 - 25
25 - 100	25 - 100
> 100	> 100
No Data	

N

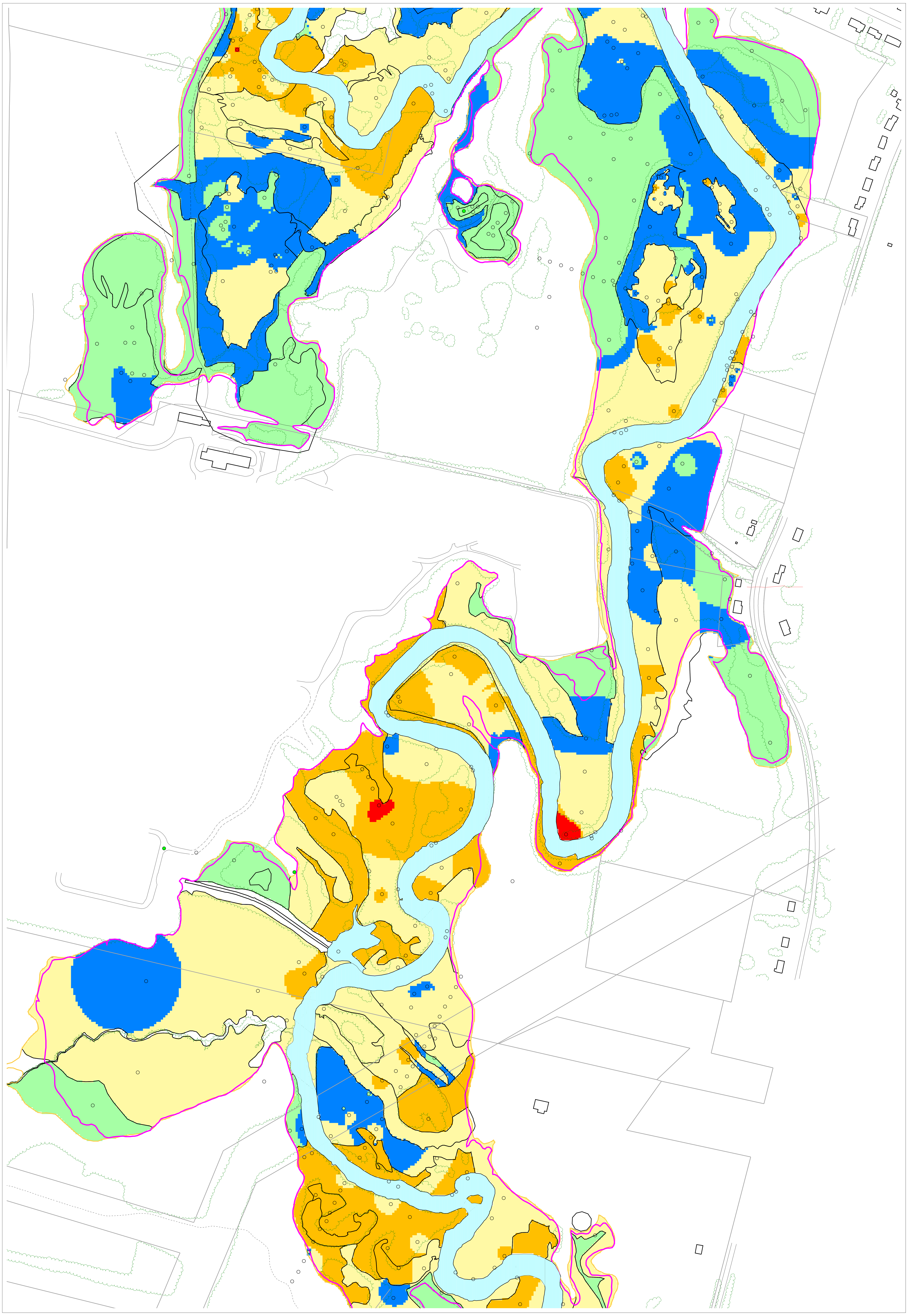
Scale in Feet

200 0 200 400 600

Ecological Risk Assessment
GE/Housatonic River Site,
Rest of River

FIGURE 2.5-6
SPATIALLY WEIGHTED tPCB CONCENTRATIONS IN FLOODPLAIN SOIL IN THE PRIMARY STUDY AREA (TILE 2 OF 7)

| d:\housatonic\ecora\swa_d1_hdl_2_7tiles_era.apr | layout - TILE 2/7 | d:\housatonic\ecora\exports\in\idw_era_2_5-6.eps | 11:28 AM, 4/9/2003 |



LEGEND:

Super Habitat Boundary

I ppm PCB Isopleth (Historic)

10-Year Floodplain

Inverse Distance Weighting PCB Concentration (mg/kg)	Total PCB (tPCB) (mg/kg) (0-6" depth)
< 1	< 1
1 - 5	1 - 5
5 - 25	5 - 25
25 - 100	25 - 100
> 100	> 100
No Data	

N

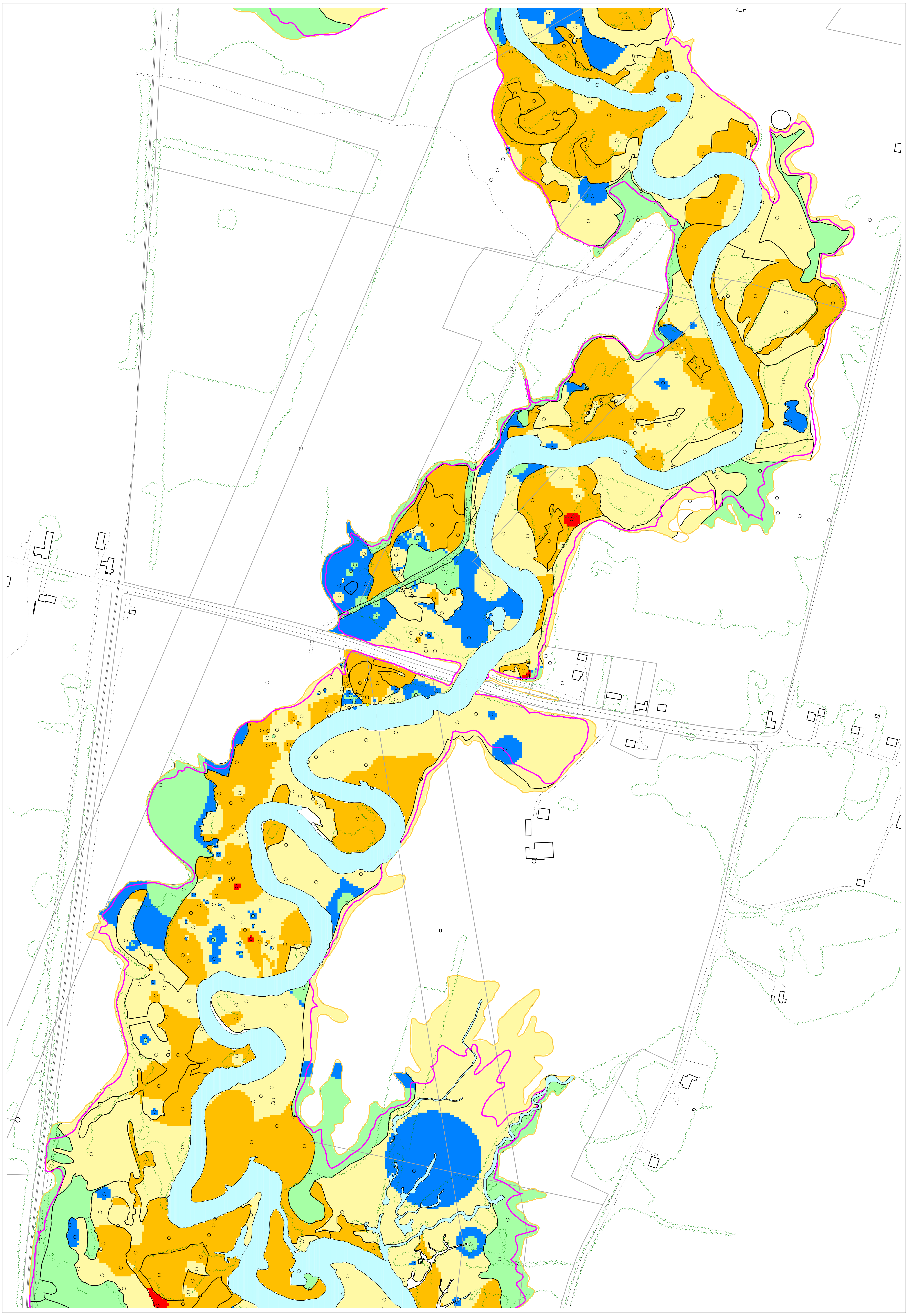
Scale in Feet

200 0 200 400 600

Ecological Risk Assessment
GE/Housatonic River Site,
Rest of River

FIGURE 2.5-7
SPATIALLY WEIGHTED tPCB CONCENTRATIONS IN FLOODPLAIN SOIL IN THE PRIMARY STUDY AREA (TILE 3 OF 7)

| d:\housatonic\ecora\swa_d1_hdl_2_7tiles_era.apr | layout - TILE 3/7 | d:\housatonic\ecora\exports\in\idw_era_2_5-7.eps | 11:29 AM, 4/9/2003 |



LEGEND:

	Inverse Distance Weighting PCB Concentration (mg/kg)	Total PCB (tPCB) (mg/kg) (0-6" depth)
Super Habitat Boundary	< 1	< 1
I ppm PCB Isopleth (Historic)	1 - 5	1 - 5
10-Year Floodplain	5 - 25	5 - 25
	25 - 100	25 - 100
	> 100	> 100
	No Data	

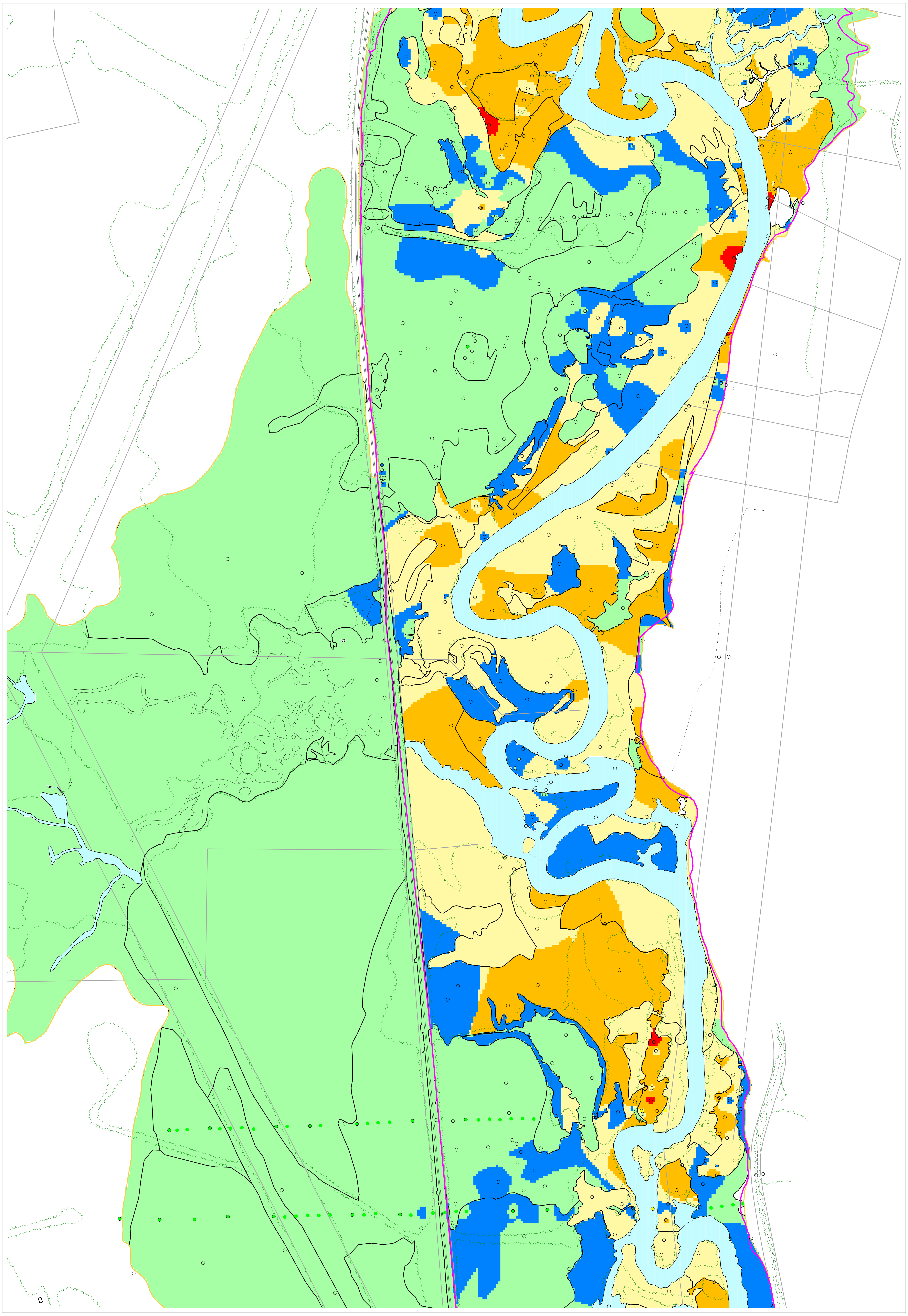
N

Scale in Feet

200 0 200 400 600

Ecological Risk Assessment
GE/Housatonic River Site,
Rest of River

FIGURE 2.5-8
SPATIALLY WEIGHTED tPCB CONCENTRATIONS IN FLOODPLAIN SOIL IN THE PRIMARY STUDY AREA (TILE 4 OF 7)



LEGEND:

	Inverse Distance Weighting PCB Concentration (mg/kg)	Total PCB (tPCB) (mg/kg) (0-6" depth)
Super Habitat Boundary	< 1	< 1
I ppm PCB Isopleth (Historic)	1 - 5	1 - 5
10-Year Floodplain	5 - 25	5 - 25
	25 - 100	25 - 100
	> 100	> 100
	No Data	

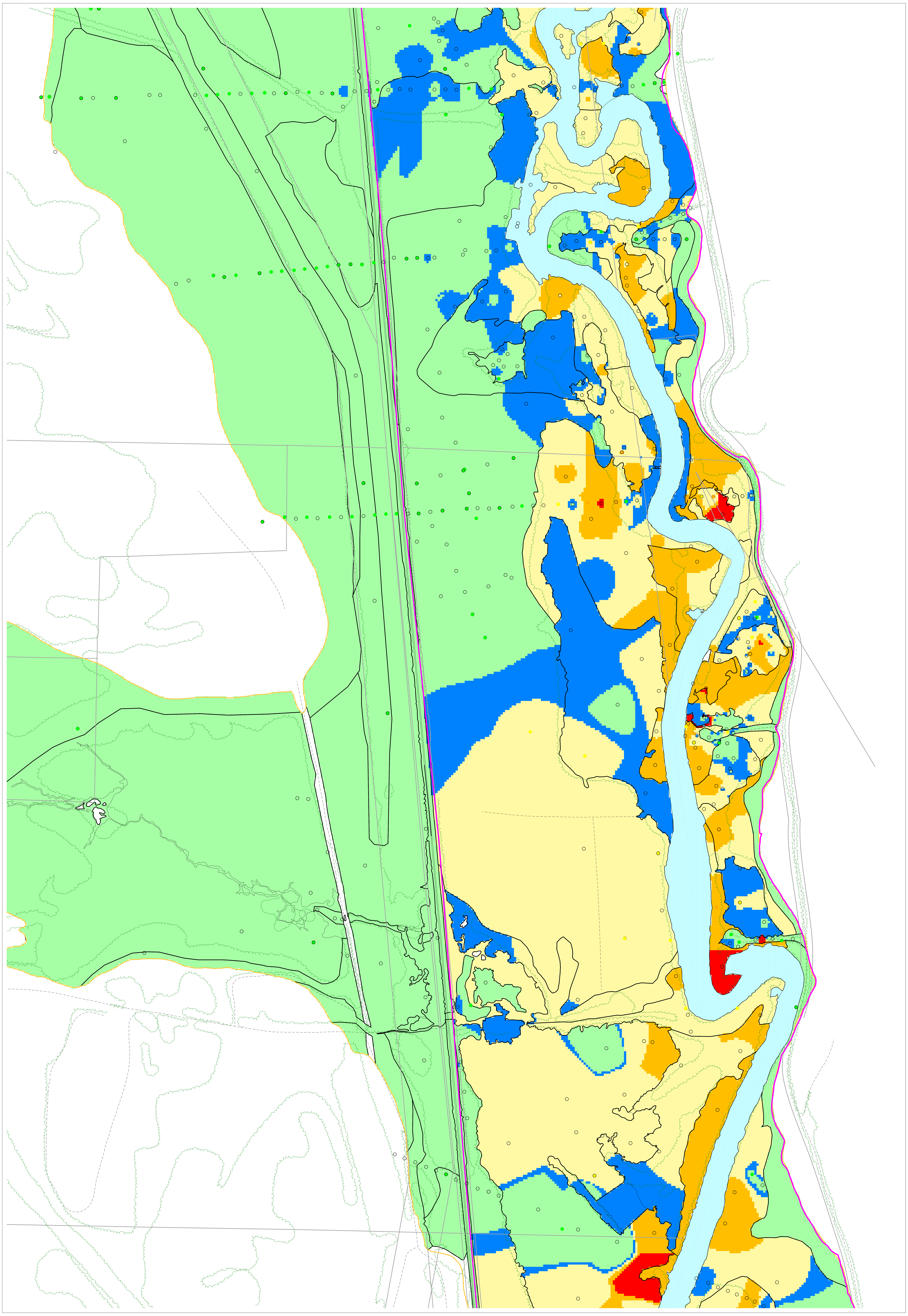
N

Scale in Feet

200 0 200 400 600

Ecological Risk Assessment
GE/Housatonic River Site,
Rest of River

FIGURE 2.5-9
SPATIALLY WEIGHTED tPCB CONCENTRATIONS IN FLOODPLAIN SOIL IN THE PRIMARY STUDY AREA (TILE 5 OF 7)



LEGEND:

	Inverse Distance Weighting PCB Concentration (mg/kg)	Total PCB (tPCB) (mg/kg) (0-6" depth)
Super Habitat Boundary	< 1	< 1
I ppm PCB Isopleth (Historic)	1 - 5	1 - 5
10-Year Floodplain	5 - 25	5 - 25
	25 - 100	25 - 100
	> 100	> 100
	No Data	

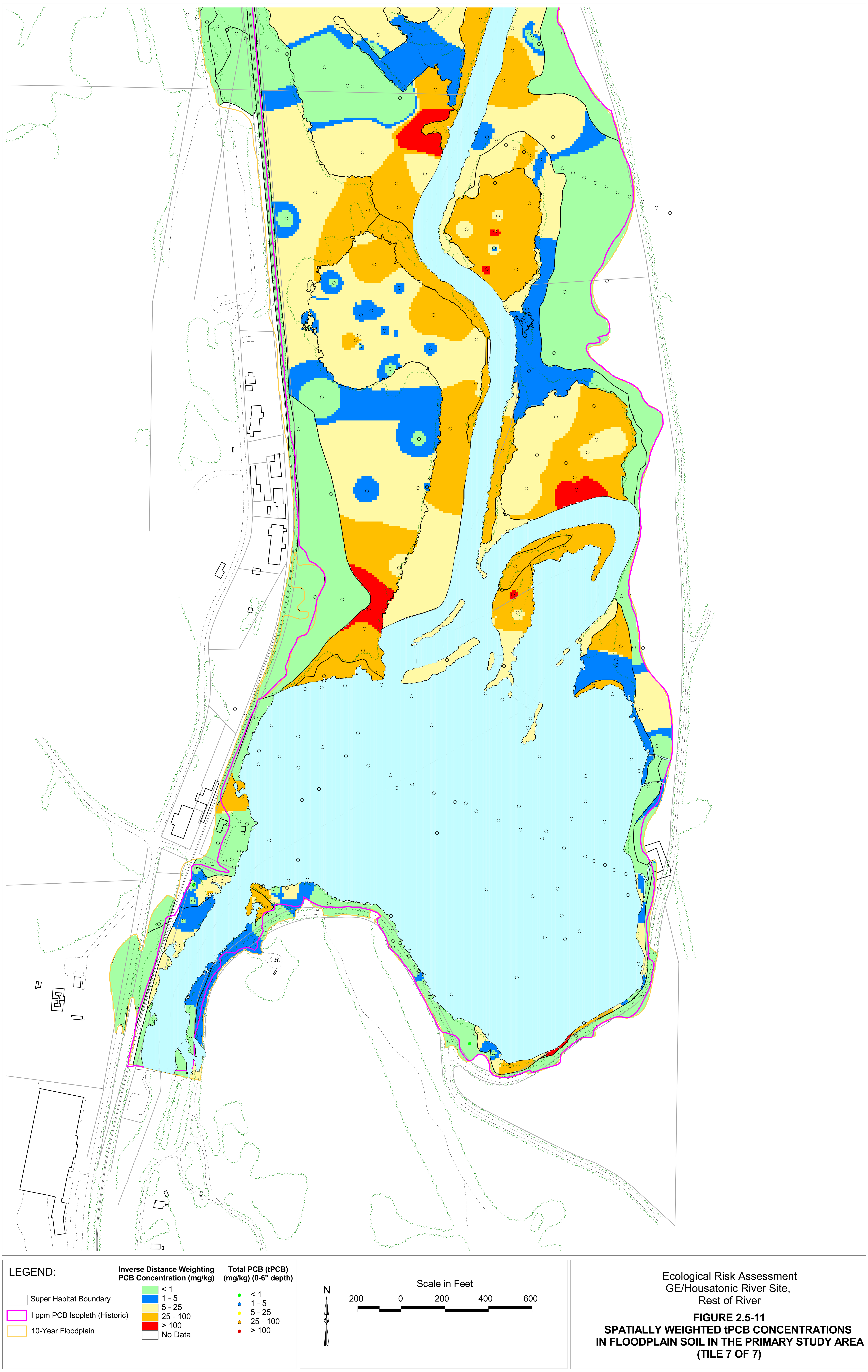
N

Scale in Feet

200 0 200 400 600

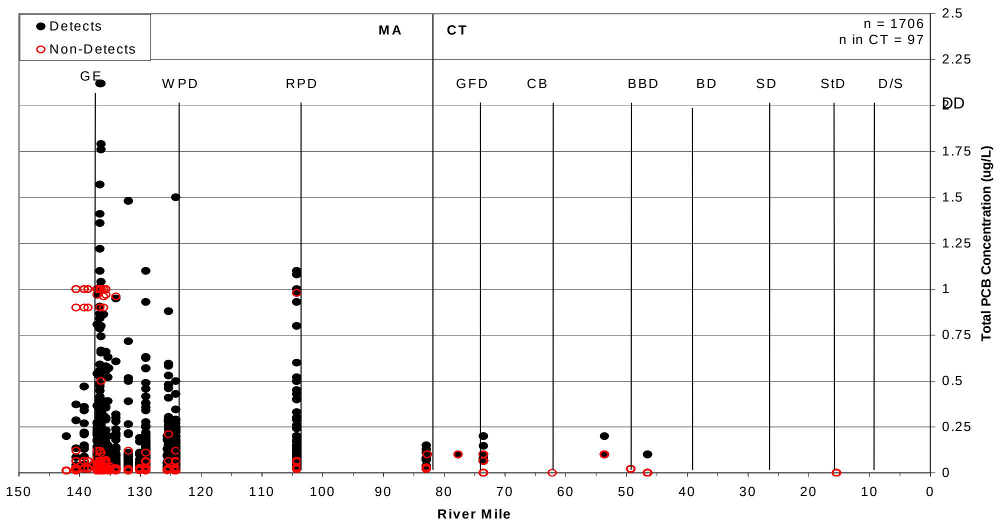
Ecological Risk Assessment
GE/Housatonic River Site,
Rest of River

FIGURE 2.5-10
SPATIALLY WEIGHTED tPCB CONCENTRATIONS IN FLOODPLAIN SOIL IN THE PRIMARY STUDY AREA (TILE 6 OF 7)



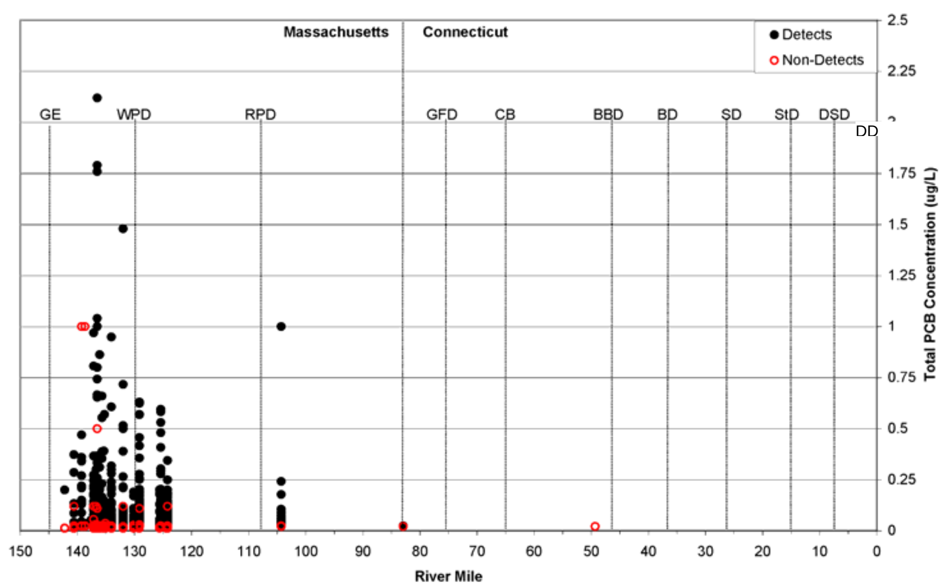
Ecological Risk Assessment
GE/Housatonic River Site,
Rest of River

FIGURE 2.5-11
SPATIALLY WEIGHTED tPCB CONCENTRATIONS IN FLOODPLAIN SOIL IN THE PRIMARY STUDY AREA
(TILE 7 OF 7)



Note: Symbols represent significant features/names of reach boundaries: GE = General Electric facility; WPD = Woods Pond Dam; RPD = Rising Pond Dam; GFD = Great Falls Dam; CB = Cornwall Bridge; BBD = Bulls Bridge Dam; BD = Bleachery Dam; SD = Shepaug Dam; STD = Stevenson Dam; DD = Derby Dam.

Figure 2.5-12 Total PCB Concentrations Measured in all Surface Water Samples Collected from the Housatonic River Since 1980



Note: Symbols represent significant features/names of reach boundaries: GE = General Electric facility; WPD = Woods Pond Dam; RPD = Rising Pond Dam; GFD = Great Falls Dam; CB = Cornwall Bridge; BBD = Bulls Bridge Dam; BD = Bleachery Dam; SD = Shepaug Dam; STD = Stevenson Dam; DD = Derby Dam.

Figure 2.5-13 Total Surface Water PCB Concentrations by River Mile (Data Collected Since 1996)

2.5.2.4 Biota

Biological tissue sampling was conducted to support both the human health and ecological risk assessments and modeling study. In general, most tissue samples collected were analyzed for tPCBs and PCB congeners, dioxins/furans, and organic carbon (OC) pesticides. Figures 2.5-14 and 2.5-15 present the distribution of tPCB concentrations for a majority of the biota used to evaluate PCB exposure in the baseline ERA.

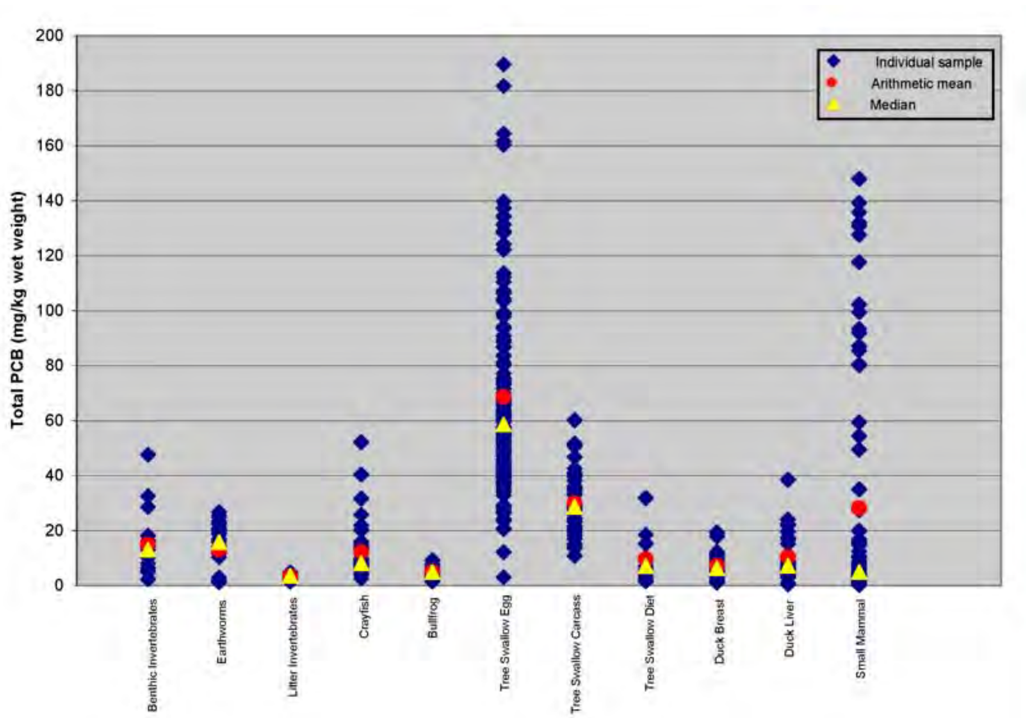
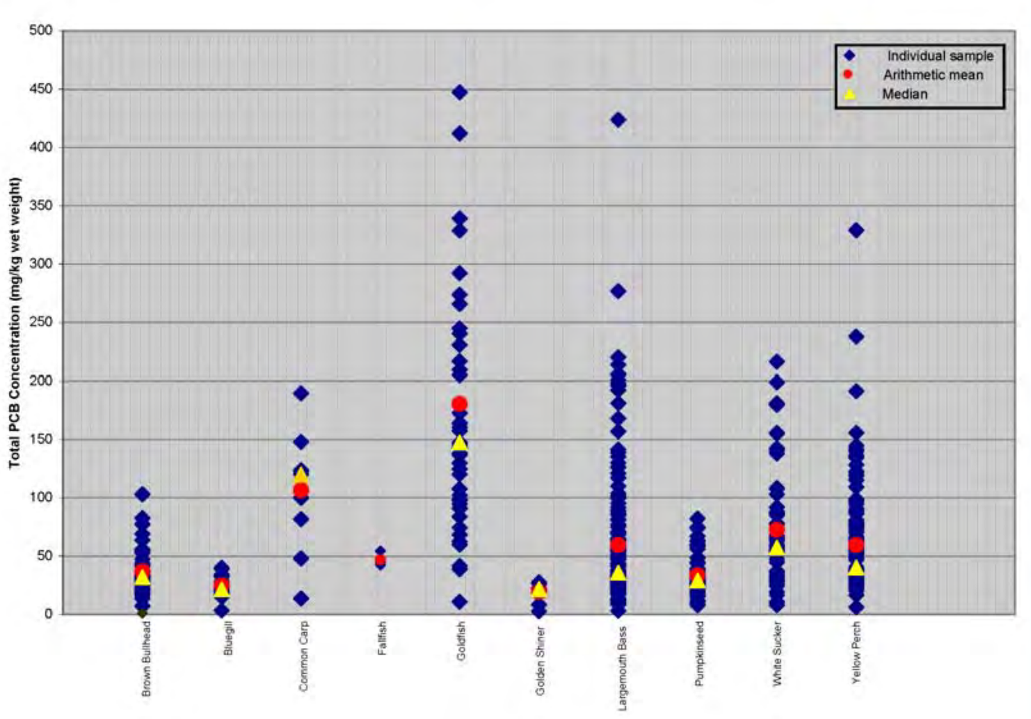


Figure 2.5-14 Total PCB Concentration (mg/kg wet weight) in Selected Biota (Excluding Fish) for Reaches 5 and 6



Note: Includes young-of-year fish.

Figure 2.5-15 Total PCB Concentration (mg/kg wet weight) in Reaches 5 and 6 Fish

2.5.3 Identification of Exposure Pathways

Considerable variability was observed in PCB concentrations in various media in the Housatonic River watershed. This variability is primarily attributable to the following factors:

- Differences in PCB concentrations among various abiotic exposure media (soil, sediment, water), particularly the small-scale heterogeneity observed in PCB concentrations in sediment.
- Analytical variability within a medium, which has been assessed and quantified (see Appendix C.11).
- Species-specific physiology, such as lipid content and metabolic requirements of the animals.
- Differences in life history and foraging behavior that affect duration and magnitude of PCB exposures.
- Position in aquatic or terrestrial food webs, thus affecting degree of biomagnification.

Sediment-associated invertebrates have significant exposures to PCBs because they remain in continuous contact with the sediment bed, which contains relatively high PCB concentrations. Water column invertebrates also accumulate PCBs, either through respiration of PCBs in the water column, or by ingestion of contaminated suspended particulate matter. Overall, food ingestion is the dominant pathway of PCB uptake for aquatic organisms in the Housatonic River.

Fish species exhibit interspecies variation in PCB concentrations. This partly reflects the differences in PCB concentrations in the abiotic media to which the fish are exposed. For example, forage fish tend to have lower PCB body burdens compared to benthic fish, which are in contact with contaminated sediment and porewater. However, the main reason for the interspecies differences is not direct contact with PCB-contaminated media; rather it is the differences in the dietary uptake patterns. Biomagnification in the food web is also a major factor controlling the PCB concentrations in fish. Biomagnification represents trophic-level differences in PCB concentrations and is measured as the increase in lipid-based contaminant concentrations in predators over those in prey (Russell et al. 1999). The mean tPCB and lipid-normalized tPCB concentrations in whole fish, by reach, are presented in Figures 2.5-16 and 2.5-17.

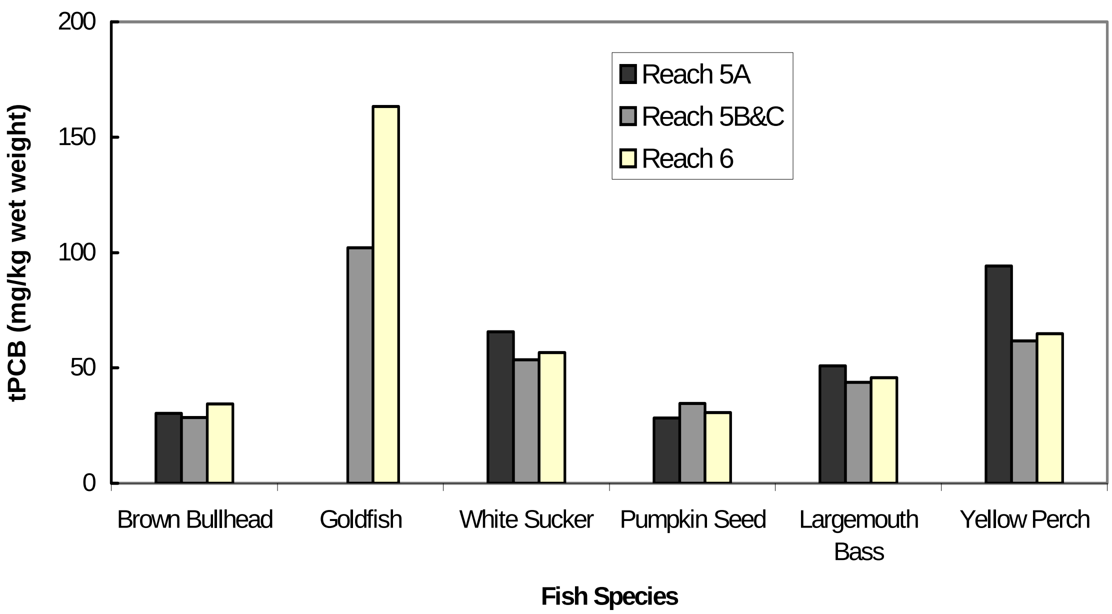


Figure 2.5-16 EPA Fish Collections (1998-2000) - Median tPCB Concentrations - All Ages by Subreach in the PSA

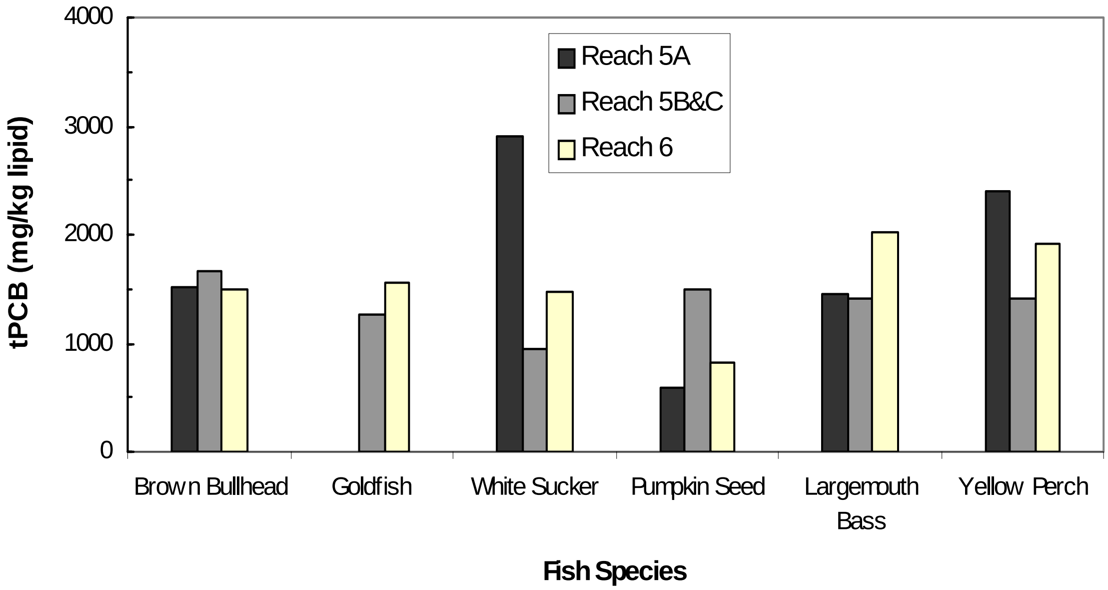


Figure 2.5-17 EPA Fish Collections (1998-2000) - Median Lipid-Normalized PCB Concentrations - All Ages by Subreach in the PSA

Organism foraging behavior plays a substantial role in the bioaccumulation of PCBs. Species that remain in proximity to the areas of higher PCB concentrations (e.g., main channel sediment) have increased exposure relative to those that use habitats such as distal floodplains or woodlands. Some species (e.g., wood frogs) have high exposures during specific life history stages but may migrate to less-contaminated habitats as adults. Other organisms (e.g., ducks, large raptors) may have exposures to highly contaminated prey as both juveniles and adults, but effectively “dilute” their exposures due to their large home ranges and/or seasonal residency in the Housatonic River watershed.

2.5.4 Changes in PCB Congener Patterns

Because PCBs constitute a group of contaminants rather than a single contaminant, their fate in the environment is complex. Some congeners are subject to degradation to a greater extent than others, with the transformation of those congeners and the potential creation of, or enhancement of, other congeners. In addition, congeners have different rates and extent of exchange in different media, resulting in differential rates and patterns of transport.

In the Housatonic River, the predominant congeners are the highly chlorinated congeners associated with the release of Aroclor 1260, and to a lesser degree, Aroclor 1254. The more highly chlorinated congeners are more resistant to degradation. A number of studies have shown that under laboratory conditions, PCB congeners in sediment samples from the Housatonic River can degrade to varying degrees, with the losses of some congeners and increases in the degradation product congeners (Bedard and May 1996). However, the congener data collected from the river sediment and floodplain soil do not support degradation as a major removal process.

During the release and transport of PCBs in the river, the level of chlorination of the congeners controls, in part, the distribution and exchange of the congeners among the solid and liquid phases. Increasing the degree of chlorination decreases the solubility of the congener and increases its tendency to sorb to solid phases. As a result, surface water samples have congener distributions that have a higher percentage of the less-chlorinated congeners compared to the congeners measured on the particulate matter in the same sample. Similarly, the less-chlorinated congeners are present at a higher percentage in porewater than those found in the sediment from which the water is extracted. The effect of this partitioning phenomenon is that PCBs tend to fractionate during transport and over time, with the loss of less-chlorinated congeners and the retention of more highly chlorinated ones. In the Housatonic River, however, the PCBs discharged from the facility were dominated by the more highly chlorinated congeners, primarily those associated with Aroclor 1260. As a result, only limited changes in the congener distribution are observed from differential congener transport.

In 2001, EPA and GE conducted a joint sampling effort to investigate site-specific PCB partitioning behavior in the Housatonic River. The program entailed synoptic collection of sediment and porewater, and in a complementary effort, synoptically in surface water and suspended solids. The synoptic nature of the collections and analyses allowed a detailed assessment of partitioning behavior and an assessment of shifts in congener distributions among media. Findings from the study include:

- The analyses of approximately 50 paired bulk sediment/porewater samples indicate a shift in the PCB homolog profiles between media. In bulk sediment, the homolog profile averaged 5.9 chlorines per biphenyl (Cl/BP), with hepta-PCBs having the largest contribution to tPCB mass. In contrast, synoptic porewater samples had an average of 5.3 Cl/BP, with hexa-PCBs having the largest contribution to tPCB mass. This pattern reflects the congener-dependent partitioning behavior described above.

- Spatial trends in chlorination level (which may be used as a surrogate for alterations in congener distributions related to chemical properties) were evaluated in sediment and porewater. No trend with distance downstream was observed in porewater. A modest reduction in chlorination level was observed in bulk sediment, however. Typically, the majority of Cl/BP ranged from 6 to 6.5 for samples collected within Reach 5A of the PSA, whereas downstream samples (Reaches 5B and 5C) usually had 5.5 to 6 Cl/BP. This confirms that changes in the congener distribution with distance from the source are possible, but limited because of the highly chlorinated nature of PCBs in the source media.

- Surface water particulate matter in samples collected from four locations (Pomeroy Avenue, West Branch, New Lenox Road, Woods Pond) exhibited congener/homolog distributions comparable to bulk sediment. The particulate organic matter yielded an average of 6.0 Cl/BP, compared to 5.9 Cl/BP for bulk sediment.

- Surface water samples from the same four locations yielded dissolved PCB profiles (4.7 Cl/BP) that were slightly “lighter” than porewater samples (5.3 Cl/BP), primarily because of an increased percentage of tri-PCBs.

These findings support the conceptual model that PCB congener distributions will differ in aqueous and particulate media, primarily because of contaminant properties that favor partitioning to solids (and reduced solubility) for higher chlorinated congeners. Some spatial variation in the partitioning behavior for sediment is apparent, but does not dominate the kinetics. Therefore, it appears that physical transport of PCBs (via bedload and suspended particulate matter at higher flows, and diffusive flux at lower flows) is the dominant fate process, with dechlorination of PCB mixtures a relatively minor process.

A more extensive evaluation of congener patterns in sediment, soil, and tissue, using multivariate classification analysis (Euclidean distance) and principal components analysis (PCA), was performed. These analyses included a broad range of media, including floodplain soil, bank soil, bed sediment, suspended sediment, and tissue (e.g., bullfrogs, fish, tree swallows, crayfish, and ducks). The analyses were conducted to investigate the similarity of congener patterns within and among groups of samples for the purpose of measuring the differences between groups and the level of consistency within groups. Overall, the congener evaluation (Appendix C.7) indicated that differences in profiles are sometimes evident, but that most media exhibit congener profiles similar to Aroclor 1260 across all reaches. Some changes in congener profiles were observed both spatially and across media, with the latter differences larger than the former.

2.5.5 Fate and Transport of Dioxins/Furans

The following discussion presents an overview of the general fate and transport mechanisms associated with dioxins/furans that were retained as COPCs in all media (see Section 2.4).

2.5.5.1 Transport and Partitioning

Dioxins and furans, similar to PCBs, are characterized by low solubility, low vapor pressure, and high affinity for organic carbon (log K_{oc} values as high as 7.39), which suggests that they will strongly adsorb to sediment or soil and that their vertical movement in either medium will be limited. The leaching of dioxins and furans is unlikely if water is the only transporting medium; however, saturation of sorbed sites and the presence of organic solvents or petroleum may result in vertical migration in sediment or soil.

Volatilization from soil during warm months may also be a major partitioning mechanism. In general, the higher the degree of chlorination, the lower the relative degree of volatilization from soil or water.

In the atmosphere, dioxins and furans are typically adsorbed to particulates with the vapor-phase tending to be negligible (Paustenbach et al. 1991). Vapor pressure and ambient temperature are the two environmental factors controlling the phase of congeners in the atmosphere. Congeners having a vapor pressure greater than 10^{-4} mm Hg will exist primarily in the vapor phase. Dioxins and furans have relatively long residence times in the atmosphere and are removed by wet, dry, and gas-phase (vapor phase onto plant surface) deposition (ATSDR 1998). Contamination of plant foliage via atmospheric deposition is the primary mechanism of accumulation in terrestrial plants.

Dioxin and furan adsorption to particulates in the water column increases with increasing chlorination. Dioxins and furans are removed from the water column primarily by binding with particulates, sediment, or biota and, to a lesser extent, by volatilization (Paustenbach et al. 1992).

Resuspension of sediment-bound dioxins and furans can increase their transport and availability for uptake by aquatic biota. The primary route of exposure to dioxins and furans for lower trophic-level organisms is uptake from water. Bioaccumulation appears to increase with increasing chlorination up to T(tetra)CDDs and TCDFs. For higher trophic-level organisms, the predominant route of exposure is via food chain transfer.

2.5.5.2 Transformation and Degradation

Photolysis of dioxins and furans in sediment or soil is a relatively slow process when compared with aquatic photolysis rates. However, the addition of organic solvents to contaminated sediment or soil can enhance photolytic transformation. Field and laboratory studies have shown that several microorganisms (e.g., fungi and bacteria) are capable of degrading different congeners. In general, the rate of biodegradation decreases with increasing chlorination.

In the atmosphere, dioxin and furan reactions with hydroxyl radicals appear to be the most significant source of transformation. Vapor-phase dioxins and furans may also undergo photolytic degradation. The estimated half-life for TCDD reactions with hydroxyl radicals is 2 to 8 days, and the estimated photolytic lifetime ranges from 1 to 7 days (ATSDR 1998). OCDDs and OCDFs, with their low vapor pressure, partition to the particulate phase. Atmospheric photodegradation of these highly chlorinated congeners is less likely.

Dioxins and furans in aquatic environments are primarily associated with particulate matter. Photodegradation of bound dioxins and furans occurs near the water's surface and decreases with water depth. Biodegradation in the water column does not appear to be a significant transformation mechanism. Limited biodegradation of dioxins and furans has been observed in sediment, with degradation rate decreasing with increasing chlorination.

2.6 EFFECTS ON RECEPTORS

There are a number of chemical stressors that may have an adverse impact upon organisms found in the Housatonic River PSA. The Pre-ERA identified 24 COPCs that are of interest (Appendix B, Section 2.3). A short review of toxicity mechanisms and the possible effects to aquatic and terrestrial organisms for PCBs and dioxins/furans follows. A more detailed, receptor-specific review of COC toxicity is presented in each of the assessment endpoint appendices (Appendices D through K).

2.6.1 Polychlorinated Biphenyls (PCBs)

The toxicology of PCBs varies considerably among congeners, depending on the number and location of chlorines on the biphenyl molecule, and also between animal species due to differences in absorption, metabolism, mechanism of action, and potential toxic effects (Eisler and Belisle 1996).

PCB congeners vary in toxicity in many ways, including mode of action, potency, and potential for interaction. PCB congeners may interact with each other and with other chemicals when combined in a complex commercial PCB mixture. Lethal and sublethal effects of PCBs on mammals, birds, and aquatic life are discussed in detail in the appropriate assessment endpoint appendices; a general summary of PCB-associated effects is presented in Table 2.6-1. The following discussion of PCB toxicology focuses primarily on the general mechanisms of PCB toxicity.

PCB congeners differ in their biological activities, and different animal species vary in their sensitivity to the individual congeners. Multiple and diverse mechanisms are involved in the toxicological responses of animals to PCB exposures. The mechanism of Ah-receptor binding is an initial step in producing toxic effects, and is the basis for the World Health Organization's (WHO) toxic equivalency factors (TEFs) approach for ranking the relative potency of PCBs, PCDDs, and PCDFs (Van den Berg et al. 1998). The WHO TEFs only apply to Ah-receptor-mediated biochemical responses and toxic effects. The relationship between PCB molecular structure and the potential for toxic effects independent of Ah-receptor mediation is not clearly understood. Research through the 1990s found increasing evidence for alternative mechanisms for several PCB-induced effects such as neurotoxicity and disruption of neutrofil function independent of Ah-receptor mediation (ATSDR 2000). In addition, there is a third category, where PCB toxicity may be initiated by both Ah-receptor-dependent and independent mechanisms.

Table 2.6-1

Common Effects of PCB Exposure Observed in Various Animals

System Affected	Specific Effect
Hepatic effects	▪ Hepatomegaly, bile duct hyperplasia ▪ Widespread (e.g., rabbit) or focal (e.g., mouse) necrosis ▪ Lipid accumulation, fatty degeneration ▪ Induction of microsomal monooxygenases and other enzymes ▪ Decreased activity of membrane ATPases ▪ Depletion of fat-soluble vitamins ▪ Porphyria
Gastrointestinal effects	▪ Hyperplasia and hypertrophy of gastric mucosa ▪ Gastric ulceration and necrosis ▪ Proliferation and invasion of intestinal mucosa (monkey) ▪ Hyperplasia, hemorrhage, necrosis (hamster, cow)
Respiratory system	▪ Chronic bronchitis, chronic cough
Nervous system	▪ Alterations in catecholamine levels ▪ Impaired behavioral responses ▪ Developmental deficits ▪ Depressed spontaneous motor activity ▪ Numbness in extremities
Skin	▪ Chloracne ▪ Edema, alopecia
Immunotoxicity	▪ Lymphoid involution (spleen, lymph nodes, especially thymus) ▪ Subsequent reduction of circulating lymphocytes ▪ Suppressed antibody responses ▪ Enhanced susceptibility to viruses ▪ Suppression of natural killer cells
Endocrine system	▪ Altered levels of circulating steroids ▪ Estrogenic, antiestrogenic, antiandrogenic effects ▪ Decreased levels of plasma progesterone ▪ Adrenocortical hyperplasia ▪ Thyroid pathology, changes in circulating thyroid hormones
Reproduction	▪ Increased length of estrus ▪ Decreased libido ▪ Embryo and fetal effects following in utero exposure
Carcinogenesis	▪ Promoter ▪ Attenuation of some carcinogens

Source: Hansen 1994

PCBs are able to induce hepatic Phase I enzymes (CYP oxygenases) and Phase II enzymes (e.g., UDP glucuronyltransferases, epoxide hydrolase, glutathione transferase). Most commercial PCB mixtures induce both 3-methylcholanthrene type (CYP1A1 and 1A2) and phenobarbital-type (CYP2B1, 2B2, and 3A) CYPs. Non-ortho and mono-ortho PCBs can assume a coplanar molecular configuration and bind to the Ah receptor causing CYP1A1/1A2 induction in rodents (Safe 1994). Effects from PCBs involving the Ah-receptor-initiated mechanisms include body weight wasting, thymic atrophy, porphyria, and porphyria cutanea tardea (Safe 1994).

There are many examples of the complexity of the relationship between PCB molecular structure and toxic effects independent of Ah-receptor initiation. For example, some PCBs with two ortho chlorines and lateral chlorines induce both types of CYPs but demonstrate little Ah-receptor affinity. Di-ortho PCBs with one or two para chlorines predominantly induce CYP2B1/2B2/3A and have no affinity for the Ah receptor (Connor et al. 1995). The induction of phenobarbital-type CYPs by PCBs is independent of the Ah receptor. PCBs with at least two ortho chlorines and one or two para chlorines are the most potent CYP inducers.

Neurological and neurodevelopmental effects involving changes in brain dopamine levels are PCB-induced effects that are Ah-receptor independent. Scientists have hypothesized that the effect on dopamine levels is related to decreased dopamine synthesis by PCB inhibition of certain enzymes or decreased dopamine uptake into vesicles (ATSDR 2000). It is also possible that a connection exists between disruption of Ca^{2+} homeostatic mechanisms and neurological and neurodevelopmental effects. It is clear that Ah-receptor-independent mechanisms are important in the induction of neurotoxic effects by PCBs.

In vitro studies have indicated that PCBs can induce functional changes such as degranulation in neutrofils (ATSDR 2000). These functional changes may be related to PCB toxicity such as immunological effects and tissue damage. Immunological effects that involve neutrofils include defenses against pathogens and inflammatory responses leading to tissue injury.

There are a number of effects that involve both Ah-receptor-dependent and -independent mechanisms. These include liver hypertrophy, neurodevelopmental, or reproduction effects involving changes in steroid hormone homeostasis and/or thyroid hormone disruption, immunological effects, and cancer (Safe 1994; ATSDR 2000).

Safe (1994) reviewed numerous studies of PCB-induced hepatoxicity in mammals including rats, mice, rabbits, guinea pigs, monkeys, and mink exposed to Aroclors including 1221, 1242, 1248, 1254, and 1260. From these studies, it appears that PCB-induced liver toxicity is mediated by both Ah-receptor-dependent and -independent mechanisms.

Reproductive impairment following PCB exposure has been observed in mink, one of the most sensitive mammals to PCB toxicity (Eisler and Belisle 1996; Moore et al. 1999). Although congeners with high Ah-receptor affinity are more potent than congeners with low Ah-receptor affinity, there is evidence that Ah-receptor-independent mechanisms may be involved.

Review of the scientific literature indicates that animals exposed to PCBs have an increased risk of cancer. Lifetime oral exposures to a number of commercial PCB mixtures (Aroclors 1016, 1242, 1254, and 1260) have produced liver tumors in female rats and Aroclor 1260 has produced liver tumors in male rats. Mixtures with high chlorine content such as Aroclor 1254 were generally more potent than mixtures with low chlorine content such as Aroclor 1016 (Mayes et al. 1998).

2.6.2 Dioxins/Furans

Many halogenated aromatic compounds have been described as exhibiting dioxin-like behavior, such as polychlorinated dibenzofurans (PCDFs) and some coplanar polychlorinated biphenyls (PCBs), based on similarities in toxicity and mechanism of action. The primary toxic mechanism of action is binding of the PCDD, PCDF, or coplanar PCB compound to the Ah receptor (described in the previous section). Because 2,3,7,8-TCDD binds to the Ah receptor with a high affinity and has a high toxic potency, it has been the focus of experimental toxicity studies. EPA, regulatory agencies in other countries, and international organizations such as WHO use a TEF approach to reflect the varied toxicity of the different PCDDs, PCDFs, and PCBs.

The impact of dioxins in the environment is directly related to their highly lipophilic and hydrophobic nature as well as to the toxic effects of these compounds on plants and animals. These toxic effects have been extensively studied in the laboratory and through evaluation of animals exposed to dioxins in the environment. The toxicology of PCDD/PCDF varies considerably between congeners and between animal species in absorption, metabolism, mechanism of action, and potency of toxic and carcinogenic effects. The following discussion of PCDD and PCDF toxicology focuses primarily on the general mechanisms of toxicity.

2,3,7,8-TCDD and equivalents share a mechanism of toxicity that initially involves binding of the individual congener to the cystolic Ah receptor in all animal species. After initial binding, the ligand-receptor complex is translocated to the nucleus of the cell. It then becomes associated with the DNA and causes transcription of one or more contaminated genes (EPA 1993). The physiological effects that follow are species-specific but there are many similarities, including the induction of enzyme systems such as cytochrome P4501A1, "wasting syndrome," decreased immunocompetence, reproductive effects, edema, and mortality. The Ah receptor is present in all mammalian and bird species that have been tested, as well as in many species of fish. It is unclear whether the Ah receptor is present in amphibians and reptiles.

A protein similar to the Ah receptor has been identified in terrestrial invertebrates, but there is no evidence to support the existence of an Ah-receptor type protein in aquatic invertebrates (EPA 1993).

2,3,7,8-TCDD toxicity and the toxicity of the other 74 individual PCDD congeners is mediated by the Ah receptor. Differences between species in sensitivity to 2,3,7,8-TCDD may be related to the size and binding efficiency of the Ah receptor, pharmacokinetic differences between species, and additional contributing factors. Ah-receptor affinity is determined by the chlorine substitution pattern of the individual dioxin congener. 2,3,7,8-TCDD is substituted in all four lateral positions and has the highest affinity for the Ah receptor. Less active congeners have an additional one, two, or four nonlateral chlorine substituents or have lateral chlorine substituents removed. 2,3,7,8-TCDD and structurally related halogenated aromatic compounds induce microsomal hepatic enzymes such as hepatic aryl hydrocarbon hydroxylase (AHH) and ethoxyresorufin-O-deethylase (EROD). Both AHH and EROD are markers of CYP1A1 activity. Increased synthesis of cytochrome P4501A1 (CYP1A1) is induced by an individual dioxin congener binding to the Ah receptor. CYP1A1 functions in the detoxification or activation of endogenous and exogenous chemicals. Cytochrome P4501A2 (CYP1A2) is only induced in hepatic tissue and has a similar function to CYP1A1.

Effects observed in the offspring of animals exposed to 2,3,7,8-TCDD include fetal/newborn mortality, structural malformations, impaired development of the reproductive system, neurodevelopmental effects, immunotoxicity, and thymic atrophy. Impaired development of the reproductive system and neurobehavioral effects in the developing organism are the most sensitive endpoints of 2,3,7,8-TCDD exposure (ATSDR 1998).

2,3,7,8-TCDD is a potent animal carcinogen and has tested positive for carcinogenicity in 19 different studies in four animal species: mice, rat, hamsters, and fish (Huff 1992; Johnson et al. 1992). EPA classifies 2,3,7,8-TCDD as a B2, probable human carcinogen (EPA 2002).

The exact mechanism of how 2,3,7,8-TCDD causes carcinogenicity is not well understood but the evidence indicates that direct DNA damage through formation of DNA adducts is not the mechanism. The carcinogenicity of 2,3,7,8-TCDD is thought to involve the Ah receptor. 2,3,7,8-TCDD is considered a nongenotoxic carcinogen and has tested as not mutagenic in the Salmonella/Ames test. 2,3,7,8-TCDD is a potent tumor promoter and is either a weak initiator or a non-initiator. 2,3,7,8-TCDD and the other carcinogenic dioxin congeners are whole and complete carcinogens as tested in mice, rats, hamsters, and fish.

PCDD/PCDFs disrupt normal homeostatic processes that regulate cell growth and differentiation. These disruptions produce a wide range of toxic effects and histopathological changes. The PCDD/PCDF congeners vary in many ways including affinity for the Ah receptor, potency, and potential for interaction.

2.6.3 2,3,7,8-TCDD Toxic Equivalence (TEQ)

The polychlorinated halogenated (PCH) congeners (including both PCBs and dioxins/furans) have different toxicity potencies, and there may be synergistic and/or antagonistic effects among the congeners. To estimate the relative toxicity of PCH mixtures, a system of toxic equivalency factors (TEFs) has been developed. This approach is based on in vivo and in vitro toxicity of each of the PCH congeners in relation to 2,3,7,8-TCDD, which is considered to be the most toxic of the PCH class of chemicals (Van den Berg et al. 1998; Birnbaum and DeVito 1995; Safe 1994). There are a number of assumptions made when using the TEF approach. These include: (1) PCH congeners are Ah-receptor antagonists and their toxicological potency is mediated by their binding affinity; and (2) no interaction occurs between the congeners and thus, the sum of the individual congener effects accounts for the potency of the PCH mixture. The overall effect of these assumptions is a potency estimate or toxic equivalency (TEQ) value. To generate a TEQ, the following equation (equation modified from Van den Berg et al. 1998) is used:

$$TEQ = \sum_{n=1}^{7}[PCDD_n \ x \ TEF_n] + \sum_{p=1}^{10}[PCDF_p \ x \ TEF_p] + \sum_{q=1}^{12}[PCB_q \ x \ TEF_q]$$

where:

TEQ = Toxic equivalence

$PCDD_n$ = Polychlorinated dibenzo-p-dioxin congener concentration

$PCDF_p$ = Polychlorinated dibenzo-p-furan congener concentration

PCB_q = Polychlorinated biphenyl congener concentration

$TEF_{n,p,q}$ = Toxic equivalency factor for appropriate individual PCDD/PCDF and PCB congeners, respectively.

There are a number of TEF approaches available in the scientific literature for PCHs (e.g., Van den Berg et al. 1998; Kennedy et al. 1996; Safe 1994; NATO 1988). For this ERA, the TEFs presented by Van den Berg et al. (1998) were adopted. TEF values were developed for those compounds that: (1) show a structural relationship to PCDDs and PCDFs; (2) bind to the Ah receptor; (3) elicit an Ah-receptor-mediated biochemical and toxic response; and (4) are persistent and accumulate in the food chain (Van den Berg et al. 1998; Birnbaum and DeVito 1995).

The Van den Berg et al. TEFs are the most recently proposed and are based on the best available science in terms of identifying specific endpoints consistent with the mode of action of each of the congeners. They have also been widely accepted and applied in the scientific literature since 1998 (Dyke and Stratford 2002; Lindstrom et al. 2002). Van den Berg et al. (1998) present TEF values for use in deriving TEQ for mammals, fish, and birds as predators (Table 2.6-2).

Table 2.6-2

TEF Values for Mammals, Fish, and Birds as Predators

No.	Congener	TEF		
		Mammals	Fish	Birds
1	PCB-77	0.0001	0.0001	0.05
2	PCB-81	0.0001	0.0005	0.1
3	PCB-126	0.1	0.005	0.1
4	PCB-169	0.01	0.00005	0.001
5	PCB-105	0.0001	<0.000005*	0.0001
6	PCB-114	0.0005	<0.000005*	0.0001
7	PCB-118	0.0001	<0.000005*	0.00001
8	PCB-123	0.0001	<0.000005*	0.00001
9	PCB-156	0.0005	<0.000005*	0.0001
10	PCB-157	0.0005	<0.000005*	0.0001
11	PCB-167	0.00001	<0.000005*	0.00001
12	PCB-189	0.0001	<0.000005*	0.00001
13	1,2,3,4,6,7,8-HpCDD	0.01	0.001	<0.001*
14	1,2,3,4,6,7,8-HpCDF	0.01	0.01	0.01
15	1,2,3,4,7,8,9-HpCDF	0.01	0.01	0.01
16	1,2,3,4,7,8-HxCDD	0.1	0.5	0.05
17	1,2,3,4,7,8-HxCDF	0.1	0.1	0.1
18	1,2,3,6,7,8-HxCDD	0.1	0.01	0.01
19	1,2,3,6,7,8-HxCDF	0.1	0.1	0.1
20	1,2,3,7,8,9-HxCDD	0.1	0.01	0.1
21	1,2,3,7,8,9-HxCDF	0.1	0.1	0.1
22	1,2,3,7,8-PeCDD	1	1	1
23	1,2,3,7,8-PeCDF	0.05	0.05	0.1
24	2,3,4,6,7,8-HxCDF	0.1	0.1	0.1
25	2,3,4,7,8-PeCDF	0.5	0.5	1
26	2,3,7,8-TCDF	0.1	0.05	1
27	2,3,7,8-TCDD	1	1	1
28	OCDD	0.0001	<0.0001*	0.0001
29	OCDF	0.0001	<0.001*	0.0001

*Values that are less than should be considered to be the upper limit for use in any TEQ calculation.

Source: Van den Berg et al. 1998

2.7 CONCEPTUAL MODEL

A conceptual model is a written description and visual representation of predicted relationships between ecological entities and the stressors to which they may be exposed. In essence, the conceptual model presents a series of working hypotheses regarding how the stressors might affect ecological components at the site. Risk hypotheses are specific assumptions about potential risk to assessment endpoints and may be based on theory and logic, empirical data, and mathematical or probability models. The hypotheses are formulated using professional judgment and available information of the ecosystem at risk, potential stressor sources and characteristics, and observed or predicted effects on assessment endpoints. Conceptual models include ecosystem processes that influence receptor responses, or exposure scenarios that qualitatively link land use activities to stressors and describe primary, secondary, and tertiary exposure pathways, ecological effects, and ecological receptors (EPA 1998).

The development of the conceptual model is a complex, non-linear process, with many parallel activities that result in modifications to the conceptual model as additional information becomes available. The objectives of the conceptual model presented here are to illustrate the important relationships within the Housatonic study area, and to specify exposure scenarios evaluated in the ERA, as a refinement of the conceptual model outlined in the SI Work Plan. The model was refined based upon physical, chemical, and biological information collected during the investigation, and on the body of scientific knowledge on the COPCs that has also evolved in the interim. The following discussion presents an overview of the primary exposure pathways, risk questions/testable hypotheses, and a visual representation of the predicted relationships between ecological receptors and contaminant stressors (see Figure 2.7-1).

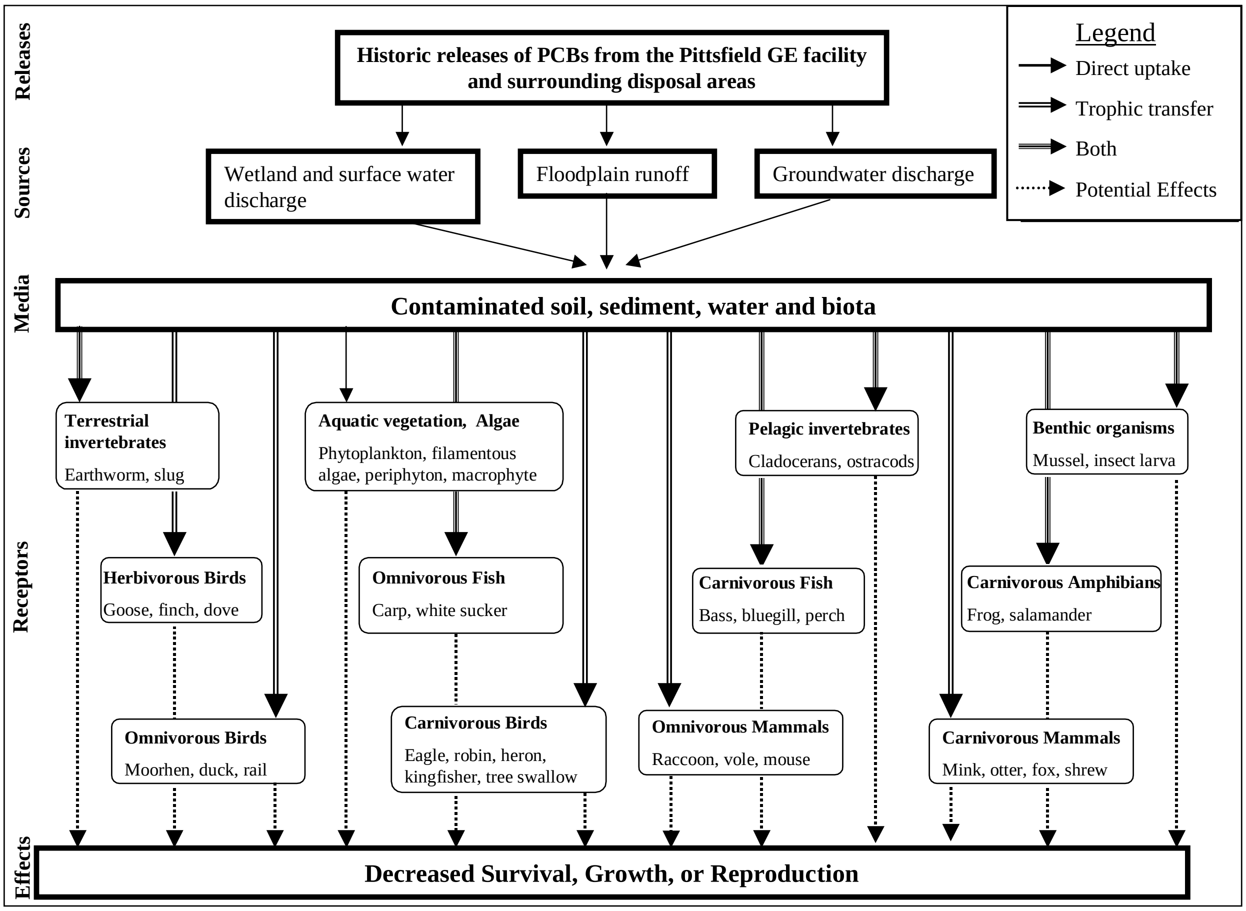


Figure 2.7-1 Housatonic River Ecological Risk Assessment Conceptual Model: Principal Exposure Pathways for PCBs

2.7.1 Exposure Pathways

Exposure of receptors to COPCs is possible through various pathways including absorption through gills, dermal contact, ingestion of sediment, ingestion of surface water, ingestion of soil, ingestion of contaminated food, and inhalation of volatilized substances. Sediment may become resuspended if hydrodynamics disturb the sediment bed and distribute suspended sediment outside the river when floodplains are inundated. Organisms may also be exposed to chemical contaminants through trophic transfer. Organisms lower in the food chain may ingest and accumulate a contaminant, which is then passed on when they are consumed by higher food chain predators.

Benthic and soil communities are at risk of direct exposure to PCBs and several other COPCs (e.g., dioxins, furans, lead, mercury, PAHs). Species in these communities are exposed to COPCs through direct contact with interstitial porewater, ingestion of sediment particles, and ingestion of organisms that have also been exposed to contaminants.

Pelagic organisms in the Housatonic River system are exposed to COPCs through dermal and gill contact with surface water; ingestion of water, suspended sediment, and organic matter; ingestion of sediment for bottom-feeding fish; and ingestion of other benthic and pelagic organisms. Uptake of PCBs by fish occurs mainly through the gills and the gastrointestinal tract (Shaw and Connell 1984). Most PCB accumulation in top fish predators can be attributed to the food pathway (Thomann 1989). Other species, such as amphibians, are also exposed to PCB-contaminated surface water. The early life stages of these organisms are entirely aquatic and, because the skin is a respiratory surface during this phase, dermal exposure may be important.

Insectivorous, carnivorous, and piscivorous birds and mammals that reside, or partially reside, within the PSA are exposed to PCBs principally through diet and trophic transfer. PCBs are highly bioaccumulative substances that increase in concentration as they are passed up the food chain. For organisms inhabiting the Lake St. Clair ecosystem, Haffner et al. (1994) showed that PCB concentrations increased from 935 µg/kg in sediment, to 1,360 µg/kg in bivalves, to 7,240 µg/kg in oligochaetes, and to 64,900 µg/kg in predatory gar pike. MacKay (1989) has also noted the food chain biomagnification of PCBs for several piscivorous birds. The avian and mammalian predators of the Housatonic River study area would similarly be expected to accumulate PCBs from the prey they consume. Water, sediment, and soil consumption from foraging activities likely contribute less to PCB exposure.

The exposure pathways for other COPCs depend largely on their chemical and physical properties. Highly lipophilic substances, such as dioxins and furans, will behave similarly to PCBs, partitioning to sediment and being upwardly mobile in the food chain.

Figure 2.7-1 characterizes the ecosystem in the Housatonic River PSA, as well as the major exposure pathways for COPCs.

As a component of the development of the site conceptual model, testable hypotheses or "risk questions" are developed to provide the basis for the study design and selection of measurement endpoints. These hypotheses represent statements regarding anticipated ecological effects and define the focus of the individual studies. In general, the primary question to be asked by the risk hypothesis is "what probabilities are associated with effects of differing magnitudes as a result of exposure of the assessment endpoint to the COPC?" The three major lines of evidence used to answer this question are:

- Comparison of an estimated or measured exposure concentration of a COPC to concentrations known from the literature to be toxic to receptors associated with the assessment endpoint.

- Comparison of laboratory bioassay results using media from the site to the results using media from a reference site, and/or comparing in situ toxicity test results at the site to results at a reference location, or comparisons of results across a concentration gradient.

- Comparison of observed effects in the receptors in the field, with observations in similar receptors at reference locations, or across a concentration gradient (e.g., exposure modeling).

2.8 SELECTION OF ASSESSMENT AND MEASUREMENT ENDPOINTS

The selection of endpoints for consideration in an ERA requires identification of ecological characteristics that may be adversely affected by site contaminants. In an ERA, two types of endpoints are required – assessment endpoints and measurement endpoints. Assessment endpoints represent specific ecological values deemed important to protect; measurement endpoints are the tools used to determine the outcome for the assessment endpoints.

2.8.1 Assessment Endpoints

Assessment endpoints are unambiguous statements or goals concerning specific ecological characteristics (e.g., reproductive effects on aquatic organisms) that are to be evaluated and protected (EPA 1994, 1998). Assessment endpoints determine the foundation for the ERA because they:

- Provide guidance for evaluating the site and the extent of contamination.
- Establish a basis for assessing the potential risks to identified receptors.
- Assist in the identification of the ecological structure and function at the site.

Each site or area evaluated in an ERA has the potential to be biologically unique; therefore, there is no universal list of assessment endpoints (Suter 1993). Because it is not practical or possible to directly evaluate risks to all of the individual components of the ecosystem at a site, assessment endpoints should focus the risk assessment on particular components of the ecosystem that could be adversely affected by contaminants from the site (EPA 1997). According to EPA's *Ecological Risk Assessment Guidance for Superfund* (EPA 1997):

> "Assessment endpoints for the baseline ERA must be selected based on the ecosystems, communities, and/or species potentially present at the site. The selection of assessment endpoints depends on:
>
> - The contaminants present and their concentration;
> - Mechanisms of toxicity of the contaminants to different groups of organisms;
> - Ecologically relevant receptor groups that are potentially sensitive or highly exposed to the contaminant and attributes of their natural history; and
> - Potentially complete exposure pathways."

To guide this process, EPA (1998) provides further detail on the criteria that assessment endpoints should satisfy:

> ***Ecological relevance*—**Assessment endpoints must reflect biologically important characteristics of the ecosystem and should be functionally related to other important components of the system. Ecologically relevant assessment endpoints are particularly valuable for identifying potential cascading adverse effects resulting from the loss or reduction of a species or guild. For example, an alteration of the benthic community is of concern not only to the benthos, but also to higher trophic levels because of disruption at the base of the food web. Alternatively, an alteration at the higher trophic levels may reflect an integration of effects throughout the ecosystem.

> ***Susceptibility to known or potential stressors*—**There should be a cause-effect linkage between the assessment endpoint and the magnitude of the contaminant stressor.

> ***Relevance to management goals*—**The selection of endpoints that reflect societal values and management goals, while not scientifically based, ensures that the risk assessment will have utility for the risk management decisions that must be made. Management goals are desired characteristics of the ecosystem deemed to have value to the public. For example, fish abundance and biomass may be used as indicators of whether fisheries are being adequately protected. The status of the benthic invertebrate community in the study area is often a good indicator of the overall productivity of the aquatic ecosystem, making it a relevant endpoint for maintaining a viable fishery in an area.

In addition, specific assessment endpoints should define the ecological value in sufficient detail to identify measures needed to answer specific questions or to test specific hypotheses (EPA 1997). An assessment endpoint must be definable in a practical context, and requires both an entity (that can be clearly defined) and an attribute (that can be measured or assessed). The operational definition ensures that the assessment endpoint can be linked with a measured response.

Ultimately, the value of an ERA depends on whether it can be used to determine if a baseline risk is present and to support appropriate managerial decisions. Therefore, the selection of assessment endpoints is fundamental in determining the utility of the risk assessment process. Once assessment endpoints are selected and the conceptual model of exposure is developed, testable hypotheses and measurement endpoints are developed to determine whether or not a potential threat to the assessment endpoints exists (EPA 1997).

2.8.2 Measurement Endpoints

A measurement endpoint is defined as "a measurable ecological characteristic that is related to the valued characteristic chosen as the assessment endpoint." Measurement endpoints link the conditions existing on-site to the goals established by the assessment endpoints through the integration of modeled, literature, field, or laboratory data (Maughan 1993).

"Measurement endpoints are frequently numerical expressions of observations (e.g., toxicity test results, community diversity measures) that can be compared statistically to a control or reference site to detect adverse responses to a site contaminant" (EPA 1997). Measurement endpoints can include measures of exposure (e.g., contaminant concentrations in water or tissues) as well as measures of effect.

It is desirable to have more than one measurement endpoint for each assessment endpoint, thereby providing multiple lines of evidence for the evaluation. However, the primary consideration for selecting measurement endpoints should always be how many and which lines of evidence are appropriate to support risk management decisions at the site. Once it has been determined which lines of evidence are required to answer questions concerning the assessment endpoint, the measurement endpoints by which the questions or test hypotheses will be examined are selected (EPA 1997).

In selecting an appropriate measurement endpoint to represent an assessment endpoint, the following criteria are considered (Suter 1991):

- Corresponds to or is predictive of an assessment endpoint.
- Readily measurable.
- Appropriate to site scale, exposure pathways, and temporal dynamics.
- Diagnostic.
- Broadly applicable.
- Standard.

In particular, measurement endpoints that address both sensitivity and likely exposure to stressors are relevant to management concerns (EPA 1998).

With the selection of measurement endpoints, the conceptual model development is essentially completed. The conceptual model, which is discussed in Section 2.7, is then used to guide the study design and development of data quality objectives (DQOs).

Over a period of several years preceding the Consent Decree, EPA, GE, and other stakeholders discussed available information on contaminants and the Housatonic River ecosystem and determined the assessment endpoints appropriate for the ERA. Past discussions and written comments between these parties demonstrated that while some parties expressed a preference for measurement endpoints using controlled studies, others had a preference for field-based observations and studies. The EPA SIWP addressed both of these preferences by including both a field and a controlled study component for assessment endpoints, where possible and/or appropriate. In addition, GE supplemented the measurement endpoints in the EPA SIWP with studies they conducted independent from agency review, but subject to EPA oversight. GE requested that these studies be incorporated into this ERA, and EPA has done so where the study was determined to be relevant to the assessment endpoint. The assessment and associated measurement endpoints that were used by EPA to evaluate potential ecological risks resulting from PCBs, and possibly other contaminants in the Lower Housatonic River that were established in the EPA SI Work Plan (WESTON 2000), are presented in Table 2.8-1. The independent studies that GE conducted are summarized in Table 2.8-2.

The conceptual model for the site demonstrates the complexity of the ecosystem being evaluated. It was necessary to develop assessment endpoints that were representative of the varying habitats and exposure pathways that exist at the site, and for which there is the potential for differing baseline risk to occur (i.e., a deepwater riverine reach versus a forested floodplain). In addition, many studies conducted as part of this investigation included multiple measurement endpoints in the design. Rather than list these individual measurement endpoints separately, the assessment endpoint and principal measurement endpoints are presented in Table 2.8-1. A listing of all the measurement endpoints included in the design is presented in the SOPs for the individual studies (WESTON 2000). In some cases, the investigators added additional endpoints during the conduct of the study. These are discussed in the individual investigator reports, and, where relevant to the assessment endpoint, in the appropriate assessment endpoint appendix.

Table 2.8-1

Ecological Assessment and Measurement Endpoints

Receptor	Assessment Endpoint	Measurement Endpoint
Benthic Invertebrates	Community structure, survival, growth, and reproduction	Community composition; species richness, abundance, and biomass and other metrics compared with similar metrics at reference locations.
		Sediment Quality Triad evaluation—Evaluation includes benthic community composition, sediment toxicity testing, and sediment chemistry.
		Sediment macroinvertebrate chronic toxicity testing using *Hyalella azteca* to determine survival, growth, and reproduction; and *Chironomus tentans* to determine survival, growth, and emergence. In situ toxicity studies using *C. tentans*, *Daphnia magna*, *H. azteca*, and *Lumbriculus variegatus* to determine survival and growth. (Growth evaluated only in *C. tentans*.) Toxicity Identification Evaluation (TIE) laboratory 24-hour study using *Ceriodaphnia dubia* to determine survival for different porewater fractions of contaminant classes.
		Comparison of sediment chemistry with sediment quality values (SQVs) and tissue chemistry with tissue effects thresholds.
Amphibians	Community condition, survival, reproduction, development, and maturation	Semiquantitative sampling of larval amphibians in breeding habitats with different sediment concentrations of stressors. Endpoints include species richness per habitat type; species abundance; gross pathology; and body, tail, and total length measurements.
		Surveys of vernal pools to quantitate amphibians entering vernal pools and determine breeding behavior and condition; egg laying, hatching success, and larval growth and development; metamorphosis and emigration.

Table 2.8-1

Ecological Assessment and Measurement Endpoints (Continued)

Receptor	Assessment Endpoint	Measurement Endpoint
Amphibians (cont'd)		Amphibian toxicity tests designed with exposure over a gradient of stressor concentrations in site sediment. Toxicity endpoints include morphology of embryos and juveniles, limb development, skin maturation, and tail resorption of *Rana pipiens* and *R. sylvatica*. Gravidity of females; egg count; necrotic eggs; oocyte maturity; sperm count, morphology, and viability; fertilization rate; embryo viability; hatching success; mortality; and teratogenesis of *Rana pipiens* collected from the study area over a contamination gradient and compared with an external control.
		In situ amphibian toxicity study evaluated how multiple stressors (including population density and PCB exposure) affect survival and growth of larval *Rana sylvatica*.
Fish	Survival, growth, and reproduction	Determine the possible extent of adverse effects by comparing the concentrations of COCs in sediment to the concentrations reported in the literature to cause adverse effects on the survival, growth, or reproduction of fish.
		Compare the concentrations of COCs in fish tissues to the concentrations in fish tissues that may result in adverse effects, based on site-specific fish toxicity studies.
		Compare the concentrations of COCs in fish tissues to concentrations documented in the literature to result in adverse effects.
		Evaluate field survey information (fish biomass study, ecological characterization study, and largemouth bass habitat and reproduction study) to qualitatively assess potential effects.
Insectivorous Birds	Survival, growth and reproduction	Reproductive performance of tree swallows (*Tachycineta bicolor*) based on the nest box study conducted in areas of varying stressor sediment concentrations. Parameters for evaluation include nest building, egg presence/absence, number of eggs, and hatching success.
		Comparison of site-specific tissue concentrations in tree swallows with reference area concentrations and with residue effects levels from literature.
		Quantitative comparison of daily intakes based on dietary intake of stressors by tree swallows, American robins *(Turdus migratorius)*, and wood duck *(Aix sponsa)* using site-specific stressor levels in invertebrates and comparison with literature-based effect values.
		American robin productivity within the PSA and reference areas was evaluated and compared to associated PCB exposure. Metrics assessed included clutch size, hatching and fledgling success, and PCB concentrations in robin eggs.

Table 2.8-1

Ecological Assessment and Measurement Endpoints (Continued)

Receptor	Assessment Endpoint	Measurement Endpoint
Piscivorous Birds	Survival, growth, and reproduction	Quantitative comparison of daily intakes based on dietary intake of stressors by belted kingfishers and osprey using site-specific fish tissue concentrations and site-specific stressor levels in other aquatic-related food items (e.g., crayfish and frogs), with literature-based effect values.
		Belted kingfisher nests within and adjacent to the PSA were identified and monitored for productivity (i.e., number of eggs, number of eggs hatched, and fledgling success). Habitat suitability and modeled PCB exposure were also evaluated and related to nest productivity.
Piscivorous Mammals	Survival, growth, and reproduction	Mink toxicity tests using Housatonic River fish in the diet. Toxicity endpoints include body weight, food intake rate, length of gestation, reproductive success (measured by number of females whelping, newborns/female, litter weight, etc.), survival, histopathology, presence/absence of jaw lesions, organ weights, and various biochemical endpoints.
		Quantitative evaluation of mink and otter presence using scent posts and snow tracking. (two separate studies)
		Quantitative comparison of daily intakes based on dietary intake of stressors by mink and river otter using site-specific stressor levels in fish and other aquatic prey with literature-based effect values.
Omnivorous and Carnivorous Mammals	Survival, growth and reproduction	Reproductive evidence in trapped small mammals (e.g., examination of placental scars to determine number of litters, and number/litter).
		Quantitative comparison of daily intakes based on dietary intake of stressors by northern short-tailed shrews and red fox using site-specific stressor levels in soil invertebrates and small mammals with literature-based effect values.
		Demographic characteristics of short-tailed shrew populations were assessed at six locations within the PSA that spanned a range of PCB soil concentrations. Population characteristics measured at each location included survival rate, sex ratio, reproduction and growth rate, and body mass.
Special Status Species (Endangered, Threatened)	Survival, growth, and reproduction	Quantitative comparison of daily intakes based on dietary intake of stressors using site-specific media concentrations and comparison with literature-based effect values.

Table 2.8-2

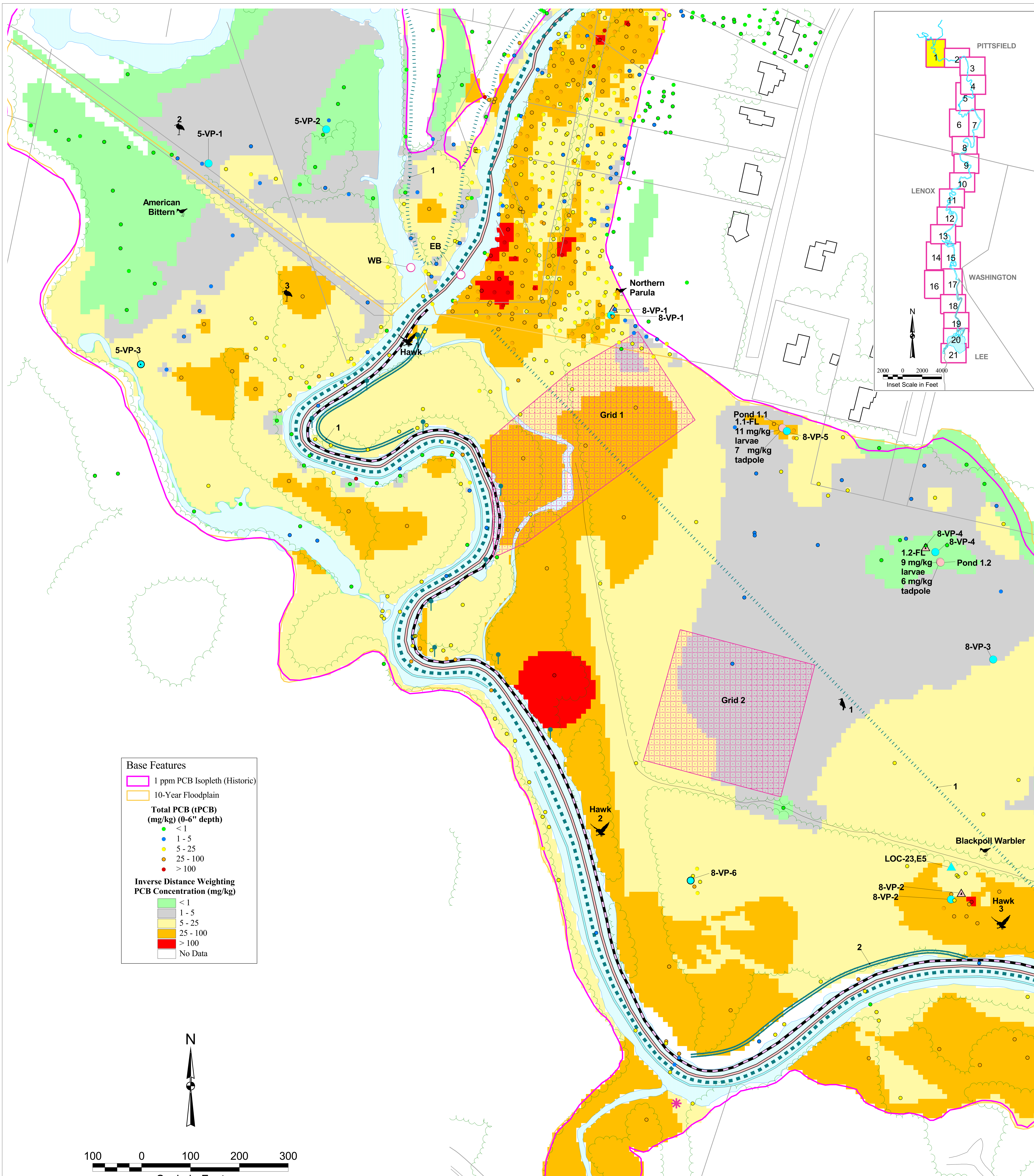
Summary of GE Ecological Studies

Study	Objectives
Robin Productivity in the Housatonic River Watershed (Henning, et al. 2002)	▪ Document reproductive output of robins in the PSA and reference areas. ▪ Evaluate exposure of eggs and young to PCBs. ▪ Evaluate relationships between exposure and reproductive output.
Productivity and Density of Belted Kingfishers on the Housatonic River (Henning 2002)	▪ Evaluate kingfisher productivity in situ in a system with known PCB contamination. ▪ Determine whether estimated PCB dose, habitat quality, phenology, and/or nest density were significant predictors of reproductive success.
Experimental Analysis of the Context-Dependent Effects of Early Life-Stage PCB Exposure on *Rana sylvatica* (Resatarits 2002)	▪ Determine the effects and interactions of PCB exposure and density-dependence on the growth and development of amphibian offspring.
Spatial and Demographic Effects on Tree Swallow Nest Quality and Reproductive Success (Robertson and Jones 2002)	▪ Determine the effects of: (1) inter-nest spacing, (2) proximity to edge, (3) settlement and nest-building date, (4) availability of nesting material, (5) history of the nest-box and nest-box grid, and (6) female and male age, on both nest quality and reproductive success.
Demography of Short-Tailed Shrew Populations Living on Contaminated Sites (Boonstra 2002)	▪ Assess whether PCBs adversely affect population demography of short-tailed shrew living in a natural environment.
Evaluation of Mink – Presence/Absence, Distribution, and Abundance in the Housatonic River Floodplain (BBL 2002)	▪ Qualitatively determine the presence/absence, abundance, and distribution of free-ranging mink in the PSA.
Evaluation of Largemouth Bass Habitat, Population Structure, and Reproduction in the Housatonic River, Massachusetts (R2 2002)	▪ Determine if largemouth bass (LMB) population in the study reach is self-sustaining. ▪ Determine if the LMB population is dependent on tributary recruitment. ▪ Identify which attributes of growth, size-class structure, and reproduction of the LMB population are similar to LMB populations in other systems.
Northern Leopard Frog (*Rana pipiens*) Egg Mass Survey (ARCADIS G&M 2003)	▪ Determine whether adult leopard frogs are failing to reproduce successfully in the field under natural conditions.

Several field surveys were conducted to provide information specifically on species presence. Although field surveys can also be used to assess community condition, the majority of the field surveys (with limited exceptions) were designed for community characterization and were not intended to be used as lines of evidence; therefore, they are not included in Table 2.8-1.

Tissue samples were collected for contaminant analyses for a number of species in support of the ecological exposure assessment, human health risk assessment, and PCB fate and effects modeling. Endpoints typically associated with residue effects range from general toxicity to reproductive effects and lethality. Where comparable literature-based residue effects data were identified through various literature and toxicity database searches, these were incorporated to provide a comparison of site-specific tissue data with literature-based effects levels in the risk assessment to provide additional lines of evidence. Figures 2.8-1 to 2.8-22 present locations for all field studies and biological samples collected in support of the ERA. Surficial soil and sediment PCB concentrations information is also provided.

Although many of the endpoints presented are linked to organism-level effects (e.g., survival and reproduction), these endpoints are expected to be strong indicators of potential local population-level effects (e.g., viability of the benthic community within the Housatonic River study area) (EPA 1992, 1999). Extrapolation from organism-level to population-level effects may be logically achieved based on the predictive nature of the endpoint and/or through the use of process-based models.



Notes:

- Line features representing fish studies are for graphic presentation of the length of reach or area sampled and are not meant to represent actual sample locations.

- Certain sample types such as macroinvertebrates, crayfish, predators, and shredders are located too close to one another to be distinguishable at map scale and thus for graphic presentation, have symbols provided with separation distance.

- In most cases, sample locations were derived using GPS and thus have varying levels of inherent horizontal accuracy, depending on equipment employed at the time of the particular effort. These points are presented herein over base features and hydrography derived from orthophotography, which can have a higher level of geographic accuracy and may not represent specific hydrologic conditions at the time the samples were collected.

- Except where horizontal coordinates are provided in the GE database, the locations of GE samples have been approximated from cited references.

- All tissue concentrations provided are tPCB mg/kg wet weight.

LEGEND

Invertebrates
EPA Ecological Risk Assessment - Weston, 2003
Macroinvertebrate Sample Location
Crayfish Sample Location
Sediment Quality Station
Benthic Predator Sample Location
Benthic Shredder Sample Location
EPA Sediment Toxicity Study - Burton/WSU, 2003
Toxicity Test Location
EPA Ecological Characterization - Woodlot, 2002
Earthworm/Leaf Litter Tissue Sample Location
Dragonfly Survey Transect Location

Reptiles/Amphibians
EPA Bullfrog Tissue Study - Weston, 2003
Bullfrog Tissue Sample Location
EPA Wood Frog Study - Fort, 2000
Wood Frog Target Site (Vernal Pools)
EPA Leopard Frog Study - Fort, 2000
Leopard Frog Target Site
GE Housatonic River Study - Stewart , 1980,1982
Snapping Turtle Sample Location
Bullfrog Sample Location
GE Wood Frog Study - Old Dominion Univ., 2002
Wood Frog Enclosure
Wood Frog Egg Mass Location
GE Leopard Frog Study - ARCADIS, 2003
Leopard Frog Egg Mass Location

Fish
EPA Ecological Characterization Report - Woodlot, 1998
Fish Characterization and Tissue Sample Reach
Fish Sample Area
EPA Fish Biomass Study - Woodlot, 2000
Fish Sample Reach
Fish Sample Area
EPA Koi/Goldfish Study, USFWS, 2003
Fish Sample Area
EPA Mink Feeding Study, Bursian/Tillitt, 2003
Fish Sample Area
EPA Bass/Bluegill Study, Tillitt, 2003
Fish Sample Area
GE Housatonic River Study - Stewart, 1980, 1982
Fish Sample Reach
GE Adult Fish Study - BBL, 1990, 1998, & 2002
Fish Sample Reach
GE Biennial YOY Fish Study - BBL, 1994-2002
Fish Sample Reach
GE Bass Study - R2, 2002
Bass Index Sites
Bass Study Sites
Bass Transect Locations

Birds
EPA - Ecological Characterization Report - Woodlot, 2003
Wood Duck Embryo Sample Location
Waterfowl Trap Location
Wood Duck Nest Box Location
Rare Bird Survey
Owl/Hawk Survey
Marsh Bird Survey
Forest Bird Survey
Kingfisher Nest Location
EPA - Tree Swallow Report - Custer, 2002
Swallow Box Location
Bird Tissue Sample Location
GE - Robin Report - ARCADIS G&M, 2002
Robin Tissue Sample Location
GE - Kingfisher Report - ARCADIS G&M, 2002
Kingfisher Nest Location

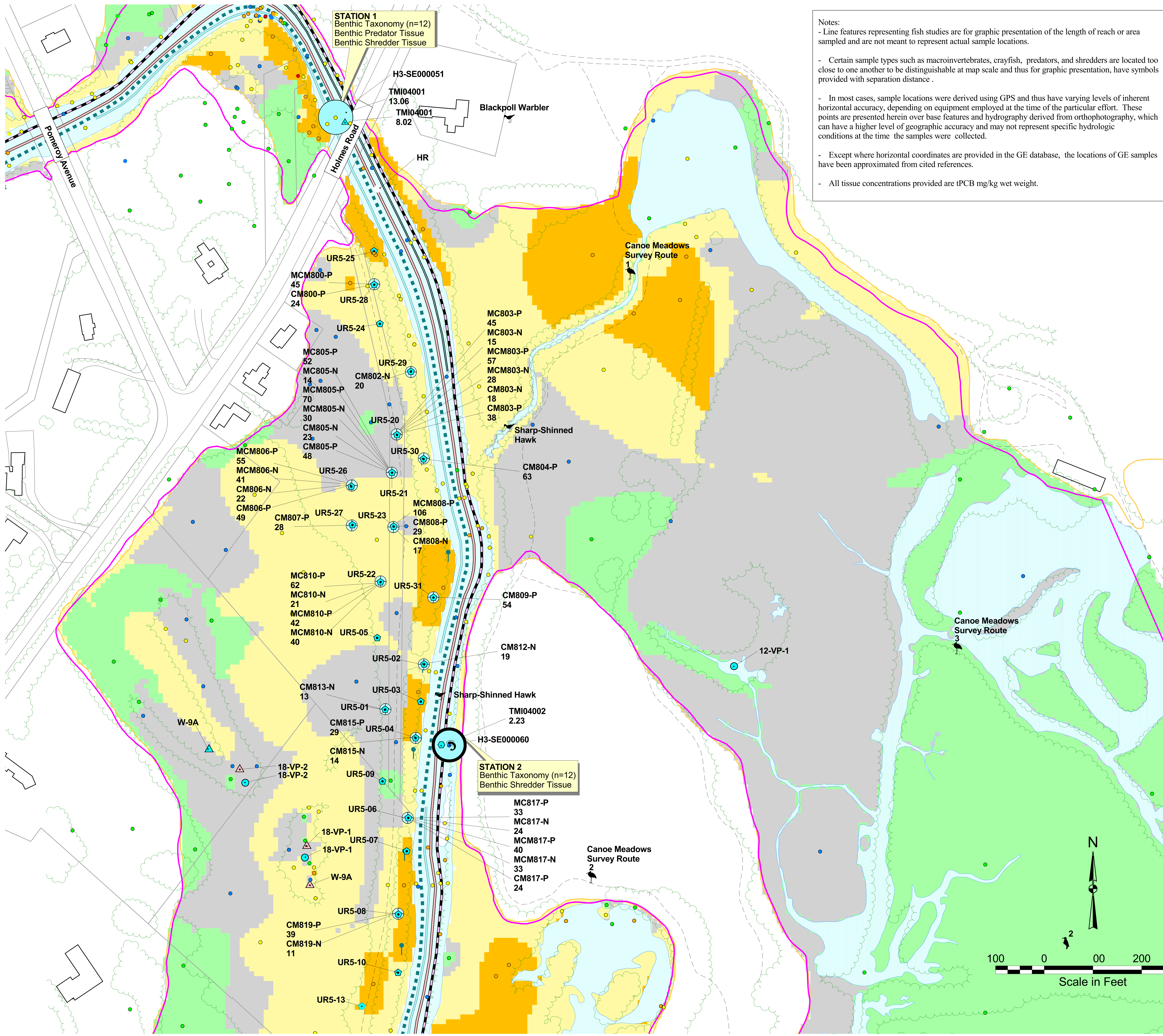
Mammals
EPA - Ecological Characterization Report - Woodlot, 2003
Bat Transect
Short-Tailed Shrew Tissue Sample Location
White-Footed Mouse Tissue Sample Location
Mink/Otter Scent Post Location
Snow Track Transects
GE Small Mammal Study - Boonstra, 2002
Small Mammal Grid Area
GE Otter/Mink Report - ARCADIS G&M, 2003
Otter Sprainting Station Location
Otter Camera Location
Otter Track Location
Mink Track Location
Mink Searching Area

Ecological Risk Assessment
GE/Housatonic River Project Site
Rest of River

FIGURE 2.8-1
BIOLOGICAL SAMPLE LOCATIONS IN THE PRIMARY STUDY AREA
(TILE 1 OF 21)

Ecological Risk Assessment
GE/Housatonic River Project Site
Rest of River

FIGURE 2.8-2
BIOLOGICAL SAMPLE LOCATIONS IN THE PRIMARY STUDY AREA
(TILE 2 OF 21)



Notes:

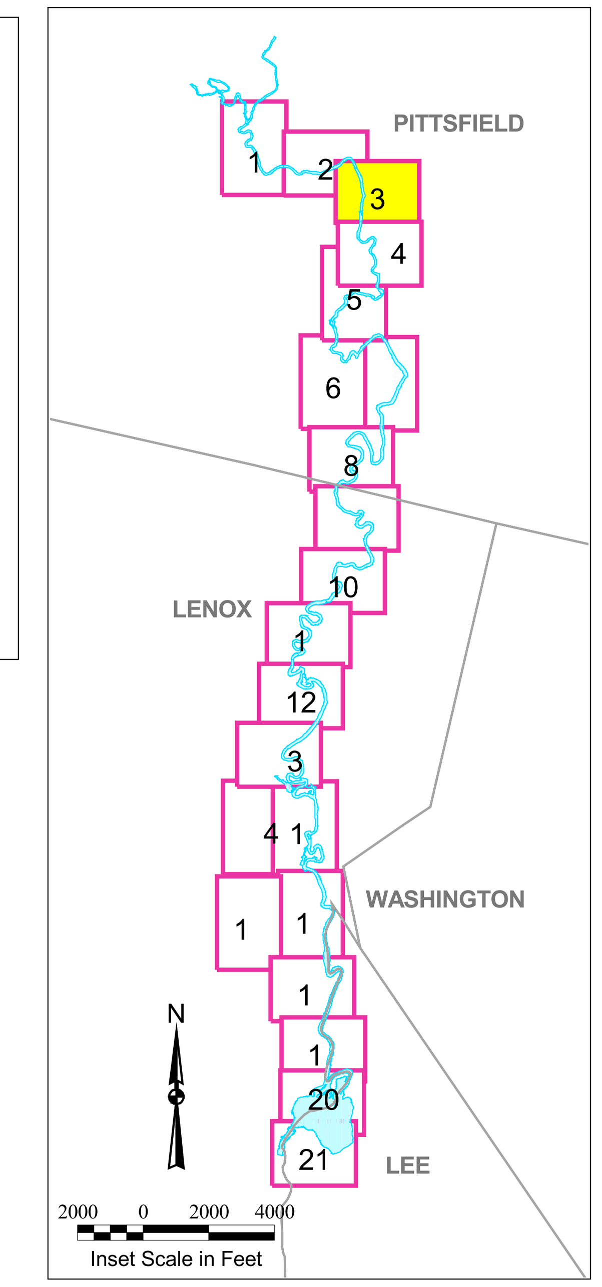
- Line features representing fish studies are for graphic presentation of the length of reach or area sampled and are not meant to represent actual sample locations.

- Certain sample types such as macroinvertebrates, crayfish, predators, and shredders are located too close to one another to be distinguishable at map scale and thus for graphic presentation, have symbols provided with separation distance .

- In most cases, sample locations were derived using GPS and thus have varying levels of inherent horizontal accuracy, depending on equipment employed at the time of the particular effort. These points are presented herein over base features and hydrography derived from orthophotography, which can have a higher level of geographic accuracy and may not represent specific hydrologic conditions at the time the samples were collected.

- Except where horizontal coordinates are provided in the GE database, the locations of GE samples have been approximated from cited references.

- All tissue concentrations provided are tPCB mg/kg wet weight.



Base Features

1 ppm PCB Isopleth (Historic)

10-Year Floodplain

Total PCB (tPCB) (mg/kg) (0-6" depth)

- < 1
- 1 - 5
- 5 - 25
- 25 - 100
- > 100

Inverse Distance Weighting PCB Concentration (mg/kg)

- < 1
- 1 - 5
- 5 - 25
- 25 - 100
- > 100
- No Data

LEGEND

Invertebrates

EPA Ecological Risk Assessment - Weston, 2003
Macroinvertebrate Sample Location
Crayfish Sample Location
Sediment Quality Station
Benthic Predator Sample Location
Benthic Shredder Sample Location

EPA Sediment Toxicity Study - Burton/WSU, 2003
Toxicity Test Location

EPA Ecological Characterization - Woodlot, 2002
Earthworm/Leaf Litter Tissue Sample Location
Dragonfly Survey Transect Location

Reptiles/Amphibians

EPA Bullfrog Tissue Study - Weston, 2003
Bullfrog Tissue Sample Location

EPA Wood Frog Study - Fort, 2000
Wood Frog Target Site (Vernal Pools)

EPA Leopard Frog Study - Fort, 2000
Leopard Frog Target Site

GE Housatonic River Study - Stewart, 1980, 1982
Snapping Turtle Sample Location
Bullfrog Sample Location

GE Wood Frog Study - Old Dominion Univ., 2002
Wood Frog Enclosure
Wood Frog Egg Mass Location

GE Leopard Frog Study - ARCADIS, 2003
Leopard Frog Egg Mass Location

Fish

EPA Ecological Characterization Report - Woodlot, 1998
Fish Characterization and Tissue Sample Reach
Fish Sample Area

EPA Fish Biomass Study - Woodlot, 2000
Fish Sample Reach
Fish Sample Area

EPA Koi/Goldfish Study, USFWS, 2003
Fish Sample Area

EPA Mink Feeding Study, Bursian/Tillitt, 2003
Fish Sample Area

EPA Bass/Bluegill Study, Tillitt, 2003
Fish Sample Area

GE Housatonic River Study - Stewart, 1980, 1982,
Fish Sample Reach

GE Adult Fish Study - BBL, 1990, 1998, & 2002
Fish Sample Reach

GE Biennial YOY Fish Study - BBL, 1994-2002
Fish Sample Reach

GE Bass Study - R2, 2002
Bass Index Sites
Bass Study Sites
Bass Transect Locations

Birds

EPA - Ecological Characterization Report - Woodlot, 2003
Wood Duck Embryo Sample Location
Waterfowl Trap Location
Wood Duck Nest Box Location
Rare Bird Survey
Owl/Hawk Survey
Marsh Bird Survey
Forest Bird Survey
Kingfisher Nest Location

EPA - Tree Swallow Report - Custer, 2002
Swallow Box Location
Bird Tissue Sample Location

GE - Robin Report - ARCADIS G&M, 2002
Robin Tissue Sample Location

GE - Kingfisher Report - ARCADIS G&M, 2002
Kingfisher Nest Location

Mammals

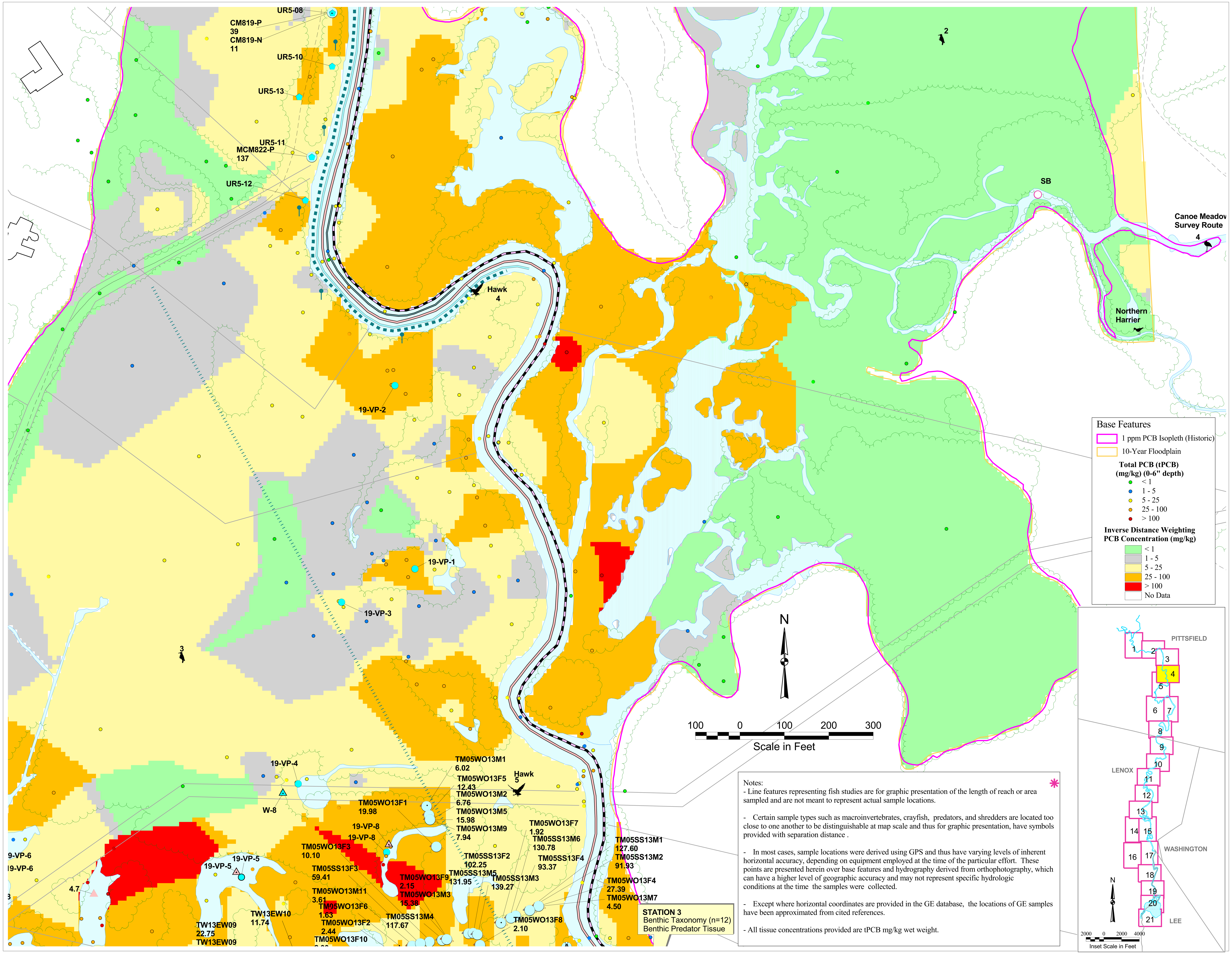
EPA - Ecological Characterization Report - Woodlot, 2003
Bat Transect
Short-Tailed Shrew Tissue Sample Location
White-Footed Mouse Tissue Sample Location
Mink/Otter Scent Post Location
Snow Track Transects

GE Small Mammal Study - Boonstra, 2002
Small Mammal Grid Area

GE Otter/Mink Report - ARCADIS G&M, 2003
Otter Spraining Station Location
Otter Camera Location
Otter Track Location
Mink Track Location
Mink Searching Area

Ecological Risk Assessment
GE/Housatonic River Project Site
Rest of River

FIGURE 2.8-3
BIOLOGICAL SAMPLE LOCATIONS IN THE PRIMARY STUDY AREA (TILE 3 OF 21)



Notes:

- Line features representing fish studies are for graphic presentation of the length of reach or area sampled and are not meant to represent actual sample locations.
- Certain sample types such as macroinvertebrates, crayfish, predators, and shredders are located too close to one another to be distinguishable at map scale and thus for graphic presentation, have symbols provided with separation distance .
- In most cases, sample locations were derived using GPS and thus have varying levels of inherent horizontal accuracy, depending on equipment employed at the time of the particular effort. These points are presented herein over base features and hydrography derived from orthophotography, which can have a high level of geographic accuracy and may not represent specific hydrologic conditions at the time the samples were collected.
- Except where horizontal coordinates are provided in the GE database, the locations of GE samples have been approximated from cited references.
- All tissue concentrations provided are tPCB mg/kg wet weight.

LEGEND

Invertebrates
EPA Ecological Risk Assessment - Weston, 2003
Macroinvertebrate Sample Location
Crayfish Sample Location
Sediment Quality Station
Benthic Predator Sample Location
Benthic Shredder Sample Location
EPA Sediment Toxicity Study - Burton/WSU, 2003
Toxicity Test Location
EPA Ecological Characterization - Woodlot, 2002
Earthworm/Leaf Litter Tissue Sample Location
Dragonfly Survey Transect Location

Reptiles/Amphibians
EPA Bullfrog Tissue Study - Weston, 2003
Bullfrog Tissue Sample Location
EPA Wood Frog Study - Fort, 2000
Wood Frog Target Site (Vernal Pools)
EPA Leopard Frog Study - Fort, 2000
Leopard Frog Target Site
GE Housatonic River Study - Stewart, 1980, 1982
Snapping Turtle Sample Location
Bullfrog Sample Location
GE Wood Frog Study - Old Dominion Univ., 2002
Wood Frog Enclosure
Wood Frog Egg Mass Location
GE Leopard Frog Study - ARCADIS, 2003
Leopard Frog Egg Mass Location

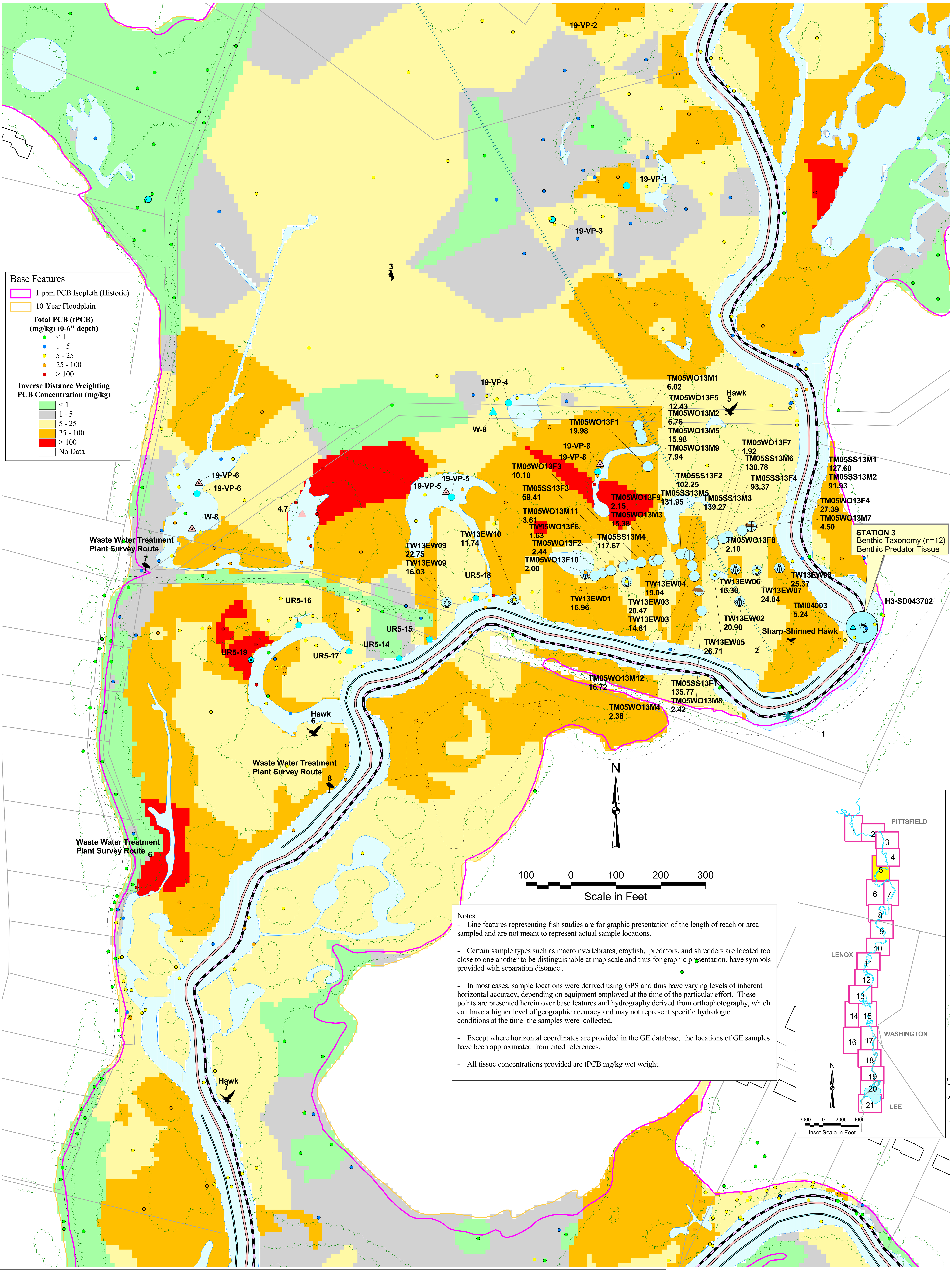
Fish
EPA Ecological Characterization Report - Woodlot, 1998
Fish Characterization and Tissue Sample Reach
Fish Sample Area
EPA Fish Biomass Study - Woodlot, 2000
Fish Sample Reach
Fish Sample Area
EPA Koi/Goldfish Study, USFWS, 2003
Fish Sample Area
EPA Mink Feeding Study, Bursian/Tillitt, 2003
Fish Sample Area
EPA Bass/Bluegill Study, Tillitt, 2003
Fish Sample Area
GE Housatonic River Study - Stewart, 1980, 1982,
Fish Sample Reach
GE Adult Fish Study - BBL, 1990, 1998, & 2002
Fish Sample Reach
GE Biennial YOY Fish Study - BBL, 1994-2002
Fish Sample Reach
GE Bass Study - R2, 2002
Bass Index Sites
Bass Study Sites
Bass Transect Locations

Birds
EPA - Ecological Characterization Report - Woodlot, 2003
Wood Duck Embryo Sample Location
Waterfowl Trap Location
Wood Duck Nest Box Location
Rare Bird Survey
Owl/Hawk Survey
Marsh Bird Survey
Forest Bird Survey
Kingfisher Nest Location
EPA - Tree Swallow Report - Custer, 2002
Swallow Box Location
Bird Tissue Sample Location
GE - Robin Report - ARCADIS G&M, 2002
Robin Tissue Sample Location
GE - Kingfisher Report - ARCADIS G&M, 2002
Kingfisher Nest Location

Mammals
EPA - Ecological Characterization Report - Woodlot, 2003
Bat Transect
Short-Tailed Shrew Tissue Sample Location
White-Footed Mouse Tissue Sample Location
Mink/Otter Scent Post Location
Snow Track Transects
GE Small Mammal Study - Boonstra, 2002
Small Mammal Grid Area
GE Otter/Mink Report - ARCADIS G&M, 2003
Otter Spraining Station Location
Otter Camera Location
Otter Track Location
Mink Track Location
Mink Searching Area

Ecological Risk Assessment
GE/Housatonic River Project Site
Rest of River

FIGURE 2.8-4
BIOLOGICAL SAMPLE LOCATIONS IN THE PRIMARY STUDY AREA
(TILE 4 OF 21)



Notes:

- Line features representing fish studies are for graphic presentation of the length of reach or area sampled and are not meant to represent actual sample locations.

- Certain sample types such as macroinvertebrates, crayfish, predators, and shredders are located too close to one another to be distinguishable at map scale and thus for graphic presentation, have symbols provided with separation distance.

- In most cases, sample locations were derived using GPS and thus have varying levels of inherent horizontal accuracy, depending on equipment employed at the time of the particular effort. These points are presented herein over base features and hydrography derived from orthophotography, which can have a higher level of geographic accuracy and may not represent specific hydrologic conditions at the time the samples were collected.

- Except where horizontal coordinates are provided in the GE database, the locations of GE samples have been approximated from cited references.

- All tissue concentrations provided are tPCB mg/kg wet weight.

LEGEND

Invertebrates
EPA Ecological Risk Assessment - Weston, 2003
- Macroinvertebrate Sample Location
- Crayfish Sample Location
- Sediment Quality Station
- Benthic Predator Sample Location
- Benthic Shredder Sample Location

EPA Sediment Toxicity Study - Burton/WSU, 2003
- Toxicity Test Location

EPA Ecological Characterization - Woodlot, 2002
- Earthworm/Leaf Litter Tissue Sample Location
- Dragonfly Survey Transect Location

Reptiles/Amphibians
EPA Bullfrog Tissue Study - Weston, 2003
- Bullfrog Tissue Sample Location

EPA Wood Frog Study - Fort, 2000
- Wood Frog Target Site (Vernal Pools)

EPA Leopard Frog Study - Fort, 2000
- Leopard Frog Target Site

GE Housatonic River Study - Stewart, 1980,1982
- Snapping Turtle Sample Location
- Bullfrog Sample Location

GE Wood Frog Study - Old Dominion Univ., 2002
- Wood Frog Enclosure
- Wood Frog Egg Mass Location

GE Leopard Frog Study - ARCADIS, 2003
- Leopard Frog Egg Mass Location

Fish
EPA Ecological Characterization Report - Woodlot, 1998
- Fish Characterization and Tissue Sample Reach
- Fish Sample Area
- Fish Sample Area

EPA Fish Biomass Study - Woodlot, 2000
- Fish Sample Reach
- Fish Sample Area

EPA Koi/Goldfish Study, USFWS, 2003
- Fish Sample Area

EPA Mink Feeding Study, Bursian/Tillitt, 2003
- Fish Sample Area

EPA Bass/Bluegill Study, Tillitt, 2003
- Fish Sample Area

GE Housatonic River Study - Stewart, 1980, 1982
- Fish Sample Reach

GE Adult Fish Study - BBL, 1990, 1998, & 2002
- Fish Sample Reach

GE Biennial YOY Fish Study - BBL, 1994-2002
- Fish Sample Reach

GE Bass Study - R2, 2002
- Bass Index Sites
- Bass Study Sites
- Bass Transect Locations

Birds
EPA - Ecological Characterization Report - Woodlot, 2003
- Wood Duck Embryo Sample Location
- Waterfowl Trap Location
- Wood Duck Nest Box Location
- Rare Bird Survey
- Owl/Hawk Survey
- Marsh Bird Survey
- Forest Bird Survey
- Kingfisher Nest Location

EPA - Tree Swallow Report - Custer, 2002
- Swallow Box Location
- Bird Tissue Sample Location

GE - Robin Report - ARCADIS G&M, 2002
- Robin Tissue Sample Location

GE - Kingfisher Report - ARCADIS G&M, 2002
- Kingfisher Nest Location

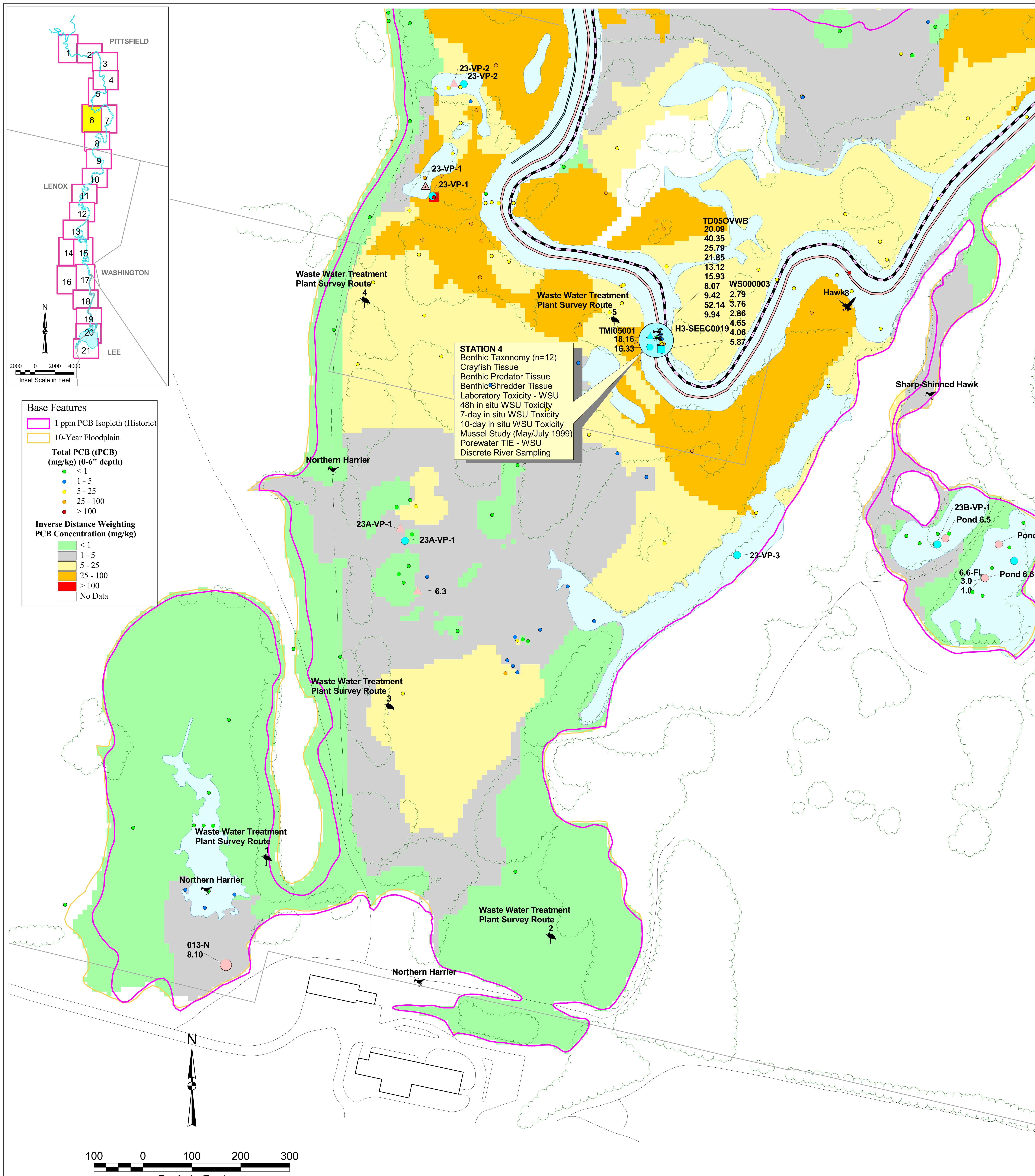
Mammals
EPA - Ecological Characterization Report - Woodlot, 2003
- Bat Transect
- Short-Tailed Shrew Tissue Sample Location
- White-Footed Mouse Tissue Sample Location
- Mink/Otter Scent Post Location
- Snow Track Transects

GE Small Mammal Study - Boonstra, 2002
- Small Mammal Grid Area

GE Otter/Mink Report - ARCADIS G&M, 2003
- Otter Sprainting Station Location
- Otter Camera Location
- Otter Track Location
- Mink Track Location
- Mink Searching Area

Ecological Risk Assessment
GE/Housatonic River Project Site
Rest of River

FIGURE 2.8-5
BIOLOGICAL SAMPLE LOCATIONS IN THE PRIMARY STUDY AREA
(TILE 5 OF 21)



Notes:

- Line features representing fish studies are for graphic presentation of the length of reach or area sampled and are not meant to represent actual sample locations.

- Certain sample types such as macroinvertebrates, crayfish, predators, and shredders are located too close to one another to be distinguishable at map scale and thus for graphic presentation, have symbols provided with separation distance .

- In most cases, sample locations were derived using GPS and thus have varying levels of inherent horizontal accuracy, depending on equipment employed at the time of the particular effort. These points are presented herein over base features and hydrography derived from orthophotography, which can have a higher level of geographic accuracy and may not represent specific hydrologic conditions at the time the samples were collected.

- Except where horizontal coordinates are provided in the GE database, the locations of GE samples have been approximated from cited references.

- All tissue concentrations provided are tPCB mg/kg wet weight.

LEGEND

Invertebrates
EPA Ecological Risk Assessment - Weston, 2003
Macroinvertebrate Sample Location
Crayfish Sample Location
Sediment Quality Station
Benthic Predator Sample Location
Benthic Shredder Sample Location
EPA Sediment Toxicity Study - Burton/WSU, 2003
Toxicity Test Location
EPA Ecological Characterization - Woodlot, 2002
Earthworm/Leaf Litter Tissue Sample Location
Dragonfly Survey Transect Location

Reptiles/Amphibians
EPA Bullfrog Tissue Study - Weston, 2003
Bullfrog Tissue Sample Location
EPA Wood Frog Study - Fort, 2000
Wood Frog Target Site (Vernal Pools)
EPA Leopard Frog Study - Fort, 2000
Leopard Frog Target Site
GE Housatonic River Study - Stewart , 1980,1982
Snapping Turtle Sample Location
Bullfrog Sample Location
GE Wood Frog Study - Old Dominion Univ., 2002
Wood Frog Enclosure
Wood Frog Egg Mass Location
GE Leopard Frog Study - ARCADIS, 2003
Leopard Frog Egg Mass Location

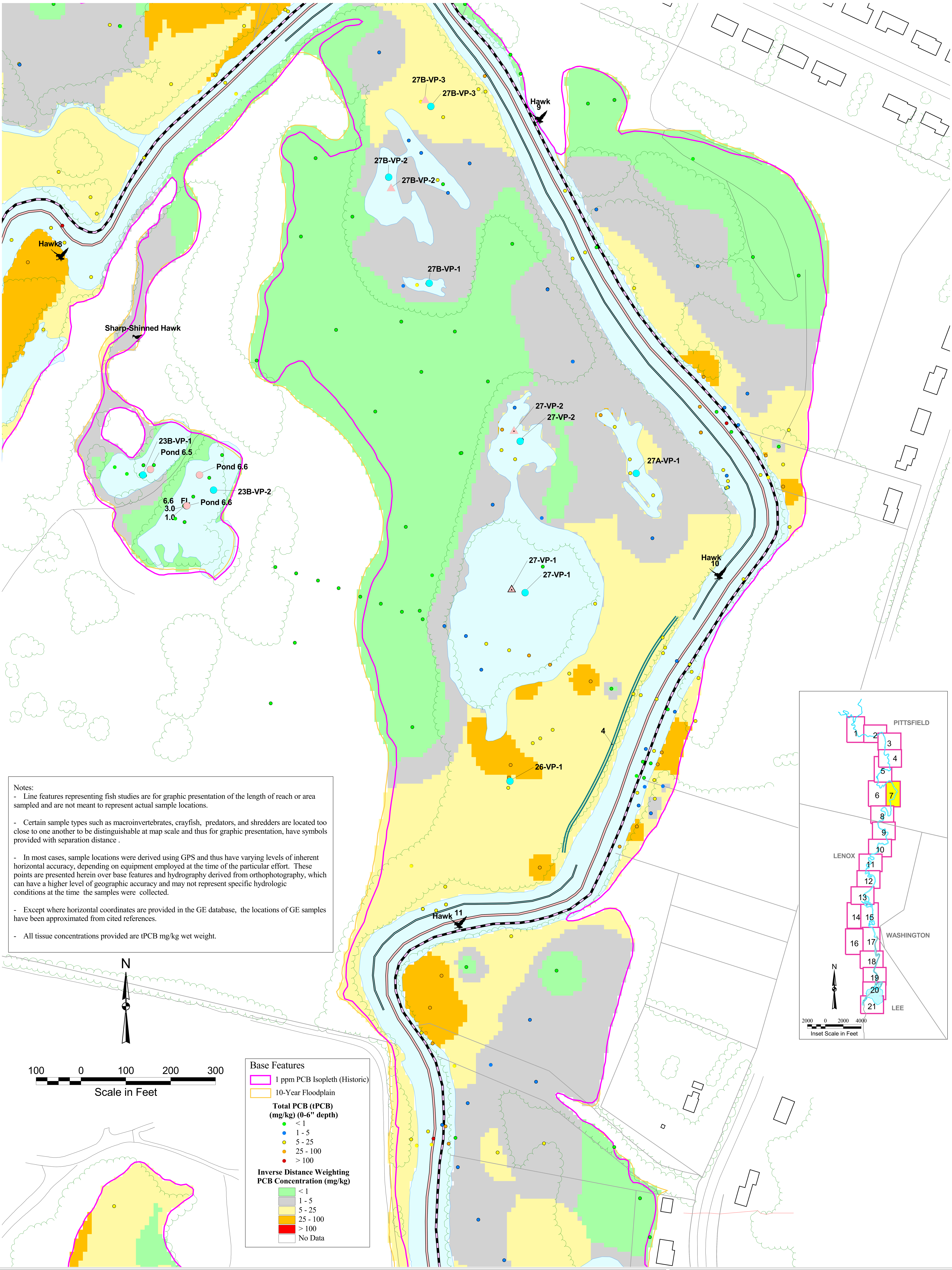
Fish
EPA Ecological Characterization Report - Woodlot, 1998
Fish Characterization and Tissue Sample Reach
Fish Sample Area
EPA Fish Biomass Study - Woodlot, 2000
Fish Sample Reach
Fish Sample Area
EPA Koi/Goldfish Study, USFWS, 2003
Fish Sample Area
EPA Mink Feeding Study, Bursian/Tillitt, 2003
Fish Sample Area
EPA Bass/Bluegill Study, Tillitt, 2003
Fish Sample Area
GE Housatonic River Study - Stewart, 1980, 1982
Fish Sample Reach
GE Adult Fish Study - BBL, 1990, 1998, & 2002
Fish Sample Reach
GE Biennial YOY Fish Study - BBL, 1994-2002
Fish Sample Reach
GE Bass Study - R2, 2002
Bass Index Sites
Bass Study Sites
Bass Transect Locations

Birds
EPA - Ecological Characterization Report - Woodlot, 2003
Wood Duck Embryo Sample Location
Waterfowl Trap Location
Wood Duck Nest Box Location
Rare Bird Survey
Owl/Hawk Survey
Marsh Bird Survey
Forest Bird Survey
Kingfisher Nest Location
EPA - Tree Swallow Report - Custer, 2002
Swallow Box Location
Bird Tissue Sample Location
GE - Robin Report - ARCADIS G&M, 2002
Robin Tissue Sample Location
GE - Kingfisher Report - ARCADIS G&M, 2002
Kingfisher Nest Location

Mammals
EPA - Ecological Characterization Report - Woodlot, 2003
Bat Transect
Short-Tailed Shrew Tissue Sample Location
White-Footed Mouse Tissue Sample Location
Mink/Otter Scent Post Location
Snow Track Transects
GE Small Mammal Study - Boonstra, 2002
Small Mammal Grid Area
GE Otter/Mink Report - ARCADIS G&M, 2003
Otter Sprainting Station Location
Otter Camera Location
Otter Track Location
Mink Track Location
Mink Searching Area

Ecological Risk Assessment
GE/Housatonic River Project Site
Rest of River

FIGURE 2.8-6
BIOLOGICAL SAMPLE LOCATIONS IN THE PRIMARY STUDY AREA
(TILE 6 OF 21)



LEGEND

Invertebrates

EPA Ecological Risk Assessment - Weston, 2003
- Macroinvertebrate Sample Location
- Crayfish Sample Location
- Sediment Quality Station
- Benthic Predator Sample Location
- Benthic Shredder Sample Location

EPA Sediment Toxicity Study - Burton/WSU, 2003
- Toxicity Test Location

EPA Ecological Characterization - Woodlot, 2002
- Earthworm/Leaf Litter Tissue Sample Location
- Dragonfly Survey Transect Location

Reptiles/Amphibians

EPA Bullfrog Tissue Study - Weston, 2003
- Bullfrog Tissue Sample Location

EPA Wood Frog Study - Fort, 2000
- Wood Frog Target Site (Vernal Pools)

EPA Leopard Frog Study - Fort, 2000
- Leopard Frog Target Site

GE Housatonic River Study - Stewart , 1980,1982
- Snapping Turtle Sample Location
- Bullfrog Sample Location

GE Wood Frog Study - Old Dominion Univ., 2002
- Wood Frog Enclosure
- Wood Frog Egg Mass Location

GE Leopard Frog Study - ARCADIS, 2003
- Leopard Frog Egg Mass Location

Fish

EPA Ecological Characterization Report - Woodlot, 1998
- Fish Characterization and Tissue Sample Reach
- Fish Sample Area

EPA Fish Biomass Study - Woodlot, 2000
- Fish Sample Reach
- Fish Sample Area

EPA Koi/Goldfish Study, USFWS, 2003
- Fish Sample Area

EPA Mink Feeding Study, Bursian/Tillitt, 2003
- Fish Sample Area

EPA Bass/Bluegill Study, Tillitt, 2003
- Fish Sample Area

GE Housatonic River Study - Stewart, 1980, 1982
- Fish Sample Reach

GE Adult Fish Study - BBL, 1990, 1998, & 2002
- Fish Sample Reach

GE Biennial YOY Fish Study - BBL, 1994-2002
- Fish Sample Reach

GE Bass Study - R2, 2002
- Bass Index Sites
- Bass Study Sites
- Bass Transect Locations

Birds

EPA - Ecological Characterization Report - Woodlot, 2003
- Wood Duck Embryo Sample Location
- Waterfowl Trap Location
- Wood Duck Nest Box Location
- Rare Bird Survey
- Owl/Hawk Survey
- Marsh Bird Survey
- Forest Bird Survey
- Kingfisher Nest Location

EPA - Tree Swallow Report - Custer, 2002
- Swallow Box Location
- Bird Tissue Sample Location

GE - Robin Report - ARCADIS G&M, 2002
- Robin Tissue Sample Location

GE - Kingfisher Report - ARCADIS G&M, 2002
- Kingfisher Nest Location

Mammals

EPA - Ecological Characterization Report - Woodlot, 2003
- Bat Transect
- Short-Tailed Shrew Tissue Sample Location
- White-Footed Mouse Tissue Sample Location
- Mink/Otter Scent Post Location
- Snow Track Transects

GE Small Mammal Study - Boonstra, 2002
- Small Mammal Grid Area

GE Otter/Mink Report - ARCADIS G&M, 2003
- Otter Sprainting Station Location
- Otter Camera Location
- Otter Track Location
- Mink Track Location
- Mink Searching Area

Ecological Risk Assessment
GE/Housatonic River Project Site
Rest of River

FIGURE 2.8-7
BIOLOGICAL SAMPLE LOCATIONS IN THE PRIMARY STUDY AREA (TILE 7 OF 21)

| c:\gepitt\era_060204\era_biotia__21tiles_bigfonts.apr | T7 | c:\gepitt\era_060204\exports\in\fig2-8-7_101504.eps | 10:40 AM, 10/28/2004 |

Ecological Risk Assessment
GE/Housatonic River Project Site
Rest of River

FIGURE 2.8-8
BIOLOGICAL SAMPLE LOCATIONS IN THE PRIMARY STUDY AREA (TILE 8 OF 21)

c:\gepitt\era_060204\era_biotia__21tiles_bigfonts.apr | T8 | c:\gepitt\era_060204\exports\in\fig2-8-8_101504.eps | 11:11 AM, 10/27/2004 |

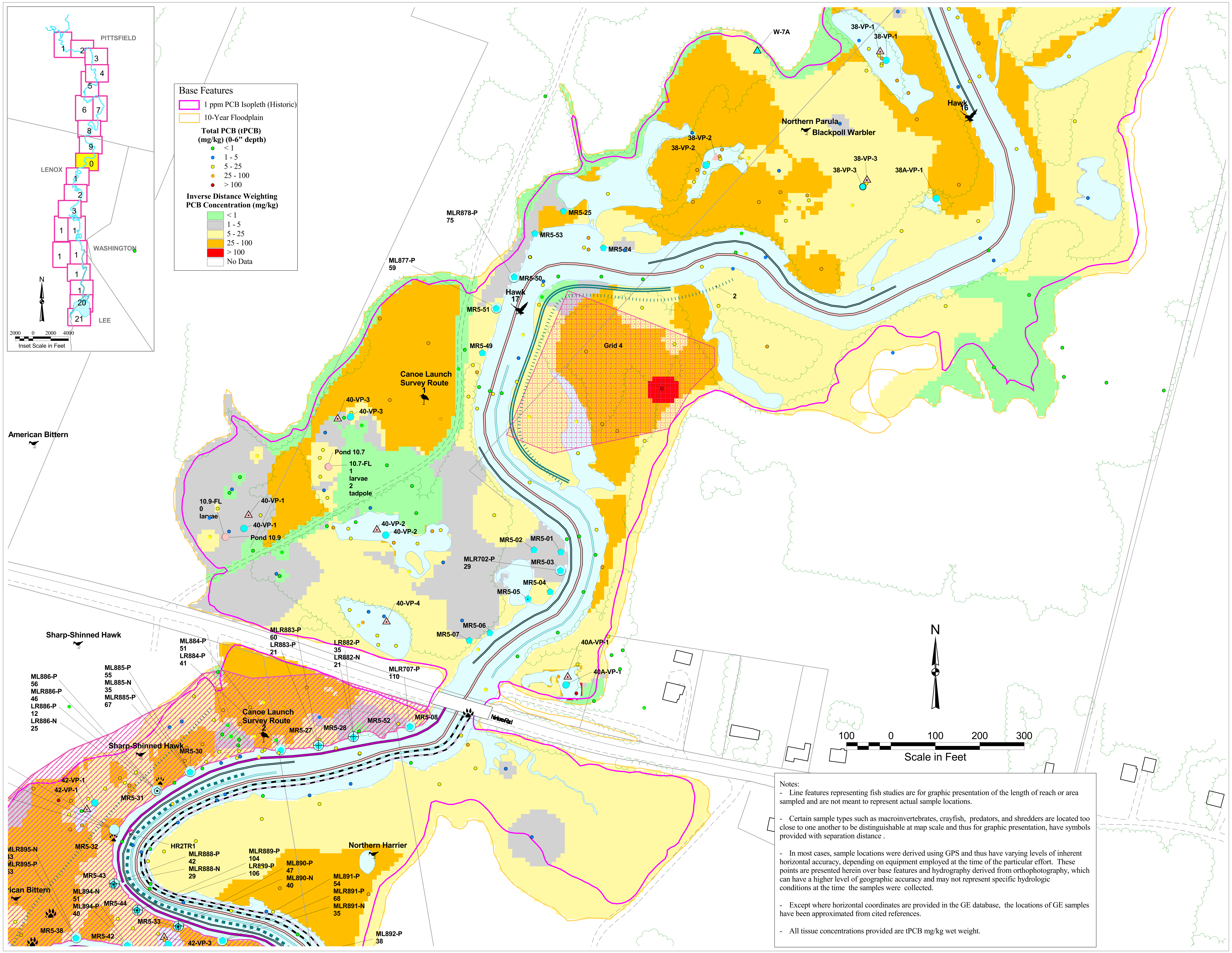
Notes:

- Line features representing fish studies are for graphic presentation of the length of reach or area sampled and are not meant to represent actual sample locations.
- Certain sample types such as macroinvertebrates, crayfish, predators, and shredders are located too close to one another to be distinguishable at map scale and thus for graphic presentation, have symbols provided with separation distance .
- In most cases, sample locations were derived using GPS and thus have varying levels of inherent horizontal accuracy, depending on equipment employed at the time of the particular effort. These points are presented herein over base features and hydrography derived from orthophotography, which can have a higher level of geographic accuracy and may not represent specific hydrologic conditions at the time the samples were collected.
- Except where horizontal coordinates are provided in the GE database, the locations of GE samples have been approximated from cited references.
- All tissue concentrations provided are tPCB mg/kg wet weight.

Ecological Risk Assessment
GE/Housatonic River Project Site
Rest of River

FIGURE 2.8-9
BIOLOGICAL SAMPLE LOCATIONS IN THE PRIMARY STUDY AREA (TILE 9 OF 21)

c:\gepitt\era_060204\era_biotia__21tiles_bigfonts.apr | T9 | c:\gepitt\era_060204\exports\in\fig2-8-9_101504.eps | 11:12 AM, 10/27/2004 |



LEGEND

Invertebrates

EPA Ecological Risk Assessment - Weston, 2003
- Macroinvertebrate Sample Location
- Crayfish Sample Location
- Sediment Quality Station
- Benthic Predator Sample Location
- Benthic Shredder Sample Location

EPA Sediment Toxicity Study - Burton/WSU, 2003
- Toxicity Test Location

EPA Ecological Characterization - Woodlot, 2002
- Earthworm/Leaf Litter Tissue Sample Location
- Dragonfly Survey Transect Location

Reptiles/Amphibians

EPA Bullfrog Tissue Study - Weston, 2003
- Bullfrog Tissue Sample Location

EPA Wood Frog Study - Fort, 2000
- Wood Frog Target Site (Vernal Pools)

EPA Leopard Frog Study - Fort, 2000
- Leopard Frog Target Site

GE Housatonic River Study - Stewart, 1980, 1982
- Snapping Turtle Sample Location
- Bullfrog Sample Location

GE Wood Frog Study - Old Dominion Univ., 2002
- Wood Frog Enclosure
- Wood Frog Egg Mass Location

GE Leopard Frog Study - ARCADIS, 2003
- Leopard Frog Egg Mass Location

Fish

EPA Ecological Characterization Report - Woodlot, 1998
- Fish Characterization and Tissue Sample Reach
- Fish Sample Area

EPA Fish Biomass Study - Woodlot, 2000
- Fish Sample Reach
- Fish Sample Area

EPA Koi/Goldfish Study, USFWS, 2003
- Fish Sample Area

EPA Mink Feeding Study, Bursian/Tillitt, 2003
- Fish Sample Area

EPA Bass/Bluegill Study, Tillitt, 2003
- Fish Sample Area

GE Housatonic River Study - Stewart, 1980, 1982,
- Fish Sample Reach

GE Adult Fish Study - BBL, 1990, 1998, & 2002
- Fish Sample Reach

GE Biennial YOY Fish Study - BBL, 1994-2002
- Fish Sample Reach

GE Bass Study - R2, 2002
- Bass Index Sites
- Bass Study Sites
- Bass Transect Locations

Birds

EPA - Ecological Characterization Report - Woodlot, 2003
- Wood Duck Embryo Sample Location
- Waterfowl Trap Location
- Wood Duck Nest Box Location
- Rare Bird Survey
- Owl/Hawk Survey
- Marsh Bird Survey
- Forest Bird Survey
- Kingfisher Nest Location

EPA - Tree Swallow Report - Custer, 2002
- Swallow Box Location
- Bird Tissue Sample Location

GE - Robin Report - ARCADIS G&M, 2002
- Robin Tissue Sample Location

GE - Kingfisher Report - ARCADIS G&M, 2002
- Kingfisher Nest Location

Mammals

EPA - Ecological Characterization Report - Woodlot, 2003
- Bat Transect
- Short-Tailed Shrew Tissue Sample Location
- White-Footed Mouse Tissue Sample Location
- Mink/Otter Scent Post Location
- Snow Track Transects

GE Small Mammal Study - Boonstra, 2002
- Small Mammal Grid Area

GE Otter/Mink Report - ARCADIS G&M, 2003
- Otter Spraining Station Location
- Otter Camera Location
- Otter Track Location
- Mink Track Location
- Mink Searching Area

Notes:

- Line features representing fish studies are for graphic presentation of the length of reach or area sampled and are not meant to represent actual sample locations.

- Certain sample types such as macroinvertebrates, crayfish, predators, and shredders are located too close to one another to be distinguishable at map scale and thus for graphic presentation, have symbols provided with separation distance .

- In most cases, sample locations were derived using GPS and thus have varying levels of inherent horizontal accuracy, depending on equipment employed at the time of the particular effort. These points are presented herein over base features and hydrography derived from orthophotography, which can have a higher level of geographic accuracy and may not represent specific hydrologic conditions at the time the samples were collected.

- Except where horizontal coordinates are provided in the GE database, the locations of GE samples have been approximated from cited references.

- All tissue concentrations provided are tPCB mg/kg wet weight.

Ecological Risk Assessment
GE/Housatonic River Project Site
Rest of River

FIGURE 2.8-10
BIOLOGICAL SAMPLE LOCATIONS IN THE PRIMARY STUDY AREA (TILE 10 OF 21)

Ecological Risk Assessment
GE/Housatonic River Project Site
Rest of River

FIGURE 2.8-11
BIOLOGICAL SAMPLE LOCATIONS IN THE PRIMARY STUDY AREA (TILE 11 OF 21)

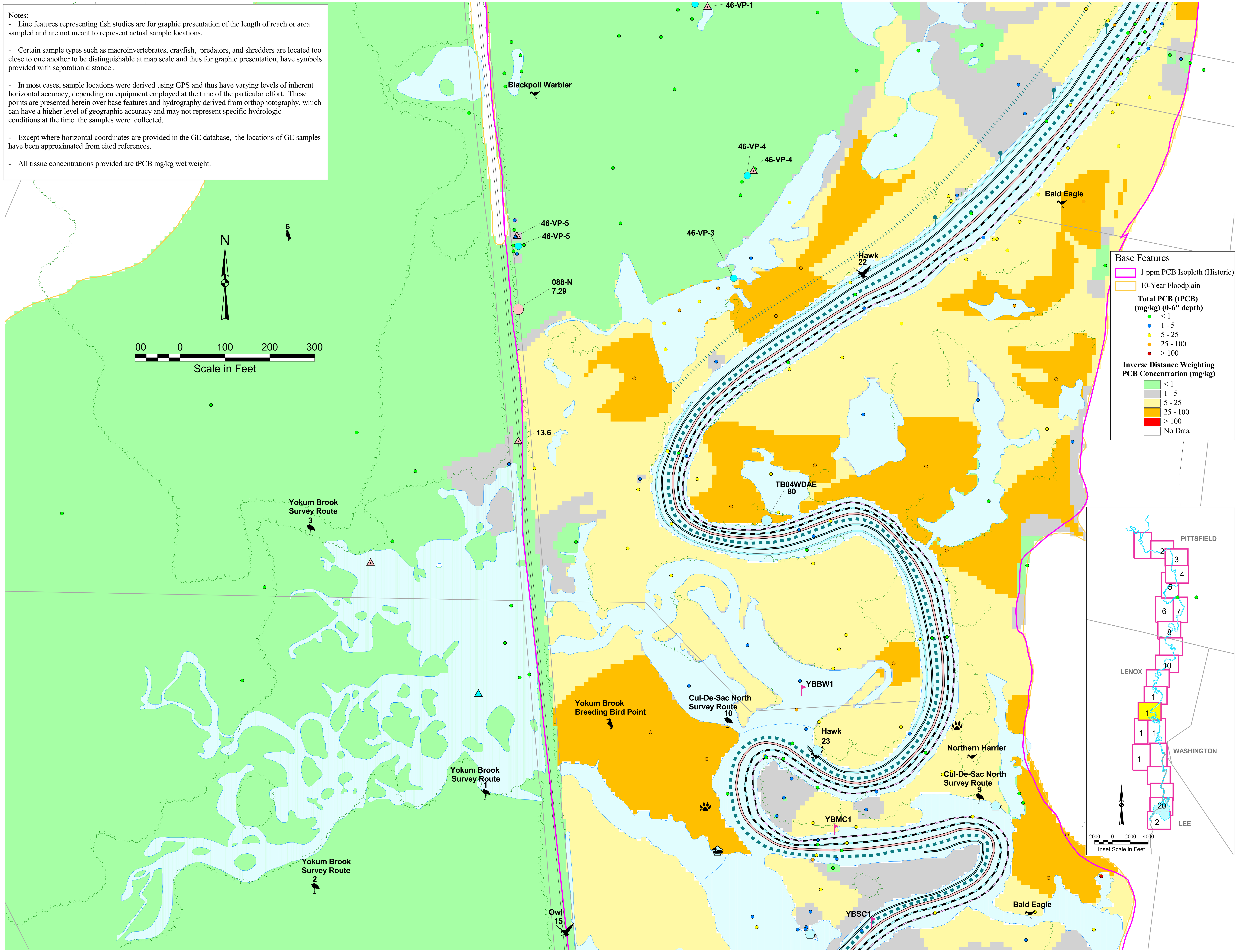
Notes:

- Line features representing fish studies are for graphic presentation of the length of reach or area sampled and are not meant to represent actual sample locations.
- Certain sample types such as macroinvertebrates, crayfish, predators, and shredders are located too close to one another to be distinguishable at map scale and thus for graphic presentation, have symbols provided with separation distance .
- In most cases, sample locations were derived using GPS and thus have varying levels of inherent horizontal accuracy, depending on equipment employed at the time of the particular effort. These points are presented herein over base features and hydrography derived from orthophotography, which can have a high level of geographic accuracy and may not represent specific hydrologic conditions at the time the samples were collected.
- Except where horizontal coordinates are provided in the GE database, the locations of GE samples have been approximated from cited references.
- All tissue concentrations provided are tPCB mg/kg wet weight.

Ecological Risk Assessment
GE/Housatonic River Project Site
Rest of River

FIGURE 2.8-12
BIOLOGICAL SAMPLE LOCATIONS IN THE PRIMARY STUDY AREA (TILE 12 OF 21)

c:\gepitt\era_060204\era_biotia__21tiles_bigfonts.apr | T12 | c:\gepitt\era_060204\exports\in\fig2-8-12_101504.eps | 11:18 AM, 10/27/2004 |



LEGEND

Invertebrates

EPA Ecological Risk Assessment - Weston, 2003
- Macroinvertebrate Sample Location
- Crayfish Sample Location
- Sediment Quality Station
- Benthic Predator Sample Location
- Benthic Shredder Sample Location

EPA Sediment Toxicity Study - Burton/WSU, 2003
- Toxicity Test Location

EPA Ecological Characterization - Woodlot, 2002
- Earthworm/Leaf Litter Tissue Sample Location
- Dragonfly Survey Transect Location

Reptiles/Amphibians

EPA Bullfrog Tissue Study - Weston, 2003
- Bullfrog Tissue Sample Location

EPA Wood Frog Study - Fort, 2000
- Wood Frog Target Site (Vernal Pools)

EPA Leopard Frog Study - Fort, 2000
- Leopard Frog Target Site

GE Housatonic River Study - Stewart, 1980, 1982
- Snapping Turtle Sample Location
- Bullfrog Sample Location

GE Wood Frog Study - Old Dominion Univ., 2002
- Wood Frog Enclosure
- Wood Frog Egg Mass Location

GE Leopard Frog Study - ARCADIS, 2003
- Leopard Frog Egg Mass Location

Fish

EPA Ecological Characterization Report - Woodlot, 1998
- Fish Characterization and Tissue Sample Reach
- Fish Sample Area

EPA Fish Biomass Study - Woodlot, 2000
- Fish Sample Reach
- Fish Sample Area

EPA Koi/Goldfish Study, USFWS, 2003
- Fish Sample Area

EPA Mink Feeding Study, Bursian/Tillitt, 2003
- Fish Sample Area

EPA Bass/Bluegill Study, Tillitt, 2003
- Fish Sample Area

GE Housatonic River Study - Stewart, 1980, 1982,
- Fish Sample Reach

GE Adult Fish Study - BBL, 1990, 1998, & 2002
- Fish Sample Reach

GE Biennial YOY Fish Study - BBL, 1994-2002
- Fish Sample Reach

GE Bass Study - R2, 2002
- Bass Index Sites
- Bass Study Sites
- Bass Transect Locations

Birds

EPA - Ecological Characterization Report - Woodlot, 2003
- Wood Duck Embryo Sample Location
- Waterfowl Trap Location
- Wood Duck Nest Box Location
- Rare Bird Survey
- Owl/Hawk Survey
- Marsh Bird Survey
- Forest Bird Survey
- Kingfisher Nest Location

EPA - Tree Swallow Report - Custer, 2002
- Swallow Box Location
- Bird Tissue Sample Location

GE - Robin Report - ARCADIS G&M, 2002
- Robin Tissue Sample Location

GE - Kingfisher Report - ARCADIS G&M, 2002
- Kingfisher Nest Location

Mammals

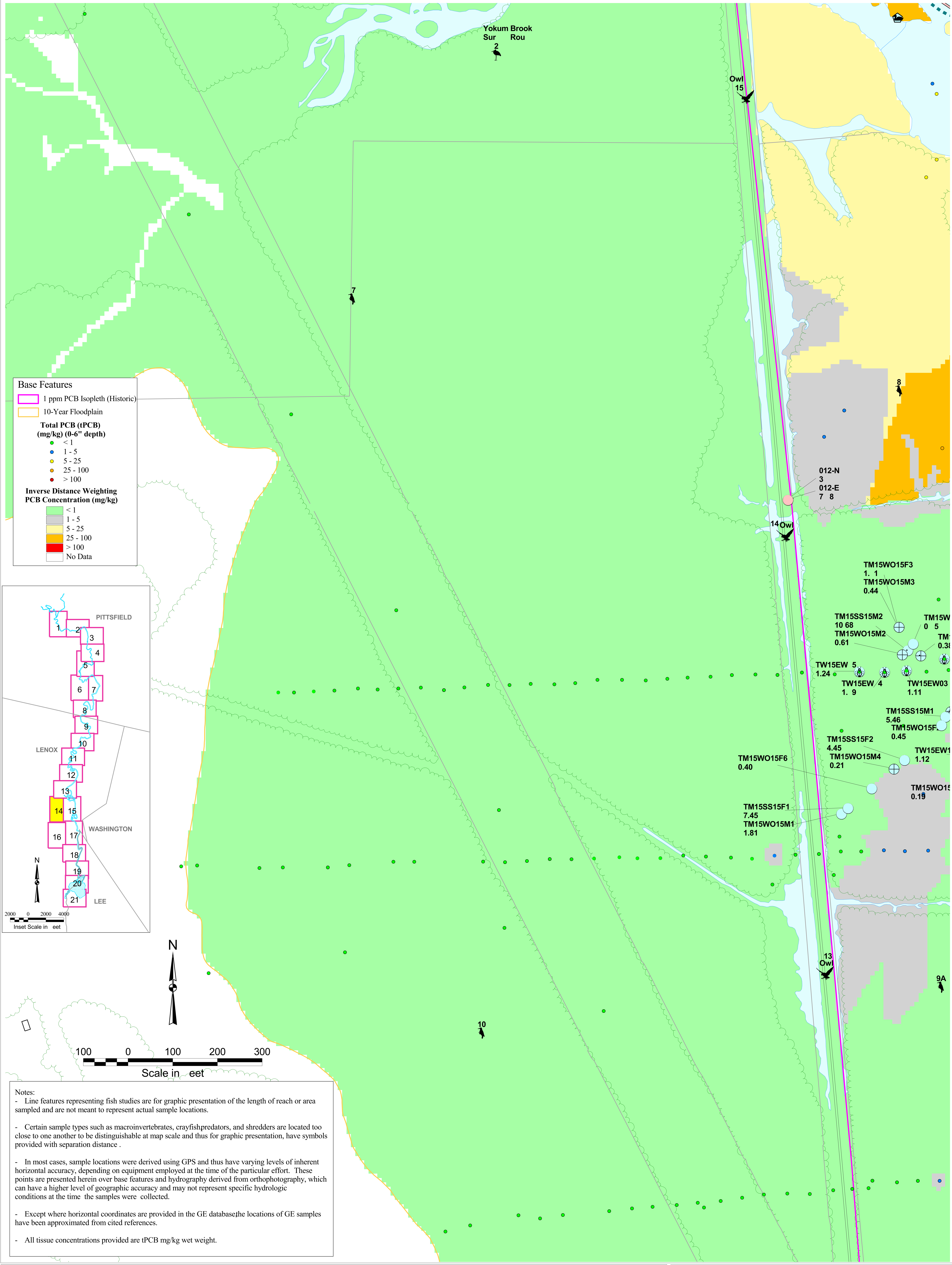
EPA - Ecological Characterization Report - Woodlot, 2003
- Bat Transect
- Short-Tailed Shrew Tissue Sample Location
- White-Footed Mouse Tissue Sample Location
- Mink/Otter Scent Post Location
- Snow Track Transects

GE Small Mammal Study - Boonstra, 2002
- Small Mammal Grid Area

GE Otter/Mink Report - ARCADIS G&M, 2003
- Otter Spraining Station Location
- Otter Camera Location
- Otter Track Location
- Mink Track Location
- Mink Searching Area

Ecological Risk Assessment
GE/Housatonic River Project Site
Rest of River

FIGURE 2.8-13
BIOLOGICAL SAMPLE LOCATIONS IN THE PRIMARY STUDY AREA
(TILE 13 OF 21)



Notes:

- Line features representing fish studies are for graphic presentation of the length of reach or area sampled and are not meant to represent actual sample locations.
- Certain sample types such as macroinvertebrates, crayfishpredators, and shredders are located too close to one another to be distinguishable at map scale and thus for graphic presentation, have symbols provided with separation distance .
- In most cases, sample locations were derived using GPS and thus have varying levels of inherent horizontal accuracy, depending on equipment employed at the time of the particular effort. These points are presented herein over base features and hydrography derived from orthophotography, which can have a higher level of geographic accuracy and may not represent specific hydrologic conditions at the time the samples were collected.
- Except where horizontal coordinates are provided in the GE database, the locations of GE samples have been approximated from cited references.
- All tissue concentrations provided are tPCB mg/kg wet weight.

LEGEND

Invertebrates
EPA Ecological Risk Assessment - Weston, 2003
Macroinvertebrate Sample Location
Crayfish Sample Location
Sediment Quality Station
Benthic Predator Sample Location
Benthic Shredder Sample Location
EPA Sediment Toxicity Study - Burton/WSU, 2003
Toxicity Test Location
EPA Ecological Characterization - Woodlot, 2002
Earthworm/Leaf Litter Tissue Sample Location
Dragonfly Survey Transect Location
Reptiles/Amphibians
EPA Bullfrog Tissue Study - Weston, 2003
Bullfrog Tissue Sample Location
EPA Wood Frog Study - Fort, 2000
Wood Frog Target Site (Vernal Pools)
EPA Leopard Frog Study - Fort, 2000
Leopard Frog Target Site
GE Housatonic River Study - Stewart , 1980,1982
Snapping Turtle Sample Location
Bullfrog Sample Location
GE Wood Frog Study - Old Dominion Univ., 2002
Wood Frog Enclosure
Wood Frog Egg Mass Location
GE Leopard Frog Study - ARCADIS, 2003
Leopard Frog Egg Mass Location

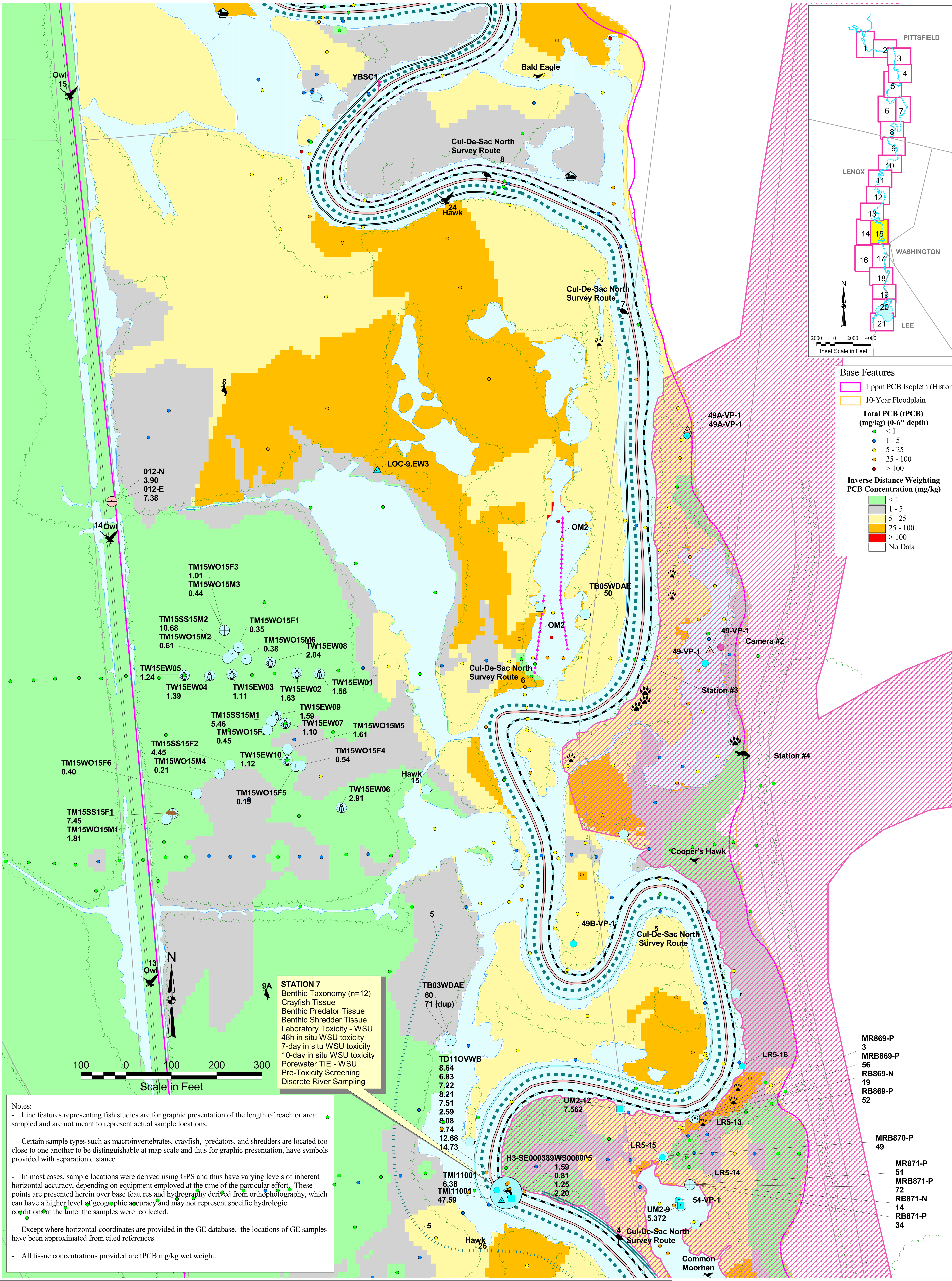
Fish
EPA Ecological Characterization Report - Woodlot, 1998
Fish Characterization and Tissue Sample Reach
Fish Sample Area
EPA Fish Biomass Study - Woodlot, 2000
Fish Sample Reach
Fish Sample Area
EPA Koi/Goldfish Study, USFWS, 2003
Fish Sample Area
EPA Mink Feeding Study, Bursian/Tillitt, 2003
Fish Sample Area
EPA Bass/Bluegill Study, Tillitt, 2003
Fish Sample Area
GE Housatonic River Study - Stewart, 1980, 1982
Fish Sample Reach
GE Adult Fish Study - BBL, 1990, 1998, & 2002
Fish Sample Reach
GE Biennial YOY Fish Study - BBL, 1994-2002
Fish Sample Reach
GE Bass Study - R2, 2002
Bass Index Sites
Bass Study Sites
Bass Transect Locations

Birds
EPA - Ecological Characterization Report - Woodlot, 2003
Wood Duck Embryo Sample Location
Waterfowl Trap Location
Wood Duck Nest Box Location
Rare Bird Survey
Owl/Hawk Survey
Marsh Bird Survey
Forest Bird Survey
Kingfisher Nest Location
EPA - Tree Swallow Report - Custer, 2002
Swallow Box Location
Bird Tissue Sample Location
GE - Robin Report - ARCADIS G&M, 2002
Robin Tissue Sample Location
GE - Kingfisher Report - ARCADIS G&M, 2002
Kingfisher Nest Location

Mammals
EPA - Ecological Characterization Report - Woodlot, 2003
Bat Transect
Short-Tailed Shrew Tissue Sample Location
White-Footed Mouse Tissue Sample Location
Mink/Otter Scent Post Location
Snow Track Transects
GE Small Mammal Study - Boonstra, 2002
Small Mammal Grid Area
GE Otter/Mink Report - ARCADIS G&M, 2003
Otter Spraining Station Location
Otter Camera Location
Otter Track Location
Mink Track Location
Mink Searching Area

Ecological Risk Assessment
GE/Housatonic River Project Site
Rest of River

FIGURE 2.8-14
BIOLOGICAL SAMPLE LOCATIONS IN THE PRIMARY STUDY AREA
(TILE 14 OF 21)



LEGEND

Invertebrates
EPA Ecological Risk Assessment - Weston, 2003
Macroinvertebrate Sample Location
Crayfish Sample Location
Sediment Quality Station
Benthic Predator Sample Location
Benthic Shredder Sample Location
EPA Sediment Toxicity Study - Burton/WSU, 2003
Toxicity Test Location
EPA Ecological Characterization - Woodlot, 2002
Earthworm/Leaf Litter Tissue Sample Location
Dragonfly Survey Transect Location

Reptiles/Amphibians
EPA Bullfrog Tissue Study - Weston, 2003
Bullfrog Tissue Sample Location
EPA Wood Frog Study - Fort, 2000
Wood Frog Target Site (Vernal Pools)
EPA Leopard Frog Study - Fort, 2000
Leopard Frog Target Site
GE Housatonic River Study - Stewart, 1980,1982
Snapping Turtle Sample Location
Bullfrog Sample Location
GE Wood Frog Study - Old Dominion Univ., 2002
Wood Frog Enclosure
Wood Frog Egg Mass Location
GE Leopard Frog Study - ARCADIS, 2003
Leopard Frog Egg Mass Location

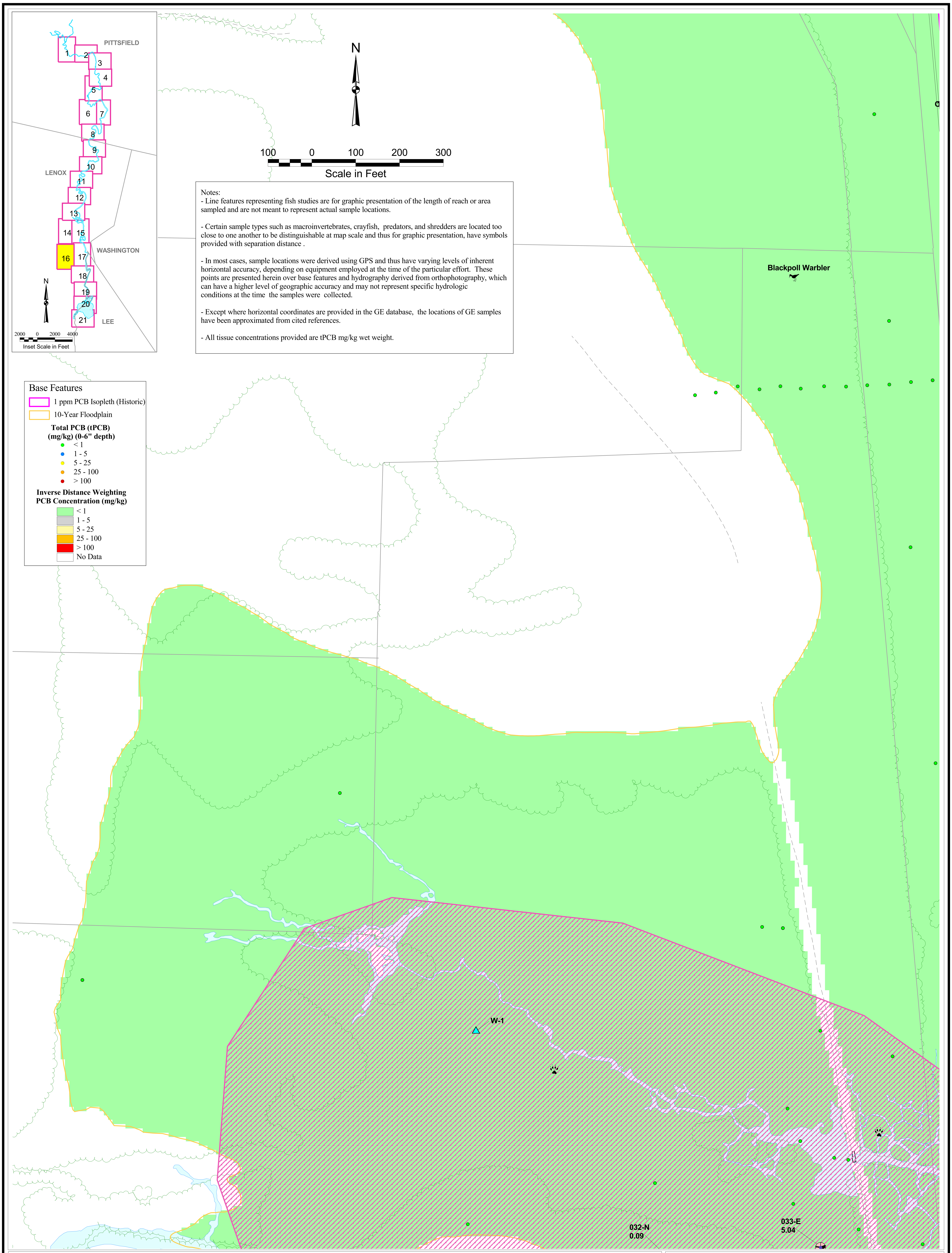
Fish
EPA Ecological Characterization Report - Woodlot, 1998
Fish Characterization and Tissue Sample Reach
Fish Sample Area
EPA Fish Biomass Study - Woodlot, 2000
Fish Sample Reach
Fish Sample Area
EPA Koi/Goldfish Study, USFWS, 2003
Fish Sample Area
EPA Mink Feeding Study, Bursian/Tillitt, 2003
Fish Sample Area
EPA Bass/Bluegill Study, Tillitt, 2003
Fish Sample Area
GE Housatonic River Study - Stewart, 1980, 1982
Fish Sample Reach
GE Adult Fish Study - BBL, 1990, 1998, & 2002
Fish Sample Reach
GE Biennial YOY Fish Study - BBL, 1994-2002
Fish Sample Reach
GE Bass Study - R2, 2002
Bass Index Sites
Bass Study Sites
Bass Transect Locations

Birds
EPA - Ecological Characterization Report - Woodlot, 2003
Wood Duck Embryo Sample Location
Waterfowl Trap Location
Wood Duck Nest Box Location
Rare Bird Survey
Owl/Hawk Survey
Marsh Bird Survey
Forest Bird Survey
Kingfisher Nest Location
EPA - Tree Swallow Report - Custer, 2002
Swallow Box Location
Bird Tissue Sample Location
GE - Robin Report - ARCADIS G&M, 2002
Robin Tissue Sample Location
GE - Kingfisher Report - ARCADIS G&M, 2002
Kingfisher Nest Location

Mammals
EPA - Ecological Characterization Report - Woodlot, 2003
Bat Transect
Short-Tailed Shrew Tissue Sample Location
White-Footed Mouse Tissue Sample Location
Mink/Otter Scent Post Location
Snow Track Transects
GE Small Mammal Study - Boonstra, 2002
Small Mammal Grid Area
GE Otter/Mink Report - ARCADIS G&M, 2003
Otter Spprainting Station Location
Otter Camera Location
Otter Track Location
Mink Track Location
Mink Searching Area

Ecological Risk Assessment
GE/Housatonic River Project Site
Rest of River

FIGURE 2.8-15
BIOLOGICAL SAMPLE LOCATIONS IN THE PRIMARY STUDY AREA (TILE 15 OF 21)



LEGEND

Invertebrates
EPA Ecological Risk Assessment - Weston, 2003
Macroinvertebrate Sample Location
Crayfish Sample Location
Sediment Quality Station
Benthic Predator Sample Location
Benthic Shredder Sample Location
EPA Sediment Toxicity Study - Burton/WSU, 2003
Toxicity Test Location
EPA Ecological Characterization - Woodlot, 2002
Earthworm/Leaf Litter Tissue Sample Location
Dragonfly Survey Transect Location

Reptiles/Amphibians
EPA Bullfrog Tissue Study - Weston, 2003
Bullfrog Tissue Sample Location
EPA Wood Frog Study - Fort, 2000
Wood Frog Target Site (Vernal Pools)
EPA Leopard Frog Study - Fort, 2000
Leopard Frog Target Site
GE Housatonic River Study - Stewart , 1980,1982
Snapping Turtle Sample Location
Bullfrog Sample Location
GE Wood Frog Study - Old Dominion Univ., 2002
Wood Frog Enclosure
Wood Frog Egg Mass Location
GE Leopard Frog Study - ARCADIS, 2003
Leopard Frog Egg Mass Location

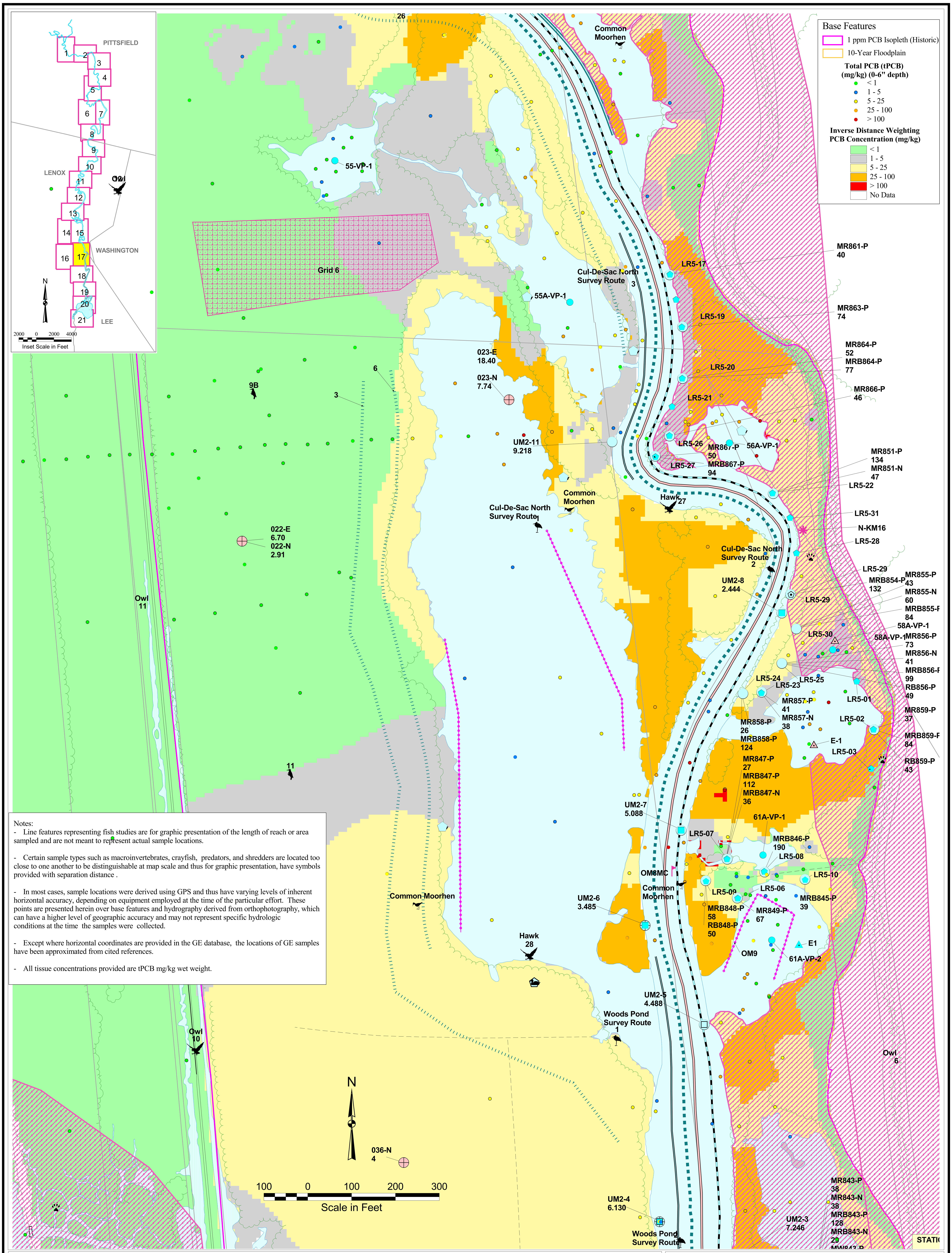
Fish
EPA Ecological Characterization Report - Woodlot, 1998
Fish Characterization and Tissue Sample Reach
Fish Sample Area
EPA Fish Biomass Study - Woodlot, 2000
Fish Sample Reach
Fish Sample Area
EPA Koi/Goldfish Study, USFWS, 2003
Fish Sample Area
EPA Mink Feeding Study, Bursian/Tillitt, 2003
Fish Sample Area
EPA Bass/Bluegill Study, Tillitt, 2003
Fish Sample Area
GE Housatonic River Study - Stewart, 1980, 1982
Fish Sample Reach
GE Adult Fish Study - BBL, 1990, 1998, & 2002
Fish Sample Reach
GE Biennial YOY Fish Study - BBL, 1994-2002
Fish Sample Reach
GE Bass Study - R2, 2002
Bass Index Sites
Bass Study Sites
Bass Transect Locations

Birds
EPA - Ecological Characterization Report - Woodlot, 2003
Wood Duck Embryo Sample Location
Waterfowl Trap Location
Wood Duck Nest Box Location
Rare Bird Survey
Owl/Hawk Survey
Marsh Bird Survey
Forest Bird Survey
Kingfisher Nest Location
EPA - Tree Swallow Report - Custer, 2002
Swallow Box Location
Bird Tissue Sample Location
GE - Robin Report - ARCADIS G&M, 2002
Robin Tissue Sample Location
GE - Kingfisher Report - ARCADIS G&M, 2002
Kingfisher Nest Location

Mammals
EPA - Ecological Characterization Report - Woodlot, 2003
Bat Transect
Short-Tailed Shrew Tissue Sample Location
White-Footed Mouse Tissue Sample Location
Mink/Otter Scent Post Location
Snow Track Transects
GE Small Mammal Study - Boonstra, 2002
Small Mammal Grid Area
GE Otter/Mink Report - ARCADIS G&M, 2003
Otter Sprainting Station Location
Otter Camera Location
Otter Track Location
Mink Track Location
Mink Searching Area

Ecological Risk Assessment
GE/Housatonic River Project Site
Rest of River

FIGURE 2.8-16
BIOLOGICAL SAMPLE LOCATIONS IN THE PRIMARY STUDY AREA (TILE 16 OF 21)



Notes:

- Line features representing fish studies are for graphic presentation of the length of reach or area sampled and are not meant to represent actual sample locations.
- Certain sample types such as macroinvertebrates, crayfish, predators, and shredders are located too close to one another to be distinguishable at map scale and thus for graphic presentation, have symbols provided with separation distance .
- In most cases, sample locations were derived using GPS and thus have varying levels of inherent horizontal accuracy, depending on equipment employed at the time of the particular effort. These points are presented herein over base features and hydrography derived from orthophotography, which can have a higher level of geographic accuracy and may not represent specific hydrologic conditions at the time the samples were collected.
- Except where horizontal coordinates are provided in the GE database, the locations of GE samples have been approximated from cited references.
- All tissue concentrations provided are tPCB mg/kg wet weight.

LEGEND

Invertebrates
EPA Ecological Risk Assessment - Weston, 2003
Macroinvertebrate Sample Location
Crayfish Sample Location
Sediment Quality Station
Benthic Predator Sample Location
Benthic Shredder Sample Location
EPA Sediment Toxicity Study - Burton/WSU, 2003
Toxicity Test Location
EPA Ecological Characterization - Woodlot, 2002
Earthworm/Leaf Litter Tissue Sample Location
Dragonfly Survey Transect Location

Reptiles/Amphibians
EPA Bullfrog Tissue Study - Weston, 2003
Bullfrog Tissue Sample Location
EPA Wood Frog Study - Fort, 2000
Wood Frog Target Site (Vernal Pools)
EPA Leopard Frog Study - Fort, 2000
Leopard Frog Target Site
GE Housatonic River Study - Stewart , 1980,1982
Snapping Turtle Sample Location
Bullfrog Sample Location
GE Wood Frog Study - Old Dominion Univ., 2002
Wood Frog Enclosure
Wood Frog Egg Mass Location
GE Leopard Frog Study - ARCADIS, 2003
Leopard Frog Egg Mass Location

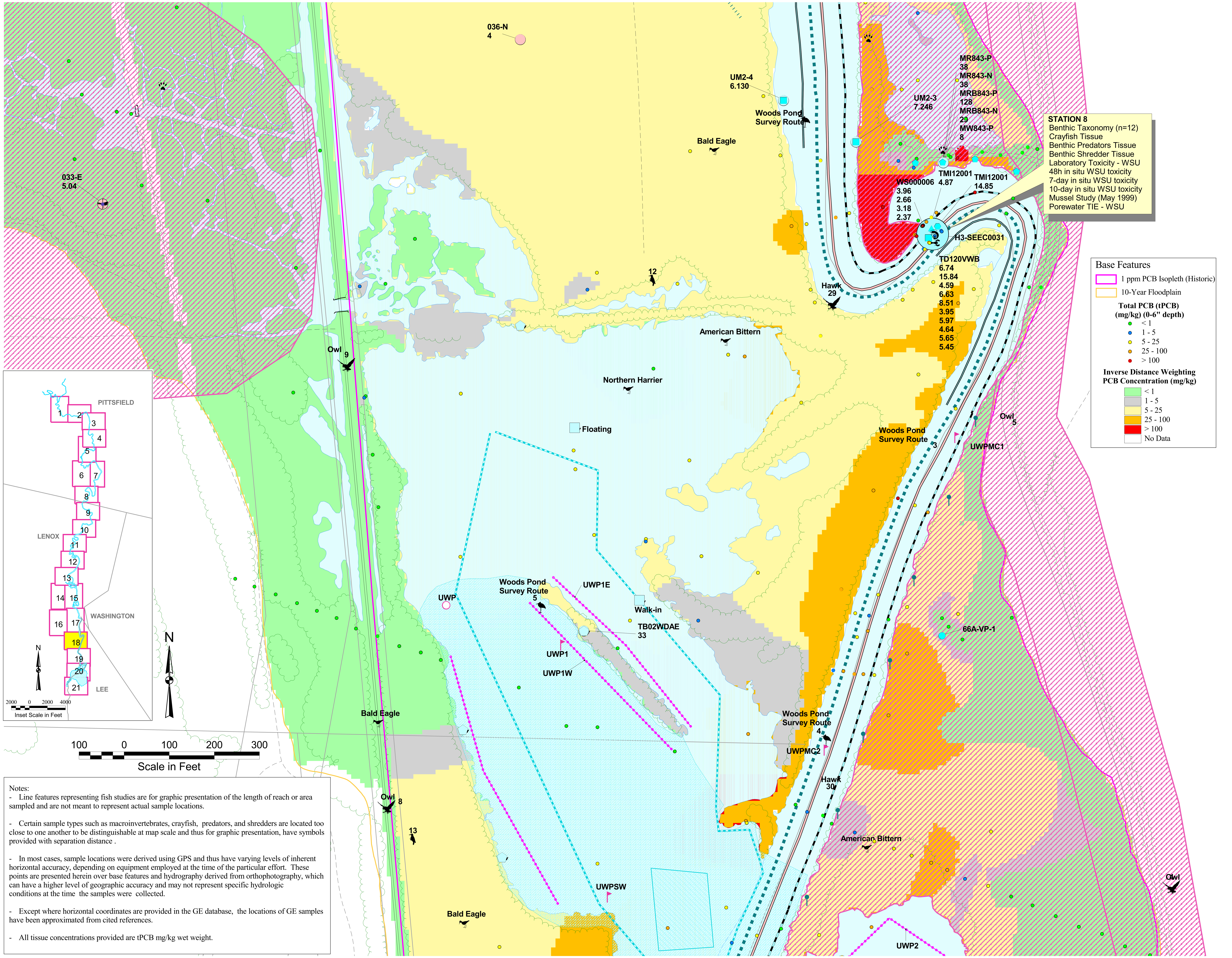
Fish
EPA Ecological Characterization Report - Woodlot, 1998
Fish Characterization and Tissue Sample Reach
Fish Sample Area
EPA Fish Biomass Study - Woodlot, 2000
Fish Sample Reach
Fish Sample Area
EPA Koi/Goldfish Study, USFWS, 2003
Fish Sample Area
EPA Mink Feeding Study, Bursian/Tillitt, 2003
Fish Sample Area
EPA Bass/Bluegill Study, Tillitt, 2003
Fish Sample Area
GE Housatonic River Study - Stewart, 1980, 1982
Fish Sample Reach
GE Adult Fish Study - BBL, 1990, 1998, & 2002
Fish Sample Reach
GE Biennial YOY Fish Study - BBL, 1994-2002
Fish Sample Reach
GE Bass Study - R2, 2002
Bass Index Sites
Bass Study Sites
Bass Transect Locations

Birds
EPA - Ecological Characterization Report - Woodlot, 2003
Wood Duck Embryo Sample Location
Waterfowl Trap Location
Wood Duck Nest Box Location
Rare Bird Survey
Owl/Hawk Survey
Marsh Bird Survey
Forest Bird Survey
Kingfisher Nest Location
EPA - Tree Swallow Report - Custer, 2002
Swallow Box Location
Bird Tissue Sample Location
GE - Robin Report - ARCADIS G&M, 2002
Robin Tissue Sample Location
GE - Kingfisher Report - ARCADIS G&M, 2002
Kingfisher Nest Location

Mammals
EPA - Ecological Characterization Report - Woodlot, 2003
Bat Transect
Short-Tailed Shrew Tissue Sample Location
White-Footed Mouse Tissue Sample Location
Mink/Otter Scent Post Location
Snow Track Transects
GE Small Mammal Study - Boonstra, 2002
Small Mammal Grid Area
GE Otter/Mink Report - ARCADIS G&M, 2003
Otter Spraint ing Station Location
Otter Camera Location
Otter Track Location
Mink Track Location
Mink Searching Area

Ecological Risk Assessment
GE/Housatonic River Project Site
Rest of River

FIGURE 2.8-17 BIOLOGICAL SAMPLE LOCATIONS IN THE PRIMARY STUDY AREA (TILE 17 OF 21)



Notes:
- Line features representing fish studies are for graphic presentation of the length of reach or area sampled and are not meant to represent actual sample locations.

- Certain sample types such as macroinvertebrates, crayfish, predators, and shredders are located too close to one another to be distinguishable at map scale and thus for graphic presentation, have symbols provided with separation distance .

- In most cases, sample locations were derived using GPS and thus have varying levels of inherent horizontal accuracy, depending on equipment employed at the time of the particular effort. These points are presented herein over base features and hydrography derived from orthophotography, which can have a higher level of geographic accuracy and may not represent specific hydrologic conditions at the time the samples were collected.

- Except where horizontal coordinates are provided in the GE database, the locations of GE samples have been approximated from cited references.

- All tissue concentrations provided are tPCB mg/kg wet weight.

LEGEND

Invertebrates
EPA Ecological Risk Assessment - Weston, 2003
Macroinvertebrate Sample Location
Crayfish Sample Location
Sediment Quality Station
Benthic Predator Sample Location
Benthic Shredder Sample Location
EPA Sediment Toxicity Study - Burton/WSU, 2003
Toxicity Test Location
EPA Ecological Characterization - Woodlot, 2002
Earthworm/Leaf Litter Tissue Sample Location
Dragonfly Survey Transect Location

Reptiles/Amphibians
EPA Bullfrog Tissue Study - Weston, 2003
Bullfrog Tissue Sample Location
EPA Wood Frog Study - Fort, 2000
Wood Frog Target Site (Vernal Pools)
EPA Leopard Frog Study - Fort, 2000
Leopard Frog Target Site
GE Housatonic River Study - Stewart, 1980, 1982
Snapping Turtle Sample Location
Bullfrog Sample Location
GE Wood Frog Study - Old Dominion Univ., 2002
Wood Frog Enclosure
Wood Frog Egg Mass Location
GE Leopard Frog Study - ARCADIS, 2003
Leopard Frog Egg Mass Location

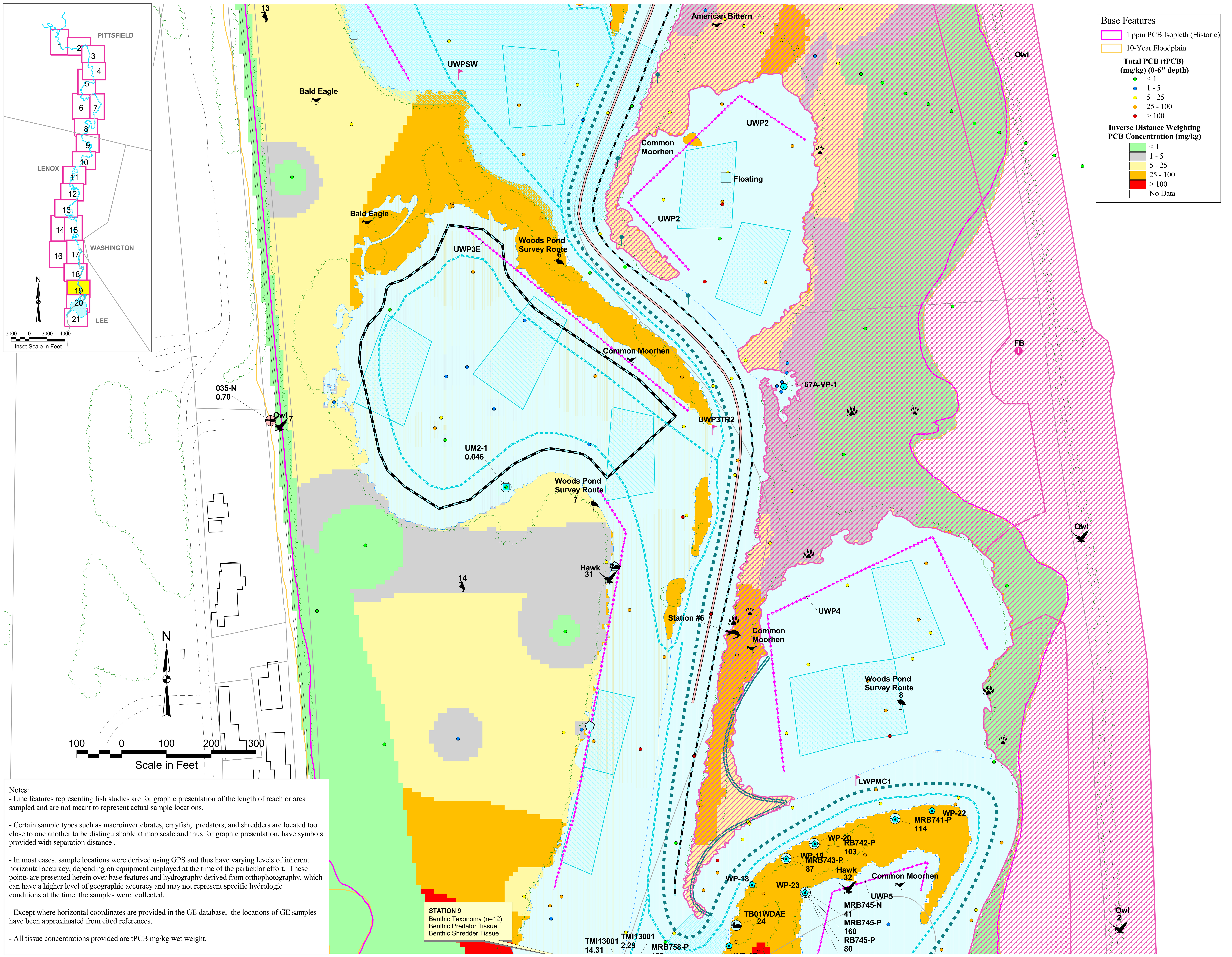
Fish
EPA Ecological Characterization Report - Woodlot, 1998
Fish Characterization and Tissue Sample Reach
Fish Sample Area
EPA Fish Biomass Study - Woodlot, 2000
Fish Sample Reach
Fish Sample Area
EPA Koi/Goldfish Study, USFWS, 2003
Fish Sample Area
EPA Mink Feeding Study, Bursian/Tillitt, 2003
Fish Sample Area
EPA Bass/Bluegill Study, Tillitt, 2003
Fish Sample Area
GE Housatonic River Study - Stewart, 1980, 1982,
Fish Sample Reach
GE Adult Fish Study - BBL, 1990, 1998, & 2002
Fish Sample Reach
GE Biennial YOY Fish Study - BBL, 1994-2002
Fish Sample Reach
GE Bass Study - R2, 2002
Bass Index Sites
Bass Study Sites
Bass Transect Locations

Birds
EPA - Ecological Characterization Report - Woodlot, 2003
Wood Duck Embryo Sample Location
Waterfowl Trap Location
Wood Duck Nest Box Location
Rare Bird Survey
Owl/Hawk Survey
Marsh Bird Survey
Forest Bird Survey
Kingfisher Nest Location
EPA - Tree Swallow Report - Custer, 2002
Swallow Box Location
Bird Tissue Sample Location
GE - Robin Report - ARCADIS G&M, 2002
Robin Tissue Sample Location
GE - Kingfisher Report - ARCADIS G&M, 2002
Kingfisher Nest Location

Mammals
EPA - Ecological Characterization Report - Woodlot, 2003
Bat Transect
Short-Tailed Shrew Tissue Sample Location
White-Footed Mouse Tissue Sample Location
Mink/Otter Scent Post Location
Snow Track Transects
GE Small Mammal Study - Boonstra, 2002
Small Mammal Grid Area
GE Otter/Mink Report - ARCADIS G&M, 2003
Otter Spraining Station Location
Otter Camera Location
Otter Track Location
Mink Track Location
Mink Searching Area

Ecological Risk Assessment
GE/Housatonic River Project Site
Rest of River

FIGURE 2.8-18
BIOLOGICAL SAMPLE LOCATIONS IN THE PRIMARY STUDY AREA (TILE 18 OF 21)



Notes:

- Line features representing fish studies are for graphic presentation of the length of reach or area sampled and are not meant to represent actual sample locations.

- Certain sample types such as macroinvertebrates, crayfish, predators, and shredders are located too close to one another to be distinguishable at map scale and thus for graphic presentation, have symbols provided with separation distance .

- In most cases, sample locations were derived using GPS and thus have varying levels of inherent horizontal accuracy, depending on equipment employed at the time of the particular effort. These points are presented herein over base features and hydrography derived from orthophotography, which can have a higher level of geographic accuracy and may not represent specific hydrologic conditions at the time the samples were collected.

- Except where horizontal coordinates are provided in the GE database, the locations of GE samples have been approximated from cited references.

- All tissue concentrations provided are tPCB mg/kg wet weight.

LEGEND

Invertebrates

EPA Ecological Risk Assessment - Weston, 2003
- Macroinvertebrate Sample Location
- Crayfish Sample Location
- Sediment Quality Station
- Benthic Predator Sample Location
- Benthic Shredder Sample Location

EPA Sediment Toxicity Study - Burton/WSU, 2003
- Toxicity Test Location

EPA Ecological Characterization - Woodlot, 2002
- Earthworm/Leaf Litter Tissue Sample Location
- Dragonfly Survey Transect Location

Reptiles/Amphibians

EPA Bullfrog Tissue Study - Weston, 2003
- Bullfrog Tissue Sample Location

EPA Wood Frog Study - Fort, 2000
- Wood Frog Target Site (Vernal Pools)

EPA Leopard Frog Study - Fort, 2000
- Leopard Frog Target Site

GE Housatonic River Study - Stewart, 1980, 1982
- Snapping Turtle Sample Location
- Bullfrog Sample Location

GE Wood Frog Study - Old Dominion Univ., 2002
- Wood Frog Enclosure
- Wood Frog Egg Mass Location

GE Leopard Frog Study - ARCADIS, 2003
- Leopard Frog Egg Mass Location

Fish

EPA Ecological Characterization Report - Woodlot, 1998
- Fish Characterization and Tissue Sample Reach
- Fish Sample Area

EPA Fish Biomass Study - Woodlot, 2000
- Fish Sample Reach
- Fish Sample Area

EPA Koi/Goldfish Study, USFWS, 2003
- Fish Sample Area

EPA Mink Feeding Study, Bursian/Tillitt, 2003
- Fish Sample Area

EPA Bass/Bluegill Study, Tillitt, 2003
- Fish Sample Area

GE Housatonic River Study - Stewart, 1980, 1982,
- Fish Sample Reach

GE Adult Fish Study - BBL, 1990, 1998, & 2002
- Fish Sample Reach

GE Biennial YOY Fish Study - BBL, 1994-2002
- Fish Sample Reach

GE Bass Study - R2, 2002
- Bass Index Sites
- Bass Study Sites
- Bass Transect Locations

Birds

EPA - Ecological Characterization Report - Woodlot, 2003
- Wood Duck Embryo Sample Location
- Waterfowl Trap Location
- Wood Duck Nest Box Location
- Rare Bird Survey
- Owl/Hawk Survey
- Marsh Bird Survey
- Forest Bird Survey
- Kingfisher Nest Location

EPA - Tree Swallow Report - Custer, 2002
- Swallow Box Location
- Bird Tissue Sample Location

GE - Robin Report - ARCADIS G&M, 2002
- Robin Tissue Sample Location

GE - Kingfisher Report - ARCADIS G&M, 2002
- Kingfisher Nest Location

Mammals

EPA - Ecological Characterization Report - Woodlot, 2003
- Bat Transect
- Short-Tailed Shrew Tissue Sample Location
- White-Footed Mouse Tissue Sample Location
- Mink/Otter Scent Post Location
- Snow Track Transects

GE Small Mammal Study - Boonstra, 2002
- Small Mammal Grid Area

GE Otter/Mink Report - ARCADIS G&M, 2003
- Otter Sprainting Station Location
- Otter Camera Location
- Otter Track Location
- Mink Track Location
- Mink Searching Area

Ecological Risk Assessment
GE/Housatonic River Project Site
Rest of River

FIGURE 2.8-19
BIOLOGICAL SAMPLE LOCATIONS IN THE PRIMARY STUDY AREA (TILE 19 OF 21)

c:\gepitt\era_060204\era_biotia__21tiles_bigfonts.apr | T19 | c:\gepitt\era_060204\exports\n\fig2-8-19_101504.eps | 11:25 AM, 10/27/2004 |

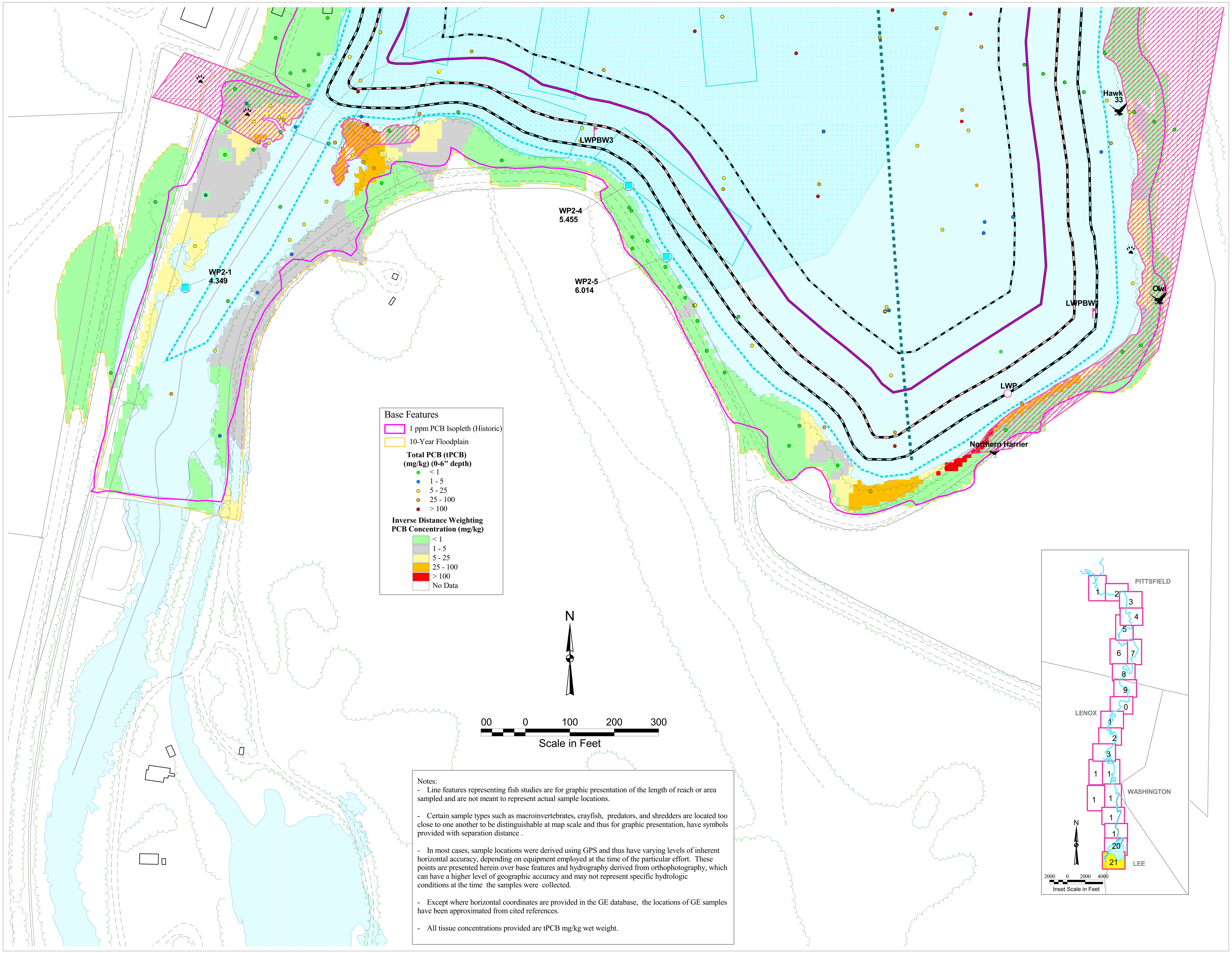
Notes:

- Line features representing fish studies are for graphic presentation of the length of reach or area sampled and are not meant to represent actual sample locations.
- Certain sample types such as macroinvertebrates, crayfish, predators, and shredders are located too close to one another to be distinguishable at map scale and thus for graphic presentation, have symbols provided with separation distance .
- In most cases, sample locations were derived using GPS and thus have varying levels of inherent horizontal accuracy, depending on equipment employed at the time of the particular effort. These points are presented herein over base features and hydrography derived from orthophotography, which can have a higher level of geographic accuracy and may not represent specific hydrologic conditions at the time the samples were collected.
- Except where horizontal coordinates are provided in the GE database, the locations of GE samples have been approximated from cited references.
- All tissue concentrations provided are tPCB mg/kg wet weight.

Ecological Risk Assessment
GE/Housatonic River Project Site
Rest of River

FIGURE 2.8-20
BIOLOGICAL SAMPLE LOCATIONS IN THE PRIMARY STUDY AREA (TILE 20 OF 21)

c:\gepitt\era_060204\era_biotia__21tiles_bigfonts.apr | T20 | c:\gepitt\era_060204\exports\in\fig2-8-20_101504.eps | 11:26 AM, 10/27/2004 |



Notes:
- Line features representing fish studies are for graphic presentation of the length of reach or area sampled and are not meant to represent actual sample locations.
- Certain sample types such as macroinvertebrates, crayfish, predators, and shredders are located too close to one another to be distinguishable at map scale and thus for graphic presentation, have symbols provided with separation distance .
- In most cases, sample locations were derived using GPS and thus have varying levels of inherent horizontal accuracy, depending on equipment employed at the time of the particular effort. These points are presented herein over base features and hydrography derived from orthophotography, which can have a higher level of geographic accuracy and may not represent specific hydrologic conditions at the time the samples were collected.
- Except where horizontal coordinates are provided in the GE database, the locations of GE samples have been approximated from cited references.
- All tissue concentrations provided are tPCB mg/kg wet weight.

LEGEND

Invertebrates
EPA Ecological Risk Assessment - Weston, 2003
Macroinvertebrate Sample Location
Crayfish Sample Location
Sediment Quality Station
Benthic Predator Sample Location
Benthic Shredder Sample Location
EPA Sediment Toxicity Study - Burton/WSU, 2003
Toxicity Test Location
EPA Ecological Characterization - Woodlot, 2002
Earthworm/Leaf Litter Tissue Sample Location
Dragonfly Survey Transect Location

Reptiles/Amphibians
EPA Bullfrog Tissue Study - Weston, 2003
Bullfrog Tissue Sample Location
EPA Wood Frog Study - Fort, 2000
Wood Frog Target Site (Vernal Pools)
EPA Leopard Frog Study - Fort, 2000
Leopard Frog Target Site
GE Housatonic River Study - Stewart, 1980, 1982
Snapping Turtle Sample Location
Bullfrog Sample Location
GE Wood Frog Study - Old Dominion Univ., 2002
Wood Frog Enclosure
Wood Frog Egg Mass Location
GE Leopard Frog Study - ARCADIS, 2003
Leopard Frog Egg Mass Location

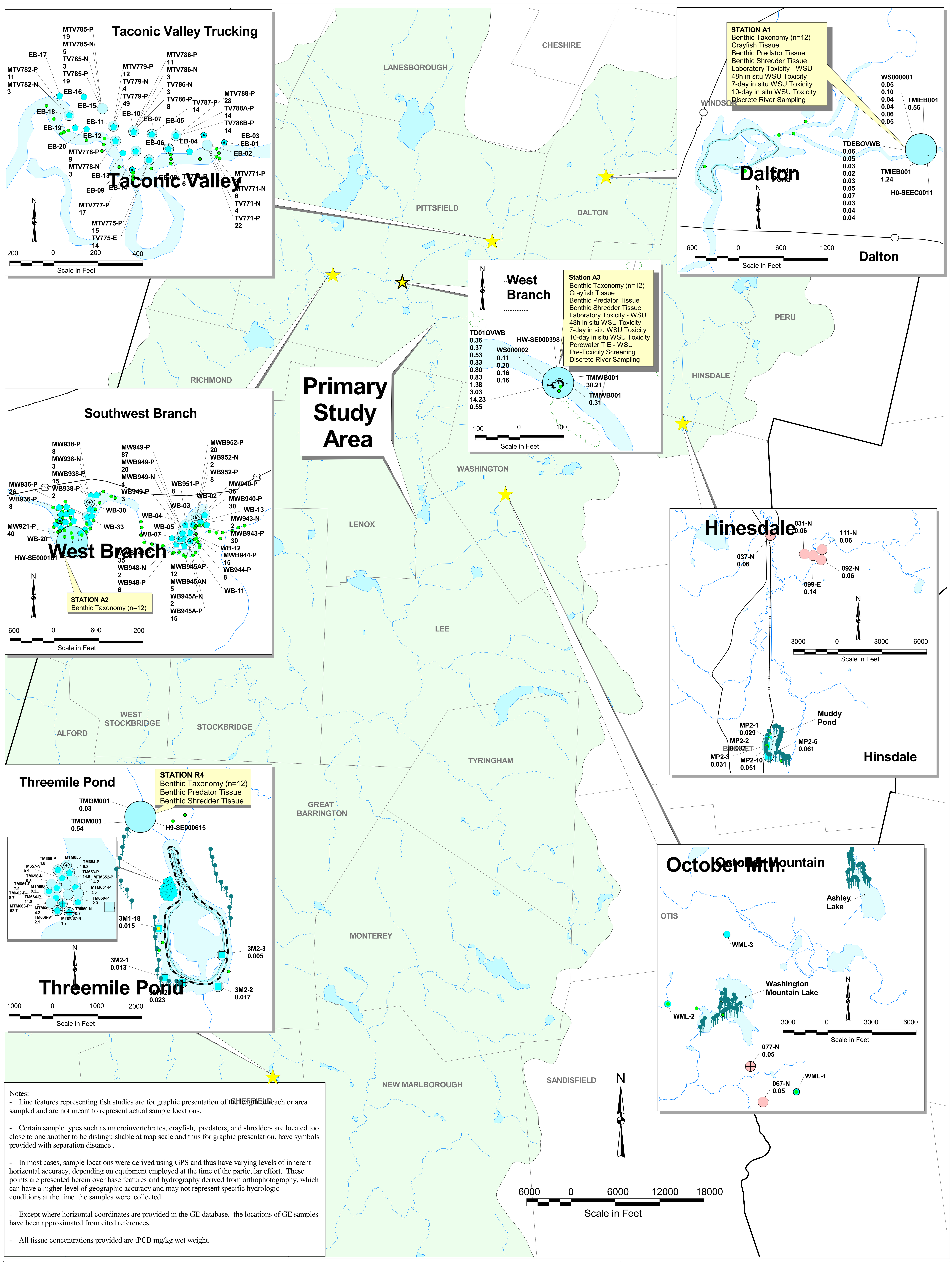
Fish
EPA Ecological Characterization Report - Woodlot, 1998
Fish Characterization and Tissue Sample Reach
Fish Sample Area
EPA Fish Biomass Study - Woodlot, 2000
Fish Sample Reach
Fish Sample Area
EPA Koi/Goldfish Study, USFWS, 2003
Fish Sample Area
EPA Mink Feeding Study, Bursian/Tillitt, 2003
Fish Sample Area
EPA Bass/Bluegill Study, Tillitt, 2003
Fish Sample Area
GE Housatonic River Study - Stewart, 1980, 1982,
Fish Sample Reach
GE Adult Fish Study - BBL, 1990, 1998, & 2002
Fish Sample Reach
GE Biennial YOY Fish Study - BBL, 1994-2002
Fish Sample Reach
GE Bass Study - R2, 2002
Bass Index Sites
Bass Study Sites
Bass Transect Locations

Birds
EPA - Ecological Characterization Report - Woodlot, 2003
Wood Duck Embryo Sample Location
Waterfowl Trap Location
Wood Duck Nest Box Location
Rare Bird Survey
Owl/Hawk Survey
Marsh Bird Survey
Forest Bird Survey
Kingfisher Nest Location
EPA - Tree Swallow Report - Custer, 2002
Swallow Box Location
Bird Tissue Sample Location
GE - Robin Report - ARCADIS G&M, 2002
Robin Tissue Sample Location
GE - Kingfisher Report - ARCADIS G&M, 2002
Kingfisher Nest Location

Mammals
EPA - Ecological Characterization Report - Woodlot, 2003
Bat Transect
Short-Tailed Shrew Tissue Sample Location
White-Footed Mouse Tissue Sample Location
Mink/Otter Scent Post Location
Snow Track Transects
GE Small Mammal Study - Boonstra, 2002
Small Mammal Grid Area
GE Otter/Mink Report - ARCADIS G&M, 2003
Otter Spraining Station Location
Otter Camera Location
Otter Track Location
Mink Track Location
Mink Searching Area

Ecological Risk Assessment
GE/Housatonic River Project Site
Rest of River

FIGURE 2.8-21
BIOLOGICAL SAMPLE LOCATIONS IN THE PRIMARY STUDY AREA (TILE 21 OF 21)



Notes:

- Line features representing fish studies are for graphic presentation of the length of reach or area sampled and are not meant to represent actual sample locations.

- Certain sample types such as macroinvertebrates, crayfish, predators, and shredders are located too close to one another to be distinguishable at map scale and thus for graphic presentation, have symbols provided with separation distance .

- In most cases, sample locations were derived using GPS and thus have varying levels of inherent horizontal accuracy, depending on equipment employed at the time of the particular effort. These points are presented herein over base features and hydrography derived from orthophotography, which can have a higher level of geographic accuracy and may not represent specific hydrologic conditions at the time the samples were collected.

- Except where horizontal coordinates are provided in the GE database, the locations of GE samples have been approximated from cited references.

- All tissue concentrations provided are tPCB mg/kg wet weight.

LEGEND

Invertebrates
EPA Ecological Risk Assessment - Weston, 2003
Macroinvertebrate Sample Location
Crayfish Sample Location
Sediment Quality Station
Benthic Predator Sample Location
Benthic Shredder Sample Location
EPA Sediment Toxicity Study - Burton/WS, 2003
Toxicity Test Location
Reptiles/Amphibians
EPA Bullfrog Tissue Study - Weston, 2003
Bullfrog Tissue Sample Location
EPA Wood Frog Study - Fort, 2000
Wood Frog Target Site (Vernal Pools)

Fish
EPA Ecological Characterization Report - Woodlot, 1998
Fish Characterization and Tissue Sample Reach
Fish Sample Area
EPA Bass/Bluegill Study, Tillitt, 2003
Fish Sample Area
Birds
EPA - Tree Swallow Report - Custer, 2002
Swallow Box Location
Bird Tissue Sample Location
GE - Robin Report - ARCADIS G&M, 2002
Robin Tissue Sample Location

Mammals
EPA - Ecological Characterization Report - Woodlot, 2003
Mink/Otter Scent Post Location
Snow Track Transects

Reference Site Location
Housatonic River Watershed

Total PCB (tPCB) (mg/kg) (0-6" depth)
< 1
1 - 5
5 - 25
25 - 100
> 100

Ecological Risk Assessment
GE/Housatonic River Project Site
Rest of River

FIGURE 2.8-22
BIOLOGICAL SAMPLE LOCATIONS IN THE STUDY REFERENCE AREAS

c:\gepitt\era_060204\era_biotia__21tiles_bigfonts.apr | Reference | c:\gepitt\era_060204\exports\in\fig2-8-22_101504.eps | 12:13 PM, 10/27/2004 |

2.9 WEIGHT-OF-EVIDENCE APPROACH TO ANALYSIS

Inferences in ERAs are often made by weight-of-evidence (WOE) rather than traditional scientific standards of proof (EPA 1992). The WOE approach is a process by which measurement endpoints are related to an assessment endpoint to evaluate whether significant risk is posed to the environment (Menzie et al. 1996). A formal WOE can range from a simple qualitative assessment to a highly quantitative evaluation; however, no matter what form the WOE takes, it should provide documentation of the thought process used when assessing potential ecological risk.

The term "line of evidence" as used in this discussion follows the definition of "Information derived from different sources or by different techniques that can be used to describe and interpret risk estimates" provided in the *Guidelines for Ecological Risk Assessment* (EPA 1998). Unlike the term "weight-of-evidence," this definition does not imply assignment of qualitative or quantitative weightings to information. The three general lines of evidence under which most measurement endpoints fall are (Hull and Suter 1994; Suter et al. 1995):

- Biological survey data that indicate the state of the receiving environment.
- Media toxicity data that indicate whether the contaminated media are toxic (i.e., laboratory or in situ toxicity testing).
- Single contaminant toxicity data that indicate the toxic effects of the concentration measured in site media (e.g., exposure modeling).

For this ERA, lines of evidence include measurement endpoints (see Table 2.8-1) that will be formally evaluated using the WOE approach and additional technical information from site-specific studies and literature reviews that will be considered during risk characterization.

Two or three general lines of evidence were generally considered in evaluating potential risk for each assessment endpoint. A more detailed presentation of the specific lines of evidence used in this risk assessment is provided in the appendix for each assessment endpoint.

The WOE approach used in this ERA for each of the assessment endpoints follows the approach originally described in the *Massachusetts Weight-of-Evidence Special Report* (Menzie et al. 1996).

According to Menzie et al. (1996), WOE is reflected in three characteristics of measurement endpoints: (1) the weight assigned to each measurement endpoint; (2) the magnitude of response observed in the measurement endpoint; and (3) the degree of concurrence among outcomes of multiple measurement endpoints for a given assessment endpoint.

First, weights are assigned to measurement endpoints based on 10 attributes (summarized in Table 2.9-1) related to: (1) strength of association between assessment and measurement endpoints; (2) data and study quality; and (3) study design and execution. In either a quantitative or qualitative WOE analysis, the process of assigning weights to measurement endpoints can incorporate two elements:

1. The relative importance assigned to each attribute, a process referred to as "attribute scaling."
2. The score that each measurement endpoint receives with respect to each attribute, typically referred to as "attribute weighting."

For this ERA, it was assumed that all attributes were of equal importance so there was no "attribute scaling" conducted. The second element of the measurement endpoint weighing process, "attribute weighting," was performed for measurement endpoints using a qualitative scale ranging from low to high and following "attribute weighting" guidelines provided in Menzie et al. (1996, Table 2). This process, even when following the guidelines, is somewhat subjective and was accomplished using the combined professional judgment of the ERA team members.

After assigning a weight for each of the 10 attributes, a total measurement endpoint value was determined by averaging the 10 attribute weights. Consistency in the weighting process was ensured by assigning each attribute weight a numerical score of 1 (low) through 5 (high). The final qualitative measurement endpoint value was determined by applying the following classification scale to the arithmetic average of the attribute weights: 1-1.49 (Low), 1.50-2.49 (Low/Moderate), 2.50-3.49 (Moderate), 3.50-4.49 (Moderate/High), and ≥4.5 (High).

This process is further described in the appendix for each assessment endpoint. Figure 2.9-1 provides a generic example of the measurement endpoint weighting process used to evaluate each assessment endpoint.

Table 2.9-1

Attributes for Judging Measurement Endpoints

1. Strength of Association Between Assessment and Measurement Endpoints
Biological linkage between measurement endpoint and assessment endpoint—This attribute refers to the extent to which the measurement endpoint is representative of, correlated with, or applicable to the assessment endpoint. If there is no biological linkage between a measurement endpoint (e.g., a study that may have been performed for some other purpose) and the assessment endpoint of interest, then that study should not be used to evaluate the stated assessment endpoint. Biological linkage pertains to similarity of effect, target organ, mechanism of action, and level of ecological organization. **Correlation of stressor to response**—This attribute relates to the degree to which a correlation is observed between levels of exposure to a stressor and levels of response and the strength of that correlation. **Utility of measure**—This attribute relates to the ability to judge results of the study against well-accepted standards, criteria, or objective measures. As such, the attribute describes the applicability, certainty, and scientific basis of the measure, as well as the sensitivity of a benchmark in detecting environmental harm. Examples of objective standards or measures for judgment might include ambient water quality criteria, sediment quality criteria, biological indices, and toxicity or exposure thresholds recognized by the scientific or regulatory community as measures of environmental harm.
2. Data and Overall Study Quality
Quality of data and overall study—This attribute reflects the degree to which data quality objectives and other recognized characteristics of high quality studies are met. The key factor affecting the quality of the data is the appropriateness of data collection and analysis practices. The key factor of the quality of the study is the appropriateness and implementation of the experimental design and the minimization of confounding factors. If data are judged to be of poor or no quality, the study would be rejected for use in the ERA.

Table 2.9-1

Attributes for Judging Measurement Endpoints (Continued)

3. Design and Execution
Site-specificity—This attribute relates to the extent to which media, species, environmental conditions, and habitat types that are used in the study design reflect the site of interest.
Sensitivity of the measurement endpoint to detecting changes—This attribute relates to the ability to detect a response in the measurement endpoint, expressed as a percentage of the total possible variability that the endpoint is able to detect. Additionally, this attribute reflects the ability of the measurement endpoint to discriminate between responses to a stressor and those resulting from natural or design variability and uncertainty.
Spatial representativeness—This attribute relates to the degree of compatibility or overlap between the study area, locations of measurements or samples, locations of stressors, and locations of ecological receptors and their points of potential exposure.
Temporal representativeness—This attribute relates to the temporal compatibility or overlap between the measurement endpoint (when data were collected or the period for which data are representative) and the period during which effects of concern would be likely to be detected. Also linked to this attribute is the number of measurement or sampling events over time and the expected variability over time.
Quantitativeness—This attribute relates to the degree to which numbers can be used to describe the magnitude of response of the measurement endpoint to the stressor. Some measurement endpoints may yield qualitative or hierarchical results, while others may be more quantitative.
Use of a standard method—The extent to which the study follows specific protocols recommended by a recognized scientific authority for conducting the method correctly. Examples of standard methods are study designs or chemical measures published in the Federal Register or the Code of Federal Regulations, developed by ASTM, or repeatedly published in the peer-reviewed scientific literature, including impact assessments, field surveys, toxicity tests, benchmark approaches, toxicity quotients, and tissue residue analyses. This attribute also reflects the suitability and applicability of the method to the endpoint and the site, as well as the need for modification of the method.

Source: Menzie et al. 1996.

Score each measurement endpoint from low to high
Assessment Endpoint:____

Attribute	Measurement Endpoint A	Measurement Endpoint B	Measurement Endpoint C
I. Relationship between Measurement and Assessment Endpoints			
▪ Degree of Association	Moderate	High	High
▪ Stressor/Response	High	Moderate	High
▪ Utility of Measure	Moderate	High	High
II. Data Quality			
▪ Quality of data	High	High	High
III. Study Design			
▪ Site-specificity	High	High	High
▪ Sensitivity	Moderate	Low	High
▪ Spatial representativeness	Moderate	High	Moderate
▪ Temporal representativeness	Low	Low	Moderate
▪ Quantitativeness	High	High	High
▪ Use of a standard method	Moderate	Moderate	Moderate
Total Endpoint Value	Moderate	Moderate	Moderate-High

Figure 2.9-1 Example Endpoint Weighting Sheet

To ensure that the selected measurement endpoints would result in the achievement of the study objectives, a preliminary WOE was conducted in preparing the SIWP. Therefore, it is expected that low attribute weights will not typically be assigned if the study was conducted as planned, and total endpoint values will typically be in the moderate to high range.

The second step of the Menzie et al. (1996) approach is to evaluate the magnitude of response in the measurement endpoint, considering two questions:

- Does the measurement endpoint indicate the presence or absence of risk (yes, no, or undetermined)?
- Is the response low or high?

Figure 2.9-2 illustrates a matrix for an assessment endpoint that provides a simple communication tool summarizing the conclusions of the WOE evaluation of the magnitude of response.

Assessment Endpoint: ______________________________

Measurement Endpoints	Weighting Score (High, Moderate, Low)	Evidence of Harm (Yes, No, Undetermined)	Magnitude (High, Intermediate, Low)
Endpoint A			
Endpoint B			
Endpoint C			

Adapted from: Menzie et al. 1996.

Figure 2.9-2 Scoring Sheet for Evidence of Harm and Magnitude

The third step of the WOE process evaluates the degree of concurrence among measurement endpoints by plotting the output of the two preceding steps on a matrix for all measurement endpoints associated with a given assessment endpoint (see Figure 2.9-3). The matrix allows easy visual examination of agreements or divergences among measurement endpoints, facilitating interpretation with respect to the assessment endpoint. Logical connections, interdependence, and correlations among endpoints should also be considered when evaluating concurrence. The generalized matrix shown in Figure 2.9-3 is used for each assessment endpoint to illustrate the results of the WOE assessment of risks of PCBs and other COCs. Completed matrices specific to each assessment endpoint are presented in the respective appendix for each endpoint and each summary section of the report.

Assessment Endpoint: **Maintenance of a benthic community that can serve as a prey base for local fish**

Harm/Magnitude	Low Weight	Moderate Weight	High Weight
Yes/High			A
Yes/Low		B	
Undetermined			
No/Low	C		
No/High			

Notes:
Use letter designations to place measurement endpoints in boxes.
→ Indicates increasing confidence in weight.

Adapted from: Menzie et al., 1996.

Figure 2.9-3 Example of Qualitative Assessment

2.10 EXTRAPOLATION OF RISK ESTIMATES FOR SELECTED ENDPOINTS DOWNSTREAM OF THE PSA

Because of the decline in PCB mass and concentrations and the associated decrease in the amount of data collected downstream of the PSA, the detailed approach followed in assessing ecological risks in the PSA was not appropriate or possible. An estimate of potential ecological risks was developed using mapping (GIS) techniques and threshold concentrations that, if exceeded, would indicate potential risk to six selected target groups: benthic invertebrates, amphibians, warm-water fish, trout, mink, and bald eagles. These target groups were selected based on the risks for these organisms observed in the PSA, and the occurrence of these organisms in the reaches downstream.

For each of these groups, a maximum acceptable threshold concentration (MATC) for tPCBs in the appropriate medium was developed, based primarily on the detailed risk assessment performed for the PSA. The MATC was then compared to available medium-specific data for areas downstream of the PSA to Long Island Sound. Areas of exceedances (HQ > 1), indicating potential risk, were plotted on maps of the river. The specific approaches developed for each of the six target groups are discussed in the appropriate appendices and summary sections of this report.

2.11 REFERENCES

ATSDR (Agency for Toxic Substances and Disease Registry). 1995. *Toxicological Profile for Polycyclic Aromatic Hydrocarbons*. Prepared by Research Triangle Institute. August 1995.

ATSDR (Agency for Toxic Substances and Disease Registry). 1997. *Polychlorinated Biphenyls (PCBs)*. www.atsdr.cdc.gov/tfacts17.html. June 12, 2001.

ATSDR (Agency for Toxic Substances and Disease Registry). 1998. *Toxicological Profile Chlorinated Dibenzo-p-Dioxins*. Prepared by Research Triangle Institute. December 1998.

ATSDR (Agency for Toxic Substances and Disease Registry). 2000. *Toxicological Profile for Polychlorinated Biphenyls*. November 2000.

Aulerich, R.J., R.K. Ringer, and J. Safronoff. 1986. Assessment of primary versus secondary toxicity of Aroclor 1254 to mink. *Archives of Environmental Contamination and Toxicology* 15:393-399.

BBL (Blasland, Bouck & Lee, Inc.). 2002. *Evaluation of Mink – Presence/Absence, Distribution, and Abundance in the Housatonic River Floodplain*. Report prepared for the General Electric Company.

BBL (Blasland, Bouck & Lee, Inc.) and QEA (Quantitative Environmental Analysis, LLC). 2003. *Housatonic River – Rest of River RCRA Facility Investigation Report*, Volume 1. Prepared for General Electric Company. January 2003.

Bedard, D. and R. May. 1996. Characterization of the polychlorinated biphenyls in the sediments of Woods Pond: Evidence for microbial dechlorination of Aroclor 1260 in situ. *Environ. Sci. Technol.* 30:237-245.

Birnbaum, L.S. and M.J. DeVito. 1995. Use of toxic equivalency factors for risk assessment for dioxins and related compounds. *Toxicology* 105:391-401.

Boonstra, R. 2002. *Demography of Short-Tailed Shrew Populations Living on PCB Contaminated Sites*. Report prepared for General Electric Company.

Callahan, M.A., M.W. Slimak, N.W. Gabel, I.P. May, C.F. Fowler, J.R. Freed, P. Jennings, R.L. Durfee, F.C. Whitmore, B. Maestri, W.R. Mabey, B.R. Holt, and C. Gould. 1979. *Water-Related Environmental Fate of 129 Priority Pollutants*. Chapter 36. EPA-440/4-79-029a.

Chadwick & Associates, Inc. 1994. *Aquatic Ecology Assessment of the Housatonic River, Massachusetts 1993.* Chadwick & Associates, Inc., Littleton, CO.

Chem Risk. 1997. *Work Plan for the Ecological Risk Assessment of the Housatonic River:* Volumes I and II. Chem Risk (24 May 1997).

Connor, K., S. Safe, and C.R. Jefcoate. 1995. Structure-dependent induction of CYP2B by polychlorinated biphenyl congeners in female Sprague-Dawley rats. *Biochem. Pharmacol.* 50(11):1913-1920.

Cornelissen, G., H. Rigterink, M.M.A. Ferdinandy, and P. van Noort. 1998. Rapidly desorbing fractions of PAHs in contaminated sediments as a predictor of the extent of bioremediation. *Environmental Science and Technology* 32:966-970.

Dyke, P.H. and J. Stratford. 2002. Changes to TEF schemes can have significant impacts on regulation and management of PCDD/F and PCB. *Chemosphere* 47:103-116.

Eisler, R. 1986. *Polychlorinated Biphenyl Hazards to Fish, Wildlife, and Invertebrates: A Synoptic Review*. Contaminant Hazard Reviews-7. 81 pp. PB86-170057.

Eisler, R. 1987. *Polycyclic Aromatic Hydrocarbon Hazards to Fish, Wildlife, and Invertebrates: A Synoptic Review*. Contaminant Hazard Reviews, Biological Report 85(1.11). May 1987.

Eisler, R. and A. Belisle. 1996. *Planar PCB hazards to fish, wildlife, and invertebrates: A synoptic review contaminant hazard reviews*. Biological Report 31.

EPA (U.S. Environmental Protection Agency). 1980. *Ambient Water Quality Criteria Document: Polychlorinated Biphenyls.* p. B-2. EPA 440/5-80-068. As cited in ATSDR 2000.

EPA (U.S. Environmental Protection Agency). 1992. *Framework for Ecological Risk Assessment.* Risk Assessment Forum, Washington, DC. EPA/630/R-92/001.

EPA (U.S. Environmental Protection Agency). 1993. *Wildlife Exposure Factors Handbook.* EPA/600/R-93/187a. Office of Research and Development, Washington, DC.

EPA (U.S. Environmental Protection Agency). 1994. *A Review of Ecological Assessment Case Studies from a Risk Assessment Perspective,* Volume II. Office of Research and Development, Washington, DC. EPA/630/R-94-003.

EPA (U.S. Environmental Protection Agency). 1996. *PCBs: Cancer Dose-Response Assessment and Application to Environmental Mixtures*. National Center for Environmental Assessment. Office of Research and Development, Washington, DC. EPA/600/P-96/001F.

EPA (U.S. Environmental Protection Agency). 1997. *Ecological Risk Assessment Guidance for Superfund: Process for Designing and Conducting Ecological Risk Assessments. Interim Final.* Solid Waste and Emergency Response, U.S. Environmental Protection Agency. EPA-540-R-97-006.

EPA (U.S. Environmental Protection Agency). 1998. *Guidelines for Ecological Risk Assessment*. Risk Assessment Forum, Washington, DC. EPA/630/R-95/002F.

EPA (U.S. Environmental Protection Agency). 1999. Ecological Risk Assessment and Risk Management Principles for Superfund Sites. OSWER Directive 9285.7-28P.

EPA (U.S. Environmental Protection Agency). 2002. Integrated Risk Information System.

Haffner, G.D., M. Tomczak, and R. Lazar. 1994. Organic contaminant exposure in the Lake St. Clair food web. *Hydrobiologia* 281:19-27.

Hansen, L.G. 1994. Halogenated Aromatic Compounds. In *Basic Environmental Toxicology*. L.G. Cockerham, and B.S. Shane eds. CRC Press. pp. 199-230. ISBN 0-8493-8851-1.

Harris, S.L. 1997. *Inorganic and Organic Constituents and Grain-Size Distribution in Streambed Sediment and Ancillary Data for the Connecticut, Housatonic, and Thames River Basins Study Unit, 1992-1994.* USGS Open-File Report 96-397.

Henning, M.H. 2002. *Productivity and Density of Belted Kingfishers on the Housatonic River*. Prepared by ARCADIS G&M, Inc. Prepared for General Electric Company.

Henning, M.H., S. Robinson, and K. Jenkins. 2002. *Robin Productivity in the Housatonic River Watershed*. Prepared by ARCADIS G&M, Inc. Prepared for General Electric Company.

Hornshaw, T.C., R.J. Aulerich, and H.E. Johnson. 1983. Feeding Great Lakes fish to mink: Effects on mink and accumulation and elimination of PCBs by mink. *Journal of Toxicology and Environmental Health* 11:933-946. As cited in EPA 1996.

Huff, J. 1992. 2,3,7,8-TCDD: A potent and complete carcinogen in experimental animals. *Chemosphere.* 25:173-176.

Hull, R.N. and G.W. Suter II. 1994. *Using the Weight-of-Evidence Approach for Ecological Risk Assessment at a DOE Facility.* SETAC Denver, CO. November 1, 1994.

HVA (Housatonic Valley Association). 2001. *The Housatonic River Watershed*. URL http://www.hvathewatershedgroup.org/HousatonicFactSheet.htm.

Johnson, R., J. Tietge, and S. Botts. 1992. Carcinogenicity of 2,3,7,8-TCDD to medaka. *Toxicologist.* 12(1), 476.

Kennedy, S.W., A. Lorenzen, and R.J. Norstrom. 1996. Chicken embryo hepatocyte bioassay for measuring cytochrome P4501a-based 2,3,7,8-tetrachlorodibenzo-p-dioxin equivalent concentrations in environmental samples. *Environmental Science and Technology* 30:706-715.

Lindstrom, G., L.S. Haug, T. Nicolaysen, and E. Dybing. 2002. Comparability of world-wide analytical data on PCDDs, PCDFs and non-ortho PCBs in samples of chicken, butter and salmon. *Chemosphere* 47:139-146.

MacDonald, D.D., C.G. Ingersoll, and T.A. Berger. 2000. Development and evaluation of consensus-based sediment quality guidelines for freshwater ecosystems. *Archives of Environmental Contamination and Toxicology* 39:20-31.

Mackay, D. 1989. Modeling the long-term behavior of an organic contaminant in a large lake application of PCBs in Lake Ontario. *Journal of Great Lakes Research* 15:283-297.

MassWildlife. 2002. http://www.state.ma.us/dfwele/dfw/dfw_wd/htm.

Maughan, J.T. 1993. *Ecological Assessments of Hazardous Waste Sites*. Van Nostrand Reinhold, New York, NY.

Mayes, B.A., E.E. McConnell, and B.H. Nell. 1998. Comparative carcinogenicity in Sprague Dawley rats of polychlorinated biphenyl mixtures Aroclor 1016, 1242, 1254, and 1260. *Toxicol. Sci.* 41(1):62-76.

Menzie, C., M.H. Henning, J. Cura, K. Finkelstein, J. Gentile, J. Maughan, D. Mitchell, S. Petron, B. Potocki, S. Svirsky, and P. Tyler. 1996. Special Report of the Massachusetts Weight-of-Evidence Workgroup: A Weight-of-Evidence Approach for Evaluating Ecological Risks. *Human and Ecological Risk Assessment* 2(2):277-304.

Moore, D.R.J., B.E. Sample, G.W. Suter, B.R. Parkhurst, R.S. Teed. 1999. A Probabilistic Risk Assessment of the Effects of Methylmercury and PCBs on Mink and Kingfishers along East Fork Poplar Creek, Oak Ridge, Tennessee, USA. *Environmental Toxicology and Chemistry*: vol. 18, No. 12, pp. 2941-2953.

NATO (North Atlantic Treaty Organization). 1988. *Pilot study on international information exchange on dioxins and related compounds. International toxicity equivalent factor (I-TEF) method of risk assessment for complex mixtures of dioxins and related compounds*. Committee on the Challenges of Modern Society, Report 176. August 1988.

Olcott, P.G. 1995. *Ground Water Atlas of the United States: Connecticut, Maine, Massachusetts, New Hampshire, New York, Rhode Island, Vermont: Carbonate Rock Aquifers*. U.S. Geological Survey Publications HA 730-M.

Paustenbach, D.J., R.J. Wennig, V. Lau, N.W. Harrington, D.K Rennix, and A.H. Parson. 1992. Recent developments on the hazards posed by 2,3,7,8-tetrachlorodibenzo-p-dioxin in soil: Implication for setting risk-based cleanup levels at residential and industrial sites. *Journal of Toxicology and Environmental Health* 36(2):103-149. As cited in ATSDR 1998.

Resetarits, W.J. 2002. *Experimental Analysis of the Context-Dependent Effects of Early Life-Stage PCB Exposure on Rana sylvatica.* Report prepared for General Electric Company.

Robertson, R.J and J. Jones. 2002. *Spatial and Demographic Effects on Tree Swallow Nest Quality and Reproductive Success*. Report prepared for General Electric Company.

Russell, R.W., F.A.P.C. Gobas, and G.D. Haffner. 1999. Role of chemical and ecological factors in trophic transfer of organic chemicals in aquatic food webs. *Environmental Toxicology and Chemistry* 18:1250-1257.

R2 (R2 Resource Consultants Inc.). 2002. *Evaluation of Largemouth Bass Habitat, Population Structure, and Reproduction in the Upper Housatonic River, Massachusetts*. Report prepared General Electric Company.

Safe, S.H. 1994. Polychlorinated biphenyls (PCBs): Environmental impact, biochemical and toxic responses, and implications for risk assessment. *Critical Reviews in Toxicology* 24(2): 87-149.

SCS (Soil Conservation Service, U.S. Department of Agriculture). 1970. Soil Survey of Litchfield County, Connecticut. Storrs, CT, USA.

SCS (Soil Conservation Service, U.S. Department of Agriculture). 1979. Soil Survey of New Haven County, Connecticut. Storrs, CT, USA.

SCS (Soil Conservation Service, U.S. Department of Agriculture). 1981. Soil Survey of Fairfield County, Connecticut. Storrs, CT, USA.

SCS (Soil Conservation Service, U.S. Department of Agriculture). 1988. Soil Survey of Berkshire County, Massachusetts, Amherst, MA, USA.

Shaw, G.R. and D.W. Connell. 1984. Factors controlling bioaccumulation of PCBs. In *PCBs and the Environment, Volume 1*. National Technical Information Service, Springfield, VA. pp. 501-516.

Suter, G.W., II. 1991. Ecological Endpoints. In *Ecological Assessment of Hazardous Waste Sites: A Field and Laboratory Reference*. U.S. Environmental Protection Agency. EPA/540/R-92/003.

Suter, G.W, II. 1993. *Ecological Risk Assessment.* Lewis Publishers, Boca Raton, FL. ISBN 0-87371-875-5.

Suter, G.W., II, B.E. Sample, D.S. Jones, T.L. Ashwood, and J.M. Loar. 1995. *Approach and Strategy for Performing Ecological Risk Assessments for the U.S. Department of Energy's Oak Ridge Reservation.* Oak Ridge National Laboratory, Oak Ridge, TN. ES/ER/TM-33/R2.

Swain, P.C. and K.B. Kearsley. 2000. *Classification of the Natural Communities of Massachusetts.* Massachusetts Natural Heritage and Endangered Species Program, Westborough, MA, USA.

Thomann, R.V. 1989. Bioaccumulation model of organic chemical distribution in aquatic food chains. *Environmental Science and Technology* 23:699-707.

Van den Berg, M., L. Birnbaum, A.T.C. Bosveld, B. Brunstrom, P. Cook, M. Freely, J.P. Giesy, A. Hanberg, R. Hasegawa, S.W. Kennedy, T. Kubiak, J.C. Larsen, F.X.R van Leeuwen, A.K.

Djien Liem, C. Nolt, R.E. Peterson, L. Poellinger, S. Safe, D. Schrenk, D. Tillitt, M. Tysklind, M. Younes, F. Waern, and T. Zacharewski. 1998. Toxic equivalency factors (TEFs) for PCBs, PCDDs, PCDFs for humans and wildlife. *Environmental Health Perspectives* 106(12):775-792.

Weatherbee, P.B. 1996. *Flora of Berkshire County Massachusetts*. The Berkshire Museum, Pittsfield, MA, USA.

Weatherbee, P.B. and G.E. Crow. 1992. Natural plant communities of Berkshire County, Massachusetts. *Rhodora* 94:171-209.

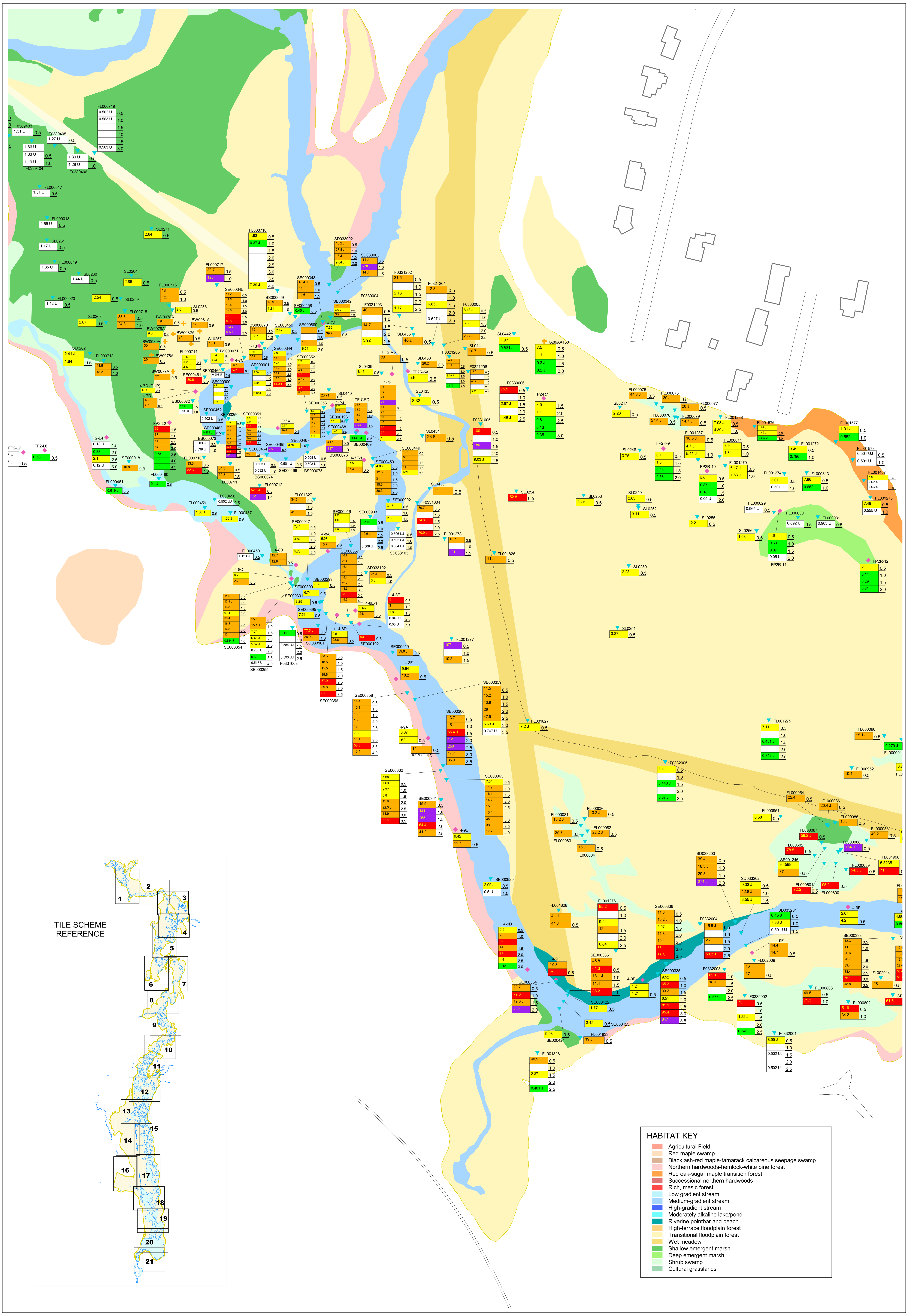
WESTON (Roy F. Weston, Inc.). 1998. *Source Area Characterization Report.* Prepared for U.S. Army Corps of Engineers and U.S. Environmental Protection Agency. DCN GEPM-072198-AABA.

WESTON (Roy F. Weston, Inc.). 2000. *Supplemental Investigation Work Plan for the Lower Housatonic River*. Prepared for U.S. Army Corps of Engineers and U.S. Environmental Protection Agency. 22 February 2000. DCN GEP2-020900-AAME.

WESTON (Weston Solutions, Inc.). 2004. *Modeling Framework Design: Modeling Study of PCB Contamination in the Housatonic River*. Prepared for U.S. Army Corps of Engineers and U.S. Environmental Protection Agency. DCN GE-042104-ACDP.

ATTACHMENT 2.1

TOTAL PCB RESULTS—REACHES 5 THROUGH 9



LEGEND:

- EPA - Corps of Engineers Sample Location
- EPA - START Sample Location
- GE Sample Location
- Open Water
- 10-year Floodplain

PCB Concentration (ppm)	result value	lower depth value
< 1	0.82	0.5
1 - 10	2.83	1.0
10 - 50	45.9	1.5
50 - 100	87.1	2.0
> 100	160	2.5
no sample		3.0
non-detect	0.65 U	3.5

result flag

NOTE:
Result flag indicates --
U = not detected at reported value
J = estimated detected value
UJ = estimated non-detected value

N

Scale in Feet

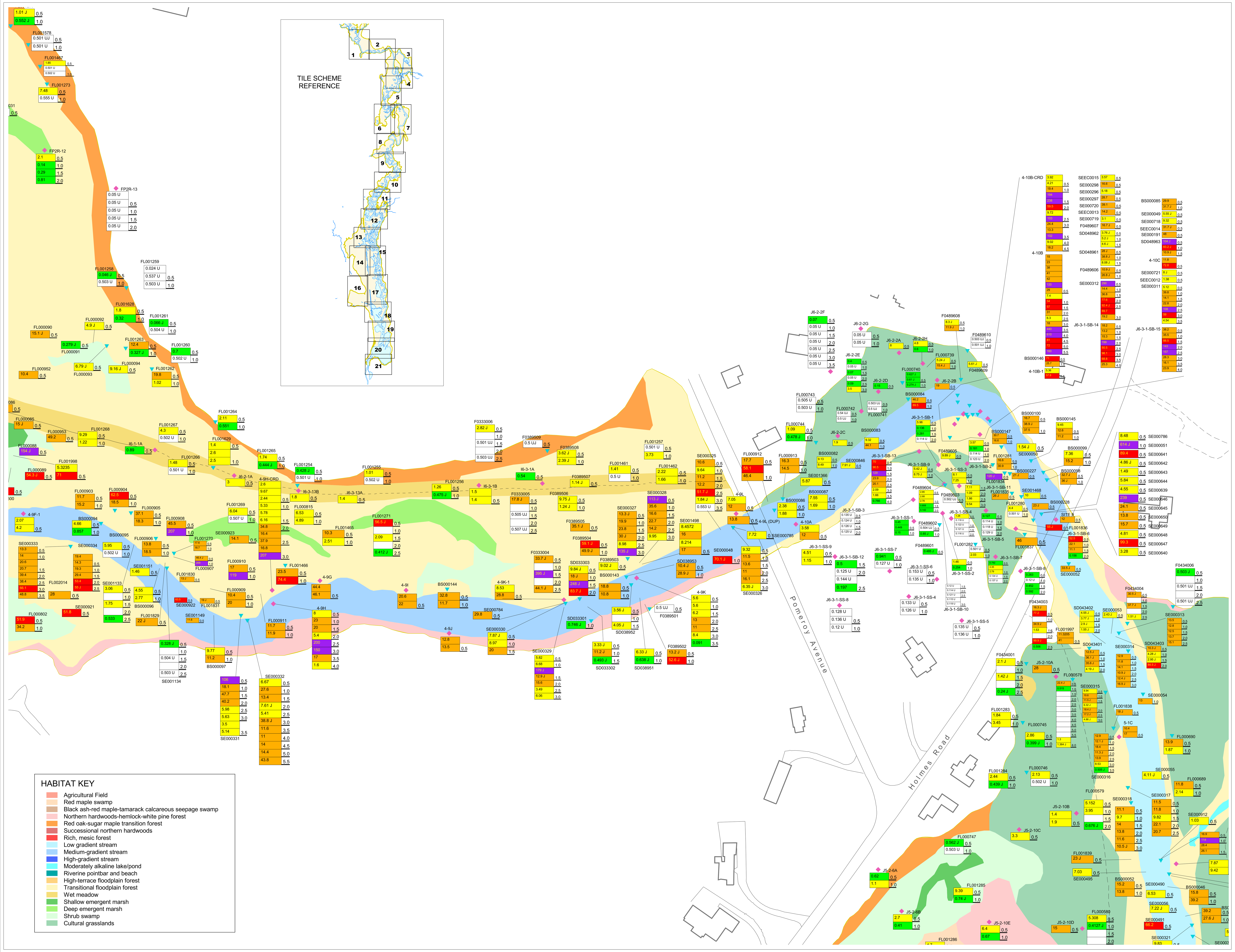
100 0 100 200 300

Housatonic River Project
Pittsfield, Massachusetts

TOTAL PCB RESULTS
REACHES 5 AND 6
TILE 1/21

NOTE: Datamart as of September 30, 2002

| O:\gepitt\aprs\reach5&6_APRs\stacks1a_sept02.apr | layout - tile 1 | o:\gepitt\epsfiles\portrait24x36\r\stacks1_sept02p.eps | 10:09 AM, 12/20/2002 |



LEGEND:

EPA - Corps of Engineers Sample Location

EPA - START Sample Location

GE Sample Location

Open Water

10-year Floodplain

PCB Concentration (ppm)

PCB Concentration (ppm)	result value	lower depth value
< 1	0.82	0.5
1 - 10	2.83	1.0
10 - 50	45.9	1.5
50 - 100	87.1	2.0
> 100	160	2.5
no sample		3.0
non-detect	0.65 U	3.5

result flag

NOTE: Result flag indicates --
U = not detected at reported value
J = estimated at or below reported value
UJ = estimated at or below reported value

N

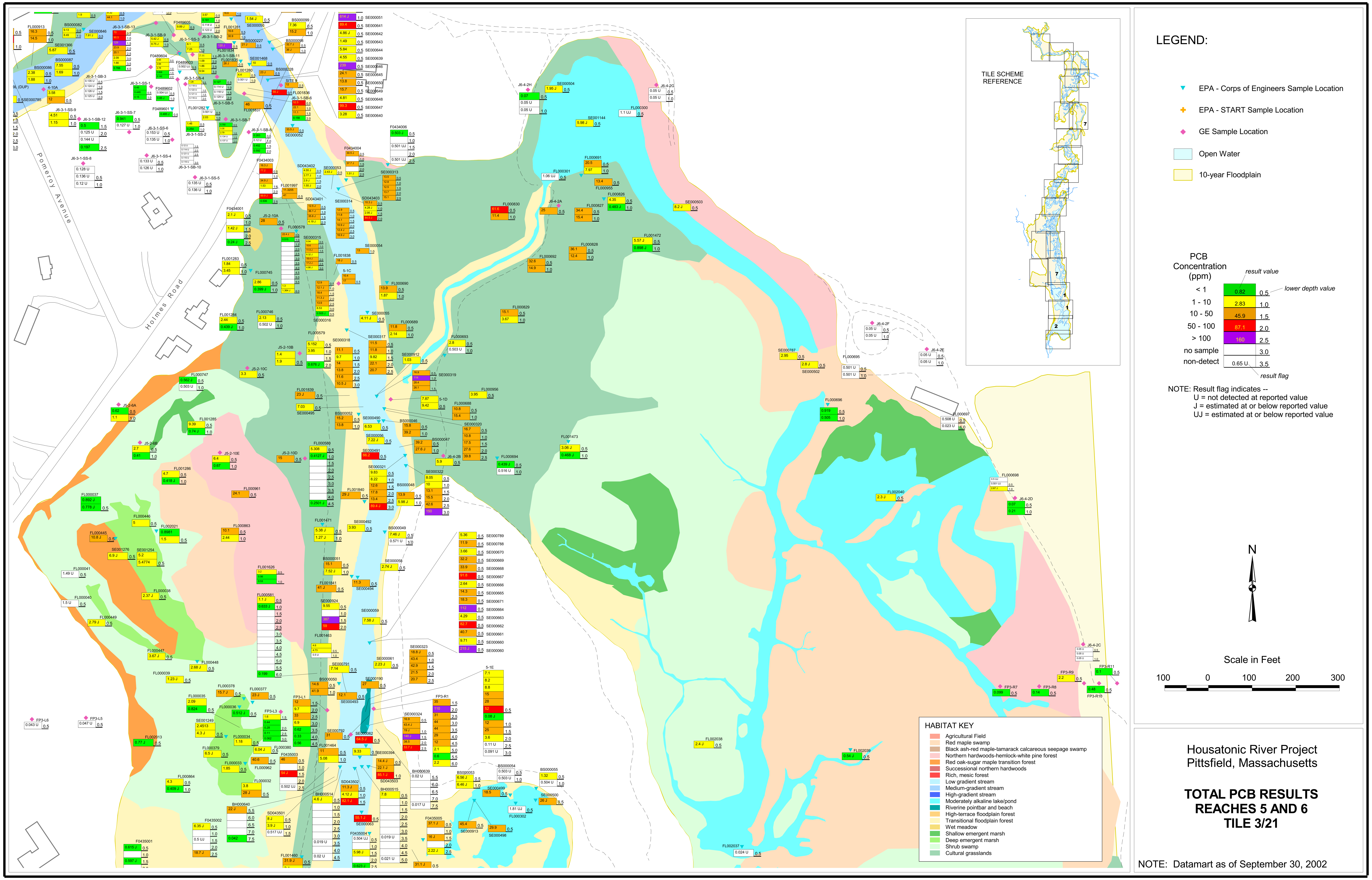
Scale in Feet

100 0 100 200 300

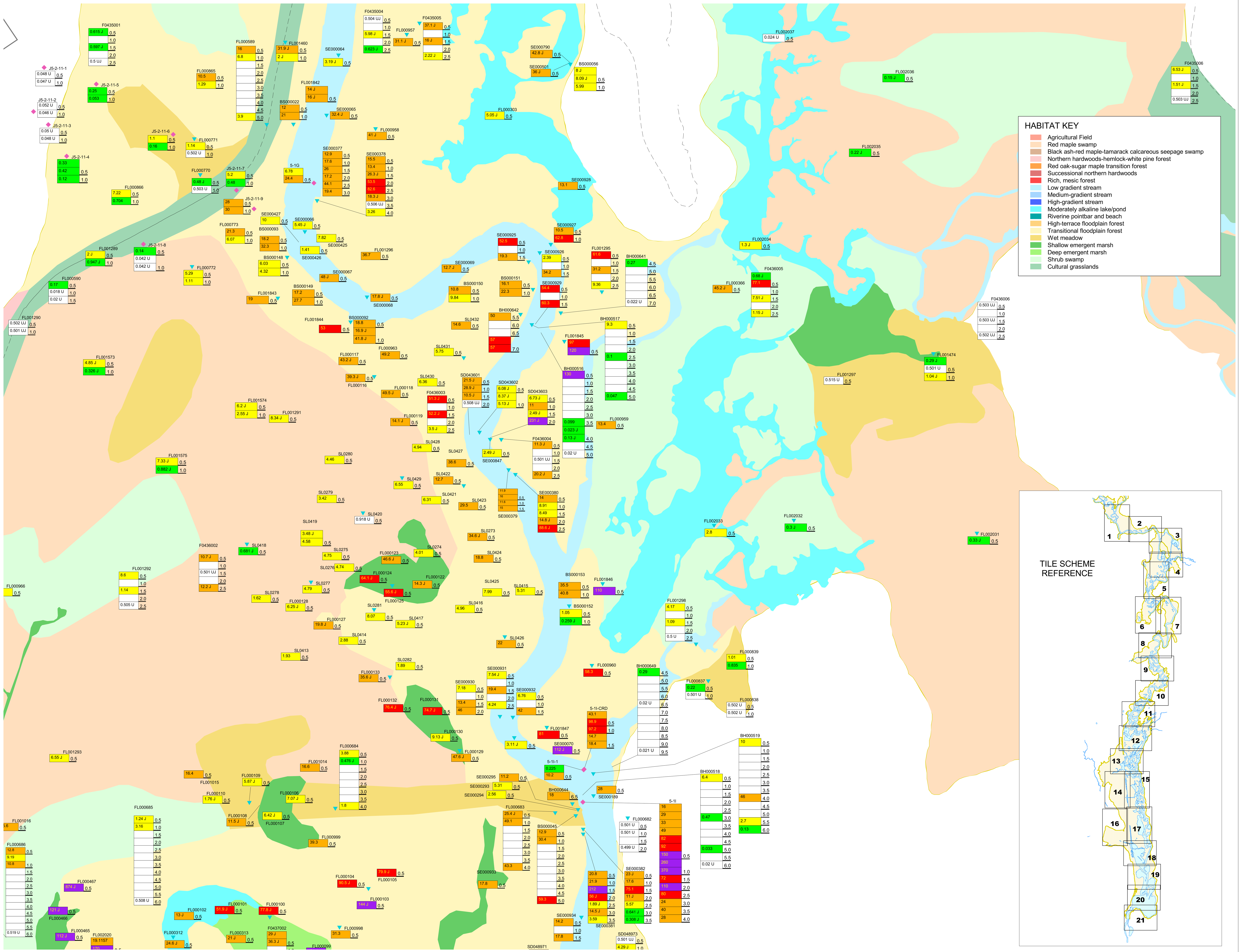
Housatonic River Project
Pittsfield, Massachusetts

TOTAL PCB RESULTS
REACHES 5 AND 6
TILE 2/21

NOTE: Datamart as of September 30, 2002



| O:\gepitt\aprs\reach5&6_APRs\stacks1a_sept02.apr | layout - tile 3 | o:\gepitt\epsfiles\landscape24x36\n\stacks3_sept02.eps | 10:12 AM, 12/20/2002 |



LEGEND:

EPA - Corps of Engineers Sample Location

EPA - START Sample Location

GE Sample Location

Open Water

10-year Floodplain

PCB Concentration (ppm)

PCB Concentration (ppm)	result value	lower depth value
< 1	0.82	0.5
1 - 10	2.83	1.0
10 - 50	45.9	1.5
50 - 100	87.1	2.0
> 100	160	2.5
no sample		3.0
non-detect	0.65 U	3.5

result flag

NOTE: Result flag indicates --
U = not detected at reported value
J = estimated at or below reported value
UJ = estimated at or below reported value

N

Scale in Feet

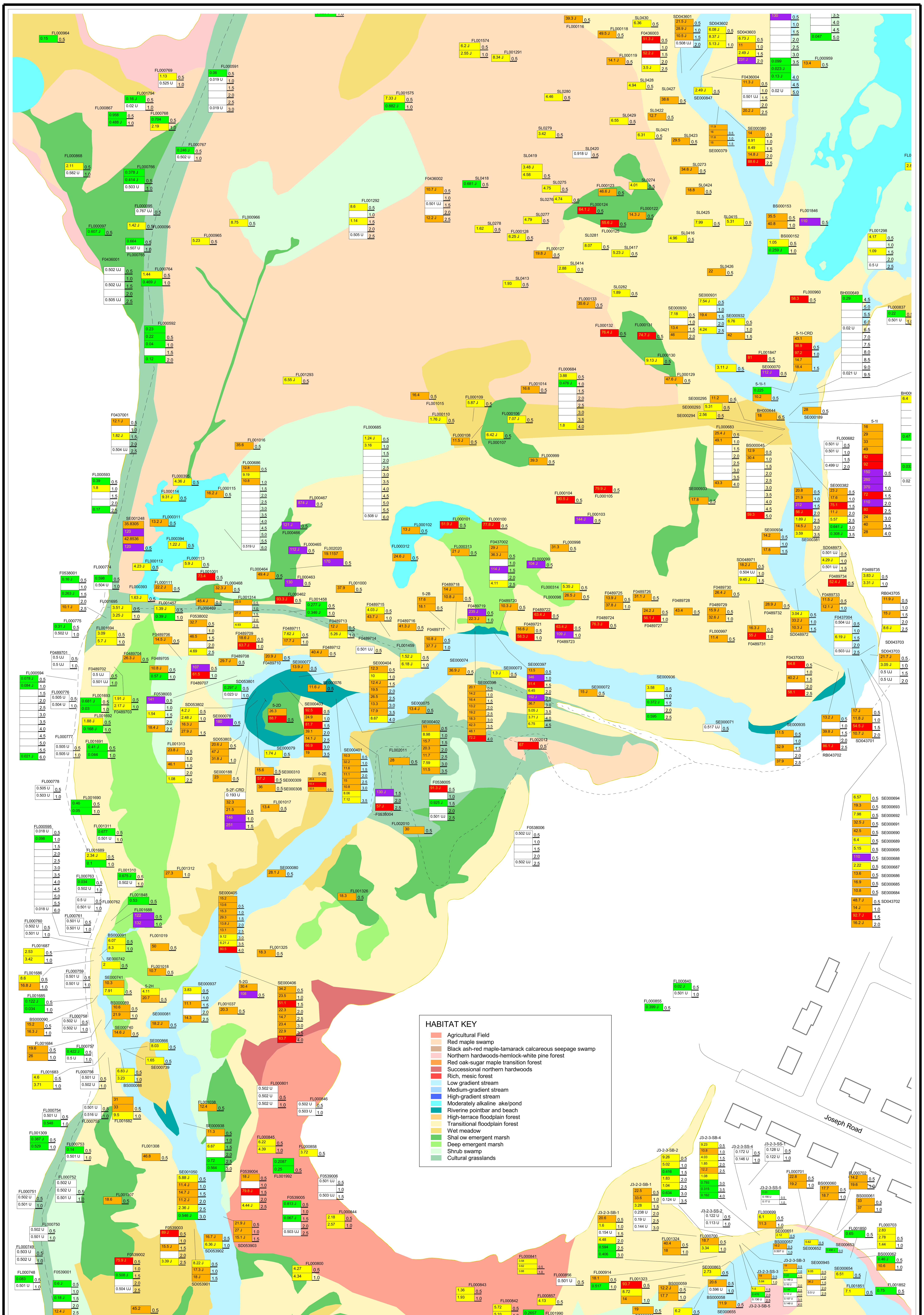
100 0 100 200 300

Housatonic River Project
Pittsfield, Massachusetts

TOTAL PCB RESULTS
REACHES 5 AND 6
TILE 4/21

NOTE: Datamart as of September 30, 2002

| O:\gepitt\aprs\reach5&6_APRs\stacks1b_sept02.apr | layout - tile 4 | o:\gepitt\epsfiles\landscape24x36\r\stacks4_sept02l.eps | 10:16 AM, 12/20/2002 |



LEGEND:

EPA - Corps of Engineers Sample Location

EPA - START Sample Location

GE Sample Location

Open Water

10-year Floodplain

PCB Concentration (ppm)

PCB Concentration (ppm)	result value	lower depth value
< 1	0.82	0.5
1 - 10	2.83	1.0
10 - 50	45.9	1.5
50 - 100	87.1	2.0
> 100	160	2.5
no sample		3.0
non-detect	0.65 U (result flag)	3.5

NOTE:
Result flag indicates --
U = not detected at reported value
J = estimated detected value
UJ = estimated non-detected value

N

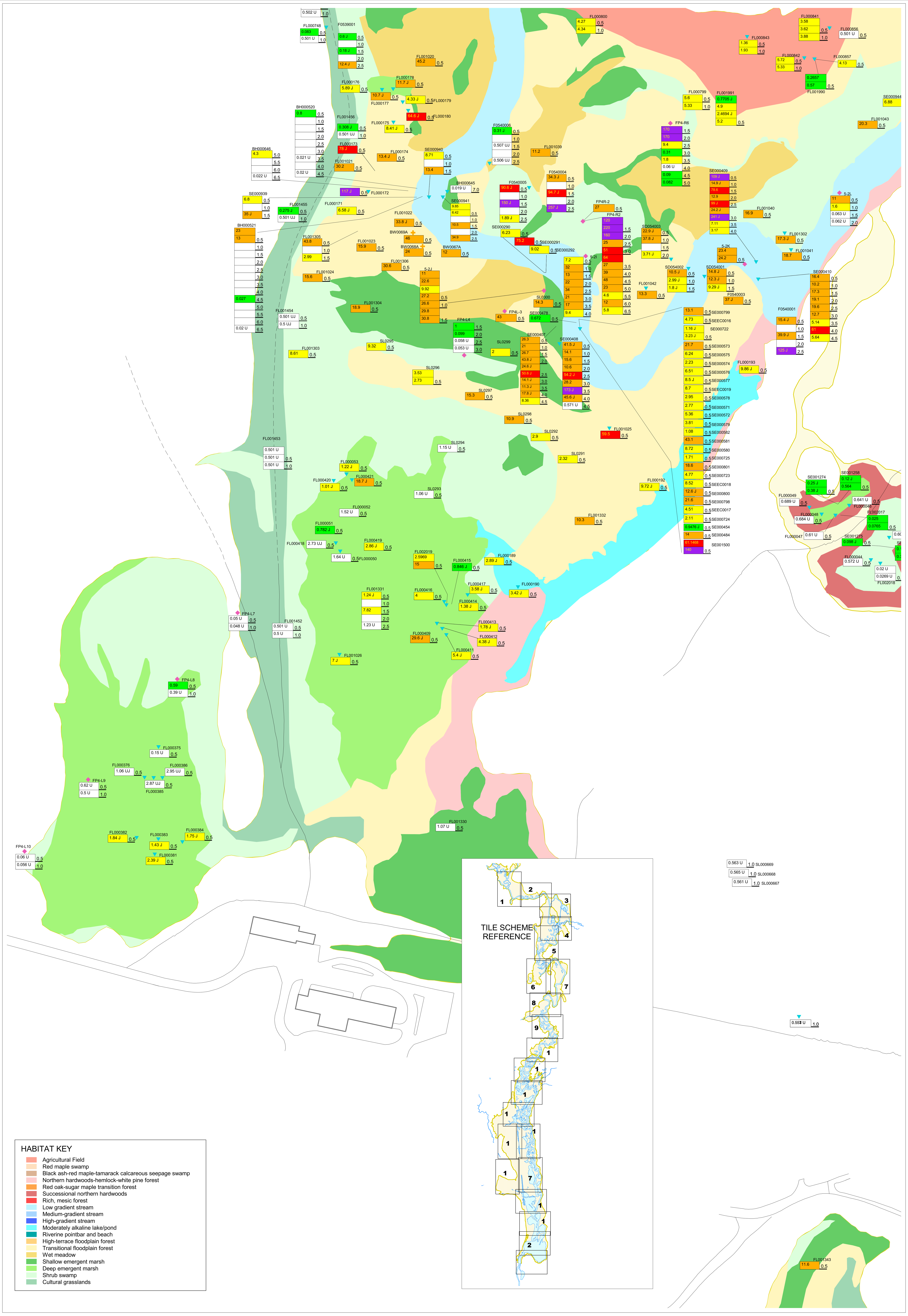
Scale in Feet

100 0 100 200 300

Housatonic River Project
Pittsfield, Massachusetts

TOTAL PCB RESULTS REACHES 5 AND 6 TILE 5/21

NOTE: Datamart as of September 30, 2002



LEGEND:

- EPA - Corps of Engineers Sample Location
- EPA - START Sample Location
- GE Sample Location
- Open Water
- 10-year Floodplain

PCB Concentration (ppm)

	result value	lower depth value
< 1	0.82	0.5
1 - 10	2.83	1.0
10 - 50	45.9	1.5
50 - 100	87.1	2.0
> 100	160	2.5
no sample		3.0
non-detect	0.65 U	3.5

result value; lower depth value; result flag

NOTE:
Result flag indicates --
U = not detected at reported value
J = estimated detected value
UJ = estimated non-detected value

Scale in Feet

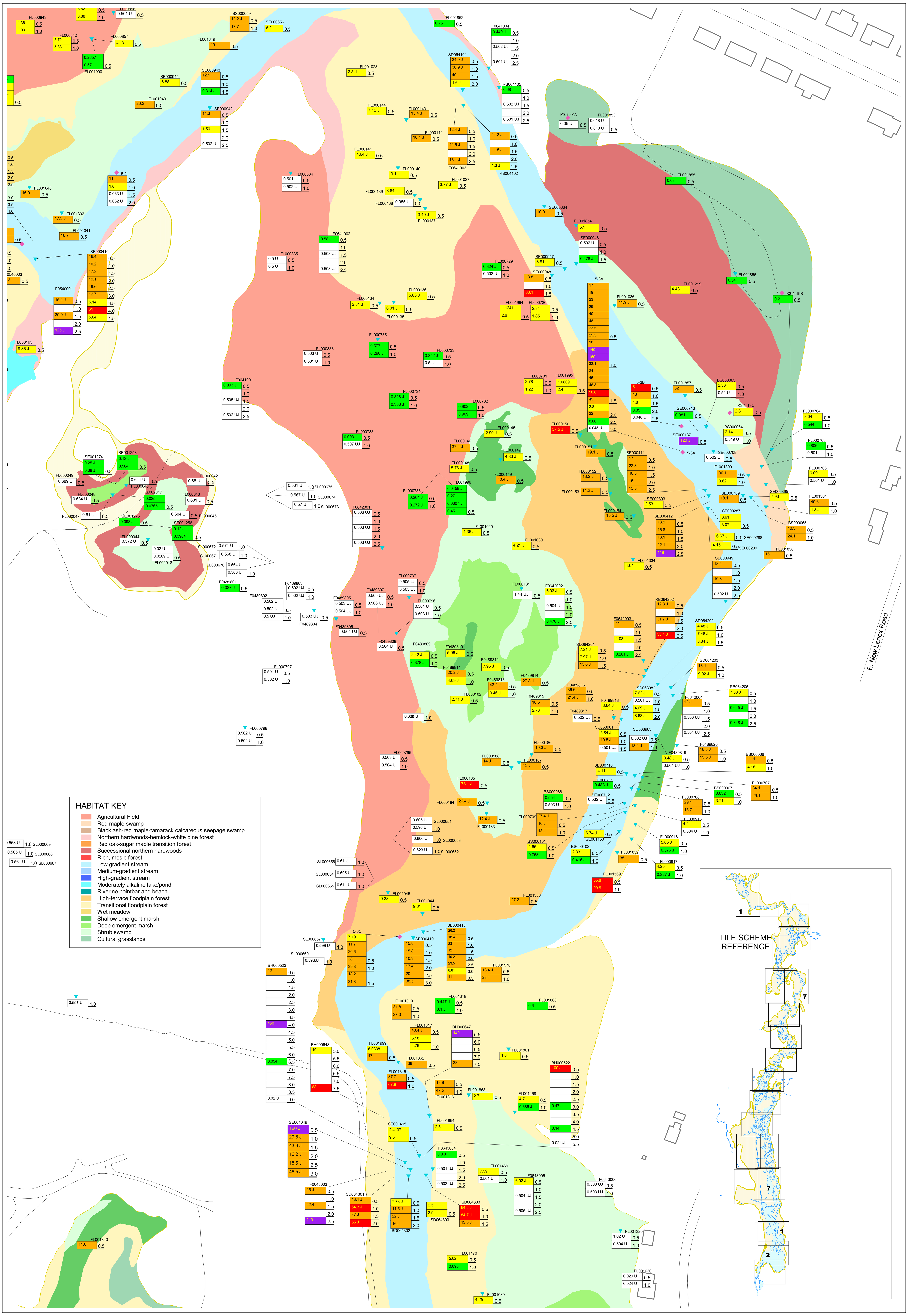
100 0 100 200 300

Housatonic River Project
Pittsfield, Massachusetts

**TOTAL PCB RESULTS
REACHES 5 AND 6
TILE 6/21**

NOTE: Datamart as of September 30, 2002

| O:\gepitt\aprs\reach5&6_APRs\stacks2_sept02.apr | layout - tile 6 | o:\gepitt\epsfiles\portrait24x36\r\stacks6_sept02t.eps | 10:18 AM, 12/20/2002 |



LEGEND:

- EPA - Corps of Engineers Sample Location
- EPA - START Sample Location
- GE Sample Location
- Open Water
- 10-year Floodplain

PCB Concentration (ppm)

PCB Concentration (ppm)	result value	lower depth value
< 1	0.82	0.5
1 - 10	2.83	1.0
10 - 50	45.9	1.5
50 - 100	87.1	2.0
> 100	160	2.5
no sample		3.0
non-detect	0.65 U	3.5

result flag

NOTE:
Result flag indicates --
U = not detected at reported value
J = estimated detected value
UJ = estimated non-detected value

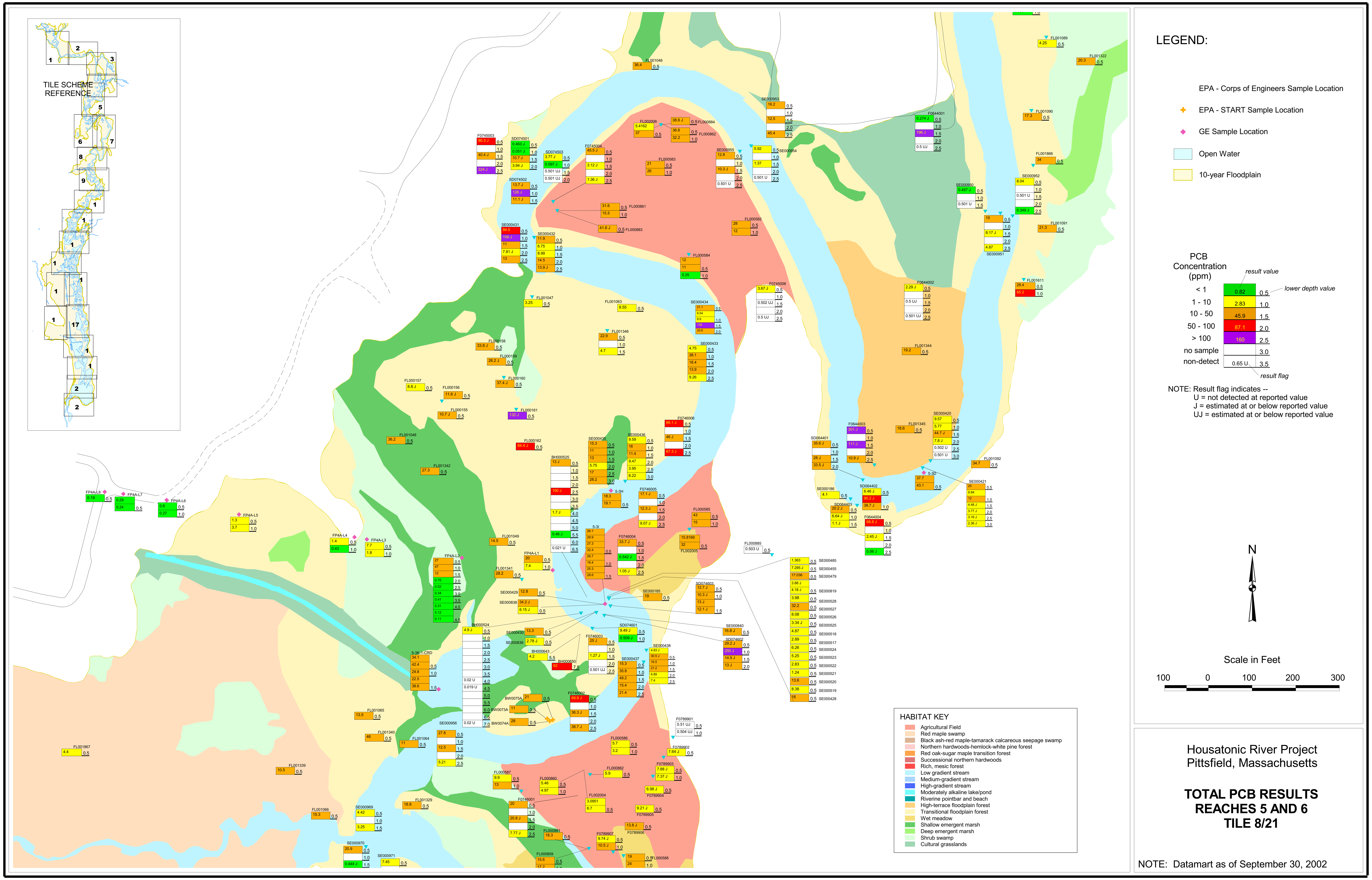
Scale in Feet

100 0 100 200 300

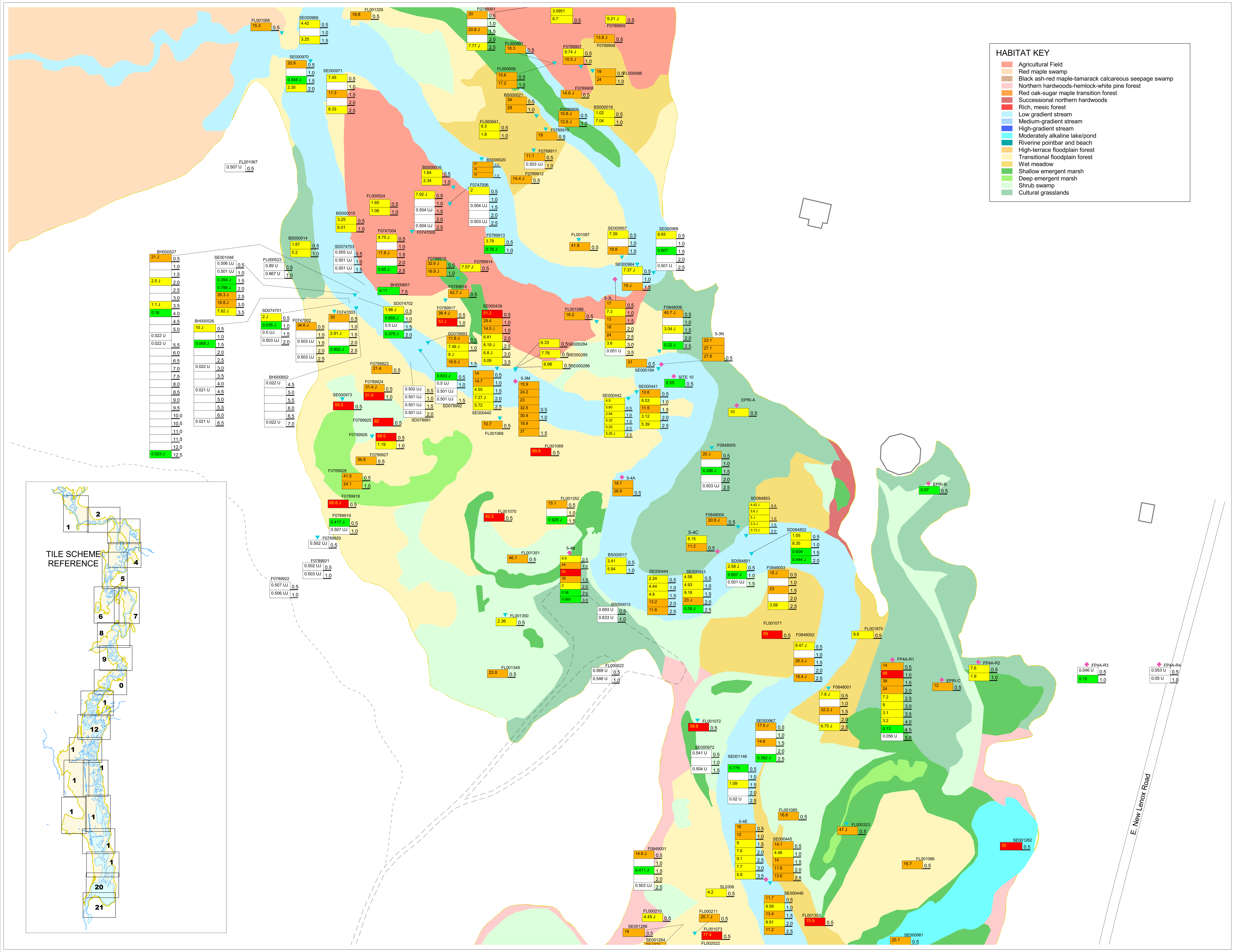
Housatonic River Project
Pittsfield, Massachusetts

TOTAL PCB RESULTS
REACHES 5 AND 6
TILE 7/21

NOTE: Datamart as of September 30, 2002



| O:\gepitt\aprs\reach5&6_APRs\stacks2_sept02.apr | layout - tile 8 | o:\gepitt\epsfiles\landscape24x36\m\stacks8_sept02l.eps | 10:20 AM, 12/20/2002 |



LEGEND:

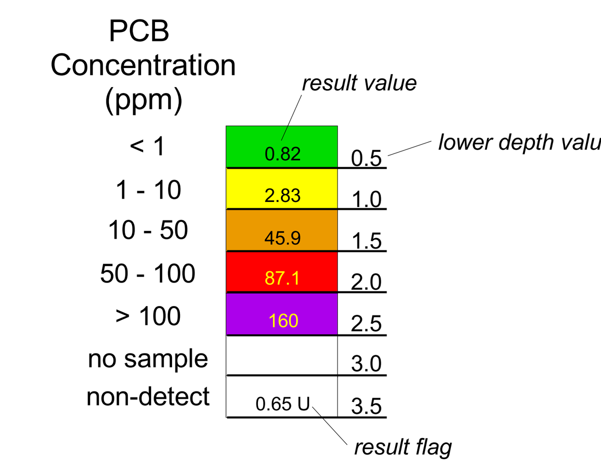
EPA - Corps of Engineers Sample Location

EPA - START Sample Location

GE Sample Location

Open Water

10-year Floodplain



NOTE: Result flag indicates --
U = not detected at reported value
J = estimated at or below reported value
UJ = estimated at or below reported value

N

Scale in Feet

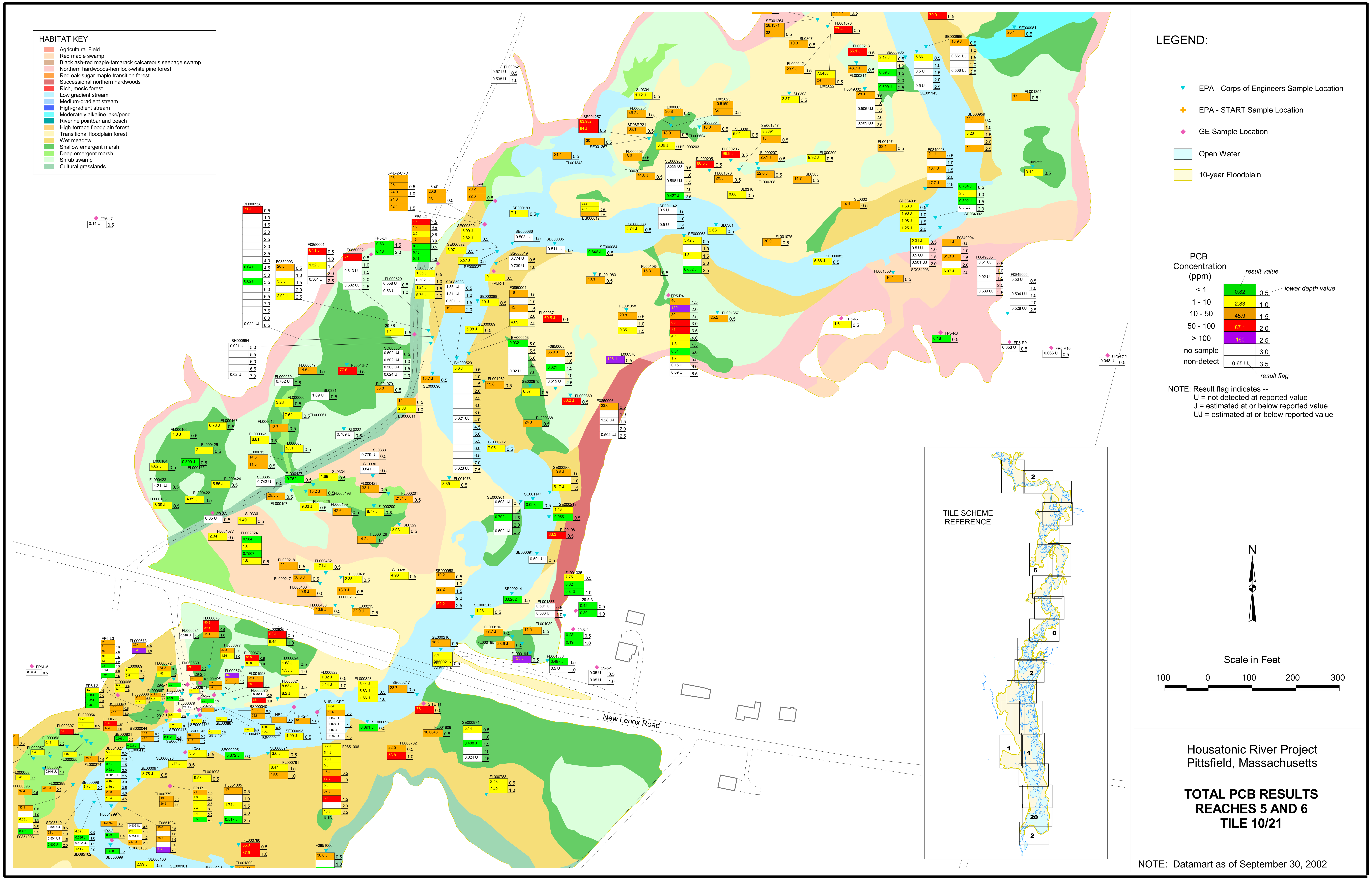
100 0 100 200 300

Housatonic River Project
Pittsfield, Massachusetts

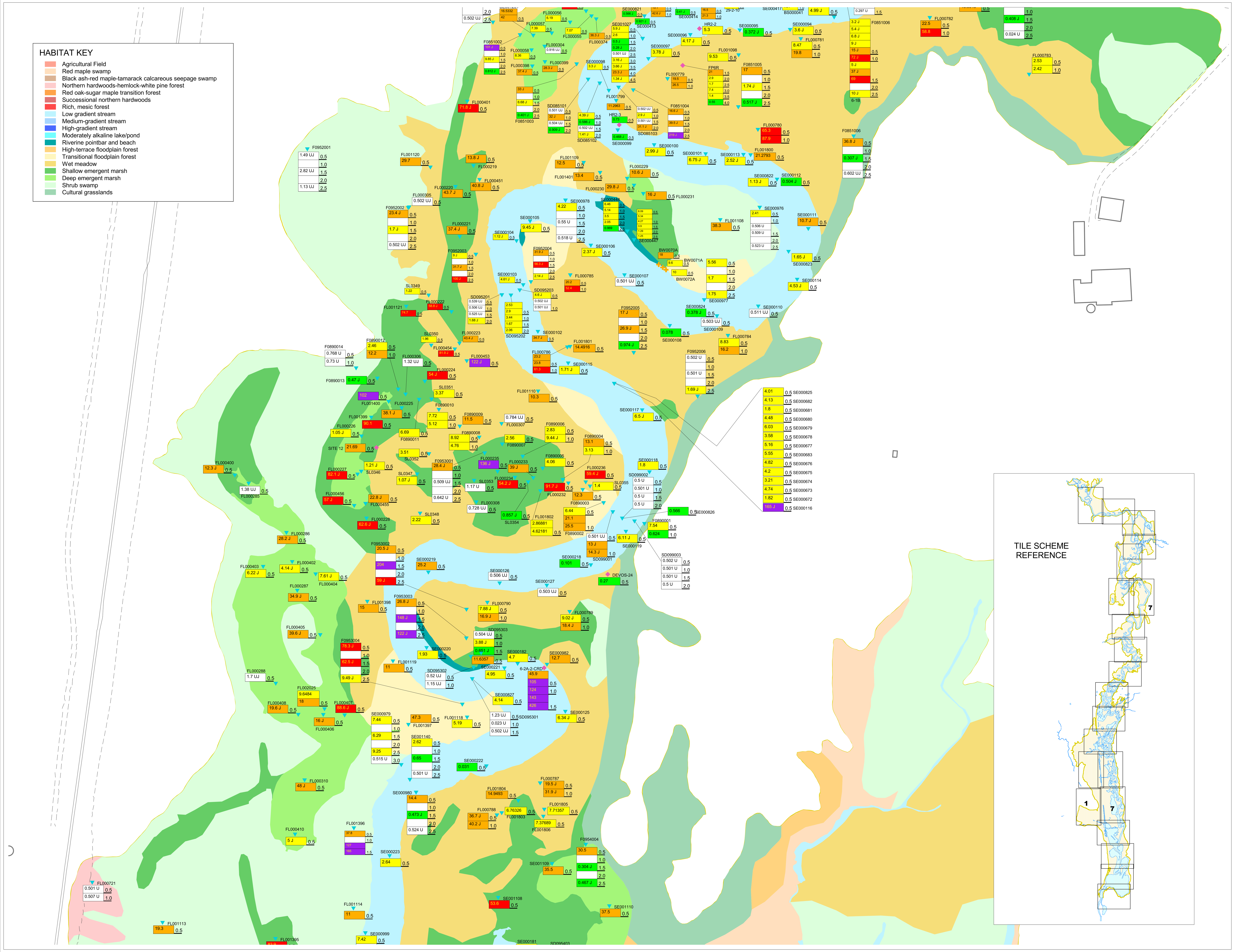
TOTAL PCB RESULTS
REACHES 5 AND 6
TILE 9/21

NOTE: Datamart as of September 30, 2002

| O:\gepitt\aprs\reach5&6_APRs\stacks2_sept02.apr | layout - tile 9 | o:\gepitt\epsfiles\landscape24x36\r\stacks9_sept02l.eps | 10:21 AM, 12/20/2002 |



HABITAT KEY
Agricultural Field
Red maple swamp
Black ash-red maple-tamarack calcareous seepage swamp
Northern hardwoods-hemlock-white pine forest
Red oak-sugar maple transition forest
Successional northern hardwoods
Rich, mesic forest
Low gradient stream
Medium-gradient stream
High-gradient stream
Moderately alkaline lake/pond
Riverine pointbar and beach
High-terrace floodplain forest
Transitional floodplain forest
Wet meadow
Shallow emergent marsh
Deep emergent marsh
Shrub swamp
Cultural grasslands
New Lenox Road
TILE SCHEME REFERENCE
LEGEND:
EPA - Corps of Engineers Sample Location
EPA - START Sample Location
GE Sample Location
Open Water
10-year Floodplain
PCB Concentration (ppm)
result value
lower depth value
< 1
1 - 10
10 - 50
50 - 100
> 100
no sample
non-detect
result flag
NOTE: Result flag indicates --
U = not detected at reported value
J = estimated at or below reported value
UJ = estimated at or below reported value
N
Scale in Feet
100 0 100 200 300
Housatonic River Project
Pittsfield, Massachusetts
TOTAL PCB RESULTS
REACHES 5 AND 6
TILE 10/21
NOTE: Datamart as of September 30, 2002



LEGEND:

- EPA - Corps of Engineers Sample Location
- EPA - START Sample Location
- GE Sample Location
- Open Water
- 10-year Floodplain

PCB Concentration (ppm)

PCB Concentration (ppm)	result value	lower depth value
< 1	0.82	0.5
1 - 10	2.83	1.0
10 - 50	45.9	1.5
50 - 100	87.1	2.0
> 100	160	2.5
no sample		3.0
non-detect	0.65 U	3.5

result flag

NOTE: Result flag indicates --
U = not detected at reported value
J = estimated at or below reported value
UJ = estimated at or below reported value

N

Scale in Feet

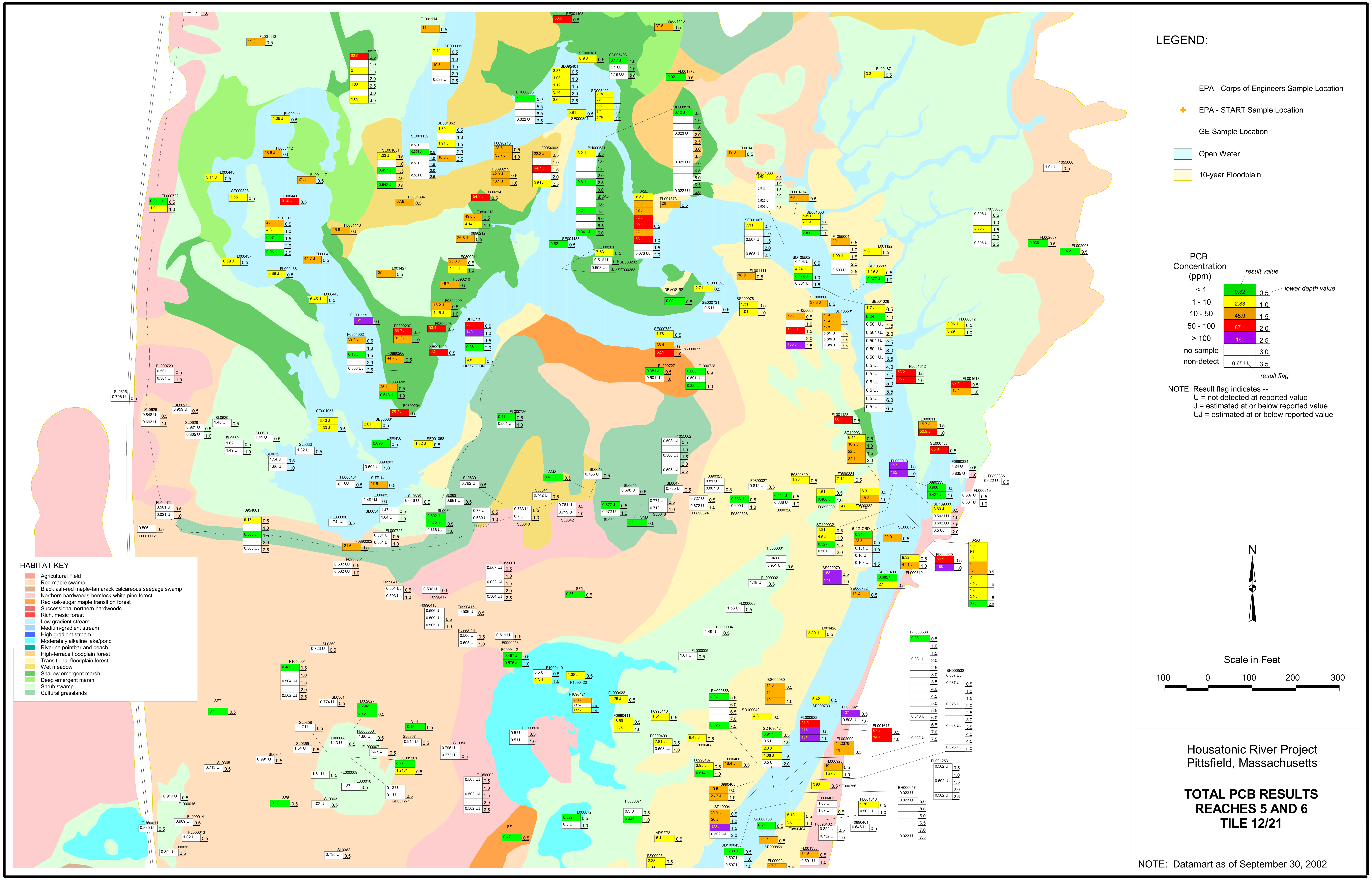
100 0 100 200 300

Housatonic River Project
Pittsfield, Massachusetts

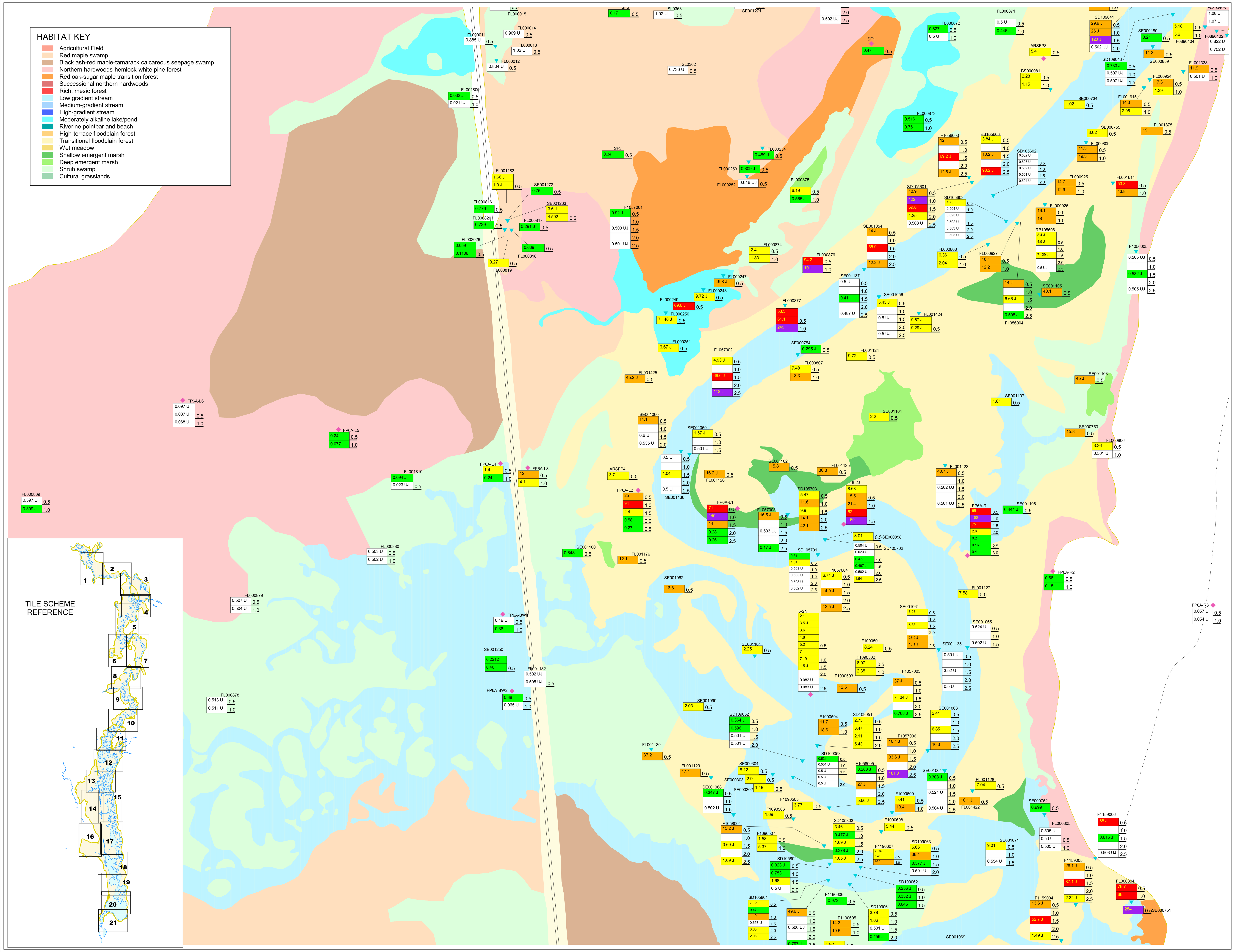
TOTAL PCB RESULTS
REACHES 5 AND 6
TILE 11/21

NOTE: Datamart as of September 30, 2002

| O:\gepittf\aprs\reach5&6_APRs\stacks3_sept02.apr | layout - tile 11 | o:\gepittf\epsfiles\landscape24x36\rir\stacks11_sept02.eps | 10:24 AM, 12/20/2002 |



| O:\gepittf\apr\reach5&6_APRs\stacks3_sept02.apr | layout - tile 12 | o:\gepittf\epsfiles\landscape24x36\r\stacks12_sept02l.eps | 10:25 AM, 12/20/2002 |



LEGEND:

EPA - Corps of Engineers Sample Location

EPA - START Sample Location

GE Sample Location

Open Water

10-year Floodplain

PCB Concentration (ppm)

	result value	lower depth value
< 1	0.82	0.5
1 - 10	2.83	1.0
10 - 50	45.9	1.5
50 - 100	87.1	2.0
> 100	160	2.5
no sample		3.0
non-detect	0.65 U (result flag)	3.5

NOTE: Result flag indicates --
U = not detected at reported value
J = estimated at or below reported value
UJ = estimated at or below reported value

N

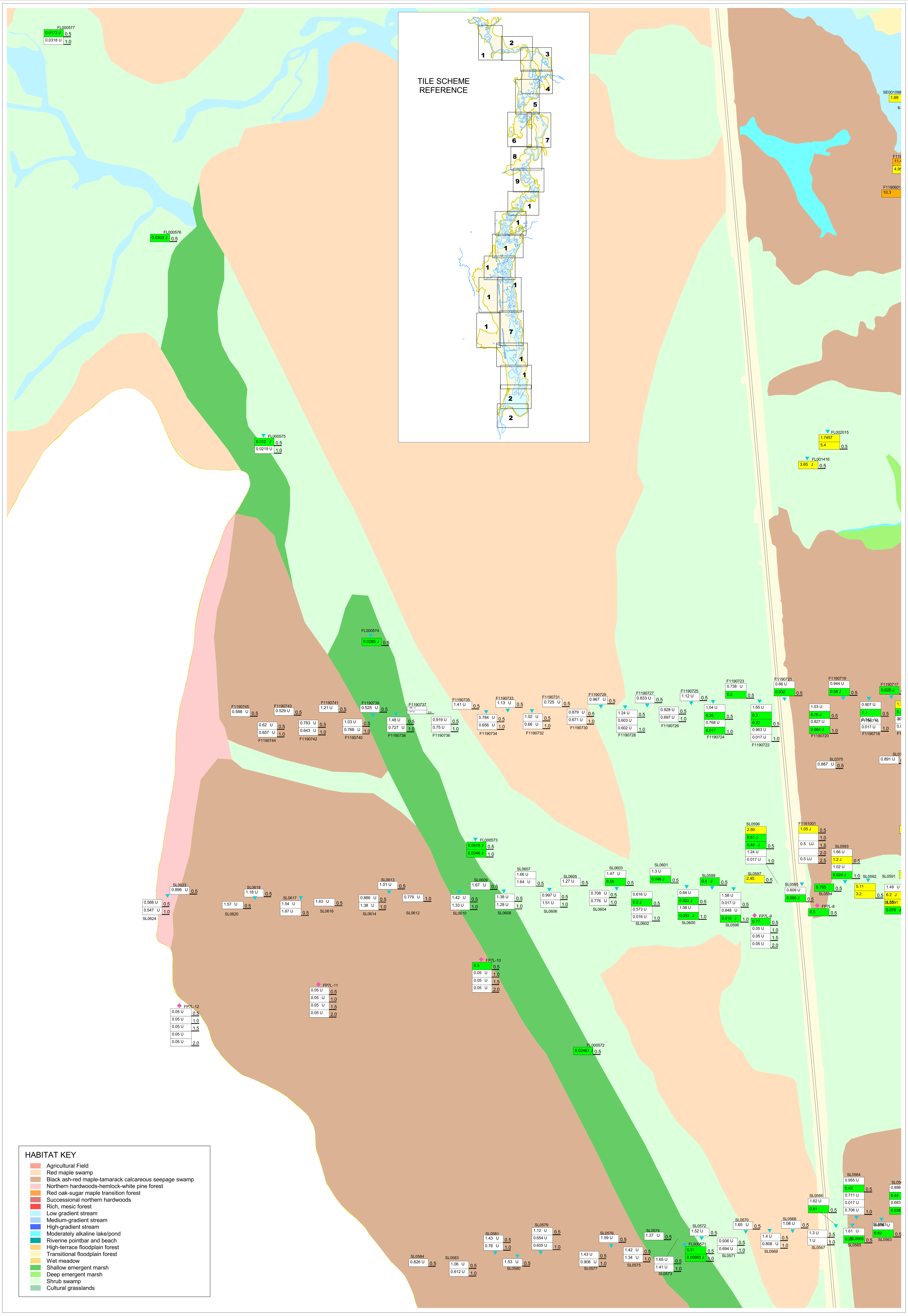
Scale in Feet

100 0 100 200 300

Housatonic River Project
Pittsfield, Massachusetts

TOTAL PCB RESULTS REACHES 5 AND 6 TILE 13/21

NOTE: Datamart as of September 30, 2002



LEGEND:

EPA - Corps of Engineers Sample Location
EPA - START Sample Location
GE Sample Location
Open Water
10-year Floodplain

PCB Concentration (ppm)

PCB Concentration (ppm)	result value	lower depth value
< 1	0.82	0.5
1 - 10	2.83	1.0
10 - 50	45.9	1.5
50 - 100	87.1	2.0
> 100	160	2.5
no sample		3.0
non-detect	0.65 U (result flag)	3.5

NOTE:
Result flag indicates --
U = not detected at reported value
J = estimated detected value
UJ = estimated non-detected value

Scale in Feet

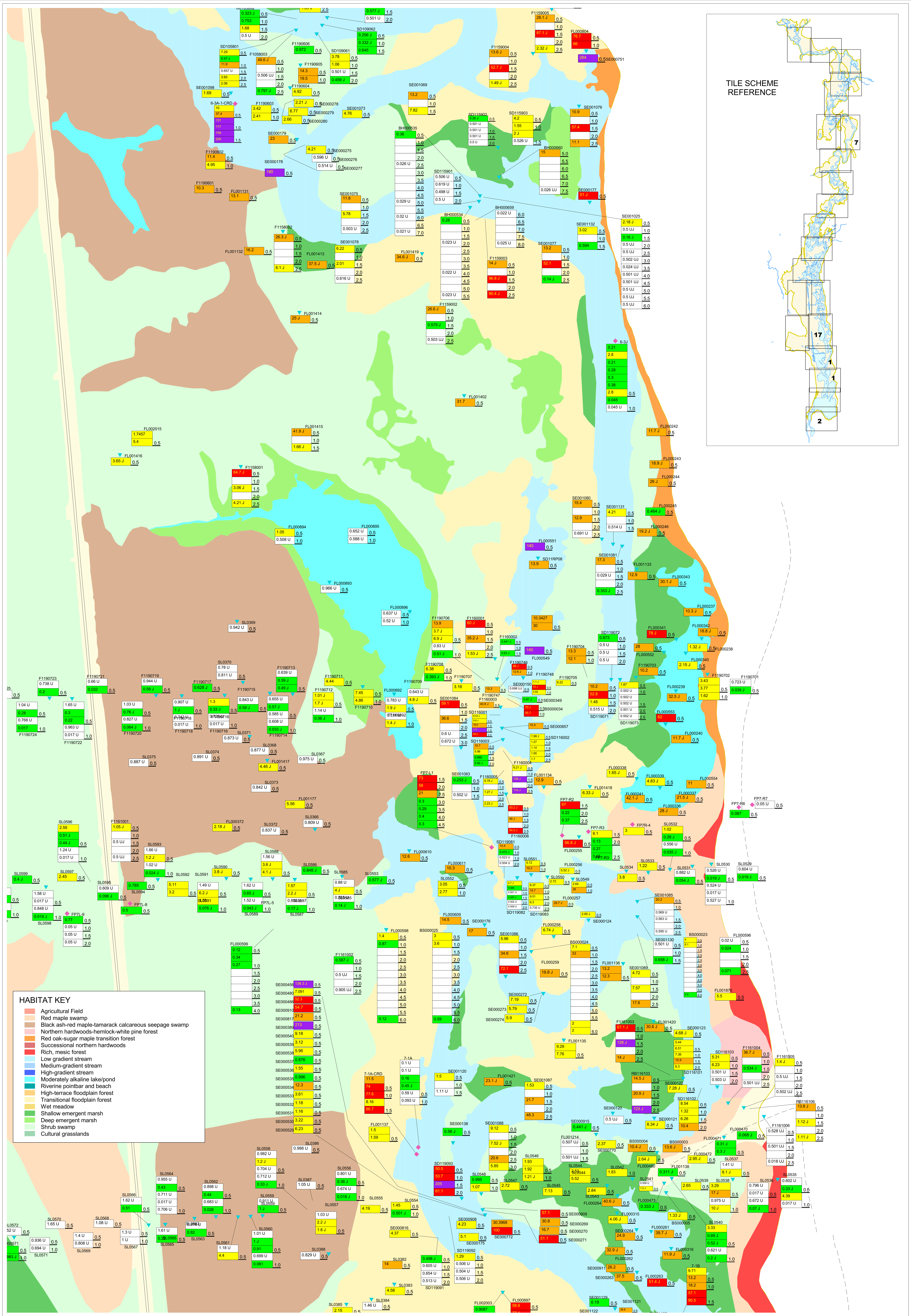
100 0 100 200 300

Housatonic River Project
Pittsfield, Massachusetts

TOTAL PCB RESULTS REACHES 5 AND 6 TILE 14/21

NOTE: Datamart as of September 30, 2002

| O:\gepittsf\aprs\reach5&6_APRs\stacks4_sept02.apr | layout - tile 14 | o:\gepitt\epsfiles\portrait24x36\n\stacks14_sept02.eps | 10:28 AM, 12/20/2002 |



LEGEND:

EPA - Corps of Engineers Sample Location

EPA - START Sample Location

GE Sample Location

Open Water

10-year Floodplain

PCB Concentration (ppm)

	result value	lower depth value
< 1	0.82	0.5
1 - 10	2.83	1.0
10 - 50	45.9	1.5
50 - 100	87.1	2.0
> 100	160	2.5
no sample		3.0
non-detect	0.65 U	3.5

result flag

NOTE:
Result flag indicates --
U = not detected at reported value
J = estimated detected value
UJ = estimated non-detected value

Scale in Feet

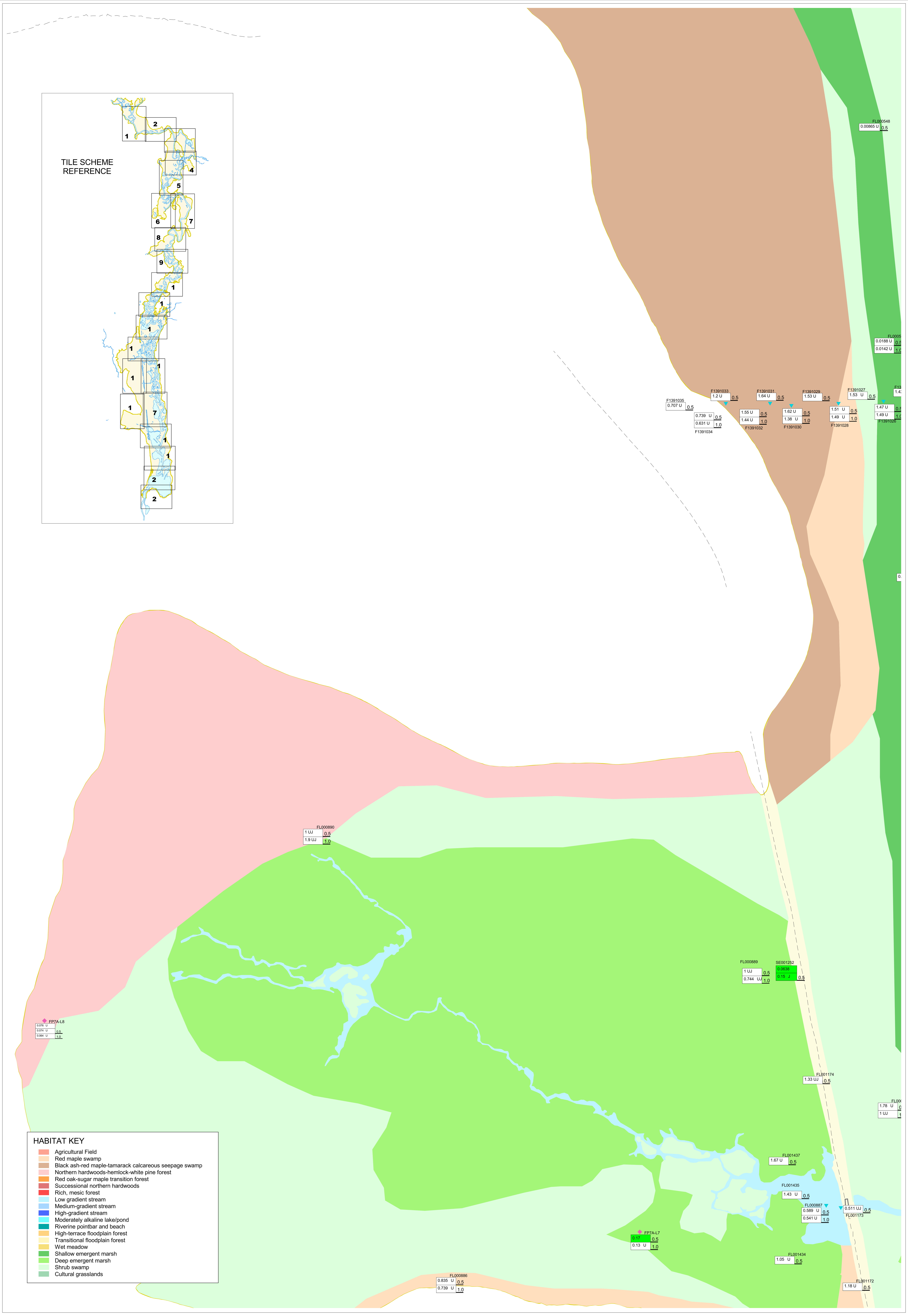
100 0 100 200 300

Housatonic River Project
Pittsfield, Massachusetts

TOTAL PCB RESULTS
REACHES 5 AND 6
TILE 15/21

NOTE: Datamart as of September 30, 2002

| O:\gepitt\aprs\reach5&6_APRs\stacks4_sept02.apr | layout - tile 15 | o:\gepitt\epsfiles\portrait24x36\r\stacks15_sept02l.eps | 10:29 AM, 12/20/2002 |



LEGEND:

- EPA - Corps of Engineers Sample Location
- EPA - START Sample Location
- GE Sample Location
- Open Water
- 10-year Floodplain

PCB Concentration (ppm)

< 1	0.82	0.5
1 - 10	2.83	1.0
10 - 50	45.9	1.5
50 - 100	87.1	2.0
> 100	160	2.5
no sample		3.0
non-detect	0.65 U	3.5

result value — *lower depth value* — *result flag*

NOTE:
Result flag indicates --
U = not detected at reported value
J = estimated detected value
UJ = estimated non-detected value

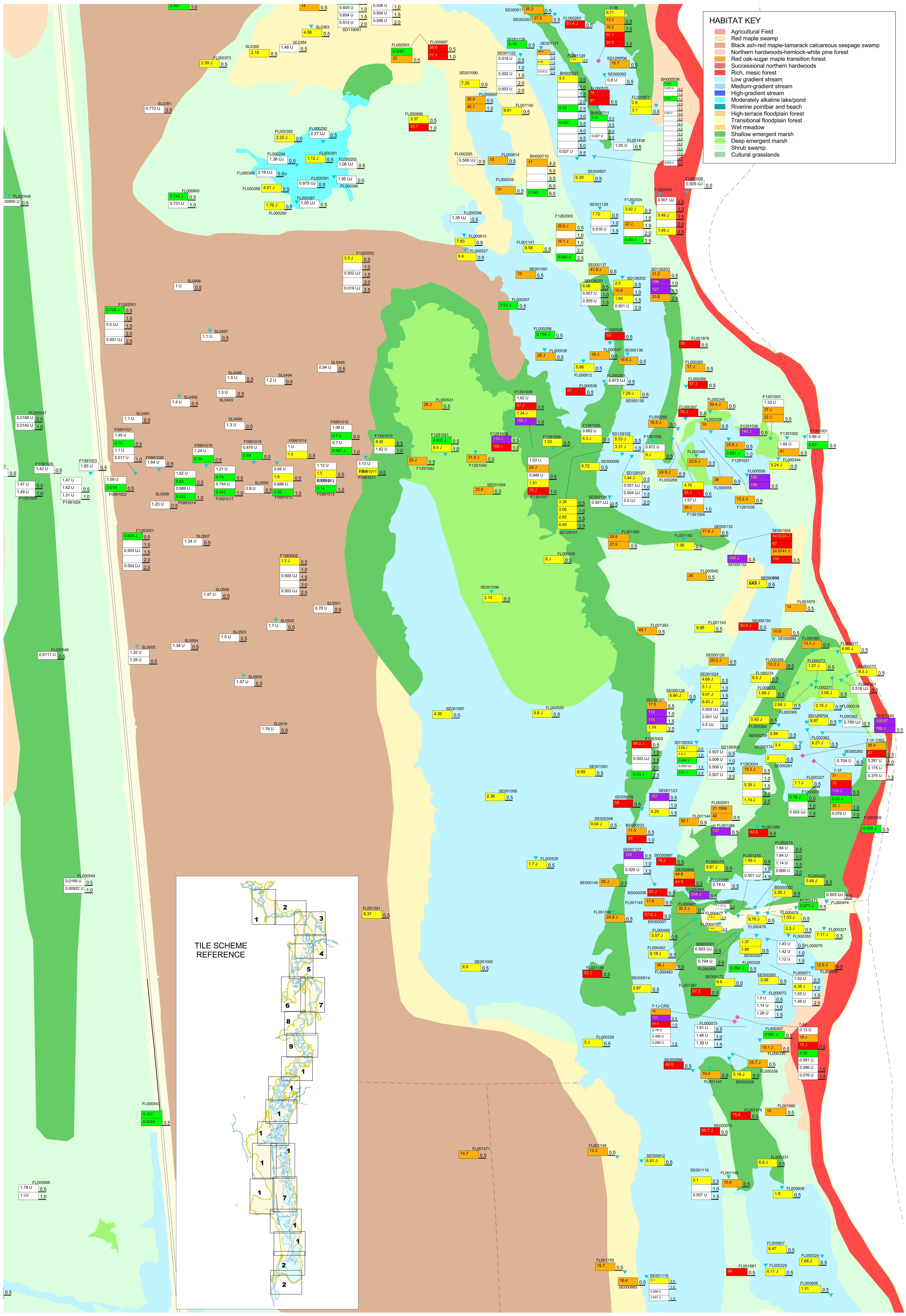
Scale in Feet

100 0 100 200 300

Housatonic River Project
Pittsfield, Massachusetts

TOTAL PCB RESULTS
REACHES 5 AND 6
TILE 16/21

NOTE: Datamart as of September 30, 2002



LEGEND:

- EPA - Corps of Engineers Sample Location
- EPA - START Sample Location
- GE Sample Location
- Open Water
- 10-year Floodplain

PCB Concentration (ppm)

PCB Concentration (ppm)	result value	result flag
< 1	0.82	0.5
1 - 10	2.83	1.0
10 - 50	45.9	1.5
50 - 100	87.1	2.0
> 100	160	2.5
no sample		3.0
non-detect	0.65 U	3.5

lower depth value

NOTE:
Result flag indicates --
U = not detected at reported value
J = estimated detected value
UJ = estimated non-detected value

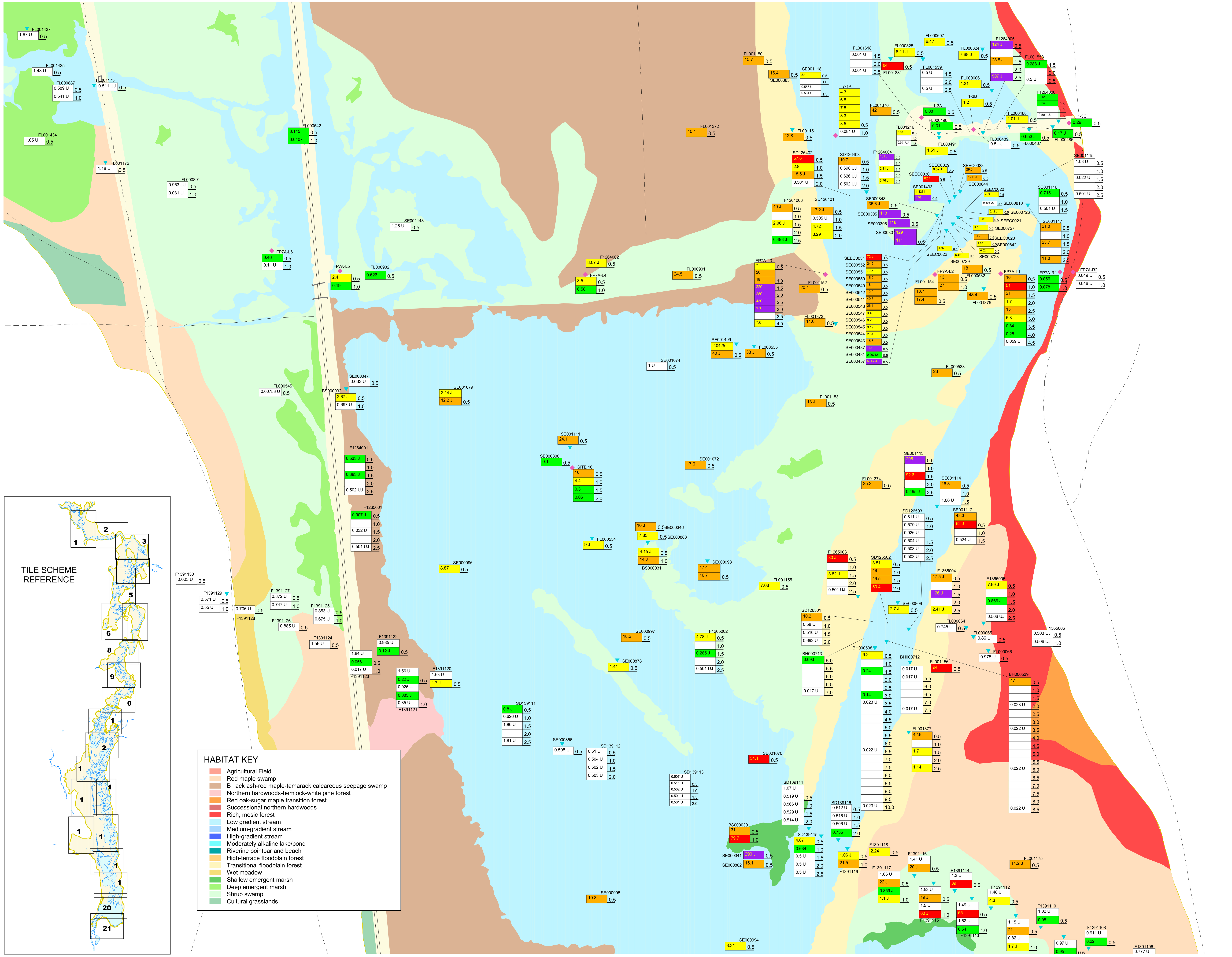
Scale in Feet

100 0 100 200 300

Housatonic River Project
Pittsfield, Massachusetts

TOTAL PCB RESULTS
REACHES 5 AND 6
TILE 17/21

NOTE: Datamart as of September 30, 2002



LEGEND:

EPA - Corps of Engineers Sample Location

EPA - START Sample Location

GE Sample Location

Open Water

10-year Floodplain

PCB Concentration (ppm)

PCB Concentration (ppm)	result value	lower depth value
< 1	0.82	0.5
1 - 10	2.83	1.0
10 - 50	45.9	1.5
50 - 100	87.1	2.0
> 100	160	2.5
no sample		3.0
non-detect	0.65 U (result flag)	3.5

NOTE: Result flag indicates --
U = not detected at reported value
J = estimated at or below reported value
UJ = estimated at or below reported value

N

Scale in Feet

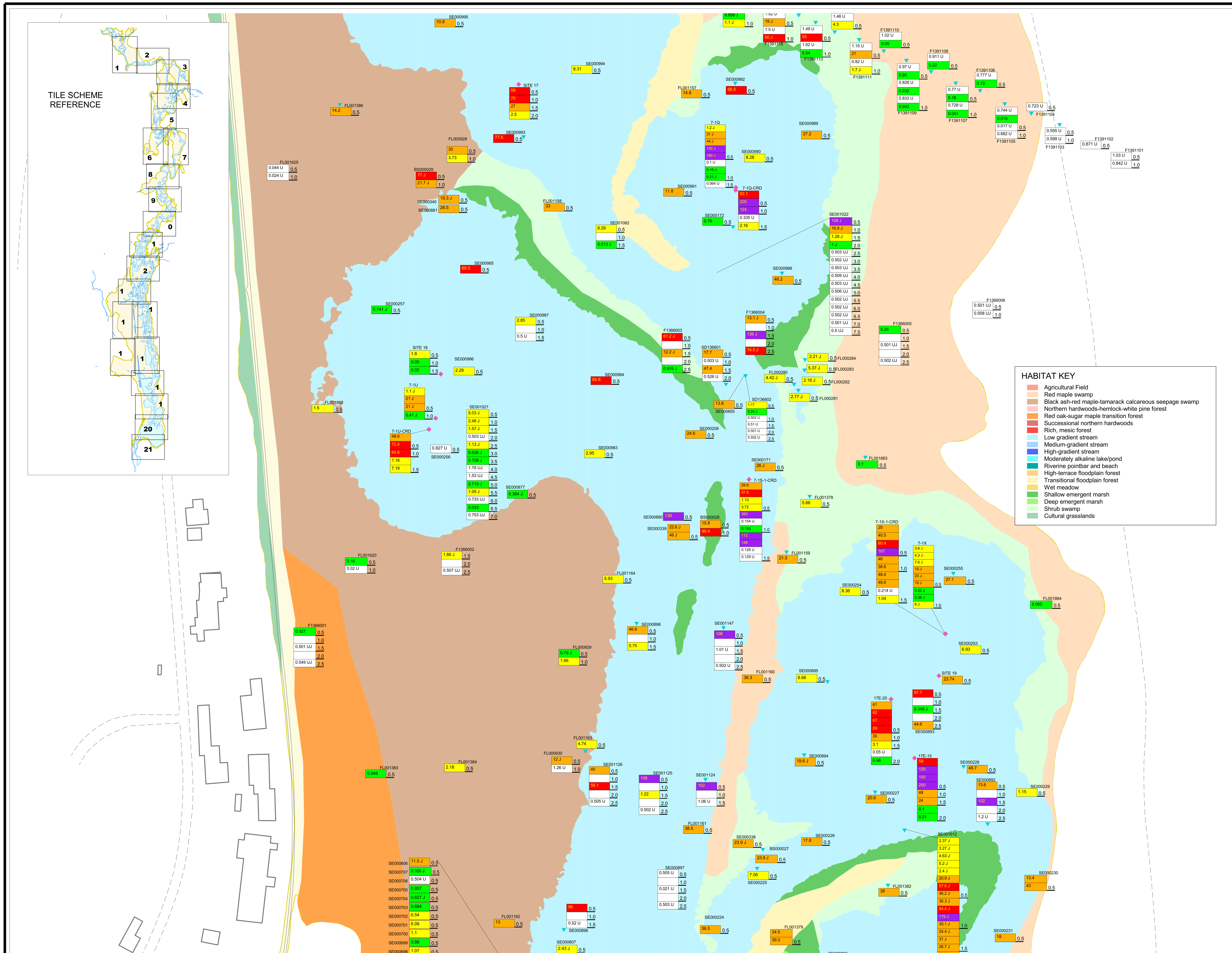
100 0 100 200 300

Housatonic River Project
Pittsfield, Massachusetts

TOTAL PCB RESULTS
REACHES 5 AND 6
TILE 18/21

NOTE: Datamart as of September 30, 2002

| O:\gepitt\aprs\reach5&6_APRs\stacks5_sept02.apr | layout - tile 18 | o:\gepitt\epsfiles\landscape24x36\r\stacks18_sept02l.eps | 10:33 AM, 12/20/2002 |



LEGEND:

- EPA - Corps of Engineers Sample Location
- EPA - START Sample Location
- GE Sample Location
- Open Water
- 10-year Floodplain

PCB Concentration (ppm)

PCB Concentration (ppm)	result value	lower depth value
< 1	0.82	0.5
1 - 10	2.83	1.0
10 - 50	45.9	1.5
50 - 100	87.1	2.0
> 100	160	2.5
no sample		3.0
non-detect	0.65 U	3.5

result flag

NOTE: Result flag indicates --
U = not detected at reported value
J = estimated at or below reported value
UJ = estimated at or below reported value

N

Scale in Feet

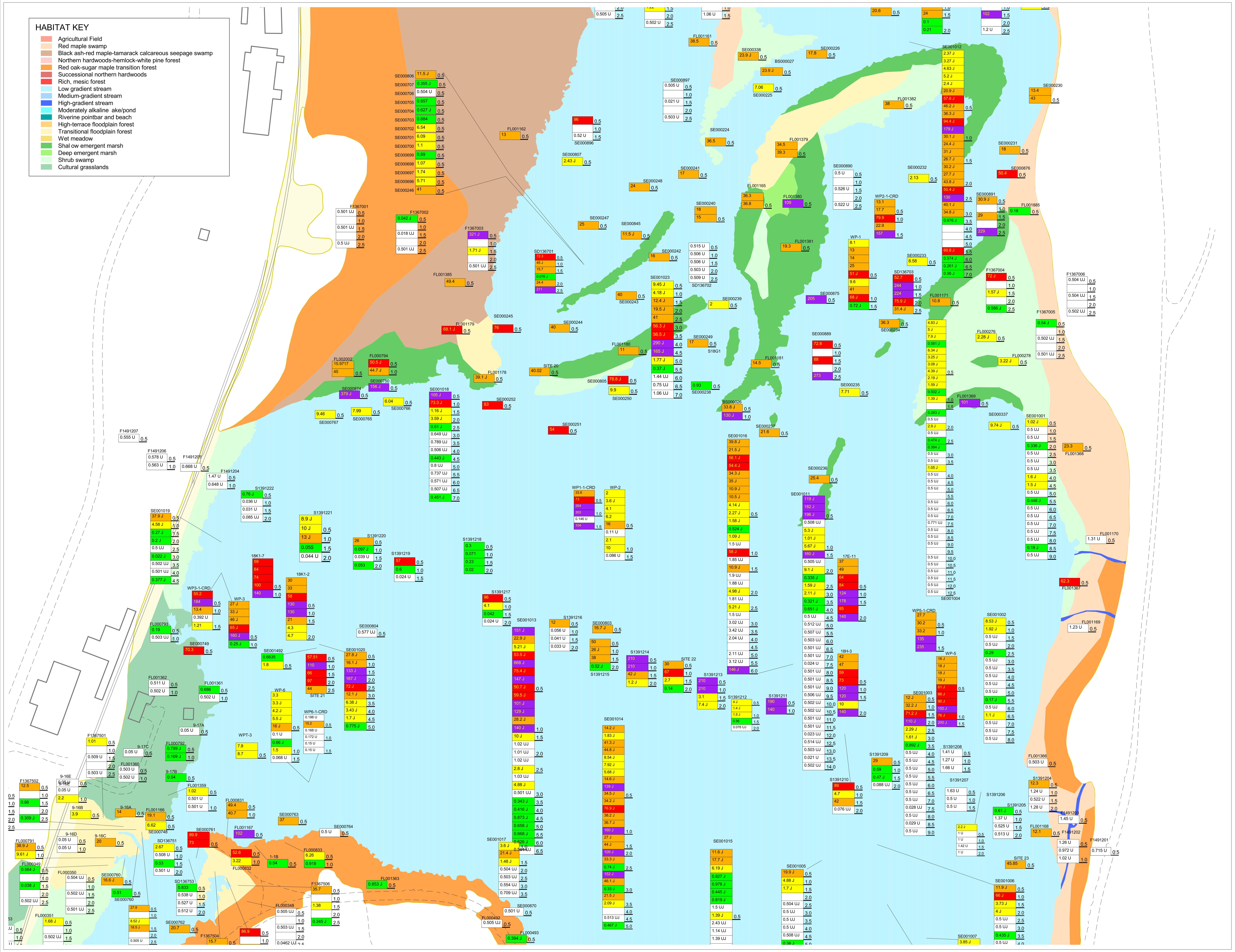
100 0 100 200 300

Housatonic River Project
Pittsfield, Massachusetts

TOTAL PCB RESULTS
REACHES 5 AND 6
TILE 19/21

NOTE: Datamart as of September 30, 2002

| O:\gepitt\apns\reach5&6_APRs\stacks5_sept02.apr | layout - tile 19 | o:\gepitt\epsfiles\landscape24x36\r\stacks19_sept02.eps | 10:34 AM, 12/20/2002 |



LEGEND:

EPA - Corps of Engineers Sample Location

EPA - START Sample Location

GE Sample Location

Open Water

10-year Floodplain

PCB Concentration (ppm)

result value / lower depth value / result flag

PCB Concentration (ppm)	result value	lower depth value
< 1	0.82	0.5
1 - 10	2.83	1.0
10 - 50	45.9	1.5
50 - 100	87.1	2.0
> 100	160	2.5
no sample		3.0
non-detect	0.65 U	3.5

NOTE: Result flag indicates --
U = not detected at reported value
J = estimated at or below reported value
UJ = estimated at or below reported value

N

Scale in Feet

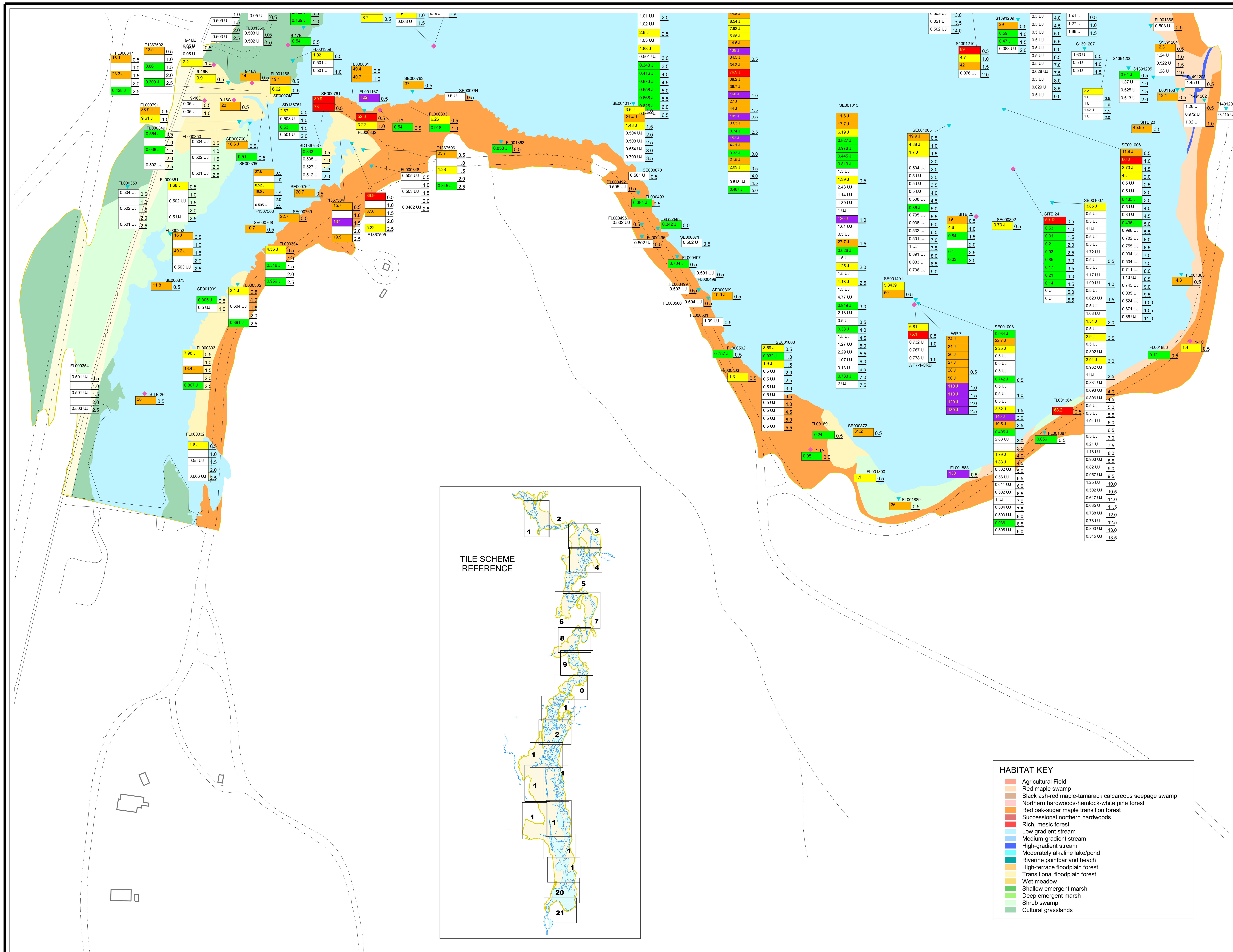
100 0 100 200 300

Housatonic River Project
Pittsfield, Massachusetts

TOTAL PCB RESULTS
REACHES 5 AND 6
TILE 20/21

NOTE: Datamart as of September 30, 2002

| O:\gepitt\apr\reach5&6_APR\stacks5_sept02.apr | layout - tile 20 | c:\gepitt\epsfiles\landscape24x36\r\stacks20_sept02l.eps | 10:35 AM, 12/20/2002 |



LEGEND:

- EPA - Corps of Engineers Sample Location
- EPA - START Sample Location
- GE Sample Location
- Open Water
- 10-year Floodplain

PCB Concentration (ppm)

	result value	lower depth value
< 1	0.82	0.5
1 - 10	2.83	1.0
10 - 50	45.9	1.5
50 - 100	87.1	2.0
> 100	160	2.5
no sample		3.0
non-detect	0.65 U (result flag)	3.5

NOTE: Result flag indicates --
U = not detected at reported value
J = estimated at or below reported value
UJ = estimated at or below reported value

N

Scale in Feet

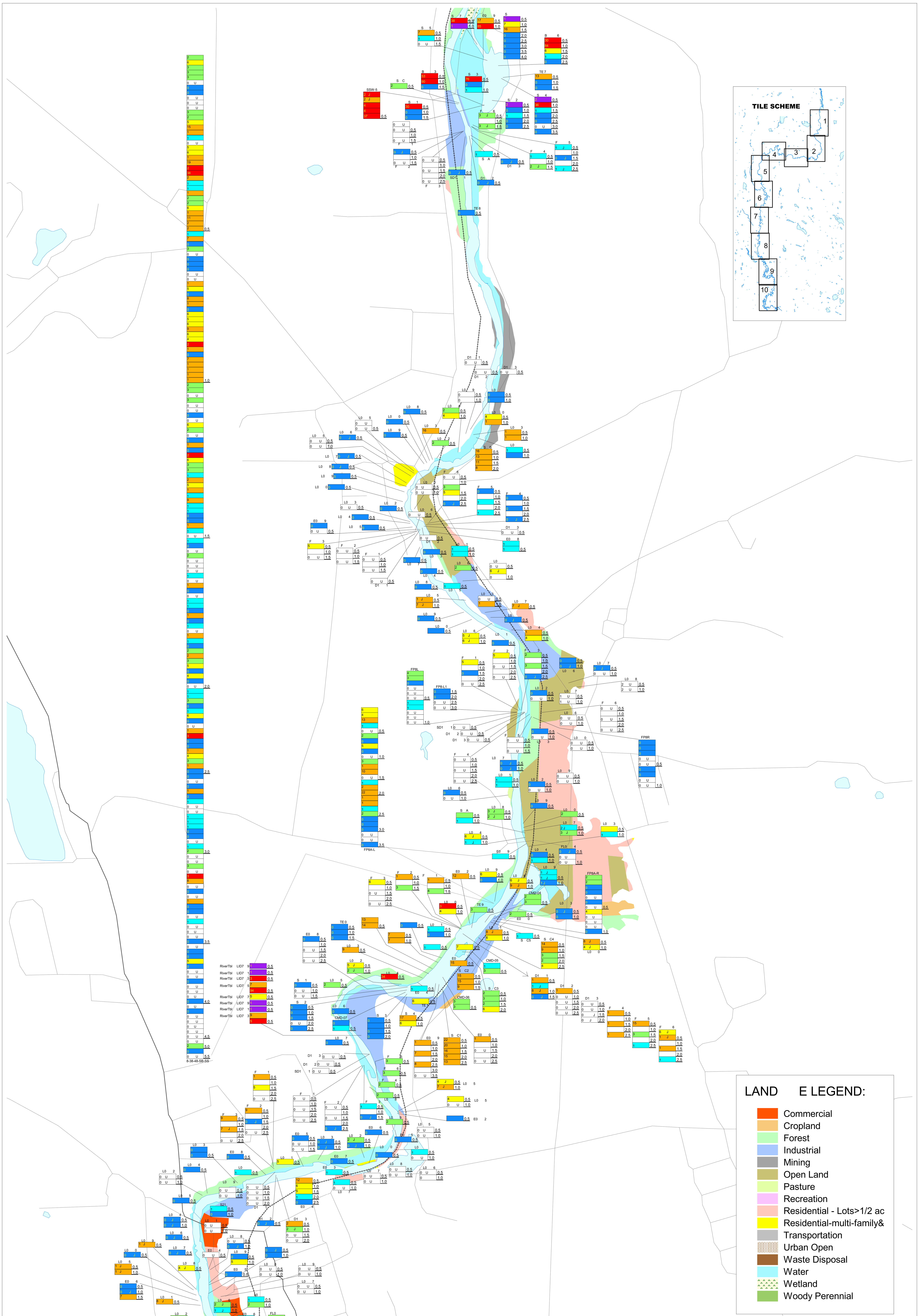
100 0 100 200 300

Housatonic River Project
Pittsfield, Massachusetts

TOTAL PCB RESULTS
REACHES 5 AND 6
TILE 21/21

NOTE: Datamart as of September 30, 2002

| O:\gepitt\apr\reach5&6_APRs\stacks5_sept02.apr | layout - tile 21 | o:\gepitt\epsfiles\landscape24x36\r\stacks21_sept02l.eps | 10:36 AM, 12/20/2002 |



LEGEND:

GE

EPA - Corps of Engineers

Roads

Railroads

River

PCB Concentration (ppm)

result value

lower depth value

result flag

PCB Concentration (ppm)	result value	lower depth value
< 1	0 1	0.5
1 - 2	1 5	1.0
2 - 4	2 9	1.5
4 - 7	6 9	2.0
-5	2 7	2.5
- 0	6 9	3.0
> 0	185	3.5
no sample		4.0
non-detect	0 U	4.5

NOTE:
Result flag indicates --
U = not detected at reported value
J = estimated detected value
U = estimated non-detected value

N

Scale in eet

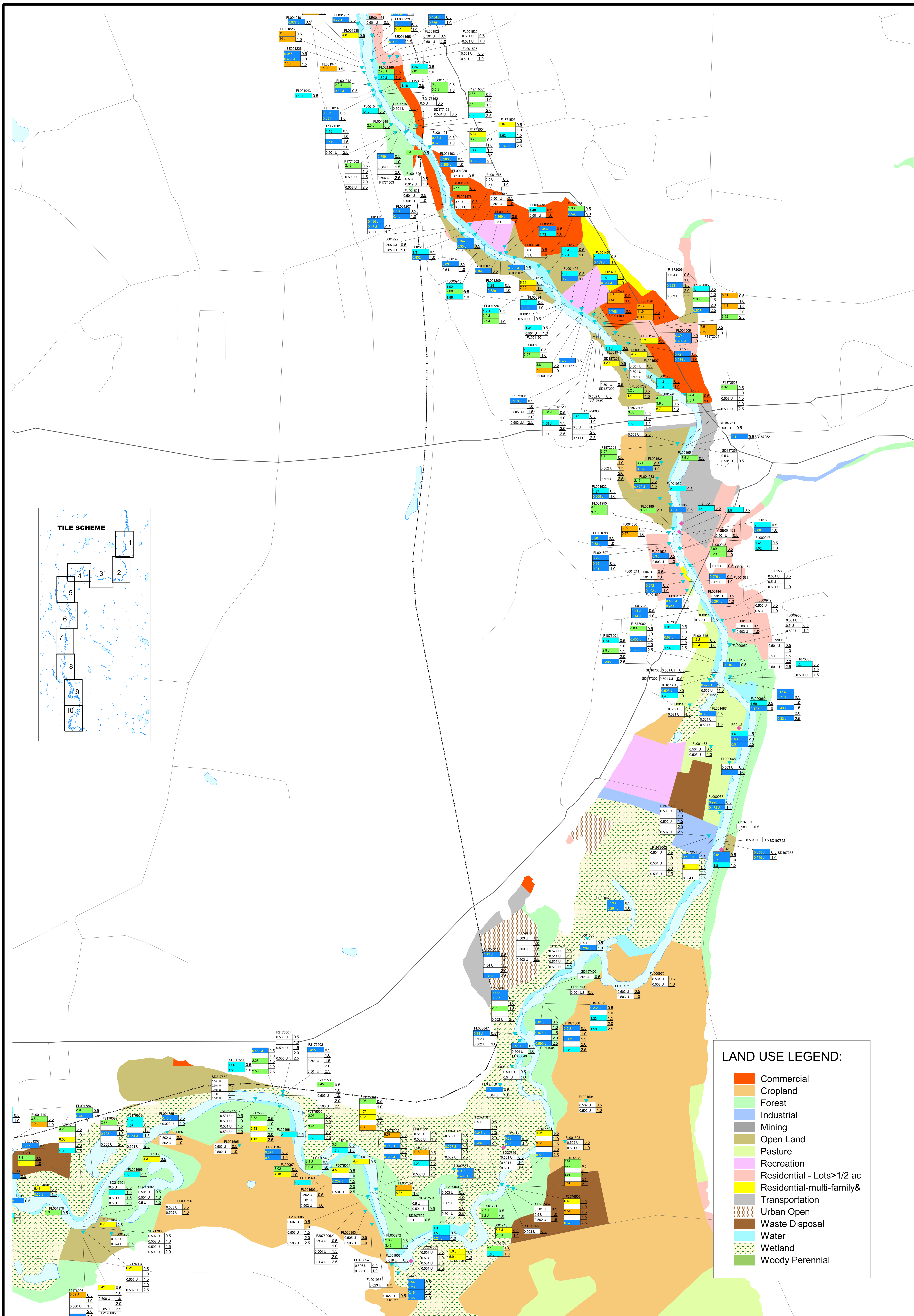
500 0 500 1 0 1 0

Housatonic River Project
Pittsfield, Massachusetts

TOTAL PCB RESULTS
REACH 7, AND 8
TILE 1/10

NOTE: Datamart as of November , 2

| O:\gepitt\aprs\r789_stacks_nov02.apr | layout - tile1 | o:\gepitt\epsfiles\plots\in\r789stacks1_nov02p.eps | 3:40 PM, 12/4/2002 |



LEGEND:

GE

EPA - Corps of Engineers

Roads

Railroads

River

PCB Concentration (ppm)

result value

lower depth value

< 1 0.51 0.5
1 - 2 1.45 1.0
2 - 4 2.89 1.5
4 - 7 6.09 2.0
7 - 25 23.7 2.5
25 - 100 68.9 3.0
> 100 186 3.5
no sample 4.0
non-detect 0.65 U 4.5

result flag

NOTE:
Result flag indicates --

U = not detected at reported value

J = estimated detected value

UJ = estimated non-detected value

Scale in Feet

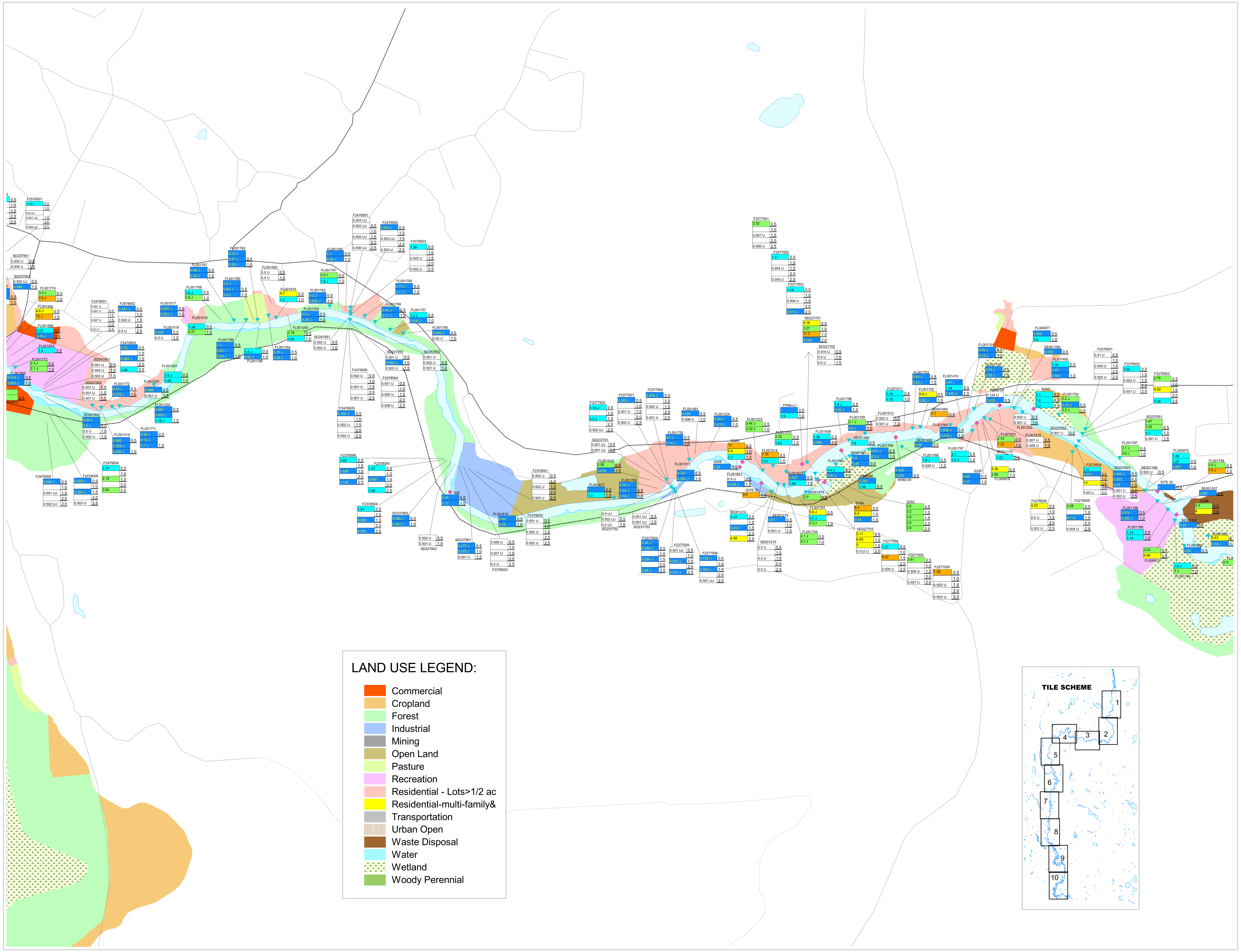
500 0 500 1000 1500

Housatonic River Project
Pittsfield, Massachusetts

TOTAL PCB RESULTS
REACH 7, AND 8
TILE 2/10

NOTE: Datamart as of November 13, 2002

| O:\gepittt\aprs\r789_stacks_nov02.apr | layout - tile2 | o:\gepittt\epsfiles\plots\in\r789stacks2_nov02p.eps | 3:39 PM, 12/4/2002 |



LEGEND:

- GE
- EPA - Corps of Engineers
- Roads
- Railroads
- River

PCB Concentration (ppm)

PCB Concentration (ppm)	result value	depth
< 1	0.51	0.5
1 - 2	1.45	1.0
2 - 4	2.89	1.5
4 - 7	6.09	2.0
7 - 25	23.7	2.5
25 - 100	68.9	3.0
> 100	185	3.5
no sample		4.0
non-detect	0.65 U	4.5

result value; lower depth value; result flag

NOTE: Result flag indicates --
U = not detected at reported value
J = estimated at or below reported value
UJ = estimated at or below reported value

N

Scale in Feet

500 0 500 1000 1500

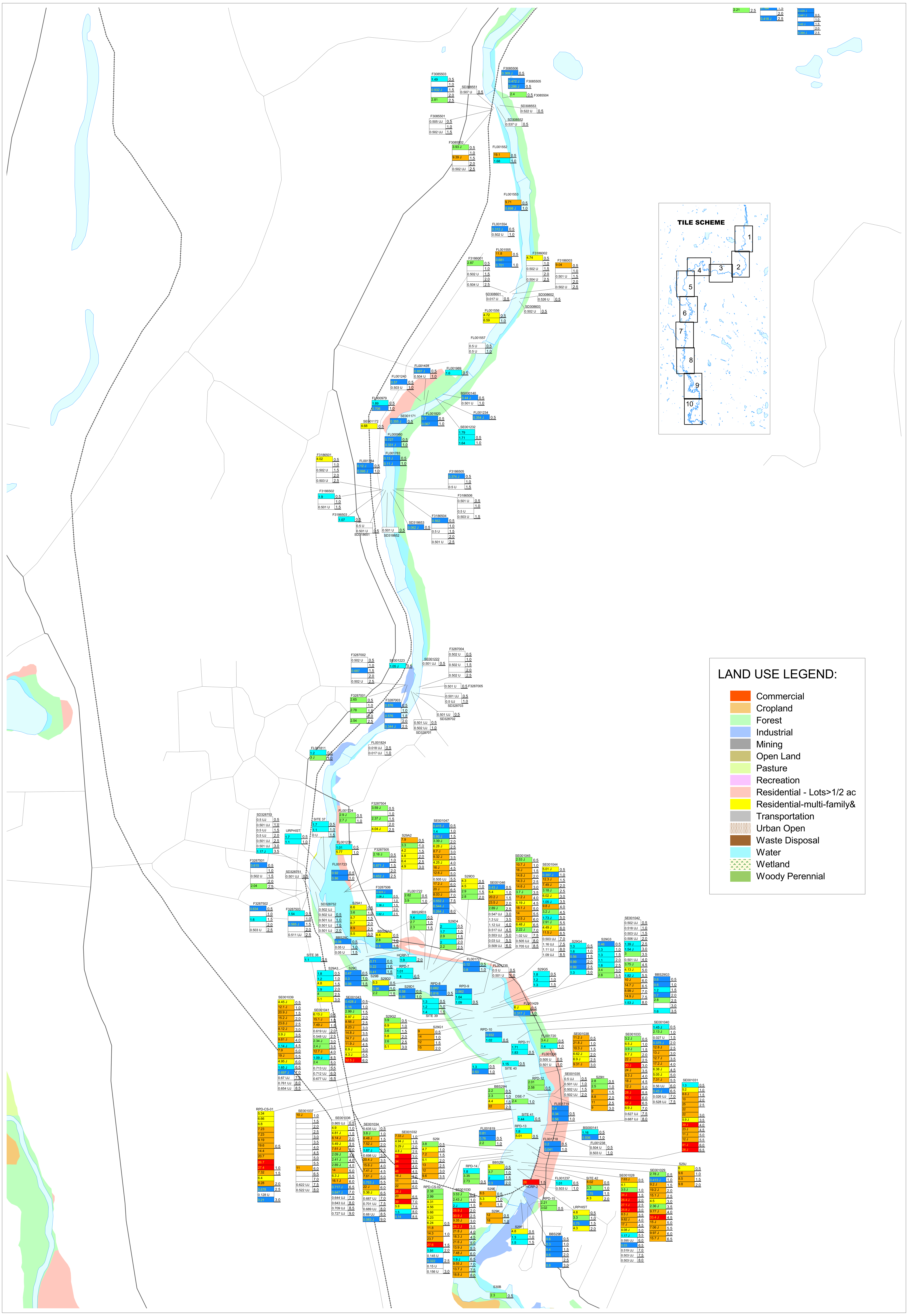
Housatonic River Project
Pittsfield, Massachusetts

TOTAL PCB RESULTS
REACH 7, AND 8
TILE 3/10

NOTE: Datamart as of November 13, 2002

| O:\gepitt\aprs\r789_stacks_nov02.apr | layout - tile3 | o:\gepitt\epsfiles\plots\in\r789stacks3_nov02l.eps | 3:38 PM, 12/4/2002 |

| O:\gepitt\aprs\r789_stacks_nov02.apr | layout - tile4 | o:\gepitt\epsfiles\plots\in\r789stacks4_nov02l.eps | 3:41 PM, 12/4/2002 |



LEGEND:

GE

EPA - Corps of Engineers

Roads

Railroads

River

PCB Concentration (ppm)

< 1	0.51	0.5
1 - 2	1.45	1.0
2 - 4	2.89	1.5
4 - 7	6.09	2.0
7 - 25	23.7	2.5
25 - 100	68.9	3.0
> 100	186	3.5
no sample		4.0
non-detect	0.65 U	4.5

result value

lower depth value

result flag

NOTE:
Result flag indicates --

U = not detected at reported value

J = estimated detected value

UJ = estimated non-detected value

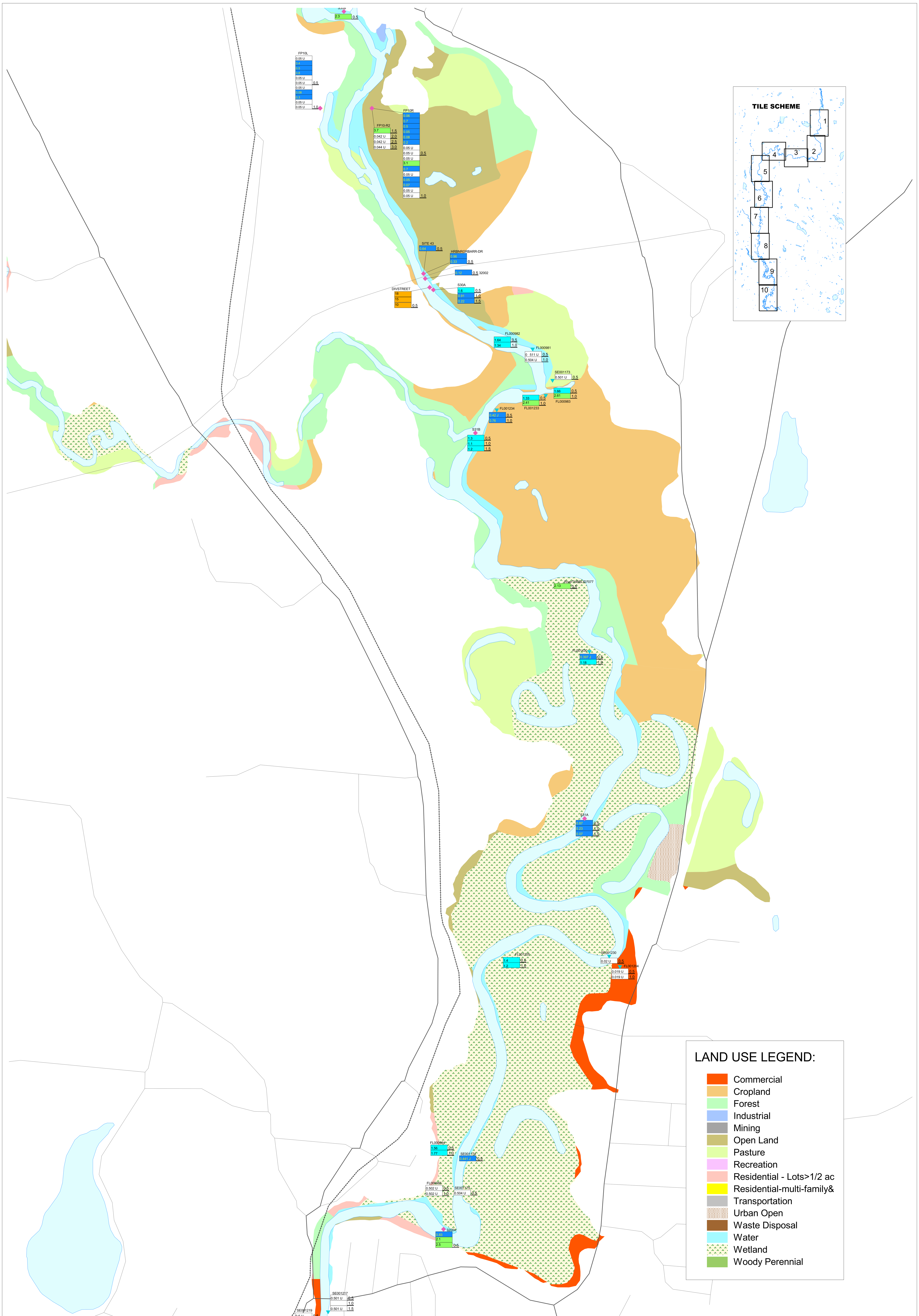
Scale in Feet

500 0 500 1000 1500

Housatonic River Project
Pittsfield, Massachusetts

TOTAL PCB RESULTS
REACH 7, AND 8
TILE 5/10

NOTE: Datamart as of November 13, 2002



LEGEND:

- GE
- EPA - Corps of Engineers
- Roads
- Railroads
- River

PCB Concentration (ppm)

	result value	lower depth value
< 1	0.51	0.5
1 - 2	1.45	1.0
2 - 4	2.89	1.5
4 - 7	6.09	2.0
7 - 25	23.7	2.5
25 - 100	68.9	3.0
> 100	186	3.5
no sample		4.0
non-detect	0.65 U	4.5

result flag

NOTE:
Result flag indicates --
U = not detected at reported value
J = estimated detected value
UJ = estimated non-detected value

Scale in Feet

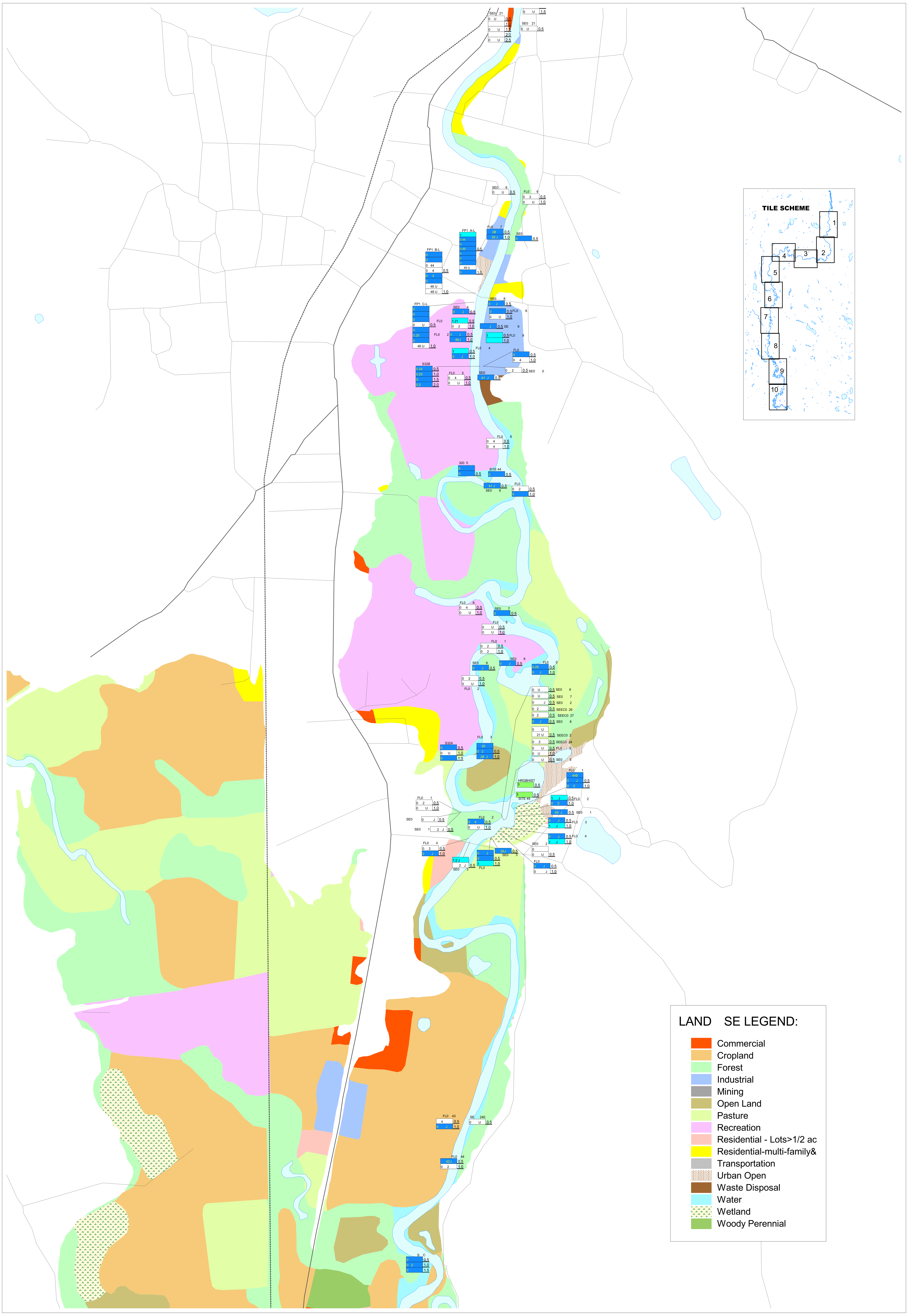
500 0 500 1000 1500

Housatonic River Project
Pittsfield, Massachusetts

TOTAL PCB RESULTS
REACH 7, 8 AND 9
TILE 6/10

NOTE: Datamart as of November 13, 2002

| O:\gepittl\aprs\r789_stacks_nov02.apr | layout - tile6 | o:\gepittl\epsfiles\plots\in\r789stacks6_nov02p.eps | 3:44 PM, 12/4/2002 |



LEGEND:

GE

EPA - Corps of Engineers

Roads

Railroads

River

PCB Concentration (ppm)

result value
lower depth value
result flag

< 1
-2
2 - 4
4 - 7
-25
25 - 0
> 0
no sample
non-detect

NOTE:
Result flag indicates --
U = not detected at reported value
J = estimated detected value
U = estimated non-detected value

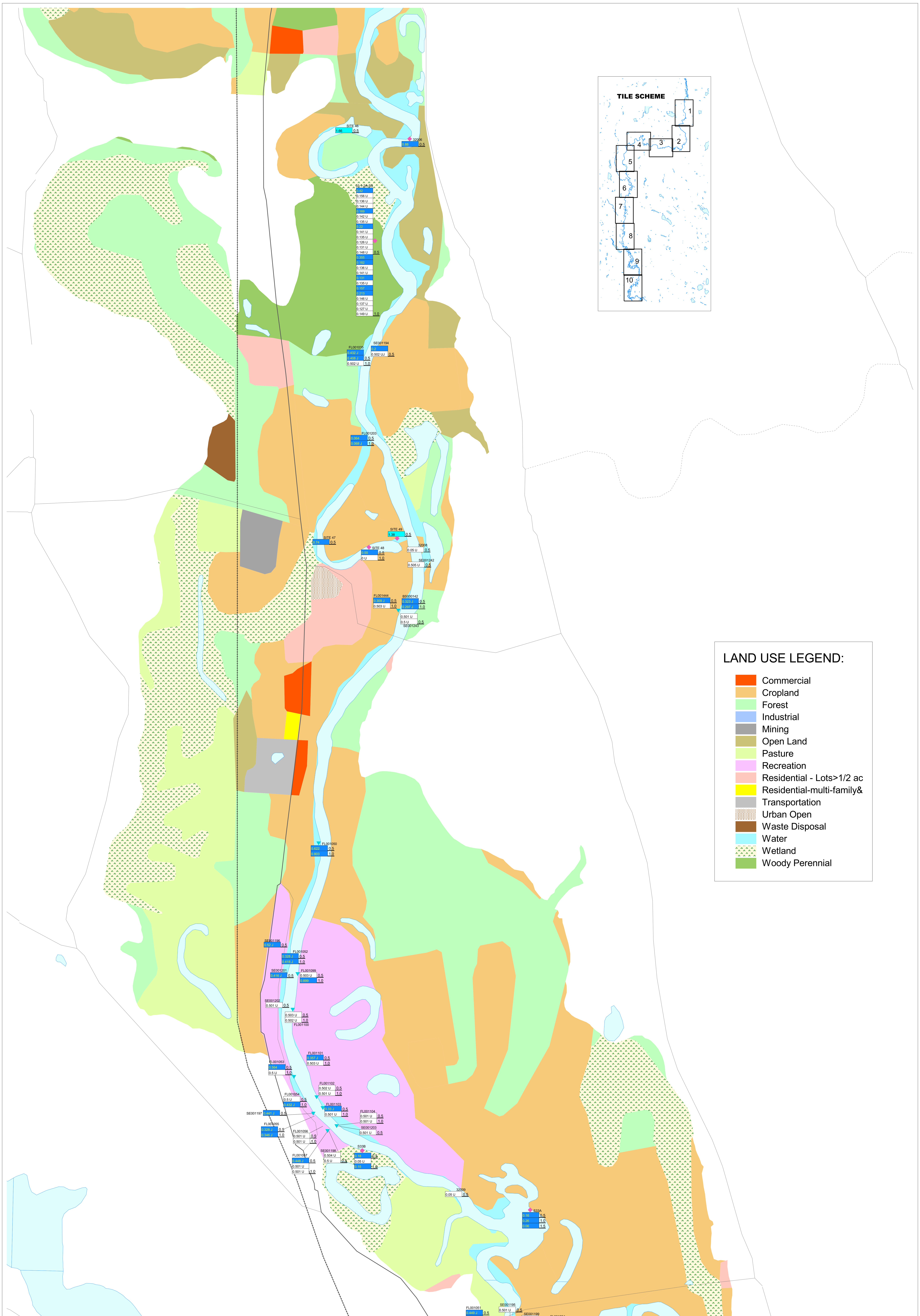
Scale in Feet

500 0 500 1 0 1 0

Housatonic River Project
Pittsfield, Massachusetts

TOTAL PCB RESULTS
REACH 7, 8 AND 9
TILE 7/10

NOTE: Datamart as of November , 20 2



LEGEND:

GE

EPA - Corps of Engineers

Roads

Railroads

River

PCB Concentration (ppm)

< 1	0.51	0.5
-2	1.45	1.0
2 - 4	2.89	1.5
4 - 7	6.09	2.0
7 - 25	23.7	2.5
25 - 100	68.9	3.0
> 100	186	3.5
no sample		4.0
non-detect	0.65 U	4.5

result value

lower depth value

result flag

NOTE:
Result flag indicates --

U = not detected at reported value

J = estimated detected value

UJ = estimated non-detected value

Scale in Feet

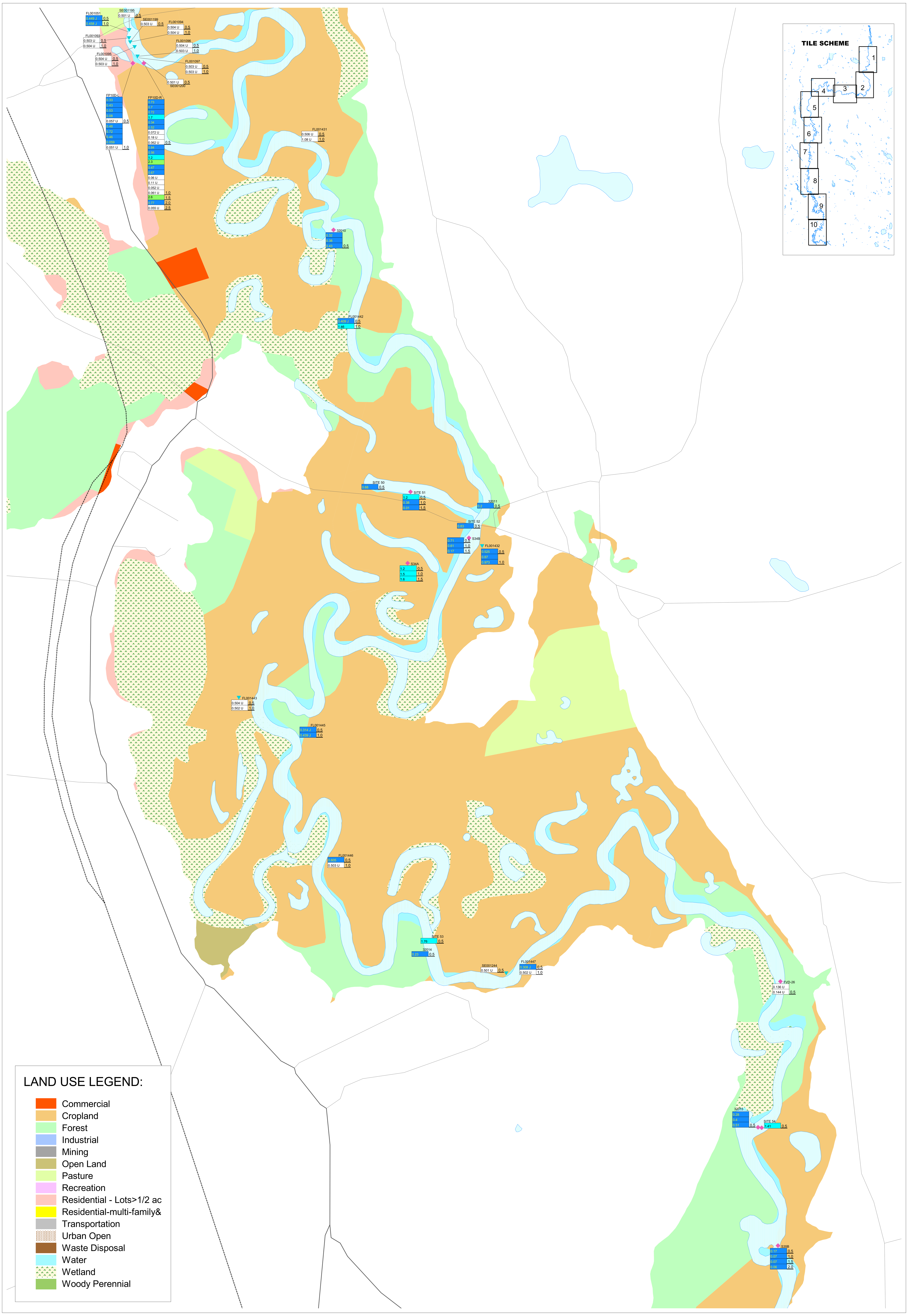
500 0 500 1000 1500

Housatonic River Project
Pittsfield, Massachusetts

TOTAL PCB RESULTS
REACH 7, 8 AND 9
TILE 8/10

NOTE: Datamart as of November 13, 2002

| O:\gepitt\aprs\r789_stacks_nov02.apr | layout - tile8 | o:\gepitt\epsfiles\plots\in\r789stacks8_nov02p.eps | 3:47 PM, 12/4/2002 |



LEGEND:

- GE
- EPA - Corps of Engineers
- Roads
- Railroads
- River

PCB Concentration (ppm)

	result value	lower depth value
< 1	0.51	0.5
-2	1.45	1.0
2 - 4	2.89	1.5
4 - 7	6.09	2.0
7 - 25	23.7	2.5
25 - 100	68.9	3.0
> 100	185	3.5
no sample		4.0
non-detect	0.65 U	4.5

result flag

NOTE:
Result flag indicates --
U = not detected at reported value
J = estimated detected value
UJ = estimated non-detected value

Scale in Feet

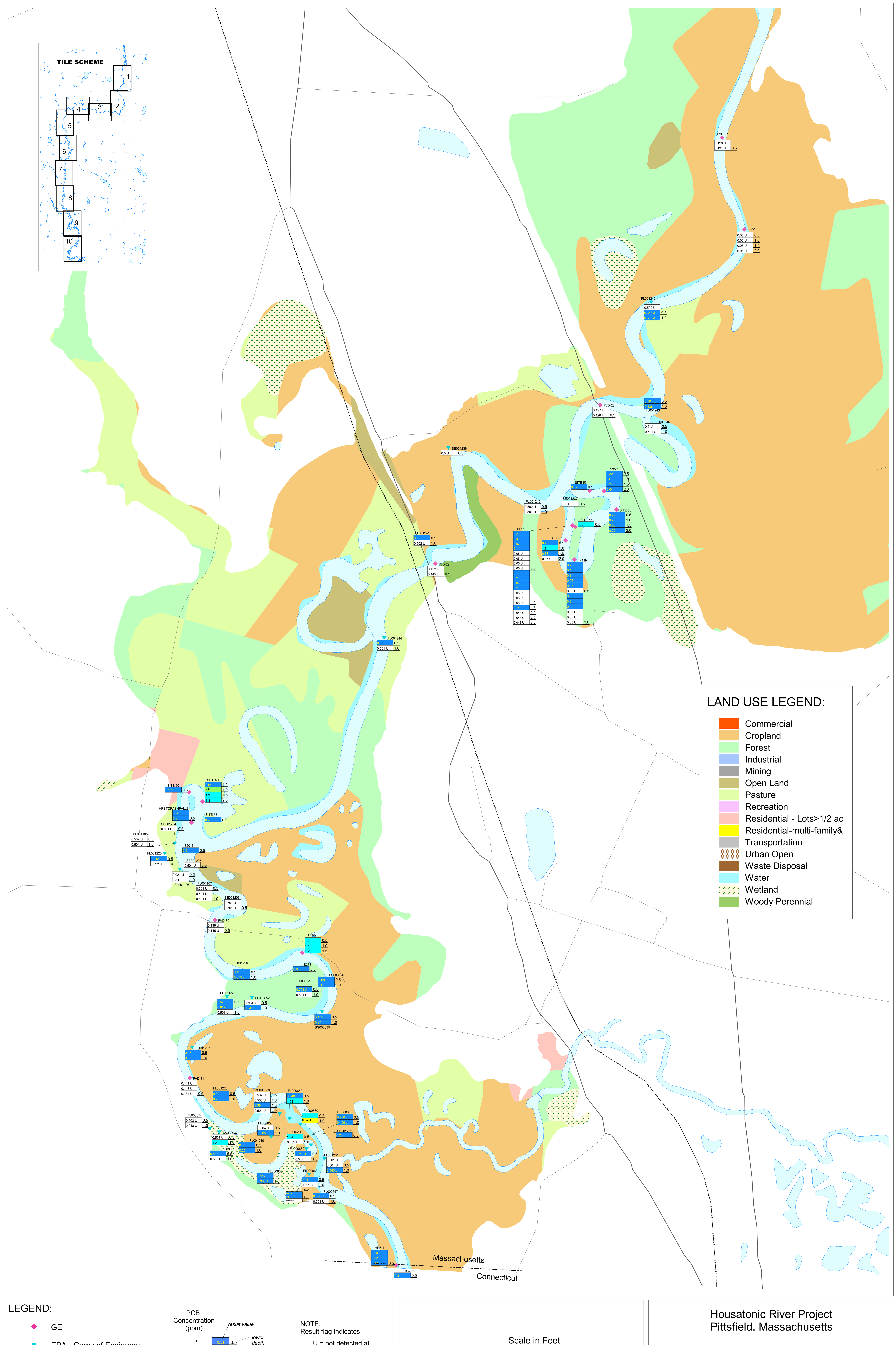
500 0 500 1000 1500

Housatonic River Project
Pittsfield, Massachusetts

TOTAL PCB RESULTS
REACH 7, 8 AND 9
TILE 9/10

NOTE: Datamart as of November 13, 2002

| O:\gepitt\aprs\r789_stacks_nov02.apr | layout - tile9 | o:\gepitt\epsfiles\plots\in\r789stacks9_nov02p.eps | 3:49 PM, 12/4/2002 |



LEGEND:

- GE
- EPA - Corps of Engineers
- Roads
- Railroads
- River

PCB Concentration (ppm)

< 1	0.51	0.5	result value / lower depth value
-2	1.45	1.0	
2 - 4	2.89	1.5	
4 - 7	6.09	2.0	
-25	23.7	2.5	
25 - 100	68.9	3.0	
> 100	186	3.5	
no sample		4.0	
non-detect	0.65 U	4.5	result flag

NOTE:
Result flag indicates --
U = not detected at reported value
J = estimated detected value
UJ = estimated non-detected value

Scale in Feet
500 0 500 1000 1500

Housatonic River Project
Pittsfield, Massachusetts

TOTAL PCB RESULTS
REACH 7, 8 AND 9
TILE 10/10

NOTE: Datamart as of November 13, 2002

| O:\gepitt\aprs\r789_stacks_nov02.apr | layout - tile10 | o:\gepitt\epsfiles\plots\in\r789stack10_nov02p.eps | 3:55 PM, 12/4/2002 |

3. ASSESSMENT ENDPOINT—COMMUNITY STRUCTURE, SURVIVAL, GROWTH, AND REPRODUCTION OF BENTHIC INVERTEBRATES

Highlights

Conceptual Model

- Benthic invertebrates include organisms that burrow within the sediment bed (i.e., infauna) and those found at the sediment-water interface (i.e., epifauna).
- Sediment and biota tissue are the most relevant exposure media, with the water column of lesser importance.

Exposure

- PCBs, PAHs, and some metals were retained as COCs.
- COCs were measured in tissue, sediment, and water at up to 13 sediment quality locations, synoptic with biological effects information. Chemistry data were also collected at numerous other locations throughout the PSA to provide a broader characterization of exposure.

Effects

- Site-specific toxicity tests (laboratory and in situ) indicate adverse responses, relative to both reference locations and negative controls.
- Benthic community appears altered at multiple locations with elevated PCB concentrations, relative to reference locations.
- Toxicity and community endpoints both indicate that Individual taxa exhibit a range of sensitivity to PCBs.
- Toxicological impacts are significantly correlated with PCB exposures.

Risk

- Comparison of exposure concentrations to literature effects benchmarks (sediment, tissue, water) suggests intermediate to high risk, particularly due to PCBs. The magnitude of risk varies by sampling location and reach.
- Toxicity identification evaluation (TIE) implicates non-polar organics (e.g., PCBs) as causal agent in toxicity tests.
- Contaminants other than PCBs and dioxins/furans do not exhibit concentration gradients consistent with the observed pattern of effects.
- Weight-of-evidence (WOE) approach used to characterize risks indicates adverse impacts to benthos throughout the PSA. Lower risks are predicted downstream of the PSA (sediment risk levels of intermediate or higher are limited to pockets of depositional sediment). Risks are low in Connecticut.

3.1 INTRODUCTION

The purpose of this section is to summarize the current and potential risks posed to benthic invertebrates exposed to contaminants of concern (COCs) in the Housatonic River, focusing on total PCBs (tPCBs) and other COCs originating from the General Electric (GE) facility in Pittsfield, MA.

A Pre-ERA was conducted to narrow the scope of the ERA by identifying contaminants, other than tPCBs, that pose potential risks to aquatic biota in the Primary Study Area (PSA) (Appendix B). Further screening was conducted to refine the list of contaminants of potential concern (COPCs) to those that were specifically relevant to benthic invertebrates. COCs that screened through to the risk assessment for benthic invertebrates were tPCBs, several metals, several polycyclic aromatic hydrocarbons (PAHs), and dibenzofuran.

This section is organized as follows.

- **Section 3.1 (Introduction and Conceptual Model)—**Describes the conceptual model for benthic invertebrates, including selection of representative taxa and establishment of measurement and assessment endpoints (Figure 3.1-1).
- **Section 3.2 (Exposure Assessment)—**Describes the quantification of exposures, both specific to locations for which linked biological effects information was collected (n = 13) as well as for the broader study area (Figure 3.1-2).
- **Section 3.3 (Effects Assessment)—**Describes the potential effects to benthic invertebrates exposed to site COCs in the PSA, and summarizes the range of benchmarks (effects thresholds) derived from both the literature and site-specific studies (Figure 3.1-3).
- **Section 3.4 (Risk Characterization)—**Integrates the exposure and effects assessments, characterizes risk for benthic invertebrates using three main lines of evidence, discusses sources of uncertainty, and presents an extrapolation of risks beyond the PSA to areas downstream of Woods Pond (Figure 3.1-4).

> This section provides a summary of the ERA for benthic invertebrates, which is presented in detail in Appendix D.

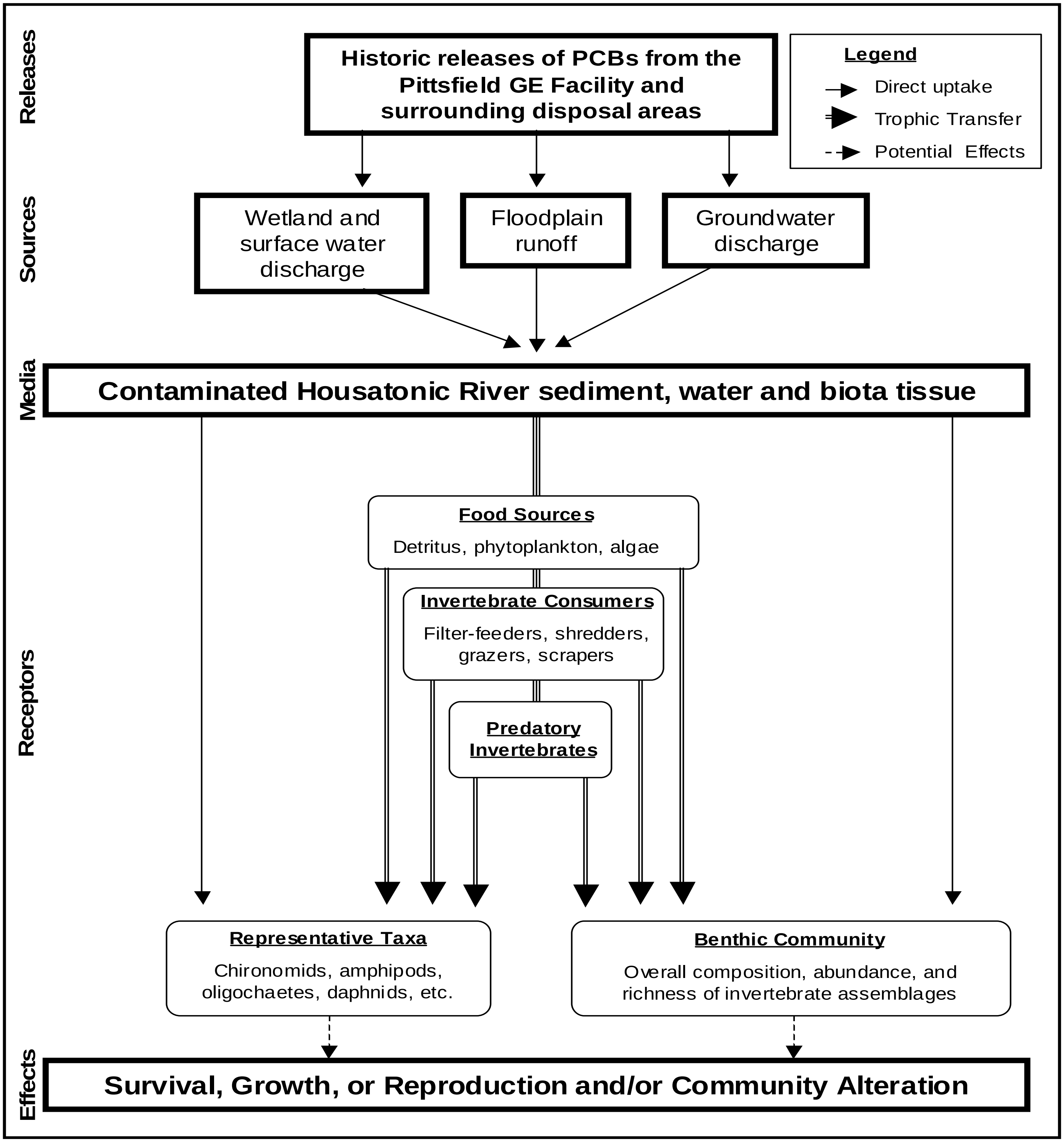


Figure 3.1-1 Conceptual Model Diagram: Exposure Pathways for Benthic Invertebrates Exposed to Contaminants of Concern (COCs) in the Housatonic River

Exposure

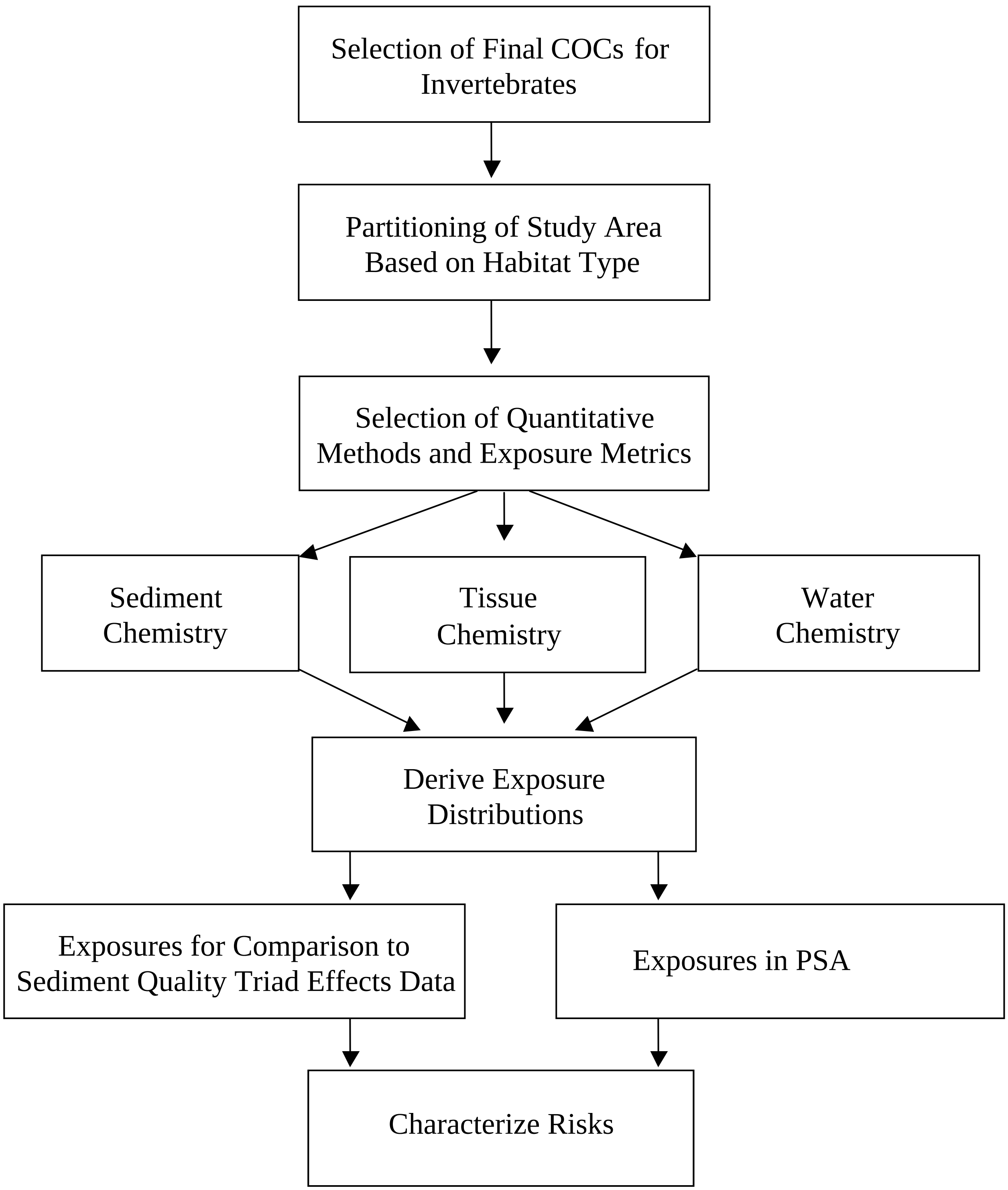


Figure 3.1-2 Overview of Approach Used to Assess Exposure of Benthic Invertebrates to Contaminants of Concern (COCs) in the Housatonic River

Effects

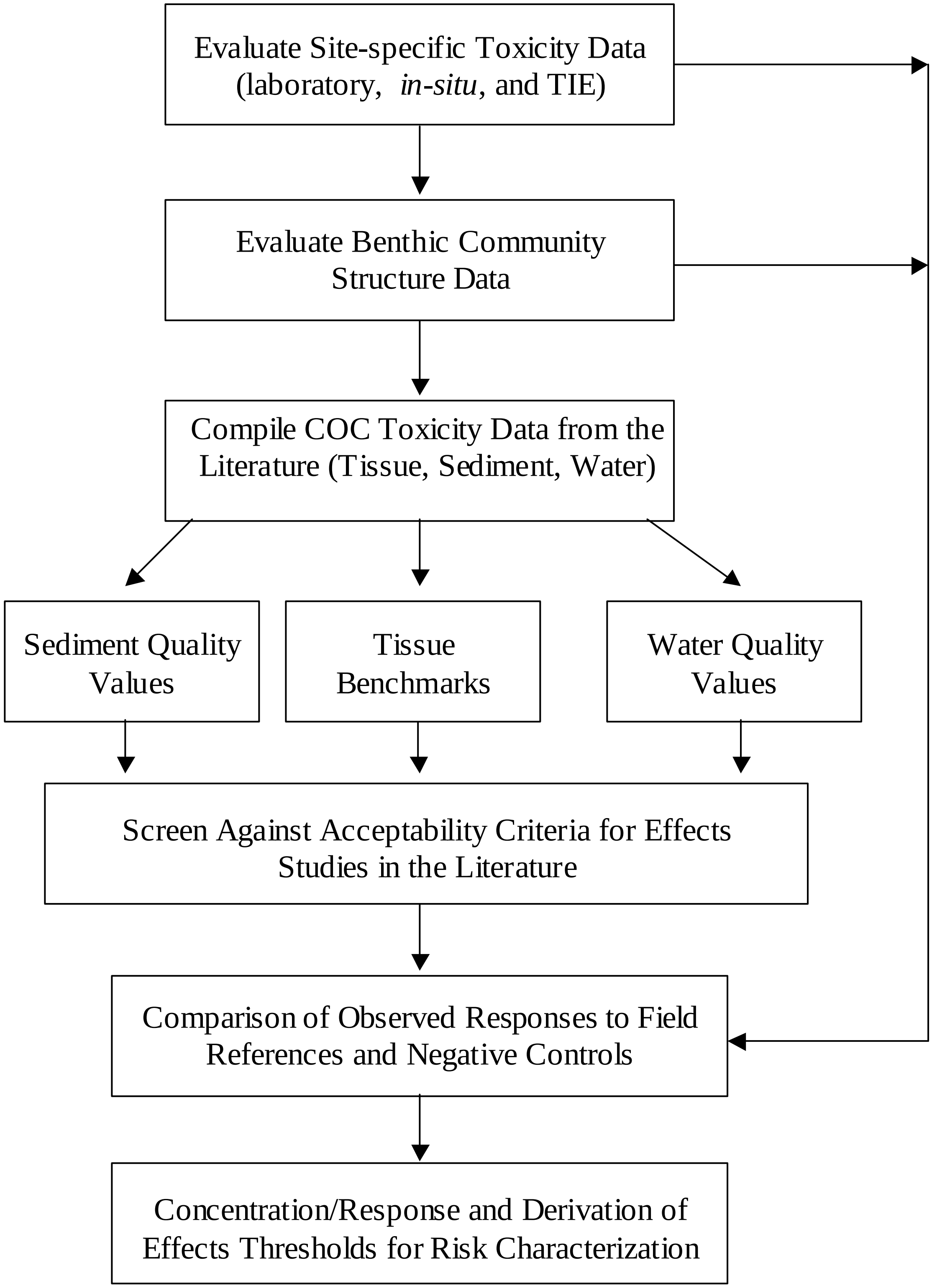


Figure 3.1-3 Overview of Approach Used to Assess the Effects of Contaminants of Concern (COCs) to Benthic Invertebrates in the Housatonic River

Risk Characterization

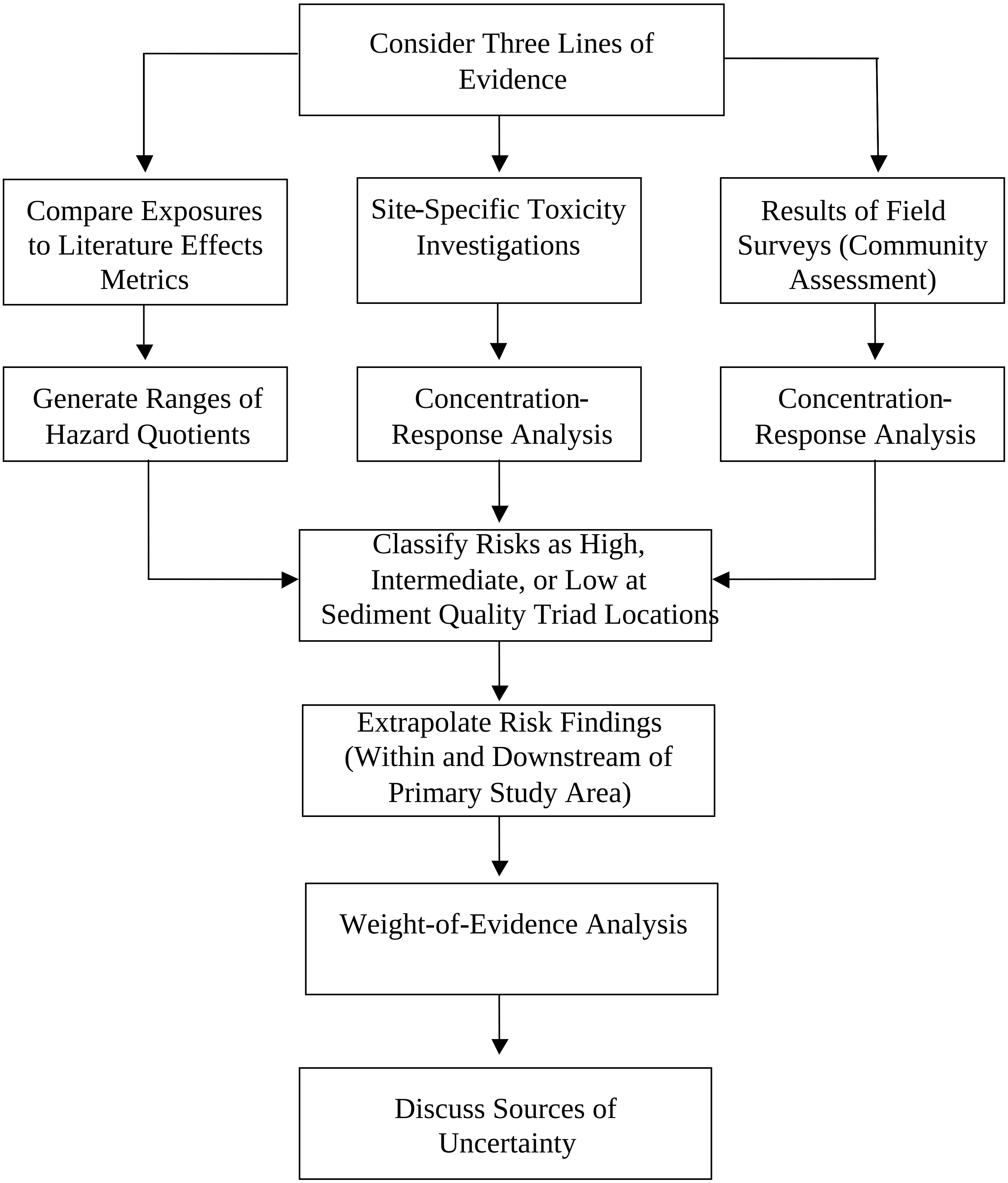


Figure 3.1-4 Overview of Approach Used to Characterize the Risks of Contaminants of Concern (COCs) to Benthic Invertebrates in the Housatonic River

3.1.1 Conceptual Model

The conceptual model presented in Figure 3.1-1 illustrates the exposure pathways for benthic invertebrates in the PSA. The benthic invertebrate receptor category includes all organisms that are exposed (in whole or in part) via sediment and porewater uptake pathways. This definition includes organisms that obtain significant exposure from both the sediment bed (particulates and pore water) and the water column (suspended particulates and water column). Therefore, the term "benthic invertebrates" used throughout the ERA is inclusive of both sediment infauna and epifauna. For benthic invertebrates, the dominant abiotic exposure media were sediment (solid phase and/or associated porewater) and surface water. Concentrations of COCs in tissues of benthic invertebrates were also considered; the latter integrate exposures from food consumption and direct uptake from abiotic media.

The problem formulation (Section 2) identified species used in toxicity tests as surrogates for the Housatonic River freshwater benthic community (i.e., *Chironomus tentans*, *Hyalella azteca*, *Daphnia magna*, *Ceriodaphnia dubia*). In addition, the freshwater oligochaete *Lumbriculus variegatus* was used as a bioaccumulation test species. Two other studies targeting individual taxa were conducted (dragonfly surveys, freshwater mussel study), but neither study collected biological effects data. Therefore, risks to dragonflies and shellfish in the river were characterized based on risks to representative invertebrate species that were studied more intensively and considered representative of the local benthic community in its entirety.

The measurement endpoints used to evaluate the assessment endpoint are presented below.

Measurement Endpoints for Benthic Invertebrates

- Determine, based on field studies, the extent to which reductions in benthic community abundance, biomass, taxa richness, taxa diversity, and other community metrics have occurred, including species-specific indications of adverse effects. Determine if these changes can be related to exposure to PCBs or other COCs in the sediment of the river.
- Determine, based on in situ and laboratory toxicity studies performed for this ERA, the extent to which the exposure to PCBs and other COCs in the river sediment may result in adverse impacts to survival, growth, and/or reproduction of representative benthic taxa.
- Determine, based on effects information from the literature, the extent to which the concentrations of PCBs and other COCs in Housatonic River sediment and/or water may cause adverse impacts to the benthic community.
- Determine, based on a combination of in situ tissue measurements and literature effects values, the extent to which the concentrations of PCBs bioaccumulated in the tissues of the benthic organisms will cause effects to survival, growth, or reproduction of one or more benthic taxa.

The approach used to characterize risks to benthic invertebrates was based on the Sediment Quality Triad. The Sediment Quality Triad is based on synoptic measurement of sediment chemistry, site-specific sediment toxicity, and benthic invertebrate community structure (Long and Chapman 1985; Chapman 1996).

Sediment Quality Triad Components Investigated in this Study

Standard Triad Components:

- Site-specific toxicity studies (laboratory and in situ); multiple species (*Hyalella magna*, *Chironomus tentans*, *Daphnia magna*), multiple test durations (48-h, 10-d, 42-d).
- Benthic macroinvertebrate community structure.
- Abiotic media chemistry (sediment, overlying water, and porewater).

Additional Components:

- Bioaccumulation assessment (chemistry in resident invertebrates [predators and shredders]; 7-d bioaccumulation assessment to deposit-feeding invertebrates in laboratory [oligochaete *Lumbriculus variegatus*]).
- Toxicity Identification Evaluations (TIEs).

A summary of the studies conducted and their linkage to the ERA is provided in Figure 3.1-5.

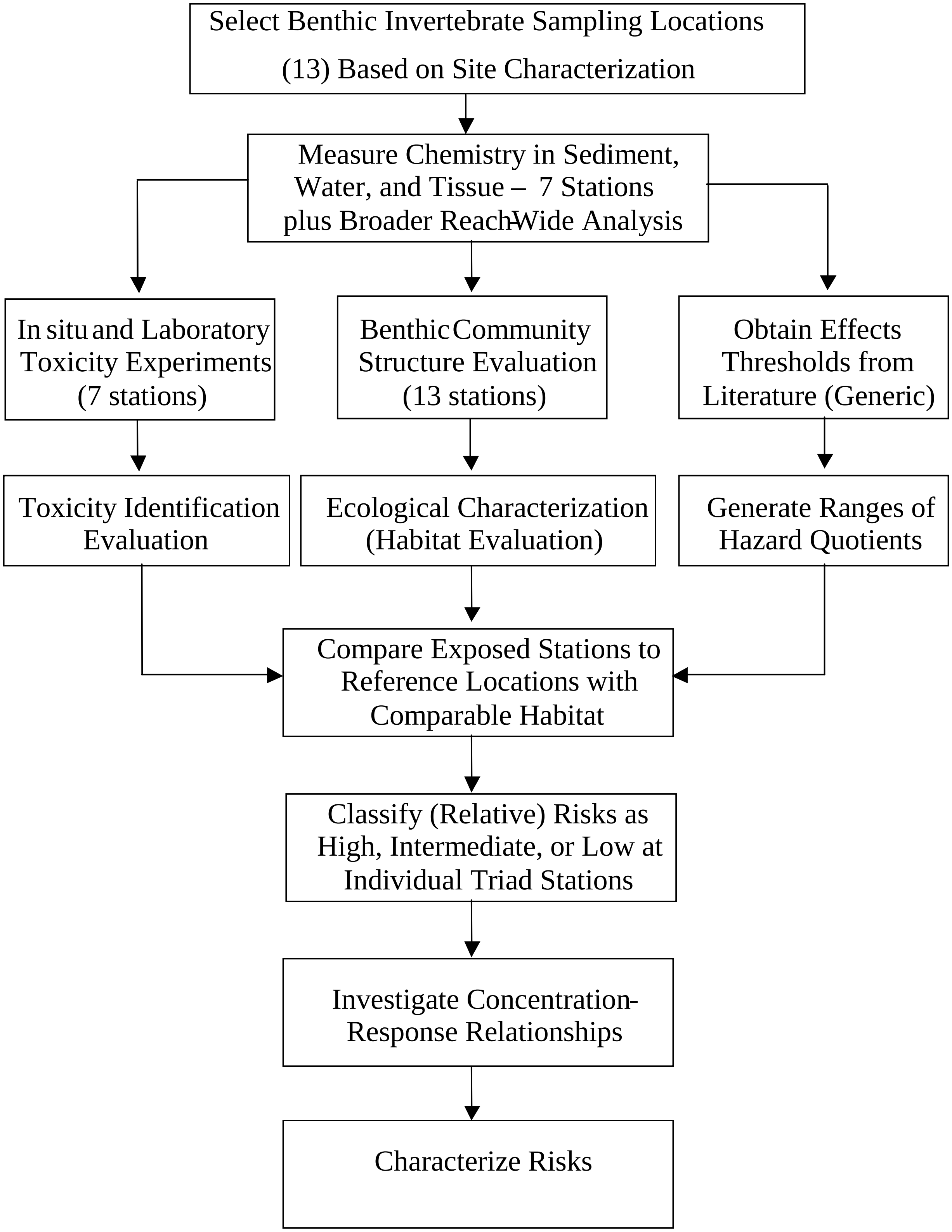


Figure 3.1-5 Summary of Studies Conducted in Conjunction with Ecological Risk Assessment for Benthic Invertebrates, and Linkage to ERA

3.2 EXPOSURE ASSESSMENT

The approach used to assess exposure of benthic invertebrates to tPCBs and other COCs in the Housatonic River PSA is shown in Figure 3.1-2. Unlike higher trophic level receptors, a complex exposure model was not required. Instead, exposures were assessed as either the COC concentrations in abiotic site media (i.e., sediment, water), or as the tissue body burdens that represent integrated exposure from all sources.

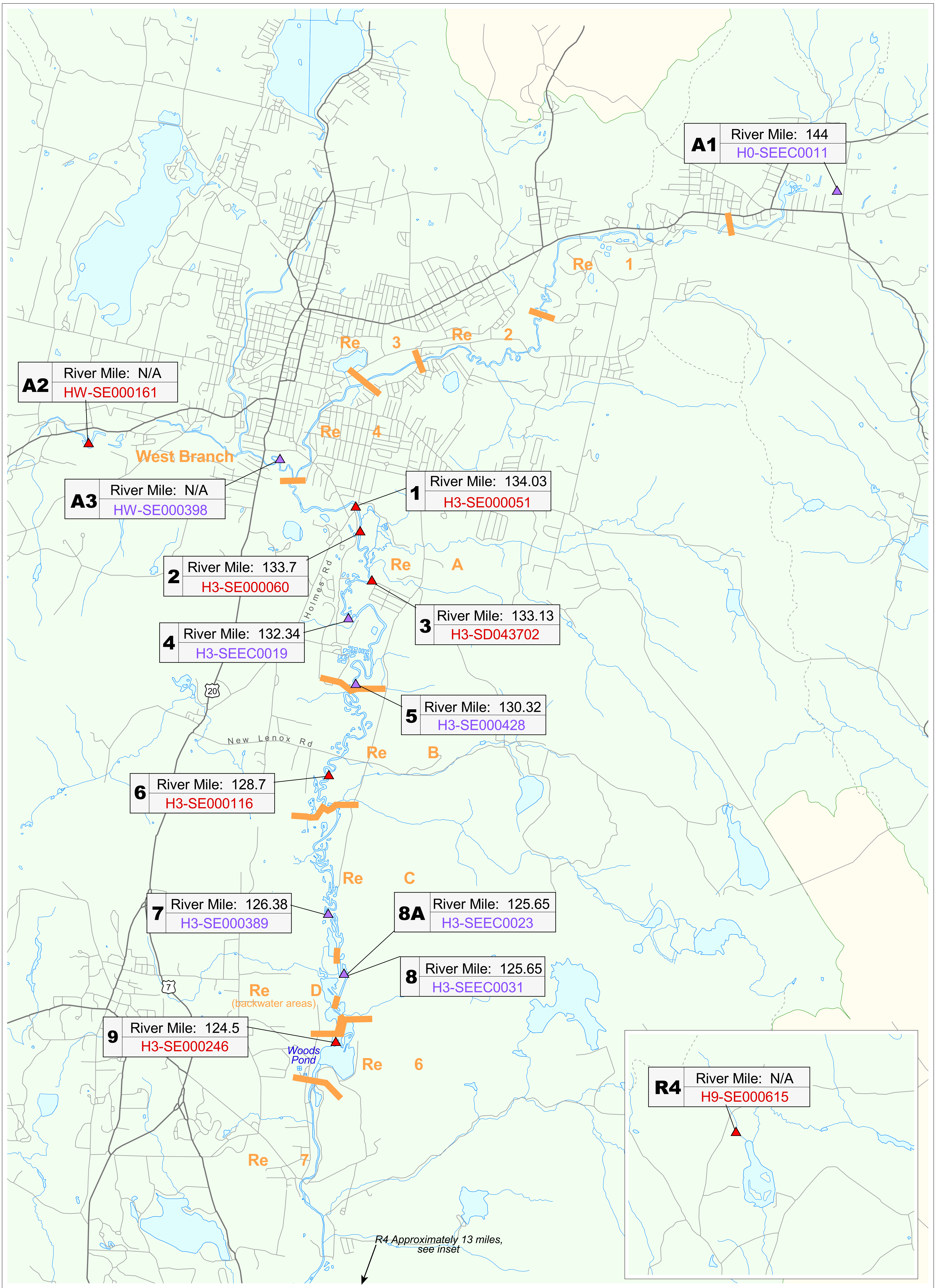
To match exposure data with effects, data were collected at 13 benthic community sampling locations and 7 sediment toxicity locations (Figure 3.2-1).

Summary of Benthic Invertebrate Sampling Locations

- Upstream Reference Locations: A1, A2, A3 (arranged north to south).
- Contaminated Locations on Housatonic River: 1 to 9 (arranged north to south).
- Downstream (watershed) Reference Location: R4 (Threemile Pond).

3.2.1 Selection of COCs for Benthic Invertebrates

The contaminants initially considered in the benthic invertebrate exposure assessment (COPCs) were identified in the Pre-ERA (Appendix B). The invertebrate Pre-ERA included screening on a reach-by-reach basis and subdivision of COPCs by major hydrological/geomorphological category. A receptor-specific screening stage (Section D.2.1.1) identified contaminants of concern (COCs) specific to benthic invertebrates in the PSA.



LEGEND:

- Benthic Community Stations Without Toxicity Testing
- Benthic Community Stations With Toxicity Testing
- Reach Breaks
- Roads
- Hydrography
- Housatonic River Basin Boundary

Notes: - Code in eft box of each ID represents "simplified" nomenclature used for benthic invertebrate ERA.
- Station 8A was positioned 12 meters from Station 8, and was tested for aboratory toxicity only.

N W E S

2500 0 2500 5000 Feet

0.5 0 0.5 1 Miles

Ecological Risk Assessment
GE/Housatonic River Site
Rest of River

FIGURE 3.2-1
BENTHIC INVERTEBRATE SAMPLING LOCATIONS AND SIMPLIFIED STATION IDENTIFIERS

The sediment COCs for benthic invertebrates are presented below. PCBs were retained as a sediment COC in all PSA reaches. PAHs were retained throughout the PSA, although the number of individual PAH compounds retained was greater for Reaches 5A and 5C, relative to the other reaches. Dibenzofuran was retained only for Reach 5A. Metals were retained in Reaches 5C and 6 only. Organochlorine pesticides were eliminated from further consideration in the invertebrate portion of the ERA.

Contaminants of Concern for Benthic Invertebrates

- Chlorinated organic compounds – tPCBs, dioxins/furans.
- Metals – antimony, barium, cadmium, chromium, copper, lead, mercury, silver, and tin.
- Semivolatile organic compounds (SVOCs) – dibenzofuran.
- PAHs – numerous individual PAH compounds, including low- and high-molecular weight PAHs.

Surface water COPCs identified in the Pre-ERA (Appendix B) included dioxins/furans, PCBs, and silver. Therefore, the water chemistry screening did not result in any additional contaminants that were not already considered as sediment COCs for invertebrates.

3.2.2 Types of Exposure Data

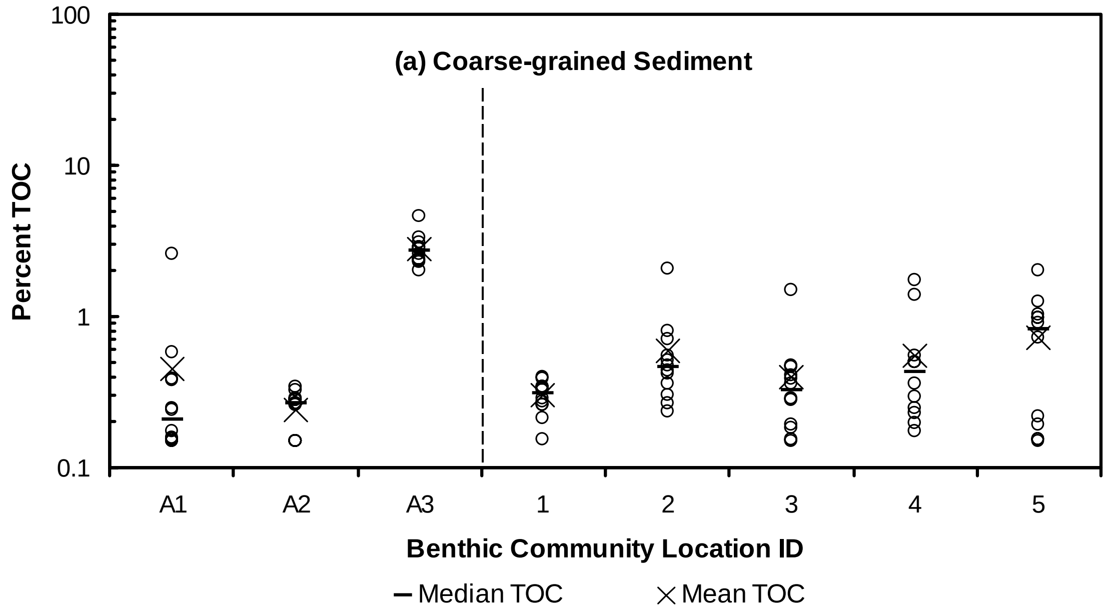
Use of the Sediment Quality Triad approach assumes correspondence between exposure and effects endpoints. Exposure-effects relationships were investigated using the single “most synoptic” chemistry value paired with each toxicity endpoint. Attachment D.5 presents a supplemental analysis using the median of all sediment data collected in the area surrounding each sampling location for comparison. In most cases, the two approaches yielded similar results, demonstrating that the interpretations in the risk assessment were not an artifact of the exposure data set selected.

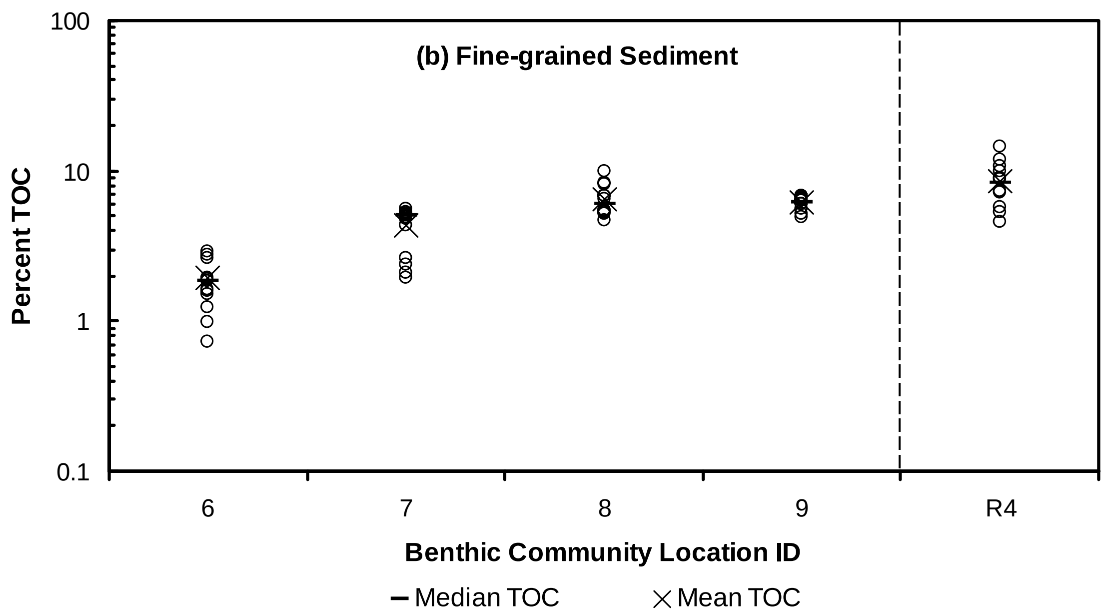
3.2.3 Habitat Characterization

The preliminary results of physical and ecological investigations were used to group sampling locations with similar properties for the exposure assessment to discriminate between substrate-related responses and those attributable to contaminants.

> ***Rationale for Grouping of Benthic Sampling Locations in ERA***
>
> Grouping sampling locations for the exposure assessment was done for three primary reasons:
>
> - To determine the appropriateness of reference locations for making statistical comparisons to exposed locations.
> - To provide a means of separating physical and ecological characteristics in a manner consistent with both the exposure and effects assessment.
> - To provide a tool to assess if there are other influences that were not well characterized.

Both physical substrate parameters (e.g., substrate type, organic carbon content, sediment particle size distribution (see Figures 3.2-2 and 3.2-3) and habitat parameters (e.g., riparian vegetation, macrophyte coverage) were used to evaluate the benthic sampling locations. A clear change in substrate and habitat type was observed between Locations 5 and 6, which is coincident with the transition in river regime from Reach 5A to Reach 5B, and the location of the Pittsfield wastewater treatment plant (WWTP) outfall.





Note: Locations A1, A2, A3, and R4 represent less contaminated "reference" areas.

Figure 3.2-2 Total Organic Carbon (TOC) Concentrations in Sediment Collected at Benthic Invertebrate Grab Sampling Locations

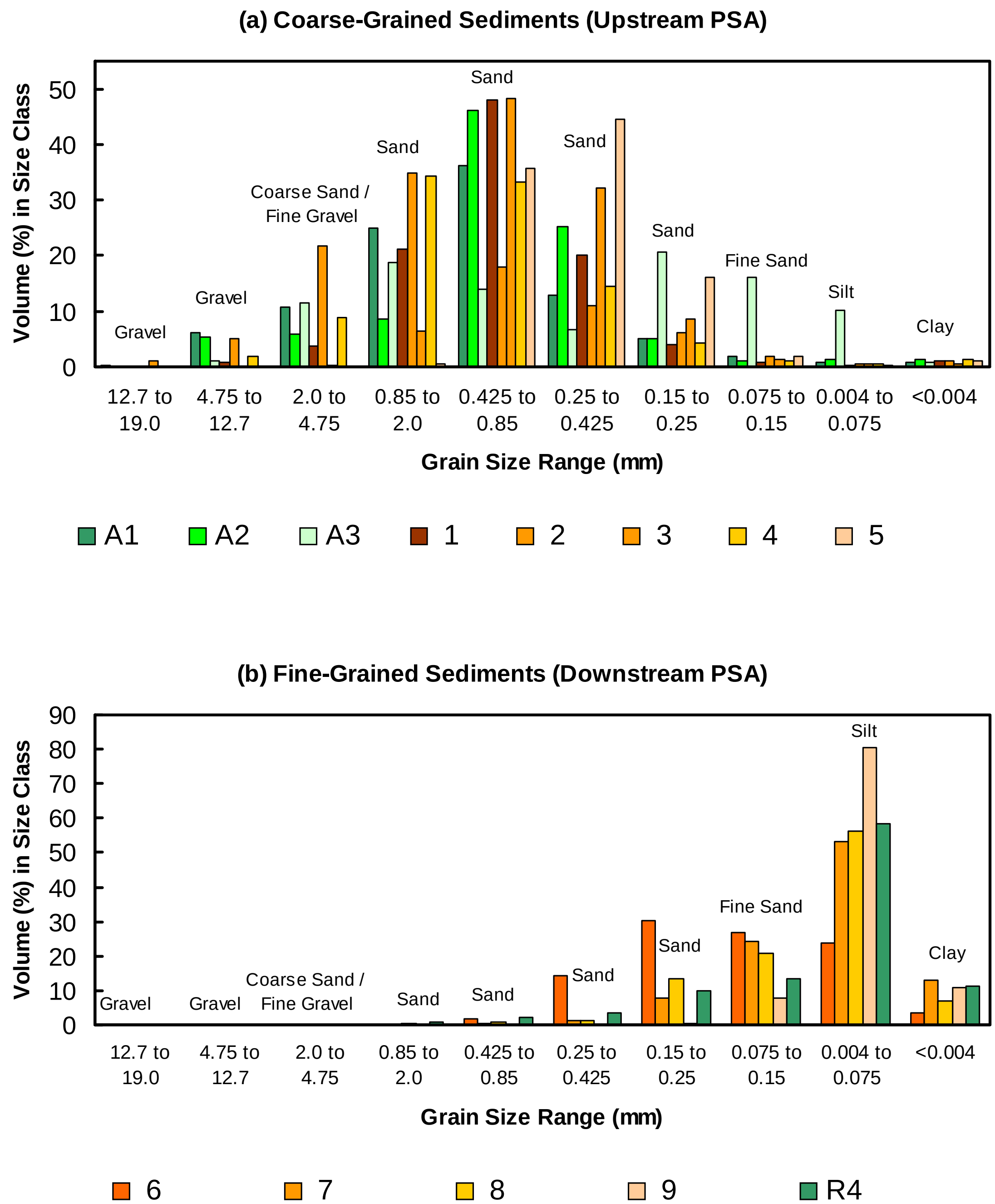


Figure 3.2-3 Mean Particle Size Distributions in Benthic Community Grab Samples in Coarse-Grained and Fine-Grained Substrates

Grouping of Benthic Sampling Locations

Benthic sampling locations in Figure 3.2-1 were assigned to one of the following categories:

- "Coarser" Reference Locations (C/R) – Low total organic carbon (TOC) (typically less than 1%), sandy sediment found either upstream of influence from the GE facility or on the West Branch. Three locations (A1, A2, A3).
- "Coarser" Contaminated Locations (C/C) – Low TOC (typically less than 1%), sandy sediment found between the confluence and the Pittsfield WWTP. Five locations (1, 2, 3, 4, 5).
- "Finer" Reference Locations (F/R) – High TOC, silty sediment found outside the PSA at Threemile Pond (Location R4).
- "Finer" Contaminated Locations (F/C) – High TOC (typically a few percent or greater), silty sediment found downstream of the Pittsfield WWTP. Four locations (6, 7, 8, 9).

3.2.4 Assessment of Sediment Chemistry

3.2.4.1 Sources of Sediment Data

There are multiple sources of sediment data, each with a varying degree of correspondence to various effects metrics.

Sediment Data Sources Used in Benthic Invertebrate ERA

- Benthic Community Grab Samples (1999) – 12 replicate samples taken at each of 13 locations and analyzed for PCBs and other parameters; synoptic with benthic community structure samples.
- Laboratory Toxicity Samples (1999) – Composite samples (mixture of five cores) collected; samples were submitted for laboratory toxicity tests and analyses including tPCBs and TOC.
- In situ Toxicity Samples – Composite samples taken in a similar fashion to the laboratory samples; and at different time periods (i.e., end of 48-hour, 7-day, and 10-day exposure periods). All samples analyzed for tPCBs; 7-day samples also analyzed for PCB congeners, PAHs, pesticides, metals, chlorinated benzenes, and TOC.
- Other Sediment Samples – Included discrete river sampling, screening samples prior to the biological investigations, and collected with supplemental studies, such as the Wright State University (WSU) TIE investigation. Other data not considered synoptic were used only for screening, characterizing generic exposures in the PSA, or for extrapolating dose-response relationships.

3.2.4.2 Distribution and Concentrations of PCBs

3.2.4.2.1 Benthic Community Grabs

Individual replicate concentrations of tPCBs for each benthic sampling location are presented in Figure 3.2-4. The data indicate highly elevated tPCB concentrations in the C/C sites, with median concentrations of approximately 5 to 25 mg/kg. In the F/C sites, median PCB concentrations were significantly lower (pooled variance t-test; $p < 0.001$). There was considerable variability in tPCB concentrations between replicates at most locations (Figure 3.2-4), indicative of small-scale variability in PCB concentrations.

3.2.4.2.2 Toxicity Test Samples

Concentrations of tPCBs were measured in sediment in conjunction with laboratory and in situ toxicity and bioaccumulation tests conducted between May and July 1999 (EVS 2003). As with the benthic community grab samples, tPCB concentrations were quite variable within locations across the four toxicity sampling events (Figure 3.2-5).

3.2.4.2.3 Other Sediment Samples

Figure 3.2-6 depicts the spatial distribution of tPCB concentrations within the PSA. The data indicate that median PCB concentrations are highest in the upstream reaches of the PSA, and decrease with distance from the GE facility. The median concentrations are lowest just downstream of the WWTP, but increase moving farther downstream to Woods Pond. Figure 3.2-7 compares the median PCB concentrations observed synoptic with the benthic community and toxicity data sets to the larger data set including all main channel PSA sediment samples. In coarse-grained sediment, the community and toxicity grab samples are reasonably precise and unbiased indicators of the average PCB concentrations within Reach 5A. However, in fine-grained sediment, the sample concentrations associated with the toxicity tests are greater than the average PCB concentrations, representing the 80th to 90th percentile of the tPCB distribution in fine-grained PSA sediment. Conversely, the fine-grained sediment sample concentrations associated with benthic community locations are lower than the average PCB concentrations in fine-grained sediment, representing the 20th to 60th percentile of the tPCB distribution in fine-grained reaches of the PSA.

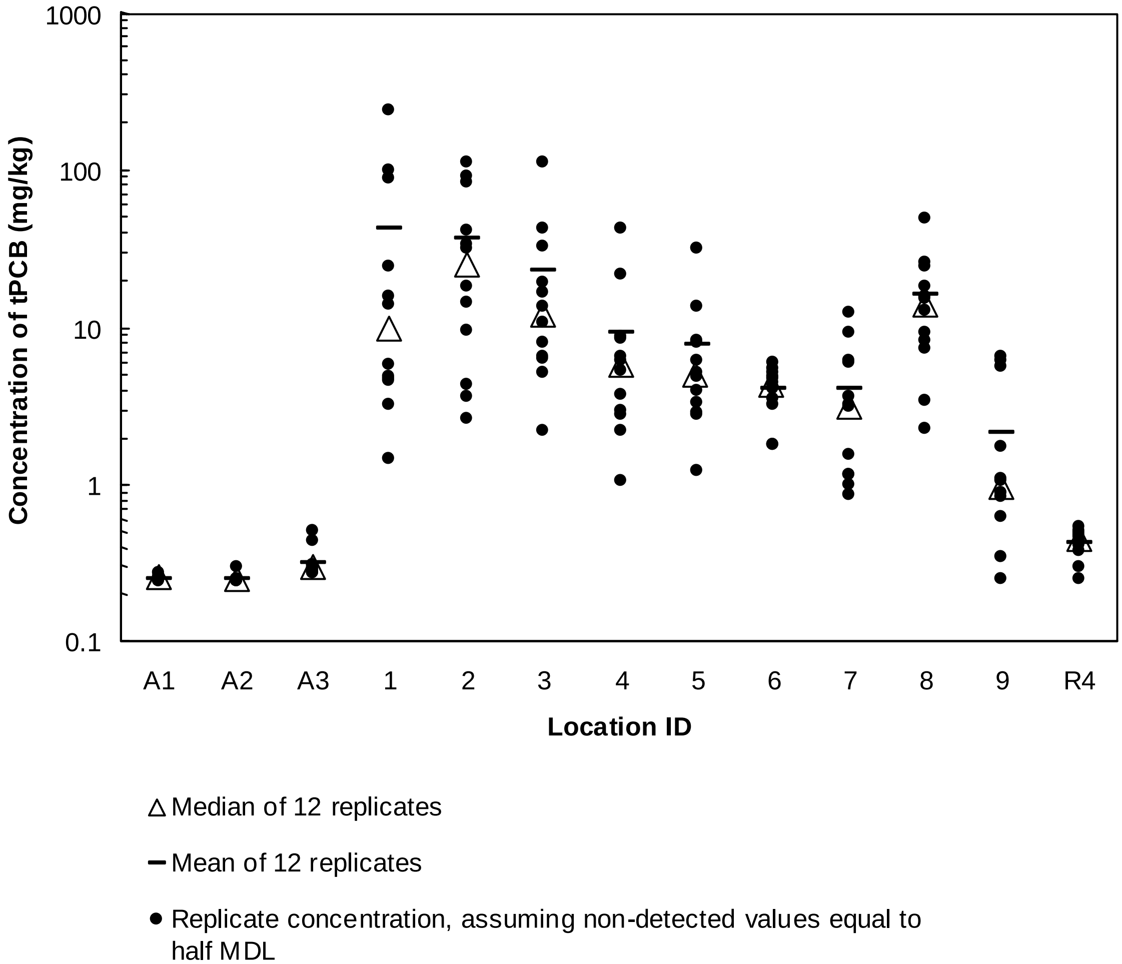


Figure 3.2-4 Concentrations of tPCBs in Sediment by Sampling Location for Individual Benthic Community Grab Samples, and Associated Measures of Central Tendency

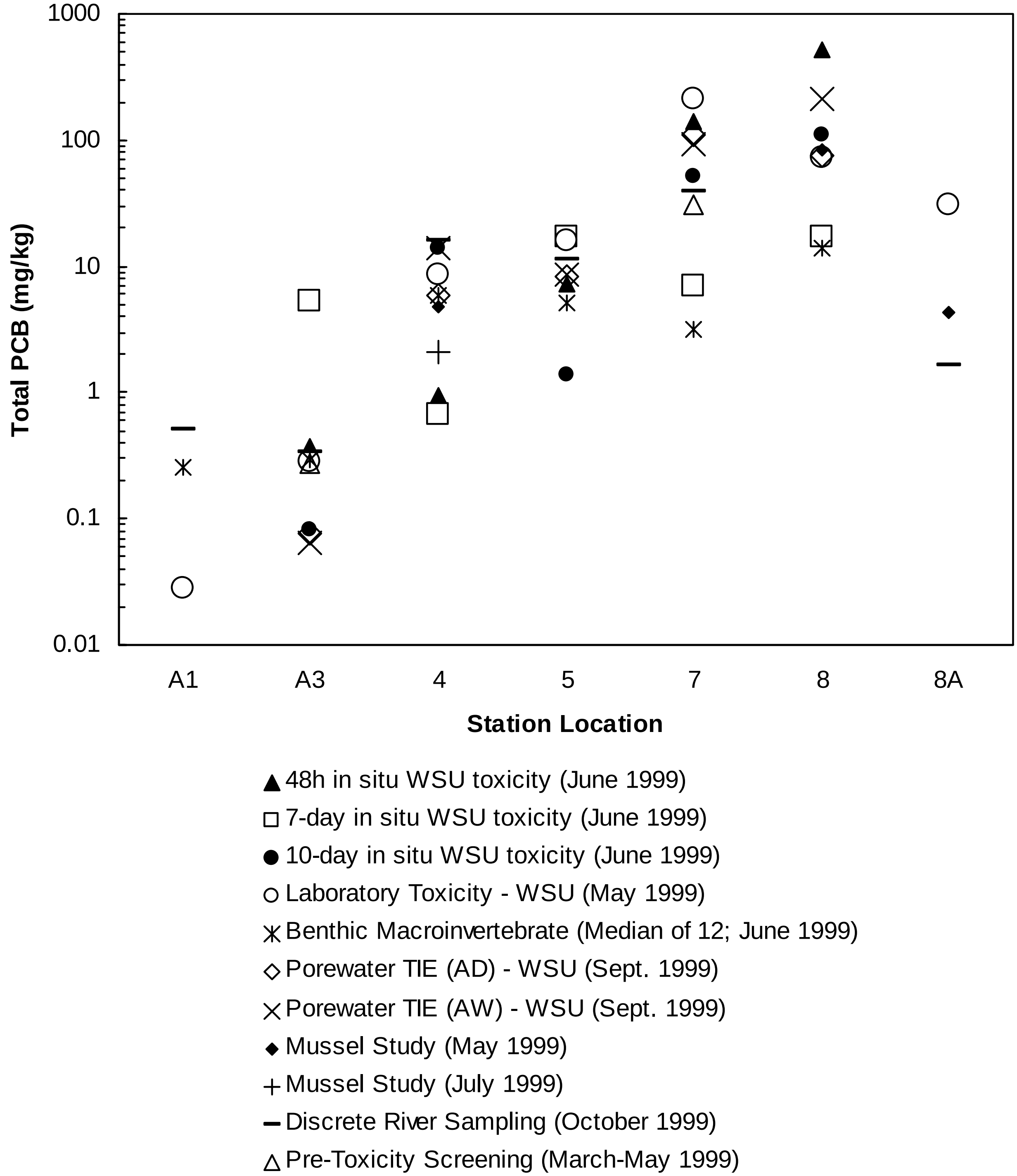
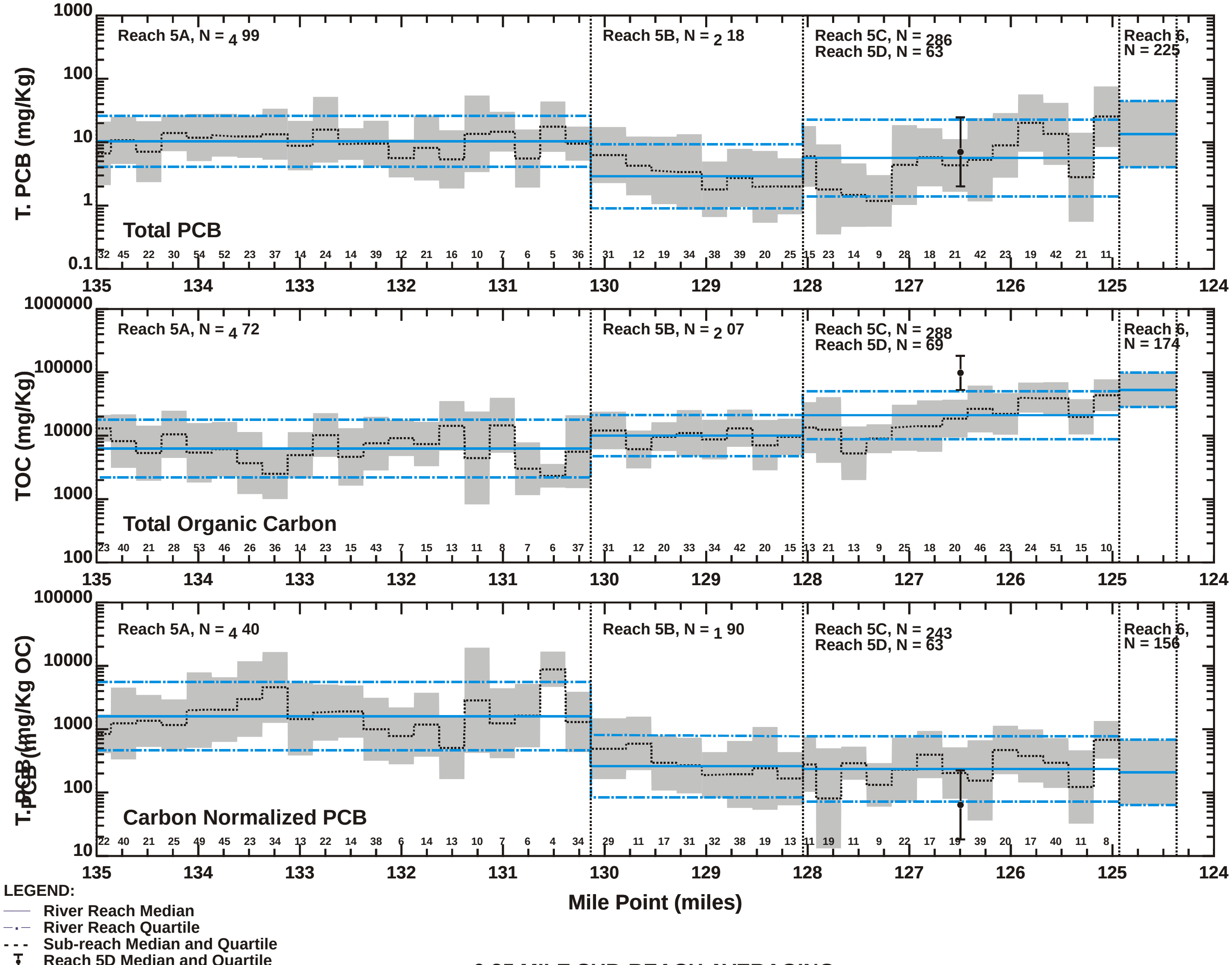


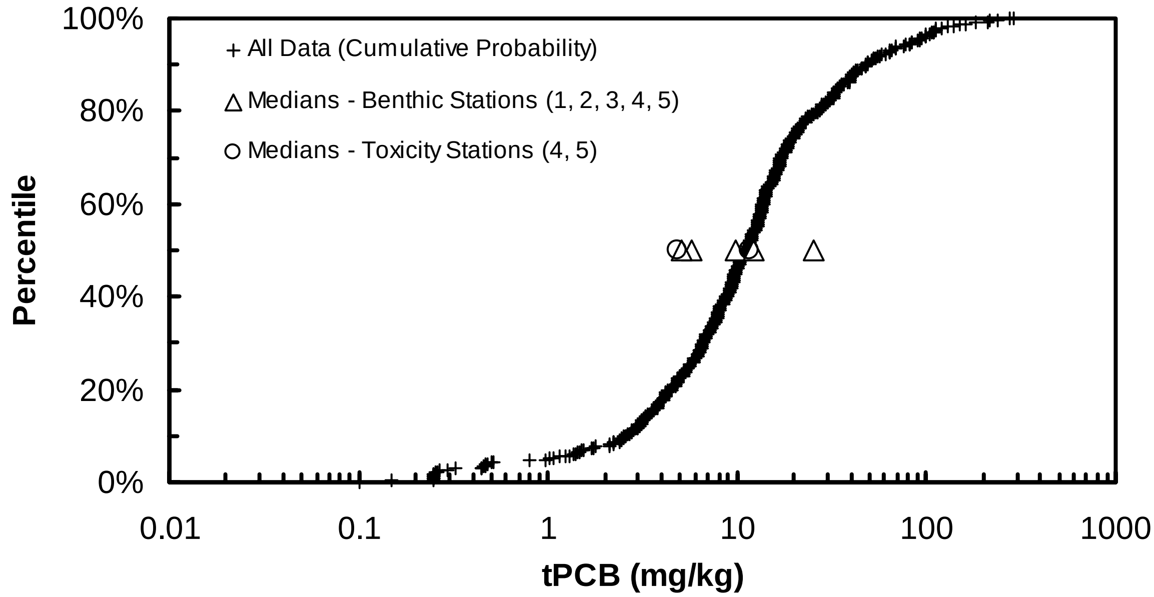
Figure 3.2-5 Comparison of tPCB Concentrations in Sediment Collected at Benthic Toxicity Sampling Locations, from Various Sampling Efforts Conducted in 1999



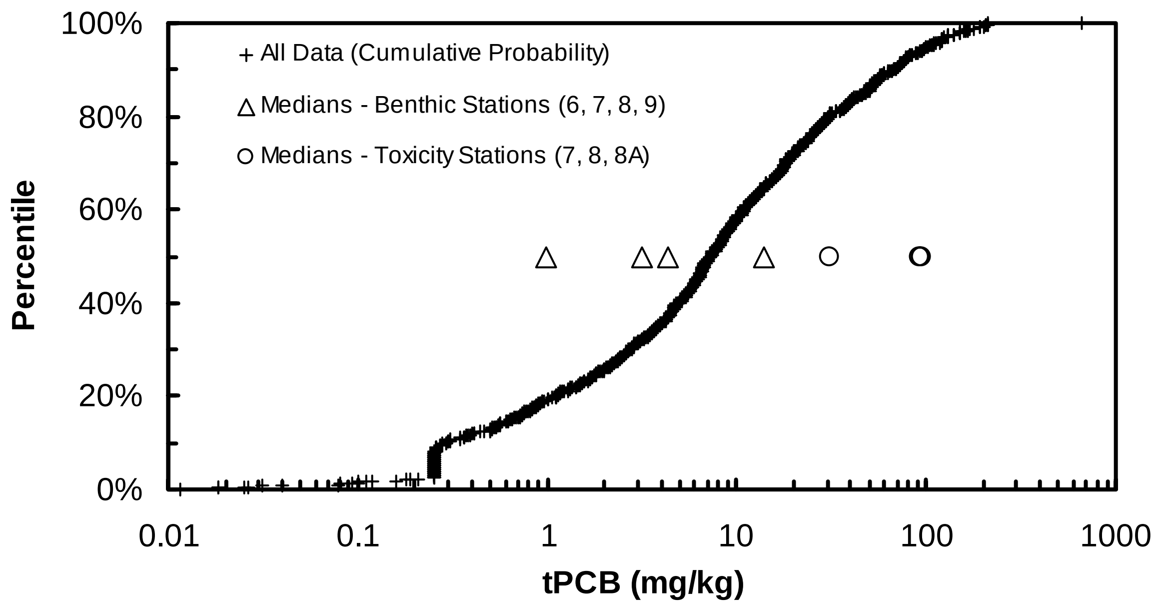
0.25 MILE SUB-REACH AVERAGING
Median and Quartiles of PCB and TOC in the Housatonic River (NOTE: Reach 5D is Backwaters)
(All Data, Half DL, Surface Layer: 0 - 6")

Figure 3.2-6 Medians and Quartiles of PCB and TOC in the Housatonic River PSA, Subdivided by River Reach and 0.25 Mile Subreaches

(a) Coarse-Grained Sediments (Reach 5A)



(b) Fine-Grained Sediments (Reaches 5B, 5C, 5D, and 6)



Note: tPCB Concentrations plotted assume non-detected values equal half DL. The shape of the cumulative probability curve is only sensitive to this assumption below 1 mg/kg.

Figure 3.2-7 Comparison of tPCB Concentrations from PSA-Wide Sampling to Benthic Community and Toxicity Grab Samples

For areas downstream of Woods Pond, the tPCB concentrations are lower than in the PSA (by approximately an order of magnitude). Sediment tPCB concentrations downstream of the Connecticut border were generally below 1 mg/kg, reflecting the general trend of decreasing concentration with distance downstream.

3.2.4.3 Distribution and Concentration of Other COCs

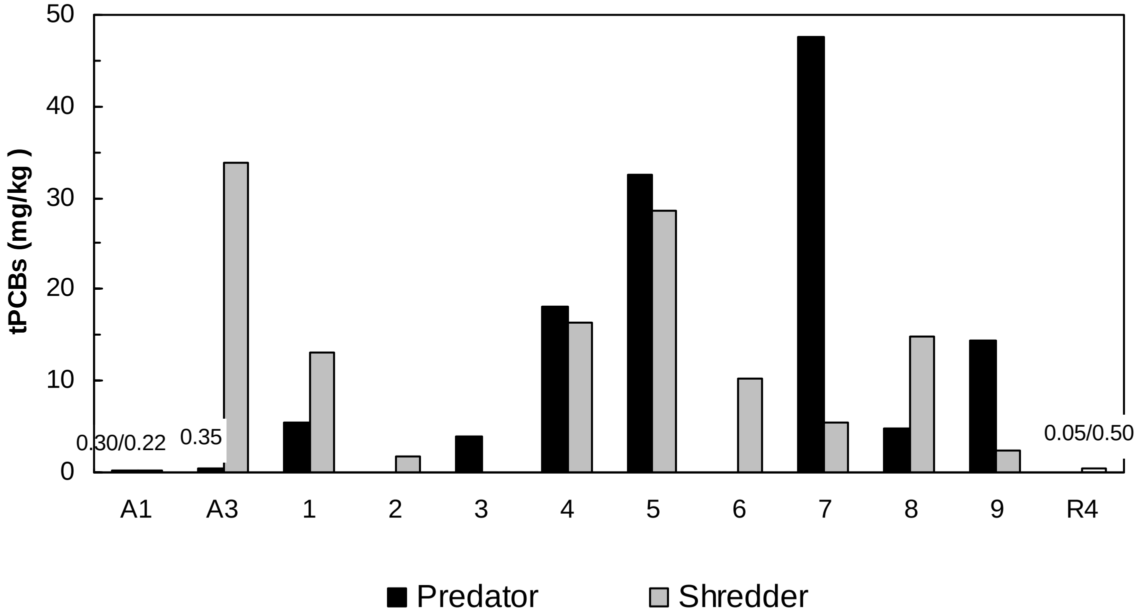
Fewer sediment data are available for other COCs; however, there were sufficient data to characterize patterns of concentrations throughout the PSA (Appendix D; Figures D.2-18 through D.2-30). A brief summary of spatial patterns for these COCs is provided below.

Summary of Patterns of Other COCs in Sediment Data

- Dioxin/furan concentrations are elevated at downstream fine-grained locations relative to upstream and/or reference locations, and are positively correlated with tPCB concentrations.
- Dibenzofurans were detected primarily in Reach 5A . Most concentrations were in the 0.1 to 1.0 mg/kg range. This COC was eliminated from the ERA (rationale provided in Appendix D).
- Total PAH concentrations were highly variable, both spatially and between sampling events. The median concentrations were greatest near the urbanized areas of the Housatonic River watershed, and were lowest at Location A1 and at the Woods Pond headwaters.
- Metals concentrations were typically lower at upstream sites relative to those associated with the fine-grained sediment found downstream. Metals concentrations were significantly correlated ($p < 0.05$) with TOC concentrations and with concentrations of other metals.

3.2.5 Assessment of Tissue Chemistry

Benthic tissue data are less abundant than data for abiotic media, nevertheless, the available data provide a measure of the site-specific bioavailability of the COCs. Figure 3.2-8 presents the distribution of tPCB concentrations by sampling location and tissue type, for samples collected at the benthic sampling locations. Most reference samples had tPCB tissue concentrations below 1.0 mg/kg ww. In contrast, contaminated locations had elevated concentrations ranging from 2 to 48 mg/kg ww.



Notes: Text labels indicate detected values close to zero. Missing values with no text labels indicate locations where no analysis was completed due to insufficient sample volume.

Tissue concentrations are wet weight.

Figure 3.2-8 Concentrations of tPCBs in Benthic Invertebrate Tissues by Location and Functional Feeding Group

Sources of Benthic Invertebrate Tissue Chemistry Data

- Analysis of composite samples of "predators" and "shredders," respectively, conducted in 1999 by EPA. Each tissue sample was analyzed for lipids and a number of organic contaminants, including PCBs (as congeners, as Aroclors, and as tPCBs), dioxins/furans, and pesticides.
- Data from the 7-day in situ bioaccumulation study (EVS 2003) conducted with the oligochaete worm *Lumbriculus variegatus* at 6 sampling locations.
- Academy of Natural Sciences of Philadelphia long-term historical tissue PCB monitoring of dobsonfly, caddisfly, and stonefly nymphs/larvae collected downstream of the PSA near Cornwall, CT (BBL & QEA 2003).

3.2.6 Surface Water Chemistry

Surface water data have limited application to the benthic ERA due to the uncertainty in extrapolating from water chemistry to effects in sediment-dwelling biota. Because of this uncertainty, the only data considered relevant were those collected synoptic with effects measurements. Unfiltered overlying site water was collected in conjunction with the 7-d in situ toxicity testing (EVS 2003) and evaluated for tPCBs, PCB congeners, PAHs, pesticides, and metals. Only tPCBs were measured for the 48-h and 10-d exposures.

Summary of Water Chemistry COC Concentrations

- Concentrations of tPCBs at upstream reference locations were less than 10 nanograms per liter (ng/L); concentrations at contaminated locations ranged from approximately 100 ng/L to 300 ng/L.
- Concentrations of furans followed the spatial patterns in tPCBs, with total detected furans below 10 picograms per liter (pg/L) at upstream reference locations, and concentrations of 60 to 120 pg/L at contaminated locations.
- No dibenzo-p-dioxins or silver were detected in the samples collected in conjunction with the 7-day in situ tests, and concentrations of other metals in surface water were low relative to Ambient Water Quality Criteria (EVS 2003).

3.3 EFFECTS ASSESSMENT

The effects assessment for benthic invertebrates (Figure 3.1-3) is based primarily on the site-specific biological investigations performed at the 13 benthic sampling locations. Both toxicity assessments (i.e., laboratory toxicity, in situ toxicity, and TIE) and community evaluations (i.e., benthic macroinvertebrate community composition) were compared to appropriate field references to determine whether the contaminated locations exhibited biological impairment. The association between effects metrics and sediment tPCB concentrations was also explored.

The effects assessment also includes an overview of the literature on the effects of tPCBs and other COCs on survival, growth, and reproduction of benthic invertebrates. Literature was reviewed to derive effects metrics for tissue, sediment, and water. In recognition of the uncertainty inherent in threshold effects concentrations for these media, ranges of benchmarks were derived instead of relying on single effects thresholds.

An integrated discussion of the effects metrics and maximum acceptable threshold concentrations (MATCs) calculated from these metrics is provided in the Risk Characterization.

3.3.1 Sediment Toxicity Testing

3.3.1.1 Methods

Wright State University (WSU) conducted site-specific in situ and laboratory toxicity testing of Housatonic River sediment (EVS 2003) at seven locations, six of which were co-located with the benthic macroinvertebrate sampling locations (Figure 3.2-1).

Toxicity Test Methods

Laboratory testing (EPA 2000 protocols)

- Chronic 42-d bulk sediment test using a freshwater amphipod (*Hyalella azteca*). Duration and endpoints were 28 days, 35 days, and 42 days for survival; 28 days and 42 days for growth; 35 days and 42 days for reproduction.
- Chronic 43-d sediment test using a freshwater midge (*Chironomus tentans*). Duration and endpoints were 20 days for survival and growth, and 23 days to 43 days for mortality and emergence.

In situ testing (including both sediment and water-only exposures):

- 48-h toxicity test using a freshwater cladoceran (*Daphnia magna*, 48 hours old). Endpoint: survival.
- 48-h and 10-d toxicity test using a freshwater midge (*Chironomus tentans*, 8 to 12 days post-hatch). Endpoint: survival.
- 48-h and 10-d toxicity test using a freshwater amphipod (*Hyalella azteca*, 7 to 14 days old). Endpoint: survival.

In situ Bioaccumulation:

- 48-h survival and 7-d bioaccumulation test using a freshwater oligochaete worm (*Lumbriculus variegatus*, multiple ages). Endpoint: survival and tissue bioaccumulation.

3.3.1.2 Results

Table 3.3-1 presents the results of the toxicity tests for most endpoints. Toxicological responses for each test type and treatment are also presented graphically (Figure 3.3-1 to Figure 3.3-11).

Summary of Site-Specific Toxicity Outcomes

In situ exposures (acute mortality endpoints):

- No to low toxicity at both reference locations (Locations A1, A3) and at the most upstream "contaminated" location with the lowest PCB concentration (Location 4).
- Toxicity was evident in multiple tests for the remaining three contaminated locations (Locations 5, 7, 8), with the magnitude of response generally greatest at the two locations with the highest PCB concentrations.
- Intermediate effects were observed in some water-column exposures, but the greatest effects were observed in the sediment-exposure treatments.

Laboratory exposures (chronic lethal and sublethal endpoints):

- Overall frequency of toxicity greater than for acute in situ exposures.
- Minor indications of responses (relative to negative control) in both reference locations (Locations A1, A3). These responses consisted mainly of marginal reductions in *Hyalella* reproduction and *Chironomus* survival and growth endpoints.
- Effects were much greater at all four contaminated locations (e.g., *Chironomus* toxicity, *Hyalella* mortality, and reproductive effects).
- Conclusion of "high" toxicity at all four contaminated locations.

Table 3.3-1

Results of Pairwise Statistical Tests Comparing Exposed Locations to Negative Control (T-Ctrl) and Reference (A1, A3) Sediment (Water-Only Exposures Excluded)

Location	4			5			7			8A			8		
Pairwise Comparison	**T-Ctrl**	**A1**	**A3**	**T-Ctrl**	**A1**	**A3**	**T-Ctrl**	**A1**	**A3**	**T-Ctrl**	**A1**	**A3**	**T-Ctrl**	**A1**	**A3**
28-d *Hyalella* survival	Yes	Yes	Yes	N/A	N/A	N/A	Yes	Yes	Yes	Yes	Yes	Yes	Yes	Yes	Yes
35-d *Hyalella* survival	Yes	Yes	Yes	N/A	N/A	N/A	Yes	Yes	Yes	Yes	Yes	Yes	Yes	Yes	Yes
42-d *Hyalella* survival	Yes	No	Yes	N/A	N/A	N/A	Yes	Yes	Yes	Yes	Yes	Yes	Yes	Yes	Yes
28-d *Hyalella* dry weight	No	No	No	N/A	N/A	N/A	NC	NC	NC	No	No	No	No	No	No
42-d *Hyalella* dry weight	No	No	No	N/A	N/A	N/A	NC	NC	NC	No	No	No	No	No	No
42-d *Hyalella* young per female	Yes	No	No	N/A	N/A	N/A	NC	NC	NC	Yes	No	Yes	Yes	Yes	Yes
42-d *Hyalella* mean young	Yes	No	Yes	N/A	N/A	N/A	NC	NC	NC	Yes	Yes	Yes	Yes	Yes	Yes
20-d *Chironomus* survival	Yes	Yes	Yes	N/A	N/A	N/A	Yes	Yes	Yes	Yes	Yes	Yes	Yes	Yes	Yes
43-d *Chironomus* emergence	Yes	Yes	Yes	N/A	N/A	N/A	Yes	Yes	Yes	Yes	Yes	Yes	Yes	Yes	Yes
20-d *Chironomus* dry weight	Yes	Yes	Yes	N/A	N/A	N/A	NC	NC	NC	NC	NC	NC	Yes	Yes	Yes
20-d *Chironomus* ash-free dry weight	Yes	Yes	Yes	N/A	N/A	N/A	NC	NC	NC	NC	NC	NC	Yes	Yes	Yes
48-h *Hyalella* survival (sediment)	No	No	No	No	No	No	No	Yes	No	N/A	N/A	N/A	No	No	No
10-d *Hyalella* survival (sediment)	No	No	No	No	No	No	No	Yes	Yes	N/A	N/A	N/A	No	Yes	Yes
48-h *Chironomus* survival (sediment)	No	No	No	No	No	No	No	No	No	N/A	N/A	N/A	No	No	No

Table 3.3-1

Results of Pairwise Statistical Tests Comparing Exposed Locations to Negative Control (T-Ctrl) and Reference (A1, A3) Sediment (Water-Only Exposures Excluded) (Continued)

Location	4			5			7			8A			8		
Pairwise Comparison	**T-Ctrl**	**A1**	**A3**	**T-Ctrl**	**A1**	**A3**	**T-Ctrl**	**A1**	**A3**	**T-Ctrl**	**A1**	**A3**	**T-Ctrl**	**A1**	**A3**
10-d *Chironomus* survival (sediment)	No	No	No	No	No	No	Yes	Yes	Yes	N/A	N/A	N/A	Yes	Yes	Yes
48-h *Daphnia* survival (sediment)	No	Yes	No	No	No	No	Yes	Yes	Yes	N/A	N/A	N/A	Yes	Yes	Yes
48-h *Lumbriculus* survival (sediment)	No	No	No	No	No	No	No	No	No	N/A	N/A	N/A	No	No	No

Yes = Statistically different at alpha = 0.05

No = Not statistically different at alpha = 0.05

NC = Sublethal endpoint not calculable (due to zero survival in treatment)

N/A = Not applicable; not tested for endpoint/location combination

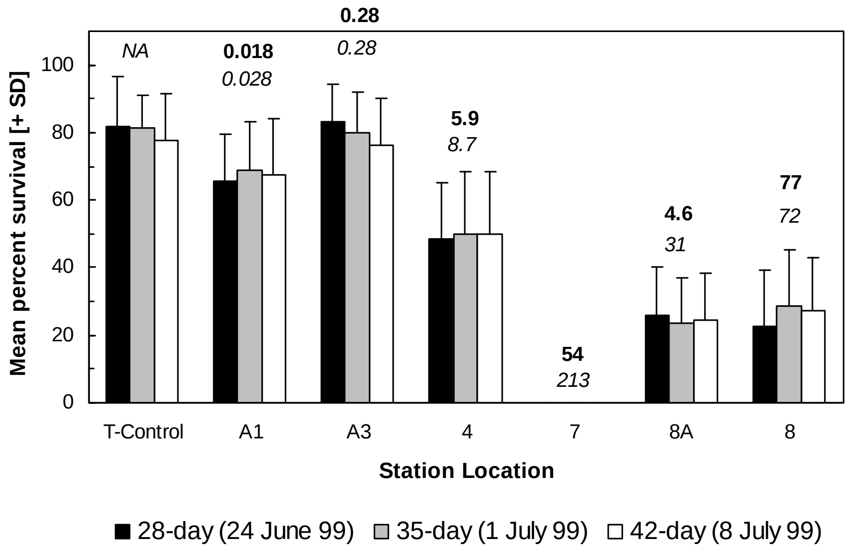


Figure 3.3-1 Survival of *Hyalella azteca* in Chronic Laboratory Toxicity Tests, at Three Time Periods (28 days, 35 days, 42 days)

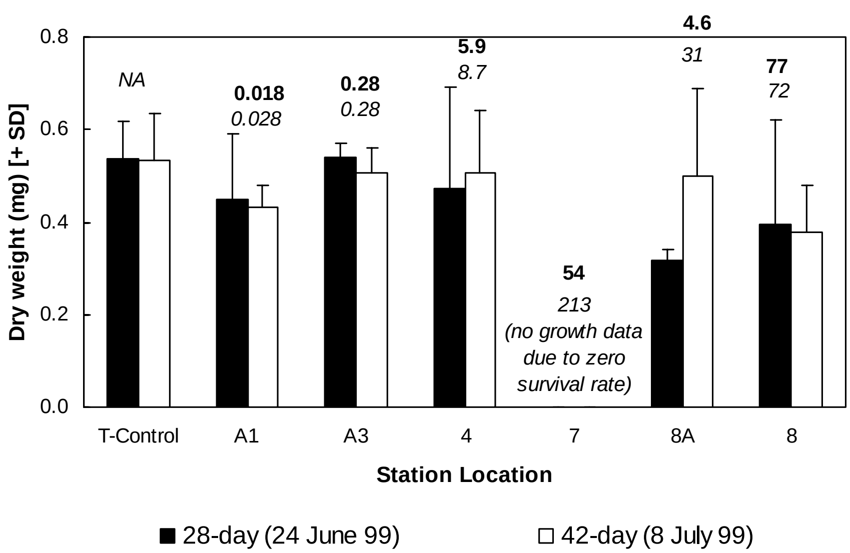


Figure 3.3-2 Growth of *Hyalella azteca* in Chronic Laboratory Toxicity Tests, at Two Time Periods (28 days, 42 days)

Notes: Labels represent tPCB concentration (mg/kg) in sediment. Values in **bold** represent median tPCB concentration (from all measurements made within 5 meters of location in 1999; see Appendix D). Values in *italics* represent "most synoptic" tPCB concentration; single concentration measured closest to toxicity test in space/time.

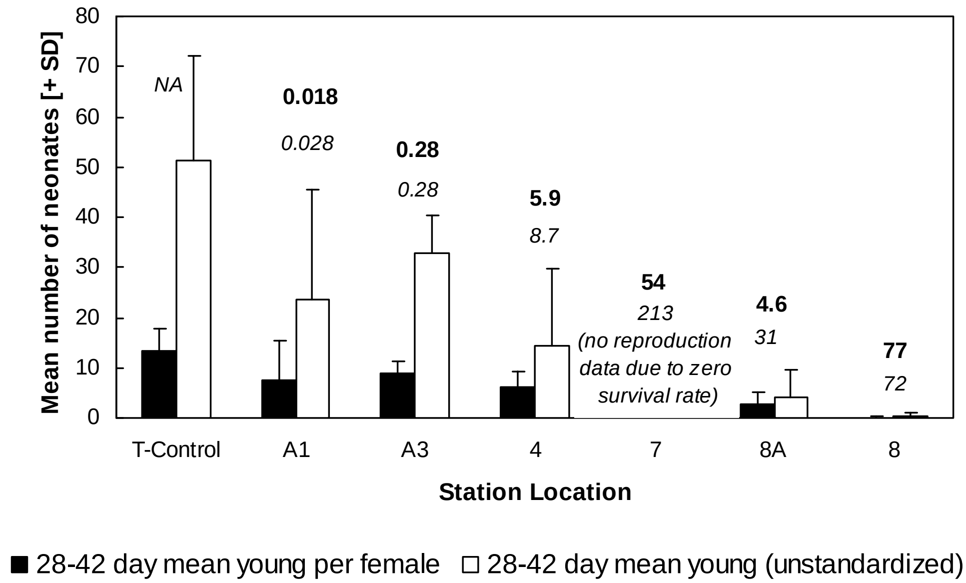


Figure 3.3-3 Reproduction of *Hyalella azteca* in Chronic Laboratory Toxicity Tests, Based on Mean Number of Young (Days 28-42)

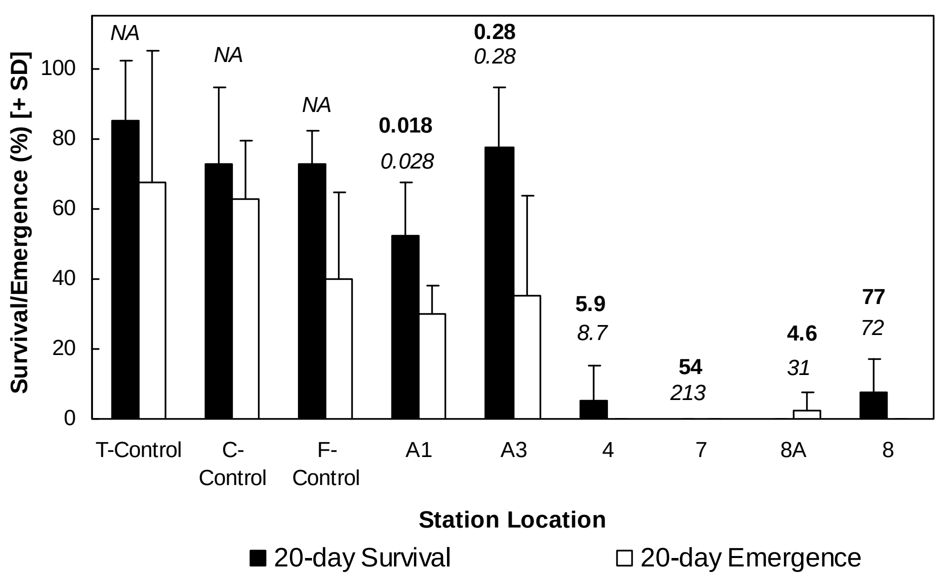


Figure 3.3-4 Survival and Emergence of *Chironomus tentans* in Chronic Laboratory Toxicity Tests (43 days)

Notes: Labels represent tPCB concentration (mg/kg) in sediment. Values in **bold** represent median tPCB concentration (from all measurements made within 5 meters of location in 1999; see Appendix D). Values in *italics* represent "most synoptic" tPCB concentration; single concentration measured closest to toxicity test in space/time. T-Control, C-Control, and F-Control are negative laboratory controls ("Trout Farm", "Cellulose", and "Florissant", respectively).

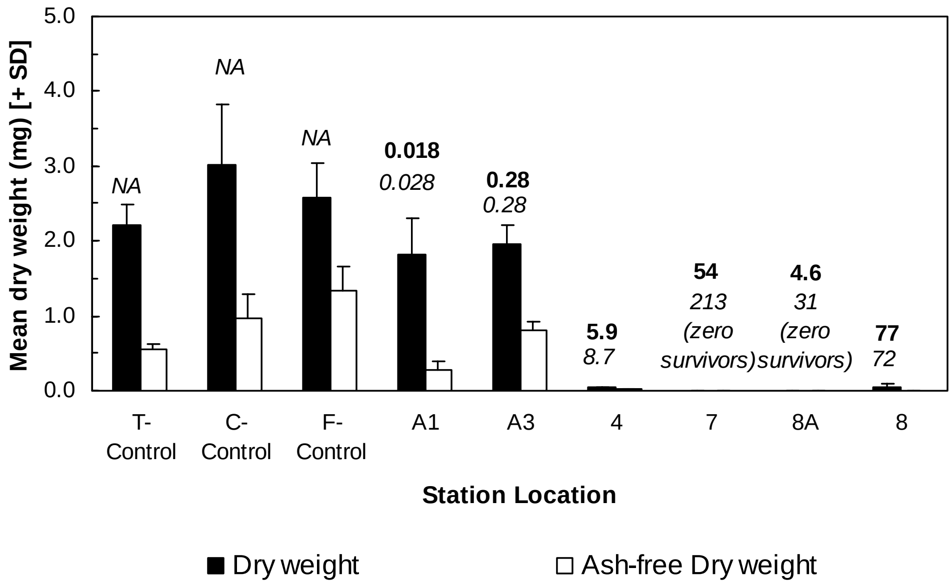


Figure 3.3-5 Growth Endpoints for *Chironomus tentans* in Chronic Laboratory Toxicity Test (20 days)

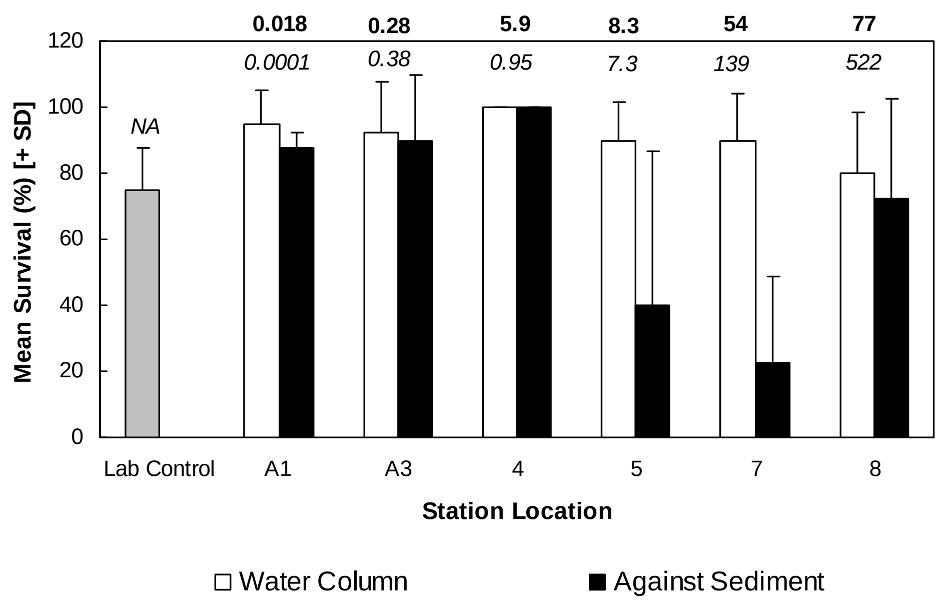


Figure 3.3-6 Survival of *Hyalella azteca* in 48-h Low Flow In Situ Toxicity Tests Conducted 14-16 June 1999

Notes: Labels represent tPCB concentration (mg/kg) in sediment. Values in **bold** represent median tPCB concentration (from all measurements made within 5 meters of location in 1999; see Appendix D). Values in *italics* represent "most synoptic" tPCB concentration; single concentration measured closest to toxicity test in space/time. T-Control, C-Control, and F-Control are negative laboratory controls ("Trout Farm", "Cellulose", and "Florissant", respectively).

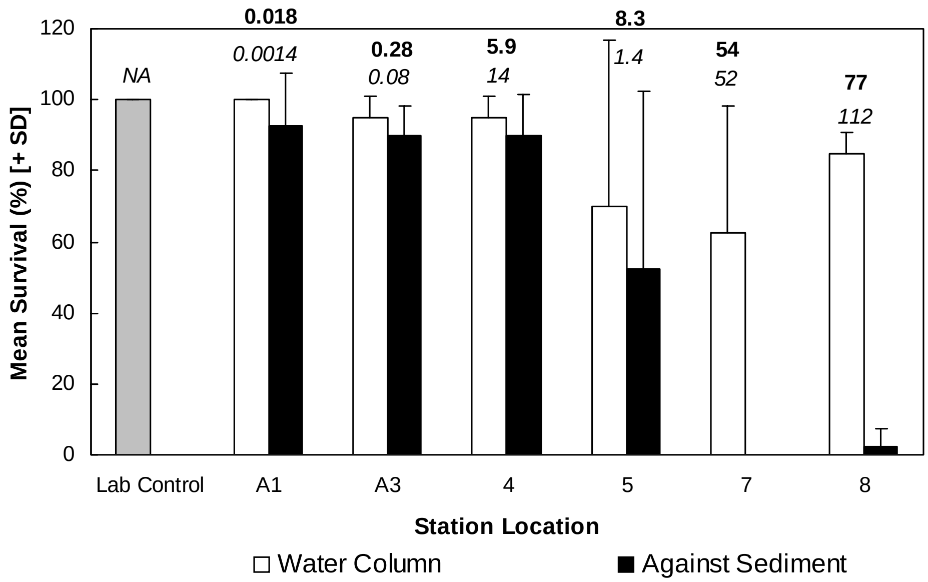


Figure 3.3-7 Survival of *Hyalella azteca* in 10-d Low Flow In Situ Toxicity Tests Conducted 17-27 June 1999

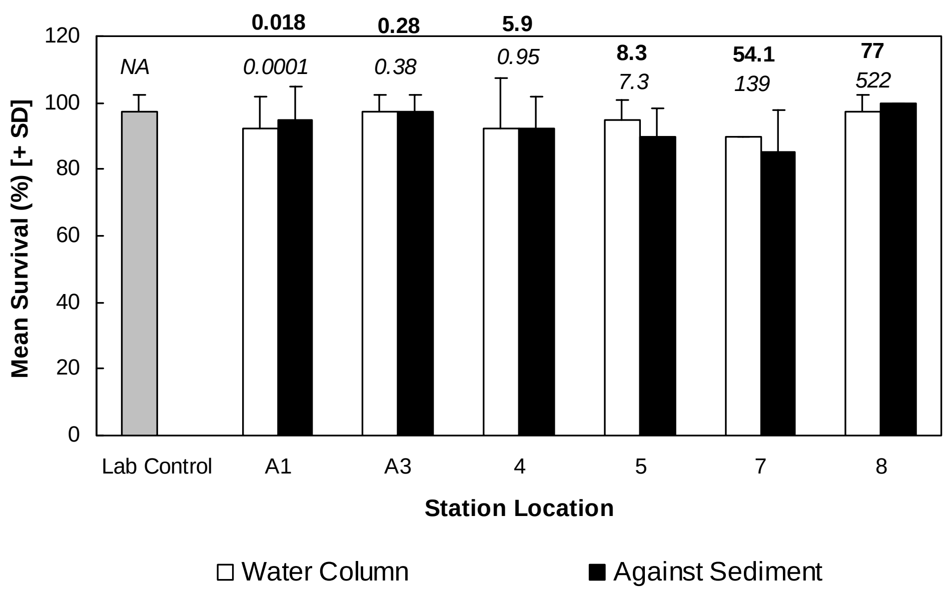


Figure 3.3-8 Survival of *Chironomus tentans* in 48-h Low Flow In Situ Toxicity Tests Conducted 14-16 June 1999

Notes: Labels represent tPCB concentration (mg/kg) in sediment. Values in **bold** represent median tPCB concentration (from all measurements made within 5 meters of location in 1999; see Appendix D). Values in *italics* represent "most synoptic" tPCB concentration; single concentration measured closest to toxicity test in space/time.

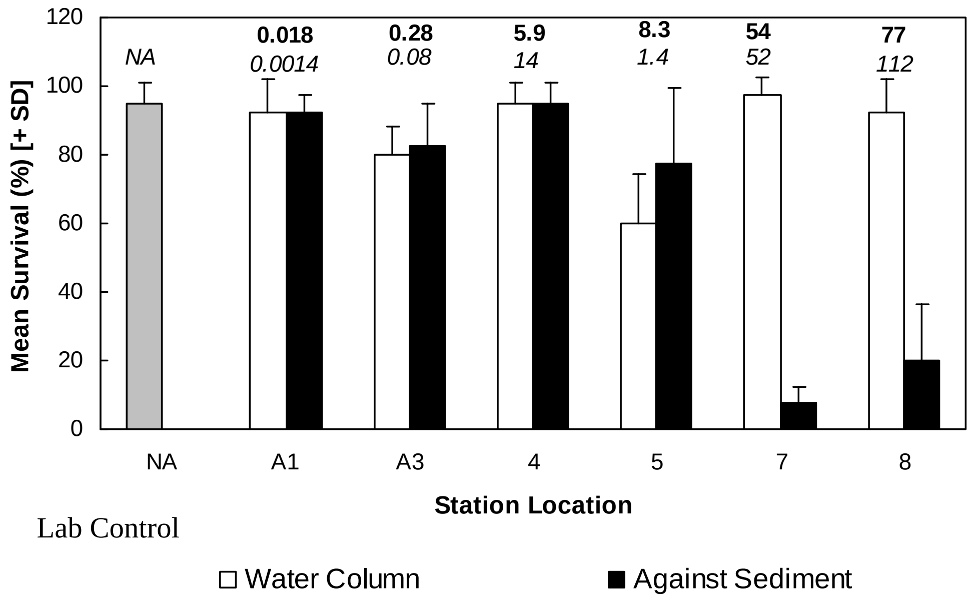


Figure 3.3-9 Survival of *Chironomus tentans* in 10-d Low Flow In Situ Toxicity Tests Conducted 17-27 June 1999

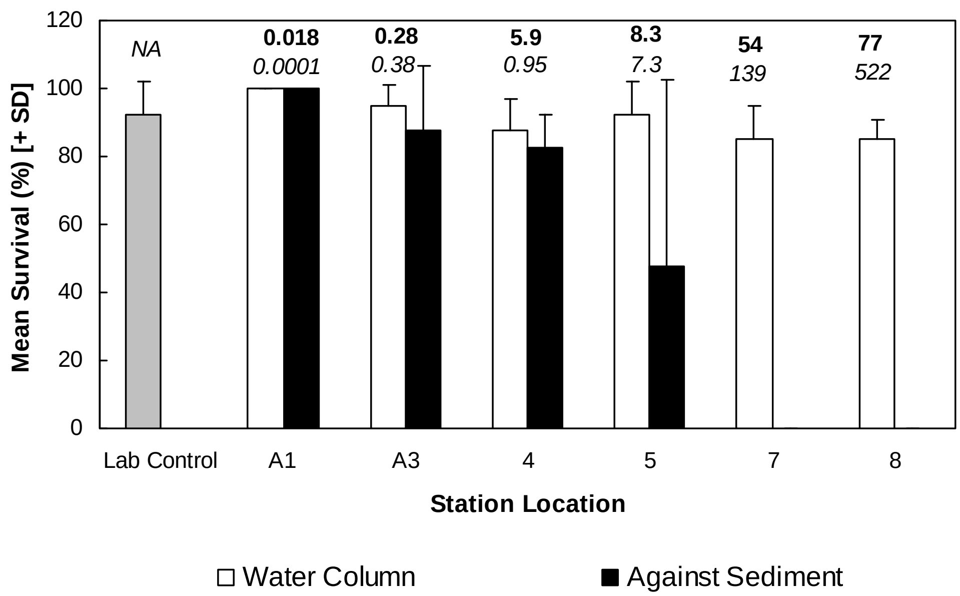


Figure 3.3-10 Survival of *Daphnia magna* in 48-h Low Flow In Situ Toxicity Tests Conducted 14-16 June 1999

Notes: Labels represent tPCB concentration (mg/kg) in sediment. Values in **bold** represent median tPCB concentration (from all measurements made within 5 meters of location in 1999; see Appendix D). Values in *italics* represent "most synoptic" tPCB concentration; single concentration measured closest to toxicity test in space/time.

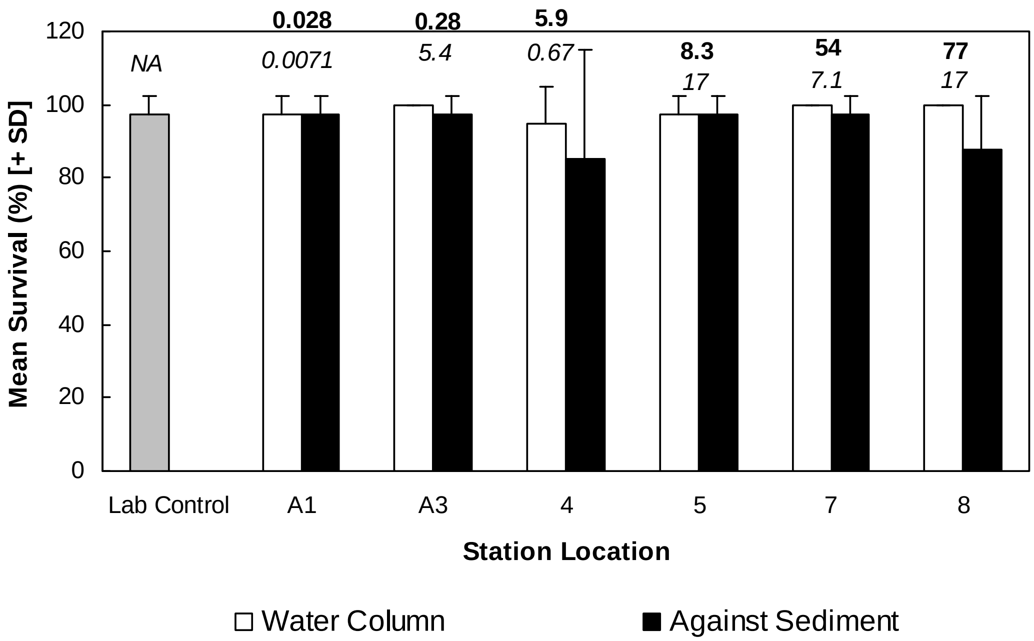


Figure 3.3-11 Survival of *Lumbriculus variegatus* in 48-h Low Flow In Situ Toxicity Tests Conducted 14-16 June 1999

Notes: Labels represent tPCB concentration (mg/kg) in sediment. Values in **bold** represent median tPCB concentration (from all measurements made within 5 meters of location in 1999; see Appendix D). Values in *italics* represent "most synoptic" tPCB concentration; single concentration measured closest to toxicity test in space/time.

A comparative approach (i.e., relative to reference) was used to distinguish background field reference responses from those observed at contaminated locations. In this assessment, comparisons were made not to the negative control, but to the two upstream reference locations (Locations A1, A3). Despite the small responses observed in the laboratory toxicity endpoints for Locations A1 and A3, the comparative assessment (Table 3.3-2) still indicated a moderate to strong incremental toxicity associated with contaminated PSA sediment. The rationale for assigning the overall toxicity ratings for each location is described in Appendix D (Section D.3.1.6).

The three most downstream locations (7, 8, and 8A) consistently demonstrated toxicity for numerous endpoints.

3.3.2 Dose-Response Analysis – Toxicity Test Endpoints

The relationship between toxicity test endpoints and COC concentrations was evaluated statistically. The assessment focused on the relationship between PCBs and toxicity endpoints because other lines of evidence indicated a high probability that PCBs were a causal agent for toxicity to benthic invertebrates. However, the other COCs were also considered.

This section emphasizes dose-response using the most synoptic sediment PCB exposure concentration at each location. An alternative analysis, using the median exposure concentration is presented in Attachment D.5. Generally, the two approaches yield comparable results (i.e., most endpoints within a factor of 2); the “median” analysis yielded effects thresholds that were slightly lower than the “most synoptic” analysis.

Methods for Evaluating Dose-Response for Toxicity Data

- Individual Endpoint Analysis – Each toxicity endpoint was investigated individually using conventional descriptive statistics that related degree of effect to PCB concentrations (e.g., LC_{50}, IC_{20}, NOAEL, LOAEL).
- Combined Endpoint Analysis – The toxicological endpoints were integrated using a general linear modeling approach to identify similarities and differences in dose-response relationships across species and endpoints.

Table 3.3-2

Summary of Housatonic River Sediment Toxicity, Relative to Reference Responses

Sampling Location (ID, Location, WESTON ID)			Chronic Laboratory Endpoints (>20 days duration)		Acute In Situ Endpoints (≤10 days duration)			Overall Assessment
			H. azteca Laboratory (Survival, Growth, Reproduction)	*C. tentans* Laboratory (Survival, Emergence, Growth)	*H. azteca* In situ Survival (Water, Sediment)	*C. tentans* In situ Survival (Water, Sediment)	*D. magna* In situ Survival (Water, Sediment)	
4	1.5 miles below Holmes Road	019	◉ - ○ - ◉	● - ● - ●	○ - ○	○ - ○	○ - ○	◉
5	Near WWTP Discharge	428	NA - NA - NA	NA - NA - NA	◉ - ◉	○ - ○	○ - ◉	◉
7	2 miles below New Lenox Road	389	● - NA - NA	● - ● - ●	◉ - ●	○ - ●	○ - ●	●
8A	½ mile above Woods Pond	023	● - ◉ - ●	● - ● - ●	NA - NA	NA - NA	NA - NA	●
8	½ mile above Woods Pond	031	● - ◉ - ●	● - ● - ●	○ - ●	◉ - ●	○ - ●	●

○ = Negligible to low toxicity: less than 20% effect size relative to upstream background (A1, A3). Overall assessment – negligible indication of ecological risk.

◉ = Moderate toxicity: 20 to 50% effect size relative to upstream background (A1, A3). Overall assessment – ecological effects possible, but not conclusive.

● = High toxicity; greater than 50% effect size relative to upstream background (A1, A3). Overall assessment – strong indication of ecological effects.

NA = Endpoint not measured due to complete mortality in treatment (Location 7), or samples not collected at location (Locations 5, 8A).

3.3.2.1 Approach 1: Calculation of Individual Toxicity Test Endpoints

Point estimates were calculated for each toxicity test, including LC_{20} and LC_{50} values for survival endpoints, and IC_{20} and IC_{50} values for sublethal response endpoints (e.g., growth, reproductive success). For each data set, test endpoints were calculated based on comparison to both negative controls and reference sediment (Locations A1, A3). Full statistics are presented in Appendix D; a graphical presentation of toxicity thresholds relative to reference Location A1 is depicted in Figure 3.3-12.

Although there are small differences in the toxicity threshold values calculated using different statistical methods (i.e., choice of extrapolation model or choice of reference sediment), the data indicate an increase in the frequency and magnitude of adverse biological responses with increasing sediment tPCB concentration. The following ranges of tPCB concentrations and associated responses were developed, based on comparisons of contaminated location responses to reference locations:

- <3 mg/kg – Some sensitive endpoints exhibited apparent responses, but the magnitude of responses was not large. These subtle responses were difficult to evaluate precisely due to statistical power limitations, caused in part by the high variability in some treatments.
- 3 to 10 mg/kg – Numerous endpoints indicated ecologically significant responses, with many LC_{50}/EC_{50} values falling in this range. Statistically significant responses were observed in most *Hyalella* and *Chironomus* life-cycle endpoints at PCB concentrations below 8.7 mg/kg.
- 10 to 30 mg/kg – Nearly all toxicity endpoints indicated large (>50%) responses relative to reference locations. The only endpoints that did not exhibit large responses in this concentration range were either growth endpoints or were short-term (48-h) tests and/or with tolerant species.
- >30 mg/kg – The dose-response analyses indicated that most survival and reproduction endpoints exhibited very large reductions at these concentrations, with complete mortality of some species.

Dose/response modeling was also conducted using individual chemistry values based on median sediment PCB concentrations (Attachment D.5). These tests had results similar to those presented above (i.e., most LC_{50}/EC_{50} values were in the 3 to 30 mg/kg tPCBs range).

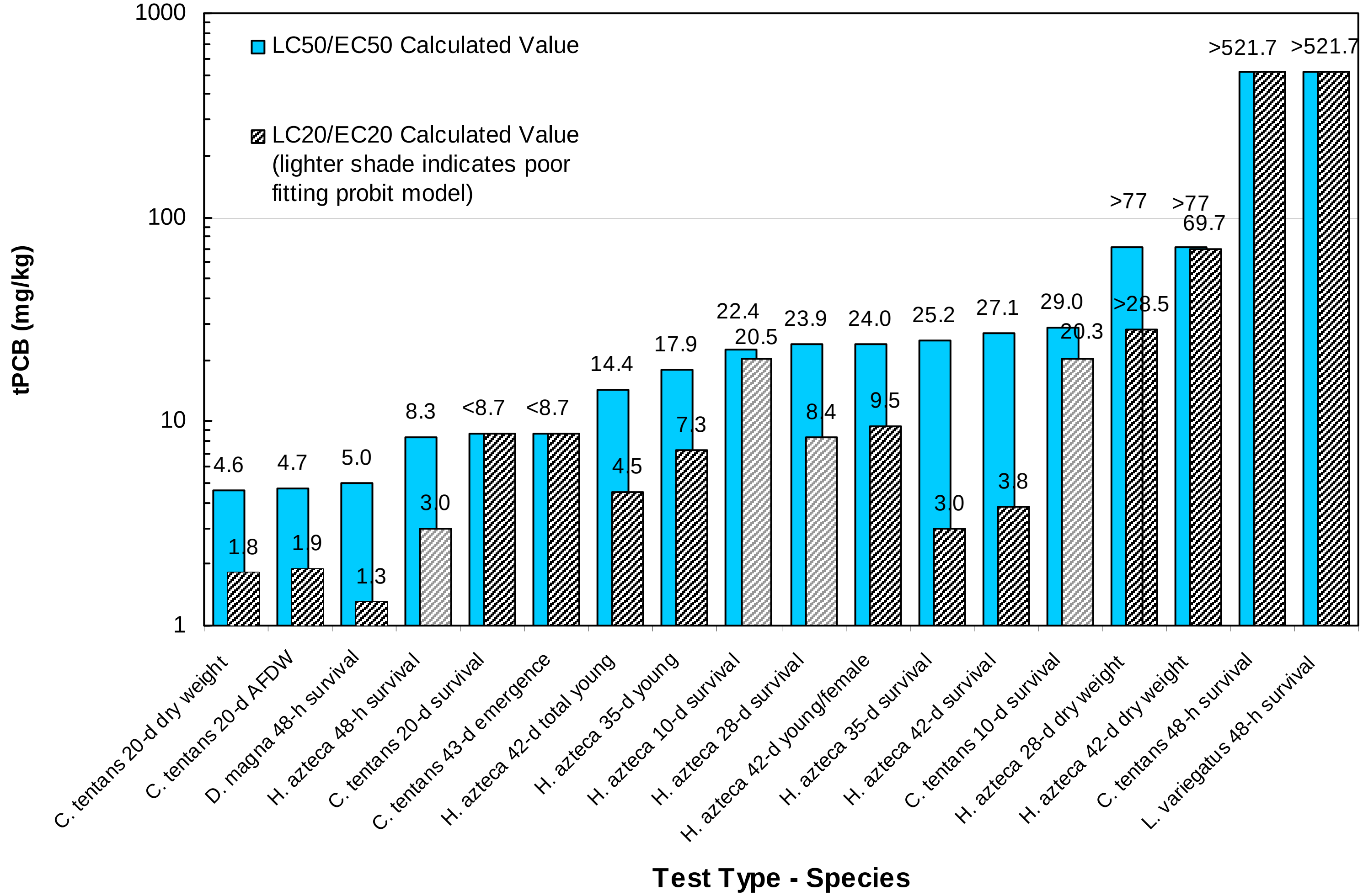


Figure 3.3-12 Statistical Endpoints for Toxicity Data, with Comparisons to Location A1 (sorted by LC_{50}/EC_{50} value): Calculations Made Using "Most Synoptic" Exposure Data Set Only

Threshold effects concentrations were calculated using the individual endpoint data, in order to allow derivation of site-specific hazard quotients in the Risk Characterization, and to support the development of maximum acceptable threshold concentrations (MATCs). The following decision rules were applied to develop site-specific sediment PCB thresholds based on the integrated toxicity test data:

- Separate acute and chronic MATCs – Acute and chronic endpoints were considered separately, and separate toxicity thresholds were derived for each type. The in situ exposures were considered acute, because they were no more than 10 days in duration. The laboratory exposures (including survival, growth, and reproduction endpoints) were considered chronic, because they were over 40 days in duration and covered the reproductive period (i.e., life cycle tests).

- Eliminate redundant endpoints – Some toxicological endpoints include multiple measurements of the same type of effect for the same test species (e.g., dry-weight vs. ash-free dry-weight). Therefore, only a single value was used for each effect type for each test species.

- Use "most synoptic data" – Threshold derivations emphasized dose-response identified using only the "most synoptic" exposures for the toxicity tests.

- Use only tests endpoints that exhibit dose-response relationships – The threshold derivation considered only test endpoints that demonstrate acceptable dose-response relationships. Criteria for inclusion of an endpoint in threshold derivation for benthic invertebrates are described in Section D.3.2.1.4.

- Comparison to reference – A derived threshold must be equal to or greater than the concentration at reference locations, provided the reference concentration represents a true reference condition. Also, comparisons to field references (rather than negative controls) were applied for derivation of sediment threshold values, since field references account for physicochemical factors that may mediate sediment toxicity.

An objective of the threshold derivation was to identify PCB concentrations corresponding to "low," "intermediate," and "high" risk, consistent with risk definitions used elsewhere in the ERA. For the wildlife receptors in the Housatonic River ERA, the effect sizes corresponding to intermediate and high risk were 10% and 20%, respectively. For aquatic receptors, larger threshold effect sizes of 20% and 50% were applied, for the following reasons:

- The reproductive strategies and life histories of aquatic invertebrate populations make them less vulnerable to small effects on reproduction or survival. Aquatic species are more likely to have compensatory processes (such as benthic drift) to offset localized impacts, and have larger numbers of breeding individuals per area, each with a large number of eggs produced per female.

- The statistical power of the invertebrate toxicity tests was not adequate to identify effect sizes as small as 10%, given the number of test replicates and the background variation in response.

The lower chronic toxicity threshold, corresponding to the transition between low and intermediate risk, was therefore based on a 20% effect size in the life-cycle invertebrate toxicity tests. Because there are only two species (*Hyalella, Chironomus*) upon which to base a chronic toxicity threshold, it was not possible to develop a species sensitivity distribution. Therefore, the most sensitive endpoint from these two species was used to develop the sediment PCB threshold concentration. The lower of the values from the two test organisms (i.e., from the *Chironomus* test) were selected as the EC thresholds for chronic toxicity (EC_{20} = 2.0 mg/kg; EC_{50} = 4.7 mg/kg).

Threshold PCB Concentrations Derived from Chronic Toxicity Data

- Chronic Toxicity (Intermediate Risk Threshold) – 2.0 mg/kg tPCB, based on impaired growth (EC_{20}) of *Chironomus*. This value is supported by the chronic *Hyalella* survival endpoint, which yielded a 20% response at 3.1 mg/kg tPCB.
- Chronic Toxicity (High Risk Threshold) – 4.7 mg/kg tPCB, based on impaired growth (EC_{50}) of *Chironomus*. This value is supported by the *Hyalella* life cycle test, which indicated a 20% response to survival and reproduction (both young and young per female) at this approximate concentration, in addition to reduced survival.

Separate thresholds were developed to represent the level at which acute toxicity was observed in multiple species in the in situ tests. Acute toxicity thresholds were developed for both intermediate and high risk levels, based on LC_{20} and LC_{50} values, respectively. The following decisions were applied in acute threshold derivation:

- The 48-h *Hyalella* survival endpoint was removed, because this endpoint did not meet the criteria for acceptable dose-response (i.e., anomalous dose-response shape was observed).
- The 48-h *Chironomus* survival endpoint was removed, because the endpoint did not display acceptable dose-response, and because a longer duration test (10-d) was available that exhibited acceptable dose-response.

- The 48-h *Lumbriculus* survival endpoint was not considered because the test duration was insufficient for this organism to achieve maximum PCB bioaccumulation or toxicity, and because the endpoint did not display acceptable dose-response.

- LC_{50} values, using the Trimmed Spearman Karber (TSK) method, from the remaining endpoints (10-d *Hyalella* survival, 10-d *Chironomus* survival, and 48-h *Daphnia* survival) were considered.

- LC_{20} values, using the Probit method, from the remaining endpoints (10-d *Hyalella* survival, 10-d *Chironomus* survival, and 48-h *Daphnia* survival) were considered.

- The average LC_{20} value and the average LC_{50} value were each derived from comparisons to both A1 and A3; the geometric mean of the three values was used as the acute toxicity threshold PCB value.

The derived thresholds for acute responses were 10.0 mg/kg tPCB (based on EC_{20} values) and 17.5 mg/kg tPCB (based on EC_{50} values). These sediment concentrations both represent a relatively high level of response, because they incorporate acute exposure durations, a lethal test endpoint, and are based on an average of multiple affected species, rather than the most sensitive species.

Threshold PCB Concentrations Derived from Acute Toxicity Data

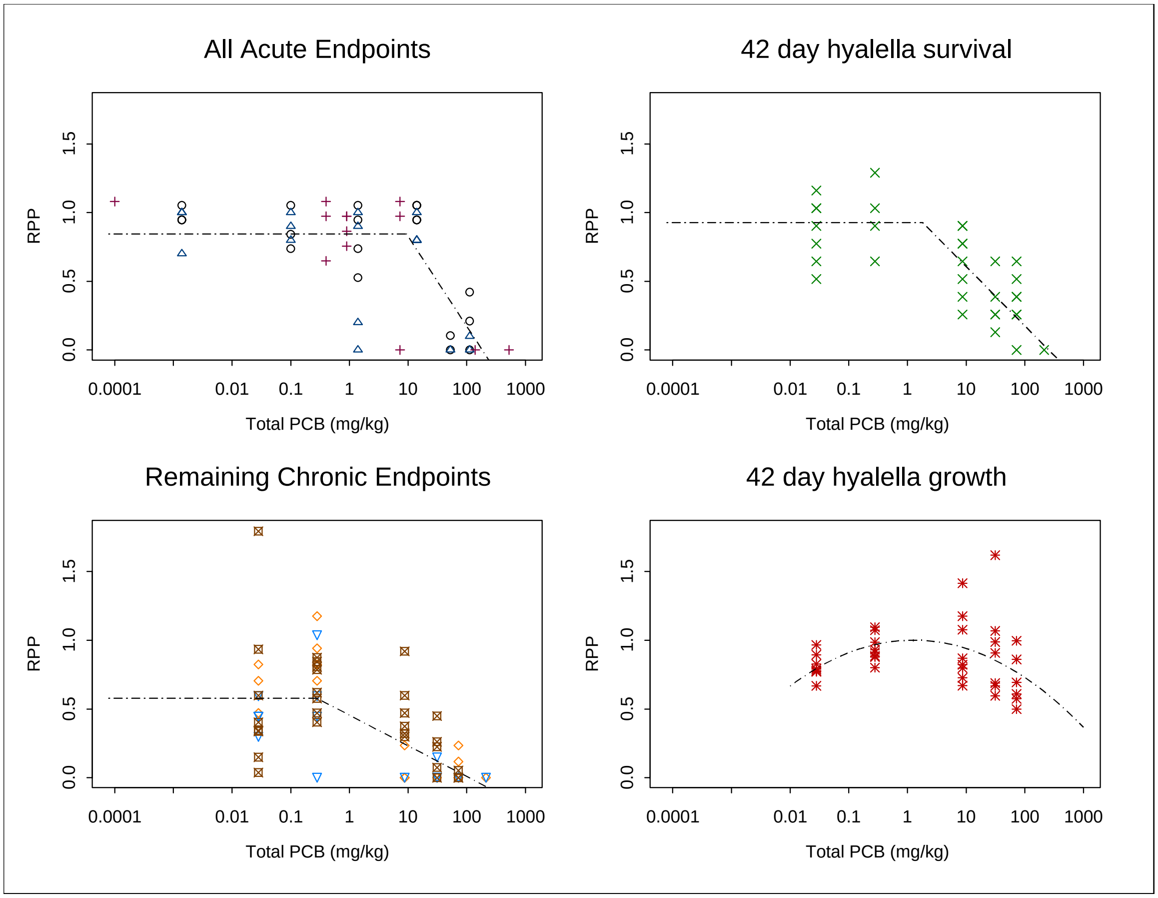
- Acute Toxicity (Intermediate Risk Threshold) – 10 mg/kg, based on the geometric mean LC_{20} (Probit method) from 10-d *Hyalella*, 10-d *Chironomus*, and 48-h *Daphnia* in situ exposures.
- Acute Toxicity (High Risk Threshold) – 17.5 mg/kg, based on the geometric mean LC_{50} (Trimmed Spearman Karber method) from 10-d *Hyalella*, 10-d *Chironomus*, and 48-h *Daphnia* in situ exposures.

3.3.2.2 Approach 2: General Linear Model of Dose-Response

The assessment of individual endpoints is sensitive to test variability, which can mask broader trends in toxicity of PCBs. Therefore, a supplemental approach was applied that combined the toxicity results from various endpoints to identify the overall trend(s) in dose-response observed. The endpoints for all toxicity tests were standardized so that the response variables were equivalent (i.e., responses represented the proportion of their control mean response). This transformation of all endpoints to the relative performance proportion (RPP) values standardized results from different toxicological endpoints to similar ranges and facilitated the search for a single unified model among all endpoints. The results of the general linear modeling are depicted in Figure 3.3-13. Overall, the linear modeling indicated that seven of eight toxicity endpoints evaluated were significantly correlated with log-transformed PCB concentration. Differences between acute endpoints and chronic endpoints were observed; these are likely related to the greater sensitivity of chronic endpoints in toxicity tests. The modeling procedure enabled the identification of threshold tPCB concentrations in sediment; these findings are in agreement with the summary of individual test endpoints provided in Section 3.3.2.1. In summary, sediment tPCB concentrations above 3 mg/kg indicate significant adverse effects for sensitive (chronic) endpoints, and tPCB concentrations in the 10 to 30 mg/kg tPCBs range may result in acute mortality to multiple organisms.

3.3.2.3 Relationships of Effects with Other COCs

The data for other COCs were also evaluated qualitatively to assess whether these contaminants were likely to have contributed to the toxicity observed. The spatial patterns in COC concentrations were compared against the spatial pattern in toxicity. Other COCs (other than tPCBs) do not explain the patterns of toxicity observed in the in situ toxicity tests. One possible exception is for dioxins and furans, which correlate strongly with tPCBs (i.e., co-contaminants in PCB mixtures).



Notes: "Remaining" chronic endpoints include 20-d *Chironomus survival*, 43-d *Chironomus* emergence, and 42-d Hyalella reproduction (young-per-female).

Calculations made using "most synoptic" exposure data set only.

Figure 3.3-13 Segmented Linear Regression Models Applied to Toxicity Data, Relating Relative Performance Proportion (RPP) to Bulk Sediment tPCB Concentrations (mg/kg)

Comparison of Other COC Trends to Toxicity Trends

- PAHs – Most PAH data show a reduction in PAH concentrations with distance downstream that is the reverse of the pattern of toxic responses. Therefore, with the possible exception of Location 7, there is no evidence that PAHs were a major contributor to sediment toxicity.
- Dioxins/Furans – These contaminants exhibited a spatial distribution similar to the pattern of toxicity. This is likely due to co-occurrence between PCBs and dioxins/furans in environmental samples.
- Metals – Metals generally exhibited a pattern of increasing concentration with distance downstream, which matched the pattern of toxic responses. However, the metals concentrations varied with TOC and particle size distributions. When metals concentrations were normalized to the substrate differences (thus accounting for lower bioavailability in downstream areas), there was no indication that metals concentrations were responsible for observed effects. This was confirmed by the low hazard quotients (HQs) for these metals, and the results of the TIE.

3.3.3 Toxicity Identification Evaluations (TIEs)

Wright State University (EVS 2003) conducted TIE testing in 1999 to identify classes of contaminants causing toxic responses. The daphnid (*Ceriodaphnia dubia*) was exposed to porewater from Housatonic River sediment for 48 hours. Several TIE treatments were initiated, including baseline tests, oxidant reduction addition tests, EDTA chelation addition, pH-adjusted filtration, pH-adjusted aeration, and pH-adjusted C_{18} solid phase extraction (SPE). None of the individual treatments provided a definitive identification of the toxic agent; however, integration of the results of various treatments provided strong indications that non-polar organic compounds (most likely PCBs) were responsible for the toxicity.

The indications of PCB toxicity from the Phase I TIE are consistent with the magnitude of exceedances of sediment quality values (SQVs) and water quality criteria for COCs observed in site media.

Rationale for Implication of Non-Polar Organic Compounds as Toxic Agent in TIE Treatments

- Significant reduction in toxicity in the pH-adjusted/filtration treatments – Higher survival in the filtration test was attributed to organic colloids in the samples being filtered out and/or pH-mediated toxicity alteration of organic compounds. Filterable compounds can include non-polar organics, such as PAHs, PCBs, and some metals.
- Significant reduction in toxicity in the pH-adjusted C_{18} SPE treatments – The results of these treatments indicated that the filtration reduced the toxicity of the original samples. Therefore, this test implicated non-polar organics, pesticides, and/or some metals.
- EDTA treatments – These treatments did not result in a reduction of toxicity. This provides evidence against metals as the dominant causal agent.
- Sediment and porewater chemistry – PCB concentrations in TIE treatments were observed to be well above upper-bound sediment quality guidelines and water quality criteria applied to porewater. Conversely, PAH concentrations in these TIE treatments were below applicable benchmarks (i.e., Swartz [1999] sediment quality guidelines for total PAHs).
- The two samples demonstrated to be the most toxic in the initial 24-h screening toxicity test had the highest sediment tPCB concentrations.

3.3.4 Tissue PCB Effects Thresholds

PCB tissue concentrations associated with lethal or sublethal effects in aquatic invertebrates were identified from the literature to estimate threshold tissue concentrations beyond which adverse effects might occur. The review focused on data for Aroclors 1260 and 1254, in addition to tPCBs. No studies conducted specifically for Aroclor 1260 were identified, and limited information was found for tPCBs and Aroclor 1254. Because tissue effects data were limited, both freshwater and marine/estuarine invertebrate species were considered in the review, and only a subset of studies deemed appropriate for threshold derivation (screening rationale provided in Appendix D) were retained.

Figure 3.3-14 presents the tissue effects data in order of increasing tissue concentration to illustrate studies where effects did and did not occur. The majority of data were for mortality; however, there were also data for growth, development, behavior, physiological, and cellular endpoints.

Based on these data, it appears that adverse effects are unlikely at tissue concentrations at or below 3 mg/kg, are likely to occur to sensitive organisms above 10 mg/kg, and that there is some uncertainty regarding the probability of effects between these concentrations.

3.3.5 Sediment Quality Values (SQVs)

Numerical sediment quality guidelines (SQGs) for freshwater ecosystems have been developed using a variety of approaches. Each approach has certain advantages and limitations which influence their application in the sediment quality assessment process (MacDonald et al. 2000). Recognizing the limitations, SQVs were used in the benthic invertebrate ERA as an additional line of evidence, rather than as a conclusive statement regarding the toxicity of COCs in Housatonic River sediment. A summary of the values used in the ERA is provided in Appendix D (Table D.3-11).

3.3.6 Benthic Macroinvertebrate Community Evaluation

3.3.6.1 Methods

Multiple metrics and statistical approaches were considered in the evaluation of benthic community data.

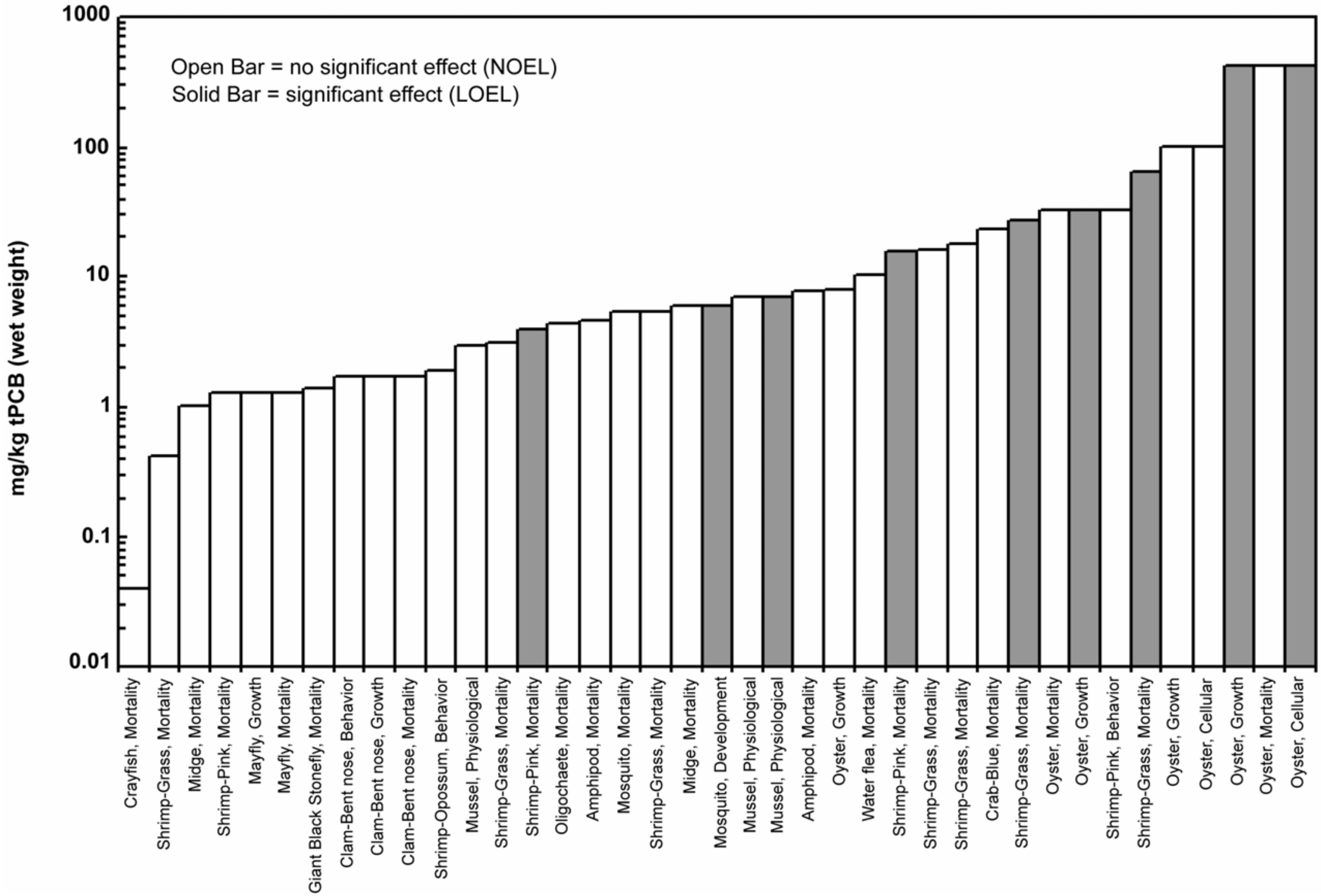


Figure 3.3-14 Combined Effects and No-Effects Levels for PCB Concentrations (mg/kg wet weight) in Benthic Invertebrate Tissue Samples – tPCBs and Aroclor 1254

Statistical Approaches Applied in Benthic Macroinvertebrate Community Assessment

Comparison of benthic assemblages between contaminated locations and reference locations. Tools used to make these comparisons included:

- Average rank plots, combining relevant summary metrics in a nonparametric multivariate approach.
- Multidimensional scaling (MDS) plots, using the same summary metrics in a parametric multivariate approach.
- Univariate tests using selected summary metrics.
- Analysis of the relationship between sediment COC concentrations and benthic community structure indices, using a regression/correlation approach.
- Multiple regression analyses (of individual taxa abundances and community indices) combining contaminant and habitat variables as independent variables.
- Species sensitivity distributions, integrating effect size and percentage of taxa affected in the benthic community.

Using the screening rationale provided in Appendix D (Attachment D.3), six benthic community metrics were included in multivariate statistical analyses.

Benthic Community Metrics Evaluated

Multivariate Assessment:

- Organism abundance (number of animals per replicate or location).
- Taxonomic richness (number of unique taxa per replicate or location).
- "EPT" relative abundance (mayflies, stoneflies, caddisflies).
- Relative abundance of tolerant dipterans.
- Relative abundance of tolerant oligochaetes.
- Relative abundance of tolerant gastropods.

Univariate Assessment:

- Total abundance.
- Biomass.
- Abundance of individual taxa, evaluated using a species sensitivity distribution.
- Taxonomic richness.
- Taxonomic diversity (Shannon-Wiener H', Simpson's Index, modified Simpson's Index).
- Modified Hilsenhoff Biotic Index (MHBI).

3.3.6.2 Results

A summary of the benthic macroinvertebrate community results is presented below. Overall, the evaluation indicated a high degree of variability, both within and among locations. Despite within-location variability, some significant differences between locations were observed that were consistent across the metrics. Specifically, for most metrics, the coarse-grained contaminated locations exhibited impaired benthic communities relative to the three coarse-grained reference locations; impairment was most pronounced at Locations 3 through 5. In fine-grained locations, no impairment to total abundance or richness metrics was observed; however, impairment was identified using diversity measures and using a species sensitivity distribution approach. No habitat differences were identified that would explain the differences in benthic assemblages observed in either coarse- or fine-grained substrates.

The results of the benthic community evaluation are summarized as follows:

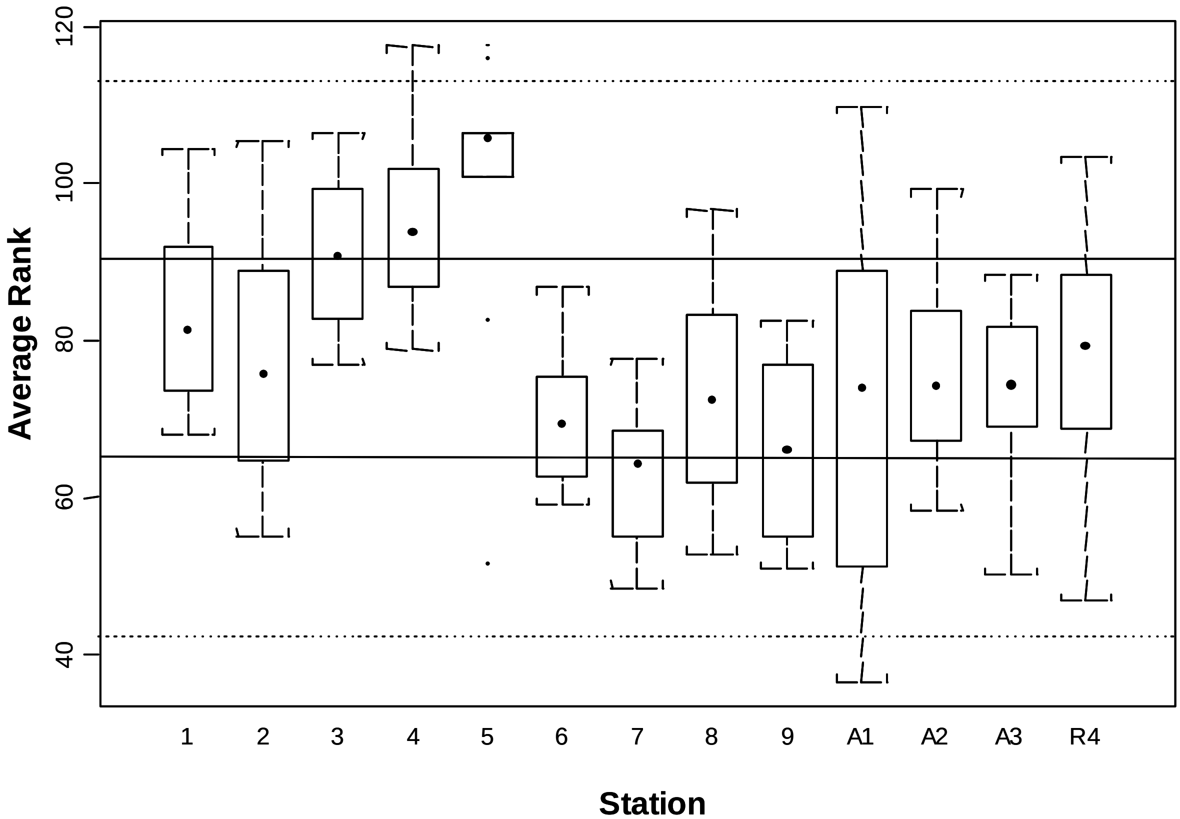
- Average Rank Plots (Figure 3.3-15) – Median ranks at Locations 3, 4, and 5 were significantly higher than all reference locations, indicating degraded conditions for the six metrics evaluated (Section D.3.7.2). Although the median ranks for Locations 1 and 2 were higher than at the coarse-grained reference sites, these differences were not statistically significant. Fine-grained locations did not differ significantly from reference locations.

- Multidimensional Scaling (Figure 3.3-16) – In the MDS plot, all coarse-grained contaminated (C/C) locations (Locations 1 through 5) were set apart from the remaining locations, suggesting community alteration (Section D.3.7.3 and D.3.7.4). Locations 1 and 2 indicated benthic communities that were different but not consistently degraded relative to the coarse-grained reference sites. The MDS analysis did not indicate benthic community alteration at the fine-grained locations.

- ANOVA (Total Abundance) – The analysis indicated that all five coarse-grained contaminated locations had significantly lower total abundances than coarse-grained reference locations (Section D.3.7.5). There was no indication of impairment in fine-grained sediment relative to reference, however.

- ANOVA (Taxa Richness) – In coarse-grained sediment, all five contaminated locations yielded significantly lower taxa richness relative to references; differences were somewhat more pronounced for Locations 3 through 5, compared to Locations 1 and 2 (Section D.3.7.5). No evidence of ecological disruption in the fine-grained sediment was observed.

- ANOVA (Taxa Diversity) – In coarse-grained sediment, the ANOVA of Shannon-Wiener H’ and Simpson’s Index indicated that all C/C locations were significantly different from the pooled reference Locations A1 and A3 (Section D.3.7.5). Only Location 5 was significantly different from reference Location A2. In fine-grained sediment, the ANOVAs of Shannon-Wiener H’, Simpson’s Index, and Simpson’s Modified Index all indicated that Locations 6, 8, and 9 differed significantly from the reference Location R4 ($p < 0.05$).



Notes: The box for each location is the interquartile range (1st through 3rd quartiles), and the whiskers are the entire range.

The median of the average ranks is shown as a black dot within the box.

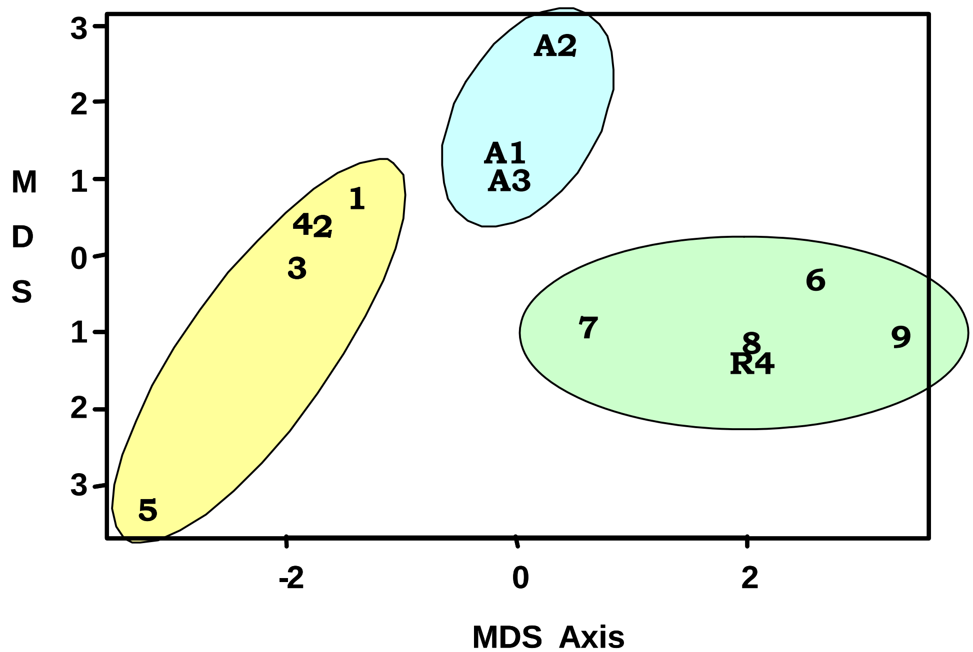
Statistical definitions of symbols are provided in Appendix D.

Locations with high average ranks indicate unfavorable conditions for the six benthic community metrics assessed (e.g., low taxonomic richness, high percentage of tolerant taxa).

_________ 25th and 97.5th Percentiles of the appropriate Random Median Distribution.

.... 25th and 97.5th Percentiles of the Random Replicate Average Results.

Figure 3.3-15 Average Ranks Analysis for Six Benthic Community Metrics, with Equal Weighting Assigned to Each Metric



Note: Shaded ovals group locations based on habitat and PCB exposure conditions:

Yellow = Coarse-grained contaminated (C/C) locations;

Blue = Coarse-grained reference (C/R) locations;

Green = Fine grained locations (F/C and F/R).

Figure 3.3-16 Multidimensional Scaling for Benthic Community Health Metrics, Showing Metric Medians on MDS Plot

- MHBI Metric – There was no compelling evidence of incremental habitat degradation due to PCBs at any of the contaminated locations using the MHBI metric (Section D.3.7.6). However, the appropriateness of this metric was questionable for the study area because the MHBI was not developed to address effects of PCBs, and because the reference locations indicated a high "background" proportion of pollution-tolerant taxa.

- Multiple Regression Analyses – Both EPA and GE conducted multiple regression analyses to attempt to separate the effects of substrate characteristics from contaminant effects (Section D.3.7.7). In this approach, the substrate influences (in addition to PCB concentrations) were considered quantitatively in additive linear models. The results of the multivariate regression analysis demonstrate that the abundance of certain taxa, notably chironomids, declines with increasing PCB concentrations, regardless of substrate type. In contrast, other taxa, such as snails, sphaeriid clams, and oligochaete worms, appear to tolerate higher substrate concentrations of PCBs. The GE analysis indicated that physical-chemical parameters account for a statistically significant proportion of the variability in broad community metrics, but also indicated that the fraction of variability that is explained by physical-chemical parameters is low. In these significant regressions, these physical-chemical parameters, taken together, accounted for 9.6 to 30.2% of the variability in community metrics. PCB concentrations accounted for a small fraction, 1.0 to 6.8%, of the overall variability in these indices.

- The EPA analyses indicated that the fraction of variability explained by PCBs increased when individual taxa abundances (rather than summary metrics for all taxa combined) were considered. In fine-grained sediment, six of 32 taxa exhibited strong logarithmic regressions ($p \leq 0.005$) between abundance and sediment tPCB concentration; the R-squared values for these significant regressions ranged from 13% to 39% (using ½ DL replacement for non-detects). In coarse grained sediment, more than half of individual taxa exhibited strong logarithmic regressions ($p \leq 0.005$) between abundance and sediment tPCB concentration; the R-squared values for these significant regressions ranged from 8% to 38% (using ½ DL replacement for non-detects).

- Biomass Assessment – Overall, the biomass assessment yielded similar findings to the abundance assessment, in that lower biomass was evident in coarse-grained contaminated locations compared to reference locations (Section D.3.7.8). No impairment was evident in fine-grained sediment; however, the very high variability in taxonomic distributions among fine-grained locations suggests that habitat variations may limit the ability to detect perturbations.

- Species Sensitivity Distributions – Partitioning of the benthic community data into individual taxa revealed a wide range of organism sensitivities to PCBs (Section D.3.7.9; Attachment D.7). The SSD and associated regressions of abundance versus PCB concentrations identified alterations in benthic communities in both coarse- and fine-grained sediment. In both habitats, although the abundance of some taxa increased with increasing tPCB concentration, a greater number of taxa responded adversely to increased contamination. The overall community sensitivity to PCBs was observed to be greater in coarse-grained sediment relative to fine-grained sediment (Figure 3.3-17).

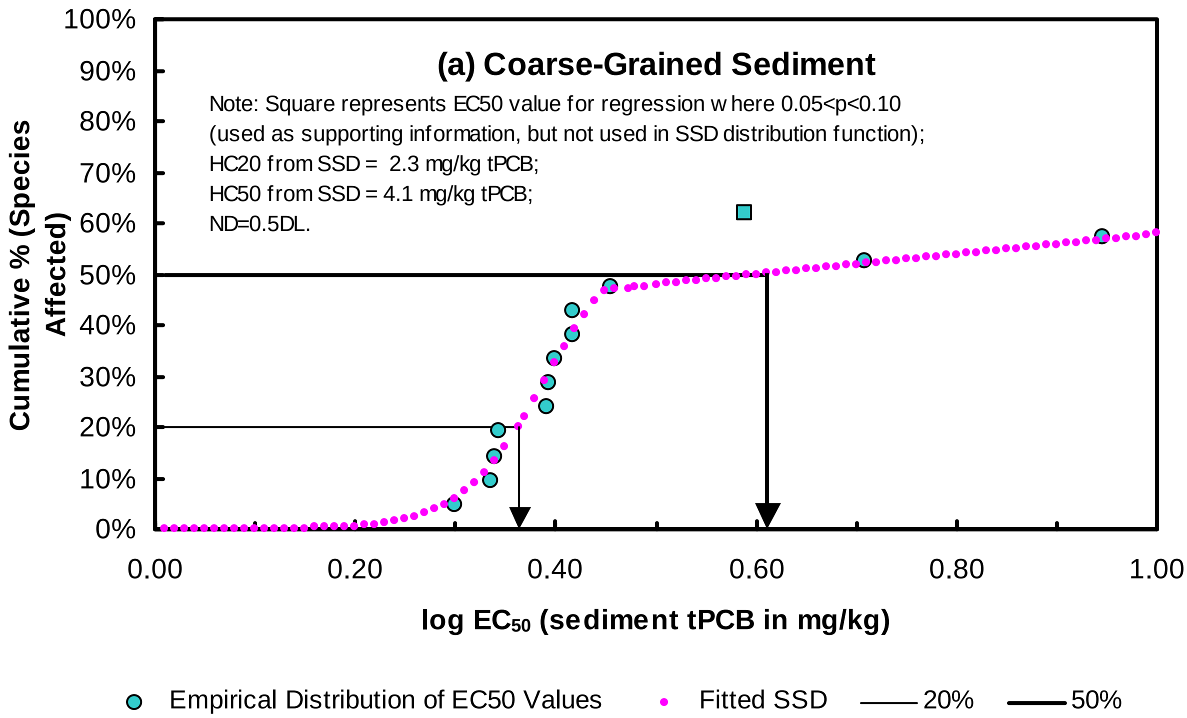
3.3.7 Dose-Response Analysis – Benthic Community Assemblages

The dose-response assessment for benthic community assemblages was conducted using only the PCB data collected synoptic with the benthic community grab sampling. Some community metrics (e.g., biomass, abundance, richness) exhibited weaker dose-response relationships relative to other metrics, particularly in fine-grained substrate. However, consideration of only summary measures tends to mask significant relationships that occur at lower levels of taxonomic organization. The abundances of specific taxa (Section D.3.7.7.1) and the species sensitivity distribution (SSD) analyses (Section D.3.7.9; Attachment D.7) indicated stronger PCB-related responses. The latter methods are more sensitive tools for detection of biological responses related to PCB contamination.

The assessment of diversity measures (Section D.3.7.5.3; Attachment D.8) also indicated PCB-related responses. All three diversity indices (Shannon Wiener H', Simpson's Index, Simpson's Modified Index) indicated significant inverse relationships to sediment tPCB concentration in both coarse- and fine-grained substrates. The relationship between PCB concentration and diversity was stronger in coarse-grained sediment than in fine-grained sediment (Figure 3.3-18), similar to the findings of the SSD analysis.

The lines of evidence from various benthic community measurement endpoints were used to identify threshold PCB concentrations associated with low, intermediate, and high risk, similar to the assessment of invertebrate toxicity endpoints (Section D.3.2.1.4). The effect sizes were 20% for the transition between low and intermediate risk, and 50% for the transition between intermediate and high risk. Considerations in the selection of a PCB threshold for community endpoints included:

- EC_{20} and EC_{50} values from regressions of abundance, richness, and diversity against tPCB concentrations (coarse- and fine-grained sediment) (See Section D.3.7.5).
- HC_{20} and HC_{50} values from the species sensitivity distributions (coarse- and fine-grained sediment) (See Section D.3.7.9).



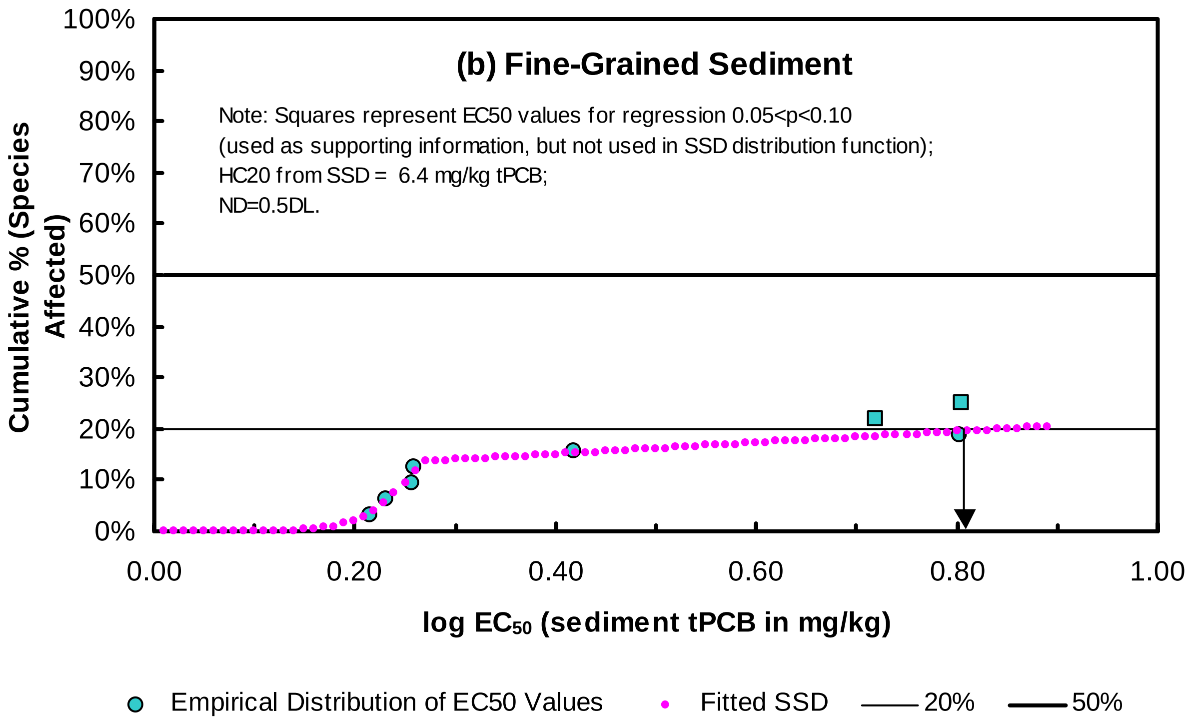
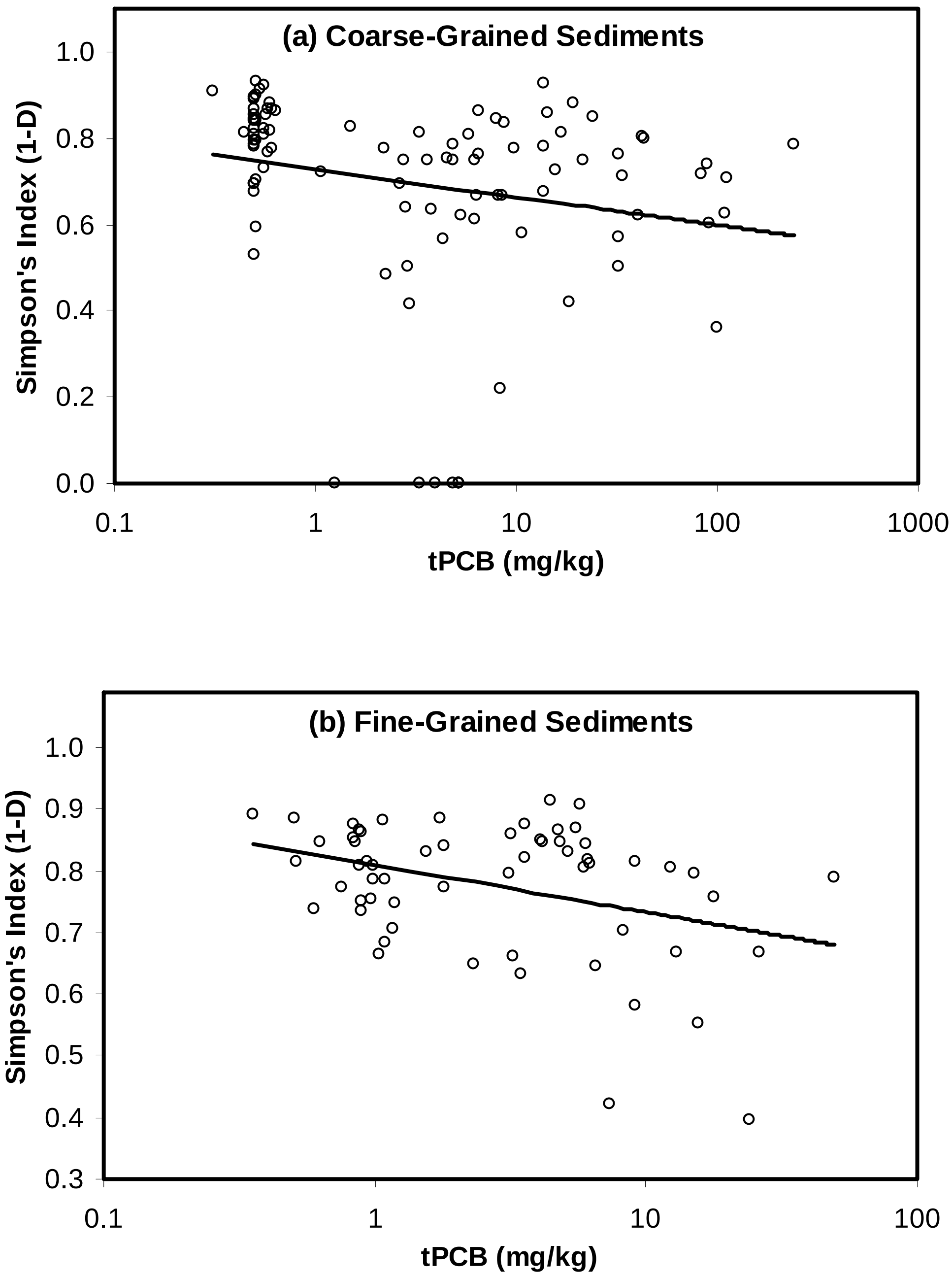


Figure 3.3-17 Species Sensitivity Distributions Fitted to Coarse- and Fine-Grained Sediment and Associated HC_{20}/HC_{50} Concentrations for tPCB



Note: tPCB concentrations for non-detected samples set to one-half detection limit. Regressions for both coarse- and fine-grained equations are statistically significant (p-values of 0.011 and 0.014, respectively).

Figure 3.3-18 Simpson's Diversity Index Versus Sediment tPCB Concentration in Benthic Community Grab Samples

- Semi-quantitative evaluation of multivariate endpoints (cluster analysis, MDS, relative rank plots), and linkage to median PCB concentrations observed at exposed locations (See Sections D.3.7.2 through D.3.7.4). The multivariate assessments indicated that most locations with median tPCB concentrations of greater than 5 mg/kg tPCB exhibited community alterations, whereas locations below 5 mg/kg tPCB did not.

Derivation of Threshold PCB Concentrations from Benthic Community Endpoints

- Intermediate Risk Threshold – 5.6 mg/kg tPCB, based on the geometric mean of the following "intermediate" effects levels:
  - HC_{20} value of 2.3 mg/kg tPCB in coarse-grained sediment from SSD.
  - HC_{20} value of 6.4 mg/kg tPCB in fine-grained sediment from SSD.
  - EC_{20} concentration of 13.4 mg/kg tPCB for taxa richness in coarse-grained sediment.
  - EC_{20} concentration of 5.8 mg/kg tPCB for total abundance in coarse-grained sediment.
  - EC_{20} concentration of 4.7 mg/kg tPCB for most sensitive diversity index (Shannon-Wiener H' in coarse-grained sediment).

- High Risk Threshold – 27.9 mg/kg tPCB, based on the geometric mean of the following "large" effects levels:
  - HC_{50} of 4.1 mg/kg in coarse-grained sediment from the SSD.
  - EC_{50} concentration for taxa richness (141.5 mg/kg tPCB) in coarse-grained sediment.
  - EC_{50} concentration for total abundance (37.3 mg/kg tPCB) in coarse-grained sediment.

No pattern in other COC concentrations (or habitat variables) appeared to contribute to the impaired benthic communities observed in coarse-grained contaminated sediment.

3.4 RISK CHARACTERIZATION

> ***Benthic Invertebrate Risk Characterization***
>
> - Integrates exposure and effects assessments.
> - Summarizes three major lines of evidence and conduct WOE for adverse effects to benthic invertebrates.
> - Discusses sources of uncertainty.
> - Extrapolates risk findings to other species and portions of the Housatonic River downstream of the PSA.

The risk characterization evaluates the likelihood that adverse effects have occurred as a result of exposure to COCs. Three lines of evidence were used in the risk characterization for benthic invertebrates (Figure 3.1-4):

- **Field surveys (i.e., benthic community structure) –** To the extent possible, responses due to COCs were distinguished from those related to other factors such as grain size or habitat type.
- **Comparison of exposure to effects levels or benchmarks –** Exposure and effects were integrated by relating the two terms quantitatively (e.g., hazard quotient [HQ] was derived using SQVs from the literature and/or site-specific effects thresholds compared to site-specific contaminant concentrations).
- **Site-specific toxicity studies –** Effects due to COCs were evaluated using in situ and laboratory toxicity tests.

These three lines of evidence were independent, allowing for a robust weight-of-evidence (WOE) assessment of the potential for risk using the approach of Menzie et al. (1996). The section concludes with a discussion of sources of uncertainty in the assessment of risks of COCs to invertebrates, and the conclusions of the risk characterization.

To characterize risk in a probabilistic manner (i.e., similar to the wildlife receptors) the following decision rules were applied:

- Low Risk: Probability (EC_{20}) < 20%
- High Risk: Probability (EC_{50}) > 50%
- Intermediate Risk: All other outcomes

The probabilities in the above framework are based on distributions of sediment tPCB chemistry data from discrete measurements in the PSA. The sediment MATC was used to mark the transition between low and intermediate risk, because it was developed based on consideration of EC_{20} values from toxicity and benthic community endpoints.

Site-specific thresholds representing the transition between low and intermediate risk, and the transition between intermediate and high risk, were developed considering both toxicity and benthic community findings:

- Intermediate Risk (MATC) Thresholds - The site-specific thresholds representing intermediate risk for toxicity endpoints and benthic community endpoints were 2.0 mg/kg tPCB and 5.6 mg/kg tPCB, respectively. These values were combined (using the rounded geometric mean) to yield a site-specific MATC of 3 mg/kg tPCB.
- High Risk Threshold - The high risk threshold was derived using the EC_{50} of chronic laboratory toxicity values (4.7 mg/kg tPCB), the EC_{50} of acute in situ toxicity values (17.5 mg/kg), and the EC_{50} of benthic community endpoints (27.9 mg/kg tPCB). The rounded geometric mean of these three thresholds (13 mg/kg) was used to mark the transition between intermediate and high risk in the above framework.

In addition, an intermediate acute toxicity threshold of 10 mg/kg was derived based on the in situ *Daphnia*, *Hyalella*, and *Chironomus* LC_{20} values. The acute LC_{20} value was not incorporated into the MATC calculation because acute lethality to multiple species represents a level of biological response that is greater than the protection level that is the goal of an MATC.

3.4.1 Field Surveys

The benthic invertebrate community study (Section 3.3.6 and 3.3.7) directly assessed the assemblages of organisms found throughout the PSA, and related these assemblages to concentrations of COCs and other stressors. After controlling for sediment particle size distributions and organic carbon content, significant differences between coarse-grained contaminated sites and coarse-grained references were observed. The differences were most pronounced in the abundances of specific indicator taxa (e.g., chironomid genera), as indicated by the species sensitivity distribution findings (Attachment D.7). Diversity metrics were moderately sensitive for detecting PCB-related responses (Attachment D.8), and total abundance and richness metrics were less sensitive metrics. The magnitude of response observed in fine-grained substrate was lower than that observed in coarse-grained sediment.

There are several possible explanations for the less pronounced community responses observed in the downstream fine-grained sediment within the PSA, including:

- Microhabitat variation – Unlike the coarse-grained sediment, the fine-grained portions of the PSA exhibited considerable inter-location differences in invertebrate communities. These variations may have masked PCB-related responses.

- Lower sediment concentrations of PCBs – The concentrations of tPCB in the benthic community sampling program were lower than those measured in other sampling efforts (e.g., toxicity studies). As shown in Figure 3.2.4, the median sediment tPCB concentration was generally in the 1-10 mg/kg range in the fine-grained sediment collected synoptic with the benthic community grabs. Although responses were observed for some biological metrics (e.g., diversity, abundance of sensitive individual taxa) over this concentration range, the statistical power for detecting differences in total abundance or other broad community metrics is low given the other sources of variability in the study.

3.4.2 Comparison of Contaminant Concentrations to Benchmarks

HQs were used to quantify the degree to which contaminant concentrations exceeded benchmarks considered protective of the assessment endpoint. Two types of HQ calculations were made:

- Comparison to Literature-Derived Thresholds – To address the uncertainty in benchmarks, the HQ assessment considered multiple benchmarks, and calculated a range of HQs. Sediment HQs were also calculated using the median value of all applicable benchmarks to provide a measure of central tendency. The extreme values of the HQ distribution are referred to as upper-bound and lower-bound HQs.

- SQVs derived from the literature are generally conservative, and hazard quotients greater than one based on literature SQVs must be interpreted in this context. Only literature-based HQs were derived for COCs other than PCBs.

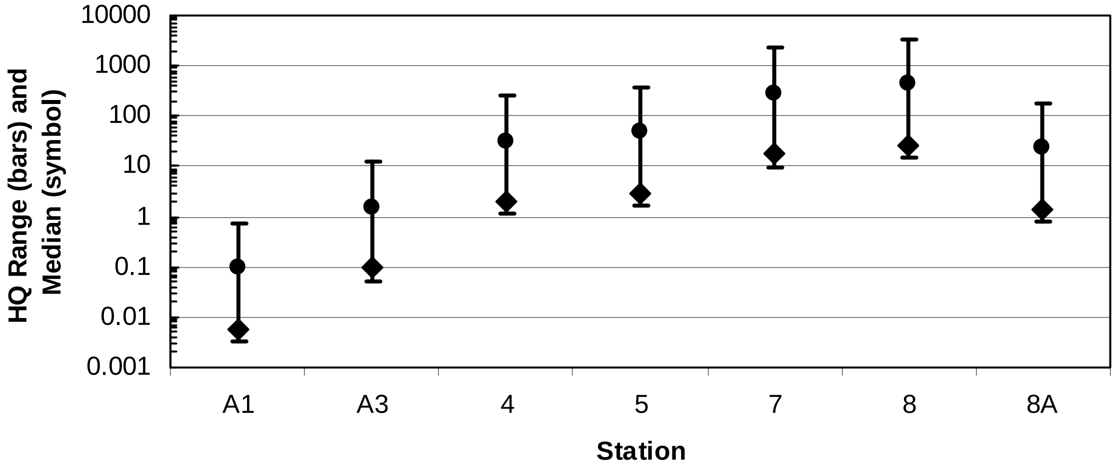
- Comparison to Site-Specific Thresholds – HQs based on site-specific effects thresholds are more reliable indicators of potential effects in the Housatonic River. The proportions of sediment samples exceeding the MATC of 3 mg/kg and/or the high risk threshold of 13 mg/kg were evaluated.

3.4.2.1 Sediment HQs

Figure 3.4-1 shows the ranges of HQs for the PCB measurements made at the seven toxicity testing locations in 1999. Within the time period of March to October 1999, 11 sampling events were conducted that were relevant to the effects data. The bars for each location indicate that the range of benchmarks derived from the literature (and thus HQs) is more than two orders of magnitude. The median SQV-based HQs for the contaminated locations are all greater than one, usually by a large amount.

Figure 3.4-1 also depicts HQs derived using the site-specific effects threshold (MATC) of 3 mg/kg. From a comparison of the two types of HQs, it is apparent that site-specific thresholds for toxicity observed in the Housatonic River fall within the range of values found in the literature, but are toward the higher end of SQVs (and therefore the lower end of HQs). All contaminated locations yielded HQ values greater than one.

Figure 3.4-2 presents the cumulative distribution of sediment tPCB concentrations throughout the PSA (by reach) compared against the site-specific criteria for intermediate and high risk. All reaches indicate at least intermediate risk, and Reach 6 indicates high risk. In all reaches, the percentage of sediment samples exceeding the MATC exceeds 50%.

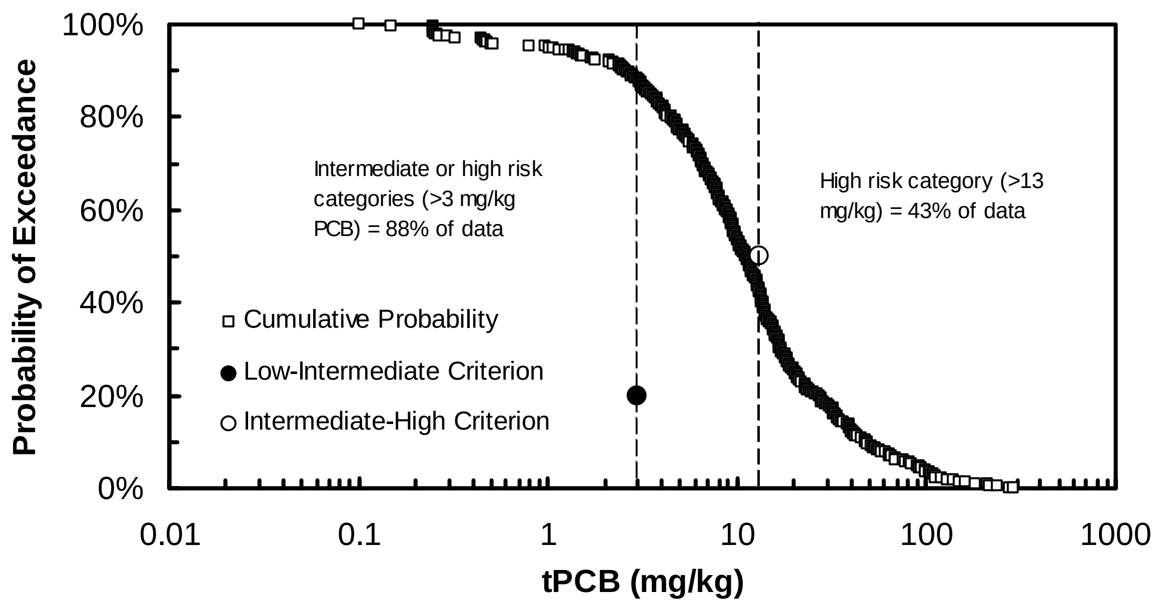


Notes: Circle symbol and error bars represent the median and range of HQs calculated using literature-derived sediment quality values. Ranges do not reflect variation in exposure concentrations, since the median sediment chemistry value represents a point estimate.

Diamond symbols represent HQs calculated using site-specific sediment effects benchmark (MATC) of 3 mg/kg.

Figure 3.4-1 Hazard Quotients (Median and Range) Based on Median Sediment Chemistry for tPCBs, for Samples Collected in 1999 Close to Sediment Quality Triad Locations

(a) Reach 5A



(b) Reach 5B

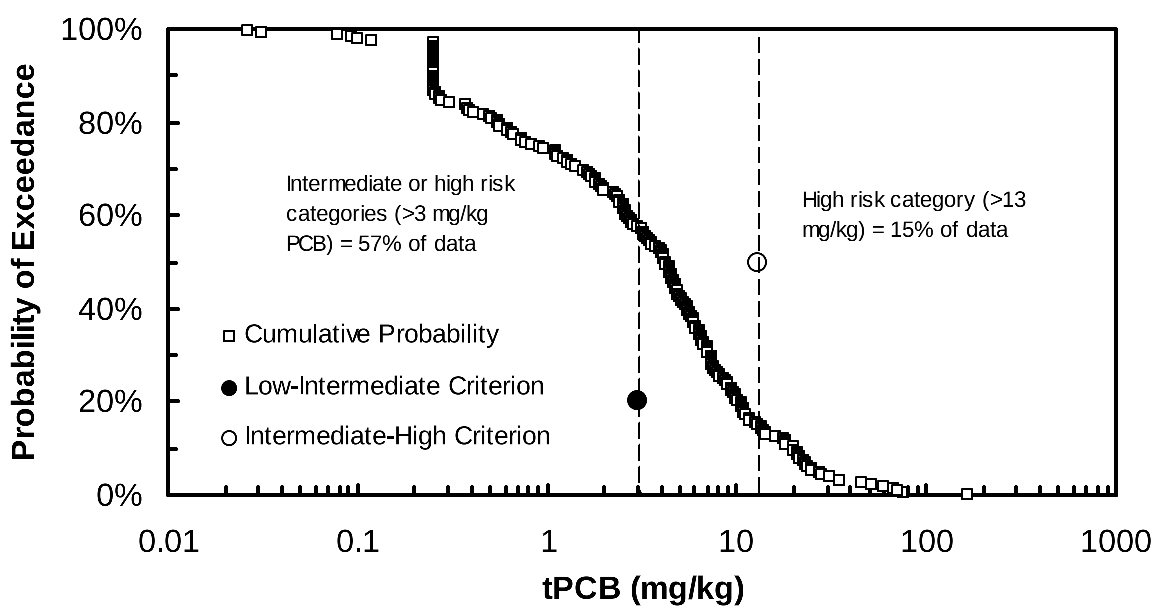
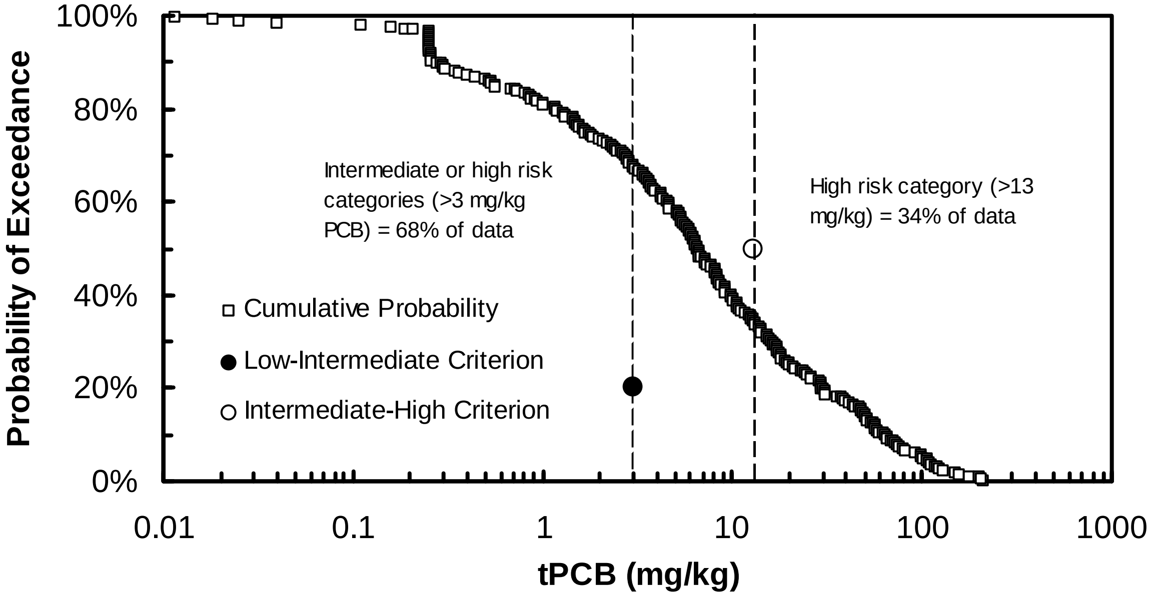


Figure 3.4-2 Cumulative Frequency of PSA Sediment tPCB Concentrations Relative to Site-Specific Intermediate and High Risk Thresholds

(c) Reach 5C



(d) Reach 5D

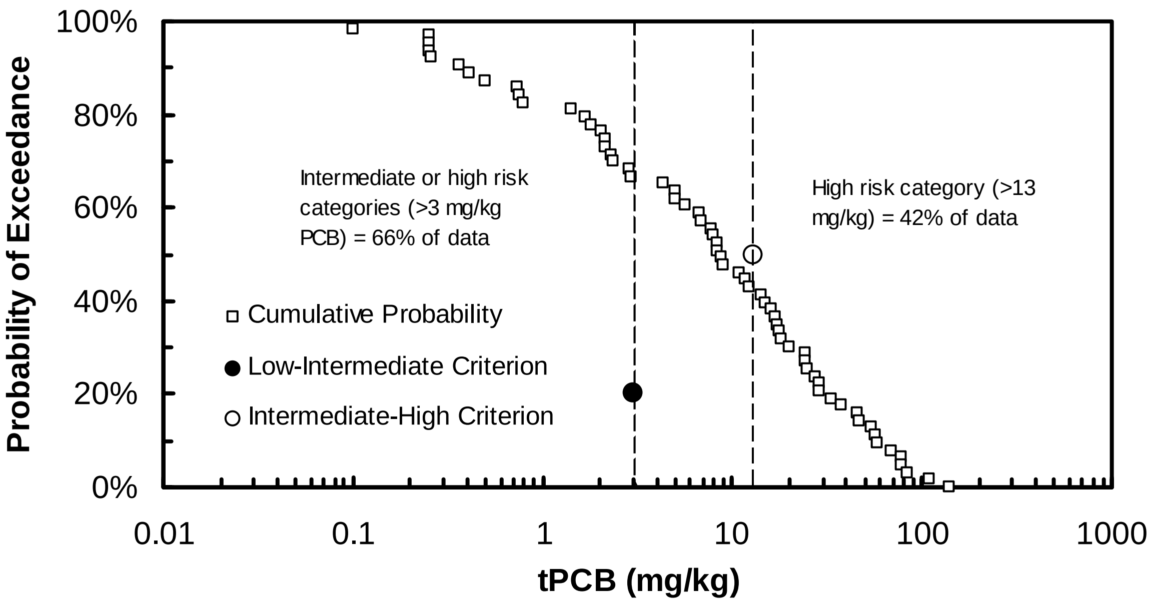


Figure 3.4-2 Cumulative Frequency of PSA Sediment tPCB Concentrations Relative to Site-Specific Intermediate and High Risk Thresholds (Continued)

(e) Woods Pond

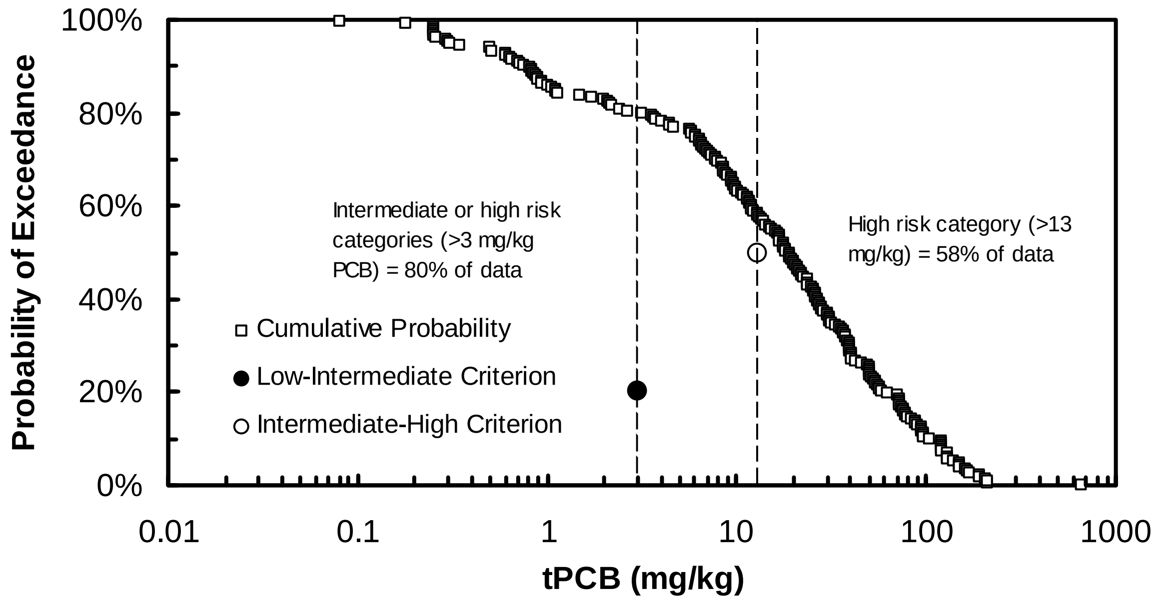


Figure 3.4-2 Cumulative Frequency of PSA Sediment tPCB Concentrations Relative to Site-Specific Intermediate and High Risk Thresholds (Continued)

HQs were also determined for other COCs that have SQVs. HQs were calculated for Sediment Quality Triad location concentration data, and also for broader reach-wide data.

Sediment HQs for Other COCs at Triad Sampling Locations

- Median antimony HQs were below 1 and maximum antimony HQs barely exceeded 1 at downstream locations.
- Barium HQs barely exceeded 1, and only at downstream locations.
- Median cadmium HQs exceeded 1 only at Location 7, and maximum HQs were 10 or less even at the most contaminated sites.
- Median chromium concentrations barely exceeded 1 at downstream locations, and maximum HQs were below 10.
- Maximum copper and lead HQs were 10 or less, even at the most contaminated locations.
- Mercury and silver exhibited median HQs between 1 and 10 at most downstream locations.
- The HQs for total PAHs also indicated low risk at locations from these compounds, with median HQs below 3 at all locations. However, the wide range of PAH SQVs resulted in higher HQs (i.e., greater than 10) if lower-bound SQVs are applied.

The broader PSA data indicated HQs that were equal to or lower than those described above for the sampling locations. For example, the PAH HQs were much lower using the broader PSA data, with median HQs below 1 for all reaches and both substrate types.

Overall, the HQ assessment for sediment indicated that the chemical hazard for tPCBs was much higher than for other COCs. The median literature-based HQ for tPCB was often 100 to 1,000, compared to other COCs that rarely exceeded an HQ of 10. This finding is in agreement with the TIE conclusions, which implicated PCBs and/or other non-polar organics as the dominant causative agents in toxicity tests. When HQs based on site-specific tPCB effects information are considered, risks are moderate to high for most sediment found within the PSA.

3.4.2.2 Water Chemistry

HQs for PCBs were calculated by comparing the PCB water column data derived from the toxicity study (EVS 2003) to water quality criteria for PCBs. The median HQs for both reference locations (A1, A3) were less than 1.0 in all three sampling events. In contrast, the PCB concentrations at contaminated locations exhibited median HQs that were elevated and fairly consistent among locations and across monitoring events (i.e., median HQ of approximately 10). The maximum HQs, using worst-case PCB benchmarks, were approximately 100. Overall, the results indicate a moderately high hazard based on PCB chemistry in the water column, with negligible risk from other water column contaminants.

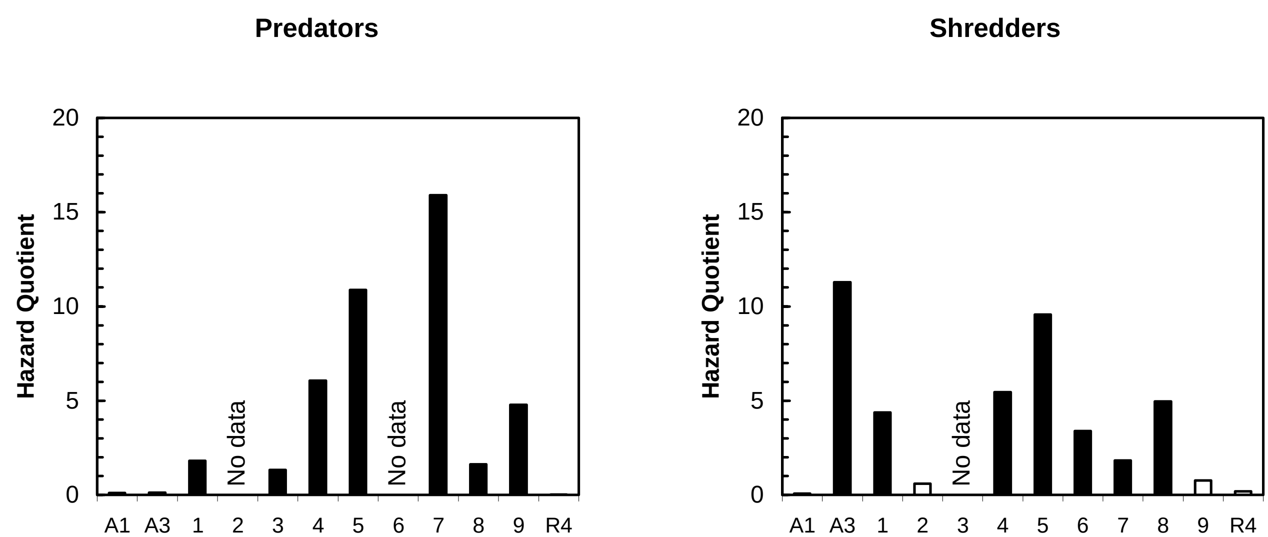
3.4.2.3 Tissue Chemistry

HQs were derived for tissue PCB burdens in benthic invertebrates sampled near the Sediment Quality Triad locations. Two sets of HQs were derived, representing different levels of conservatism (Figure 3.4-3). One set of HQs was based on comparison of observed tissue concentrations to an effects threshold (intermediate risk) of 3 mg/kg ww tPCBs, which represents the lowest concentration at which significant adverse effects were found in the literature. Nearly all HQs derived in this manner were greater than 1.0, and three HQs were greater than 10. The second method compared observed tissue concentrations to a 10 mg/kg ww tPCB threshold (high risk), a concentration that the literature review suggested would cause impacts to numerous species. Even with this relaxed benchmark, most HQs still exceeded 1.0.

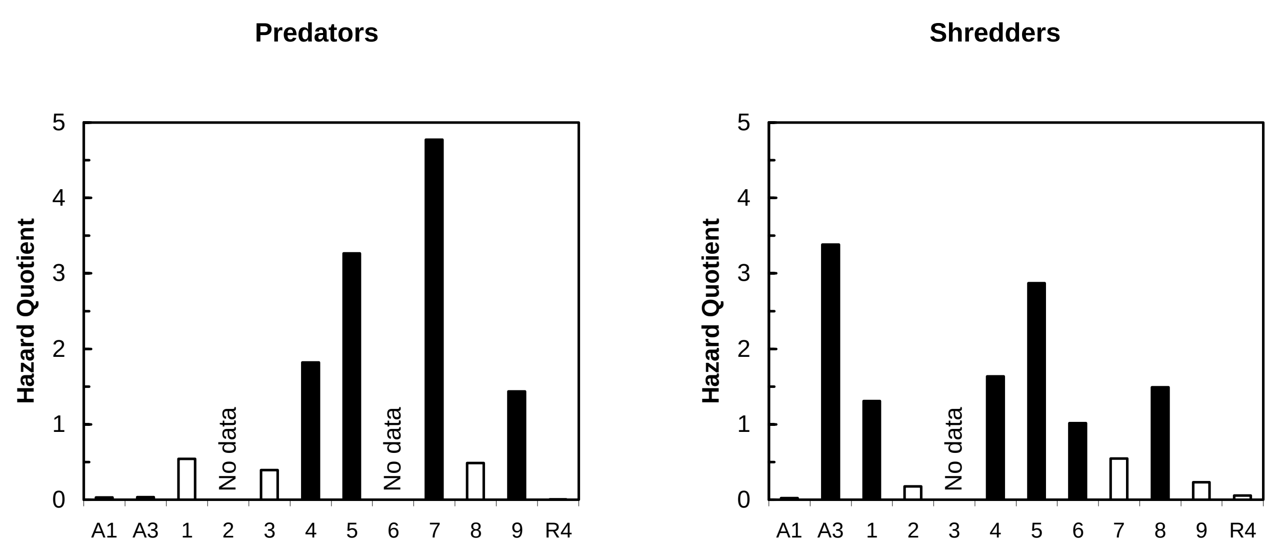
3.4.3 Site-Specific Toxicity Study Results

Both the in situ and laboratory toxicity tests (Section 3.3.1) exhibited significant adverse effects in both coarse- and fine-grained sediment, relative to both negative controls and field reference sediment. The only toxicity test endpoints that did not show significant adverse responses at the highest tPCB concentrations were: (a) limited exposure pathways, such as water-only exposures; (b) short test durations; and/or (c) tolerant test species, such as freshwater oligochaetes used for bioaccumulation testing. The large number of endpoints indicating significant toxicity (even for some acute lethal endpoints), and the high magnitude of response at the highest PCB concentrations (100% mortality in some treatments), indicate that risks to invertebrates are high in localized areas within all reaches of the PSA (see Figure 3.4-2). The evaluation of dose-response (Section 3.3.2) and the TIE study (Section 3.3.3) both indicated that non-polar organics (principally PCBs) were likely the dominant toxic agents in the toxicity tests.

(a) Relative to lower-bound effects threshold of 3 mg/kg (HQ > 1 in black)



(b) Relative to upper-bound effects threshold of 10 mg/kg (HQ > 1 in black)



Note: Tissue concentrations are wet weight.

Figure 3.4-3 Hazard Quotients for tPCB Tissue Residues in Benthic Invertebrates, Relative to Two Effects Thresholds Derived from Literature Studies

3.4.4 Weight-of-Evidence (WOE) Procedure for Assessing Risk from PCBs in the Housatonic River PSA

A formal WOE process was applied to determine whether PCBs pose a significant risk to the Housatonic River benthos. The three-phase approach of Menzie et al. (1996) and the Massachusetts Weight-of-Evidence Workgroup was applied for this purpose, in which WOE was reflected in the following three characteristics: (1) the weight assigned to each measurement endpoint; (2) the magnitude of response observed in the measurement endpoint; and (3) the concurrence among outcomes of the multiple measurement endpoints.

A discussion of attributes considered in the WOE is provided in Section 2, and the detailed rationales for weighting of measurement endpoints for benthic invertebrates are provided in Appendix D (Table D.4-1). A summary of the weightings for each attribute is provided in Table 3.4-1. The chemistry endpoints yielded the lowest overall values because of lower site-specificity and some uncertainties in the biological association between the measurement endpoints and the assessment endpoint(s). The toxicity testing endpoints yielded the highest overall values, due in part to the high sensitivity to detecting changes (i.e., ability of the endpoint to detect variation in the stressor) and the high utility of the measures (i.e., ability to judge measurement endpoint results against well-accepted performance-based measures). The benthic community structure endpoints had intermediate weighting values. Although these endpoints were site-specific, collected at a time when effects would be expected, and were measures of the community structure component of the assessment endpoint, the potential for the confounding effects of other factors in the direct attribution of the response to the stressor reduced the utility of these endpoints to some degree.

The magnitude of the response in the measurement endpoint is considered together with the measurement endpoint weight in judging the overall WOE (Menzie et al. 1996). This requires assessing the strength of evidence that ecological harm has occurred, as well as an indication of the magnitude of response, if present. The weighting scores, evidence of harm, and magnitudes of responses were combined in a matrix format and are presented in Table 3.4-2.

A graphical method was used for displaying concurrence among measurement endpoints (Table 3.4-3). The method entailed plotting the nine symbols representing the toxicity (T), benthic community (B), and chemistry (C) endpoints in a matrix, with the weight of the measurement endpoint and the degree of response as axes. These graphics indicate that the majority of endpoints suggest some risk for benthic communities in both coarse- and fine-grained sediment. The plots also indicate that several of the endpoints suggest a high degree of risk with a relatively high weight (e.g., toxicity endpoints). The conclusion from interpretation of Table 3.4-3 is that there is a moderate to high risk to much of the benthic community indicated by the WOE evaluation.

Table 3.4-1

Weighting of Measurement Endpoints for Benthic Invertebrate WOE – Summary

Attributes	Chemistry Endpoints			Toxicity Endpoints			Benthic Community Endpoints			
	C-1 (Water)	**C-2 (Sediment)**	**C-3 (Tissue)**	**T-1 (Lab)**	**T-2 (In-situ)**	**T-3 (TIE)**	**B-1 (Metrics)**	**B-2 (Multivar)**	**B-3 (MHBI)**	**B-4 (SSD)**
1. Degree of Biological Association	Low	Low	Moderate	Moderate	Mod/High	Moderate	Moderate	Moderate	Low/Mod	Moderate
2. Stressor/Response	Low	Low/Mod	Moderate	Mod/High	Mod/High	Mod/High	Low/Mod	Low/Mod	Low/Mod	Moderate
3. Utility of Measure	Low	Low/Mod	Moderate	High	Mod/High	Mod/High	Low/Mod	Low/Mod	Low/Mod	Low/Mod
4. Data Quality	High	High	High	High	High	High	High	High	High	High
5. Site Specificity	Low/Mod	Low/Mod	Low/Mod	Mod/High	High	Mod/High	High	High	High	High
6. Sensitivity to Detecting Changes	Low/Mod	Low/Mod	Moderate	High	Mod/High	Mod/High	Low/Mod	Low/Mod	Low/Mod	Moderate
7. Spatial Representativeness	Moderate	High	Moderate	Moderate	Mod/High	Low	Mod/High	Mod/High	Mod/High	Mod/High
8. Temporal Representativeness	Moderate	High	Low/Mod	Moderate	Moderate	Low/Mod	Moderate	Moderate	Moderate	Moderate
9. Quantitativeness	Moderate	Moderate	Moderate	Mod/High	Mod/High	Moderate	Mod/High	Mod/High	Moderate	Mod/High
10. Standard Method	Moderate	High	Mod/High	High	Mod/High	Mod/High	Mod/High	Mod/High	Moderate	Moderate
Overall Endpoint Value	**Low/Mod**	**Moderate**	**Moderate**	**Mod/High**	**Mod/High**	**Moderate**	**Moderate**	**Moderate**	**Moderate**	**Mod/High**

C. Chemical Measures [C-1. Concentration of PCB in overlying water relative to literature thresholds for effects; C-2. Concentration of PCB in the sediment relative to literature thresholds for effects; C-3. Concentration of PCB in invertebrate tissues relative to literature thresholds for effects].

T. Toxicological Measures [T-1. Sediment toxicity to multiple invertebrate species, as measured in laboratory toxicity tests; T-2. Sediment toxicity to multiple invertebrate species, as measured in the in situ toxicity tests; T-3. Indications of PCB as toxicity driver in TIE investigations].

B. Benthic Community Measures [B-1. Abundance, richness, and biomass of invertebrates, relative to reference locations of comparable substrate and habitat (ANOVA analysis); B-2. Benthic community structure, as assessed using multivariate assessment of key benthic metrics (rank analysis and multidimensional scaling); B-3. Water quality assessment using modified Hilsenhoff Biotic Index (MHBI) indicator of organic pollution; B-4. Species sensitivity distribution analysis of individual taxa].

Table 3.4-2

Evidence of Harm and Magnitude of Effects for Measurement Endpoints Related to Maintenance of a Healthy Benthic Invertebrate Community

Measurement Endpoints	Weighting Value	Evidence of Harm	Magnitude of Harm (High, Intermediate, or Low)	
			Coarse-Grained	Fine-Grained
C. Chemistry Endpoints				
C-1. Concentration of PCB in overlying water in relation to concentrations reported (in literature) to be harmful to benthic invertebrates.	Low/Moderate	Yes	Intermediate	Intermediate
C-2. Concentration of PCB in sediment in relation to concentrations reported (in literature) to be harmful to benthic invertebrates.	Moderate	Yes	High	High
C-3. Concentration of PCB in invertebrate tissues in relation to concentrations reported (in literature) to be harmful to benthic invertebrates.	Moderate	Yes	High	High
T. Toxicity Endpoints				
T-1. Sediment toxicity to multiple invertebrate species, as measured in the laboratory toxicity tests (*Hyalella*, *Chironomus*).	Moderate/High	Yes	High	High
T-2. Sediment toxicity to multiple invertebrate species, as measured in the in situ toxicity tests (*Hyalella*, *Chironomus*, *Daphnia*).	Moderate/High	Yes	High	High
T-3. Indications of PCB as toxicity driver in TIE investigations (*Ceriodaphnia*).	Moderate	Yes	–	Intermediate

Table 3.4-2

Evidence of Harm and Magnitude of Effects for Measurement Endpoints Related to Maintenance of a Healthy Benthic Invertebrate Community (Continued)

Measurement Endpoints	Weighting Value	Evidence of Harm	Magnitude of Harm (High, Intermediate, or Low)	
			Coarse-Grained	Fine-Grained
B. Benthic Community Endpoints				
B-1. Abundance, richness, biomass, and diversity of assemblages, relative to reference stations of comparable substrate and habitat (ANOVA and regression analyses).	Moderate	Yes	Intermediate	Low
B-2. Benthic community structure, as assessed using multivariate analysis of benthic metrics (rank analysis, multidimensional scaling) and multiple regression analyses.	Moderate	Yes	Intermediate	Low
B-3. Water quality assessment using modified Hilsenhoff Biotic Index (MHBI) indicator of organic pollution	Moderate	No	–	–
B-4. Evaluation of individual taxa using species sensitivity distribution (SSD) analysis of abundance endpoint.	Moderate/High	Yes	High	Intermediate

Table 3.4-3

Weight-of-Evidence Risk Analysis Summary Indicating Concurrence Among Endpoints for Coarse-Grained and Fine-Grained Sediment

Assessment Endpoint:	Community, structure, survival, growth, and reproduction of benthic invertebrates

(a) Coarse-grained contaminated (C/C) sediment

	Weighting Factors (increasing confidence or weight)				
Harm/Magnitude	**Low**	**Low/Moderate**	**Intermediate**	**Moderate/High**	**High**
Yes/High			C-2, C-3	T-1, T-2, B-4	
Yes/Intermediate			B-1, B-2		
Yes/Low					
Undetermined			T-3		
No Harm			B-3		

(b) Fine-grained contaminated sediment (F/C)

	Weighting Factors (increasing confidence or weight)				
Harm/Magnitude	C-1 **Low**	**Low/Moderate**	**Intermediate**	**Moderate/High**	**High**
Yes/High			C-2, C-3	T-1, T-2	
Yes/Intermediate		C-1	T-3	B-4	
Yes/Low			B-1, B-2		
Undetermined					
No Harm			B-3		

Note: See Tables 3.4-1 and 3.4-2 for definitions of endpoint codes.

3.4.5 Sources of Uncertainty

Some of the major sources of uncertainty associated with the assessment of risks of tPCBs and TEQ to benthic invertebrates are briefly summarized below. A more complete discussion is presented in Appendix D.

- Small-scale spatial/temporal variability in COC exposure concentrations.
- Inconsistencies in exposure concentrations across site-specific studies.
- Occurrence of elevated PCB concentrations (tissue and sediment) at Reference Location A3 (West Branch).
- Effects benchmarks derived from the literature.
- Small-scale habitat heterogeneity.
- Varying species sensitivity to COCs.

These uncertainties were addressed in the ERA by conducting multiple assessments (e.g., applying different statistical models and exposure assumptions). The general concordance of the findings using many different data processing assumptions provides confidence that the derived MATCs are not based on spurious statistical outcomes.

3.4.6 Analysis of Risk Downstream of the PSA

Because of the more limited amount and spatial coverage of biological and contaminant concentration data downstream of the PSA, the more rigorous approach followed in assessing ecological risks in the PSA was not appropriate or possible. Risk estimates for downstream of the PSA were derived by comparing observed sediment and tissue concentrations with maximum acceptable threshold concentrations (MATCs) for tPCBs developed from the Sediment Quality Triad.

Use of MATCs to Assess Risks from PCBs Downstream of the PSA

- The sediment MATC of 3 mg/kg tPCB was used to mark the transition between low and intermediate risk to benthic invertebrates downstream of the PSA.
- The tissue MATC of 3 mg/kg ww tPCB was used to mark the transition between low and intermediate risk to benthic invertebrates downstream of the PSA.

Using the MATC values, potential risks to benthic invertebrates are predicted to occur in limited areas downstream of Woods Pond to Rising Pond, where pockets of sediment contaminated with higher concentrations of PCBs have accumulated. Below Rising Pond and through the remainder of Massachusetts and Connecticut, sediment does not contain concentrations of PCBs that are sufficiently elevated to represent a potential risk to benthic invertebrates. PCB concentrations in tissue of caddisflies, dobsonflies, and stoneflies (BBL and QEA 2003) collected from Cornwall, CT, were at or below the toxicity threshold of 3 mg/kg ww, and therefore support the conclusions based on application of the sediment MATC (i.e., low risk).

3.4.7 Extrapolation to Other Species

The benthic invertebrate ERA included the entire benthic community; benthic community composition analysis was a measurement endpoint considered in the weight-of-evidence assessment. Furthermore, the species sensitivity distribution approach explicitly considered the range of sensitivities within a benthic community. Individual species were also used in toxicity tests as surrogates for the Housatonic River freshwater benthic community. Both the status of sensitive taxa and community composition are considered indicators of overall health and productivity of the benthic community. As a result, no explicit extrapolation to other species was required.

3.4.8 Conclusions

Overall, the benthic ERA indicates intermediate to high risk to benthic invertebrates in the PSA based on a WOE evaluation of multiple Sediment Quality Triad endpoints. Furthermore, the available data suggest that PCBs are the primary chemical stressor responsible for such impairment. The confidence in the conclusion is moderate to high, based upon the concordance in predictions of risk from multiple measurement endpoints.

Evidence of ecological risk comes from the sediment toxicity tests, which not only indicated significant toxicological effects in multiple appropriate indicator species and endpoints, but also indicated a correlation between the level of effect and sediment PCB concentration (in both laboratory and in situ tests). This correlation was consistent with the TIE results, which implicated non-polar organics as the dominant toxic agent in Housatonic River sediment.

Community alterations, as identified by abundance, richness, and diversity metrics, SSD results, and multivariate measures, were observed in coarse-grained sediment at concentrations similar to those that exhibited sediment toxicity. The evidence of effects on benthic community structure was not as clear for fine-grained sediment; however, diversity measures and abundances of sensitive taxa were inversely related to tPCB concentrations in fine-grained sediment. However, the proportion of variation explained by PCBs is low, due to natural variations in benthic communities, microhabitat variations, and small-scale variability in PCB exposure data.

The magnitude of risk to benthic invertebrates in the Housatonic River varies spatially, primarily as a function of sediment PCB concentration and also in relation to sediment characteristics, primarily organic carbon content and grain size. The WOE assessment of benthic invertebrate endpoints indicates intermediate risk at concentrations of 3 mg/kg tPCB or higher, and that risks are high at 13 mg/kg tPCB or higher. These concentrations are in concordance with the upper tail of the SQV distribution for tPCBs identified in a literature review (i.e., 1 to 10 mg/kg). The spatial distribution of tPCB concentrations in the PSA (Figure 3.2-6) and the cumulative probability distribution of tPCB concentrations in each reach (Figure 3.2-7) indicate that the majority of the sediment in the PSA exceeds one or both of these threshold effects levels. In the downstream reaches of the PSA, risks were lower; however, the tPCB data indicate that a high percentage of samples still exceed the site-specific thresholds described above (i.e., approximately 75% of samples above the MATC). Downstream of Woods Pond, risks are reduced relative to the PSA, and are low downstream of Rising Pond.

3.5 REFERENCES

BBL (Blasland, Bouck & Lee, Inc.) and QEA (Quantitative Environmental Analysis, LLC). 2003. *Housatonic River – Rest of River RCRA Facility Investigation Report*. Prepared for General Electric Company. January 2003.

Chapman, P.M. 1996. Presentation and interpretation of Sediment Quality Triad data. *Ecotoxicology* 5:327-339.

EPA (U.S. Environmental Protection Agency). 2000. *Methods for Measuring the Toxicity and Bioaccumulation of Sediment-Associated Contaminants with Freshwater Invertebrates.* EPA 600/R-99/064. 192 pp.

EVS Environment Consultants. 2003. *Assessment of in situ Stressors and Sediment Toxicity in the Lower Housatonic River.* Final Report, adapted from a study by G.A. Burton, 2001, Institute for Environmental Quality, Wright State University, Dayton Ohio.

Long, E.R., and P.M. Chapman. 1985. A Sediment Quality Triad: measures of sediment contamination, toxicity and infaunal community composition in Puget Sound. *Marine Pollution Bulletin* 16:405-415.

MacDonald, D.D., C.G. Ingersoll, and T.A. Berger. 2000. Development and evaluation of consensus-based sediment quality guidelines for freshwater ecosystems. *Arch. Environ. Contam. Toxicol*. 39:20-31.

Menzie, C., M.H. Henning, J. Cura, K. Finkelstein, J. Gentile, J. Maughan, D. Mitchell, S. Petron, B. Potocki, S. Svirsky, and P. Tyler. 1996. Special report of the Massachusetts Weight-of-Evidence Workgroup: A weight of evidence approach for evaluating ecological risks. *Human and Ecological Risk Assessment* 2:277-304.

Swartz, R.C. 1999. Consensus sediment quality guidelines for polycyclic aromatic hydrocarbon mixtures. *Environ. Tox. Chem.* 18(4):780-787.

4. ASSESSMENT ENDPOINT—COMMUNITY CONDITION, SURVIVAL, REPRODUCTION, DEVELOPMENT, AND MATURATION OF AMPHIBIANS

Highlights of the Amphibian ERA

Conceptual Model

The assessment endpoint is the survival, development, and reproduction of amphibians in the PSA. Amphibians, including leopard frogs and wood frogs, selected as representative species for the ERA, are exposed to contaminants of concern (COCs) via diet and possibly dermal absorption.

Exposure

Exposure of the representative species to tPCBs, dioxins and furans, metals, and PAHs was determined through three site-specific studies that evaluated reproductive performance and developmental effects. Routes of exposure and rates of bioaccumulation were also assessed.

Effects

Reproductive performance and early developmental effects were assessed using a number of measurement endpoints in frogs from contaminated areas in the PSA and frogs from reference areas from the Housatonic River watershed and external reference sources. These effects were compared to those reported in the literature to identify similarity of responses for COCs, types of effects, and mechanisms of effects.

Risk

There is a high probability of risk of ecologically significant effects at PCB concentrations observed in the PSA. There were significant correlations between adverse effects in late larval-stage wood frogs and PCB concentrations in sediment and tissue. Leopard frogs appear more acutely sensitive than wood frogs, with strong indications of toxicity observed through the range of tPCB concentrations tested in the PSA. These findings suggest that amphibian populations are impacted throughout much of the PSA. The indications of community responses from the population studies (i.e., localized depressions of richness and abundance near high tPCB vernal pools, and high incidence of malformations observed) substantiate these findings.

4.1 INTRODUCTION

This section summarizes the current and potential risks posed to amphibians exposed to COPCs in the Housatonic River and floodplain. A Pre-ERA (Appendix B) was conducted to identify COPCs, other than tPCBs, posing potential risks to aquatic biota in the PSA. The COPCs were then screened for specific relevance to the amphibian community, and the contaminants of concern (COCs) that were retained for the detailed risk assessment were tPCBs, six metals, several polycyclic aromatic hydrocarbons (PAHs), and dibenzofurans.

A step-wise approach was used to assess the risks of these COCs to amphibians in the Housatonic River watershed. The four main steps in this process included:

1. Derivation of a conceptual model (Figure 4.1-1).
2. Assessment of exposure of amphibians to COCs (Figure 4.1-2).
3. Assessment of the effects of COCs on amphibians (Figure 4.1-3).
4. Characterization of risks to amphibians (Figure 4.1-4).

> *The detailed ecological risk assessment for amphibians is provided in Appendix E.*

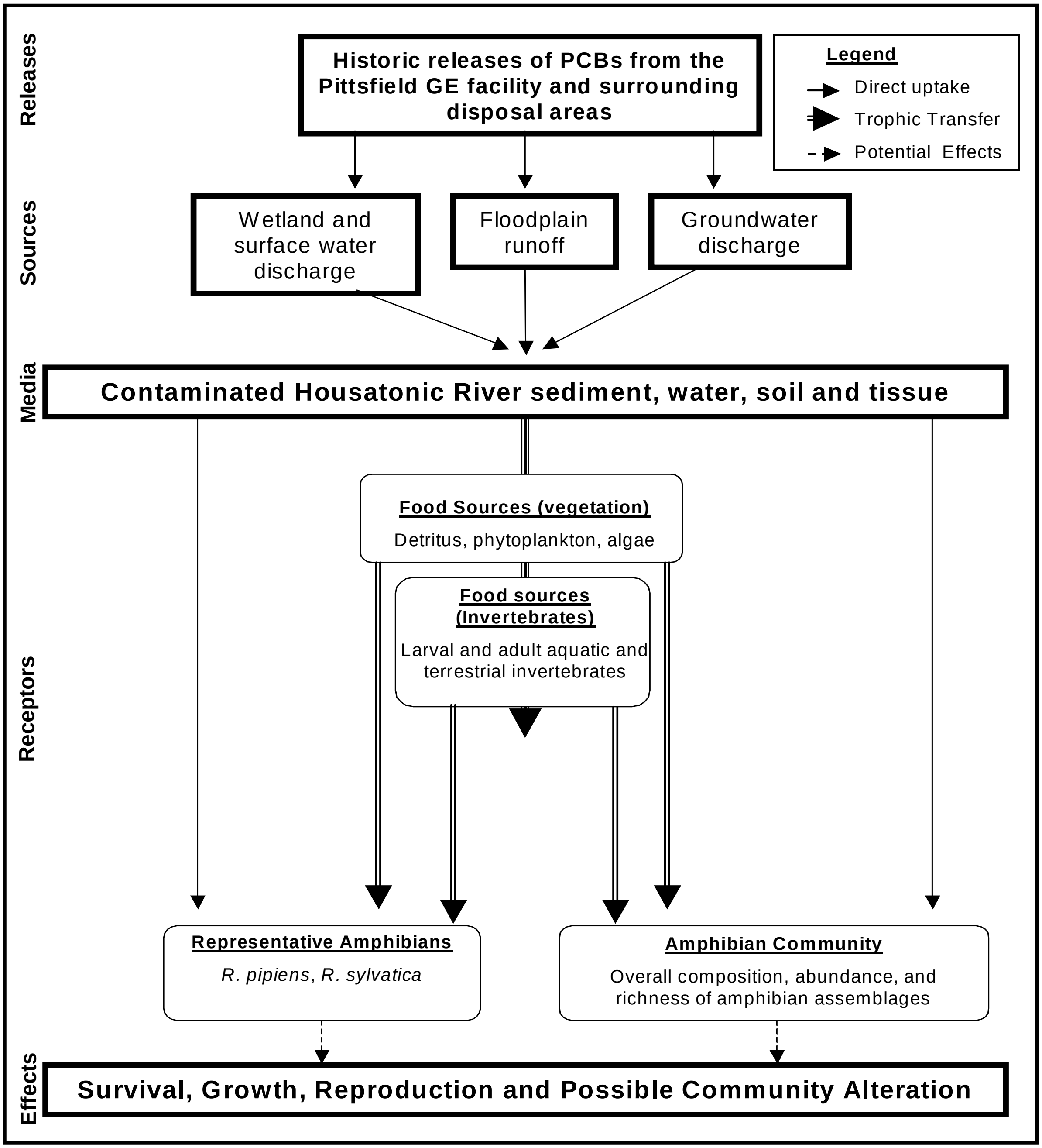


Figure 4.1-1 Conceptual Model Diagram: Exposure Pathways for Amphibians Exposed to Contaminants of Concern (COCs) in the Housatonic River PSA

Exposure

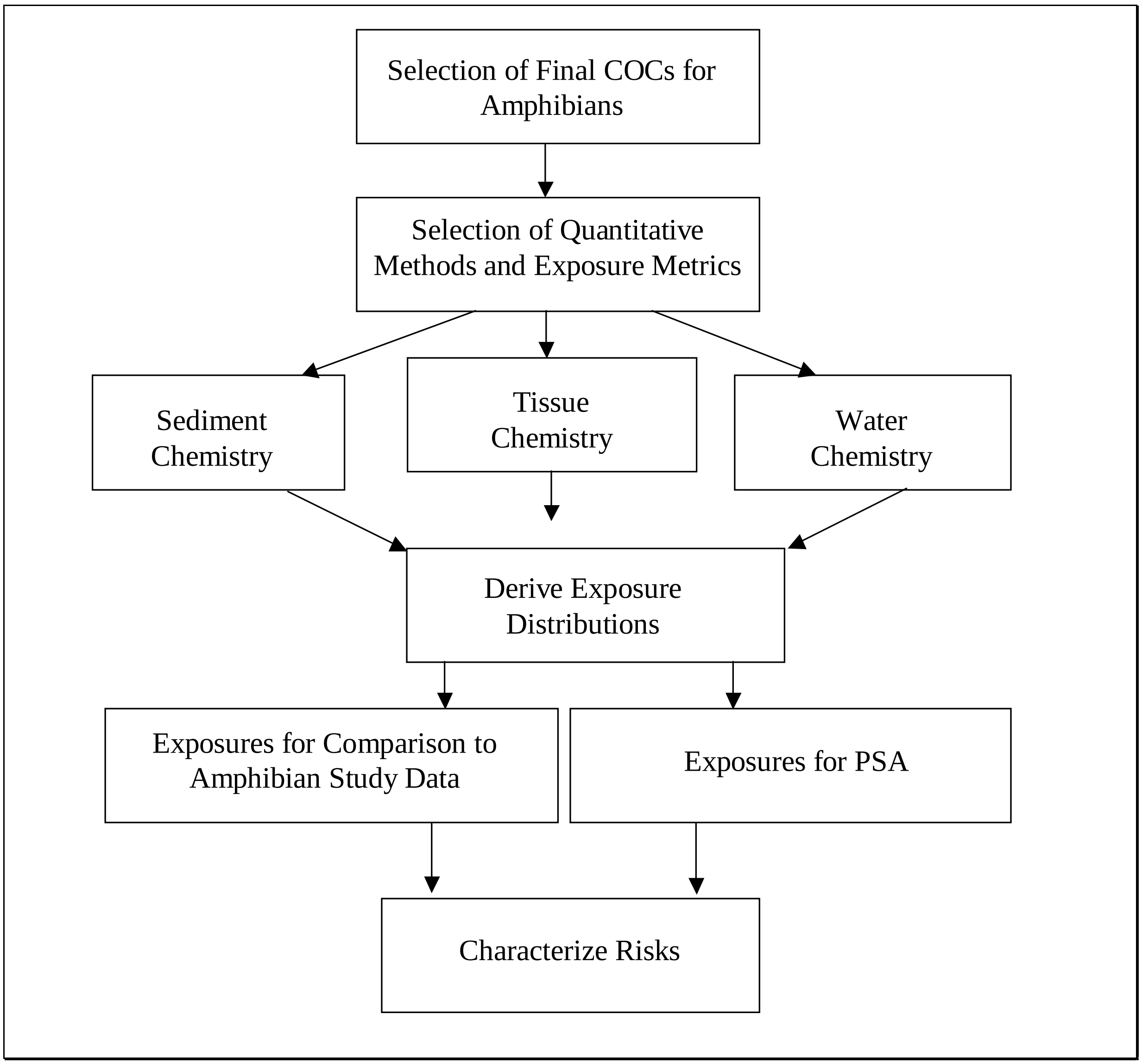


Figure 4.1-2 Overview of Approach Used to Assess Exposure of Amphibians to Contaminants of Concern (COCs) in the Housatonic River PSA

Effects

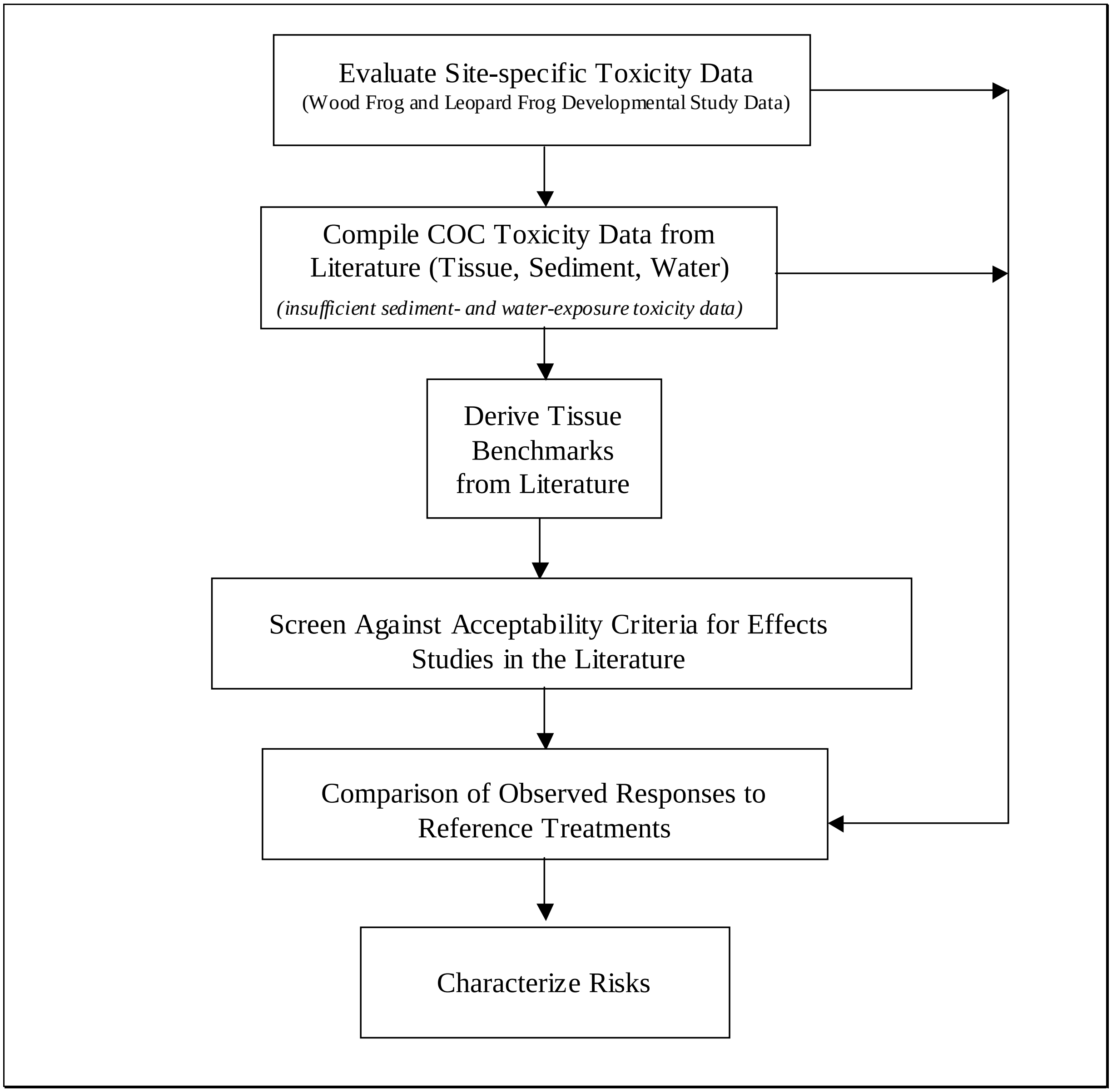


Figure 4.1-3 Overview of Approach Used to Assess the Effects of Contaminants of Concern (COCs) to Amphibians in the Housatonic River PSA

Risk Characterization

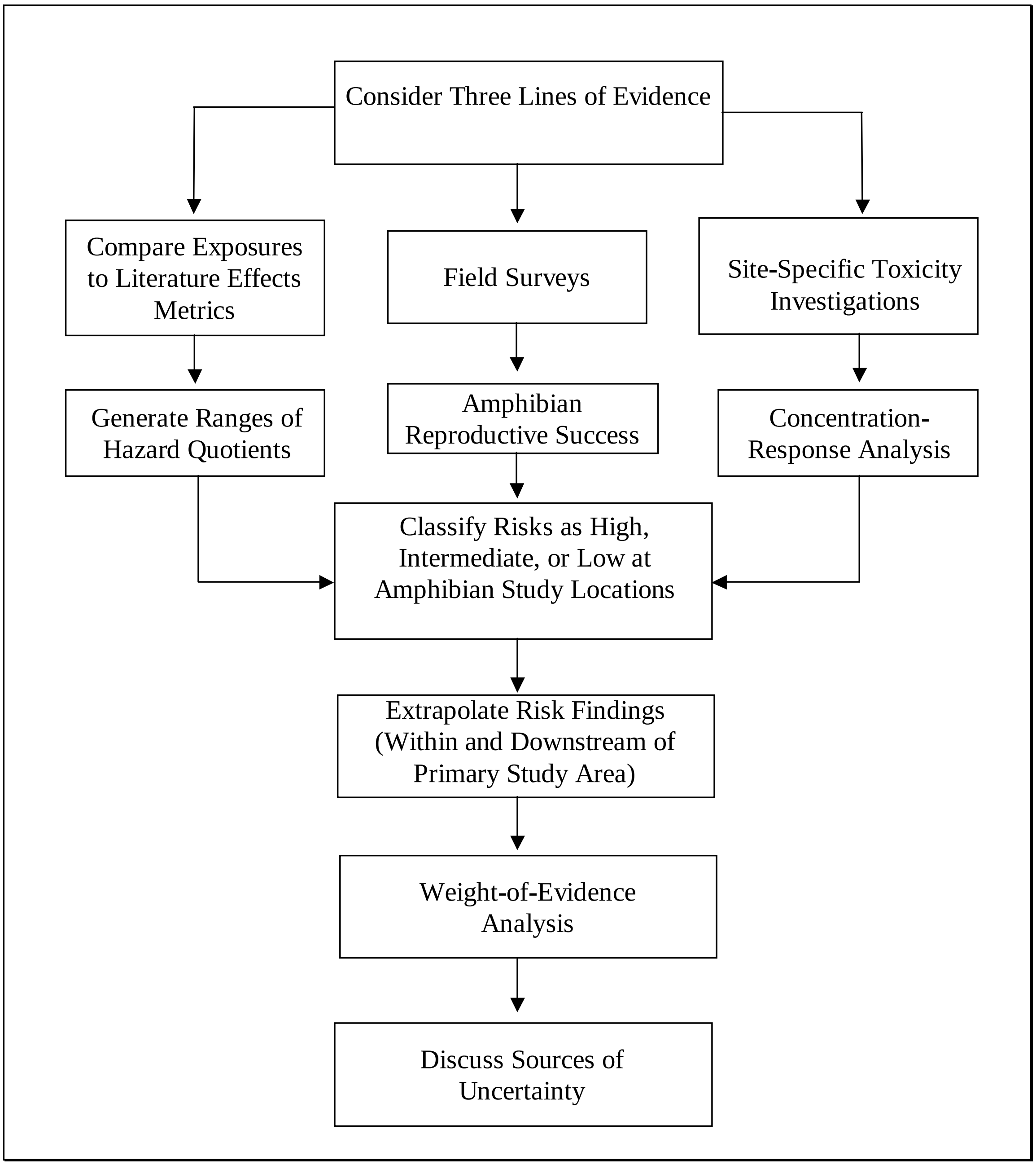


Figure 4.1-4 Overview of Approach Used to Characterize the Risks of Contaminants of Concern (COCs) to Amphibians in the Housatonic River PSA

4.2 CONCEPTUAL MODEL

The COCs identified for amphibians exhibit both direct (i.e., contact with contaminated source media) and indirect (i.e., food web bioaccumulation, maternal transfer) exposure pathways. The conceptual models presented in Figures 4.2-1 and 4.2-2 illustrate the exposure pathways for amphibians. The amphibian assessment focused on life stages that are in direct contact with sediment. For amphibian larvae, the dominant abiotic exposure media were sediment (solid phase and/or porewater) and surface water. Concentrations of COCs in tissues of amphibians provide an organism-based measure of bioavailability, and provide an additional line of evidence to consider along with the effects data gathered in the two frog developmental studies (FEL 2002a, 2002b).

Section 2, Problem Formulation, identified two indigenous species to be used in toxicity tests: leopard frogs (*Rana pipiens*) and wood frogs (*Rana sylvatica*). Detailed life history profiles for these species are provided in Appendices A and E.

The assessment endpoint that is the subject of this section is the maintenance of local populations of amphibians by ensuring the survival, reproduction, and development of local species. The measurement endpoints used to evaluate the assessment endpoint are presented below.

Measurement Endpoints for Amphibians

- Semiquantitative sampling of larval amphibians in breeding habitats with different sediment concentrations of stressors. Endpoints include species richness per habitat type; species abundance; gross pathology; and body, tail, and total length measurements.
- Surveys of vernal pools to quantify amphibians entering vernal pools and determine breeding behavior and condition; egg laying, hatching success, and larval growth and development; metamorphosis and emigration.
- Amphibian toxicity tests designed with exposure over a gradient of tPCB concentrations in site sediment. Toxicity endpoints include morphology of embryos and juveniles, limb development, and tail resorption of *Rana pipiens*.
- Gravidity of females; egg count; necrotic eggs; oocyte maturity; sperm count, morphology, and viability; fertilization rate; embryo viability; hatching success; mortality; and teratogenesis of amphibians collected from the study area compared with a control or reference.

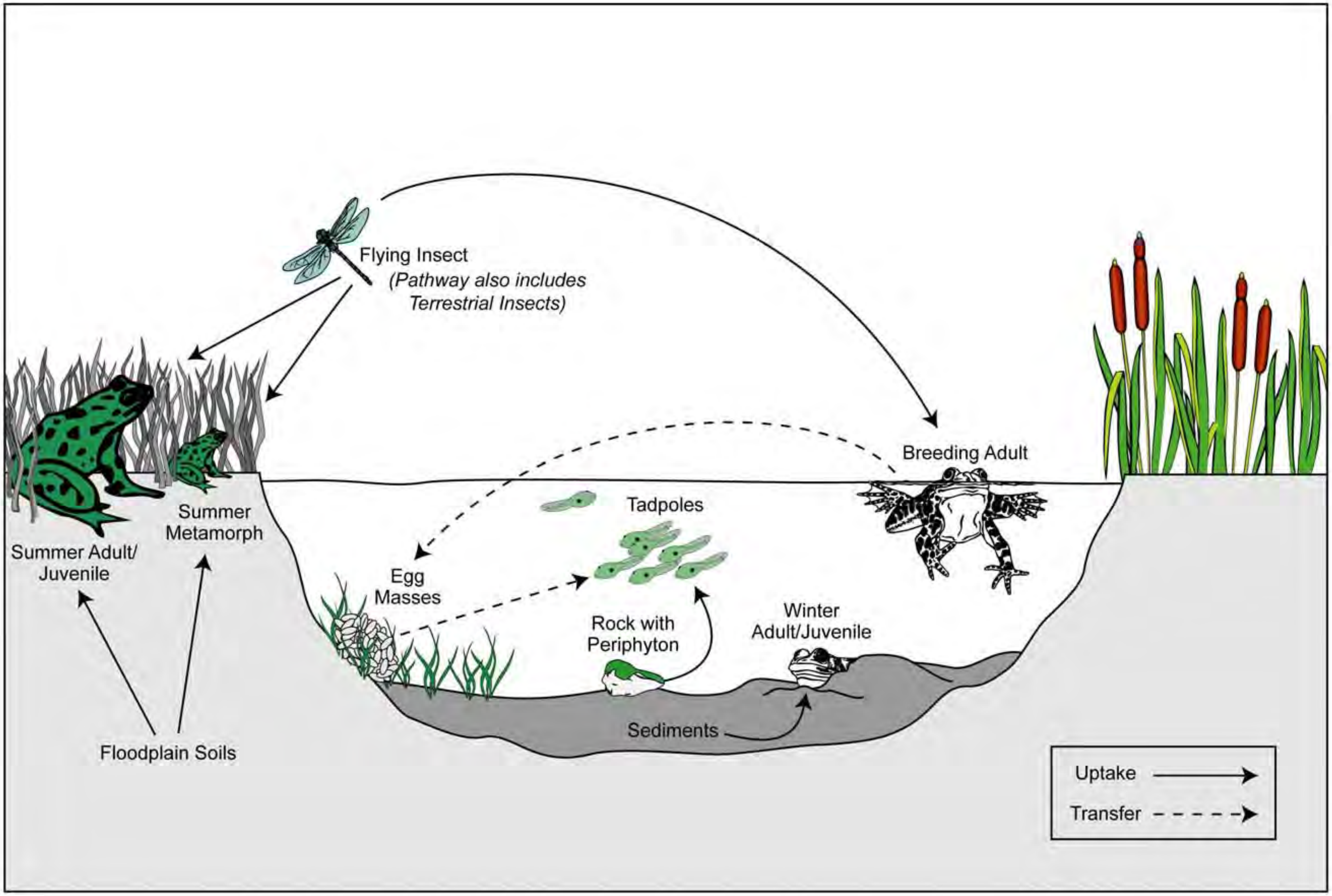


Figure 4.2-1 Leopard Frog Exposure Pathways

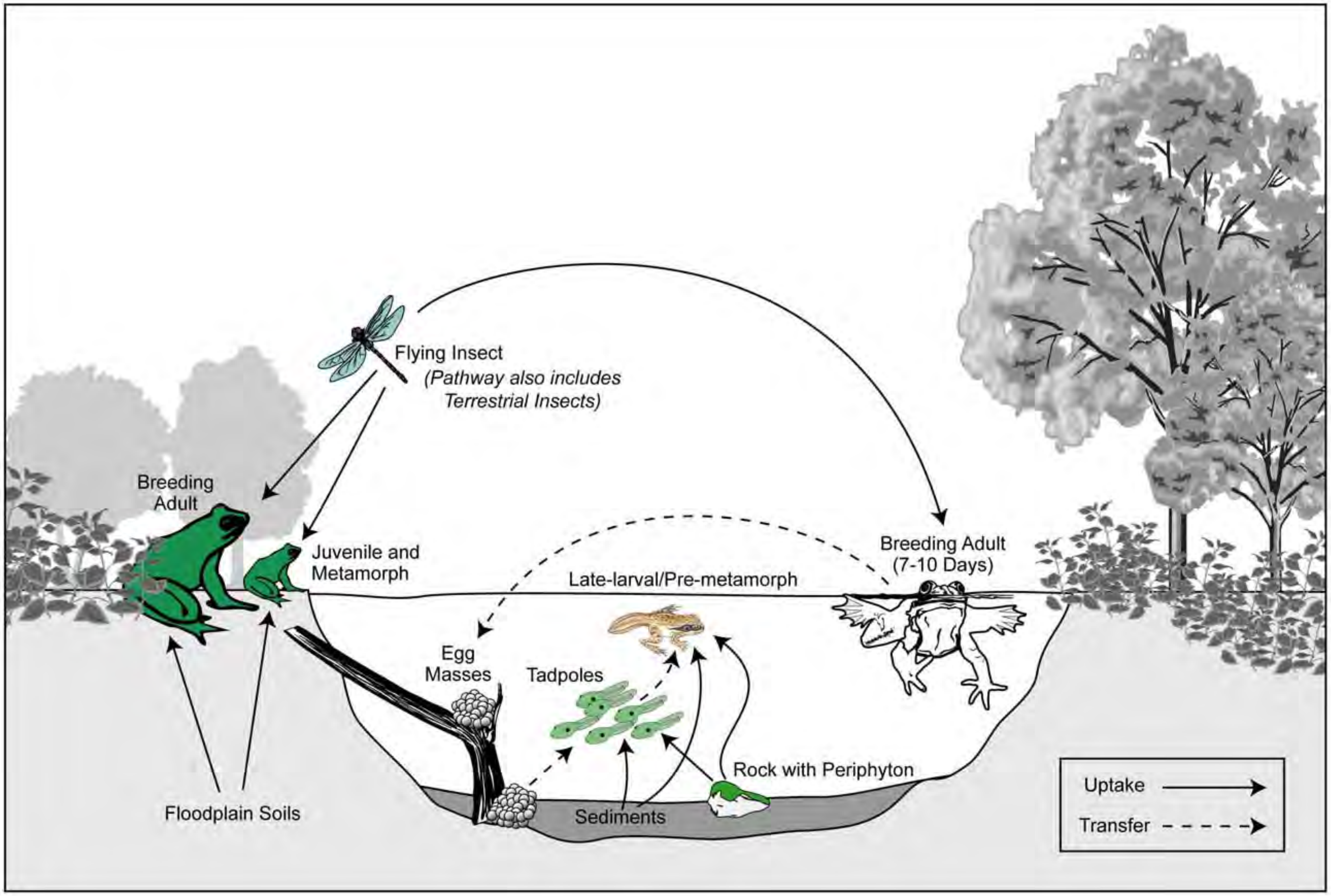


Figure 4.2-2 Wood Frog Exposure Pathways

4.2.1 Amphibian Developmental Studies

Three separate site-specific studies, two by EPA and one by GE, were conducted to evaluate reproductive performance and developmental effects in frogs exposed to PCBs and other COCs. The studies focused on reproduction, early development, and maturation (metamorphosis) in northern leopard frogs, and development and maturation in wood frogs. Various reproductive and developmental endpoints were assessed, such as gravidity of female frogs, egg mass fertilization and hatching success, larval and metamorph mortality, growth, and incidence of larval and metamorph malformation. Bioaccumulation of COCs in amphibian tissue was also evaluated.

4.2.2 Leopard Frog Study: EPA

This study (see Figure 4.2-3 for an overview of the study design) was intended to evaluate the reproductive fitness of adult leopard frogs and to monitor development of hatchlings through metamorphosis. Adult frogs were collected from nine sampling areas within the PSA; in addition to conducting an assessment of reproductive condition, the female frogs were to be fertilized in the laboratory, with the resulting larvae to serve as test organisms in the developmental portion of the study. However, this study plan could not be completed because the females had virtually no mature oocytes and fertilization was unsuccessful. In addition, the males exhibited a high proportion of malformed sperm heads. The study design was modified, and egg masses/hatchlings at five of the nine original sampling stations were collected and returned to the laboratory for developmental evaluation. In addition, external frogs were obtained from a commercial supplier and egg masses were cultured as control treatments. Larval mortality, malformation, growth, and incidence of metamorphosis were recorded. Figure 4.2-3 illustrates the general study design.

The study included cross-over treatments, in which control egg masses were cultured in contaminated sediment; the resulting larvae remained in the test media and were observed through metamorphosis. Also included were sediment spiking treatments, in which control egg masses were cultured in reference site sediment that had been spiked with 30 mg/kg Aroclor 1260. For both studies, tissue concentrations of COCs were measured in samples taken at the various leopard frog life stages (adult whole body, egg mass/ovary, and larvae whole body).

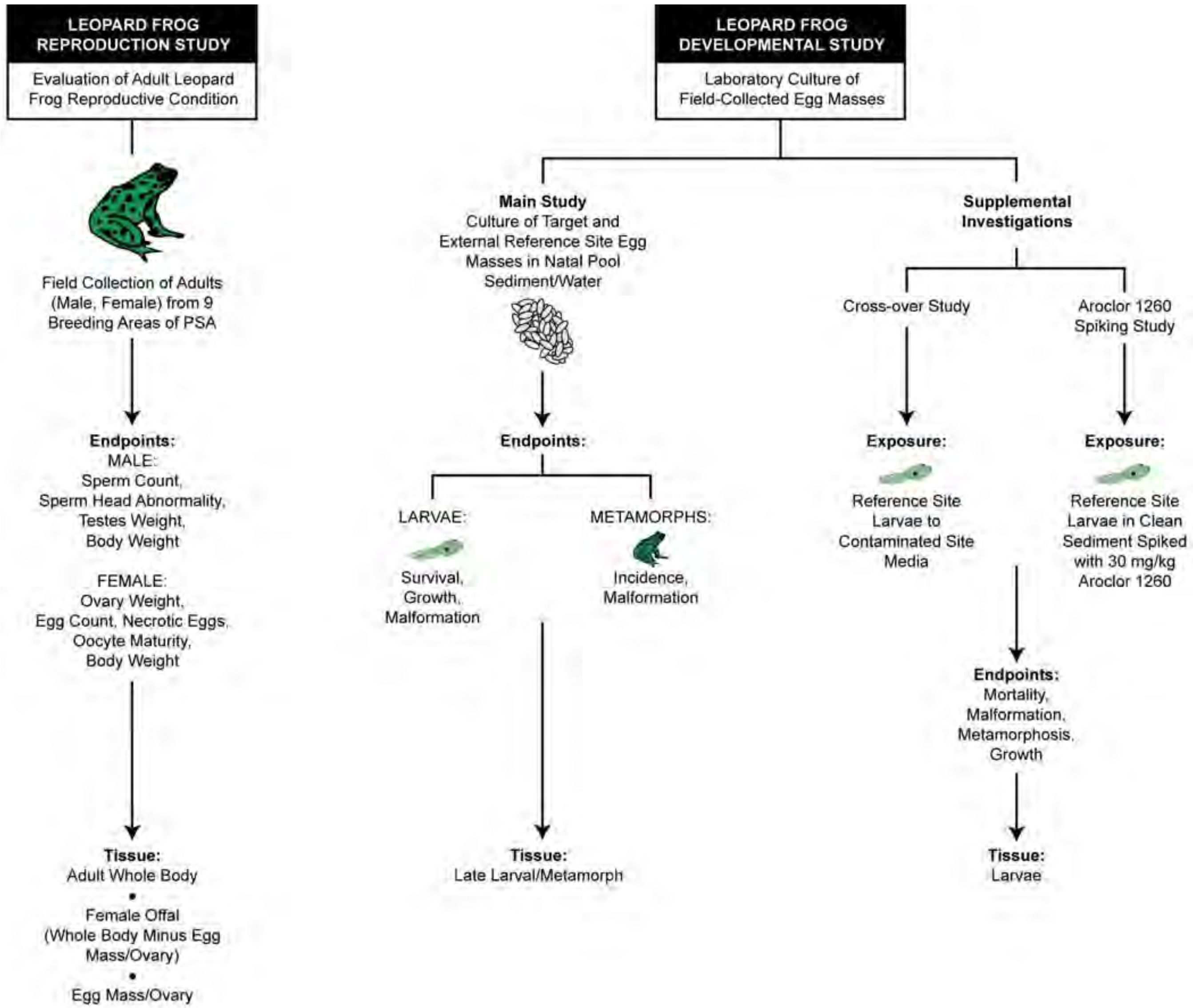


Figure 4.2-3 General Model of Leopard Frog Vernal Pool (VP) Reproduction and Development Study

4.2.3 Wood Frog Study Design (EPA Studies)

The wood frog study evaluated the growth, development, and maturation of wood frogs. The study design (Figure 4.2-4) combined laboratory exposures to vernal pool water and sediment and assessment of field-collected animals, and consisted of three separate phases.

In Phase I of the study, egg masses were collected from contaminated and reference vernal pools and cultured in the laboratory in sediment and water collected from the associated contaminated or reference pool. Egg mass fertilization, egg counts, egg weight, hatching success, larval growth, percent metamorphosis, and malformations were the endpoints for this phase. Egg mass tissue and metamorph samples were analyzed for tPCBs.

In Phase II, following natural hatching of egg masses in the pools, tadpoles were collected to assess in situ development. Endpoints for this phase included larval growth and malformations. Larval tissue samples from the first and third collection were analyzed for tPCBs.

In Phase III of the wood frog study, wood frog metamorphs were collected from the vernal pools, and individual weight, gender, and malformations were recorded. Tissue samples (one composite per pool) were analyzed for tPCBs and other COCs (PAHs, OC-pesticides, metals, and dioxins/furans).

4.2.4 Context-Dependent Wood Frog Study: GE

The objectives of this study were to evaluate the potential independent and interactive effects of larval density and PCB exposure on the survival and growth of wood frog larvae. Egg masses were collected from five vernal pools within the PSA and transported to a building at the GE facility, where they were maintained until hatching. Composite tissue samples were collected from each pool and analyzed for whole body PCB concentration. These hatchling (i.e., larvae) samples were considered to be indicative of the maternal transfer of contaminants.

Hatchling tissue tPCB concentrations and larval density were the basis for the creation of the experimental treatment groups. The study design specified three tissue concentrations (high, low, very low) and three levels of initial larval density (i.e., 200, 400, and 800). Each of these nine combinations of concentration and density was exposed in each of two vernal pools

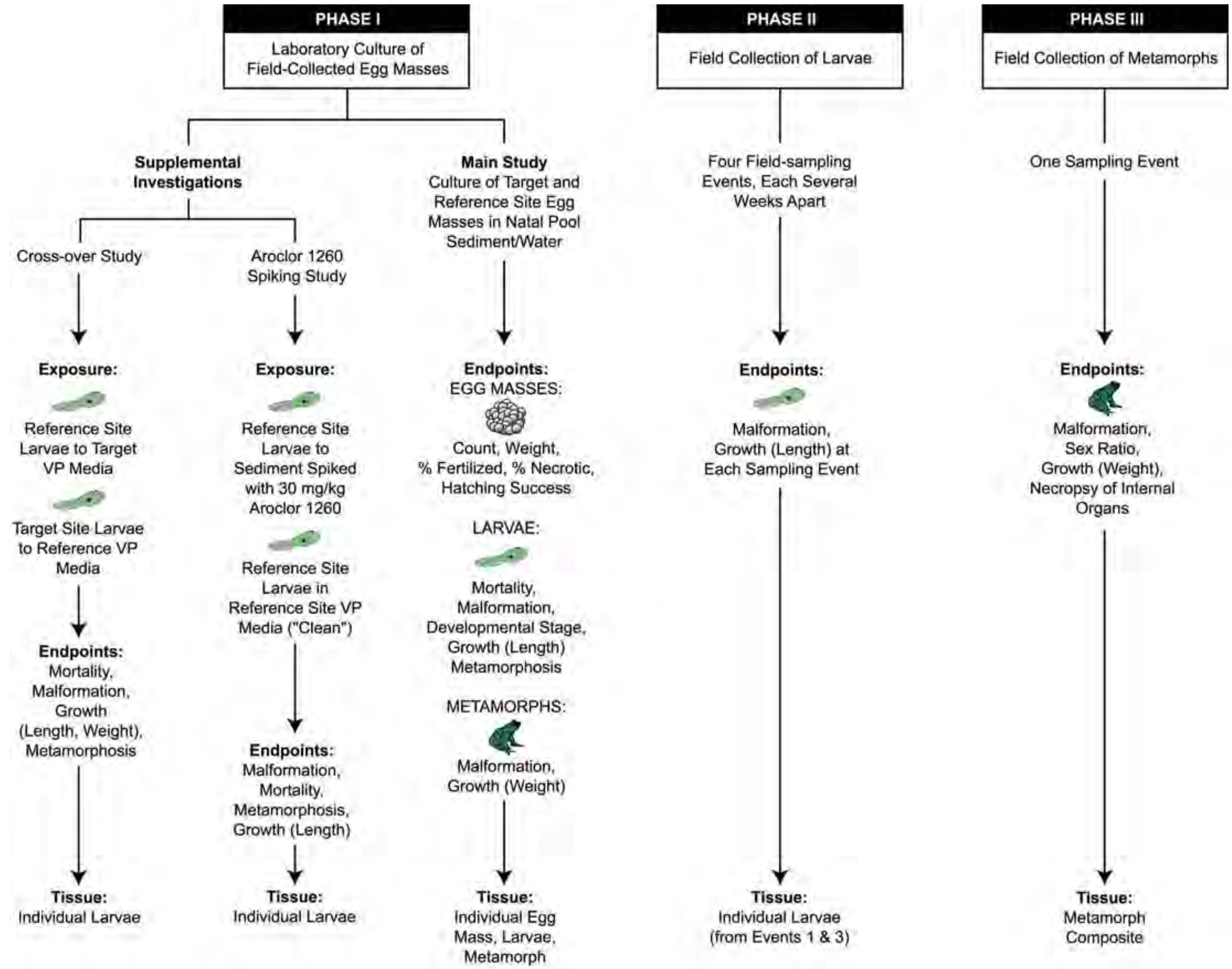


Figure 4.2-4 General Model of Wood Frog Vernal Pool (VP) Reproduction and Development Study

(23b-VP-1 and 23b-VP-2), thus providing a total of 18 experimental treatments. These two vernal pools were chosen because they supported natural populations of wood frogs, had very low concentrations of tPCBs, and were believed to be deep enough to hold water longer than most of the other floodplains in the PSA.

The survival and growth of both tadpoles and metamorphs were monitored, and the results were used to test whether any combination of initial larval density, hatchling tissue tPCB concentration, or vernal pool sediment tPCB concentration affected the number and weight of juveniles of each life stage.

4.3 EXPOSURE ASSESSMENT

Exposures to site contaminants were assessed as either the COC concentrations in sediment or water, or as tissue body burdens that represent integrated exposure from all sources. Routes of exposure were assessed to determine the contribution from maternal transfer and from bioaccumulation during later developmental stages.

To match exposure data with effects-based measures, many of the sediment tPCB data considered were from sampling conducted as part of the EPA studies that evaluated reproductive performance and developmental effects in frogs exposed to tPCBs and other COCs. Additional exposure data included spatially weighted sediment tPCB concentrations. These were calculated using all available sediment data, to develop average concentrations based on habitat types preferred by wood frogs and leopard frogs during the reproductive period. Additional sediment data were included in the GE wood frog study for the five pools of original egg mass selection and for the two experimental ponds used for the larval development portion of the study.

4.3.1 Selection of COCs for Amphibians

The contaminants initially considered in the amphibian exposure assessment were identified in the Pre-ERA (Appendix B). A refined screening was subsequently conducted on the sediment data collected in support of the two EPA amphibian studies. The sediment COCs retained for the amphibian assessment are presented below.

Contaminants of Concern for Amphibians

- Chlorinated organic compounds – tPCBs, dioxins/furans.
- Metals – Cadmium, chromium, copper, lead, mercury, silver.
- PAHs – Some individual PAH compounds, including low and high molecular weight PAHs.

4.3.2 Exposure Data

The approach used to characterize exposure to amphibians is shown in Figure 4.1-2. Dose-response relationships were investigated using both (1) the single "most synoptic" chemistry value paired with each toxicity endpoint; and (2) a combined data set, based on a spatially weighted average exposure concentration generated from all relevant sediment data. In most cases, the two approaches yielded similar results and helped to reduce the uncertainty associated with the use of either particular source of data.

4.3.3 Habitat Characterization

Sediment characterization was based on examination of gross physical parameters (such as total organic carbon [TOC] and grain size distributions) known to affect contaminant bioavailability. In addition, potential effects of the Pittsfield Wastewater Treatment Plant (WWTP) discharge were also examined with respect to the backwater habitats. Evaluation of effects from the WWTP was based on the assumption that backwater habitats receive a portion of their sediment load from the river main stem; therefore, influence from the WWTP may be carried into the backwaters.

Vernal Pool/Backwater Substrate Evaluation

- Vernal pool and backwater sediment is much richer in organic matter than main channel sediment, particularly in the upper Reach 5 area. The range of TOC for the amphibian sampling areas was 1.7 to 59.1%, with a median TOC concentration of 7% in the wood frog vernal pools, and 6% in the leopard frog sampling areas.
- Grain size distributions for the wood frog sampling areas in the PSA were fairly homogeneous and similar to reference sites. However, comparison of backwater sediment characteristics at locations in the leopard frog study indicated that the reference station had more coarse-grained sediment than did the sampling areas in the PSA.
- The relationship between tPCBs and TOC appeared qualitatively similar upstream and downstream of the WWTP (no excess organic enrichment attributed to the WWTP, thus, no confounding influence).

4.3.4 Assessment of Sediment Chemistry

4.3.4.1 Sources of Sediment Data

The sediment data used in the amphibian assessment included all floodplain samples collected from depths of 0 to 6 inches; all floodplain vernal pool samples collected in 1998 and 1999 to characterize floodplain sediment tPCB contamination; and all leopard frog and wood frog samples collected as part of the EPA amphibian developmental studies.

Sediment Data Sources Used in Amphibian ERA

- ***Vernal Pool Characterization Study:*** Sediment sampling in 66 temporary and permanent pools was conducted in 1998 and 1999 to characterize PCB contamination in floodplain habitats of the PSA and to select "representative" pools for amphibian developmental studies. PCB concentrations were fairly high; approximately 78% of the pools evaluated within the PSA had sediment tPCB concentrations > 5 mg/kg.
- ***Spatially Weighted Sediment/Floodplain Soil Concentrations*:** All available surficial sediment tPCB data were combined using spatial weighting to estimate average concentrations and exposure in wood frog vernal pools and leopard frog ponds and backwaters. The spatial weighting integrated wetland habitat types into the inverse distance weighting (IDW) procedure (Appendix C.3).
- ***Amphibian Developmental Studies*:** The sediment chemistry data with greatest synopticity with the effects endpoints in the amphibian ERA were data collected in conjunction with collection of the amphibian tissues (egg masses and larvae) during the developmental studies.
- ***GE Wood Frog Study*:** The source of the sediment chemistry data used in the study is not known, but the tPCB concentrations in the two experimental ponds known to be low: ≤ 0.3 mg/kg dw were similar to EPA concentrations. PCB concentrations in the 5 egg mass source ponds ranged from 0.5 to 30.8 mg/kg wet weight (ww) (0.3 to 32 mg/kg ww).

4.3.4.2 Distribution and Concentrations of PCBs

4.3.4.2.1 Vernal Pool Characterization Data

The 1998 to 1999 ecological characterization sampling resulted in more than 500 samples analyzed for tPCBs. Many of these samples were collected from vernal pools, with less than 10% collected from pool perimeters. A few of these samples were analyzed for other COCs: six for PCB congeners, seven for PAHs, eight for metals, and eight for dioxins/furans.

4.3.4.2.2 Spatially Weighted Average Exposure Concentrations

All surficial (0-15 cm) data collected in the PSA, combined with detailed habitat type maps and an understanding of site-specific hydrodynamics, were used to estimate spatially weighted surficial PCB concentrations. The spatial weighting approach is summarized in Appendix C.3, and the results are shown on Figures 2.5-5 through 2.5-11 of the ERA.

An exposure point concentration (EPC) was computed as the spatially weighted arithmetic mean of the cells contained within each leopard or wood frog sampling area boundary. Spatially weighted EPCs for the leopard frog sampling areas ranged from 0.4 to 44 mg/kg, with six of the nine areas greater than 20 mg/kg tPCBs. Spatially weighted EPCs for the wood frog vernal pools ranged from 0.2 to 99.5 mg/kg; approximately two-thirds were greater than 10 mg/kg tPCBs (see Figure 4.3-1).

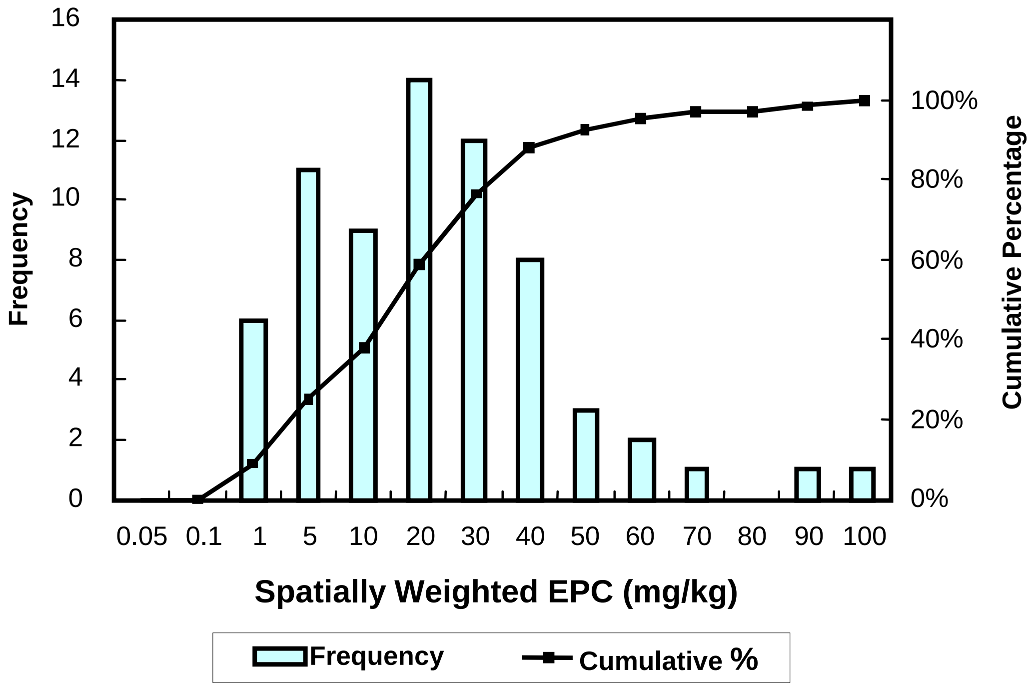


Figure 4.3-1 Frequency Distribution and Cumulative Percentage of Sediment tPCB Exposure Point Concentrations for 66 PSA Temporary and Permanent Pools (Based on EPA Spatially Weighted Data)

4.3.4.2.3 Data Collected in Support of the EPA Amphibian Developmental Studies

The sediment chemistry data with greatest synopticity with the effects endpoints in the amphibian ERA were data collected in conjunction with the collection of amphibian tissues (egg masses and larvae) during the EPA developmental studies. Figure 4.3-2 shows the sediment tPCB concentrations for both amphibian developmental studies. Pools 23b-VP-1 and 23b-VP-2, used as experimental ponds in the GE study, also are shown in Figure 4.3-2.

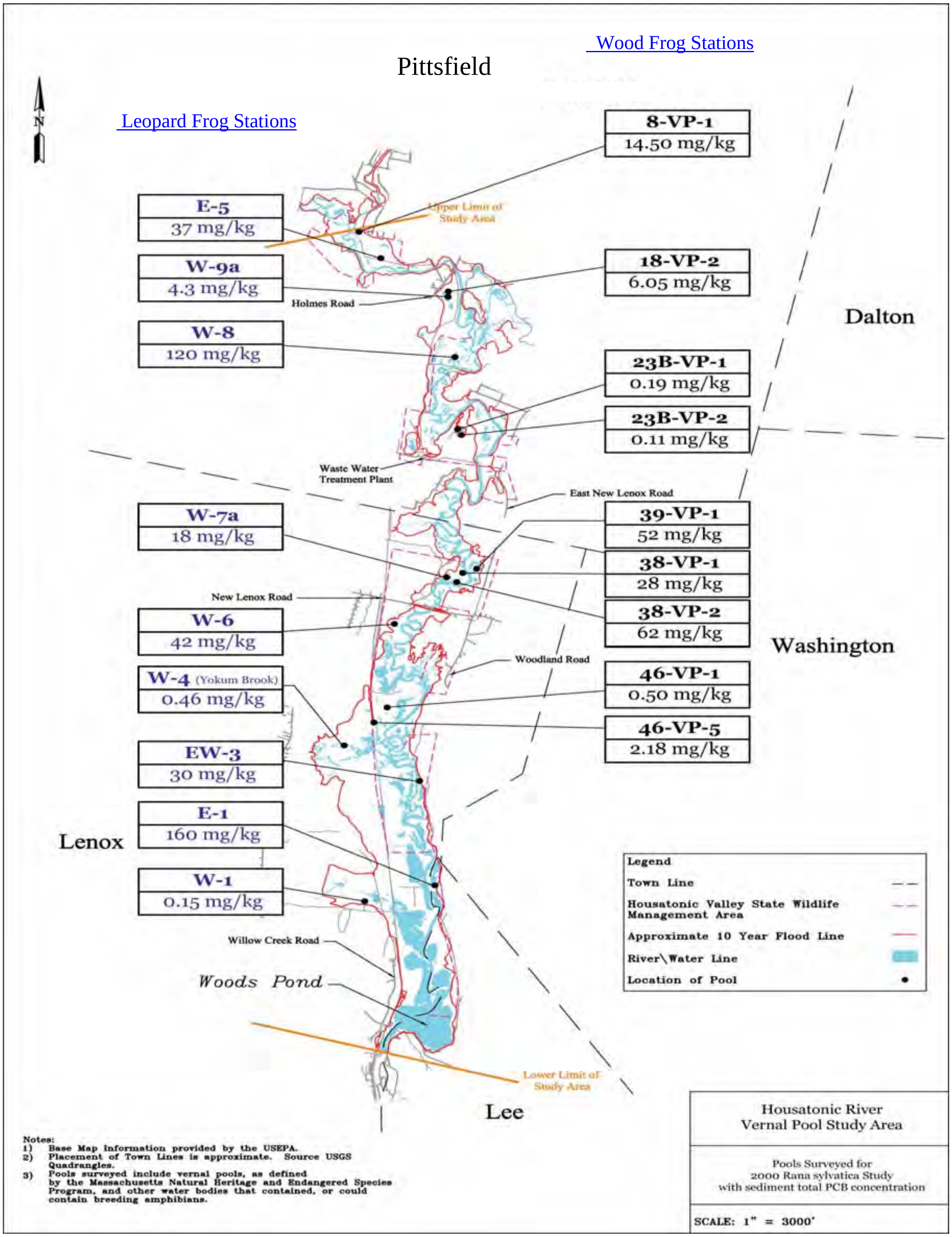


Figure 4.3-2 Total Sediment PCB Concentrations for Wood Frog Vernal Pool Study (mean, n =2) and Leopard Frog Reproduction/Development Study

Summary of Occurrence of Other COCs in Sediment Data

- Metals concentrations at leopard frog sites were similar to the Muddy Pond reference station; only Station E-1 had elevated metals concentrations. Metals in the wood frog vernal pools were similar to the reference stations.
- Total PAH concentrations were elevated at most leopard frog stations, relative to the reference site. PAH concentrations in the wood frog vernal pools were elevated at three stations. Elevated PAH compounds included acenaphthylene, benzo(a)anthracene, benzo(a)pyrene, benzo(b)fluoranthene, benzo(k)fluoranthene, and pyrene.

4.3.5 Surface Water Chemistry Assessment

Because of the uncertainty in relating water chemistry to effects on amphibians, the most relevant water data were those collected in conjunction with effects measurements (from EPA studies) conducted for the two EPA amphibian developmental studies.

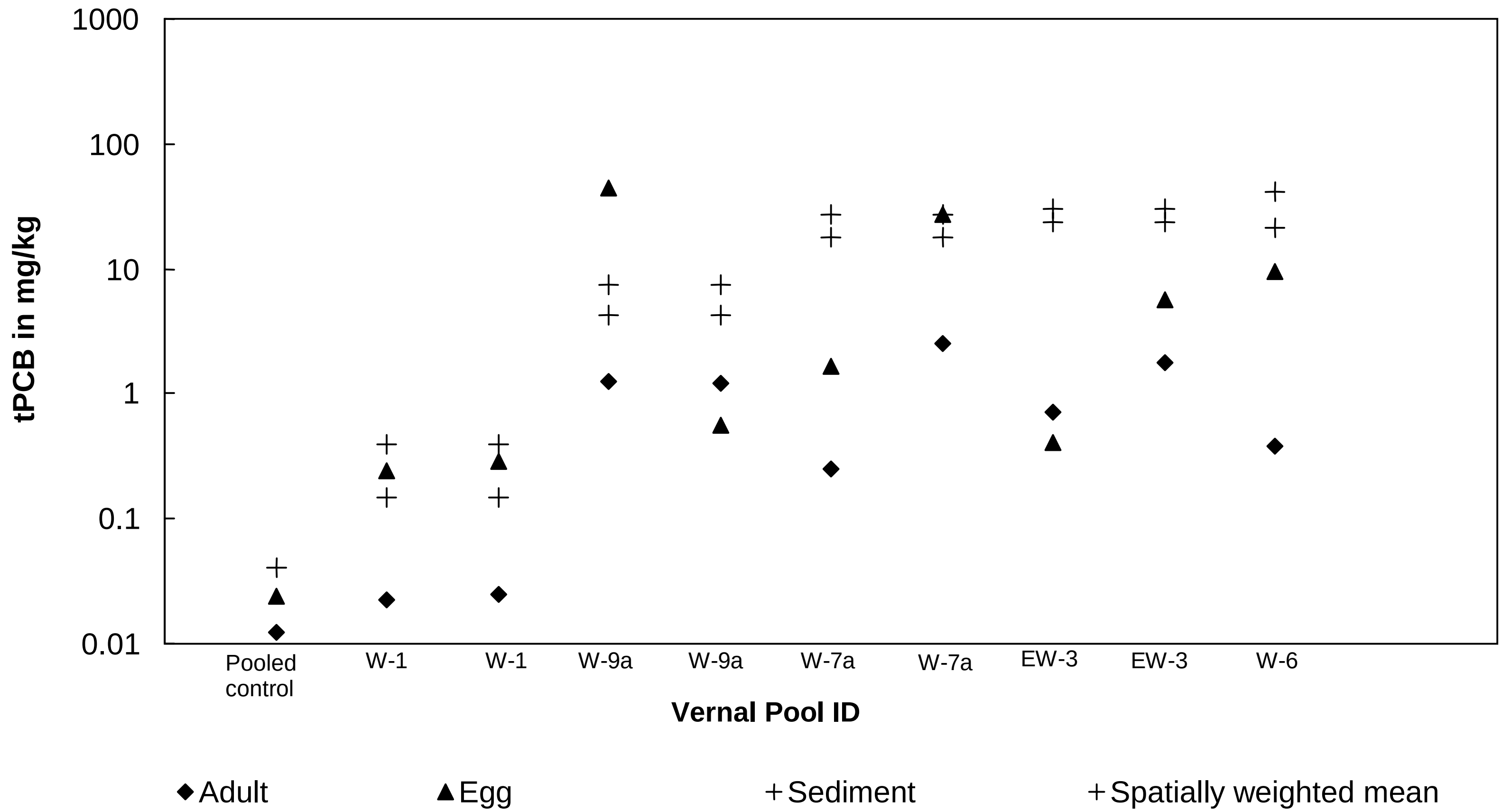
Surface water tPCB concentrations were lowest in the reference stations and the target stations with low sediment tPCB concentrations (0.01 to 0.03 μg/L for the wood frog study, 0.013 μg/L for the leopard frog study). Elevated water tPCB concentrations corresponded to amphibian sampling areas with elevated tPCB concentrations (0.1 to 0.47 μg/L for the wood frog study, 0.03 to 0.41 μg/L for the leopard frog study).

Elimination of Other COCs in Surface Water Data

- All metals screened in the sediment assessment were not detected in the two water samples, except for zinc. The measured value for zinc (17 μg/L) is below both the EPA federal and British Columbia provincial criteria for protection of aquatic life. Therefore, it was not retained as a COC.
- All PAHs were screened out of the water assessment because they were not detected.

4.3.6 Tissue Chemistry Assessment

Figure 4.3-3 presents the distribution of tPCB concentrations by sampling location and tissue type for adult samples collected at the leopard frog stations. Adult whole body tissue concentrations ranged from 0.15 to 5.4 mg/kg ww at PSA sites; Figure 4.3-3 shows whole body



Notes: Data include adult offal. N=2 for offal/egg mass samples (except for station W-6; n=1). N=1 for adult whole body samples.
All tissue concentrations are wet weight.

Figure 4.3-3 Comparison of Leopard Frog Tissue tPCB Concentrations (Laboratory) to Sampling Area Bulk Sediment Concentrations

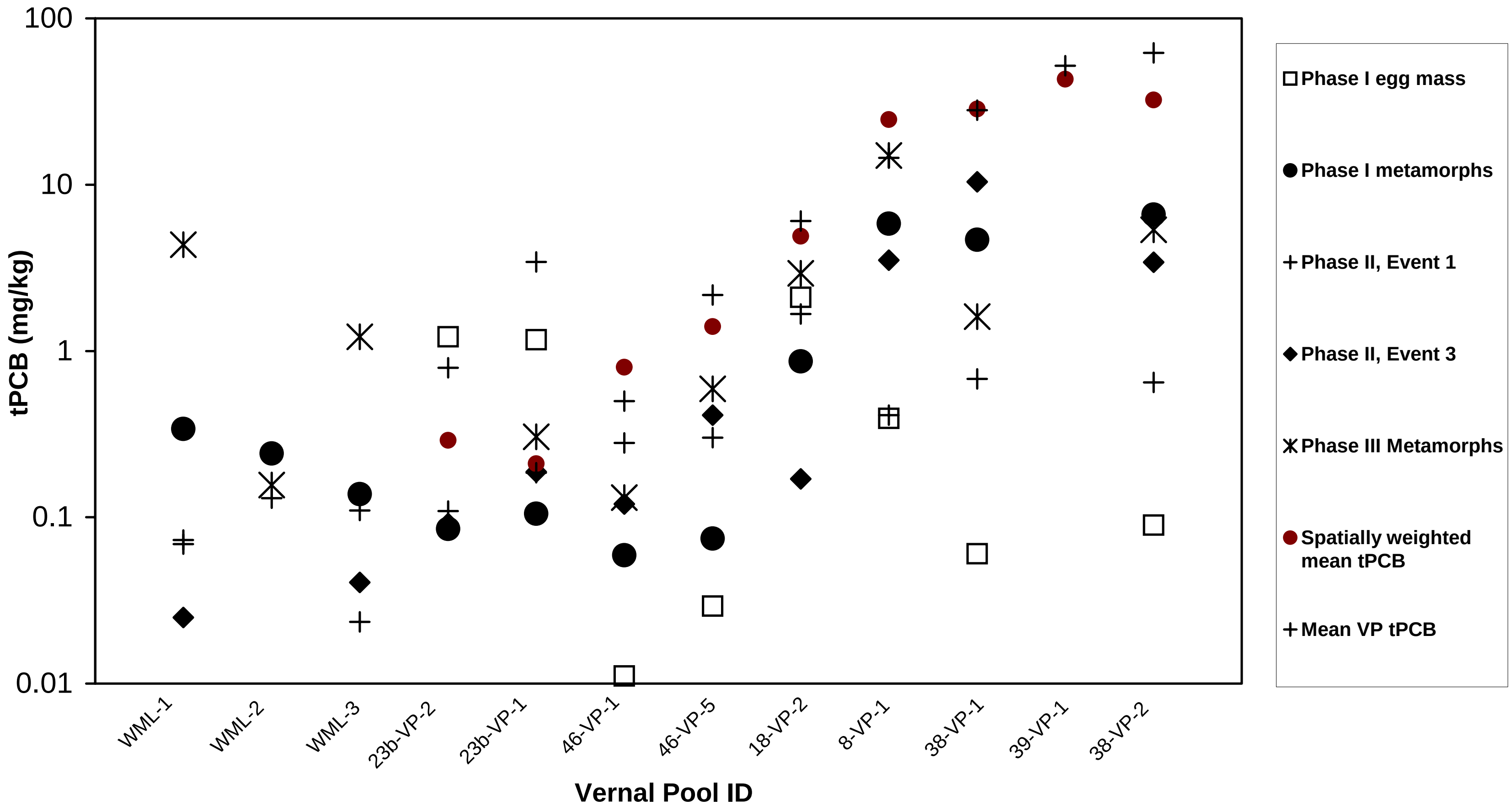
samples only from sites with associated offal and egg mass/ovary samples. For all tissue sample types, there was a direct relationship between sediment tPCB concentration and tissue tPCB concentration. Tissue samples from the cross-over and Aroclor 1260-spiked treatments confirmed the importance of the sediment uptake pathway.

Sources of Amphibian Tissue Chemistry Data

- ***Leopard frog study (EPA)*:** Nine composite whole body samples were analyzed for tPCBs. Nine individual offal samples with associated egg mass/ovary tissue removed were analyzed for tPCBs (Figure 4.3-3). Five offal samples were analyzed for other COCs. Five composite larval samples were analyzed for tPCBs. Six composite larval samples were analyzed for tPCBs from the cross-over and Aroclor 1260 spike treatments. All tissue samples confirmed contaminant bioavailability from sediment.
- ***Wood frog study (EPA)*:** 15 egg mass samples, 13 Phase I metamorph samples, 20 Phase II larval samples (from two discrete sampling events), 10 Phase III metamorph samples (five of these samples analyzed for other COCs) were analyzed for tPCBs. Four larval tissue samples from cross-over treatments and one larval tissue sample from the Aroclor 1260 spike treatment were also analyzed for tPCBs. Results of analysis showed that PCBs are bioavailable at all life stages, but that egg mass tissue concentration is not related to sediment tPCBs. However, all other tissue samples showed a trend of increasing contaminant uptake with increasing exposure concentration and duration in the vernal pools.
- ***Wood frog study (GE)***: Composite tissue samples were collected at two events during the juvenile period: at the hatchling stage (1 – 2 days post-hatch) and at the early larval stage (approximately 11 – 12 days post-hatch), one from each of the five vernal pool stations during the first event, and 4 composites during the second event (Pool 10.9 [EPA 40-VP-1] was not sampled).

Wood frog tissue concentrations (EPA study) for various life stages are included in Figure 4.3-4. PSA site egg mass tissue concentrations were unrelated to sediment tPCB concentrations. Contaminant concentrations in wood frog egg masses are more representative of the female's exposure prior to moving into the pool to breed. Phase I metamorph (laboratory-cultured) tPCB concentrations were related to sediment tPCB concentration. Phase II larval tissue samples (from collection event 1) were not related to sediment tPCB concentration. However, later-stage larval samples (from Phase II, collection event 3) were related to sediment tPCB concentrations. Phase III metamorph (field-collected) samples were also related to sediment tPCB concentrations.

In the cross-over and spiking studies, tissue tPCB concentrations were elevated in all samples exposed to PCBs and were much lower in the reference exposures.



Notes: No tissue samples for 39-VP-1; no Phase II, event 3 tissue data for WML-2; Phase I egg mass tissue values for WML-1, WML-2, and WML-3 were non-detect (ND) – values shown reflect detection limits (0.007, 0.004, and 0.008, respectively).

All tissue concentrations are wet weight.

* WML-1 Phase III metamorph sample is anomalous.

Figure 4.3-4 Comparison of Tissue tPCBs (in Various Phases of the Wood Frog Developmental Study) with Mean Vernal Pool and Spatially Weighted Mean tPCBs (Logarithmic Scale)

Hatchling tissue concentrations in the GE study ranged from 0.26 mg/kg ww to 11.2 mg/kg ww; no other COC tissue data were included in the report (n = 5 composites). The hatchlings from three of the five pools were then selected for placement in the in situ vernal pool enclosures. Just prior to placing the test organisms in the enclosures, larval tissue samples representing animals from the three hatchling test concentrations were analyzed for PCB body burden. Larval tissue concentrations ranged from 1.4 to 7.2 mg/kg ww tPCBs. A fourth larval tissue sample was also analyzed and contained 6.1 mg/kg ww tPCBs, although hatchlings from this location were not evaluated in the experimental treatments.

4.4 EFFECTS ASSESSMENT

The effects assessment for amphibians (Figure 4.1-3) emphasizes the site-specific field investigations; these studies provided direct indications of the bioavailability, toxicity, and effects of site-specific COCs. The effects assessment also includes an overview of the literature on the effects of tPCBs and other COCs on survival, growth, and reproduction of amphibians. Detailed evaluation of dose-response relationships, for both toxicity assessments and amphibian community structure assessments, are presented in the risk characterization section (Section 4.5).

4.4.1 Sediment Toxicity

4.4.1.1 Data Evaluation

Approximately 50 discrete endpoints, representing each major life stage of leopard and wood frogs, were initially evaluated for the ERA data analysis. The approach used to evaluate these endpoints was based on two objectives:

- Determination of relative sensitivity of various life stages.
- Evaluation of COC dose-response relationships.

The wood frog data were fairly well suited to the use of inferential statistics in the evaluation of relationships. The leopard frog data, however, required a more qualitative approach.

4.4.1.1.1 Leopard Frog Data

Non-parametric Spearman's correlation coefficients were calculated to evaluate the relationship between adult tissue concentrations and sediment tPCB concentrations. Data from the cross-over and spiked studies were evaluated via hypothesis testing for differences among groups (always a two-sample comparison of a control treatment and a target treatment of interest).

4.4.1.1.2 Wood Frog Data: EPA Study

Non-parametric Spearman's correlation coefficients were calculated to evaluate the relationships between biological effects and PCB concentration at sensitive life stages. The results were compared against literature-based effects data, and provided a check against spurious correlations.

As with the leopard frog data, the cross-over and spiked study data were evaluated via hypothesis testing for differences among groups (a two-sample comparison of a control treatment and a target treatment of interest).

4.4.1.1.3 Wood Frog Data: GE Study

Multivariate and univariate analysis of variance (MANOVA and ANOVA) were used to evaluate the interaction of vernal pool, hatchling tissue tPCB concentration, and initial larval density on the survival and growth of the test organisms. In addition, correlation analyses were used to evaluate potential relationships between hatchling and larval tissue concentration and the sediment tPCB concentration of the vernal pool from which the egg masses were collected.

4.4.1.2 Results

4.4.1.2.1 Leopard Frog Study

Reproductive Fitness: Adult leopard frogs were collected between March 25 and April 22, 2000. The timing of collection of adult leopard frogs from the target stations coincided with the normal onset of reproductive receptiveness and initiation of breeding activity. Insufficient numbers of frogs were present at the three reference areas; therefore, control animals from nearby in Vermont were purchased from Carolina Biological Supply (CBS), a commercial supplier.

Female frog gravidity was recorded, and mature (gravid) females were induced to produce egg masses; fertilization was then attempted on these egg masses. The number of eggs produced per female, rates of necrosis, and oocyte developmental stage distribution were determined. Sperm count, morphology, and overall viability were also assessed. The eggs were monitored for fertilization, morphology, and coloration.

Male body weight and sperm count did not appear to be related to exposure media tPCB concentrations; however, there was a strong relationship between incidence of sperm head abnormalities and sediment tPCB concentrations (Table 4.4-1 and Figure 4.4-1).

Findings regarding reproductive fitness of the female leopard frogs included (see Table 4.4-2):

- Leopard frogs from contaminated areas in the PSA showed signs of reproductive stress.

Table 4.4-1

Summary of Male Adult Leopard Frog Reproductive Health

Sampling Area ID	Sediment tPCBs (mg/kg)*		Mean Total Water PCBs (μg/L)	Mean Male Body Weight (g) (std. dev.)	Mean Testes Weight (% of Total Body Weight) (std. dev.)	Mean % Abnormal Sperm Heads (std. dev.)	Mean Sperm Count x 10^6/ g Gonad Tissue (std. dev.)
	tPCBs	Sp. Wt. PCBs					
C1	-	-	-	40.6 (6.29)	0.176 (0.016)	0.42 (0.32)	5.38 (1.25)
C2	-	-	-	40.8 (4.75)	0.126 (0.083)	0.89 (0.51)	7.88 (5.94)
C3	-	-	-	35.9 (1.48)	0.179 (0.039)	2.36 (0.49)	3.61 (0.56)
pooled C1, C2, C3	-	-	-	39.2 (4.91)	0.162 (0.051)	1.14 (0.95)	5.60 (3.40)
W-1	0.15	0.4	0.013	36.5 (4.08)	0.163 (0.044)	4.33 (0.76)	5.06 (2.86)
W-4	0.46	0.4	0.013	37.2 (NA)	0.140 (NA)	3.15 (NA)	4.48 (NA)
W-9a	4.3	7.5	0.013	40.7 (4.82)	0.147 (0.023)	8.26 (2.39)	2.42 (0.87)
W-7a	18	27.6	0.03	34.2 (4.93)	0.106 (0.040)	12.0 (4.72)	3.52 (1.67)
EW-3	30	23.8	0.41	31.4 (4.28)	0.112 (0.026)	49.5 (10.8)	7.87 (0.28)
W-6	42	21	0.22	39.6 (5.28)	0.138 (0.058)	37.3 (6.26)	2.01 (1.30)
W-8	120	43.5	0.14	34.8 (11.8)	0.089 (0.010)	42.7 (2.39)	6.08 (0.20)
E-1	160	26.6	0.24	41.4 (7.39)	0.113 (0.075)	14.3 (4.43)	3.83 (3.26)

* tPCBs = Value from amphibian developmental studies; Sp. Wt. PCBs = mean tPCBs for each sampling area based on spatial weighting of sediment data.

std. dev. = Standard deviation.

Table 4.4-2

Summary of Female Adult Leopard Frog Reproductive Health

Sampling Area ID	Sediment tPCBs (mg/kg)*		Total Water PCBs (µg/L)	Mean Female Tissue PCBs (mg/kg) (std. dev.)	Mean Egg Mass/Ovary PCBs (mg/kg) (std. dev.)	Mean Female Body Weight (g) (std. dev.)	Proportion Gravid	Mean Ovary Weight (% of Body Weight) (std. dev.)	Mean Total Egg Count (std. dev.)	Mean % of < Stage III Oocytes (std. dev.)	Mean % of Stage VI Oocytes (std. dev.)
	tPCBs	Sp. Wt. PCBs									
C1	-	-	-	0.012 (0.014)	0.036 (0.007)	77.41 (5.55)	4/4	30.78 (3.87)	1264 (908)	23.01 (32.75)	56.75 (25.20)
C2	-	-	-	0.011 (0.013)	0.012 (0.005)	79.32 (15.26)	4/4	23.32 (5.71)	2811 (1342)	27.82 (8.93)	65.25 (7.22)
C3	-	-	-	0.017 (0.010)	0.024 (0.003)	76.10 (12.33)	5/5	32.14 (5.44)	119 (49)	0	89.46 (6.16)
Pooled C1, C2, C3	-	-	-	0.015 (0.010)	0.024 (0.011)	77.61 (11.14)	13/13	29.01 (6.14)	1174 (1339)	14.62 (21.85)	72.50 (20.78)
W-1	0.15	0.4	0.013	0.022 (NA)	0.240 (NA)	48.83(18.28)	1/5	4.56 (4.59)	1008 (961)	84.86 (30.29)	0.44 (0.89)
W-4	0.46	0.4	0.013	-		43.26 (10.69)	2/2	6.23 (0.36)	1038 (110)	28.38 (7.90)	2.22 (0.60)
W-9a	4.3	7.5	0.013	1.260 (NA)	45.086 (NA)	51.59 (12.04)	0/3	3.88 (1.28)	1238 (799)	99.70 (0.52)	0
W-7a	18	27.6	0.03	1.407 (1.636)	14.219 (17.801)	52.43 (12.87)	5/5	21.35 (2.88)	2918 (1663)	20.00 (44.72)	4.88 (3.04)
EW-3	30	23.8	0.41	-		55.73 (4.97)	2/3	13.04 (5.67)	419 (436)	67.93 (29.50)	1.02 (1.37)
E-5	37	19.6	0.043	-		50.33 (NA)	0/1	1.25 (NA)	177 (NA)	100	0
W-6	42	21	0.22	0.386 (NA)	9.477 (NA)	57.35 (12.66)	2/5	4.65 (1.99)	2401 (841)	97.52 (3.39)	0.06 (0.13)
W-8	120	43.5	0.14	-		52.66 (11.66)	0/1	5.03 (NA)	307 (NA)	100	0
E-1	160	26.6	0.24	-		37.91 (9.73)	0/4	3.17 (0.97)	1168 (733)	99.26 (1.55)	0

* tPCBs = Value from amphibian developmental studies; Sp. Wt. PCBs = mean tPCBs for each sampling area based on spatial weighting of sediment data.
std. dev. = Standard deviation.
NA = Not applicable; only 1 replicate.
Note: All tissue concentrations are wet weight.

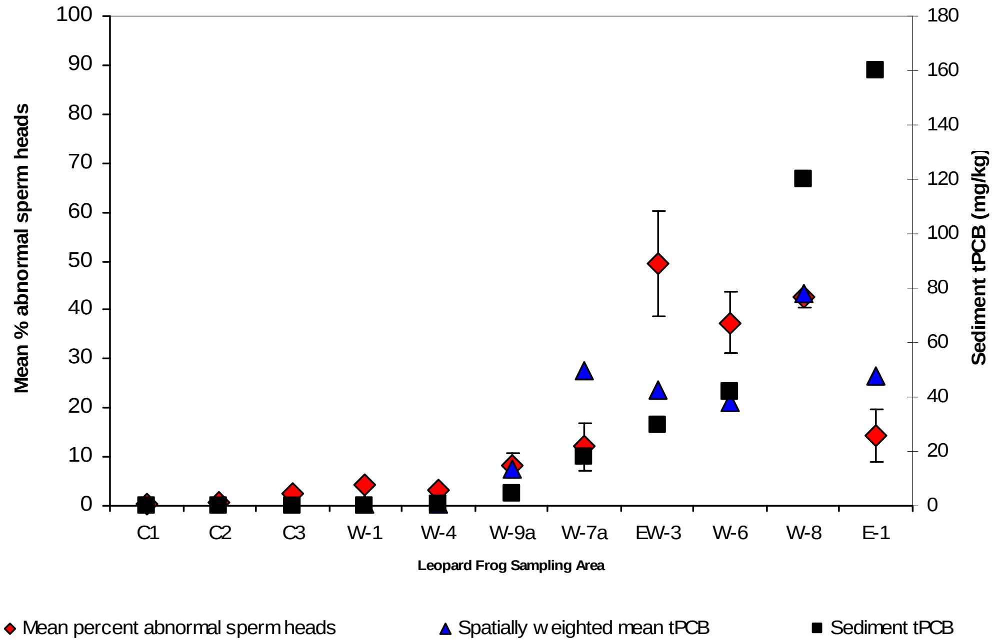


Figure 4.4-1 Comparison of Percent Abnormal Sperm Heads (Mean) from Male Adult Chemical Analysis Leopard Frogs, with Mean Sediment tPCBs and Spatially Weighted Mean tPCBs

- None of the females collected from Sites E-5, W-9a, W-8, and E-1 (37.0, 4.3, 120.0, and 160.0 mg/kg dw sediment tPCBs, respectively) were found to be gravid.
- Few of the PSA sites produced female specimens that possessed any biologically significant quantity of Stage VI oocytes (mature eggs capable of fertilization), with the exception of Station W-7a (Figure 4.4-2).
- Even though more advanced oocytes were found in specimens containing greater concentrations of ovary tPCBs, only a few Stage VI oocytes were found, indicating that the final stage of maturation may have been inhibited.
- Control frogs (collected in Vermont) were larger than most contaminated site frogs, and had relatively larger ovaries. Because the control frogs were collected in a similar climate, region, and timeframe, differences in body sizes were not expected.

In summary, the evidence supporting impairment of reproductive fitness related to PCB exposure includes:

- Low rates of egg maturation.
- Poor egg mass fertilization from field-collected female frogs.
- High incidence of sperm head abnormalities in males from pools with high sediment PCB concentrations.

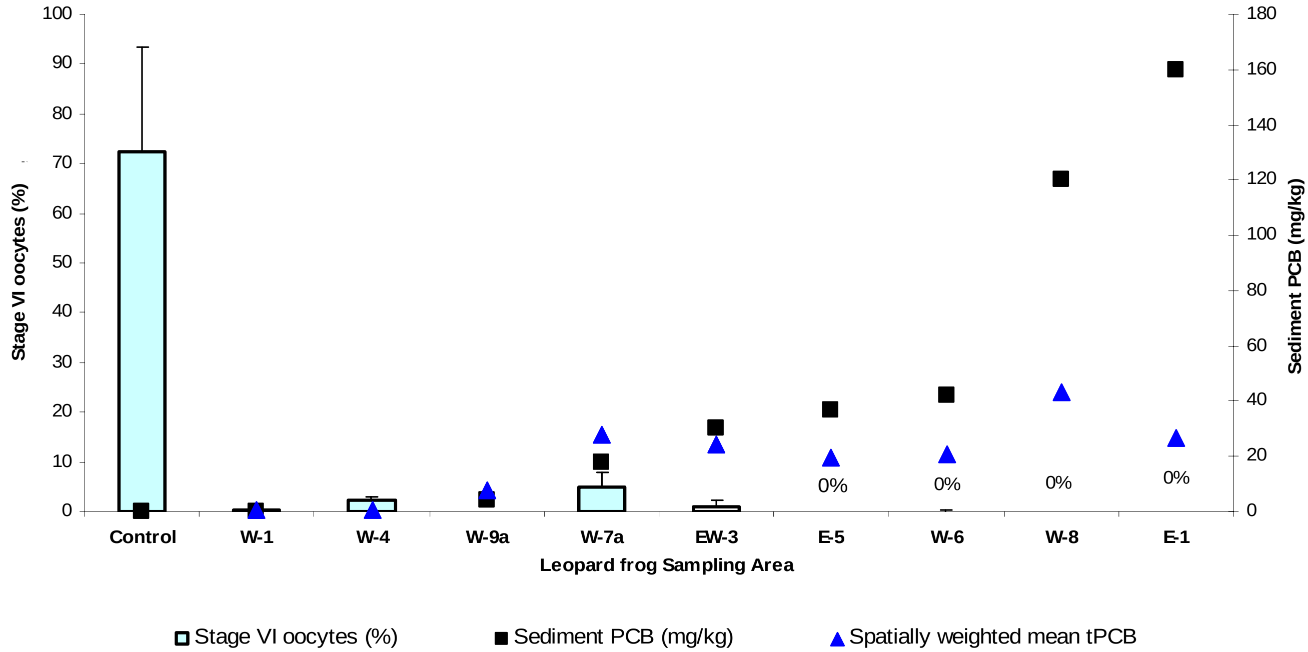


Figure 4.4-2 Comparison of Mean Percent of Oocytes at Stage VI (Mature) for Female Leopard Frogs, with Mean Sediment tPCBs and Spatially Weighted Mean tPCBs

Developmental Endpoints: Because of poor egg fertilization success, where most of the females from the PSA sampling areas were reproductively unfit, the study design was modified to include the field collection of egg masses from the leopard frog sampling areas, and to raise them in the laboratory, as was done for wood frogs. Egg masses and hatchlings were collected at five of the nine contaminated sampling areas. No egg masses were found in the two locations with the highest sediment PCB concentrations (Stations W-8 and E-1), or at the reference areas. Therefore, control egg masses were obtained from the female leopard frogs from CBS, which were fertilized in the laboratory.

Larval endpoints measured in the study included:

- Malformations
- Growth
- Metamorphosis.
- Mortality.
- Days to reach Gosner developmental Stage 26.
- Developmental stage reached at EOT.

Gosner Stage 26 is the point in development when tadpoles go from a relatively immobile embryo to an active, feeding tadpole. EOT was used to designate the shorter test duration (the last day that larval growth and malformation were measured). Final test duration refers to the last day that larval mortality and metamorph data were recorded.

Table 4.4-3 and Table 4.4-4 show the responses of the four developmental endpoints from the main study, as well as the results of the cross-over and Aroclor 1260-spiked treatments. Findings with respect to the leopard frog developmental endpoints are summarized in the following paragraphs. The study results indicate that some endpoints demonstrate a very strong toxic response to PCBs, even at low concentrations.

- Mortality was high (85 to 100%) for larvae raised in contaminated sediment regardless of PCB concentration, compared to control larvae (44%) raised in Muddy Pond sediment and water.
- The incidence of larval malformations was low (0 to 3.4%) in sampling locations with tPCB concentrations below 1 mg/kg, and higher (46 to 54%) in sampling locations with tPCBs greater than 20 mg/kg. Malformations were similar to those observed in studies of exposure of other frog species to PCBs and similar contaminants (Birge et al. 1978; Eisler and Belisle 1996; Gutleb et al. 1999, 2000).
- Larval developmental delay was observed in leopard frogs raised in contaminated sediment. There was an obvious relationship between sediment tPCBs and the amount of time for the larvae to reach Stage 26 (±1) (Figure 4.4-3). Larvae from the two most contaminated stations (W-7a and W-6) never developed beyond Stage 26 (±1).
- Extended time to metamorphosis, and a low incidence of metamorphosis, was observed in juveniles from the PSA sites.

Table 4.4-3

Summary of Leopard Frog Larval Development Endpoints Data at End-of-Test

Sampling Area ID[a]		Test Duration[b]	Initial Larval Count	Sediment tPCBs (mg/kg)[c]		Water tPCBs (µg/L)	Living Larvae at End of Test (EOT)	Mean % Larval Mortality at EOT	Mean % Metamorph at EOT	Mean % Larval Malformed (Based on Surviving Larvae)	Days to Reach Stage 25 (±2)	Develop. Stage at EOT	Mean Larval Growth: Length at EOT (cm)
				tPCBs	Sp. Wt. PCBs								
EW-3		22	13	30	23.8	0.41	NA	100	NA	NA	NA	22	NA
W-6		91	98	42	21	0.22	11	88.8	0	54.5	91	25-27	4.48
W-1		105	105	0.15	0.4	0.013	14	86.7	0	0	49	37-40	3.88
W-7a		105	105	18	27.6	0.03	16	84.8	0	45.8	105	25-27	4.47
W-4		111	210	0.46	0.4	0.013	15	92.8	0.83	0	55	36-37	4.55
MP Ref.		69	160	0.04	-	0.013	125	21.8	0	3.4	13	38	4.30
Cross-over Study	C1 Target	91	40	120	-	0.14	19	52.5	2.5	25.9	91	26	4.06
	C3 Target[d]	69	80	120	-	0.14	70	12.5	0	26.1	69	32	4.39
	Reference[d]	69	160	0.04	-	0.013	125	21.8	0	3.4	69	38	4.3
Aroclor 1260 Spike Study	Spiked	23	80	30	-	-	57	28.7	NA	29.8	NA	NA	NA
	Control	23	80	0.04	-	-	59	26.2	NA	0	NA	NA	NA

[a]Sampling areas arranged in order of increasing test duration.

[b]Durations vary for endpoints; larval malformation and growth had shorter test durations than larval mortality/metamorphosis. Test durations shown here are for the malformation/growth endpoints. Last day of test duration shown here is used as end-of-test (EOT) for a given sampling area.

[c]tPCBs = Value from amphibian developmental studies; Sp. Wt. PCBs = mean tPCBs for each sampling area based on spatial weighting of sediment data.

[d]Treatments used in hypothesis testing.

NA = Not applicable.

Table 4.4-4

Summary of Leopard Frog Larval Development Endpoints at Final Test Duration

Sampling Area ID[a]		FEL Site ID	Final Test Duration[b]	Initial Larval Count	Sediment tPCBs (mg/kg)[c]		Living Larvae and Metamorphs at End of Test (EOT)	Final Mean % Larval Mortality	Living Metamorphs (EOT)	Mean % Metamorph (at Final Test Duration)
					tPCBs	Sp. Wt. PCBs				
EW-3		37	28	13	30	23.8	NA	100	0	0
W-6		35	128	98	42	21	8	91.8	3	3.1
W-1		39	142	105	0.15	0.4	9	91.4	0	0
W-7a		34	142	105	18	27.6	13	87.6	1	0.95
W-4		36	148	210	0.46	0.4	9	95.7	2	0.95
MP Ref.		40	106	160	0.04	-	90	43.8	9	5.6
Cross-over Study	C1 Target	-	128	40	120	-	19	60.0	2	5.0
	C3 Target[d]	-	106	80	120	-	70	32.5	7	8.8
	Reference[d]	-	106	160	0.04	-	125	43.8	6	5.6
Aroclor 1260 Spike Study	Spiked	-	23	80	30	-	57	28.7	NA	NA
		-	23	80	0.04	-	59	26.2	NA	NA

[a]Sampling areas arranged in order of increasing test duration.

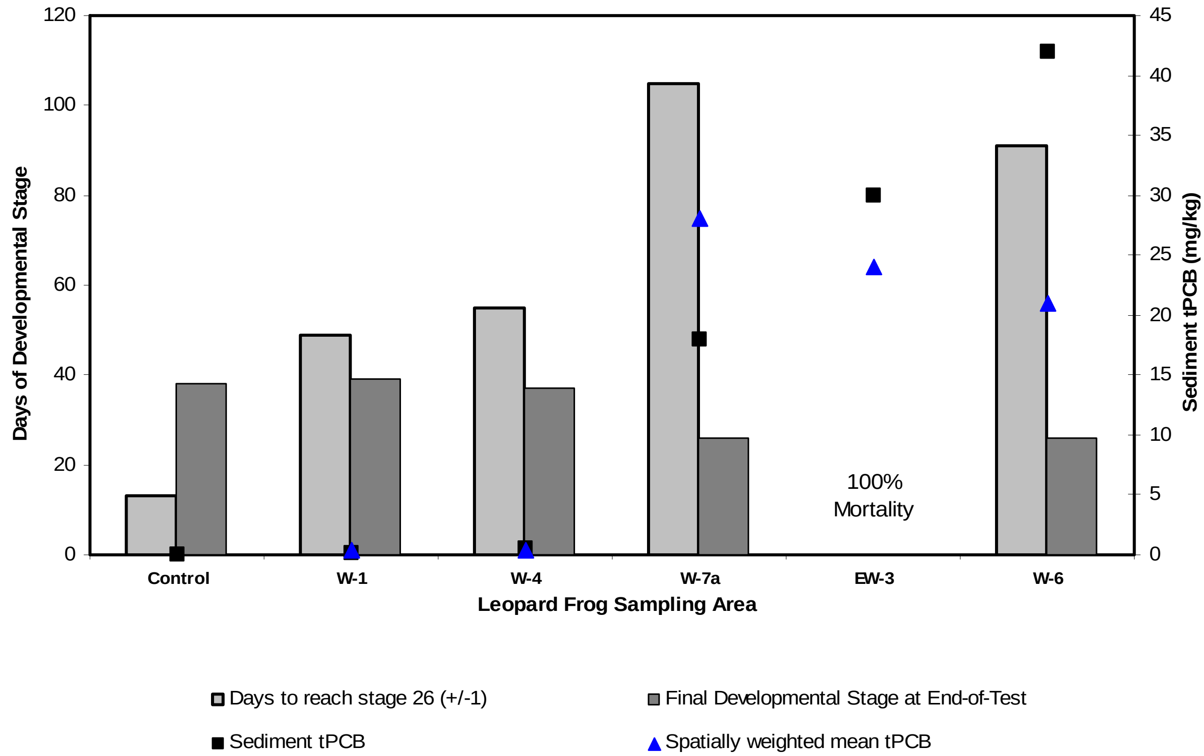
[b]Test durations for the larval mortality/metamorphosis endpoints were longer than for the larval malformation/growth endpoints. Endpoint measures in this table correspond to final test durations.

Control

[c]tPCBs = Value from amphibian developmental studies; Sp. Wt. PCBs = mean tPCBs for each sampling area based on spatial weighting of sediment data.

[d]Treatments used in hypothesis testing.

NA = Not applicable.



Test durations:

Control	69 days
W-1	105 days
W-4	111 days
W-7a	105 days
EW-3	22 days
W-6	91 days

Figure 4.4-3 Days to Gosner Developmental Stage 26 (±1) and Final Developmental Stage Reached at End-of-Test, with Sediment tPCBs and Spatially Weighted Mean tPCBs (FEL 2002b): 2000 Leopard Frog Study

Summary of Northern Leopard Frog Toxicity Study

Adult Reproductive Fitness Endpoints:

- Both male and female adult frogs showed signs of reproductive stress, with the females showing more severe effects. Males exhibited a high incidence of malformed sperm in the higher-sediment tPCB sites (up to 50%). Females had virtually no mature eggs (Stage VI, which the eggs must reach in order for fertilization to occur). Incidences of immature oocytes (Stage III or smaller) were high in the sites with high concentrations of sediment tPCBs (up to 99% Stage III).

Developmental Endpoints:

- High sensitivity to acute endpoints: larval mortality very high (88 to 100% in the PSA treatments, 44% in the control treatment); low incidence of metamorphosis (0 to 6% in the PSA treatments, 6% in the control treatment, but 63% in the water-only control treatment).
- Minor indications of reduced endpoint performance for larval malformations.
- High incidence of larval developmental delay, such that subsequent environmental changes (i.e., decreased water temperature) may prohibit animals developing in situ from reaching metamorphosis.

4.4.1.2.2 Wood Frog Study: EPA

Table 4.4-5 presents the results of statistical tests of significance (comparisons to reference stations) for wood frog toxicity test endpoints; additional detail and graphical presentation of these results are included in Appendix E. Results are shown for both the most synoptic (discrete) and spatially weighted (SW) sediment tPCB concentrations.

Phase I: In general, there were no significant relationships found between egg mass tissue concentration and any of the egg mass endpoints (such as hatching success or percent fertilization). Egg mass tissue concentrations or any egg mass endpoints were not significantly related to sediment tPCB concentration. These results (relative to other life stages) suggest that maternal transfer in wood frogs is not the dominant exposure pathway through which toxicity was manifested in this study.

The larval development and metamorphosis component of the Phase I wood frog study produced mixed results. The pattern of responses appears to be related to the exposure duration and/or the organism life stage. Hatchling stages indicated no dose-response relationships, whereas larval

Table 4.4-5

Statistical Analysis Results: Wood Frog Reproduction and Development Studies

Variable/Endpoint (shading indicates significant relationship)		Sample Size (n)[a]	Statistical Test	Results	Data Source[b]
Phase I Egg Mass	tPCB sediment (discrete) and tPCB egg mass tissue	11	Spearman's	r = 0.36, p > 0.05	FEL 2002a: Appendix C
	tPCB sediment (SW) and egg mass tissue	8	Spearman's	r = -0.19, p > 0.05	FEL 2002a: Appendix C and EPA
	tPCB water and tPCB egg mass tissue	11	Spearman's	r = 0.31, p > 0.05	FEL 2002a: Appendix C
Egg Mass Viability	tPCB sediment (discrete) and mean egg weight	12	Spearman's	r = -0.014, p > 0.05	FEL 2002a: Appendix C
	tPCB sediment (SW) and mean egg weight	9	Spearman's	r = 0.067, p > 0.05	FEL 2002a: Appendix C
	tPCB sediment (discrete) and % fertilized eggs	12	Spearman's	r = -0.29, p > 0.05	FEL 2002a: Appendix C
	tPCB sediment (SW) and % fertilized eggs	9	Spearman's	r = -0.28, p > 0.05	FEL 2002a: Appendix C
	tPCB water and % fertilized eggs	12	Spearman's	r = -0.10, p > 0.05	FEL 2002a: Appendix C
	tPCB egg mass tissue and % fertilized eggs	11	Spearman's	r = -0.27, p > 0.05	FEL 2002a: Appendix C
	tPCB sediment (discrete) and hatching success	12	Spearman's	r = 0.18, p > 0.05	FEL 2002a: Appendix C
	tPCB sediment (SW) and hatching success	9	Spearman's	r = -0.067, p > 0.05	FEL 2002a: Appendix C
	tPCB water and hatching success	12	Spearman's	r = -0.26, p > 0.05	FEL 2002a: Appendix C
	tPCB egg mass tissue and hatching success	11	Spearman's	r = -0.08, p > 0.05	FEL 2002a: Appendix C

Table 4.4-5

Statistical Analysis Results: Wood Frog Reproduction and Development Studies (Continued)

Variable/Endpoint (shading indicates significant relationship)		Sample Size (n)[a]	Statistical Test	Results	Data Source[b]
Phase II Larvae, Event 1	tPCB sediment (discrete)and tPCB Event 1 larvae tissue	10	Spearman's	r = 0.29, p > 0.05	FEL 2002a: Appendix D
	tPCB sediment (SW) and tPCB Event 1 larvae tissue	8	Spearman's	r = -0.26, p > 0.05	FEL 2002a: Appendix D
	tPCB water and tPCB Event 1 larvae tissue	10	Spearman's	r = 0.098, p > 0.05	FEL 2002a: Appendix D
Phase II Larvae, Event 3	tPCB sediment (discrete)and tPCB Event 3 larvae tissue	11	Spearman's	r = 0.89, p < 0.002	FEL 2002a: Appendix D
	tPCB sediment (SW) and tPCB Event 3 larvae tissue	8	Spearman's	r = 0.74, p = 0.05	FEL 2002a: Appendix D
	tPCB water and tPCB Event 3 larvae tissue	10	Spearman's	r = 0.69, p < 0.05	FEL 2002a: Appendix D
Phase II Larval Growth and Development (Field-Collected Animals), Event 4	tPCB sediment (SW) and Event 4 larval malformations	8	Spearman's	r = 0.83, p = 0.02	FEL 2002a: Appendix D
Larval Development, Metamorphosis, and Mortality	tPCB sediment (discrete) and Phase I larval mortality (day 95)	11	Spearman's	r = -0.41, p > 0.05	FEL 2002a: Appendix C
	tPCB sediment (SW) and Phase I larval mortality (day 95)	8	Spearman's	r = -0.88, p < 0.02	FEL 2002a: Appendix C
	tPCB water and Phase I larval mortality (day 95)	11	Spearman's	r = -0.65, p < 0.05	FEL 2002a: Appendix C
	tPCB egg mass tissue and Phase I larval mortality (day 95)	11	Spearman's	r = 0.19, p > 0.05	FEL 2002a: Appendix C
	tPCB sediment (discrete) and Phase I larval mortality (EOT)	11	Spearman's	r = -0.41, p > 0.05	FEL 2002a: Appendix C
	tPCB sediment (SW) and Phase I larval mortality (EOT)	8	Spearman's	r = -0.57, p > 0.05	FEL 2002a: Appendix C

Table 4.4-5

Statistical Analysis Results: Wood Frog Reproduction and Development Studies (Continued)

Variable/Endpoint (shading indicates significant relationship)		Sample Size (n)[a]	Statistical Test	Results	Data Source[b]
	tPCB water and Phase I larval mortality (EOT)	11	Spearman's	r = -0.43, $p > 0.05$	FEL 2002a: Appendix C
	tPCB egg mass tissue and Phase I larval mortality (EOT)	11	Spearman's	r = 0.21, $p > 0.05$	FEL 2002a: Appendix C
	tPCB sediment (discrete) and Phase I larval metamorphosis (day 95)	11	Spearman's	r = 0.43, $p > 0.05$	FEL 2002a: Appendix C
	tPCB sediment (SW) and Phase I larval metamorphosis (day 95)	8	Spearman's	r = 0.57, $p > 0.05$	FEL 2002a: Appendix C
	tPCB water and Phase I larval metamorphosis (day 95)	11	Spearman's	r = 0.42, $p > 0.05$	FEL 2002a: Appendix C
	tPCB egg mass tissue and Phase I larval metamorphosis (day 95)	11	Spearman's	r = -0.13, $p > 0.05$	FEL 2002a: Appendix C
	tPCB sediment (discrete) and Phase I larval metamorphosis (EOT)	11	Spearman's	r = 0.41, $p > 0.05$	FEL 2002a: Appendix C
	tPCB sediment (SW) and Phase I larval metamorphosis (EOT)	8	Spearman's	r = 0.57, $p > 0.05$	FEL 2002a: Appendix C
	tPCB water and Phase I larval metamorphosis (EOT)	11	Spearman's	r = 0.43, $p > 0.05$	FEL 2002a: Appendix C
	tPCB egg mass tissue and Phase I larval metamorphosis (EOT)	11	Spearman's	r = -0.21, $p > 0.05$	FEL 2002a: Appendix C
	tPCB sediment (discrete) and no. of days to 50% mortality Phase I larvae	11	Spearman's	r = 0.53, $p > 0.05$	FEL 2002a: Appendix C
	tPCB sediment (SW) and no. of days to 50% mortality Phase I larvae	8	Spearman's	r = 0.81, $p < 0.05$	FEL 2002a: Appendix C
	tPCB water and no. of days to 50% mortality Phase I larvae	11	Spearman's	r = 0.71, $p < 0.02$	FEL 2002a: Appendix C
	tPCB egg mass tissue and no. of days to 50% mortality Phase I larvae	11	Spearman's	r = -0.30, $p > 0.05$	FEL 2002a: Appendix C

Table 4.4-5

Statistical Analysis Results: Wood Frog Reproduction and Development Studies (Continued)

Variable/Endpoint (shading indicates significant relationship)		Sample Size (n)[a]	Statistical Test	Results	Data Source[b]
	tPCB sediment (discrete) and % Phase I larval malformation Gosner stage 20-24	11	Spearman's	r = 0.80, p = 0.005	FEL 2002a: Appendix C
	tPCB sediment (SW) and % Phase I larval malformation Gosner stage 20-24	8	Spearman's	r = 0.74, p = 0.05	FEL 2002a: Appendix C
	tPCB water and % Phase I larval malformation Gosner stage 20-24	11	Spearman's	r = 0.77, p < 0.01	FEL 2002a: Appendix C
	tPCB tissue and % Phase I larval malformation Gosner stage 20-24	11	Spearman's	r = 0.53 p > 0.05	FEL 2002a: Appendix C
	tPCB sediment (discrete) and Phase 1 mean metamorph weight	11	Spearman's	r = 0.56, p > 0.05	FEL 2002a: Appendix C
	tPCB sediment (SW) and Phase 1 mean metamorph weight	8	Spearman's	r = 0.26, p > 0.05	FEL 2002a: Appendix C
	tPCB water and Phase 1 mean metamorph weight	11	Spearman's	r = 0.67, p < 0.05	FEL 2002a: Appendix C
	tPCB egg mass tissue and Phase 1 mean metamorph weight	11	Spearman's	r = 0.66, p < 0.05	FEL 2002a: Appendix C
	tPCB sediment (discrete) and Phase 1 metamorph malformations	11	Spearman's	r = 0.84, p < 0.005	FEL 2002a: Appendix C
	tPCB sediment (SW) and Phase 1 metamorph malformations	8	Spearman's	r = 0.81, p < 0.05	FEL 2002a: Appendix C
	tPCB water and Phase 1 metamorph malformations	11	Spearman's	r = 0.73, p < 0.02	FEL 2002a: Appendix C
	tPCB egg mass tissue and Phase 1 metamorph malformations	11	Spearman's	r = 0.49, p > 0.05	FEL 2002a: Appendix C
	tPCB metamorph tissue and Phase 1 metamorph malformations	11	Spearman's	r = 0.54, p > 0.05	FEL 2002a: Appendix C
Phase I Metamorphs	tPCB sediment and tPCB metamorph tissue	11	Spearman's	r = 0.55, p > 0.05	FEL 2002a: Appendix C

Table 4.4-5

Statistical Analysis Results: Wood Frog Reproduction and Development Studies (Continued)

Variable/Endpoint (shading indicates significant relationship)		Sample Size (n)[a]	Statistical Test	Results	Data Source[b]
	tPCB sediment (SW) and tPCB metamorph tissue	8	Spearman's	r = 0.76, $p < 0.05$	FEL 2002a: Appendix C
	tPCB water and tPCB metamorph tissue	11	Spearman's	r = 0.67, $p > 0.05$	FEL 2002a: Appendix C
	tPCB egg mass tissue and tPCB metamorph tissue	11	Spearman's	r = 0.16, $p > 0.05$	FEL 2002a: Appendix C
Phase III Metamorphs	tPCB sediment (discrete) and tPCB metamorphs (all)	10	Spearman's	r = 0.43, $p > 0.05$	FEL 2002a: Appendix E
	tPCB sediment (discrete) and tPCB metamorphs (exclude WML-1)	9	Spearman's	r = 0.70, $p = 0.05$	FEL 2002a: Appendix E
	tPCB sediment (SW) and tPCB metamorphs (exclude WML-1)	8	Spearman's	r = 0.76, $p < 0.05$	FEL 2002a: Appendix E
	tPCB water and tPCB metamorphs (all)	10	Spearman's	r = 0.74, $p < 0.05$	FEL 2002a: Appendix E
	tPCB water and tPCB metamorphs (exclude WML-1)	9	Spearman's	r = 0.81, $p < 0.02$	FEL 2002a: Appendix E
Phase III Metamorph Development (Field-Collected Animals)	tPCB sediment and Phase III metamorph mean weight	10	Spearman's	r = 0.25, $p > 0.05$	FEL 2002a: Appendix E
	tPCB metamorph tissue (excluding reference) and Phase III metamorph mean weight	9	Spearman's	r = 0.37, $p > 0.05$	FEL 2002a: Appendix E
	tPCB sediment (discrete) and sex ratios	10	Spearman's	r = -0.77, $p < 0.02$	FEL 2002a: Appendix E
	tPCB sediment (SW) and sex ratios	8	Spearman's	r = -0.91, $p = 0.005$	FEL 2002a: Appendix E
	tPCB metamorph tissue (excluding WML-1) and sex ratios	9	Spearman's	r = -0.91, $p < 0.005$	FEL 2002a: Appendix E

Table 4.4-5

Statistical Analysis Results: Wood Frog Reproduction and Development Studies (Continued)

Variable/Endpoint (shading indicates significant relationship)		Sample Size (n)[a]	Statistical Test	Results	Data Source[b]
	tPCB sediment (discrete) and % malformation Phase III metamorphs	10	Spearman's	r = 0.93, $p < 0.001$	FEL 2002a: Appendix E
	tPCB sediment (SW) and % malformation Phase III metamorphs	8	Spearman's	r = 0.93, $p < 0.005$	FEL 2002a: Appendix E
	tPCB metamorph tissue (excluding WML-1) and % malformation Phase III metamorphs	9	Spearman's	r = 0.85, $p < 0.01$	FEL 2002a: Appendix E
	tPCB sediment (discrete) and % female gonadal malformation,* Phase III metamorphs	10	Spearman's	r = 0.95, $p < 0.002$	FEL 2002a: Appendix E
	tPCB sediment (SW) and % female gonadal malformation,* Phase III metamorphs	8	Spearman's	r = 0.95, $p < 0.005$	FEL 2002a: Appendix E
	tPCB metamorph tissue (excluding WML-1) and % female gonadal malformation,[c] Phase III metamorphs	9	Spearman's	r = 0.88, $p < 0.005$	FEL 2002a: Appendix E
	tPCB metamorph tissue and % female gonadal malformation, [c] Phase III metamorphs	10	Spearman's	r = 0.72, $p < 0.05$	FEL 2002a: Appendix E
	Metamorph sex ratio and % female gonadal malformation [c]	10	Spearman’s	r = -0.93, $p< 0.002$	FEL 2002a: Appendix E

[a] Sample size (n) refers to the number of samples being compared. For example, in the first row of the table, the “11” refers to 11 tPCB sediment samples and 11 tPCB egg mass samples.

[b] FEL 2002a: Appendix C: *Phase I – Specimen Inventory List, Developmental Data, Crossover Data, and Spike Data.*

FEL 2002a: Appendix D: *Phase II – Specimen Inventory List, Growth Data, and Larval Malformation Data.*

FEL 2002a: Appendix E: *Phase III – Specimen Inventory List, Sex Ratio/Abnormality/Weight Data.*

[c] Although the relationship between female gonadal malformation and total incidence of malformation is arguably correlated, these comparisons are still of interest. There are many types of malformations that a juvenile could show; however, the malformed females had a high incidence of gonadal aberrations that increased in relation to increasing sediment and tissue tPCB concentration. Gonadal malformations can lead to sterility of the females.

malformations at Gosner developmental Stages 20 to 24, and metamorphs exhibited indications of toxicity (Figure 4.4-4). The malformation endpoint appeared sensitive and was significantly correlated with sediment, water, and tissue tPCB concentrations (see Figure 4.4-5 and Figure 4.4-6).

Cross-Over and Aroclor 1260-Spiked Treatments: These components of the wood frog study confirmed the importance of vernal pool media as an exposure pathway:

- Reference site larvae raised in sediment and water from their native reference site locations had low tissue PCB concentrations, while larvae from the same reference sites raised in PSA vernal pool media had elevated tissue PCB concentrations.
- PSA pool larvae raised in their native (contaminated) media had elevated tissue PCB concentrations, while larvae from the same pools raised in reference site media had low tissue PCB concentrations. These results indicate that uptake from sediment is more important than maternal transfer.

The cross-over study confirmed the overall findings of the Phase I main study, and indicated that mortality and metamorphosis endpoints were not significantly affected by PCB exposures, while moderate toxicity was observed for the malformation endpoint. The percentage of malformed metamorphs was higher in the spiked treatment than in the control treatment, indicating that exposure to the spiked sediment had an adverse effect on metamorph malformation; however, larval mortality and the incidence of larval metamorphosis appeared unaffected.

Phase II: The Phase II larvae showed a similar pattern of responses to that of the Phase I animals. While only growth and malformation were evaluated in Phase II, the effects for the two endpoints were similar in the two study phases. Mean larval length at each station increased between sampling events 1 and 3 (as the larvae grew); and mean length was similar between PSA and reference pools, showing no apparent relationship to tissue PCB concentration. The findings of the Phase II study show that PCB accumulation and the incidence of malformations increase with exposure duration.

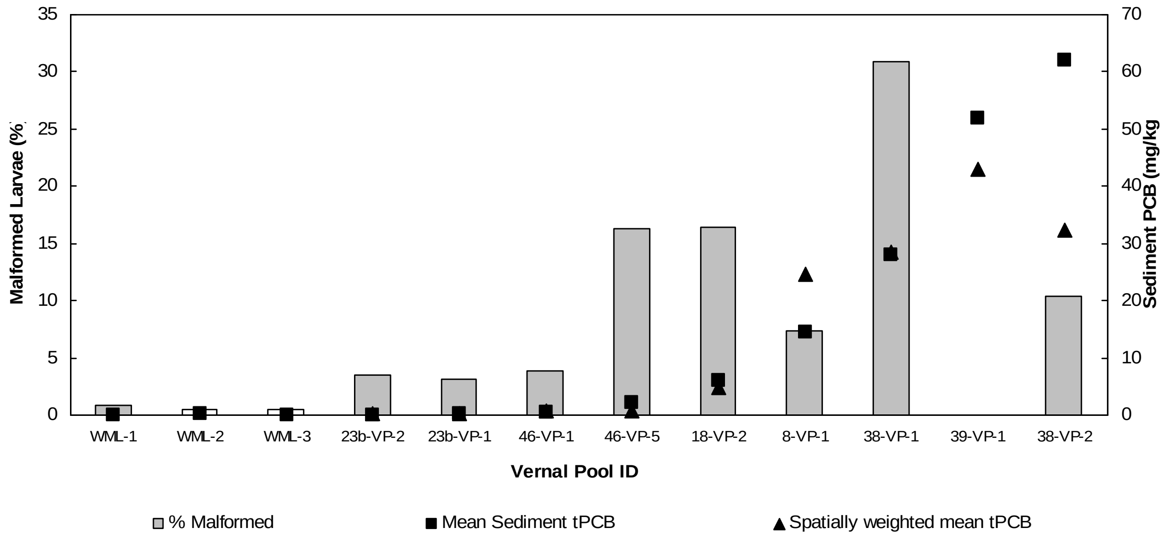


Figure 4.4-4 Comparison of Phase I Larval Wood Frog Malformations as Gosner Developmental Stage 20-24 to Mean Sediment tPCBs and Spatially Weighted Mean tPCBs (FEL 2002a)

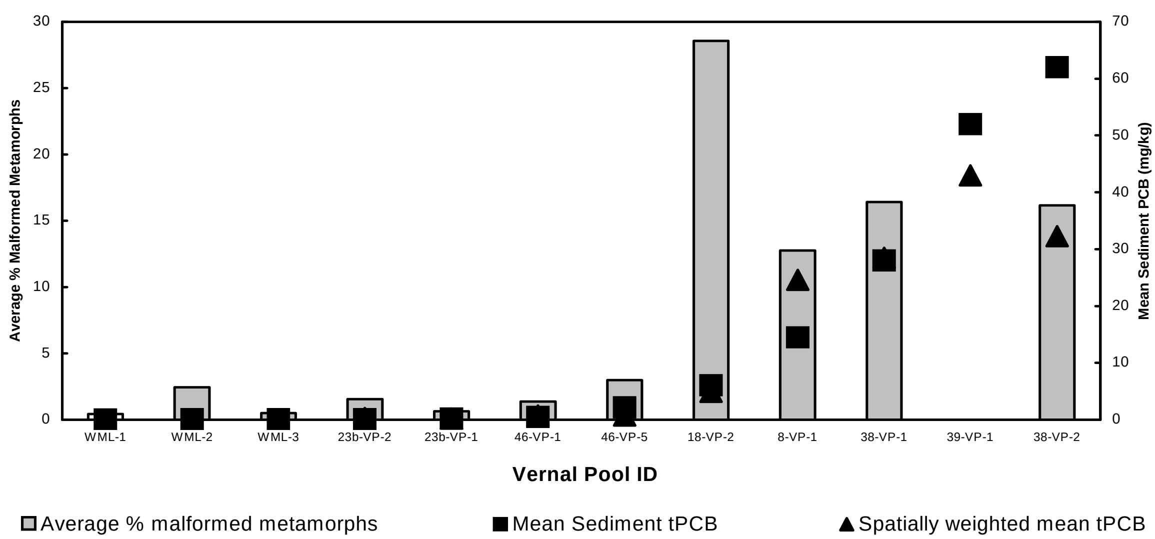
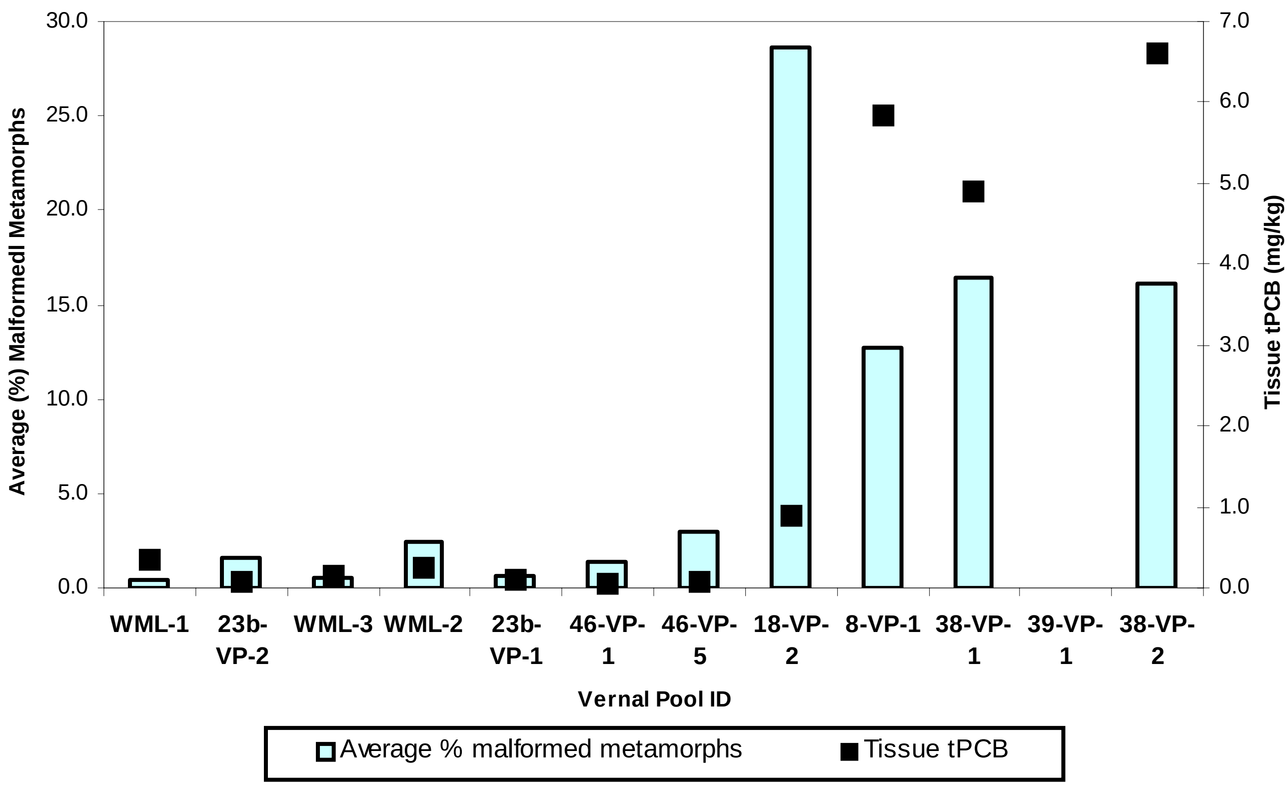


Figure 4.4-5 Incidence of Malformation in Phase I Wood Frog Metamorphs, with Mean Sediment tPCBs and Spatially Weighted Mean tPCBs (FEL 2002a)



Notes: No tissue sample for 39-VP-1; sediment tPCB analyses for WML-1, WML-2, and WML-3 were non-detect (ND). Numbers shown represent detection limits (0.069, 0.13, and 0.11, respectively).

Tissue concentrations are wet weight.

Figure 4.4-6 Incidence of Malformation in Phase I Wood Frog Metamorphs, Phase I Metamorph Tissue tPCBs (FEL 2002a)

Phase III: This phase of the study was an in situ exposure, using the same pools where egg masses for Phase I were collected. Endpoints included incidence of malformation, growth, and sex ratio (number of males to females). As with the first two study phases, there was no significant relationship between metamorph weight and sediment or tissue PCB concentration. However, the sex ratios changed with increasing sediment concentration. In general, as the PCB concentration in sediment or tissue increased, the proportion of males to females decreased. There was a significant correlation between skewed sex ratios and sediment and tissue tPCB concentrations (Figure 4.4-7 and Figure 4.4-8).

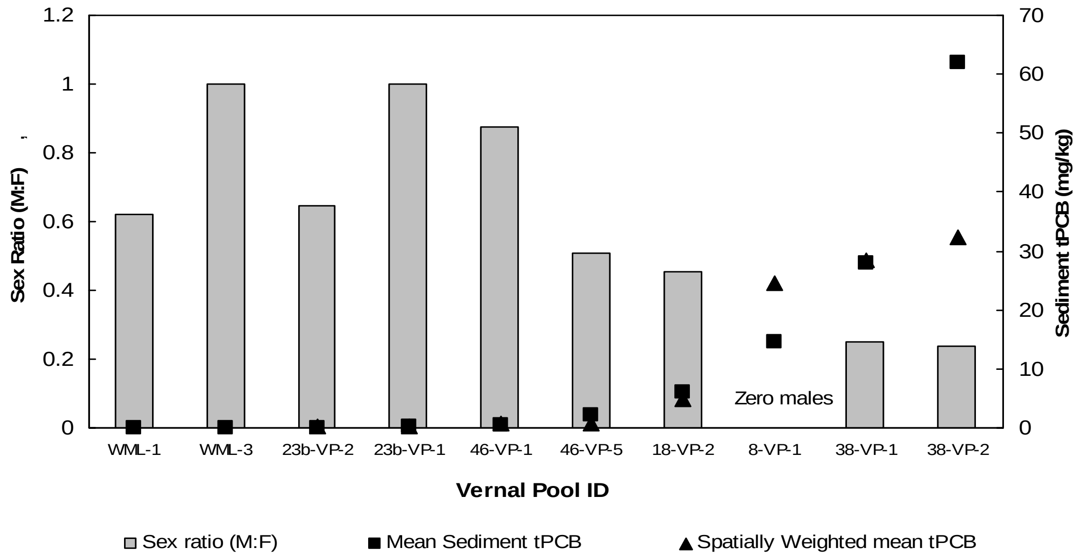
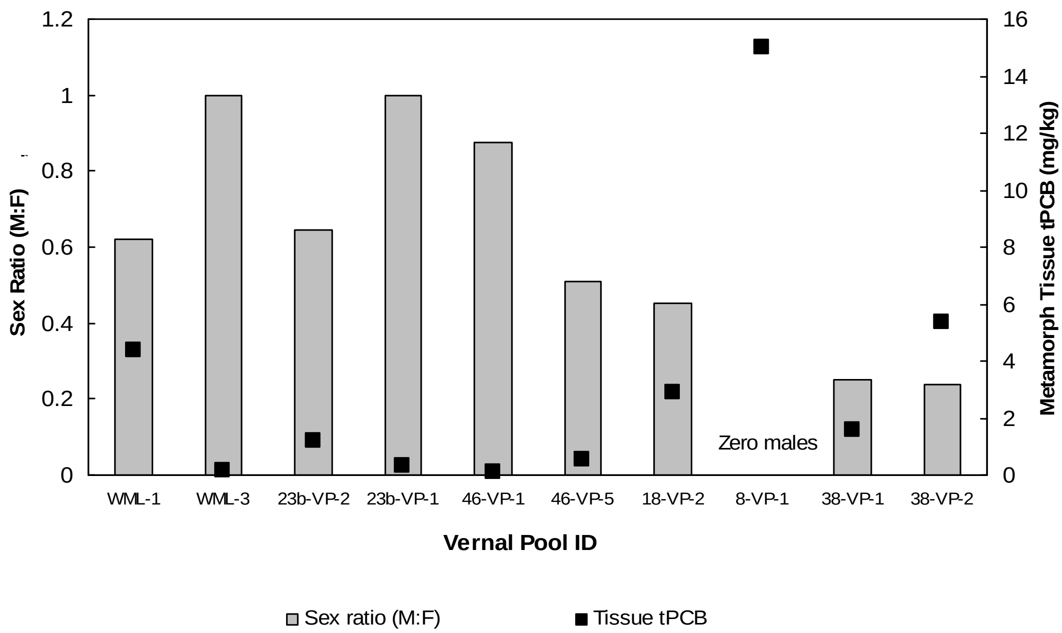


Figure 4.4-7 Male to Female Ratio in Phase III Wood Frog Metamorphs, with Sediment tPCBs and Spatially Weighted Sediment tPCBs (FEL 2002a)



Note: Tissue concentrations are wet weight.

Figure 4.4-8 Male to Female Ratio in Phase III Wood Frog Metamorphs, with Tissue tPCBs (FEL 2002a)

The incidence of juvenile malformations was highest in this phase (both internal and external malformations were assessed). Incidence of malformation was correlated with sediment and metamorph tissue tPCB concentration. There was a significant relationship between metamorph malformation and sediment and tissue tPCB concentrations (Figure 4.4-9 and Figure 4.4-10).

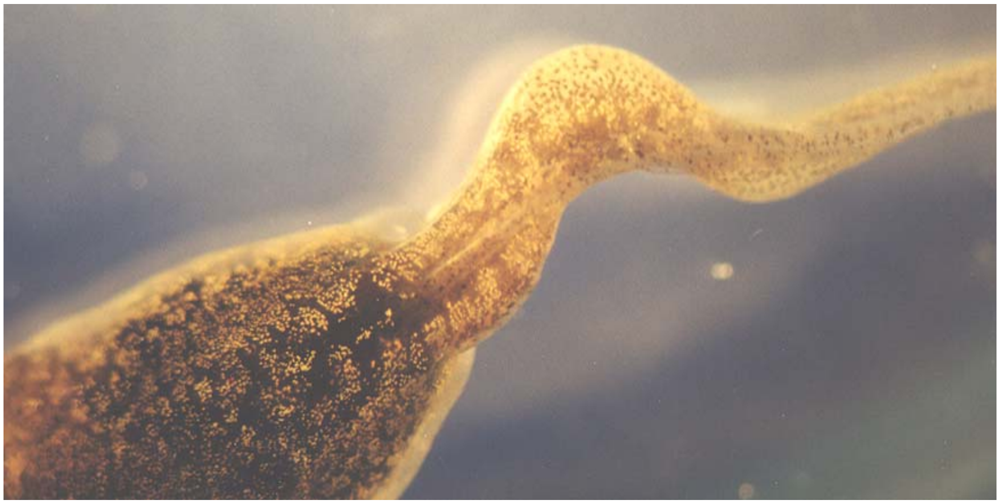
The incidence of metamorph malformation is expected to be significant at the population level because a high degree of malformations could lead to reduced population recruitment at local and regional scales (Ouellet 2000). The types of malformations observed in the metamorphs may affect survivorship by interfering with swimming, hopping, foraging, and predator avoidance (see following photos).

Summary of Wood Frog Toxicity

- No observed toxicity in egg mass viability; egg fertilization, hatching success, and egg counts were unaffected by vernal pool tPCBs, or egg mass tissue tPCB concentrations.
- Contaminant effects were not observed in early-stage juveniles, although high mortality in the reference animals makes it difficult to assess the acute sensitivity of the wood frogs. Incidence of metamorphosis appeared unaffected.
- Manifestation of effects increased with time spent in the vernal pools. Late-stage larvae/metamorphs (laboratory-cultured and field-collected) had elevated levels of both internal and external malformations, with magnitude of response related to sediment and tissue tPCB concentrations.
- Metamorphs collected after in situ exposure in natal pools showed alteration in sex ratio in relation to sediment and tissue tPCB concentrations.

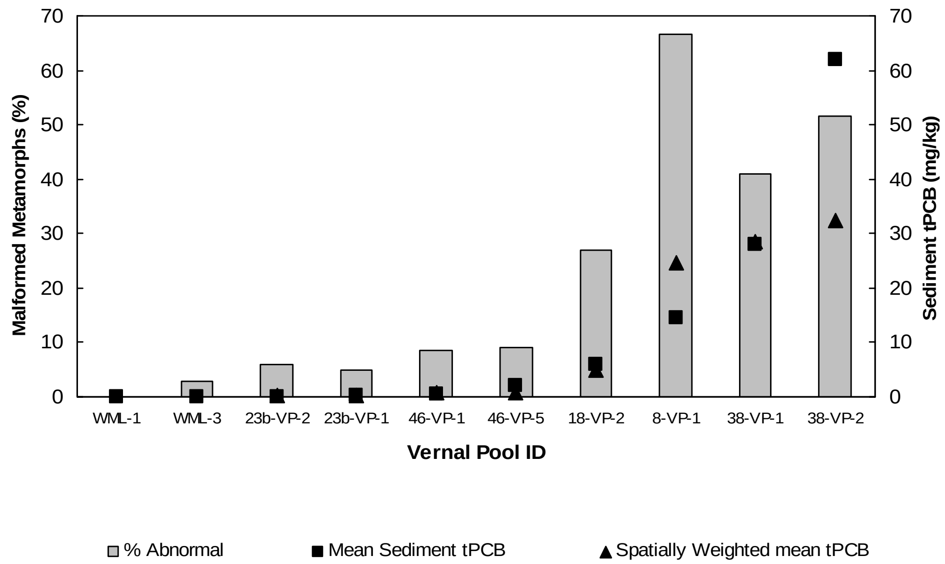
4.4.1.2.3 Wood Frog Study: GE

The study reported that there was not a statistically significant relationship between vernal pool sediment tPCB concentrations and juvenile tissue concentrations. This finding concurs with that of the EPA study, wherein natal pool sediment tPCB was shown to be unrelated to egg mass tissue concentration. Overall, there was a high degree of uncertainty associated with this study, due mostly to an inadequate evaluation of relevant exposure pathways and study duration.



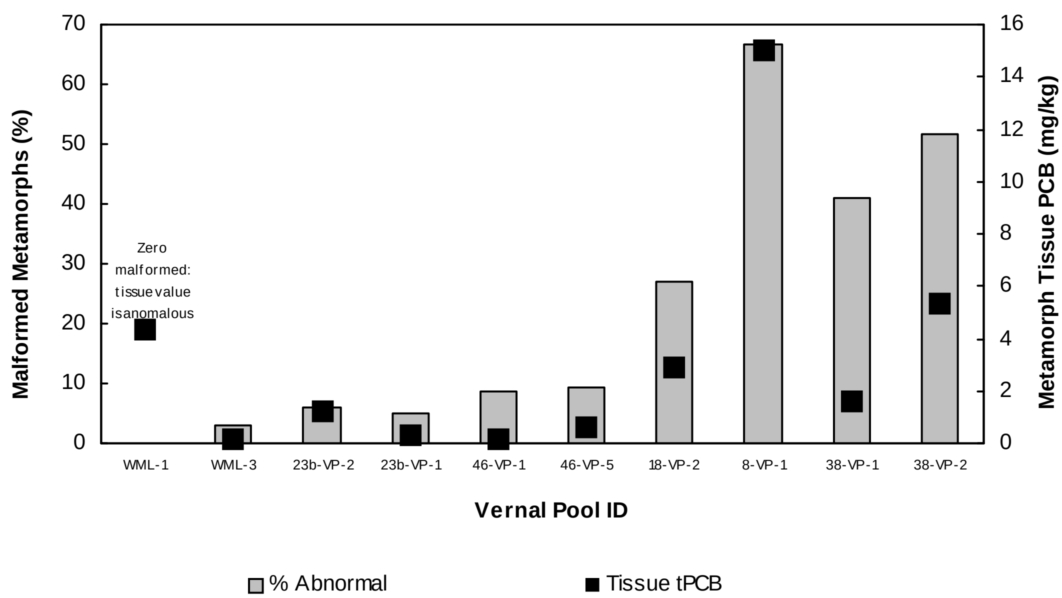
Example of Axial Flexure and Notochord Lesions

Example of Normal Tail



Notes: Sediment tPCB analyses for WML-1, WML-2, and WML-3 were non-detect (ND) – numbers shown represent detection limits (0.069, 0.13, and 0.11 mg/kg, respectively).

Figure 4.4-9 Percent Malformation in Phase III Wood Frog Metamorphs, with Sediment tPCBs (FEL 2002a)



Note: Tissue concentrations are wet weight.

Figure 4.4-10 Percent Malformation in Phase III Wood Frog Metamorphs, with Tissue tPCBs (FEL 2002a)

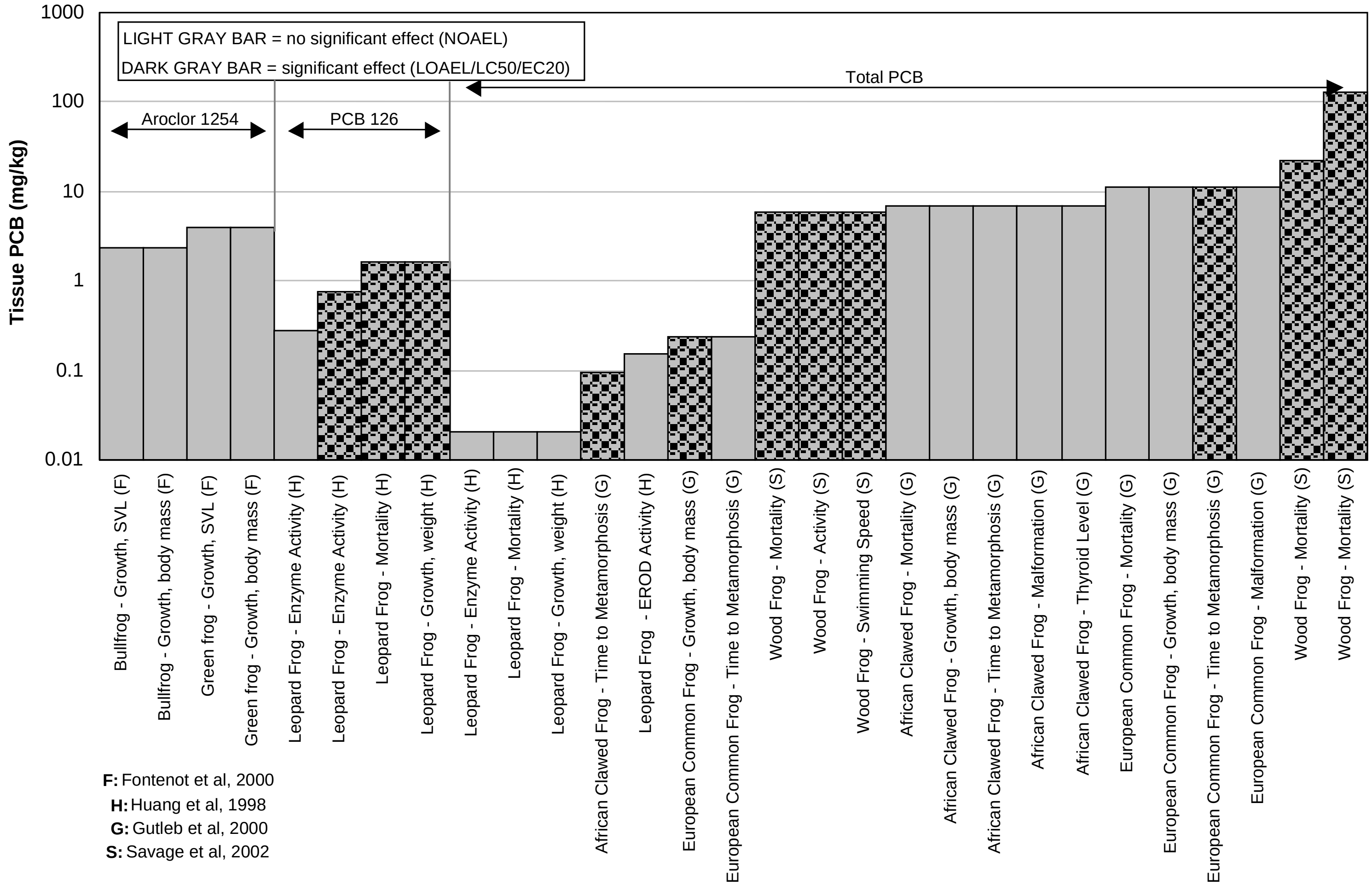
4.4.2 PCB Effect Thresholds

Data were compiled on sediment and tissue PCB concentrations associated with lethal or sublethal effects in representative amphibians from the literature. The purpose was to estimate threshold tissue concentrations where adverse effects might occur in Housatonic River amphibians. The review focused on data for Aroclor 1254 and PCB-126 (considered one of the more toxic congeners) in addition to tPCBs.

Figure 4.4-11 shows the distribution of no effect and effect tissue concentrations. Based on this distribution, it is unlikely that adverse effects will occur at tissue concentrations below 1 mg/kg ww, and it is likely that they will occur above 10 mg/kg ww.

Overall, the upper and lower bounds of the effect concentration ranges from the EPA wood frog study closely match those derived from the literature. Specifically, 1 mg/kg ww was the approximate tissue concentration where ecologically significant adverse effects began to occur, and responses became frequent and more severe at approximately 10 mg/kg ww.

No tissue effects threshold could be established for the leopard frogs, due to the difficulty in establishing relevant biological linkages between tissue data and effects endpoints. However, the leopard frogs appeared more acutely sensitive than the wood frogs.



Note: Tissue concentrations are wet weight.

Figure 4.4-11 Summary of Available Literature Effects Data on PCB Tissue Residues in Anuran Amphibians

4.5 RISK CHARACTERIZATION

The risk characterization evaluates the likelihood that adverse effects may occur as a result of amphibian exposure to tPCBs and/or other COCs. Three broad categories of measurement endpoints in the Housatonic River amphibian risk assessment were used to develop the risk characterization:

- Endpoints based on field surveys (i.e., amphibian community structure) – For these endpoints, care was exercised to discriminate between responses related to COCs and those related to other factors such as substrate or habitat type.

- Endpoints based on site-specific toxicity study results – These endpoints directly evaluated biological responses to COCs.

- Endpoints that compared field-measured exposures to effects levels or benchmarks – For these endpoints, the risk characterization integrated exposure and effects by relating the two terms quantitatively (e.g., hazard quotient [HQ] method for tissue chemistry data and derivation of dose-response relationships for toxicity data).

These three categories of endpoints were independent, allowing for a robust weight-of-evidence (WOE) assessment of the potential for risk using the approach of Menzie et al. (1996).

All three lines of evidence suggested some degree of harm to amphibians in the Housatonic River. In addition, for each line of evidence, there were indications that PCBs are primarily responsible for the observed patterns of responses.

4.5.1 Dose-Response Analysis – Toxicity Test Endpoints

A statistical assessment was conducted to quantify the relationship between toxicity test endpoints and COC concentrations measured concurrent with the wood frog study. The assessment focused on the relationship between PCBs and toxicity endpoints because other lines of evidence indicated a high probability that PCBs (as opposed to other COCs) were a causal agent for toxicity to amphibians within the Housatonic River PSA.

Endpoints Selected for Dose-Response Analysis

Regardless of study phase, the late larval/metamorph endpoints were consistently correlated with contaminant media concentrations. Therefore, the following endpoints were selected for the more detailed statistical assessment:

- Phase I metamorph percent malformed larvae (compared to sediment and Phase I metamorph tissue tPCB concentrations).
- Phase III percent malformed metamorphs (compared to Phase III metamorph tissue and sediment tPCB concentrations).
- Phase III metamorph sex ratio (proportion of females) (compared to sediment tPCBs and Phase III metamorph tissue tPCB concentrations).

4.5.1.1 Calculation of Individual Toxicity Test Endpoints

Comparisons based on magnitude of effects for various endpoints deemed biologically relevant were considered. Effects observed at frequencies of 20% and 50% were selected as indicators of moderate and major toxic effects, respectively.

Three sets of exposure data (two sediment, one tissue) were used to evaluate tPCB dose-response relationships. Summary metrics (e.g., EC_{20}, EC_{50}) were calculated for each endpoint based on sediment tPCB concentrations measured concurrent with the tests, and also with spatially weighted sediment tPCB concentrations. In addition, tissue tPCB concentrations were compared to effects.

4.5.1.2 Results of Dose-Response Analysis

4.5.1.2.1 Sediment

Ecologically significant adverse effects in late stage juvenile wood frogs occurred in the sediment tPCB concentration range of 9.54 to 59.3 mg/kg, and statistically significant responses of lesser magnitude were observed at 0.52 mg/kg tPCBs and lower. An MATC of 3.27 mg/kg was established for sediment.

Estimated Maximum Acceptable Threshold Concentrations

- Most endpoints followed a fairly smooth (typically sigmoidal) dose-response, which could be fit using the probit model.
- Concordance was observed among endpoints for sediment concentrations causing significant effects (i.e., 50% responses occurred at sediment tPCB concentrations of 9.54 to 59.3 mg/kg).
- Concordance was observed among endpoints for tissue residues causing significant effects (50% responses occurred at tissue tPCB concentrations of 3.09 to 6.54 mg/kg ww).
- The MATC for amphibians was based on the integration of metamorph malformation and sex ratios endpoints. These were integrated because it was difficult to determine which endpoint is more sensitive with respect to population-level impacts. Furthermore, the two endpoints may act in concert to limit the viability of local wood frog populations.
- A sediment MATC of 3.27 mg/kg tPCBs was determined based on the EC_{20} value for the Phase III metamorph malformation endpoint. A 20% incidence of malformation was considered biologically relevant, hence, the use of the 20% effect size for this endpoint in the calculation of the sediment MATC. The EC_{20} for the sex ratio endpoint was 0.52 and 0.61 mg/kg tPCBs (based on measured and spatially weighted sediment tPCB concentrations, respectively). There is uncertainty, however, associated with whether a 20% effect size is of biological relevance for sex ratio; therefore, a sediment MATC based on the sex ratio EC_{20} may be overly conservative.

4.5.1.2.2 Tissue

The threshold concentration range for amphibian tissues was 0.60 mg/kg ww to 6.54 mg/kg ww tPCBs, and was based on the sex ratio endpoint (both an EC_{20} and EC_{50}) and the Phase III metamorph malformation endpoint (an EC_{50} point estimate). The tissue dose-response curve predicted significant risk in the range of 1 to 10 mg/kg ww. At tissue concentrations >10 mg/kg ww, adverse ecological effects are expected to occur with a high degree of certainty.

4.5.2 Biological Community Endpoints

4.5.2.1 Amphibian Community Evaluation: EPA

Population responses of amphibians were measured in field studies of amphibian communities conducted in 1999 and 2000 (Woodlot Alternatives, Inc. 2003). Detailed data were collected for wood frogs (e.g., numbers of frogs entering and leaving pools, numbers of metamorphs captured leaving the pools). In addition, species abundance, richness, and presence of malformations were assessed for multiple species in selected vernal pools.

The findings included:

- Species richness was lower in the vernal pools with higher average sediment tPCB concentrations; density and biomass (on a per m^2 basis) were lower in the more contaminated vernal pools.
- Salamanders appeared to be sensitive to tPCBs, appearing in lower numbers in vernal pools with high sediment tPCB concentrations.
- Gross malformation rates in adults (wood frogs and spotted salamanders) and metamorphs (wood frogs) were low. However, malformation rates in larval wood frogs were high in all pools.

4.5.2.2 Leopard Frog Egg Mass Survey: GE

In the spring of 2003, ARCADIS G&M, Inc. (ARCADIS) conducted a survey of leopard frog egg masses occurring in the vernal pool and backwater habitats of the PSA. The primary objective of the survey was to determine whether adult leopard frogs are reproducing successfully in the PSA; the metric chosen for evaluation of leopard frog reproductive health was the presence/absence of egg masses within breeding habitats.

The study concluded that there was no relationship between pool sediment tPCB concentration and the presence/absence of egg masses. The investigators also concluded that there was no evidence of reproductive impairment in leopard frogs within the PSA, even though 61% of the suitable leopard frog habitat identified did not contain any egg masses. In addition, there were 2.4 leopard frog egg masses per hectare (ha) reported in the GE study, in comparison to the 277 egg masses per hectare reported by Hine et al. (1981) and 58 egg masses per hectare reported by Gilbert et al. (1994).

4.5.2.3 Amphibian Community Measures Observed During Developmental Study Field Sampling: EPA

Additional evidence for population responses of amphibians was derived from anecdotal information from field studies collected in support of the FEL developmental studies. No egg masses were found at three of the leopard frog sampling locations and one of the wood frog vernal pools. Sediment tPCB concentrations at these areas were among the highest of the concentrations measured for the two studies: between 50 and 160 mg/kg. In addition, female leopard frogs were not found at three contaminated sampling locations. Sediment PCB concentrations at two of these areas were over 100 mg/kg dw tPCBs.

4.5.3 Comparison of Tissue Chemistry Data to Benchmarks

As an additional line of evidence, hazard quotients (HQs) were used to quantify the degree to which amphibian tissue COC concentrations exceeded the literature-based effects and site-specific thresholds deemed protective of assessment endpoints. In theory, adverse ecological responses are possible if any HQ exceeds 1.0 (i.e., if exposure exceeds the lower threshold level). Separate HQs were calculated for each tissue type and species. Tissue HQs were based on comparison of observed tissue residues to an effects threshold of 1 mg/kg ww tPCBs, which represents a conservative interpretation of the LOAELs at which significant adverse effects were found in the literature.

4.5.3.1 Leopard Frog HQs

Table 4.5-1 presents the range of HQs for leopard frog tissues, using the literature-derived LOAEL. Based on the comparison to the LOAEL and the 10 mg/kg ww concentration at which ecologically significant effects are expected with a high degree of certainty, the abundance of tissue HQs between 1 and 10 indicates a likelihood for adverse effects.

4.5.3.2 Wood Frog HQs: EPA Study

Wood frog HQs based on concentrations of tPCBs measured in egg masses were relatively low. The low-to-marginal HQs indicate that the hazard for this life stage is fairly low. This finding is consistent with the lack of significant toxicity observed in the egg mass toxicity endpoints, such as hatching success, percent fertilization, and percent necrotic eggs. Tissue HQs based on Phase II wood frog tadpole concentrations were variable, ranging between <0.1 and 10.

Table 4.5-1

Hazard Quotients for Leopard Frog PCB Tissue Residues, Based on Effects Reported in the Literature (LOAEL 1 mg/kg ww tPCBs)

Sampling Area ID	Life Stage or Tissue Type	HQ
Muddy Pond Reference	Adult chemical analysis (whole body)	0.03
	Adult experimental (female whole body minus ovaries/egg masses)	0.012
	Ovary/egg mass (from adult experimental female)[a]	0.024
	Larvae-to-metamorphs[b]	NA
W-1	Adult chemical analysis (whole body)	0.15
	Adult experimental (female whole body minus ovaries/egg masses)	0.023
	Ovary/egg mass (from adult experimental female)[a]	0.26
	Larvae-to-metamorphs	NA
W-4	Adult chemical analysis (whole body)	0.34
	Adult experimental (female whole body minus ovaries/egg masses)	NA
	Ovary/egg mass (from adult experimental female)[a]	NA
	Larvae-to-metamorphs[b]	1.4
W-9a	Adult chemical analysis (whole body)	3.59
	Adult experimental (female whole body minus ovaries/egg masses)	1.24
	Ovary/egg mass (from adult experimental female)[a]	5.05
	Larvae-to-metamorphs[b]	NA
W-7a	Adult chemical analysis (whole body)	2.11
W-7a	Adult experimental (female whole body minus ovaries/egg masses)	1.4
	Ovary/egg mass (from adult experimental female)[a]	6.61
	Larvae-to-metamorphs[b]	1.11
EW-3	Adult chemical analysis (whole body)	4.26
	Adult experimental (female whole body minus ovaries/egg masses)	1.23
	Ovary/egg mass (from adult experimental female)[a]	1.52
	Larvae-to-metamorphs[b]	0.96

Table 4.5-1

Hazard Quotients for Leopard Frog PCB Tissue Residues, Based on Effects Reported in the Literature (LOAEL 1 mg/kg ww tPCBs) (Continued)

Sampling Area ID	Life Stage or Tissue Type	HQ
E-5	Adult chemical analysis (whole body)	1.31
	Adult experimental (female whole body minus ovaries/egg masses)	NA
	Ovary/egg mass (from adult experimental female)[a]	NA
	Larvae-to-metamorphs[b]	NA
W-6	Adult chemical analysis (whole body)	1.78
	Adult experimental (female whole body minus ovaries/egg masses)	0.386
	Ovary/egg mass (from adult experimental female)[a]	9.45
	Larvae-to-metamorphs[b]	0.67
W-8	Adult chemical analysis (whole body)	5.39
	Adult experimental (female whole body minus ovaries/egg masses)	NA
	Ovary/egg mass (from adult experimental female)[a]	NA
	Larvae-to-metamorphs	NA
E-1	Adult chemical analysis (whole body)	3.10
	Adult experimental (female whole body minus ovaries/egg masses)	NA
	Ovary/egg mass (from adult experimental female)[a]	NA
	Larvae-to-metamorphs	NA

Sampling areas arranged in order of increasing sediment PCB concentration.

NA = No sample available because specimens were not found.

[a]Egg mass/ovary HQs based on a geometric mean of the two tissue concentrations per station. This was done because of the large difference between the two concentrations for a given station.

[b]HQs for larvae-to-metamorph samples cannot all be compared to one another, as the specimens were not all the same age when the samples were collected. Animals from sampling areas W-6, W-4, and EW-3 are comparable; animals from sampling areas W-7a and W-1 are comparable.

Phase III wood frog metamorphs (12 to 15 weeks old) had tissue HQs exceeding 1.0 in several samples, with maximum HQs above 10. This study phase represented sediment exposure over the entire juvenile period, and exhibited the most pronounced toxicological responses.

4.5.4 Weight-of-Evidence Procedure for Assessing Risk from PCBs in the Housatonic River PSA

A WOE process (Menzie et al. 1996) was applied to determine whether PCBs pose a significant risk to amphibians in the Housatonic River PSA. The rationale for weighting of measurement endpoints is provided in Appendix E, along with a discussion of attributes considered. Integrated assessments of potential adverse impacts to amphibians based on FEL's leopard and wood frog studies are shown in Tables 4.5-2 and 4.5-3. A summary of the weighting for each attribute is provided in Table 4.5-4. The chemistry endpoints yielded the lower overall values due to low-to-moderate site specificity and some uncertainty with the linkage between the measurement endpoints and the assessment endpoint(s). The toxicity testing endpoints yielded the highest overall weighting, due to the site specificity and high degree of biological relevance in the reproductive endpoints. The three field studies of biological community endpoints had intermediate values.

The magnitude of the response in the measurement endpoint is considered together with the measurement endpoint weight in judging the overall WOE. The weighting values, evidence of harm, and magnitude of response were combined in a matrix format and are presented in Table 4.5-5.

A graphical method was used for displaying the degree of concurrence among measurement endpoints (Table 4.5-6). The 12 symbols representing the chemistry (C), wood frog toxicity (W), leopard frog toxicity (L), field biology (B), and (P) population model endpoints were displayed in a matrix, with the weight of the measurement endpoint and the degree of response as the axes.

Table 4.5-2

Integrated Assessment of Potential for Adverse Impacts to Amphibian Populations (Leopard Frog Study)

Sampling Area	Sediment tPCBs (mg/kg)		Adult Reproductive Health[a]	Larval Development[b]	Tissue Concentration	Overall Rating
	Sampling Area tPCBs	Spatially Weighted tPCBs				
W-1	0.15	0.4	◉	●	⊙	◉
W-4	0.46	0.4	●	●	◉	●
W-9a	4.3	7.5	●	NA	●	●
W-6	42	21	●	●	◉	●
EW-3	30	23.8	●	✖	◉	●
E-1	160	26.6	●	NA	◉	●
W-7a	18	27.6	◉	●	●	●
W-8	120	43.5	●	NA	◉	●

Sampling areas sorted by spatially weighted tPCB concentration.

⊙ = Negligible-to-low toxicity: negligible indication of ecological risk. No exceedances of tissue benchmark (1 mg/kg tPCBs).

◉ = Moderate toxicity; ecological effects possible, but not conclusive. At least 1 exceedance of tissue benchmark.

● = High toxicity; strong indication of potential ecological effects. At least 1 tissue concentration is ≥10x the tissue benchmark.

✖ = Very strong toxic response.

[a]Includes 6 endpoints: Adult body weight (male and female), sperm count, % abnormal sperm heads, egg count, % mature oocytes.

[b]Includes 5 endpoints: Larval growth, % metamorphosis, % malformation, growth, and days to reach Gosner Stage 26±1.

Table 4.5-3

Integrated Assessment of Potential for Adverse Impacts to Amphibian Populations (Wood Frog Study)

Sampling Area ID	Sediment tPCBs (mg/kg)		Egg Mass[a]	Early larvae (up to Gosner Stage 20-24)[b]	Mid- to Late-stage larvae (after Gosner Stage 24)[c]	Metamorph[d]	Tissue tPCBs	Overall Rating
	VP PCBs	Spatially Weighted tPCBs						
23b-VP-1	0.19	0.2	⊙	⊙	◉	⊙	◉	⊙
23b-VP-2	0.11	0.3	◉	⊙	⊙	⊙	◉	⊙
46-VP-5	2.18	0.7	⊙	⊙	⊙	⊙	⊙	⊙
46-VP-1	0.5	0.8	⊙	⊙	◉	⊙	⊙	⊙
18-VP-2	6.05	4.9	⊙	⊙	⊙	●	◉	◉
8-VP-1	14.5	24.6	◉	◉	◉	●	●	●
38-VP-1	28	28.5	⊙	◉	⊙	●	●	●
38-VP-2	62	32.3	⊙	◉	⊙	●	◉	◉
39-VP-1[e]	52	43	NA	NA	NA	NA	NA	NA

Sampling area sorted by spatially weighted tPCB concentration.

⊙ = Negligible-to-low toxicity: negligible indication of ecological risk. No exceedances of lower or upper tissue benchmarks (1 and 10 mg/kg tPCBs).

◉ = Moderate toxicity; ecological effects possible, but not conclusive. At least 1 exceedance of lower tissue benchmark.

● = High toxicity; strong indication of potential ecological effects. At least 1 exceedance of upper tissue benchmark.

✖ = Very strong toxic response.

[a]Includes 4 endpoints: Phase I egg mass weight (total), % fertilized, %viable, and % hatching success.

[b]Includes 4 endpoints: Phase I early larval malformation (Gosner Stage 20-24); Phase II, Event 1 and 2 larval abundance, % malformed and growth.

[c]Includes 4 endpoints: Phase I larval mortality at day 95; Phase II, Event 3 and 4 larval abundance, % malformed and growth.

[d]Includes 3 endpoints: Phase I malformed metamorphs, Phase III malformed metamorphs, Phase III sex ratio.

[e]No frogs were found in this pool for collection and subsequent study.

Table 4.5-4

Weighting of Measurement Endpoints for Amphibian Weight-of-Evidence Evaluation

Measurement Endpoints	Endpoint Group: Chemistry	Endpoint Group: Wood Frog Toxicology (W)				Endpoint Group: Leopard Frog Toxicology (L)				Endpoint Group: Biology		
Attributes	**C (tissue)**	**W-1 (hatchling)**	**W-2 (larvae)**	**W-3 (metamorph)**	**W-4 (GE juveniles)**	**L-1 (hatchling)**	**L-2 (larvae)**	**L-3 (metamorph)**	**L-4 (adult)**	**B-1 (community)**	**B-2 (GE egg mass survey)**	**B-3 (anecdotal)**
I. Relationship between Measurement and Assessment Endpoints												
1. Degree of Association	Mod	Mod/High	Mod/High	Mod/High	Low	Mod/High	Mod/High	Mod/High	Mod/High	Mod/High	Mod	Low
2. Stressor/Response	Mod	Mod/High	Mod/High	Mod/High	Low	Mod/High	Mod/High	Mod/High	Mod/High	Mod	Mod	Low/Mod
3. Utility of Measure	Mod	Mod	Mod/High	Mod/High	Mod/High	Mod	Mod	Mod	Mod	Mod	Mod	Low/Mod
II. Data Quality												
4. Data Quality	Mod/High	Mod/High	Mod/High	Mod/High	Low	Mod/High	Mod/High	Mod/High	Mod/High	Mod/High	Mod	Mod/High
III. Study Design												
5. Site Specificity	Low/Mod	Mod/High	Mod/High	Mod/High	Low/Mod	Mod	Mod	Mod	Mod	Mod/High	Mod/High	Mod/High
6. Sensitivity	Mod	Mod/High	Mod/High	Mod/High	Low	Mod	Mod	Mod	Mod	Low/Mod	Low/Mod	Low/Mod
7. Spatial Representativeness	High	High	High	High	Low	Mod/High	Mod/High	Mod/High	Mod/High	High	High	Mod/High
8. Temporal Representativeness	High	Mod/High	Mod/High	Mod/High	Low/Mod	Mod	Mod	Mod	Mod	Mod	Mod	Mod
9. Quantitative Measure	Low	High	High	High	Low/Mod	Mod	Mod	Mod	Mod	High	Low	Low
10. Standard Method	Mod/High	Mod/High	Mod/High	Mod/High	Low/Mod	Mod/High	Mod/High	Mod/High	Mod/High	Mod/High	Mod/High	
Overall Endpoint Value	**Mod**	**Mod/High**	**Mod/High**	**Mod/High**	**Low/Mod**	**Mod/High**	**Mod/High**	**Mod/High**	**Mod/High**	**Mod/High**	**Mod**	**Mod**

Mod/High

C. Chemical Measures
C. Concentration of PCBs in frog tissues in relation to levels reported to be harmful to amphibians

W. Wood Frog Toxicological Measures
W-1. Sediment toxicity to hatchling/late embryo life stages
W-2. Sediment toxicity to larval life stages
W-3. Sediment toxicity to late larval/metamorph life stage
W-4. GE Context-Dependent Wood Frog Study (hatchlings, tadpoles, and metamorphs evaluated)

L. Leopard Frog Toxicological Measures
L-1. Sediment toxicity to hatchling/late embryo life stages
L-2. Sediment toxicity to larval life stages
L-3. Sediment toxicity to late larval/metamorph life stage
L-4. Sediment toxicity to adult leopard frogs (reproductive health)

B. Biology
B-1. Vernal pool community study
B-2. GE leopard frog egg mass survey
B-3. Anecdotal observations during collections for reproductive study

Note: Sections E.4.7.1.1, E.4.7.1.2, E.4.7.1.3 present a detailed narrative discussion of the rationale used to support the attribute ranks contained in this table.

Table 4.5-5

Evidence of Harm and Magnitude of Effects for Measurement Endpoints Related to Maintenance of Amphibian Populations in the Lower Housatonic River

Measurement Endpoints	Weighting Value (High, Moderate, Low)	Evidence of Harm (Yes, No, Undetermined)	Magnitude (High, Intermediate, Low)
C. Chemical Measures			
C. Concentration of PCBs in frog tissues in relation to levels reported to be harmful to amphibians.	Moderate	Yes	Low
W. Wood Frog Toxicological Measures			
W-1. Sediment toxicity to hatchling/late embryo life stages.	Mod/High	No	-
W-2. Sediment toxicity to larval life stages.	Mod/High	Yes	Intermediate
W-3. Sediment toxicity to late larval/metamorph life stage.	Mod/High	Yes	High
W-4. GE Study (juvenile wood frogs)	Low/Mod	Undetermined	-
L. Leopard Frog Toxicological Measures			
L-1. Sediment toxicity to hatchling/late embryo life stages.	Mod/High	Yes	Low
L-2. Sediment toxicity to larval life stages.	Mod/High	Yes	High
L-3. Sediment toxicity to late larval/metamorph life stage.	Mod/High	Yes	High
L-4. Sediment toxicity to adult leopard frogs (reproductive health).	Mod/High	Yes	High
B. Biological Measures			
B-1. Vernal pool community study.	Mod/High	Yes	Low
B-2. GE leopard frog egg mass survey	Moderate	Undetermined	-
B-3. Anecdotal observations during collections for reproductive study.	Moderate	Yes	Low

Table 4.5-6

Risk Analysis for Amphibians Exposed to tPCBs and Other COCs in the Housatonic River PSA

Assessment Endpoint:	Community condition, survival, reproduction, development, and maturation of amphibians.

Harm/Magnitude	Weighting Factors (increasing confidence or weight)				
	Low	**Low/Moderate**	**Moderate**	**Moderate/High**	**High**
Yes/High				L-2, L-3, L-4, W-3	
Yes/Intermediate				W-2	
Yes/Low			C, B-3	L-1, B-1	
Undetermined		W-4	B-2		
No Harm				W-1	

C = Chemistry (tissue).

W = Wood frog study (1 = hatchling, 2 = larvae, 3 = metamorphs, 4 = GE Study).

L = Leopard frog study (1 = hatchling, 2 = larvae, 3 = metamorphs, 4 = adult).

B = Field study (1 = community, 2 = GE egg mass survey, 3 = anecdotal).

The resulting plots show that 9 out of the 12 endpoints indicated some degree of risk. The potential for the two GE studies to determine risk to amphibians was judged to be undetermined due to limitations in the study designs. The only endpoint that did not indicate potential risk was the earliest life stage wood frog toxicity endpoint. The plots also indicate that four endpoints exhibited a high degree of risk combined with a moderate to high confidence rating.

4.5.5 Sources of Uncertainty

Some of the major sources of uncertainty associated with the assessment of risks of tPCBs and TEQ to amphibians are briefly summarized below. A more complete list is presented in Appendix E.

4.5.5.1 EPA Studies

- Mobility of the animals and exposure to varying concentrations of tPCBs.
- Small-scale variability in exposure concentrations.
- Derivation of effects thresholds from the literature.
- Reduced performance of the reference animals potentially masking a natural sensitivity in the wood frogs with respect to mortality.
- Lack of replication in the leopard frog data.
- Extrapolation of individual-level effects to the level of population.

4.5.5.2 GE Studies

Wood Frog Study

- Lack of examination of contaminated media (i.e., sediment and water) as an exposure pathway.
- Potential confounding effects of larval density and predators on density and survivorship in the vernal pools.

Leopard Frog Egg Mass Survey

- Presence of egg masses is not necessarily indicative of a healthy reproducing population.
- Mobility of the animals and exposure to varying concentrations of tPCBs.
- No measurements of COC concentrations in the egg masses.

4.5.6 Conclusions

Overall, the amphibian ERA indicates significant risk to frog species in the Housatonic River PSA based on a WOE evaluation of multiple effects endpoints and their associated contaminant media. The available data implicate tPCBs as the stressor responsible for this impairment. The confidence in these conclusions is moderate to high, based on the concordance in predictions of risk from multiple measurement endpoints. The most compelling evidence for ecological risk comes from the two frog toxicity studies, which not only exhibited significant toxicological effects in both frog species and endpoints, but which also indicated a correlation between level of effect and sediment tPCB concentration in media during the developmental exposure of wood frogs.

4.5.6.1 Population Modeling

A stochastic population model was developed to determine whether effects from tPCBs on individual wood frogs influence wood frog populations within the Housatonic River PSA (Appendix E, Attachment E.3). The impact of tPCBs on the wood frog population was assessed by comparing population projections from a base population model (i.e., a wood frog population in the absence of tPCBs), with projections from population models that included the effect of tPCBs on population vital rates (see FEL 2002b). Two projection comparisons were performed based on simulations of (1) a non-declining base population, and (2) a declining base population. All models were constructed using RAMAS Metapop (Akçakaya 2002). Sensitivity analyses were also conducted on fertility and mortality rates, vital rates, environmental correlation, dispersal rates, and density-dependence to assess the robustness of the models to changes.

Findings from the population modeling were:

- Total PCBs have an impact on wood frog population growth and abundance.
- Total PCBs hasten population decline, reduce population numbers, and increase the likelihood of extinction.
- Data collected in the PSA provide field evidence supporting the population-level effects of tPCBs seen in the simulations.
- The relationship between sediment tPCB concentrations and adult male and female density indicate that increased tPCB concentration leads to decreased density—particularly for adult females.
- The sensitivity analyses were performed to assess the robustness of the model projections in the face of uncertainty regarding several important assumptions. The assumptions analyzed included vital rates, environmental correlation, dispersal rates, and density-dependence. Projections of time to extinction and terminal extinction risk were sensitive to all of these assumptions except for dispersal rates. The impact of tPCBs on population decline was not sensitive to any of these assumptions. Instead, the increased risk due to tPCB contamination relative to the uncontaminated base models was maintained in each sensitivity study. The models are robust in projecting the increased risk of population decline and extinction due to tPCB contamination.

4.5.6.2 Risks Within the PSA

The goal of the amphibian risk assessment was to determine the maximum concentration of tPCBs at which the resident amphibian populations are protected. This concentration could be termed the maximum acceptable threshold concentration (MATC). The MATC allows for some risk to individual amphibians, but those risks are considered to be such that the local population is not believed to be affected at or below the MATC. In calculating the MATC, first the threshold effect sizes were defined based on magnitude of effect. For the amphibians, effect sizes of 20% and 50% were considered the relevant effect sizes for analysis, based on their ecological relevance for amphibians. Most amphibians, unlike higher trophic-level predators, have a reproductive strategy wherein large numbers of young are produced, and minimal parental care of the young is typical. Higher juvenile output *and* mortality characterizes amphibians relative to animals located higher in the food web. There is a compensation mechanism in many amphibian species resident in the PSA, such that an effect size of 5% to 10% may be of less biological relevance to the local population (whereas this could be a critical effect size in a higher-order predator that has more limited reproductive resources).

The MATC for amphibians was based on the integration of the two sensitive endpoints (metamorph malformation and sex ratios). The rationale for integrating the point estimates of the two endpoints was that it is difficult to determine which endpoint is more sensitive with respect to population-level impacts. Furthermore, the two endpoints may act in concert to limit the viability of local wood frog populations. For example, of the Phase III wood frog metamorphs that were malformed, a higher proportion of the total number of malformed animals was female (the proportion of malformed males ranged from 0 to 42% at the four pools with the highest sediment tPCBs; the proportion of malformed females ranged from 33 to 67%).

4.5.6.2.1 Sediment

A sediment MATC of 3.27 mg/kg tPCBs was determined based on the results of the point estimate calculations presented in Appendix E, Section E.4.3.3. This concentration was just below the EC_{20} values for the Phase III metamorph malformation endpoint. The MATC of 3.27 mg/kg for this endpoint is believed to provide adequate protection for other amphibian species.

4.5.6.2.2 Tissue

The threshold concentration range for amphibian tissues was 0.60 mg/kg ww to 6.54 mg/kg ww tPCBs, and was based on the sex ratio endpoint (both an EC_{20} and EC_{50}) and the Phase III metamorph malformation endpoint (an EC_{50} point estimate). A tissue MATC of 1 mg/kg ww, therefore, was selected to provide a balance between the protection of other amphibian receptors and the lower tissue MATC of approximately 0.65 mg/kg ww tPCBs.

MATCs for PCBs Used to Assess Risks to Amphibians

- The soil and sediment MATC of 3.27 mg/kg tPCBs was used as a measure of the potential for adverse effects to amphibians. This concentration was developed in the risk assessment for the PSA using multiple lines of evidence (e.g., amphibian community studies, in situ and laboratory-exposure toxicity testing) and was selected as the concentration at which some sensitive endpoints exhibited apparent responses, but the magnitude of responses was not large. Above a concentration of 3.27 mg/kg tPCBs, numerous endpoints indicated ecologically significant responses.
- The tissue MATC of 1 mg/kg ww tPCBs was used as a conservative measure of the potential for adverse effects to amphibians. This concentration was developed considering the frequency of adverse effects observed in the literature studies, in the site-specific studies, and in an effort to compensate for the uncertainty associated with the sensitivity of salamander species.

The WOE assessment of amphibian endpoints indicated a high probability of risk of ecologically significant effects at PCB concentrations observed in the PSA vernal pools included in the studies. Extrapolation to other areas of the PSA suggests that amphibian populations are impacted throughout much of the PSA, with leopard frogs impacted at a wide range of sediment concentrations (likely due to prolonged contact with sediment PCB concentrations, which were not measured in the study), and with stronger responses from wood frogs expected in the more highly contaminated vernal pools.

A total of 533 floodplain soil samples (although referred to as “soil,” these samples occur in habitats that are either temporarily or semi-permanently inundated; they were not labeled as “sediment” so as to distinguish them from samples collected in the river proper versus the floodplain) were collected from temporary and permanent pools in the PSA (Figures E.1.6-1 to E.1.6-21, ERA Volume 1, Section 1). These pools provide breeding habitat for wood frogs, leopard frogs, and other amphibians. Using these data, and spatially weighted data, the percentage of each reach posing low, intermediate, and high risk to amphibians is shown below.

Reach	tPCB Data Source	Low Risk <3.27 mg/kg tPCBs	Intermediate Risk 3.27-9.54 mg/kg tPCBs	High Risk >9.54 mg/kg tPCBs
5A	Discrete	35%	20%	45%
5A	Spatially Weighted	21%	22%	57%
5B	Discrete	16%	22%	62%
5B	Spatially Weighted	5%	10%	85%
5C	Discrete	56%	14%	30%
5C	Spatially Weighted	42%	20%	38%

4.5.6.3 Extrapolation of Risk Estimates Downstream of the PSA

Potential risks to amphibians living in the river and floodplain downstream of the PSA were evaluated. The first step in this process was developing a detailed list of amphibians known or expected to occur along the Housatonic River between the Woods Pond Dam and Long Island Sound. Potential amphibian habitat was then identified using USGS topographical maps, National Wetland Inventory [NWI] maps, and recent aerial photos. Breeding habitats were identified based on map classifications shown on NWI maps, analysis of aerial photos, or limited field reconnaissance surveys.

Ten salamander and nine toad and frog species could potentially occur downstream of the PSA. Leopard frogs, green frogs, bullfrogs, and red-spotted newts are likely the most common species within the Housatonic River and the associated semi-permanent pools and backwaters. Wood frogs and spotted salamanders are likely the most common breeding amphibians within temporary vernal pools in the floodplain and adjacent forests.

The risk assessment focused on vernal pools, but such detailed data were not available downstream of the PSA, thus, the parameter of interest was tPCBs in floodplain soil and sediment in general. Sediment was included to account for more aquatic amphibians such as bullfrogs, and to account for the aquatic life phases of leopard frogs. A soil and sediment MATC of 3.27 mg/kg tPCBs was derived from the amphibian risk assessment conducted for the PSA, as described above and, in more detail, in Appendix E, Section E.4.3. Inverse Distance Weighting (IDW) was used to interpolate PCB concentrations to the limit of the 100-year floodplain (10-

year floodplain contours are not available downstream of the PSA) using the 0- to 6-inch depth data from the floodplain downstream of the PSA.

Areas where the 3.27-mg/kg tPCB threshold was exceeded, indicating potential risks to amphibians due to PCBs in floodplain soil and sediment, are indicated in Appendix E, Figures E.4-7 and E.4-8. Several large areas of potential risk are located in the area between Woods Pond and Rising Pond, with only small isolated areas of potential risk downstream of Rising Pond.

4.6 REFERENCES

Akçakaya, H.R. 2002. *RAMAS Metapop: Viability Analysis for Stage-structured Metapopulations* (Version 4.0). Applied Biomathematics, Setauket, New York.

Berven, K.A. 1990. Factors affecting population fluctuations in larval and adult stages of the wood frog (*Rana sylvatica*). *Ecology* 71:1599-1608.

Birge, W.J., J.A. Black, and A.G. Westerman. 1978. Effects of polychlorinated biphenyl compounds and proposed PCB-replacement products on embryo-larval stages of fish and amphibians. University of Kentucky. Lexington, KY. pp. 1-33.

Burkett, R.D. 1984. An ecological study of the cricket frog, *Acris crepitans. Vert. Ecol. Syst.* 10:89-103.

DeGraaf, R.M. and D.D. Rudis. 1983. *Amphibians and Reptiles of New England: Habitats and Natural History*. University of Massachusetts Press, MA.

Eisler, R. and A.A Belisle. 1996. Planar PCB Hazards to Fish, Wildlife, and Invertebrates: A Synoptic Review. National Biological Service Report 31, Washington, DC.

FEL (Fort Environmental Laboratories, Inc.). 2002a. *Final Report – Frog Reproduction and Development Study. 2000 Rana sylvatica Vernal Pool Study*. Study protocol no.: WESR01–RSTS03–1. Prepared by Fort Environmental Laboratories Inc., Stillwater, OK.

FEL (Fort Environmental Laboratories, Inc.). 2002b. *Final Report – Frog Reproduction and Development Study. 2000 Rana pipiens Reproduction and Development Study*. Study protocol no.: WESR01–RSTS03–1. Prepared by Fort Environmental Laboratories Inc., Stillwater, OK.

Fontenot, L.W., G.P. Noblet, J.M. Akins, M.D. Stephens, and G.P. Cobb. 2000. Bioaccumulation of polychlorinated biphenyls in ranid frogs and northern water snakes from a hazardous waste site and a contaminated watershed. *Chemosphere* 40:803-809.

Gilbert, M., R. Leclair, Jr., and R. Fortin. 1994. Reproduction of the northern leopard frog (*Rana pipiens*) in floodplain habitat in the Richelieu River, P. Quebec, Canada. *Journal of Herpetology* 28(4):465-470.

Ginzburg, L.R., L.B. Slobodkin, K. Johnson, and A.G. Bindman. 1982. Quasiextinction probabilities as a measure of impact on population growth. *Risk Analysis* 2:171-181.

Gosner, K. 1960. A simplified table for staging anuran embryos and larvae with notes on identification. *Herpetologica* 16(2):183-190.

Gutleb, A.C., J. Appelman, M.C. Bronkhorst, J.H.J. van den Berg, A. Spenkelink, A. Brouwer, and A.J. Murk. 1999. Delayed effects of pre- and early-life time exposure to polychlorinated biphenyls on tadpoles of two amphibian species (*Xenopus laevis* and *Rana temporaria*). *Environ. Toxicol. Pharm.* 8:1-14.

Gutleb, A.C., J. Appelman, M. Bronkhorst, J.H.J. van den Berg, and A.J. Murk. 2000. Effects of oral exposure to polychlorinated biphenyls (PCBs) on the development and metamorphosis of two amphibian species (*Xenopus laevis* and *Rana temporaria*). *Sci. Tot. Environ*. 262:147-157.

Hayes, T.B. 2000. Endocrine Disruption in Amphibians. In Chapter 10A, *Ecotoxicology of Amphibians and Reptiles.* D.W. Sparling et al., editors. SETAC Press, Pensacola, FL.

Hine, R.L., B.L. Les, and B.F. Hellmich. 1981. Leopard frog populations and mortality in Wisconsin, 1974-76. Wis. Dep. Nat. Resour., Tech. Bull. No. 122.

Huang, Y.W., M.J. Melancon, R.E. Jung, and W.H. Karasov. 1998. Induction of cytrochrome P450-associated monooxygenases in northern leopard frogs, *Rana pipiens*, by 3,3’4,4’5-pentachlorobiphenyl. *Environ. Tox. Chem*. 17(8):1564-1569.

Menzie, C., M.H. Henning, J. Cura, K. Finkelstein, J. Gentile, J. Maughan, D. Mitchell, S. Petron, B. Potocki, S. Svirsky, and P. Tyler. 1996. Special report of the Massachusetts Weight-of-Evidence Workgroup: A weight of evidence approach for evaluating ecological risks. *Human and Ecological Risk Assessment* 2:277-304.

Mikamo, K. and Witschi, E. 1964. Masculinization and breeding of the WW *Xenopus*. *Experentia* 20:622-623.

Ouellet, M. 2000. Chapter 11, Amphibian Deformities: Current State of Knowledge. In *Ecotoxicology of Amphibians and Reptiles*. Society of Environmental Toxicology and Chemistry (SETAC), Pensacola, FL. 904 p.

Reeder, A.L., G.L. Foley, D.K. Nichols, L.G. Hansen, B. Wikoff, S. Faeh, J. Eisold, M.B. Wheeler, R. Warner, J.E. Murphy, and V.R. Beasley. 1998. Forms and prevalence of intersexuality and effects of environmental contaminants on sexuality in cricket frogs (*Acris crepitans*). *Environ. Health Persp.* 106:261-266.

Richards, C.M. and Nace, G.W. 1978. Gynogenetic and hormonal sex reversal used in tests of the XX-XY hypothesis of sex determination in *Rana pipiens*. *Growth* 42:319-331.

Rosenshield, M.L., M.B. Jofre, and W.H. Karasoy. 1999. Effects of polychlorinated biphenyl 126 on green frog (*Rana clamitans*) and leopard frog (*Rana pipiens*) hatching success, development, and metamorphosis. *Environmental Toxicology and Chemistry* 18(11):2478-2486.

Savage, W.K., F.W. Quimby, and A.P. DeCaprio. 2002. Lethal and sublethal effects of polychlorinated biphenyls on *Rana sylvatica* tadpoles. *Environmental Toxicology and Chemistry* 21(1):168-174.

Sparling, D.W., G. Linder, and C.A. Bishop. Editors. 2000. *Ecotoxicology of Amphibians and Reptiles*. Society of Environmental Toxicology and Chemistry (SETAC), Pensacola, FL. 904 p.

Taylor A.C. and J.J. Kollros. 1946. Stages in the normal development of *Rana pipiens* larvae. *The Anatomical Record* 94:7-23.

Woodlot Alternatives, Inc. 2002. *Ecological Characterization of the Housatonic River*. Prepared for U.S. Environmental Protection Agency.

Woodlot Alternatives, Inc. 2003. *Amphibian Reproductive Success within Vernal Pools Associated with the Housatonic River, Pittsfield to Lenoxdale, Massachusetts*. Report prepared for Weston Solutions, Inc. DCN: 99-1275.

5. ASSESSMENT ENDPOINT – SURVIVAL, GROWTH, AND REPRODUCTION OF FISH

Highlights

Exposure

- COCs evaluated were total PCBs (tPCBs), 2,3,7,8-TCDD TEQ, and polycyclic aromatic hydrocarbons (PAHs).
- Whole body fish concentrations in five representative warmwater fish species were used to evaluate exposures to both tPCBs and TEQ in the PSA. Measured or estimated whole body concentrations in warmwater and coldwater fish species were considered for habitats downstream of Woods Pond.
- Sediment concentrations were used to evaluate risks to fish from PAHs.

Effects

- Site-specific toxicity tests (Phase I and Phase II) indicate adverse effects at contaminated locations and general PCB dose-dependency.
- Literature review indicates that PCB and TEQ threshold tissue concentrations from the literature are in the same range as those from site-specific toxicity tests.
- Field surveys (fish community and reproduction studies) indicate lack of substantial effects (e.g., reproductive failure), but cannot be used to quantify lesser-magnitude responses.
- Malformations in fish (i.e., lesions in goldfish and deformities in other adult fish) have been observed and may be contaminant-related, but cannot be linked to a specific COC and have unknown implications for local population responses.

Risk Characterization

- An intermediate probability of adverse impacts to fish from tPCBs and/or TEQ is predicted throughout the PSA. Endpoints such as reproduction and development are likely affected, but mortality of adults is unlikely.
- Risks attributable to PAHs are low.
- Impacts downstream of Woods Pond are limited to marginal risks for coldwater fish (trout).
- Magnitude of impact is predicted to be intermediate in all PSA reaches; adverse effects, although high in probability, may not be impacting local population abundance under current environmental conditions.

5.1 INTRODUCTION

The purpose of this section is to characterize and quantify the current and potential risks posed to fish, focusing on tPCBs and other COCs. A Pre-ERA (Appendix B) was conducted to identify contaminants, other than tPCBs, that posed potential risks to aquatic biota in the PSA. These

COPCs were further screened for specific relevance to fish, and the following COC groups were retained for evaluation in the detailed assessment: tPCBs, dioxin-like TEQ, and polycyclic aromatic hydrocarbons (PAHs).

A stepwise approach was used to assess the risks of COCs to fish in the Housatonic River. The four main steps in this process included the following:

1. Development of a conceptual model (Figure 5.1-1).
2. Assessment of exposure of fish to COCs (Figure 5.1-2).
3. Assessment of the effects of COCs to fish (Figure 5.1-3).
4. Characterization of risks to fish (Figure 5.1-4).

> This section provides a summary of the ERA for fish, which is presented in detail in Appendix F.

5.1.1 Conceptual Model

The COCs identified for fish exhibit both direct (i.e., contact with contaminated media) and indirect (i.e., food web bioaccumulation) pathways, with emphasis on the latter pathway for PCBs. The conceptual model presented in Figure 5.1-1 illustrates the exposure pathways for fish in the PSA. Tissue concentrations reflect the net COC uptake from food, sediment, overlying water, and pore water, and therefore, integrate all primary exposure pathways of interest. For PAHs, sediment was considered the most relevant exposure medium. Field (1996) and Varanasi et al. (1989) found that fish tissue residue concentrations of parent PAH compounds do not provide a useful measure of exposure to fish. Therefore, sediment, rather than tissue, exposure concentrations were used for linkage to effects.

Five fish species, representing different trophic levels and exposure routes, were selected as the representative species for the ERA.

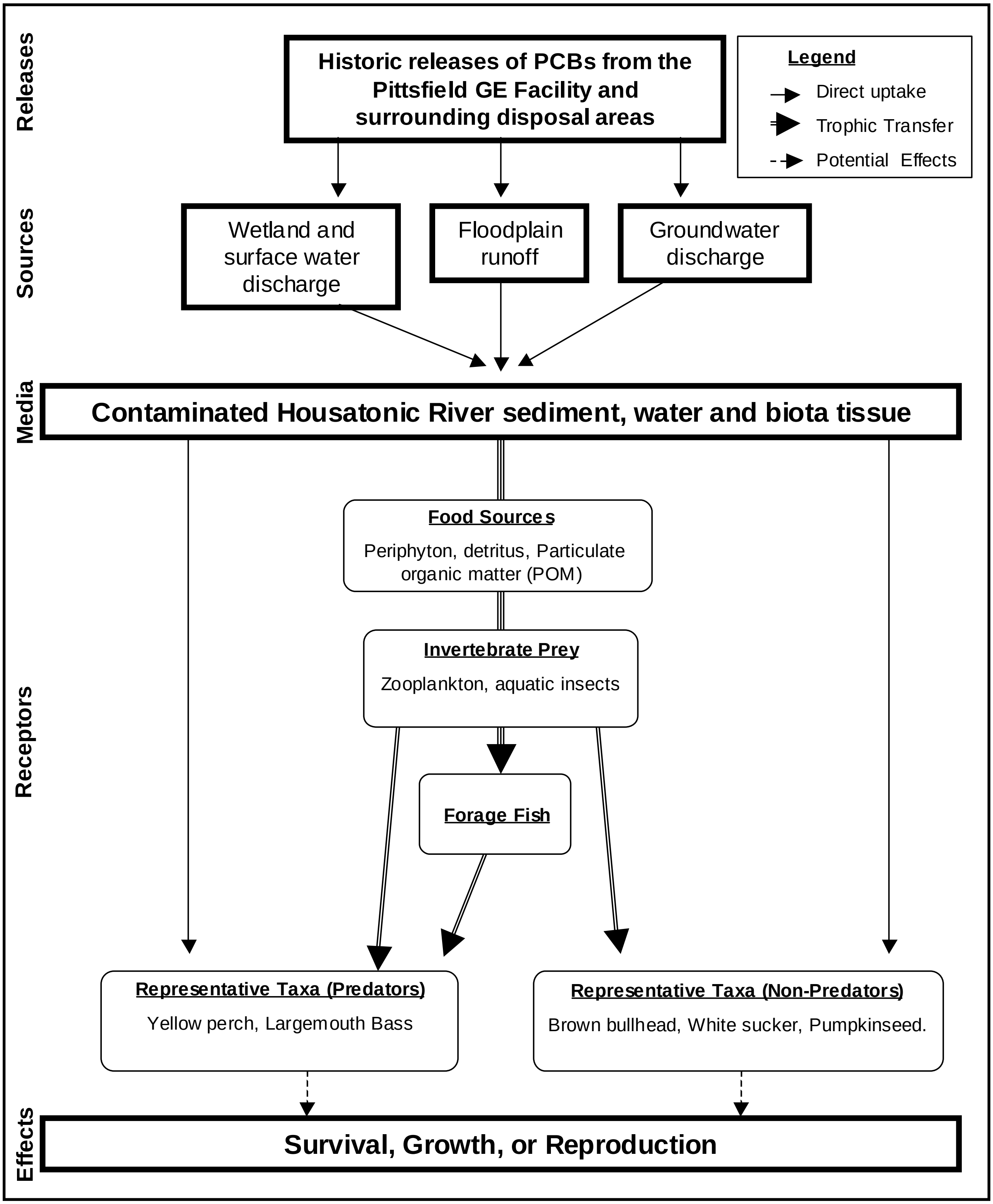


Figure 5.1-1 Conceptual Model Diagram: Exposure Pathways for Fish Exposed to COCs in the Housatonic River

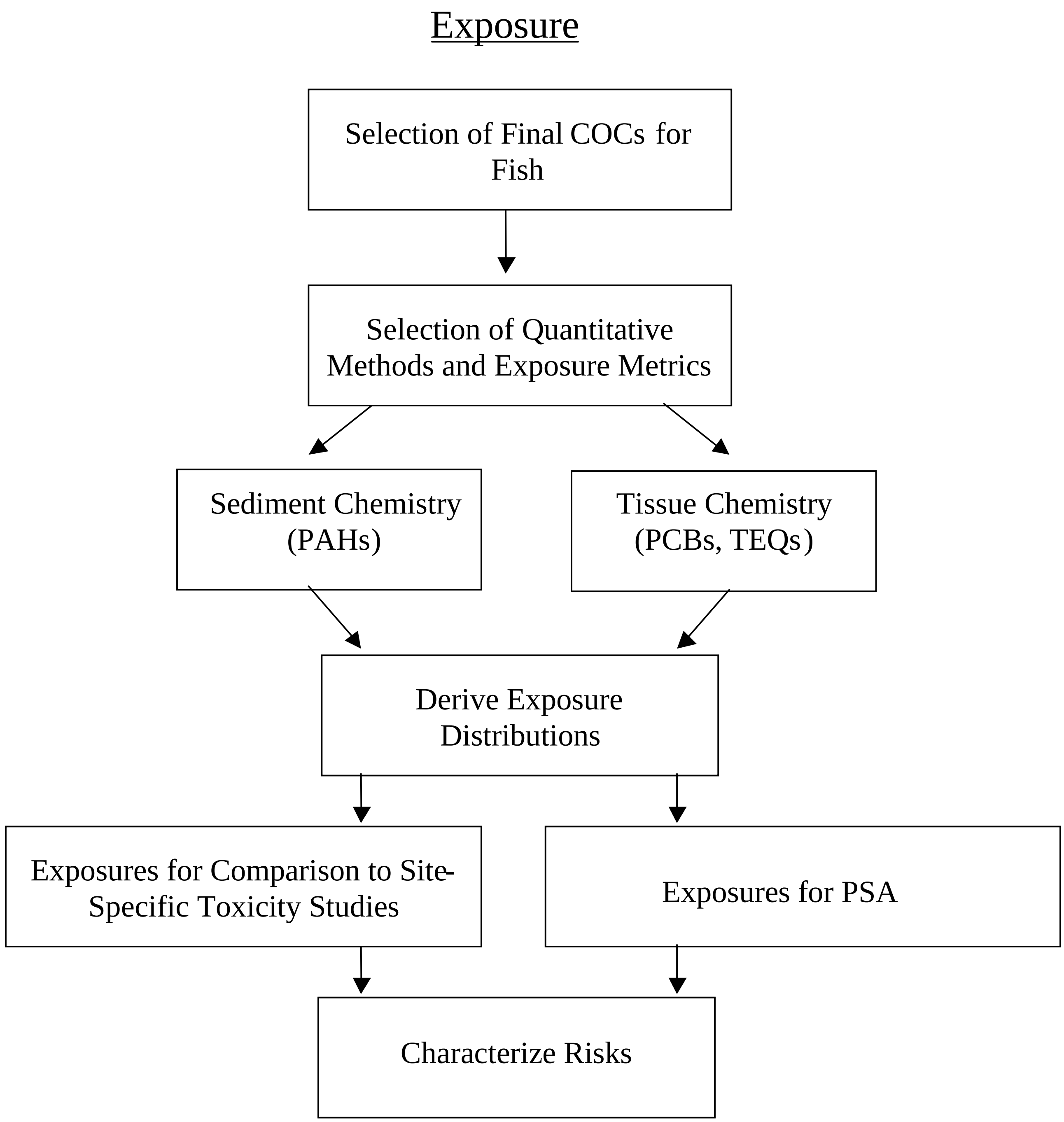


Figure 5.1-2 Overview of Approach Used To Assess Exposure of Fish to COCs in the Housatonic River

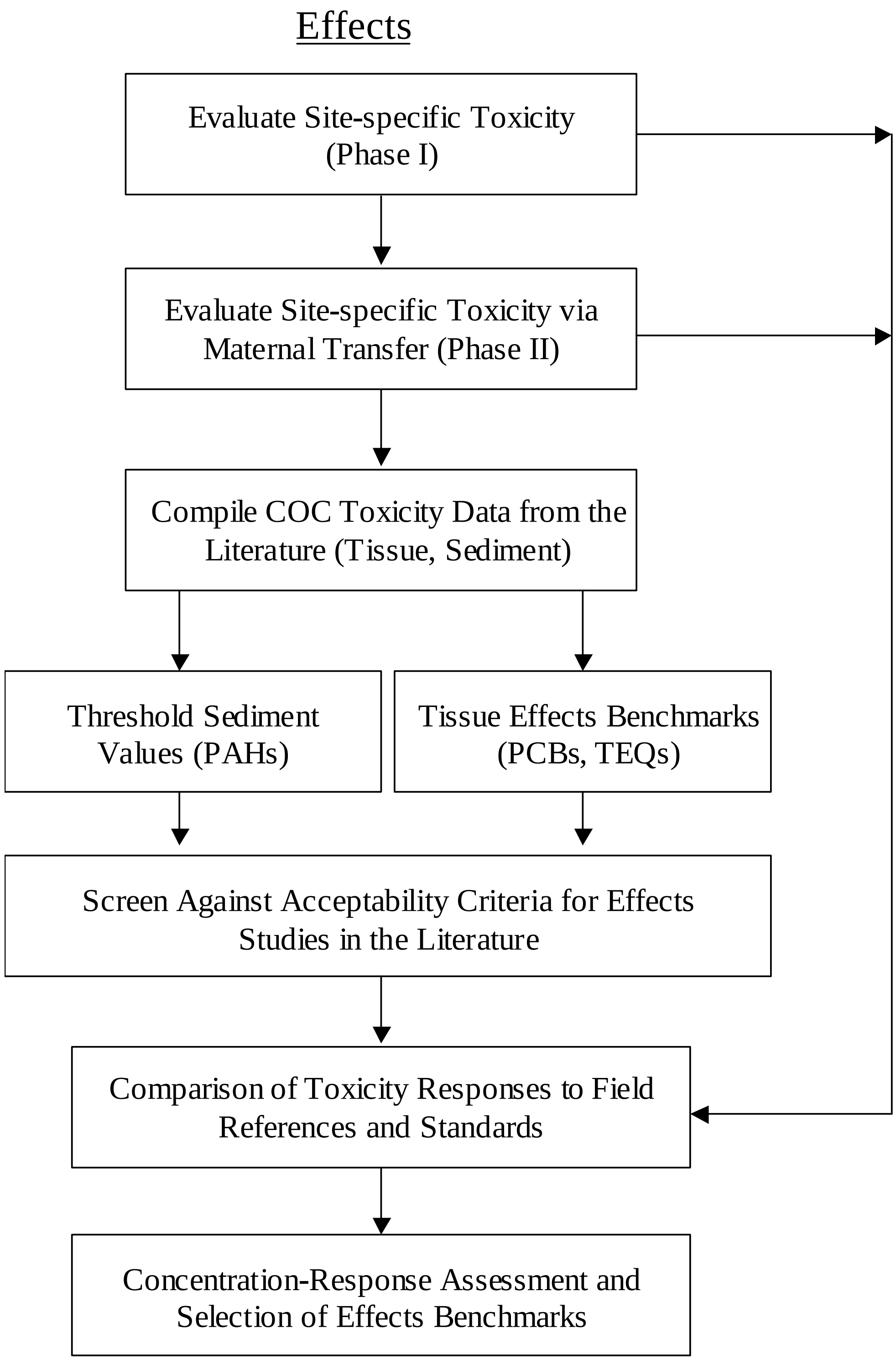


Figure 5.1-3 Overview of Approach Used To Assess the Effects of COCs to Fish in the Housatonic River

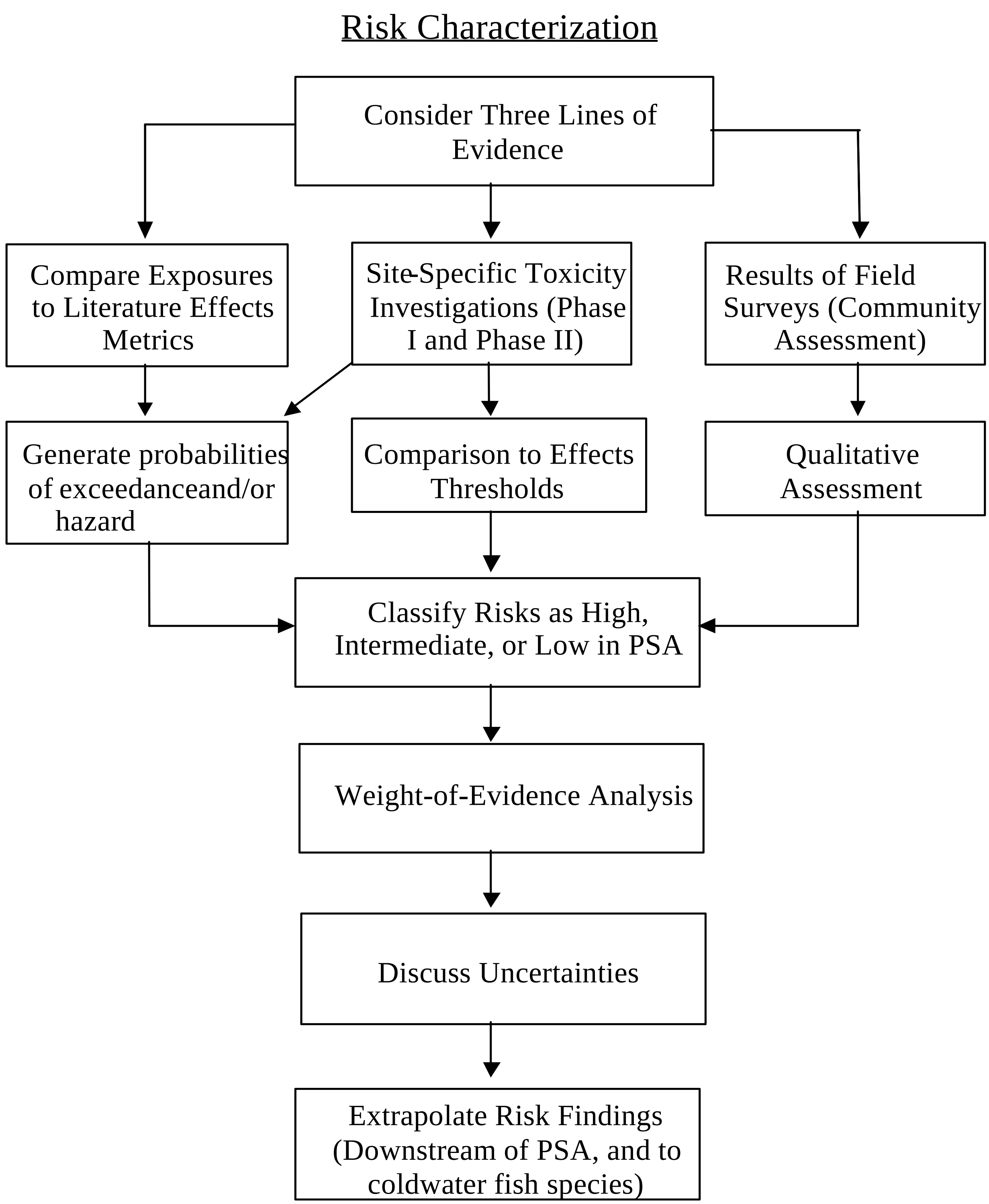


Figure 5.1-4 Overview of Approach Used To Characterize the Risks of COCs to Fish in the Housatonic River

Representative Species for Primary Study Area

- Largemouth bass (*Micropterus salmoides*) – predator
- Yellow perch (*Perca flavescens*) – predator
- Brown bullhead (*Ameiurus nebulosis*) – bottom feeder
- White sucker (*Catostomus commersoni*) – bottom feeder
- Pumpkinseed (*Lepomis gibbosus*) – forage fish

Criteria considered in selecting representative fish species included trophic level and feeding preferences, abundance and biomass in the study area, availability of site-specific data, and availability of toxicological data. Because trout have greater importance downstream of the PSA (due to the presence of suitable coldwater habitat), a separate analysis for trout was conducted as a part of the evaluation of risk downstream of the PSA.

The assessment endpoint that is the subject of this section is the survival, growth, and reproduction of fish. The selected measurement endpoints were:

Measurement Endpoints for Fish

- Determine the extent of effects by comparing the concentrations of COCs in sediment to the concentrations reported to affect the survival, growth, or reproduction of fish.
- Compare the concentrations of COCs in fish tissues to the tissue concentrations that may cause adverse effects, based on site-specific fish toxicity studies.
- Compare the concentrations of COCs in fish tissues to concentrations documented in the literature to result in adverse effects.
- Evaluate field survey information (fish biomass study, ecological characterization study, largemouth bass habitat and reproduction study, field observations of fish deformities) to qualitatively assess potential effects.

The approach used to characterize risks to fish was based upon evaluation of site-specific toxicity investigations, chemical measurements of fish tissue and sediment, biological/ community assessments, and literature reviews.

Summary of Studies Used To Characterize Risks to Fish

- Phase I site-specific studies – Investigated contaminant accumulation and reproductive and developmental effects in largemouth bass and in their offspring from the PSA, Rising Pond, and a reference area.
- Phase II site-specific studies – Simulated the maternal transfer of contaminants to developing oocytes, and assessed reproductive and developmental responses.
- Sampling and analysis of fish tissue.
- Sampling and analysis of sediment.
- Field studies – EPA Fish Community and Ecological Characterization Study (Appendix A) and GE Largemouth Bass Community, Reproduction, and Habitat Study.
- Evaluation of adult fish health, including histopathology (Blazer 2004), viral analyses (Coll 2004), and gross external deformities observed during sampling.
- Literature review – Evaluated the range of PCB, TEQ, and PAH concentrations observed to result in adverse effects to fish.

5.2 EXPOSURE ASSESSMENT

In the exposure assessment, COPCs that were retained in the Pre-ERA (Appendix B) were screened specifically for relevance to fish. The most relevant exposure data were those collected within the PSA. More limited tissue data were also available for areas downstream of the PSA. In some cases, upstream data were used to standardize downstream data for use in the ERA (e.g., lipid-based conversions of fillet PCB concentrations to whole-body concentrations).

5.2.1 Screening of COPCs for Fish

The Pre-ERA (Appendix B) developed separate lists of COPCs for fish tissue, water, and sediment. Water and sediment screening included comparisons to thresholds considered protective of aquatic life, including invertebrates and fish. A receptor-specific screen was conducted to refine the COPC list. The first step in this subsequent screening was to ensure that no bioaccumulative COPCs that could be of concern to fish were eliminated prematurely. A detailed discussion of the screening of COPCs is presented in Appendix F; the contaminants retained as COCs for fish were:

COCs for Fish

- Chlorinated organic compounds – PCBs as tPCBs and TEQ, dioxins/furans expressed as TEQ equivalents.
- PAHs – Total PAH, benzo(g,h,i)perylene, indeno(1,2,3-cd)pyrene, phenanthrene, anthracene, benzo(a)anthracene, pyrene, fluorene, and fluoranthene.

All pesticides, including 4-4’-DDE, were eliminated from further consideration in the fish assessment based on comparisons of measured tissue concentrations to threshold effects levels from the literature. All metals, including mercury, were also eliminated from consideration as a tissue-based COPC in the Tier I Pre-ERA based on several lines of evidence including comparison with background concentrations and comparisons of fish tissue concentrations to threshold levels for fish health documented in the literature (Attachment F.1).

5.2.2 Tissue Chemistry Assessment (Exposure to PCBs and TEQ)

The most robust data set for fish tissue concentrations of tPCBs and TEQ was collected by EPA from September 1998 through October 2000. Other fish tissue tPCB data collected within the PSA by GE and others from 1977 to 2002 are also available. These additional data sets were evaluated, and either the inclusion or exclusion of these data would result in very similar risk conclusions.

Fish Chemistry Types Considered in the ERA

- CM – Composite samples – represent the combination of multiple fish, typically young-of-year or other small fish.
- WB – Whole body samples – represent the analysis of single larger fish, often for species that were not considered in the human health risk assessment (e.g., white sucker).
- WB-R – Whole body reconstituted samples – represent individual fish concentrations that were calculated/estimated using separate fillet and offal measurements.

5.2.2.1 Total PCBs

Table 5.2-1 presents summary statistics for fish tissue tPCB concentrations in the PSA by sample type and species, for all representative species. In general, there were relatively consistent fish tissue concentrations across the entire PSA. The composite fish tissue samples had lower PCB concentrations than other tissue types; the small fish in the composite samples were typically representative of younger fish or of species that are small even as adults (i.e., dace).

Table 5.2-2 presents summary statistics for lipid-normalized tPCB concentrations by sample type, species, and reach. The mean and median concentrations in Table 5.2-2 indicate that the highest PCB concentrations (i.e., median >2,000 mg/kg lipid) were observed in either adult (WB-R) predator fish (due to biomagnification in the food web) or in adult bottom fish (due to direct contact with contaminated sediment and associated benthic prey). PCB concentrations increased with increasing age of fish (see Figure 5.2-1) and were positively correlated with lipid content.

Table 5.2-1

Total PCB Concentrations in Representative Species Fish Tissue (mg/kg wet weight [ww]) in the PSA; Data from EPA Tissue Collections (1998-2000)

Sample Type	Species	Sample Count	Min	25th Percentile	Median	Mean	75th Percentile	Max
WB-R	BB	43	7.19	25.3	32.3	37.6	45.6	103
	LB	38	10.9	42.3	67.8	97.1	125	424
	PS	51	7.82	23.2	34.6	36.7	44.7	82.1
	YP	75	6.11	61.3	76.1	87.3	104	329
WB	BB	2	20.9	21.3	21.7	21.7	22.0	22.4
	GF*	42	10.8	95.5	143	163.6	215	447
	LB	26	3.03	22.7	36.5	57.1	78.4	220
	WS	57	7.96	36.2	56.6	70.6	86.5	216
CM	LB	12	9.03	19.9	26.1	27.9	36.3	51.2
	PS	9	8.8	26.4	27.5	26.2	27.9	35.1
	YP	15	16.5	27.4	31.0	31.4	35.7	46.9

*Goldfish (GF) were not selected as a representative species, but were included because of large sample size and high tPCB concentrations.

BB Brown Bullhead
LB Largemouth Bass
PS Pumpkinseed
WS White Sucker
YP Yellow Perch
CM Composite
WB Whole Body
WB-R Reconstituted Whole Body

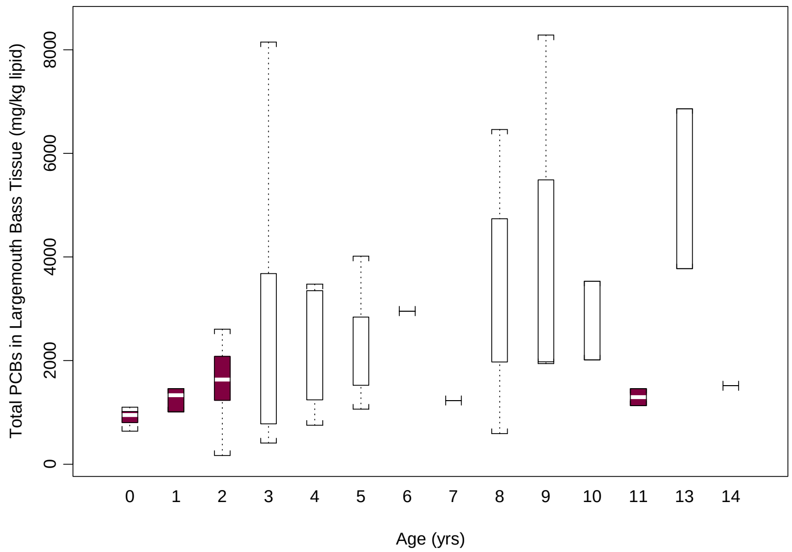
Table 5.2-2

Lipid-Normalized* tPCB Concentrations (mg/kg lipid) for Representative Species in the PSA; Data from EPA Tissue Collections (1998-2000)

Sample Type	Species	Sample Count	Min	25th Percentile	Median	Mean	75th Percentile	Max
WB-R	BB	43	224	1010	1520	2160	1870	14700
	LB	38	591	1600	2490	2960	3720	8280
	PS	51	210	692	1270	1370	1960	3600
	YP	75	154	1410	2060	2510	2920	9990
WB	BB	2	2030	NA	2060	2060	NA	2090
	GF*	42	578	1050	1480	1550	1770	4710
	LB	26	168	837	1420	1980	2270	8150
	WS	57	252	927	1480	2780	2900	44700
CM	LB	12	636	936	1080	1440	1490	3580
	PS	9	664	758	854	998	1070	1760
	YP	15	681	990	1210	1340	1410	3350

*Goldfish (GF) were not selected as a representative species, but were included because of large sample size and high tPCB concentrations.

BB	Brown Bullhead
LB	Largemouth Bass
PS	Pumpkinseed
WS	White Sucker
YP	Yellow Perch
CM	Composite
WB	Whole Body
WB-R	Reconstituted Whole Body



The shaded box represents the interquartile range, the white bar represents the median, and the whiskers extend to the full range of the data.

Figure 5.2-1 Box-and-Whisker Plot of Largemouth Bass tPCB Concentrations (Lipid-Normalized) Versus Age

Fish tissue PCB data for samples collected downstream of Woods Pond were also evaluated to determine patterns in concentrations for the Rest of River area. Overall, fish tissue tPCB concentrations were significantly lower downstream of the PSA (Table 5.2-3) relative to those measured in the PSA. Details on the various sources of downstream fish tissue data, and on the non-EPA data sets considered within the PSA, are provided in Appendix F.

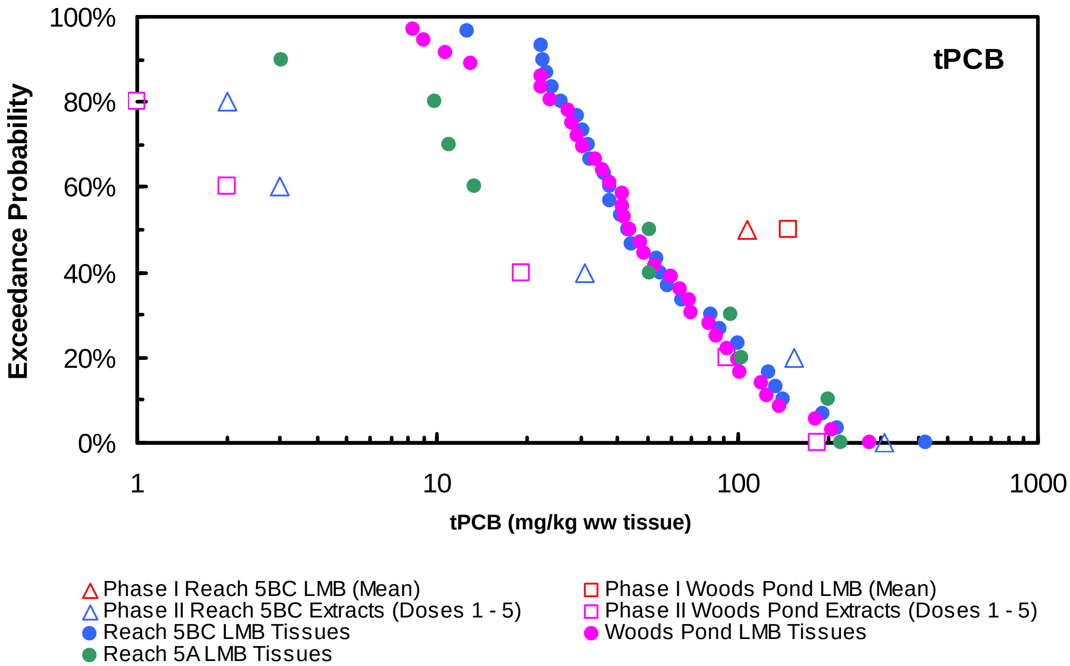
Because largemouth bass were evaluated in the site-specific toxicity studies and were also collected in large numbers in the EPA field collections, the respective concentrations can be compared directly. Figure 5.2-2 compares the probability distributions of largemouth bass tPCB concentrations against the respective exposure concentrations for Phase I and Phase II toxicity studies in PSA reaches. The tPCB concentration ranges are similar, and all but the extreme upper tail of the sampling distribution was represented in the Phase II dosing regime. The mean tPCB concentrations in the Phase I study represent the upper 20th percentile of the overall sampling distribution, and therefore, are slight overestimates of the mean concentrations observed in the field.

Table 5.2-3

Summary of tPCB Concentrations (mg/kg ww) from EPA Samples Collected in Reach 8

Sample Type	Species	Sample Count	Min	25th Percentile	Median	Mean	75th Percentile	Max
WB-R	BB	7	3.46	3.58	3.83	4.97	3.93	12.5
	LB	17	1.30	23.0	28.8	38.2	40.6	145
	PS	13	5.87	12.7	13.7	14.6	14.9	26.0
	YP	6	13.3	23.5	31.8	50.0	41.5	158
WB	LB	14	12.8	18.1	22.4	24.2	29.3	41.5
CM	LB	5	9.98	10.5	10.6	11.9	13.0	15.3
	PS	5	9.74	9.98	10.4	10.5	10.7	11.8
	YP	5	8.08	8.70	8.91	9.62	11.2	11.2

BB Brown Bullhead
LB Largemouth Bass
PS Pumpkinseed
YP Yellow Perch
CM Composite
WB Whole Body
WB-R Reconstituted Whole Body



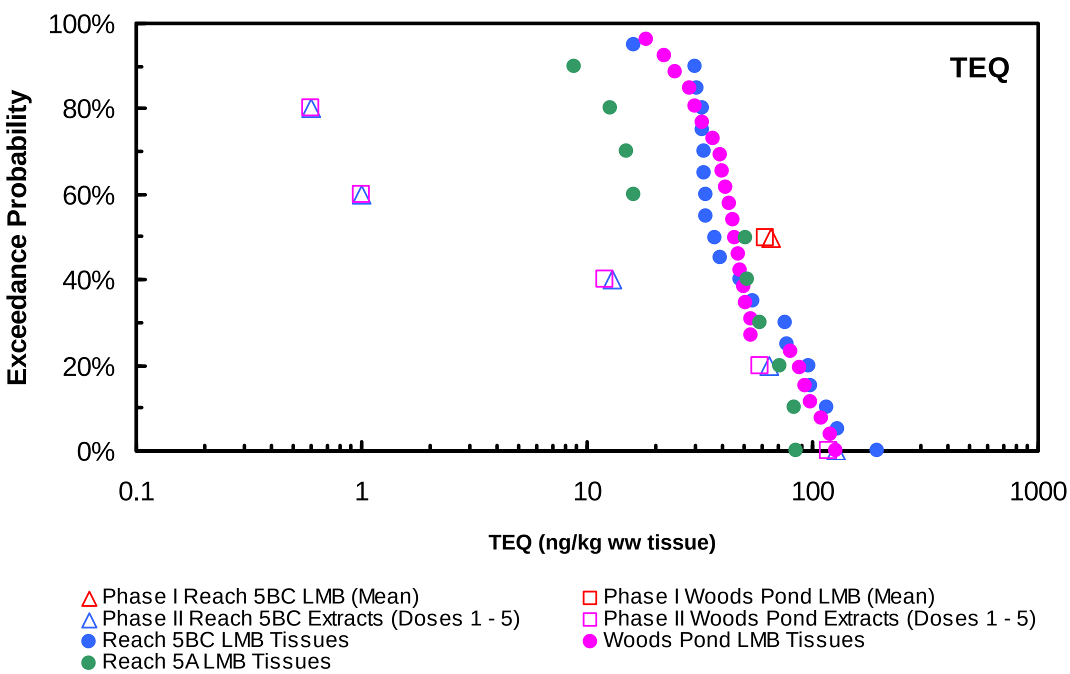


Figure 5.2-2 Comparison of tPCB and TEQ Exposure Concentrations from USGS Toxicity Studies to Distributions of EPA Field Collections of Largemouth Bass in the PSA

5.2.2.2 2,3,7,8-TCDD TEQ

Fish tissue TEQ was calculated using the approach outlined in Appendix C.10 (Van den Berg et al. 1998). Table 5.2-4 presents the summary statistics for TEQ concentrations (representative species only) in PSA fish tissue by sample type and species, with DL substitution for non-detects (see Appendix C.2). As with tPCBs, there are trends of increasing TEQ concentrations with age and fish size.

For TEQ, the variability in concentrations across PSA largemouth bass samples was somewhat lower than for tPCBs (Figure 5.2-2), with the vast majority of data between 10 and 100 ng/kg TEQ. Although representative of central tendencies in field data, the Phase I exposure concentrations do not span the full range of TEQ found in the field. However, the highest three doses in the Phase II study cover nearly the entire range of concentrations observed in the field study.

5.2.3 Sediment Chemistry Assessment (Exposure to PAHs)

Because PAHs are readily metabolized by most aquatic biota, including fish (Johnson 2000; Johnson et al. 2002), no data were collected on fish tissue concentrations in the PSA for the eight individual parent PAHs retained as COCs, or for total PAHs. Exposure for these contaminants, therefore, was assessed based on sediment concentrations only. The median total PAH concentrations were below 10 mg/kg in all PSA reaches. However, localized observations of elevated PAH concentrations were observed in Reach 5A and Reach 5C, and the mean total PAH concentrations exceeded 10 mg/kg in those reaches. Appendix F provides further details on PAH concentration distributions in sediment.

Table 5.2-4

TEQ in Representative Species Fish Tissue in the PSA with DL Substitution for NDs (ng/kg ww); Data from EPA Fish Collections (1998-2000)

Sample Type	Species	Sample Count	Min	25th Percentile	Median	Mean	75th Percentile	Max
WB-R	BB	31	22.9	48.8	62.2	70.3	86.2	152
	LB	29	20.3	48.0	58.5	78.2	100	196
	PS	31	24.3	33.6	41.3	44.5	51.3	108
	YP	45	17.8	74.2	91.7	102	122	246
WB	BB	1	43.3	NA	NA	NA	NA	43.3
	GF*	29	37.8	69.0	104	118	142	288
	LB	15	12.6	23.6	37.5	43.0	54.1	86.9
CM	LB	12	29.2	36.8	41.9	46.1	57.2	63.1
	PS	9	27.7	31.7	32.7	36.1	42.0	47.1
	YP	15	34.2	41.3	42.9	45.8	52.3	63.1

DL=detection limit

ND=non-detect

*Goldfish (GF) were not selected as a representative species, but were included because of large sample size, high tPCB concentrations, and observation of lesions in Woods Pond goldfish.

BB Brown Bullhead
LB Largemouth Bass
PS Pumpkinseed
YP Yellow Perch
CM Composite
WB Whole Body
WB-R Reconstituted Whole Body

5.3 EFFECTS ASSESSMENT

This section describes the literature and site-specific studies used to characterize the effects of PCBs and other COCs to fish. Results of site-specific fish toxicity studies and literature effects levels were synthesized to develop tissue concentration ranges at which adverse developmental effects can be expected in the representative fish species in the Housatonic River.

Three sources of data were considered in the development of tissue effects thresholds for PCBs and TEQ (Figure 5.1-3). These include the following:

- General Literature—The literature review evaluated the range of PCB and TEQ concentrations observed to cause adverse effects to ecologically relevant endpoints in fish, such as reproduction and development.
- Phase I Site-Specific Studies—These studies investigated contaminant accumulation and effects in largemouth bass from the PSA and in their offspring. Adult fish collected from the PSA were spawned and the development of their offspring was monitored for survival, development, gross abnormalities, and biochemical alterations.
- Phase II Site-Specific Studies—These studies investigated the effects of maternal transfer of contaminants to developing oocytes. Contaminants extracted from largemouth bass tissue were injected into largemouth bass, medaka, and rainbow trout eggs and developmental effects were monitored.

Other site-specific studies were conducted, including: field assessment of largemouth bass reproduction, habitat, and population demographics; fish biomass and community assessments; and evaluation of deformities and disease in field-collected fish. However, these field studies were not used in the development of tissue effects thresholds for PCBs and TEQ because the studies did not specifically evaluate dose-response relationships. The results are described qualitatively, and therefore, were considered in the weight-of-evidence evaluation in the risk characterization.

Documented effects of PCBs on fish include mortality, growth-related effects, behavioral responses, biochemical alterations, and adverse reproductive effects. Of particular concern are the effects of dioxin-like PCB congeners that have the same toxic mechanism as 2,3,7,8-TCDD (Walker and Peterson 1991; Zabel et al. 1995).

5.3.1 Derivation of Literature Tissue Effects Metrics

A literature review was conducted to develop threshold effects concentrations for species that occur in the Housatonic River.

5.3.1.1 Total PCBs

A total of 39 scientific papers, 6 of which met the screening criteria for relevance to this ERA (Table 5.3-1), were reviewed to identify the range of tPCB concentrations associated with adverse effects on survival, growth, and reproductive success in fish.

Table 5.3-1

Criteria Used To Screen Available Studies for Determining Threshold Body Burdens

Criterion	Decision	
	Accept	Reject
PCBs and TCDD		
Body burden data	Reported (whole body preferred)	Not reported or reported fillet concentrations only
Endpoints	Population-level reproductive, development and survival effects	Chemical level (i.e., enzyme induction) effects
Exposure route	Studies mimicking maternal transfer, exposure of eggs, juveniles, and adults via diet, water and/or sediment	Intraperitoneal injection of adults
Statistics	Study included a control	No control
PCBs only		
PCB type	Aroclor 1254, Aroclor 1260, Clophen A50, tPCBs	Other PCB mixtures or individual congeners, or when co-occurring contaminants present

Reported tissue concentration LOAELs ranged from 1.53 mg/kg ww for increased mortality in lake trout (*Salvelinus namaycush*) sac fry (Berlin et al. 1981) to 125 mg/kg ww for fry mortality in brook trout (*Salvelinus fontinalis*) (Mauck et al. 1978) (Figure 5.3-1).

Based on the lines of evidence approach (EPA 1999), a threshold effects concentration of 61 mg/kg ww tPCBs for egg/sac-fry tissue was chosen. This value corresponds to the following:

- The average concentration for all effects reported in the studies used (61 mg/kg ww).
- The highest NOAEL reported in the studies used (71 mg/kg ww).
- The geometric mean of the paired NOAEL/LOAELs reported in the studies used (92.9 mg/kg ww).

Based on site-specific and literature information, a factor of 0.5 was applied to scale the selected egg/sac-fry tissue concentration to a whole body concentration for warmwater fish (Section F.3.5.2.2). As a result, a whole body tissue concentration of 31 mg/kg ww is recommended and is expected to be protective of reproductive and developmental endpoints for fish species in the PSA. Attachment F.4 provides additional details for the derivation of this threshold.

5.3.1.2 2,3,7,8-TCDD TEQ

A total of 19 papers, 11 of which met the screening criteria, were reviewed to identify the range of TEQ concentrations associated with adverse effects on survival, growth, and reproductive success in fish. Most studies provided egg concentrations, with only one of the papers reporting adult female whole body contaminant concentrations.

These data are summarized in Figure 5.3-2. There are two general groups of results—a lower group at approximately 50 ng/kg ww, and a higher group between 400 and 1,200 ng/kg ww. Basing a threshold concentration for adult whole body on the lower group (which corresponds to the lowest LOAEL observed, and the 10th percentile of egg concentrations at which effects were observed) may be overly conservative due to the known sensitivity of the trout species (e.g., lake trout) used in those studies. Conversely, basing the threshold concentration on the higher group may not be protective against adverse effects for all the species of concern in the Housatonic River.

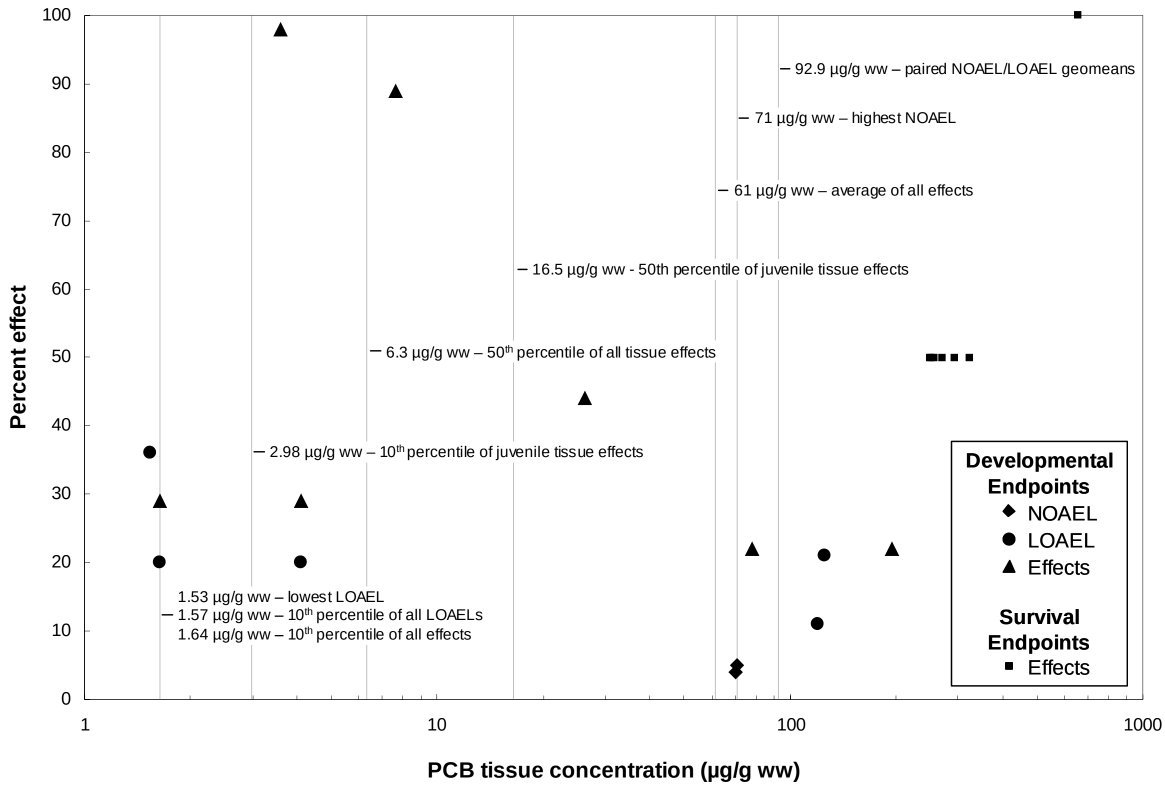


Figure 5.3-1 Literature-Based PCB Fish Tissue Effects Concentrations

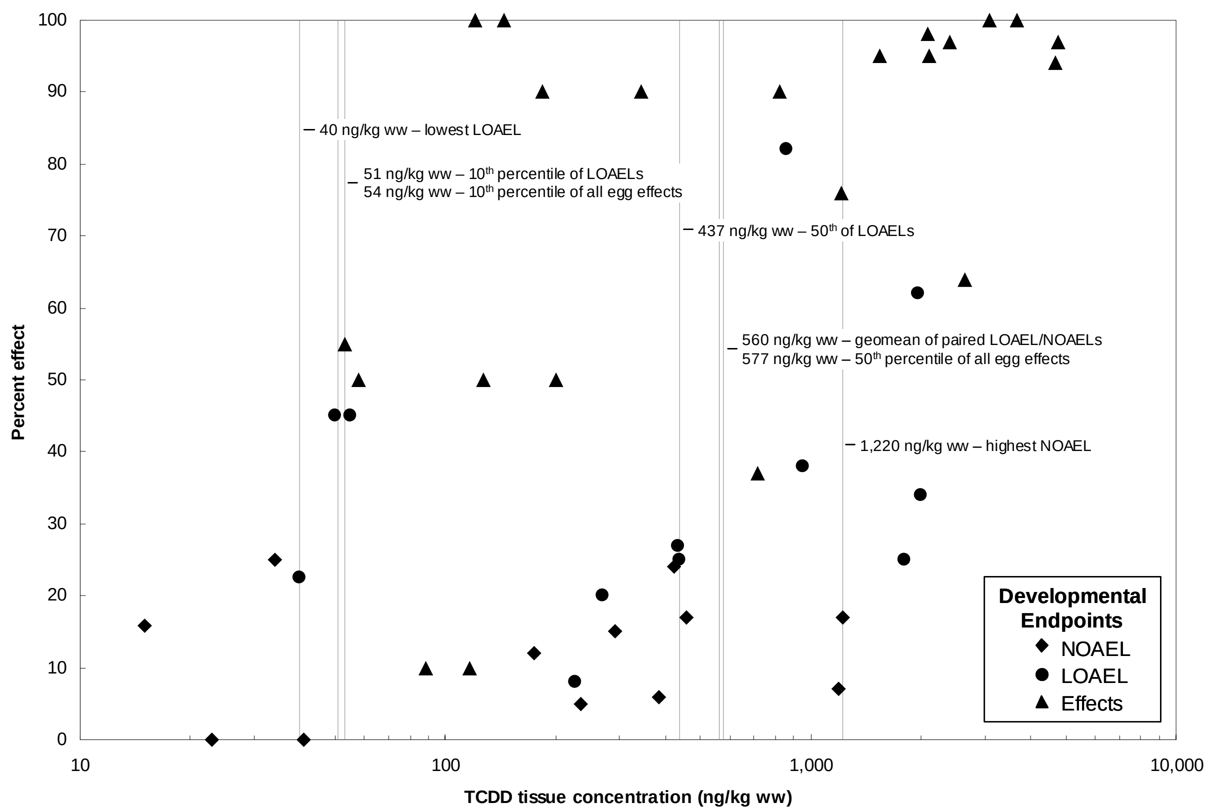


Figure 5.3-2 Literature-Based TCDD (TEQ) Fish Tissue Effects Concentrations

An egg effects threshold of 100 ng/kg ww TEQ, which is intermediate between the high and low groups discussed above, was selected for the Housatonic River PSA fish. This concentration represents the level at which early lifestage mortality starts to increase in several species, including warmwater fish species (see Attachment F.4).

Using the same conversion factor of 0.5 to scale from egg to whole body concentration (Section F.3.5.2.2), a whole body tissue concentration of 50 ng/kg ww TEQ was derived and is expected to be protective of sensitive reproductive and developmental endpoints in PSA fish.

5.3.2 Site-Specific Toxicity Studies

5.3.2.1 Housatonic River Fish Toxicity Study - Phase I

5.3.2.1.1 Methods

The Phase I study for the Housatonic River fish reproductive health assessment was conducted by the USGS Columbia Environmental Research Center (CERC) (Tillitt et al. 2003a). The study investigated contaminant-associated effects in fish collected from the PSA and spawned under controlled conditions.

Adult largemouth bass were collected from two Housatonic River locations within the PSA (Reach 5C and Reach 6), from Rising Pond (Reach 8) and from the Threemile Pond reference location. Adult fish (both pre- and post-spawning) were analyzed for a number of biochemical, histological, and morphological characteristics. Laboratory-reared offspring of adult largemouth bass from the PSA were monitored for survival, developmental delays and deformities, growth, and biochemical effects.

5.3.2.1.2 Results

Adult largemouth bass from the Housatonic River sites exhibited multiple sublethal effects at frequencies of occurrence that were statistically different from those observed in bass from the reference location.

Phase I Adult Largemouth Bass – Effects Observed

- Elevated EROD levels in livers.
- Thickened lobule wall in testes.
- Elevated occurrence of macrophage aggregates.
- Reduced growth in females.
- Reduced estrogen levels.

These responses are indicative of effects due to organic contaminants. Although not necessarily ecologically significant, they are indications of biological and chemical alterations that may lead to reproductive effects.

Effects were also observed in offspring. At 15 days post swim-up, deformities were observed in fry from all three reaches on the Housatonic River, while none were observed in fry from the reference location.

Phase I Largemouth Bass Offspring – Effects Observed

- Survival – Reduced survival from hatch to swim-up, or reduced survival post swim-up.
- Development – Developmental delays (increased days to swim-up).
- Growth – Reduced growth from swim-up to 15 days post swim-up.
- Deformities – Increase in eye deformities from hatch to swim-up; shortened opercula; tail deformities; external swim bladders.

In summary, the effects observed in the Phase I study were suggestive of PCB-related toxicosis. The Phase I fish toxicity study identified reproductive effects including reduced survival and growth, as well as developmental delays and deformities, in Housatonic River offspring. Specific abnormalities were observed in Housatonic River fish that were not observed in the fish from the Threemile Pond reference location. Although responses were not all consistent across all exposed reaches, the adults and offspring both exhibited a suite of symptoms that was consistent with PCB-related toxicity (Tillitt et al. 2003a,b)

5.3.2.1.3 Dose-Response and Effects Threshold Development

Not all Phase I endpoints exhibited a dose-response; however, for adult largemouth bass the following endpoints were found to exhibit a dose-response related to PCBs:

- The number of gonadal histopathologies (i.e., lobule wall thickness) increased with increasing adult whole-body tissue concentration.
- The number of macrophage aggregates increased with increasing adult whole-body tissue concentration.
- Post-spawning female length decreased with increasing tissue concentration (although no difference in post-spawning male weight between stations was observed).

Dose-response relationships were not observed for the following endpoints: adult post-spawn weight (female and male), adult male post-spawn length, post-spawn GSI (female and male), post-spawn LSI (female and male), testosterone levels, and vitellogenin levels.

For Phase I offspring, several dose-response relationships were observed:

- Survival at swim-up was significantly lower in offspring of fish from sites with the highest tissue burdens of tPCBs.
- Growth, measured as increased length and weight from hatch to swim-up, was significantly lower in offspring of fish from sites with the highest tPCB concentrations.
- The rate of eye deformities (at swim-up) was highest in the offspring of fish from the sites with the highest tPCB concentrations.
- Offspring of fish from sites with the highest tPCB concentrations had a higher rate of shortened operculae (at 15 days post swim-up).
- The number of days for development to swim-up were also highest in offspring of fish from the site with the second highest tPCB concentrations.

There were no clear dose-response relationships in the numbers of other individual pathologies.

Both tPCB and TEQ tissue effects thresholds were derived from the Phase I study results. The threshold derived from the Phase I study was based on tissue concentrations of 45 mg/kg ww tPCBs (or 38 ng/kg ww TEQ), representing the lowest tissue concentration measured in non-reference locations. The effect sizes of the Phase I endpoints used to derive this threshold were variable, although in the range of 10 to 30%, which is consistent with the definition of intermediate risk used in the risk characterization. Because effects were observed in the lowest exposure location on the Housatonic River, it is possible that adverse effects may also occur at lower tissue concentrations than the above threshold. However, these effects would likely occur at a rate lower than the level used to designate “intermediate” risk (i.e., 20%), or would be for biological endpoints that have uncertain relevance to local population endpoints (e.g., endocrine disruption responses; enzyme induction; macrophage aggregates.

5.3.2.2 Housatonic River Fish Toxicity Studies – Phase II

5.3.2.2.1 Methods

In Phase II of the fish toxicity studies conducted by USGS CERC (Tillitt et al. 2003b), organic contaminants present in largemouth bass from the Phase I studies were injected into cultured eggs of largemouth bass, medaka, and rainbow trout. This study provided a simulation of maternal transfer of PCB contamination to offspring.

The assessment of lethal and sublethal effects in fish was focused on the later (i.e., swim-up and post swim-up) stages of development. Percent survival was determined for each treatment group and compared to controls. Lengths and weights of largemouth bass and medaka were measured at the end of the experiment (i.e., 15 days post swim-up) and compared to controls. Each fish was examined for deformities at important life stages.

5.3.2.2.2 Results

The following effects were observed in offspring of largemouth bass, medaka, and rainbow trout exposed to extracts from the Housatonic River and to PCB-126 and TCDD standards.

Survival and Growth

Statistically significant reductions in survival were most evident in fish exposed to PCB and TCDD standards. Reduced survival was also observed in medaka exposed to Housatonic River extracts. Survival was not affected in largemouth bass and rainbow trout exposed to Housatonic River extracts. High mortality was observed in largemouth bass control fish between 3 and 15 days post swim-up (fish did not successfully transition to exogenous feeding); the post swim-up assessment for bass, therefore, was terminated. Largemouth bass and medaka length and weight (growth endpoints) were not affected by exposure to extracts or standards.

LD_{50}s were calculated for trials passing acceptance criteria for largemouth bass (swim-up), medaka (3 and 15 days post swim-up), and rainbow trout (600 DTU) for Reaches 5BC and 6, PCB-126, and/or TCDD. Overall, medaka at 15 days post swim-up exhibited the lowest LD_{50}s, relative to other species, for all extracts and standards, with the exception of TCDD. The overall results (i.e., order of magnitude difference in TEQ-based LD_{50}s between site extracts and standards) indicate that the Housatonic River extracts are more toxic than would be predicted on the basis of an additive model of dioxin-like toxicity alone.

Individual Deformities

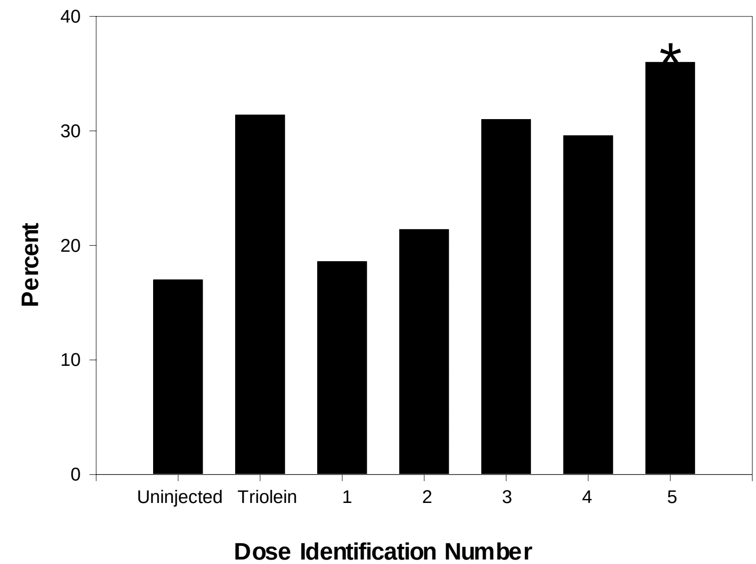
At certain stages of development, largemouth bass, medaka, and rainbow trout were examined for a variety of abnormalities. Several abnormalities exhibited an apparent dose-related or threshold response to high doses of Housatonic River extracts or standards, relative to control fish.

Overall, increased rates of deformities (craniofacial deformities, spinal deformities, swim bladder deformities, hemorrhage, pericardial edema, peritoneal edema, yolk sac edemas, and delayed development) were most evident in fish at swim-up and post swim-up following in ovo exposure to PCB and TCDD standards. Fish exposed in ovo to high doses of these standards exhibited a variety of gross pathologies that are characteristic of PCB and dioxin exposure. Fish exposed in ovo to high doses of Housatonic River extracts exhibited similar types of gross pathologies as the dioxin-like standards.

Some of the deformities observed in fish were only weakly related to tPCB or TEQ concentrations for one species/life stage/treatment combination. The lack of a dose-response in fish injected with Housatonic River extracts and/or PCB and TCDD standards and the occurrence of these deformities in fish injected with control and reference site extracts indicates that these abnormalities are not the most reliable markers of PCB exposure.

Total Abnormalities

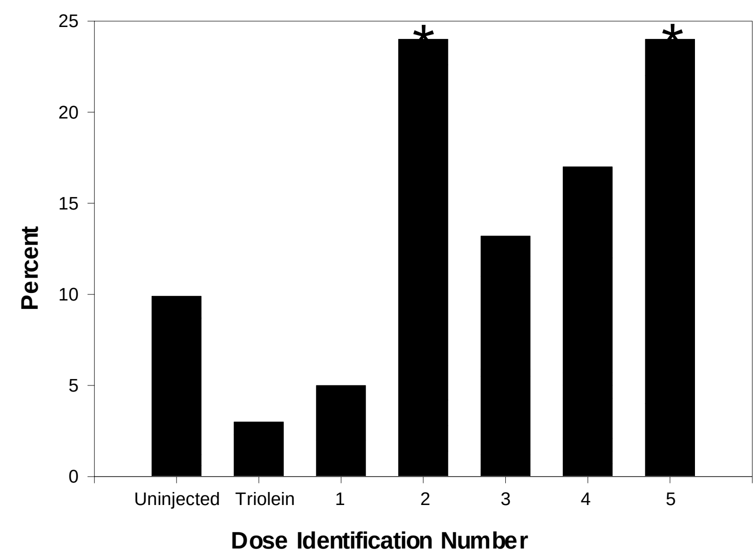
To provide an overall picture of the relationship between PCB exposure and the occurrence of abnormalities, the proportion of fish exhibiting one or more abnormalities was compared to control fish. Fish exposed to PCB and TCDD standards exhibited significantly higher percentages of abnormalities, relative to control fish. Similar, but more variable, dose-response relationships were observed in fish exposed to Housatonic River extracts. Figures 5.3-3 to 5.3-7 summarize the dose-response relationships for the PSA reaches.



Notes: Bar height indicates percentage of fish affected with one or more pathologies.
Asterisks indicate significant differences from negative controls (uninjected and triolein).
Doses are 1, 2, 19, 93, and 185 mg/kg ww tPCBs and 0.6, 1, 12, 59, and 118 ng/kg ww TEQ for Dose IDs 1-5, respectively.

Source: Adapted from Tillitt et al. 2003b.

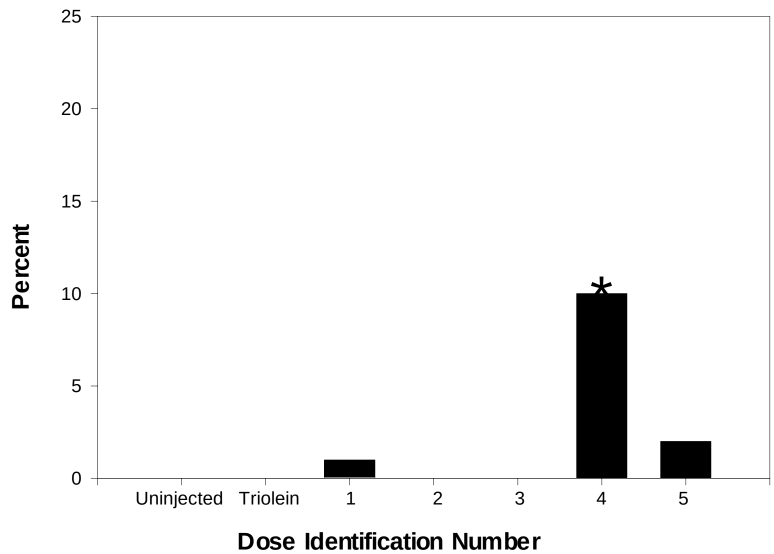
Figure 5.3-3 Effects of in Ovo Exposure to Increasing Doses of Reach 6 Extracts on Largemouth Bass at Swim-Up



Notes: Bar height indicates percentage of fish affected with one or more pathologies.
Asterisks indicate significant differences from negative controls (uninjected and triolein).
Doses are 2, 3, 31, 155, and 310 mg/kg ww tPCBs and 0.6, 1, 13, 64, and 128 ng/kg ww TEQ for Dose IDs 1-5, respectively.

Source: Adapted from Tillitt et al. 2003b.

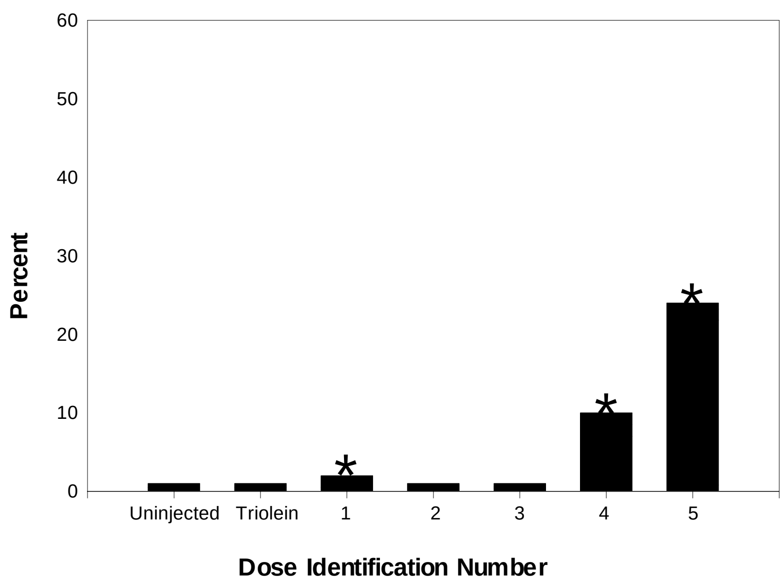
Figure 5.3-4 Effects of in Ovo Exposure to Increasing Doses of Reach 5BC Extracts on Medaka at 5d Post Swim-Up



Notes: Bar height indicates percentage of fish affected with one or more pathologies.
Asterisks indicate significant differences from negative controls (uninjected and triolein).
Doses are 1, 2, 19, 93, and 185 mg/kg ww tPCBs and 0.6, 1, 12, 59, and 118 ng/kg ww TEQ for Dose IDs 1-5, respectively.

Source: Adapted from Tillitt et al. 2003b.

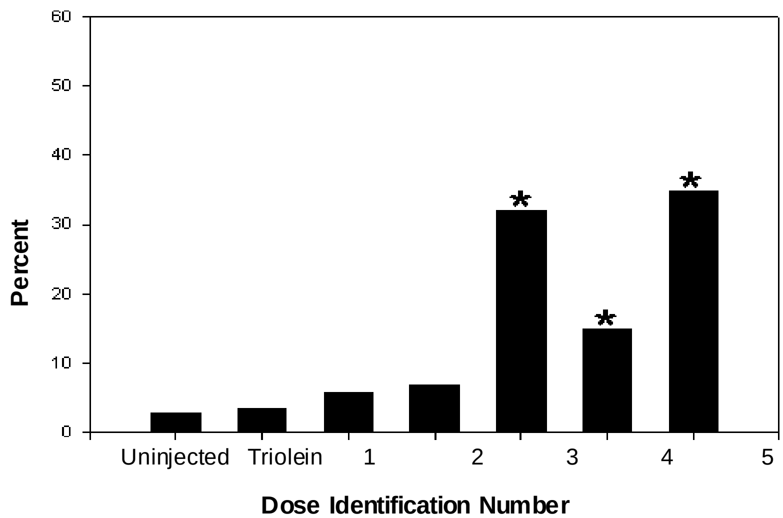
Figure 5.3-5 Effects of in Ovo Exposure to Increasing Doses of Reach 6 Extracts on Medaka at 5d Post Swim-Up



Notes: Bar height indicates percentage of fish affected with one or more pathologies.
Asterisks indicate significant differences from negative controls (uninjected and triolein).
Doses are 1.4, 2.8, 28, 70, and 139 mg/kg ww tPCBs and 0.6, 1, 11, 29, and 57 ng/kg ww TEQ for Dose IDs 1-5, respectively.

Source: Adapted from Tillitt et al. 2003b.

Figure 5.3-6 Effects of in Ovo Exposure to Increasing Doses of Reach 5BC Extracts on Rainbow Trout at 600 DTU



Notes: Bar height indicates percentage of fish affected with one or more pathologies.
Asterisks indicate significant differences from negative controls (uninjected and triolein).
Doses are 0.8, 1.7, 17, 42, and 83 mg/kg ww tPCBs and 0.5, 1, 11, 27, and 53 ng/kg ww TEQ for Dose IDs 1-5, respectively.

Source: Adapted from Tillitt et al. 2003b.

Figure 5.3-7 Effects of in Ovo Exposure to Increasing Doses of Reach 6 Extracts on Rainbow Trout at 600 DTU

Biochemical Effects

Cytochrome P450 induction was evaluated qualitatively in largemouth bass, medaka, and rainbow trout tissues using immunochemical histological techniques. Cytochrome P450 induction was observed in fish exposed to both standards and Housatonic River extracts. Rainbow trout was the most sensitive test species, exhibiting apparent dose-related increases in cytochrome P450 induction. Largemouth bass did not appear to exhibit a dose-related induction of cytochrome P450 following exposure to Housatonic River extracts.

5.3.2.2.3 Dose-Response and Effects Threshold Development

Although the Phase I study identified a suite of effects that were consistent with PCB-related toxicity, the Phase II study evaluated the cause-effect linkage more directly. The results of the Phase II study indicated that fish exposed to Housatonic River extracts exhibited decreased survival and increased abnormalities and biochemical alterations, in response to high doses of tPCBs and TEQ. The patterns of responses observed were not always consistent across species and treatments; however, gross pathologies observed were characteristic of PCB-related effects reported in the literature and corresponded with a number of the effects observed in the Phase I study.

ED_{50}s derived from the Phase II study results were used to develop thresholds for Housatonic River extracts. ED_{50} values were calculated using raw data from the Phase II studies (Attachment F.7), using methods described in Appendix F (Section F.3.5.2.1) and are presented in Table 5.3-2. For calculation of an effects threshold, calculated ED_{50} values were increased to the highest concentration tested if an acceptable dose-response relationship was not observed (Table 5.3-3). Criteria for an acceptable dose-response relationship were:

- The highest dose must be statistically different from the control (either the pooled control or triolein control, depending on whether the controls were significantly different).
- The largest magnitude of adverse responses must be observed in either of the highest two dose levels.

Because Tillitt et al (2003b) emphasize TEQ as the exposure measure, TEQ was converted to tPCB concentrations using linear regression (i.e., TEQ versus tPCB doses); regression equations used for the conversions are provided in Appendix F (Table F.3-9).

Rather than selecting an individual ED_{50} concentration as a threshold value, the entire distribution of ED_{50} values was considered. The procedures applied in the derivation of the egg-based maximum acceptable threshold concentration (MATC) for tissue are described in Appendix F (Section F.3.5.2.2). Based on these criteria, ED_{50} concentrations in eggs were calculated as 131 mg/kg ww tPCBs and 100 ng/kg ww TEQ.

As described in Section F.3.5.2.2, a value of 0.5 was selected to extrapolate from egg to whole body tissue concentration.

Table 5.3-2

Calculated ED_{50} Values (tPCBs and TEQ) for Largemouth Bass, Medaka, and Rainbow Trout Exposed in Ovo to Housatonic River Extracts and PCB-126 and 2,3,7,8-TCDD Standards

Endpoint[a]	Species	Life Stage	Extract	Concentration[b]	TEQ (ng/kg)
ED_{50}	Largemouth bass	Swim-up	Reach 6 (Trial 1)	136.58 µg tPCBs/g egg	87
			PCB-126 (Trial 2)	657.2 ng PCB-126/g egg	3,286
	Medaka	Swim-up	Reach 5BC (Trial 1)	82.49 µg tPCBs/g egg (uninjected) 34.11 µg tPCBs/g egg (triolein)	34 (uninjected) 14 (triolein)
			Reach 5BC (Trial 3)	43.78 µg tPCBs/g egg	18
			PCB-126 (Trial 2)	46.6 ng PCB-126/g egg (uninjected) 64.4 ng PCB-126/g egg (triolein)	233 (uninjected) 322 (triolein)
			PCB-126 (Trial 3)	44.8 ng PCB-126/g egg	224
			2,3,7,8-TCDD (Trial 3)	2,667 ng TCDD/g egg	2,667
		15 d post swim-up	Reach 5BC (Trial 1)	48.62 µg tPCBs/g egg	20
			Reach 5BC (Trial 3)	9.91 µg tPCBs/g egg (uninjected) 22.25 µg tPCBs/g egg (triolein)	4.0 (uninjected) 9.1 (triolein)
			PCB 126 (Trial 3)	31 ng PCB-126/g egg	155
	Rainbow trout	600 DTU	Reach 6 (Trial 1)	11.85 µg tPCBs/g egg	7.6
			Reach 5BC (Trial 4)	116.9 µg tPCBs/g egg	48
			PCB-126 (Trial 1)	24.2 ng PCB-126/g egg	121
			PCB-126 (Trial 2)	87.0 ng PCB-126/g egg	435
			2,3,7,8-TCDD (Trial 1)	294 pg TCDD/g egg	294
			2,3,7,8-TCDD (Trial 2)	152 pg TCDD/g egg	152

[a]ED_{50} endpoints were based on the combined effects observed per fish (pathology and mortality combined) for each species/life stage/treatment/dose combination.

[b]Total PCB concentrations were interpolated from non-standardized concentrations through linear regression. PCB-126 and 2,3,7,8-TCDD concentrations were converted using toxicity equivalent factors for fish provided in Van den Berg et al. (1998). All concentrations are wet weight.

Table 5.3-3

Summary of ED_{50} Values Used in Phase II Tissue Threshold Calculations

Species	TEQ (ng/kg ww)	tPCBs (mg/kg ww)	Reach	Stage	Trial
Largemouth bass	118	185.0	Reach 6	Swim-Up	Trial 1
Medaka	118	185.0	Reach 6	Swim-Up	Trial 2
Medaka	118	185.0	Reach 6	Post Swim-Up	Trial 2
Medaka	20[a]	48.6[a]	Reach 5BC	Post Swim-Up	Trial 1
Medaka	128	310.0	Reach 5BC	Post Swim-Up	Trial 2
Medaka	128	310.0	Reach 5BC	Post Swim-Up	Trial 3
Medaka	14[a]	34.1[a]	Reach 5BC	Swim-Up	Trial 1
Medaka	128	310.0	Reach 5BC	Swim-Up	Trial 2
Medaka	18[a]	43.8[a]	Reach 5BC	Swim-Up	Trial 3
Medaka	161	90.0	Reach 8	Post Swim-Up	Trial 2
Medaka	161	90.0	Reach 8	Post Swim-Up	Trial 3
Medaka	161	90.0	Reach 8	Post Swim-Up	Trial 4
Medaka	161	90.0	Reach 8	Swim-Up	Trial 3
Medaka	161	90.0	Reach 8	Swim-Up	Trial 4
Rainbow Trout	57	139.0	Reach 5BC	600 DTU	Trial 3
Rainbow Trout	48[a]	117.0[a]	Reach 5BC	600 DTU	Trial 4
Rainbow Trout	7.6[a]	11.9[a]	Reach 6	600 DTU	Trial 1
Rainbow Trout	53	83.0	Reach 6	600 DTU	Trial 2
Rainbow Trout	145	81.0	Reach 8	600 DTU	Trial 1
Average (All)	100.3	131.2			
Average (Sensitive Only[a])	21.5	51.1			

[a] Sensitive trials defined as those for which the ED_{50} was based on a clear dose-response within the range of concentrations tested. Trials with no dose-response within range tested were set equal to highest concentration tested.

Conversion of tPCB Egg Threshold to Whole Body Threshold

- The mean ED_{50} value for tPCB Phase II fish toxicity (largemouth bass, medaka, and rainbow trout) was 131 mg tPCBs/kg ww egg.
- The mean ED_{50} value for TEQ Phase II fish toxicity (largemouth bass, medaka, and rainbow trout) was 100 ng TEQ/kg ww egg.
- A review of egg to whole body conversion factors for PCBs and TEQ was conducted using site-specific and literature information, yielding an estimate of 0.5. This value was used to extrapolate tPCB egg concentrations to whole body concentrations.
- The threshold egg concentrations were converted to whole body adult tissue concentrations of 66 mg tPCBs/kg ww and 50 ng TEQ/kg ww.

5.3.2.2.4 Study Conclusions

A high degree of variability was observed in many of the parameters evaluated in the Phase II study. Despite this variability, however, an overall pattern of PCB-related toxicity was apparent. The types of abnormalities observed in fish exposed to Housatonic River extracts in the Phase II study corresponded with the abnormalities reported in the Phase I study, as well as with dioxin-like effects documented in the literature.

Because the 2,3,7,8-TCDD and PCB-126 standards are contaminants with high toxic potencies (i.e., high TEF values) relative to most individual congeners and congener mixtures, it is expected that the magnitude of effects observed with the standards would be greater than those observed at similar concentrations of the Housatonic River extracts. When ED_{50} concentrations were normalized using TEQ, however, the Housatonic River extracts were more toxic than the standards. The increased toxicity associated with the Housatonic River extracts could be attributed to synergistic toxicity of the PCB mixtures, as well as the effects of PCBs that are not incorporated into the TEQ model (Tillitt, personal communication 2003).

5.3.3 Derivation of Literature-Based Sediment Effects Metrics for PAHs

Exposure of fish to PAHs in sediment can result in reproductive, developmental, and carcinogenic effects.

Potential Effects of PAHs to Fish

- Reproductive and Developmental Effects – PAHs have been shown to be immunotoxic and to have adverse effects on reproduction (reduced egg fertility, increased fry mortality) and development, with egg and larval stages the most vulnerable.
- Carcinogenic Effects – A number of studies of bottom-dwelling fish, including tomcod, English sole, Pacific staghorn sculpin, rock sole, brown bullhead, and winter flounder indicate a link between sediment exposures and hepatic neoplasms.

The assessment of fish toxicity of PAHs is complicated by the fact that PAHs are readily metabolized by most aquatic animals, including teleost fish (Johnson 2000; Johnson et al. 2002). Because tissue concentrations are not a reliable predictor of adverse effects in fish, the relationship between exposure to contaminated sediment and effects can be used to derive an effects threshold (detailed in Appendix F; Attachment F.5).

Sediment Threshold for Total PAH

- Most relevant threshold (10 mg/kg PAH) – Based on brown bullhead effects observed over changing sediment PAH levels at an industrial site on the Black River, Ohio.

The 10 mg/kg threshold was considered the most appropriate value for application to the Housatonic ERA because of common environments (i.e., freshwater), species (i.e., brown bullhead), and endpoints of interest. In addition to the threshold for total PAHs, thresholds of 0.92, 0.68, and 0.64 mg/kg were identified for individual PAHs (phenanthrene, benzo(a)anthracene, and indeno(1,2,3-cd)pyrene, respectively); these values were based on the no observed effect concentrations (NOECs) from EPA (2000) (Attachment F.5).

5.3.4 Site-Specific Field Studies of Populations and Communities

5.3.4.1 EPA Fish Community Studies

A survey of fish biomass in the main stem and Woods Pond (Reaches 5 and 6) was conducted by EPA in fall 2001 to generate information for use in the modeling study and risk assessments

(Woodlot 2002). Biomass estimates were developed for each species for both prey-sized fish (fish <10 cm) and all other size classes pooled (fish >10 cm). For largemouth bass (only), separate biomass estimates were made for each age of 11 age classes (0+ to 10+ age classes). Also, capture frequency data were compiled in 10 cm size class increments (all species). Separate estimates were developed for each species/size class group in each of five reaches. Results of the EPA biomass study are presented in Woodlot (2002) and summarized in Table F.3-14. Chadwick (1993, 1994) also conducted an earlier biomass study that generally yielded lower biomass estimates (Table F.3-15). An ecological characterization of the Housatonic River PSA (Appendix A.1) was also conducted to describe the aquatic and ecological habitats throughout the study area. These results are in general agreement with those of the biomass study (above).

Based on these and other biological surveys, it is clear that the five representative fish species chosen for this assessment (i.e., largemouth bass, pumpkinseed, yellow perch, white sucker, brown bullhead) are found in suitable habitats of the PSA. There is also evidence that these species are self-sustaining; therefore, total reproductive failure or other obvious effects to the fish populations are not apparent. However, meaningful statistical assessment of PCB or TEQ relationship to fish community parameters was not feasible and no quantitative conclusions can be drawn regarding the health of the Housatonic River fish community based on these field surveys.

5.3.4.2 GE Largemouth Bass Community and Reproduction Study

A field study was conducted in the summer and early fall of 2000 and 2001 to assess largemouth bass habitat, community structure, and reproduction in the Housatonic River (R2 Resource Consultants Inc., 2002; Reiser et al. 2004). The objectives of the study were to:

- Determine if the largemouth bass population was self-sustaining.
- Determine if the largemouth bass population depended on tributary recruitment.
- Determine if the characteristics of the largemouth bass population were similar to those observed in other systems.

These objectives were broader than the measurement endpoints identified for the fish ERA. As such, the study was not designed to determine whether site contaminants have had adverse impacts on fish individuals or populations. Instead, the study was designed to evaluate whether major alterations of fish communities could be identified.

5.3.4.2.1 Habitat Surveys

In the spring of 2000, habitat surveys were conducted at 13 sites located in the main river channel, backwater areas, three major branches, and six tributaries. In 2001, surveys were conducted at 15 sites located in the backwater areas. The aquatic habitat assessment indicated that suitable largemouth bass spawning habitat is abundant in the PSA, particularly within Woods Pond and shallow backwater areas of the PSA (R2 Resource Consultants Inc. 2002). The results indicate that in the main channel largemouth bass habitat (including backwaters) is of good quality and that the tributaries generally have poor habitat, with the exception of Moorewood and Yokun Brooks.

5.3.4.2.2 Nest Surveys

Nest surveys were conducted to determine if largemouth bass in the Housatonic River were successfully reproducing and to assess the condition of young-of-year (YOY) bass. Metrics included reproductive activity, relative abundance of YOY, and YOY growth rates. Quantitative estimates of egg-to-fry or nesting success could not be made for individual nests because on subsequent survey dates individual nests frequently could not be relocated (R2 Resource Consultants Inc. 2002). Accordingly, measures of reproductive success were generally qualitative, and relied on interpretation of nest and brood observations, growth rate data, and catch-per-unit-effort (CPUE) of YOY bass. Interpretation of the nest condition data vis-à-vis overall reproductive success is difficult due to the lack of monitoring over time at most nests. Therefore, it is not evident whether the low percentage of nests with fry is attributable to natural biological factors (e.g., spawning not yet occurred; eggs consumed by predators; offspring moved off the nest), or if productivity of nests was reduced due to contaminants or other factors.

The estimated mean growth rates for YOY bass are less than those reported for other systems in northeastern North America (e.g., Micajah Pond, MA; Dryden Lake, NY. Quabbin Reservoir, MA). The reduced growth of YOY, however, does not translate to older age classes because overall growth rates of PSA fish were comparable to other systems.

CPUE estimates of YOY largemouth bass were computed for seven main channel, six transition, and seven backwater habitats within the PSA. Although the CPUE estimates from the Housatonic River are within the ranges observed at other sites, the estimates fall toward the lower end of the ranges, and quantitative comparisons are difficult due to the confounding effect of habitat differences among locations. As designed, the GE study could identify large (e.g., order-of-magnitude) reductions in YOY abundance, but smaller response sizes (i.e., 20% to 50% reductions) cannot be evaluated precisely given the variability and uncertainty in the estimates of CPUE.

The observance of nests and fry, combined with the unsuitable habitat of some of the tributaries, and the barriers to fish passage from less-contaminated areas (i.e., below Woods Pond Dam), confirm that bass are successfully reproducing in the PSA. The data provide only an indication that reproduction is occurring, however, not an evaluation of the actual reproductive rate. No egg or fish tissue was collected to determine contaminant concentrations in the study. The apparent self-sustaining nature of this population may be assisted by the low mortality rate of the adults (due to lack of harvesting pressure). CPUE data for YOY bass indicate that recruitment of bass from PSA habitats occurs and is likely sufficient to prevent large population-level responses under the current stressor regime.

5.3.4.2.3 Population Structure and Adult Condition

In 2000, 133 largemouth bass were collected from the main channel (120 fish), backwater areas (3 fish), and East and West Branches (10 fish). In 2001, 239 largemouth bass were collected from the main channel and backwater areas located between the confluence of the East and West Branches and Woods Pond. Generally, largemouth bass were found at all sites sampled, except for selected tributaries.

Adult growth and condition were evaluated using Bayesian analyses and comparisons of parameters to North American largemouth bass populations. The PSA adult growth estimates fell within the Bayesian 80^{th} and 90^{th} percentile credibility intervals from all populations of largemouth bass. The mean length-at-age of Housatonic River largemouth bass is also similar to that found in other northern systems (Scott and Crossman 1973, 1998). Relative weights for Housatonic River bass, assessed over several size ranges, all indicated fish condition within the range for all populations (Reiser et al. 2004).

The population demographics data from the 2000 and 2001 sampling events were reviewed to assess whether any obvious population-level impacts were evident. The metrics evaluated by R2 Resource Consultants Inc. (2002) do not allow for a definite conclusion regarding whether any population-level responses have occurred, but can be used as indicators of whether responses are sufficiently large to suggest major adverse population conditions, such as year-class failures or low recruitment (Anderson and Neumann 1996). The age analyses indicated that the bass population consists primarily of large older fish. In both sampling years, a low proportion of fish were collected in the intermediate size class range of 18 to 28 cm length, corresponding to age 2+ fish (R2 Resource Consultants Inc. 2002). The largemouth bass population displays characteristics similar to that of an unexploited population of this species. It appears that only low levels of recruitment of the young fish are currently required to sustain the local populations of large adult (age 4+) bass. The high proportion of older fish is at least partly a function of the lack of fishing pressure on the population (R2 Resource Consultants Inc. 2002), due to the lack of both a catch-and-consume fishery, and the relative lack of piscivorous mammals in the PSA.

Overall, the GE study data do not indicate large perturbations in the population structure of the largemouth bass in the Housatonic River. However, the ability to detect small population-level responses is weak due to the large interannual variations in population structure data. Whether recruitment could limit the largemouth bass populations under a condition of introduced fishing pressure was not evaluated in the GE study, nor was the quantitative effect of contaminants on recruitment levels directly evaluated.

5.3.5 Malformations in Field-Collected Fish

Incidence of deformities, erosions, lesions, and tumors (DELTs) in freshwater fish has been used as an indicator of environmental quality (Baumann et al. 1987; Smith et al. 1994).

5.3.5.1 Goldfish Lesions

During the EPA fish tissue collections, lesions were observed on bottom-feeding cyprinids, specifically goldfish (*Carassius auratus*), located in and around Woods Pond. Due to the unknown etiology of these lesions, a combined virology and histopathology assessment was conducted to identify the growths and to investigate potential causes.

A total of 10 goldfish samples were shipped to Dr. John Coll at the U. S. Fish and Wildlife Service, Fish Health Center, Lamar, PA on October 9, 2003. These samples consisted of tissues for viral and histological assays. No viral replication/activity was observed in any of the virology assays (Coll, personal communication 2004). Therefore, although some lesions were visually similar to those of carp pox, no evidence of carp pox virus was observed. Following virology, samples for histological examination were forwarded to Dr. Vicki Blazer (U.S. Geological Survey, National Fish Health Research Laboratory, Leetown, WV). The histological examination of the fish samples indicated multiple abnormalities (Blazer, personal communication 2004), including abnormalities in gills, livers, spleens, skin, and eyes.

Most of the lesions appeared to be contaminant-related, and the various abnormalities (e.g., abnormal gill filaments; altered foci and hepatic cell neoplasms in liver; and large numbers and size of macrophage aggregates in spleen and liver) all suggest a response to contaminants, most likely an organic contaminant such as PAHs or PCBs (Blazer, personal communication 2004). Although the histopathology results cannot be used to make a definitive conclusion regarding the specific organic contaminant causing the responses, the combined virology, histopathology, and contaminant exposure analyses suggest that PCBs, PAHs, or a combination of both PCBs and PAHs, are responsible for the majority of the abnormalities observed.

5.3.5.2 Assessment of DELTs During EPA Fish Sampling

During field collection of fish for tissue analysis in 1998 by EPA and USFWS, gross morphological and external DELT information was recorded. Detailed or comprehensive inspections of each fish were not conducted because this was not a primary objective of the surveys. Incidence rates for major abnormalities in most species were low. The most common abnormalities were liver abnormalities in largemouth bass and tumors in Woods Pond bottom fish. Parasitism was observed in many yellow perch, consisting of glob eye that was not observed in any other species. The incidence of this response was relatively consistent across reaches, at approximately 20% frequency in the PSA.

5.4 RISK CHARACTERIZATION

The risk characterization for fish integrates the exposure assessment (Section 5.2) and effects assessment (Section 5.3) to evaluate the assessment endpoint of survival, growth, and reproduction of fish in the Housatonic River.

The following three lines of evidence were used to develop the risk characterization in the Housatonic River fish risk assessment (Figure 5.1-4):

- **Field surveys –** Two field surveys were conducted in the study area. EPA evaluated fish abundance/biomass and conducted an ecological characterization of the site. GE independently evaluated largemouth bass reproduction, community, and habitat data.

- **Comparison of field-measured exposures to effects levels or thresholds –** For these endpoints, the risk characterization integrated exposure and effects by relating the two terms quantitatively. This method consisted of a comparison of tissue chemistry (PCBs and TEQ) to tissue effects thresholds, and sediment chemistry (PAHs) to literature-based sediment effects thresholds. Hazard quotients were calculated for PCBs by comparing observed tissue concentration data to site-specific MATCs. Probabilities of exceeding various effects threshold levels were also calculated.

- **Site-specific toxicity study results –** These endpoints (e.g., Phase I and Phase II toxicity tests) directly evaluated biological responses to COCs.

These lines of evidence allowed for a robust weight-of evidence assessment of the potential for risk using the approach of Menzie et al. (1996).

To characterize risk in a probabilistic manner in the PSA (i.e., similar to the wildlife receptors) the following decision rules were applied:

- Low Risk: Probability (moderate response) < 20%
- High Risk: Probability (large response) > 50%
- Intermediate Risk: All other outcomes

The probabilities in the above framework are based on distributions of tissue tPCB and TEQ chemistry data from discrete measurements in the PSA, or some distributions of sediment PAH data from the PSA.

5.4.1 Selection of MATCs

For COCs in fish tissue, maximum acceptable threshold concentrations (MATCs) were derived to represent the transition between low and intermediate risk, for calculation of HQs, and for extrapolation to areas downstream of the PSA. The effect sizes used to determine Phase I tissue thresholds were variable, but were generally in the 10 to 30% range (Section F.3.3). The effect sizes for the Phase II study are also variable because some of the EC_{50} values used to calculate tissue thresholds were unbounded values, and therefore, may represent effect sizes lower than 50% (Section F.3.4). These effect sizes are considered appropriate to mark the transition from low to intermediate risk, and were used to derive a site-specific MATC.

The thresholds from the site-specific toxicity studies (Phase I and Phase II) were averaged to obtain fish tissue MATCs of 55 mg/kg tPCBs and 44 ng/kg TEQ. Literature values were not included in the calculations due to higher uncertainty; however, the literature threshold values (31 mg/kg ww tPCBs; 50 ng TEQ/kg ww) were similar to the MATCs.

In addition to the MATCs, "high risk" thresholds for tPCBs and TEQ were derived by considering larger effect sizes (i.e., EC_{50} or greater) and considering the literature studies (see Section F.3.5.3 for rationale). In summary, adult fish with tissue concentrations greater than 300 ng/kg ww TEQ or 150 mg/kg ww tPCBs are at risk of large adverse responses that may include mortality of some adults and large reproductive and developmental effects.

5.4.2 Field Surveys

Based on the biomass study, ecological characterization work, and other biological surveys, it is clear that the five representative fish species chosen for this assessment are found in suitable habitats of the PSA. There is also evidence that these species are self-sustaining; therefore, total reproductive failure or other obvious effects to the fish populations are not apparent.

Because of the variability in habitat across the PSA, small-scale variability in PCB concentrations, and the small gradient in PCB tissue concentrations across the PSA, it is difficult to discriminate habitat influences from potential contaminant influences.

The GE largemouth bass study confirms that there is a reproducing community of bass in the Housatonic River. Although the full extent of tributary recruitment remains unknown, the majority of bass found in the PSA result from reproduction occurring within the PSA and associated backwaters. Reproduction is confirmed to occur in the study area, although quantitative reproductive success (i.e., egg-to-fry production) was not evaluated in the study. Growth rates are generally similar to other systems, in spite of slightly lower YOY growth rates and a lower asymptotic fish size. The current population (biomass) is dominated by older, larger fish in good condition. Although the population appears stable, the lack of typical harvesting pressure (i.e., consumption advisory, and few piscivorous mammals) places very little demand on recruitment in order to sustain overall numbers of older adults. The principal investigators (Reiser et al. 2004) speculate that the inability to detect PCB-induced population-level effects may be due to the overall resiliency of a high-fecundity species. This hypothesis was not evaluated, however, because the quantitative effects of PCBs on fry survival relative to other sources of early life stage mortality were not assessed.

Overall, the field studies did not evaluate whether PCBs are causing or may cause effects to the population structure and/or viability, nor were the studies suited to the detection of lower-level population responses.

5.4.3 Comparison of Estimated Exposures to Derived Effects Metrics

5.4.3.1 Total PCBs in PSA Fish

Summary of Tissue Effects Thresholds for tPCBs

- 31 mg/kg ww tPCBs – based on literature review (warmwater species).
- 45 mg/kg ww tPCBs – based on Phase I study (largemouth bass).
- 66 mg/kg ww tPCBs – based on Phase II study (warmwater species and rainbow trout).
- 55 mg/kg ww tPCBs – based on average of Phase I and Phase II thresholds (MATC for risk characterization of PSA fish).
- 150 mg/kg ww tPCBs – high risk threshold based on mortality of adults and large reproductive and developmental effects.
- 14 mg/kg ww tPCBs – based on Phase I/Phase II studies, and literature information on trout sensitivity (MATC for coldwater species downstream of the PSA).

Figure 5.4-1 depicts HQs for PSA fish tissue concentrations compared to the average of the site-specific (Phase I and Phase II) fish effects thresholds derived for the PSA (i.e., MATC of 55 mg/kg tPCBs). For most species and tissue types, the HQs span a range that includes 1.0. For some fish (e.g., composite samples of small perch and sunfish), the HQs are consistently below 1.0. Due to the correlation between fish age and PCB concentration, however, these species exhibit HQs > 1 in large adult specimens (i.e., WB-R samples).

Figures 5.4-2 through 5.4-6 show the cumulative distribution plots for observed whole body tPCB concentrations for each species and reach within the PSA. The circles represent effects thresholds representative of the transition between low and intermediate risk (filled circles) and intermediate and high risk (open circles). Table 5.4-1 displays the 95th percentile and exceedance probabilities for the threshold effects concentrations by species and PSA reach for observed fish tissue concentrations.

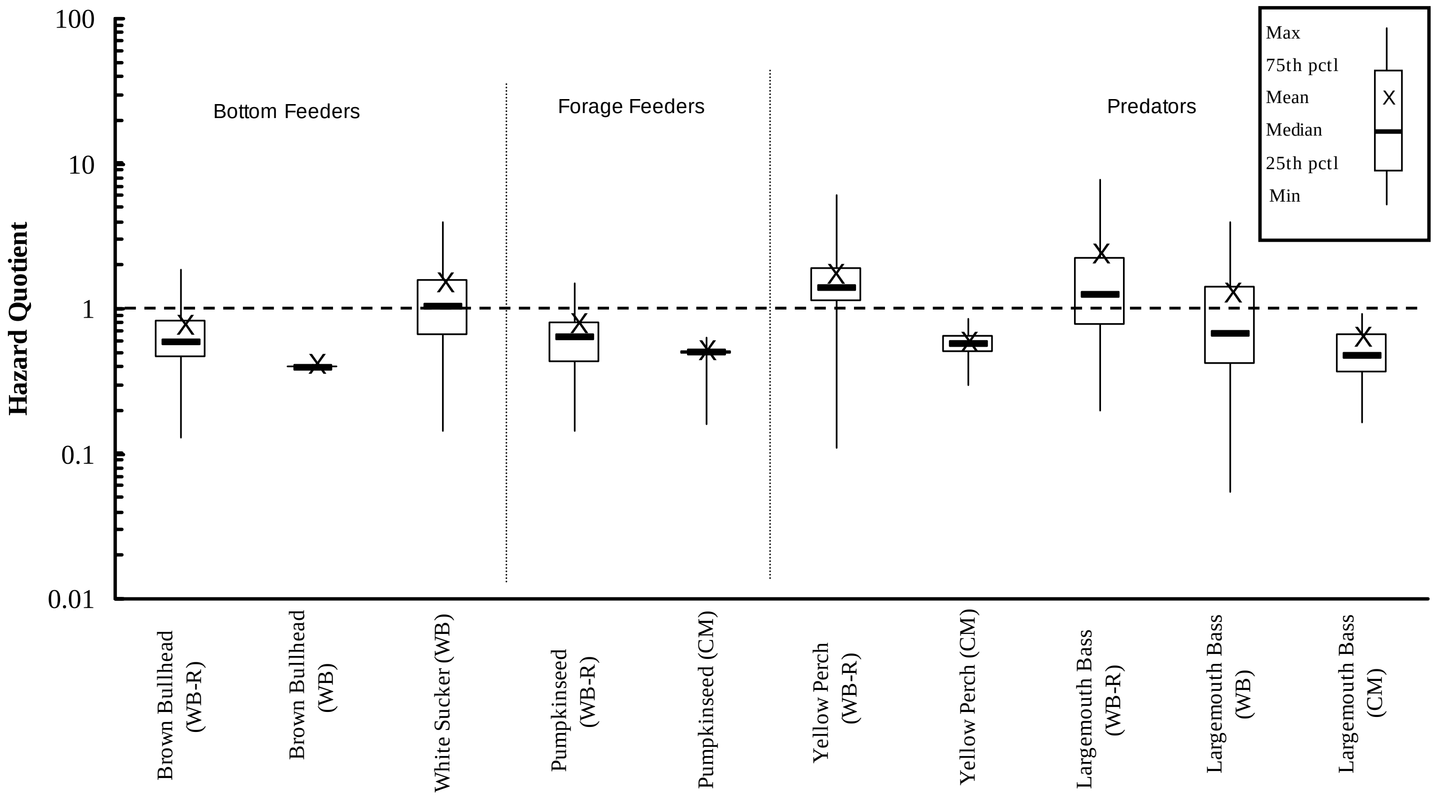
Within the PSA, Table 5.4-1 indicates a moderate to high probability of exceedance for most representative species and reaches. Figures 5.4-2 through 5.4-6 indicate that the risks for most fish species/reach combinations are intermediate, following the decision rules specified in Section 5.4.

5.4.3.2 2,3,7,8-TCDD TEQ in PSA Fish

Summary of Effects Thresholds for TEQ

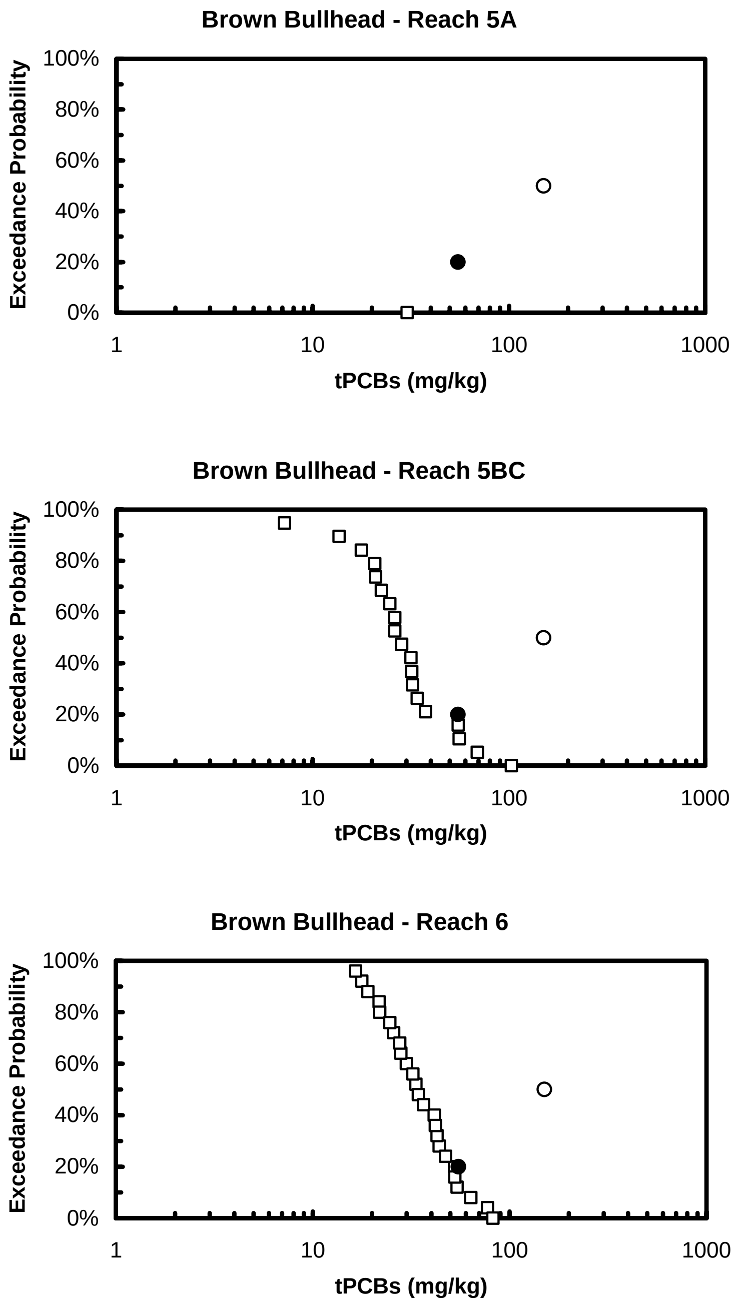
- 50 ng/kg ww TEQ – based on literature review (relevant to PSA fish species)
- 38 ng/kg ww TEQ – based on Phase I study (largemouth bass)
- 50 ng/kg ww TEQ – based on Phase II study (warmwater species and rainbow trout)
- 44 ng/kg ww TEQ – based on average of Phase I and Phase II thresholds (MATC for risk characterization of PSA fish)
- 300 ng/kg ww TEQ – high risk threshold based on mortality of adults and large reproductive and developmental effects.

Figure 5.4-7 depicts HQs for PSA fish tissue concentrations compared to the average site-specific effects threshold (i.e., 44 ng/kg ww TEQ). All 75th percentile-based HQs exceed 1, but mean and median HQs for adult fish are below 3 for all species.



WB = whole body individuals; WB-R = whole body reconstituted; CM = multiple whole body fish composites

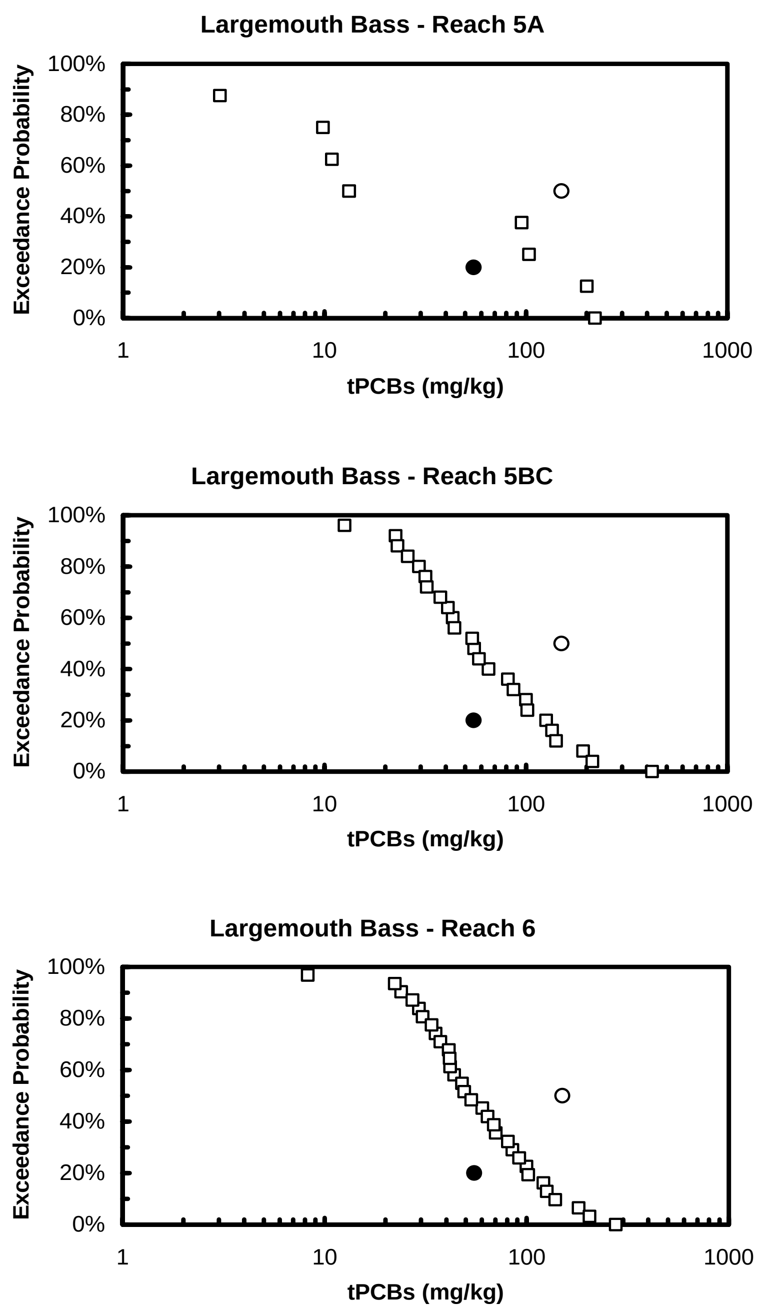
Figure 5.4-1 Hazard Quotients for tPCBs in PSA Fish Based on Comparison to the Mean Site-Specific Fish Toxicity Threshold (55 mg/kg ww tPCBs) (Logarithmic Scale)



Key: ● Threshold for low to intermediate effect.
○ Threshold for intermediate to high effect.

Note: Concentrations are wet weight.

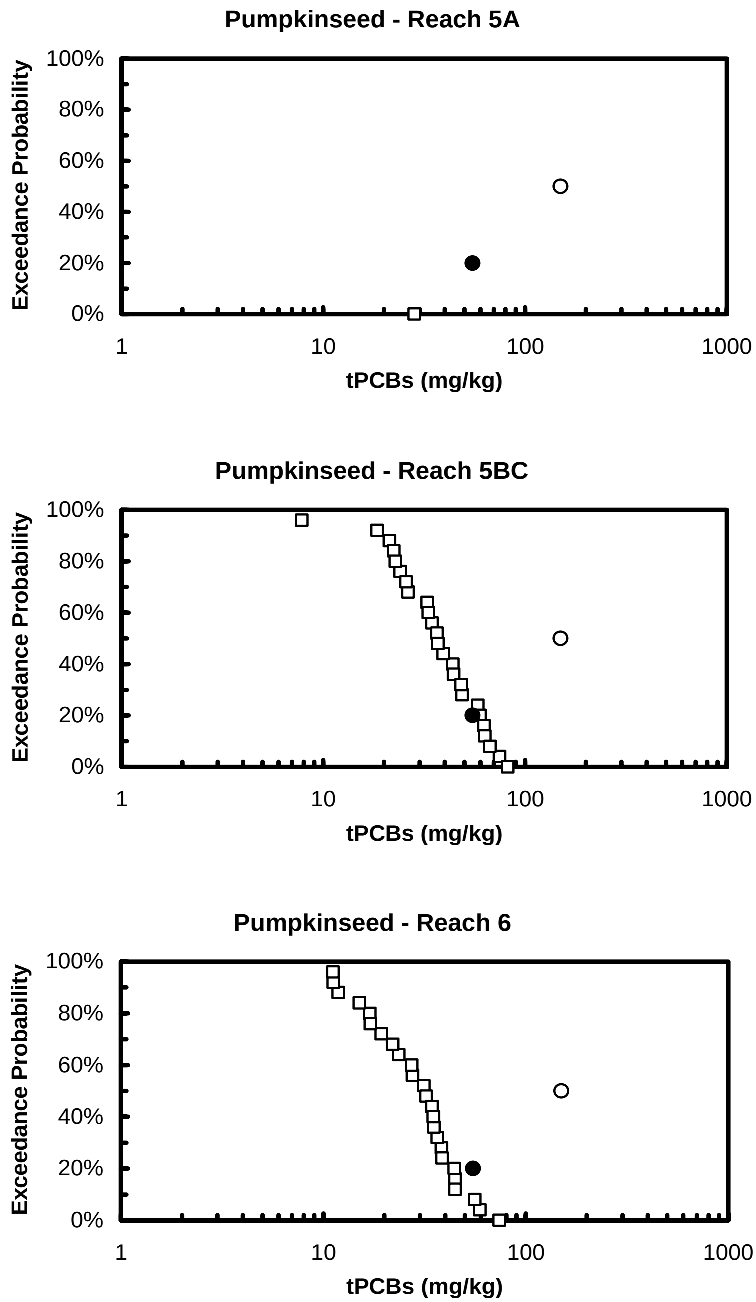
Figure 5.4-2 Complementary Cumulative Distribution Plot for tPCB Concentrations in Whole Body Tissue Compared to Effects Concentrations – Brown Bullhead



Key: ● Threshold for low to intermediate effect.
○ Threshold for intermediate to high effect.

Note: Concentrations are wet weight.

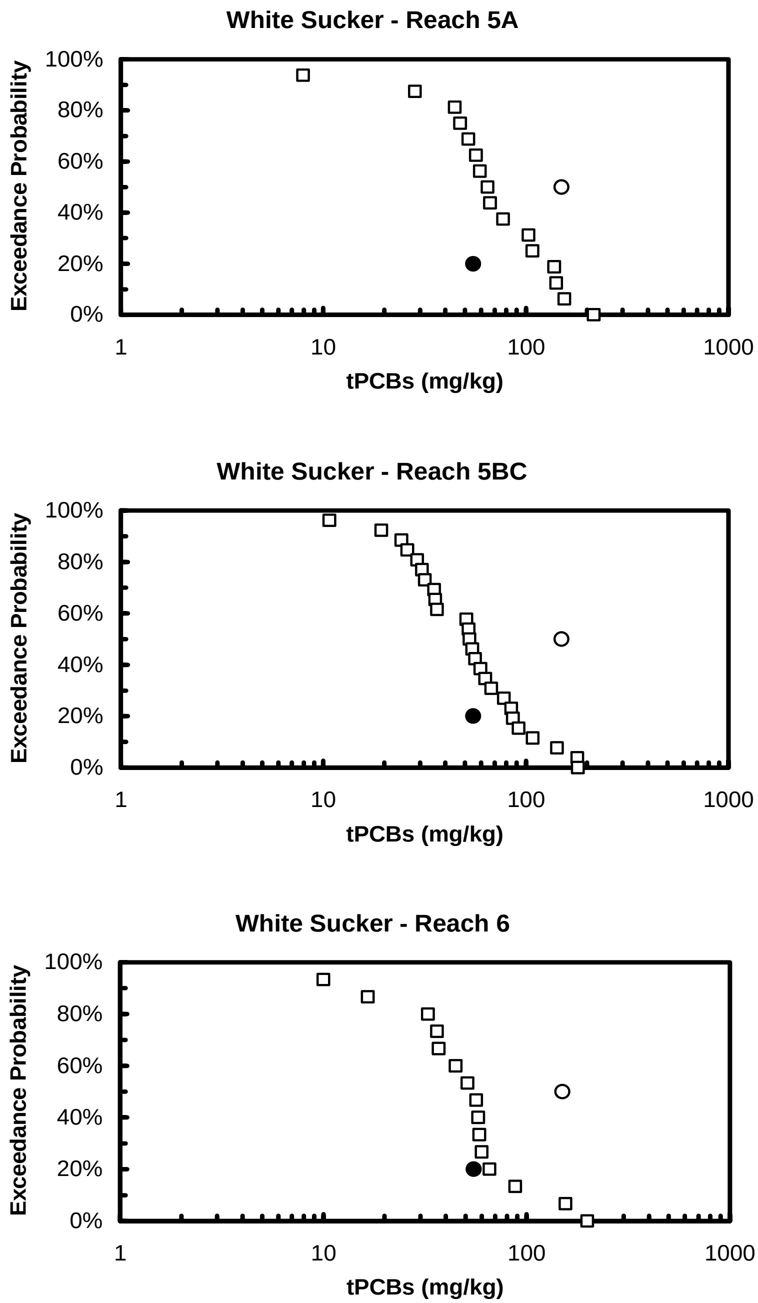
Figure 5.4-3 Complementary Cumulative Distribution Plot for tPCB Concentrations in Whole Body Tissue Compared to Effects Concentrations – Largemouth Bass



Key: ● Threshold for low to intermediate effect.
○ Threshold for intermediate to high effect.

Note: Concentrations are wet weight.

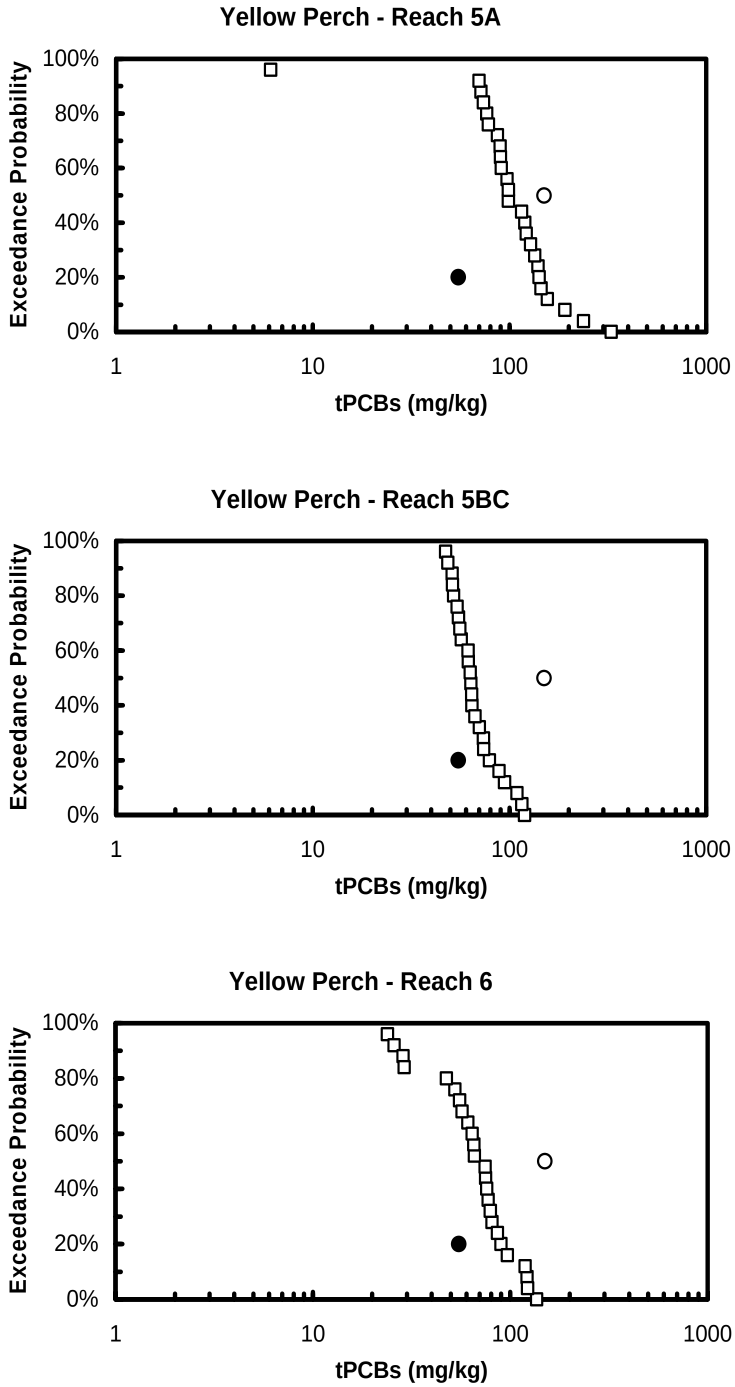
Figure 5.4-4 Complementary Cumulative Distribution Plot for tPCB Concentrations in Whole Body Tissue Compared to Effects Concentrations – Pumpkinseed



Key: ● Threshold for low to intermediate effect.
○ Threshold for intermediate to high effect.

Note: Concentrations are wet weight.

Figure 5.4-5 Complementary Cumulative Distribution Plot for tPCB Concentrations in Whole Body Tissue Compared to Effects Concentrations – White Sucker



Key: ● Threshold for low to intermediate effect.
○ Threshold for intermediate to high effect.
Note: Concentrations are wet weight.

Figure 5.4-6 Complementary Cumulative Distribution Plot for tPCB Concentrations in Whole Body Tissue Compared to Effects Concentrations – Yellow Perch

Table 5.4-1

Probabilities of Exceedances in the PSA for tPCBs

Species	Reach 5A			Reach 5BC			Reach 6		
	95th Percentile (mg/kg)	Probability of Exceeding Thresholds		95th Percentile (mg/kg)	Probability of Exceeding Thresholds		95th Percentile (mg/kg)	Probability of Exceeding Thresholds	
		55 mg/kg ww	150 mg/kg ww		55 mg/kg	150 mg/kg		55 mg/kg	150 mg/kg
BB	NA[a]	NA	NA	73	21%	0%	75	12%	0%
LB	213	50%	25%	209	52%	12%	193	48%	10%
PS	NA	NA	NA	73	28%	0%	59	12%	0%
WS	170	69%	13%	170	46%	8%	169	53%	13%
YP	228	96%	16%	114	76%	0%	123	76%	0%

[a]NA = not available

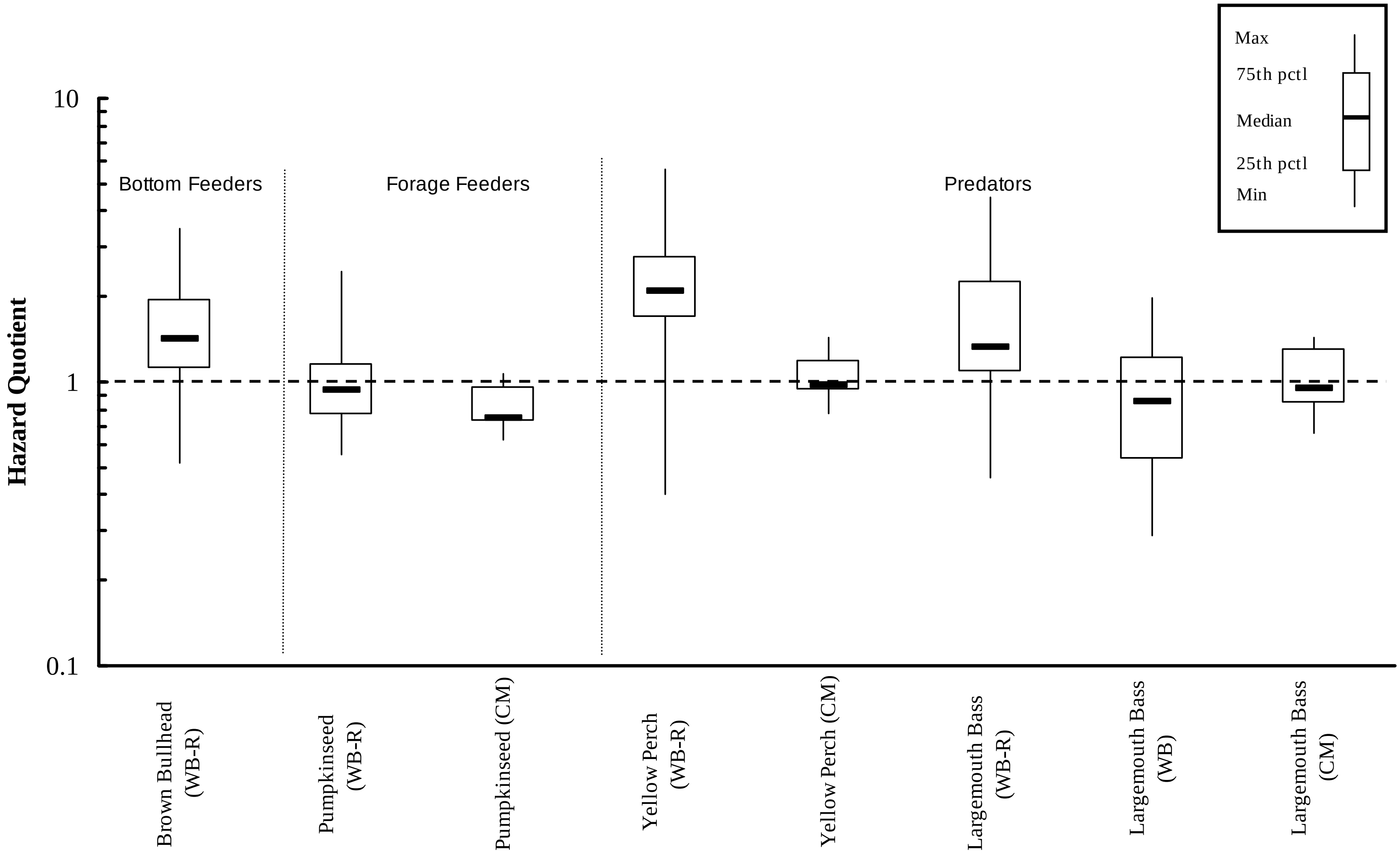
BB Brown Bullhead

LB Largemouth Bass

PS Pumpkinseed

WS White Sucker

YP Yellow Perch



WB = whole individuals; WB-R = whole body reconstituted; CM = multiple whole fish composites

Figure 5.4-7 Hazard Quotients for TEQ for Fish in Primary Study Area (PSA) Based on Comparison to the Average Site-Specific Tissue Effects Threshold (44 ng/kg ww TEQ) (Logarithmic Scale)

Figure 5.4-8 shows the cumulative distribution plots for observed whole body TEQ tissue concentrations for each species (using DL substitution for ND congeners). The circles represent effects thresholds representative of the transition between low and intermediate risk (filled circles) and intermediate and high risk (open circles). Based on the criteria for determining risk stated at the beginning of Section 5.4, Figure 5.4-8 indicates that risks are intermediate for all PSA species. Table 5.4-2 displays the exceedance probabilities for the effects concentrations, as well as the 95th percentile for exposure for each species, substituting concentrations equal to zero or to the DL, respectively, for non-detected congeners. With the exception of pumpkinseed, the majority of individual fish exceeded the MATC, irrespective of the choice of substitution method.

5.4.3.3 PAHs

Figure 5.4-9 presents the cumulative probability plots of individual PAHs and total PAH concentrations versus the sediment concentration effects thresholds. For all five PAH concentration measures, the effects thresholds pass very close to the 20th percentile of the observed distribution. Therefore, the PAH risks straddle the “low” and “intermediate” risk designations. Because the PAH effects thresholds are expected to be conservative, for reasons described in Section F.3.9.3., the overall risk of local population-level responses due to PAHs is expected to be low. Some individual level responses are possible, however, and the histopathology analyses could not definitively exclude PAHs as a potential contributor to the occurrence of the lesions in Woods Pond goldfish.

5.4.4 Site-Specific Toxicity Studies

Results and interpretation of the site-specific toxicity studies are summarized in Sections 5.3.2 and 5.3.3 and detailed in Appendix F. These studies indicate that PCBs are acting upon early life stages of fish, and causing various reproductive and developmental responses.

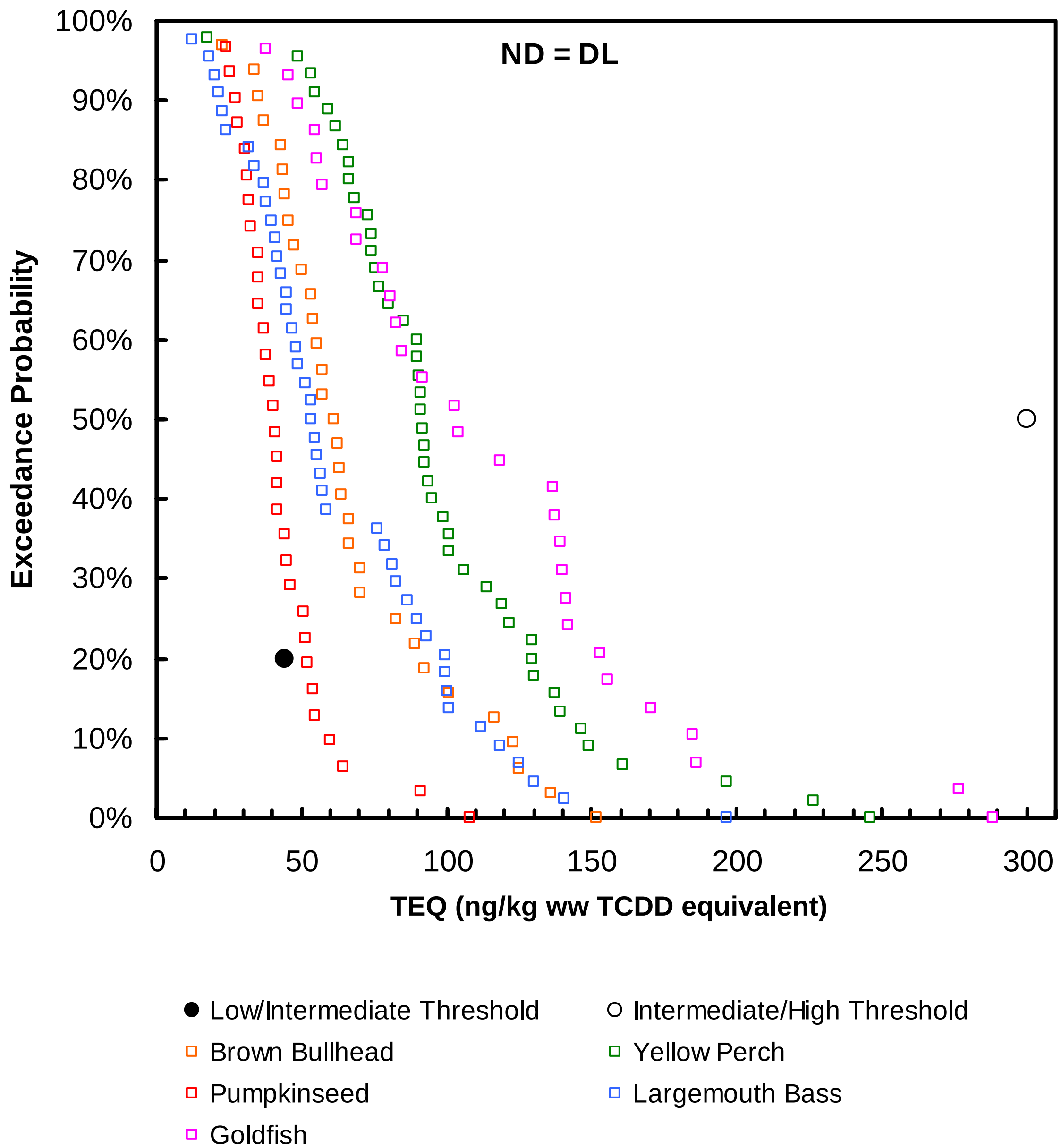


Figure 5.4-8 Complementary Cumulative Distribution Plot for TEQ Concentrations in Whole Body Tissue Compared to Effects Concentrations for All Species Using DL Substitution

Table 5.4-2

Probabilities of Exceedances for TEQ

Non-Detect Method	Species	95th Percentile (ng/kg)	Probability of Exceeding Thresholds	
			44 ng/kg ww	300 ng/kg ww
0 Substitution	BB	125	66%	0%
	LB	117	57%	0%
	PS	58.9	16%	0%
	YP	151	87%	0%
DL Substitution	BB	130	81%	0%
	LB	126	68%	0%
	PS	66	39%	0%
	YP	163	98%	0%

BB Brown Bullhead
LB Largemouth Bass
PS Pumpkinseed
YP Yellow Perch

0 substitution = Non-detected concentrations were substituted with zeroes

DL substitution = Non-detected concentrations were substituted with method detection limit

Fish Effects Thresholds:

- 44 ng/kg ww TEQ—Low/intermediate risk threshold (MATC), based on site-specific toxicity studies.
- 300 ng/kg ww TEQ—Intermediate/high risk threshold, based on site-specific toxicity studies and literature review.

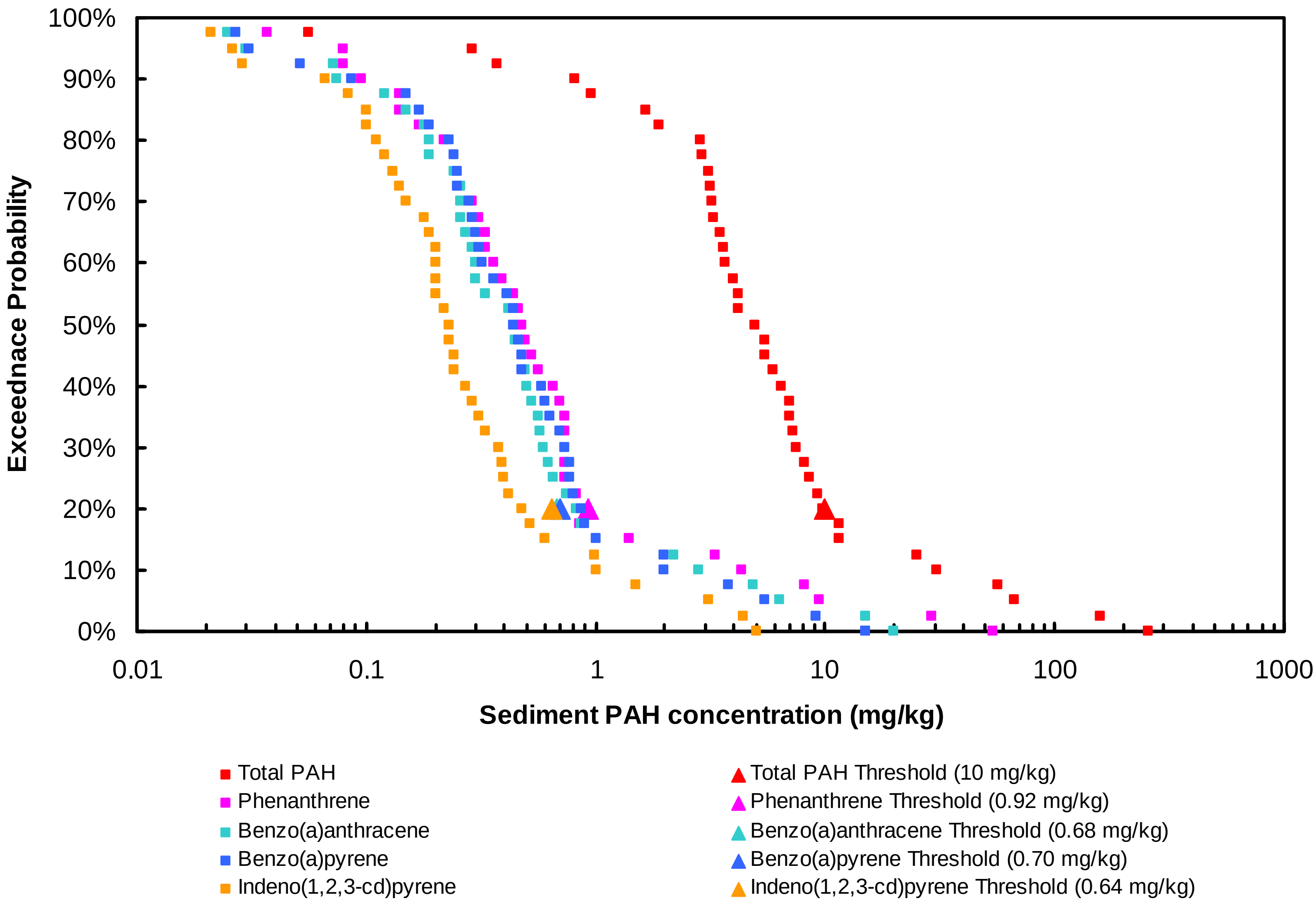


Figure 5.4-9 Sediment PAH Concentrations Relative to Thresholds Representing Intermediate Risk to Fish

5.4.5 Weight-of-Evidence Analysis

A weight-of-evidence evaluation was conducted for the multiple measurement endpoints in the fish ERA to determine whether significant risk is posed to fish from tPCBs. The attributes considered in the weight of evidence are described in Section 2, and the rationale for weighting the measurement endpoints are provided in Appendix F. A summary of the derived weightings for each attribute is provided in Table 5.4-3. The weighting values, evidence of risk, and magnitude of responses were combined in a matrix format and are presented in Table 5.4-4.

A graphical method was used for displaying concurrence among measurement endpoints (Table 5.4-5). This table illustrates that the majority of endpoints indicate, with a moderately high degree of confidence, that there are intermediate magnitude risks to fish in the PSA.

5.4.6 Downstream Extrapolation

5.4.6.1 Risks to Warmwater Fish Downstream of the PSA

As was done for the PSA, downstream risks to warmwater fish were evaluated based on concentrations of tPCBs in fish tissue. The MATC of 55 mg/kg ww tPCBs in tissue was also applied to areas downstream of Woods Pond using the available warmwater fish (e.g., bass, perch, sunfish) tissue data.

Results are provided in Appendix F (Figure F.4-11). The intermediate risks observed in the PSA decline to low risks in Reaches 7 through 9, in the section of the river between Woods Pond and the Massachusetts/Connecticut state line. Table 5.2-3 indicates that maximum individual fish PCB concentrations exceed the MATC for some species (i.e., largemouth bass, yellow perch); however, less than 20% of samples exceeded the MATC and only a single yellow perch sample exceeded the high risk threshold. Low risks were predicted for the Connecticut portion of the river, where tissue concentrations decline further.

Table 5.4-3

Weighting of Measurement Endpoints for Fish Weight-of-Evidence Evaluation – Summary

Measurement Endpoints:	T: Site-Specific Toxicity		C: Fish Tissue Chemistry			F: Field Studies		
	T-1	**T-2**	**C-1**	**C-2**	**C-3**	**F-1**	**F-2**	**F-3**
I. Relationship Between Measurement and Assessment Endpoints								
1. Degree of Association	Mod/High	Mod/High	Mod	Mod/High	Mod/High	Low/Mod	Mod	Low/Mod
2. Stressor/Response	Mod	Mod	Low/Mod	Mod	Mod	Low	Low	Low
3. Utility of Measure	Mod/High	Mod/High	Mod	Mod/High	Mod/High	Low/Mod	Low/Mod	Low
II. Data Quality								
4. Data Quality	High	High	High	Mod/High	Mod/High	High	High	Mod
III. Study Design								
5. Site Specificity	High	High	Low/Mod	High	High	High	High	High
6. Sensitivity	Low/Mod	Mod/High	Low/Mod	Mod	Mod	Low	Low	Low
7. Spatial Representativeness	Mod	Mod	Mod/High	Mod	Mod	Mod/High	Mod/High	Mod/High
8. Temporal Representativeness	Mod/High	Mod/High	Mod/High	Mod/High	Mod/High	Mod/High	Mod/High	Low/Mod
9. Quantitative Measure	High	High	Mod	Mod	Mod	Mod	Low/Mod	Low/Mod
10. Standard Method	High	High	Mod	Mod	Mod	Mod	Mod	Low/Mod
Overall Endpoint Value	**Mod/High**	**Mod/High**	**Mod**	**Mod/High**	**Mod/High**	**Mod**	**Mod**	**Low/Mod**

T: Site-Specific Toxicity. T-1 – Reproductive success in Phase I site-specific toxicity tests, relative to reference condition; T-2 – Reproductive success in Phase II site-specific toxicity tests, using dose-response analysis

C: Fish Tissue Chemistry. C-1 – Observed fish tissue concentrations relative to literature toxicity threshold; C-2 – Observed fish tissue concentrations relative to Phase I study toxicity threshold; C-3 – Observed fish tissue concentrations relative to Phase II study toxicity threshold

F: Field Community and Reproduction Studies. F-1 – EPA and GE community studies; F-2 – Largemouth bass reproduction study; F-3 – Largemouth bass demographics

Table 5.4-4

Evidence of Harm and Magnitude of Effects for Measurement Endpoints Related to Maintenance of a Healthy Fish Community

Measurement Endpoints	Weighting Value	Evidence of Harm	Magnitude of Harm (High, Intermediate, or Low)
C. Chemistry Endpoints			
C-1. Concentration of PCBs in tissues of representative species in relation to threshold concentrations reported (in literature) to be harmful to fish.	Mod	Yes	Intermediate
C-2. Concentration of PCBs in tissues of representative species in relation to threshold concentrations reported (in Phase I study; Tillitt et al. 2003a) to be harmful to fish	Mod/High	Yes	Intermediate
C-3. Concentration of PCBs in tissues of representative species in relation to threshold concentrations reported (in Phase II study; Tillitt et al. 2003b) to be harmful to fish.	Mod/High	Yes	Intermediate
T. Toxicity Endpoints			
T-1. Indications of reproductive and developmental impairment of largemouth bass spawned from Housatonic River adults in Phase I study (Tillitt et al. 2003a).	Mod/High	Yes	Intermediate
T-2. Indications of reproductive and developmental impairment of bass, medaka, and rainbow trout fry reared from eggs injected with Housatonic River extracts in Phase II study (Tillitt et al. 2003b).	Mod/High	Yes	Intermediate

Table 5.4-4

Evidence of Harm and Magnitude of Effects for Measurement Endpoints Related to Maintenance of a Healthy Fish Community (Continued)

Measurement Endpoints	Weighting Value	Evidence of Harm	Magnitude of Harm (High, Intermediate, or Low)
F. Field Study Endpoints			
F-1. Abundance and biomass of fish species observed in EPA fish collections, and GE fish community assessment.	Mod	No	--
F-2. Reproduction and nest condition metrics from GE reproduction study (R2 Resource Consultants 2002) including evidence of largemouth bass reproduction, nest conditions, YOY abundance and growth, and adult growth rate.	Mod	No	--
F-3. Largemouth bass population demographics including age-structure analysis and adult condition.	Low/Mod	No	--

Table 5.4-5

Risk Analysis Summary for Fish Exposed to tPCBs and TEQ in the Housatonic River PSA

Assessment Endpoint:	Survival, growth, and reproduction of fish

Harm/Magnitude	Weighting Factors (increasing confidence of weight)				
	Low	**Low/Moderate**	**Moderate**	**Moderate/High**	**High**
Yes/High					
Yes/Intermediate			C-1	T-1, T-2, C-2, C-3	
Yes/Low					
Undetermined					
No		F-3	F-1, F-2		

C: Fish Tissue Chemistry

C-1. Concentration of PCBs in tissues of representative species in relation to threshold concentrations reported (in literature) to be harmful to fish.

C-2. Concentration of PCBs in tissues of representative species in relation to threshold concentrations reported (in Phase I study; Tillitt et al. 2003a) to be harmful to fish.

C-3. Concentration of PCBs in tissues of representative species in relation to threshold concentrations reported (in Phase II study; Tillitt et al. 2003b) to be harmful to fish.

T: Site-Specific Toxicity

T-1. Indications of reproductive and developmental impairment of largemouth bass spawned from Housatonic River adults in Phase I study (Tillitt et al. 2003a).

T-2. Indications of reproductive and developmental impairment of bass, medaka, and rainbow trout fry reared from eggs injected with Housatonic River extracts in Phase II study (Tillitt et al. 2003b).

F: Field Community and Reproduction Studies

F-1 – Abundance and biomass of fish species observed in EPA fish collections, and GE fish community assessment.

F-2 – Reproduction and nest condition metrics from GE reproduction study (R2 Resource Consultants 2002) including evidence of largemouth bass reproduction, nest conditions, YOY abundance and growth, and adult growth rate.

F-3 – Largemouth bass population demographics including age-structure analysis and adult condition.

5.4.6.2 Risks to Coldwater Fish (Trout) Downstream of the PSA

Trout were evaluated separately from PSA fish species because of apparent differences in the sensitivity of trout to PCBs as indicated by the results of the site-specific fish toxicity studies (Tillitt et al. 2003b) and the generally higher PCB concentrations in trout due to their increased lipid content. Based on a comparison with LD_{50} values reported in the literature, Tillitt et al. (2003b) determined that the Fish Lake strain of rainbow trout used in the Phase II study was approximately three times less sensitive to the effects of TCDD and PCBs than other strains tested in the literature. In addition, other salmonid species (e.g., brown trout) may have different sensitivity to PCBs relative to trout. The PSA effects threshold of 55 mg/kg ww tPCBs, therefore, was divided by a factor of 4 to account for potential increased sensitivity of downstream coldwater species (i.e., coldwater MATC of 14 mg/kg tPCBs whole body, wet weight). The application of a factor of 4 is somewhat uncertain because the sensitivity of the trout inhabiting the downstream reaches of the Housatonic River (particularly brown trout) is unknown.

Because of the more limited database for trout, a number of extrapolations were necessary to convert available warmwater fish data and/or trout fillet data to estimated whole body concentrations for trout. These extrapolations are summarized in Appendix F (Section F.4.6.2).

Results are provided in Appendix F (Figure F.4-12). In general, some potential risk to trout from PCBs was found in Reach 7 downstream of Woods Pond Dam. These risks were marginal, and are uncertain due to the incertitude about the sensitivity differences for various trout species. Potential risk to trout was not evaluated downstream of Reach 12 due to the lack of suitable trout habitat.

5.4.7 Extrapolation to Other Species

Five fish species—largemouth bass, yellow perch, brown bullhead, white sucker, and pumpkinseed—were selected as the representative species for the ERA. The fish species were selected to include representatives of the principal trophic levels and exposure routes for fish in the PSA. The risk estimates for other warmwater species in the PSA are expected to be similar to the representative species. There are few salmonids in the PSA. Because there is evidence in the literature that salmonid species may have a higher sensitivity to the effects of PCBs and other dioxin-like COPCs (Walker et al. 1994; Zabel et al. 1995), the risk of COCs to the occasional salmonid occurring within the PSA may be greater than for warmwater species.

5.4.8 Sources of Uncertainty

Some of the major sources of uncertainty associated with the assessment of risks of tPCBs and TEQ to fish are briefly summarized below. A more complete list is presented in Appendix F.

- Potential for seasonal variation in fish tissue concentrations.
- Extrapolation of whole-body tissue concentrations from fillet data.
- Determination of injected doses in Phase II toxicity study based on assumed relationships among whole-body, ovary, and egg concentrations.
- Lack of synopticity in Phase I toxicity study.
- Development of effects thresholds from the literature.
- Potential confounding effects of other contaminants in the Phase I study.
- Extrapolation of egg thresholds to whole body burdens in the Phase II study.
- Field survey studies capable of detecting only very large responses in the local population.

5.4.9 Risk Assessment Conclusions

Two of the three lines of evidence (site-specific toxicity, fish tissue concentrations compared to MATCs) suggest intermediate risk to fish in the Housatonic River. However, the field surveys suggest that PCBs and/or other COCs are not causing obvious effects in fish populations. Therefore, the overall risk conclusion for fish is low/intermediate risk.

The lines of evidence collectively indicate that although widespread and obvious effects (e.g., acute mortality to adults or total reproductive failure) are not currently occurring, there is evidence of biological impairment during sensitive life stages. Biological indicators such as histopathology data, disease and deformity data, and induction of reproductive/endocrine biomarkers all support the conclusion that the health of individual fish has been impaired. The effect of this impairment on local fish population size, recruitment, and/or resilience to natural or anthropogenic stressors is unknown.

Field studies were inconclusive with respect to evaluating more subtle potential alterations to fish community health (e.g., reduced or delayed hatching success and reduced fry survival and growth). Moreover, the field surveys evaluated only the current status of environmental stressors and could not evaluate the potential population-level impacts under a modified condition (e.g., introduction of fishing pressure following removal of the consumption advisory).

Overall, evaluation of the fish assessment endpoint suggests that it is likely that risk to fish in the Housatonic River exists from both tPCBs and TEQ. The confidence in the numerical effects thresholds that support this conclusion is moderate. Strength in the conclusions was derived from the concordance in predictions of risk from multiple measurement endpoints; however, there is some uncertainty associated with several of the endpoints (particularly the field surveys).

5.5 REFERENCES

Anderson, R.O. and R.M. Neumann. 1996. Length, weight, and associated structural indices. In *Fisheries Techniques,* Second Edition. B.R. Murphy and D.W. Willis, Editors. American Fisheries Society, Bethesda, MD. pp. 447-481.

Baumann, P.C., W.D. Smith, and W.K. Parland. 1987. Tumor frequencies and contaminant concentrations in brown bullheads from an industrialized river and a recreational lake. *Trans. Am. Fish. Soc.* 116-79-86.

Berlin, W.H., R.J. Hesselberg, and M.J. Mac. 1981. Growth and mortality of fry of Lake Michigan lake trout during chronic exposure to PCBs and DDE. Tech. Pap. U.S. Fish Wildl. Ser. 105:11-22.

Blazer, V. 2004. Personal communication. Electronic mail from Dr. Vicki Blazer, U.S. Fish and Wildlife Service to Scott Campbell, Weston Solutions, Inc. containing Housatonic River goldfish lesion report and figures. August 2, 2004. WESTON Document Number 02-0496.

Chadwick (Chadwick & Associates). 1993. *Fisheries Investigation of the Housatonic River, Massachusetts.* Prepared for General Electric Company, Pittsfield, MA.

Chadwick (Chadwick & Associates). 1994. *Aquatic Ecology Assessment of the Housatonic River, Massachusetts.* Prepared for General Electric Company, Pittsfield, MA.

Coll, J. 2004. Personal communication. Memorandum from John Coll, U.S. Fish and Wildlife Service, Northeast Fishery Center, Lamar, PA to Scott Campbell (Weston Solutions, Inc.) regarding viral and histological assays of Housatonic River fish. January 21, 2004

EPA (U.S. Environmental Protection Agency). 1999. Development of a tissue trigger level from bioaccumulated tributyltin in marine benthic organisms: West Waterway, Harbor Island Superfund Site, Seattle, WA. U.S. Environmental Protection Agency, Region 10, Seattle, WA.

EPA (U.S. Environmental Protection Agency). 2000. Health of bullhead in an urban fishery after remedial dredging. U.S. Environmental Protection Agency, Great Lakes National Program Office. Chicago, IL. URL: http://www.epa.gov/grtlakes/sediments/Bullhead/report.html.

Field, L.J. 1996. *Use of Tissue Residue Data in Exposure and Effects Assessments for Aquatic Organisms*. National Oceanic and Atmospheric Administration, Hazardous Materials Response and Assessment Division, Coastal Resources Coordination Branch, Seattle, WA. Presented at the National Sediment Bioaccumulation Conference, September 11-13, 1996, Bethesda, MD. pp. 2-21—2-24. Website location: http://www.epa.gov/waterscience/cs/field.pdf

Johnson, L. 2000. *An analysis in support of sediment quality thresholds for polycyclic aromatic hydrocarbons (PAHs) to protect estuarine fish*. Environmental Conservation Division, Northwest Fisheries Science Center, NOAA, NMFS, Seattle WA. July 24, 2000.

Johnson, L, T. Collier, and J.E. Stein. 2002. An analysis in support of sediment quality thresholds for polycyclic aromatic hydrocarbons (PAHs) to protect estuarine fish. *Aquatic Conservation: Marine and Freshwater Ecosystems* 12:518-538.

Mauck, W.L., P.M. Mehrle, and F.L. Mayer. 1978. Effects of the polychlorinated biphenyl Aroclor® 1254 on growth, survival, and bone development in brook trout (*Salvelinus fontinalis*) *J. Fish. Res. Board Can.* 35:1084-1088.

Menzie, C., M.H. Henning, J. Cura, K. Finkelstein, J. Gentile, J. Maughan, D. Mitchell, S. Petron, B. Potocki, S. Svirsky, and P. Tyler. 1996. Special report of the Massachusetts Weight-of-Evidence Workgroup: A weight-of-evidence approach for evaluating ecological risks. *Human and Ecological Risk Assessment* 2:277-304.

R2 (R2 Resource Consultants Inc.). 2002. *Evaluation of Largemouth Bass Habitat, Population Structure, and Reproduction in the Upper Housatonic River, Massachusetts*. Report prepared for General Electric Company.

Reiser, D.W., E.S. Greenberg, T.E. Helser, M. Branton, and K.D. Jenkins. 2004. In situ reproduction, abundance, and growth of young-of-year and adult largemouth bass in a population exposed to polychlorinated biphenyls. *Environmental Toxicology and Chemistry*. 23:1762-1773.

Scott, W.B., and E.J. Crossman. 1973. *Freshwater Fishes of Canada*. Bulletin 184. Fisheries Research Board of Canada.

Scott, W.B. and E.J. Crossman. 1998. *Freshwater Fishes of Canada*. Fisheries Research Board of Canada. Galt House Publications, Ltd. I-xx + 1-966.

Smith, S.B., M.A. Blouin, and M.J. Mac. 1994. Ecological comparisons of Lake Erie tributaries with elevated incidence of fish tumors. *J. Great Lakes Res*. 20:701-716.

Tillitt, D. 2003. Personal communication (conference call with EVS Environmental Consultants, USGS study team members, Weston Solutions, Inc. and U.S. Environmental Protection Agency regarding the Phase II studies). USGS, Columbia Environmental Research Center, Columbia, MO. June 11, 2003.

Tillitt, D, D. Papoulias, and D. Buckler. 2003a. *Fish Reproductive Health Assessment in PCB Contaminated Regions of the Housatonic River, Massachusetts, USA: Investigations of Causal Linkages Between PCBs and Fish Health.* Prepared for U.S. Fish and Wildlife Service, Concord, NH, and U.S. Environmental Protection Agency, Boston, MA.

Tillitt, D, D. Papoulias, and D. Buckler. 2003b. *Fish Reproductive Health Assessment in PCB Contaminated Regions of the Housatonic River, Massachusetts, USA: Investigations of Causal Linkages Between PCBs and Fish Health.* Interim Report of Phase II Studies. Prepared for U.S. Fish and Wildlife Service, Concord, NH, and U.S. Environmental Protection Agency, Boston, MA.

Van den Berg, M., et al. 1998. Toxic equivalency factors (TEFs) for PCBs, PCDDs, PCDFs for humans and wildlife. *Environmental Health Perspectives* 106(12): 775-792.

Varanasi, U., J.E. Stein, and M. Nishimoto. 1989. Biotransformation and Disposition of Polycyclic Aromatic Hydrocarbons (PAHs) in Fish. In *Metabolism of Polycyclic Aromatic Hydrocarbons in the Aquatic Environment*, U. Varanasi, ed., CRC Press, Boca Raton, FL. pp. 94-149.

Walker, M.K. and R.E. Peterson. 1991. Potencies of polychlorinated dibenzo-*p*-dioxin, dibenzofuran, and biphenyl congeners, relative to 2,3,7,8-tetrachlorodibenzo-*p*-dioxin, for producing early life stage mortality in rainbow trout (*Oncorhynchus mykiss*). *Aquat. Toxicol*. 21:219-238.

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

Woodlot (Woodlot Alternatives). 2002. Fish Biomass Estimate for Housatonic River Primary Study Area. Prepared for U.S. Army Corps of Engineers. DCN: GE-061202-ABBF.

Zabel, E.W., P.M. Cook, and R.E. Peterson. 1995. Toxic equivalency factors of polychlorinated dibenzo-*p*-dioxin, dibenzofuran and biphenyl congeners based on early life stage mortality in rainbow trout (*Oncorhynchus mykiss*). *Aquat. Toxicol.* 31:315-328.

6. WILDLIFE ASSESSMENT HIGHLIGHTS

Highlights

- Conceptual model for wildlife indicates diet is the major route of exposure to tPCBs and TEQ. Therefore, wildlife exposure modeling approach focuses only on dietary exposure.
- Probabilistic methods used to propagate uncertainty through wildlife exposure model.
- Selection of dietary concentration variables required consideration of spatial and temporal averaging of exposure.
- Options available to characterize exposure-effects relationships for tPCBs and TEQ include developing dose-response curves, NOAELs or LOAELs, field-based thresholds, or threshold ranges.
- Weight-of-evidence approach was used to characterize risks.

6.1 OVERVIEW

The purpose of this section is to describe the general approach and methods used to estimate risks of contaminants of concern (COCs) to wildlife. The same general approach was appropriate for the majority of the wildlife endpoints. The endpoints for which this discussion applies include:

- Insectivorous Birds (Section 7)
- Piscivorous Birds (Section 8)
- Piscivorous Mammals (Section 9)
- Omnivorous and Carnivorous Mammals (Section 10)
- Threatened and Endangered Species (Section 11)

An overview of the following topics is included:

- Selection of COCs for wildlife
- Development of wildlife conceptual model
- Approach to wildlife exposure modeling
- Spatial and temporal averaging of exposure
- Probabilistic methods for propagating uncertainty
- Approach for effects assessment
- Approach for risk characterization

This section provides a comprehensive overview of the approaches used in modeling risk for wildlife to reduce repetition in each of the subsequent sections, but does not provide the technical details specific to each endpoint. The specific details are discussed in Appendix C and the wildlife risk assessment appendices (Appendices G to K).

In the COPC screening specific to wildlife species, several COPCs (primarily organochlorine pesticides) were screened out because their concentrations in the PSA are likely much lower than the measured values due to laboratory interference (see Section 2.4 and Attachment B.2). The organochlorine pesticides that were screened through the Pre-ERA, but screened out for this assessment were: 4,4'-DDE; O,p'-DDT; 4,4'-DDT; heptachlor epoxide; cis-nonachlor, trans-nonachlor, and oxychlordane (see Section B.4.6.2). It was suspected that the analytical results for these COPCs in fish tissues were, at least partially, the result of interference from PCB concentrations. To test this possibility, 10 fish tissue extracts were archived and frozen after the initial sample analyses were submitted to GERG for re-analysis using a more sensitive analytical method (GC/MS). The results indicated that pesticide concentrations in fish tissue were overestimated in the initial GC/ECD analyses.

The GC/MS analyses did not detect heptachlor epoxide in fish tissue and thus this COPC was eliminated from further consideration (previous analyses had not detected this COPC in soil, sediment, or water). Cis-nonachlor, trans-nonachlor, and oxychlordane were overestimated by 3 to 99 times in the initial analyses on which the Pre-ERA was based (Attachment B.2). Given that these COPCs have not been detected in any other medium (Table B-142), they likely pose little risk to wildlife species. However, there are no tissue-based benchmarks for nonachlor, trans-nonachlor, and oxychlordane and thus it cannot be stated with certainty that they do not pose risk to wildlife species.

Two of the DDT metabolites, 4,4'-DDE and 4,4'-DDT, were overestimated by 5 and 12 times, respectively. Given that the maximum hazard quotients calculated from the initial analyses for 4,4'-DDT across different sub-reaches of the PSA and fish size classes were less than 10 (Table B-142), it seems likely that this COPC poses no risk to wildlife species. The same logic applies to 4,4'-DDE, except in this case a few of the 324 samples used in the Pre-ERA have a quotient above 1 (Table B-142). Thus, it cannot be stated with certainty that 4,4'-DDE poses no risk to wildlife species. The remaining pesticide COPC that screened through the Pre-ERA, O,p'-DDT, was overestimated by 99 times in the analyses upon which the Pre-ERA was based, thus, it seems likely that O,p'-DDT poses no risk to wildlife species, although this cannot be stated with certainty. Methods for calculating TEQ concentrations are described in Section 6.4.

In summary, the COPCs that were retained for the probabilistic risk assessment for wildlife species were tPCBs and 2,3,7,8-tetrachlorodibenzo-p-dioxin (TCDD) toxic equivalence (TEQ). Total PCBs detected in Housatonic River media samples closely resemble the commercial PCB mixtures Aroclor 1260 and Aroclor 1254, which are similar in congener makeup. TEQ is calculated from coplanar PCB and dioxin and furan congeners using the toxic equivalency factor (TEF) approach developed by Van den Berg et al. (1998) (see Section 6.4 and Appendix C.10).

The conceptual model for the wildlife assessments is shown in Figure 6.1-1. The conceptual model outlines the ecosystem processes that qualitatively link stressor releases and primary, secondary, and tertiary exposure pathways to ecological receptors. Thus, the conceptual model provides a visual representation of the potential risk pathways to wildlife from COCs. Each representative species has a species-specific conceptual model. These are presented in Sections 7 to 11 and Appendices G to K.

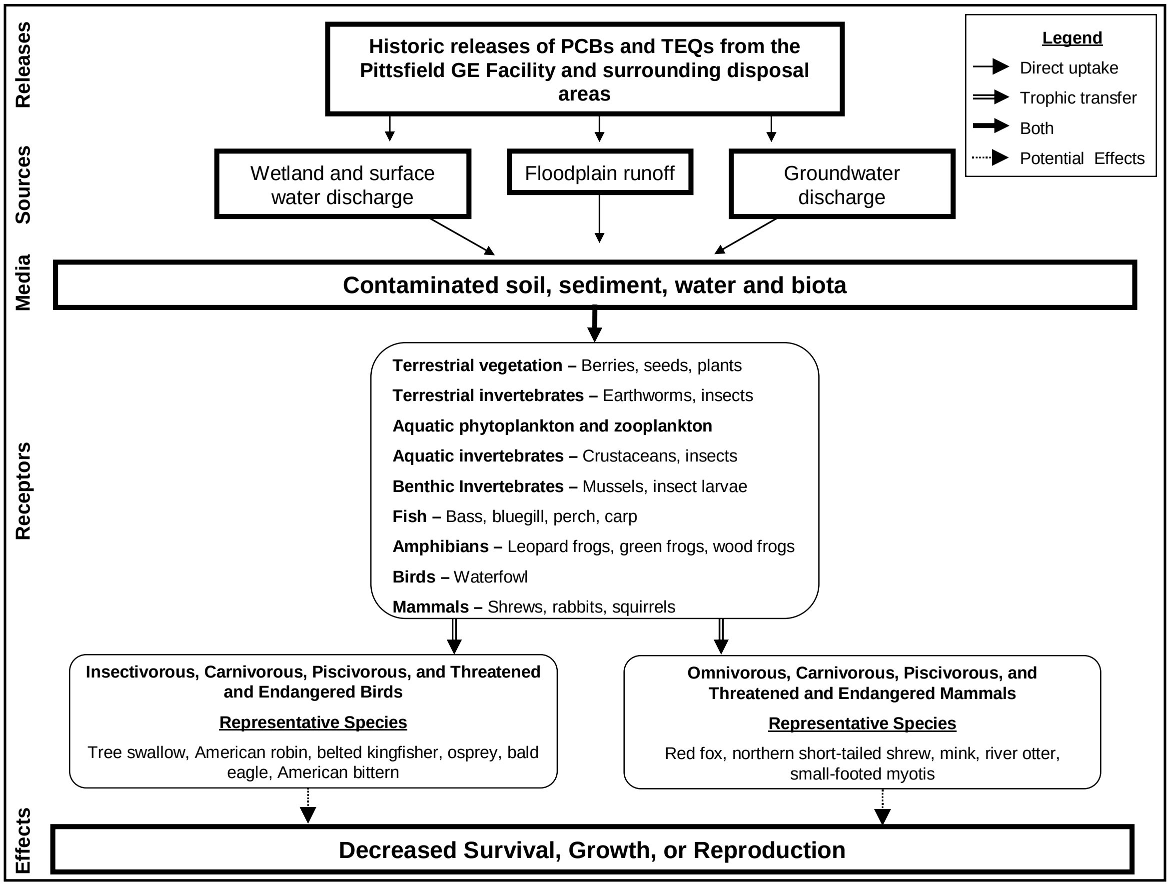


Figure 6.1-1 Conceptual Model for the Assessment of Risks from tPCBs and TEQ to Wildlife in the Housatonic River Primary Study Area

The wildlife risk assessments have three main components: the exposure assessment, the effects assessment, and risk characterization. The process used in each of these components is described below. Figure 6.1-2 depicts the framework for the exposure assessment.

During the exposure assessment, exposure of wildlife to tPCBs and TEQ in the Housatonic River Primary Study Area (PSA) was determined, beginning with a description of the exposure model. Input variables for the exposure model were established using life history information for the representative species, and concentrations of tPCBs and TEQ in prey collected in the PSA. For those input variables that were uncertain, variable, or both, distributions were used rather than point estimates. Monte Carlo and probability bounds analyses were then performed to propagate input variable uncertainties through the exposure model for each COC.

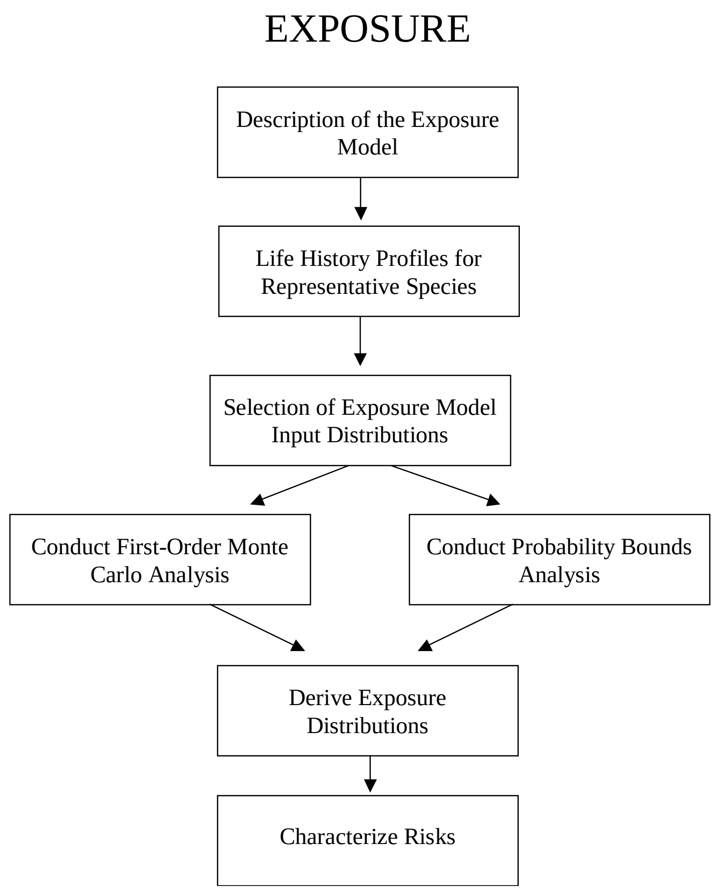


Figure 6.1-2 Framework Used to Model Exposure of Wildlife Species to Contaminants of Concern (COCs) in the Housatonic River PSA

Figure 6.1-3 shows the approach used in conducting the effects assessment. The effects assessment includes a comprehensive review of the literature on the effects of tPCBs and TEQ on survival, growth, and reproduction of representative wildlife species or reasonable surrogate species. Each of the studies was evaluated using acceptability criteria established for this ERA (see box below). Appropriate studies were then selected and used to derive the effects metric.

Acceptability Criteria for Studies of Effects on Wildlife

Were appropriate controls used?

Were appropriate statistics applied?

Were acceptable methods (e.g., laboratory methods) used?

Was there an appropriate range of exposure doses?

Was the experimental effect attributable to the COC?

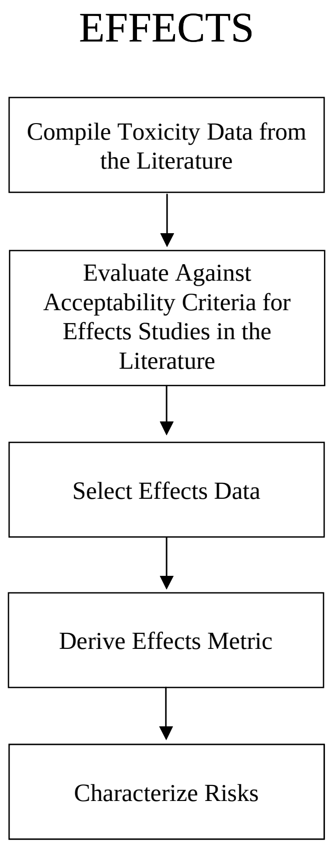


Figure 6.1-3 Approach Used to Model Effects of Contaminants of Concern (COCs) to Representative Species in the Housatonic River PSA

The final component of the wildlife risk assessments is the characterization of risk combining the results of the exposure and effects assessments, and other available lines of evidence (e.g., whole media or in situ studies, field surveys). Figure 6.1-4 presents the general approach used in the risk characterization process.

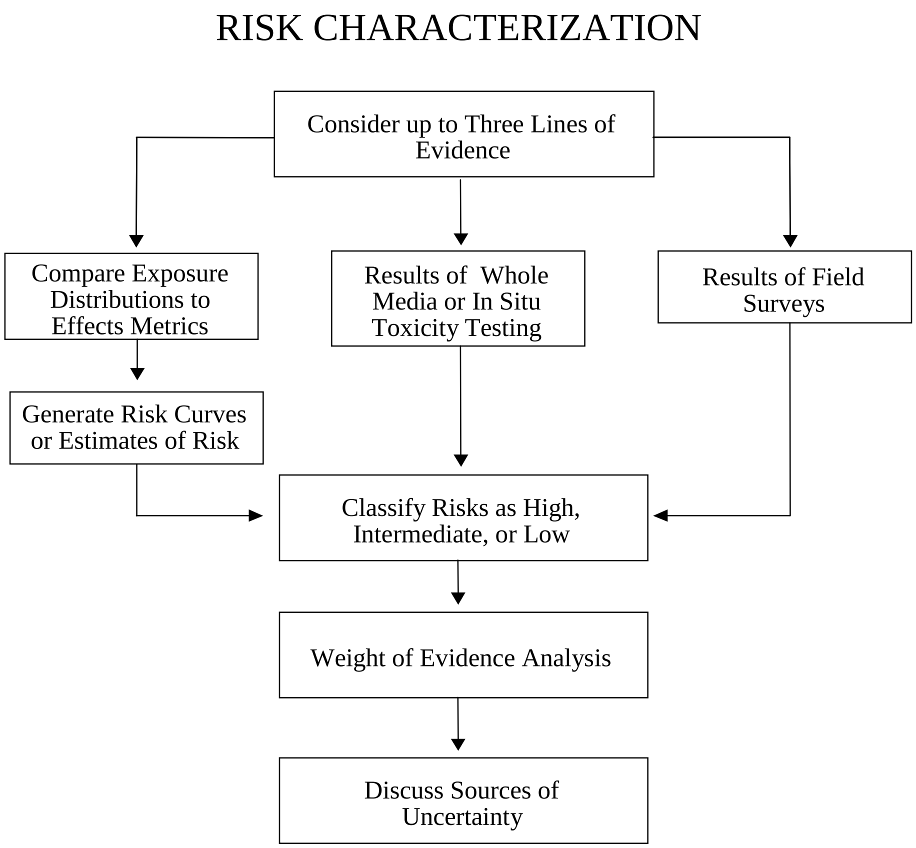


Figure 6.1-4 Approach Used to Characterize the Risks from Contaminants of Concern (COCs) to Representative Species in the Housatonic River PSA

In the risk characterization, the likelihood and magnitude of adverse effects occurring as a result of exposure of the representative wildlife species to tPCBs and TEQ was evaluated. A weight-of-evidence approach (WOE) was used to make a risk determination for each representative species. Several lines of evidence were available to characterize risks to wildlife from exposure to tPCBs and TEQ, however, not all lines of evidence were available for each species:

- Modeled Exposure and Effects – This line of evidence determines the extent to which the concentrations of tPCBs and TEQ ingested in the diet will cause adverse effects to the survival, reproduction, or growth of wildlife. Estimated exposures were compared to results of toxicological studies reported in the literature to determine if the representative wildlife species are exposed to tPCBs and TEQ at levels likely to induce adverse effects.

- Field Surveys – When available, this line of evidence was used to determine the relationship between the concentrations of tPCBs and TEQ and the abundance of wildlife in the Housatonic River floodplain.
- Whole Media or In Situ Studies – When available, this line of evidence was used to examine the relationship between tPCB and TEQ concentrations at specific sites or in whole media from the PSA and effects observed in wildlife species. This line of evidence is considered analogous to the site-specific toxicity testing line of evidence used in the assessments for aquatic receptors.

Each wildlife risk characterization includes a discussion of sources of uncertainty in the assessment of risks of COCs to wildlife, and the conclusions of the risk characterization.

6.2 WILDLIFE EXPOSURE MODEL

The approach for conducting the modeled exposure assessment for wildlife relies on the use of total daily intake models. The primary focus of the model is on ingestion of prey. The dietary exposure pathway is by far the most important exposure pathway for bioaccumulative substances such as tPCBs and TCDD and equivalence (TEQ) (Moore et al. 1997, 1999). Thus, the wildlife exposure assessments do not include environmental media in the exposure model calculations. The wildlife exposure model follows the general form:

$$TDI = FIR \bullet FT \bullet \sum_{i=1}^{n} C_i \bullet P_i \qquad \text{(Eq. 1)}$$

where:

TDI	=	Total daily intake (mg/kg bw/d tPCBs, ng/kg bw/d TEQ)
P_i	=	Proportion of i^{th} food item in the diet (unitless)
FIR	=	Normalized food intake rate (kg/kg bw/d)
C_i	=	Concentration in i^{th} food item (mg/kg tPCBs, ng/kg TEQ)
FT	=	Fraction of time in the contaminated area (unitless)

This general exposure model was customized accordingly for each representative species to reflect feeding habits, foraging range, habitat preferences, and life history. Extensive literature searches were conducted and data collected to determine the appropriate model inputs. Each of these inputs is briefly discussed below.

6.2.1 Food Intake Rate (FIR)

Data on food intake rate (FIR) are only available for a few species, primarily due to the difficulties in measuring intake for free-ranging wildlife. This assessment does not use measured food intake rates determined using captive animals, because such animals do not expend energy foraging for food and water, avoiding predators, defending territories, etc. (EPA 1993). Thus, food intake rates estimated for captive animals considerably underestimate expected food intake rates for free-ranging animals. In this assessment, allometric equations developed from measurements of free metabolic rate (*FMR*) in free-ranging animals (see text box below) were used to estimate food intake rate for each representative wildlife species. Food intake rate is derived from *FMR* using the following equation:

$$FIR(g/day) = \frac{FMR}{\sum_{i=1}^{n}(AE_i \, x \, GE_i)} \qquad \text{(Eq. 2)}$$

where AE_i is the assimilation efficiency of the i^{th} food item (unitless) and GE_i is the gross energy of the i^{th} food item (kcal/g). Where measured food intake rates are available for free-ranging animals for a representative species, the measured food intake rates are compared to the corresponding food intake rate derived using the allometric modeling approach. Such comparisons can be found in the wildlife sections and appendices.

Example Wildlife Free Metabolic Rate Equations

Birds

$$FMR(kJ/day) = 10.5 \cdot BW(g)^{0.681}$$ (Coraciiformes)

Mammals

$$FMR(kJ/day) = \log 0.221 \cdot BW(g)^{0.869}$$ (Carnivores)

where

BW = body weight (g).

(Nagy et al. 1999)

6.2.2 Body Weight (BW)

Body weights for each of the representative wildlife species were determined through data from the literature or data collected in the PSA of the Housatonic River. Data were combined from each relevant, acceptable study, and the mean and standard deviations calculated. Body weight is assumed to be a normally distributed parameter. The uncertainty associated with the variable is generally due to natural variability, rather than a lack of knowledge or data (i.e., body weight is easy to measure and data are available for each of the representative species).

6.2.3 Proportions of Dietary Items

Extensive literature searches were conducted to locate data and information on the dietary preferences of the wildlife species assessed. The information in the literature on dietary preferences was evaluated to determine relevance to representative species in the PSA and the timing of their exposures to COCs. Some wildlife species have dietary preferences that can include a large number of prey items. Therefore, only dietary items that comprise at least 10% of the total diet of each species were included in the exposure model. In these cases, dietary items for prey items comprising >10% of the diet were adjusted resulting in the sum of all dietary components equaling 100%. Because diets vary between locations and individuals and are also uncertain because of the limited data available for some species, distributions were used where appropriate for dietary variables.

6.3 SPATIAL AND TEMPORAL AVERAGING

Concentrations of COCs vary spatially and temporally in prey. The representative wildlife species forage over distances ranging from tens of meters to greater than 10 km. Thus, individuals tend to integrate spatial variation in the tissue concentrations of their prey over time. Therefore, estimates of the central tendency (i.e., arithmetic means) are used in the exposure model as an expression of the spatial and temporal averaging of concentrations of COCs in prey tissues (EPA 1999). In the Monte Carlo analysis, it was assumed that the spatially and temporally averaged exposure estimate did not vary between individuals foraging in the same area. Thus, the point estimate of centrality was the minimum of:

1. The 95% upper confidence limit (UCL) calculated using the Land H-statistic (assuming data are lognormally distributed), or
2. The maximum concentration measured.

In the probability bounds analyses, however, the uncertainty regarding the arithmetic mean was accounted for with a different procedure.

The procedure for the probability bounds analysis generally involved using the Land H-statistic to estimate the 95% lower confidence limits (LCL) and 95% UCLs on the mean (Gilbert 1987), and then using these LCLs and UCLs to derive bounds on all possible distributions that exist within this range. This approach results in an expression of the uncertainty about the true value of the arithmetic mean that arises due to the small sample size. In cases where the 95% UCL could not be estimated, or exceeded the maximum measured concentration, other techniques were used to derive the bounds on the mean (see Appendices G to K). Appendix C.5 describes the procedures for parameterizing prey concentration variables in more detail.

EPA (1992) states that because of the uncertainty associated with estimating the true average concentration for a site, "the 95% UCL of the arithmetic mean should be used for this variable." For lognormal data, EPA (1992) recommends the Land method using the H-statistic. Several authors (e.g., Ott 1995; Seiler and Alvarez 1996; Hattis and Burmaster 1994) have argued that concentrations of contaminants in environmental media tend to be lognormally distributed and that this may be expected because of mechanistic reasons. Current EPA guidance (EPA 1997; also see Haimes et al. 1994) states that distributions should be chosen for input variables on the basis of mechanistic or theoretical reasons, if possible, because such distributions have the highest degree of confidence. As a result, concentrations of contaminants in prey were assumed to be lognormally distributed in this ERA, and hence the Land H-statistic was used to estimate the 95% UCL. To determine the reasonableness of this assumption, the Shapiro-Wilk test was used to test for lognormality. Over two-thirds of the data sets used in the wildlife assessments passed the test for lognormality (i.e., $p > 0.05$), which supports the assumption of lognormality for concentrations of contaminants. That said, it is recognized that the Land method can produce high values for the UCL, particularly when data are not lognormally distributed, sample size is small, or variation is high (Singh et al. 1997; Schultz and Griffin 1999). EPA's (1992) guidance recognized this problem and recommended that the maximum detected concentration be used when the calculated UCL exceeds this value. This guidance was followed in this assessment.

6.4 TOXIC EQUIVALENCE (TEQ)

Some PCB congeners belong to a large class of chemicals called planar chlorinated hydrocarbons (PCH) that are regularly detected in the environment. The PCHs also include polychlorinated dibenzo-p-dioxins (PCDDs), and polychlorinated dibenzo-furans (PCDFs). PCHs have a common structural relationship that includes lateral halogenation (i.e., the addition of a halogen such as fluorine or chlorine to a compound), and the ability to assume a planar conformation (Figure 6.4-1).

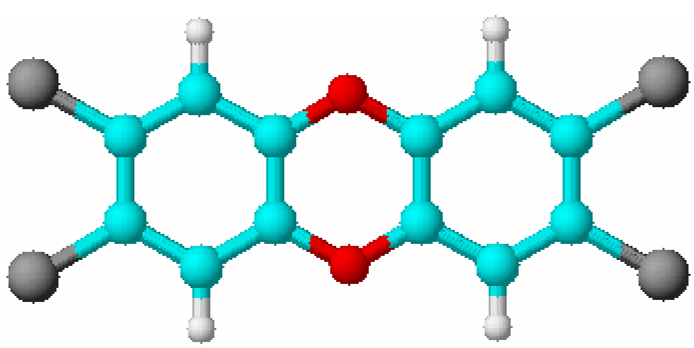


Figure 6.4-1 Molecular Structure of the Planar Chlorinated Hydrocarbon, 2,3,7,8-Tetrachlorodibenzo-p-dioxin

This structure is important because it leads to a common mechanism of action in many animal species that involves binding to the aryl hydrocarbon (Ah) receptor and elicitation of an Ah-receptor-mediated biochemical and toxic response (Van den Berg et al. 1998; Newsted et al. 1995; Safe 1994). Toxic responses include:

- Lethality.
- Hepatic lesions.
- Immunotoxicity.
- Tumor promotion.
- Adverse effects on reproduction.
- Induction of drug-metabolizing enzymes (Van den Berg et al. 1998; Newsted et al. 1995).

The planar structure determines the ability of the chemical to bind with the Ah receptor (Birnbaum and Devito 1995; Newsted et al. 1995). The Ah receptor facilitates the translocation of PCHs into the nucleus of affected cells and the binding of the PCH-Ah receptor complex to sites on the DNA (Newsted et al. 1995). Environmental degradation of PCH congeners varies due to their unique physical/chemical properties (Cogliano 1998) and thus there can be substantial differences between the congeners detected in environmental samples and the congener makeup of the original product (Cogliano 1998; Van den Berg et al. 1998). The congeners also have different toxic potencies. To address these issues and effectively estimate the relative toxicity of these mixtures, various systems have been created involving the development and use of toxic equivalency factors (TEFs) to derive toxic equivalence (TEQ) (Van den Berg et al. 1998; Safe 1990, 1994; EPA 1987, 1989, 1991; Kennedy 1996; NATO 1988a, 1988b; Ahlborg et al. 1994). The TEQ approach is based on the in vivo and in vitro toxicity of each of the PCH congeners in relation to 2,3,7,8-tetrachlorodibenzo-p-dioxin (TCDD). TCDD is considered to be the most toxic member of the PCH class of chemicals (Van den Berg et al. 1998; Birnbaum and DeVito 1995; Safe 1994). For this ERA, the TEFs proposed by Van den Berg et al. (1998) (also referred to as the World Health Organization or WHO TEFs) have been adopted (Table 6.4-1).

Table 6.4-1

World Health Organization Toxic Equivalency Factors (TEFs) for TCDD and Equivalents (Van den Berg et al. 1998)

Congener	Mammals	Fish	Birds
	TEF		
PCB-77	0.0001	0.0001	0.05
PCB-81	0.0001	0.0005	0.1
PCB-126	0.1	0.005	0.1
PCB-169	0.01	0.00005	0.001
PCB-105	0.0001	<0.000005[*]	0.0001
PCB-114	0.0005	<0.000005[*]	0.0001
PCB-118	0.0001	<0.000005[*]	0.00001
PCB-123	0.0001	<0.000005[*]	0.00001
PCB-156	0.0005	<0.000005[*]	0.0001
PCB-157	0.0005	<0.000005[*]	0.0001
PCB-167	0.00001	<0.000005[*]	0.00001
PCB-189	0.0001	<0.000005[*]	0.00001
1,2,3,4,6,7,8-HpCDD	0.01	0.001	<0.001[*]
1,2,3,4,6,7,8-HpCDF	0.01	0.01	0.01
1,2,3,4,7,8,9-HpCDF	0.01	0.01	0.01
1,2,3,4,7,8-HxCDD	0.1	0.5	0.05
1,2,3,4,7,8-HxCDF	0.1	0.1	0.1
1,2,3,6,7,8-HxCDD	0.1	0.01	0.01
1,2,3,6,7,8-HxCDF	0.1	0.1	0.1
1,2,3,7,8,9-HxCDD	0.1	0.01	0.1
1,2,3,7,8,9-HxCDF	0.1	0.1	0.1
1,2,3,7,8-PeCDD	1	1	1
1,2,3,7,8-PeCDF	0.05	0.05	0.1
2,3,4,6,7,8-HxCDF	0.1	0.1	0.1
2,3,4,7,8-PeCDF	0.5	0.5	1
2,3,7,8-TCDD	1	1	1
2,3,7,8-TCDF	0.1	0.05	1
OCDD	0.0001	<0.0001[*]	0.0001
OCDF	0.0001	<0.001[*]	0.0001

[*] Values that are "less than" should be considered to be the upper limit for use in any TEQ calculation.

These TEF values were developed for compounds that:

- Show a structural relationship to PCDDs and PCDFs.
- Bind to the Ah receptor.
- Elicit an Ah-receptor-mediated biochemical and toxic response.
- Are persistent and accumulate in the food chain (Van den Berg et al. 1998; Birnbaum and DeVito 1995).

The WHO TEFs are the most recent estimates of 2,3,7,8-TCDD equivalence and are based on current scientific research (Dyke and Stratford 2002). They have been accepted and applied in numerous jurisdictions worldwide (Dyke and Stratford 2002). Assumptions are made when using the TEF approach, including:

- PCH congeners are Ah-receptor antagonists and their toxicological potency is mediated by their binding affinity.
- No interaction occurs between the congeners and thus, the sum of the individual congener effects accounts for the potency of the PCH mixture.

The overall effect of these assumptions is a potency estimate or toxic equivalence (TEQ) value. To generate a TEQ the following equation (Equation 1- modified from Van den Berg et al. 1998) is used:

$$TEQ = \sum_{n=1}^{6}[PCDD_n \; x \; TEF_n] + \sum_{p=1}^{10}[PCDF_p \; x \; TEF_p] + \sum_{q=1}^{12}[PCB_q \; x \; TEF_q] \qquad \text{(Eq. 3)}$$

where

TEQ = Toxic equivalence

$PCDD_n$ = Polychlorinated dibenzo-p-dioxin congener concentration

$PCDF_p$ = Polychlorinated dibenzo-furan congener concentration

PCB_q = Polychlorinated biphenyl congener concentration

$TEF_{n,p,q}$ = Toxic equivalency factor for appropriate individual PCDD/PCDF and PCB congeners, respectively

Two circumstances often arise when calculating a TEQ value:

- Congener concentrations may be below the detection limit (i.e., non-detects), and
- Some congeners may not be resolved due to co-elution during analysis.

The approach used to address each of these circumstances in the ERA is discussed in the following sections.

6.4.1 Non-Detects

Congeners detected at or below the detection limit (DL) were included in the TEQ calculations by investigating three options: first, setting the value for the congener equal to zero (0), setting it to half the DL, and, finally, setting it equal to the DL (Appendix C.2). A comparison of the results of this bounding analysis provides a description of the uncertainty surrounding the TEQ value due to concentrations of one or more congeners being below the detection limit. This approach is also useful for determining the relative influence of individual non-detected congeners on the estimated TEQ value. Concentrations of tPCBs in prey in the PSA were all above the detection limit; therefore, there is no non-detect issue for tPCBs. However, treatment of non-detects remains a concern for the TEQ congeners.

6.4.2 Congener Co-Elution

The development of a TEQ using the WHO approach requires the concentrations for each of 29 unique congeners (12 PCB and 17 PCDD/PCDF congeners). During analysis of many of the tissue samples collected for the risk assessment, 2 of the 29 TEQ congeners (i.e., PCB-123 and PCB-157) co-eluted with other congeners. PCB-123 co-eluted with PCB-149 (PCB-123/149) and PCB-157 co-eluted with PCB-201 and PCB-173 (PCB-157/201/173). Assuming that the concentration of the congener PCB-123 is equal to the doublet concentration and that the concentration of PCB-157 is equal to the triplet concentration would likely overestimate the TEQ concentration. Conversely, assuming that concentrations of the two congeners (i.e., PCB-123 and PCB-157) were equal to zero would likely underestimate the TEQ concentration. These two approaches are useful to estimate TEQ bounds, but say little about the relative probabilities of values between the bounds.

Where possible, independent data sets were located for tissue types where analytical results were available for the co-eluted congeners in the Housatonic River database. Priority was given to data sets with tissue samples taken from the Housatonic River to minimize uncertainty associated with congener metabolism and environmental degradation. Only one appropriate data set was located, for fish tissue, in the Housatonic River that had unique results for each of the congeners in the doublet and triplet. Ratios of the congeners found in the independent data sets were generated and applied to the co-eluted congener data. The co-elution ratio was then multiplied by the reported result for the doublet and triplet concentrations to estimate the PCB-123 and PCB-157 concentrations for fish tissue samples. Uncertainty associated with the method for treating the co-eluted congeners includes interlaboratory variance due to different analytical methods, laboratory conditions, and analyst experience and expertise. The calculated ratios also do not account for differences between species found in the tissue database. A full description of the approach to developing the co-elution ratios from the independent data sets is provided in Appendix C.10.

Co-Elution Ratios

PCB-123/149 – 0.003/0.997

PCB-157/201/173 – 0.195/0.632/0.174

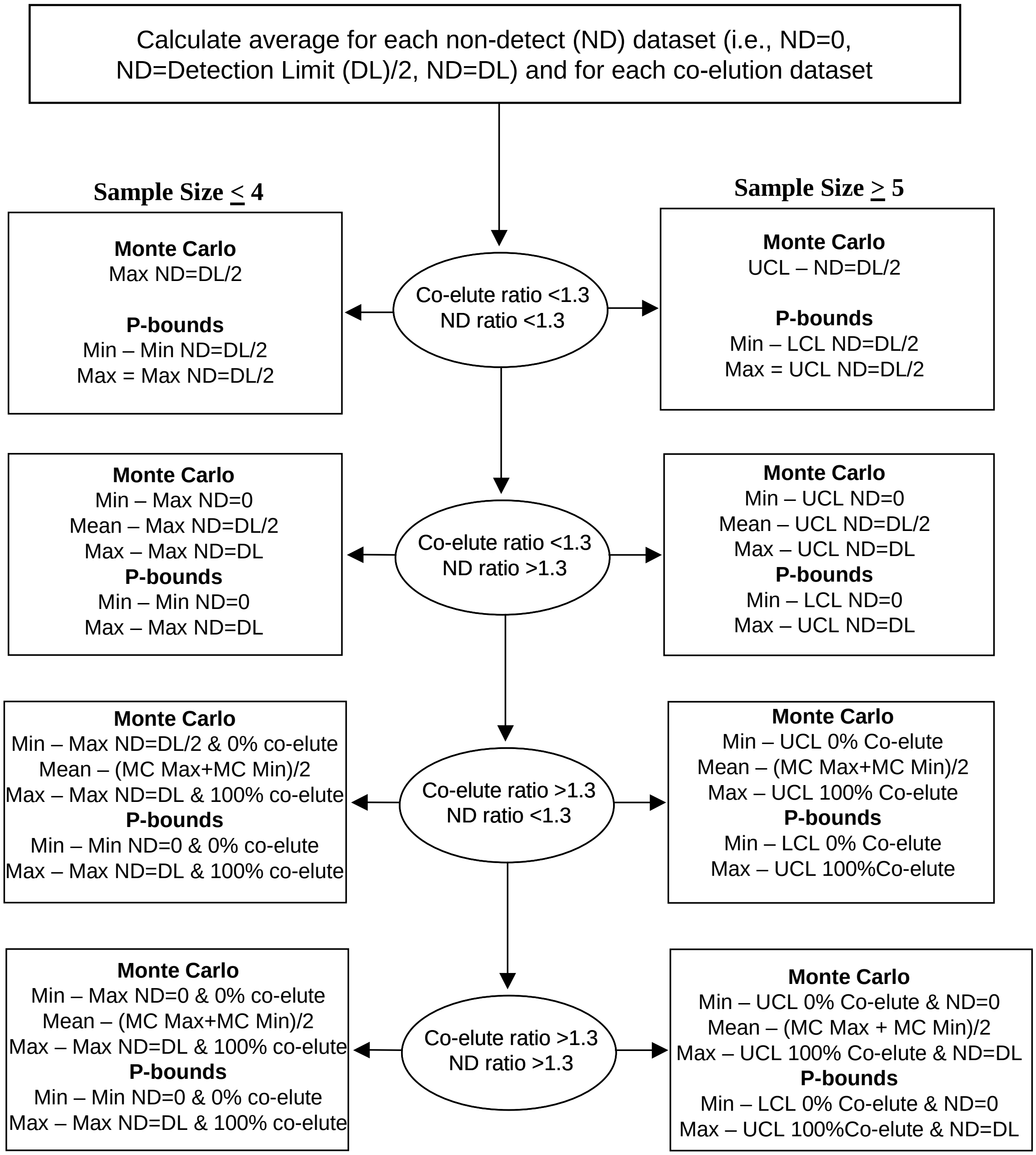
6.4.3 Summary of Decision Criteria for Estimating Exposure Point Concentrations

To deal with the uncertainty arising from co-elution or non-detect congeners when estimating exposure point concentrations (EPCs) for use in the exposure analyses, the following decision criteria (Figure 6.4-2) were developed (also see Appendix C.2):

- **Concentrations of COCs in samples where the concentration was below the detection limit** – To determine whether this source of uncertainty was important, arithmetic means were calculated for tissue concentrations assuming a concentration of zero for non-detected COCs (ND = 0), and assuming a concentration equal to the detection limit (ND = DL). If the ratio of the ND = DL mean to the ND = 0 mean was less than 1.3, this source of uncertainty was deemed unimportant. In these cases, exposure calculations were done assuming that concentrations of non-detected COCs were equal to half the detection limit (ND = ½ DL).[1] In cases where the ratio exceeded 1.3, the source of uncertainty was considered sufficiently important to incorporate in the exposure analysis. In the Monte Carlo analysis, for samples with COC concentrations below the detection limit three estimates of the EPC (i.e., estimates assuming ND = 0, ND = ½ DL, ND = DL) were used as parameters in a triangular distribution (i.e., minimum, best estimate, maximum). In the probability bounds analysis, the distribution-free range was the range spanning the LCL calculated assuming ND = 0 for the lower limit, and the UCL calculated assuming ND = DL for the upper limit.

- **Concentrations of TEQ in tissue samples (other than fish) with co-eluted congeners –** In some tissue samples, two PCB congeners required in the TEQ calculation (PCB-157 and PCB-123) co-eluted with other congeners. As a result, the concentrations of the triplet PCB-201/157/173 and the doublet PCB-149/123 are known, but not the concentrations of PCB-157 and PCB-123. This source of uncertainty was accounted for in the exposure calculations using an approach similar to that used to account for uncertainty stemming from non-detected COCs. For each tissue concentration variable, a ratio was calculated for mean TEQ concentration assuming that the concentration of PCB-157 and PCB-123 was zero, and the mean TEQ concentration assuming that the concentrations of these congeners were equal to the triplet and doublet concentrations, respectively. If the ratio was less than 1.3, this source of uncertainty was deemed unimportant. In these cases, exposure calculations were done assuming that concentrations of PCB-157 and PCB-123 were equal to the triplet and doublet concentrations, respectively. In cases where the ratio exceeded 1.3, the source of uncertainty was considered sufficiently important to incorporate in the exposure analysis. The procedures followed to accomplish this task were the same as used to deal with uncertainty due to non-detected COCs.

[1] This decision criterion supplements the procedures described in Appendix C.2.



UCL = Lower of the 95% UCL from the Land H-statistic or the dataset max
LCL = Higher of the 95% LCL from the Land H-statistic or the dataset min

Figure 6.4-2 Decision Tree for Determining Appropriate Treatment of Data with Non-Detects and Co-Elution

6.5 PROBABILISTIC RISK ASSESSMENT

6.5.1 Distribution Selection

Input distributions for the exposure analyses were generally assigned as follows: lognormal distributions for variables that were right skewed with a lower bound of zero and no upper bound (e.g., amount of COC transferred from mother to offspring via egg tissue), beta distributions for variables bounded by zero and one (e.g., proportion of a prey item in the diet), normal distributions for variables that were symmetric and not bounded by one (e.g., body weight), and point estimates for minor variables or variables with low coefficients of variation. In certain situations (e.g., poor fit of data), other distributions were fit to the data or other approaches were used. To quantify uncertainty, two approaches were used as described in Section 6.5.2, below.

6.5.2 Monte Carlo and Probability Bounds Analysis

General Risk Assessment Approaches

Deterministic Methods – Methods in which all biological, chemical, physical, and environmental parameters are assumed to be constant and can be accurately specified.

Probabilistic Methods – Methods in which important biological, chemical, physical, and environmental parameters are assumed to vary or are uncertain and therefore, are specified using distribution of possible values.

Monte Carlo and probability bounds analysis are two uncertainty propagation techniques used in the Housatonic River wildlife risk assessments. The use of probabilistic methods in risk analysis is growing rapidly and EPA has produced guidance on how to conduct such analyses in Superfund and other programs (EPA 1997, 1999). The benefit of using probabilistic methods in risk assessment is that they give the risk assessor the ability to fully characterize risk, rather than providing a best estimate or a conservatively biased estimate of risk. For example, calculating a mean risk (i.e., deterministic method) may exclude the potential for relatively rare, but serious, extreme events (e.g., species extinction). This is generally undesirable, because although rare, these events can occur and have significant impacts on individuals, communities, and populations of species. By including the entire distribution for risk, all events are considered and all of the data and information collected to characterize a situation are included. The remainder of this section provides a short overview of Monte Carlo and probability bounds analysis as applied in the wildlife risk assessments. Further technical detail on these methods can be found in Appendix C.4.

Probabilistic Methods
Monte Carlo Analysis – A technique where parameter values are drawn at random from defined input probability distributions, combined according to a model equation, and the process repeated iteratively until a stable distribution of solutions results. It is most useful when input distributions are known precisely.
Probability Bounds Analysis – Separates uncertainty and variability to obtain bounds on the result that explicitly account for the uncertainty about the input distributions.

The primary goal of a Monte Carlo analysis in the risk assessment is to characterize quantitatively, the uncertainty and variability in estimates of exposure and risk (EPA 1997). A secondary goal is to identify key sources of variability and uncertainty and to quantify the relative contribution of these sources to the overall range of wildlife exposure model results. While Monte Carlo methods are appropriate for the determination of exposure risks when input distributions are known precisely, they may not adequately represent the effects of uncertainty about how to parameterize variability in the input distributions (Ferson 1996). In many ecological risk assessments, the available data are limited and consequently the input distributions used to calculate risks are uncertain. Probability bounds analysis is a tool for separating variability and uncertainty to obtain bounds on the result that explicitly account for uncertainty about the input distributions. As in Monte Carlo analysis, the overall slopes of the bounds indicate how much variability exists in the system. The distance between the bounds, on the other hand, is an indication of the uncertainty that exists due to lack of knowledge. An example of exposure model outputs from Monte Carlo and probability bounds analyses is presented in Figure 6.5-1.

The wildlife exposure models contain multiple variables, some of which may be correlated. The assumption of independence can be inappropriate, because dependencies can affect the estimates of exposure. If correlations are not accounted for, the variance and the tails of the exposure distribution may be poorly estimated. The wildlife assessments use several approaches to address correlations between variables. These approaches include simulation of observed correlations, assumption of perfect covariance (e.g., when the diet consists of two prey items, the proportion of one item in the diet is equal to one minus the other item), or no assumptions at all about dependencies (e.g., all possible relationships between two variables can occur). The specific approach used depends on the type of data and the application. In cases where independence of variables seemed intuitively obvious (e.g., COC concentration in the prey item and proportion of that item in the diet), independence was assumed.

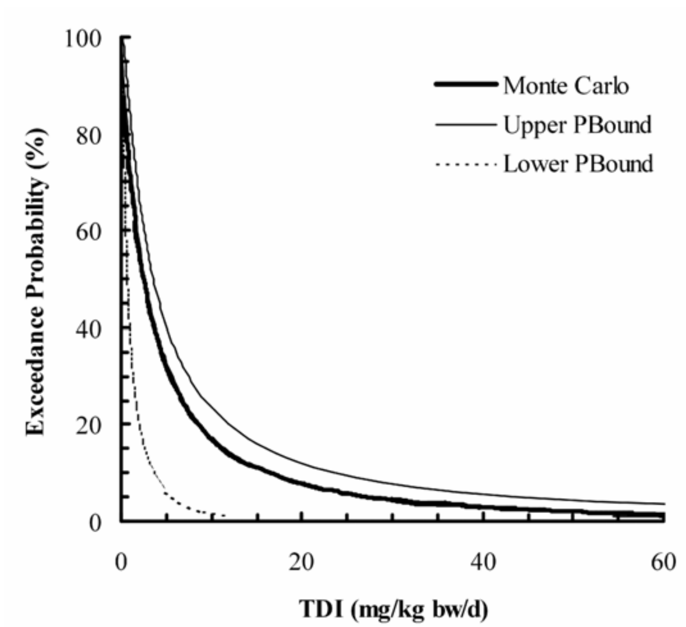


Figure 6.5-1 Example Exposure Distribution from Monte Carlo and Probability Bounds Analyses (TDI = total daily intake of tPCBs)

6.6 EFFECTS ASSESSMENT

Effects data can be characterized and summarized in a variety of ways ranging from benchmarks designed to be protective of most or all species to dose-response curves for the representative species of interest. In this ERA, effects characterization preferentially relied on concentration- or dose-response curves, but defaulted to benchmarks or other estimates of effect (e.g., no observed adverse effect level [NOAEL], or lowest observed adverse effect level [LOAEL]) when insufficient data were available to derive dose-response curves. Effects associated with growth, survival, and reproduction were generally the preferred measures of effect as they most closely relate to the assessment endpoints for wildlife. This section provides an overview of the procedures used for characterizing effects information and describes the decision criteria for choosing among them for each receptor-COC combination.

Figure 6.6-1 displays the hierarchy of decision criteria used to characterize effects for each receptor-COC combination. In all cases, the units of the effects metrics were consistent with the units of the exposure metrics. To the extent possible, effects metrics were based on long-term studies to match expected exposure durations.

The remainder of this section provides details on how the effects metrics were derived from the decision tree.

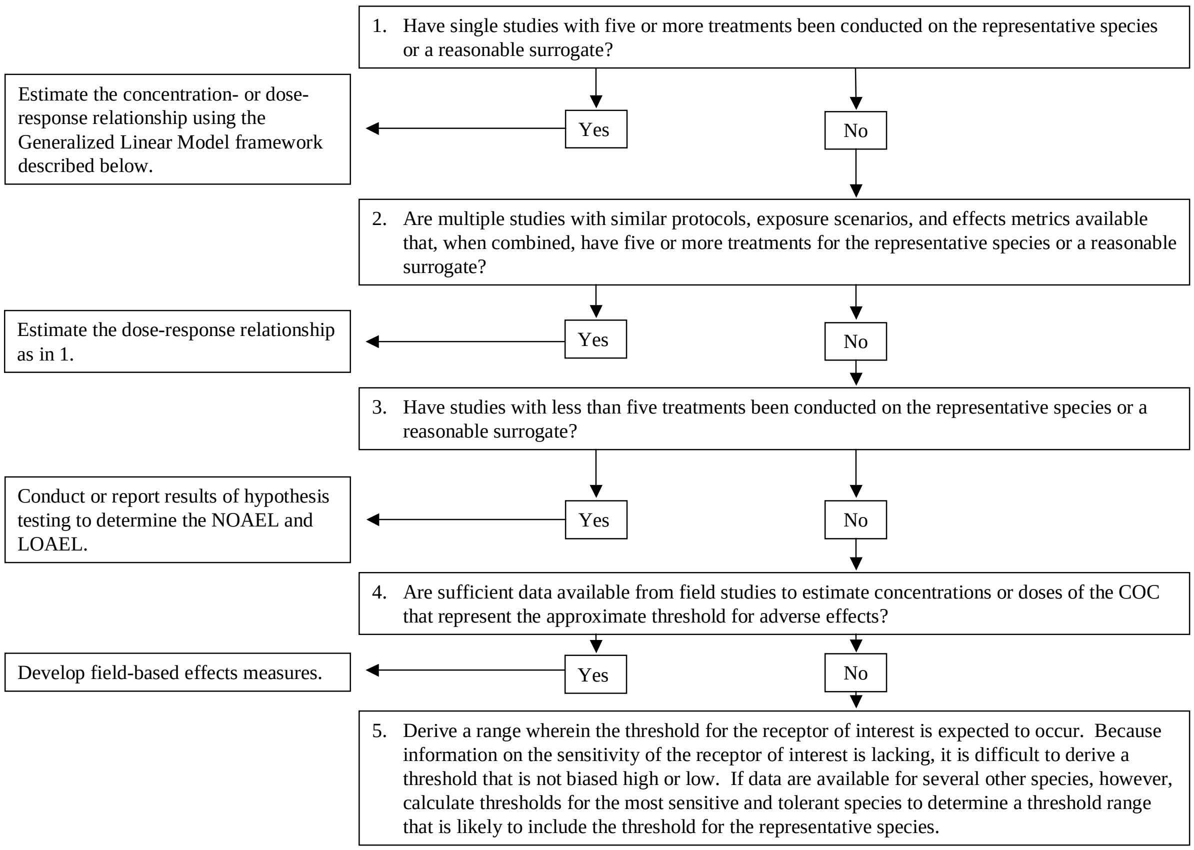


Figure 6.6-1 Decision Criteria Used to Characterize Effects for Each Wildlife Receptor-COC Combination

6.6.1 Dose-Response Relationships Using the Generalized Linear Model Framework

Most probabilistic risk assessments previously conducted estimated the probability that exposure exceeded a specified no-observed-effects or lowest-observed-effects dose. An alternative approach is to estimate the probabilities of effects of varying magnitude. To do this, a concentration- or dose-response model is required. Generally, five or more treatments are required to develop concentration- or dose-response relationships, either from a single study or from several studies that used a similar methodology. The Generalized Linear Model (GLiM) framework described by Kerr and Meador (1996) and Bailer and Oris (1997) is a useful framework for deriving these relationships. The framework involves using link functions to transform effects metrics (e.g., probit or logit link functions for quantal responses) and assigning appropriate error distributions (e.g., binomial distribution for quantal responses). Linear regression can then be conducted on the transformed data to derive the dose-response relationship. Thus, the framework can be used for all available types of response variables (Moore et al. 2000). By adding a quadratic term to the linear model, the framework can be adapted to incorporate simulation at low doses. The GLiM framework was used to derive dose-response relationships in this ERA when five or more treatments were available from a single study for the receptor of interest or a reasonable surrogate. In some cases, it was necessary to convert concentration-response relationships to dose-response relationships by multiplying the former by the food intake rate of the species (Moore et al. 1999).

Dose-response relationships are combined with the corresponding exposure distribution in risk characterization to derive risk curves that characterize the relationship between probability and magnitude of effect.

6.6.2 Hypothesis Testing to Determine LOAEL and NOAEL

Analysis of variance (ANOVA) is the most common method of estimating low-level toxic effects from chronic tests. There are several reasons for this, including the wide availability of software capable of performing ANOVA and related nonparametric tests, and the familiarity of regulators with the technique. Until recently, most toxicity-testing protocols specified experimental designs more suited to hypothesis-testing methods such as ANOVA than to regression-based approaches. However, hypothesis testing as an approach for estimating low-level toxic effects has some limitations, including:

1. NOAELs and LOAELs are test doses that do not correspond with specified effects levels from one test to the next.
2. Poor experimental design may mistakenly indicate that a contaminant is less toxic than it really is.
3. Most information available from the toxicity test is not used (Stephan and Rogers 1985; Pack 1993; Suter 1996).

As a result, hypothesis testing was not the preferred method for analysis of toxicity data in this ERA.

However, in many cases, toxicity studies with five or more treatment levels are not available for the representative species of interest or for a reasonable surrogate for tPCBs and TEQ. In those cases, the use of hypothesis testing was necessary to estimate the NOAEL and LOAEL. In many toxicological studies, these endpoints were previously determined and reported. Such studies were evaluated to determine that proper statistical procedures were followed. Where the data could be obtained from the reports or directly from the authors, the data were re-analyzed. In cases where a re-analysis was conducted, information regarding the minimal difference required to give a significant result was reported (e.g., number of replicates, test variance, α, β, test dose intervals). The percent effect associated with the LOAEL, relative to the control, was also reported.

6.6.3 Field-Based Measures of Effect and Threshold Ranges

Field-based measures of effect were derived from monitoring or in situ toxicity tests conducted on the representative species or a reasonable surrogate. There are several methods available for deriving field-based measures of effect. For benthic invertebrates, chemistry and effect data from surveys of sediment and biota in various locations have been combined to develop sediment concentrations that are generally protective or, conversely, likely associated with adverse effects (Long et al. 1995; MacDonald et al. 1996). Similar approaches can be used with wildlife species. With in situ studies, if sufficient data were available, regression-based approaches (e.g., GLiM models) could be used to link concentrations or doses with effects observed in the field.

When data are lacking on the toxicity of a particular COC to the representative species or a reasonable surrogate, threshold ranges were developed. In these cases, it is not known whether the representative species is sensitive or tolerant. Therefore, a threshold range was developed that spanned the concentrations (or doses) that would be protective of sensitive species to those that would be protective only for tolerant species. The assumption is that the threshold for the representative species lies between these two extremes. To derive a threshold range, the toxicity literature was reviewed to determine the most sensitive and the most tolerant species for which studies have been conducted. Thresholds were derived for both the most sensitive and the most tolerant species using methods similar to those used in the Pre-ERA (see Section 2.4 and

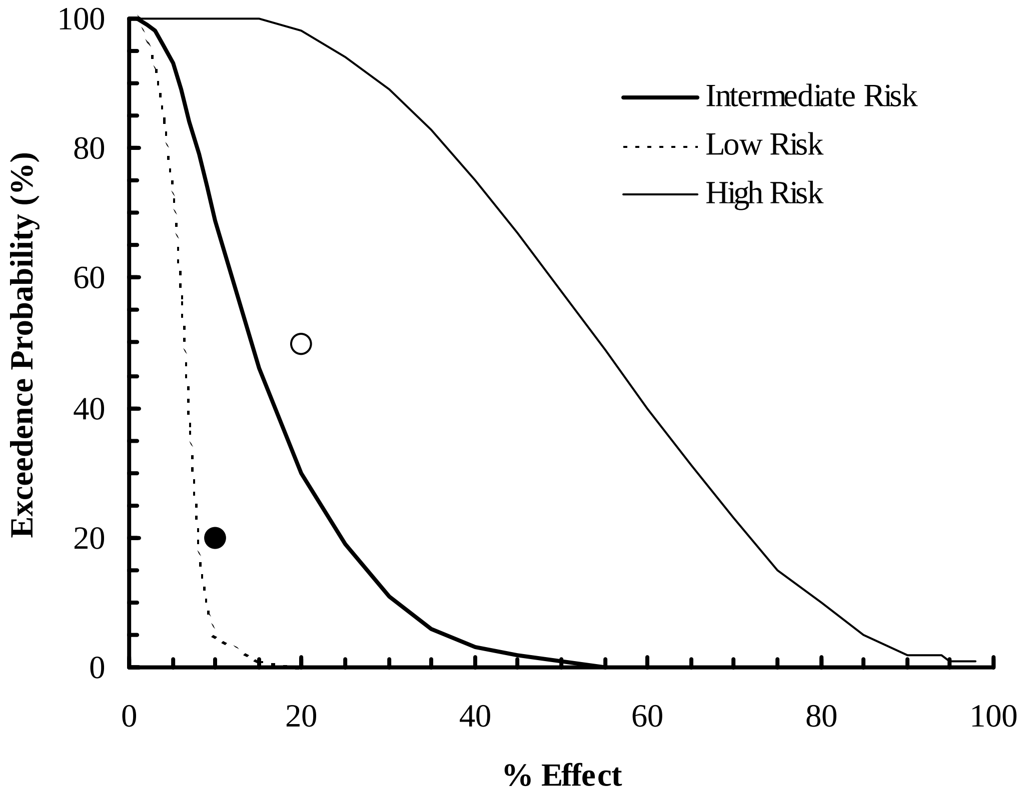
Appendix B). The two resulting thresholds become the threshold range, which was then compared to the exposure assessment results in the risk characterization.

6.7 RISK CHARACTERIZATION

6.7.1 Risk Categorization

If possible, risk should be expressed quantitatively (Wentsel et al. 1997). For example, a risk could be expressed as a 10% probability of >25% mortality for a particular species. In this ERA, quantitative expressions of risk were derived for each of the wildlife assessment endpoints (Appendices G to K) to facilitate discussion and to simplify comparisons of risk between species, COCs, and locations. The following criteria were used to categorize risks to wildlife as high, intermediate, or low:

- Scenarios with effects data for the representative species (or a reasonable surrogate):
  - If the probability of 10% or greater effect (or of exceeding the NOAEL) was less than 20%, then the risk was categorized as low (Figure 6.7-1).
  - If the probability of 20% or greater effect (or of exceeding the LOAEL) was greater than 50%, then the risk was categorized as high (Figure 6.7-1).
  - All other outcomes were categorized as intermediate risk (Figure 6.7-1).
- Scenarios with effects data for the representative threatened and endangered species (or a reasonable surrogate):
  - If the probability of 10% or greater effect (or of exceeding the NOAEL) was less than 20%, then the risk was categorized as low.
  - If the probability of 10% or greater effect (or of exceeding the NOAEL) was greater than 50%, then the risk was categorized as high.
  - All other outcomes were categorized as intermediate risk.



Note: Risk curves passing below the filled circle symbol indicate low risk, while those passing above the open circle symbol indicate high risk.

Figure 6.7-1 Example Risk Curves Indicating Low, Intermediate, and High Risk Categories

- Scenarios with threshold concentrations for the representative species (or a reasonable surrogate):

  - If the probability of exceeding the threshold for the most sensitive species was less than 20%, the risk was categorized as low.

  - If the probability of exceeding the threshold for the most tolerant species was greater than 20%, the risk was categorized as high.

  - All other outcomes were categorized as intermediate risk.

Each categorization of risk was derived from the results of the Monte Carlo exposure analyses (Figure 6.7-1). To capture the uncertainty about a risk categorization, the results from the corresponding probability bounds analysis were compared to the above criteria to determine a risk range (risk category using lower probability bound to risk category using upper probability bound).

These risk categorization criteria were based on several considerations including:

- Efroymson and Suter (1999) and others (e.g., Pack 1993) suggested that reductions in survival, growth, or reproduction of 20% or greater is indicative of significant effects to wildlife. Thus, a better than even chance (i.e., >50%) of exceeding this effect level was deemed to represent a high risk situation. However, because effects at or above the 20% level possibly may not be ecologically significant, these categorizations should be considered further in each situation. For example, a stressor causing a 20% decline in reproductive fecundity of brook trout was shown to lead to a general lowering of risks of population decline compared to unexposed conditions because the negative consequences of overcrowding were diminished (Ferson et al. 1996). Similar effects on other fish species, however, have led to population collapses (Myers et al. 1995).

- Although there are exceptions (such as threatened and endangered species), an effect level of 10% is unlikely to be ecologically significant. Thus, if the probability of exceeding this effect level is relatively low (<20%), risk is deemed to be negligible to low.

- Several studies have shown that NOAELs are generally associated with effects of 10% or greater (85% of studies examined by Moore et al. 1997), and LOAELs are generally associated with effects of 20% or greater (79% of studies examined by Moore et al. 1997) (also see Hoeckstra and Van Ewijk 1993; Pack 1993). Therefore, the decision criteria above equated NOAELs with the 10% effect level, and LOAELs with the 20% effect level.

- When toxicity data are lacking for representative species or reasonable surrogates, the toxicity threshold for representative species is assumed to be between the thresholds of the most sensitive and tolerant species tested. Thus, if the probability of exceeding the lowest threshold is low (<20%), risk is deemed to be negligible to low. Tolerant species may have thresholds one to several orders of magnitude higher than sensitive species (see effects assessment sections in Appendices G to K). Thus, at the highest threshold, it is likely that some representative species would be adversely affected, possibly quite seriously. Thus, even a relatively low probability (20% or greater) of exceeding the upper threshold may be cause for concern.

- Any effect to threatened and endangered species is cause for concern (Massachusetts Office of Environmental Affairs 1999; Massachusetts Division of Fisheries and Wildlife 2003; United States Congress 1973, Endangered Species Act). Because a LOAEL generally represents >20% effect, the criterion separating intermediate and high risk was adjusted for threatened and endangered species. Thus, a better than even (>50%) chance of exceeding a NOAEL was deemed to represent a high risk situation for threatened and endangered species.

The risk categories should not be used alone to determine whether risk management actions are necessary. Risk categories are based on the results of the Monte Carlo exposure analysis only and are as follows:

- When Monte Carlo results indicate a low-risk category (i.e., estimated exposure levels fall below the lowest risk threshold), the conclusion is that the evidence of harm is “no” and the magnitude is “low.”

- When Monte Carlo exposure estimates fall between the low-risk and high-risk thresholds, evidence of harm is expressed as “yes,” with the associated magnitude of effects being “intermediate.”

- When Monte Carlo exposure estimates are greater than the high-risk threshold the evidence of harm is expressed as “yes,” with the associated magnitude of effects being “high.”

To further characterize the uncertainty of the Monte Carlo analysis results, p-bounds results are summarized using a qualitative risk range category system that represents the possible relationships between the risk thresholds and the upper and lower p-bounds. The p-bounds analysis was conducted to illustrate both the variability and uncertitude associated with the Monte Carlo results. The following discussion provides a description of the risk range categories that could occur in the ERA:

- Low—Both p-bounds fall below the low-risk threshold.
- Low-Intermediate—The lower p-bound falls below the low-risk threshold and the upper p-bound falls between the two risk thresholds.
- Intermediate—Both p-bounds fall between the two risk thresholds.
- Intermediate-High—The lower p-bound falls between the risk thresholds and the upper p-bound falls above the upper risk threshold.
- High—Both p-bounds fall above the upper risk threshold.

6.7.2 Weight-of-Evidence Assessment

A WOE approach was used in the risk assessments for wildlife. The WOE approach is a process by which measurement endpoints are related to an assessment endpoint to evaluate whether significant harm is posed to the environment (Menzie et al. 1996). The WOE approach used in this ERA follows the approach originally described in the *Massachusetts Weight of Evidence Special Report* (Menzie et al. 1996). A detailed review of the WOE approach used in the Housatonic River ERA is provided in Section 2.9.

In general, the WOE approach is an inclusive process whereby multiple lines of evidence are considered prior to determining risk. For the wildlife risk assessments, these lines of evidence included the exposure and effects modeling results, field survey results, and/or in situ or whole media toxicity test results. For the modeling of exposure and effects line of evidence, risk categories and risk ranges were developed.

6.8 REFERENCES

Alhborg, U.S., G.C. Becking, L.S. Birnbaum, A. Brouwer, H.J.G.M. Derks, M. Feeley, G. Golor, A. Hanberg, J.C. Larsen, A.K.D. Liem, S.H. Safe, C. Schlatter, F. Wern, M. Younes, and E. Yrjanheikki. 1994. Toxic equivalency factors for dioxin-like PCBs: Report on a WHO-ECEH and IPCS consultation, December 1993. *Chemosphere* 28:1049-1067.

Bailer, A.J. and J.T. Oris. 1997. Estimating inhibition concentrations for different response scales using generalized linear models. *Environmental Toxicology and Chemistry* 16:1554-1559.

Birnbaum, L.S. and M.J. DeVito. 1995. Use of toxic equivalency factors for risk assessment for dioxins and related compounds. *Toxicology* 105:391-401.

Cogliano, V.J. 1998. Assessing the cancer risk from environmental PCBs. *Environmental Health Perspectives* 106(6):317-323

Dyke, P.H. and J. Stratford. 2002. Changes to the TEF schemes can have significant impacts on regulation and management of PCDD/F and PCB. *Chemosphere* 47:103-116.

Efroymson, R. and G.W. Suter II. 1999. Finding a niche for soil microbial toxicity tests in ecological risk assessment. *Human and Ecological Risk Assessment* 5(4):635-868.

EPA (U.S. Environmental Protection Agency). 1987. *Interim Procedures for Estimating Risk Associated with Exposure to Mixtures of Chlorinated Dibenzo-p-dioxins and Dibenzofurans (CDD:CDFs).* U.S. Environmental Protection Agency. EPA 625-3-87012.

EPA (U.S. Environmental Protection Agency). 1989. *Interim Procedures for Estimating Risks Associated with Exposure to Mixtures of Chlorinated Dibenzo-p-dioxins and Dibenzofurans and 1989 Update.* Risk Assessment Forum, EPA/625/3-89/016.

EPA (U.S. Environmental Protection Agency). 1991. *Workshop Report on Toxicity Equivalence Factors for Polychlorinated Biphenyl Congeners Risk Assessment Forum.* U.S. Environmental Protection Agency. EPA/625/3-91/020.

EPA (U.S. Environmental Protection Agency). 1992. *A Supplemental Guidance to RAGS: Calculating the Concentration Term.* Publication 9285.7-081. Office of Research and Development, Washington, DC. http://www.deq.state.ms.us/newweb/opchome.nsf/pages/ HWDivisionFiles/ $file/uclmean.pdf.

EPA (United States Environmental Protection Agency). 1993. *Wildlife Exposure Factors Handbook.* Volumes I and II. Office of Research and Development. Washington, DC. EPA EPA/600/R-93/187a, EPA/600/R-93/187b.

EPA (U.S. Environmental Protection Agency). 1997. *Guiding Principles for Monte Carlo Analysis.* U.S. Environmental Protection Agency, Office of Research and Development, Washington, DC. EPA/630/R-97/001.

EPA (U.S. Environmental Protection Agency). 1999. *Risk Assessment Guidance for Superfund,* Volume 3 – Part A, *Process for Conducting Probabilistic Risk Assessment*. Draft. Office of Solid Waste and Emergency Response, U.S. Environmental Protection Agency, Washington, DC.

Ferson, S. 1996. What Monte Carlo methods cannot do. *Human and Ecological Risk Assessment* 2(4):990-1007.

Ferson, S., L.R. Ginzburg, and R.A. Goldstein. 1996. Inferring ecological risk from toxicity bioassays. *Journal of Water, Air and Soil Pollution* 90:71-82.

Gilbert, R.O. 1987. *Statistical Methods for Environmental Pollution Monitoring*. Van Nostrand Reinhold Company, New York.

Haimes, Y.Y., T. Barry, and J.H. Lambert, Editors. 1994. When and how you can specify a probability distribution when you don't know much? *Risk Analysis* 14:661-706.

Hattis, D. and D.E. Burmaster. 1994. Assessment of variability and uncertainty distributions for practical risk analyses. *Risk Analysis* 14:713-730.

Hoeckstra, J.A. and P.H. Van Ewijk. 1993. Alternatives for the no-observed-effect-level. *Environmental Toxicology and Chemistry* 12:187-194.

Kennedy, S.W., A. Lorenzen, and R.J. Norstrom. 1996. Chicken embryo hepatocyte bioassay for measuring cytochrome P4501A-based 2,3,7,8-tetrachlorodibenzo-p-dioxin equivalent concentrations in environmental samples. *Environmental Science and Technology* 30:706-715.

Kerr, D.R. and J.P. Meador. 1996. Modeling dose response using generalized linear models. *Environmental Toxicology and Chemistry* 15:395-401.

Long, E.R., D.D. MacDonald, S.L. Smith, and F.D. Calder. 1995. Incidence of adverse biological effects within ranges of chemical concentrations in marine and estuarine sediments. *Environmental Management* 19(1):81-97.

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

Massachusetts Division of Fisheries and Wildlife. 2003. 321 CMR 10.00 and 11.00. Massachusetts Endangered Species Act Regulations. Boston, MA.

Massachusetts Office of Environmental Affairs. 1999. 310 CMR 40.00. Massachusetts Contingency Plan (MCP). Boston, MA.

Menzie, C., M.H. Henning, J. Cura, K. Finkelstein, J. Gentile, J. Maughan, D. Mitchell, S. Petron, B. Potocki, S. Svirsky, and P. Tyler. 1996. Special report of the Massachusetts Weight-of-Evidence Workgroup: A weight-of-evidence approach for evaluating ecological risks. *Human and Ecological Risk Assessment* 2(2):277-304.

Moore, D.R.J., W. Warren-Hicks, B.R. Parkhurst, R.S. Teed, R.B. Baird, R. Berger, D.L. Denton, and J.J. Pletl. 2000. Intra- and intertreatment variability in reference toxicant tests: Implications for whole effluent toxicity testing programs. *Environmental Toxicology and Chemistry* 19(1):105-112.

Moore, D.R.J., B.E. Sample, G.W. Suter, B.R. Parkhurst, and R.S. Teed. 1999. A probabilistic risk assessment of the effects of methylmercury and PCBs on mink and kingfishers along East Fork Poplar Creek, Oak Ridge, Tennessee, USA. *Environmental Toxicology and Chemistry* 18(12):2941-2953.

Moore, D.R.J., R.L. Breton, and K. Loyd. 1997. The effects of hexachlorobenzene on mink in the Canadian environment: An ecological risk assessment. *Environmental Toxicology and Chemistry* 16(5):1042-1050.

Moore, D.R.J. and P.-Y. Caux. 1997. Estimating low toxic effects. *Environmental Toxicology and Chemistry* 16(4):794-801.

Myers, R.A., N.J. Barrowman, J.A. Hutchings, and A.A. Rosenberg. 1995. Population dynamics of exploited fish stocks at low population levels. *Science* 269:1106-1108.

Nagy, K.A., I.A. Girard, and T.K. Brown. 1999. Energetics of free-ranging mammals, reptiles, and birds. *Annual Reviews of Nutrition* 19:247-77

NATO (North Atlantic Treaty Organization). 1988a. *International Toxicity Equivalence Factors (I-TEF) Method of Risk Assessment for Complex Mixtures of Dioxins and Related Compounds. Pilot Study on International Information Exchange on Dioxins and Related Compounds.* Report Number 176, August 1988. North Atlantic Treaty Organization, Committee on Challenges of Modern Society.

NATO (North Atlantic Treaty Organization). 1988b. *Scientific Basis for the Development of International Toxicity Equivalency Factor (I-TEF), Method of Risk Assessment for Assessment of Complex Mixtures of Dioxins and Related Compounds.* Report No. 178.

Newsted, J.L., J.P. Giesy, G.T. Ankley, D.E. Tillitt, R.A. Crawford, J.W. Gooch, P.D. Jones, and M.S. Denison. 1995. Development of toxic equivalency factors for PCB congeners and the assessment of TCDD and PCB mixtures in rainbow trout. *Environmental Toxicology and Chemistry* 14(5):861-871

Ott, W.R. 1995. *Environmental Statistics and Data Analysis.* Lewis Publishers, Boca Raton, FL.

Pack, S. 1993. *A Review of Statistical Data Analysis and Experimental Design in OECD Aquatic Toxicology Test Guidelines*. Organization of Economic Cooperation and Development. Paris, France.

Safe, S. 1990. Polychlorinated biphenyls (PCBs) dibenzo-p-dioxins (PCDDs), dibenzofurans (PCDFs), and related compounds: environmental and mechanistic considerations which support the development of toxic equivalence factors (TEFs). *Critical Reviews of Toxicology* 21:51-88.

Safe, S.H. 1994. Polychlorinated biphenyls (PCBs): Environmental impact, biochemical and toxic responses, and implications for risk assessment. *Critical Reviews in Toxicology* 24(2):87-149.

Schulz, T.W. and S. Griffin. 1999. Estimating risk assessment exposure point concentrations when the data are not normal or lognormal. *Risk Analysis* 19:577-584.

Seiler, F.A. and J.L. Alvarez. 1996. On the selection of distributions for stochastic variables. *Risk Analysis* 16:5-18.

Singh, A.K., A. Singh, and M. Engelhardt. 1997. The lognormal distribution in environmental applications. EPA/600/R-97/006. U.S. Environmental Protection Agency, Washington, DC.

Stephan, C.E. and J.W. Rogers. 1985. Advantages of using regression analysis to calculate results of chronic toxicity tests. In *Aquatic Toxicology and Hazard Assessment*. R.C. Bahner and D.J. Hansen, Editors. STP 737. American Society for Testing and Materials, Philadelphia, PA. pp. 377-387.

Suter, G.W., II. 1996. Toxicological benchmarks for screening contaminants of potential concern for effects on freshwater biota. *Environmental Toxicology and Chemistry* 15(7):1232-1241.

United States Congress. 1973. Endangered Species Act of 1973 (87 Statute 884). Washington, DC.

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

Wentsel, R.S., M. Simini, R.T. Checkai, D. Ludwig, and T.W. La Point. 1997. Ecological Risk Assessment Procedures for U.S. Army sites. *Environmental Toxicology and Risk Assessment: Modelling and Risk Assessment* (Sixth Volume), ASTM STP 1317, F.J. Dwyer, T.R. Doane and M.L. Hinman, Eds, American Society for Testing and Materials, West Conshohocken, PA.

7. ASSESSMENT ENDPOINT – SURVIVAL, GROWTH, AND REPRODUCTION OF INSECTIVOROUS BIRDS

Highlights

Conceptual Model

The assessment endpoint is the survival, growth, and reproduction of insectivorous birds in the Housatonic River PSA. Tree swallow, American robin, and wood duck were selected as the representative species.

Exposure

Exposure was determined from: (1) concentrations of contaminants of concern (COCs) found in prey, (2) an estimation of the daily intake of COCs from consumption of prey, and (3) for tree swallows and wood ducks only, tissue concentrations in nestlings and egg tissue, respectively. Site-specific studies were conducted to evaluate adverse effects from COCs.

Effects

Field data on PCB toxicity to tree swallows in the PSA were available. No data were available on the toxicity of TEQ to tree swallows or American robins, but a field-based effect study was available on the toxicity of TEQ to wood ducks. Surrogate species were used to estimate effects to American robins and wood ducks exposed to tPCBs and to tree swallows and American robins exposed to TEQ.

Risk

Modeled exposure and effects for tree swallows, American robins, and wood ducks suggest that they are at intermediate to high risk as a result of exposure to tPCBs and TEQ in the Housatonic River PSA. However, for tree swallows and American robins, the more highly weighted field study line of evidence suggests that if effects are occurring, they are minor for both species. There is no field study line of evidence for wood duck. Therefore, the weight of evidence (WOE) assessment suggests a finding of low risk for insectivorous birds such as tree swallows and American robin exposed to tPCBs and TEQ in the PSA. The WOE assessment for wood duck indicates that wood duck and similar species (e.g., hooded merganser) may be at high risk in the PSA due to tPCB and TEQ exposure.

7.1 INTRODUCTION

The purpose of this section is to characterize and quantify the current and potential risks posed to insectivorous birds, focusing on total PCBs (tPCBs) and other COPCs.

A Pre-ERA (Appendix B) was conducted to identify contaminants, other than tPCBs, that pose potential risks to wildlife in the PSA. A three-tiered deterministic approach was used to compare conservative estimates of exposure with conservative adverse effects benchmarks to identify COPCs for insectivorous birds. The COPCs that screened through to the probabilistic risk assessment for insectivorous birds were tPCBs and 2,3,7,8-TCDD toxic equivalence (TEQ); both were evaluated as contaminants of concern (COCs) for this assessment endpoint.

A step-wise approach was used to assess the risks of tPCBs and TEQ to insectivorous birds in the Housatonic River watershed. The four main steps in this process include:

1. Derivation of a conceptual model (Figure 7.1-1).
2. Assessment of exposure of birds to COCs (Figure 7.1-2).
3. Assessment of the effects of COCs on birds (Figure 7.1-3).
4. Characterization of risks to the insectivorous bird species (Figure 7.1-4).

> This section provides a summary of the ecological risk assessment for insectivorous birds, which is presented in detail in Appendix G.

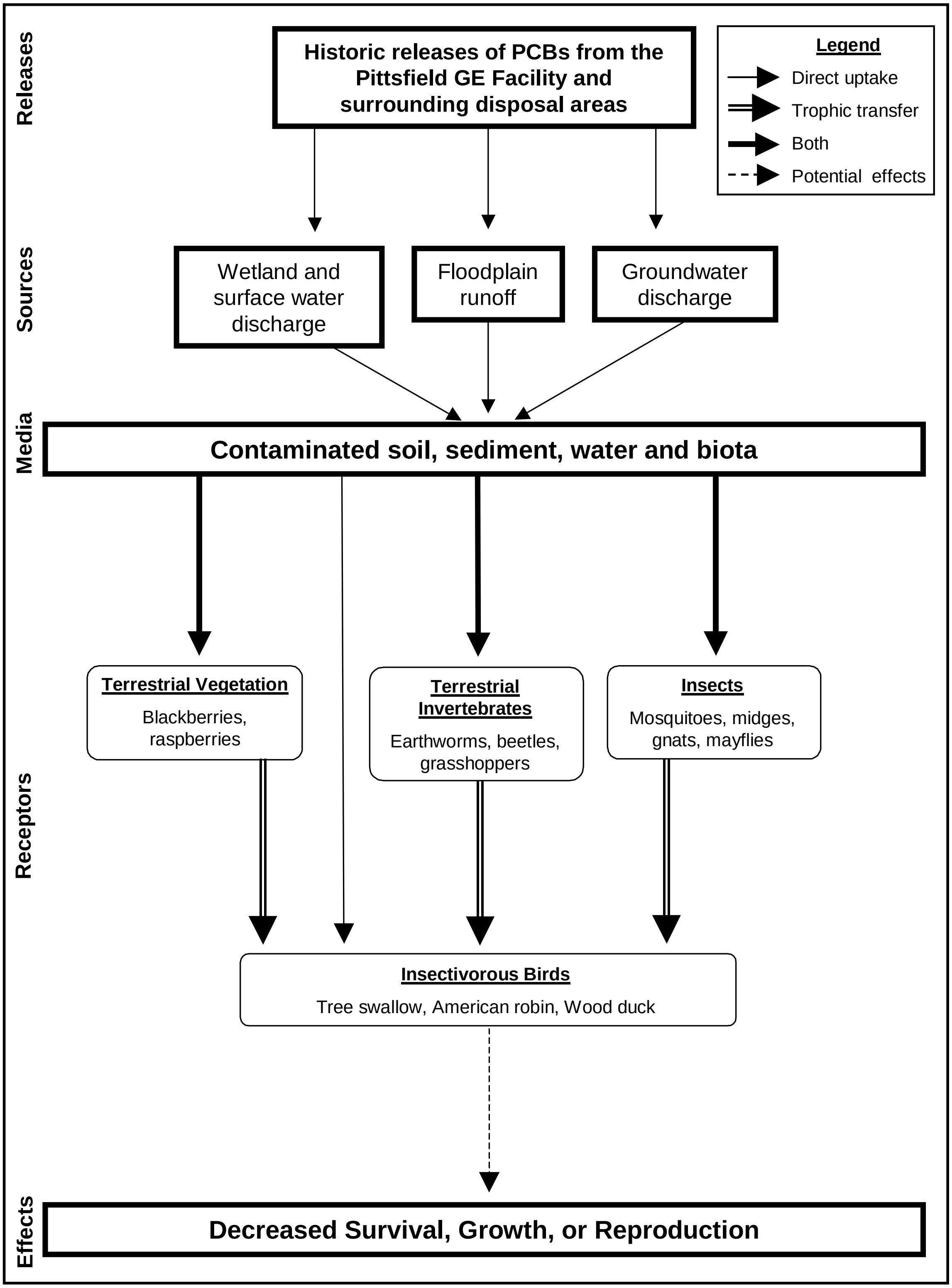


Figure 7.1-1 Conceptual Model Diagram: Exposure Pathways for Insectivorous Birds Exposed to Contaminants of Concern in the Housatonic River PSA

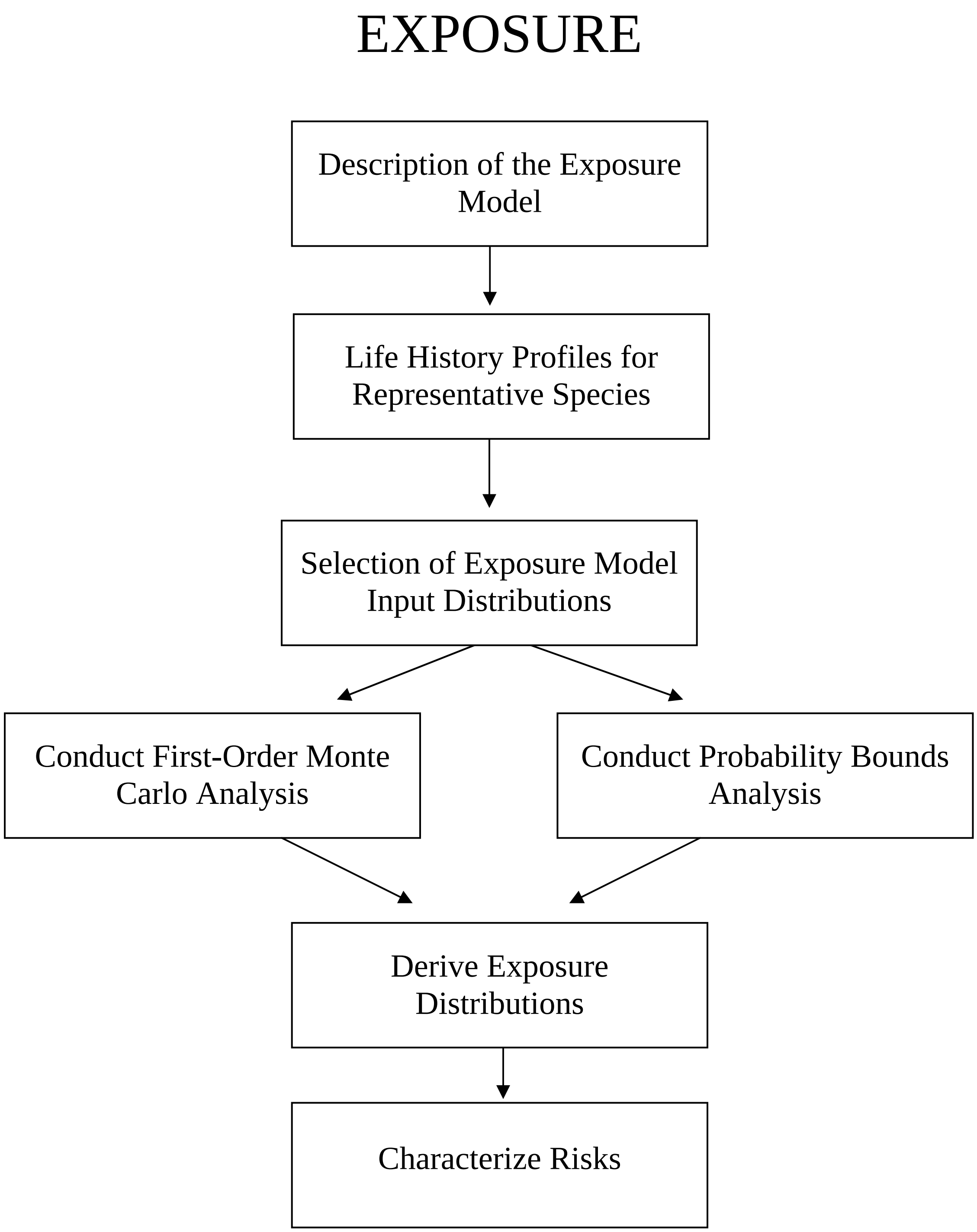


Figure 7.1-2 Overview of Approach Used to Assess Modeled Exposure of Insectivorous Birds to Contaminants of Concern in the Housatonic River PSA

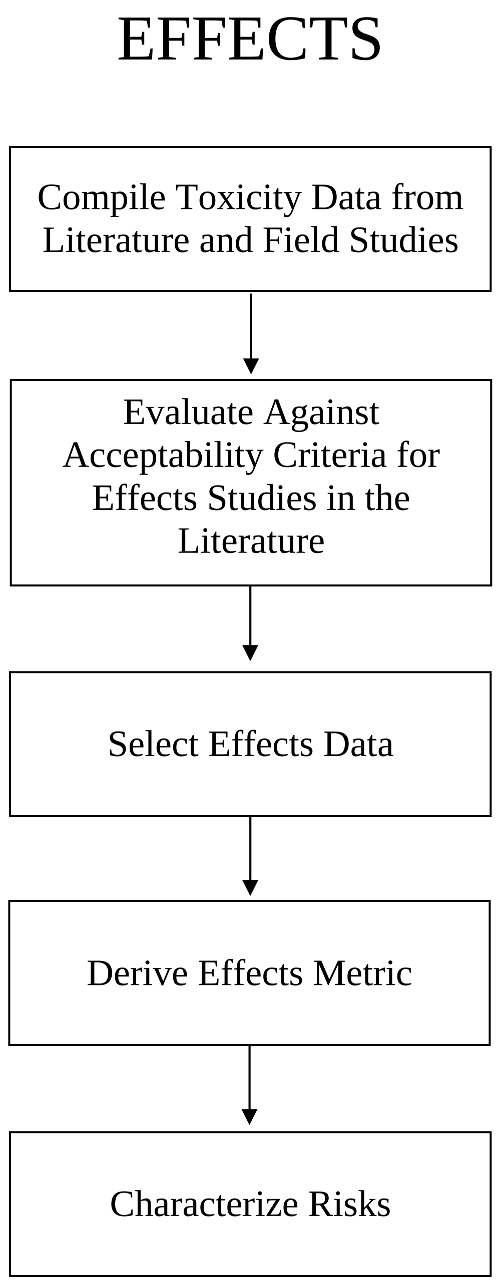


Figure 7.1-3 Overview of Approach Used to Assess the Modeled Effects of Contaminants of Concern to Insectivorous Birds in the Housatonic River PSA

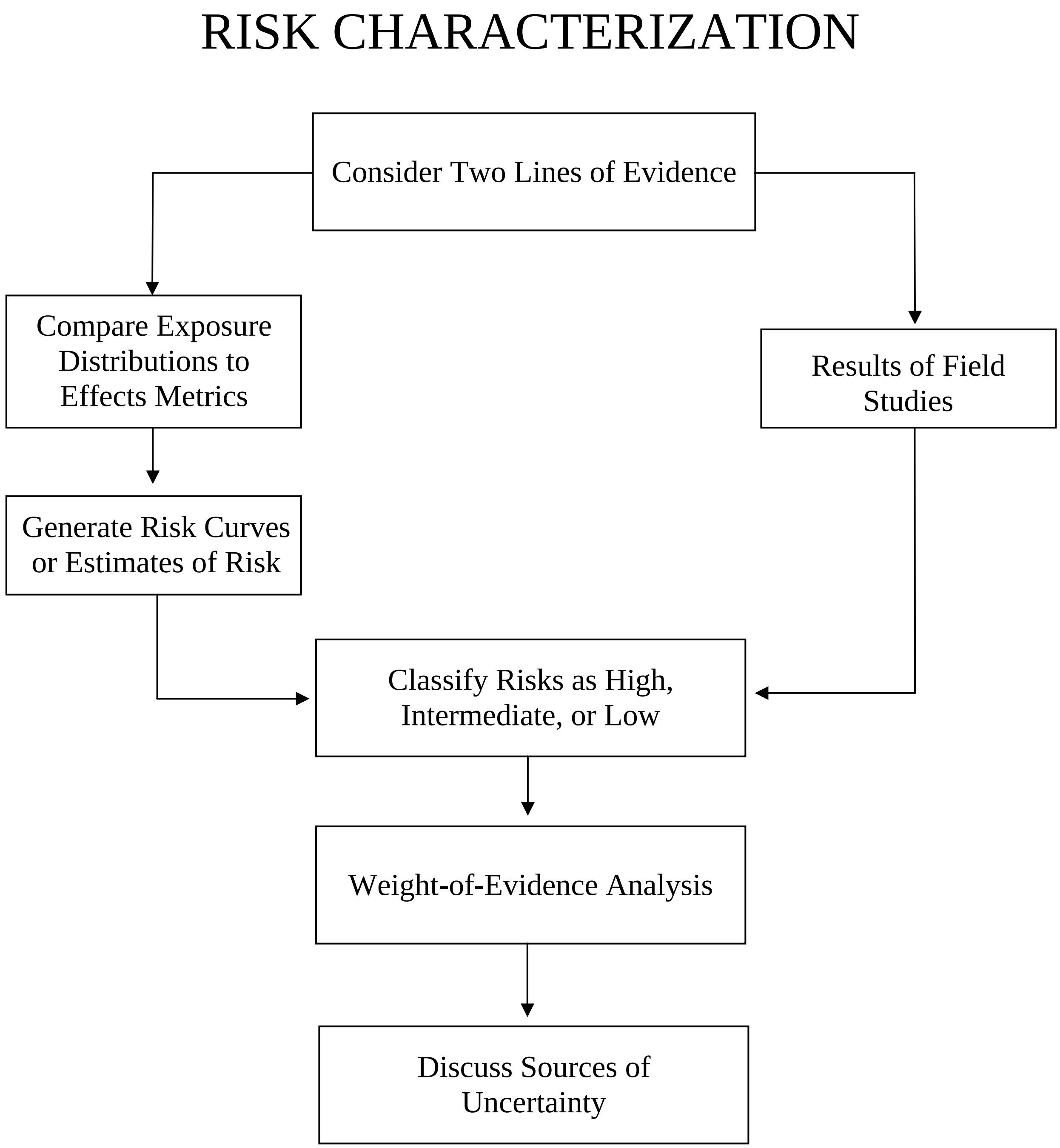


Figure 7.1-4 Overview of Approach Used to Characterize the Risks of Contaminants of Concern to Insectivorous Birds in the Housatonic River PSA

7.2 CONCEPTUAL MODEL

The conceptual model presented in Figure 7.1-1 illustrates the exposure pathways for insectivorous birds. Insectivorous birds that reside, or partially reside, within the study area are exposed to tPCBs and TEQ principally through diet and maternal transfer. Other routes of exposure, considered to be less important to overall exposure, include inhalation, water consumption, and soil ingestion (Moore et al. 1999).

The problem formulation (see Section 2) identified the tree swallow (*Tachycineta bicolor*), American robin (*Turdus migratorius*), and wood duck (*Aix sponsa*) as the representative species for insectivorous birds. Life history information for both species can be found in Sections G.2.1.3 and G.2.1.

The assessment endpoint that is the subject of this section is the survival, growth, and reproduction of insectivorous birds in the Housatonic River PSA. The measurement endpoints used to evaluate the assessment endpoint include: (1) determining, by comparisons of modeled exposure to doses reported in the literature to cause adverse effects, the extent to which the concentrations of tPCBs and TEQ ingested in the diet will cause adverse effects to the survival, growth, and reproduction of insectivorous birds; and (2) determining, by conducting field studies, the relationship between concentrations of tPCBs and TEQ in diet and the reproductive performance of insectivorous birds in the Housatonic River floodplain.

7.3 EXPOSURE ASSESSMENT

Exposure of tree swallows, American robins, and wood ducks to tPCBs and TEQ was estimated using the standard total daily intake model adapted from the *Wildlife Exposure Factors Handbook* (EPA 1993). Tree swallow exposure was also estimated using an explicit microexposure model.

The exposure assessment focused on the six locations used in the field study for tree swallows conducted by Custer (2002). The tree swallow field study was performed with nest boxes placed at six locations, three of which were located downstream of the GE facility within the PSA (Holmes Road, New Lenox Road, and Roaring Brook). Three other locations (Threemile Pond, Southwest Branch, and Taconic Valley) were expected to serve as reference locations (Custer 2002). A map of the tree swallow nest box locations is shown in Figure 7.3-1.

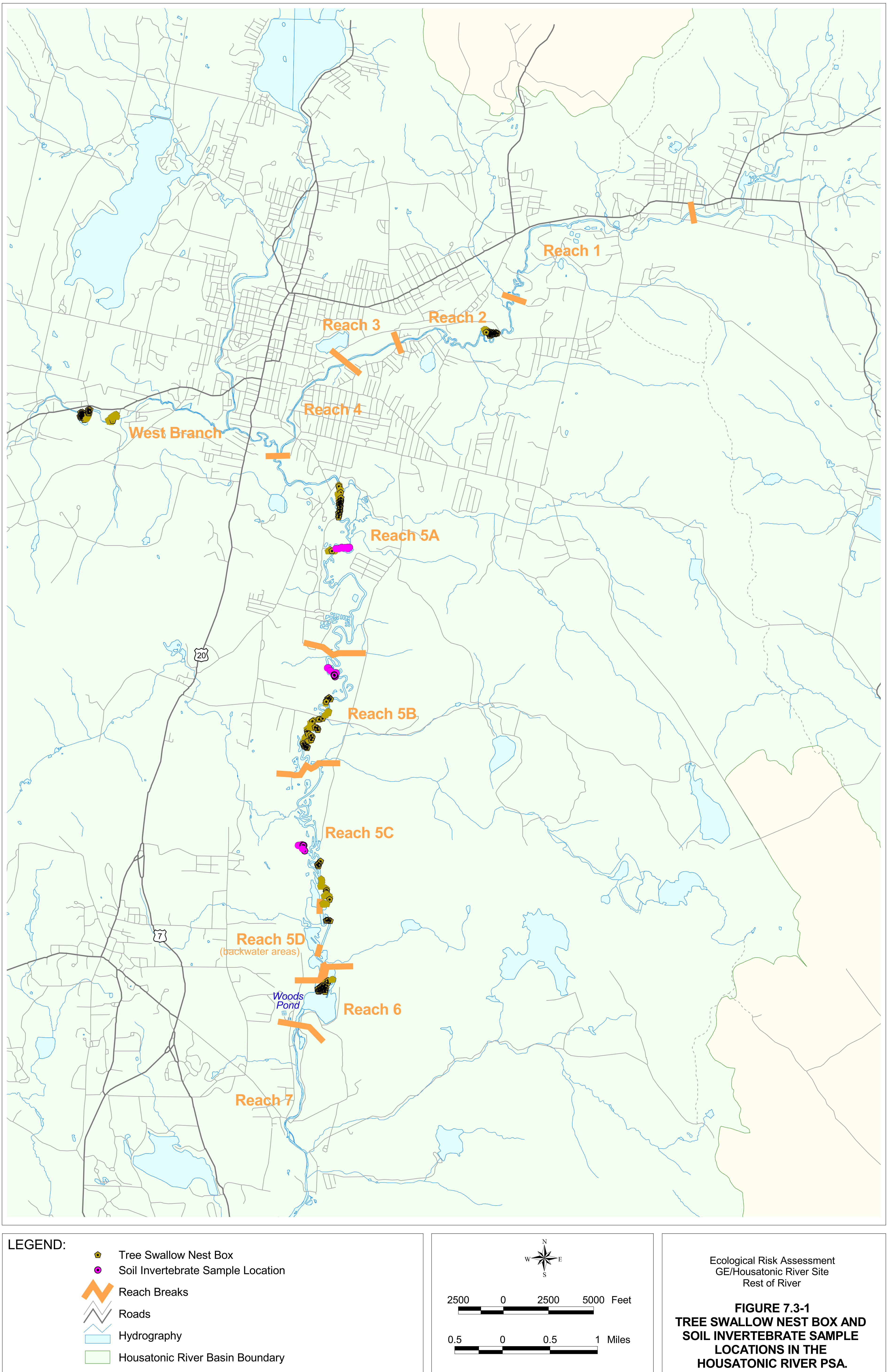
Exposure of American robins and wood ducks was estimated for three areas (Locations 13, 14, and 15; see Figure 7.3-1 and text box) in the PSA that were sampled for terrestrial invertebrates by EPA (WESTON 2002). An exposure assessment was not conducted for American robins or wood ducks in reference areas because concentrations of COCs in their prey items were not available for these locations.

Description of Locations 13, 14, and 15

Location 13 is a relatively flat area on the west shore of the river, adjacent to river mile 133, at an elevation of 965 ft (294 m). The community type is transitional floodplain forest that is flooded seasonally and is moderately well drained, with extensive vegetation cover (80%) and alluvial silt-loam soil. The PCB concentrations in floodplain soil averaged 55.2 mg/kg.

Location 14 is a relatively flat low-lying area located on the west shore of the river, adjacent to river mile 130, at an elevation of 965 ft (294 m). The community type is transitional floodplain forest that is flooded seasonally with extensive vegetation cover (70%) and fluvial silt soil. The PCB concentrations in floodplain soil averaged 26.1 mg/kg dw.

Location 15 is a flat area located on the west shore of the river, adjacent to river mile 127, at an elevation of 965 ft (294 m). Community types are circumneutral hardwood swamp and transitional floodplain forest that are flooded seasonally. This site has 60% vegetation cover, 40% leaf litter cover, and a primarily mineral soil. The PCB concentrations in floodplain soil averaged 0.484 mg/kg dw.



Ecological Risk Assessment
GE/Housatonic River Site
Rest of River

FIGURE 7.3-1
TREE SWALLOW NEST BOX AND SOIL INVERTEBRATE SAMPLE LOCATIONS IN THE HOUSATONIC RIVER PSA.

| o:\gepitt\aprs\swallows.apr | Layout - swallow/soilinvert locs 7.3-1 | o:\gepitt\plots\in\tsnb_sis_locs_7-3-1.eps | 10:21 AM, 7/3/2003 |

7.3.1 Exposure Models for Insectivorous Birds

7.3.1.1 Total Daily Intake Model

Exposure of the representative species, tree swallows, American robins, and wood duck to tPCBs and TEQ was estimated using a total daily intake model adapted from the *Wildlife Exposure Factors Handbook* (EPA 1993) and related publications. The model used in the exposure analysis was:

$$TDI = FT \times FIR \sum_{i=1}^{n} C_i \times P_i \qquad \text{(Eq. 1)}$$

where

- TDI = Total daily intake (mg/kg bw/d tPCBs, ng/kg bw/d TEQ).
- FT = Foraging time in the area of interest (unitless and set equal to one for insectivorous birds).
- FIR = Normalized food intake rate (kg/kg bw/d).
- P_i = Proportion of the *i*th dietary item (unitless).
- C_i = Concentration in food item (mg/kg for tPCBs, ng/kg for TEQ).

The model considered the food intake rates of representative species (*FIR*), the concentrations of COCs in each food item (C_i), and the proportion of diet accounted for by that food item (P_i). For those input variables that were uncertain, variable, or both, distributions were used rather than point estimates. Monte Carlo and probability bounds analyses were used to propagate uncertainties about input variables through the exposure model for each COC. A description of these techniques and the methods used to parameterize input variables is presented in Section 6.5 and Appendix C.4. The results of the Monte Carlo and probability bounds analyses are discussed in detail in Appendix G.

7.3.1.2 Microexposure Model

A microexposure model was used to determine the whole body contaminant content in the swallows as a function of time over their first 15 days of development, the time during which swallows reach adult body weight. In addition, nestling swallow tissue samples were collected between 12 and 14 days in the Custer (2002) study. The construction of the microexposure model is described in detail in Section G.2.1.8.2 of Appendix G.

7.3.1.3 Exposure Model for TEQ in Wood Duck Eggs

To model the concentration of 2,3,7,8-TCDD TEQ in wood duck eggs, the TDI model described above was applied to generate an estimated total daily intake in breeding adult female wood ducks. Prelaying female wood ducks in the PSA are likely to acquire a significant TEQ body burden by consuming contaminated prey for the 14 days between arrival in the PSA and egg laying. Wood ducks are known to lay approximately 1 egg per day and have an average brood size of 13 eggs (Grice and Rogers 1965). The concentration of TEQ mass (ng/kg) in female breeding wood duck over the 14-day prelaying period was estimated by multiplying the TDI (ng/kg bw/d) by a chemical absorption efficiency (CAE) (unitless) and summing the daily results for 14 days. The mean CAE for PCBs in avian species ranges from 0.80 to 0.97. Not all of the TEQ in adult females is subsequently transferred to the wood duck eggs. A portion of the TEQ mass will remain with the adult wood duck. The concentration of TEQ in the adult female breeding wood duck was thus multiplied by an egg:adult ratio ($CR_{e:a}$) to determine the concentration of TEQ (ng/kg) in the egg. Therefore, over a 13-day period, wood duck females will excrete TEQ congeners from their body, into their eggs. At the same time, they are adding to their body burden by foraging on contaminated prey in the PSA. To account for this, the egg model was run once per day for 13 days, and the amount of TEQ excreted by the wood duck female into the egg was subtracted from the total female body burden. Wood duck eggs have variable mass. To determine the TEQ mass excreted in each individual wood duck egg, the concentration in the egg was multiplied by the egg mass (kg). For this assessment, egg TEQ was estimated for Day 14 (first egg), Day 20 and Day 27 (last egg). Parameter distributions used in this exposure analyses are described in detail in Appendix G.

7.3.1.4 Description of Model Parameters

7.3.1.4.1 Tree Swallows

Foraging Time

The foraging range of the tree swallow is within the area of the PSA. Prey availability and abundance suggest that tree swallows are able to meet their needs exclusively within the PSA. Therefore, tree swallows were assumed to spend 100% of their foraging time within the PSA.

Body Weight (BW)

TDI Model

Tree swallows are small birds with an average adult body weight of about 20 g. Based on data cited in Dunning (1984), the mean adult body weight was estimated to be 20.1 g with standard deviation (std. dev.) of 1.58.

Microexposure Model

The body weight of juvenile birds was modeled as a function of time, reaching the adult weight at 12 to 14 days, using a logistic model from Teather (1996). In the Monte Carlo version of the microexposure model, point values were used for body weight. In the probability bounds analysis, the uncertainty surrounding body weight as a function of time was taken into account. Tables G.2-4 and G.2-5 show the body weight distributions used in the Monte Carlo and probability bounds analyses, respectively.

Food Intake Rate (FIR)

TDI Model

The food intake rate of tree swallows has not been well characterized. As a result, an allometric modeling approach, described in Appendix G, was used to estimate food intake rate for tree swallows.

Microexposure Model

The allometric relationship for food intake rate was modeled as a function of body weight. This was recalculated as the microexposure model was stepped forward in time over the life of the swallow. For each day, the corresponding point estimate of body weight (in the case of the Monte Carlo analysis) or interval estimate of body weight (in the case of the probability bounds analysis) was substituted into the allometric equation for food intake rate.

Proportion of Dietary Items (P_i)

Consumption of contaminated aquatic insects is presumed to be the primary route of exposure of swallows to tPCBs. An analysis of the diet delivered to swallow nestlings indicated that it consisted of Diptera (41.8%), mayflies (21.3%), and moths and butterflies (9.2%) by total dry mass (Blancher and McNicol 1991). A separate study also showed that mayflies and Diptera were common prey for swallows (Robertson et al. 1992). Stomach content samples were taken from birds at nest sites within the Housatonic River PSA and at reference sites (Custer 2002). These samples were used as the primary source of contaminant input concentrations for the exposure models developed in this assessment.

Maternal Transfer (TOT_MT)

Maternal transfer refers to the total amount of COCs transferred from the mother to the offspring via egg tissue. For tree swallows, maternal transfer was estimated at each site using the concentrations of COCs in the tissues of the newborn swallows (pippers) and egg samples. Table G.2-6 summarizes the ratios of the means of tPCB concentrations in eggs and pippers (due to maternal transfer only) to the tPCB concentrations in nestlings (aged 12 to 14 days). Low ratios indicate that the majority of tPCB tissue concentrations originated from feeding activity at the site. Ratios approaching or greater than one indicate that the tPCB content was primarily due to maternal transfer, and not from feeding locally over the period from birth to 14 days.[1] High ratios of tPCB concentrations in pippers and eggs to total concentrations in nestlings occurred at sites expected to be relatively uncontaminated locations.

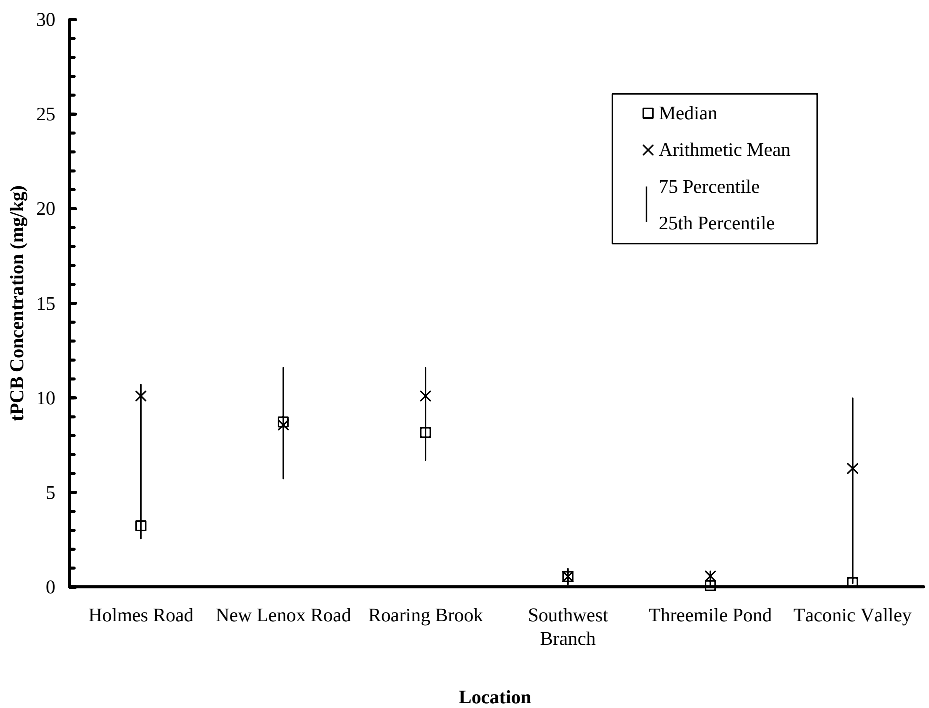
---

[1] Ratios greater than one, observed at Southwest Branch in 1998, Threemile Pond in 1999 and 2000, and Taconic Valley in 2000, suggest depuration or growth dilution over the 14 days of growth. These ratios are considered indicative of very high proportions of maternal transfer relative to intake of local contaminants.

For the tree swallow, maternal transfer was assumed to have a lognormal distribution, with location-specific mean and standard deviation. For the probability bounds analysis, probability bounds were derived using the site-specific lower 95% confidence limit (LCL) and the upper 95% confidence limit (UCL) around the means calculated using the Land H-statistic.

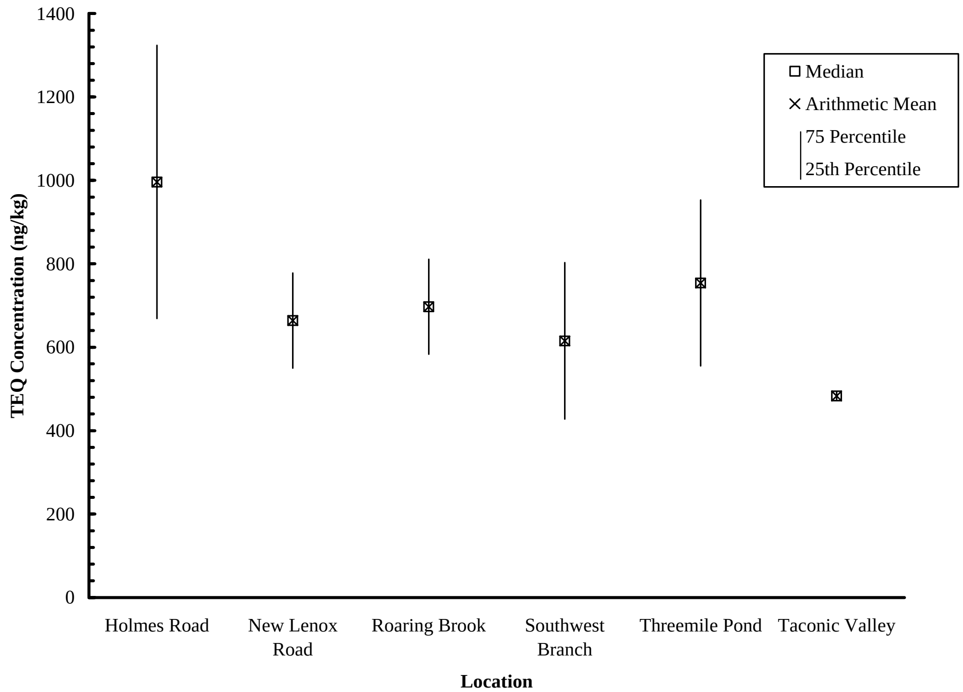
Concentrations of COCs in Food Items (C_f)

The concentrations of tPCBs and TEQ in the dietary items of tree swallows are illustrated in Figure 7.3-2 and Figure 7.3-3. The concentrations of COCs used in the exposure analyses are shown in Tables G.2-7 and G.2-27. Total PCB concentrations in the prey of tree swallows are similar at Holmes Road, New Lenox Road, and Roaring Brook. All of the other locations have lower tPCB concentrations than sites in the PSA. TEQ concentrations in prey items of tree swallows are highest at Holmes Road and lowest at the Taconic Valley.



Note: Prey concentrations are wet weight.

Figure 7.3-2 Concentration of tPCBs in Prey of Tree Swallows



Note: Prey concentrations are wet weight.

Figure 7.3-3 Concentration of TEQ in Prey of Tree Swallows

Stomach content samples were collected in 1998, 1999, and 2000 from swallows at nest sites within the Housatonic River PSA and at other sites (Custer 2002). Median tPCB concentrations in stomach contents at Holmes Road, New Lenox Road, and Roaring Brook were approximately one order of magnitude higher than at other locations, except for Taconic Valley, which had concentrations comparable to the PSA locations. Median TEQ concentrations in stomach contents at all locations were in the same order of magnitude. Statistics for concentrations of tPCBs and TEQ in stomach contents at the six locations are shown in Tables G.2-7 and G.2-8.

7.3.1.4.2 American Robins

Foraging Time

The foraging range of the American robin is within the area of the PSA. Prey availability and abundance suggest that robins are able to meet their needs exclusively within the PSA. Therefore, American robins were assumed to spend 100% of their foraging time within the PSA.

Both tPCB and TEQ concentrations in the prey items of American robins and wood ducks are highest at Location 13 and lowest at Location 15.

Body Weight (BW)

Robins monitored in Delta Marsh, Manitoba, Canada, ranged from 72 to 86 g, with females gaining slightly more weight during the incubation period (Bierman and Sealy 1985). Clench and Leberman (1978) found an average mass of 77 g when data from both sexes were pooled. The mean adult body weight of American robins was estimated to be 79.7 g, with a std. dev. of 5.53 g.

Food Intake Rate (FIR)

The food intake rate of American robins has not been well characterized. As a result, an allometric modeling approach, described in Appendix G, was used to estimate food intake rate for American robins.

Proportion of Dietary Items (P_i)

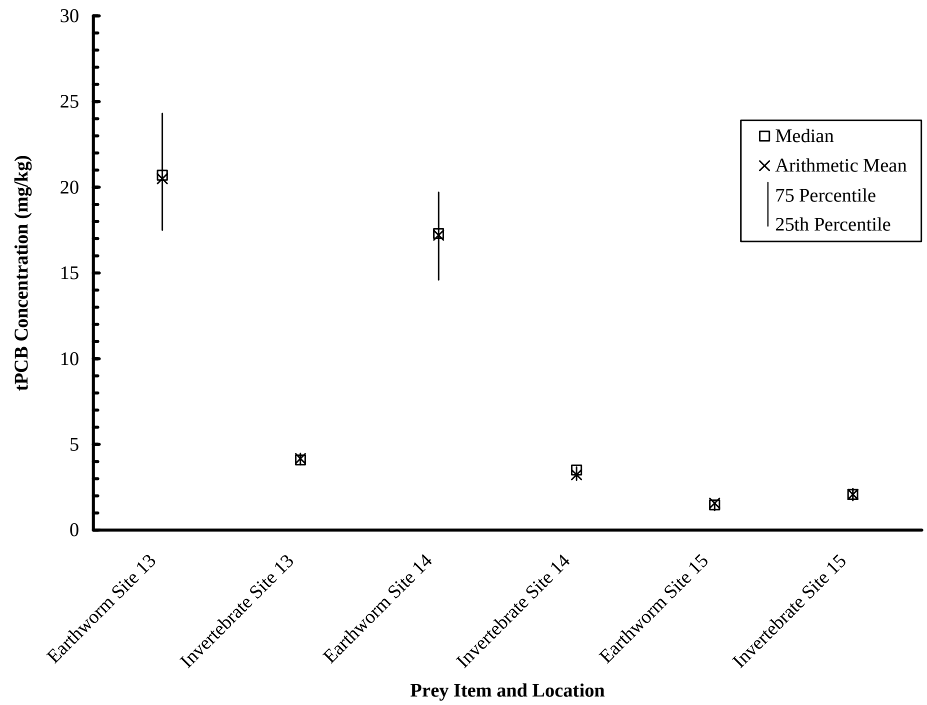
For robins, the available literature indicates that earthworms comprise about 15% of the diet on average during the spring and summer, but may range from 10 to 20%. Litter invertebrates generally comprise about 60% of the diet during spring and summer, with an approximate range of 45 to 75%. The proportion of fruit in the robin diet during spring and summer was calculated as one minus the total of earthworms and litter invertebrates.

Maternal Transfer

Maternal transfer was not considered for American robins as no effort was made to model egg concentrations of the COCs.

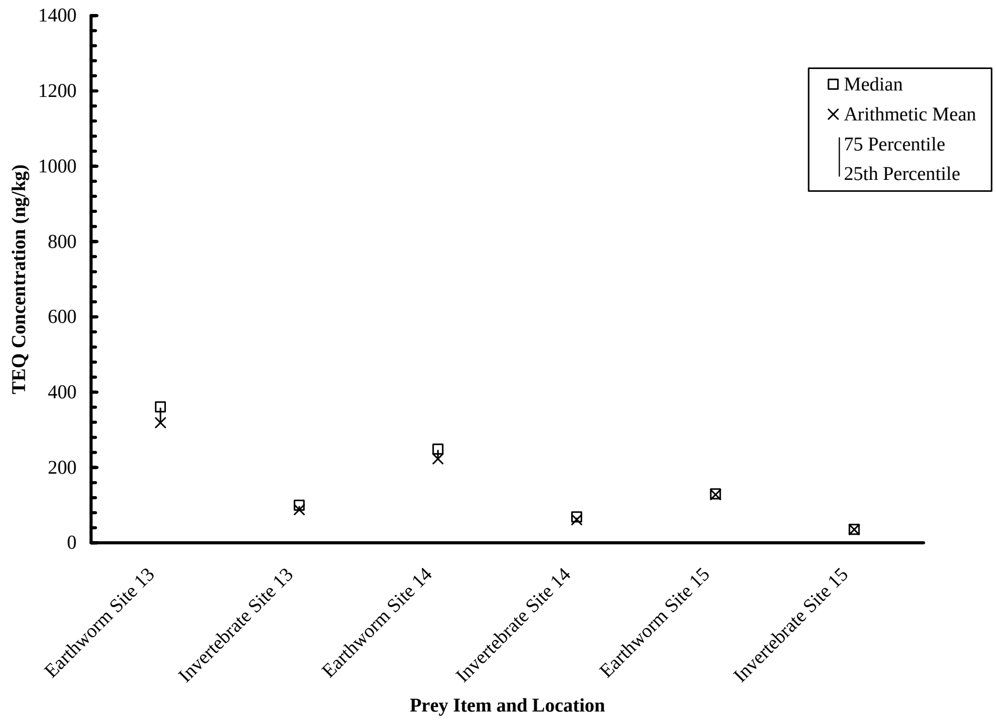
Concentrations of COCs in Food Items (C_f)

Prey items for American robin, including litter invertebrates and earthworms, were sampled in the PSA. Figures 7.3-4 and 7.3-5 indicate the range of robin prey tissue concentrations for tPCB and TEQ, respectively, sampled from the PSA. The prey with the highest concentrations of tPCB or TEQ were found at Location 13, while the lower COC concentrations were detected at Location 15.



Note: Prey concentrations are wet weight.

Figure 7.3-4 Concentration of tPCBs in Prey of American Robins



Note: Prey concentrations are wet weight.

Figure 7.3-5 Concentration of TEQ in Prey of American Robins

7.3.1.4.3 Wood Ducks

Foraging Time

The foraging range of the wood duck is within the area of the PSA. Female breeding wood ducks are known to forage within 1 km of the nest. Prey availability and abundance also suggest that wood ducks are able to meet their needs exclusively within the PSA. Therefore, wood ducks were assumed to spend 100% of their foraging time within the PSA.

Body Weight (BW)

The body weight of the wild adult female wood duck ranges from 550 to 675 g (Landers et al. 1977). Female wood ducks appear to have lowest body weight in the spring and summer. They lose weight during the breeding season due to their incorporation of body fat into egg production. Body weight data from Landers et al. (1977) for the months of March through June for female wood ducks were therefore used to calculate the mean body weight of a breeding adult female wood duck. The geometric mean weight of a breeding adult female wood duck is 564 g (n=43; std. dev. =18.07).

Food Intake Rate (FIR)

The food intake rate of the wood duck has not been well characterized. As a result, an allometric modeling approach, described in Appendix G, was used to estimate FIR for wood ducks.

Proportion of Dietary Items (Pi)

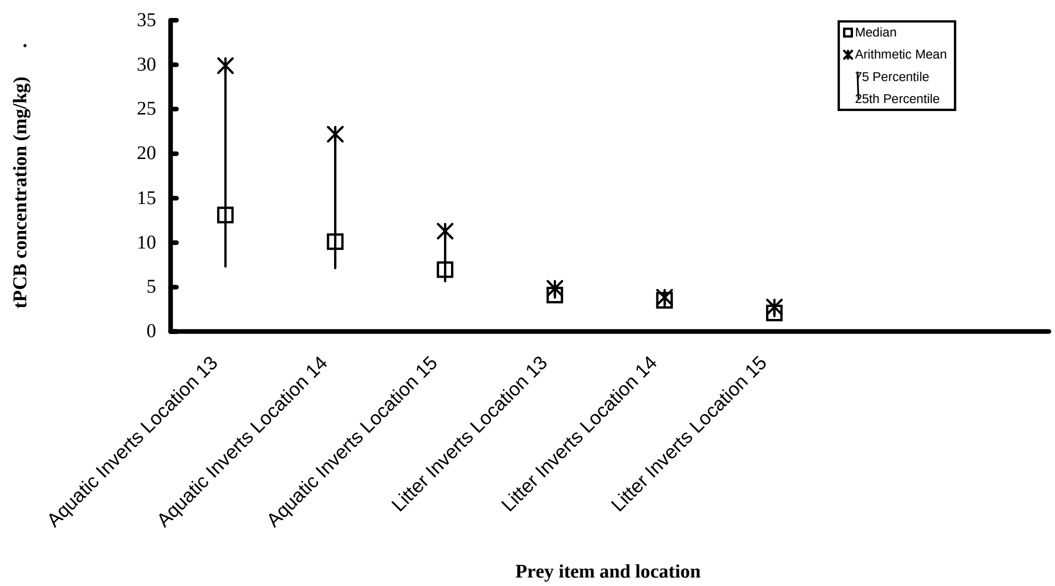
Female wood ducks are known to have a small foraging range, feed close to the nest site prior to egg laying, and substantially increase the relative percentage of invertebrates in their diet prior to and during egg laying. They are also migratory and are not found in the PSA year round. Therefore, it is anticipated that a large portion of their body burden and subsequent contaminant concentrations in eggs would be due to exposure to contaminated prey in the PSA during the breeding season. Laying female wood ducks were reported to have 76% invertebrate tissue and 22% plant tissue in their diet (Drobney and Fredrickson 1979). Of the 76% invertebrate tissue reported, 56.4% was determined to be aquatic while 19.6% was terrestrial.

Egg to Adult Concentration Ratio ($CR_{e:a}$)

For wood ducks, the potential influence of maternal body burdens on contaminant concentrations in eggs was determined by examining the ratio of chemical concentrations in the egg and adult (egg:hen) for each PCB congener and for tPCBs at each dose concentration. Bargar et al. (2001) examined the maternal transfer of three PCB congeners into eggs in the white leghorn chicken. Chicken hens were injected with PCB-189, PCB-156, and PCB-105, and the concentration in eggs was determined as a percentage of the concentration in hens. The authors determined that congener chlorination had a significant effect on excretion. The higher chlorinated PCB congener PCB-189 was detected in lower concentrations in the egg than the other two congeners (PCB-156 and PCB-105). Bargar et al. (2001) calculated a mean egg:hen concentration ratio of 0.22 and range of 0.19 to 0.26 for tPCBs in white leghorn chickens. These values are used to parameterize a triangular distribution for the egg:adult ratio ($CR_{e:a}$) used in the wood duck egg TEQ concentration model.

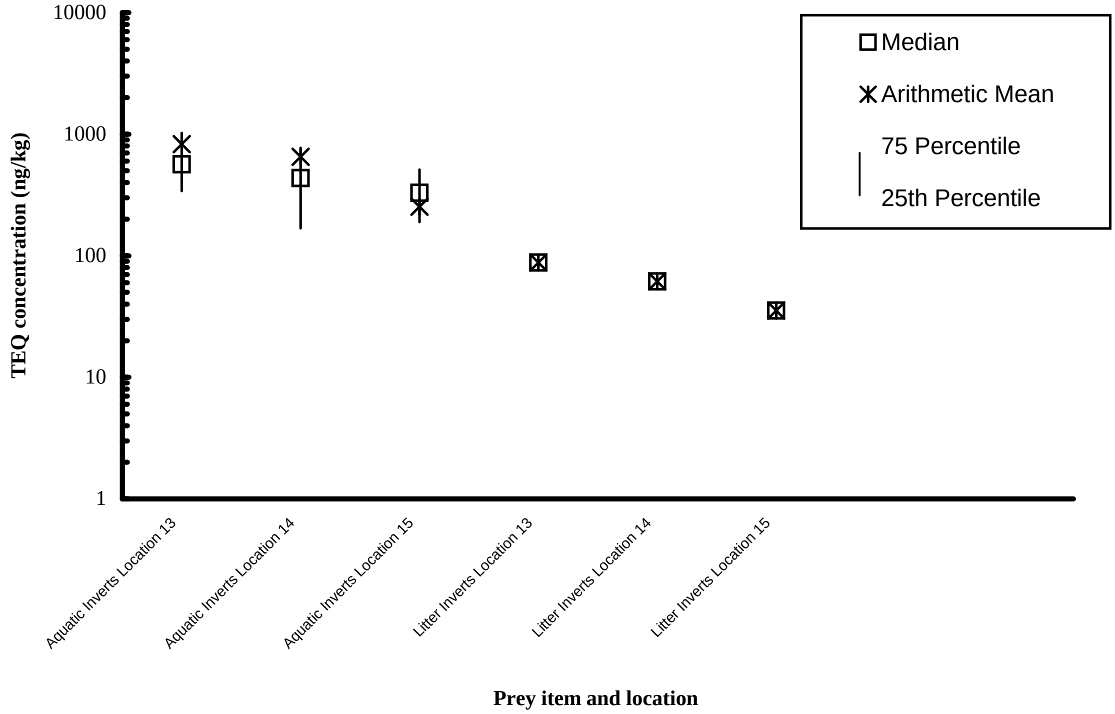
Concentrations of COCs in Food Items (C_f)

Prey items for wood ducks, including litter invertebrates and aquatic invertebrates, were sampled in the PSA at Locations 13, 14, and 15. Figures 7.3-6 and 7.3-7 indicate the range of wood duck prey tissue concentrations for tPCB and TEQ, respectively, sampled from the PSA. The prey with the highest concentrations of tPCB or TEQ were found at Location 13, while the lower COC concentrations were detected at Location 15.



Note: Prey concentrations are wet weight.

Figure 7.3-6 Concentration of tPCBs in Prey of Wood Ducks



Note: Prey concentrations are wet weight.

Figure 7.3-7 Concentration of TEQ in Prey of Wood Ducks

7.3.2 TDI Model Results

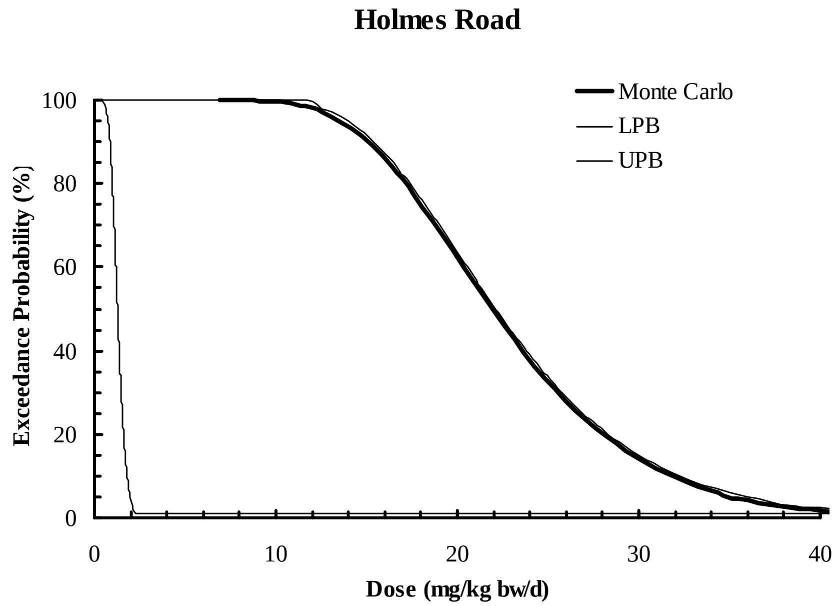
Tree Swallows

Tree swallow tPCB and TEQ exposure distributions at Holmes Road, New Lenox Road, Roaring Brook Road, Southwest Branch, Threemile Pond, and Taconic Valley are presented in Figures 7.3-8 to 7.3-19.

The Monte Carlo analysis indicated that exposure of tree swallows to tPCBs at Holmes Road (Figure 7.3-8) could range from a minimum of 6.87 to a maximum of 63.0 mg/kg bw/d. The mean exposure was 22.9 mg/kg bw/d, and the median exposure was 22.0 mg/kg bw/d; 80% of the exposure estimates were between 15.3 and 32.1 mg/kg bw/d (see Table G.2-9).

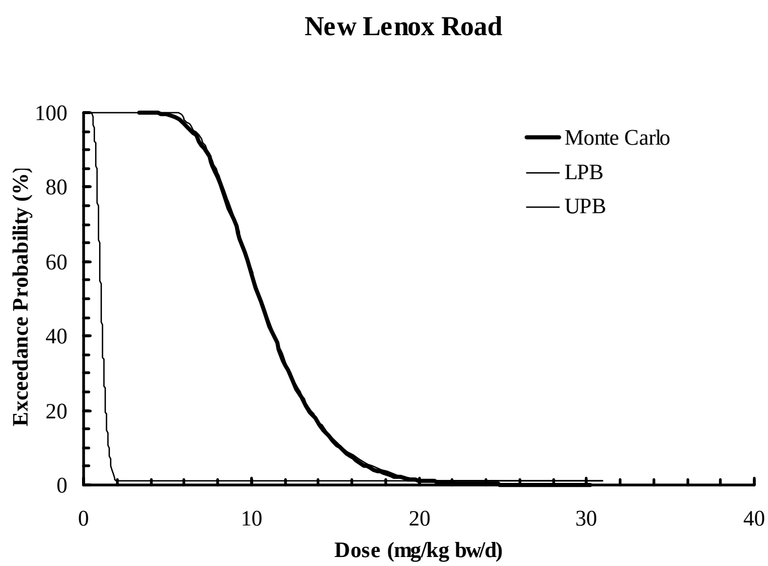
The probability bounds for tree swallows exposed at Holmes Road are also depicted in Figure 7.3-8. The 10^{th} percentile of the probability envelope formed by the lower and upper bounds ranged between 1.04 and 18.2 mg/kg bw/d. The 50^{th} percentile ranged between 1.27 and 22.1 mg/kg bw/d, and the 90^{th} percentile ranged between 1.81 and 32.3 mg/kg bw/d. In comparison, the 10^{th} percentile of the Monte Carlo output was 15.3, the 50^{th} percentile was 22.0, and the 90^{th} percentile was 32.1 mg/kg bw/d (Table G.2-9).

Exposures of tree swallows to tPCBs at the two other PSA locations, New Lenox Road and Roaring Brook, were lower than at Holmes Road, having mean total daily intakes of 11.0 and 13.3 mg/kg bw/d, respectively (Table G.2-9). Exposures at the reference locations, Southwest Branch and Threemile Pond, were very low, with mean total daily intakes of 0.73 and 1.15 mg/kg bw/d. A third location, Taconic Valley, had relatively high exposure with a mean of 14.2 mg/kg bw/d, indicating that the proximity of this location to the GE facility may invalidate its consideration as a reference. The highest concentrations of TEQ were at Holmes Road with a mean of 1,227 ng/kg bw/d. New Lenox Road and Roaring Brook had mean total daily intakes of 701 and 681 ng/kg bw/d, respectively (Table G.2-10). Mean exposure concentrations at these three locations ranged from 396 to 866 ng/kg bw/d.



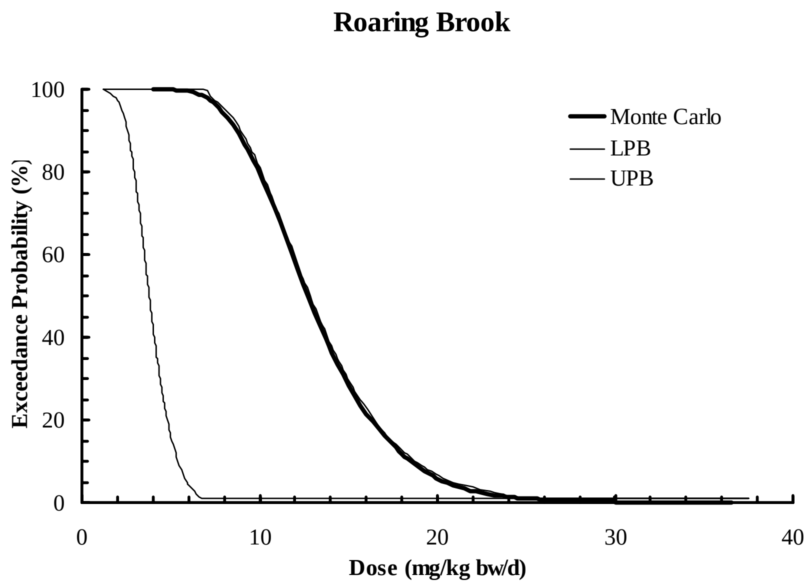
Notes:
LPB = Lower probability bound
UPB = Upper probability bound

Figure 7.3-8 Tree Swallow TDI Exposure Model for tPCBs: Results of Monte Carlo Analysis and Probability Bounds Analysis at Holmes Road



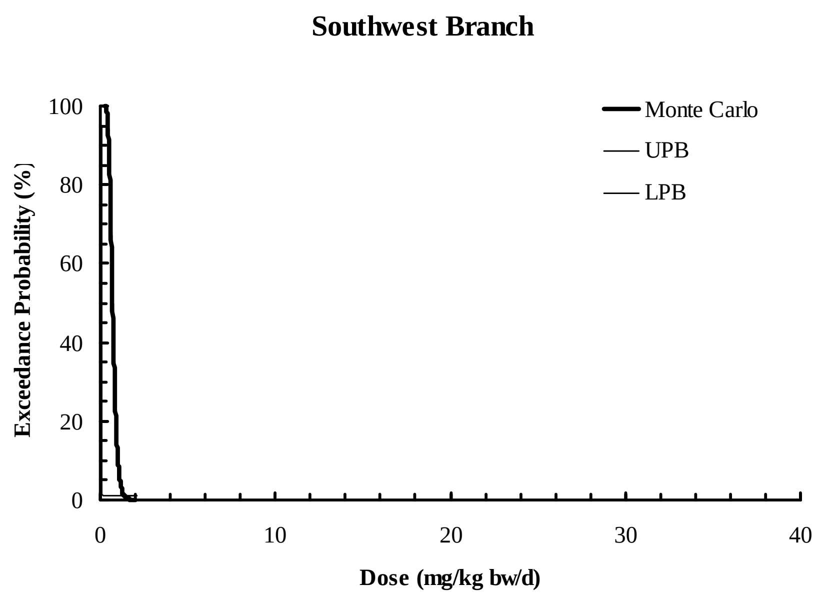
Notes:
LPB = Lower probability bound
UPB = Upper probability bound

Figure 7.3-9 Tree Swallow TDI Exposure Model for tPCBs: Results of Monte Carlo Analysis and Probability Bounds Analysis at New Lenox Road



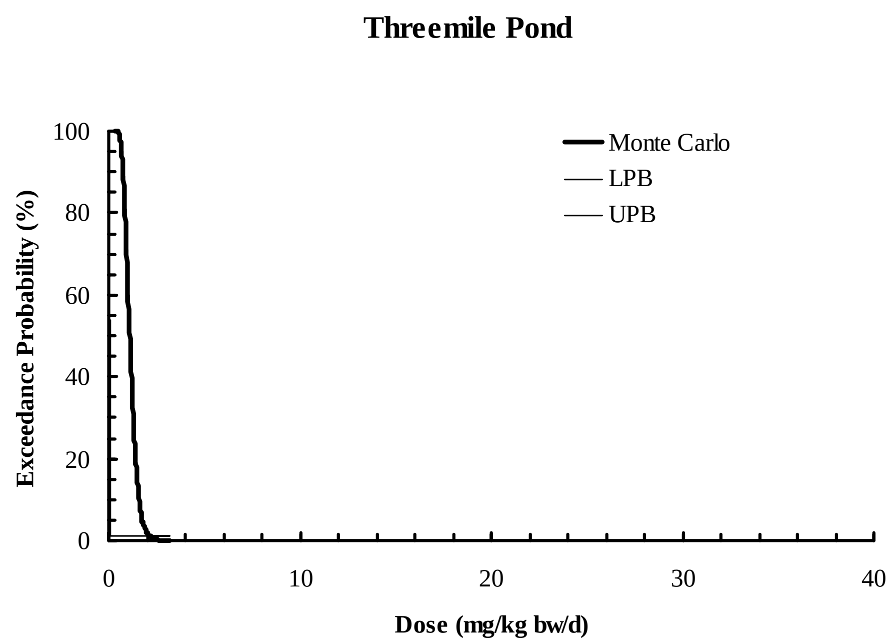
Notes:
LPB = Lower probability bound
UPB = Upper probability bound

Figure 7.3-10 Tree Swallow TDI Exposure Model for tPCBs: Results of Monte Carlo Analysis and Probability Bounds Analysis at Roaring Brook



Notes:
LPB = Lower probability bound
UPB = Upper probability bound

Figure 7.3-11 Tree Swallow TDI Exposure Model for tPCBs: Results of Monte Carlo Analysis and Probability Bounds Analysis at Southwest Branch

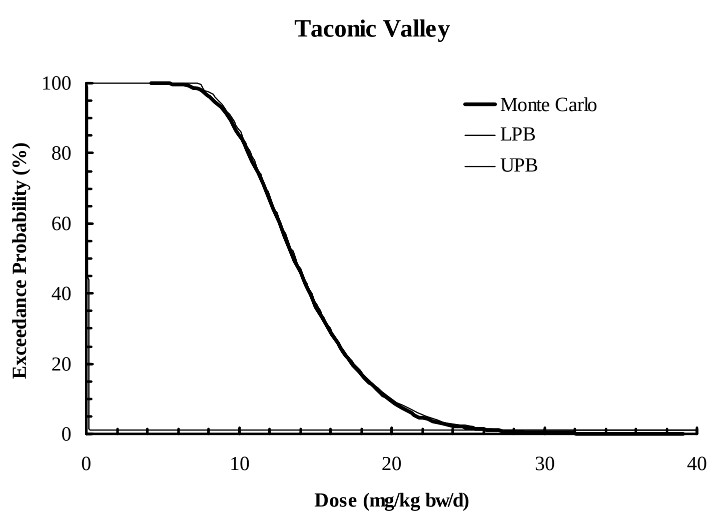


Notes:

LPB = Lower probability bound

UPB = Upper probability bound

Figure 7.3-12 Tree Swallow TDI Exposure Model for tPCBs: Results of Monte Carlo Analysis and Probability Bounds Analysis at Threemile Pond

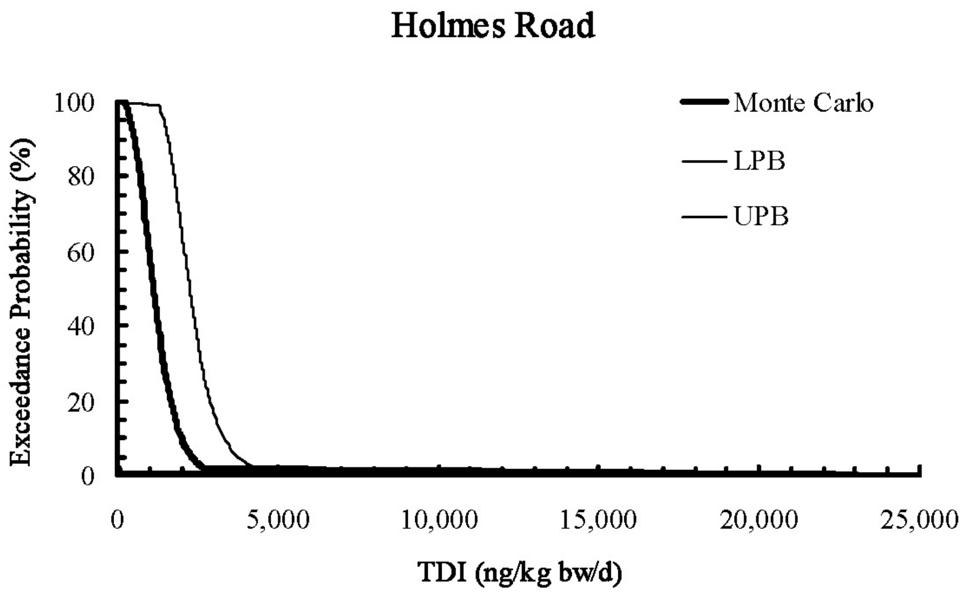


Notes:

LPB = Lower probability bound

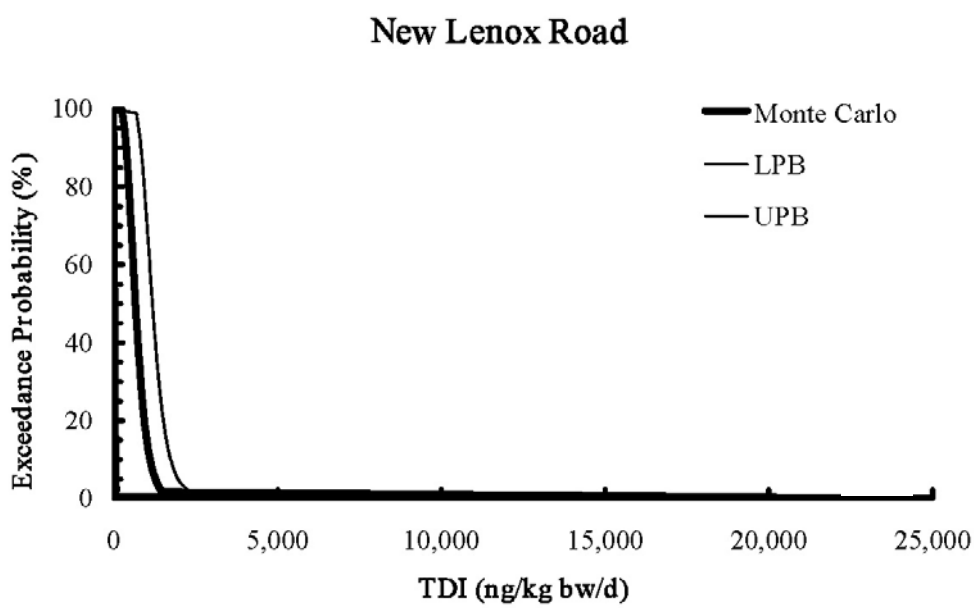
UPB = Upper probability bound

Figure 7.3-13 Tree Swallow TDI Exposure Model for tPCBs: Results of Monte Carlo Analysis and Probability Bounds Analysis at Taconic Valley



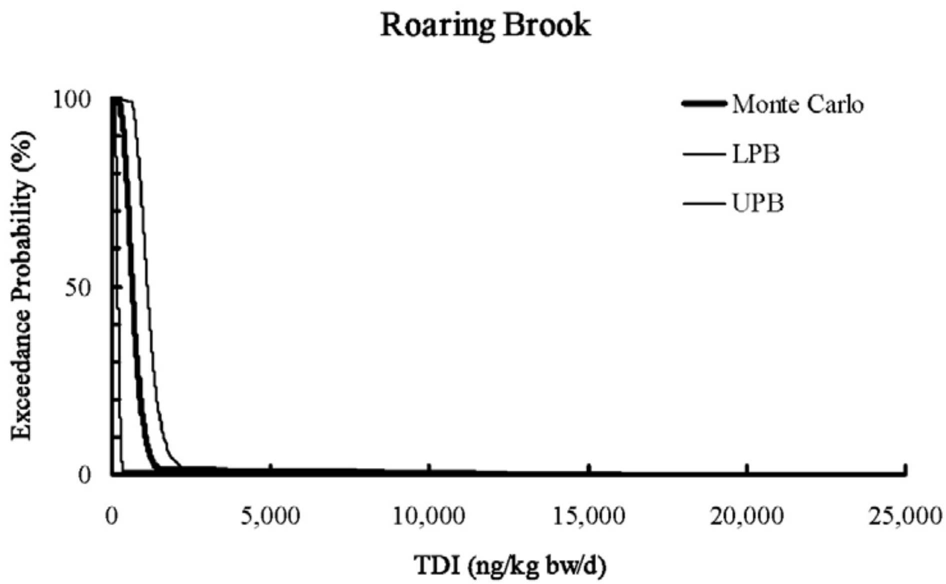
Notes:
LPB = Lower probability bound
UPB = Upper probability bound

Figure 7.3-14 Tree Swallow TDI Exposure Model for TEQ: Results of Monte Carlo Analysis and Probability Bounds Analysis at Holmes Road



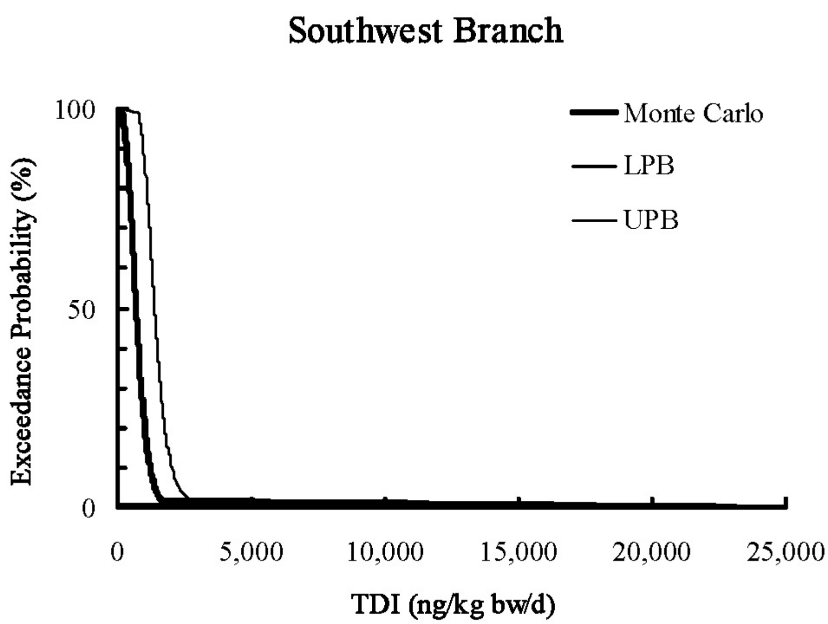
Notes:
LPB = Lower probability bound
UPB = Upper probability bound

Figure 7.3-15 Tree Swallow TDI Exposure Model for TEQ: Results of Monte Carlo Analysis and Probability Bounds Analysis at New Lenox Road



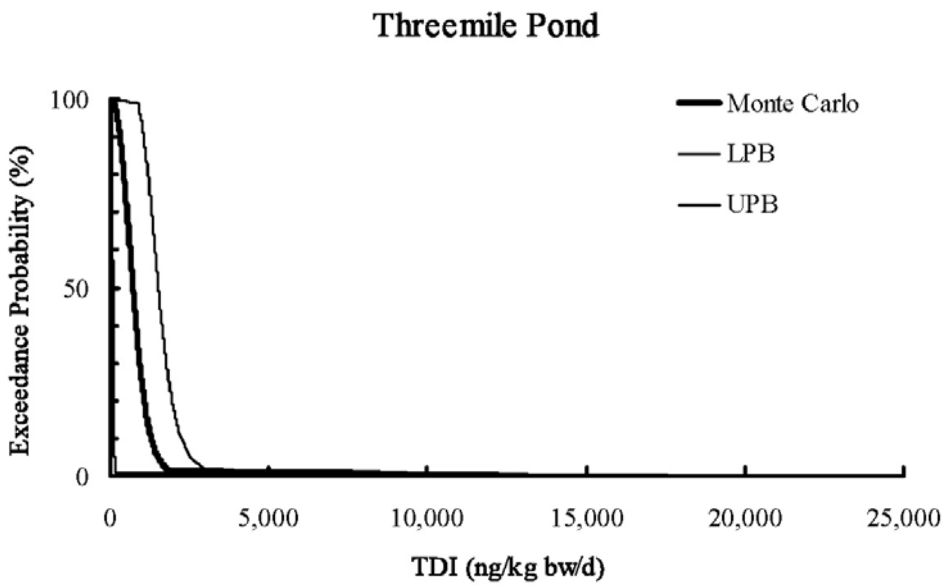
Notes:
LPB = Lower probability bound
UPB = Upper probability bound

Figure 7.3-16 Tree Swallow TDI Exposure Model for TEQ: Results of Monte Carlo Analysis and Probability Bounds Analysis at Roaring Brook



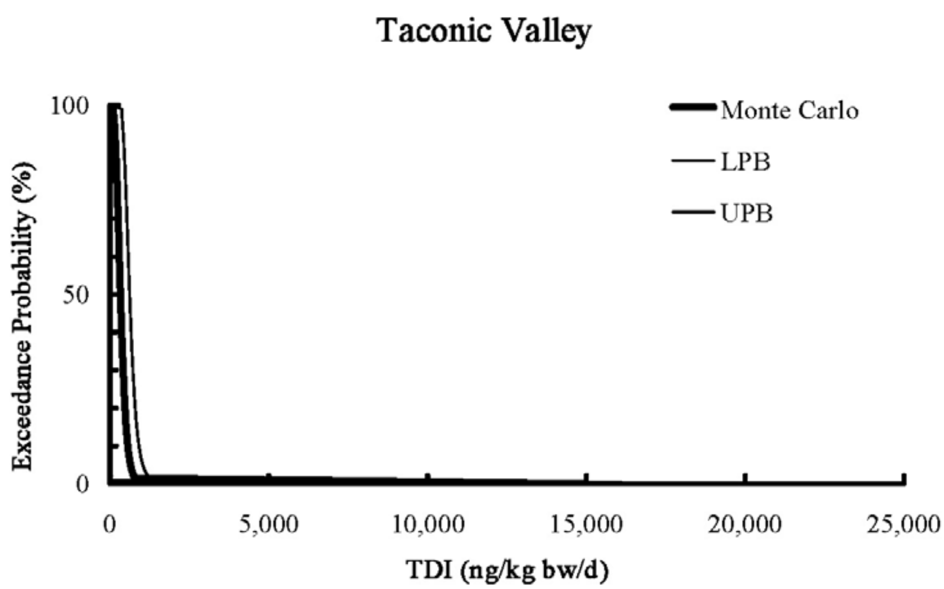
Notes:
LPB = Lower probability bound
UPB = Upper probability bound

Figure 7.3-17 Tree Swallow TDI Exposure Model for TEQ: Results of Monte Carlo Analysis and Probability Bounds Analysis at Southwest Branch



Notes:
LPB = Lower probability bound
UPB = Upper probability bound

Figure 7.3-18 Tree Swallow TDI Exposure Model for TEQ: Results of Monte Carlo Analysis and Probability Bounds Analysis at Threemile Pond



Notes:
LPB = Lower probability bound
UPB = Upper probability bound

Figure 7.3-19 Tree Swallow TDI Exposure Model for TEQ: Results of Monte Carlo Analysis and Probability Bounds Analysis at Taconic Valley

American Robins

American robin tPCB and TEQ exposure distributions at Locations 13, 14, and 15 are presented in Figures 7.5-13 to 7.5-18. Please note that these figures, which show the exposure modeling results, along with comparisons to effects metrics, are presented in Section 7.5 (Risk Characterization). This was done to avoid unnecessary duplication of figures that convey essentially the same information. American robins had the highest exposure to tPCBs and TEQ at Location 13, with mean concentrations of 5.66 mg/kg bw/d and 82.3 ng/kg bw/d, respectively (Figure 7.5-13 and Figure 7.5-16). Exposure at Locations 14 and 15 was somewhat lower for both tPCBs and TEQ (Figures 7.5-14 and 7.5-17 and Figures 7.5-15 and 7.5-18). Mean tPCB concentrations at Locations 14 and 15 were 4.69 and 1.22 mg/kg bw/d, respectively. Mean TEQ concentrations at Locations 14 and 15 were 57.5 and 33.0 ng/kg bw/d, respectively.

Wood Ducks

Wood duck tPCB exposure distributions are presented in Figures 7.5-19 to 7.5-21. Please note that these figures, which show the exposure modeling results, along with comparisons to effects metrics, are presented in Section 7.5 (Risk Characterization). This was done to avoid unnecessary duplication of figures that convey essentially the same information. Wood ducks had the highest exposure to tPCBs at Location 13. Total PCB exposure ranged from a minimum of 3.52 to a maximum of 12.2 mg/kg bw/d. The mean exposure was 6.90 mg/kg bw/d, and the median exposure was 6.80 mg/kg bw/d (Figure 7.5-19; Table G.2-39). Of the exposure estimates, 80% were between 5.48 and 8.42 mg/kg bw/d. The 10^{th} percentile of the probability envelope formed by the lower and upper bounds ranged between 1.53 and 6.87 mg/kg bw/d. The 50^{th} percentile ranged between 2.00 and 8.70 mg/kg bw/d, and the 90^{th} percentile ranged between 2.69 and 11.1 mg/kg bw/d. In comparison, the 10^{th} percentile of the Monte Carlo output was 5.48, the 50^{th} percentile was 6.80, and the 90^{th} percentile was 8.42 mg/kg bw/d. Exposure at Locations 14 and 15 were lower for tPCBs, which is consistent with the American robin analysis. Mean tPCB concentrations at Locations 14 and 15 were 5.14 and 2.67 mg/kg bw/d, respectively (Figures 7.5-20 and 7.5-21).

7.3.3 Wood Duck Egg TEQ Exposure Model Results

The predicted TEQ concentrations in wood duck eggs on days 14 (first egg), 20 (7^{th} egg), and 27 (last egg) at the three modeled locations in the PSA are shown in Table G.2-40 and Figures 7.5-22 to 7.5-30. Please note that these figures, which show the exposure modeling results, along with comparisons to effects metrics, are presented in Section 7.5 (Risk Characterization). This was done to avoid unnecessary duplication of figures that convey essentially the same information. The Monte Carlo egg model predictions indicated that the egg TEQ concentration in Location 13 ranged from a minimum of 558 to a maximum of 4,854 ng/kg. The mean exposure was 1783 ng/kg, and the median exposure was 1720 ng/kg. Of the exposure estimates, 80% were between 1,122 and 2,526 ng/kg. The 10^{th} percentile of the probability envelope formed by the lower and upper bounds ranged between 62.3 and 3,571 ng/kg. The 50^{th} percentile ranged between 80.4 and 4537 ng/kg, and the 90^{th} percentile ranged between 108 and 5737 ng/kg. In comparison, the 10^{th} percentile of the Monte Carlo output was 1,122, the 50^{th} percentile was 1,720, and the 90^{th} percentile was 2,526 ng/kg.

7.3.4 Wood Duck Egg Data

No field study was undertaken for wood ducks; however, wood duck tissue was collected from within the PSA. Five wood duck eggs were collected from nest boxes in the PSA and analyzed for tPCB and TEQ congeners (Woodlot Alternatives 2004). Figure G.1-12 indicates the locations of the nest boxes when the eggs were sampled. Table G.2-43 presents the results of these analyses. The mean tPCB concentration in the wood duck eggs was 50.6 mg/kg with a range of 32.9 to 80 mg/kg. 2,3,7,8-TCDD TEQ was calculated using the avian toxic equivalency factors in Van den Berg et al. (1998) and assuming the non-detected congeners were equal to 0, 50%, and 100% of the detection limit. The mean avian TEQ concentration (DL=DL/2) for the five eggs analyzed was 1,336 ng/kg, with a range from 703 to 2077 ng/kg. The wood duck eggs were sampled in May 2004 from an area above Woods Pond in the PSA (see Section 2.8).

7.3.5 Tree Swallow Microexposure Model Results

Tree swallow tPCB and TEQ exposure distributions at Holmes Road, New Lenox Road, Roaring Brook, Southwest Branch, Threemile Pond, and Taconic Valley are presented in Figures 7.5-1 to

7.5-12. Please note that these figures, which show the exposure modeling results, along with comparisons to effects metrics, are presented in Section 7.5 (Risk Characterization). This was done to avoid unnecessary duplication of figures that convey essentially the same information.

The Monte Carlo analysis indicated that accumulation of tPCBs by tree swallows at Holmes Road (Figure 7.5-1) could range from a minimum of 75.8 mg/kg in tissues of 14-d nestlings (5.06 mg/kg bw/d)[1] to a maximum of 595 mg/kg (39.7 mg/kg bw/d). The mean exposure was 222 mg/kg (14.8 mg/kg bw/d), and the median exposure was 215 mg/kg (14.3 mg/kg bw/d); 80% of the exposure estimates were between 154 (10.3) and 253 (15.6) mg/kg.

The probability bounds for tree swallows exposed at Holmes Road are depicted in Figure 7.3-41. The 10^{th} percentile of the probability envelope formed by the lower and upper bounds ranged between 11.5 mg/kg (0.77 mg/kg bw/d) and 215 mg/kg (14.3 mg/kg bw/d). The 50^{th} percentile ranged between 12.6 mg/kg (0.84 mg/kg bw/d) and 227 mg/kg (15.2 mg/kg bw/d), and the 90^{th} percentile ranged between 14.4 mg/kg (0.96 mg/kg bw/d) and 273 mg/kg (18.2 mg/kg bw/d). In comparison, the 10^{th} percentile of the Monte Carlo output was 154 mg/kg (10.3 mg/kg bw/d), the 50^{th} percentile was 215 mg/kg (14.3 mg/kg bw/d), and the 90^{th} percentile was 253 mg/kg (15.6 mg/kg bw/d).

Accumulation of tPCBs by tree swallows at the other two PSA sites, New Lenox Road and Roaring Brook, was lower than at Holmes Road (Tables G.2-13 and G.2-14). Accumulation at Southwest Branch and Threemile Pond was very low; however, the Taconic Valley location had a relatively high accumulation of tPCBs with a mean of 137 mg/kg (9.10 mg/kg bw/d). The highest tissue residues of TEQ were at Holmes Road with a mean of 11,580 ng/kg (772 ng/kg bw/d). New Lenox Road and Roaring Brook had mean tissue residues considerably lower than at Holmes Road (Tables G.2-15 and G.2-16). Mean tissue residues at the three other locations ranged from 3,783 ng/kg (252 ng/kg bw/d) to 8,224 ng/kg (548 ng/kg bw/d). The results of the microexposure model, when converted to daily intake units of tPCBs, are in agreement with the results of the TDI models.

---

[1] Estimated by dividing tissue residues by 15, which was the number of days included in the microexposure model.

7.3.6 Tree Swallow Tissue Data from Field Study

7.3.6.1 Tissue Residue Data

Analyses of the COC concentrations in tree swallow tissues provided direct measures of tPCB and TEQ exposure. Tissue samples were collected from eggs or just-hatched young (herein referred to as pippers) and 12- to 14-day-old nestlings in 1998 at four sites and in 1999 and 2000 at six sites (Custer 2002). These samples provided tPCBs concentration data for pippers and 12- to 14-day-old nestlings (i.e., pre-migratory full-grown birds) and TEQ concentrations for pippers.

7.3.6.2 12- to 14-day-old Bird Tissue Concentrations

Summary statistics for Custer (2002) tissue samples by location are provided in Table G.2-20. The highest median concentrations, observed at Holmes Road, New Lenox Road, and Roaring Brook, ranged from 21.5 to 36.0 mg/kg ww during the 3-year period. Median concentrations at the upstream and the other locations (Threemile Pond) ranged from 0.77 to 3.30 mg/kg ww. Total PCB concentrations in nestlings at Holmes Road, New Lenox Road, and Roaring Brook indicated that there was accumulation beyond what was observed in the eggs and pippers at these locations.

The nestling tissue data were compared to the results of the microexposure model directly because the microexposure model estimated whole body concentrations of tPCBs (mg/kg) for nestlings aged 14 days and the tissue samples were taken from nestlings aged 12 to 14 days. For all locations, the measured tissue concentrations were significantly lower than the distributions estimated by the Monte Carlo analysis of the microexposure model. This is likely partially due to the dissimilar exposure durations between the location-specific data and modeled data. However, all measured distributions were within the microexposure model probability bounds.

7.3.6.3 Pipper Tissue Concentrations

Total PCBs

Total PCB concentrations were generally higher in pippers from the PSA than at the reference locations (Table G.2-22). Median concentrations in pippers ranged from 44.9 to 80.2 mg/kg ww in the PSA; median concentrations ranged from 7.5 to 14.4 mg/kg ww in pippers at the reference sites. Concentrations of tPCBs were generally comparable among years at each site.

TEQ

Concentrations of TEQ were highest at sites in the PSA. The median concentrations ranged from 1,390 ng/kg ww at Holmes Road to 2,890 ng/kg at Roaring Brook. In comparison, median concentrations ranged from 562 to 730 ng/kg ww in pippers at the other sites (Table G.2-23) (Custer 2002).

7.4 EFFECTS ASSESSMENT

The effects assessment has two objectives. The first is to review the scientific literature for effects of tPCBs (Aroclors 1254 and 1260) and TEQ to insectivorous birds, with documented effects to the representative species in this assessment, tree swallows, American robins, and wood ducks. The other objective is to derive the effects metrics that will be used, in conjunction with the exposure assessment results, to estimate risks to insectivorous birds.

7.4.1 Review of Effects of tPCBs

The range of effect concentrations on avian species for PCBs (Aroclors 1254 and/or 1260) is summarized graphically in Figure 7.4-1. These effects concentrations were obtained from the pertinent scientific literature; details of the studies consulted and their applicability are included in Appendix G.

A field study conducted for GE examined the effects of ecological stressors (i.e., nest box density) on tree swallow reproduction (Robertson and Jones 2002). The study took place 50 km north of Kingston, Ontario, Canada. The study was not considered in this risk assessment because the study did not evaluate the effects of tPCBs or TEQ on tree swallow reproduction; therefore, it was unrelated to the assessment endpoint.

7.4.2 2,3,7,8-TCDD Equivalence (TEQ)

The range of effect concentrations on avian species for dioxin (2,3,7,8-TCDD) and dioxin toxic equivalence (TEQ) is summarized graphically in Figure 7.4-2. These effects concentrations were obtained from the pertinent scientific literature; details of the studies consulted and their applicability are included in Appendix G.

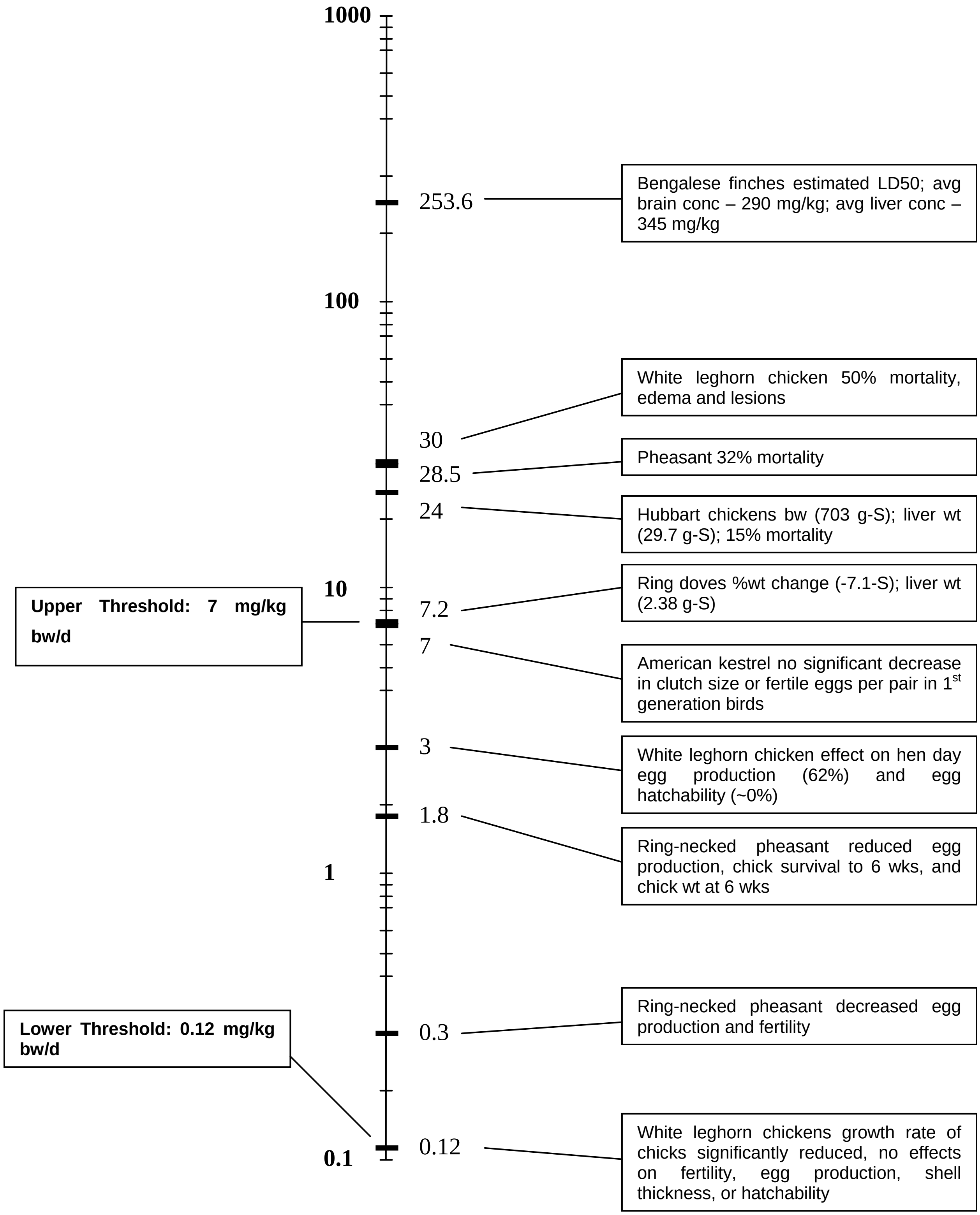


Figure 7.4-1 Effects of Aroclor 1254/1260 on Avian Species (mg/kg bw/d)

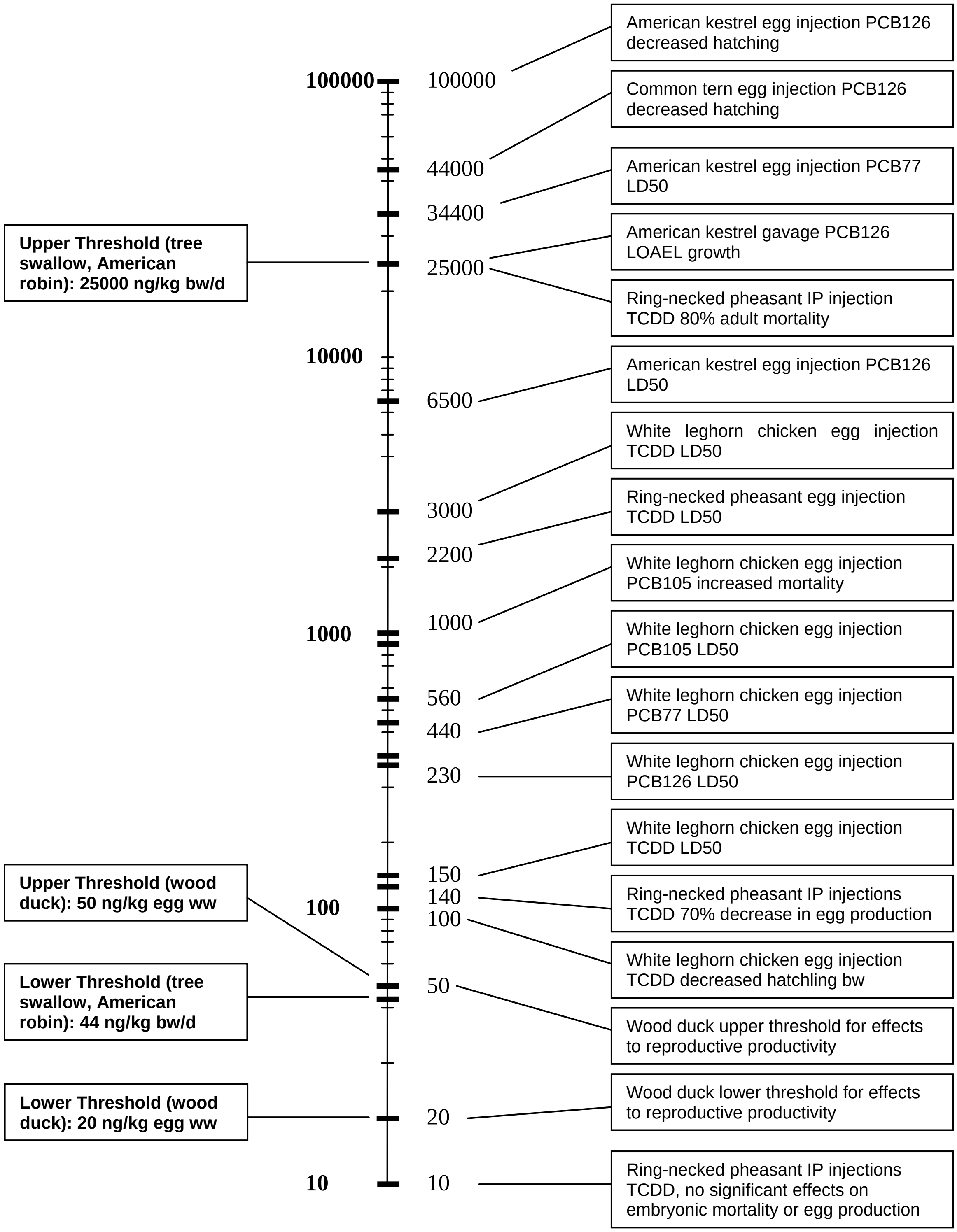


Figure 7.4-2 Effects of 2,3,7,8-TCDD TEQ on Avian Species (ng TEQ/kg bw/d)

7.4.3 Tree Swallow Field Study

A tree swallow reproduction study was conducted between 1998 and 2000. The study focused on nest box occupancy rate, clutch size, nesting success, and determination of tree swallow tissue concentrations of tPCBs and TEQ at sites inside and outside of the PSA. Tree swallows were selected for study prior to the publication of several field studies examining the impact of PCBs (e.g., Custer 2002; Custer et al. 1998; Bishop et al. 1999; McCarty and Secord 1999a) that reported tree swallows were insensitive to PCB exposure at relatively high concentrations, though low concentration effects (e.g. EROD activity) were observed.

Nest boxes were deployed in 1998 at four locations along the Housatonic River and its tributaries and in 1999 at two additional reference locations. Samples of eggs or just-hatched young (pippers) were collected for organochlorine contaminant analyses. Stomach content samples were taken from birds in 1998, 1999, and 2000 (Custer 2002). 2,3,7,8-TCDD equivalents (TEQ) were calculated using the toxic equivalency factors (TEFs) of Van den Berg et al. (1998).

Results**—**There was a significant negative relationship between tPCB concentrations in eggs and hatching success in 1999 ($p = 0.044$); however, the fit of the model was poor. In 1998 and 1999, clutches that contained dead embryos had significantly higher concentrations of tPCBs than those that hatched normally ($p < 0.001$). The geometric mean concentrations in clutches that had reduced hatching were 62.8 mg/kg ww in eggs in 1998 and 69.1 mg/kg ww in eggs in 1999.

Differences in the geometric mean concentrations of dioxins and furans between sites were similar to the pattern for tPCBs.

Conclusions**—**Although some differences in tree swallow reproduction appeared linked to contaminants in the PSA, the fecundity of tree swallows was not severely impacted. Multivariate regression models indicated that dioxins and furans in the PSA could be contributing to the observed reduced hatching success.

7.4.4 American Robin Field Study (GE)

The reproductive output of American robins was studied in the PSA and reference areas during the 2001 breeding season (Henning 2002). The objective of the study was to evaluate the relationship between reproductive success and tissue concentrations of tPCBs in eggs and young. Active robin nests were monitored to record the number of eggs and hatchlings, as well as depredation, abandonment, parental behavior, and development of young. Eggs and nestlings were analyzed for tPCB concentrations.

Results**—**Although there were large differences in tPCB concentrations in American robin eggs in the PSA compared with reference areas, clutch sizes in the PSA and reference areas were not significantly different.

Conclusions**—**Concentrations of tPCBs in American robin eggs and nestlings were significantly higher in the PSA compared to reference areas. There were, however, no significant differences in any of the measures of effects on reproduction included in this study. Further, clutch size, hatching success, and fledging success in the target and reference areas were within ranges reported for American robins (Brehmer and Anderson 1992; Kemper and Taylor 1981; Fluetsch and Sparling 1994).

7.4.5 Selection of Effects Metrics for Characterizing Risk

Effects data can be characterized and summarized in a variety of ways ranging from benchmarks designed to be protective of most or all species to concentration- or dose-response curves. A summary of the decision criteria used to derive effects metrics is provided in the text box. Further details on the decision criteria used in selecting effects metrics is provided in Section 6.6.

Hierarchical Decision Criteria for Derivation of Effects Metric

1. Have single-study bioassays with five or more treatments been conducted on the receptor of interest or a reasonable surrogate? If yes, estimate the concentration- or dose-response. If not, go to 2.
2. Are multiple bioassays with similar protocols, exposure scenarios, and effects metrics available that, when combined, have five or more treatments for the receptor of interest or a reasonable surrogate? If yes, estimate the dose-response relationship as in 1. If not, go to 3.
3. Have bioassays with less than five treatments been conducted on the receptor of interest or a reasonable surrogate? If yes, conduct or report results of hypothesis testing to determine the NOAEL and LOAEL. If not, go to 4.
4. Are sufficient data available from field studies and monitoring programs to estimate concentrations or doses of the COC that are consistently protective or associated with adverse effects? If yes, develop field-based effects metrics. If not, go to 5.
5. Derive a range where the threshold for the receptor of interest is expected to occur. Because information on the sensitivity of the receptor of interest is lacking, it is difficult to derive a threshold that is biased neither high nor low. If bioassay data are available for several other species, however, calculate a threshold for each to determine a threshold range that spans sensitive and tolerant species. That range is likely to include the threshold for the receptor of interest.

Although much work has been done to investigate the effects of tPCBs and TEQ to various bird species in controlled laboratory settings, no suitable studies were available for tree swallows, American robins, wood duck, or for bird species that could be considered reasonable surrogates. Therefore, no dose-response relationship could be established between exposure of tree swallows, American robins, or wood ducks to tPCBs (either Aroclor 1254 or 1260) or TEQ and adverse effects on mortality, growth, or reproduction. It was also not possible to establish a NOAEL or LOAEL for adverse effects from available laboratory studies. A field-based effects metric for reproductive effects was available for TEQ exposed wood ducks.

7.4.5.1 Effects of tPCBs to Tree Swallows

Based on the results of the field-based effects studies reviewed, a field-based threshold range was derived for tree swallows exposed to tPCBs. The low threshold was based on the McCarty and Secord (1999b) study where tPCB concentrations in whole body nestlings were 62.2 mg/kg ww, and the high effects threshold was based on the field study conducted by Custer (2002) where tree swallow pipper tissue concentrations of 69 mg/kg ww were associated with hatching problems. Therefore, the effects threshold range for insectivorous birds for tPCBs is 62.2 to 69 mg/kg ww.

7.4.5.2 Effects of tPCBs to American Robins and Wood Ducks

The threshold range for the reproductive success of American robins and wood ducks exposed to tPCBs selected for this assessment is 0.12 to 7.0 mg/kg bw/d based on reproductive studies conducted on white leghorn chickens (Lillie et al. 1974) and American kestrels (Fernie et al. 2001), respectively.

7.4.5.3 Effects of TEQ to Tree Swallows and American Robins

The low toxicological threshold for effects of TEQ to sensitive birds is based on the study by Nosek et al. (1992) using ring-necked pheasants. The threshold for tolerant avian species was derived from Hoffman et al. (1996) where 25,000 ng/kg bw/d TEQ was determined to be the reproductive threshold for American kestrels exposed to TEQ. Thus, the effects threshold range is 44 to 25,000 ng/kg bw/d[1] for TEQ.

7.4.5.4 Effects of TEQ to Wood Ducks

The toxicological threshold range for the effects of TEQ to wood duck is based on a field study by White and Seginak (1994). The authors examined the link between TEQ concentration in eggs and reproductive impairment in wood ducks in a contaminated wetland in central Arkansas. The authors reported that hatching success and duckling production were negatively correlated ($p < 0.001$) with egg TEQ. The authors reported a threshold range for reduced productivity of >20 to 50 ppt (ng/kg egg) for wood duck. Wood duck was identified as being more sensitive to dioxins and furans relative to some other bird species. This threshold range is similar to or higher than reported reproduction effect thresholds for TEQ in eggs in other sensitive bird species including Forster's terns (*Sterna forsteri*) (37 ng 2,3,7,8-TCDD/kg egg; Kubiak et al. 1989), white leghorn chickens (10 ng 2,3 7,8-TCDD/kg egg; Verrett 1970), and adult ringed-neck pheasants (1 ng 2,3,7,8-TCDD /kg egg; Nosek et al. 1992).

---

[1] A tissue residue threshold range was derived for TEQ by multiplying this threshold range by 15 days, which is the number of days included in the microexposure model. However, this threshold range was not used in the final assessment of risk to tree swallows because the TDI model results were used to estimate risk of TEQ to tree swallows.

The TEQ threshold range reported by White and Seginak (1994) was calculated using International Toxicity Equivalence Factors (I-TEFs) for mammals (EPA 1989). Van den Berg et al. (1998) have since updated the TEFs for dioxin and furan congeners, derived TEFs for coplanar PCB congeners, and developed TEFs specific to birds. The updated TEFs are often referred to as the World Health Organization TEFs (WHO TEFs). Because the original congener concentration data from White and Seginak (1994) are no longer available, it is not possible to recalculate the TEQ threshold range for wood duck using the WHO TEFs for birds. Using the data summary tables published by White and Seginak (1994), however, it was determined that the WHO TEQ threshold range would be no less than 0.94 times and no more than 1.77 times of the I-TEFs threshold range (see Attachment II in Appendix G). Given the similarity between the two TEQ threshold ranges and the inability to estimate a precise WHO-TEQ threshold range, the I-TEFs TEQ threshold range was retained for the wood duck assessment.

7.5 RISK CHARACTERIZATION

This section characterizes risk to insectivorous birds exposed to tPCBs and 2,3,7,8-TCDD TEQ in the PSA. The risk characterization uses two lines of evidence to determine potential ecological risks to tree swallows and American robins, and one line of evidence for wood ducks. The major lines of evidence are considered to be independent and will be combined in a weight-of-evidence assessment. The risk questions and the lines of evidence are summarized in the text box. A more detailed presentation of this information is provided in Appendix G.

Risk Questions

- Are the concentrations of tPCBs and TEQ present in the prey of insectivorous birds sufficient to cause adverse effects to individuals inhabiting the PSA of the Housatonic River?
- If so, how severe are the risks and what are their potential consequences?

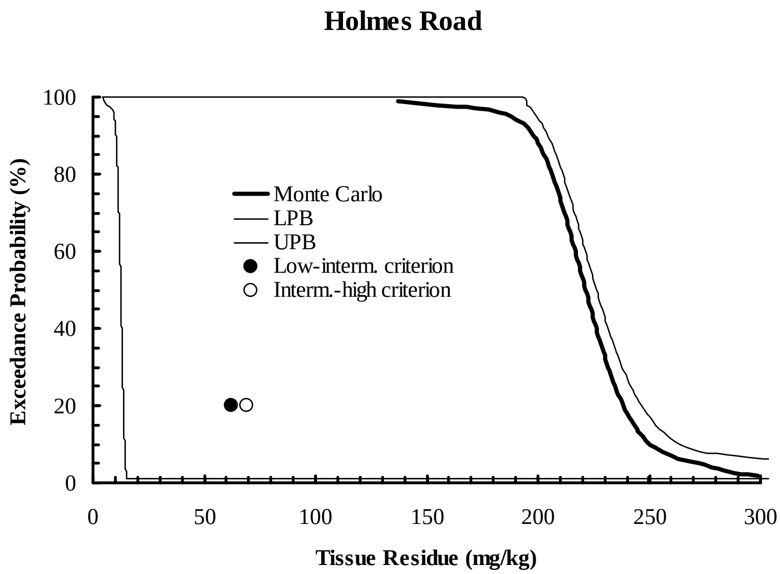
Lines of Evidence

- Field-based tree swallow and American robin reproduction studies.
- Probabilistic exposure and effects modeling.

7.5.1 Comparison of Modeled Exposures to Effects

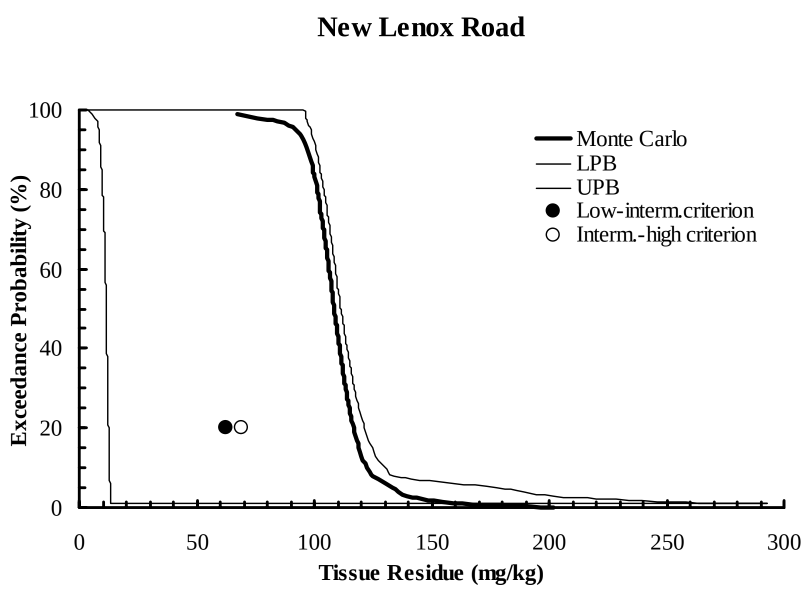
The probabilistic exposure analysis for tree swallows used a total daily intake rate modeling approach for calculating exposure to tPCBs and TEQ. A microexposure model was also employed to estimate tissue concentrations of tPCBs and TEQ in juvenile birds after the first 2 weeks of their development. The results from the microexposure model were used to estimate risk for tPCBs and the TDI model was used to estimate the risk for TEQ, to match the type of effect metric developed for each COC (Figures 7.5-1 to 7.5-12).

The probabilistic exposure analysis for American robins and wood ducks used a total daily intake model to estimate exposure to tPCBs and TEQ (Figures 7.5-13 to 7.5-21). The total daily intake model was further expanded to estimate exposure wood duck egg TEQ concentrations (Figures 7.5-22 to 7.5-30).



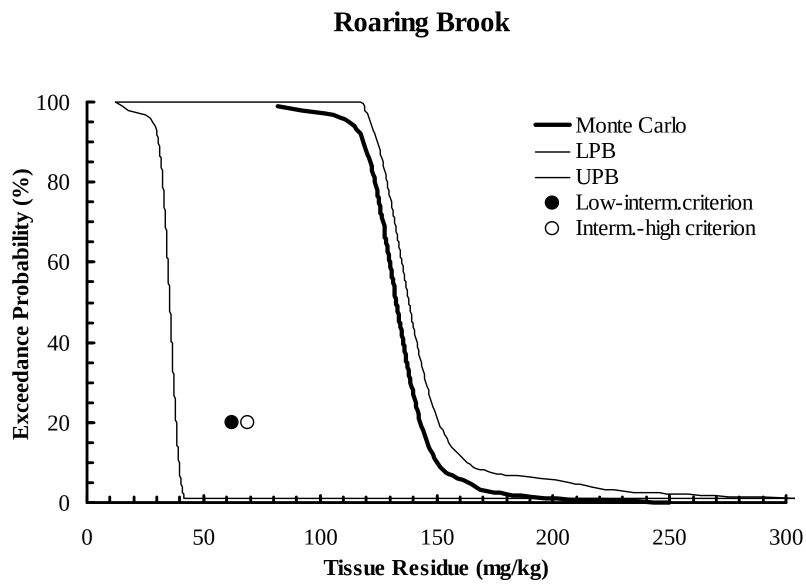
Notes:
LPB = Lower probability bound
UPB = Upper probability bound

Figure 7.5-1 Tree Swallow Microexposure Model for tPCBs: Results of Monte Carlo Analysis and Probability Bounds Analysis at Holmes Road



Notes:
LPB = Lower probability bound
UPB = Upper probability bound

Figure 7.5-2 Tree Swallow Microexposure Model for tPCBs: Results of Monte Carlo Analysis and Probability Bounds Analysis at New Lenox Road

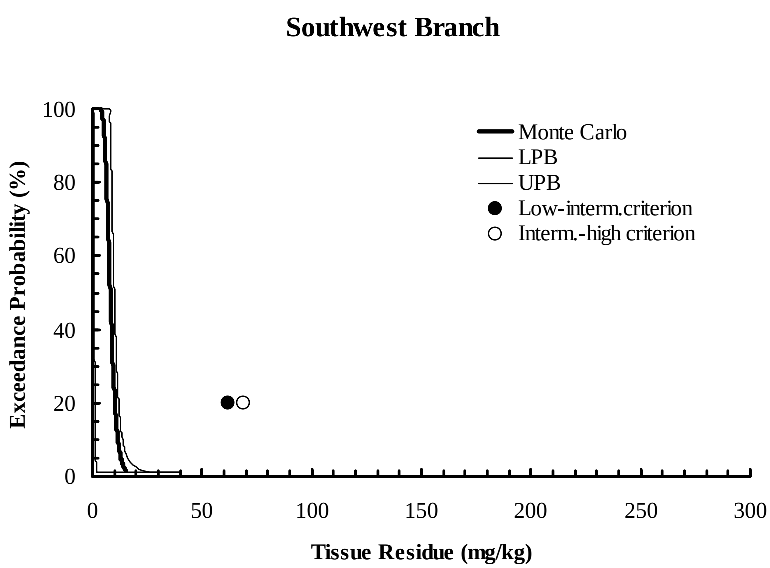


Notes:

LPB = Lower probability bound

UPB = Upper probability bound

Figure 7.5-3 Tree Swallow Microexposure Model for tPCBs: Results of Monte Carlo Analysis and Probability Bounds Analysis at Roaring Brook



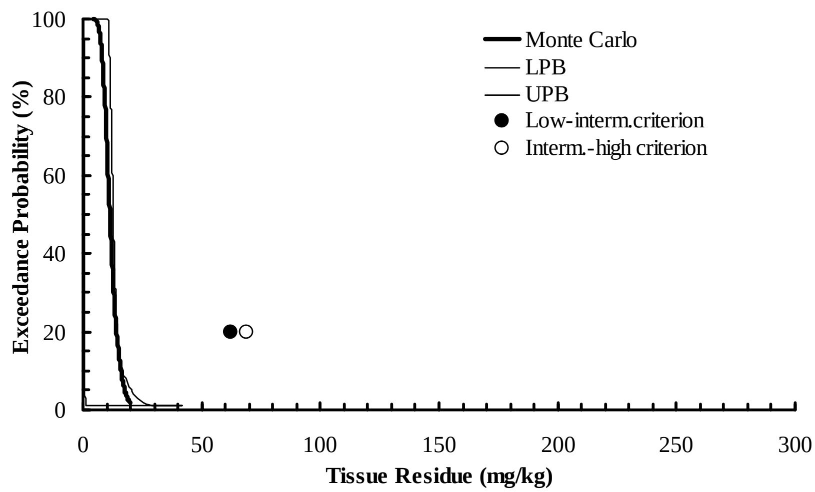
Notes:

LPB = Lower probability bound

UPB = Upper probability bound

Figure 7.5-4 Tree Swallow Microexposure Model for tPCBs: Results of Monte Carlo Analysis and Probability Bounds Analysis at Southwest Branch

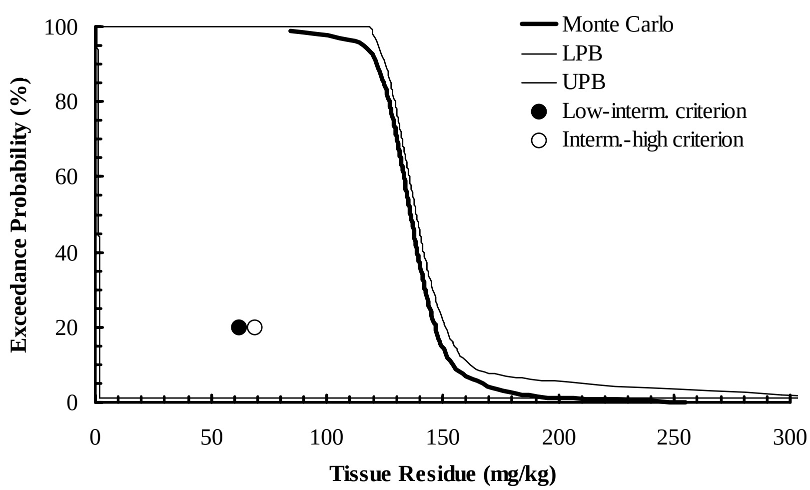
Threemile Pond



Notes:
LPB = Lower probability bound
UPB = Upper probability bound

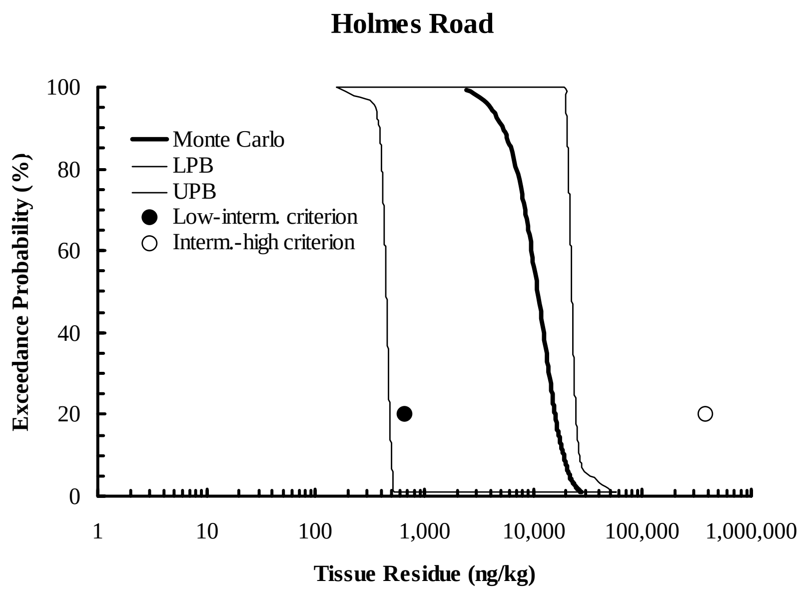
Figure 7.5-5 Tree Swallow Microexposure Model for tPCBs: Results of Monte Carlo Analysis and Probability Bounds Analysis at Threemile Pond

Taconic Valley



Notes:
LPB = Lower probability bound
UPB = Upper probability bound

Figure 7.5-6 Tree Swallow Microexposure Model for tPCBs: Results of Monte Carlo Analysis and Probability Bounds Analysis at Taconic Valley

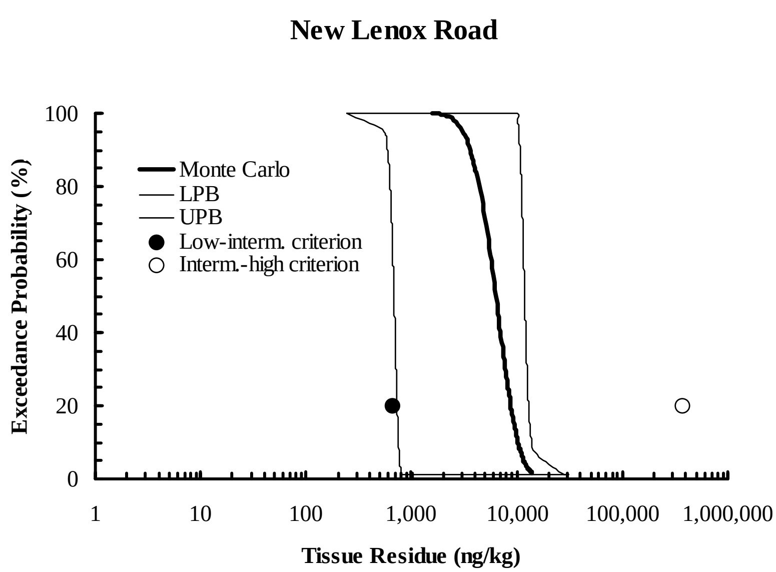


Notes:

LPB = Lower probability bound

UPB = Upper probability bound

Figure 7.5-7 Tree Swallow Microexposure Model for TEQ: Results of Monte Carlo Analysis and Probability Bounds Analysis at Holmes Road

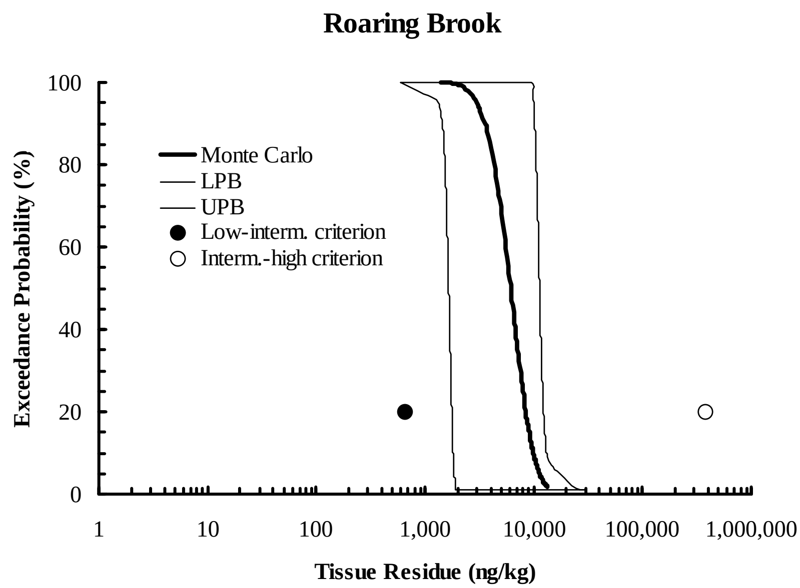


Notes:

LPB = Lower probability bound

UPB = Upper probability bound

Figure 7.5-8 Tree Swallow Microexposure Model for TEQ: Results of Monte Carlo Analysis and Probability Bounds Analysis at New Lenox Road

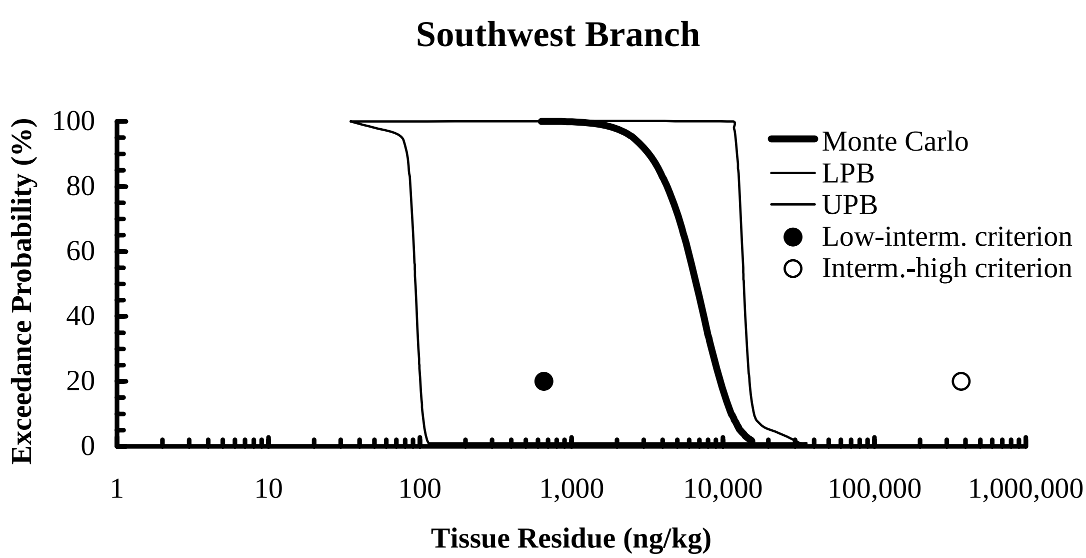


Notes:

LPB = Lower probability bound

UPB = Upper probability bound

Figure 7.5-9 Tree Swallow Microexposure Model for TEQ: Results of Monte Carlo Analysis and Probability Bounds Analysis at Roaring Brook

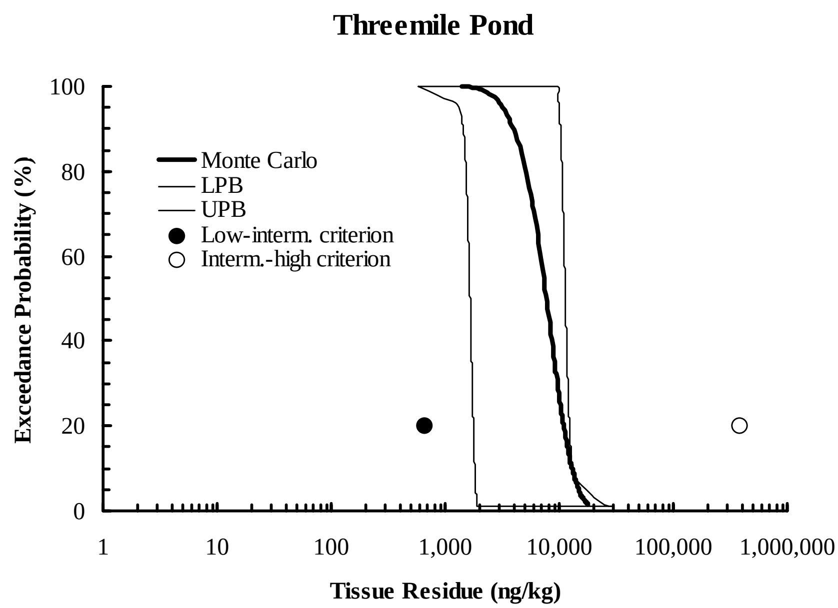


Notes:

LPB = Lower probability bound

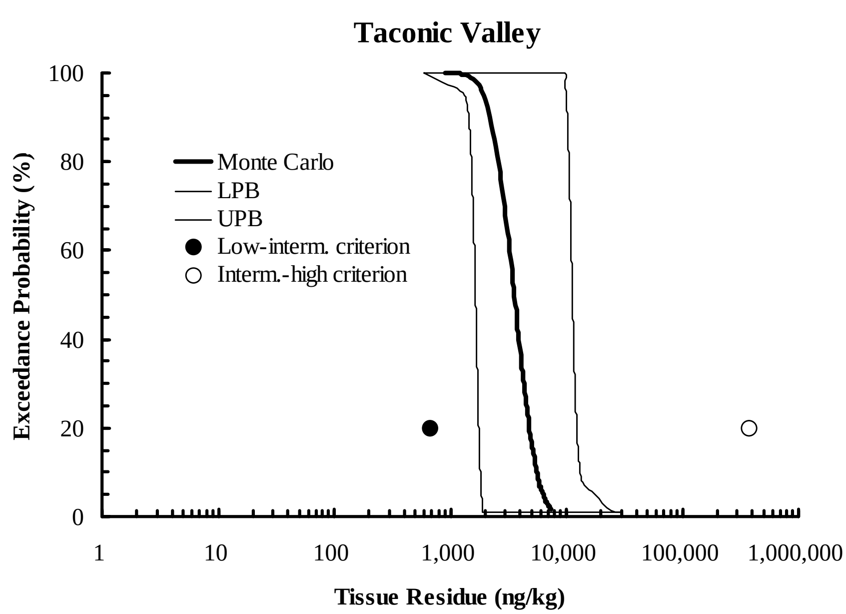
UPB = Upper probability bound

Figure 7.5-10 Tree Swallow Microexposure Model for TEQ: Results of Monte Carlo Analysis and Probability Bounds Analysis at Southwest Branch



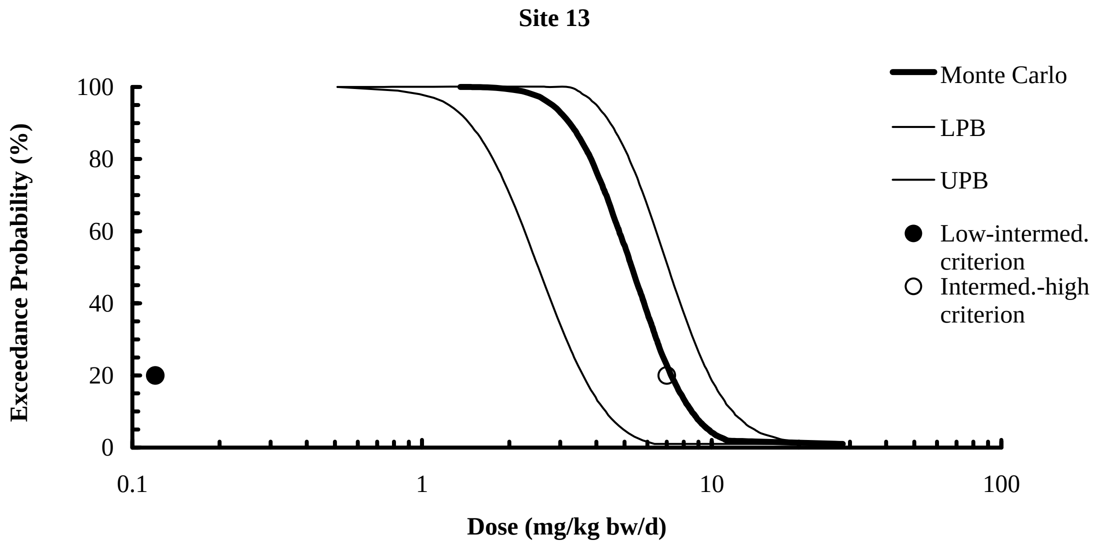
Notes:
LPB = Lower probability bound
UPB = Upper probability bound

Figure 7.5-11 Tree Swallow Microexposure Model for TEQ: Results of Monte Carlo Analysis and Probability Bounds Analysis at Threemile Pond



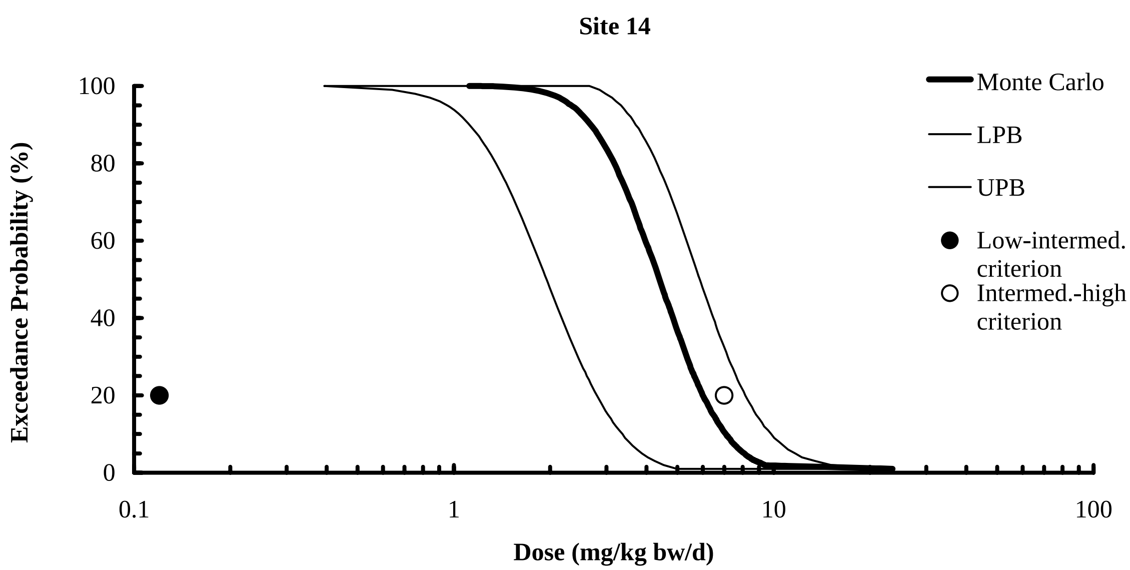
Notes:
LPB = Lower probability bound
UPB = Upper probability bound

Figure 7.5-12 Tree Swallow Microexposure Model for TEQ: Results of Monte Carlo Analysis and Probability Bounds Analysis at Taconic Valley



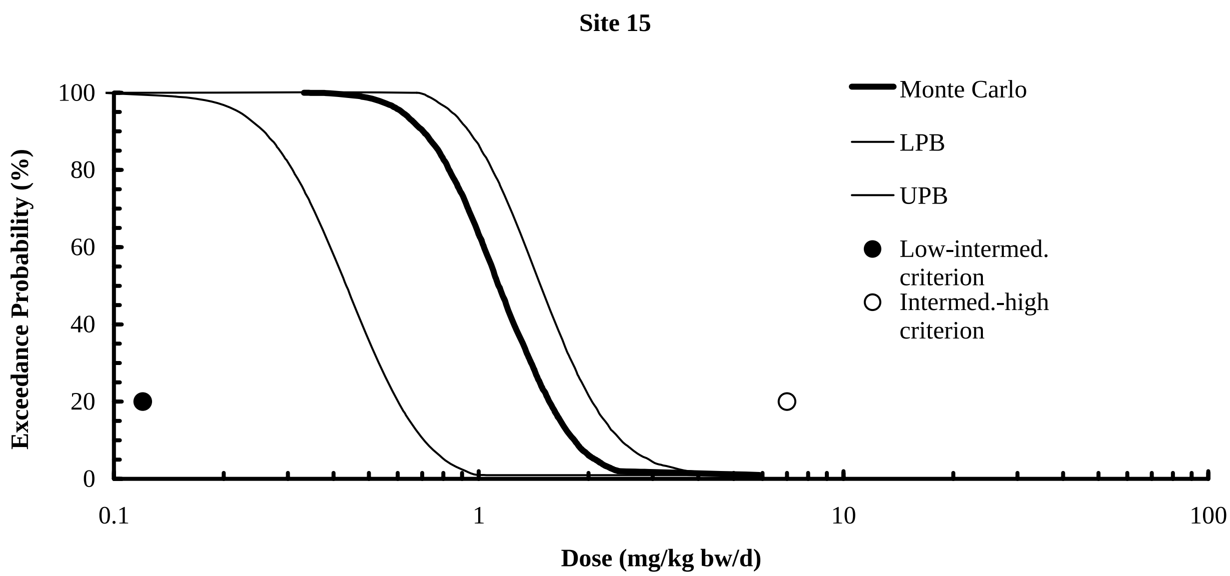
Notes:
LPB = Lower probability bound
UPB = Upper probability bound

Figure 7.5-13 Exposure of American Robins to tPCBs at Site 13 of the Housatonic River PSA



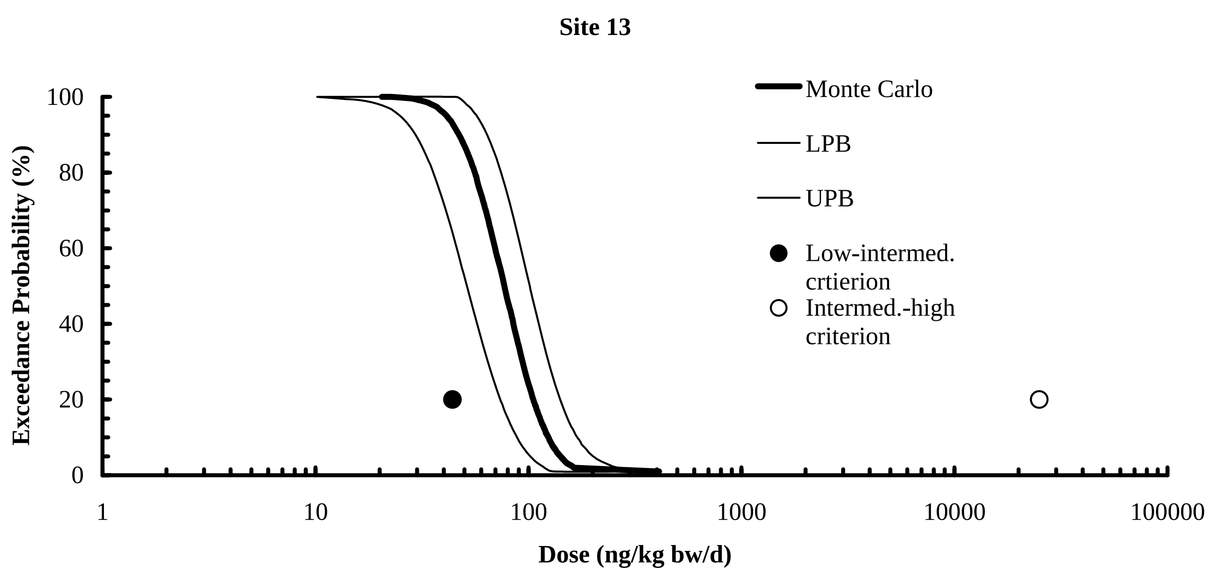
Notes:
LPB = Lower probability bound
UPB = Upper probability bound

Figure 7.5-14 Exposure of American Robins to tPCBs at Site 14 of the Housatonic River PSA



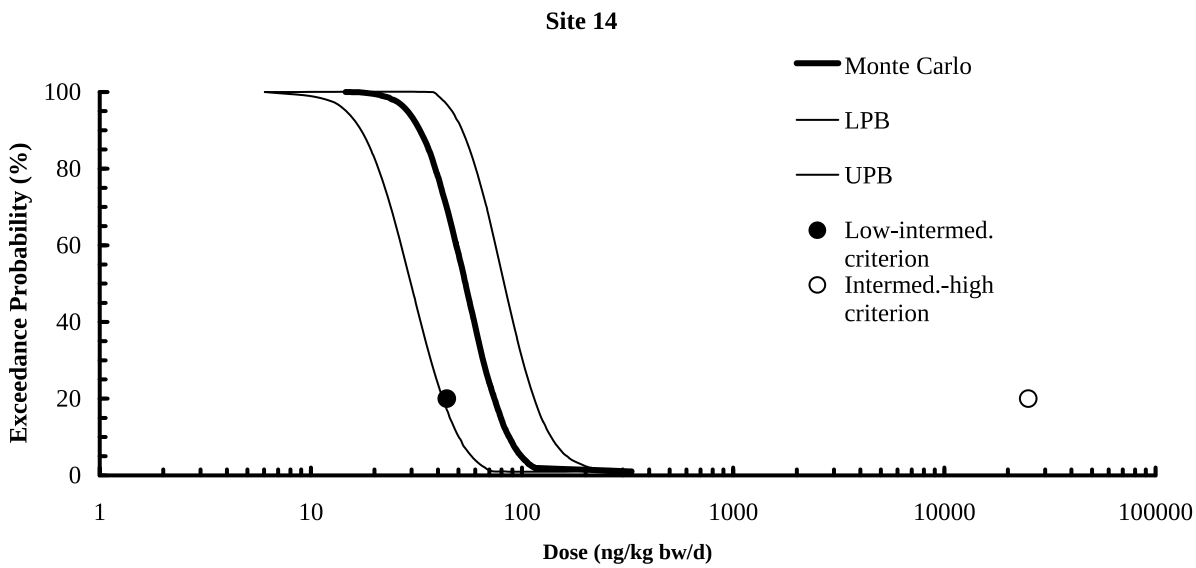
Notes:
LPB = Lower probability bound
UPB = Upper probability bound

Figure 7.5-15 Exposure of American Robins to tPCBs at Site 15 of the Housatonic River PSA



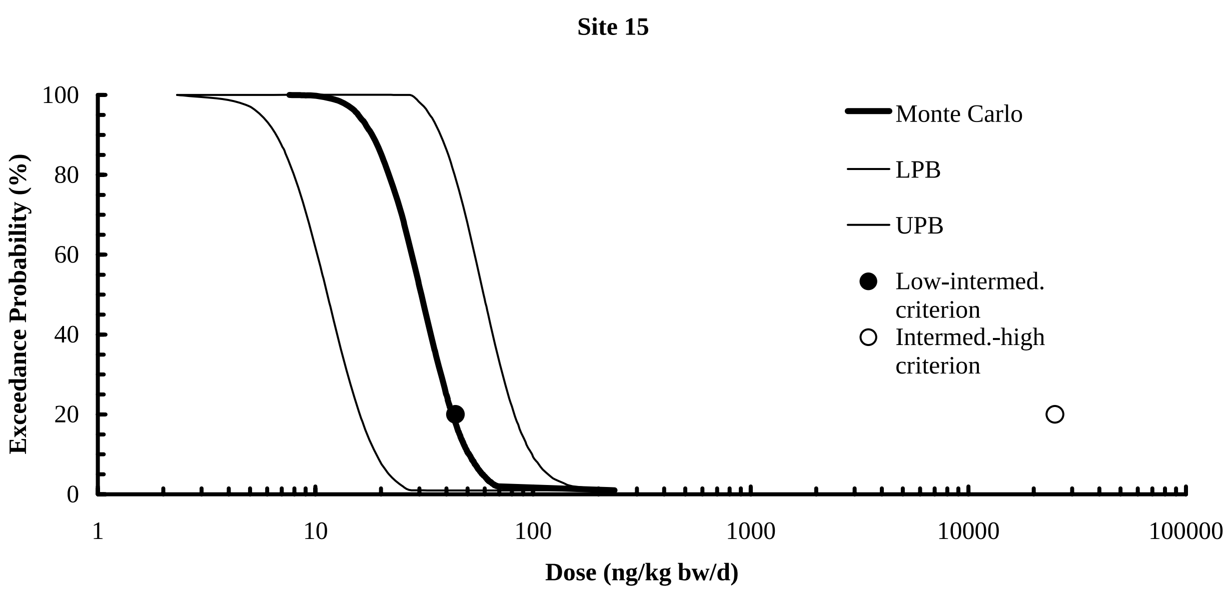
Notes:
LPB = Lower probability bound
UPB = Upper probability bound

Figure 7.5-16 Exposure of American Robins to 2,3,7,8-TCDD TEQ at Site 13 of the Housatonic River PSA



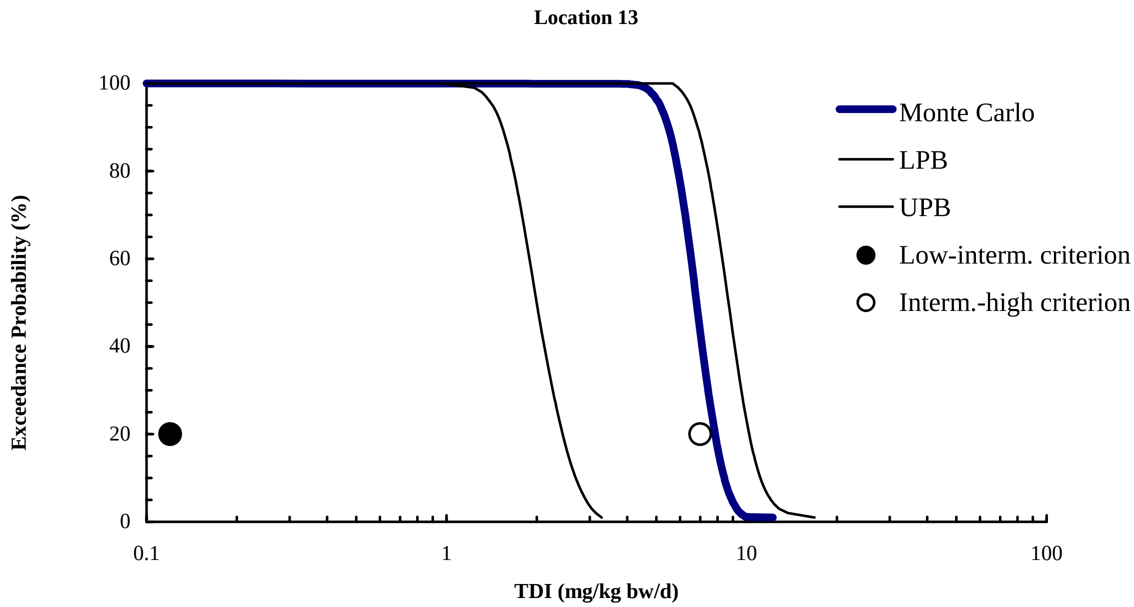
Notes:
LPB = Lower probability bound
UPB = Upper probability bound

Figure 7.5-17 Exposure of American Robins to 2,3,7,8-TCDD TEQ at Site 14 of the Housatonic River PSA



Notes:
LPB = Lower probability bound
UPB = Upper probability bound

Figure 7.5-18 Exposure of American Robins to 2,3,7,8-TCDD TEQ at Site 15 of the Housatonic River PSA

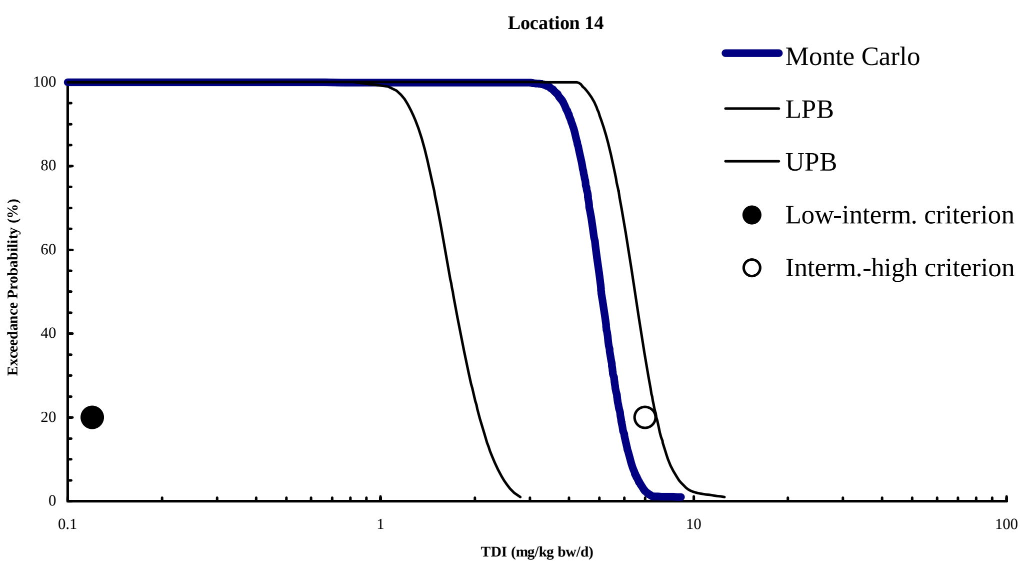


Notes:

LPB = Lower probability bound

UPB = Upper probability bound

Figure 7.5-19 Exposure of Wood Ducks to tPCBs at Location 13 of the Housatonic River PSA

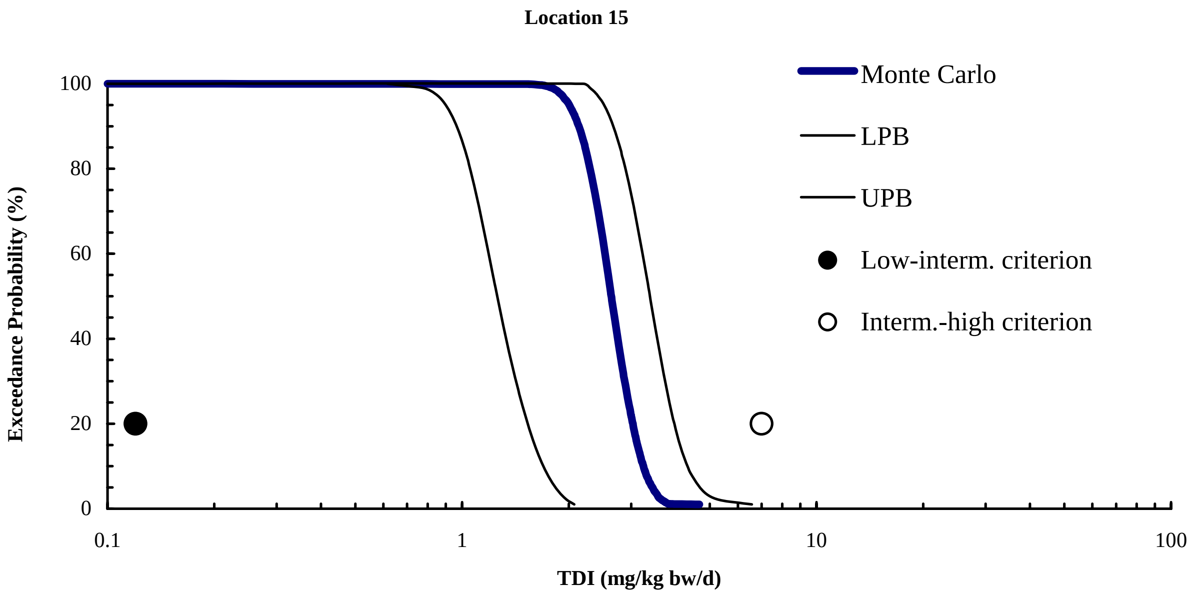


Notes:

LPB = Lower probability bound

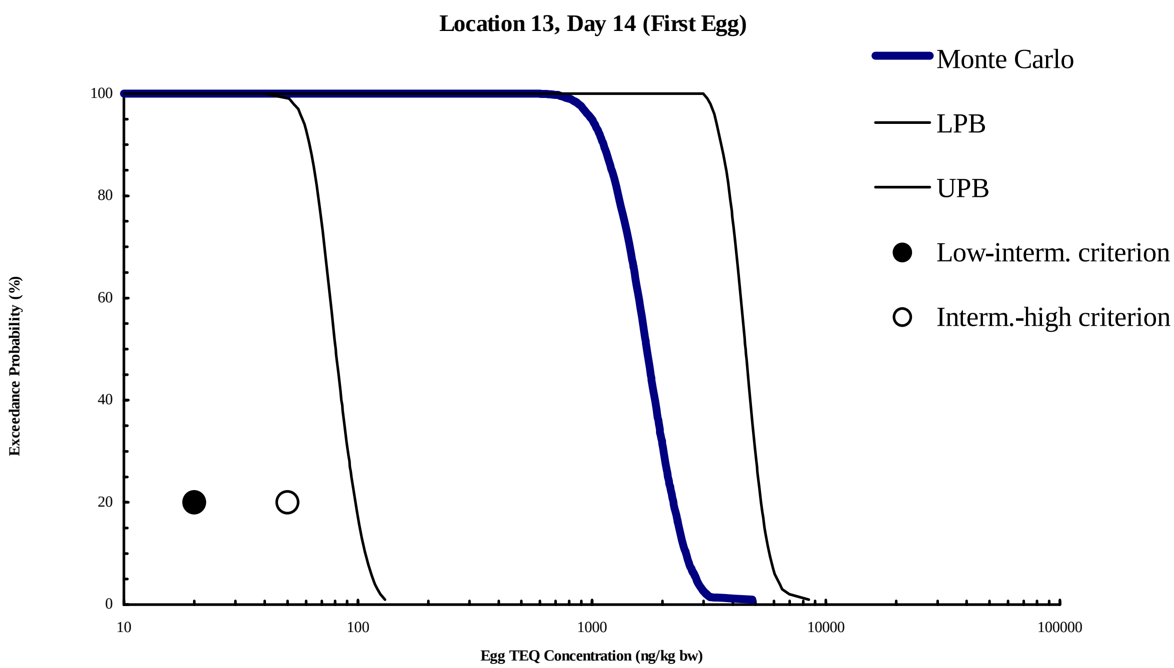
UPB = Upper probability bound

Figure 7.5-20 Exposure of Wood Ducks to tPCBs at Location 14 of the Housatonic River PSA



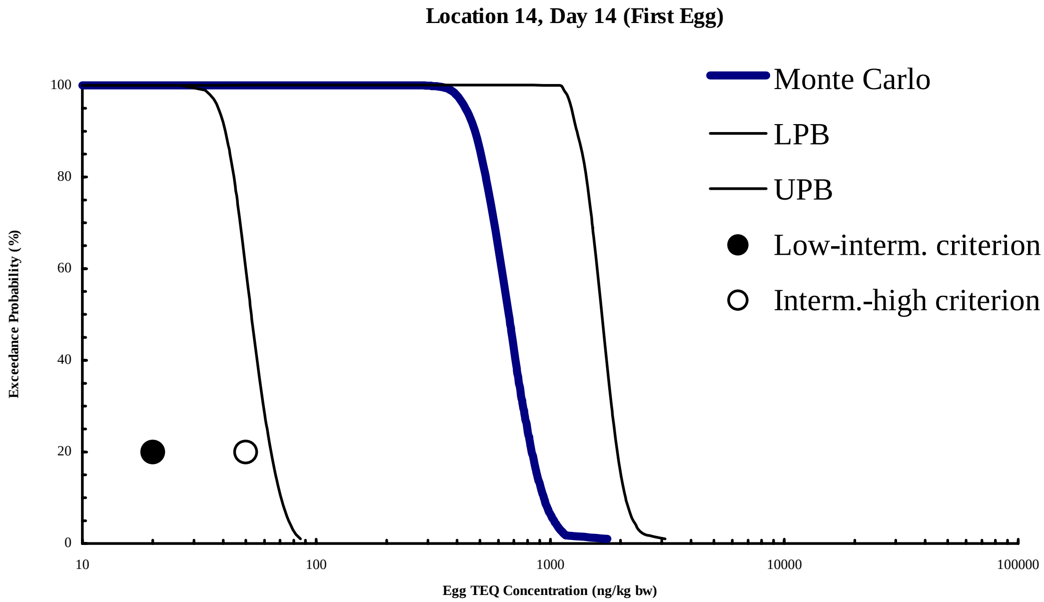
Notes:
LPB = Lower probability bound
UPB = Upper probability bound

Figure 7.5-21 Exposure of Wood Ducks to tPCBs at Location 15 of the Housatonic River PSA



Notes:
LPB = Lower probability bound
UPB = Upper probability bound

Figure 7.5-22 Exposure of Wood Duck Eggs to 2,3,7,8-TCDD TEQ at Location 13, 14 Days (First Egg) After Arriving in the PSA

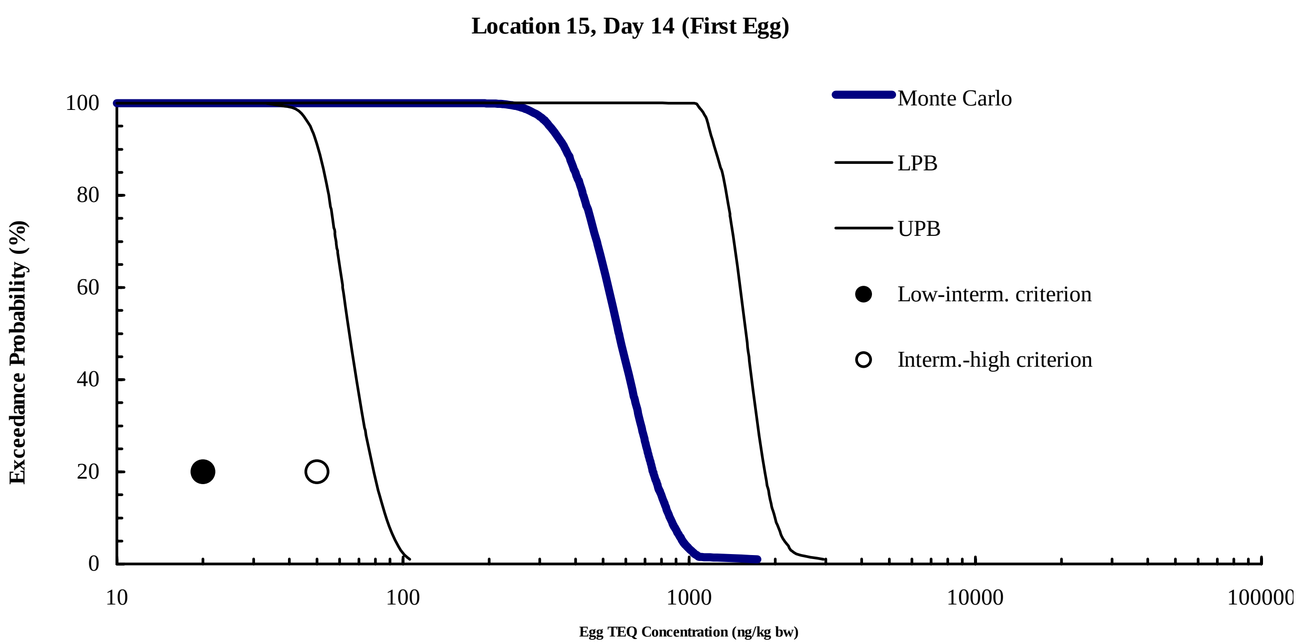


Notes:

LPB = Lower probability bound

UPB = Upper probability bound

Figure 7.5-23 Exposure of Wood Duck Eggs to 2,3,7,8-TCDD TEQ at Location 14, 14 Days (First Egg) After Arriving in the PSA

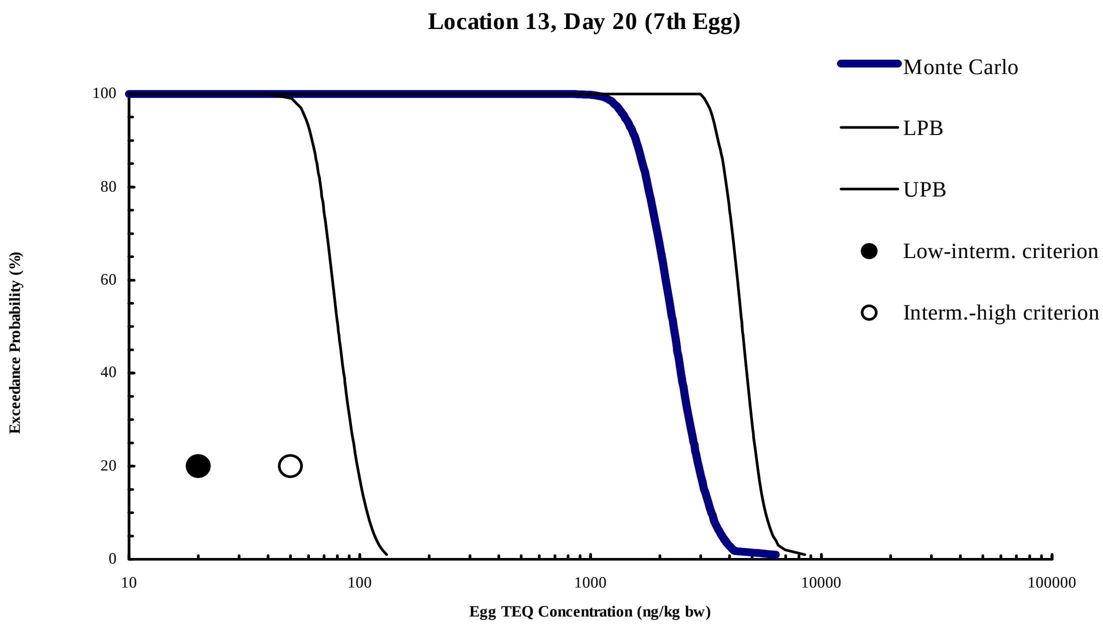


Notes:

LPB = Lower probability bound

UPB = Upper probability bound

Figure 7.5-24 Exposure of Wood Duck Eggs to 2,3,7,8-TCDD TEQ at Location 15, 14 Days (First Egg) After Arriving in the PSA

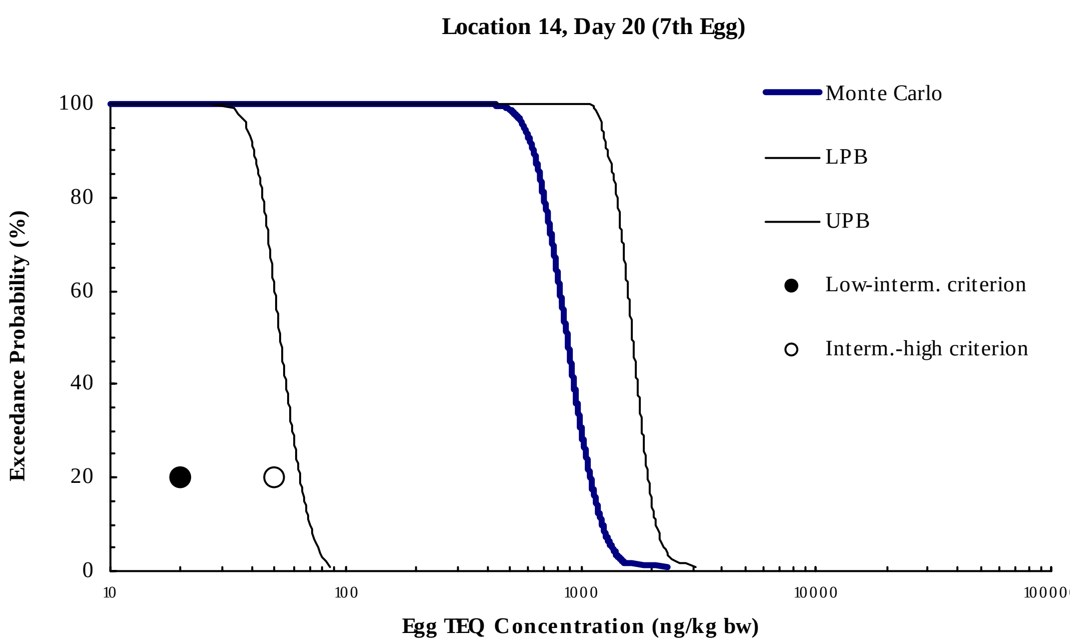


Notes:

LPB = Lower probability bound

UPB = Upper probability bound

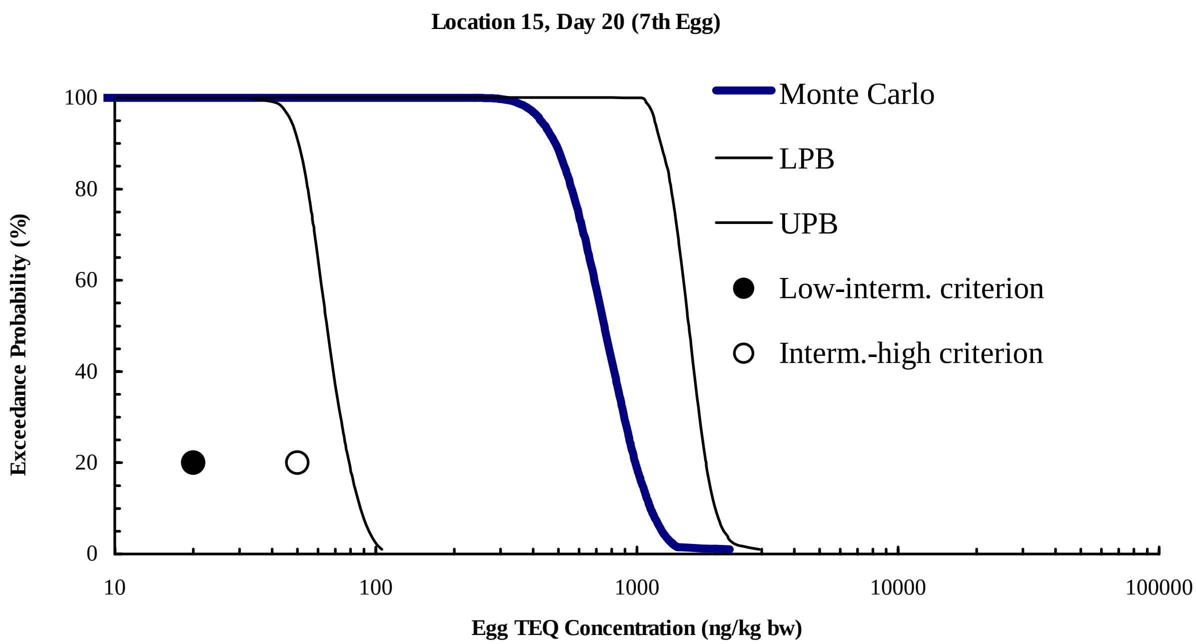
Figure 7.5-25 Exposure of Wood Duck Eggs to 2,3,7,8-TCDD TEQ at Location 13, 20 Days (7th Egg) After Arriving in the PSA



Notes: LPB = Lower probability bound

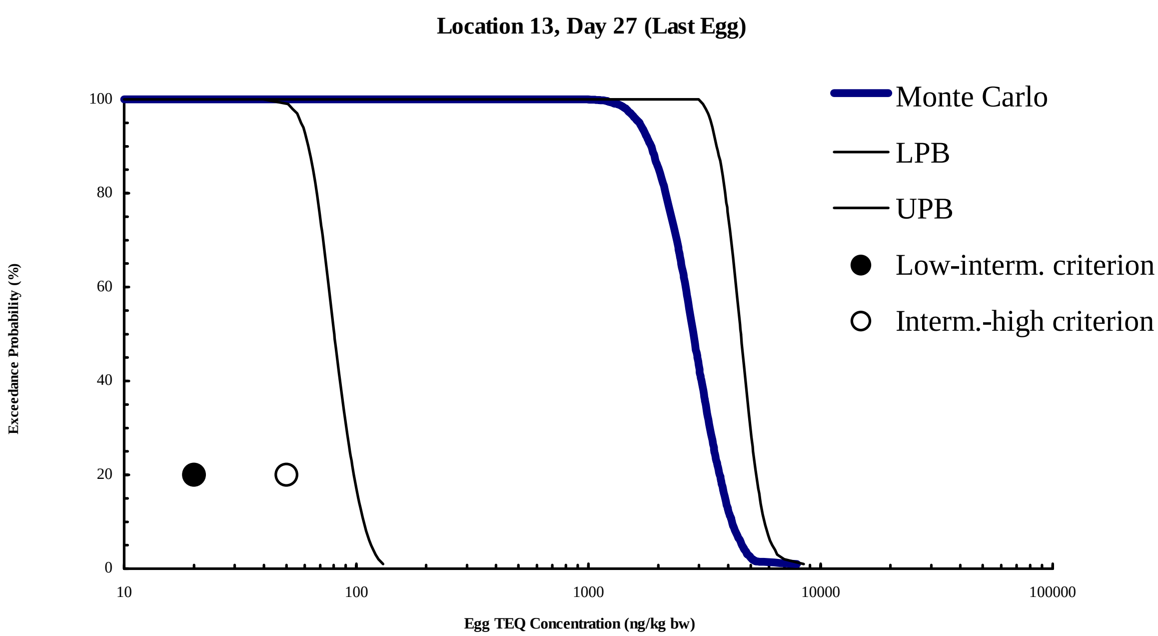
UPB = Upper probability bound

Figure 7.5-26 Exposure of Wood Duck Eggs to 2,3,7,8-TCDD TEQ at Location 14, 20 Days (7th Egg) After Arriving in the PSA



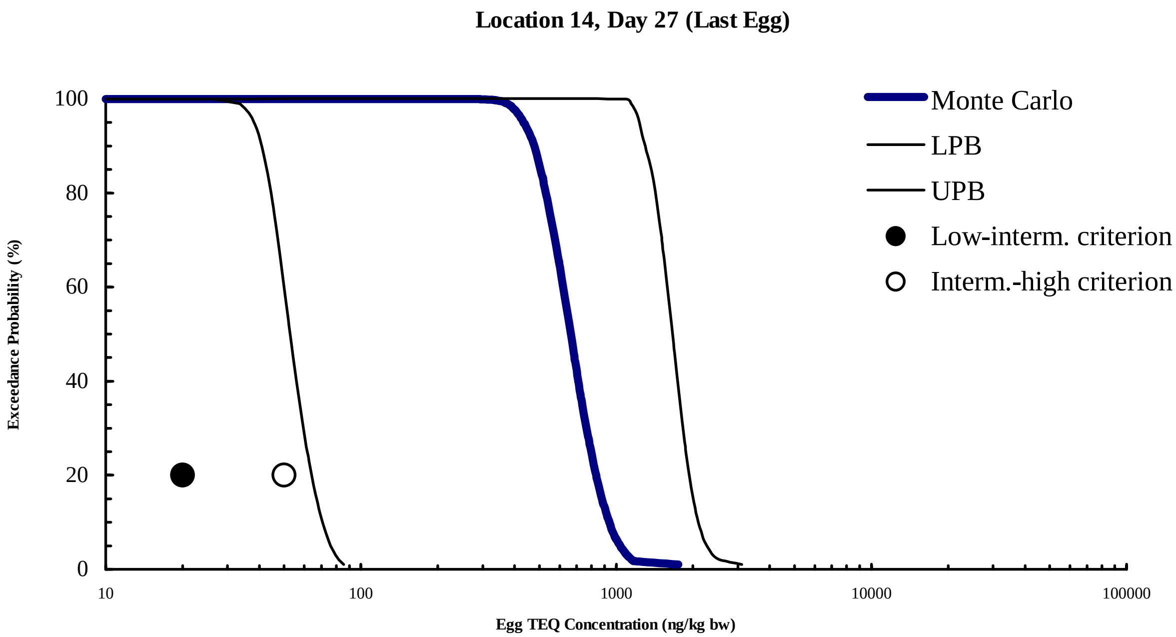
Notes: LPB = Lower probability bound
UPB = Upper probability bound

Figure 7.5-27 Exposure of Wood Duck Eggs to 2,3,7,8-TCDD TEQ at Location 15, 20 Days (7th Egg) After Arriving in the PSA



Notes:
LPB = Lower probability bound
UPB = Upper probability bound

Figure 7.5-28 Exposure of Wood Duck Eggs to 2,3,7,8-TCDD TEQ at Location 13, 27 Days (Last Egg) After Arriving in the PSA

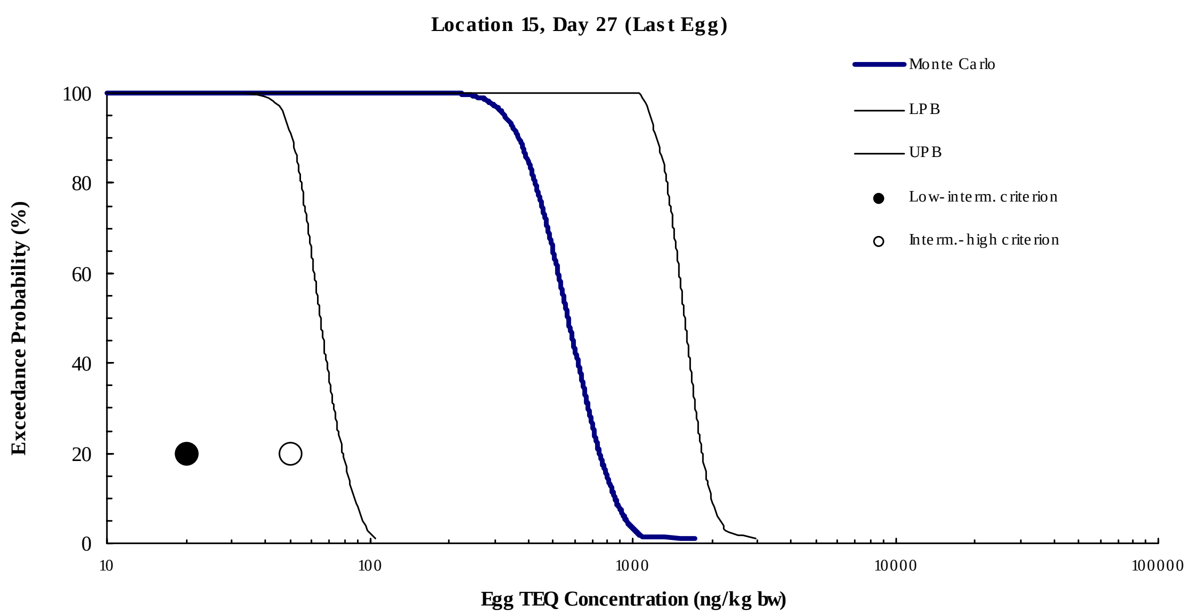


Notes:

LPB = Lower probability bound

UPB = Upper probability bound

Figure 7.5-29 Exposure of Wood Duck Eggs to 2,3,7,8-TCDD TEQ at Location 14, 27 Days (Last Egg) After Arriving in the PSA



Notes:

LPB = Lower probability bound

UPB = Upper probability bound

Figure 7.5-30 Exposure of Wood Duck Eggs to 2,3,7,8-TCDD TEQ at Location 15, 27 Days (Last Egg) After Arriving in the PSA

For each COC-location combination, a category of low, intermediate, or high risk was assigned, using the guidance below, when integrating the exposure and effects distributions.

Guidance for Integrating the Exposure and Effects Distributions

- If the probability of exceeding the lower toxicity threshold is less than 20%, the risk is considered to be low.
- If the probability of exceeding the upper toxicity threshold is greater than 20%, the risk is considered to be high.
- All other outcomes are considered to have intermediate risk.

This exercise was done separately for the results of the Monte Carlo analyses and the lower and upper bounds from the probability bounds analyses. The "risk category" refers to the level of risk based on the results of the Monte Carlo analyses. The "risk range" refers to the levels of risk based on the results of the probability bounds analyses.

The results of the risk characterization are summarized in Table 7.5-1. The highest risk to tree swallows is from exposure to tPCBs at the PSA sites and at the Taconic Valley site, which may not represent a true reference condition, with low risk at the two reference areas. As shown in Figure 7.5-6, the estimated exposure of tree swallows to tPCBs is greater than the upper bound threshold for adverse effects to this species. The risk category for tree swallows is high at the three PSA locations, and the risk range is low-high, indicating high uncertainty. The risk category and the risk range are low for the Southwest Branch and Threemile Pond reference locations; for the Taconic Valley location, the risk category is high and the risk range is low-high (Table 7.5-1). The highest risk to American robins and wood ducks is from exposure to tPCBs at Location 13. The risk category at Locations 14 and 15 is intermediate. The risk range from exposure to tPCBs at Locations 13 and 14 is intermediate-high, and at Location 15 the risk range is intermediate.

Table 7.5-1

Summary of Qualitative Risk Statements for Insectivorous Birds from the Housatonic River Study Area

Bird/Location	Qualitative Risk Statements				
	tPCBs			TEQ	
	Risk Category	Risk Range		Risk Category	Risk Range
Tree Swallow					
Holmes Road	High	Low/High		Intermediate	Intermediate
New Lenox Road	High	Low/High		Intermediate	Intermediate
Roaring Brook	High	Low/High		Intermediate	Intermediate
Southwest Branch	Low	Low		Intermediate	Low/Intermediate
Threemile Pond	Low	Low		Intermediate	Intermediate
Taconic Valley	High	Low/High		Intermediate	Intermediate
American Robin					
Location 13	High	Intermediate/High		Intermediate	Intermediate
Location 14	Intermediate	Intermediate/High		Intermediate	Low/Intermediate
Location 15	Intermediate	Intermediate		Low	Low/Intermediate
Wood Duck					
Location 13	High	Intermediate/High		High[a]	High[a,b]
Location 14	Intermediate	Intermediate/High		High[a]	High[a,b]
Location 15	Intermediate	Intermediate		High[a]	High[a,b]

[a] The qualitative risk statements were the same for wood duck eggs at Day 14, 20 and 27 at each of the locations modeled.

[b] Risk Range is considered High based on the increasing egg concentrations estimated in the Monte Carlo Analysis from Day 14 to Day 27 and the high risk range assigned to wood duck eggs on Day 14.

Tree swallows are at intermediate risk from exposure to TEQ in the PSA and in the reference locations, with risk ranges of intermediate for all sites except Southwest Branch, which had a risk range of low-intermediate (Table 7.5-1). Similarly, American robins are at intermediate risk from exposure to TEQ at Locations 13 and 14, and are at low risk at Location 15. The risk range from exposure to TEQ at Location 13 is intermediate, and at Locations 14 and 15 the risk range is low/intermediate. Wood ducks are at high risk at all locations and for all modeled eggs (Day 14 through Day 27). The risk range is also high in all cases. As described in Section 7.4.5.4, the TEQ effects metric calculated for wood duck eggs used International TEFs (I-TEFs) (EPA 1989), rather than WHO TEFs (Van den Berg et al. 1998). WHO TEFs were used to estimate exposure of wood duck to TEQ in the PSA. The TEQ effects metric for wood duck eggs would have been 0.94 to 1.77-fold higher if WHO avian TEFs had been used instead of I-TEFs (see Attachment II in Appendix G). Even with the highest possible TEQ effects metric under the WHO system (i.e., the I-TEFs threshold range multiplied by 1.77), the risk categorization would remain high for wood ducks at Locations 13, 14, and 15. There would only be a slight change in risk range for Locations 14 and 15 (intermediate to high instead of high to high) and no change in risk range for Location 13.

The complete characterization of risks of tPCBs and TEQ to insectivorous birds is presented in Appendix G.

7.5.2 Tree Swallow Field Study

The tree swallow reproduction study (Custer 2002) indicated that tree swallows did not experience serious adverse effects, despite high tissue concentrations of tPCBs and TEQ in nestlings in the PSA locations. The fecundity of tree swallows in the PSA was not significantly different from that of tree swallows generally in central Massachusetts as reported by Chapman (1955). McCarty and Secord (1999a) similarly reported large clutch sizes in tree swallows exposed to PCBs in the Upper Hudson River watershed, NY.

The geometric means of tPCB concentrations in tree swallow pippers and nestlings collected from the Housatonic River ranged from 32 to 101 mg/kg ww whole body. These are the highest concentrations reported in the literature (Custer 2002) and are substantially higher than concentrations in samples obtained from reference sites (6 to 19 mg/kg ww whole body). Total PCBs, dioxins, and furans were negatively correlated with hatching success in 1998 and 1999, but the correlations were weak.

7.5.3 American Robin Field Study (GE)

The results of the American robin field study indicated that concentrations of tPCBs in American robin eggs and nestlings were significantly higher in the PSA compared to reference areas, yet there were no significant differences in the measures of effects included in this study. Clutch size, hatching success, and fledging success all exhibited no difference in target and reference areas and were within ranges typical for American robins (Brehmer and Anderson 1992; Kemper and Taylor 1981; Fleutsch and Sparling 1994).

7.5.4 Wood Duck Egg Analyses

As discussed in Section 7.3.4, five wood duck eggs were collected from the PSA and analyzed for tPCBs and TEQ congeners. The sampling locations are shown in Figure G.1-12, and the results of these analyses are presented in Table G.2-43. The mean tPCB concentration in the eggs was 50.6 mg/kg ww (range: 32.9 to 80 mg/kg), and the mean TEQ concentration was 1,336 ng/kg ww (range: 703 to 2,077 ng/kg).

The wood duck eggs were sampled from an area above Woods Pond in the PSA. These egg TEQ concentrations would, therefore, be expected to most closely correspond spatially with exposure modeling conducted TEQ egg concentrations at Location 15, because this was the location closest to the wood duck egg sampling location. The Monte Carlo predictions for egg TEQ concentrations ranged from 155 to 1732 ng/kg with a mean of 595 ng/kg for Location 15 on Day 14, the first day of laying (Table G.2-40). By the last egg (Day 27), predictions of egg TEQ at Location 15 had received their peak with a mean of 952 ng/kg and a range of 259 to 2816 ng/kg. These results indicate the egg model may be slightly under-estimating the TEQ concentrations in wood duck eggs from the PSA. However, the measured values fell within the probability bounds for Days 14 and 27 at Location 15.

7.5.5 Weight-of-Evidence Analysis

A weight-of-evidence analysis approach was used to assess risks of tPCBs and TEQ to insectivorous birds. The three-phase approach of Menzie et al. (1996) and the Massachusetts Weight-of-Evidence Workgroup was applied for this purpose, in which weight-of-evidence was reflected in the following three characteristics: (1) the weight assigned to each measurement endpoint, (2) the magnitude of response observed in the measurement endpoint, and (3) the concurrence among outcomes of the multiple measurement endpoints (see Section 2.9 for details).

A discussion of attributes considered in the WOE is provided in Section 2, and the rationale for weighting of measurement endpoints is provided in Appendix G. A summary of how attributes were weighted for the tree swallow, American robin, and wood duck lines of evidence is provided in Tables 7.5-2 to 7.5-4.

> For tree swallows exposed to tPCBs and TEQ, the field-based reproductive study line of evidence was given a high weighting, and the modeled exposure and effects line of evidence was given a moderate weighting.
>
> For American robins, the field study was given a moderate/high weighting for both tPCBs and TEQ. The modeled exposure and effects line of evidence was given moderate value for tPCBs and TEQ.
>
> For wood ducks, the modeled exposure and effects line of evidence was given a moderate and moderate/high weighting for tPCBs and TEQ, respectively. There was no field study conducted for wood ducks in the PSA.

Table 7.5-2

Weighting of Measurement Endpoints for Tree Swallow Weight-of-Evidence Evaluation

Attributes	Field Study Custer (2002)	Modeled Exposure and Effects for tPCBs and TEQ
I. Relationship Between Measurement and Assessment Endpoints		
1. Degree of Association	High	Moderate
2. Stressor/Response	Moderate/High	Moderate
3. Utility of Measure	High	Moderate
II. Data Quality		
4. Data Quality	High	Moderate
III. Study Design		
5. Site Specificity	High	Low/Moderate
6. Sensitivity	Moderate/High	Low/Moderate
7. Spatial Representativeness	High	Moderate
8. Temporal Representativeness	High	Moderate
9. Quantitative Measure	Moderate/High	Moderate/High
10. Standard Method	High	Moderate
Overall Endpoint Value	**High**	**Moderate**

Table 7.5-3

Weighting of Measurement Endpoints for American Robin Weight-of-Evidence Evaluation

Attributes	Field Study Henning (2002)	Modeled Exposure and Effects for tPCBs and TEQ
I. Relationship Between Measurement and Assessment Endpoints		
1. Degree of Association	High	Moderate
2. Stressor/Response	Moderate	Moderate
3. Utility of Measure	Moderate/High	Moderate
II. Data Quality		
4. Data Quality	Moderate/High	Moderate
III. Study Design		
5. Site Specificity	High	Low/Moderate
6. Sensitivity	Moderate/High	Low/Moderate
7. Spatial Representativeness	High	Moderate
8. Temporal Representativeness	Moderate	Moderate
9. Quantitative Measure	Moderate/High	Moderate/High
10. Standard Method	Moderate/High	Moderate
Overall Endpoint Value	**Moderate/High**	**Moderate**

Table 7.5-4

Weighting of Measurement Endpoints for Wood Duck Weight-of-Evidence Evaluation

Attributes	Modeled Exposure and Effects for tPCBs and TEQ
I. Relationship Between Measurement and Assessment Endpoints	
1. Degree of Association	Moderate (tPCB) Moderate/High (TEQ)
2. Stressor/Response	Moderate
3. Utility of Measure	Moderate (tPCB) Moderate/High (TEQ)
II. Data Quality	
4. Data Quality	Moderate/High
III. Study Design	
5. Site Specificity	Low/Moderate
6. Sensitivity	Moderate (tPCB) Moderate/High (TEQ)
7. Spatial Representativeness	Moderate
8. Temporal Representativeness	Moderate
9. Quantitative Measure	Moderate/High
10. Standard Method	Moderate/High
Overall Endpoint Value	**Moderate (tPCB)** **Moderate/High (TEQ)**

The attribute values, evidence of harm, and magnitudes of responses for tree swallows, American robins, and wood ducks are presented in Table 7.5-5 (tPCBs) and Table 7.5-6 (TEQ). For both tree swallows and American robins exposed to tPCBs (Table 7.5-5) and TEQ (Table 7.5-6), the modeled exposure and effects line of evidence indicated that there was evidence of harm, and that the magnitude was high risk for tPCBs and intermediate risk for TEQ. For both tPCBs and TEQ, the tree swallow field study line of evidence indicated there was evidence of harm, the American robin field study line of evidence indicated that there was little evidence of harm, and in both cases the magnitude was low. For wood ducks exposed to tPCBs and TEQ, the modeled exposure and effects line of evidence indicated there was evidence of harm, and that the magnitude was high.

Table 7.5-5

Evidence of Harm and Magnitude of Effects for Insectivorous Birds Exposed to tPCBs in the Housatonic River PSA

Measurement Endpoints	Weighting Value (High, Moderate, Low)	Evidence of Harm (Yes, No, Undetermined)	Magnitude (High, Intermediate, Low)
Field Study	High (Tree swallow) Moderate/High (American robin)	Yes (Tree swallow) No (American robin)	Low (Tree swallow) -- (American robin)
Modeled Exposure and Effects	Moderate (Tree swallows) Moderate (American robin) Moderate (Wood duck)	Yes (Tree swallow) Yes (American robin) Yes (Wood duck)	High (Tree swallow) High (American robin) High (Wood duck)

Table 7.5-6

Evidence of Harm and Magnitude of Effects for Insectivorous Birds Exposed to TEQ in the Housatonic River PSA

Measurement Endpoints	Weighting Value (High, Moderate, Low)	Evidence of Harm (Yes, No, Undetermined)	Magnitude (High, Intermediate, Low)
Field Study	High (Tree swallow) Moderate/High (American robin)	Yes (Tree Swallow) No (American robin)	Low (Tree Swallow) -- (American robin)
Modeled Exposure and Effects	Moderate (Tree swallow/American robin) Moderate/High (Wood duck)	Yes (Tree swallow) Yes (American robin) Yes (Wood duck)	Intermediate (Tree swallow/American robin) High (Wood duck)

The final component in the weight-of-evidence approach addresses the concurrence among the measurement endpoints as they relate to each assessment endpoint. The methodology for evaluating concurrence involves the use of a graphical method where measurement endpoints are plotted on a matrix that also includes the weight of each endpoint and degree of response. Tables 7.5-6 and 7.5-7 depict the outcomes for the representative species exposed to tPCBs and TEQ, respectively. The analyses were conducted separately for tPCBs and TEQ.

Table 7.5-7

Risk Analysis Summary for Insectivorous Birds Exposed to tPCBs in the Housatonic River PSA

Assessment Endpoint: Survival, growth, and reproduction of insectivorous birds

Harm/Magnitude	**Weighting Factors (increasing confidence of weight)**				
	Low	**Low/Moderate**	**Moderate**	**Moderate/High**	**High**
Yes/High			MEE (Tree swallow) MEE (American robin) MEE (Wood duck)		
Yes/Intermediate					
Yes/Low					FS (Tree swallow)
Undetermined					
No				FS (American robin)	

FS=Field study

MEE=Modeled exposure and effects

Table 7.5-8

Risk Analysis Summary for Insectivorous Birds Exposed to TEQ in the Housatonic River PSA

Assessment Endpoint:	Survival, growth, and reproduction of insectivorous birds

Harm/Magnitude	Weighting Factors (increasing confidence of weight)				
	Low	**Low/Moderate**	**Moderate**	**Moderate/High**	**High**
Yes/High				MEE (Wood duck)	
Yes/Intermediate			MEE (Tree swallow) MEE (American robin)		
Yes/Low					FS (Tree swallow)
Undetermined					
No				FS (American robin)	

FS = Field Study

MEE = Modeled exposure and effects

The results from the modeled exposure and effects line of evidence suggest that tPCBs and TEQ pose intermediate to high risks to tree swallows, American robins, and wood ducks living in the PSA. However, the more highly weighted field study line of evidence suggests that if effects are occurring, they are minor for the tree swallow and American robin. Although no field study was conducted on the wood duck, measured exposure in the field suggests that the modeled exposure provides a reasonable prediction of exposure concentrations in wood duck eggs.

7.5.6 Sources of Uncertainty

Some of the major sources of uncertainty associated with the assessment of risks of tPCBs and TEQ to insectivorous birds are briefly summarized below. A more complete list is presented in Appendix G.

- Assumption that dietary exposure represented the most important pathway for exposure of insectivorous birds to COCs.
- Use of the ratio of tissue concentrations in pippers and nestlings to indirectly estimate the amount of tPCBs and TEQ transferred from mothers to offspring via egg tissue.
- Estimation of free metabolic rates using allometric equations.
- Limited sample sizes for the analyses of COC concentrations in tree swallow stomach contents.
- Limited sample sizes for the analyses of COC concentrations in some prey items, including earthworms, litter invertebrates, and benthic invertebrates.
- Co-elution of PCB congeners PCB-123 and PCB-157.
- Many TEQ congeners detected at or below the method detection limit.
- Lack of controlled laboratory studies involving tree swallows, American robins, and wood ducks requiring use of surrogate species for effects assessment.
- Weak correlation between concentrations of tPCBs in tree swallow pippers and nestlings and hatching success.
- The wood duck egg TEQ model relied upon chemical absorption efficiency (CAE), and the concentration ratio of egg to adult ($CA_{e:a}$) to calculate final egg TEQ concentrations.

7.5.7 Conclusions and Extrapolation to Other Species

The WOE analysis indicated that exposure of insectivorous birds, such as tree swallows and American robins, to tPCBs and TEQ in the PSA is unlikely to lead to adverse reproductive effects. This conclusion, however, is uncertain because the lines of evidence did not produce concordant results. The lines of evidence used in this conclusion were the field-based tree swallow and American robin reproductive studies and the comparison of modeled exposure with effects. Adverse reproductive effects may occur in some insectivorous bird species based upon the results of the comparison of wood duck modeled exposure with the effects line of evidence.

Other insectivorous bird species common to the PSA include the bank swallow, northern rough-winged swallow, barn swallow, cliff swallow, chimney swift, common nighthawk, eastern kingbird, eastern phoebe, eastern bluebird, eastern towhee, gray catbird, hermit thrush, northern mockingbird, veery, wood thrush, American black duck, mallard, common merganser, Canada goose, hooded merganser, green-winged teal, and common goldeneye. A qualitative analysis was conducted to compare exposure of tree swallows, American robins, wood ducks and other insectivorous birds to tPCBs and TEQ. The major factors that influence avian exposure to tPCBs and TEQ include:

- Foraging behavior and dietary composition.
- Foraging and home range size.
- Species body weight and other life history characteristics.

Tree swallows and other insectivorous bird species were compared using these factors and the results are provided in Appendix G.4.6. A qualitative analysis of risk to these species indicates that the cliff swallow, eastern kingbird, eastern bluebird, and eastern towhee have a similar to lower level of risk compared to the representative species; barn swallow, common nighthawk, eastern phoebe, hermit thrush, northern mockingbird, veery, and wood thrush have a similar level of risk; and bank swallow, chimney swift, northern rough-winged swallow, and gray catbird have a similar to higher level of risk compared to the tree swallow. Relative to the wood duck, hooded mergansers are expected to have a higher level of risk, American black ducks are expected to have a similar level of risk, and mallard, common merganser, green-winged teal, and common goldeneye are expected to have a lower level of risk.

ERA Summary

The weight-of-evidence analysis indicates that tree swallow and American robins are likely at low risk in the PSA as a result of exposure to tPCBs and TEQ. This conclusion, however, is uncertain. Risks to tree swallows and American robins in the PSA are intermediate to high based on modeled exposure and effects, but field studies detected no obvious adverse reproductive effects in the PSA. The weight-of-evidence indicates that wood ducks are likely at high risk to tPCB and TEQ exposure.

A qualitative analysis of risk to insectivorous bird species common to the PSA indicates that the cliff swallow, eastern kingbird, eastern bluebird, and eastern towhee have a similar to lower level of risk compared to the tree swallow; barn swallow, common nighthawk, eastern phoebe, hermit thrush, northern mockingbird, veery, and wood thrush have a similar level of risk; and bank swallow, chimney swift, northern rough-winged swallow, and gray catbird have a similar to higher level of risk compared to the tree swallow. Relative to the wood duck, hooded mergansers are expected to have a higher level of risk, American black ducks are expected to have a similar level of risk, and mallard, common merganser, green-winged teal, and common goldeneye are expected to have a lower level of risk.

7.6 REFERENCES

Blancher, P.J. and D.K. McNicol. 1991. Tree swallow diet in relation to wetland acidity. *Canadian Journal of Zoology* 69:2629-2637.

Brehmer, P.M. and R.K. Andersen. 1992. Effects of urban pesticide application on nesting success of songbirds. *Bulletin of Environmental Contamination and Toxicology* 48:352-359.

Chapman, L.B. 1955. Studies of a tree swallow colony. *Bird Banding.* 26:45-70.

Clench, M.H. and R.C. Leberman. 1978. *Weights of 151 Species of Pennsylvania Birds Analyzed by Month, Age, and Sex*. Bulletin of the Carnegie Museum of Natural History. 87 pp.

Custer, C.M. 2002. Exposure and Effects of Chemical Contaminants on Tree Swallows Nesting Along the Housatonic River, Berkshire Co., Massachusetts, 1998-2000. Final Report to U.S. Environmental Protection Agency. USGS, Upper Midwest Environmental Sciences Center, La Crosse, WI.

Drobney, R.D. and Fredrickson. L.H. 1979. Food selection by Wood ducks in relation to breeding status. *J. Wildlife Manage*. 43: 109-120.

Dunning, J.B. 1984. *Body Weights of 686 Species of North American Birds.* Eldon Publishing, Cave Creek, Arizona.

EPA (U.S. Environmental Protection Agency). 1989. Interim procedures for estimating risks associated with exposure to mixtures of chlorinated dibenzo-p-dioxins and dibenzofurans (CDDs and CDFs) and 1989 update. EPA/625/3-89/016. Risk Assessment Forum. Washington, DC.

EPA (U.S. Environmental Protection Agency). 1993. *Wildlife Exposure Factors Handbook*. EPA/600/R-93/187a. Office of Research and Development. Washington, DC.

Fernie, K.J., J.E. Smits, G.R. Bortolotti, and D.M. Bird. 2001. Reproductive success of American kestrels exposed to dietary polychlorinated biphenyls. *Environmental Toxicology and Chemistry* 20:776-781.

Fleutsch, K.M. and D.W. Sparling. 1994. Avian nesting success and diversity in conventionally and organically managed apple orchards. *Environmental Toxicology and Chemistry* 13(10):1651-1659.

Grice, D. and Rogers, J.P. 1965. The Wood Duck in Massachusetts. Final Report. Massachusetts Division of Fisheries and Game. Final Report. Federal Aid in Wildlife Restoration Project No. W-19-RHenning, M.H. 2002. *Robin Productivity in the Housatonic River Watershed, Berkshire County, Massachusetts*. Prepared for: General Electric Company, Pittsfield, Massachusetts.

Hoffman, D.J., M.J. Melancon, P.N. Klein, C.P. Rice, J.D. Eisemann, R.K. Hines, J.W. Spann, and G.W. Pendleton. 1996. Developmental toxicity of PCB 126 (3,3,4,4,5-pentachlorobiphenyl) in nestling American kestrels (*Falco sparverius*). *Fundamental and Applied Toxicology* 34:188-200.

Kemper, D.L. and J.M. Taylor. 1981. Seasonal reproductive changes in the American robin (*Turdus migratorius*) of the Pacific Northwest. *Canadian Journal of Zoology* 59:212-217.

Kubiak, T.L., H.J. Harris, L.M. Smith, T.R. Schwartz, D.L. Stalling, J.A. Trick, L. Sileo, D.E. Docherty, and T.C. Erdman. 1989. Microcontaminants and reproductive impairment of the Forster's tern on Green Bay, Lake Michigan – 1983. *Archives of Environmental Contamination and Toxicology*. 18:706-727.Lillie, R.J., H.C. Cecil, J. Bitman, and G.F. Fries. 1974. Differences in response of caged white leghorn layers to various polychlorinated biphenyls (PCBs) in the diet. *Poultry Science* 53:726-732.

McCarty, J.P., and A.L. Secord. 1999b. Reproductive ecology of tree swallows (*Tachycineta bicolor)* with high levels of polychlorinated biphenyl contamination. *Environ. Toxicol. Chem.* 18:1433-1439.

Menzie, C., M.H. Henning, J. Cura, K. Finkelstein, J. Gentile, J. Maughan, D. Mitchell, S. Petron, B. Potocki, S. Svirsky, and P. Tyler. 1996. Special report of the Massachusetts Weight-of-Evidence Workgroup: A weight-of-evidence approach for evaluating ecological risks. *Human Ecological Risk Assessment* 2:277-304.

Moore, D.R.J., B.E. Sample, G.W. Suter, B.R. Parkhurst, and R.S. Teed. 1999. A probabilistic risk assessment of the effects of methylmercury and PCBs on mink and kingfishers along East Fork Poplar Creek, Oak Ridge, Tennessee, USA. *Environmental Toxicology and Chemistry* 18(12):2941-2953.

Nosek, J.A., S.R. Craven, J.R. Sullivan, S.S. Hurley, and R.E. Peterson. 1992. Toxicity and reproductive effects of 2,3,7,8-tetrachlorodibenzo-p-dioxin in ring-necked pheasant. *Journal of Toxicology and Environmental Health* 35:187-198.

Robertson, R.J., B.J. Stuchbury, and R.R. Cohen. 1992. Tree Swallow (*Tachycineta bicolor*). In *The Birds of North America*, No. 11, A. Poole, P. Stettenheim, and F. Gill, Editors. Academy of Natural Sciences, Philadelphia, and American Ornithologists Union, Washington, DC.

Roberston, R.J., and J. Jones. 2002. Spatial and Demographic Effects on Tree Swallow Nest Quality and Reproductive Success. Department of Biology, Queen's University. Kingston, Ontario. Prepared for General Electric Company, April 2002.

Teather, K. 1996. Patterns of growth and asymmetry in nestling tree swallows. *Journal of Avian Biology* 27:302-310.

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

Verrett, M.J. 1970. Witness statement. Page 190. in Hearings before the subcommittee on energy, natural resources and the environment of the committee on commerce. U.S. Senate, Serial 9-60. U.S. Gov. Print. Off., Washington, D.C.

WESTON (Roy F. Weston, Inc.). 2002. *Rest of River Site Investigation Data Report*. Prepared for U.S. Army Corps of Engineers and U.S. Environmental Protection Agency. DCN GE-080202-ABDK.

White, D.H. and Seginak, J.T. 1994. Dioxin and furans linked reproductive impairment in Wood ducks. *J. Wildlife Management* 58(1):100-106.

8. ASSESSMENT ENDPOINT—SURVIVAL, GROWTH, AND REPRODUCTION OF PISCIVOROUS BIRDS

Highlights

Conceptual Model

Piscivorous birds, including the osprey and belted kingfisher, selected as representative species for the ERA, are exposed to contaminants of concern (COCs) via diet and trophic transfer. The assessment endpoint is the survival, growth, and reproduction of piscivorous birds in the Housatonic River PSA.

Exposure

COC intake by osprey and belted kingfisher was highest in Reaches 5 and 6 of the PSA, while exposure in the reference areas was much lower.

Effects

No information was available specifically on the toxicity of tPCBs and TEQ to belted kingfisher or osprey. A threshold range spanning sensitive and tolerant surrogate species was used instead for both tPCBs and TEQ.

Risk

Osprey are at high risk as a result of exposure to tPCBs and intermediate risk as a result of exposure to TEQ in the Housatonic River PSA. In these high-risk areas, modeled exposure of ospreys to PCBs is greater than doses that cause adverse effects in the most tolerant species studied. The weight-of-evidence (WOE) conclusion of high risk is uncertain because other lines of evidence (e.g., field surveys, in situ or whole media studies) were unavailable.

While modeled exposure and effects indicated high risk to belted kingfishers as a result of exposure to tPCBs and intermediate risk as a result of exposure to TEQ, a field study of kingfisher productivity indicated that the birds were reproducing in the PSA. The WOE assessment for belted kingfisher concluded that this species is at intermediate risk. This conclusion, however, is uncertain because of the lack of concordance between the two lines of evidence.

8.1 INTRODUCTION

This section summarizes the current and potential risks to piscivorous birds. A Pre-ERA was conducted to narrow the scope of the ERA by identifying contaminants, other than tPCBs, that pose potential risks to aquatic biota and wildlife in the PSA (Appendix B). Subsequent to the Pre-ERA, several other contaminants of potential concern (COPCs) (primarily organochlorine pesticides) were screened out because their actual concentrations in the PSA were likely much lower than the measured values due to laboratory interference (see Section 2.4). Contaminants of concern (COCs) that were retained for the risk assessment for piscivorous birds were total

PCBs (tPCBs) and 2,3,7,8-tetrachlorodibenzo-p-dioxin (2,3,7,8-TCDD) toxic equivalence (TEQ).

8.1.1 Overview of Approach

A step-wise approach was used to assess the risks of tPCBs and TEQ to piscivorous birds in the Housatonic River watershed. The four main steps in this process include:

1. Derivation of a conceptual model (Figures 8.1-1 and 8.1-2).
2. Assessment of exposure of birds to COCs (Figure 8.1-3).
3. Assessment of the effects of COCs on birds (Figure 8.1-4).
4. Characterization of risk to the piscivorous avian species (Figure 8.1-5).

The detailed ecological risk assessment for piscivorous birds is provided in Appendix H.

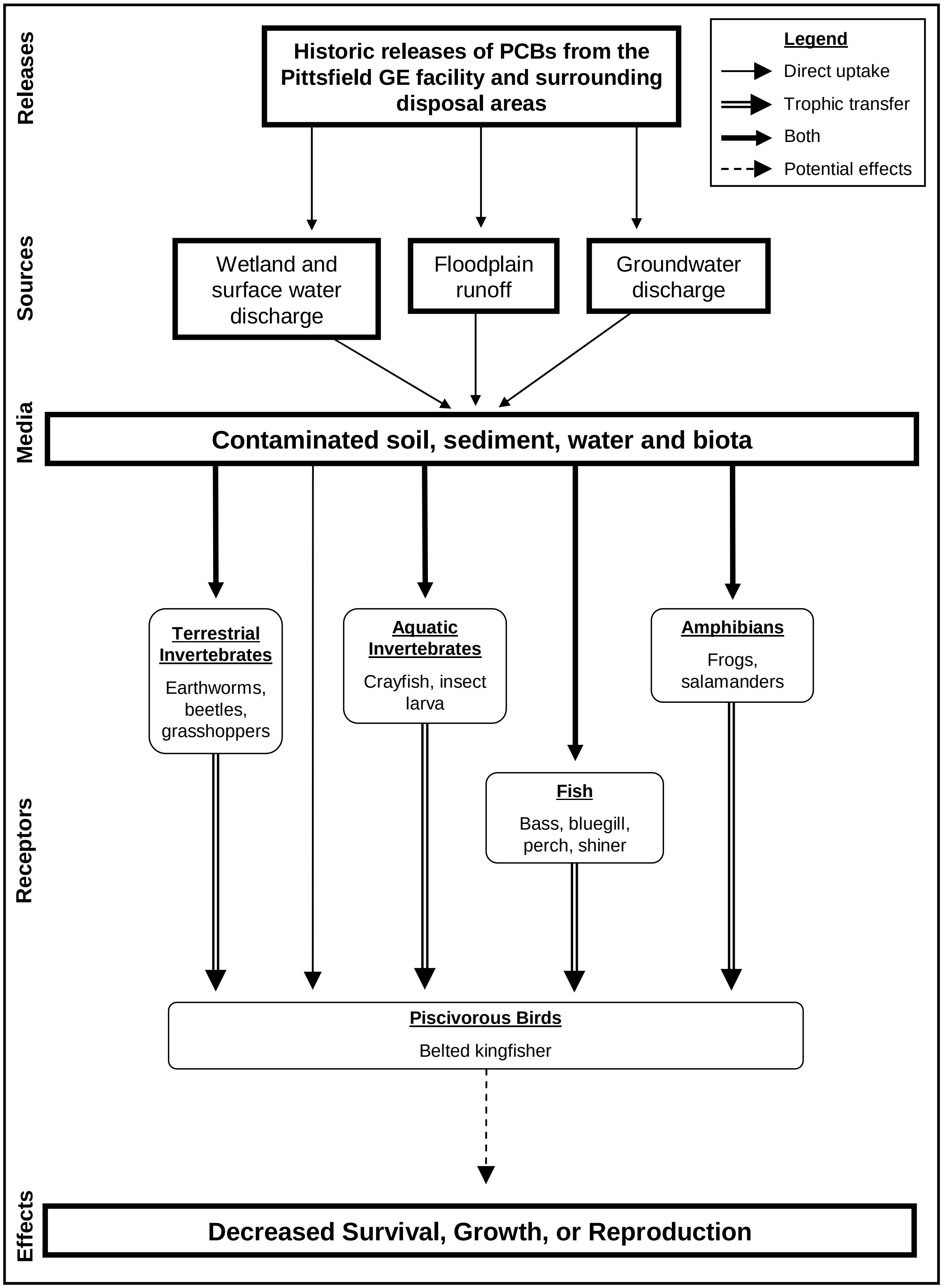


Figure 8.1-1 Conceptual Model for Exposure of Belted Kingfisher to tPCBs and TEQ in the Housatonic River PSA

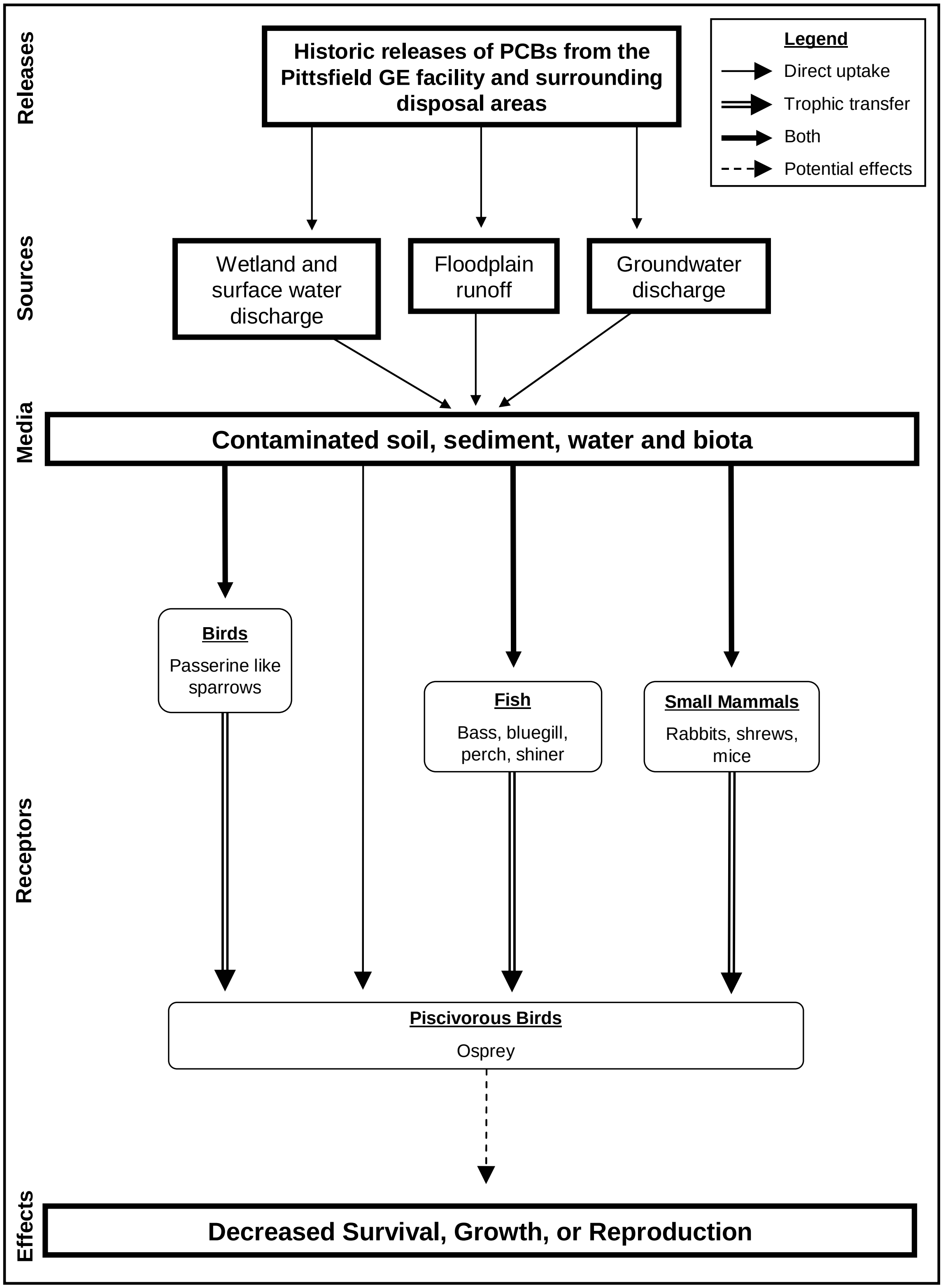


Figure 8.1-2 Conceptual Model for Exposure of Osprey to tPCBs and TEQ in the Housatonic River PSA

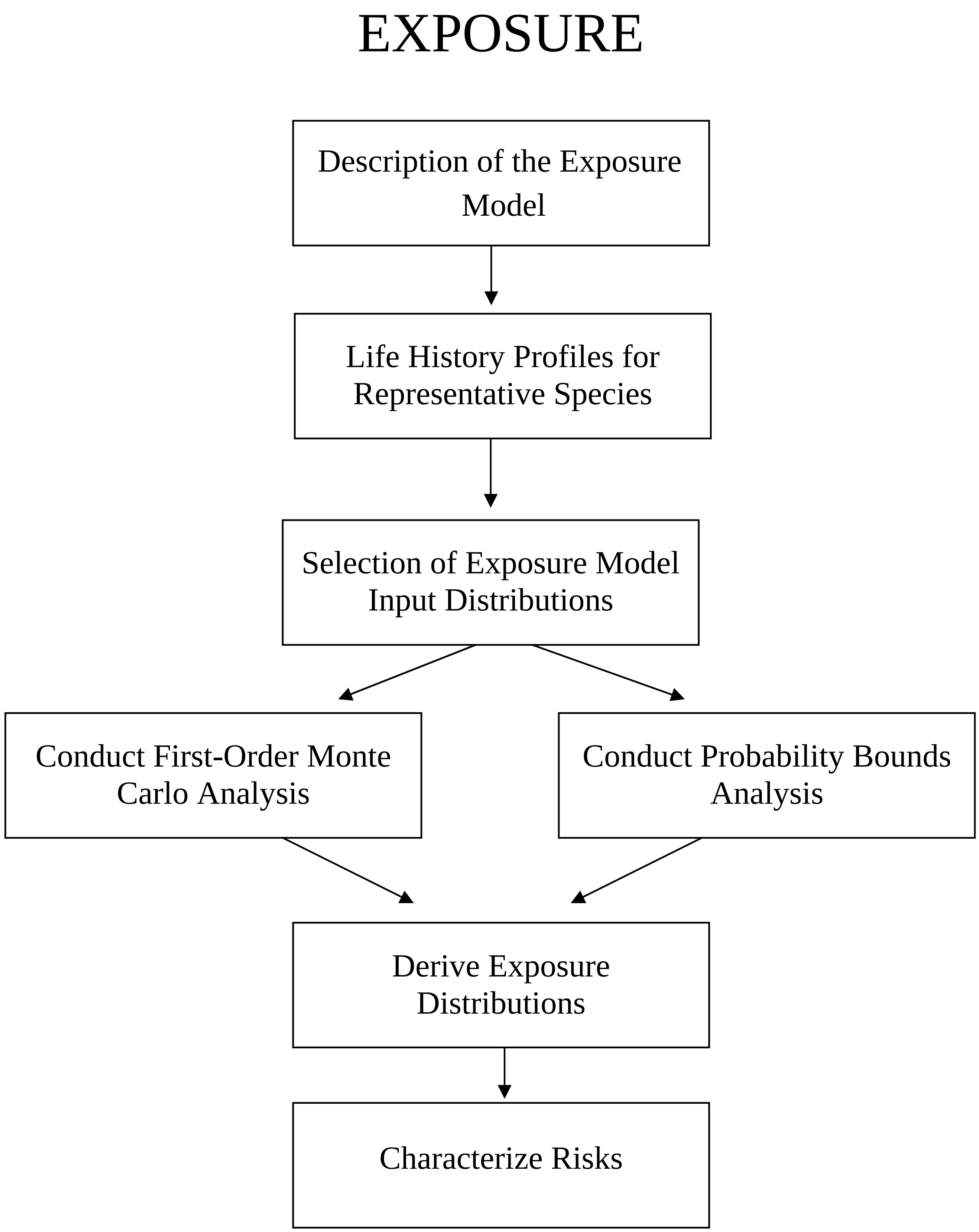


Figure 8.1-3 Overview of Approach Used to Assess Modeled Exposure of Piscivorous Birds to COCs in the Housatonic River PSA

EFFECTS

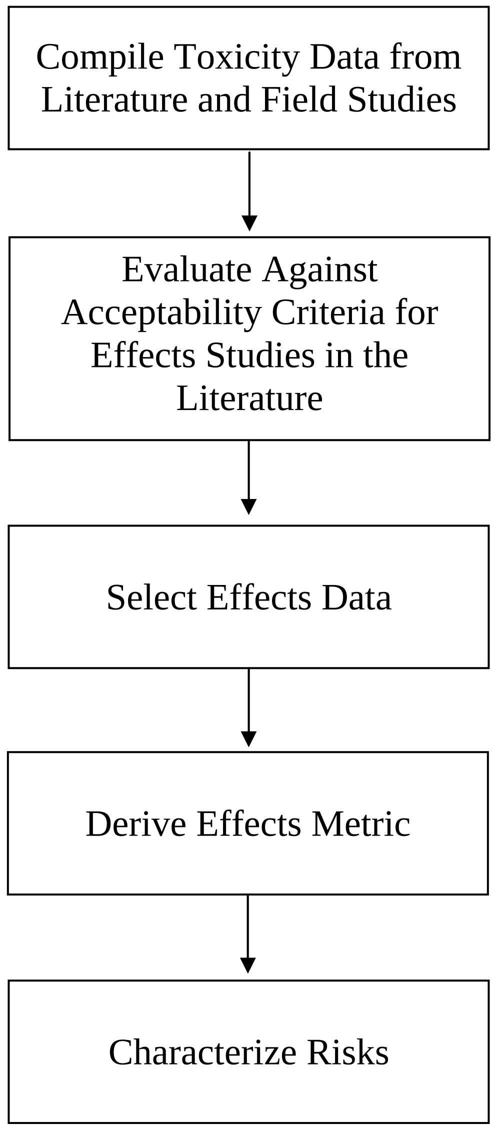


Figure 8.1-4 Overview of Approach Used to Assess the Modeled Effects of COCs to Piscivorous Birds in the Housatonic River PSA

RISK CHARACTERIZATION

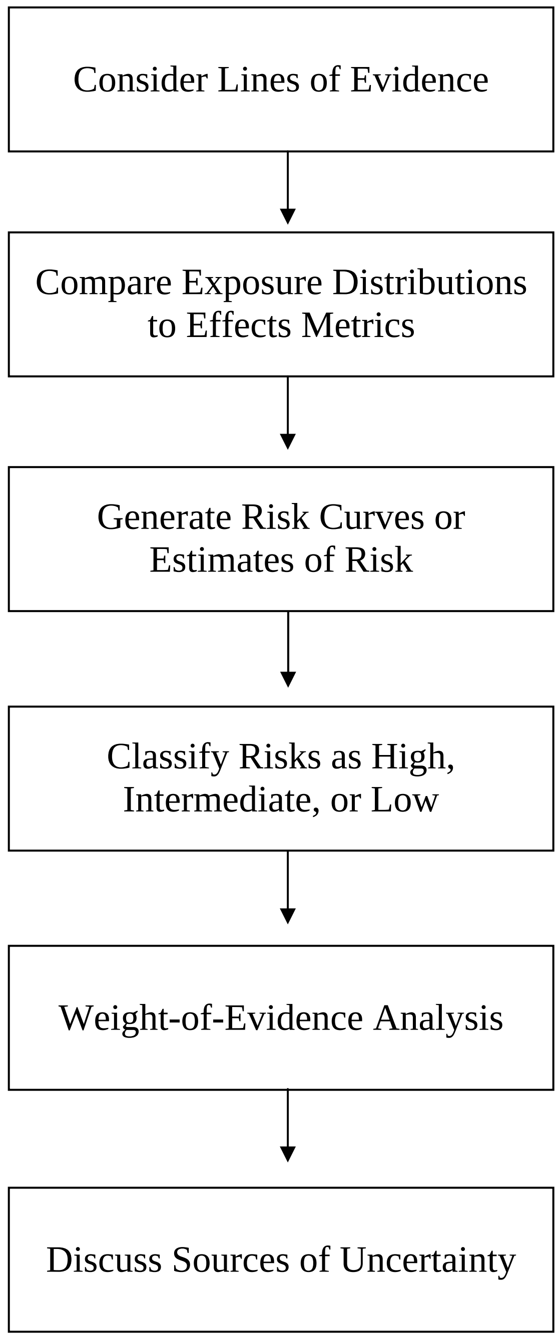


Figure 8.1-5 Overview of Approach Used to Characterize the Risks of COCs to Piscivorous Birds in the Housatonic River PSA

8.2 CONCEPTUAL MODEL

The conceptual model presented in Figures 8.1-1 and 8.1-2 illustrates the exposure pathways for piscivorous birds exposed to tPCBs and TEQ in the PSA. Piscivorous birds that reside, or partially reside, within the study area are exposed to tPCBs and TEQ principally through diet and trophic transfer. Other routes of exposure, considered to be less important to overall exposure, include inhalation, water consumption, and sediment ingestion (Moore et al. 1999).

The problem formulation (see Section 2 of the ERA) identified the belted kingfisher (*Ceryle alcyon*) and osprey (*Pandion haliaetus*) as the representative species for piscivorous birds potentially exposed to tPCBs and TEQ from consumption of contaminated prey in the PSA. Kingfishers have been observed nesting and breeding in the PSA, while observations of ospreys suggest that birds foraging in the PSA are currently transients. The PSA contains suitable habitat for osprey, with abundant prey, so there is a high likelihood that as the Massachusetts and Connecticut osprey population continues to expand, they may nest in the PSA. A pair of osprey were observed exhibiting courtship behavior in the PSA during the EPA field investigation.

Life history profiles for belted kingfisher and osprey are summarized in the following text boxes. Additional life history information on these species is provided in Appendix H.

The assessment endpoint that is the subject of this section is the survival, growth, and reproduction of piscivorous birds in the Housatonic River PSA. The measurement endpoints used to evaluate the assessment endpoint were based on the determination of the extent to which the concentrations of PCBs and TEQ ingested in the diet will impact the survival, reproduction, or growth of piscivorous birds. The assessment for piscivorous birds includes both a site-specific field study of kingfisher reproductive success, and comparisons of modeled exposures to doses reported in the literature to cause adverse effects.

Life History of Belted Kingfisher

The belted kingfisher is a pigeon-sized member of the family Alcedinidae and is a common bird in North America, excluding the far north and the higher elevations of the Rocky Mountains.

Habitat - Prefers foraging areas with clear water and visibility unobstructed by turbidity or aquatic vegetation. Size of territory depends upon the availability of prey, ranging from 0.5 to 1.36 miles (0.8 to 2.2 km) of shoreline.

Diet - Principal prey is fish, but may also feed on berries and other small animals, including mollusks, crustaceans, insects, amphibians, reptiles, young birds, and small mammals.

Life History of Osprey

Ospreys, also known as fish hawks or fishing eagles, are the only species in the family Pandionidae. The range of these raptorial birds covers almost all of North America, except for the extreme north.

Habitat - Use both fresh and saltwater ecosystems, but primarily the latter (Rattner et al. 2001). Ospreys are tree nesters, but have also adapted to man-made structures. Foraging ranges from 1.7 km to 15 km, depending on prey availability.

Diet - Almost exclusively piscivorous, preferring medium-sized fish (13 to 40 cm). On rare occasions, osprey will take dead fish or prey on small mammals, reptiles, and crustaceans.

8.3 EXPOSURE ASSESSMENT

This assessment focuses on Reaches 5 (excluding Reach 5A for osprey) and 6 (Woods Pond), also referred to as the Primary Study Area (PSA). Where possible, the exposure assessment for belted kingfisher was also conducted for two reference locations: East Branch of the Housatonic River, upstream of Dalton (hereinafter termed "upstream reference area"), and Threemile Pond in Sheffield, MA. Threemile Pond was the only reference area used in the exposure assessment for osprey because the habitat at the upstream reference area is not suitable for this species.

8.3.1 Exposure Model

The exposure model for kingfisher and osprey focuses on the ingestion of tPCBs and 2,3,7,8-TCDD TEQ through the diet. Other exposure routes (e.g., water, air) were considered to be of much less importance for tPCBs and TEQ, and were excluded from the analyses. The equation used to estimate exposure was adapted from the *Wildlife Exposure Factors Handbook* (EPA 1993) and related publications:

$$TDI = FT \times FIR \sum_{i=1}^{n} C_i \times P_i \qquad \text{(Eq. 1)}$$

where

- TDI = Total daily intake (mg/kg bw/d tPCBs, ng/kg bw/d TEQ).
- FT = Foraging time in the PSA (unitless).
- FIR = Normalized food intake rate (kg/kg bw/d).
- C_i = Concentration in the *i*th prey item (mg/kg for tPCBs, ng/kg for TEQ).
- P_i = Proportion of the *i*th prey item in the diet (unitless).

Because of differences in the size of their foraging ranges, exposure analyses for kingfisher were conducted separately for Reaches 5 and 6, and analyses for osprey were conducted for the PSA as a whole. The upstream reference area and Threemile Pond reference area were included for comparative purposes. Monte Carlo and probability bounds analyses were used to propagate input variable uncertainties through the exposure model for each COC. The results of these analyses are discussed in detail in Appendix H.

The input variable distributions used in the exposure models for piscivorous birds are summarized in the following sections and presented in detail in Appendix H.

8.3.1.1 Foraging Time (FT)

The foraging ranges of the two representative species are within the size of the PSA. Prey availability and an abundance of suitable foraging habitat suggest that the birds that forage in the PSA are able to meet their needs exclusively within this section of the river and floodplain. The assessment of risk to piscivorous birds inhabiting the PSA of the Housatonic River therefore focuses on those birds that spend 100% of their time foraging within the PSA.

8.3.1.2 Body Weight (BW)

Body weights of belted kingfishers vary only slightly with sex (Hamas 1994). Dunning (1993) reported a body weight range of 125 to 215 g with a mean of 148 g. Female ospreys are generally larger than males, weighing an average of 1.6 kg and 1.4 kg, respectively (Rattner et al. 2001).

8.3.1.3 Food Intake Rate (FIR)

The food intake rate of belted kingfishers has not been well characterized. The field-based estimate of the daily food intake rate of free-living adult kingfishers (0.50 g/g-day, Alexander 1977) was close to the 30^{th} percentile of the modeled food intake rate (see below) for these birds. Nagy (1987) and Nagy et al. (1999) derived allometric equations for estimating the metabolic rate of free-living birds using the following general equation:

$$\text{FMR (kJ/day)} = a \times BW(g)^{b} \qquad \text{(Eq. 2)}$$

The slope (*a*) and power (*b*) terms in Equation 2 were based on the error statistics derived from regression analysis of the data reported in Nagy et al. (1999). There were insufficient data to generate an allometric equation for Coraciiformes, of which belted kingfishers are members, so the equation for all birds was used.

The food intake rate of ospreys has not been well characterized either. The field-based measurements of the daily food intake rate of adult male ospreys (0.21 g/g-day, Poole 1983) were close to the 25th percentile of the modeled food intake rate described below. There were insufficient data to generate an allometric equation for Falconiformes, of which osprey are members, so the equation for Charadriiformes was used.

These input variable distributions are depicted in Figures H. 2-1 and H.2-10 in Appendix H. The body weights (*BW*) for these birds are described above. The results of the calculation were then converted to kcal/kg bw/d.

Food intake rate is derived from *FMR* using the following equation:

$$FIR(kg / kg\ bw / day) = FMR / \sum_{i=1}^{n} \left(AE_i \times GE_i \right) \qquad \text{(Eq. 3)}$$

where AE_i is the assimilation efficiency of the *i*th food item (unitless) and GE_i is the gross energy in the *i*th food item (kcal/g). For kingfisher, mean assimilation efficiencies were 77% for aquatic invertebrates and 79% for fish, and the mean gross energies were 1.1 kcal/g wet weight for aquatic invertebrates and 1.2 kcal/g wet weight for fish. For osprey, the mean assimilation efficiency of fish was 79% and the mean gross energy of fish was 1.2 kcal/g wet weight (EPA 1993). Point estimates were used for AE_i and GE_i in the Monte Carlo and probability bounds analyses because of their relatively small coefficients of variation (i.e., CV <10%).

8.3.1.4 Proportion of Dietary Items (P_i)

The principal prey of kingfishers is fish, but they also feed on berries and a variety of other small animals, including mollusks, crustaceans, insects, amphibians, reptiles, young birds, and small mammals (Hamas 1994). The exposure model uses a diet with a mean composition of 86% fish and 14% crayfish in Reach 5. Distributions of the proportion of kingfishers' dietary items in Reach 5 are depicted in Figure H.2-1. In Woods Pond and Threemile Pond, crayfish were not included in the kingfisher diet. The primary reasons for this include:

- The lack of observations of crayfish when conducting other field surveys during the last 3 years at these locations.
- The presence of aquatic vegetation, which conceals crayfish from kingfishers.
- The abundance of cyprinids and centrachids of forage size, which live in the shallow areas and are visually attractive to hunting kingfishers.

Therefore, the percent contribution of fish in the diet of kingfishers in Woods Pond and Threemile Pond was assumed to be 100%, and is not included in Figure H.2-1.

Osprey prefer to forage in shallow waters in lakes and rivers where fish occur near the surface and may be easily seen (DeGraaf and Yamasaki 2001). For this exposure assessment, it was assumed that fish account for 100% of the osprey diet. Proportion of fish in the osprey diet is a point estimate and therefore, is not shown in Figure H.2-10.

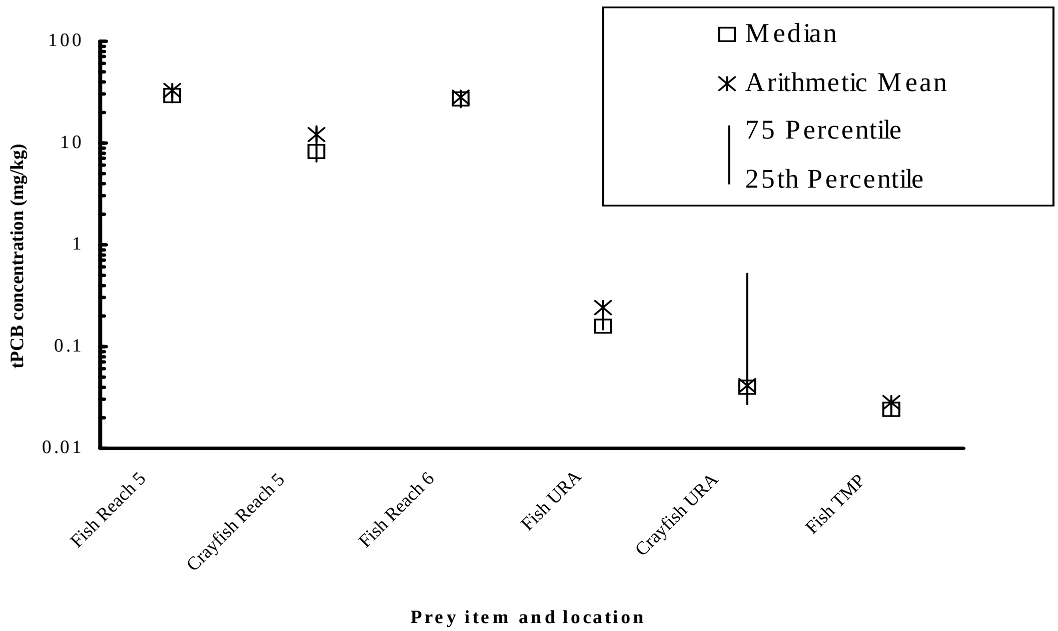
8.3.1.5 Concentration of COCs in Dietary Items (C_i)

The concentrations of tPCBs and TEQ in the prey of kingfisher and osprey are illustrated in Figures 8.3-1 to 8.3-4. The concentrations of COCs used in the exposure analyses are shown in Tables H.2-4, H.2-5, H.2-12, and H.2-13 of Appendix H.

8.3.2 Exposure Model Results

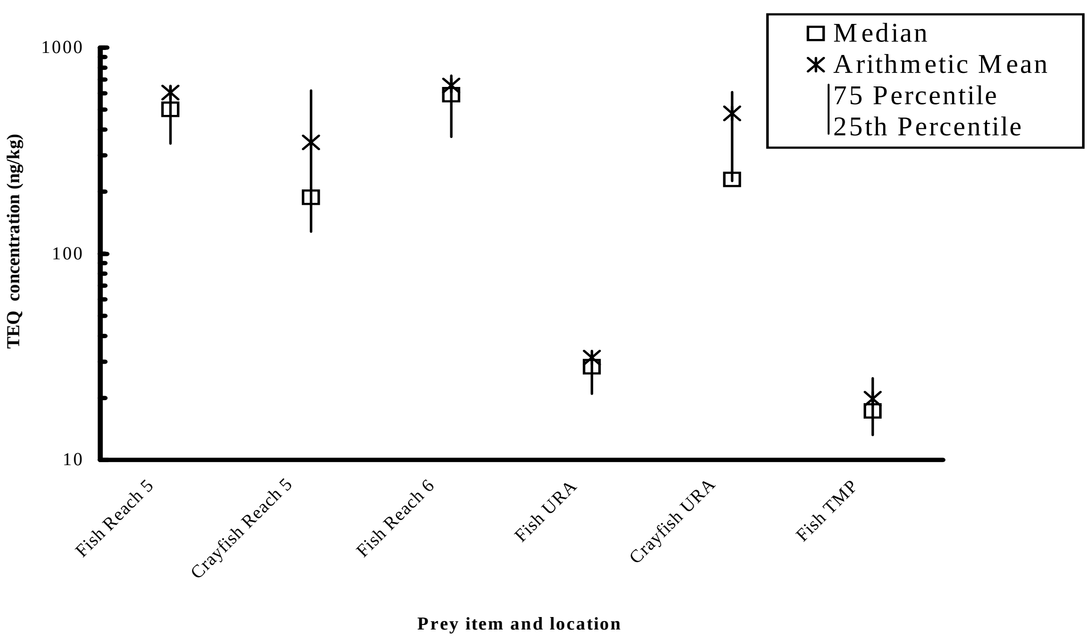
The exposure model results for belted kingfisher and osprey exposed to tPCBs and TEQ are discussed in greater detail in Section 2 of Appendix H.

Belted kingfisher had the highest modeled exposure to tPCBs in Reach 5 (Figure 8.5-1) and Reach 6 (Figure 8.5-2), whereas exposures in the reference areas (Figures 8.5-3 and 8.5-4) were substantially lower. Belted kingfisher had the highest modeled exposure to TEQ in Reach 5 (Figure 8.5-5) and Reach 6 (Figure 8.5-6), whereas exposures in the reference areas (Figures 8.5-7 and 8.5-8) were again substantially lower. Please note that these figures, which show the exposure modeling results, along with comparisons to effects metrics, are presented in Section 8.5 (Risk Characterization). This was done to avoid unnecessary duplication of figures that convey essentially the same information.



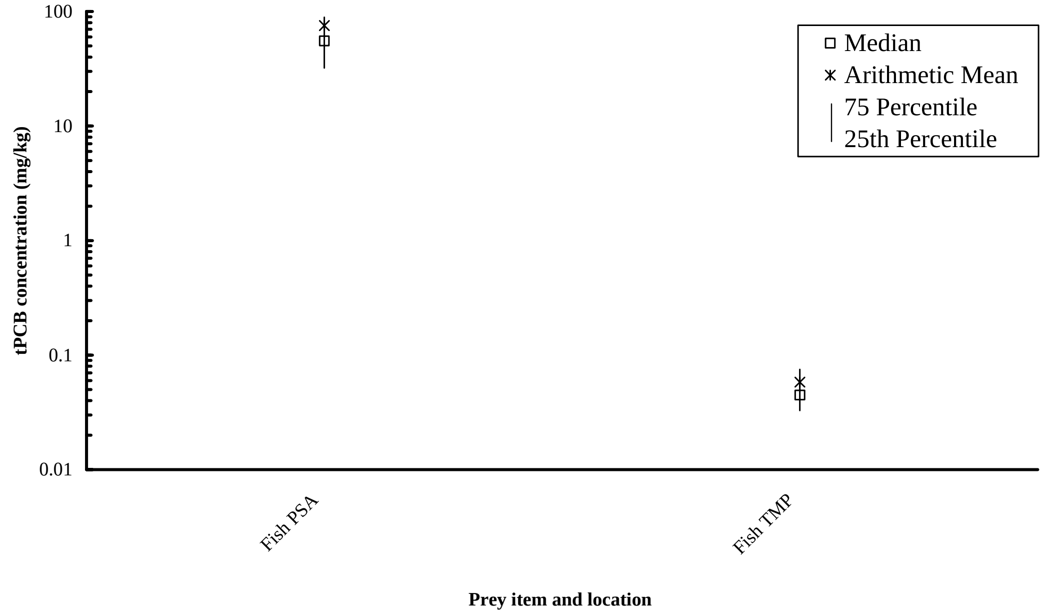
Notes: Error bars indicate interquartile range.
URA = Upstream reference area; TMP = Threemile Pond reference area.
Tissue concentrations are wet weight.

Figure 8.3-1 Concentrations of tPCBs in the Belted Kingfisher Diet in the Housatonic River PSA and Reference Areas



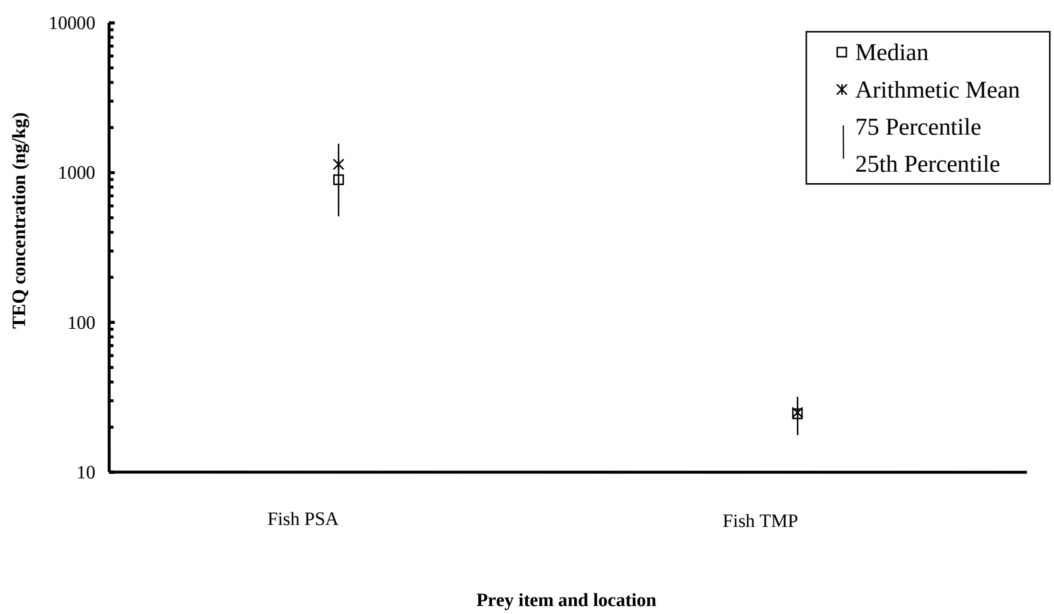
Notes: Error bars indicate interquartile range.
URA = Upstream reference area; TMP = Threemile Pond reference area.
Tissue concentrations are wet weight.

Figure 8.3-2 Concentrations of TEQ in the Belted Kingfisher Diet in the Housatonic River PSA and Reference Areas



Notes: Error bars indicate interquartile range.
PSA = Primary Study Area; TMP = Threemile Pond reference area.
Tissue concentrations are wet weight.

Figure 8.3-3 Concentrations of tPCBs in the Osprey Diet in the Housatonic River PSA and Reference Areas



Notes: Error bars indicate interquartile range.
PSA = Primary Study Area; TMP = Threemile Pond reference area.
Tissue concentrations are wet weight.

Figure 8.3-4 Concentrations of TEQ in the Osprey Diet in the Housatonic River PSA and Reference Areas

The exposure distributions for ospreys exposed to tPCBs in the PSA and the Threemile Pond reference area are presented in Figures 8.5-9 and 8.5-10, respectively. The exposure distributions for ospreys exposed to TEQ in the PSA and the Threemile Pond reference area are presented in Figures 8.5-11 and 8.5-12, respectively. Please note that these figures, which show the exposure modeling results, along with comparisons to effects metrics, are presented in Section 8.5 (Risk Characterization). This was done to avoid unnecessary duplication of figures that convey essentially the same information. Ospreys had the highest modeled exposure to tPCBs and TEQ in the PSA, while exposures in the reference areas were substantially lower.

Tables H.2-6 and H.2-14 present a summary of tPCB exposure model results for belted kingfishers and ospreys, respectively. Results for the TEQ exposure model for piscivorous birds are presented in Tables H.2-7 and H.2-15, respectively. A complete discussion of the exposure model results, including Monte Carlo and probability bounds analyses and figures and tables, is presented in Appendix H.

8.4 EFFECTS ASSESSMENT

The effects assessment has two objectives. The first is to review the scientific literature for effects of tPCBs (mainly Aroclor 1254 and 1260 mixtures) and TEQ to piscivorous birds, with documented effects to the representative species in this assessment, kingfisher and osprey, of primary interest. The other objective is to derive the effects metrics that will be used, in conjunction with the exposure assessment results, to estimate risks to kingfisher and osprey.

Figures 8.4-1 and 8.4-2 illustrate the ranges of effects of tPCBs and TEQ, respectively, to various avian species.

8.4.1 Effects Metrics

Effects data are ideally summarized as dose-response curves for each representative species. For this assessment, however, data were insufficient to generate dose-response curves, NOAELs and LOAELs, or field based measures of effect. Therefore, a threshold range for surrogate species was used to represent the effects of tPCBs and TEQ to piscivorous birds. This approach establishes a range of toxic effects thresholds for the most sensitive and tolerant avian species known and assumes that the thresholds for the representative species are within these bounds. Further details on the decision criteria used in selecting effects metrics are presented below and in Section 6.6.

8.4.1.1 Total PCBs

Based on a review of avian toxicity literature, white leghorn chickens were the most sensitive avian species to the reproductive effects of tPCBs (Lillie et al. 1974) and the most reproductively tolerant avian species to tPCBs was the American kestrel (Fernie et al. 2001). The resulting threshold range for the reproductive success of piscivorous birds exposed to tPCBs selected for this assessment is 0.12 to 7.0 mg/kg bw/d.

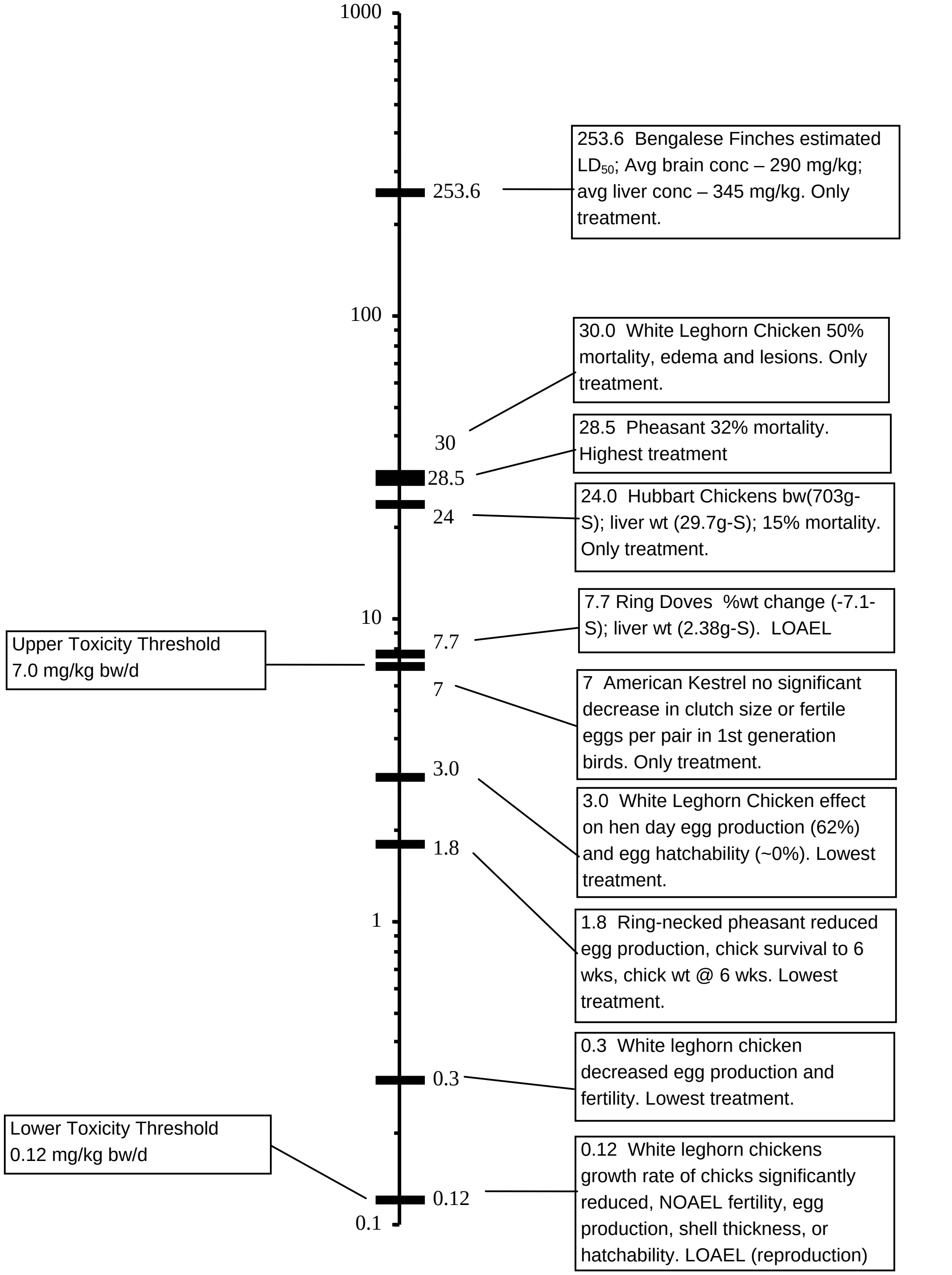
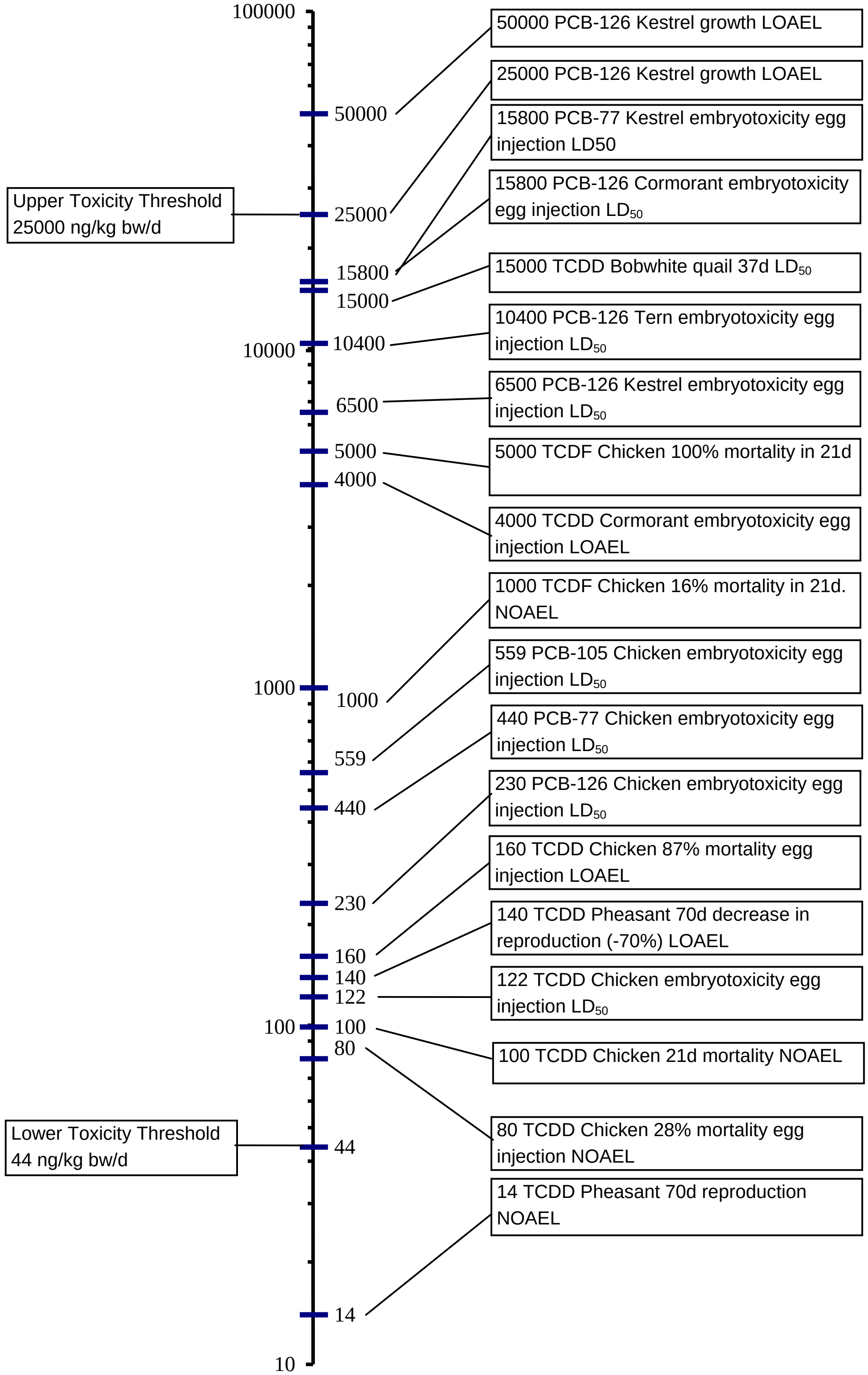


Figure 8.4-1 Effects of Aroclor 1254/1260 on Avian Species (mg/kg bw/d)



1
2 **Figure 8.4-2 Effects of 2,3,7,8-TCDD TEQ on Avian Species (ng TEQ/kg bw/d)**

Hierarchical Decision Criteria for Derivation of Effects Metric

- Have single-study bioassays with five or more treatments been conducted on the receptor of interest or a reasonable surrogate? If yes, estimate the concentration- or dose-response relationship. If not, go to step 2.
- Are multiple bioassays with similar protocols, exposure scenarios, and effects metrics available that, when combined, have five or more treatments for the receptor of interest or a reasonable surrogate? If yes, estimate the dose-response relationship as in step 1. If not, go to step 3.
- Have bioassays with less than five treatments been conducted on the receptor of interest or a reasonable surrogate? If yes, conduct or report results of hypothesis testing to determine the NOAEL and LOAEL. If not, go to step 4.
- Are sufficient data available from field studies and monitoring programs to estimate concentrations or doses of the COC that are consistently protective or associated with adverse effects? If yes, develop field-based effects metrics. If not, go to step 5.
- Derive a range where the threshold for the receptor of interest is expected to occur. Because information on the sensitivity of the receptor of interest is lacking, it is difficult to derive a threshold that is neither biased high nor low. If bioassay data are available for several other species, however, calculate a threshold for each to determine a threshold range that spans sensitive and tolerant species. That range is likely to include the threshold for the receptor of interest.

8.4.1.2 2,3,7,8-TCDD Toxic Equivalence (TEQ)

The lower toxicological threshold for the effects of TEQ to sensitive birds is based on ring-necked pheasants (Nosek et al. 1992) and the upper threshold for tolerant species is based on the American kestrel (Hoffman et al. 1996). The resulting threshold range for the reproductive success of piscivorous birds exposed to TEQ is 44 to 25,000 ng/kg bw/d.

8.4.2 Belted Kingfisher Field Study

A belted kingfisher reproduction study was performed in the PSA during the 2002 breeding season (Henning 2002). The objective of the study was to evaluate the relationship between reproduction of kingfishers and exposure of adult and nestling kingfishers to tPCBs. Nine belted kingfisher burrows were monitored during this study, three of which were depredated before the young could fledge. When depredated nests were excluded, fledging rates were consistent with the results of the only other kingfisher study reported in the literature (Brooks and Davis 1987). No significant relationships were observed between estimated tPCB dose and reproductive output (p>0.05), although this does not necessarily support a conclusion of no adverse effects to the reproductive success of belted kingfishers. See Section 8.5.2 and Section H.4.2 of Appendix H for discussion of this study.

8.5 RISK CHARACTERIZATION

This section characterizes risks to kingfisher and osprey exposed to tPCBs and 2,3,7,8-TCDD TEQ in the PSA of the Housatonic River. The risk characterization includes a comparison of probabilistic exposure estimates to relevant effect metrics, a review of the findings of the belted kingfisher field study, a summary of the weight-of-evidence (WOE) assessment, and a discussion of risks to other piscivorous birds foraging in the PSA.

Risk Questions

- Are the concentrations of tPCBs and TEQ present in the prey of piscivorous birds sufficient to cause adverse effects to individuals inhabiting the PSA of the Housatonic River?
- If so, how severe are the risks and what are their potential consequences?

Lines of Evidence

- Probabilistic exposure and effects modeling.
- Field study of the reproductive success of belted kingfishers in the PSA.

8.5.1 Comparison of Estimated Exposures to Laboratory-Derived Effect Doses

Exposure was assessed separately in Reaches 5 and 6 for kingfisher, and in the PSA (Reaches 5B, 5C, 5D, and 6 combined) for osprey. For each receptor-COC-area combination, a category of low, intermediate, or high risk was assigned using the guidance below, when integrating the exposure and effects distributions.

Guidance for Integrating the Exposure and Effects Distributions

- If the probability of exceeding the lower toxicity threshold is less than 20%, the risk is considered to be low.
- If the probability of exceeding the upper toxicity threshold is greater than 20%, the risk is considered to be high.
- All other outcomes are considered to have intermediate risk.

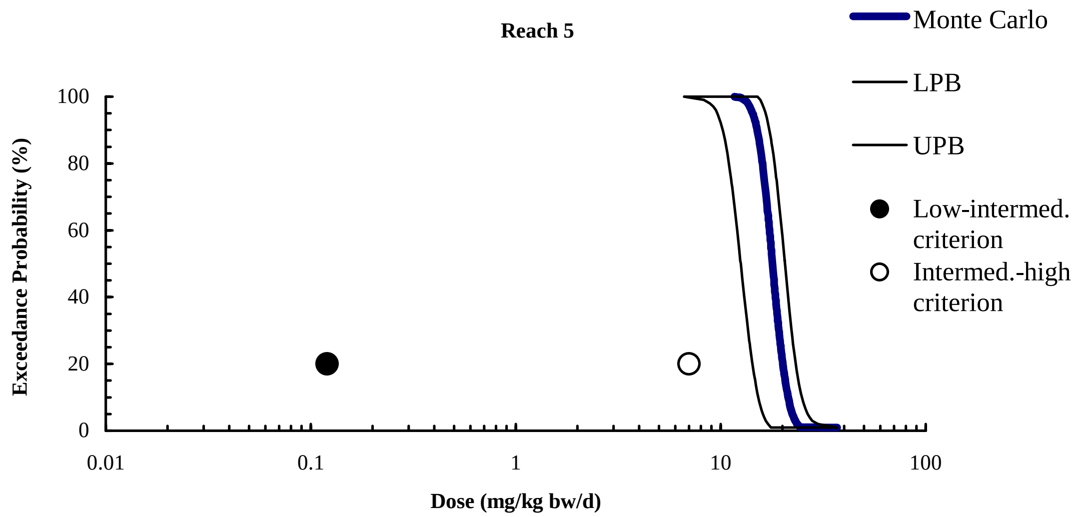
This exercise was done separately for the results of the Monte Carlo analyses and the lower and upper bounds from the probability bounds analyses. The "risk category" refers to the level of risk based on the results of the Monte Carlo analysis. The "risk range" refers to the levels of risk based on the results of the probability bounds analyses.

The results of the risk characterization for kingfisher and osprey, as described by the modeled exposure and effects line of evidence, are summarized in Tables 8.5-1 and 8.5-2, and in Figures 8.5-1 to 8.5-12. The risk category and risk range for kingfisher exposed to tPCBs in the PSA are both high. The risk category for kingfisher is estimated to be intermediate in the upstream reference area and low in the Threemile Pond reference area with risk ranges of low-intermediate and low, respectively.

Table 8.5-1

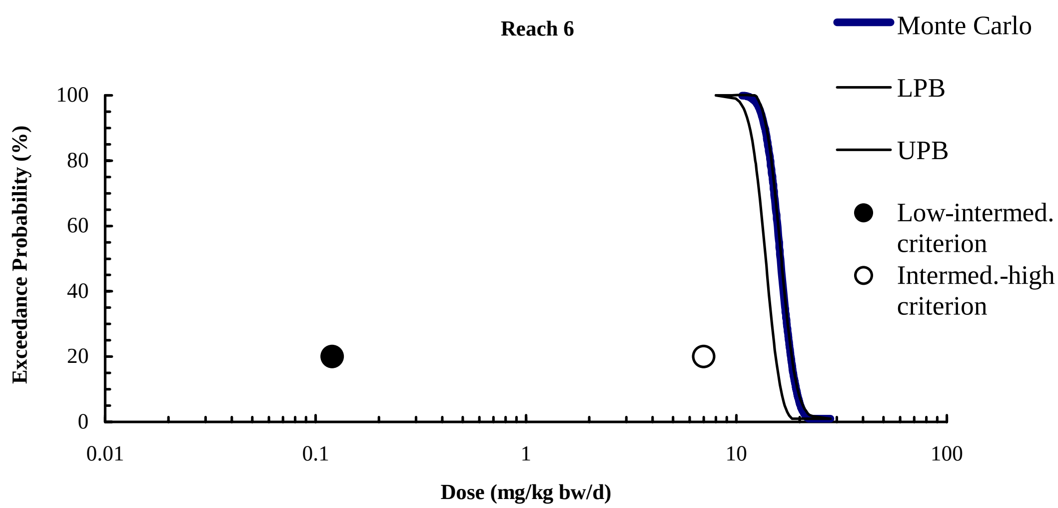
Summary of Qualitative Risk Statements for Belted Kingfisher from the Housatonic River PSA

Bird/Location	**Qualitative Risk Statements**				
	tPCBs			**TEQ**	
	Risk Category	**Risk Range**		**Risk Category**	**Risk Range**
Belted Kingfisher					
Reach 5	High	High		Intermediate	Intermediate
Reach 6	High	High		Intermediate	Intermediate
Upstream Reference Area	Intermediate	Low-intermediate		Intermediate	Low-intermediate
Threemile Pond	Low	Low		Low	Low



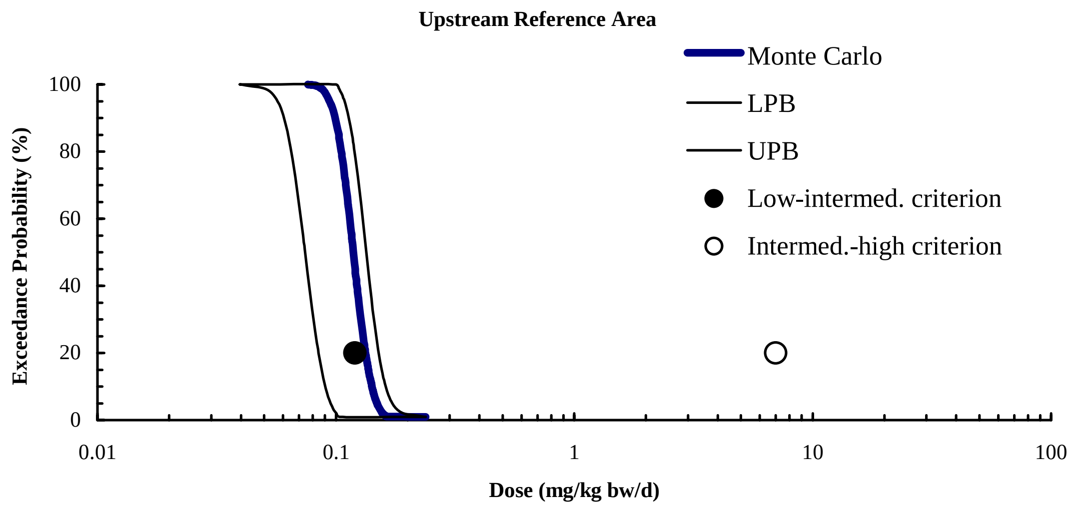
LPB = Lower probability bound
UPB = Upper probability bound

Figure 8.5-1 Risk to Belted Kingfisher from Exposure to tPCBs in Reach 5 of the Housatonic River PSA



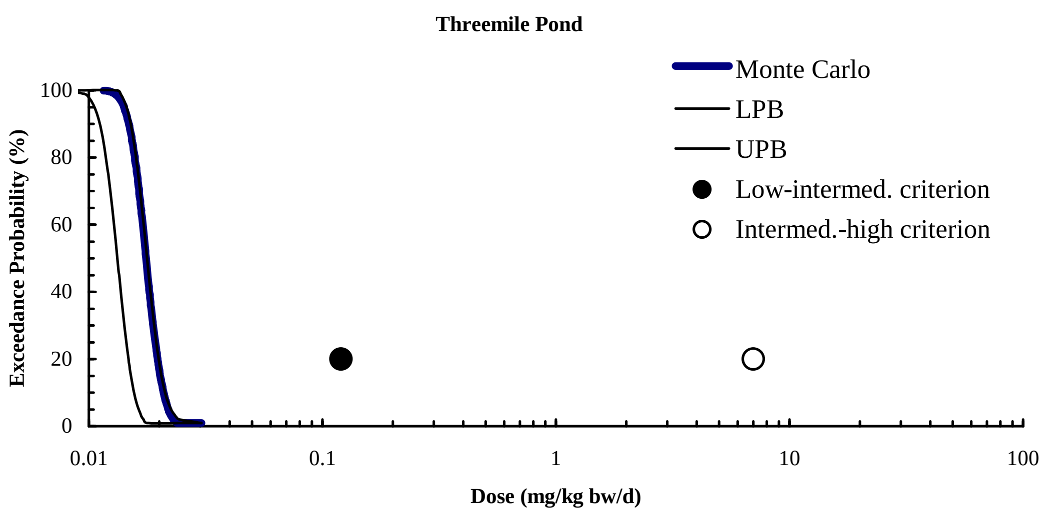
LPB = Lower probability bound
UPB = Upper probability bound

Figure 8.5-2 Risk to Belted Kingfisher from Exposure to tPCBs in Reach 6 of the Housatonic River PSA



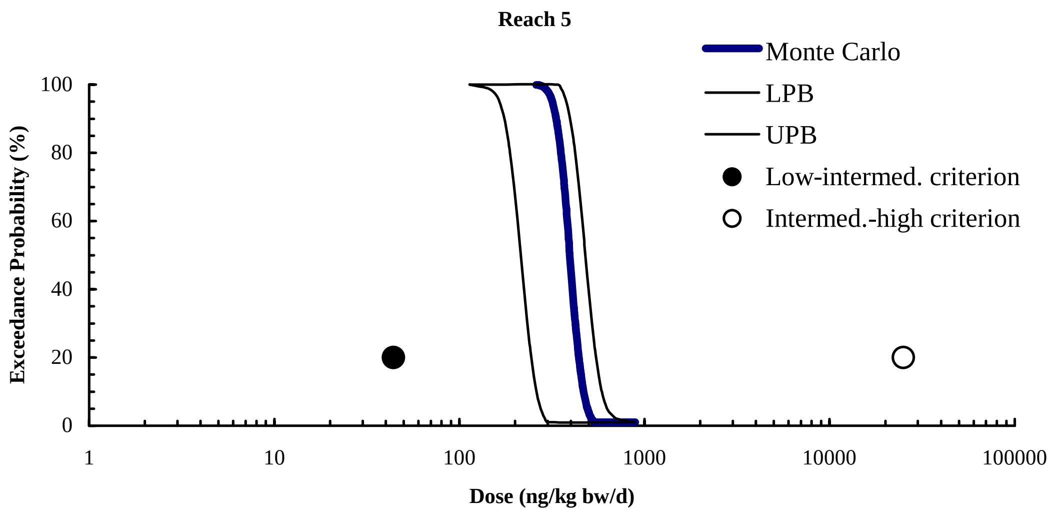
LPB = Lower probability bound
UPB = Upper probability bound

Figure 8.5-3 Risk to Belted Kingfisher from Exposure to tPCBs in the Housatonic River Upstream Reference Area



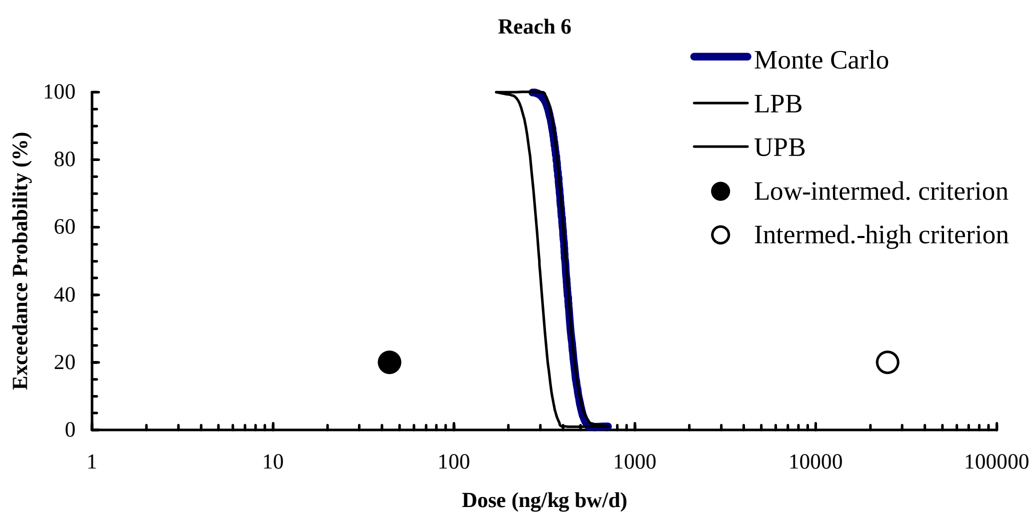
LPB = Lower probability bound
UPB = Upper probability bound

Figure 8.5-4 Risk to Belted Kingfisher from Exposure to tPCBs in the Threemile Pond Reference Area



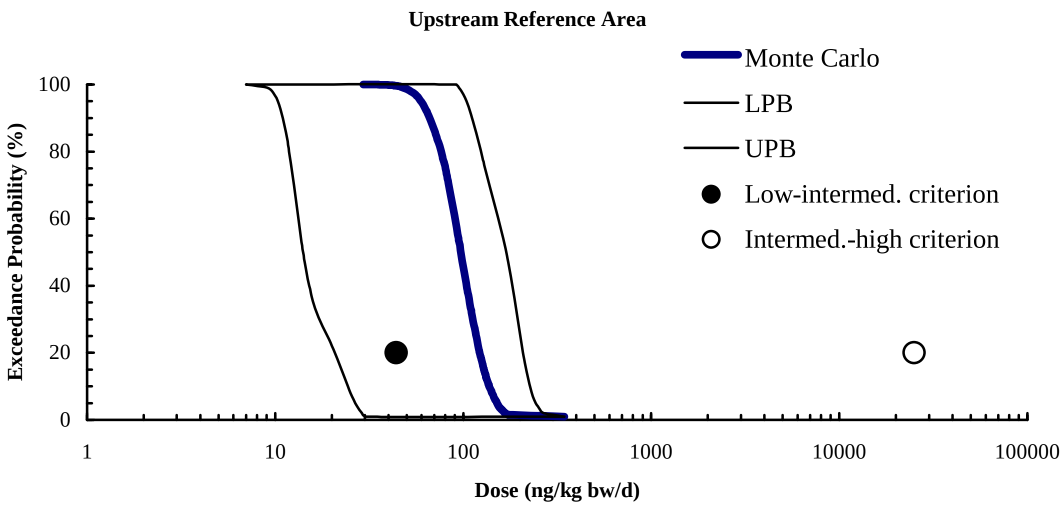
LPB = Lower probability bound
UPB = Upper probability bound

Figure 8.5-5 Risk to Belted Kingfisher from Exposure to TEQ in Reach 5 of the Housatonic River PSA



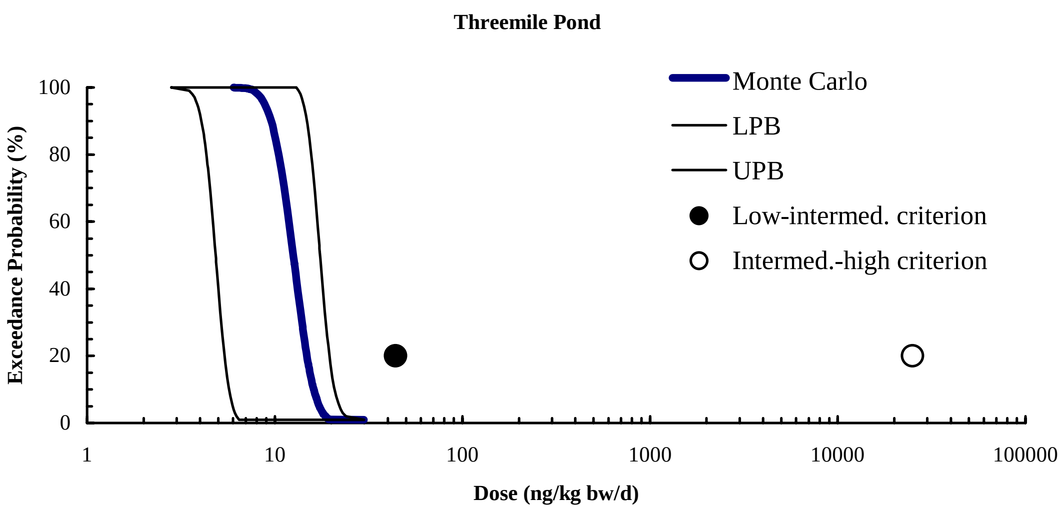
LPB = Lower probability bound
UPB = Upper probability bound

Figure 8.5-6 Risk to Belted Kingfisher from Exposure to TEQ in Reach 6 of the Housatonic River PSA



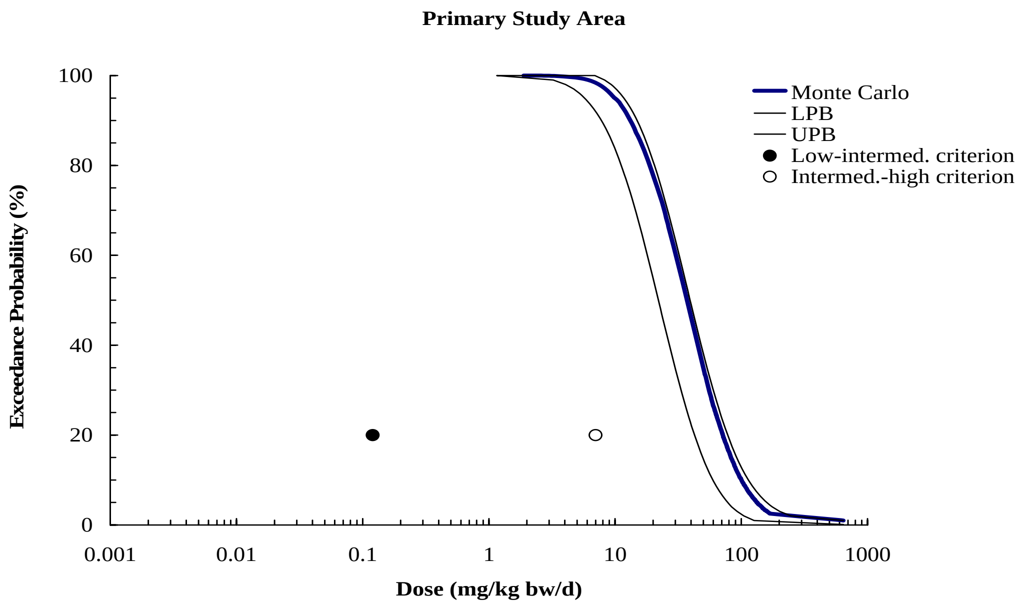
LPB = Lower probability bound
UPB = Upper probability bound

Figure 8.5-7 Risk to Belted Kingfisher from Exposure to TEQ in the Housatonic River Upstream Reference Area



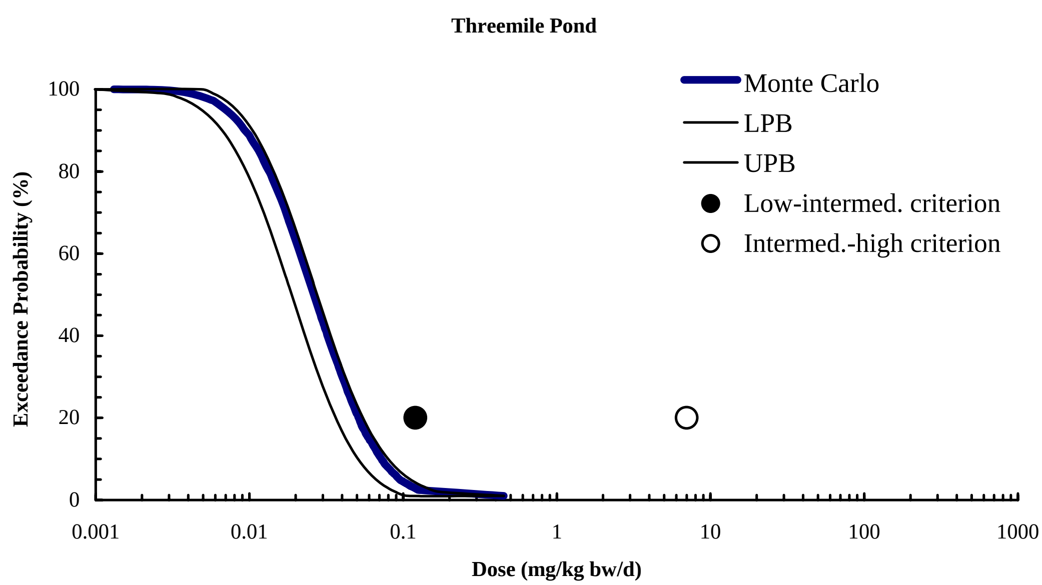
LPB = Lower probability bound
UPB = Upper probability bound

Figure 8.5-8 Risk to Belted Kingfisher from Exposure to TEQ in the Threemile Pond Reference Area



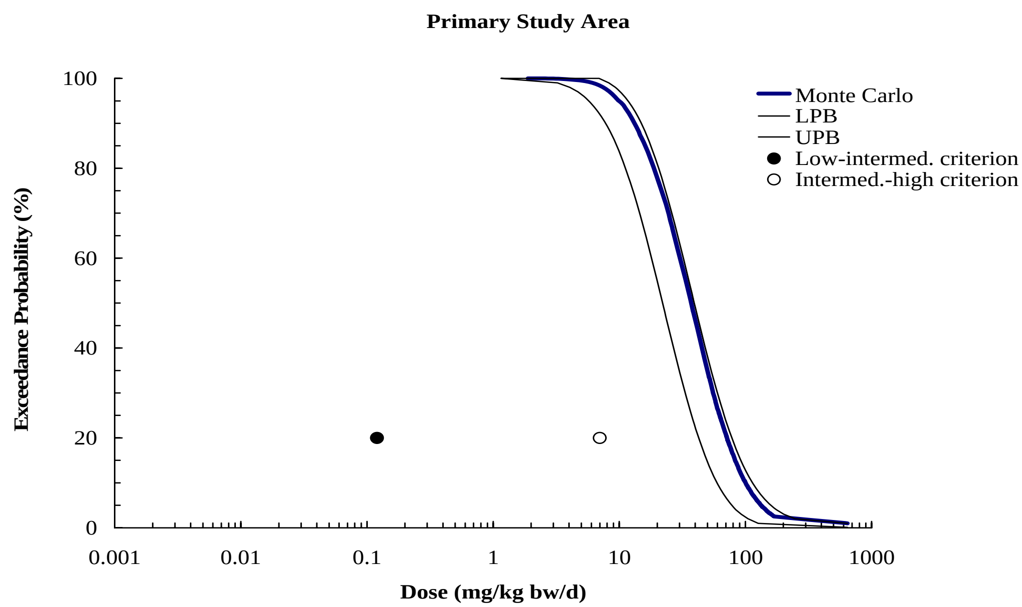
LPB = Lower probability bound
UPB = Upper probability bound

Figure 8.5-9 Risk to Osprey from Exposure to tPCBs in Reaches 5 and 6 of the Housatonic River PSA



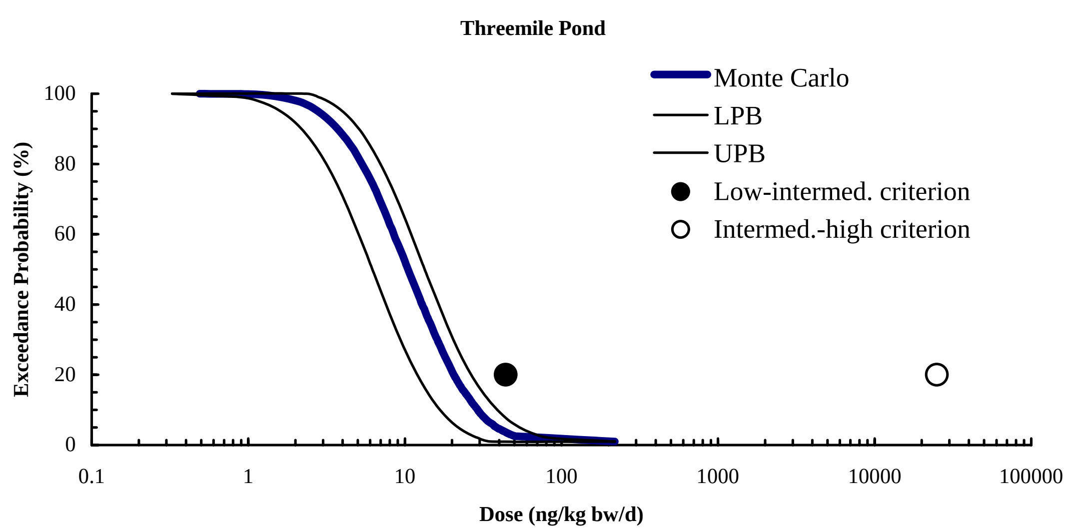
LPB = Lower probability bound
UPB = Upper probability bound

Figure 8.5-10 Risk to Osprey from Exposure to tPCBs in the Threemile Pond Reference Area



LPB = Lower probability bound
UPB = Upper probability bound

Figure 8.5-11 Risk to Osprey from Exposure to TEQ in Reaches 5 and 6 of the Housatonic River PSA



LPB = Lower probability bound
UPB = Upper probability bound

Figure 8.5-12 Risk to Osprey from Exposure to TEQ in the Threemile Pond Reference Area

Table 8.5-2

Summary of Qualitative Risk Statements for Ospreys from the Housatonic River PSA

Bird/Location	Qualitative Risk Statements				
	tPCBs			TEQ	
	Risk Category	Risk Range		Risk Category	Risk Range
Osprey					
PSA	High	High		Intermediate	Intermediate
Threemile Pond	Low	Low		Low	Low

As shown in Figure 8.5-9, the tPCB exposure curve for osprey in the PSA is well above the upper toxicity threshold. This means that the estimated daily intake of tPCBs by osprey is greater than the intake known to cause adverse effects in the most tolerant bird species studied. The risk category and risk range for osprey exposed to tPCBs in the PSA are both high (Table 8.5-2).

Both kingfisher and osprey were determined to be at intermediate risk to TEQ in the PSA. The complete characterization of risks of piscivorous birds to tPCBs and TEQ is presented in Appendix H.

8.5.2 Belted Kingfisher Field Study

A study of belted kingfisher reproduction in the PSA was performed by GE during the 2002 breeding season. The objective of the study was to determine the relationship between tPCB dose and reproductive success. More information on this study can be found in Henning (2002).

Henning (2002) reported that there were no significant relationships between estimated tPCB dose and any of the endpoints ($p>0.05$). The range of estimated total daily intakes in the PSA was narrow, however, and could not be replicated from the data, providing insufficient basis on which to evaluate a dose-response relationship. Based on the results of this study, the kingfisher population in the Housatonic River appears to be breeding successfully, with fledging rates and population densities that, when depredated nests are excluded, are similar to what was reported in the only other comparable study from the literature (Brooks and Davis 1987).

8.5.3 Weight-of-Evidence Analysis

A WOE analysis procedure was used to assess risks of tPCBs and TEQ to piscivorous birds. The three-phase approach of Menzie et al. (1996) and the Massachusetts Weight-of-Evidence Workgroup was applied for this purpose, in which WOE was reflected in the following three characteristics: (1) the weight assigned to each measurement endpoint; (2) the magnitude of response observed in the measurement endpoint; and (3) the concurrence among outcomes of the multiple measurement endpoints (see Section 2.9 for details).

The rationale for evaluating measurement endpoints is provided in Section 2.9 and Appendix H, along with a discussion of attributes considered in the WOE. The measurement endpoint weighting scores for kingfisher and osprey are presented in Tables 8.5-3 and 8.5-4, respectively. The overall weighting value, evidence of harm, and magnitude of response for kingfisher exposed to tPCBs and TEQ are presented in Tables 8.5-5 and 8.5-6, respectively. The corresponding information for osprey is presented in Tables 8.5-7 and 8.5-8. For both tPCBs and TEQ, the modeled exposure and effects line of evidence was given a moderate weighting. The belted kingfisher field study was also given a moderate weighting.

Table 8.5-3

WOE Evaluation for Measurement Endpoints for Belted Kingfisher

Attributes	Modeled Exposure and Effects for tPCBs and TEQ	GE Kingfisher Field Study (Henning 2002)
I. Relationship Between Measurement and Assessment Endpoints		
1. Degree of Association	M	M/H
2. Stressor/Response	M	L/M
3. Utility of Measure	M	M
II. Data Quality		
4. Data Quality	M/H	L/M
III. Study Design		
5. Site Specificity	L/M	M/H
6. Sensitivity	M	L/M
7. Spatial Representativeness	M	M/H
8. Temporal Representativeness	M	M
9. Quantitative Measure	M/H	M
10.Standard Method	M/H	M/H
Overall Endpoint Value	**M**	**M**

L = low
L/M = low to moderate
M = moderate
M/H = moderate to high
H = high

Table 8.5-4

WOE Evaluation for Measurement Endpoints for Osprey

Attributes	Modeled Exposure and Effects for tPCBs and TEQ
I. Relationship Between Measurement and Assessment Endpoints	
1. Degree of Association	M
2. Stressor/Response	M
3. Utility of Measure	M
II. Data Quality	
4. Data Quality	M/H
III. Study Design	
5. Site Specificity	L/M
6. Sensitivity	M
7. Spatial Representativeness	M
8. Temporal Representativeness	M
9. Quantitative Measure	M/H
10.Standard Method	M/H
Overall Endpoint Value	**M**

L = low
L/M = low to moderate
M = moderate
M/H = moderate to high
H = high

Table 8.5-5

Evidence of Harm and Magnitude of Effects for Belted Kingfisher Exposed to tPCBs in the Housatonic River PSA

Measurement Endpoints	Weighting Value (High, Moderate, Low)	Evidence of Harm (Yes, No, Undetermined)	Magnitude (High, Intermediate, Low)
Modeled Exposure and Effects	Moderate	Yes	High
Belted Kingfisher Field Study (Henning 2002)	Moderate	No	---

Table 8.5-6

Evidence of Harm and Magnitude of Effects for Belted Kingfisher Exposed to TEQ in the Housatonic River PSA

Measurement Endpoints	Weighting Value (High, Moderate, Low)	Evidence of Harm (Yes, No, Undetermined)	Magnitude (High, Intermediate, Low)
Modeled Exposure and Effects	Moderate	Yes	Intermediate
Belted Kingfisher Field Study (Henning 2002)	Moderate	No	---

Table 8.5-7

Evidence of Harm and Magnitude of Effects for Osprey Exposed to tPCBs in the Housatonic River PSA

Measurement Endpoints	Weighting Value (High, Moderate, Low)	Evidence of Harm (Yes, No, Undetermined)	Magnitude (High, Intermediate, Low)
Modeled Exposure and Effects	Moderate	Yes	High

Table 8.5-8

Evidence of Harm and Magnitude of Effects for Osprey Exposed to TEQ in the Housatonic River PSA

Measurement Endpoints	Weighting Value (High, Moderate, Low)	Evidence of Harm (Yes, No, Undetermined)	Magnitude (High, Intermediate, Low)
Modeled Exposure and Effects	Moderate	Yes	Intermediate

A graphical method was used for displaying concurrence among measurement endpoints. The analyses were conducted separately for belted kingfisher and osprey exposed to tPCBs and TEQ. Tables 8.5-9 and 8.5-10 depict the WOE for belted kingfisher exposed to tPCBs and TEQ, respectively, in the Housatonic River PSA. The results from the modeled exposure and effects lines of evidence suggest that tPCBs and TEQ pose a risk to belted kingfisher inhabiting the PSA of the Housatonic River. The belted kingfisher field study, however, suggests that the reproductive success of these birds may not be significantly impaired by exposure to tPCBs and TEQ in the PSA. Therefore, the WOE assessment supports a conclusion of intermediate risk to belted kingfisher exposed to tPCBs and TEQ in the PSA. Confidence in this conclusion is not high because the two lines of evidence (modeled exposure and effects, kingfisher field study) produced conflicting risk estimates. Risks in the reference areas for these COCs are generally low. The WOE assessment supports a conclusion of high risk for osprey exposed to tPCBs and intermediate risk for osprey exposed to TEQ in the PSA. This conclusion, however, is based on a single line of evidence (modeled exposure and effects) and therefore, is uncertain (Tables 8.5-11 and 8.5-12).

Table 8.5-9

Risk Analysis Summary for Belted Kingfisher Exposed to tPCBs in the Housatonic River PSA

Assessment Endpoint:	Survival, growth, and reproduction of piscivorous birds.

Harm/Magnitude	Weighting Factors (increasing confidence of weight)				
	Low	**Low/ Moderate**	**Moderate**	**Moderate/High**	**High**
Yes/High			MEE		
Yes/Intermediate					
Yes/Low					
Undetermined					
No			FS		

MEE = Modeled exposure and effects

FS = Field study

Table 8.5-10

Risk Analysis Summary for Belted Kingfisher Exposed to TEQ in the Housatonic River PSA

Assessment Endpoint: Survival, growth, and reproduction of piscivorous birds.

	Weighting Factors (increasing confidence of weight)				
Harm/Magnitude	**Low**	**Low/ Moderate**	**Moderate**	**Moderate/High**	**High**
Yes/High					
Yes/Intermediate			MEE		
Yes/Low					
Undetermined					
No			FS		

MEE = Modeled exposure and effects

FS = Field study

Table 8.5-11

Risk Analysis Summary for Osprey Exposed to tPCBs in the Housatonic River PSA

Assessment Endpoint:	Survival, growth, and reproduction of piscivorous birds.

	Weighting Factors (increasing confidence of weight)				
Harm/Magnitude	**Low**	**Low/ Moderate**	**Moderate**	**Moderate/High**	**High**
Yes/High			MEE		
Yes/Intermediate					
Yes/Low					
Undetermined					
No					

MEE = Modeled exposure and effects

Table 8.5-12

Risk Analysis Summary for Osprey Exposed to TEQ in the Housatonic River PSA

Assessment Endpoint: Survival, growth, and reproduction of piscivorous birds.

Harm/Magnitude	Weighting Factors (increasing confidence of weight)				
	Low	Low/ Moderate	Moderate	Moderate/High	High
Yes/High					
Yes/Intermediate			MEE		
Yes/Low					
Undetermined					
No					

MEE = Modeled exposure and effects

8.5.4 Sources of Uncertainty

Some of the major sources of uncertainty associated with the assessment of risks of tPCBs and TEQ to piscivorous birds are briefly summarized below. A more complete list is presented in Appendix H.

- Estimation of free metabolic rates using allometric equations.
- Limited sample sizes for the analyses of COC concentrations in some prey items, specifically, crayfish.
- Co-elution of PCB congeners PCB-123 and PCB-157.
- Lack of toxicity studies involving the representative species.
- Lack of reference site, lack of information regarding nest search intensity, inability to determine clutch size, insufficient visits to the nests during the reproductive cycle in the GE belted kingfisher field study (Henning 2002).
- Lack of an appropriately developed dose-response relationship in the GE belted kingfisher field study (Henning 2002).

8.5.5 Extrapolation to Other Species

Potential risks to other piscivorous birds (e.g., pied-billed grebe, great blue heron) are addressed in Section 11 and Appendix K.4.5.

8.5.6 Summary and Conclusions

The WOE analysis indicated that exposure of belted kingfisher and osprey, to tPCBs and TEQ in the PSA could lead to adverse reproductive effects. For belted kingfisher, the lines of evidence used to support this conclusion were the comparison of modeled exposure and effects to piscivorous birds and the field study of kingfisher reproductive success in the PSA. The modeled exposure and effects line of evidence was used to support the conclusion for osprey.

For the assessment of risks to kingfisher, both lines of evidence were considered. The modeled exposure and effects line of evidence indicated that kingfisher in the PSA are likely to receive a tPCB dose greater than what the most tolerant species studied can endure. For TEQ, the risk picture is less clear because the threshold range for this COC is very wide and the exposure estimates for kingfishers fell within this range. Thus, without effects data specific to kingfisher, it is difficult to make definitive conclusions about the risks of TEQ to this species. The field study of kingfisher productivity, however, indicated that these birds are able to reproduce in the PSA. Therefore, kingfishers are considered to be at intermediate risk in the PSA as a result of exposure to tPCBs and TEQ. The conclusion of intermediate risk to kingfisher is uncertain because the two lines of evidence did not give concordant results.

For osprey, only the modeled exposure and effects line of evidence was available to assess risk to this species because a site-specific study of the effects of COCs in the PSA to osprey was not conducted. As with kingfisher, this line of evidence indicated that osprey in the PSA are likely to receive a tPCB dose that is greater than what the most tolerant avian species studied could be exposed to without significant effects. The risks due to exposure to TEQ are unclear because the estimates for exposure also fell within the toxicity threshold range. The PSA contains suitable habitat for osprey with abundant prey. Therefore, osprey are estimated to be at risk in the PSA as a result of exposure to tPCBs and TEQ.

ERA Summary

The WOE analysis suggests that belted kingfishers may be at intermediate risk as a result of exposure to tPCBs and TEQ in the Housatonic River PSA. Although modeled exposure and effects indicated high risk for tPCBs and intermediate risk for TEQ, a field study of kingfisher productivity indicated that the birds were reproducing in the PSA. The conclusion of intermediate risk to kingfisher is highly uncertain because the two lines of evidence did not give concordant results.

Ospreys may be at high risk from exposure to tPCBs and intermediate risk from exposure to TEQ in the Housatonic River PSA. In the PSA, exposure of piscivorous birds to tPCBs is greater than concentrations that caused adverse effects in the most tolerant species studied. The conclusion of high risk to osprey is uncertain because only one line of evidence was available.

8.6 REFERENCES

Alexander, G.R. 1977. Food and vertebrate predators on waters in north central lower Michigan. *Michigan Academician* 10:187-195.

Brooks, R.P. and W.J. Davis. 1987. Habitat selection by breeding belted kingfishers. *American Midland Naturalist* 117:63-70.

DeGraaf, R.M. and M. Yamasaki. 2001. *New England Wildlife: Habitat, Natural History, and Distribution*. University Press of New England, Hanover, NH, USA.

Dunning, J.C. Jr. 1993. *CRC Handbook of Avian Body Masses*. CRC Press, Boca Raton, FL.

EPA (U.S. Environmental Protection Agency). 1993. *Wildlife Exposure Factors Handbook*. EPA/600/R-93/187a. Office of Research and Development, Washington, DC.

Hamas, M.J. 1994. Belted Kingfisher *(Ceryle alcyo*n). In *The Birds of North America, No. 84*. A. Poole and G. Gill, Editors. The Birds of North America, Inc., Philadelphia, PA.

Henning, M.H. 2002. *Productivity and Density of Belted Kingfishers on the Housatonic River, Berkshire County, Massachusetts.* Prepared for: General Electric Company, Pittsfield, Massachusetts.

Hoffman, D.J., M.J. Melancon, P.N. Klein, C.P. Rice, J.D. Eisemann, R.K. Hines, J.W. Spann, and G.W. Pendleton. 1996. Developmental toxicity of PCB-126 (3,3,4,4,5-pentachlorobiphenyl) in nestling American kestrels (*Falco sparverius*). *Fundamental and Applied Toxicology* 34:188-200.

Lillie, R.J., H.C. Cecil, J. Bitman, and G.F. Fries. 1974. Differences in response of caged white leghorn layers to various polychlorinated biphenyls (PCBs) in the diet. *Poultry Science* 53:726-732.

Menzie, C., M.H. Henning, J. Cura, K. Finkelstein, J. Gentile, J. Maughan, D. Mitchell, S. Petron, B. Potocki, S. Svirsky, and P. Tyler. 1996. Special report of the Massachusetts Weight-of-Evidence Workgroup: A weight-of-evidence approach for evaluating ecological risks. *Human Ecological Risk Assessment* 2:277-304.

Moore, D.R.J., B.E. Sample, G.W. Suter, B.R. Parkhurst and R.S. Teed. 1999. A probabilistic risk assessment of the effects of methylmercury and PCBs on mink and kingfishers along East Fork Poplar Creek, Oak Ridge, Tennessee, USA. *Environ. Toxicol. Chem*. 18:2941-2953.

Nagy, K.A. 1987. Free metabolic rate and food requirement scaling in mammals and birds. *Ecological Monographs* 57:111-128.

Nagy, K.A., I.A. Girard, and T.K. Brown. 1999. Energetics of free-ranging mammals, reptiles, and birds. *Annual Review of Nutrition* 19:247-277.

Nosek, J.A., S.R. Craven, J.R. Sullivan, S.S. Hurley, and R.E. Peterson. 1992. Toxicity and reproductive effects of 2,3,7,8-tetrachlorodibenzo-p-dioxin in ring-necked pheasant. *Journal of Toxicology and Environmental Health* 35:187-198.

Poole, A.F. 1983. Courtship feeding, clutch size, and egg size in ospreys: A preliminary report. In Bird, D. M.; Seymour, N. R.; Gerrard, J. M., eds. *Biology and management of bald eagles and ospreys*. St. Anne de Bellvue, Quebec: Harpell Press; pp. 243-256.

Rattner, B.A., N.H. Golden, J.L. Pearson, J.B. Cohen, L.J. Garrett, M.A. Ottinger, and R.M. Erwin. 2001. *Biological and Ecotoxicological Characteristics of Terrestrial Vertebrate Species Residing in Estuaries*. http://www.pwrc.usgs.gov/bioeco/default.htm.

9. ASSESSMENT ENDPOINT – SURVIVAL, GROWTH, AND REPRODUCTION OF PISCIVOROUS MAMMALS

Highlights

Conceptual Model

The assessment endpoint is the survival, growth, and reproduction of piscivorous mammals in the Housatonic River PSA. The piscivorous mammals selected as representative species for the ERA (mink and river otter) are exposed to tPCBs and TEQ via diet and trophic transfer.

Exposure

Exposure of the representative species to COCs (tPCBs and TEQ) was determined from concentrations found in prey items and an estimation of the daily intake of COCs from consumption of prey.

Effects

Data on toxicity of tPCBs to mink were used to derive a dose-response relationship. The existing data on toxicity of TEQ to mink were insufficient to derive a dose-response relationship; therefore, upper and lower toxicity thresholds were derived. No tPCB or TEQ toxicity data were available for otter. River otter were assumed to have a similar sensitivity to tPCBs and TEQ. A site-specific feeding study was conducted to evaluate adverse effects to mink from exposure to Housatonic River COCs in their diet.

Risk

Mink and river otter are at high risk as a result of exposure to tPCBs and TEQ in the Housatonic PSA. The risk remains high even for those individuals who forage only a fraction of their time in the PSA.

9.1 INTRODUCTION

The purpose of this section is to characterize and quantify the current and potential risks posed to piscivorous mammals. A Pre-ERA (Appendix B) was conducted to identify contaminants, other than tPCBs, that pose potential risks to aquatic biota and wildlife in the PSA. A three-tiered deterministic approach was used to screen COPCs. Subsequent to the Pre-ERA, several other COPCs (primarily organochlorine pesticides) were screened out because their actual concentrations in the PSA were likely much lower than the measured values due to laboratory interference problems (see Appendix I.1, Attachment B.2 and Section 2.4). These COPCs were evaluated further for each assessment endpoint, and the contaminants of concern (COCs) that were retained for piscivorous mammals were tPCBs and 2,3,7,8-TCDD toxic equivalence (TEQ).

A step-wise approach was used to assess the risks of tPCBs and TEQ to piscivorous mammals in the Housatonic River watershed. The four main steps in this process include:

1. Derivation of a conceptual model (Figure 9.1-1).
2. Assessment of exposure of piscivorous mammals to COCs (Figure 9.1-2).
3. Assessment of the effects of COCs on piscivorous mammals (Figure 9.1-3).
4. Characterization of risks to the piscivorous mammalian community (Figure 9.1-4).

The detailed ecological risk assessment for piscivorous mammals is provided in Appendix I.

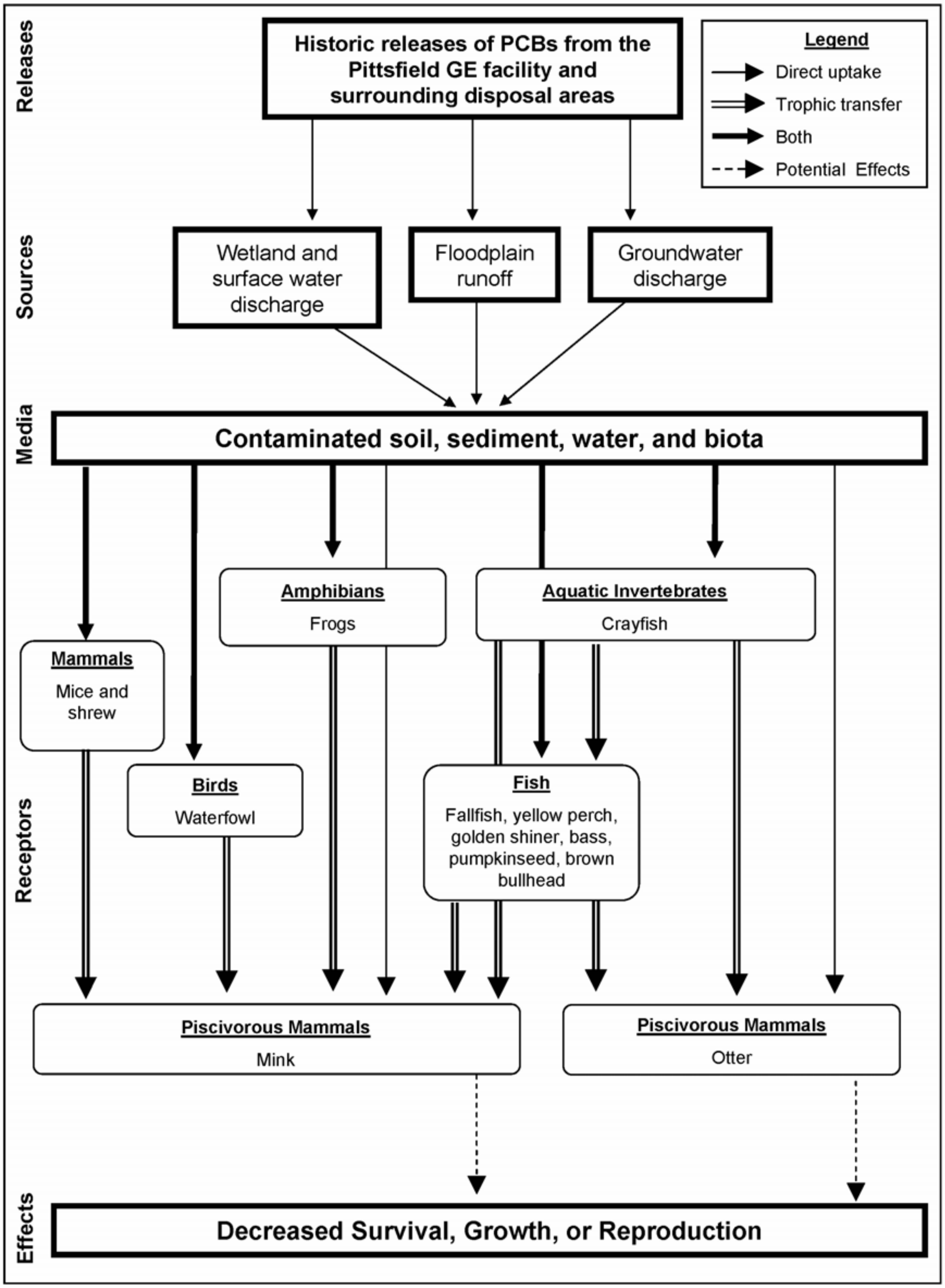


Figure 9.1-1 Conceptual Model Diagram: Exposure Pathways for Piscivorous Mammals Exposed to COCs in the Housatonic River PSA

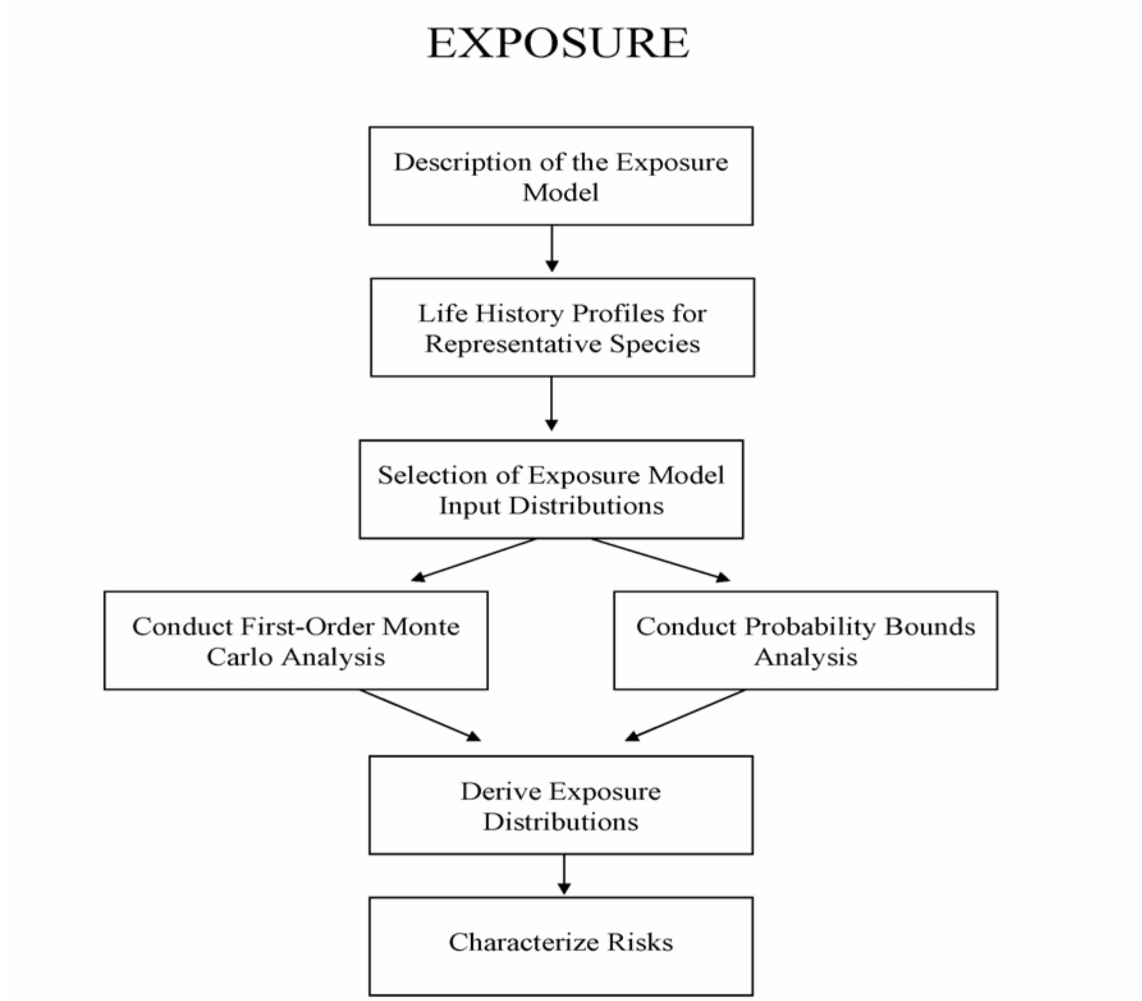


Figure 9.1-2 Overview of Approach Used to Assess Modeled Exposure of Piscivorous Mammals to Contaminants of Concern (COCs) in the Housatonic River PSA

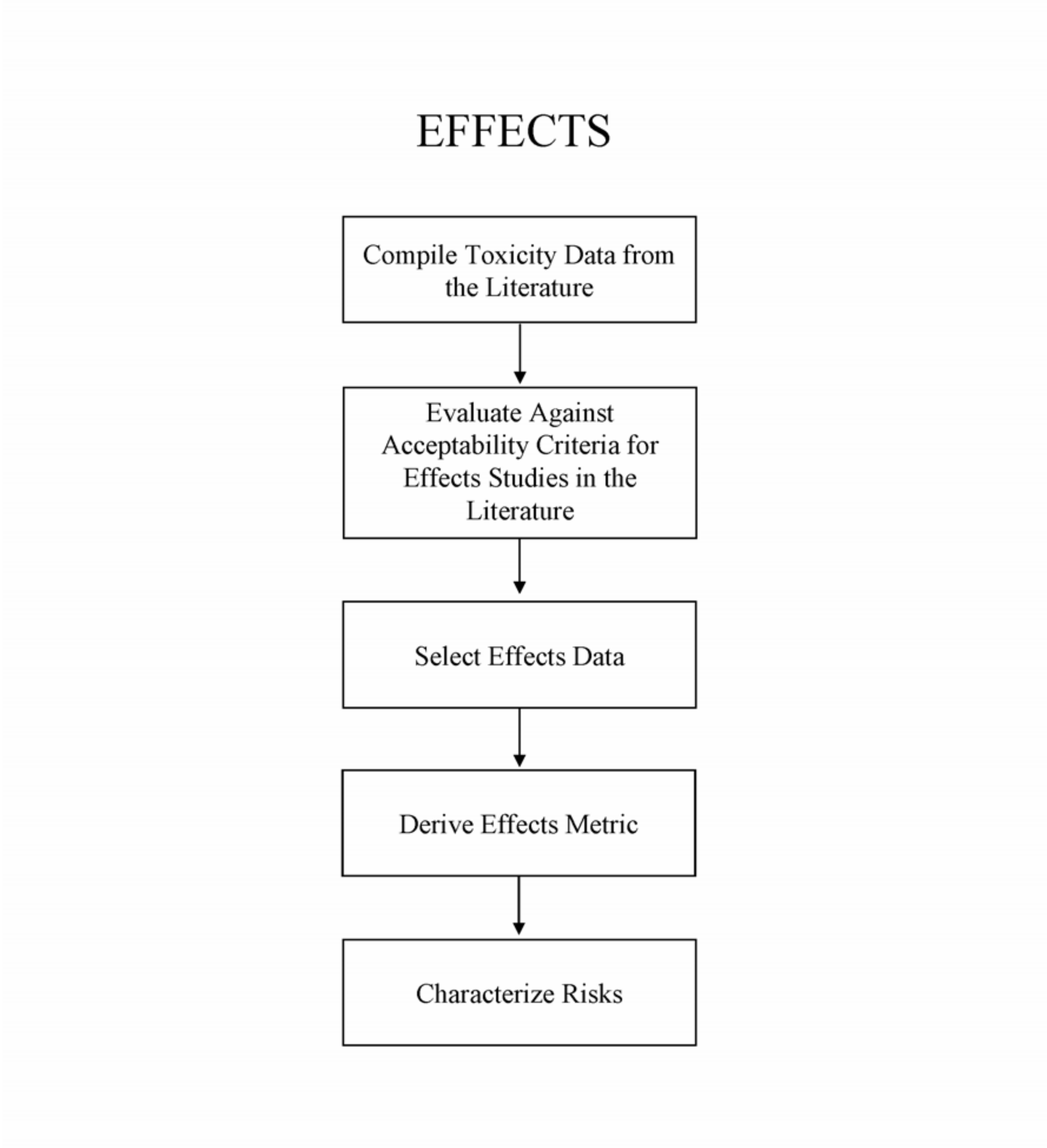


Figure 9.1-3 Overview of Approach Used to Assess the Modeled Effects of Contaminants of Concern (COCs) to Piscivorous Mammals in the Housatonic River PSA

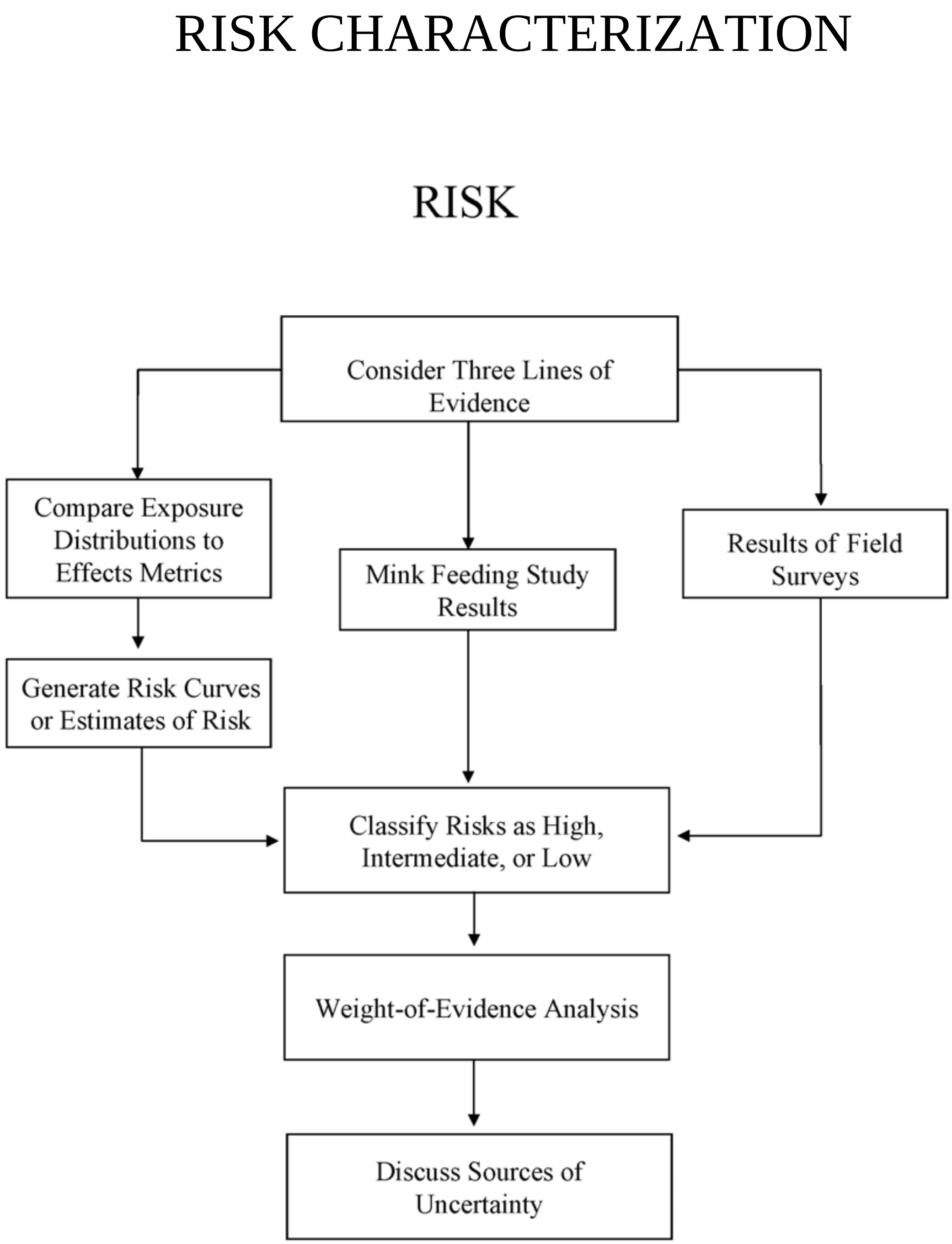


Figure 9.1-4 Overview of Approach Used to Characterize the Risks of Contaminants of Concern (COCs) to Piscivorous Mammals in the Housatonic River PSA

9.2 CONCEPTUAL MODEL

The conceptual model presented in Figure 9.1-1 illustrates the exposure pathways for piscivorous mammals exposed to tPCBs and TEQ in the PSA. Piscivorous mammals that reside, or partially reside, within the study area are exposed to tPCBs and TEQ principally through diet and trophic transfer. Other routes of exposure, considered to be less important to overall exposure, include inhalation, water consumption, and sediment ingestion (Moore et al. 1999).

The problem formulation (see Section 2) identified mink (*Mustela vison*) and river otter (*Lutra canadensis*) as the representative species for piscivorous mammals exposed to tPCBs and TEQ from consumption of contaminated prey. Life history profiles for mink and river otter are summarized in the following text boxes.

The assessment endpoint that is the subject of this section is the survival, growth, and reproduction of piscivorous mammals in the Housatonic River PSA. The measurement endpoints used to evaluate the assessment endpoint included: (1) determining the extent to which the concentrations of tPCBs and TEQ ingested in the diet impact the survival, reproduction, or growth of piscivorous mammals by comparisons to doses reported in the literature to cause adverse effects; (2) determining, by conducting quantitative field surveys, the abundance of piscivorous mammals in the Housatonic River relative to appropriate uncontaminated reference areas within the watershed; and (3) determining, by conducting a feeding study using fish collected from the PSA, whether a diet of site-specific fish has an adverse effect on the survival and reproduction of farm-raised mink.

Life History of Mink

Mink are small, fur-bearing animals with characteristic elongated bodies, short legs, and long tails. Mink are one of the most widespread mammalian carnivores, with a range spanning much of the continental USA and Canada.

Habitat – Require access to open water such as streams, tidal flats, marshes, shallow rivers, lakes, and swamps. Also suitable cover in the form of overhanging vegetation, rock crevices, exposed roots, log jams, and undercut banks.

Home Range - Adult males occupy home ranges exclusive of other adult males, but that may include the home ranges of one or more females. Home range size varies from 309 to 776 ha for males, and from 7.8 to 20.4 ha for females. Riverine home ranges are linear (between 1.0 and 6.0 km of shoreline); those in marsh habitats tend to be more circular.

Dietary Habits - Primary food items include fish, small mammals, benthic invertebrates, birds, and amphibians. Opportunistic; diet varies depending upon the availability of prey items. Mean percentage of prey items in diet: fish, 23%; mammals, 15%; birds, 11.0%; invertebrates, 36%; and amphibians and reptiles, 15.0%.

Life History of River Otter

River otter are long-bodied, short-legged, semi-aquatic mustelids that occur throughout most of Canada and the continental United States. Male otter in the eastern United States are quite large and range in weight from 8 to 11 kg. Females range from 7.5 to 8 kg.

Habitat - Remain close to aquatic habitats such as lakes, marshes, streams, seashores, rivers, creeks, and bayous. In New England, preferentially select riverine and lacustrine systems. Have numerous denning and nesting sites within home range, used over the course of the year. Denning and resting sites may be located in log jams, riparian vegetation, snow or ice cavities, riprap, talus rock, boulders, brush and log piles, undercut banks, and dens constructed by other animals.

Home Range - Average size of the home range for adult otter is about 30 km of shoreline. Lactating females have the smallest home ranges. Other than family groups, are typically solitary. Will form temporary associations that may include related or unrelated individuals. Home ranges overlap extensively, with some otter sharing essentially the same home range.

Dietary Habits - Diet somewhat variable; primarily consists of aquatic animals, particularly fish; other prey includes crayfish, amphibians, turtles, birds, small mammals, and insects. Prefer to forage in shallow water and eat primarily slow-moving, shallow-dwelling fish, such as chubs, suckers, catfish, daces, darters, and schooling fish such as bluegill and other sunfish.

9.3 EXPOSURE ASSESSMENT

This exposure assessment evaluates exposure of piscivorous mammals to tPCBs and 2,3,7,8-TCDD toxic equivalence (TEQ) in Reaches 5 (confluence to Woods Pond) and 6 (Woods Pond), together referred to as the Primary Study Area (PSA) of the Housatonic River and for two reference areas on the East Branch of the Housatonic River in Dalton, MA (herein referred to as the "upstream reference area") and at Threemile Pond located in Sheffield, MA.

The exposure analysis for mink was carried out separately for Reach 5 and Reach 6 of the PSA because the foraging range of mink approximates the lengths of those river sections. However, the foraging range of river otter is larger; therefore, the exposure analysis for river otter was conducted with Reaches 5 and 6 combined.

9.3.1 Exposure Model

Exposure of mink and river otter to tPCBs and TEQ was estimated using a total daily intake model adapted from the *Wildlife Exposure Factors Handbook* (EPA 1993) and related publications. The model used in the exposure analysis was:

$$TDI = FT \cdot FIR \sum_{i=1}^{n} C_i \cdot P_i \qquad \text{(Eq. 1)}$$

where

TDI	=	Total daily intake (mg/kg bw/d tPCBs, ng/kg bw/d TEQ).
FIR	=	Normalized food intake rate (kg/kg bw/d).
FT	=	Foraging time in PSA (unitless).
C_i	=	Concentration in *i*th food item (mg/kg tPCBs, ng/kg TEQ).
P_i	=	Proportion of the *i*th food item in the diet (unitless).

The models consider the food intake rates of the representative species (*FIR*), the concentrations of COCs in each food item (C_i), and the proportion of the diet accounted for by that food item (P_i). For those input variables that are uncertain, variable, or both, distributions are used rather than point estimates. Monte Carlo and probability bounds analyses are the methods used to propagate uncertainties about input variables in the exposure model for each COC. A description of these techniques and the methods used to parameterize input variables is presented in Section 6.5 and Appendix C. The distributions used in the exposure analyses for mink and river otter and the results of these analyses are discussed in detail in Appendix I. A brief description of these variables is provided below.

9.3.1.1 Body Weight (BW)

Average body weights (wet weight of wild animals) of female mink range from 550 g (Mitchell 1961) to 970 g (Hornshaw et al. 1983) and males range from 630 to 1,000 g (Whitaker and Hamilton 1998). For both the Monte Carlo and probability bounds analyses, the mean weight of females was estimated to be 685 g.

Whitaker and Hamilton (1998) reported that body weights of river otter ranged from 8 to 11 kg (average of 9.2 kg) for males and from 7.5 to 8.0 kg (average of 7.9 kg) for females in eastern United States populations. In the Monte Carlo and probability bounds analyses, body weight was assumed to be normally distributed with a mean of 8.63 kg.

9.3.1.2 Food Intake Rate (FIR)

The daily energy requirements of mink vary depending on environmental conditions and the stage of the reproductive cycle. For the purpose of the ERA for piscivorous mammals, *FIR* was estimated using an allometric equation rather than using literature-reported values for captive mink.

Food intake rate (*FIR*) is derived using the following equation:

$$FMR\ (kJ/d) = a \cdot BW(g)^{b} \qquad \text{(Eq. 2)}$$

where *FMR* is the free-living metabolic rate and *BW* is the body weight. Food intake rate is derived from *FMR* using the following equation:

$$FIR = \frac{FMR}{\sum_{i=1}^{n} AE_i \cdot GE_i} \qquad \text{(Eq. 3)}$$

where AE_i is the assimilation efficiency of *i*th food item (unitless) and *GE* is the gross energy of *i*th food item (kcal/kg).

The gross energies of various wildlife food sources are summarized in the *Wildlife Exposure Factors Handbook* (EPA 1993). Average assimilation efficiency for mammals consuming fish and amphibians is 0.91, for invertebrates it is 0.87, and for birds and mammals it is 0.84 (EPA 1993; Grodzinski and Wunder 1975; Barrett and Stueck 1976). No data were available for assimilation efficiency of mammals consuming amphibians, but it is likely to be similar to that for mammals consuming fish.

9.3.1.3 Proportions of Dietary Items (P_i)

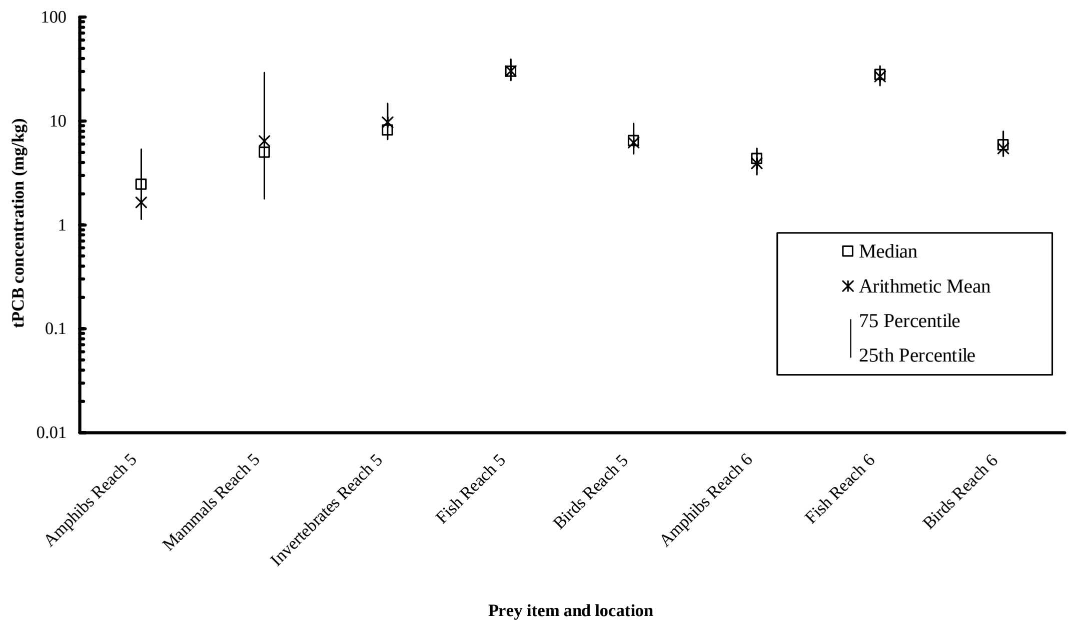
The primary food items in the mink diet include small mammals, fish, benthic invertebrates (crayfish), birds (waterfowl), and amphibians (Alexander 1977; Burgess and Bider 1980; Cowan and Reilly 1973; Gilbert and Nancekivell 1982; Hamilton 1959, 1940; Melquist et al. 1981; Proulx et al. 1987) (Table I.2-2). Combining the available data, an average of 23% (range of 0 to 64.7%) of the mink diet consists of fish. Mammals on average comprise 15% of the diet (range of 0 to 25%). Reptiles and amphibians also constitute an average of 15% (range of 0 to 30%) of the diet, and birds (i.e., waterfowl) 11% (range of 0 to 39%) of the diet. Invertebrates constitute an average of 36% of the diet (range of 0 to 54%). Based on published information, fish used in the exposure analyses were limited to a minimum length of 7 cm and a maximum length of 20 cm.

River otter have a less variable diet than mink. The river otter diet is dominated by aquatic prey, particularly fish (Melquist and Hornocker 1983; Greer 1955; Anderson and Woolf 1987; Sheldon and Toll 1964). In this ERA, the mean proportion of crayfish in the diet was assumed to be 20% (range of 8 to 24%). Fish were assumed to compose the remainder of the diet. Based on published information, the exposure analysis for otter used tissue samples for fish ranging in length from 2 to 50 cm.

9.3.1.4 Concentrations of COCs in Prey (C_i)

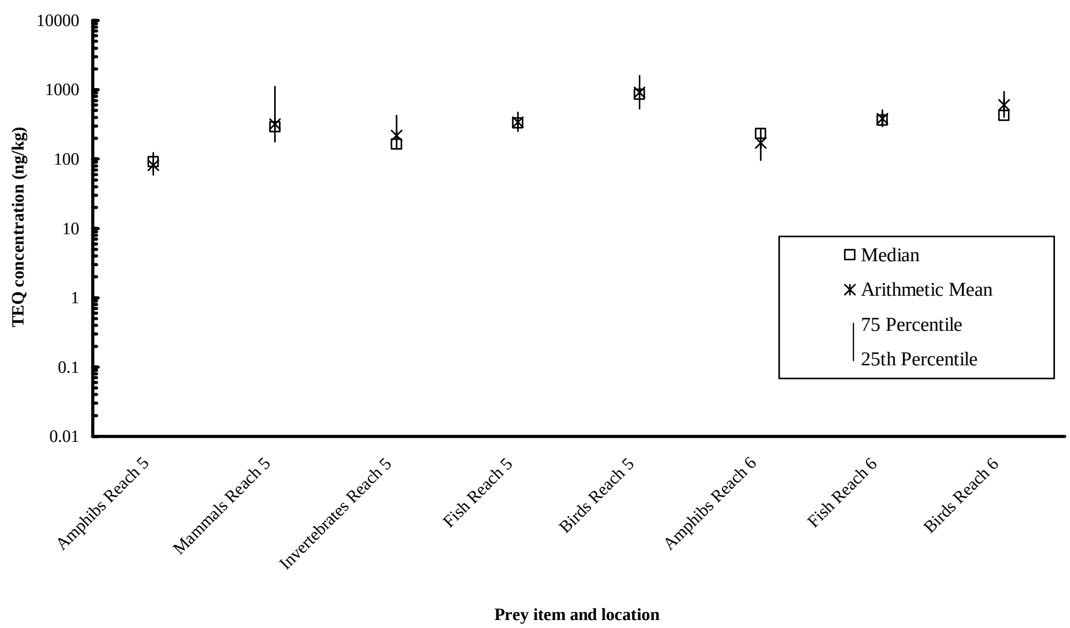
The median concentrations of tPCBs in mink prey from the PSA range from 2.45 mg/kg ww in amphibians from Reach 5 to 29.9 mg/kg ww in fish from the same location. Median TEQ levels in mink prey range from 91.6 ng/kg ww in amphibians from Reach 5 to 858 ng/kg ww in birds from the same location. Concentrations of tPCBs and TEQ in prey of mink are presented in

Figures 9.3-1 and 9.3-2, respectively. Concentrations of tPCBs and TEQ in prey of river otter are presented in Figures 9.3-3 and 9.3-4, respectively. The input variables for concentrations of COCs in prey of mink and river otter are shown in Tables I.2-4, I.2-6, I.2-13, and I.2-15.



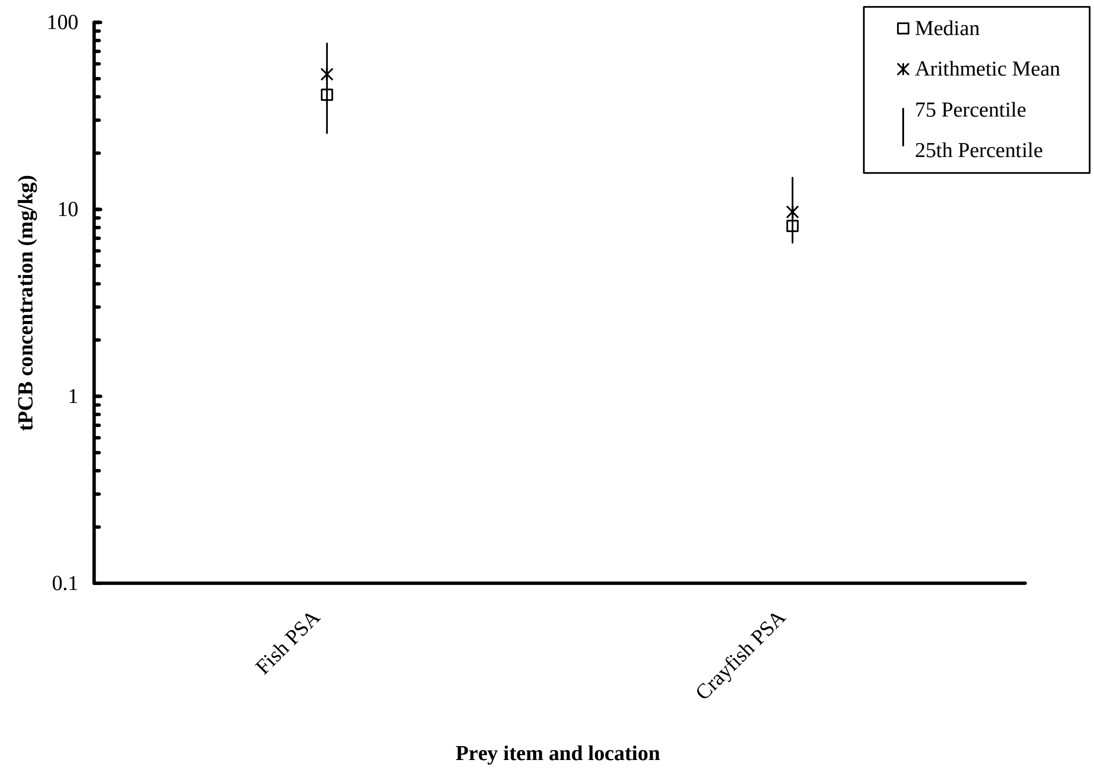
Note: Prey concentrations are wet weight.

Figure 9.3-1 Concentrations of tPCBs in Prey of Mink



Note: Prey concentrations are wet weight.

Figure 9.3-2 Concentrations of TEQ in Prey of Mink



Note: Prey concentrations are wet weight.

Figure 9.3-3 Concentrations of tPCBs in Prey of River Otter

Note: Prey concentrations are wet weight.

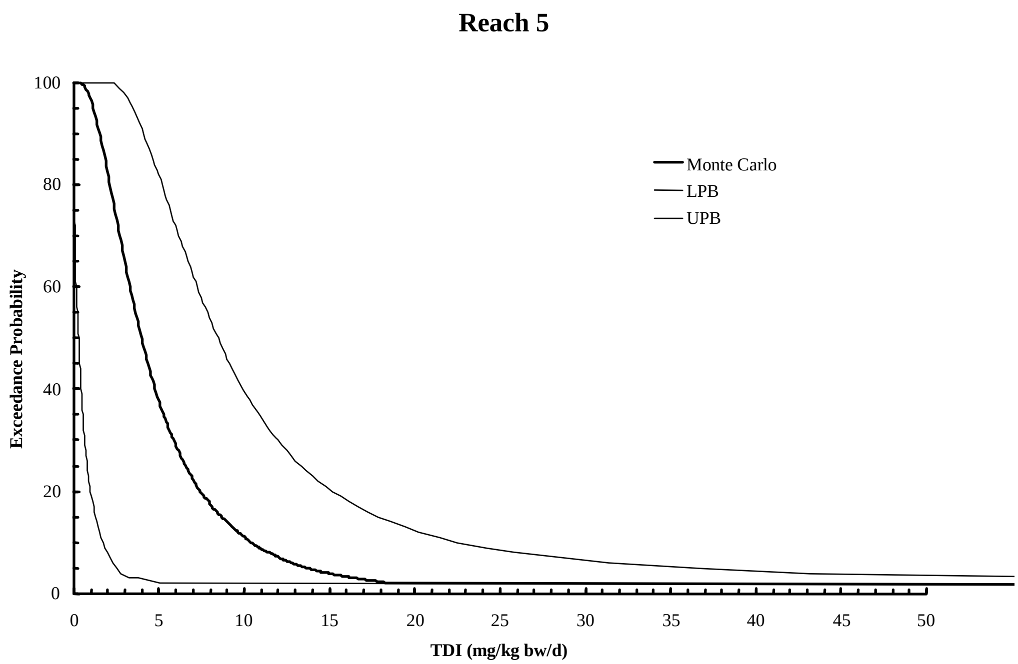
Figure 9.3-4 Concentrations of TEQ in Prey of River Otter

9.3.2 Results of Exposure Assessments

Exposure distributions for exposure of mink and river otter to tPCBs and TEQ in Reaches 5 and 6, and in reference areas, are presented in Figures 9.3-5 through 9.3-18.

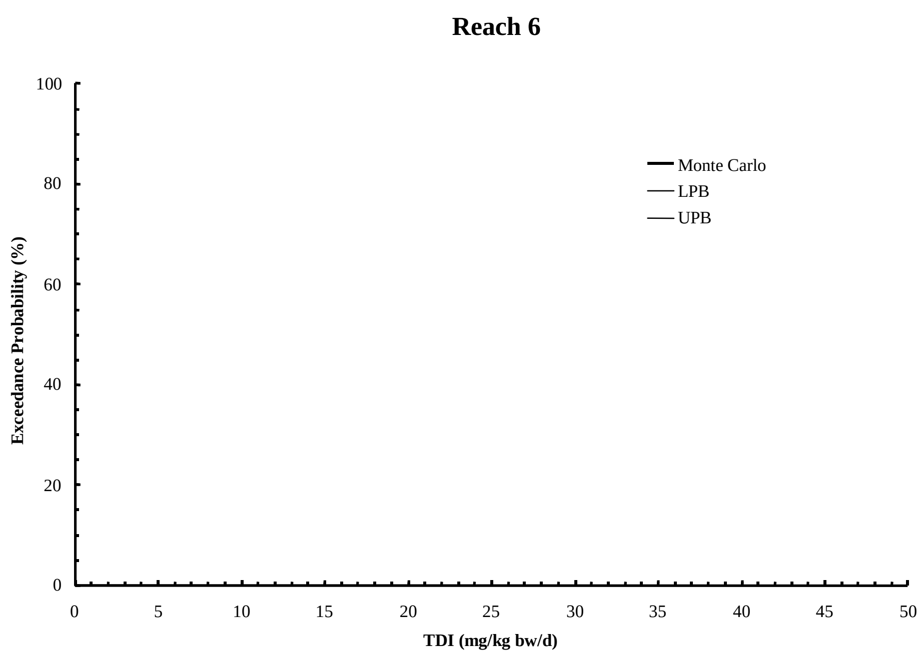
Figure 9.3-5 depicts the cumulative distribution and probability bounds of tPCB intake rates for mink in Reach 5. The Monte Carlo analysis indicated that exposure of mink to tPCBs could range from a minimum of 0.308 to a maximum of 82.5 mg/kg bw/d. The mean exposure was 5.29 mg/kg bw/d and the median exposure was 3.97 mg/kg bw/d. The 50th percentile of the probability envelope formed by the lower and upper bounds ranged between 0.292 and 8.47 mg/kg bw/d.

Figure 9.3-13 depicts the cumulative distribution and probability bounds of tPCB intake rates for river otter in Reaches 5 and 6. The Monte Carlo analysis indicated that exposure of otter to tPCBs foraging in the PSA 100% of the time could range from a minimum of 0.251 to a maximum of 111 mg/kg bw/d. The mean exposure was 8.42 mg/kg bw/d and the median exposure was 6.02 mg/kg bw/d (Table I.2-14). The 50th percentile ranged between 3.27 and 14.2 mg/kg bw/d.



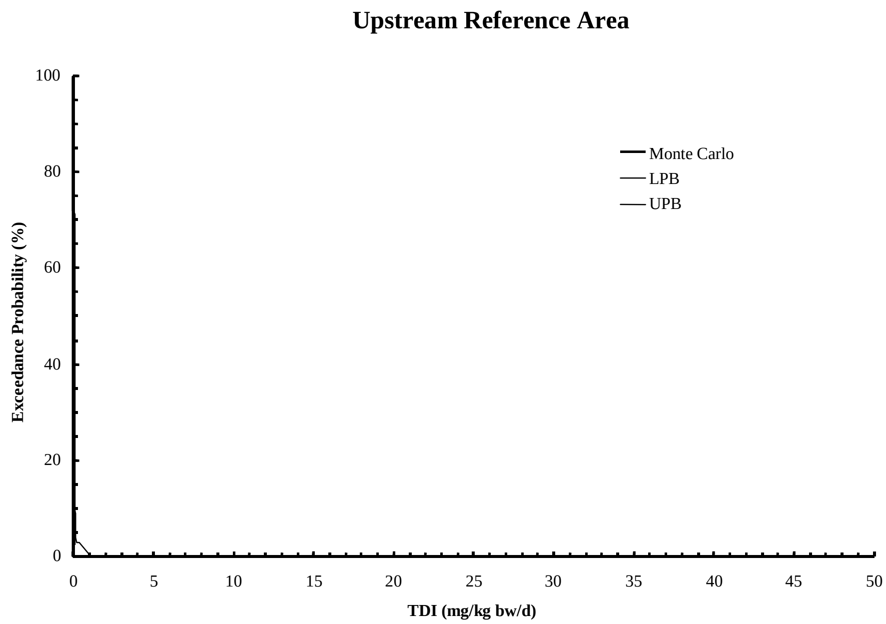
LPB = Lower probability bound
UPB = Upper probability bound

Figure 9.3-5 Exposure of Mink to tPCBs in Reach 5 of the Housatonic River



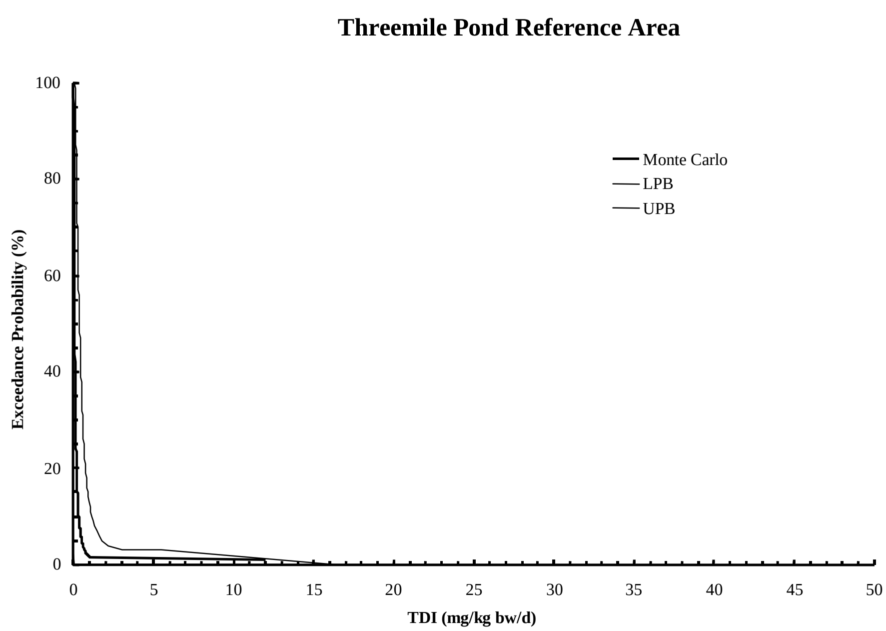
LPB = Lower probability bound
UPB = Upper probability bound

Figure 9.3-6 Exposure of Mink to tPCBs in Reach 6 of the Housatonic River



LPB = Lower probability bound
UPB = Upper probability bound

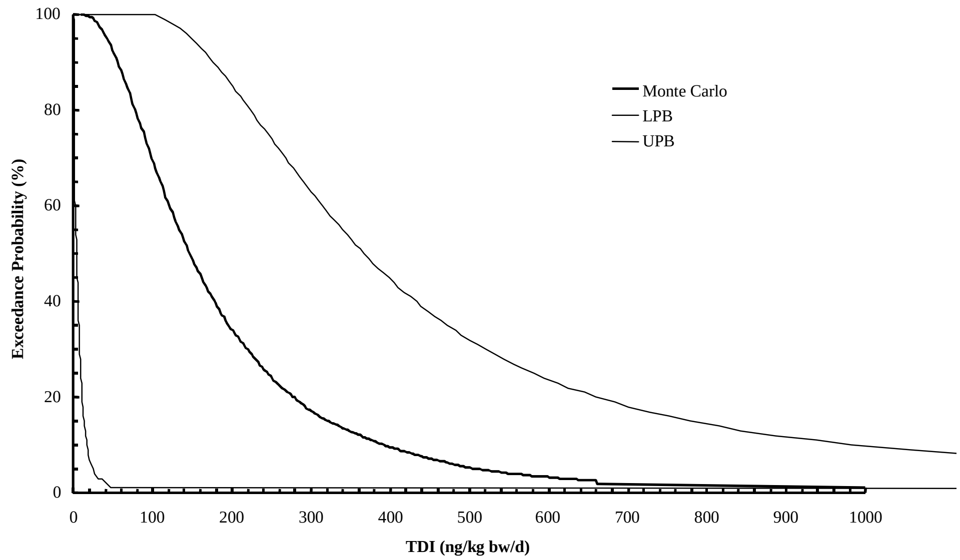
Figure 9.3-7 Exposure of Mink to tPCBs in the Housatonic River Upstream Reference Area



LPB = Lower probability bound
UPB = Upper probability bound

Figure 9.3-8 Exposure of Mink to tPCBs in the Threemile Pond Reference Area

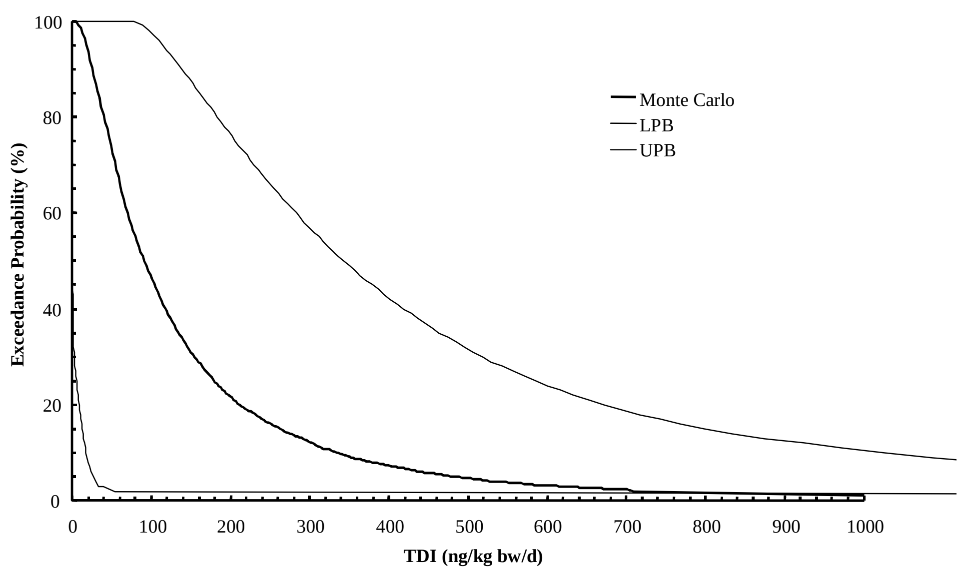
Reach 5



LPB = Lower probability bound
UPB = Upper probability bound

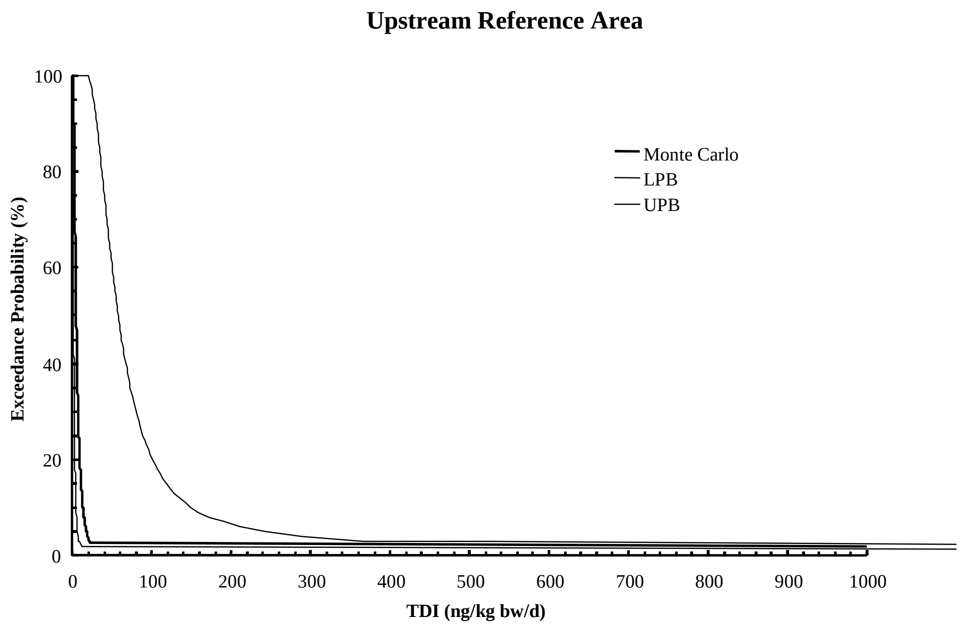
Figure 9.3-9 Exposure of Mink to 2,3,7,8-TCDD TEQ in Reach 5 of the Housatonic River

Reach 6



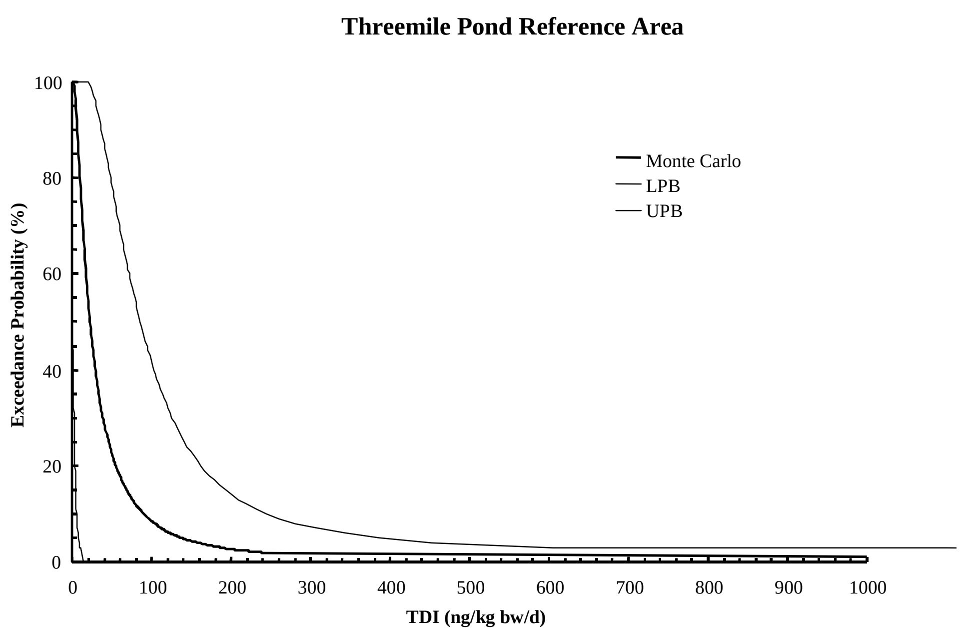
LPB = Lower probability bound
UPB = Upper probability bound

Figure 9.3-10 Exposure of Mink to 2,3,7,8-TCDD TEQ in Reach 6 of the Housatonic River



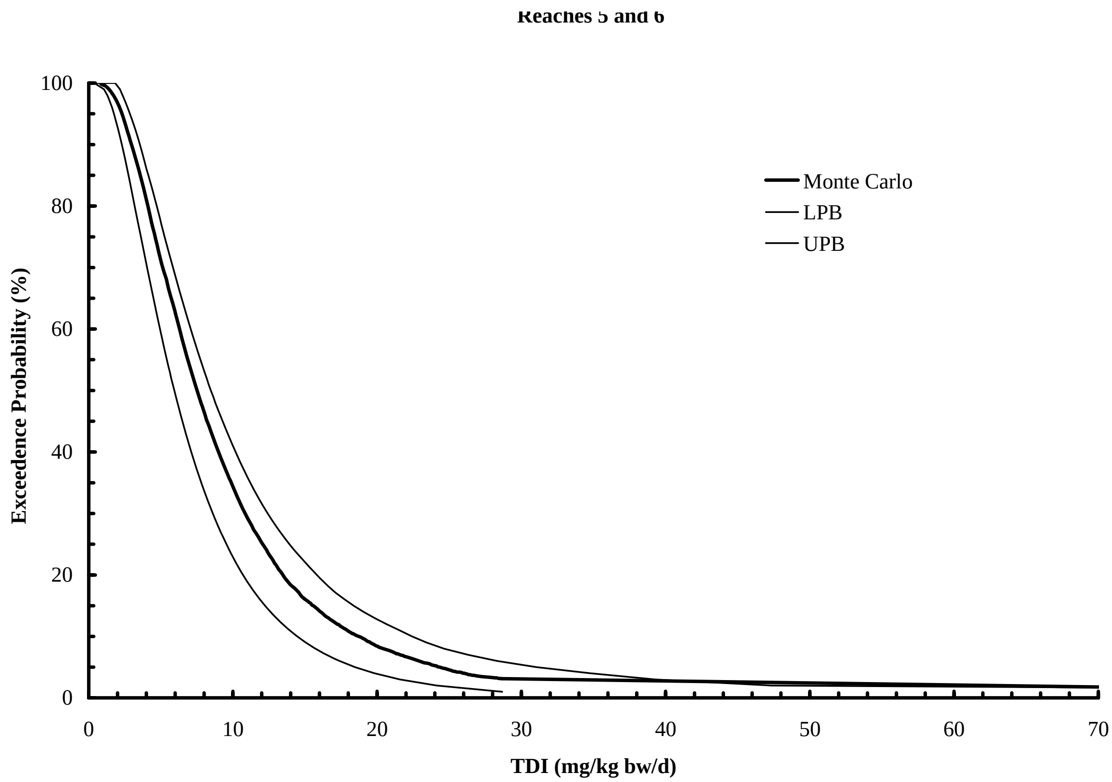
LPB = Lower probability bound
UPB = Upper probability bound

Figure 9.3-11 Exposure of Mink to 2,3,7,8-TCDD TEQ in the Housatonic River Upstream Reference Area



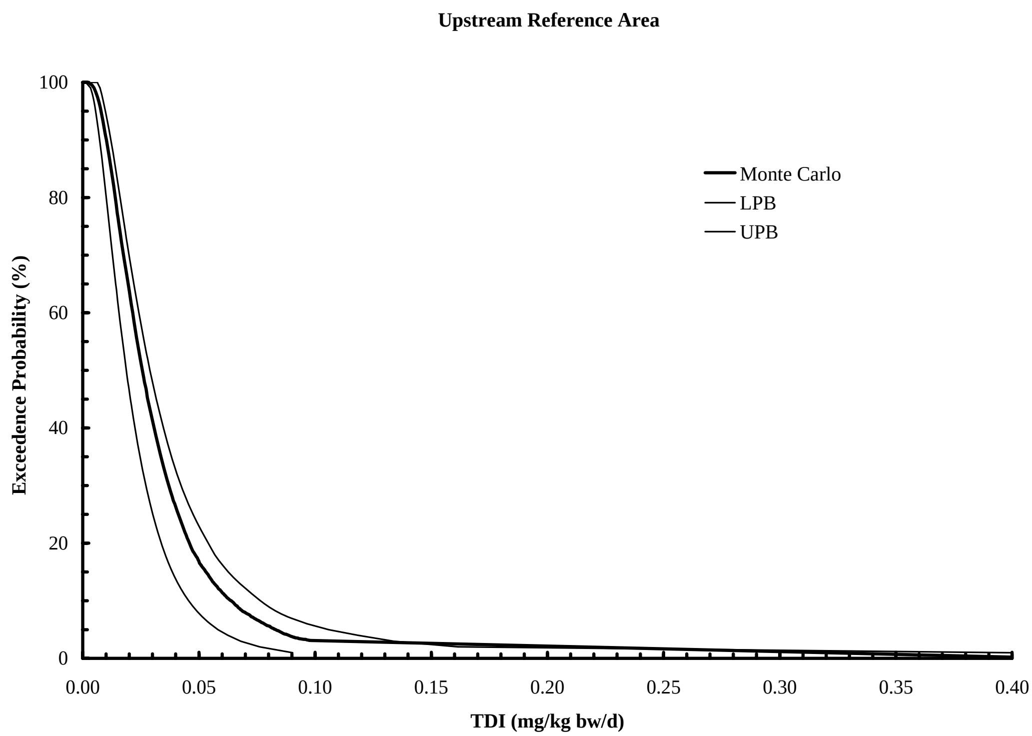
LPB = Lower probability bound
UPB = Upper probability bound

Figure 9.3-12 Exposure of Mink to 2,3,7,8-TCDD TEQ in the Threemile Pond Reference Area



LPB = Lower probability bound
UPB = Upper probability bound

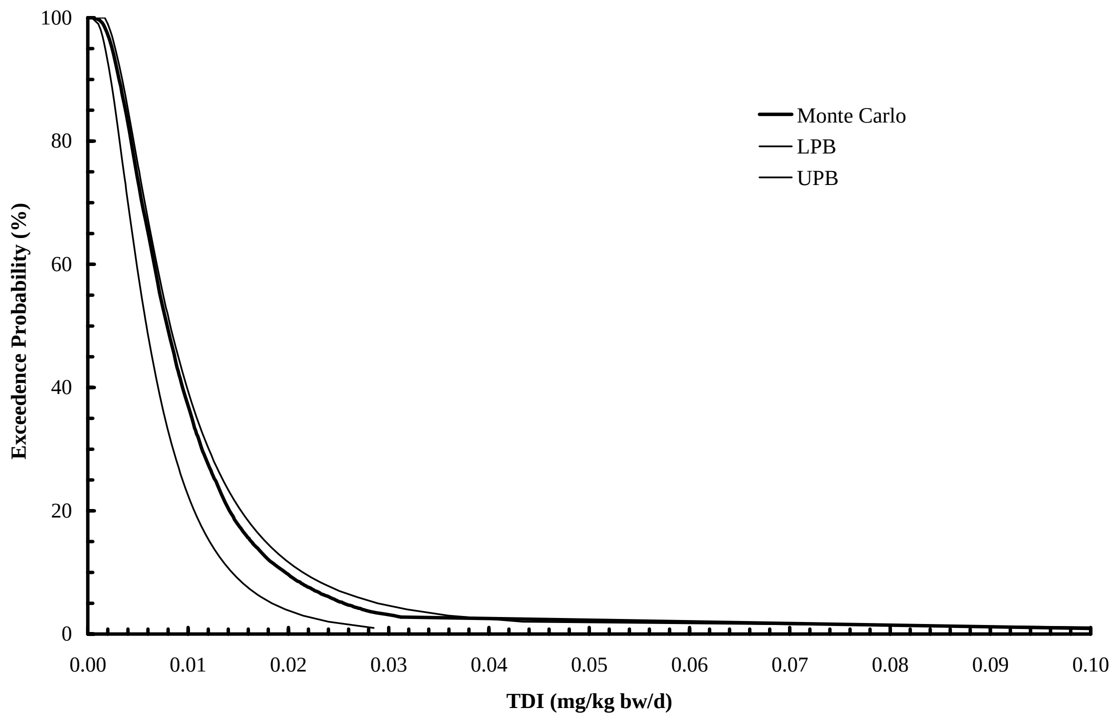
Figure 9.3-13 Exposure of River Otter to tPCBs in Reaches 5 and 6 of the Housatonic River



LPB = Lower probability bound
UPB = Upper probability bound

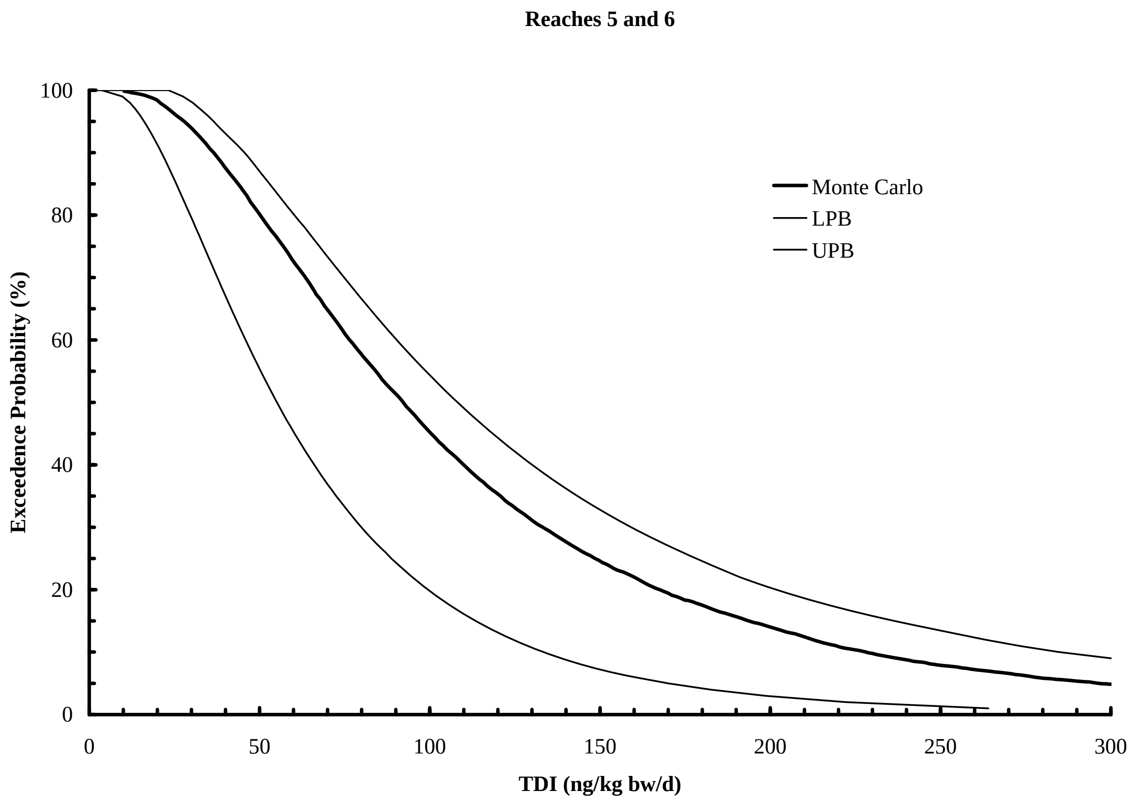
Figure 9.3-14 Exposure of River Otter to tPCBs in the Housatonic River Upstream Reference Area

Threemile Pond Reference Area



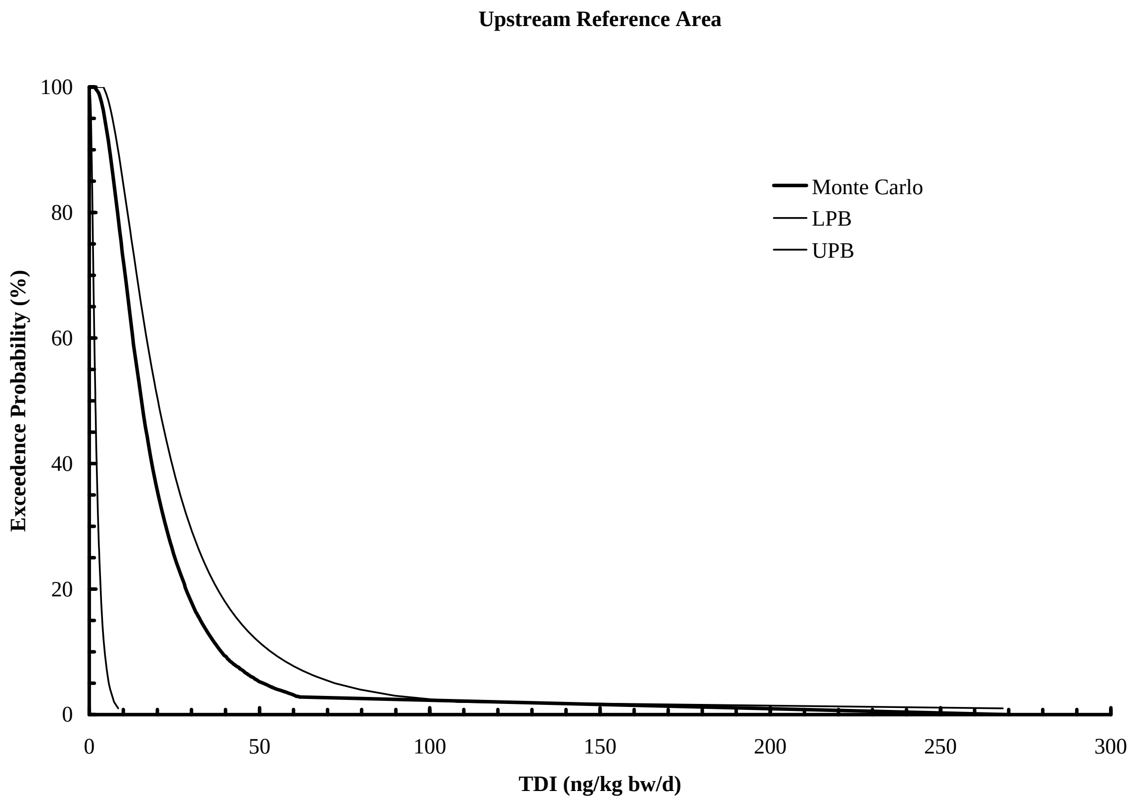
LPB = Lower probability bound
UPB = Upper probability bound

Figure 9.3-15 Exposure of River Otter to tPCBs in the Threemile Pond Reference Area



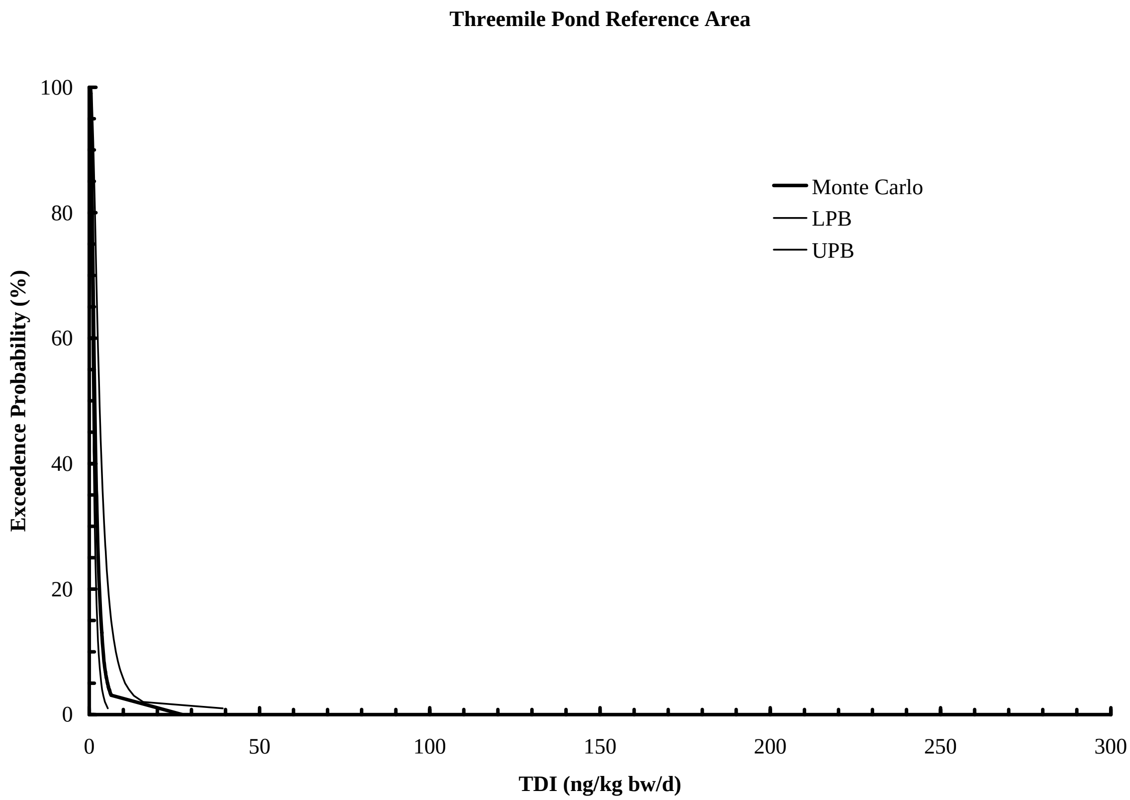
LPB = Lower probability bound
UPB = Upper probability bound

Figure 9.3-16 Exposure of River Otter to 2,3,7,8-TCDD TEQ in Reaches 5 and 6 of the Housatonic River



LPB = Lower probability bound
UPB = Upper probability bound

Figure 9.3-17 Exposure of River Otter to 2,3,7,8-TCDD TEQ in the Upstream Reference Area



LPB = Lower probability bound
UPB = Upper probability bound

Figure 9.3-18 Exposure of River Otter to 2,3,7,8-TCDD TEQ in the Threemile Pond Reference Area

9.4 EFFECTS ASSESSMENT

The effects assessment has two objectives. The first is to review the scientific literature for effects of tPCBs (mainly Aroclor 1254 and 1260 mixtures) and TEQ to piscivorous mammals, with documented effects to the representative species of primary interest in this assessment, mink and river otter. The other objective is to derive the effects metrics that will be used, in conjunction with the exposure assessment results, to estimate risks to the representative piscivorous mammal species.

9.4.1 Review of Toxicity from the Literature

Appendix I includes a detailed review of the scientific literature on the effects of dietary tPCBs and TEQ to piscivorous mammals. The discussion focuses on ecologically relevant effects endpoints such as survival, growth, and reproduction. A summary of reproduction effects for tPCBs and TEQ is presented in Figures I.3-1 and I.3-2 and Table I.3-1.

9.4.2 Mink Feeding Study

To evaluate whether contaminants in the prey of piscivorous mammals foraging in the PSA may cause adverse effects on the survival, reproduction, and/or growth of exposed individuals, a long-term feeding study was performed by researchers at Michigan State University (MSU) (Bursian et al. 2003), the results of which are summarized below.

9.4.2.1 Methodology

Fish were collected from the PSA and mixed with ocean herring in varying proportions to derive a control (uncontaminated) diet and five treatment diets containing target concentrations ranging from 0.25 to 4 mg/kg tPCBs. The diets were fed to captive adult female mink for approximately 160 days, beginning approximately 2 months prior to mating and continuing through whelping of the kits. Some kits were exposed for an additional 6 months following whelping. A variety of endpoints were measured during the study including feed consumption rate, mating success, gestation length, number of kits born, adult and kit survival, body weights, organ weights, and tissue histology. Histopathology of the jaws of mink kits and biochemical parameters were also measured. The latter endpoints are discussed separately in Sections 9.4.3 and 9.4.4.

9.4.2.2 Results and Discussion

Consumption of diets containing COCs derived from Housatonic River fish had no significant effect on breeding or whelping success. Average litter size and kit survival at birth and 3 weeks of age were also not affected by the exposure treatments. However, decreased survival of kits in the highest tPCB treatment group (3.7 mg/kg tPCBs, 68.5 ng/kg TEQ) at 6 weeks of age (i.e., 46% lower compared to controls) was statistically significant ($p < 0.05$) compared to kits in the control and 1.6 mg/kg tPCB treatment groups (Table I.3-2). Regression analysis indicated significant relationships ($p < 0.05$) between kit survival from 0 to 6 weeks and tPCB concentration or TEQ concentration in the diet. For tPCBs, the dietary LC_{10} and LC_{20} were 0.231 and 0.984 mg/kg, respectively. The corresponding dietary LD_{10} and LD_{20} were 0.0247 and 0.105 mg/kg bw/d, respectively. For TEQ, the dietary LC_{10} and LC_{20} were 4.22 and 16.2 ng/kg, respectively. The corresponding dietary LD_{10} and LD_{20} were 0.452 and 1.73 ng/kg bw/d, respectively.

Absolute and relative (expressed as a percentage of body weight) brain, heart, spleen, liver, kidney, and adrenal gland weights of adult females and kits were not significantly different between treatment groups at necropsy. The results of the histological examination of the tissues of major internal organs of the adult female mink and their kits did not show remarkable changes attributable to the treatment diets.

9.4.3 Mink Jaw Lesion Study

The purpose of this study was to determine whether the dietary treatments induced lesions that have been previously observed in other studies of mink fed PCB-126 and TCDD (Bursian and Yamini 2003). The evaluation was conducted on 6-month-old kits necropsied at the end of the mink feeding study.

9.4.3.1 Methodology

The skulls of 6-month-old mink kits (36 kits collected) were processed using a routine histotechnology method. Jaws from 36 kits were examined for pathologies. The observed lesions were graded as mild, moderate, or severe based on the number and size of foci of squamous cell proliferation in maxilla and mandibles.

9.4.3.2 Results and Discussion

The results of the jaw lesion study indicate that dietary concentrations of PCB-126 as low as 54 ng/kg in the diet (0.96 mg tPCBs/kg diet) can induce maxillary and mandibular squamous cell proliferation. Exposure of mink to higher concentrations of PCB-126 for longer periods of time, as would be expected in the Housatonic River ecosystem, would undoubtedly cause increased severity of the lesions leading to erosion of the mandible and maxilla with concomitant loss of teeth (Bursian and Yamini 2003).

9.4.4 Mink Enzyme Study

Tillitt et al. (2003) performed a study to measure hepatic O-dealkylase activities associated with cytochrome P450 (CYP) isozymes induced in mink fed diets containing fish collected from the PSA as part of the MSU feeding study by Bursian et al. (2003). Specific activities were measured against four separate substrates to measure the induction of CYP enzymes in maternal and F1 generations of the exposed mink. The induction of CYP enzymes is a good indicator of exposure to coplanar PCBs, dioxins, and furans, and indicates a first level of toxicological response (Aulerich et al. 2003). Hepatic activities were measured because the majority of detoxification of xenobiotics occurs in the liver.

9.4.4.1 Methodology

In the feeding study by Bursian et al. (2003), 36 offspring (kits), along with the adults, were used at 6 weeks after whelping. Another 36 kits were used at 6 months post whelping. The livers were removed and placed in 1.2-ml cryovials and frozen in liquid nitrogen. Frozen liver samples from the parental generation, 6-week-old offspring, and 6-month-old offspring were transmitted to the Columbia Environmental Research Center (CERC) for analysis. The analyses consisted of microsomal preparation and various O-dealkylase assays. All procedures were executed according to CERC Standard Operating Procedures (SOPs) and QA/QC procedures.

9.4.4.2 Results and Discussion

Induction of CYP2B-related activity in mink (benzyloxyresorufin-O-deethylase or BROD and pentoxyresorufin-O-deethylase or PROD) was not substantial at any of the doses of fish from the Housatonic River. Further, none of the increases in BROD or PROD activities occurred in a dose-dependent fashion. Thus, the amounts of di- to tetra-ortho-chloro-substituted PCBs (PCB congeners thought to be responsible for CYP2B-related enzyme inductions) were either below a threshold of activation of these enzymes in the dietary treatments or the enzyme induction pathways were saturated.

Induction of CYP1A1-related hepatic enzyme activities (ethoxycoumarin-O-deethylase or ECOD and ethoxyresorufin-O-deethylase or EROD) was observed to occur in a dose-dependent manner in all ages of mink examined. Significant increases in these Ah-receptor-regulated enzymes were observed even in treatments with only a small amount of fish from the Housatonic River (0.44%) in their diets. These results confirm the known sensitivity of mink to the effects of tPCBs and other related dioxin-like compounds. The results also confirm that only a small amount of fish (< 0.5%) from the Housatonic River would be required in the diets of mink to activate Ah receptor pathways and processes in mink (Tillitt et al. 2003).

9.4.5 Effects Metrics for Characterizing Risk

Effects data can be summarized in a variety of ways ranging from benchmarks designed to be protective of most or all species to dose-response curves. A summary of the decision criteria used to derive effects metrics is provided in the text box. Further details on the decision criteria used in selecting effects metrics are provided in Section 6.6.

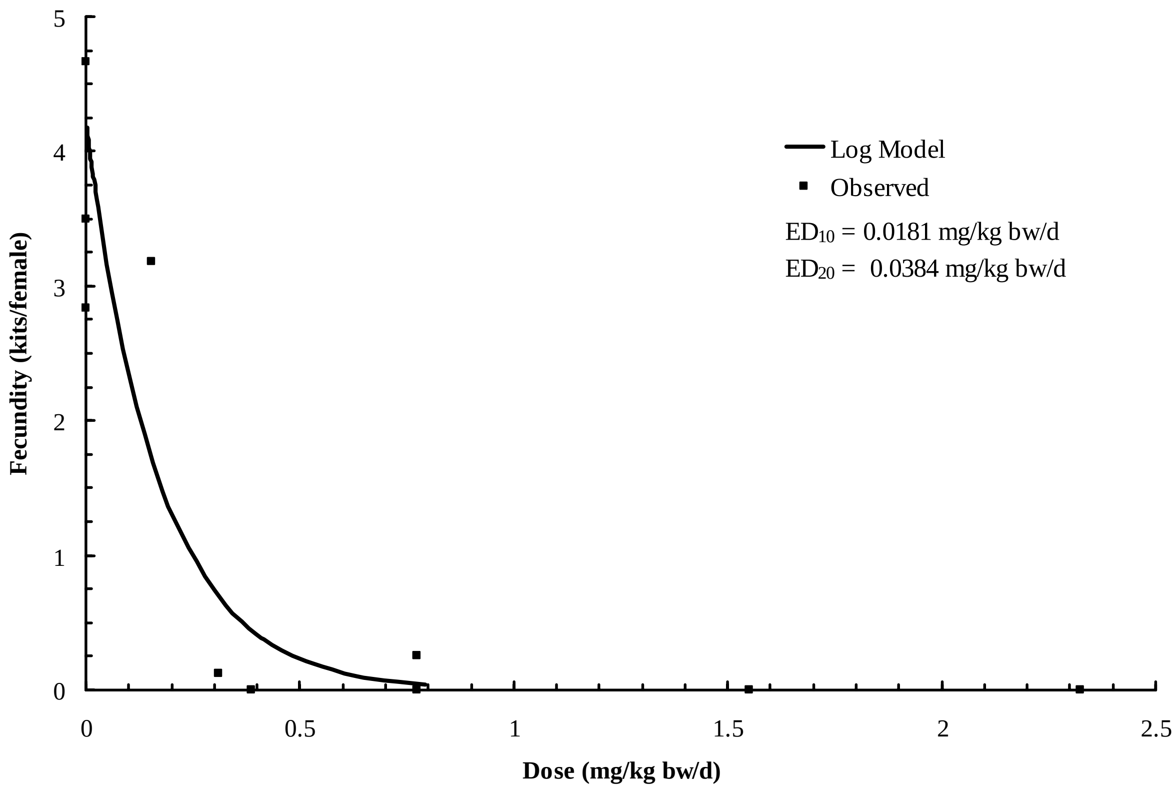
Hierarchical Decision Criteria for Derivation of Effects Metrics

1. Have single-study bioassays with five or more treatments been conducted on the receptor of interest or a reasonable surrogate? If yes, estimate the concentration- or dose-response relationships. If not, go to 2.
2. Are multiple bioassays with similar protocols, exposure scenarios and effects metrics available that, when combined, have five or more treatments for the receptor of interest or a reasonable surrogate? If yes, estimate the dose-response relationship as in 1. If not, go to 3.
3. Have bioassays with less than five treatments been conducted on the receptor of interest or a reasonable surrogate? If yes, conduct or report results of hypothesis testing to determine the NOAEL and LOAEL. If not, go to 4.
4. Are sufficient data available from field studies and monitoring programs to estimate concentrations or doses of the COC that are consistently protective or associated with adverse effects? If yes, develop field-based effects metrics. If not, go to 5.
5. Derive a range where the threshold for the receptor of interest is expected to occur. Because information on the sensitivity of the receptor of interest is lacking, it is difficult to derive a threshold that is biased neither high nor low. If bioassay data are available for several other species, however, calculate a threshold for each to determine a threshold range that spans sensitive and tolerant species. That range is likely to include the threshold for the receptor of interest.

In this ERA, data were available to derive dose-response curves for mink exposed to tPCBs. There were insufficient data to derive dose-response relationships for TEQ. A field-based threshold range was derived instead. There were no toxicity data for river otter. Mink toxicity data were used as surrogate estimates of toxicity of tPCBs and TEQ to river otter.

9.4.5.1 Effects of tPCBs to Mink and River Otter

Acceptable studies for deriving a dose-response curve were Aulerich and Ringer (1977) and Aulerich et al. (1985). Figure 9.4-1 presents the dose-response curve for reduced fecundity of mink exposed to tPCBs. The dose-response curve indicates that 10% and 20% declines in fecundity would be expected at doses of 0.0181 and 0.0384 mg/kg bw/d, respectively.



Note: Symbols indicate raw data.

Figure 9.4-1 Dose Response Curve for Effects of tPCBs on Fecundity of Mink

9.4.5.2 Effects of TEQ to Mink and River Otter

The studies by Heaton et al. (1995), Hochstein et al. (1998, 2001), and Aulerich et al. (1988) involved exposing mink to fish contaminated with TEQ and other contaminants. Based on a review of these studies, adverse effects on growth in kits begin at concentrations of 3.6 ng/kg bw/d (lower toxicity threshold; Heaton et al. 1995). The highest dose that did not cause adverse effects was 36 ng/kg bw/d (upper toxicity threshold; Hochstein et al. 2001). Thus, the threshold range, based on studies that used field-collected fish, is 3.6 to 36 ng/kg bw/d for piscivorous mammals.

9.5 RISK CHARACTERIZATION

This section characterizes the risk to piscivorous mammals exposed to tPCBs and TEQ in the PSA of the Housatonic River. The risk characterization uses three lines of evidence to determine ecological risks to mink and two lines of evidence to determine ecological risks to river otter (see text box). The major lines of evidence are considered to be independent and are combined in a weight-of-evidence (WOE) assessment. The key risk questions and the lines of evidence are summarized in the text box.

Risk Questions

- Are the concentrations of tPCBs and TEQ present in the prey of piscivorous mammals sufficient to cause adverse effects to individuals inhabiting the PSA of the Housatonic River?
- If so, how severe are the risks and what are their potential consequences?

Lines of Evidence

- Use of semi-qualitative field surveys.
- Probabilistic exposure and effects modeling.
- Feeding study using fish from the PSA (mink only).

9.5.1 Field Surveys

9.5.1.1 EPA Study

The mammalian community in the PSA was studied by EPA over a 4-year period, from 1998 to 2001. A brief summary of these surveys is presented here; detailed information is presented in Appendix A. Surveys were conducted to record presence, relative abundance, and habitat usage for small and large mammals including mink and river otter. A variety of field survey techniques including small mammal trapping, snow tracking, and scent-post station surveys were used to characterize the mammalian community.

The Housatonic River PSA offers an abundance of habitat that meets the requirements for mink and river otter. Based on the experience of the field personnel and the substantial number of hours spent in the study area from 1998 to 2001, far fewer observations of mink and otter, or their sign, occurred than would be normally expected in a riverine system such as the PSA.

9.5.1.2 GE Study

The General Electric (GE) company studied the presence or absence and possible distribution of wild mink in the Housatonic River Study Area from the spring of 2001 to the spring of 2003 (Bernstein et al. 2003). River otter (*Lutra canadensis*) were included in the study of winter of 2002/2003. The methods were similar to those used in the EPA surveys, and consisted of looking for tracks in soft sand at scent post stations in the spring, summer, and fall (mink only), and snow tracking (mink and otter) during the winter months.

In summary, the GE report cited incidences of mink and river otter signs in the PSA. However, the study had several limitations, which lead to conclusions that are not supported by the data. These limitations included the failure to discuss the implications of the disproportionate number of sightings in winter versus other seasons, apparent ineffectiveness of scent posts, no established reference areas outside the PSA, lack of tracking expertise and experience, empty traps, no results from motion-sensitive camera trials (i.e., no mink or river otter observed in the snow-free months trials), uncertainty in determining sex of mink from tracks, and uncertainty in attributing different sets of tracks to separate individuals.

9.5.2 Comparison of Estimated Exposures to Laboratory-Derived Effects Doses

Exposure was assessed for mink and river otter in the PSA. Because Reaches 5 and 6 combined roughly correspond to the size of the home range for otter, these reaches were combined for the river otter analysis. For mink, the assessment was conducted separately for each reach because of their smaller foraging range

For each receptor-COC combination, a category of low, intermediate, or high risk was assigned following integration of the exposure and effects distributions. This exercise was done separately for the results of the Monte Carlo analyses and each of the lower and upper bounds from the probability bounds analyses. The “risk category” refers to the level of risk based on the results of the Monte Carlo analysis. The “risk range” refers to the levels of risk based on the results of the probability bounds analyses.

Guidance for Determining Level of Risk to Representative Species

Risk Curves for Mink and River Otter Available

- If the probability of 10% or greater effect was less than 20%, then the risk was low.
- If the probability of 20% or greater effect was greater than 50%, then the risk was high.
- All other outcomes were considered to have intermediate risk.

Risk Curves for Mink and River Otter Not Available

- If the probability of exceeding the lower toxicity threshold was less than 20%, the risk was low.
- If the probability of exceeding the upper toxicity threshold was greater than 20%, the risk was high.
- All other outcomes for the lower and upper thresholds were considered to have intermediate risk.

The results of the risk characterization are summarized in Table 9.5-1. Figures 9.5-1 through 9.5-14 depict the risk curves for mink and river otter exposed to tPCBs in Reaches 5 and 6, including the Monte Carlo results and both the lower and upper probability bounds (LPB and UPB, respectively). The highest risk to mink and river otter is from exposure to tPCBs in Reaches 5 and 6. Risks were much lower in the reference areas. The exposure in Reaches 5 and 6 is so high that individuals foraging 10% of their time at those locations would experience high risk.

Table 9.5-1

Summary of Qualitative Risk Statements for Piscivorous Mammals from the Housatonic River Primary Study Area (Reaches 5 and 6) and Nearby Reference Areas

Location	Qualitative Risk Statements				
	tPCBs			TEQ	
	Risk Category	Risk Range		Risk Category	Risk Range
Mink					
Reach 5	High	High		High	Intermediate/High
Reach 6	High	High		High	Intermediate/High
Upstream Reference Area	Intermediate	Low/Intermediate		High	Intermediate/High
Threemile Pond	Intermediate	Low/High		Intermediate	Low/Intermediate
River Otter					
Reaches 5 and 6	High	High		High	High
Upstream Reference Area	Intermediate	Intermediate		Intermediate	Low/Intermediate
Threemile Pond	Low	Low		Low	Low/Intermediate

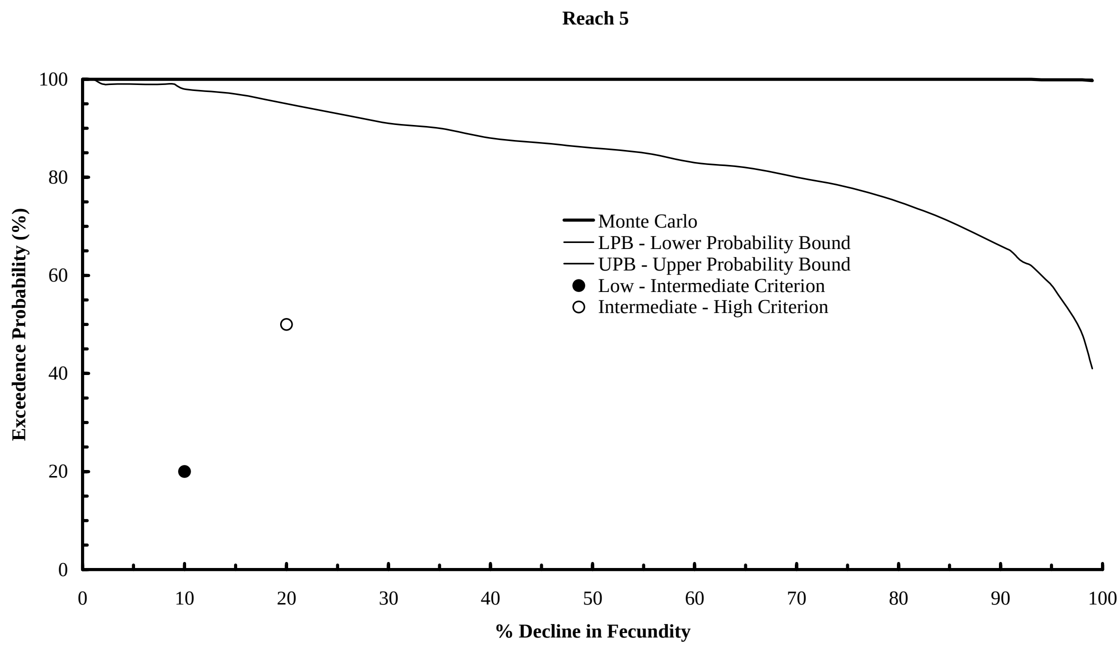


Figure 9.5-1 Total Risk to Mink Exposed to tPCBs in Reach 5 of the Housatonic River

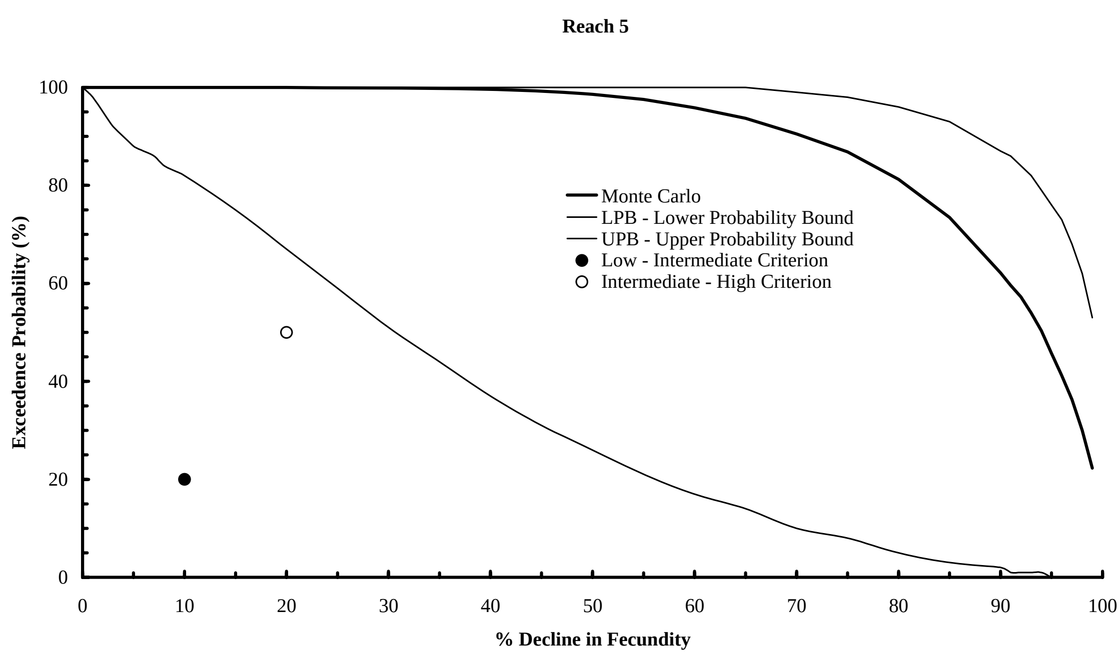


Figure 9.5-2 Total Risk to Mink Exposed to tPCBs in Reach 5 (10% Foraging Time)

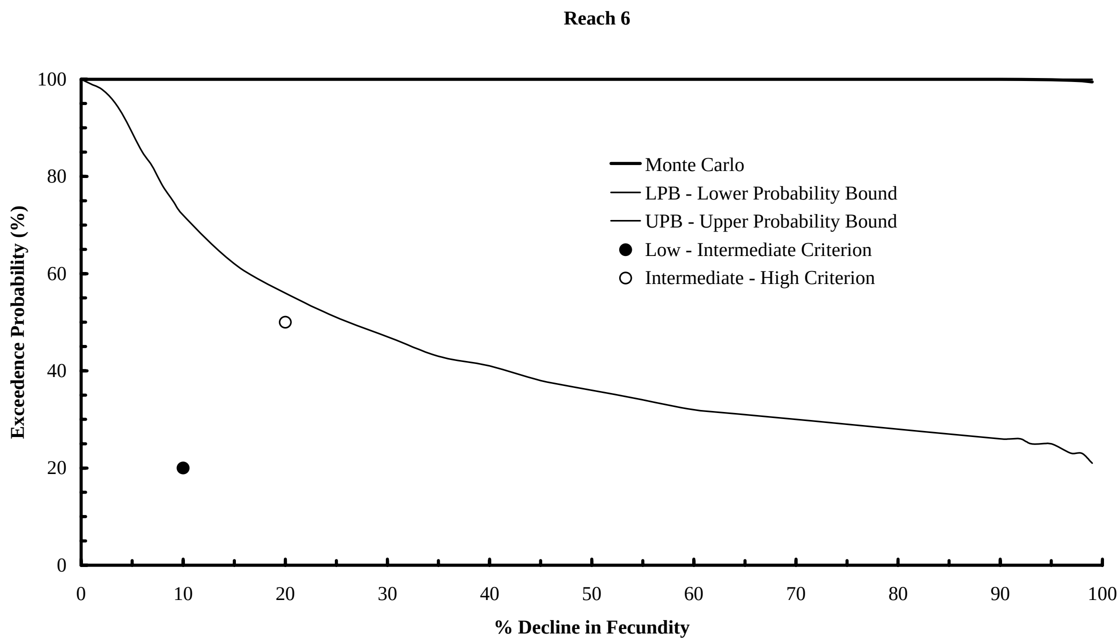


Figure 9.5-3 Total Risk to Mink Exposed to tPCBs in Reach 6 of the Housatonic River

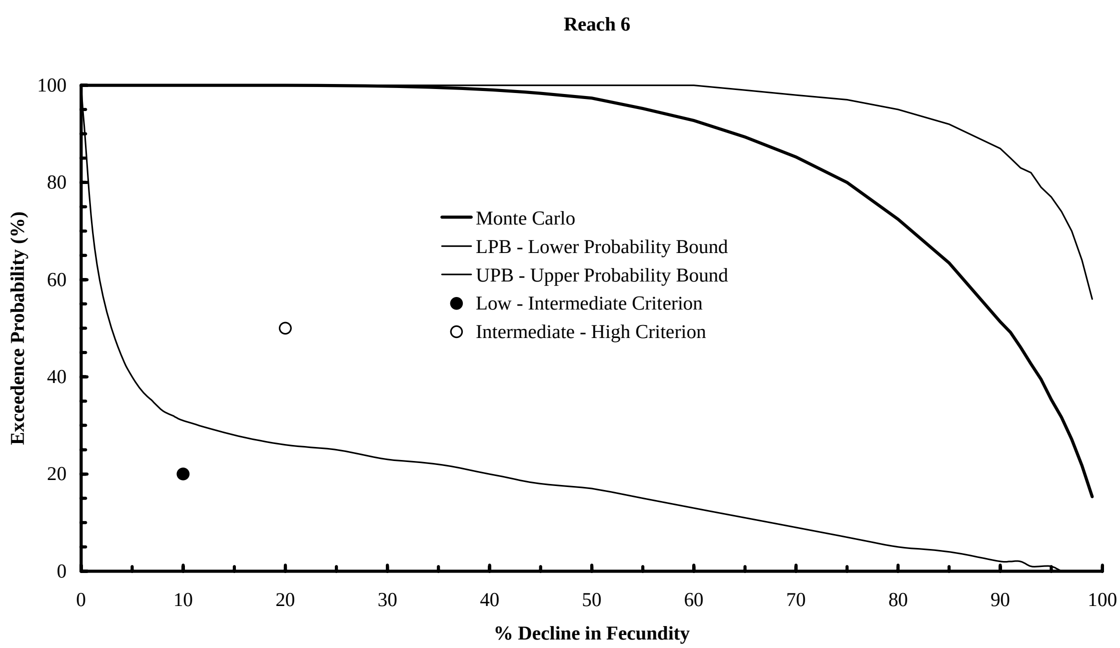


Figure 9.5-4 Total Risk to Mink Exposed to tPCBs in Reach 6 (10% Foraging Time)

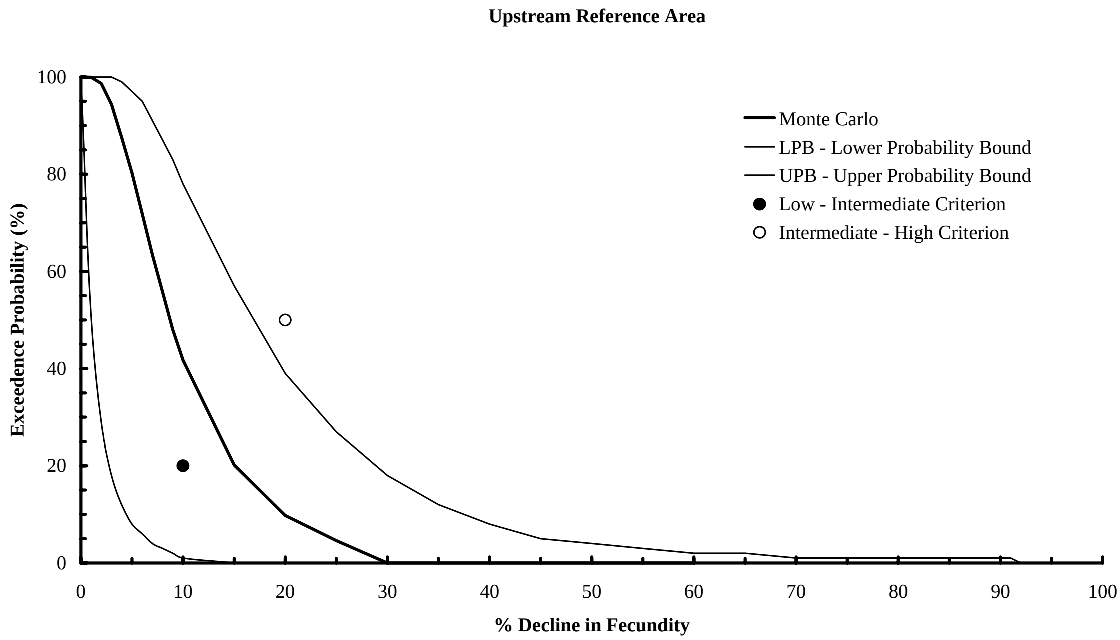


Figure 9.5-5 Total Risk to Mink Exposed to tPCBs in the Upstream Reference Area

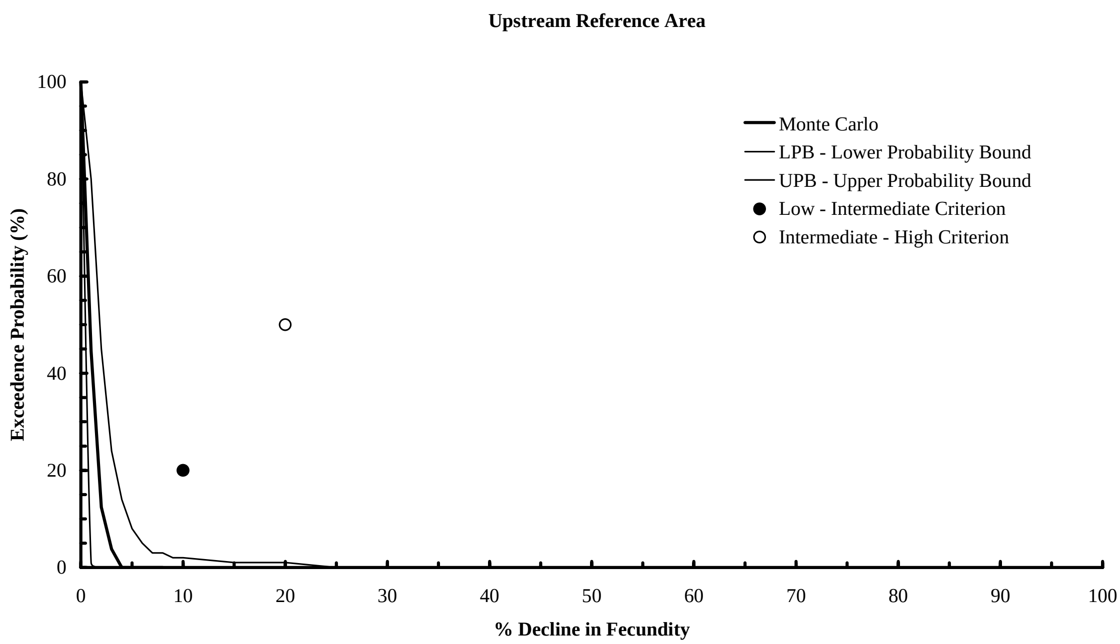


Figure 9.5-6 Total Risk to Mink Exposed to tPCBs in the Upstream Reference Area (10% Foraging Time)

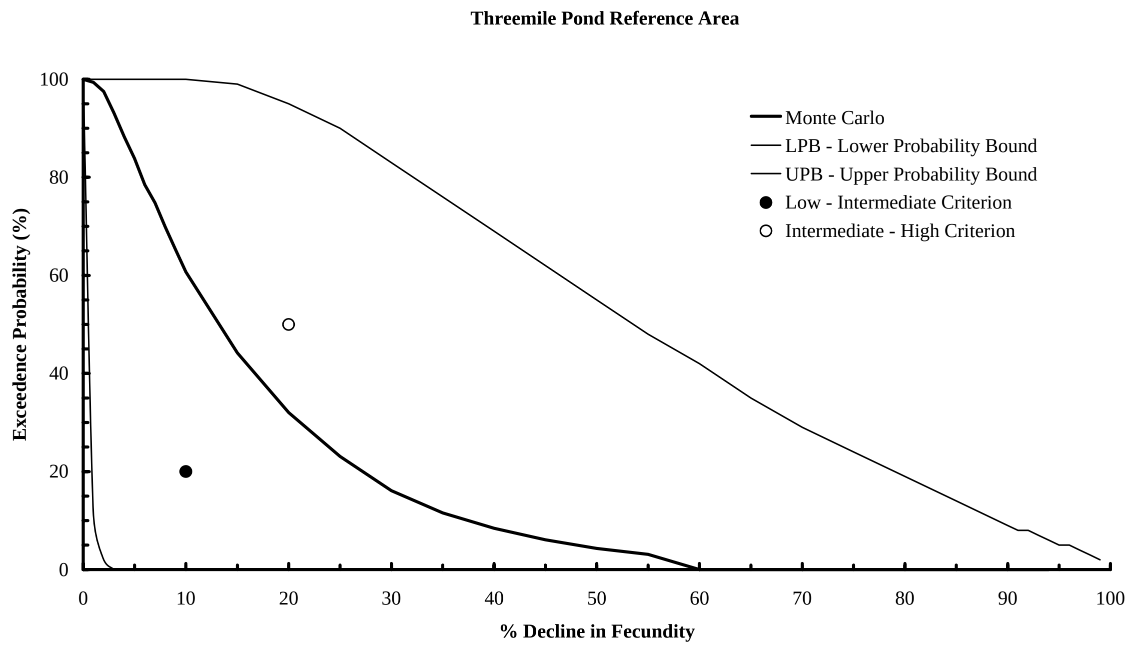


Figure 9.5-7 Total Risk to Mink Exposed to tPCBs in the Threemile Pond Reference Area

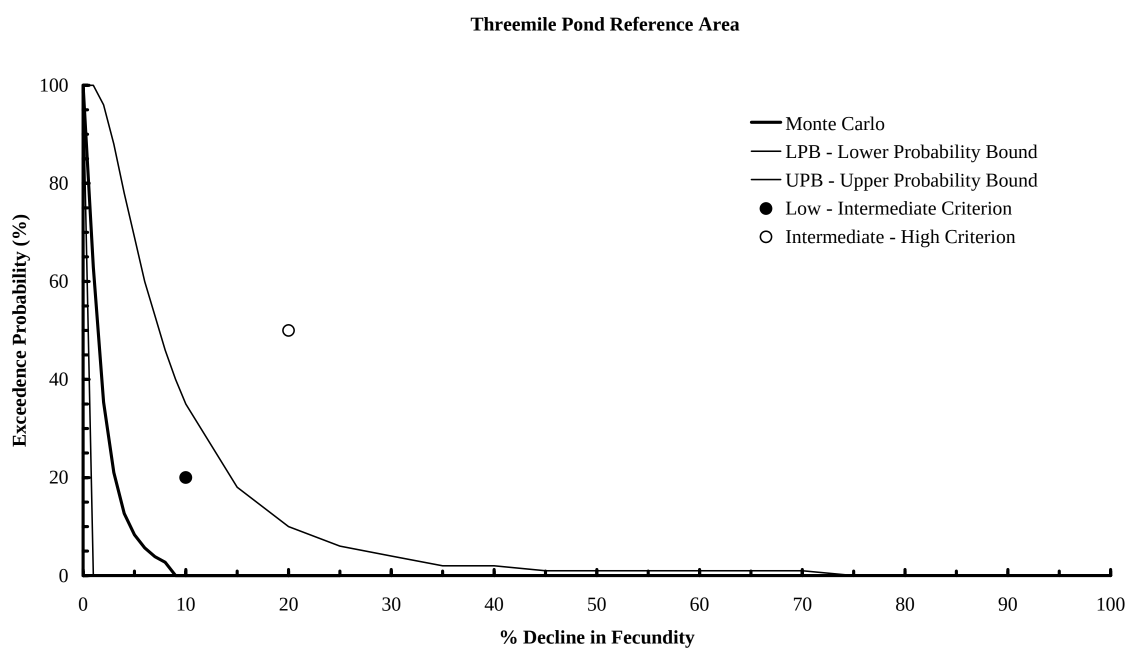
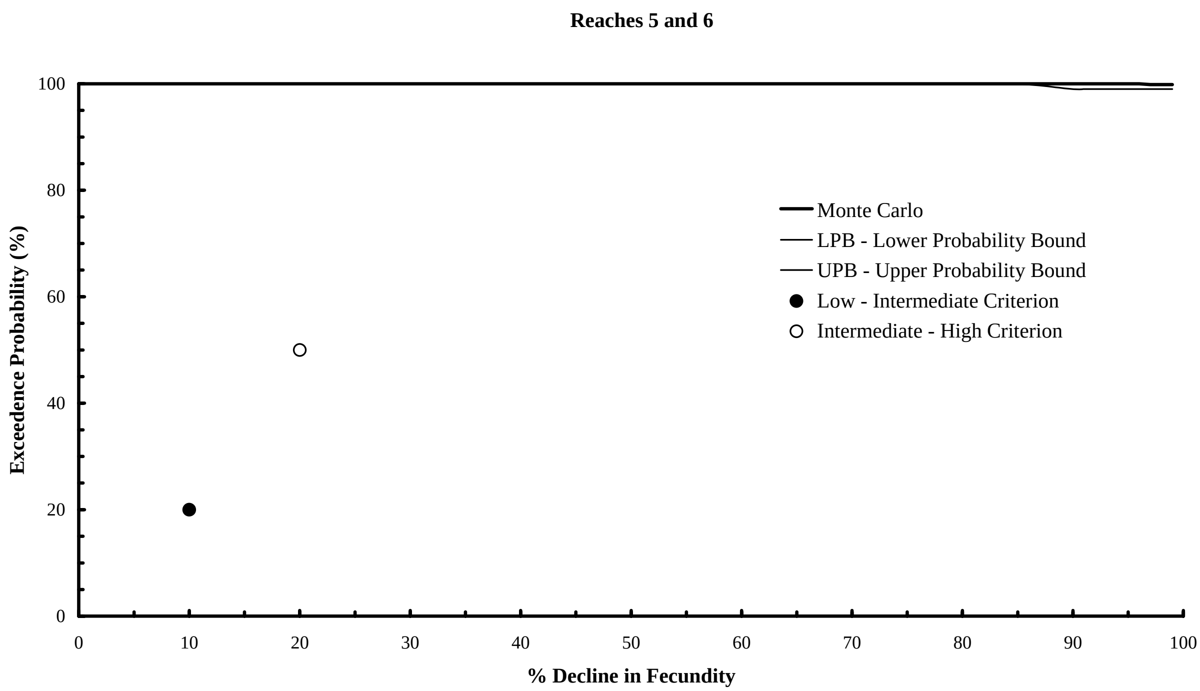


Figure 9.5-8 Total Risk to Mink Exposed to tPCBs in the Threemile Pond Reference Area (10% Foraging Time)



Note: The LPB and UPB overlap the Monte Carlo line.

Figure 9.5-9 Total Risk to River Otter Exposed to tPCBs in Reaches 5 and 6 of the Housatonic River

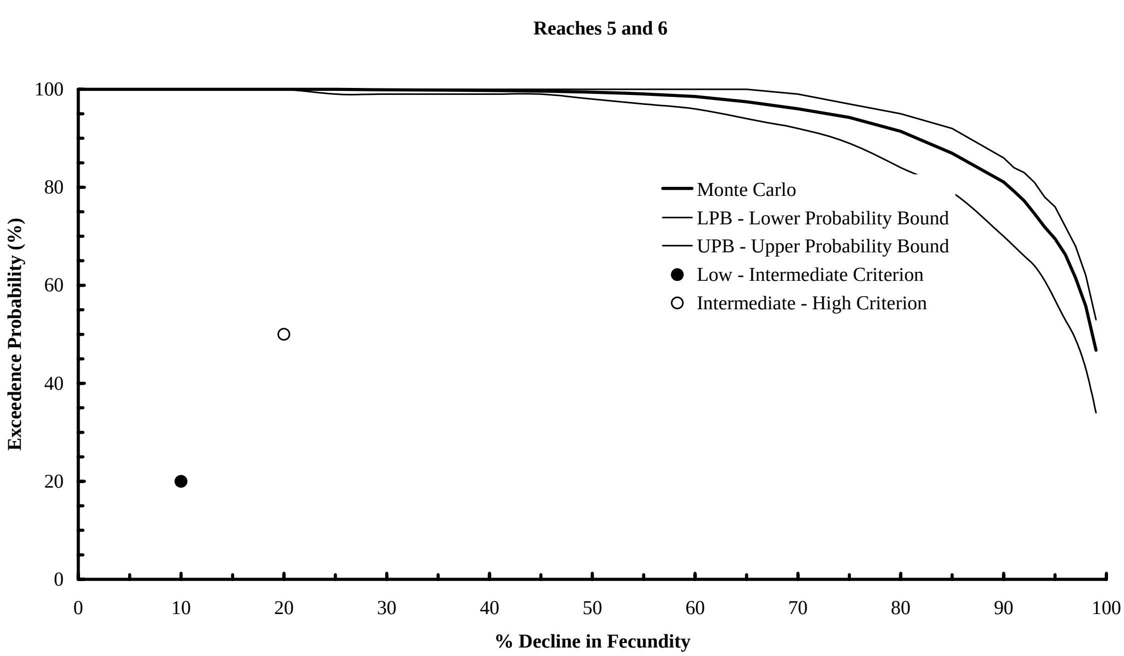


Figure 9.5-10 Total Risk to River Otter Exposed to tPCBs in Reaches 5 and 6 (10% Foraging Time)

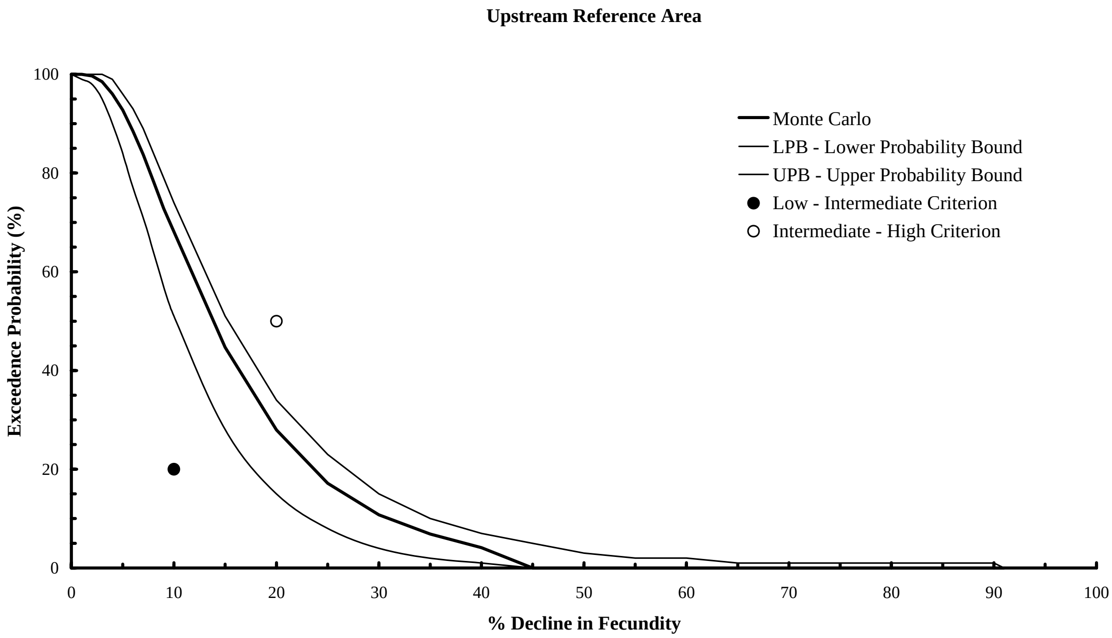


Figure 9.5-11 Total Risk to River Otter Exposed to tPCBs in the Upstream Reference Area

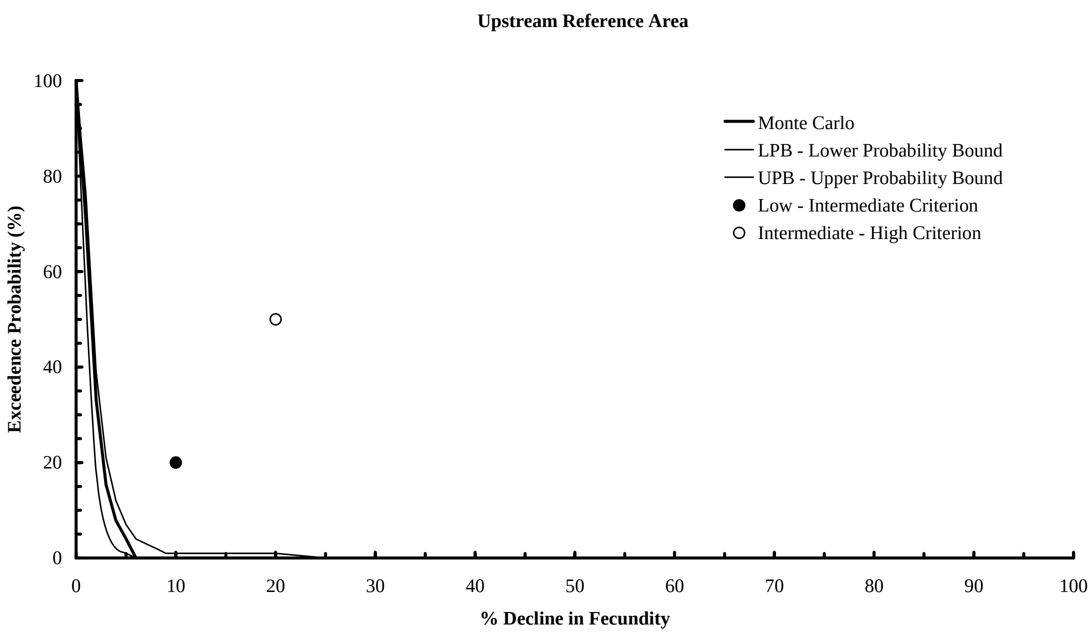


Figure 9.5-12 Total Risk to River Otter Exposed to tPCBs in the Upstream Reference Area (10% Foraging Time)

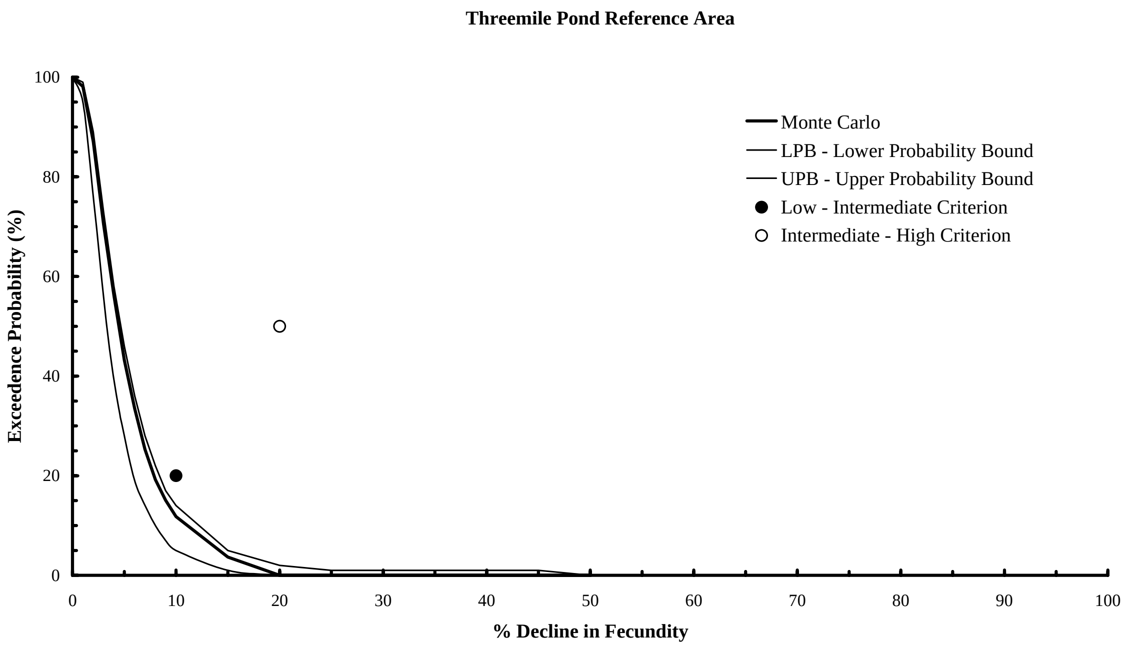


Figure 9.5-13 Total Risk to River Otter Exposed to tPCBs in the Threemile Pond Reference Area

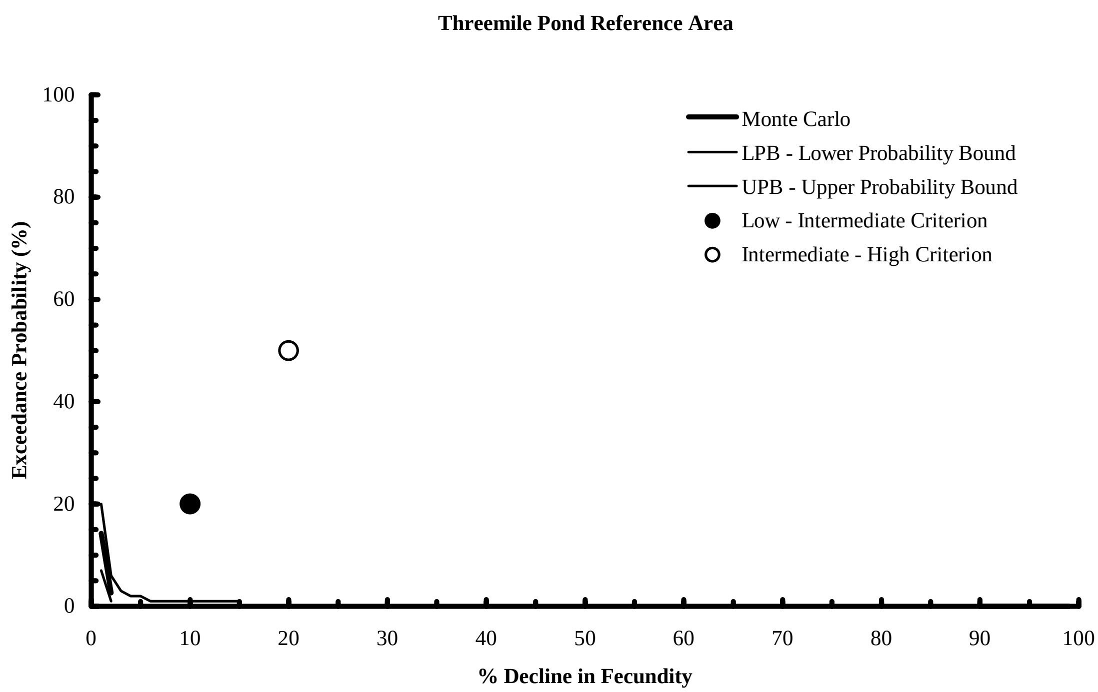


Figure 9.5-14 Total Risk to River Otter Exposed to tPCBs in the Threemile Pond Reference Area (10% Foraging Time)

9.5.3 Mink Feeding Study

Kit survival at 6 weeks of age was significantly decreased in the 3.7 mg/kg tPCBs (68.5 ng/kg TEQ) treatment group. In this treatment, less than 4% of the diet was derived from Housatonic River fish, which is well below what mink typically consume in the wild (23% on average with a range from 0 to 65%). Further, regression analysis indicated significant relationships ($p < 0.05$) between kit survival from 0 to 6 weeks and tPCB concentration or TEQ concentration in the diet. For tPCBs, the dietary LC_{10} and LC_{20} were 0.231 and 0.984 mg/kg, respectively. For TEQ, the dietary LC_{10} and LC_{20} were 4.22 and 16.2 ng/kg, respectively. The tPCB and TEQ concentrations in fish in the PSA are approximately two orders of magnitude above the tPCB and TEQ LC_{10}s.

The histopathological examination of kit jawbones revealed that jaw lesions were apparent at tPCB treatments as low as 0.96 mg/kg diet (0.88% Housatonic River fish in the diet) (Bursian and Yamini 2003). There was also a dose-response relationship between frequency of jaw lesions in mink kits and dietary tPCB and TEQ concentration.

9.5.4 Weight-of-Evidence Analysis

For the WOE analysis, the three-phase approach of Menzie et al. (1996) and the Massachusetts Weight-of-Evidence Workgroup was used, in which WOE was expressed with the following three characteristics: (1) the weight assigned to each measurement endpoint; (2) the magnitude of response observed in the measurement endpoint; and (3) the concurrence among outcomes of the multiple measurement endpoints.

Each measurement endpoint was evaluated and assigned a qualitative weight in Appendix I, along with a discussion of the reason for the value assigned (Tables 9.5-2 and 9.5-3). The magnitude of the response in the measurement endpoint is considered together with the measurement endpoint value in developing the overall WOE (Menzie et al. 1996). This requires assessing the strength of evidence that ecological harm has occurred, as well as an indication of the magnitude of response, if present. The weighting values, evidence of harm, and magnitude of response were combined in a matrix format and are presented in Tables 9.5-4 and 9.5-5 for mink and Tables 9.5-6 and 9.5-7 for river otter.

Table 9.5-2

Weighting of Measurement Endpoints for Mink WOE Evaluation

Attributes	Field Surveys		Feeding Study	Modeled Exposure and Effects	
	EPA	GE		tPCBs	TEQ
I. Relationship Between Measurement and Assessment Endpoints					
1. Degree of Association	L/M	L/M	H	M/H	M/H
2. Stressor/Response	L/M	L/M	H	H	M
3. Utility of Measure	L/M	L	H	H	M
II. Data Quality					
4. Data Quality	H	L	H	M/H	M/H
III. Study Design					
5. Site Specificity	H	M/H	M	L/M	L/M
6. Sensitivity	M	L/M	M/H	H	M
7. Spatial Representativeness	H	M/H	M/H	M	M
8. Temporal Representativeness	H	H	M/H	M/H	M/H
9. Quantitative Measure	M/L	L/M	H	H	M/H
10. Standard Method	H	M	H	M/H	M/H
Overall Endpoint Value	**M/H**	**M**	**H**	**M/H**	**M**

L = low; L/M = low-moderate; M = moderate; M/H = moderate-high; H = high

Table 9.5-3

Weighting of Measurement Endpoints for River Otter WOE Evaluation

Attributes	Field Surveys		Modeled Exposure and Effects	
	EPA	**GE**	**tPCBs**	**TEQ**
I. Relationship Between Measurement and Assessment Endpoints				
1. Degree of Association	L/M	L/M	M/H	M/H
2. Stressor/Response	L/M	L/M	H	M
3. Utility of Measure	L/M	L	H	M
II. Data Quality				
4. Data Quality	H	L	M/H	M/H
III. Study Design				
5. Site Specificity	H	M/H	L/M	L/M
6. Sensitivity	M	L/M	H	M
7. Spatial Representativeness	H	M/H	M	M
8. Temporal Representativeness	H	L/M	M/H	M/H
9. Quantitative Measure	L/M	L/M	H	M/H
10. Standard Method	H	M	M/H	M/H
Overall Endpoint Value	**M/H**	**L/M**	**M/H**	**M**

L = low; L/M = low-moderate; M = moderate; M/H = moderate-high; H = high

Table 9.5-4

Evidence of Harm and Magnitude of Effects for Mink Exposed to tPCBs in the Housatonic River PSA

Measurement Endpoints		Weighting Value (High, Moderate, Low)	Evidence of Harm (Yes, No, Undetermined)	Magnitude (High, Intermediate, Low)
Field Surveys	EPA	Moderate/High	Yes	High
	GE	Moderate	No	---
Feeding Study		High	Yes	High
Modeled Exposure and Effects		Moderate/High	Yes	High

Table 9.5-5

Evidence of Harm and Magnitude of Effects for Mink Exposed to TEQ in the Housatonic River PSA

Measurement Endpoints		Weighting Value (High, Moderate, Low)	Evidence of Harm (Yes, No, Undetermined)	Magnitude (High, Intermediate, Low)
Field Surveys	EPA	Moderate/High	Yes	High
	GE	Moderate	No	---
Feeding Study		High	Yes	High
Modeled Exposure and Effects		Moderate	Yes	High

Table 9.5-6

Evidence of Harm and Magnitude of Effects for River Otter Exposed to tPCBs in the Housatonic River PSA

Measurement Endpoints		Weighting Value (High, Moderate, Low)	Evidence of Harm (Yes, No, Undetermined)	Magnitude (High, Intermediate, Low)
Field Surveys	EPA	Moderate/High	Yes	High
	GE	Low/Moderate	No	---
Modeled Exposure and Effects		Moderate/High	Yes	High

Table 9.5-7

Evidence of Harm and Magnitude of Effects for River Otter Exposed to TEQ in the Housatonic River PSA

Measurement Endpoints		Weighting Value (High, Moderate, Low)	Evidence of Harm (Yes, No, Undetermined)	Magnitude (High, Intermediate, Low)
Field Surveys	EPA	Moderate/High	Yes	High
	GE	Low/Moderate	No	---
Modeled Exposure and Effects		Moderate	Yes	High

The three lines of evidence for mink and two lines of evidence for river otter indicated that the elevated concentrations of tPCBs and TEQ in the PSA of the Housatonic River are causing adverse effects of high magnitude to mink and river otter. The field surveys indicated that mink and river otter are rarely present in the PSA, except during winter, and likely have not established home territories close to the main channel despite suitable mink and otter habitat. The MSU feeding study indicated that feeding adult female mink with a diet containing as little as approximately 1% fish from the PSA would cause a 20% decrease in kit survival from 0 to 6 weeks of age. Further, the jaw lesion study indicated that erosion of the jaw occurs at even lower doses and exhibits a dose-response relationship.

The high risks evident from the feeding study are further supported by the modeled exposure and effects line of evidence. The estimated potential for exposure is so high that even individual mink and otter that only forage in the PSA for short periods of time (less than or equal to 10% of foraging time) are at an intermediate or higher risk from tPCBs and TEQ.

A graphical method was used for displaying concurrence among measurement endpoints. Tables 9.5-8 and 9.5-9 depict the outcome for mink exposed to tPCBs and TEQ, respectively. Tables 9.5-10 and 9.5-11 depict the outcome for river otter exposed to tPCBs and TEQ, respectively. The measurement endpoints for mink and river otter have a high degree of concurrence.

Table 9.5-8

Risk Analysis Summary for Mink Exposed to tPCBs in the Housatonic River PSA

Assessment Endpoint: Survival, growth, and reproduction of piscivorous mammals

	Weighting Factors (increasing confidence of weight)				
Harm/Magnitude	**Low**	**Low/Moderate**	**Moderate**	**Moderate/High**	**High**
Yes/High				MEE, FS-EPA	MFS
Yes/Intermediate					
Yes/Low					
Undetermined					
No			FS-GE		

FS-EPA = Field surveys by EPA
FS-GE = Field surveys by GE
MFS = Mink feeding study
MEE = Modeled exposure and effects

Table 9.5-9

Risk Analysis Summary for Mink Exposed to TEQ in the Housatonic River PSA

Assessment Endpoint: Survival, growth, and reproduction of piscivorous mammals

Harm/Magnitude	Weighting Factors (increasing confidence of weight)				
	Low	**Low/Moderate**	**Moderate**	**Moderate/High**	**High**
Yes/High			MEE	FS-EPA	MFS
Yes/Intermediate					
Yes/Low					
Undetermined					
No			FS-GE		

FS-EPA = Field surveys by EPA
FS-GE = Field surveys by GE
MFS = Mink feeding study
MEE = Modeled exposure and effects

Table 9.5-10

Risk Analysis Summary for River Otter Exposed to tPCBs in the Housatonic River PSA

Assessment Endpoint:	Survival, growth, and reproduction of piscivorous mammals

Harm/Magnitude	**Weighting Factors (increasing confidence of weight)**				
	Low	**Low/Moderate**	**Moderate**	**Moderate/High**	**High**
Yes/High				MEE, FS-EPA	
Yes/Intermediate					
Yes/Low					
Undetermined					
No		FS-GE			

FS-EPA = Field surveys by EPA
FS-GE = Field surveys by GE
MEE = Modeled exposure and effects

Table 9.5-11

Risk Analysis Summary for River Otter Exposed to TEQ in the Housatonic River PSA

Assessment Endpoint: Survival, growth, and reproduction of piscivorous mammals

Harm/Magnitude	Weighting Factors (increasing confidence of weight)				
	Low	Low/Moderate	Moderate	Moderate/High	High
Yes/High			MEE	FS-EPA	
Yes/Intermediate					
Yes/Low					
Undetermined					
No		FS-GE			

FS-EPA = Field surveys by EPA
FS-GE = Field surveys by GE
MEE = Modeled exposure and effects

9.5.5 Development of the Maximum Acceptable Threshold Concentration (MATC)

The MATC of 0.984 mg/kg tPCBs in fish (whole body, wet weight) is the LC_{20} for kit survival from 0 to 6 weeks determined from the regression analyses of the results of the Bursian et al. (2003) study of the toxicity of Housatonic River fish to mink.

9.5.6 Sources of Uncertainty

Some of the major sources of uncertainty associated with the assessment of risks of tPCBs and TEQ to piscivorous mammals are briefly summarized below. A more complete list is presented in Appendix I.

- Assumption that dietary exposure represented the most important pathway for exposure of mink and river otter to COCs.
- Estimation of free metabolic rates using allometric equations.

- Lack of stomach contents or other dietary analyses for mink in the PSA.
- Limited sample sizes for the analyses of COC concentrations in some prey items.
- Lack of data on concentrations of tPCBs and TEQ in crayfish and mammals in Reach 6.
- Limited data on concentrations of tPCBs and TEQ in crayfish, birds, and mammals from reference areas.
- Assumption that mink and river otter forage 100% of their time in the PSA.
- Effects metrics used to estimate risk to mink and river otter via exposure models were derived for Aroclor 1254 mixtures.
- Difference in PCB-126 concentrations in mink diet for MSU study and in likely Housatonic River prey species.
- Lack of information needed to confirm track identification (i.e., multiple measurements with a scale) and sex determination in the GE mink and river otter study.
- Lack of reference areas in the GE mink and river otter study.

9.5.7 Comparison to Other Piscivorous Mammals

There are no piscivorous mammals other than mink and river otter in the PSA.

9.5.8 Risk Downstream of PSA

The risk for mink and river otter associated with exposure to tPCBs downstream of the PSA was assessed by comparing concentrations of tPCBs in prey fish (5 to 20 cm) in Reaches 7 to 16 to the MATC developed specifically for mink (also used for river otter).

Fish tissue data were obtained from sampling efforts conducted during 1998 to 2002. The results of the analysis are presented in Figures I.5-15 and I.5-16. Potential risk to mink and river otter exists in river sections from Woods Pond to the end of Reach 15.

9.5.9 Conclusions

For mink, data from three major lines of evidence were available, including field surveys, mink feeding study, and exposure and effects modeling. Two lines of evidence were available for river otter (field surveys, exposure and effects modeling). In general, the weight-of-evidence analysis indicates a high risk for mink and river otter to tPCBs and TEQ in the PSA.

Field surveys by EPA and GE were conducted to determine the presence of mink and river otter in the PSA. Signs of mink and river otter were observed in the PSA, but nearly always in winter, suggesting that mink and river otter that are present in the PSA are there on a transient basis.

The mink feeding study was designed to determine the effects on growth and reproduction of captive mink fed a diet containing fish from the PSA. The results from this study indicated that feeding adult female mink with a diet containing as little as approximately 1% fish from the PSA caused a 20% decline in kit survival from 0 to 6 weeks of age. Because mink in the wild typically consume between 0 and 65% fish in their diet (mean 23%), the associated risk is correspondingly higher. Further, the jaw lesion study indicates that erosion of the jaw occurs at even lower doses and exhibits a dose-response relationship.

The modeling of exposure and effects line of evidence was used to determine the level of risk to the representative mammal species, mink and river otter. The effects characterization developed a dose-response curve to describe the potential effects of tPCBs to mink and river otter. Toxicity benchmarks based on mink studies were developed for TEQ. The dose-response curve for effects of tPCBs to mink and river otter indicated that 10% and 20% declines in fecundity would be expected at doses of 0.0181 and 0.0384 mg/kg bw/d, respectively. For TEQ benchmarks (reduction in kit growth), the lower threshold was set at 3.6 ng/kg bw/d and the upper threshold was set at 36 ng/kg bw/d. The modeled exposure results indicated that the daily intake rates of tPCBs by mink and river otter were far greater than the toxicity thresholds. This means that mink and river otter feeding in the PSA receive tPCB doses that cause adverse reproductive effects. A similar conclusion was reached for TEQ.

ERA Summary

The weight-of-evidence analysis indicates a high risk for mink and river otter exposed to tPCBs and TEQ in the PSA.

The risk continues to be elevated for individuals that forage only a small fraction of their time in the PSA.

The MATC for piscivorous mammals is 0.984 mg/kg tPCBs in diet.

Downstream of the PSA (Reach 6), a screening level ERA indicated that mink and river otter may be at risk from exposure to tPCBs and TEQ as far as Reach 15.

9.6 REFERENCES

Alexander, G.R. 1977. Food of vertebrate predators on trout waters in north central lower Michigan. *The Michigan Academician* 10:181-195.

Anderson, E.A. and A. Woolf. 1987. River otter food habits in northwestern Illinois. *Transactions of the Illinois State Academy of Science* 80:115-118.

Aulerich, R.J., and R.K. Ringer. 1977. Current status of PCB toxicity to mink, and effect on their reproduction. *Archives of Environmental Contamination and Toxicology* 6:279-292.

Aulerich, R.J., S.J. Bursian, W.J. Breslin, B.A. Olson, and R.K. Ringer. 1985. Toxicological manifestations of 2,4,5,2',4',5'-, 2,3,6,2',3',6'-, and 3,4,5,3',4',5'-hexachlorobiphenyl and Aroclor® 1254 in mink. *Journal of Toxicology and Environmental Health* 15:63-79.

Aulerich, R.J., S.J. Bursian, and A.C. Napolitano. 1988. Biological effects of epidermal growth factor and 2,3,7,8-tetrachlorodibenzo-p-dioxin on developmental parameters of neonatal mink. *Archives of Environmental Contamination and Toxicology* 17:27-31.

Aulerich, R.J., S. Bursian, and B. Yamini. 2003. Hepatic 0-dealkylase activities from induction of cytochrome P-450 in mink and their offspring fed diets containing contaminated fish from the Housatonic River. USGS Biochemistry and Physiology Branch Final Laboratory Report FY 2003-30-4. 18 pp.

Barrett, G.W., and K.L. Stueck. 1976. Caloric ingestion rate and assimilation efficiency of the short-tailed shrew, *Blarina brevicauda*. *Ohio Journal of Science* 76:25-26.

Bernstein, P.; M. Chamberlain; Arcadis G&M, Inc.; BBL Sciences, Inc.; Branton Environmental Consulting Report. 2003. *Evaluation of Piscivorous Mammals - Presence/Absence, Distribution, and Abundance in the Housatonic River Floodplain*. Prepared for General Electric Company.

Burgess, S.A. and J.R. Bider. 1980. Effects of stream habitat improvements on invertebrates, trout populations, and mink activity. *Journal of Wildlife Management* 44(4):871-880.

Bursian, S.J., R.J. Aulerich, B. Yamini, and D.E. Tillitt. 2003. *Dietary Exposure of Mink to Fish from the Housatonic River: Effects on Reproduction and Survival*. Michigan State University, Department of Animal Science. Revised Final Report. June 10, 2003.

Bursian, S.J., and B. Yamini. 2003. *Dietary Exposure of Mink to Fish from the Housatonic River. Inducement of Manidibular and Maxillary Squamos Cell Proliferation* Report submitted to Weston Solutions, Inc. March 18, 2003.

Cowan, W.F. and J.R. Reilly. 1973. Summer and fall foods of mink on the J. Clark Salyer National Wildlife Refuge. *Prairie Naturalist* 5:20-24.

EPA (U.S. Environmental Protection Agency). 1993. *Wildlife Exposure Factors Handbook.* Volume 1 of 2. EPA/600/R-93/187a. U.S. Environmental Protection Agency, Office of Research and Development, Washington, DC.

Gilbert, F. F. and E.G. Nancekivell. 1982. Food habits of mink (*Mustela vison*) and otter (*Lutra canadensis*) in northeastern Alberta. *Canadian Journal of Zoology* 60:1282-1288.

Greer, K.R. 1955. Yearly food habits of the river otter in the Thompson Lakes region, northwestern Montana, as indicated by scat analysis. *American Midland Naturalist* 54:299-313.

Grodzinski, W., and B.A. Wunder. 1975. Ecological Energetics of Small Mammals. In *Small Mammals: Their Productivity and Population Dynamics*. F.B. Golley, K. Petrusewicz, and L. Ryszkowksi, Editors. Cambridge University Press, Cambridge, MA.

Hamilton, W.J. Jr. 1940. The summer food of minks and raccoons on the Montezuma Marsh, New York. *Journal of Wildlife Management* 4:80-84.

Hamilton, W.J. Jr. 1959. Foods of mink in New York. *New York Fish and Game Journal* 6(1):77-85.

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

Hochstein, J.R., S.J. Bursian, and R.J. Aulerich. 1998. Effects of dietary exposure to 2,3,7,8-tetrachlorodibenzo-p-dioxin in adult female mink (*Mustela vison*). *Archives of Environmental Contamination and Toxicology* 35:348-353.

Hochstein, J.R., J.A. Render, S.J. Bursian, and R.J. Aulerich. 2001. Chronic toxicity of dietary 2,3,7,8-tetrachlorodibenzo-p-dioxin to mink. *Veterinary and Human Toxicology* 43:134-39.

Hornshaw, T.C., R.J. Aulerich, and H.E. Johnson. 1983. Feeding Great Lakes fish to mink: Effects on mink and accumulation and elimination of PCBs by mink. *Journal of Toxicology and Environmental Health* 11:933-946.

Melquist, W.E., J.S. Whitman, and M.G. Hornocker. 1981. Resource partitioning and coexistence of sympatric mink and river otter populations. In *Worldwide Furbearer Conference Proceedings,* Vol. I. J.A. Chapman and D. Pursley, Editors. Frostberg, MD.

Melquist, W.E., and M.G. Hornocker. 1983. Ecology of river otters in west central Idaho. *Wildlife Monograph 83*. 60 pp.

Menzie, C., M.H. Henning, J. Cura, K. Finkelstein, J. Gentile, J. Maughan, D. Mitchell, S. Petron, B. Potocki, S. Svirsky, and P. Tyler. 1996. Special report of the Massachusetts Weight-of-Evidence Workgroup: A weight-of-evidence approach for evaluating ecological risks. *Human and Ecological Risk Assessment* 2:277-304.

Mitchell, J.L. 1961. Mink movements and populations on a Montana river. *Journal of Wildlife Management* 25:48-54.

Moore, D.R.J., B.E. Sample, G.W. Suter, B.R. Parkhurst, and R.S. Teed. 1999. A probabilistic risk assessment of the effects of methylmercury and PCBs on mink and kingfishers along East Fork Poplar Creek, Oak Ridge, Tennessee, USA. *Environmental Toxicology and Chemistry* 18:2941-2953.

Proulx, G., J.A. McDonnell, and F.F. Gilbert. 1987. The effect of water level fluctuations on muskrat, *Ondatra zibethicus*, predation by mink, *Mustela vison*. *Canadian Field-Naturalist* 101(1):89-92.

Sheldon, W.G., and W.G. Toll. 1964. Feeding habits of the river otter in a reservoir in central Massachusetts. *Journal of Mammalogy* 45:449-455.

Tillitt, D. E., S. R. Birke, and D. K. Nicks. 2003. Hepatic O-dealkylase activities from induction of cytochrome P-450 in mink and their offspring fed diets containing contaminated fish from the Housatonic River. USGS, Columbia Environmental Research Center, Biochemistry and Physiology Branch, Final Laboratory Report FY 2003 – 30 – 4, 2003. 16 p.

Whitaker, J.O., Jr., and W.J. Hamilton Jr. 1998. *Mammals of the Eastern United States*. 3rd edition, Cornell University Press, Ithaca, NY, USA.

10. ASSESSMENT ENDPOINT—SURVIVAL, GROWTH, AND REPRODUCTION OF OMNIVOROUS AND CARNIVOROUS MAMMALS

Highlights

Conceptual Model

The assessment endpoint is the survival, growth, and reproduction of omnivorous and carnivorous mammals in the Housatonic River PSA. Omnivorous and carnivorous mammals, including red fox and northern short-tailed shrew, are exposed to tPCBs and TEQ via trophic transfer. These two species were selected as representative species for the ecological risk assessment (ERA).

Exposure

Exposure of the representative species to tPCBs and TEQ was determined from concentrations of these contaminants of concern (COCs) in prey items and an estimation of the daily intake of the COCs from consumption of prey, and from tissue concentrations in small mammals.

Effects

No data were available on toxicity of tPCBs and TEQ to red fox and northern short-tailed shrew. Surrogate species were used to estimate effects and to generate dose-response curves for each species. Field studies were performed by EPA and GE to evaluate effects as well.

Risk

The weight-of-evidence for red fox suggests, based on two lines of evidence, an intermediate risk to fox exposed to tPCBs and TEQ in the PSA. This finding is uncertain because, although fox were commonly observed during the EPA field surveys, a foraging rate of 50% in Reach 5 was used in the exposure modeling, and species-specific measures of effects were not available.

For short-tailed shrew, data from three major lines of evidence were available, including field surveys, a population demography field study of short-tailed shrew, and exposure and effects modeling. The weight-of-evidence analysis indicates an intermediate risk for short-tailed shrews exposed to tPCBs and TEQ in the PSA. This conclusion, however, is also uncertain because of the lack of definitive findings about whether effects are occurring in two of the lines of evidence (i.e., EPA field surveys and GE population demography field study), and the lack of species-specific measures of effects.

Other omnivorous and carnivorous mammal species are expected to have either higher levels of risk (e.g., smoky shrew, short-tailed weasel, long-tailed weasel), similar levels of risk (e.g., masked shrew, gray fox, fisher), and in one case (coyote), a lower level of risk.

10.1 INTRODUCTION

The purpose of this section is to characterize and quantify the current and potential risks posed to omnivorous and carnivorous mammals. A Pre-ERA (Appendix B) was conducted to identify contaminants, other than tPCBs, that pose potential risks to aquatic biota and wildlife in the PSA. A three-tiered deterministic approach was used to screen COPCs. In the COPC screening specific to this endpoint, several other COPCs (primarily organochlorine pesticides) were screened out because their actual concentrations in the PSA were likely much lower than the measured values due to laboratory interference (see Section 2.4). The COPCs that screened through as COCs for the probabilistic risk assessment for omnivorous and carnivorous mammals were tPCBs and 2,3,7,8-TCDD toxic equivalence (TEQ).

A step-wise approach was used to assess the risks of tPCBs and TEQ to omnivorous and carnivorous mammals in the Housatonic River watershed. The four main steps in this process include the following:

1. Derivation of a conceptual model (Figures 10.1-1 and 10.1-2).
2. Assessment of exposure of red fox and northern short-tailed shrew to COCs (Figure 10.1-3).
3. Assessment of the effects of COCs on red fox and northern short-tailed shrew (Figure 10.1-4).
4. Characterization of risks to red fox and northern short-tailed shrew community (Figure 10.1-5).

This section provides a summary of the ERA for omnivorous and carnivorous mammals, which is presented in detail in Appendix J.

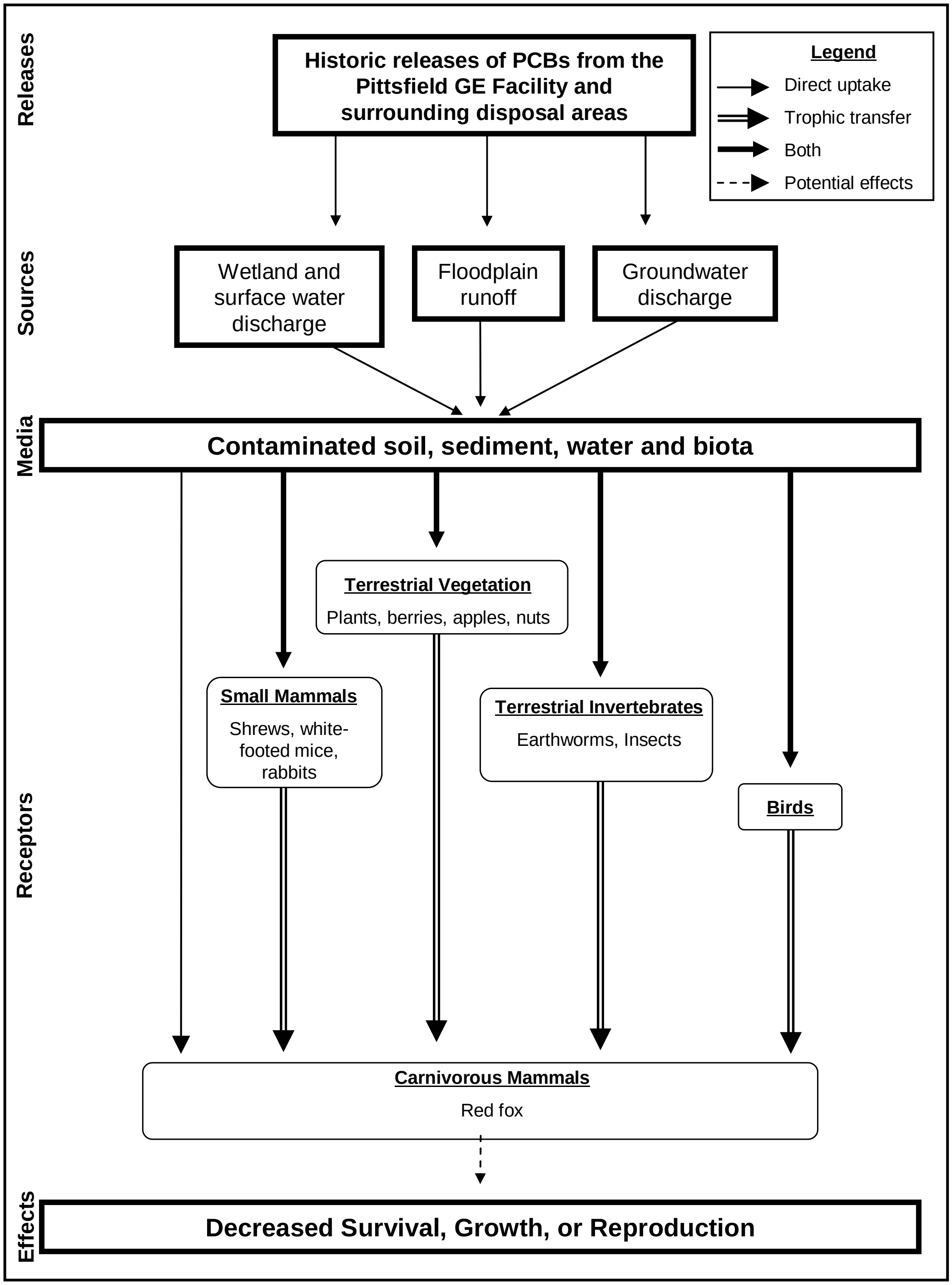


Figure 10.1-1 Conceptual Model Diagram: Exposure Pathways for Red Fox Exposed to Contaminants of Concern (COCs) in the Housatonic River PSA

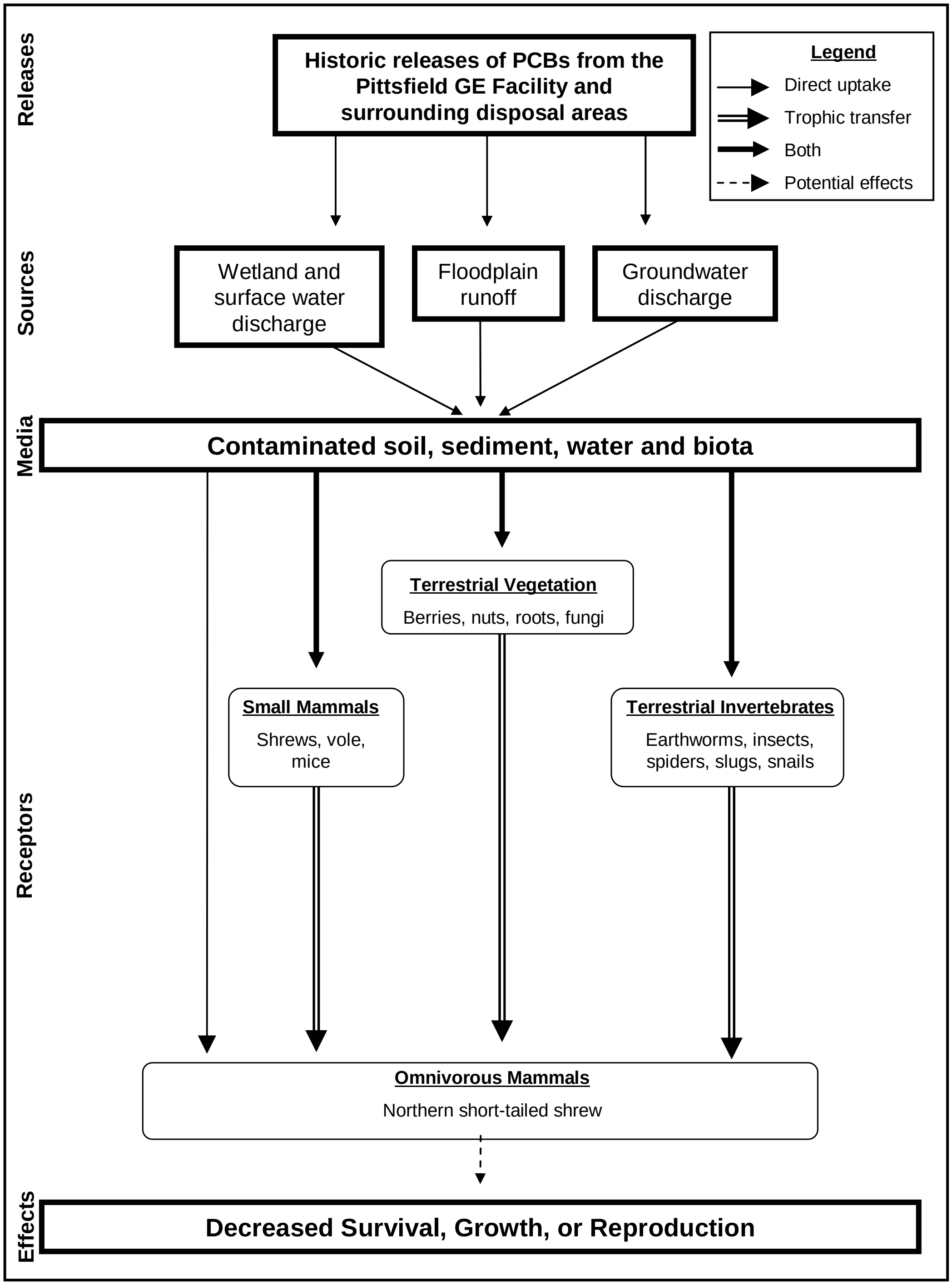


Figure 10.1-2 Conceptual Model Diagram: Exposure Pathways for Northern Short-Tailed Shrew Exposed to Contaminants of Concern (COCs) in the Housatonic River PSA

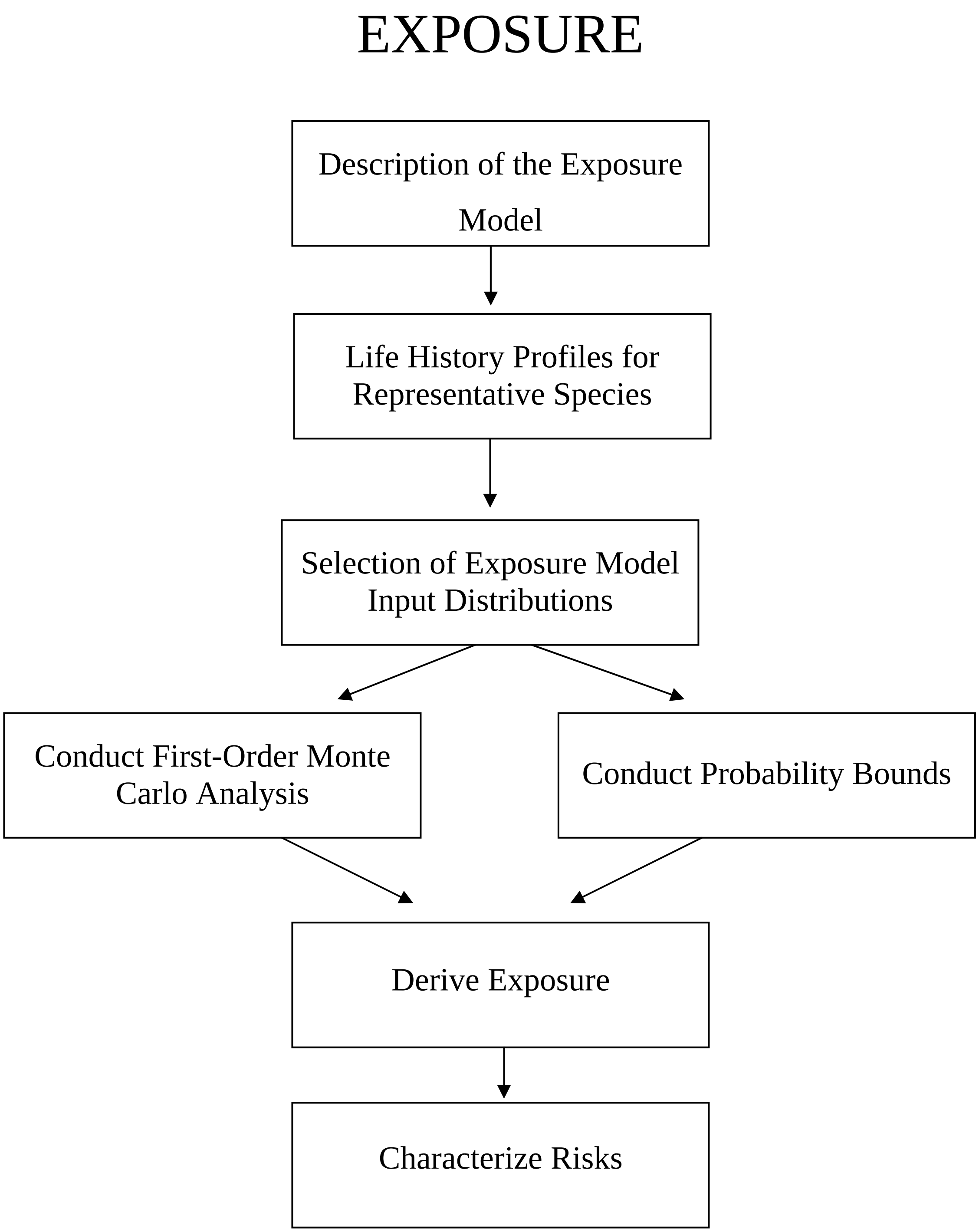


Figure 10.1-3 Framework Used to Model Exposure of Wildlife Species to Contaminants of Concern (COCs) in the Housatonic River PSA

EFFECTS

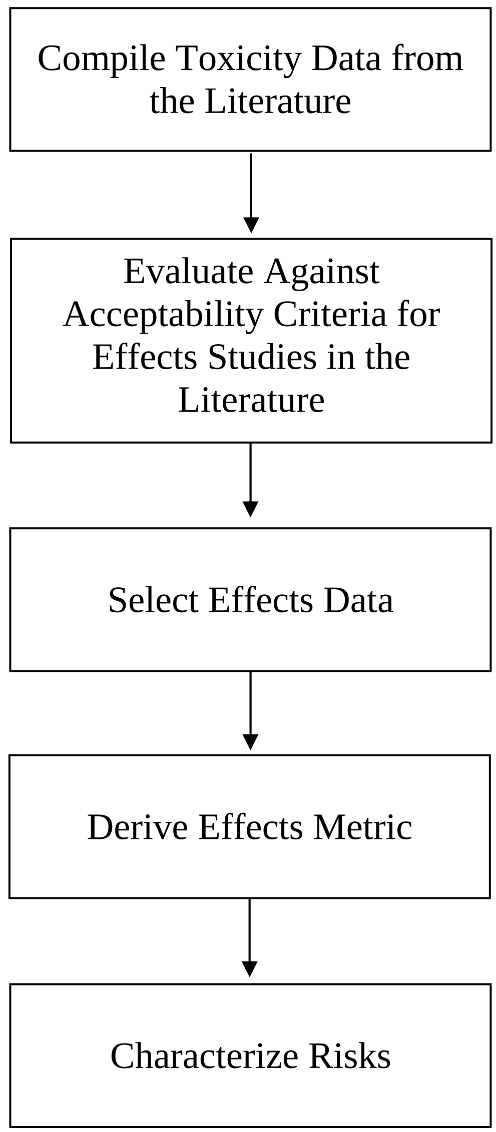


Figure 10.1-4 Approach Used to Model Effects of Contaminants of Concern (COCs) to Representative Species in the Housatonic River PSA

RISK CHARACTERIZATION

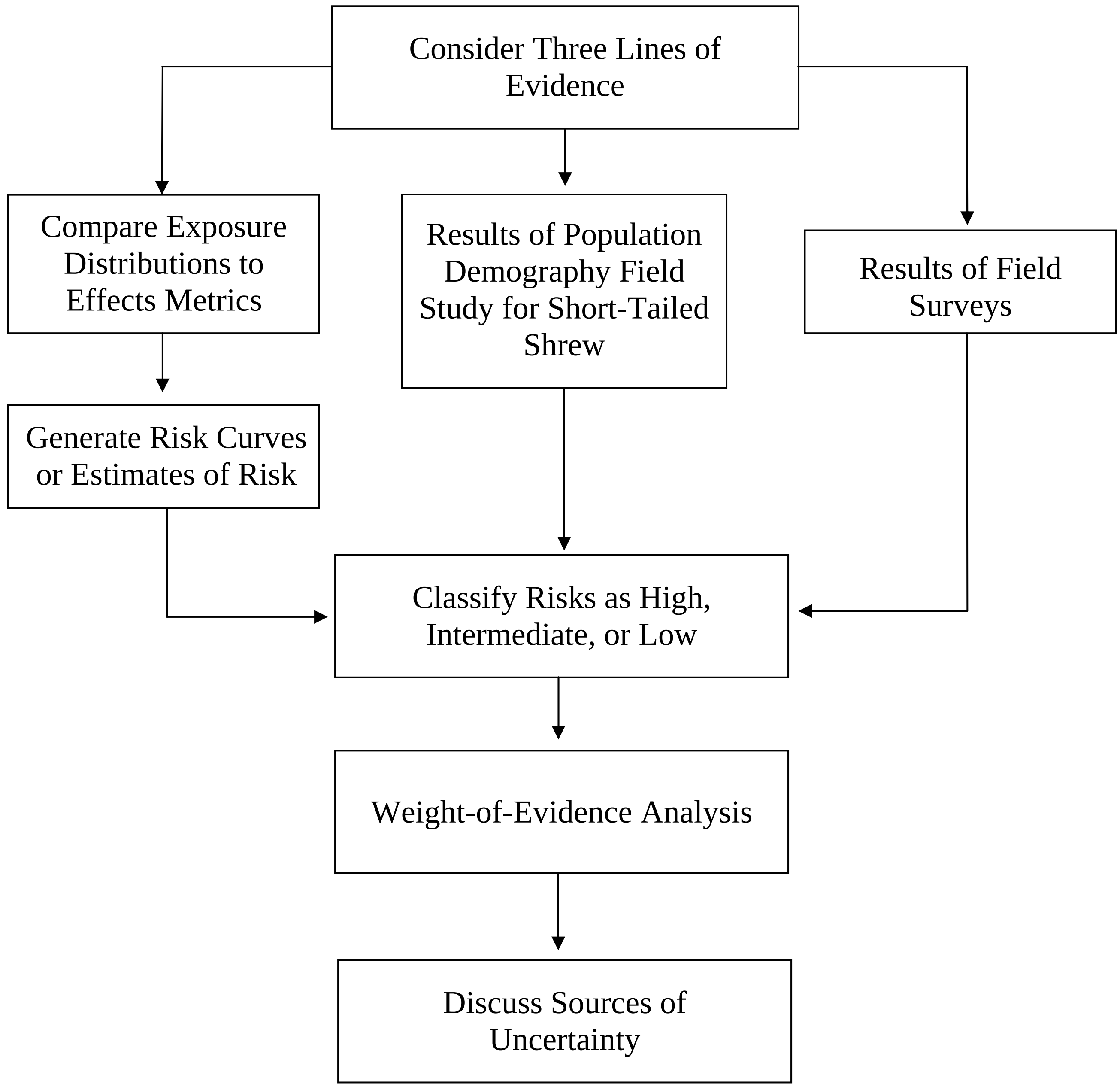


Figure 10.1-5 Overview of Approach Used to Characterize the Risks of Contaminants of Concern (COCs) to Omnivorous and Carnivorous Mammals in the Housatonic River PSA

10.2 CONCEPTUAL MODEL

The conceptual models presented in Figures 10.1-1 and 10.1-2 illustrate the exposure pathways for red fox and northern short-tailed shrews, respectively, exposed to tPCBs and TEQ in the PSA. Omnivorous and carnivorous mammals that reside, or partially reside, within the study area are exposed to tPCBs and TEQ principally through diet as a result of trophic transfer.

The problem formulation (see Section 2) identified the red fox (*Vulpes vulpes*) as the representative species for carnivorous mammals; the northern short-tailed shrew (*Blarina brevicauda*) was selected as the representative species for omnivorous mammals. Life history profiles for the red fox and short-tailed shrew are provided in the text boxes.

The assessment endpoint is the survival, growth, and reproduction of omnivorous and carnivorous mammals in the Housatonic River PSA. The measurement endpoints used to evaluate the assessment endpoint include: (1) determining, by comparisons to doses reported in the literature to cause adverse effects, the extent to which the concentrations of tPCBs and TEQ ingested in the diet will cause adverse effects to the survival, growth, or reproduction of omnivorous and carnivorous mammals, and (2) determining, by conducting field surveys, the relationship between the concentrations of tPCBs and TEQ and survival, reproduction, and relative abundance of omnivorous and carnivorous mammals in the Housatonic River floodplain.

Life History of Red Fox

The red fox is a common dog-sized canine that occurs in many habitats and is the most widely distributed carnivore in the world. In North America, the red fox is found throughout the United States and Canada, but not in the southeast coastal region, extreme southwest, parts of the central states, or the Pacific coastal regions. The typical pelage color of fox is red and it can be identified by its characteristic bushy, white-tipped tail, pointed muzzle, and prominent ears.

Habitat - Occupies a variety of habitats, but preferred habitat is a mix of forest, cropland, and pastureland, habitats common in the PSA. The availability of suitable prey as well as suitable den sites is also important. Prefer to locate dens in forested areas, but within a short distance of open areas and usually within 100 m of a source of open water.

Home Range - Maintains territory throughout the year and is considered nonmigratory. Average home range for adults in Maine was 14.7 km^2 (range = 6.0-27.5 km^2), average home range in Ontario was 9 km^2 but varied from 5 to 20 km^2. Mean territory sizes reported in EPA (1993) ranged from 100 to 2,000 hectares (1 to 20 km^2). Adults traverse most of their territory on a routine basis, but focus activities around dens, preferred hunting areas, food supplies, and resting areas.

Dietary Habits - Diet varies throughout the year depending on food availability. Includes almost all available animals as prey such as insects, fish, reptiles, amphibians, birds, small mammals, and carrion. Although typically identified as a carnivore, can consume considerable amounts of plant materials, particularly in the summer and fall. Plant material in the diet includes berries, apples, and nuts.

Life History of Northern Short-Tailed Shrew

The northern short-tailed shrew is a small energetic mouse-like animal with dark slate-colored pelage found throughout the northcentral and northeastern United States extending into southern Canada. It is easily identified as a shrew by its long pointed snout, small black eyes, concealed ears, and five toes on each foot.

Habitat - Occupies a variety of habitats, including wetlands and uplands, and is common in areas with abundant vegetative cover in both forested and open habitats.

Home Range - Home range of 0.06 acre (0.024 ha) in central New York State. Other estimates of home range size vary from 0.25 to 0.5 acres (0.1 to 0.2 ha) in areas of low prey density in winter months during nonbreeding periods to 0.07 to 0.17 acres (0.03 to 0.07 ha) in areas of high prey density with a minimum of territory overlap. Does not migrate seasonally, remaining in home range.

Dietary Habits - Earthworms and insects comprise most of the diet, with earthworms reported to be the most important item in the diet. Other invertebrates in the diet are mainly obtained from the leaf litter layer, and consist of millipedes, insect larvae, spiders, slugs, snails, and other mollusks. Plant materials, including nuts, berries, roots, and fungi, and occasional small mammals are also a component of the diet.

10.3 EXPOSURE ASSESSMENT

Trophic transfer and exposure through ingestion of contaminated prey are the major exposure pathways for red fox and short-tailed shrew exposed to tPCBs and TEQ. Other routes of exposure, considered to be negligible contributors to overall exposure, include inhalation, water consumption, and soil/sediment ingestion (Moore et al. 1999).

Fox were observed throughout the PSA from 1998 to 2001 (Appendix A). The exposure analysis was carried out for all of Reach 5 of the PSA because the foraging range of red fox is large. Because short-tailed shrews have a much smaller foraging range, the exposure analysis was performed for three locations in the PSA (Locations 13, 14, and 15, Figure J.1-8) that represent the range of COC concentrations found in the PSA.

Figure J.2-5 presents the cumulative frequency distribution for the inverse distance weighted (IDW) tPCB concentrations in soil samples collected in the PSA (See Appendix C.3 for IDW approach). The arithmetic mean of the soil concentrations for Locations 13, 14, and 15 are identified on the figure as well, providing an indication of the representativeness of the locations in comparison to the soil concentrations in the remainder of the PSA. These three locations were selected to span the range of tPCB soil concentrations in the PSA. Twenty-two percent of the IDW soil concentrations in the PSA were below the mean tPCB soil concentration at Location 15. Four percent of the soil tPCB concentrations in the PSA were above the mean concentration at Location 13.

Description of Sampling Locations 13, 14, and 15

Location 13 is a relatively flat area on the west shore of the river, adjacent to river mile 133, at an elevation of 965 ft (294 m). The community type is transitional floodplain forest that is flooded seasonally and is moderately well drained, with extensive vegetation cover (80%) and alluvial silt-loam soil. PCB concentrations in soil averaged 55.2 mg/kg.

Location 14 is a relatively flat low-lying area on the west shore of the river, adjacent to river mile 130, at an elevation of 965 ft (294 m). The community type is transitional floodplain forest that is flooded seasonally, with extensive vegetation cover (70%) and fluvial silt soil. PCB concentrations in soil averaged 26.1 mg/kg.

Location 15 is a flat area on the west shore of the river, adjacent to river mile 127, at an elevation of 965 ft (294 m). Community types are circumneutral hardwood swamp and transitional floodplain forest that are flooded seasonally. This site has 60% vegetation cover, 40% leaf litter cover, and a primarily mineral soil. PCB concentrations averaged 0.484 mg/kg.

10.3.1 Exposure Model

Exposure of the representative species, red fox and northern short-tailed shrew, to tPCBs and TEQ was estimated using a total daily intake model adapted from the *Wildlife Exposure Factors Handbook* (EPA 1993) and related publications. The model used in the exposure analysis was:

$$TDI = FT \cdot FIR \sum_{i=1}^{n} C_i \cdot P_i \qquad \text{(Eq. 1)}$$

where

TDI	=	total daily intake (mg/kg bw/d tPCBs, ng/kg bw/d TEQ)
FIR	=	normalized food intake rate (kg/kg bw/d)
FT	=	foraging time in PSA (unitless)
C_i	=	concentration in *i*th food item (mg/kg tPCBs, ng/kg TEQ)
P_i	=	proportion of the *i*th food item in the diet (unitless)

For those input variables that are uncertain, variable, or both, distributions are used rather than point estimates. Monte Carlo and probability bounds analyses were used to propagate uncertainties about input variables through the exposure model for each COC. A description of these techniques and the methods used to parameterize input variables is presented in Section 6.5 and Appendix C.4. Input distributions to the exposure analyses and the results of the Monte Carlo and probability analyses are discussed in detail in Appendix J.

10.3.1.1 Body Weight (BW)

As with many mammalian species, the red fox exhibits sexual dimorphism in body size. Males are typically 10% larger and 20% to 30% heavier than females (Storm et al. 1976; Lariviere and Pasitschniak-Arts 1996; Voigt 1987). The average body weight of adult female red fox used in the exposure assessment was 3.87 kg (std. dev. = 0.322g).

The northern short-tailed shrew can weigh over 22 grams (George et al. 1986; Burt and Grossenheider 1980, as cited in EPA 1993). As part of the ecological characterization of the PSA (Appendix A), 58 adult short-tailed shrews of both sexes were caught during small mammal trapping in 1998 to 2001. The body weights ranged from 15 to 27 g (mean = 21.9 g). The average weight of adult female shrews used in the exposure assessment was 22.3 g (std. dev. = 2.87 g).

10.3.1.2 Food Intake Rate (FIR)

An allometric modeling approach, described below, was used to estimate food intake rate for red fox and short-tailed shrew.

Food intake rate (FIR) was derived using the following equation:

$$\text{FIR(kg/kg bw/d)} = \frac{\text{FMR}}{\sum_{i=1}^{n} \text{AE}_i \cdot \text{GE}_i} \qquad \text{(Eq. 3)}$$

where *FMR* is the free metabolic rate, AE_i is the assimilation efficiency of *i*th food item (unitless) and GE_i is the gross energy of *i*th food item (kcal/kg).

The gross energies of various wildlife food sources are summarized in the *Wildlife Exposure Factors Handbook* (EPA 1993). The assimilation efficiency for mammals consumed by mammals is 84% (Castro et al. 1989). The assimilation efficiency of earthworms consumed by mammals is not known. The mean assimilation efficiency for insects consumed by small mammals is 87% (Bryant and Bryant 1988). This value was used to represent the assimilation efficiency for earthworms consumed by mammals. Point estimates were used for these variables in the Monte Carlo and probability bounds analyses because of their relatively small coefficients of variation (i.e., CV<10%).

10.3.1.3 Proportions of Dietary Items (P_i)

Studies reporting the dietary composition of the red fox in North America show that the proportion of dietary items varies according to season (Table J.2-3). Most studies found that mammals constitute the majority of the diet of the red fox, with the percentage in the diet as high as 92% in the spring (Knable 1974). For this assessment, mammals represent approximately 76% of the average diet for all seasons (Figure 10.3-1). Other food items including birds, invertebrates, and vegetation were not included as part of the exposure model because the dietary items represent a relatively small portion of the diet (e.g., birds and invertebrates) or the contribution to overall exposure is negligible (e.g., vegetation).

As with the red fox, there is variation in the proportion of dietary items reported for the short-tailed shrew (Table J.2-7). Earthworms comprised between 5% and 31% of the diet of short-tailed shrew, whereas insects and small mammals were reported as high as 61% and 24% of the diet, respectively (Hamilton 1941, as cited in EPA 1993; Linzey and Linzey 1973; Eadie 1944).

10.3.1.4 Foraging Time (FT)

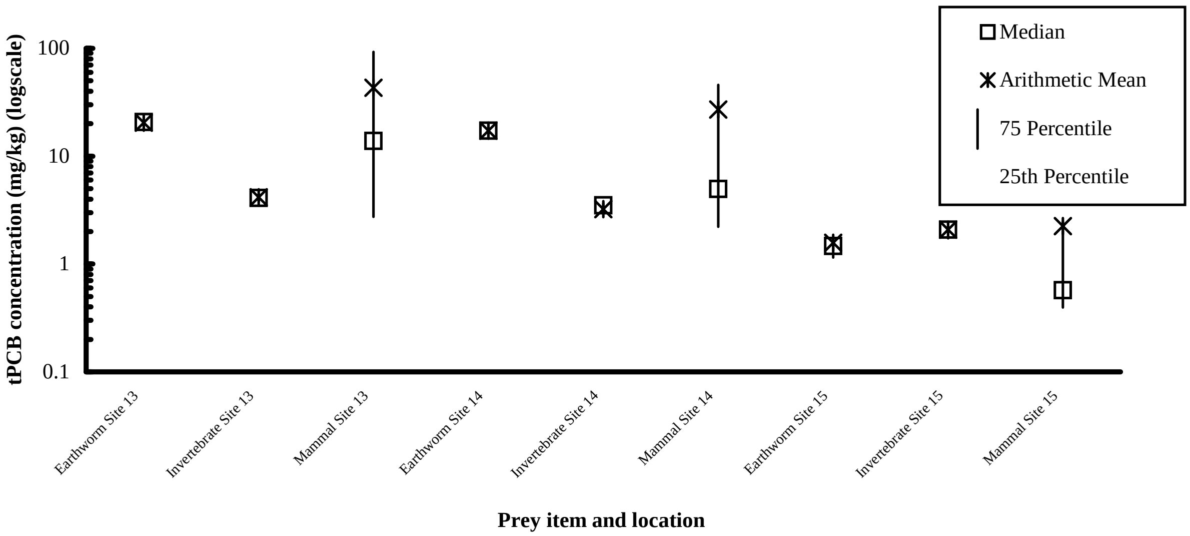
Because of its large home range in comparison to the size and configuration of the floodplain in Reach 5, the red fox is expected to spatially and temporally average exposure inside and outside the PSA, potentially experiencing areas of high contamination along with areas of low or no contamination. As a result, it was estimated that red fox spend up to 50% of their time foraging in the PSA (see Appendix J.2.1.5.2).

The foraging range for northern short-tailed shrew is smaller than the areas represented by Locations 13, 14, and 15, respectively. Therefore, shrews are expected to have 100% of their foraging range within each of Locations 13, 14, and 15 in the PSA.

10.3.1.5 Concentrations of COCs in Prey

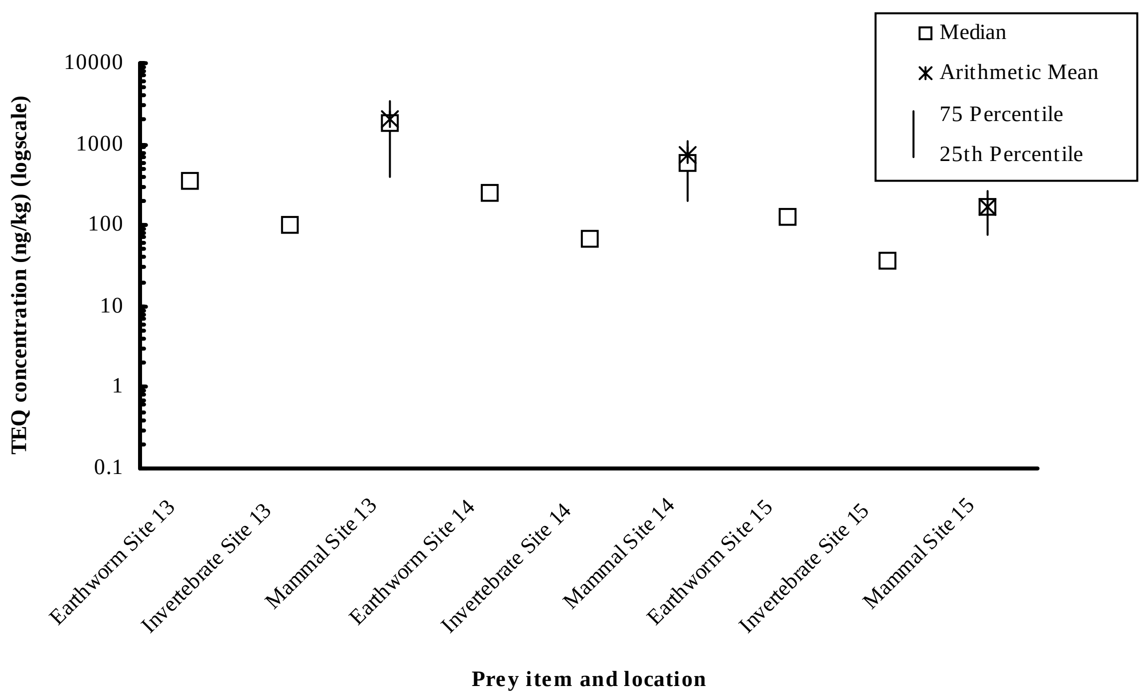
Mammals such as white-footed mouse and short-tailed shrew are the major dietary items for red fox. The median concentration of tPCBs in mammals measured in Reach 5 is 4.98 mg/kg ww.

The diet for northern short-tailed shrew includes earthworms, litter invertebrates, and mammals. Concentrations of tPCBs and TEQ, respectively, in these prey at Locations 13, 14, and 15 are presented in Figures 10.3-1 and 10.3-2. TEQ concentration in earthworms was measured in one composite sample of 20 to 45 earthworms at each location. Data on concentrations of TEQ in litter invertebrate prey were not available; therefore, the concentrations of TEQ in prey were extrapolated from concentrations in earthworms.



Notes: Error bars indicate interquartile range.
Prey concentrations are wet weight.

Figure 10.3-1 Concentrations of tPCBs in Prey of Northern Short-Tailed Shrew (n=1 for invertebrates and earthworms)



Notes: Error bars indicate interquartile range.
Prey concentrations are wet weight.

Figure 10.3-2 Concentrations of TEQ in Prey of Northern Short-Tailed Shrew (n=1 for invertebrates and earthworms)

The input variables for concentrations of COCs in prey of red fox and short-tailed shrew are shown in Tables J.1-1 to J.1-4.

10.3.2 Results of Exposure Assessments

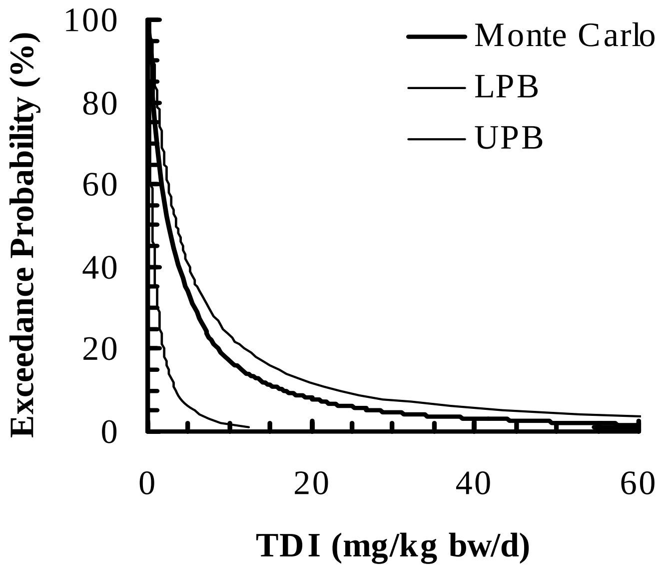
Figures 10.3-3 to 10.3-10 present exposure distributions for red fox and short-tailed shrew to tPCBs and TEQ.

Figure 10.3-3 depicts the cumulative distribution of tPCB intake rates for red fox in Reach 5. The Monte Carlo analysis indicated that exposure of red fox to tPCBs could range from a minimum of 0.0220 to a maximum of 82.5 mg/kg bw/d. The mean exposure was 6.25 mg/kg bw/d and the median exposure 2.68 mg/kg bw/d.

The probability bounds estimated for red fox foraging in Reach 5 are also depicted in Figure 10.3-3. The 50th percentile of the probability envelope ranged from 0.607 to 3.54 mg/kg bw/d. In comparison, the 50th percentile of the Monte Carlo simulation was 2.68 (Table J.2-4).

Short-tailed shrew living at Locations 13 and 14 had the highest exposure to tPCBs. Red fox foraging in Reach 5 had slightly less exposure to tPCBs than shrews at Locations 13 and 14. Red fox in Reach 5 and short-tailed shrew at Locations 13 and 14 had the highest exposures to TEQ. For both tPCBs and TEQ, short-tailed shrew foraging at Location 15 had the lowest exposure.

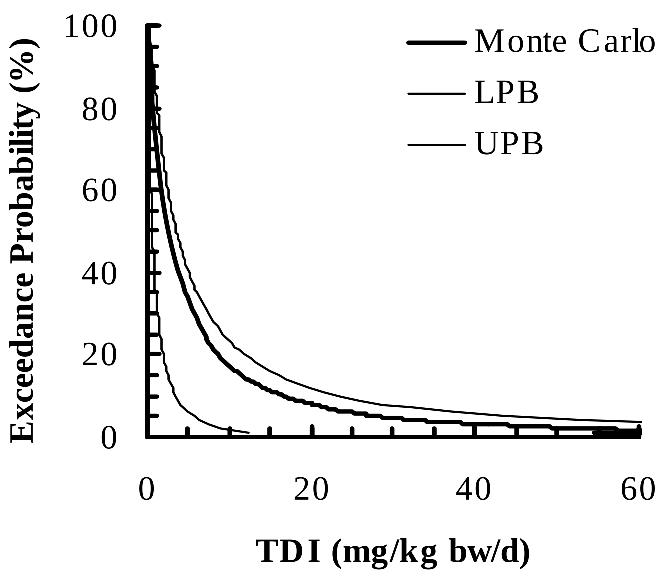
Reach 5



Notes: LPB = Lower Probability Bound
. UPB = Upper Probability Bound

Figure 10.3-3 Exceedance Probability Distribution for Red Fox Exposed to tPCBs in Reach 5 of the PSA

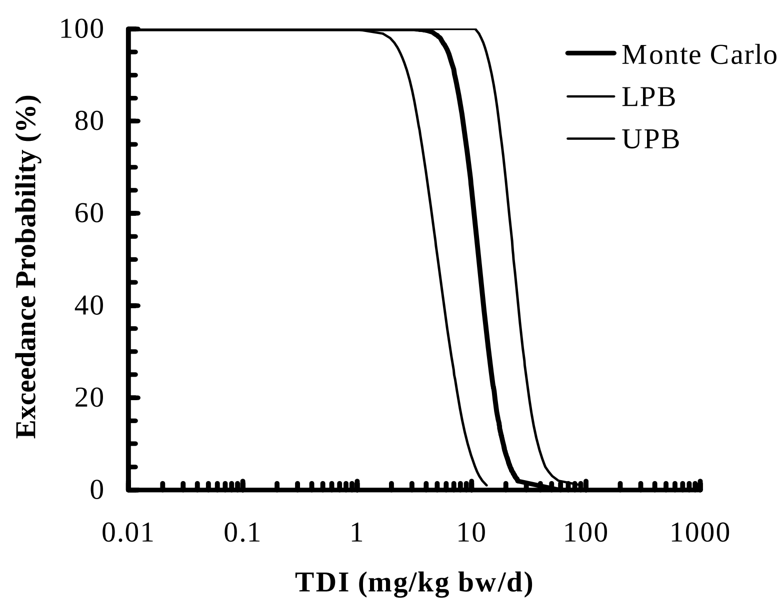
Reach 5



Notes: LPB = Lower Probability Bound
. UPB = Upper Probability Bound

Figure 10.3-4 Exceedance Probability Distribution for Red Fox Exposed to TEQ in Reach 5 of the PSA

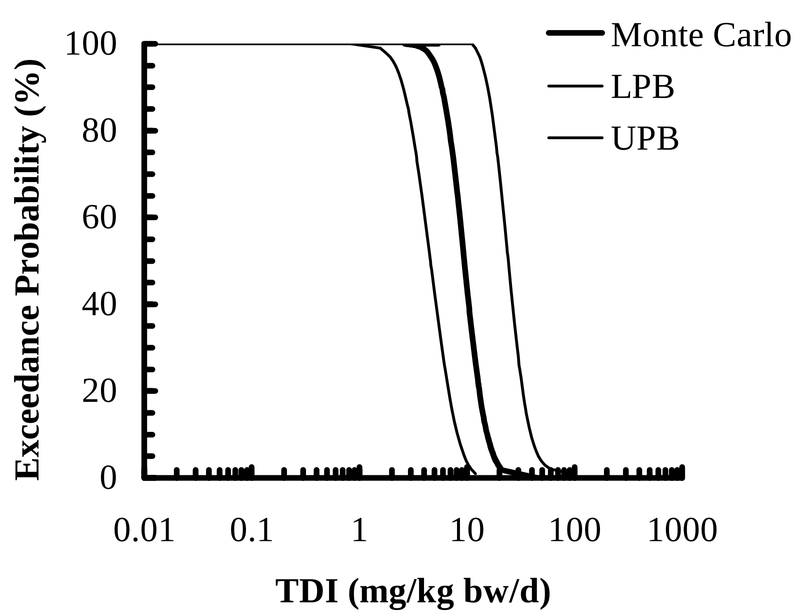
Location 13



Notes: LPB = Lower Probability Bound
. UPB = Upper Probability Bound

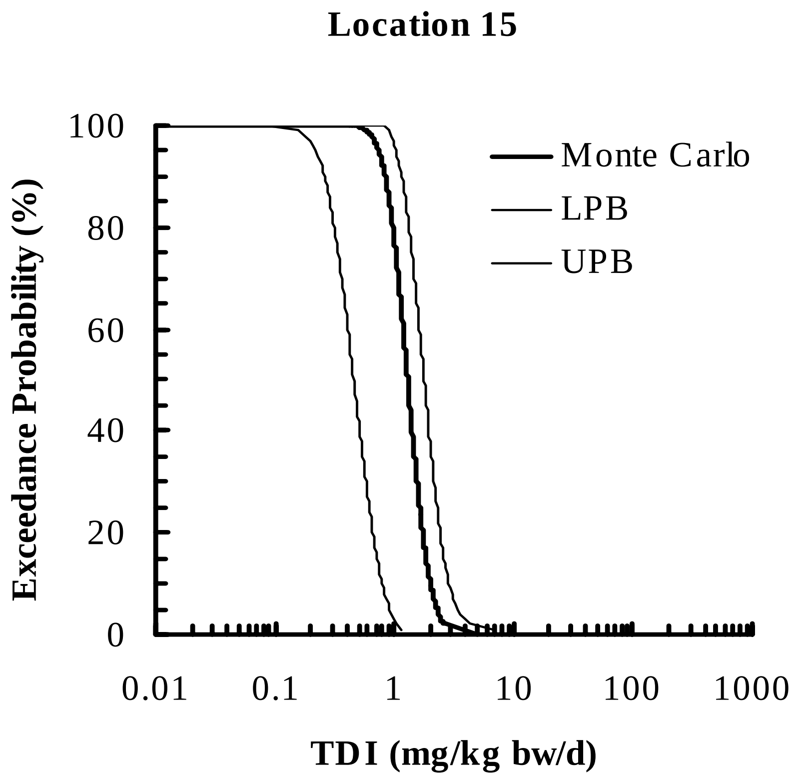
Figure 10.3-5 Exceedance Probability Distribution for Short-Tailed Shrew Exposed to tPCBs at Location 13 of the PSA

Location 14



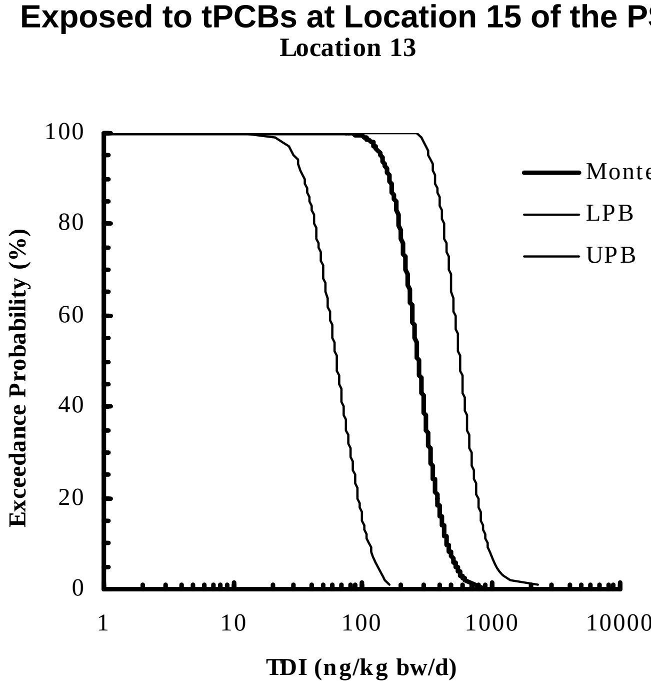
Notes: LPB = Lower Probability Bound
. UPB = Upper Probability Bound

Figure 10.3-6 Exceedance Probability Distribution for Short-Tailed Shrew Exposed to tPCBs at Location 14 of the PSA



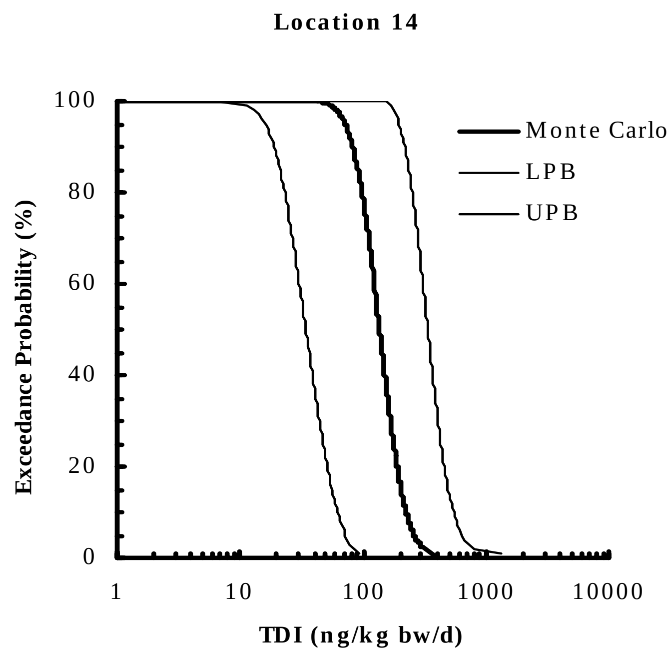
Notes: LPB = Lower Probability Bound
. UPB = Upper Probability Bound

Figure 10.3-7 Exceedance Probability Distribution for Short-Tailed Shrew Exposed to tPCBs at Location 15 of the PSA



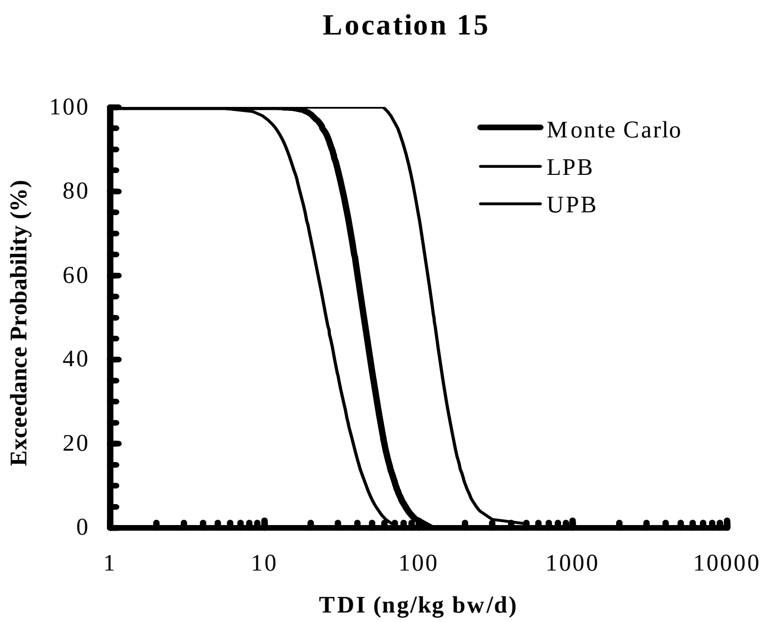
Notes: LPB = Lower Probability Bound
. UPB = Upper Probability Bound

Figure 10.3-8 Exceedance Probability Distribution for Short-Tailed Shrew Exposed to TEQ at Location 13 of the PSA



Notes: LPB = Lower Probability Bound
. UPB = Upper Probability Bound

Figure 10.3-9 Exceedance Probability Distribution for Short-Tailed Shrew Exposed to TEQ at Location 14 of the PSA



Notes: LPB = Lower Probability Bound
. UPB = Upper Probability Bound

Figure 10.3-10 Exceedance Probability Distribution for Short-Tailed Shrew Exposed to TEQ at Location 15 of the PSA

10.4 EFFECTS ASSESSMENT

The effects assessment has two objectives. The first is to review the scientific literature for effects of tPCBs (mainly Aroclors 1254 and 1260) and TEQ to omnivorous and carnivorous mammals, with documented effects to the representative species in this assessment, red fox and short-tailed shrew, of primary interest. The other objective is to derive the effects metrics that will be used, in conjunction with the exposure assessment results, to estimate risks to red fox and short-tailed shrew.

A discussion of the chemical features that elicit the toxic response and the mode of action is presented below.

Effects of tPCBs and TEQ on Mammals

Types of effects to mammals from exposure to tPCBs and TEQ include:

- Hormone induction
- Decreases in body and organ weight
- Reduced fertility
- Reduced litter size
- Reduced survival at birth or weaning
- Mortality

10.4.1 Review of Toxicity from the Literature

Appendix J includes a detailed review of the scientific literature on the effects of dietary tPCBs and TEQ to omnivorous and carnivorous mammals. The discussion focuses on ecologically relevant effects endpoints such as survival, growth, and reproduction. A summary of reproductive effects for tPCBs and TEQ is presented in Figures J.3-1 and J.3-2 and Tables J.3-2 and J.3-3.

10.4.2 Effects Metrics for Characterizing Risk

Effects data can be summarized in a variety of ways ranging from benchmarks designed to be protective of most or all species to dose-response curves. A summary of the decision criteria used to derive effects metrics is provided in the text box. Further details on the decision criteria used in selecting effects metrics are provided in Section 6.6.

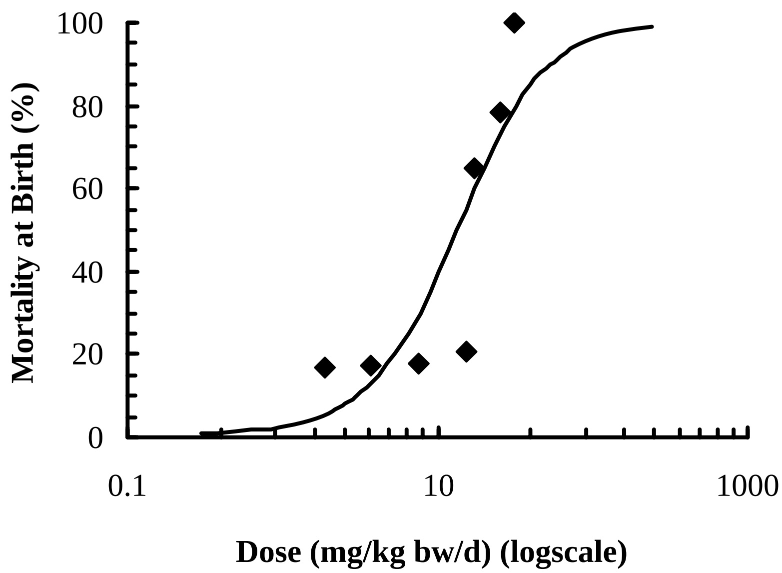
Hierarchical Decision Criteria for Derivation of Effects Metric

1. Have single-study bioassays with five or more treatments been conducted on the receptor of interest or a reasonable surrogate? If yes, estimate the concentration- or dose-response. If not, go to 2.
2. Are multiple bioassays with similar protocols, exposure scenarios and effects metrics available that, when combined, have five or more treatments for the receptor of interest or a reasonable surrogate? If yes, estimate the dose-response relationship as in 1. If not, go to 3.
3. Have bioassays with less than five treatments been conducted on the receptor of interest or a reasonable surrogate? If yes, conduct or report results of hypothesis testing to determine the NOAEL and LOAEL. If not, go to 4.
4. Are sufficient data available from field studies and monitoring programs to estimate concentrations or doses of the COC that are consistently protective or associated with adverse effects? If yes, develop field-based effects metrics. If not, go to 5.
5. Derive a range where the threshold for the receptor of interest is expected to occur. Because information on the sensitivity of the receptor of interest is lacking, it is difficult to derive a threshold that is neither biased high nor low. If, however, bioassay data are available for several other species, calculate a threshold for each to determine a threshold range that spans sensitive and tolerant species. That range is likely to include the threshold for the receptor of interest.

In this ERA, data were available to derive dose-response curves using surrogate mammals for the representative species.

10.4.2.1 Effects of tPCBs to Red Fox and Short-Tailed Shrew

The Spencer (1982) study was used for the derivation of a dose-response curve based on mortality at birth. Figure 10.4-1 presents the dose-response curve for mortality of rats at birth. The dose-response curve indicates that 10% and 20% declines in mortality at birth would be expected at doses of 3.05 and 5.37 mg/kg bw/d, respectively.

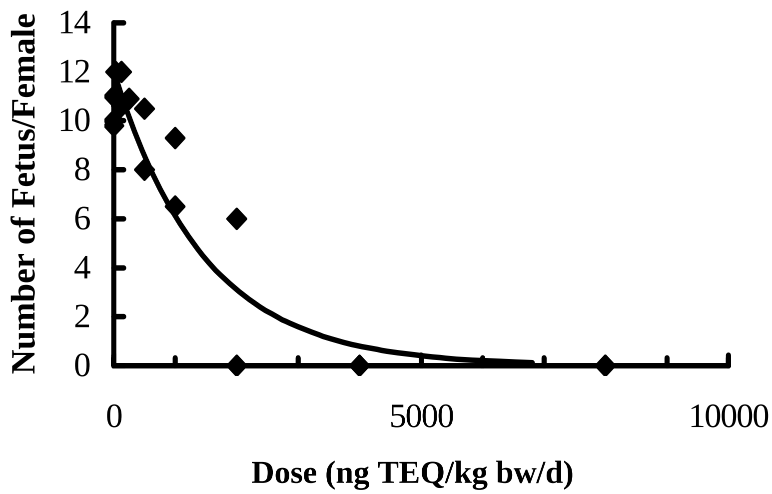


Note: Symbols indicate raw data.

Figure 10.4-1 Dose-Response Curve for Effects of tPCBs on Mortality at Birth of Rats

10.4.2.2 Effects of TEQ to Red Fox

The Khera and Ruddick (1973) and Sparschu et al. (1971) studies were used for the derivation of a dose-response curve based on reproductive effects. Figure 10.4-2 presents the dose-response curve for reproductive fecundity of rats exposed to TEQ. The dose-response curve indicates that 10% and 20% declines in reproductive fecundity would be expected at doses of 156 and 330 ng/kg bw/d TEQ, respectively.

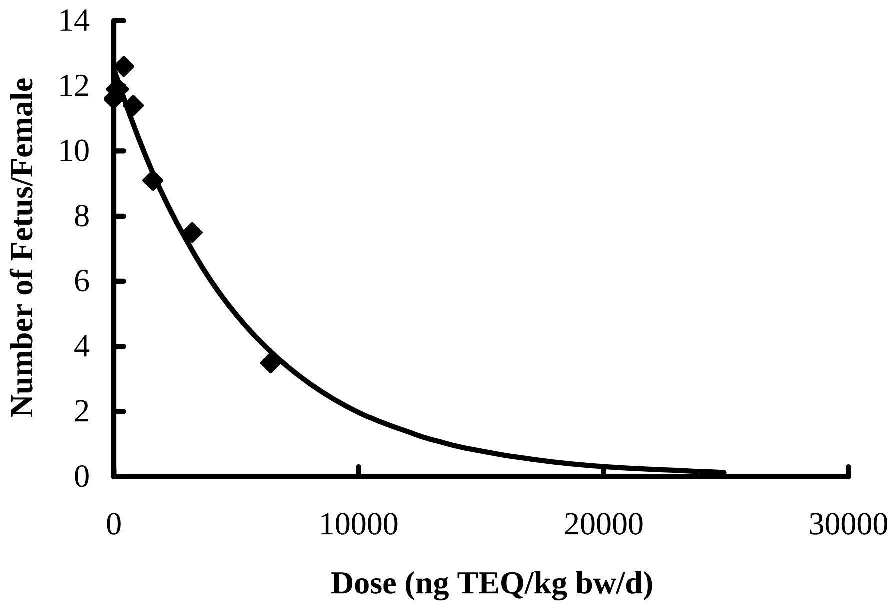


Note: Symbols indicate raw data.

Figure 10.4-2 Dose-Response Curve for Effects of TEQ on Reproductive Fecundity of Rat

10.4.2.3 Effects of TEQ to Short-Tailed Shrew

The Marks et al. (1989) study was used for the derivation of a dose-response curve based on reproductive effects. Figure 10.4-3 presents the dose-response curve for reproductive fecundity of mice exposed to TEQ. The dose-response curve indicates that 10% and 20% declines in reproductive fecundity would be expected at doses of 570 and 1207 ng/kg bw/d TEQ, respectively.



Note: Symbols indicate raw data.

Figure 10.4-3 Dose Response Curve for Effects of TEQ on Reproductive Fecundity of Mouse

10.5 RISK CHARACTERIZATION

This section characterizes risk to red fox and short-tailed shrews exposed to tPCBs and TEQ in the PSA. The risk characterization uses two and three lines of evidence to determine potential ecological risks to red fox and short-tailed shrew, respectively. The major lines of evidence are considered to be independent and will be combined in a weight-of-evidence assessment. The key risk questions and the lines of evidence are summarized in the text box.

Key Risk Questions

- Are the concentrations of tPCBs and TEQ present in the prey of omnivorous and carnivorous mammals sufficient to cause adverse effects to individuals inhabiting the PSA?
- If so, how severe are the risks and what are their potential consequences?

Lines of Evidence

- Use of semi-quantitative biological field surveys.
- Probabilistic exposure and effects modeling.
- Population demography field study for short-tailed shrew.

10.5.1 Field Surveys (Performed by EPA)

The mammalian community in the PSA was studied by EPA over a 4-year period, from 1998 to 2001 (see Appendix A). Surveys were conducted to record presence, relative abundance, and habitat usage for small and large mammals including short-tailed shrew and red fox. A variety of field survey techniques including small mammal trapping, snow tracking, and scent-post station surveys were used to characterize the mammalian community.

Observations of omnivorous and carnivorous mammals including coyotes, red fox, raccoons, white-footed mice, short-tailed shrews, and little brown bats were common, all of which were observed in forested and nonforested habitats as well as riverine, shoreline, wetland, upland, and residential habitats. Other carnivorous mammals observed in the PSA included bobcats, fishers, and long-tailed weasels. Omnivorous mammals were one of the most abundant groups of mammals observed in the PSA. Common omnivores included white-footed mice, raccoons, striped skunks, Virginia opossums, and black bears. The short-tailed shrew was the most abundant insectivorous mammal observed in the PSA. The semi-quantitative nature of the field surveys and lack of reference locations in the surveys preclude the development of relationships between abundance of representative species and concentrations of COCs.

10.5.2 Comparison of Estimated Exposures to Laboratory-Derived Effects Doses

Red fox exposure was assessed for all of Reach 5 and short-tailed shrew exposure was estimated in three areas (Locations 13, 14, and 15, Figure J.1-8) in the PSA.

For each receptor-COC combination, a category of low, intermediate, or high risk was assigned following integration of the exposure and effects distributions. This exercise was done separately for the results of the Monte Carlo analyses and each of the lower and upper bounds from the probability bounds analyses. The "risk category" refers to the level of risk based on the results of the Monte Carlo analysis. The "risk range" refers to the levels of risk based on the results of the probability bounds analyses. The 10% and 20% effects doses for each species and COC are presented in Section 10.4.

Guidance for Determining Level of Risk to Representative Species

- If the probability of 10% or greater effect is less than 20%, then the risk to omnivorous and carnivorous mammals was considered low.
- If the probability of 20% or greater effect is greater than 50%, then the risk to omnivorous and carnivorous mammals was considered high.
- All other outcomes were considered to have intermediate risk.

The results of the risk characterization are summarized in Table 10.5-1. Figures 10.5-1 to 10.5-8 present the risk curves for red fox exposed in Reach 5 and short-tailed shrew exposed at Locations 13, 14, and 15 to tPCBs and TEQ.

Table 10.5-1

Summary of Qualitative Risk Statements for Omnivorous and Carnivorous Mammals from the Housatonic River Study Area

Mammal / Location	Qualitative Risk Statements				
	tPCBs			TEQ	
	Risk Category	Risk Range		Risk Category	Risk Range
Red Fox					
Reach 5	Intermediate	Low-Intermediate		Intermediate	Low- Intermediate
Short-Tailed Shrew					
Location 13	High	Intermediate -High		Low	Low-Intermediate
Location 14	High	Intermediate-High		Low	Low
Location 15	Low	Low-Intermediate		Low	Low

The results of the risk characterization showed that the highest risk to omnivorous and carnivorous mammals is from exposure to tPCBs at Locations 13 and 14. The risk category for short-tailed shrew at Locations 13 and 14 was high, and the risk range, as determined by the probability bounds analysis, ranged from intermediate to high. Risk to shrews at Location 15 was low. The risk category for exposure of short-tailed shrew to TEQ at Location 13 is low; the risk range is low to intermediate. Short-tailed shrew exposed to TEQ at Locations 14 and 15 have a risk category of low. Both the upper and lower bound of the risk ranges for Locations 14 and 15 are low. Red fox had an intermediate risk category for both exposure to tPCBs and TEQ. The risk range for both COCs for red fox is low to intermediate.

Reach 5

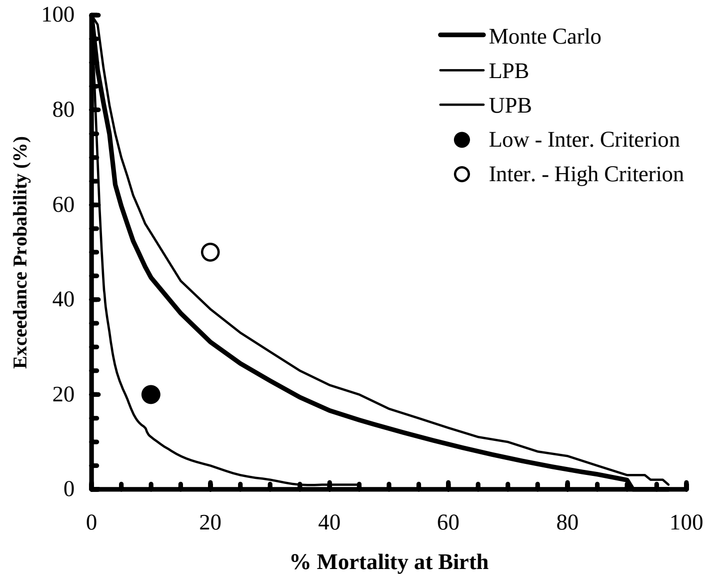


Figure 10.5-1 Risk Function for Red Fox Exposed to tPCBs in Reach 5 of the Housatonic River

Reach 5

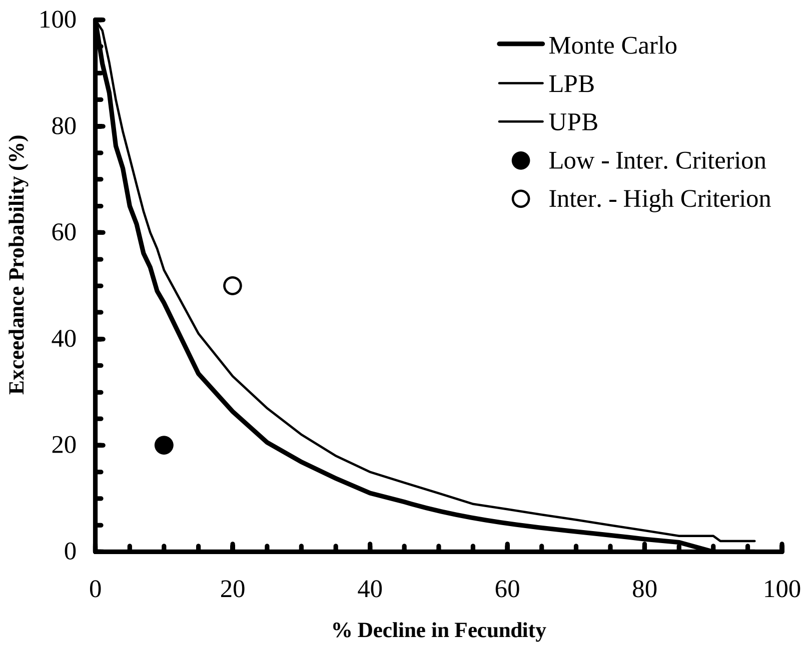


Figure 10.5-2 Risk Function for Red Fox Exposed to TEQ in Reach 5 of the Housatonic River

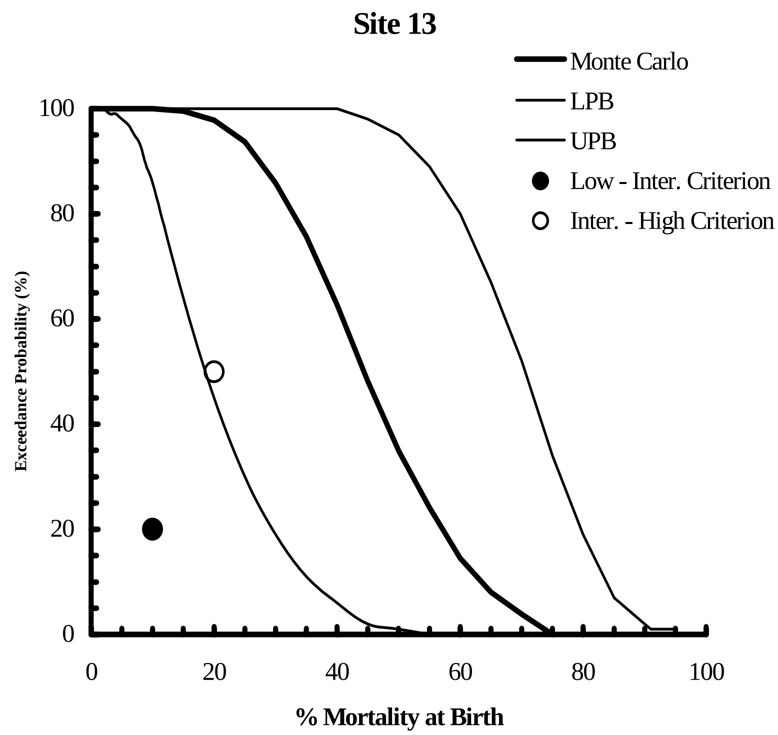


Figure 10.5-3 Risk Function for Short-Tailed Shrew Exposed to tPCBs at Location 13 of the PSA

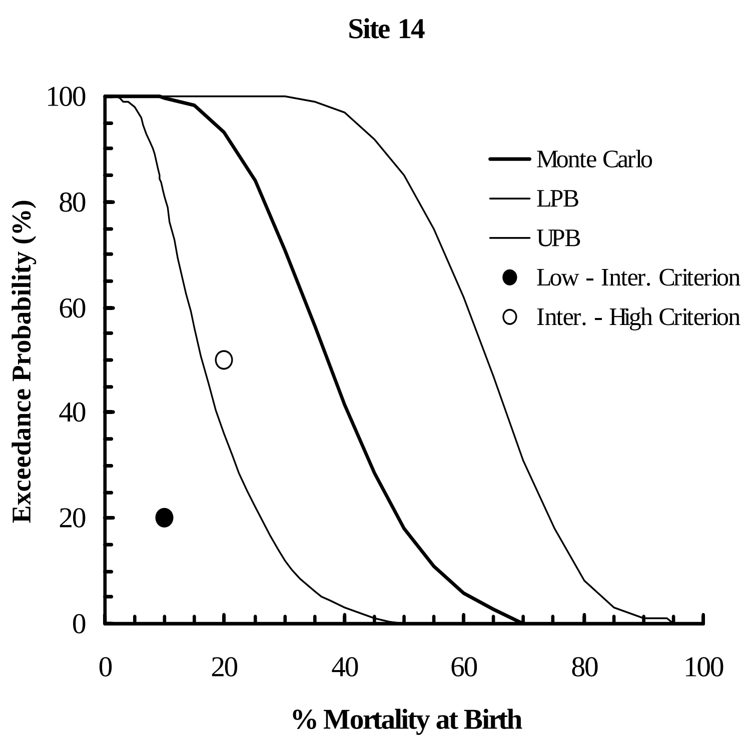


Figure 10.5-4 Risk Function for Short-Tailed Shrew Exposed to tPCBs at Location 14 of the PSA

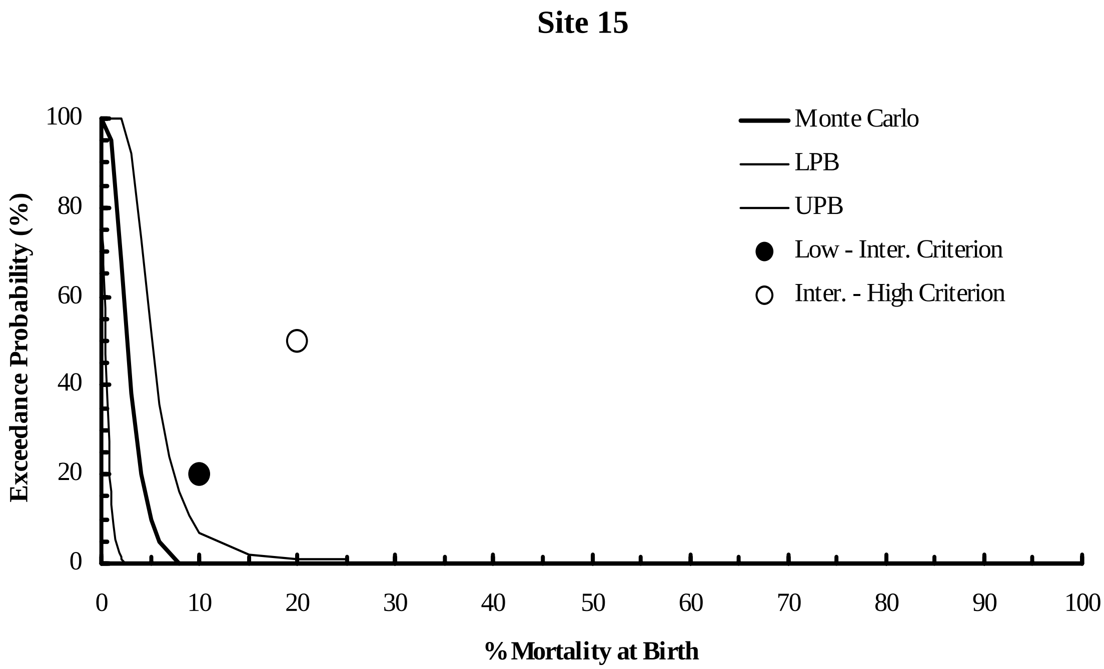


Figure 10.5-5 Risk Function for Short-Tailed Shrew Exposed to tPCBs at Location 15 of the PSA

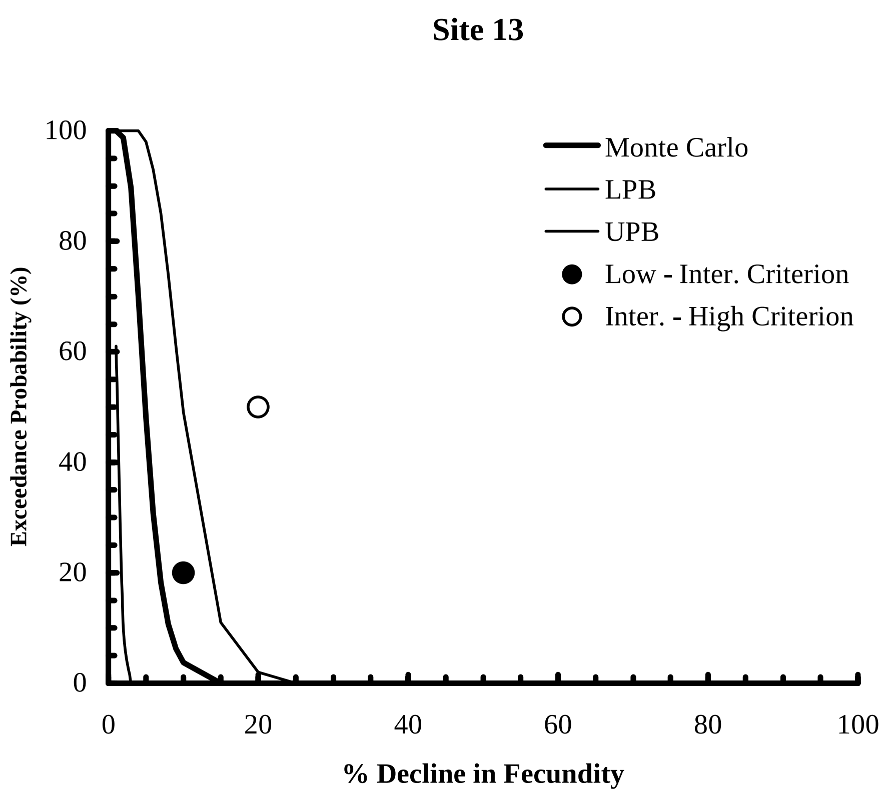


Figure 10.5-6 Risk Function for Short-Tailed Shrew Exposed to TEQ at Location 13 of the PSA

Site 14

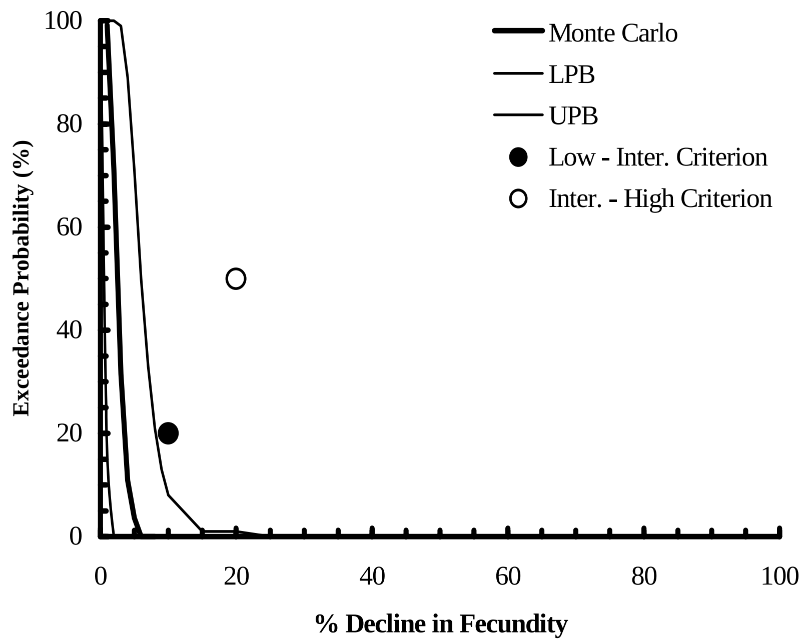


Figure 10.5-7 Risk Function for Short-Tailed Shrew Exposed to TEQ at Location 14 of the PSA

Site 15

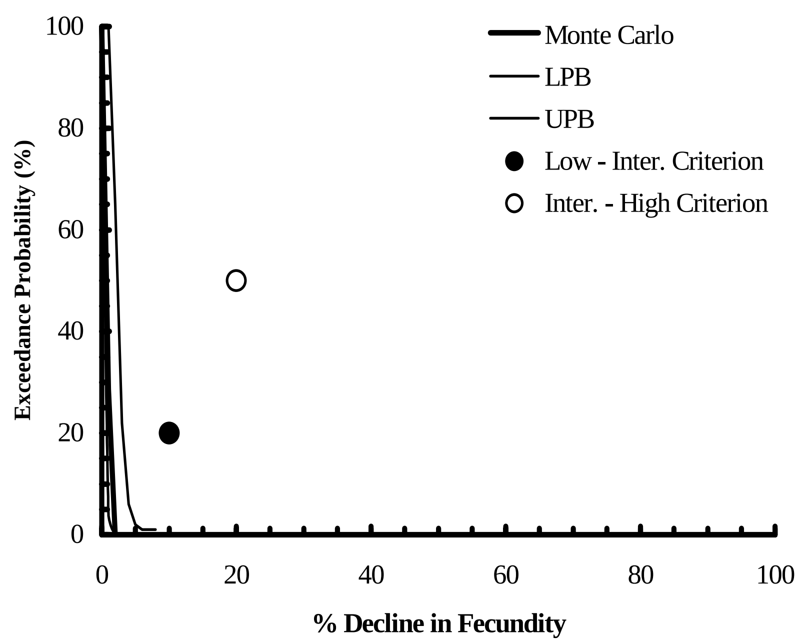


Figure 10.5-8 Risk Function for Short-Tailed Shrew Exposed to TEQ at Location 15 of the PSA

10.5.3 Population Demography Field Study (Performed by GE)

A population demography field study was performed in 2001 along a 16-km reach of the Housatonic River between Pittsfield and Woods Pond (Boonstra 2002). The study objectives included evaluating population density, survival, rates of reproduction, sex ratio, and growth rates of short-tailed shrew. More information on this study, including a detailed description of the study methods and data analysis procedures, is provided in Boonstra (2002).

Based on the results of this study, Boonstra (2002) suggested that population characteristics of short-tailed shrew living on more contaminated tPCB sites in the PSA are not negatively affected compared to those living on less contaminated sites. Variations in tPCB concentrations among the sites did not cause differences in population demography of short-tailed shrew.

There were several confounding factors in the Boonstra (2002) study, including flooding of the floodplain area (i.e., location within the floodplain) and habitat quality, each of which may have had significant effects on population demographics. The lack of reproduction rate data in the Boonstra (2002) study makes it impossible to know if shrew populations in the PSA are maintaining themselves through natural production or immigration. The use of body weight to imply reproductive status may not be appropriate because it is insensitive to potential reproductive impairments. These factors limit the strength of conclusions from the study.

EPA conducted a re-analysis of the Boonstra data to verify the relationship between shrew survival and concentrations of tPCBs. The results of the analyses indicated a significant relationship between concentrations of tPCBs in soil and survival of shrews from summer to autumn for males, females, and males and females combined. The results of the analyses indicated a significant relationship between soil concentrations of tPCBs and shrew survival. The regression model indicated that survival was reduced in the “high” contaminated grids compared to the “low” contaminated grids.

10.5.4 Weight-of-Evidence Analysis

For the WOE analysis, the three-phase approach of Menzie et al. (1996) and the Massachusetts Weight-of-Evidence Workgroup was used, in which weight-of-evidence was reflected in the following three characteristics: (a) the weight assigned to each measurement endpoint, (b) the magnitude of response observed in the measurement endpoint, and (c) the concurrence among outcomes of the multiple measurement endpoints.

A discussion of attributes considered in the WOE is provided in Section 2, and the rationales for weighting of measurement endpoints are provided in Appendix J. A summary of the derived weightings is provided in Table 10.5-2 and Table 10.5-3.

For red fox, the field surveys and the modeled exposure and effects lines of evidence for tPCBs and TEQ were both given values of moderate.

For short-tailed shrew, the field surveys were given a moderate value and the population demography study and the modeled exposure and effects lines of evidence for tPCBs and TEQ were both given values of moderate/high.

Table 10.5-2

Weighting of Measurement Endpoints for Red Fox Weight-of-Evidence Evaluation

Attributes	Field Surveys	Modeled Exposure and Effects for tPCBs and TEQ
I. Relationship Between Measurement and Assessment Endpoints		
1. Degree of Association	L/M	M
2. Stressor/Response	L/M	M/H
3. Utility of Measure	L/M	M/H
II. Data Quality		
4. Data Quality	H	M/H
III. Study Design		
5. Site Specificity	M/H	L/M
6. Sensitivity	M	H
7. Spatial Representativeness	M/H	M
8. Temporal Representativeness	M/H	M
9. Quantitative Measure	L/M	H
10. Standard Method	H	M/H
Overall Endpoint Value	**M**	**M/H**

L = low; L/M = low-moderate; M = moderate; M/H = moderate/high; H = high

Table 10.5-3

Weighting of Measurement Endpoints for Short-Tailed Shrew Weight-of-Evidence Evaluation

Attributes	Field Surveys	Population Demography Field Study*	Modeled Exposure and Effects for tPCBs and TEQ
I. Relationship Between Measurement and Assessment Endpoints			
1. Degree of Association	M	M/H	M
2. Stressor/Response	L/M	M/H	M/H
3. Utility of Measure	L/M	M/H	M/H
II. Data Quality			
4. Data Quality	H	M/H	M/H
III. Study Design			
5. Site Specificity	M/H	H	L/M
6. Sensitivity	M	M/H	H
7. Spatial Representativeness	M/H	M/H	M
8. Temporal Representativeness	M/H	M	M
9. Quantitative Measure	L/M	M	H
10. Standard Method	H	H	M/H
Overall Endpoint Value	**M**	**M/H**	**M/H**

L = low; L/M = low-moderate; M = moderate; M/H = moderate/high; H = high

The magnitude of the response in the measurement endpoint is considered together with the measurement endpoint weight in judging the overall weight-of-evidence (Menzie et al. 1996). This requires assessing the strength of evidence that ecological harm has occurred, as well as an indication of the magnitude of response, if present. The weighting values, evidence of harm, and magnitudes of responses were combined in a matrix format and are presented in Tables 10.5-4 to Table 10.5-7.

The field surveys indicated that red fox and short-tailed shrew are likely common in the PSA. However, it is not known whether these species would be more abundant in the absence of COCs, or if they are abundant because of immigration from less contaminated areas.

EPA's reanalysis of the data, the short-tailed shrew population study (Boonstra 2002) showed that tPCBs may be having effects on survival of short-tailed shrews. Other demographic parameters, however, do not appear to be affected by tPCB concentrations in soil.

The results from the modeled exposure and effects line of evidence suggest that there is an intermediate risk for fox exposed to tPCBs and TEQs in the PSA. There is a high risk to short-tailed shrew exposed to tPCBs at Locations 13 and 14, and a low risk at Location 15. There was no appropriate risk to short-tailed shrew exposed to TEQ is low at Locations 13, 14, and 15.

Table 10.5-4

Evidence of Harm and Magnitude of Effects for Red Fox Exposed to tPCBs in the Housatonic River PSA

Measurement Endpoints	Weighting Value (High, Moderate, Low)	Evidence of Harm (Yes, No, Undetermined)	Magnitude (High, Intermediate, Low)
Field Surveys	Moderate	Undetermined	--
Modeled Exposure and Effects	Moderate/High	Yes	Intermediate

Table 10.5-5

Evidence of Harm and Magnitude of Effects for Short-Tailed Shrew Exposed to tPCBs in the Housatonic River PSA

Measurement Endpoints	Weighting Value (High, Moderate, Low)	Evidence of Harm (Yes, No, Undetermined)	Magnitude (High, Intermediate, Low)
Field Surveys	Moderate	Undetermined	--
Population Demography Field Study	Moderate/High	Yes	Intermediate
Modeled Exposure and Effects	Moderate/High	Yes	High

Table 10.5-6

Evidence of Harm and Magnitude of Effects for Red Fox Exposed to TEQ in the Housatonic River PSA

Measurement Endpoints	Weighting Value (High, Moderate, Low)	Evidence of Harm (Yes, No, Undetermined)	Magnitude (High, Intermediate, Low)
Field Surveys	Moderate	Undetermined	--
Modeled Exposure and Effects	Moderate/High	Yes	Intermediate

Table 10.5-7

Evidence of Harm and Magnitude of Effects for Short-Tailed Shrew Exposed to TEQ in the Housatonic River PSA

Measurement Endpoints	Weighting Value (High, Moderate, Low)	Evidence of Harm (Yes, No, Undetermined)	Magnitude (High, Intermediate, Low)
Field Surveys	Moderate	Undetermined	--
Modeled Exposure and Effects	Moderate/High	No	--

A graphical method was used for displaying concurrence among measurement endpoints. Tables 10.5-8 and 10.5-11 depict the outcome for red fox and short-tailed shrew exposed to tPCBs and TEQ. The potential for harm to shrew and red fox based on field surveys was undetermined but the population demography line of evidence indicated an intermediate level of risk for shrew. Based on the modeled exposure and effects assessment for shrew, there is a high potential for adverse effects resulting from tPCB exposure and no appreciable potential for adverse effects resulting from exposure to TEQ.

Table 10.5-8

Risk Analysis Summary for Red Fox Exposed to tPCBs in the Housatonic River PSA

Assessment Endpoint:	Survival, growth, and reproduction of omnivorous and carnivorous mammals

Harm/Magnitude	**Weighting Factors (increasing confidence of weight)**				
	Low	**Low/Moderate**	**Moderate**	**Moderate/High**	**High**
Yes/High					
Yes/Intermediate				MEE	
Yes/Low					
Undetermined			FS		
No					

FS = Field surveys
MEE = Modeled exposure and effects

Table 10.5-9

Risk Analysis Summary for Short-Tailed Shrew Exposed to tPCBs in the Housatonic River PSA

Assessment Endpoint:	Survival, growth, and reproduction of omnivorous and carnivorous mammals

Harm/Magnitude	**Weighting Factors (increasing confidence of weight)**				
	Low	**Low/Moderate**	**Moderate**	**Moderate/High**	**High**
Yes/High				MEE	
Yes/Intermediate				PDFS	
Yes/Low					
Undetermined			FS		
No					

FS = Field surveys
MEE = Modeled exposure and effects
PDFS = Population demography field study for short-tailed shrew

Table 10.5-10

Risk Analysis Summary for Red Fox Exposed to TEQ in the Housatonic River PSA

Assessment Endpoint:	Survival, growth, and reproduction of omnivorous and carnivorous mammals

Harm/Magnitude	Weighting Factors (increasing confidence of weight)				
	Low	**Low/Moderate**	**Moderate**	**Moderate/High**	**High**
Yes/High					
Yes/Intermediate				MEE	
Yes/Low					
Undetermined			FS		
No					

FS = Field surveys

MEE = Modeled exposure and effects

Table 10.5-11

Risk Analysis Summary for Short-Tailed Shrew Exposed to TEQ in the Housatonic River PSA

Assessment Endpoint:	Survival, growth, and reproduction of omnivorous and carnivorous mammals

Harm/Magnitude	**Weighting Factors (increasing confidence of weight)**				
	Low	**Low/Moderate**	**Moderate**	**Moderate/High**	**High**
Yes/High					
Yes/Intermediate					
Yes/Low					
Undetermined			FS		
No				MEE	

FS = Field surveys
MEE = Modeled exposure and effects

10.5.5 Sources of Uncertainty

Some of the major sources of uncertainty associated with the assessment of risks of tPCBs and TEQ to omnivorous/carnivorous mammals are briefly summarized below. A more complete list is presented in Appendix J.

- Estimation of free metabolic rates using allometric equations.
- Limited sample sizes for the analyses of COC concentrations in some prey items.
- Co-elution of PCB congeners 123 and 157.
- Lack of toxicity studies involving the representative species.
- Lack of feeding studies involving prey and food items from the PSA for the representative species.
- Lack of reproduction rate data in the Boonstra (2002) short-tailed shrew study.
- Use of body weight to imply reproductive fitness in the Boonstra (2002) short-tailed shrew study.
- Analytical variability in the chemical analysis of prey tissues.

10.5.6 Extrapolation to Other Species

Other omnivorous and carnivorous species common to the area include smoky shrews, masked shrews, coyotes, gray fox, fishers, short-tailed weasels, and long-tailed weasels (see Appendix A). Exposure and sensitivity to COCs are the two factors used to estimate risk to omnivorous and carnivorous mammals. As noted in this ERA, effects studies conducted on short-tailed shrew and red fox are not available. Similarly, effects data are not available for other omnivorous and carnivorous species living in the Housatonic River area. As a result, the same surrogate effects data used to estimate effects to short-tailed shrew and red fox would be used for other omnivorous and carnivorous species. A qualitative analysis was conducted to compare exposure of red fox and short-tailed shrew and other omnivorous and carnivorous mammals to tPCBs and TEQ. The major factors that influence mammalian exposure to tPCBs and TEQ include the following:

- Foraging behavior and dietary composition.
- Foraging and home range size.
- Species body weight and other life history characteristics.

Representative species and other mammal species were compared using these factors. Results are provided in the ERA Summary text boxes.

10.5.7 Conclusions

The weight-of-evidence evaluation for red fox suggests, based on two lines of evidence, an intermediate risk to fox exposed to tPCBs and TEQ in the PSA. This finding is uncertain because although fox were commonly observed during the field surveys, a foraging rate of 50% in Reach 5 was used in the exposure and effects modeling, and species-specific measures of effects were not available.

For short-tailed shrew, data from three major lines of evidence were available, including field surveys, a population demography field study, and exposure and effects modeling. The weight-of-evidence analysis for the exposure and effects modeling indicates an intermediate risk for short-tailed shrews exposed to tPCBs and no appreciable risk to short-tailed shrew as a result of TEQ exposure in the PSA. This conclusion, however, is also uncertain because of the lack of definitive findings in the field surveys about whether effects are occurring and the lack of species-specific measures of effects.

The field surveys and the conclusions made in the Boonstra (2002) study contradict the results from the modeling of exposure and effects line of evidence. However, the results of the supplemental analyses of the data from the Boonstra (2002) study on survival of short-tailed shrews are in agreement with the modeling results, suggesting that there are intermediate risks occurring from exposure to COCs in the contaminated areas of the PSA.

A quantitative assessment downstream of the PSA was not conducted for red fox and short-tailed shrew because concentrations of tPCBs and TEQ in prey are not available. Based on the lower soil and sediment concentrations, risks to omnivorous and carnivorous mammals are likely to be much reduced below Woods Pond relative to the PSA. To test this hypothesis, a hockey stick regression model was fit to the survival data from Boonstra (2002) for males and females combined. The independent variable was arithmetic mean tPCB concentration in soil for each of the six grids. Based on the analysis, the MATC for shrews exposed to tPCB in soil is 21.1 mg/kg (dw) (Figure J.4-9). Twenty-one percent of soil concentrations in PSA habitats suitable for shrews exceed the MATC threshold. No soil concentrations exceeding the MATC were measured downstream of the PSA.

ERA Summary for Red Fox

The weight-of-evidence analysis indicates an intermediate risk for red fox exposed to tPCBs and TEQ in the PSA. This finding is uncertain because although fox were commonly observed during the field surveys, a foraging rate of 50% in Reach 5 was used, and species-specific measures of effects were not available.

Other carnivorous mammal species common to the PSA include coyote, gray fox, fisher, short-tailed weasel, and long-tailed weasel. A qualitative analysis of risk on these species indicates that short-tailed weasel and long-tailed weasel have higher levels of risk; gray fox and fisher have similar levels of risk; and coyote has a lower level of risk compared to the representative species.

ERA Summary for Short-Tailed Shrew

Three lines of evidence were evaluated when assessing risk to the short-tailed shrew. The weight-of-evidence analysis indicates high risk for short-tailed shrews based on modeled exposure to tPCBs and no appreciable risks for short-tailed shrew as a result of TEQ exposure in the PSA. The shrew population demography study indicated an intermediate level of risk to shrews based on tPCB exposure. Risk to the short-tailed shrew based on field surveys was undetermined. Because field surveys did not identify risks, and species-specific measures of effects were not available for the modeled exposure and effects assessment, there is uncertainty associated with the determination of high risk to short-tailed shrew.

Other omnivorous mammal species common to the PSA include smoky shrew and masked shrew. A qualitative analysis of risk on these species indicates that smoky shrew have higher levels of risk; and masked shrew have similar levels of risk.

10.6 REFERENCES

Boonstra, R. 2002. Demography of short-tailed shrew population living on PCB contaminated sites. Division of Life Sciences, University of Toronto at Scarborough, Ontario, Canada.

Bryant, D.M. and V. M. Bryant. 1988. Assimilation efficiency and growth of nestling insectivores. *Arch. Hydrobiol*. 130: 268-274.

Burt, W.H., and R.P. Grossenheider. 1980. *A field guide to the mammals of North America north of Mexico*. Houghton Mifflin Co., Boston, MA.

Castro, G., N. Stoyan, and J.P. Myers. 1989. Assimilation efficiencies in birds: A function of taxon or food type? *Comp. Biochem. Physiol*. 92:271-278.

Eadie, W.R. 1944. The short-tailed shrew and field mouse predation. *Journal of Mammalogy* 25:359-364.

EPA (U.S. Environmental Protection Agency). 1993. *Wildlife Exposure Factors Handbook*. Volumes I and II. Office of Research and Development. Washington, DC. EPA EPA/600/R-93/187a, EPA/600/R-93/187b. (99-0040)

George, S.B., J.R. Choate, and H.H. Genoways. 1986. *Blarina brevicauda*. p. 1-9. *In* T.E. Lawlor (ed.) Mammalian Species Number 261. American Society of Mammalogists, Baltimore, MD.

Hamilton, W.J., Jr. 1941. The foods of small forest mammals in eastern United States. *Journal of Mammalogy* 22:250-263.

Khera, K.S. and J.A. Ruddick. 1973. Polychlorodibenzo-p-dioxins: Perinatal effects and the dominant lethal test in Wistar rats. *Toxicol*. 120:70-84.

Knable, A.E. 1974. Seasonal trends in the utilization of major food groups by the red fox (*vulpes fulva*) in Union County, Illinois. Trans. III. State Acad. Sci. 66:113-115.

Lariviere, S. and M. Pasitschniak-Arts. 1996. *Vulpes vulpes*. Mammalian Species No. 537. American Society of Mammalogist. 11pp.

Linzey, D.W., and A.V. Linzey. 1973. Notes on food of small mammals from Great Smokey Mountains National Park, Tennessee-North Carolina. J. Elisha Mitchell Science Society 89:6-14.

Marks, T.A., G.L. Kimmel and R.E. Staples. 1989. Influence of symmetrical polychlorinated biphenyl isomers on embryo and fetal development in mice. *Fund. Appl. Tox*. 13:681-693.

Menzie, C., M.H. Henning, J. Cura, K. Finkelstein, J. Gentile, J. Maughan, D. Mitchell, S. Petron, B. Potocki, S. Svirsky and P. Tyler. 1996. Special report of the Massachusetts weight-of – evidence workgroup: A weight-of-evidence approach for evaluating ecological risks. *Human Ecol. Risk Assess*. 2:277-304.

Moore, D.R.J., B.E. Sample, G.W. Suter, B.R. Parkhurst and R.S. Teed. 1999. A probabilistic risk assessment of the effects of methylmercury and PCBs on mink and kingfishers along East Fork Poplar Creek, Oak Ridge, Tennessee, USA. Environ. Toxicol. Chem. 18: 2941-2953.

Sparschu, G.L., F.L. Dunn, and V.K. Rowe. 1971. Study of the Terotogenicity of 2,3,7,8-tetrachlorodibenzo-p-dioxin in the rat. *Fd. Cosmet. Toxicol*. 9:405-412.

Spencer, F. 1982. An assessment of the reproductive toxic potential of Aroclor 1254 in female Sprague Dawley rats. *Bull. Environ. Contam. Tox*. 28-290-297.

Storm, G.L., R.D. Andrews, R.L. Phillips, R.A. Bishop, D.B. Siniff and J.R. Tester. 1976. Morphology, reproduction, dispersal and mortality of Midwestern red fox populations. *Wild. Monogr*. 49, 82 pp.

Voigt, D.R. 1987. Red fox *In* M. Novak, J. A. Baker, M. E. Obbard, and B. Mallock, eds., *Wild Furbearer Management and Conservation in North America*, pp. 378-392. Toronto: Ontario Ministry of Natural Resources, Toronto, Ontario.

11. ASSESSMENT ENDPOINT—SURVIVAL, GROWTH, AND REPRODUCTION OF THREATENED AND ENDANGERED SPECIES

Highlights

Conceptual Model

The assessment endpoint is the survival, growth, and reproduction of T&E species. The measurement endpoints include comparisons to doses reported in the literature to cause adverse effects and field surveys to determine the abundance of T&E species in the Housatonic River PSA. T&E species, including bald eagle, American bittern, and small-footed myotis, selected as representative species for the ERA, are exposed to these COCs via diet and trophic transfer.

Exposure

Exposure of the representative species to tPCBs and TEQ was determined from concentrations found in prey items and an estimation of the daily intake of these COCs from consumption of prey.

Effects

Limited data were available on the toxicity of tPCBs and TEQ to bald eagle, American bittern, and small-footed myotis. Toxicity thresholds for bald eagle were based on field studies in which effects to eggs were measured. Surrogate species were used to develop toxicity thresholds for American bitterns. Sufficient surrogate data were available to generate effects dose-response curves for small-footed myotis.

Risk

Bald eagle, American bittern, and small-footed myotis in the Housatonic PSA are at risk as a result of exposure to tPCBs and TEQ. In particular, bald eagles are at high risk in the PSA. Other similar, but not T&E, species common to the PSA have either higher levels of risk (e.g., least bittern, green heron); similar levels of risk (e.g., great blue heron, Indiana bat, little brown bat); or a lower level of risk (e.g., sora) compared to the representative species.

11.1 INTRODUCTION

The purpose of this section of the ecological risk assessment (ERA) is to characterize and quantify the current and potential risks posed to rare, threatened, and endangered (T&E) species. A Pre-ERA (Appendix B) was conducted to identify contaminants, other than tPCBs, that pose potential risks to aquatic biota and wildlife in the PSA. These COPCs were evaluated further for each assessment endpoint, and the contaminants of concern (COCs) that were retained in the probabilistic risk assessment for T&E species were tPCBs and 2,3,7,8-tetrachlorodibenzo-p-dioxin (2,3,7,8-TCDD) toxic equivalence (TEQ).

11.1.1 Overview of Approach

A step-wise approach was used to assess the risks of tPCBs and TEQ to T&E species in the Housatonic River watershed. The four main steps in this process include:

1. Derivation of a conceptual model (Figure 11.1-1).
2. Assessment of exposure of T&E species to COCs (Figure 11.1-2).
3. Assessment of the effects of COCs on T&E species (Figure 11.1-3).
4. Characterization of risks to the T&E species community (Figure 11.1-4).

The detailed ecological risk assessment for T&E species is provided in Appendix K.

11.1.2 Conceptual Model

The conceptual model presented in Figure 11.1-1 illustrates the exposure pathways for T&E species exposed to tPCBs and TEQ in the PSA. T&E species that reside, or partially reside, within the PSA are exposed to tPCBs and TEQ principally through diet. Fish, amphibians, invertebrates, mammals, and birds comprise the major dietary items for T&E species.

The problem formulation (Section 2) identified the bald eagle (*Haliaeetus leucocephalus*), American bittern (*Botaurus lentiginosus*), and small-footed myotis (*Myotis leibii*) as representative T&E species. American bitterns have been observed during the breeding season in suitable nesting habitat; therefore, they were chosen for inclusion because of the potential for nesting. Similarly, bald eagles nest downstream, have attempted to nest in the PSA, and have ample habitat available for nesting in the PSA. Small-footed myotis may occur in the PSA as well because of their known range and the suitability of habitat. Life history profiles for the bald eagle, American bittern, and small-footed myotis are presented in the following text boxes. Additional life history information on these species is provided in Appendix K, Sections K.2.1.5, K.2.1.6, and K.2.1.7, respectively.

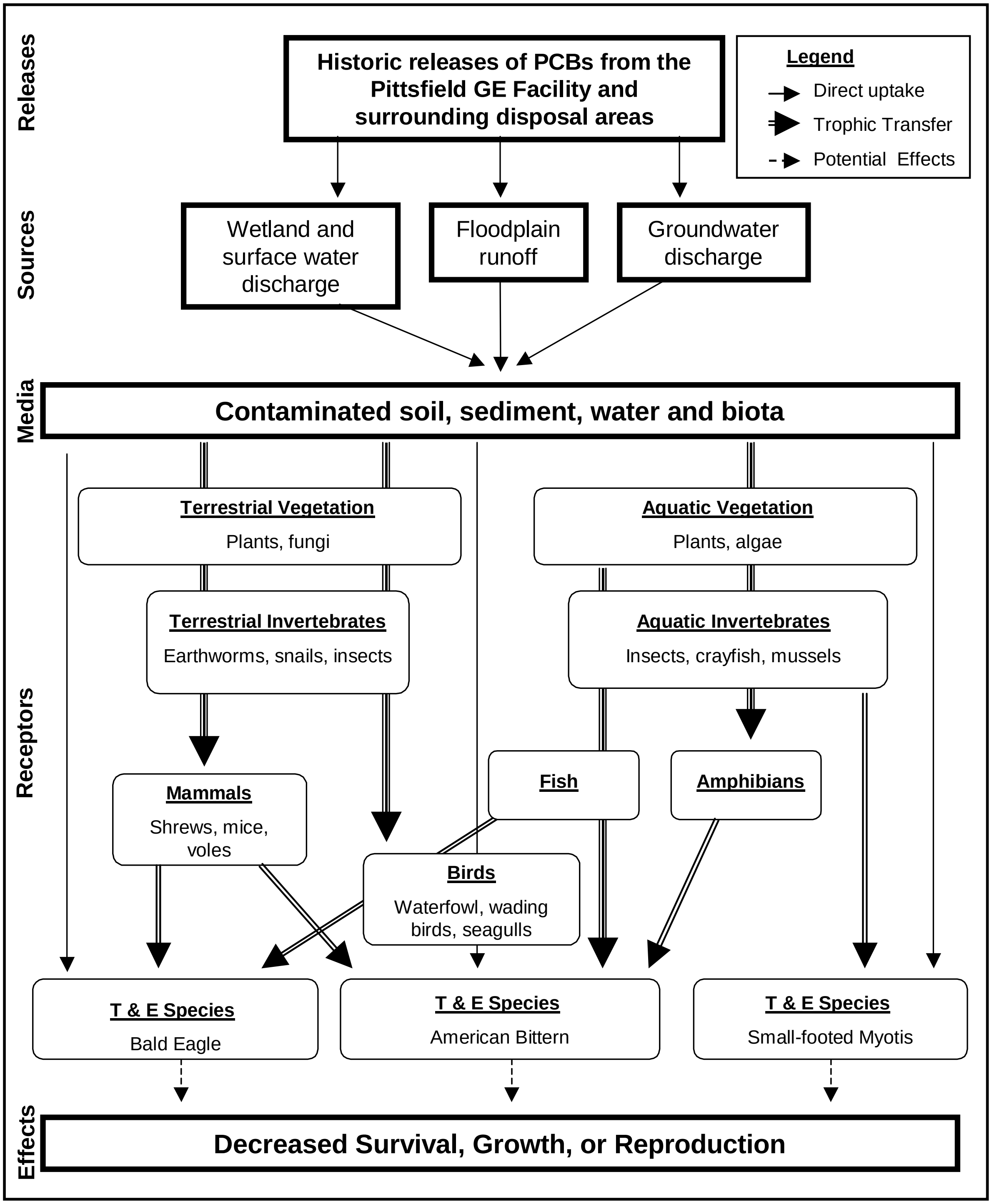


Figure 11.1-1 Conceptual Model Diagram: Exposure Pathways for T&E Species Exposed to COCs in the Housatonic PSA

Exposure

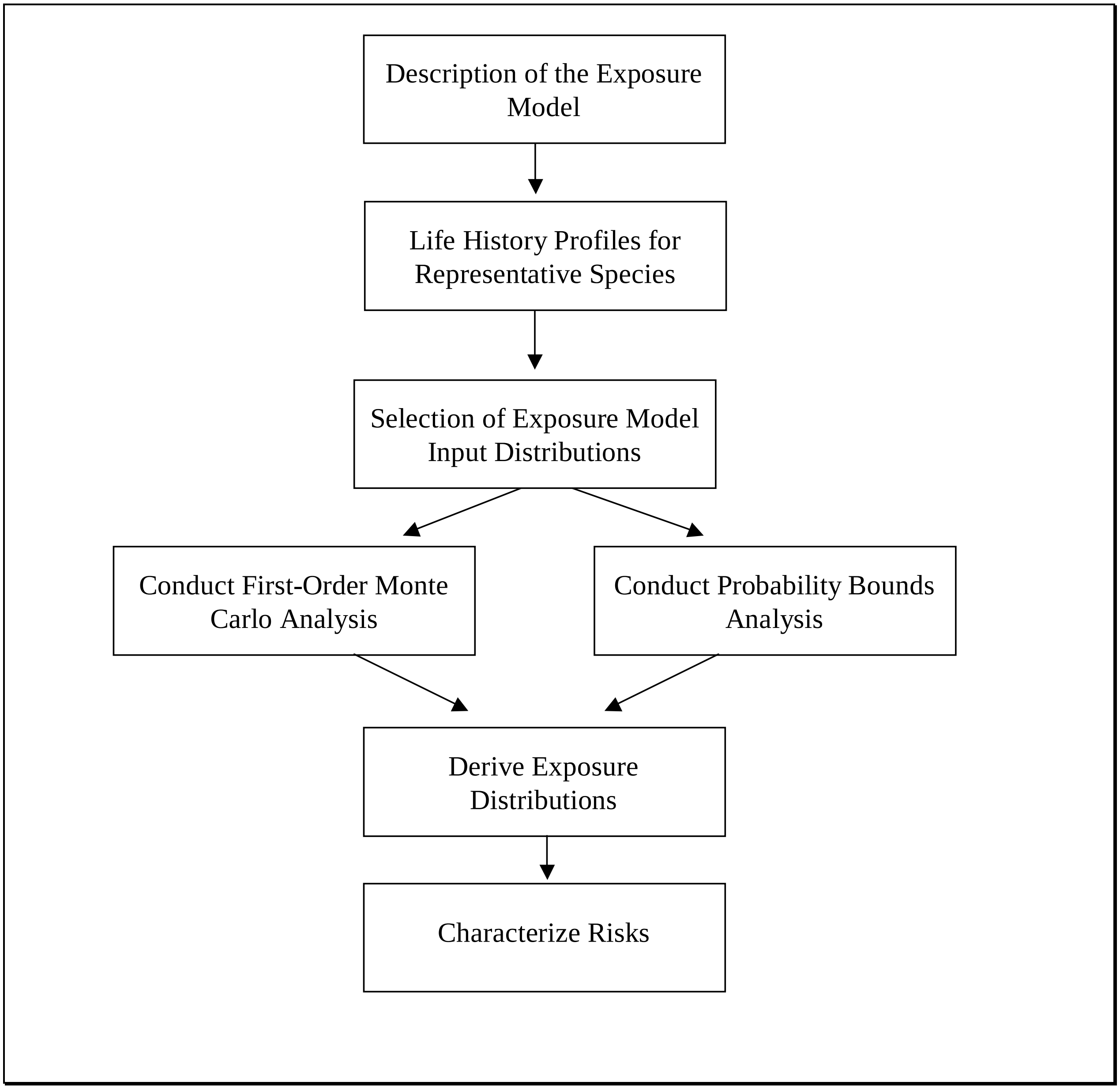


Figure 11.1-2 Approach Used to Assess Modeled Exposure of T&E Species to COCs

Effects

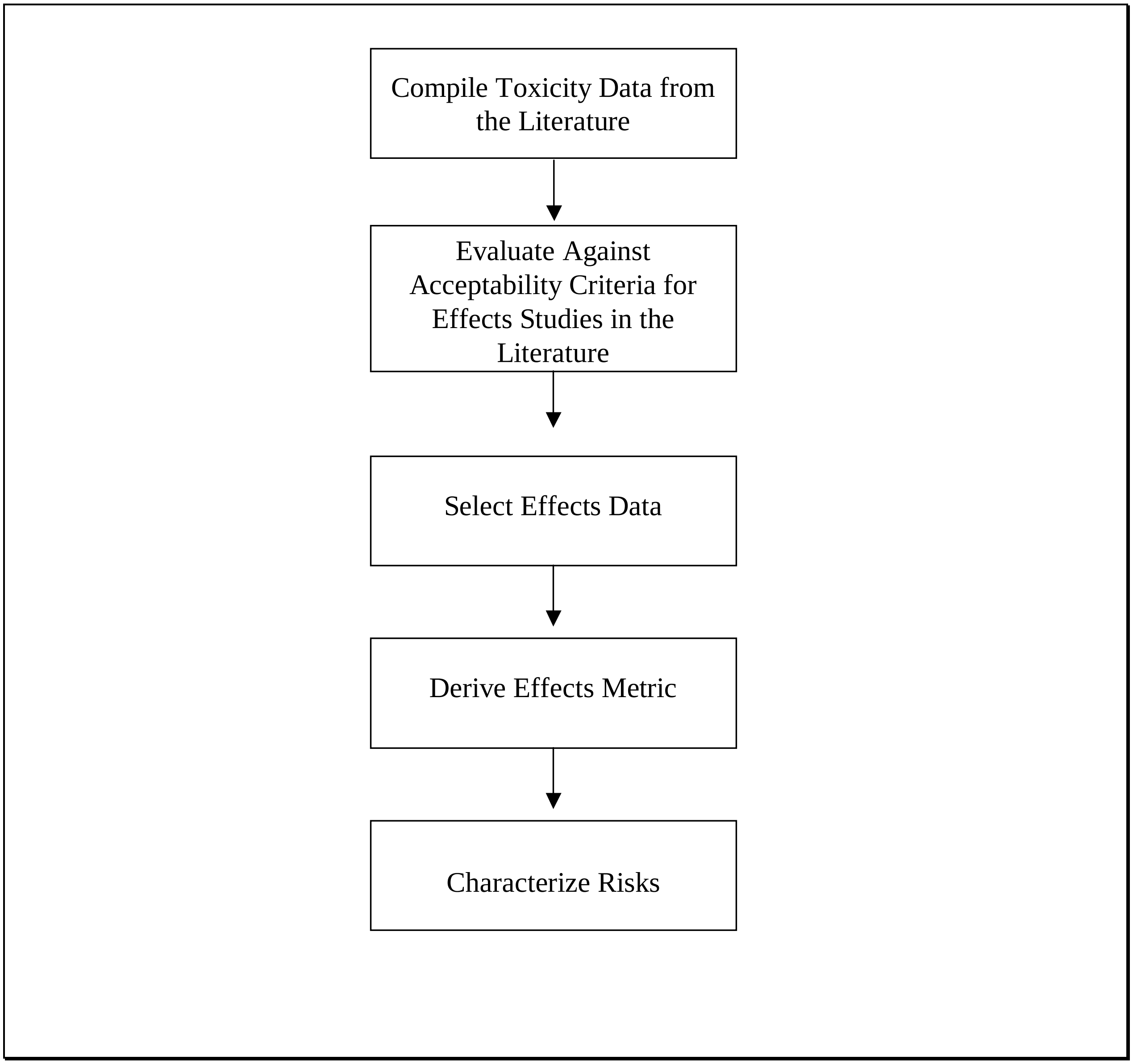


Figure 11.1-3 Approach Used to Assess the Modeled Effects of COCs to T&E Species

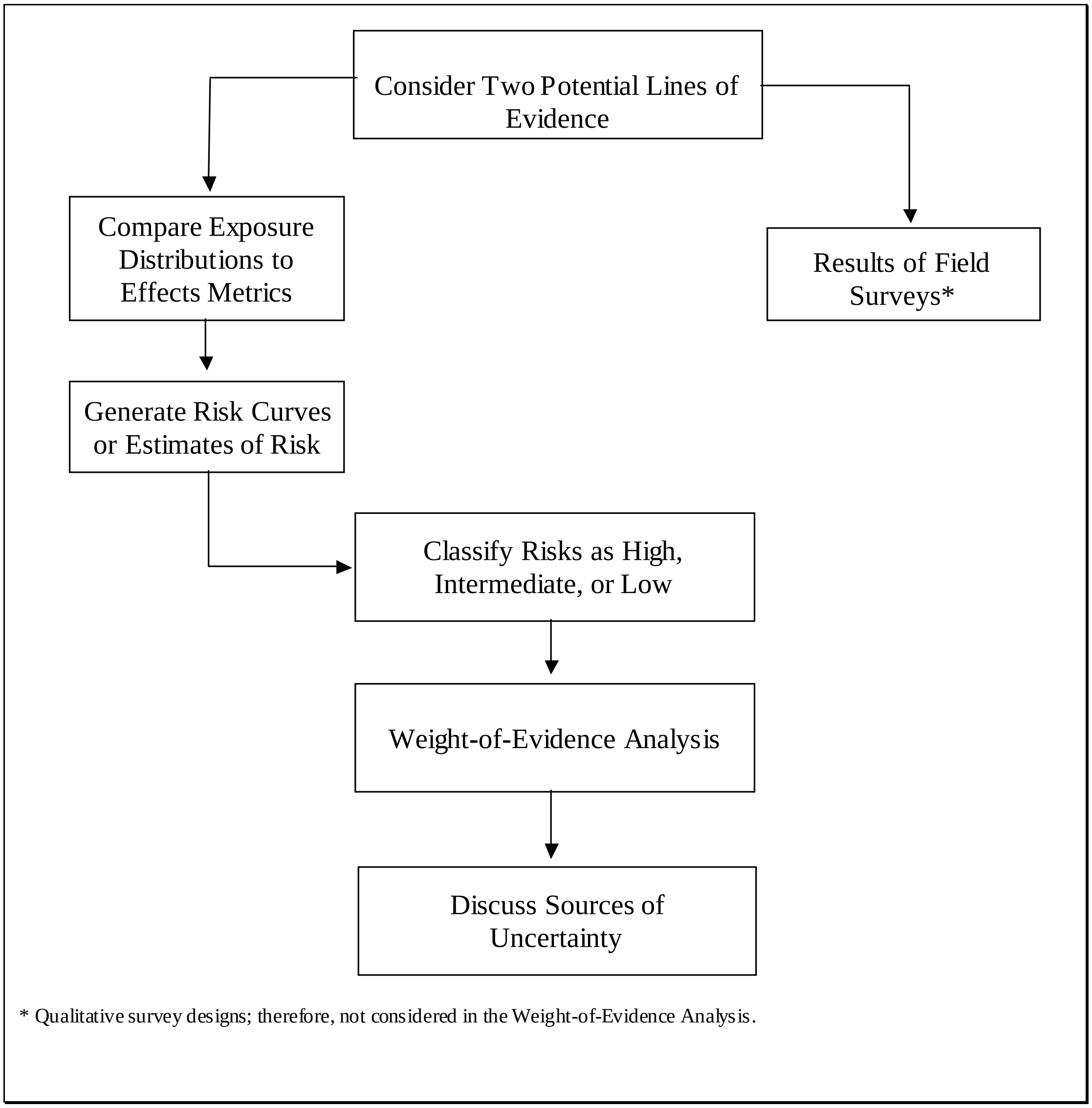


Figure 11.1-4 Approach Used to Characterize Risks of COCs to T&E Species

The assessment endpoint, that is the subject of this section, is the survival, growth, and reproduction of T&E species in the Housatonic River PSA. The potential lines of evidence considered in the evaluation of the assessment endpoint included (1) comparing modeled exposure to doses of tPCBs and TEQ ingested in the diet reported in the literature to cause adverse effects to the survival, reproduction, or growth of omnivorous and carnivorous mammals; and (2) determining, by conducting field surveys, the qualitative abundance of T&E species in the Housatonic River floodplain.

Life History of Bald Eagle

The bald eagle is one of the largest and most conspicuous birds of prey in North America. The bald eagle is currently federally listed as Threatened in all of the 48 lower states, but is more restrictively listed as Endangered by several New England states, including Massachusetts and Connecticut.

- **Habitat** – Habitat use varies depending on the region, but proximity to large bodies of water with suitable foraging opportunities is essential; thus bald eagles are generally restricted to coastal areas, lakes, and rivers. Relatively open canopies, some type of habitat edge, and the availability of super-story trees with stout horizontal perching branches providing good access to nests are preferred habitat features for breeding pairs.
- **Home Range** –Large home ranges, average size of 4,645 ± 2,224 acres; linear (riverine) foraging distances of 1.9 to 4.3 miles. Nesting bald eagles were reported to generally forage within 0.3 mile of the nest, ranging up to 1.9 miles from the nest.
- **Dietary Habits** –Feed primarily over water on aquatic prey; opportunistic feeders, consuming a variety of live prey and scavenging carrion. Fish taken primarily from shallow water form the largest percentage of diet. Fish consumption is 17.1% to 90.1%, depending on location, season, and prey availability. Birds, particularly waterfowl, can form large portions of the diet, more commonly during the winter and in coastal habitats. Mammal species average 4.9% of prey; but are reported to be as much as 11.7%, or as little as 0%.

Life History of American Bittern

The American bittern is a mid-sized heron of freshwater marshes. American bittern populations have been declining since the 1960s primarily as a result of habitat loss and wetland degradation. Massachusetts has included the American bittern on its list of Endangered species.

- **Habitat** – Use a wide range of freshwater wetlands with diversity of types of vegetation interspersed with open water.
- **Home Range** – Varies with geographic area and availability of preferred habitat and prey species. Average home ranges of 315 acres in Minnesota, with the birds using a 61-acre core area more than 50% of the time. In Massachusetts breeding occurs in scattered localities in Berkshire County. Nests built in dense emergent vegetation over water with depths ranging from 2 to 8 inches, consisting of a 6- to 10-inch-high platform of reeds or grasses bent down and lined with fine grasses.
- **Dietary Habits** – Prey upon insects, crayfish, amphibians, fish, and small mammals. Insect prey consists primarily of adult and nymphal dragonflies, giant waterbugs, water scorpions, water beetles, and grasshoppers. Fish species vary with availability and include eels, catfish, pickerel, sunfish, suckers, killifish, sticklebacks, and perch, typically from 10 to 100 mm in length.

Life History of Small-Footed Myotis

The small-footed myotis is a small bat. The small-footed myotis is listed as a Species of Special Concern by the Massachusetts Natural Heritage and Endangered Species Program (MNHESP).

- **Habitat** – Use buildings, overhanging rocks, and caves as summer roosts and maternity sites. Females and young roost in small colonies in rock crevices; males are solitary. Hibernate hanging from walls or underneath fallen rock from November to March, usually in the foothills of mountains up to (2,000 feet) in elevation, in coniferous woodlands.
- **Home Range** – Home range is unknown. It is assumed that home ranges are similar to other *Myotis* species (Indiana bat has a home range of 52 to 95 ha for pregnant and lactating females).
- **Dietary Habits** – Little is known about feeding habits; however, believed to be similar to other *Myotis* species. Flies, beetles, bugs, leafhoppers, and flying ants have been found in stomachs. Likely opportunistic feeders, exploiting available food resources. Fly low to the ground (1 to 3 m) when feeding, along forest openings, including waterways.

11.1.2.1 Studies Used in Support of the ERA

The following studies were used to support the ERA for T&E species:

- Fish samples used in the exposure assessment were collected by EPA and GE between 1994 and 2000. During this period, a total of 321 fish samples were collected from Reach 5 of the Housatonic River. Reach 5 ranges from the confluence of the East and West Branches to the headwaters of Woods Pond. A total of 247 fish samples were collected from Reach 6 (Woods Pond). Reference areas were also sampled during this time. A total of 76 fish samples were taken from the upstream reference area, and 73 fish samples were taken from the Threemile Pond reference area. Fish samples in the preferred size range for bald eagles and American bitterns were used in the exposure assessment.

- Crayfish samples were collected between September and October 1999, from four locations in the PSA and two reference areas. A total of 153 crayfish were collected: 90 from Reach 5 and 63 from the two reference areas. A subset of 60 crayfish samples was selected for tissue analysis of tPCBs and other contaminants. These samples were used in the American bittern exposure assessment.

- Twenty-five waterfowl samples were collected in Reaches 5C, 5D, and 6 in 1999 and were used in the bald eagle exposure analysis.

- Small mammal samples were collected in Reaches 5A, 5B, and 5C and were used in bald eagle and American bittern exposure analysis.

- Amphibian data from two studies were used to develop amphibian tissue concentrations for use in the American bittern exposure assessment. Bullfrog and leopard frog tissue data were available from several sites located throughout the PSA.

- Benthic invertebrate tissue samples collected in Reaches 5A, 5B, 5C, 5D and 6 were used in the American bittern exposure assessment. Tree swallow gut content samples from Reaches 5A, 5B, and 5C (also composed of emerging invertebrates) were used in the small-footed myotis exposure assessment.

11.2 EXPOSURE ASSESSMENT

All of the bald eagle sightings in the PSA occurred south of New Lenox Road, primarily at Woods Pond and the backwaters north of Woods Pond (Appendix A). Bald eagles would not be expected to regularly utilize the more shallow and narrow northern sections of the river. Therefore, the exposure area assumed for bald eagles was the southern portion of the PSA, from the more downstream portion of Reach 5B to Woods Pond. This entire area was not subdivided because individual bald eagles would likely forage throughout this area.

American bittern habitat occurs throughout the PSA, and bitterns have been observed from Canoe Meadows Wildlife Sanctuary south to Woods Pond (Appendix A). Home range sizes and habitat requirements for this species are such that individuals forage predominantly within one subreach. Therefore, for the PCB analyses, the PSA was split into four reaches: Reach 5A, Reach 5B, Reach 5C, and Reaches 5D and 6 combined. For the TEQ analyses, samples from the PSA were combined into one analysis because the smaller sample sizes did not allow for statistically robust analyses to be conducted for each subreach.

Little is known about the home range size of the small-footed myotis. The exposure area assumed for small-footed myotis was the entire PSA because of the small number of dietary samples from each reach.

11.2.1 Exposure Model

Exposure of T&E species to tPCBs and TEQ was estimated using a total daily intake model adapted from the *Wildlife Exposure Factors Handbook* (EPA 1993). The model used in the exposure analysis was:

$$TDI = FT \cdot FIR \sum_{i=1}^{n} C_i \cdot P_i \qquad \text{(Eq. 1)}$$

where

TDI	=	total daily intake (mg/kg bw/d tPCBs, ng/kg bw/d TEQ)
FIR	=	normalized food intake rate (kg/kg bw/d)
FT	=	foraging time in Primary Study Area (unitless)
C_i	=	concentration in *i*th food item (mg/kg tPCBs, ng/kg TEQ)
P_i	=	proportion of the *i*th food item in the diet (unitless)

The models consider the food intake rates (FIRs) of the representative species (*FIR*), the concentrations of COCs in each food item (C_i), and the proportion of the diet accounted for by that food item (P_i). For those input variables that are uncertain, variable, or both, distributions are used rather than point estimates. Monte Carlo and probability bounds analyses are the methods used to propagate uncertainties about input variables through the exposure model for each COC. A description of these techniques and the methods used to parameterize input variables is presented in Section 6.5. The distributions used in the exposure analyses for T&E species and the results of these analyses are discussed in detail in Appendix K. A brief description of these variables is provided below.

11.2.1.1 Input Variables

11.2.1.1.1 Body Weight *(BW)*

Adult male bald eagles average 4.13 kg in weight and adult females average 5.4 kg (Dunning 1992; EPA 1993; Buehler 2000; Canadian Wildlife Service 2000). For this risk assessment, the weight of female bald eagles was used because the effects endpoint is reproductive impairment. The mean weight was 5.35 kg with a standard deviation of 0.40 g.

The typical weight of an adult American bittern ranges from 370 g to >800 g (Gibbs et al. 1992; Dunning 1992). In the Monte Carlo analyses, the mean BW of 707 g with a standard deviation of 183 g was used (Dunning 1992).

Small-footed myotis typically weigh 5 to 7 g, although their weights can range from 3 to 8 g (Kurta 1995). A mean BW of 6 g with a standard deviation of 0.7 g was used in the exposure analyses for small-footed myotis.

11.2.1.1.2 Food Intake Rate *(FIR)*

FIR was derived from the estimated metabolic rate of free-living eagles using data from Nagy (1987) and Nagy et al. (1999).

Estimated values for free-flying eagles from Connecticut (Craig et al. 1988) were 0.12 to 0.14 g/g bw/d, while the median FIR from the allometric equation was 0.158 g/g bw/d.

Nagy (1987) and Nagy et al. (1999) derived allometric equations for estimating the free metabolic rate (FMR) of free-living mammals in kilojoules per day using the following general equation:

$$FMR\ (kJ/d) = a \cdot BW(g)^b \qquad \text{(Eq. 2)}$$

where *FMR* (kcal/kg bw/d) is the free-living metabolic rate, and *BW (g)* is the body weight in grams. The slope (*a*) and power (*b*) distributions were based on the error statistics reported in Nagy et al. (1999).

FIR is derived from FMR using the following equation:

$$FIR = \frac{FMR}{\sum_{i=1}^{n} AE_i \cdot GE_i} \qquad \text{(Eq. 3)}$$

where *AE* is the assimilation efficiency of *i*th food item (unitless) and GE_i is the gross energy of *i*th food item (kcal/kg).

The gross energies of various wildlife food sources are summarized in the *Wildlife Exposure Factors Handbook* (EPA 1993). The mean gross energy is 1.6 for invertebrates, 1.2 for fish and amphibians, and 1.8 for birds and mammals.

The mean assimilation efficiency for dietary items consumed by birds is 79% for fish and amphibians, 78% for mammals, and 72% for invertebrates. Point estimates were used for these variables in the Monte Carlo and probability bounds analyses because of their relatively small coefficients of variation (i.e., CV<10%).

11.2.1.1.3 Proportions of Dietary Items (P_i)

The proportions of prey items in bald eagle diets are listed in Table K.2-1. Fish comprise a mean of 77.5% of the diet, bird species are approximately 17%, and mean percentage of mammals consumed is approximately 5%.

An analysis of the stomach contents of 160 individuals reported that the American bittern diet consisted of invertebrates (23%), amphibians (21%), fish (21%), crayfish (19%), small mammals (10%), and snakes (5%) (Cottam and Uhler 1945, as cited in Gibbs et al. 1992). Based on this information, the proportion in the diet for this analysis was parameterized to allow the diet to equal 100%, resulting in a diet of 24.5% invertebrates, 20.2% macroinvertebrates (crayfish), 22.3% fish, and 22.3% amphibians (Table K.2-9).

Along the Housatonic River, the small-footed myotis likely forages on small emergent aquatic insects, as does the little brown bat. For this exposure assessment, the proportion of invertebrates in the diet was assumed to be a point estimate, with invertebrates accounting for 100% of the small-footed myotis diet.

11.2.1.1.4 Foraging Time (FT)

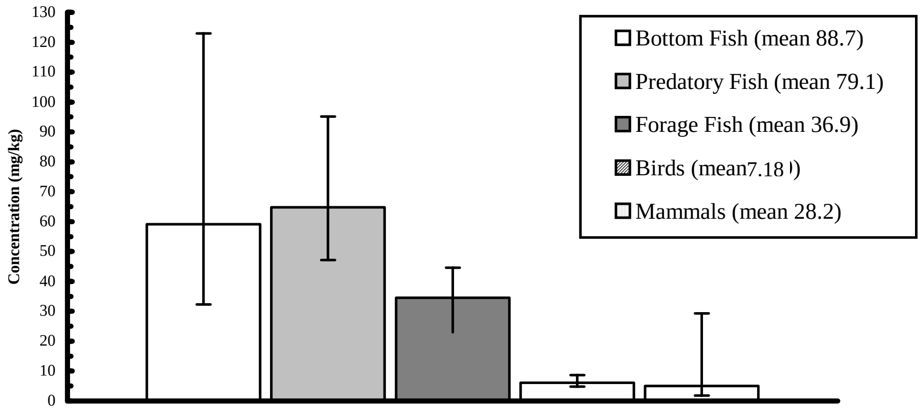
Bald eagles nesting in the PSA would be expected to forage entirely within the PSA, as they generally forage within 0.3 miles (0.5 km) of the nest, with a maximum reported foraging distance of up to 5.0 miles (8.0 km) from their nest (Bowerman et al. 1995; Stratus 1999). American bittern were also assumed to spend 100% of their time foraging in the PSA.

Little is known about home range size of the small-footed myotis. Small-footed myotis feed predominantly over water on emergent insects; therefore, it was assumed that small-footed myotis would forage 100% of the time in the PSA.

11.2.1.2 Concentrations of COCs in Prey

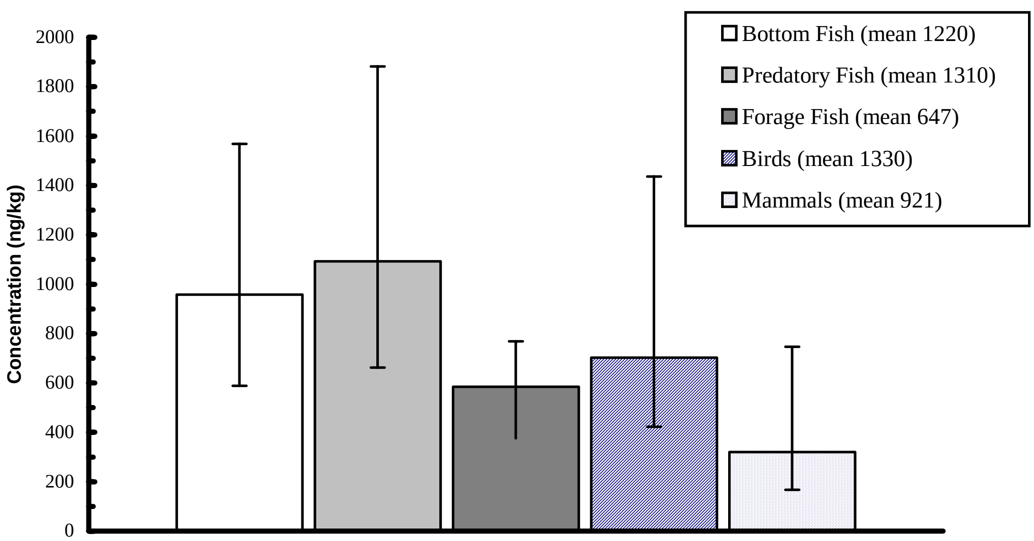
The median, 25^{th} and 75^{th} percentile concentrations of tPCBs and TEQ of the dietary items for bald eagles are presented in Figures 11.2-1 and 11.2-2, respectively.

Similar statistics for concentrations of tPCBs and TEQ in American bittern prey from Reaches 5A, 5B, 5C, and 5D and 6 are presented in Figures 11.2-3 and 11.2-4, respectively.



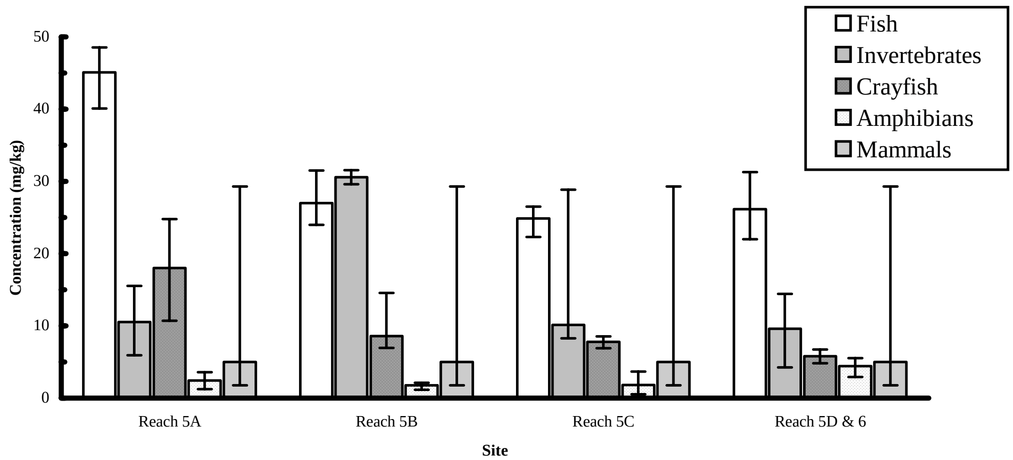
Note: Tissue concentrations are wet weight.

Figure 11.2-1 Median Concentrations of tPCBs in Prey of Bald Eagles



Note: Tissue concentrations are wet weight.

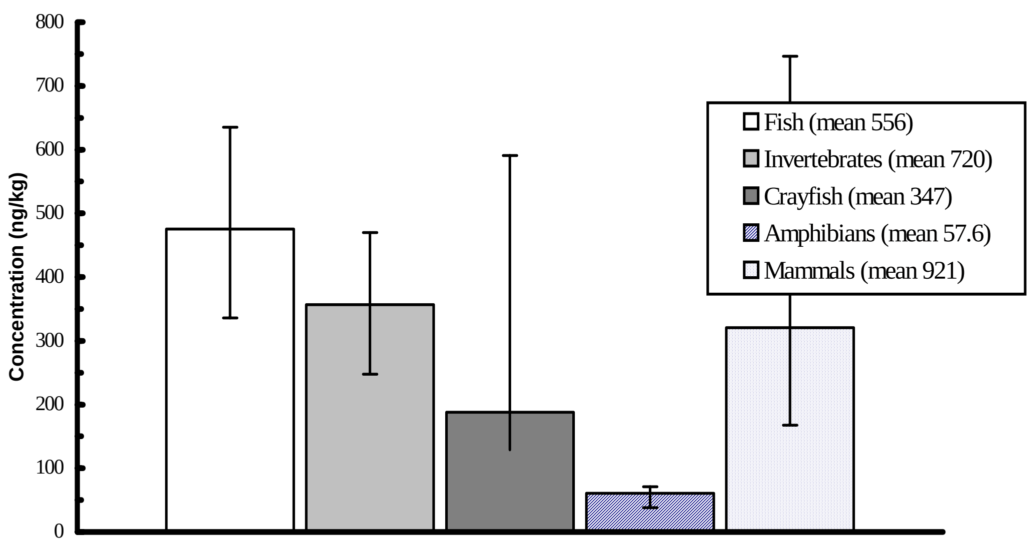
Figure 11.2-2 Median Concentrations of TEQ in Prey of Bald Eagles



Fish	Invertebrates	Crayfish	Amphibians	Mammals
Reach 5A, mean 43.03	Reach 5A, mean 10.51	Reach 5A, mean 21.67	Reach 5A, mean 2.774	Reach 5A, mean 28.21
Reach 5B, mean 27.68	Reach 5B, mean 30.56	Reach 5B, mean 12.28	Reach 5B, mean 2.375	Reach 5B, mean 28.21
Reach 5C, mean 22.62	Reach 5C, mean 21.37	Reach 5C, mean 8.223	Reach 5C, mean 2.586	Reach 5C, mean 28.21
Reach 5D&6, mean 25.43	Reach 5D&6, mean 9.081	Reach 5D&6, mean 6.799	Reach 5D&6, mean 4.242	Reach 5D&6, mean 28.21

Note: Tissue concentrations are wet weight.

Figure 11.2-3 Median Concentrations of tPCBs in Prey of American Bittern



Note: Tissue concentrations are wet weight.

Figure 11.2-4 Median Concentrations of TEQ in Prey of American Bittern

Small-footed myotis prey items were not directly sampled in the PSA. Because both species are aerial insectivores, concentrations of tPCBs in invertebrates were obtained from samples of tree swallow (*Tachycineta bicolor*) gut contents (Custer 2002). The median concentration of these samples was 7.10 mg/kg for tPCBs and 564 ng/kg ww for TEQ.

The input variables for concentrations of COCs in prey of bald eagle, American bittern, and small-footed myotis are shown in Tables K.2-4, K.2-5, K.2-12, K.2-13, K.2-20, and K.2-21.

11.2.2 Results of Exposure Assessments

Examples of exposure distributions for exposure to tPCBs and TEQ for bald eagles in the southern PSA, American bittern in Reaches 5 and 6, and small-footed myotis in the PSA are presented in Figures 11.2-5 through 11.2-20.

11.2.2.1 Bald Eagle

Figure 11.2-5 depicts the cumulative distribution of tPCB intake rates by bald eagles in the southern PSA. The Monte Carlo analysis indicated that the mean exposure was 13.2 mg/kg bw/d, and the median exposure 13.0 mg/kg bw/d. Eighty percent of the exposure estimates were between 10.2 and 16.5 mg/kg bw/d (Table K.2-6).

The probability bounds estimated for bald eagles foraging in the southern PSA are depicted in Figures 11.2-5 and 11.2-6. The 50th percentile of the P-bounds analysis ranged between 5.41 and 17.9 mg/kg bw/d, and the 90th percentile ranged between 8.27 and 24.2 mg/kg bw/d. Details of these distributions are provided in Table K.2-6.

The Monte Carlo analysis indicated that the concentration of tPCB in the first bald eagle egg could range from a minimum of 35.0 mg/kg ww to a maximum of 168 mg/kg ww. The mean estimated concentration was 80.5 mg/kg ww, and the median was 79.0 mg/kg ww. Eighty percent of the egg concentration estimates were between 60.9 and 102 mg/kg ww. Figure 11.2-7 depicts the cumulative distribution for tPCB in the first bald eagle egg in the southern PSA.

The probability bounds estimated for tPCB concentration in the first bald eagle egg for eagles nesting in the southern PSA are depicted in Figure 11.2-8. The 10th percentile of the probability envelope formed by the lower and upper bounds ranged between 18.0 and 88.0 mg/kg ww. The 50th percentile ranged between 26.6 and 117 mg/kg ww, and the 90th percentile ranged between 44.2 and 174 mg/kg ww. In comparison, the 10th percentile of the Monte Carlo output was 60.9, the 50th percentile was 79.0, and the 90th percentile was 102 mg/kg ww.

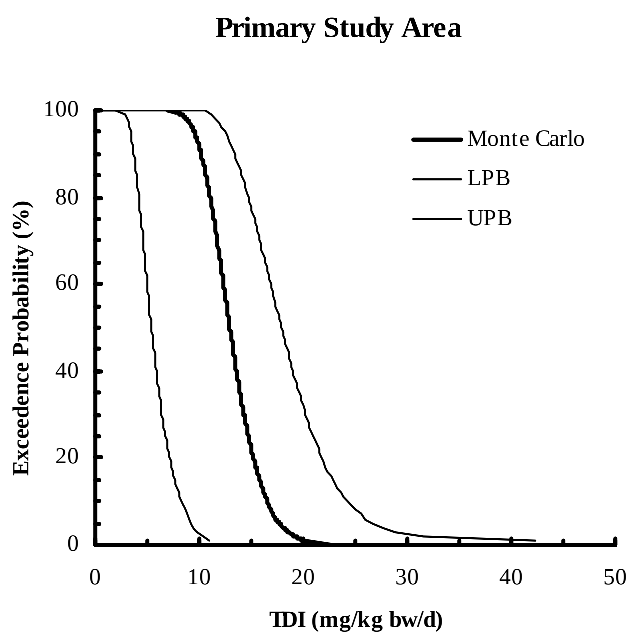


Figure 11.2-5 Total Daily Intake (TDI) of tPCBs by Bald Eagles in the Housatonic River Primary Study Area

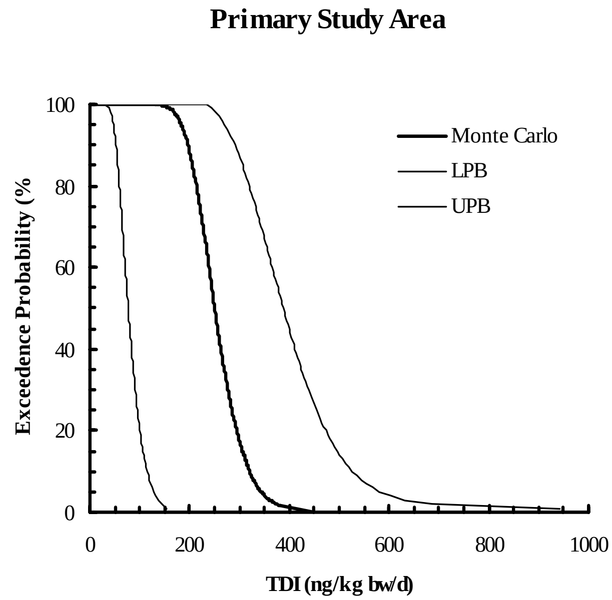
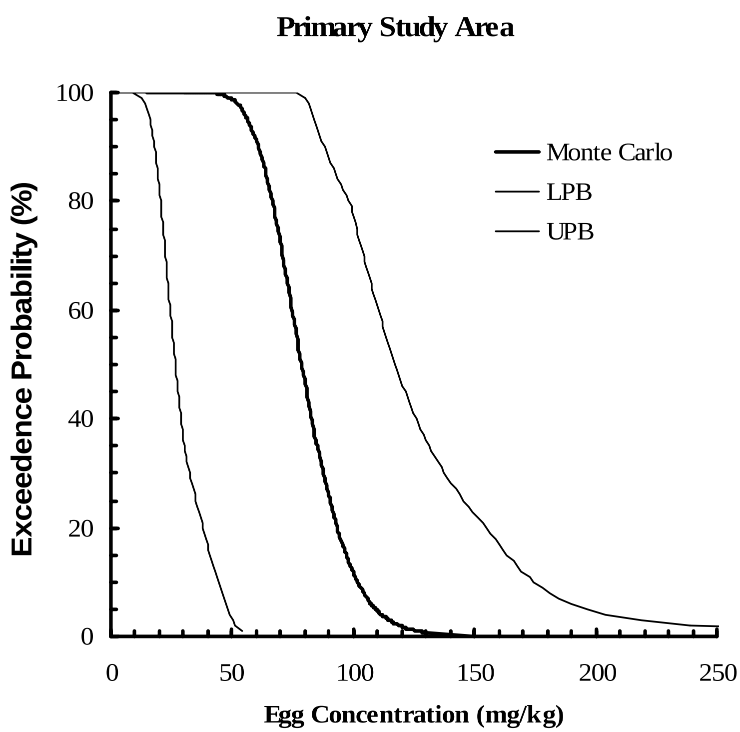
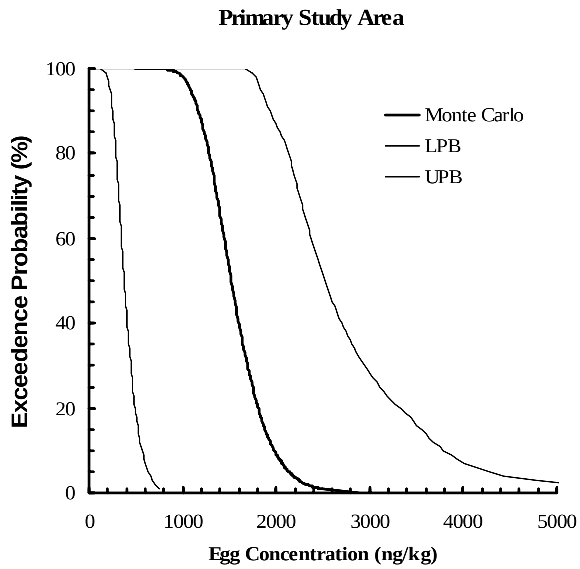


Figure 11.2-6 Total Daily Intake (TDI) of TEQ by Bald Eagles in the Housatonic River Primary Study Area



Note: Egg concentrations are wet weight.

Figure 11.2-7 Bald Eagle Egg Exposure to PCBs in the Housatonic River Primary Study Area



Note: Egg concentrations are wet weight.

Figure 11.2-8 Bald Eagle Egg Exposure to TEQ in the Housatonic River Primary Study Area

11.2.2.2 American Bittern

The Monte Carlo analysis indicated that exposure of American bittern to tPCBs ranged from a minimum of 4.70 to a maximum of 18.6 mg/kg bw/d. The mean exposure was 9.24 mg/kg bw/d (Table K.2-14).

The probability bounds estimated for American bittern foraging in the PSA are depicted in Figures 11.2-9 through 11.2-18. In Reach 5A, the 50^{th} percentile ranged between 4.42 and 9.19 mg/kg bw/d, and the 90^{th} percentile ranged between 5.49 and 11.7 mg/kg bw/d. In comparison, the 50^{th} percentile of the Monte Carlo analysis was 9.07, and the 90^{th} percentile was 11.4 mg/kg bw/d (Table K.2-14). Exposures of American bittern to tPCBs in Reaches 5B, 5C, and 5D, and 6 were similar or lower than in Reach 5A.

Mean exposure of American bittern to TEQ was 372 for the PSA (Table K.2-16). Figures 11.2-13 and 11.2-18 depict the cumulative distribution for TEQ intake, as well as the probability bounds.

Mean egg concentrations in the PSA after 45 days were 37.0, 31.4, 36.2, and 26.2 mg/kg for Reaches 5A, 5B, 5C, and 6, respectively. The estimated TEQ egg concentration for American bitterns over time is shown in Figure 11.2-18; the mean concentration was 1,490 ng/kg.

Reach 5a

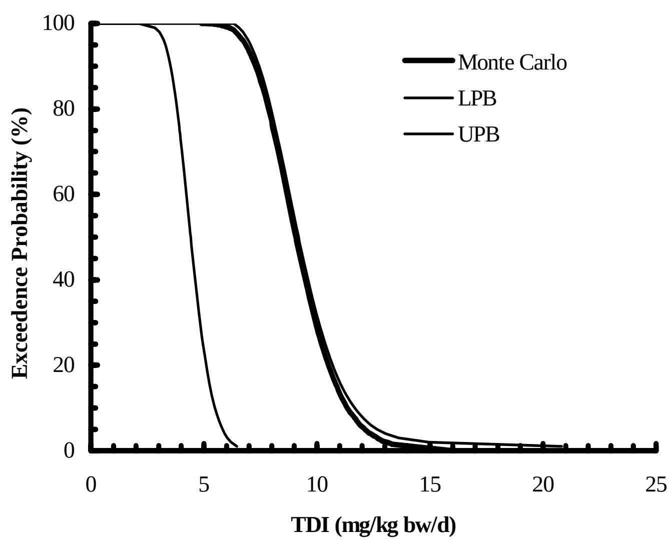


Figure 11.2-9 Total Daily Intake (TDI) of tPCBs by American Bittern in Reach 5A of the Housatonic River Primary Study Area

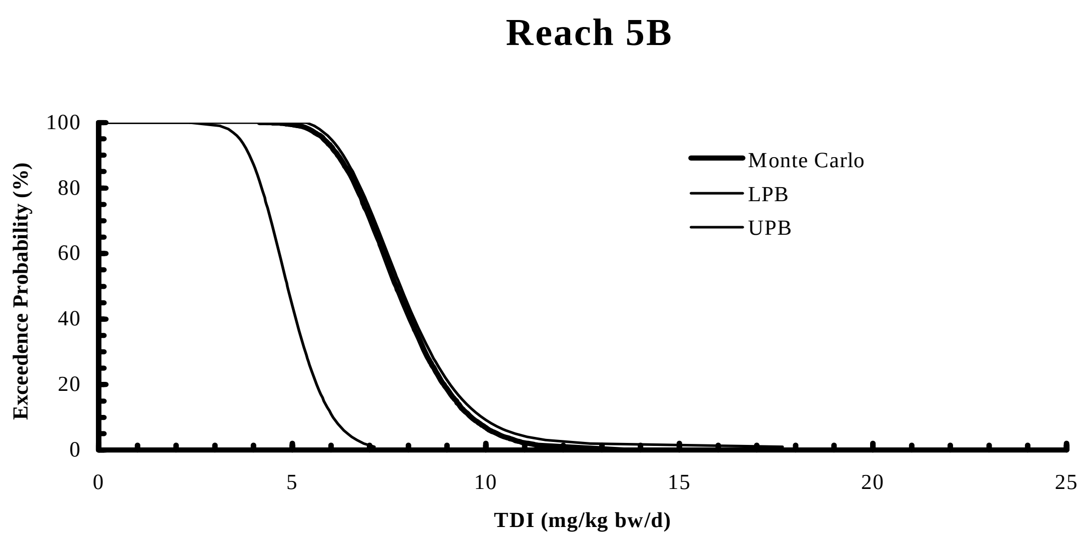


Figure 11.2-10 Total Daily Intake (TDI) of tPCBs by American Bittern in Reach 5B of the Housatonic River Primary Study Area

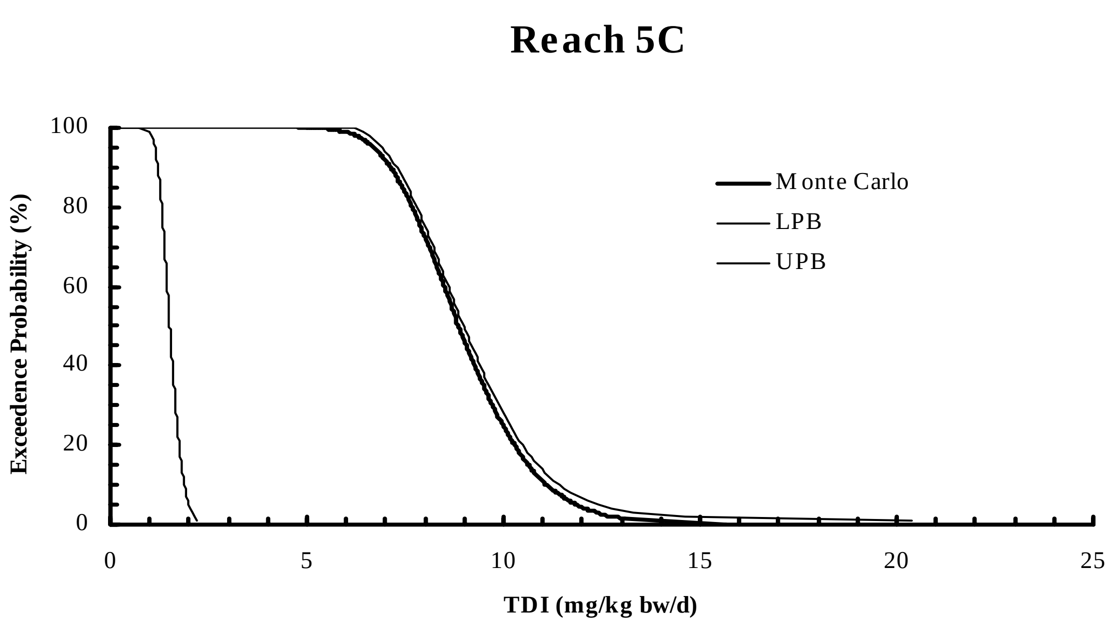


Figure 11.2-11 Total Daily Intake (TDI) of tPCBs by American Bittern in Reach 5C of the Housatonic River Primary Study Area

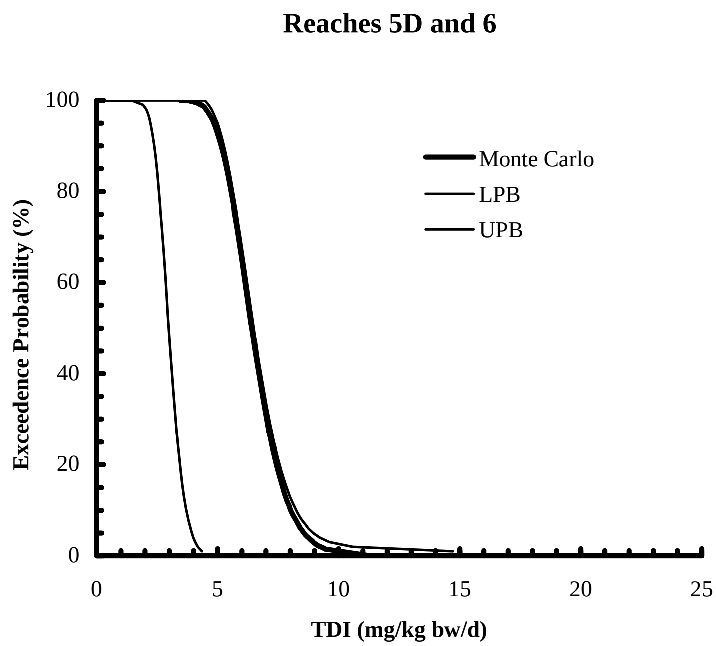


Figure 11.2-12 Total Daily Intake (TDI) of tPCBs by American Bittern in Reaches 5D and 6 of the Housatonic River Primary Study Area

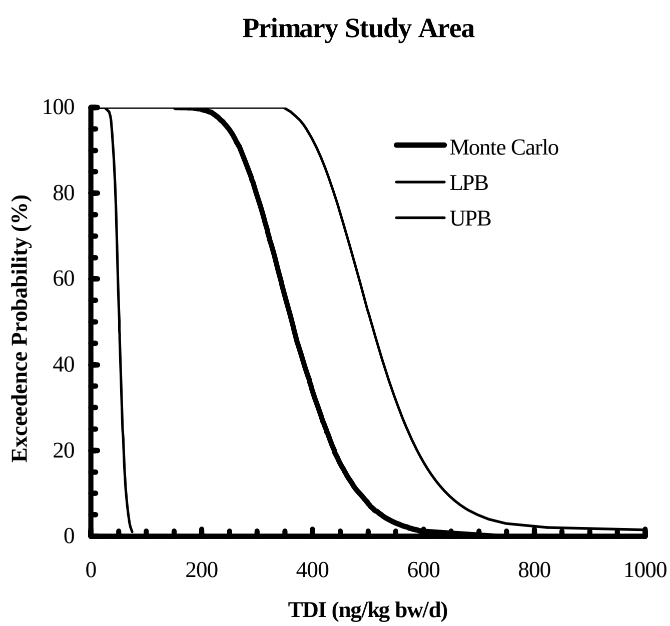
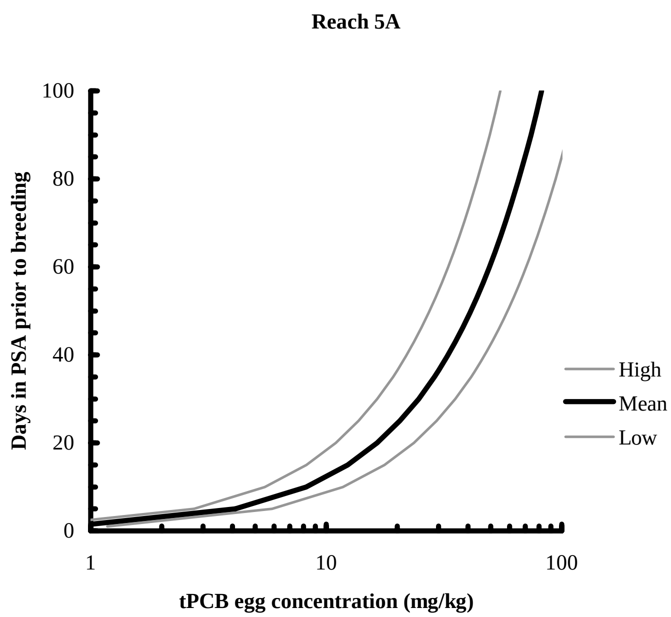
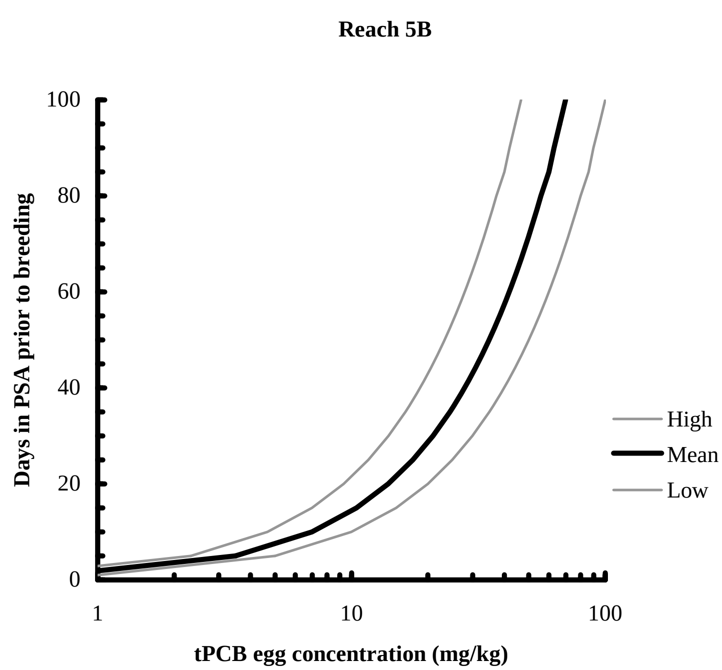


Figure 11.2-13 Total Daily Intake (TDI) of TEQ by American Bittern in the Housatonic River Primary Study Area



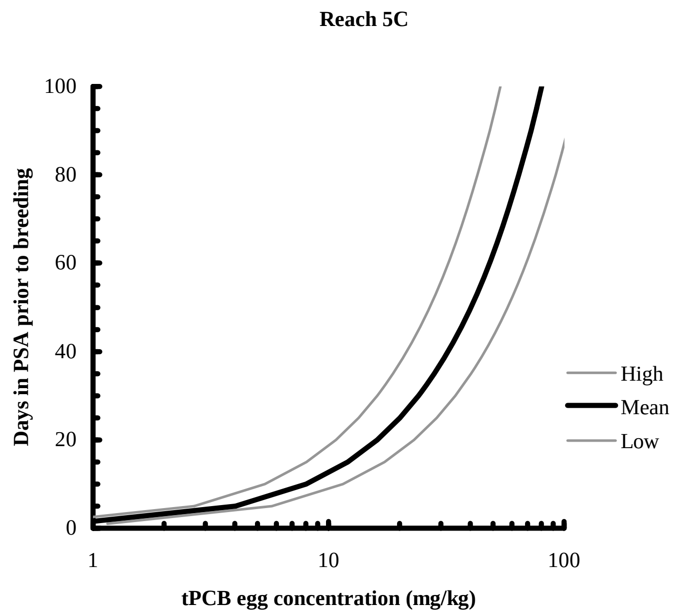
Note: Egg concentrations are wet weight.

Figure 11.2-14 American Bittern Egg Exposure to tPCBs in Reach 5A of the Housatonic River Primary Study Area



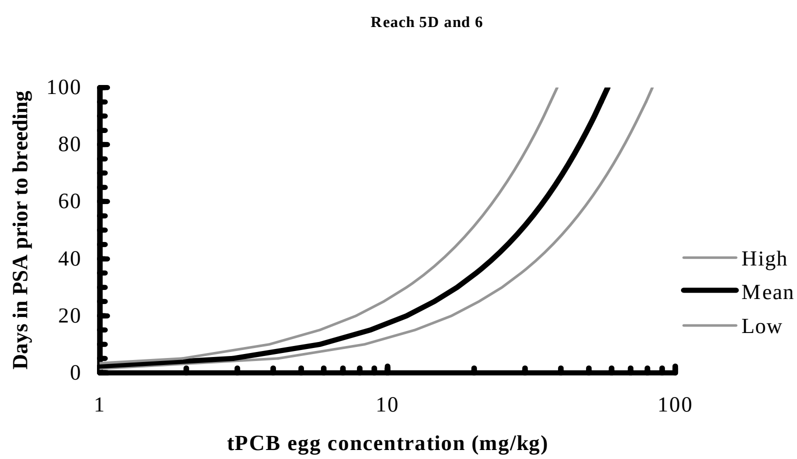
Note: Egg concentrations are wet weight.

Figure 11.2-15 American Bittern Egg Exposure to tPCBs in Reach 5B of the Housatonic River Primary Study Area



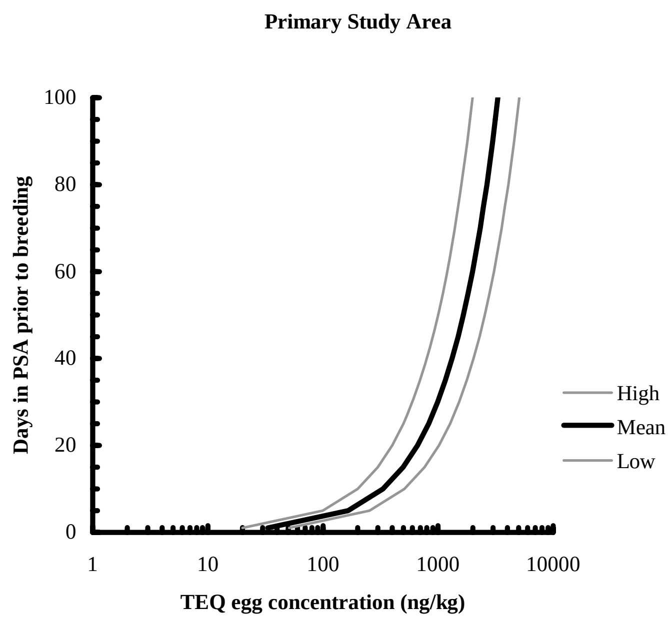
Note: Egg concentrations are wet weight.

Figure 11.2-16 American Bittern Egg Exposure to tPCBs in Reach 5C of the Housatonic River Primary Study Area



Note: Egg concentrations are wet weight.

Figure 11.2-17 American Bittern Egg Exposure to tPCBs in Reaches 5D and 6 of the Housatonic River Primary Study Area



Note: Egg concentrations are wet weight.

Figure 11.2-18 American Bittern Egg Exposure to TEQ in Reaches 5 and 6 of the Housatonic River Primary Study Area

11.2.2.3 Small-Footed Myotis

11.2.2.3.1 Total PCBs

The Monte Carlo analysis indicated that the mean exposure of small-footed myotis to tPCBs was 16.7 mg/kg bw/d, and the median exposure was 14.5 mg/kg bw/d (Table K.2-22). Figure 11.2-19 depicts the cumulative distribution for small-footed myotis in Reach 5.

The probability bounds estimated for small-footed myotis foraging in Reach 5 are depicted in Figure 11.2-19. The 50th percentile of the probability envelope formed by the lower and upper bounds ranged between 4.05 and 15.0 mg/kg bw/d. In comparison, the 50th percentile of the Monte Carlo output was 14.5 mg/kg bw/d (Table K.2-22).

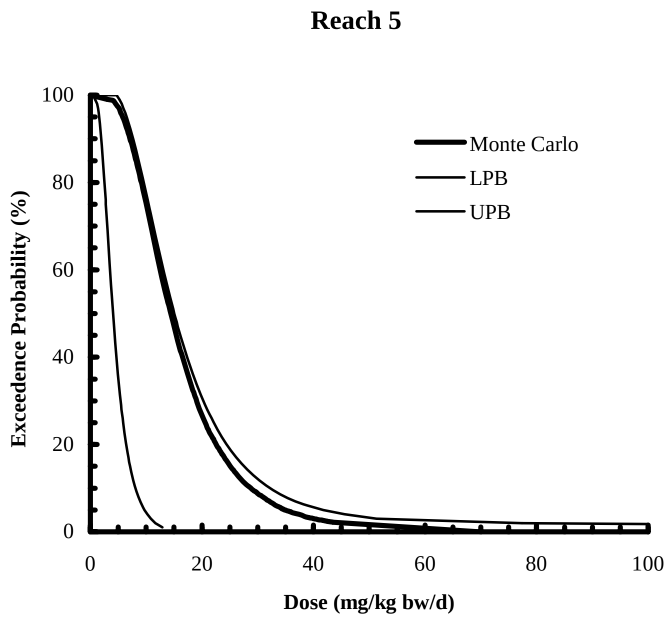


Figure 11.2-19 Total Daily Intake (TDI) of tPCBs by Small-Footed Myotis in Reach 5 of the Housatonic River Primary Study Area

11.2.2.3.2 TEQ

The Monte Carlo analysis indicated that mean exposure of small-footed myotis to TEQ was 1,130 mg/kg bw/d, and the median exposure was 936 ng/kg bw/d (Table K.2-24). Figure 11.2-20 depicts the cumulative distribution for small-footed myotis in Reach 5.

The probability bounds estimated for small-footed myotis foraging in the PSA are depicted in Figure 11.2-20. The 50th percentile of the probability envelope formed by the lower and upper bounds ranged between 38.1 and 1,910 ng/kg bw/d. In comparison, the 50th percentile of the Monte Carlo output was 936 ng/kg bw/d (Table K.2-24).

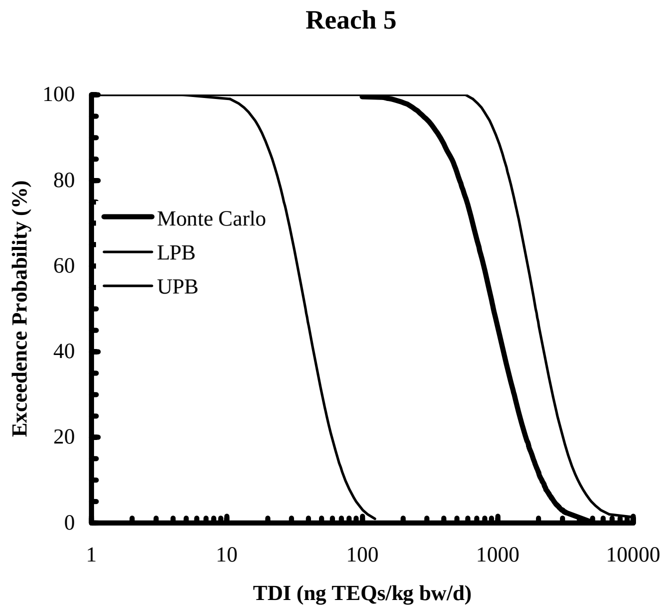


Figure 11.2-20 Total Daily Intake (TDI) of TEQ by Small-Footed Myotis in Reach 5 of the Housatonic River Primary Study Area

11.3 EFFECTS ASSESSMENT

The purpose of the effects assessment is to review the scientific literature and to derive the most appropriate metrics for effects of tPCBs and TEQ to T&E species. The effects metric is used, in conjunction with the exposure assessment, to estimate risks to T&E species exposed to tPCBs and TEQ in the Housatonic River PSA. This section focuses on effects that have an influence on the long-term maintenance of T&E species populations (i.e., mortality or impairment of reproduction or growth).

A brief review of the scientific literature on the effects of tPCBs and TEQ to T&E species from dietary exposure is presented in the following sections. The discussion focuses on ecologically relevant effect endpoints such as survival, growth, and reproduction of T&E species. A summary of reproduction effects for tPCBs and TEQ is presented in Table K.3-1. The effects metrics used for this assessment are also presented.

Field survey data are presented in the risk characterization, although the surveys were not designed to characterize risks to T&E species from COCs, rather they were designed to verify presence of species and/or suitable habitats and specific habitats used. Effects-based field surveys for T&E species were not performed because of the potential impacts to these species from the surveys themselves, and the necessarily small sample sizes that would preclude statistical analysis of data. Each of the T&E species chosen for assessment occurs in small numbers, which limits the types and amount of data that can be collected. Barring an acute effect from the COCs, such as observed mortality, small sample sizes can lead to incorrect conclusions.

11.3.1 Total PCBs

Laboratory studies on the toxicity of PCBs to bald eagles and American bittern have not been conducted. However, studies using other avian species were available. Appendix H provides detailed descriptions of dietary and in ovo exposures of PCBs and TEQ to surrogate bird species.

Laboratory studies on raptor species demonstrated that PCBs cause adverse effects. American kestrels (*Falco sparverius*) dosed in ovo had decreased reproductive success, including suppression of egg laying, delays in clutch initiation, smaller clutch sizes, and reduced fledgling survival (Fernie et al. 2001a). Numerous field studies have found that organochlorine compounds negatively impact the reproductive success of raptors and piscivorous birds (see overview in Donaldson et al. 1999). Toxicological effects include reduced hatching success, malformation, edema, and reduced organ and body weight (Elliott et al. 1996).

Hoffman et al. (1986) found a negative correlation between embryonic weight and tPCB residues in eggs of black-crowned night herons nesting in the San Francisco Bay. Other effects associated with PCB exposure in black-crowned night herons included reduced femur to body weight ratio, increased edema, and increased hepatic aryl hydrocarbon hydroxylase activity (Hoffman et al. 1993). Laporte (1982) reported that mean tPCB concentrations of 15 mg/kg ww in eggs negatively impacted great blue heron reproductive success in Quebec.

No PCB toxicology studies have been conducted on small-footed myotis; however, studies have been conducted for little brown bats and big brown bats. Studies have shown that bats accumulate PCBs from their diet (Clark and Lamont 1976a; Clark and Lamont 1976b; Clark and Prouty 1976; Clark 1978; Clark and Stafford 1981). PCBs are also known to have adverse reproductive effects on bats (Clark and Lamont 1976a; Clark and Lamont 1976b; Clark 1978). Wild captured female big brown bats that produced dead young contained significantly higher concentrations of PCBs (1.99 mg/kg ww) than those that produced live young (0.56 mg/kg ww) (Clark and Lamont 1976b).

11.3.2 2,3,7,8-TCDD Toxic Equivalence (TEQ)

Several researchers estimated NOAEL and LOAEL values for TEQ for bald eagles (Giesy et al. 1995, Bowerman et al. 1995, Elliott et al. 1996). Elliott et al. (1996) conducted a study of the effects of TEQ using incubated bald eagle eggs taken from nests in British Columbia. Based on the Elliott et al. (1996) study, the NOAEL is 135 ng/kg ww TEQ in eggs and the corresponding LOAEL is 400 ng/kg ww TEQ in eggs.

Elliott et al. (1989) found that a TEQ of 230 ng/kg ww in the eggs of great blue herons caused reduced reproductive success. The same study found TEQ concentrations of 11, 14, 34, 64, and 79 ng/kg ww in eggs to have no effect on hatching success. Based on these results, the NOAEL for TEQ in American bittern eggs is 79 ng/kg ww, and the LOAEL is 230 ng/kg ww TEQ.

For myotis, the dose-response curve for TEQ was derived using the results of Khera and Ruddick (1973) and Sparschu et al. (1971). Khera and Ruddick (1973) treated pregnant Wistar rats with several doses of TEQ and reported a dose-related decrease in live fetuses; 100% embryonic lethality was reported when animals were exposed to a dose of 4,000 ng TEQ/kg bw/d. Sparschu et al. (1971) made similar observations in Sprague-Dawley rats fed several doses of TCDD, finding that the number of viable fetuses decreased and the total number of resorptions increased dose-dependently, starting at 125 ng TEQ/kg bw/d.

11.3.3 Effects Metrics for Characterizing Risk

A summary of the decision criteria used to derive effects metrics is provided in the text box. In this ERA, data were available to derive dose-response curves using surrogate mammals for the small-footed myotis. Toxicity threshold ranges were developed for bald eagles and American bittern. Details on the decision criteria used in selecting effects metrics is provided in Section 6.6 of the ERA.

Hierarchical Decision Criteria for Derivation of Effects Metric

1. Have single-study bioassays with five or more treatments been conducted on the receptor of interest or a reasonable surrogate? If yes, estimate the concentration- or dose-response. If not, go to 2.
2. Are multiple bioassays with similar protocols, exposure scenarios and effects metrics available that, when combined, have five or more treatments for the receptor of interest or a reasonable surrogate? If yes, estimate the dose-response relationship as in 1. If not, go to 3.
3. Have bioassays with less than five treatments been conducted on the receptor of interest or a reasonable surrogate? If yes, conduct or report results of hypothesis testing to determine the NOAEL and LOAEL. If not, go to 4.
4. Are sufficient data available from field studies and monitoring programs to estimate concentrations or doses of the COC that are consistently protective or associated with adverse effects? If yes, develop field-based effects metrics. If not, go to 5.
5. Derive a range where the threshold for the receptor of interest is expected to occur. Because information on the sensitivity of the receptor of interest is lacking, it is difficult to derive a threshold that is biased neither high nor low. If bioassay data are available for several other species, however, calculate a threshold for each to determine a threshold range that spans sensitive and tolerant species. That range is likely to include the threshold for the receptor of interest.

11.3.3.1 Effects of tPCBs to Bald Eagle

A threshold value of 20 mg/kg ww in bald eagle eggs was suggested in the recent assessment of the Fox River/Green Bay system (Stratus 1999). That value is consistent with other raptor studies that suggest tPCBs have higher egg thresholds for reproductive effects than does DDE (Helander et al. 1982; Peakall et al. 1990; Nobel and Elliott 1990). Therefore, the field-based threshold selected for tPCB in bald eagle eggs was 20 mg/kg ww.

11.3.3.2 Effects of TEQ to Bald Eagle

Using the Elliott et al. (1996) study, the toxicity threshold for TEQ in eggs is the NOAEL of 135 ng/kg ww.

11.3.3.3 Effects of tPCBs to American Bittern

The threshold for toxic effects to herons is in the range of 4 to >6 mg/kg ww PCBs in eggs. American bitterns can be reasonably represented by black-crowned night and great blue herons. For this assessment, a NOAEL of 4.9 mg/kg ww PCBs in eggs was selected for American bitterns.

11.3.3.4 Effects of TEQ to American Bittern

Elliott et al. (1989) found that a TEQ of 230 ng/kg ww in the eggs of great blue herons caused reduced reproductive success. The same study found TEQ concentrations of 11, 14, 34, 64, and 79 ng/kg ww in eggs to have no effect on hatching success. The NOAEL selected for TEQ in eggs was 79 ng/kg ww and the LOAEL was 230 ng/kg ww TEQ in eggs.

11.3.3.5 Effects of tPCBs to Small-Footed Myotis

The Spencer (1982) study was used for the derivation of a dose-response curve based on mortality at birth. Figure 11.3-1 presents the dose-response curve for mortality of rats at birth. The dose-response curve indicates that 10% and 20% declines in mortality at birth would be expected at doses of 3.05 and 5.37 mg/kg bw/d, respectively.

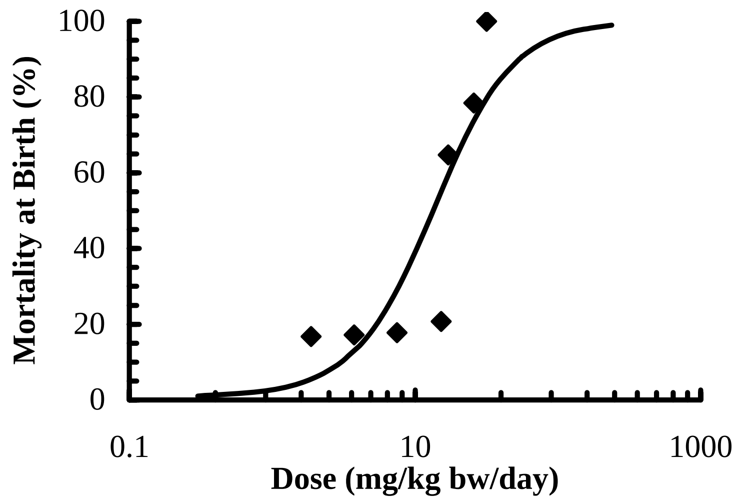


Figure 11.3-1 Dose-Response Curve for Effects of tPCBs on Mortality at Birth of Rats

11.3.3.6 Effects of TEQ to Small-Footed Myotis

Because of the similarity of the protocols, the Khera and Ruddick (1973) and Sparschu et al. (1971) studies were combined for the derivation of a dose-response curve based on reproductive effects. Figure 11.3-2 presents the dose-response curve for reproductive fecundity of rats exposed to TEQ. The dose-response curve indicates that 10% and 20% declines in reproductive fecundity would be expected at doses of 156 and 330 ng/kg bw/d TEQ, respectively.

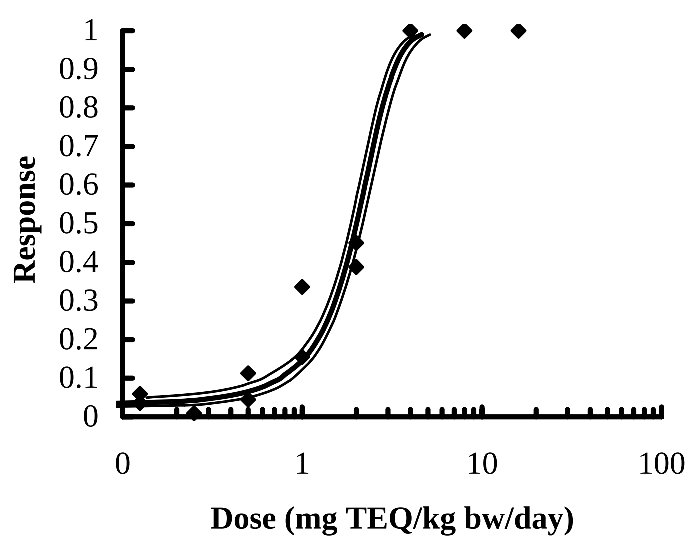


Figure 11.3-2 Dose-Response Curve for Effects of TEQ on Mortality at Birth of Rats

11.4 RISK CHARACTERIZATION

This section characterizes risk to T&E species exposed to tPCBs and TEQ in the PSA of the Housatonic River. The risk characterization discusses two potential lines of evidence, field surveys and modeled exposure and effects, to determine potential ecological risks to T&E species. The risk questions and the two potential lines of evidence are summarized in the text box.

> ***Risk Questions***
>
> - Are the concentrations of tPCBs and TEQ present in the prey of T&E species sufficient to cause adverse effects to individuals inhabiting the PSA of the Housatonic River?
> - If so, how severe are the risks and what are their potential consequences?
>
> ***Lines of Evidence***
>
> - Use of qualitative field surveys (not considered in the weight-of-evidence analysis).
> - Probabilistic exposure and effects modeling.

11.4.1 Field Surveys

Details of field surveys conducted in support of the ERA are provided in Appendix A. This section provides a brief summary of that information.

The avian community in the PSA was studied over a 4-year period, from 1998 to 2001. Surveys were conducted to record presence, abundance, and habitat usage for each major group of birds. Bald eagles were not observed during raptor surveys in the PSA or any of the three reference areas. However, incidental bald eagle observations were made in the PSA (primarily in the vicinity of Woods Pond) and at the Threemile Pond reference area. The PSA provides suitable nesting and foraging habitat for bald eagles. In the mid 1990s, a pair of bald eagles constructed a nest at Woods Pond (T. Gulo, MassWildlife, personal communication 2001). The nest was reportedly destroyed during an April snowstorm and the pair did not attempt to re-nest. American bitterns were not observed during marsh bird surveys in the PSA, and no marsh bird surveys were conducted outside of the PSA. Incidental observations of American bitterns occurred in the PSA and at Washington Mountain Lake during the breeding season, and one individual was heard calling in the PSA, indicating intent to breed in the area.

Bat surveys were conducted to determine presence in the PSA by recording their echolocation calls. The majority of the recorded calls were likely little brown bat; however, a small number of the calls had parameters that suggested small-footed myotis rather than little brown bats or Indiana bats. Small-footed myotis observations in the PSA have not been confirmed, and no bat surveys were conducted in reference areas. Suitable summer habitat for small-footed myotis is present in and adjacent to the study area, and it is likely that the species occurs there.

Any differences in population structure between PSA and reference locations cannot be evaluated due to the overall low population size of T&E species, few numbers of sightings, and qualitative study design. The number of observations for these species by definition is expected to be low and the lack of observations in one location does not necessarily reflect the suitability of that habitat or the absence of an individual.

11.4.2 Comparison of Estimated Exposures to Laboratory-Derived Effects Doses

For the bald eagle, exposure was assessed for the southern PSA, while exposure of American bittern was estimated for each reach, and exposure to small-footed myotis was estimated for the entire PSA.

Risks to eagles were estimated in two ways: by comparing modeled TDIs for adult eagles to a toxicity threshold developed using a surrogate species, the American kestrel; and by comparing modeled egg tissue concentrations to a toxicity threshold based on bald eagle field studies. Because a surrogate species was used to estimate the risk for adult eagles (resulting in higher uncertainty) and because effects to eggs have direct consequences for T&E species, risk characterization for eagles from exposure to tPCBs and TEQs is based on modeled egg tissue concentrations.

Risks to American bitterns were also estimated in two ways: by comparing modeled TDIs for adult bitterns to a toxicity threshold developed using surrogate species, white leghorn chickens and pheasants (both sensitive) and the American kestrel (tolerant); and by comparing modeled egg tissue concentrations to a toxicity threshold, which was based on a great blue heron field study. Because non-heron surrogate species were used to estimate the risk for adult bitterns (resulting in higher uncertainty) and because effects to eggs have direct consequences for T&E species, overall risk to bitterns from tPCBs and TEQs was based on modeled egg tissue concentrations.

For each receptor-COC combination, a category of low, intermediate, or high risk was assigned using the guidance in the text box following integration of the exposure and effects distributions. This exercise was done separately for the results of the Monte Carlo analyses and each of the lower and upper bounds from the probability bounds analyses. The "risk category" refers to the level of risk based on the results of the Monte Carlo analyses. The "risk range" refers to the levels of risk based on the results of the probability bounds analyses. The toxicity thresholds or the 10% and 20% effects doses for each species and COC are presented in Section 11.3.

Guidance for Determining Level of Risk to Bald Eagle

- If the probability of exceeding the toxicity threshold was less than 20%, the risk to bald eagle was considered to be low.
- If the probability of exceeding the toxicity threshold was greater than 50%, the risk to bald eagle was considered to be high.
- All other outcomes are considered to have intermediate risk.

Guidance for Determining Level of Risk to American Bittern

- If the probability of exceeding the lower toxicity threshold was less than 20%, the risk to American bittern was considered to be low; or if a bittern would not be present in the PSA for a long enough time prior to breeding to accumulate sufficient levels of PCBs or TEQ to exceed the threshold in eggs, then the risk was considered to be low.
- If the probability of exceeding the upper toxicity threshold was greater than 20%, the risk to American bittern was considered to be high; or if a bittern would be present in the PSA for a long enough time prior to breeding to accumulate sufficient levels of PCBs or TEQ to exceed the threshold in eggs, then the risk was considered to be high.
- All other outcomes are considered to have intermediate risk; or if a bittern would be present in the PSA for a period of time where it would accumulate levels of PCBs and TEQ between those known to cause low to high risk, then the risk was considered to be intermediate.

Guidance for Determining Level of Risk to Small-Footed Myotis

- If the probability of 10% or greater effect is less than 20%, then the risk to small-footed myotis is low.
- If the probability of 20% or greater effect is greater than 50%, then the risk to small-footed myotis is high.
- All other outcomes are considered to have intermediate risk.

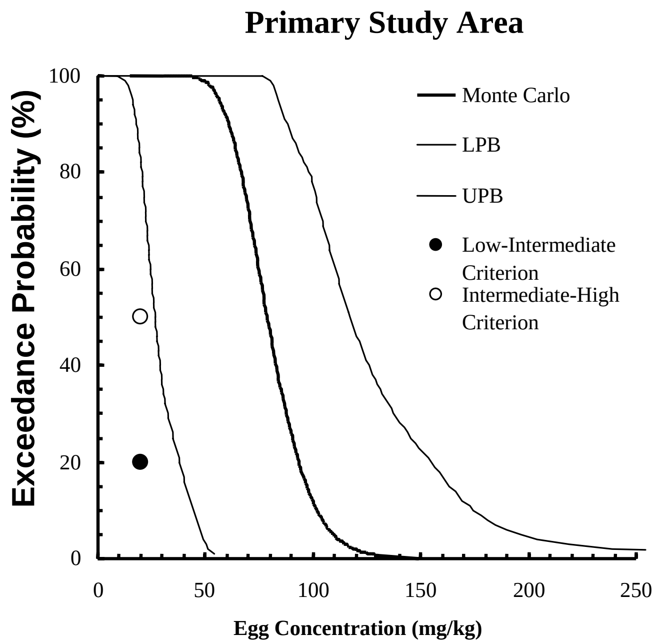
The results of the risk characterization are summarized in Table 11.4-1. Figures 11.4-1 through 11.4-2 are risk curves for bald eagles exposed to tPCBs and TEQ in the PSA. Figures 11.4-3 through 11.4-7 show American bittern exposed to tPCBs and TEQ in the PSA, and Figures 11.4-8 and 11.4-9 show small-footed myotis exposed to tPCBs and TEQ in the PSA.

Table 11.4-1

Summary of Qualitative Risk Statements for T&E Species from the Housatonic River Study Area

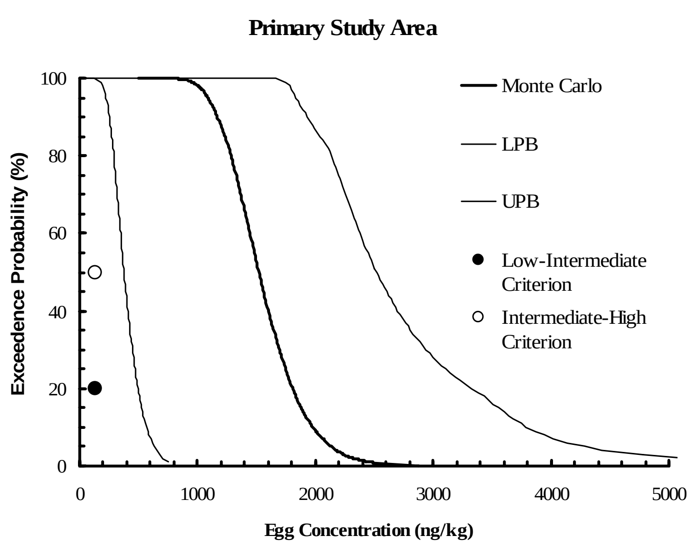
Bird / Location	Qualitative Risk Statements			
	tPCBs		TEQ	
	Risk Category[a]	Risk Range[b]	Risk Category[a]	Risk Range[b,c]
Bald Eagle				
Southern PSA	High	High	High	High
American Bittern				
Reach 5A	High	High	High	High
Reach 5B	High	High	High	High
Reach 5C	High	High	High	High
Reach 5D and 6	High	High	High	High
Small-Footed Myotis				
Reaches 5 and 6	High	Intermediate - High	High	Low - High

[a]Risk category is the risk level based on First Order Monte Carlo (FOMC).
[b]Risk range is the range of risk encompassed by the upper and lower probability bounds (UPB and LPB).
[c]P-bounds were not calculated for bittern eggs; risk is based on estimates for exceeding toxicity thresholds in eggs.



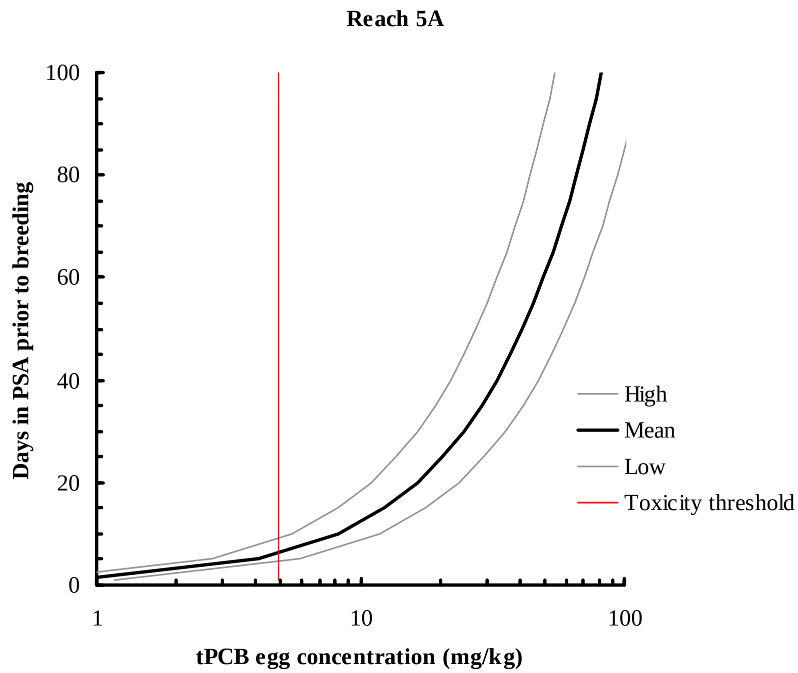
Note: Egg concentrations are wet weight.

Figure 11.4-1 Risk for Bald Eagle Eggs Exposed to tPCBs in the Housatonic River Primary Study Area



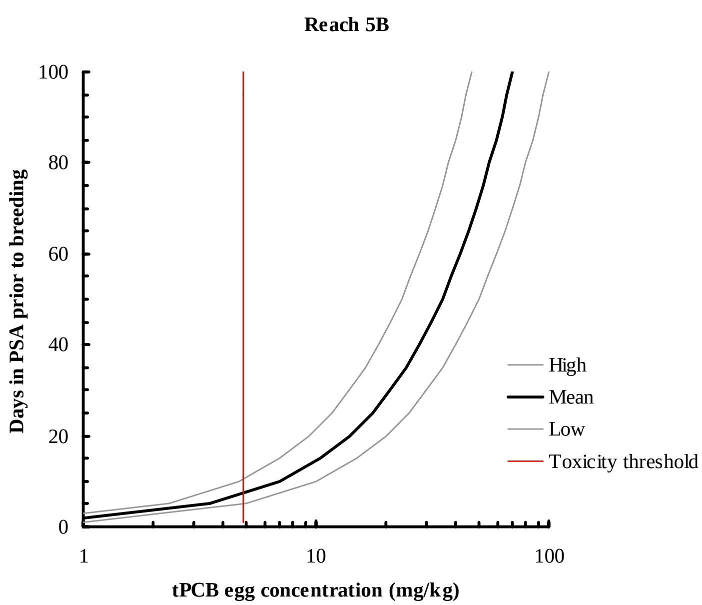
Note: Egg concentrations are wet weight.

Figure 11.4-2 Risk for Bald Eagle Eggs Exposed to TEQ in the Housatonic River Primary Study Area



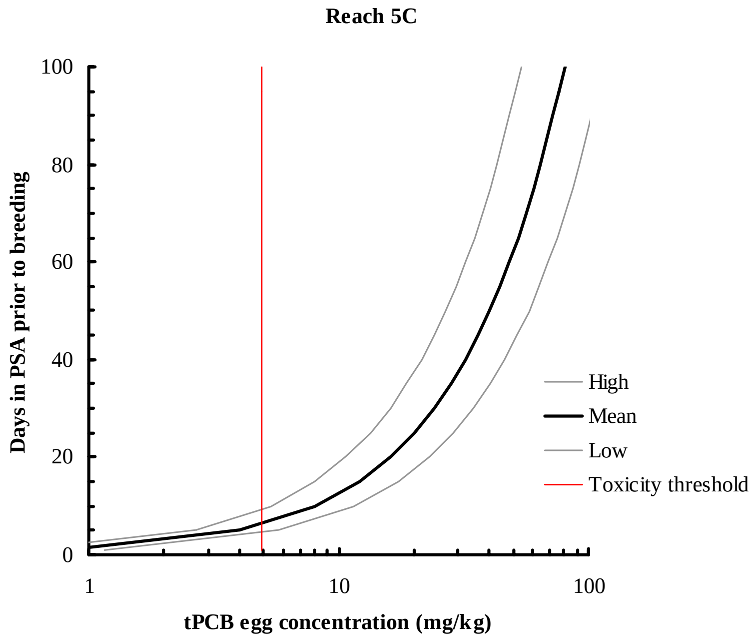
Note: Egg concentrations are wet weight.

Figure 11.4-3 Risk for American Bittern Eggs Exposed to tPCBs in Reach 5A of the Housatonic River Primary Study Area



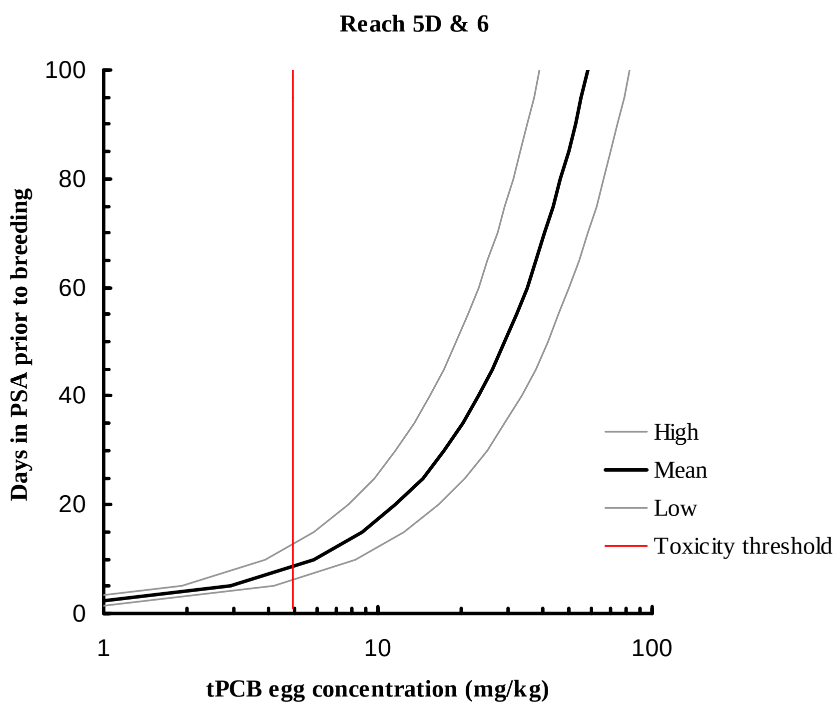
Note: Egg concentrations are wet weight.

Figure 11.4-4 Risk for American Bittern Eggs Exposed to tPCBs in Reach 5B of the Housatonic River Primary Study Area



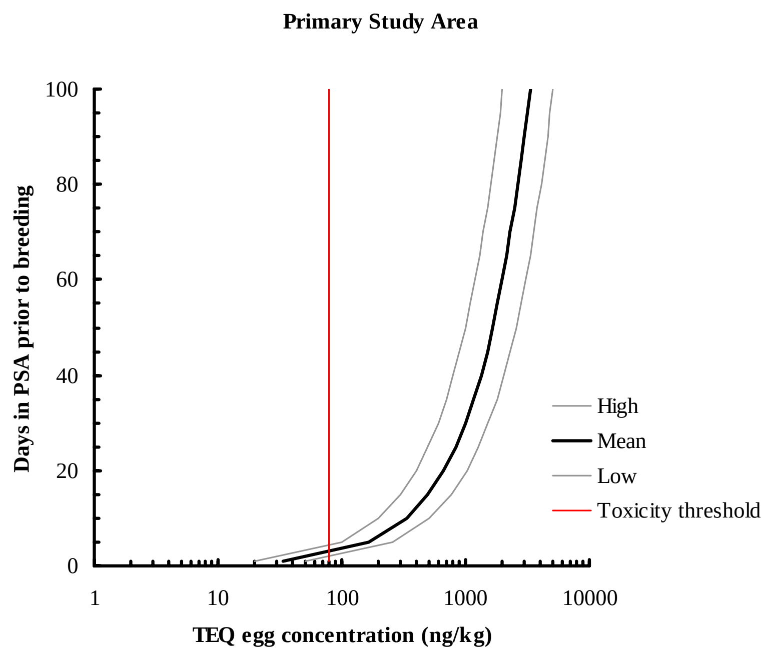
Note: Egg concentrations are wet weight.

Figure 11.4-5 Risk for American Bittern Eggs Exposed to tPCBs in Reach 5C of the Housatonic River Primary Study Area



Note: Egg concentrations are wet weight.

Figure 11.4-6 Risk for American Bittern Eggs Exposed to tPCBs in Reaches 5D and 6 of the Housatonic River Primary Study Area



Note: Egg concentrations are wet weight.

Figure 11.4-7 Risk for American Bittern Eggs Exposed to TEQ in the Housatonic River Primary Study Area

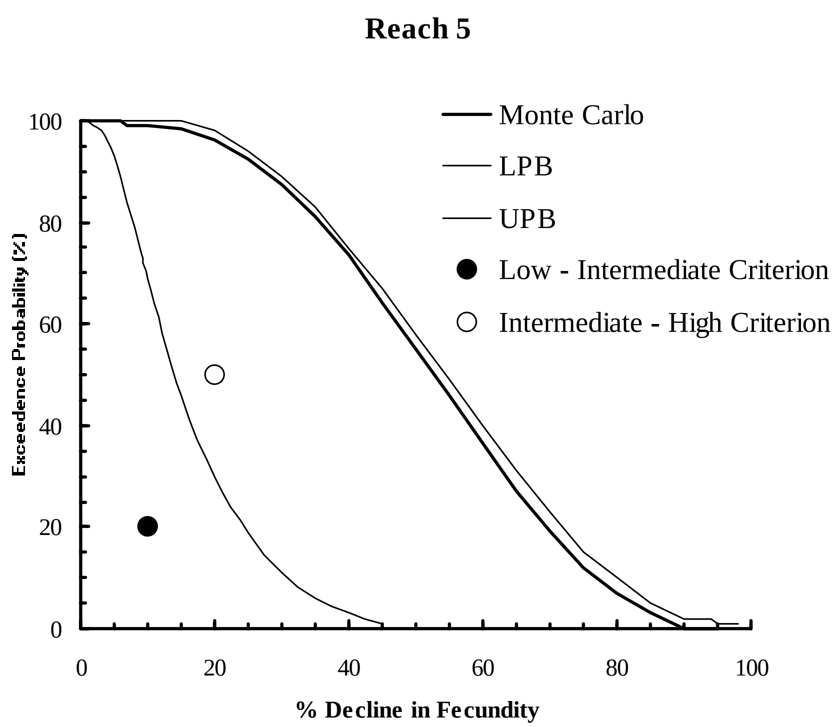


Figure 11.4-8 Risk Curves for Small-Footed Myotis Exposed to tPCBs in Reach 5 of the Housatonic River Primary Study Area

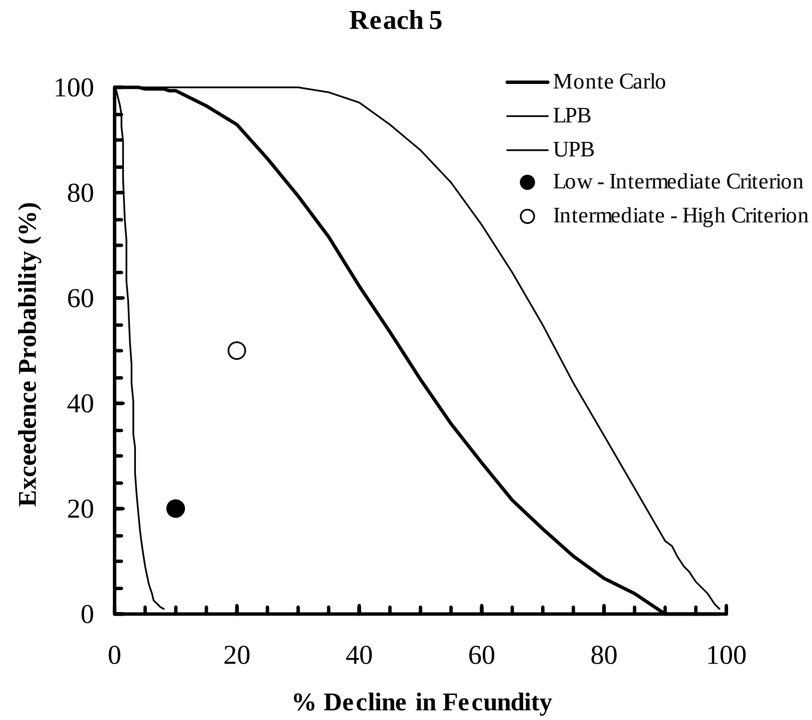


Figure 11.4-9 Risk Curves for Small-Footed Myotis Exposed to TEQ in Reach 5 of the Housatonic River Primary Study Area

The results of the risk characterization showed that the highest risk to T&E species is to bald eagles and American bitterns from exposure to tPCBs and TEQ. The analysis for bald eagles associated with exposure to tPCBs downstream of Woods Pond indicated that bald eagles would only potentially be at risk in Reach 8 (Rising Pond). The risk to bald eagles nesting and wintering downstream of the PSA is low. The risk category for small-footed myotis was high for both tPCB and TEQ. The risk range for small-footed myotis, as determined by the probability bounds analysis, ranged from intermediate to high for tPCBs and low to high for TEQ.

11.4.3 Weight-of-Evidence Analysis

A weight-of-evidence analysis was used to evaluate the lines of evidence described in the preceding sections for T&E species. The three-phase approach of Menzie et al. (1996) and the Massachusetts Weight-of-Evidence Workgroup was applied for this purpose, in which weight-of-evidence was reflected in the following three characteristics: (a) the weight assigned to each measurement endpoint, (b) the magnitude of response observed in the measurement endpoint, and (c) the concurrence among outcomes of the multiple measurement endpoints. As noted previously, field surveys were qualitative and therefore not used in this analysis.

The rationale for weighting of measurement endpoints is provided in Appendix K, along with a discussion of attributes considered in the weight-of-evidence. A summary of how attributes were weighted for the bald eagle, American bittern, and small-footed myotis lines of evidence is provided in Tables 11.4-2 to 11.4-4. For both tPCBs and TEQ, the modeled exposure and effects lines of evidence were given a moderate/high value for bald eagles. American bitterns and small-footed myotis each received a moderate value.

Table 11.4-2

Weighting of Measurement Endpoints for Bald Eagle Weight-of-Evidence Evaluation

Attributes	Modeled Exposure and Effects for Bald Eagles Exposed to tPCBs and TEQ
I. Relationship Between Measurement and Assessment Endpoints	
1. Degree of Association	M/H (tPCBs and TEQ)
2. Stressor/Response	H (tPCBs and TEQ)
3. Utility of Measure	M/H (tPCBs and TEQ)
II. Data Quality	
4. Data Quality	M/H (tPCBs and TEQ)
III. Study Design	
5. Site Specificity	M (tPCBs and TEQ)
6. Sensitivity	M/H (tPCBs and TEQ)
7. Spatial Representativeness	M (tPCBs and TEQ)
8. Temporal Representativeness	M (tPCBs and TEQ)
9. Quantitative Measure	M/H (tPCBs and TEQ)
10. Standard Method	M/H (tPCBs and TEQ)
Overall Endpoint Value	**M/H (tPCBs and TEQ)**

H = High, M/H = Moderate to High, M = Moderate, M/L = Moderate to Low, L = Low

Table 11.4-3

Weighting of Measurement Endpoints for American Bittern Weight-of-Evidence Evaluation

Attributes	Modeled Exposure and Effects for American Bitterns Exposed to tPCBs and TEQ
I. Relationship Between Measurement and Assessment Endpoints	
1. Degree of Association	M (tPCBs and TEQ)
2. Stressor/Response	M (tPCBs and TEQ)
3. Utility of Measure	M (tPCBs and TEQ)
II. Data Quality	
4. Data Quality	M/H (tPCBs and TEQ)
III. Study Design	
5. Site Specificity	L/M (tPCBs and TEQ)
6. Sensitivity	M (tPCBs and TEQ)
7. Spatial Representativeness	M (tPCBs and TEQ)
8. Temporal Representativeness	M/H (tPCBs and TEQ)
9. Quantitative Measure	M/H (tPCBs and TEQ)
10. Standard Method	M/H (tPCBs and TEQ)
Overall Endpoint Value	**M (tPCBs and TEQ)**

H = High, M/H = Moderate to High, M = Moderate, M/L = Moderate to Low, L = Low

Table 11.4-4

Weighting of Measurement Endpoints for Small-Footed Myotis Weight-of-Evidence Evaluation

Attributes	Modeled Exposure and Effects for Small-Footed Myotis Exposed to tPCBs and TEQ
I. Relationship Between Measurement and Assessment Endpoints	
1. Degree of Association	M (tPCBs and TEQ)
2. Stressor/Response	M (tPCBs and TEQ)
3. Utility of Measure	M (tPCBs and TEQ)
II. Data Quality	
4. Data Quality	M/H (tPCBs and TEQ)
III. Study Design	
5. Site Specificity	L/M (tPCBs and TEQ)
6. Sensitivity	M (tPCBs and TEQ)
7. Spatial Representativeness	M (tPCBs and TEQ)
8. Temporal Representativeness	M/H (tPCBs and TEQ)
9. Quantitative Measure	H (tPCBs and TEQ)
10. Standard Method	M/H (tPCBs and TEQ)
Overall Endpoint Value	**M (tPCBs and TEQ)**

H = High, M/H = Moderate to High, M = Moderate, M/L = Moderate to Low, L = Low

The weighting, evidence of harm, and magnitudes of responses were combined in a matrix format and are presented in Tables 11.4-5 and 11.4-6.

Table 11.4-5

Evidence of Harm and Magnitude of Effects for T&E Species Exposed to tPCBs in the Housatonic River PSA

Measurement Endpoints	Weighting Value (High, Moderate, Low)	Evidence of Harm (Yes, No, Undetermined)	Magnitude (High, Intermediate, Low)
Modeled exposure and effects, Bald Eagle	Moderate/High	Yes	High
Modeled exposure and effects, American Bittern	Moderate	Yes	High
Modeled exposure and effects, Small-Footed Myotis	Moderate	Yes	Intermediate

Table 11.4-6

Evidence of Harm and Magnitude of Effects for T&E Species Exposed to TEQ in the Housatonic River PSA

Measurement Endpoints	Weighting Value (High, Moderate, Low)	Evidence of Harm (Yes, No, Undetermined)	Magnitude (High, Intermediate, Low)
Modeled exposure and effects, Bald Eagle	Moderate/High	Yes	High
Modeled exposure and effects, American Bittern	Moderate	Yes	High
Modeled exposure and effects, Small-Footed Myotis	Moderate	Yes	Intermediate

A graphical method was used for displaying concurrence among measurement. Tables 11.4-7 and 11.4-8 depict the outcome for T&E species exposed to tPCBs and TEQ, respectively. The field survey line of evidence was not included as it is inconclusive.

Table 11.4-7

Risk Analysis Summary for T&E Species Exposed to tPCBs in the Housatonic River PSA

Assessment Endpoint: Survival, growth, and reproduction of T&E species

Harm/Magnitude	Weighting Factors (increasing confidence of weight)				
	Low	**Low/ Moderate**	**Moderate**	**Moderate/ High**	**High**
Yes/High			AB	BE	
Yes/Intermediate			SFM		
Yes/Low					
Undetermined					
No					

BE = bald eagle
AB = American bittern
SFM = small-footed myotis

Table 11.4-8

Risk Analysis Summary for T&E Species Exposed to TEQ in the Housatonic River PSA

Assessment Endpoint: Survival, growth, and reproduction of T&E species

Harm/Magnitude	Weighting Factors (increasing confidence of weight)				
	Low	**Low/Moderate**	**Moderate**	**Moderate/High**	**High**
Yes/High			AB	BE	
Yes/Intermediate			SFM		
Yes/Low					
Undetermined					
No					

BE = bald eagle
AB = American bittern
SFM = small-footed myotis

11.4.4 Sources of Uncertainty

Some of the major sources of uncertainty associated with the assessment of risks of tPCBs and TEQ to T&E species are briefly summarized below. A more complete list is presented in Appendix K.

- Use of allometric equations to estimate free metabolic rate and food intake rate.
- Limited samples of tissue contaminant concentrations for some prey items.
- Co-elution of some PCB congeners, requiring estimation of coplanar congener concentrations.
- Simplifying assumption that eagles do not metabolize PCBs.
- Estimation of adult body burden assuming that a breeding adult would arrive in the PSA with no tPCBs in the body.
- Lack of, or limited, toxicity studies involving the representative species.
- Lack of direct feeding studies for T&E species.

11.4.5 Conclusions

11.4.5.1 Risks in the PSA

The weight-of-evidence analysis indicates that T&E species such as bald eagles, American bitterns, and small-footed myotis are at risk in the PSA as a result of exposure to tPCBs and TEQ. The risks for bald eagles and American bitterns exposed to tPCBs are high. The risk for small-footed myotis exposed to tPCBs and TEQ are intermediate.

Other T&E species that occur in the area include one mussel (triangle floater); three dragonflies (riffle snaketail, zebra clubtail, and arrow clubtail); a turtle (wood turtle); three salamanders (Jefferson salamander, four-toed salamander, and northern spring salamander); three hawks (northern harrier, sharp-shinned hawk, and Cooper's hawk); two warblers (northern parula and blackpoll warbler); a wading bird (common moorhen); and a shrew (northern water shrew). Some of these species were assessed in other appendices, and the risks were compared to other, more appropriate assessment endpoints (e.g., salamanders assessed in Appendix E, Amphibians).

A qualitative analysis was conducted to compare exposure of representative species and other similar species to tPCBs and TEQ. The major factors that influence exposure to tPCBs and TEQ include the following:

- Foraging behavior and dietary composition.
- Foraging and home range size.
- Species body weight and other life history characteristics.

Surrogate effects data used to estimate effects to bald eagles were also used to estimate risk for other piscivorous raptors, data for American bittern were used for other wading birds, and data from small-footed myotis for other bat species.

Results are provided in the following text box.

ERA Results for Other Piscivorous Raptors, Wading Birds, and Bats Living in the PSA

The other piscivorous raptor that occurs in the PSA is the osprey. Risk to osprey is characterized in Appendix H.

Other piscivorous wading bird species that could occur in the PSA include the least bittern, green heron, great blue heron, king rail, least rail, sora, and pied-billed grebe. A qualitative analysis of risk to these species indicates that the great blue heron and king rail are expected to have a similar level of risk compared to the American bittern.

The wading birds that have similar diets but are smaller and have higher metabolisms—such as least bittern, green heron, Virginia rail, and pied-billed grebe—are expected to have a higher level of risk than the American bittern. Wading birds that consume plant material, such as the sora, are expected to have low levels of risk.

Other bat species, especially those in the myotis family (little brown bat, Indiana bat, and northern myotis) are expected to have a similar level of risk as the small-footed myotis.

11.4.5.2 Development of Maximum Acceptable Threshold Concentration (MATC)

The MATC of 30.41 mg/kg tPCBs in fish (whole body, wet weight) was developed as the concentration at which bald eagle TDI would exceed the toxicity threshold for eggs (20 mg/kg ww tPCB). The TDI was calculated assuming that overwintering eagles would consume 83.4% fish and 16.1% waterfowl (Stalmaster and Plettner 1992). The waterfowl concentration was assumed to be zero, as waterfowl overwintering on the Housatonic River are likely to have migrated there from northern locations outside the study area.

11.4.5.3 Risk Estimates Downstream of Woods Pond

Risks to bald eagles due to contaminants in the river and floodplain downstream of the PSA were also assessed. The data indicate that contamination in these media declines substantially downstream of the PSA.

11.4.5.3.1 Risk for Bald Eagles Wintering Downstream of Woods Pond

The risk for bald eagles from exposure to tPCBs downstream of Woods Pond was assessed by comparing concentrations of tPCBs in prey fish in Reaches 7 to 16 to the MATC of 30.41 mg/kg ww tPCBs in fish developed specifically for bald eagles. The results of the analysis that indicate that bald eagles would be at risk only in Reach 8 (Rising Pond) are presented in Figure K.4-5. This conclusion is conservative, in that it assumes bald eagles would consume fish only from Rising Pond. However, this is unlikely because Rising Pond is considerably smaller than a typical bald eagle foraging area. It is likely that eagles nesting at Rising Pond would be exposed through a foraging area including but greater than Rising Pond. The concentrations in fish tissue in the adjacent subreaches of river are not known, but are expected to be elevated.

11.4.5.3.2 Risk for Bald Eagles Breeding Downstream of Woods Pond

Figure K.4-5 presents the assessment of risk to bald eagles exposed to tPCBs downstream of Woods Pond. Bald eagles are known to breed downstream of Woods Pond. In particular, one bald eagle pair nested and raised one chick in Reach 15, just south of Interstate 84, in 2001.

Risk from exposure to tPCBs was estimated for bald eagles nesting at this location. Bald eagles have a linear (riverine) foraging distance of 1.9 to 4.3 miles (3.1 to 6.9 km) (Craig et al. 1988). Therefore, bald eagles nesting near Interstate 84 could potentially be foraging in Reach 15 (Lake Zoar) and the southern section of Reach 14 (Lake Lillinonah). The estimated low, moderate, and high tPCB intake rates for bald eagles foraging in this area averaged 0.022 mg/kg bw/d, 0.025 mg/kg bw/d, and 0.243 mg/kg bw/d, respectively. These values fall below the lower toxicity threshold of 0.7 mg/kg bw/d.

Risks from TEQ to adult bald eagles in the PSA were intermediate. Because TEQ concentrations in downstream prey species are reduced to the same degree as PCB concentrations, it is assumed that risk from TEQ to bald eagles breeding downstream would be low.

11.5 REFERENCES

Bowerman, W.W., J.P. Giesy, D.A. Best, and V.J. Kramer. 1995. A review of factors affecting productivity of bald eagles in the Great Lakes region: Implications for recovery. *Environmental Health Perspectives* 103(Suppl 4):51-59.

Buehler, D.A. 2000. Bald Eagle (*Haliaeetus leucocephalus*). pp. 1–39. In Poole, A. and F. Gill (ed.) *The Birds of North America*. Vol. 506. The Birds of North America, Inc., Philadelphia, PA, USA.

Canadian Wildlife Service. 2000. Hinterland Who's Who: Bald Eagle. URL http://www.cws-scf.ec.gc.ca/hww-fap/bald/bald.html (accessed 13 May 2002).

Clark, D.R., Jr. 1978. Uptake of dietary tPCB by pregnant big brown bats and their fetuses. *Bulletin of Environmental Contamination and Toxicology* 19:707–714.

Clark, D.R., Jr., and C.J. Stafford. 1981. Effects of DDE and PCB (Aroclor 1260) on experimentally poisoned female little brown bats (*Myotis lucifugus*): Lethal brain concentrations. *Journal of Toxicology and Environmental Health* 7:925–934.

Clark, D.R., Jr., and R.M. Prouty. 1976. Organochlorine residues in three bat species from four localities in Maryland and West Virginia, 1973. *Pesticides Monitoring Journal* 10:44-53.

Clark, D.R., Jr., and T.G. Lamont. 1976a. Organochlorine residues in females and nursing young of the big brown bat. *Bulletin of Environmental Contamination and Toxicology* 15(1):1-8.

Clark, D.R., Jr., and T.G. Lamont. 1976b. Organochlorine residues and reproduction in the big brown bat. *Journal of Wildlife Management* 40(2):249-254.

Cottam, C. and F.M. Uhler. 1945. Birds in Relation to Fishes. U.S. Fish and Wildlife Service Leaflet, No. 272.

Craig, R.J., E.S. Mitchell, and J.E. Mitchell. 1988. Time and energy budgets of bald eagles wintering along the Connecticut River. *Journal of Field Ornithology* 59:22-32.

Custer, C.M. 2002. Exposure and effects of chemical contaminants on tree swallows nesting along the Housatonic River, Berkshire Co., Massachusetts, 1998-2000. Final Report to U.S. Environmental Protection Agency. USGS, Upper Midwest Environmental Sciences Center, La Crosse, WI.

Donaldson, G.M., J.L. Shutt, and P. Hunter. 1999. Organochlorine contamination in bald eagle eggs and nestlings from the Canadian Great Lakes. A*rchives of Environmental Contamination and Toxicology* 36:70-80.

Dunning, J.B., Jr. (ed.) 1992. CRC *Handbook of Avian Body Mass*. CRC Press, Boca Raton, FL, USA.

Elliott, J.E., R.W. Butler, R.J. Norstrom, and P.E. Whitehead. 1989. Environmental contaminants and reproductive success of great blue herons in British Columbia 1986-87. *Environmental Pollution* 59:91-114.

Elliott, J.E., R.J. Norstrom, A. Lorenzen, L.E. Hart, H. Philibert, S.W. Kennedy, J.J. Stegeman, G.D. Bellward, and K.M. Cheng. 1996. Biological effects of polychlorinated dibenzo-p-dioxins, dibenzofurans, and biphenyls in bald eagle (*Haliaeetus leucocephalus*) chicks. *Environmental Toxicology and Chemistry* 15:782-793.

EPA (U.S. Environmental Protection Agency). 1993. *Wildlife Exposure Factors Handbook.* Volume 1 of 2. EPA/600/R-93/187a. Office of Research and Development, Washington, DC, USA.

Fernie, K.J., J.E. Smits, G.R. Bortolotti, and D.M. Bird. 2001a. In ovo exposure to polychlorinated biphenyls: reproductive effects on second-generation American kestrels. A*rchives of Environmental Contamination and Toxicology* 40:544-550.

Fernie, K.J., J.E. Smits, G.R. Bortolotti, and D.M. Bird. 2001b. Reproductive success of American kestrels exposed to dietary polychlorinated biphenyls. *Environmental Toxicology and Chemistry* 20:776-781.

Gibbs, J.P., S. Melvin, and F.A. Reid. 1992. American Bittern (*Botaurus lentiginosus*). In *The Birds of North America*, No. 18. A. Poole, P. Stettenheim, and F. Gill (eds). The Birds of North America, Inc. Philadelphia, PA, USA.

Giesy, J.P., W.W. Bowerman, M.A. Mora, D.A. Verbrugge, R.A. Othoudt, J.L. Newsted, C.L. Summer, R.J. Aulerich, S.J. Bursian, J.P. Ludwig, G.A. Dawson, T.J. Kubiak, D.A. Best, and D.E. Tillitt. 1995. Contaminants of fishes from Great Lakes-influenced sections and above dams of three Michigan rivers: III. Implications for health of bald eagles. A*rchives of Environmental Contamination and Toxicology* 29(3):309-321.

Gulo, T. 2001. Massachusetts Division of Fisheries and Wildlife. Personal Communication to Woodlot Alternatives, Inc.

Helander, B., M. Olsson, and L. Reuterhard. 1982. Residue levels of organochlorine and mercury compounds in unhatched eggs and their relationship to breeding success in white-tailed sea eagles in Sweden. *Holarctic Ecology* 5:349-366.

Hoffman, D.J., B.A. Rattner, C.M. Bunck, A. Krynitsky, H.M. Ohlendorf, and R.W. Lowe. 1986. Association between PCBs and lower embryonic weight in black-crowned night herons in San Francisco Bay. *Journal of Toxicology and Environmental Health* 19:383-391.

Hoffman, D.J., G.J. Smith, and B.A. Rattner. 1993. Biomarkers of contaminant exposure in common terns and black-crowned night herons in the Great Lakes. *Environmental Toxicology and Chemistry* 12:1095-1103.

Khera, K.S. and J.A. Ruddick. 1973. Polychlorodibenzo-p-dioxins: Perinatal effects and the dominant lethal test in Wistar rats. *Toxicology* 120:70-84.

Kurta, A. 1995. *Mammals of the Great Lakes Region*. The University of Michigan Press. Ann Arbor, MI. 392 pp.

Laporte, P. 1982. Organochlorine residues and eggshell measurements of great blue heron eggs from Quebec. *Colonial Waterbirds* 5:95-103.

Menzie, C., M.H. Henning, J. Cura, K. Finkelstein, J. Gentile, J. Maughan, D. Mitchell, S. Petron, B. Potocki, S. Svirsky, and P. Tyler. 1996. Special report of the Massachusetts Weight-of-Evidence Workgroup: A weight of evidence approach for evaluating ecological risks. *Human and Ecological Risk Assessment* 2(2):277-304.

Nagy, K.A., I.A. Girard, and T.K. Brown. 1999. Energetics of free-ranging mammals, reptiles, and birds. A*nnu. Rev. Nutr*. 19:247-277.

Nagy, K.A. 1987. Free metabolic rate and food requirement scaling in mammals and birds. *Ecological Monographs* 57:111-128.

Nobel, D.G. and J.E. Elliott. 1990. Levels of contaminants in Canadian raptors, 1966 to 1988; effects and temporal trends. *Canadian Field Naturalist* 104:222-243.

Peakall, J.D., D.G. Noble, J.E. Elliott, J.D. Somers, and G. Erickson. 1990. Environmental contaminants in Canadian Peregrine Falcons, *Falco peregrines*: a toxicological assessment. *Canadian Field-Naturalist* 104:244–254.

Sparschu G.L., F.L. Dunn, R.W. Lisowe, and V.K. Rowe. 1971. Study of the effects of high levels of 2,4,5-trichlorophenoxyacetic acid on fetal development in the rat. *Food and Cosmetics Toxicology* 9:527-530.

Spencer, F. 1982. An assessment of the reproductive toxic potential of Aroclor 1254 in female Sprague-Dawley rats. *Bulletin of Environmental Contamination and Toxicology* 28:290-297.

Stratus (Stratus Consulting, Inc.). 1999. I*njuries to Avian Resources, Lower Fox River/Green Bay Natural Resource Damage Assessment*. Final Report. Prepared for U.S. Fish & Wildlife Services, U.S. Department of the Interior, and the U.S. Department of Justice. May 7, 1999.

12. RISK SUMMARY

Highlights of Risk Summary

- Total PCBs (tPCBs) and other COCs in the PSA of the Housatonic River pose intermediate or high risks to some assessment endpoints, and low risk to others.
- Risk is high for amphibians and piscivorous mammals. Confidence in this conclusion is high because (1) multiple lines of evidence with concordant results were available; (2) models used to estimate risk were not conservative; and (3) consideration of uncertainty indicates a high probability of effects.
- Risk is high for some insectivorous birds (wood duck).
- Risk is high for selected threatened and endangered birds (bald eagle and American bittern) and intermediate for a T&E mammal species (small-footed myotis). There is uncertainty regarding these conclusions because corroborating lines of evidence were generally not available.
- Risk is intermediate to high for benthic invertebrates, but risk varies across reaches and at smaller spatial scales. Confidence in this conclusion is high because multiple lines of evidence with concordant results were available.
- Risk is intermediate to high for piscivorous birds (osprey and belted kingfisher).
- Risk is intermediate for omnivorous and carnivorous mammals (red fox and short-tailed shrew).
- Risk is low to intermediate for warmwater fish in the PSA, and confidence in this conclusion is moderate. Two of the three major lines of evidence (site-specific toxicity, fish tissue chemistry relative to effects thresholds) suggest intermediate risk to the fish. However, the field surveys suggest that PCBs and/or other COCs are not causing obvious effects to fish populations.
- Risk is low for some insectivorous birds (e.g., tree swallow and American robin), but confidence in this conclusion is not high because of conflicting results between the available lines of evidence.
- Other species not included in the quantitative risk assessments may also be at risk in the PSA.
- Assessment of risks to the most susceptible endpoints downstream of the PSA indicates that benthic invertebrates, amphibians, coldwater fish, mink, river otter, and bald eagles may be at risk.

12.1 OVERVIEW

The assessment of ecological risks of COCs in the Housatonic River to aquatic life and wildlife is described in Sections 3 through 11, and presented in more detail in Appendices D through K. The amount of information considered in this assessment is large, and the analyses and interpretation complex. The purpose of this section is to summarize the major findings of the

ERA and to discuss the implications of these findings for biota in the Primary Study Area (PSA) and downstream of the PSA.

Section 12.2 summarizes the risk assessment findings for each assessment endpoint. The first part of this presentation (Section 12.2.1) discusses the results of the weight-of-evidence approach for each of the 8 assessment endpoints evaluated in the risk assessment. The WOE approach is a process by which measurement endpoints are related to an assessment endpoint to evaluate whether significant risk is posed to the environment (Menzie et al. 1996). A formal WOE can range from a simple qualitative assessment to a highly quantitative evaluation; however, no matter what form the WOE takes, it should provide documentation of the thought process used when assessing potential ecological risk.

The term “line of evidence” as used in this ERA follows the definition of “Information derived from different sources or by different techniques that can be used to describe and interpret risk estimates” provided in the *Guidelines for Ecological Risk Assessment* (EPA 1998). Unlike the term “weight-of-evidence,” this definition does not imply assignment of qualitative or quantitative weightings to information. The three general lines of evidence under which most measurement endpoints fall are (Hull and Suter 1994; Suter et al. 1995):

- Biological (field) survey data that indicate the state of the receiving environment.
- Media toxicity data that indicate whether the contaminated media are toxic (i.e., laboratory or in situ toxicity testing).
- Comparisons of individual contaminant exposures (measured or estimated) to toxicity thresholds (e.g., exposure modeling or comparison of site media concentrations to effects thresholds).

Two or three general lines of evidence were considered in evaluating potential risk for most assessment endpoints. The WOE approach used in this ERA for each of the assessment endpoints follows the approach originally described in the *Massachusetts Weight-of-Evidence Special Report* (Menzie et al. 1996).

Following the WOE discussion, Section 12.2.2 presents a discussion of hazard quotients (HQs) that were calculated for each receptor for the two COCs of greatest concern, tPCBs and TEQ, to facilitate comparison of risks between assessment endpoints in the PSA. The HQ analysis includes estimates of uncertainty to provide an indication of both the magnitude of risk for each COC receptor combination and the amount of uncertainty about each risk estimate.

Following the HQ analysis, the assessment of risks conducted for areas downstream of the PSA is described in Section 12.2.3. As is apparent from the preceding sections of the ecological risk assessment, risks to some assessment endpoints vary within the PSA as well as downstream, due to the small-scale variability in sediment and, to a lesser degree, floodplain soil concentrations. Section 12.2 concludes with a brief discussion of possible reasons for the differences in risk between assessment endpoints for the most influential COCs, tPCBs and TEQ.

Section 12.3 discusses the broader implications of the risk assessment findings summarized in Section 12.2. Issues addressed include:

- The risk assessment described in Sections 3 through 11 and Appendices D through K focused the majority of quantitative analyses on selected species, termed "representative species." There are, however, many other species that occur in the watershed of the Housatonic River. Section 12.3 begins with a discussion of estimates of the potential risks posed by COCs to these other species.

- In addition to effects on survival, growth, and reproduction of individuals in the Housatonic River, there are a number of other possible impacts of COCs on aquatic life and wildlife that were not addressed in the individual assessments (i.e. indirect effects, narrowing of the genetic pools for exposed species). These topics are briefly addressed in Section 12.3.

Section 12.4 provides a discussion acknowledging that there are many sources of uncertainty in an ecological risk assessment, even in assessments (such as this ERA) that have a great deal of available information. The preceding sections and the appendices described the sources of uncertainty for each assessment endpoint and their potential influence on risk estimates and the confidence in those risk estimates. Section 12.4 summarizes the most important sources of uncertainty, particularly those that were common to many assessment endpoints.

Section 12.5 concludes with a listing of the major findings of the Housatonic River Ecological Risk Assessment.

12.2 SUMMARY OF THE ASSESSMENT ENDPOINT CONCLUSIONS

The problem formulation stage of the ERA (Section 2.8) identified the assessment endpoints considered important in the Housatonic River ERA. Each of the assessment endpoints was evaluated and conclusions made regarding the potential for adverse effects (see Sections 3 through 11 and Appendices D through K). Table 12.2-1 provides a short summary for each of the assessment endpoints and the conclusions reached in the ERA for the PSA. Tables 12.2-2 to 12.2-16 indicate the results of the weight-of-evidence assessments for each assessment endpoint.

12.2.1 Results of Weight-of-Evidence Evaluation

12.2.1.1 Benthic Invertebrates

The WOE results for the benthic invertebrate assessment endpoint are shown in Table 12.2-2. In this WOE table, the measurement endpoints for the three lines of evidence: water, sediment, and tissue chemistry (C), toxicity tests (T), benthic community measures (B) are listed, as are the weighting of the measurement endpoint, evidence of harm, and magnitude of response. This table indicates that the majority of endpoints suggest some risk for benthic communities in both coarse- and fine-grained sediment. The evidence for harm is somewhat stronger in coarse-grained substrate compared to fine-grained substrate; in situ studies indicated larger perturbations to the benthic community in coarse-grained substrate than in fine-grained substrate. The conclusion from the WOE evaluation is that there is an intermediate to high risk to much of the benthic community in the PSA.

12.2.1.2 Amphibians

The results of the WOE assessment for amphibians are presented in Table 12.2-3. In the amphibian WOE matrix, the measurement endpoints for the three lines of evidence: tissue chemistry (C); wood frog toxicity tests (W) and leopard frog toxicity tests (L); and field surveys (B) are listed. As shown in the table, many of the measurement endpoints indicated some degree of risk. Evidence of harm to amphibians based on the two GE studies was undetermined due to limitations in the study designs. The only endpoint that did not indicate potential risk was the earliest life stage wood frog toxicity endpoint, for which there is a mechanistic explanation for the lack of response. Four endpoints exhibited a high degree of risk combined with a moderate to high confidence rating.

Table 12.2-1

Ecological Assessment Endpoints and Conclusions for the Primary Study Area Portion of the Lower Housatonic River

Receptor	Assessment Endpoint	Conclusions
Benthic Invertebrates	Community structure, survival, growth, and reproduction	The benthic invertebrate ERA demonstrates intermediate to high risk from tPCBs based on a weight-of-evidence evaluation of multiple effects endpoints. The pronounced toxicity (laboratory and in situ) observed in PSA sediment was supported by toxicity identification evaluation (TIE) findings, alterations to macroinvertebrate community structure, reduced benthic diversity, and large exceedances of effects benchmark values for invertebrate tissues, sediment, and water.
Amphibians	Community condition, survival, reproduction, development, and maturation	The amphibian ERA indicates risk to frogs from tPCBs is high, based on a weight-of-evidence evaluation of multiple effects endpoints. The literature-derived tissue thresholds for tPCBs were supported by site-specific toxicity studies, skewed sex ratios, malformations, and other effects that implicated tPCBs as the causal agent. Sediment toxicity tests indicated a correlation between level of effect and tPCB concentration.
Fish	Survival, growth, and reproduction	The fish ERA documented intermediate risk to fish from both tPCBs and TEQ, and low risk from PAHs, based on a weight-of-evidence evaluation. The PCB tissue thresholds identified in the literature and from site-specific toxicity studies were exceeded by fish tissue concentrations measured in the PSA for all representative species except brown bullhead. For some species (e.g., yellow perch), the majority of individual fish concentrations exceeded the respective benchmarks for both tPCBs and TEQ. Despite the high probability of responses, the magnitude of adverse responses is expected to be low to intermediate. The toxicity data suggest that reproductive and developmental effects occur in many spawns of PSA fish, but do not necessarily occur in all spawns. Field studies suggest that effects to fish at the community or local population level are not severe, and that populations of largemouth bass are reproducing and self-sustaining. The findings from the literature reviews and site-specific toxicity studies are consistent with the results of field studies. The latter cannot rule out low-level impact to local fish populations, but indicate that local populations are not experiencing obvious effects (e.g., reproductive failure). The incidence of abnormalities such as tumors is indicative of potential health impairment that has uncertain relevance to the local population.

Table 12.2-1

Ecological Assessment Endpoints and Conclusions for the Primary Study Area Portion of the Lower Housatonic River (Continued)

Receptor	Assessment Endpoint	Conclusions
Insectivorous Birds	Survival, growth, and reproduction	The weight-of-evidence analysis indicates that some insectivorous birds, such as tree swallows and American robins, are likely at low risk in the PSA as a result of exposure to tPCBs and TEQ. Risks to tree swallows and robins in the PSA are predicted to be intermediate to high based on modeling of exposure and effects, but field studies of tree swallows and American robins detected no obvious adverse reproductive effects to these birds in the PSA. The weight-of-evidence assessment relied more heavily on the results of the site-specific field studies, but the conclusion of low risk is uncertain because the lines of evidence did not give concordant results. Risks are predicted to be high for wood ducks exposed to tPCBs and TEQ in the PSA based on modeling of exposure and effects. A limited sampling program indicated that measured concentrations of TEQ in wood duck eggs were similar to model predictions, thus reducing uncertainty for this line of evidence. No other lines of evidence were available for this representative species.
Piscivorous Birds	Survival, growth, and reproduction	The weight-of-evidence analysis indicates that ospreys are at high risk from exposure to tPCBs and intermediate risk from exposure to TEQ in the Housatonic River PSA. In the PSA, exposure of ospreys to tPCBs is greater than doses that caused adverse effects in the most tolerant bird species studied. The conclusion of high risk to ospreys is uncertain because only one line of evidence was available. Belted kingfishers are considered to be at intermediate risk as a result of exposure to tPCBs and TEQ in the Housatonic River PSA. While modeled exposure and effects indicated high risk for tPCBs and intermediate risk for TEQ, a field study of belted kingfisher productivity indicated that the birds were reproducing in the PSA. The conclusion of intermediate risk to belted kingfishers is uncertain because the two lines of evidence did not give concordant results.
Piscivorous Mammals	Survival, growth, and reproduction	The weight-of-evidence analysis indicates that piscivorous mammals (i.e., mink and river otter) are at high risk in the PSA as a result of exposure to tPCBs and TEQ. Evidence for this conclusion includes limited sightings of mink and otter in the PSA, except during winter, despite availability of appropriate habitat and evidence that they are common in nearby reference areas; results of the feeding study which showed effects on kit survival and jaw lesions in surviving mink at a much smaller fraction of fish in the diet (<1%) than would be expected of mink foraging in the PSA; and modeling of exposure and effects, which predicted high risks to mink and otter foraging in the PSA. Risks to mink and otter are elevated even for mink and otter that forage only a small fraction (e.g., 10%) of their time in the PSA.

Table 12.2-1

Ecological Assessment Endpoints and Conclusions for the Primary Study Area Portion of the Lower Housatonic River (Continued)

Receptor	Assessment Endpoint	Conclusions
Omnivorous and Carnivorous Mammals	Survival, growth, and reproduction	The weight-of-evidence analysis indicates an intermediate risk for red fox and short-tailed shrews exposed to tPCBs in the PSA. The field surveys indicated that omnivorous and carnivorous mammals, including short-tailed shrew and red fox, are common in some areas of the PSA. Modeling of exposure and effects predicted fox to be at intermediate risk as a result of exposure to tPCBs and TEQ in the PSA. Modeling of exposure and effects indicated that short-tailed shrews exposed to tPCBs are at high risk in some PSA locations and low risk in others. The risks posed by TEQ to short-tailed shrews in the PSA are lower. Analysis of the population demography field study by EPA suggests that short-tailed shrews are being affected by tPCBs in the more contaminated locations in the PSA. The conclusion of intermediate risk for fox is uncertain because of high uncertainty about several inputs used in the modeling exercise and because the two available lines of evidence did not produce concordant results. The conclusion of intermediate risk for short-tailed shrews is supported by the modeling and population demography study lines of evidence.
Threatened and Endangered Species	Survival, growth, and reproduction	Based on modeling of exposure and effects, the risk to bald eagles and American bitterns is high as a result of exposure to tPCBs and TEQ. The risks to small-footed myotis exposed to tPCBs and TEQ are intermediate.

Table 12.2-2

Evidence of Harm and Magnitude of Effects for Measurement Endpoints Related to Maintenance of the Benthic Community

Measurement Endpoints	Weighting Value	Evidence of Harm?	Magnitude of Harm (High, Intermediate, or Low)	
			Coarse-Grained	Fine-Grained
C. Chemistry Endpoints				
C-1. Concentration of PCB in overlying water in relation to concentrations reported (in literature) to be harmful to benthic invertebrates.	Low/Moderate	Yes	Intermediate	Intermediate
C-2. Concentration of PCB in sediment in relation to concentrations reported (in literature) to be harmful to benthic invertebrates.	Moderate	Yes	High	High
C-3. Concentration of PCB in invertebrate tissues in relation to concentrations reported (in literature) to be harmful to benthic invertebrates.	Moderate	Yes	High	High
T. Toxicity Endpoints				
T-1. Sediment toxicity to multiple invertebrate species, as measured in the laboratory toxicity tests (*Hyalella*, *Chironomus*).	Moderate/High	Yes	High	High
T-2. Sediment toxicity to multiple invertebrate species, as measured in the in situ toxicity tests (*Hyalella*, *Chironomus*, *Daphnia*).	Moderate/High	Yes	High	High
T-3. Indications of PCB as toxicity driver in TIE investigations (*Ceriodaphnia*).	Moderate	Yes	–	Intermediate
B. Benthic Community Endpoints				
B-1. Abundance, richness, biomass, and diversity of assemblages, relative to reference stations of comparable substrate and habitat (ANOVA and regression analyses).	Moderate	Yes	Intermediate	Low
B-2. Benthic community structure, as assessed using multivariate analysis of benthic metrics (rank analysis, multidimensional scaling) and multiple regression analyses.	Moderate	Yes	Intermediate	Low
B-3. Water quality assessment using modified Hilsenhoff Biotic Index (MHBI) indicator of organic pollution	Moderate	No	–	–
B-4. Evaluation of individual taxa using species sensitivity distribution (SSD) analysis of abundance endpoint.	Moderate/High	Yes	High	Intermediate

Table 12.2-3
Evidence of Harm and Magnitude of Effects for Measurement Endpoints Related to Maintenance of Amphibian Populations in the Housatonic River PSA

Measurement Endpoints	Weighting Value (High, Moderate, Low)	Evidence of Harm (Yes, No, Undetermined)	Magnitude (High, Intermediate, Low)
C. Chemical Measures			
C. Concentration of PCB in frog tissues in relation to levels reported to be harmful to amphibians.	Moderate	Yes	Low
W. Wood Frog Toxicological Measures			
W-1. Sediment toxicity to hatchling/late embryo life stages.	Mod/High	No	-
W-2. Sediment toxicity to larval life stages.	Mod/High	Yes	Intermediate
W-3. Sediment toxicity to late larval/metamorph life stage.	Mod/High	Yes	High
W-4. GE Study (juvenile wood frogs)	Low	Undetermined	-
L. Leopard Frog Toxicological Measures			
L-1. Sediment toxicity to hatchling/late embryo life stages.	Mod/High	Yes	Low
L-2. Sediment toxicity to larval life stages.	Mod/High	Yes	High
L-3. Sediment toxicity to late larval/metamorph life stage.	Mod/High	Yes	High
L-4. Sediment toxicity to adult leopard frogs (reproductive health).	Mod/High	Yes	High
B. Biology			
B-1. Vernal pool community study.	Mod/High	Yes	Low
B-2. GE leopard frog egg mass survey	Low/Mod	Undetermined	-
B-3. Anecdotal observations during collections for reproductive study.	Moderate	Yes	Low

In addition, a population model was constructed for wood frogs to determine whether effects from PCBs on individual wood frogs influence the populations within the PSA. A 10-year simulation, both with and without the effects of PCBs, was conducted. The model demonstrated that effects observed in the toxicity studies would result in population level impacts.

The conclusion is that there is a significant risk to amphibians as indicated by the preponderance of the evidence, the relative weights of the measurement endpoints, and the population modeling. The "no risk" value of measurement endpoint W-1 does not diminish the overall conclusion, because the study demonstrated that the embryo/early larval life stages are fairly insensitive to the effects of maternally transferred PCBs relative to later juvenile life stages exposed to contaminated media.

12.2.1.3 Fish

The WOE results for fish in the PSA are shown in Table 12.2-4. In the fish WOE matrix, the measurement endpoints for the three lines of evidence: site-specific toxicity tests (T); fish tissue chemistry (C); and field surveys (F) are listed. This table illustrates that the majority of endpoints indicate, with a moderately high degree of confidence, that there are intermediate risks to fish in the PSA. Although a high probability of adverse impacts to fish from tPCBs and/or TEQ is predicted throughout the PSA, the impacts predicted are for sensitive sublethal endpoints (reproduction and development); mortality of adults is unlikely except at the extreme upper tail of the observed fish tissue PCB concentration distribution. Large and obvious local population or community level responses, such as total reproductive failure or year-class failures, are not predicted for any reach. Adverse effects, although high in probability, have been observed at the individual level only; the linkage of these observed responses to local population health and resiliency over the long term is uncertain, given the low precision of the field studies conducted. The field studies conducted in the PSA (fish community and reproduction studies) support lack of obvious effects to the population, but cannot be used to assess lesser impacts. PSA fish exhibit a suite of physical malformations in both young and adults of some species (e.g., lesions, tumors) that appear to be related to contaminants. These deformities, although of uncertain relevance to local population abundance, may reflect an impairment of overall fish health in the PSA.

Table 12.2-4

Evidence of Harm and Magnitude of Effects for Measurement Endpoints Related to Maintenance of a Healthy Fish Community

Measurement Endpoints	Weighting Value (High, Moderate, Low)	Evidence of Harm (Yes, No, Undetermined)	Magnitude (High, Intermediate, Low)
C. Chemistry Endpoints			
C-1. Concentration of PCB in tissues of representative species in relation to threshold concentrations reported (in literature) to be harmful to fish.	Moderate	Yes	Intermediate
C-2. Concentration of PCB in tissues of representative species in relation to threshold concentrations reported (in Phase I toxicity study) to be harmful to fish	Moderate/High	Yes	Intermediate
C-3. Concentration of PCB in tissues of representative species in relation to threshold concentrations reported (in Phase II toxicity study) to be harmful to fish.	Moderate/High	Yes	Intermediate
T. Toxicity Endpoints			
T-1. Indications of reproductive and developmental impairment of largemouth bass spawned from Housatonic River adults in Phase I toxicity study.	Moderate/High	Yes	Intermediate
T-2. Indications of reproductive and developmental impairment of bass, medaka, and rainbow trout fry reared from eggs injected with Housatonic River extracts in Phase II toxicity study.	High	Yes	Intermediate
F. Field Study Endpoints			
F-1. Abundance and biomass of fish species observed in EPA fish collections and GE fish community assessment.	Moderate	No	–
F-2. Reproduction and nest condition metrics from GE reproduction study including evidence of largemouth bass reproduction, nest conditions, YOY abundance and growth, and adult growth rate.	Moderate	No	–
F-3. Largemouth bass population demographics including age-structure analysis and adult condition.	Low/Moderate	No	–

12.2.1.4 Insectivorous Birds

The WOE results for exposure of representative insectivorous birds (i.e., tree swallow, American robin, and wood duck) to tPCBs are presented in Table 12.2-5 and for exposure to TEQ in Table 12.2-6. Two lines of evidence are presented in the table: the field studies, and modeled exposure and effects. The results from the modeled exposure and effects line of evidence suggest that tPCBs and TEQ pose intermediate to high risk to tree swallows living in the PSA. However, the field study line of evidence suggests that, although effects are likely occurring, they are minor. The uncertainty concerning the field-based threshold range for tPCBs likely means that risks of this COC are overestimated for the PSA. Even the upper end of the tPCB range is associated with equivocal evidence for adverse effects to tree swallows. For TEQ, the threshold range is quite broad. The available evidence from field studies indicates that tree swallows are tolerant to exposure to persistent organochlorines such as tPCBs and TEQ. If the tree swallow threshold is near the upper end of the threshold range, then the current modeled exposure and effects line of evidence is overestimating risks of TEQ to tree swallows. Thus, the WOE assessment supports a finding of low risk for tree swallows exposed to tPCBs and TEQ in the PSA. This conclusion, however, is uncertain because of the conflicting results in the WOE assessment.

Table 12.2-5

Evidence of Harm and Magnitude of Effects for Insectivorous Birds Exposed to tPCBs in the Housatonic River PSA

Measurement Endpoints	Weighting Value (High, Moderate, Low)	Evidence of Harm (Yes, No, Undetermined)	Magnitude (High, Intermediate, Low)
Field Study	High (Tree swallow) Moderate/High (American robin)	Yes (Tree swallow) No (American robin)	Low (Tree swallow) – (American robin)
Modeled Exposure and Effects	Moderate (Tree swallow) Moderate (American robin) Moderate (Wood duck)	Yes (Tree swallow) Yes (American robin) Yes (Wood duck)	High (Tree swallow) High (American robin) High (Wood duck)

Table 12.2-6

Evidence of Harm and Magnitude of Effects for Insectivorous Birds Exposed to TEQ in the Housatonic River PSA

Measurement Endpoints	Weighting Value (High, Moderate, Low)	Evidence of Harm (Yes, No, Undetermined)	Magnitude (High, Intermediate, Low)
Field Study	High (Tree swallow) Moderate/High (American robin)	Yes (Tree swallow) No (American robin)	Low (Tree swallow) – (American robin)
Modeled Exposure and Effects	Moderate (Tree swallow) Moderate (American robin) Moderate/High (Wood duck)	Yes (Tree swallow) Yes (American robin) Yes (Wood duck)	Intermediate (Tree swallow) Intermediate (American robin) High (Wood duck)

The results from the modeled exposure and effects lines of evidence suggest that tPCBs and TEQ pose an intermediate to high risk to American robins inhabiting the PSA. The American robin field study, however, suggests that reproductive success is not being impaired by the tPCBs and TEQ in the PSA. The uncertainty in the modeled exposure and effects line of evidence, outlined below, likely means the approach overestimates the risks of tPCBs and TEQ to American robins in the PSA. The WOE assessment therefore supports a conclusion of low risk to American robins exposed to tPCBs and TEQ in the PSA. This conclusion, however, is uncertain because of the conflicting results in the WOE assessment.

The results from the modeled exposure and effects line of evidence suggest that tPCBs and TEQ pose high risk to wood ducks living in the PSA. A limited sampling program indicated that measured concentrations of TEQ in wood duck eggs were similar to model predictions, thus reducing uncertainty for this line of evidence. However, there are no demographic data on the wood duck population in the PSA to provide an additional line of evidence. The WOE assessment suggests a finding of high risk for wood duck and similar species (e.g., hooded merganser) residing in the PSA.

12.2.1.5 Piscivorous Birds

The WOE analysis indicates that exposure of piscivorous birds, such as the belted kingfisher and osprey (Tables 12.2-7 and 12.2-8), to tPCBs and TEQ in the PSA, could lead to adverse reproductive effects in some species. The two lines of evidence used to support this conclusion were the field study of belted kingfisher productivity and the comparison of modeled exposure with effects to piscivorous birds (for belted kingfisher and osprey).

Table 12.2-7

Evidence of Harm and Magnitude of Effects for Piscivorous Birds Exposed to tPCBs in the Housatonic River PSA

Measurement Endpoints	Weighting Value (High, Moderate, Low)	Evidence of Harm (Yes, No, Undetermined)	Magnitude (High, Intermediate, Low)
Modeled Exposure and Effects	Moderate	Yes (Belted kingfisher) Yes (Osprey)	High (Belted kingfisher) High (Osprey)
Belted kingfisher Field Study (Henning 2002)	Moderate	No (Belted kingfisher)	–

Table 12.2-8

Evidence of Harm and Magnitude of Effects for Piscivorous Birds Exposed to TEQ in the Housatonic River PSA

Measurement Endpoints	Weighting Value (High, Moderate, Low)	Evidence of Harm (Yes, No, Undetermined)	Magnitude (High, Intermediate, Low)
Modeled Exposure and Effects	Moderate	Yes (Belted kingfisher) Yes (Osprey)	Intermediate (Belted kingfisher) Intermediate (Osprey)
Belted kingfisher Field Study (Henning 2002)	Moderate	No (Belted kingfisher)	–

For the assessment of risks to belted kingfishers, both lines of evidence were available. The modeled exposure and effects line of evidence indicated that belted kingfishers in the PSA are likely to receive a tPCB dose greater than what the most tolerant species known from the literature can be exposed to without effects. For TEQ, the risk picture is less clear because the threshold range for this COC is very wide and the exposure estimates for belted kingfishers fell within this range. Thus, without effects data specific to belted kingfishers, it is difficult to make definitive conclusions about the risks of TEQ to this species. The field study of belted kingfisher productivity, however, indicated that these birds are able to reproduce in the PSA. This line of evidence was given an equal weighting to the exposure and effects modeling line of evidence, despite concerns about the design and conduct of the field study. Therefore, belted kingfishers are considered to be at intermediate risk in the PSA as a result of exposure to tPCBs and TEQ. The conclusion of intermediate risk to belted kingfishers is uncertain because the two lines of evidence did not give concordant results.

For osprey, only the modeled exposure and effects line of evidence was available to assess risk to these birds. As with belted kingfishers, this line of evidence indicated that ospreys in the PSA are likely to receive a tPCB dose that is greater than what the most tolerant species known in the literature can be exposed to without effects. The risks due to exposure to TEQ are unclear, as the estimates for exposure also fell within the toxicity threshold range. However, a site-specific study that investigated the effects of COCs on osprey in the PSA was not conducted. The PSA contains suitable habitat for ospreys, with abundant prey: one pair was observed in the PSA exhibiting courtship behavior during the breeding season but did not nest successfully. Ospreys are considered to be at risk in the PSA as a result of exposure to tPCBs and TEQ.

12.2.1.6 Piscivorous Mammals

The results of the WOE assessment for piscivorous mammals are presented for tPCBs and TEQ, respectively, in Tables 12.2-9 and 12.2-10. All three lines of evidence: field studies (EPA), mink feeding study, and modeled exposure and effects, indicate that the elevated concentrations of tPCBs and TEQ in the PSA of the Housatonic River are causing adverse effects of high magnitude to mink and river otter. The field surveys indicate that mink and river otter are rarely present in the PSA, except during winter when they are occasionally present as transients, and have not established home territories close to the main channel despite suitable mink and otter habitat. The MSU site-specific feeding study indicated that feeding adult female mink with a diet containing approximately 0.9% fish from the PSA caused a 20% reduction in kit survival from 0 to 6 weeks of age. Because mink in the wild typically consume fish as 20% or more of their diet, the associated risk from exposure in the field is correspondingly higher. In addition, other components of the mink diet in the PSA (e.g., crayfish) have high concentrations of tPCBs and TEQ. Further, the jaw lesion study indicated that erosion of the jaw occurs at even lower doses and exhibits a dose-response relationship. The occurrence of jaw lesions coincides with the induction of Ah-receptor-regulated enzymes (ECOD and EROD) also in a dose-response manner.

Table 12.2-9

Evidence of Harm and Magnitude of Effects for Piscivorous Mammals Exposed to tPCBs in the Housatonic River PSA

Measurement Endpoints		Weighting Value (High, Moderate, Low)	Evidence of Harm (Yes, No, Undetermined)	Magnitude (High, Intermediate, Low)
Field Surveys	EPA	Moderate/High	Yes (Mink) Yes (River otter)	High (Mink) High (River otter)
	GE	Moderate	No (Mink) No (River otter)	– (Mink) – (River otter)
Feeding Study		High	Yes (Mink)	High (Mink)
Modeled Exposure and Effects		Moderate/High	Yes (Mink) Yes (River otter)	High (Mink) High (River otter)

Table 12.2-10

Evidence of Harm and Magnitude of Effects for Piscivorous Mammals Exposed to TEQ in the Housatonic River PSA

Measurement Endpoints		Weighting Value (High, Moderate, Low)	Evidence of Harm (Yes, No, Undetermined)	Magnitude (High, Intermediate, Low)
Field Surveys	EPA	Moderate/High	Yes (Mink) Yes (River otter)	High (Mink) High (River otter)
	GE	Moderate	No (Mink) No (River otter)	– (Mink) – (River otter)
Feeding Study		High	Yes (Mink)	High (Mink)
Modeled Exposure and Effects		Moderate/High	Yes (Mink) Yes (River otter)	High (Mink) High (River otter)

The high risks evident from the feeding study are further supported by the modeled exposure and effects line of evidence. The estimated potential for exposure is so high that even individual mink and otter that forage in the PSA only for short periods of time (10% of foraging time or less) are at an intermediate or higher risk from tPCBs and TEQ.

12.2.1.7 Omnivorous and Carnivorous Mammals

The WOE results for omnivorous and carnivorous mammals are shown in Table 12.2-11 for tPCB and Table 12.2-12 for TEQ. Data from three lines of evidence were available: field surveys, a population demography study of short-tailed shrew, and exposure and effects modeling.

The weight-of-evidence analysis indicates, based on modeling results, high risk for short-tailed shrews exposed to tPCBs and no appreciable risk to short-tailed shrew exposed to TEQs in the PSA. This conclusion, however, is uncertain because of the lack of species-specific measures of effects in the literature used for the exposure and effects modeling. The conclusions in the short-tailed shrew demographic study at the site conducted for GE are not in agreement with the results from the modeling of exposure and effects line of evidence. However, the results of the supplemental analyses conducted by EPA of the data from the GE demographic study of short-tailed shrews are in agreement with the modeling results, suggesting that there are intermediate effects from exposure to COCs in the contaminated areas of the PSA. Risk to short-tailed shrew based on field surveys was undetermined.

The WOE also suggests, based on the modeling results for red fox, an intermediate risk to fox exposed to tPCBs and TEQ in the PSA. This assessment is uncertain because measures of effects of tPCBs and TEQ to fox were not available in the literature and there are no other lines of evidence, because the field study was not designed to distinguish the potential impacts from exposure to contaminants.

Table 12.2-11

Evidence of Harm and Magnitude of Effects for Omnivorous and Carnivorous Mammals Exposed to tPCBs in the Housatonic River PSA

Measurement Endpoints	Weighting Value (High, Moderate, Low)	Evidence of Harm (Yes, No, Undetermined)	Magnitude (High, Intermediate, Low)
Field Surveys	Moderate	Undetermined (Shrew) Undetermined (Red fox)	— (Shrew) — (Red fox)
Population Demography Field Study	Moderate/High	Yes* (Shrew)	Intermediate (Shrew)
Modeled Exposure and Effects	Moderate/High	Yes (Shrew) Yes(Red fox)	High (Shrew) Intermediate (Red fox)

* Based on EPA's re-analysis of the demographic study data.

Table 12.2-12

Evidence of Harm and Magnitude of Effects for Omnivorous and Carnivorous Mammals Exposed to TEQ in the Housatonic River PSA

Measurement Endpoints	Weighting Value (High, Moderate, Low)	Evidence of Harm (Yes, No, Undetermined)	Magnitude (High, Intermediate, Low)
Field Surveys	Moderate	Undetermined (Shrew) Undetermined (Red fox)	— (Shrew) — (Red fox)
Modeled Exposure and Effects	Moderate/High	No (Shrew) Yes (Red fox)	— (Shrew) Intermediate (Red fox)

12.2.1.8 Threatened and Endangered Species

The results of the WOE evaluation for threatened and endangered species using a single line of evidence, modeled site-specific exposures and effects, are shown in Table 12.2-13 and Table 12.2-14 for tPCBs and TEQ, respectively.

Table 12.2-13

Evidence of Harm and Magnitude of Effects for T&E Species Exposed to tPCBs in Housatonic River PSA

Measurement Endpoints	Weighting Value (High, Moderate, Low)	Evidence of Harm (Yes, No, Undetermined)	Magnitude (High, Intermediate, Low)
Modeled exposure and effects, Bald Eagle	Moderate/High	Yes	High
Modeled exposure and effects, American Bittern	Moderate	Yes	High
Modeled exposure and effects, Small-Footed Myotis	Moderate	Yes	Intermediate

Table 12.2-14

Evidence of Harm and Magnitude of Effects for T&E Species Exposed to TEQ in the Housatonic River PSA

Measurement Endpoints	Weighting Value (High, Moderate, Low)	Evidence of Harm (Yes, No, Undetermined)	Magnitude (High, Intermediate, Low)
Modeled exposure and effects, Bald Eagle	Moderate/High	Yes	High
Modeled exposure and effects, American Bittern	Moderate	Yes	High
Modeled exposure and effects, Small-Footed Myotis	Moderate	Yes	Intermediate

The weight-of-evidence analysis indicates that T&E species such as bald eagles, American bitterns, and small-footed myotis are at risk in the PSA as a result of exposure to tPCBs and TEQ. The risks for bald eagles and American bittern exposed to tPCBs and TEQ are high. The risks to small-footed myotis exposed to tPCBs and TEQ are intermediate. However, there is less uncertainty associated with the bald eagle evaluation than with the American bittern and small-footed myotis evaluations.

12.2.2 Hazard Quotient Analyses

The assessments described in Sections 3 through 11 and Appendices D through K were conducted using various lines of evidence including, in many cases, different measurement endpoints and effects metrics. It is clear that risks posed by COCs in the PSA vary between species; however, the degree of variability is not clear from these discussions because of the differing endpoints and metrics used. To facilitate comparison of risks among aquatic life and wildlife receptors and to give a broad summary of the findings of the risk assessment, risks were converted to probabilistic hazard quotients (HQs). These HQs are the expected environmental concentration or dose of a contaminant divided by its estimated low or no toxic effect concentration or dose. Higher quotients indicate greater risk, although the magnitude of risk is not necessarily linearly proportional to the magnitude of the HQ. The methods used to calculate the probabilistic HQs and the results of these analyses for each endpoint are discussed in this section.

12.2.2.1 Aquatic Assessment Endpoints

12.2.2.1.1 Benthic Invertebrates

For benthic invertebrates, HQs were calculated for Reaches 5A, 5B, 5C, 5D, and 6 in the PSA. Using the data on tPCB concentrations in sediment, medians, means, 25^{th} and 50^{th} percentiles, and minimum and maximum concentrations were determined for each reach. Hazard quotients were calculated by dividing each of these summary statistics by the site-specific effects benchmark for benthic invertebrates exposed to tPCBs in sediment. The sediment effects threshold used in the derivation of the HQs was the MATC of 3 mg/kg tPCB, which represents the threshold between low and intermediate risk, based on an evaluation of site-specific toxicity and benthic community findings (Section 3 and Appendix D). The results are plotted in Figure 12.2-1.

These results indicate that intermediate (or greater) risks to benthic invertebrates were observed in all reaches of the PSA, with HQs for tPCBs greater than 1 for both mean and median tPCB concentrations. Probability distributions of sediment concentrations in each reach, relative to the intermediate-risk threshold (i.e., MATC) of 3 mg/kg tPCB and a high-risk sediment threshold of 13 mg/kg tPCB, are presented in Appendix D (Figure D.4-2). Predicted risks were greatest in the upstream (Reach 5A) and Woods Pond (Reach 6) sediment; the latter reach was assigned a high-risk rating because more than half the sediment samples exceeded the high-risk sediment threshold. Due to the considerable small-scale variation in sediment tPCB concentrations, HQs derived for individual samples span approximately 4 orders of magnitude (approximately 0.01 to 100). Because benthic invertebrates are much less mobile than fish and wildlife, they do not spatially and temporally integrate their exposures to the same degree. Therefore, the hazard quotient results for benthic invertebrates indicate that the majority of individuals are at risk (i.e., HQ > 1), but that some individuals in less contaminated areas are not.

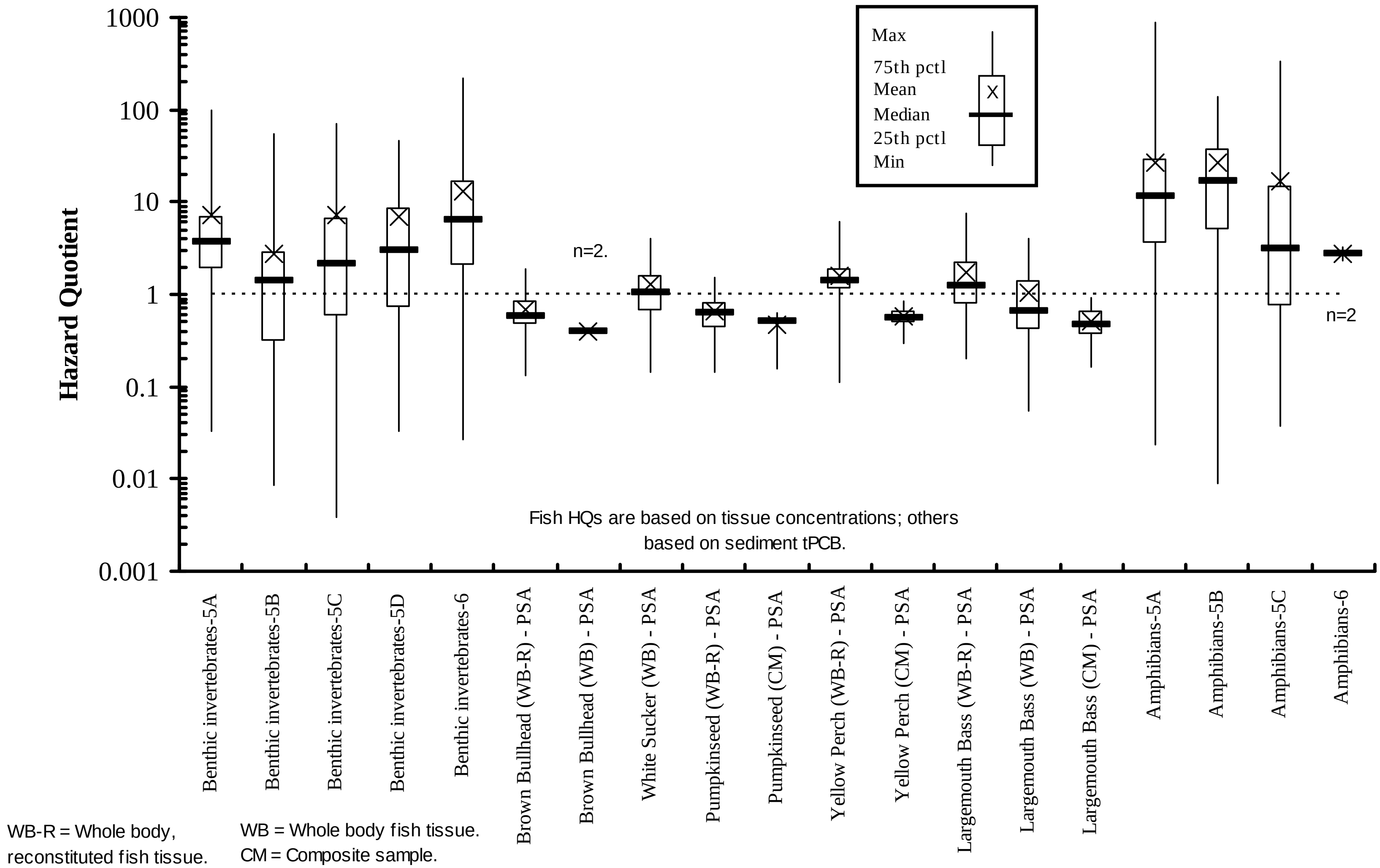


Figure 12.2-1 Hazard Quotients for Aquatic Biota Exposed to tPCBs in the Housatonic River PSA

12.2.2.1.2 Amphibians

For amphibians, HQs were calculated for Reaches 5A, 5B, 5C, 5D, and 6 in the PSA, using methods similar to those described for benthic invertebrates. Hazard quotients were calculated by dividing summary statistics for vernal pool sediment concentrations by the site-specific effects benchmark for amphibians exposed to tPCBs in sediment (3.27 mg/kg tPCBs) (Section 4 and Appendix E). This approach does not address adult leopard frog exposures that likely occur in river and backwater sediment. The results are plotted in Figure 12.2-1. These results indicate significant risk in all reaches of the PSA, with HQs above 1 for both mean and median tPCB concentrations. Predicted risks were greatest in the upstream (Reach 5A) vernal pool habitats. Because of the variation in sediment PCB concentrations between the vernal pools, HQs for the subreaches span about 4 orders of magnitude (approximately 0.01 to 100). The hazard quotient results for amphibians indicate that the majority of individuals are at risk (i.e., HQ > 1), with greater HQs (i.e., HQ > 5) in a large percentage of vernal pools (about 50% in Reaches 5A and 5B).

12.2.2.1.3 Fish

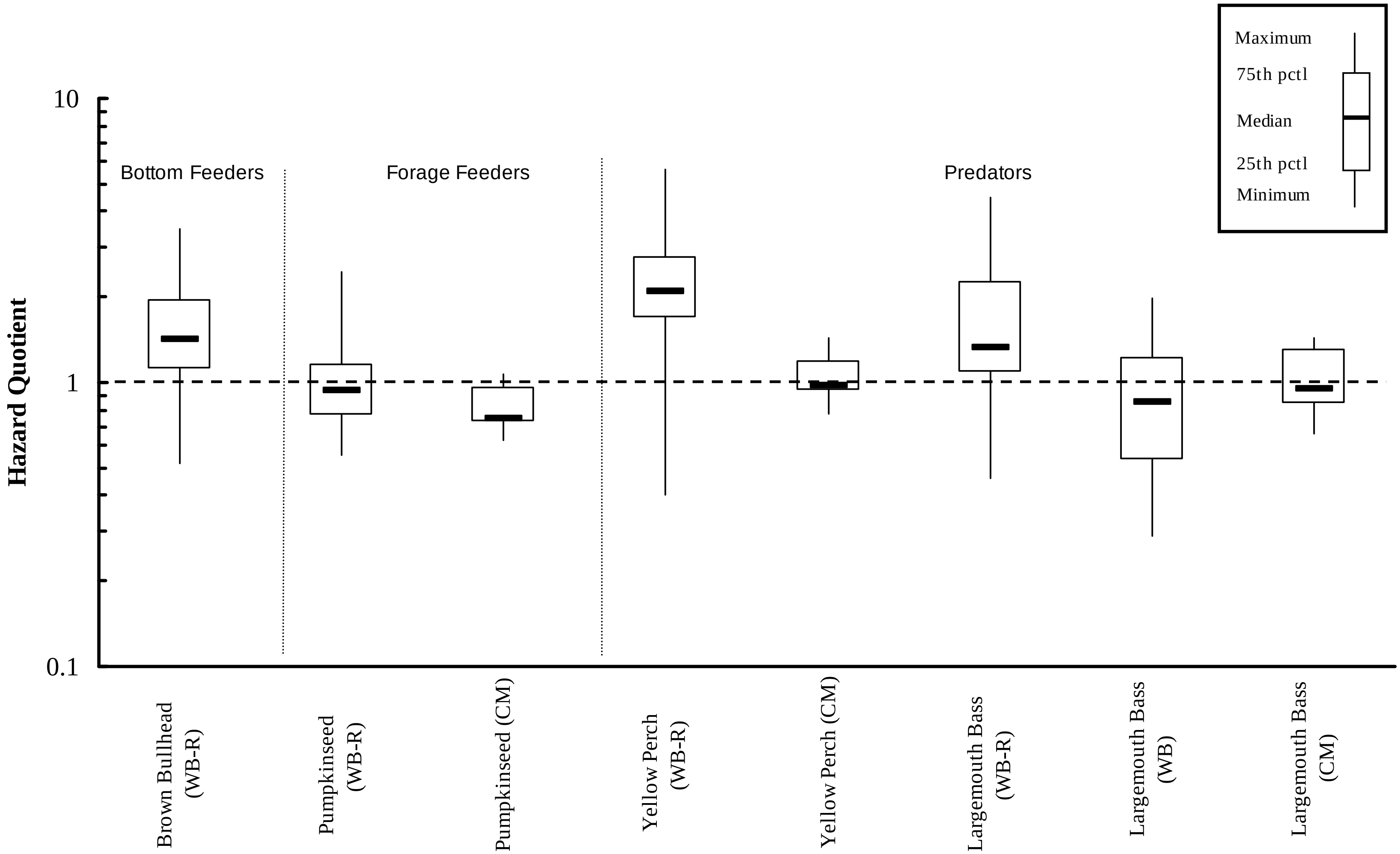
For fish, HQs were calculated separately for the five representative warmwater species (Section 5 and Appendix F) by dividing summary statistics for exposure by the site-specific tissue effects benchmark protective of all warmwater PSA fish (55 mg/kg ww tPCB; MATC derived from site-specific toxicity studies and supported by literature review findings). The results are plotted in Figure 12.2-1. These results indicate that risk occurs in all reaches of the PSA, with both mean and median HQs for tPCBs above 1 for adult fish (i.e., whole body reconstituted tissue concentrations of some species). Predicted risks were greatest in adult fish and in predator fish at the top of the food web. Due to the variation in fish tissue tPCB concentrations, hazard quotients for the reaches span about an order of magnitude, with most HQ values between 0.3 and 3; the lower bound of this range represents primarily younger age classes that have not yet accumulated their maximum tPCB burdens and fish species near the lower end of the food chain. Thus, the hazard quotient results for fish indicate that predatory species are at risk (i.e., HQ > 1) once they reach their maximum adult tPCB concentrations. The ERA indicates that these HQs are indicative of sublethal (e.g., reproductive and developmental) responses to offspring; the pathway for the manifestation of effects is through the maternal transfer of tPCBs to eggs. Acute mortality to adults is not expected for most fish.

In addition to tPCBs, fish HQs were derived and plotted for TEQ (Figure 12.2-2). The effects benchmark applied in this analysis was derived from the site-specific toxicity studies (44 ng/kg ww TEQ) (Appendix F). The magnitudes and probabilities of risk for TEQ are generally similar to tPCB risks.

12.2.2.1.4 Summary

For aquatic receptors (benthic invertebrates, amphibians, and fish), the HQs presented in Figures 12.2-1 and 12.2-2 are not conservative. Although sensitive species were considered in the derivation of the effects thresholds, no additional safety factors were used to develop the effects metrics. The thresholds used in HQ calculations (MATCs) represent concentrations demonstrated to cause adverse responses to organisms in site-specific studies. Thus, HQ exceedances of 1 are cause for concern. In addition, some individual-level responses, such as endocrine responses in reproducing adult fish, incidence of lesions or tumors due to contaminants, have been observed at exposure concentrations lower than the MATCs. Overall, for aquatic organisms in the ERA, risk levels range from intermediate to high, depending on species, reach, and local exposure conditions.

The HQs presented in Figures 12.2-1 and 12.2-2 depict ranges in risk attributable to variation in exposures (tissue or sediment) but do not convey variability or incertitude in the site-specific effects threshold values. The uncertainties associated with the MATC derivation are discussed in detail in Appendices D to F. Due to these uncertainties, a hazard quotient of 1 should not be interpreted as a definitive conclusion that no risk is possible.



1

2

3 **Figure 12.2-2 Hazard Quotients for Fish Exposed to 2,3,7,8-TCDD TEQ in the Housatonic River PSA**

12.2.2.2 Wildlife Assessment Endpoints

For wildlife, probabilistic HQs were calculated as follows:

- The distributions from the Monte Carlo analyses for total daily intake of COCs by representative species were each divided by the corresponding effects metrics used to estimate risks in Sections 7 through 11 and Appendices G through K. In the case of a dose-response curve effects metric (e.g., mink exposed to tPCBs), the effects metric was specified as a uniform distribution of dose ranging from 10 to 20% effect. A similar approach was used for field-based effects metrics (e.g., tree swallows exposed to tPCBs), and threshold ranges (e.g., belted kingfishers exposed to TEQ). For wood ducks and bald eagles, the exposure medium and the effects metric were concentration in eggs, as described in Appendices G and K.

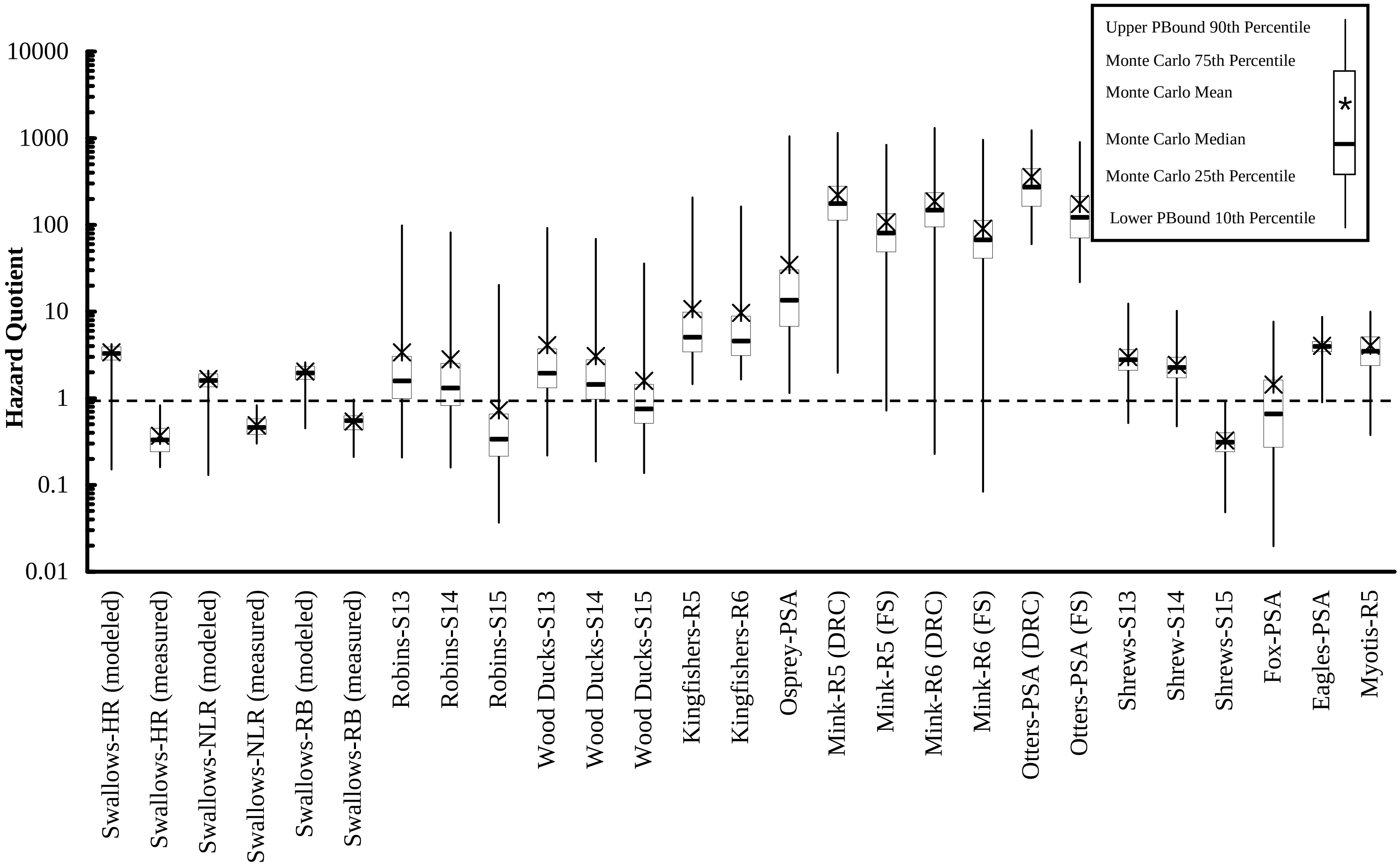
- A similar approach was used with the results of the probability bounds analysis, except that the effects metric was specified as a distribution-free range.

- The analyses were done for both tPCBs and TEQ.

- Modified box-and-whisker plots were developed for each representative species in the PSA. For species with smaller foraging ranges, the analyses were done for different areas within the PSA. Included in the plots (Figures 12.2-3 and 12.2-4) are the median HQ from the Monte Carlo analysis (the thick line bisecting each box), the mean HQ from the Monte Carlo analysis (star symbol), the 25th and 75th percentile HQs from the Monte Carlo analysis (the bottom and top of each box), the 10th percentile HQ from the lower bound of the probability bounds analysis (bottom whisker), and the 90th percentile from the upper bound of the probability bounds analysis (top whisker).

- Probabilistic HQ plots were also developed for tree swallows exposed to tPCBs using measured concentrations in 12-day nestlings (data from Custer 2002) as an estimate of exposure. This was done to facilitate comparison of risks using the microexposure modeling approach and using measured concentrations from birds in the PSA. Insufficient data were available to perform a similar analysis for TEQ. For the measured concentrations data, an empirical histogram was specified for each PSA location using the available 3 years of data for the Monte Carlo portion of the analyses. The empirical histograms were divided by the same threshold range as used for HQs based on the microexposure model outputs. For the probability bounds analyses, 95% Kolmogorov-Smirnov confidence limits were calculated for each empirical histogram. Both the upper and lower confidence limits were divided by the same threshold range as used for HQs based on the microexposure model outputs. Using the results of these analyses, modified box-and-whisker plots were developed as previously described (Figures 12.2-3 and 12.2-4).

In addition to plots developed for mink and otter exposed to tPCBs and TEQ using the results of literature-based dose-response curves, plots were also developed using the results of the mink feeding study conducted by Bursian et al (2003). In this case, the denominator was the 10 to



1
2
3 **Figure 12.2-3 Hazard Quotients for Wildlife Exposed to tPCBs in the Housatonic River PSA**

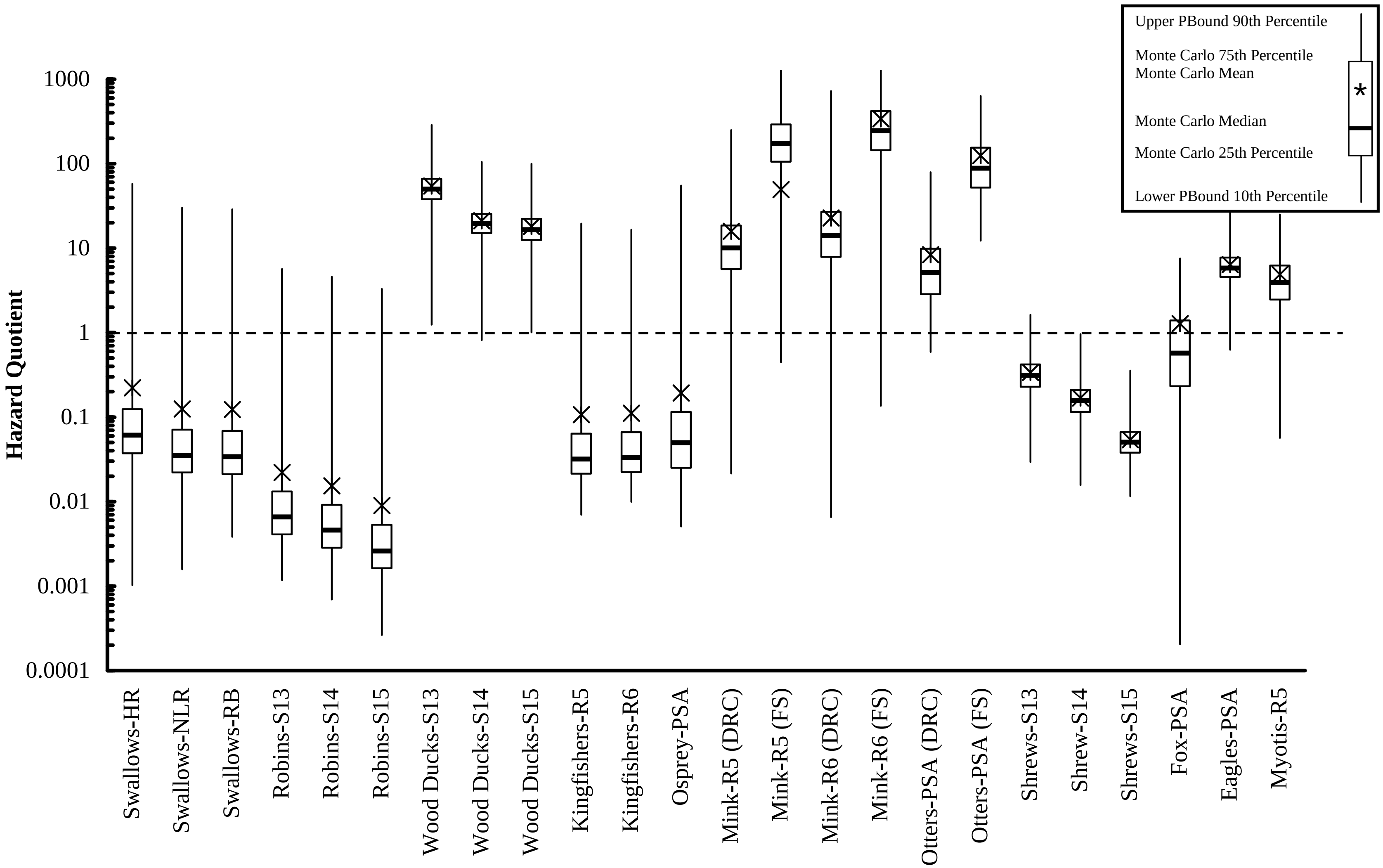


Figure 12.2-4 Hazard Quotients for Wildlife Exposed to 2,3,7,8-TCDD TEQ in the Housatonic River PSA

20% effects dose range from the regression analyses conducted using the results of the Bursian et al. (2003) study. Otherwise, the approach was as described above. The results are shown in Figures 12.2-3 and 12.2-4. For wildlife species other than tree swallow, mink, and river otter, it was not possible to derive field-based or feeding-study-based HQs because the required data were not available.

The HQs presented in Figures 12.2-3 and 12.2-4 depict ranges in risk attributable to variation in exposures (tissue or sediment) but do not convey variability or incertitude in the effects metrics. However, unlike many HQs derived in other risk assessments, the probabilistic HQs generated for the wildlife receptors are not biased to be conservative. No safety factors were used to estimate the effects metrics (except in the case of the bald eagle). The uncertainties associated with the effects metric or derivation of the MATC are discussed in detail in Appendices G to K. Due to these uncertainties, an HQ of 1 should not be interpreted as a definitive conclusion that no risk is possible.

In summary, the wildlife HQs presented in Figures 12.2-3 and 12.2-4 indicate the following:

- Wildlife risks from tPCBs are generally higher than risks from TEQ by 1 to several orders of magnitude. However, for wood ducks, the risks from TEQ exceeded those from tPCBs.

- The comparison of HQ plots for tree swallows indicates that risks are higher using the results of the microexposure model than is the case using measured concentrations in tree swallow nestlings. Thus, the microexposure model appears to be overestimating exposure for tPCBs. The HQ plots using measured concentrations in nestlings supports the weight-of-evidence conclusion that risks of tPCBs to tree swallows is low (see Section 7, Appendix G and Table 12.2-1).

- Wildlife risks from tPCBs and TEQ are highest for mink and river otter, with mean and median HQs between 100 and 500 for tPCBs, and between 5 and 10 for TEQ using the results of the literature-based dose-response curve. The HQs for tPCBs were similar (tPCBs) or higher (TEQ) when the results of the feeding study were used to derive the effects range. Wood ducks are also at high risk from exposure to TEQ.

- The risks from tPCBs and TEQ for most wildlife species have some uncertainty associated with them to the extent that the bottom or top whiskers span an HQ of 1, and there are additional uncertainties associated with the effects metric that are not addressed in the HQ calculation. The whiskers are the extreme representations of uncertainty associated with exposure because they are tail outputs from the probability bounds analyses, a technique designed to propagate all forms of uncertainty (e.g., inability to precisely specify distribution type or parameter values for a distribution). Thus, the boxes in Figures 12.2-3 and 12.2-4 should be interpreted as representing a reasonable range within which the HQ estimate occurs for the receptor of interest, and the whiskers should be interpreted as representing the extremes within which the HQ could occur given the sources of uncertainty and variability in the exposure assessment.

The HQ analyses for aquatic biota and wildlife indicate that risks of tPCBs vary widely between representative species and assessment endpoints. Section 12.3 explores the fundamental reasons why this might occur.

12.2.3 Estimates of Risk Downstream of the PSA

Because of the reduced contaminant concentrations downstream of the PSA, the size of the area of concern, and shifts in aquatic habitat types associated both with river gradient and location of dams, a different approach than that applied in the PSA was followed to assess ecological risks of tPCBs to biota in areas downstream of the PSA. The assessment of ecological risks was conducted using mapping (GIS) techniques and threshold concentrations that indicate risk for seven taxonomic groups or species selected based on the outcome of the evaluations performed in the PSA and the habitat characteristics found downstream. These groups are benthic invertebrates, amphibians, warmwater fish, trout, mink, river otter, and bald eagles. A screening assessment for the downstream floodplain was also conducted for short-tailed shrews exposed to tPCBs in soil. Prey tissue concentration data downstream of the PSA are lacking for shrews, making a more refined assessment not feasible.

For each of these groups, a maximum acceptable threshold concentration (MATC) for tPCBs was developed based primarily on the detailed risk assessment performed for the PSA. Each MATC was then compared to media-specific data for areas downstream of the PSA to Long Island Sound. Areas with MATC exceedances, indicating potential risk, were plotted on maps of the river. The methods used for each of the seven representative groups and the results of the analyses are discussed in the following sections (see Table 12.2-15 for a summary of results).

Table 12.2-15

Summary of the Assessment of Risks Conducted for Biota Exposed to tPCBs in the Lower Housatonic River Downstream of the PSA[a]

Reach	Potentially at Risk (MATC Exceeded)						
	Benthic Invertebrates	**Amphibians**	**Warmwater Fish**	**Trout**	**Mink**	**River Otter**	**Eagle**
7	Yes	Yes	No	Yes[b]	Yes	Yes	No
8	Yes	Yes	No	No suitable habitat	Yes	Yes	Yes[c]
9	No	Yes	No	No	Yes	Yes	No
10	No	Limited suitable habitat	No	No suitable habitat	Yes	Yes	No
11	No	Limited suitable habitat	No	No	Yes	Yes	No
12	No	Limited suitable habitat	No	No	Yes	Yes	No
13	No	Limited suitable habitat	No	No suitable habitat	Yes	Yes	No
14	No	Limited suitable habitat	No	No suitable habitat	Yes	Yes	No
15	No	Limited suitable habitat	No	No suitable habitat	Yes	Yes	No
16	No	Limited suitable habitat	Insufficient data but unlikely	No suitable habitat	Insufficient data but unlikely	Insufficient data but unlikely	Insufficient data but unlikely

[a] The downstream assessment for short-tailed shrews indicated that they are not at risk from exposure to tPCBs in soil in any reach downstream of the PSA.

[b] Risks to coldwater fish are more uncertain than for warmwater fish because the trout species commonly occurring downstream of the PSA were not directly assessed in the ERA.

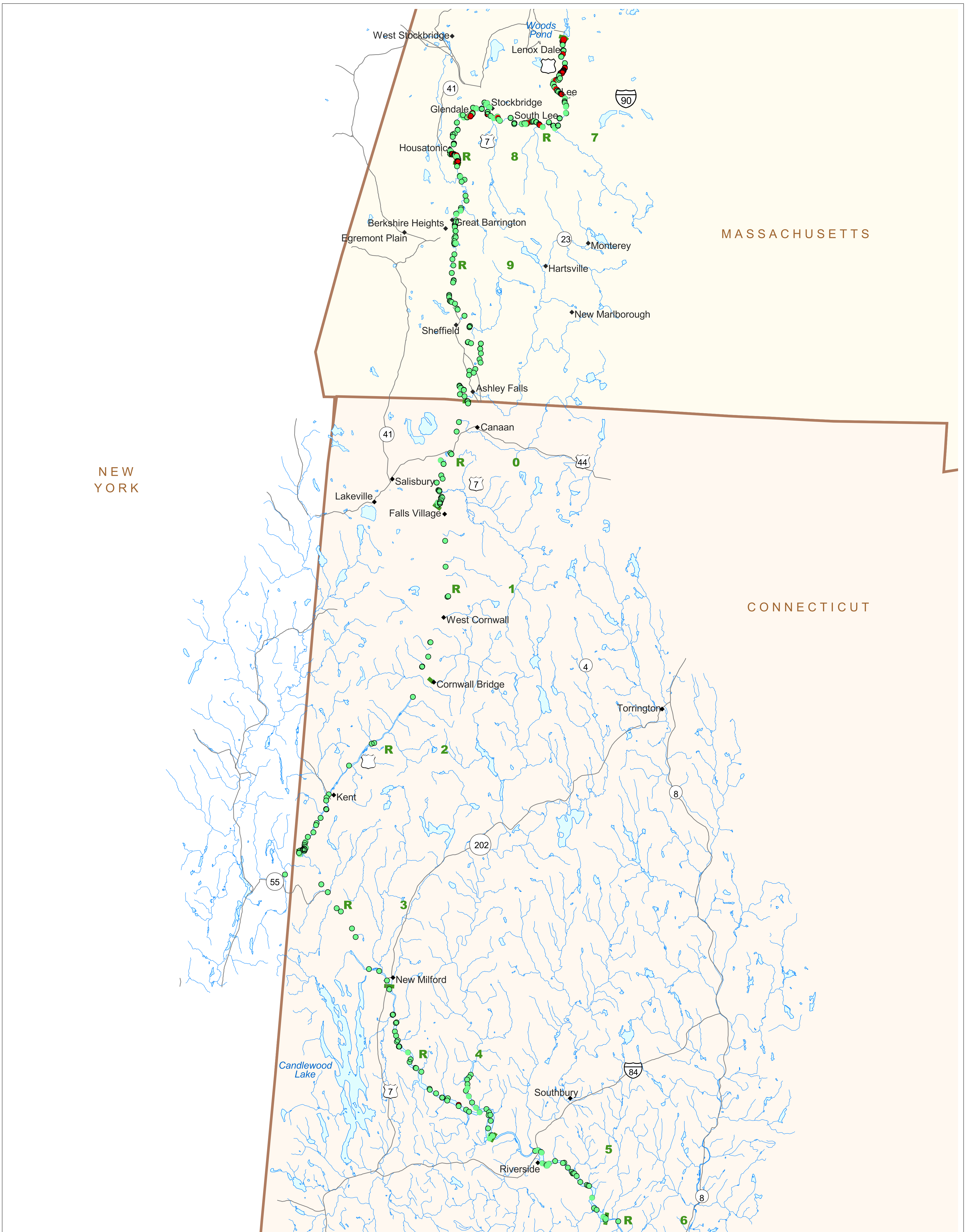
[c] Reach 8 is Rising Pond. Although eagles would be at risk foraging there, Rising Pond is smaller than an eagle's total foraging area and the prey concentrations outside of Rising Pond are not known; however, concentrations of tPCBs in prey consumed from the river and floodplain outside of Rising Pond are unlikely to be zero. Therefore, there is uncertainty about this risk estimate.

Where insufficient data for the medium of interest were available for a reach to estimate the risk for a species, other available data were examined to determine the likelihood of the concentrations of tPCBs in the medium of interest being high enough to pose a risk. In all cases, concentrations were lower than in other reaches with available data for the medium of interest; therefore, it is unlikely that the receptor is at risk in those reaches for which data in the medium of interest were not available.

12.2.3.1 Benthic Invertebrates

For benthic invertebrates, the medium of interest was river sediment. An MATC of 3 mg/kg tPCBs was used to define the transition between low and intermediate risk of adverse affects to benthic invertebrates downstream of the PSA (see Section 3 and Appendix D for a description of the derivation of the MATC). This concentration was developed using multiple lines of evidence (e.g., benthic community studies, in situ and laboratory toxicity testing, bioaccumulation testing, Sediment Quality Triad). Above the MATC concentration of 3 mg/kg tPCBs, numerous endpoints indicated ecologically significant responses, with many LC_{50}/EC_{50} values falling in the range of 3 to 30 mg/kg tPCBs. High risk is predicted at sediment concentrations of 13 mg/kg tPCB or greater, including acute lethal responses to multiple taxa.

The MATC of 3 mg/kg tPCBs was compared to recent surficial sediment data downstream of the PSA, and the results were plotted (Figure 12.2-5) to indicate samples above and below the MATC.



NOTES:
Risk to benthic invertebrates is based on a maximum acceptable threshold concentration (MATC) of 3.0 mg/kg total PCB (tPCB) (dry weight) in surficial sediments (0-6 inches).

LEGEND:

BENTHIC INVERTEBRATE RISK
- < 3 mg/kg
- >= 3 mg/kg

- Town
- Reach Break
- Roads
- Housatonic River Basin Hydrology
- State Boundary

N W E S

2 0 2 4 6 8 Miles

Ecological Risk Assessment
GE/Housatonic River Site
Rest of River

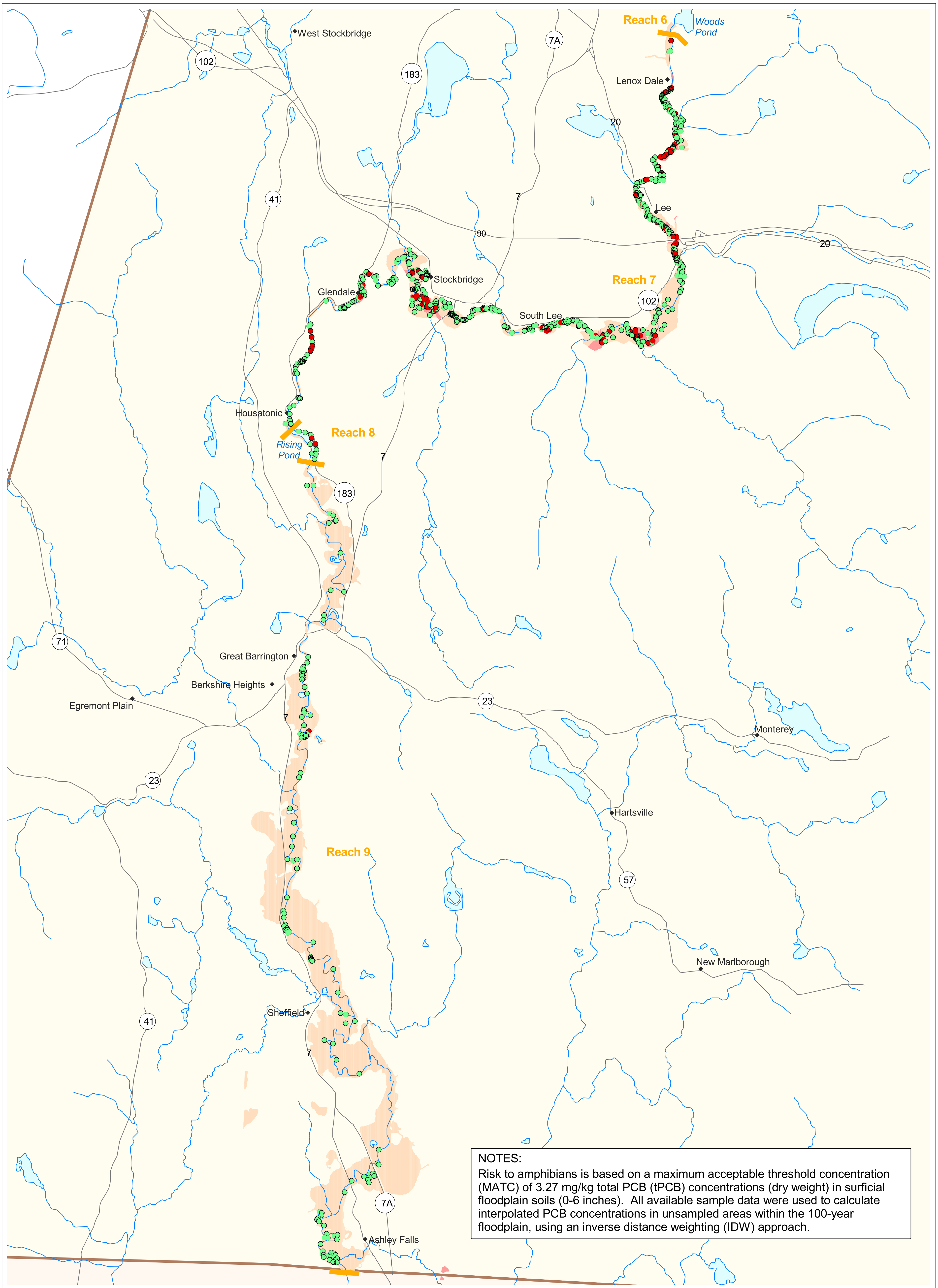
FIGURE 12.2-5
ASSESSMENT OF RISK TO BENTHIC INVERTEBRATES EXPOSED TO tPCBs DOWNSTREAM OF THE PSA

In general, potential risks to benthic invertebrates occur in limited areas downstream of Woods Pond to Rising Pond. These areas are depositional and tend to have higher concentrations of tPCBs. Below Rising Pond, sediment does not contain concentrations of tPCBs that represent a potential risk to benthic invertebrates. The latter conclusion is supported by comparison of field-collected invertebrate tissue residue data (West Cornwall, CT) to literature-derived PCB tissue thresholds.

12.2.3.2 Amphibians

Many species of amphibians are primarily exposed to PCBs in the floodplain, particularly in vernal pools and other low-lying wet areas. A sediment/floodplain soil MATC of 3.27 mg/kg tPCB was derived from the amphibian risk assessment conducted for the PSA (Section 4 and Appendix E). Inverse distance weighting was used to interpolate tPCB concentrations to the limit of the 100-year floodplain (10-year floodplain contours are not available downstream of the PSA) using the 0- to 6-inch (0- to 15-cm) depth data from the floodplain downstream of the PSA; a separate analysis was conducted for sediment in a manner analogous to that described above for benthic invertebrates.

Areas where the 3.27 mg/kg tPCB threshold was exceeded, estimating risks to amphibians due to PCB concentrations in floodplain soil and sediment, are shown in Figures 12.2-6 and 12.2-7. There are several large floodplain areas located in the area between Woods Pond and Rising Pond where risk is estimated, with only small isolated areas exceeding the MATC downstream of Rising Pond. Downstream of the Massachusetts/Connecticut state line, the risk mapping for amphibians was conducted for sediment only because the concentrations in floodplain soil had decreased below the MATC throughout Reach 9 upstream of the border, and the extent of the floodplain is limited in Connecticut.



NOTES:
Risk to amphibians is based on a maximum acceptable threshold concentration (MATC) of 3.27 mg/kg total PCB (tPCB) concentrations (dry weight) in surficial floodplain soils (0-6 inches). All available sample data were used to calculate interpolated PCB concentrations in unsampled areas within the 100-year floodplain, using an inverse distance weighting (IDW) approach.

LEGEND:

AMPHIBIAN RISK

SEDIMENT SAMPLES

- < 3.27 mg/kg
- >= 3.27 mg/kg

FLOODPLAIN SOIL

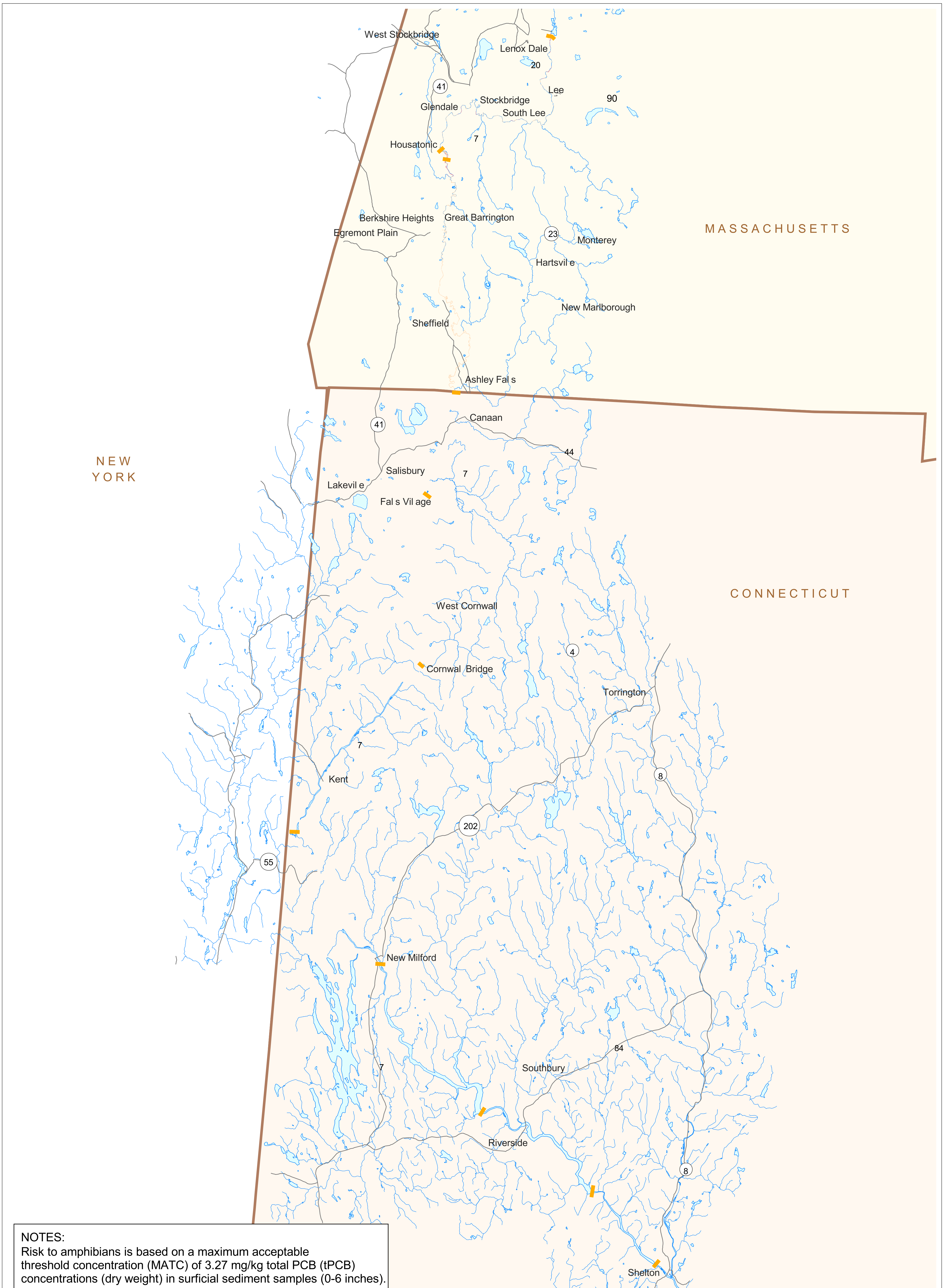
- < 3.27 mg/kg
- >= 3.27 mg/kg

- Town
- Reach Breaks
- Roads
- Housatonic River Basin Hydrology
- State Boundary

0.5 0 0.5 1 1.5 2 Miles

Ecological Risk Assessment
GE/Housatonic River Site
Rest of River

FIGURE 12.2-6
ASSESSMENT OF RISK TO AMPHIBIANS EXPOSED TO tPCBs DOWNSTREAM OF THE PSA IN MASSACHUSETTS



NOTES:
Risk to amphibians is based on a maximum acceptable threshold concentration (MATC) of 3.27 mg/kg total PCB (tPCB) concentrations (dry weight) in surficial sediment samples (0-6 inches).

LEGEND:

Town

Reach Breaks

AMPHIBIAN RISK

Roads

< 3.27 mg/kg
>= 3.27 mg/kg

Housatonic River Basin Hydrology

State Boundary

N W E S

2 0 2 4 6 8 Miles

Ecological Risk Assessment
GE/Housatonic River Site
Rest of River

FIGURE 12.2-7
ASSESSMENT OF RISK TO AMPHIBIANS IN SEDIMENT EXPOSED TO tPCBs DOWNSTREAM OF THE PSA IN CONNECTICUT

12.2.3.3 Warmwater Fish

As was done for the PSA, risk to fish downstream was evaluated based on concentrations of tPCBs in fish tissue. The MATC of 55 mg/kg tPCB in tissue (whole body, wet weight) developed for the PSA based on site-specific effects (Section 5 and Appendix F) was applied to areas downstream of the PSA using the available (e.g., bass, perch, sunfish) tissue data for warmwater species. Each downstream reach (Reaches 7 through 16) was evaluated as a unit, and the mean adult fish tissue concentration in each reach was compared with the MATC to estimate risk. Only data collected since 1998 were used in this analysis. The results are shown in Figure 12.2-8.

In some cases, it was necessary to extrapolate from the available data to develop an adult tissue concentration for comparison with the MATC. Young-of-year (YOY) largemouth bass data were extrapolated up to estimated adult concentrations by applying a factor of 3.5; similarly, yellow perch YOY data were scaled up by a factor of 2.5. Both ratios were calculated from the more extensive database available for the PSA. In addition, the majority of filet data were extrapolated to estimated whole body concentrations using the factor of 2.3 developed by Bevelhimer et al. (1997). Brown bullhead filet data were extrapolated to estimated whole body concentrations using the factor of 1.5 developed by EPA (1999a).

The evaluation of risk to warmwater fish species downstream of the PSA indicated no risks in any of the reaches below the PSA.

NOTES:
Risk to warmwater fish is based on a maximum acceptable threshold concentration (MATC) of 55 mg/kg total CB (tPCB) concentrations (wet weight) in whole body tissue.

- Only fish collected in 1998 to the present (2002) were included.
- Young-of-year bass composites were scaled by a factor of 3.5.
- Young-of-year perch composites were scaled by a factor of 2.5.
- Brown bullhead filet samples were scaled by 1.5.
- Warmwater filet samples were scaled by 2.3.

LEGEND:

WARMWATER FISH RISK
- < 55 mg/kg
- >= 55 mg/kg

- Town
- Reach Break
- Roads
- Housatonic River Basin Hydrology
- State Boundary

2 0 2 4 6 8 Miles

Ecological Risk Assessment
GE/Housatonic River Site
Rest of River

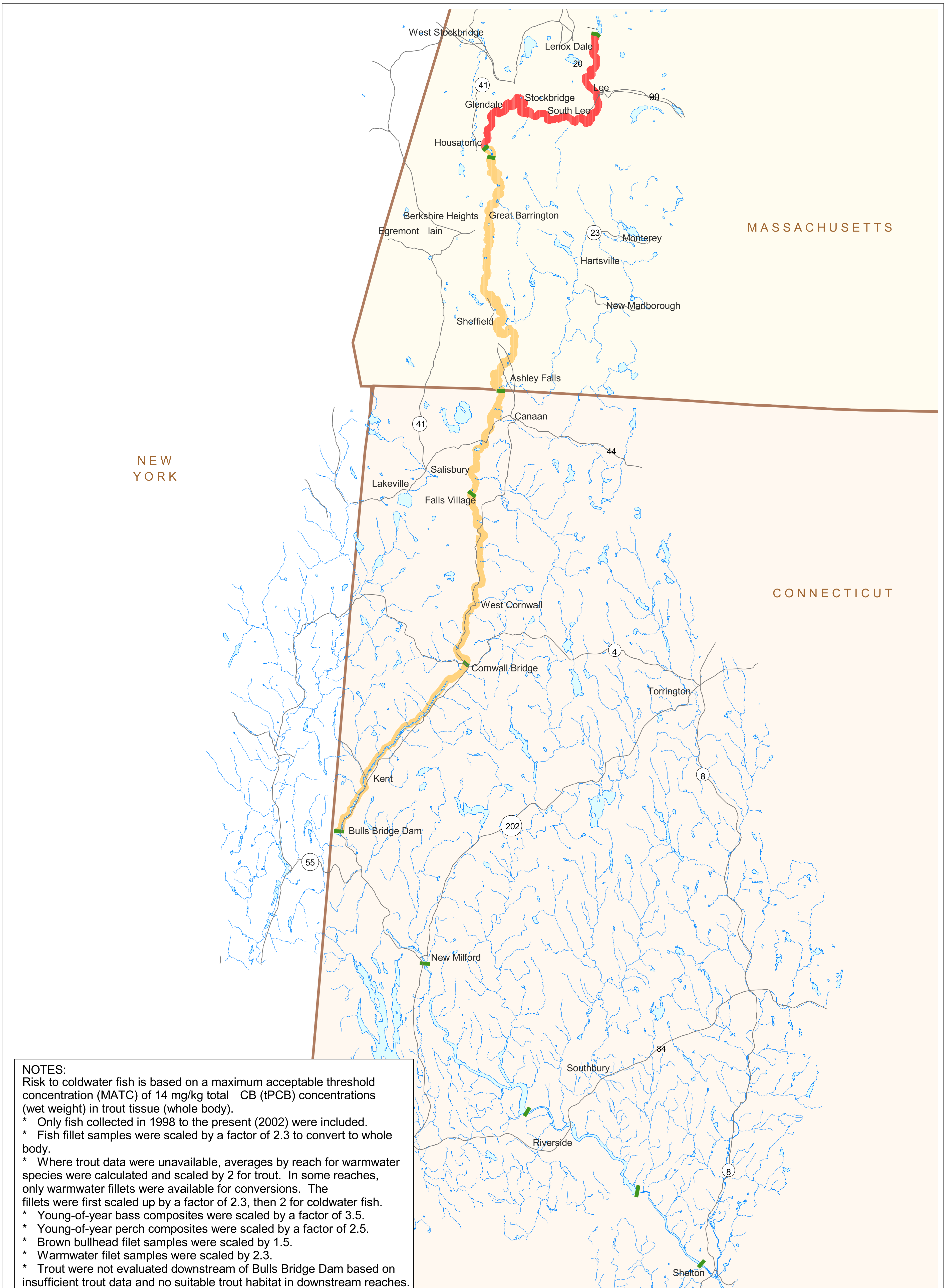
FIGURE 12.2-8
ASSESSMENT OF RISK TO WARM WATER FISH EXPOSED TO tPCBs DOWNSTREAM OF THE PSA

12.2.3.4 Trout

Trout were evaluated separately from warmwater fish species because of differences in habitat requirements and differences in the sensitivity of some trout species to tPCBs documented in the literature. Some trout species/strains have been documented to have greater sensitivity to PCBs and dioxins (Walker et al. 1994; Zabel et al. 1995) relative to the species considered in the development of the 55 mg/kg tPCB warmwater MATC. Trout also tend to have higher tPCB concentrations because of their higher lipid content. The strain of rainbow trout used in the site-specific toxicity tests (Tillitt et al. 2003b) is less sensitive than some other strains of trout that have been used in toxicity testing. Based on a comparison with LD_{50} values reported in the literature, Tillitt et al. (2003b) determined that the Fish Lake strain of rainbow trout used in the Phase II study was approximately three times less sensitive to the effects of TCDD and PCB than other strains and/or species reported in the literature. To protect more sensitive strains and species of trout resident in the downstream reaches of the Housatonic River (e.g., brown trout), the PSA toxicity threshold value of 55 mg/kg tPCB (whole body, wet weight) was divided by a factor of 4 to account for inter- and intra-species differences. Accordingly, a tissue MATC of 14 mg/kg ww tPCB for coldwater species was applied.

Because of the more limited database for trout, a number of extrapolations were necessary to convert available warmwater fish data and/or trout filet data to estimated whole body concentrations for trout. These extrapolations included all of the extrapolation factors discussed above for warmwater fish species and an additional extrapolation factor of 2.0 to estimate trout whole body tPCB concentrations from measured tPCB concentrations in warmwater fish samples. This factor was developed from the available smallmouth bass and trout data from the Trout Management Area (Reach 11, in part) in Connecticut. The Amrhein et al. (1999) factor of 1.47 was used to convert the trout filet data to estimated whole body concentrations.

The results of this evaluation are shown in Figure 12.2-9. Trout are potentially at risk in Reach 7, but not in reaches with suitable habitat farther downstream. This assessment has high uncertainty due to the number of extrapolations required and the low magnitude of exceedance of the MATC value. Potential risk to trout was not evaluated downstream of Reach 12 due to lack of suitable trout habitat.



NOTES:
Risk to coldwater fish is based on a maximum acceptable threshold concentration (MATC) of 14 mg/kg total CB (tPCB) concentrations (wet weight) in trout tissue (whole body).
* Only fish collected in 1998 to the present (2002) were included.
* Fish fillet samples were scaled by a factor of 2.3 to convert to whole body.
* Where trout data were unavailable, averages by reach for warmwater species were calculated and scaled by 2 for trout. In some reaches, only warmwater fillets were available for conversions. The fillets were first scaled up by a factor of 2.3, then 2 for coldwater fish.
* Young-of-year bass composites were scaled by a factor of 3.5.
* Young-of-year perch composites were scaled by a factor of 2.5.
* Brown bullhead filet samples were scaled by 1.5.
* Warmwater filet samples were scaled by 2.3.
* Trout were not evaluated downstream of Bulls Bridge Dam based on insufficient trout data and no suitable trout habitat in downstream reaches.

LEGEND:

COLD WATER FISH RISK
< 14 mg/kg
>= 14 mg/kg

Town
Reach Break
Roads
Housatonic River Basin Hydrology
State Boundary

2 0 2 4 6 8 Miles

Ecological Risk Assessment
GE/Housatonic River Site
Rest of River

**FIGURE 12.2-9
ASSESSMENT OF RISK
TO TROUT EXPOSED TO
tPCBs DOWNSTREAM
OF THE PSA**

12.2.3.5 Mink

An MATC of 0.984 mg/kg tPCBs in fish (whole body, wet weight) was derived for mink and river otter. The MATC is the LC_{20} from the regression analysis conducted on mink kit survival from 0 to 6 weeks (see Section I.3.2) in the feeding study by Bursian et al. (2003). This feeding study was conducted to determine the toxicity of Housatonic River fish to mink (Section 9 and Appendix I).

Habitat suitability for mink was determined for the reaches downstream of the PSA (Appendix A). According to this analysis, potential mink habitat is ubiquitous along Reaches 7 to 16 and includes all areas except high gradient stream, calcareous rock cliff, cultural grassland, agricultural cropland, and residential/industrial development.

Mean fish concentrations were calculated for each river reach downstream of the PSA using available whole body fish tissue data from samples collected since 1998. The analysis was restricted to fish with an overall body length between 7 and 20 cm, corresponding to the size commonly preyed on by mink. In some cases, it was necessary to use YOY data for bass and perch, and these were extrapolated to adult concentrations using the factors discussed above.

For this analysis, it was assumed that the mink were exposed to the mean fish concentration in the downstream reaches for 59% of the diet because the mink diet contains on average 23% fish and 36% invertebrates, the majority of which are crayfish. No crayfish data were available for the downstream reaches; however, within the PSA, crayfish tPCB concentrations were similar to fish concentrations in the size range consumed by mink. Therefore, the assumption of 59% of the dietary exposure at the mean fish concentrations in the downstream reaches is reasonable. This risk estimate likely underestimates the mink exposure in the downstream reaches, as it was assumed that the concentration of tPCBs in the remaining 41% of the diet was 0.

The results of the evaluation for mink are shown in Figure 12.2-10. Potential risk to mink due to consumption of contaminated fish occurs from the Woods Pond Dam downstream to the end of Reach 15.

NOTES:
Risk to mink is based on a maximum acceptable threshold concentration (MATC) of 0.984 mg/kg total CB (tPCB) concentration (wet weight) in whole body fish tissue 7-20 cm in length.

* Only fish collected in 1998 to the present (2000) were included.
* Young-of-year bass composites were scaled by a factor of 3.5.
* Young-of-year perch composites were scaled by a factor of 2.5.
* Trout filet samples were scaled by 1.47.
* Brown bullhead filet samples were scaled by 1.5.
* Warmwater filet samples were scaled by 2.3.

LEGEND:

MINK RISK
- < 0.984 mg/kg
- >= 0.984 mg/kg

- Town
- Reach Break
- Roads
- Housatonic River Basin Hydrology
- State Boundary

2 0 2 4 6 8 Miles

Ecological Risk Assessment
GE/Housatonic River Site
Rest of River

FIGURE 12.2-10
ASSESSMENT OF RISK TO MINK EXPOSED TO tPCBs DOWNSTREAM OF THE PSA

| O:\gepitt\aprs\era_species_082504.apr | layout - mink i4-17 | o:\gepitt\epsfiles\plots\in\fig_12-2-10.eps | 3:46 PM, 11/2/2004 |

12.2.3.6 River Otter

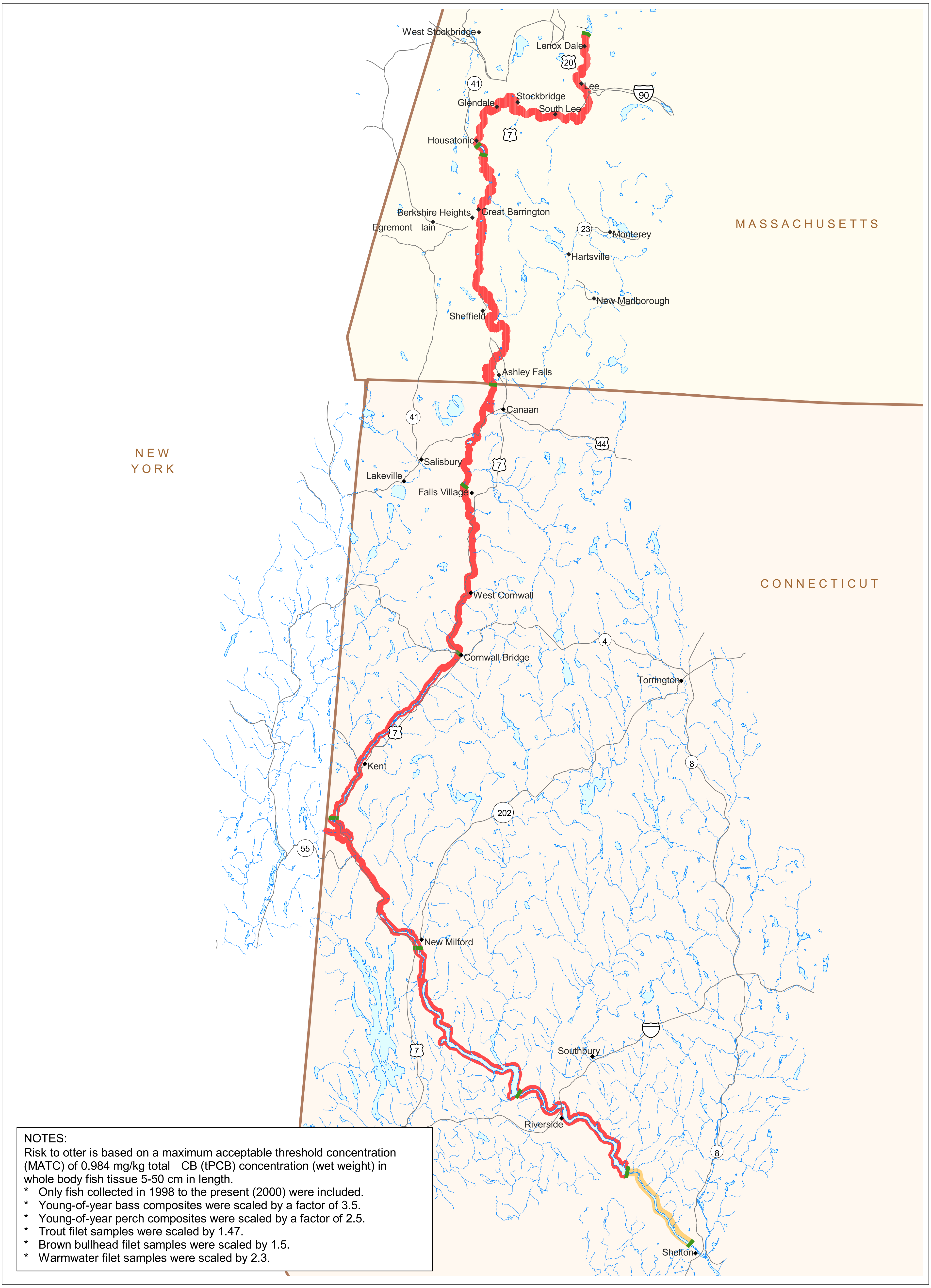
The mink MATC of 0.984 mg/kg tPCB in fish (whole body, wet weight) was also used for river otter. Potential river otter habitat downstream of the PSA is less abundant than for mink and is associated with larger wetland systems, with slower flowing water, or with impounded water. Any places where the river is impounded, or near a lake or pond, there is potential river otter habitat. Mean fish concentrations were calculated for such areas in river reaches downstream of the PSA using available whole body fish tissue data from samples collected since 1998. The analysis was restricted to fish with an overall body length between 5 and 50 cm, corresponding to the size commonly preyed on by otter. In some cases, it was necessary to use YOY data for bass and perch, and these were extrapolated to adult concentrations using the factors discussed above.

For this analysis it was assumed that otter were exposed to the mean fish concentrations in the downstream reaches for 100% of the diet because the majority (80%) of the otter diet is fish, with most of the remainder (8 to 20%) composed of crayfish. No crayfish data were available for the downstream reaches; however, within the PSA, crayfish tPCB concentrations were similar to fish concentrations in the size range consumed by otter. Therefore, the assumption of 100% of the dietary exposure at the mean fish concentrations in the downstream reaches was reasonable.

The results of this evaluation for otter are shown in Figure 12.2-11. Potential risk to otter due to consumption of contaminated fish occurs from the Woods Pond Dam downstream to the end of Reach 15.

12.2.3.7 Bald Eagle

An MATC of 30.4 mg/kg tPCBs (whole body fish tissue, wet weight) (Appendix K) was developed for wintering bald eagles, assuming that the eagle diet was composed of 78% fish, and that the remainder of the diet included other non-aquatic species that were assumed, for the purpose of this analysis, to be uncontaminated.



NOTES:
Risk to otter is based on a maximum acceptable threshold concentration (MATC) of 0.984 mg/kg total CB (tPCB) concentration (wet weight) in whole body fish tissue 5-50 cm in length.
* Only fish collected in 1998 to the present (2000) were included.
* Young-of-year bass composites were scaled by a factor of 3.5.
* Young-of-year perch composites were scaled by a factor of 2.5.
* Trout filet samples were scaled by 1.47.
* Brown bullhead filet samples were scaled by 1.5.
* Warmwater filet samples were scaled by 2.3.

LEGEND:

OTTER RISK
- < 0.984 mg/kg
- >= 0.984 mg/kg

- Town
- Reach Break
- Roads
- Housatonic River Basin Hydrology
- State Boundary

N W E S

2 0 2 4 6 8 Miles

Ecological Risk Assessment
GE/Housatonic River Site
Rest of River

FIGURE 12.2-11
ASSESSMENT OF RISK TO OTTER EXPOSED TO tPCBs DOWNSTREAM OF THE PSA

This concentration was compared with the available fish tissue concentration data from areas downstream of the PSA, in some cases applying scaling factors as discussed above for other receptors. Only data from samples collected since 1998 were used, and fish less then 12 cm total length were excluded from the analysis (unless appropriately scaled) to reflect the common size of fish preyed on by eagles.

A more in-depth analysis was performed for Reaches 14 and 15 where bald eagles have nested. Bald eagles on average consume a summer diet consisting of 78.2% fish, 16.3% birds, and 5% mammals (see Section K2.1.5). Mammal and bird tPCB concentrations were not available for downstream reaches. Total PCB concentrations for these prey items were estimated in three ways to give high, moderate, and low concentrations. High concentrations assumed that waterfowl and mammals from downstream would have tPCB concentrations equal to those in the PSA. Low concentrations assumed that waterfowl and mammals from downstream would have tPCB concentrations of zero. A moderate concentration was developed by determining fish-to-mammal and fish-to-bird ratios based on concentrations in the PSA. Mammal tPCB concentrations in the PSA are on average 75% of the total fish concentration, and waterfowl tPCB concentrations averaged 15% of the total fish concentration. Therefore, moderate tPCB concentrations downstream were 0.539 mg/kg for mammals and 0.108 mg/kg for birds.

The results of the evaluation are shown in Figure 12.2-12. Potential risks to bald eagles from consuming contaminated fish in areas downstream of the PSA are restricted to Reach 8, corresponding to Rising Pond. However, Rising Pond is smaller than the typical eagle foraging area so this estimate of risk is conservative. It is likely that eagles nesting at Rising Pond would be exposed through a foraging area including but greater than Rising Pond. The concentrations of COCs in fish tissue in the adjacent subreaches of the river are not known, but are expected to be elevated. Therefore, the risks to eagles with a foraging area associated with Rising Pond are uncertain. In addition, the more in-depth analysis specific to Reaches 14 and 15 also did not show risk in the foraging area of the nesting bald eagles.

NOTES:
Risk to eagles is based on a maximum acceptable threshold concentration (MATC) of 30.4 mg/kg total CB (tPCB) concentration (wet weight) in whole body fish tissue greater than or equal to 12 cm.
* Only fish collected in 1998 to the present (2000) were included.
* Young-of-year bass composites were scaled by a factor of 3.5.
* Young-of-year perch composites were scaled by a factor of 2.5.

LEGEND:

EAGLE RISK
- < 30.4 mg/kg
- >= 30.4 mg/kg

- Town
- Reach Break
- Roads
- Housatonic River Basin Hydrology
- State Boundary

2 0 2 4 6 8 Miles

Ecological Risk Assessment
GE/Housatonic River Site
Rest of River

FIGURE 12.2-12
ASSESSMENT OF RISK TO BALD EAGLE EXPOSED TO tPCBs DOWNSTREAM OF THE PSA

| O:\gepitt\aprs\era_species_082504.apr | layout - eagle k4-5 | o:\gepitt\epsfiles\plots\in\fig_12-2-12.eps | 3:31 PM, 11/2/2004 |

12.3 SPECIES SENSITIVITY AND MECHANISMS OF TOXICITY

There is a large amount of variability in the toxicokinetic responses of different species to tPCBs and 2,3,7,8-TCDD equivalence (TEQ). Numerous studies have shown that tPCBs and TEQ may cause a variety of adverse effects (e.g., Bosveld and Van den Berg 1994; Newsted et al. 1995; Van den Berg et al. 1998). Effects may include:

- Lethality.
- Hepatic lesions.
- Immunotoxicity.
- Tumor promotion.
- Adverse effects on reproduction.
- Induction of drug-metabolizing enzymes.

How PCB and TEQ congeners cause these effects, and the ability of different species to defend against these contaminants is less clear. A brief description of the primary toxic mechanism of coplanar PCBs, dioxins, and furans is provided in Section 12.2.3.1 (see also Section 2), as is information on the relative sensitivities of biota to tPCBs and TEQ.

For some biological endpoints, the linkage between the adverse response and a specific contaminant or group of contaminants is strong. In other cases, multiple contaminants may elicit similar responses, and causation is less clear. For example, the observation of fully external swim bladders (in fish from contaminated reaches of the Housatonic River) is consistent with studies of dioxin-like contaminants, but not other COCs (Tillitt et al. 2003a,b). Conversely, the observation of lesions and hepatic alterations in bottom fish is consistent with field studies of both PCBs and PAHs.

Although sensitivity to COCs undoubtedly explains some of the differences in effects and resulting risk experienced by biota in the PSA (see Figures 12.2-1 through 12.2-4), other factors also play a role. For example, higher trophic level biota that may forage exclusively in the PSA (e.g., belted kingfisher, mink) have higher exposures to tPCBs and TEQ than do biota with foraging areas of which only a portion is in the PSA (e.g., red fox).

Also, the composition and toxicity of the congener mixture that biota are exposed to changes with trophic level. The latter issue is briefly discussed in Section 12.2.3.2 and in more detail in Appendix C.7.

12.3.1 Mechanism of Action and Sensitivity of Species to tPCBs and TEQ

Some chlorinated PCBs, dioxins, and furans belong to a large class of chemicals called planar chlorinated hydrocarbons (PCHs) that are regularly detected in the environment. PCHs have a common structural relationship that includes lateral halogenation (i.e., the addition of chlorine to the compound) and the ability to assume a planar conformation. This structure is important as it leads to a common mechanism of action in many animal species involving binding to the aryl hydrocarbon (Ah) receptor and elicitation of an Ah-receptor-mediated biochemical and toxic response (Van den Berg et al. 1998; Newsted et al. 1995; Safe 1994). The planar conformation is the factor that controls the ability of the the chemical to bind with the Ah receptor (Birnbaum and Devito 1995; Newsted et al. 1995). The Ah receptor facilitates the translocation of PCHs into the nucleus of affected cells and the binding of the PCH-Ah receptor complex to sites on the DNA (Newsted et al. 1995).

Exposure to PCBs and other organic toxins in vertebrates elicits a response of the cytochrome P450 system with associated mixed function oxidases (MFO). MFOs enhance the elimination of some hydrophobic chemicals through a series of oxidative reactions (Eisler 2000). However, the MFO system is less capable of breaking down congeners with chlorine substitution at the 2, 3, 7, and 8 positions. As a result, these coplanar congeners show resistance to metabolic breakdown in many higher organisms (Bosveld and Van den Berg 1994).

The development of the cytochrome P450 system varies between species of vertebrates. Therefore, some species may be more sensitive to tPCBs and TEQ than others, even within taxonomic families. For example, mustelids may vary widely in sensitivity to PCBs (Leonards et al. 1997). Mink are among the most sensitive species to PCBs known (Aulerich et al. 1985; Giesy and Kannan 1998). Conversely, ferrets are much less sensitive to PCBs than mink (Bleavins et al. 1980). There are, however, few data available for other mustelid species. Foxes and dogs have been shown to have an unusual P450 isoenzyme that allows them to degrade PCB-153 more efficiently than rats and monkeys (Georgii et al. 1994). In general, fish are less capable of metabolizing PCBs than most birds and mammals (Van den Berg et al. 1998). Despite their reduced ability to metabolize PCBs, fish are relatively insensitive to mono-ortho PCBs, compared to birds and mammals (Van den Berg et al. 1998).

Fewer studies have been conducted on amphibians than on mammals, fish, and birds. The ability of amphibians to metabolize organic contaminants appears to be comparable to that of fish, but lower than that of rats (Eisler 2000). In amphibians, effects on the neutrophil function (i.e., immunosuppression) may be important (Angermann and Matsumura 1999). There is evidence that indicates that amphibians contain the Ah receptor, but it is not as well described because of limited research (Jung 1997). PCB-126 induces cytochrome P450 activity in both leopard frogs and green frogs (Huang et al. 2001). Amphibians have a cytochrome P450 mixed function oxidase that is less active and well developed than mammals, but that does not appear to be significantly different from other vertebrates (Eisler 2000). Benthic invertebrate toxicity is not mediated by an Ah receptor mechanism, and TEF systems have not been developed for amphibians or benthic invertebrates.

CCME (1999) reviewed the toxicology literature for mammals and birds and suggested that bird species may be less sensitive to the effects of PCBs than mammals. Some bird species such as tree swallows appear to be quite tolerant of elevated exposures to tPCBs and TEQ (Custer et al. 1998; Bishop et al. 1995, 1999; McCarty and Secord 1999). Substantial differences in sensitivity to PCBs between bird species have also been noted. Barron et al. (1995) determined that differences in the genetic expression of the Ah receptor were the dominant factor explaining differences in PCB sensitivity of the bird species examined. Brunström (1988, 1989) examined the sensitivity of numerous species of avian embryos to coplanar PCBs and concluded that interspecies differences in sensitivity were due to differences in the Ah receptor ligands.

Therefore, the available literature indicates that there are differences in sensitivity of biota, and that these differences may be partially attributed to differences in development of the cytochrome P450 system and other factors. The differences in sensitivity of biota partially explain why, for example, mink are experiencing much greater effects from exposure to tPCBs and TEQ in the PSA than are belted kingfishers (Figures 12.2-3 and 12.2-4), despite the two species having similar diets. Similarly, tree swallows and small-footed myotis have similar diets, yet the more tolerant tree swallows are likely to experience lower risks from exposure to tPCBs and TEQ than does small-footed myotis (Figures 12.2-3 and 12.2-4).

Although differences in sensitivity of biota can partly be explained on the basis on differential development of Ah receptor and cytochrome P450 systems, toxicity from non-coplanar congeners not associated with these systems is also important. Observations of effects in aquatic receptors that are either unrelated to Ah receptor interactions (e.g., benthic invertebrates) or are greater than would be predicted on the basis of congeners with TEF values only (e.g., fish) are evidence of this. Detailed knowledge of PCB toxicity for all 209 PCB congeners does not exist; differential toxicity to non-coplanar congeners may explain some of the interspecies differences in toxicity observed.

12.3.2 Congener Composition and Toxicity to Biota

Environmental degradation (or weathering) of PCH congeners varies due to the unique physical/chemical properties of each congener (Cogliano 1998). This can cause differences between the congeners detected in environmental samples and the congener makeup of the original product or Aroclor (Cogliano 1998; Van den Berg et al. 1998). In the Housatonic River PSA, PCB composition exhibits little spatial variability within a medium (e.g., in sediment between reaches), although there are some shifts in composition across media (Appendix C.7). Between receptor groups, PCB congener composition may exhibit some variations. The change in the congener composition in prey tissue can produce differences in toxicological responses in exposed predator species relative to the effects observed in the laboratory for species exposed to technical mixtures (e.g., Aroclor 1254 or 1260).

In the site-specific fish studies conducted for this assessment, the congener composition in fish was found to be more toxic than would be expected from studies exposing fish to Aroclor 1254 and 1260 commercial mixtures (Appendix F). Conversely, the results of the mink feeding study (Appendix I) indicated that the congener composition in fish tissue from the PSA was less toxic than would be expected from mink toxicity studies conducted with mixtures expected to be similar. For example, the LD_{10} and LD_{20} for tPCBs estimated from the mink feeding study are 1.4 to 2.7 times higher than the LD_{10} and LD_{20} for tPCBs estimated from the results of chronic feeding studies conducted with Aroclor 1254 (see Section I.3.3.2 in Appendix I).

12.4 BROADER IMPLICATIONS

The weight-of-evidence assessments briefly described in Sections 3 through 11, and in more detail in Appendices D through K, indicate that COCs in the PSA of the Housatonic River, particularly tPCBs, are causing risks to many of the species chosen to represent the assessment endpoints (see Figures 12.2-1 through 12.2-4, Table 12.2-1). Risks from COCs, however, may potentially extend beyond adverse effects to survival, growth, and reproduction of representative species. The purpose of this section is to explore the implications of the risks of COCs to representative species demonstrated in the preceding sections. This section begins by extending the ecological risk assessment to species that occur in the Housatonic River watershed, but that had not been considered explicitly in the quantitative ecological risk assessments previously described in Sections 3 through 11. This section is followed by a general discussion of the possible broader ecological implications of this risk assessment.

12.4.1 Implications for Other Species in the Primary Study Area

The purpose of this section is to qualitatively compare exposure of the representative species to other species that were identified in the Ecological Characterization (Appendix A) to occur in the PSA for tPCBs and TEQ. The major factors that influence exposure to tPCBs and TEQ and that were considered in the analysis include:

- Dietary composition.
- Foraging behavior and home range.
- Size, metabolism, and life history characteristics.
- Sensitivity to COCs.

The following sections briefly compare these factors between the representative species and other species in their foraging groups. The comparison highlights similarities and differences, and their potential to influence exposure and thus risks from tPCBs and TEQ. This comparison does not consider differences in sensitivity between representative species and other species in their foraging groups because toxicity data to make this comparison are lacking. Table 12.4-1 summarizes this qualitative risk assessment. More discussion is presented in Appendices D through K.

Table 12.4-1

Comparison of Risks of tPCBs and TEQ to Representative and Other Species in the Housatonic River PSA

Category	Representative Species and Risk Category in PSA[a]	Other Species	Size	Residency	Foraging/ Home Range	Diet	Life History/ Miscellaneous	Level of Risk Compared to Representative Species					References
								Lower	Lower to Similar	Similar	Similar to Higher	Higher	
Top Predator Fish	Largemouth bass/ Intermediate to High		10-16 inches (21-41 cm)	Year-round	n/a	Aquatic insects, fish, crayfish		n/a					Hartel et al. 2002
		Smallmouth bass	8-13 inches (20-33 cm)	Year-round	n/a	Aquatic invertebrates, fish	Prefers cooler, clearer, rockier areas than largemouth bass (LMB), but similar			X			Hartel et al. 2002
		Black crappie	8-12 inches (20-30 cm)	Year-round	n/a	Aquatic invertebrates, fish	Prefers cooler, clearer, rockier areas than LMB, but diet is similar			X			Hartel et al. 2002
		Rock bass	6-8 inches (15-20 cm)	Year-round	n/a	Aquatic invertebrates, fish, crayfish	Diet and some habitat preferences are similar, particularly to young (3-4 y.o.) LMB			X			Hartel et al. 2002
		Chain pickerel	15-24 inches (38-61 cm)	Year-round	n/a	Invertebrates, fish	Similar diet compared to LMB			X			Hartel et al. 2002
Omnivorous Bottom Feeders	Brown bullhead/ Intermediate to High		8-14 inches (20-36 cm)	Year-round	n/a	Animal and plant material (up to 40% plants, up to 60% filamentous algae)		n/a					Hartel et al. 2002; Gunn et al. 1977
		Yellow bullhead	8-12 inches (20-30 cm)	Year-round	n/a	Insects, crustaceans, mollusks, small fish, plant material (up to 40% plants)	Similar habitat, although typically prefers clearer water than brown bullhead. Diet is similar			X			Hartel et al. 2002

Table 12.4-1

Comparison of Risks of tPCBs and TEQ to Representative and Other Species in the Housatonic River PSA (Continued)

Category	Representative Species and Risk Category in PSA[a]	Other Species	Size	Residency	Foraging/ Home Range	Diet	Life History/ Miscellaneous	Level of Risk Compared to Representative Species: Lower	Lower to Similar	Similar	Similar to Higher	Higher	References
Omnivorous Bottom Feeders (cont.)	Brown bullhead/ Intermediate to High (cont.)	Common carp	24 inches (61 cm)	Year-round	n/a	Animal and plant material	Habitat and diet similar to brown bullhead			X			Hartel et al. 2002
		Goldfish	5-13 inches (13-33 cm)	Year-round	n/a	Animal and plant material	Habitat and diet similar to brown bullhead			X			Hartel et al. 2002
	White sucker/ Intermediate to High		12-24 inches (30-61 cm)	Year-round	n/a	Aquatic invertebrates, larval insects, detritus				n/a			Hartel et al. 2002
		Longnose sucker	12-15 inches (30-38 cm)	Year-round	n/a	Aquatic invertebrates, algae	Prefers cooler, cleaner stream reaches than white sucker. Diet is similar to white sucker			X			Hartel et al. 2002
		Creek chubsucker	9 inches (23 cm)	Year-round	n/a	Omnivorous	Habitat and diet similar to white sucker			X			Hartel et al. 2002
Forage Fish	Pumpkinseed/ Intermediate to High		4-5 inches (10-13 cm)	Year-round	n/a	Aquatic invertebrates				n/a			Hartel et al. 2002
		Bluegill	5-7 inches (13-18 cm)	Year-round	n/a	Aquatic invertebrates, some small fish	Habitat and diet similar to pumpkinseed			X			Hartel et al. 2002
		Redbreast sunfish	4-8 inches (10-20 cm)	Year-round	n/a	Larvae and adult aquatic insects, terrestrial insects, some small fish	Habitat and diet similar to pumpkinseed			X			Hartel et al. 2002

Table 12.4-1

Comparison of Risks of tPCBs and TEQ to Representative and Other Species in the Housatonic River PSA (Continued)

Category	Representative Species and Risk Category in PSA[a]	Other Species	Size	Residency	Foraging/ Home Range	Diet	Life History/ Miscellaneous	Level of Risk Compared to Representative Species					References
								Lower	Lower to Similar	Similar	Similar to Higher	Higher	
Amphibians	Wood frog/ High		18 g	Year-round, larvae: 3 months	64.5 m^2	Insects, beetles, flies, slugs, snails, spiders, bugs, moth larvae, and earthworms	Small home range, but establishes territories >1,000 m from breeding pools	n/a					DeGraaf and Yamasaki 2001; Hunter et al. 1999
		Northern spring peeper	1.0-2.7 g	Year-round, larvae: 3 months	23 m^2	Spiders (up to 50%), ants, beetles, mites, ticks, springtails, caterpillars, slugs, and snails	Territories usually established near suitable breeding sites. Late summer/fall migrations to hibernation sites may be further away				X		DeGraaf and Yamasaki 2001; Hunter et al. 1999
		Spotted salamander	14 g	Year-round, larvae: 3 months	10-14 m^2	Forest floor invertebrates: earthworms, slugs, snails, spiders, millipedes, centipedes, larval and adult insects	Home ranges usually established within 200 m of breeding site				X		Petranka 1998; DeGraaf and Yamasaki 2001; Hunter et al. 1999

Table 12.4-1

Comparison of Risks of tPCBs and TEQ to Representative and Other Species in the Housatonic River PSA (Continued)

Category	Representative Species and Risk Category in PSA[a]	Other Species	Size	Residency	Foraging/ Home Range	Diet	Life History/ Miscellaneous	Level of Risk Compared to Representative Species					References
								Lower	Lower to Similar	Similar	Similar to Higher	Higher	
Amphibians (cont.)	Wood frog/ High (cont.)	Jefferson salamander	11 g	Year-round, larvae: 3 months		Small invertebrates, worms, spiders, insects, crustaceans	Home ranges typically within 250 m from breeding pond, but have been recorded up to 624 m away				X		Petranka 1998; DeGraaf and Yamasaki 2001; Hunter et al. 1999
	Northern leopard frog/ High		38 g	Year-round, larvae: 3 months	5 to 53 m nightly movement	Beetles (up to 50%), lepidopteran larvae, bugs, grasshoppers, ants, spiders, crayfish, snails	Semi-terrestrial spending summer month in damp fields and woods, hibernates and breeds in permanent bodies of water			n/a			DeGraaf and Yamasaki 2001; Hunter et al. 1999
		Pickerel frog	Not found	Year-round, larvae: 3 months	Not found	Insects, beetles, caterpillars, true bugs, ants, spiders, snails, crayfish, and amphipods	Habitat preferences and feeding habits similar to leopard frog				X		DeGraaf and Yamasaki 2001; Hunter et al. 1999
		Green frog	30-70 g	Year-round, larvae: 1-2 years	61 m^2 home range, 2-3 m breeding territory	Adults: plants, spiders, beetles, true bugs, wasps and bees, mosquitoes, flies, midges and gnats, mayflies, moths, and butterflies	Green frogs are more aquatic than leopard frogs. They do enter the floodplain to access seasonally available food resources				X		DeGraaf and Yamasaki 2001; Hunter et al. 1999; Stewart and Sandison 1973; Jenssen and Klimstra 1966.

Table 12.4-1

Comparison of Risks of tPCBs and TEQ to Representative and Other Species in the Housatonic River PSA (Continued)

Category	Representative Species and Risk Category in PSA[a]	Other Species	Size	Residency	Foraging/ Home Range	Diet	Life History/ Miscellaneous	Level of Risk Compared to Representative Species					References
								Lower	Lower to Similar	Similar	Similar to Higher	Higher	
Amphibians (cont.)	Northern leopard frog/ High (cont.)	Eastern newt	2-3 g	Year-round, larvae: 2 months, efts 3-7 years	Adults: captured within 7 m of original capture sites, efts: 270 m^2 up to 800 m from breeding sites	Adults: mayflies, damselflies, dragonflies, mosquitoes, midges, gnats, water fleas, amphipods, bivalves, and clams Newt larvae efts: snails, slugs, mites, ticks, beetles, beetle larvae, flies, mosquitoes, midges, gnats, maggots, wasps, and bees	Larval period is spent in pools; metamorph into terrestrial eft stage that lasts 2-7 years. A second metamorphosis occurs when adults return to pools and transform into aquatic adults				X		Petranka 1998; DeGraaf and Yamasaki 2001; Hunter et al. 1999; Burton 1977; MacNamara 1977; Bellis 1968; Healy 1975
Insectivorous Birds	Tree swallow/ Intermediate to High		20.1 g (range 16.5-25.5 g)	6 months	Defend 10-15 m around nest; feed within 300-400 m of nest	Primarily emergent insects, such as flies, mosquitoes, midges, gnats, mayflies, and beetles	Feed on emergent insects over bodies of water			n/a			DeGraaf and Yamasaki 2001; Robertson et al. 1992; Ehrlich et al. 1988; Martin et al. 1951
		Bank swallow	13.5 g	5 months	Territory limited to immediate vicinity of the nest entrance	Almost entirely insects	Nest in exposed and eroding riverbanks and in gravel pits				X		DeGraaf and Yamasaki 2001; Ehrlich et al. 1992

Table 12.4-1

Comparison of Risks of tPCBs and TEQ to Representative and Other Species in the Housatonic River PSA (Continued)

Category	Representative Species and Risk Category in PSA[a]	Other Species	Size	Residency	Foraging/ Home Range	Diet	Life History/ Miscellaneous	Level of Risk Compared to Representative Species					References
								Lower	Lower to Similar	Similar	Similar to Higher	Higher	
Insectivorous Birds (cont.)	Tree swallow/ Intermediate to High (cont.)	Northern rough-winged swallow	16 g	5 months	Nest within 1 km of water	Entirely insects	Nest in exposed and eroding riverbanks and in gravel pits				X		DeGraaf and Yamasaki 2001; Ehrlich et al. 1992
		Barn swallow	19 g	5 months	Seldom feed more than 0.8 km from nest site	Insects, occasionally berries and seeds	Nest under bridges in PSA and feed over river and fields			X			DeGraaf and Yamasaki 2001; Ehrlich et al. 1992
		Cliff swallow	21 g	5 months	Foraging range typically within 1 km	Almost entirely insects, but occasionally gorge on berries	Nest under bridges in PSA and feed over river and fields	X					DeGraaf and Yamasaki 2001; Ehrlich et al. 1992
		Chimney swift	23 g	5 months	Foraging range up to several kilometers	Flying insects					X		DeGraaf and Yamasaki 2001; Ehrlich et al. 1992; Sibley 2001
		Common nighthawk	62 g	5 months	Territory 2-23 ha	Flying insects, especially flying ants, mosquitoes, moths, grasshoppers	Nests on rooftops in town, feeds over fields and water			X			DeGraaf and Yamasaki 2001; Ehrlich et al. 1992
		Eastern kingbird	40 g	6 months	Territory 5.7-14.2 ha	Flying insects, some fruit	Commonly nests in trees overhanging the river in the PSA, capture insects over the river. Also occurs in agricultural areas, over fields	X					DeGraaf and Yamasaki 2001; Ehrlich et al. 1992; Martin et al. 1951

Table 12.4-1

Comparison of Risks of tPCBs and TEQ to Representative and Other Species in the Housatonic River PSA (Continued)

Category	Representative Species and Risk Category in PSA[a]	Other Species	Size	Residency	Foraging/ Home Range	Diet	Life History/ Miscellaneous	Level of Risk Compared to Representative Species: Lower	Lower to Similar	Similar	Similar to Higher	Higher	References
Insectivorous Birds (cont.)	Tree swallow/ Intermediate to High (cont.)	Eastern phoebe	20 g	7 months	0.3 ha in an Illinois flood-plain; 1.3-2.9 ha in settled area	92-97% flying insects from spring through fall, mostly berries and seeds in winter				X			DeGraaf and Yamasaki 2001; Ehrlich et al. 1992; Martin et al. 1951
	American robin/High		77 g	Year-round	Territory: 0.4 ha; foraging range for nestlings and fledg-lings; 0.15 and 0.81 ha	50-90% animal matter in spring and summer, switches to plants (berries) in fall and winter. Prey includes earthworms, butterflies, moths, beetles, and ants		n/a					DeGraaf and Yamasaki 2001; Sallabanks and James 1999; Ehrlich et al. 1992; Weatherhead and McRae 1990; Martin et al. 1951
		Eastern bluebird	31 g	8 months	Territory 2.2-3.5 ha	60% or more animal matter year-round, up to 80-95% in spring and summer. Prey includes beetles, grasshoppers, crickets, and caterpillars	Diet similar to robin except earthworms rarely consumed		X				DeGraaf and Yamasaki 2001; Ehrlich et al. 1992; Martin et al. 1951
		Eastern towhee	40 g	6 months	Territory 0.26-2.4 ha	50:50 plant and animal in summer and fall. Mostly terrestrial insects, seeds, and berries	Tends to consume considerably less animal matter than robin	X					DeGraaf and Yamasaki 2001; Ehrlich et al. 1992; Martin et al. 1951

Table 12.4-1

Comparison of Risks of tPCBs and TEQ to Representative and Other Species in the Housatonic River PSA (Continued)

Category	Representative Species and Risk Category in PSA[a]	Other Species	Size	Residency	Foraging/ Home Range	Diet	Life History/ Miscellaneous	Level of Risk Compared to Representative Species					References
								Lower	Lower to Similar	Similar	Similar to Higher	Higher	
Insectivorous Birds (cont.)	American robin/High (cont.)	Gray catbird	37 g	7 months	Territory 0.06-0.32 ha	40-80% animal matter in spring and summer, fall diet nearly 80% plants (berries). Prey includes largely terrestrial insects (ants, beetles), caterpillars, and grasshoppers	Diet similar to robin except earthworms rarely consumed				X		DeGraaf and Yamasaki 2001; Ehrlich et al. 1992; Martin et al. 1951
		Hermit thrush	31 g	7 months	Territory 0.06-3.34 ha	93 and 85% animal matter in spring and summer, respectively. Dominant prey includes beetles, ants, caterpillars, flies, and insects	Diet similar to robin with the exception of earthworms			X			DeGraaf and Yamasaki 2001; Ehrlich et al. 1992; Martin et al. 1951
		Northern mockingbird	49 g	Year-round	Territory 0.25-0.5 ha in summer	70-85% animal matter in spring and summer. Dominant prey includes beetles, ants, bees, wasps, and grasshoppers	Diet similar to robin except earthworms rarely consumed			X			DeGraaf and Yamasaki 2001; Ehrlich et al. 1992; Martin et al. 1951

Table 12.4-1

Comparison of Risks of tPCBs and TEQ to Representative and Other Species in the Housatonic River PSA (Continued)

Category	Representative Species and Risk Category in PSA[a]	Other Species	Size	Residency	Foraging/ Home Range	Diet	Life History/ Miscellaneous	Level of Risk Compared to Representative Species					References
								Lower	Lower to Similar	Similar	Similar to Higher	Higher	
Insectivorous Birds (cont.)	American robin/High (cont.)	Veery	31 g	5 months	Territory 0.1-3+ ha	60-95% animal matter in spring and summer. Dominant prey includes beetles, ants, caterpillars, spiders, and grasshoppers	Diet similar to robin except earthworms rarely consumed			X			DeGraaf and Yamasaki 2001; Ehrlich et al. 1992; Martin et al. 1951
		Wood thrush	47 g	6 months	Territory 0.08-2.8 ha	60-95% animal matter in spring and summer. Dominant prey includes beetles, ants, caterpillars, spiders, and grasshoppers	Diet similar to robin except earthworms rarely consumed			X			DeGraaf and Yamasaki 2001; Ehrlich et al. 1992; Martin et al. 1951

Table 12.4-1

Comparison of Risks of tPCBs and TEQ to Representative and Other Species in the Housatonic River PSA (Continued)

Category	Representative Species and Risk Category in PSA[a]	Other Species	Size	Residency	Foraging/ Home Range	Diet	Life History/ Miscellaneous	Level of Risk Compared to Representative Species					References
								Lower	Lower to Similar	Similar	Similar to Higher	Higher	
Insectivorous Birds (cont.)	Wood Duck / High	Wood Duck	0.546 kg	9 months March - November	Foraging area: 1 km from nest for breeding females	Breeding female wood duck diet is comprised of 76% invertebrates, 23% vegetation in spring – early summer. Dominant prey includes beetles, flies, snails, and isopods Males and non-breeding females generally have a high proportion of vegetation, particularly seeds							Drobney and Fredrickson 1979; Gilmer et al. 1978; Grice and Rogers 1965; Landers et al. 1977.
		American Black Duck	0.72 – 1.6 kg	6 months: November to April	Foraging area: 2 ha	Vegetation, particularly seeds, aquatic and terrestrial invertebrates, and occasional fish	Diet similar to wood ducks, but has greater body weight and foraging range			X			Anderson and Titman 1992; Bellrose 1981; Longcore et al. 2000

Table 12.4-1

Comparison of Risks of tPCBs and TEQ to Representative and Other Species in the Housatonic River PSA (Continued)

Category	Representative Species and Risk Category in PSA[a]	Other Species	Size	Residency	Foraging/ Home Range	Diet	Life History/ Miscellaneous	Level of Risk Compared to Representative Species					References
								Lower	Lower to Similar	Similar	Similar to Higher	Higher	
Insectivorous Birds (cont.)	Wood Duck / High (cont.)	Canada Goose	0.95 – 9 kg	Year round resident	NA	Terrestrial vegetation except during the pre-nesting and nesting season when they are partially insectivorous	Larger than wood duck and consumes primarily vegetation	X					Ehrlich et al. 1988; Mowbray et al. 2002
		Common goldeneye	0.6 – 1.3 kg	Migratory visitor	Territory: Variable depending upon habitat quality. No defense of specific territory.	Primarily consists of aquatic invertebrates with lesser amounts of vegetation and fish	Aside from fish, diet is similar to the wood duck	X					Eadie et al. 1995
		Common merganser	0.9 – 2.2 kg	Migratory visitor	Prefer wide spacing between breeding pairs. No quantitative estimate available.	Consists primarily of fish and some aquatic invertebrates	Wood ducks do not consume fish	X					Mallory and Metz 1999
		Hooded merganser	0.75 kg	8-9 months	NA	Aquatic invertebrates with lesser quantities of fish	Aside from fish, diet is similar to the wood duck					X	Bellrose 1981

Table 12.4-1

Comparison of Risks of tPCBs and TEQ to Representative and Other Species in the Housatonic River PSA (Continued)

Category	Representative Species and Risk Category in PSA[a]	Other Species	Size	Residency	Foraging/ Home Range	Diet	Life History/ Miscellaneous	Level of Risk Compared to Representative Species					References
								Lower	Lower to Similar	Similar	Similar to Higher	Higher	
Insectivorous Birds (cont.)	Wood Duck / High (cont.)	Green-winged Teal	0.14 – 0.5 kg	Migratory visitor	NA	Composed primarily of seeds and vegetation with some aquatic invertebrates	Upper range of body weight is similar to wood duck. Consume less aquatic invertebrates than wood duck	X					Johnson 1995; Bellrose 1981
		Mallard	1.3 kg	Year round resident	Foraging range: 16 ha	Aquatic and terrestrial plants and aquatic invertebrates	Greater body weight and foraging range. Diet is similar to wood duck	X					Bellrose 1981; Godfrey 1986; Drilling et al. 2002
Omnivorous and Carnivorous Mammals	Northern short-tailed shrew/High		20.5 g	Year-round	0.024-0.2 ha	Common prey includes insects and earthworms. Also forage on snails, slugs, crustaceans, small mammals, fungi, and, rarely, vegetation	Occurs in damp woodlands and fields			n/a			DeGraaf and Yamasaki 2001; Whitaker and Hamilton 1998; Kurta 1995; EPA 1993; Whitaker and Ferraro 1963; Hamilton 1941; Linzey and Linzey 1973; Eadie 1944; Eadie 1948
		Smoky shrew	6.1-11 g	Year-round	Not found	Insectivorous; also salamanders, young mice, vegetable matter	Habitat preferences and diet similar to short-tailed shrew					X	DeGraaf and Yamasaki 2001; Whitaker and Hamilton 1998; Kurta 1995

Table 12.4-1

Comparison of Risks of tPCBs and TEQ to Representative and Other Species in the Housatonic River PSA (Continued)

Category	Representative Species and Risk Category in PSA[a]	Other Species	Size	Residency	Foraging/ Home Range	Diet	Life History/ Miscellaneous	Level of Risk Compared to Representative Species: Lower	Lower to Similar	Similar	Similar to Higher	Higher	References
Omnivorous and Carnivorous Mammals (cont.)	Short-tailed shrew/High (cont.)	Masked shrew	4.0-6.5 g	Year-round	0.16-0.28 ha	Predominantly insectivorous; also mollusks, annelids, dead bodies of larger animals, salamanders, young mice, *Endogone*, vegetable matter	Similar diet but more commonly in dryer uplands, meadows, old fields, and fencerows			X			DeGraaf and Yamasaki 2001; Whitaker and Hamilton 1998; Kurta 1995
	Red fox/ Intermediate		3.4-6.4 kg	Year-round	60-600 ha	Up to 30% plants in summer and fall, the remainder being small mammals, birds, and insects. Mammals average 76% of diet for all seasons	Prefers open agricultural land and forest edges	n/a					DeGraaf and Yamasaki 2001; Whitaker and Hamilton 1998; Kurta 1995; EPA 1993; Martin et al. 1951; Powell and Case 1982; Knable 1974; Korschgen 1959; Hockman and Chapman 1983; Dibello et al. 1990
		Coyote	9.1-22.7 kg	Year-round	1000-4000 ha	78% mammals, 21% fruit, 10% insects, and 3% birds by frequency in 1,500 scats from Adirondacks	Broad habitat requirements, open fields, agricultural land, forested areas	X					DeGraaf and Yamasaki 2001; Whitaker and Hamilton 1998; Kurta 1995; Martin et al. 1951

Table 12.4-1

Comparison of Risks of tPCBs and TEQ to Representative and Other Species in the Housatonic River PSA (Continued)

Category	Representative Species and Risk Category in PSA[a]	Other Species	Size	Residency	Foraging/ Home Range	Diet	Life History/ Miscellaneous	Level of Risk Compared to Representative Species: Lower	Lower to Similar	Similar	Similar to Higher	Higher	References
Omnivorous and Carnivorous Mammals (cont.)	Red fox/ Intermediate (cont.)	Gray fox	3.2-5.9 kg	Year-round	85-3200 ha	85-95% animal matter throughout the year (e.g., rabbit, squirrel)	Most common in forested areas			X			DeGraaf and Yamasaki 2001; Whitaker and Hamilton 1998; Kurta 1995; Martin et al. 1951
		Fisher	3.6-5.5 kg	Year-round	1500-3500 ha	Nearly 100% animal matter, including small mammals, squirrels, rabbits, porcupine, birds, reptiles, and amphibians	Prefer forested areas with closed canopies				X		DeGraaf and Yamasaki 2001; Whitaker and Hamilton 1998; Kurta 1995; Martin et al. 1951
		Long-tailed weasel	85-270 g	Year-round	31.9-160 ha	78% small mammals (mice, voles, shrews), 17% rabbits; also birds (up to 10%), squirrels, snakes, invertebrates	Terrestrial				X		DeGraaf and Yamasaki 2001; Whitaker and Hamilton 1998; Kurta 1995
		Short-tailed weasel	50-150 g	Year-round	Males: 17.0-25.0 ha Females: 10.1-14.9 ha	75% small mammals (mice, voles, shrews); also squirrels, rabbits, birds, amphibians, snakes, invertebrates	Terrestrial				X		DeGraaf and Yamasaki 2001; Whitaker and Hamilton 1998; Kurta 1995

Table 12.4-1

Comparison of Risks of tPCBs and TEQ to Representative and Other Species in the Housatonic River PSA (Continued)

Category	Representative Species and Risk Category in PSA[a]	Other Species	Size	Residency	Foraging/ Home Range	Diet	Life History/ Miscellaneous	Level of Risk Compared to Representative Species: Lower	Lower to Similar	Similar	Similar to Higher	Higher	References
Threatened and Endangered Species[b]	American bittern/High		370-500 g	5 months	Varies with geographic area, preferred habitat availability and prey species >3 ha reported for Michigan and 24.7 ha core use reported for Minnesota	Amphibians, small snakes, crayfish, insects, small fish	Consume a wide variety of prey items allowing them to hunt in varying habitats			n/a			Gibbs et al. 1992; Brown and Dinsmore 1986; Azure 1998 in Deschant et al. 2001; Gibbs et al. 1992 in DeGraaf and Yamasaki 2001
		Great blue heron	2,400 g	Year-round	Will feed kilometers from nest	Approximately 70% fish, also insects, crayfish, small mammals, and amphibians				X			DeGraaf and Yamasaki 2001; Ehrlich et al. 1992; Martin et al. 1951
		Green heron	210 g	5 months	Defends territory a few feet from nest	Crayfish approximately 50% of diet, 25% aquatic insects, and 20% small fish	Separate feeding territories may be vigorously defended					X	DeGraaf and Yamasaki 2001; Ehrlich et al. 1992; Martin et al. 1951

Table 12.4-1

Comparison of Risks of tPCBs and TEQ to Representative and Other Species in the Housatonic River PSA (Continued)

Category	Representative Species and Risk Category in PSA[a]	Other Species	Size	Residency	Foraging/ Home Range	Diet	Life History/ Miscellaneous	Level of Risk Compared to Representative Species: Lower	Lower to Similar	Similar	Similar to Higher	Higher	References
Threatened and Endangered Species (cont.)	American bittern/High (cont.)	Pied-billed grebe	450 g	7 months	Defends area 50 m around nest	Primarily animal matter, including crayfish, small fish, mollusks, aquatic insects						X	DeGraaf and Yamasaki 2001; Ehrlich et al. 1992; Martin et al. 1951
		Sora	75 g	5 months	Distances between nest ranges from 1.2 to 25 m	60% animal matter spring and fall. Prey includes beetles, snails, spiders, and crustaceans		X					DeGraaf and Yamasaki 2001; Odum 1977; Tanner and Hendrickson 1956 in DeGraaf and Yamasaki 2001; Berger 1951 in Degraaf and Yamasaki 2001; Ehrlich et al. 1992; Martin et al. 1951
		Virginia rail	85 g	5 months	Territorial during pair formation and nest establish-ment	90-95% animal matter spring and fall. Prey includes beetles, snails, spiders, true bugs, and diptera larvae. Also crustaceans, dragonfly and damselfly larvae, ants, grasshoppers, crickets, and small fish						X	DeGraaf and Yamasaki 2001; Ehrlich et al. 1992; Martin et al. 1951.

Table 12.4-1

Comparison of Risks of tPCBs and TEQ to Representative and Other Species in the Housatonic River PSA (Continued)

Category	Representative Species and Risk Category in PSA[a]	Other Species	Size	Residency	Foraging/ Home Range	Diet	Life History/ Miscellaneous	Level of Risk Compared to Representative Species: Lower	Lower to Similar	Similar	Similar to Higher	Higher	References
Threatened and Endangered Species (cont.)	Small-footed myotis/High		6 g, range 5-7 g	Year-round, but hibernate	Travel < 40 km between summer and winter grounds	Primarily midges, caddisflies, moths, butterflies, and beetles	Observed longevity of 12 years in the wild			n/a			DeGraaf and Yamasaki 2001; Griffith and Gates 1985; Anthony and Kunz 1977; Belwood and Fenton 1976; Kurta 1995; van Zyll de Jong 1985 in DeGraaf and Yamasaki 2001
		Big brown bat	15-24 g	Year-round, but hibernate	Forages 1 to 2 km from day roosts	Specialize in capturing beetles, but also take true bugs, wasps, bees, flies, mosquitoes, midges, gnats, moths, and butterflies	Travels short distances, usually no more than 48 to 80 km from maternity colony to hibernaculum site			X			DeGraaf and Yamasaki 2001; Whitaker 1995; Griffith and Gates 1985; Barbour and Davis 1969 in DeGraaf and Yamasaki 2001; Mills et al. 1975 in DeGraaf and Yamasaki 2001; Kurta 1995; Kurta and Baker 1990 in Degraaf and Yamasaki 2001; Kurta 1995
		Indiana myotis	6-11 g	Year-round, but hibernate.	Female range over 52-95 ha	Includes terrestrial insects as well as emergent aquatic insects	Forages in foliage of tree crowns along river and lake shores			X			DeGraaf and Yamasaki 2001; Kurta 1995

Table 12.4-1

Comparison of Risks of tPCBs and TEQ to Representative and Other Species in the Housatonic River PSA (Continued)

Category	Representative Species and Risk Category in PSA[a]	Other Species	Size	Residency	Foraging/ Home Range	Diet	Life History/ Miscellaneous	Level of Risk Compared to Representative Species: Lower	Lower to Similar	Similar	Similar to Higher	Higher	References
Threatened and Endangered Species (cont.)	Small-footed myotis/High (cont.)	Little brown bat	6-12 g	Year-round, but hibernate	Unknown	Midges, mayflies, caddisflies, and mosquitoes. Also beetles, moths, stoneflies, true bugs, and termites	Consume 1.8-3.7 g of food per night			X			DeGraaf and Yamasaki 2001; Griffith and Gates 1985; Anthony and Kunz 1977; Belwood and Fenton 1976; Kurta 1995

[a]For representative species with multiple assessments in the PSA, the highest risk category is listed here.

[b] Several of the species included in this section (i.e., sora, great blue heron, green heron, Virginia rail, northern myotis, little brown bat) are not threatened and endangered species either federally or in Massachusetts and Connecticut (Appendix A). They are included in the discussion of T&E species because they are taxonomically and ecologically similar to either American bittern or to small-footed myotis.

n/a = not applicable.

12.4.1.1 Benthic Invertebrates

The benthic invertebrate ERA included the entire benthic community; benthic community composition analysis was a measurement endpoint considered in the weight-of-evidence assessment, and the species sensitivity distribution method explicitly considered the range in observed organism sensitivities. Individual species were also used in toxicity tests as surrogates for the Housatonic River freshwater benthic community (i.e., *Chironomus tentans, Hyalella azteca, Daphnia magna, Ceriodaphnia dubia*). The toxicity test species and endpoints encompass a range of toxicological sensitivities; similar variation in sensitivity can be expected in the field. Both the status of sensitive taxa and community composition are considered indicators of the overall health and productivity of the benthic community. Thus, there is no need to extrapolate the findings of the benthic invertebrate assessment described previously to other benthic invertebrate species in the PSA.

12.4.1.2 Amphibians

Certain amphibian species that were not studied may be more susceptible to the effects of tPCBs because of their life history characteristics. For example, blue-spotted (*Ambystoma laterale*) and spotted salamanders (*Ambystoma maculatum*) have a lifestage as aquatic carnivorous, bottom-dwelling larvae. Thus, they could potentially bioaccumulate PCBs more quickly than herbivorous amphibians. The larvae of these two species forage on the bottom of vernal pools, and have greater opportunities to be in contact with contaminated sediment than do pelagic frog tadpoles. The salamander larvae feed on an assortment of planktonic animals, and then shift to larger aquatic worms, insect larvae, small crustaceans, and tadpoles as they grow larger (Hunter et al. 1999).

Blue-spotted and spotted salamanders also have longer larval periods, lasting between 70 to 100 days (Whitford and Vinegar 1966), than do wood frogs, which have a larval period averaging 67 days (Hunter et al. 1999). Salamanders appeared in lower numbers in vernal pools with high sediment tPCB concentrations (Woodlot, 2003a). Several salamander species occur in contaminated habitat in the PSA, including the spotted salamander, the Jefferson salamander (*Ambystoma jeffersonianum*, formerly considered a variety of blue-spotted salamander), and the four-toed salamander (*Hemidactylium scutatum*), the latter two of which are Species of Special Concern (www.state.ma.us/dfwele/dfw/nhesp/nhrare.htm).

12.4.1.3 Fish

Five fish species—largemouth bass (*Micropterus salmoides)*, yellow perch (*Perca flavescens)*, brown bullhead (*Ameiurus nebulosis)*, white sucker (*Catostomus commersoni)*, and pumpkinseed (*Lepomis gibbosus*)—were selected as the representative species for the ERA. The fish species were selected to include representatives of the principal trophic levels and exposure routes for fish in the PSA.

There is evidence in the literature that salmonid species may have a higher sensitivity to the effects of PCBs and other dioxin-like COPCs (Walker et al. 1994; Zabel et al. 1995). The use of rainbow trout in the site-specific toxicity testing program (Phase II), combined with effects data from the literature (Appendix F), provides a high degree of confidence that the ERA included an evaluation of fish species with equal or greater sensitivities than the representative species listed above. However, the procedure used to establish MATCs for fish in the PSA placed a low weight on studies conducted with fish species known to be highly sensitive (e.g., lake trout) to avoid an overly conservative assessment. Risks to coldwater fisheries (e.g., trout) downstream of the PSA were explicitly evaluated using benchmarks developed for salmonids; the uncertainty in these downstream risk estimates is high due to the number of extrapolations required. The risk of COCs to the occasional salmonid occurring within the PSA is considered to be intermediate. The PSA, however, is considered to be a warmwater fishery, and thus salmonid abundance is expected to be low in this portion of the river, even in the absence of chemical stressors.

12.4.1.4 Insectivorous Birds

The weight-of-evidence assessment indicated that exposure of insectivorous birds, such as tree swallows and American robins, to tPCBs and TEQ is high but unlikely to lead to adverse reproductive effects. Confidence in this conclusion, however, is not high because the two available lines of evidence for both species did not produce concordant results. The weight-of-evidence assessment for wood ducks indicated that they are at high risk from exposure to tPCBs and TEQ in the PSA. This conclusion, however, is based on only one line of evidence – modeling of exposure and effects. Uncertainty for this line of evidence, however, is relatively low for wood ducks exposed to TEQ because: (1) measured concentrations of TEQ in eggs corresponded well to model predictions, and (2) the effects metric for TEQ in eggs was based on a field study conducted using wood ducks.

The tree swallow was chosen as the one of the representative species for insectivorous birds. This species is common in the PSA and other areas in the watershed of the Housatonic River (Appendix A). Other insectivorous bird species that are comparable to tree swallows and are common to the PSA include the bank swallow (*Riparia riparia*), northern rough-winged swallow (*Stelgidopteryx serripennis*), barn swallow (*Hirundo rustica*), cliff swallow (*Hirundo pyrrhonota*), chimney swift (*Chaetura pelagica*), common nighthawk (*Chordeiles minor*), eastern kingbird (*Tyrannus tyrannus*), and eastern phoebe (*Sayornis phoebe*).

Compared to the tree swallow, the cliff swallow is expected to have a similar to lower level of risk from exposure to tPCBs and TEQ. The cliff swallow has a similar diet and is of similar size, but has a much larger foraging range that may dilute exposure compared to tree swallows. The eastern kingbird is also expected to have a similar to lower level of risk because it is twice the size of the tree swallow and, therefore, has a lower metabolic and food intake rate. The lower food intake rate will likely lead to reduced exposure. Eastern kingbirds and tree swallows have a similar diet and foraging range. The barn swallow and eastern phoebe have a similar level of risk compared to tree swallows because they have similar body sizes and diet. The bank swallow, chimney swift, and northern rough-winged swallow have a similar to higher level of risk from exposure to tPCBs and TEQ because they have a higher proportion of insects in the diet and/or are smaller than tree swallows.

Insectivorous birds that are more comparable to American robins and are common to the area include the eastern bluebird (*Sialia sialis*), eastern towhee (*Pipilo erythrophthalmus*), gray catbird (*Dumetella carolinensis*), hermit thrush (*Catharus guttatus*), northern mockingbird (*Mimus polyglottos*), veery (*Catharus fuscescens*), and wood thrush (*Hylocichla mustelina*) (see Appendix A).

Compared to American robins, eastern bluebirds and eastern towhees are expected to experience lower to similar levels of risk from exposure to tPCBs and TEQ. Eastern bluebirds consume similar prey compared to American robins, but have a larger foraging range that would dilute exposure to tPCBs and TEQ in the PSA. Eastern towhees consume less animal matter than American robins. Because animal matter contains higher concentrations of COCs than vegetation, eastern towhee exposure to tPCBs and TEQ will likely be lower.

The level of risk for the hermit thrush, northern mockingbird, veery, and wood thrush is expected to be similar to American robins. With the exception of earthworms in the robin diet, the dietary preferences of these birds are similar to the American robin. The absence of earthworms, a major dietary source of contaminants, will decrease their exposure to tPCBs and TEQ. However, their smaller body sizes result in higher food intake rates and thus greater exposure to tPCBs and TEQ through diet compared to American robins.

Gray catbirds are expected to experience similar to higher levels of risk compared to American robins. With the exception of earthworms in the robin diet, gray catbirds consume similar prey and have a foraging range comparable to American robins. Their smaller body size results in a higher food intake rate and greater exposure to tPCBs and TEQ through diet compared to American robins.

There are several waterfowl species comparable to the wood duck that have been observed in the PSA. These include American black duck (*Anas rubripes*), mallard (*Anas platyrhynchos*), common merganser (*Mergus merganser*), Canada goose (*Branta canadensis*), hooded merganser (*Lophodytes cucullatus*), green-winged teal (*Anas crecca*), and common goldeneye (*Bucephala clangula*) (see Appendix A).

Compared to wood duck, hooded mergansers are expected to have a higher risk from exposure to tPCBs and TEQ due to their similar body size and consumption of a greater proportion of aquatic associated prey (i.e., aquatic invertebrates and fish). American black ducks are expected to have a similar level of risk due to the inclusion of small quantities of fish in the black duck diet, and lower metabolism due to higher body weight. Mallard, common merganser, green-winged teal, and common goldeneye are expected to have lower risk to tPCB and TEQ exposure.

12.4.1.5 Piscivorous Birds

The weight-of-evidence assessment indicates that risks of tPCBs and TEQ to belted kingfisher are low; however, risks of these COCs to osprey are high and could lead to adverse reproductive effects.

The belted kingfisher and osprey were chosen to represent piscivorous birds inhabiting the Housatonic River area. Belted kingfisher and osprey are common piscivorous birds in the PSA. Great blue herons are also found in the PSA, and are discussed in Appendix K with other piscivorous birds (e.g., American bittern).

12.4.1.6 Piscivorous Mammals

Mink and river otter, the representative species for piscivorous mammals, are the only piscivorous mammals commonly found in the watershed of the Housatonic River (Woodlot, 2003b) (Appendix A).

12.4.1.7 Omnivorous and Carnivorous Mammals

The weight-of-evidence assessment indicates that omnivorous and carnivorous mammals, such as red fox and short-tailed shrew, are at risk in the PSA as a result of exposure to tPCBs and, to a lesser extent, TEQ. Risks to short-tailed shrews exposed to tPCBs at Locations 13 and 14 are high.

The northern short-tailed shrew and red fox were chosen to represent omnivorous and carnivorous mammals inhabiting the Housatonic River area. Other omnivorous and carnivorous species common to the area include the smoky shrew (*Sorex fumeus*), masked shrew (*Sorex cinereus*), coyote (*Canis latrans*), gray fox (*Urocyon cinereoargenteus*), fisher (*Martes pennanti*), long-tailed weasel (*Mustela frenata*), and short-tailed weasel (*Mustela erminea*) (see Appendix A).

Masked shrews are expected to experience a level of risk similar to northern short-tailed shrews. Both animals have similar foraging behaviors and ranges. Masked shrews are smaller than northern short-tailed shrews; therefore, they have a higher metabolism that increases exposure to contaminants. Masked shrews prefer to inhabit dry upland areas that are less contaminated than the damp woodlands and fields of the PSA where northern short-tailed shrews are found.

Compared to northern short-tailed shrews, smoky shrews are expected to experience higher levels of risk from exposure to tPCBs and TEQ. These shrews have similar foraging preferences and life history characteristics. Smoky shrews are much smaller than northern short-tailed shrews and thus have a higher metabolic rate that increases exposure to tPCBs and TEQ.

Coyotes have a larger body size and foraging range that decreases their exposure to tPCBs and TEQ. Considering these characteristics, coyotes are expected to experience lower risks from exposure to tPCBs and TEQ.

Gray and red foxes are expected to experience similar risks from exposure to tPCBs and TEQ. Gray fox have a larger foraging range than red fox and that may decrease their exposure to tPCBs and TEQ. Gray fox, however, have a greater reliance on animal matter and thus greater exposure to tPCBs and TEQ.

Fisher, long-tailed weasels, and short-tailed weasels are expected to experience similar to higher levels of risk from exposure to tPCBs and TEQ compared to the red fox. Fisher and red fox have similar body weights. Animal matter constitutes nearly 100% of the fisher diet compared to an average of 76% of the red fox diet. This greater consumption of animal matter increases the fisher's exposure to tPCBs and TEQ. The diets of long- and short-tailed weasels are similar to red fox, but weasels have smaller foraging ranges, which increases their exposure to tPCBs and TEQ in the PSA. Their smaller body weight results in a higher metabolism that further increases exposure to tPCBs and TEQ.

12.4.1.8 Threatened and Endangered Species

The weight-of-evidence assessment indicates that threatened and endangered (T&E) species such as bald eagles, American bitterns, and small-footed myotis are at risk in the PSA as a result of exposure to tPCBs and TEQ. The risks for bald eagles and American bitterns exposed to tPCBs and TEQ are high. The risks for small-footed myotis exposed to tPCBs and TEQ are intermediate.

The bald eagle, American bittern, and small-footed myotis were chosen to represent T&E species that are likely to be highly exposed to COCs in the Housatonic River PSA. Other T&E species that occur in the area include one mussel (triangle floater); three dragonflies (riffle snaketail, zebra clubtail, and arrow clubtail); a turtle (wood turtle); three salamanders (Jefferson salamander, four-toed salamander, and northern spring salamander); three hawks (northern harrier, sharp-shinned hawk, and Cooper's hawk); two warblers (northern parula and blackpoll warbler); a wading bird (common moorhen); and a shrew (northern water shrew). Some of these species were qualitatively assessed in other appendices and compared to other, more appropriate, assessment endpoints (e.g., amphibians for salamanders).

The level of risk for soras[1] is expected to be lower than for American bitterns. The sora and American bittern have similar life history characteristics and habitat preferences. However, the sora consumes more vegetable matter and less animal matter than the American bittern. This decreases the sora’s exposure to tPCBs and TEQ.

Great blue herons and king rails are expected to experience a similar level of risk as American bitterns. Great blue herons consume more fish than American bitterns. Fish contain higher concentrations of tPCBs than other prey. However, great blue herons have a larger body size than American bitterns resulting in a slower metabolism and lower accumulation of contaminants. In addition, the reproductive strategy for great blue herons suggests that few individuals from an entire rookery would be exposed in the PSA, lessening the risk to the local population. King rails consume prey that would have lower concentrations of COCs than American bitterns. Therefore, although they have a smaller body size and higher metabolism, their exposure to tPCBs and TEQ is expected to be lower.

The least bittern, green heron, Virginia rail, and pied-billed grebe are expected to experience higher levels of risk compared to the American bittern. The foraging and life history characteristics of these birds are similar to the American bittern. However, these birds are much

---

[1] Several of the species included in this section (i.e., sora, great blue heron, green heron, Virginia rail, northern myotis, little brown bat) are not threatened and endangered species either federally or in Massachusetts and Connecticut (Appendix A). They are included in the discussion of T&E species because they are taxonomically and ecologically similar to either American bittern or to small-footed myotis.

smaller than the American bittern. Their smaller body sizes result in a higher metabolism and greater exposure to tPCBs and TEQ.

The Indiana bat, northern myotis, and little brown bat are expected to have similar levels of risk as the small-footed myotis. These species belong to the same genus (*Myotis*) and have similar foraging behaviors and life histories.

All of the above information is summarized in greater detail in Table 12.4-1.

12.4.2 Ecological Implications and Other Concerns

As with most ecological risk assessments of contaminated sites, the ecological risk assessment for the Housatonic River is an assessment of the direct effects of COCs on a species-by-species basis. The following discussion places the ecological risk assessment in a broader ecological context by examining populations, ecological interactions and functions, and other issues of concern to decision makers and the public. The section begins with a brief discussion of the regulatory objectives of EPA and other agencies as they pertain to ecological protection goals for contaminated sites.

12.4.2.1 Protection Goals

Recently, there has been considerable interest in regulatory agencies and elsewhere for assessing risks at higher levels of organization, such as the population, community, or ecosystem (e.g., Environment Canada 1997; EPA 1997; Landis et al. 1998; EPA 1999b). Assessment of risks at higher levels of organization is useful because it furthers the understanding of the seriousness of risks posed by COCs, an important consideration in developing appropriate risk management responses. Assessment of risk at higher levels of organization, however, is not an easy task (Moore and Bartell 2000). The desire to understand risk at higher levels of organization should not be misinterpreted to mean that risks must be demonstrated at higher levels of organization (e.g., population or higher) to be of concern to EPA and other agencies at the Housatonic River, or other assessments of contaminated sites. As stated in EPA 1999b:

> 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 line-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. Data from discrete field and laboratory studies, if properly planned and appropriately interpreted, can be used to estimate local population or community level effects.

In addition, the Massachusetts Contingency Plan at 310 CMR 40.0995 (4) states:

> (b) The Stage II Environmental Risk Characterization shall identify environmental resources associated with the disposal site, such as wetlands, aquatic and terrestrial habitat, fisheries, or rare and endangered species, and shall evaluate whether the release of oil and/or hazardous material has adversely impacted, or may adversely impact the ecological functions which support those resources.
>
> 1. The evaluation shall focus on ecological functions at the spatial scale of the disposal site.
>
> 2. The relevance of potential impacts shall be judged at the spatial scale of the disposal site (e.g., effects on subpopulations that use the site as habitat) rather than the proportional significance of the site to regional environmental resources.

The concentrations of contaminants at this site, compared to most sites assessed under hazardous waste regulatory standards, are very high. By assessing aquatic life and wildlife that are exposed to COCs, the risk assessment evaluated whether the contaminated habitats (i.e., the Primary Study Area) are functioning as would normal habitats in the absence of contaminants. Vital functions include providing adequate food and shelter and sustaining normal reproductive success. The central question, for purposes of this assessment, is whether the exposed local populations are thriving in the contaminated habitat, not whether the larger regional population is surviving in spite of it.

12.4.2.2 Ecological Implications

Populations of organisms are controlled by the balance between positive processes (e.g., growth, reproduction, immigration) and negative processes (e.g., starvation, death from predation, toxicant effects, emigration) (Taub 1989). Growth and reproduction are often controlled by food supply and availability of adequate habitat. Mortality may be caused by predators and other stressors in the environment. The dynamics of populations exposed to COCs have important implications toward other interacting organisms and functions (Landis et al. 1998). Examples include:

- **Removal of Predators Compensates for Direct Effects of Contaminants—**A toxic chemical may increase food supply by reducing abundance of competitors or by eliminating predators. In the Housatonic River ERA, the modeling of exposure and effects line of evidence indicated that some species are experiencing risk from exposure to tPCBs and TEQ (e.g., short-tailed shrews, largemouth bass), yet are fairly abundant in the PSA. One possible explanation for this lack of concordance between measurement endpoints is that elimination of predators (e.g., mink, river otter) may be compensating for the direct effects due to tPCBs and TEQ.

- **Fishing Ban Artificially Enhances Recruitment—**For fish, the fishing ban imposed on the Housatonic River limits human predation on the fish stocks, therefore likely compensating for the effects of tPCBs on recruitment to older age classes. Elimination of predators, however, does not necessarily benefit each prey population. For example, some prey populations may decline as a result of increased competition from other prey populations (Bartell et al. 1992).

- **Immigration Compensates for Direct Effects of Contaminants—**The elimination of individuals from a habitat creates openings that may be exploited by individuals emigrating from surrounding habitats. Many species can migrate to the PSA or within the PSA from areas of lower contamination to areas of higher contamination and compensate for losses of organisms due to toxic effects; therefore, the presence of a normal number of animals in a contaminated area does not necessarily demonstrate that the population is unaffected or that the habitat is providing normal support functions. If contaminant concentrations are such that organisms cannot reproduce normally or thrive in the affected area without immigration from other areas, then those effects are viewed by EPA as unacceptable. Immigration is likely the process that explains the infrequent sightings of mink tracks in winter on the edges of the PSA. In winter, juvenile and young adult mink often emigrate to other habitats, particularly those that are not already occupied by mink, have an abundant food supply, and offer ideal habitat (i.e., the PSA). The risk assessment for the PSA indicates, however, that these mink are unlikely to survive and reproduce in the PSA.

- **Populations Exposed to COCs May Be More Vulnerable to Other Stressors in the Future—**The studies conducted to support the ERA were done during a period of regulatory restrictions, such as the fish, turtle, and frog advisories that have been in place in the Housatonic River in Massachusetts since 1982. Several authors (e.g., Evans et al. 1990; Edwards et al. 1990) have surmised that lake trout populations in the Great Lakes remained stable in the first half of the century despite fishing pressure and the influence of other anthropogenic stressors. Lake trout populations declined to very low levels by the 1970s, however, because of additional stressors such as contaminants and introduction of sea lampreys. Thus, a fish population can often tolerate some anthropogenic stresses, but if the combination of stresses becomes too great, the population crashes. Density-independent stressors, such as toxic chemicals that reduce

> fitness at all population densities, lower the capacity of populations to respond to otherwise favorable conditions or to tolerate other stressors (Evans et al. 1990). Control of sea lampreys, reductions in contaminant concentrations, and reduced fishing pressure have not restored lake trout populations to their historical levels (Edwards et al. 1990).

Many other indirect effects may occur as a result of the presence of tPCBs and TEQ in the PSA. For example, Wu et al. (1993) and Spromberg et al. (1998) have shown that subpopulations in patches removed from contamination may be affected by contaminated patches even if there is no transfer of contaminant (the so-called "action at a distance"). Alternatively, COCs transferred outside the PSA (e.g., by downstream transport, migration of birds) can augment exposures or cause effects to organisms outside the PSA (e.g., hawks preying on migrating birds).

12.4.2.2.1 Genetic Diversity

Chemical adaptation has been cited as a compensatory mechanism that can enable populations to survive at a site (Shugart 1996). This mechanism leads to selection of genotypes that are resistant to a COC as a result of the elimination of sensitive individuals. The result is a population with an altered genetic makeup, generally with a less diverse genetic pool. It is likely that such reductions in the genetic pool cause alterations that may reduce a population's resilience, making it more susceptible to other stresses in the future.

12.4.2.2.2 Immune System Effects

Non-coplanar PCBs can influence the activity of neutrophils (a type of white blood cell) through mechanisms unrelated to the Ah receptor. In studies with rat- and human-derived neutrophils, researchers have found that non-coplanar PCBs can activate biochemical pathways that lead to the production of reactive oxygen species (ROS) (Fischer et al. 1998). Although the production of ROS is a normal function of neutrophils (it is designed to destroy bacteria and viruses and to break down tissue damaged by burns, chemicals, and physical injuries), when inappropriately activated by PCBs, this neutrophil function can initiate harmful effects on healthy tissues because of the destructive nature of ROS. Because neutrophils are among the first white blood cells sent to sites of infection or inflammation, exposure to PCBs may weaken an animal's immune and inflammatory responses.

12.4.2.2.3 Deformities and Disease

Organism abundance and the self-sustaining nature of local populations are often used as assessment endpoints in ERAs. However, the presence of a self-sustaining population does not provide conclusive evidence that no harm has occurred. Individual-level responses, such as deformities and cancerous growths, may impair the overall population "health" without necessarily resulting in responses to reproduction or local population abundance. Individual-level responses are of potential concern from both a human perspective (i.e., deformed fish are considered a less valuable resource) and from an ecological perspective (i.e., tumors and lesions often serve as biomarkers indicating that reproductive thresholds are being approached). The endocrine responses observed in adult PSA fish species, neoplastic lesions in Woods Pond goldfish, and other deformities in multiple PSA fish species, all provide indications of potential harm attributable to COCs that may not be resulting in obvious impacts at the local population level.

12.5 SOURCES OF UNCERTAINTY

The assessment of risks of COCs to aquatic and wildlife species in the Housatonic River contains uncertainties. Each source of uncertainty can influence the estimates of risk; therefore, it is important to describe and, when possible, specify the magnitude and direction of such uncertainties. The sources of uncertainty associated with the assessment of risks of tPCBs and TEQ to each assessment endpoint were summarized in Sections 3 through 11 and Appendices D through K. This material is not repeated here. In this section, the most significant sources of uncertainty commonly encountered throughout the ERA are described. The sources of uncertainty are grouped by phase of the ERA (i.e., problem formulation, exposure assessment, effects assessment, risk assessment).

12.5.1 Problem Formulation

The problem formulation is intended to define the linkages between stressors, potential exposure, and predicted effects on ecological receptors. As such, the conceptual model provides the scientific basis for selecting assessment and measurement endpoints to support the risk assessment process. Potential uncertainties arise from lack of knowledge regarding ecosystem functions, failure to adequately address spatial and temporal variability in the evaluations of sources, fate and effects, omission of stressors, and overlooking secondary effects (EPA 1998). The types of uncertainties associated with the conceptual model that links contaminant sources to effects include those associated with the identification of COCs, environmental fate and transport of COCs, exposure pathways, receptors at risk, and ecological effects. Of these, the identification of exposure pathways probably represents the primary source of uncertainty in the conceptual model. The detailed ecological characterization performed at this site has greatly reduced the uncertainties associated with problem formulation, yet some uncertainties remain and are described below:

- The Housatonic River and surrounding floodplain have received chemical inputs since the industrialization of the area. In addition to tPCBs and TEQ, other contaminants identified in water, soil, and sediment samples include metals (e.g., mercury, lead, chromium); pesticides (e.g., aldrin, DDT, toxaphene, parathion, 2,4-D); and semivolatile organic compounds (e.g., PAHs, chlorinated benzenes, anilines, phenols). The conservative screening level assessment indicated that only tPCBs and TEQ present a potential risk to wildlife species; therefore, only these COCs were included in the probabilistic risk assessments for wildlife. Several other COCs were screened through for aquatic life, although none were as influential as tPCBs. Additive, synergistic, and antagonistic effects due to exposure to multiple contaminants were not considered.

- In this assessment, it was assumed that dietary exposure represented the most important pathway for the exposure of wildlife to COCs. Although unlikely to provide a major contribution to the risk, other pathways could increase exposure and perhaps increase risk slightly (Moore et al. 1999). Deterministic calculations were conducted in which estimates of exposure to COCs via drinking water and inhalation were included in the exposure model, but were not included in the assessment because inclusion of these routes did not substantially increase overall exposure of wildlife to the COCs. This issue was less important for aquatic life because these assessments were conducted for multiple media exposures. The aquatic life endpoints considered tissue burdens that integrated exposures from all sources and/or evaluated exposures from the abiotic media (sediment, overlying water, porewater) deemed most relevant to exposure.

12.5.2 Exposure Assessment

The exposure assessment is intended to describe the actual or potential co-occurrence of stressors with receptors. As such, the exposure assessment identifies the exposure pathways and the intensity and extent of contact with stressors for each receptor or group of receptors at risk. The exposure models for wildlife were energetics-based models requiring information on body weight, free living metabolic rate, proportions of food items in the diet, and the concentrations of COCs in these food items. Each of these variables has associated uncertainties, most of which were propagated through the exposure models. Exposure of fish species to COCs was assessed using measured wet weight whole body tissue concentrations. Exposure of benthos to COCs was assessed as either the COC concentrations in abiotic site media (i.e., sediment, water) or as the tissue body burdens that represent integrated exposure from all sources.

- The greatest uncertainty in the benthic invertebrate and amphibian exposure assessments was the potential for small-scale variability in exposure concentrations to complicate the development of dose-response relationships, additionally confounded by analytical variability (Appendix C.11). For studies that had replicate measurements of tPCBs at a given station over a short period (e.g., benthic macroinvertebrate sampling), the spatial variability was quantified and considered explicitly in the derivation of dose-response relationships. Where spatial replication was not available, characterization of variability required the incorporation of additional data sets.

- Tissue chemistry data (tPCBs, TEQ) were relied upon in the characterization of exposures to fish species and piscivorous wildlife. Total PCB concentrations in fish exhibit seasonal fluctuations; these are sometimes related to lipid content changes that occur during reproductive life history stages. To minimize additional variability in PCB concentrations due to spawning events and other seasonal effects, the vast majority of the PCB data for this project were collected in the late summer and early fall. Potential uncertainties (over time and space) in fish tissue chemistry were ameliorated through the collection of a large number of samples (multiple species, ages, and spanning multiple years of collections). Therefore, the ERA was conducted with a robust data set (including a confirmatory analysis with non-EPA data sets) that limited the probability of spurious outcomes in the exposure assessments.

- The Monte Carlo sensitivity analyses suggested that the parameters of the free metabolic rate (*FMR*) allometric equation were generally the most influential variables on predicted total daily intakes of COCs. However, no direct measurements of free metabolic rate or food intake rate (other than for captive animals) were available for most of the representative wildlife species. Therefore, free metabolic rates were estimated using allometric equations. The use of allometric equations introduces some degree of uncertainty into the exposure estimates because they are subject to model-fitting error and are based on species different from the representative species used in this assessment. Given the lack of data on species specific to this assessment, it is difficult to judge the magnitude of the uncertainty introduced by the use of the allometric models. The uncertainty due to model-fitting error was propagated in the uncertainty analyses by using distributions as inputs for the allometric slope and power terms.

- Sample sizes, while composites, were limited for the analyses of COC concentrations in some prey items, including earthworms, litter invertebrates, and benthic invertebrates. To address this uncertainty in the Monte Carlo analyses for wildlife, the upper confidence limit (UCL) or data set maximum (see Section 6.4 and Appendix C.5) was used as an estimate of COC concentrations in prey items. The potential magnitude of the uncertainty associated with small sample sizes for COC concentrations is unknown, but this approach likely overestimated exposure. The probability bounds analysis used an unbiased approach (e.g., distribution free range from LCL to UCL) to deal with sample size uncertainty.

12.5.3 Effects Assessment

The effects assessment is intended to describe the effects caused by stressors, link them to the assessment endpoints, and evaluate how effects change with fluctuations in the levels (i.e., concentrations or doses) of the various stressors. In this assessment, the effects of tPCBs and other COCs to representative species were assessed. There are several sources of uncertainty in the assessment of effects, including measurement errors, extrapolation errors, and data gaps.

- For fish, benthos, and amphibians, the effects benchmarks derived from the literature had a high degree of uncertainty due to the need to extrapolate across sites and species. This uncertainty was explicitly addressed in the weight-of-evidence evaluation.

- The site-specific fish toxicity studies indicated variations in the dose-response relationships observed across species, reaches, and treatments, and introduced uncertainty into the development of effects thresholds.

- The methodology used to mimic maternal transfer used in the fish Phase II studies has been recently developed and has not been widely applied as an environmental monitoring technique; therefore, there are potential uncertainties inherent to extrapolating these laboratory-based results to Housatonic River fish. Similarly, the extrapolation of concentrations of tPCBs in egg to whole body concentrations has a degree of uncertainty associated with it. The magnitude and direction of the uncertainty is unknown.

- The greatest potential source of uncertainty for the fish and wildlife effects assessments was associated with the lack of toxicity studies involving the representative species. The direct assessment of effects to benthos, amphibians, largemouth bass, and mink included studies on the effects of tPCBs to reproduction and/or survival for these representative species. There were, however, no toxicity studies available for many other representative species exposed to tPCBs or TEQ. As a result, laboratory studies involving other species were often used to estimate effects to representative species. To address uncertainty in the effects assessments, threshold ranges were used in which effects to tolerant and sensitive species were considered. It was assumed that the toxicity thresholds for the representative species lie within these ranges.

- The effects metrics used to estimate risk from the literature were derived for Aroclor 1254 and Aroclor 1260 mixtures, and more information was available for Aroclor 1254 than for Aroclor 1260. Some uncertainty is inherent in extrapolating from studies using commercial Aroclor mixtures to the specific congener patterns observed in weathered mixtures in the PSA of the Housatonic River. The potential magnitude and direction of the uncertainty associated with this extrapolation are unknown.

- TEQ is an expression of the planar chlorinated hydrocarbons (PCHs). TEQ are derived from an equation that combines the relative potency of each congener into a single concentration. The potencies of individual congeners are not known precisely and were estimated based on a combination of data and professional judgment (Van den Berg et al. 1998). Although there is uncertainty in these calculations, this approach has been accepted and applied in numerous jurisdictions worldwide. The potential magnitude and direction of the uncertainty associated with this approach are unknown. The potential toxic effects of congeners not evaluated by this method are poorly understood and can not be quantified.

12.5.4 Risk Characterization

A weight-of-evidence procedure was used to assess risks of tPCBs and TEQ to the assessment endpoints in the Housatonic River PSA. The analysis follows the methodology proposed by the Massachusetts Weight-of-Evidence Workgroup (Menzie et al. 1996; see Section 2.9 for details).

In general, the weight-of-evidence approach is an inclusive process whereby multiple lines of evidence are considered prior to determining risk. For the wildlife risk assessments, these lines of evidence included the exposure and effects modeling results and, in some cases, field survey results, and/or in situ or whole media toxicity test results. For the fish and benthic invertebrate risk assessments, available lines of evidence included field survey results (e.g., community evaluation for benthos), site-specific toxicity tests, and comparison of tissue and sediment concentrations to benchmarks (both from the literature and site-specific benchmarks).

- Uncertainty in the risk characterization arises from the absence of one or more of the available lines of evidence. In the case of belted kingfishers, data on two of the three major lines of evidence were available, i.e., comparison of modeled exposure and effects and the belted kingfisher field study. Threatened and endangered species had only the modeled exposure and effects line of evidence to support the risk assessment. The consequence of the lack of multiple concurring lines of evidence is less confidence in the conclusion regarding risk. For example, the risk characterization of mink had three major lines of evidence available, thus providing high confidence in the risk conclusions. The risk characterization for fish and benthic invertebrates also had three major lines of evidence available.

- Uncertainty for individual lines of evidence was sometimes sufficiently large to render the line of evidence of limited use in the ERA. For example, the community evaluation for fish was confounded by large habitat variations combined with small overall gradients and large small-scale variation in PCB concentrations. These factors made derivations of dose-response relationships unfeasible (i.e., due to very low statistical power for determining contaminant-induced effects) and limited the studies to qualitative assessments.

12.6 ERA CONCLUSIONS

- Weight-of-evidence assessments indicated that many aquatic and wildlife species in the PSA of the Housatonic River are experiencing intermediate to high risk as a result of exposure to tPCBs and other COCs. Confidence in this conclusion is high for benthic invertebrates, amphibians, and piscivorous mammals because multiple lines of evidence gave concordant results.

- The risks of tPCBs and other COCs likely extend beyond the representative species considered in the quantitative risk assessments described herein. Qualitative risk assessments indicated that many other species in the PSA are potentially at risk. Further, there are likely indirect effects (e.g., changes in predator-prey relationships, changes in metapopulation dynamics) occurring inside and outside the PSA as a result of the direct impacts caused by tPCBs and other COCs.

- An assessment of risk downstream of the PSA indicated that tPCBs could potentially be causing adverse effects to benthic organisms in depositional areas as far as Reach 8, amphibians in floodplain areas as far as Reach 9, trout in Reach 7, mink as far as Reach 15, otter as far as Reach 15, and bald eagle in Reach 8. However, the magnitude of risks in these areas is lower than in the PSA.

12.7 REFERENCES

Amrhein, J.F., C.A. Stow and C. Wible. 1999. Whole-fish versus filet polychlorinated-biphenyl concentrations: An analysis using classification and regression tree models. *Environmental Toxicology and Chemistry* 18:1817-1823.

Anderson, M.G. and R.D. Titman. 1992. Spacing in breeding waterfowl: ecology and behavior. Pages 251-289 in Batt, B.D.J., AD Afton, M.G. Anderson, C.D. Ankney, D.H. Johnson, K.A. Kadlec, and G.L. Krapu, editors. *The Ecology and Management of Breeding Waterfowl*. University of Minnesota Press, Minneapolis.

Angermann, J.E., and F. Matsumura. 1999. Effects of PCBs and Cold Exposure on Immune Function and Hepatic Glycogenolysis in *Rana pipiens*. Society of Environmental Toxicology and Chemistry (SETAC). Presentation, SETAC 20th Annual Meeting, Nashville, Tennessee.

Anthony, E.P.L. and T.H. Kunz. 1977. Feeding strategies of the little brown bat in southern New Hampshire. *Ecology* 58:775-786.

Aulerich, R.J., S.J. Bursian, W.J. Breslin, B.A. Olson, and R.K. Ringer. 1985. Toxicological manifestations of 2,4,5-, 2',4',5'-, 2,3,6,2',3',6'-, and 3,4,5,3',4',5'-Hexachlorobiphenyl and Aroclor 1254 in mink. *Journal of Toxicology and Environmental Health* 15:63-79.

Azure, D.A. 1998. Aspects of American Bittern ecology in northwest Minnesota. M.S. Thesis. University of North Dakota, Grand Forks, North Dakota. 139 pp. Cited in: J.A. Deschant, M.L. Sondreal, D.H. Johnson, L.Dl, Igl, C.M. Goldade, A.L. Zimmerman, and B. R. Euliss. 2001. *Effects of Management Practices on Grassland Birds: American Bittern.* Northern Prairie Wildlife Research Center, Jamestown, ND, USA. Available at: http://www.npwrc.usgs.gov/resource/literatr/grasbird/ambi/ambi.htm (Version 17FEB2000).

Barbour, R.W. and W.H. Davis. 1969. *Bats of America.* University of Kentucky Press. Lexington, Kentucky. 286 pp. Cited in: R.M. DeGraaf, and M. Yamasaki. 2001. *New England Wildlife: Habitat, Natural History, and Distribution.* University Press of New England, Hanover, New Hampshire.

Barron, M.G., H. Galbraith, and D. Beltman. 1995. Comparative reproductive and developmental toxicology of PCBs in birds. *Comparative Biochemistry and Physiology* 112C(1)1-14.

Bartell, S.M., R.H. Gardner and R.V. O'Neill. 1992. *Ecological Risk Estimation.* Lewis Publishers, Boca Raton, FL.

Bellis, E.D. 1968. Summer movement of red-spotted newts in a small pond. *Journal Herpetology* 1:86-91.

Bellrose, F.C. 1981. Ducks, geese and swans of North America. Stackpole Books, Harrisburg, PA.

Belwood, J.J. and M.B. Fenton. 1976. Variation in the diet of *Myotis lucifugus. Canadian Journal of Zoology* 54:1674-1678.

Berger, A.J. 1951. Nesting density of Virginia and sora rails in Michigan. *Condor* 53:202. Cited in: R.M. DeGraaf, and M. Yamasaki. 2001. *New England Wildlife: Habitat, Natural History, and Distribution.* University Press of New England, Hanover, NH.

Bevelhimer, M.S., J.J. Bauchamp, B.E. Sample and G.R. Southworth. 1997. *Estimation of Whole-Fish Contaminant Concentrations From Fish Fillet Data.* ES/ER/TM-202. Oak Ridge National Laboratory, Oak Ridge, TN.

Birnbaum, L.S., and M.J. DeVito. 1995. Use of toxic equivalency factors for risk assessment for dioxins and related compounds. *Toxicology* 105:391-401.

Bishop, C.A., M.D. Koster, A.A. Chek, D.J.T. Hussell, and K. Jock. 1995. Chlorinated hydrocarbons and mercury in sediments, red-winged blackbirds (*Agelaius phoeniceus*) and tree swallows (*Tachycineta bicolor*) from wetlands in the Great Lakes-St. Lawrence River basin. *Environmental Toxicology and Chemistry* 14:491-501.

Bishop, C.A., H.A. Mahony, S. Trudeau, and K.E. Pettit. 1999. Reproductive success and biochemical effects in tree swallows (*Tachycineta bicolour*) exposed to chlorinate hydrocarbon contaminants in wetlands of the Great Lakes and St. Lawrence River basin, USA and Canada. *Environmental Toxicology and Chemistry* 18:263-271.

Bleavins, M.R., R.J. Aulerich, and R.K. Ringer. 1980. Polychlorinated biphenyls (Aroclor 1016 and 1242): effects on survival and reproduction in mink and ferret. *Archives of Environmental Contamination and Toxicology* 9:627-635.

Bosveld, A.T.C, and M. Van den Berg. 1994. Effects of polychlorinated biphenyls: dibenzo-p-dioxins and dibenzofurans on fish-eating birds. *Environmental Reviews* 2:147-166.

Brown, M. and J.J. Dinsmore. 1986. Implications of marsh size and isolation for marsh bird management. *Journal of Wildlife Management* 50:392-397.

Brunström, B. 1988. Sensitivity of embryos from duck, goose, herring gull, and various chicken breeds to 3,3’, 4,4’-tetrachlorobiphenyl. *Poultry Science* 67:52-57.

Brunström, B. 1989. Toxicity of coplanar polychlorinated biphenyls in avian embryos. *Chemosphere* 19(1-6):765-768.

Bursian, S.J., R.J. Aulerich, B. Yamini, and D.E. Tillitt. 2003. Dietary Exposure of Mink to Fish from the Housatonic River: Effects on Reproduction and Survival. Michigan State University, Department of Animal Science. Final Report. June 10, 2003.

Burton, T.M. 1977. Population estimates, feeding habits, and nutrient and energy relationships of *Notophthalmus v. viridescens,* in Mirror Lake, New Hampshire. *Copeia*. 1977:139-143.

CCME (Canadian Council of Ministers of the Environment). 1999. Canadian Tissue Residue Guidelines for the Protection of Wildlife Consumers of Aquatic Biota: Polychlorinated Biphenyls (PCBs). In: *Canadian Environmental Quality Guidelines 1999.* Canadian Council of Ministers of the Environment, Winnipeg.

Cogliano, V.J. 1998. Assessing the cancer risk from environmental PCBs. *Environmental Health Perspectives* 106(6):317-323

Custer, C.M., T.W. Custer, P.D. Allen, K.L. Stromborg, and M. Melancon. 1998. Organochlorine contaminants and tree swallows nesting along the Fox River and in Green Bay, Wisconsin. *Environmental Toxicology and Chemistry* 17:1786-1798.

Custer, C.M. 2002. Exposure and Effects of Chemical Contaminants on Tree Swallows Nesting Along the Housatonic River, Berkshire Co., Massachusetts, 1998-2000. Final Report to U.S. Environmental Protection Agency. USGS, Upper Midwest Environmental Sciences Center, La Crosse, WI.DeGraaf, R.M. and M. Yamasaki. 2001. *New England Wildlife: Habitat, Natural History, and Distribution.* University Press of New England, Hanover, NH.

Deschant, J.A., M.L. Sondreal, D.H. Johnson, L.Dl, Igl, C.M. Goldade, A.L. Zimmerman, and B. R. Euliss. 2001. *Effects of Management Practices on Grassland Birds: American Bittern.* Northern Prairie Wildlife Research Center, Jamestown, North Dakota. Available at: http://www.npwrc.usgs.gov/resource/literatr/grasbird/ambi/ambi.htm (Version 17FEB2000).

DiBello, F., S. Arthur, and W. Krohn. 1990. Food habits of sympatric coyotes, red foxes, and bobcats in Maine. *Canadian Field Naturalist* 104:403-408.

Drilling, N., R. Titman, and F. McKinney. 2002. Mallard (*Anas platyrhynchos*). *In* The Birds of North America, No. 658 (A. Poole and F. Gill, eds.). The Birds of North America, Inc., Philadelphia, PA.

Drobney, R.D. and Fredrickson. L.H. 1979. Food selection by Wood ducks in relation to breeding status. *J. Wildlife Manage*. 43: 109-120.

Eadie, J.M., M.L. Mallory, and H.G. Lumsden. 1995. Common Goldeneye (*Bucephala clangula*). *In* The Birds of North America, No. 170 (A. Poole and F. Gill, eds.). The Academy of Natural Sciences, Philadelphia, PA, and The American Ornithologists' Union, Washington, D.C.

Eadie, W.R. 1944. The short-tailed shrew and field mouse predation. *Journal of Mammalogy* 25:359-364.

Eadie, W.R. 1948. Shrew-mouse predation during low mouse abundance. *Journal of Mammalogy* 29:35-37.

Edwards, C.J., R.A. Ryder, and T.R. Marshall. 1990. Using lake trout as a surrogate of ecosystem health for oligotrophic waters of the Great Lakes. *Journal of Great Lakes Research* 16:591-608.

Ehrlich, P.R., D.S. Dobkin, and D. Wheye. 1988. *The Birder's Handbook – A Field Guide to the Natural History of North American Birds.* Fireside Press, New York.

Ehrlich, P.R., D.S. Dobkin, and D. Wheye. 1992. *The Birder's Handbook – A Field Guide to the Natural History of North American Birds.* Fireside Press. New York.

Eisler, R. 2000. *Handbook of Chemical Risk Assessment: Health Hazards to Humans, Plants, and Animals. Volume 2 – Organics.* Lewis Publishers, Boca Raton, FL. ISBN 1-56670-506-1.

Environment Canada. 1997. *Environmental Risk Assessments of Priority Substances und the Canadian Environmental Protection Act: Guidance Manual.* Chemicals Evaluation Division, Commercial Chemicals Evaluation Branch, Environment Canada, Ottawa, Canada.

EPA (U.S. Environmental Protection Agency). 1993. *Wildlife Exposure Factors Handbook. Volumes I and II.* Office of Research and Development, Washington, DC. EPA EPA/600/R-93/187a, EPA/600/R-93/187b.

EPA (U.S. Environmental Protection Agency). 1997. *Ecological Risk Assessment Guidance for Superfund: Process for Designing and Conducting Ecological Risk Assessments.* Interim Final Report. Solid Waste and Emergency Response. EPA 540-R-97-006. June 1997.

EPA (U.S. Environmental Protection Agency). 1998. *Guidelines for Ecological Risk Assessment.* Risk Assessment Forum, Washington, DC. EPA/630/R-95/002F.

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

EPA (U.S. Environmental Protection Agency). 1999b. Issuance of Final Guidance: Ecological Risk Assessment and Risk Management Principles for Superfund Sites; Signed 10/7/99. Memorandum from Stephen D. Luftig, Office of Emergency and Remedial Response, Washington, DC.

Evans, D.O., G.J. Warren, and V.W. Cairns. 1990. Assessment and management of fish community health in the Great Lakes: synthesis and recommendations. *Journal of Great Lakes Research* 16:639-669.

Fischer, L., R.F. Seegal, P. Ganey, I.N. Pessah, and P.R.S. Kodavanti. 1998. Toxicity of non-coplanar PCBs. *Toxicological Sciences* 41:49-61.

Georgii, S., G. Bachour, K. Failing, U. Eskins, I. Emadfa, and W.L. Budde. 1994. Polychlorinated biphenyl congeners in foxes in Germany from 1983 to 1991. *Archives of Environmental Contamination and Toxicology* 26:1-6.

Gibbs, J.P., S. Melvin, and F.A. Ried. 1992. American Bittern (*Botaurus lentiginosus*). In: *The Birds of North America*, No. 18. A. Poole, P. Stettenheim, and F. Gill, Editors. The Birds of North America, Inc. Philadelphia, PA. pp.1-11.

Giesy, J.P., and K. Kannan. 1998. Dioxin-like and non-dioxin-like toxic effects of polychlorinated biphenyls (PCBs): implications for risk assessment. *Critical Reviews in Toxicology* 28(6):511-569.

Gilmer, D.S., Ball, I.J., Cowardin, L.M. Mathisen, J.E. and Reichmann, J.H. 1978. Natural cavities used by wood ducks in north-central Minnesota. *J. Wildlife Management* 42:288-298.

Godfrey, W.E. 1986. *The Birds of Canada*. National Museum of Natural Sciences, Ottawa, Canada.

Grice, D. and Rogers, J.P. 1965. The Wood Duck in Massachusetts. Final Report. Massachusetts Division of Fisheries and Game. Final Report. Federal Aid in Wildlife Restoration Project No. W-19-R.

Griffith, L.A. and J.E. Gates. 1985. Food habits of cave-dwelling bats in the central Appalachians. *Journal of Mammalogy* 66:451-460.

Gunn, J.M., S.U. Qadri, and D.C. Mortimer. 1997. Filamentous algae as a food source for the brown bullhead. *Journal of the Fisheries Research Board of Canada* 34:396-401.

Hamilton, W.J., Jr. 1941. The foods of small forest mammals in eastern United States. *Journal of Mammalogy* 22:250-263.

Hartel, K.E., D.B. Halliwell, and A.E. Launer. 2002. *Inland Fishes of Massachusetts. Massachusetts Audubon Society.* Lincoln, Massachusetts.

Healy, W.R. 1975. Terrestrial activity and home range in efts of *Notophthalmus v. viridescens. American Midland Naturalist* 93:131-138.

Hockman, J.G., and J.A. Chapman. 1983. Comparative feeding habits of red foxes (*Vulpes vulpes*) and gray foxes (*Urocyon cinereoargentus*) in Maryland. *American Midland Naturalist* 110:267-285.

Huang, Y., D.J. Hoffman, and W.H. Karasov. 2001. Oxidative stress in PCB 126-exposed frogs, *Rana pipiens.* Presentation, SETAC 22nd Annual Meeting, Baltimore, MD.

Hull, R.N. and G.W. Suter II. 1994. *Using the Weight-of-Evidence Approach for Ecological Risk Assessment at a DOE Facility.* SETAC Denver, CO. November 1, 1994.

Hunter, M.L., A.J.K. Calhoun, and M. McCollough. 1999. *Maine Amphibians and Reptiles.* The University of Maine Press, Orono, Maine. 252 pp.

Jenssen, T.A., and W.D. Klimstra. 1966. Food habits of the green frog, *Rana clamitans*, in southern Illinois. *American Midland Naturalist* 76:169-182.

Johnson, K. 1995. Green-winged Teal (*Anas crecca*). *In* The Birds of North America, No. 193 (A. Poole and F. Gill, eds.). The Academy of Natural Sciences, Philadelphia, and The American Ornithologists' Union, Washington, D.C.

Jung, R.E., and M.K. Walker. 1997. Effects of 2,3,7,8-tetrachlorodibenzo-p-dioxin (TCDD) on development of anuran amphibian. *Environmental Toxicology and Chemistry* 16:230-240.

Knable, A.E. 1974. Seasonal trends in the utilization of major food groups by the red fox (*vulpes fulva*) in Union County, Illinois. *Trans. III. State Acad.* Sci. 66:113-115.

Korschgen, L.J. 1959. Food habits of the red fox in Missouri. *Journal of Wildlife Management* 23:168-176.

Kurta, A. 1995. *Mammals of the Great Lakes Region.* The University of Michigan Press. Ann Arbor, MI. 392 pp.

Kurta, A. and R.H. Baker. 1990. *Eptesicus fuscus.* Mammalian Species No. 356:1-10. American Society of Mammalogists. Shippensburg, Pennsylvania.

Landers, J.L., T.T. Fendley, and A.S. Johnson. 1977. Feeding ecology of wood ducks in South Carolina. *J. Wildlife Management* 41: 118-127.

Landis, W.G., D.R.J. Moore, and S.B. Norton. 1998. Ecological risk assessment: looking in, looking out. In: *Pollution Risk Assessment and Management.* P.E.T. Douben (Ed). John Wiley & Sons, Toronto.

Leonards, P.E.G., Y. Zierikzee, U.A. Brinkman, W.P. Cofino, N.M. van Straalen, and B. van Hattum. 1997. The selective dietary accumulation of planar polychlorinated biphenyls in the otter (*Lutra lutra*). *Environmental Toxicology and Chemistry* 16(9):1807-1815.

Linzey, D.W. and A.V. Linzey. 1973. Notes on food of small mammals from Great Smokey Mountains National Park, Tennessee-North Carolina. *J. Elisha Mitchell Science Society* 89:6-14.

Longcore, J. R., D. G. McAuley, G. R. Hepp, and J. M. Rhymer. 2000. American Black Duck (*Anas rubripes*). *In* The Birds of North America, No. 481 (A. Poole and F. Gill, eds.). The Birds of North America, Inc., Philadelphia, PA.

MacNamara, M.C. 1977. Food habits of terrestrial adult migrants and immature red efts of the red-spotted newt, *Notophthalmus viridescens. Herpetologica* 33:127-132.

Mallory, M. and K. Metz. 1999. Common Merganser (*Mergus merganser*). In The Birds of North America, No. 442 (A. Poole and F. Gill, eds.). The Birds of North America, Inc., Philadelphia, PA.

Martin, A.C., H.S. Zim, and A.L. Nelson. 1951. *American Wildlife and Plants - A Guide to Wildlife Food Habits.* Dover Publications, Inc., New York.

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

Menzie, C., M.H. Henning, J. Cura, K. Finkelstein, J. Gentile, J. Maughan, D. Mitchell, S. Petron, B. Potocki, S. Svirsky, and P. Tyler. 1996. Special report of the Massachusetts Weight-of-Evidence Workgroup: A weight-of-evidence approach for evaluating ecological risks. *Human and Ecological Risk Assessment* 2:277-304.

Mills, R.S., G.W. Barrett, and M.P. Farrell. 1975. Population dynamics of the big brown bat (*Eptesicus fuscus*) in southwestern Ohio. *Journal of Mammalogy* 56:591-604. Cited in: R.M.

DeGraaf, and M. Yamasaki. 2001. *New England Wildlife: Habitat, Natural History, and Distribution.* University Press of New England, Hanover, New Hampshire.

Mowbray, T. B., C. R. Ely, J. S. Sedinger, and R. E. Trost. 2002. Canada Goose (Branta canadensis). In The Birds of North America, No. 682 (A. Poole and F. Gill, eds.). The Birds of North America, Inc., Philadelphia, PA.

Moore, D.R.J., and S.M. Bartell. 2000. Estimating ecological risk of multiple stressors: advanced methods and difficult issues. In: *Multiple Stressors in Ecological Risk and Impact Assessment: Approaches to Risk Estimation.* S.A. Ferenc and J.A. Foran (Eds). Society of Environmental Toxicology and Chemistry (SETAC).

Moore, D.R.J., B.E. Sample, G.W. Suter, B.R. Parkhurst, and R.S. Teed. 1999. A probabilistic risk assessment of the effects of methylmercury and PCBs on mink and kingfishers along East Fork Poplar Creek, Oak Ridge, Tennessee. *Environmental Toxicology and Chemistry* 18:2941-2953.

Newsted, J.L., J.P. Giesy, G.T. Ankley, D.E. Tillitt, R.A. Crawford, J.W. Gooch, P.D. Jones, and M.S. Denison. 1995. Development of toxic equivalency factors for PCB congeners and the assessment of TCDD and PCB mixtures in rainbow trout. *Environmental Toxicology and Chemistry* 14(5):861-871

Odum, R.R. 1977. Sora (*Porzana carolina*). In: *Management of migratory shore and upland game birds in North America*, pp. 57-65. G.C. Sanderson, Editor. Washington, D.C.

Petranka, J.W. 1998. *Salamanders from the United States and Canada.* Smithsonian Institution Press. Washington, DC.

Powell, D.G. and R.M. Case. 1982. Food habits of the red fox in Nebraska. *Trans. Nebr. Acad. Sci. and Affil. Soc.* 10:13-16.

Robertson, R.J., B.J. Stutchbury, and R.R. Cohen. 1992. Tree Swallow (*Tachycineta bicolor*). In: *The Birds of North America, No. 11*, pp. 1-26. A. Poole, P. Stettenheim, and F. Gill, Editors. The Birds of North America, Inc., Philadelphia, Pennsylvania.

Safe, S.H. 1994. Polychlorinated Biphenyls (PCBs): Environmental impact, biochemical and toxic responses, and implications for risk assessment. *Critical Reviews in Toxicology* 24(2):87-149.

Sallabanks, R. and F.C. James. 1999. American Robin (*Turdus migratorius*) In: *The Birds of North America, No. 642*, pp. 1-27. A. Poole, P. Stettenheim, and F. Gill, Editors. The Birds of North America, Inc., Philadelphia, Pennsylvania.

Shugart, L.R. 1996. Molecular markers to toxic agents. In: *Ecotoxicology: A Hierarchical Treatment*, pp. 154-156. M.C. Newman and C.H. Jagoe, Editors. CRC Lewis Publishers, New York.

Sibley, D.A. *The Sibley Guide to Bird Life & Behavior.* 2001. Alfred A. Knopf, Inc., New York.

Spromberg, J.A., B.M. John, and W.G. Landis. 1998. Metapopulation dynamics: Indirect effects and multiple discrete outcomes in ecological risk assessment. *Environmental Toxicology and Chemistry* 17(8):1640-1649.

Stewart, M.M. and P. Sandison. 1973. Comparative food habits of sympatric mink frogs, bullfrogs, and green frogs. *Journal of Herpetology* 6:241-244.

Suter, G.W., II, B.E. Sample, D.S. Jones, T.L. Ashwood, and J.M. Loar. 1995. *Approach and Strategy for Performing Ecological Risk Assessments for the U.S. Department of Energy's Oak Ridge Reservation.* Oak Ridge National Laboratory, Oak Ridge, TN. ES/ER/TM-33/R2.

Tanner, W.D. and G.O. Hendrickson. 1956. Ecology of the sora in Clay County, Iowa. *Iowa Bird Life* 26(4):78-81. Cited in: R.M. DeGraaf, and M. Yamasaki. 2001. *New England Wildlife: Habitat, Natural History, and Distribution.* University Press of New England, Hanover, New Hampshire.

Taub, F.B. 1989. Standardized aquatic microcosms. *Environmental Science and Technology* 23(9):1064-1066.

Tillitt, D, D. Papoulias, and D. Buckler. 2003a. *Fish Reproductive Health Assessment in PCB Contaminated Regions of the Housatonic River, Massachusetts, USA: Investigations of Causal Linkages Between PCBs and Fish Health.* Final Report of Phase I Studies. Prepared for U.S. Fish and Wildlife Service, Concord, New Hampshire and U.S. Environmental Protection Agency, Boston, Massachusetts. July 2, 2003.

Tillitt, D, D. Papoulias, and D. Buckler. 2003b. *Fish Reproductive Health Assessment in PCB Contaminated Regions of the Housatonic River, Massachusetts, USA: Investigations of Causal Linkages Between PCBs and Fish Health.* Final Report of Phase II Studies. Prepared for U.S. Fish and Wildlife Service, Concord, New Hampshire and U.S. Environmental Protection Agency, Boston, Massachusetts. July 8, 2003.

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

van Zyll de Jong, C.G. 1985. *Handbook of Canadian Mammals – Bats.* National Museum of Natural Sciences. Ottawa, Canada. 212 pp. Cited in: R.M. DeGraaf, and M. Yamasaki. 2001. *New England Wildlife: Habitat, Natural History, and Distribution.* University Press of New England, Hanover, New Hampshire.

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

Weatherhead, P.J. and S.B. McRae. 1990. Brood care in American robins: implications for mixed reproductive strategies by females. *Anim. Behav.* 39:1179-1188.

Whitford, A.G., and A. Vinegar. 1966. Homing, survivorship, and overwintering of larvae in *Ambystoma maculatum. Copeia.* 1966:515-519.

Whitaker, J.O. 1995. Food of the big brown bat *Eptesicus fuscus* from maternity colonies in Indiana and Illinois. *American Midland Naturalist* 134:346-360.

Whitaker, J.O., Jr. and M.G. Ferraro. 1963. Summer food of 220 short-tailed shrews from Ithaca, New York. *Journal of Mammalogy* 44:419.

Whitaker, J.O. and W.J. Hamilton. 1998. *Mammals of the Eastern United States.* Cornell University Press. Ithaca, New York.

Woodlot (Woodlot Alternatives, Inc.). 2003a. *Amphibian Reproductive Success Within Vernal Pools Associated with the Housatonic River, Pittsfield to Lenoxdale, Massachusetts*.

Woodlot (Woodlot Alternatives, Inc.). 2003b. *Ecological Characterization of the Housatonic River*, Appendix A of the *Ecological Risk Assessment of the General Electric (GE)/Housatonic River Site, Rest of River.* Prepared by Weston Solutions, Inc. Prepared for U.S. Environmental Protection Agency and U.S. Army Corps of Engineers.

Wu, J., J.L. Vankat, and Y. Barlas. 1993. Effects of patch connectivity and arrangement on animal metapopulation dynamics: a simulation study. *Ecological Modeling* 65:221-254.

Zabel, E.W., P.M. Cook, and R.E. Peterson. 1995. Toxic equivalency factors of polychlorinated dibenzo-*p*-dioxin, dibenzofuran and biphenyl congeners based on early life stage mortality in rainbow trout (*Oncorhynchus mykiss*). *Aquat. Toxicol.* 31:315-328.