# An Evaluation of Human Health Risks from Contaminants in Presque Isle Bay

Erie, Pennsylvania

Prepared for: Pennsylvania Department of Environmental Protection Meadville, PA

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# TABLE OF CONTENTS

EX	ECU	FIVE SUMMARY	1
1.	INT	TRODUCTION	5
1	.1.	Overview of Risk Assessment	5
1	.2.	Site Location and History	7
1	.3.	Recreational Uses of the Bay	9
1	.4.	Fish Consumption Advisories	0
2.	СН	EMICALS OF POTENTIAL CONCERN 1	.1
2	.1.	Sampling Data	2
2	.2.	Selection of COPCs 1	6
3.	EX	POSURE ASSESSMENT	1
3	.1.	Conceptual Site Model	1
3	.2.	Exposure Pathways 2	1
3	.3.	Potential Exposure Groups	4
3	.4.	Calculation of Exposure Point Concentrations	1
3	.5.	Quantification of Exposure	2
4.	TO	XICITY ASSESSMENT	4
4	.1.	Oral and Dermal CSFs	4
4	.2.	Oral and Dermal RfDs for Non-carcinogenic Effects	5
4	.3.	Evaluation of Potential Exposure to Lead	5

5.	RIS	K CHARACTERIZATION	37
	5.1.	Direct Contact with Contaminated Sediment	37
	5.2.	Ingestion of Contaminants in Fish Tissue	39
6.	UN	CERTAINTY ANALYSIS	41
	6.1.	Hazard Identification	41
	6.2.	Exposure Assessment	43
	6.3.	Toxicity Assessment	48
	6.4.	Risk Characterization	49
7.	CO	NCLUSIONS	49
	7.1.	Risks Associated with Direct Contact with Contaminated Sediments	50
	7.2.	Consumption of Contaminated Fish	50
	7.3.	Main Conclusions	51
8.	RE	FERENCES	53

## LIST OF TABLES

MAIN REPORT	
Table 1: Summary of Exposure Groups and Pathways Evaluated in this HHRA	2
Table 2: Fish Consumption Advisories for Areas within the Lake Erie Basin <sup>(1)</sup>	11
Table 3: Inorganic and Organic Analytes Measured in Sediments	13
Table 4: Summary of Fish Species, Sampling Information and Residence Time	14
Table 5: Inorganic and Organic Analytes Measured in Fish Tissue <sup>(1)</sup>	16
Table 6: COPCs with Maximum Values Exceeding the Residential Soil RSLs	18
Table 7: COPC Summary of Contaminants with Maximum Values Greater than the Applicable Screening Level.	
Table 8: Potential Exposure Pathways for Presque Isle Bay	
Table 9: Exposure Point Concentrations for COPCs in Sediment	
Table 10: Summary of Noncancer and Cancer Risks for Direct Contact with Sediments for Presque Isle Bay Recreational Users	Water 38
Table 11: Summary of Noncancer and Cancer Risks for Ingestion of Fish	40
APPENDIX	

#### APPENDIX

Table 2-1	Sediment Sampling Data and COPC Selection Summary
Table 2-2	Sediment Constituents/Parameters Not Included in Current HHRA
Table 2-3	Bluegill Tissue Sampling Data and COPC Selection Summary
Table 2-4	Brown Bullhead Tissue Sampling Data and COPC Selection Summary
Table 2-5	Burbot Tissue Sampling Data and COPC Selection Summary
Table 2-6	Channel Catfish Tissue Sampling Data and COPC Selection Summary

- Table 2-7
   Common Carp Tissue Sampling Data and COPC Selection Summary
- **Table 2-8** Lake Trout Tissue Sampling Data and COPC Selection Summary
- Table 2-9
   Largemouth Bass Tissue Sampling Data and COPC Selection Summary
- **Table 2-10** Northern Pike Tissue Sampling Data and COPC Selection Summary
- Table 2-11
   Pumpkinseed Fish Tissue Sampling Data and COPC Selection Summary
- Table 2-12
   Smallmouth Bass Tissue Sampling Data and COPC Selection Summary
- Table 2-13
   Walleye Tissue Sampling Data and COPC Selection Summary
- **Table 2-14** White Bass Tissue Sampling Data and COPC Selection Summary
- Table 2-15
   White Sucker Tissue Sampling Data and COPC Selection Summary
- **Table 2-16** Yellow Perch Tissue Sampling Data and COPC Selection Summary
- Table 2-17
   Additional Constituents/Parameters Measured in Channel Catfish Tissue Excluded from HHRA
- **Table 2-18** Summary of Chemicals that are Additive in Nature
- Table 2-19
   Summary of Chemical Constituents in Fish Tissue with Federal/State Fish Consumption Advisory Levels
- Table 3-1
   Parameters and Values Included in Exposure Calculations Dermal Contact with Contaminated Sediment Adult Recreational Water Users
- Table 3-2
   Parameters and Values Included in Exposure Calculations Incidental Ingestion of Contaminated Sediment- Adult Recreational Water Users
- Table 3-3
   Parameters and Values Included in Exposure Calculations Dermal Contact with Contaminated Sediment Child Recreational Water Users
- Table 3-4
   Parameters and Values Included in Exposure Calculations Incidental Ingestion of Contaminated Sediment- Child

   Recreational Water Users
   Parameters
- Table 3-5
   Parameters and Values Included in Exposure Calculations Fish consumption Adult Anglers
- Table 3-6
   Parameters and Values Included in Exposure Calculations Fish consumption Children of Adult Anglers
- **Table 3-7**Exposure Point Concentrations (µg/kg) by Fish Species
- Table 4-1
   Summary of Cancer-Related Toxicity Values for COPCs in Sediment

- Table 4-2
   Summary of Noncancer Toxicity Values for COPCs in Sediment
- Table 4-3
   Summary of Noncancer and Cancer Toxicity Values for COPCs in Fish Tissue
- **Table 4-4** Assumptions and Results of the Adult Lead Model
- Table 4-5
   Summary of Assumptions and Results Used for the Child IEUBK Lead Model
- Table 5-1
   Summary of Cancer and Noncancer Risks by COPC and Exposure Route Adult Recreational Water Users RME
- Table 5-2
   Summary of Cancer and Noncancer Risks by COPC and Exposure Route Adult Recreational Water Users CTE
- Table 5-3
   Summary of Cancer and Noncancer Risks by COPC and Exposure Route Child Recreational Water Users RME
- Table 5-4
   Summary of Cancer and Noncancer Risks by COPC and Exposure Route Child Recreational Water Users CTE
- **Table 6-1** Summary of Qualitative Uncertainty Analysis
- Table 7-1
   Comparison of Contaminants in Fish Tissue Samples from Presque Isle Bay to other Sampling Locations within Lake Erie

# LIST OF FIGURES

## MAIN REPORT

Figure 1: Overview of the Four-Stage Risk Assessment Process Followed for this HHRA
Figure 2: Aerial photo of Presque Isle Bay with Area of Concern Boundary7
Figure 3: Map of Presque Isle Bay showing boat launch and marina locations

## APPENDIX

Figure 2-1:	Summary of Grab Sediment Sampling Locations in Presque Isle Bay
Figure 2-2:	Summary of Fish Sampling Locations in Presque Isle Bay and Lake Erie
Figure 3-1:	Conceptual Site Model - Potential Exposure Pathways

# List of Acronyms

ABS	Absorption Efficiency Factor
ALM	Adult Lead Model
AOC	Area of Concern
ATSDR	Agency for Toxic Substances and Disease Registry
BCF	Bioconcentration Factor
BHC	Hexachlorocvclohexane
Cal EPA	California Environmental Protection Agency
CDC	Centers for Disease Control
COPC	Chemicals of Potential Concern
CSF	Cancer Slope Factor
CSM	Conceptual Site Model
CTE	Central Tendency Exposure
DDD	Dichlorodiphenyldichloroethane
DDE	Dichlorodiphenyldichloroethylene
DDT	Dichlorodiphenyltrichloroethane
ELCR	Excess Lifetime Cancer Risk
EPC	Exposure Point Concentration
FDA	Food and Drug Administration
HEAST	Health Effects Assessment Summary Tables
HHRA	Human Health Risk Assessment
HI	Hazard Indices
HQ	Hazard Quotient
IEUBK	Integrated Exposure Uptake Biokinetic
IRIS	Integrated Risk Information System
NCEA	National Center for Exposure Assessment
PADEP	Pennsylvania Department of Environmental Protection
PAHs	Polycyclic Aromatic Hydrocarbons
PbB	Blood Lead Concentration
PCBs	Polychlorinated Biphenyls
PIB	Presque Isle Bay
PIBPAC	Presque Isle Bay Public Advisory Committee
PPRTV	Provisional Peer Reviewed Toxicity Values
RfD	Reference Dose
RME	Reasonable Maximum Exposure
RSL	Recommended Screening Level
95% UCL	95 <sup>th</sup> Percent Upper Confidence Limit
USEPA	United State Environmental Protection Agency
VOCs	Volatile Organic Compounds

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#### **EXECUTIVE SUMMARY**

## **ES.1** Overview

This report summarizes the results of a human health risk assessment (HHRA) for Presque Isle Bay in Erie, Pennsylvania. This is a companion document to the Ecological Risk Assessment of Presque Isle Bay prepared by Limnotech, Incorporated. (Limnotech 2011). The purpose of this HHRA was to develop estimates of current human health risks due to contact with contaminated sediments and from fish consumption utilizing existing datasets (i.e., sediment sampling database and fish species collected for fish advisory program). Both noncarcinogenic (i.e., liver, developmental and kidney toxicity) and carcinogenic (probability of developing cancer over a lifetime) risks were then compared to guidelines developed for the U.S. Environmental Protection Agency (EPA) Superfund program. These risk estimates were developed to support the policy and decision-making process.

The overall objectives of this risk assessment include to:

- compare the levels of contaminants in sediment and fish tissue to screening levels established by EPA in order to determine which contaminants should be included in the risk estimate process;
- estimate the current (or baseline) human health risks associated direct contact with Presque Isle Bay sediments and consumption of fish; and
- determine which exposure pathways and contaminants contribute most to human health risks.

The datasets utilized for this HHRA included sampling data collected between 2004 and 2010 for selected metals, polycyclic aromatic hydrocarbons (PAHs), polychlorinated biphenyls (PCBs), and pesticides in sediment and fish tissue. Fish tissue data utilized in this risk assessment included species sampled from Presque Isle Bay as well as Lake Erie. It should be noted that both the sediment and fish tissue sampling data were collected for purposes other than this human health risk assessment which may increase the uncertainty of the risk calculations.

Only a limited subset of the chemical constituents were retained and used to develop chemicalspecific risk estimates. As recommended by EPA, chemical constituents were screened from inclusion in the risk assessment using established risk-based screening levels (USEPA 2011a, 2011b). For the sediment data, this resulted in a total of nine contaminants being included within the risk assessment. For fish tissue sampling results, the number of chemicals retained varied between zero for pumpkinseed and bluegill species to 16 for lake trout.

To minimize the likelihood of underestimating risks, conservative, health-protective assumptions were incorporated into the identification of exposure scenarios, the estimates of exposure, and

the use of toxicity values. These are reflected in the reasonable maximum exposure (RME) scenarios presented in the report which represent high-end exposures that are likely to occur. This risk assessment also includes central tendency exposure (CTE) estimates which correspond to average exposures experienced by affected populations. Table 1 summarizes the exposure groups and pathways that were evaluated in this HHRA based on the most likely and significant exposures and data availability.

	In-wate	Fish Tissue	
	Dermal contact	Incidental ingestion	Ingestion
Adult recreational water user	•	•	
Child recreational water user	•	•	
Adult recreational angler			•
Adult urban/subsistence angler			•
Children of recreational angler			•
Children of urban/subsistence angler			•

Table 1: Summary of Exposure Groups and Pathways Evaluated in this HHRA

Potential cancer and noncancer risks were calculated for each chemical retained in the risk assessment for the above exposure scenarios. Noncancer effects were evaluated by calculating the hazard quotient (HQ) which represents the estimated exposure level divided by the reference dose (RfD). An HQ less than 1 indicates that exposures are not likely to be associated with adverse noncarcinogenic health effects while values above 1 may be of concern. HQs were summed across exposure pathways and chemicals to develop summary hazard indices (HIs). These are interpreted in a similar manner to the HQs.

For cancer risks, the endpoint is the ELCR or excess lifetime cancer risk, representing the probability of developing cancer over a lifetime due to exposure to a carcinogen. These values are calculated as the product of the lifetime exposure level to a chemical and its established cancer slope factor (CSF). Carcinogenic effects were summed across exposure pathways and across multiple chemicals. Estimated total cancer risks (summed across all chemicals) were

compared to a  $10^{-4}$  to  $10^{-6}$  (1 in 10,000 to 1 in 1 million) risk range representing the target range required by EPA as part of the Superfund program (USEPA 1991a). Cancer risks in the  $10^{-5}$  (1 in 100,000) range and higher are generally considered to be of concern

## ES. 2 Summary of Results

- 1. Overall, these results show that the main exposure route for contaminants in Presque Isle Bay is through fish consumption. These risks were several orders of magnitude greater than those associated with direct contact with contaminated sediments.
- 2. The cancer and noncancer risk estimates generated from consumption of fish tissue were highly dependent on the fish species and location (refer to Table 11). Based on the dataset utilized in this HHRA, several species from Lake Erie contributed to higher risks compared to species from Presque Isle Bay. These findings include:
  - Lake trout and smallmouth bass represented the fish species with the highest cancer and noncancer risk estimates. These species are likely to have a higher residence time and thus represent exposures to chemical constituents that occurred mainly from open water areas of the lake.
    - The summative noncancer risk for lake trout was approximately 3 for the typical or CTE estimate and 184 for the high-end or RME estimate (target level = 1.0). This latter value indicates that the estimated exposure to this chemical from consuming fish is 184 times greater than the level recommended by the EPA.
    - The summative cancer risk for lake trout was  $5 \times 10^{-5}$  (5 in 100,000) and  $5 \times 10^{-3}$  (5 in 1,000) for the CTE and RME estimates respectively (target level < 1 in 100,000).
    - The contaminant with the highest contribution to the noncancer and cancer risk estimates for lake trout and smallmouth bass was Arochlor 1254 or 1260.

(It should be noted that the cancer and noncancer risk estimates include the assumption of a single species diet and that all fish consumed originates from Lake Erie. These assumptions are conservative in nature and likely to overestimate the cancer and noncancer risks from consumption of fish. It should also be considered that these risk estimates are based on a limited sampling of fish tissue.)

- **3.** Values for certain fish species from Presque Isle Bay were also greater than the applicable cancer and noncancer risk thresholds. These results include:
  - Common carp and largemouth bass were the species with the highest associated risks.
    - The summative noncancer risk for common carp was 6 for the typical or CTE estimate and 48 for the high-end or RME estimate.

- The summative cancer risk for lake trout was  $3 \times 10^{-5}$  (3 in 100,000) and  $8 \times 10^{-4}$  (8 in 10,000) for the CTE and RME estimates respectively (target level = 1 in 100,000).
- The contaminant with the highest contribution to the noncancer and cancer risk estimates for common carp was Arochlor 1254.
- The concentration of chemical constituents measured in panfish, including bluegill and pumpkinseed species, were all below the applicable fish tissue screening levels. Therefore, risk estimates were not calculated for these species.

(It should be noted that the cancer and noncancer risk estimates include the assumption of a single species diet and that all fish consumed originates from Lake Erie. These assumptions are conservative in nature and likely to overestimate the cancer and noncancer risks from consumption of fish. It should also be considered that these risk estimates are based on a limited sampling of fish tissue.)

- 4. The cancer and noncancer risk estimates for direct contact with contaminated sediments from Presque Isle Bay were generally below the target risk levels for all exposure groups evaluated in this HHRA. All chemical-specific and cumulative excess lifetime cancer risk estimates were below  $1 \times 10^{-5}$  (1 in 100,000) and all chemical-specific and cumulative hazard indices were below 1.0. The exception to this was the RME cancer risk estimate for child recreational water users which was  $4 \times 10^{-5}$  (4 in 100,000). This value is mainly driven by dermal exposure from total PCBs. It should be noted that these risk estimates are conservative in nature and likely to overestimate the risk (the uncertainties associated with these estimates are discussed in section 6 of this report).
- **5.** The uncertainties associated with this risk assessment should be considered in utilizing the results for risk management decisions. A summary of the uncertainties inherent to this HHRA are discussed in section 6 of this report. The major uncertainties noted include the:
  - small dataset from which the risk estimates were drawn (i.e., data for certain fish species included one composite sample of five individual fish);
  - lack of specific data for the environmental media to which exposure groups are more likely to contact (i.e., for children beach sediment is a more likely exposure media compared to in-water sediment on which the risk estimates are based); and
  - lack of site-specific information on fish consumption patterns within the study area.

## 1. INTRODUCTION

#### 1.1. Overview of Risk Assessment

This HHRA has been prepared in support of the investigation to address potential human health risks associated with contaminated environmental media and fish consumption in Presque Isle Bay. This HHRA presents the potential for current cancer risks and noncancer health hazards to people who may be exposed to contaminants. The overall goals of this assessment are to:

- compare the levels of chemicals of potential concern (COPCs) in sediment and fish tissue to EPA screening levels in order to determine which constituents should be carried through the full risk assessment process;
- compare the estimated human health risks from consuming fish from Presque Isle Bay with those from Lake Erie;
- determine which exposure pathways lead to the highest human health risks; and
- quantify the current (or baseline) human health risk associated with the COPCs using existing dataset.

Potential human health risks were characterized based on COPC concentrations detected in sediment samples collected in 2005 and fish tissue samples from various species collected between 2004 and 2010. The sampling and analytical details are summarized in this report and presented in detail elsewhere (PADEP 2006). Both CTE and RME estimates were included in order to represent both typical exposures (representative of the average exposures that are likely to occur) and conservative exposures (representative of the maximal exposure that is reasonably likely to occur).

The procedures and guidelines followed in this HHRA are consistent with those outlined by EPA. This methodology includes a four-stage process: hazard identification, exposure assessment, toxicity assessment, and risk characterization (see Figure 1). This HHRA was conducted in a manner consistent with the following documents:

- *Risk Assessment Guidance for Superfund: Human Health Evaluation Manual, Part A* (USEPA 1989);
- Human Health Evaluation Manual, Supplemental Guidance: Standard Default Exposure Factors (USEPA 1991b);
- *Guidance Manual for the Integrated Exposure Uptake Biokinetic Model for Lead in Children. Office of Solid Waste and Emergency* (USEPA 1994);
- *Guidance for Assessing Chemical Contaminant Data for Use in Fish Advisories Volume 1 Fish Sampling and Analysis*, 3<sup>rd</sup> ed. (USEPA 2000);
- Human Health Toxicity Values in Superfund Risk Assessments (USEPA 2003a);
- *Recommendations of the Technical Review Workgroup for Lead for an Approach to Assessing Risks Associated with Adult Exposures to Lead in Soil* (USEPA 2003b);
- *Risk Assessment Guidance for Superfund: Human Health Evaluation Manual, Part E, Supplemental Guidance for Dermal Risk Assessment* (USEPA 2004).



#### Toxicity Assessment

- Identify the appropriate cancer and noncancer toxicity parameters
- Identify COPCs without toxicity parameters
- Identify alternative methods of assessing toxicity (i.e., lead)

#### **Exposure Assessment**

Identify exposure groups

•

- Identify exposure pathways
- Calculate exposure point concentrations
- Estimate CTE and RME intakes/dose
- Use the IEUBK and adult lead model to estimate lead exposures

#### **Risk Characterization**

- Calculate cancer risks for carcinogenic COPCs and sum by exposure route
- Calculate noncancer HQs for COPCs with noncancer effects and sum by exposure route
- Sum cancer risks across COPCs
- Sum noncancer risks across
   COPCs
- Compare summed cancer and noncancer risks to target risk levels

#### Figure 1: Overview of the Four-Stage Risk Assessment Process Followed for this HHRA

## **1.2. Site Location and History**

Presque Isle Bay is located in northwestern Pennsylvania at the southeastern end of Lake Erie (refer to Figure 2). The bay is approximately 4.5 miles long and 1.5 miles wide across at its widest point with an average depth of 13 feet. Access to Lake Erie occurs through a narrow dredged channel at the southeastern end of the bay. The bay is bordered by the City of Erie on the southern shore, Presque Isle State Park on the northern shore and Millcreek township on the western side (PADEP 2002).



Figure 2: Aerial photo of Presque Isle Bay with Area of Concern Boundary

The drainage basin for PIB is approximately 25 square miles consisting mainly of urban and industrial land uses within the City of Erie and the townships of Millcreek, Summit, Greene and Harborcreek. Approximately two-thirds of the water flowing into the bay originates from two main tributaries: Mill Creek and Cascade Creek. Approximately 80 percent of this watershed is comprised of urban land usage (Foyle 2006).

Mud-dominated sediments comprise much of the bay and are known to be contaminated with polycyclic aromatic hydrocarbons, metals and hydrocarbons (Batelle 1994, 1997; Diz 2002; PADEP 2002, 2006) originating from historical sources along the bay and within the watershed (Foyle 2006). Prior to the City of Erie's changes in its wastewater treatment, conveyance and collection system untreated wastewater from industrial, commercial and residential sources was able to reach the bay through combined sewer overflows. Additionally, stormwater runoff from sources within the urbanized watershed has also contributed to pollutant loading of the bay. Many of these contaminants have decayed over the years through natural biodegradation processes, however, substances such as heavy metals and persistent organics still remain in the sediment.

In 1991, PIB was designated as the 43rd Area of Concern (AOC) due to two beneficial use impairments including: restrictions on dredging (due to contaminant concentrations in sediments) and fish tumors and other deformities (PADEP 2002, 2006). Sediment sampling studies have been conducted since the 1980s by the Pennsylvania Department of Environmental Protection (PADEP) in collaboration with community partners and other governmental agencies. While differences exist across these studies, similar conclusions were reached including that bay sediments were found to contain widespread but low levels of polycyclic aromatic hydrocarbons (PAHs) and several heavy metals (i.e., nickel, lead and cadmium). Sediment dredged from the navigation channel and turning basin within the bay by the U.S. Army Corp of Engineers has consistently met the requirements for open lake disposal in Lake Erie (PADEP 2002 and 2006).

The major concern with regard to fish began in the 1980s, when the United States Fish and Wildlife Service began receiving reports of "tumorous" growths on brown bullhead catfish caught within the bay. Since these fish are non-migratory and bottom-dwelling these are in direct and prolonged contact with contaminated sediments (Blazer, et al. 2009a). A number of studies have been conducted on brown bullheads in the bay to examine the rates of both internal and external tumors, their migration habits, and a potential causal relationship between these tumors and sediment contaminants. While the risk factors for the tumors in bay bullheads have yet to be elucidated, these studies taken together show a trend of decreasing tumors in brown bullheads since 1990 (PADEP 2002). It should be noted, however, that the rate of tumor incidence still appears to be higher in PIB compared to non-AOC reference locations (Blazer et al. 2009a and 2009b).

In 2002, PIB was the first AOC in the United States to be designated as in the Recovery Stage. This was based on the determination by the PA DEP in conjunction with the PIB Public Action Committee (PAC) that natural attenuation, rather than active remediation within the AOC, would provide the most practical and cost-effective method for removing the restrictions on dredging activities. This determination, along with the downward trend in

fish tumors during the 1990s, contributed to the re-designation of PIB as an AOC in the Recovery Stage (PADEP, 2006).

Evaluation of Presque Isle Bay sediment and fish populations continued after the redesignation in 2002 (PADEP 2006). This HHRA along with an evaluation of the ecological health (Limnotech 2011) of the bay is an additional dataset that adds to the body of data and research that exists for Presque Isle Bay to assist in the policy and decision-making process.

## 1.3. Recreational Uses of the Bay

Presque Isle Bay has many recreational uses including fishing, boating, sailing, and other water-related activities. Fishing on the bay is prevalent and occurs through access from piers, docks, boats, ice (in winter) and from the shoreline at many locations, as shown in Figure 3. There are numerous public and private marinas providing boat access to both the bay and to Lake Erie. While there are no designated swimming areas or beaches along the shores of the bay, swimming access from the shoreline or boats is likely to occur. Additionally, water skiing and the use of personal watercraft (jet skis) are common within the bay.

The bay is particularly attractive to anglers throughout much of the year. Depending on the season, anglers will commonly pursue opportunities to catch panfish, perch, bass, muskellunge, walleye, northern pike, crappie, or steelhead salmon (PADCNR 2011). Popular shore fishing locations include the Waterworks and Ferry Dock ponds, East and West piers, Perry Monument, North Pier, lagoons and all boat landings. Fishing along the north shore of the bay within Presque Isle State Park is permitted throughout almost the entire length of the park. Many areas along the bay's shore in the park are suitable for wading due to the shallow depths of near-shore areas.

On the western and southern shores of the bay, fishing occurs on-shore and at numerous public docks and piers at various access points located to both west and east of the mouth of Cascade Creek. In addition to these popular areas, anglers also attain fishing access at the public piers located at Dobbins Landing, Liberty Street Dock, Bay Harbor Marina, and the South Pier.

Ice fishing occurs on the bay when there is sufficient ice which usually occurs during the months of December or January. The most popular location for ice fishing includes the head (western end) of the bay, Misery Bay and Horseshoe Pond since these tend to be the first areas to develop a thick enough layer of ice.



Figure 3: Map of Presque Isle Bay showing boat launch and marina locations.

#### 1.4. Fish Consumption Advisories

The 2012 fish advisory for Presque Isle Bay recommends limiting the number of meals of specific sport fish in order to reduce the exposure to mercury and polychlorinated biphenyls (PCBs) (PA Fish and Boat 2011). Table 2 shows the 2012 fish consumption advisory for areas within the Lake Erie basin. In order to limit PCB exposure it is recommended that the following fish be consumed at a rate of one meal per month: smallmouth bass, northern pike, white perch, freshwater drum, bowfin, carp, Coho salmon and steelhead (Rainbow Trout).

Pennsylvania has issued a general, statewide health advisory for recreationally caught sport fish. This advisory recommends no more than one meal (one-half pound) per week of sport fish caught in the state's waterways. This general advice was issued to protect against eating large amounts of fish that have not been tested or that may contain unidentified contaminants (PA Fish and Boat 2011). Other aquatic species from the bay with the potential to be consumed include clams, mussels and turtles. In the state of Pennsylvania the harvesting of live mussels and clams is prohibited. Currently there are no restrictions on the consumption of turtles caught within the Lake Erie basin. However, the advisory does warn consumers that small amounts of PCBs have been found in snapping turtles and that these tend to accumulate in fat and internal organs. The advisory therefore recommends that consumers remove fat and internal organs before consuming turtle meat (PA Fish and Boat 2011).

Advisory Area	Species	Meal	Contaminant	
		frequency		
Lake Erie - Open Waters	Walleye, Coho salmon <sup>(2)</sup> , Steelhead <sup>(2)</sup> (Rainbow trout), Smallmouth bass, White perch, White bass, Lake whitefish, Carp under 20", Freshwater drum, Lake trout and Channel catfish	1 meal/month	PCBs	
	Carp over 20 inches	Do not eat	PCBs	
Lake Erie – Presque Isle Bay	Smallmouth bass, Northern pike, White perch, Freshwater drum, Bowfin, Carp, Coho salmon <sup>(2)</sup> and Steelhead <sup>(2)</sup> (rainbow trout)	1 meal/month	PCBs	
Conneaut Creek (Erie County) SR 0215 bridge to PA/OH border	Smallmouth bass	2 meals/month	Mercury	

#### Table 2: Fish Consumption Advisories for Areas within the Lake Erie Basin<sup>(1)</sup>

Notes:

(1) Fish and Boat Commission. 2012 Fish Consumption Advisory

(2) Salmon and trout are migratory. They may be found seasonally in Presque Isle Bay or Lake Erie tributary streams. Trout, salmon and other fish, whether caught in the lake or elsewhere, should be treated as Lake Erie fish.

# 2. CHEMICALS OF POTENTIAL CONCERN

The purpose of this section is to identify the chemicals detected at the site that will be included in the overall HHRA. The COPCs were selected by comparing the maximum detected concentrations to the appropriate screening criteria. Chemical concentrations that exceeded the screening criteria were retained and included in the overall risk characterization while those chemicals below the criteria were excluded from further evaluation. Tables 2-1 through 2-16 summarize the results of this evaluation for sampling data for both sediment and fish tissue data.

## 2.1. Sampling Data

The dataset used in this HHRA included only those matrices relevant for direct human health exposure pathways: surface sediment (0 to 10 centimeter (cm) in depth) and fish tissue. The most recent set of sediment sampling data, collected in 2005, was utilized in this HHRA (PADEP 2006). It should be noted that this data was collected for purposes other than a human health risk assessment and thus the best available data was used whenever possible. For example, ideally on-shore sediment sampling data would be most appropriate to determine a young child's exposure to on-shore sediment through ingestion and direct contact. However, since this data was not available, in-water sediment sampling data was used as a surrogate. It is likely that this provides a more conservative (i.e., higher) estimate of human health exposures.

Contaminant sampling within pore water or the water column was not conducted and thus, this potential exposure pathway could not be evaluated in this HHRA. It is likely that this would be a minor or insignificant exposure pathway for contaminants of concern.

## 2.1.1. Sediment Sampling Summary

Table 2-1 (Appendix) summarizes the sediment sampling data that was utilized in this HHRA. The dataset used in this HHRA was collected from September 12 through September 15 in 2005 from a comprehensive sediment survey (PADEP 2006). Partners in the survey included PADEP, PIBPAC, Pennsylvania Sea Grant, Gannon University, the Regional Science Consortium and the Erie County Department of Health. Funding for the study was provided by the Great Lakes National Program Office and directed by MacDonald Environmental Services Ltd.

In this survey, a total of 32 surficial samples and four core sediment samples were collected (PADEP 2006). The surficial samples were collected from the top 10 centimeters of sediment using a Van Veen grab sampler. Twelve of the samples were collected based on historical locations while twenty samples were collected from randomly selected locations (refer to Figure 2-1 in Appendix). Two of the four cores were cut into 5 cm sections to a depth of 80 cm and subsequently analyzed. The remaining two core samples were archived.

Only data relevant to the exposure scenarios were included in the risk assessment. Since contact with sediments is only likely to occur during wading and swimming, only near-shore sampling sites were included in the analysis. For the purposes of this HHRA near-shore was considered to be those samples collected from areas with a depth of 10 feet or less. Sampling sites from the center of the bay and within the dredging zone were

excluded from the analysis. Additionally, only surficial samples collected from the top 10 cm of sediment were included. The two core samples were not included in this analysis. This resulted in a total of 14 sample sites being included in the risk assessment as shown in Figure 2-1.

The sediment contaminants included in this HHRA include those summarized in Table 3. Additional compounds or parameters were quantified in sediment samples but not included in this risk assessment (refer to Table 2-2 in Appendix). These constituents or quality parameters were excluded due to their lack of correlation with human health risks or, in the case of alkyl-PAHs, due to lack of information that would allow human health risks to be quantified.

Metals	PAHs	PCBs	Pesticides
Arsenic Cadmium Chromium (total) Copper Lead Mercury Nickel Zinc	Acenaphthene Acenaphthylene Anthracene Benzo(a)anthracene Dibenzo(a,h)anthracene Benzo(a)pyrene Benzo(b)fluoranthene Benzo(c)pyrene Benzo(c)pyrene Benzo(c)pyrene Benzo(c)fluoranthene Chrysene Fluoranthene Fluorene Indeno(1,2,3-c,d)pyrene Naphthalene Perylene Phenanthrene Pyrene	Total PCBS PCB008 PCB018 PCB028 PCB044 PCB052 PCB066 PCB087 PCB101 PCB105 PCB118 PCB128 PCB128 PCB138 PCB153 PCB170 PCB180 PCB180 PCB187 PCB195 PCB206 PCB209	Aldrin Chlordane, technical grade Dieldrin o,p'-DDD p,p'-DDD o,p'-DDE p,p'-DDE o,p'-DDT p,p'-DDT Endosulfan-alpha Endosulfan-beta Endrin Heptachlor Heptachlor epoxide Hexachlorobenzene Hexachlorobenzene Hexachlorobutadiene Hexachlorocyclohexane-γ Hexachlorocyclopentadiene Methoxychlor Mirex Nonachlor, trans- (chlordane)

**Table 3: Inorganic and Organic Analytes Measured in Sediments** 

#### 2.1.2. Fish Tissue Sampling Summary

Fish tissue data that was included in this HHRA was originally collected expressly for the purpose of and in accordance with DEP's Fish Consumption Advisory Program (PADEP 2010). Tables 2-3 through 2-16 summarize the results of these sampling surveys. Fish species were collected during various time periods from February 8, 2004 through

November 10, 2010 from three approximate sampling locations in Presque Isle Bay and Lake Erie (refer to Figure 2-2). Samples collected in Presque Isle Bay occurred primarily via electrofishing while sampling in Lake Erie occurred via gillnets, trot lines and/or angling until the required number of fish of the target species were caught. Table 4 summarizes the fish species (common name), date of sampling and area of sampling. The last two columns of this table indicate the assumption of where the fish was likely to reside for the majority of its life. Table 4 summarizes the fish tissue species included in this risk assessment, the location of sampling and the assumption of where each species is likely to spend most of its life (Presque Isle Bay or Lake Erie).

One fish tissue sample represents ten scaled, skin-on fillets from a composite of five individuals of the fish species being targeted. Channel catfish and burbot samples consisted of ten skinless fillets. As per PA DEP guidelines, all fish in the composite were of the same species and approximately the same size, (i.e., lengths of all fish in the composite were within 75 percent of the length of the largest fish) (PADEP 2010).

Category	Common Name	Year	Area caught	Assumption of "residence time" of fish	
				PIB	LAKE
	Bluegill	2004	PIB	Х	
		2004	LEW		
		2006	LEW		
	Lake trout	2007	LEW		Х
		2008	LEW		
		2010	LEW		
	Largemouth bass	2005	PIB	Х	
Due deten/Come/		2006	PIB		
Other Species	Northern Pike	2010	PIB	X	
Other Species	Pumpkinseed sunfish	2004	PIB	X	
		2004	LEE		
		2005	LEE		
	Smallmouth hass	2006	LEE		v
	Smallmouth bass	2007	LEE		Λ
		2008	LEW		
		2010	LEW		

Table 4: Summary of Fish Species, Sampling Information and Residence Time

Category	Common Name	Year	Area caught	Assumption of "residence time" of fish		
				PIB	LAKE	
		2007	LEW			
	Walleye	2008	LEW		Х	
		2010	LEE			
	White bass	2004	LEW		X	
		2004	LEW			
	Vallaw Darah	2005	LEW			
		2006	LEW/PIB		v	
	renow Perch	2007	LEW		Λ	
		2008	LEW			
		2010	LEW			
	Brown bullhead	2005	PIB	X		
	Durk of	2007	LEE		v	
Bottom	Burbot	2008	LEE		Λ	
dwelling		2004	LEE			
species	Channel Catfish	2005	LEE		Х	
<sup>o</sup> r <sup>onos</sup>	pecies	2010	LEW			
	Common carp	2010	PIB	X		
	White sucker	2007	LEE		Х	

 Table 4: Summary of Fish Species, Sampling Information and Residence Time (cont.)

The fish tissue samples were analyzed by validated methods and included the chemical constituents summarized in Table 6. The chemical constituent concentration was determined as the mass of chemical per wet weight of fish tissue except for those chemical constituents as noted in the table. In addition to these constituents, channel catfish were analyzed for a total of 22 radioactive isotopes in 2010 (refer to Table 2-17 within the Appendix). The analytical results showed no levels of these radioactive isotopes within any of the fish tissue samples.

Metals	Arochlors	Pesticides
Barium <sup>(2)</sup>	Arochlor 1221	Aldrin
Cadmium	Arochlor 1232	alpha-BHC
Chromium (total)	Arochlor 1242	alpha-Chlordane
Copper	Arochlor 1248	gamma-Chlordane
Lead	Arochlor 1254	Chlordene
Mercury	Arochlor 1260	4,4'-DDD
Selenium		4,4'-DDE
Strontium <sup>(2)</sup>		4,4'-DDT
		O,P-DDD
		O,P-DDE
		O,P-DDT
		Methoxychlor
		Mirex <sup>(3)</sup>
		cis-Nonachlor
		trans-Nonachlor
		Oxychlordane
		Endrin
		Heptachlor
		Heptachlor epoxide
		gamma-GHC (Lindane)
		Dieldrin

Table 5: Inorganic and Organic Analytes Measured in Fish Tissue<sup>(1)</sup>

Notes:

(1) Chemical concentrations measured as mass of chemical per wet weight of fish tissue.

(2) Barium and strontium analyzed only in tissue of Channel Catfish.

(3) Mirex was measured in all species except bluegill and pumpkinseed.

#### 2.2. Selection of COPCs

Inclusion or exclusion of chemical constituents in the subsequent risk assessment was based on the guidelines established by EPA (USEPA 1989). This guidance recommends utilizing screening criteria to limit the number of chemicals that are carried through the quantitative risk assessment while ensuring that all chemicals that may contribute to the overall risk are still included (USEPA 1989).

In order to achieve this objective the results of the sediment analyses were screened against the EPA Region 3 Risk-based Screening Levels (RSLs) to determine whether the constituents should be included in the next stage of the risk assessment (USEPA 2011a, 2011b). These screening values are likely to be conservative and protective of human health since these are based on residential exposures and assume that the exposure frequency is 365 days per year and the exposure duration is 30 years.

#### 2.2.1. COPCs in Sediment

The residential soil RSLs were chosen utilizing the following selection criteria:

- 1. If available, 1/10 of the value of the non-carcinogenic RSL was obtained from the residential table for soil (HQ = 0.1);
- 2. If available, the carcinogenic RSL was obtained from the non-residential table for soil (target risk =  $1 \times 10^{-6}$ );
- 3. the screening level was selected by choosing the lower (more stringent) value of the two values identified in steps 1 and 2.

Table 2-1 (Appendix) shows the results of comparing the maximum measured value of each constituent in sediment to the appropriate RSL. As a result of this evaluation, a total of nine constituents (two metals, six PAHs and total PCB congeners) were found to have a maximum concentration greater than the applicable RSL and were subsequently carried forward in the risk assessment for the direct contact with sediment exposure pathway. Table 6 summarizes the screened COPCs, the maximum detected value, the location of the maximum value and the number of values detected above the RSL.

The chromium concentration in sediment was measured and reported as total chromium. There are no RSLs or toxicity values available for total chromium. Instead the RSL for trivalent chromium was used for screening purposes. Studies have demonstrated that hexavalent chromium tends to reduce to trivalent chromium in anaerobic conditions and in the presence of reducing agents such as  $S^{-2}$  and Fe.<sup>+2</sup> A study by Graham, et al. found that the Cr(VI)-reducing capacity of sediments was strongly correlated to the acid volatiles content of the sediments (Graham 2009) and thus trivalent chromium is more prevalent in the environment (ATSDR 2008). In risk assessments, it is often assumed that the ratio of Cr VI to Cr III is 1:6 (reference). The RSLs for hexavalent and trivalent chromium in residential soil are 0.29 mg/kg (cancer effects) and 12,000 mg/kg (for noncancer effects) respectively. The uncertainty associated with using the toxicity parameters for trivalent chromium is further discussed in the Uncertainty Section 7.3.1 ("Use of Trivalent Chromium Toxicity Parameters for Total Chromium.").

Only two (PCB 105 and PCB 118) of the 19 polychlorinated biphenyls (PCBs) that were evaluated had applicable RSLs. Therefore, the RSL for total high risk PCBs was used to evaluate this class of compounds. This value represents the sum of the concentration of the 19 PCB congeners at each location (refer to Table 2-18 in Appendix). Two of the PCBs measured are considered to be dioxin-like PCBs and were included in the total PCB concentration. These two congeners were measured in concentrations well below the applicable RSL.

Total chlordanes and total DDT and its derivatives were summed and compared to the screening levels for chlordane and DDT respectively. Total chlordanes included the sum of the concentrations of chlordane, heptachlor, heptachlor epoxide, and nonachlor at each sampling site. The summed concentration did not exceed the RSL for chlordane. Similarly, the sum of DDT and its derivatives included the summed concentrations of six derivatives as shown in Table 2-18. Likewise, the total concentration of all derivatives did not exceed the RSL for DDT.

For other chemical constituents without RSLs, structural analogy was utilized in that the RSL for a chemical with a similar structure was substituted. These were based on the surrogates for toxicity values available from the PADEP toxicity database (PADEP 2011). These substitutions included: acenaphthene for acenaphthalene and benzo(g,h,i)perylene; pyrene for benzo(e)pyrene and perylene; and anthracene for phenanthrene.

Chemical	EPA Region 3 RSL <sup>(1)</sup> (mg/kg)	Maximum sediment concentration (mg/kg)	Location of sample with maximum value	Number of samples above the RSL
Arsenic	0.39	30.1	47-PIP	14/14
Benzo(a)anthracene	0.15	2.2	15-PIB/27-MC	13/14
Benzo(a)pyrene	0.015	2.7	15-PIB	14/14
Benzo(b)fluoranthene	0.15	2.7	15-PIB	14/14
Benzo(k)fluoranthene	1.5	2.9	15-PIB	3/14
Dibenz(a,h)anthracene	0.015	0.44	39-PIB	14/14
Indeno(1,2,3-c,d)pyrene	0.15	3.1	15-PIB	14/14
Lead	40	127	18-PIB	11/14
Total PCBs	0.22	0.37	35-PIB	1/14

Table 6: COPCs with Maximum Values Exceeding the Residential Soil RSLs

Notes:

(1) USEPA Region 3 Risk-Based Screening Levels for Residential Soil (USEPA, 2011a)

#### 2.2.2. COPCs in Fish Tissue

Chemical concentrations in fish tissue were screened against either the EPA Region 3 RSLs or other fish consumption advisory levels that are utilized by the PADEP to develop advisories for Pennsylvania lakes and tributaries (Anderson 1993, PADEP 2010, FDA 2011). If a COPC had multiple screening levels, the lowest value of the RSL or fish consumption advisory level was utilized. The fish tissue screening RSLs (USEPA 2011b) were chosen utilizing the following selection criteria:

- 1. If available, 1/10 of the value of the non-carcinogenic RSL was obtained from the table for fish tissue (HQ = 0.1) (USEPA 2011b);
- 2. If available, the carcinogenic RSL was obtained from the table for fish tissue (target risk =  $1 \times 10^{-6}$ ) (USEPA 2011b);
- 3. If available, the fish consumption advisory level was selected (Anderson 1993, USEPA 1997, FDA 2011);
- 4. the screening level was selected by choosing the lower (more stringent) of the values identified in steps 1 through 3.

Tables 2-3 through 2-16 (Appendix) show the results of comparing the maximum measured value of each constituent in fish tissue to the Region 3 RSL values or fish consumption advisory levels as detailed above (USEPA, 2011b). Based on this review, a total of 20 constituents were found to have a maximum concentration greater than the applicable RSL or fish consumption advisory level in at least one fish species and were subsequently carried forward in the risk assessment for the fish consumption exposure pathway. Table 7 summarizes the contaminants that were identified as COPCs by fish species. Lake trout and smallmouth bass were the species with the highest number of maximum values greater than the screening levels at 16 and 14 respectively. Bluegill and pumpkinseed (panfish) did not have chemical concentrations that exceeded the screening levels. Those chemicals identified as COPCs were included in the risk estimates.

As with the sediment samples, the chromium concentrations in fish tissue were reported as total chromium. Since there are no screening levels or toxicity values for total chromium, the screening level for trivalent chromium was used as a surrogate since the majority of chromium in the environment is likely to be in the trivalent form as previously discussed (ATSDR 2008, Graham 2009). This approach is further supported by a study which found the bioconcentration factor (BCF) for Cr(VI) in fish muscle to be less than 1.0 which suggests hexavalent chromium is not likely to bioaccumulate in fish tissue (USEPA 1998).

It should be noted that there were numerous chemicals for which the method detection limit was greater than that of the screening value. These chemical constituents were not included in the risk assessment since the concentration in fish tissue could not be ascertained. This is further discussed in the Uncertainty Analysis section 6.1.3 entitled "Detection Limits Greater Than the RSLs."

	Bluegill	Brown Bullhead	Burbot	Channel catfish	Common carp	Lake Trout	LM bass	North Pike	Pumpki nseed	SM Bass	Walleye	White Bass	White sucker	Yellow Perch
Aldrin										•				
Arochlor 1254			•							•	٠			
Arochlor 1260				•	٠	٠	•			•	٠	•	٠	
α-BHC						•				٠	٠			
$\alpha$ -Chlordane				•		•								
γ-chlordane						•								
4,4'-DDD				•		•				•		•		
4,4'-DDE				•		٠	•	٠		•	•		•	
4,4'-DDT				•		•				٠		•	•	
O,P-DDT						•				•				
Dieldrin				•		٠		٠		٠	٠	•		
Heptachlor				•		•				•				
Heptachlor Epoxide						•				•				
Mercury			•	•		•	•			●	●		•	●
Mirex											٠			
Cis-Nonachlor						•								
Trans-Nonachlor						•				•				
Oxychlordane						•								
Selenium		•	•	•	•	•	•	•		•	•		•	•
Strontium				•				•						

# Table 7: COPC Summary of Contaminants with Maximum Values Greater than the Applicable Screening Level.

## 3. EXPOSURE ASSESSMENT

The exposure assessment stage involves the estimation of the magnitude, frequency and duration of current and future human exposures for each complete exposure pathway.

## 3.1. Conceptual Site Model

The conceptual site model (CSM) for Presque Isle Bay is shown in Figure 3-1. The purpose of the conceptual site model is to identify complete and incomplete exposure pathways. A detailed account of the rationale for including or excluding exposure pathways and receptors is provided in the next two sections (3.2 and 3.3) and summarized in Table 8 below.

## **3.2. Exposure Pathways**

Exposure pathways are defined as the means by which a person comes into contact with a chemical within environmental media. In order for an exposure pathway to be complete the following four elements must be present (USEPA 1989):

- a source of contamination;
- a mechanism for transport of a substance from the source to the air, surface water, groundwater and/or soil;
- a point where people come in contact with contaminated air, surface water, groundwater or soil; and
- a route of entry into the body.

If all four of these elements are met, the pathway is considered complete and potentially included in the next stages (toxicity assessment, exposure assessment, risk characterization) of the risk assessment. If any elements are missing, the pathway is considered incomplete and would not be included in the next stages of the risk assessment. The rationale for including and excluding pathways is provided below. Table 8 summarizes the exposure pathways and the rationale for including or excluding each within this risk assessment.

Media	Exposure Pathway	Pathway	Pathway Retained	Rationale
	Dermal contact	Dermal contact with contaminated sediments during water-related activities	Yes	Dermal contact with sediment is a potentially complete exposure pathway.
Sediment	Ingestion	Incidental ingestion of contaminated sediments during water-related activities	Yes	Incidental ingestion of sediment is a potentially complete exposure pathway.
	Inhalation	Inhalation of re-entrained sediment into air No		This is likely to be a minor or insignificant exposure pathway for all exposure groups
	Γ	1		1
Fish	Ingestion	Ingestion of contaminated fish tissue by anglers, their families and other fish consumers	Yes	Consumption of fish is likely to be a significant exposure pathway.
Waterfowl	Ingestion	Ingestion of contaminated tissue from waterfowl and other aquatic organisms	No	No comprehensive data available. Many duck/goose species in PA are migratory making it difficult to isolate PIB as a contaminant source.
	Ingestion	Ingestion of clams and mussels.		No comprehensive data available to evaluate
Clams and Mussels			No	Exposure pathway is likely to be incomplete (see advisory notice in section 1.4)
Turtles	Ingestion	Ingestion of contaminated turtle	No	No comprehensive data available to
		meat.		evaluate
	Dermal contact	Dermal contact with chemicals in water while swimming, wading, etc.		These are likely to be minor or insignificant exposure pathways.
Surface Water	Ingestion	Incidental ingestion of surface water while swimming/wading	No	Most organic chemicals have minor to negligible solubility in water.
	Inhalation	Inhalation Water while swimming/wading Inhalation of vapors of VOCs/semi-VOCs from surface		No comprehensive data on chemical concentrations in surface water were available.

**Table 8: Potential Exposure Pathways for Presque Isle Bay** 

## 3.2.1. Potentially Complete and Significant Pathways

Consumption of contaminated fish was considered to be a complete and potentially significant pathway. Presque Isle Bay and Lake Erie anglers were considered to be a group that is likely to have exposure to chemical contaminants. This is likely to include their families as well.

Anglers may also come into contact with chemical constituents through direct contact with sediments, direct contact with water and inhalation of vapors from surface water. These, however, are likely to be relatively minor sources of exposure. The most significant exposure in this group includes the consumption of contaminated fish. This may also hold true for the family members of anglers who also consume fish from the bay. Children of adult anglers were considered a separate exposure group in this HHRA.

## 3.2.2. Potentially Complete and Negligible Pathways

Inhalation of vapors or dust from contaminated sediment by exposure groups is considered to be a negligible pathway. Particulate and vapor concentration in the ambient air is likely to be low due to dilution and mixing within the area. There is no comprehensive data for the bay which specifically looks at the flux of various semivolatile organic compounds (semi-VOCs) from water to air. The PA DEP conducts regular air monitoring for hazardous air pollutants at a site located in Presque Isle State Park. These concentrations, however, represent air concentrations from all sources and are not exclusively representative of volatilization from surface water.

Indirect exposure due to vapor intrusion (movement of vapors from soil/sediment to indoor structures) was not considered a complete exposure pathway for residential receptors since residential receptors are located greater than 100 feet horizontally from the source of soil/sediment contamination (PADEP 2002). While there are some individuals that reside in houseboat structures at various marinas within the bay, it is unlikely that significant amounts of vapors would accumulate in these structures from movement of chemicals from surface water to inside the houseboat structure.

## 3.2.3. Incomplete Pathways

Water from Presque Isle Bay is not used as a source of drinking water and, therefore, ingestion of contaminated drinking water was considered to be an incomplete pathway. The City of Erie Water Authority supplies potable water to properties located within the City limits. Additionally, the City of Erie Codified Ordinances, Part Nine - Streets, Utilities and Public Services Code, Title Five - Sewers and Water, Article 947 Non-Used Aquifers regulations indicate that "no well or spring located on a property shall be used as a source for drinking water or agricultural purposes." The regulations of the ordinance state that no owner, lessee or other person shall use any groundwater source for drinking water or agricultural purposes.

## 3.2.4. Exposure Pathways Not Evaluated

Comprehensive data on chemical concentrations within the water column were not available to evaluate exposures in this risk assessment. Therefore, the exposure pathway of incidental ingestion of water could not be assessed. It is likely, however, that this would be a negligible pathway of exposure due to the hydrophobicity of many of the COPCs; the small amounts of chemicals that would be ingested from the water column; and the potential low probability of incidental ingestion of bay water.

The exposure pathway of consuming contaminated waterfowl was not evaluated. Duck and goose hunting is allowed at limited times and locations in-season within areas of Presque Isle Bay. However, due to the migratory nature of waterfowl and lack of adequate contaminant concentrations in waterfowl tissue, this potential pathway could not be assessed within this risk assessment.

The exposure pathway of consuming contaminated clams or mussels was not evaluated. It was assumed that this pathway would be a nonexistent or rare exposure since the state of Pennsylvania prohibits the harvesting of live mussels and clams.

## 3.3. Potential Exposure Groups

The goal of this risk assessment is to identify and characterize the predominant and most significant receptor groups rather than identifying every possible group that may exposed no matter how insignificant. Based on the current and most common usages of Presque Isle Bay the primary receptor groups include recreational water users and Presque Isle Bay anglers. While additional receptor groups could have been developed, it is likely that the receptor groups focused on in this HHRA include the dominant and most likely exposure pathways (i.e., groups with the highest potential exposures).

#### 3.3.1. Adult Recreational Water Users

Recreational water users may be exposed to contaminanted sediments while swimming, wading, boating, fishing, and other activities. While there are no public beaches on the bay, it is likely that swimming does occur at various locations. Potential exposures associated with recreational water use include: dermal contact with contaminated sediments, incidental ingestion of contaminated sediments, dermal contact with water, incidental ingestion of water and inhalation of chemicals from surface water.

The exposure assumptions included in the dose and intake calculations were based either on default values (USEPA 1991b, 2004) or best professional judgement using site-

specific assumptions. The assumption parameters used to calculate the intakes for adult recreational water users are summarized in Tables 3-1 and 3-2 of the Appendix.

In order to calculate an exposure estimate for sediment ingestion, default soil ingestion values from the EPA Exposure Factor Handbook were included (USEPA 2011c). These default values include 100 mg/day (95<sup>th</sup> percentile value for RME calculation) and 50 mg/day (mean value for CTE calculation) for adults. The fraction of contaminated soil or sediment ingested was conservatively assumed to be 0.5 for the RME estimate and 0.3 for the CTE estimate. This is based on the assumption that recreational water users would likely have exposures from other areas such as work or home due to dividing their time between various locations.

The average adult recreational water user was assumed to typically wear a short-sleeved shirt, shorts and no shoes. Thus, the exposed skin surface area (5,700 cm<sup>2</sup>) was the sum of the average of the 50th percentile surface area for adult males and females for the hands, forearms, calves and feet. This value is the recommended exposed surface area for both CTE and RME estimates (USEPA 2004).

The soil-to-skin adherence factor was assumed to be 0.07 milligrams per square meter (mg/cm<sup>2</sup>) and 0.3 mg/cm<sup>2</sup> for the CTE and RME estimates respectively (USEPA 2004, 2011c). These values correspond to the recommended soil adherence factors, or mass of soil that adheres per surface area of skin, for an adult resident. EPA recommends a similar approach for sediments as for soils due to a lack of detailed studies concerning dermal exposures to sediments (USEPA 2004).

The exposure duration assumptions for the CTE and RME estimates included EPA default values (USEPA 1991b). A value of 9 years was used for the CTE calculation which represents the median length of time an individual stays at one residence in the U.S. (USEPA 1991b). For RME estimates, a value of 30 years was included representing the 90<sup>th</sup> percentile value for the length of time an adult lives at one residence in the U.S.

The values included for exposure frequency for adults were based on best professional judgment. The exposure frequency assumed for the CTE estimate was based on adults that would come into contact with bay sediments an average of 38 days across a year. This value is derived from an individual conducting water-related activities 2 times per week for 13 weeks during the summer and 12 times during the spring and fall months. For the higher-end or RME estimate it was concluded that individuals would come into contact with bay sediments an average of 81 days per year. This value is based on a frequency rate of 5 days per week across 13 weeks (65 days) during the summer and 1 day per week for 16 weeks (16 days) for the spring and fall.

## 3.3.2. Child Recreational Water Users

Tables 3-3 and 3-4 summarize the assumptions included in the exposure calculations for exposure to sediment for child recreational water users (6 months to 6 years old).

In order to calculate an exposure estimate for sediment ingestion, default soil ingestion values from the EPA Exposure Factor Handbook were utilized (USEPA 2011c). These values include 200 mg/day (RME value) and 100 mg/day (CTE value). The fraction of contaminated soil or sediment ingested from Presque Isle Bay was conservatively assumed to be 0.5 for the RME estimate and 0.3 for the CTE estimate. This is based on the assumption that recreational water users would likely have exposures from other areas such as work or home due to dividing their time between various locations.

In order to calculate an exposure estimate for sediment ingestion, default soil ingestion values from the EPA Exposure Factor Handbook were included (USEPA 2011c). These default values include 200 mg/day (RME value) and 100 mg/day (CTE value) which represent the mean and 95<sup>th</sup> percentile value for soil ingestion for this age group. The fraction of contaminated soil or sediment ingested was conservatively assumed to be 0.5 for the RME estimate and 0.3 for the CTE estimate. This is based on the assumption that recreational water users would likely have exposures from other areas such as work or home due to dividing their time between various locations.

The child water user was assumed to typically wear a short-sleeved shirt, shorts, and no shoes. Thus, the exposed skin surface area (2,800 cm<sup>2</sup>) was the average of the 50th percentile surface area for the forearms, hands, legs, and feet for males and females for children aged 6 months to 6 years (EPA 2004, 2011c).

The soil-to-skin adherence factor was assumed to be 0.2 milligrams per square meter (mg/cm<sup>2</sup>)-event and 3.3 mg/cm<sup>2</sup>-event for the CTE and RME estimates, respectively (USEPA 2004, 2010). These values correspond to the mean and 95<sup>th</sup> percentile recommended soil adherence factors for a child resident. EPA currently recommends the identical approach for sediments as for soils since there is a lack of data concerning dermal exposures to sediments (USEPA 2004).

The values for exposure duration, body weight and averaging time included EPA default values (USEPA 1991b). The exposure duration was 6 years which is the default value recommended by EPA for children aged 0 through 7 years (USEPA 1989, 1991b). The average body weight included for both CTE and RME estimates was 15 kg, the average body weight of children under 7 in the United States (USEPA 1991b). For cancer

estimates, 70 years, the default average lifetime value for the U.S. population was used (USEPA 1991b).

The same assumptions used to estimate the values for exposure frequencies in adults were also used for children. These assumptions are discussed in the previous section.

## 3.3.3. Presque Isle Bay Anglers and Their Families

Presque Isle Bay anglers were considered to be a group that is likely to have exposure to chemical constituents within the bay. This may also hold true for their family members. Anglers may come into contact with chemical constituents while fishing through direct contact with sediments, direct contact with water and inhalation of vapors from surface water. The assumptions included in these exposure calculations are identical to those outlined in the previous section entitled "recreational water users." These, however, are likely to be relatively minor sources of exposure.

The most significant exposure in this group includes the consumption of contaminated fish. This also holds true for family members of anglers who consume fish. The exposure group that may receive the highest exposure includes children which was an exposure group considered in this risk assessment. Based on the location of sampling and habits of each species, the fish species that were considered to reside primarily in Presque Isle Bay included:

- Blue gill
- Largemouth bass
- Northern pike
- Pumpkinseed sunfish
- o Brown bullhead
- Common carp

For the purposes of this risk assessment bay anglers were considered to be recreational anglers and urban/subsistence anglers. The CTE calculation was considered to represent the recreational angler while the RME estimate would include a higher end urban/subsistence angler. The exposure assumptions used to calculate the RME and CTE estimates are consistent with the EPA document entitled *Guidance for Assessing Chemical Contaminant Data for Use in Fish Advisories* (USEPA 2000) and the *Estimated Per Capita Fish Consumption in the United States* report (USEPA 2002). These rates were estimated from a national dietary study and may not be representative site-specific consumption patterns. Additional uncertainties associated with these ingestion rates are discussed in Section 7.2.2.1. "Fish Consumption Rates."

#### 3.3.3.1. CTE Calculation – Adult Recreational Anglers

Recreational anglers include those who fish in Presque Isle Bay for both sport and non-sport fish. While anglers may not exclusively limit their fishing to the bay to include Lake Erie and its tributaries, individuals were assumed to fish primarily from the bay for the purposes of this risk assessment. Anglers are likely to fish through a variety of means including from boats, the shoreline and various public piers and docks located along the bay.

The ingestion rate for fish used in this calculation was a value of 17.5 grams/day. This value corresponds to the average ingestion rate for uncooked freshwater and estuarine finfish for adults (age 18 and older) within the United States (USEPA 2000 and 2002). This assumption represents an average of 2.3 fish meals per month (28 meals per year) and includes a serving size of 227 grams (8 ounces) per meal for an average 70 kg. adult for every month of the year. A single species diet was assumed in this calculation and all consumed fish originated from the study area. No reduction in chemical concentration was considered for the cooking and cleaning of fish.

The assumptions used for both dermal contact and incidental ingestion of sediment for this group are the same assumptions used for the CTE calculation for adult recreational water users as outlined in Section 3.3.1 and Tables 3-1 and 3-2 within the Appendix. The exposure duration used in this calculation was 9 years which represents the default value used by EPA to represent the average time a U.S. resident resides at their current residence (USEPA 1991b). An average body weight of 70 kg was used in the exposure estimates which correspond to the value for an average adult residing in the U.S.

#### 3.3.3.2. CTE Calculation – Children of Adult Recreational Anglers

This exposure group represents the children of adult recreational anglers who consume fish caught from the study area. The fish consumption rate used for children was assumed to be proportional by body weight to that of the adult angler resulting in an intake rate of 3.75 grams per day (15 kg/70 kg  $\times$  17.5 grams/day = 3.75 grams/day). The exposure frequency included is 365 days per year to correspond to the use of an annual average consumption rate. The additional parameters used in these calculations correspond to the default values for body weight, exposure duration and averaging time as discussed in the section describing the assumptions for child recreational water users.

#### 3.3.3.3. RME Calculation – Adult Urban/Subsistence Anglers

The RME calculation for fish consumption is for the high end fish consumer. There is no comprehensive survey data regarding fish consumption of anglers within Presque Isle Bay. It is possible that there is a population that consumes fish close to that of a subsistence angler. A recent focus group survey of anglers within the Great Lakes by Lauber, et al., suggested that "urban sites have significant subpopulations of anglers from different ethnic and cultural backgrounds with different fish consumption norms (Lauber, et al. 2011)." In the same survey urban anglers were more likely to consume large amounts of fish if they could not easily obtain food through other means. The exposure pathways for the urban angler are identical to that of the recreational angler but include higher end values for the exposure parameters.

To account for this possibility the RME calculation uses the default EPA fish consumption rate of a subsistence angler (USEPA 2002) of 142.4 g/day. This value corresponds to the 99<sup>th</sup> percentile ingestion rate for uncooked freshwater and estuarine finfish for adults (age 18 and older) within the United States (USEPA 2000, 2002). This assumption represents an average of 19 fish meals per month (228 meals per year) and includes a serving size of 227 grams (8 ounces) per meal for an average 70 kg adult.

In calculating the estimated intakes of chemical constituents from ingestion of fish two additional assumptions included a single species diet and that all fish consumed were caught from the Presque Isle Bay area. No reduction in chemical concentration was considered for the cooking and cleaning of fish. The exposure duration used in this calculation was 30 years which represents the default values used by EPA to represent the 90<sup>th</sup> percentile (high-end) estimate of time a U.S. resident lives at their current residence (USEPA 1991b).

#### 3.3.3.4. RME Calculation – Children of Adult Urban/Subsistence Anglers

This exposure group represents the children of adult urban/subsistence anglers who consume fish caught from the study area. Limited information is available about fish consumption for children 6 months to 7 years of age. The national dietary study, on which the adult fish consumption rates are based, does not include consumption information for young children. Therefore, the fish consumption rate used for children was assumed to be proportional by body weight to that of the adult angler resulting in an intake rate of 30.5 grams per day (15 kg/70 kg × 142.4 grams/day = 30.5 grams/day). The exposure frequency included is 365 days per year to correspond to the use of an annual average consumption rate. The additional parameters used in

these calculations correspond to the default values for body weight, exposure duration and averaging time as discussed in the section describing the assumptions for child recreational water users.

## 3.3.4. Lake Erie Anglers and Their Families

The exposure assumptions used to calculate cancer and non-cancer risks for Lake Erie Anglers and their families (adult and children) are identical to those used for Presque Isle Bay anglers. There is no specific data available to include site-specific parameters about percentage of fish caught in the lake versus the bay. Therefore, for the purposes of this HHRA, risk estimates are based on the assumption that the person consumes one species which are all taken from Lake Erie.

The fish species included in these estimates included those that are assumed to spend a majority of their lives within Lake Erie. For the purposes of this risk assessment, these species included:

- Burbot
- Channel catfish
- Lake trout
- Smallmouth bass
- Walleye
- White bass
- Yellow perch
- White sucker

# 3.3.5. Exposure Groups not Included in this Analyses

Other groups may be exposed to bay contaminants through work-related activities such as individuals performing dredging activities in the bay and researchers collecting sediment for research purposes. These groups were not considered in this particular analysis. Contact with sediment contaminants among these groups is likely to be of a limited and short-term nature (i.e., less than one year) and lower than other exposure groups included in this analysis. This assumption is based not only on the limited nature of the work but also includes the assumption that workers would limit their exposure to contaminated sediments through various protective measures.

Residents living near or on Presque Isle Bay were not considered as a separate exposure group since their exposure is likely to be low unless they participate in activities that put them in contact with contaminated media. Such individuals would be included in the risk assessment due to their specific activities such as participating in water-related activities and consuming fish.

## 3.4. Calculation of Exposure Point Concentrations

#### 3.4.1. Exposure Point Concentrations for Sediment

Table 9 summarizes the EPCs used in estimating risk for direct contact exposure to sediment for the designated exposure groups according to EPA guidelines (EPA 1992). In calculating EPCs, the exposure area concept was utilized which includes the assumption that over a long period of time a receptor would contact all parts of the exposure area. For RME intake and dose estimates, the 95% upper confidence limit (UCL) of the mean was utilized as the EPC. The 95% UCL of the mean provides a conservative estimate of the average concentration of a chemical across an exposure area. For central tendency exposure (CTE) estimates of intakes and dose, the arithmetic mean value for each constituent was included as the EPC.

UCLs were calculated using the most current version of the ProUCL software (version 4.1) (USEPA, 2010a). The software evaluates the data distribution (i.e., normal, lognormal, or gamma or nonparametric) using various goodness-of-fit tests in order to calculate the appropriate 95% UCL of the mean (USEPA 2010a). ProUCL requires at least seven values in order to calculate an appropriate UCL. For sediment data that contained non-detect values, one-half the detection limit was substituted for the non-detect value. This maintains a conservative risk assessment approach since this method is likely to overestimate the EPC.

Chemical	CTE Arithmetic Mean (mg/kg)	RME 95% UCL of the Mean (mg/kg)	Maximum Sediment Concentration (mg/kg)
Arsenic	8.9	13.2 <sup>(1)</sup>	30.1
Benzo(a)anthracene	1.0	1.3(2)	2.2
Benzo(a)pyrene	1.2	$1.5^{(2)}$	2.7
Benzo(b)fluoranthene	1.3	$1.7^{(2)}$	2.7
Benzo(k)fluoranthene	1.2	1.6 <sup>(2)</sup>	2.9
Dibenz(a,h)anthracene,	0.23	$0.29^{(2)}$	0.44
Indeno(1,2,3-c,d)pyrene	1.3	1.6 <sup>(2)</sup>	3.1
Lead	69.3	85.8 <sup>(2)</sup>	127
Total PCBs	0.07	0.11 <sup>(2)</sup>	0.37

**Table 9: Exposure Point Concentrations for COPCs in Sediment** 

Notes:

<sup>(1)</sup> Data distribution as determined by ProUCL

<sup>(2)</sup> 95% approximate gamma distribution

<sup>(3)</sup> 95% Student's t-UCL

#### 3.4.2. Exposure Point Concentrations for Fish Tissue

Table 3-7 (Appendix) summarizes the EPCs included for chemical constituents in fish tissue. Due to the limited number of samples (sample size between 1 and 5 composites with each composite representing five individual fish of the same species), the EPC used for the RME estimates included the maximum value for each constituent within each of the 12 species regardless of year. EPCs for CTE estimates included the arithmetic mean or maximum value (if only 1 composite was evaluated) of each constituent within fish species regardless of year.

#### 3.5. Quantification of Exposure

The basic equations used to calculate cancer risk and noncancer hazard estimates for the identified exposure scenarios are taken from various guidance documents (USEPA 1989, 1991b, 2004, 2009 and 2010b).

#### 3.5.1. Dermal Contact with Sediment

The dermal absorbed dose is estimated as the dose that crosses the skin and is systemically absorbed. A dermal absorption factor (ABS) is included in this equation to account for the proportion of the chemical that is likely to be absorbed across the skin surface. This dose was estimated from the following equation (USEPA 2004):

$$Intake = \frac{CS \times SA \times AF \times ABS \times EF \times ED \times CF}{BW \times AT}$$

where:

Intake	= dermal absorbed dose of COPC (mg/kg-day, calculated)
CS	= concentration of COPC in sediment (mg/kg)
SA	= surface area of the skin exposed to sediment $(cm^2)$
AF	= soil/sediment adherence factor
ABS	= dermal absorption coefficient – COPC-specific (unitless)
EF	= exposure frequency (days/year)
ED	= exposure duration (years)
CF	= conversion factor (1E-06 kg/mg)
BW	= body weight (kg)
AT	= averaging time (days)

The specific assumptions and values included in these calculations are summarized in Tables 3-1 and 3-3 in the Appendix.

#### **3.5.2.** Incidental Ingestion of Sediment

The ingested intake of COPCs in sediment is estimated by the following equation:

Intake = 
$$\frac{CS \times IR \times FI \times EF \times ED \times CF}{BW \times AT}$$

where:

Intake	= ingested	daily i	intake of	COPCs in	sediment	(mg/kg-day.	calculated)
	0						

CS = concentration of COPC in sediment (mg/kg)

IR = ingestion rate of sediment (mg/day)

- FI = fraction of exposure attributed to site sediment (unitless)
- EF = exposure frequency (days/year)
- ED = exposure duration (years)
- CF = conversion factor (1E-06 kg/mg)
- BW = body weight (kg)
- AT = averaging time (days)

The specific assumptions and values included in these calculations are summarized in Tables 3-2 and 3-4 in the Appendix.

#### 3.5.3. Ingestion of Fish Tissue

The ingested intake of COPCs from fish is estimated by the following equation:

$$Intake = \frac{C_{Fish} \times IR \times EF \times ED \times CF}{BW \times AT}$$

where:

 $\begin{array}{ll} Intake &= ingested \ daily \ intake \ of \ COPCs \ from \ fish \ (mg/kg-day, \ calculated) \\ C_{Fish} &= concentration \ of \ COPC \ in \ fish \ (mg/kg) \\ IR &= ingestion \ rate \ of \ fish \ (g/day) \\ EF &= exposure \ frequency \ (days/year) \\ ED &= exposure \ duration \ (years) \\ CF &= conversion \ factor \ (kg/g) \\ BW &= body \ weight \ (kg) \\ AT &= averaging \ time \ (days) \end{array}$ 

The specific assumptions and values included in these calculations are summarized in Tables 3-5 and 3-6 in the Appendix.

## 4. TOXICITY ASSESSMENT

Toxicity values used in HHRAs quantify the dose-response relationship for a chemical. These values include cancer slope factors (CSFs) and noncancer reference doses (RfDs), both of which are specific to the route of exposure (USEPA 2003a). Tables 4-1 through 4-3 summarize the toxicity values, EPA weight-of-evidence for cancer classification, target organ and health effects and other pertinent information for selected COPCs. The source for these toxicity values was chosen based on the hierarchy as recommended by EPA and includes:

- 1. Integrated Risk Information System;
- 2. EPA's Provisional Peer-Reviewed Toxicity Values (PPRTVs); and
- 3. Other peer-reviewed toxicity values which may include California Environmental Protection Agency (CALEPA), Agency for Toxic Substances Disease Registry (ATSDR) Minimal Risk Levels, and Health Effects Assessment Summary Tables (HEAST) Toxicity Values.

## 4.1. Oral and Dermal CSFs

EPA has developed CSFs specific to the oral route of exposure. In accordance with EPA guidance (1989), this risk assessment uses route-to-route extrapolation to estimate dermal CSFs from oral CSF values in order to estimate the risk associated with dermal contact with

contaminated soil. This extrapolation is done by dividing the oral CSF by a constituentspecific oral absorption factor. To calculate a dermal CSF for a particular chemical, the oral CSF is divided by the oral absorption efficiency value (GIABS) (USEPA 2004, 2010). The adjusted CSFs for dermal exposure are summarized in Table 4-2.

## 4.2. Oral and Dermal RfDs for Non-carcinogenic Effects

Oral reference doses are expressed in units of daily dose (mg/kg-day) and incorporate uncertainty factors to account for limitations in the quality or quantity of available data. The EPA defines the RfD or RfC as an estimate of the daily maximum level of exposure to human populations (including sensitive sub-populations) that is likely to be without an appreciable risk of adverse effects across a lifetime (USEPA 1989). The oral RfD provides a benchmark against which human intakes (via ingestion) are compared.

In this risk assessment, dermal RfDs were extrapolated from the oral RfD values using the appropriate oral absorption factors. In order to calculate a dermal RfD for a specific chemical, the oral RfD is multiplied by the oral absorption efficiency value expressed in decimal form (USEPA 2004). The absorption efficiencies and the adjusted RfDs used are presented in Table 4-2.

## 4.3. Evaluation of Potential Exposure to Lead

#### 4.3.1. Lead Exposure in Adults Using the Adult Lead Model

The EPA Adult Lead Model (ALM) was utilized to determine if nonresidential lead exposures at the site pose a significant risk (USEPA 1994b, 2003b, 2007, 2009). This method focuses on estimating blood lead concentrations (PbB) in fetuses carried by women exposed to average concentrations of lead measured in environmental media (adult exposure to soil; ultimate receptor is fetus). Unlike the IEUBK model, the ALM does not consider contributions from other environmental media but it does account for a non-zero baseline blood-lead level. The default baseline blood lead levels were assumed.

This method is based on a probability model for PbB in adult women exposed to lead in environmental media coupled with an estimated constant of proportionality between fetal and maternal PbBs, a geometric mean fetal PbB concentration and an empirically determined geometric standard deviation. The statistical terms used in the method allow the user to estimate an average adult PbB such that a fetus has not more than a five percent probability of PbB exceeding 10  $\mu$ g/dL. Soil lead levels with no more than a five percent chance that the blood lead level in a fetus will exceed 10  $\mu$ g/dL are considered to be below the risk threshold (USEPA 2003b).

Table 4-4 summarizes the parameters selected for use in the ALM for assessing lead exposures for adults. As above, intake rates and exposure frequencies for contact with site media are the same as the exposure parameters utilized for non-lead chemicals (Tables 3-1 and 3-2), and the biokinetic modeling parameters are the recommended defaults (USEPA 2003b).

#### 4.3.2. Lead Exposure in Children Using the IEUBK Model

Toxicity values are not available to evaluate the noncancerous health risks associated with lead so it must be evaluated using a separate methodology. EPA considers the development of a reference dose (RfD) to be inappropriate because no threshold has been established for the most sensitive noncancer effects of lead in infants and young children (USEPA 1994). The Integrated Exposure Uptake Biokinetic (IEUBK) Model was developed to predict the probable blood lead level for children between 6 months and 7 years of age who have been exposed to lead through environmental media (air, water, soil, dust, and diet) (USEPA 2010b). This model utilizes separate components for exposure, absorption and the biokinetic transfer of lead to all tissues of the body and calculates age-specific blood lead concentrations for children. According to EPA recommendations, model results protective of human health include those for which the probability of a blood level >10  $\mu$ g/dL blood lead level was selected based on studies indicating that exposures resulting in blood lead levels at or above this concentration may present an increased health risk to children (CDC 1991, 2002).

For the current evaluation, input values selected for the parameters in the IEUBK model are summarized in Table 4-5. The model was run with a combination of EPA default parameters and site-specific information for lead in sediment and fish as noted in the table. Upper level values were included in the model run to represent an RME scenario. This model considers additional sources of lead exposure such as outdoor air  $(1 \ \mu g/m^3)$ , drinking water (4  $\mu g/L$ ) and maternal blood lead level at birth (1  $\mu g/dL$ ).

EPA has a goal of limiting exposure to lead in soil such that "a typical (or hypothetical) child or group of similarly exposed children would have an estimated risk of no more than 5 percent of exceeding a 10  $\mu$ g/dL blood lead level" (USEPA 1994). The 10  $\mu$ g/dL blood lead level was selected based on studies indicating that exposures resulting in blood lead levels at or above this concentration may present an increased health risk to children (CDC 1991, 2002). The results of the model run are included in Table 4-5 and discussed in the next section.

#### 5. RISK CHARACTERIZATION

#### 5.1. Direct Contact with Contaminated Sediment

#### 5.1.1. Characterization of Cancer Risks

Quantification of cancer risks involves the calculation of ELCRs or excess lifetime cancer risks. These values represent the probability of an individual developing cancer over a 70-year lifetime associated with exposure to a cancer-causing chemical. An ELCR of  $1 \times 10^{-6}$  indicates that an exposed individual has a one in a million increased risk of developing cancer as a result of exposure to the specified chemical.

ELCRs for evaluation of dermal contact and ingestion pathways were calculated for each COPC using the following formula:

$$ELCR = \sum_{i=1}^{n} (LADI_i \text{ or } LADD_i) \times CSF_i$$

The CSFi is expressed in units of mg/kg-day)<sup>-1</sup> for each compound and the lifetime average daily intake (LADI) and lifetime average daily dose (LADD) are expressed in units of mg/kg-day for each compound. The resultant product, or ELCR, is dimensionless since the units cancel out.

Table 10 summarizes the estimated CTE and RME summative cancer risk estimates for contact with sediments by exposure group. The majority of risk estimates across exposure groups are below the risk level of  $1 \times 10^{-5}$ . However, the cancer risk estimate for the high-end child exposure group was higher than this level at  $3.7 \times 10^{-5}$ . This summative risk was driven by the dermal exposure pathway.

The cancer risks by COPC are summarized in Tables 5-1 through 5-4 in the Appendix. For the dermal contact pathway, benzo(a)pyrene contributed the most to the summative cancer risk followed by arsenic. For the incidental ingestion pathway, the converse was true with arsenic having the highest contribution to the risk estimates followed by benzo(a)pyrene. This was true for all exposure groups.

 Table 10: Summary of Noncancer and Cancer Risks for Direct Contact with Sediments for

 Presque Isle Bay Recreational Water Users

	N	oncancer ri	sks	Cancer risks			
	HQ <sub>dermal</sub>	HQ <sub>oral</sub>	HI	ELCR <sub>dermal</sub>	<b>ELCR</b> <sub>oral</sub>	<b>ELCR</b> <sub>sum</sub>	
Adult RME	1.9E-02	9.4E-03	2.9E-02	6.5E-06	2.5E-06	9.0E-06	
Adult CTE	1.4E-03	8.9E-04	2.2E-03	1.6E-07	7.8E-08	2.4E-07	
Child RME	4.8E-01	8.8E-02	5.7E-01	3.3E-05	4.7E-06	3.7E-05	
Child CTE	8.9E-03	8.3E-03	1.7E-02	7.1E-07	4.9E-07	1.2E-06	
Adult Lead Model <sup>(3)</sup> RME	y that fetal P <0.6%						
Adult Lead Model <sup>(3)</sup> CTE	Probability that fetal PbB > PbB <sub>t</sub> = <0.4%						
Child IEUBK Model <sup>(4)</sup>	Probabi	lity that child μg/dL < 0.3 9	l PbB > 10 ‰				

Notes:

(1) No available RfD or RfC

(2) No cancer slope factor available for oral exposures

(3) The Adult Lead Model (EPA, 2009) was used to assess the noncancer risks from lead exposure.

(4) The Child IEUBK Model was used to assess the noncancer risk from lead exposure

Acronyms

HQ = hazard quotient

HI = Hazard Index (sum of HQs across exposure pathways)

ELCR = excess lifetime cancer risk

#### 5.1.2. Characterization of Noncancer Risks

Risk characterization of noncancer effects of a chemical involves comparing the ratio of the Average Daily Intake (ADI) or the Average Daily Dose (ADD) to the RfD for the ingestion or dermal contact routes. This ratio is referred to as the Hazard Quotient (HQ) and is calculated as follows:

$$HQ_i = \frac{ADI_i \text{ or } ADD_i}{RfD_i}$$

HQs for the same chemical but from different exposure pathways were calculated by summing across all HQs. To calculate the cumulative HI, which represents the adverse effects associated with simultaneous exposure to all detected chemicals, all the calculated chemical-specific HQs were summed to derive a hazard index for each chemical. The HIs

were summed across exposure pathways for all COPCs since there is considerable overlap between the systemic effects.

The noncancer risk estimates for recreational water users are summarized in Table 10. The HIs across all exposure pathways for each COPC were less than 1 indicating that noncarcinogenic effects from contact with contaminated sediments are not likely to occur. Arsenic and total PCBs were the only COPCs which had established RfDs allowing for noncancer risks to be calculated. Total PCBs contributed the most to noncancer risks for deremal contact while arsenic had the higher contribution to incidental ingestion of sediments.

## 5.1.3. Adult Lead Exposures (Noncancer Effects)

Results of the adult lead model for adult recreational water users are summarized in Table 10 and in Table 4-4 (Appendix). The modeling results included the updated adult female PbB estimates from the 2000-2004 NHANES III Study (USEPA 2009b). The modeling results estimated the probability that fetal PbB would exceed 10  $\mu$ g/dL on-site to be less than one percent for both the CTE and RME calculation for adult recreational water users. This result suggests that females exposed to lead through direct contact with sediment within the study area have a low probability of developing blood lead levels that would cause harm to the fetus (USEPA 2003b).

#### 5.1.4. Lead Exposures in Children (Noncancer Effects)

The results of the IEUBK model show a low probability of risks from lead exposure among children exposed to sediments and fish tissue within the study area (refer to Table 10 and Table 4-5 in the Appendix). The model was run including both EPA default values and site specific assumptions. The results include exposures from contact with contaminated sediments and ingestion of contaminated fish. It also includes exposures to lead from other sources such as outdoor air, drinking water and from maternal exposures. The results of the IEUBK model suggest that the probability of a child developing blood lead levels of 10  $\mu$ g/dL and above is less than 0.3 percent considering site-specific and other exposures. This is well below the EPA target level of five percent.

#### 5.2. Ingestion of Contaminants in Fish Tissue

#### 5.2.1. Characterization of Cancer Risks

The procedure for calculating cancer risks for contaminants in fish tissue was identical to that utilized for the exposure pathways for sediment. The summative cancer risks by exposure group are summarized in Table 11. The summative excess cancer risks for the

high end exposure scenarios (urban/subsistance anglers) were very significant varying from  $1.0 \times 10^{-4}$  to  $4.9 \times 10^{-3}$  for adult anglers. The fish species showing the highest summative cancer risk was lake trout at  $4.9 \times 10^{-3}$ . The main COPC driving these risk estimates was either Arochlor 1254 or 1260 or both (refer to Tables 5-5 through 5-8 in the Appendix). Summative CTE estimates for cancer risk in adult anglers were in the range of  $1 \times 10^{-6}$  to  $\times 10^{-5}$ . These values are still within the range of concern as delineated by EPA.

		Noncan (summa	cer risks tive HIs)		Cancer risks (summative ELCRs)				
Fish species	Adult		Child		Ad	lult	Child		
	RME	СТЕ	RME	СТЕ	RME	СТЕ	RME	СТЕ	
Presque Isle Bay	Anglers a	nd Their	Children	n					
Bluegill				No	COPCs				
Brown bullhead <sup>(1)</sup>	0.41	0.05	0.47	0.05	n/a	n/a	n/a	n/a	
Common carp	48.2	6.0	48.3	6.0	8.0E-04	3.0E-05	1.6E-04	2.0E-05	
Largemouth bass	11.4	1.6	11.4	1.5	1.0E-04	2.8E-06	2.6E-05	1.9E-06	
Northern pike	0.67	0.08	0.67	0.08	1.0E-04	3.6E-06	1.9E-05	2.4E-06	
Pumpkinseed				No	COPCs				
Lake Erie Angle	rs and The	eir Child	ren						
Burbot	10.8	0.95	10.8	0.91	1.3E-04	2.7E-06	2.6E-05	1.8E-06	
Channel catfish	100	8.5	99.8	8.4	2.1E-03	5.0E-05	4.0E-04	3.3E-05	
Lake trout	183.7	3.2	183.8	3.2	4.9E-03	5.0E-05	1.0E-03	3.1E-05	
Smallmouth bass	132.8	12.5	150.1	12.5	3.1E-03	7.0E-05	6.1E-04	4.7E-05	
Walleye	39.4	3.2	39.3	3.2	7.4E-04	2.0E-05	1.4E-04	1.2E-05	
White bass	26.4	3.1	26.4	3.1	5.9E-04	2.0E-05	1.2E-04	1.4E-05	
White sucker	30.5	3.5	30.5	3.5	4.6E-04	2.0E-05	9.4E-05	1.1E-05	
Yellow perch <sup>(2)</sup>	3.1	0.27	3.1	0.27	n/a	n/a	n/a	n/a	

Table 11: Summary of Noncancer and Cancer Risks for Ingestion of Fish

Notes:

(1) only selenium found over the RSL - no cancer toxicity value is available

(2) only selenium and mercury found over the RSL – no cancer toxicity values available

## 5.2.2. Characterization of Noncancer Risks

The procedure for calculating noncancer risks for contaminants in fish tissue was identical to that utilized for the exposure pathways for sediment. The summative noncancer risks, or hazard indices, by exposure group are summarized in Table 11. Lake trout was the species with a consistently higher hazard index across exposure groups. The summative hazard index for RME estimates was approximately 184 for both adult anglers and children of adult anglers. These values are much higher than the target level established by EPA which is 1. As with cancer risk estimates, Arochlor 1254, 1260 or both was the COPC with the highest contribution to the noncancer risk estimates.

## 6. UNCERTAINTY ANALYSIS

Uncertainties are inherent in any human health risk assessment due to the use of environmental sampling results, modeling approaches, assumptions regarding exposure, and the toxicity of particular constituents. This risk assessment has incorporated site-specific information, where feasible, in order to reduce the uncertainty associated with those assumptions. However, in many instances, there was little quantitative information to include in terms of site-specific assumptions for Presque Isle Bay and/or Lake Erie.

Analysis of the critical areas of uncertainty in risk assessment provides context for better understanding the assessment conclusions by identifying the uncertainties expected to most significantly affect the results. In this risk assessment, where assumptions were made, the uncertainty errs on the conservative side in order to protect human health (i.e., overestimate human health risks). Table 6-1 within the Appendix summarizes the major sources of uncertainty in this risk assessment and provides a qualitative judgment on the magnitude of each source in terms of its likelihood to under- or over-estimate human health risks.

## 6.1. Hazard Identification

#### 6.1.1. Data Accuracy and Site Characterization

A major concern of any risk assessment is the accuracy and completeness of COPC identification, both in terms of ensuring that all contaminants have been correctly identified as COPCs, and ensuring that concentrations are adequately quantified. In order to maintain precision and accuracy of sampling and analytical procedures, EPA-approved sampling and analytical procedures were followed in order to characterize the site. All samples were collected and analyzed following appropriate quality assurance/quality control procedures.

The accuracy of COPC identification is directly related to the quality of COPC characterization data, including information on contaminant identification, location, and concentrations. The sampling data included in this analysis was collected for purposes other than an HHRA. As a result, samples were not necessarily collected in areas specific to exposure activities (i.e., swimming, wading, fishing) such as those characterized in this study. Based on best professional judgment, the sampling data were grouped in an attempt to best reflect exposure areas. It is possible, however, that sample locations could have been included for an exposure that may overestimate potential exposure for some populations while underestimating potential exposure for others.

A limitation of the fish tissue data is that in many instances, the chemical concentration was based on one composite sample of five individual fish. This adds uncertainty to the risk assessment in that it is difficult to ascertain how representative this composite is for the species as a whole in Presque Isle Bay. Less uncertainty exists for those species which included more than one composite in the analysis.

Sediment and fish tissue samples were collected from the site over a limited number of days. Although these data were collected during the spring, summer and fall, they represent a snapshot in time and may not be representative of concentrations present at other times of the year under different conditions.

## 6.1.2. Screening of COPCs Using RSLs

The screening criteria used at the Site were chosen to represent conservative and reasonable screening criteria as established by EPA Region 3. The screening process was designed to identify those constituents that were site-specific and likely to exceed conservative risk-based criteria for residential use. These criteria therefore, include the assumption that an individual would be exposed to sediments 350 days per year for 30 years. A level of uncertainty exists with chemicals that do not have a specific RSL. In this instance, a surrogate value was used which represents a chemical constituent with a similar structure that is assumed to pose the identical human health risks.

A number of uncertainties exist in the selection of COPCs for inclusion of the risk assessment including those associated with sampling/analytical procedures; the number of samples for use to estimate the COPC and the selection of the appropriate screening criteria.

## 6.1.3. Detection Limits Greater than the RSLs

Uncertainty exists in the screening and evaluation of chemicals that had method detection limits exceeding the EPA Region 3 RSLs. Site-specific RSLs for some chemicals are exceptionally low, and in some instances, may not be attainable with currently available laboratory methods. For fish tissue samples, detection limits exceeded the screening levels for many of the COPCs (see Table 2-3 through 2-16). Chemicals that were not detected were not carried through the entire risk assessment evaluation. If chemicals were present at concentrations above the screening levels but below the detection limits, it is possible that these chemicals could contribute to unacceptable risks.

## 6.1.4. Use of Trivalent Chromium Toxicity Parameters for Total Chromium

Chromium was analyzed as total chromium in all media. However, screening values only exist for the hexavalent and trivalent species. For the purposes of this risk assessment it was assumed that the majority of the total chromium measured in sediment and fish tissue was in the trivalent form. Therefore, screening levels for trivalent chromium were used to determine whether chromium would be carried forward in the risk assessment. While there is uncertainty in this approach it is likelihood of underestimating the risk from hexavalent chromium is minimal. As previously discussed, the majority of chromium within a reducing environment is the trivalent form and hexavalent chromium has a low BCF in fish tissue indicating that it is unlikely to bioaccumulate (ATSDR 2008, USEPA 1998).

#### 6.1.5. Use of Structural Analogy to Determine Surrogate Screening Levels

Some chemical constituents in sediment and fish tissue did not have an associated screening level. In this instance structural analogy was used to screen this particular chemical. This is generally due to the lack of toxicity information available for this specific chemical constituent. For chemical constituents in sediment the following substitutions were made: pyrene for benzo(e)pyrene and perylene; anthracene for phenanthrene; and acenaphthene for benzo(g,h,i)perylene and acenaphthalene. The use of surrogate screening levels may under- or overestimate the risk associated with a particular chemical.

#### 6.2. Exposure Assessment

In order to estimate the amount of a COPC for a particular receptor a number of assumptions must be made about the duration and frequency of exposure and characteristics inherent to a particular receptor (i.e., body weight, skin surface exposed to soil). Although effort has been taken to apply site-specific and receptor-specific exposure factors, for those with limited

data, EPA defaults were used in a number of cases. These recommended defaults are also based on limited data and are chosen to represent conservative estimates. It is likely that the actual exposure factors are much lower than the default values suggested by the EPA resulting in an overestimation of the human health risks.

## 6.2.1. Lack of Data to Evaluate Surface Water Exposure

There was no data available to evaluate exposure from contact with chemicals in surface water. Therefore, exposure pathways such as inhalation, dermal contact and incidental ingestion of contaminated water could not be evaluated. This is likely to underestimate the risk, however, the impact is likely to be negligible. Many of the chemicals of concern are lipophilic in nature and not likely to be present in appreciable concentrations within the water column.

## 6.2.2. Exposure Assessment from Fish Consumption

## 6.2.2.1. Fish Consumption Rates

There was little quantitative information on fish consumption rates in the Presque Isle Bay area therefore, fish consumption rates were based on the national per capita consumption of estuarine and freshwater fish (USEPA 2002). The 90th and 99th percentile ingestion rates for children and adults were selected to evaluate potential risks over a range of possible ingestion rates. The extent to which these assumptions correspond to consumption patterns in the study area is unknown.

## 6.2.2.2. Use of Single Fish Species Consumption Pattern

Risk estimates were based on the consumption of individual fish species and tissue types. However, it is very likely that an individual's diet would include multiple fish species. A mixed-diet scenario was not evaluated for this risk assessment because of the lack of species-specific consumption data for the study area.

## 6.2.2.3. Use of Fillets to Represent All Fish Consumption Patterns

Bioaccumulation of chemicals in fish tissue will differ depending on the chemical. Organic compounds, especially lipophilic chemicals, tend to accumulate in fatty tissues while metals tend to accumulate in muscle and other tissues (PA Fish and Boat 2011, Gutenmann 1992). The chemicals with the greatest contribution to the cumulative cancer risk and with the highest noncancer HQ are the Arochlors, which are organic compounds that accumulate preferentially in fatty tissue. Diets consisting of different fish parts result in varying levels of risk to the consumer. Using only whole body or fillet tissue with skin to evaluate risk from all types of fish tissue diets is likely to overestimate chemical exposure from consumption of contaminated fish. Since PCBs contribute to the vast majority of risks from tissue consumption, this uncertainty could have a significant impact on the conclusions of this HHRA. Alternatively, chemicals such as methyl mercury preferentially accumulate in muscle tissue, which means concentrations of mercury in fillet tissue would likely be higher than concentrations of mercury in whole body fish tissue.

## 6.2.2.4. Assumption of Residence Time of Fish Species

For the purposes of this HHRA the fish species were assumed to be denizens of either Presque Isle Bay or Lake Erie based on sampling location and habits of each species. Species that were considered to reside mostly in Presque Isle Bay included: bluegill, largemouth bass, northern pike, pumpkinseed sunfish, brown bullhead and common carp. Fish species assumed to reside in Lake Erie included: burbot, channel catfish, lake trout, smallmouth bass, walleye, white bass and white sucker. A summary of these assumptions is included in Table 4 (page 14 and 15 of this report).

Since many fish are migratory in nature (i.e., burbot and smallmouth bass during spawning) (Grazio, 2012) and migrate between the lake and bay on a seasonal basis, their exposure may represent chemicals from time spent in both the Bay and Lake waters.

## 6.2.2.5. Sample Size and Length of Fish Collected for Study

The sample size of the fish analyzed in this study represents another source of uncertainty in the risk estimates. Limited numbers of fish of an individual species were collected between 2004 and 2010. Each composite represents ten fillets that were collected at a given period of time. This small sample size and sampling period may not adequately represent the concentration of contaminants in fish species within the general study area.

Fish were collected such that composite samples included fish species of similar lengths and therefore, age (PADEP 2010). The length, and in essence, the age of fish, is positively correlated with the contaminant body burden concentration within fish tissue (Gutenmann et al. 1992; Young, et al. 1994). The risk estimates in this HHRA are based on various fish species of a given length. Individuals that consume fish of a smaller or longer length than those included in this study may have risks that lower or higher than estimated in this report. Fish species with size limit regulations are more

likely to be better represented in the data since the sampled fish are close to the allowable size limits.

## 6.2.3. Exposure Point Concentration

The sampling data included in this analysis was collected for purposes other than an HHRA and, therefore, may have limitations in terms of adequately characterizing all the human exposure scenarios evaluated in this report. In order to account for these uncertainties upper bound estimates or maximum values were included in this evaluation so as not to understate any potential risk.

Uncertainties associated with the exposure assessment include calculation of exposure point concentrations and selection of exposure parameters. The RME approach was utilized in this HHRA to characterize risks. The use of high-end values as exposure parameters, including the 95% UCLs, prevents an underestimation of the health risks. In addition, the maximum value of the COPCs were utilized for the RME estimates which is likely to overestimate the intake and dose calculations from exposure to sediment and thus overestimate the health risks.

For some chemicals in sediment samples, the calculation of average exposure point concentrations relied upon sample data where the concentration was reported as below the detection limit. These chemicals were assumed to be present at a concentration equal to one-half the detection limit in order to calculate an EPC. This practice increases the uncertainty of the resulting exposure point concentrations because the actual sample concentration may range from zero to the full detection limit.

## 6.2.4. Extrapolation of Chemical Concentrations Over Time

Another source of uncertainty in this risk assessment involves the use of the average chemical concentrations for fish and sediment collected over a short period of time to estimate human exposure durations of 9, 30 and 70 years. If average chemical concentrations in these media have changed over time, or are likely to change in the future, the risk estimates presented in this report may either underestimate or overestimate the risk to individuals. The existing historical data on sediment contamination in Presque Isle Bay suggests that many of the chemical concentrations are decreasing over time (PADEP 2005). If this trend continues, the extrapolation of current chemical concentrations into the future is likely to overestimate the human health risks from exposure to sediment and fish tissue.

#### 6.2.5. Exposure Duration and Frequency

Exposure duration is defined as the time period over which an individual is exposed to one or more contaminants. Two defaults were used for the risk assessment: 70 years, which represents the average lifetime exposure duration; 30 years, which represents the 90th percentile length of time that an individual resides at one residence; and 9 years which represents the median amount of time an individual resides at a given residence These parameters are conservative default values obtained from EPA guidance documents and are typically used to estimate CTEs and RME. These values may overestimate the risks for actual receptors.

The frequency of residents' exposure to bay sediments was determined by using best professional judgment with consideration of the weather conditions in western Pennsylvania. This value is higher than the default value indicated in the exposure factor handbook of 12 days per year. The exposure frequency utilized in the exposure estimates may overestimate the exposure contact with sediments for some but underestimate it for individuals frequently participating in water activities,

## 6.2.6. Use of Dermal Absorption Factors for Soil

The bioavailability of COPCs in sediment was considered by using the dermal absorption factors for soil in the dose calculations (USEPA 2011a). Unlike soil, sediments are consistently water-covered, more likely to wash off, and consequently tend to have a shorter contact time on skin than soil. As a result, dose calculations may be overestimated. Default absorption factors were included for those chemicals that did not have a specific one available. For example, inorganic chemicals, such as chromium was considered to have a value of 0.1. This assumption is likely to overestimate the actual amount of chromium absorbed through the skin.

Also, the greater the moisture content of the sediment, the greater the difference between wet vs. dry weight contaminant concentration. Because estimates of sediment adherence reflect wet weight (i.e., in situ), and the estimated intakes are based on sediment sample results recorded in dry weight, the resulting risk estimates are over-estimated in direct proportion to the moisture content of the sediment. Conversely, increased moisture content increases the ability of sediment to adhere to skin and may also affect the relative percent absorbed. Therefore, EPA recommends the use of the same dermal absorption fraction for sediments as for soil until more information becomes available (EPA 2004a).

#### 6.3. Toxicity Assessment

#### 6.3.1. Toxicity Values

The toxicity information used in the health risk assessment adds a degree of uncertainty to the risk estimates. The uncertainties specific to the toxicity assessment are associated with: the toxicity studies that form the basis for the toxicity values recommended by EPA and the lack of sufficient toxicity data to develop toxicity values for certain substances. In order to reduce the extent of this uncertainty to the extent possible, the most current toxicity values were utilized in this risk assessment (USEPA 2011a, 2011b). The extrapolation used in developing toxicity values may contribute to uncertainty in the risk estimates. An additional source of uncertainty originates from toxicity values that are chemical-specific and do not take into account interaction with other chemicals.

The toxicity values (i.e., RfDs and CSFs) used in this risk assessment were developed by the EPA for regulatory purposes and are intended to represent upper-bound estimates of toxicity. For example, most of the RfDs incorporate large uncertainty factors which are intended to lie well below the true threshold for toxicity in humans. While this helps ensure the protectiveness of decisions based on the RfD, it should be recognized that a dosage exceeding the RfD (i.e., a HQ > 1.0) does not necessarily indicate the likelihood for toxicity given the level of uncertainty within various elements of the risk assessment process.

Similarly, the CSFs developed by the EPA incorporate a number of conservative choices in risk extrapolation. These include the assumption of a linear, non-threshold doseresponse relationship for cancer, interpretation of animal carcinogenicity data, and dosemetrics for extrapolation of results from rodents to humans. As a result, estimates of lifetime cancer risks including these values reflect conservative upper bound estimates of risk associated with specific exposures. They may be extrapolated from high-dose to low-dose models, laboratory animal studies, and/or subchronic studies.

## 6.3.2. PCB Congeners and Arochlor Mixtures

In this risk assessment, two different classes of PCBs were measured. In sediment samples, 19 PCB congeners were measured. Only two of the congeners had specific toxicity values. The approach used was to sum the concentrations of all PCB congeners to develop a total PCB concentration. This total concentration was then compared to the toxicity values for high risk PCBs. Only two of the congeners (PCB 105 and PCB 118) are considered to be dioxin-like congeners. Therefore, this approach is likely to overestimate the risks associated with PCB congeners in sediment.

In fish tissue, a total of 6 Arochlors were measured including Arochlor 1221, 1232, 1242, 1248, 1254 and 1260. These represent commercial mixtures of PCBs with the last two digits of the Arochlor representing the percentage by weight of chlorine in the mixture. The reference dose for Arochlor 1254 was used as a surrogate for Arochlor 1260 which does not have an associated reference dose. This approach adds uncertainty to the calculation and is likely to overestimate the noncarcinogenic risk associated with exposure to Arochlor mixtures.

## 6.3.3. DDD, DDE, DDT and its Derivatives

DDT and its derivatives, DDD and DDE, were measured in both sediment and fish tissue samples. For noncarcinogenic risk estimates, a conservative approach was employed which involved the summation of DDT, DDD, and DDE per sample (total DDT) and the use of the RfD associated with DDT to calculate an HQ. Alternatively, only DDT could have been used in the HQ because it alone has an RfD. DDT has been identified as having a hepatic health endpoint as based on the RfD value, and therefore the treatment of DDT and its derivatives will affect the HQ and the HI for hepatic toxicity.

#### 6.4. Risk Characterization

The summation of HQs and ELCRs across chemicals and pathways are primary uncertainties in the risk characterization. Summation of HQs across different COPCs is most properly applied to compounds that induce the same effects by the same mechanism. However, in the absence of information on the toxicity of specific chemical mixtures, it is assumed that ELCRs and HQs are additive (i.e., cumulative) (EPA 1989). One of the limitations of this approach for noncarcinogens is that the effects of a mixture of chemicals are generally unknown and it is possible that the interactions could be synergistic, antagonistic, rather than additive. Additionally, the estimated values of the RfDs have different accuracy and precision and are not based on the same severity or effect.

#### 7. CONCLUSIONS

As part of this human health risk assessment excess lifetime cancer risks and hazard indices were calculated for direct contact with sediments from Presque Isle Bay and ingestion of fish from Presque Isle Bay and Lake Erie. These risks were compared to the target levels established by EPA of 10<sup>-5</sup> (cancer risks) and 1.0 (noncancer risks) for the following exposure groups:

- PIB Adult recreational water users (RME and CTE);
- PIB Child recreational water users (RME and CTE);

- PIB and Lake Erie Adult urban/subsistence anglers (RME)
- PIB and Lake Erie Adult recreational anglers (CTE)
- PIB and Lake Erie Children of urban/subsistence anglers (RME)
- PIB and Lake Erie Children of recreational anglers (CTE)

## 7.1. Risks Associated with Direct Contact with Contaminated Sediments

A quantitative analysis was conducted to evaluate risks to recreational water users for direct contact of contaminants in sediments. All chemical-specific and cumulative excess lifetime cancer risk estimates were below  $1 \times 10^{-5}$  and all chemical-specific and cumulative hazard indices were below 1.0 with the exception of the RME cancer estimate child recreational water users. The estimate for this exposure group was  $4 \times 10^{-5}$  and mainly driven by dermal exposure from total PCBs. It should be noted that these risk estimates are conservative in nature and likely to overestimate the risk (the uncertainties associated with these estimates are discussed in section 6 of this report). The results of the Adult Lead and IEUBK models show that lead concentrations measured in bay sediments pose insignificant noncancer health risks to child and adult female populations.

## 7.2. Consumption of Contaminated Fish

Cancer and noncancer risks were analyzed for the consumption of 14 separate fish species using sampling data gathered in Presque Isle Bay and Lake Erie between 2004 and 2010. The total number of COPCs varied by fish species from zero to 16. For this particular dataset, lake trout and smallmouth bass were the fish species with the highest and second highest cancer and noncancer risks compared to the other twelve species. Panfish, including both pumpkinseed and bluegill species, had the lowest concentrations and lowest risks of all fish species evaluated. The contaminants with the largest contribution to the summative risk estimates included Arochlor 1254 and 1260. In all cases, these COPCs contributed more than 50 percent of the overall cancer and noncancer risk estimates (data not shown).

Fish tissue sampling data from the current study was compared with data collected at other areas within Lake Erie (refer to Table 7-1 in Appendix) (Carlson et al. 2000, Perez-Fuentetaja et al. 2006 and Sadraddini et al. 2011). There was not enough data from the current study or information in the comparison studies from which to conduct statistical analyses. A qualitative comparison shows that many of the concentrations measured in fish tissue in Presque Isle Bay were comparable or lower than those measured in other studies.

The results of this comparison should be used with caution due to the limited amount of sampling data and differences in study methodology. Other issues that should be noted include differences in sampling time period, sample size, and species evaluated.

## 7.3. Main Conclusions

- 1. Overall, these results show that the main exposure route for contaminants in Presque Isle Bay is through fish consumption. These risks were several orders of magnitude greater than those associated with direct contact with contaminated sediments.
- 2. The cancer and noncancer risk estimates generated from consumption of fish tissue were highly dependent on the fish species and location (refer to Table 11). Based on the dataset utilized in this HHRA, several species from Lake Erie contributed to higher risks compared to species from Presque Isle Bay. These findings include:
  - Lake trout and smallmouth bass represented the fish species with the highest cancer and noncancer risk estimates. These species are likely to have a higher residence time and thus represent exposures to chemical constituents that occurred mainly from open water areas of the lake.
    - The summative noncancer risk for lake trout was approximately 3 for the typical or CTE estimate and 184 for the high-end or RME estimate (target level = 1.0). This latter value indicates that the estimated exposure to this chemical from consuming fish is 184 times greater than the level recommended by the EPA.
    - The summative cancer risk for lake trout was  $5 \times 10^{-5}$  (5 in 100,000) and  $5 \times 10^{-3}$  (5 in 1,000) for the CTE and RME estimates respectively (target level = 1 in 100,000).
    - The contaminant with the highest contribution to the noncancer and cancer risk estimates for lake trout and smallmouth bass was Arochlor 1254 or 1260.

(It should be noted that the cancer and noncancer risk estimates include the assumption of a single species diet and that all fish consumed originates from Lake Erie. These assumptions are conservative in nature and likely to overestimate the cancer and noncancer risks from consumption of fish. It should also be considered that these risk estimates are based on a limited sampling of fish tissue.)

- **3.** Values for certain fish species from Presque Isle Bay were also greater than the applicable cancer and noncancer risk thresholds. These results include:
  - Common carp and largemouth bass were the species with the highest associated risks.
    - The summative noncancer risk for common carp was 6 for the typical or CTE estimate and 48 for the high-end or RME estimate.

- The summative cancer risk for lake trout was  $3 \times 10^{-5}$  (3 in 100,000) and  $8 \times 10^{-4}$  (8 in 10,000) for the CTE and RME estimates respectively (target level = 1 in 100,000).
- The contaminant with the highest contribution to the noncancer and cancer risk estimates for common carp was Arochlor 1254.
- The concentration of chemical constituents measured in panfish, including bluegill and pumpkinseed species, were all below the applicable fish tissue screening levels. Therefore, risk estimates were not calculated for these species.
- 4. The cancer and noncancer risk estimates for direct contact with contaminated sediments from Presque Isle Bay were generally below the target risk levels for all exposure groups evaluated in this HHRA. All chemical-specific and cumulative excess lifetime cancer risk estimates were below  $1 \times 10^{-5}$  (1 in 100,000) and all chemical-specific and cumulative hazard indices were below 1.0. The exception to this was the RME cancer risk estimate for child recreational water users which was  $4 \times 10^{-5}$  (4 in 100,000). This value is mainly driven by dermal exposure from total PCBs. It should be noted that these risk estimates are conservative in nature and likely to overestimate the risk (the uncertainties associated with these estimates are discussed in section 6 of this report).
- **5.** The uncertainties associated with this risk assessment should be considered in utilizing the results for risk management decisions. A summary of the uncertainties inherent to this HHRA are discussed in section 6 of this report. The major uncertainties noted include the:
  - small dataset from which the risk estimates were drawn (i.e., data for certain fish species included one composite sample of five individual fish);
  - lack of specific data for the environmental media to which exposure groups are more likely to contact (i.e., for children beach sediment is a more likely exposure media compared to in-water sediment on which the risk estimates are based); and
  - lack of site-specific information on fish consumption patterns within the study area.

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