Eighteenmile Creek
Great Lakes Area of Concern (AOC)
Niagara County, New York

FINAL

Bioaccumulation Modeling and Ecological Risk Assessment

Prepared for

US Army Corps of Engineers Buffalo District and

Niagara County Soil & Water Conservation District

Prepared by

E Risk Sciences, LLP

12 Holton Street

Allston, MA 02134

Katherine von Stackelberg

and

US Army Corps of Engineers
Karl Gustavson

January 2012

Table of Contents

List of Tables

List of Figures

Appendices

EXECUTIVE SUMMARY

This report presents the results of the bioaccumulation modeling and associated field sampling efforts at the Eighteenmile Creek Area of Concern (AOC) in response to a request from the USACE Buffalo District. We developed a bioaccumulation model describing the movement of PCB congeners from sediment and water exposure sources through the aquatic food web. The model does not address PCB fate and transport or the relationship between sediment and water. In addition, the model evaluates potential impacts to several ecological receptors.

We used the *TrophicTrace* model based on an aquatic food web model originally developed by Dr. Frank Gobas and colleagues at Simon Frasier University together with ecological risk equations. The *TrophicTrace* model predicts fish tissue concentrations, ecological receptor daily doses, and toxicity quotients for fish and higher order ecological receptors. The *TrophicTrace* model allows users a range of up to four input values to describe uncertainty using interval mathematics. The model generates a "probable" range (based on a median and/or mean) and a "possible" range (based on lower and upper confidence limits).

Sediment and fish tissue were collected to parameterize and calibrate the food web model. Surface sediments (approximately top 6 inches) were collected from below Burt Dam (sediment samples from above Burt Dam were available from recent EPA sampling). Three fish species were collected from above and below Burt Dam and analyzed for PCBs and lipid contents. The ages and stomach contents of these fish were also analyzed to support the food web modeling. We compared modeled fish tissue concentrations to observed tissue concentrations for the two sections of Eighteenmile Creek above and below Burt Dam. The model shows good agreement with measured fish tissue PCB concentrations to within a factor of two or less across the modeled species (pumpkinseed, brown bullhead, largemouth bass) in both River sections.

The comparison of fish body burdens to toxicity reference values from the literature indicate that it is likely that fish in the study area experience exposures that exceed no-effect threshold levels. Although it is less likely that fish body burdens exceed actual effect levels, that cannot be ruled out. We predict toxicity quotients for avian and mammalian ecological receptors based on a comparison of predicted average daily doses to literature-based toxicity reference values. The results for the heron show predicted toxicity quotients that fall below one. Predicted toxicity quotients for the kingfisher show a low potential for exposures to exceed a no-effect threshold level, but it is unlikely that the kingfisher will experience exposures that exceed effect levels. Predicted toxicity quotients for the mink suggest there is a low potential for mink to exceed no-effect threshold levels, and while it is less likely that these exposures exceed actual effect levels, potential exceedances cannot be ruled out.

We conclude that the *TrophicTrace* model adequately predicts fish tissue concentrations in the study area (based on the available data), and that these tissue concentrations are associated with the potential for exposure to PCBs for several receptors (e.g., fish, mink) to exceed no-effect levels, but are less likely to exceed effect levels. Predicted toxicity quotients are slightly higher for Section 2 above Burt Dam than for Section 1 below Burt Dam.

1.0 INTRODUCTION

Eighteenmile Creek is one of forty-three areas of concern (AOCs) established within the Great Lakes due to loss of "beneficial uses" from degraded water quality. The AOC encompasses Eighteenmile Creek from its entry into Lake Ontario, upstream to the Burt Dam (approximately 2 miles). The AOC has three identified use impairments linked to sediment contamination: (1) restrictions on fish and wildlife consumption; (2) degradation of benthos; and (3) restrictions on dredging activities.

Studies since the 1970s have indicated elevated levels of polychlorinated biphenyls (PCBs), chlorinated pesticides, and metals in surficial sediments throughout most of the AOC (for a Summary, see Chapter 4 of the Remedial Action Plan [NYSDEC 1997]). Several studies have documented potential risk to human, aquatic organism, and terrestrial wildlife receptors. New York State Department of Health (NYSDOH) has designated Eighteenmile Creek with its most stringent "Do Not Eat" fish advisory on the basis of PCB contamination. Lake Ontario is subject to other less stringent, species-specific fish advisories related to the presence of PCBs, Mirex, and dioxins/furans (NYSDOH, 2009). The U.S. Army Corps of Engineers (USACE) Buffalo District conducted an evaluation of the toxicity and bioaccumulation of persistent organics in samples from the lower reach collected in 2003 (USACE Buffalo District, 2008); this study indicated that Dichlorodiphenyltrichloroethane (DDE) likely presented a chronic toxicity risk relative to selected freshwater toxicity threshold values and was bioaccumulating at higher than anticipated levels. PCBs were also found to be bioaccumulating. Dioxins were detected in sediment samples and predicted to cause potential wildlife bioaccumulation risks based on an equilibrium partitioning approach used by the New York State Department of Environmental Conservation (NYSDEC).

Invertebrate bioaccumulation testing also suggests that organic contaminants moving through the food web are creating environmental risks (Karn et al., 2004). Specific contamination sources to the creek have not been fully delineated. However, recent investigations by NYSDEC have focused on a contamination source in Lockport, NY, near the upper reach at the Erie Canal (approximately 12 miles upstream of Burt Dam). During investigations in the 1980s and early-1990s, elevated levels of PCBs were detected in sediments near this facility and fish tissue contaminant levels were also elevated (samples above 2 mg/kg total PCBs wet weight) in the creek reach above the Burt Dam (NYSDEC, 1997).

In 2008, a study on the Beneficial Use Impairments of Eighteenmile Creek (Ecology and Environment, 2008) concluded that the impairment was largely due to PCB contamination. This study evaluated contaminant levels in brown bullhead collected below the Burt Dam and at a reference station (Oak Orchard Creek). It showed elevated tissue residues in Eighteenmile Creek fish compared to Oak Orchard Creek, with PCBs exceeding literature-based critical tissue concentrations for PCBs while dioxins/furans did not exceed critical levels (Ecology and Environment, 2008). That report concluded, "Overall, these results suggest that bullhead from Eighteenmile Creek may be at risk from elevated tissue residues of PCBs but not from dioxins/furans" (p. 3-29). A risk evaluation for fish-eating wildlife from PCBs and dioxins/furans was conducted as part of the investigation. The results indicate small excess risk from dioxins/furans to mink with much greater risk from PCBs. Slightly elevated risk to fish-eating birds was indicated for PCBs, but not dioxins/furans (p. 3-36). Risks from chlorinated pesticides were not evaluated in this study.

To date, there have been several data collection efforts in and upstream of the AOC to define PCB levels in sediments, surface water, and biota. However, they have been limited in scope and have not focused on understanding PCB bioaccumulation, movement in the aquatic food web, and consequent environmental risks. Developing such an understanding will assist site managers as they move toward greater resolution on the nature of impairments at the site, develop remedial actions, and ultimately delist the area.

The US Army Engineer Research and Development Center (USAERDC) has completed the bioaccumulation modeling effort presented here in response to a request from the USACE Buffalo District. This is the Final Report of the food web bioaccumulation modeling effort, focusing specifically on PCB contamination in the AOC.

1.1 Components of Modeling Effort

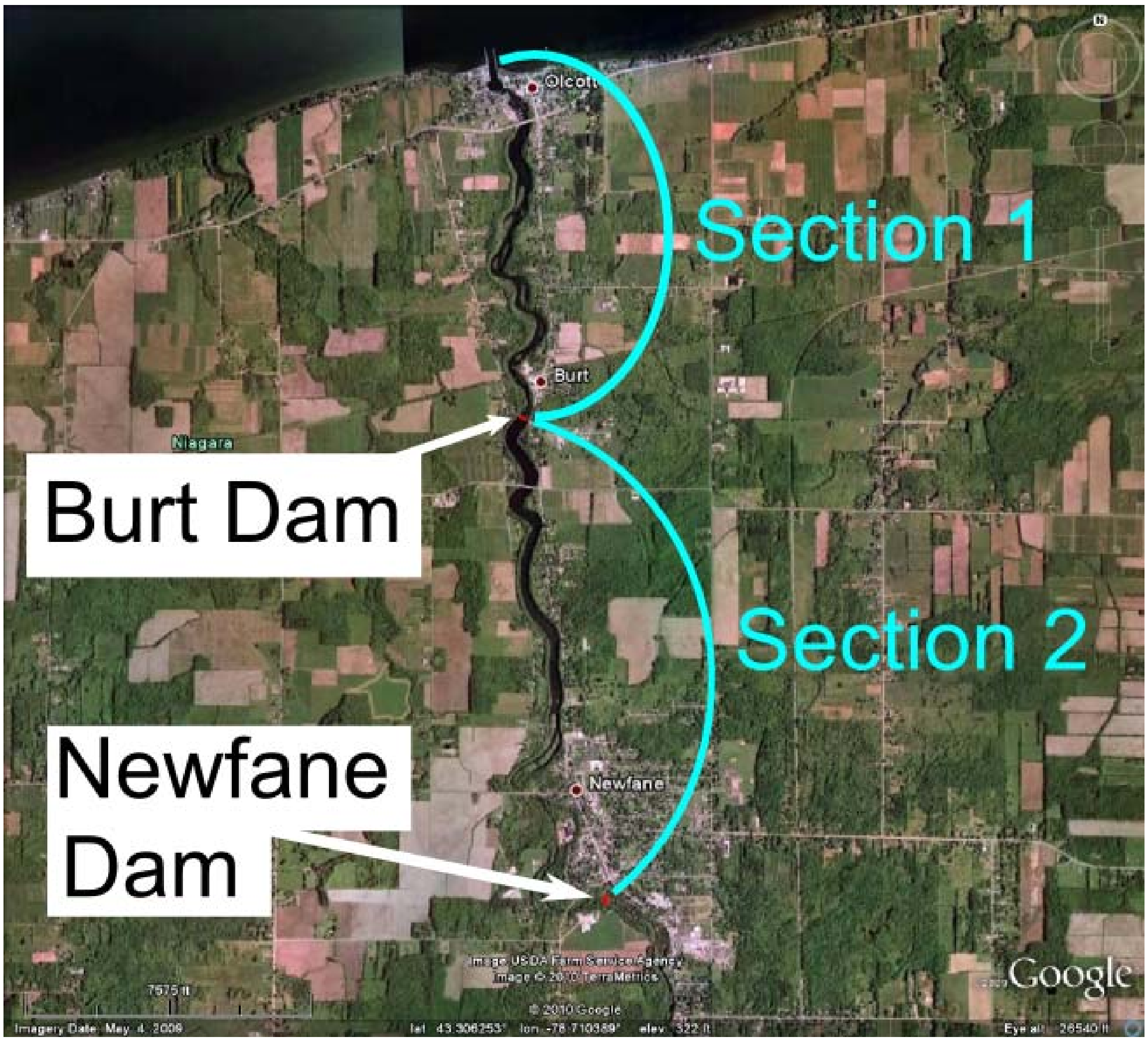
This final report is supported by two interim memoranda:

> 1) Final Data Gaps Memorandum, dated August 3, 2010. This memo provided a description of the food web bioaccumulation modeling to be performed and an associated review of existing contaminant data for Eighteenmile Creek to identify data gaps with respect to spatial resolution, contaminants, and types of organisms used to inform the bioaccumulation modeling effort. This memo recommended sampling/analysis efforts to support development of the modeling effort.
>
> 2) Final Conceptual Site Model (CSM) Memorandum, dated January 21, 2011. This memorandum describes the CSM, providing an overview of the physical, chemical, and biological aspects of the system that are modeled, including site-specific assumptions used to establish modeling conditions.

In this Final Report, we summarize the field sampling effort and resulting analytical data, describe the *TrophicTrace* model and its parameterization, and present model output, including risk estimates to terrestrial aquatic and terrestrial wildlife receptors. As appropriate, critical components of the earlier memoranda are summarized.

2.0 FIELD SAMPLING SUMMARY AND DATA RESULTS

As described in greater detail below, the area below Burt Dam is designated as Section 1 and the area above Burt Dam as Section 2 (See Figure 1). Based on sampling needs identified in the August 3, 2010 Final Data Gaps Memorandum, sediment and fish tissue sampling was conducted in the Eighteenmile Creek AOC, above and below Burt Dam. The sediment data from these and other efforts are used to represent exposure concentrations in the *TrophicTrace* model, and the fish tissue concentrations are used as the empirical basis for model calibration and validation.



d as
n of

PCB contamination is depicted using the sum of detected congeners (rather than individual or sum of Aroclors). It is well known that Aroclors represent the mixture of congeners released to the environment originally, but over time the congener composition changes due to weathering, selective dechlorination, and magnification of particular congeners resistant to dechlorination. Thus, congener-based analyses were selected to depict PCB contamination (see Section 3.3 for a more detailed explanation of PCB toxicity as it relates to congeners versus Aroclors).

2.1 Description of Fish Sampling and Analysis

Fish sampling took place in the Eighteenmile Creek AOC, above and below Burt Dam, on six days between September 13 and 30, 2010. Fish were collected by the U.S. Fish and Wildlife Service (USFWS), Lower Great Lakes Fish and Wildlife Conservation Office in Amherst, NY. Boat mounted electrofishing gear or minnow traps were used for collection. Per the CSM (See the Final CSM Memorandum), largemouth bass, brown bullhead, and pumpkinseed were targeted. These fish species represent different trophic levels, have different feeding strategies, and their tissue concentrations can be simulated in the *TrophicTrace* Model (see Section 3).

Sampling was performed according the Fish Tissue Field Sampling Plan included in Appendix 1. The field sampling plan describes targeted species, size ranges, and numbers of fish along with sampling contingencies. A summary of the collected fish is included in Table 1.

Table 1. Characteristics of Collected Fish

	Number Collected		**Size (inches) (min-max, geometric mean)**		**Weight (grams) (min-max, geometric mean)**	
	Section 1	Section 2	Section 1	Section 2	Section 1	Section 2
Largemouth Bass	10	10	11.2 - 15.0, 12.9	12.6 - 15.0, 13.6	364 - 956, 547	492 - 884, 607
Brown Bullhead	9	10	8.9 - 11.1, 10.5	9.5 - 11.9, 10.7	146 - 312, 244	188 - 450 284
Pumpkinseed (5-fish composite)	10	11	NA	NA	NA	NA

NA: Not applicable. Weight and length measurements were not taken on the pumpkinseed to be composited. Composites were 5 fish of <4 inches in length (A2R2PKN5 was a composite of 4 fish, see Appendix 2).

Target Species, Size, and Number

For all three species, the targeted number of fish was collected. While the bullhead were slightly larger than those targeted, this does not affect the overall basis for their inclusion which was a close association with sediments and sediment derived food sources (see stomach content analysis). The largemouth bass were within the targeted (contingency) size range. For pumpkinseed, the contingency to expand to alternate *Lepomis* spp. was used. The collection included a mixture of bluegill, green sunfish, and pumpkinseed, all below 4 inches in length. At that size range, *Lepomis* spp. share a similar trophic level and feeding preference. Overall, the size of fish sampled from Sections 1 and 2 was quite similar. Appendix 2, the fish sampling field collection log, contains descriptions of each fish and composite.

Targeted Sampling Areas

Sampling occurred in the creek sections above and below Burt Dam. Each section was divided into 3 equal reaches in an effort to collect fish equally throughout the section (see Figures 1 through 3 of Appendix 1). This was intended to permit an evaluation of the relationship between sediment and fish contaminants in a per-reach basis. However, fish could not be equally sampled from each reach, either because they were not equally distributed or collection techniques were not equally effective in all reaches. Thus, the sampling plan's contingency to collect fish throughout the whole section was used. Table 2 summarizes the distribution of sampled fish within the designated reaches of each section. Appendix 3 to this report presents locations of the electroshocking runs or traps along with the fish collected at those locations.

Table 2: Numbers of Fish (or Composites) Collected in Individual Reaches of Sections 1 and 2

		Reach 1	Reach 2	Reach 3
Section 1	Largemouth Bass	3	4	3
	Brown Bullhead	4	0	5
	Pumpkinseed	3	3	4
Section 2	Largemouth Bass	3	7	0
	Brown Bullhead	0	10	0
	Pumpkinseed	0	8	3

Fish Processing and Shipping

Several field days were required to achieve targeted numbers and species in the two river sections. Fish were processed according to procedures documented in Appendix 1. Following each day's sampling, fish were bagged and labeled. Fish were then frozen at the USFWS, Lower Great Lakes Fish and Wildlife Conservation Office until shipping. Chain of custody forms are provided in Appendix 4.

Fish Lipid and Contaminant Analyses

Whole fish were submitted to the Environmental Chemistry Laboratory at USAERDC for analysis of PCB congeners and lipid. Fish samples were thawed, contents of stomach removed (see Section 2.3), and the entire fish (or composite for the pumpkinseed samples) was ground using a meat grinder. Approximately 5g of fish were weighed into vials and extracted overnight with 10% acetone in hexane in a sonic bath. Extracts were filtered followed by cleanup using florisil (SW846 3620 modified) and sulfuric acid (SW846 3665). Extracts were concentrated to 2mL and analyzed by Gas Chromatography/Electron Capture Detector (GC/ECD) using dual columns (SW846 8082).

2.2 Description of Sediment Sampling and Analysis

Sediment sampling took place on October 26, 2010. Sediment surface grab samples were collected by the USACE Buffalo District at 16 locations throughout Section 1 (See Table 3). Field duplicate samples were also collected at two locations (EMC-4 and EMC-12). Sediment samples were analyzed for PCBs to represent sediment exposure concentrations. The Section 1 sampling and analysis will be further described below.

Sediments in Section 2 in the Burt Dam reservoir were collected for PCB analysis in May 2010 by the Great Lakes National Program Office (GLNPO). In addition to an extensive vibracoring effort in the Burt Dam reservoir, surface sediment samples were collected using a petite ponar grab (that device was also used in Reach 1 sampling). Section 2 sediment sampling and analyses are described in Eighteenmile Creek Site Characterization Data Summary Report (CH2MHill 2011)[1]. Briefly, ponar grab samples were collected at 27 locations and analyzed for TOC and PCB congeners, among other analytes (See Table 2-1b in CH2MHill 2011). The location of sediment sampling is depicted in Figures 2-1 and 2-2 of the Data Summary Report. The ponar sampling data will be used in this report to represent sediment PCB exposure concentrations in Section 2 (see Section 3.2.1 for further discussion).

Table 3. Sediment Sampling Locations in Section 1

Sample	Latitude	Longitude
EMC-1	43 20.316	78 43.127
EMC-2	43 20.222	78 42.999
EMC-3	43 20.146	78 42.931
EMC-4	43 20.043	78 42.978
EMC-5	43 19.936	78 42.958
EMC-6	43 19.825	78 42929
EMC-7	43 19.745	78 42.942
EMC-8	43 19.619	78 43.016
EMC-9	43 19.519	78 43 073
EMC-10	43 19.437	78 43.038
EMC-11	43 19.352	78 42.969
EMC-12	43 19.279	78 42.876
EMC-13	43 19.177	78 42.859
EMC-14	43 19.115	78 42.953
EMC-15	43 19.059	78 43.022
EMC-16	43 18.912	78 42.907

Section 1 Sediment Sampling and Analysis

Section 1 sediment sampling was performed according to the Sediment Field Sampling Collection Plan included in Appendix 5. Field notes from that effort are included in Appendix 6. Collection was typically by petite ponar. Sediment samples were collected in 8 ounce jars for contaminant analysis and 4 ounce jars for analysis of total organic carbon (TOC). At site EMC-16 near the Fisherman's Park area, the sediment was predominantly gravel (with some minor sand mixed in), so sampling was accomplished by scoop. Field personnel stated that sampling using the ponar sampler was to a depth of approximately 3-6 inches. When poor recoveries of sediment occurred (typically due to hard or gravely substrate), the location was moved to achieve appropriate sample volume. Actual sampling locations varied slightly (20-30 feet) from the locations designated in the sampling plan. Actual sampling locations are presented in Table 3.

[1]In CH2MHill (2011), the area between Burt and Newfane Dam is designated as sections 2 and 3

Sediment samples were processed according to procedures documented in Appendix 5. Sediments were collected, processed, and shipped on the same day. Samples were packed on ice and shipped via overnight delivery to the USAERDC chemistry laboratory for contaminant and TOC analyses. Chain of custody forms are provided in Appendix 7.

Sediments were analyzed for PCB congeners. Approximately 15g of sediment were extracted by Accelerated Solvent Extraction (SW846 3545) followed by clean up with acid (SW846 3665). Extract volume was adjusted to 2mL and extracts were analyzed using GC/ECD with dual columns (SW846 8082).

2.3 Description of Fish Stomach Content and Aging Analyses

Prior to grinding fish for contaminant extraction, stomach contents (not the stomach itself) were removed and examined at 160x magnification to determine prey items and establish their percent volume or mass of the entire gut contents. Individual bullhead and largemouth bass were aged using either scales or pectoral spines. See Appendix 8 for a complete description of methods and results.

2.4 Results from Analyses of Sediments and Fish

Sediment Chemical Analysis Results

Table 4 provides a summary of the PCB and TOC analytical data for Section 1 and 2 of Eighteenmile Creek. Total PCB and TOC data from individual samples collected from Section 1 are presented in Appendix 9 (Section 2 data, not collected as part of this effort, are included here for ease in comparison). Figure 2 presents these results graphically, interpolating PCB results between data points. Data files from the USAERDC chemistry laboratory with the full suite of congeners, Aroclors, and quality control samples and information have been provided electronically to the sponsor. For the purposes of data analysis and modeling, non-detect values were assigned ½ the detection limit.

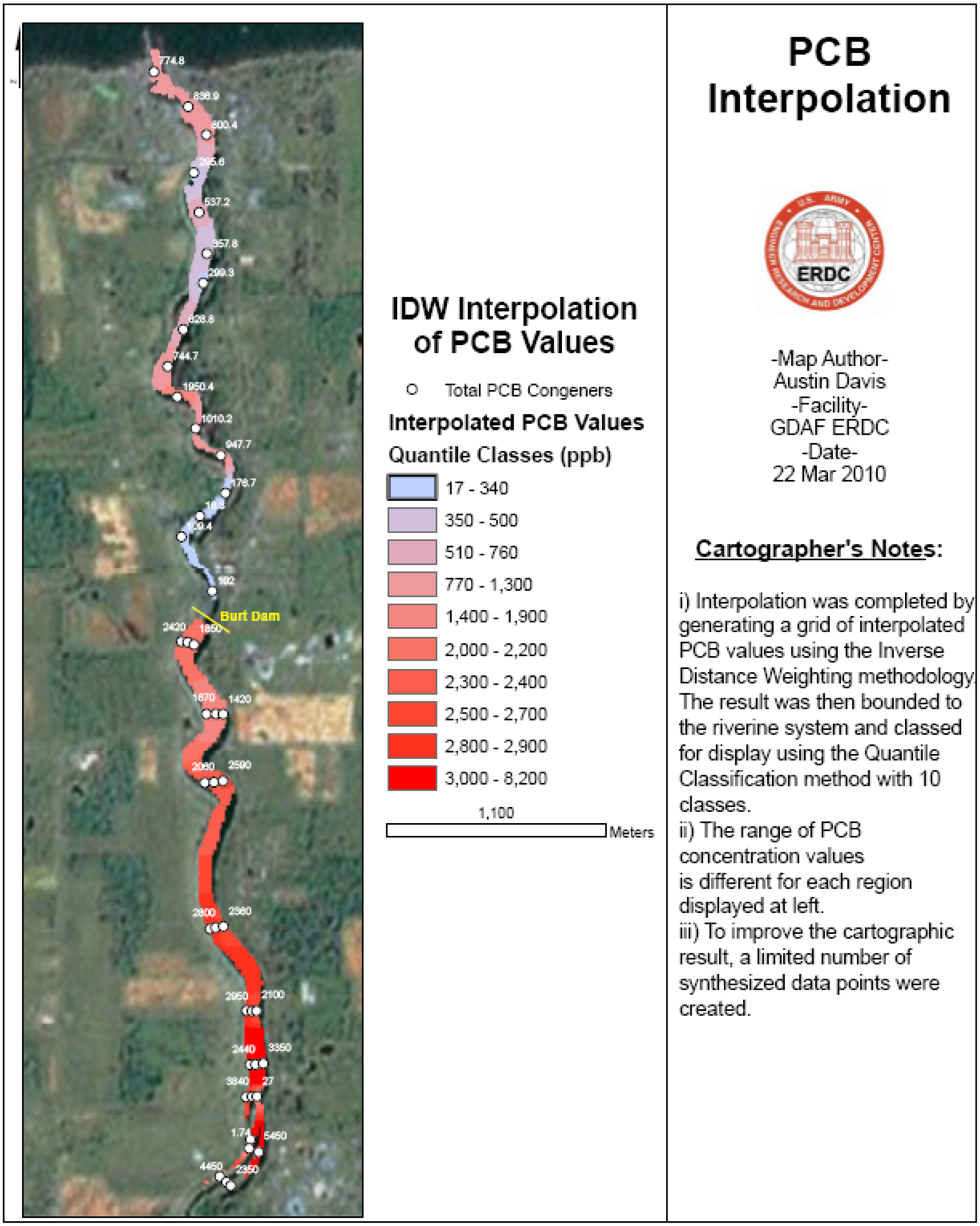


Figure 2: PCB Concentrations in Sediment. Figure depicts total PCB concentrations interpolated across the section (shown as colors) and as individual sampling points (white circles). See Appendix 9 for individual sample values.

Table 4: Summary Statistics for Sediment at Eighteenmile Creek.

PCB Concentration							
Section	n	Average	Standard Deviation	5% LCL	95% UCL	Min	Max
		ug/kg dry weight					
1	16	601	484	343	859	16.3	1950
2	27	2640	1691	1971	3309	1.7	8300

TOC-normalized PCB Concentration					
Section	n	Average TOC-norm	Standard Deviation TOC-norm	Min TOC-norm	Max TOC-norm
		mg/kg TOC-normalized			
1	16	33.5	17.5	1.3	55.8
2	27	60.3	40.6	0.07	203

Total Organic Carbon (TOC)						
Section	n	Average TOC	Min TOC	Max TOC	5%LCL	95% UCL
		Percent				
1	16	1.6	0.69	3.8	1.2	1.9
2	27	4.5	0.34	9.3	3.8	5.2

Fish Tissue Chemical Analysis Results

Table 5 provides summary statistics for the fish tissue data, and Table 6 provides a summary of the lipid data. Total PCB and lipid data from individual samples are presented in Appendix 9. Data files from the USAERDC chemistry laboratory with the full suite of congeners, Aroclors, and quality control samples and information have been provided electronically to the sponsor.

Table 5: Summary of Total PCB Concentrations Based on Sum of Congeners (mg/kg wet weight) from Whole Fish Collected in Each Section of Eighteenmile Creek

Section	Number of fish	Arithmetic Mean	Standard Deviation	Minimum	Maximum	Geometric Mean
Largemouth Bass (LMB)						
1	10	2.5	1.8	0.7	6.3	1.9
2	10	8.6	9.4	1.8	34.6	6.2
Brown Bullhead (BB)						
1	9	1.2	0.5	0.4	2.1	1.1
2	10	2.7	1.4	1.1	6.3	2.4
Pumpkinseed (PKSD) [1]						
1	10	2.4	1.2	1.2	5.1	2.2
2	10[2]	2.4	1.1	0.8	4.3	2.2

[1]Pumpkinseed represent a composite of individual *Lepomis* spp. See section 2.1. Eleven pumpkinseed composites were collected from Reach 2. PKN11 was collected just below the Newfane Dam, well above the reservoir and area with sediment chemistry. Therefore PKN11 chemistry was not included.

Table 6. Summary of Lipid Data (Percent) from Eighteenmile Creek Fish

Species	Number of fish	Average	Standard Deviation	Minimum	Maximum	5% LCL	95% UCL
LMB	20	2.9	1.2	1.2	5.1	2.3	3.4
BB	19	3.1	1.3	0.6	5.5	2.4	3.7
PKSD	20	2.4	0.9	1.0	3.9	2.0	2.8

Fish Stomach Content and Aging Analysis Results

Table 7 provides a summary of the composition of the diets of largemouth bass and brown bullhead from the site. Table 8 summarizes the ages of the collected largemouth bass and bullhead. The relationship between the age group of the fish and lipid normalized PCB total PCB concentrations is presented in Figure 4. A discernible relationship of PCB concentration with age is noted only for largemouth bass in Section 2 (however, only two age groups were collected in that Section). This observation is mostly driven by the anomalously high PCB concentration in the single fish. Overall, fish tissue PCB concentrations do not appear to vary as a function of age in the sampled population.

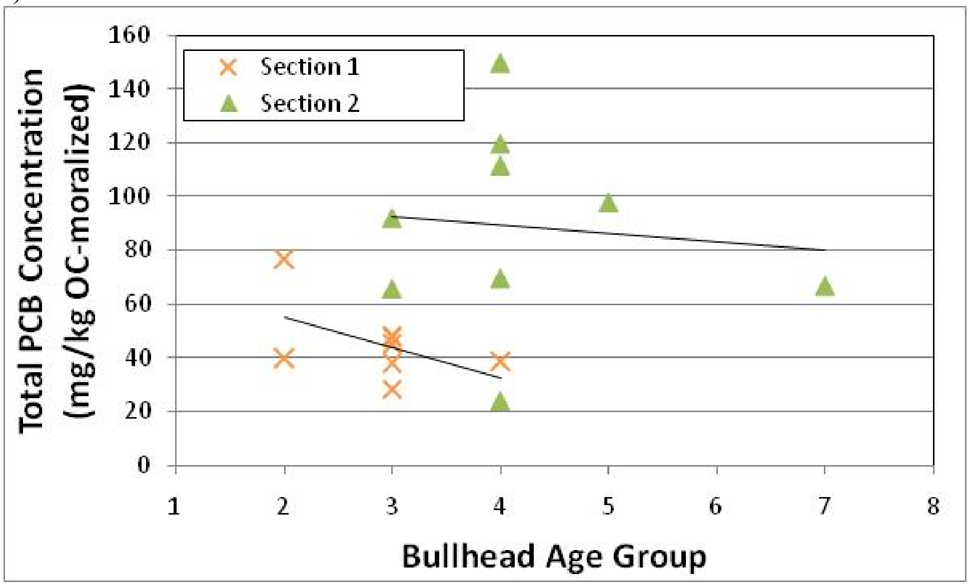
Table 7: Composition of the Diets of Largemouth Bass and Brown Bullhead from Eighteenmile Creek. Overall prey number, food volume, and food weight are means (and standard deviations). Prey frequency (Freq), number, weight (bass), and estimated volume (bullhead) are percentages of total value. See Appendix 8 for details.

	Largemouth Bass N = 20				Brown Bullhead N = 19			
% With Food	60				89			
Prey Number	0.90 (0.19)				4.16 (0.95)			
Food Volume (mm^3)	1.92 (0.58)				1.80 (0.43)			
Food Weight (g)	1.77 (0.52)				2.02 (0.47)			
Prey	Freq	Number	Weight	RI	Freq	Number	Volume	RI
Algae & Detritus	-	-	-	-	5	16.35	64.13	51.78
Vascular Plant	5	5.5	0.4	0.01	32	7.69	11.56	28.09
Seed	-	-	-	-	16	3.85	0.41	0.25
Bryozoa	-	-	-	-	5	11.30	0.03	0.02
Physidae	-	-	-	-	21	17.79	2.44	9.00
Ancylidae	-	-	-	-	5	1.20	0.03	T
Planorbidae	-	-	-	-	5	1.20	0.03	T
Gastropoda (UNID)	-	-	-	-	5	1.20	0.06	T
Sphaeridae	-	-	-	-	10	2.40	0.03	0.01
Dreissenidae	5	5.5	0.2	0.01	5	1.20	0.01	T
Invertebrate (UNID)	-	-	-		5	8.65	0.06	0.03
Cambaridae	30	38.9	70.0	92.16	16	6.25	10.20	10.07
Aranea	5	5.5	2.2	0.07	-	-	-	-
Anisoptera	-	-	-	-	5	1.20	0.06	T
Gyrinidae	5	5.5	0.2	0.01	-	-	-	-
Coleoptera (UNID)	5	5.5	1.8	0.06	5	1.20	0.06	T
Hydroptilidae	-	-	-	-	5	1.20	0.03	T
Trichoptera (UNID)	-	-	-	-	16	3.85	0.12	0.07
Chironomidae	-	-	-	-	16	3.85	0.09	0.05
Diptera (pupae)	-	-	-	-	5	1.20	0.03	T
Insecta (UNID)	-	-	-	-	5	1.20	0.03	T
Cyprinidae	5	5.5	6.6	0.20	-	-	-	-
Centrarchidae	5	5.5	3.8	0.12	5	1.20	2.35	0.14
Percidae	-	-	-	-	5	1.20	2.94	0.17
Perciform (UNID)	-	-	-	-	5	1.20	0.23	0.01
Fish (UNID)	20	22.2	14.7	7.36	5	1.20	0.03	T
Vertebrate (UNID)	-	-	-		5	1.20	5.00	0.30
Total	n/a	99.6	99.9	100	n/a	98.8	99.96	99.99

Table 8: Numbers of Largemouth Bass and Brown Bullhead from Eighteenmile Creek for Each Age Group

Age Group	Largemouth Bass	Brown Bullhead
II	-	2
III	-	7
IV	9	6
V	9	1
VI	1	-
VII	-	1

a)



b)

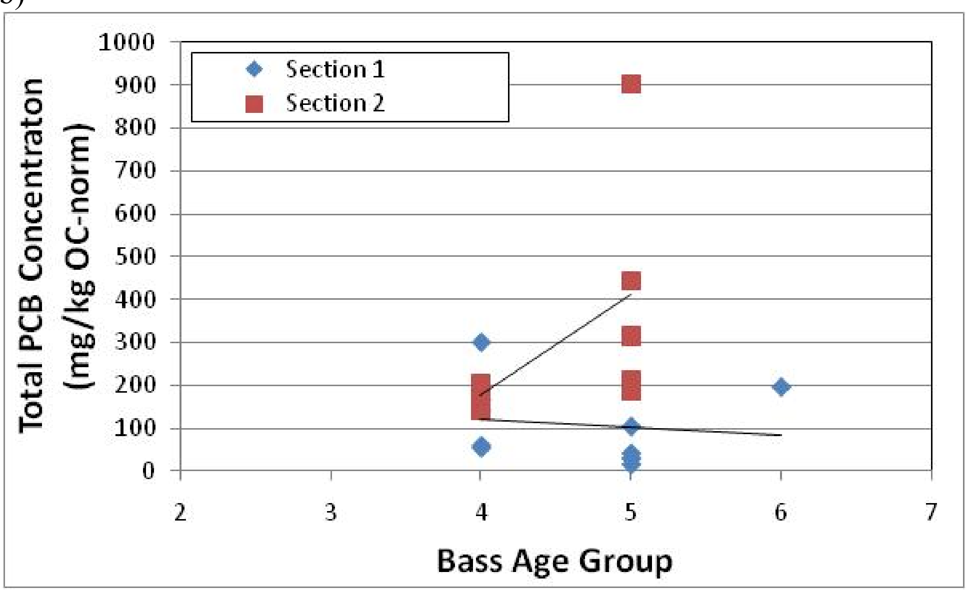


Figure 3: Relationship Between Fish Age Group and Lipid-normalized PCB Concentration. Individual data points are presented for a) bullhead and b) largemouth bass in both sections of the Eighteenmile Creek study area. Linear trend lines for each section are also included.

3.0 *TROPHICTRACE* MODEL DEVELOPMENT AND RESULTS

We developed a bioaccumulation and ecological risk model using the *TrophicTrace* food web bioaccumulation model. The *TrophicTrace* model is a steady-state model that predicts the expected body burden in fish and then uses these results as inputs to the ecological risk model to predict average daily doses to ecological receptors. Finally, the model estimates toxicity quotients (TQs) by dividing the receptor's average daily dose by the toxicity reference values from the literature. TQs are estimated for ecological receptors exposed to expected (e.g., average) conditions and also develops uncertainty bounds around predicted central estimates. The bioaccumulation modeling portion of *TrophicTrace* and its underlying mathematical structure (Gobas 1993) are well-accepted and have been used in a number of regulatory applications (Gustavson et al. 2011). Appendix B of the Data Gaps Memorandum provides more detailed background information regarding the *TrophicTrace* Bioaccumulation Model.

Ecological risk can be evaluated in different ways. Evaluating potential adverse effects across a population of receptors requires evaluating multiple lines of evidence. There are assessment endpoints -- the ecological endpoints that we are interested in protecting (e.g., growth, reproduction and sustainability of fish and wildlife populations) -- and measurement endpoints (e.g., those aspects of ecosystem function that we measure to evaluate the assessment endpoints -- the lines of evidence). Evidence ranges from the type of modeling presented here (e.g., comparisons of predicted body burdens and/or doses to no- and lowest-observed effect levels from the literature) to various kinds of field studies. If the modeling results presented here suggest the potential for receptors to experience exposures in the field that are likely to exceed no- or lowest observed effect levels from the literature, then that would provide an indication that additional field studies may be warranted in order to draw stronger conclusions concerning the potential for adverse effects to occur as a result of exposures in the study area.

3.1 Modeling Area

The model is run separately for each of the two creek sections described earlier: Eighteenmile Creek from Lake Ontario to Burt Dam (Section 1) and an upper reach from Burt Dam to the Newfane Dam (Section 2) (Figure 1). The definition of the two sections assumes that fish populations will not interact and only be exposed to conditions in those sections.

The AOC (Section 1) and the Burt Dam reservoir area (Section 2) are the closest in environmental conditions, habitat, and fishery, so they are appropriate conditions to fulfill the SOW objective "to evaluate organic contaminant bioaccumulation, trophic transfer and consequent risks in creek sections above and below Burt Dam of the Eighteenmile Creek." The Burt Dam Reservoir extends approximately 2 miles before more typical stream morphology continues for another mile to the Newfane Dam. The Newfane Dam along with the relatively swift shallower bedrock and gravel channel below the Newfane dam are hydraulically significant features and serve as impediments to fish movement. Fish and sediment in Section 2 were collected from the reservoir (Figure 2), not all the way up to Newfane dam. Modeling results are most applicable to the area encompassed by sampling. Conditions are more complex upstream of Newfane Dam with more typical stream reach/run morphology; these areas will possess fewer fine-grained sediments, support a different fishery, and exhibit a different dynamic of contaminant exposure between modeled organisms, sediments, dietary constituents, and water. However, as described earlier, contamination source areas and impacted receptors extend further upstream of the modeled sections to the city of Lockport at the Erie Canal.

3.2 Assessment and Measurement Endpoints

Assessment endpoints are selected to represent those aspects of the study area ecosystem that are to be protected and potentially at risk from exposures to PCBs. In general, ecological risk assessment endpoints relate to populations rather than individuals, and should reflect ecosystem structure and function at higher levels. The assessment endpoints selected for this study area include:

- maintenance and sustainability of fish that serve as a prey base for other fish and wildlife (as represented by forage fish as described in Section 3.3.2); and,

- maintenance and sustainability of wildlife (as represented by specific receptors as described in Section 3.3.2).

Measurement endpoints are those aspects of the ecosystem that can be measured. For this study area, the primary concern is potential exposures of ecological receptors to PCB concentrations in sediment. Available measurements include PCB concentrations in sediment and fish. Using the *TrophicTrace* bioaccumulation model, we first predict expected PCB concentrations in fish (and compare the results to data), and then predict expected average daily doses to higher-order receptors and compare those to no- and lowest-observed effect levels from the literature.

3.3 Model Parameterization

To the extent possible, development and parameterization of the *TrophicTrace* model relied on site-specific data. This section describes the modeling assumptions and data used to parameterize the *TrophicTrace* model.

The *TrophicTrace* model predicts the mean expected fish tissue contaminant concentration and this is the basis of all model prediction-data comparisons. However, the mean observed fish tissue contaminant concentration is uncertain as reflected by the standard error on the mean from the data. The model is designed to capture the uncertainty in the mean by allowing model inputs for each module (e.g., bioaccumulation, ecological risk) to be specified as a range, representing the uncertainty in the mean estimate of that input. Fuzzy set theory or interval analysis (Zadeh, 1965; Zimmerman, 1985) can be used to propagate uncertainty in a mathematical model when there is insufficient information to use a more sophisticated framework (e.g., probabilistic approaches) or when the equations are too complex to allow for analytical approaches (Hammonds et al., 1994). Fuzzy set approaches have been used for risk assessment applications (Guyonnet et al., 1999; Huang et al., 1999; Lee et al., 1994) and for fate and transport studies (Dou et al, 1995; Bardossy et al., 1995).

When possible, each model input value is defined by three (in some cases, four) numbers. These values represent two ranges: a probable range (the likeliest range based on average input values) and a possible range, analogous to an upper and lower 95% confidence limit. In terms of predictions, the probable range represents the best estimate of predicted fish body burdens, average daily doses, and resulting TQs based on using central tendency estimates for each input value in the model. The possible range represents the lowest and highest possible predicted risks based on using a 5% lower confidence and 95% upper confidence interval of the mean, respectively, for each input value. We use the probable TQ range to make a determination of potential risk, and use the possible range to describe our confidence in those conclusions.

Figure 4 provides a graphical representation of how the final TQ ranges are calculated. The highest possible TQ value (and prior to that, average daily dose, and prior to that, predicted fish body burden) is derived by simultaneously combining the series of all upper bound input estimates. Note that it is unlikely that all input values would simultaneously occur at the 95% upper confidence limit value in a steady-state fashion (that is, ecological receptors would consume fish exposed to a 95% upper confidence sediment concentration consistently over time and so on). In the same manner, the lowest possible predicted value represents a lower bound on the risk estimate, derived by simultaneously combining all 5% lower confidence level values for model inputs. The possible range is calculated for the purpose of uncertainty analysis to provide perspective on how high (or low) predicted TQs could be given our uncertainty in the input values.

For this analysis, the probable input range is represented by a single value (the mean value of the input variable) for most of the inputs except for temperature, Log K_{ow}, dissolved water concentration, and the Toxicity Reference Values (TRVs). The possible range reflects the uncertainty in the central tendency input value as reflected by a 95% confidence interval on either side of the mean. The exception to this is the input values for the TRVs. As described in Section 3.3.1, there is more than one mean TRV available based on individual studies from the literature, so these inputs are based on at most four values (low, medium, medium-high-and high), but in some cases only two. For the dissolved water concentration, the data do not support development of more than one value (and PCB uptake from water is expected to have minimal influence on fish tissue concentration), so a single value was used.

The specific inputs selected to represent environmental conditions, contaminant exposures, and the food webs are described in the rest of this section. Table 11 at the end of this section provides an overall summary of all inputs to the model.

Probable Range for Predicted Average Daily Dose	Probable Range for Toxicity Reference Value
10	5
20	30

Example Calculation

10/5 = 2	within the range
10/30 = 0.3	← Low
20/5 = 4	← High
20/30 = 0.6	within the range

Predicted Toxicity Quotient Probable Range Low	Predicted Toxicity Quotient Probable Range High
0.3	4

Figure 2: Example of the Calculation to Estimate the Probable Range of Predicted Toxicity Quotients. Probable and possible ranges for outputs (e.g., fish tissue concentrations, average daily doses for ecological receptors and toxicity quotients across fish and ecological receptors) are generated through combinations of inputs in the food web equations based on interval mathematics. For example, since ecological risk (measured as a toxicity quotient [TQ]) is described mathematically by dividing a predicted average daily dose (for ecological receptors) or a fish tissue body burden (for fish) by a toxicity reference value (TRV), the lowest probable TQ is obtained by dividing the lowest predicted average daily dose by the highest TRV; the highest probable TQ is obtained by dividing the highest predicted average daily dose by the lowest TRV. The same is true for the possible range.

3.3.1 Environmental Inputs

Sediment and Water Exposure Concentrations

Sediment PCB and TOC concentrations were derived from sampling conducted in May (Section 2) and October 2010 (Section 1), which correspond closely to the September 2010 fish sampling that occurred in both Sections. Water concentrations were derived from site literature. Only a single freely dissolved water concentration data point of 80 ng/L was available; this datum was obtained in 1998 at a location just below Burt Dam in Section 1 (Garabedian et al., 2001).[2] This value was used as the input concentration for dissolved water in Section 1, and proportionally increased to 100 ng/L. The 25% increase in concentration for Section 2 was a simple estimate derived considering the higher surface sediment PCB concentrations in Section 2, the location of the single sample (at the base of the dam, water could realistically be considered either Section 1 or 2 water), and the small role that direct water uptake plays with respect to bioaccumulation through the food web. The need for additional water data is examined further in Section 4,

[2] Since 2002, the U.S. Environmental Protection Agency (EPA) has collected Eighteenmile Creek surface water samples for analysis of PCBs and other pollutants (e.g. Zevin 2011). However, these analyses are of "whole water" samples, and do not represent the freely dissolved PCB fraction which is needed for food web modeling.

Discussion of Uncertainty. Table 9 summarizes the sources of exposure concentration data for each Section. Sediment concentrations are represented in the model by three values: we selected the mean value from the data to represent the probable input range, and the lower and upper 95% confidence limits of the mean to represent the possible range.

Table 9: Summary of Data Sources for Exposure Input Concentrations

	Sediment	**Water**
Section 1	USAERDC and USACE Buffalo District October 2010 PCB congener analysis of surface sediments.	Garabedian et al., 2001; sample collected by NYSDEC in November 1998.
Section 2	GLNPO May 2010 PCB congener analysis of surface sediments (CH2MHill 2011); data received electronically in October 2010 from Ecology and Environment.	Extrapolated from single measurement from Section 1.

Temperature
Several parameters within the food web model are temperature dependent, including feeding rate and growth rate. Temperature inputs were derived from a number of sampling programs at Eighteenmile Creek and reflect an April to October average, which is the most active foraging time for fish. Historical temperature data for the site were compiled in the Data Gaps Analysis (see 18miCk_water.xlsx, submitted August 2010) from sampling conducted by US EPA in 2004, 2005, 2006; NYSDEC and U.S. Geological Survey (USGS) in 1990; and USGS back to the mid-1970's. Four input values are used to represent the range in expected temperatures in Eighteenmile Creek over the period from April to October. The probable range for temperature is based on the median and mean of the available data; the possible range is represented by the lower and upper 95% confidence limits on the mean.

Log K_{ow}
The octanol-water partitioning coefficient (Log K_{ow}) is an indicator of the hydrophobicity of a compound and its partitioning between water and organic phases (e.g., organic carbon and lipid). The term is used to derive several rate constants in the model. We used four input values to represent the range of expected Log K_{ow} values. The probable -- likeliest -- range for Log K_{ow} is 6 to 6.2. Aroclor 1248 (Log K_{ow} approximately 6) is the primary Aroclor at the site; a mixture of 20% Aroclor 1260 and 80% Aroclor 1248 has an approximate Low K_{ow} of 6.2.[3] The possible range is represented by a low of 5.8 and a high of 6.4. Those values represent the uncertainty in the mean (e.g., the best estimate for the mean value of Log K_{ow} is 6 to 6.2, but that mean could be as high as 6.4 or as low as 5.8, given specific, but unknown, conditions in Eighteenmile Creek).

[3] Aroclor 1260 was found in fewer than half of the samples analyzed between the Burt Dam and Newfane Dam by in the May 2010 sediment sampling effort (CH2MHill 2011). When Aroclor 1260 was found in surface sediments (top 1 foot), it was approximately 20% of the total Aroclor present.

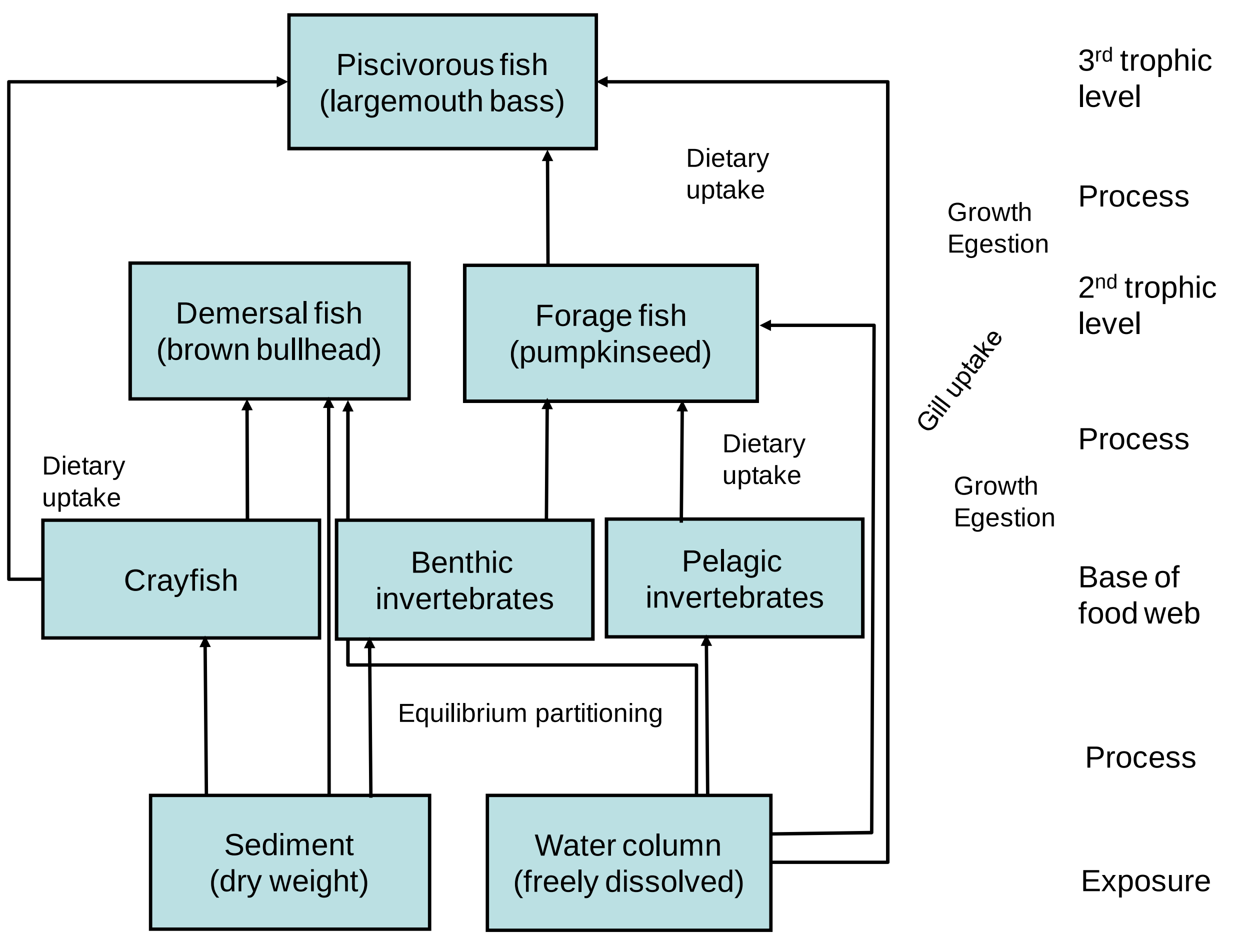
Benthos-Sediment Accumulation Factor (BSAF)

The BSAF assumed in the modeling is based on a site-specific laboratory study using sediments obtained from Section 1 in Eighteenmile Creek. This laboratory study exposed *Lumbriculus variegatus* to site sediments and developed ratios of lipid-normalized benthic invertebrate concentrations to TOC-normalized sediment concentrations (USACE Buffalo District 2008). That study recommends a BSAF of 1.7 for total PCBs. We therefore represented both the probable and possible ranges for BSAF using 1 as the lower bound and 1.7 as the upper bound for the crayfish and benthic invertebrate food web compartments. A BSAF = 1 was maintained for the sediment compartment which, based on gut contents analyses, was a significant dietary component of the brown bullhead.

3.3.2 Food Web Composition and Exposures

The CSM memorandum introduced the proposed food web and provided qualitative information on food web composition. Since that time, the foodweb model was modified based on input from the sponsor and stakeholders (e.g., upper level terrestrial receptors were modified to include known inhabitants of the area)and from stomach content analyses of collected fish. More quantitative details regarding the modeling framework are provided in this section.

Figure 5: Conceptual Site Model for the Aquatic Food Web at Eighteenmile Creek



Aquatic Food Web

Figure 5 presents a simplified CSM for the aquatic food web in Eighteenmile Creek, and Table 11 provides a summary of all the input data to the *TrophicTrace* model (except for dietary preferences, found in Table 10). The goal is to develop a modeling framework that captures exposures to PCBs in sediment. Because PCBs are known to bioaccumulate, it is also important to include fish species that consume other fish, and to focus on permanent residents of Eighteenmile Creek rather than transient species.[4] The proposed food web starts with invertebrates that serve as a prey base for forage, demersal, and piscivorous fish.

Benthic Invertebrates

Contaminant concentrations in benthic invertebrates are assumed to be in equilibrium with those in local sediments. We model two kinds of benthic invertebrates and a detrital sediment compartment as food sources. The first benthic invertebrate compartment (benthos) represents the range of infaunal and epifaunal organisms typically found in riverine systems. Lipid content for the benthos is based on a large database of lipid across sieved organisms from the Hudson River sampling program (US EPA 2000). As mentioned, a site-specific benthic BSAF of 1.7 was applied to the equilibrium partitioning equation to estimate concentrations in this compartment.

The second compartment for benthic invertebrates is represented by crayfish, based on the stomach contents analysis (see Table 7) that showed that largemouth bass, in particular, consume a large amount of crayfish. Crayfish lipid content was obtained from Gewurtz et al. (2000) and Lin et al. (2000). Across the two studies, data from 43 crayfish was available for parameterizing the *TrophicTrace* model. The BSAF of 1.7 was also applied to crayfish. There were not site-specific data on crayfish contaminant or lipid concentration.

The third "invertebrate" compartment (sediment) is represented by detrital matter and vegetation found on the surface of sediment. The stomach contents analysis (Table 7) of brown bullhead collected from the site showed a very high dietary portion of detritus and vegetation (an observation similar to other studies on bullhead diets [e.g., Eddy and Surber, 1947]). A BSAF was not applied to this compartment as consuming these items is analogous to direct sediment consumption. Equilibrium partitioning from sediment was used to estimate concentrations in this compartment, together with an assumption of lipid content equivalent to that found in the benthos compartment.

Pelagic Invertebrates

Pelagic invertebrates are assumed to be in equilibrium with dissolved-phase water concentrations.

[4] Eighteenmile Creek is an important recreational salmonid fishery. The salmonids spend the majority of their life cycle in the open water of Lake Ontario, returning to Eighteenmile Creek to spawn. Because they are primarily open water residents, they are not appropriate to include in food web modeling that focuses on Eighteenmile Creek sediment. In addition, salmonid species are routinely stocked; the resident warm-water species analyzed in this study are not (Niagara County Soil and Water Conservation District 2007).

Forage Fish
The forage fish sampling program focused on small forage fish (*Lepomis spp*.) less than 4 inches in size. In this size range, the primary food source is zookplankton with some epibenthic invertebrates, depending on the species (Mittelbach 1984).

Brown Bullhead (*Ameriurus nebulosus*)
The next feeding guild that is important to capture is the demersal fish, such as brown bullhead. Collected fish ranged from approximately 9 to 12 inches (Table 1). In this size range, bullhead consume primarily benthic food sources and a small percentage of pelagic invertebrates. Stomach content analysis of the collected fish (Table 7) shows that bullhead consumed primarily sediment-associated detrital matter, with approximately 10% pelagic invertebrates (e.g., snails [Physidae] and other pelagic invertebrates) and 10% crayfish (e.g., Cambaridae).

Largemouth Bass (*Micropterus salmoides*)
The piscivorous feeding guild is represented by largemouth bass. Collected fish ranged from 11 to 15 inches (Table 1), which is the size range that would be attractive to anglers and larger ecological receptors. In this size range, bass consume primarily smaller forage fish and benthic invertebrates, including crayfish. As shown in Table 7, largemouth bass from the site consume greater than 90% of their diet as crayfish.

Table 10: Dietary Preferences (% of Diet) for the Modeled Species

Species	Pelagic	Sediment	Benthos	Crayfish	PKSD	Notes
PKSD	80		20			BSAF probable and possible ranges = 1 to 1.7
BB	10	80		10		BSAF = 1
LMB				92	8	BSAF probable and possible ranges = 1 to 1.7
Kingfisher			12	10	78	46% -100%fish; 5% - 41% invertebrates
Heron					100	100% fish
Mink				16.5	34	18.8% - 34% fish; 13.9% - 16.5% inverts; large proportion of small mammals

Terrestrial Food Web
Fish and invertebrates that may have accumulated contaminants from sediments in Eighteenmile Creek also serve as a prey base for ecological receptors, including fish eating birds and mammals. Figure 6 presents a simplified conceptual model for terrestrial receptors that consume fish. The selected species were chosen based on the following criteria:

- Observed in the study area or could occur in the study area
- Fish consumers
- Life histories and foraging strategies that lead to potential exposures from Eighteenmile Creek
- Modeling parameters are readily available (e.g., knowledge of quantitative foraging preferences, etc.)

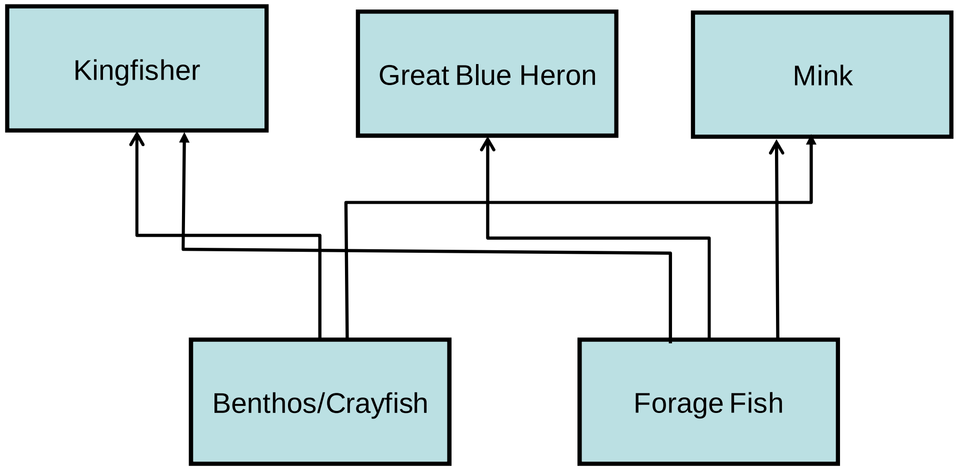
We evaluate potential exposures and hazards to two avian receptors known to inhabit Eighteenmile Creek: the belted kingfisher (*Ceryle alcyon*) and the great blue heron (*Ardea herodias*). One mammalian receptor is selected: the mink (*Mustela vison*). The primary reference for body weight, ingestion rate, and dietary preferences, the three key inputs, is obtained from the US EPA's Wildlife Exposure Factors Handbook and data provided therein (US EPA, 1993a and 1993b). A summary of dietary preferences is found in Table 10, and Table 11 provides a summary of all inputs to the *TrophicTrace* model.

Belted Kingfisher
The belted kingfisher is a medium-sized bird, measuring about 13 in (33 cm) (Peterson 1980). It is blue-gray with a ragged bushy crest and broad gray breastband. It generally feeds on fish that swim near the surface or in shallow water. The kingfisher may also feed on crayfish, and in times of food shortages it can feed on a variety of invertebrates and vertebrates. Kingfishers nest in burrows that they excavate in embankments. Kingfisher are found throughout the study area (Ecology and Environment, 2007, p 7-33,7-47; also as documented on the Atlas 2000 website for block 1980C; http://www.dec.ny.gov/cfmx/extapps/bba/). Body weight, ingestion rates, and dietary preferences for the kingfisher were obtained from the US EPA Wildlife Exposure Factors Handbook (US EPA, 1993a and 1993b) and references contained therein.

Great Blue Heron
The great blue heron is one of the largest wading birds found in upper New York State. It can stand over 4 ft high (average 42 to 52 in) with a wing span of 6 to 7 ft. It has a blue-gray color and adults are white about the head. Their long legs, necks, and bills are adapted for wading in the shallow water and stabbing prey. Fish are the preferred prey of great blue herons, but they also eat amphibians, reptiles, crustaceans, insects, birds, and mammals. Great blue heron have been observed throughout New York State, and have been observed in the study area (http://www.guides.nynhp.org/guide.php?id=6752&part=3; Ecology and Environment, 2007, p 7-47; Atlas 2000 website http://www.dec.ny.gov/cfmx/extapps/bba/). Body weight, ingestion rates, and dietary preferences for the heron were obtained from the US EPA Wildlife Exposure Factors Handbook (US EPA, 1993a and 1993b) and references contained therein.

Mink
The proposed mammalian receptor is the mink. The mink is a small carnivore that is widely distributed throughout New York State (http://nyfalls.com/wildlife/Wildlife-mammals-weasel-like.html) and is found throughout the study area as well (Ecology and Environment. 2007, p 7-47). Generally, mink are opportunistic in their feeding habits and prey varies according to seasonal abundance of prey and habitat. They feed on a variety of prey including fish, aquatic invertebrates, and small mammals. Their sensitivity to PCBs is well understood.

Figure 6: Conceptual Site Model for Eighteenmile Creek Terrestrial Foodweb



Further information on the feeding strategies and life histories of the selected species is provided in Appendix 10. The values used in the modeling are provided in Table 11.

3.4 Effects Assessment

This section provides a general overview of the toxicology of PCBs and describes the methods used to characterize particular toxicological effects of PCBs on aquatic and terrestrial organisms. TRVs are levels of exposure associated with either Lowest Observed Adverse Effects Levels (LOAELs) or No Observed Adverse Effects Levels (NOAELs). They provide a basis for judging the potential effects of measured or predicted exposures that are above or below these levels.

The toxicity of PCBs has been shown to manifest itself in many different ways, among various species of animals. Typical responses to PCB exposure in animals include wasting syndrome, hepatotoxicity, immunotoxicity, neurotoxicity, reproductive and developmental effects, gastrointestinal effects, respiratory effects, dermal toxicity, and mutagenic and carcinogenic effects. Some of these effects are manifested through endocrine disruption.

PCBs are typically present in the environment as complex mixtures. These mixtures consist of discrete PCB molecules that are individually referred to as PCB congeners. PCB congeners are often introduced into the environment as commercial mixtures known as Aroclors. PCB toxicity varies significantly among different congeners and is dependent on a number of factors. Two significant factors relate to the chemical structure of the PCB congener, including the degree of chlorination and the position of the chlorines on the biphenyl structure (Safe et al., 1985). In general, higher chlorine content typically results in higher toxicity, and PCB congeners that are chlorinated in the ortho position are typically less toxic than congeners chlorinated in the meta and para positions. Furthermore, metabolic activation is believed to be the major process contributing to PCB toxicity.

This ecological risk assessment focuses on effects that relate to the survival, growth, and reproduction of individuals within the local populations of fish and wildlife species. Reproductive effects are defined broadly herein to include egg maturation, spawning, egg hatchability, survival of fish larvae, and offspring survival.

Reproductive effects tend to be the most sensitive endpoint for animals exposed to PCBs. Indeed, toxicity studies in vertebrates indicate a relationship between PCB exposure, as demonstrated by aryl hydrocarbon hydroxylase (AHH) induction, and functions that are mediated by the endocrine system, such as reproductive success. A possible explanation for the relationship between AHH activity and reproductive success may be due to a potential interference from the P450-dependent MFO with the ability of this class of P450 proteins to regulate sex steroids. In fact, the induction of cytochrome P450 isozymes from PCB exposure has been shown to alter patterns of steroid metabolism (Spies et al., 1990). As another example, the maternal hepatic AHH activity of the flatfish, *Paralichthys stellatus*, at the time of spawning, was found to be inversely related to three reproductive functions: egg viability, fertilization success, and successful development from fertilization through hatching (Long and Buchman, 1990).

Historically, the most common approach for assessing the ecological impact of PCBs has involved estimating exposure and effects in terms of totals or Aroclor mixtures. It is important to note that, since different PCB congeners may be metabolized at different rates through various enzymatic mechanisms, when subjected to processes of environmental degradation and mixing, the identity of Aroclor mixtures is altered (McFarland and Clarke, 1989). Therefore, depending on the extent of breakdown, the environmental composition of PCBs may differ significantly from the original Aroclor mixture released into the environment. Furthermore, commercial Aroclor mixtures used in laboratory toxicity studies may not represent true environmental exposures. Thus, there are some unquantifiable uncertainties associated with estimating the ecological effects of PCBs using a combination of congener-based exposure data and Aroclor or "total PCB" based effects data (See Discussion in Section 4).

3.4.1 TRVs

TRVs can be developed on the basis of no-observed adverse effect levels (NOAELs) and lowest observed effect levels (LOAELs). These two alternative TRV values reflect the range of uncertainty that exists between the presence and absence of an adverse effect. If the HQ based on the NOAEL does not exceed a value of one, it is concluded that the chemical does not pose a hazard. If the HQ based on the LOAEL exceeds a value of one, it is expected that the chemical could pose a significant hazard. If the HQ based on the LOAEL is less than one but the HQ based on the NOAEL is greater than one, the chemical is probably close to a level that could cause adverse effects, but whether or not significant effects would actually occur cannot be judged with certainty. TRVs for the present risk assessments are developed on the basis of NOAELs and LOAELs to provide perspective on the range of potential effects relative to modeled exposures. TRVs used in this analysis are shown in Table 11.

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

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

- TRVs for terrestrial receptors (e.g., birds and mammals) are expressed as daily dietary doses (e.g., mg/kg whole body wt/d).

TRVs for Fish
No laboratory studies were identified that examined toxicity of PCBs to the selected fish species. A low and high NOAEL and LOAEL were selected to provide perspective on the range of potential toxicity quotients based on three studies. A study by Bengtsson (1980) on the minnow is selected as the lowest appropriate NOAEL for development of the TRV. In this study, fish were exposed to Clophen 50 (a commercial mixture with a chlorine content of 50%) in food for 45 days. Hatchability was significantly reduced in fish with an average total PCB concentration of 170 mg/kg (measured on day 171 of the experiment), but not in fish with an average concentration of 15 mg/kg. Because the experimental study measured the actual concentration in fish tissue, rather than estimating the dose on the basis of the concentration in external media (e.g., food, water, or sediment, or injected dose), a subchronic-to-chronic uncertainty factor is not applied. Because results of studies of dioxin-like PCBs on fish eggs have shown another species of minnow to be of intermediate sensitivity compared to all other fish species tested, an interspecies uncertainty factor of 10 is applied to the LOAEL (170 mg/kg) and NOAEL (15 mg/kg) from this study to develop TRVs for fish.

A second study (Hansen et al., 1974) that exposed sheepshead minnow to Aroclor 1254 for 28 days developed a NOAEL of 1.9 mg/kg wet weight and a LOAEL of 9.3 mg/kg wet weight based on significantly reduced fry hatchability. This study is used to develop TRVs for the current assessment.

A study by the USACE (1988) was also selected for TRV development. In that study, spawning and fecundity was significantly reduced in fathead minnow (*Pimephales promelas*) exposed in the laboratory to sediments from Sheboygan Harbor in Wisconsin (exact mixture of PCBs unknown but an environmental mixture rather than a pure Aroclor). Because the experimental study measured the actual concentration in fish tissue, rather than estimating the dose on the basis of the concentration in external media (e.g., food, water, or sediment), a subchronic-to-chronic uncertainty factor is not applied. Effects were observed at 13.7 mg/kg wet weight (LOAEL), but not at 5.3 mg/kg wet weight (NOAEL) for total PCBs.

Finally, we developed TRVs using several field studies by Adams et al. (1989, 1990 and 1992) who evaluated the redbreast sunfish (*Lepomis auritus*), a species in the same family as the pumpkinseed and largemouth bass. The NOAEL for growth was reported as being significantly different from a one downstream location, but no comparison to the reference sites was provided.

Growth is a relevant endpoint, and the NOAEL for growth, 0.3 mg/kg, and the LOAEL for growth, 0.4 mg/kg, was used in this assessment based on total PCBs. A subchronic-to-chronic uncertainty factor is not applied because the experimental study measured the actual concentration in fish tissue, rather than estimating the dose on the basis of the concentration in external media.

Avian Receptors

No laboratory studies were identified that examined the toxicity of PCBs in the diet of either the kingfisher or the great blue heron or a bird in the same order. Following the methodology established by The Great Lakes Water Quality Initiative to develop wildlife criteria among studies that are similar (e.g., in exposure duration, etc.), preference is given to laboratory studies with wildlife species (USEPA, 1995). A study by Dahlgren et al. (1972) on the ring-necked pheasant is selected for development of TRVs since this study reports TRVs for a wildlife species and provides both a NOAEL and a LOAEL. In this study, egg production was significantly reduced in birds fed a dietary dose of 7.1 mg PCBs/kg/day, but was not reduced at a dose of 1.8 mg/kg/day over the course of 16 weeks. Egg production by hens fed PCBs at the LOAEL was 32-97% that of control hens. Because gallinaceous birds, such as the pheasant, are among the most sensitive of avian species to the effects of PCBs, an interspecies uncertainty factor is not applied. Because a hatching period is a short-term event by nature, a subchronic-to-chronic uncertainty factor is not applied.

Chapman (2003) developed recommended avian TRVs on behalf of US EPA Region 5 and these are within the range of the values selected here.

Mink

The TRVs for mink are based primarily on a study by Aulerich and Ringer (1977) which has been used as the basis for TRV development at numerous sites and for many studies (Hope 1999; Blankenship et al., 2008; Chapman 2003). In this study, mink were exposed by diet to several doses of Aroclor 1254 (0, 1, 5, 15 mg/kg in feed) for up to 130 d and a dose of 2 mg/kg for up to 298 d through a critical reproductive life stage. Conversions of concentrations in the diet to a daily dose were based on a normalized ingestion rate of 0.15 kg/kg/d (based on assumptions of a food consumption rate of 0.15 kg/d and a body weight of 1.0 kg). No adverse effects were observed on the number of kits per female at a dose level of 1 mg/kg in feed (or 0.15 mg PCB/kg/d). At this dose, the number of kits per female was 4.3, which was not a statistically significant difference compared to three different sets of controls in which the number of kits ranged from 4.1 to 6.0. At a dose of 2 mg/kg of Aroclor 1254 in feed (or 0.3mg PCB/kg/d), adverse effects were observed including a reduction in the number of kits per female. However, when Aroclors 1221, 1242, and 1016 (41 percent chlorine) were tested at dietary concentrations of 2 mg/kg (or 0.3 mg PCB/kg/d), no effects were observed on reproduction. Thus, since Aroclor 1254 was found to be more toxic than other Aroclors tested, the NOAEL and LOAEL values of 1 and 2mg PCB/kg (or 0.15 and 0.3 mg PCB/kg/d) of Aroclor 1254 in feed should be considered a conservative estimate of the NOAEL and LOAEL, respectively (Blankenship et al., 2008), and these values are appropriate for Aroclor 1248, as well, as studies have shown that Aroclor 1248 is just as toxic as Aroclor 1254 in an *in vitro* bioassay (Chapman 2003 citing Tillitt et al., 1992). Since the study considered dietary exposure during the sensitive and ecologically relevant time period of reproduction, the 0.15 and 0.30 mg PCB/kg/d doses were considered to be chronic dietary-based NOAELs and LOAELs, respectively.

Chapman (2003) developed recommended mink TRVs on behalf of US EPA Region 5 based on a meta-analysis of several studies. He developed a recommended NOAEL of 0.5 mg/kg-d and a LOAEL of 0.6 mg/kg-d for environmental mixtures of PCB congeners based on total PCBs.

Table 11: Summary of Input Data to the *TrophicTrace* Model (dietary preference data found in Table 10)

Environmental	**Units**	**Possible Low**	**Probable Low**	**Probable High**	**Possible High**	**Reference(s)**
Total PCB Congeners Section 1	mg/kg dw	341	599	599	857	Data
TOC Section 1	%	1.23	1.61	1.61	1.98	Data
Dissolved Water Section 1	ng/L	80	80	80	80	Data
Total PCB Congeners Section 2	mg/kg dw	1971	2640	2640	3309	Data
TOC Section 2	%	3.79	4.51	4.51	5.23	Data
Dissolved Water Section 2	ng/L	100	100	100	100	Data
Temperature (both sections)	Deg C	15	18	20	22	Data
Log K_{ow}	L/kg	5.8	6	6.2	6.4	PCB Congeners
Benthic BSAF	unitless	1.0	1.0	1.7	1.7	USACE, 2008
Biota						
Benthic lipid	%	1.8	2.2	2.2	2.6	US EPA, 2000
Crayfish lipid	%	2.41	2.83	2.83	3.25	Lin et al. 2004; Gewurtz et al. 2000
Pelagic lipid	%	0.01	0.2	0.2	0.8	US EPA, 2000

PKSD lipid	%	2.0	2.4	2.4	2.8	Data
PKSD weight	gr	4.0	5.8	5.8	7.6	Data
PKSD TRV NOAEL	mg/kg ww	0.3	1.5	1.9	5.3	Adams et al. 1992; Bengtsson, 1980; Hansen et al. 1974; USACE, 1988
PKSD TRV LOAEL	mg/kg ww	0.4	9.3	13.7	17.0	Adams et al. 1992; Hansen et al. 1974; Bengtsson, 1980; USACE, 1988
BB lipid	%	2.4	3.1	3.1	3.7	Data
BB weight	gr	237	274	274	312	Data
BB TRV NOAEL	mg/kg ww	0.3	1.5	1.9	5.3	Adams et al. 1992; Bengtsson, 1980; Hansen et al. 1974; USACE, 1988
BB TRV LOAEL	mg/kg ww	0.4	9.3	13.7	17.0	Adams et al. 1992; Hansen et al. 1974; Bengtsson, 1980; USACE, 1988
LMB lipid	%	2.3	2.9	2.9	3.4	Data
LMB weight	Gr	515	601	601	686	Data
LMB TRV NOAEL	mg/kg ww	0.3	1.5	1.9	5.3	Adams et al. 1992; Bengtsson, 1980; Hansen et al. 1974; USACE, 1988
LMB TRV LOAEL	mg/kg ww	0.4	9.3	13.7	17.0	Adams et al. 1992; Hansen et al. 1974; Bengtsson, 1980; USACE, 1988
Kingfisher						
Weight	kg	0.14	0.15	0.15	0.16	US EPA 1993a; 1993b
Ingestion rate	kg/day	0.055	0.058	0.058	0.06	US EPA 1993a; 1993b
TRV (NOAEL)	mg/kg-d	0.6	0.6	1.8	1.8	Chapman 2003; Dahlgren et al. 1972
TRV(LOAEL)	mg/kg-d	1.2	1.2	7.1	7.1	Chapman, 2003; Dahlgren et al. 1972
Great Blue Heron						
Weight	kg	2.04	2.29	2.29	2.57	US EPA 1993a; 1993b
Ingestion rate	kg/day	0.28	0.35	0.35	0.43	US EPA 1993a; 1993b

TRV (NOAEL)	mg/kg-d	0.6	0.6	1.8	1.8	Chapman, 2003; Dahlgren et al. 1972
TRV (LOAEL)	mg/kg-d	1.2	1.2	7.1	7.1	Chapman, 2003; Dahlgren et al. 1972
Mink						
Weight	kg	0.55	0.83	1.02	1.36	US EPA 1993a; 1993b
Ingestion rate	kg/day	0.12	0.13	0.13	0.15	US EPA 1993a; 1993b
TRV (NOAEL)	mg/kg-d	0.04	0.15	0.15	0.5	Blankenship et al. 2008
TRV(LOAEL)	mg/kg-d	0.3	0.3	0.6	0.6	Hope, 1999; Blankenship et al. 2008; Chapman, 2003

3.5 Results and Ecological Risk Characterization

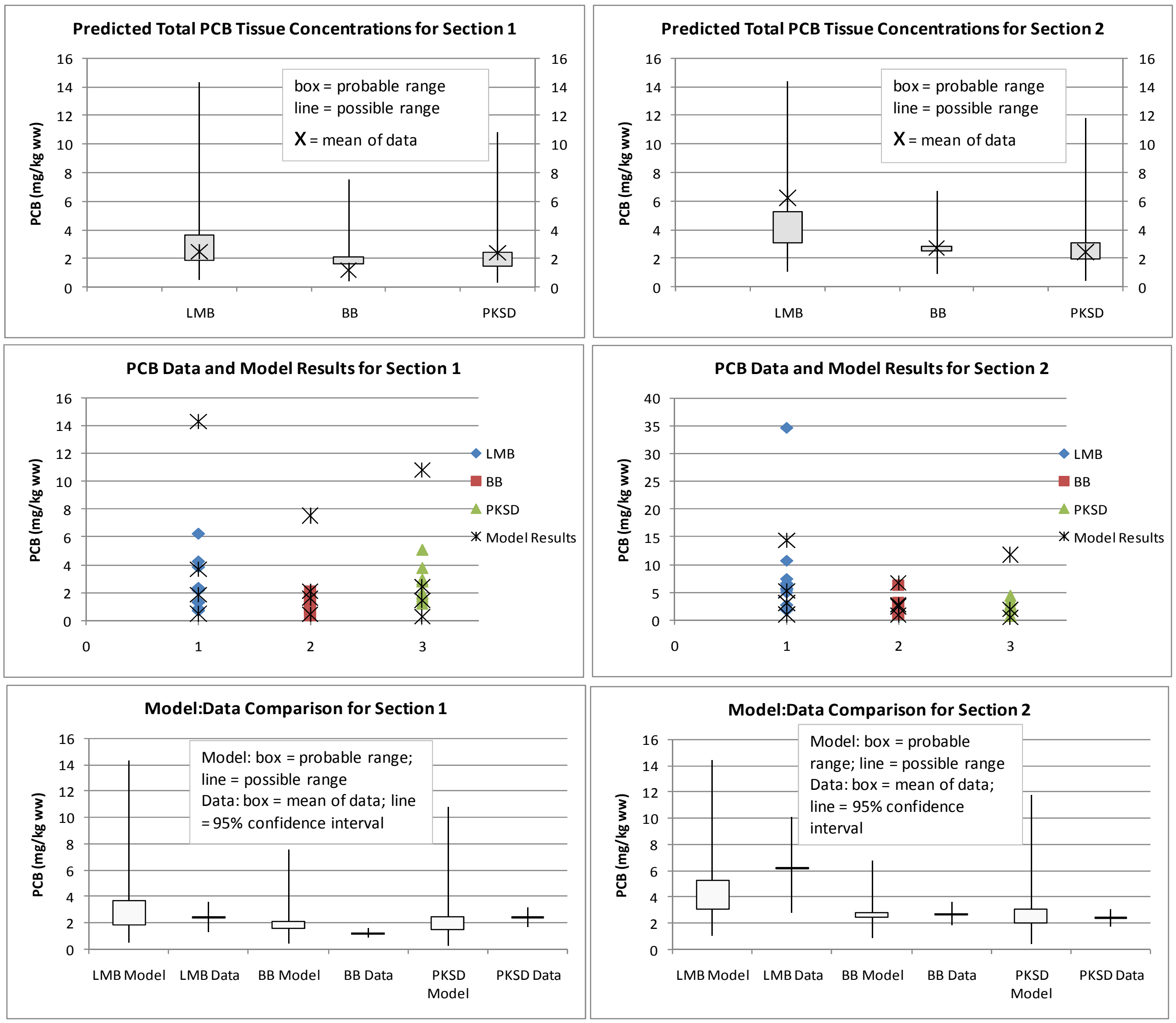
This section provides the results of the *TrophicTrace* modeling. The summary statistics for the contaminant concentrations of collected fish (Table 4) are used as the basis for comparing tissue concentrations predicted by the model. Below, we present the modeled tissue concentrations and comparisons to the empirical data. Finally, we present the predicted toxicity quotients resulting from comparisons of predicted doses (or body burdens) to receptors as compared to TRVs from the literature.

3.5.1 Predicted Tissue Concentrations

Figure 7 shows the results for the *TrophicTrace* bioaccumulation modeling using three different graphical formats for the same results. The top row of graphs shows the predicted tissue concentrations as compared to the mean of the data. The box represents the range of probable predicted concentrations; while the lines depict the range of possible (incorporating uncertainty) predicted concentrations. The possible and probable ranges are estimated by using ranges for input values. The second row of graphs shows the same results, but this time compared to individual data points. This graph shows that for Section 2, there is one LMB sample that is significantly higher than the remaining data set. The asterisks in this set of graphs represent the range of model output (the top and bottom asterisks represent the possible range, while the two center asterisks represent the probable range). Finally, the bottom row of graphs shows the model predictions as compared to the data using the same box format. In this set of graphs, the data are represented by the mean (center line) and the 95% confidence interval (lines).

These graphs show that the model predictions are well within the range of data, particularly for the most probable predicted range. As noted previously, the model predicts the expected average fish tissue contaminant concentration with associated uncertainty. Thus, the possible range reflects the uncertainty in the predicted mean, and is greater than the 95% confidence interval of the data itself. This is because the model reflects the underlying uncertainty in the model inputs rather than just the variability of the data.

Figure 7: Predicted Fish Tissue versus Observed Fish Tissue Concentrations. Top row: graphs show the predicted tissue concentration as compared to the mean of the data. The box represents the range of probable predicted concentrations, while the lines depict the range of possible predicted concentrations. The asterisk is the mean of the data. Second row: graphs show the same results, but individual data points are compared. Bottom row: model predictions are compared to empirical data using the same box format as the top row, but empirical fish data are represented by the mean (center line/box) and the 95% confidence interval of the mean (vertical lines).



3.5.2. Discussion on Model Performance and Implications

The *TrophicTrace* model was parameterized, run, and the results compared to the available data without going through an explicit calibration process. That is, we parameterized the model and compared the results to data, and having found an acceptable agreement between the predictions and data, we then used the predicted fish body burdens to estimate ecological TQs. Table 12 below shows the relative percent difference for the probable predicted range from the *TrophicTrace* model as compared to the mean value from the data. This table shows that for Section 1, *TrophicTrace* probable range (expected value) predicted body burdens were within less than 25% to greater than 1% of the data, a very close prediction. For brown bullhead, the model somewhat overpredicted observed tissue concentrations by between 15 and 27%, while for largemouth bass, the model predicted to within 13% less and 20% more than the mean value from the data.

In Section 2, the model predicted to within 10% less and 12% more than the data for pumpkinseed, while brown bullhead model performance was slightly better, predicting to within less than 5% and greater that 1% relative to the mean. The largemouth bass model somewhat underpredicted body burdens, but achieved a prediction within 8% of the mean value. Note that there is one largemouth bass sample that is considered, statistically, an outlier (although, of course, all data were included in the derivation of mean concentrations at the site). Overall, these results demonstrate excellent agreement with the data. In general, this is considered excellent performance for a food web model that required no explicit calibration to site data.

There are no known substantive differences in potential food web bioaccumulation dynamics between Sections 1 and 2; i.e., both areas are equally well predicted by the bioaccumulation model. This suggests that the environmental conditions that varied between sections (contaminant and TOC concentrations) accounted for the differences seen in fish tissue concentrations. Still, there are uncertainties associated with model parameterization (see Section 4). Reducing these uncertainties would provide confidence that model performance results from a refined relationship between sediments and fish, rather than an inspired selection of model parameters. Optimally, a further verification data set would be collected to verify the model's performance.

Table 12: Relative Percent Difference between Actual and Predicted Fish Tissue Concentrations

Relative Percent Difference	PKSD	BB	LMB
Section 1	-25% to 1%	15% to 27%	-13% to 20%
Section 2	-10% to 12%	-5% to 1%	-34% to -8%

3.5.3 Ecological Effects Risk Characterization

A toxicity quotient (TQ) is estimated as the ratio between the predicted contaminant concentration in an organism (for fish) and dose (for birds and mammals) and the literature-derived TRV. A predicted TQ is not an actual measure of risk, but simply a convenient method for indicating exceedance of a TRV (Hope 1999). TRVs can be based on NOAELs and/or

LOAELs. These alternative TRV values reflect the range of uncertainty that exists in the actual threshold between the presence and absence of an adverse effect. If the TQ based on the NOAEL does not exceed a value of one, it is concluded that the chemical does not pose a hazard (a conclusion of no significant risk [NSR]). If a TQ based on the LOAEL exceeds a value of one, it is expected that exposure could pose a hazard, that is, adverse effects cannot be ruled out. If the TQ based on the LOAEL is less than one but the TQ based on the NOAEL is greater than one, predicted exposures are probably close to levels that could cause adverse effects, but whether or not significant effects would actually occur cannot be judged with certainty. Therefore, the conclusion is that there is a low potential for risk.

The LOAEL-based comparison, by definition, reflects an exceedance of an effect level; therefore, potential risk of adverse effects is presumed to be directly proportional to the degree of exceedance of the LOAEL for the probable range with the following conclusions: no significant risk (NSR) for $TQ \leq 1$, low potential risk for $1 < TQ \leq 10$, moderate for $10 < TQ \leq 100$, and high for $TQ > 100$.

The probable range (box) of predicted TQs is used to determine the potential for adverse effects, while the possible range (line) is used to determine our confidence in that result. That is, a probable range less than one but a possible range greater than one indicates a lower degree of confidence in the results than if both the possible and probable ranges fall below one. TQs are estimated as the average expected TQ across the population -- for example, if we were to sample a random fish and measure the body burden, we would expect the result to fall in the probable range, but the result for any random fish could fall in the possible range as that reflects our uncertainty. Table 13 below provides the interpretation of the risk characterization results.

Table 13: Matrix of Interpretation of LOAEL- and NOAEL-based Exceedances for the Probable (Conclusion of Potential Risk) and Possible (Uncertainty) Ranges

		<<-- Confidence in Conclusion -->>		
Probable Range	**Conclusion**	**Possible range upper $N \leq 1$**	**Possible range upper $1 < N \leq 10$**	**Possible range upper $10 < N \leq 100$**
$N \leq 1$	No potential for exposures to exceed a NOAEL	High	Moderate	Low
$1 < N \leq 10$	Low potential for exposures to exceed a NOAEL	------	High	Moderate
$10 < N \leq 100$	Moderate potential for exposures to exceed a NOAEL	------	-------	High
$N > 100$	High potential for exposures to exceed a NOAEL	------	------	------

Probable Range	**Conclusion**	**Possible range upper $L \leq 1$**	**Possible range upper $1 < L \leq 10$**	**Possible range upper $10 < L \leq 100$**
$L \leq 1$	NSR	High	Moderate	Low
$L \leq 1; N > 1$	Low potential risk	High	Moderate	Low
$1 < L \leq 10$	Low potential risk	------	High	Moderate
$10 < L \leq 100$	Moderate potential risk	------	-------	High
$L > 100$	High potential risk	------	------	------

Figure 8 shows the results of the comparison of predicted daily doses and/or body burdens for the ecological receptors as compared to literature-derived TRVs. As for Figure 7, the box represents the probable range, while the lines represent the possible range incorporating uncertainty around the mean estimate (although the line does not, strictly speaking, represent a 95% confidence interval in statistical terms, it is analogous to that). The red line is the threshold value of one: results above this line indicate a potential for exposures (NOAEL-based comparison) and/or effects (LOAEL-based comparison). Table 13 presents these results in a tabular format.

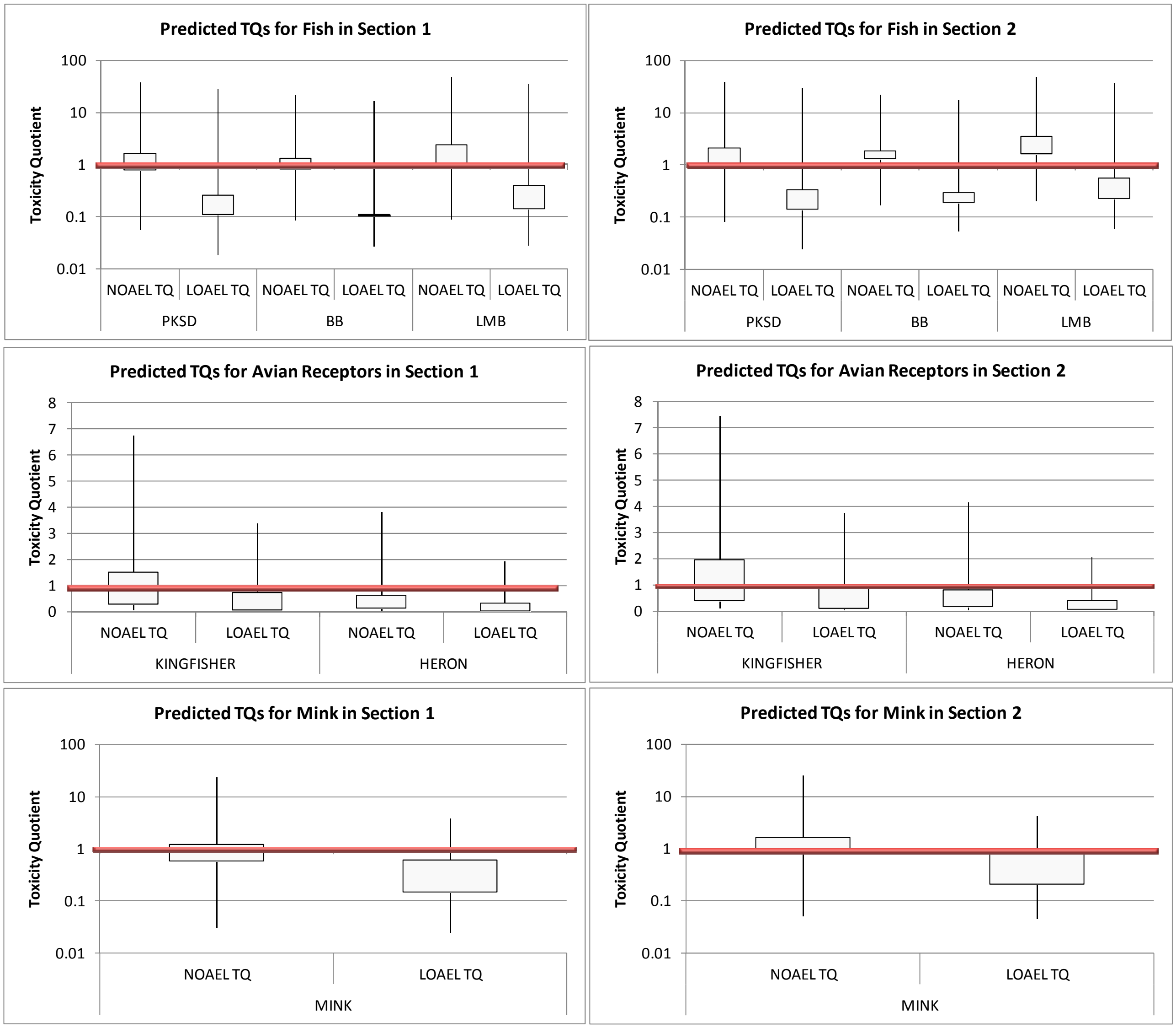
Results for Fish in Section 1

The probable range results for pumpkinseed, a forage fish that serves as a prey base for piscivorous fish, and largemouth bass, a top-level predator, show NOAEL-based comparisons between one and ten, but the LOAEL-based comparisons are all below one. Because the true effect level could theoretically occur at any exposure between the NOAEL and LOAEL, we conclude there is a low potential for risk. The possible upper bound for both the NOAEL- and LOAEL-based comparisons are greater than ten but less than 100, indicating moderate confidence in the results (i.e., there is moderate confidence in the conclusion of low potential for risk). For the brown bullhead, a dermersal fish, both the LOAEL- and NOAEL-based comparisons for the probable range fall below one, indicating no potential for exposures to exceed a no-effect level and a conclusion of no significant risk. However, since the possible upper bound for both is greater than ten, we have low confidence in the conclusion of no significant risk (e.g., the body burden of a randomly selected fish could easily exceed either a NOAEL- or a LOAEL, and the potential for adverse effects cannot be ruled out).

Results for Fish in Section 2

In Section 2, there is a low potential for risk for all fish species given that the NOAEL-based comparisons for the probable range exceed one, but the LOAEL-based comparisons do not (indicating that effects could occur within that spectrum). The possible upper bounds exceed ten across species and endpoints, indicating moderate confidence in the conclusion that there is a low potential for risk.

Figure 8: Predicted Toxicity Quotients at Eighteenmile Creek



Results for Avian Receptors in Section 1

Kingfisher in Section 1 show a low potential for risk, since the NOAEL-based comparison for the probable range slightly exceeds one but the LOAEL-based comparison does not. The possible upper bound for both the NOAEL- and LOAEL-based comparisons are greater than one but less than ten, indicating a high confidence in this conclusion (i.e., the potential for risk across the population is low given that even the upper bound is less than ten although greater than one).

For the heron, both the NOAEL- and LOAEL-based comparisons for the probable range fall below one, indicating no potential for exposures to exceed no-effect levels and a conclusion of no significant risk. The possible range comparisons for both endpoints are greater than one but less than ten, indicating moderate confidence in the conclusion (i.e., exceedances cannot be ruled out).

Results for Avian Receptors in Section 2

Kingfisher in Section 2 show a low potential for risk, since the NOAEL-based comparison for the probable range exceeds one but the LOAEL-based comparison does not. The possible upper bound for both the NOAEL- and LOAEL-based comparisons are greater than one but less than ten, indicating a high confidence in this conclusion (i.e., the potential for risk across the population is low given that even the upper bound is less than ten although greater than one).

Table 14: Results of the Ecological Risk Assessment

	Low Possible Predicted TQ		Low Probable Predicted TQ		High Probable Predicted TQ		High Possible Predicted TQ		Conclusion	Confidence
	<<---- Results for Section 1 ------>>									
	NOAEL	LOAEL	NOAEL	LOAEL	NOAEL	LOAEL	NOAEL	LOAEL		
PKSD	0.1	0.02	0.8	0.1	2	0.3	36	27	Low potential risk	Moderate
BB	0.1	0.03	0.8	0.1	1.3	0.1	21	16	Low potential risk	Moderate
LMB	0.1	0.03	0.99	0.1	2	0.4	48	36	Low potential risk	Moderate
Kingfisher	0.1	0.02	0.3	0.1	1.5	0.7	7	3	Low potential risk	Moderate
Heron	0.02	0.005	0.1	0.03	0.6	0.3	4	2	No potential for exposures to exceed NOAELs; NSR	Moderate
Mink	0.03	0.03	0.6	0.2	1.2	0.6	23	4	Low potential risk	Moderate to High
	<<---- Results for Section 2 ------>>									
	Low Possible Predicted TQ		Low Probable Predicted TQ		High Probable Predicted TQ		High Possible Predicted TQ		Conclusion	Confidence
	NOAEL	LOAEL	NOAEL	LOAEL	NOAEL	LOAEL	NOAEL	LOAEL		
PKSD	0.1	0.03	1.0	0.1	2	0.3	39	30	Low potential risk	Moderate
BB	0.2	0.1	1.3	0.2	2	0.3	22	17	Low potential risk	Moderate
LMB	0.2	0.1	2	0.2	4	0.6	48	36	Low potential risk	Moderate
Kingfisher	0.1	0.03	0.4	0.1	2	1.0	7	4	Low potential risk	High
Heron	0.03	0.01	0.2	0.04	0.8	0.4	4	2	No potential for exposures to exceed NOAELs; NSR	Moderate
Mink	0.05	0.04	0.8	0.2	2	0.8	26	4	Low potential risk	Moderate to High

For the heron, both the NOAEL- and LOAEL-based comparisons for the probable range fall below one, indicating no potential for exposures to exceed no-effect levels and a conclusion of no significant risk. The possible range comparisons for both endpoints are greater than one but less than ten, indicating moderate confidence in the conclusion (i.e., exceedances cannot be ruled out).

Results for Mink in Section 1

The results for mink in Section 1 indicate a low potential for risk, since the NOAEL-based comparison for the probable range exceeds one but the LOAEL-based comparison does not. The possible upper bound for the NOAEL-based comparison is greater than ten, indicating high confidence that the NOAEL-based comparison is likely greater than one. The possible upper bound for the LOAEL-based comparison is greater than one but less than ten, indicating moderate confidence in the conclusion of low potential risk.

Results for Mink in Section 2

The results for mink in Section 2 indicate a low potential for risk, since the NOAEL-based comparison for the probable range exceeds one but the LOAEL-based comparison does not. The possible upper bound for the NOAEL-based comparison is greater than ten, indicating high confidence that the NOAEL-based comparison is likely greater than one. The possible upper bound for the LOAEL-based comparison is greater than one but less than ten, indicating moderate confidence in the conclusion of low potential risk.

3.5.4 Discussion of Ecological Effects Results

For fish, the NOAEL- and LOAEL-based comparisons across the probable and possible ranges indicate that the body burdens of fish from either creek section could exceed a NOAEL or a LOAEL. Although the probable range for the LOAEL-based comparisons all fall below one, the NOAEL-based comparisons for pumpkinseed and largemouth bass are greater than one, and the possible ranges are not only greater than one, but greater than ten. Indeed, looking at the data collected for this study, we did observe one largemouth bass sample of approximately 35 mg/kg, twice the highest observed LOAEL for this species (17 mg/kg), and over two orders of magnitude higher than the lowest observed LOAEL (0.4 mg/kg), confirming that it is possible for a randomly collected fish to have a predicted LOAEL-based TQ greater than ten.

Avian receptors show less of a range in the results. Even the highest predicted TQs for the possible range are less than ten. The probable range only exceeds one for the kingfisher and is less than one for the heron. Taken together, we conclude there is a low potential for risk to kingfisher and a negligible potential for risk to the heron.

For mink, there is a wide range in NOAEL-based TRVs. When predicted doses are compared to these TRVs, the result is that for the most part, comparisons fall below one or are between one and ten, except for the highest possible NOAEL-based comparison, which is greater than ten across both Sections of the study area. We therefore conclude there is a low potential for risk to mink foraging across the study area. This conclusion also includes a consideration of the actual TRVs, which, as shown in Table 11, show an overlap between LOAELs and NOAELs. That is, the highest NOAEL (0.5 mg/kg-d) is higher than the lowest LOAEL of 0.3 mg/kg-d. Both studies are reasonable, well-conducted, and explored endpoints relevant to reproduction.

The modeling approach developed here is termed a "bottom-up" study. Another approach, termed "top-down," involves collecting field observations that may indicate impacts from exposures within the study area, and these studies could be used as additional lines of evidence with which to evaluate the conclusions of the modeling study. These include efforts such as wildlife studies, benthic invertebrate community studies and population analyses.

The 2008 bioaccumulation study on Eighteenmile Creek AOC sediments (USACE Buffalo District 2008) found that PCBs are accumulating in invertebrates. That report also indicated that several samples exceeded the probable effect level (PEL) in sediments developed by Environment Canada (2003) of 277 µg/kg Total PCBs. The current dataset indicate that sediment concentrations on a mean basis, including the lower 5% confidence limit on the mean, all exceed this value as well. Further, based on empirical data from laboratory bioaccumulation experiments on field collected sediments, USACE Buffalo District (2008) indicated that PCBs in surficial sediments throughout most of the AOC (Section 1 below Burt Dam) PCBs in surface sediments are "bioaccumulating in benthic invertebrates, and are likely to bioaccumulate in predator fish and higher trophic levels". The results of this study are generally in concert with that finding.

4.0 DISCUSSION OF UNCERTAINTY

The modeling presented here is based on the best available information and data collected up to this point. The model reflects our understanding of the relationship between sediment (and nominally water) contaminant exposure concentrations and fish body burdens on the basis of one sampling event. Consequently, uncertainties remain that are briefly described in this section.

4.1 Surface Water Dissolved-phase Contaminant Concentrations

There is only one dissolved phase water PCB concentration available within the study area to support the model's depiction of freely dissolved PCBs in surface water. Soluble contaminants are an important pathway for contaminant uptake and bioaccumulation in aquatic organisms. One option to estimate this parameter is to predict water concentrations from sediment concentrations based on equilibrium partitioning theory. Alternatively, synoptic water sampling can be conducted to empirically derive an average exposure concentration over a relevant timeframe. In our case, only a single emprical data point was available. This is suboptimal and creates uncertainty in the depiction of soluble contaminants and estimation of fish tissue contaminant concentrations.

4.2 Contaminant Concentrations at the Base of the Food Web

The stomach contents analyses conducted as part of this effort revealed that largemouth bass, in particular, consume a higher than anticipated amount of crayfish. Site-specific contaminant data do not exist for crayfish, so this part of the model utilized literature values for lipid content, and predicted crayfish concentrations were never verified by field data. Similarly, benthic and pelagic invertebrate inputs were all based on literature values. Such information could be relatively easily obtained from a collection and analysis effort.

4.3 Fish Foraging Strategies and Dietary Exposure

The two primary factors influencing contaminant uptake into fish tissue are exposure concentrations and fish foraging strategies. Fish body burdens reflect both what fish consume (which will vary seasonally according to prey and bioamass availability) and may vary spatially depending on their movements and exposure history. Fish integrate exposures over particular spatial and temporal scales, and these are assumed rather than known. For example, the model assumes essentially equal exposures within each section as reflected by the mean and 95% confidence limit on the mean used as inputs to the model. If there was evidence of preferential foraging within a section, these exposure estimates could be refined. This uncertainty is not readily informed without, for example, fish tracking studies. However, those studies are resource intensive and still may not provide information capable of refinng the understanding of bioaccumulation from contaminated sediments.

Similarly, dietary preferences of the fish also represent an uncertainty. Although stomach content analysis was conducted on the fish sampled during Fall 2010, that represents one snapshot in time and does not address seasonality in feeding and potential spatial differences in feeding. A more refined understanding of diet (and hence dietary exposures of contaminants) during the course of the year could be developed from additional sampling during the spring and summer periods. Such information could be relatively easily obtained from a collection and analysis effort.

4.4 Terrestrial Receptor Foraging Strategies and Dietary Exposures

All exposure parameters used for terrestrial receptors in this analysis (e.g., kingfisher, great blue heron, and mink body weight, ingestion rate, and dietary preferences) were obtained from the literature as referenced in the US EPA Wildlife Exposure Factors Handbook (US EPA 1993a; 1993b). Although the uncertainty in these estimates was captured to some extent through the use of ranges as inputs rather than deterministic values, there is still uncertainty as to the composition of these diets.

4.5 Toxicity Reference Values

There is significant uncertainty inherent in the TRVs that are used in the analysis to predict risk. This is due to several factors, including:

- *Use of laboratory versus field studies.* In general, TRVs developed from laboratory studies are used because the exposure of concern can be isolated in the species of concern. However, this introduces uncertainty given that environmental conditions and contaminant exposures differ significantly between the laboratory and field environments.
- *Use of Aroclor TRVs.* As mentioned above, PCB congener composition differed across commercial mixtures of aroclors. Then these aroclors were released into the environment and underwent environmental processes that differed from site to site. Consequently, PCB mixtures in the laboratory used to develop TRVs can vary greatly from those found in the environment.
- *Attribution of effects related to cooccuring contaminants (and/or other stressors) in field studies.* While field studies better represent exposure to environmentally weathered PCBs, there may or may not be cooccuring unmeasured contaminants responsible for observed effects. One way to reduce this uncertainty is to use field-collected sediments in a laboratory setting, although it is difficult to measure all possible contaminants and to develop statistically robust models based on large numbers of potential predictors.
- *Species differences in effects studies (e.g., pheasant, chicken, etc.).* Often, TRV studies are conducted on reference species (e.g., chicken, pheasant) or TRVs are based on field studies that of necessity focus on the key receptors at those locations (e.g., falcon) that may or may not adequately represent species at other areas.

TRVs can reflect no- observed adverse effect levels or lowest-observed adverse effect levels, and both are used in this report. However, it is possible, particularly when values are taken from across the published literature and are not from the same study, for NOAELs to exceed LOAELs, and this is the case here. This reflects the very real and true uncertainty in those values -- a NOAEL from one study could exceed a LOAEL from another study, yet both are from well-designed studies and reflect endpoints relevant to the population.

5.0 REFERENCES

Adams SM, Crumby WD, Greeley Jr MS, Ryon MG and EM Schilling. 1992. Relationships between physiological and fish population responses in a contaminated stream. *Env Tox Chem* 11:1549-1557.

Adams SM, Shepard KL, Greeley Jr MS, Jimenez BD, Ryon MG, Shugart LR and JF McCarthy. 1989. The use of bioindicators for assessing the effects of pollutant stress on fish. *Marine Env Res* 28:459-464.

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

Aulerich RJ and R Ringer. 1977. Current status of PCB toxicity in mink and effect on their reproduction. *Arch. Environ. Contam. Toxicol.* 6:279-292.

Bardossy A and Duckstein L. 1995. Fuzzy rule-based modeling in geophysical, economic, biological, and engineering systems. Boca Raton, FL: CRC Press.

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

Blankenship AL, Kaya DP, Zwiernik MJ, Holem RR, Newsted JL, Hecker M and JP Giesy. 2008. Toxicity reference values for mink exposed to 2,3,7,8-tetrachlodibenzo-p-dioxin (TCDD) equivalents (TEQs). *Ecotox Env Saf* 69:325–349.

Chapman, J. 2003. EPA Region 5 recommended mink and avian toxicity reference values. Memorandum and Appendix D. March 6, 2003.

CH2MHill. 2011. Data Summary Report. Site Characterization. Eighteenmile Creek Area of Concern. Niagara County, New York. March 2011.

Dahlgren RB, Linder RL and Carlson CW. 1972. Polychlorinated biphenyls: Their effects on penned pheasants. *Env Health Perspect* 1:89-101.

Dou C, Woldt W, Bogardi I and Dahab M. 1995. Steady-state groundwater flow simulation with imprecise parameters. *Water Resources Research* 31(11):2709-2719.

Ecology and Environment, Inc. 2007. Eighteenmile Creek State of the Basin Report.

Ecology and Environment, Inc. 2008. Draft Beneficial Use Impairment Investigation for Eighteenmile Creek, Niagara County, New York.

Eddy S and Surber T. 1947. Northern Fishes: With Special Reference to the Upper Mississippi Valley. The University of Minnesota Press. Minneapolis, MN.

Environment Canada. 2003. Canadian Environmental Quality Guidelines. http://www.ccme.ca/assets/pdf/e1_062.pdf

Garabedian B, Estabrooks F, Swart J, Bopp RF. 2001. Final Report, Eighteenmile Creek Sediment Study: Summary of August 17-20 and November 3, 1998 Results. NYSDEC, Division of Water, December 2001.

Gewurtz SB, Lazar R and Haffner D. 2000. Comparison of polycyclic aromatic hydrocarbon and polychlorinated biphenyl dynamics in benthic invertebrates of Lake Erie, USA. *Env Tox Chem* 19(12):2943–2950.

Gobas F. 1993. A model for predicting the bioaccumulation of hydrophobic organic chemicals in aquatic food-webs: Application to Lake Ontario. *Ecological Modeling* 69(1-2):1-17.

Gustavson KE, von Stackelberg T, Linkov I, Bridges, T. 2011. *Bioaccumulation models: State of the application at large Superfund sites*. DOER Technical Notes Collection. ERDC TN-DOER-R17. Vicksburg, MS: U.S. Army Engineer Research and Development Center.

Guyonnet D, Côme B, Perrochet P and Parriaux A. 1999. Comparing two methods for addressing uncertainty in risk assessments. *Journal of Environmental Engineering* 7:660-666.

Hammonds JS, Hoffman FO and Bartell SM. 1994. An introductory guide to uncertainty analysis in environmental and health risk assessment. Environmental Restoration Program, Oak Ridge National Laboratory. ES/ER/TM-35/RI.

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

Hope BK. 1999. Assessment of risk to terrestrial receptors using uncertain analysis—A case Study. *Hum. Ecol. Risk Assess.* 5(1):145-170.

Huang GH, Chen Z, Tontiwachwuthikul P, Chakma A. 1999. Environmental risk assessment for underground storage tanks through an interval parameter fuzzy relation analysis approach. *Energy Sources*, 21(1-2):75-96.

Karn R, Escalon L, Lotufo G. 2004. Sediment Sampling, Biological Analyses, and Chemical Analyses for Eighteenmile Creek AOC, Olcott, New York. Vol I-II. US Army Corps of Engineers, Engineer Research and Development Center, Environmental Laboratory, March.

Lee YW, Dahab MF, Bogardi I. 1994. Nitrate risk assessment using fuzzy set approach. *Journal of Environmental Engineering* 121(3):245-256.

Lin H-I, Berzins DW, Myers L, George WJ, Abdfelghani A and Watanabe KH. 2000. Bayesian approach to parameter estimation for a crayfish (Procambarus Spp.) bioaccumulation model. *Env Tox Chem* 23(9):2259-2266.

Long ER and Buchman MF. 1990. A comparative evaluation of selected measures of biological effects of exposure of marine organisms to toxic chemicals. In: Biomarkers of Environmental Contamination. Pp 355-394. J.F. McCarthy, and L.R. Shugart, Editros. Lewis Publishers. Boca Raton, FL.

McFarland VA and Clarke JU. 1989. Environmental occurrence, abundance and potential toxicity of polychlorinated biphenyl congeners: considerations for a congener-specific analysis. *Environ Health Perspect* 81:225-239.

Mittelbach GG. 1984. Predation and resource partitioning in two sunfishes (Centrarchidae). *Ecology* 65(2):499-513.

Niagara County Soil and Water Conservation District. 2007. Eighteenmile Creek Remedial Action Plan (RAP) 2006 Status Report. March 2007. Available at: http://www.eighteenmilerap.com/2006%20Status%20Report%20%28Final%29.pdf.

New York State Department of Environmental Conservation (NYSDEC). 1997. Eighteenmile Creek Remedial Action Plan. August 1997.

NYSDOH. 2009. Chemicals in Sportfish and Game: 2009-2010 Health Advisories. http://www.health.state.ny.us/environmental/outdoors/fish/docs/fish.pdf

Peterson, R.T. 1980. A Field Guide to the Birds East of the Rockies. Fourth Edition. Houghton Mifflin Company. Boston, MA. 384 pp.

Safe S, Bandiera S, Sawyer T, Robertson L, Safe L, Parkinson A, Thomas PE, Ryan DE, Reik LM, Levin W, Denomme MA and Fujita T. 1985. PCBs: structure – function relationships and mechanisms of action. *Env Health Perspect* 60: 47-56.

Spies RB, Stegeman JJ, Rice DW, Woodlin B, Thomas P, Hose JE, Cross JN and Prieto M. 1990. Sublethal responses of *Platichthys stellatus* to organic contamination in San Francisco Bay with emphasis on reproduction in biomarkers of environmental contamination. pp. 87-121. J.F. McCarthy and L.R. Shugart, Editors. Lewis Publishers. Boca Raton, FL.

Tillitt D, Ankley G, Giesy J, Ludwig J, Kurita-Matsuba H, Weseloh D, Ross P, Bishop C, Sileo L, Stromborg K, Larson J and Kubiak T. 1992. Polychlorinated biphenyl residues and egg mortality in doublecrested cormorants from the Great Lakes. *Environ Toxicol Chem* 11:1281-1288.

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

US Army Corps of Engineers, Buffalo District. 2008. Concentrations, Bioaccumulation and Bioavailability of Contaminants in Surface Sediment. Eighteenmile Creek, Great Lakes Area of Concern (AOC), Niagara County, New York, June.

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

United States Environmental Protection Agency (USEPA). 2000. Phase 2 Report: Further Site Characterization and Analysis, Volume 2D – Revised Baseline Modeling Report, Hudson River PCBs Reassessment RI/FS. Prepared by TAMS Consultants, Inc., Limno-Tech, Inc., Menzie-Cura, Inc. and Tetra Tech, Inc. Available from www.epa.gov/hudson click on "reassessment reports."

United States Environmental Protection Agency (USEPA). 1993a. Wildlife Exposure Factors Handbook. Volume I of II. USEPA Office of Research and Development. Washington, D.C. EPA/600/R-93/187a.

United States Environmental Protection Agency (USEPA). 1993b. Wildlife Exposure Factors Handbook. Volume II of II. USEPA Office of Research and Development. Washington, D.C. EPA/600/R-93/187b.

United States Geological Survey (USGS). 1971-2008. Online electronic data. URLs (accessed 22 Jan 2010):
http://waterdata.usgs.gov/nwis/inventory?agency_code=USGS&site_no=04219767
http://waterdata.usgs.gov/nwis/inventory?agency_code=USGS&site_no=04219765

Zadeh LA. 1965. Fuzzy Sets. *Inform. and Control*, 8:338-353.

Zevin. P. 2011. Field Data Report Lake Ontario Tributaries 2009-2010. Dated March 28, 2011.

Zimmermann HJ. 1991. Fuzzy Set Theory and its Application. Boston: 2nd Edition, Kluwer Academic Publisher.